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Grays River Watershed and Biological Assessment

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Grays River Watershed and Biological Assessment

Final Report

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Summary

The Grays River Watershed and Biological Assessment was funded to address degradation and loss of spawning habitat for chum salmon (*Onchorhynchus keta*) and fall Chinook salmon (*Onchoryhnchus tshawytscha*). In 1999, the National Marine Fisheries Service listed lower Columbia River chum salmon as a threatened Evolutionarily Significant Unit (ESU) under the *Endangered Species Act of 1973* (ESA). The Grays River watershed is one of two remaining significant chum salmon spawning locations in this ESU. Runs of Grays River chum and Chinook salmon have declined significantly during the past century, largely because of damage to spawning habitat associated with timber harvest and agriculture in the watershed. In addition, approximately 20–25% of the then-remaining chum salmon spawning habitat was lost during a 1999 channel avulsion that destroyed an important artificial spawning channel operated by the Washington Department of Fish and Wildlife (WDFW). Although the lack of stable, high-quality spawning habitat is considered the primary physical limitation on Grays River chum salmon production today, few data are available to guide watershed management and channel restoration activities. The objectives of the Grays River Watershed and Biological Assessment project were to 1) perform a comprehensive watershed and biological analysis, including hydrologic, geomorphic, and ecological assessments; 2) develop a prioritized list of actions that protect and restore critical chum and Chinook salmon spawning habitat in the Grays River based on comprehensive geomorphic, hydrologic, and stream channel assessments; and 3) gain a better understanding of chum and Chinook salmon habitat requirements and survival within the lower Columbia River and the Grays River.

The watershed-based approach to river ecosystem restoration relies on a conceptual framework that describes general relationships between natural landscape characteristics, watershed-scale habitat-forming processes, aquatic habitat conditions, and biological integrity. In addition, human land-use impacts are factored into the conceptual model because they can alter habitat quality and can disrupt natural habitat-forming processes. In this model (Figure S.1), aquatic habitat—both instream and riparian—is viewed as the link between watershed conditions and biologic responses. Based on this conceptual model, assessment of habitat loss and the resultant declines in salmonid populations can be conducted by relating current and historical (e.g., natural) habitat conditions to salmonid utilization, diversity, and abundance. In addition, assessing disrupted ecosystem functions and processes within the watershed can aid in identifying the causes of habitat change and the associated decline in biological integrity. In this same way, restoration, enhancement, and conservation projects can be identified and prioritized.

A watershed assessment is primarily a landscape-scale evaluation of current watershed conditions and the associated hydrogeomorphic riverine processes. The watershed assessment conducted for this project focused on watershed processes that form and maintain salmonid habitat. Landscape metrics describing the level of human alteration of natural ecosystem attributes were used as indicators of water quality, hydrology, channel geomorphology, instream habitat, and biotic integrity. Ecological (watershed) processes are related to and can be predicted based on specific aspects of spatial pattern. This study evaluated the hydrologic regime, sediment delivery regime, and riparian condition of the sub-watersheds that comprise the upper Grays River watershed relative to their natural range of conditions. Analyses relied primarily on available geographic information system (GIS) data describing landscape characteristics such as climate, vegetation type and maturity, geology and soils, topography, land use, and road

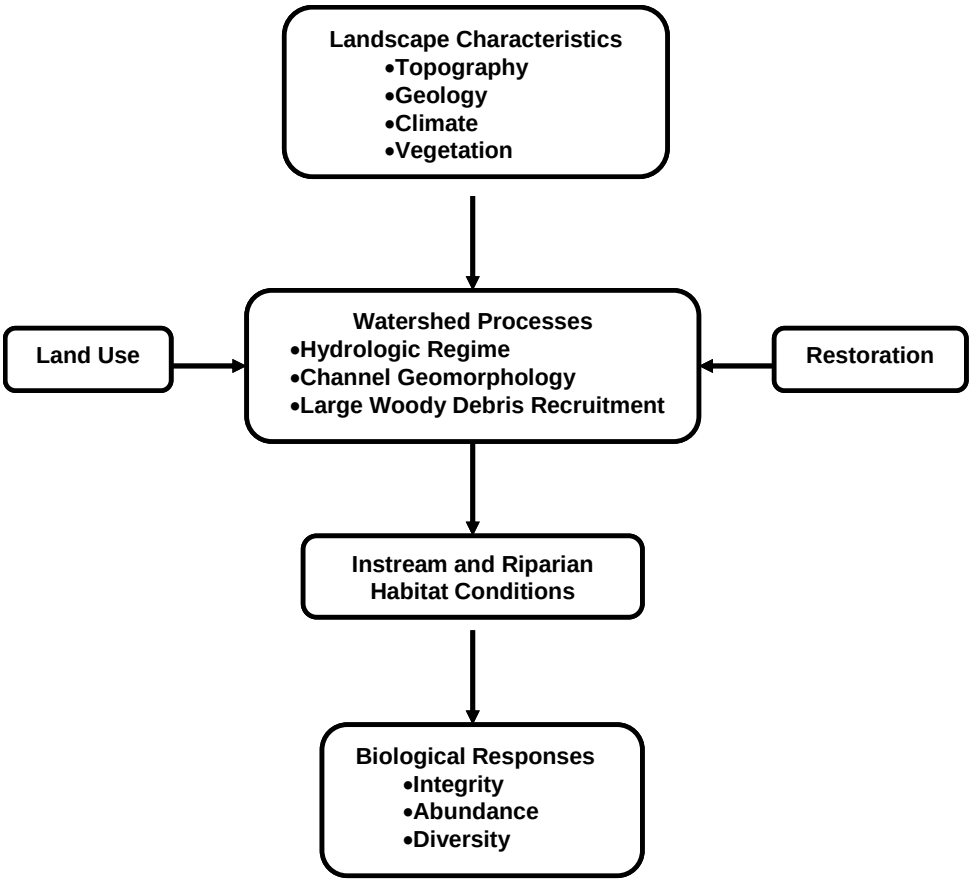

Figure S.1. Conceptual Model Illustrates Linkages Between the Landscape Characteristics, Watershed Processes, Habitat Conditions, and Biological Responses, as Well as the Interaction of Human Land-Use Activities and Restoration Efforts.

density. In addition to watershed-scale landscape characteristics, the study area was also evaluated on the riparian scale, with appropriate landscape variables analyzed within riparian buffers around each stream or river channel.

Included in the overall watershed assessment are field habitat surveys and analyses of the physical and hydrological characteristics of primary chum and fall Chinook salmon spawning areas and spawning habitat availability and use. This assessment is a significant step in a comprehensive program to ensure the survival and recovery of Columbia River chum salmon in its most productive system and builds on existing recovery planning efforts for these ESA-listed salmonids within the Grays River and the lower Columbia River. This assessment also provides a basis for the recovery of other fish species in the Grays River, including coho salmon, winter steelhead, coastal cutthroat trout, and Pacific lamprey. The conservation, enhancement, and restoration recommendations developed in this project are based on scientific assessment of historical and current watershed conditions, while at the same time considering local community stakeholder interests.

S.1 Grays River Watershed Characterization

S.1.1 Introduction

The Grays River is a tributary of the lower Columbia River and is located in Washington State a short distance upstream from the mouth of the Columbia River, primarily in Pacific and Wahkiakum counties (Figure S.2). The Grays River watershed encompasses a total of 79,635 acres (124 mi^2 or 322 km^2). The Grays River watershed can be divided into an upper and lower watershed, with the delineation at the upper extent of tidal influence, typically at the State Route 4 (SR 4) highway bridge, approximately 18.5 km upstream from the confluence with the Columbia River. The focus of this study is the upper watershed; spawning habitat analyses focus on a localized spawning area immediately above SR 4.

The watershed consists of 1,298 miles of stream channel that drain southwesterly to the Columbia River. The Grays River watershed has an average drainage-density of 10.8 miles of stream channel per square mile. The northern portion of the watershed lies within the Willapa Hills. The upper watershed is generally steep and is dissected by a dense stream network. Unconfined floodplain areas are found in the lower mainstem and locally within the watershed interior along the West, East, and South Forks of the Grays River. The watershed ranges in elevation from sea level in the southwest to over 3,000 ft above sea level in the headwaters.

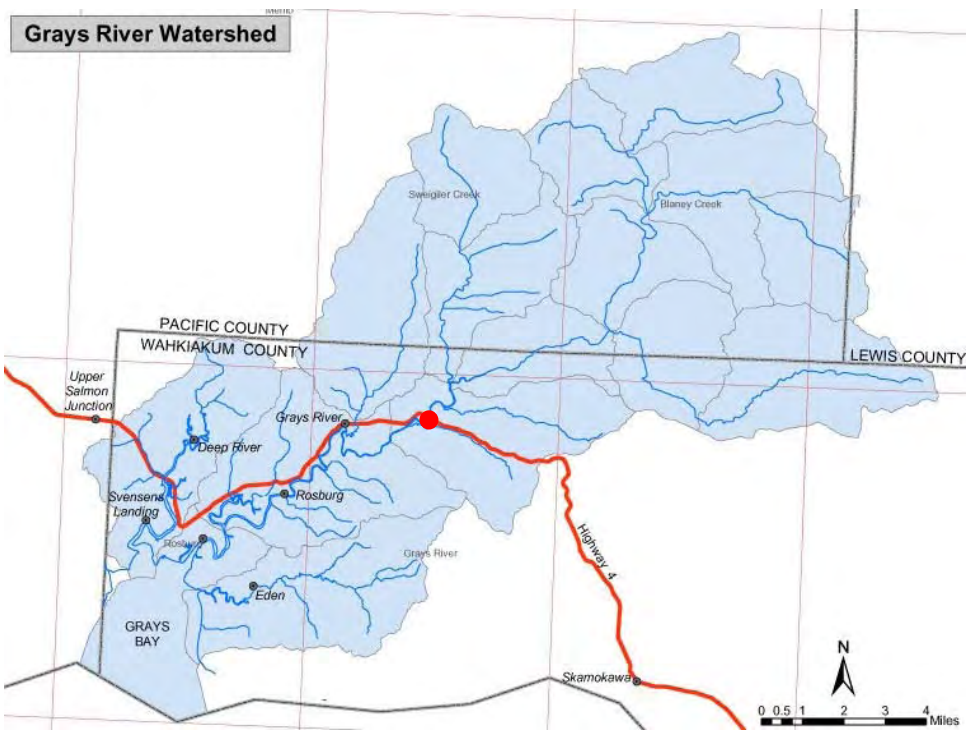

Figure S.2. Grays River Watershed Showing the Upper and Lower Portions of the Watershed, Delineated at the State Route 4 crossing (red circle). This study focuses on the upper watershed.

S.1.2 Methods and Results

S.1.2.1 Natural Watershed Characteristics

Conditions of the natural landscape in the upper Grays River watershed were identified through standard approaches of historical ecology. Historical ecology examines influences of the natural landscape as well as human social and cultural influences on that landscape. The General Land Office (GLO) mapped the lower Columbia River and Puget Sound regions between 1850 and 1895, prior to or during the earliest stages of human settlement. GLO survey records represent the earliest written description of landscape characteristics of the Grays River watershed. The human history of the Grays River Valley, including the history of logging in the watershed, is based on oral history interviews and archival material provided by members of the local community.

The Grays River watershed historically consisted almost entirely of old-growth coniferous-dominated forest, divided between two distinct forest types (Figure S.3). The Sitka-spruce zone occupies approximately 30,080 acres or 37% of the watershed, mostly at lower elevations in the lower river floodplain valley. This zone also includes some natural meadows and wetlands. The Sitka-spruce zone can be characterized by proximity to the ocean, mild temperatures, 60–80 in. annual precipitation, and frequent summer fog. The western hemlock zone occupies approximately 49,555 acres or 62% of the Grays River watershed, mostly on the steeper slopes in the upland areas of the watershed. This zone can be described as a wet maritime climate with 80–120 in. of precipitation per year.

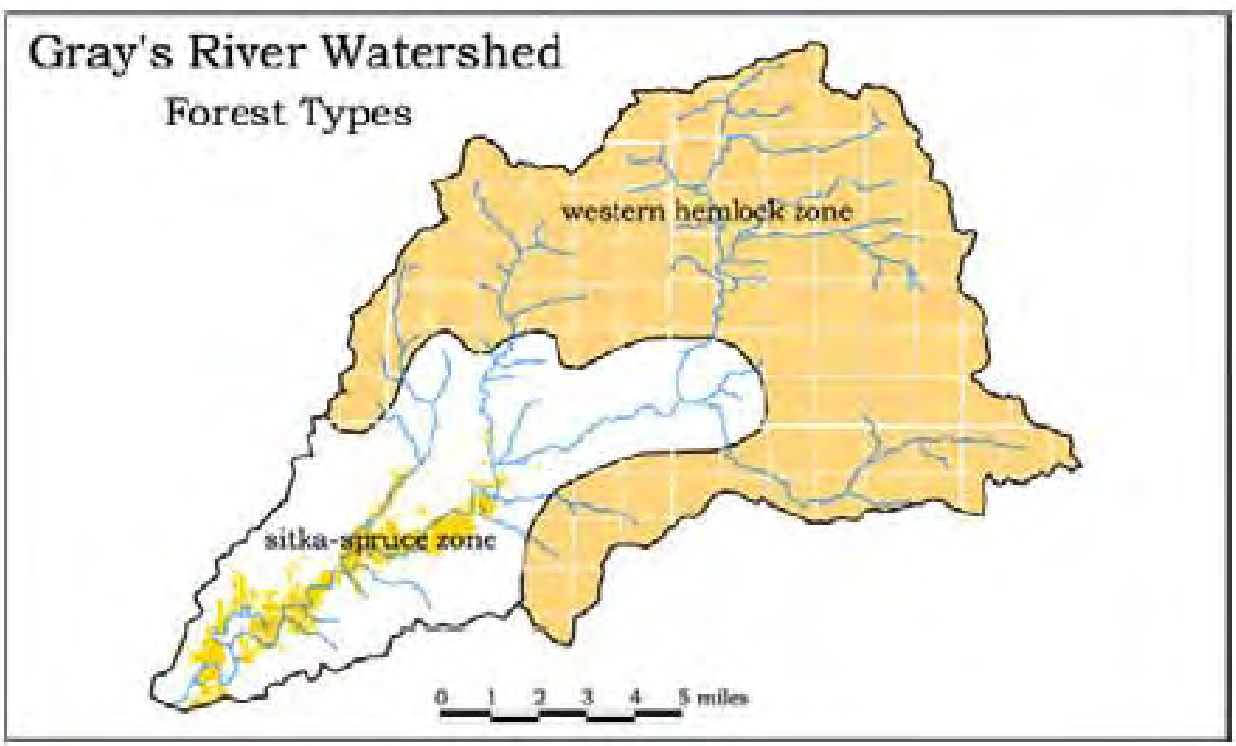

Figure S.3. Grays River Watershed Vegetation Zone Map

S.1.2.2 Current Watershed Characteristics

Currently, the upper Grays River watershed is dominated almost completely by privately owned forestland. Little unlogged, natural old-growth forest remains within the upper Grays River watershed (Figure S.4). Over 90% of the upper Grays River watershed is forested, with about two-thirds of the area classified as coniferous-dominated forest. At any one time, about 5–10% of the watershed can be classified as recently disturbed (clear-cut or burned). Only a small fraction (0.2%) of the upper Grays River watershed is classified as agricultural, with all of that land use located in the lower mainstem subbasin, at the very downstream end of the study area.

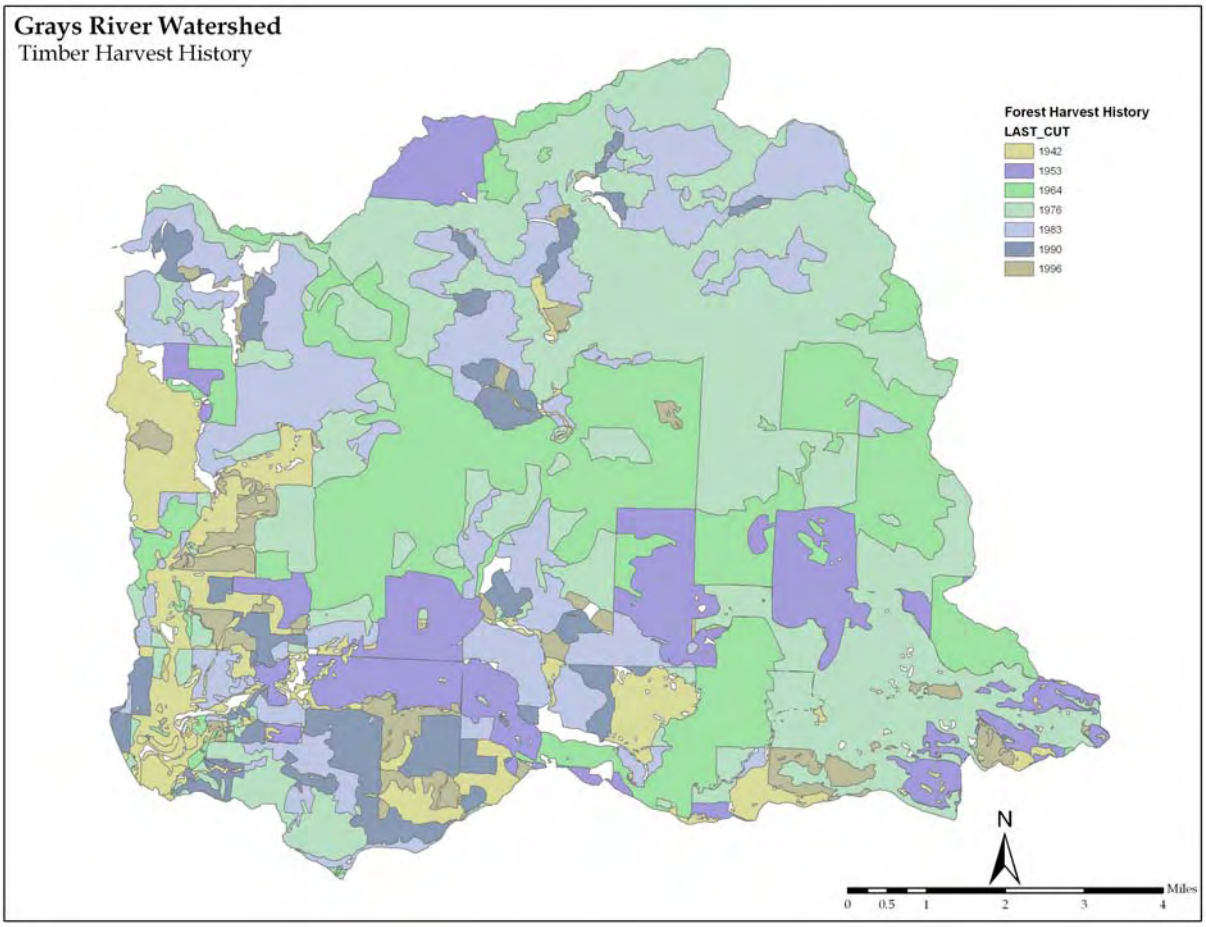

Figure S.4. Upper Grays River Watershed Timber Harvest History

Grays River riparian areas were delineated using a GIS at 30-m and 100-m linear scales based on field surveys and GIS analysis of landscape-scale data. All subbasin riparian areas were classified as either *impaired* or *moderately impaired*. Riparian areas of the upper Grays River watershed are composed almost entirely of deciduous-dominated forest (mainly alder and maple). In most subbasins, the riparian zone has been logged such that only a fraction of the native coniferous forest cover remains. Fragmentation of the riparian corridor by road crossings was found to be impaired (>1 road crossing per kilometer of stream length). Although the number of stream-road crossings is high within the watershed, relatively few fish passage barriers are present within the upper Grays River watershed.

S.2 Hydrologic Assessment

A hydrologic analysis was conducted to evaluate impacts of harvest timing, intensity, and location on runoff and streamflow.

S.2.1 Methods

The 43-year historical precipitation record available for the Grays River was used as input data for the Distributed Hydrology Soil Vegetation Model (DHSVM). Land-cover scenarios were generated based on timber harvest scenarios from 1942 to 1996 and included a baseline of pristine, mature forest conditions. The DHSVM was used in a scenario method, with each scenario consisting of one land-cover state held constant through the entire precipitation record. Thirteen model output locations were selected throughout the watershed. Effects of timber harvest on duration of peak flow, summer low flow, and spawning season (November-December) low flow were examined.

The DHSVM provides a dynamic representation of watershed processes at the spatial scale described by digital elevation model (DEM) data (typically 10–90 m horizontal resolution). The modeled landscape is divided into computational grid cells centered on DEM nodes. This characterization of topography is used to model topographic controls on absorbed shortwave radiation, precipitation, air temperature, and downslope moisture redistribution. Vegetation characteristics and soil properties are assigned to each model grid cell and may vary spatially throughout the basin. In each grid cell, the modeled unit is composed of a column of vegetation and soil. At each time step, the model provides simultaneous solutions to energy and water balance equations for every grid cell in the watershed. Individual grid cells are hydrologically linked through surface and subsurface flow routing. The model predicts temporal and spatial changes in hydrological properties such as snow cover and depth, soil moisture, ground saturation, soil temperature, subsurface water movement, recharge to deep groundwater, surface runoff, and simplified one-dimensional channel flow.

Exceedance probability plots of daily streamflow simulated by the DHSVM for each of the nine vegetation scenarios were created at each of the 13 assessment locations. Cumulative density functions (CDF) of daily streamflow simulated by the DHSVM were estimated at each of the hydrological assessment locations individually using a Gaussian-kernel–based nonparametric density estimation technique. Streamflows corresponding to specific frequency of exceedances were then “looked-up” from these CDFs. The exceedance probability plot for the lowermost site in the project area (State Route 4) is presented as an example (Figure S.5).

S.2.2 Timber Harvest History in the Grays River Watershed

Prior to 1942, approximately 89% of the upper Grays River watershed was covered by mature conifers and approximately 11% was covered with conifers of 65% of full height. Harvest during that year was along the West Fork and the South Fork. By 1953, timber harvesting had progressed to the headwaters of West Fork, North Fork, and South Fork. Areas along the Grays River mainstem below the confluence of the East and South Forks also were harvested. By 1964, much of the watershed was harvested, mainly the central areas, with approximately 54% of the watershed covered by conifers 12 m or less in height and with 23% of the watershed in clear-cuts. Area in fully mature timber cover in 1964 was 1.82%, the lowest in the watershed’s history. The most severe watershed-wide impact to vegetation height and LAI (a regrowing overstory leaf-area index, the ratio of overstory leaves divided by the land

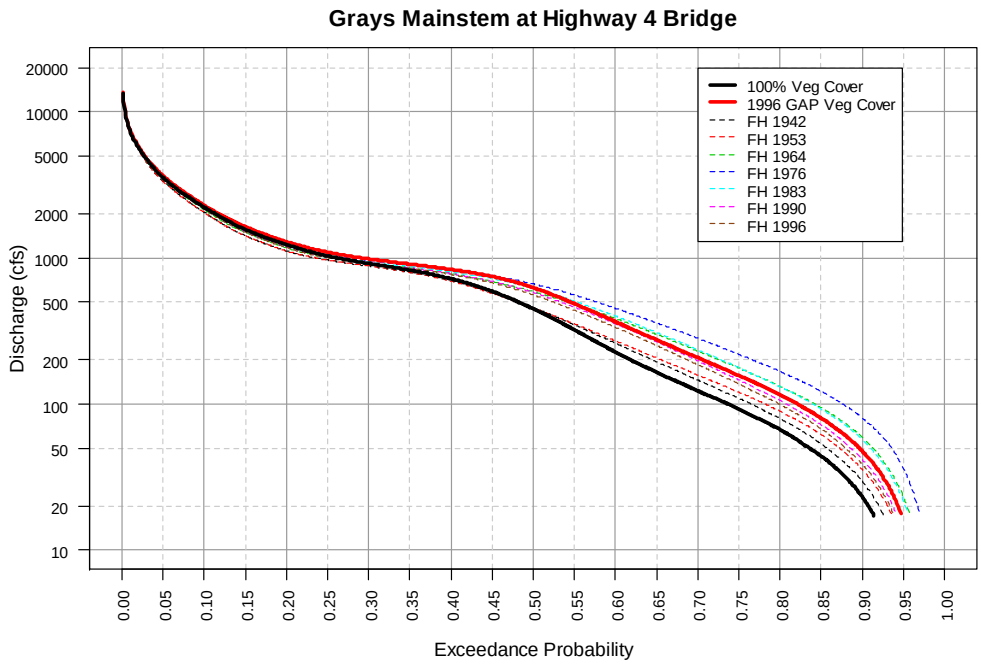

Figure S.5. Exceedance Probability Plot for Daily Streamflow for Grays Mainstem at State Route 4 Bridge

surface area) was in 1976, when the pattern of harvesting over the entire watershed continued. By 1976, more than 75% of the watershed was covered with conifers 12 m or less in height, with 28% in clear-cuts (Figure S.6). Since 1976, the watershed vegetation cover has recovered somewhat, with the fully mature area at 17% in 1990 and 13.6% in 1996. Clear-cuts have reduced in area to about 3.9%. By 1996, approximately 72% of the watershed was covered by conifers exceeding 15 m in height.

S.2.3 Effects of Timber Harvest on the Annual Hydrologic Regime

Relative difference in daily streamflow corresponding to different exceedance probabilities for each of the nine vegetation scenarios at all 13 assessment locations were computed by assuming the streamflow corresponding to the pre-harvest 100% fully mature vegetation cover as the baseline and expressing differences from this baseline in percentage. Changes in the hydrological response are more pronounced in the low flow region (50% or greater exceedance) than in the high flow region at all assessment locations (see Table S.1, Figure S.5 as an example).

The greatest change in streamflow response in the watershed was observed for the 1964 and 1976 vegetation scenarios, both of which were results of extensive harvesting throughout the watershed. Changes in 75% and 90% exceedance flows are considered minor. The change in streamflow response at all assessment points except Crazy Johnson was affected most for the 1976 vegetation scenario. Streamflow response varies among assessment points in response to differences in drainage area and the spatial and temporal distribution of timber harvest intensity.

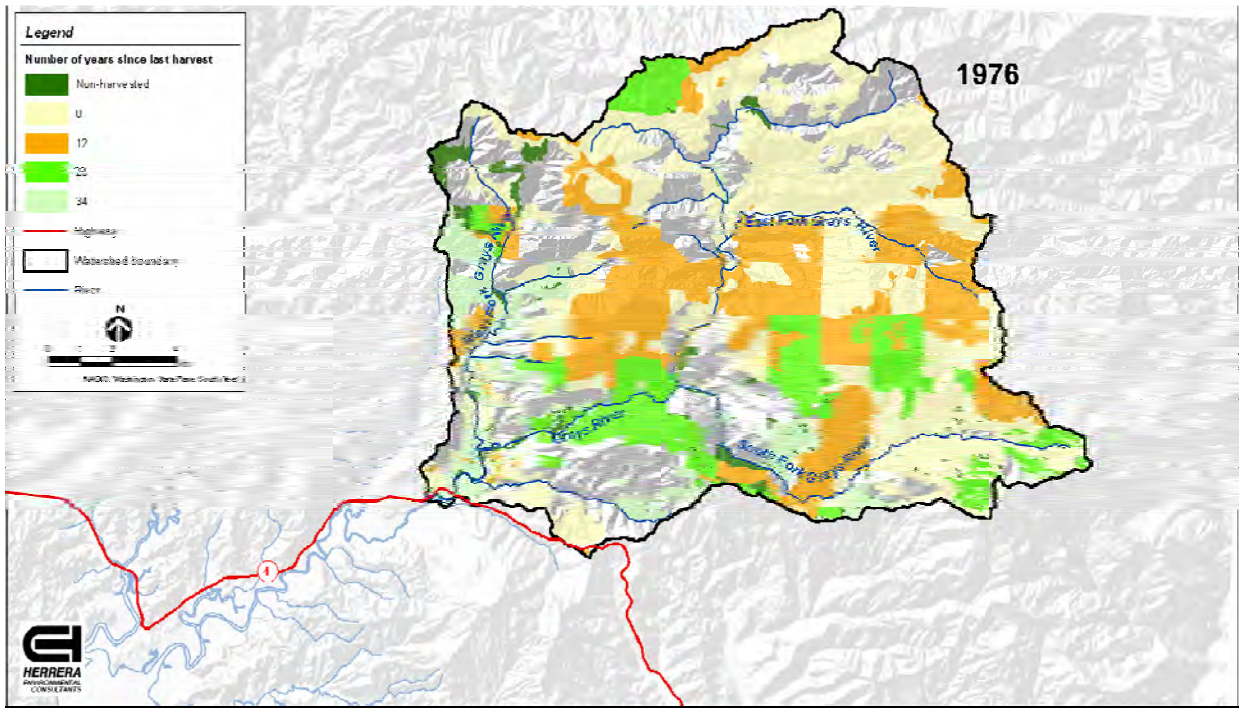

Figure S.6. Forest Harvest History for 1976

Table S.1. Relative Change in Percentage of Exceedance Flow in the Grays Mainstem at the State Route 4 Bridge

Vegetation Scenario	% Exceedance Flow				
	90	75	50	25	10
100%[(a)]	0	0	0	0	0
Gap Analysis[(b)]	106	70	39	6	3
FH1942[(c)]	28	19	2	-6	-7
FH1953[(c)]	54	30	1	-6	-8
FH1964[(c)]	156	91	31	-2	-4
FH1976[(c)]	247	137	48	2	1
FH1983[(c)]	147	93	38	0	-1
FH1990[(c)]	81	59	29	-2	-2
FH1996[(c)]	67	49	24	-3	-3

(a) A pre-harvest scenario with fully mature conifers covering the entire watershed.
(b) Gap Analysis-identified gaps in biodiversity protection that may be filled by establishing new preserves or changes in land-use practices.
(c) Spatial dataset of forest clearings (harvests) determined for a specific year.

S.2.4 Effects of Timber Harvest on Bankfull Flows

Bankfull flows were estimated with DHSVM-simulated streamflow using a 1.4-year return period. Return period of a discharge equal to the bankfull flow corresponding to the baseline vegetation scenario (100% vegetation cover) was estimated for each hydrological assessment location under the remaining vegetation scenarios. Change in the return period of bankfull flow ranged from a 0.11-year reduction corresponding to the 1976 vegetation scenario at East Fork to a 0.03-year increase corresponding to the 1942 forest harvest scenario for Fossil Creek. At all hydrological assessment points except Crazy Johnson, the largest change from baseline was that corresponding to the 1976 forest harvest scenario. The frequency of occurrence and duration of bankfull flows was also estimated. Timber harvest intensity had minimal effect on either of these hydrological parameters.

S.2.5 Effects of Timber Harvest on Low Flows

To analyze the relative change in low flows in the Grays River Basin, a peak under threshold approach was adopted to identify flow sequences in the daily-simulated streamflow series that fall below a threshold to construct a partial-duration series. The threshold for this analysis was set to the 90% exceedance flow (10th percentile daily mean discharge) under the baseline vegetation condition (100% vegetation cover) at each hydrological assessment location. Effects on low flow for the full annual flow record, summer low flows (July through September) and the spawning season (November-December) were examined.

Mean duration of low flows shows changes from a reduction of 67% from baseline at Crazy Johnson (location 13) corresponding to the 1964 forest harvest scenario to an increase of 27% from baseline at North Fork (location 12) corresponding to the 1996 forest harvest scenario (Table S.2). Greatest absolute change from baseline among all forest harvest scenarios corresponded to the 1976 forest harvest except at Crazy Johnson, which showed the greatest absolute change corresponding to the 1964 forest harvest scenario.

Table S.2. Relative Change in Percentage of Mean Duration of Low Flows at Selected Locations

Vegetation Scenario	Hydrological Assessment Location Numbers												
	1	2	3	4	5	6	7	8	9	10	11	12	13
100%[a]	0	0	0	0	0	0	0	0	0	0	0	0	0
Gap Analysis[b]	-51	-53	-49	-50	-51	-50	-50	-49	-51	-51	-53	-55	-52
FH1942[c]	19	19	16	19	20	19	22	18	20	17	25	27	-2
FH1953[c]	6	-7	7	9	8	4	1	5	6	2	5	-6	-14
FH1964[c]	-29	-17	-25	-29	-28	-29	-26	-23	-14	-19	-16	-32	-67
FH1976[c]	-56	-51	-24	-39	-60	-55	-60	-36	-37	-51	-35	-40	-28
FH1983[c]	-19	-12	-4	-21	-27	-22	-24	-15	-19	-22	-19	-16	-7
FH1990[c]	9	10	8	11	12	9	15	8	9	5	17	17	1
FH1996[c]	9	12	10	9	10	11	9	12	11	4	22	27	1

(a) A pre-harvest scenario with fully mature conifers covering the entire watershed.
(b) Gap Analysis-identified gaps in biodiversity protection that may be filled by establishing new preserves or changes in land-use practices.
(c) Spatial dataset of forest clearings (harvests) determined for a specific year.

Relative change in mean duration of summer low flows with the 100% vegetation cover scenario as the baseline is shown in Table S.3. Relative change in mean duration of summer low flows was small for the 1942 forest harvest scenario except at Crazy Johnson (location 13) and, to some extent, at hydrological assessment locations 3, 9, and 10. The greatest absolute changes occurred corresponding to the 1976 forest harvest scenario, except at Crazy Johnson, where it was corresponding to the 1964 forest harvest scenario.

Relative change in mean duration of spawning period low flows is shown in Table S.4. Change in mean duration of spawning period low flows varied from a reduction of 100% (no flows below the 90% exceedance mean daily flow corresponding to the 100% vegetation scenario) at hydrological assessment

Table S.3. Relative Change in Percentage of Mean Duration of Summer Low Flows at Selected Locations

Vegetation Scenario	Hydrological Assessment Location Numbers												
	1	2	3	4	5	6	7	8	9	10	11	12	13
100%[a]	0	0	0	0	0	0	0	0	0	0	0	0	0
Gap Analysis[b]	-59	-63	-59	-58	-59	-58	-59	-57	-59	-59	-62	-64	-58
FH1942[c]	0	-1	-7	-2	1	-2	3	-4	-6	-6	-2	-1	-16
FH1953[c]	-19	-34	-20	-19	-20	-22	-26	-18	-20	-23	-24	-37	-26
FH1964[c]	-48	-43	-49	-49	-51	-51	-49	-45	-40	-45	-46	-56	-71
FH1976[c]	-76	-89	-49	-71	-81	-75	-86	-68	-72	-81	-60	-61	-50
FH1983[c]	-48	-41	-42	-49	-52	-49	-51	-44	-48	-52	-46	-46	-23
FH1990[c]	-17	-18	-18	-16	-14	-16	-12	-16	-19	-22	-12	-12	-15
FH1996[c]	-16	-13	-16	-16	-15	-13	-16	-12	-16	-21	-6	-3	-14

(a) A pre-harvest scenario with fully mature conifers covering the entire watershed.
(b) Gap Analysis-identified gaps in biodiversity protection that may be filled by establishing new preserves or changes in land-use practices.
(c) Spatial dataset of forest clearings (harvests) determined for a specific year.

Table S.4. Relative Change in Percentage of Mean Duration of Spawning Period Low Flows at Selected Locations

Vegetation Scenario	Hydrological Assessment Location Numbers												
	1	2	3	4	5	6	7	8	9	10	11	12	13
100%[a]	0	0	0	0	0	0	0	0	0	0	0	0	0
Gap Analysis[b]	28	14	11	12	12	15	10	16	7	23	13	-13	-5
FH1942[c]	18	24	19	14	14	18	16	28	26	22	25	15	-6
FH1953[c]	11	16	6	10	16	6	11	-2	-2	0	35	13	18
FH1964[c]	22	61	53	18	18	44	51	23	23	15	1	9	-43
FH1976[c]	-78	-100	-23	-79	-79	-78	-100	-78	-78	-100	-77	-46	36
FH1983[c]	67	50	64	61	50	55	51	57	57	61	24	63	-12
FH1990[c]	14	5	-7	7	7	11	10	12	12	5	18	-4	-15
FH1996[c]	-3	9	-14	-8	-8	-6	-10	1	3	6	10	4	-5

(a) A pre-harvest scenario with fully mature conifers covering the entire watershed.
(b) Gap Analysis-identified gaps in biodiversity protection that may be filled by establishing new preserves or changes in land-use practices.
(c) Spatial dataset of forest clearings (harvests) determined for a specific year.

locations on Fossil Creek (location 2), South Fork (location 7), and East Fork (location 10) corresponding to the 1976 forest harvest scenario, to an increase of 67% at the SR 4 bridge (location 1) corresponding to the 1983 forest harvest scenario. The greatest absolute change for all hydrological assessment locations was corresponding to the 1976 forest-harvest scenario except for the West Fork (location 3) and North Fork (location 12), where it was corresponding to the 1983 forest harvest scenario, and Crazy Johnson (location 13), where it was corresponding to the 1964 forest harvest scenario.

S.2.6 Hydrologic Assessment Summary and Conclusions

- Different locations in the Grays River watershed respond differently to historical forest harvest events based on timber harvest severity, the hydrologic connectivity of individual watershed locations, and the spatial location of harvest events relative to areas contributing runoff to each assessment location.

- Maximum impacts on runoff production and streamflow would occur under the 1976 land-use scenario, the peak intensity of harvest activities. However, even under this land-use scenario, impacts to flood flows were minor.

- Timber harvest had the most significant effects on low-flow magnitude and duration. Timber harvest tended to decrease duration of low flows. For example, mean duration of low flows decreased by 27% to 67% across all locations and timber harvest scenarios. Maximum duration of low flows decreased by 8% to 90% from baseline. The impact of timber harvest on low flows decreased under land-use scenarios after 1976 associated with reduced harvest rates.

- Timber harvest decreased the maximum duration of spawning period low flows in chum and fall Chinook salmon spawning areas by 18% to 94% from baseline. The mean duration of spawning period low flows in chum and fall Chinook salmon spawning areas ranged from an increase of 67% to a decrease of 79% from baseline.

- Impacts of timber harvest on bankfull flood return interval and mean duration were relatively minor.

- Effects of roads associated with timber harvest on peak flows were not examined in this study. Roads can have significant effects on peak flows.

S.3 Geomorphic Assessment

The geomorphic analysis examined the effects of historical and current land-use practices on erosion and sedimentation within the channel network, the ways in which these effects have influenced the sediment budget of the upper watershed, resulting responses in the mainstem Grays River upstream of State Highway 4, and past and future implications for salmon habitat. The objective of this investigation was to assess how sediment production in the upper Grays River watershed and alterations to floodplains have affected the morphology of the channel network. The elements of this geomorphic investigation included 1) analysis of mass wasting and surface erosion, 2) construction of a sediment budget, 3) geomorphic characterization of the channel network, and 4) analysis of channel sensitivity and response potential.

S.3.1 Upland Sediment Production

The assessment of current and historical geomorphic conditions within the upper Grays River watershed was conducted at both the watershed scale and reach scale using standard geomorphic assessment methods. Historical trends in sediment production from hillslopes were reconstructed at the watershed scale from Grays River timber harvest records, historical aerial photographs, and previous studies of natural background erosion rates for undisturbed forested basins. Aerial photographs were used also to evaluate the reach-scale response of alluvial valley segments to historical increases in sediment loading related to land-use practices. Historical records and aerial photographs were augmented with field investigations and data collection. Sediment generation within the Grays River watershed was evaluated also in terms of the dominant geomorphic processes observed during the field reconnaissance of the watershed and from aerial photographic analysis of principal sediment source terrains. Geomorphic processes responsible for sediment delivery to the channel network included in this analysis were classified as mass-wasting events (landslides), surface erosion from disturbed hillslopes, road-surface erosion, and soil creep. Records of historical land-use activities were used to reconstruct a historical sediment budget for the watershed, which is followed by a description of current channel conditions and channel response to increased sediment inputs.

An inventory of historical landslides in the upper Grays River watershed was compiled from 1970, 1996, and 2003 aerial photographs. A total of 216 landslides were identified in the historical photographs. Landslides were classified as channelized debris flows, shallow translational slides, or deep-seated rotational slides. Debris flows and translational slides were observed to be the most common landslide forms in the upper Grays River watershed. A GIS-based analysis of potential debris-flow initiation at road crossings was performed by comparing the road network with the headwater channel network in the upper Grays River watershed (Figure S.7).

Surface erosion of landslide areas was evaluated based on hillslope gradient, geology, and forest-harvest characteristics. Relatively minor surface erosion was observed in recently clear-cut areas during the field reconnaissance surveys. However, significant surface gully erosion was observed on road embankments and on the surfaces of recent landslide areas. In all cases, gullies were located below culverts that apparently discharge concentrated flow onto bare hillslope surfaces. Because timber entrained in landslides limits gully development on large landslide deposits and gully erosion tends to develop relatively soon if a landslide occurs on slopes where timber has been removed (e.g., clear-cut hillslopes), gully development observed in the upper Grays River watershed was interpreted to be at least partially the result of timber removal on hillslopes. Gully incision of large rotational landslides tends to accelerate the delivery of stored sediment to the channel network. The gullies observed on the surface of the large rotational landslides surveyed in the upper Grays River watershed suggest that the numerous historical landslides in the watershed may be a chronic source of sediment, particularly where runoff from roads is concentrated onto landslide surfaces.

Annual rates of sediment yield from mass wasting (Table S.5), average delivery ratios (ratio of sediment contributed to channels to sediment mobilized by landslides on the hillslope), and the lag time between timber harvest and the peak in mass wasting were estimated. Average delivery ratios for rotational, translational, and debris flow landslide types were 0.25, 0.43, and 0.87, respectively. The frequency distribution of the historical landslide inventory indicates that 72% of the landslides occurred between 10 and 20 years after timber harvest. Based on the landslide inventory data and results from

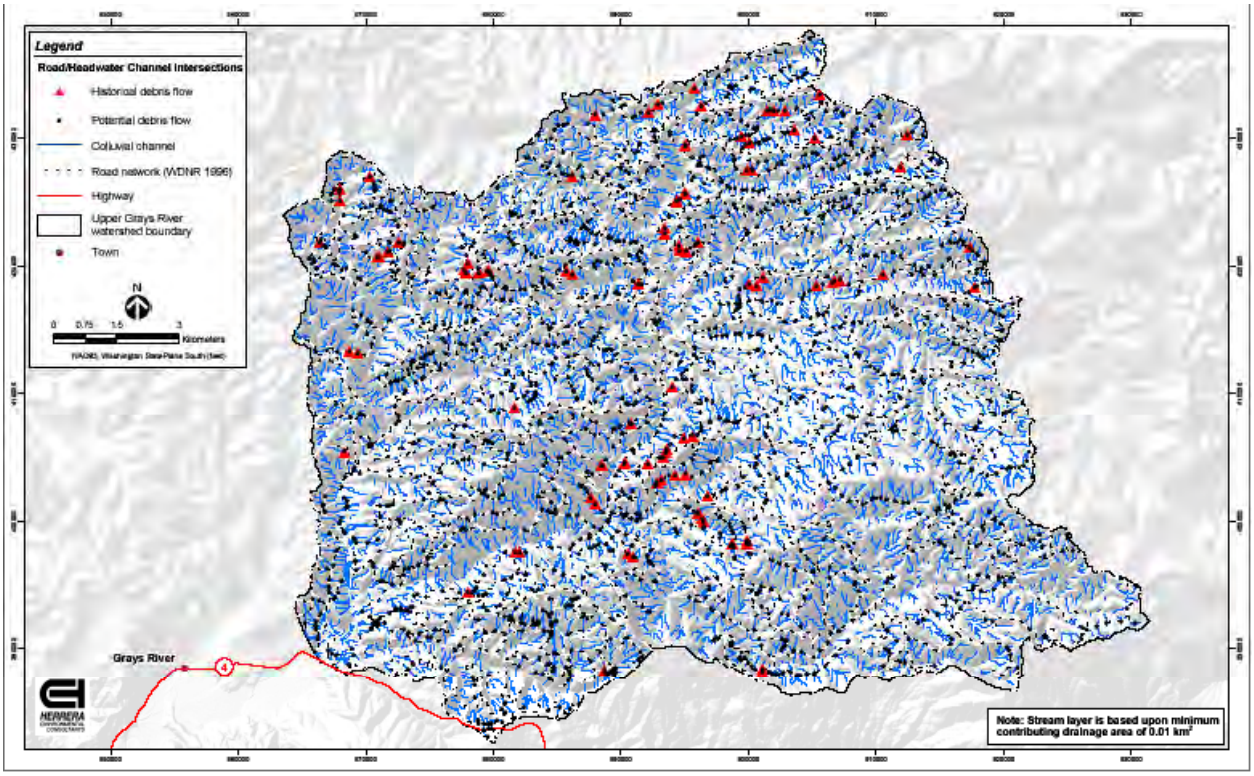

Figure S.7. Locations of Historical and Potential Debris Flow Initiation Based on GIS-Derived Intersections of Roads and Colluvial Channels in the Upper Grays River Watershed, Washington

Table S.5. Historical Sediment Yield to the Upper Grays River Watershed from Landslides

Aerial Photograph Year	Sediment Yield (tons/year)
1970	209,000
1996	221,000
2003	238,000
Sediment yield for 1970 was extrapolated for the entire watershed area due to limited aerial coverage (76% of watershed).	

previous studies, it is anticipated that mass wasting should peak approximately 10 years after timber harvest and deliver the majority of landslide-derived sediment to the channel network over the period between 10 and 30 years after timber harvest.

Results of the upper Grays River watershed landslide inventory indicate that roads also contributed significantly to historical mass wasting, either directly through slope modifications or indirectly by concentrating storm runoff on unstable slopes. Roughly half of all landslide areas delineated from the aerial photographs studied were found to intersect roads. In addition, visual inspection of the aerial photographs during delineation found that approximately three-fourths of all landslides were associated with roads.

Erodibility ratings were generated for the upper Grays River watershed by combining the GIS coverage-layers for soil erodibility factor (K) and slope. Approximately 37% of the Grays River watershed is rated with high soil erosion potential, 9% of the watershed is rated with a moderate soil erosion potential, and the remaining 54% of the watershed is rated as low for soil erosion potential (Figure S.8). The majority of current timber harvest activities are located in areas with relatively high erosion potential.

S.3.2 Upper Grays River Watershed Sediment Budget

Historical and contemporary (2003) sediment budgets were constructed for the upper Grays River watershed (Table S.6). Historical sediment budgets were constructed for each of the eight reference years in which timber harvest data are available (see Hydrologic Assessment). Background sediment yield is the long-term average sediment yield from a basin under natural geomorphic conditions prior to anthropogenic disturbance. Background sediment yield was determined from field measures, GIS analyses, and literature estimates. Under background conditions, the primary geomorphic processes that generate sediment in mountain watersheds are soil creep and mass wasting. Current sediment yield was estimated for mass wasting, roads, and soil creep.

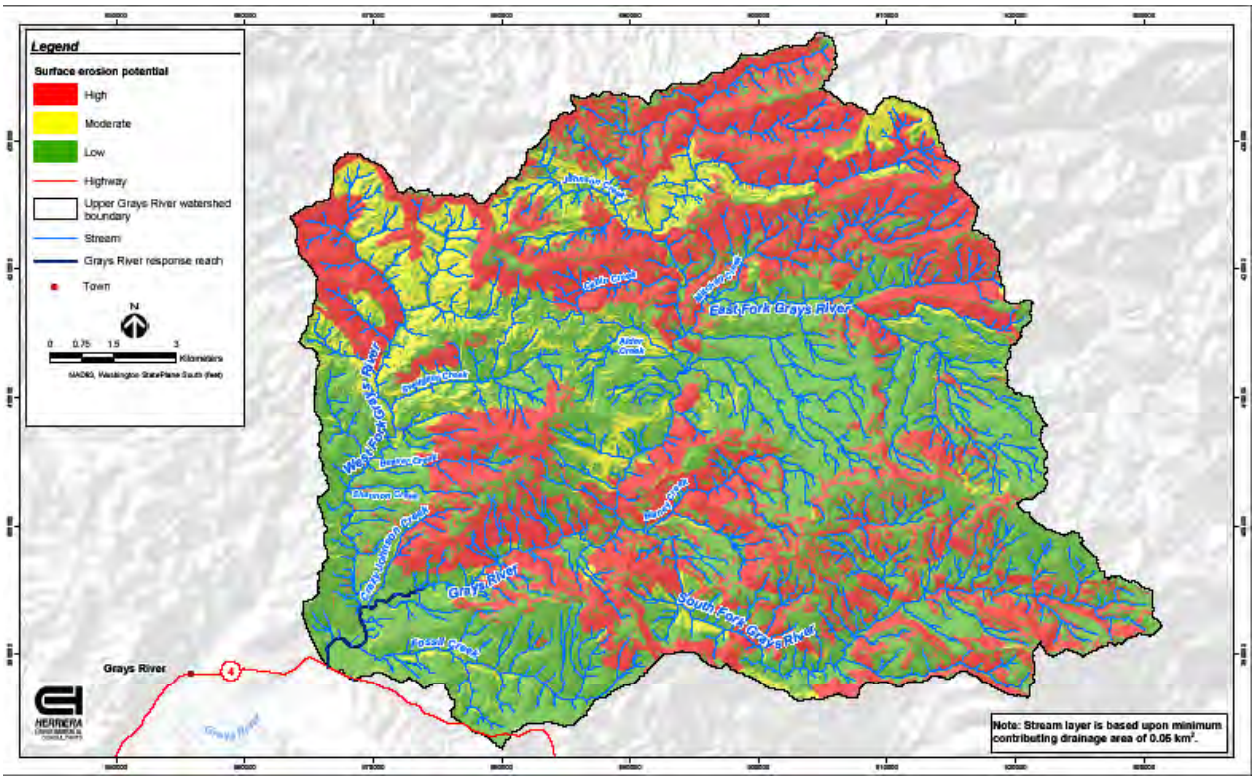

Figure S.8. Ground Surface Erosion Potential Based on Whole Soil Erosion Factors and Slope in the Upper Grays River Watershed

Table S.6. Average Sediment Yield in the Upper Grays River Watershed Under Current (2003) Conditions and Long-Term Background Conditions

Sediment Source	Sediment-Yield (tons/km^2/year)
2003 Mass wasting	1,032
2003 Roads	41
2003 Soil creep	26
2003 Total	1,099
Long-term background (range)	56–125

Reconstruction of the sediment budget indicates that the total sediment yield for the upper Grays River watershed was nearly constant from 1942 through 1964 and ranged from approximately 13,000 to 18,000 tons/year. The estimated sediment yield increased sharply to 87,000 tons/year by 1976 and continued to climb at an unprecedented rate to more than 250,000 tons/year by 2003 (Figure S.9a). Although the total sediment yield for the watershed has increased every year, local sediment yield from some subbasins declined between 1996 and 2003. The majority of the sediment delivered to the channel network is derived from mass-wasting processes brought about by road construction and use associated with timber harvest operations (Figure S.9b). Erosion from roads and soil creep contribute a relatively minor proportion of the total sediment yield. Sediment yield to the channel network is projected to reach a maximum of approximately 290,000 tons/year in 2005 and decline to approximately 24,000 tons/year by 2025 but remain approximately 85% higher than the background erosion rate of 13,000 tons/year predicted by the analysis.

Based on prior studies of bedload attrition, most of the coarse sediment delivered to the upper Grays River channel network by mass-wasting processes is likely converted to suspended load after moving relatively short distances (1 to 2 km) through the channel network, and certainly by the time sediment reaches the mainstem (Gorley) response reach.

Given the current timber harvest rotation in the upper Grays River watershed of about 30-40 years, sediment production will likely remain relatively high and ultimately will also likely result in significant long-term upland soil loss and possible reduced timber productivity.

S.3.3 Geomorphic Characterization of the Channel Network

The ratio between transport capacity and sediment supply influences numerous characteristics of a stream channel. Change in this ratio as a result of increased sediment supply in the Grays River is expected to increase channel width, scour depth, slope, and sediment storage, and decrease channel depth, grain size, roughness, and sinuousity. Change in the availability of LWD can also affect channel structure and function, and can convert a response reach to a transport reach. A reduction in the number and effectiveness of response reaches would result in greater sediment discharge to the lower mainstem of the Grays River. A process-based channel classification was applied to the upper Grays River watershed to predict the distribution of geomorphic channel types and identify transport versus response reaches.

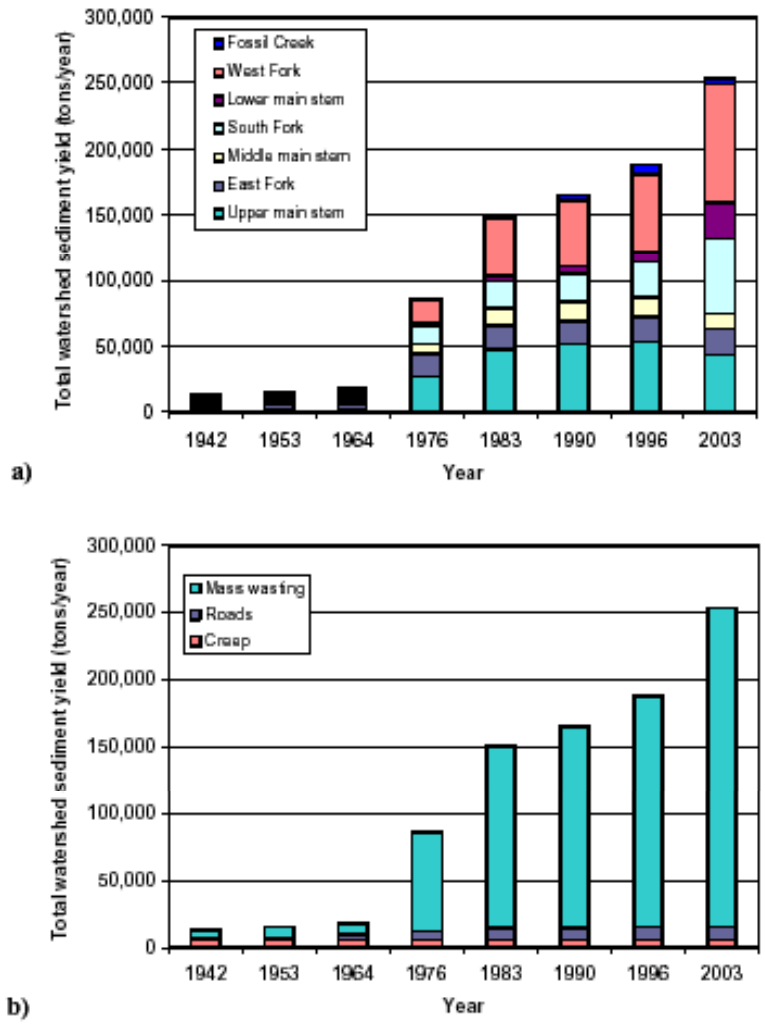

Figure S.9. Predicted Temporal Variation in Total Upper Grays River Watershed Sediment Yield by (a) Subbasin Group and (b) Source

Transport capacity was quantified throughout the entire Grays River channel network in terms of an index of unit stream power (Figure S.10). Unit stream power (i.e., transport capacity) decreases downstream through the channel network, with the lowest values found in the low-gradient alluvial reaches of the mainstem Grays River and major tributaries.

To assess how increases in sediment production have affected channel morphology and the extent of channel change, a field reconnaissance of select reaches in the upper Grays River watershed was completed. Channel surveys documented evidence of sediment storage, routing, and channel response and included descriptions of the channel morphology, condition of instream sediment, the character and extent of LWD loading, and the recruitment potential of adjacent upper Grays River watershed.

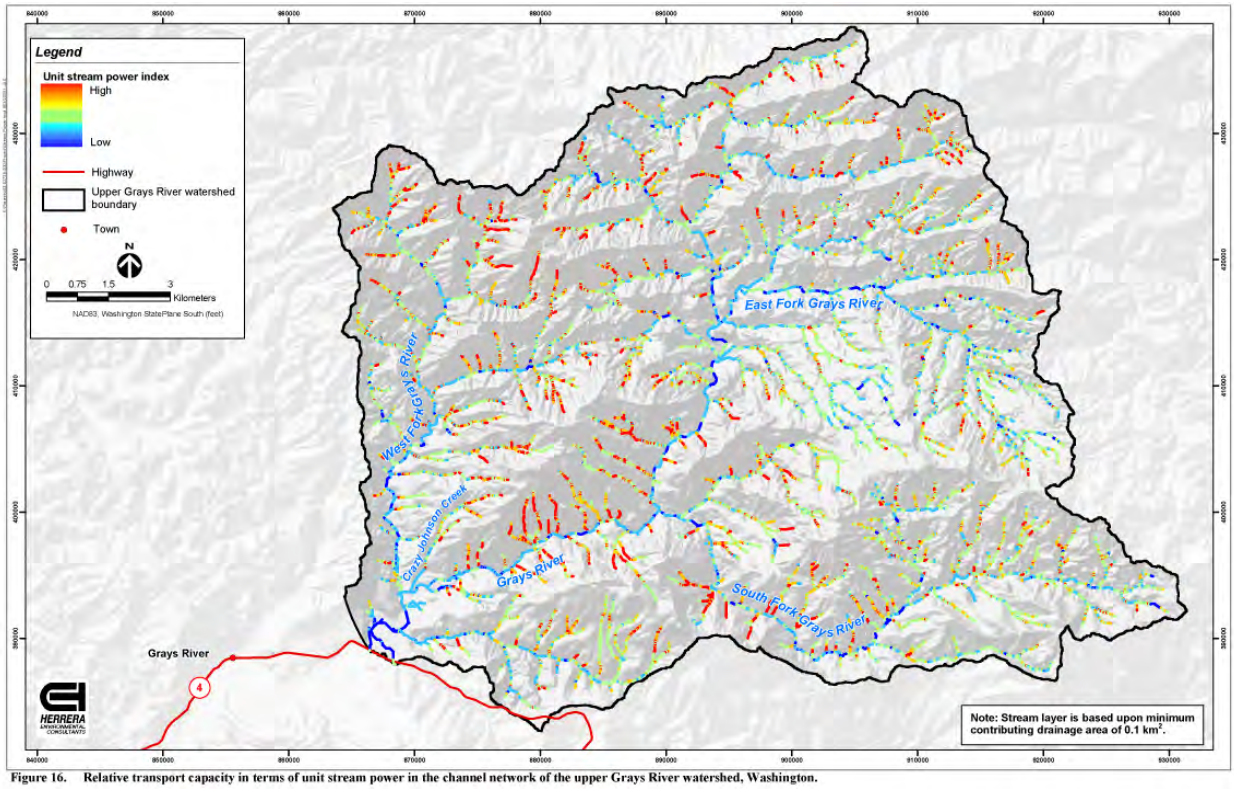

Figure 16. Relative transport capacity in terms of unit stream power in the channel network of the upper Grays River watershed, Washington.

Figure S.10. Relative Transport Capacity in Terms of Unit Stream Power in the Channel Network of the Upper Grays River Watershed, Washington

Historical analysis of channel position focused on the response reach segments of the mainstem Grays River, delineated from digital geo-referenced aerial photographs obtained from the Washington Geospatial Data Archive (WA-GDA; Figure S.11). Two forms of channel migration were identified on the Grays River, lateral migration and avulsion. The geomorphic response of the mainstem Grays River was evaluated quantitatively in terms of temporal changes in reach-scale meander geometry (i.e., sinuosity, meander wavelength, radius of curvature, and migration rates), as well as qualitatively through observations of changes in vegetation patterns and channel response. The 1999 channel avulsion in the lower Grays Gorley reach appears to have been triggered by the cumulative effects of an increased sediment influx, channel confinement by levees, and the mobilization of stored sediment by above-average precipitation in the years preceding the avulsion.

Both the rate and direction of lateral channel migration varied considerably during the period of historical record. Reach-average lateral migration rates were greatest in the Gorley sub-reach of the mainstem, where lateral migration rates varied from 0.72 to 5.13 m/year and the highest erosion rates were measured between 1996 and 2003 (Table S.7). Reach-average migration rates were relatively lower in the West Fork and South Fork response reaches, ranging from 0.64 m/year in the West Fork to 1.35 m/year in the South Fork.

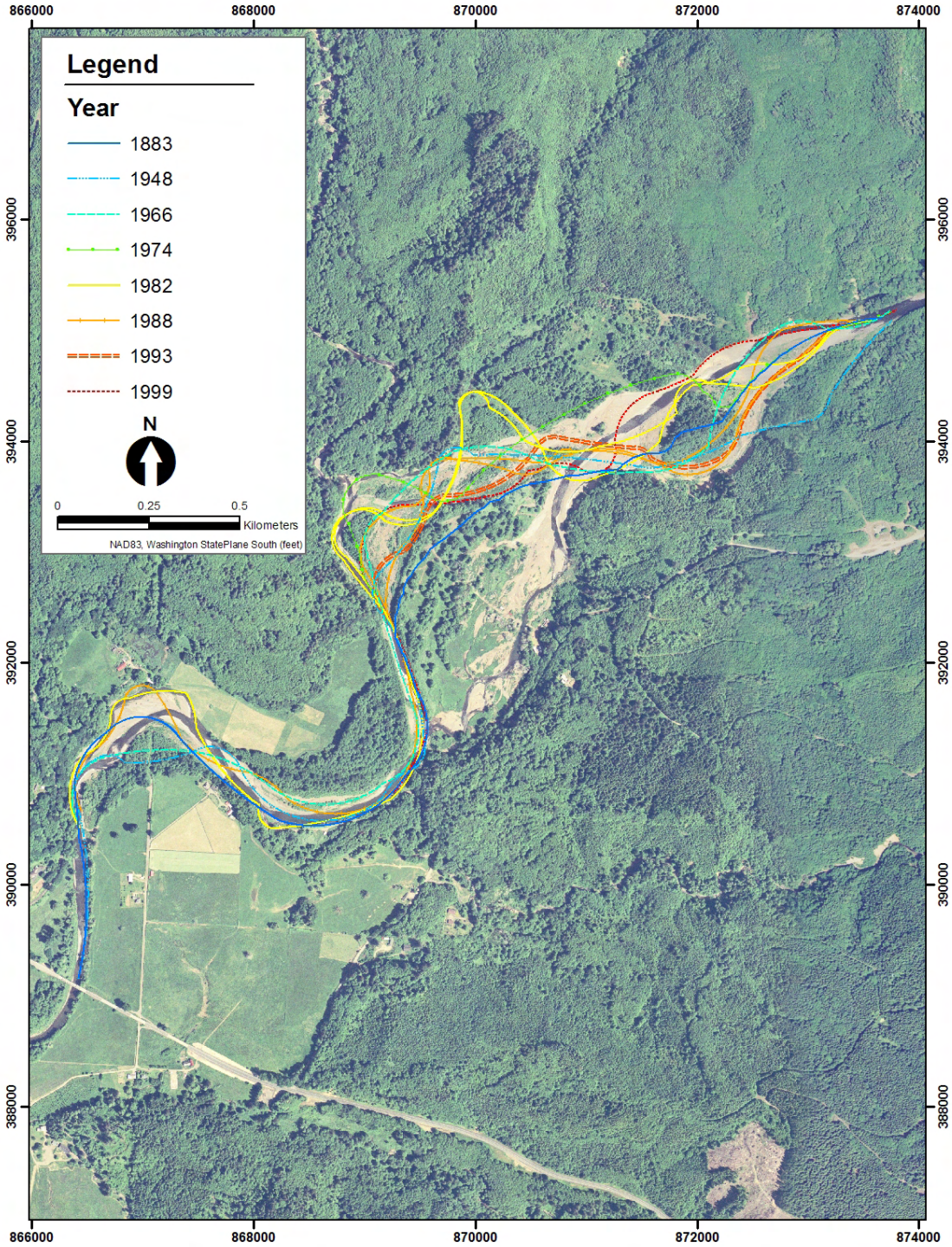

Figure S.11. Historical Locations for 1883–1999, Grays River Response Reach

Table S.7. Erosion Rates and Lateral Channel Migration Rates for the Mainstem, West Fork, and South Fork Grays River (Herrera 2005)

Reach/Sub-Reach	Photographic Period	Erosion Rate (m^2/year)	Lateral Migration Rate (m/year)
SR 4 sub-reach	1939–1966	394	0.17
	1966–1960	3,858	1.71
	1970–1982	3,488	1.46
	1982–1996	3,978	1.57
	1996–2003	8,684	3.43
Gorley sub-reach	1939–1966	1,709	0.72
	1966–1960	3,364	1.38
	1970–1982	3,291	1.29
	1982–1996	6,691	2.71
	1996–2003	11,292	5.13
West Fork response reach	1970–1996	1,595	1.06
	1996–2003	553	0.64
South Fork response reach	1970–1996	3,041	1.35
	1996–2003	1,253	1.01

Observations from the field studies and analytical efforts can be generalized into a conceptual model of the temporal relations among watershed disturbance, sediment yield, and channel response within the upper Grays River watershed (Figure S.12). Sediment yield to the channel network increases sharply above natural background levels shortly after the onset of timber harvest activities and reaches a peak that lags behind the peak harvest rate by approximately 25 years. Added to this period is the 10-year lag between harvest and peak in landslide frequency. This results in the overall 35-year lag time between timber harvest and channel response observed in the Grays River. The magnitude of channel change tends to increase with time due to the mobilization of sediment stored within the channel network during high-magnitude storm events. This reduction in sediment storage is often magnified by the removal of instream LWD and harvest of riparian forests that would otherwise supply LWD to the channel network. Under this conceptual model and the inherent uncertainties involved with predicting sediment production and transport, channel adjustment could continue for at least 50 years after present timber harvest activities.

S.3.4 Geomorphic Assessment Summary and Conclusions

- The upper Grays River watershed is composed of relatively erosive soils and is susceptible to mass-wasting events when vegetation is disturbed or removed. Relatively steep, confined channels that efficiently route sediment to the broad alluvial valley of the lower Grays River dominates the upper Grays River stream network.

- Current sediment production in the watershed is approximately an order of magnitude greater than the background erosion rate typical of forested watersheds in the coastal region.

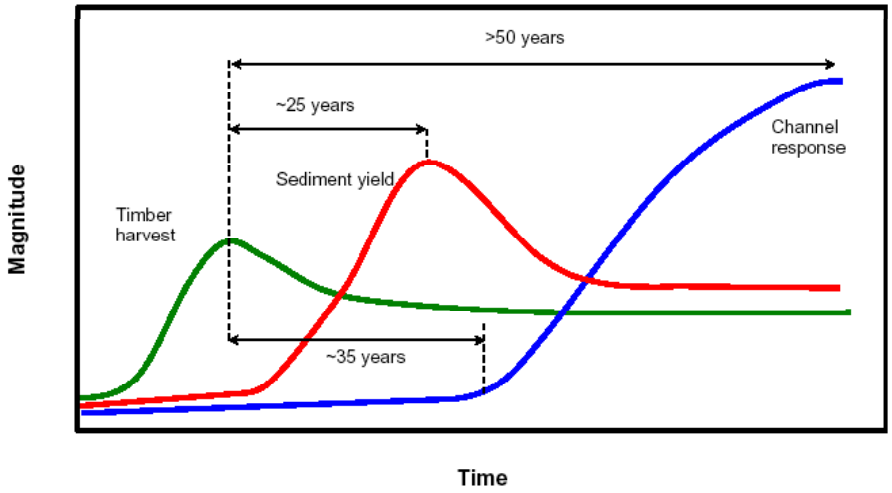

Figure S.12. Conceptual Model of Temporal Trends in Sediment Yield to the Upper Grays River Channel Network and Channel Response within the Grays River Response Reach to an Assumed Historical Timber Harvest

- Sediment production (mass wasting and surface erosion) lags approximately 10 to 30 years behind forest clearing (timber harvest). Because the maximum rate of forest harvest occurred between 1976 and 1983, the peak in sediment production may not have yet occurred.

- Headwater stream reaches that could moderate downstream sediment flux contain high sediment loads, limited channel complexity, and a significant lack of sediment storage potential. This lack of instream sediment storage is due mainly to an absence of instream large woody debris (LWD) and the loss of LWD recruitment resulting from degraded riparian forest conditions. Restoration of riparian forests and instream LWD recruitment potential is a critical action to support long-term ecosystem health.

- The large increase in sediment supply has had the most noticeable impact in the Grays River Gorley reach upstream of State Route 4 where much of the sediment from the upper watershed is deposited.

- Channel response in the Grays River lags behind forest harvest by approximately 30 to 50 years. Channel response has been influenced by dike and levee construction intended to control channel migration and flooding. Artificial confinement and high sediment loads combined to create unstable conditions that led to the catastrophic 1999 channel avulsion.

- Without significant efforts to implement more sustainable land management practices, it is likely that excess sediment delivery, instability of the lower river channel, and detrimental impacts on habitat and property will continue. If historical levels of timber harvest are not significantly reduced, soil loss may severely reduce the long-term productivity of the upper watershed.

S.4 Aquatic Integrity

S.4.1 Methods

A biological assessment was conducted using the benthic index of biological integrity (BIBI), a multimetric index based on attributes of the benthic invertebrate community. The standard BIBI is composed of 10 metrics of taxa richness and diversity, population attributes, disturbance tolerance, and feeding and other habits. Separate evaluation criteria for streams and rivers were used. A total of 16 sites were evaluated throughout the watershed using both the stream and river criteria.

S.4.2 Summary and Conclusions

Using the stream criteria, that 75% of sites had fair to very poor biological integrity, with only 25% of sites having good integrity and no sites with excellent integrity (Figure S.13). Using the river criteria, all sites were rated as fair to very poor, with 56% fair; no sites were found to have good or excellent biological integrity. Results suggest that the cumulative effects of human land-use activities in the watershed have compromised the biological integrity of the Grays River aquatic ecosystem.

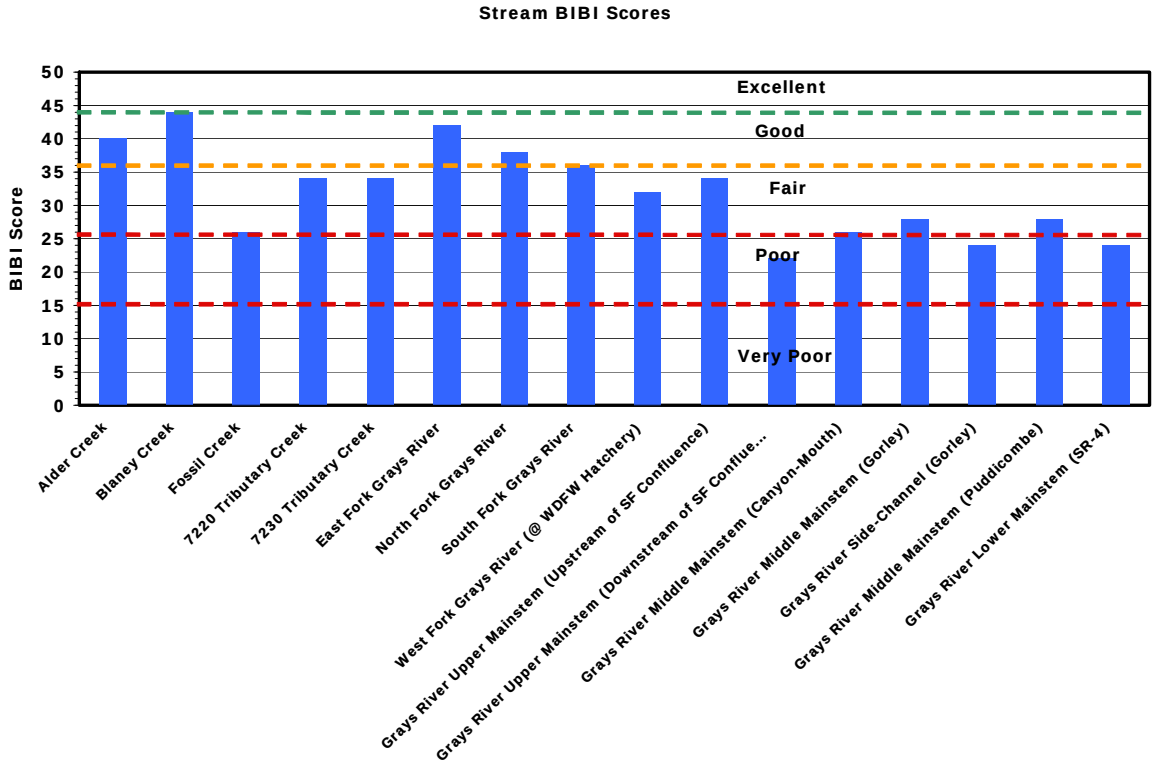

Figure S.13. 2004 Stream BIBI Scores for Grays River

S.5 Salmon Spawning Habitat

A salmon spawning analysis was conducted to evaluate the potential and actual spawning habitat use of the Grays River study area by chum and fall Chinook salmon. Potential spawning habitat availability was evaluated at multiple flows to characterize habitat quality and estimated redd capacity of spawning reaches. Chum salmon spawning consistency from 2002 through 2005 was examined and contrasted spawning and non-spawning velocity, depth, and substrate habitat characteristics of chum and fall Chinook salmon as well as hyporheic and geomorphic characteristics of chum salmon.

Salmon spawning habitat analyses focused on chum salmon spawning areas in the Grays River, West Fork, and Crazy Johnson Creek immediately upstream of SR 4. This area comprises the majority of fall Chinook salmon spawning area in the Grays River as well.

S.5.1 Potential Spawning Habitat Availability

S.5.1.1 Methods

To identify areas suitable for chum and fall Chinook salmon in the Grays River, spawning limits were developed based on an extensive review of existing literature and a focused limits analysis of spawning habitat use in the study area during 2002 through 2005. Two sets of limits were developed—those within which chum and fall Chinook salmon have been documented to spawn and, within these limits, a more restrictive range in which these species were most likely to spawn (probable use). The outer spawning use limits were used to quantify total potential spawning area for each species. The more restrictive probable use limits were used to distinguish potential high- versus low-quality habitat, where quality scores range from 0.0 (at least one habitat variable is beyond outer use limits) to 2.0 (all three habitat variables are within the probable use range). Available spawning area was identified using modeled velocity and depth estimates and empirical substrate and hyporheic temperature data. Velocity, depth, and substrate were used to delimit spawning from non-spawning area. Hyporheic temperature was also used to delimit chum salmon spawning area because literature sources confirm the importance of hyporheic exchange to spawning chum salmon. From available habitat area, the potential redd capacity for each species was quantified. Current active channels as well as the channel abandoned during the 1999 avulsion and associated floodplain areas also were evaluated across a range of flows characteristic of spawning conditions in these streams.

Velocity and depth estimates were produced by the Modular Aquatic Simulation System Two-Dimensional (MASS2) hydrodynamic model, a depth-averaged hydrodynamic and transport model for river and coastal systems. Velocity and depth estimates were produced by MASS2 for the 90, 50, and 10% exceedance daily mean discharges for the chum salmon spawning season (November 1–December 31). These discharge values were produced by the DHSVM described in the hydrological assessment. Substrate grain size and hyporheic and river water temperature data were collected from all reaches of the study area. The d_{50} (the grain size in millimeters of which 50% of particles are finer) metric was used to describe substrate particle size. The temperature of the hyporheic environment was characterized as the difference between river and hyporheic temperatures ($\Delta T = T_H - T_R$). Minimum and maximum redd capacity was estimated by dividing available spawning habitat by redd size with two estimates of inter-redd spacing.

S.5.1.2 Spawning Habitat Availability

Of the total wetted area, 61.6% (12.22 ha) is within spawning limits for both species, 27.9% (5.54 ha) is within only chum salmon limits, 2.7% (0.54 ha) is within only fall Chinook salmon limits, and 7.8% (1.55 ha) is beyond the spawning limits for both species (Table S.8). Total spawning area available for chum and fall Chinook salmon is 17.76 ha (89.5%) and 12.75 ha (64.2%), respectively. Velocity, depth, and substrate do not appear to be limiting to chum salmon, with only 6.3, 4.4, and 0.0% of the total wetted area beyond use limits, respectively. ΔT was more limiting to chum salmon spawning habitat availability than were velocity, depth, or substrate. Total available chum salmon spawning area based on all four habitat variables is 5.88 ha, or 29.6% of the wetted area. Spawning habitat is more limited by velocity, depth, and substrate for fall Chinook salmon than for chum salmon, with 11.6, 7.6, and 25.4%, respectively, beyond use limits.

Table S.8. Spawning Area Availability for Chum and Fall Chinook Salmon by Stream Section Based on Velocity, Depth, and Substrate

Stream Reach	Total Area	Non-Spawning		Chum Only		Fall Chinook Only		Both Species	
	ha	ha	%	ha	%	ha	%	ha	%
Total	19.85	1.55	7.8	5.54	27.9	0.54	2.7	12.22	61.6
Crazy Johnson	0.92	0.12	12.6	0.76	82.5	0.00	0.0	0.05	4.9
Upper West Fork	3.18	0.16	5.2	0.64	20.2	0.11	3.4	2.26	71.2
Lower West Fork	2.29	0.23	10.1	0.16	7.1	0.02	1.1	1.87	81.7
Upper Grays	4.10	0.63	15.3	1.69	41.1	0.19	4.7	1.59	38.8
Lower Grays	9.36	0.41	4.4	2.29	24.5	0.21	2.2	6.45	68.9

S.5.1.3 Spawning Habitat Quality

Of the wetted area suitable for chum salmon spawning based on all four habitat characteristics, 12.4% (0.73 ha) is of the highest quality (Figure S.14). Habitat quality patterns for fall Chinook salmon were similar to those for chum salmon. Of the total area suitable for fall Chinook salmon spawning based on velocity, depth, and substrate, 26.1% (3.33 ha) is of the highest quality (Figure S.15).

S.5.1.4 Effects of Flow on Habitat Availability

Of the wetted area available, 89.7% (13.12 ha), 88.9% (17.76 ha) and 70.3% (19.10 ha) of the total wetted area was suitable for chum salmon spawning based on velocity, depth, and substrate at the 90, 50, and 10% exceedance flows, respectively. Spawning area increased in all reaches from the 90% to the 50% exceedance flow and increased in all reaches except the upper and lower Grays from the 50% to the 10% exceedance flow. When restricted by ΔT, 31.3% (4.58 ha), 29.6% (5.88 ha), and 21.1% (5.75 ha) of the total wetted area was suitable for chum salmon spawning at the 90, 50, and 10 % exceedance flows, respectively (Table S.9).

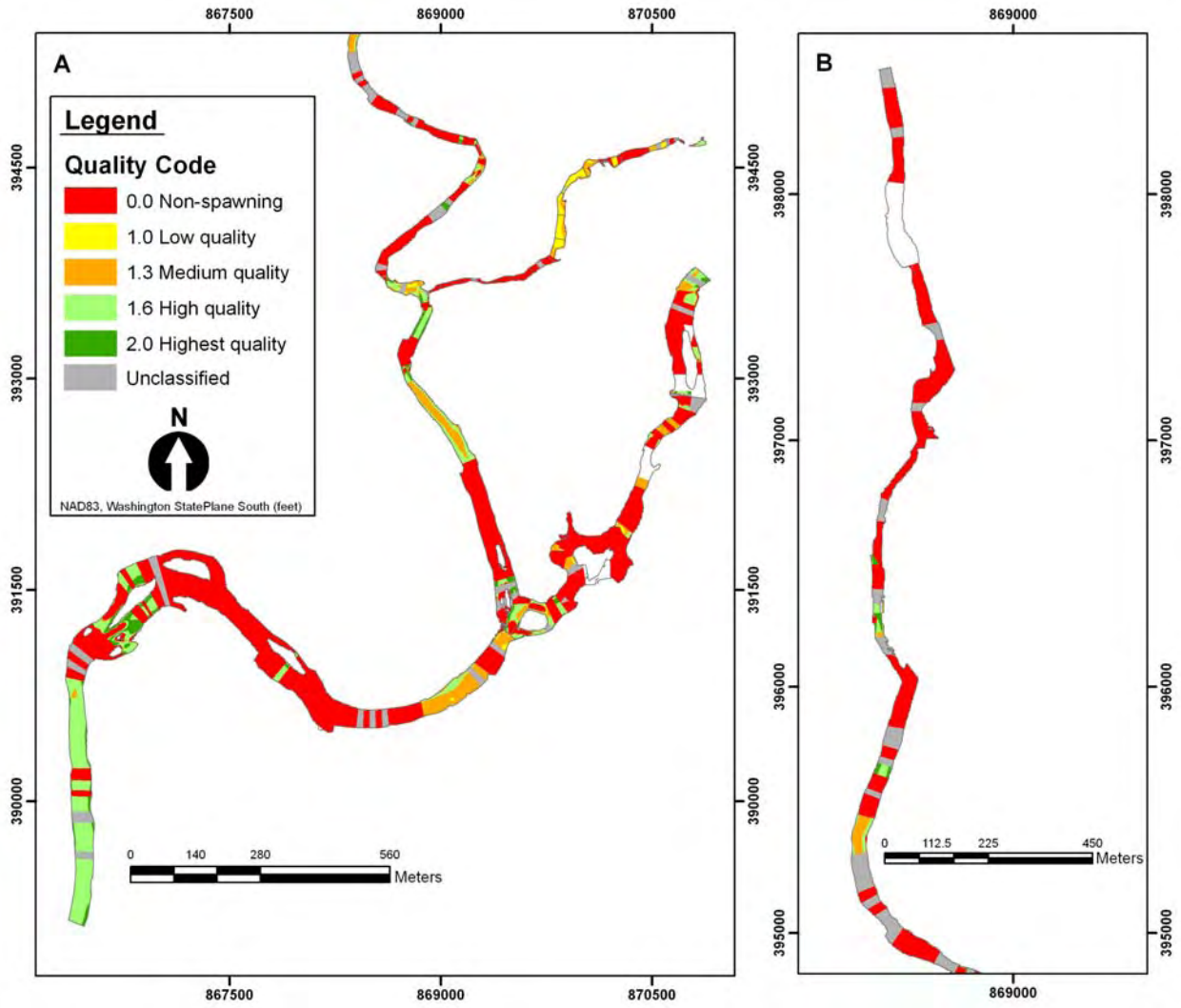

Figure S.14. Habitat Quality of Chum Salmon Spawning Areas Based on Velocity, Depth, Substrate and Hyporheic Temperature in A) the Study Area Including the Lower West Fork and B) the Upper West Fork

At the 90, 50, and 10 % exceedance flows, 54.5% (7.99 ha), 64.2% (12.75 ha), and 60.7% (16.50 ha) of the wetted area was suitable for fall Chinook salmon spawning based on velocity, depth, and substrate (Table S.9). The spatial distribution of suitable spawning areas was similar among all flows, with general areas of suitability or unsuitability increasing laterally across the channel.

S.5.1.5 Redd Capacity

Total chum salmon redd capacity estimates for the study area based on velocity, depth, and substrate range from 16,924 to 67,824 redds. Redd capacity in highest-quality habitat based on these variables ranges from 3,105 to 12,442 redds. Limiting available habitat by ΔT greatly reduces potential redd capacity. Using all four spawning habitat variables, total redd capacity and redd capacity in highest-quality habitat range from 5,590 to 22,405 redds and from 695 to 2,786 redds, respectively (Table S.10).

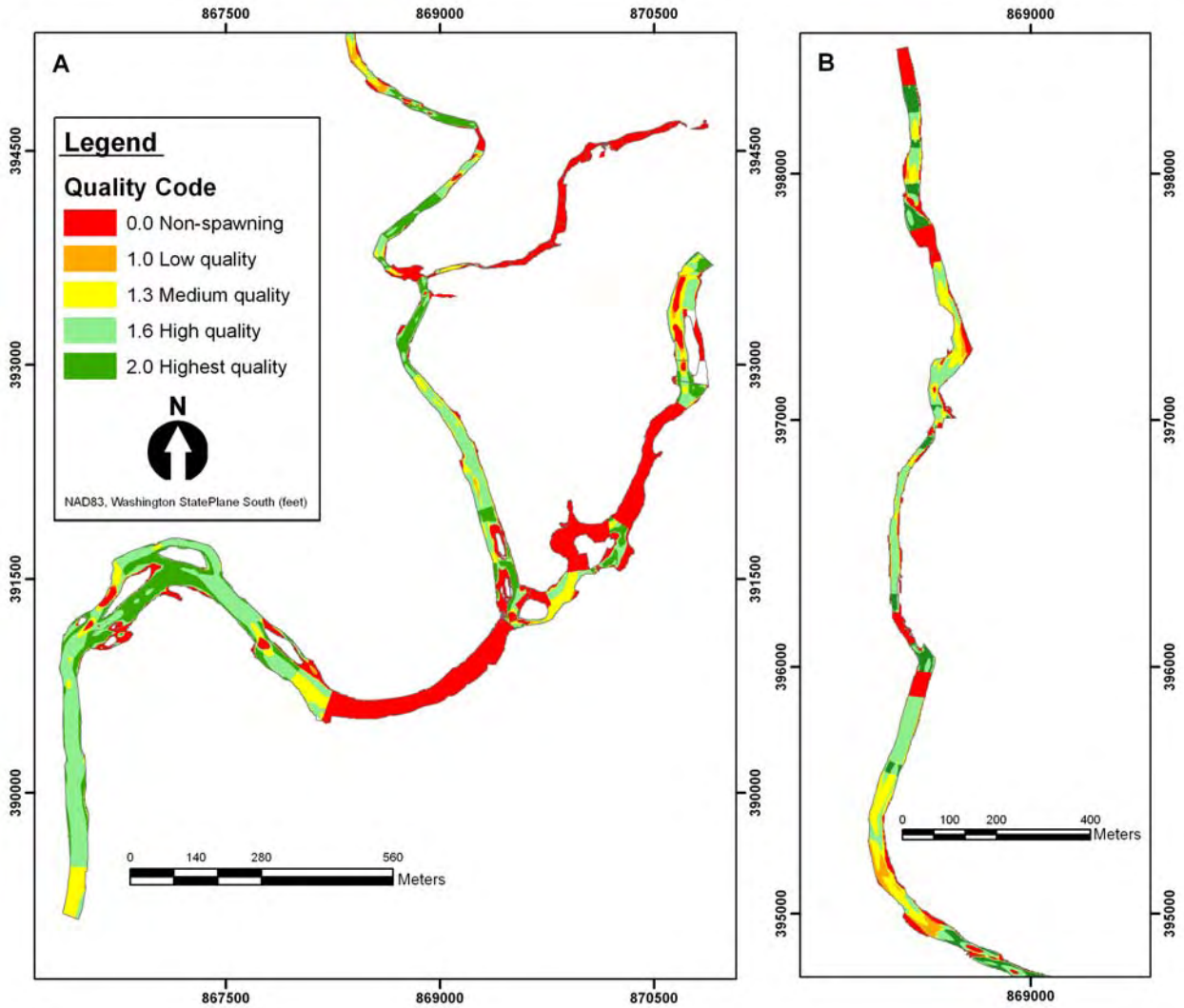

Figure S.15. Quality of Fall Chinook Salmon Spawning Habitat Based on Velocity, Depth, and Substrate in A) the Study Area Including the Lower West Fork and B) the Upper West Fork

Table S.9. Effect of Flow on Chum and Fall Chinook Salmon Spawning Habitat Availability. Chum salmon spawning habitat is based on substrate, velocity, depth, and ΔT; fall Chinook salmon spawning habitat is based only on substrate, velocity, and depth.

Species	% Exceedance					
	90		50		10	
	ha	%	ha	%	ha	%
Chum	4.58	31.3	5.88	29.6	5.75	21.1
Fall Chinook	7.99	54.5	12.75	64.2	16.50	60.7

Table S.10. Range of Chum and Fall Chinook Salmon Redd Capacities (number of redds). Redd capacities were generated from two redd area estimates.

Stream Reach	Chum		Fall Chinook	
	All	Highest Quality[(a)]	All	Highest Quality[(a)]
Total	5,590–22,405	695–2,786	4,904–19,615	1,281–5,123
Crazy Johnson	400–1,603	0–0	15–62	1–3
Upper West Fork	571–2,290	105–420	912–3,646	246–983
Lower West Fork	790–3,168	124–496	727–2,908	200–801
Upper Grays	638–2,557	48–191	685–2,738	133–533
Lower Grays	3,190–12,786	419–1,679	2,558–10,231	702–2,809
(a) Highest quality = quality score = 2.0.				

Total fall Chinook salmon redd capacity estimates for the study area based on velocity, depth, and substrate range from 4,904 to 19,615 redds (Table S.10). Redd capacity in highest-quality habitat ranges from 1,281 to 5,123 redds.

S.5.2 Spawning Habitat Use

S.5.2.1 Methods

Chum and fall Chinook salmon spawning was quantified during 2002 through 2005 by the Pacific States Marine Fisheries Commission (2002–2004) and the Washington Department of Fish and Wildlife (2005). In 2004 and 2005, each study reach was divided into 50-m-long sampling sections and numbers of spawning chum salmon were recorded weekly. Fall Chinook salmon redds were mapped during one week each in 2003 and 2005.

To examine chum salmon consistency of use, a spawning use index was developed because chum salmon spawning occurrence data were collected using different methods among years. Sampling sections were rated as 0 (no use), 1 (moderate use), or 2 (heavy use) for each year and then averaged across years. The hypothesis that spawning was randomly distributed among sampling sections throughout the study area among years (i.e., there was no spatial consistency in spawning among years) was tested by conducting a Monte Carlo simulation of expected average spawning scores under a random distribution across the study area.

To identify habitat differences between chum salmon spawning and non-spawning sections, each 50-m sampling section was classified as spawning or non-spawning for 2004 and 2005. Single variable comparisons (t-test or the non-parametric equivalent) were used to identify differences in habitat variables. A forward stepwise linear regression analysis was also conducted of the four-year average spawning use score described above as an overall summary of spawning relationships with habitat variables. Differences in velocity, depth, substrate, and hyporheic temperature characteristics between fall Chinook salmon redd locations and randomly selected non-redd locations were examined using t-tests as well.

Collection of velocity, depth, substrate, and hyporheic temperature data were as described above. In addition, vertical hydraulic gradient, hydraulic conductivity (K), and specific discharge (q) were measured in each sampling section. From the channel bathymetry, riffle:pool bedforms were identified,

channel bed slope, proximity to the nearest riffle, and width to depth ratio were characterized to examine differences between chum salmon spawning and non-spawning sections.

Across all four years, the observed and expected average spawning use score frequency distributions were significantly different ($\chi^2 = 76.3$, N = 8, $P < 0.005$). Spawning was more consistent than expected, with more sections having no spawning or more spawning than expected and fewer sections than expected of the predicted most likely spawning occurrence score under a random distribution (Figure S.16).

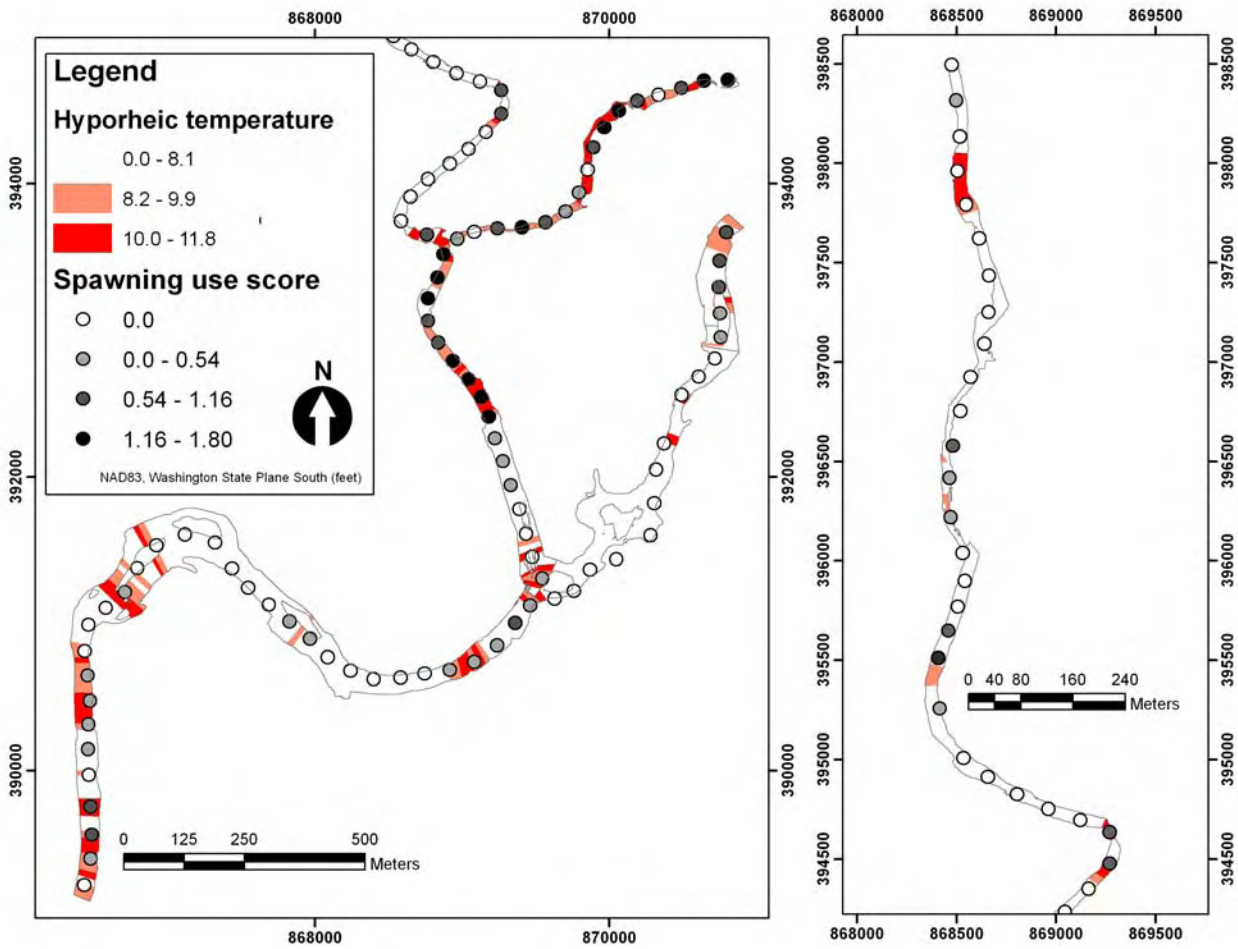

Figure S.16. Hyporheic Temperature and Chum Salmon Four-Year Average Spawning Use Score Across the Grays River Study Area. Possible spawning score values range from 0.0 (no use) to 2.0 (highest use).

S.5.2.2 Habitat Characteristics of Spawning Areas

In 2004 and 2005, tests of individual habitat variables across all reaches combined indicated that spawning and non-spawning sections differed in hyporheic temperature, with average hyporheic temperature in spawning areas 0.6°C and 1.2°C warmer, respectively, than non-spawning areas (Figure S.16). In 2005, average VHG was 0.22 cm/cm higher in spawning than in non-spawning sections; VHG did not differ in 2004. Velocity, depth, and substrate, hydraulic conductivity and specific discharge did not differ

in either year. In the stepwise linear regression of the four-year average spawning use score with velocity, depth, substrate, and hyporheic temperature, higher hyporheic temperatures and larger substrates were associated with more consistent use. Chum salmon spawning habitat use varied somewhat among reaches. For example, during both years, spawning areas were significantly shallower in the Upper Grays and deeper in the Lower West Fork than non-spawning areas. Also, VHG was significantly higher in spawning sections in the Lower Grays in both years, but not in any other reach.

From 0.5 to 3.5 riffle:pool bedforms were identified among study reaches. Riffle:pool spacing ranged from 10.2 (Lower Grays) to 34.1 (Lower West Fork) channel widths. Riffle:pool bedform spacing of 5 to 7 channel widths has been widely documented across streams of varying size and channel type, with values of 0.2 to 3 in channels with naturally high levels of large woody debris. Overall, spawning areas tended to be on descending slopes, but otherwise, relationships between spawning areas and geomorphic variables were complex and varied among reaches (Figure S.17). Spawning sections tended to occur in the areas characterized by negative slopes closer to pool bottoms (53.9%, two-year average) or closer to riffle crests (38.4%, two-year average). Location of spawning sections relative to bedform features varied among reaches.

Fall Chinook salmon redd locations were identified only in the Upper West Fork and Lower Grays. When all redd sites were combined, no differences between fall Chinook salmon redd and non-redd locations were detected in velocity, depth, substrate, or hyporheic temperature. Redd locations in the Lower Grays had significantly higher velocities than non-redd sites.

S.5.3 Salmon Spawning Assessment Summary and Conclusions

- The majority of aquatic habitat in the Grays River study area appears suitable for chum and fall Chinook salmon spawning based on velocity, depth, and substrate. However, much of the available habitat may be of relatively poor quality.

- Area with elevated hyporheic temperature appears to be more limiting to available chum salmon spawning habitat than are velocity, depth, or substrate.

- Available spawning area appears to be sufficient to support historical chum and fall Chinook salmon populations in the Grays River study area. It was estimated that chum and fall Chinook salmon redd capacity ranges from 5,590 to 22,405 redds and 4,904 to 19,615 redds, respectively, based on suitability for velocity, depth, substrate, and hyporheic temperature (chum salmon only).

- Chum salmon spawning areas tended to be located on negative (descending) slopes, closer to pool bottom bedforms than to riffle crests, and in channels with lower width-to-depth ratios.

- Bedform associations with chum salmon spawning in the Grays River study area are complex and variable among study reaches. Geomorphologic factors at broader scales than riffle-to-pool bedforms likely contribute significantly to hyporheic exchange within Grays River chum salmon spawning areas.

- Spawning habitat quality rather than habitat quantity may limit chum and fall Chinook salmon production and recovery in the Grays River.

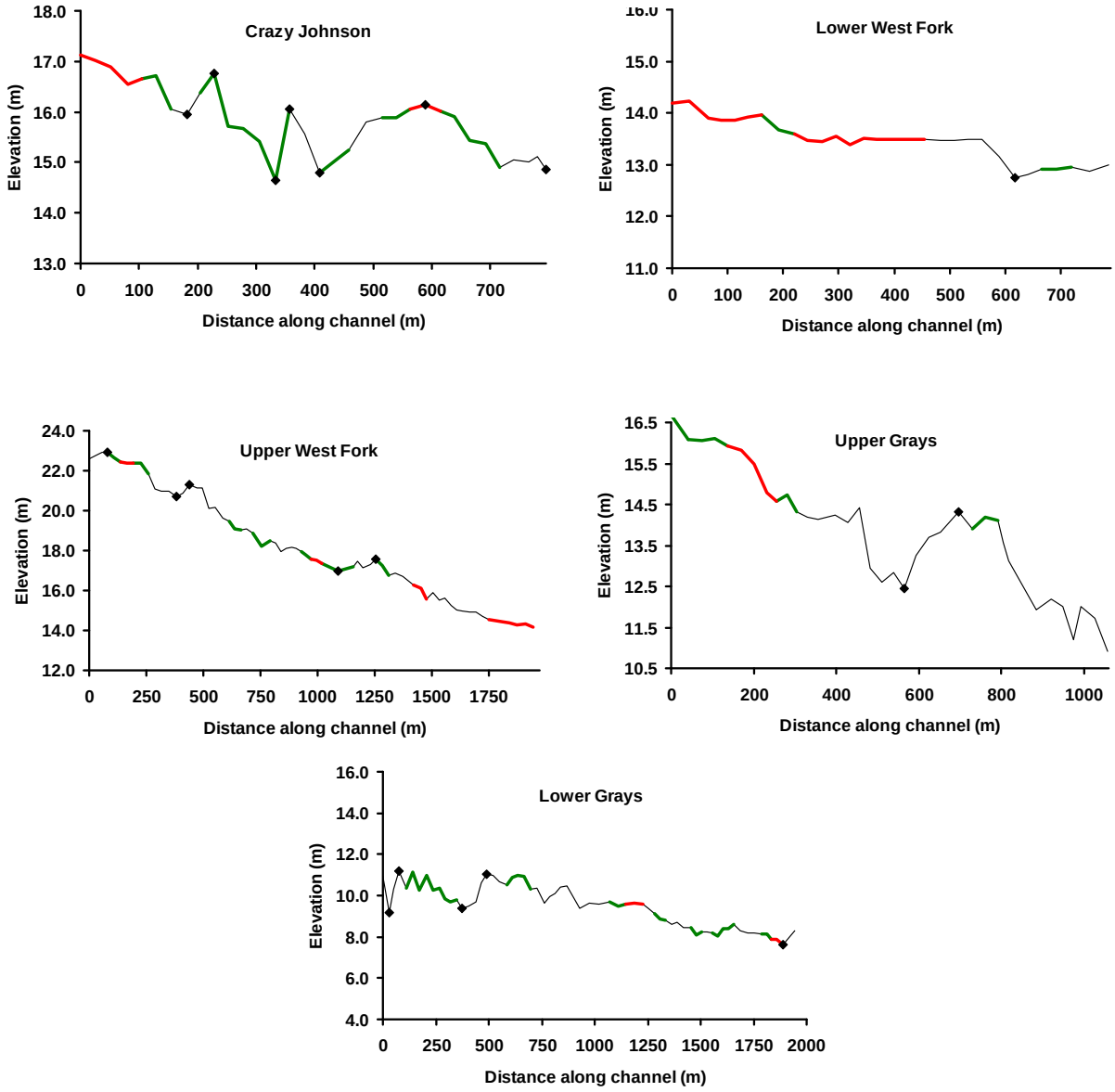

Figure S.17. Longitudinal Plots of A) Crazy Johnson Creek, B) Upper West Fork, C) Lower West Fork, D) Upper Grays, and E) Lower Grays Showing Sections Used for Chum Salmon Spawning in One (green) or Both (red) Years and Riffle Crest and Pool Bottom Bedforms (diamonds). The proportion of channel length to elevation change is held constant across all plots.

S.6 Integration of Study Findings

S.6.1 Effects on Habitat Forming Processes

A conceptual model that describes general relationships between natural and human land-use induced landscape characteristics, habitat-forming processes, and aquatic responses (see Figure S.1) was used to guide the Grays River watershed assessment. Assessment and restoration of land-use impacts on aquatic

resources require that differences be examined between historical and current watershed characteristics, identifying effects of land use on the critical habitat forming processes, hydrology, sediment delivery, and large woody debris (LWD).

Due to the inherent nature of its soils and hillslopes, the Grays River watershed is prone to erosion where vegetation has been removed and has the potential to yield large quantities of sediment. In the past, dense forests mantled the watershed and moderated both the production of sediment from hillslopes and the routing of sediment through the channel network. LWD in stream channels recruited from forested riparian areas trapped in-channel sediment, stabilized stream channels, moderated stream temperatures, and contributed to bed and habitat diversity and system productivity.

Changes in the hydrologic regime of the Grays River watershed associated with timber harvest in the upper watershed were relatively minor. In contrast, dramatic changes in delivery of sediment and LWD to stream channels have occurred in response to timber harvest and logging roads in the upper watershed as well as agricultural practices lower in the watershed. Localized effects of timber harvest and associated logging roads have increased the frequency and intensity of local mass wasting events and debris flows, which contribute to the sediment load delivered to the stream network. Although landslide-derived sediment is coarser than sediment derived from road surfaces, landslides delivered 20 to 200 times more sediment than roads during the 20 to 30 years following timber harvest and most of the coarser material is broken down to suspended load during fluvial transport.

Reconstruction of the sediment budget indicates that the total sediment yield for the upper Grays River watershed was nearly constant from 1942 through 1964 and ranged from approximately 13,000 to 18,000 tons/year. The estimated sediment yield increased sharply to approximately 87,000 tons/year by 1976 and continued to climb to more than 250,000 tons/year by 2003, indicating an order-of-magnitude increase in sediment production.

Hydrologic modeling in this study did not examine the effects of roads on streamflow. Road networks associated with timber harvest have been shown to have dramatic local effects on upper watershed drainages as well as moderated effects on flows lower within a watershed. The extensive road network has likely increased the magnitude of peak flows as well as altered their timing by intercepting shallow subsurface flow and moving it more rapidly to the channel system. In addition to direct effects on aquatic habitat and organisms, these effects may interact with changes in sediment and LWD availability as well. While this study did not document these effects, they are likely to be occurring within the Grays River watershed.

S.6.2 Effects on Channel Condition and Aquatic Habitat

The increased sediment production resulting from timber harvest and associated road construction has significantly affected downstream channel processes. A variety of channel responses have been observed, including channel aggradation and flooding. Deforestation and construction of floodplain levees and dikes within the Grays River watershed have impaired altered fluvial processes. Flood-control measures in lower reaches have resulted in a loss of floodplain connectivity and side-channel development, which have reduced channel habitat complexity and access to historical off-channel habitats. In addition, channel instability due to aggradation and riparian corridor clearing, loss of riparian function, and separation of the active Grays River channel from its floodplain have led to instream habitat degradation.

The loss of instream LWD and LWD recruitment potential from degraded riparian areas has had a significant impact on instream habitat quality, complexity, diversity, and channel geomorphology. The interaction of LWD, sediment, and water has profound effects on channel form and instream processes. Instream LWD provides for local sediment storage and transport capacity by increasing hydraulic roughness and capturing sediment behind channel-spanning LWD and logjams. Additional channel instability has resulted from the harvest of riparian forest vegetation and the loss of instream LWD.

Aggradation and natural straightening of the channel are typical morphological responses to an increase in sediment loading. The local increase in slope caused by continued aggradation (as well as confinement by levees) will shift the depositional front of the mainstem downstream. Confinement and straightening of the mainstem response reach by levees increase the local transport capacity (through increases in both slope and flow depth, which, in turn, increase shear stress) and shift future sediment deposition and channel response downstream to the SR 4 sub-reach. Additional channel avulsions are likely to occur if measures are not taken to maintain and raise existing levees and revetments concurrently with the anticipated sediment aggradation.

If channel response in the Grays River mainstem lags behind harvest by approximately 35 years as predicted by the geomorphic assessment, the current instability within the mainstem can be expected to continue and possibly increase through 2030. Response reaches throughout the upper Grays River watershed may be prone to continued instability and avulsions as long as sediment supply exceeds threshold transport rates. The remobilization and routing of sediment stored within the channel network may extend this response period through the latter half of the twenty-first century.

Results of the biological assessment suggest that cumulative effects of human land-use activities in the watershed have compromised the biological integrity of the Grays River aquatic ecosystem. These findings are consistent with existing habitat assessments suggesting aquatic habitat degradation throughout the Grays River watershed.

S.6.3 Effects on Salmonid Spawning

Most of the aquatic habitat in the Grays River study area was found to be suitable for chum and fall Chinook salmon spawning based on velocity, depth, substrate, and hyporheic temperature (chum salmon only) and potential redd capacity appears to be sufficient to support historical runs of these species. Decline of Grays River chum and fall Chinook salmon therefore may be a function of spawning habitat quality rather than quantity. The geomorphic assessment identified significant increases in sediment delivery and decreases LWD availability associated with timber harvest in the upper Grays River watershed. These changes have had profound effects on aquatic habitat throughout the watershed and in particular in the Grays River response reach. This reach is a primary chum salmon spawning area in the watershed and is located at a break (decrease) in channel-bed slope between a steep canyon outlet and the tidally influenced lower river. As an unconfined response reach, this area is prone to excess sediment deposition and is particularly sensitive to increases in sediment supply. Artificial channel confinement, low quantities of instream LWD, elevated fine sediment loading, and degraded riparian-floodplain conditions exacerbate this problem and collectively lead to reduced habitat diversity and stability. Increased sediment load and reduced in-channel LWD availability are widespread through the watershed and include all chum and fall Chinook salmon spawning areas. Thus, upper watershed timber harvest and lower river agricultural activities have resulted in altered habitat-forming processes and degraded aquatic habitat in all chum and fall Chinook salmon spawning areas in the Grays River.

The most significant limiting factors to salmon within the upper watershed and lower Grays River are likely substrate instability and excessive fine sediments. Elevated sediment loads and the lack of instream LWD throughout the watershed, especially within the response reach, have produced unstable sediment-laden channels lacking structure and diversity associated with LWD that do not fully support native salmon populations. Elevated sediment loads and lack of instream LWD contribute to reduced incubation success through redd scour and reduced water quality, respectively.

A third significant factor that may be reducing chum salmon spawning habitat quality in the Grays River is a lack of habitat diversity. Excess sediment and reduced LWD may have effects on channel bedform diversity and hyporheic exchange. In the salmon spawning habitat analysis, reduced riffle:pool frequencies were documented in all spawning study reaches when compared to stream channels in naturally forested Pacific Northwest watersheds. Bedform diversity is one of several important mechanisms encouraging exchange between surface and subsurface flows. In addition, agricultural practices such as levee and dike construction separate the river from its floodplain and restrict natural channel migration and off-channel habitat development. LWD contributes to channel bedform diversity and system productivity through its effects on flow and contributions of nutrients and habitat structure.

S.7 Restoration Alternatives

Actions focused on restoration of altered habitat-forming processes, sediment delivery and LWD recruitment, are key to salmonid recovery. The overall long-term goal for the upper Grays River watershed is to conserve and restore habitat-forming processes while minimizing risks to property and infrastructure. This goal can be achieved with a comprehensive watershed restoration plan and by focusing land-use activities in those portions of the watershed where they will not adversely affect hydrologic, geomorphic, or ecological processes. In developing restoration plans, streams and rivers must be recognized as integral components of the watershed landscape and not as discrete channels that can be "fixed" without addressing the connected issues within contributing upland areas. The following objectives are recommended to focus watershed restoration efforts toward this goal:

1. Limit upland land-use activities that trigger sediment production in the upper watershed.

2. Restore functional riparian forests and long-term LWD recruitment potential.

3. Improve floodplain connectivity and function in the Grays River response reach.

These objectives address the historical changes in watershed processes and the resultant degraded habitat conditions documented in this and other studies within the upper Grays River watershed.

Because achieving this goal may take 50–100 years or more, shorter term rehabilitation projects can be implemented to bridge the gap between current degraded conditions and future desired conditions. Thus, priority actions are those that restore habitat-forming processes (long term) and maintain existing habitats (short term). Restoration actions to meet these objectives are summarized below.

Based on adaptive management experience from the Pacific Northwest, it is likely that a multiple-option approach will be necessary over the long term. Whatever option(s) chosen, the solution must be fish-friendly, sustainable, cost-effective, and technically feasible. Local support for restoration activities is critical to the long-term success of most restoration efforts. Landowner cooperation is often a limiting

factor and, as such, some restoration alternatives are better left to local groups to implement, with technical guidance provided by scientists and engineers as required.

To realize the stated restoration goals and objectives for the Grays River watershed, the following restoration activities are recommended:

- Protect existing intact and functional refugia habitat. Crazy Johnson Creek is one example of a critical refugia subbasin within the Grays River system.
- Limit timber harvest on steep slopes and in areas with soil types that are most susceptible to mass-wasting and surface erosion.
- Decommission inactive forest roads and restore natural drainage patterns.
- Construct channel-spanning LWD and/or engineered logjam (ELJ) structures throughout the upper watershed to store sediment, moderate sediment routing, and improve local aquatic habitat.
- Implement active reforestation (conifers) of floodplain and riparian forests throughout the watershed to restore natural, self-sustaining processes that supply and recruit functional LWD to the channel.
- Reestablish riparian buffers throughout the watershed and protect these critical areas from human disturbances such as logging and roads.
- Lengthen the timber harvest rotation period (80–100 years or more) to better mimic the natural disturbance regime.
- Establish an unconstrained channel migration zone (CMZ) and floodplain in the Grays River response reach immediately upstream of State Route 4.
- Construct ELJ structures to create stable forested islands and complex instream habitat within the floodplain and CMZ of the Grays River response reach.
- Support and strengthen existing local efforts to build and maintaining grass-roots community involvement and cooperation within the watershed.

A well-designed monitoring program is essential to determine the effectiveness of restoration projects as well as allow for adaptive management as additional restoration activities, changes, and improvements are identified for the Grays River watershed. Each restoration action should be considered a full-scale field experiment with effectiveness monitoring to provide feedback on the success of the action and allow for modifications.

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1.0 Introduction

This report documents a three-year study conducted by Pacific Northwest National Laboratory (PNNL) for the Bonneville Power Administration. The study sought to perform a comprehensive watershed and biological assessment of the Grays River watershed in Washington State. Based on the results of the comprehensive watershed analysis, PNNL researchers developed a prioritized list of actions to protect and restore critical chum and Chinook salmon spawning habitat in the Grays River.

1.1 Background

Stocks of chum (*Oncorhynchus keta*) and fall Chinook (*Oncorhynchus tshawytscha*) salmon in the lower Columbia River declined dramatically during the past century (LCFRB 2004). In 1999, the National Marine Fisheries Service (NMFS) listed lower Columbia River chum salmon as a threatened Evolutionarily Significant Unit (ESU) under the *Endangered Species Act of 1973* (ESA; NMFS 1999). Since ESA listing, lower Columbia River ESU chum salmon populations have generally remained stable but low, with average runs in the lower Columbia River less than 1% of their historical run size (LCFRB 2004). Only two significant populations of chum salmon remain in the lower Columbia River ESU. These critical chum salmon populations are located in the Grays River and downstream from Bonneville Dam. Lower Columbia River ESU chum populations below Bonneville Dam have been adversely affected by operation of the dam (NOAA 2004), which could reduce the likelihood of survival and recovery of this portion of the chum salmon ESU and increase the importance of the Grays River population.

The Grays River watershed once was noted for its large runs of chum salmon. Although currently larger than other populations, the Grays River chum salmon run is a fraction of its historical size. Historically, the chum salmon run size in the Grays and Chinook River population ranged from 8,000 to 14,000 fish (LCFRB 2004), with most spawning occurring in the Grays River. The decline of the Grays River chum salmon population has been documented in the *Water Resources Inventory Area (WRIA) 25 Salmon and Steelhead Habitat Limiting Factors Analysis* (WCC 2002) and in the Northwest Power Planning Council (NPPC) *Grays River Sub-Basin Plan* (LCFRB 2004).

The Grays River chum salmon population has declined significantly in the past several decades as a result of damage to spawning habitat associated with human land-use activities in the upper watershed (LCFRB 2004). Channel clearing, floodplain agriculture and associated riparian clearing and diking, and logging and associated road construction on unstable ground in the watershed produced landslides, erosion and channel instability and loss of riparian function, and were largely responsible for the decline in Grays River chum and Fall Chinook salmon populations (WDFW 2001; WCC 2002; LCFRB 2004). The few stable chum salmon spawning areas that remain in the Grays River are subject to variable conditions that further threaten these declining populations. The lack of stable spawning habitat has been identified as the primary physical limitation on chum salmon spawning in the Grays River (WCC 2002). In addition, approximately 20–25% of the remaining spawning habitat was lost during a 1999 channel avulsion that destroyed an important artificial spawning channel operated by the Washington Department of Fish and Wildlife (WDFW) in the Gorley reach of the upper Grays River (WDFW 2000).

1.2 Study Scope

The Grays River Watershed and Biological Assessment Project was initiated by the Bonneville Power Administration (BPA) based on the recommendation of the Lower Columbia Fish Recovery Board (LCFRB). The primary purpose of the project was to address information needs for the recovery of ESA-listed Grays River chum salmon. Information gathered during this project would benefit ESA-listed Grays River fall Chinook salmon as well. Although the lack of stable spawning habitat is believed to be the primary physical limitation on Grays River chum salmon production (WDFW 2001; WCC 2002; LCFRB 2004), numerous additional sources of aquatic habitat degradation may have contributed to this species' decline and may limit its recovery in the Grays River. This project was initiated to collect critical data needed to guide watershed management and habitat restoration activities for these species.

The ultimate goal of this project was to provide the technical basis for the enhancement and restoration of ecosystem processes that support native salmonid populations in the Grays River watershed, primarily chum and fall Chinook salmon. This study was specifically designed to address the decline in chum salmon populations and make recommendations for recovery of these populations. The study was aimed at addressing critical information needs, including the location and characteristics of salmonid spawning habitat and hydrologic, geomorphic, and land-use factors in the upper Grays River watershed that influence spawning habitat availability and condition. One important information need that was not included within the scope of the study is an assessment of juvenile fall Chinook salmon rearing habitat. Because the primary focal species was chum salmon, which emigrate soon after emergence, the project scope focused on evaluation of spawning habitat. Successful recovery of fall Chinook salmon in the Grays River may depend on improved understanding of rearing as well as spawning habitat. The specific objectives of the project were to 1) perform a comprehensive watershed and biological assessment, including hydrologic, geomorphic and ecological assessments; 2) gain a better understanding of chum and Chinook salmon spawning habitat requirements, and 3) develop a prioritized list of actions to protect and restore critical chum and Chinook salmon spawning habitat in the Grays River.

The watershed assessment includes geomorphologic and hydrological analyses of the upper Grays River watershed and analyses of the primary chum and Chinook salmon spawning habitat within the Grays River. Included in the assessment are analyses of the physical and hydrological characteristics of primary spawning areas and chum and fall Chinook salmon spawning habitat availability and use. This assessment is a significant step in a comprehensive program to ensure the survival and recovery of Columbia River chum salmon in its most productive system and builds on existing recovery planning efforts for these ESA-listed salmonids within the Grays River and the lower Columbia River. This assessment also provides a basis for the recovery of other fish species in the Grays River, including coho salmon, winter steelhead, coastal cutthroat trout, and Pacific lamprey. The conservation, enhancement, and restoration recommendations developed in this project are based on scientific assessment of historical and current watershed conditions, while at the same time considering local community stakeholder interests.

1.3 Project Approach

There is broad acceptance within the scientific community that the most effective approach to salmon recovery should be watershed-based. Conservation and restoration efforts need to address all the life-history stages of the species of concern. A watershed-based approach encompasses almost all of the critical components of the freshwater ecosystem that supports native salmonid species. It has been widely recognized that restoration and conservation measures developed in a watershed-ecosystem context are more likely to be successful (Nehlsen et al. 1991; Reeves et al. 1991, 1995; Naiman et al. 1992, 1995; Doppelt et al. 1993; FEMAT 1993; Lichatowich et al. 1995; Beechie et al. 1996, 2003; Frissell and Bayles 1996; Stouder et al. 1997; Williams et al. 1997; Beechie and Bolton 1999; Pess et al. 1999, 2002, 2003).

Successful river ecosystem restoration requires a comprehensive assessment of current conditions. The assessment should include an analysis of current ecosystem structure and function, addressing the level of disruption of natural habitat-forming processes. Key elements of a watershed assessment include the analysis of hydrologic and geomorphic characteristics of the river system. These analyses provide a framework for restoration planning. In addition, the biological integrity of the river system must be assessed to evaluate the level of productive capacity in relation to focal species abundance and diversity. The goal of these assessment efforts is to identify alterations to key ecosystem processes that are responsible for creating and sustaining salmonid habitat, as well as management actions necessary to conserve and/or restore the river to a properly functioning condition. Restoration of focal salmonid species populations must be subordinate to the primary goal of restoring a river ecosystem capable of supporting multiple salmonid species as well as other species within the salmonid food web (Beechie et al. 2003).

The watershed-based approach to river ecosystem restoration relies on a conceptual framework that describes general relationships between natural landscape characteristics, watershed-scale habitat-forming processes, aquatic habitat conditions, and biological integrity. In addition, human land-use impacts are factored into the conceptual model because they can alter habitat quality and can disrupt the natural habitat-forming processes. In this model (see Figure 1.1), aquatic (instream and riparian) habitat is viewed as the link between watershed conditions and biologic responses. Based on this conceptual model, assessment of habitat loss and the resultant declines in salmonid populations can be conducted by relating current and historical (e.g., natural) habitat conditions to salmonid utilization, diversity, and abundance. In addition, assessing disrupted ecosystem functions and processes within the watershed can aid in identifying the causes of habitat change and the associated decline in biological integrity. In this same way, restoration, enhancement, and conservation projects can be identified and prioritized (Beechie et al. 2003).

Watershed assessment is broadly defined as compiling and analyzing technical information on watershed landscape conditions and land-use activities. Watershed assessment can also be described as an approach for understanding watershed processes, their influences on the riverine ecosystem, and the effect of human activities on the natural watershed ecosystem. This assessment process is typically one of the first steps in a long-term *ecosystem management* program. The watershed assessment process should provide the foundation for a better understanding of how a watershed is functioning and how the

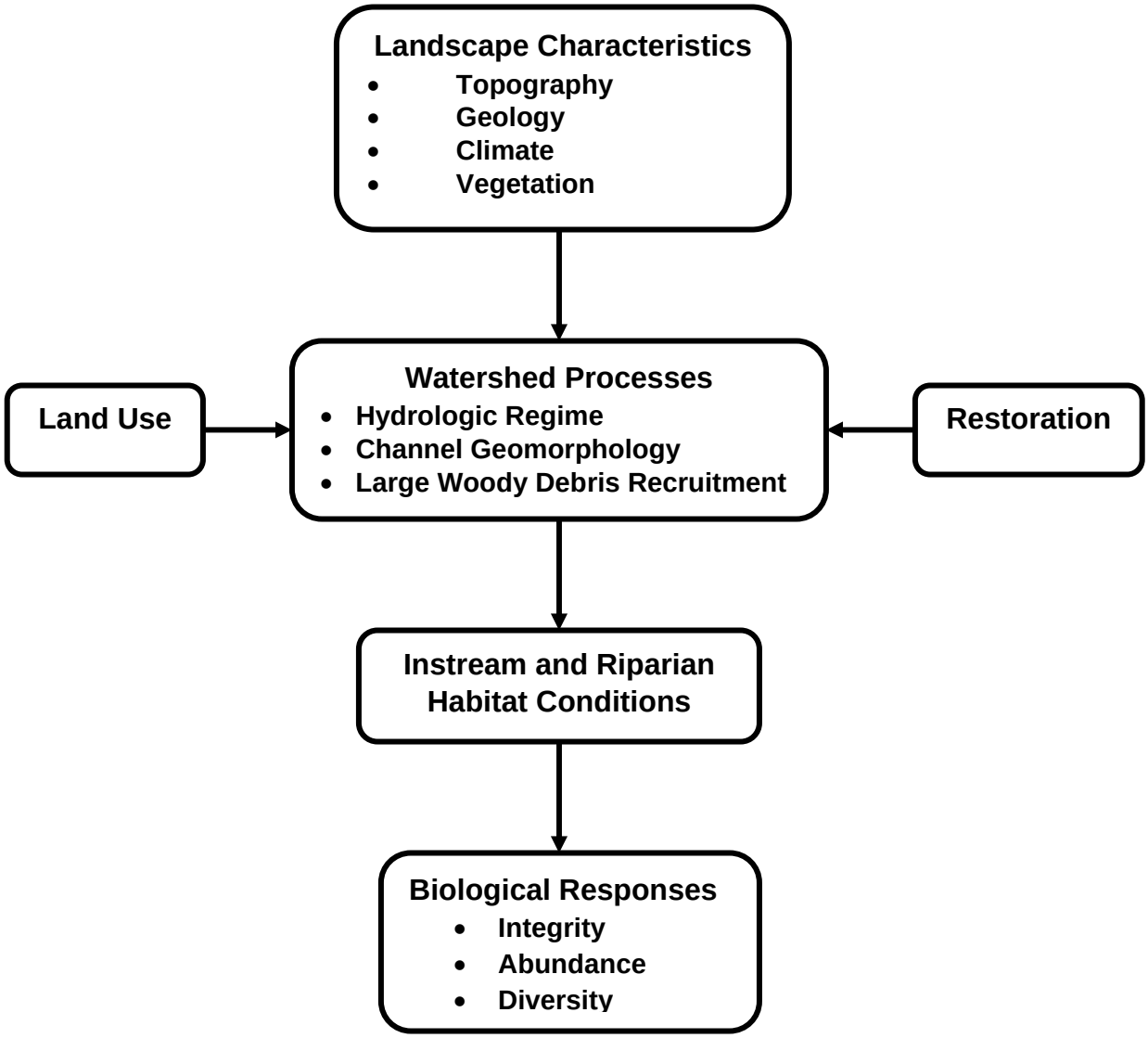


Figure 1.1. Conceptual Model Illustrates Linkages Between the Landscape Characteristics, Watershed Processes, Habitat Conditions, and Biological Responses, as Well as the Interaction of Human Land-Use Activities and Restoration Efforts (adapted from Beechie et al. 2003)

watershed has changed as a result of human activities. The watershed analysis process essentially provides the framework for implementing ecosystem management, linking scientific data to land-use planning decision-making (Montgomery et al. 1995).

There is significant scientific evidence that landscape-scale factors influence aquatic ecosystem structure and function as well as biological integrity across a wide range of spatial and temporal scales (Naiman et al. 1992; Gergel et al. 2002; Allan 2004). The linkage between landscape characteristics and salmonids (habitat quality, diversity, and abundance) has also been demonstrated (Thompson and Lee 2000; Paulsen and Fisher, 2001; Pess et al. 2002).

In developing our approach to this watershed assessment of the Grays River, guidance from numerous sources was used. The *Federal Guide for Watershed Analysis* is subtitled *Ecosystem Analysis at the Watershed Scale* (IAC 1994, 1995). The state of Washington also provides guidance on watershed

assessment and analysis. The Joint Natural Resources Cabinet (JNRC) document, *Guidance on Watershed Assessment for Salmon* (JNRC 2001), provides broad guidance on conducting watershed-based studies specifically related to salmon recovery efforts. The Washington State Department of Natural Resources (WA-DNR) also provides guidance on conducting watershed assessments in areas where timber harvest is the primary land-use activity, via its manual, *Standard Methodology for Conducting Watershed Analysis*, published by the WA-DNR Forest Practices Board (WA-DNR 1994). The state of Oregon also provides guidance on watershed assessment in the *Oregon Watershed Assessment Manual* (GWEB 1999).

Natural ecosystem processes tend to operate at a wide range of spatial and temporal scales within a watershed. In addition, the natural ecological *disturbance regimes* (e.g., flooding, fire, wind) also operate at different spatial and temporal scales within a watershed (Naiman et al. 1992). In general, there is a relationship between the spatial scale of watershed processes or disturbance regimes and the degree of variability or time scales over which the processes vary. In riverine ecosystems, hydrogeomorphic processes such as floods, landslides, and channel avulsions act as the primary ecological disturbance regime and play a key role in creating, modifying, and destroying aquatic and riparian habitat (Montgomery and Bolton 2003). The hydrogeomorphic disturbance regime also establishes the physical habitat template that determines potentially successful behavioral and life-history strategies of aquatic organisms (Poff and Ward 1989; Poff 1990, 1996; Poff et al. 1997).

Developing an understanding of natural variability and the level of uncertainty is one of the primary goals of the watershed assessment process and is critical to the successful implementation of almost any river ecosystem restoration program. In general, restoration plans should be designed to incorporate natural variability and disturbance rather than try to “control” watershed processes or “engineer” riverine structure (NRC 1992). For example, engineering of a fixed geomorphic channel form does not restore the hydrogeomorphic processes that create natural channel geomorphology and habitat features. The current approach to river restoration advocated by the scientific community is to address, not just manage, the symptoms and construct site-specific projects, focusing on the causes and restoring natural processes throughout the watershed (Beechie et al. 1996, 2003; Beechie and Bolton 1999).

The natural variability (spatial and temporal scales) of river ecosystems provides an important part of the conceptual foundation for conservation, enhancement, and restoration planning. It is also because of this variability in natural processes and the uncertainty regarding the outcomes of selected management approaches that river restoration is often difficult to accomplish. Uncertainty arises when all the factors influencing complex river ecosystems are not completely understood or are unpredictable. Natural variability contributes to uncertainty when our knowledge of natural processes and disturbance regimes are incomplete, which is almost always the case. Even the most comprehensive watershed assessments rarely, if ever, have the benefit of complete knowledge of the natural environmental conditions or the effects of anthropogenic activities on the riverine ecosystem under study (Wissmar and Bisson 2003).

Spatial and temporal variations in landscape processes and disturbances create and maintain the dynamic mosaic of habitat conditions within a watershed that support local salmonid populations. Although specific habitat features (e.g., pools and riffles) are spatially and temporally transient, the dynamic processes that shape and change local conditions generally lead to persistent characteristics within the riverine landscape when viewed over a wider spatial context (Montgomery and Bolton 2003). This illustrates the concept of *dynamic equilibrium* that is common to most lotic ecosystems.

It is to these unique watershed characteristics that salmonid stocks have adapted over the long term (Reeves et al. 1995). This also implies that salmonid species and populations have adapted to spatially and temporally variable habitats, and that such environmental variability is likely important to their long-term survival and therefore should be preserved (Beechie et al. 1996). Thus, spatial and temporal variability of natural watershed processes must be considered in any restoration effort.

Because salmonids are adapted to spatially and temporally diverse and varying habitat conditions, it does not make sense to manage for the same conditions in all locations or to expect conditions to remain the same at any one location. It is therefore important to recognize the appropriate spatial and/or temporal scale for each component of the watershed assessment. An integrated and comprehensive watershed assessment will contain multiple and overlapping scales of analysis. The result of the assessment process will be a suite of habitat conditions that should be protected, enhanced, or restored over the entire watershed (salmon) landscape.

As a general principle, *protection* or *conservation* of intact, functional watersheds or subwatersheds should be a high priority, according to the National Research Council (NRC 1992, 1996) and researchers in the field (Beechie and Bolton 1999; Roni et al. 2002; Beechie et al. 2003). Maintaining high-quality habitat is typically much easier than restoring degraded habitat (NRC 1992; Franklin 1993; Roni et al. 2002). These protected areas can also serve as "reference areas" or "templates" for restoration of already degraded habitat. These areas can provide fundamental information on natural processes and ecosystem structure, enable evaluation of temporal and spatial changes, and provide a basis for estimating attainable future conditions for restoration. Protection of aquatic resources involves preventing anthropogenic alterations to the structure and function of the ecosystem (NRC 1992). Ideally, this would involve protection of biological, chemical, and physical processes that maintain salmonid habitat. The mechanisms for protection include regulation, acquisition, conservation easements, and stewardship.

Watershed restoration activities should be prioritized according to their potential effectiveness as well as likelihood for longevity of response and probability of success. However, the effectiveness of various restoration techniques is not fully understood. Inherent variability in ecological data can make it difficult to detect real change, and long-term monitoring data needed to evaluate restoration success are not always available. Lack of effective monitoring, including an emphasis on physical rather than biological factors, has limited our understanding of aquatic ecosystem response to restoration. During prioritization of restoration activities within a watershed, landscape position is important to consider. Some argue that restoration should proceed downstream from headwater reaches, because the successful restoration of downstream areas often depends on restoring upstream processes and upland land use. Minimizing adverse impacts on small headwater streams and restoring their ecological connections are of critical importance to improvements downstream. Often, effects of disturbances such as floods are magnified as they progress downstream. In addition, stream restoration without modification of upslope land uses are often ineffective. Obviously, many criteria may be weighed when prioritizing restoration, enhancement, and conservation efforts (NRC 1992).

Management strategies also should be developed based on the guiding principles of ecosystem management and the specific results of watershed assessments. Watershed management is continuous and needs a multidisciplinary, flexible approach. A watershed management framework that supports community partnering, uses sound science, and takes well-planned actions should achieve positive results. Some would argue that we lack sufficient knowledge about the way in which watersheds work to implement an ecosystem-based management strategy. On the contrary, even if no further research were

done, the current level of knowledge is adequate to do a much better job of aquatic resource conservation than has been done in the past. In general, watershed protection is more effective than restoration or rehabilitation. Therefore, conservation efforts should move forward and should not be delayed while awaiting "better" data. It must be acknowledged that science will never answer all the questions. Science is a process, not an endpoint. Rather than being paralyzed by the unknown as more ecosystems are degraded or more species are lost, action should be taken to apply the *best available science* toward conservation of the remaining functioning habitat. In conjunction with this conservation action, scientific research should continue so that the level of knowledge improves and *adaptive management* is supported in the long term. This also should reinforce the *precautionary principle*, in that watersheds and aquatic resources should not be managed to the limit of their functional capacity but instead should be managed conservatively (Franklin 1993).

1.4 Report Organization

This report is organized following the conceptual format of a watershed assessment. Chapter 2 describes current and historical conditions in the Grays River watershed, with additional detail provided in Appendix A. The hydrologic and geomorphic analyses are summarized in Chapters 3 and 4, respectively. Additional detail regarding the geomorphic assessment is provided in Appendixes B through D. A detailed analysis of chum and fall Chinook salmon spawning habitat use, including potential and actual use and relationships between spawning and hyporheic and geomorphic characteristics of spawning habitats, is presented in Chapter 5, with additional detail provided in Appendixes E through H. An analysis of aquatic macroinvertebrate community integrity is included in Appendix I. Chapter 6 integrates the findings of the hydrologic, geomorphic, salmon spawning habitat, and aquatic integrity analyses and presents a discussion of restoration needs including a prioritized list of restoration recommendations. References are in Chapter 7. Appendix J provides a log of study area photographs compiled by PNNL researchers from field visits over the course of the project.

2.0 Grays River Watershed Characterization

The watershed approach to the assessment of riverine ecosystems has a foundation in landscape ecology and has been widely employed in natural resource management. Landscape ecology emphasizes the interaction between spatial patterns and ecological processes. This approach has been shown to be relevant for assessing the condition of rivers and their associated watersheds. Watershed-scale landscape characteristics describing the level of human alteration of natural ecosystem attributes can be used as indicators of water quality, hydrology, channel geomorphology, instream habitat, and biotic integrity (Roth et al. 1996; Allan et al. 1997; Gergel et al. 2002; Liu and Taylor 2002; Allan 2004; Fu et al. 2005; Kearns et al. 2005). The scale of analysis is especially critical in watershed assessment (Roth et al. 1996; Allan et al. 1997; Allan 2004). Studies to date indicate that riverine watershed assessment should be conducted at multiple spatial scales and should include the subbasin or subwatershed scale as well as the riparian or floodplain scale (Roth et al. 1996; Allan et al. 1997; Gergel et al. 2002; Allan 2004; Kearns et al. 2005). That is the approach used for the upper Grays River watershed assessment project.

2.1 Background

The Grays River is a tributary of the lower Columbia River and is located in Washington State a short distance upstream from the mouth of the Columbia River in the counties of Pacific, Cowlitz, and Wahkiakum (Figure 2.1). The Grays River watershed encompasses a total of 79,635 acres (124 mi^2 or

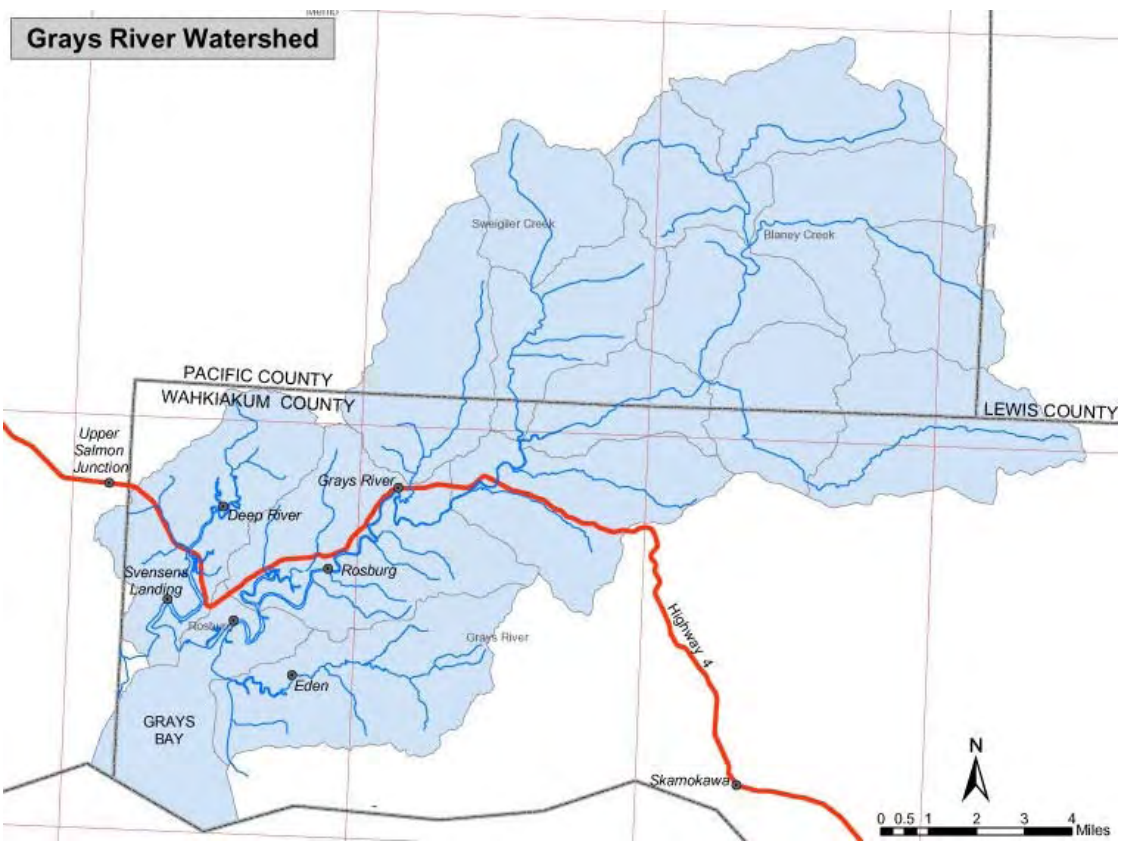


Figure 2.1. Upper Grays River Watershed

322 km^2). The Grays River watershed can be divided into an upper and lower watershed, with the delineation usually being made at the upper extent of tidal influence, typically at the State Route 4 (SR 4) highway bridge, approximately 18.5 km upstream from the confluence with the Columbia River.

The watershed consists of 1,298 miles of stream channel that drain southwesterly to the Columbia River. The Grays River watershed has an average drainage density of 10.8 miles of stream channel per square mile. The northern portion of the watershed lies within the Willapa Hills. The upper watershed is generally steep and is dissected by a dense stream network. Unconfined floodplain areas are found in the lower mainstem and locally within the watershed interior along the West, East, and South Forks of the Grays River. The watershed ranges in elevation from sea level in the southwest to over 3,000 ft above sea level in the headwaters.

2.2 Natural Watershed Characteristics

Historically, the Grays River watershed consisted almost entirely of old-growth coniferous-dominated forest (Scott 2001). Natural forest cover in the Grays River watershed was split between two distinct forest types (Figure 2.2). The Sitka-spruce zone occupies approximately 30,080 acres or 37% of the watershed, mostly at lower elevations in the lower river floodplain valley (Franklin and Dyrness 1969). The lower river valley also included some natural meadows and wetlands. The dominant tree species of this zone, in order of abundance, are Sitka spruce, western hemlock, Douglas fir, and black cottonwood. Regionally in Western Washington, the Sitka-spruce zone is contained in a narrow belt extending from the Pacific Ocean into adjacent river valleys and can be characterized by proximity to the ocean, mild temperatures, 60–80 in. annual precipitation, and frequent summer fog (Franklin and Dyrness 1969). The western hemlock zone occupies approximately 49,555 acres or 62% of the Grays River watershed, mostly on the steeper slopes in the upland areas of the watershed. This zone can be described as a wet maritime climate with 80–120 in. of precipitation per year (Franklin and Dyrness 1969). The western hemlock forest consists predominantly of western hemlock, Douglas fir, and western red cedar. Hardwoods such as red alder and big-leaf maple are typically found in disturbed areas in both forest zones (Franklin and Dyrness 1969).

Several methods can be used to recreate the natural landscape of an area as part of a comprehensive watershed assessment project or to reconstruct a degraded landscape as part of a restoration effort. This approach is generally known as *historical ecology*. Historical ecology includes aspects of landscape ecology, archaeology, and ethno-botany, as well as an understanding of the social, cultural, political, and economic influences on the natural environment (Eagan and Howell 2001). It is very important to consider the influences of aboriginal people on the landscape, especially in areas where native tribes manipulated the natural environment (e.g., controlled fires). From this perspective, the “natural” landscape is often described as a “historical” landscape because of the influences of native peoples that inhabited most areas of the Pacific Northwest (Eagan and Howell 2001).

The earliest written description of landscape characteristics can be found in the General Land Office (GLO) survey records. The GLO, as part of its cadastral survey of the western states and territories, mapped the lower Columbia River and Puget Sound regions between 1850 and 1895. Because these surveys were conducted prior to or during the earliest stages of settlement, GLO maps and field notes are a unique resource for characterizing the historical environment. GLO survey records have been used to reconstruct the natural/historical landscape in several areas of the United States (Galatowitsch 1990; Nelson et al. 1997; Collins et al. 2003).

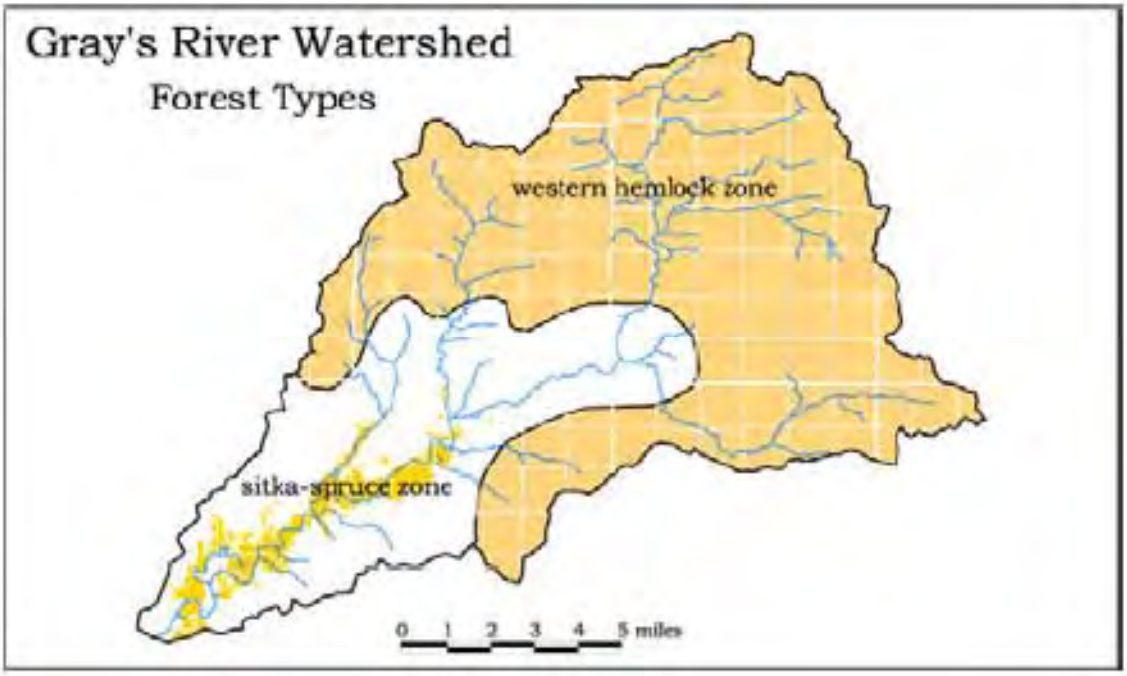


Figure 2.2. Grays River Watershed Vegetation-Zone Map (Scott 2001)

GLO maps and field notes contain a wealth of information on historical landscape characteristics. However, these historical records must be used properly. GLO maps and field notes generally record observations only along the "section" boundaries and navigable channels that the survey teams were required to walk. GLO surveys were conducted using "rectangular" survey methods. Section lines were surveyed at 1-mi intervals, creating sections that were 1 mi square or approximately 640 acres. Survey "monuments" or other marks were typically made at half-mile intervals to delineate "quarter-sections" or smaller parcels in multiples of 40 acres. Each survey team consisted of a deputy surveyor, two chainmen, and flagman, and a marker (or axe-man). The two chainmen dragged a 66-ft chain (each chain consisted of 100 links, each 7.92 in. long, such that a mile was equal to 80 chains) along each section line to measure distances, and the flagman marked the end point of each chain length. The marker was responsible for blazing (with an axe) "witness" or "bearing" trees or constructing a suitable monument (i.e., rock cairn, post). Section surveys took on the order of 5 to 15 days to complete and involved walking over 120 mi, as the 60 mi of section lines had to be walked at least twice ("over and back") to get an accurate line survey. Section lines were surveyed from north to south, requiring the survey team to walk from the north edge of the township to the south edge five times per township. Needless to say, this was difficult work, especially in mountainous and wet terrain (Eagan and Howell 2001).

Although these methods seem crude by today's standards in which global positioning system (GPS) equipment is typically used, the GLO surveys are generally quite accurate. GLO maps have proven to be quite accurate when compared to present-day maps of the same areas. Much depended on the quality of the survey teams with respect to survey accuracy. Survey teams also constructed township maps to complement their filed notes. Some of these maps were quite detailed, although some were a bit lacking, depending on the skill and dedication of the survey team. In general, major streams, rivers, wetlands, and

lakes were mapped, as well as cultural features and basic topographic characteristics. Survey teams also noted the width of stream and river channels crossed during surveys. These widths are roughly equivalent to present-day wetted or bankfull widths. Survey teams typically extrapolated channels between crossings, often drawing in meandering streams or rivers. However, channel locations should be considered accurate at only the intersection of section lines. Channels shown on GLO maps in between sections boundaries have the most uncertainty (Eagan and Howell 2001).

It is essential that the GLO maps and field notes be used together, as in many cases maps may not reflect all the observations noted in the field survey logs. For each section line walked, deputy surveyors were required to record distances (in "chains") between section corners. In addition, surveyors were required to note landscape features such as streams or rock formations. In addition, types of trees and sizes of timber were also noted, as well as cultural features such as homesteads, trails, or cleared areas. At each section corner, surveyors were required to note the distance and direction to at least two "witness" or "bearing" trees (also noting species and size of tree) that were used to mark the section corner. Survey teams also had to note the predominant vegetation and soil type along each section line walked. In addition, if a tree was encountered along a section line, this "line" tree was noted by species and size in the field notes. Lakes, mountains, and rivers off the section lines were also frequently noted if they were easily observed and were considered significant. Finally, each deputy surveyor was required to write a general description of the township and section natural characteristics and cultural features. It is these notes, especially the notes of vegetation (e.g., bearing, witness, and line trees) that provides the basis for historical reconstruction of the natural landscape (Eagan and Howell 2001).

In general, surveyors used common names or traditional names for trees and under-story vegetation, which can lead to confusion in identifying trees or shrubs by their scientific names (Eagan and Howell 2001). Although useful in reconstructing the historical landscape, bearing or witness trees are not necessarily the most reliable proxies for recreating the historical forest characteristics because survey teams did not select these trees randomly but instead used specific criteria. These were generally trees that were at least 6 cm in diameter that were located close to section or quarter corners.

Research in the Puget Sound region indicates that bearing/witness trees tended to underrepresent smaller-diameter tree species such as vine maple (*Acer circinatum*). In spite of this and other bias issues, when combined with information on soil and wetland conditions, GLO tree-species data can be fairly representative of historical forest characteristics (Collins et al. 2003). In general, bearing or witness trees provide a good representative database of the species present in the area at the time of the survey but not necessarily the stand age (forest maturity) or average tree diameter/size. Line trees, because they were encountered randomly, do provide a good representation of both species and stand age or tree size (i.e., diameter). Figure 2.3 shows an example of a GLO survey map of the upper Grays River watershed. All available GLO maps and survey notes are included in Appendix A.

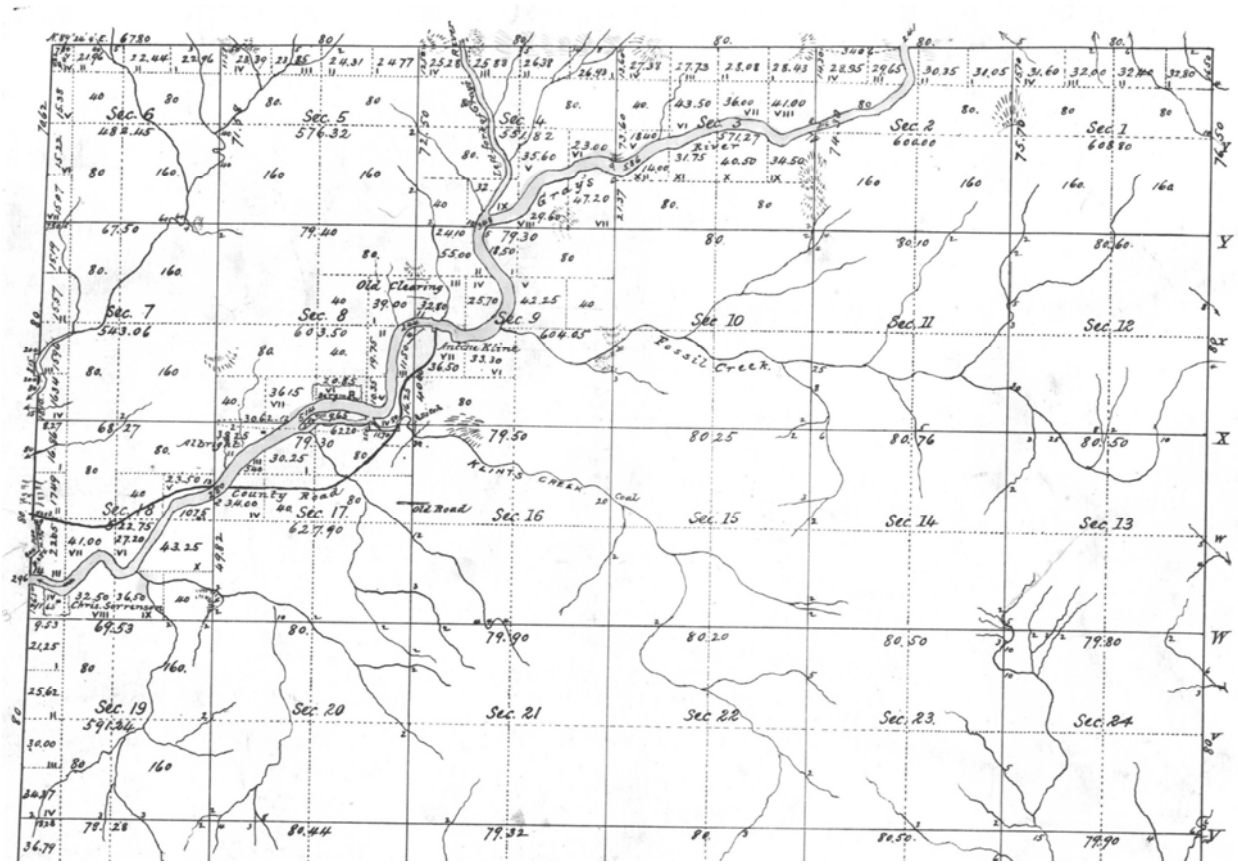


Figure 2.3. Grays River Watershed General Land Office Survey Map (BLM 2004)

The following is an excerpt from the 1882–3 GLO survey of the Grays River watershed in Township 10 North - Range 7 West:

> *"The great portion of this township consists of high rolling land, covered with dense, heavy timber of Hemlock, Fir, Spruce, Cedar, Alder, and Maple, all of which will be very valuable as lumber. The forest has a dense undergrowth of vine maple and huckleberry."*
>
> *"The soil is generally a yellow (sandy) loam on the hills and of a good 2nd-rate quality. There are considerable quantities of good bottom lands in Sections 3, 4, 8, 9, 17, 18, 35, and 36."*
>
> *"Grays River is navigable for small steamers and scows to within 20 chains of the west boundary (of the township). Thence to the east boundary of Section 3 (near the Gorley reach) it is navigable for small boats for about six of seven months during the year, when the settlers get easy access to market with their produce. There are now some 15 or 20 settlers in the township."*
>
> *Lewis D.W. Shelton, U.S. Deputy Surveyor*

The following is an excerpt from the 1882-3 GLO survey of the upper Grays River watershed in Township 11 North - Range 7 West:

> *"This township consists chiefly of high rolling hills and broken land covered with a forest of Hemlock, Fir, Cedar, and Spruce. The land is a good 2nr rate quality except in Sections 28, 29, 30, 31, and 32 in which there are some good bottom lands of 1st rate quality. There are no settlers in the township."*
>
> *Lewis D.W. Shelton, U.S. Deputy Surveyor*

A review of all available GLO maps and field notes (circa 1882–1884) indicates that the upper Grays River watershed consisted of mature, coniferous-dominated forest with a dense under-story. The dominant tree species appear to have been western hemlock, fir (Douglas and silver), and cedar, followed in abundance by spruce, alder, and maple. Vine maple, huckleberry, salmonberry, and devil's club. Cottonwood was frequently noted in "bottom-land" (i.e., floodplain) areas within the Gorley reach and downstream.

The mainstem of the Grays River downstream of the Gorley reach was measured at 2.41 chains or approximately 160 ft and the West Fork (at the confluence) was measured at 1 chain of about 66 ft in width. Multiple channels and numerous tributary streams were noted in this section of the survey. Fossil Creek was measured at 25 links (approximately 16 ft) in width near the confluence, and the "river bottom" (i.e., floodplain) was measured at approximately 8 chains (over 500 ft in width) in this confluence area where the West Fork, Fossil Creek, and the mainstem Grays come together. Upstream of the Gorley reach (near the mouth of the canyon) the river was measured at 3.15 chains or approximately 200 ft in width, indicating a relatively wide floodplain area. Within the canyon, the river was measured at 1.60 chains or approximately 100 ft in width. These measurements are comparable to current conditions.

A summary of "line-trees" noted in the GLO survey notes includes the following:

- hemlock (30 in. in diameter)
- fir (60 in. in diameter)
- spruce (50 in. in diameter)
- maple (24 in. in diameter)
- fir (70 in. in diameter)
- fir (75 in. in diameter)
- hemlock (50 in. in diameter)
- fir (50 in. in diameter)
- fir (50 in. in diameter)
- hemlock (70 in. in diameter).

To more accurately reconstruct the historical (natural) landscape conditions in the study area, we supplemented the GLO maps and field notes for the upper Grays River watershed with data obtained from the 1909 Timber-Cruise Survey records (Washington State Archives 2005). These timber-cruise records show the species composition (only conifers were included in these surveys) and estimated quantity of

timber (in board-feet) available in a survey area. Using that data, the following estimate of forest composition was developed for the upper Grays River watershed at the turn of the century:

- western hemlock (38%)
- western red cedar (23%)
- Douglas fir (23%)
- silver fir (14%)
- spruce (2%).

Based on the data from timber-cruise records and GLO surveys, the historical or natural landscape of the upper Grays River was characterized by old-growth coniferous-dominated forests, with western hemlock being the dominant species. A dense undergrowth of vine maple and huckleberry also was present. Riparian areas likely had a large portion of alder, maple, and cottonwood, especially in areas frequently disturbed by flooding and channel migration. Spruce and cedar were also most commonly found in riparian areas, floodplains, and wetland areas.

The human history of the Grays River Valley, including the history of logging in the watershed, is described in detail in Appendix A of this report. That historical record is based on oral-history interviews and archival material provided by members of the local community.

2.3 Current Watershed Characteristics

Currently, the upper Grays River watershed is dominated almost completely by privately owned forestland (Figure 2.4). There is no federal forestland in the upper Grays River watershed, and the Washington State Department of Natural Resources owns only a small portion of the watershed, with almost all of that land located in the South and West Fork subwatersheds. Figure 2.5 illustrates the history of timber-harvest activities in the watershed. Little unlogged, natural old-growth forest remains within the upper Grays River watershed. Most areas have been logged multiple times.

The current (2003) land-cover characteristics are shown in Figure 2.6 and summarized in Table 2.1. Currently, over 90% of the upper Grays River watershed is forested, with about two-thirds of the area classified as coniferous-dominated forest. At any one time, about 5–10% of the watershed can be classified as recently disturbed; either clear-cut or burned. Only a small fraction (0.2%) of the upper Grays River watershed is classified as agricultural, with all of that land use located in the lower mainstem subbasin, at the very downstream end of the study area.

We compiled the Grays River watershed assessment at the subwatershed scale. Each major tributary subwatershed is composed of several, smaller subbasins (see Figures 2.7 and 2.8). The primary subwatersheds evaluated included the North Fork, East Fork, South Fork, Upper Mainstem, West Fork, Fossil Creek, and the Middle Mainstem. The Lower Mainstem, which is tidally influenced (downstream of the SR 4 highway bridge), was not included in the study area of this project.

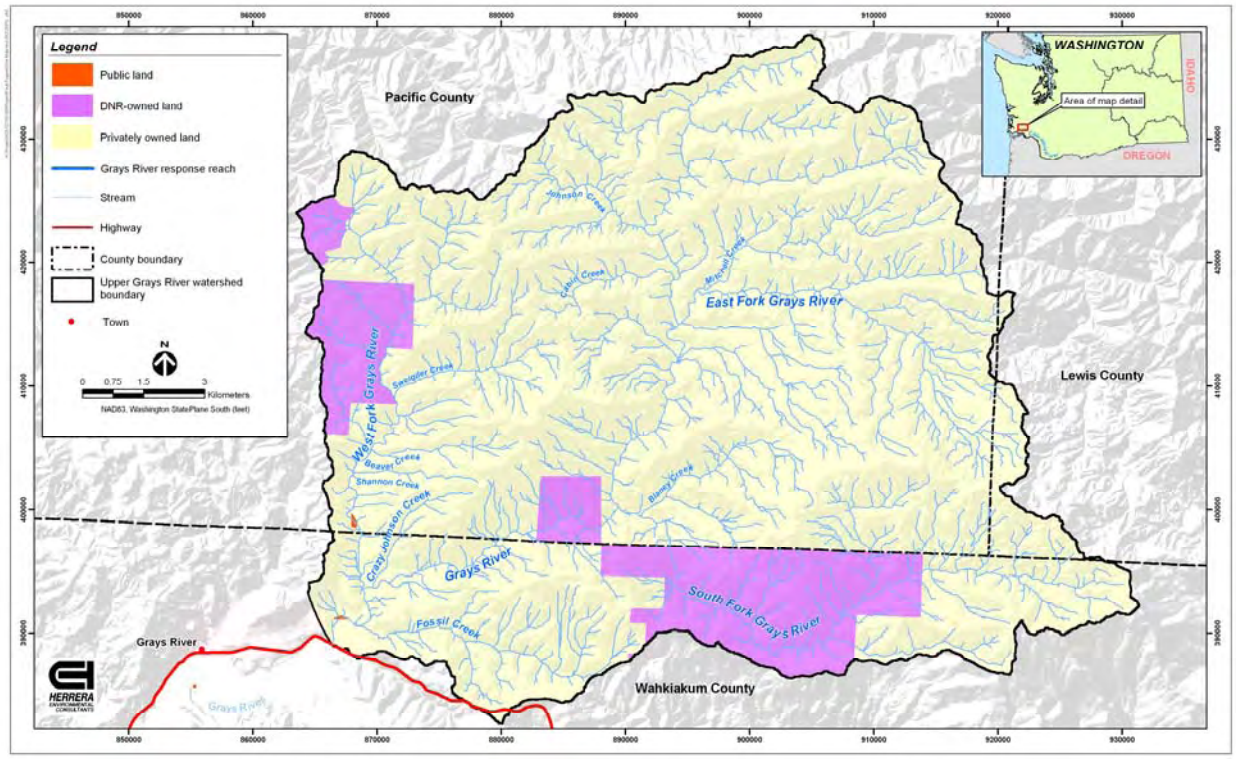


Figure 2.4. Upper Grays River Watershed Land Ownership

The hydrologic conditions of the watershed can have a significant impact on channel geomorphology and instream habitat conditions, as well as influencing the overall quantity and quality of available habitat. The hydrologic regime can be influenced by both upstream and local subbasin conditions. As is the case with hydrology, sediment supply and transport can also have a significant impact on channel geomorphology and instream habitat conditions. Excessive instream sedimentation has been shown to cause degradation of salmonid spawning habitat. Elevated sediment loading from hillslopes has been linked to land-use activities such as timber harvest and logging roads. In addition, these same land-use activities can result in an increase in mass-wasting sediment delivery to stream systems. Natural sediment production (e.g., erodibility factor) and transport is influenced by geologic factors, soil type, vegetation, and other physiographic characteristics of the landscape. Erosional and hillslope processes largely determine the local sediment loading. The hydrologic analysis and geomorphic analysis components of the Grays River watershed assessment are covered in Chapters 3 and 4 of this report. The main landscape factors used in the hydrologic and geomorphic analyses include

- subbasin land cover and land use (Figure 2.6)
- subbasin road density (Figure 2.9)
- riparian fragmentation (Figure 2.10)
- riparian land cover and land use (Figure 2.11).

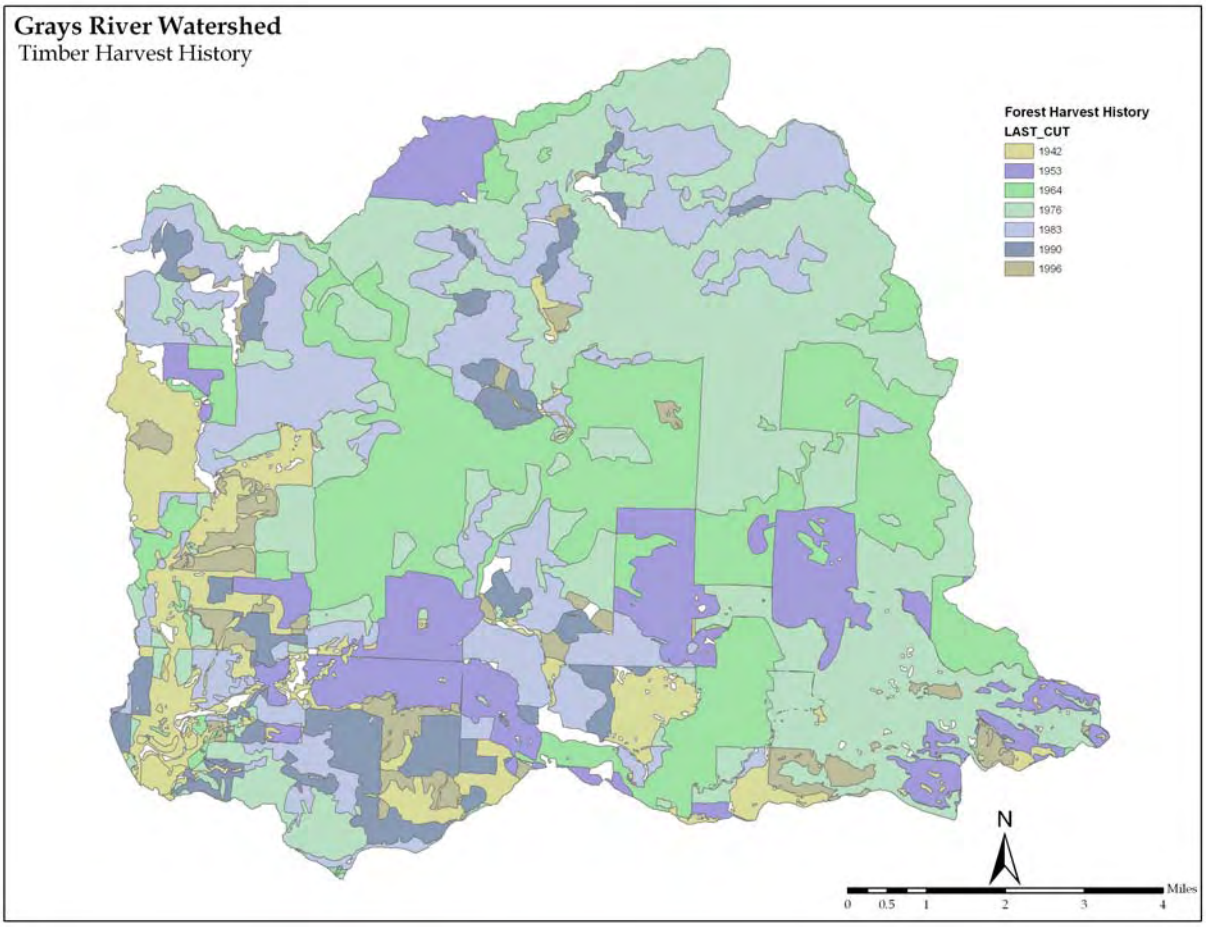


Figure 2.5. Upper Grays River Watershed Timber Harvest History

The Grays River riparian areas were delineated (buffered) using the GIS at two spatial scales (30 m and 100 m). Figure 2.11 illustrates the delineated (30-m) riparian zone. The riparian areas of the upper Grays River watershed are almost entirely composed of deciduous-dominated forest (mainly alder and maple). In most subbasins, the riparian zones have, until recently, been logged such that there is only a fraction of the native conifers (cedar, spruce, and hemlock) that originally dominated these streamside areas. The riparian condition is closely linked to large woody debris (LWD) supply and recruitment process. Of significance is the lack of very large conifers within most riparian corridors. This situation has a major influence on LWD recruitment potential, especially with respect to large, "key" pieces of LWD. Only in segments of the lower mainstem subbasin has the riparian zone been completely lost due to agricultural and rural-residential land-use activities. Even here, some remnants of the native riparian-floodplain forest still exist, most notably a few stands of large cottonwood upstream of the SR 4 highway bridge. In general, riparian conditions in the upper Grays River system mirror those found in other areas of the Pacific Northwest where past timber-harvest practices have encroached into the riparian corridor.

Riparian-condition analysis was conducted at the sub-basin scale. Table 2.2 summarizes the riparian corridor conditions for each subbasin in the study area based on field surveys and GIS analysis of landscape-scale data. All subbasin were classified as either *impaired* or *moderately impaired* due to the conditions described above. Riparian surveys conducted as part of the subbasin planning process

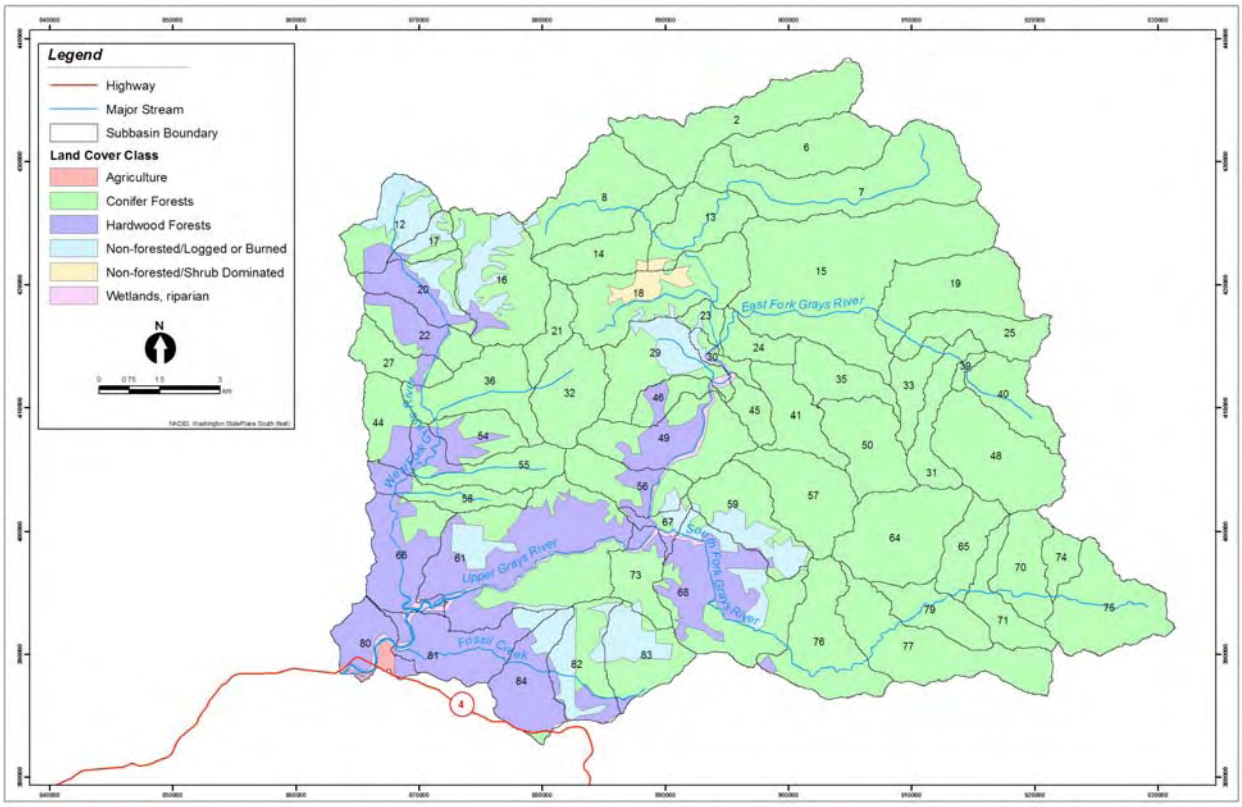


Figure 2.6. Upper Grays River Watershed Land-Cover Characteristics

Table 2.1. Land Cover in the Upper Grays River Watershed

Subbasin	Agriculture	Conifer Forest	Deciduous Forest	Cleared/Burned	Shrub	Wetland
East Fork	0.0%	100.0%	0.0%	0.0%	0.0%	0.0%
Fossil Creek	0.1%	23.3%	49.1%	26.3%	0.0%	1.2%
North Fork	0.0%	96.3%	0.0%	0.7%	3.0%	0.0%
South Fork	0.0%	85.3%	8.5%	5.7%	0.0%	0.5%
Upper Mainstem	0.0%	54.7%	31.5%	9.8%	0.0%	4.0%
Lower Mainstem	14.8%	0.0%	74.6%	0.0%	0.0%	10.5%
West Fork	0.0%	63.3%	25.0%	11.3%	0.0%	0.4%
Grays Total	0.2%	77.2%	14.6%	6.5%	0.5%	0.9%

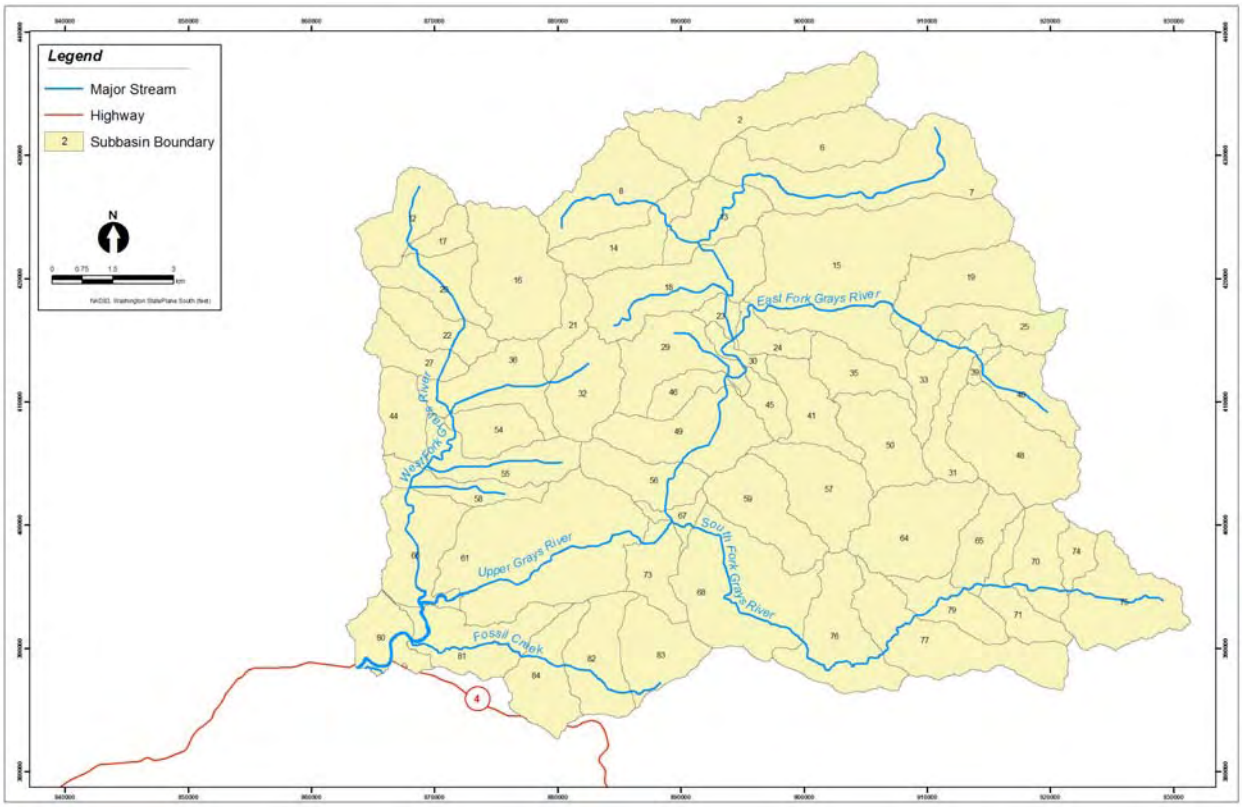


Figure 2.7. Upper Grays River Subbasins

(LCFRB 2004) also found that most of the riparian corridors in the upper Grays River basin were dominated by young deciduous forest (mostly alder and maple), and many were narrow and highly fragmented by logging roads and clear-cut areas, especially in headwater or seasonal channels.

Fragmentation of the riparian corridor is also of concern from a hydrologic and ecological perspective. For each of the road crossings shown in Figure 2.10, there is a culvert or bridge that provides a pathway for water, sediment, and organic debris (e.g., LWD) to pass under the road. In many cases, these structures also provide a route for fish passage upstream and/or downstream within the river network. If not designed, installed, and maintained properly, these structures can become migration barriers for aquatic biota. These stream-road crossings also fragment the riparian corridor, which can have an impact on the ecological integrity of the stream ecosystem. In general, the level of fragmentation is high, which is typical of intensively logged watersheds in the Pacific Northwest. Table 2.3 summarizes the riparian fragmentation conditions in the study area. In general, streams with more than one road crossing per kilometer of stream length are considered impaired (FEMAT 1993). Most of the streams in the upper Grays River watershed exceed this benchmark and are considered impaired with respect to riparian corridor fragmentation.

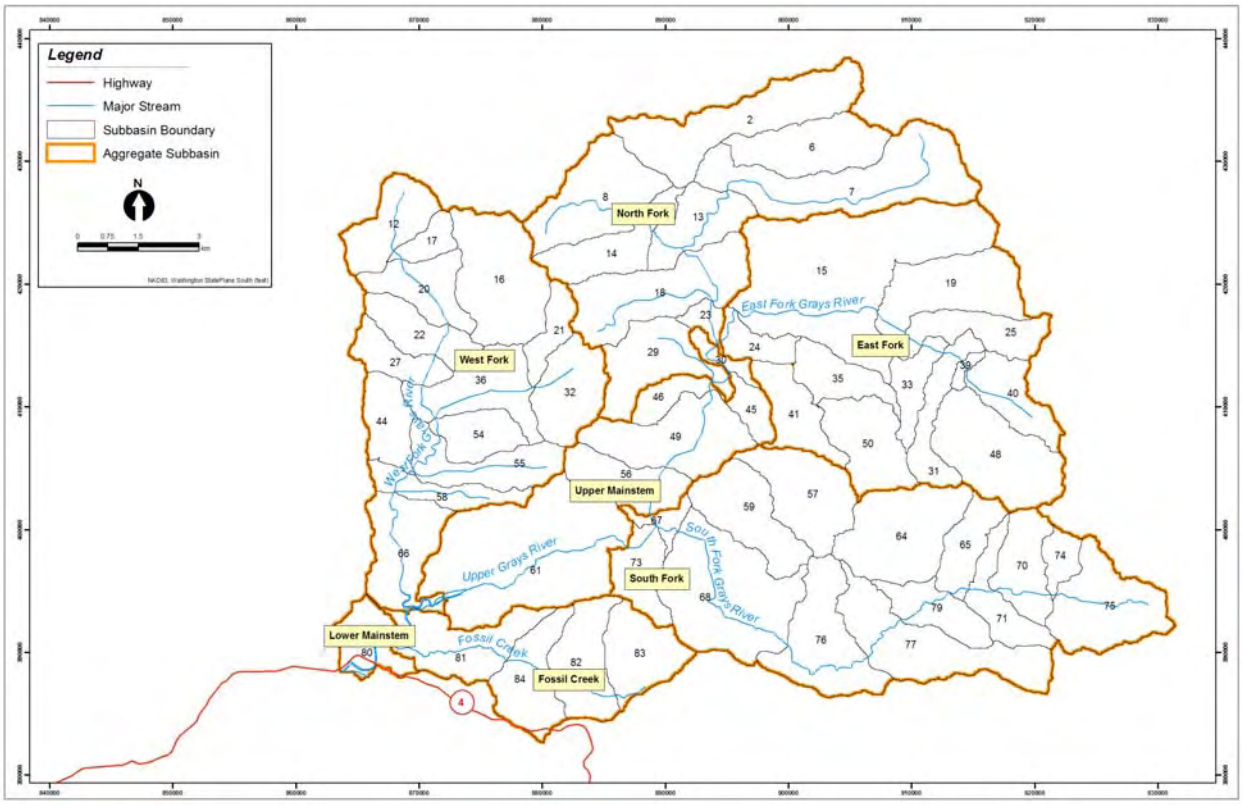


Figure 2.8. Upper Grays River Subwatersheds

Although the number of stream-road crossings is high within the watershed, data obtained from WA-DNR (2005) show relatively few fish passage barriers (i.e., impassable culverts) within the upper Grays River watershed. Most of the existing barriers are on small, high-gradient, headwater streams. There is one confirmed barrier (RAMP ID 24439) and several potential barriers on Mitchell Creek (a tributary of the North Fork). There is also an old diversion structure and a culvert that are fish-passage barriers on Alder Creek (RAMP IDs 3149 and 3224). The Alder Creek project is also scheduled to remove a flow diversion weir located on the upper mainstem that is likely a partial barrier during low-flow conditions. These barriers are scheduled for replacement in 2006–2007. There are also several potential barrier culverts on multiple tributaries of the West Fork and on Blaney Creek (a South Fork tributary). A perched culvert (fish-passage barrier) on an unnamed tributary of the upper mainstem (RAMP ID 24) under the mainline (7200) logging road at the junction of the 7210 and 7220 spur roads should also be replaced (WA-DNR 2005).

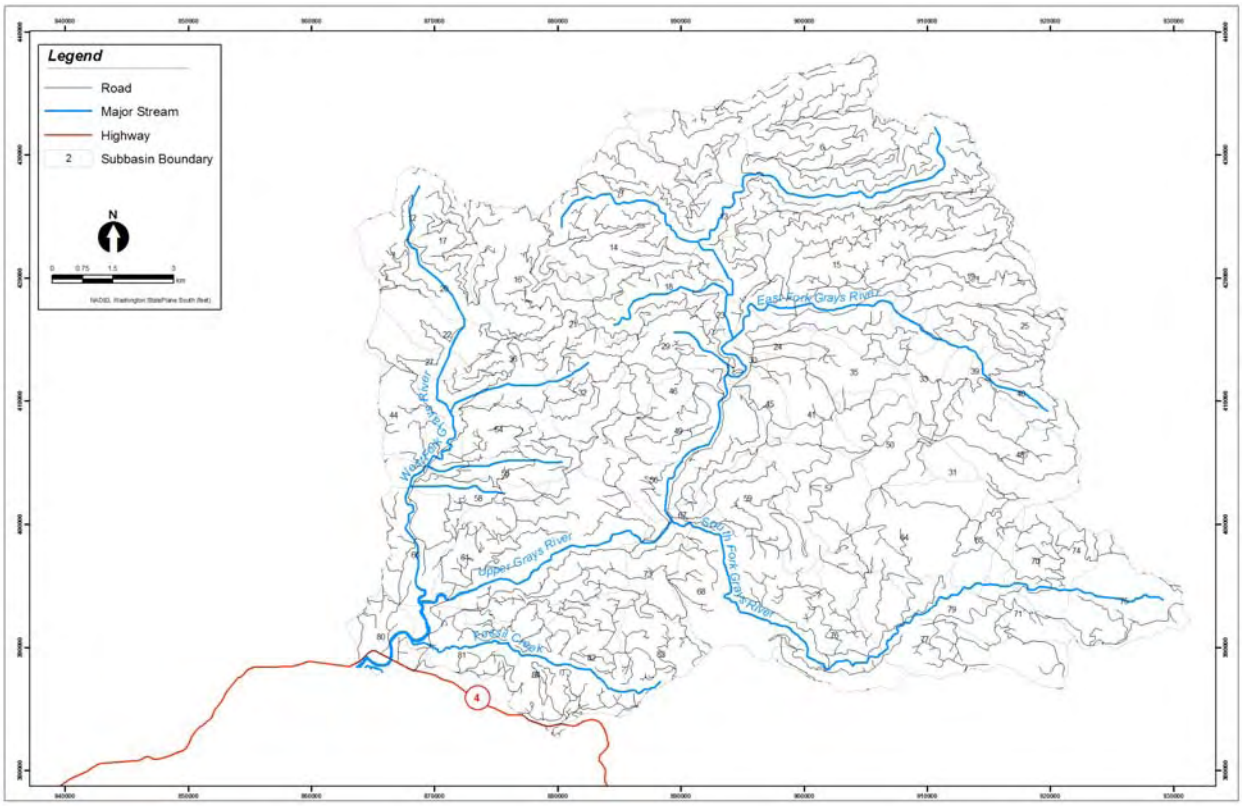


Figure 2.9. Road Network in the Upper Grays River Watershed

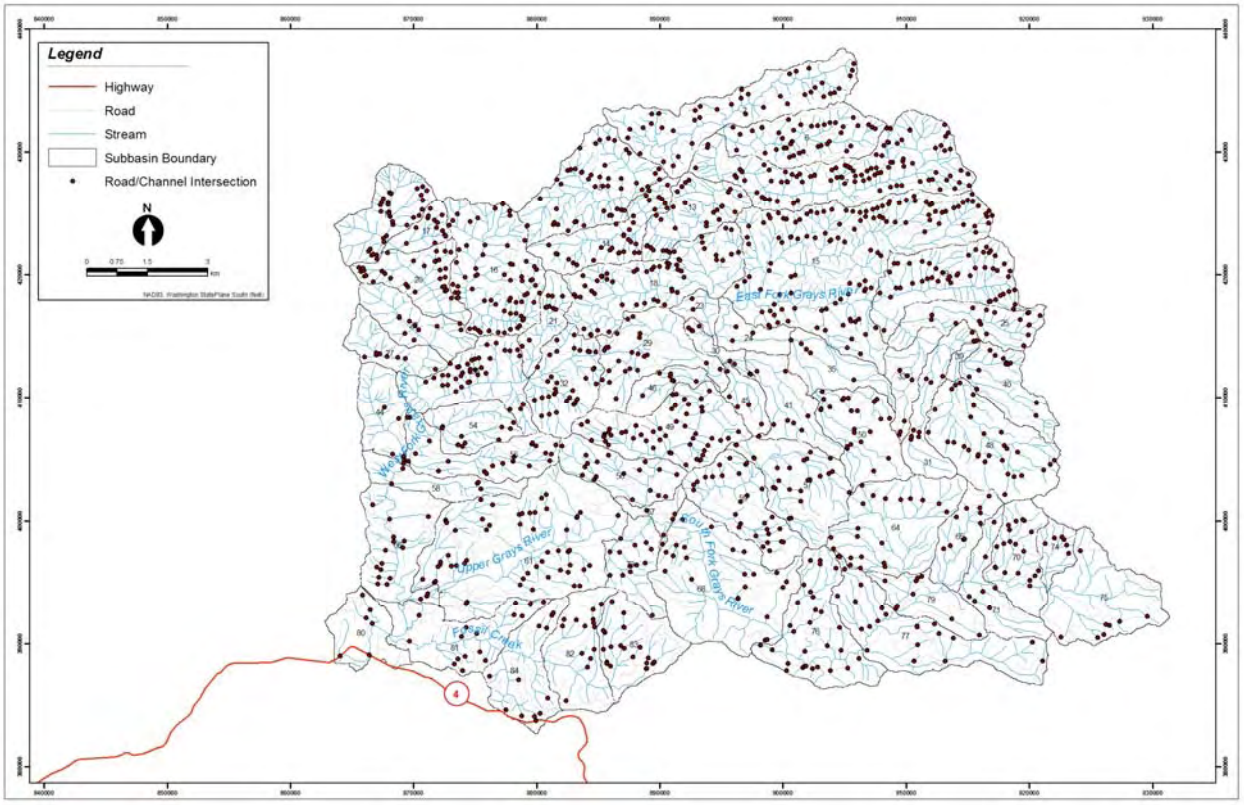


Figure 2.10. Stream-Road Crossings in the Upper Grays River Watershed

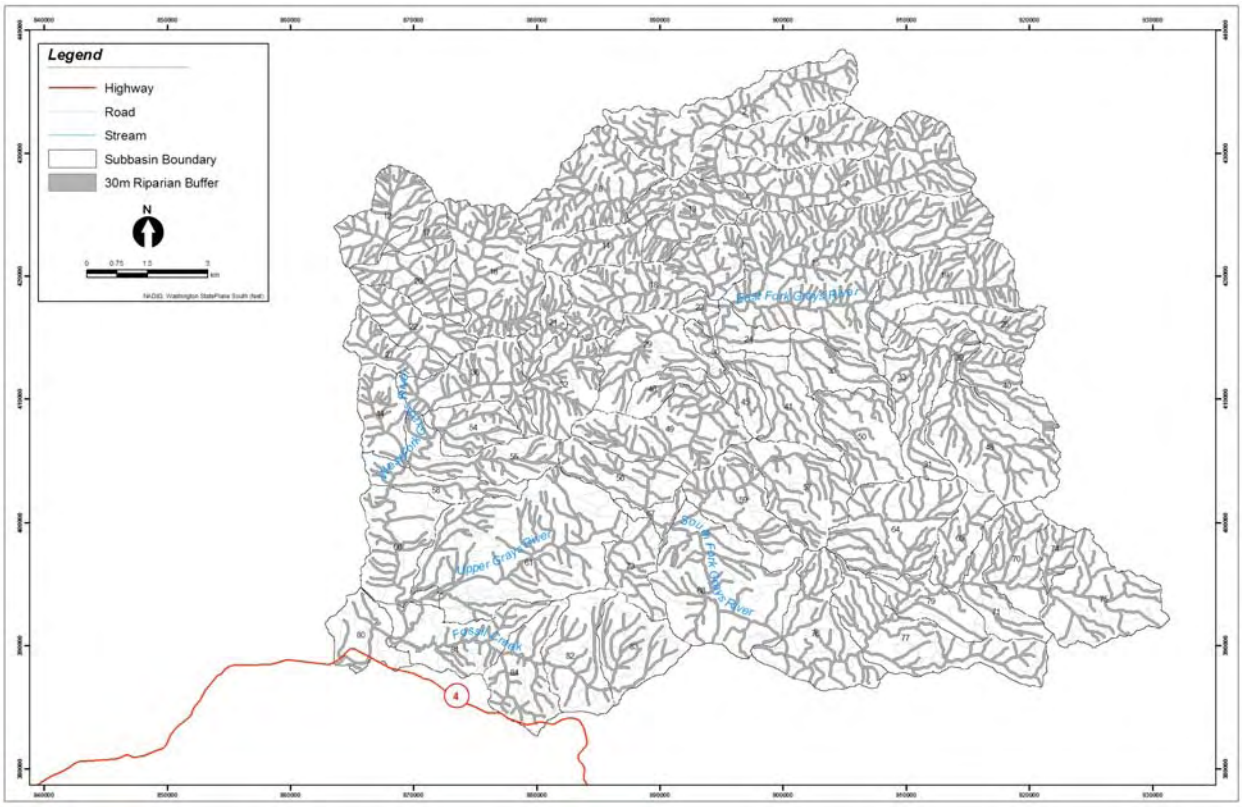


Figure 2.11. Upper Grays River Watershed Riparian Zones

Table 2.2. Riparian Corridor Conditions in the Upper Grays River Watershed

Subbasin	% Mature Coniferous Forest	Riparian Functional Condition
East Fork	40–60%	Moderately Impaired
North Fork	30–50%	Impaired
South Fork	40–60%	Moderately Impaired
Upper Mainstem	30–50%	Impaired
Fossil Creek	10–30%	Impaired
Lower Mainstem	<20%	Impaired
West Fork	20–40%	Impaired

Table 2.3. Stream-Road Crossings in the Upper Grays River Watershed

Subbasin	Number of Road Crossings	Stream Channel Length (km)	Fragmentation (road crossings/km)
East Fork	211	207.3	1.1
North Fork	279	193.9	1.4
South Fork	170	207.2	0.8
Upper Mainstem	126	108.7	1.2
Fossil Creek	41	51.7	0.8
Lower Mainstem	4	5.9	0.7
West Fork	234	201.8	1.2
Total	1,065	976.5	7.2

3.0 Grays River Watershed Hydrological Assessment

The objective of the hydrological assessment was to examine how timber harvest in the upper Grays River watershed has affected the natural hydrologic flow regime of the river system. The Distributed Hydrology Soil Vegetation Model (DHSVM) was selected to evaluate the impacts of harvest timing, intensity, and location on runoff and streamflow. The DHSVM is a process-based, continuous simulation model that can be used to evaluate the influence of land-use change over a range of flow regimes, from summer low flows to winter peak flows.

3.1 Background

The Grays River watershed typically receives rainfall from moist frontal systems originating in the Pacific Ocean. Precipitation records have been recorded at the Washington Department of Fish and Wildlife (WDFW) Grays River Salmon Hatchery since 1962. Approximately 77% of precipitation falls during the winter months from October through March (Figure 3.1). Annual precipitation measured at the hatchery for the period 1962 to 2004 ranged from 191 to 346 cm, with a mean of 279 cm (Figure 3.2). Precipitation increases with elevation in the watershed, from approximately 200 cm near the mouth of the Grays River to 300 cm in the upper watershed.

The variability in monthly precipitation for the period of record is shown in Figure 3.3. Although annual rainfall totals are similar, the distinct pattern of rainfall over a given year can differ greatly from year to year. For example, the rainfall total for 1998 resulted from a number of moderately high monthly totals, whereas rainfall in 1999 began and ended with consecutive months of very high rainfall, thereby displaying a similar monthly average but one that results from greater variability (Figure 3.3).

Limited discharge data are available for the Grays River watershed. Until recently, there have been no active streamflow gauges in the Grays River watershed since the 1970s. However, between the 1950s and 1970s, four U.S. Geological Survey (USGS) gauges were active within the Grays River watershed. These historical streamflow gauges were sited on the South Fork Grays River (24 years of flow data from 1956 to 1979), the mainstem Grays River just downstream of its confluence with the South Fork (5 years of flow data from 1956 to 1960), the West Fork Grays River (21 years of flow data from 1949 to 1969), and the mainstem Grays River near the town of Grays River, Washington. In 2004, a new streamflow gauge was established by the Washington State Department of Ecology at the covered bridge just downstream of the SR 4 highway bridge.

Flood-frequency curves have been developed for two locations on the river (West Consultants 2004). This analysis was conducted under contract for the Grays River Habitat Enhancement District. An analysis of streamflow data from the available gauges and correlation with long-term streamflow records from the nearby Naselle River watershed provide the best available estimate of flood frequency values for the Grays River (West Consultants 2004). The results from three analytical methods were compared to determine the most appropriate method of estimating flood frequency values for the mainstem of the Grays River at SR 4 (West Consultants 2004). Flood frequency values were calculated using the data area transfer procedure, the USGS regional regression equation, and a localized regression analysis (West Consultants 2004). Evaluating the produced results and assumptions inherent in using these methods

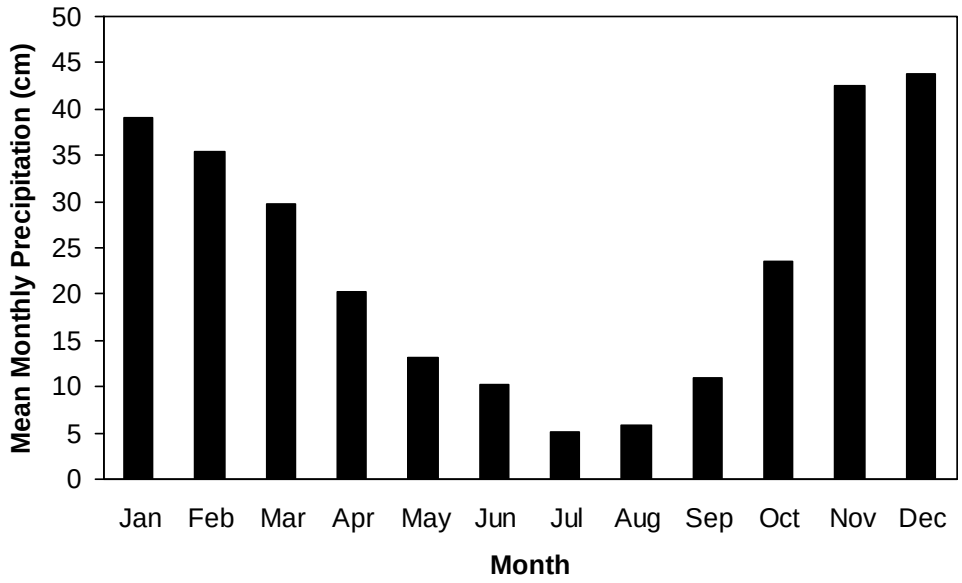


Figure 3.1. Mean Monthly Rainfall Totals in the Grays River Watershed at the WDFW Weather Station on the West Fork Grays River Based on Data from 1962 Through 2004

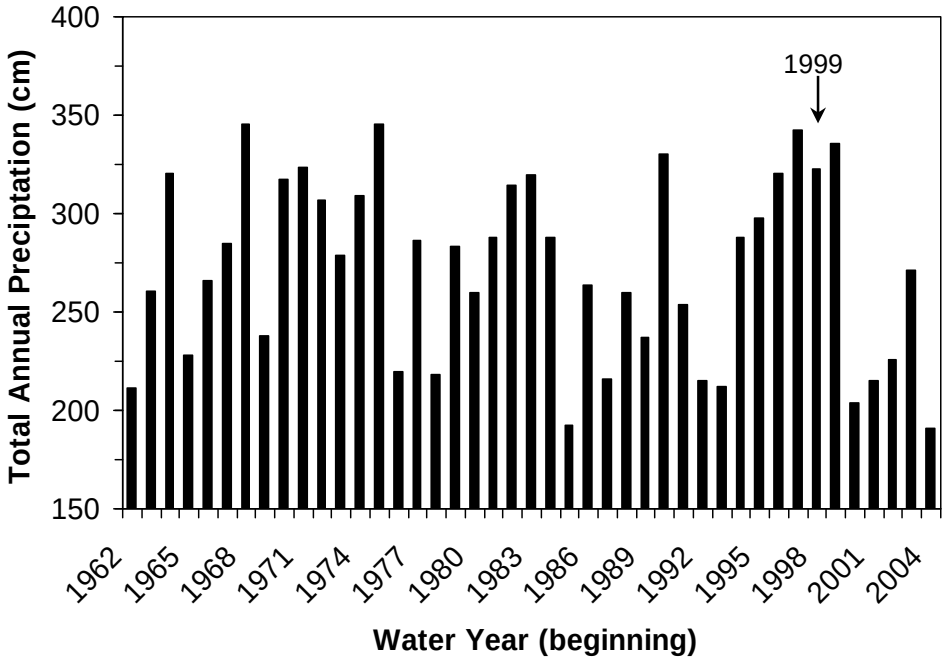


Figure 3.2. Annual Rainfall Totals in the Grays River Watershed at the WDFW Weather Station on the West Fork Grays River from 1962 Through 2004

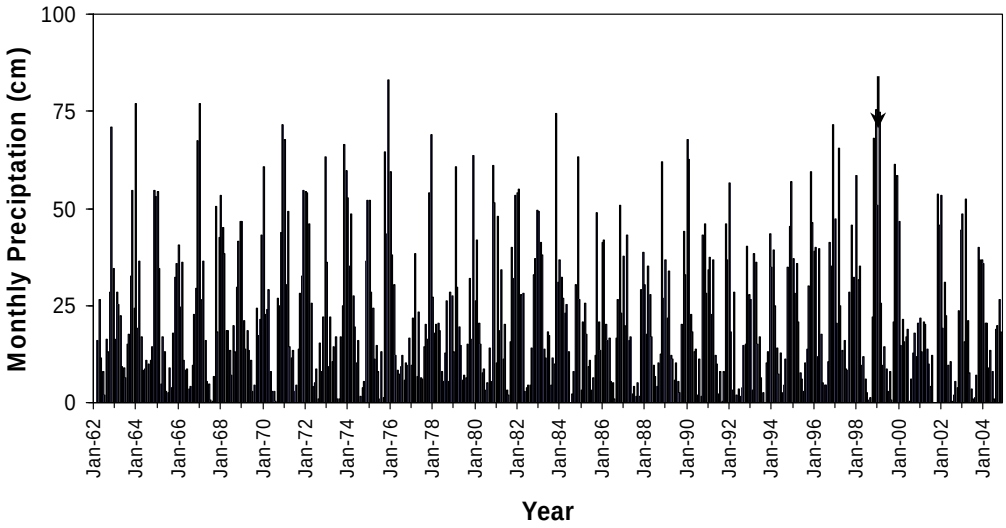


Figure 3.3. Monthly Rainfall Totals in the Grays River Watershed at the WDFW Weather Station on the West Fork Grays River from 1962 Through 2004

indicated that the data transfer procedure produced the most reliable flood frequency values when applied to the Naselle River gauge near Naselle, Washington (West Consultants 2004). Table 3.1 summarizes the results of the flood frequency analysis. Figure 3.4 shows a comparison between Grays River and Naselle River (USGS Gauge 12010000) streamflow data for selected years.

3.2 Methods

The DHSVM is a physically based distributed parameter model that provides an integrated representation of watershed processes at the spatial scale described by digital elevation model (DEM) data (Wigmosta et al. 2002). The DHSVM has been used in a number of studies that focused on prediction of streamflow, evapotranspiration and other surface energy and moisture fluxes (Wigmosta et al. 1994; Haddeland and Lettenmaier 1995; Kenward and Lettenmaier 1997; Dubin 1999; Wigmosta and Lettenmaier 1999; Kenward et al. 2000; Westrick et al. 2000, 2001, 2002). The model has been used also to study the interactions between climate and hydrology (Wigmosta et al. 1995; Arola and Lettenmaier 1996; Nijssen et al. 1997) and the potential impacts of climate change on water resources (Leung et al. 1996; Leung and Wigmosta 1999; Wigmosta and Leung 2001). Furthermore, the model has proven to be an important tool

Table 3.1. Estimated Flood Frequency Discharge for the Mainstem Grays River at SR 4, Estimated Using Data Transfer Procedure and Naselle River Gauge Data

Recurrence Interval	Discharge (cfs)
2-year	8,590
10-year	14,300
50-year	18,500
100-year	20,200
500-year	24,300
Source: West Consultants (2004).	

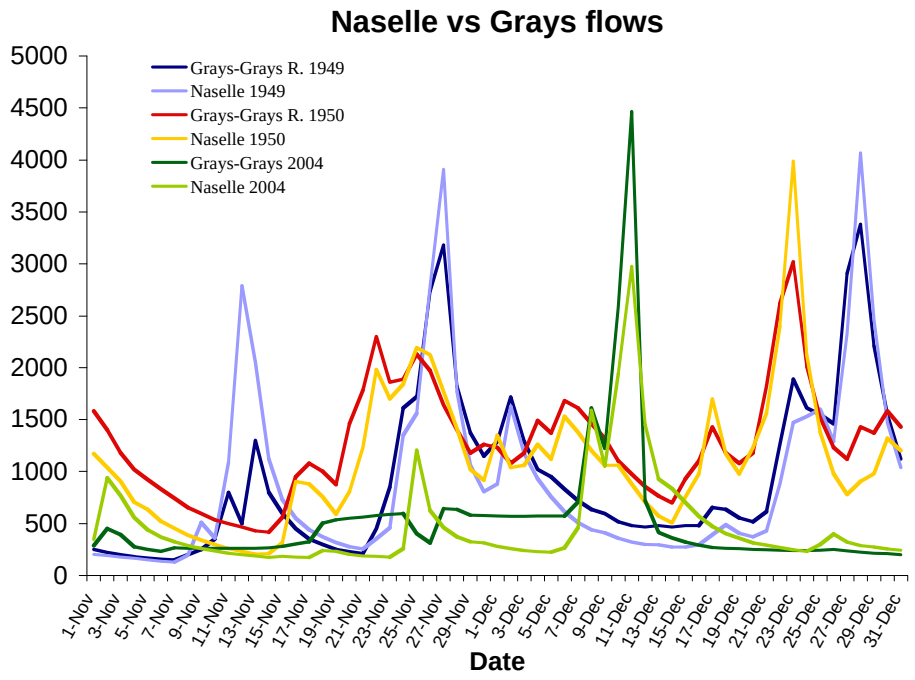


Figure 3.4. Comparison of Grays River and Naselle River Streamflow Data

for assessment of the effects of forest management activities on watershed processes—especially floods (Storck et al. 1995; Lamarche and Lettenmaier 1998, 2001; Storck et al. 1998, 1999; Bowling et al. 2000; Storck 2000; Bowling and Lettenmaier 2001; Wigmosta and Perkins 2001; Vanshaar and Lettenmaier 2001; Vanshaar et al. 2002; Waichler et al. 2005).

Waichler et al. (2005) conducted a detailed test of the DHSVM in small watersheds in the H. J. Andrews Experimental Forest, Oregon, examining the model's skill in simulating observed forest treatment effects on streamflow. These watersheds, located in the rain–snow transition zone, underwent road and clearcut treatments during 1959–1966 and subsequent natural regeneration. The DHSVM was applied with 10-m and 1-hr resolution to most of the period of record, 1958–1998. Overall efficiency in simulating hourly streamflow was good, exceeding 0.7 as measured by the Nash-Sutcliffe goodness-of-fit measure where 1.0 indicates perfect simulation of observed data (Nash and Sutcliffe 1970). In the analysis, mean annual error was less than 10% and statistical analyses of simulated and observed peak flows yielded similar characterizations of treatment effects. This was the first test of the DHSVM against observations of both control and treated watersheds in a classic paired-basin study involving a long period of forest regrowth and hydrologic recovery.

For the Grays River analysis, the DHSVM was used in a scenario method that involves comparing the difference between model outputs where the land-cover state in each scenario is held constant through time. Nine land-cover scenarios based on historical data were used to represent the history and intensity of harvest activities in the basin. The model was run for each scenario using the same 43-year record of meteorological data from the Grays River Hatchery. This approach is not equivalent to a historical simulation because land use is held constant. The major advantage to this approach is that all scenarios have been affected by the same sequence of weather and climate conditions; the differencing controls for these effects and allows the impacts of land-use change to be isolated. Output from the DHSVM for each scenario was obtained at specified output sites located at key watershed subbasin outlets (Figure 3.5).

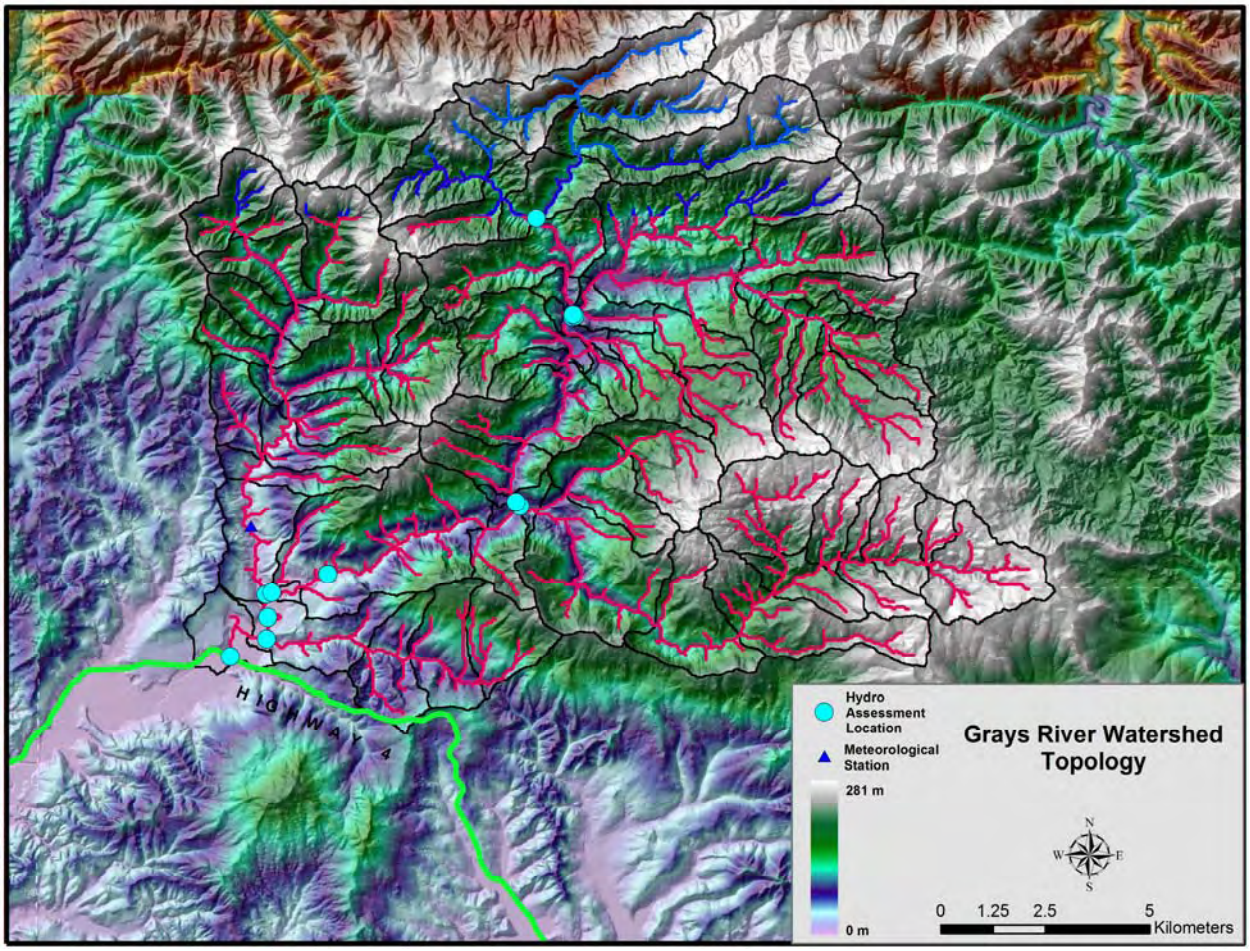


Figure 3.5. Upper Grays River Watershed Hydrologic Model (DHSVM) Coverage. Blue lines show the stream network derived from the underlying digital elevation data obtained from the U.S. Geological Survey 1:24,000 7½-minute quadrangles. Black lines delineate the subbasin boundaries. Yellow circles denote outlet points used in the hydrological assessment.

Daily meteorological data for the Grays River Hatchery cooperative station, obtained from the Western Regional Climate Center, were used to provide meteorological input to the DHSVM. This is the only meteorological station located within the Grays River watershed with a long period of record. Data at this station consist of precipitation and minimum and maximum air temperature. Daily meteorological data from the Grays River Hatchery station were disaggregated to a three-hourly timescale for use by the DHSVM. Solar radiation and atmospheric long-wave radiation required for the DHSVM were estimated from sun-earth geometry and corrected for atmospheric effects using relationships based on diurnal air temperature ranges.

The DHSVM provides a dynamic representation of watershed processes at the spatial scale described by DEM data (typically 10–90 m horizontal resolution). The modeled landscape is divided into computational grid cells centered on DEM nodes (Figure 3.6). This characterization of topography is used to model topographic controls on absorbed shortwave radiation, precipitation, air temperature, and downslope moisture redistribution. Vegetation characteristics and soil properties are assigned to each model

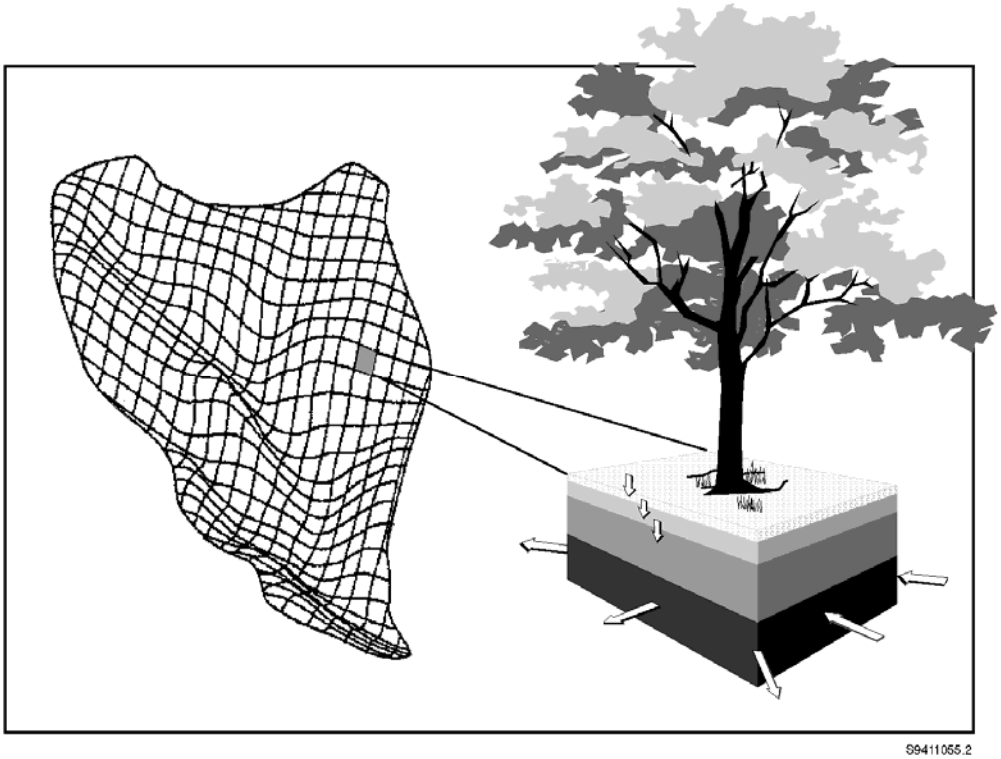


Figure 3.6. Model Representation of a Watershed. DEM data are used to model topographic controls on absorbed solar radiation, precipitation, air temperature, and downslope water movement. Grid cells are allowed to exchange water with their adjacent neighbors, resulting in a three-dimensional redistribution of surface and subsurface water across the landscape (Wigmosta et al. 2002).

grid cell and may vary spatially throughout the basin. In each grid cell, the modeled land surface can be composed of a combination of vegetation and soil. At each time step, the model provides simultaneous solutions to energy and water balance equations for every grid cell in the watershed. Individual grid cells are hydrologically linked through surface and subsurface flow routing. The model predicts temporal and spatial changes in hydrological properties such as snow cover and depth, soil moisture, ground saturation, soil temperature, subsurface water movement, recharge to deep groundwater, surface runoff, and simplified one-dimensional channel flow.

Canopy snow interception and release is modeled by the DHSVM using a one-layer mass-and-energy-balance model. Snow accumulation and melt below the canopy (or in the open) are simulated using a two-layer mass-and-energy-balance model that explicitly incorporates the effects of topography and vegetation cover on the energy and mass exchange at the snow surface. Evapotranspiration is represented using a two-layer canopy model with each layer partitioned into wet and dry areas. Unsaturated moisture movement through multiple rooting zone soil layers is calculated using Darcy's Law. Discharge from the lower rooting zone recharges the local (grid cell) water table (Figure 3.7). Each grid cell exchanges water with its adjacent neighbors as a function of local hydraulic conditions, resulting in a transient, three-dimensional representation of surface and saturated subsurface flow. Return flow and saturation overland flow are generated in locations where grid cell water tables intersect the ground surface. Flow in stream channels is routed using a cascade of linear channel reservoirs.

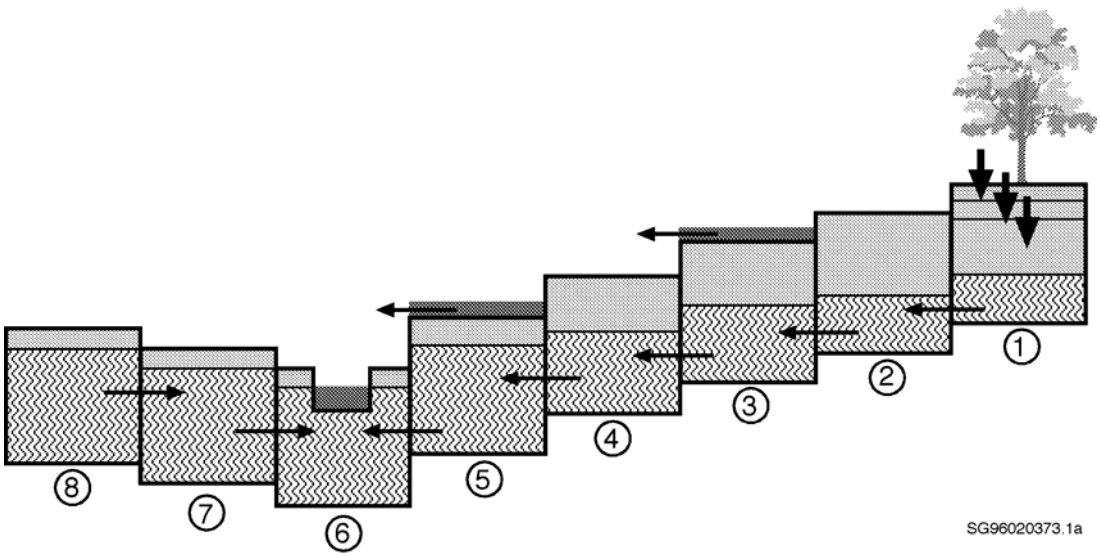


Figure 3.7. Model Representation of Downslope Water Movement to a Stream Channel. Dark shading represents surface water in the form of overland flow (model cells 3 and 5) or as channel flow (model cell 6). Light shading represents unsaturated soil, while the wave pattern corresponds to saturated soil below the water table. In each grid cell, percolation from the lower rooting zone recharges local grid cell water tables (shown by the downward arrows in cell 1). Grid cells exchange water with adjacent neighbors resulting in the downslope movement of water (horizontal arrows) to stream channels. The stream channel receives subsurface flow when grid cell water tables rise above the elevation of the channel bed (model cell 6). Surface flow may be reinfiltrated in downslope grid cells where the zone of soil saturation is below the ground surface. Surface flow from cells 3 and 5 reinfiltrate in cells 4 and 6, respectively (Wigmosta et al. 2002).

Digital elevation data for the upper Grays River watershed were assembled by mosaicking USGS 7½-minute quadrangles that contained elevation data at 10-m spatial resolution. The mosaicked 10-m DEM was aggregated to produce a 60-m DEM by averaging elevation values of individual 10-m grid cells. This aggregation was necessary to hold the grid cells in DHSVM simulations to a manageable number and at the same time to provide sufficient spatial detail for fully distributed hydrological modeling. The number of grid cells within the upper Grays River watershed was 64,252. Digital terrain processing algorithms (Tarboton 2000) were used on the 60-m DEM to automatically delineate stream network and corresponding subbasins for the upper Grays River watershed (see Figure 3.5). Several of the terrain data layers are used to provide input to the DHSVM.

Data from the Soil Survey Geographic (SSURGO) Database available from the Natural Resources Conservation Service (NRCS) were used in the hydrological assessment to parameterize soil type distribution and parameter estimates within the upper Grays River watershed. SSURGO data mapping scales generally range from 1:12,000 to 1:63,360. It is the most detailed level of soil mapping carried out by the NRCS. SSURGO Data Use Information (USDA 1995) was used to compute soil properties based on texture information included in the SSURGO data. These soil properties were used as baseline properties for the DHSVM. The soil map for the upper Grays River watershed is shown in Figure 3.8.

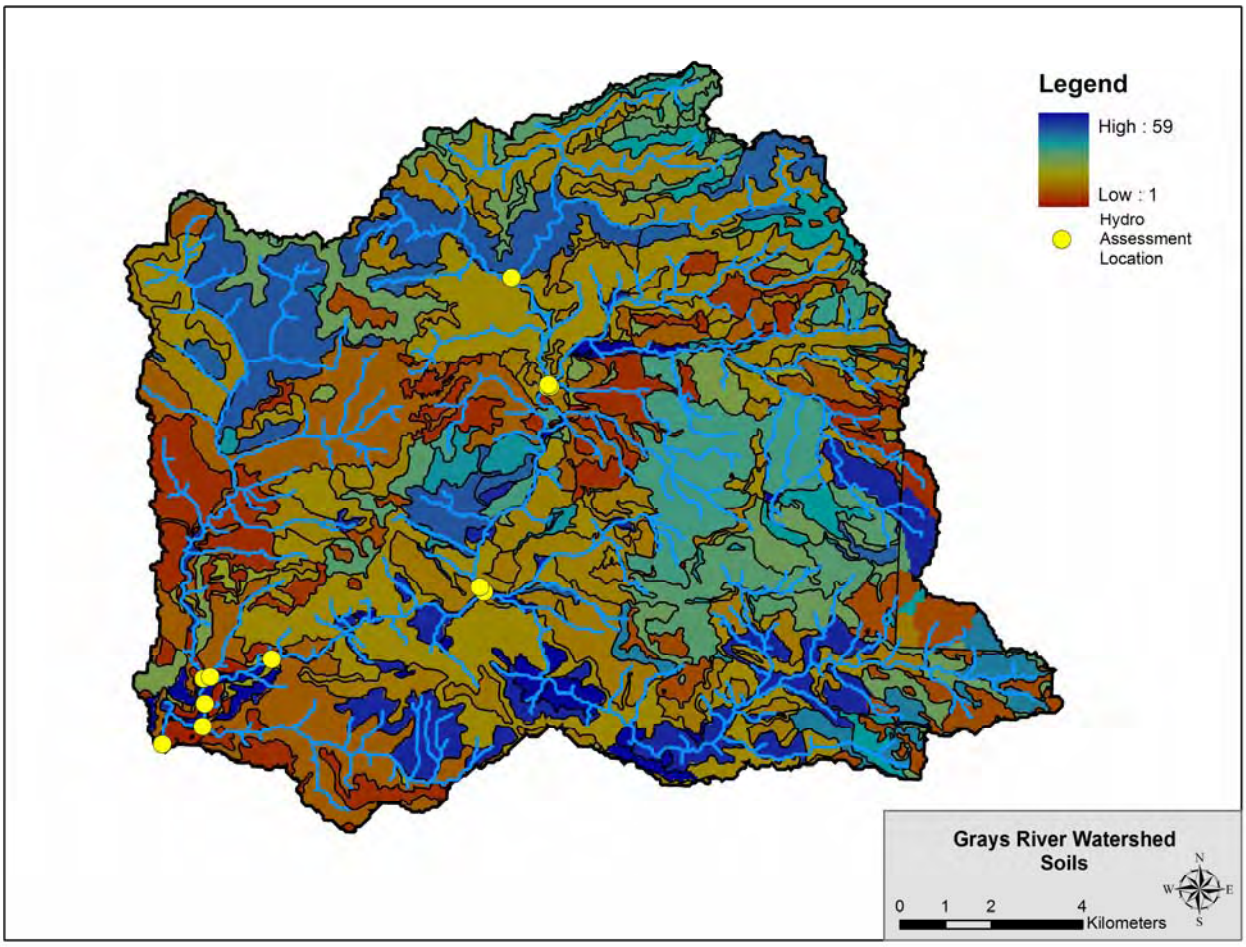


Figure 3.8. Distribution of Soils in the Upper Grays River Watershed. The map was created by integrating individual quadrangles of the SSURGO database.

The Grays River watershed has a significant history of forest harvest activity beginning as early as 1905 and continuing today. The harvesting activity in the watershed has varied through time, with a peak occurring in the 1976–1983 timeframe where 9.1 km^2/year were being extracted (Scott 2001). The original watershed was dominated by two primary vegetation zones consisting of 37% Sitka spruce and 62% western hemlock (Franklin and Dyrness 1969). Data used to reconstruct the forest harvest history were assembled largely by the use of historical aerial photography, forest history literature, and local observation (Scott 2001). A 1996 vegetation status was obtained using Landsat7 satellite imagery from the Washington State Gap Analysis project, which classified 30-m^2 multi-band image pixels to a possible 16 land-cover types within the Grays River watershed (WDFW 1996). Through the use of various methodologies, primarily aerial photography/imagery and satellite-based observations, spatial datasets of forest clearings were determined for the following years: 1942, 1953, 1964, 1976, 1983, 1990, and 1996. Maps showing the forest clearing history in these periods are shown in Figures 3.9 through 3.15.

In addition to the forest harvest scenarios described above, vegetation cover from the Washington Gap Analysis (WA-DNR 2001) was used to develop a pre-harvest scenario with fully mature conifers covering the entire watershed. This provided a baseline or reference scenario for the hydrological assessment.

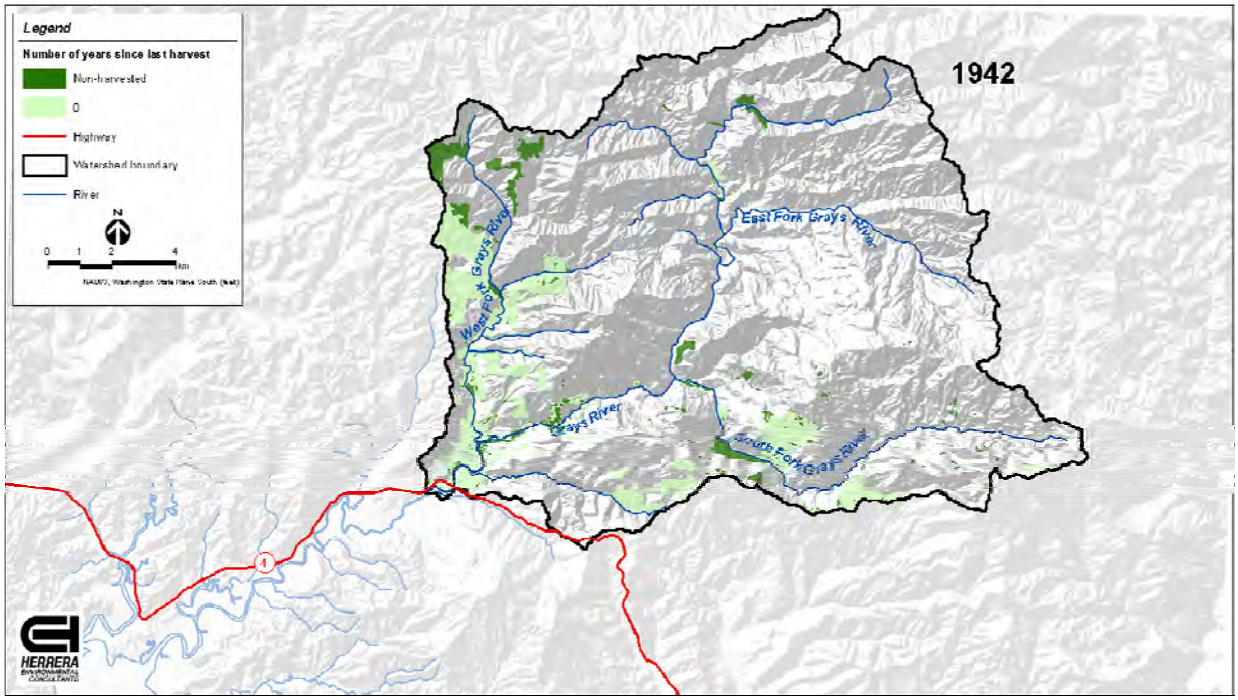


Figure 3.9. Forest Harvest History for 1942 (Herrera 2005)

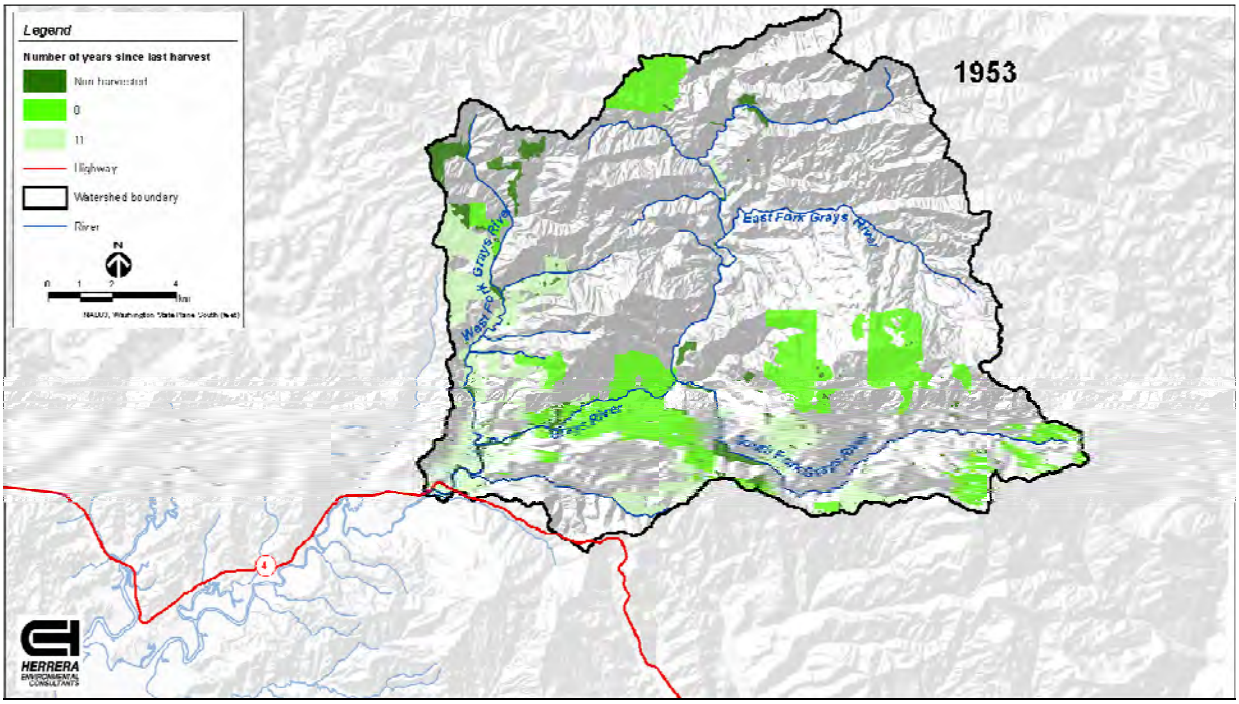


Figure 3.10. Forest Harvest History for 1953 (Herrera 2005)

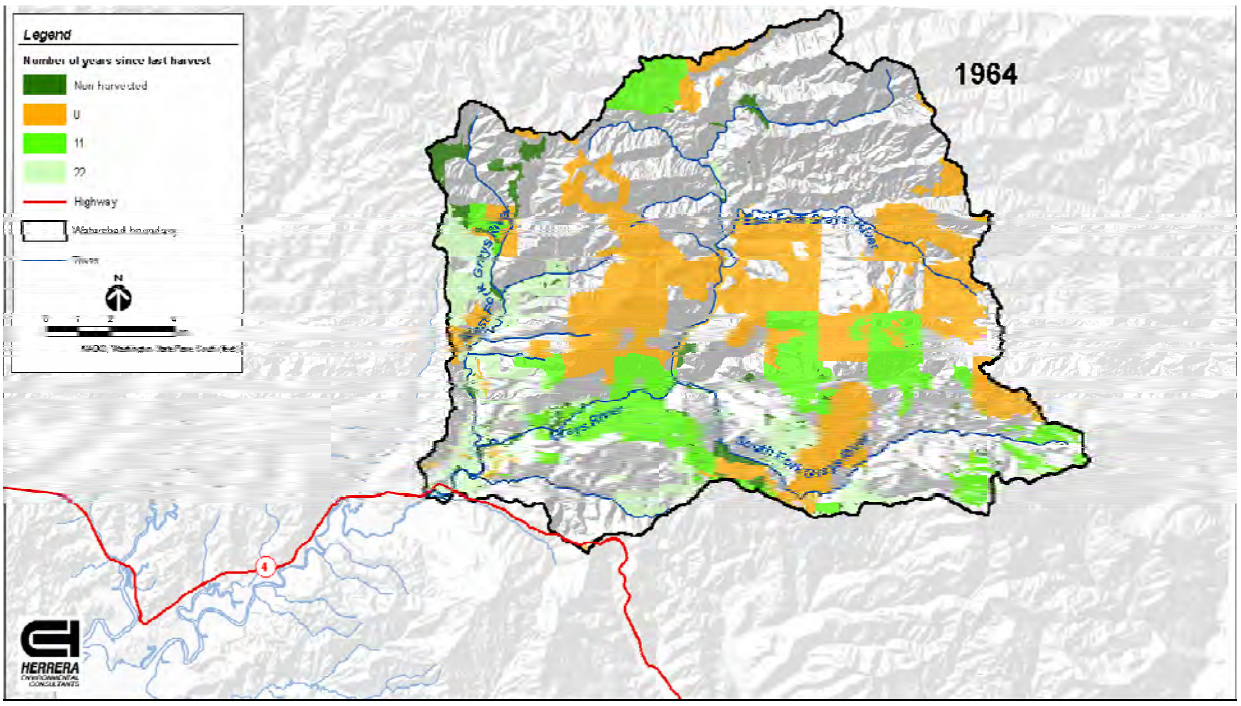


Figure 3.11. Forest Harvest History for 1964 (Herrera 2005)

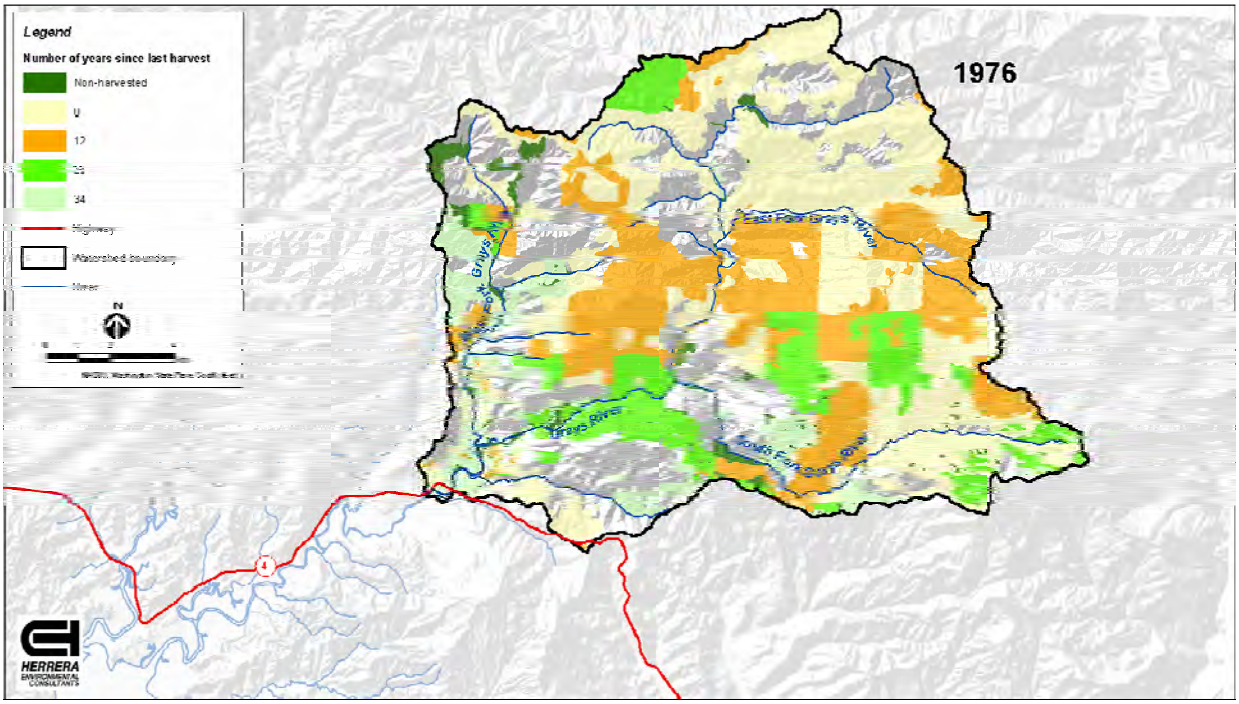


Figure 3.12. Forest Harvest History for 1976 (Herrera 2005)

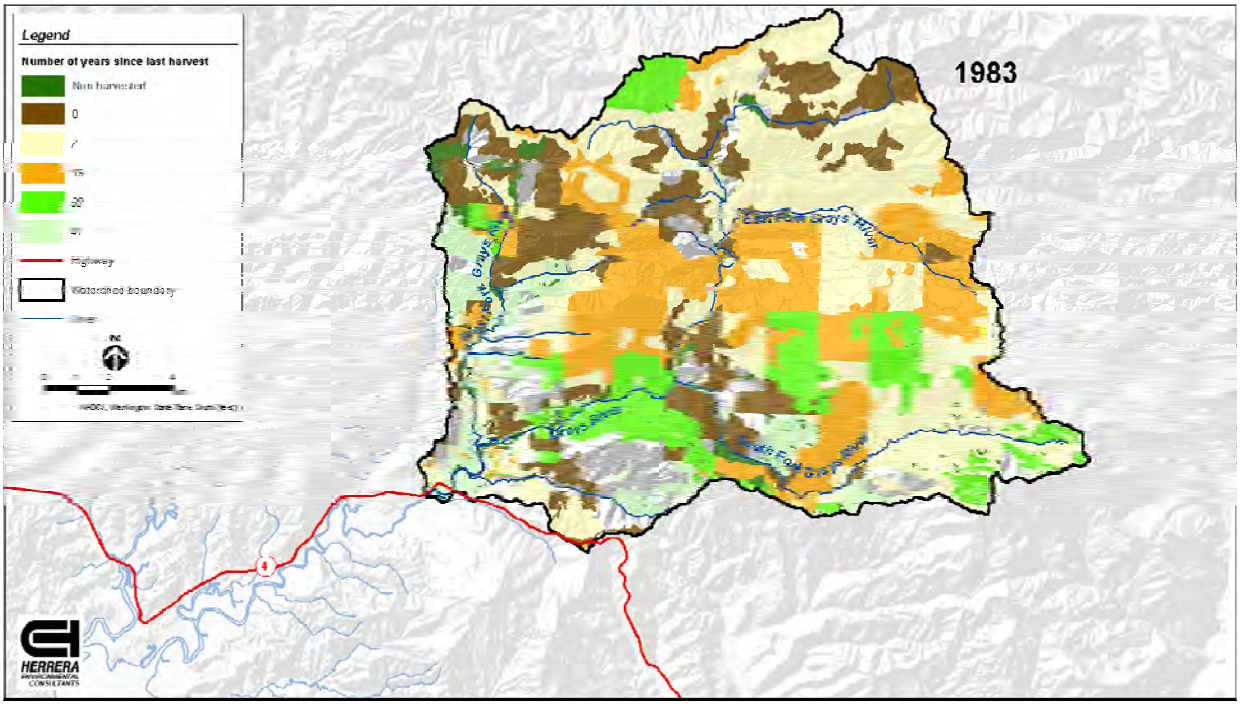


Figure 3.13. Forest Harvest History for 1983 (Herrera 2005)

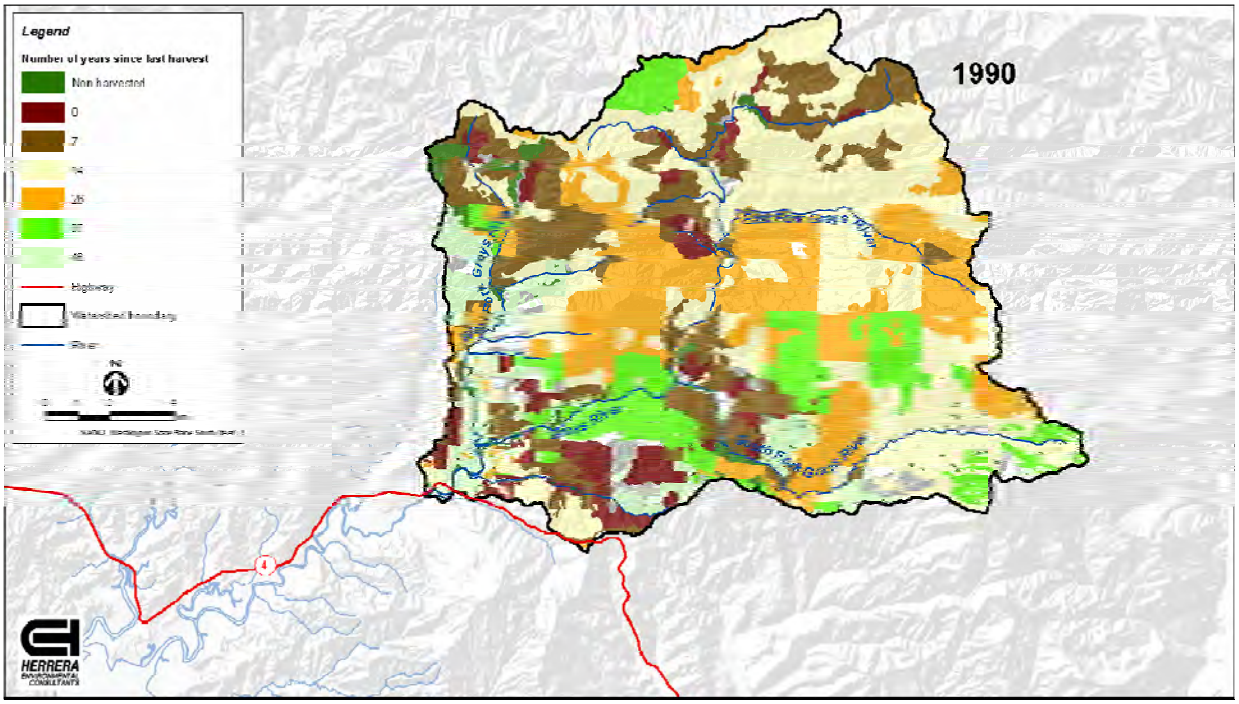


Figure 3.14. Forest Harvest History for 1990 (Herrera 2005)

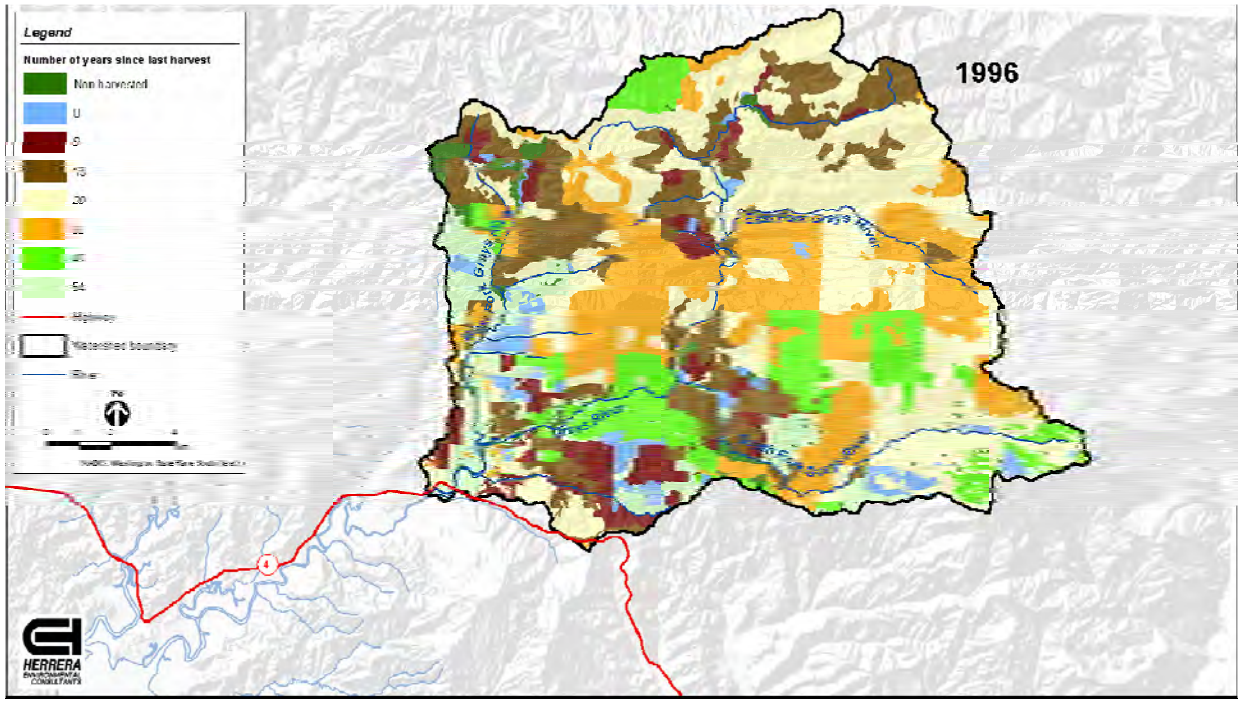


Figure 3.15. Forest Harvest History for 1996 (Herrera 2005)

The spatial forest harvest and regrowth history provides a key linkage to reconstructing the hydrological history of the watershed in terms of hydrograph response under varying landscape conditions. Overstory height and leaf area index provide dominant controls on the transfer of water and energy at the land surface. Using a methodology demonstrated at the H. J. Andrews Experimental Forest, Oregon, these two vegetation-based variables were estimated for a range of years from 1942–1996 and used as parameters to the DHSVM. First, a regrowing overstory leaf-area index (LAI), or the ratio of overstory leaves divided by the land surface area, was determined by using the following exponential growth function (Richards 1959; Duan 1996; Waichler et al. 2005):

$$LAI_o = LAI_{max}(1 - e^{-B_1 t}) \tag{3.1}$$

where $LAI_{max} = 11.25$, t is time in fractional years, and $B_1 = 0.065$. This derived variable provides an indicator of precipitation interception and rates of evapotranspiration.

The second derived vegetation-based variable was regrowing overstory height, which was calculated with the following exponential growth function (Richards 1959; Duan 1996; Waichler et al. 2005):

$$H_o = 1.0 + H_{max}(1 - e^{-B_1 t}) \tag{3.2}$$

where $H_{max} = 20$ m, t is time in fractional years, and $B_1 = 0.05$. This derived variable provides an estimate of forest stand heights according to the determined harvest histories. The constants provided for both LAI_{max} and H_{max} were determined to be values of old growth forest.

Using raster-based spatial processing methods in the GIS, ArcInfo version 9.1 (Environmental Research Systems Inc., Redlands, California), a series of scripts was developed to retrieve forest harvest history for each of the seven defined years and apply both the overstory leaf-area index and overstory height for the current year, while considering cut and growth histories for the previous known years (see Figures 3.9 through 3.15). Table 3.2 shows the attributes extracted from the GIS forest harvest history data and defines specific classifications for cut histories and land–cover or land-use changes used in the DHSVM.

The resulting GIS output data provided two 10-m resolution raster datasets, the LAI and vegetation height maps, for each of the seven defined years. These raster datasets match the extents and resolution for the existing DEM data used in the DHSVM. The LAI and vegetation height maps derived from forest harvest analysis were reclassified to produce 26 distinct vegetation types. The vegetation maps also were aggregated to 60-m resolution to match the spatial resolution of DHSVM simulations. These vegetation maps are shown in Figures 3.16 through 3.22. Table 3.3 contains the LAI and vegetation heights corresponding to the 26 vegetation types. Lighter shades denote more recent harvests that correspond to lower heights and LAI. Progressively darker shades during later years show forest patches that grew back to greater heights and acquired greater LAI values.

The DHSVM was set up to carry out fully distributed hydrological modeling in the upper Grays River watershed. The model was driven using the meteorological record at the Grays River Hatchery station for the period extending from October 1, 1962, through March 31, 2005, and parameterized using the SSURGO soils data and 1996 GAP vegetation data. The model parameters were manually calibrated using a visual comparison between modeled and measured streamflow at USGS gauge 14250500, West Fork Grays River near Grays River, Washington.

Nine DHSVM scenario simulations were carried out for the hydrological assessment. All scenario simulations spanned the same period (October 1, 1962, through March 31, 2005). The difference between individual scenario simulations was the vegetation scenario used as input to the DHSVM. These vegetation scenarios are 100% fully mature conifer cover (pre-harvest scenario), Washington Gap cover, and the seven forest harvest covers derived from the 1942, 1953, 1964, 1976, 1983, 1990, and 1996 information. The vegetation map or spatial distribution of vegetation within the upper Grays River watershed remained fixed in time during the corresponding DHSVM scenario simulations. Therefore, the set of DHSVM scenario simulations provides information regarding the effect of vegetation change on the hydrological response to the range of meteorological conditions observed within the watershed. These sets of runs are very useful to derive statistical measures related to magnitude, frequency, and durations of streamflow within the Grays River watershed. However, due to the vegetation scenarios remaining fixed during the 43-year simulations, streamflows simulated by the model cannot be compared directly to historical streamflow because vegetation cover was affected during this period due to logging activities. The DHSVM could have been used with varying vegetation patterns during the 43-year simulations. However, we chose to not adopt that approach. This latter simulation would have yielded just a few years' worth of streamflow corresponding to a particular vegetation scenario, thereby biasing the hydrological response to only those meteorological conditions that were observed during the few years that vegetation scenario existed and thus reducing the confidence in estimation of the range of hydrological response from DHSVM simulations for that vegetation scenario.

Table 3.2. Forest Harvest Geographic Information System Attributes Used To Derive Forest Harvest Vegetation Maps for Input to the Distributed Hydrology Soil Vegetation Model

Code	Definition
25	Natural growth
50	Cut before 1942
100	Cut between 1942-1953
200	Cut between 1953-1964
270	Was farmland in 1953-1964, now forest
300	Cut between 1964-1976
310	Cut in 1942 and again between 1964-1976
370	Farmland between 1964-1976, now in forest
390	River bar between 1964-1976, now forest
400	Cut between 1976-1983
410	First harvest between 1964-1976, second harvest between 1976-1983
420	First harvest between 1953-1964, second harvest between 1976-1983
430	Power Lines, first harvest between 1964-1976, second harvest between 1976-1983
470	Farmland up to 1976-1983, now in forest
500	Cut between 1983-1990
510	First harvest between 1942-1953, second harvest between 1983-1990
520	First harvest between 1964-1976, second harvest between 1983-1990
530	First harvest between 1953-1964, second harvest between 1983-1990
570	Farmland up to 1983-1990, now in forest
600	Cut between 1990-1996
610	First harvest between 1942-1953, second harvest between 1990-1996
620	First harvest between 1953-1964, second harvest between 1990-1996
630	First harvest between 1964-1976, second harvest between 1990-1996
700	Farmland
710	Harvest in 1942-1953, now in farmland
720	Harvest in 1953-1964, now in farmland
730	Harvest in 1964-1976, now in farmland
750	Harvest in 1983-1990, now in farmland
760	Harvest in 1990-1996, now in farmland
800	Residential
810	Harvest between 1942-1953, now residential
820	Harvest between 1953-1964, now residential
830	Harvest between 1964-1976, now residential
840	Harvest between 1976-1983, now residential
841	Harvest between 1942-1953, again between 1976-1983, now residential
843	Harvest between 1964-1976, again between 1976-1983, now residential
850	Harvest between 1983-1990, now residential
860	Harvest before 1942, again between 1990-1996, now residential
862	Harvest between 1953-1964, again between 1990-1996, now residential
900	River bars
910	Rock pit
913	Harvested between 1964-1976, now rock pit
914	Harvested between 1976-1983, now rock pit
915	Harvested between 1983-1990, now rock pit
916	Harvested between 1990-1996, now rock pit
924	Landslide, harvested between 1964-1976, now landslide
933	Pond, harvested between 1964-1976, now pond
950	Columbia River mudflats
999	Old growth
998	Old growth (from 1942 mapping)

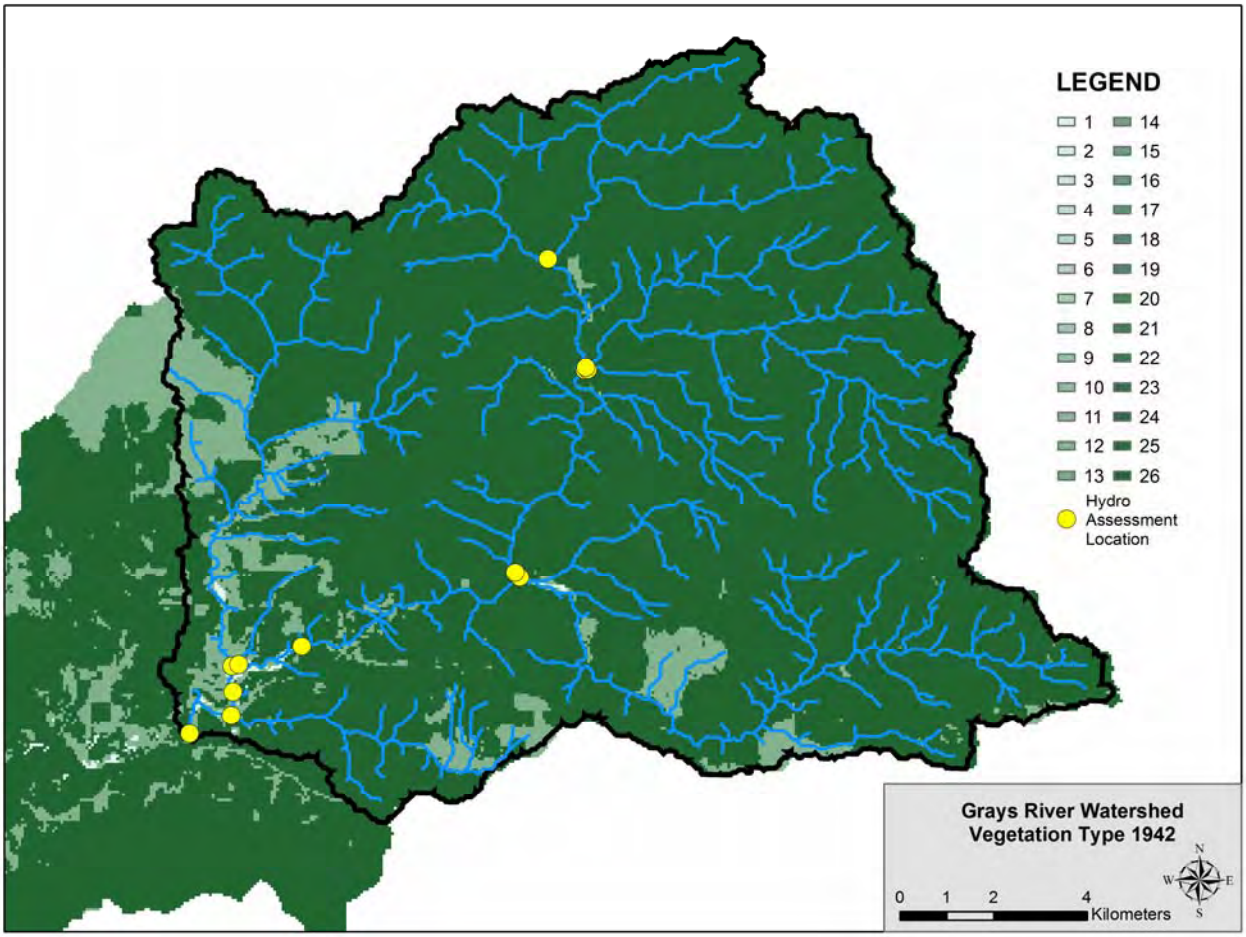


Figure 3.16. Vegetation Map Corresponding to the 1942 Forest Harvest Scenario

The nine DHSVM scenario simulations that spanned the entire period of meteorological records in the Grays River basin provide the complete range of hydrological response to the nine vegetation scenarios under the complete observed meteorological regime, thereby providing a much larger range (43 water-years) of hydrological response for a given vegetation scenario. The change in hydrological response under the nine vegetation scenarios was analyzed at a set of 13 points on the stream network (Figure 3.23). Names for each number location are provided in Table 3.4. The metric used for change detection was streamflow at several exceedance probabilities at each of the selected assessment locations.

Durations of floods and low flows at the hydrological assessment locations also were simulated. Durations were computed using a peaks-over-threshold method to derive the partial-duration series of floods using the mean daily simulated streamflow as described below. A similar approach, peaks-under-threshold, was used to derive the partial-duration series of low flows from mean daily simulated discharges.

3.3 Results

Table 3.5 shows percentages of the 26 vegetation types for the forest harvest scenarios. Lower vegetation type numbers represent shorter conifers with smaller LAI that are in various stages of regrowth depending on the time elapsed since the last harvesting activity in those areas. Table 3.5 summarizes the basin-wide patterns of harvesting at different times since 1942. It also shows the effects of regrowth; for

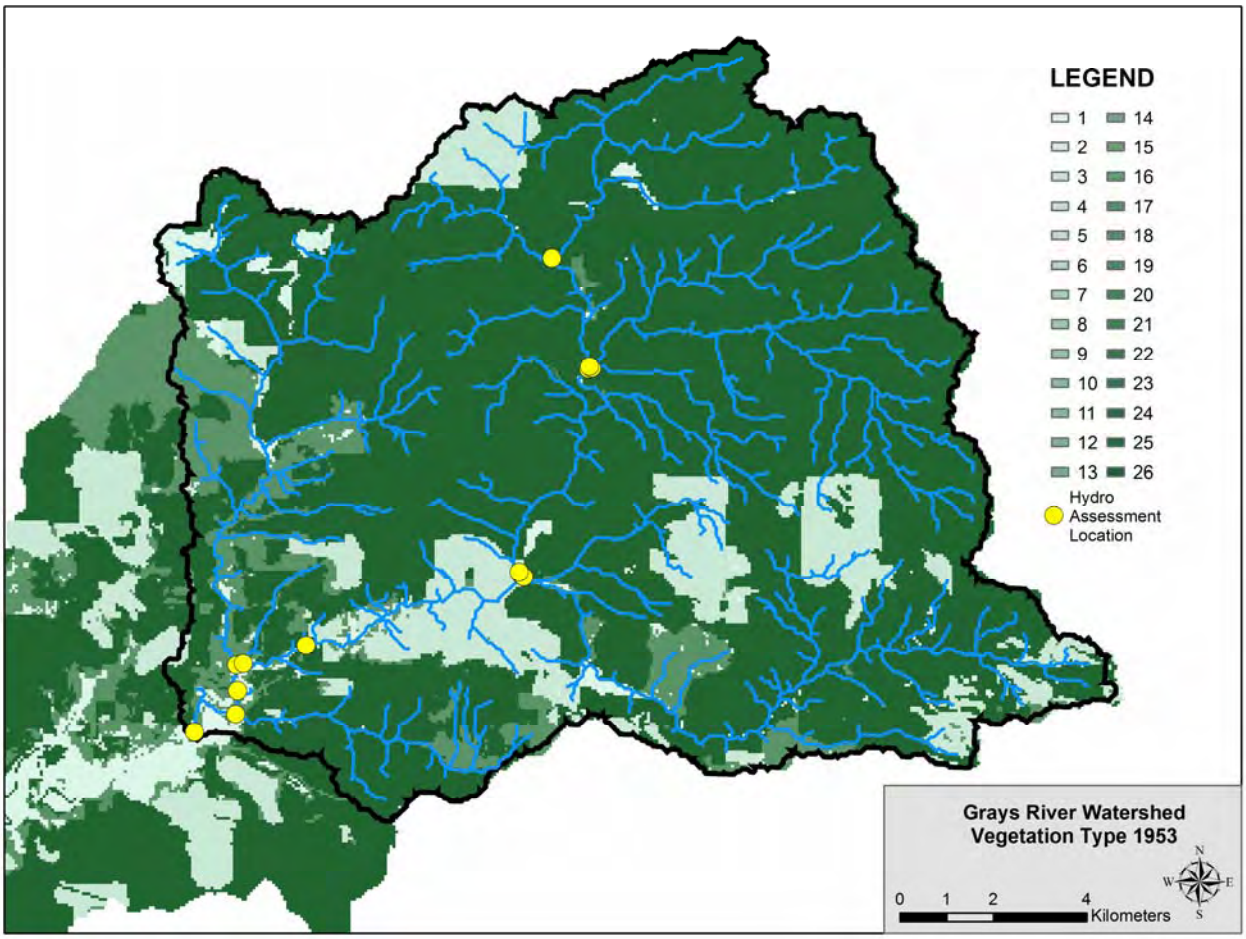


Figure 3.17. Vegetation Map Corresponding to the 1953 Forest Harvest Scenario

example, approximately 11% of vegetation type 12 in 1942 growing to vegetation type 17 in 1953, although the cut-regrowth pattern can become very complex in later years and may not be attributable to a single cause.

Changes in vegetation cover and associated parameters over time within the upper Grays River watershed due to forest harvest were shown in Figures 3.16 through 3.22. As of 1942, approximately 89% of the upper Grays River watershed was covered by mature conifers (vegetation type 26) and approximately 11% was covered with conifers of 65% of full height (vegetation type 12). In 1942, most of the harvest was along the West Fork and the South Fork. In 1953, harvesting progressed to the headwaters of West Fork, North Fork, and South Fork. Areas along the Grays River mainstem below the confluence of East and South Forks also were harvested. In 1964, much of the watershed was harvested, mainly the central areas. In 1964, approximately 54% of the watershed was covered by conifers 12 m or less in height, with 23% clear-cuts. The fully mature cover in 1964 was the smallest at 1.82%.

The most severe watershed-wide impact to vegetation height (and LAI) was in 1976, when the pattern of harvesting over the entire watershed continued with only 1.84% fully mature forest left in the watershed. In 1976, more than 75% of the watershed was covered with conifers 12 m or less in height (60% of mature height) with 28% clear-cuts. Since 1976, the watershed vegetation cover has recovered with the

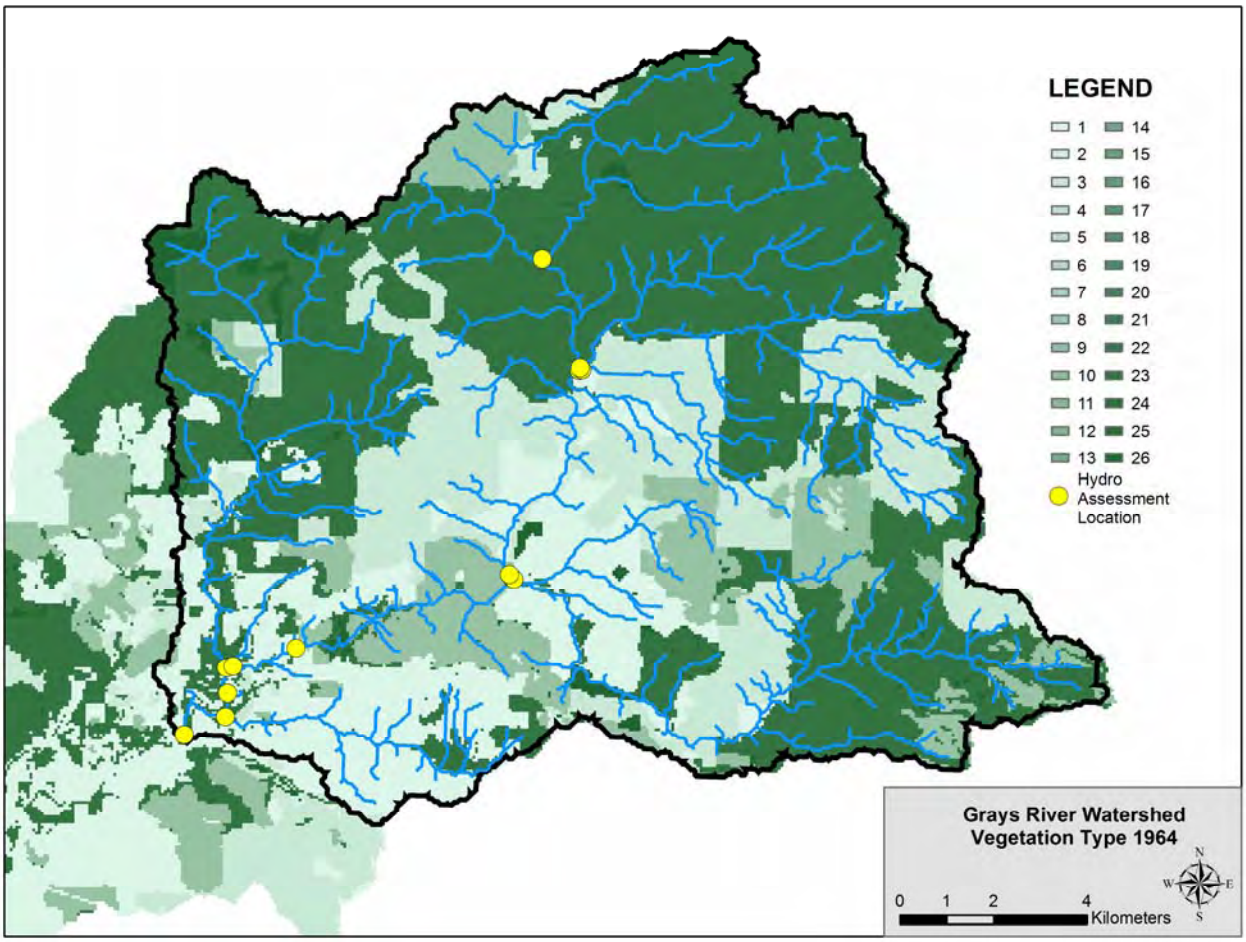


Figure 3.18. Vegetation Map Corresponding to the 1964 Forest Harvest Scenario

fully mature area at 17% in 1990 and 13.6% in 1996. Fully mature cover recovered to 17% of the watershed area in 1990, but decreased to 13.6% in 1996. Clear-cuts have reduced in area to about 3.9%. By 1996, approximately 72% of the watershed was covered by conifers exceeding 15 m in height.

Exceedance probability plots of daily streamflow simulated by the DHSVM for each of the nine vegetation scenarios were created at each of the 13 assessment locations. Cumulative density functions (CDF) of daily streamflow were estimated at each of the hydrological assessment locations individually using daily streamflow simulated by the DHSVM using a Gaussian-kernel–based nonparametric technique (Silverman 1986; Sheather and Jones 1991). Streamflows corresponding to specific frequency of exceedances were then "looked-up" from these CDF data. The flow exceedance probability plots are shown in Figures 3.24 through 3.36.

Changes in the hydrological response are more pronounced in the low flow region (50% or greater exceedance) than in the high flow region at all assessment locations. The reason for this effect is that evaporation, the main effect of vegetation on runoff, is relatively low during high flow. Consequently, the effect of change in vegetation is also small during high flow periods, as shown by the exceedance probability plots (see Figures 3.24 through 3.36).

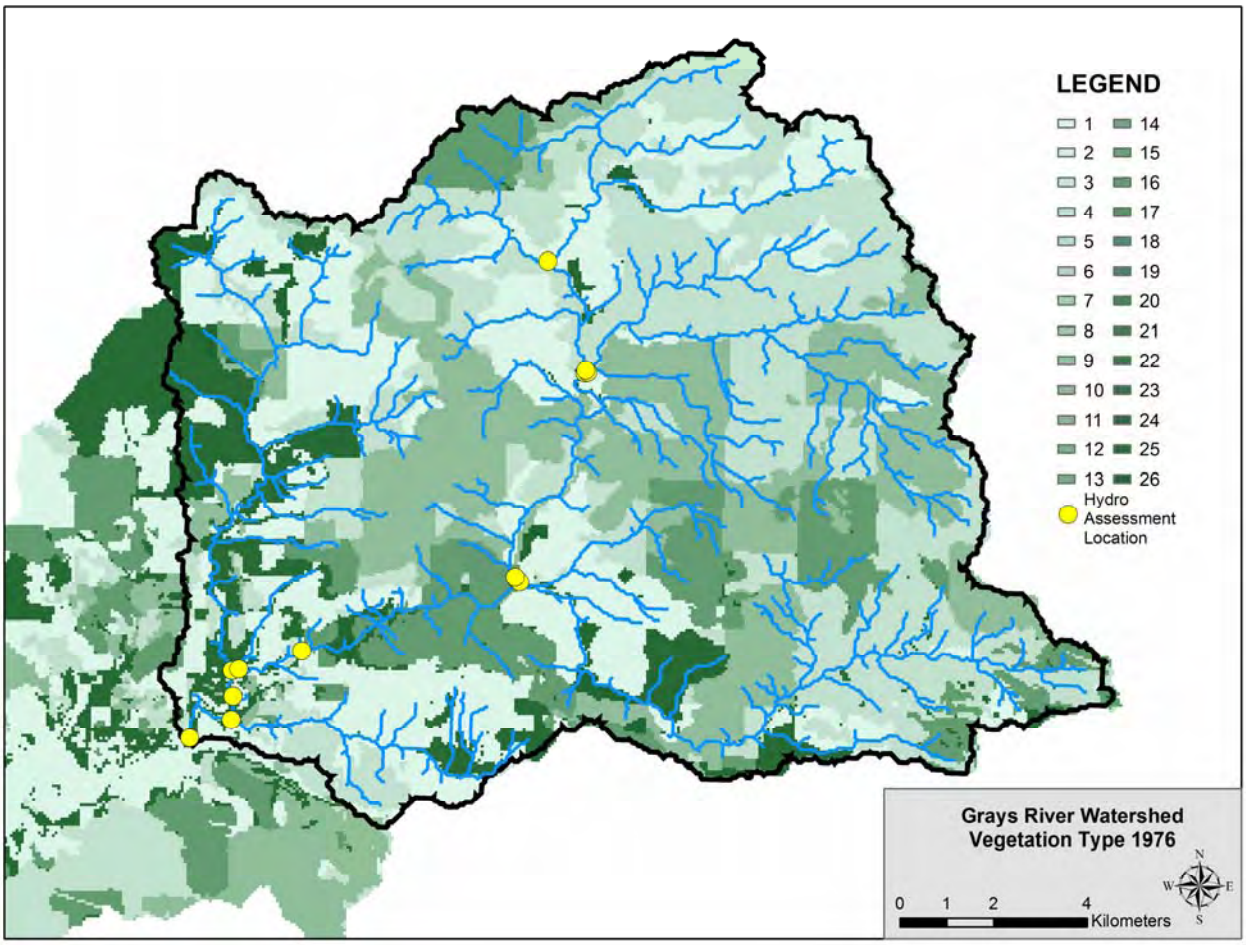


Figure 3.19. Vegetation Map Corresponding to the 1976 Forest Harvest Scenario

Tables 3.6 through 3.10 show the relative difference in daily streamflow corresponding to different exceedance probabilities for each of the nine vegetation scenarios at all 13 assessment locations. These metrics are computed by assuming the streamflow corresponding to the pre-harvest 100% fully mature vegetation cover as the baseline and expressing differences from this baseline in percentage.

The greatest change in streamflow response in the watershed was observed for the 1964 and 1976 vegetation scenarios, both of which were results of extensive harvesting throughout the watershed. Changes in 75% and 90% exceedance flows are considered minor. Except for Crazy Johnson (hydrologic assessment location number 13), the change in streamflow response at all assessment points was affected the greatest for the 1976 vegetation scenario. Streamflow responses at the assessment points are affected by changes in their respective contributing areas. Harvest in the immediate contributing area of the assessment point will affect the response the greatest. The contributing area of Crazy Johnson was extensively harvested in 1964 and probably recovered only slightly by 1976. The greatest changes in Crazy Johnson low flows appear to happen in response to the 1964 harvesting.

Only small areas were harvested within the North Fork (hydrological assessment location number 12) between 1942 and 1953. The 10% exceedance flow for the North Fork increased 28% to 69% greater than baseline between the 1942 and 1953 vegetation scenarios. Extensive harvesting occurred in the

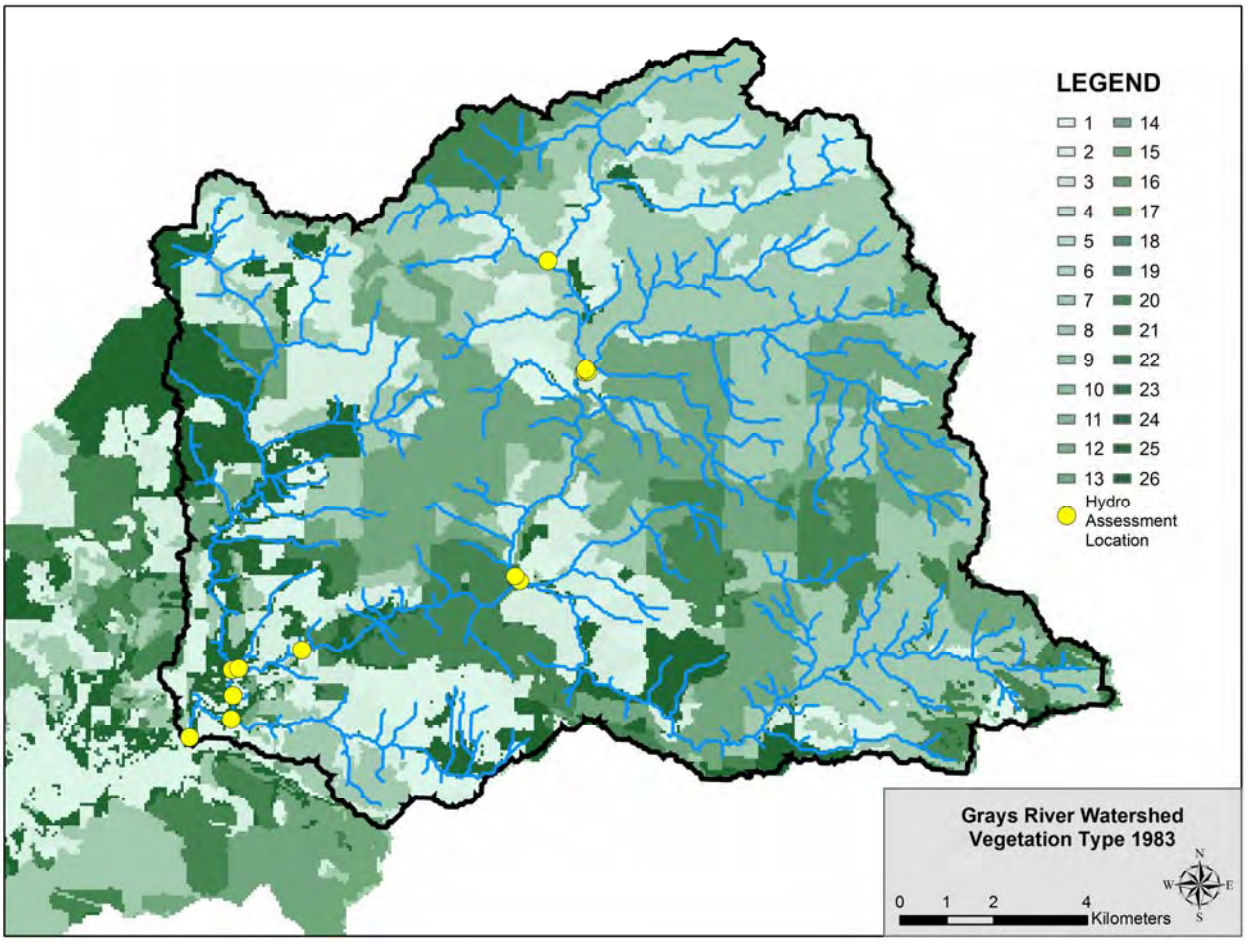


Figure 3.20. Vegetation Map Corresponding to the 1983 Forest Harvest Scenario

contributing area of North Fork in 1964 and 1976. The 1976 harvesting was the most severe, and the 10% exceedance flow increased from 69% greater than baseline for the 1953 vegetation scenario to 168% greater than baseline for the 1964 vegetation scenario. The corresponding increase from baseline for the 1976 vegetation scenario was 229%.

There was minor harvesting in the East Fork (hydrological assessment location number 10) contributing area until 1953, and the increase from baseline in 10% exceedance flow was only 27% for the 1942 vegetation scenario and 38% for the 1953 vegetation scenario. However, in response to the extensive harvesting in 1964 within the East Fork contributing area, the 10% exceedance flow increased 168% greater than the baseline. Continued harvesting between 1964 and 1976 in the East Fork contributing area increased the 10% exceedance flow corresponding to the 1976 vegetation scenario to 229% greater than the baseline.

Bankfull flow is defined as a discharge that fills a stream to the top of its banks (Leopold et al. 1964). A more commonly accepted definition of bankfull discharge is that discharge at which channel maintenance is most effective—that is, the geomorphological process at this discharge results in the average morphological characteristics of channels (Dunne and Leopold 1978). Leopold (1994) states that

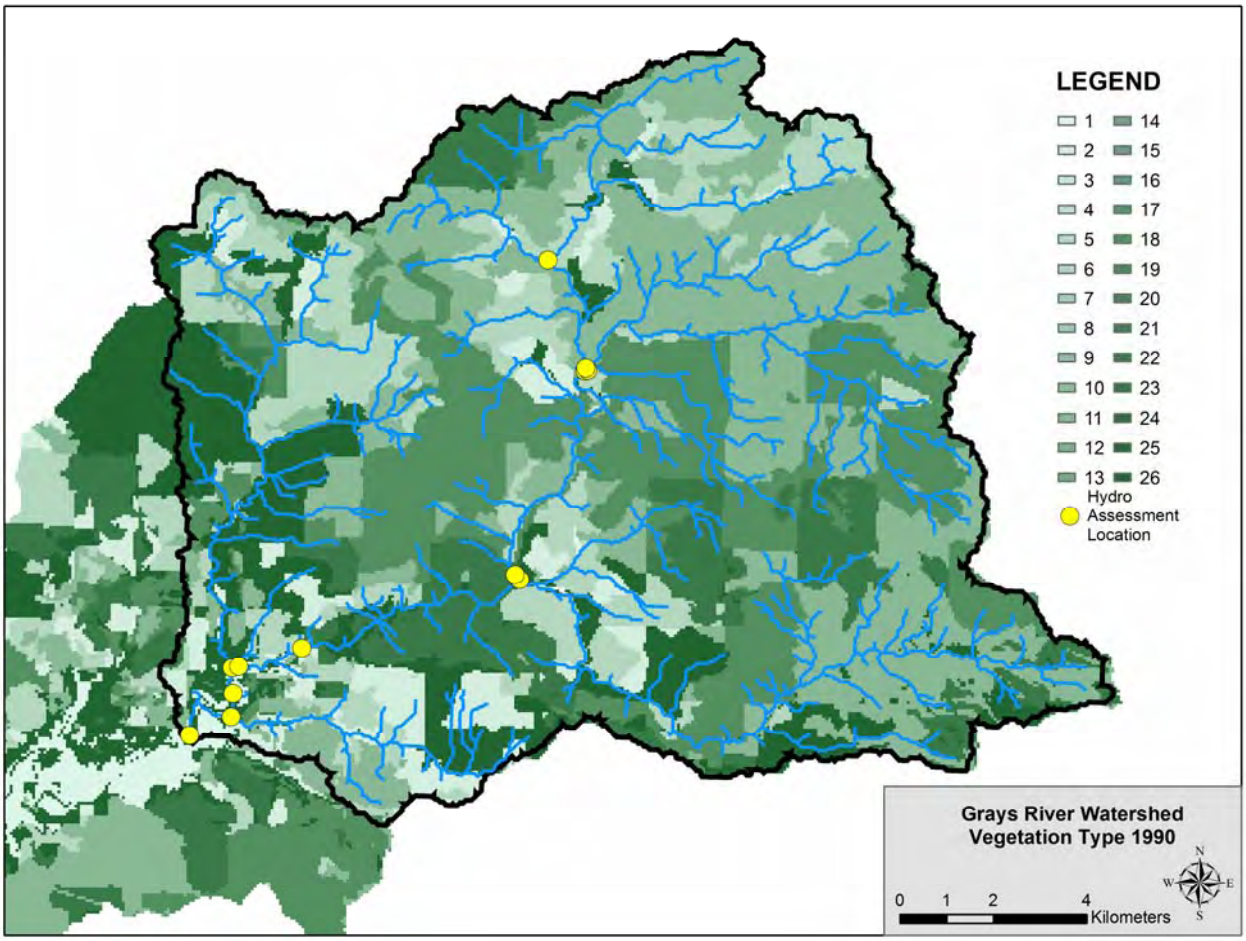


Figure 3.21. Vegetation Map Corresponding to the 1990 Forest Harvest Scenario

return periods of bankfull flow range between 1 and 2.5 years, with 1.5 years being the average. In the Pacific Northwest (the states of Washington, Oregon, and Idaho), Castro and Jackson (2001) estimated that the average return period of bankfull flow is 1.4 years.

Bankfull flows were estimated with DHSVM-simulated streamflow using the recommended (Castro and Jackson 2001) 1.4-year return period at the hydrological assessment locations defined for this study. To assess changes in bankfull flow under various vegetation scenarios, return period of a discharge equal to the bankfull flow corresponding to the baseline vegetation scenario (100% vegetation cover) was estimated from the frequency distribution of annual maximum discharge for each hydrological assessment location under the remaining vegetation scenarios. The return period of a discharge equal to bankfull flow corresponding to the baseline vegetation scenario at all hydrological assessment locations for each vegetation scenario is shown in Table 3.11.

Change in the return period of bankfull flow ranged from a 0.11-year reduction corresponding to the 1976 vegetation scenario at East Fork (location 10) to a 0.03-year increase corresponding to the 1942 forest harvest scenario for Fossil Creek (location 2). At all hydrological assessment points except Crazy

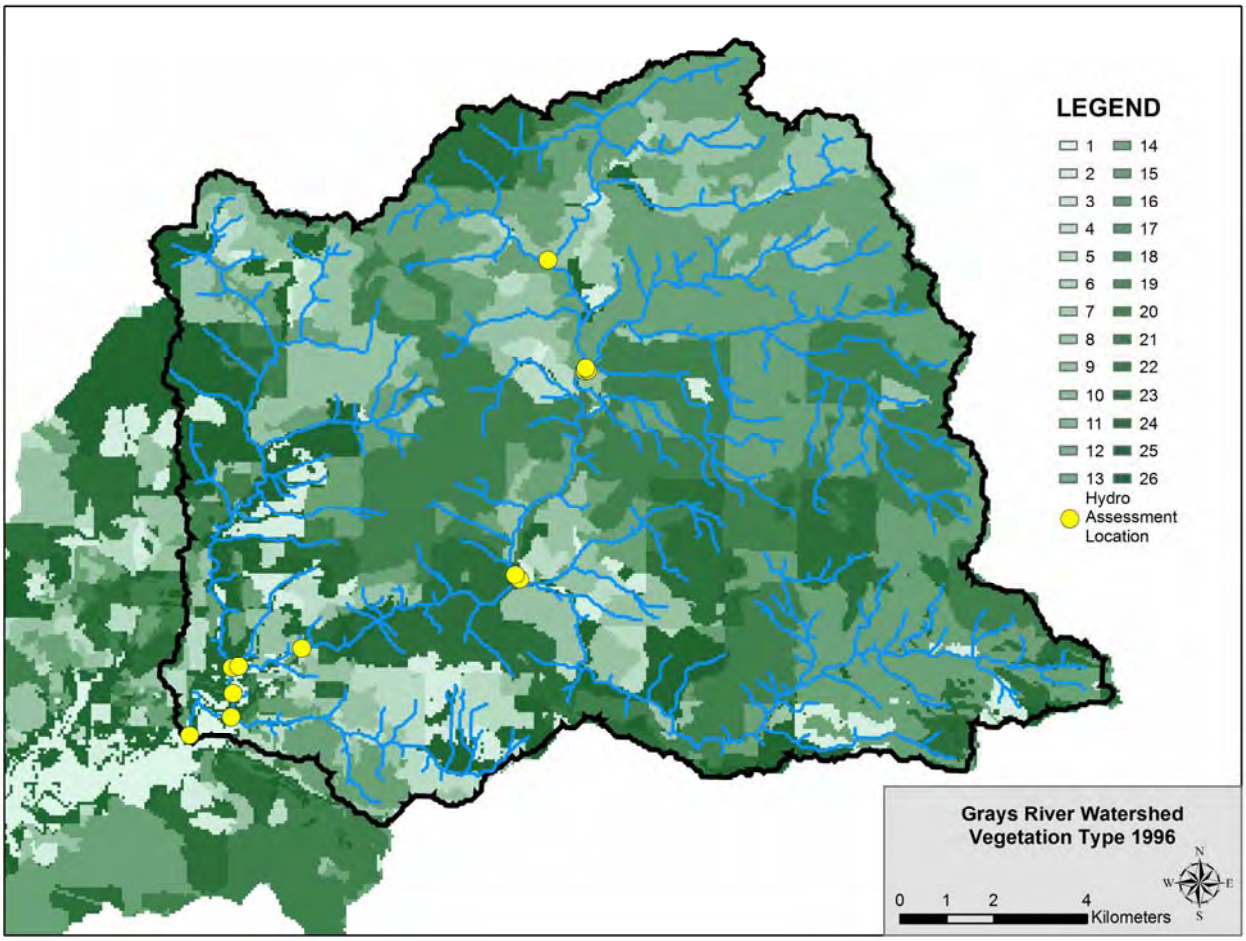


Figure 3.22. Vegetation Map Corresponding to the 1996 Forest Harvest Scenario

Table 3.3. Vegetation Height and Leaf Area Index for Vegetation Types in the Upper Grays River Watershed

Vegetation Class	Height (m)	LAI (-)	Vegetation Class	Height (m)	LAI (-)
1	0.00	0.00	14	14.67	8.89
2	2.93	1.99	15	15.28	9.17
3	4.65	3.12	16	15.82	9.43
4	5.44	3.63	17	16.07	9.54
5	7.61	4.98	18	16.54	9.75
6	8.26	5.38	19	16.76	9.84
7	8.88	5.75	20	17.35	10.09
8	10.04	6.42	21	17.70	10.23
9	11.56	7.27	22	18.43	10.52
10	12.02	7.52	23	18.55	10.56
11	13.27	8.18	24	19.09	10.75
12	13.65	8.38	25	19.66	10.93
13	14.01	8.56	26	20.00	11.25

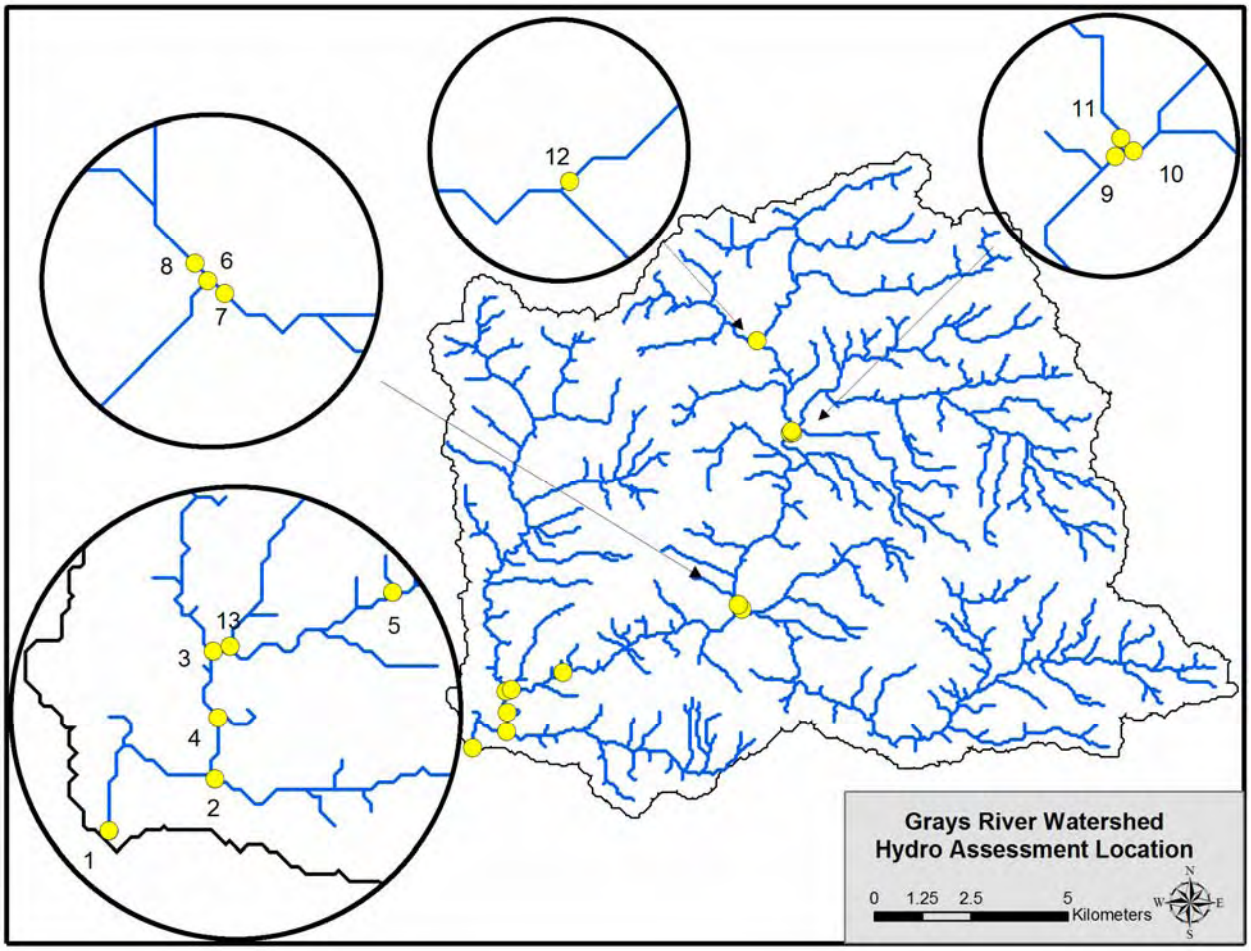


Figure 3.23. Hydrological Assessment Locations on the Stream Network

Table 3.4. Hydrological Assessment Location Names on the Stream Network

Location Number	Name
1	Grays Mainstem at SR 4 Bridge
2	Outflow from Fossil Creek
3	Outflow from West Fork
4	Grays Mainstem below West Fork and Gorley Reach
5	Grays Mainstem at Canyon Mouth and above Gorley Reach
6	Grays Mainstem above Canyon and below South Fork Confluence
7	South Fork
8	Grays Mainstem above South Fork Confluence
9	Grays Mainstem below East Fork Confluence
10	East Fork
11	Grays Mainstem above East Fork Confluence
12	North Fork
13	Crazy Johnson

Table 3.5. Percentage Cover of Vegetation Types for Individual Forest Harvests

Vegetation Type	FH1942	FH1953	FH1964	FH1976	FH1983	FH1990	FH1996
1	0.40	5.57	23.24	28.01	14.17	3.94	3.95
2	0.00	0.00	0.00	0.00	13.91	6.09	4.23
3	0.00	11.97	19.22	0.00	0.00	0.00	0.00
4	0.00	0.00	0.00	28.15	0.00	0.00	0.00
5	0.00	0.00	0.00	0.00	0.00	0.00	6.10
6	0.00	0.00	0.00	0.00	0.00	13.91	0.00
7	0.00	0.03	0.00	0.00	0.00	0.00	0.00
8	0.00	0.00	0.00	0.00	28.06	0.00	0.00
9	0.00	0.00	11.97	0.00	0.00	0.00	13.92
10	0.00	0.00	0.00	19.23	0.00	0.00	0.00
11	0.00	0.00	0.00	0.00	0.00	27.99	0.00
12	10.65	0.00	0.00	0.00	0.00	0.00	0.00
13	0.00	0.00	0.03	0.00	0.00	0.00	0.00
14	0.00	0.00	0.00	0.00	19.21	0.00	0.00
15	0.00	0.00	0.00	0.00	0.00	0.00	27.97
16	0.00	0.00	0.00	12.01	0.00	0.00	0.00
17	0.00	10.76	0.00	0.00	0.00	0.00	0.00
18	0.00	0.00	0.00	0.00	0.00	19.20	0.00
19	0.00	0.01	0.00	0.00	0.00	0.00	0.00
20	0.00	0.00	0.00	0.00	12.00	0.00	0.00
21	0.00	0.00	0.00	0.00	0.00	0.00	19.09
22	0.00	0.00	0.00	0.00	0.00	11.60	0.00
23	0.00	0.00	43.71	0.00	0.00	0.00	0.00
24	0.00	0.00	0.00	0.00	0.00	0.00	11.19
25	0.00	0.00	0.00	10.77	0.00	0.00	0.00
26	88.95	71.66	1.82	1.84	12.64	17.26	13.56

Johnson, the largest change from baseline was that corresponding to the 1976 forest harvest scenario. Return period of bankfull flows in Crazy Johnson were affected greatest by the Gap vegetation scenario, closely followed by the 1964 forest harvest scenario.

To estimate the frequency of occurrence of bankfull flows, a partial-duration series was constructed using the peaks-over-threshold method. All flood peaks are identified that exceed the bankfull discharge for all hydrological assessment locations for all vegetation scenarios. Table 3.12 lists the mean number of consecutive days, or duration, per year at or above bankfull flows for all hydrological assessment locations for all vegetation scenarios. Table 3.13 shows the percentage change in mean number of consecutive days, or duration, per year at or above bankfull flows with the 100% vegetation scenario as the baseline.

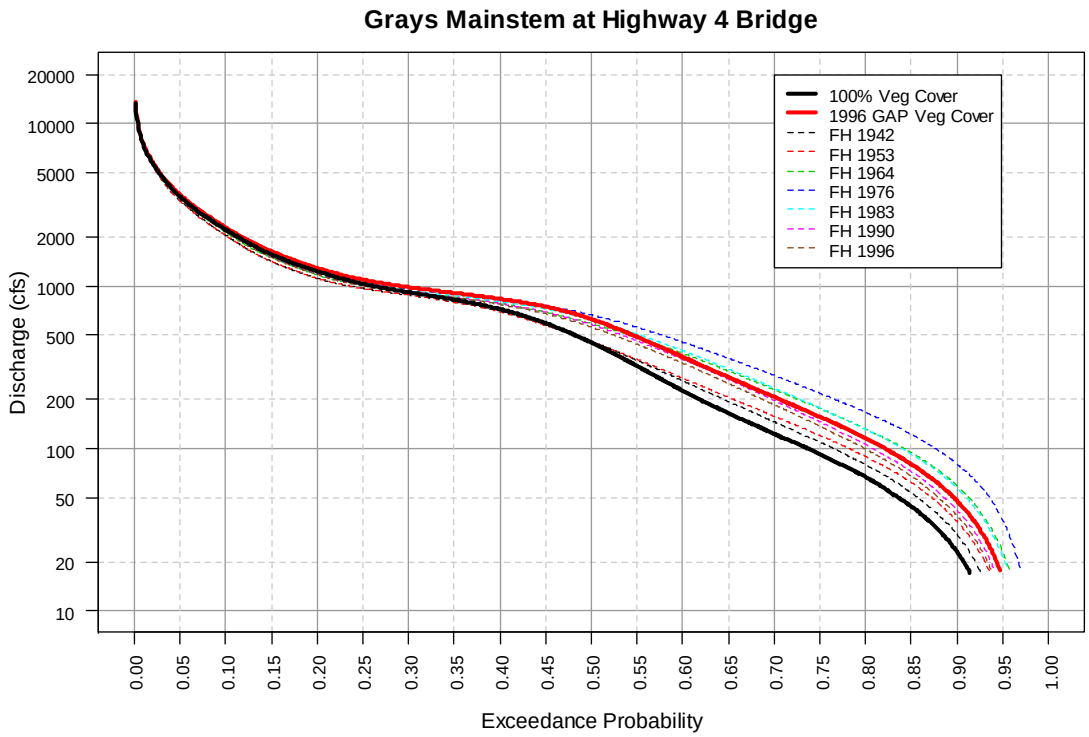


Figure 3.24. Exceedance Probability Plot for Daily Streamflow for Grays Mainstem at State Route 4 Bridge

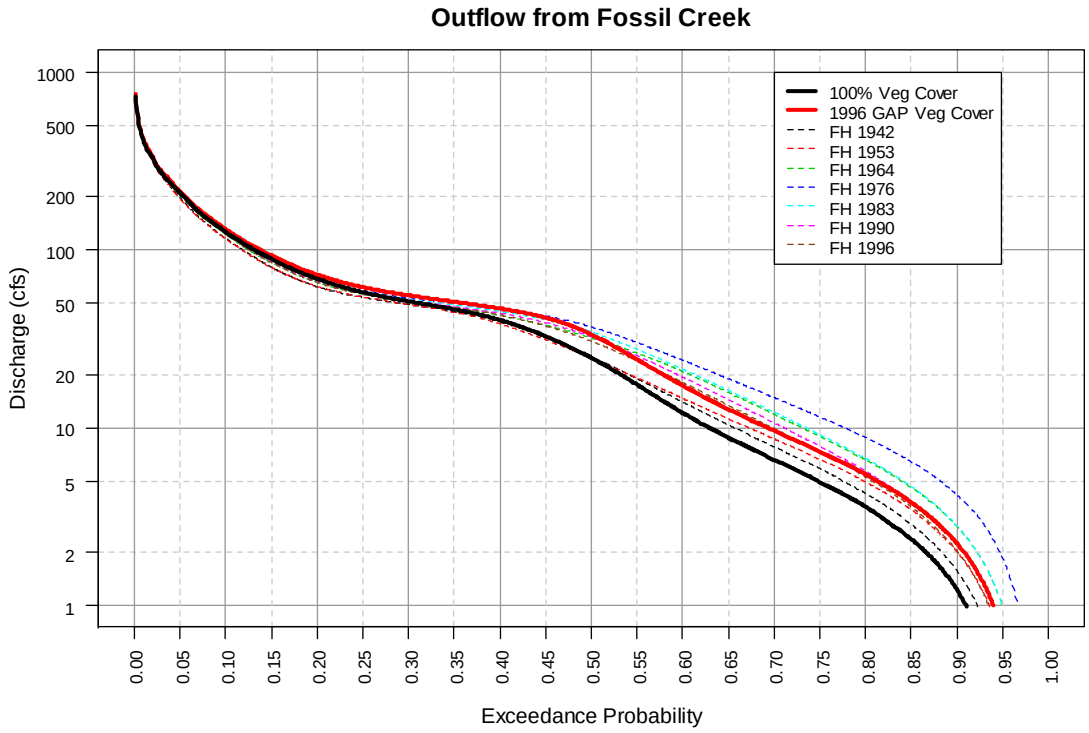


Figure 3.25. Exceedance Probability Plot for Daily Streamflow for Fossil Creek

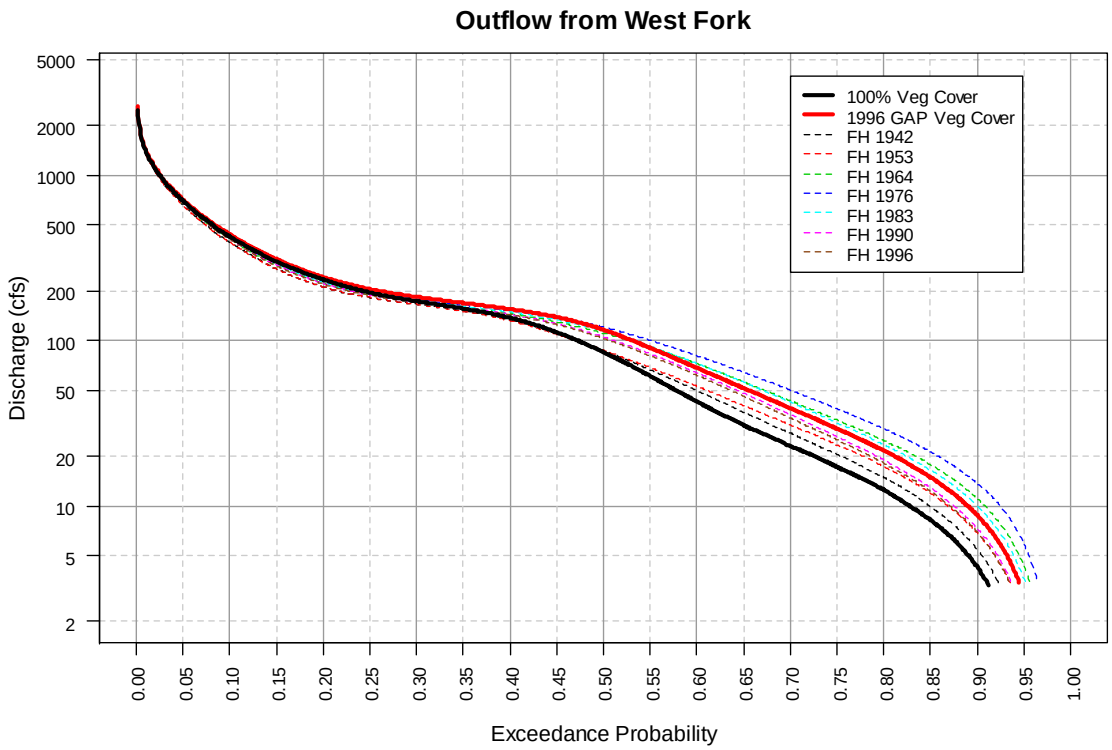


Figure 3.26. Exceedance Probability Plot for Daily Streamflow for West Fork

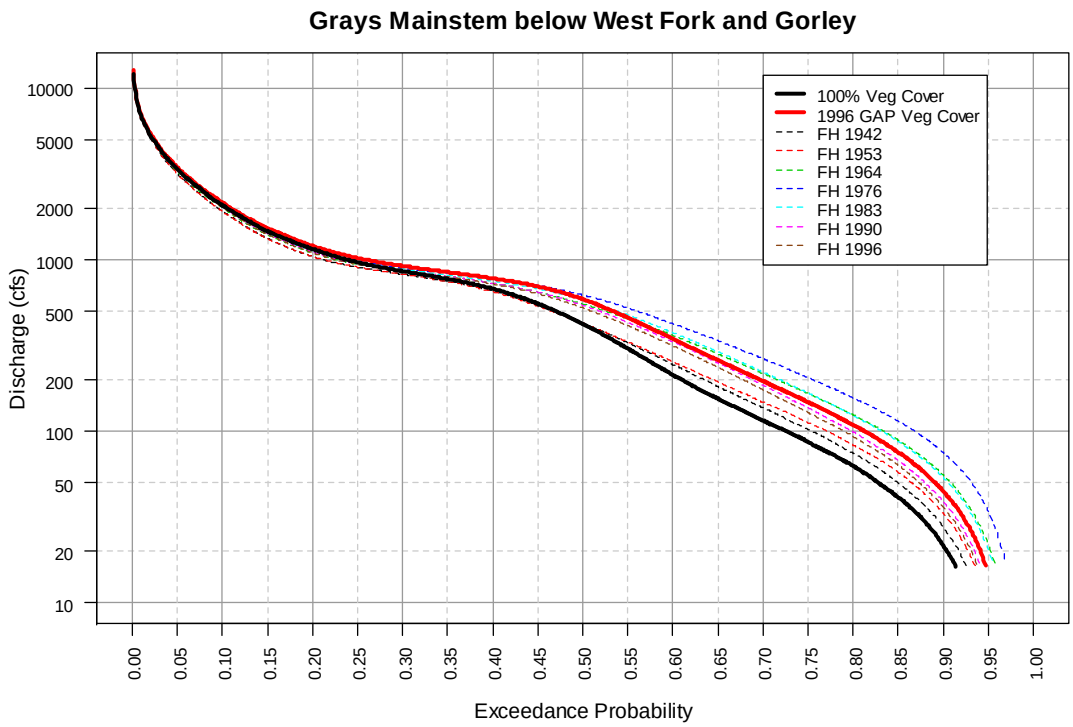


Figure 3.27. Exceedance Probability Plot for Daily Streamflow for Grays Mainstem Below West Fork and Gorley Reach

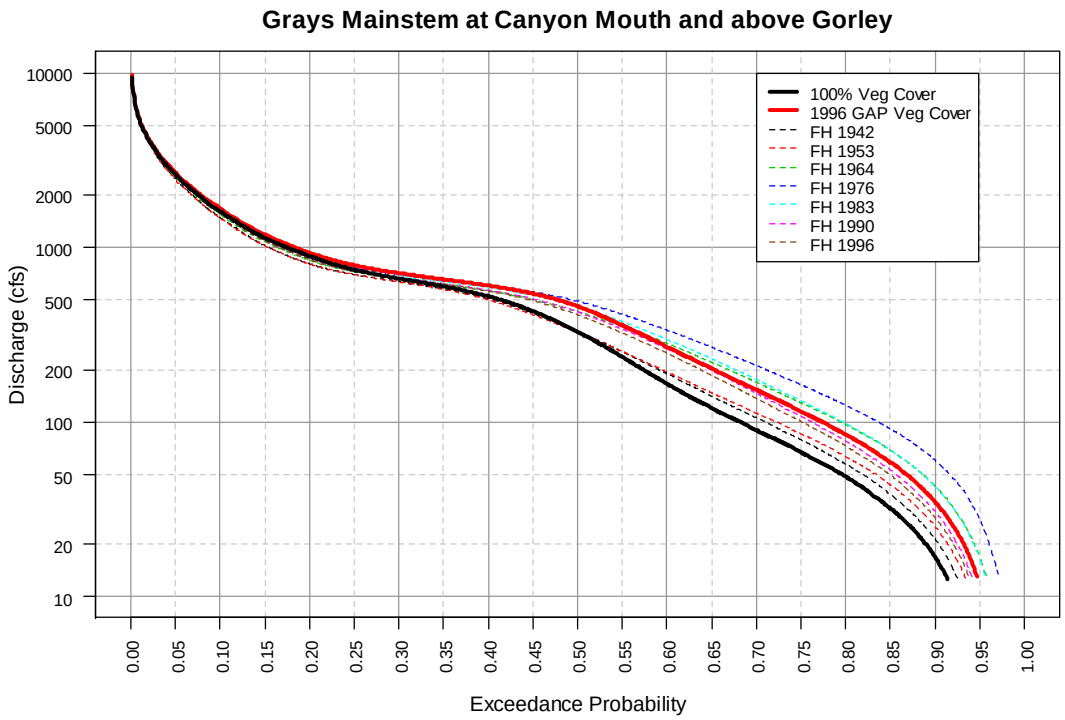


Figure 3.28. Exceedance Probability Plot for Daily Streamflow for Grays Mainstem at Canyon Mouth and Above Gorley Reach

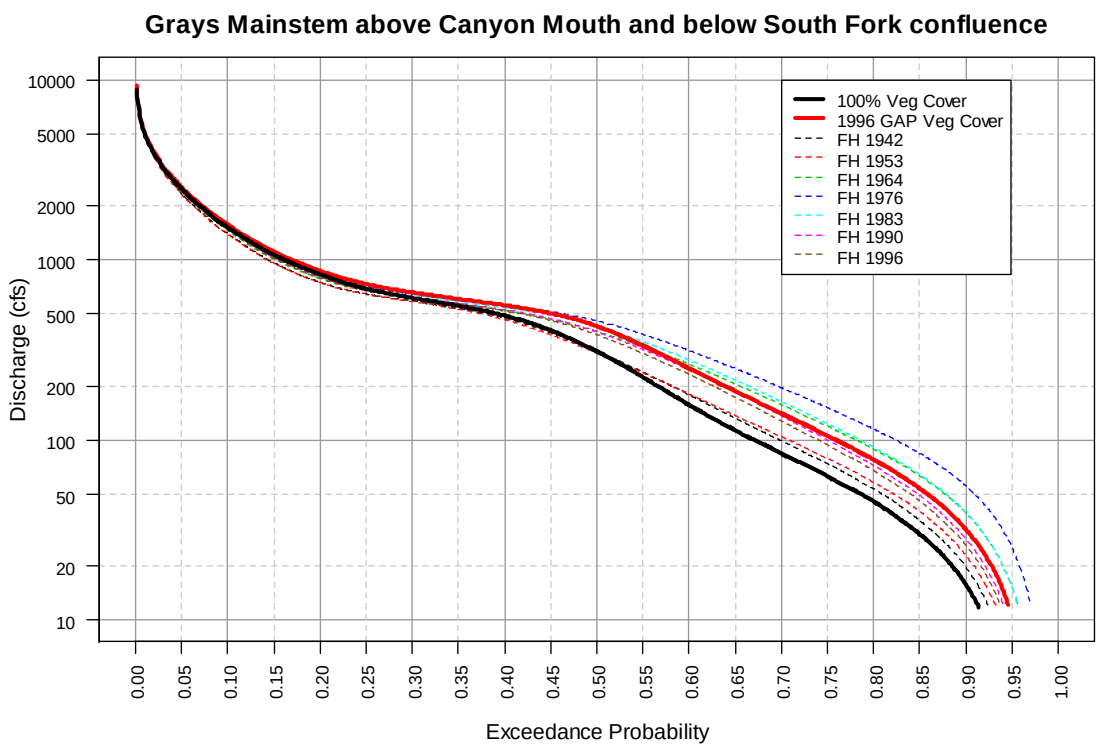


Figure 3.29. Exceedance Probability Plot for Daily Streamflow for Grays Mainstem Above Canyon Mouth and Below South Fork Confluence

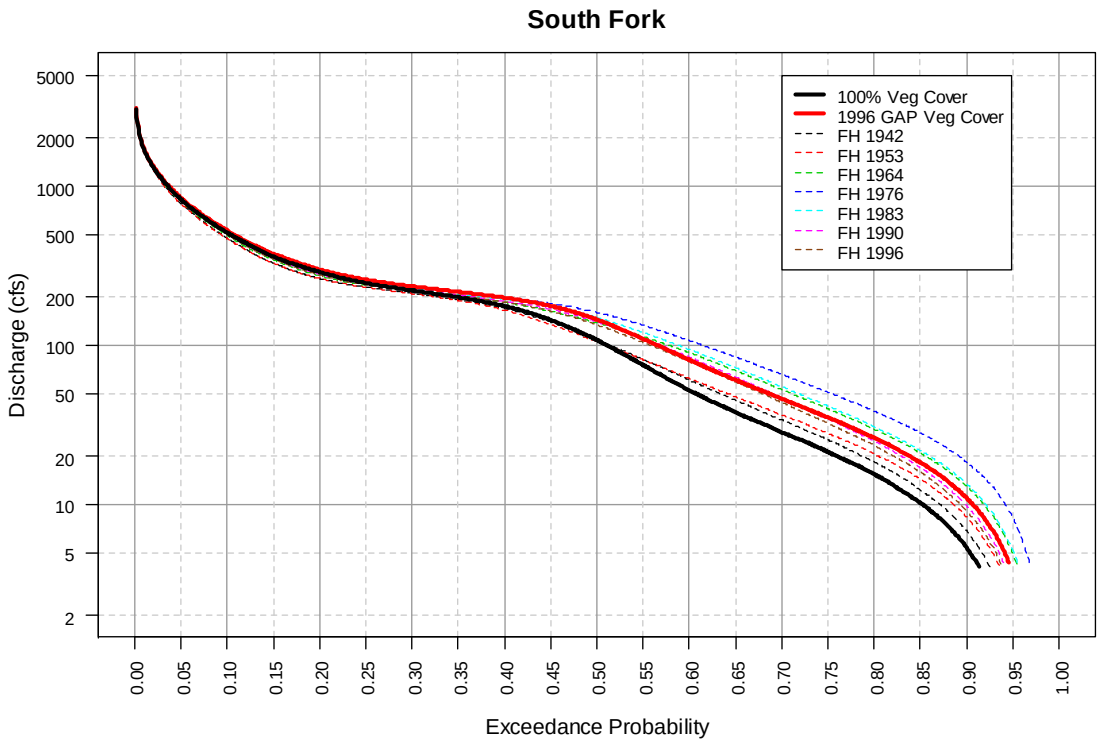


Figure 3.30. Exceedance Probability Plot for Daily Streamflow for South Fork

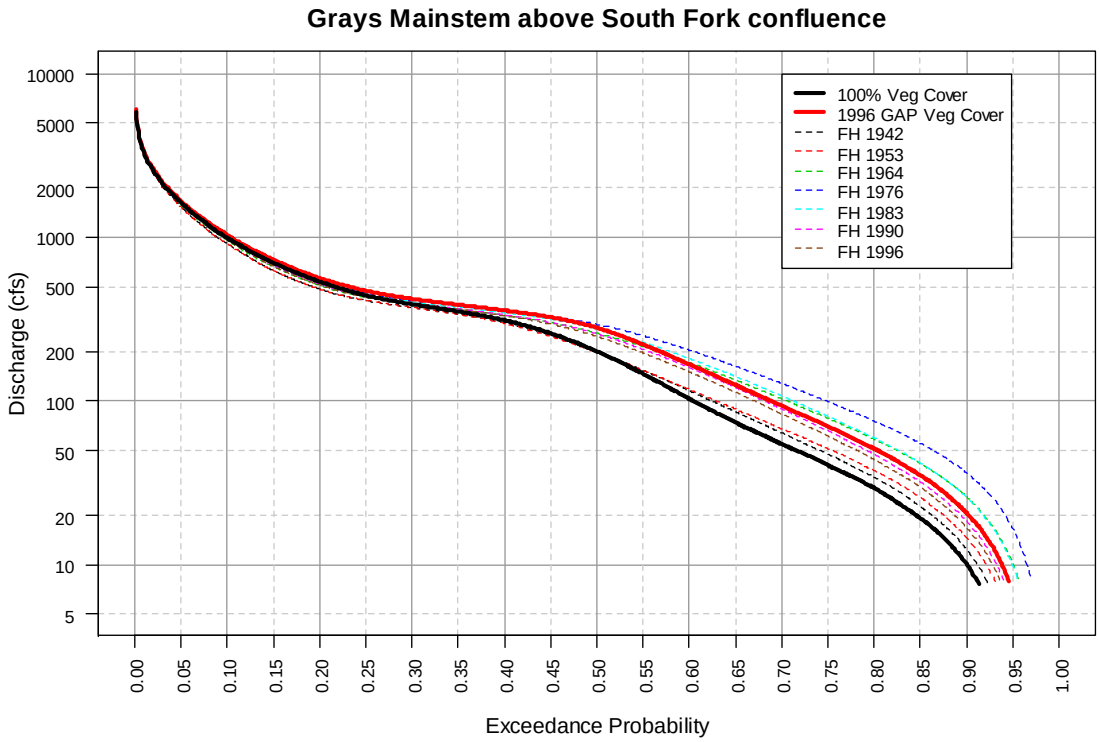


Figure 3.31. Exceedance Probability Plot for Daily Streamflow for Grays Mainstem Above South Fork Confluence

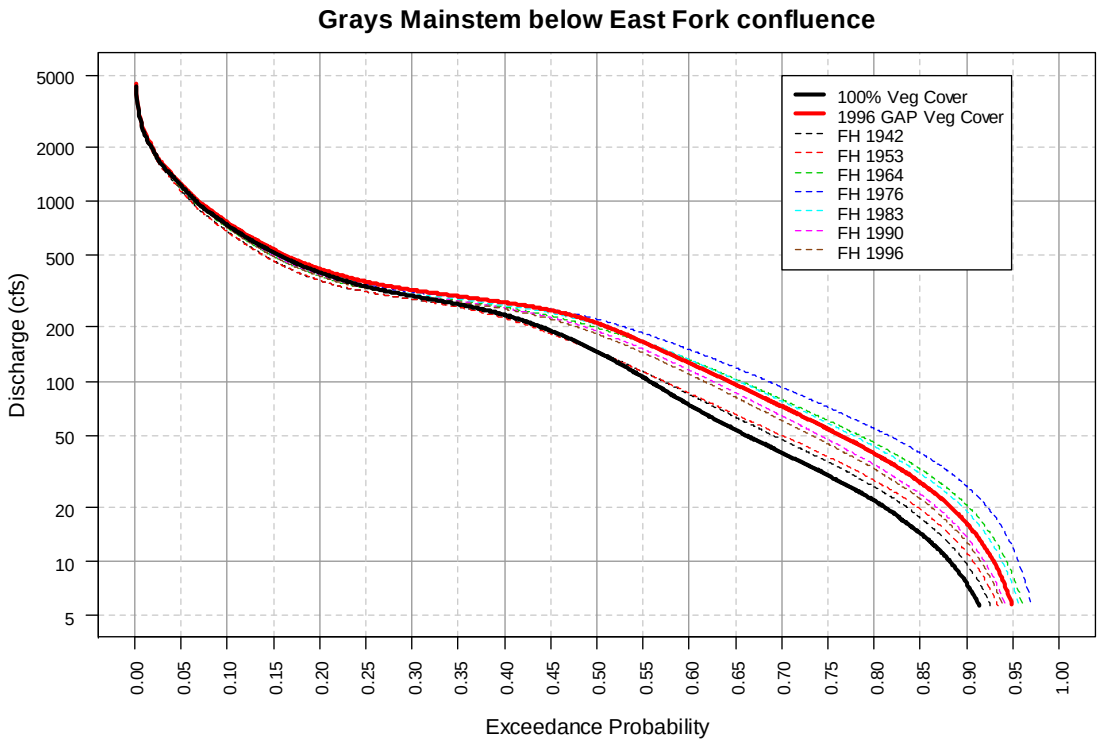


Figure 3.32. Exceedance Probability Plot for Daily Streamflow for Grays Mainstem Below East Fork Confluence

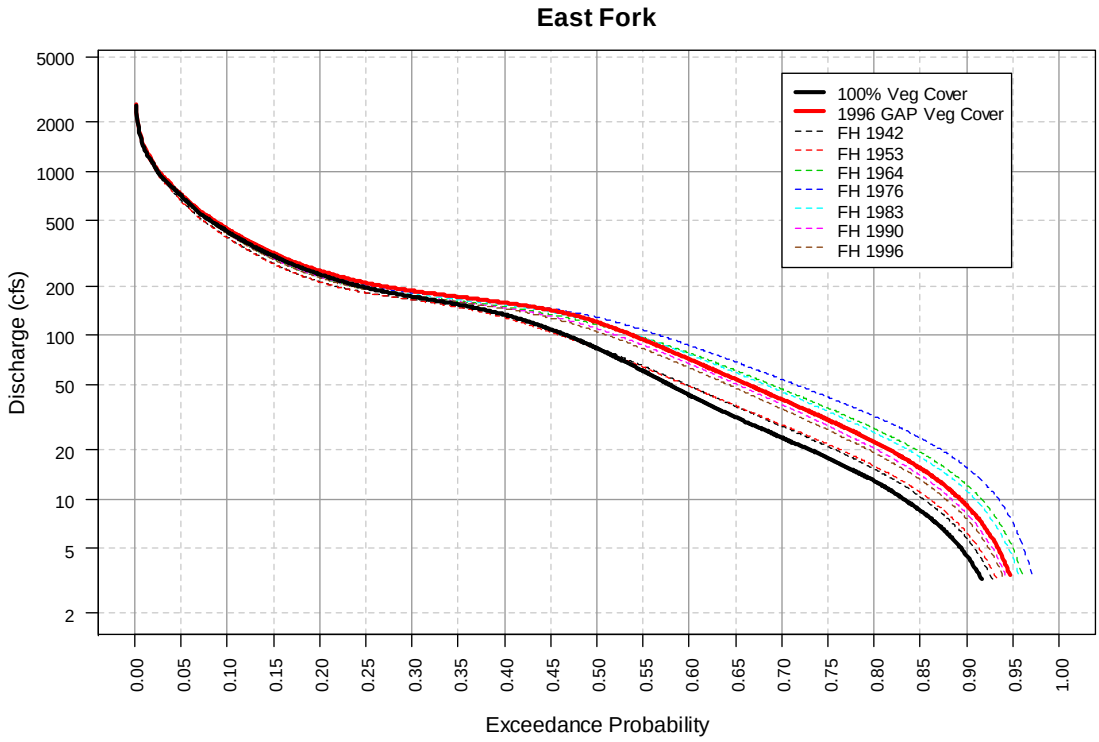


Figure 3.33. Exceedance Probability Plot for Daily Streamflow for East Fork

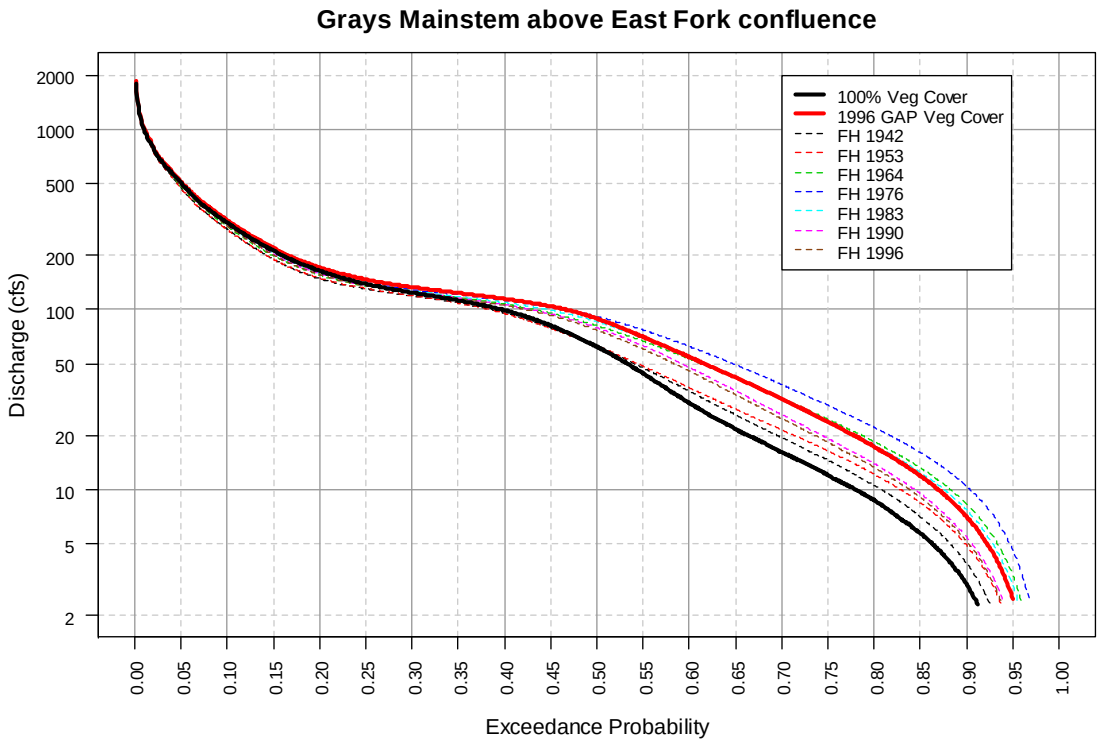


Figure 3.34. Exceedance Probability Plot for Daily Streamflow for Grays Mainstem Above East Fork Confluence

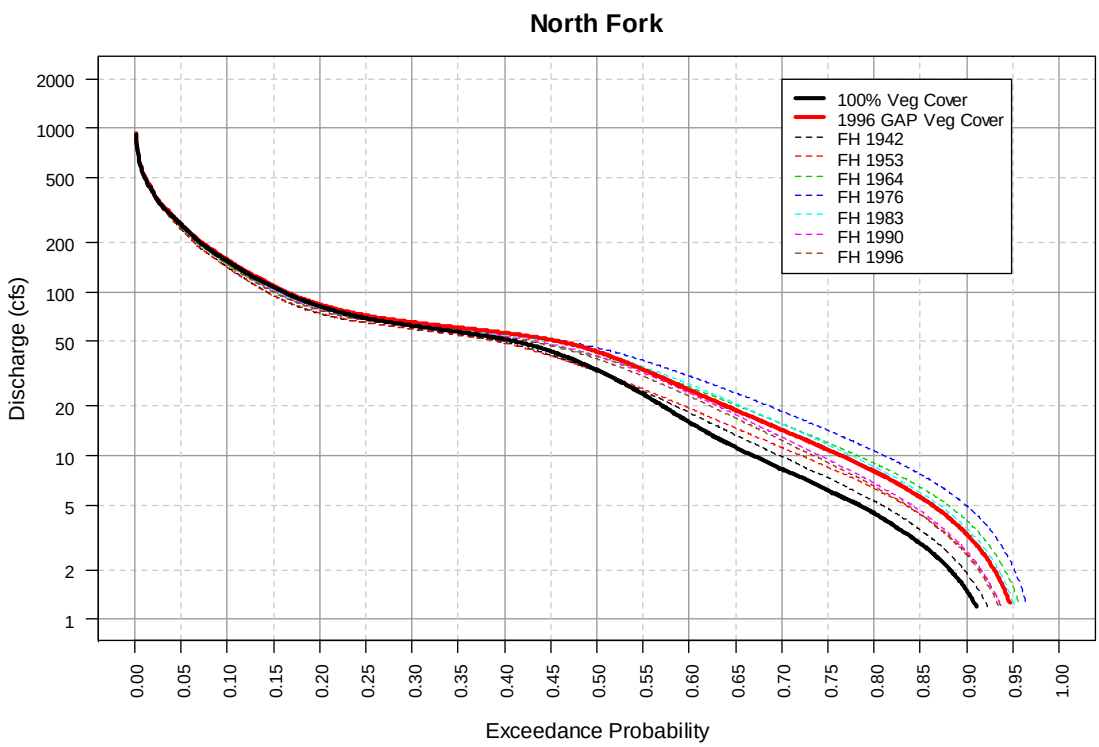


Figure 3.35. Exceedance Probability Plot for Daily Streamflow for North Fork

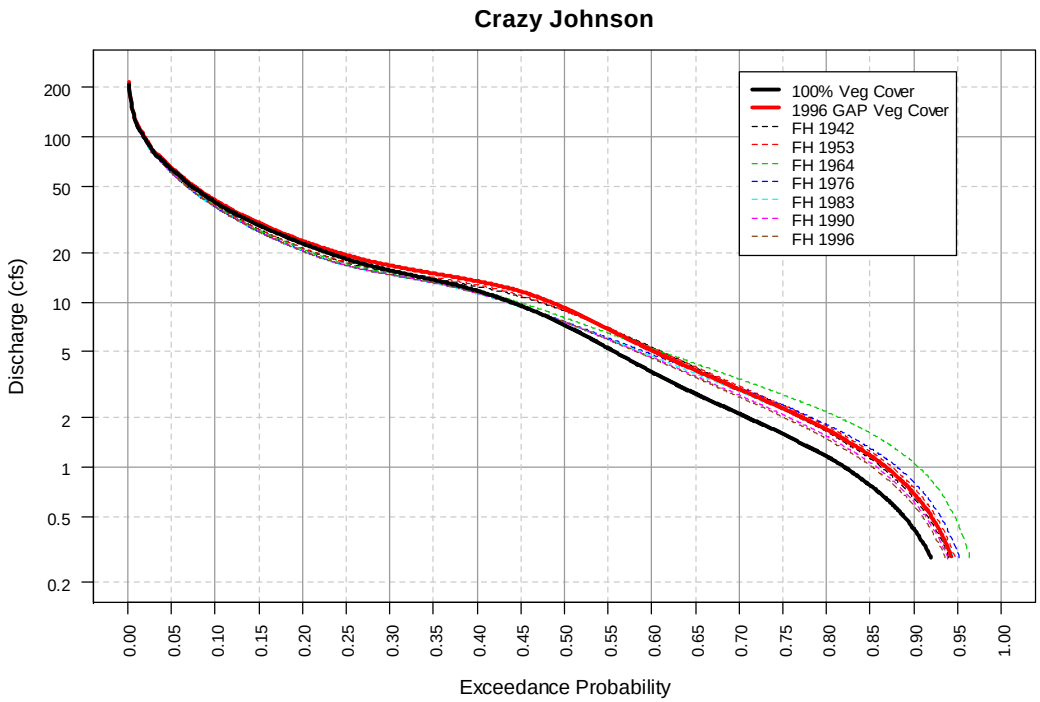


Figure 3.36. Exceedance Probability Plot for Daily Streamflow for Crazy Johnson

Table 3.6. Relative Change in Percentage in the 90% Exceedance Streamflow

Vegetation Scenario	Hydrological Assessment Location Numbers												
	1	2	3	4	5	6	7	8	9	10	11	12	13
100%	0	0	0	0	0	0	0	0	0	0	0	0	0
GAP	106	82	105	106	107	102	105	106	115	102	138	121	64
FH1942	28	28	28	27	26	25	27	23	29	27	31	28	54
FH1953	54	63	61	53	49	45	55	45	48	38	65	69	78
FH1964	156	127	160	157	156	150	143	158	173	171	180	168	151
FH1976	247	242	221	247	258	252	244	261	248	245	251	229	92
FH1983	147	128	134	148	154	152	152	156	152	148	159	140	67
FH1990	81	81	70	80	82	79	84	83	81	80	84	74	46
FH1996	67	66	63	67	68	66	70	68	70	67	74	65	39

Mean duration of flows at or above bankfull in the Grays River basin ranged from 1.38 days to 1.79 days for the baseline scenario. There was only a slight change in mean duration of bankfull flows corresponding to the forest harvest scenarios. This behavior is consistent with only slight changes in magnitudes of high flow across the forest harvest scenarios, as shown in the exceedance probability plots of streamflow simulated daily.

The maximum number of consecutive days, or duration, above bankfull flow did not change from vegetation scenario to vegetation scenario at any hydrological assessment location. The maximum duration above bankfull in the 43-year hydrology simulations for the hydrological assessment locations was three for locations 2, 9, 10, 11, and 12, and four for locations 1, 3, 4, 5, 6, 7, 8, and 13.

Table 3.7. Relative Change in Percentage in the 75% Exceedance Streamflow

Vegetation Scenario	Hydrological Assessment Location Numbers												
	1	2	3	4	5	6	7	8	9	10	11	12	13
100%	0	0	0	0	0	0	0	0	0	0	0	0	0
GAP	70	49	71	71	71	68	65	71	81	71	98	77	42
FH1942	19	19	20	19	18	17	20	17	19	18	21	19	41
FH1953	30	34	36	30	27	25	31	26	27	21	35	38	49
FH1964	91	80	93	92	92	90	87	93	102	102	104	95	73
FH1976	137	132	124	137	143	140	138	144	139	135	144	132	49
FH1983	93	85	85	93	97	96	95	99	95	93	99	90	39
FH1990	59	59	53	58	60	59	62	62	58	57	60	54	30
FH1996	49	48	46	49	50	49	52	51	50	48	52	47	26

Table 3.8. Relative Change in Percentage in the 50% Exceedance Streamflow

Vegetation Scenario	Hydrological Assessment Location Numbers												
	1	2	3	4	5	6	7	8	9	10	11	12	13
100%	0	0	0	0	0	0	0	0	0	0	0	0	0
GAP	39	36	36	39	40	38	35	39	44	45	43	30	27
FH1942	2	1	2	1	1	0	1	0	1	1	0	-2	22
FH1953	1	0	3	1	-1	-2	-2	-1	0	-1	1	-2	22
FH1964	31	30	30	30	30	29	28	29	36	39	31	20	11
FH1976	48	49	41	47	50	47	49	46	51	54	47	37	5
FH1983	38	40	33	38	40	38	38	38	41	43	38	29	4
FH1990	29	31	23	28	30	28	30	28	30	31	28	21	4
FH1996	24	24	21	24	25	23	25	23	25	26	24	17	4

Table 3.9. Relative Change in Percentage in the 25% Exceedance Streamflow

Vegetation Scenario	Hydrological Assessment Location Numbers												
	1	2	3	4	5	6	7	8	9	10	11	12	13
100%	0	0	0	0	0	0	0	0	0	0	0	0	0
GAP	6	7	5	6	6	6	5	7	6	7	6	4	6
FH1942	-6	-6	-7	-6	-6	-6	-6	-7	-6	-7	-6	-6	-5
FH1953	-6	-6	-6	-6	-6	-7	-6	-7	-6	-7	-5	-6	-4
FH1964	-2	-2	-2	-2	-2	-3	-3	-2	-2	-2	-2	-3	-7
FH1976	2	2	0	2	3	3	2	3	2	3	2	0	-9
FH1983	0	0	-2	0	0	0	0	1	0	0	-1	-1	-9
FH1990	-2	-1	-3	-2	-1	-1	-1	-1	-2	-2	-2	-2	-9
FH1996	-3	-2	-4	-2	-2	-2	-2	-2	-2	-2	-2	-3	-9

Table 3.10. Relative Change in Percentage in the 10% Exceedance Streamflow

Vegetation Scenario	Hydrological Assessment Location Numbers												
	1	2	3	4	5	6	7	8	9	10	11	12	13
100%	0	0	0	0	0	0	0	0	0	0	0	0	0
GAP	3	4	3	4	4	4	3	4	3	4	3	2	2
FH1942	-7	-8	-7	-8	-8	-7	-8	-7	-8	-8	-8	-8	-4
FH1953	-8	-8	-7	-8	-8	-8	-8	-8	-8	-9	-8	-8	-4
FH1964	-4	-3	-3	-3	-3	-3	-4	-3	-3	-2	-5	-5	-5
FH1976	1	1	0	1	2	2	1	2	1	1	0	0	-7
FH1983	-1	0	-1	-1	0	0	-1	0	-1	-1	-2	-2	-7
FH1990	-2	-1	-3	-2	-1	-1	-1	-1	-2	-2	-3	-3	-7
FH1996	-3	-2	-3	-3	-2	-2	-2	-2	-3	-3	-4	-4	-7

Table 3.11. Return Period (years) of Bankfull Streamflow. Bankfull flows were estimated as 1.4-year discharges from annual maximum daily discharge data corresponding to the 100% vegetation cover scenario.

Vegetation Scenario	Hydrological Assessment Location Numbers												
	1	2	3	4	5	6	7	8	9	10	11	12	13
100%	1.40	1.40	1.40	1.40	1.40	1.40	1.40	1.40	1.40	1.40	1.40	1.40	1.40
GAP	1.33	1.33	1.33	1.33	1.33	1.33	1.33	1.33	1.32	1.33	1.32	1.33	1.34
FH1942	1.42	1.43	1.42	1.42	1.42	1.42	1.42	1.42	1.42	1.42	1.42	1.42	1.38
FH1953	1.40	1.40	1.40	1.40	1.40	1.40	1.40	1.40	1.41	1.41	1.40	1.39	1.36
FH1964	1.34	1.35	1.34	1.34	1.34	1.34	1.34	1.34	1.33	1.33	1.34	1.34	1.35
FH1976	1.30	1.30	1.31	1.30	1.30	1.30	1.30	1.30	1.30	1.29	1.30	1.30	1.38
FH1983	1.32	1.33	1.33	1.32	1.32	1.32	1.32	1.32	1.32	1.32	1.32	1.33	1.40
FH1990	1.35	1.35	1.36	1.35	1.35	1.35	1.35	1.35	1.35	1.35	1.36	1.36	1.41
FH1996	1.36	1.37	1.37	1.36	1.36	1.36	1.36	1.36	1.36	1.36	1.36	1.37	1.41

Table 3.12. Mean Number of Consecutive Days at or Above Bankfull Streamflow

Vegetation Scenario	Hydrological Assessment Location Numbers												
	1	2	3	4	5	6	7	8	9	10	11	12	13
100%	1.39	1.58	1.47	1.38	1.38	1.41	1.39	1.48	1.64	1.64	1.65	1.60	1.79
GAP	1.41	1.59	1.49	1.39	1.39	1.40	1.39	1.50	1.66	1.66	1.62	1.63	1.76
FH1942	1.39	1.59	1.48	1.39	1.39	1.40	1.40	1.49	1.63	1.62	1.67	1.60	1.82
FH1953	1.40	1.57	1.49	1.38	1.38	1.40	1.40	1.49	1.63	1.62	1.66	1.61	1.81
FH1964	1.40	1.58	1.50	1.41	1.39	1.43	1.41	1.49	1.62	1.62	1.63	1.60	1.82
FH1976	1.40	1.59	1.49	1.42	1.41	1.43	1.39	1.50	1.61	1.64	1.62	1.60	1.85
FH1983	1.41	1.56	1.50	1.40	1.40	1.43	1.39	1.50	1.62	1.62	1.64	1.60	1.85
FH1990	1.41	1.58	1.48	1.39	1.39	1.44	1.40	1.48	1.64	1.64	1.66	1.62	1.85
FH1996	1.41	1.58	1.49	1.39	1.39	1.43	1.40	1.48	1.64	1.64	1.66	1.63	1.83

Table 3.13. Percentage Change in Mean Number of Consecutive Days at or Above Bankfull Streamflow

Vegetation Scenario	Hydrological Assessment Location Numbers												
	1	2	3	4	5	6	7	8	9	10	11	12	13
100%	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
GAP	1.0	1.0	1.6	0.9	0.9	-0.2	0.5	1.6	1.0	1.0	-2.0	2.0	-1.8
FH1942	0.0	1.0	1.1	0.4	0.4	-0.2	0.7	1.1	-0.4	-1.0	1.4	0.0	1.5
FH1953	0.5	-0.6	1.6	0.0	0.0	-0.2	1.2	1.1	-0.4	-1.0	0.3	0.6	1.3
FH1964	0.5	0.0	2.2	2.1	0.9	1.9	1.6	1.1	-1.0	-1.0	-1.0	0.4	1.5
FH1976	0.5	1.0	1.6	2.6	2.1	1.9	0.5	1.6	-1.6	0.0	-2.0	0.0	3.3
FH1983	1.0	-1.0	2.2	1.4	1.4	1.9	0.5	1.6	-1.0	-1.0	-0.4	0.0	3.3
FH1990	1.5	0.0	1.1	0.4	0.9	2.4	1.2	0.5	0.3	0.3	0.3	1.2	3.3
FH1996	1.0	0.0	1.6	0.4	0.4	1.9	0.7	-0.1	0.3	0.3	0.3	1.9	2.4

To analyze the relative change in low flows in the Grays River basin, a peak under threshold approach was adopted (Ashkar et al. 1998) to identify flow sequences in the daily-simulated streamflow series that fall below a threshold to construct a partial-duration series. The threshold for this analysis was set to the 90% exceedance flow (10th percentile daily mean discharge) under the baseline vegetation condition (100% vegetation cover) at each hydrological assessment location. Relative changes in mean duration of low flows for all hydrological assessment locations for the nine vegetation scenarios are shown in Table 3.14.

Table 3.14. Relative Change in Percentage of Mean Duration of Low Flows at Selected Locations

Vegetation Scenario	Hydrological Assessment Location Numbers												
	1	2	3	4	5	6	7	8	9	10	11	12	13
100%	0	0	0	0	0	0	0	0	0	0	0	0	0
GAP	-51	-53	-49	-50	-51	-50	-50	-49	-51	-51	-53	-55	-52
FH1942	19	19	16	19	20	19	22	18	20	17	25	27	-2
FH1953	6	-7	7	9	8	4	1	5	6	2	5	-6	-14
FH1964	-29	-17	-25	-29	-28	-29	-26	-23	-14	-19	-16	-32	-67
FH1976	-56	-51	-24	-39	-60	-55	-60	-36	-37	-51	-35	-40	-28
FH1983	-19	-12	-4	-21	-27	-22	-24	-15	-19	-22	-19	-16	-7
FH1990	9	10	8	11	12	9	15	8	9	5	17	17	1
FH1996	9	12	10	9	10	11	9	12	11	4	22	27	1

Mean duration of low flows shows changes from a reduction of 67% from baseline at Crazy Johnson corresponding to the 1964 forest harvest scenario to an increase of 27% from baseline at North Fork corresponding to the 1996 forest harvest scenario. Greatest absolute change from baseline among all forest harvest scenarios corresponded to the 1976 forest harvest except at Crazy Johnson, which showed the greatest absolute change corresponding to the 1964 forest harvest. These changes also are consistent with the exceedance probability plots presented previously. Table 3.15 shows the relative change in maximum duration of low flows (maximum number of consecutive days below the 90% exceedance flow) for each hydrological assessment location under the nine vegetation scenarios.

Table 3.15. Relative Change in Percentage of Maximum Duration of Low Flows at Selected Locations

Vegetation Scenario	Hydrological Assessment Location Numbers												
	1	2	3	4	5	6	7	8	9	10	11	12	13
100%	0	0	0	0	0	0	0	0	0	0	0	0	0
GAP	-58	-60	-58	-58	-58	-58	-58	-58	-58	-58	-58	-68	-52
FH1942	-9	-9	-9	-9	-9	-9	-9	-9	-8	-8	-8	-8	-11
FH1953	-49	-53	-50	-48	-48	-47	-49	-46	-46	-44	-50	-53	-26
FH1964	-73	-63	-73	-73	-72	-71	-70	-72	-73	-73	-73	-73	-86
FH1976	-89	-90	-82	-88	-90	-89	-90	-87	-87	-89	-82	-79	-65
FH1983	-67	-62	-63	-67	-69	-68	-69	-68	-69	-69	-71	-66	-19
FH1990	-16	-16	-14	-15	-16	-16	-15	-17	-16	-48	-14	-14	-11
FH1996	-13	-13	-13	-13	-13	-13	-13	-13	-13	-45	-13	-12	-9

Changes in maximum duration of low flows ranged from a reduction of 90% from baseline at Fossil Creek to a reduction of 8% at hydrological assessment locations 9 through 12 corresponding to the 1942 forest harvest scenario. The greatest absolute change in maximum duration of low flows at all hydrological assessment locations was that corresponding to the 1976 forest harvest scenario except at Crazy Johnson, where the greatest absolute change was that corresponding to the 1964 forest harvest scenario.

To analyze the behavior of summer low flows, a partial-duration series was constructed from the simulated flows for the months of July through September using the peaks-under-threshold approach with the threshold set to 90% exceedance daily flow. Relative change in mean duration of summer low flows with the 100% vegetation cover scenario as the baseline is shown in Table 3.16. Table 3.17 shows the relative change in maximum duration of summer low flows with the 100% vegetation cover as the baseline for all hydrological assessment locations in the Grays River basin.

Table 3.16. Relative Change in Percentage of Mean Duration of Summer Low Flows at Selected Locations

Vegetation Scenario	Hydrological Assessment Location Numbers												
	1	2	3	4	5	6	7	8	9	10	11	12	13
100%	0	0	0	0	0	0	0	0	0	0	0	0	0
GAP	-59	-63	-59	-58	-59	-58	-59	-57	-59	-59	-62	-64	-58
FH1942	0	-1	-7	-2	1	-2	3	-4	-6	-6	-2	-1	-16
FH1953	-19	-34	-20	-19	-20	-22	-26	-18	-20	-23	-24	-37	-26
FH1964	-48	-43	-49	-49	-51	-51	-49	-45	-40	-45	-46	-56	-71
FH1976	-76	-89	-49	-71	-81	-75	-86	-68	-72	-81	-60	-61	-50
FH1983	-48	-41	-42	-49	-52	-49	-51	-44	-48	-52	-46	-46	-23
FH1990	-17	-18	-18	-16	-14	-16	-12	-16	-19	-22	-12	-12	-15
FH1996	-16	-13	-16	-16	-15	-13	-16	-12	-16	-21	-6	-3	-14

Table 3.17. Relative Change in Percentage of Maximum Duration of Summer Low Flows at Selected Locations

Vegetation Scenario	Hydrological Assessment Location Numbers												
	1	2	3	4	5	6	7	8	9	10	11	12	13
100%	0	0	0	0	0	0	0	0	0	0	0	0	0
GAP	-35	-38	-35	-35	-35	-35	-35	-35	-35	-35	-35	-50	-27
FH1942	0	0	0	0	0	0	0	0	0	0	0	0	-8
FH1953	-28	-37	-28	-28	-28	-25	-28	-24	-24	-18	-30	-36	-8
FH1964	-58	-46	-63	-58	-57	-54	-53	-57	-58	-58	-59	-59	-85
FH1976	-91	-97	-72	-88	-93	-91	-93	-89	-90	-95	-72	-67	-54
FH1983	-57	-54	-52	-57	-58	-58	-59	-57	-58	-60	-54	-50	-8
FH1990	-8	-8	-8	-8	-8	-8	-8	-8	-8	-22	-8	-8	-8
FH1996	-8	-8	-8	-8	-8	-8	-8	-8	-8	-22	-8	-8	-8

Relative change in mean and maximum duration of summer low flows was small for the 1942 forest harvest scenario except at Crazy Johnson and, to some extent, at hydrological assessment locations 3, 9, and 10. The greatest absolute changes occurred corresponding to the 1976 forest harvest scenario, except at Crazy Johnson, where it was corresponding to the 1964 forest harvest scenario.

A similar analysis was conducted on streamflow during the spawning period, taken as the months of November and December, to estimate durations of flow below the 90% exceedance daily flow. We examined the 90% exceedance flow during this period because we expected more significant hydrologic effects of timber harvest on low flows than on high flows. Flow levels affect the wetted area available, the distribution of spawning fish and the availability of velocities and depths suitable for spawning salmonids. Low flows can limit fish access to spawning sites and limit available spawning area, concentrate spawning fish, increasing the potential for redd superimposition. Effects of flow on spawning habitat availability is examined in detail in Chapter 5. Relative changes in mean and maximum durations of spawning period low flows are shown in Tables 3.18 and 3.19.

Table 3.18. Relative Change in Percentage of Mean Duration of Spawning Period Low Flows at Selected Locations

Vegetation Scenario	Hydrological Assessment Location Numbers												
	1	2	3	4	5	6	7	8	9	10	11	12	13
100%	0	0	0	0	0	0	0	0	0	0	0	0	0
GAP	28	14	11	12	12	15	10	16	7	23	13	-13	-5
FH1942	18	24	19	14	14	18	16	28	26	22	25	15	-6
FH1953	11	16	6	10	16	6	11	-2	-2	0	35	13	18
FH1964	22	61	53	18	18	44	51	23	23	15	1	9	-43
FH1976	-78	-100	-23	-79	-79	-78	-100	-78	-78	-100	-77	-46	36
FH1983	67	50	64	61	50	55	51	57	57	61	24	63	-12
FH1990	14	5	-7	7	7	11	10	12	12	5	18	-4	-15
FH1996	-3	9	-14	-8	-8	-6	-10	1	3	6	10	4	-5

Table 3.19. Relative Change in Percentage of Maximum Duration of Spawning Period Low Flows at Selected Locations

Vegetation Scenario	Hydrological Assessment Location Numbers												
	1	2	3	4	5	6	7	8	9	10	11	12	13
100%	0	0	0	0	0	0	0	0	0	0	0	0	0
GAP	-29	-47	-29	-29	-29	-29	-47	-24	-29	-29	-39	-53	-18
FH1942	-29	-24	-29	-29	-29	-29	-29	-29	-29	-29	-33	-29	-41
FH1953	-47	-47	-47	-47	-47	-41	-47	-41	-41	-41	-50	-47	-47
FH1964	-59	-53	-59	-59	-59	-59	-59	-59	-59	-59	-61	-59	-65
FH1976	-94	-100	-71	-94	-94	-94	-100	-94	-94	-100	-94	-76	-53
FH1983	-53	-47	-53	-53	-53	-59	-59	-59	-59	-59	-61	-53	-47
FH1990	-35	-35	-41	-41	-41	-41	-35	-41	-41	-41	-39	-35	-41
FH1996	-35	-29	-41	-41	-41	-41	-35	-41	-41	-41	-39	-35	-35

Change in mean duration of spawning period low flows varied from a reduction of 100% (no flows below the 90% exceedance mean daily flow corresponding to the 100% vegetation scenario) at hydrological assessment locations on Fossil Creek, South Fork, and East Fork corresponding to the 1976 forest harvest scenario, to an increase of 67% at the SR 4 bridge corresponding to the 1983 forest harvest scenario. The greatest absolute change for all hydrological assessment locations was corresponding to the 1976 forest-harvest scenario except for the West Fork (location 3) and North Fork (location 12), where it was corresponding to the 1983 forest harvest scenario, and Crazy Johnson, where it was corresponding to the 1964 forest harvest scenario.

3.4 Discussion

The DHSVM was used in fully distributed mode to simulate the streamflow responses of the upper Grays River watershed under a number of vegetation scenarios that correspond to historical forest harvesting within the watershed. A probabilistic analysis of daily streamflow at several locations within the watershed clearly showed the range of impacts that may result from heterogeneous changes in vegetation pattern within the watershed. The results of this modeling clearly show that the extent and intensity of forest harvest and its relative hydrological connectivity with respect to the streamflow location being analyzed is very important in any hydrological scenario analysis.

Because it is a fully distributed model, the DHSVM can account for the hydrological connectivity explicitly and allows the assessment to clearly articulate the level of impact at different points on the stream network in response to spatially heterogeneous change. DHSVM simulations described above do not represent historical streamflow regime during 1963–2004 water years because all scenario runs used a vegetation pattern that was held fixed throughout the simulation period. These vegetation patterns correspond to one of the nine vegetation patterns described above and thus individually represent the range of hydrologic response expected over almost the complete period of record (42 water years) of meteorological observations.

Explicit analysis of change over time cannot be addressed by the DHSVM simulation carried out for this study. However, useful statements regarding the hydrologic regime can be made based on the change in several hydrological measures (flows corresponding to several exceedance probabilities, bankfull

flows, low flows, summer low flows, and spawning period low flows). Although a direct comparison with historical streamflow within the Grays River basin is not possible due to the reasons stated above, the simulations allow us to derive useful information regarding hydrologic response in the Grays River watershed.

Different parts of the Grays River watershed respond differently to forest harvest and associated change in the spatial distribution of vegetation. These differences occur due to differences in response related to whether the spatial location of the harvest directly affects the hydrology in the contributing area of an assessment location.

The results of hydrologic modeling indicate that the greatest impact of vegetation scenarios is reflected in mid to low flows. Impacts on larger flood flows are relatively minor. This behavior is expected because vegetation affects the streamflow primarily through evapotranspiration. During floods, conditions in the basin are expected to be wet and humid, limiting evapotranspiration. Low flows generally occur during dry conditions, when evapotranspiration is greater. Also, the relative magnitude of evapotranspiration during floods is small and during low flows is larger. Therefore, floods are affected less by changes in vegetation cover compared to low flows.

As an example, this result is illustrated for the 1999 flooding-event that caused the Gorley avulsion. Although this was a catastrophic event, the flood flows were not significantly higher than typical flooding events that occur relatively frequently on the Grays River. However, the effect of flooding at a local scale can be quite severe, especially if human infrastructure (i.e., levees, dikes) designed to protect property fail, as was the case during the 1999 avulsion when a floodplain dike was breached.

Figure 3.37 shows the streamflow hydrographs corresponding to the nine DHSVM scenario simulations during the 1999 avulsion event. The peak discharge occurred on December 15, 1999. The peak discharge corresponding to the 100% vegetation cover scenario was 10,831 cfs, 82% greater than the bankfull flow at this location. The DHSVM simulated discharge on December 15, 1999, for all vegetation scenarios is shown in Table 3.20. To compare the severity of the discharge during the 1999 avulsion event, return periods of annual floods equal to the discharges listed in Table 3.21 were estimated from the frequency distribution of annual maximum discharge used previously to estimate the bankfull flows.

The DHSVM accurately reproduced the timing of the December 15, 1999, avulsion event. Estimated streamflow was 82% greater than 100% forest cover bankfull discharge. This was a significant but not exceptional flood event, with an estimated return period of 8–14 years relative to 100% forested conditions. These results support the Washington State Department of Fish and Wildlife analysis of Cramer (2000) that dike failure resulted from a combination of high flows, channel bed aggradation, inadequate dike design, and limited maintenance.

In general, the magnitude, frequency, and duration of flooding events and peak flows have been affected relatively little by forest harvesting. However, due to reduced vegetation cover and correspondingly greater exposed soil surfaces, sediment generation areas are potentially more extensive within the Grays River watershed. This implies that sediment delivery may be greater to the channels, thereby increasing the sediment concentration during floods. In addition, localized effects of runoff from harvested areas and forest roads can have a significant hydrologic influence at the stream channel scale.

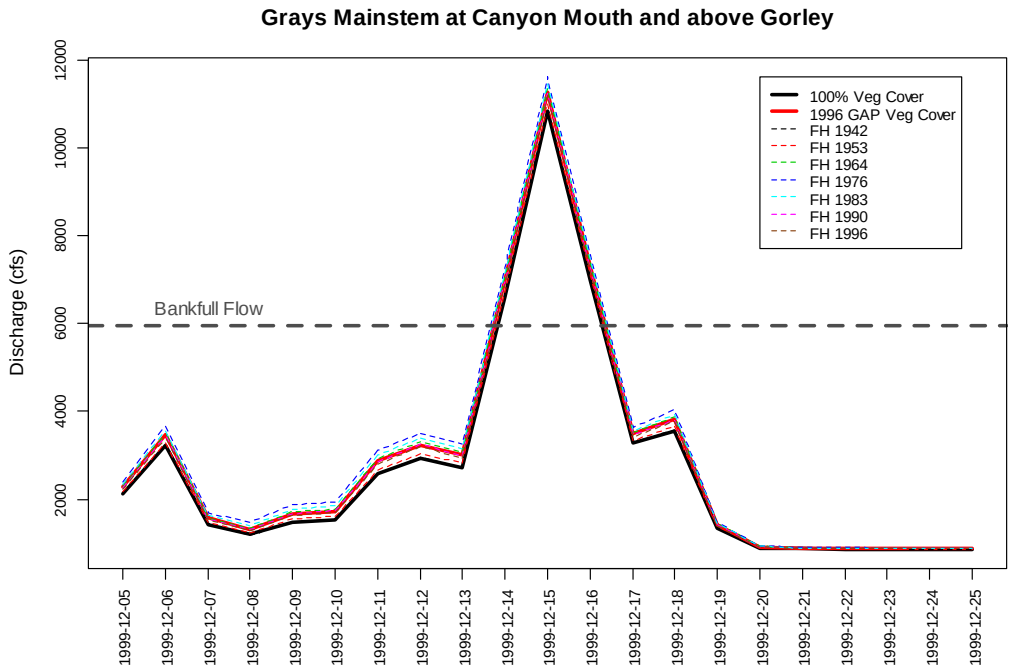


Figure 3.37. Streamflow Hydrograph of Grays River Mainstem at Canyon Mouth During 1999 Avulsion Event

Table 3.20. Distributed Hydrology Soil Vegetation Model Simulated Discharge During the 1999 Avulsion Event

Vegetation Scenario	DHSVM Simulated Discharge on December 15, 1999 (cfs)
100%	10,831
GAP	11,265
FH1942	10,865
FH1953	10,983
FH1964	11,355
FH1976	11,610
FH1983	11,442
FH1990	11,243
FH1996	11,171

This influence includes greater stream power to transport sediment from upland source areas through the stream network. Elevated streamflow in these headwater channels also could facilitate debris flows and streambank erosion on a localized scale.

There was only a slight change in mean duration of bankfull flows corresponding to the forest harvest scenarios. This behavior is consistent with only slight changes in magnitudes of high flow across the forest harvest scenarios. Low flows appear to be more affected by forest cover changes.

Table 3.21. Return Period of the 1999 Avulsion Event Discharge for all Vegetation Scenarios

Vegetation Scenario	Return Period (years)
100%	7.8
GAP	10.6
FH1942	8.0
FH1953	8.7
FH1964	11.3
FH1976	13.9
FH1983	12.1
FH1990	10.4
FH1996	9.9

Duration of low flows during the spawning period was shorter under the timber-harvest scenarios when compared to baseline (historical) conditions, ranging from a reduction of 100% from baseline at hydrological assessment locations on Fossil Creek, South Fork, and East Fork corresponding to the 1976 forest harvest scenario to a reduction of 18% at Crazy Johnson corresponding to the GAP vegetation scenario. The greatest absolute change at all hydrological assessment locations occurred corresponding to the 1976 forest harvest scenario except for Crazy Johnson, where it occurred corresponding to the 1964 forest harvest scenario.

The DHSVM simulation results indicate the maximum impacts to runoff production and streamflow would occur under the 1976 land-use scenario, which represents the peak intensity of harvest activities. Impacts to flood flows were minor, and the impact on low flows decreased with later land-use scenarios that reflect a reduction in the annual rate of harvest. These results are similar to the Thomas and Meghan (1998) analysis of timber harvesting in the H. J. Andrews Experimental Forest. The treatment in the study basin was road construction followed 4 years later by harvesting 25% of the basin. They showed treatment effects diminished with time and concluded that flow increases were detectable only for flow less than the 2-year recurrence interval.

The DHSVM analysis did not specifically evaluate the effects of roads on runoff production and streamflow. The road network has likely altered local peak flows by intercepting shallow subsurface flow and moving it more rapidly to the channel system. Roads can have dramatic effects on peak flows, but interactions between flows and roads are complex (Wigmosta and Perkins 2001) and localized effects seen in the upper watershed may not continue into the lower watershed. However, the greatest hydrologic impact of harvest related activities is likely through increased sediment yield, both by direct erosion from road surfaces and, more important, through a dramatic increase in mass wasting triggered by both road construction and timber harvest.

In naturally forested watersheds of the Pacific Northwest, the rainfall intensity rarely exceeds the infiltration capacity of the forest soils. In addition, evapotranspiration accounts for almost half of the precipitation that falls on the forest canopy. The result is that there is little, if any, surface runoff in naturally forested watersheds, with most recharge to surface waters occurring slowly via interflow or groundwater pathways. Hillslope processes in the Pacific Northwest tend to be dominated by these subsurface flow paths.

Timber-harvest activities, including clear-cut logging and road construction, have been shown to have the potential to alter the hydrologic regime of forested watersheds in the Pacific Northwest (Harr et al. 1975, 1982; Harr and McCorison 1979; Harr 1986; Naiman et al. 1992; LaMarche and Lettenmaier 1998; Thomas and Megahan 1998; Beschta et al. 2000; Bowling et al. 2000; Bowling and Lettenmaier 2001), as well as influence the ecology of the native forests (Franklin et al. 2002). Based on these hydrologic alterations, there is a general perception that the land-cover changes resulting from logging have resulted in an increased frequency and severity of flooding in watersheds affected by timber harvest land use. However, an evaluation of this body of research indicates that this perception is not always evident from hydrologic data or modeling scenarios (Bowling et al. 2000). There is a demonstrated, consistent increase in watershed water-yield in harvested basins, due largely to a decrease in evapotranspiration in clear-cut areas (Bowling et al. 2000).

Research does indicate that during so-called *rain-on-snow* events, there can be a significant increase in runoff from clear-cut areas as compared to areas of mature coniferous forest (Harr 1981, 1986; Harr et al. 1982). Although these events are relatively infrequent, they can occur in the southwestern Washington region. The removal of mature forest canopy can cause substantial increases in both snow accumulation before rain-on-snow events and snowmelt during those events (Storck et al. 1999). The greater accumulation of snow in clear-cut areas prior to rain-on-snow events is a result of less ablation of snow that otherwise would have been intercepted by the forest canopy and lost to the atmosphere prior to the rainfall-melt event. The increased snowmelt rate that occurs during rain-on-snow events is largely due to greater heat transfer in clear-cut areas resulting from more exposure to surface winds (Harr 1981, 1986; Storck et al. 1999).

Some studies have shown an increase in peak streamflow after timber-harvest activities (e.g., clear-cutting and road construction) in forested watersheds (Bowling et al. 2000). However, other studies have found no trend in peak flows (Harr et al. 1982; Harr 1986; Thomas and Megahan 1998). Still other studies indicate that increases in peak flow occur only for intermediate-sized storms in areas already saturated by previous precipitation events (Harr et al. 1975). Changes in peak streamflow resulting from clear-cutting and logging roads tend to be dependent on a variety of environmental factors. These factors include rainfall intensity, antecedent soil moisture, and pre-rainfall snow accumulation as well as the physiographic conditions of the watershed (Bowling et al. 2000). Detection of peak flow increases due to clear-cut logging are complicated also by the hydrogeomorphic effects of logging roads, which can effectively short-circuit the natural stream drainage network (Bowling et al. 2000).

Because logging roads are so closely associated with timber-harvest activities, it is often difficult to separate their hydrologic effects from that of logging alone. In spite of this complexity, several studies have found statistically significant increases in peak streamflow associated with forest-road networks (Harr et al. 1975; Jones and Grant 1996; LaMarche and Lettenmaier 1998; Jones et al. 2000; Bowling and Lettenmaier 2001). Therefore, the logging road network in the Grays River watershed may be affecting streamflow beyond that detected by the DHSVM associated with changes in watershed vegetation. The most significant hydrologic changes associated with logging roads may be on peak flows.

Logging roads have been shown to increase peak streamflow by modifying the mechanisms (i.e., shifting slower subsurface flows to more rapid surface runoff) and timing of hillslope runoff (Jones and Grant 1996; LaMarche and Lettenmaier 2001; Wemple and Jones 2003). In addition, less drainage area is needed to initiate channel formation in roaded areas than in natural, forested areas (Montgomery 1994). Roads also cause more rapid and more frequent “gully” (erosional) channel development (Wemple et al.

1996). In forested watersheds, road networks tend to be highly connected, hydrologically, to the natural stream channel network, resulting in more efficient transfer of precipitation volume to the stream system, resulting in generally more frequent, longer duration, and higher peak flows (Dunne and Leopold 1978).

Logging roads tend to be highly compacted surfaces and therefore have a much lower infiltration rate than the natural (forest) soils they have replaced. This results in an increase in surface runoff from forest roads (Reid and Dunne 1984; Montgomery 1994; Wigmosta and Perkins 2001; Wemple and Jones 2003). LaMarche and Lettenmaier (2001) found that the interception of subsurface flows by forest roads that cut into the natural soil profile, and sometimes into the underlying bedrock, were responsible for a majority of the runoff that was created in the study watershed.

Most of the runoff attributed to roads appears to be from that portion of the road network that is directly connected to the natural drainage system (Montgomery 1994; Wemple et al. 1996; Croke and Mockler 2001; Wemple and Jones 2003). Recognizing the hydrologic effects of logging roads is critical to understanding the hydrogeomorphic processes in watersheds affected by forest management. Forest roads and timber-harvest activities can have a synergistic effect on hydrologic alterations as well as on sediment production and transport (Montgomery 1994; Luce and Black 2001; Wemple et al. 2001). Clearly, road design, construction, and maintenance practices that reduce the alteration of the natural hydrologic regime will aid in controlling sediment production and transport.

4.0 Grays River Watershed Geomorphic Analysis

Rivers are dynamic ecosystems that are subject to an array of hydrologic and geomorphic influences exhibiting a significant range of spatial and temporal variability. Hydrogeomorphic processes play key roles in creating, maintaining, modifying, and destroying instream, floodplain, and riparian habitat. For the most part, these processes also act as the primary ecological disturbances that shape the river ecosystem and control dynamic-equilibrium conditions. Of particular importance are the processes involving water, sediment, and large woody debris (LWD), and how differences in historical, current, and potential future conditions are related to local conditions, watershed characteristics, and the influences of human activities (Montgomery and Bolton 2003). An understanding of these hydrogeomorphic processes is critical to developing an effective suite of conservation, enhancement, and restoration activities within the watershed.

The dynamic nature of hydrogeomorphic processes and the resulting variability in aquatic, riparian, and floodplain habitat characteristics are highly dependent on the spatial location within the river channel network (Montgomery and Buffington 1997). Headwater channels have a different suite of habitat-forming processes and disturbance regimes than mid-river mainstem channels, which are different from lower-river floodplain areas (Figure 4.1). In addition, local features along a channel network can enhance or dampen hydrogeomorphic variability. The longitudinal differences in hydrogeomorphic processes and disturbance within a river system will have a significant bearing on what restoration options should be considered for different segments of a river system (Montgomery and Bolton 2003).

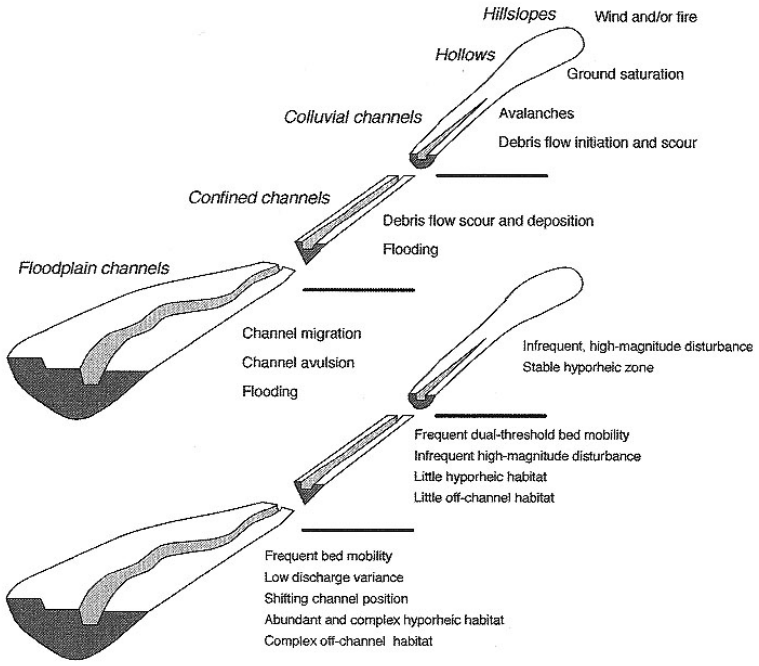


Figure 4.1. Disturbance Processes in River Systems (Montgomery 1999)

4.1 Upland Sediment Production

4.1.1 Background

Sediment production is a result of hillslope and erosional processes (i.e., mass wasting events, landslides, and debris flows), as well as instream processes including streambank erosion, flooding events, avulsions, and channel migration. Sediment source areas, delivery processes, transport, and storage mechanisms can be highly variable within a river system. Sediment production from hillslopes begins with the chemical and mechanical weathering of bedrock to create colluvium. The rate at which colluvium is produced is dependent on the regional tectonics, bedrock lithology, precipitation, ambient temperature, and vegetation. Much of the bedrock exposed at the surface within the upper Grays River watershed is highly weathered and is prone to erosion where vegetation has been removed. Hence, the watershed has the potential to yield large quantities of sediment. In the past, the native forest land-cover that dominated the watershed moderated both the production of sediment from hillslopes and the routing of sediment through the channel network. In most situations, sediment produced from the weathering of bedrock tends to be mobilized down-slope by soil creep, surface runoff, and mass-wasting events. Mobilized sediment either is deposited at the base of slopes or enters the channel network, where it is routed downstream by fluvial processes.

The morphology and dynamics of a river system are strongly influenced by the sediment supply in a watershed and transport capacity of the river. Changes in the balance between sediment supply and transport capacity can influence channel characteristics at several scales including habitat-scale features (e.g., pools and riffles), reach-scale channel types, and channel patterns (e.g., braided or single channel). In addition, changes in streamflow and/or sediment sources (whether due to natural or anthropogenic causes) can shift channel patterns and processes (Montgomery and Bolton 2003).

In large part, the routes and frequency of sediment delivery from the various sources in the watershed govern the sediment regime of a river system. In headwater channels, where debris-flow processes dominate sediment delivery, the natural sediment supply tends to be highly variable and largely episodic (e.g., caused by infrequent catastrophic events that deliver large pulses of sediment to the channel network). The sediment delivered to natural headwater channels is often stored within those small channels or downstream in larger tributary streams because of the typically high abundance of LWD that acts to trap most sediment within the upper watershed. Lowland, floodplain segments of natural river systems are usually insulated from upland hillslope processes and tend to have a relatively consistent supply of sediment that is often related to high streamflow (i.e., bankfull) events (Church 2002).

The morphology of lower-gradient river channels can change in response to changes in the sediment supply or delivery mechanisms of sediment from the upper watershed. If the intensity or frequency of hillslope processes (e.g., mass-wasting events or debris flows) increases, the sediment supply can dramatically increase as well (Montgomery 1999). Human land-use activities, such as timber harvest, can result in an increase in sediment production from upland areas due to an increased in landslides and from surface erosion from logging roads (Bilby et al. 1989; Campbell and Doeg 1989; Stott 1997; Dunne 2001). In mountain-rivers of the Pacific Northwest, hillslope areas tend to be highly connected to their headwater stream channels and, therefore, can deliver sediment to the river channel network relatively efficiently (Montgomery 1999).

In addition to an increase in sediment sources and the quantity of sediment delivered to stream channels, there also can be a decrease in sediment storage within stream channels. The loss of floodplain connectivity and side-channel areas within a river system can significantly reduce sediment storage capacity (Ward et al. 1999, 2001, 2002). Instream LWD, especially large logjams, can also be very effective in storing sediment within stream channels (Montgomery and Bolton 2003). In the Pacific Northwest region, local sediment storage by large logjams can be a significant morphological control feature within a river system (Montgomery et al. 1996). Even small instream sediment storage elements, such as small logjams or individual pieces of LWD, can store substantial quantities of sediment, especially when considered cumulatively (Montgomery and Buffington 1997; Abbe 2000; Abbe and Montgomery 2003; Reeves et al. 2003). The loss of instream LWD and logjams from headwater channels and mid-river reaches can trigger channel instability and result in the delivery of pulses of stored sediment to downstream segments of the river (Bilby 1984; Bilby and Ward 1989; Montgomery et al. 2003).

Recruitment of LWD and other organic material also results from hillslope and instream processes, as well as from windthrow and blowdown events. Woody debris deposited in streams by debris flows or other hillslope processes typically forms jams or dams within the upper watershed channel network. These natural features can create persistent, long-term instream sediment storage *nodes* or reservoirs within the river channel-floodplain system. Debris jams and dams also provide significant instream habitat complexity. In addition, debris jams and LWD dams tend to inhibit downstream propagation of sediment pulses. Furthermore, instream LWD and debris jams typically result in the creation of steps in the channel profile and terraces on the floodplain or valley floor. These features can persist in place even if the LWD is lost due to washout or after long-term decay. The role of LWD and debris jams also changes over time as the river channel evolves and the active channel locations shift (Abbe and Montgomery 2003).

In large part, upstream processes control the hydrology and sediment loading of downstream reaches of a river system. Instream LWD and debris jams naturally control sediment output to downstream reaches. In natural systems, sediment output tends to be relatively steady and generally shows little evidence of large, episodic inputs (Leopold et al. 1964). Abundant instream LWD plays a key role in moderating sediment flux from upstream when natural hillslope mass-wasting and debris-flow events do occur. On the other hand, landslides and debris flows that occur in areas with a lack of abundant LWD, such as is the case in watersheds impacted by long-term timber-harvest activities, often result in more frequent and larger episodic sediment flux to downstream channels and habitat features (Abbe and Montgomery 1996).

4.1.2 Methods

The assessment of current and historical geomorphic conditions within the upper Grays River watershed was conducted at both the watershed scale and reach scale using standard geomorphic assessment methods (Kondolf and Piegay 2003). Historical trends in sediment production from hillslopes were reconstructed at the watershed scale from timber-harvest records and historical aerial photographs, as well as from previous studies of natural background erosion rates for undisturbed forested basins. Aerial photographs were also used to evaluate the reach-scale response of alluvial valley segments to historical increases in sediment loading related to land-use practices. Historical records and aerial photographs were augmented with field investigations and data collection. Standard methods were employed in the sediment budget analysis (Reid and Dunne 1996).

Sediment generation within the Grays River watershed was also evaluated in terms of the dominant geomorphic processes observed during the field reconnaissance of the watershed and from aerial photographic analysis of principal sediment source terrains. Geomorphic processes responsible for sediment delivery to the channel network and included within this analysis are classified as mass-wasting events or landslides, surface erosion from disturbed hillslopes, road-surface erosion, and soil creep.

A 1990 WA-DNR Forest Practice Board and Technical Review Team memo (WA-DNR 1990) identified the Grays River watershed as having "exceeded the threshold for public resources" due to the following:

- slope instability and mass wasting due to timber-harvest activities on steep slopes
- surface erosion and sedimentation from logging roads
- aggradation and siltation of the Grays River, along with a loss of fish habitat
- degradation or loss of riparian habitat.

This chapter of the report begins with a discussion of sediment generation from mass wasting, road-surface erosion, and soil creep. Records of historical land-use activities were then used to reconstruct a historical sediment budget for the watershed, which is followed by a description of current channel conditions and channel response to increased sediment inputs. Details on methods used in the geomorphic assessment can be found in *Grays River Watershed Geomorphic Analysis* (Herrera 2005).

4.1.2.1 Current Hillslope Conditions

Current mass wasting and hillslope surface-erosion conditions were documented during a field investigation of the upper Grays River watershed conducted during October 2004. Field conditions are documented in the photographic log (Appendix B). The field investigation focused on landslides identified in the 2003 aerial photographs to quantify failure mechanisms, in-place and run-out volumes, landslide depth, and the estimated quantity of sediment delivered to the channel network (Herrera 2005).

Bulk sediment samples were collected from representative landslide deposits located throughout the upper Grays River watershed to characterize typical grain-size distributions of colluvial sediment supplied to the channel system. The sediment samples are representative of the various types of mass-wasting features found in the upper watershed and the parent rock type from which the deposits originated (Table 4.1). Sediment samples were submitted to a geotechnical laboratory for sieve analysis of sediment greater than 0.067 mm (see Herrera 2005, Appendix D).

Results of the sieve analyses are presented as cumulative grain-size distribution curves (Figure 4.2). Sediment dominated by fines tends to reach a cumulative fraction of 50% or greater within the sand domain (0.067 to 2.0 mm). Coarser sediment has a small cumulative fraction in the sand domain and an increasing fraction in the gravel domain.

A comparison of the cumulative particle-size distribution curves and lithology indicates that sediment samples can be aggregated into two groups based on lithology (Figure 4.2). With the exception of sample Grays-1, all of the Crescent basalt samples have weathered to a relatively coarse-grained distribution. Landslide deposits derived from the remaining geologic units (intrusive rocks, volcanic tuff, and marine sediments) include a greater proportion of fine-grained particles in their distribution (Figure 4.2). The exception is Grays-4, which is notably finer grained than the other three.

Table 4.1. Summary of Grays River Watershed Sediment Sample Characteristics (Herrera 2005)

Sample Identification	Lithology	D_{10} (mm)	D_{50} (mm)	D_{90} (mm)
Grays-1	Evb(gr) basalt	0.03	0.55	35.70
Grays-2	Evb(gr) basalt	0.06	25.44	106.49
Grays-3	Evb(gr) basalt	1.76	35.65	117.41
Grays-4	Evb(gr) basalt	0.45	29.81	104.65
Grays-5	Eib intrusive	0.03	1.09	109.02
Grays-8	Eib intrusive	0.01	2.22	27.35
Grays-7	Evt tuff	0.02	0.11	0.69
Grays-9	Evt tuff	0.02	1.37	50.20
Grays-6	Em(1) marine sediment	0.02	0.77	20.31
Grays-10	Em(1) marine sediment	0.04	3.46	19.77

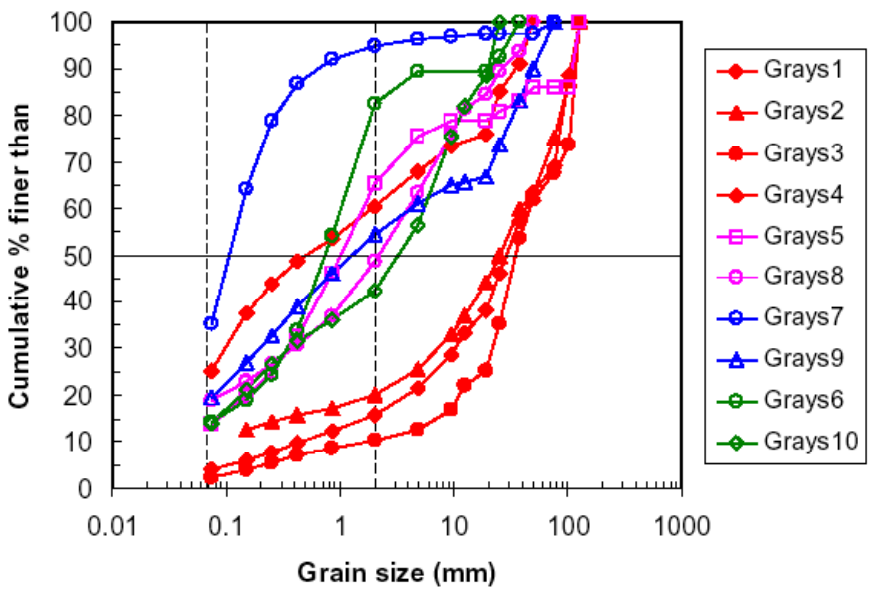


Figure 4.2. Grain-Size Distributions for Bulk Sediment Samples Collected from Grays River Watershed Landslide Deposits (Herrera 2005)

4.1.2.2 Landslide Inventory and Analysis

An inventory of historical landslides in the upper Grays River watershed was compiled from 1970, 1996, and 2003 aerial photographs (Herrera 2005). A total of 216 landslides were identified in the historical photographs (Appendix E). Figure 4.3 shows a view of typical landslides from the upper Grays River watershed. Figures 4.4 and 4.5 show close-up views of landslides. The 1996 and 2003 photo series cover the entire study area, whereas the 1970 set covers approximately 76% of the watershed. The 1970 landslide inventory was adjusted in proportion to watershed coverage to account for the missing areas and allow comparison with the 1996 and 2003 data sets.

Figure 4.3. Typical Translational Landslides in the Upper Grays River Watershed

Figure 4.4. Landslide Debris on Forest Road in the Upper Grays River Watershed

Figure 4.5. Shallow Translational Landslide from the Upper Grays River Watershed

Landslides were classified according to the three principal types of mass-wasting features identified in the aerial photographs and during the field reconnaissance. These landslide types include *channelized debris flows*, *shallow translational slides*, and *deep-seated rotational slides* (see Figure 4.6). Debris flows and translational slides were observed to be the most common landslide forms in the upper Grays River watershed.

Narrow mass-wasting features confined within low-order channels were classified as debris flows (Photos MW-2-4, MW-2-5, and MW-3-3 in Appendix B). Debris flows commonly scour steep channels to bedrock and may increase in bulk by entraining stored sediment as they travel downstream through the channel network.

Translational landslides observed during the field reconnaissance were typically wider than debris flows; these often begin on hillslopes (within bedrock) in the absence of convergent topography (Photos MW-4-1 and MW-4-2 in Appendix B). In some situations, upon reaching stream channels, shallow translational slides can transform into debris flows. These dual failures were observed during the landslide inventory and were classified as debris flows.

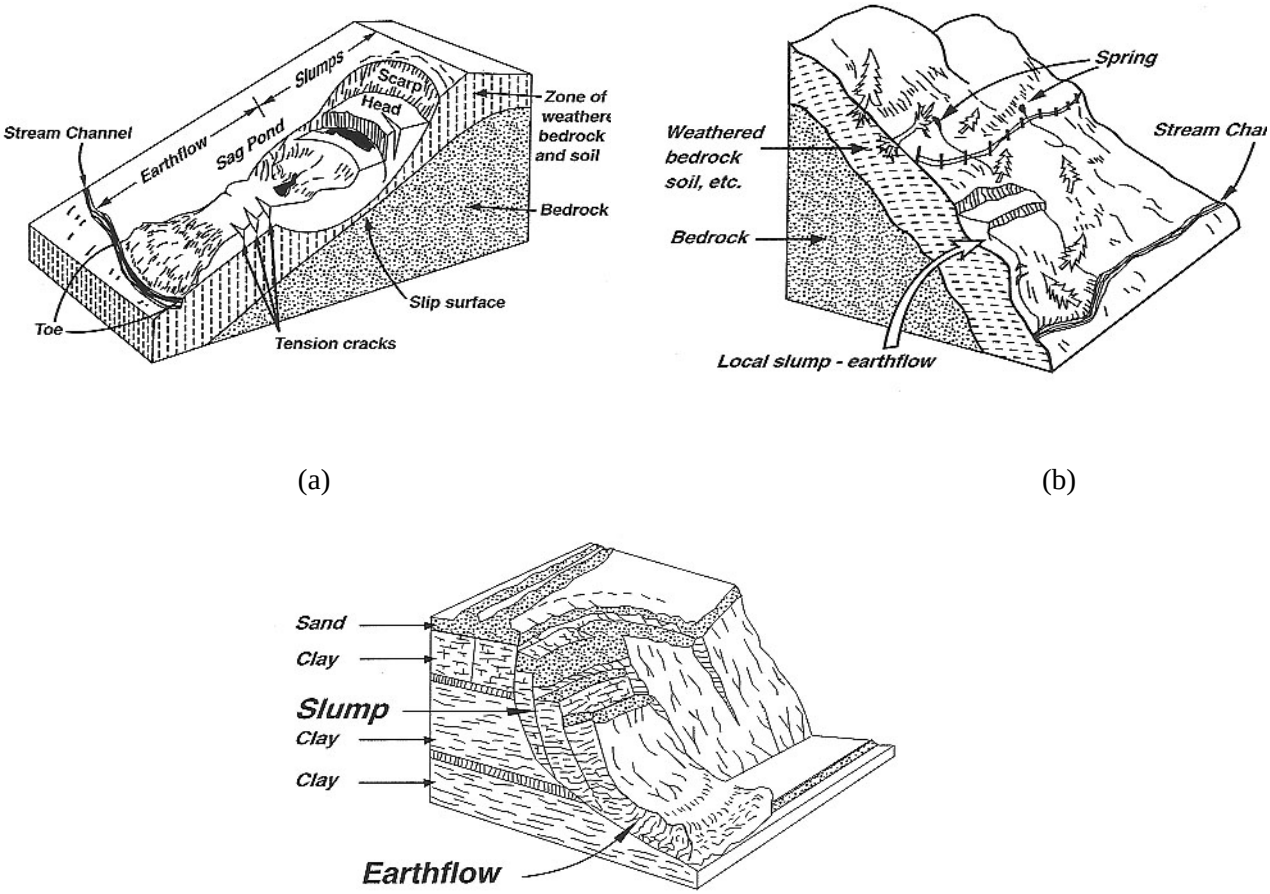


Figure 4.6. Examples of (a) Earth Flow, (b) Soil Creep and Shallow Translational Landslide, and (c) Deep-Seated Rotational Landslide (modified from Hobbs et al. 2002)

Rotational landslides were observed less frequently than either debris flows or translational slides and tended to exhibit visible head-scarps and spoon-shaped failure masses (Photos MW-2-4 and MW-2-5 in Appendix B). Field reconnaissance surveys included a detailed investigation of several large-scale rotational failures identified in aerial photographs and on geologic maps of the upper Grays River watershed (Photo MW-2-1 in Appendix B). Figures 4.7, 4.8, and 4.9 show three views of one of the surveyed rotational failures. Figure 4.10 shows a close-up view of another surveyed hillslope failure. Most of these large failures terminate on the valley floor and can intersect the active stream channel. However, in most of the surveyed sites it appears that only a small fraction of the total volume of sediment mobilized has been delivered to the channel network.

In general, large landslides have the potential to form valley-spanning dams that can cause significant realignment of the channel and local convexities in the channel longitudinal profile, which could interrupt sediment transport through the affected reach. Large boulders armoring the channel bed and a convex channel profile observed in the field at the terminus of one of these large rotational failures suggest that these landslides can deliver sediment volumes and particle sizes exceeding the transport capacity of some channels. At one site, lag-boulders exhumed from the landslide deposit by the winnowing of finer sediment during channel incision formed a step-pool stream channel morphology. Large woody debris at the front of the landslide also appeared to play a significant role in creating and sustaining the observed step-pool morphology. In many situations, armoring of the bed by lag boulders and LWD inhibits incision of these large landslide deposits by dissipating flow energy. Consequently, valley-spanning landslides can act as large sediment capacitors by both shielding landslide deposits from erosion and trapping sediment upstream of the deposit. The discontinuity in sediment transport through reaches affected by these large landslides was considered in the analyses of sediment routing.

Surface erosion of landslide areas was evaluated based on hillslope gradient, geology, and forest-harvest characteristics. Relatively minor surface erosion was observed in recently clear-cut areas during the field reconnaissance surveys. However, significant surface gully erosion was observed on road embankments and on the surfaces of recent landslide areas (Photos MW-2-2 and MW-2-3 in Appendix B). In all cases, gullies were located below culverts that apparently discharge concentrated flow onto bare hillslope surfaces.

Figure 4.7. Headscarp of a Rotational Hillslope Failure in the Upper Grays River Watershed

Figure 4.8. Headscarp of a Rotational Hillslope Failure in the Upper Grays River Watershed

Figure 4.9. Headscarp of a Rotational Hillslope Failure in the Upper Grays River Watershed

Figure 4.10. Hillslope Failure in the Upper Grays River Watershed

Previous studies in old-growth forests of the Olympic Peninsula found that timber entrained in landslides provided an effective means of limiting gully development on large landslide deposits (Abbe 2000). Where timber has been removed (e.g., clear-cut hillslopes), gully erosion tends to develop relatively soon if a landslide occurs on that slope. In general, gully development observed in the upper Grays River watershed is interpreted to be at least partially the result of timber removal on hillslopes. Gully incision of large rotational landslides tends to accelerate the delivery of stored sediment to the channel network. The gullies observed on the surface of the large rotational landslides surveyed in the upper Grays River watershed, suggest that the numerous historical landslides that exist throughout the watershed may be a chronic source of sediment, particularly where runoff from roads is concentrated onto landslide surfaces.

Annual rates of sediment yield from mass wasting were calculated from new landslides that occurred between the dates of the historical aerial photographs studied. Relative errors in the calculated sediment yield are reduced when the time elapsed between aerial photographs is significantly less than the time for revegetation of hillslopes disturbed by landslides. The 7-year span between the 1996 and 2003 photographs meets this criterion; however, some landslides that occurred early during the 26 years elapsed between the 1970 and 1996 photographs may not have been identified.

The general form of the equation for sediment yield to the channel network from landslides is given by

$$S_{LS} = \frac{A_{LS} D \Phi \rho}{\Delta t \ A_S} \tag{4.1}$$

where S_{LS} = sediment yield from landslides (tons/square kilometer/year)
A_{LS} = area of delineated landslide features in each map unit (square meters)
A_S = sample area for each year of aerial photographic coverage (square kilometers)

D = average landslide depth (meters)
Φ = delivery ratio
ρ = sediment density (tons/cubic meter)
Δt = duration of sediment yield to the channel network (years).

The average landslide depth used in the analysis was based on field measurements of landslide features made during the field reconnaissance (Table 4.2). Rotational landslides were found to have the greatest depth. The depth of translational landslides ranged from 1.5 to 12 m, with an average depth of 6 m. Debris flows were the shallowest of all the landslide types and had an average depth of 3 m.

Table 4.2. Average Depth and Sediment Delivery Ratios for Upper Grays River Watershed Landslide Types (Herrera 2005)

Landslide Type	Number Measured	Average Depth (m)	Average Delivery Ratio
Rotational	3	20	0.25
Translational	5	6	0.43
Debris flow	9	3	0.87

A significant fraction of the sediment mobilized by landslides may be deposited on slopes, so short-term sediment production rates may not be equivalent to rates of sediment delivery to the channel network. The delivery ratio (Φ) is the ratio between the amount of sediment contributed to channels and that mobilized by landslides on the hillslope. Average delivery ratios for each landslide type were determined from field measurements of landslide scar volumes and the sediment volume remaining on slopes. Delivery ratios were highest for debris flows and lowest for large, deep-seated landslides (Table 4.2).

The delivery ratios computed from our field investigation are consistent with results from prior studies within the region. Nelson and Booth (2002) found that delivery ratios decreased with landslide volume, ranging from 65% for landslides between 256 m^3 and about 1,000 m^3 to 100% for small failures along the stream bank. Large, deep-seated landslides typically experience relatively small displacements and persist for thousands of years as a source of chronic sediment generation (Dietrich et al. 1982; Reid and Dunne 1996). Alternatively, debris flows may convey the majority of their sediment loads to the channel network because of greater water content and confinement within low-order hollows (Dietrich et al. 1982). Sediment volume delivered to the channel network was converted to metric tons using a density (ρ) of 1.7 tons/m^3 (Madej 1982).

The time parameter (Δt) in Equation (4.1) was evaluated from the rate of regrowth and stabilization of disturbed areas, as shown on the time series of aerial photographs and inferred from the age of vegetation observed growing on recent landslides during the field reconnaissance. The tracking of landslide features through the aerial survey record suggests 20 years as an appropriate recovery time for the revegetation and stabilization of landslide scars and associated deposits. Smith et al. (1983) also found that surface erosion processes are active for 20 years on landslide scars in coastal British Columbia. Based on these findings, 20 years was used for Δt in Equation (4.1) to calculate the annual sediment yield from recent landslides.

The annual sediment yield calculated from the historical landslide inventory ranged from 209,000 to 238,000 tons/year for the entire upper Grays River watershed (Table 4.3). The calculated sediment yield (tons/year) represents the total mass of landslide-derived sediment delivered to the channel network during a one-year period. Sediment yield calculated from the landslide inventory was greatest in 2003. However, the relatively small scale and poor quality of the 1970 photos allow delineation of only the most recent and largest of the landslide features that existed at that time. Hence, the sediment yield calculated for 1970 should be considered a minimum rate.

Table 4.3. Historical Sediment Yield to the Upper Grays River Watershed from Landslides (Herrera 2005)

Aerial Photograph Year	Sediment Yield (tons/year)
1970	209,000
1996	221,000
2003	238,000
Sediment yield for 1970 was extrapolated for the entire watershed area due to limited aerial coverage (76% of watershed).	

The lag time between timber harvest and the peak in mass wasting was estimated from the variation in landslide frequency with time since harvest. The frequency distribution of the historical landslide inventory indicates that 72% of the landslides occurred between 10 and 20 years after timber harvest (Figure 4.11). Because the landslides could have occurred anytime between the photograph date and previous harvest, these estimates represent maximum lag times between harvest and sediment production from landslides. It is also likely that the timing and intensity of winter storms also influences the lag time between timber harvest and the peak in mass-wasting events.

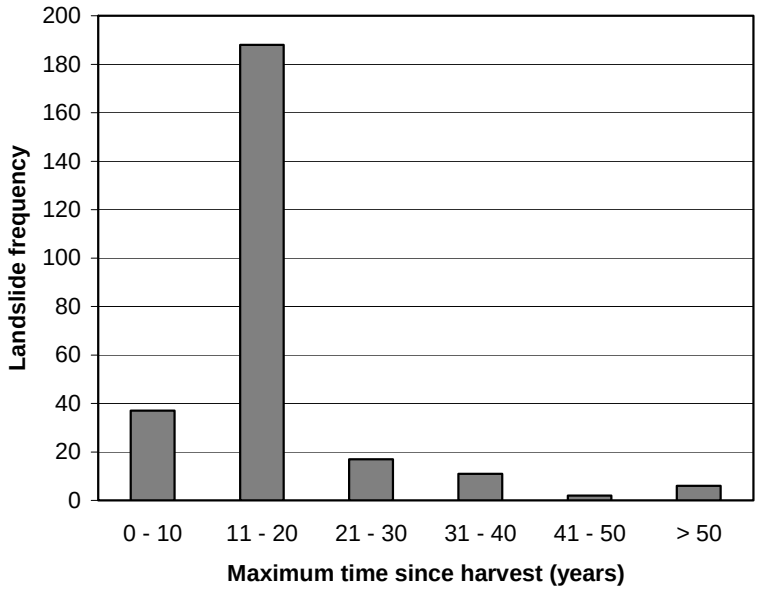


Figure 4.11. Variation in Landslide Frequency with Maximum Time Since Timber Harvest in the Upper Grays River Watershed (Herrera 2005)

The lag time measured between timber harvest and the onset of major landslides in the upper Grays River watershed is comparable to the time after harvest for root decay to reach a maximum (Figure 4.12). Schmidt et al. (2001) found that the minimum root strength occurred between 6 and 10 years after harvest in the Oregon Coast Range (Figure 4.13). Root strength can decline to 10% of the original strength within 5 years and weaken further to 5% of the original strength 10 years after harvest (Schmidt et al. 2001). Regrowth provided only 10% of the original root strength 10 to 12 years after harvest (Schmidt et al. 2001).

The combination of root strength decline from decay and increase from new regrowth provides an estimate of when the minimum root strength is likely to occur following harvest. The results of Schmidt et al. (2001) suggest that the greatest potential for storm-driven landslides occurs within the decade following harvest. The root decay and regrowth curves of Schmidt et al. (2001) also suggest that the frequency-based lag time inferred from the aerial photographs may overestimate the actual lag time.

Based on the landslide inventory data and results from prior studies, it is anticipated that mass wasting should peak approximately 10 years after timber harvest and deliver the majority of landslide-derived sediment to the channel network over the period between 10 and 30 years after timber harvest.

Results of the upper Grays River watershed landslide inventory indicate that roads also contributed significantly to historical mass wasting, either directly through slope modifications or indirectly by concentrating storm runoff on unstable slopes. Roughly half of all landslide areas delineated from the aerial photographs studied (i.e., 1970, 1996, and 2003) were found to intersect roads (Figure 4.13). In addition, visual inspection of the aerial photographs during delineation found that approximately three-fourths of all landslides were associated with roads (Table 4.4). Based on the average length of landslides and the slope distance between roads, this proportion is greater than that expected from a random distribution. Results suggest that road construction and road runoff may reactivate old landslides in addition to initiating new landslides, particularly in areas previously harvested.

When grouped according to proximity to roads, landslides associated with roads showed approximately the same frequency distribution as landslides unrelated to roads, with a peak in landslides between 10 and 20 years for both sample populations (Figure 4.14). However, landslides unrelated to roads appear to occur over a broader period, between 10 and 30 years after timber harvest.

As part of the overall geomorphic assessment, a GIS-based analysis of potential debris-flow initiation at road crossings was performed by comparing the road network with the headwater channel network in the upper Grays River watershed. Headwater channels dominated by debris flow processes were delineated in GIS from grid cells in the 10-m digital elevation model (DEM) with a drainage area between 0.05 and 0.2 km^2. Research indicates that roads constructed over hollows where runoff is concentrated over colluvial deposits have the potential to initiate debris flows (Stock and Dietrich 2003). Within the upper Grays River watershed, many of these hollows have already failed as debris flows at road crossings. Results of the GIS analysis indicate the presence of an estimated 1,797 road crossings within the study area capable of generating debris flows. Of these intersections, 98 are associated with historical landslides delineated from aerial photographs (Figure 4.14), and the remaining road crossings represent potential mass-wasting sites.

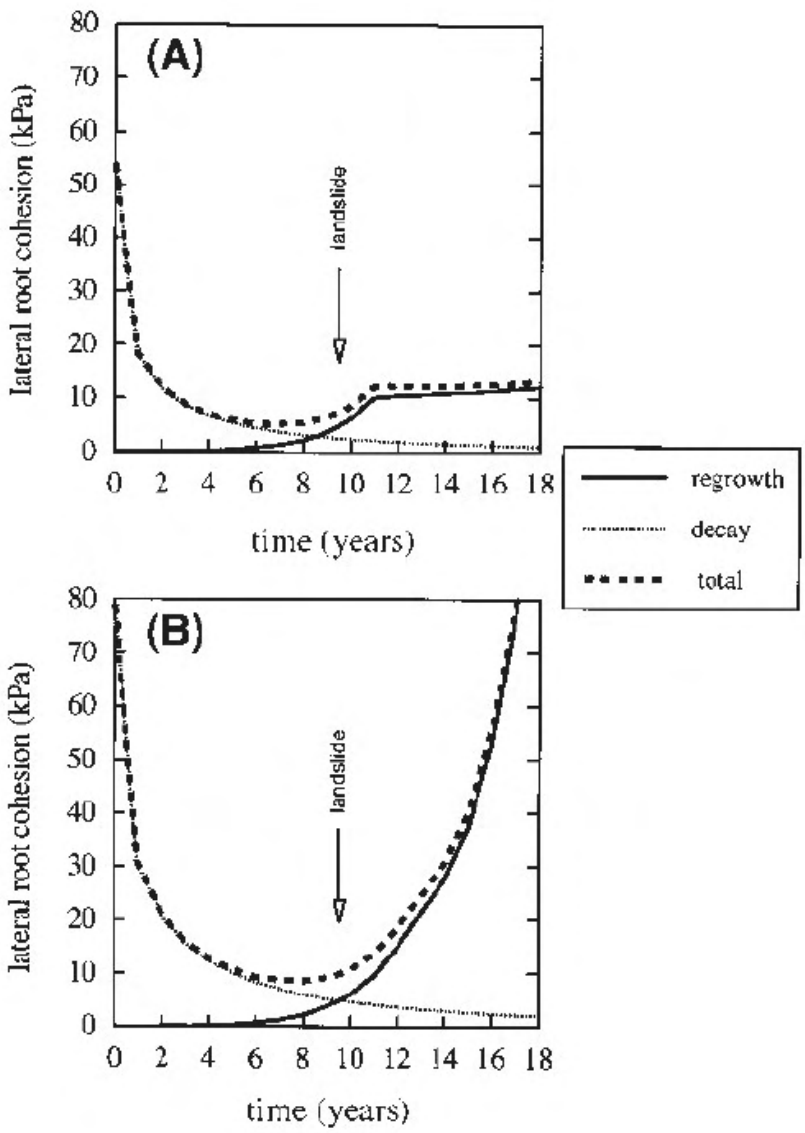


Figure 4.12. Root Growth and Decay Contributions of Lateral Root Cohesion After Clear-Cut Logging for Two Sites in the Oregon Coast Range (Schmidt et al. 2001)

Results of the landslide inventory also indicate a high degree of geomorphic coupling between hillslopes and the channel network. The majority of landslides (69-97%) terminated within 60 m (200 ft) of an active stream channel (see Table 4.4 and Appendix D). Hillslope-channel coupling is common in mountain channel networks such as the upper Grays River watershed, which are confined by adjacent hillslopes. Confined reaches lacking the buffering effects of a floodplain are more effective at routing the sediment generated by landslides. The strong hillslope-channel coupling in the upper Grays River watershed indicates a high response potential of the channel network to hillslope disturbance and sediment generation within the watershed. Appendix D provides a detailed summary of the landslide inventory conducted for the upper Grays River watershed.

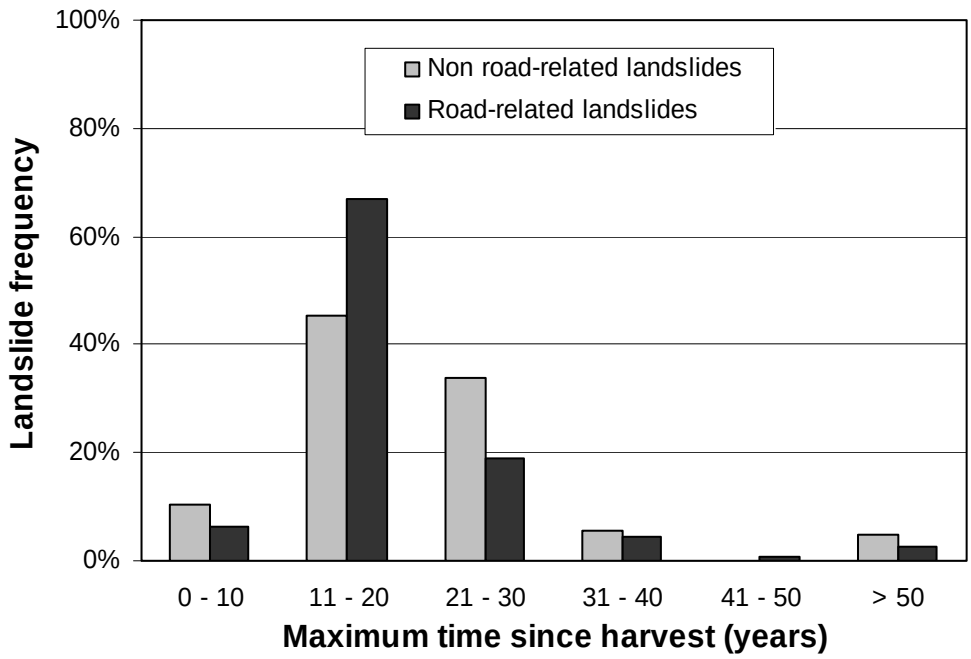


Figure 4.13. Landslide Association with Roads in the Upper Grays River Watershed (Herrera 2005)

Table 4.4. Landslide Association with Roads and Channels in the Upper Grays River Watershed (Herrera 2005)

Photo Year	Slides Intersecting Roads (GIS)	Slides Associated with Roads (visual)	Slides within 60 m of Channel Network (GIS)
1970	41%	–	69%
1996	52%	–	93%
2003	56%	76%	97%

4.1.2.3 Mass-Wasting Map Units

Mass-wasting map units (MWMU) were developed by correlating sediment yield from landslides with landscape characteristics (i.e., hillslope gradient, vegetation cover, soils, and underlying geology) and land-use practices (e.g., timber harvest and logging roads) that previous studies have linked to an increase in landslide frequency (Madej 1982; Grant and Wolff 1991; Jakob 2000). Although hillslope gradient and geology are assumed constant through time, there have been temporal and spatial variations in timber-harvest patterns throughout the upper Grays River watershed. MWMU provide a means of modeling the effects of both landscape characteristics and timber-harvest rotation on patterns of historical sediment yield from mass wasting.

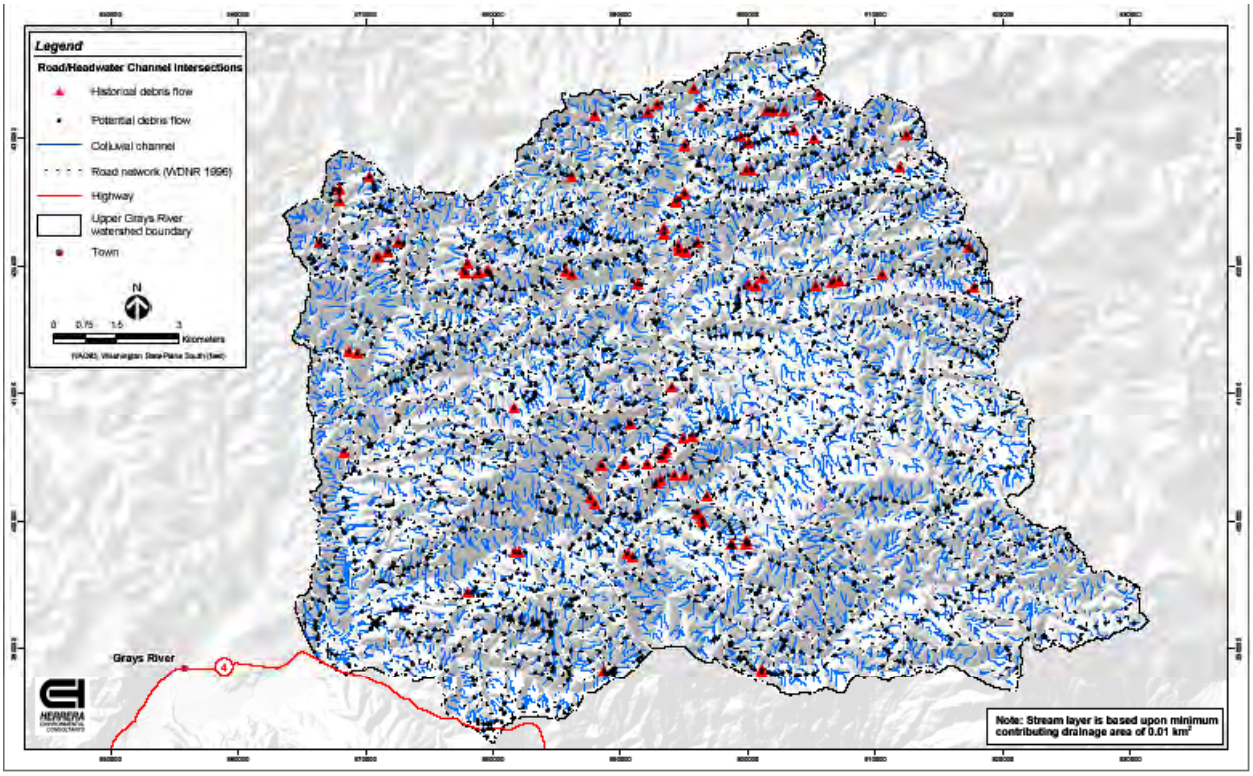


Figure 4.14. Locations of Historical and Potential Debris Flow Initiation Based on GIS-Derived Intersections of Roads and Colluvial Channels in the Upper Grays River Watershed, Washington (Herrera 2005)

Using GIS-based analysis tools, sediment yield from mass wasting was tabulated according to geology, forest age, and hillslope gradient into 24 MWMU. The MWMU were defined from geomorphic criteria developed for the analysis:

- two geologic classes (Crescent Formation basalt and marine sediments/intrusive rocks)
- four forest-age classes (0–10 years, 10–30 years, 30–60 years, and >60 years)
- three hillslope-gradient classes (0–30%, 30–65%, and >65%).

Geologic classes were defined according to the grain-size distribution of local colluvium and landslide deposits derived from weathered bedrock. Forest-age categories were defined based on 1) the anticipated time lag between timber harvest and peak in landslide frequency, when root strength is at a minimum (approximately 10 years), and 2) the duration of sediment delivery from landslides (20 years). The hillslope gradient coverage was derived from the 10-m digital elevation model of the watershed. Combining the GIS layers for slope, geology, and forest-age created each MWMU. A shorthand notation for the MWMU was developed based on geology, forest age, and slope. For instance, MWMU 123 signifies geology class 1 (Crescent basalt), harvest-age class 2 (10–30 years), and hillslope-gradient class 3 (slope >65%). Sediment yield from mass wasting (tons/square kilometer/year) was calculated for each MWMU in GIS using the intersection of the 2003 landslide inventory with geology, forest age, and slope layers. Sediment yield computed for each MWMU is tabulated in Table 4.5.

Table 4.5. Upper Grays River Watershed Sediment Yield (tons/square kilometer/year) by Mass-Wasting Map Unit (Herrera 2005)

Slope and Forest-Age Class	Geology 1			Geology 2		
	0%<S<30%	30%<S<65%	S>65%	0%<S<30%	30%<S<65%	S>65%
H<10 yrs	30 (111)	30 (112)	30 (113)	30 (211)	30 (212)	30 (213)
10<H<30 yrs	145 (121)	737 (122)	22,391 (123)	137 (221)	2,041 (222)	33,659 (223)
30<H<60 yrs	1,822 (131)	5,998 (132)	17,456 (133)	1,822 (231)	6,792 (232)	17,456 (233)
H>60 yrs	30 (141)	30 (142)	30 (143)	30 (241)	30 (242)	30 (243)
Note: Map-unit shorthand shown in parentheses (see text for explanation).						

Recent landslide features were not identified in either the youngest or the oldest forest-age classes except for map unit 143, in which two landslide polygons were found. Therefore, a background sediment yield of 30 tons/km^2/year was assigned to these 12 MWMU. An anomalously low sediment yield from MWMU 133 (the map unit with the smallest drainage area) was replaced with the sediment yield from a comparable map unit (233) in geology class 2.

Three large landslides (two in MWMU 331 and one in MWMU 231) occurred between 1996 and 2003 and were an order of magnitude larger than any other landslide feature. These three landslides are deep-seated bedrock failures unrelated to the road network and occurred several decades after local harvest activities. Deep-seated landslides such as these occur naturally in the region and likely comprise a major component of the background sediment yield. Topographic features suggesting ancient landslides of a similar scale were also observed during field surveys (Herrera 2005). Inclusion of these landslides in sediment budget calculations, however, significantly skews the sediment yield for MWMU representing older forest-age classes, and causes annual sediment yield rates to fluctuate by about 50,000 tons/km^2/year. Based on the results of several methods used to estimate background sediment yield (presented later in Chapter 4), the recurrence interval of such features is thought to far exceed the period analyzed in this study, and these landslides were therefore removed from the sample population.

Results from this MWMU analysis indicate that the potential for sediment yield from mass wasting is defined primarily by slope and time since harvest. Sediment yield from mass wasting is highest on slopes greater than 65%. Results indicate that sediment yield for a given geologic and hillslope gradient class is highest where harvest occurred between 10 and 30 years ago and between 30 and 60 years ago. Findings indicating a peak in sediment yield between 30 and 60 years are inconsistent with the well-established peak in landslide frequency measured 11–20 years following harvest (see Figures 4.11 and 4.12). The map units where maximum sediment yields were calculated to occur between 30 and 60 years ago (131, 132, 231, and 232) were small in areal extent (i.e., less than 4 km^2) and contained relatively few land-slides spanning an order of magnitude in size. Estimates of sediment yield from these map units are expected to have high relative errors, but because of their small areas, they do not have important impacts on total watershed sediment yield. Sediment yield results by MWMU were used for further analyses to reconstruct historical trends in sediment yield from the upper Grays River watershed.

4.1.2.4 Assessment of Surface Erosion from Disturbed Hillslopes

Surface erosion occurs where bare soil is exposed to rain and wind, particularly after the loss of vegetative cover. Surface erosion may also be triggered by the concentration of surface runoff from road drainage or logging activities. The factors that influence surface erosion include soil type, hillslope gradient, rainfall intensity and duration, vegetative cover, and soil compaction (WA-DNR 1997). The potential for surface erosion from disturbed hillslopes in the upper Grays River watershed was evaluated based on soil erodibility and hillslope gradient.

The *erodibility* of soil is expressed in terms of an empirical K factor developed from the universal soil loss equation. The K factor is based on the clay, silt, sand, and organic content of soil. Soils high in silt and fine sand content typically have a higher K value and are more erosive (WA-DNR 1997).

A GIS coverage of soil type in the upper Grays River watershed prepared by the Natural Resources Conservation Service (NRCS 2004) includes two types of K factors. The erodibility of fine soils less than 2 mm in particle diameter is characterized by the fine-earth factor, K_f. The reduction in erosion from the effects of shielding by rock fragments is accounted for in the whole-soil factor, K_w. Due to the presence of rock fragments in the upper Grays River watershed soils, surface erosion potential was evaluated using K_w values (NRCS 2004).

The use of hillslope gradient in the analysis of surface erosion accounts for the effects of gravity, which causes particles detached by rain-splash or sheet-wash to move down slope. Sheet-wash and rill-erosion also tend to become more effective erosional agents with increasing gradient. The combination of K_w factors and slope categories was used by WA-DNR (1997) to define low, moderate, and high erodibility ratings (Table 4.6).

Table 4.6. Erodibility Ratings Based on K_w Factors and Slope (WA-DNR 1997)

Slope Class	K_w <0.25 Not Easily Detached	0.25 <K_w <0.40 Moderately Detached	K_w >0.40 Easily Detached
<30%	Low	Low	Moderate
30 – 65%	Low	High	High
>65%	Moderate	High	High

Erodibility ratings were generated for the upper Grays River watershed by combining the GIS coverage-layers for both K_w factors and slope (Figure 4.15). The erodibility ratings shown on the erosion potential map reflect the maximum surface erosion from a bare hillslope as a function of soil type and slope and do not account for variations in vegetative cover, land use, or precipitation.

Approximately 37% of the Grays River watershed is rated with high soil erosion potential, 9% of the watershed is rated with a moderate soil erosion potential, and the remaining 54% of the watershed is rated as low for soil erosion potential. The majority of current timber-harvest activities (between 1996 and 2003) are located in areas with relatively high erosion potential.

Field observations of surface erosion in areas of recent timber harvest indicate that sediment eroded by rain-splash and sheet-wash is displaced a relatively short distance down slope and deposited in transient storage sites provided in hummocky terrain and organic material remaining on disturbed

hillslopes. The majority of surface erosion from gullies was observed on road embankments and on the surface of recent landslides. Hence, sediment yield from disturbed surfaces is not accounted for in the assessment of mass wasting and roads; it is expected to be an insignificant component of the total sediment yield for the watershed and therefore is not included in the sediment budget. However, the erodibility ratings and surface erosion potential map (Figure 4.15) provides valuable information on the potential for chronic erosion from landslide and road surfaces.

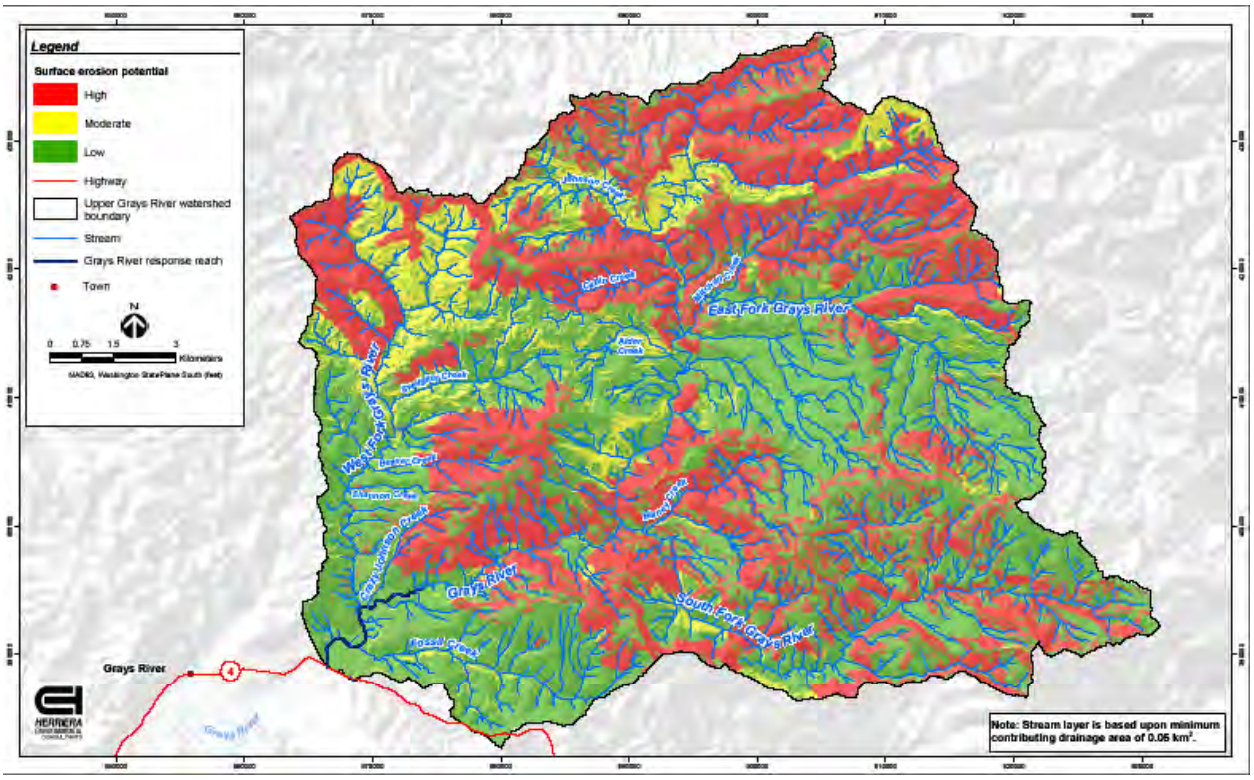


Figure 4.15. Ground Surface Erosion Potential Based on Whole Soil Erosion Factors and Slope in the Upper Grays River Watershed (Herrera 2005)

4.1.2.5 Assessment of Surface Erosion from Roads

Forest roads provide several necessary functions, including access for timber harvest and recreation, as well as fire detection and suppression. However, these same roads result in multiple unintended negative impacts on the forest ecosystem and associated aquatic resources. The construction and use of roads can be a major source of sediment in forested basins. Sediment that reaches streams, wetlands, or lakes can have an impact on water quality, fish, and other aquatic life (WA-DNR 2003).

The specific impacts of logging roads, some of which have already been discussed, include increased surface-water runoff (Harr et al. 1975; Jones and Grant 1996; Wemple et al. 1996; Bowling and Lenttenmeier 1997), road-surface erosion (Reid and Dunne 1984; Duncan et al. 1987; Bilby et al. 1989), and landslide initiation (Megahan and Kidd 1972; Swanson and Dyrness 1975; Sessions et al. 1987; Megahan 1991; Spies et al. 2002). The typical forest road network interfaces with the natural stream-channel network to influence the routing of water and sediment, as well as soil movements (i.e., landslides, debris flows, and mass-wasting events) on forest hillslopes (Figure 4.16).

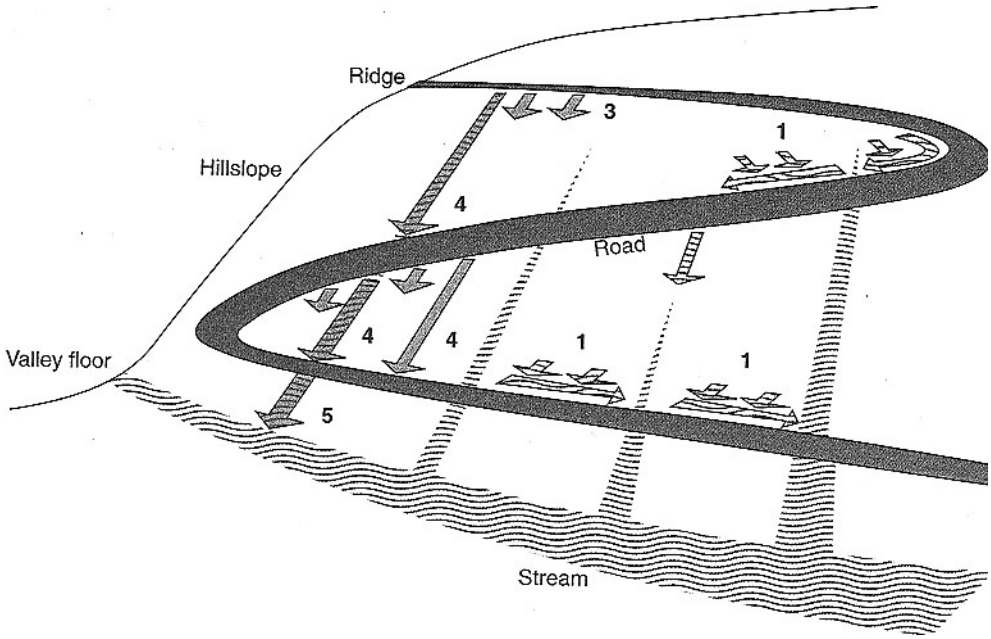


Figure 4.16. Interactions Between Forest Roads and Stream Channels in Terms of Water and Sediment. Roads can intercept surface and subsurface water flows (1), alter water flow-paths (2), initiate soil movement and mass-wasting events (3), aid in transporting sediment on hillslopes (4), and act as a conduit for sediment to be deposited in stream channels (5) or valley floors. (Modified from Spies et al. 2002)

Increased sediment from roads is caused by three major erosion processes: surface erosion, gully erosion, or mass wasting (i.e., landslides). Each of these processes can be important. However, surface erosion occurs on all roads, whereas gullies and landslides are limited to specific locations on steep slopes and/or unique geologic and soil conditions. Surface erosion tends to produce fine-grained sediment (sand, silt, clay) that can harm fish and other aquatic organisms if it enters streams (WA-DNR 2003).

Surface erosion is defined as the detachment of individual soil particles by a force such as raindrop impact, overland flow of water, wind, or gravity. Detachment of soil particles depends not only on the amount of external force applied but also on how well the soil particles tend to resist separation. This latter factor is an inherent soil property termed soil erodibility and is strongly influenced by the texture (grain size) of the exposed soil. Generally, gravelly or cohesive soils are not as easily eroded as sandy or silty soils (WA-DNR 2003).

In naturally forested watersheds, a thick layer of duff protects the soil from surface erosion, and most rainfall and snowmelt infiltrates into the soil. However, construction of forest roads in mountainous terrain can lead to high rates of surface erosion due to (WA-DNR 2003):

- removal of vegetative cover and surface protection
- construction of cut and fill slopes that are steeper than the original hillslope to obtain a relatively level driving surface

- greatly increased potential for overland water flow due to soil compaction and concentration of runoff

- interception of groundwater by the cut slope, which is usually the primary cause of sediment transport from the roadway.

Compacted road surfaces, long lengths of roads without cross-drains, areas with heavy rainfall, and soils prone to gully formation are more likely to result in transport of eroded sediment off the road prism (Figure 4.17). Transport of sediment to a stream is most likely to occur when the road is close to a stream, there is a steep slope between the road and the stream, and/or there are few obstructions to slow down or trap the sediment. Sediment is likely to be trapped or deposited before it enters a stream if it is produced from roads far from a stream, or from roads with a vegetative buffer or in areas with a topographic low-point between the road and the stream (WA-DNR 2003).

Figure 4.17. Typical Forest-Road Prism (WA-DNR 2003)

On a newly constructed road, each component of the road prism is typically exposed to erosional processes. Over time, the cut-slope and fill-slope areas may revegetate and erosion from these sources may be reduced. In most established road networks, fill-slope areas tend to have nearly 100% vegetative cover, and do not deliver sediment to streams. However, the road-tread surfaces and ditches, and to a lesser extent the cut-slope area, can continue to produce sediment as long as the road is in use. Research has shown that the most important factors determining how much sediment is produced from the road tread are how much the road is used, and the amount and type of road surfacing (WA-DNR 2003). The amount of cover on the cut-slope, armoring in the ditch, and whether or not these surfaces have been re-graded recently, also affect erosion from these components. In addition to these factors, the configuration of the road drainage system, particularly whether or not road drainage reaches the stream network, determines if sediment produced from roads has the potential to affect aquatic resources (WA-DNR 2003).

In Washington State, the WA-DNR has implemented a Road Maintenance and Abandonment Plan (RMAP) program for forest roads. One of the goals of the RMAP program is to ensure forest roads are maintained in a way that helps protect fish and aquatic organisms from the harmful effects of sediment produced from the road system. The RMAP road-maintenance practices are designed to improve many

aspects of the road network by reducing the likelihood of road landslides, culvert plugging, road surface erosion, and fish passage barriers. The Washington Road Surface Erosion Model (WA-DNR 2003) has been designed based on the surface erosion assessment in the Watershed Analysis Manual (WA-DNR 1997) to support a number of assessment and monitoring needs related to roads. The primary motivation is for use as a monitoring tool, but the model can also be used to estimate the amount of sediment supplied from the surface erosion component of their road system and the relative effectiveness of different measures to reduce road surface erosion to meet established RMAP goals (WA-DNR 2003).

The Washington Road Surface Erosion Model produces a long-term average for sediment production resulting from surface erosion of logging roads. The erosion calculations are based on a set of empirical relationships that have been developed from research on road erosion. Based on measurements of road surface erosion it has been determined that the amount of sediment delivered to streams from forest roads is influenced by a number of factors including the physical setting, the proximity of the road to a stream, the condition of the road, the amount and intensity of rainfall and the amount and type of traffic. The actual quantity of sediment eroded from a particular road segment varies greatly from year to year as a result of differences in precipitation, traffic, and maintenance activities. Due to the complex nature of forest road networks, our ability to measure or predict all of these factors precisely at each location is limited. However, it is useful to predict where roads have the potential to produce relatively high amounts of sediment based on our current understanding of road erosion processes and typical conditions of each road segment. The model output, in average annual tons of sediment per year, allows road managers to identify road segments that are most likely to produce larger amounts of sediment, and to determine the relative sediment savings from a variety of management practices (WA-DNR 2003).

The construction and operation of logging roads can comprise a significant fraction of the total sediment yield from harvested basins (Reid and Dunne 1984, 1996; Madej 1982; Bilby et al. 1989; Skaugset al. 2002). Based on a review of these and other studies, WA-DNR (1997) compiled a list of factors that influence sediment production from road surfaces and delivery to the channel network. These factors include

- road density
- surface area of the road and disturbed road corridor
- traffic use
- road surface erodibility
- distance to the nearest channel.

As part of the Grays River geomorphic assessment, physical parameters on the road network were collected and mapped using GIS and in the field to evaluate surface erosion from roads. Current conditions of roads were characterized by measuring road width, centerline slope, cross-slope condition (i.e., crowned, in, or away from bank), and road-base condition. Measurements also included the height, slope, and fraction of vegetative cover on cut slopes and the dominant grain size of sediment stored in roadside ditches.

4.1.2.6 Road Density Analysis

Road density is defined as the total length of roads in a particular subbasin divided by the area of that subbasin. Road density is often used as a general measure or metric of human influence in watersheds where forest management (i.e., timber harvest) is the primary land-use activity. Although road density

can provide a useful measure of overall forest-management impact, it may not provide a complete picture of the level of impact. For example, while reducing road density within a watershed is often cited as a sediment-reduction goal, a program that concentrates solely on reducing overall road density may not achieve significant sediment reduction in receiving waters. As has been discussed, the condition of road surfaces, the characteristics of cut and fill slopes associated with forest roads, the proximity of roads to stream channels, and the connectivity of the road and stream network (e.g., cross-culverts, ditches) all are important factors that must be considered when targeting roads for sediment-reduction treatments or decommissioning (WA-DNR 1997, 2003).

Using GIS-based forest road data (WA-DNR 1996; The Campbell Group 2004) for the upper Grays River watershed a road-density analysis was performed. Seven of the 60 subbasins within the upper Grays River watershed were chosen at random for this road-density analysis (Figure 4.18). Roads represented in the 1996 WA-DNR (GIS) road layer within each of these subbasins were compared to the roads on the 2004 Campbell Group maps. Roads present in the 2004 map but not in the 1996 map were digitized in GIS and added to the *master GIS road layer*. A subset of the added roads was labeled as *added harvest roads* if they were associated with active harvest operations identified in the 1996 and 2003 aerial photos. Roads present on the 1996 WA-DNR map but absent from the 2004 Campbell Group map (and which did not appear to be active during the 2003 aerial photo analysis) were digitized and labeled as *decommissioned roads* in the GIS database. An empirical relationship was derived for the added road

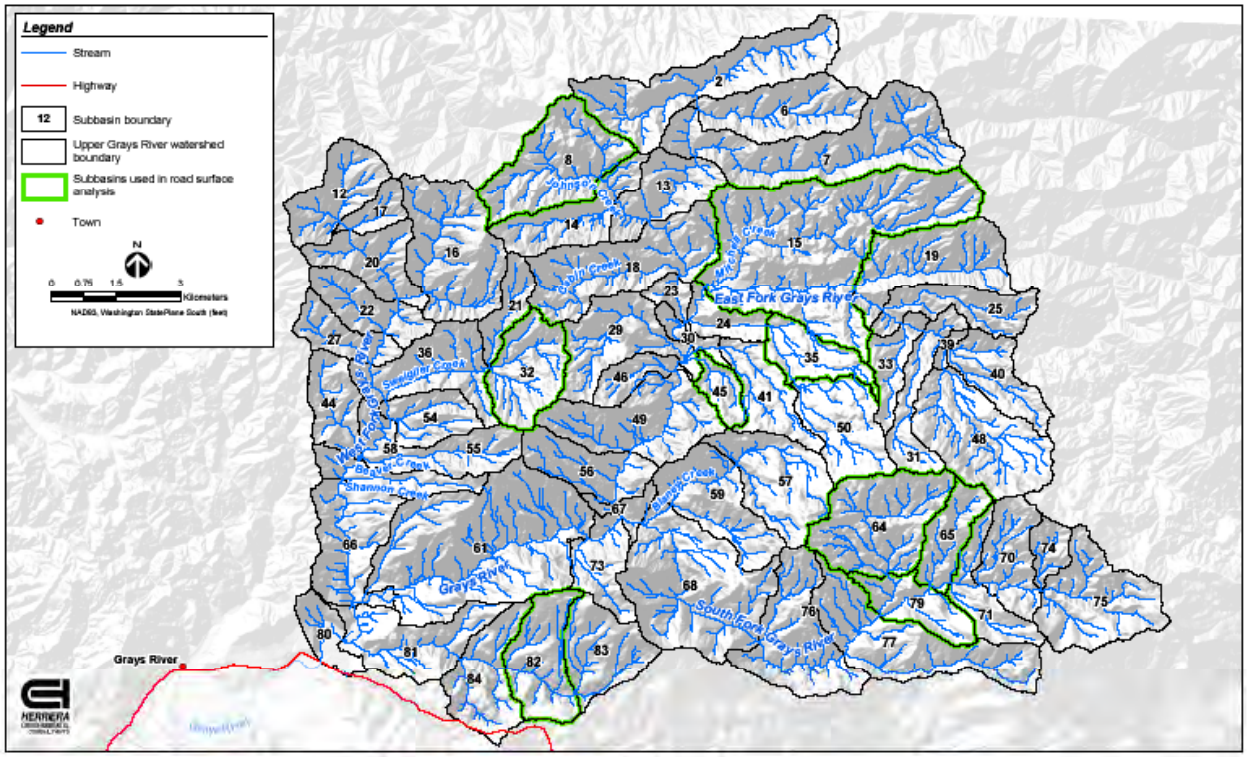


Figure 4.18. Upper Grays River Watershed Subbasins Used for Analysis of Road Surface Erosion Sediment Production (Herrera 2005)

density as a function of the fraction of the subbasin that had been harvested between 1993 and 2003 (Figure 4.19). The road length for the nonharvested subbasins delineated in the 1996 road coverage was not adjusted. Abandoned or decommissioned road lengths for the remaining harvested and non-harvested subbasins were characterized using the average decommissioned road densities of the seven sample subbasins and two additional representative subbasins. Representative road-density data for the sampled upper Grays River watershed subbasins are presented in Table 4.7. For comparison, the average road density for the southwestern Washington region is 4.6 mi/mi^2 (WA-DNR 2003).

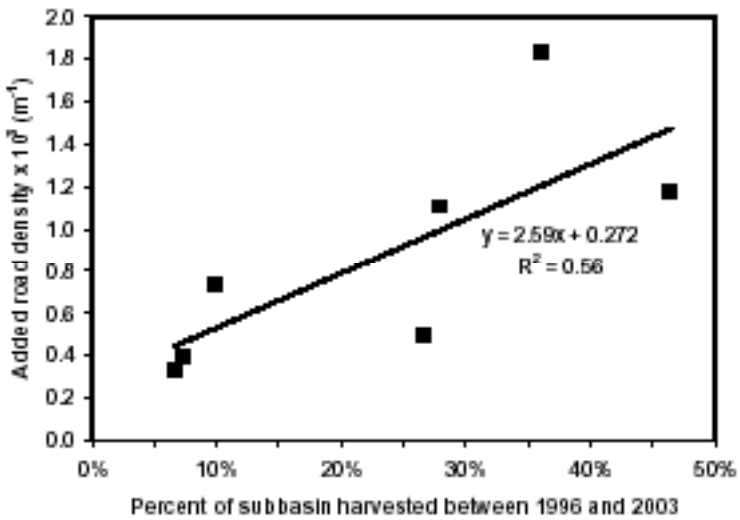


Figure 4.19. Linear Regression for Added Road Density Associated with Harvesting Between 1996 and 2003 (Herrera 2005)

Table 4.7. Road Densities for Representative Subbasins in Upper Grays River Watershed (Herrera 2005)

Subbasin #	Subbasin Drainage Area (km^2)	Percent of Subbasin Harvested Between 1996 and 2003	1996 Road Density (m^{-1}) x 10^{-4}	2003 Road Density (m^{-1}) x 10^3	Added Road Density (m^{-1}) x 10^3	Added Harvest Road Density (m^{-1}) x 10^3	Decommissioned Road Density (m^{-1}) x 10^3
8	5.53	6.6	3.75	4.03	0.33	0.14	0.05
15	15.24	0	4.29	4.45	0.20	0.00	0.04
32	3.69	0	3.65	3.95	0.30	0.00	0.00
45	1.19	10	2.86	3.60	0.74	0.51	0.00
64	5.95	36	1.78	3.55	1.83	1.58	0.06
65	2.30	47	3.13	4.13	1.17	0.96	0.17
79	2.51	27	2.79	3.29	0.50	0.38	0.01
82	--	--	--	--	--	--	0.00
35	--	--	--	--	--	--	0.20

Road density computed as road length per basin area.

4.1.2.7 Road Surface Erosion

For the upper Grays River watershed, erosion from road surfaces and adjacent slopes and ditches within the road corridor (i.e., the road prism) was evaluated using a baseline erosion rate and several adjustment factors that correlate erosion rates with road characteristics (WA-DNR 1997 and 2003). Roads were also classified according to use as mainline, secondary, non-active (inactive), and decommissioned. The factors represent empirical relationships between characteristics of the road prism and the difference in erosion rates relative to reference conditions. The *reference road* characteristics include (WA-DNR 1997)

- in-sloped with ditch
- native surface road tread and ditch
- general-use traffic
- cut-slope 1:1 and fill-slope gradient 1.5:1
- initial ground cover density of zero on cut and fill slopes
- sustained grade of 5–7%.
- average cross-drain spacing of 500 ft.

The reference erosion rate from roads for the upper Grays River watershed was developed from 10 representative subbasins (see subbasins 8, 15, 32, 35, 45, 64, 65, 79, 82, and 84 on Figure 4.18). Road maps, aerial photographs, and field observations of road conditions were used to classify roads within the subbasins into four groups based on traffic use. These groups included

- Mainline = active roads with evidence of heavy logging traffic

- Secondary = active (non-mainline) roads that are present on the 1996 and 2004 maps as well as on aerial photographs

- Non-active or Inactive = roads with little evidence of active use that are present on the 1996 and 2004 maps but not highly visible from aerial photographs

- Decommissioned = roads that are present on the 1996 maps but not on 2004 maps and not visible in aerial photographs.

Because there are few mainline roads in the basin (e.g., 7200 road), the length of mainline roads was delineated for the entire upper Grays River watershed using only the 1996 GIS coverage. Field observations and comparisons with the 2003 aerial photos and the 2004 road maps indicated that no significant mainline road lengths were added between 1996 and 2004. This was confirmed in discussions with WA-DNR and the Campbell Group. Approximately 2% of roads in the upper Grays River basin are mainline roads. Of the remaining active roads in the 10 representative subbasins, 85% were classified as secondary and 15% as non-active (inactive) roads. This proportion of secondary and non-active roads delineated in the 10 sampled subbasins was then applied to all subbasins in the upper Grays River watershed. The length of decommissioned roads in each subbasin was estimated based on the representative subbasin data. The total area of road surface in each subbasin was then calculated from the delineated road length and the average width of road types measured during field investigations. The width of decommissioned roads was characterized using the average width of the nonactive roads. Average road widths measured in the field are shown in Table 4.8.

Table 4.8. Average Road Width Measured in the Field (Herrera 2005)

Road Category	Average Road Width (m)
Mainline	8.1
Secondary	6.1
Non-Active	5.4
Decommissioned	5.4

Reference erosion rates were calculated for each road type and for each component of the road prism using the proportions given in Table 4.13. The baseline erosion rate is given by the following equation (WA-DNR 1997):

$$E_{bl} = (F_{rp})*(L_r)*W_r)*(E_{ref}) \tag{4.2}$$

where E_{bl} = baseline erosion rate for each component of each road type (tons/year)
F_{rp} = road prism factor (Table 4.13)
L_r = road length (meters)
W_r = road width (meters)
E_{ref} = reference erosion rate (tons/acre).

The reference erosion rate selected for the analysis was based on erosion criteria for sediment production (i.e., low, moderate, and high) the parent material, and road age (Table 4.10). A reference erosion rate of 30 tons/acre/year was used for this analysis. Baseline erosion rates for each road type and prism component were then adjusted for vegetative cover, surface material, traffic use, and precipitation.

Table 4.9. Proportions of Total Long-Term Average Road Erosion Rates Attributed to the Components of the Standard Road Prism (WA-DNR 1997)

Component of Standard Road Prism	Proportion of Total Long-Term Average Road Erosion Rates
Tread	40%
Cutslope ditch	40%
Hillslope	20%

4.1.2.8 Cover Factor

Cover factors for cut and fill slopes refer to all surfaces other than bare soil (e.g., vegetation, rock, slash, and erosion control materials). Because reference road conditions are defined with un-vegetated cut and fill slopes, cover protecting these slopes will tend to reduce the baseline erosion rate. Correction factors for different ground cover densities are shown in Table 4.11. Average ground cover densities measured during field road surveys are summarized for each road prism component in Table 4.12.

Table 4.10. Reference Erosion Rates of Road Prism per Year by Road Age and Parent Material, Based on Reference Road Conditions (WA-DNR 1997)

General Category	Parent Material	Erosion Rates (tons acre) of Road Prism per Year	
		Road Age	
		New 0-2 Years	Old >2 Years
High	Mica schist Volcanic ash Highly weathered sedimentary	110	60
High/Moderate	Quartzite Coarse-grained granite	110	30
Moderate	Fine-grained granite Moderately weathered rock Sedimentary rocks	60	30
Low	Competent granite Basalt Metamorphic rocks Relatively unweathered rocks	20	10

Table 4.11. Correction Factors for Ground Cover Density (F_{gc}) on Cut and Fill Slopes (WA-DNR 1997)

Ground Cover Density	F_{gc} Factor
>80%	0.18
50%	0.37
30%	0.53
20%	0.63
10%	0.77
0%	1.00

Table 4.12. Average Ground Cover Densities (F_{gc}) Measured from Road Surveys Performed in the Upper Grays River Watershed in October 2004

Road Category	Average Ground Cover Density, F_{gc} (%)		
	Road Tread	Cutslope Ditch	Hillslope
Mainline	0	92	81
Secondary	20	68	84
Non-Active	60	45	85
Decommissioned	>80	>80	>80

4.1.2.9 Surface Material Factor

The reference erosion rate is based on a road surface of native material. Therefore, improvements to the road surface material will reduce erosion from the road surface relative to the baseline erosion rate. Correction factors for different surface materials are shown in Table 4.13. The surface material of most roads in the upper Grays River watershed are made up of gravel greater than 6 inches deep; therefore, a correction factor of 0.20 was applied to Equation (4.2) (WA-DNR 1997).

Table 4.13. Road Tread Correction Factors (F_{rt}) for Road Surface Materials (WA-DNR 1997)

Surfacing Material	F_{rt} Factor
Paved	0.03
Dust-oil	0.15
Gravel, >6 inches deep	0.2
Gravel, 2-6 inches deep	0.5
Native soil/rock	1.00

4.1.2.9.1 Traffic Use and Precipitation Factor

Traffic use is one of the greatest factors affecting the generation of sediment from road surfaces (Reid and Dunne 1984). Traffic can grind the road surface material into smaller particles, which can be transported more easily by rainfall runoff. While traffic rate determines the quantity of sediment available for transport, precipitation determines the transport capacity. Correction factors for traffic use by annual precipitation categories are presented in Table 4.14. Annual precipitation in the Grays River watershed is approximately 2,800 mm/year. The traffic-precipitation factors used in the analysis are those for annual precipitation of 1,200 to 3,000 mm as shown in Table 4.14 (WA-DNR 1997).

Table 4.14. Traffic and Precipitation Factors (F_{tp}) for Traffic Use Based on Annual Precipitation (WA-DNR 1997)

Traffic Use/ Road Category	Annual Precipitation and F_{tp} Factor		
	<1,200 mm	1,200-3,000 mm	>3,000 mm
Heavy traffic/active mainline	20	50	120
Moderate traffic/active secondary	2	4	10
Light traffic/non-active	1	1	1
No traffic/decommissioned	0.02	0.05	0.1

The ground cover, road tread, and traffic-precipitation factors specified for the road network were applied to the reference erosion rate to calculate the adjusted erosion rate for the road prism. The adjusted erosion rate of the tread component of the road prism is given by (WA-DNR 1997):

$$E_{adj} = (E_{ref})*(F_{gc})*(F_{rt})*(F_{tp}) \tag{4.3}$$

where E_{adj} = adjusted erosion rate for the road tread (tons/year)
E_{ref} = reference erosion rate (tons/acre)
F_{gc} = ground cover density factor
F_{rt} = road tread surface factor
Ft_p = traffic and precipitation factor.

The adjusted erosion rates for the cut-slope or ditch and the fill-slope components were calculated using Equation (4.3) without the *Frt* and *Ftp* factors. The adjusted erosion rates calculated from Equation (4.3) represents the average sediment production rate from all roads within the upper Grays River watershed. The adjusted erosion rate for each road type was then divided by the total road length to calculate an average sediment production rate per length of road prism (tons/meter/year).

4.1.2.10 Sediment Yield from Roads

Sediment generated from the road prism is delivered to the stream network by water flowing in roadside ditches, gullies, culverts, or on surfaces as shallow overland flow (Skaugset et al. 2002). Only a portion of the sediment produced from the road network reaches the stream network. The portion of sediment delivered to the stream system is highly dependent on drainage characteristics of the road prism. The two main characteristics of the road prism are the cross-sectional shape and the connection of ditches to stream channels. The cross-sectional shape of the road prism can be one of four types:

- in-sloped with ditch
- crowned with ditch
- out-sloped with ditch
- out-sloped with no ditch.

Road surveys from the October 2004 field investigation indicate approximately 39% of roads are in-sloped (both with and without a ditch), 38% are crowned with a ditch, and 24% are out-sloped with a ditch. Results indicate approximately half of the road surfaces drain into the slope and concentrate water along a ditch or swale at the toe of the cut-slope. Sediment yield to the channel network depends on the relative degree of connectivity of road ditches and culverts to the stream network. Roadside ditches that drain directly to a stream can deliver 100% of their sediment load to the channel network (WA-DNR 1997). Approximately 10% of sediment discharge from the road prism to within 200 ft (60 m) of a stream channel is delivered to the channel network (WA-DNR 1997). Sediment yield to the channel network was calculated using the following delivery ratios (WA-DNR 1997):

- 1.0 if road drains directly to a stream channel via ditch or gully
- 0.1 if road drains onto hillslope within 200 ft of a stream
- 0.0 if road drains onto hillslope more than 200 ft from a stream.

A GIS comparison of road and stream coverage-layers indicates, on average, 80% of roads in the upper Grays River basin are located within 200 ft (60 m) of a stream channel. Field observations indicate drainage from roughly half of the roads within 200 ft (60 m) of a stream drain to the stream channel. Based on these observations, 50% the roads within 200 ft (60 m) of a stream were assigned a delivery ratio of 1.0 and the other half were assigned a delivery ratio of 0.1. Therefore, the combined delivery ratio for roads within 200 ft (60 m) of a stream channel is 0.55.

The average sediment yield (tons/year) from roads was then calculated for each upper Grays River subbasin using the lengths of road-types for each subbasin. The average sediment yield from roads for each subbasin is given by (WA-DNR 1997):

$$S_r = 0.55\left(\frac{L_{R200}}{L_R}\right)\left(E_{adj\text{-}m}L_m + E_{adj\text{-}m}L_m + E_{adj\text{-}m}L_m + E_{adj\text{-}m}\right) \tag{4.4}$$

where S_r = average sediment yield from all roads in each subbasin (tons/year)
L_{R200} = length of roads within 200 ft (60 m) of the channel network (meters)
L_R = total length of roads in the subbasin (meters)

$E_{adj\text{-}m}$ = adjusted sediment production rate per length of road prism for mainline (*m*), secondary (*s*), non-active (*n*), and decommissioned (*d*) roads (tons/meter/year)

L_m = road length for each road type (meters).

A summary of data used in the analysis of surface erosion from roads, as well as results of the analysis, is presented in Table 4.15.

Table 4.15. Summary of Surface Erosion Factors and Rates for Roads (Herrera 2005)

Road Type	2003 Road Length (m)	Road Width (m)	Baseline Erosion Rate (tons/year)	Ground Cover Density Factor (F_{gc})	Road Tread Factor (F_{rt})	Traffic ppt Factor (F_{tp})	Adjusted Erosion Rate (tons/year)	Normalized Sediment Production Rate (tons/krn/year)
Mainline	**42,011**	**8.1**					**10,334**	**246**
Tread			1,006	1.00	0.2	50	10,062	
Cutslope			1,006	0.18			181	
Hillslope			503	0.18			91	
Active Secondary	**717,073**	**6.1**					**12,495**	**17.4**
Tread			12,962	0.63	0.2	4	6,533	
Cutslope			12,962	0.37			4,796	
Hillslope			6,481	0.18			1,167	
Non-Active	**126,542**	**5.4**					**1,413**	**11.2**
Tread			2,036	0.37	0.2	1	151	
Cutslope			2,036	0.53			1,079	
Hillslope			1,018	0.18			183	
Decommissioned	**15,737**	**5.4**					**69**	**4.37**
Tread			253	0.18	0.2	0.05	0	
Cutslope			253	0.18			46	
Hillslope			127	0.18			23	
Basin Totals	**901,364**		**40,010**				**24,311**	**27.0**

Results of this analysis, presented in Table 4.16, indicate that an average of 27.0 tons/year is generated from each kilometer of the road network within the upper Grays River watershed. Mainline and active secondary roads together generate 86% of the total sediment from road surfaces but comprise only 0.5% and 80% of the road network length, respectively.

Table 4.16. Summary of Surface Erosion Factors and Rates for Roads Within the Upper Grays River Watershed (Herrera 2005)

Road Type	2003 Road Length (m)	Adjusted Erosion Rate (tons/year)	Normalized Sediment Production Rate (tons/km/year)
Mainline	42,011	10,334	246
Active secondary	717,073	12,495	17.4
Inactive	126,542	1,413	11.2
Decommissioned	15,737	69	0.4
Watershed totals	901,364	24,311	27.0

4.2 Upper Grays River Watershed Sediment Budget

4.2.1 Background Sediment Yield

Background sediment yield is the long-term average sediment yield from a basin under natural geomorphic conditions prior to anthropogenic disturbance. Knowledge of the background sediment yield is important for understanding the increase in sediment yield caused by anthropogenic disturbance. Under natural (background) conditions, the primary geomorphic processes that generate sediment in mountain watersheds are soil creep and mass wasting (Dietrich and Dunne 1978). The background sediment yield for the upper Grays River watershed was evaluated using two approaches:

1. contemporary field studies and sediment budgets computed for undisturbed basins
2. long-term tectonic uplift rates under steady-state conditions.

4.2.1.1 Sediment Budget Approach

Soil creep is the slow downslope movement of the soil mantle under the influence of gravity. Traditionally, sediment production by tree wind-throw and animal burrowing are also included in soil creep (WA-DNR 1997). Soil creep introduces sediment along the entire length of a channel at an average rate of approximately 1.5 mm/year in undisturbed watersheds in western Washington (WA-DNR 1997). A summary of previous studies of soil creep in undisturbed watersheds is compiled in Table 4.17.

Table 4.17. Soil Creep Rates and Watershed Characteristics from Previous Studies (Herrera 2005)

Reference	Precipitation (mm/year)	Gradient (%)	Creep Rate (mm/year)
Madej (1982)	1,300–1,700	0.2–5	1.5
Everett (1963)	990	25–65	0.0–2.2
Young (1960)	1,400	35–55	1.0–2.0
Kirkby (1967)	1,710	15–35	1.0–3.0
Lewis (1976)	3,050–4,500	35–32	1.8–2.3
Eyles and Ho (1970)	2,540	18	3.2

The factors required to calculate sediment yield from soil creep are length of the stream channel, average soil depth, and creep rate. Stream length (L) was estimated from GIS applications and is doubled to account for creep along both stream banks. Soil depth (D) can be generalized over the subbasin using soil maps and field reconnaissance. An average soil depth of 1.3 m was computed from eight field measurements of the soil profile at road cuts that were not located near colluvial hollows. The average annual sediment yield from soil creep for the upper Grays River watershed is given by

$$S_c = \frac{2\rho LDC}{A_d} \tag{4.5}$$

where S_c = average annual sediment yield from soil creep (tons/square kilometer/year)
ρ = bulk density of soil (1.7 tons/m^3)
L = total length of the channel network (902,248 m)
D = average soil depth (1.3 m)

C = creep rate (0.0015 m/year)
A_d = drainage area of the upper Grays River watershed (230 km^2).

Sediment yield from soil creep was calculated for each subbasin within the watershed using the length of stream channels in each subbasin and then summed for the entire watershed. Based on field measurements of soil depth, length of the channel network, and the assumed creep rate, sediment yield from soil creep in the upper Grays River watershed calculated from Equation (4.5) is 26 tons/km^2/year.

Mass-wasting processes such as landslides, debris flows, and channel bank erosion are the second major source of sediment quantified for the assessment of background sediment yield (WA-DNR 1997). Sediment yield calculated from previous studies of mass wasting in undisturbed watersheds is compiled in Table 4.18.

Table 4.18. Sediment Yield from Mass Wasting in Undisturbed Watersheds from Previous Studies (Herrera 2005)

Reference	Precipitation (mm/year)	Location	Sediment Yield from Mass Wasting (tons/km^2/year)
Dietrich and Dunne (1978)	3,400	Oregon Coast Range	17.2
Madej (1982)	1,300–1,700	Kitsap Peninsula, WA	15
Roberts and Church (1986)[(a)]	1,500–4,220	Queen Charlotte Islands, BC	15.3–122
Grant and Wolff (1991), total sediment yield	2,300	Western Cascade Range, OR	18–170[(b)]

(a) Various sources cited in Roberts and Church (1986). Cubic meters converted to tons using a density of 1.7 tons/m^3.
(b) Includes total sediment yield from the watershed measured as suspended load and bedload in two watersheds with 9 and 30 years of sediment yield data, respectively.

The average background sediment yield from mass wasting in undisturbed watersheds, as reported from research, is approximately 30 tons/km^2/year. The combination of sediment yield from soil creep and mass wasting results in a total background sediment yield of approximately 56 tons/km^2/year. This rate is comparable to the total sediment yield measured by Grant and Wolff (1991) for undisturbed watersheds in the western Cascade Range of Oregon.

4.2.1.2 Tectonic Uplift Approach

The average background sediment yield can also be estimated from long-term tectonic uplift rates for the region. Tectonic uplift of the Olympic Mountains and the Coast Range has occurred over the past 15 million years (Brandon et al. 1998; Kelsey et al. 1996). These mountain ranges are considered to be in a condition of topographic steady state (Willett and Brandon 2002), whereby uplift equals erosion, and feedbacks between these processes moderate a steady topographic form at the regional scale. Although uplift rates for southern Washington within the region of the Grays River study area have not been measured, geochemical data indicate a long-term uplift rate in the southern Olympic Mountains of 0.0001 m/year (Brandon et al. 1998). The minimum uplift rate for the northern Oregon Coast Range is also 0.0001 m/year (Kelsey et al. 1996). Based on these studies, a long-term rock uplift rate of 0.0001 mm/year was used for the analysis.

The long-term background erosion rate was calculated from the uplift rate by

$$S_U = U * \rho * (10^6 \text{ m}^2/\text{km}^2) \tag{4.6}$$

where S_U = average, long-term sediment yield (tons/square kilometer/year)
U = long-term rock uplift rate (0.0001 m/year)
ρ = bedrock density (2.5 tons/m^3)
10^6 = the conversion from square meters to square kilometers.

The steady-state assumption of Equation (4.6) requires that all of the sediment eroded from hillslopes is delivered to the channel network and is transported out of the watershed. Therefore the background sediment yield is equivalent to the long-term erosion rate. The long-term sediment yield calculated from Equation (4.3) must be modified to account for the fraction of sediment that leaves the watershed through solution weathering. Based on the assumption that approximately half of the sediment mass leaves the watershed in solution as dissolved load (Selby 1993), the average background sediment yield calculated from long-term uplift rates (S_U) is 125 tons/km^2/year.

Although slightly higher, the background sediment yield calculated from tectonic uplift is comparable with results from the contemporary measurements of sediment yield from undisturbed watersheds described above in the sediment budget approach. The difference in computed sediment yield is within the relative errors for the two approaches, which arise from the variability in the assumed creep rate used in the sediment budget approach and the assumed uplift rate used in the tectonic approach.

4.2.2 Sediment Budget Computation

Contemporary and historical sediment budgets were constructed for the Grays River watershed to compute both temporal and spatial trends in sediment yield within the watershed. Sediment yield was calculated for each of 60 subbasins delineated within the watershed. To quantify sediment yield to specific alluvial segments of the channel network, the 60 subbasins were aggregated into seven main subbasin groups:

- upper mainstem Grays River
- middle mainstem Grays River
- lower mainstem Grays River
- East Fork
- South Fork
- West Fork
- Fossil Creek.

4.2.2.1 Contemporary Sediment Budget

A contemporary sediment budget was constructed for the upper Grays River watershed using results from the detailed assessment of sediment yield for 2003. Sediment yield from mass wasting, roads, and soil creep were combined to compute the annual sediment yield for each aggraded subbasin group in the watershed (Appendix G). Results of the sediment budget indicate that the contemporary sediment yield in the upper Grays River watershed is about an order of magnitude greater than the long-term background

sediment yield expected for the region (Table 4.19). In addition, several local, short-term sediment-yields, calculated solely from mass wasting, were more than two orders of magnitude greater than estimated background rates.

Table 4.19. Average Sediment Yield in the Upper Grays River Watershed Under Current (2003) Conditions and Long-Term Background Conditions (Herrera 2005)

Sediment Source	Sediment Yield (tons/km^2/year)
2003 Mass wasting	1,032
2003 Roads	41
2003 Soil creep	26
2003 Total	1,099
Long-term background (range)	56–125

4.2.2.2 Historical Sediment Budget

Historical sediment budgets were constructed for each subbasin group by synthesizing historical timber-harvest records for the upper Grays River watershed with earlier correlations established between the time since harvest and sediment yield from mass wasting and roads. Sediment yield from soil creep, which is assumed to be independent of timber-harvest practices, was also included in the historical sediment budget. The results of the detailed 2003 analysis of sediment yield provided the basis for reconstruction of the historical sediment budget.

Historical sediment budgets were constructed for each of the eight reference years in which timber-harvest data are available (Figures 4.20 through 4.26). Sediment yield from mass wasting was calculated for each subbasin in GIS using an overlay of historical harvest data and the mass-wasting map units. Sediment yield from roads was calculated using the 2003 sediment yield normalized by the fraction of the subbasin previously harvested, by each reference year. Estimates of sediment yield from roads are conservative because this method assumes that roads generate sediment at a constant rate from the time of road construction. Sediment yield from mass wasting, roads, and soil creep were then combined to compute the annual sediment yield for each subbasin group.

The reconstruction of the historical sediment yield for each subbasin group is presented graphically in Figure 4.27a. Total watershed sediment yield by source is shown in Figure 4.27b. Detailed results of the historical sediment budget are included in Appendix G.

Reconstruction of the sediment budget indicates that the total sediment yield for the upper Grays River watershed was nearly constant from 1942 through 1964 and ranged from approximately 13,000 to 18,000 tons/year. The estimated sediment yield increased sharply to 87,000 tons/year by 1976 and continued to climb at an unprecedented rate to more than 250,000 tons/year by 2003 (Figure 4.27a). Although the total sediment yield for the watershed has increased every year, local sediment yield from the Fossil Creek, middle mainstem, and upper mainstem subbasins declined between 1996 and 2003 (Figure 4.27a).

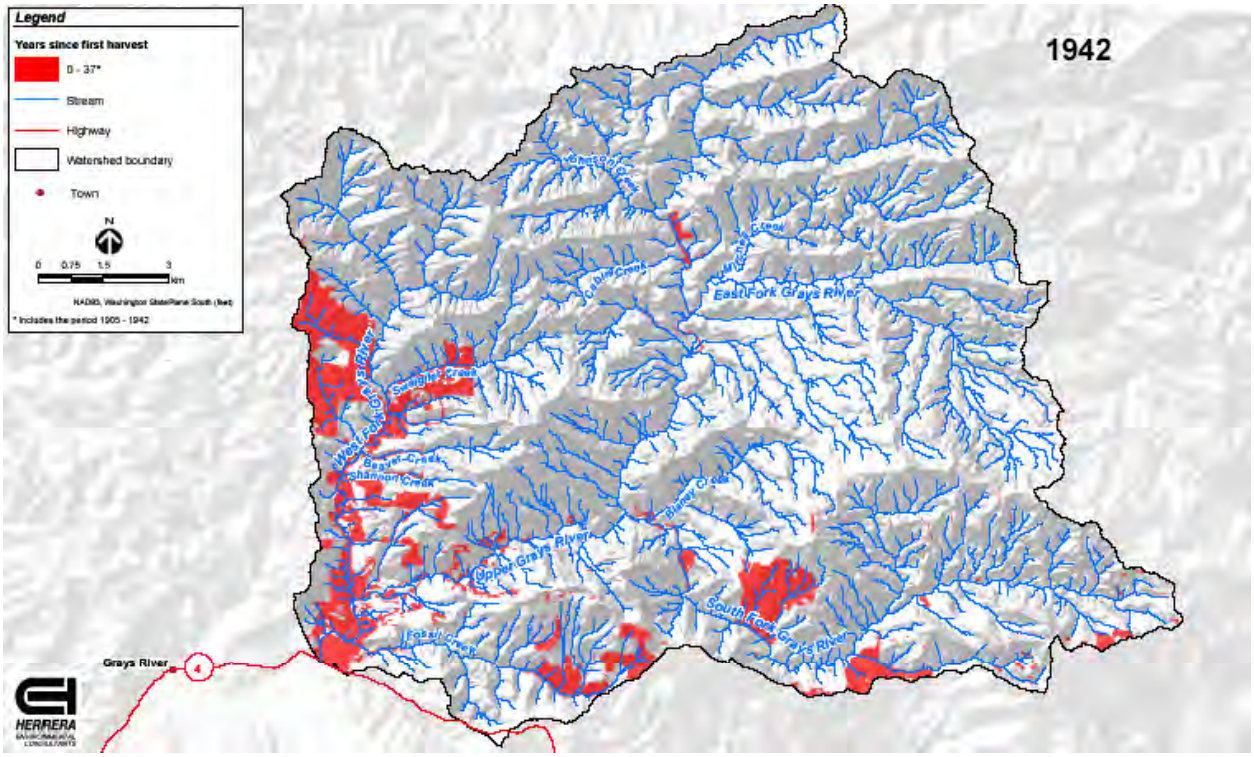


Figure 4.20. Upper Grays River Watershed Forest-Age Distribution in 1942

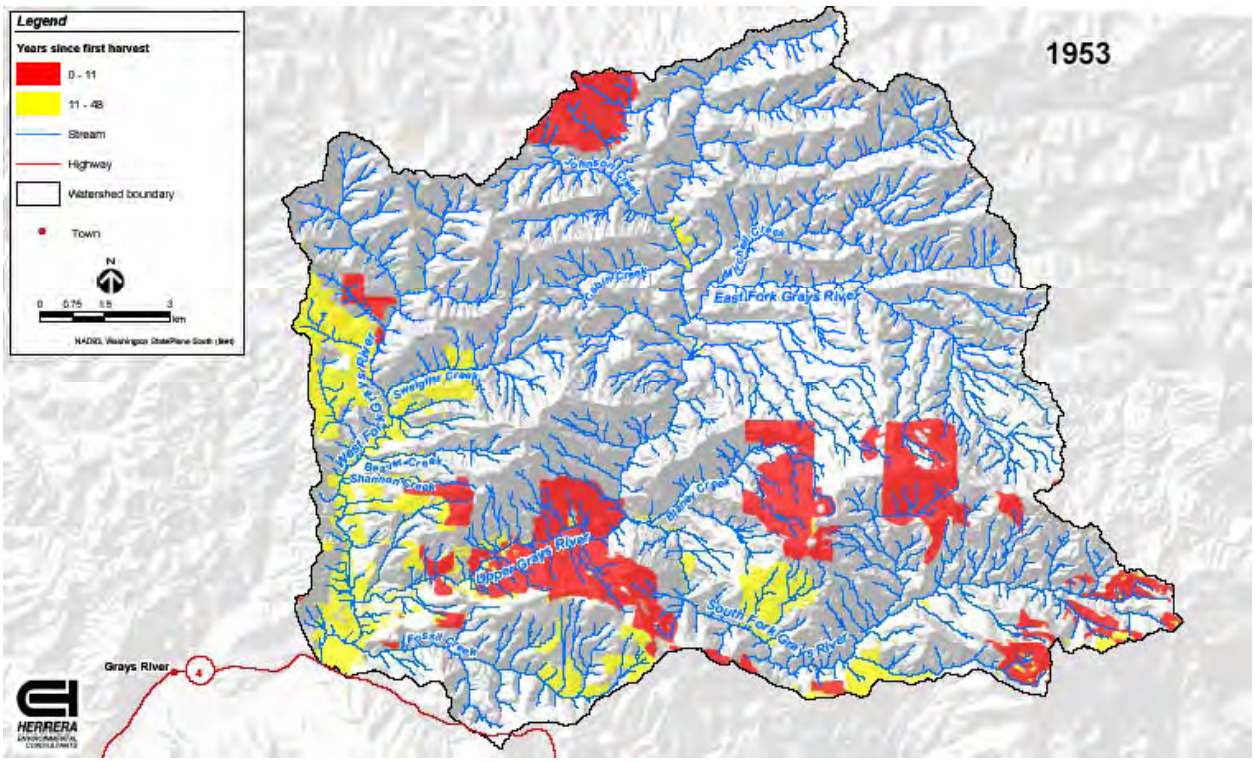


Figure 4.21. Upper Grays River Watershed Forest-Age Distribution in 1953

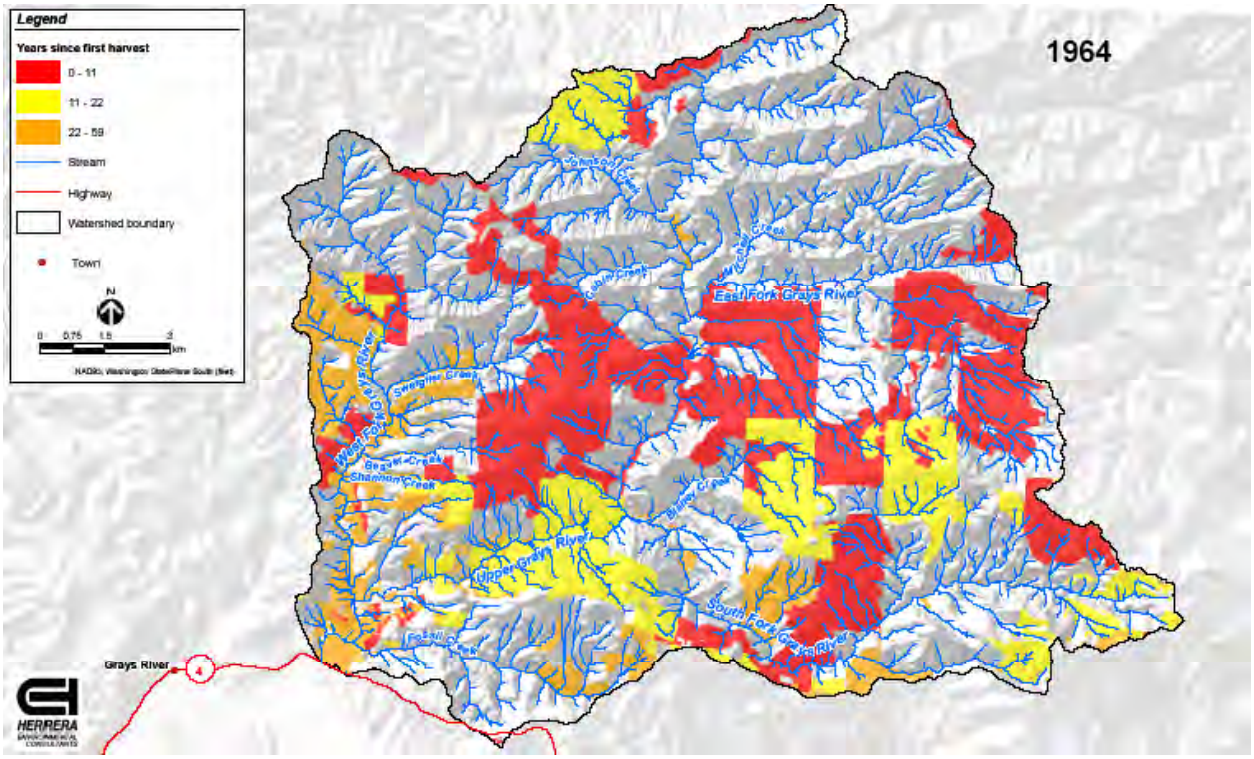


Figure 4.22. Upper Grays River Watershed Forest-Age Distribution in 1964

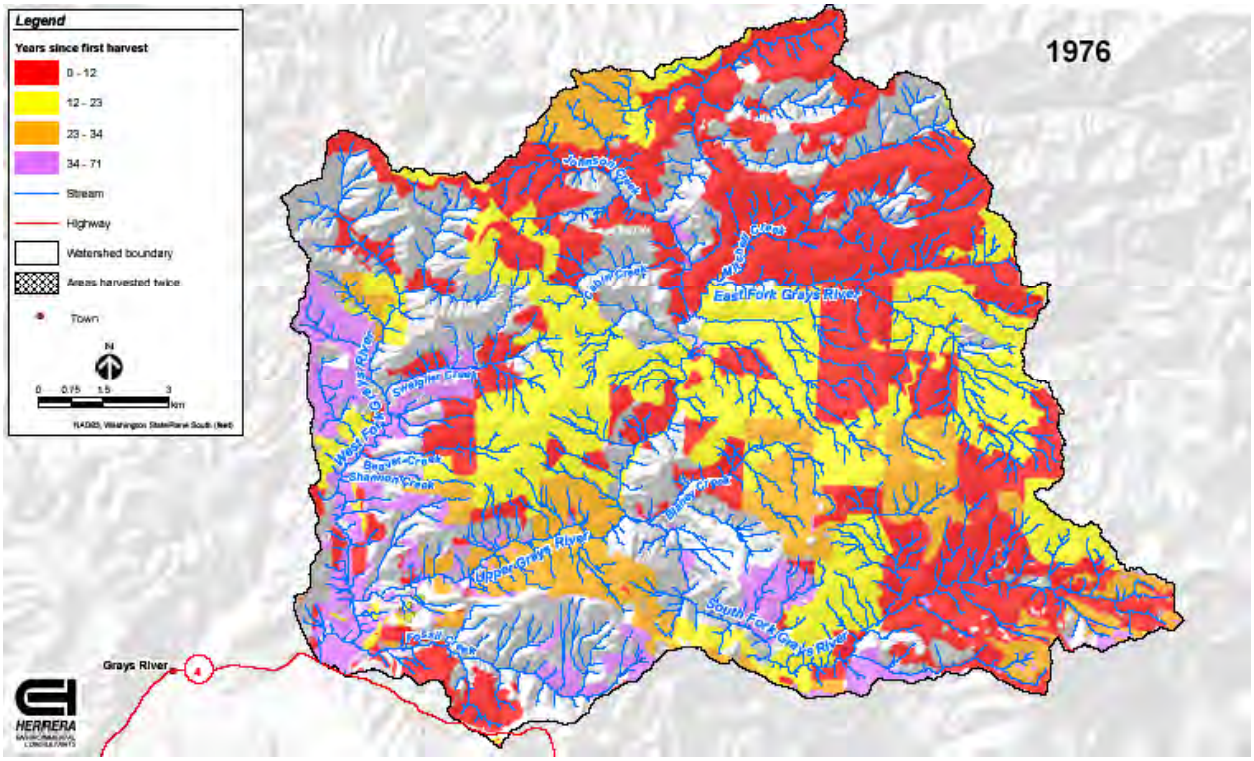


Figure 4.23. Upper Grays River Watershed Forest-Age Distribution in 1976

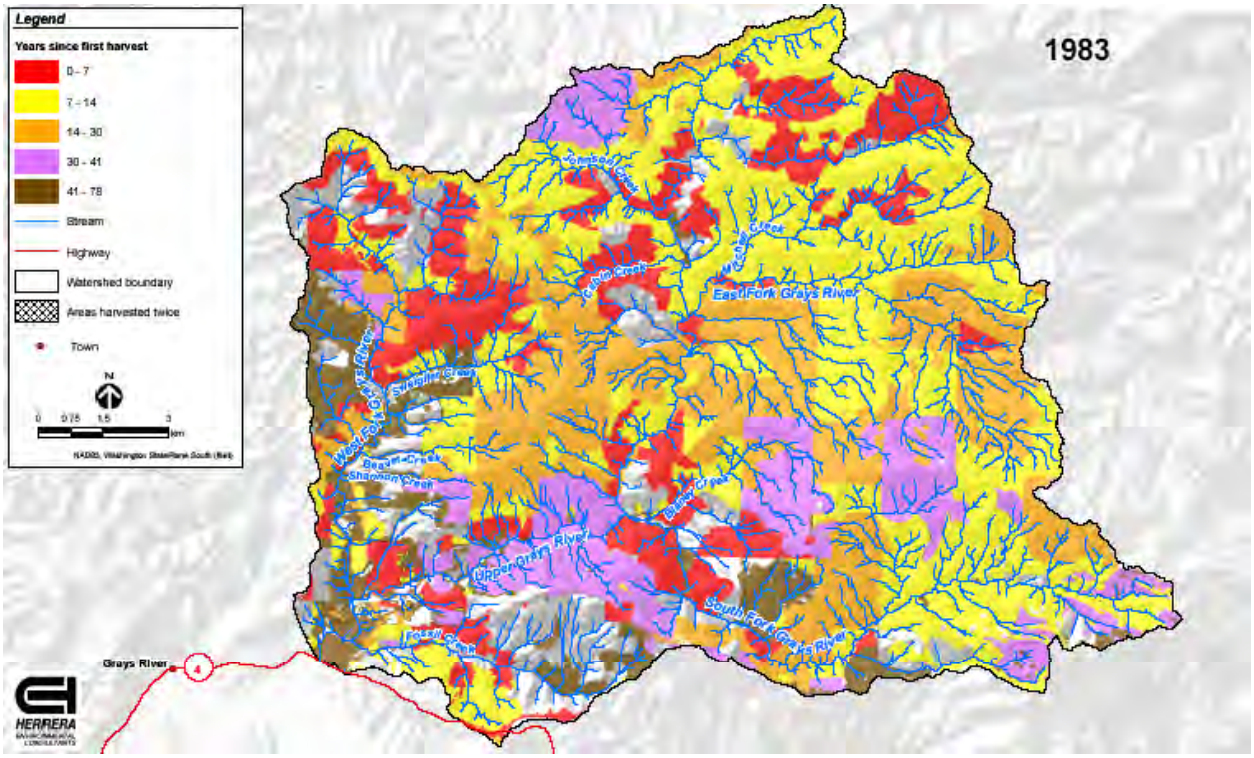


Figure 4.24. Upper Grays River Watershed Forest-Age Distribution in 1983

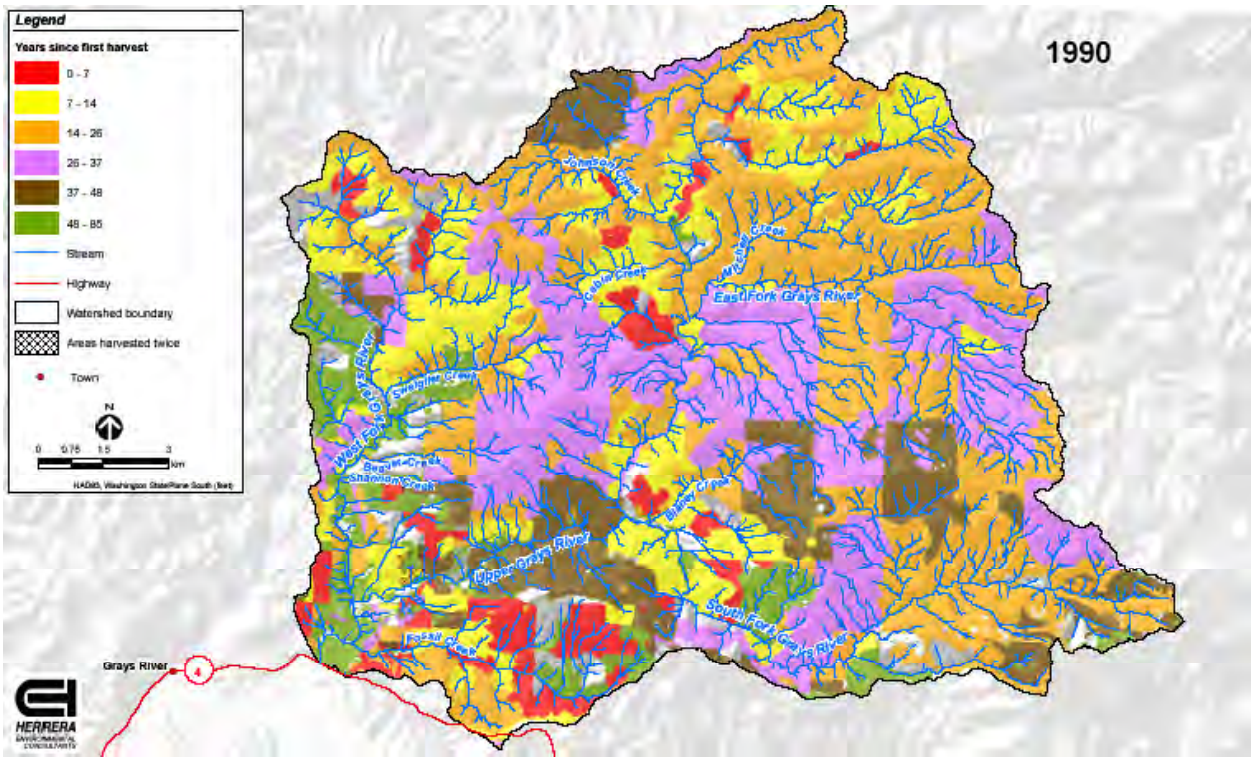


Figure 4.25. Upper Grays River Watershed Forest-Age Distribution in 1990

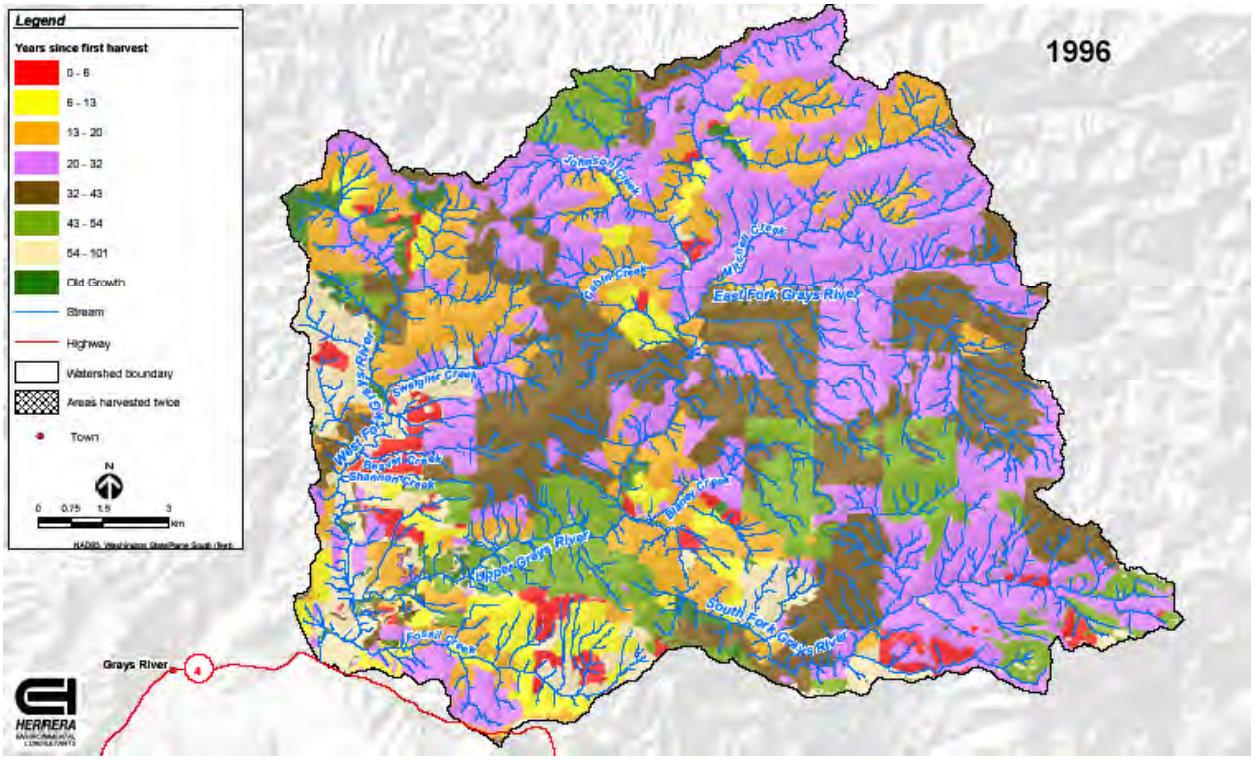


Figure 4.26. Upper Grays River Watershed Forest-Age Distribution in 1996

Results indicate that most of the sediment delivered to the channel network is derived from mass-wasting processes (Figure 4.27b). Roads and soil creep contribute a relatively minor proportion of the total sediment yield; however, earlier analyses identified roads as a likely contributor to landslides for several decades following initial timber-harvest activities. Hence, the jump in sediment yield between 1964 and 1976 may be linked to increased road construction.

The grain-size distribution of sediment delivered to the channel network varies among the different sediment sources. The analysis of sediment samples collected from landslide deposits and observations during the field reconnaissance indicate that landslides are characterized by a poorly sorted grain-size distribution that includes both fines and coarse gravel. Deep-seated landslides, which incorporate bedrock, produce coarser grain-size distributions. Sediment delivered to the channel network by soil creep is expected to have the same grain-size distribution as shallow landslides, which consist of colluvium. In contrast, the grain-size distribution of sediment originating from road surfaces should be dominated by fines. However, surface runoff and bank erosion associated with the road prism can also deliver some coarse sediment to the channel network.

Although landslide-derived sediment is coarser than sediment derived from road surfaces, landslides delivered 20 to 200 times more sediment than roads during the 20 to 30 years following timber harvest (Herrera 2005). Therefore, the volume of fine sediment delivered by mass-wasting processes may exceed by many times the quantity of fine sediment generated from road surfaces during this same period.

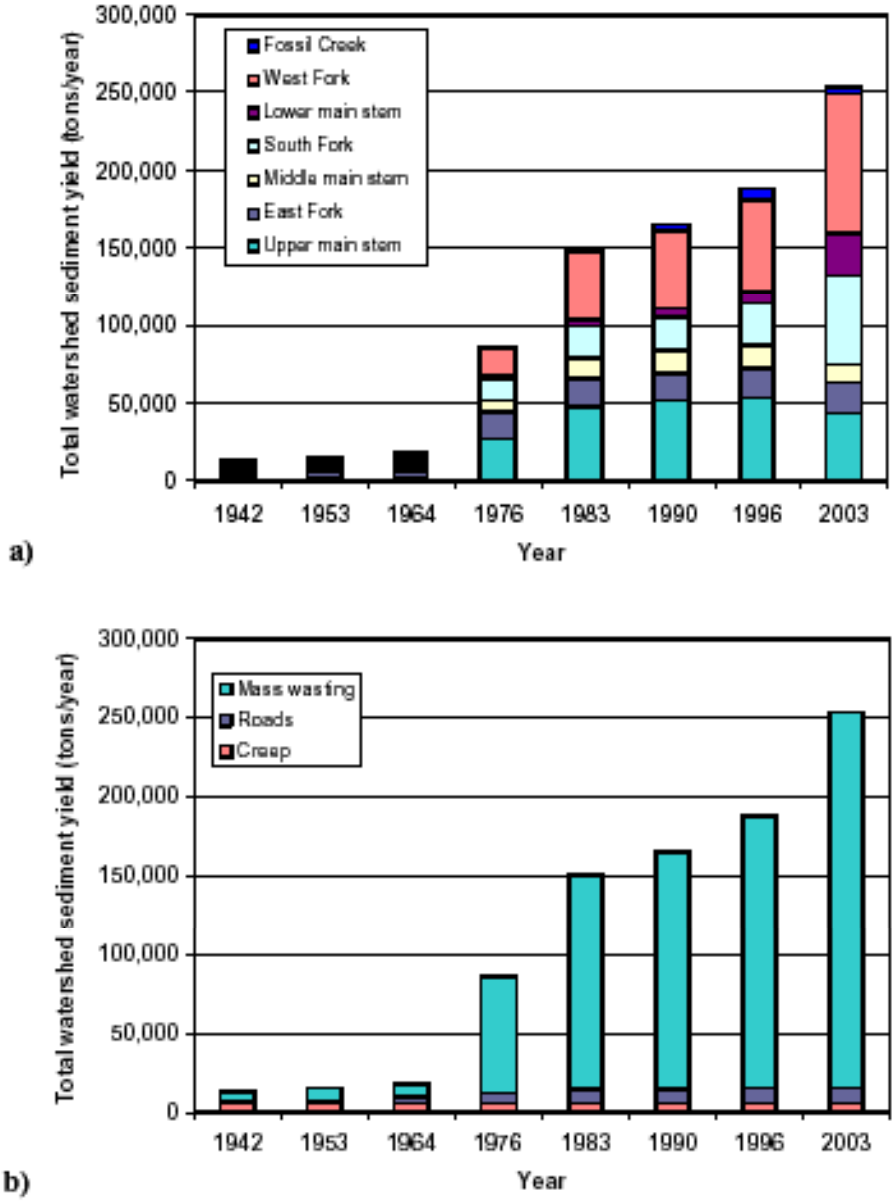


Figure 4.27. Predicted Temporal Variation in Total Upper Grays River Watershed Sediment Yield by (a) Subbasin Group and (b) Source

Most of the poorly sorted sediment supplied by mass wasting is broken down to suspended load by particle attrition during fluvial transport. Based on a sediment budget constructed for a small basin underlain by basalt in the Oregon Coast Range, Dietrich and Dunne (1978) determined that about half of the sediment delivered to the channel network by creep and mass wasting was coarser than 2 mm. Of this fraction, about 80% broke down to suspended load (less than 2 mm) during transport to the mouth of the basin. Tumbling mill experiments conducted by Collins and Dunne (1989) with basalt gravel obtained from rivers draining the Crescent Formation yielded a mass reduction of about 2% per kilometer of transport for clasts greater than 0.5 mm. In a tumbling mill experiment performed by Perkins (1989) on

unsorted sediment derived from landslide deposits, particle attrition accounted for a 58% reduction in the mass of bedload-size particles (greater than 0.5 mm) and a 50% increase in the fraction of suspended-load particles finer than 0.5 mm after only 1.6 km of simulated transport. In a similar experiment, Sutherland et al. (2002) reported a reduction in the mass of bedload particles that was 2 to 10 times greater for fresh landslide material than for ambient river gravel.

Based on these prior studies of bedload attrition, most of the coarse sediment delivered to the upper Grays River channel network by mass-wasting processes is likely converted to suspended load after moving relatively short distances (1 to 2 km) through the channel network, and certainly by the time sediment reaches the mainstem (Gorley) response reach.

The contemporary sediment yield calculated for 2003 in the upper Grays River watershed is consistent with the findings of Grant and Wolff (1991), which indicate that the average annual production of sediment after clear-cutting was about 4 to 12 times greater than the undisturbed rate and remained higher than pre-harvest rates more than 20 years later. Based on their computed trends, Grant and Wolff (1991) forecast sediment production to remain elevated above background rates for 30 years after harvest. Montgomery et al. (2000) found sediment yield from landslides in the Oregon Coast Range at 3 to 9 times the natural background rate for the region.

Given the current timber-harvest rotation in the upper Grays River watershed of about 30 to 40 years, sediment production will likely remain relatively high and ultimately will also likely result in significant long-term upland soil loss and possible reduced timber productivity.

4.3 Geomorphic Characterization of the Channel Network

4.3.1 Background

In rivers and larger streams, the *floodplain* is an integral part of the stream-riparian ecosystem. The floodplain is generally a relatively flat, low-lying area adjacent to the main channel of a river or stream. Floods are part of the natural hydrologic regime and result in over-bank flows that temporarily inundate the low-lying areas adjacent to the stream or river. Flooding can also result in streambank erosion and channel migration (Richards et al. 1993). The flood-induced processes of erosion and subsequent deposition are essential to the creation and maintenance of many riparian areas, especially those in the lower reaches of large streams and rivers (Scott et al. 1996; Gurnell 1997; Poole 2002; Naiman et al. 2005).

Flooding generally occurs in so-called *flood hazard zones* where flooding events can threaten private property or human life. These flood hazard zones are usually regulated to protect people and property. However, in some cases structural measures are used to facilitate human activities within the floodplain or flood hazard zone. Dikes and levees are the most common of these measures. In most cases, these measures are only partially effective in preventing flooding, and the best method of protection is to avoid human activity within the floodplain (Poff et al. 1997; Naiman and Bilby 1998; Bolton and Shellberg 2001; Ward et al. 2002). Minimizing floodplain modification and development within the flood hazard zone is generally the best management strategy to both ensure human safety and support ecological function. Based on this understanding, the floodplain should be considered a part of the stream-riparian corridor.

Typically, the riparian zone is considered to include the entire extent of the floodplain, an integral part of the riparian corridor in low-gradient streams and rivers. A *floodplain* is defined as the low gradient area adjacent to the stream or river channel that becomes inundated with over-bank flows during larger storm events. This area begins at the edge of the bankfull channel and typically receives over-bank flows during larger storm events. The frequency of inundation depends on the size of the stormflow event and the topography of the floodplain. The so-called 100-year flood is often used as the defining flood-flow for purposes of delineating the nominal extent of the floodplain (Dunne and Leopold 1977). According to Bayley (1995), the floodplain is that part of the river floodplain ecosystem that is regularly flooded and dried, and it represents a type of wetland. Well-developed, complex floodplains are characteristic in large river systems where there are long periods of seasonal flooding, oxbow lakes, wetlands, a diverse forest community and moist soils (Gregory et al. 1991; Naiman et al. 1992; Poff et al. 1997; Naiman and Bilby 1998). Flood events of different size and frequency play a vital role in maintaining a diversity of riparian plant species and aquatic habitat (Junk et al. 1989; Poff et al. 1997; Swanson et al. 1998). Biological productivity is often enhanced in floodplains because sediment and nutrients are deposited during the advance and retreat of floodwaters (Bayley 1995). Small floods transport fine sediments and organic matter downstream and laterally, and help create and maintain the stream-riparian ecosystem. Intermediate and large floods create opportunities for larger organic material input and allow for the nourishment and establishment of pioneer plant species (Gregory et al. 1991; Naiman et al. 1992; Poff et al. 1997; Naiman and Bilby 1998; Naiman et al. 2000).

Because of the dynamic equilibrium of stream systems, in addition to a floodplain area, many streams may also have a characteristic *channel migration zone* (CMZ). Although floodplain area does not always delineate the CMZ (Rapp et al. 2003), they often do overlap and can be useful in defining the limits of channel migration for most streams. Channel migration is the process of a river or stream channel moving laterally across or within its floodplain. During this process, the stream erodes one bank while depositing sediment along the other, resulting in a net lateral movement of the channel. These processes typically occur gradually, often resulting in significant channel movement over a long period of time. Ongoing lateral channel migration typically results in the development of meander bends, which themselves often migrate downstream over time. Channel migration can also occur abruptly in a process called *avulsion*, which is most common in large rivers (Dunne and Leopold 1977).

Over time, streams move back and forth across the valley floor in a process called lateral migration. The CMZ is the lateral extent of likely channel movement over the past 100-year period or where aquatic or wetland habitat could possibly exist at some time in the future (Pollock and Kennard 1998; WA DNR 1999; Naiman et al. 2000; Bolton and Shellberg 2001; Rapp et al. 2003). The CMZ usually includes the lower terraces and the toe of hill slopes adjacent to the floodplain where the stream is likely to meander (Pollock and Kennard 1998). The CMZ delineates the area over which the channel may occupy based on its dynamic equilibrium characteristics. The CMZ can be identified using historical records, aerial photos, or field verification of paleochannels or channel features such as braided channels, overflow channels, progressive meandering, meander cutoffs, oxbows, side channels, or channel avulsions (Dunne and Leopold 1977). Consideration should also be given to the natural or historical levels of LWD found in the stream channel. Streams with abundant LWD and/or beaver activity, such as natural Pacific

Northwest lowland streams, often have a more active CMZ (Pollock and Kennard 1998). Often, the 100-year floodplain delineation is used to delineate both the floodplain area and CMZ for a river or stream (Bolton and Shellberg 2001). Due to natural topography and geomorphology, some streams are constrained to narrow valleys or ravines. These streams typically have a smaller (or nonexistent) floodplain and a constrained CMZ. Streams within narrow valleys or ravines may not move laterally and thus do not have a CMZ. In these streams the edge of the bankfull channel typically corresponds to the start of the active floodplain. The floodplain itself may also be relatively narrow, depending on valley form and topographic constraints. Bankfull flow is the streamflow (approximated by the flow of the 2-year flood recurrence interval) that has the capability of moving a significant amount of sediment or bedload (Dunne and Leopold 1977).

The *hyporheic zone* is another critical component of the stream-riparian ecosystem. The hyporheic zone is the saturated substrata beneath a stream or river channel and under the riparian zone where groundwater and surface water mix. Properties of both groundwater and surface water are blended in the hyporheic zone, significantly changing the water's chemical composition and often stimulating unique biological activity (Naiman et al. 2000). The complex mix of cobble, gravel, and sediment provide a wide range of microhabitats that vary in nutrient and oxygen content. A myriad of specialized insects and microorganisms take advantage of this habitat, some never emerging for their entire life cycle. Important biological activities (such as denitrification or the removal of excess nitrogen) take place in the hyporheic zone, mediated by specialist biota. In addition, the hyporheic zone plays an important role in aquatic food webs by storing particulate organic matter, for processing by benthic invertebrates and other instream biota.

Diverse stream and floodplain characteristics, as well as native plant communities exist within riparian areas due to the interaction of the natural hydrogeomorphic processes such as flooding events and inputs of organic and inorganic material from upland areas and hillslopes (Gregory et al. 1991; Naiman et al. 1992; Scott et al. 1996; Gurnell 1997; Naiman et al. 2000; Rot et al. 2000). In turn, the species composition and stand age (e.g., maturity) of vegetation growing within the riparian corridor and floodplain areas of a river can be a major influencing factor on channel morphology and hydrogeomorphic processes (Hawk and Zobel 1974; Harwood and Brown 1993; Montgomery and Bolton 2003).

In the Pacific Northwest, the interaction of stream channel hydrogeomorphic processes (e.g., flooding and channel migration) and riparian vegetation results in a river valley that is typically characterized as being a dynamic mosaic of landscape patches (Figure 4.28). These patches are natural transient features of the river valley produced by the combined actions of the river and vegetation successional processes. These processes include plant growth, organic-matter (OM) processing, and LWD recruitment, as well as sediment production and storage (Harwood and Brown 1993; Knighton and Nanson 1993; Fetherston et al. 1995; Ward et al. 2002; Latterell et al. 2006; Van Pelt et al. 2006).

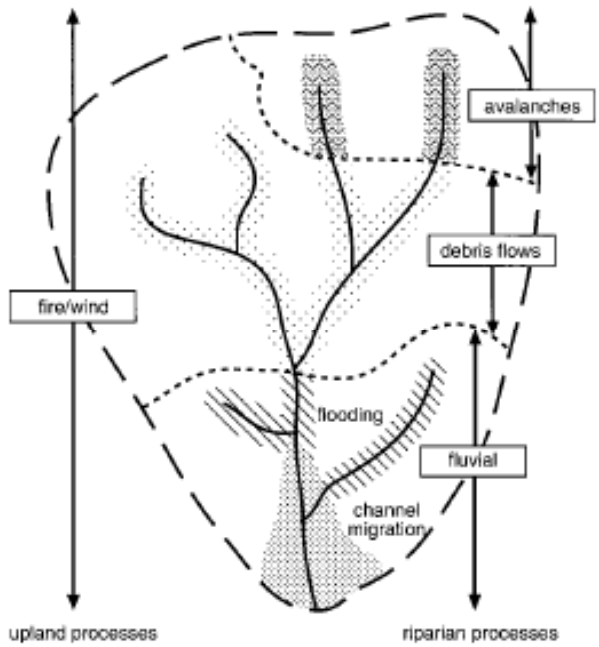


Figure 4.28. Typical Processes Found Within the Stream-Riparian Ecosystem of the Pacific Northwest (Naiman et al. 2000)

4.3.2 Grays River Geomorphic Channel Analysis

Predictions of alluvial channel response to disturbance can be made by considering general relationships between discharge (Q), transport capacity (Q_c), sediment supply rate (Q_s), and various channel metrics. Building on historical research in fluvial geomorphology (Mackin 1948; Leopold and Maddock 1953; Lane 1955; Leopold et al. 1964; Shumm 1971), Montgomery and Buffington (1998) developed a comprehensive conceptual relationship that expresses the ratio of transport capacity to sediment supply (Q_c/Q_s) in terms of nine response variables:

$$\frac{Q_c}{Q_s} \propto \frac{DnD_{50}\,p}{WD_s S_s S\lambda} \tag{4.7}$$

where D = channel depth
n = Manning's roughness coefficient
D_{50} = the median bed-surface grain size
p = sinuosity (the ratio of channel to valley length)
W = channel width
D_s = scour depth
S_s = sediment storage
S = channel slope
λ = meander wavelength.

Equation (4.7) implies that the changes in the ratio of transport capacity to sediment supply will evoke changes in response variables. For example, an increase in sediment supply relative to transport capacity would result in a net reduction in the numerator product or a net increase in the denominator

product. With a net increase in sediment supply, numerator variables (depth, grain size, roughness, and sinuosity) would diminish, while denominator variables (width, scour depth, slope, and sediment storage) would tend to rise. Table 4.20 illustrates the application of Equation (4.7) to qualitatively predict trends in channel changes caused by a declining ratio of transport capacity to sediment supply ($\Delta(Q_c/Q_s) < 0$) in a typical river system (Herrera 2005).

Table 4.20. Predicted Channel Response to a Declining Ratio of Transport Capacity to Sediment Supply (Montgomery and Buffington 1997)

Forcing Function	Response Variable								
	Width W	Depth D	Scour Depth D_s	Grain Size D_{50}	Roughness n	Slope S	Sediment Storage S_s	Meander Wavelength λ	Sinuosity p
$\Delta(Q_c/Q_s) < 0$	+	–	+	–	–	+	+	+	–
Signs indicate relative change in response variables (plus sign = increase, minus sign = decrease).									

Land-use practices that alter transport capacity and sediment supply can initiate channel responses through changes in the response variables. Response segments generally have alluvial channels and floodplains but can include bedrock channel segments that periodically store alluvium (in which case there is typically some evidence of periodic sediment storage such as alluvial floodplains or terraces). The larger the size and number of channel response segments, the greater the moderating effect on sediment flux downstream through a basin.

Debris jams and LWD deposited on the floodplain and in the river also contribute structure and function to the river ecosystem (Figure 4.29). LWD jams are frequently instrumental in creating the braided, multichannel morphology common to many rivers. LWD jams also promote and regulate channel avulsions and floodplain sloughs. Sediment storage also continues to be a functional attribute of debris jams in the lower reaches of rivers, including estuarine areas. Very large, key pieces of LWD are especially important in larger rivers. These key pieces of LWD trap smaller debris and form jams, eventually creating forested islands within the river channel complex. It is generally accepted that these reforested floodplains can develop naturally recruited LWD jams within 50 to 100 years (Collins and Montgomery 2002). LWD and debris jams are also ecologically important in rivers that are characterized by a single, meandering channel. In these systems, debris jams provide habitat complexity, create and maintain off-channel habitat, and provide streambank protection (Collins et al. 2002). LWD not only can act as a significant grade-control element limiting incision but also can be very effective at trapping bed material and storing large volumes of sediment (Abbe and Montgomery 2003). Response reaches, however, can also be converted to transport reaches when wood debris and riparian vegetation are removed or discharge increases (Herrera 2005).

Wood is a natural and ecologically important feature of streams and rivers that flow through forested watersheds such as those of the Pacific Northwest coastal ecoregion (Naiman 1992; Naiman and Bilby 1998; Naiman et al. 2000; Gregory et al. 2003). Large woody debris can be found throughout the riverine landscape. Recruitment of LWD and other organic material results from hillslope mass-wasting events (e.g., landslides and debris flows) and instream processes including streambank erosion, flooding events, avulsions, and channel migration, as well as from windthrow and blowdown events (Reeves et al. 2003). Woody debris provides physical roughness elements within the channel and is largely responsible for the formation of instream physical habitat features such as pools, which are vital to several salmonid life

stages. Instream LWD also traps organic material such as leaf litter and salmon carcasses, which form the base of the food web in most Pacific Northwest stream systems. In short, LWD is a critical to the ecological structure and function of streams and rivers in the Pacific Northwest. Changes in LWD recruitment, transport, and residence of instream woody debris can significantly affect the ecological condition of lotic ecosystems (Bilby and Ward 1989 & 1991; Maser and Sedell 1994; Gippel 1995; Naiman et al. 2000; Gurnell et al. 2002).

In addition to habitat formation, woody debris deposited in streams by debris flows or other hillslope processes typically forms jams or dams within the upper watershed channel network. These natural features can create persistent, long-term instream sediment storage nodes within the river channel-floodplain system. Debris jams and dams also provide significant instream habitat complexity. In addition, debris jams and LWD dams tend to inhibit downstream propagation of sediment pulses. Furthermore, instream LWD and debris jams typically result in the creation of steps in the channel profile and terraces on the floodplain or valley floor. These features can persist in place even if the LWD is lost due to washout or after long-term decay. The role of LWD and debris jams also changes over time as the river channel evolves and the active channel locations shift (Lancaster et al. 2001; May and Gresswell 2003; Montgomery and Bolton 2003; Montgomery et al. 2003).

Due to the increase in flow energy or stream power, the size of LWD that can be considered stable tends to increase as one moves downstream in a river system (Bilby and Ward 1989). The configurations in which LWD is typically organized in river systems also changes as one moves from upstream to downstream segments of a river. In headwater channels and tributary streams, even relatively small pieces of LWD typically remain within the channel near where they fell because of the lack of sufficient streamflow to move them. Single pieces of LWD tend to be more common features of smaller channels, where they provide ecological function (Montgomery et al. 2003).

In larger channels downstream, single pieces of LWD tend to be more mobile and woody debris tends to accumulate in logjams that are made up of multiple pieces of LWD. The influence of very large, key pieces of LWD on stabilizing other debris in logjams increases as the river channel widens downstream. Stable, key pieces of LWD normally are necessary for the formation of logjams and anchor these logjams in position (Abbe and Montgomery 1996; Collins et al. 2002; Montgomery et al. 2003).

Channel islands are also common riverine features that are created by LWD deposited within the active channel. Instream LWD and debris jams also play a significant role in the creation of side channels and other habitat features located on channel margins. Finally, debris jams are one of the primary causes of channel avulsions and the anastomosing pattern found in many river systems (Harwood and Brown 1993; Knighton and Nanson 1993; Montgomery 1999; Abbe and Montgomery 1996; Collins et al. 2002; Montgomery et al. 2003; Montgomery and Abbe 2006).

In large part, upstream processes control the hydrology and sediment loading of downstream reaches of a river system. Instream LWD and debris jams naturally control sediment output to downstream reaches. In natural systems, sediment output tends to be relatively steady and generally shows little evidence of large, episodic inputs (Leopold et al. 1964). Abundant instream LWD plays a key role in

moderating sediment flux from upstream subbasins when natural hillslope mass wasting and debris-flow events do occur. On the other hand, landslides and debris flows that occur in areas with a lack of abundant LWD, such as is the case in watersheds impacted by long-term timber-harvest activities, often result in more frequent and larger episodic sediment flux to downstream channels and habitat features (Abbe and Montgomery 1996).

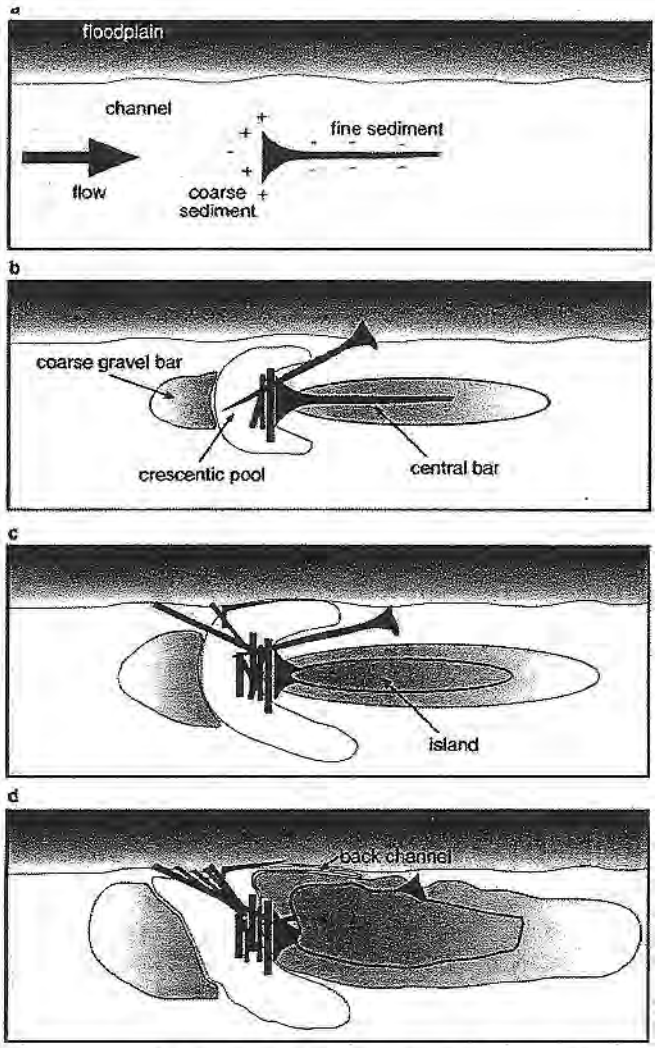
The existing body of research indicates that the most effective way to restore the natural abundance and diversity of LWD to streams and rivers is to reestablish the riparian forest structure that can provide wood to the system through natural recruitment processes. In some cases, however, interim LWD inputs and man-made solutions may be necessary until the forest structure is restored to a self-sustaining level (a process that can take decades). This is especially critical where LWD is a limiting factor inhibiting salmonid recovery (Beechie et al. 1996; Beechie et al. 2000; Collins and Montgomery 2002; Abbe et al. 2003; Montgomery and Bolton 2003; Beechie et al. 2003; Gregory et al. 2003).

A reduction in the number and effectiveness of response reaches would result in greater sediment discharge to the lower mainstem of the Grays River. Potential responses to land use practices are summarized in Table 4.21.

4.3.2.1 Delineation of Channel Network and Geomorphic Channel Units

The process-based classification of Montgomery and Buffington (1997) was applied to the upper Grays River watershed to predict the distribution of geomorphic channel types and divide the channel network into geomorphic channel units that would be useful for evaluating the impacts from changes in land use. The Montgomery and Buffington (1997) classification of channel-reach morphology was developed for mountain channels. The classification synthesizes stream morphologies into five fluvial channel types along opposing continua of increasing sediment supply and decreasing sediment transport capacity. The classification ranges from cascade and step-pool channels, which are supply-limited (the stream's energy to move sediment exceeds the delivery of sediment to the channel), to plane-bed, pool-riffle, and dune-ripple (sand-bedded) channels, which are transport-limited (sediment supply exceeds the river's capacity for transport). The classification also recognizes colluvial channels, which are transport-limited reaches (also termed *supply reaches*) located high in the channel network above cascade channels. Hence, there is a peak in transport capacity between the tips of the channel network and the low-gradient response reaches (Brummer and Montgomery 2003).

A number of researchers have identified the topographic signature of the transition from channels dominated by debris flow to fluvial dominated channels as an inflection in the relationship between contributing drainage area and channel slope (Montgomery and Foufoula-Georgiou 1993; Montgomery and Buffington 1998; Stock and Dietrich 2003). An analysis of the relationship between these channel parameters was used to define the threshold between colluvial and fluvial channels in the upper Grays River channel network. The channel network of the upper Grays River watershed was defined with a minimum drainage area of 0.01 km^2 using a 10-m digital elevation model derived from U.S. Geological Survey 1:24,000-scale topographic maps. The log-bin-averaged drainage area-to-slope relationships are based on the channel network and define the transitional domains between debris flow and fluvial channels (Figure 4.30).



Morphological stages in alluvial topography associated with construction of a woody debris (bar-apex) jam. (a) Deposition of an especially large tree with the root wad intact. (b) Formation of a coarse gravel bar upstream, a crescent-shaped pool immediately upstream of the root wad, and a downstream central bar of finer sediments along the axis of the tree. (c) Island development along the central bar. (d) Integration into the broader floodplain. Modified from Abbe and Montgomery (1996).

Figure 4.29. Typical Morphological Stages in Alluvial Landform Associated with Large Woody Debris and Logjams (Naiman et al. 2000)

Table 4.21. Summary of Channel Response to Land-Use Practices (Herrera 2005)

Disturbance	Change	Potential Channel Response
Upland forest clearing	Increase in sediment supply to stream	Aggradation/sedimentation
		Channel widening
	Increase in discharge to stream	Channel incision and widening
Riparian forest clearing and agricultural conversion	Destabilization of banks	Channel widening
	Increase of local sediment supply	Accelerated channel migration
	Reduction in functional wood debris recruitment	Channel sedimentation, increased sediment storage
		Decrease in channel complexity, reduction in roughness
		Increase in turbidity
Road construction and maintenance	Increase in fine sediment production	Infilling of coarse bed sediment with fines, reduction in bed-surface grain size, and roughness
		Increase in turbidity
	Increase in drainage density	Channel incision and widening
Channel clearing	Increase in stream gradient	Down-cutting, incision, and head-cuts
		Channel simplification, reduction in roughness
		Bank destabilization, increase in sediment supply
		Initial channel narrowing followed by channel widening

Debris flows are clearly dominant in channels with contributing drainage areas less than 0.05 km^2, and a clear fluvial domain exists where drainage areas exceed approximately 0.2 km^2. The transitional domain between colluvial and fluvial channels indicates where channels dominated by debris flow transition toward cascade and supply-limited fluvial channels. Additionally, a fluvial response domain is apparent as concavity increases where drainage area exceeds approximately 20 km^2, which corresponds to slopes less than 0.02. The upper Grays River (Gorley) response reach falls within this domain. Within the fluvial response domain, confined reaches of the mainstem Grays River follow a slope consistent with the fluvial transport domain, representative of their high transport capacity relative to sediment supply.

Based on the inflection in the relationship between contributing drainage area and channel slope, the threshold between colluvial and fluvial channels in the upper Grays River channel network was assumed to occur at 0.1 km^2. Where drainage areas contributing to the channel network exceed 0.1 km^2, the range in channel slope reported by Montgomery and Buffington (1997) was used to predict the reach morphology throughout the Grays River channel network (Table 4.26). The spatial distribution of channel types predicted by slope is illustrated in Figure 4.31.

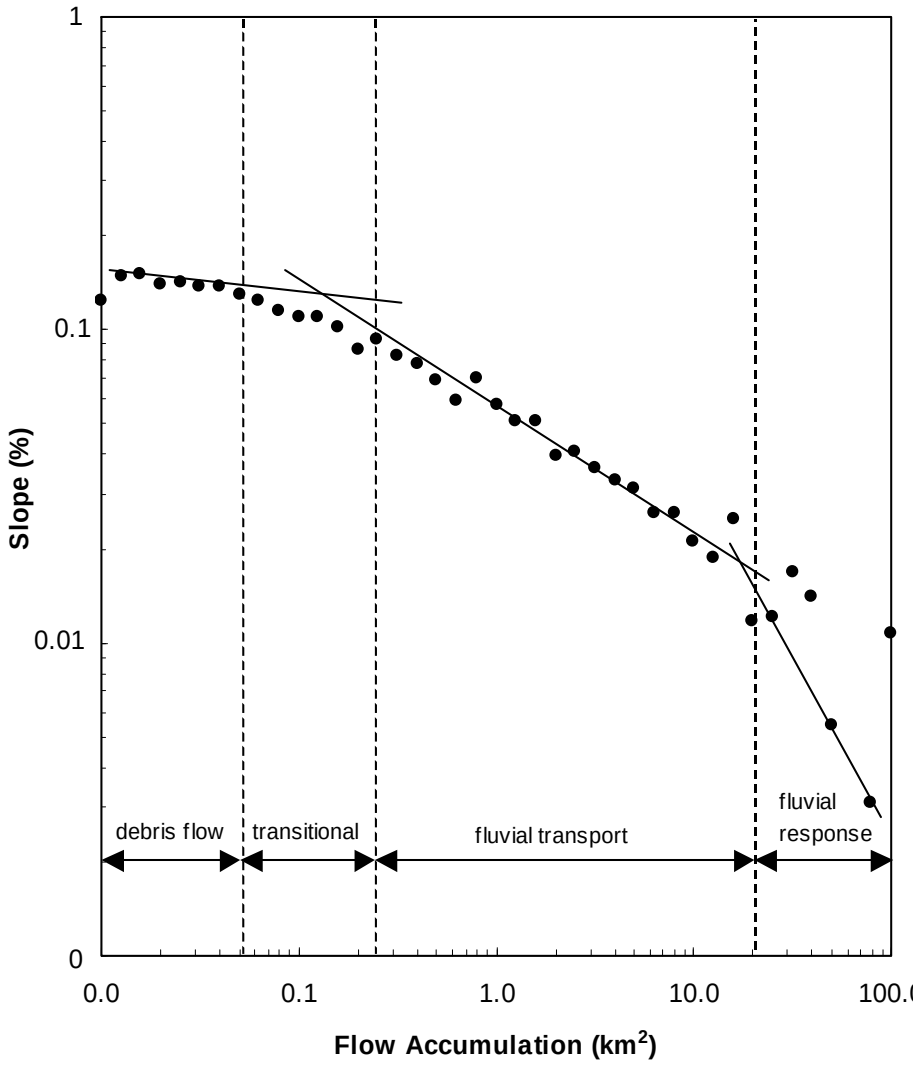


Figure 4.30. Log-Bin-Averaged Local Slope Derived from 10-Meter Grid Size Digital Elevation Model for the Upper Grays River Watershed (Herrera 2005). Channel divisions are based on Stock and Dietrich (2003).

Table 4.22. Slope Range and Contributing Drainage Area Used to Predict Grays River Channel Network Reach Morphology (Herrera 2005)

Reach Type	Slope	Contributing Drainage Area (km^2)
Pool-riffle	<0.02	>0.1
Plane bed	0.02–0.04	>0.1
Step pool	0.04–0.08	>0.1
Cascade	>0.8	>0.1
Colluvial	–	>0.01 and <0.1
Source: Adapted from Montgomery and Buffington (1997).		

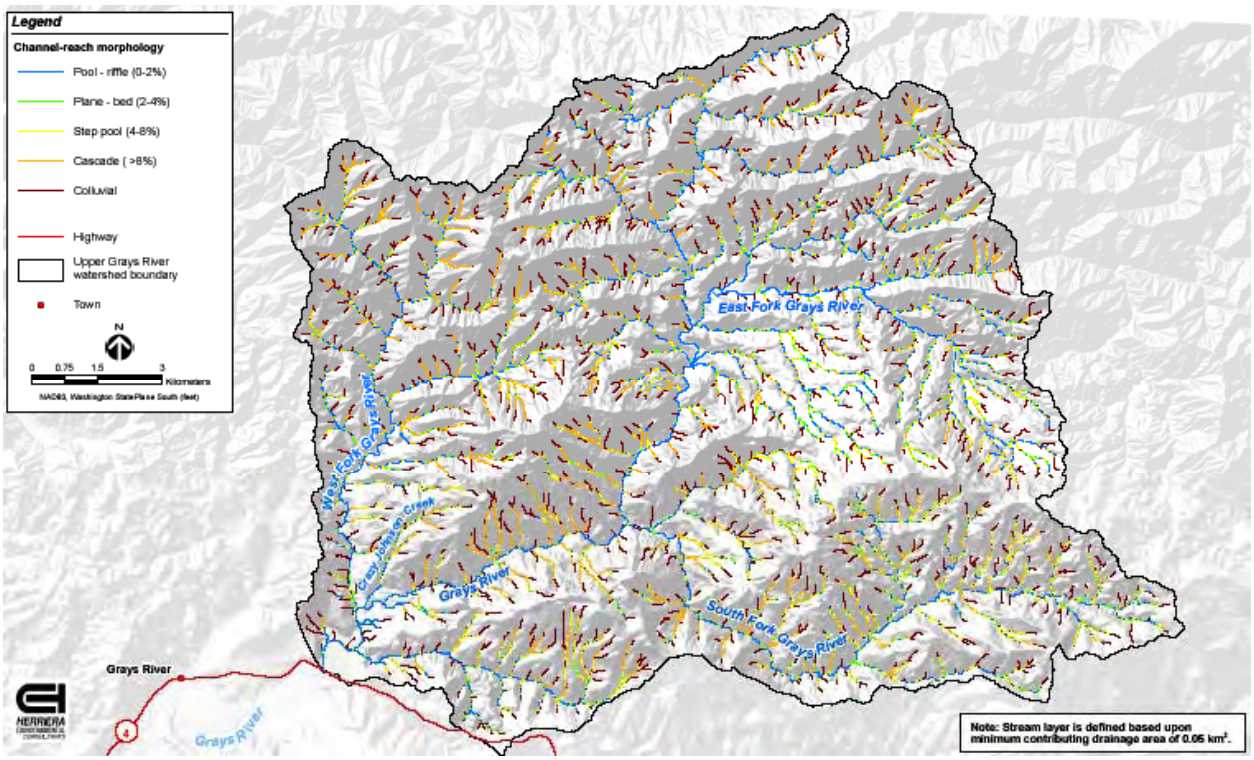


Figure 4.31. Predicted Spatial Distribution of Channel-Reach Morphology Based on Channel Slope (Montgomery and Buffington 1997) in the Upper Grays River Watershed, Washington (Herrera 2005)

Although sediment supply throughout mountain channel networks may be difficult to quantify, transport capacity can be estimated from empirical relations and digital elevation models. Spatial variations in the relative magnitude of sediment transport capacity were evaluated throughout the upper Grays River channel network to distinguish transport and response reaches. Transport capacity varies with the rate of energy expenditure per unit area of the channel bed, which is defined in terms of unit stream power:

$$\omega = \frac{\gamma QS}{W} \tag{4.8}$$

where γ = the unit weight of water
Q = discharge
S = slope
W = channel width.

Bankfull discharge may be assumed to vary with drainage area A:

$$Q \propto A^d \tag{4.9}$$

where the exponent d has been determined empirically to range from 0.7 for semi-arid regions to 1.0 for humid landscapes such as the Grays River watershed (Dunne and Leopold 1978; Leopold 1994; Rice

1998; Whiting et al. 1999). Regime theory and empirical data (Leopold and Maddock 1953; Ibbitt 1997; Knighton 1998) indicate that channel width varies as the square root of discharge (and thus can be expressed as a function of drainage area):

$$W \propto \sqrt{A} \tag{4.10}$$

Combining Equations (4.8) through (4.10) allows unit stream power to be recast in terms of parameters that are readily measured from digital elevation models:

$$\omega \propto S\sqrt{A} \tag{4.11}$$

Equation (4.11) was used to define the magnitude of transport capacity throughout the entire Grays River channel network in terms of an index of unit stream power. Results indicate that unit stream power (i.e., transport capacity) decreases downstream through the channel network, with the lowest values found in the low-gradient alluvial reaches of the mainstem Grays River and major tributaries (Figure 4.32). The pattern of unit stream power in the Grays River channel network watershed is consistent with other field studies, which have found unit stream power decreases from headwater channels to lower-gradient alluvial reaches (Magilligan 1992; Lecce 1997; Knighton 1999; Brummer and Montgomery 2003).

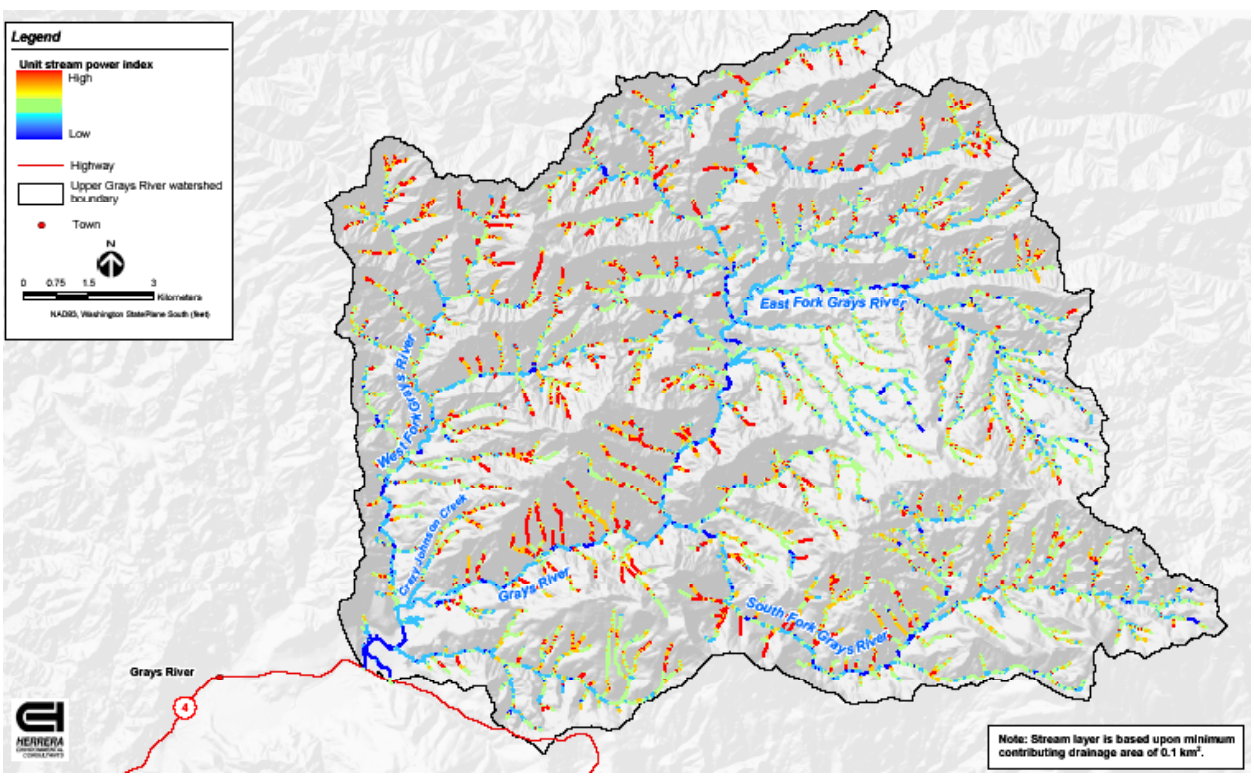


Figure 4.32. Relative Transport Capacity in Terms of Unit Stream Power in the Channel Network of the Upper Grays River Watershed, Washington (Herrera 2005)

Montgomery and Buffington (1997) demonstrated that the downstream sequence in which channel types are arranged affects the potential for a disturbance to affect a particular reach. Transport reaches are defined as morphologically resilient channels with high transport capacities relative to sediment supply (i.e., bedrock, cascade, and step-pool channels) that readily convey increases in sediment supply. In contrast, response segments are channels with a low ratio of transport capacity to sediment supply (i.e., plane-bed and pool-riffle channels) in which significant morphologic adjustment results when sediment supply is increased.

The spatial distribution of channel types predicted from slope and drainage area, along with patterns of unit stream power, were used to identify transport and response reaches in the upper Grays River watershed. In general, transport reaches are located high within the channel network, whereas response reaches are located in the larger alluvial valley segments.

To characterize the geomorphic conditions that have resulted from basin-wide changes in land use, the channel network in the upper Grays River watershed was divided into the following nine distinct geomorphic channel units based on the predicted distribution of channel types and the observed channel conditions:

- colluvial channels
- headwater transport channels
- tributary response channels
- East Fork Grays River
- South Fork Grays River
- West Fork Grays River
- lower Crazy Johnson Creek
- mainstem Grays River
- Grays River response reach.

Table 4.23 summarizes the data on channel units in the upper Grays River watershed. Figure 4.33 presents the locations of these geomorphic channel units within the upper Grays River watershed.

4.3.2.2 Current Channel Conditions

A field reconnaissance of the upper Grays River channel network was completed to assess how increases in sediment production have affected channel morphology and the extent to which channel segments may have changed. Because the entire channel network could not be observed, a survey was completed of selected reaches of each geomorphic channel unit. Field reconnaissance of selected survey reaches was conducted in March, September, and October 2004. Channel surveys extending a minimum of 20 bankfull widths in length were completed for each of the geomorphic channel units except the mainstem Grays River (Figure 4.33).

Surveys of the Grays River were focused on the Grays River response reach, and observations of the main stem Grays River channel unit were limited to locations where the mainstem was easily observable from forest roads and bridges.

Table 4.23. Length and Percentage of Each Geomorphic Channel Unit in the Upper Grays River Watershed (Herrera 2005)

Geomorphic Channel Unit	Length (km^2)	Percentage of Total Channel Network	Percentage of Response Channel Network
Colluvial channels	623.2	67.4	–
Headwater transport channels	157.0	17.0	–
Tributary response channels	109.4	11.8	77.9
East Fork Grays River	5.2	0.6	3.7
South Fork Grays River	7.8	0.8	5.6
West Fork Grays River	7.0	0.8	5.0
Lower Crazy Johnson Creek	0.3	0.0	0.2
Mainstem Grays River	11.2	1.2	4.8
Grays River response reach	4.0	0.3	2.8
Total channel network[(a)]	925.1	–	–
Response channel network[(b)]	140.4	15.7	–

(a) The channel network of the upper Grays River watershed was defined with a minimum drainage area of 0.01 km^2 using a 10-m digital elevation model derived from U.S. Geological Survey 1:24,000-scale topographic maps.

(b) The response channel network includes the tributary response channels, East Fork, South Fork, and West Fork Grays River, lower Crazy Johnson Creek, and 10.7 km of the mainstem Grays River geomorphic channel units.

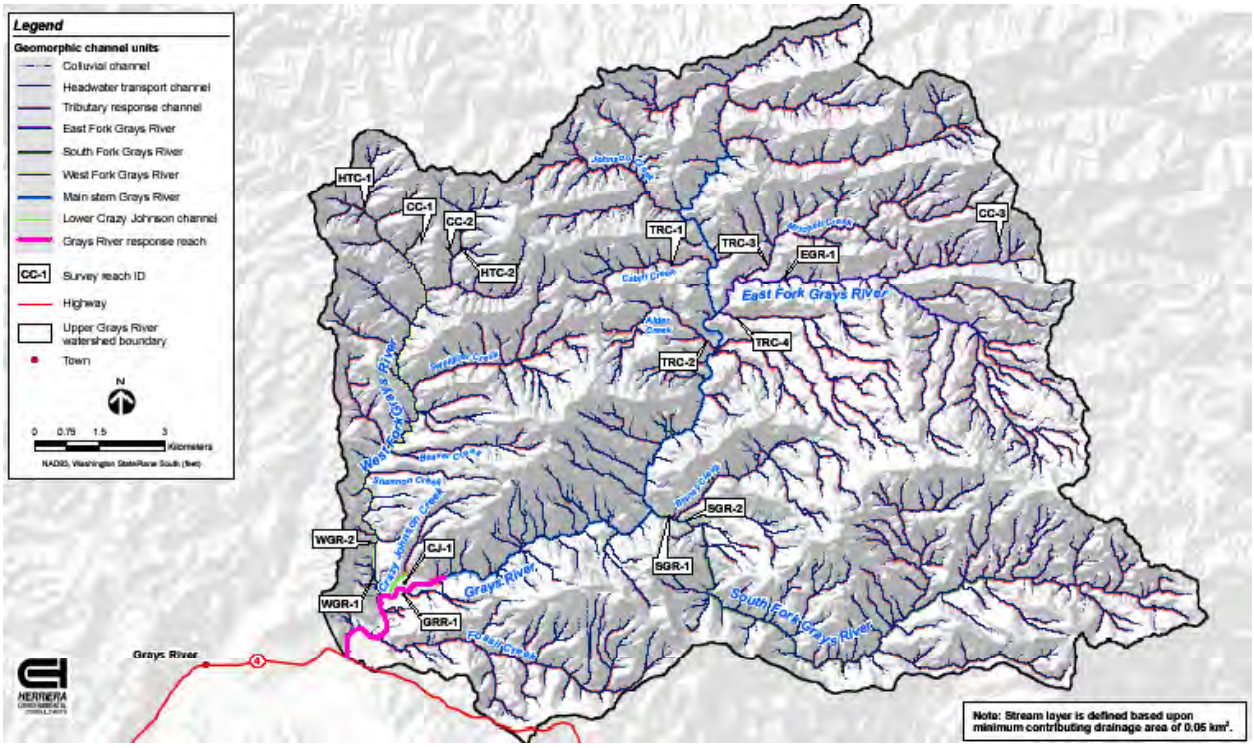


Figure 4.33. Geomorphic Channel Units and Locations of Survey Reaches Within the Upper Grays River Watershed, Washington (Herrera 2005)

To determine the extent of disturbance or degradation that has resulted from timber harvest within the channel network, the channel surveys documented evidence of sediment storage, routing, and channel response. Surveys included descriptions of the channel morphology, the conditions of instream sediment, and riparian vegetation. Additionally, the channel surveys included measurements of local channel slope, bankfull width and depth, and valley confinement. The location of each survey reach was georeferenced using a hand-held GPS instrument. Photographs of survey reach conditions are included in Appendix B, and summaries of each geomorphic channel unit survey reach are presented in Appendix C.

Wolman pebble counts of both the bed surface and subsurface were also conducted at select reconnaissance sites to characterize the grain-size distribution of streambed sediment (Wolman 1954). Reach average values of bed surface sediment were obtained by randomly selecting at least 100 surface grains (from both exposed bars and submerged locations) with the tip of an index finger. The medial axis of each grain was measured and recorded, and the grain was then discarded to avoid sampling the same particle more than once. If feasible, bed surface sampling occurred across the entire width of the channel. Surface pebble counts were located at the top of riffle channel features or at the upstream end of exposed gravel bars when subsurface counts were also made. Subsurface pebble counts were conducted at the upstream end of exposed gravel bars by first removing surface sediment to a depth of the largest grain size over a 1-m^2 area. The subsurface sediment grain size was measured using the same method employed for surface sampling. Pebble count results are included in Appendix C and Appendix D.

4.3.2.2.1 Colluvial Channels

Colluvial channels are steep headwater portions of the stream channel network that are largely dominated by hillslope processes such as soil creep and small-scale slope instability. Sediment transport in colluvial channels is limited by intermittent flow, large sediment grain sizes, woody debris, and riparian vegetation. Sediment delivery to the alluvial channel network occurs primarily through episodic debris flows (Swanson et al. 1982; Montgomery et al. 1996; Montgomery and Buffington 1998).

Colluvial channels in the upper Grays River watershed were delineated as headwater channels with a contributing drainage area less than 0.1 km^2 (Table 4.23). Colluvial channels in the upper Grays River watershed total approximately 623 km and make up about 67% of the channel network.

Evidence of mass wasting and debris flows were common in colluvial channels observed in the upper Grays River watershed during field reconnaissance (Herrera 2005). Many colluvial channels, such as survey reach CC-1, exhibited evidence of debris flows such as channels scoured to bedrock (Photo C-1-1 in Appendix B). In other colluvial channels (e.g., survey reach CC-2), upstream mass wasting has deposited significant sediment that is slowly being transported farther downstream (Photo CC-2-1 in Appendix B). Where evidence of debris flows was not observed (e.g., survey reach CC-3), LWD created steps in the channel profile and provided significant sediment storage. However, the recruitment potential of riparian forests adjacent to colluvial channels was typically poor due to historical timber-harvest activities that have left these areas lacking in mature coniferous vegetation. Because LWD increases channel roughness, the absence of future sources of LWD may increase the run-out distance of debris flows and sediment delivery to the channel network.

4.3.2.2.2 Headwater Transport Channels

Headwater transport channels include cascade and step-pool channels where transport capacity typically exceeds sediment supply. Headwater transport channels are frequently located between colluvial channels and response channels lower in the channel network and serve as a conduit to transport sediment derived through upland mass-wasting processes. These are typically high-gradient channels that in this analysis were delineated where channel gradient exceeds 4% and contributing drainage area exceeds 0.1 km^2 (Herrera 2005). Headwater transport channels in the upper Grays River watershed total approximately 157 km and make up 17% of the channel network (Table 4.23).

Headwater transport channels observed during the channel survey were generally confined step-pool channels (Herrera 2005). Channel steps were typically formed from boulders and occasionally from LWD (Photos HTC-1-1 and HTC-1-2 in Appendix B). The LWD recruitment potential from adjacent riparian forests was fair to poor due to historical timber-harvest activities that have left these areas lacking in mature coniferous vegetation (Photo HTC-2-1 in Appendix B). Although headwater transport channels are theoretically supply-limited, the widespread extent of bar deposition observed in survey reach HTC-1 indicates high rates of sediment loading that locally overwhelm transport capacity.

4.3.2.2.3 Tributary Response Channels

Tributary response channels include plane-bed and pool-riffle channels flowing to the mainstem or a fork of the Grays River with channel gradients less than 4% (Herrera 2005). These alluvial response channels are sensitive to changes in influences on channel morphology such as sediment load, discharge, and the character of riparian vegetation. Tributary response channels in the upper Grays River watershed total approximately 109 km and make up approximately 12% of the channel network and 78% of the alluvial response reaches (Table 4.23).

Tributary response channels are dominated by confined to moderately confined plane-bed channels (Photo TRC-1-1 in Appendix B). Where the channel was moderately confined, lateral instability and channel widening was commonly observed (Photo TRC-2-1 in Appendix B). Loose deposits of fine-grained sediment (silt to fine-grained gravel) composed of marine siltstone were observed in a number of the survey reaches and covered up to 40% of the streambed surfaces in depths up to 0.5 m locally (Photo TRC-1-1 and TRC 1-2 in Appendix B). Clasts composed of the marine siltstone are weak and easily broken down.

Again, due to historical timber-harvest activities, LWD loading was typically moderate to poor, and little functional wood was observed within the active channel of observed tributary response channels (Photo TRC-3-1 in Appendix B). Where functional LWD was observed, it was composed of relict old-growth LWD that entered the channel network prior to the harvest of the adjacent riparian forest. The general lack of functional LWD is significant because LWD plays a large role in determining the channel response to changes in sediment supply or transport capacity (Montgomery et al. 1996; Abbe 2000; Abbe and Montgomery 2003).

In all tributary response channels observed, the adjacent riparian community was dominated by immature deciduous vegetation with little or no recruitment potential for functional LWD (Herrera 2005). The poor recruitment potential typical of the adjacent riparian community indicates that the potential for

these response channels to function as capacitors that regulate the flux of sediment through the channel network will continue to diminish as relict LWD breaks down, further increasing sediment delivery to downstream reaches (Photo TRC-4-1 in Appendix B).

4.3.2.2.4 East Fork Grays River

The East Fork Grays River geomorphic channel unit includes the alluvial response portion of the river that extends 4.0 km east from the confluence of the East Fork and mainstem Grays River (Figure 4.33). The East Fork Grays River geomorphic channel unit is defined as the portion of the East Fork with a channel gradient less than 4% and includes plane-bed and pool-riffle reaches (Herrera 2005). The East Fork Grays River geomorphic channel unit makes up less than 1% of the total channel network in the upper Grays River watershed and only 3.7% of the alluvial response reaches (Table 4.23).

The surveyed reach of the East Fork Grays River is a moderately confined plane-bed channel exhibiting little complexity (Photos EGR-1-1 and EGR-1-2 in Appendix B). The channel substrate is predominantly gravel- and cobble-bedded with occasional boulders and lateral bars consisting of sand and fine gravel. Sand deposits were found on the active floodplain surface, indicating floodplain connectivity at high-flow stages and fine sediment storage within the active floodplain. A side channel observed at the left edge of the floodplain is not directly connected by surface flows to the main channel but contained flowing water, indicating hyporheic (subsurface) flow through floodplain sediments.

The role of LWD is limited in the survey reach. No functional LWD was observed in the 500-m survey reach, and the riparian forest consists primarily of deciduous forest with little future LWD recruitment potential. An increase in LWD loading would increase both floodplain connectivity and sediment storage potential within the channel, but this could be accomplished only by importing large functional LWD (Herrera 2005). Otherwise, up to several centuries of successional forest development will be required to reintroduce functional LWD to these reaches.

4.3.2.2.5 South Fork Grays River

The South Fork Grays River geomorphic channel unit includes 7.5 km of the South Fork extending east from its confluence with the mainstem Grays River (Figure 4.33). The South Fork Grays River geomorphic channel unit is defined as the portion of the South Fork with a channel gradient less than 4% and includes both plane-bed and pool-riffle reaches (Herrera 2005). The South Fork Grays River geomorphic channel unit makes up about 1% of the total channel network in the upper Grays River watershed and just over 5% of the alluvial response reaches (Table 4.23).

Field reconnaissance of the South Fork Grays River identified two distinct channel conditions. Survey reach SGR-2 was observed to be a complex, multiple-threaded pool-riffle channel network (Photo SGR-2-2 in Appendix B). The surveyed reach is moderately confined and dominated by forested islands, mid-channel bars, buried debris jams, and high levels of LWD loading. Complex valley jams such as these typically create steps in the channel profile, producing steep transverse hydraulic gradients between channels across the floodplain (Abbe and Montgomery 2003). In the South Fork, debris-jam steps are up to 1.5 m in height and extend almost the full valley width in some locations, up to 50 m (Photo SGR-2-1 in Appendix B). Much of the wood within these jams, however, is relict and in a partial or significant state of decay. Recruitment potential in this reach is relatively limited due to immature (deciduous-dominated) riparian forests within the valley bottom and along the margins of the channel migration zone.

As the relict LWD providing complexity and structure to the channel continues to break down and is not replaced when lost from the system, sediment currently stored by this LWD will be conveyed downstream and transport capacity within the reach will increase.

In contrast to the complexity observed in survey reach SGR-2, the lower of the two surveyed reaches (SGR-1) is a moderately confined predominantly cobble-bedded plane-bed channel (Photo SGR-1-1 in Appendix B). Large bar deposits within the reach consist of coarse-grained gravel and cobble. As with other reaches, LWD loading is low within this survey reach and provides little hydraulic complexity or opportunity for sediment storage. The floodplain areas adjacent to the channel are primarily deciduous-dominated forest and provide overall poor LWD recruitment potential (Herrera 2005). The elevation of these floodplain surfaces is variable, however, and terraces ranging up to 3 m above the current bankfull stage indicate a wide range of historical channel elevations.

At the downstream end of survey reach SGR-1, portions of a 4-m-high floodplain terrace (3 m above bankfull stage), which appears to have formed behind a debris dam resulting from a landslide, remain at the left margin and within the central part of the floodplain valley. The extent of this terrace provides an indication of historical sediment loads through this reach (Photo SGR-1-2 in Appendix B). The stratigraphy of the terrace includes clayey silt layers and beds of sand with coarser alluvial deposits, representing a higher-energy depositional environment at the upstream end of the terrace (Photos SGR-1-3 and SGR-1-4 in Appendix B). Samples of wood collected from the debris dam and the terrace deposit were sent to an analytical laboratory for radiocarbon dating (Figure 4.34). Results from these samples, along with the observed extent of the remaining portions of the terrace surface, the terrace height, and the dimensions of the floodplain valley, were used to calculate historical deposition rates behind the debris dam. See Appendix D for radiocarbon dating data.

Radiocarbon dating of LWD from the debris dam deposit at the downstream end of the terrace formation (Sample Grays RC 1) and from wood exhumed from within the terrace deposit (Sample Grays RC 2) at 2 m above current bankfull stage (1 m below the top of the terrace surface) indicate that the terrace began forming between 210 and 330 years ago (275 years before present ±60 years), and deposition continued through at least 130 and 250 years ago (205 years before present ±60 years). The reported radiocarbon dates of the debris dam and terrace sample (275 years and 205 years before present, respectively) provide a reasonable estimate of the duration of deposition to the terrace sample elevation. Based on this duration and the relative position of the terrace sample within the deposit, a period of 100 years was estimated for historical deposition rate calculations.

Based on the distribution of remaining terrace surfaces, the width of the floodplain, and the observed terrace height, terrace dimensions were estimated to be 300 m in length, 120 m in width, and 4 m in depth, with a total sediment volume of approximately 144,000 m^3. Assuming a depositional period of 100 years, the depositional rate behind the debris jam was estimated at approximately 1,440 m^3/year, or 2,450 tons/year. With a subbasin area of 52 km^2 for the South Fork Grays River, the depositional rate responsible for formation of the terrace behind the debris dam was 47 tons/km^2/year. This rate is close to estimates of historical sediment production rates determined in the assessment of mass wasting and surface erosion, which range from 56 to 125 tons/km^2/year, and supports the interpretation that sediment production has increased by many orders of magnitude since the onset of timber harvest.

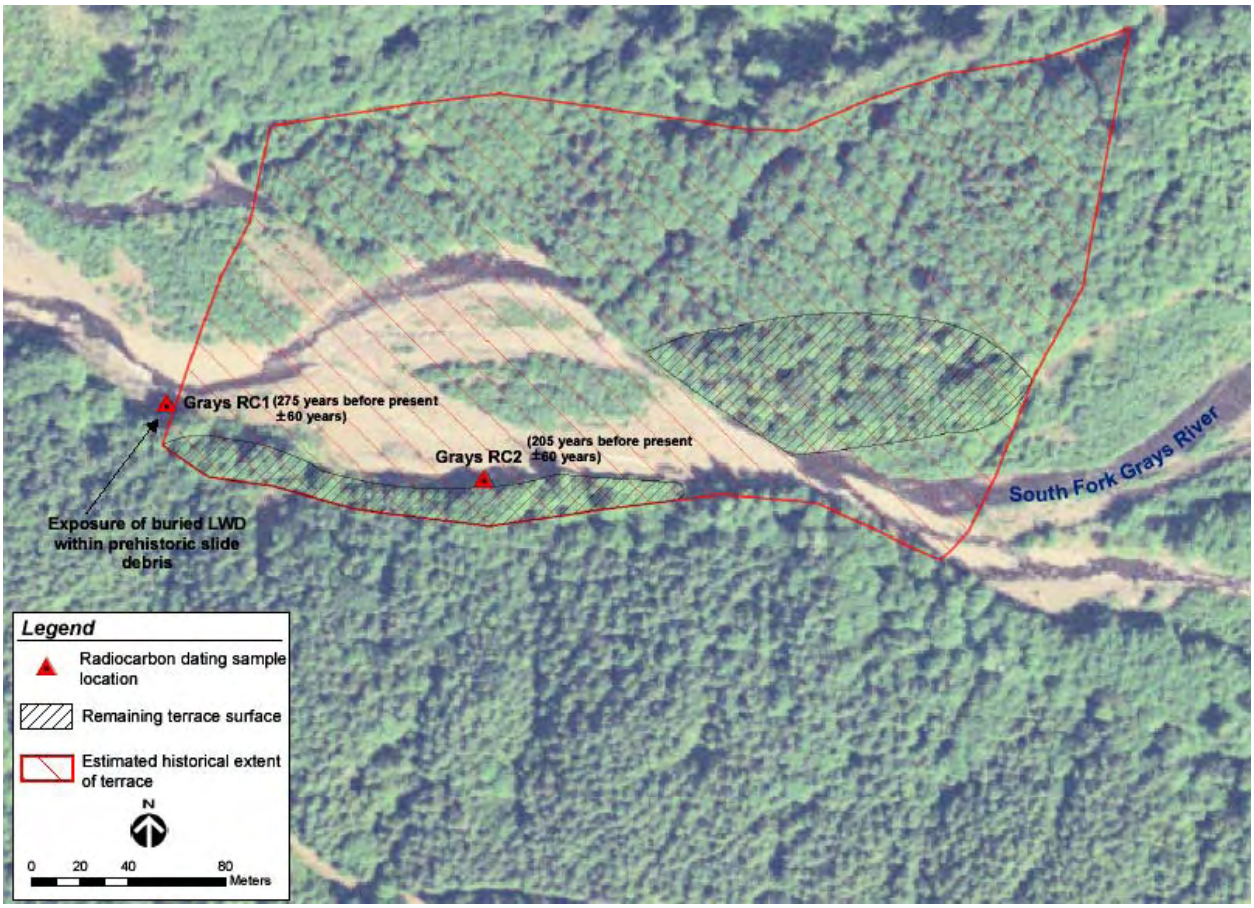


Figure 4.34. Locations of Radiocarbon Dating Samples and the Extent of Remaining and Estimated Historical Terrace Surfaces Upstream of the Observed Debris Dam Evidence in the South Fork Grays River, Washington (Herrera 2005)

4.3.2.2.6 West Fork Grays River

The West Fork Grays River geomorphic channel unit includes 7.0 km of the West Fork extending north from the pre-avulsion confluence with the mainstem Grays River (Figure 4.33). The West Fork Grays River geomorphic channel unit is defined as the portion of the West Fork with a channel gradient less than 4% and includes plane-bed and pool-riffle reaches. The West Fork geomorphic channel unit makes up less than 1% of the total channel network in the upper Grays River watershed and 5% of the alluvial response reaches (Table 4.23).

A reconnaissance stream survey of the West Fork Grays River was completed between the West Fork and mainstem Grays River confluence and the Grays River state fish hatchery at West Fork. The West Fork exhibits two distinct characters over this distance. Between the confluence with the mainstem Grays River and RKM 1.25 (survey reach WFG-1), the West Fork is a moderately confined, predominantly straight, plane-bed channel with little channel complexity. The channel is aligned against bedrock outcrops at the floodplain valley margin along much of this reach (Photo WGR-1-1 in Appendix B). The gravel and cobble substrate material is loose, and lateral bar deposits are not armored and consist of finer-grained sediments. Although these conditions indicate moderate to high sediment loading, there is

limited sediment storage in the simple channel configuration, which lacks functional LWD and recruitment potential from the adjacent deciduous-dominated riparian forest.

In contrast, survey reach WFG-2 contains moderate to high levels of LWD loading, and LWD jams in the reach frequently impart significant hydraulic complexity and provide considerable sediment storage in the active channel (see Figure 4.33 and Appendices B and C). The channel in this reach is predominantly a moderately confined, gravel-bedded, pool-riffle reach (Photo WGR-2-1 in Appendix B). Lateral channel instability is evidenced by undercut riparian vegetation and bank erosion into floodplain valley margins (Photo WGR-2-3 in Appendix B). Within the current alignment, the channel position appears to be unstable as a result of the combination of locally recruited LWD and high sediment loads. There is evidence of recent avulsion where the old channel alignment has been abandoned at the left valley margin and the current channel alignment has become established through an area of moderate to low LWD recruitment potential (Photo WGR-2-2 in Appendix B). Significant deposits of very loose marine siltstone sand and gravel were observed to fill pools and cover more consolidated substrate material in many locations. These conditions are indicative of high sediment delivery to this reach.

A record of regular dredging at the bridge to the Grays River fish hatchery also indicates high sediment delivery to the reach. According to discussions with Grays River state fish hatchery personnel, dredging of 50 m of the West Fork Grays River upstream of the fish hatchery bridge has removed at least 2,500 m^3/year (4,300 tons/year) of bedload sediment annually from 1999 to 2003. The 4,300 tons/year dredged annually from the West Fork amounts to about 5% of the estimated sediment production of approximately 89,000 tons/year in the West Fork Grays River subbasin. Because bedload sediment is thought to generally constitute approximately 5-20% of the total sediment transported through a typical channel network, the quantity of the dredged material suggests that sediment yield estimates are reasonably accurate (Dietrich and Dunne 1978; Montgomery 1999; Hicks and Gomez 2003).

4.3.2.2.7 Lower Crazy Johnson Creek

Lower Crazy Johnson Creek is a gravel- and sand-bed channel fed by a 1.8-km^2 drainage area (Figure 4.33). The lower reach of the Crazy Johnson Creek is located on the mainstem Grays River floodplain. Streambanks are typically composed of alluvial sand, gravel, and cobble sediment, indicating historical deposition in this area from the mainstem Grays River. Beaver-dam ponds are common along Crazy Johnson Creek, creating backwater environments. Beaver-dam ponds are often correlated with occurrences of LWD (Photo CJ-1-1 in Appendix B). These beaver ponds create hydraulic gradients across the floodplain, and upwelling in Crazy Johnson Creek was clearly visible downstream of most ponds. Portions of the length of Crazy Johnson Creek are likely inundated by floodwaters during high flows on the mainstem Grays River. Such high flow events would increase the ground water table in floodplain deposits and increase the hyporheic recharge, or subsurface flow, to Crazy Johnson Creek.

4.3.2.2.8 Mainstem Grays River

The mainstem Grays River geomorphic channel unit includes approximately 12 km of the Grays River upstream of RKM 22.5 (Figure 4.33). The mainstem geomorphic channel unit is defined as the portion of the Grays River with a channel gradient less than 4%, although the unit also includes the confined bedrock reach from RKM 22.5 to RKM 27.0 where local gradients exceed 4%.

The mainstem Grays River geomorphic channel unit makes up 1.3% of the total channel network length in the upper Grays River watershed and 4.8% of the total length of the alluvial response reaches (Table 4.23). The mainstem Grays River is a moderately confined to confined, predominantly cobble-bedded, plane-bed channel that lacks significant hydraulic complexity. Large lateral bar deposits are visible within the channel, even where it is confined, indicating high levels of sediment loading. Instream LWD loading is generally low, as is the recruitment potential from adjacent riparian forests.

4.3.2.2.9 Grays River Response Reach

The Grays River response reach includes approximately 4 km of the Grays River from the SR 4 highway bridge at RKM 18.5 to the downstream end of the bedrock canyon at RKM 22.5 (Figure 4.33). The present confluence of the mainstem and West Fork Grays River is used to divide the Grays River response reach into two subreaches: the SR 4 subreach and the Gorley subreach. The SR 4 subreach (RKM 18.5 to 20.3) extends from the highway bridge to the present confluence of the West Fork and mainstem Grays River at the downstream end of the 1999 avulsion path. This subreach includes the two downstream-most meanders of the Grays River response reach. The Gorley subreach (RKM 20.3 to 22.5) extends upstream from the present confluence of the West Fork and mainstem Grays River (at the downstream end of the 1999 avulsion path) to the downstream end of the bedrock canyon, and includes both the pre- and post-avulsion mainstem channel alignments and the three upstream-most meanders in the response reach.

Channel gradients throughout the response reach are generally less than 2%. The response reach makes up 0.3% of the total channel network length in the upper Grays River watershed and 2.8% of the total length of the alluvial response reaches (Table 4.23). The field survey of the response reach focused primarily on the Gorley subreach and extended from the present confluence of the West Fork and mainstem Grays River (the downstream end of the 1999 avulsion path) to the downstream end of the bedrock canyon, including both the pre- and post-avulsion mainstem channel alignments.

Before levee construction in the 1960s, the lateral channel migration zone (CMZ) along the Grays River response reach likely included the entire floodplain and was limited only by the bedrock walls of the valley margin. Levees constructed in the area of the West Fork confluence in the 1960s reduced the width of the historical CMZ from approximately 750 m to 200 m. The downstream segment of the pre-avulsion mainstem alignment is confined to a narrow, 50-m-wide corridor by levees constructed on both sides of the channel. A series of channel avulsions in 1999 (discussed in greater detail later in this report) breached the levees where they connected with bedrock at the left floodplain margin. This (1999) avulsion reconnected the channel with the former floodplain (WDFW 2000).

Valley width continues to increase downstream through the SR 4 subreach to over 900 m at the highway bridge. A series of streambank rip-rap revetments (or *groins*) was constructed between 1982 and 1996 (date based on aerial photographs) along the outside meander of the SR 4 subreach upstream of the confluence with Fossil Creek. A levee constructed earlier (1960s) was along the shoreline opposite the revetments. Construction of the revetments and levees has reduced the CMZ in this area of the SR 4 subreach from approximately 600 m to 350 m.

In the downstream-most section of the SR 4 subreach, natural bedrock outcroppings (Figure 4.33) maintain the channel meander pattern and constrain the CMZ of the river. The streambanks along the section of the SR 4 subreach just upstream of the highway bridge have been armored to prevent channel migration and maintain the channel alignment with the SR 4 highway bridge. This was likely done during bridge construction.

Widespread streambank erosion and channel adjustment are evident throughout the channel alignment created by the 1999 avulsion in the Gorley subreach. Trees buried in growth position and floodplain surfaces at a wide range of elevations throughout the survey reach indicate significant changes in vertical channel elevation (Photo GRR-1-2 in Appendix B). When the levee was breached during the December 1999 avulsion event, it is estimated that approximately 1 m of aggradation occurred over much of the floodplain surface that previously had been isolated from the active channel since at least 1966 (Photo GRR-1-4 in Appendix B). Significant deposition is also evident in pre-avulsion channel alignments upstream of the avulsion. Channel aggradation within this segment during the 1999 storms likely contributed to flows overtopping and breaching the levee. Appendix J contains an analysis of the 1999 avulsion compiled by WDFW hydraulic engineers (WDFW 2000).

The mainstem response reach is a pool-riffle channel lacking significant complexity (Photo GRR-1-1 in Appendix B). Large lateral and mid-channel bars of fine and coarse sediments indicate high sediment loading and suggest a high response potential to any future increased LWD loading. Grade control or hydraulic complexity evoked by channel obstructions would provide storage and sorting of alluvial sediment, yet few roughness elements such as functional LWD were observed within the study reach. Individual pieces of LWD resting on bar tops appear readily mobile at high-flow stages. In contrast, historical logjams exhumed from the floodplain by bank erosion in a number of locations throughout the response reach suggest that historical LWD loading was greater than at present (Photo GRR-1-3 in Appendix B). These ancient logjams are consistent with conditions that prevailed in the old-growth floodplain forest once found along the river.

An analysis of historical channel conditions focuses primarily on the SR 4 subreach and the Gorley subreach within the 4-km response reach of the mainstem Grays River (RKM 18.5 to 22.5 in Figure 4.33). Historical channel patterns have been influenced by timber harvest, levee construction, bank protection structures, and the operation of upstream splash dams. Levees and bank revetments constructed along the mainstem Grays River have affected channel patterns since at least the 1960s.

Splash dams functioned to store water and timber that was then released during large storm events to enable log to be moved downstream (see Figure 4.35 and Appendix A). This process had catastrophic impacts on channel conditions. Splash dams are known to have been operated for at least several seasons on the South Fork, West Fork, and mainstem Grays River during the period from 1880 to 1910 and were used within the upper Grays River watershed as late as 1937 (Sedell and Luchessa 1982). Splash-damming caused significant bed scour that removed spawning gravel and LWD, widened the channel, and reduced channel complexity. In addition, channel clearing prior to splash-dam log-drives removed obstructions such as LWD, natural debris jams, boulders, and riparian vegetation and compounded the habitat degradation associated with splash-damming (Sedell and Luchessa 1982). Because these practices were followed before detailed descriptions of the river morphology and aerial photographs were made, major portions of the Grays River had likely been severely altered by the time they were first documented.

Figure 4.35. Splash Dam near the Mouth of the South Fork Grays River, Circa 1937 (Wahkiakum County Archives)

4.3.2.3 Historical Channel Conditions

Historical analysis of channel position focused on the response reach segments of the mainstem Grays River. The historical alignments of active channels were delineated from digital georeferenced aerial photographs (Kondolf and Piegay 2003). Orthorectified USGS digital orthophoto quarter-quadrangles (DOQQ) from 1996 were obtained from the Washington Geospatial Data Archive (WA-GDA). Channel migration patterns inferred from the sequence of aerial photographs were compared with the 2003 survey completed for the response reach of the lower mainstem of the Grays River. The aerial photographs used in the historical analysis of channel migration patterns are summarized in Table 4.24. Aerial photographs were georeferenced to the 1996 USGS DOQQ using first-order polynomial (affine) transformation in the GIS. Control points were placed evenly along the channel in each photograph to maximize georeferencing accuracy in this area. Road bends, road crossings, corners of buildings, bedrock outcrops, and large trees were used to guide control point placement for georeferencing. Six or more control points typically were used to georeference more recent photo sets. Five or more control points typically were used for older photo sets because older photo sets had fewer distinct features in common with the 1996 orthorectified photo set.

Table 4.24. Aerial Photographs Used in the Analysis of Historical Channel Migration Patterns in the Grays River (Herrera 2005)

Date of Aerial Photograph	Source	Scale
1939	U.S. Army Corps of Engineers	1:10,200
1966	U.S. Army Corps of Engineers	Unknown
1970	Washington Department of Natural Resources	1:12,000
1982	Washington Department of Natural Resources	1:12,000
1996	U.S. Geological Survey (DOQQ)	1:12,000
2003	Washington Department of Natural Resources	1:32,000
DOQQ = Digital orthophoto quarter-quadrangles.		

The boundaries of historical active channel alignments (channel polygons) were digitized from the georeferenced photographs at a scale of approximately 1:2,000, although the scale was adjusted between 1:750 and 1:10,000 to aid in interpretation. The extent of the unvegetated channel was used to define channel polygons to account for seasonal variability in river height and width. To minimize small differences in channel edge due to georeferencing error, one unvegetated channel polygon was created in GIS from the 1996 aerial photo set. This polygon was copied and modified to create the 1982 unvegetated channel. The process was repeated by modifying the unvegetated channel of one photo set to create the unvegetated channel of the previous photo set. Other photo sets were taken into account when unvegetated channels were digitized. For instance, the same channel edge was used when the previous and next aerial photographs indicated no change in the channel boundary. Side channels that were visibly carrying water or were unvegetated were included in the unvegetated channel layer if they were over 10 m wide. Forested islands larger than approximately 1,000 square meters were not included as part of the unvegetated channel. The historical extent of the active channel is presented in Figures 4.36 and 4.37. Channel positions for each photo year are shown in detail in Appendix B.

4.3.2.4 Channel Migration Analysis

Two principal forms of channel migration were identified on the Grays River: *lateral migration* and *avulsion*. Lateral migration involves persistent movement of the channel across the valley, particularly at the outer edge of the meanders. Lateral migration leaves a geomorphic legacy of scroll bars and erosion scars matching the channel curvature. Avulsion involves the rapid relocation of the channel to a new alignment across the floodplain. A shift in channel location between aerial photographs was classified as an avulsion if vegetation between the pre- and post-avulsion channel was not disturbed (i.e., forest greater in age than the time span between the two photo sets) by channel movement.

Historical channel alignments of the Grays River were reconstructed from aerial photographs and information collected during the field reconnaissance (Herrera 2005). The geomorphic response of the mainstem Grays River was evaluated quantitatively in terms of temporal changes in reach-scale meander geometry (i.e., sinuosity, meander wavelength, radius of curvature, and migration rates), as well as qualitatively through observations of changes in vegetation patterns and channel response (i.e., gravel bar deposition, channel aggradation, and avulsion). The following sections describe the methods and results of the historical channel analysis.

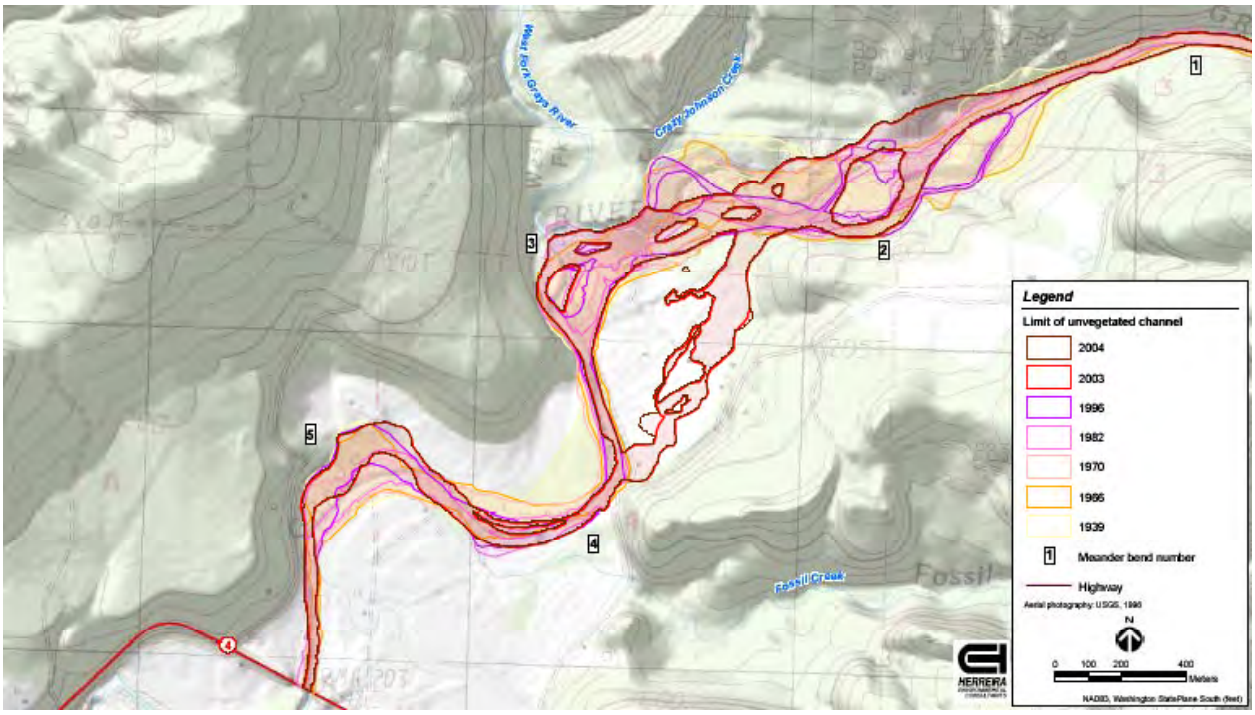


Figure 4.36. Historical Channel Locations for 1939–2004, Grays River Response Reach (Herrera 2005)

4.3.2.4.1 Lateral Migration

Two methods were used to quantify lateral migration in the study reach through the period of aerial photographic record. The first method, based on a linear analysis, was used where lateral migration of recognizable channel features occurred between subsequent periods. An alternative method, based on a polygon analysis, quantified erosion by subreach by determining the difference in spatial extent of the unvegetated channels between photographs (Kondolf and Piegay 2003).

Rates of historical lateral channel migration were calculated from the change in position of the outside edge of meander bends and the time elapsed between aerial photographs. The outside edge of the meander bend was defined using the extent of the unvegetated channel delineated from aerial photographs. Average annual rates of lateral bank erosion were calculated from the distance between the outward and downstream translation of a meander bend apex and the length of time between aerial photographs (Kondolf and Piegay 2003).

The five meander bends in the mainstem response reach were classified as translational, stable, or unstable, based on the historical pattern of lateral channel migration. A *translational meander* is defined as progressive meander growth and erosion of the outside bank identified in three or more consecutive aerial photographs. A *stable meander* is defined by lateral migration that varies by less than approximately one channel width between aerial photographs. An *unstable meander* is identified by erosion of the outside bank for less than three consecutive aerial photographs. For most of the unstable meanders, the direction of channel migration reversed between aerial photographs.

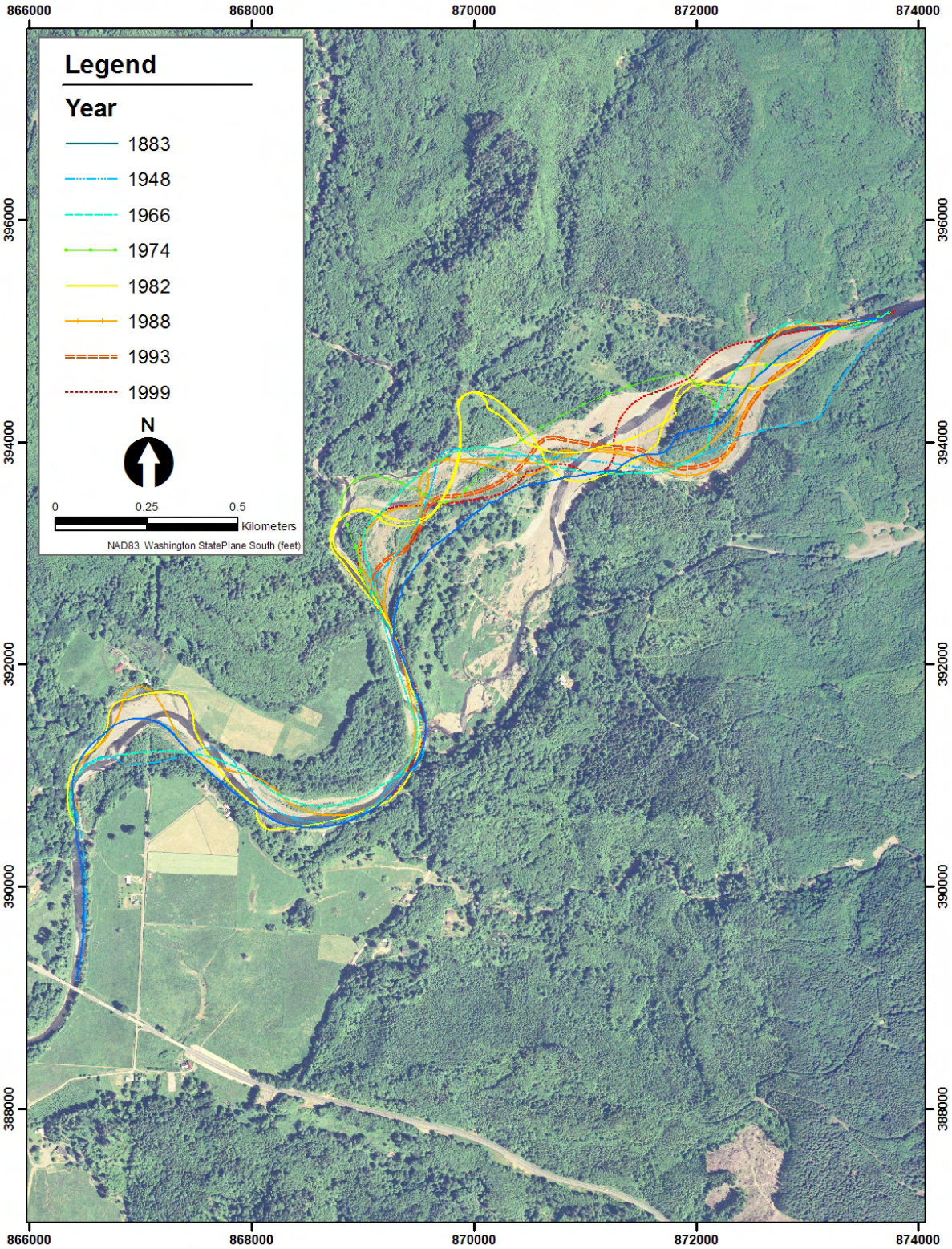


Figure 4.37. Historical Channel Locations for 1883–1999, Grays River Response Reach

Historical channel migration patterns for each meander bend in the Grays River mainstem response reach are summarized in Table 4.25. Both the rate and direction of lateral channel migration varied considerably during the period of historical record. Although the radius of curvature of Bend 1 increased through the historical period of record, the meander apex remained relatively fixed through the bedrock-confined segment. The meander in Bend 4 was also stable between 1996 and 2003, where aerial photographs indicate a series of groin walls constructed along the outside (left) bank of this meander between 1982 and 1996 (Figure 4.36). Bend 4 is characterized by an unstable migration pattern prior to 1982. Unstable migration patterns were also identified in Bend 2 for the period of historical record, despite partial confinement by bedrock along the southern margin of the valley.

Table 4.25. Linear Analysis of Lateral Channel Migration Rates for the Mainstem Grays River (Herrera 2005)

Meander Bend	Migration Pattern	Photographic Period	Lateral Bank Erosion (m)	Average Migration Rate (m/year)
1	Stable	1939–2003	–	–
2	Unstable	1939–2003	–	–
3	Translational	1970–1996	66	2.6
4	Unstable Stable	1939–1982 1996–2003	– –	– –
5	Translational	1970–2003	109	3.8

Translational migration patterns (i.e., progressive meander growth identified in three or more consecutive aerial photographs) were limited to Bends 3 and 5 for the period of historical record (Figure 4.38). Approximately 66 m of bank erosion occurred in Bend 3 between 1970 and 1996. The average annual migration rate in Bend 3 for this period was 2.6 m/year. The highest rate of average annual channel migration (3.8 m/year) was measured in Bend 5 between 1970 and 2003, where 109 m of bank erosion was measured during this 33-year period.

A polygon analysis of lateral channel migration rates was performed for the Gorley and SR 4 subreaches of the Grays River mainstem response reach and the West Fork and South Fork response reaches. Coverage by historical aerial photographs in the West and South Forks was limited to 1970, 1996, and 2003. Reach-average lateral migration rates were calculated from the change in the area of unvegetated channel polygons between aerial photo sets. Spatial erosion rates (m^2/year) were calculated for each channel reach by dividing the difference in polygon area by the time elapsed between aerial photographs. Reach-average lateral migration rates (m/year) were calculated by dividing the spatial erosion rates by the average reach length between photographs.

Results of the polygon analysis of lateral channel migration are presented in Table 4.26. The polygon method differs from the linear method in that it first quantifies the reach-average area over which erosion of the floodplain surface has occurred. Reach-average erosion rates were greatest in the mainstem Grays (SR 4 and Gorley subreaches), with the highest erosion rates measured between 1996 and 2003. Reach-average lateral migration rates were greatest in the Gorley subreach of the mainstem, where lateral migration rates varied from 0.72 to 5.13 m/year. Reach-average migration rates were relatively lower in the West Fork and South Fork response reaches, ranging from 0.64 m/year in the West Fork to 1.35 m/year in the South Fork.

Table 4.26. Polygon Analysis of Erosion Rates and Lateral Channel Migration Rates for the Mainstem, West Fork, and South Fork Grays River (Herrera 2005)

Reach/Subreach	Photographic Period	Erosion Rate (m^2/year)	Lateral Migration Rate (m/year)	Average Lateral Migration Rate (m/year)
SR 4 subreach	1939–1966	394	0.17	–
	1966–1960	3,858	1.71	–
	1970–1982	3,488	1.46	–
	1982–1996	3,978	1.57	–
	1996–2003	8,684	3.43	1.67
Gorley subreach	1939–1966	1,709	0.72	–
	1966–1960	3,364	1.38	–
	1970–1982	3,291	1.29	–
	1982–1996	6,691	2.71	–
	1996–2003	11,292	5.13	2.25
West Fork response reach	1970–1996	1,595	1.06	–
	1996–2003	553	0.64	0.85
South Fork response reach	1970–1996	3,041	1.35	–
	1996–2003	1,253	1.01	1.18

Reach-average lateral migration rates calculated from the polygon method are lower than lateral migration rates measured using the linear method. The average lateral migration rates measured by the linear method (Bends 3 and 5) are 1.5 to 2 times greater than the reach-average rates measured in the mainstem using the polygon method. The linear method best represents bank erosion rates, whereas the polygon method better represents the residence time of sediment stored in the floodplain between sediment-transporting events.

4.3.2.4.2 Avulsion

Channel avulsions typically occur in response to an increase in the local floodplain relief following channel aggradation (Knighton 1998). In rivers that regularly recruit large wood from their banks, the reduction in transport capacity caused by the flow resistance of large logjams can initiate channel aggradation and force periodic avulsions that maintain an anastomosing (multithread) channel network. The multiple channels that develop in forested floodplains increase the availability of aquatic habitat.

Channel aggradation and avulsion can also occur in response to an increase in sediment supply from logging operations (in the absence of an increase in transport capacity). The overtopping of channel banks following the loss of flood conveyance in aggrading channels can initiate an avulsion. However, the shallow braided channels and fine sediment that characterize most aggrading rivers may lack the habitat conditions selected by spawning salmonids.

Channel avulsions can be triggered also by feedback between flow dynamics and meander development. A gradual increase in meander amplitude reduces the channel gradient and corresponding transport capacity, thereby inhibiting further bank migration. Reduction below a threshold transport rate can cause

a channel to rapidly aggrade and avulse (Knighton 1998). The increase in channel length during meander extension in the 1970s and 1980s may have reduced the local transport capacity in Bends 2 and 3 and promoted local aggradation prior to the 1999 avulsion.

Field observations by WDFW (2000) indicate that the two neck avulsions through Bends 2 and 3 occurred in December 1999 (Appendix J). Both avulsions initiated at locations upstream of meander point bars and cut across the necks of the meanders (Figure 4.38). The avulsion in Bend 2 (point A in Figure 4.38) occurred sometime before December 3, 1999, and shifted the channel alignment to the north of the forested island. Field observations during high-flow conditions on December 3 document flow on both sides of the island through Bend 2; however, the majority of flow at this time was directed through the northern alignment and into the eastern end of the levee at a nearly perpendicular angle (point B in Figure 4.38). Lateral erosion of the levee and undermining of the levee toe, as observed on December 3, preceded the second and larger avulsion through Bend 3, which occurred on December 15, 1999 (Appendix J). The avulsion through Bend 3 reoccupied a former side channel along the eastern margin of the floodplain and rejoined the mainstem by breaching the downstream end of the levee, where the levee formerly connected with bedrock (point C in Figure 4.38) of the valley margin.

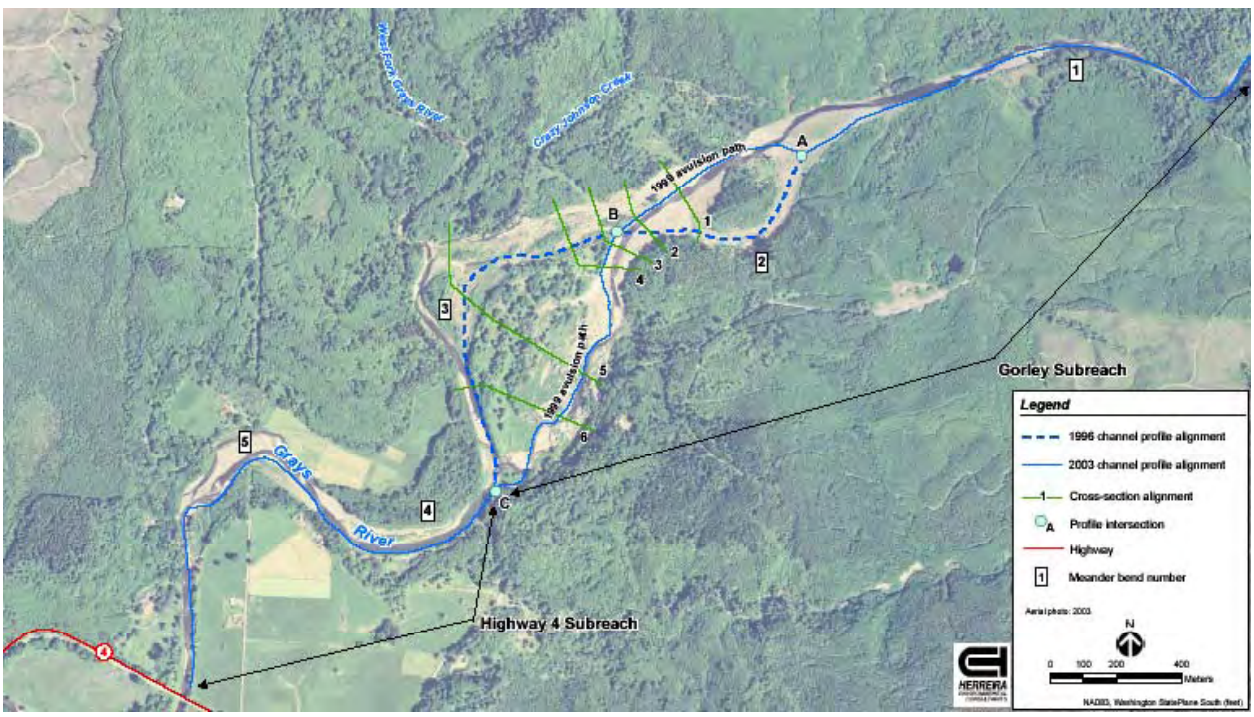


Figure 4.38. Avulsion Paths and Locations of Longitudinal Profile and Cross Sections Based on 2004 PNNL Topographic Survey of Grays River Response Reach (Herrera 2005)

4.3.2.4.3 Meanders

Historical trends in the radius of curvature (r_c) of meanders can be used to evaluate channel stability. For instance, stable meanders are typically characterized by a large r_c value, whereas channel instability in the form of bank erosion and avulsion can occur as a meander bend tightens and meander amplitude increases. Temporal changes in r_c values can also be used to forecast trends in bank erosion and the potential for channel avulsion.

Temporal variations in meander geometry and channel stability of the Grays River response reach were evaluated in terms of the radius of curvature of channel meanders measured from historical aerial photographs, based on methods developed by Nanson and Hickin (1983). Channel width (w) was measured at several inflection points along the channel. The average channel width was then used as the length of an arc (Δx) placed around the outside of the meander bend, starting from the point of maximum curvature (c_{max}). Two arcs with length Δx were traced on the outside of the bend to create points a_1, a_2, b_1, and b_2. Two circles were fitted through a_1, c_{max}, and a_2; and b_1, c_{max}, and b_2. The average of the radii of these two circles (r_1, r_2) is reported as r_c of the meander bend. Figure 4.39 illustrates this methodology.

The radius of curvature was measured for each of the five meander bends identified in the 1939–1996 aerial photographs and the three bends evident in the 2003 photographs. Bend 1 is located at the upstream extent of the Gorley subreach within the bedrock-confined segment. Bend 2 is located downstream of Bend 1 along the southern bedrock margin of the floodplain (Figure 4.40). Bend 3 is located north of the Gorley property and includes the Crazy Johnson Creek subreach along the upstream segment of Bend 3 and the confluence with the West Fork at the downstream segment of Bend 3. A rock revetment defines the left bank along the inside of Bend 3. The northern and western extents of Bend 3 are defined by bedrock along the valley edge in the vicinity of the West Fork confluence (Figure 4.40). The meander neck between Bends 3 and 4 is confined by levees on both sides. The outside (left) bank of Bend 4 is also armored with a gravel/rock revetment and a series of groin walls constructed between the 1982 and 1996 aerial photographs (Figure 4.40). Bend 5 is located upstream of the SR 4 bridge and is unconfined by levees but is partially confined by bedrock along the right bank (Figure 4.40).

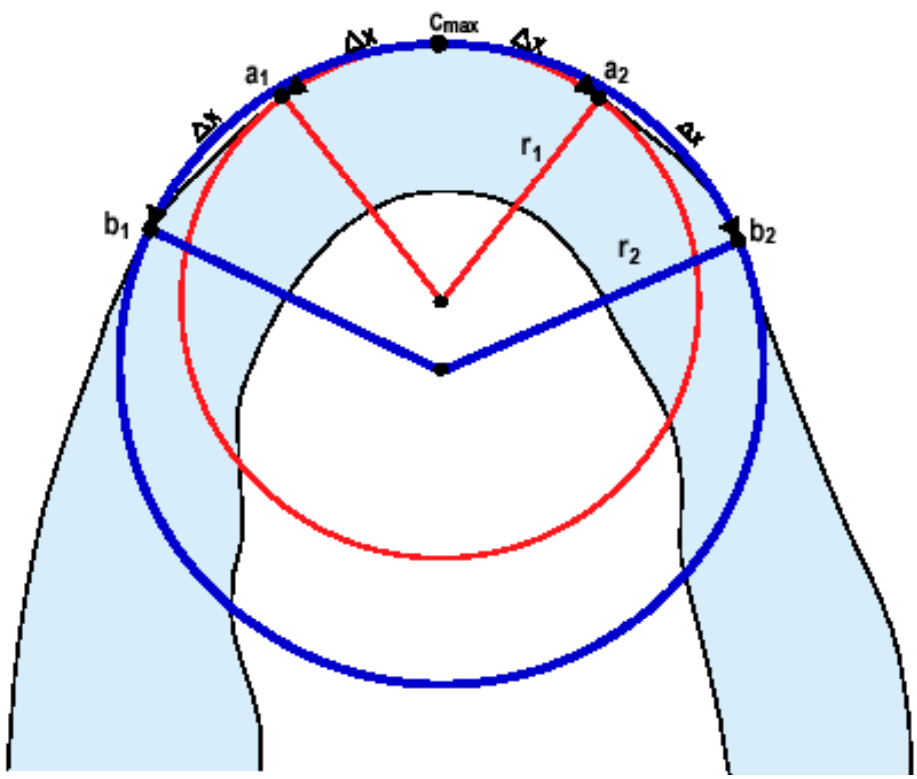


Figure 4.39. Measurement of Radius of Curvature of a Meander Bend (adapted from Nanson and Hickin 1983)

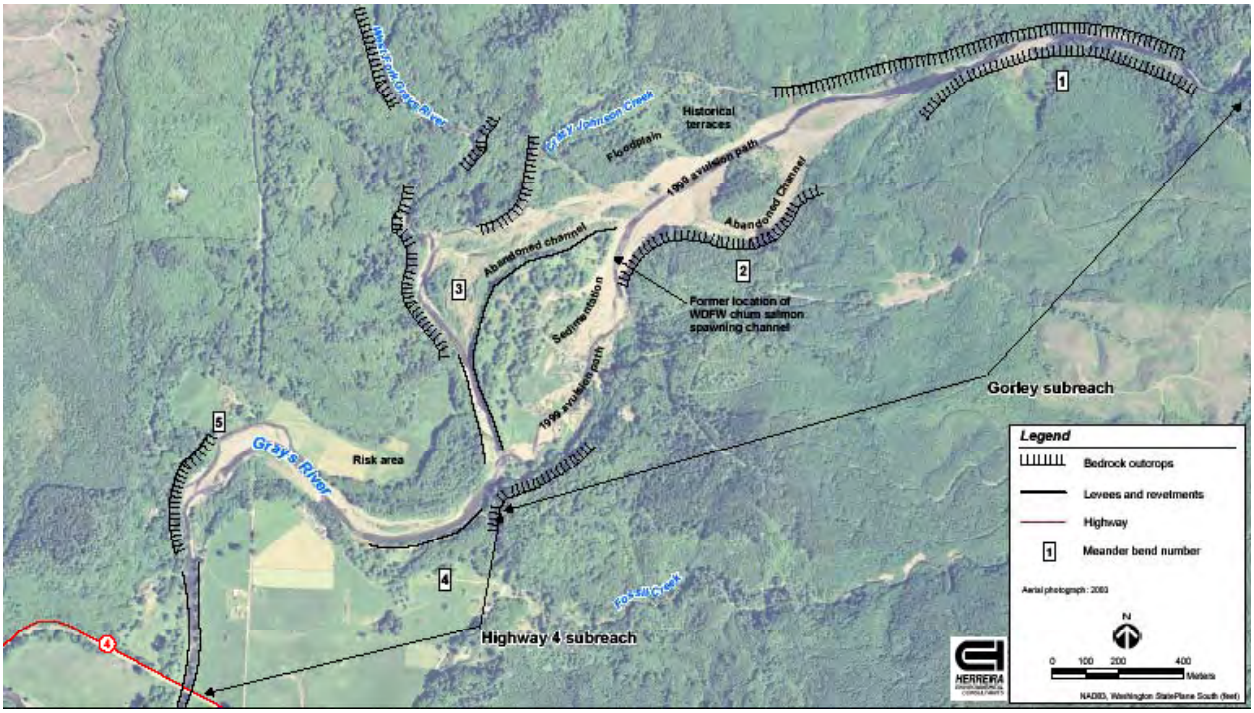


Figure 4.40. Bedrock and Levee Controls on Channel Migration in the Grays River Response Reach (Herrera 2005)

The radii of curvature of these five meander bends measured during the period of historical record are summarized in Table 4.27 and Figure 4.41. Original data for each year of the analysis are presented and discussed in detail in Herrera (2005).

The largest radius of curvature (prior to the 1999 avulsion) was measured in Bend 1. The radius of curvature in Bend 1 increased consistently during the period of historical record as the channel alignment progressively straightened downstream of the canyon opening. In contrast, the radii of curvature in Bends 2 and 3 declined steadily through the 1970s and 1980s, until the 1999 avulsions. The channel avulsions in 1999 forced the abandonment of Bends 2 and 3 and straightened the channel alignment.

Table 4.27. Radii of Curvature for Each Grays River Meander Bend Measured for the Period of Record, 1939–2003 (Herrera 2005)

Bend Number	Radius of Curvature (m)					
	1939	1966	1970	1982	1996	2003
1	247	458	417	427	648	672
2	202	314	195	194	168	–[(a)]
3	120	132	272	142	91	–[(a)]
4	112	124	132	119	131	216
5	79	74	95	112	99	103

(a) Bends 2 and 3 were cut off by the 1999 avulsion events.

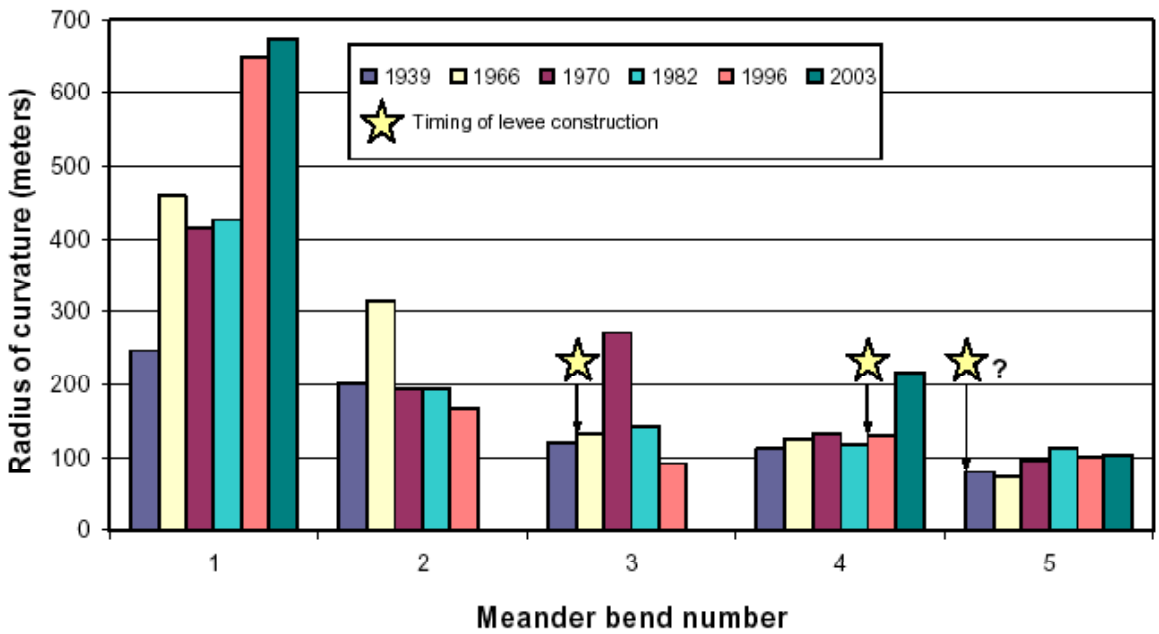


Figure 4.41. Average Radius of Curvature for Five Meander Bends on the Grays River, Washington, During the Period of Record, 1939-2003 (Herrera 2005)

Consequently, the radii of curvature are not reported for Bends 2 and 3 in 2003 (Table 4.27). The radius of curvature of Bend 4 remained relatively stable during the period of record. Although there was significant movement of the channel alignment through Bend 5, the radius of curvature has remained relatively unchanged through the period of record.

Meander geometry was also evaluated from trends in bend curvature by normalizing the radius of curvature by the channel width (r_c/w). Normalizing the radius of curvature by width allows for a scale-independent comparison of results with prior studies (Kondolf and Piegay 2003). In a study of the Beatton River in British Columbia, Nanson and Hickin (1983) showed that the meander migration rate (bank erosion) is greatest where r_c/w falls between 2.5 and 4 (Figure 4.42). Bank erosion is considerably less for r_c/w outside this range. Bagnold (1960) found that when r_c/w falls below a threshold of 2, the formation of large eddies generates flow resistance that inhibits bank erosion and continued meander migration, thereby shifting channel adjustment from bank erosion to avulsion between meander bends. Results from similar studies indicate that the decline in bank erosion in tight meander bends, where r_c/w is less than 2, can lead to a channel avulsion through a chute or neck cutoff of the meander. In a study of Welsh rivers, Lewis and Lewin (1983) found that the maximum frequency of chute and neck avulsions coincided with r_c/w between 1 and 2.

Historical trends in bend curvature for Bends 2 and 3 leading up to the 1999 avulsions on the mainstem Grays River are presented in Figure 4.43 (Herrera 2005). Although r_c/w in Bend 2 fluctuated about the upper envelope of the critical range, r_c/w in Bend 3 (the site of the larger avulsion) declined to 2.0 (or possibly lower) prior to the 1999 avulsion. Based on a comparison with similar studies, the reduction in r_c/w through Bend 3 from 1970 to 1996 may have contributed to the 1999 avulsion.

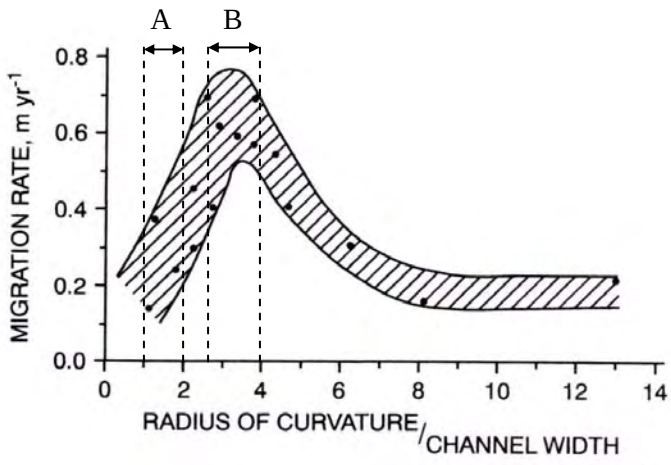


Figure 4.42. Meander Migration Rate Versus Bend Curvature (r_c/w) for the Beatton River, British Columbia (Nanson and Hickin 1983), Illustrating Optimal Range of Channel Avulsions for $1 < r_c/w < 2$ (A) and Maximum Migration Rate for $2.5 < r_c/w < 4$ (B)

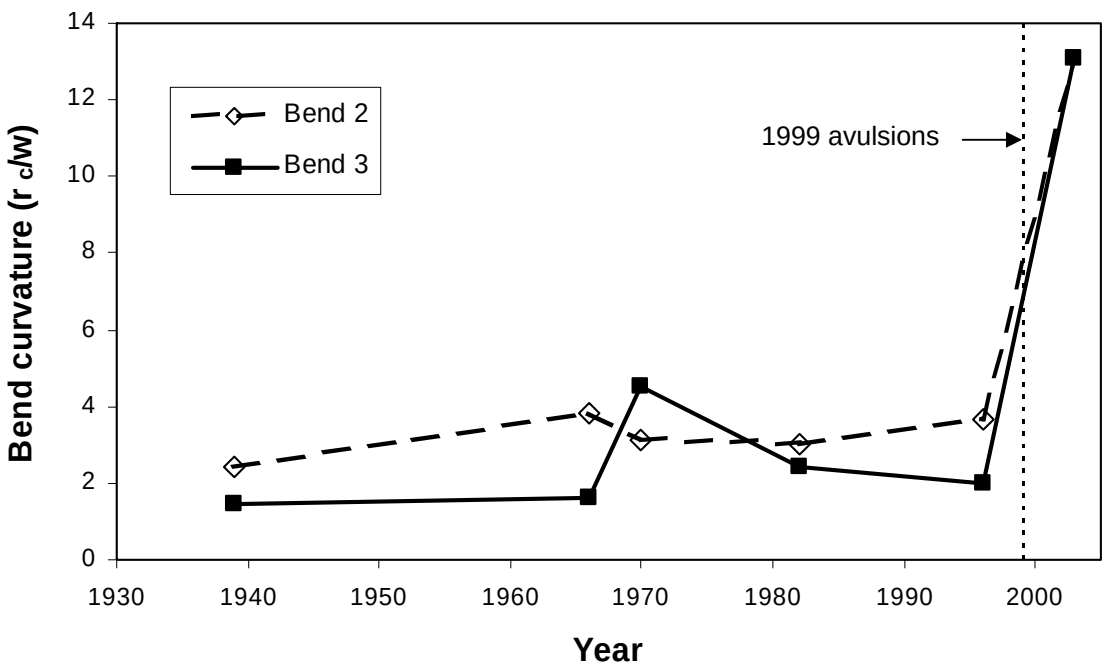


Figure 4.43. Temporal Variation in Meander Bend Curvature Relative to the 1999 Avulsions in the Grays River, Washington (Herrera 2005)

4.3.2.4.4 Sinuosity

Sinuosity of the mainstem Grays River was calculated from the ratio of channel length to straight-line valley length (a channel flowing along the centerline of a valley has a sinuosity of 1). Channel and valley lengths were measured in GIS from the digitized historical channels for the period of record (1939–2003). The historical trends in sinuosity are presented in Figure 4.44. Results indicate that sinuosity of the mainstem declined steadily from 1.37 to 1.33 between 1939 and 1966. Meander growth after 1966

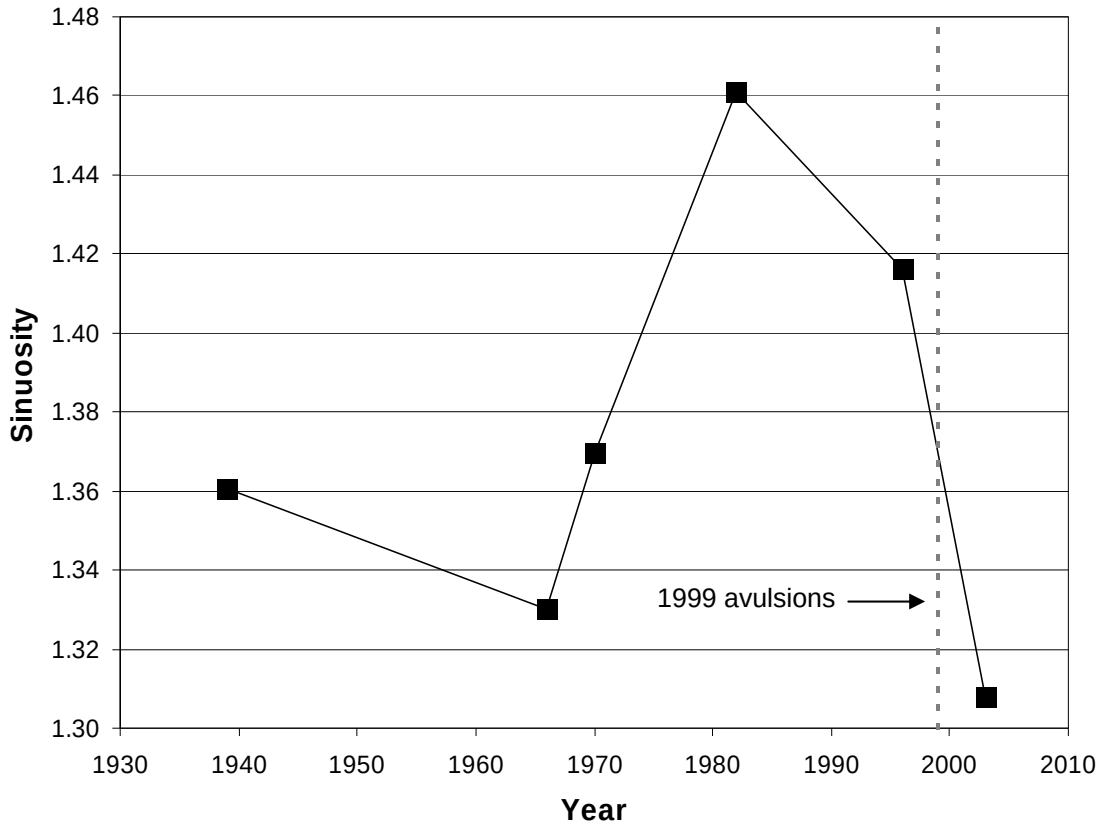


Figure 4.44. Temporal Variation in Sinuosity of the Mainstem Grays River

caused a rapid increase in sinuosity to 1.46 by 1982. Thereafter, sinuosity decreased rapidly to 1.42 by 1996 and declined further to 1.31 by 2003 following the channel avulsion event of 1999.

The reduction in sinuosity through 1966 is consistent with the trends predicted due to the increase in sediment supply from timber harvest. The increase in sinuosity during the 1970s followed the construction of levees in the 1960s. The levees locked portions of the channel into their 1960s alignment and focused migration to shorter, unconstrained segments, where meander growth between 1966 and 1982 increased sinuosity. Sinuosity declined sharply after the 1999 avulsions as a result of channel shortening.

4.3.2.4.5 Vertical Channel Adjustment

Vertical changes in the elevation of alluvial surfaces caused by the 1999 avulsion were assessed by comparing profiles and cross sections constructed along the channel alignments delineated from the 1996 and 2003 aerial photographs (Figure 4.45). Longitudinal profiles were constructed using surface elevations at the edge of water in the 2004 survey of the response reach (Figure 4.45). Cross sections were also constructed from topographic data derived from this survey (Appendix I). In Figure 4.45, the 2003 profile extends from Bend 1 at the upstream end of the Gorley subreach to the bridge at the downstream end of the SR 4 subreach. The 1996 profile extends from point A, crosses the 2003 alignment at point B, and joins the profile at point C (Herrera 2005).

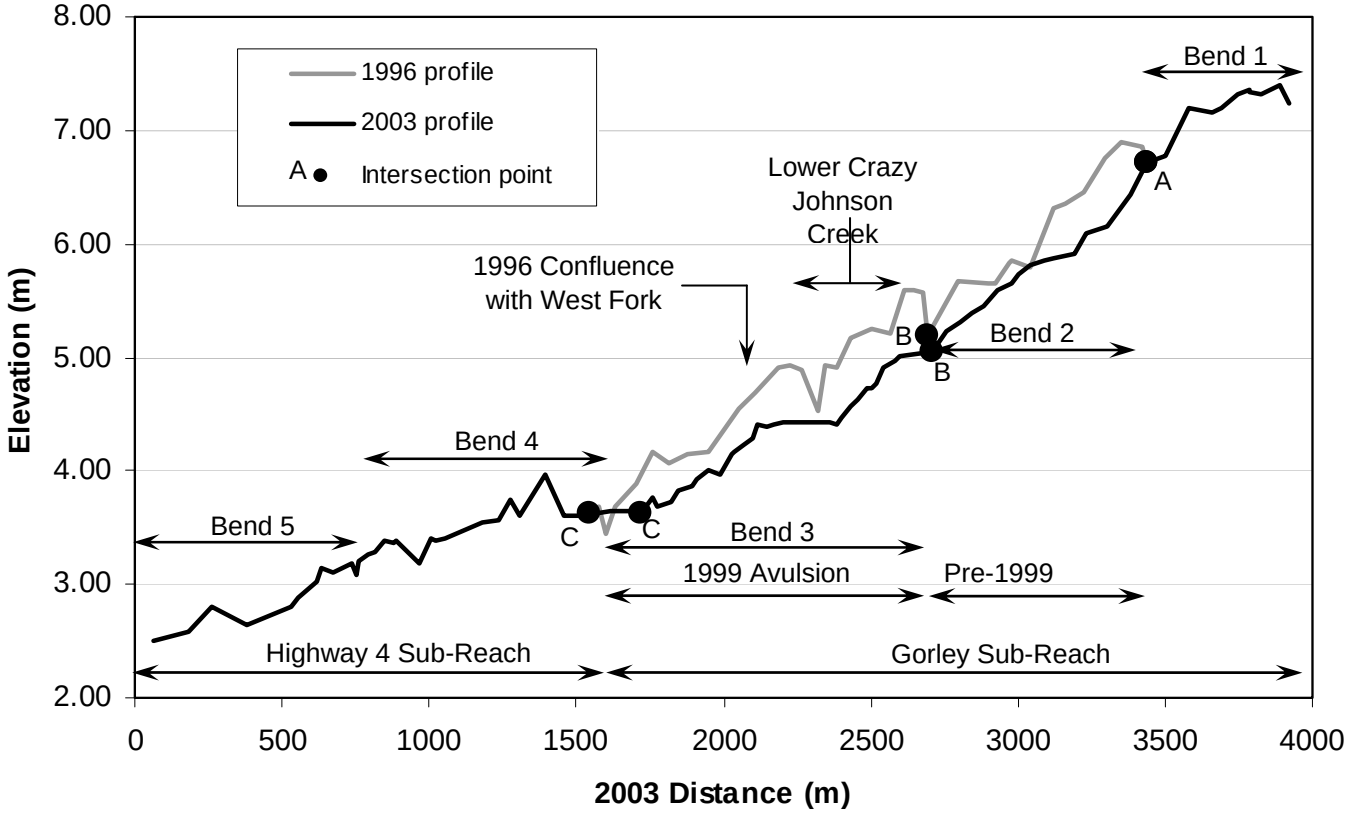


Figure 4.45. Channel Profile of the Grays River Response Reach from State Route 4 to the Bedrock Canyon (Herrera 2005)

The 1999 avulsions decreased the length of the mainstem by approximately 117 m (the offset at point C in Figure 4.44) and increased the reach gradient from 0.16% to 0.18%. Most of the channel shortening and increase in gradient occurred through Bend 3. Subsequent meander development in the new channel above point C since the 2003 aerial photograph has increased the channel length through Bend 3. A comparison of the 1996 and 2003 profiles in Figure 4.44 suggests that the 1999 avulsions were triggered either by aggradation downstream of points A and B along the old (1996) channel alignment, by levee failure and relocation through Bend 3 to an ancient channel alignment at a lower elevation, or by a combination of these two mechanisms.

Field reconnaissance confirms the recent deposition of sediment along the 1996 alignment downstream of point B. A comparison of the profiles indicates up to 0.6 m of aggradation along 1,900 m of the river between 1996 and 2003. Based on the average width of the active channel in 1996 (100 m), this change in bed elevation represents the deposition of approximately 114,000 m^3 (194,000 tons) of sediment during this 7-year period, or approximately 10% of the estimated 1.5 million tons of sediment delivered to the entire channel network of the upper Grays River watershed (but not necessarily routed to the lower response reach) during this same period. Observations of historical channel conditions are summarized in Table 4.28.

4.4 Geomorphic Analysis Conclusions

4.4.1 Analysis of Channel Sensitivity and Response Potential

The mainstem response reach of the Grays River (RKM 18.5 to 22.5) has been significantly altered from its natural condition by accelerated sediment supply stemming from historical land-use practices within the upper Grays River watershed and by the construction of floodplain levees. Results of the geomorphic assessment indicate that the lower mainstem is in a state of dynamic adjustment to the altered sediment regime and channel confinement by levees. The lower mainstem is located at a point of reduction in channel gradient at the transition from the moderately confined bedrock canyon to the tidally influenced reach between the SR 4 highway bridge and the Columbia River. The reduction in transport capacity at this break in slope, combined with a reduction in channel confinement, has made the mainstem reach particularly sensitive to minor increases in sediment supply. In addition, the lack of instream and floodplain hydraulic complexity (e.g., LWD and debris jams) exacerbates the problems resulting from the elevated sediment load.

Because fluvial systems are typically threshold-dominated, the response to cumulative effects can be abrupt when threshold conditions are exceeded (Abbe and Montgomery 1996; Montgomery and Buffington 1998; Montgomery 1999; Montgomery and Bolton 2003; Montgomery et al. 2003). The 1999 avulsion represents the most significant historical response of the mainstem channel to date, but events of similar magnitude are likely to continue and may even progress downstream as sediment stored in the Gorley subreach is transported to the SR 4 subreach.

4.4.2 Mainstem Response to Historical Watershed Disturbance

Historical variations in harvest rate, estimated sediment yield, and channel response variables are illustrated in Figure 4.46. The assessment of mass wasting and surface erosion indicates that the majority of sediment supplied to the channel network is generated by mass-wasting processes brought about by road construction and road use associated with timber-harvest operations. Based on the temporal

Table 4.28. Summary of Historical Observations for the Mainstem Grays River (Herrera 2005)

Location	1939–1966	1966–1970	1970–1982	1982–1996	1996–2003
Between Bend 1 and Bend 2	Wide active channel; north bank migrates south.	No significant change in channel position.	Active channel narrows.	Active channel widens to the north.	Secondary channel to the south is abandoned.
Bend 2	Inside bank retreats south; side channel to Crazy Johnson Creek abandoned.	Minor shift to north of outside bend of active channel.	Active channel width decreases; thalweg translates to north of vegetated island; secondary channel forms south of island.	Thalweg moves to the south; north alignment becomes secondary channel; dense vegetation on mid-channel island.	Thalweg abandons south channel and avulses to the north through vegetated island.
Bend 3 (upstream)	Thalweg contacts bedrock point and migrates south; vegetated island forms southeast of bedrock.	No significant change in channel location.	Meander translates downstream and contacts bedrock; radius of curvature tightens and flows through Crazy Johnson Creek.	Thalweg abandons small radius of curvature in north bend and erodes through vegetated island; flow splits between Bends 2 and 3.	Bend abandoned during large avulsion through levee and Gorley property.
Bend 3 (downstream)	Thalweg migrates northwest against bedrock at West Fork confluence; levees constructed along inside bend and between Bends 3 and 4 in early 1960s.	Migration rate to northwest decreases; mid-channel islands densely vegetated.	Channel continues to erode to the west; radius of curvature tightens; sinuosity increases.	Split flow captures Crazy Johnson Creek and West Fork Grays River and joins mainstem between Bends 3 and 4.	Bend abandoned by large avulsion; West Fork confluence with mainstem moves to Bend 4; channel shortened by 177 m; slope increases.
Bend 4	Inside bend retreats south across point bar.	No significant change.	Channel erosion to the south in the downstream portion of Bend #4.	Levee and groin wall constructed by 1996 along outside (left) bank.	No significant change.
Bend 5	Small high-flow channel at tight bend in pasture; erosion inside the downstream bend; no change in alignment at bridge.	Tight north bend abandoned and vegetated; no significant change in channel at bridge.	Bend migrates to the northeast (slightly upstream); no significant change in reach at bridge.	Channel erodes to the north into 1996 alignment; no significant change in reach at bridge.	Upstream reach of bend migrates across point bar to northeast; no significant change in reach at bridge.

relations among timber harvest and landslide frequency, and assumptions used in this geomorphic analysis, historical sediment yield to the channel network appears to lag behind harvest operations by approximately 25 years. For example, the increase in the harvest rate in the 1950s corresponds to the predicted increase in sediment yield through the late 1970s and 1980s (Figure 4.46). Likewise, the relatively stable harvest rate through the 1960s corresponds with moderate increases in sediment yield in the late 1980s, and the sharp increase in harvest rate in the 1970s is followed by the increase in predicted sediment yield in the late 1990s (Figure 4.46).

The sediment budget developed during this study does not route sediment through the channel network to response reaches, nor does the analysis account for all sediment storage within the watershed. However, the historical response of the mainstem Grays River was used as a basis to infer the lag time between sediment yield to the channel network and the supply of coarse sediment to the mainstem response reach. The response time was then used to project future responses within the mainstem to watershed disturbance.

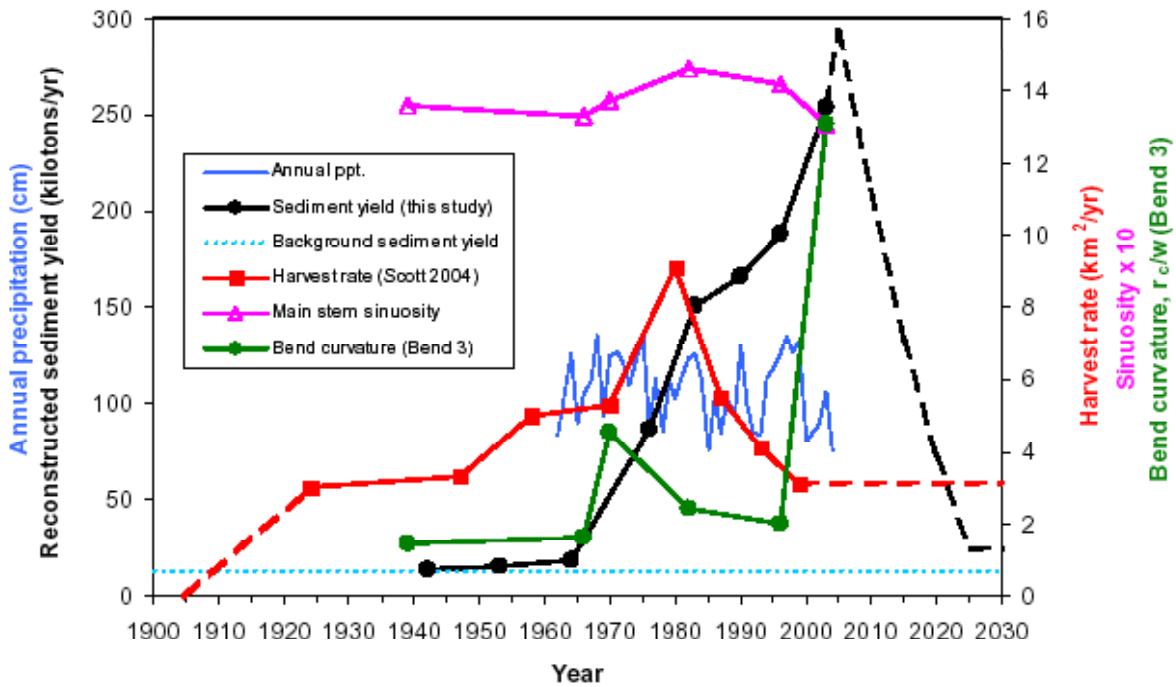


Figure 4.46. Composite Plot Illustrating Relations Between Forcing Functions and Response Variables (Herrera 2005)

Field observations indicate that large quantities of sediment are currently stored in headwater transport reaches and tributary floodplains. The additional lag time between sediment yield and delivery to response reaches will depend on the distance from the source, as well as transport rates and sediment capacitance (storage potential) within the intervening channel network.

The apparent lag time between timber harvest and sediment yield was used to forecast future trends in sediment yield to the channel network. Assuming that the 1999 harvest rate of 3.1 km^2/year remains

constant, and using a lag time of 25 years, sediment yield (S_y) expressed in kilotons/year is correlated to the harvest rate (H) expressed in square kilometers/year according to the empirical relation

$$S_y = 251*\ln(H) - 260 \quad (r^2 = 0.98, \text{n=7}) \tag{4.12}$$

Based on the assumed harvest rate and lag time, sediment yield to the channel network is projected to reach a maximum of approximately 290,000 tons/year in 2005 and decline to approximately 24,000 tons/year by 2025 but remain approximately 85% higher than the background erosion rate of 13,000 tons/year predicted by the analysis. The elevated erosion rate is attributed to sediment yield from mass wasting and surface erosion from the existing road network.

Temporal trends in channel morphology in the mainstem Grays River provide information on the lag time between sediment yield and the supply of coarse sediment to the mainstem. In addition, changes in channel form have been significantly influenced by floodplain modifications and levee construction. Both sinuosity and bend curvature of the mainstem increased shortly after levee construction in the 1960s and then declined by the early 1980s. The decline in sinuosity may be a response to the increase in sediment yield during the 1970s. If so, this response suggests a lag time of about 10 years between the basin-wide increase in sediment yield and the onset of channel adjustment in the mainstem channel.

When combined with the lag between harvest and sediment yield, the information developed during this study suggests 35 years as the characteristic response time for the cumulative effects of basin-wide timber harvest to significantly affect the mainstem response reach. Significant channel adjustment is expected to continue beyond this period.

This study also does not account for the potential effects of climate variability on landslide frequency. The natural variability in annual precipitation between wet and dry years is on the order of 2 to 5 years (Figure 4.46), which is significantly shorter than the predicted lag times between timber harvest, sediment yield, and channel response (i.e., 25 years). The 1999 avulsion followed 6 years of above-average precipitation but also occurred after 35 years of rising sediment yield within the watershed (Figure 4.46). Climate variability may provide a second-order control on channel response by mobilizing stored sediment and forcing channel change when conditions are near a threshold. The correspondence of the 1999 avulsion with the posited 35-year lag time between harvest and the onset of mainstem channel response suggests that the avulsion event was triggered by the cumulative effects of an increased sediment influx, channel confinement by levees, and the mobilization of stored sediment by above-average precipitation in the years preceding the avulsion.

4.4.3 Factors Influencing Future Sediment Yield

Past and future land-use (e.g., timber harvest and forest road management) practices in the upper Grays River watershed will continue to influence the rate of sediment yield to the Grays River channel network. The current trend of second- and third-growth harvest rotation is expected to reduce the reported 10-year lag between the loss of root strength and the peak in landslide frequency because smaller second- and third-growth roots are typically weaker than roots of old-growth trees. However, sediment yield might decline if the frequency of harvest rotation or the overall harvest rate is reduced. The harvest rate within the upper Grays River watershed has declined since 1980, and future sediment yield to the channel network is predicted to decline as well (Figure 4.46). Sediment yield could also be reduced by

eliminating timber harvest from slopes steeper than about 65%, where sediment yield from mass wasting is estimated to be 1,000 times greater than the yield on slopes less than 65%.

In addition, contemporary rates of sediment production by mass wasting are not sustainable with long-term soil production rates and could lead to the eventual depletion of soil on steep slopes, which could severely impair future timber production and other recreational and wildlife uses within the watershed. In a comparative study of debris-flow characteristics in old-growth and industrial forests, Bunn (2003) found that the sustained short-rotation harvest of headwater basins and removal of old-growth wood from headwater channels is diminishing soil depth on hillslopes and leaving headwater channels with an increased sediment flux and a consequent increase in sediment output to low-gradient response reaches.

Field observations in the upper Grays River watershed indicate that some hillslopes have already been stripped to bedrock by widespread mass wasting (Photo MW-4-1 in Appendix B). Past timber-harvest practices that removed instream LWD (either by snagging or splash-damming) and stripped floodplains of large trees severely reduced potential sediment storage sites within headwater channels of the upper Grays River watershed by disrupting the natural self-sustaining processes that recruit LWD to channels. Past timber-harvest practices not only increased the rate of sediment yield to the channel network but also accelerated the delivery of this sediment to the mainstem response reach by eliminating most of the natural sediment capacitance (i.e., instream storage) provided by LWD and debris jams. Under current conditions, sediment storage locations within the watershed can be expected to continue to shift from hillslopes to the response reaches of the channel network.

4.4.4 Projected Channel Response

This analysis predicts that channel response in the Grays River mainstem lags behind upland harvest activity by approximately 35 years. Therefore, the current instability within the mainstem (posited to be related to the 1980s spike in the timber-harvest rate) can be expected to continue and possibly increase through 2030. Response reaches throughout the upper Grays River watershed will likely be prone to continued instability and more avulsions so long as sediment supply exceeds threshold transport rates and the lack of instream storage remains the same.

Confinement and straightening of the mainstem response reach by existing levees tends to increase the local transport capacity (through increases in both slope and flow depth, which in turn increase shear stress) and shift future sediment deposition and channel response downstream of the SR 4 subreach. Additional channel avulsions are likely to occur in the response reach if measures are not taken to maintain and raise existing levees and revetments concurrently with the anticipated sediment aggradation. Alternatively, the removal of levees and the restoration of floodplain access would also reduce the likelihood of catastrophic avulsion such as occurred in 1999.

The current decreasing trend in channel sinuosity and increasing trend in meander bend curvature are also expected to continue within the Grays River mainstem. The reduction in sinuosity and development of a multi-threaded, braided channel visible in the 1996 aerial photographs signal a shift toward a transport-limited regime. Aggradation and natural straightening of the channel are typical morphological responses to an increase in sediment loading. The local increase in slope caused by continued aggradation (as well as confinement by levees) will likely shift the depositional front of the mainstem downstream from the Gorley subreach. The downstream advance will eventually force Bends 4 and 5 to

respond (Figure 4.47). Aggradation is likely to occur in the mainstem at the confluence with the West Fork, where there is a local decrease in slope (Figure 4.47). Backwater propagation up the West Fork would initiate aggradation in the diked reach between Bends 3 and 4 (Figure 4.47). The reduction in channel depth that follows would increase the likelihood of an avulsion through Bend 4 (Figure 4.47). Realignment of the channel through Bend 4 and continued downstream migration of the depositional front could trigger an avulsion through Bend 5 and threaten the SR 4 highway bridge crossing.

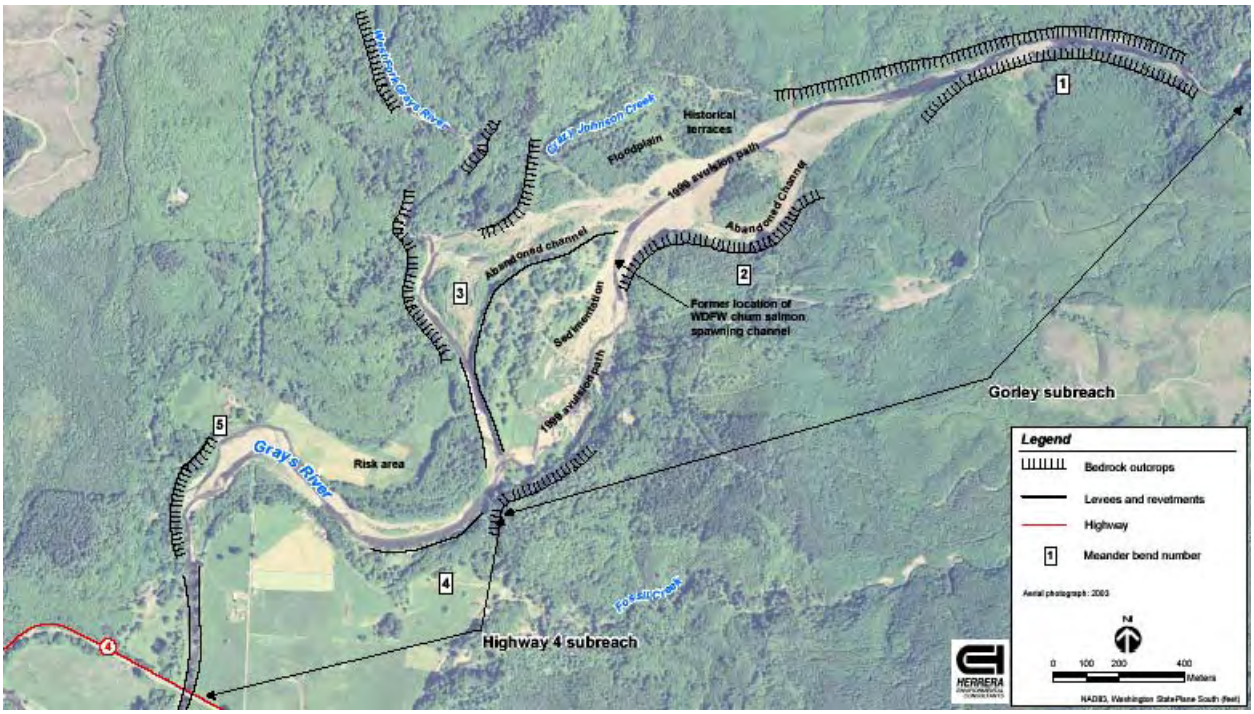


Figure 4.47. Grays River Response Reach (Herrera 2005)

The response to increased sediment supply is compounded by levees within the mainstem floodplain. In general, the levees restrict the natural tendency toward channel migration and floodplain sediment deposition. Isolation of the channel from the floodplain accelerates local aggradation and increases the potential for channel avulsion. Although levee construction may have initially provided short-term stability to portions of the channel and floodplain, channel confinement and floodplain isolation by the levees has forced channel adjustments to the shorter, unconfined segments of the river. Consequently, the floodplain may experience periods of channel stability punctuated by high-magnitude variability in channel configuration.

The 1999 avulsion to an ancient channel alignment of lower elevation suggests that the former channel alignment through Bend 3 has been aggrading (Figure 4.47). The sediment capacitance made available by the avulsion through the Gorley subreach may provide a temporary delay for future avulsions in Bend 3. Historically, the inflection point between Bends 3 and 4 has remained relatively stable due to levee construction and periodic dredging (Figure 4.47). However, the reduction in flows through this confined segment caused by the 1999 aggradation could accelerate aggradation and force an avulsion through Bend 4, where the levee contacts bedrock.

The observed response of the mainstem Grays River is analogous to historical channel changes in the lower Skokomish River of the southeastern Olympic Peninsula following extensive timber harvesting. Stover and Montgomery (2001) identified three phases of channel response to historical disturbance in the Skokomish River basin. The first phase involved rapid channel incision of nearly 0.5 m following riparian timber harvesting and removal of instream wood. The second phase was characterized by fluctuations in bed-surface elevation of up to 1 m that coincided with widespread timber harvest and road development in the basin during the 1940s and 1960s. Stover and Montgomery (2001) attributed the oscillations in bed elevation to sediment pulses moving through the channel network. The sediment pulses were linked to concurrent timber-harvest activities in the upper basin and the release of sediment stored in the nearby south fork following the harvest of riparian forests and removal of instream wood. Channel filling, increased channel width, and fining of bed sediment characterized the third phase of channel response through at least the end of the study in the late 1990s (Stover and Montgomery 2001).

In contrast with results of the analysis for the upper Grays River, the onset of channel aggradation on the Skokomish began rapidly, approximately 10 years after the commencement of intense upstream timber harvesting, and continued at a steady pace through 1997, at the end of the study period (Stover and Montgomery 2001). Results of the Stover and Montgomery (2001) study suggest a minimum of 50 years for the lower response reach of the Skokomish River to adjust to the influx of sediment from timber harvesting. Timber harvesting and road construction in the headwaters continue to contribute to ongoing aggradation and recurrent flooding within the Skokomish River valley (Stover and Montgomery 2001).

Observations from the field studies and analytical efforts can be generalized into a conceptual model of the temporal relations among watershed disturbance, sediment yield, and channel response within the upper Grays River watershed (Figure 4.48). Sediment yield to the channel network increases sharply above natural background levels shortly after the onset of timber-harvest activities and reaches a peak that lags behind the peak harvest rate by approximately 25 years. Added to this period is the 10-year lag between harvest and peak in landslide frequency. This results in the overall 35-year lag time between timber harvest and channel response observed in the Grays River. Channel response to increased sediment yield may include aggradation, decreased sinuosity, increased bend curvature, and increased frequency of flooding and channel avulsions.

The onset of channel change may occur rapidly in alluvial reaches after timber harvest (10 years in the case of the Skokomish River) or several decades following harvest (35 years as indicated by the Grays River analysis). The magnitude of channel change tends to increase (despite the decline in sediment yield) with time due to the mobilization of sediment stored within the channel network during high-magnitude storm events. This reduction in sediment storage is often magnified by the removal of instream LWD and harvest of riparian forests that would otherwise supply LWD to the channel network.

Under this conceptual model and the inherent uncertainties involved with predicting sediment production and transport, channel adjustment could continue for at least 50 years after present timber-harvest activities. Channel response in the Grays River watershed is already 25 years out from the peak in harvest rate. Based on results of Stover and Montgomery (2001), channel adjustment in the lower Grays River could continue for at least another 25 years, under the current (2003) timber-harvest rate. Changes in timber-harvest rates and methods, as well as improvements in forest road construction, maintenance, and decommissioning could significantly reduce the period of channel instability and sediment loading.

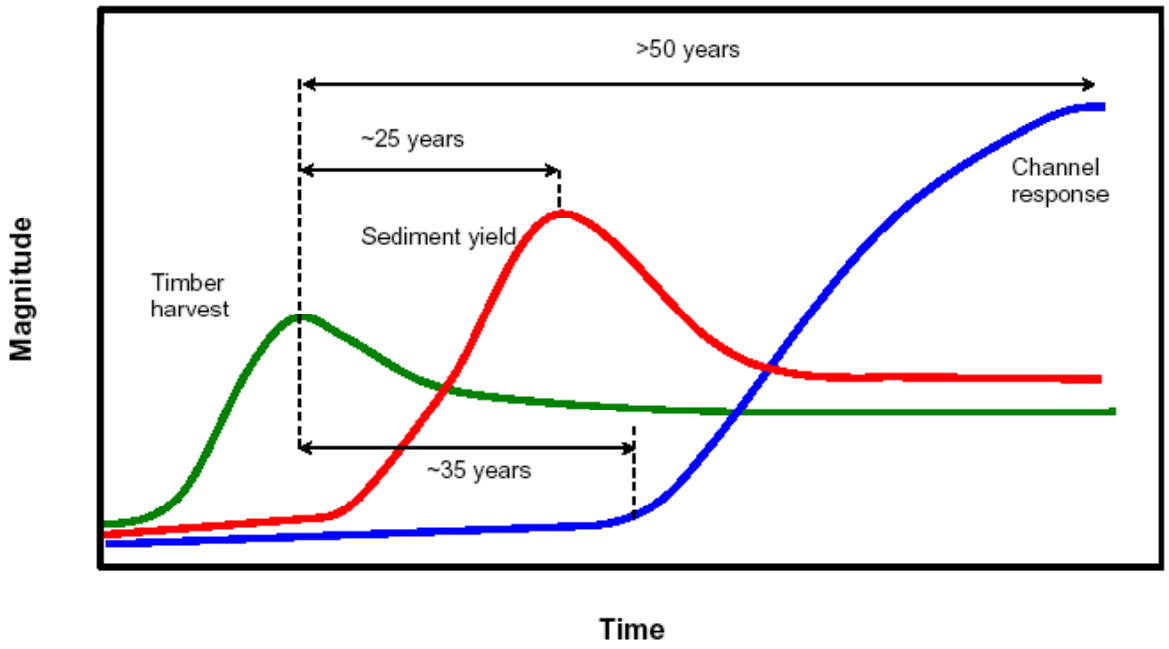


Figure 4.48. Conceptual Model of Temporal Trends in Sediment Yield to the Upper Grays River Channel Network and Channel Response within the Grays River Response Reach to an Assumed Historical Timber Harvest (Herrera 2005)

The magnitude of impacts on fish habitat caused by increased sediment yield and channel instability within the Grays River mainstem response reach will be determined by the ability of restoration efforts to counteract the destructive effects of past and ongoing land use activities within the upper watershed. Rivers with high sediment loads can support productive fish populations if they contain abundant instream LWD, which promotes pool formation, protective cover, and substrate diversity, as well as maintenance of an island-braided morphology with unrestricted channel migration into floodplain forests for self-sustaining wood recruitment.

5.0 Chum and Fall Chinook Salmon Spawning Analysis

Chum (*Oncorhynchus keta*) and fall Chinook (*Oncorhynchus tshawytscha*) salmon populations in the lower Columbia River have declined dramatically during the last century (LCFRB 2004). In 1999, the lower Columbia River evolutionarily significant units (ESUs) of these species were listed as threatened under the Endangered Species Act (ESA; NMFS 1999a, 1999b). Chum salmon populations have generally remained stable but low in recent years, with average runs in the lower Columbia River less than 1% of their historical run size (LCFRB 2004). Only two significant populations of chum salmon remain in this ESU, the Grays River and downstream from Bonneville Dam. Although the Chinook salmon ESU includes populations of both fall and spring runs and are distributed more widely than chum salmon, production in the ESU is largely hatchery driven with few naturally-producing wild populations, abundance trends for most populations are negative, and about half of the populations are very small (LCFRB 2004). The populations of these species below Bonneville Dam suffer adverse effects from operation of the dam and the Federal Columbia River Power System (NOAA 2004). This may reduce the likelihood of survival and recovery of this portion of the chum salmon ESU and increase the importance of the Grays River population.

Although currently stronger than others, the Grays River chum and fall Chinook salmon populations are a fraction of their historic sizes. The Grays River watershed was once noted for its large chum and fall Chinook salmon populations. The historic run size for chum salmon in the Grays/Chinook river population ranged from 8,000 to 14,000 fish (LCFRB 2004), with most spawning occurring in the Grays River. Between 1959 and 2002, an average of 1,120 chum salmon spawned in the Grays River, with fewer than 500 fish per year in approximately one-third of those years (T. Fisher, Fisher Fisheries Ltd., Oregon City, Oregon, unpublished data; Roler 2002). Fall Chinook salmon historic run size has been estimated at 1,500 to 10,000 fish, with only 100 to 300 individuals returning to spawn in recent years (LCFRB 2004). Channel clearing, floodplain agriculture and associated riparian clearing and diking, and logging and associated road construction on unstable ground in the watershed produced landslides, erosion and channel instability and loss of riparian function, and were largely responsible for the decline in Grays River chum and fall Chinook salmon populations (WCC 2001; WDFW 2001). In December 1999, an avulsion of the Grays River channel occurred in the immediate vicinity of the chum salmon spawning area, destroying an artificial spawning channel that supported approximately 25% (Hymer 1993) of the remaining chum salmon population, abandoning a portion of the Grays River that supported chum salmon spawning, and delivering tons of gravel and silt into downstream spawning areas. This event further destabilized spawning habitat and heightened the need to protect and restore remaining Grays River spawning habitat. The few stable chum salmon spawning areas that remain in the Grays River are subject to variable conditions that further threaten these declining populations.

The lack of stable spawning habitat has been identified as the primary physical limitation on chum salmon spawning in the Grays River (Roler 2002). Data are lacking on how and where this problem can best be addressed. The Grays River Watershed Assessment Project was initiated to address information needs for protection and restoration of ESA-listed chum and fall Chinook salmon. One of the primary objectives of the project was to gain a better understanding of chum and Chinook salmon spawning habitat requirements within the Grays River watershed. Research reported in this chapter was designed to address this information need. Our findings identify potential spawning habitat availability for chum and fall Chinook salmon, habitat characteristics limiting spawning, and redd capacity of the Grays River study

area. This information will help guide Grays River watershed management and restoration efforts for the conservation of chum and fall Chinook salmon in this watershed as well as other regions within the lower Columbia River Basin.

In this chapter, we focus on the significant chum salmon spawning areas in the Grays River watershed (Figure 5.1), which comprise the majority of fall Chinook salmon spawning area in the Grays River as well. These areas occur primarily in the Grays River as well as two tributaries immediately upstream of State Route (SR) 4 in Wahkiakum County, Washington. Chum salmon spawn in the mainstem Grays River from river kilometer (RKM) 15.3 to RKM 21.0, the West Fork Grays River (hereafter West Fork) downstream from the Grays River Hatchery (RKM 3.0), and Crazy Johnson Creek, a springbrook within the floodplain of the Grays River (hereafter Crazy Johnson) (LCFRB 2004; Figure 5.1). Within the study area, most of the fall Chinoook spawning in the Grays River occurs as well; some spawning also occurs downstream of the SR 4 Bridge outside of the study area.

This chapter consists of four main sections. In Section 5.1, we evaluate potential spawning habitat availability in the Grays River study area by chum and fall Chinook salmon at multiple flows, characterize spawning habitat quality, and estimate redd capacity of spawning reaches. In Section 5.2, we examine spawning consistency from 2002 through 2005, contrast spawning habitat characteristics of spawning and non-spawning areas during 2004 and 2005, and compare predictions made in Section 5.1 to actual spawning use. In Section 5.3, we evaluate hyporheic and geomorphic characteristics of chum salmon spawning areas. In Section 5.4, we present conclusions and implications of Chapter 5 for chum and fall Chinook salmon recovery in the Grays River.

5.1 Potential Spawning Habitat Availability and Redd Capacity for Chum and Fall Chinook Salmon

The Grays River study area is one of the most productive chum salmon spawning areas in the lower Columbia as well as an important fall Chinook salmon spawning area. The significance of this area to recovery of these species has been well-recognized. However, relatively little is known regarding spawning habitat availability and distribution within the Grays River study area. Effective identification of available spawning habitat for these species will guide habitat management and restoration actions for their recovery.

Habitat characteristics of salmonids including chum and fall Chinook salmon have been well-studied over the past 60 years. The importance of water velocity, depth and substrate for salmonid spawning has been widely recognized, and upper and lower spawning habitat limits for chum and fall Chinook salmon have been defined for these variables (e.g., Hale et al. 1985; Raleigh et al. 1986; Bjornn and Reiser 1991). However, these variables have limited ability to differentiate between salmonid spawning and non-spawning areas (Geist and Dauble 1998). The role of hyporheic exchange and associated stream bed morphology to spawning habitat selection of these species have been recognized (e.g., Leman 1993; Geist 2000; Geist et al. 2002) and used to improve salmonid spawning habitat models (Geist and Dauble 1998). Chum and fall Chinook salmon have generally been associated with areas of active hyporheic exchange characterized by elevated hyporheic temperatures, vertical flow, and well-oxygenated water (Bakkala 1970; Vincent-Lang 1984; Vining et al. 1985; Bjornn and Reiser 1991; Healey 1991; Salo 1991; Leman 1993; O'Brien 2006).

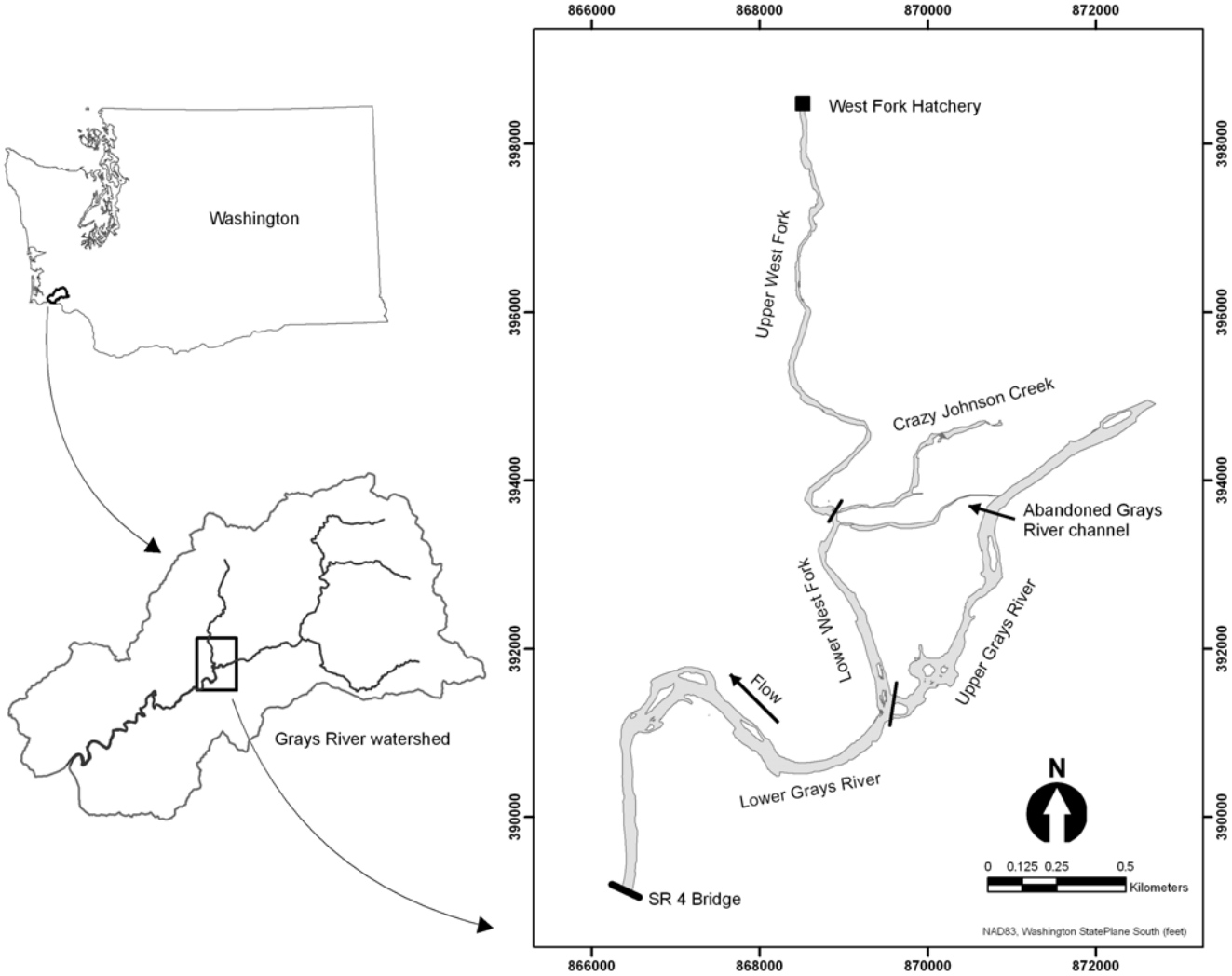


Figure 5.1. Grays River Study Area in Southwestern Washington, U.S.A. The five study reaches, Crazy Johnson Creek, Upper West Fork, Lower West Fork, Upper Grays, and Lower Grays are identified and labeled, with bold lines separating upper and lower reaches. In 1999, the Grays River channel avulsed, abandoning the western channel and what is now the lower West Fork channel and forming the new southern channel.

To identify areas suitable for chum and fall Chinook salmon in the Grays River, we developed spawning limits based on an extensive review of existing literature and a focused limits analysis of spawning habitat use in the study area during 2002 through 2005. From available habitat area, we quantified the potential redd capacity for each species. We evaluated current active channels as well as the channel abandoned during the 1999 avulsion and associated floodplain areas across a range of flows characteristic of spawning conditions in these streams. Our specific objectives were to address chum and fall Chinook salmon spawning information needs by 1) identifying where and how much potential spawning area was available in the Grays River study area, 2) characterizing spawning habitat quality, and 3) quantifying the theoretical redd capacity of the Grays River study area. In addition, detailed analysis of factors limiting spawning habitat availability is presented in Appendix H.

5.1.1 Methods

Because we hypothesized that study reaches would differ in chum salmon habitat use and availability, we stratified the study area based on reach differences in stream order, valley confinement (width), gradient, and hydrologic connectivity. These stream channel characteristics largely control the structure and function of stream channels and hyporheic exchange (Ward 1989; Rosgen 1994; Kasahara and Wondzell 2003) and are well recognized in stream classification systems (Vannote et al. 1980; Ward 1989; Stanford and Ward 1988, 1993; Rosgen 1994). We evaluated five active stream reaches (Table 5.1):

- upper Grays River (avulsion to West Fork confluence; hereafter upper Grays)
- lower Grays River (West Fork confluence to the SR 4 bridge; hereafter lower Grays)
- upper West Fork (West Fork Hatchery to Crazy Johnson)
- lower West Fork (Crazy Johnson to Grays River)
- Crazy Johnson.

Table 5.1. Study Area Stream Reach Characteristics

Reach	Reach Length (km)	Order[(a)]	Bankfull Width (m)[(b)]	Gradient (% slope)
Crazy Johnson	0.8	1, 2	18 ± 3	0.23
Upper West Fork	1.8	5	36 ± 10	0.45
Lower West Fork	0.8	5	46 ± 7	0.27
Upper Grays[(c)]	0.9	6	157 ± 53	0.63
Abandoned channel	0.8	6	41 ± 10	0.53
Lower Grays	1.9	6	76 ± 18	0.18

(a) Strahler (1964).
(b) Post-avulsion, estimated at the 10% exceedance flow (3924.9 cfs), ± 2 SE.
(c) Including inundated floodplain areas excluding lower Crazy Johnson, lower West Fork, and abandoned channel.

We present velocity, depth, substrate and hyporheic data primarily for the five focal study area stream reaches. The upper Grays River remains highly unstable following the 1999 channel avulsion (see Appendix H). The channel in this reach changed location between the bathymetry survey conducted in spring 2004, substrate data collection in July 2005, and hyporheic temperature data collection in November 2005. We include analyses of spawning habitat availability, quality and redd capacity only for

the portions of this reach that overlapped spatially. Current spawning habitat conditions likely differ somewhat from the conditions during which our data were collected. In addition, our bathymetry reflects the geomorphic channel bed. Beaver dams present in Crazy Johnson are often removed during winter flood flows but are usually in place when chum and fall Chinook salmon spawn. Our modeled velocity and depth estimates do not reflect conditions produced by these transitory beaver dams, and our substrate characterization reflects fine sediments collected in ponded areas. Although the channel abandoned during the 1999 avulsion no longer carries perennial flow, we evaluated its spawning habitat potential as well.

5.1.1.1 Habitat Characteristics

To obtain velocity and depth estimates, we used the Modular Aquatic Simulation System Two-Dimensional (MASS2) hydrodynamic model, a depth-averaged hydrodynamic and transport model for river and coastal systems (Perkins and Richmond 2004b, 2004c). MASS2 solves the depth-averaged equations for the conservation of mass and momentum in two dimensions using the finite-volume method. The channel system is represented on a multiblock grid using an orthogonal, curvilinear, full-transformation coordination system in a conservation form as described by Richmond et al. (1986). The model requires a river bottom elevation at each individual grid location. The bathymetric surface is developed using geographic information system (GIS) software from survey data and is sampled directly for each grid node location. GIS and grid generation utilities are used to facilitate the creation of a bathymetry surface and the computational grid. Boundary condition data (inflows and downstream stage) were provided from simulations by the Distributed Hydrology Soil Vegetation Model (DHSVM) described in Chapter 3. Model outputs for the entire spatial domain are in the form of GIS grids of wetted cells, water velocity, and water depth at approximately a 1-m^2 resolution.

Survey data for bathymetry development were collected using real-time kinematic (RTK) GPS and a single-beam, survey-grade fathometer. Vertical accuracy ranged from 4 to 12 cm. Survey transect width ranged from 16 to 37 m among and within reaches. To provide sufficient detail for modeling, transect data were collected at narrower widths in areas of channel change such as bends and gradient changes. Additional channel-influenced features such as side channels, dikes, levees, and floodplains were surveyed as necessary. Resulting data were combined with U.S. Geological Survey 10-m digital elevation model (DEM) data to produce a continuous three-dimensional surface. Anisotropic kriging was used to establish contours at a 10-m contour interval using an inverse distance weighting method to combine the topographic datasets. Bottom elevations from each computational mesh cell were extracted from the continuous three-dimensional surface to use as inputs to the MASS2 model. See Appendix F for additional details regarding bathymetric data collection and processing.

To develop stage-discharge relationships for MASS2 model calibration, piezometers were installed in the streambed at all spatial domain boundaries. An additional piezometer was installed above water at the West Fork hatchery to record atmospheric pressure. The river piezometers were installed such that the 30 cm-long screens were open to the river from 0 to 30 cm above the riverbed (see Appendix F for additional installation detail). Each piezometer was equipped with a Solinst (Solinst Canada Ltd., Georgetown, Canada) pressure transducer with an accuracy ±1.0–2.5 cm to record total barometric pressure. Pressure transducers were suspended from piezometer caps and located with their sensors near the bottom of the piezometer screens. At each location, the distance from the top of the piezometer to the sensor tip and the distance from the top of the piezometer to the adjacent riverbed were recorded. The elevations of the piezometer tops and adjacent riverbed were surveyed using either RTK GPS or laser

transit, allowing pressure readings to be converted to stream stage using a known datum. Correction of total barometric pressure by subtracting atmospheric pressure produced continuous readings of stream stage at each location. Stream discharge was estimated during 2004–05 at all model input locations using the cross-section procedure with velocity measured at 0.4 ft from the bed (Gallagher and Stevenson 1999). Stage-discharge relationships were developed for all MASS2 input locations and were used in model calibration to adjust Manning roughness coefficients (see Appendix F).

We used estimates produced by the DHSVM of the 90, 50, and 10% exceedance daily mean discharges for the chum salmon spawning season (November 1 – December 31) at all upstream boundary locations in the study area as input parameters to the MASS2 model. To obtain exceedance discharge values, three-hour model outflows were aggregated to daily flow, and spawning period daily flows were averaged for each year of the 42-year period of record. Exceedance values were derived from these spawning period flows. DHSVM estimates of the 90, 50, and 10% exceedance flows at the SR 4 Bridge were 349.4, 1058.6, and 3924.9 cfs, respectively. Velocity and depth grids were produced from MASS2 for the entire spatial domain for each of the input flows. Our analyses used depth and velocity outputs from the 50% exceedance flow model run except when examining spawning habitat availability and quality at all three flows.

Substrate grainsize data were collected for every geomorphic feature longer than 10 m in each of the study reaches as well as the abandoned upper Grays River channel. Feature length was estimated, and feature boundaries were recorded with a GPS when possible; dense canopy cover on the West Fork and Crazy Johnson precluded GPS positioning in some cases. We evaluated substrate using the pebble count method in which 100 substrate particles are randomly selected and measured (secondary diameter) per geomorphic feature (e.g., riffle, pool) (Wolman 1954). One hundred particles adequately characterize the particle size distribution of an entire feature if particle selection is random and spatially distributed across the feature to avoid bias (Wolman 1954). We selected general pebble collection locations based on a systematic approach of three to six transects, depending on the feature size, spaced evenly throughout the geomorphic feature. Pebbles were collected every 1 m or 2 m along each transect. At each pebble collection location, pebbles were collected randomly. All grainsize measurements were made with a metal template or gravelometer containing square openings in 1/2-phi size classes from 128 mm (−7 phi) to 2 mm (−1 phi) (Hey and Thorne 1983). A cumulative grain size distribution was developed for each pebble count, and the d_{50} (the grain size in millimeters of which 50% of particles are finer) was identified for each pebble count. The d_{50} is a widely used substrate metric in salmonid spawning studies (Kondolf and Wolman 1993). A GIS shapefile of d_{50} values was developed by hand digitizing geomorphic features into polygon coverages of the wetted channel for each flow.

We characterized the hyporheic environment by quantifying the difference between river and hyporheic temperatures ($\Delta T = T_H - T_R$). The difference between hyporheic and river temperatures has been used to characterize chum and fall Chinook spawning areas (Geist et al. 2002), with chum salmon spawning associated with hyporheic temperatures warmer than river water and therefore positive ΔT values. Areas with positive ΔT characteristics were identified and used together with velocity, depth, and substrate classifications to identify areas that appeared suitable for chum salmon spawning. Additional hyporheic characteristics associated with chum salmon spawning, including hyporheic temperature, vertical hydraulic gradient (VHG), hydraulic conductivity (K, the flow rate through porous media), and q (specific discharge, the product of VHG and K), are examined in detail in Section 5.3.

During November and December 2005, hyporheic and river water temperature were measured at a total of 634 points—386 points on the Grays River, 227 points on the West Fork, and 66 points on Crazy Johnson. River and hyporheic river temperature were collected every 12.5 m longitudinally along the thalweg of each reach. Temperature data were collected similarly from the abandoned Grays River channel. To compare temperature data to modeled velocity and depth data, we extrapolated temperature data for temperature point locations to the area bounded laterally by the wetted channel perimeter at the 50% exceedance flow and longitudinally to channel cross sections perpendicular to the thalweg located midway between adjacent temperature point locations. During the sampling period, discharge ranged from 255 to 468 cfs (approximately the 90% exceedance flow) at the Grays River gauge station 1.5 km downstream from the study area (Washington Department of Ecology Site 25B060).

At each temperature location, a customized temperature probe was driven into the riverbed following the methods of Geist et al. (2002). A thermistor (Model 44033, Omega Engineering, Stamford, Connecticut) was mounted in a 1.8-cm–inside-diameter drive rod with an extension cable exiting the upper end of the drive rod and attached to a temperature reader (Model 866C, Omega Engineering, Stamford, Connecticut). The lower 20 cm of the drive rod were perforated with 3-mm-diameter holes at the lower end to allow water infiltration. A post-pounder was used to pound the drive rod into the bed such that the thermistor tip was 30 cm into the bed. After temperature equilibration (2–20 minutes), hyporheic temperature was recorded, the drive rod was extracted from the bed, and river water temperature was recorded. The thermistor and temperature indicator had stated accuracies of ±0.1°C and ±0.3%, respectively. The resolution of the temperature indicator was 0.1°C.

5.1.1.2 Spawning Habitat Availability, Quality, and Redd Capacity

We conducted an extensive review of the literature to identify empirical estimates of velocity, depth, and substrate spawning limits for chum and fall Chinook salmon (Appendix G). Habitat suitability indices (HSIs) for both species for all three variables have been developed for use in instream flow studies (Hale et al. 1985; Raleigh et al. 1986) and have been used successfully for estimating fall Chinook salmon spawning habitat availability (Hanrahan et al. 2004). However, considerably more information is now available for both species than was originally used to develop the HSI limits. Further, most of the available data including those used to develop the HSI are from large river systems and may not apply to the Grays River. We disregarded empirical evidence of spawning use in rare instances if the habitat conditions appeared to represent extreme outliers.

We summarized velocity, depth, substrate, and ΔT in known Grays River chum and fall Chinook salmon spawning areas. Spawning surveys in the study area were conducted by the Pacific States Marine Fisheries Commission (2002–2004) and the Washington Department of Fish and Wildlife (2005). In 2002 and 2003, chum salmon locations of individual and mass redds were recorded with a global positioning system (GPS) weekly throughout the spawning season for each species. In 2004 and 2005, each study reach was divided into 50-m-long sections (referred to hereafter as sampling sections) and the numbers of carcasses and live fish actively engaged in spawning were recorded weekly. Carcasses were marked to avoid recounting. In 2004, spawning count data were obtained for all reaches during only one week of the chum salmon spawning season. In 2003 and 2005, fall Chinook salmon redd locations were recorded with a GPS.

We defined polygons with 1-m-width buffers around chum salmon individual redds located in 2002 and 2003 and 3-m-width buffers around mass redds located in 2002. Individual redds were not included

unless they occurred within 5 m of a second redd or mass redd. Mass redd polygons identified in 2003 that extended beyond the wetted channel were reduced in scope to the wetted channel boundary. Mass redd polygons were not used if they were large (>150 m long). Thus, digitized spawning polygons represent areas where we had confidence in the location and extent of spawning occurrences, and some spawning areas were not assessed. Average velocity, depth, substrate (d_{50}), and ΔT were calculated for these polygons and contrasted to the remaining non-spawning area. MASS2 velocity and depth outputs were obtained from the model output for the 50% exceedance flow. Chum salmon spawning habitat use in 2004 and 2005 was assessed by identifying sampling sections in which spawning use was concentrated and redds represented a significant portion of each section. Specifically, we identified sampling sections in 2005 in which 25% of a sampling section area could have been redds. These sections were identified by multiplying half the total spawning fish count for the section by the area of one redd (2.26 m^2; Burner 1951). The proportion of the total sampling section area represented by potential redd area was then calculated. Although we estimated the percentage of each section that could have been redds, these percentages may be overestimates because of redd superimposition. Because we did not have total spawning season counts for all sections in 2004, we selected the same number of sections in 2004 as we did in 2005, that contained the most concentrated spawning during the week for which we had data and identified spawning use limits analogous to the 2005 25% use limit.

For fall Chinook salmon, we examined spawning habitat characteristics for the individual redd locations recorded in 2003 and 2005. Individual and mass redd chum salmon polygons and fall Chinook salmon redd locations were used to examine differences in habitat variables between species. We conducted Wilcoxon rank sum tests for differences in velocity, depth, substrate, and ΔT. This test was used to address data non-normality. Chum and fall Chinook salmon habitat use is examined in greater detail in Section 5.2.

We compared habitat use from the literature and actual use to develop use limits for each species for velocity, depth, and substrate. We developed two sets of limits (Table 5.2), those within which chum and fall Chinook salmon have been documented to spawn, and within these limits, a more restrictive range in which these species were most likely to spawn (see next paragraph). Although we examined actual spawning habitat use by chum and fall Chinook salmon in the study area, we relied primarily on the literature-based limits we established to finalize our limits because 1) both species are currently under-populated relative to historic populations and current habitat use may not reflect potential use; and 2) our 2004–2005 spawning estimates were based on 50-m sampling sections, of which very few contain sufficient redds to represent a significant portion of the section. In contrast, literature values were usually based on habitat conditions of individual redd locations. We used the outer spawning use limits to quantify total potential spawning area for each species. In addition, we identified habitat characteristics limiting spawning area for each species (see Appendix H). Once potential spawning habitat area was quantified, we applied literature values for redd size and density to estimate the total potential redd capacity of the Grays River study area.

The more restrictive limits (referred to hereafter as probable use) were developed from literature sources reporting confidence intervals, standard deviations, or other metrics suggesting that this category represented habitat use by 65–80% of individuals within a spawning population (Appendix G). Probable use limits were used to distinguish potential high- versus low-quality habitat. To develop an overall quality rating for spawning habitat, we scored habitat as 2, 1, or 0 for each variable (velocity, depth, and

Table 5.2. Spawning Limits for Chum and Fall Chinook Salmon

Variable	Beyond Lower Limit	Use Limit Category			Beyond Upper Limit
		Within Lower Limit	Probable Use	Within Upper Limit	
Chum Salmon					
Velocity (cm/sec)	None	<12	12–100	100–110	>110
Depth (cm)	<12	12–18	18–50	>50	None
Substrate (d_{50}; mm)	None	<14	14–42	42–62	>62
Fall Chinook Salmon					
Velocity (cm/sec)	<15	15–30	30–100	100–130	>130
Depth (cm)	<15	15–20	20–70	>70	None
Substrate (d_{50}; mm)	<11	11–21	21–48	48–78	>78

substrate) if it was probable, within use limits but not probable, or beyond use limits, respectively. We calculated the geometric mean of the three scores to obtain an overall spawning quality score, where five potential scores are possible:

- 0.0 = lowest quality (at least one habitat variable score is zero)
- 1.0 = all three habitat variable scores are 1
- 1.3 = two scores of 1, one score of 2
- 1.6 = one score of 1, two scores of 2
- 2.0 = highest quality (all three habitat variable scores are 2).

The geometric mean is a useful metric because it allows for limited compensation among variables but has a value of zero if any of the three habitat variable scores is zero (beyond use limits) (USFWS 1981; Hanrahan et al. 2004).

Spawning habitat availability was determined for chum salmon in two steps, by limiting total potential spawning area (wetted area) by velocity and depth estimates and empirical d_{50} substrate data by applying the spawning limits criteria, and then further restricting potential spawning area by ΔT. We limited spawning area by ΔT in a separate step because although the role of hyporheic exchange with river water to spawning salmon site selection has been recognized, specific hyporheic temperature limits desired by salmon have not been defined. Total usable area was calculated per stream reach and GIS attribute files were exported to a spreadsheet for further analysis and summarization. Because actual use of the Grays River study area by fall Chinook salmon did not indicate a spawning association with ΔT, we limited suitable habitat for this species by velocity, depth, and substrate but not ΔT.

For chum salmon, we classified ΔT in a GIS classification environment using a combination of the Jenks natural breaks method (Jenks 1967), examining the range of values and the relationship between river and hyporheic temperature, and considering the accuracy of the thermistors and electronic readers used to collect the data. Based on this analysis, we determined that ΔT values less than 0.9°C could not be distinguished from zero. Therefore, we considered sites with ΔT greater than 0.8°C to have potential desirability for chum salmon. Because the entire length of Crazy Johnson is dominated by hyporheic flow with elevated temperatures, and river water temperatures are considerably warmer than at any other location in the study area, ΔT was ineffective at identifying areas with elevated hyporheic temperatures.

We examined hyporheic temperature and reclassified some areas with ΔT values between 0.0 and 0.8°C as chum spawning areas if hyporheic temperatures were greater than 10.4°C, a temperature value higher than any observed river temperature in Crazy Johnson. In the West Fork and Grays River, river waters generally reflected air temperature and were minimally influenced by hyporheic temperatures.

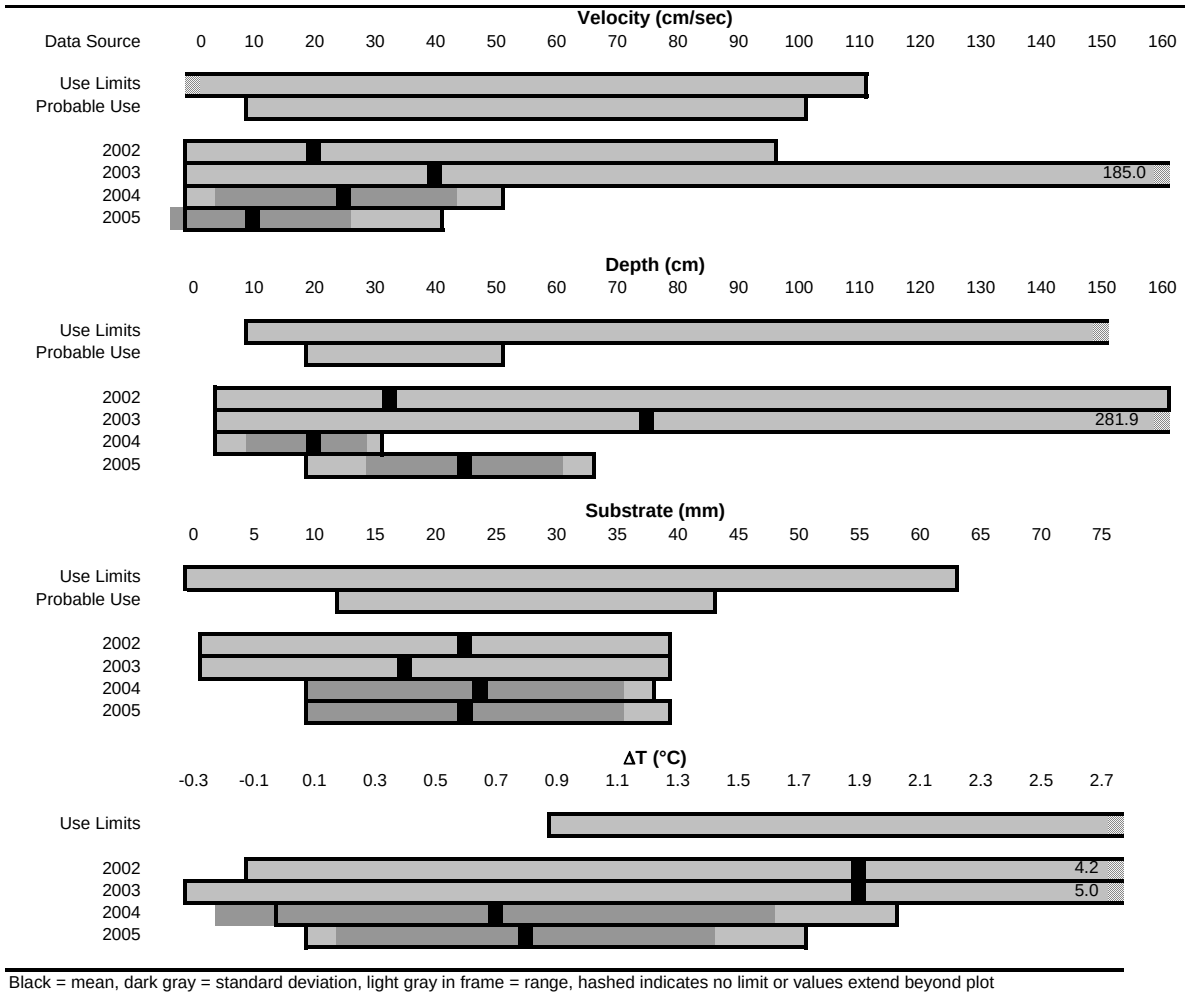
Spawning area availability in the upper Grays River channel abandoned during the 1999 avulsion was evaluated by applying the same limits restrictions methods as were used for the five focal study reaches. However, only substrate and temperature data were available for this reach because the channel was disconnected from the flowing Grays River at the 50% exceedance flow used in analyses, so no MASS2 velocity or depth estimates could be produced. Channel boundaries were approximated using a constant width estimated visually from the bathymetry and GPS locations for substrate and hyporheic data to create raster grids for each variable in a GIS. Although total area of the channel is likely overestimated, area percentages are proportional to the distribution of linear substrate and hyporheic data inputs and are therefore relatively insensitive to actual channel area.

Salmon redd size and spacing has been used successfully to estimate total redd capacity for a defined spawning habitat area (Hanrahan et al. 2004; O'Brien 2006). Burner (1951) quantified chum and fall Chinook salmon redd size in lower Columbia River tributaries, including the Grays River for chum salmon and the Toutle River for fall Chinook salmon. He found chum and fall Chinook salmon redd sizes in these rivers to be 2.62 m^2 and 6.5 m^2, respectively. He also recommended that redd density should be estimated as four times the average redd size to incorporate inter-redd spacing. We estimated minimum and maximum redd capacity by dividing available spawning habitat by redd size with no inter-redd spacing and with that recommended by Burner (1951) for each species. We estimated redd capacity for spawning habitat limited by only velocity, depth, and substrate (three variables), and with the further limitation of hyporheic behavior (four variables; chum salmon). We provide redd capacity estimates for total spawning area available and total area of high-quality spawning habitat to provide a range of estimates acknowledging the uncertain use of lower-quality habitats. The highest quality habitats (quality code 2.0) is within the probable use range of all three habitat variables (velocity, depth, and substrate) and may provide a more accurate estimate of spawning area used by chum and fall Chinook salmon.

5.1.2 Results

5.1.2.1 Spawning Habitat Use by Chum and Fall Chinook Salmon

Spawning habitat use by chum salmon in 2002 and 2003 was characterized for 37 and 31 spawning polygons, respectively. Spawning habitat was characterized in all reaches except the upper Grays in 2002 and the upper Grays and lower West Fork in 2003. In these reaches, no spawning polygons were available; therefore, no spawning areas could be defined with certainty. Spawning use was generally within established limits but included some areas that were shallower, faster, or with ΔT values lower than use limits (Figure 5.2; see Appendix H for additional detail). Modeled velocity in spawning areas ranged from 0.0 to 194.2 cm/sec in 2002 at the 90% exceedance flow and from 0.0 to 233.3 cm/sec in 2003 at the 50% exceedance flow, with velocity in portions of one spawning area located at the confluence of the upper Grays and lower West Fork reaches extending beyond the upper velocity limit in 2003. Portions of some spawning areas in each reach were more shallow than the 12-cm limit, with depths as low as 3.6 cm modeled in 2002 and 4.4 cm in 2003 at the 90% and 50% exceedance flows, respectively. Depths as shallow as this are unlikely to be used by chum salmon and are more likely a result of model or spawning polygon inaccuracy. No spawning areas exceeded substrate limits in either



Black = mean, dark gray = standard deviation, light gray in frame = range, hashed indicates no limit or values extend beyond plot

Velocity and depth estmates for 2002, 2004 obtained from MASS2 model outputs for the 90% exceedance flow, for 2003, 2005 from the 50% exceedance flow output

Figure 5.2. Chum Salmon Spawning Habitat Limits and Habitat Characteristics of Spawning Areas During 2002-2005

year. ΔT ranged from −0.1°C to 4.2°C in 2002 and −0.3°C to 5.0°C in 2003, with portions of some spawning areas in each reach lower than the 0.9°C limit we set for chum salmon. It is not known whether chum salmon will spawn in areas that do not have elevated hyporheic temperatures or if these areas are a result of limitations in the resolution of spawning or hyporheic temperature data.

In 2004 and 2005, spawning use by chum salmon was within established limits for velocity, depth, and substrate spawning limits in most cases. In 2005, 5 sampling sections contained sufficient chum salmon redds to represent at least 25% of sampling section area. These sections were in Crazy Johnson (4 sections) and the lower West Fork (1 section). In these sections, velocity, depth, and substrate ranged from 1.2 to 37.5 cm/sec, 19.8 to 64.0 cm, and 2.5 to 21.2 mm, respectively (Figure 5.2). In 2004, the 5 sections with the highest percentage of redd area were in the lower West Fork (1 section), Crazy Johnson (3 sections), and the upper West Fork (1 section). Velocity, depth, and substrate in these sections ranged from 0.3 to 51.5 cm/sec, 6.1 to 30.2 cm, and 10.2 to 38.0 mm, respectively (Figure 5.2). In 2004, one section in Crazy Johnson had an average depth of 6.10 cm, shallower than the use limit of 12 cm.

Whether spawning fish used depths this shallow or concentrated in areas that were deeper than average is not known. No sections had average velocities or substrates that exceeded use limits during either year. ΔT across all sections ranged from 0.0°C to 2.0°C. ΔT extended beyond the established spawning limit (ΔT > 0.8°C) primarily in Crazy Johnson, in which both river and hyporheic temperatures were elevated by warm water sources upstream relative to river water temperatures in adjacent reaches.

Habitat characteristics of 17 fall Chinook salmon redds were examined, two in 2003 and 15 in 2005. Redds were located in the upper West Fork (9 redds) and the lower Grays (8 redds). Velocity, depth, and substrate at these redds ranged from 51.4 to 115.2 cm/sec, 29.6 to 160.2 cm, and 7.3 to 28.3 mm, respectively (Figure 5.3). Spawning use by fall Chinook salmon was within established velocity, depth, and substrate spawning limits at all but one redd. At this redd, substrate extended beyond use limits, with average substrate d_{50} finer than the limit. ΔT across all redds ranged from −1.0°C to 4.3°C. Wilcoxon signed rank test results indicated that fall Chinook salmon spawning areas were significantly faster ($P < 0.0001$, difference = 40.6 cm/sec) and deeper ($P < 001$, difference = 37.3 cm) than chum salmon spawning areas (Figure 5.4) and were not significantly different in substrate size or ΔT.

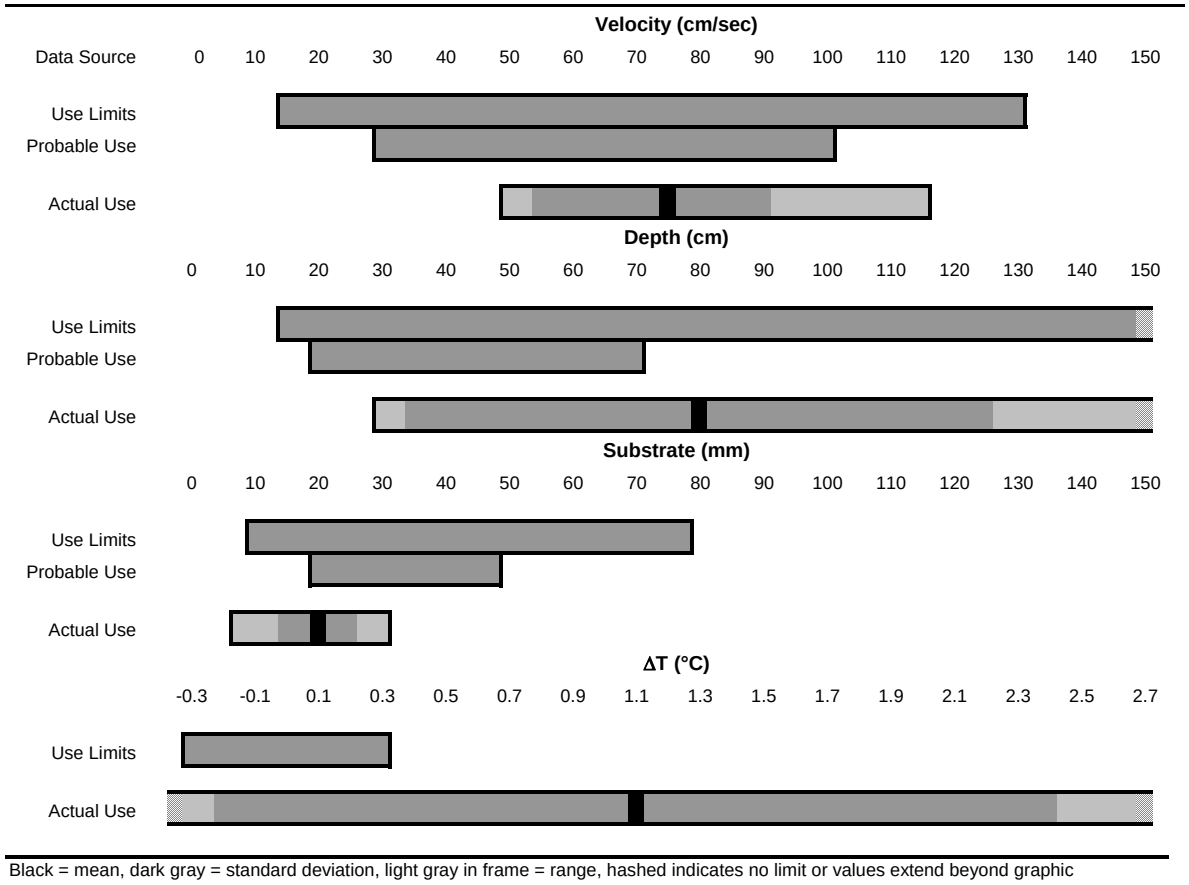


Figure 5.3. Fall Chinook Salmon Spawning Habitat Limits and Redd Habitat Characteristics in the Grays River Study Area During 2003 and 2005

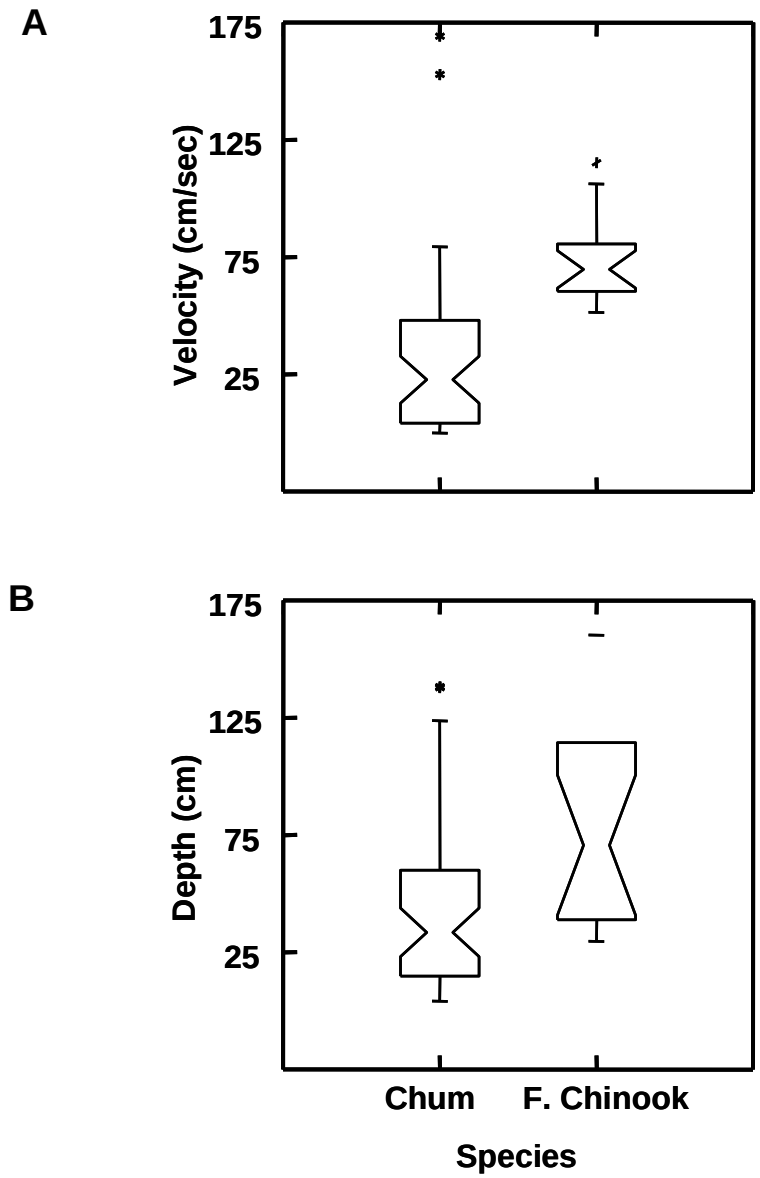


Figure 5.4. Comparison of A) Velocities (cm/sec) and B) Depths (cm) Used by Chum and Fall Chinook Salmon in the Grays River Study Area. Differences are significant at $P < 0.0001$ and $P < 0.001$, respectively.

5.1.2.2 Spawning Habitat Availability Limited by Velocity, Depth, and Substrate

Of the total wetted area, 61.6% (12.22 ha) is within spawning limits for both species, 27.9% (5.54 ha) is within only chum salmon limits, 2.7% (0.54 ha) is within only fall Chinook salmon limits, and 7.8% (1.55 ha) is beyond the spawning limits for both species (Table 5.3; Figure 5.5). Total spawning area available for chum and fall Chinook salmon is 17.76 ha (89.5%) and 12.75 ha (64.2%), respectively. See Appendix H for additional analyses regarding velocity, depth, and substrate limitations on spawning habitat availability.

Table 5.3. Spawning Area Availability for Chum and Fall Chinook Salmon by Stream Reach Based on Velocity, Depth, and Substrate

Stream Reach	Total Area	Non-Spawning		Chum Only		Fall Chinook Only		Both Species		Chum Total		Fall Chinook Total	
	ha	ha	%	ha	%	ha	%	ha	%	ha	%	ha	%
Total	19.85	1.55	7.8	5.54	27.9	0.54	2.7	12.22	61.6	17.76	89.5	12.75	64.2
Crazy Johnson	0.92	0.12	12.6	0.76	82.5	0.00	0.0	0.05	4.9	0.81	87.4	0.05	5.4
Upper West Fork	3.18	0.16	5.2	0.64	20.2	0.11	3.4	2.26	71.2	2.90	91.4	2.37	74.5
Lower West Fork	2.29	0.23	10.1	0.16	7.1	0.02	1.1	1.87	81.7	2.03	88.8	1.90	82.5
Upper Grays	4.10	0.63	15.3	1.69	41.1	0.19	4.7	1.59	38.8	3.28	80.0	1.79	43.6
Lower Grays	9.36	0.41	4.4	2.29	24.5	0.21	2.2	6.45	68.9	8.74	93.3	6.66	71.0

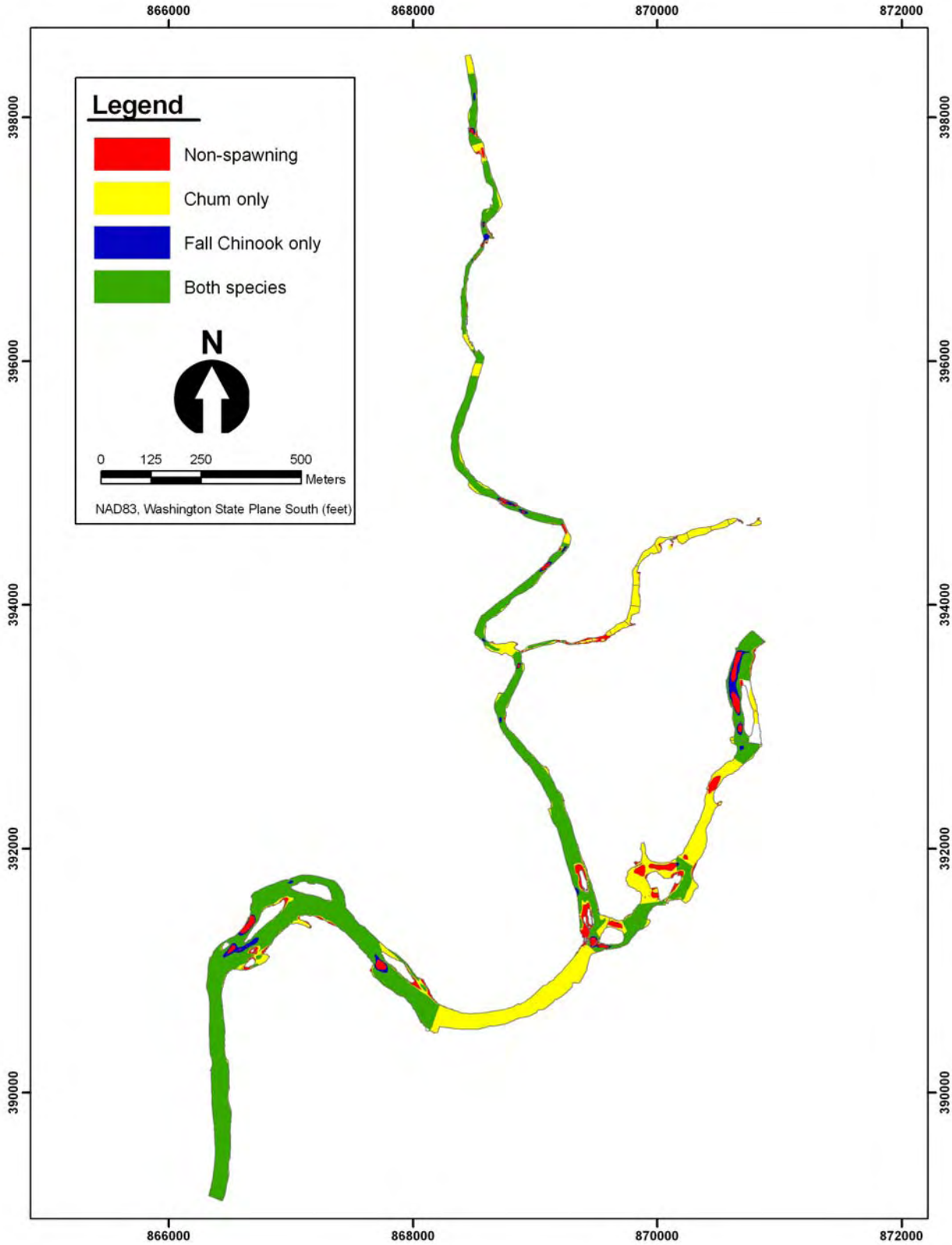


Figure 5.5. Chum and Fall Chinook Salmon Spawning Areas Based on Velocity, Depth, and Substrate (d_{50}) Limits

Stream reaches differ somewhat in spawning habitat availability, with Crazy Johnson differing the most from the other reaches (Table 5.3, Figure 5.5). Most of Crazy Johnson (82.5%) is suitable only for chum salmon. Crazy Johnson, the lower West Fork, and the upper Grays have more area that is unsuitable for either species (12.6, 10.1, 15.3%, respectively), whereas the upper West Fork and lower Grays have less area that is unsuitable (5.2% and 4.4%, respectively). The majority of all reaches are suitable for chum salmon (80.0% to 93.3%). Less than half of Crazy Johnson and the upper Grays is suitable for fall Chinook salmon (5.4% and 43.7% suitable, respectively).

Velocity, depth, and substrate do not appear to be limiting to chum salmon, with only 6.3, 4.4, and 0.0% of the total wetted area beyond use limits, respectively (Table 5.4, Figure 5.6). Velocity is within spawning limits for chum salmon in 84.1% (upper Grays) to 100.0% (Crazy Johnson) of the total area available. Because there is no lower velocity limit, all area beyond spawning use limits is of higher velocity than can be used by chum salmon. Depth is within chum salmon spawning limits in 84.6% (Crazy Johnson) to 97.5% (upper Grays) of the total area available. Because there is no upper depth limit, all of the area beyond use limits is shallower than can be used by chum salmon. All of the substrate in the Grays River study area is within spawning limits for chum salmon.

Spawning habitat is more limited by velocity, depth, and substrate for fall Chinook salmon than for chum salmon, with 11.6, 7.6, and 25.4%, respectively, beyond use limits (Table 5.5, Figure 5.7). Most (88.4%) of the Grays River study area is within fall Chinook salmon spawning velocity limits. Area within fall Chinook salmon spawning velocity limit ranges from 18.8% (Crazy Johnson) to 95.3% (lower West Fork), with velocities that are too slow for fall Chinook salmon in most of Crazy Johnson. Of the unsuitable area in other reaches, more area is too slow in the lower Grays and both reaches of the West Fork and too fast in the upper Grays. Across the study area, 92.4% is within fall Chinook salmon spawning depth limits. Among reaches, available area within depth limits ranges from 76.7% (Crazy Johnson) to 95.2% (lower Grays). In all reaches, relatively little area is too shallow. Unlike chum salmon, spawning habitat for fall Chinook salmon is most limited by substrate. Across the study area, 74.6% of the Grays River is within fall Chinook salmon spawning substrate limits, ranging from 16.4% (Crazy Johnson) to 100.0% (lower West Fork).

5.1.2.3 Chum Salmon Spawning Habitat Availability Limited by All Habitat Variables

Hyporheic temperature ranged from 5.0°C to 11.8°C (Figure 5.8a). Colder temperatures occurred primarily in the upper Grays reach, likely due to the later sampling date in this reach (two weeks later). Areas where warm hyporheic water was found include Crazy Johnson, the lower end of the lower Grays, and smaller areas in the other three reaches (Figure 5.8a). Almost the entire length of Crazy Johnson had elevated hyporheic temperatures.

River water temperatures ranged from 5.1°C to 10.2°C (Figure 5.8b). Temperatures in Crazy Johnson were considerably warmer than anywhere else in the study area (Figure 5.8b). Crazy Johnson river temperatures ranged from 9.5°C to 10.2°C whereas the maximum river water temperature found in the other reaches was 7.7°C. Crazy Johnson and/or warm hyporheic upwelling in the West Fork at and below the Crazy Johnson confluence increased river temperatures in the lower West Fork by approximately 0.5°C, which in turn appeared to increase temperatures in the lower Grays (by approximately 0.6°C) just below their confluence. We also observed warming trends associated with elevation and time of sampling, associated with air temperature. These secondary patterns had relatively minor influences on overall temperature patterns in the study area.

Table 5.4. Spawning Habitat Limitation by Habitat Characteristic for Chum Salmon

Stream Reach	Not Used		Within Lower Limit		Probable Use		Within Upper Limit		Not Used		Total Within Limits	
	ha	%	ha	%	ha	%	ha	%	ha	%	ha	%
	Velocity (cm/sec)											
	---		<12		12–100		100–110		>110		0–110	
Crazy Johnson			0.78	78.5	0.21	21.5	0.00	0.0	0.00	0.0	0.99	100.0
Upper West Fork			0.20	6.2	2.74	85.0	0.09	2.8	0.19	6.0	3.03	94.0
Lower West Fork			0.06	2.7	2.19	95.2	0.02	0.9	0.03	1.2	2.27	98.8
Upper Grays			0.19	4.5	3.07	74.0	0.24	5.7	0.66	15.9	3.50	84.1
Lower Grays			0.27	2.9	8.63	91.1	0.17	1.8	0.40	4.2	9.07	95.8
Total			1.50	7.4	16.84	83.6	0.52	2.6	1.28	6.3	18.86	93.7
	Depth (cm)											
	<12		12–18		18–50		>50		---		>12	
Crazy Johnson	0.15	15.4	0.13	12.8	0.37	38.0	0.33	33.8			0.83	84.6
Upper West Fork	0.10	3.0	0.24	7.6	1.63	50.5	1.25	38.9			3.13	97.0
Lower West Fork	0.23	10.0	0.21	9.2	1.23	53.4	0.63	27.4			2.07	90.0
Upper Grays	0.17	4.1	0.26	6.2	1.38	33.3	2.34	56.4			3.98	95.9
Lower Grays	0.24	2.5	0.42	4.5	2.18	23.1	6.63	70.0			9.23	97.5
Total	0.88	4.4	1.26	6.3	6.80	33.8	11.18	55.6			19.24	95.6
	Substrate (d_{50}; mm)											
	---		<14		14–42		42–62		0–62		>62	
Crazy Johnson			0.80	83.6	0.16	16.4	0.00	0.0	0.00	0.0	0.96	100.0
Upper West Fork			1.23	38.8	1.94	61.2	0.00	0.0	0.00	0.0	3.17	100.0
Lower West Fork			1.00	43.6	1.25	54.6	0.04	1.7	0.00	0.0	2.28	100.0
Upper Grays			1.98	48.0	1.15	27.9	0.99	24.1	0.00	0.0	4.13	100.0
Lower Grays			2.54	27.0	6.74	71.5	0.14	1.5	0.00	0.0	9.42	100.0
Total			7.55	37.8	11.23	56.3	1.17	5.9	0.00	0.0	19.95	100.0

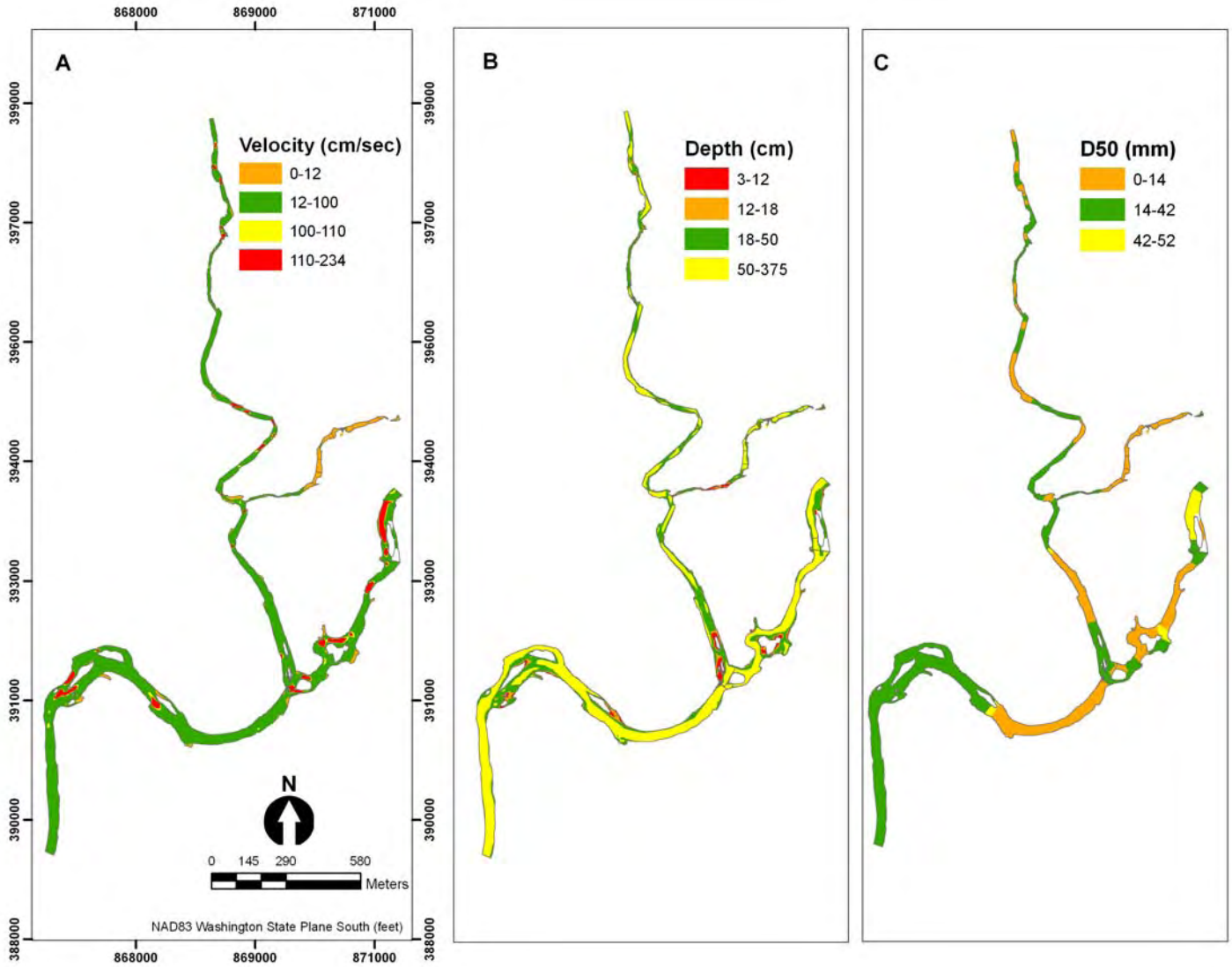


Figure 5.6. Chum Salmon Spawning Limits. A) Velocity, cm/sec; (B) Depth, cm; (C) Substrate, d_{50}; mm. Colors refer to potential use categories: Not used (red), Within lower limits (orange), Within upper limits (yellow), and Probable use (green).

Table 5.5. Spawning Habitat Limitation by Habitat Characteristic for Fall Chinook Salmon

Stream Reach	Not Used		Within Lower Limit		Probable Use		Within Upper Limit		Not Used		Total Within Limits	
	ha	%	ha	%	ha	%	ha	%	ha	%	ha	%
	Velocity (cm/sec)											
	<15		15–30		30–100		100–130		>130		15–130	
Crazy Johnson	0.80	81.2	0.08	7.7	0.11	11.0	0.00	0.0	0.00	0.0	0.18	18.8
Upper West Fork	0.26	8.1	0.47	14.7	2.20	68.4	0.22	6.7	0.07	2.1	2.89	89.8
Lower West Fork	0.11	4.6	0.46	20.1	1.68	73.2	0.05	2.0	0.00	0.2	2.19	95.3
Upper Grays	0.23	5.6	0.32	7.6	2.71	65.3	0.60	14.4	0.30	7.1	3.63	87.3
Lower Grays	0.38	4.0	0.79	8.4	7.72	81.6	0.38	4.0	0.19	2.0	8.90	94.0
Total	1.78	8.8	2.12	10.5	14.43	71.7	1.24	6.2	0.56	2.8	17.79	88.4
	Depth (cm)											
	<15		15–20		20–70		>70		---		>15	
Crazy Johnson	0.23	23.3	0.08	8.3	0.47	47.6	0.20	20.8			0.76	76.7
Upper West Fork	0.21	6.6	0.23	7.0	2.17	67.3	0.61	19.0			3.01	93.4
Lower West Fork	0.34	14.7	0.20	8.5	1.63	71.0	0.13	5.8			1.96	85.3
Upper Grays	0.29	7.0	0.23	5.5	2.02	48.7	1.61	38.7			3.86	93.0
Lower Grays	0.45	4.8	0.35	3.7	3.26	34.4	5.41	57.2			9.02	95.2
Total	1.52	7.6	1.08	5.4	9.55	47.4	7.97	39.6			18.60	92.4
	Substrate (d_{50}; mm)											
	<11		11–21		21–48		48–78		>78		11–78	
Crazy Johnson	0.80	83.6	0.04	4.4	0.11	12.0	0.00	0.0	0.00	0.0	0.16	16.4
Upper West Fork	0.44	13.7	1.68	53.1	1.05	33.2	0.00	0.0	0.00	0.0	2.74	86.3
Lower West Fork	0.00	0.0	1.17	51.3	1.11	48.7	0.00	0.0	0.00	0.0	2.28	100.0
Upper Grays	1.87	45.4	0.50	12.2	0.96	23.3	0.79	19.1	0.00	0.0	2.25	54.6
Lower Grays	1.96	20.8	1.19	12.7	6.27	66.6	0.00	0.0	0.00	0.0	7.46	79.2
Total	5.07	25.4	4.59	23.0	9.51	47.6	0.79	3.9	0.00	0.0	14.89	74.6

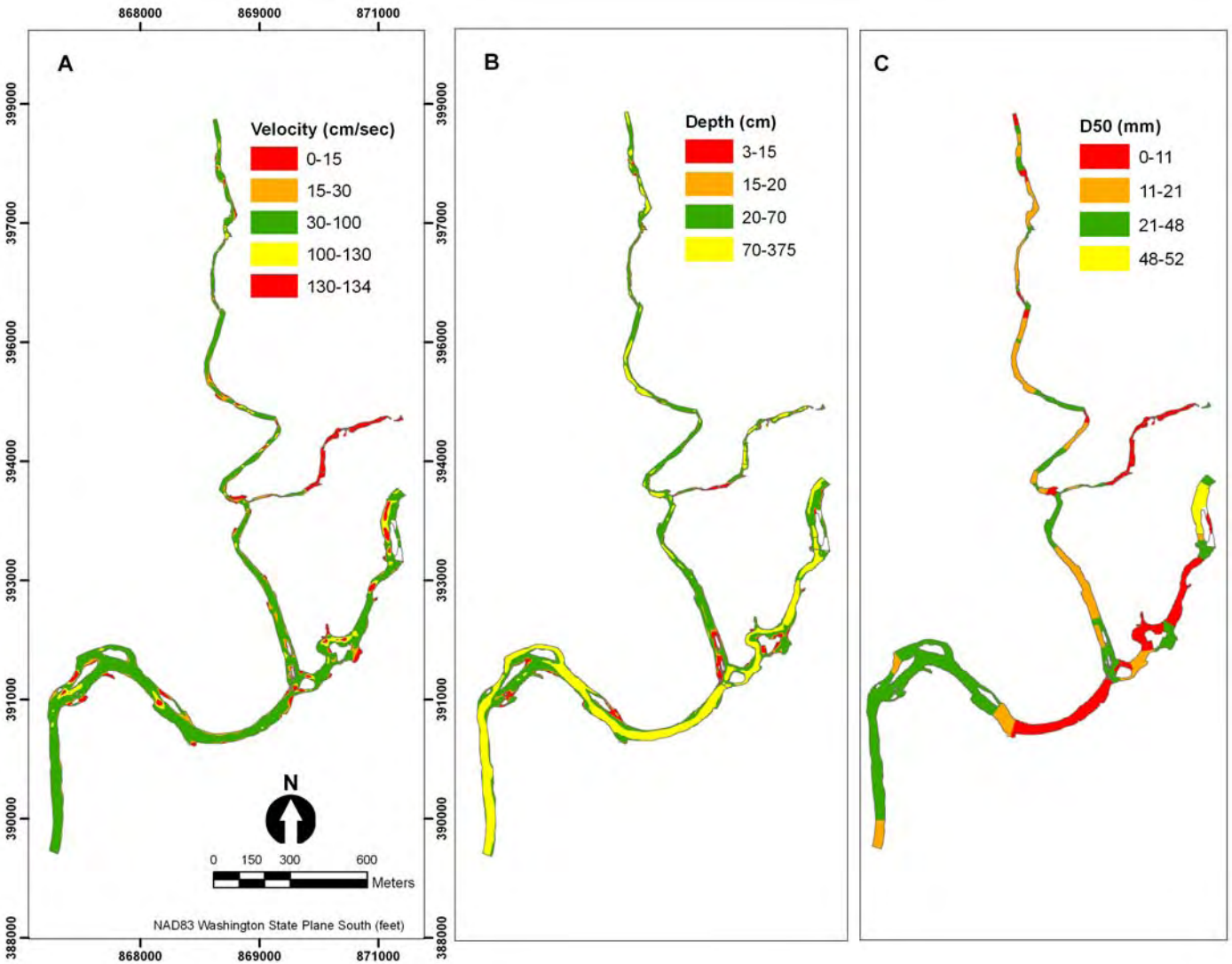


Figure 5.7. Fall Chinook Salmon Spawning Limits. (A) velocity, cm/sec; (B) depth, cm; (C) substrate, d_{50}; mm. Colors refer to potential use categories: Not used (red), within lower limits (orange), within upper limits (yellow), and probable use (green).

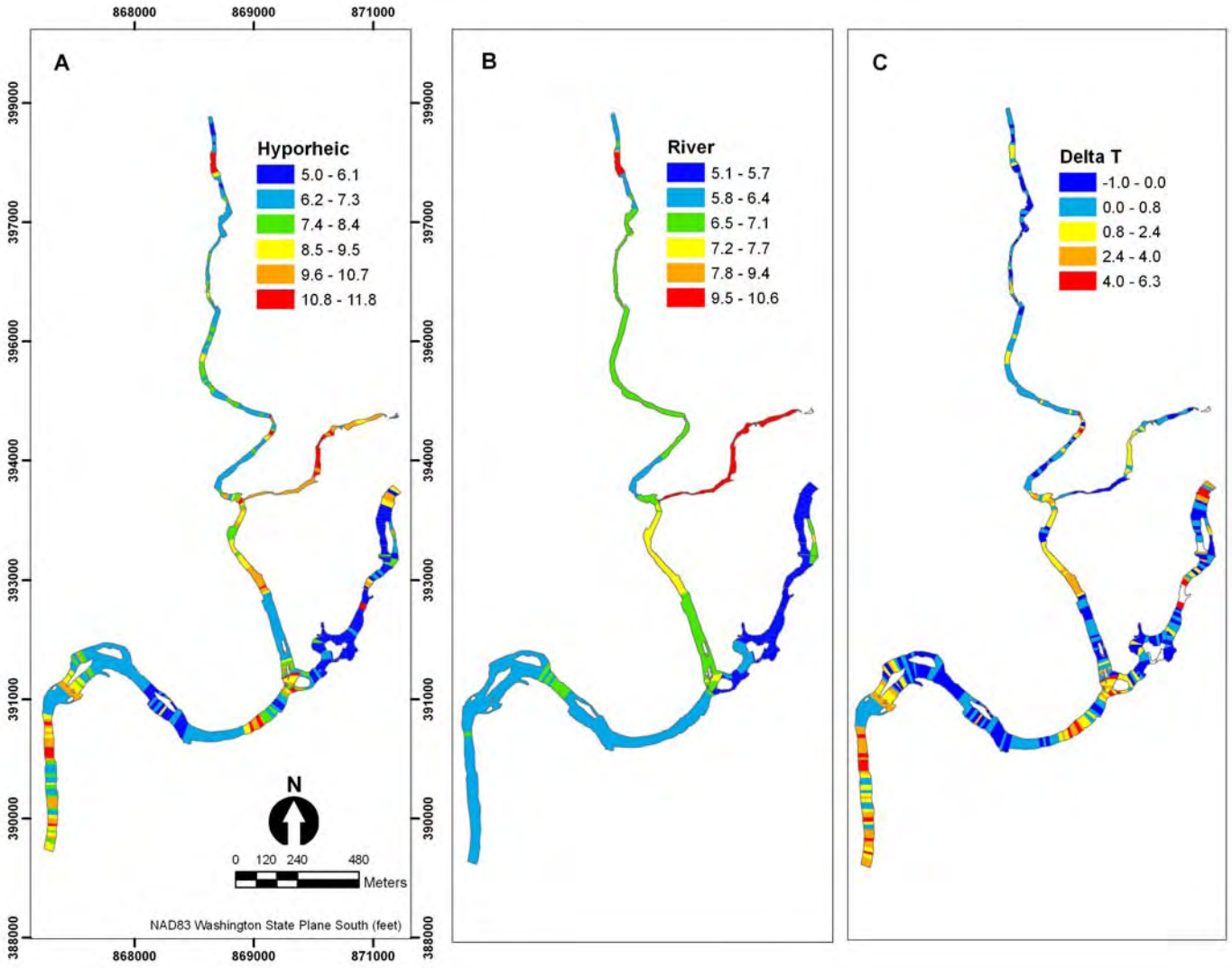


Figure 5.8. Grays River Study Area Water Temperatures. (A) hyporheic; (B) river; (C) difference between hyporheic and river (ΔT). ΔT values greater than 0.8°C were considered suitable for chum salmon spawning.

ΔT ranged from −1.0 to 6.3°C throughout the study area (Figure 5.8c). Zones of large positive ΔT values were identified in each reach, with particularly extensive areas with elevated hyporheic temperatures in the middle of Crazy Johnson and in the lower third of the lower Grays. Other warm hyporheic areas were identified near the confluences of the West Fork and Crazy Johnson and the West Fork and the Grays River (Figure 5.8c). Area characterized by hyporheic temperatures that were elevated compared to river water represented 33.9% (6.52 ha) of the total area, ranging from 21.8% (upper West Fork) to 50.3% (Crazy Johnson; Table 5.6).

Table 5.6. Area Characterized by the Difference Between Hyporheic and River Water Temperature (ΔT)

ΔT (°C)	>0.8	
Stream Reach	ha	%
Total	6.53	33.9
Crazy Johnson	0.44	50.3
Upper West Fork	0.68	21.8
Lower West Fork	0.93	40.9
Upper Grays	0.84	23.0
Lower Grays	3.64	39.0

ΔT was more limiting to chum salmon spawning habitat availability than were velocity, depth, or substrate. Total available chum salmon spawning area based on all four habitat variables is 5.88 ha, or 29.6% of the wetted area (Table 5.7). Among stream reaches, total available spawning area based on all four habitat variables ranges from 16.3% (upper Grays) to 46.9% (Crazy Johnson) of the total wetted area (Table 5.7).

Table 5.7. Chum Salmon Total and High-Quality (Quality Score = 2.0) Spawning Habitat Area by Stream Reach Limited by Velocity, Depth, Substrate (VDS), and by These Variables and Hyporheic Temperature (All). See text for explanation of habitat quality scores.

	VDS					All				
	Spawning		Highest Quality			Spawning		High Quality		
Stream Reach	Area (ha)	% of Wetted Area	Area (ha)	% of Spawning	% of Wetted Area	Area (ha)	% of Wetted Area	Area (ha)	% of Spawning	% of Wetted Area
Total	17.76	89.5	3.26	18.4	16.4	5.88	29.6	0.73	12.4	3.7
Crazy Johnson	0.81	87.4	0.05	5.7	5.0	0.43	46.9	0.00	0.0	0.0
Upper West Fork	2.90	91.4	0.89	30.5	27.9	0.60	18.9	0.11	18.3	3.6
Lower West Fork	2.03	88.8	0.53	26.3	23.4	0.83	36.2	0.13	15.7	5.7
Upper Grays	3.28	80.0	0.33	10.0	8.0	0.67	16.3	0.05	7.5	1.4
Lower Grays	8.74	93.3	1.47	16.8	15.7	3.35	35.8	0.44	13.2	4.7

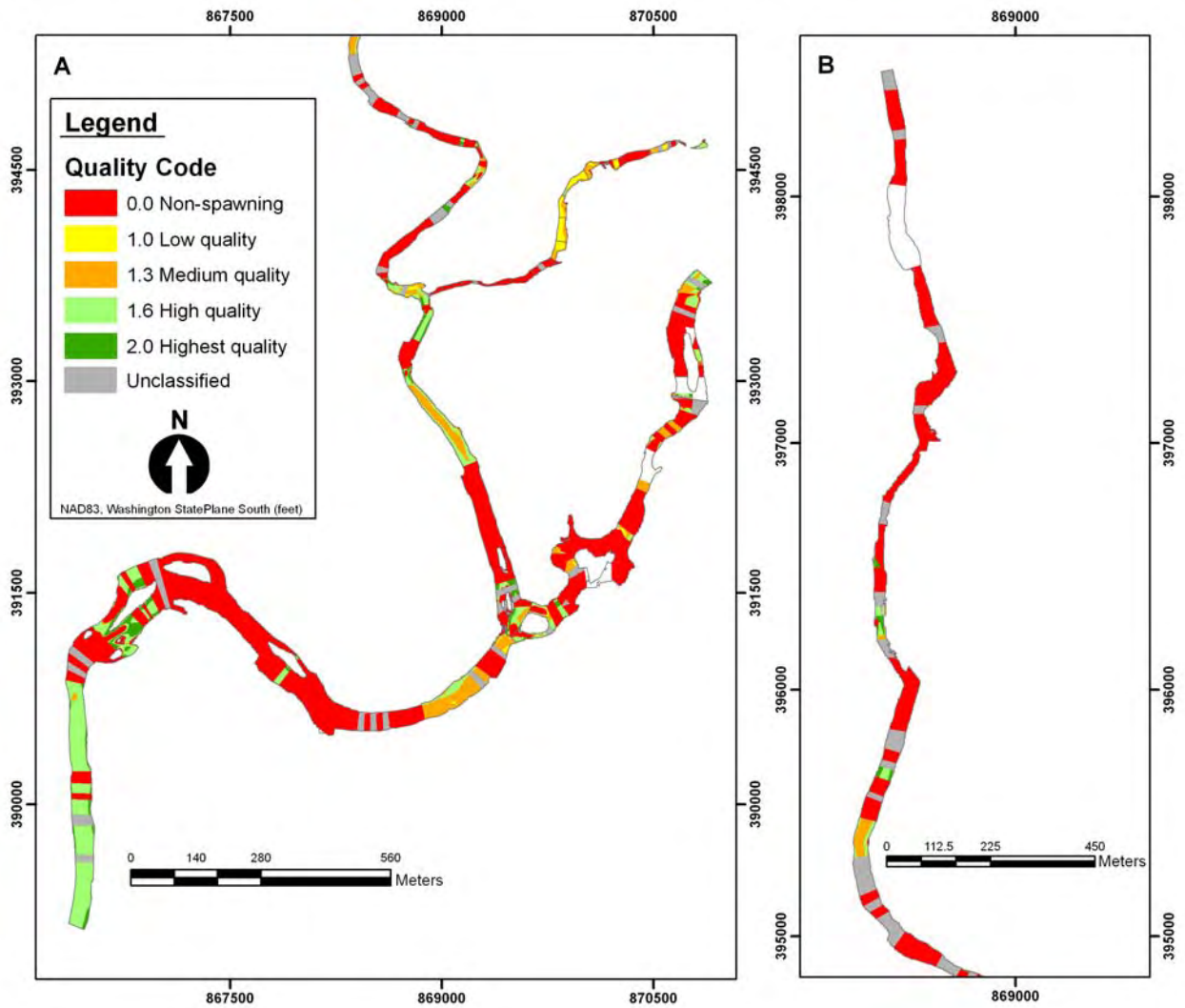


Figure 5.9. Habitat Quality of Chum Salmon Spawning Areas Based on Velocity, Depth, Substrate and Hyporheic Temperature in A) the Study Area Including the Lower West Fork and B) the Upper West Fork. Gray areas could not be classified because hyporheic and river temperature differences were within the range of equipment uncertainty.

5.1.2.4 Habitat Quality

Of the habitat suitable for chum salmon spawning based on velocity, depth, and substrate, 18.4% (3.26 ha) is of the highest quality (quality code 2.0; Table 5.7, Figure 5.9). Among reaches, 5.7% to 30.5% of spawning habitat is of the highest quality. Of the wetted area suitable for spawning based on all four habitat characteristics, 12.4% (0.73 ha) is of the highest quality. Among reaches, 0.0 to 18.3% is of the highest quality. See Appendix H for additional detail regarding habitat quality in spawning areas.

Habitat quality patterns for fall Chinook salmon were similar to those for chum salmon. Of the total area suitable for fall Chinook salmon spawning based on velocity, depth, and substrate, 26.1% (3.33 ha) is of the highest quality (Table 5.8, Figure 5.10). Among reaches, highest-quality habitat suitable for these characteristics ranges from 4.8% to 27.5%. See Appendix H for additional detail.

Table 5.8. Fall Chinook Salmon Total and High Quality (Quality Score = 2.0) Spawning Habitat Area by Stream Reach Limited by Velocity, Depth, and Substrate. See text for explanation of habitat quality scores.

	Spawning		Highest Quality		
	Area (ha)	% of Wetted Area	Area (ha)	% of Spawning	% of Wetted Area
Total	12.75	64.2	3.33	26.1	16.8
Crazy Johnson	0.05	5.4	0.00	4.8	0.2
Upper West Fork	2.37	74.5	0.64	27.0	20.1
Lower West Fork	1.89	82.5	0.52	27.5	22.8
Upper Grays	1.79	43.7	0.35	19.4	8.4
Lower Grays	6.65	71.0	1.83	27.4	19.5

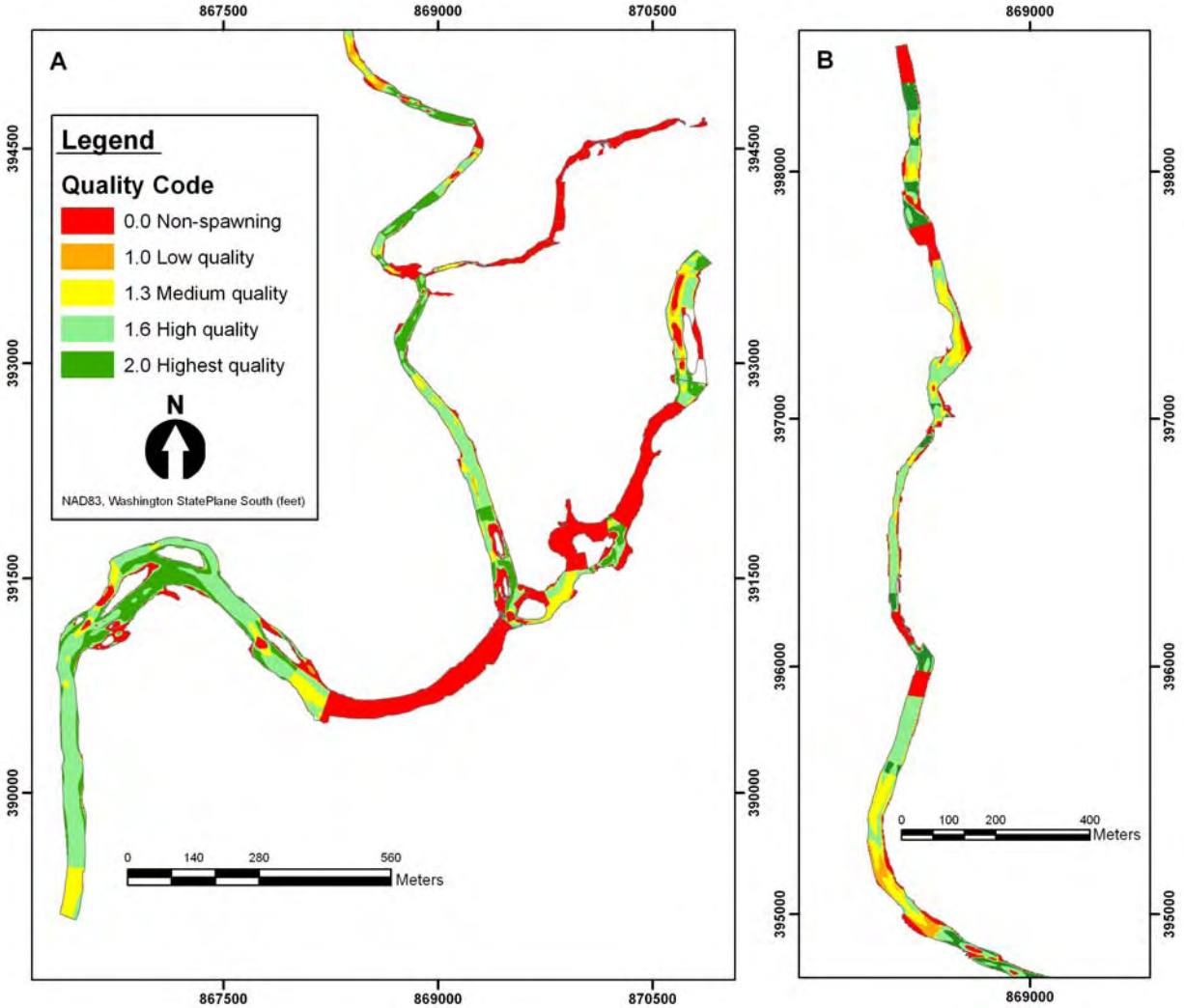


Figure 5.10. Quality of Fall Chinook Salmon Spawning Habitat Based on Velocity, Depth, and Substrate in A) the Study Area Including the Lower West Fork and B) the Upper West Fork

5.1.2.5 Effect of Flow on Spawning Habitat

Total wetted area available in the five study reaches at 90, 50, and 10% exceedance flows were 14.74, 19.80, and 28.20 ha, respectively (see Appendix H for additional detail). Wetted area increased by 34.3% from the 90% to the 50% exceedance flow and by 42.4% from the 50% to the 10% exceedance flow. Among reaches, wetted area increases from the 90% to the 50% exceedance flows were distributed relatively uniformly, with a somewhat lower than average increase in the upper Grays reach and somewhat higher than average increase in Crazy Johnson. From the 50% to the 10% exceedance flow, wetted area increases were also relatively uniform, with a somewhat lower than average increase in the upper Grays. At the 10% exceedance flow, the abandoned channel and a considerable portion of the upper Grays floodplain area between Crazy Johnson and the upper Grays were inundated, contributing an additional 1.96 and 11.00 ha of wetted area, respectively.

Of the wetted area available, 89.7% (13.12 ha), 88.9% (17.76 ha) and 70.3% (19.10 ha) of the total wetted area was suitable for chum salmon spawning based on velocity, depth, and substrate at the 90, 50, and 10% exceedance flows, respectively (Table 5.9). Spawning area increased in all reaches from the 90% to the 50% exceedance flow and increased in all reaches except the upper and lower Grays from the 50% to the 10% exceedance flow. In these reaches, spawning area decreased and stayed the same, respectively. Of the floodplain area, 43.5% (4.52 ha) is suitable for chum salmon spawning based on velocity and depth.

When restricted by ΔT, 31.3% (4.47 ha), 29.6% (5.88 ha), and 21.1% (5.75 ha) of the total wetted area was suitable for chum salmon spawning at the 90, 50, and 10% exceedance flows, respectively (Table 5.9). Consistent with spawning area based on velocity, depth, and substrate, spawning area increased in all reaches from the 90% to the 50% exceedance flow and increased in all reaches except the upper and lower Grays from the 50% to the 10% exceedance flow. The spatial distribution of suitable spawning areas was similar between the 90% and 50% exceedance flows but not between the 50% and 10% exceedance flows, in which large areas of the upper West Fork and upper and lower Grays reaches became unsuitable for spawning as flow increased (Figure 5.11).

At the 90, 50, and 10% exceedance flows, 54.6% (7.99 ha), 64.2% (12.75 ha), and 60.7% (16.50 ha) of the wetted area was suitable for fall Chinook salmon spawning based on velocity, depth, and substrate (Table 5.10). The spatial distribution of suitable spawning areas was similar among all flows, with general areas of suitability or unsuitability increasing laterally across the channel (Figure 5.12). Spawning area increased with flow in all reaches as well. More than 50% of the total wetted area was suitable for spawning at all three flows in both reaches of the West Fork and the lower Grays whereas only 0.1, 5.4, and 16.6% of Crazy Johnson was suitable for spawning at the 90, 50, and 10% exceedance flows, respectively. The abandoned upper Grays channel and other floodplain areas contributed 0.45 (23.1% of wetted area) and 2.66 ha (24.2% of wetted area) of additional spawning area at the 10% percentile flow.

Table 5.9. Effect of Flow on Chum Salmon Spawning Habitat Availability Based Velocity, Depth, and Substrate (VDS) and These Variables Plus Hyporheic Temperature (All)

Reach	Percent Exceedance Flow											
	90				50				10			
	VDS		All		VDS		All		VDS		All	
	ha	%	ha	%	ha	%	ha	%	ha	%	ha	%
Total[a]	13.12	89.7	4.58	31.3	17.76	88.9	5.88	29.6	19.10	70.3	5.75	21.1
Crazy Johnson	0.68	84.1	0.40	40.0	0.81	91.4	0.43	46.9	1.14	94.0	0.53	43.8
Upper West Fork	2.23	88.2	0.37	14.7	2.90	91.2	0.60	19.0	3.18	72.0	0.67	15.3
Lower West Fork	1.45	83.0	0.66	37.6	2.03	89.0	0.83	36.5	2.82	92.7	1.26	41.6
Upper Grays	2.29	91.2	0.55	22.1	3.28	80.5	0.67	16.4	3.18	53.3	0.56	9.4
Upper Grays AC[b]	No flow				No flow				1.59	83.4	0.15	8.1
Upper Grays FL[b]	No flow				No flow				4.52[c]	43.5	--	--
Lower Grays	6.47	91.9	2.59	36.9	8.74	93.5	3.35	35.8	8.78	70.0	2.71	21.6

(a) Excluding upper Grays abandoned channel and floodplain.
(b) AC = abandoned channel, FL = floodplain.
(c) Based on velocity and depth only.

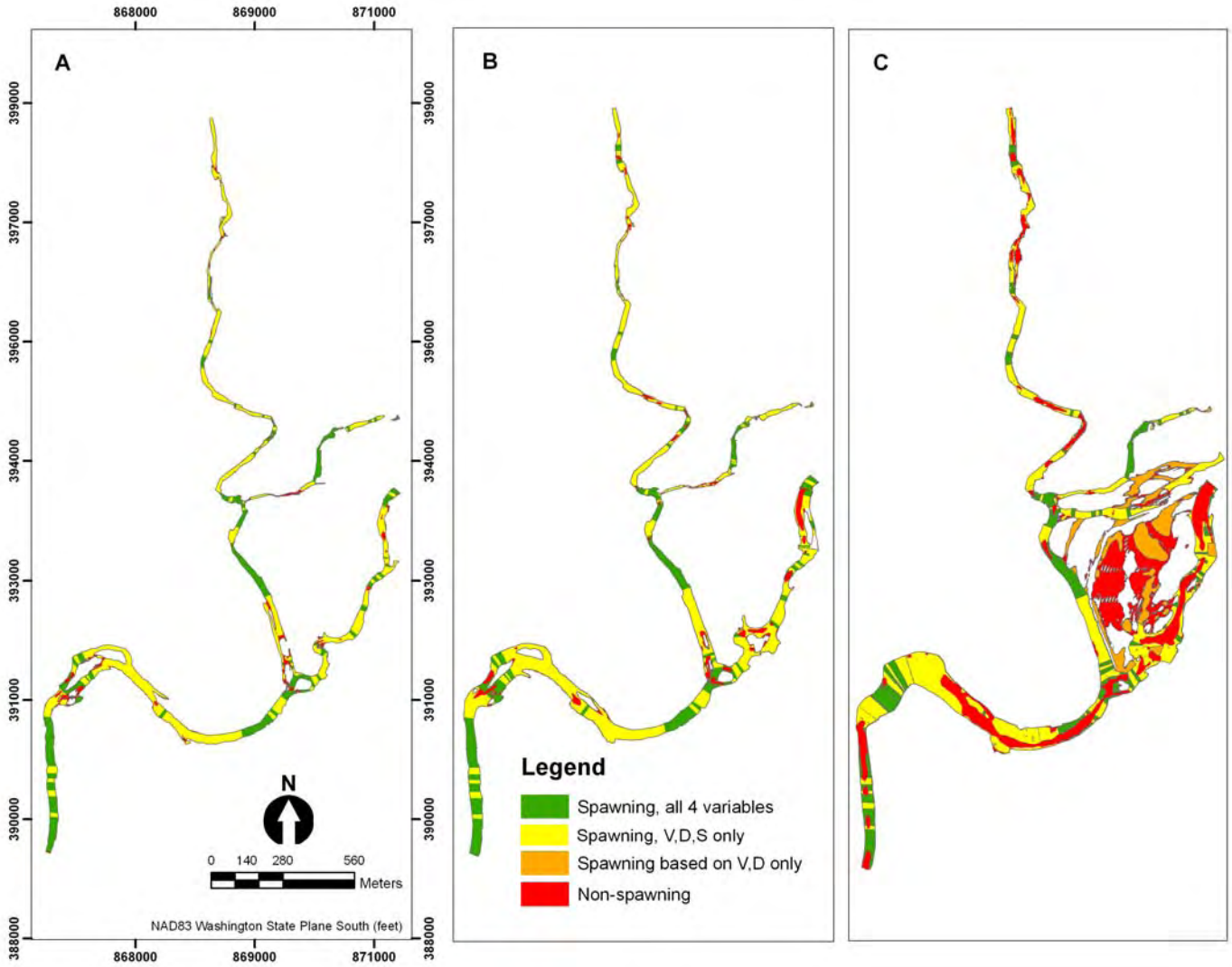


Figure 5.11. Chum Salmon Spawning Habitat at the a) 90%, b) 50%, and c) 10% Exceedance Flows for November 1–December 31. Shown are spawning habitat based on four variables, velocity (V), depth (D), substrate (S), and hyporheic temperature (green), V,D, and S only (yellow), V and D only (orange), and non-spawning habitat (red).

Table 5.10. Effect of Flow on Fall Chinook Salmon Spawning Habitat Availability Based on Velocity, Depth, and Substrate

Stream Reach	Percent Exceedance Flow					
	90		50		10	
	ha	%	ha	%	ha	%
Total [(a)]	7.99	54.5	12.75	64.2	16.50	60.7
Crazy Johnson	0.00	0.0	0.05	5.4	0.20	16.6
Upper West Fork	1.34	53.2	2.37	74.5	2.88	65.3
Lower West Fork	1.08	61.8	1.89	82.5	2.94	96.7
Upper Grays	1.10	43.7	1.79	43.7	1.97	32.8
Upper Grays AC[(b)]	No flow		No flow		0.45	23.1
Upper Grays FL[(b)]	No flow		No flow		2.66[(c)]	24.2
Lower Grays	4.46	63.4	6.65	71.0	8.53	68.0

(a) Excluding upper Grays abandoned channel and floodplain.
(b) AC = abandoned channel, FL = floodplain.
(c) Based on velocity and depth only.

5.1.2.6 Potential Spawning Habitat in the Abandoned Channel

Potential chum salmon spawning habitat in the channel abandoned during the 1999 avulsion was limited by ΔT but not substrate (Table 5.11, Figures 5.13a and b). Approximately 11.1% of the estimated wetted area in the abandoned channel was suitable for chum salmon spawning based on substrate and hyporheic temperature (Figure 5.13c). However, of this area, only 0.9% had substrate in the probable use range, suggesting limited substrate habitat quality throughout most of the abandoned channel. In contrast, spawning area for fall Chinook salmon was limited by substrate (Table 5.11). Of the available wetted area, 44.5% was potentially suitable for fall Chinook salmon spawning (Figure 5.13d), of which 15.3% had substrate that was in the probable use range.

5.1.2.7 Redd Capacity

Chum salmon redd capacity estimates for the study area range from 695 redds for highest-quality spawning habitat (quality score = 2.0) limited by all four habitat variables, with inter-redd spacing, to 67,824 redds for all spawning habitat limited by only velocity, depth, and substrate and with no inter-redd spacing (Table 5.12). Limiting available habitat by ΔT greatly reduces potential redd capacity. For the study area, habitat defined by velocity, depth, and substrate has a total capacity (all habitat qualities combined) of 16,924 to 67,824 redds, compared to 5,590 to 22,405 redds in habitat limited by these variables and ΔT. Total redd capacity in highest-quality habitat using all four spawning habitat variables ranges from 695 to 2,786 redds. Among reaches, redd capacity in highest-quality habitat using all four spawning habitat variables ranges from Crazy Johnson, with zero redds at either redd area, to the lower Grays, with more than half of all redds (419–1,679 redds).

Across the study area, fall Chinook salmon redd capacity estimates for the study area based on velocity, depth, and substrate range from 4,904 to 19,615 redds. Redd capacity in highest-quality habitat ranges from 1,281 to 5,123 redds (Table 5.13). Total redd capacity among reaches ranges from the lower Grays with 2,558 to 10,231 redds, to Crazy Johnson with 15 to 62 redds. Redd capacity in highest-quality habitat among reaches ranges from the lower West Fork (246–983 redds), to Crazy Johnson, with only 1 to 3 redds.

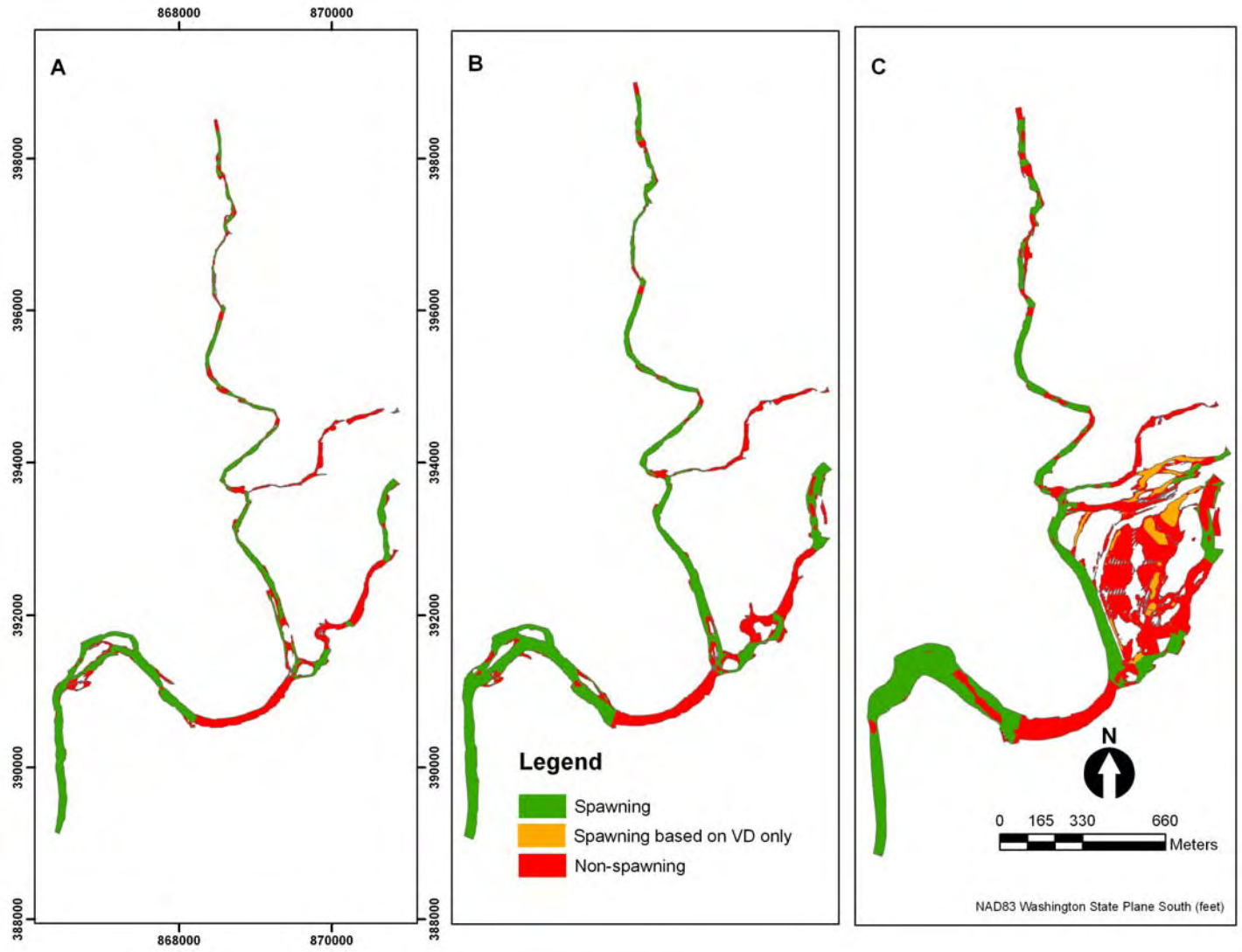


Figure 5.12. Fall Chinook Salmon Spawning Habitat at the a) 90%, b) 50%, and c) 10% Exceedance Flows for November 1–December 31. Shown are spawning habitat based on three variables, velocity (V), depth (D), and substrate (S) (green), V and D only (orange), and non-spawning habitat (red).

Table 5.11. Total Spawning Area Availability in the Abandoned Upper Grays Channel Based on Substrate and Hyporheic Temperature, Limiting Factors, and Proportion of Spawning Area Within Probable Use Substrate Limits

Species	Limiting Factor	Non-Spawning		Spawning		Probable Use	
		ha	%	ha	%	ha	%
Chum							
	Substrate	0.00	0.0	2.94	100.0	1.07	36.3
	Hyporheic temperature	2.61	88.9	0.33	11.1		
	Both	2.61	88.9	0.33	11.1	0.03	0.9
Fall Chinook							
	Substrate	1.63	55.5	1.31	44.5	0.45	15.3

5.1.3 Discussion

Based on the limits we established, the majority of wetted area in the Grays River study area was suitable for chum and fall Chinook salmon spawning when limited by velocity, depth, and substrate. Of the total available area, 61.6% (12.22 ha) was within spawning limits for both species, 27.9% (5.54 ha) was within only chum salmon limits, 2.7% (0.54 ha) was within only fall Chinook salmon limits, and 7.8% (1.55 ha) was beyond the spawning limits for both species. Among reaches, 80.0% (upper Grays) to 93.3% (lower Grays) of available area was suitable for chum salmon and 5.4% (Crazy Johnson) to 82.5% (lower West Fork) was suitable for fall Chinook salmon. Substrate was never limiting for chum salmon and 93.7 and 95.6% of the total area was within velocity and depth limits, respectively. Area within velocity, depth, and substrate limits for fall Chinook salmon were 88.4, 92.4, and 74.6%, respectively.

Availability of area characterized by elevated hyporheic temperature may be the limiting spawning characteristic for chum salmon in the study area. We assumed that spawning chum salmon required areas with elevated hyporheic temperatures and found that most areas of concentrated spawning were associated with elevated hyporheic temperatures. In the Pacific Northwest, spawning site selection by chum salmon has been associated with areas of warm upwelling hyporheic exchange (Salo 1991; Bjornn and Reiser 1991; Geist et al. 2002). Salmon spawning has also been associated with geomorphological features of the streambed that promote hyporheic exchange (Bjornn and Reiser 1991; Geist and Dauble 1998; McHugh and Budy 2004). If chum salmon require elevated hyporheic temperatures, area suitable for chum salmon is reduced from 89.5% (17.76 ha) to 29.6% (5.88 ha). Other researchers have found that salmon spawning occurs in a relatively small proportion of otherwise suitable spawning habitat due to limitations in the availability of hyporheic flow (Leman 1993; Geist and Dauble 1998; Geist et al. 2002). However, the extent to which hyporheic exchange strictly limits spawning habitat availability is not thoroughly understood. Spawning salmonids may not be limited to areas of geomorphically-induced hyporheic exchange because alteration of streambed morphology during redd construction can induce hyporheic exchange at the scale of the individual redd (White 1990). Associations between chum salmon spawning and hyporheic exchange are explored further in Sections 5.2 and 5.3.

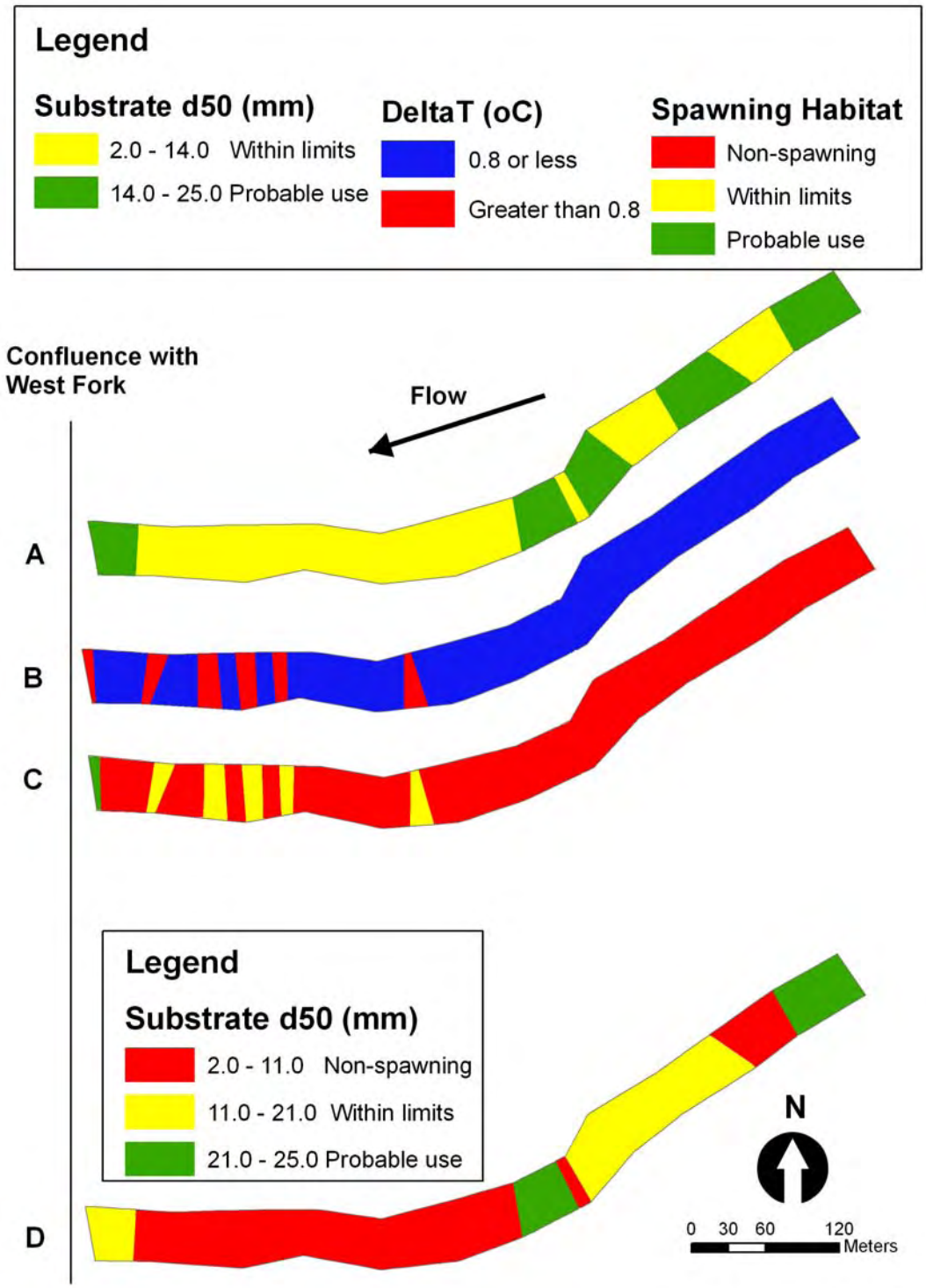


Figure 5.13. Chum and Fall Chinook Spawning Habitat Characteristics of the Upper Grays River Abandoned Channel. Pictured are spawning habitat characteristics (A) substrate (d_{50}, mm), (B) ΔT, and (C) spawning habitat availability for chum salmon and (D) substrate (d_{50}, mm) distribution for fall Chinook salmon.

Table 5.12. Range of Chum Salmon Redd Capacities (number of redds) for Spawning Habitat Area Estimated by Velocity, Depth, and Substrate (VDS) and These Variables Plus Hyporheic Temperature (All). Each range of calculated values is based on two redd area estimates, 10.50 m^2/redd and 2.62 m^2/redd.

Stream Reach	VDS		All	
	All	Highest[(a)] Quality	All	Highest[(a)] Quality
Total	16,924–67,824	3,105-12,442	5,590–22,405	695–2,786
Crazy Johnson	771–3,092	44-176	400–1,603	0–0
Upper West Fork	2,781–11,145	844-3,382	571–2,290	105–420
Lower West Fork	1,933–7,748	509-2,041	790–3,168	124–496
Upper Grays	3,124–12,519	311-1,248	638–2,557	48–191
Lower Grays	8,333–33,397	1,396-5,596	3,190–12,786	419–1,679
(a) Highest quality = quality score 2.0.				

Table 5.13. Range of Fall Chinook Salmon Redd Capacities (number of redds) for Spawning Habitat Area Estimated by Velocity, Depth, and Substrate. Range of redd capacities is presented for all suitable habitat and highest quality habitat (quality score = 2.0). Each range of calculated values is based on two redd area estimates, 26.0 m^2/redd and 6.5 m^2/redd.

	All	Highest Quality
Total	4,904–19,615	1,281-5,123
Crazy Johnson	15–62	1-3
Upper West Fork	912–3,646	246-983
Lower West Fork	727–2,908	200-801
Upper Grays	685–2,738	133-533
Lower Grays	2,558–10,231	702-2,809

Fall Chinook salmon in the Grays River study area did not demonstrate a clear spawning association with elevated hyporheic temperature. This is in contrast to several studies in which an association with elevated hyporheic temperature or upwelling flow was made (Burner 1951; Bjornn and Reiser 1991; Geist 2000; Connor et al. 2003). However, fall Chinook salmon also have been shown to spawn in down-welling areas or areas where hyporheic temperatures were similar to river water temperatures (Russell et al. 1983; Geist et al. 2002; Hanrahan 2006; Geist et al. in review). The hyporheic requirements of spawning fall Chinook salmon appear to be less well understood than those of other salmon species (Smirnov 1975; Geist 2000). Vronskii and Leman (1991) suggested that flow magnitude may be more important than direction to fall Chinook salmon during spawning site selection. Geist et al. (2002) hypothesized that when chum and fall Chinook salmon spawn in sympatry, differences in hyporheic preference may reduce competition for spawning sites and fall Chinook salmon redd loss by the later-arriving chum salmon. We found considerable overlap in hyporheic characteristics of chum and fall Chinook salmon because both species spawned in areas with elevated hyporheic temperatures. However, populations of both species have been depressed for decades, potentially reducing competitive and therefore selective pressure favoring spawning habitat segregation. In addition, our characterization of the hyporheic environment in fall Chinook salmon spawning habitat may be inaccurate due to the low number of redds available for evaluation.

Although considerable area was suitable for spawning by one or both species, most was not of the highest quality for either species. For chum and fall Chinook salmon, only 18.4% (3.26 ha) and 26.1% (3.33 ha), respectively, of available spawning area defined by velocity, depth, and substrate was of high quality (quality score = 2.0, all variables in the probable use range). Of spawning habitat defined by all four variables, 12.4% (0.73 ha) and 29.1% (2.07 ha) was of high quality for these species. However, we found that chum and fall Chinook salmon may use lower quality habitat even when high quality suitable habitat is available. Thus, our quality scores based on probable use limits derived from other river systems may not fully describe chum salmon habitat selection in the Grays River. Habitat selection by the reduced spawning populations relative to historic levels may differ from historic habitat use, habitat use reflects habitat availability, which may differ from other river systems as well as historic habitat conditions in the Grays River study area, and limiting factors such as hyporheic exchange may affect use ranges of other habitat characteristics.

Spawning habitat availability for chum and fall Chinook salmon varied with flow. For chum salmon, spawning habitat availability based on velocity, depth, substrate, and the difference between hyporheic and river temperature was lowest at the 90% exceedance flow (4.58 ha) and was similar at the 50% and 10% exceedance flows (5.88 and 5.75 ha, respectively). Among reaches, habitat was always least available at the 90% exceedance flow and the relationship was inconsistent between the 50% and 10% exceedance flows. For fall Chinook salmon, spawning habitat availability based on velocity, depth, and substrate increased with flow from the 90% (7.99 ha) to the 50% (12.75 ha) to the 10% (16.50 ha) exceedance flow. This pattern was seen in all individual reaches as well. At higher flows, wetted area increased proportionally throughout the study area. In addition, at the 10% exceedance flow, large floodplain and abandoned channel areas became inundated, greatly increasing wetted area at this flow. For chum salmon overall and in some reaches, velocity and depth in some areas exceeded spawning limits at the 10% exceedance flow but did not at the 50% exceedance flow, and total available spawning area decreased in spite of the increased wetted area. In contrast, for fall Chinook salmon, increasing flows increased wetted area without disproportionately increasing area with depths or velocities that exceeded spawning limits, thereby increasing net spawning area. Thus, proportionally more spawning area is available at higher flows for fall Chinook salmon than for chum salmon.

The abandoned channel may contain considerable spawning habitat for chum and fall Chinook salmon if adequate flow velocities and depths could be restored. All of the channel length contained substrate suitable for chum salmon, and 44.5% of the channel length contained suitable substrate for fall Chinook salmon spawning. Although only 11.1% (approximately 0.33 ha) had elevated hyporheic temperatures preferred by chum salmon, hyporheic exchange is largely induced by head differences associated with flowing water. The general vicinity of the abandoned channel appears to have strong subsurface flow (see Section 5.2 for additional discussion), and an artificial spawning channel, destroyed during the 1999 channel avulsion, had been constructed nearby to intercept hyporheic flow (Hymer 1993). If flow were restored to the channel, one might expect the area and extent of hyporheic exchange to increase. Much of the area with unsuitable substrates limiting potential spawning area for fall Chinook salmon was in pools ponded by beaver dams. Restoration of active flow may also remove beaver dams and associated fine sediments, improving substrate conditions for spawning fall Chinook salmon.

We estimate the chum salmon redd capacity to be 5,590 to 22,405 redds in the Grays River study area based on available area that is suitable for velocity, depth, substrate, and hyporheic exchange and depending on redd size/density. If chum salmon spawn only in high quality habitat, redd capacity is 695 to 2,706 redds. The historic run size for chum salmon in the Grays/Chinook river population ranged from

8,000 to 14,000 fish (LCFRB 2004). Thus, redd capacity appears to be sufficient to support the average historic run size, but most of this habitat may not be high quality habitat. In addition, redd superimposition and clumping are characteristics of chum salmon spawning behavior. Use of available spawning habitat may therefore be inefficient, with some suitable spawning areas unused and others overused by exceptionally high densities of spawning fish. Thus, redd success in some areas may be relatively low. Studies of habitat quality and redd success would be required to fully evaluate the adequacy of available spawning area to support historic run sizes.

We estimate the fall Chinook salmon redd capacity to be 4,904 to 19,615 redds in the Grays River study area based on the redd sizes and spawning habitat limit criteria for the three habitat characteristics we analyzed. Within the highest quality habitat, redd capacity is estimated to be 1,281 to 5,123 redds. The Grays and Chinook rivers historically had a run size of 1,500 to 10,000 fish (LCFRB 2004). Therefore, our analysis suggests that sufficient spawning habitat is currently available to support historic runs of fall Chinook salmon, based on velocity, depth, and substrate habitat limits. However, if habitat quality restricts salmonid redd selection and production, sufficient high quality habitat may not available to support the larger historic annual run sizes. In addition, considerable research suggests that fall Chinook salmon select spawning habitats that are characterized by active hyporheic exchange (e.g., Bjornn and Reiser 1991; Geist 2000; Geist et al. 2002). This suggests that our redd capacity estimates may be considerable overestimates of actual redd capacity. Reasons for the limited return of fall Chinook salmon to the Grays River are uncertain but likely include poor incubation habitat quality and therefore reduced incubation success.

Factors other than spawning habitat may be limiting chum and fall Chinook salmon production and recovery in the Grays River. The most significant limiting factors within the Grays River system may be substrate stability and excessive fines, each of which reduces incubation success and juvenile production (Chapman 1988; Platts et al. 1989; Scrivener and Brownlee 1989; Montgomery et al. 1996). The extent of colmation, or substrate sedimentation, is directly linked to land use practices that increase sediment loading, such as timber harvest and associated road construction (Karr and Schlosser 1978, Platts et al. 1989). Brunke and Gonser (1997) suggest that the balance between bed scour and colmation can be disrupted by a variety of anthropogenic activities. Excessive sediment delivery to the stream channel network has been documented in the Grays River watershed (Chapter 4; Herrera 2005), and we observed extreme channel instability in some portions of the study area. We found that substrate in the Grays River was generally suitable for both chum and fall Chinook spawning. However, the metric we examined (d_{50}) provides an indication of a salmon's ability to move surface sediments during redd construction (Kondolf 2000), but is not informative regarding substrate quality for incubating and emerging embryos. The extent that each of these processes has reduced incubation success in the Grays River remains unknown. Watershed recovery to reduce sediment delivery following land use modifications can require decades (Platts et al. 1989). Therefore, information needs critical to recovery of Grays River spawning habitats include 1) spatially explicit evaluation of redd survival and success to identify relationships between redd siting and geomorphic processes and 2) restoration approaches that mitigate and reduce sediment delivery for the short term.

The Grays River represents one of two remaining significant populations of chum salmon in the lower Columbia River and contains ESA-listed fall Chinook salmon, as well. Our findings suggest that sufficient spawning habitat is available to support historic run sizes of both species, but that much of the available habitat may not be high quality. Because chum salmon fry migrate out of freshwater almost immediately after emergence, the Grays River provides primarily spawning and incubation habitat. In

contrast, fall Chinook salmon may rear for up to one year. This suggests that spawning habitat quality rather than quantity, or habitat factors associated with juvenile rearing in the case of fall Chinook salmon, may be limiting recovery of these species. Management, protection, and restoration of spawning habitat quality may therefore be critical to successful recovery of these ESUs, and without focused effort these populations may continue to decline in the Grays River.

5.2 Chum and Fall Chinook Salmon Spawning Habitat Use

Salmonids spawn in areas characterized by specific velocity, depth, and substrate ranges (Bjornn and Reiser 1991) and tend to spawn in areas of active hyporheic exchange with surface waters (e.g., Bonnell 1991; Salo 1991; Leman 1993; Geist 2000). The habitat characteristics chosen serve to maximize incubation success of eggs buried within the substrate. Factors influencing redd success include water flow and resultant water quality in the incubation environment and probability of redd scour, burial, or disturbance by later-arriving spawning individuals (Shumway et al. 1964; Chapman 1988; Montgomery et al. 1996). The ranges of habitat characteristics selected vary among species and individuals associated with differences in body size and therefore physical capability (Witzel and MacCrimmon 1983; Heggberget et al. 1988; Bjornn and Reiser 1991; Kondolf and Wolman 1993; Fukushima and Smoker 1998). Because desirable habitat characteristics are temporally stable over short time frames, the locations of salmonid spawning areas may be somewhat static (Hoopes 1971; Fukushima 1994; Fukushima and Smoker 1998). Thus, spawning areas for each species may be able to be predicted based on habitat characteristics.

In Section 5.1, we determined potential spawning area availability based on velocity, depth, substrate, and hyporheic temperature limits from the literature and a limited assessment of actual chum and fall Chinook salmon spawning habitat use in the Grays River study area (see also Appendix G). We concluded that if our habitat classification criteria reliably predicted spawning habitat availability for these species, spawning habitat availability appeared to be sufficient to support historic chum and fall Chinook run sizes; therefore, available spawning area (habitat quantity) did not appear to limit recovery of these populations in the Grays River. One assumption we made in Section 5.1 was that chum salmon spawning was restricted to areas of elevated hyporheic temperatures relative to river temperature based on studies documenting such spawning habitat use (Kobayashi 1968; Leman 1993; Geist et al. 2002; Geist et al. in review). In this section we examine chum and fall Chinook salmon habitat use with respect to these spawning habitat characteristics.

Our objectives in this section were to 1) examine the consistency of chum salmon spawning locations during 2002–2005; 2) conduct a detailed analysis of chum and fall Chinook salmon spawning habitat selection based on velocity, depth, substrate, and hyporheic temperature; and 3) compare chum and fall Chinook salmon spawning habitat use in the Grays River study area to that predicted in Section 5.1. Associated with Objective 2, we also examined hyporheic temperature use by fall Chinook and its potential to effect spawning segregation of chum and fall Chinook salmon.

5.2.1 Methods

5.2.1.1 Spawning Use

Chum salmon spawning was quantified during 2002 through 2005 and fall Chinook salmon spawning was quantified in 2003 and 2005. Spawning surveys in the study area were conducted by the Pacific

States Marine Fisheries Commission (2002–2004) and the Washington Department of Fish and Wildlife (2005). However, methods varied each year, making inter-annual comparisons difficult. In 2002, chum salmon locations of individual and mass redds were recorded with a global positioning system (GPS) weekly throughout the spawning season for each species. A mass redd was an area with multiple redds where individual redds could not be counted accurately. The areal extent of mass redds was not recorded in 2002. In 2003, individual redd locations and three or four boundary points for each mass redd were recorded weekly as GPS points, allowing for better approximation of each mass redd area. In 2004, spawning survey methods were changed from redd counts to spawner counts because of problems with redd superimposition and difficulty quantifying redds and mass redds. Therefore, in 2004 and 2005, each study reach was divided into 50-m-long sampling sections. Within each sampling section, the numbers of carcasses and live fish actively engaged in spawning were recorded weekly. Carcasses were marked to avoid recounting. In 2004, spawning count data were obtained for all reaches during only one week of the chum salmon spawning season. We visually compared the spatial distribution of spawning fish across reaches and sections for which more than one weekly count was obtained and established that the spatial distribution among reaches and sections did not vary across the spawning season. Therefore the week for which we had data for all reaches was representative of the entire season. In 2005, summed weekly counts for the entire spawning season were used. In 2003 and 2005, fall Chinook salmon individual redd locations were recorded with a GPS.

Because chum salmon spawning occurrence data were collected using different methods among years, we developed a spawning use index to examine spatial consistency of use. Within each year, thresholds were established based on examination of frequency histograms of the data, and each sampling section was scored as 0 (no spawning), 1, or 2 (high spawning). Spawning use scores were averaged across the four years to obtain an average spawning use score for each sampling section in the study area.

To model the relationship between chum salmon spawning use and habitat variables, we classified each 50-m sampling section as spawning or non-spawning for 2004 and 2005. We did not include the spawning data for 2002 and 2003 in these analyses because total count per section data were not available. Based on examination of spawning count frequency histograms, we characterized a section as spawning if it contained 5% or more of the total spawning count for that year. Spawning was not equally distributed among reaches, particularly in 2004. Therefore, the 5% classification criterion produced few sections classified as spawning in some reaches. To examine spawning habitat use within reach, we therefore reclassified sections within these reaches (Crazy Johnson, lower Grays, and upper West Fork) as spawning if they contained 2% or more of the total spawning count. This criterion produced more balanced numbers of spawning and non-spawning sections in each reach while still reflecting the distribution of concentrated spawning in each reach.

Fall Chinook salmon redds were mapped during one week each in 2003 and 2005. Spawning counts were very low in both years—in 2003, 2 redds were mapped in the project area, both in the upper West Fork, and in 2005, 15 redds were mapped, 7 in the upper West Fork and 8 in the lower Grays. Because spawning use of the Grays River study area by fall Chinook salmon was extremely low when redd location data were collected, we compared individual spawning locations obtained via GPS to randomly selected non-spawning locations within the same reaches. Velocity, depth, and substrate characteristics of each redd and non-redd location were averaged over a 5-m^2 area, the approximate size of a fall Chinook salmon redd (Burner 1951).

5.2.1.2 Spawning Habitat

Average spawning season discharge (November–December) in the Grays River watershed during 2004 was intermediate between the 90% and 50% exceedance flows and, during 2005, closely matched the 50% exceedance flow. We used the Distributed Hydrology Soil Vegetation Model (DHSVM, described in Chapter 3) to obtain an estimate of the 90% and 50% exceedance discharges at all upstream boundary locations in the study area. These discharges were used as inputs to the MASS2 model. The 42-year precipitation record available from the Grays River Hatchery Climate Center site (Western Regional Climate Center, Site 453333) on the West Fork at the upper bound of the project area was used as input data to the DHSVM. Model development criteria were based on 1996 vegetation coverage data obtained from Scott (2001).

We used the MASS2 hydrodynamic model to produce velocity and depth estimates at a 1-m^2 resolution. We used MASS2 outputs from the 90% and 50% exceedance flow model runs for 2004 and from the 50% exceedance flow for 2005, respectively, to match actual flows for those years. The 90% and 50% exceedance flows at the lower end of the project area were estimated to be 349.4 and 1058.6 cfs, respectively. Although analyses of 2004 chum salmon spawning habitat use were initially conducted with both flows, we present only the results for the 90% exceedance flow because the two outcomes were similar. Velocity, depth, and flow data for model calibration (see below) were collected as described in Section 5.1.

Within each of the study reaches, we identified distinct geomorphic features (e.g., riffles, pools) longer than 10 m. For each feature, pebble counts were conducted to determine the cumulative grainsize distribution (see Section 5.1 for additional detail). We calculated the d_{84} (the grain size in mm of which 84% of particles are finer), a metric that is widely used in salmonid spawning studies (Kondolf and Wolman 1993) and has been associated with salmonid spawning habitat selection (McHugh and Budy 2004). From the substrate data based on geomorphic features, we calculated 50-m sampling section d_{84} averages weighted by section area. Although not used in our analyses, we also report d_{50} values because this metric also is commonly reported in salmonid spawning studies.

Hyporheic temperature was recorded every 12.5 m along the thalweg of each study reach (4 sample points per spawning use section; see Section 5.1 for additional information). Hyporheic temperature was mapped at a total of 634 sites, 386 sites on the Grays River, 227 sites on the West Fork, and 66 sites on Crazy Johnson. Data for each temperature point location were extrapolated to the area bounded laterally by the wetted channel perimeter at the 50% exceedance flow and longitudinally to channel cross sections perpendicular to the thalweg located midway between adjacent temperature point locations. Average hyporheic temperature weighted by area was calculated for each sampling section and used in analyses of chum salmon spawning. Hyporheic data for sampling points nearest redd and non-redd locations were used in fall Chinook salmon analyses.

5.2.1.3 Analyses

To evaluate the consistency of chum salmon spawning among years, we first calculated Pearson correlation coefficients of spawning use score for each year pair. We then tested the hypothesis that spawning was randomly distributed among sampling sections throughout the study area among years (i.e., there was no spatial consistency in spawning among years) by conducting a Monte Carlo simulation of expected average spawning scores under a random distribution across the study area. For each year, we

calculated expected probabilities of a 0, 1, or 2 score per section for each year based on actual probabilities of observation for that year. We conducted 1,000 simulations of a random spawning distribution for each year and averaged the four years to produce an expected distribution of average spawning use scores. We tested our hypothesis by conducting a χ^2 test of the observed versus expected counts for each possible average spawning use score value.

To examine habitat use for chum salmon, we tested whether average sampling section values of individual habitat variables differed between spawning and non-spawning habitat during 2004 and 2005 by conducting a t-test (depth, velocity, substrate) or Wilcoxon rank sum test (hyporheic temperature) for each variable. The rank sum test was conducted for hyporheic temperature because of data non-normality. All analyses were conducted within year and at two spatial scales, across and within reaches. In these analyses of the entire study area, spawning was defined using the 5% spawning criterion for 2004 and 2005 spawning data. We also conducted a forward stepwise linear regression analysis (probability to enter and leave of 0.5 and 1.0, respectively) of the four-year average spawning use score as an overall summary of spawning relationships with habitat variables. We used a similar approach to examine differences in chum salmon spawning habitat use among reaches. For analyses of 2004 sampling section use, we used the 2% spawning criterion because sample sizes were extremely low in some reaches. For analyses of 2005 sampling sections, we used the 5% criterion. For fall Chinook salmon, we conducted t-tests on habitat values of individual spawning and non-spawning redd points across and within reaches.

In all statistical analyses, we considered differences to be significant at $P < 0.05$. JMP (Version 6, 2005, JMP USA, SAS Worldwide Headquarters, Cary, North Carolina) was used for all two-sample tests and Systat (Systat 8.0, 1998, SPSS Inc., Point Richmond, California) was used to construct boxplots and normal probability plots.

We used two approaches to compare predicted and actual spawning habitat use by chum and fall Chinook salmon. First, we constructed a confusion matrix of actual versus predicted spawning use per sampling section for each species for each year of available data (Kohavi and Provost 1998). A confusion matrix summarizes the outcome of a binary prediction model, where two types of prediction errors are possible, false positives and false negatives. Predicted spawning per sampling section was based on spawning habitat limits established in Section 5.1. Sections were predicted as suitable for spawning based on 1) all four habitat variables combined (ALL), 2) velocity, depth, and substrate combined (VDS), and 3) hyporheic temperature only (T_H). For each set of predictions, we considered a section suitable for spawning if 25% or more of available area was suitable for all variables in the set. This limit was applied because 25% of the available area is sufficient habitat area for the number of redds observed in the section. For each set, we calculated prediction success (the proportion of the total that is predicted correctly) and Cohen's kappa (κ), a measure of the proportion of all possible cases that are predicted correctly after accounting for chance effects (Manel et al. 2001). κ has the advantage of being relatively insensitive to prevalence (the proportion of positive spawning occurrences), unlike prediction success (Manel et al. 2001). We applied the evaluation criteria of Landis and Koch (1977) for κ: poor $\kappa < 0.40$, good $0.40 < \kappa < 0.75$, excellent $\kappa > 0.75$.

Our second approach was to calculate a modified weighted usable area (WUA) habitat index of spawning area per section based on methods used to predict habitat use (Section 5.1). The WUA is a combined rating of habitat quality based on individual ratings for each habitat characteristic (Bovee 1982). We examined the correlation between WUA estimates and spawning use, consistent with Gallagher and Gard (1999) and Beecher et al. (2002), across and within stream reaches. This method has

the advantages that it is not based on the somewhat arbitrary classification of spawning use or the limitations of a binary classification, and it can be conducted at multiple nested scales with different criteria per scale. The WUA index value per sampling section was derived from habitat quality index values for each habitat variable weighted by section area. We did not factor habitat quantity into the final WUA metric because our sections were similar in area. Also, we expected spawning habitat use to relate more to habitat quality than quantity because spawner density per sampling section was low for most sections and spawning area was not limiting.

5.2.2 Results

5.2.2.1 Chum Salmon Spatial Spawning Consistency, 2002–2005

Spawning use score correlations were highly significant for all year pairs except 2002–2004 (Table 5.14), indicating that chum salmon tend to spawn in consistent locations across years. Locations that received a high frequency of spawning use across the four years included Crazy Johnson, the West Fork immediately above and below Crazy Johnson, and several discrete locations in the Grays River (Figure 5.14). Across all four years, the observed and expected average spawning use score frequency distributions were significantly different ($\chi^2 = 76.3$, N = 8, $P < 0.005$). Observed spawning produced higher than expected frequencies of average spawning use scores of 0.0, 1.25, and 1.50, and lower than expected frequencies of 0.25 and 0.50 (Figure 5.15). Spawning was therefore more consistent than expected, with more sections having no spawning (average score = 0.0) or more spawning than expected (score = 1.25 or 1.50) across the four years, and fewer sections than expected of the predicted most likely spawning occurrence score under a random distribution (score = 0.5).

Table 5.14. Pearson Correlation Coefficients of Spawning Scores Among Year Pairs for 2002 Through 2004

	2003	2004	2005
2002	0.321*	−0.022	0.272*
2003		0.295*	0.459*
2004			0.394*
* Highly significant at $P < 0.01$.			

5.2.2.2 Chum Salmon Spawning Habitat Selection – 2004, 2005

Average velocities in chum salmon spawning sections in 2004 and 2005 were 31.9 and 45.8 cm/sec, respectively and average depths in these years were 35.8 and 61.6 cm, respectively (Table 5.15). Average substrate d_{84} values were 45.0 and 40.9 mm, and d_{50} values were 21.2 and 17.8 mm in these years, respectively. Average hyporheic temperatures in chum spawning sections in 2004 and 2005 were 8.4°C and 8.6°C, respectively.

In 2004 and 2005, tests of individual habitat variables across all reaches combined indicated that spawning and non-spawning sections differed in hyporheic temperature, with average hyporheic temperature in spawning areas 0.6°C and 1.2°C warmer, respectively, than non-spawning areas (Table 5.16, Figure 5.16). In 2004, spawning and non-spawning areas also differed in depth, with spawning areas 10.2 cm shallower than non-spawning areas (Table 5.16, Figure 5.17).

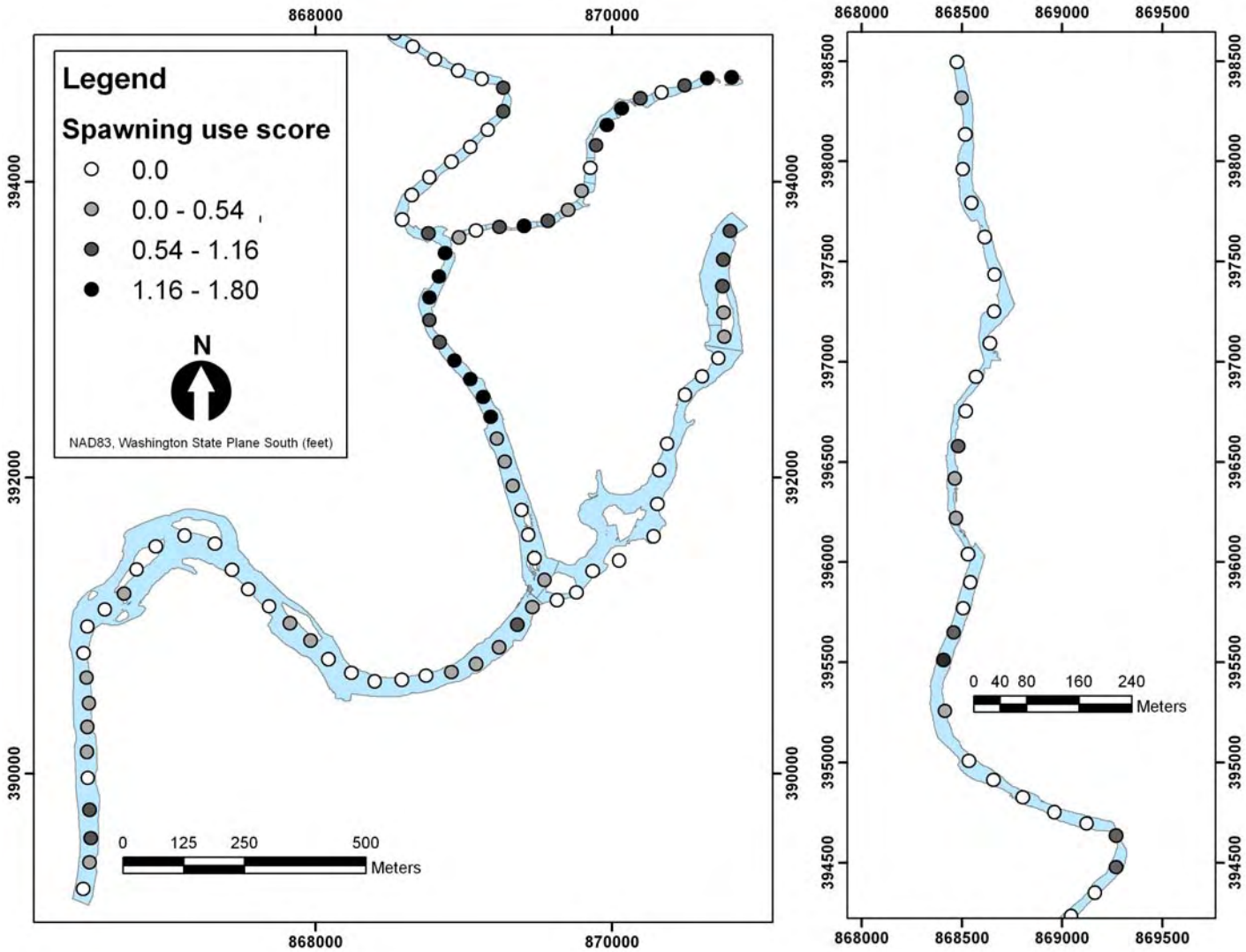


Figure 5.14. Distribution of Chum Salmon Spawning During 2002 Through 2005. Four-year average spawning score is shown. Possible values range from 0.0 (no use) to 2.0 (highest use).

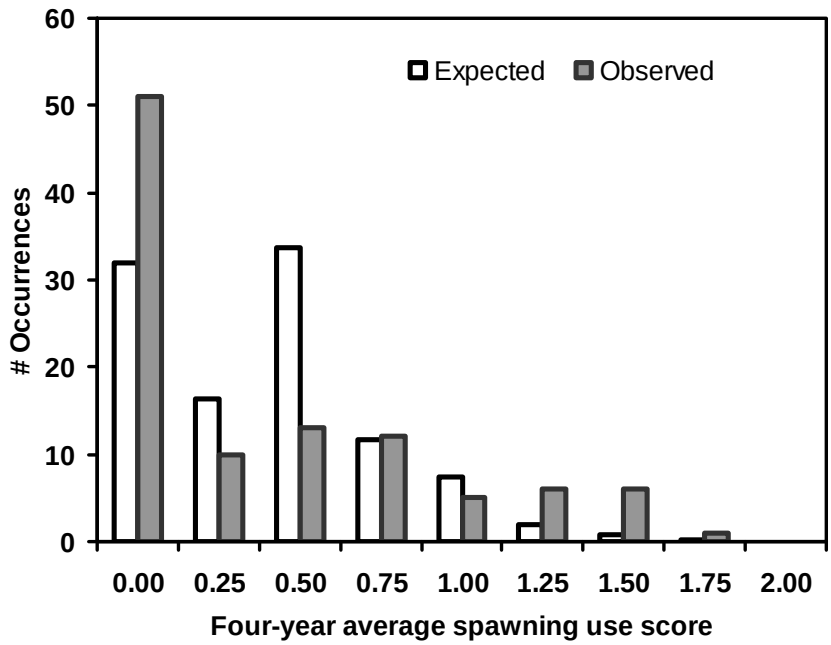


Figure 5.15. Observed (gray) and Expected (white) Frequency Distributions of Possible Chum Salmon Average Spawning Use Scores During 2002 Through 2005. Possible spawning score values range from 0.0 (no use) to 2.0. N = 104 stream sections; several sections were excluded due to missing data for one or more years. Observed use was higher than expected for value 0.0 (no use) and values 1.25–1.75 (highest use observed), indicating a non-random spatial distribution of spawning use among the four years.

Table 5.15. Chum Salmon Spawning Habitat Characteristics in the Grays River Study Area in 2004 and 2005. Velocity and depth estimates were produced by modeling the 90% and 50% exceedance flows, respectively.

Habitat Variable	Mean (range)	
	2004	2005
Velocity (cm/sec)	31.9	45.8
	(0.2–81.2)	(1.2–98.4)
Depth (cm)	35.8	61.6
	(6.2–78.6)	(11.2–155.5)
d_{84} (mm)	45.0	40.9
	(15.0–95.2)	(15.0–86.6)
d_{50} (mm)	21.2	17.8
	(2.0–51.5)	(2.0–44.2)
T_H (°C)	8.4	8.6
	(5.8–11.2)	(5.5–11.0)

Table 5.16. Habitat Differences (Spawning - Non-Spawning) Between Chum Salmon Spawning and Non-Spawning Sampling Sections Across the Study Area and by Stream Reach During 2004 and 2005, with Spawning Areas Defined as Containing ≥5% of Total Spawning. *P* values presented are t-test probabilities.

Reach	N	T_H		Depth[a]		Velocity[a]		d_{84}	
		2004	2005	2004	2005	2004	2005	2004	2005
Study Area	109[b]	**0.6***	**1.2*****	**-10.2***	-2.4	-2.4	-8.6	3.2	-3.9
Crazy Johnson	16	-0.2	0.0	-16.8	-2.5	8.4	-10.4	**16.7***	-13.0
Upper West Fork	28	0.6	**1.7*****	-6.0	7.4	1.1	-5.9	-3.2	-2.1
Lower West Fork	15	**1.5****	**1.7****	**9.2***	**16.8*****	-4.8	0.9	-0.9	4.1
Upper Grays	11	-0.2	-0.6	**-26.1***	**-29.9***	19.0	3.7	18.7	5.8
Lower Grays	33	**0.9***	0.9	10.9	10.9	5.1	-1.9	0.7	-4.1

(a) Estimates for 2004 and 2005 produced by the MASS2 model with 90% and 50% exceedance flow inputs, respectively.
(b) N for d_{84} = 105.
Asterisks denote *P* values, P < 0.05*, P < 0.01**, or P < 0.001***.
T_H = hyporheic temperature, d_{84} = substrate particle size.

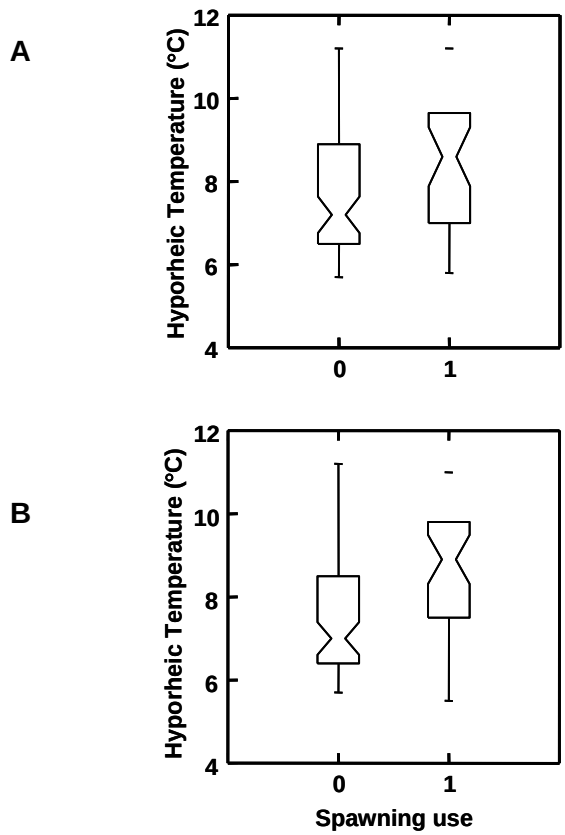


Figure 5.16. Hyporheic Temperature in Chum Salmon Non-Spawning (0) Versus Spawning (1) Sections During 2004 (A) and 2005 (B). Differences are highly significant (*P* < 0.04, 0.0002, respectively). The medians of the distributions are shown as the centers of the notches and the lower and upper quartiles are illustrated as hinges of the box plots. Whiskers indicate the range of values within 1.5 * the interquartile range. The notches represent an approximate 95% confidence interval around the median.

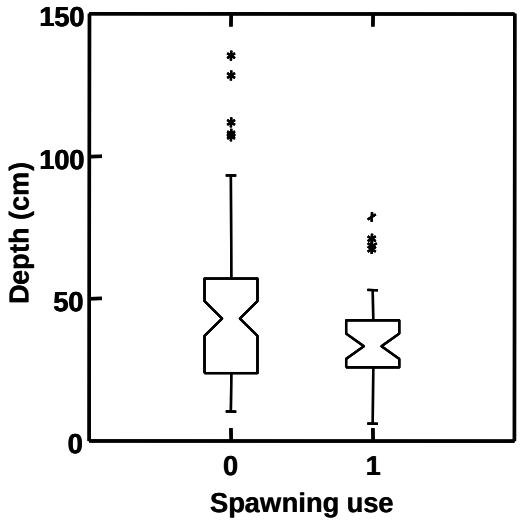


Figure 5.17. Depth in Chum Salmon Non-Spawning (0) and Spawning (1) Sections During 2004. Difference is highly significant ($P < 0.02$). The medians of the distributions are shown as the centers of the notches and the lower and upper quartiles are illustrated as hinges of the box plots. Whiskers indicate the range of values within 1.5 * the interquartile range. The notches represent an approximate 95% confidence interval around the median.

In the stepwise linear regression of the four-year average spawning use score with all four habitat variables, hyporheic temperature and substrate (d_{84}) were significantly associated with spawning use, with higher hyporheic temperatures and larger substrates associated with more consistent use (N = 100, F = 11.03, $P < 0.0001$, $r^2 = 0.288$, T_H: $P < 0.0001$, d_{84}: $P < 0.04$; Figure 5.18). Depth and velocity were marginally nonsignificant (depth: $P < 0.08$, velocity: $P < 0.06$), with spawning use associated with shallower depths and slower velocities.

Tests of individual variables within reaches revealed chum salmon spawning habitat differences among reaches. In three of the five reaches (upper West Fork [2005 only], lower West Fork [both years], lower Grays [2004 only]), hyporheic temperature was significantly higher in spawning than non-spawning sections and was the only difference between spawning and non-spawning sections in the upper West Fork and the lower Grays (Table 5.16). In the lower West Fork, hyporheic temperature and depth were significantly different in both years. In the upper Grays, depth was the only habitat variable that differed between spawning and non-spawning areas. Comparison of depth differences in these two reaches revealed that spawning areas were shallower in the upper Grays and deeper in the lower West Fork than non-spawning areas (Figure 5.19). Crazy Johnson spawning habitat characteristics were different from all other reaches. In 2004, substrate differed significantly between spawning and non-spawning sections, with spawning section substrate d_{84} 16.7 mm larger than non-spawning areas, but in 2005, no habitat variables differed significantly between spawning and non-spawning sections (Table 5.16).

5.2.2.3 Fall Chinook Salmon Spawning Habitat Selection – 2005

Average velocity, depth, and substrate d_{84} at fall Chinook salmon redds were 73.3 cm/sec, 79.0 cm, and 44.6 mm, respectively (Table 5.17). Average hyporheic temperature associated with fall Chinook salmon redds was 7.8°C.

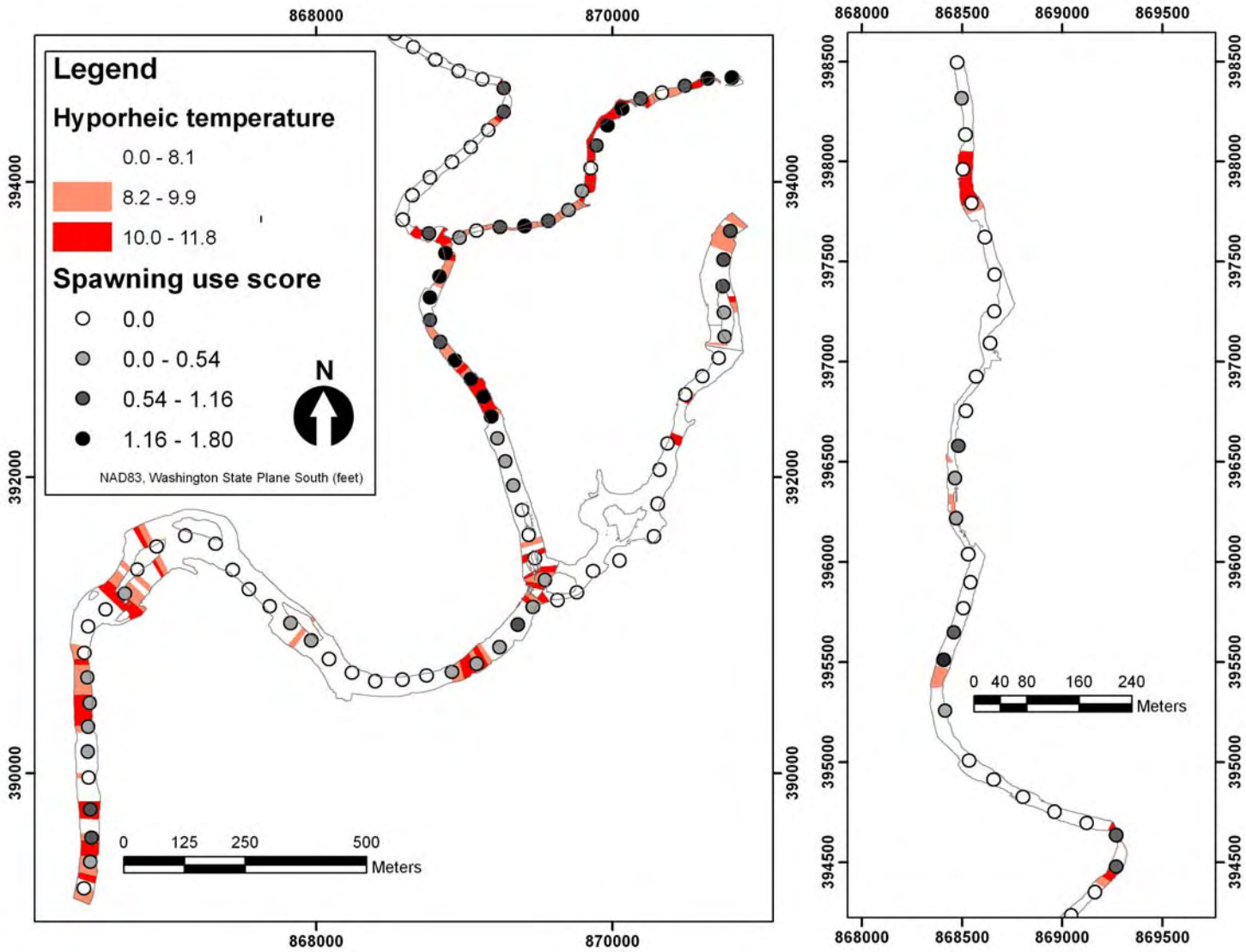


Figure 5.18. Hyporheic Temperature and Chum Salmon Four-Year Average Spawning Use Score Across the Grays River Study Area. Possible spawning score values range from 0.0 (no use) to 2.0 (highest use).

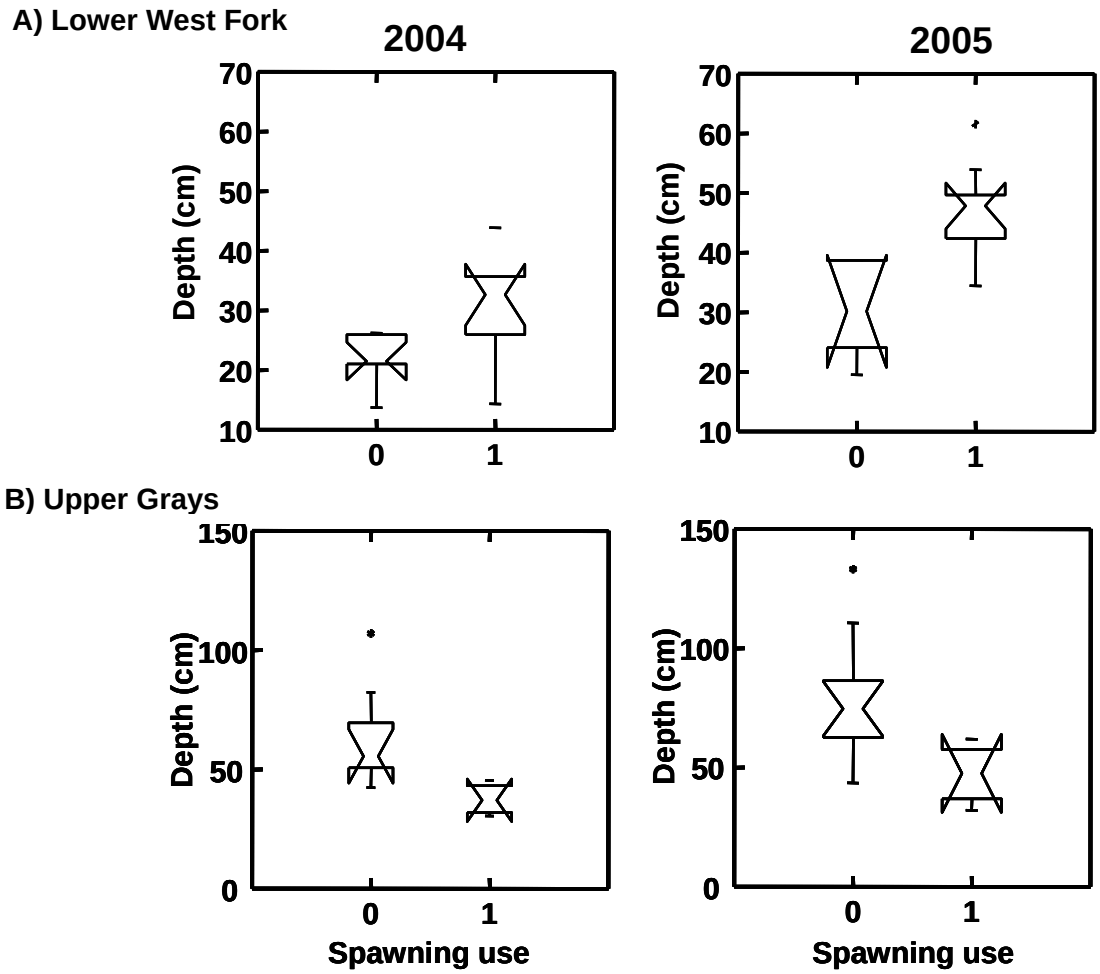


Figure 5.19. Depth in Chum Salmon Non-Spawning (0) and Spawning (1) Sections in the Lower West Fork (A) and Upper Grays (B) During 2004 (left) and 2005 (right). All differences are significant ($P < 0.02$ or less). The medians of the distributions are shown as the centers of the notches and the lower and upper quartiles are illustrated as hinges of the box plots. Whiskers indicate the range of values within 1.5 * the interquartile range. The notches represent an approximate 95% confidence interval around the median.

In t-test analyses with all redds combined, no differences between redd and non-redd locations were detected for any of the four habitat variables. Similarly, no differences were detected between redd and non-redd locations for any of the four variables in the upper West Fork. For redd locations in the lower Grays, velocities were significantly faster (20.1 cm/sec; N = 16, t-statistic = 2.11, $P < 0.05$) than non-redd locations (Figure 5.20).

Fall Chinook salmon did not appear to select spawning habitat based on hyporheic temperature. Both of the two fall Chinook salmon redds mapped during 2003 were in sections with hyporheic temperatures similar to river temperatures. In contrast, 10 of the 15 fall Chinook salmon redds mapped in 2005 were in areas with elevated hyporheic temperature (6 of 8 in the lower Grays, 4 of 7 in the upper West Fork).

Table 5.17. Fall Chinook Spawning Habitat Characteristics in the Grays River Study Area in 2005, Based on 15 Redds Located in 2003 and 2005. Velocity and depth estimates are those produced by modeling the 50% exceedance flow.

Habitat Variable	Mean	Range
Velocity (cm/sec)	73.3	51.4–115.2
Depth (cm)	79.0	29.6–160.2
d_{50} (mm)	19.9	7.3–28.3
d_{84} (mm)	44.6	19.4–56.3
T_H (°C)	7.8	6.0–11.5

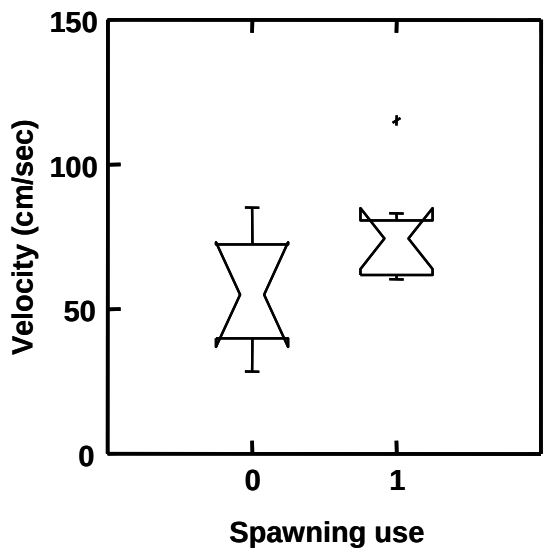


Figure 5.20. Velocity Difference Between Fall Chinook Salmon Non-Spawning (0) and Spawning (1) Areas in the Lower Grays River. Difference is significant ($P < 0.05$). N = 15 redds. The medians of the distributions are shown as the centers of the notches and the lower and upper quartiles are illustrated as hinges of the box plots. Whiskers indicate the range of values within 1.5 * the interquartile range. The notches represent an approximate 95% confidence interval around the median.

5.2.2.4 Predicted Versus Actual Spawning Habitat Use

The confusion table metrics prediction success and κ indicate moderate to poor success in predicting chum salmon spawning use by stream section when κ is evaluated using Landis and Koch's (1977) scoring system (Table 5.18). The binary hyporheic variable, T_H was almost as effective at predicting spawning use as all four variables combined (ALL), and was considerably more effective than using velocity, depth, and substrate combined (VDS) in both years. In 2004, prediction success of T_H, ALL, and VDS were 57.4, 58.3, and 32.4%, respectively. Our ability to predict chum salmon spawning use using T_H and ALL was generally better during 2005 than 2004—prediction success of T_H, ALL, and VDS in 2005 were 68.5, 69.4, and 31.5%, respectively (Table 5.18).

Table 5.18. Predictive Ability of Spawning Habitat Use by Stream Section for Chum and Fall Chinook Salmon for 2004 and 2005. Confusion matrix measures of occurrence (prevalence) and predictive ability (prediction success, Cohen's kappa [κ]) are presented. Prevalence and prediction success are presented as percentages.

	Year	Prevalence	Confusion Matrix Metric	Habitat Variables		
				All	VDS	T_H
Chum	2004	32.4	Prediction Success	58.3	32.4	57.4
			κ	0.12	−0.16	0.13
	2005	34.2	Prediction Success	69.4	31.5	68.5
			κ	0.36	−0.18	0.36
Fall Chinook	2005	17.9	Prediction Success	38.8	52.2	31.3
			κ	−0.02	0.02	−0.11
VDS = velocity, depth, substrate, T_H = hyporheic temperature; All = V,D,S, T_H.						

Correlations were highly significant and positive between our WUA metric and chum salmon spawning distribution across the study area during 2004 and 2005 (Table 5.19). The only habitat quality factor contributing to the WUA metric that was also significantly correlated with spawning distribution was hyporheic temperature in both 2004 and 2005 (Table 5.19). Depth was marginally nonsignificant in 2005.

Correlations between our WUA metric and chum salmon spawning use were highly significant and positive in most but not all of the study reaches (Table 5.19). In 2004, WUA was significantly correlated with spawning in the upper West Fork and marginally nonsignificantly correlated ($P < 0.09$) with spawning in the lower West Fork but not correlated with spawning in the remaining three reaches. Habitat factors significantly correlated with WUA in 2004 included velocity in Crazy Johnson, depth in the lower West Fork, and hyporheic temperature in both reaches of the West Fork. In 2005, WUA was highly significantly correlated with spawning in all reaches except the upper Grays. The only habitat factor also significantly correlated with WUA in 2005 was hyporheic temperature, in all reaches except Crazy Johnson (marginally nonsignificant; $P < 0.06$) and the upper Grays (nonsignificant).

Fall Chinook salmon prediction success and κ values indicate low success in predicting spawning use. Using Landis and Koch's (1977) scoring system for κ, our predictive ability was "poor." In contrast to chum salmon, combined velocity, depth, and substrate (VDS) was more effective than all four variables combined (ALL) or the hyporheic behavior variable (T_H; Table 5.18). Prediction success of the three prediction metrics, VDS, ALL, T_H, were 52.2, 38.8, and 31.3%, respectively.

The WUA approach for comparing predicted and actual spawning indicates a lack of ability to correctly predict the locations of fall Chinook salmon spawning. Correlations were nonsignificant between our WUA metric and fall Chinook salmon spawning distribution among sections across the study area (N = 67) and in the two reaches in which spawning occurred, the upper West Fork (N = 34) and the lower Grays (N = 33), during 2005. No correlations between spawning use and any of the four contributing habitat quality factors were significant across the study area or in either of the two reaches in which spawning occurred.

Table 5.19. Correlation Between Chum Salmon Spawning Count During 2004 and 2005 and Weighted Useable Area (WUA) Score and the Habitat Factors Contributing to the WUA Score. Values presented are significance probabilities (P) ≤ 1.0. P values are presented only if correlation coefficients are positive indicating a positive association between spawning use and habitat quality.

	N		WUA		Velocity		Depth		Substrate		T_H	
	2004	2005	2004	2005	2004	2005	2004	2005	2004	2005	2004	2005
All	107	108	**0.002**	**0.0002**	NS	NS	0.08	NS	NS	NS	**0.006**	**<0.0001**
Crazy Johnson	15	16	NS	**0.007**	**0.01**	NS	NS	NS	NS	NS	NS	0.06
Upper West Fork	34	30	**0.001**	**0.003**	NS	NS	0.09	NS	NS	NS	**0.03**	**0.0005**
Lower West Fork	15	15	0.09	**0.009**	NS	NS	**0.03**	NS	NS	NS	**0.01**	**0.005**
Upper Grays	10	14	NS	NS	NS	NS	1.0	NS	NS	NS	NS	NS
Lower Grays	33	33	NS	**0.008**	NS	NS	NS	NS	NS	NS	NS	**0.006**

WUA= wetted useable area score, T_H = hyporheic temperature, NS = nonsignificant.

5.2.3 Discussion

Chum salmon spawn in consistent locations in the Grays River. During 2002 through 2005, spawning chum salmon were more likely to use specific areas each year than would be expected due to chance. Spawning score correlations were highly significant for all year pairs except 2002–2004, and more stream sections had either no spawning or more spawning than was predicted from a random use model. Our findings are consistent with others (Hunter 1959; Hoopes 1971; Fukushima 1994; Fukushima and Smoker 1998), who found that salmonid species including chum, sockeye (*O. nerka*), pink salmon (*O. gorbuscha*), and taimen (*Hucho perryi*) spawned in consistent locations across years. Salmonid spawning has been associated with specific river characteristics, such as the head and tail or riffle structures associated with a heterogeneous bedform and islands (Bjornn and Reiser 1991; Leman 1993; Geist and Dauble 1998; Fukushima 2001). These areas of morphological diversity are associated with active hyporheic exchange with surface water (Geist and Dauble 1998; Fukushima 2001). Chum salmon likely return to specific areas in the Grays River because they consistently have the hyporheic characteristics chum salmon seek.

Hyporheic temperature was a more important habitat factor for spawning chum salmon than velocity, depth, or substrate in the Grays River. The importance of hyporheic temperature, often associated with upwelling hyporheic flow, has been recognized by others (Geist and Dauble 1998). Some have suggested that the presence of upwelling water may be the single most important factor in salmonid spawning habitat selection (Burner 1951; Leman 1993). In Section 5.1, we found that most (89.5%) of the wetted area within the study area was within chum salmon use ranges for velocity, depth, and substrate. Our results do not suggest that velocity, depth, and substrate are unimportant to chum salmon. Rather, and consistent with Geist and Dauble (1998), our results suggest that in the Grays River these variables are ineffective at differentiating spawning from non-spawning habitat because suitable spawning conditions are widely available and chum salmon will spawn in a wide range of velocities, depths, and substrates. Hyporheic flow may be the most limiting factor in chum salmon spawning habitat because chum salmon typically spawn in small streams (Johnson et al. 1997) in which other habitat variables may be less limiting.

Chum salmon selected spawning habitats based on different habitat characteristics in different reaches within the Grays River study area. For example, depth was identified as an important habitat variable in two of the five study reaches, and in these reaches, chum salmon spawned in shallower than average habitats in the upper Grays and deeper than average habitats in the lower West Fork. The upper Grays is a diverse channel, of which a considerable portion is narrow, incised, deep, and unsuitable for chum salmon spawning. In contrast, the lower West Fork is a wide, homogeneous channel of moderate depth that is oversized for current flows since its abandonment by the Grays River following the 1999 avulsion. These reaches provide different proportions of depth in suitable ranges for chum salmon, and therefore chum salmon use of depth differs in the two reaches. That habitat use is a function of habitat availability is a well-recognized paradigm in ecology, and methods of habitat use characterization have recognized the dependence of use on availability (Baldrige and Amos 1982; Litvaitis et al. 1996). Our findings emphasize the importance of quantifying habitat availability as well as a stratified study design to recognize differences among streams and reaches that differ in morphology and hydrology.

Chum salmon spawn at high densities in Crazy Johnson in spite of the relatively low quality habitat it provides. This creek has lower velocities and depths and more fine particulate substrate than the other four reaches in the Grays River, and in Section 5.1, we rated 85.4% of the available area in the lowest two

quality categories for velocity, depth, and substrate (combined). This stream was also one of only two study area reaches in which we did not find a significant association with hyporheic temperature. For these reasons, chum salmon spawning in this creek may appear anomalous. However, chum salmon use of Crazy Johnson is likely associated with the influence of hyporheic upwelling on hydrology and water quality, in spite of the lack of association we found. Crazy Johnson is a springbrook emerging from within the Grays River floodplain, with river water temperatures that were 1°C to 4°C warmer than in adjacent reaches during our sampling. Habitat associations with spawning in Crazy Johnson differed from the other reaches. Within relatively poor-quality habitat, chum salmon may be spawning in higher-quality areas, because we found some association between chum salmon spawning and higher velocities and deeper areas. Chum salmon have previously been documented spawning in areas covered by up to 30 cm of fine sediment when hyporheic upwelling was present (Vining et al. 1985). Concentrated chum salmon spawning in Crazy Johnson may also be associated with higher channel stability and lower vulnerability to scour than other Grays River spawning areas because its watershed is small and it is rarely inundated by Grays River flood flows (see Chapters 3 and 4). Ultimately, the concentrated use of Crazy Johnson for chum salmon spawning may emphasize the importance of the availability of elevated hyporheic temperatures and channel stability to successful chum salmon spawning.

We did not find associations between fall Chinook salmon spawning in the Grays River study area during 2003 and 2005 and any of the habitat characteristics most commonly used to characterize spawning areas for this species. We did not detect differences between redd sites and non-redd sites based on velocity, depth, substrate, or hyporheic temperature across the study area. The only association we detected between fall Chinook salmon spawning and the habitat variables we examined was with velocity in the lower Grays, where fall Chinook salmon selected redd sites that were of higher velocity than randomly selected non-redd sites. Although associations between hyporheic exchange and fall Chinook salmon spawning have been well documented (Geist et al. 2000; Geist et al. 2002), we did not find that fall Chinook salmon redd sites differed in hyporheic temperature from non-spawning sites. The specific nature of the hyporheic environment preferred by fall Chinook salmon is not fully understood, and this species has been associated with both upwelling and downwelling areas (often associated with cold and warm hyporheic temperatures, respectively). However, unlike in this study, other studies have linked fall Chinook salmon spawning with areas of either elevated or non-elevated hyporheic temperatures but not both, within a particular stream system.

Our study found no evidence that chum and fall Chinook salmon segregate based on differences in hyporheic temperature, in contrast to Geist et al. (in review). These authors suggested that fall Chinook salmon may select downwelling areas where hyporheic and river temperatures do not differ when spawning in sympatry with chum salmon to reduce competition for redd sites and avoid redd destruction by the later-arriving chum salmon. Although we found that chum salmon, as expected, tended to spawn in areas with elevated hyporheic temperatures, we found no association between fall Chinook salmon spawning areas and hyporheic temperature. However, differences in velocity and depth preferences in the Grays River study area may have contributed to spatial segregation of these species. In Section 5.1, we found that fall Chinook salmon spawned in deeper and faster water than chum salmon. Fall Chinook salmon are larger than chum salmon and therefore have a greater ability to withstand higher velocities during redd construction. Because of data limitations associated with our spawning data, we are unable to confirm spatial segregation between the species, and potential mechanisms contributing to segregation are speculative.

We were moderately successful in predicting chum salmon spawning locations when hyporheic temperature was a predictor variable. In Section 5.1, hyporheic temperature largely determined our spawning location predictions because it was the most limiting variable. Of the total wetted area available, approximately 89.5% was suitable for spawning with respect to velocity, depth, and substrate (combined), whereas only 33.9% was characterized by elevated hyporheic temperature. Our ability to predict spawning locations more precisely was limited by the resolution of our spawning and hyporheic data. Based on our findings, predicting chum salmon spawning in other streams using hyporheic temperature alone should result in moderate prediction success and could potentially be increased by increasing the resolution of the hyporheic temperature data. Locating areas of elevated hyporheic temperature, often associated with upwellings, can be challenging because hyporheic exchange is driven by multiple processes of varying scale, from fine scale (White 1990) to the habitat unit scale (Vaux 1962; Thibodeaux and Boyle 1987; Cardenas et al. 2004) to the stream reach scale (Brunke and Gonser 1997; Baxter and Hauer 2000; Harvey and Wagner 2000). For example, examination of our 2002 and 2003 chum salmon redd location data revealed spawning on the stream margins (PNNL unpublished data) that may be associated with localized upwelling. Our 2004 and 2005 spawning data and hyporheic temperature methods were unable to capture these areas of lateral variability within the stream channel. However, quantification and understanding of geomorphic influences on hyporheic flow and salmonid spawning are improving (Geist and Dauble 1998; Poole et al. 2006) and may increasingly contribute to prediction of salmonid spawning site selection.

We were not able to successfully predict fall Chinook salmon spawning locations in the Grays River study area. In the Grays River, 44.5% and 28.5% of the upper West Fork and lower Grays, respectively, was classified as suitable for these four habitat characteristics (Section 5.1). That we had poor ability to predict the locations of fall Chinook salmon redds is not surprising for two reasons. First, most of the available wetted area has suitable velocity, depth, and substrate for fall Chinook spawning (Section 5.1). Most redd and non-redd locations were within useable ranges for these habitat variables. Second, we had little predictive ability because our redd location data were limited (17 redds), and we did not find associations between habitat characteristics widely associated with salmon spawning, on which our predicted redd locations were based. The habitat factor most limiting available spawning area, hyporheic temperature, does not appear to be limiting for fall Chinook. Either this species is willing to use areas with and without elevated hyporheic temperatures in the Grays River study area, or some other hyporheic characteristic, such as specific discharge, is influencing redd site selection.

In Section 5.1, we concluded that available spawning area with velocity, depth, substrate, and hyporheic characteristics suitable for chum salmon appeared to be sufficient to support historic chum salmon populations in the Grays River study area. Assuming that all four habitat variables must be within the suitability ranges we established in Section 5.1, redd capacity ranged from 5,590 to 22,405 redds, with a capacity of 695 to 2,786 redds in the highest quality habitat (Table 5.12). This study includes several findings pertinent to our redd capacity estimates in Section 5.1. First, it may be appropriate to focus on high-quality (probable use) habitat, because for example, we found that chum salmon generally avoided deeper areas that are otherwise suitable. If this is the case, sufficient high-quality habitat is not available to support historic chum salmon runs. Second, of the Pacific salmon, chum salmon are characterized by closer inter-redd spacing and redd superimposition (Burner 1951). Therefore, it is unlikely that chum salmon in the Grays would spawn only at the largest redd size/lowest density we considered, and the lower redd capacity values we presented in Section 5.1 may therefore be underestimates. Third, we considered any area with hyporheic temperatures significantly warmer than river water to be suitable for

chum salmon spawning. The magnitude and other characteristics of hyporheic exchange desired by chum are not fully understood. In other studies, spawning salmonid site selection has been associated with substrate permeability and hyporheic seepage rate (Carline 1980; Geist 2000). Many areas within the study area with elevated hyporheic temperatures and within other habitat variable limits were not used for spawning. Lack of use of these areas may be due to current chum salmon underpopulation of the Grays River or to some hyporheic flow inadequacy that we were unable to identify.

The primary cause of Grays River chum salmon decline may be spawning habitat quality rather than quantity. The Grays River is in a degraded condition, with many reaches (including spawning reaches) classified as moderately impaired or impaired by the Integrated Watershed Assessment process used in the Pacific Northwest watershed planning program (LCFRB 2004). Characteristics of the Grays River and its tributaries that may contribute to degraded spawning habitat quality include excess bedload, channel and bank erosion, and channel instability (Chapter 4; LCFRB 2004; Herrera 2005). These habitat alterations contribute increased fine sediments loads (Herrera 2005) that are detrimental to salmonid incubation success (Cedarholm et al. 1981; Tagart 1984; Scrivener and Brownlee 1989). Fine sediment reduces water flow through the redds, reducing water quality (summarized in Chapman 1988). Excess bedload including fine and coarse sediments may bury redds, reducing water quality and entombing juvenile salmonids as they attempt to emerge (Lapointe et al. 2000). Scour destroys redds, washing eggs downstream (Montgomery et al. 1996; DeVries 1997; Lapointe et al. 2000). Future study of chum salmon population decline and restoration alternatives in the Grays River should include identification of the nature, location, and extent of 1) successful chum salmon production and 2) threats to incubation success.

Restoration and protection of hyporheic exchange may benefit chum salmon in the Grays River system. Hyporheic exchange is the habitat factor most limiting spawning area for chum salmon as long as flow conditions create velocities and depths within suitable ranges. Land use effects in the upper Grays watershed and study area have altered habitat-forming processes, potentially affecting bedform and subsurface flow. Loss of large woody debris and sinuosity and excess bedload deposition have likely homogenized the Grays River channel bedform. Because these factors contribute to bedform diversity (Montgomery et al. 1995; Fukushima 2001) and therefore hyporheic exchange, their management in the Grays River may contribute to increased hyporheic exchange and, hence, total available spawning area for chum salmon. This may be important if substrate permeability is currently limiting incubation success of chum salmon in the Grays River. Salmonids select redd sites that have strong subsurface flow, maintaining high water-quality conditions for incubating juveniles (Shumway et al. 1964; Lapointe et al. 2004). It may also be important if reduced bedform diversity has resulted in reduced magnitude of upwelling flows. Reduced hyporheic exchange may weaken the chemical and physical cues chum salmon use to select spawning sites, reduce incubation success, and decrease the extent of suitable spawning area.

The relative consistency in chum salmon spawning among years suggests that restoration and protection efforts could target the spawning areas we identified. While availability of spawning habitat does not appear to be limiting for chum salmon, the specific causes of chum salmon decline are unknown. However, known habitat deficiencies in the Grays system include lack of large woody debris, unstable stream banks and bed, and excess bedload (LCFRB 2004; Herrera 2005). Prior to full understanding of the causes of the Grays River chum salmon decline, restoration of habitat forming processes associated with these habitat conditions could be initiated. These could include improvements in upper watershed timber harvest and road management practices to reduce fine sediment loads, streambank stabilization to enhance instream hydraulic and habitat diversity and channel stability, and floodplain and riparian

reforestation to reduce scouring flows and restore large woody debris recruitment processes. Committed restoration efforts will be required to reverse the declining trend in the chum salmon spawning in the Grays River. Although we identified important areas for restoration, watershed-wide management actions will be required to restore natural processes in the Grays River (Doppelt et al. 1993; Beechie et al. 2005), and will also require improved understanding of the specific causes of the chum salmon decline.

5.3 Hyporheic and Geomorphic Characteristics of Chum Salmon Spawning Habitat

Chum salmon tend to spawn in areas of active hyporheic exchange with river water (e.g., Leman 1988; Bonnell 1991; Salo 1991; Leman 1993; Geist et al. 2002). Hydrologic factors influencing redd success include water flow and resultant water quality in the incubation environment (Shumway et al. 1964; Chapman 1988). Chum salmon tend to spawn in areas of warm upwelling water (Burner 1951; Kobayashi 1968; Rukhlov 1969; reviewed by Bjornn and Reiser 1991; Bonnell 1991; Geist et al. 2002) and may use some combination of a temperature difference between river and hyporheic water, accelerating flow exchange between the bed and river, and chemical cues to select spawning sites (Hara 1982; Crisp 1990; Salo 1991). Chum salmon also use areas of groundwater upwelling, in which their specific cues may differ (Rukhlov 1969; Leman 1993 and sources therein). Subsurface bed characteristics such as the pressure difference between surface and subsurface environments and substrate permeability may also affect chum salmon perception of hyporheic conditions and therefore spawning site selection.

Salmonid spawning has been associated with specific geomorphologic characteristics of the river channel. Geist and Dauble (1998) identified associations between salmon spawning and complex channels that tend to be characterized by geomorphic features such as multiple channels, bars and islands, diverse riffle:pool bedform, and areas of slope change. Features associated with spawning include the head or tail of riffle structures, heterogeneous bedform, channel concavity, and channels with high sinuosity (Vaux 1962; Bjornn and Reiser 1991; Leman 1993; Bauman 1998; Geist and Dauble 1998; Baxter and Hauer 2000; Geist 2000; Fukushima 2001; McHugh and Budy 2004; Hanrahan 2006; O'Brien 2006). Leman (1993) found that chum salmon tended to spawn on the inside of meander bends and others (Salo 1991; Fukushima and Smoker 1998) have found that chum salmon prefer to spawn at the lower end of riffles.

Chum salmon associations with geomorphic properties of the streambed are likely due to the strong role of geomorphology in inducing hyporheic exchange. Advective transport of surface water into the streambed due to pressure differences across the sediment surface have been well modeled (Vaux 1968; Thibodeaux and Boyle 1987), producing downwelling areas at the head of riffles and upwelling areas at riffle:pool transitions (Harvey and Bencala 1993; Hill et al. 1998; Kasahara and Wondzell 2003). Bedform-induced hyporheic exchange has the potential to occur at multiple scales associated with topographic diversity ranging from minor obstructions such as individual logs (White 1990) to broad-scale geomorphic knickpoints (Wroblicky et al. 1998). Other geomorphic properties of the stream channel that induce hyporheic exchange include channel and valley width (Stanford and Ward 1993), meander bends (Wroblicky et al. 1998; Cardenas and Wilson 2004), secondary channels (Wondzell and Swanson 1996), and buried paleochannels (Stanford and Ward 1988).

Although associations have been made between chum salmon spawning areas and geomorphic and hyporheic characteristics of the streambed, improved understanding of the hyporheic characteristics important to chum salmon, and geomorphic structures that induce those hyporheic characteristics, is needed. Improved quantification of the linkage between chum salmon and its spawning habitat, and elucidation of importation hyporheic-geomorphic mechanisms, will direct spawning habitat restoration and management of declining chum salmon populations. In this study, we hypothesized that chum salmon spawning sites would be associated with a hyporheic environment characterized by elevated subsurface temperatures relative to river water temperatures, upward head differentials (positive vertical hydraulic gradient [VHG]), and high substrate permeability. Further, chum salmon spawning sites are expected to be associated with riffle:pool bedform locations associated with hyporheic upwellings, such as descending slopes closer to riffle crests than pool bottoms. Our specific objectives were to 1) examine relationships between chum salmon spawning site selection and several characteristics of the hyporheic environment, 2) test whether geomorphic features including channel width:depth ratio, bed slope, and proximity to riffle crests differ between chum salmon spawning areas and non-spawning areas, and 3) test whether these geomorphic features were associated with elevated hyporheic temperatures.

5.3.1 Methods

5.3.1.1 Spawning Use

Spawning use data were collected in 2004 and 2005 and manipulated as described in Section 5.2. Consistent with Section 5.2, spawning data collected in 2004 were classified applying the 5% spawning criterion for study area-wide analyses and the 2% spawning criterion for within-reach analyses. All analyses with 2005 data were conducted using the 5% spawning criterion.

5.3.1.2 Hyporheic Characteristics

We quantified four metrics of hyporheic exchange. Hyporheic temperature (T_H) data were collected as described in Section 5.2. Vertical hydraulic gradient (VHG) is a unitless index of the head pressure difference between surface and subsurface environments (Freeze and Cherry 1979). VHG is positive when there is potential for flow from the hyporheic zone into the river (upwelling) and negative when there is potential for flow from the river into the hyporheic zone (downwelling). Hydraulic conductivity (K), the potential flow rate through porous media, was quantified by conducting slug tests (Butler 1998) along the thalweg of each reach in conjunction with VHG measurements. The geometric mean was calculated from the three K estimates per site. Specific discharge (q), also know as the apparent velocity, was calculated as the product of VHG and K to provide a simplified estimate of the flow rate within the hyporheic zone (see below).

VHG and hydraulic conductivity (K) were mapped during November and December, 2005. Data were collected at 37 sites on the Grays River, 57 sites on the West Fork, and 17 sites on Crazy Johnson. Sample points were located every 50 m adjacent to the thalweg throughout the five study reaches, and locations were recorded with a GPS. During the sampling period, discharge ranged from 255 to 468 cfs (approximately the 90% exceedance flow) at the Grays River gauge station 1.5 km downstream from the study area (Washington Department of Ecology Site 25B060).

Slug tests and VHG measurements were conducted inside 180-cm-long stainless steel piezometers. Each piezometer had a 30-cm-long screened interval. Piezometers were installed such that the top of the

screen was 15 cm below the surface of the riverbed to the approximate depth of a chum salmon redd egg pocket. The screens had a slot size of 0.038, which corresponds to an open area of 168 cm^2 per screen. The piezometer was pounded into the riverbed following the methods of Geist et al. (1998). Following piezometer installation, the piezometer was developed by pumping approximately 2.5 L of river water from the piezometer.

To perform a slug test, an airtight pressure-regulating wellhead assembly was threaded to the top of the piezometer. The assembly consisted of a 5.0-cm ball valve coupled to a 20.0-cm section of schedule-40 PVC pipe containing a small valve stem for pressurizing. A data logging pressure transducer (PT2X, Instrumentation Northwest, Inc., Kirkland, Washington) was lowered into the piezometer to measure the change in hydraulic head during the test. A slug test consists of pressurizing the piezometer interior, depressing the water level inside the piezometer, releasing the pressure, and recording the pressure response (rising water level) over time. Based on preliminary results and on past research in similar sedimentary environments, intragravel flow was assumed to be laminar (Reynolds numbers less than unity; Vaux 1968). The fundamental hydraulic response of hyporheic flow to the slug tests was overdamped (i.e., approximating an exponential decay response), and was therefore analyzed using the Bouwer and Rice method (Bouwer and Rice 1976; Bouwer 1989; Butler 1998; Weight and Wittman 1999).

Hydraulic conductivity was determined using the equation

$$K = \frac{r_c^2 \ln(R_e/R)}{2L_e} \frac{1}{t} \ln\left(\frac{H_0}{H_t}\right) \tag{5.1}$$

where r_c = radius of the well casing (cm),
R_e/R = dimensionless ratio of radius of gravel envelope to distance away from the well over which the average value of K is being measured (obtained as outlined in Fetter 1994)
L_e = length of the screen or open section of the well (cm)
H_t = drawdown at time t
H_0 = drawdown at time t = 0
t = time from H_0.

Slug tests were summarized using the geometric mean hydraulic conductivity (Butler 1998). Standard error in hydraulic conductivity measurements was computed as the sum of the standard error associated with linear regressions using the Bouwer and Rice method and the standard deviation of replicate samples.

VHG was determined as the ratio of the difference in head pressure between the hyporheic zone and river to the distance from the riverbed to the top of the piezometer screen:

$$\text{VHG} = \frac{\Delta h}{L} = \frac{[\text{hyporheic head - river head}]}{\text{distance from riverbed to top of piezometer screen}} \tag{5.2}$$

where Δh was determined by suspending a pressure transducer (PT2X, Instrumentation Northwest, Kirkland, Washington) from the top of the piezometer and recording hyporheic and river head, and determining the difference between the two head values. PT2X sensors record water level with an accuracy of ± 0.6 cm.

5.3.1.3 Geomorphic Characteristics

We evaluated geomorphic characteristics including width:depth ratio, bedform slope, and proximity to the nearest riffle crest within a riffle:pool bedform. The average width:depth ratio was calculated for three transects perpendicular to the thalweg across each 50-m stream section. The MASS2 model (see Section 5.1 and Appendix F) was used to produce water depths at a 1-m by 1-m resolution for the entire wetted channel. Although bankfull depths and channel width are commonly used to define the ratio (Gordon et al. 2004), we used depth outputs from the 19% exceedance flow model run to approximate bank full water depths. While this flow is lower than bankfull, visual examination of the bathymetry for all reaches indicated that channel width did not increase greatly between the 10% exceedance flow and bankfull flow and width:depth at these two flows are proportional. Average water depth for each transect and transect length were calculated in a GIS environment.

The channel thalweg for each reach was identified using Arc GIS Hydro tools (Version 1.1 Beta 2, ESRI, Redlands, California). To characterize the longitudinal profile of each reach, we identified elevations from the MASS2 bathymetry for points at the upstream and downstream boundaries and center of each 50-m section on the thalweg. The longitudinal resolution of the elevation data for bedform identification and spawning analyses was approximately 25 m. Pool bottom and riffle crest bedforms were identified following the method of O'Neill and Abrahams (1984). This method uses the standard deviation of elevation differences of each series, where a series is an uninterrupted sequence of elevation changes of the same sign, to separate minor bed undulations from riffle:pool bedforms. Bedforms larger than the defined tolerance value are identified as riffle crests or pool bottoms. To factor in scale-dependent associations between riffle:pool structures and channel size, we applied Hanrahan's (2006) channel width:depth adjustment to the tolerance value for each of the five reaches. Specifically, the tolerance value for each reach was obtained by multiplying the standard deviation of elevation differences by the inverse of the width:depth ratio. Tolerance values used for Crazy Johnson, the upper West Fork, the lower West Fork, upper Grays, and lower Grays were 0.34, 0.34, 0.25, 0.94, and 1.15, respectively.

Once bedforms were identified for each reach, we characterized each thalweg point by slope of the bedform and relative proximity to riffle crests and pool bottoms. Channel slope was quantified for each bedform (riffle, pool). The riffle proximity index (RPI; Hanrahan 2006) provides an indication of whether thalweg points are within a riffle or pool. The RPI is calculated as

$$\mathrm{RPI} = 1 - \frac{\mathrm{Relev} - \mathrm{Telev}}{\mathrm{Relev} - \mathrm{Pelev}} \tag{5.3}$$

where Relev is the nearest riffle crest elevation, Telev is the thalweg point elevation, and Pelev is the nearest pool bottom elevation. The RPI ranges from 0.0 to 1.0, with values greater than 0.5 associated with riffle crests. These data for thalweg points were used in chum salmon spawning analyses. To examine associations between hyporheic temperature and geomorphic factors, hyporheic temperature sampling points were characterized by width:depth ratio, bedform slope, and RPI value as well. Only temperature sampling points located adjacent to the thalweg were used in analyses.

5.3.1.4 Analyses

To examine relationships between chum salmon spawning use and the hyporheic environment, pairwise Pearson's correlations were conducted on each pair of hyporheic variables to examine

relationships among them. Data from individual sampling points were used in correlation analyses. To examine associations between hyporheic characteristics and chum salmon spawning habitat use, we first tested whether individual hyporheic variables averaged over 50-m sections differed between spawning and non-spawning sections during 2004 and 2005. Because VHG, K, and q were not normal, we conducted the Wilcoxon rank sum test for these variables. All analyses were conducted within year and across and within reach.

We used a similar approach to examine geomorphic differences between chum salmon spawning and non-spawning stream sections. We calculated riffle:pool spacing by dividing the reach length by the average reach width. This channel length, in channel width units, was then divided by the number of riffle:pool pairs in the reach. Because geomorphic variables were not normal, the Wilcoxon rank sum test was conducted to test for differences between spawning and non-spawning sampling sections. Within a riffle:pool bedform, four bedform types can be described—positive slope within a riffle, negative slope within a riffle, negative slope within a pool, and positive slope within a pool. We characterized the relationship between spawning and riffle:pool scale bedform by examining the percentage of spawning sections occurring in each bedform type in 2004 and 2005. We examined relationships between hyporheic temperature and geomorphic properties of the streambed by conducting simple linear regression analyses with individual geomorphic variables, followed by forward stepwise regression of all geomorphic variables. Model entrance and exit criteria were $P < 0.10$. JMP Version 6, 2005 (JMP USA, SAS Worldwide Headquarters, Cary, North Carolina) was used for all analyses. Boxplots were constructed in Systat (Systat 8.0, 1998, SPSS Inc., Point Richmond, California).

5.3.2 Results

5.3.2.1 Chum Salmon Spawning Associations with Hyporheic Characteristics

Across the study area, pairwise Pearson's correlations between hyporheic VHG, K, q, and T_H revealed a strong correlation between VHG and q ($P < 0.0001$). This was expected because q is a product of VHG and K. No other variable pairs were significantly correlated. VHG and q were also strongly positively correlated in each individual reach ($P < 0.0001$). VHG and T_H were significantly positively correlated in Crazy Johnson ($P < 0.04$) and the lower West Fork ($P < 0.02$), marginally nonsignificantly correlated in the lower Grays ($P < 0.08$). No VHG or K data were collected in the upper Grays River. No other hyporheic variable pairs were significantly correlated within reach.

Hyporheic characteristics of spawning sections are summarized in Table 5.20. Hyporheic temperatures associated with spawning sections during 2004 and 2005 averaged 8.4°C and 8.6°C, respectively. Average VHG values associated with spawning during these were 0.038 and 0.128, average K values were 0.137 and 0.119 cm/sec, and average q values were 0.050 and 0.024 cm/sec, respectively.

In Section 5.2, we found that hyporheic temperature was significantly higher in spawning than in non-spawning sections over the entire study area in 2004 and 2005. In this study, we found that over the study area, in 2004, spawning and non-spawning sections did not differ significantly in VHG, q, or K (Table 5.21). Across the study area in 2005, average VHG was 0.22 cm/cm higher in spawning than in non-spawning sections ($N = 96$, $P < 0.03$; Figure 5.21). Spawning and non-spawning sections did not differ in K or q.

Table 5.20. Summary Statistics of Hyporheic Variables Across all Sections, Spawning, and Non-Spawning Sections in the Grays River Study Area Based on 2004 and 2005 Sampling Section Classifications

Variable	All		Spawning		Non-Spawning	
	Mean	SD	Mean	SD	Mean	SD
2004						
T_H (°C)	7.8	1.5	8.4	1.6	7.7	1.5
VHG (cm/cm)	−0.008	0.468	0.038	0.381	−0.029	0.505
K (cm/sec)	0.1248	0.058	0.137	0.053	0.120	0.060
q (cm/sec)	−0.001	0.058	0.005	0.050	−0.004	0.061
2005						
T_H (°C)			8.6	1.5	7.5	1.4
VHG (cm/cm)			0.128	0.159	−0.090	0.573
K (cm/sec)			0.119	0.059	0.127	0.059
q (cm/sec)			0.016	0.024	−0.011	0.070

T_H = hyporheic temperature, VHG = vertical hydraulic gradient, K = hydraulic conductivity, q = specific discharge

Table 5.21. Hyporheic Differences (Spawning – Non-Spawning) Between Chum Salmon Spawning and Non-Spawning Sections Across the Study Area and by Stream Reach During 2004 and 2005. *P* values presented are Wilcoxon rank sum normal approximation z-score probabilities.

	T_H (°C)[(a)]		VHG (cm/cm)		K (cm/sec)		q (cm/sec)	
	2004	2005	2004	2005	2004	2005	2004	2005
Study Area	**0.6***	**1.2*****	0.067	**0.218***	0.017	-0.007	0.009	0.027
Crazy Johnson	-0.2	0.0	0.064	0.299	0.061	-0.030	-0.002	0.029
Upper West Fork	0.6	**1.7*****	-0.024	0.014	-0.019	0.006	-0.004	0.002
Lower West Fork	**1.5****	**1.7****	0.116	0.266	0.007	-0.037	0.015	0.049
Upper Grays	-0.2	-0.6	--	--	--	--	--	--
Lower Grays	**0.9***	0.9	**0.341***	**0.357***	0.022	-0.017	0.028	0.043

(a) Results presented are t-test *P* values; results presented in greater detail in Section 5.2.
Asterisks denote *P* values, P < 0.05*, P < 0.01**, or P < 0.001***.
T_H = hyporheic temperature, VHG = vertical hydraulic gradient, K = hydraulic conductivity, q = specific discharge.

Relationships between chum salmon spawning and hyporheic characteristics differed among reaches (Table 5.21). Hyporheic temperature was higher in several reaches, as reported in Section 5.2. In Crazy Johnson, spawning and non-spawning sections did not differ in any hyporheic characteristic. In the upper West Fork, T_H was significantly higher in spawning sections in 2005 (1.7°C) but not in 2004. T_H in spawning sections in the lower West Fork was higher in both years (2004: 1.5°C, 2005: 1.7°C). Only T_H was quantified in the upper Grays and did not differ significantly between spawning and non-spawning sections in either year. In the lower Grays, T_H was significantly higher in spawning than in

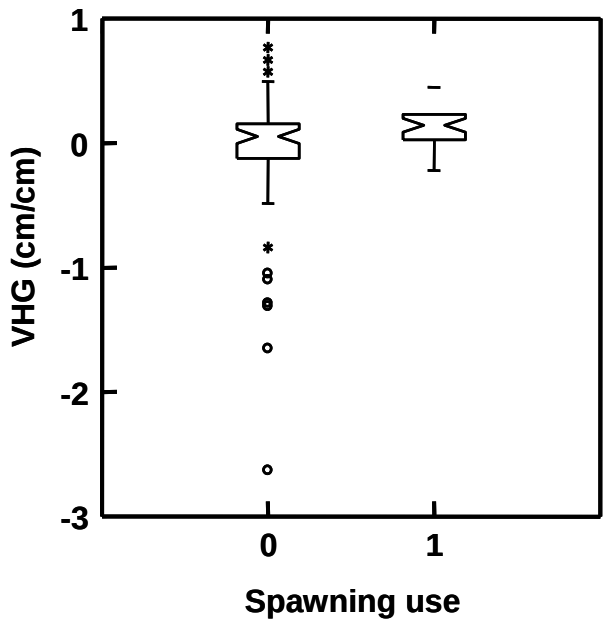


Figure 5.21. Difference in Vertical Hydraulic Gradient (VHG; $P < 0.03$) Between Spawning (1) and Non-Spawning (0) Sections Across the Grays River Study Area in 2005. Difference was significant at $P < 0.05$. The medians of the distributions are shown as the centers of the notches and the lower and upper quartiles are illustrated as hinges of the box plots. Whiskers indicate the range of values within 1.5 * the interquartile range. The notches represent an approximate 95% confidence interval around the median.

non-spawning sections in 2004 (0.9°C) and marginally nonsignificantly different ($P < 0.06$) in 2005. VHG was also significantly higher in the lower Grays in spawning sections in both years (2004: 0.34 cm/cm, 2005: 0.36 cm/cm; Figure 5.22) and q was marginally nonsignificant ($P < 0.07$) in 2004.

5.3.2.2 Geomorphic Features of Chum Salmon Spawning Areas

From 0.5 to 3.5 riffle:pool bedforms were identified among study reaches (Figure 5.23). Riffle:pool spacing ranged from 10.2 (lower Grays) to 34.1 (lower West Fork) channel widths (Table 5.22).

Sampling section geomorphic characteristics are summarized in Table 5.23. Average width:depth ratio in spawning sections in 2004 and 2005 were 44.4 and 45.0, respectively, compared to 47.8 across all sections. Average bedform feature slope in which spawning sections occurred was −0.0037 and −0.0043, respectively, in these years, compared to 0.0029 across all sections. The average riffle proximity index value of spawning sections in these years was 0.51 and 0.42, respectively, compared to 0.49 across all sections.

Overall, spawning areas tended to be on descending slopes, but otherwise, relationships between spawning areas and geomorphic variables were complex and varied among reaches (Figure 5.24; Table 5.24). Across the study area, chum salmon spawning and non-spawning sections in 2004 did not differ in width:depth, slope, or RPI (Table 5.24). In contrast, spawning sections in 2005 had lower slopes and were somewhat closer to pool bottoms than non-spawning sections.

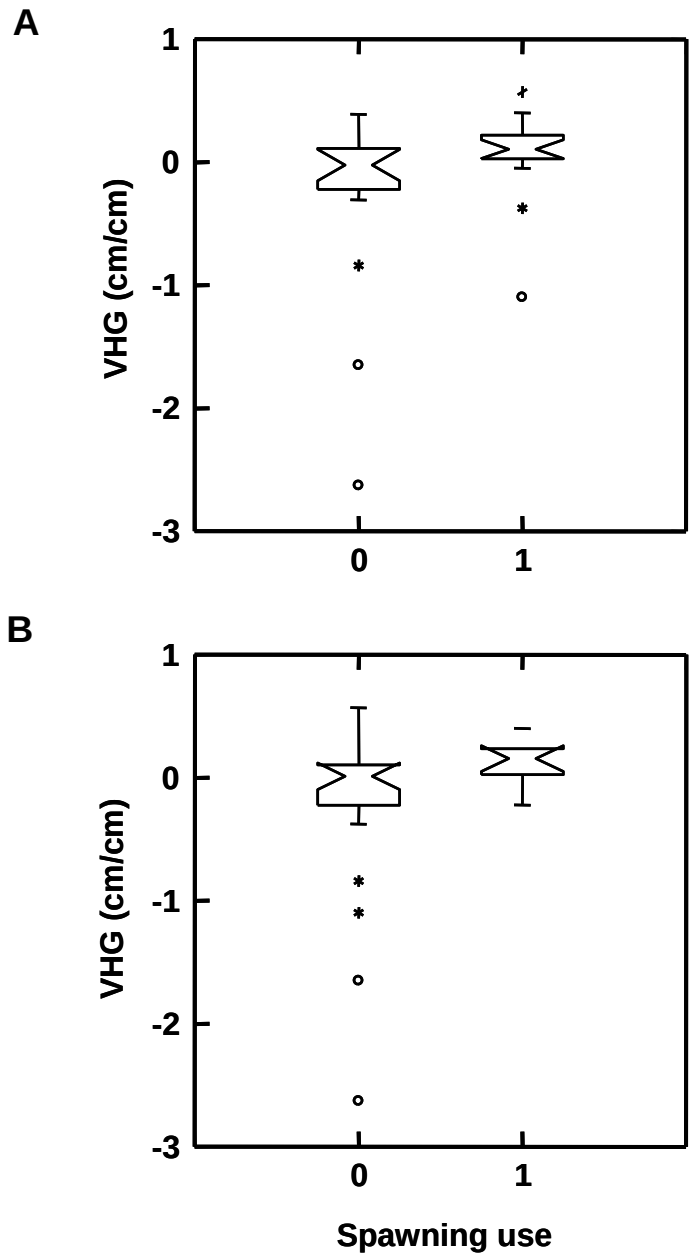


Figure 5.22. Significant Difference in Vertical Hydraulic Gradient (VHG) Between Spawning (1) and Non-Spawning (0) Sections in the Lower Grays in A) 2004 ($P < 0.04$) and B) 2005 ($P < 0.04$). The medians of the distributions are shown as the centers of the notches and the lower and upper quartiles are illustrated as hinges of the box plots. Whiskers indicate the range of values within 1.5 * the interquartile range. The notches represent an approximate 95% confidence interval around the median.

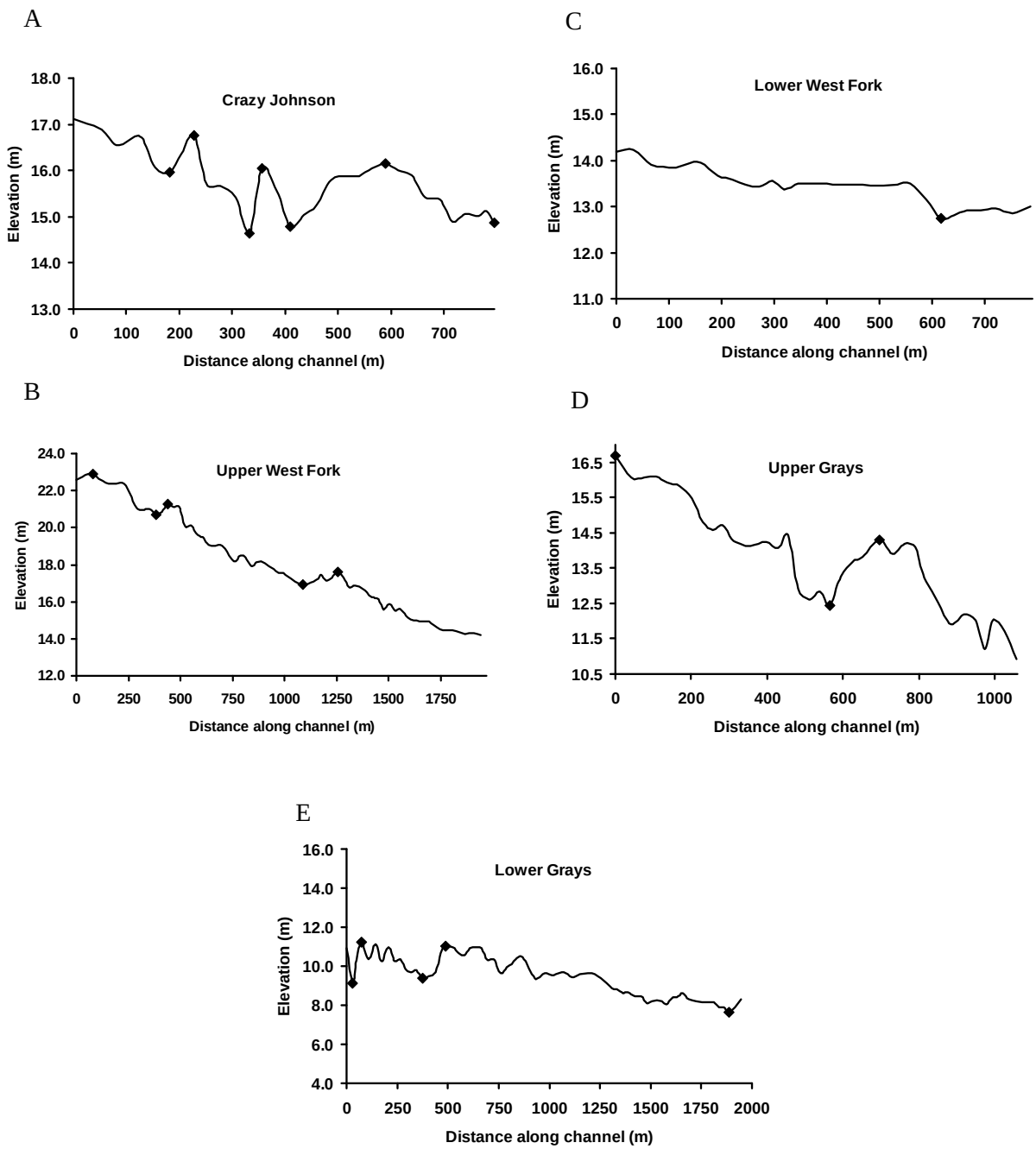


Figure 5.23. Longitudinal Plots of A) Crazy Johnson Creek, B) Upper West Fork, C) Lower West Fork, D) Upper Grays, and E) Lower Grays Showing Riffle Crest and Pool Bottom Bedforms (diamonds). The proportion of channel length to elevation change is held constant across all plots.

Table 5.22. Number and Spacing of Riffle (R):Pool (P) Bedforms in the Grays River Study Area. Bedform spacing is in channel-widths.

Reach	No of R:P Bedforms	R:P Spacing (channel widths)
Crazy Johnson	3.5	12.6
Upper West Fork	2.5	21.6
Lower West Fork	0.5	34.1
Upper Grays	1	13.9
Lower Grays	2.5	10.2

Table 5.23. Summary Statistics of Geomorphic Variables Across all Sections in the Study Area and of Spawning and Non-Spawning Sections During 2004 and 2005

Variable	All		Spawning		Non-Spawning	
	Mean	SD	Mean	SD	Mean	SD
2004						
W:D	47.8	16.2	44.4	10.2	46.1	14.7
Slope	−0.0029	0.0089	−0.0037	0.0037	−0.0019	0.0104
RPI	0.49	0.32	0.51	0.33	0.46	0.31
2005						
W:D			45.0	12.1	48.2	17.5
Slope			−0.0043	0.0090	−0.0017	0.0090
RPI			0.42	0.30	0.53	0.32

In 2004, spawning and non-spawning sections in Crazy Johnson Creek did not differ in any of the geomorphic characteristics studied (Table 5.24). In 2005, spawning sections were on lower slopes closer to riffle crests than were non-spawning sections. In 2004 and 2005 in the upper West Fork, spawning sections were in bedform features with higher width:depth ratios and in 2004 were also on lower slopes than non-spawning sections. In the lower West Fork in 2004, spawning and non-spawning sections did not differ in any of the three geomorphic characteristics studied. In 2005, the spawning sections in the lower West Fork tended to be on bedforms with lower width:depth ratios and on lower slopes than non-spawning sections. Spawning and non-spawning sections in the upper Grays were closer to riffle crests than non-spawning sections in 2005 but did not differ in any other geomorphic characteristic in either year. In the lower Grays in 2004, spawning and non-spawning sections did not differ in any of the three geomorphic characteristics studied, but in 2005, spawning sections were on lower slopes and were closer to pool bottoms than non-spawning sections than non-spawning sections.

Of the four potential bedform types differentiated by sign of the slope and the RPI (Figure 5.25), spawning sections tended to occur in the areas characterized by negative slopes closer to pool bottoms (53.9%, two-year average) and closer to riffle crests (38.4%, two-year average; Table 5.25). Location of spawning sections relative to bedform features varied among reaches. Spawning sections in Crazy

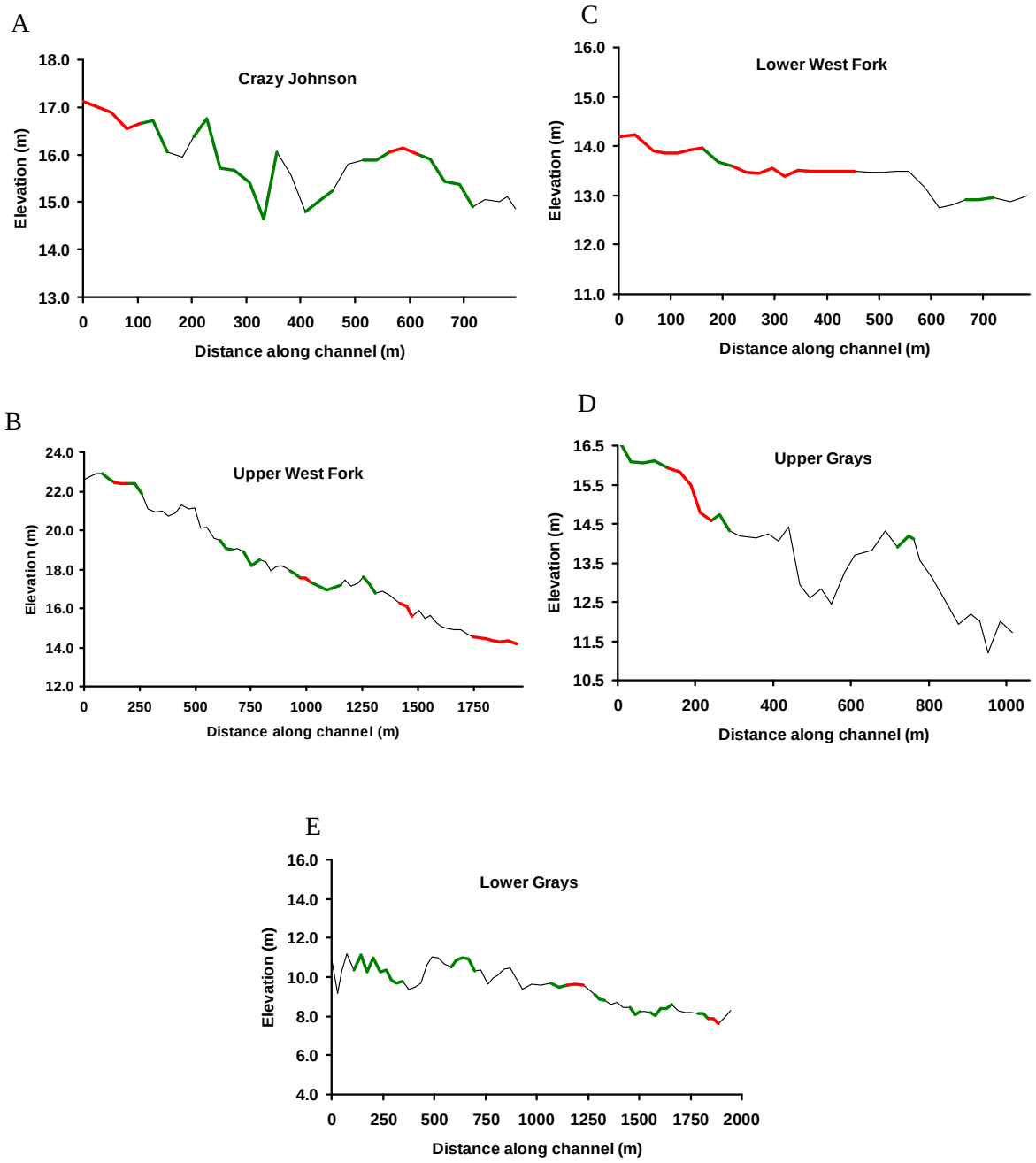


Figure 5.24. Longitudinal Plots of A) Crazy Johnson Creek, B) Upper West Fork, C) Lower West Fork, D) Upper Grays, and E) Lower Grays Showing Sections Used for Chum Salmon Spawning in One (green) or Both (red) Years. The proportion of channel length to elevation change is held constant across all plots.

Table 5.24. Bedform Differences (Spawning - Non-Spawning) Between Chum Salmon Spawning and Non-Spawning Sampling Sections Across the Study Area and by Stream Reach During 2004 and 2005. *P* Values presented are Wilcoxon rank sum normal approximation z-score probabilities.

Reach	W:D		Slope		RPI	
	2004	2005	2004	2005	2004	2005
Study Area	-1.7	-3.1	-0.0018	**−0.0026****	0.04	**−0.10***
Crazy Johnson	-2.7	0.9	-0.0042	-0.0006	0.10	**0.26***
Upper West Fork	**6.9*****	**5.5***	**−0.0038*****	-0.0021	-0.07	-0.09
Lower West Fork	-2.3	**−5.1*****	-0.0012	**−0.0026*****	-0.14	-0.36
Upper Grays	-5.3	-2.2	-0.0034	-0.0033	**0.39*****	0.14
Lower Grays	-3.3	-3.3	-0.0014	**−0.0048***	-0.00	**−0.19***

W:D = width to depth ratio, RPI = riffle proximity index
Asterisks denote *P* values, P < 0.05*, P < 0.01**, or P < 0.001***.

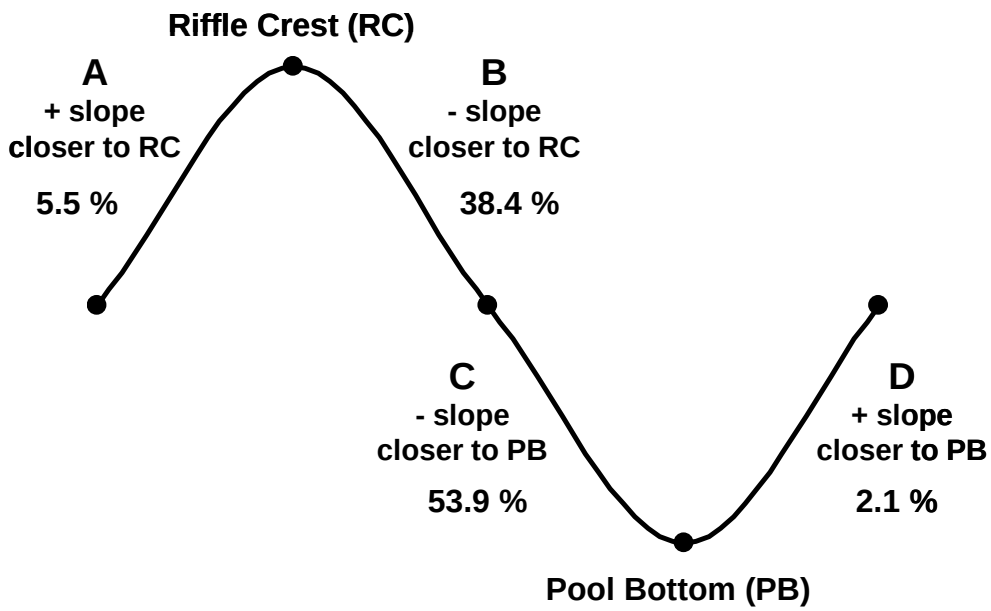


Figure 5.25. Conceptual Diagram of a Riffle Pool Bedform Showing Bedform Types A-D Differentiated by Slope and Riffle Proximity, and Average 2004–2005 Chum Salmon Spawning Section Occurrence by Type. Bedform types are further described in Table 5.26.

Table 5.25. Chum Salmon 2004 and 2005 Sampling Section Count and Percentage of Total by Bedform Type in the Grays River Study Area. Bedform type is characterized by sign of the slope and riffle proximity index (RPI), where RPI values > 0.5 are closer to riffle crests than pool bottoms. Bedform types A-D are illustrated in Figure 5.25.

	Bedform Type															
	A				B				C				D			
	+ Slope				− Slope				− Slope				+ Slope			
	Closer to Riffle				Closer to Riffle				Closer to Pool				Closer to Pool			
	Spawn		Non-Spawn		Spawn		Non-Spawn		Spawn		Non-Spawn		Spawn		Non-Spawn	
	No.	% Spawn	No.	% Non-Spawn	No.	% Spawn	No.	% Non-Spawn	No.	% Spawn	No.	% Non-Spawn	No.	% Spawn	No.	% Non-Spawn
2004																
Study Area	5	7.1	22	15.2	27	38.6	45	31.0	36	51.4	70	48.3	2	2.9	8	5.5
Crazy Johnson	3	21.4	4	22.2	6	42.9	3	16.7	3	21.4	10	55.6	2	14.3	1	5.6
Upper West Fork	0	0.0	8	15.7	6	37.5	18	35.3	10	62.5	23	45.1	0	0.0	2	3.9
Lower West Fork	2	11.1	4	33.3	0	0.0	0	0.0	16	88.9	8	66.7	0	0.0	0	0.0
Upper Grays	0	0.0	2	14.3	8	100.0	5	35.7	0	0.0	6	42.9	0	0.0	1	7.1
Lower Grays	0	0.0	4	8.0	7	50.0	19	38.0	7	50.0	23	46.0	0	0.0	4	8.0
2005																
Study Area	3	3.9	25	18.0	29	38.2	47	33.8	43	56.6	58	41.7	1	1.3	9	6.5
Crazy Johnson	3	21.4	4	22.2	7	50.0	2	11.1	4	28.6	9	50.0	0	0.0	3	16.7
Upper West Fork	0	0.0	8	18.6	8	50.0	14	32.6	7	43.8	20	46.5	1	6.3	1	2.3
Lower West Fork	0	0.0	6	50.0	0	0.0	0	0.0	18	100.0	6	50.0	0	0.0	0	0.0
Upper Grays	0	0.0	3	13.6	8	100.0	11	50.0	0	0.0	7	31.8	0	0.0	1	4.5
Lower Grays	0	0.0	4	9.1	6	30.0	20	45.5	14	70.0	16	36.4	0	0.0	4	9.1

Johnson and the upper Grays were more likely to be closer to riffle crests, whereas spawning sections in the lower West Fork were more likely to be closer to pool bottoms (Table 5.25). The upper West Fork and the lower Grays were inconsistent between years but tended to have more spawning sections closer to pool bottoms. Very few spawning sections were located on positive slopes. Only in Crazy Johnson were significant numbers of spawning sections on positive slopes, with 35.7% and 21.4% of spawning sections on positive slopes in 2004 and 2005, respectively.

5.3.2.3 Geomorphic Factors Associated with Elevated Hyporheic Temperature

Across the study area in linear regression models with individual geomorphic variables, hyporheic temperature was significantly associated with width:depth ratio and RPI and marginally nonsignificantly related to feature slope (Table 5.26). Higher temperatures were associated with lower slopes and lower width:depth ratios and were located closer to pool bottoms. However, no variable explained significant variation in hyporheic temperature. In the stepwise multiple linear regression analysis of all three geomorphic variables across the study area, all three variables were significantly associated with hyporheic temperature (Model $P < 0.01$, $r^2 = 0.111$; RPI: $P < 0.0001$, $r^2 = 0.079$, WD: $P < 0.003$, $r^2 = 0.024$, slope: $P < 0.01$, $r^2 = 0.016$), with warmer temperatures associated with lower RPI values, width:depth ratios, and bed slopes.

Table 5.26. Regression Model Results of Hyporheic Temperature with Individual Geomorphic Variables Significance. *P* (above) and r^2 values (below) are presented.

	RPI	W:D	Slope
Study Area	**<0.0001** 0.08	**0.0001** 0.04	0.08
Crazy Johnson	NS	**<0.0001** 0.594	NS
Upper West Fork	**0.04** 0.039	NS	NS
Lower West Fork	**0.02** 0.076	**0.02** 0.074	**0.02** 0.070
Upper Grays	**0.04** 0.109	NS	NS
Lower Grays	**0.0001** 0.206	NS	NS
RPI = riffle proximity index, W:D = width to depth ratio.			

The relationship between hyporheic temperature and geomorphic variables differed among study reaches (Table 5.26). Although in each reach at least one geomorphic variable was significant, data figures are presented only for significant relationships with $r^2 > 0.10$. In Crazy Johnson, higher hyporheic temperatures were significantly associated with lower width:depth ratios but neither of the other geomorphic variables ($r^2 = 0.594$; Figure 5.26). In the stepwise regression with all three variables, only width:depth ratio was significantly associated with hyporheic temperature ($P < 0.0001$, $r^2 = 0.607$). In the upper West Fork and the upper and lower Grays, higher hyporheic temperature was associated only with lower RPI values, but this variable only explained significant variation in the two Grays River reaches (Table 5.26; Figure 5.27). Stepwise models for the upper West Fork (UWF) and upper Grays

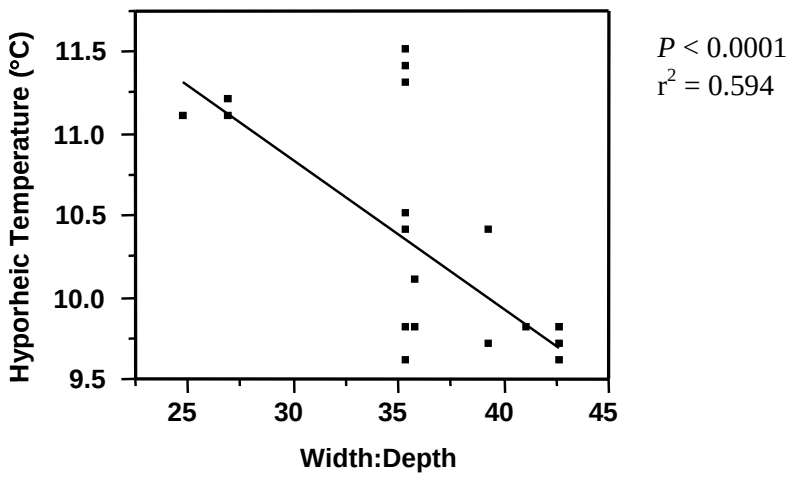


Figure 5.26. Significant Relationship Between Hyporheic Temperature and Width:Depth Ratio in Crazy Johnson Creek

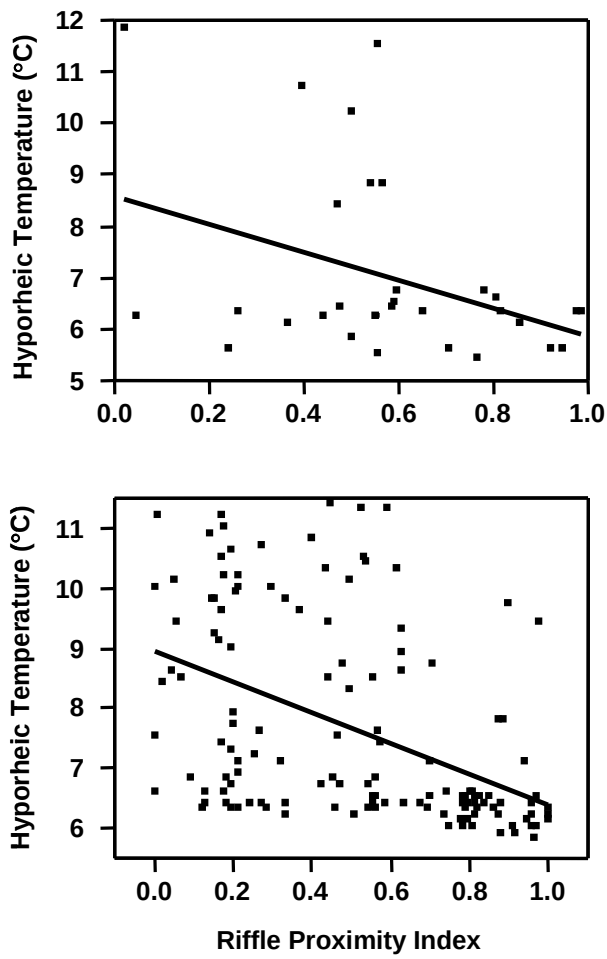


Figure 5.27. Relationship Between Hyporheic Temperature and Riffle Proximity Index (RPI) in the A) Upper Grays and B) Lower Grays River Reaches

(UG) included only RPI (UWF: $P < 0.04$, $r^2 = 0.05$; UG: $P < 0.04$, $r^2 = 0.139$). In the lower West Fork, higher temperatures were weakly associated with all three geomorphic variables in individual regression models (Table 5.26), but the stepwise regression model for this reach contained only RPI ($P < 0.02$, $r^2 = 0.092$). In the lower Grays, the stepwise model included RPI and slope (RPI: $P < 0.0001$, $r^2 = 0.212$; slope: $P < 0.2$, $r^2 = 0.241$).

5.3.3 Discussion

The strongest relationship we found between chum salmon spawning and any hyporheic or geomorphic variable was with hyporheic temperature. In Section 5.2, we established an association between hyporheic temperature and chum salmon spawning across the study area during both years and in all reaches except Crazy Johnson and the upper Grays. Chum spawning sections tended to be located in areas with hyporheic temperatures 1 to 5°C warmer than non-spawning areas and river water. Others have documented a chum salmon association with groundwater and strong hyporheic exchange (Leman 1993; Geist et al. 2002). Warmer incubation temperatures have been shown to affect growth rate and therefore emergence timing (Leman 1993) and to protect developing embryos from freezing conditions (Crisp 1990; Bjornn and Reiser 1991).

We found only a limited relationship between chum salmon spawning and VHG, in contrast to the findings of others (Baxter and Hauer 2000; Geist et al. 2002). We found an association between spawning section location and VHG in the lower Grays River reach but not in the other four study reaches. Although chum salmon prefer warm hyporheic water (reviewed by Bjornn and Reiser 1991; Bonnell 1991; Geist et al. 2002; Section 5.2) and may prefer flowing water (Tautz and Groot 1975), strongly vertical flow may be less important. Hill et al. (1998) found that VHG values in riffles tended to be negative toward the riffle crest and positive toward the downstream end, with VHG values of zero in the transitional area. Hyporheic flow may be immediately subsurface in some portions of a riffle. In these areas, spawning fish may be able to intercept warmer subsurface flows during redd construction in areas with VHG values that are marginally positive to zero. Because we found spawning sections located throughout riffles, our VHG values may be expected to be a range of values in areas with potentially accessible warm hyporheic water. In the Crazy Johnson, hyporheic and river water is dominated by water that is of subsurface origin and warmer than river temperatures in other reaches. In this reach, we found no associations between spawning and hyporheic temperature or VHG, yet this stream consistently attracted the highest spawning densities in the study area (Section 5.2). Our observations in Crazy Johnson suggest that chum salmon may seek warm hyporheic water or water with specific chemical cues, but the direction of hyporheic flow may be less important.

We found chum salmon spawning primarily on negative (descending) slopes. An association between chum salmon spawning and riffles has been well documented by other researchers (Bjornn and Reiser 1991; Salo 1991; Leman 1993; Fukushima and Smoker 1998). Although we did find an association between chum salmon and descending slopes, the relationship was not consistent among years or reaches. This inconsistency may be a function of our method of quantifying spawning use. In most studies of salmonid spawning habitat, known spawning area boundaries are delimited and habitat characteristics are contrasted with adjacent non-spawning areas. For many salmonids, individual redds are identifiable and spawning areas at the scale of redds or redd clusters can be described. Chum salmon are unusual for their close inter-redd spacing and practice of redd superimposition (Burner 1951). Although identification of individual redds was attempted, the practice was ineffective for chum salmon in the Grays River. For this reason, spawning individuals were counted within arbitrarily-designated

50-m stream sections. Consequently, our spawning sections may have contained considerable non-spawning area as well as diverse geomorphological features. Our method is preferred for identifying broad areas suitable for spawning when little *a priori* spawning information is available and for species in which discrete redd identification is difficult, and represents a potentially less biased approach to characterizing spawning habitat. However, identifying strong habitat associations can be challenging when spawning areas are small relative to our unit of sampling (50-m reaches) or located in atypical areas within the broader sampling unit.

In the Grays River study area, chum salmon spawning sections tended to be closer to pool bottoms than riffle crests. Of all chum salmon spawning sections averaged over the two years, 53.9% were on descending slopes closer to pool bottoms and 38.4% were on descending slopes closer to riffle crests. The remaining 7.6% were on positive slopes. We expected chum salmon spawning to be closer to riffle crests because bedform-induced upwellings occur at the upstream ends of riffles (Hill et al. 1998; Fernald et al. 2001). Our finding suggests that multiple geomorphic mechanisms of hyporheic exchange are occurring in the Grays River study area associated with riffle:pool bedform morphology. Hill et al. (1998) also found lateral hyporheic exchange associated with meander bends, with upwellings associated with riffle:pool transitions at the lower end of riffles. Chum salmon spawning has been shown to occur in a narrow band on the inside bank of channel meanders associated with this source of subsurface flow (Leman 1993). Because we did not incorporate channel sinuosity or other aspects of lateral channel structure and sampled adjacent to the thalweg, we were not able to detect this source of hyporheic inflow.

Elevated hyporheic temperatures were also closer to pool bottoms and were weakly associated with descending slopes in some reaches. This is consistent with our findings that chum salmon spawning was more likely to occur associated with elevated hyporheic temperatures and on the lower end of riffles. Upwelling hyporheic water has been closely associated with channel slope and sharp changes in slope by others (Vaux 1968; White et al. 1987; Harvey and Bencala 1993; Wroblicky et al. 1998). Wroblicky et al. (1998) suggested that the magnitude of change in streambed slope required to induce hyporheic exchange may scale with stream size. Although hyporheic temperature responded to different geomorphic variables across the stream reaches in our analyses, differences in geomorphic control of hyporheic exchange is to be expected, given differences in channel morphology and size among the reaches (Kasahara and Wondzell 2003). They found that geomorphic mechanisms available to promote hyporheic exchange varied among streams of differing size and confinement. Variation in stream reach morphologies within the study site may also explain our ability to associate geomorphic characteristics with hyporheic exchange better in some reaches than in others. In some reaches, geomorphic mechanisms we did not examine may be particularly important, such as multiple channels and island bars.

Considerable variation in the spatial distribution of chum salmon spawning could not be explained by the geomorphic characteristics we examined. We suggest that much of the warm hyporheic water we detected in study area may be present due to broader-scale processes. Broad-scale variation in valley morphology has been shown to induce hyporheic exchange (Larkin and Sharp 1992, Stanford and Ward 1993; Baxter and Hauer 2000; Fernald et al. 2001). Stanford and Ward (1993) described alternating constrained and unconstrained alluvial reaches as beads on a string along the river continuum. The upstream end of each bead or wide valley reach is typically an area of hyporheic downwelling whereas the downstream end is upwelling. Baxter and Hauer (2000) found bull trout (*Salvelinus confluentus*) spawning associated with downwelling areas in bounded alluvial valleys (BAVs), reaches with relatively wide valleys and active hyporheic exchange. Wide valley reaches may also contain multiple channels with complex subsurface exchange between them, and island, bar, and side channel structures that

promote hyporheic exchange (Wondzell and Swanson 1996; Geist and Dauble 1998; Fernald et al. 2001; Kasahara and Wondzell 2003). These flowpaths may be controlled by variation in substrate permeability forming preferential subsurface pathways (Stanford and Ward 1993). Kasahara and Wondzell (2003) found that in a model of a wide alluvial valley, secondary channels controlled the location and direction of subsurface flow paths, and removal of secondary channels reduced hyporheic discharge by 25%. The upper Grays reach is a wide-valley, low-gradient response reach immediately downstream from a steep constricted reach. All other reaches in the study area except the upper West Fork are either within this wide-valley response reach (Crazy Johnson, lower West Fork) or immediately downstream from it (lower Grays). As such, they may be affected by large-scale geomorphic processes associated with changes in valley width and bedslope beginning at the upstream end of the study area.

Subsurface alluvial structure and the presence of bedrock or other confining materials may also affect hyporheic exchange as well as the inflow of deeper groundwater sources (Stanford and Ward 1993). Wondzell and Swanson (1996) found that subsurface flow was promoted by a combination of small-scale bedform and groundwater input from adjacent hillslopes. Bauman (1998) found that deeper groundwater inputs disrupted expected patterns of association between bedform and hyporheic temperature and hydraulic conductivity. We observed a dense clay outcropping throughout the middle reach of the upper Grays channel, which may explain the lack of hyporheic exchange we observed in this area. In contrast, hyporheic temperatures in Crazy Johnson Creek were significantly warmer than at any other sites in the study area, suggesting a deeper rather than shallow groundwater source that is less influenced by river water temperatures. Subsurface features may direct flow through the floodplain of the upper Grays reach including Crazy Johnson and the lower West Fork. The floodplain of the upper Grays reach has been described as an unusually rich source of groundwater in the State of Washington (G. Johnson, WDFW, Olympia, Washington). Until it was destroyed by the avulsion, an artificial spawning channel had been constructed at the upstream end of the upper Grays reach because subsurface water could be easily intercepted, and two springs (Crazy Johnson, Gorley) occur in the immediate vicinity. We conclude that the hyporheic environment of the Grays River chum spawning area is complex and diverse as a result of multiple processes interacting over multiple scales. Although we did identify some weak associations between hyporheic exchange and riffle:pool scale bedform, we suggest that broad-scale processes associated with valley morphology and phreatic groundwater input make significant contributions to hyporheic exchange in the Grays River.

We also found high hyporheic temperatures weakly associated with low width:depth ratios in Crazy Johnson and the lower West Fork. In valleys of varied width, downwelling tends to occur at the upstream ends of wide valley sections as the valley widens into an unconstricted reach, and upwelling tends to occur at the downstream end as the valley narrows again (as discussed above; Stanford and Ward 1993; Brunke and Gonser 1997; Baxter and Hauer 2000). Baxter and Hauer (2000) suggest that the presence of knickpoints at the bounded end of alluvial valleys tended to constrain hyporheic flow and promote upwelling exchange, and Stanford and Ward (1993) suggest that subsurface flow may be progressively more constricted by fine sediment and reduced permeability as the valley becomes confined. In channels downstream from the wide valley upper Grays reach, we hypothesized (above) that significant downwelling occurs as the river exits the narrow canyon above the study area and transitions into the lower gradient upper Grays response reach. At the downstream end of the upper Grays reach, including the lower West Fork reach, the active floodplain narrows, constricted by agricultural dikes and revetments that may function as constraints and promote upwelling analogous to knickpoints in naturally bounded stream reaches. Alternatively, the landscape position of this channel at the downstream end of an active

downwelling response reach may be responsible for upwelling in these reaches, with channel width coincidental. Warmer hyporheic temperatures in Crazy Johnson occur in the upper half of the reach, associated with several significant upwellings. This portion of the reach is incised into a floodplain terrace of the Grays River. In contrast, the lower half that is not characterized by elevated hyporheic temperatures flows through the active floodplain of the Grays River immediately associated with the abandoned channel. Consequently, the geomorphology of the lower Crazy Johnson may be influenced by Grays River flood flows and associations between width:depth ratios and elevated hyporheic temperature in this reach may be coincidental.

Considerable variation remained in our hyporheic temperature data that could not be explained by bedform. Riffle:pool bedform topography has been well documented to generate hyporheic exchange (Harvey and Bencala 1993; Hill et al. 1998; Kasahara and Wondzell 2003). We suggest that the limited hyporheic exchange associated with bedform we documented may be partially the result of interactions among multiple processes such as broader scale channel morphology and groundwater inputs that may also influence the distribution of spawning habitat, as discussed earlier. In addition, Malard et al. (2002) and Baxter and Hauer (2000) found considerable fine scale hyporheic variation associated with lateral heterogeneity in the channel bathymetry. Similarly, Bauman (1998) found limited associations between hyporheic temperature and hydraulic conductivity and bedform due to phreatic groundwater input and high variability in hydraulic conductivity at fine scales. Although our sampling was of a resolution sufficient to detect riffle:pool bedforms, it was not designed to detect lateral bedform structure and associated hyporheic behavior.

The limited association we found between hyporheic exchange and bedform may also be a function of relatively low bedform diversity. Riffle:pool bedform spacing of 5 to 7 channel widths has been widely documented across streams of varying size and channel type (summarized in Montgomery et al. 1995). However, these channel width spacings reflect free-formed bed structures in channels without forcing structures. Montgomery et al. (1995) found that riffle:pool channels with high levels of large woody debris had riffle:pool spacing values of 0.2 to 3 channel widths, with more frequent spacing in wider and steeper channels. Bedload accumulation upstream of large woody debris creates riffle crests, whereas scour downstream from large woody debris scours pools. The resulting topographic complex encourages hyporheic exchange through creation of differential head pressures above and below the substrate. Thus, riffle:pool structure spacing in the Grays River system would be expected to occur at these frequencies with historic large woody debris loading. We found riffle:pool spacing of 10.2 to 34.1 channel widths across all reaches, and the relative lack of large woody debris in all reaches has been documented (Herrera 2005). Our study reaches in general appear to lack the degree of bedform structuring that would be expected in historically forested watersheds such as the Grays River. Although hyporheic-exchange–generating bedforms occur in the absence of large woody debris, the riffle:pool spacing and therefore frequency of exchange are greatly reduced in the absence of large woody material (Montgomery et al. 1995).

Bedform may not induce hyporheic exchange in substrates that have low hydraulic conductivity. Fulton (1970) suggested that riffles in lower Columbia River tributaries became compacted and full of fine sediments following extensive timber harvest beginning in the early 1900s. He concluded that riffles once used by spawning salmon may no longer be suitable. The magnitude and direction of subsurface flow is influenced by the permeability of the bed material and will be limited by subsurface materials with low permeability (Vaux 1968). Hyporheic flow paths are associated with hydraulic conductivity, and colmation, or clogging of upper substrate layers, can lead to consolidation of the sediment matrix,

decreased permeability, and retardation of hyporheic exchange (Brunke and Gonser 1997). The extent of colmation is directly linked to land use practices that increase sediment loading (Karr and Schlosser 1978, Platts et al. 1989). Platts et al. (1989) found that sedimentation of fall Chinook salmon (*O. tschawytscha*) and steelhead (*O. mykiss*) spawning areas in an Idaho river was directly associated with logging and road construction. They found that after more than twenty years following a logging moratorium and watershed rehabilitation, substrate sediment levels had declined significantly. However, they concluded that complete recovery to pre-logging conditions would require additional watershed recovery and occurrence of multiple flood flows capable of resorting stored gravel substrates and transporting fine material downstream. Anderson et al. (2005) suggested that the influence of hydraulic conductivity, and therefore substrate sedimentation, on hyporheic exchange may be higher in lower-gradient streams such as the lower West Fork where exchange processes associated with bedform diversity are reduced.

Salmonid incubation success has been associated with flow velocities within the incubation environment. Thus, specific discharge and hydraulic conductivity may provide some indication of the relative quality of spawning substrates for successful embryo incubation and emergence. In spawning sections identified in 2004 and 2005, we found average hydraulic conductivity values of 0.137 and 0.119 cm/sec and specific discharge values of 0.005 and 0.016 cm/sec, respectively. These values are within the range of values found in other salmonid spawning areas (Baxter and Hauer 2000; Geist 2000; Zimmermann and Lapointe 2005) and suggest that flow potential through spawning areas may be sufficient to support incubating salmonids with moderate survival to emergence (Cooper 1965). However, there is no threshold velocity that insures survival of incubating embryos (Lapointe et al. 2004) because incubation success also depends on dissolved oxygen and fine sediment levels as well as other substrate characteristics within the egg pocket (Cooper 1965; Sowden and Power 1985). Although Cooper (1965) found a strong association between salmonid incubation success and specific discharge, he also found that gravel size and grading affected egg survival at a constant specific discharge and that suspended sediment levels in interstitial flow reduced salmonid survival. In addition, clogging of interstitial pore spaces in downwelling areas by fine sediments had been shown to reduce overall influx to the streambed (Packman and MacKay 2003). Excess fine sediments within a stream system associated with land use changes can penetrate into the surface armoring layer and reduce surface-hyporheic exchange throughout the stream system (Schalchli 1992). Because river water is usually saturated with dissolved oxygen, interaction of groundwater with river water may be critical for maintenance of water quality within the incubation environment (Malcolm et al. 2004). Although salmonids have the potential to disrupt the relatively thin armored layer, sedimented capping layers can reform quickly. And, fine sediment penetration is deeper in larger gravel substrates typical of salmonid spawning areas (Schalchli 1992). Without information regarding these additional characteristics of chum and fall Chinook salmon redds, one can not definitively associate redd success in the Grays River with hydraulic characteristics of the substrate.

Chum salmon may benefit from increased hyporheic exchange associated with restoration of bedform diversity. The importance of hyporheic exchange to spawning site selection has been the focus of this study; however, other important benefits of increased hyporheic exchange include increased incubation success, buffering of low and high temperatures, and increased stream productivity (Crisp 1990; Bjornn and Reiser 1991; Alexander and Caissie 2003). Although the importance of large woody debris for pool formation has been well recognized, its role in inducing hyporheic exchange has not. Lack of large woody debris may have reduced the extent of hyporheic exchange in the Grays River, and may partially explain the decline of salmonids in the watershed. Large woody debris is lacking throughout the watershed, and may be affecting not just chum and fall Chinook salmon but also coho salmon

(*O. kisutch*), steelhead, and cutthroat trout (*O. clarki*) in smaller streams of the upper watershed. The benefits of hyporheic exchange may be more significant in these smaller streams where the proportion of hyporheic flow to river flow is higher (Kasahara and Wondzell 2003). In addition, the bedform of smaller streams is more responsive to large woody debris than larger streams. Thus, lack of large woody debris and associated hyporheic exchange may be more significant impacts on the quality of salmonid habitat in these smaller streams.

5.4 Conclusions

5.4.1 Chum and Fall Chinook Salmon Spawning Habitat Availability and Redd Capacity

The majority of aquatic habitat in the Grays River study area appears suitable for chum salmon spawning based on velocity, depth, substrate particle size, and hyporheic temperature and for fall Chinook salmon based on velocity, depth, and substrate. Of the total available area, 61.6% (12.22 ha) is within spawning limits for both species, 27.9% (5.54 ha) is only within chum salmon limits, 2.7% (0.54 ha) is only within fall Chinook salmon limits, and 7.8% (1.55 ha) is beyond the spawning limits for both species. Hyporheic temperature was much more limiting to chum salmon available spawning habitat than were velocity, depth and substrate. Although considerable area was suitable for spawning, relatively little was of high quality for either species. For chum salmon, only 3.7% (0.73 ha) of the total wetted area was of the highest quality such that all four habitat characteristics were within the probable use range for this species. For fall Chinook salmon, only 16.8% (3.33 ha) of the total wetted area was of the highest quality such that velocity, depth and substrate were within the probable use range for this species.

Spawning habitat availability for chum and fall Chinook salmon varied with flow. For chum salmon, spawning habitat availability was lowest at the 90% exceedance flow (4.58 ha) and was similar or slightly higher at the 50% and 10% exceedance flows (5.80 and 5.75 ha, respectively), whereas for fall Chinook salmon, spawning habitat availability increased substantially as flow increased from the 90% (7.99 ha) to the 50% (12.75 ha) to the 10% (16.50 ha) exceedance flow. The abandoned channel offers potential spawning area for chum or fall Chinook salmon based on evaluation of only substrate and hyporheic exchange. We found that all of the abandoned channel contained suitable substrate for chum salmon, and 11.1% (approximately 0.33 ha) currently has elevated hyporheic temperatures preferred by this species. For fall Chinook, 44.5% (1.31 ha) had suitable spawning substrate.

Available spawning area appears to be sufficient to support historic chum and fall Chinook salmon populations in the Grays River study area. We estimate that chum and fall Chinook salmon redd capacity ranges from 5,590 to 22,405 redds and 4,904 to 19,615 redds, respectively, based on suitability for velocity, depth, substrate, and hyporheic exchange (chum salmon) and velocity, depth, and substrate (fall Chinook salmon). However, redd capacity in high spawning habitat may be insufficient to support historic run sizes of both species.

Factors other than spawning habitat quantity may be limiting chum and fall Chinook salmon production and recovery in the Grays River. Although sufficient spawning habitat, and therefore potential redd capacity, is available to support historic run sizes of both species, much of the available spawning habitat is beyond the probable use limits we established based on existing literature, and therefore may represent relatively low-quality habitat. Thus, habitat quality associated with substrate stability and excessive fine sediments may be the most significant factor limiting chum and fall Chinook salmon

recovery within the Grays River system. These habitat conditions have been conclusively linked to reduced incubation success and juvenile production, as well as increased sediment delivery associated with timber harvest, as documented in Chapter 4.

5.4.2 Habitat Characteristics of Chum Salmon Spawning Areas

Chum salmon spawn in consistent locations in the Grays River. During 2002 through 2005, spawning chum salmon were more likely to use specific areas each year than would be expected due to chance. Hyporheic temperature was a more important habitat factor for spawning chum salmon than velocity, depth, or substrate in the Grays River. Hyporheic temperature appears to drive chum salmon spawning habitat selection in the Grays River because it is the most limiting habitat factor. Chum salmon selected spawning habitats based on different habitat characteristics in different reaches of the Grays River study area. Reaches differed in the relative availability of each spawning habitat characteristic and therefore factors limiting spawning habitat availability differed among reaches. Our findings from Section 5.2 suggest that chum salmon may have a maximum spawning depth, and that the suitability limits we applied in Section 5.1 may overestimate available habitat. Chum salmon concentrate spawning in relatively low-quality habitat in Crazy Johnson Creek, likely associated with the strong presence of warm hyporheic flows. We were moderately successful in predicting chum salmon spawning locations when hyporheic temperature was a predictor variable.

5.4.3 Hyporheic and Geomorphic Characteristics of Chum Salmon Spawning Areas

The strongest relationship we found between chum salmon spawning and any hyporheic or geomorphic variable was with hyporheic temperature. We also found a limited relationship between spawning and vertical hydraulic gradient (VHG), observed only in the lower Grays River reach. Chum salmon spawned primarily on negative (descending) slopes. We also found that chum salmon spawning sections tended to be in riffles closer to pool bottoms, although considerable spawning occurred on the upper half of riffles as well. This may be because multiple geomorphic mechanisms of hyporheic exchange occur associated with riffle:pool bedform morphology. Consistent with our finding of chum salmon spawning tending to be lower on riffles, we found an association between elevated hyporheic temperatures and descending slopes and pool bottoms.

Relationships between chum salmon spawning and warm hyporheic temperatures suggestive of upwelling areas and riffle:pool bedform characteristics were weaker and less consistent than we expected. We suggest that some of the warm hyporheic water we detected in the study area may be present due to broader-scale processes such as variation in valley morphology that have been associated with hyporheic exchange, and the presence of phreatic groundwater sources. Wide valley reaches are more likely to contain multiple channels, islands, and bar structures that promote complex subsurface exchange. Subsurface alluvial topography and the presence of bedrock or other confining materials may also affect hyporheic exchange as well as the inflow of shallow phreatic groundwater into the Grays River study area. Considerable evidence exists to suggest that localized groundwater inputs contribute significant subsurface flow. These inputs likely influence chum salmon spawning distribution in the Grays River.

Riffle:pool scale bedform has been shown clearly to induce hyporheic exchange in streams. The limited hyporheic exchange we documented may be due to the loss of bedform diversity associated with a relative lack of large woody debris. We found riffle:pool spacing of 10.2 to 34.1 channel widths across all reaches. Riffle:pool frequencies in streams with high levels of large woody debris are typically 0.2 to

3 channel widths. Bedform-induced hyporheic exchange may also be lacking in the Grays River if substrates have low hydraulic conductivity. lower Columbia River tributaries have been characterized as compacted and full of fine sediments due to extensive timber harvest many decades ago.

5.4.4 Habitat Characteristics of Fall Chinook Salmon Spawning Areas

In the Grays River study area, fall Chinook salmon spawning and non-spawning areas differed in depth and velocity in some cases but not in hyporheic temperature or substrate. We conclude that chum and fall Chinook salmon differ in the spawning habitat characteristics they use in the Grays River study area, with fall Chinook salmon spawning in deeper and faster water than chum salmon. Our findings are consistent with others, that the range of fall Chinook salmon depth and velocity spawning preferences include faster and deeper water than chum salmon. We were not able to successfully predict fall Chinook salmon spawning locations in the Grays River study area. There are several potential reasons for our lack of predictive success. First, our spawning data were at the scale of individual redds. Extrapolation of one or several redds to a 50-m-long stream section likely misclassified sections as spawning when a significant portion of the habitat may have been unsuitable for fall Chinook salmon spawning (and therefore classified as non-spawning based on our evaluation in Section 5.1). Second, the very low number of spawning fall Chinook salmon in the Grays in 2003 and 2005 may not have spawned in habitat considered typical of the species as documented from other river systems.

5.4.5 Implications for Chum and Fall Chinook Salmon Recovery in the Grays River

Our findings suggest that available spawning habitat may be sufficient to support historic run sizes of both species. However, of the available spawning habitat for these species, relatively little was of high quality. Chum salmon fry emigrate out of freshwater almost immediately after emergence. The Grays River therefore provides primarily spawning and incubation habitat for chum salmon whereas fall Chinook salmon rear for up to one year. This suggests that for chum salmon, spawning habitat quality rather than quantity may be limiting recovery. For fall Chinook salmon, both spawning and juvenile rearing habitat quality may be limiting recovery.

Chum salmon may benefit from increased hyporheic exchange associated with restoration of bedform diversity. Hyporheic exchange is the habitat factor most limiting spawning area for chum salmon. Land use effects in the upper Grays watershed and study area have altered habitat-forming processes, potentially affecting bedform and subsurface flow. Loss of large woody debris has likely reduced bedform diversity of Grays River channels. Restoration of hydraulic and bedform diversity in the Grays River study area may contribute to increased hyporheic exchange and therefore total available spawning area for chum salmon.

The most significant limiting factors within the Grays River system may be factors such as substrate stability and excessive fines, which reduce incubation success and juvenile production. Clogging of upper substrate layers can lead to consolidation of the sediment matrix, decreased permeability, and retardation of hyporheic exchange. Reduced flow to the egg pocket region of salmonid redds reduces survival, and sedimentation of surface layers impedes alevin emergence from redds. The balance between bed scour and sedimentation can be disrupted by a variety of anthropogenic activities such as timber harvest and associated road construction. The extent that each of these processes has reduced incubation success in the Grays River is unknown and would require further study to resolve.

6.0 Integration of Assessment Findings

The upper Grays River watershed assessment was based on the conceptual model shown in Figure 6.1 (Beechie et al. 2003) and discussed in detail earlier in this report. The approach associated with the model is based on linkages between landscape-level (watershed-scale) parameters and the natural disturbance regime that controls habitat-forming processes in the watershed. These processes or controlling factors have a strong influence on the ecological and biological integrity of the riverine ecosystem.

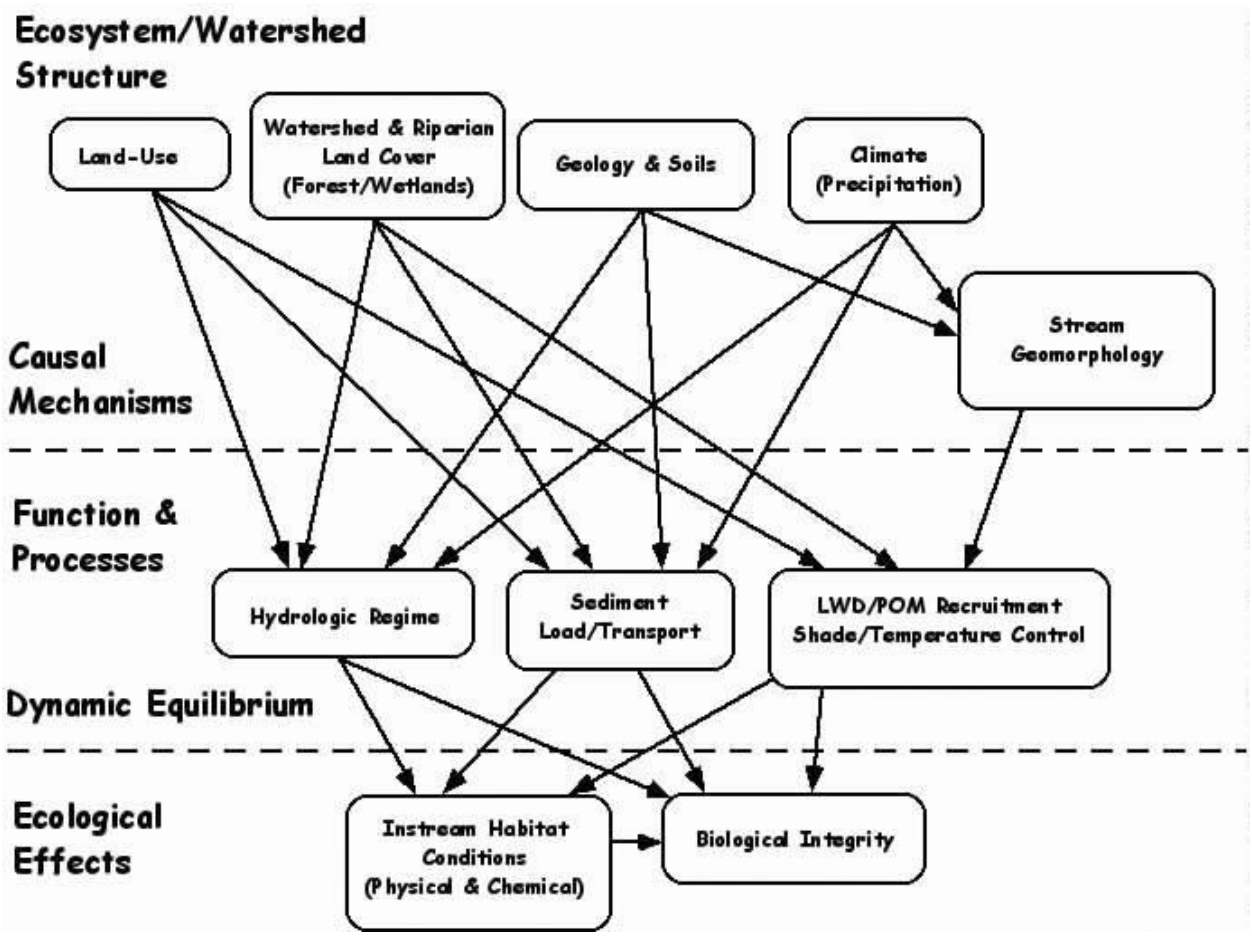


Figure 6.1. Watershed Conceptual Model (based on Beechie et al. 2003)

The Grays River is vitally important to the recovery of lower Columbia River chum salmon because it currently has the most viable population of these ESA-listed salmon remaining in the region (LCFRB 2004). In addition, fall Chinook and coho salmon and steelhead also use the Grays River watershed. All Grays River salmon and steelhead will need to be restored to a high level of viability to meet regional recovery objectives (LCFRB 2004). During the past century, the numbers of naturally spawning salmon and steelhead have declined to levels far below historical numbers because of the disruption of normal riverine processes by timber harvest and agriculture in the watershed and riparian corridor. Habitat-limiting factors include the loss of floodplain habitat connectivity, loss of instream complexity and

diversity, stream channel and streambank instability, loss of riparian function and LWD recruitment potential, elevated sediment loading, and altered streamflow conditions (WCC 2002).

The disruption of normal riverine processes has also resulted in degradation of habitat quality and destabilization of the primary area where chum salmon spawn in the Grays River (WDFW 2001; WCC 2002; LCFRB 2004). Chum salmon are known to prefer to spawn in areas with low velocities that contain springs, seeps, or upwelling (hyporheic) flow (Geist et al. 2002). The primary chum salmon spawning area in the Grays River watershed is located at a break in channel-bed slope between a steep canyon outlet and the tidally influenced lower river. This *response reach* of the river represents a transition to an unconfined depositional zone. The response reach is particularly sensitive to increases in sediment supply. The lower-gradient and alluvial nature of this reach makes it prone to excess sediment deposition. Loss of habitat diversity and complexity due to artificial channel confinement, low quantities of instream LWD, elevated fine sediment loading, and degraded riparian-floodplain conditions exacerbate this problem.

In 1985, the WDFW constructed artificial spawning channels within a diked area in the Gorley Springs section of the river to increase chum salmon production in the Grays River. The dike at Gorley Springs was constructed in the early 1960s in a natural deposition zone of the Grays River, disconnecting the river from its floodplain and reducing sediment storage capacity. The artificial spawning channels proved successful in terms of spawning adult returns, accounting for 30–50% of the chum salmon produced in the river system (WDFW 2001). However, this engineered solution proved to be short-lived. In December 1999, an avulsion of the Grays River breached the dike upstream of the chum salmon spawning channels, destroying them. This one catastrophic event led to the loss of approximately 20–25% of the remaining naturally reproducing chum salmon in the lower Columbia River (WDFW 2000). Previous investigations have determined that much of the remaining spawning habitat is unstable and sediment-laden, and the few stable spawning areas in the Grays River are subject to extremely variable conditions that further threaten these salmon populations. The WDFW investigated the avulsion and determined that the dike had exacerbated aggradation in the confined channel along this reach, resulting in the eventual overtopping of the dike and the avulsion (WDFW 2000). With only two population centers remaining for chum salmon in the lower Columbia River, this event emphasized the potential consequences of random catastrophic events on the continued existence of this population.

Land-use practices throughout the upper Grays River watershed, particularly timber harvest and forest road construction in the upland areas, have increased the number of landslides, debris flows, and surface erosion that have significantly increased sediment loading into the upper Grays River system. In addition, channel instability (due to aggradation), loss of riparian function, and separation of the active Grays River channel from its floodplain have led to instream habitat degradation. The loss of stable instream LWD is of special concern, considering the elevated sediment load and the lack of instream storage capacitance (e.g., channel-spanning LWD and logjams).

Historically, the watershed was dominated by old-growth Sitka spruce, western hemlock, and Douglas fir. Commercial timber harvest began in the Grays River watershed in 1905. The harvesting of second-growth forest had begun by 1953; by 1983, more than 95% of the old-growth forest within the Grays River watershed had been harvested. Currently less than 2% of the original old growth forest remains, and a dense network of forest roads covers the watershed. Due to the inherent nature of its soils and hillslopes, the Grays River watershed is prone to erosion where vegetation has been removed and has the potential to yield large quantities of sediment. In the past, dense forests mantled the watershed and

moderated both the production of sediment from hillslopes and the routing of sediment through the channel network. Timber harvest activities removed mature conifers from riparian zones, removed instream LWD and stripped floodplains of large trees, severely reducing potential sediment storage sites within channels of the Grays River by disrupting the natural, self-sustaining processes that recruit LWD to stream channels.

In summary, forest clearing and the associated construction of roads have significantly altered a variety of landscape processes. Timber harvest and forest road construction and operation in the upper Grays River watershed have significantly increased landslide activity and surface erosion on road and clear-cut sites relative to naturally forested areas. This has resulted in excessive rates of fine sediment production, with subsequent delivery to the channel network (Herrera 2005). The stream channel network of the upper watershed is also deficient in LWD, which exacerbates the fine sediment-loading problem due to the loss of natural sediment storage capacity within the stream system (Herrera 2005).

The increased sediment production resulting from timber harvest and road construction has significantly affected downstream channel processes. A variety of channel responses have been observed, including channel aggradation and flooding. Additional channel instability has resulted from the harvest of riparian forest vegetation and the loss of instream LWD. The harvest of riparian vegetation removes the most immediate source of LWD, which can act not only to provide channel stability and habitat complexity but also to effectively trap bed material and store large volumes of sediment (Herrera 2005).

Deforestation and construction of floodplain levees and dikes within the Grays River watershed have impaired geomorphic processes and, in turn, salmonid habitat. Timber-harvest activities, including logging road construction and operation, have resulted in increased sediment production, channel instability, and a decline in channel complexity, which have severely impaired salmonid habitat and increased flood and erosion risks to landowners in the lower Grays River valley (Herrera 2005). Flood-control measures have resulted in a loss of floodplain connectivity and side-channel development, which have reduced channel habitat complexity and access to historical off-channel habitats (LCFRB 2004).

6.1 Hydrology

A hydrologic model (the DHSVM) was used to simulate the streamflow responses of the upper Grays River watershed under a number of vegetation scenarios that corresponded to historical forest management conditions within the watershed. A probabilistic analysis of daily streamflow at several locations within the watershed clearly showed the range of impacts that can result from heterogeneous changes in vegetation pattern within the watershed due to timber-harvest activities.

The model simulations conducted do not represent historical streamflow regime during 1963-2004 water years because all scenario runs used a vegetation pattern that was held fixed throughout the simulation period (see Figure 6.2). These vegetation patterns correspond to one of several vegetation scenarios that represent specific periods of time in the history of forest management in the upper Grays River watershed. These model runs individually represent the range of hydrologic response expected over almost the complete period of record (42 water years) of meteorological observations.

The model simulations conducted for this study cannot address explicit analysis of change over time. However, useful statements regarding the hydrologic regime can be made based on the change in several hydrological measures (flows corresponding to several exceedance probabilities, bankfull flows, low

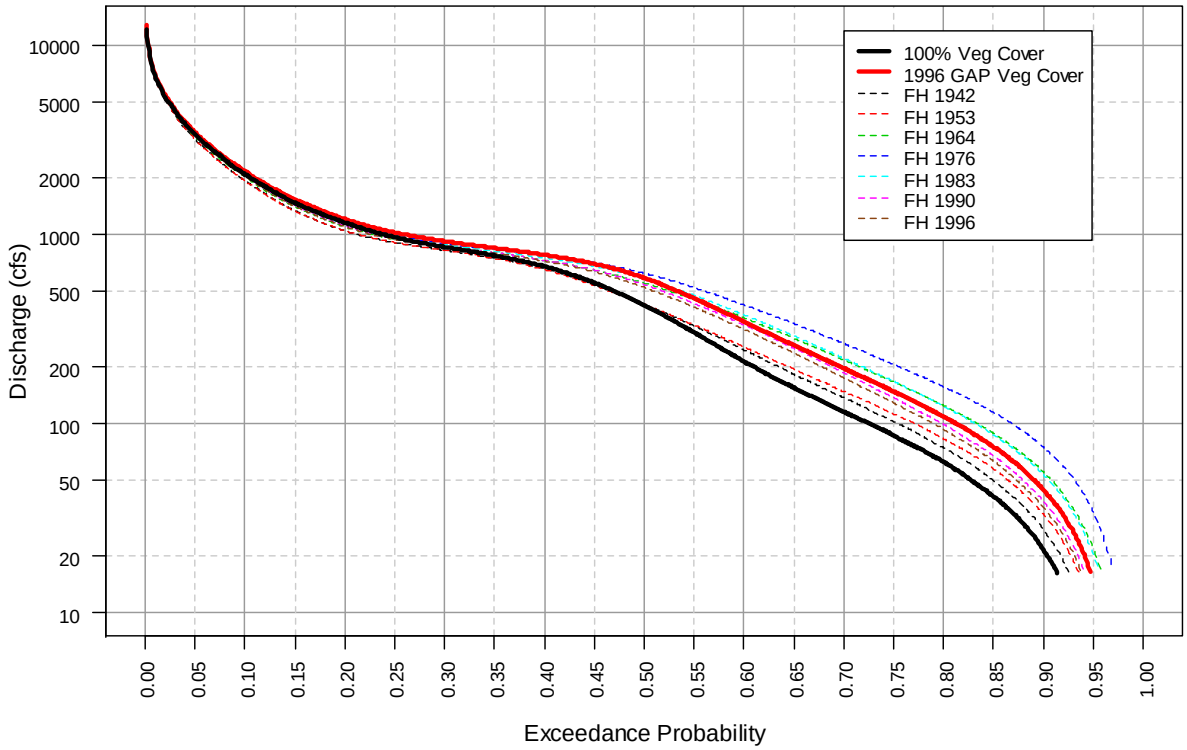


Figure 6.2. Exceedance Probability Plot for Daily Streamflow for Grays Mainstem Below West Fork and Gorley Reach

flows, summer low flows, and spawning period low flows). Although a direct comparison with historical streamflow within the Grays River Basin is not possible due to the reasons stated above, the simulations allow us to derive useful information regarding hydrologic response in the Grays River watershed.

The results of hydrologic modeling indicate that the greatest impact of vegetation scenarios is reflected in mid to low flows and not in the higher flood flows. This lack of influence of timber harvest on flooding frequency or intensity was not unexpected because changes in vegetation patterns typically influence streamflow primarily through the evapotranspiration process. During floods, conditions in the basin are expected to be wet and humid, limiting evapotranspiration. Therefore, rainfall events that cause flooding are not influenced greatly by vegetation conditions (Harr et al. 1982).

This result was illustrated specifically for the December 1999 flooding event that caused the Gorley avulsion. Although this was a catastrophic event, the flood flows were not significantly higher than typical flooding events that occur regularly on the Grays River during the rainy season. However, the effect of flooding at a local scale can be quite severe, especially if human infrastructure (e.g., levees, dikes) designed to protect property fail, as was the case during the 1999 avulsion when a floodplain dike was breached. The DHSVM accurately reproduced the timing of the flows that led to the December 1999 avulsion event. Estimated streamflow was 82% greater than 100% forest cover bankfull discharge. This was a significant but not exceptional flood event, with an estimated return period of 8–14 years relative to 100% forested conditions. These results support the conclusion that dike failure resulted from a combination of high flows, long-term streambed aggradation, poor dike design, and inadequate dike maintenance (WDFW 2000).

In general, the magnitude, frequency, and duration of flooding events and peak flows are affected relatively little by forest harvesting. However, due to reduced vegetation cover (e.g., clear-cut areas) and correspondingly greater exposed soil surfaces, source areas for sediment generation are potentially more extensive within the upper Grays River watershed. In addition, the extensive network of logging roads also increases the surface area of exposed sediment, as well as providing a direct connection to headwater stream channels. This implies that sediment delivery may be greater to the channels, thereby increasing the sediment concentration and loading during floods. Although flood frequency, duration, and intensity may not change on a basin-wide basis due to timber-harvest activities, these floods still provide the stream power necessary to mobilize and transport sediment and debris. In addition, localized effects of runoff from harvested areas and forest roads can have a significant hydrologic influence at the more local, stream-channel scale. This influence includes the greater stream power to transport sediment from upland source areas through the stream network. Elevated streamflow in these headwater channels can also facilitate greater debris flows and streambank erosion on a localized scale.

The model analysis also indicated that there was only a minor change in the intensity and mean duration of mid-range or bankfull flows corresponding to the various forest harvest scenarios. This behavior is consistent with the relatively small changes in magnitudes of high flow across the forest harvest scenarios. It is these bankfull flow events that can cause stream channel realignment, streambed scour, and streambank erosion that can have an impact on instream habitat conditions and biota.

In contrast to higher flows (e.g., flooding events or bankfull flows), low flows generally occur during the dry season, when evapotranspiration is greater. In addition, relative magnitude of evapotranspiration during low flows is larger than during peak-flow events. Therefore, low flows are affected more by changes in vegetation cover compared to high flows. The change in maximum duration of spawning-period low flows varied depending on the location in the river system and the vegetation scenario.

The DHSVM simulation results indicate the maximum impacts to runoff production and streamflow would occur under the 1976 land-use scenario, which represents the peak intensity of timber-harvest activities. Impacts to flood flows were minor, and the impact on low flows only moderate. These results are essentially mirrored by those of the Thomas and Meghan (1998) analysis of timber harvesting in the H. J. Andrews Experimental Forest in Oregon. The hydrologic impacts of timber harvest are relatively minor, and the effects are relatively short-lived, decreasing as forest regrowth occurs.

Different parts of the Grays River watershed respond differently to forest harvest and associated change in the spatial distribution of vegetation. These differences occur due to differences in response related to whether the spatial location of the harvest directly affects the hydrology in the contributing area of an assessment location.

Although the DHSVM hydrologic modeling analysis did not specifically evaluate the effects of roads on runoff production and streamflow, the extensive road network has likely increased the magnitude of peak flows as well as altered their timing by intercepting shallow subsurface flow and moving it more rapidly to the channel system. In addition, surface runoff from logging roads can have a significant effect on local streamflow (Bowling et al. 2000). However, the greatest hydrologic impact of harvest-related activities is likely through increased sediment yield, both by direct erosion on road surfaces and, more important, through a dramatic increase in hillslope mass wasting events from a combination of timber removal and construction of the associated road network. This conclusion is supported by the geomorphic analysis conducted as part of this project work (Herrera 2005).

6.2 Geomorphology

Historical sediment budgets were constructed for the subbasins of the upper Grays River watershed by synthesizing historical timber harvest records with correlations established between the time since harvest and sediment yield from mass-wasting events and forest road surface erosion data. Sediment yield from soil creep, which is assumed to be independent of timber harvest practices, also was included in the historical sediment budget. Historical sediment budgets were constructed for each of the eight reference years in which timber harvest data are available (Scott 2004). Sediment yield from mass wasting, roads, and soil creep were then combined to compute the annual sediment yield (Herrera 2005).

Reconstruction of the sediment budget indicates that the total sediment yield for the upper Grays River watershed was nearly constant from 1942 through 1964 and ranged from approximately 13,000 to 18,000 tons/year. The estimated sediment yield increased sharply to approximately 87,000 tons/year by 1976 and continued to climb to more than 250,000 tons/year by 2003, indicating an order-of-magnitude increase in sediment production (Herrera 2005).

The findings of the geomorphic assessment indicate that most of the sediment delivered to the channel network is derived from hillslope mass-wasting processes in clear-cut areas of the upper watershed. Roads and soil creep contribute a relatively minor proportion of the total sediment yield. However, the geomorphic assessment identified roads as a likely contributor to landslide and debris-flow initiation. Hence, the jump in sediment yield between 1964 and 1976 may be linked to increased road construction as well as timber-harvest activities (Herrera 2005).

The grain-size distribution of sediment delivered to the channel network varies among the different sediment sources. The analysis of sediment samples collected from landslide deposits and observations during the field reconnaissance indicate that landslides are characterized by a poorly sorted grain-size distribution that includes both fines and coarse gravel. Deep-seated landslides, which incorporate bedrock, produce coarser grain-size distributions. Most of the poorly sorted sediment supplied by mass wasting is broken down to suspended load by particle attrition during fluvial transport. Sediment delivered to the channel network by soil creep is expected to have the same grain-size distribution as shallow landslides, which consist of colluvium. In contrast, the grain-size distribution of sediment originating from road surfaces should be dominated by fines. However, surface runoff and bank erosion associated with the road prism also can deliver some coarse sediment to the channel network (Herrera 2005).

Although landslide-derived sediment is coarser than sediment derived from road surfaces, landslides delivered 20 to 200 times more sediment than roads during the 20 to 30 years following timber harvest. Hence, the volume of fine sediment delivered by mass-wasting processes may exceed by many times the quantity of fine sediment generated from road surfaces during this same time period (Herrera 2005).

The contemporary sediment yield calculated for 2003 in the upper Grays River watershed is consistent with the findings of Grant and Wolff (1991), which indicate that the average annual production of sediment after clear-cutting was about 4 to 12 times greater than the undisturbed rate and remained higher than pre-harvest rates more than 20 years later. Based on their computed trends, Grant and Wolff (1991) forecast sediment production to remain elevated above background rates for 30 years after harvest. Montgomery et al. (2000) found sediment yield from timber harvest-related landslides in the Oregon Coast Range at 3 to 9 times the natural background rate for the region. Given the current timber harvest

rotation in the upper Grays River watershed of about 40 years, sediment production will remain relatively high and ultimately could result in significant soil loss and reduced timber productivity (Herrera 2005).

Past and future timber harvest practices will continue to influence the rate of sediment yield to the Grays River channel network. The current trend of second- and third-growth harvest rotation is expected to reduce the reported 10-year lag between the loss of root strength and the peak in landslide frequency because smaller second- and third-growth roots are typically weaker than roots of old-growth trees. However, sediment yield might decline if the frequency of harvest rotation or the overall harvest rate is reduced. The harvest rate within the upper Grays River watershed has declined since 1980, and future sediment yield to the channel network is predicted to decline as well. Sediment yield could be reduced by also eliminating timber harvest from slopes steeper than about 65%, where sediment yield from mass wasting is estimated to be 1,000 times greater than the yield on slopes less than 65% (Herrera 2005).

Contemporary rates of sediment production by mass wasting are not sustainable with long-term soil production rates and could lead to the eventual depletion of soil on steep slopes, which would, in turn, severely impair timber production and other recreational and wildlife uses within the watershed. In a comparative study of debris-flow characteristics in old-growth and industrial forests, Bunn (2003) found that the sustained short-rotation harvest of headwater basins and removal of old-growth wood from headwater channels is diminishing soil depth on hillslopes and leaving headwater channels with an increased sediment flux and a consequent increase in sediment output to low-gradient response reaches.

Field observations in the upper Grays River watershed indicate that some hillslopes already have been stripped to bedrock by widespread mass wasting. Past timber harvest practices that removed instream wood (either by snagging or splash-damming) and stripped floodplains of large trees severely reduced potential sediment storage sites within headwater channels of the upper Grays River watershed by disrupting the natural, self-sustaining processes that recruit wood to channels. Past timber harvest practices not only increased the rate of sediment yield to the channel network but also accelerated the delivery of this sediment to the mainstem response reach by eliminating most of the natural sediment storage capacity provided by instream wood (Herrera 2005).

If channel response in the Grays River mainstem lags behind harvest by approximately 35 years, as predicted by the geomorphic assessment, the current instability within the mainstem (posited to be related to the 1980-1990 spike in the timber harvest rate) can be expected to continue and possibly increase through 2030. Response reaches throughout the upper Grays River watershed may be prone to continued instability and more avulsions so long as sediment supply exceeds threshold transport rates. The remobilization and routing of sediment stored within the channel network may extend this response period through the latter half of the twenty-first century (Herrera 2005).

Aggradation and natural straightening of the channel are typical morphological responses to an increase in sediment loading. The local increase in slope caused by continued aggradation (as well as confinement by levees) will shift the depositional front of the mainstem downstream. Confinement and straightening of the mainstem response reach by levees increase the local transport capacity (through increases in both slope and flow depth, which, in turn, increase shear stress) and shift future sediment deposition and channel response downstream to the SR 4 sub-reach. Additional channel avulsions are likely to occur if measures are not taken to maintain and raise existing levees and revetments concurrently with the anticipated sediment aggradation. However, a more long-term solution to this problem would be to remove or set back any levees that are currently encroaching on floodplain areas (Herrera 2005).

The response to increased sediment supply is compounded by levees within the mainstem floodplain. In general, the levees restrict the natural tendency toward channel migration and floodplain sediment deposition. Isolation of the channel from the floodplain accelerates local aggradation and increases the potential for channel avulsion. Although levee construction may have initially provided short-term stability to portions of the channel and floodplain, channel confinement and floodplain isolation by the levees has forced channel adjustments to the shorter, unconfined segments of the river. Consequently, the floodplain may experience periods of channel stability punctuated by high-magnitude variability in channel configuration (Herrera 2005).

The observed response of the mainstem Grays River is analogous to historical channel changes in the lower Skokomish River of the southeastern Olympic Peninsula following extensive timber harvesting. Stover and Montgomery (2001) identified three phases of channel response to historical disturbance in the Skokomish River Basin. The first phase involved rapid channel incision of nearly 0.5 m following riparian timber harvesting and removal of instream wood. The second phase was characterized by fluctuations in bed-surface elevation of up to 1 m that coincided with widespread timber harvest and road development in the basin during the 1940s and 1960s. Stover and Montgomery (2001) attributed the oscillations in bed elevation to sediment pulses moving through the channel network. The sediment pulses were linked to concurrent timber harvest activities in the upper basin and the release of sediment stored in the nearby south fork following the harvest of riparian forests and removal of instream wood. Channel filling, increased channel width, and fining of bed sediment characterized the third phase of channel response through at least the end of the study in the late 1990s (Stover and Montgomery 2001).

In contrast to results of the analysis for the upper Grays River, the onset of channel aggradation on the Skokomish began rapidly, approximately 10 years after the commencement of intense upstream timber harvesting, and continued at a steady pace through 1997, at the end of the study period. Results of the Stover and Montgomery (2001) study suggest a minimum of 50 years for the lower response reach of the Skokomish River to adjust to the influx of sediment from timber harvesting. Timber harvesting and road construction in the headwaters continue to contribute to ongoing aggradation and recurrent flooding within the Skokomish River valley.

Observations from the field studies and modeling efforts can be generalized into temporal relations among watershed disturbance, sediment yield, and channel response within the upper Grays River (Figure 6.3). Sediment yield to the channel network increases sharply above natural background levels shortly after the onset of timber harvest activities and reaches a peak that lags behind the peak harvest rate by approximately 25 years. Included in this period is the 10-year lag between harvest and peak in landslide frequency. Channel response to increased sediment yield may include aggradation, decreased sinuosity, increased bend curvature, and increased frequency of flooding and channel avulsions (Herrera 2005).

The onset of channel change may occur rapidly in alluvial reaches after timber harvest or several decades following harvest. The magnitude of channel change continues to increase (despite the decline in sediment yield) due to the mobilization of sediment stored within the channel network during high-magnitude storm events. This reduction in sediment storage is magnified by the removal of instream LWD and harvest of riparian forests that would otherwise supply LWD to channels. Under this conceptual model and with the inherent uncertainties involved in predicting sediment production and transport, channel adjustment could continue for at least 50 years after current timber harvest activities. Channel response in the Grays River watershed is already 25 years out from the peak in harvest rate.

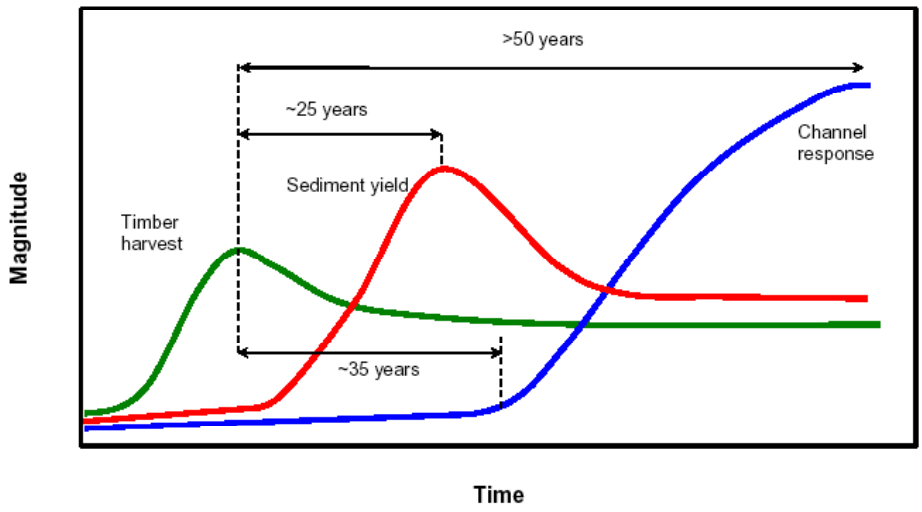


Figure 6.3. Temporal Trends in Sediment Delivery Processes to the Upper Grays River Channel Network and Channel Response in Apparent Response to Historical Timber Harvest (Herrera 2005)

Based on results of Stover and Montgomery (2001), channel adjustment in the lower Grays River could continue for at least another 25 years, under the current (2003) timber harvest rate. Changes in timber harvest rates and methods, as well as improvements in forest road construction, maintenance, and decommissioning could significantly reduce the period of channel instability and sediment loading.

The magnitude of impacts on fish habitat caused by increased sediment yield and channel instability within the Grays River mainstem response reach will be determined by the ability of restoration efforts to counteract the destructive effects of past and ongoing land-use activities within the upper watershed. Rivers with high sediment loads can support productive fish populations if they contain abundant instream LWD, which promotes pool formation, protective cover, substrate diversity, and channel migration into floodplain forests for self-sustaining LWD recruitment (Herrera 2005).

The goal of improving salmonid habitat and reducing risks to property and infrastructure can be achieved by restoring riparian forests and instream roughness (i.e., LWD) to appropriate channel reaches and by focusing development, timber harvest, and agriculture toward portions of the watershed where they will not adversely affect geomorphic processes. In addition, the establishment of an unconstrained channel migration zone (CMZ) in the Grays River response reach upstream of SR 4 through a program that is sensitive to landowners and the community, and that focuses land acquisition, conservation easements, and levee setback or removal toward areas with high potential for flood and erosion risks and high potential for habitat restoration (Herrera 2005).

Current channel conditions in the upper Grays River watershed present significant opportunities for habitat restoration. The riparian zones of the upper watershed are beginning to recover from historical logging and will eventually provide natural LWD recruitment. In the interim, instream LWD installation could bridge the gap and provide both needed habitat functions and sediment storage capacity. However, successful restoration depends on recognizing and accommodating ongoing geomorphic processes such as channel migration and fluctuating sediment loads.

The most direct and immediate actions to restore habitat are the reintroduction of functional wood debris and reestablishment of mature riparian forests. Returning functional LWD to the Grays River response reach would increase sediment storage, increase channel complexity and pool frequency, diffuse flood peaks, and increase channel length. Hard points within the floodplain created by stable wood debris or installed logjams would allow islands or riparian forests to mature within the channel migration zone. The placement of stable LWD structures in probable flow paths would also disperse and bifurcate flow, decreasing the erosive potential of flows while providing the habitat benefits of increased hydraulic complexity where functional wood and recruitment potential is currently lacking. These hard points also promote the formation of deep channels and pools in place of shallow, braided channels that typically result from increased sediment loading. Intentional placement of engineered logjam (ELJ) structures also would provide predictability of locations of future flooding and channel changes, such as avulsions, thus improving flood and erosion management (Herrera 2005).

In summary, the findings of the Grays River watershed geomorphic analysis indicate that the mainstem river has been significantly altered from its natural condition by accelerated sediment supply within the upper Grays River watershed and by the construction of floodplain levees in the middle and lower river system (Herrera 2005). Sediment loading to the river system appears to be at least an order of magnitude greater than the natural background levels found in undisturbed forested watersheds of the region (Herrera 2005). Response time for the cumulative effects of basin-wide timber harvest to significantly affect the mainstem response reach appears to be approximately 35 years (see Figure 6.3). Thus, current instability within the Grays River mainstem can be expected to continue and possibly increase through 2030. The 1999 avulsion represents the most significant historical response of the mainstem channel to date, but events of similar magnitude are likely to continue and progress downstream as sediment stored in this response reach is transported downstream to Highway 4 and into the lower mainstem (Herrera 2005). The magnitude of channel change continues to increase due to the mobilization of sediment stored within the channel network during high-magnitude storm events. This reduction in sediment storage is magnified by the removal of instream wood and harvest of riparian forests throughout the Grays River watershed (Herrera 2005).

6.3 Effects of Land Use on the Aquatic Environment of the Grays River Watershed

6.3.1 Current Aquatic Habitat Conditions

Aquatic habitat degradation in the upper Grays River watershed associated with timber harvest has been well documented (WCC 2002; LCFRB 2004). Effects of timber harvest on upper watershed channel characteristics documented during this project (Herrera 2005) are summarized in Section 6.2. The most significant effects of timber harvest and associated road construction activities are on sediment delivery processes and LWD recruitment affecting aquatic habitat throughout the upper watershed. Upper Grays River channels are unstable and turbid during flood events (WCC 2002). Previous evaluations of the Grays River concluded that alterations to aquatic habitat resulting from altered land use included elevated peak flows, inadequate summer low flows, passage obstruction due to low summer flows and culverts, potential nutrient limitation due to reduced salmonid escapement and lack of LWD, lack of quality pool habitat, excessive fine sediments in substrates, and poor bedload retention associated with lack of LWD (WCC 2002). Our hydrologic assessment suggests that effects on the hydrologic regime are less severe than impacts on sediment delivery and LWD processes. We found that hydrologic changes in the upper

Grays River watershed associated with timber harvest hydrologic regime include increased low flows and insignificant effects on high flows. This is in contrast to IWA process findings of reduced summer low flows and elevated high flows (WCC 2002). Whether increased summer low flows have improved salmonid passage at existing obstructions is unknown. We also found that the duration of low flows during the November–December spawning period was reduced during periods of timber harvest. In general, lower flows during the spawning season can concentrate fish and increase the occurrence of redd superimposition, restrict access to shallow spawning areas, and increase the risk of egg mortality due to freezing (McNeil 1967; Vadas 2000). In Chapter 5, we found that higher water levels were generally associated with increased spawning habitat availability within the range of flows we examined. We can not conclude with certainty, however, whether any of these effects have occurred as a result of reduced duration of spawning period low flows in the Grays River.

In the lower Grays River, primary habitat-forming processes affected by agricultural practices and timber harvest are sediment delivery and LWD density. Specifically, bank hardening (dikes, levees) and riparian and floodplain clearing are responsible for these impairments. Aspects of degraded habitat identified previously (WCC 2002) include bank and bed instability, low habitat diversity with few pools, and low water conditions restricting access and producing excessively high summer water temperatures. Our findings (Chapter 5 and discussed in Section 6.3.3 below) also suggest that bedform diversity is lacking, documenting reduced riffle:pool frequency and the potential for reduced hyporheic exchange. Although the 1999 avulsion restored floodplain width within the upper spawning reach, the new Grays River channel has not stabilized and currently provides limited spawning habitat. Channel structure and diversity also have been affected by agricultural practices, resulting in loss of backwater and secondary habitats and quality pools. Habitat-limiting factors in priority reaches for chum, fall Chinook, and coho salmon include habitat diversity, sediment, temperature, and channel stability. Finally, aquatic habitat in the lower Grays River suffers from impacts delivered from the upper watershed. These include excessive bedload, producing shifting unstable channels and substrates containing excessive sediments.

6.3.2 Aquatic Species Status

The biological assessment conducted as part of this project (Appendix I) documented fair to poor biological integrity at most of the sample sites, with very few sites having excellent integrity. These results suggest that cumulative effects of human land-use activities in the watershed have compromised the biological integrity of the Grays River aquatic ecosystem and are consistent with existing habitat assessments (Wade 2002). Fish population trends in the watershed are consistent as well, in that all focal fish populations (salmon, trout, Pacific lamprey) for which assessments have been conducted were found to be depressed (WDFW 2000, 2002). Specifically, the WDFW salmonid stock inventory (SaSI) rated population trends for chum and fall Chinook salmon, cutthroat trout, and winter steelhead as depressed. The status of coho salmon and Pacific lamprey are unknown but expected to be depressed.

6.3.3 Spawning Capacity and Factors Affecting Spawning Habitat Availability

We found that most of the aquatic habitat in the Grays River study area was suitable for chum and fall Chinook salmon spawning based on velocity, depth, and substrate. Of the total available area, 89.5% was within chum salmon spawning limits and 64.2% was within fall Chinook salmon spawning limits. We concluded that redd capacity appeared to be sufficient to support historical runs of chum (8,000 to 14,000 fish; LCFRB 2004) and fall Chinook salmon (1,500 to 10,000 fish; LCFRB 2004). We also found that

considerable suitable spawning habitat was available for both chum and fall Chinook salmon across a range of flows, suggesting that the minor hydrologic changes modeled in Chapter 3 are unlikely to significantly affect spawning habitat availability.

We identified several habitat factors that may limit the availability of spawning habitat for chum and fall Chinook salmon. First, hyporheic temperature was more limiting to available spawning habitat than velocity, depth, and substrate, if spawning chum salmon require areas with warmer hyporheic water typically associated with upwelling and fall Chinook salmon require areas where surface and hyporheic temperatures do not differ, typically associated with downwelling. The hyporheic temperature requirement reduced the area suitable for chum salmon to 29.6%. Although others (Leman 1993; Geist and Dauble 1998; Geist et al. 2002) have found that salmon spawning habitat availability is limited by hyporheic flow, we did not document an association between fall Chinook salmon spawning and hyporheic temperature.

Second, our findings suggest that channel bedform diversity may be lacking in lower Grays River spawning reaches. We did not find a strong association between bedform and hyporheic exchange or salmon spawning, yet the role of bedform in these processes is well documented and theoretically based (Vaux 1962; Leman 1993; Geist 2000; Baxter and Hauer 2000; McHugh and Budy 2004). We found riffle:pool spacing of 10.2 to 34.1 channel widths across all reaches, whereas Montgomery et al. (1995) suggested that riffle:pool channels have riffle:pool spacing values of 0.2 to 3 when high levels of bedform-forcing LWD are present. Our study reaches in general appear to lack the degree of bedform structuring that would be expected in historically forested watersheds such as the Grays River. Simplified channel bedforms are associated with limited numbers and depth of pools and limited hyporheic exchange.

6.3.4 Factors Reducing Spawning Habitat Quality

Decline of Grays River chum and fall Chinook salmon may be a function of spawning habitat quality rather than quantity. Considerable evidence (LCFRB 2004; Herrera 2005) suggests that increased sediment delivery to focal spawning areas is largely responsible for a decline in quality of salmon spawning areas in the Grays River. Fine sediments are detrimental to salmonid incubation success (Cedarholm et al. 1981; Tagart 1984; Scrivener and Brownlee 1989), reducing water flow through redds and water quality (summarized in Chapman 1988). Fine and coarse sediments may bury redds, entombing juvenile salmonids as they attempt to emerge (Lapointe et al. 2000). The extent of colmation, or substrate sedimentation, is directly linked to land-use practices that increase sediment loading, such as timber harvest and associated road construction (Karr and Schlosser 1978, Platts et al. 1989). Sedimentation of spawning riffles associated with timber harvest in the lower Columbia tributaries has been documented (Fulton 1970). Riffles once used by spawning salmon may no longer be suitable due to compaction and loss of hyporheic exchange because hyporheic exchange is reduced in substrates containing excess sediments. Although hydraulic conductivity and specific discharge values we documented in the Grays River study area (Chapter 5) were generally within the range found by others in salmonid spawning habitats, the relationship between flow velocities through salmonid redds and incubation success is complex and dependent on multiple factors including several we did not measure. Additional study would be required to assess effects of fine sediment on incubation success in the Grays River. Because timber harvest and associated activities produce excess sediment throughout the watershed, impacts are

expected on all focal fish species. However, sedimentation of spawning substrates may be a particular problem in the Grays River spawning reach because it is a low-gradient response reach already prone to sedimentation.

A second significant factor that may be reducing the quality of salmon spawning areas and potentially salmon spawning success is channel and substrate instability. Although documenting channel instability was not an objective of this project, channel conditions documented in the geomorphic assessment (Chapter 4) as well as observations and challenges experienced during spawning habitat assessment (Chapter 5) confirm that channels within spawning area contain excessive bedload and little LWD. The Upper Grays (Gorley) reach is particular unstable (Appendix H). The WCC (2002) identified lack of stable spawning habitat as the primary physical limitation on current chum salmon production. Activities such as LWD removal from riparian areas, floodplains, and channels have contributed to channel instability throughout the watershed. Excess coarse bedload also is associated with laterally unstable channels (Herrera 2005). In addition, excess fine sediment may decrease the flow magnitude necessary to mobilize streambed sediments thereby reducing the stability of the streambed and potentially leading to scour of spawning habitat (Wilcock et al. 2001).

The 1999 avulsion, as well as channel changes since then (see Sections 4.4, 5.2), provides graphic evidence of channel instability in the Grays River. Scour has reduced incubation success and can cause complete redd loss flows (Montgomery et al. 1996; DeVries 1997; Lapointe et al. 2000). Brunke and Gonser (1997) suggested that the natural balance between colmation (substrate sedimentation) and bed scour can be disrupted by a variety of anthropogenic activities. Because land-use activities produce unstable channels throughout the watershed, channel stability impacts are expected on all focal fish species. However, channel instability may be a particular problem in the Grays River Gorley reach because it is naturally dynamic and prone to channel instability.

The third significant factor that may be reducing chum salmon spawning habitat quality in the Grays River is a lack of habitat diversity associated with agricultural practices such as levee and dike construction and LWD removal. Channel constriction separates the river from its floodplain and restricts natural channel migration and off-channel habitat development. LWD contributes to channel bedform diversity and system productivity through its effects on flow and contributions of nutrients and habitat structure. Complex floodplain morphologies, including islands, bars, and secondary channels, have been associated with hyporheic exchange and therefore salmon spawning habitat (Wondzell and Swanson 1996; Geist and Dauble 1998; Fernald et al. 2001; Kasahara and Wondzell 2003). Kasahara and Wondzell (2003) found that in a model of a wide alluvial valley, removal of secondary channels reduced hyporheic discharge by 25%. Complex off-channel habitats also are important juvenile rearing habitats and have been associated with increased productivity (Beechie et al. 1994). Bedform diversity associated with inchannel LWD also has been associated with salmon spawning habitat (Montgomery et al. 1995). Although the 1999 avulsion partially restored floodplain width within the upper Grays River spawning reach, lower reaches remain constrained.

6.3.5 Salmonid Recovery in the Grays River

Although much is known regarding the effects of timber harvest and agriculture on channel conditions and salmonid spawning habitat in the Grays River watershed, information needs critical to salmonid recovery remain. We estimated that sufficient spawning habitat is currently available to support historical run sizes. However, most of the available spawning area may be of relatively low quality for

both chum and fall Chinook salmon. In addition, factors such as redd superimposition (characteristic of chum salmon) in high-quality spawning habitat may reduce efficiency of spawning habitat use, and our estimates of habitat availability may be overestimates. Redd success in high-quality habitat may be relatively low if redd superimposition occurs in these areas. Substrate quality may reduce incubation success in some spawning areas such that some spawning habitat is ultimately unproductive. Fine-scale study of chum salmon spawning habitat use and redd success would be required to fully evaluate the adequacy of available spawning area to support historical run sizes. Successful restoration of the lower Grays River may require some additional evaluation of the nature, location, and extent of successful chum salmon production and threats to incubation success.

Protection and management of existing spawning areas may be important to short-term survival of chum and fall Chinook salmon in the Grays River. For example, chum salmon spawning is concentrated in Crazy Johnson Creek in spite of the apparent low-quality habitat it provides. This may be due to the dominating influence of hyporheic flow, providing strong homing cues and desired flow conditions, and possibly higher incubation success rates. Crazy Johnson Creek may also be more stable and less vulnerable to excessive sediment delivery and scouring flood flows from the upper Grays watershed because it is rarely inundated by the Grays River (Herrera 2005). Chum salmon spawn in consistent locations (see Section 5.3), allowing for focused attention on spawning habitats currently supporting production. Focus on existing spawning areas may stabilize the chum salmon population for the short term.

The need for habitat restoration to address declining chum salmon and other fish populations in the Grays River has been clearly identified in the *Grays River Sub-Basin Plan* (LCFRB 2004) and the WRIA-25 habitat limiting factors analysis (WCC 2002). Actions focused on restoration of sediment delivery and LWD recruitment processes are key to salmonid recovery. Removal of channel constrictions in the lower river will reactivate habitat-forming processes across the Grays River floodplain in key salmonid spawning reaches of the lower river. Because watershed recovery to reduce sediment delivery and restore LWD loading following land-use modifications will require decades (Platts et al. 1989), short-term actions may be required to maintain existing spawning areas as well. Thus, priority actions are those that restore habitat-forming processes (long term) and maintain existing habitats (short term).

6.4 Grays River Watershed Restoration and Salmon Recovery

In general, the term *restoration* is used to describe a range of aquatic resource improvement projects. However, it is important to acknowledge the difference between integrated watershed restoration and instream habitat enhancement or rehabilitation efforts. In developing restoration plans, there also needs to be a clear understanding of the linkage between watershed land-use activities and the condition of the aquatic ecosystem. Streams and rivers need to be seen as integral components of the watershed landscape and not as discrete channels that can be “fixed” without addressing the connected issues within contributing upland areas (Lichatowich 1999).

Restoration, by its scientific definition, means to return the resource to some level of pre-disturbance condition (Bradshaw 1996), although it is used in most cases to encompass a broad range of activities. In contrast, habitat *enhancement* or *rehabilitation* is defined as the improvement of habitat from existing conditions (Bradshaw 1996). Restoration can be classified as active or passive. Passive techniques seek to restore watershed processes or habitat conditions by curtailing detrimental land uses or human activities, protecting functional habitat, or establishing conditions that will allow the ecosystem to recover

naturally. Acquisition and conservation efforts fall into this category (NRC 1996). Active restoration typically involves more direct human intervention and usually seeks a more rapid recovery of ecosystem structure and function. Instream habitat (LWD) enhancement projects, logging-road decommissioning, and riparian reforestation projects are example of active restoration (NRC 1996).

There is also a widely held recognition that a comprehensive, holistic ecosystem approach is necessary if the recovery of endangered salmonid populations is to be successful. It is also generally accepted that a long-term restoration effort will be required to restore the natural structure and function of these impacted rivers and streams. Specifically, a process-based approach to watershed restoration is widely accepted as the most effective in the long term (Figure 6.4). In addition to long-term restoration efforts, short-term measures will be needed to bridge the gap between current degraded conditions and future restored functional ecosystems. Short-term efforts include improving forest management practices as well as active instream and riparian enhancement or rehabilitation measures (Beechie et al. 2003).

In general, the application of appropriate instream habitat enhancement or rehabilitation techniques such as LWD installation does not hinder long-term, process-based natural recovery that may already be under way. In most cases, these active restoration techniques tend to bridge the gap between current degraded habitat conditions and future self-sustaining natural habitat conditions.

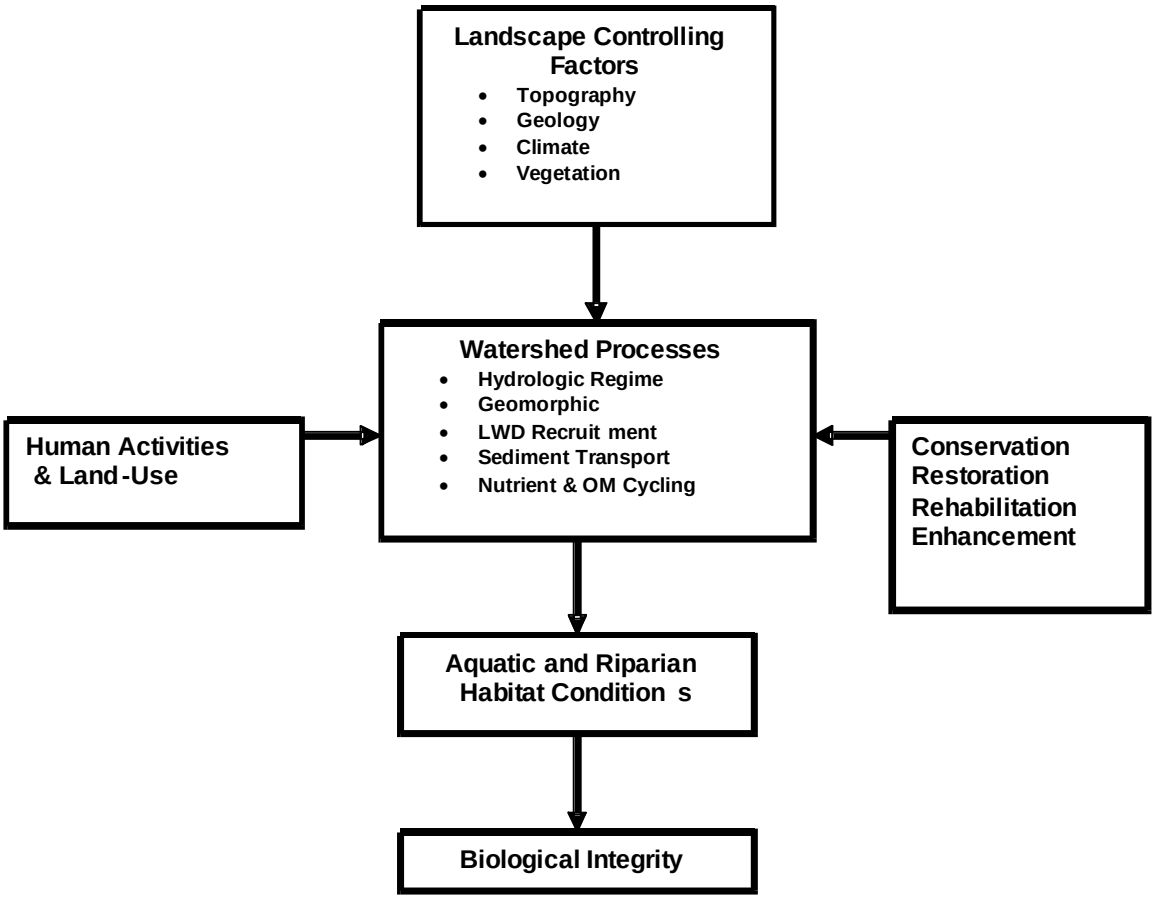


Figure 6.4. Conceptual Model Illustrating the Linkages Between the Landscape, Watershed Processes, Habitat Conditions, and Biological Integrity, as Well as the Interaction of Human Land-Use Activities and Restoration Efforts (adapted from Beechie et al. 2003)

Most scientists and resource managers agree that achieving a significant improvement in aquatic resource quality likely will require an integrated watershed approach and a combination of coordinated (long-term) restoration and (short-term) rehabilitation efforts. These efforts also will require recognition of the linkages between landscape conditions, physical processes, and biological integrity within the watershed ecosystem. Programs and projects that are cooperatively developed and implemented by stakeholders, resource managers, regulatory agencies, jurisdictions, and scientists likely will be the most successful. Most successful rehabilitation projects follow a sequence of watershed-based assessment, problem description, rehabilitation prescriptions, and implementation (Murphy 1995; Beechie and Bolton 1999; Beechie et al. 2003; Montgomery et al. 2003).

In addition to an integrated conservation, restoration, rehabilitation, and enhancement program that addresses watershed impacts, a comprehensive monitoring program is also needed. Monitoring is a foundation of an effective *adaptive management* program. Adaptive management provides a feedback mechanism so that restoration efforts can be modified or fine-tuned over time. Monitoring should include several comprehensive measures of river health (i.e., physical, chemical, and biological components of water quality).

The following key principles should be taken into account when developing a comprehensive watershed management and restoration plan:

1. River and stream ecosystems are spatially and temporally dynamic. The variability of these systems is largely driven by their natural disturbance regime and, as a result, is not static (e.g., dynamic equilibrium concept). Native biota have adapted to this natural disturbance regime and the resultant dynamic environment.

2. The resiliency of river and stream ecosystems is influenced by the type of disturbance, as well as by the intensity, duration, and return-frequency of the disturbance. Ecosystems are also interconnected in space and time.

3. A dynamic mosaic of habitat types and stages of development across the watershed landscape is critical to maintaining biological diversity. This holds true from the longitudinal, lateral, and vertical perspectives of riverine ecosystems.

4. Human (land-use activities) disturbance is typically very different from natural disturbance and can overwhelm the resistance and resilience of aquatic and forest ecosystems, often causing significant long-term alterations in ecosystem structure and function.

5. Humans are an integral part of the terrestrial and aquatic ecosystems. Land-use influences and socio-economic factors must be accounted for in all aspects of watershed (ecosystem) management. In most cases, human-related disturbance, such as timber-harvest activities, can be managed to minimize the impact on aquatic ecosystems and native biota.

Riverine ecosystem restoration should be approached from a watershed perspective and with the recognition that landscape-scale (habitat-forming) processes should be the focus of long-term recovery efforts. Restoration efforts should be carried out at the scale of these habitat-forming processes. With rivers and streams, restoration efforts must include the longitudinal (headwaters to estuary continuum), lateral (floodplain perspective), <u>and</u> vertical (instream structure and hyporheic influences). In general,

processes in the upper watershed should be addressed prior to or concurrently with downstream issues. Finally, recovery efforts that are focused on salmonids must address all applicable life-history stages for all species of concern. Trying to "fix" spawning habitat without also dealing with rearing habitat or migration barriers is a recipe for failure (Murphy 1995; Beechie and Bolton 1999; Beechie et al. 2003; Montgomery et al. 2003).

6.4.1 Background

Timber-harvest activities and logging-road operations and maintenance have the potential to alter almost all of the environmental components that affect the quantity and quality of instream and riparian habitat, as well as physiochemical water quality and biological food web interactions of the affected aquatic ecosystem (FEMAT 1993; NRC 1996; WA-DNR 2001). Early studies in southeastern Alaska (Murphy et al. 1986), the Alsea watershed in Oregon (Hall et al. 1987), and Carnation Creek, British Columbia (Hartman and Scrivener 1990), established the foundation for understanding the impacts of forest management on stream ecosystems.

Since those early studies, much has been learned about the potential impacts of timber-harvest activities and logging roads on aquatic ecosystem quality in general and fish habitat in particular (Meehan 1991). In general, any short-term benefits to aquatic resources from logging activities (e.g., short-term improved growth rates of juvenile salmonids in streams where riparian areas are logged, allowing more sunlight to reach the stream, which stimulates greater primary productivity) are offset by the long-term negative cumulative impacts (Murphy et al. 1986).

These negative impacts include

- an increase in the extent and frequency of landslides and hillslope mass-wasting events due to logging on steep slopes
- chronic surface erosion from logging roads, resulting in elevated fine sediment inputs into receiving waters
- more frequent and more severe debris flows in headwater channels impacted by clear-cuts and logging roads
- more frequent bankfull discharges and more severe flooding events
- degraded water quality (higher turbidity and sediment loading)
- migration barriers caused by logging-road culverts (improperly designed, installed, and/or maintained)
- stream channel instability due to elevated sediment loading and higher flows
- loss of riparian shading and stream temperature regulation
- loss of mature coniferous trees from riparian zones, resulting in a reduced LWD recruitment potential from logged riparian corridors

- loss of instream LWD and the resultant degradation of instream rearing habitat and channel complexity
- floodplain disconnection and loss of off-channel habitat due to channel modifications (i.e., dikes and levees) and encroachment.

This watershed assessment study has clearly demonstrated that almost all of these impacts are present in the upper Grays River watershed. Timber-harvest activities, including clear-cut logging and forest road construction, and construction of levees and dikes within the floodplain of the Grays River have impaired the natural watershed processes and, in turn, degraded or destroyed salmonid habitat. Historical land-use practices have resulted in increased sediment production, channel instability, and a decline in channel complexity (e.g., LWD), which have severely impaired salmonid habitat and increased flood and erosion risks to landowners in the Grays River valley.

Regardless of the legacy of historical land-use activities, human demands on the upper Grays River watershed will continue into the future. In all likelihood, timber harvest and related activities (road construction, operation, and maintenance) will be a major factor in the watershed for many years to come. Given that, it is also likely that it will not be possible to fully restore the natural disturbance regime that would sustain the river ecosystem and the biota that depend on its processes. Therefore, human land-use activities such as forest management need to be viewed in the context of a modified disturbance regime that can, if managed properly, provide a sustainable level of ecosystem structure and function. The challenge is to make the human influenced disturbance regime mimic the natural regime as much as is possible. To do this, human activities will need to be modified to maintain rather than hinder or exacerbate the natural ecological processes of the watershed (Reeves et al. 2002).

Although forest-management operations can vary with ownership and company policy, some general characteristics can be viewed in comparison to natural conditions. The natural disturbance regime of the coastal range ecosystem is dominated by wildfire and flooding events that occur at varying temporal frequencies (e.g., pulses) in a “patchy” spatial pattern. This disturbance regime tends to differ markedly from the human-influenced regime in areas where forest management dominates (Hobbs et al. 2002).

Under natural disturbance conditions, large quantities of standing or downed wood are generally scattered over the landscape after a major forest fire. This LWD is often delivered, along with hillslope sediments, to stream channels and valley bottoms by landslides and debris flows (Benda and Cundy 1990). The LWD delivered in these hillslope processes can be a significant portion of the total quantity of LWD present in a stream (Ralph et al. 1994). Once the sediment load declines to natural levels following the disturbance event, complex, high-quality instream habitat typically develops (Meehan 1991; May and Gresswell 2003; Reeves et al. 2003). On the other hand, mass-wasting events associated with timber harvest tend to contain mostly sediment, as a majority of the trees located within the disturbance zone have already been removed during harvest operations. In addition, most landslides and debris flows that occur in managed forests tend to be associated with forest road. These disturbance events also can deliver large loads of sediment to stream channels (Beschta 1978; Bilby et al. 1989; Montgomery 1994; Borga et al. 2005). As a consequence, stream channels in watersheds dominated by forest management tend to be simpler and lack the functional attributes that abundant instream LWD provides (Meehan 1991).

Natural and timber harvest-influenced disturbance regimes also tend to differ significantly with regard to the frequency and intensity of disturbances. The interval between disturbance events is ecologically significant because it affects the range of conditions that can develop and persist within the

ecosystem. In general, the period between major disturbances, such as wildfire and flooding, especially very widespread or intense disturbances, tends to be relatively long. This allows for a wide range of ecological conditions to develop in aquatic ecosystems. Timber harvest disturbances generally have much shorter intervals (i.e., harvest rotation cycle) and therefore do not support the development of naturally diverse ecological conditions. As a result, the instream habitat conditions needed to support a naturally diverse salmonid population may not develop in the relatively short disturbance interval common to watersheds being managed for timber harvest (Hobbs et al. 2002).

In addition to the temporal differences between natural and human-influenced disturbance regimes outlined above, there are spatial-scale variations. The spatial distribution of forest successional stages under natural conditions was heavily skewed toward mature or old-growth forest, with only an estimated 15-25% of a forest in early successional stages at any one time (Franklin et al. 2002). In watersheds such as the Grays, this relationship is reversed, with most of the forest in the early successional stages. These types of forest successional characteristics have been demonstrated to be much less supportive of salmonid populations than the natural forest pattern (FEMAT 1993; Reeves et al. 1995; Hobbs et al. 2002). Although timber harvest generally disturbs a relatively small area, these disturbed areas can be spread across the landscape in great numbers. In contrast, natural disturbance events such as wildfire tend to be larger in scale and concentrated in few areas. The widespread presence of the timber harvest disturbance regime over relatively large areas has subjected a greater percentage of watersheds to disturbance at any one time than would have happened under natural disturbance spatial patterns, which has resulted in more degraded instream salmonid habitat throughout the Pacific Northwest landscape (Reeves et al. 2002).

Managing forests based on the natural disturbance regime should produce the diverse, patchy landscape that has supported salmonid populations throughout history. In the natural disturbance environment, there always will be places where habitat is suitable and places where it is not. Salmonid species have evolved to exploit this natural variability (Benda et al. 1998; Rieman and Dunham 2000; Rieman et al. 2000).

6.4.2 Grays River Watershed Restoration Goals

The overall long-term goal for the upper Grays River watershed is to conserve and restore natural ecosystem structure and function (i.e., habitat-forming processes) that are essential to the aquatic ecosystem and long-term salmonid survival, with specific emphasis on chum salmon stocks. To achieve this goal of a self-sustaining watershed, supportive of native salmonid populations, may take 50–100 years or more. In the interim, rehabilitation projects can be implemented to bridge the gap between current degraded conditions and a future properly functioning system. These short-term measures should improve local habitat conditions until natural watershed processes are functional. It is these natural, habitat-forming processes that exist across the watershed landscape that will maintain the aquatic ecosystem over time (Roni et al. 2002).

An overall goal of improving salmonid habitat and reducing risks to property and infrastructure can be achieved with a comprehensive watershed restoration plan and by focusing development, timber harvest, and agriculture on those portions of the watershed where they will not adversely affect

hydrologic, geomorphic, or ecological processes. Specifically, the following objectives are recommended to focus watershed restoration efforts toward this goal (Figure 6.5):

1. Improve floodplain connectivity and function in the Grays River response reach. These improvements can control sediment deposition and channel migration by restoring riparian forests, reintroducing natural levels of LWD to the channel, and retiring existing levees (through either setbacks, proactive breeching, removal, or allowing river processes to erode levees) to reconnect floodplain and off-channel habitat with the mainstem river channel.

2. Limit upland land-use activities (clear-cut logging and logging roads) that trigger sediment production in the upper watershed. Utilize best available forest practice techniques for all timber harvest activities and use best available road maintenance practices for all forest roads.

3. Restore functional riparian forests and long-term LWD recruitment potential. In the interim, enhance instream structural complexity (LWD placement) to increase sediment storage within tributaries and improve instream habitat.

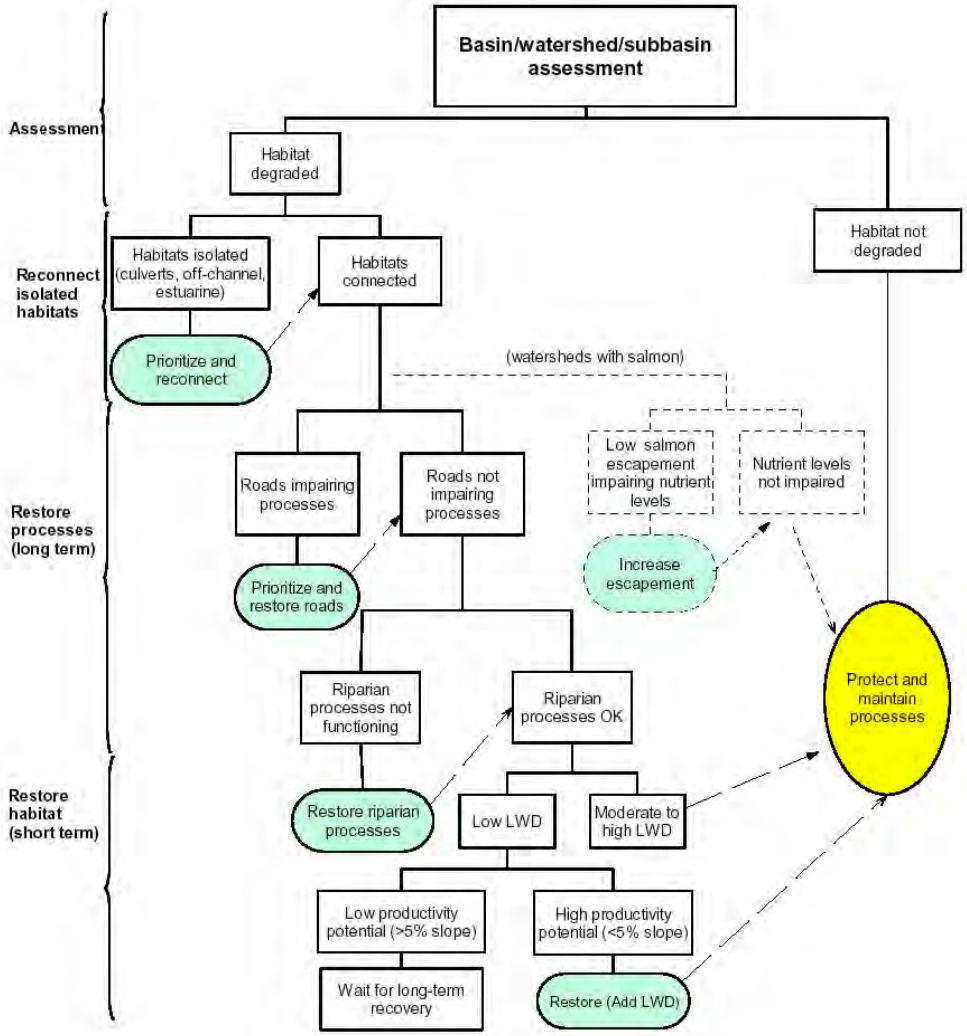


Figure 6.5. Watershed Restoration Flow-Chart (Beechie et al. 2003)

Each of these objectives addresses the historical changes in watershed processes and the resultant degraded habitat conditions documented by this and other studies within the upper Grays River watershed. Current channel conditions in the upper Grays River watershed present significant opportunities for restoring habitat-forming processes. However, successful restoration depends on recognizing and accommodating ongoing geomorphic processes such as channel migration and fluctuating sediment loads, as well as managing land-use activities such that they are sustainable and compatible with the natural disturbance regime (Roni et al. 2002).

6.4.3 Restoration Options

After gathering data on natural, historical, and current watershed and instream habitat conditions in the upper Grays River watershed, the next step is to integrate this information into a comprehensive watershed restoration and salmon recovery plan. A key part of this plan is the development of a prioritized list of restoration project options, including specific conservation, enhancement, and rehabilitation components.

If there were a way to explicitly model the linkage between changes in watershed processes, habitat conditions, and biological response (e.g., salmonid population characteristics), it would be a simple matter of running the model to see which restoration actions were most effective in attaining salmon recovery goals. However, because of the inherent complexity of natural riverine ecosystems and the limitations of current modeling capabilities, such explicit models do not exist. Therefore, the approach to restoration that is recommended is to employ scientific experience and best professional judgment to develop these prioritized restoration options (Beechie et al. 2003).

Each restoration option or project should be considered a full-scale field experiment (Roni et al. 2002). As such, effectiveness monitoring is a must, and feedback should be used to modify restoration efforts to optimize results over the long term (e.g., adaptive management).

The upper Grays River watershed restoration plan could include a wide range of options, including

- implementation of no active restoration measures and hope that revised timber harvest and road maintenance regulations are effective in supporting natural recovery
- acquisition and protection of key “refugia” sub-watersheds in the upper Grays watershed – The West Fork and Crazy Johnson Creek are two that stand out with respect to chum salmon spawning habitat.
- active and passive restoration of riparian corridors throughout the watershed
- reconnection and restoration of floodplain and off-channel habitat (e.g., levee removal)
- enhanced road maintenance and large-scale road decommissioning targeted on specific areas (i.e., West Fork and South Fork) to reduce sediment loading to headwater streams and subsequent transport downstream
- elimination or reduction of timber harvest on steep or unstable slopes to reduce sediment input from mass-wasting events
- installation of instream structural complexity (e.g., LWD) in headwater channels to capture and hold sediment within the upper watershed

- installation of instream structural complexity (e.g., LWD, ELJ structures, and/or constructed rock structures) within the mainstem response reach to encourage the formation of multiple channels, encourage hyporheic flows, trap excess sediment, and provide more stable chum spawning habitat
- construction of floodplain sediment traps and selective removal of excess sediment from the river system
- construction of artificial spawning habitat and rearing habitat for chum and other salmonids.

Based on adaptive management experience from the Pacific Northwest, it is likely that no single option will solve all the existing problems. Instead, a multiple-option approach likely will be necessary over the long term (Roni et al. 2002). Whatever the option or options chosen, the solution must be fish-friendly, sustainable, cost-effective, and technically feasible. Local support for restoration activities also is critical to the long-term success of most restoration efforts. Landowner cooperation is often a limiting factor and, as such, some restoration alternatives are better left to local groups to implement, with technical guidance provided by scientists and engineers.

Many restoration alternatives are not friendly to fish or river ecosystems in the long run and should be avoided. While these solutions may provide quick fixes to specific problems, they create other problems, particularly for the long term, and are not sustainable holistic solutions (Beechie et al. 2003). Examples of these problematic solutions include streambank armoring and levee construction.

The findings of this study, as well as the WRIA-25 habitat limiting factors analysis (WCC 2001) and the *Grays River Sub-Basin Plan* (LCFRB 2004), recommend that immediate risks to salmon populations be addressed with short-term, localized habitat solutions while at the same time working toward long-term recovery goals on a watershed scale. Both short-term and long-term approaches are essential and are fully complementary in their contributions to overall watershed restoration and the recovery of Grays River chum salmon populations. Restoration of natural habitat-forming processes is seen as a critical part of establishing a trajectory toward a sustainable ecosystem that will ultimately benefit both fish and people in the Grays River watershed.

To realize the stated restoration goals and objectives for the Grays River watershed, the following restoration activities are recommended:

- Protect existing intact and functional refugia habitat. The establishment of so-called “salmon watershed reserves” may be a necessary short-term measure to sustain viable populations of salmonids until more watersheds are restored to functional status throughout the lower Columbia River. The Crazy Johnson is one example of a critical refugia subbasin within the Grays River system. The West Fork, although currently degraded, could also provide this function for chum salmon if rehabilitation and enhancement efforts were linked with conservation initiatives in that specific subbasin.
- Support and strengthen the efforts of the Grays River Stakeholder Group, the Grays River Habitat Enhancement District (GRHED), and the Wahkiakum County in building and maintaining grass-roots community involvement and cooperation within the watershed. Continue with the restoration efforts already underway in the watershed.
- Establish an unconstrained channel migration zone (CMZ) and floodplain in the Grays River response reach upstream of State Route 4 through a program that is sensitive to landowners and the

community, and that focuses on land acquisition, conservation easements, and levee setback or removal in areas with high potential for flooding risk and avulsion, as well as areas with high potential for habitat restoration. Remove damaged structures and related man-made materials from the mainstem response reach floodplain on the Gorley property that have become isolated after the 1999 avulsion and are at risk of being washed downstream.

- Limit timber harvest on steep slopes and in areas with soil types that are most susceptible to mass-wasting and surface erosion. Current sediment loads are at least four times greater than natural levels and up to an order of magnitude greater in some areas. Sediment yield from slopes greater than 65% was found to be (on average) two to three orders of magnitude greater than on slopes less than 65% (Herrera 2005).
- Establish more ecologically meaningful riparian buffers and protect these critical areas from human disturbances such as logging and roads. Adhere to forest practices that minimize sediment production and eliminate harvest within the CMZ and riparian management zone (RMZ). This riparian conservation should be extended to intermittent, seasonal, and non-fish-bearing headwater channels because of the hydrologic and geomorphic connection these channels have with larger, downstream fish-bearing perennial streams.
- Decommission inactive forest roads and restore natural drainage patterns to reduce the potential for debris-flow initiation at road prisms and to reduce surface erosion directly from road surfaces. Approximately 80% of all mass-wasting sites in the upper watershed are associated with forest roads. Forest roads are also the largest source of surface runoff sediment delivery to stream channels. Ensure all active logging roads are maintained in accordance with current forest-practice standards.
- Construct ELJ structures to create stable forested islands and complex instream habitat within the floodplain and CMZ of the Grays River response reach. These ELJ installations should encourage the development of a complex multi-channel morphology that is natural for that segment of the river and provides the optimal mix of spawning and rearing habitat. These ELJ structures also will provide hydraulic roughness and sediment storage and sorting.
- Construct channel-spanning LWD and/or ELJ structures throughout the upper watershed at sites where structures would store sediment, moderate sediment routing, and improve local aquatic habitat.
- Implement active reforestation (conifers) of floodplain and riparian forests throughout the watershed to restore natural, self-sustaining processes that supply and recruit functional LWD to the channel.
- Extend the period of time between timber harvest activities to coincide more with the natural disturbance regime (80–100 years or more) and attempt to mimic the spatial extent of natural forest disturbances.

6.4.4 Grays River Restoration Plan

Like many Pacific Northwest watersheds, the Grays River is in an early stage of the natural recovery process from habitat degradation caused by past land-use practices. By most accounts, it could take decades or centuries for the ecosystem to naturally recover lost habitat-forming processes. The restoration of natural watershed function, including healthy riparian forests, natural LWD recruitment, and floodplain function will almost certainly take many decades under the best of conditions. However, salmon recovery necessitates some interim actions be undertaken to kick-start these natural watershed processes. This is where short-term restoration (e.g., rehabilitation and enhancement) projects come into play.

This section will describe the recommended restoration actions for each subbasin within the upper Grays River watershed. A prioritized list of all potential restoration actions is shown in Table 6.1. Primary emphasis was placed on actions or projects that have the most influence on restoring natural, habitat-forming processes over the long term. Restoration actions that fall into the realm of rehabilitation or enhancement (i.e., involve human manipulation of the river) are considered secondary but may be desirable from a short-term perspective.

6.4.4.1 Acquisition and Conservation of Refugia Habitat

In general, protection of intact high-quality habitat should be given priority over habitat restoration (see Figure 6.5), as conservation of functional habitat is typically more cost-effective and has a higher probability of success than does enhancement or rehabilitation of degraded habitat (Roni et al. 2002). Therefore, an important initial step in the overall Grays River watershed restoration process is to identify and protect (conservation) areas of naturally functioning habitat that presently exist in the watershed. These high quality areas provide refuge for various life-history stages of biota, both aquatic and terrestrial. These so-called refugia are areas where natural habitat-forming processes are still functioning and the impacts of human land-use activities are minimal. These refugia serve a critical function within the watershed as source areas for recolonization of severely degraded habitat that is rehabilitated or previously inaccessible areas that have access restored. Refugia areas should be targeted for conservation easements and acquisition so as to protect this functional habitat from possible future degradation (FEMAT 1993; NRC 1996; Williams et al. 1997; Roni et al. 2002; Beechie et al. 2003).

The Crazy Johnson Creek sub-watershed contains important spawning and rearing habitat. This refugia area is used by chum salmon for spawning and by other salmonids for rearing habitat. The Crazy Johnson subbasin should be a high priority for acquisition or conservation easement. Another potential conservation area could be the lower West Fork stream-riparian corridor or, potentially, the entire West Fork subbasin. Although degraded due to land-use activities in the upper subbasin, the lower West Fork is currently a productive segment of the Grays River with respect to chum spawning. Providing protection for this segment of the river would allow the natural habitat-forming processes to recover naturally or with the help of active restoration efforts. Protection of only the lower West Fork stream-riparian corridor would be the less costly option, but there would always be a risk that hydrologic and geomorphic impacts (i.e., high flows and elevated sediment loads) could continue to degrade instream habitat. Protection (and restoration) of the entire West Fork would be quite costly but would likely have a much higher probability of success in the long term.

6.4.4.2 Floodplain and Channel Migration Zone Restoration

Often a key rehabilitation program component in large rivers is the reconnection of floodplain and other off-channel habitat to the existing channel system. The removal or breaching of dikes and levees to reconnect floodplain or off-channel wetland habitat has become a common rehabilitation technique (Williams et al. 1997).

The mainstem of the Grays River downstream of the canyon and upstream of the SR 4 highway bridge is the main response reach for the upper watershed. This reach is very sensitive to changes in sediment supply and, as such, is prone to channel avulsion activity. Establishing an unconstrained CMZ and floodplain within this response reach should be a high priority within the overall watershed restoration plan. This restoration should be accomplished through an effort that is sensitive to landowners and

the community and that focuses land acquisition and/or conservation easements and levee setback and/or removal in areas with high potential for flooding risk and avulsion as well as high potential for habitat restoration.

The series of levees and earthen dikes constructed during the 1960s should be considered for removal, provided that the floodplain and CMZ areas behind these levees can be acquired or conservation easements obtained. These levees are located on the Gorley and Schmand properties within the mainstem response reach. Removal of damaged structures and related man-made materials from the mainstem response reach floodplain on the Gorley property, which were damaged in the 1999 avulsion and have become isolated, should be included in this effort. These debris pose a potential water-quality problem and are at risk of being washed downstream where they could cause damage to habitat and property.

These two sections of the mainstem response reach are adjacent to the lower West Fork and Crazy Johnson Creek, which were previously recommended for protection. This combined area, if conserved and restored, would encompass the majority of chum spawning habitat in the Grays River watershed.

An area of somewhat lower priority for floodplain and CMZ reconnection is the lower segment of Fossil Creek between the confluence with the mainstem Grays River upstream to the bridge on Fossil Creek Road. This habitat area has some potential for use as spawning habitat by chum and other salmonids, although the high sediment load of Fossil Creek and the small channel area make this a medium-priority area.

Experience indicates that, in general, reconnection of floodplain, CMZ, and off-channel habitat generally has a high potential to be ecologically effective and has a high probability of success (Roni et al. 2002). In addition to floodplain reconnection, one of the first steps in the rehabilitation sequence should be the correction of fish-migration passage barriers. These barriers are most often road culverts that were designed and/or installed improperly or have become fish-passage barriers due to changes in contributing drainage areas (e.g., debris flows or high runoff flows). Replacement or rehabilitation of barriers can provide access to usable habitat for juvenile and adult salmonids as well as other aquatic species. Prioritization of fish-passage barriers is a common component of most watershed assessment programs (Slaney and Zaldokas 1997).

The construction of new or the reconnection of remnant side-channel areas for spawning and/or rearing is also a technique that has been used successfully in river and stream settings in the Pacific Northwest (Slaney and Zaldokas 1997). Figure 6.6 shows an example of this technique. Based on historical and current conditions, the Schmand reach is a potential location for this type of off-channel habitat construction project. Although this is one restoration option to consider, it should be viewed as an alternative to the preferred alternative of levee removal and full restoration of the CMZ. It should also be noted that an artificial spawning channel existed in the Gorley reach and was destroyed by the 1999 avulsion. Although constructed channels can be effective, they can also require maintenance. The dynamic nature of the Grays River response reach makes this type of approach to spawning and other off-channel habitat restoration more risky than a natural, self-sustaining approach.

6.4.4.3 Hillslope Stabilization of Previously Logged Steep Slopes

It is generally accepted that, based on the findings of the watershed assessment process, *hillslope stabilization* efforts must be initiated early in the rehabilitation process, to reduce sediment loads entering

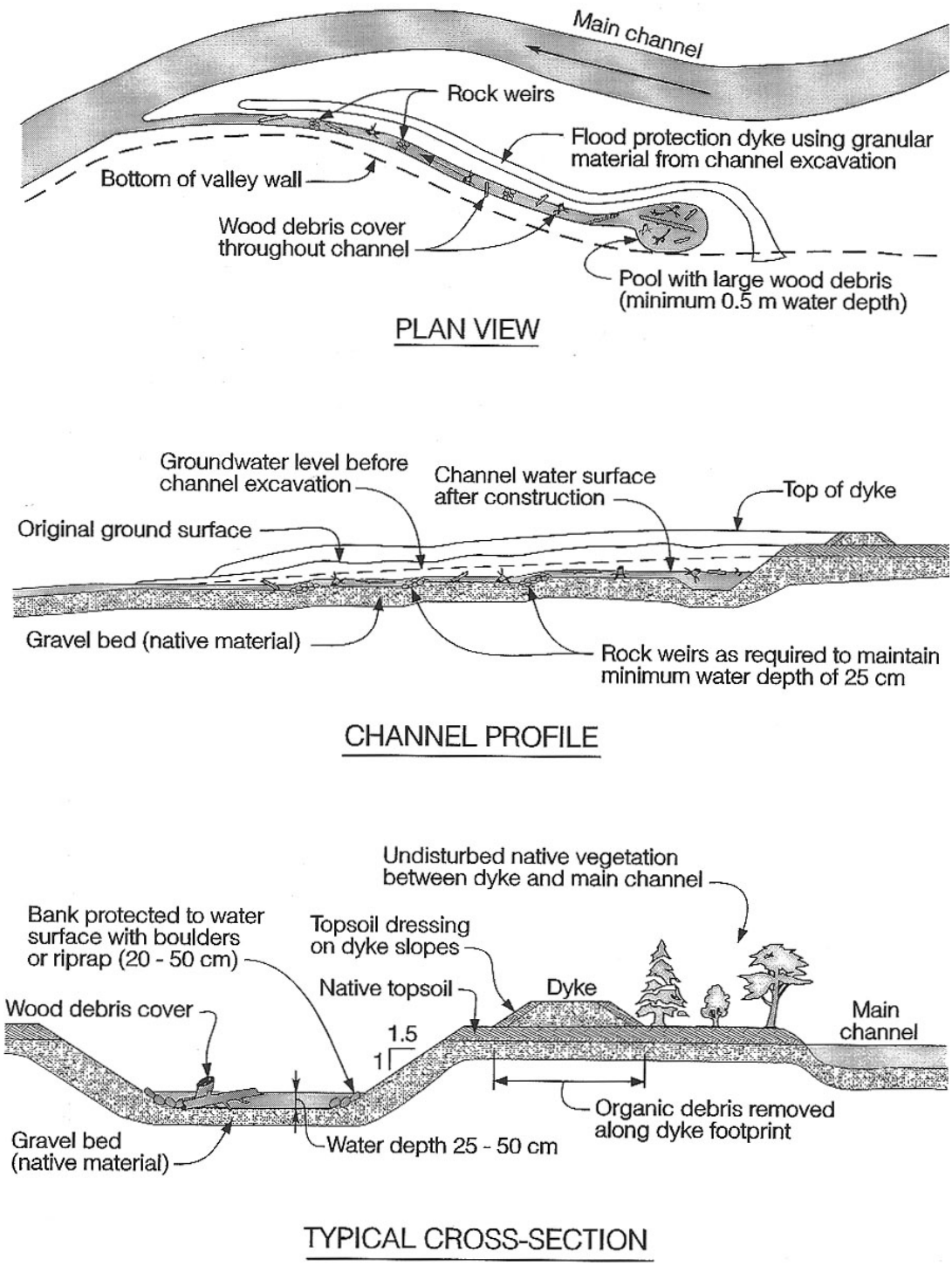


Figure 6.6. Constructed Off-Channel Habitat (Slaney and Zaldokas 1997)

the stream channel network. Hillslope stabilization measures include replanting of clear-cut slopes, surface enhancement or decommissioning of logging roads, stabilization of mass-wasting sites, and road drainage improvements. These measures should reduce or eliminate sources of sediment input into the stream network, which if not controlled could overwhelm any downstream rehabilitation projects (Slaney and Zaldokas 1997).

Logged hillslopes often experience an increase in the occurrence of mass-wasting events in the form of shallow landslides and debris flows (Swanson and Dyrness 1975; Swanston 1991; Montgomery et al. 2000; Sidle and Wu 2001). Likely reasons for this tendency have been discussed previously and include reduced soil shear strength due to loss of root strength, disturbance of soil structure due to timber harvest activities, increased soil moisture due to loss of canopy interception of precipitation, and reduced natural evapotranspiration following timber harvest activities. Steep, convergent landforms (i.e., hollows, swales, and headwater channels) tend to be more susceptible to landslides and debris flows, which can eventually deliver significant sediment loads to downstream stream channels (Montgomery et al. 2000).

Hillslope stabilization on previously logged steep slopes to reduce sediment supply and delivery to headwater channels and restore natural hillslope hydrology is one of the most critical long-term restoration actions that should be implemented in the upper Grays River watershed. Prevention of landslides and debris flows is one of the main focal areas of the Washington Forest Practices Manual (WA-DNR 2001). Recommended actions to address hillslope stabilization include the following:

- Limit timber harvest on steep slopes and in areas with soil types that are most susceptible to mass wasting and surface erosion.
- Adhere to forest practices that minimize sediment production and eliminate harvest within CMZs and RMZs.
- Strictly adhere to reforestation requirements of the forest practices manual.

6.4.4.4 Forest Road Construction and Maintenance

Construction of logging roads can greatly accelerate surface erosion rates in a watershed (Swanson and Dyrness 1975), resulting in significant loading of sediment into stream channels (Furniss et al. 1991). In addition, forest roads can significantly alter watershed hydrology (Furniss et al. 1991). Although erosion rates generally decline after completion of road construction, unpaved forest roads continually produce sediment due to surface erosion from logging-truck traffic and precipitation runoff, adding significant amounts of fine sediment to stream channels (Swanston 1991). Poorly designed, constructed, or improperly maintained forest roads on unstable slopes can also greatly increase the rate of landslides and debris flows in comparison to natural forest conditions (Swanson and Dyrness 1975; Furniss et al. 1991; Swanston 1991; Luce and Black 1999). Approximately 80% of all mass-wasting sites in the upper Grays River watershed are associated with forest roads (Herrera 2005).

Current forest management practices recommend that active forest roads be constructed and maintained such that natural surface and groundwater flow regimes are not significantly affected. This can be done by disconnecting road drainage from the natural stream network and controlling surface

runoff from roads (Furniss et al. 1997). Recommended actions to address surface erosion and hillslope stabilization from forest-road construction, operation, and maintenance include the following:

- Limit road construction on steep slopes and in areas with soil types that are most susceptible to mass wasting and surface erosion.
- Avoid road construction in proximity to stream channels except for necessary stream-crossings. Avoid constructing roads within the RMZ of a stream or wetland.
- For frequently used or heavy-haul roads, gravel should be applied to reduce surface erosion caused by timber-haul truck traffic. Roads near stream crossings also should be graveled.
- Minimize the number of stream-road crossings when constructing forest roads and ensure all road culverts are designed, installed, and maintained so as to support passage for all affected aquatic biota, especially all life stages of native fish.
- All stream crossings must be designed and installed to pass the 100-year flood event with consideration for the passage of debris (i.e., LWD).
- Grade forest roads to minimize surface runoff and erosion.
- Maintain all cut and fill slopes at a stable angle (i.e., no steeper than the angle of repose).
- Where practical and safe, design forest roads with the road surface sloped toward the outside of the road (out-sloped) such that runoff does not accumulate in uphill ditch-lines where it can concentrate and cause erosion or lead to excessive flows in headwater channels. Fill slopes must be vegetated on out-sloped roads so that sheet-flow runoff can be dissipated and infiltrated without causing hillslope erosion or instability.
- Where in-sloped roads must be used, design the drainage (ditch) system to minimize direct delivery of runoff into stream channels. Locate drainage structures upgrade from stream crossings such that runoff is directed through vegetated areas prior to discharge into a stream channel.
- In areas where hillslope instability is a problem and/or where vegetated areas are not available for runoff dissipation, diversion structures should be employed to dissipate energy and bypass runoff around erodible material and unstable slopes.
- To reduce erosion in drainage ditches, vegetate and/or armor ditches, use check-dams to dissipate flow energy, and construct sediment traps as required. Re-excavate (“clean”) ditches only when necessary. Limit the length of roadside ditches between cross-drainage structures, especially on steep forest roads.
- Use drainage relief structures and water-bars where necessary to enhance surface drainage from forest roads.
- Ensure all active logging roads are maintained in accordance with current forest practice standards (see Chapter 222-24 WAC and WA-DNR 2001).

The subbasins of greatest concern for both hillslope stabilization and forest road maintenance are listed in priority order below (Table 6.1):

- West Fork
- South Fork and Blaney Creek

- Crazy Johnson Creek
- East Fork
- North Fork
- Fossil Creek.

Decommissioning of inactive forest roads and restoration of natural drainage patterns to reduce the potential for debris-flow initiation at road prisms and to reduce surface erosion directly from road surfaces has been demonstrated to be an effective rehabilitation technique (Furniss et al. 1997; Madej 2001; Elseroad et al. 2003; Kolka and Smidt 2004). However, there is currently no common framework for prioritizing road decommissioning efforts on a regional or watershed basis (Luce et al. 2001). The decision to decommission or maintain a forest road is complex, involving many competing factors, including economics, ecosystem health, and access issues. Streamside roads that encroach on the riparian corridor (i.e., within 200 ft of a stream channel) are excellent candidates for decommissioning, as are roads that require near-continuous maintenance (Luce et al. 2001).

Decommissioning roads in areas designated as refuge habitat or areas where restoration activities will be focused is also a common strategy (Luce et al. 2001). However, based on the current body of knowledge, it appears that not all roads are equal in their impact; therefore, just reducing road density overall may not necessarily reduce the impacts of roads (e.g., sediment production and delivery, hydrologic disruptions). It appears that a more effective approach is to determine which roads are having the greatest impact on sediment loading and target those roads for decommissioning or rehabilitation (Luce et al. 2001). Road decommissioning in the West Fork and South Fork subbasins would likely have the greatest potential for sediment load reduction (Herrera 2005).

Road traffic levels also have a significant influence on surface erosion (Luce and Black 1999). Reducing traffic and/or resurfacing high-traffic roads, such as the upper Grays River mainline road, can often reduce sediment loading substantially (Reid and Dunne 1984). In addition, enhanced maintenance attention to stream-road crossings can be one of the most effective measures for reducing catastrophic sediment loading events (Montgomery 1995; Furniss et al. 1997; Madej 2001).

6.4.4.5 Riparian Corridor Restoration

Native vegetation in general, and native forest cover in particular, play a critical role in healthy watersheds (Naiman et al. 2000; NRC 2002; Naiman et al. 2005). Plant communities are dynamic. Soils, nutrients, and woody debris move from one area to another through precipitation and erosion, leaching, wind, natural and human disturbances, and a variety of other means. Plant communities in riparian areas help determine what, how much, and when materials from upland areas enter the stream ecosystem. For example, a wide, mature riparian forest will capture many soils and sediments, nutrients, and woody debris, adding richness and complexity to soil and plant communities near the water and protecting water from excessive nutrient or soil inputs (Figure 6.7). A fine balance exists between having enough and having too much of these inputs to the stream. Riparian areas, and consequently the structure, functions and processes occurring within and around the stream, are fundamentally altered when upland and riparian vegetation are removed (Naiman et al. 2000; NRC 2002; Naiman et al. 2005).

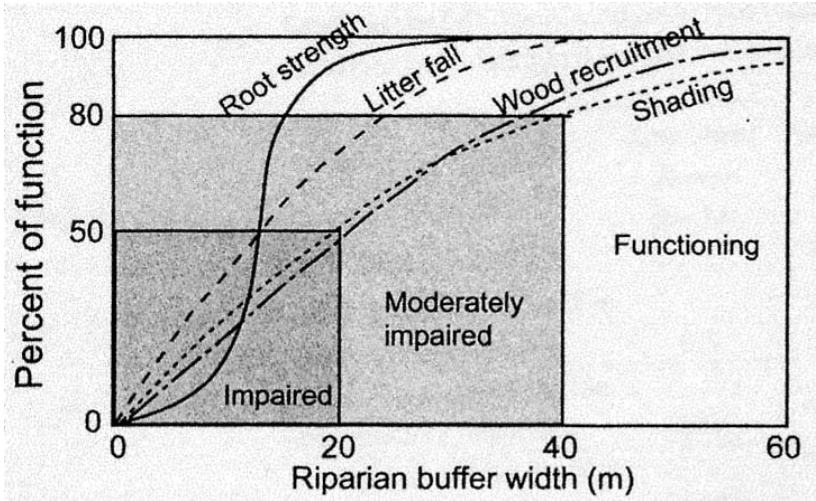


Figure 6.7. Riparian Function Impairment (FEMAT 1993; Beechie et al. 2003)

Timber harvest activities also have had some significant impacts on the riparian corridors of streams and rivers of the Pacific Northwest (Naiman et al. 2000; NRC 2002; Naiman et al. 2005). Harvest of mature conifers within the riparian zone has resulted in a decline or loss of LWD recruitment potential. Logging and road building within the riparian corridor have resulted in a loss of shade and temperature regulation in many Pacific Northwest streams. In addition, timber harvest activities have resulted in the degradation or loss of water-quality filtering capacity that natural riparian buffers afford streams and rivers (Naiman et al. 2000). Because of these cumulative impacts, active riparian rehabilitation tends to be another key component of most river-stream rehabilitation programs (Naiman et al. 2005). Riparian rehabilitation activities generally include conifer replanting or hardwood conversion projects (Slaney and Zaldokas 1997). Without active riparian rehabilitation, it will likely take decades or even centuries for some riparian functions, such as LWD recruitment (see Figure 6.8), to return to natural levels.

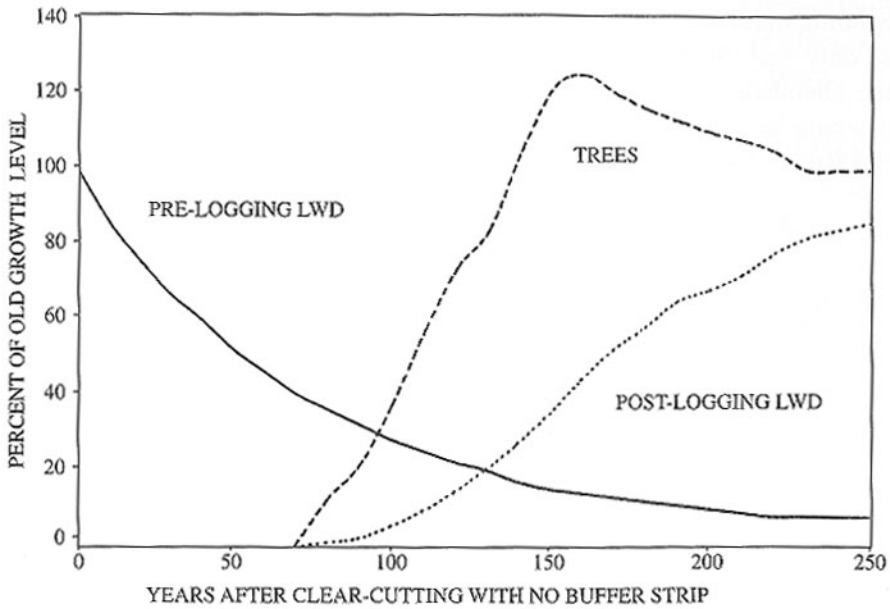


Figure 6.8. Riparian Function Recovery (Slaney and Zaldokas 1997)

As part of an overall conservation and restoration strategy for stream-riparian areas in the upper Grays River watershed, the following riparian management actions are recommended:

- Treat seasonal, non-fish-bearing stream channels the same as perennial, fish-bearing streams when determining appropriate buffer width protection criteria. Headwater channels, whether they directly support fish populations, provide habitat functions (e.g., LWD and organic matter recruitment) and are hydrologically and geomorphically (e.g., debris-flow risk and sediment delivery) connected to downstream channels that do support fish (Naiman et al. 2000; Naiman et al. 2002; Naiman et al. 2005).
- Provide increased riparian protection for floodplain and CMZ areas of the mainstem river system. The CMZ and floodplain areas found in most Pacific Northwest rivers have a very strong connectivity between the aquatic and terrestrial ecosystems that they encompass. In addition, these areas have very high habitat-forming functional value. To maintain this critical landscape function, floodplain and CMZ areas should receive the highest level of riparian protection. The goal is to reestablish a mature, fully functional floodplain forest, including forested islands and multiple channels (Montgomery and Bolton 2003; Montgomery et al. 2003; Montgomery and Abbe 2006).
- Implement an active restoration program to increase conifer density in riparian zones throughout the upper Grays River watershed. In addition to native conifers, use native cottonwoods in the mainstem response reach floodplain riparian zone.

The addition of marine-derived nutrients (MDN) also is becoming a more common rehabilitation technique. This is typically accomplished by artificially adding salmon carcasses into the stream system during fall or winter (spawning) periods. The addition of MDN in this manner has been shown to have positive effects on the instream food web, salmonid juvenile survival, and on riparian forest health (Willson and Halpuka 1995; Bilby et al. 1998; Naiman et al. 2002; Schindler et al. 2003; Wipfli et al. 2003). Consideration should be given to the addition of salmon carcasses (MDN) into streams in the upper Grays River watershed.

6.4.4.6 Mainstem Response Reach Restoration

From a hydro-geomorphic perspective, the mainstem (Gorley) response reach is a key segment of the Grays River with respect to the overall restoration strategy. Much of the sediment, LWD, and organic matter generated in the upper watershed and subsequently transported through the canyon reach is initially deposited in the response reach. As a result, this reach is the most dynamic segment of the river, with a high potential for lateral channel migration and floodplain side-channel formation. Geomorphically, the Gorley reach has been identified as the most active portion of the response reach in the Grays River system (Herrera 2005).

The mainstem response reach is also the primary area of spawning for endangered chum salmon, as well as for chinook, coho, and steelhead (WDFW 2001 and WCC 2002). This is due, in large part, to the presence of multiple, braided floodplain channels and the complexity and strength of hyporheic flow patterns that result within the floodplain of this reach. These areas are sought out by chum salmon as preferential spawning habitat. There are several spring-fed areas within the mainstem response reach that are also heavily utilized by spawning chum and other salmonids. The lower reaches of Crazy Johnson Creek and the Gorley Springs areas are the primary examples of this unique habitat feature.

The Grays River mainstem (Gorley) response reach has previously been identified by multiple studies (WCC 2002; LCFRB 2004; GRHED 2004) as a key segment of the Grays River for restoration. The Grays River subbasin plan (LCFRB 2004) identified the Gorley reach as the top chum salmon restoration priority for the entire river system. This reach was classified as Tier-1, which is a segment of the river where "…recovery measures would yield the greatest benefits towards accomplishing the biological objectives" (LCFRB 2004). The Gorley reach is located within both a high-priority (Tier-1) reach and the highest priority ("A") sub-watershed (LCFRB 2004).

As was discussed earlier, because of its location at the mouth of the Grays River canyon and the topography of the reach (it is the beginning of the relatively low-gradient lower Grays River as compared to the steeper headwater upper Grays watershed), this is the most active area of the river with regard to channel migration and floodplain dynamics. Therefore, restoration of the Gorley reach will be somewhat constrained by its position in the river system and the existing upper watershed conditions (i.e., sediment loading, lack of instream structure).

Natural restoration would likely progress without human intervention, and this is certainly a restoration option that could be selected. However, some short-term intervention may be desirable to support the immediate needs of declining salmon populations and to mitigate related flooding impacts on downstream landowners (LCFRB 2004). In short, balancing watershed restoration requiring a long-term timeframe with the more immediate need for spawning habitat stabilization for chum and flood-hazard mitigation may be the most effective course to take. Due to the importance of the Grays River chum population to the overall lower Columbia River chum ESU and the declining state of those fish runs, immediate intervention is probably warranted, even if much of the problem requires long-term solutions as outlined above (e.g., riparian forest maturation, natural LWD recruitment, and reduced sediment loading).

Currently, the sediment load from the upper watershed is significantly higher than would be present under natural (mature forest) conditions. The load has been estimated to be between four times and an order of magnitude greater than a naturally forested watershed (Herrera 2005). This is due primarily to the legacy of historical timber harvest and road-building activities in the upper watershed. Because of the lag time between sediment production (e.g., mass-wasting events, surface erosion) and the transport-deposition process, this elevated sediment load likely will continue to degrade aquatic conditions throughout the watershed for several more decades, depending on the effectiveness of the future forest management methods (Herrera 2005).

The most direct and immediate actions that can be implemented to enhance habitat quality and complexity in the mainstem response reach are the reintroduction of functional LWD and reestablishment of mature riparian forests. Returning functional LWD to the response reach would increase sediment storage, increase channel complexity and pool frequency, diffuse flood peaks, and increase channel length. Hard points within the floodplain created by stable LWD or ELJ would support the formation of floodplain islands and allow riparian forests to mature within the CMZ. These are important features of the salmon landscape within a river ecosystem (Herrera 2005).

The placement of stable LWD structures or ELJs in probable flow paths would also disperse and bifurcate flow, decreasing the erosive potential of flows while providing the habitat benefits of increased hydraulic complexity where functional wood and recruitment potential is currently lacking. These hard points also promote the formation of deep channels and pools in place of shallow, braided channels that

typically result from increased sediment loading. Intentional placement of ELJ structures also would provide predictability of locations of future flooding and channel changes, such as avulsions, thus improving flood management (Herrera 2005).

Because of historical timber harvest activities in the upper watershed, the quantity and quality of instream LWD is far below levels normally present in natural forested watersheds of the Pacific Northwest. This lack of LWD is especially significant within the mainstem response reach, where individual pieces of LWD and natural logjams would serve a critical role in trapping sediment as well as forming and maintaining instream habitat. These same functions of LWD are also lacking in the upper watershed but are magnified within the response reach due to the elevated sediment load and more frequent localized bankfull flow and flooding events that result from land-use activities in the upper watershed.

A concept plan for rehabilitation of the Grays River mainstem response reach incorporating ELJ structures, floodplain widening, and vegetation restoration is presented in Figure 6.9. The recommended restoration activities could provide for the following:

- decreased sedimentation leading to channel instability by reducing sediment input from the upper watershed
- increased length of the wetted channel in the response reach used by salmonids by increasing the number of perennial side channels
- increased pool frequency and increased rearing habitat quantity and quality
- increased hyporheic connectivity in spawning areas
- increased shade and cover by increasing the relative area of the channel beneath a forest canopy, resulting in more moderate water temperatures
- increased LWD recruitment and sediment trapping efficiency within the reach
- decreased lateral erosion and flood risks to the lower Grays River.

Based on initial field assessments in the mainstem response reach, prospective locations for ELJ installations have been identified (Figure 6.9). These are preliminary locations based on current hydraulic modeling and field surveys. Because of the dynamic nature of the response reach, an updated, detailed field assessment will be required to finalize ELJ installation positions to optimize side channel habitat creation and instream habitat complexity. Riparian revegetation should be initiated throughout the response reach, from the canyon to the SR 4 highway bridge. Floodplain restoration opportunities also are identified in Figure 6, although acquisition and/or landowner cooperation (e.g., conservation easements) still will be required before any levee removal is done. As discussed earlier, protection of Crazy Johnson Creek should be a key component of any restoration plan.

Short-term habitat benefits of this proposed plan include creation of juvenile salmonid rearing (pool) habitat, creation of side-channel salmon spawning habitat, and an increase in overall channel stability. Long-term benefits include restoration of floodplain habitat, forested islands, side channels, recruitment of future sources of LWD, and an increase in spawning and rearing habitat area.

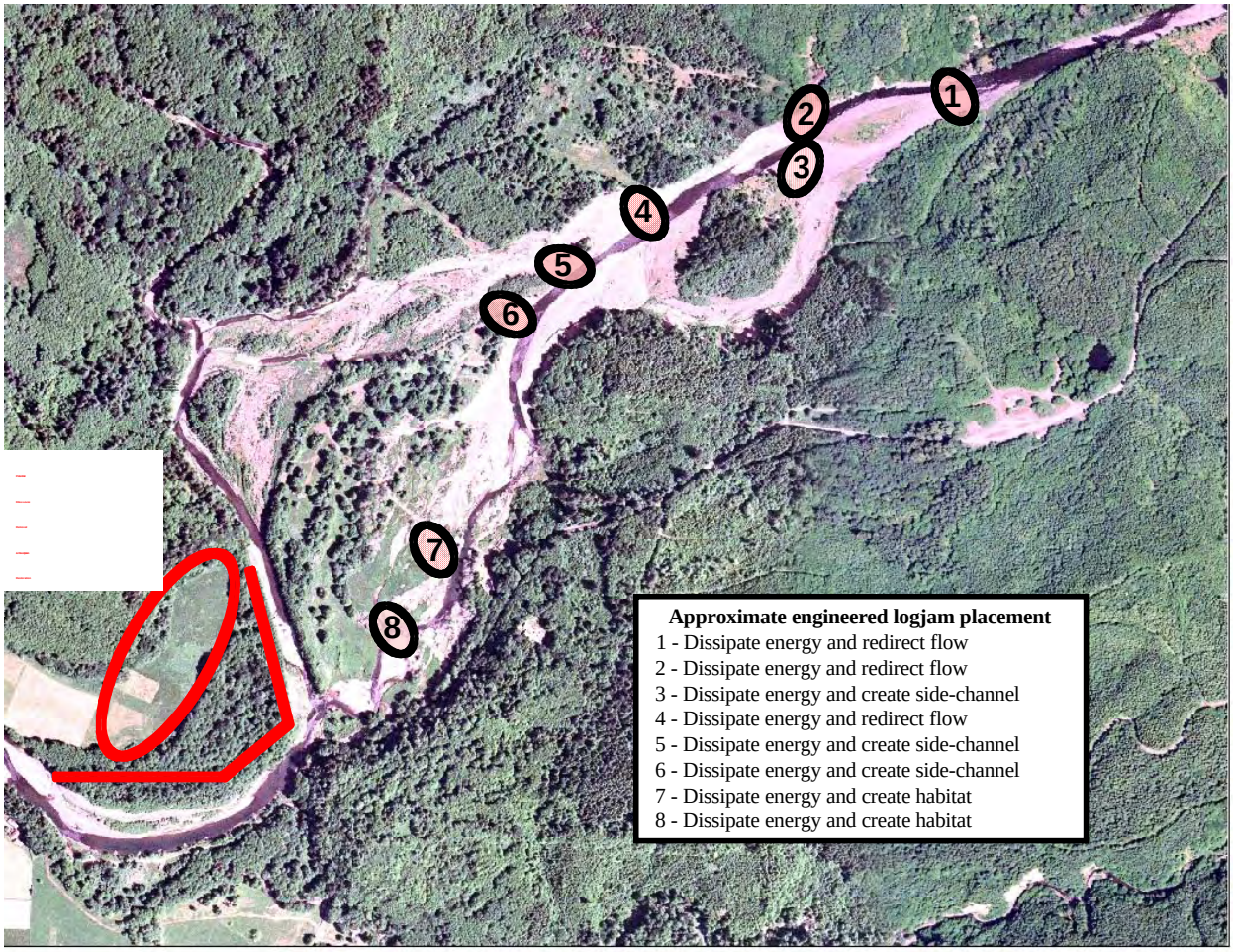


Figure 6.9. Proposed Engineered Logjam Locations in the Mainstem Response Reach of the Upper Grays River. Potential levee removal and floodplain restoration area shown in red.

The magnitude of future impacts on fish habitat caused by increased sediment yield and channel instability within the Grays River mainstem response reach will be determined by the ability of restoration efforts to counteract the destructive effects of past and ongoing land-use activities within the upper watershed. Rivers with high sediment loads can support productive fish populations if they contain abundant instream wood, which promotes pool formation, protective cover, substrate diversity, and channel migration into floodplain forests for self-sustaining LWD recruitment (Beechie et al. 2003).

This proposed option is designed to also implement actions previously identified in the *Grays River Sub-Basin Plan* (LCFRB 2004) and the WRIA-25 habitat limiting factors analysis (WCC 2002). Because complete restoration of natural watershed processes takes a very long time, this proposed project specifically addresses one of the key priorities identified in the Grays River subbasin plan (LCFRB 2004)—to address immediate risks to salmon populations with short-term habitat fixes while working toward long-term recovery goals. Moreover, this project in particular embodies a priority need articulated in the LCFRB habitat work schedule (LCFRB 2004) for this particular reach and those adjacent to it. According to the Grays River subbasin plan (LCFRB 2004), priority habitats in need of protection include salmon spawning areas and floodplain habitat in the mainstem response reach of the Grays River.

This project will also address the following limiting factors (WCC 2002) with respect to salmonid habitat:

- restoration of off-channel and side-channel spawning and rearing habitat for chum, chinook, and coho salmon and steelhead
- increased instream habitat complexity (LWD) and deposition of sediment in discrete areas of the channel
- improved seasonal stability of chum salmon spawning habitat areas in a highly dynamic and productive reach of the river
- enhancement of riparian quality to provide streambank stabilization, fine sediment filtering, and water temperature regulation.

Restoration of the mainstem response reach specifically addresses the highest-priority habitat types in need of restoration that have been identified for the Grays River watershed in this report and in previous studies (e.g., floodplain and side-channel rearing and spawning habitat). This proposed project also addresses the need for more instream habitat (LWD) complexity and improved instream sediment retention also identified in this and previous reports. Finally, the project addresses the overall need for restoration of natural riparian ecosystem structure and function. The salmonid life-cycle stages that will be affected by this project include chum, chinook, coho, and steelhead spawning, incubation (alevin), and freshwater juvenile rearing.

Several significant project uncertainties exist and must be taken into account during design, construction, and especially the monitoring phase of the proposed project. These uncertainties include the following:

- Continued timber harvest and road operations in the upper watershed likely will continue to have impacts on the project reach (excess sediment loading and lack of instream LWD) into the near future, even with the implementation of improved forest management practices (Fish and Forest requirements).
- The project reach is the most naturally dynamic segment of the mainstem Grays River and should therefore be expected to change frequently, depending on the hydrologic disturbance regime.
- There is always a risk that instream habitat structures such as ELJ structures will not function as designed due to unforeseen circumstances or will fail under extreme natural disturbance events.
- Development of additional restoration activities is dependent on landowner cooperation and stakeholder support.

6.4.4.7 Large Woody Debris Enhancement

The role of LWD as the primary structural component in rivers of the Pacific Northwest is well understood (Maser and Sedell 1994). In natural, forested watersheds, recruitment of LWD and other organic material to the stream channel occurs in conjunction with normal hillslope sediment production processes such as mass-wasting events (e.g., landslides and debris flows) and instream processes such as streambank erosion and channel migration. Woody debris deposited in streams typically forms jams or dams throughout the watershed channel network, acting as a significant grade control element limiting

incision and also trapping bed material and storing large volumes of sediment (Abbe and Montgomery 2003). These natural features can create persistent, long-term instream sediment storage sites or "nodes" within the river channel-floodplain system. Debris jams also inhibit downstream propagation of sediment pulses (Montgomery et al. 2003). Instream LWD typically creates steps in the channel profile that contribute to the characteristic riffle-pool structure that provides habitat diversity and complexity for stream fishes. Instream LWD also plays a significant role in the creation of side channels and other habitat features located on channel margins (Maser and Sedell 1994; Abbe and Montgomery 1996; Abbe and Montgomery 2003).

Debris jams and instream LWD deposited on the floodplain and in the lower reaches of river systems also contributes structure and function to the river ecosystem. LWD jams are frequently instrumental in creating the multi-channel morphology common to many low-gradient rivers. LWD jams also promote and regulate channel avulsions and floodplain sloughs. Sediment storage also continues to be a functional attribute of debris jams in the lower reaches of rivers, including estuarine areas. Very large, key pieces of LWD are especially important in larger rivers. These key pieces of LWD trap smaller debris and form jams, eventually creating forested islands within the river channel complex. It is generally accepted that these reforested floodplains can develop naturally recruited LWD jams within 50–100 years (Collins and Montgomery 2002). LWD and debris jams are also ecologically important in rivers that are characterized by a single, meandering channel. In these systems, debris jams provide habitat complexity, create and maintain off-channel habitat, and provide streambank protection (Collins et al. 2002).

ELJ structures are a common technique utilized for short-term instream LWD enhancement. These structures tend to be more flexible than rock structures and more natural to Pacific Northwest river systems such as the Grays (Abbe et al. 2003b). These ELJ structures have been effective in other Pacific Northwest rivers and tend to speed up recovery without causing long-term problems because they are natural materials and break down as the river begins to recover naturally. In contrast, hard materials such as rock can lock the channel in place and inhibit natural dynamic riverine processes (Abbe et al. 2003b).

Prior to extensive logging activities in the past century, logjams were common throughout most streams in the Pacific Northwest. These accumulations of woody material helped create stable stream channels and habitat for fish and wildlife. Only in recent years have engineers and scientists begun studying the role of logjams in river ecosystems. Mimicking how these accumulations form and function is the basis for the concept and design of ELJ structures (Abbe et al. 2003b).

ELJ structures are collections of LWD that create or redirect flow and provide stability to a downstream bar or island. ELJ installations are patterned after stable, natural logjams and can be either unanchored or anchored in place using man-made materials. ELJ structures used as a streambank-protection treatment are still considered experimental, but they are becoming increasingly popular because they integrate fish-habitat restoration with streambank protection.

Naturally occurring logjams in alluvial channels are usually formed by one or several key LWD pieces (typically very large and stable), consisting of large trees with root-wads, that stabilize and anchor other debris that is "racked" against the key members. Natural logjams typically extend above bankfull water surface and, when connected to a streambank, constitute natural streambank protection. Naturally occurring logjams may start as a single, large tree, as a large number of trees drifting together, or from an undercut, timbered streambank. Over the years, people have removed many of these naturally formed

structures for navigation, firewood, timber, and for flood-control reasons. However, natural logjams provide habitat for a wide variety of fish species during most of their life stages.

ELJ structures can be used to realign a channel or redirect flow away from a streambank to protect it from erosional forces. They are used also to increase channel roughness to reduce flow velocities and shear stress along eroding banks. ELJ structures also tend to create a hydraulic shadow, a low-velocity zone for some distance downstream that allows sediment to settle out and stabilize. After an ELJ has been installed along an eroding bank, the streambank downstream of the ELJ becomes a deposition zone rather than an erosion zone. The deposition zone tends to become vegetated and continues to expand over time. Prior to designing and constructing an ELJ structure as a streambank-protection technique, it is important to understand the existing physical characteristics and geomorphic processes present at a potential project site and along the reach. ELJ structures are best applied on long, uniform bends in alluvial channels. They are appropriate also when the mechanism of failure is toe erosion because they provide roughness and redirect erosive flows away from an eroding streambank. When applied along a bend, they are apt to grow significantly as they recruit wood, so changes to the opposite streambank should be expected (Abbe et al. 2003b).

ELJ structures also are useful in degrading channels for capturing and storing sediment and debris. ELJ structures can slow the rate of erosion in an equilibrium channel that is migrating laterally or where there is potential for a chute cutoff, although they still allow for gradual meander migration. LWD jams often occur naturally at the inlet of many side channels. Similarly, ELJ structures can be assembled at the inlet of preexisting or constructed side channels to regulate the amount of flood flow entering the side channel or to stimulate hyporheic flows. This protects the banks in the side channel, prevents the side channel from capturing the main channel, and protects existing spawning and rearing habitat in the side channel (Abbe et al. 2003b).

ELJ structures may be appropriate also when the mechanism of failure is excessive streambed scour. They should be placed upstream from the scour hole to redirect flow away from the obstacle that is creating the scour or to dissipate some of the energy that is causing the scour. They should not be placed directly in a scour hole. In tight-radius bends or other constricted reaches, they may not be very effective, and their application can further exacerbate existing erosion problems or move them upstream. Care in sizing and spacing ELJ structures is crucial to avoid creating a channel constriction (Abbe et al. 2003b).

In aggrading channels, ELJ structures may be appropriate, depending upon the severity of aggradation. They can be effective strategically if placed in an aggrading channel where they can collect and store sediment. Their presence in such circumstances will better define the low-flow channel. ELJ structures also recruit LWD, which reduces the likelihood of the jam becoming buried and ineffective over time. When a channel has been disturbed and is carrying a high bedload, ELJ structures can be constructed in upstream reaches to stabilize sediment movement. Over the long term, ELJ structures usually reduce aggradation and erosion in the downstream reach. ELJ structures can be placed either at the bank or in mid channel (Abbe et al. 2003b).

In aggrading channels, ELJ structures can create excellent fish habitat by developing deep scour pools and associated tail-out spawning areas as well as complex cover. The structural complexity and hydraulic diversity associated with ELJ structures provide ideal habitat for a variety of life stages and species of fish. For these reasons, ELJ structures receive high marks as a habitat-restoration and mitigation tool.

Successful restoration projects need to take advantage of the ability of the river to restore itself over time, applying active recovery measures to speed up the process. Some restoration structures are prone to failure, particularly in large systems. Rock structures are really not a natural feature in most Pacific Northwest river systems, and when they fail there can be negative consequences within the river environment. ELJ structures too can fail, but even when this occurs, in most cases the individual LWD elements still can provide channel roughness and habitat features missing from the river. In short, ELJ structures tend to provide value to the river even if displaced from installation locations. ELJ structures also offer a distinct advantage over most rock structures such as barbs and groins. As scour holes develop adjacent to the ELJ, the interlocking nature of ELJ structures allows them to deform and settle, effectively retaining their structural integrity. For these reasons, ELJ structures are more likely to provide value to the river system and less likely to fail than rock structures (Abbe et al. 2003b).

Still, ELJ structures can pose inherent risks to downstream infrastructure and human stream users. Careful, well-calculated design and positioning of ELJ structures can minimize all of these risks (Abbe et al. 2003b). These risks include

- safety hazards caused by the logjams or the cables that anchor them (this risk can be somewhat reduced by placing warning signs upstream from the log jams to alert boaters)
- blockage of culverts or bridge openings by LWD that has been dislodged from ELJ structures upstream
- unanticipated erosion across the channel or to the adjacent streambank from an ELJ structure
- increased channel roughness and constriction, and/or increased flood stage on a local scale.

Increasing roughness throughout the chum spawning area of the Grays River response reach is a critical need if the mainstem channel downstream from the 1999 (Gorley) avulsion is to be stabilized. Riparian revegetation of the entire floodplain also is needed to assist in the long-term stabilization of the response reach (i.e., provide self-sustaining LWD recruitment). This critical chum spawning area is immediately downstream of a high-gradient narrow canyon and is (and always will be) a highly dynamic depositional area. Whatever restoration option(s) selected must be sensitive to this dynamic and changing environment.

Because of the impacts of human activities in the watersheds of the region, the natural role of LWD as the principal structural component in forested streams and its functional role in creating habitat complexity and diversity have been almost eliminated during the last century. Human activities in the watershed, such as timber harvest, have transformed complex anastomosing channel systems with abundant LWD and diverse, complex habitat (Harwood and Brown 1993) into unstable single-thread channels with little complexity and habitat diversity (Figures 6.10 and 6.11). As a consequence, the replacement of instream LWD, the installation of ELJ structures, and streambank stabilization are often large components of stream or river rehabilitation efforts (Abbe and Montgomery 1996; Abbe et al. 1997, 2003b).

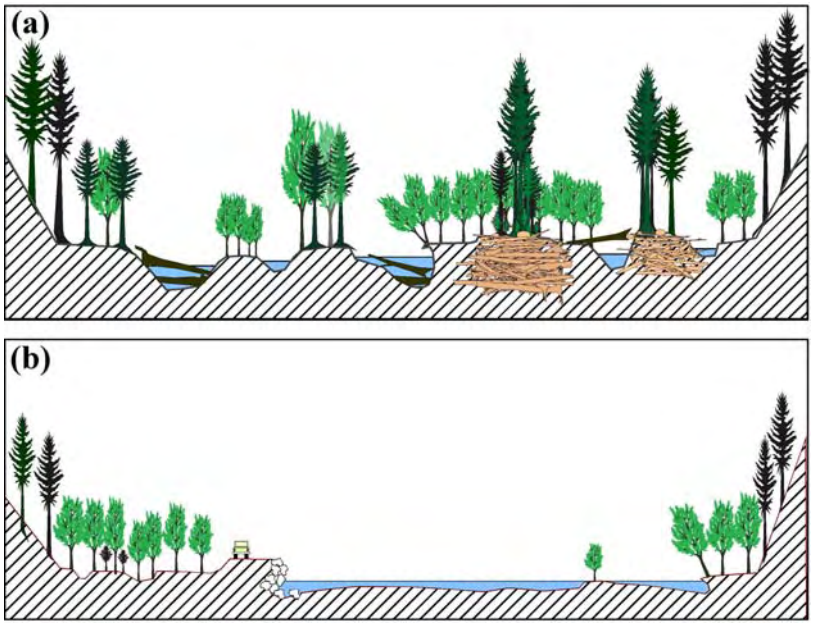


Figure 6.10. Comparison of Natural and Degraded River Systems. (A) Natural anastomosing river system with a mature riparian forest, abundant instream large woody debris (LWD), and a complex mosaic of channels. (B) Degradation of forest river systems due to direct (e.g., channelization and LWD removal) and indirect (e.g., removal of riparian vegetation and an increase in sediment supply due to upland land-use activities) human impacts (Abbe et al. 2003b).

The persistence of LWD is of fundamental ecological importance because of its influence on channel dynamics and the subsequent succession of riparian vegetation (Fetherston et al. 1995; Abbe and Montgomery 1996; Rot et al. 2000). LWD jams often control local channel hydraulics and provide refugia for salmonids against flood scour. The deposition of very large or "key" pieces of LWD, usually with intact root wads, often initiates the formation of stable LWD debris-jams that alter the local hydraulics and thereby control the spatial pattern of scour and deposition, as well as creating sites for riparian vegetation initiation. Even though LWD accumulations may appear to form at random, there is often a distinctive structural pattern (Naiman et al. 2000). Individual jams may be remarkably stable over the long term, allowing mature riparian forest patches to form and persist within an environment that is characterized by rapid channel migration and frequent disturbance.

The addition of individual pieces or clusters of LWD is also a common rehabilitation technique, especially in small streams or river tributaries where flows are not as great as found in the mainstem or rivers where ELJ installations are often the more stable LWD rehabilitation tool (Slaney and Zaldokas 1997). Salmon, especially juveniles, often respond favorably to LWD supplementation projects (Roni et al. 2002).

(a)

(b)

(c)

Figure 6.11. A Typical Engineered Logjam Project. (a) Cispus River site before ELJ installation; (b) the same Cispus River site after ELJ installation; (c) aerial view of Cispus River showing four recently installed ELJs (Abbe et al. 2003b).

Recommended locations for LWD enhancement and ELJ installations in the upper Grays River watershed include the following sites (in priority order):

- mainstem response reach
- Lower West Fork
- Lower South Fork
- Lower Fossil Creek
- Upper West Fork
- Blaney Creek
- Upper South Fork
- Upper Fossil Creek
- East Fork
- North Fork.

6.4.4.8 Restoration Monitoring

An effective monitoring program is an essential aspect of any restoration project. Restoration projects are often expensive, large-scale, and long-term. Lack of effective monitoring has limited our understanding of aquatic ecosystem response to restoration (NRC 1992), and the scale and cost of these projects makes effective monitoring essential to evaluate system response (Bryant 1995). A monitoring program incorporated from the beginning into any restoration project will allow for adaptive management and mid-course corrections throughout the life of a project, as well as provide essential guidance for improvement of future restoration projects. A well-designed monitoring program is essential to determine the effectiveness of restoration projects as well as allow for adaptive management as additional restoration activities, changes, and improvements are identified for the Grays River watershed.

Performance standards for restoration activities should be based on comparison of recovering systems to a desired distribution of natural habitat (reference) conditions (median, range, variance). We have included quantifiable attributes of chum salmon spawning habitat for which change in response to restoration activities can be directly compared to ideal or reference conditions based on literature values for healthy stream habitat and chum salmon spawning habitat (velocity, depth, substrate). Monitoring will therefore include tracking changes in essential habitat characteristics toward reference conditions following restoration activities. This approach enables evaluation of the efficacy of proposed restoration activities.

The ISAB (2003) recommends that intensive watershed monitoring be conducted at select locations, including large spatial and temporal scales when possible. Our approach is consistent with and will contribute to the Intensive Watershed Monitoring (IWM) approach to recovery monitoring recommended by the ISAB (2003). The IWM is characterized by intense focus on habitat parameters and cause-effect relationships in relatively few watersheds. Data gathered in the upper Grays River watershed would add considerable value to this database.

Proposed restoration monitoring will incorporate existing data collected for the Grays River watershed assessment project during 2004–2006. We propose to evaluate the effects of restoration actions on 1) spawning habitat availability and characteristics including depth, velocity, substrate, and bedform, 2) abundance and distribution of spawning chum salmon, and 3) geomorphic and channel flow

characteristics associated with quality fish habitat, including embeddedness, pool frequency, maximum pool depth, and bed stability. We propose to monitor restoration activities annually through 2020.

In addition to detailed and tailored monitoring of individual restoration projects (i.e., ELJ or LWD installations), the following specific monitoring activities are recommended:

- Continue salmon and steelhead spawning (WDFW) surveys.
- Implement smolt-trapping on the lower Grays River to evaluate spawning success and productivity of chum and other salmonid species.
- Conduct periodic habitat assessment surveys in selected subbasins of the upper Grays River to evaluate the long-term cumulative effects of revised forest management practices. Surveys should focus on riparian conditions, instream LWD, spawning habitat quality (i.e., fine sediment levels), and rearing habitat.
- Install continuous temperature monitors and data loggers in all major tributaries of the upper Grays River, including selected headwater streams. Data from these stations can be used to track the effectiveness of riparian restoration efforts and monitor for trends in water temperature as related to upstream forest management activities.
- Monitor turbidity in all major tributary channels, including selected headwater streams to assess the effectiveness of hillslope stabilization and forest road maintenance activities. Use stand-alone turbidity monitors and data loggers for this effort.
- Conduct a periodic watershed assessment of all subbasins in the upper Grays River watershed. The assessment should focus on landscape-scale measures of watershed condition, including forest cover characteristics, road density, road surface erosion, stream–road crossings, landslides and mass-wasting sites, and riparian corridor buffer widths. These data can be compared to the same data from this study to evaluate trends and identify problem areas.

Table 6.1. Upper Grays River Watershed Priority Restoration Summary

Project Description	Project Location	Project Type	Priority	Timeframe
Protection of spawning and rearing refugia habitat	Crazy Johnson Creek	Acquisition and Conservation	High	Long-term
Protection of spawning and rearing refugia habitat	Lower West Fork Mainstem	Acquisition and Conservation	High	Long-term
Protection of spawning and rearing refugia habitat	West Fork Sub-Watershed	Acquisition and Conservation	Low	Long-term
Restore floodplain and channel migration zone connectivity in the mainstem Grays River (includes levee removal and/or dike breaching)	Gorley Reach	Instream Watershed Process Rehabilitation	High	Long-term
Restore floodplain and channel migration zone connectivity in the mainstem Grays River (includes levee removal and/or dike breaching)	Schmand Reach	Instream Watershed Process Rehabilitation	High	Long-term
Restore floodplain and channel migration zone connectivity in Grays River tributaries (includes levee removal and/or dike breaching)	Lower West Fork	Instream Watershed Process Rehabilitation	High	Long-term
Restore floodplain and channel migration zone connectivity in Grays River tributaries (includes levee removal and/or dike breaching)	Lower Fossil Creek	Instream Watershed Process Rehabilitation	Medium	Long-term
Hillslope stabilization on previously logged steep slopes (to reduce sediment supply and delivery to headwater channels and restore natural hillslope hydrology)	West Fork Subbasin	Upland Watershed Process Rehabilitation	High	Long-term
Hillslope stabilization on previously logged steep slopes (to reduce sediment supply and delivery to headwater channels and restore natural hillslope hydrology)	South Fork and Blaney Creek Subbasin	Upland Watershed Process Rehabilitation	High	Long-term
Hillslope stabilization on previously logged steep slopes (to reduce sediment supply and delivery to headwater channels and restore natural hillslope hydrology)	East Fork Subbasin	Upland Watershed Process Rehabilitation	Medium	Long-term

Table 6.1. (contd)

Project Description	Project Location	Project Type	Priority	Timeframe
Hillslope stabilization on previously logged steep slopes (to reduce sediment supply and delivery to headwater channels and restore natural hillslope hydrology)	North Fork Subbasin	Upland Watershed Process Rehabilitation	Medium	Long-term
Hillslope stabilization on previously logged steep slopes (to reduce sediment supply and delivery to headwater channels and restore natural hillslope hydrology)	Crazy Johnson Creek Subbasin	Upland Watershed Process Rehabilitation	High	Long-term
Enhanced forest-road maintenance and decommissioning (to reduce sediment supply and delivery to headwater channels and restore natural hillslope sediment-source loading)	West Fork Subbasin	Upland Watershed Process Rehabilitation	High	Long-term
Enhanced forest-road maintenance and decommissioning (to reduce sediment supply and delivery to headwater channels and restore natural hillslope sediment-source loading)	South Fork and Blaney Creek Subbasin	Upland Watershed Process Rehabilitation	High	Long-term
Enhanced forest-road maintenance and decommissioning (to reduce sediment supply and delivery to headwater channels and restore natural hillslope sediment-source loading)	East Fork Subbasin	Upland Watershed Process Rehabilitation	Medium	Long-term
Enhanced forest-road maintenance and decommissioning (to reduce sediment supply and delivery to headwater channels and restore natural hillslope sediment-source loading)	North Fork Subbasin	Upland Watershed Process Rehabilitation	Medium	Long-term
Enhanced forest-road maintenance and decommissioning (to reduce sediment supply and delivery to headwater channels and restore natural hillslope sediment-source loading)	Crazy Johnson Creek Subbasin	Upland Watershed Process Rehabilitation	High	Long-term
Riparian corridor reforestation	Gorley Reach	Riparian Corridor Process Rehabilitation	High	Long-term

Table 6.1. (contd)

Project Description	Project Location	Project Type	Priority	Timeframe
Riparian corridor reforestation	Schmand Reach	Riparian Corridor Process Rehabilitation	High	Long-term
Riparian corridor reforestation	SR 4 Reach	Riparian Corridor Process Rehabilitation	Medium	Long-term
Riparian corridor reforestation	Lower West Fork	Riparian Corridor Process Rehabilitation	High	Long-term
Riparian corridor reforestation	Upper West Fork	Riparian Corridor Process Rehabilitation	Medium	Long-term
Riparian corridor reforestation	South Fork	Riparian Corridor Process Rehabilitation	High	Long-term
Riparian corridor reforestation	East Fork	Riparian Corridor Process Rehabilitation	Medium	Long-term
Riparian corridor reforestation	North Fork	Riparian Corridor Process Rehabilitation	Medium	Long-term
Riparian corridor reforestation	Crazy Johnson Creek	Riparian Corridor Process Rehabilitation	High	Long-term
Riparian corridor reforestation	Lower Fossil Creek	Riparian Corridor Process Rehabilitation	High	Long-term
Riparian corridor reforestation	Upper Fossil Creek	Riparian Corridor Process Rehabilitation	High	Long-term
Riparian corridor reforestation	Blaney Creek	Riparian Corridor Process Rehabilitation	Medium	Long-term
Enhancement of instream structure and habitat complexity - includes installation of engineered logjams and large woody debris additions	Gorley Reach	Instream Watershed Process Rehabilitation	High	Short-term
Enhancement of instream structure and habitat complexity - includes installation of engineered logjams and large woody debris additions	Schmand Reach	Instream Watershed Process Rehabilitation	High	Short-term
Enhancement of instream structure and habitat complexity - includes installation of engineered logjams and large woody debris additions	SR 4 Reach	Instream Watershed Process Rehabilitation	Low	Short-term
Enhancement of instream structure and habitat complexity - includes installation of engineered logjams and large woody debris additions	Lower West Fork	Instream Watershed Process Rehabilitation	High	Short-term
Enhancement of instream structure and habitat complexity - includes installation of engineered logjams and large woody debris additions	Upper West Fork	Instream Watershed Process Rehabilitation	Medium	Short-term

Table 6.1. (contd)

Project Description	Project Location	Project Type	Priority	Timeframe
Enhancement of instream structure and habitat complexity - includes installation of engineered logjams and large woody debris additions	South Fork	Instream Watershed Process Rehabilitation	Medium	Short-term
Enhancement of instream structure and habitat complexity - includes installation of engineered logjams and large woody debris additions	East Fork	Instream Watershed Process Rehabilitation	Medium	Short-term
Enhancement of instream structure and habitat complexity - includes installation of engineered logjams and large woody debris additions	North Fork	Instream Watershed Process Rehabilitation	Medium	Short-term
Enhancement of instream structure and habitat complexity - includes installation of engineered logjams and large woody debris additions	Crazy Johnson Creek	Instream Watershed Process Rehabilitation	Low	Short-term
Enhancement of instream structure and habitat complexity - includes installation of engineered logjams and large woody debris additions	Lower Fossil Creek	Instream Watershed Process Rehabilitation	High	Short-term
Enhancement of instream structure and habitat complexity - includes installation of engineered logjams and large woody debris additions	Upper Fossil Creek	Instream Watershed Process Rehabilitation	Medium	Short-term
Enhancement of instream structure and habitat complexity - includes installation of engineered logjams and large woody debris additions	Blaney Creek	Instream Watershed Process Rehabilitation	Medium	Short-term
Fish-Passage Barrier Removal	Alder Creek	Access Restoration	Medium	Short-term
Fish-Passage Barrier Removal	Mitchell Creek	Access Restoration	Medium	Short-term
Fish-Passage Barrier Removal	Unnamed Upper Mainstem Tributary (7200 mainline road at 7210 and 7220 intersection)	Access Restoration	Medium	Short-term

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Appendix A

History of the Grays River Watershed

Appendix A

History of the Grays River Watershed

The history of human activity in the Grays River Valley is important to understand as a background to this watershed assessment study. Human land-use has had an influence on current conditions on multiple spatial scales over time. This appendix provides the historical context for the analysis of current conditions described in the body of this report.

A.1 Settlement History

The Grays River watershed is located in an area of the lower Columbia River region originally the home of the Wahkiakum band of the Chinook tribe of people from which the county gets its name (Martin 1997). Two main types of settlements existed within the watershed during aboriginal times—permanent villages and temporary (fishing and hunting) camps. The latter were inhabited seasonally when weather and resources (i.e., salmon) dictated. Lewis and Clark estimated the population to be no more than 200 (Martin 1997). These native people had adapted to a river-based existence.

The first written record of European exploration of the area was from an American army captain, Robert Gray, the namesake of the Grays River and Grays Bay. He arrived in spring 1792 aboard the *Columbia Rediviva* (Martin 1997). The English explorer Captain George Vancouver also explored the lower Columbia River that same year. In 1805 and 1806, Lewis and Clark passed through the area, recording that the Wahkiakum region was "foggy, wet, and mostly disagreeable" (Martin 1997). From these early records and on through the present day, it is clear that water more than any other influence has shaped the character and history of the area.

It was not until 1844, when John Couch of Portland established a trading post at Skamokawa, that the area began to be settled (Pyle 2001). In 1853, the Washington Territory was created. In 1868, the settlement of Grays River was established, and the Grays River Post Office was built in 1872. The first school opened in Grays River in 1873. In 1901, the Grays River Grange was formed, and in 1905, the Grange Hall was built in Grays River. The Grays River covered bridge was constructed also in 1905 (Martin 1997).

Initially, it was not logging but farming that brought European settlers to the Grays River Valley. Even in a river valley dominated by huge trees, wetland swamps, and rain-driven floods, the availability of potential farmland was inducement enough for the European immigrants who settled in Grays River. Settlement took hold beginning in the 1860s. By the early 1870s, Scandinavian immigrants dominated the homesteaders. The early Grays River settlers also relied on salmon fishing for both sustenance and livelihood. Historically, salmon runs on the Grays River, especially chum, were reported to be huge (Martin 1997).

Early farms in the Grays River Valley were concentrated in the lower river floodplain, mainly around the settlements of Grays River and Rosburg (see Figures A.1 and A.2). In those days, land clearing was a monumental task, sometimes taking 20 to 30 years to completely clear a parcel of trees and drain wetland areas enough to make farming possible (Martin 1997). Flooding or freshets were common and could destroy the work of many years in a matter of hours. Fire was often the most efficient method of clearing

Figure A.1. Farming in the Grays River Valley, Circa 1906 (Martin 1997)

Figure A.2. Ahlberg Farm, Circa 1910 (Appelo Archive Center 2005)

the land of stumps after trees were felled. The practice of “char-pitting” was common (Martin 1997). This process consisted of boring holes in tree stumps (mostly spruce and cedar in the lower river valley) and repeatedly burning the stumps to get rid of them. According to Marie Fauver, local resident interviewed in 2005, these fires often burned for months before consuming the massive stumps that dotted the landscape of the lower valley.

During the early years of settlement in the Grays River area, the river was the main transportation route for people and materials coming into the valley, as well as for goods (i.e., timber and agricultural products) and people leaving the valley. The river was routinely dredged to maintain a navigation channel upstream to the town of Grays River (Appelo Archive Center 2005) and ferry boats ran between Grays River and other towns along the lower Columbia River system (see Figures A.3 through A.5).

Historically, flooding has been a part of life in the Grays River Valley from the early days of settlement until the present. The farms and homes of the lower Grays River area and the town of Grays River itself often experienced flooding during the winter rainy season (see Figures A.6 and A.7). Damage was sometimes severe, but the people of the Grays River Valley learned to adapt to the dynamic environment of the river (Appelo Archive Center 2005).

Dikes and levees were constructed to reduce the flooding of farms and homes. Local organizations were also created to oversee construction and maintenance of these levee and dike systems. In addition, according to valley resident Robert Larson (interviewed in 2005), individual property owners sometimes constructed dikes and levees in an attempt to protect their land from the flooding and freshets that were a part of living in the river floodplain area.

Figure A.3. Grays River Landing, Circa 1910 (Appelo Archive Center 2005)

Figure A.4. Dredge on Grays River, Circa 1910 (Appelo Archive Center 2005)

Figure A.5. Grays River Ferry Boats, Circa 1910 (Appelo Archive Center 2005)

Figure A.6. Grays River Flooding, Circa 1910 (Appelo Archive Center 2005)

Figure A.7. Grays River Flooding, Circa 1910 (Appelo Archive Center 2005)

A.2 Logging History

Although available farmland and abundant salmon runs were the main reasons for settlement in the lower Columbia River region, logging became a way of life for many. Initially, logging activity was focused on land-clearing for farming, but soon timber harvest became a full-time occupation for many. Carlton Appelo, local historian and a descendent of one of the local pioneers, remarked in 2005 "It took the perseverance and temperament of a Finn to clear the land of the thick, ancient forests and build a home in this area."

The transition between land-clearing for farming or grazing and commercial logging is not distinct. These activities likely went on in parallel for many years during the pioneer period. It is likely that logging commenced in the lower Grays River Valley in the 1870s, at first concentrated within the estuary and tidally influenced shorelines of the river. Early loggers used mostly hand tools and oxen to skid the logs to tidewater where they were rafted and towed to nearby sawmills. These early loggers, called bull-team loggers, consisted of a team of two to eight yoke oxen and a crew of as many hands. Tom Foss of Grays River was one of these early bull-team loggers (Martin 1997).

As with other rivers in the Willapa Hills, by the 1880s and 1890s loggers had proceeded up the Grays above tidewater, relying on seasonal freshets to sluice their logs downriver. The use of splash dams was a common practice. In general, these early logging practices had a relatively minor impact on the environment, as only selected trees were harvested and soil disturbance in removing them was minimal, although splash-damming could be quite destructive to the river channel (B. Penttila, unpublished[(a)]).

With the introduction of steam power in the late 1890s, the scale of logging began to change and loggers began harvesting timber throughout the Grays River on hillsides previously inaccessible to oxen (B. Penttila, unpublished). The technology used in logging also changed dramatically with the introduction of steam power. Steam-powered donkeys were a common feature of logging operations from 1899 through the early 1900s (Figure A.8). These steam engines took the place of oxen, pulling logs on skid roads to the nearest river access or railroad line. Steam donkeys were mounted on log sleds so that they could be moved as required (Martin 1997).

One of the best-known names in logging history was Simon Benson. He brought the first logging train to the lower Columbia in 1892 in Cathlamet (Martin 1997). The first logging railroad in Grays River was built along the south bank of the Grays River from the head of tidewater to Klints Creek by Saldern Logging Company in 1899 (B. Penttila, unpublished). From 1899 until 1908, the Saldern logging camp was located just downstream of the present State Route 4 (SR 4) highway bridge along Klints Creek (Figure A.9). After 1908, the original Saldern logging railroad passed through several corporate hands and was extended many miles before being abandoned by the Knappton Mills & Timber Company in 1924, from which KM Mountain supposedly takes its name. Much of SR 4 on the west side of KM Mountain follows this old logging railroad grade (B. Penttila, unpublished). Another company,

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(a) Bryan Penttila, *After the Old-Growth: Social and Ecological Changes in the Willapa Hills, Washington.* Master of Arts thesis (Draft), Washington State University, Pullman, Washington.

Figure A.8. Logging Steam Donkey in the Grays River Watershed

Figure A.9. Grays River Logging Camp and Railroad, Circa 1910

H.B. & A., operated a rail line along Hull Creek from the town of Grays River to the ridges below the Hull Creek lookout from around 1908 until 1923 (B. Penttila, unpublished). During this period, there were also a number of smaller railroad logging ventures within the Grays River watershed. These included Columbia River Log and Timber Company that operated between 1912 and 1918, Multnomah Lumber Company (1917-1921), North Bank Lumber Company (1913-1919), and Malone Creek Logging Company (1929-1931).

Around the turn of the century, Willamette Pulp & Paper Company also logged portions of the Grays River Valley and used the river to transport logs to the Columbia for shipment to Astoria, Oregon (B. Penttila, unpublished). Several other logging operations existed in the Grays River watershed around this period. These include Merserve, Black Brothers, Johnson, Palmer Brothers, and several other local timber outfits. In 1903, Meserve and Palmer built a wagon road about 10 miles in length to reach their camp on the North Fork of the Grays River. This was the first logging venture in the headwater portion of the upper Grays River watershed (B. Penttila, unpublished).

A majority of the timber harvested from the Grays River watershed during the turn of the century was moved to the Columbia by floating the logs downstream. In 1902, it was estimated that over 2 million logs (approximately 2.5 million board-feet) came out of the upper Grays River on the first freshet (B. Penttila, unpublished). Splash-dam operations (Figure A.10) were also common in the Grays River watershed during the early 1900s (Wendler and Deschamps 1955). There were at least five splash dams on Grays River and its tributaries: two on the South Fork operated by the Willamette Pulp & Paper Company, the Meserve splash dam on the Grays mainstem at the mouth of the canyon (the splash-dam abutments can still be seen), the Holden dam on Hull Creek, and another small splash dam on Fall Creek, a tributary of Hull Creek (Wendler and Deschamps 1955). How long each dam operated is unclear, although they typically were abandoned when logging ceased. The exact time of removal of the Meserve and Holden dams is unknown, but the two dams on the South Fork were removed by salvage loggers in the 1950s (B. Penttila, unpublished).

In 1902, it was reported that a massive jam of logs had formed at the falls within the Grays River canyon. These logs were from the camps of the Willamette Pulp & Paper Company and Black Brothers. This logjam subsequently broke during a freshet, and about 2 million board-feet of timber were washed downstream. In 1904, the Willamette Pulp & Paper Company constructed a splash dam near Blaney Creek, four miles upstream of its first dam on the South Fork. This was considered an ideal location for a dam, as the water could be raised and backed for a long distance. It was reported that there were 8 million board-feet of logs in the stream, which the two dams were expected to splash to tidewater (B. Penttila, unpublished). The lower Grays River Valley was almost completely cleared of native trees for settlement and agriculture by 1905 when full-scale commercial timber harvest began in the upper watershed (Scott 2001).

In 1905, Meserve Logging Company was logging on the upper mainstem Grays River. The company built a splash dam to drive its logs 5.5 miles downstream to the mouth of South Fork, where the Willamette Pulp & Paper Company had its two dams. The Meserve Company had about 25 million board-feet of timber behind its dam. The two companies coordinated their splash-dam releases to move timber downstream (B. Penttila, unpublished). Figure A.11 shows the land cover in the Grays River watershed prior to 1905.

Figure A.10. Splash Dam on the South Fork of the Grays River, Circa 1900

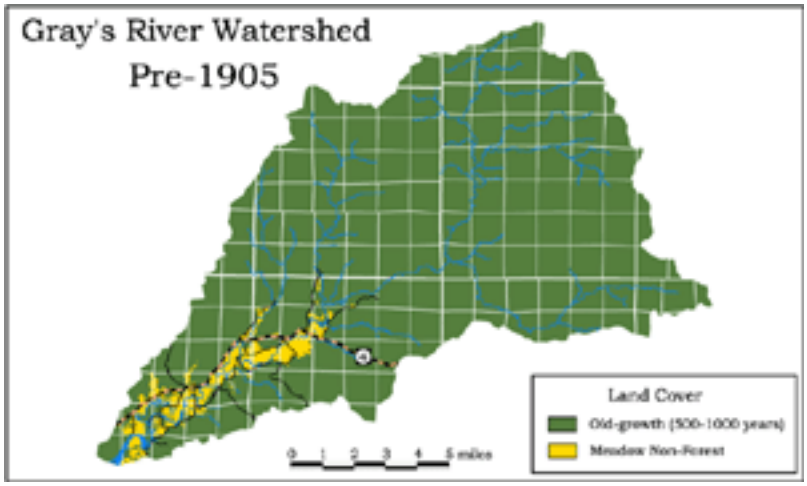


Figure A.11. Grays River Watershed and Land Cover Prior to 1905 (Scott 2001)

During World War I, the demand for timber, especially spruce for airplane construction, increased dramatically (Martin 1997). Between 1905 and 1920, areas in the upper Grays River watershed were logged, although much of the steep headwater terrain was inaccessible. During the Great Depression, logging operations in the watershed were minimal and most of the logged upland hills outside the river valley recovered naturally (Scott 2001). This second-growth forest replaced the old-growth trees that were logged between 1870 and 1920. The largest timber companies in the area at that time, Willamette Pulp & Paper and the Meserve Logging Company, ceased logging operations in the upper Grays River watershed around 1920 (B. Penttila, unpublished).

Logging resumed in earnest during World War II. The Schmand Logging Company logged spruce in the Hull Creek area during World War II. The Hull Creek spruce was used for making gliders (Martin 1997). After World War II, logging methods changed significantly, with roads and trucks replacing railroads and mechanized logging becoming the norm. New logging technology also allowed for harvesting of steeper slopes and other areas not within the capabilities of previous logging methods. In addition, selective cutting of specific species gave way to the practice of clear-cut logging. The combination of these new technologies and new logging practices opened almost all the upper Grays River watershed to logging (Scott 2001). Figure A.12 shows the land cover in the Grays River watershed prior to 1942, and Figure A.13 shows the land cover as of 1942.

Prior to 1942, the average harvest rate was 3.0 km^2/year, or approximately 1.3% of the watershed per year (Table A.1). By 1942, approximately 8% of the upper watershed had been harvested. Timber harvesting continued at a rate of about 3.3 km^2/year between 1942 and 1953 during expansion of activities into the eastern and northern portions of the watershed (Scott 2001).

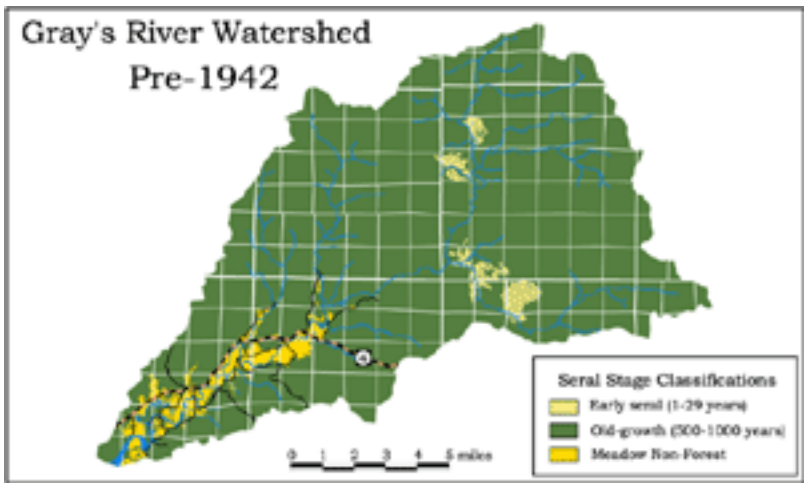


Figure A.12. Grays River Watershed and Land Cover Prior to 1942 (Scott 2001)

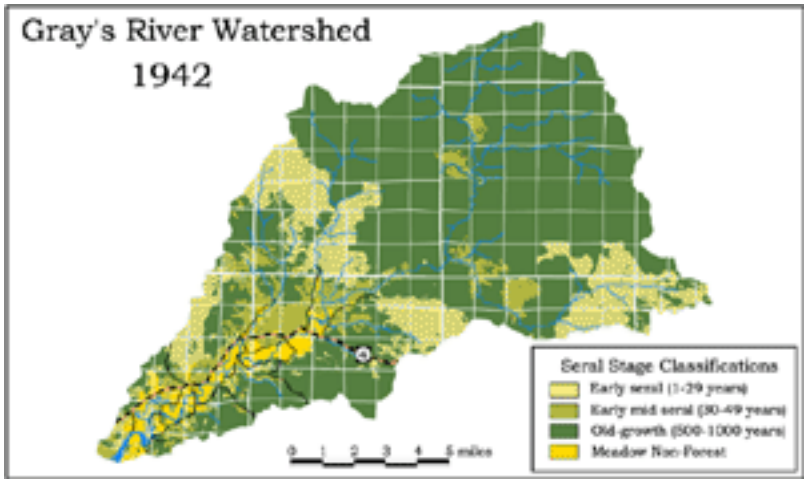


Figure A.13. Grays River Watershed and Land Cover as of 1942 (Scott 2001)

In the late 1940s, the Wirkkala Brothers Logging Company of Naselle established operations in the Grays River area (B. Penttila, unpublished master's thesis). Widespread use of roads and trucking in timber-harvest operations also began in the 1950s (Scott 2001). It was the Wirkkala Brothers who constructed the main access road (the 7000 Line) into the headwaters of the Grays River (B. Penttila, unpublished master's thesis). During and after World War II, some of the largest timber companies in the country also began to purchase land and conduct logging operations in the Grays River watershed. Crown Zellerbach operated in the Grays River area until 1985, when it was purchased and converted into a land-management operation (Pyle 2001). In 1955, the Wirkkala Brothers sold their logging operations to Weyerhaeuser Timber Company, which harvested timber in Grays River until 1983 when the operation was closed (B. Penttila, unpublished master's thesis).

It was Weyerhaeuser, working with a notion of enhancing salmon passage, that dynamited the Grays River falls in the 1950s and 1960s, which created the boulder-strewn riffle present today. Prior to that, local stories tell of the early splash-dam loggers using copious amounts of blasting powder to reshape the gorge for better log passage (B. Penttila, unpublished master's thesis).

In 1947, reforestation of logged areas became mandatory in Washington State (Martin 1997). The harvesting of second-growth forest had begun by 1953 (Scott 2001). Harvest operations expanded throughout the study area and increased to 5.0 km^2/year between 1953 and 1964 (Table A.1). Figure A.14 shows the land cover in the Grays River watershed as of 1953.

Table A.1. Summary of Forest Clearing in the Grays River Watershed (Scott 2001)

Period	Average Harvest Rate (km^2/year)	Total Percentage of Upper Watershed Harvested	Percentage of Harvest Area by Stand-Age Category		
			Old Growth	Second Growth	Third Growth
1905–1942	3.0	8.2	100.0	0.0	0.0
1942–1953	3.3	19.1	94.4	5.6	0.0
1953–1964	5.0	40.7	97.0	3.0	0.0
1964–1976	5.3	73.6	93.2	6.8	0.0
1976–1983	9.1	88.9	83.3	15.7	0.5
1983–1990	5.5	94.4	40.0	58.0	2.0
1990–1996	4.1	97.6	20.1	77.4	1.5
1996–2003	3.1	98.9	0.0	95.0	5.0

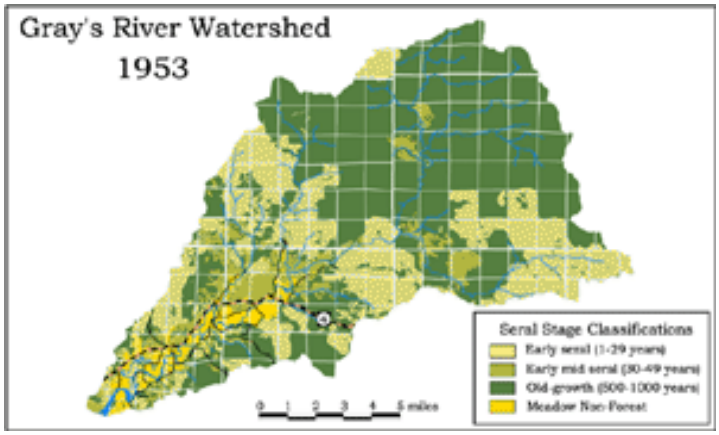


Figure A.14. Grays River Watershed and Land Cover as of 1953 (Scott 2001)

During the period 1953 to 1964, the fraction of remaining old growth declined from 59% to 39%. In 1953, second-growth forest made up 41% of the Grays River watershed and 38% of the timber in the watershed was less than 30 years old. As of 1964, second-growth forest made up 60% of the Grays River watershed and 44% of the watershed was less than 30 years old. On Columbus Day in 1962, a major windstorm hit the Pacific Northwest, blowing down millions of acres of forest throughout the region. This storm created an opportunity for timber companies to increase their harvest rates using salvage-logging operations (Scott 2001). Spurred by this abundant and inexpensive supply of lumber, the Pacific Rim export market expanded significantly. The demands of this export-driven market for lumber continued into the 1970s. The market for high-value, old-growth timber also increased dramatically during this period. During this period, 97% of the timber harvested in the upper Grays was old-growth trees (Scott 2001).

The period between 1964 and 1976 marked expansion of harvest practices to the northeast portion of the Gray River watershed. The road network was also expanded significantly during this period, providing access to the highest elevations in the watershed (Scott 2001). During this period, the average harvest rate was 5.3 km^2/year (Table 3.2). By 1976, only 18% of the original old-growth forest in the watershed remained (Scott 2001). Figure A.15 shows the land cover in the Grays River watershed as of 1964.

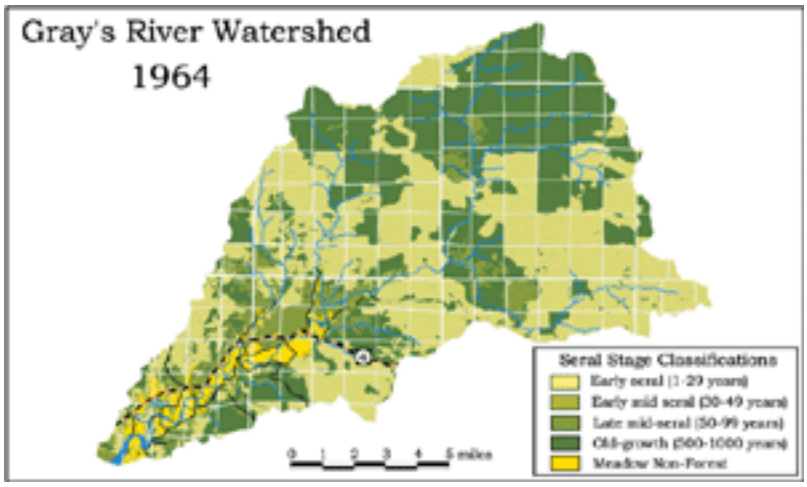


Figure A.15. Grays River Watershed and Land Cover as of 1964 (Scott 2001)

Between 1976 and 1983, timber-harvest rates peaked at 9.1 km^2/year, or 4% of the watershed per year, and included the logging of some third-growth forest. Over 95% of the old-growth forest within the Grays River watershed had been harvested by 1983 (Table 3.2). Figure A.16 shows the land cover in the Grays River watershed as of 1976, and Figure A.17 shows the land cover as of 1983.

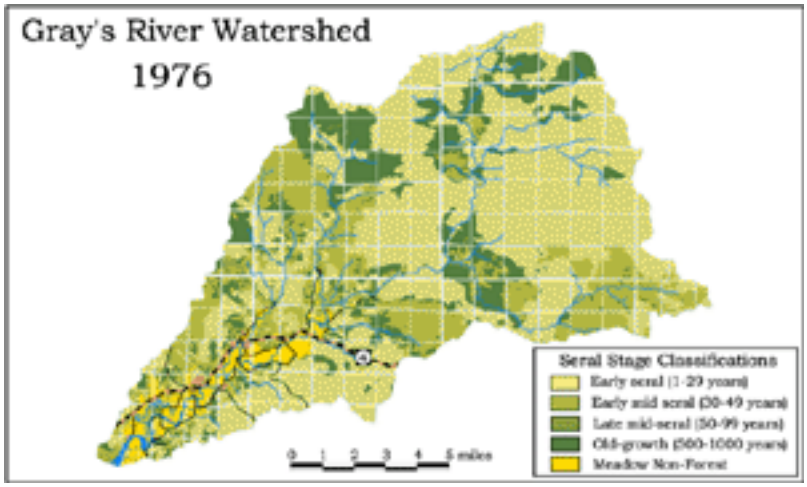


Figure A.16. Grays River Watershed and Land Cover as of 1976 (Scott 2001)

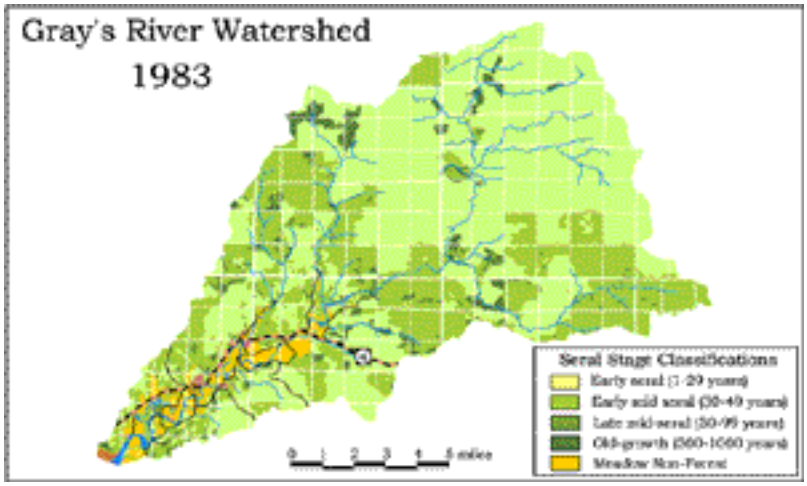


Figure A.17. Grays River Watershed and Land Cover as of 1983 (Scott 2001)

Annual harvest rates declined thereafter to 5.5 km^2/year between 1983 and 1990, 4.1 km^2/year between 1990 and 1996, and 3.1 km^2/year between 1996 and 2003 (Table A.1). Only about 2%, or 4.6 km^2 (1,137 acres), of the original old-growth forest remained as of 1996. This remaining old-growth forest exists as 400 individual fragments of the original, nearly continuous forest, ranging in size from 3.6 to 216 acres (Scott 2001). Figure A.18 shows the land cover in the Grays River watershed as of 1990, and Figure A.19 shows the land cover as of 1996. Between 1942 and 2003, approximately 95% of the forest of the Grays River watershed was clear-cut (Herrera 2005). Figure A.20 shows a summary of timber-harvest activities in the upper Grays River watershed as of 2003.

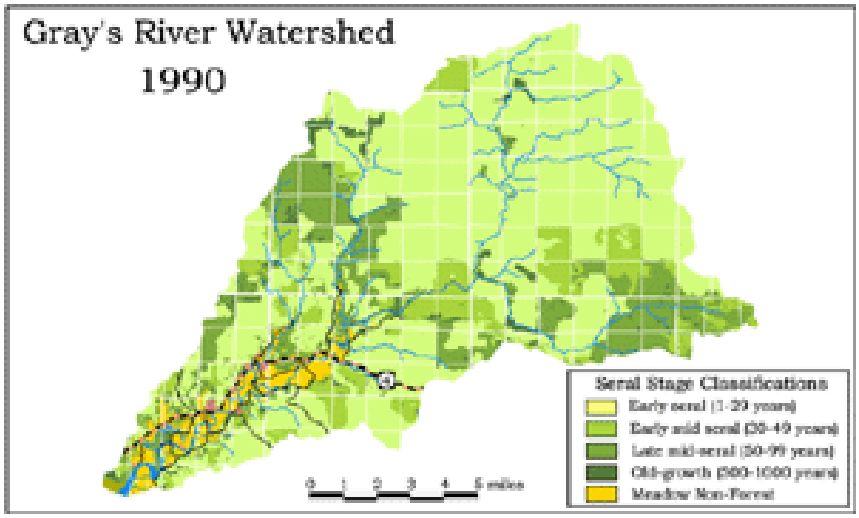


Figure A.18. Grays River Watershed and Land Cover as of 1990 (Scott 2001)

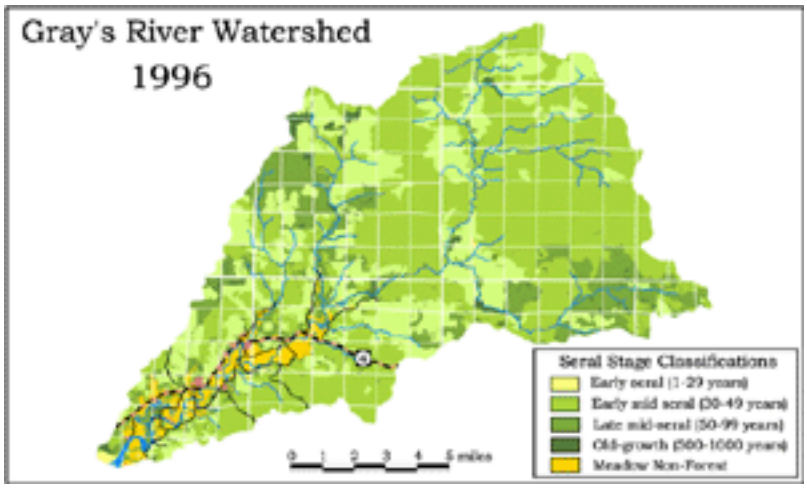


Figure A.19. Grays River Watershed and Land Cover as of 1996 (Scott 2001)

Currently, private timber companies own 95% of the land area in the upper Grays River watershed (Figure A.21). Public lands held by the state of Washington (Washington Department of Natural Resources and Washington Department of Fish and Wildlife) represent only about 5% of total land area within the upper watershed. Timber harvest is still the principal land-use activity in the watershed. There are 13 miles of paved state highway (SR 4) and 62 miles of paved county road within the watershed. Most of the roads in the upper watershed are logging access roads (Figure A.22). There are currently more than 600 miles of logging roads in the upper Grays River watershed, resulting in an overall road density of approximately 5 miles of road per square-mile of watershed area. Logging roads alone account for about 5% of the total land area in the watershed (Scott 2001).

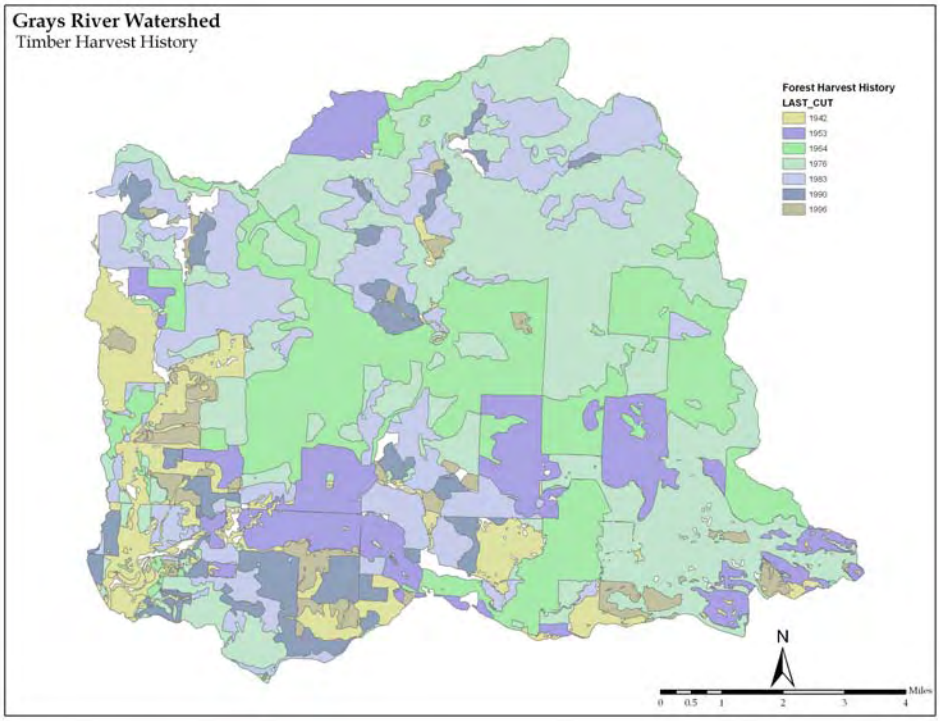


Figure A.20. Summary of Grays River Timber Harvest as of 2003 (Herrera 2005)

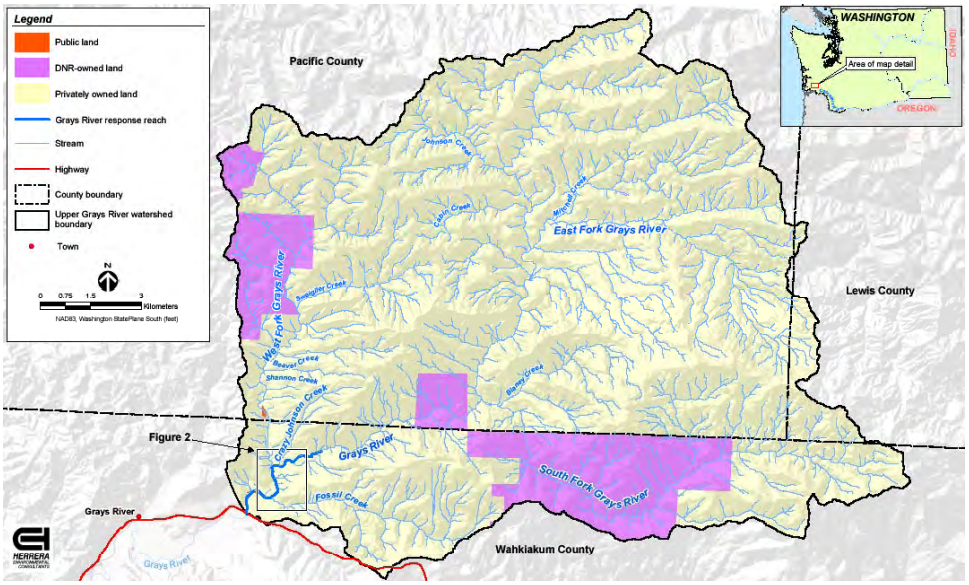


Figure A.21. Land Ownership in the Upper Grays River Watershed (Herrera 2005)

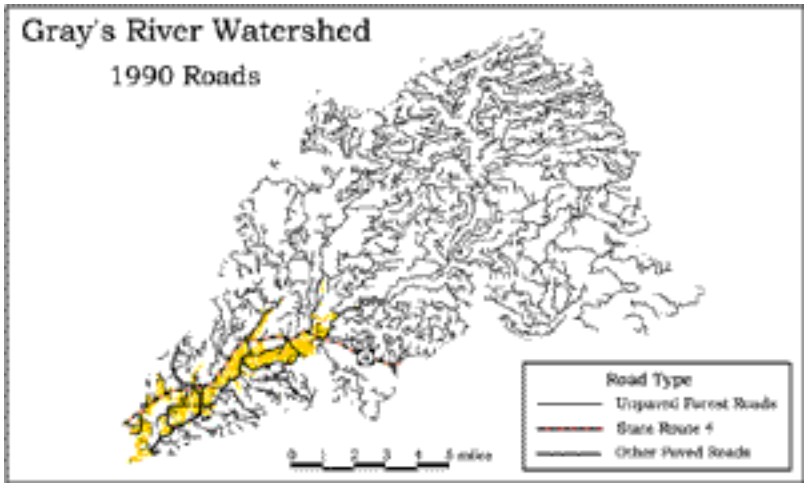


Figure A.22. Grays River Watershed Road Network (Scott 2001)

Figures A.23 through A.36 show examples of historical timber activities in the Grays River watershed. All historical photographs are courtesy of the Appelo Archive Center, Wahkiakum Community Foundation, unless noted otherwise.

Figure A.23. Logging in the Grays River Watershed, Circa 1900

Figure A.24. Logging in the Grays River Watershed, Circa 1900

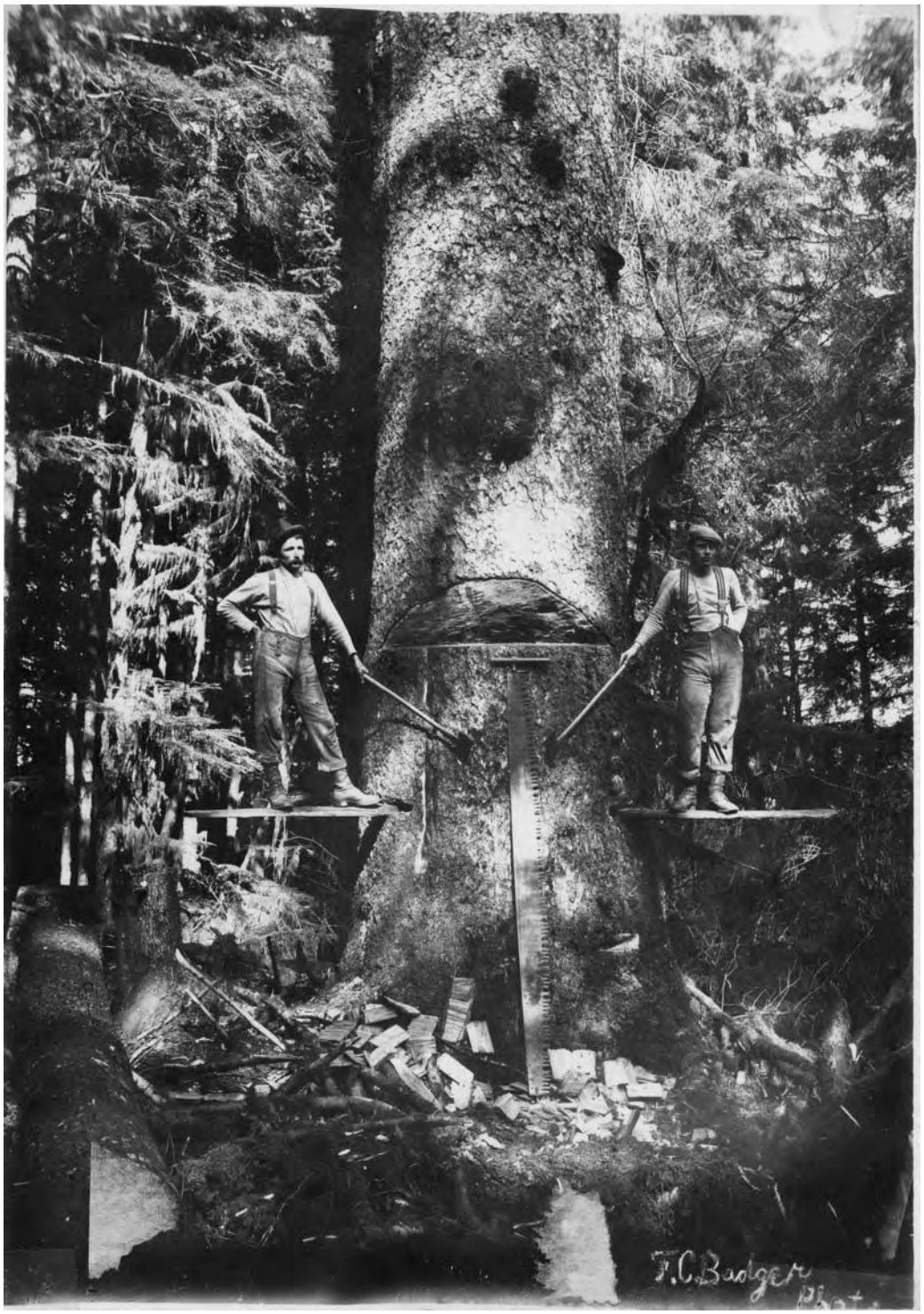


Figure A.25. Logging in the Grays River Watershed, Circa 1900

Figure A.26. Logging in the Grays River Watershed, Circa 1900

Figure A.27. Logging in the Grays River Watershed, Circa 1900

Figure A.28. Logging in the Grays River Watershed, Circa 1900

Figure A.29. Grays River Logging Camp, Circa 1910

FigureA.30. Grays River Logging Camp, Circa 1910

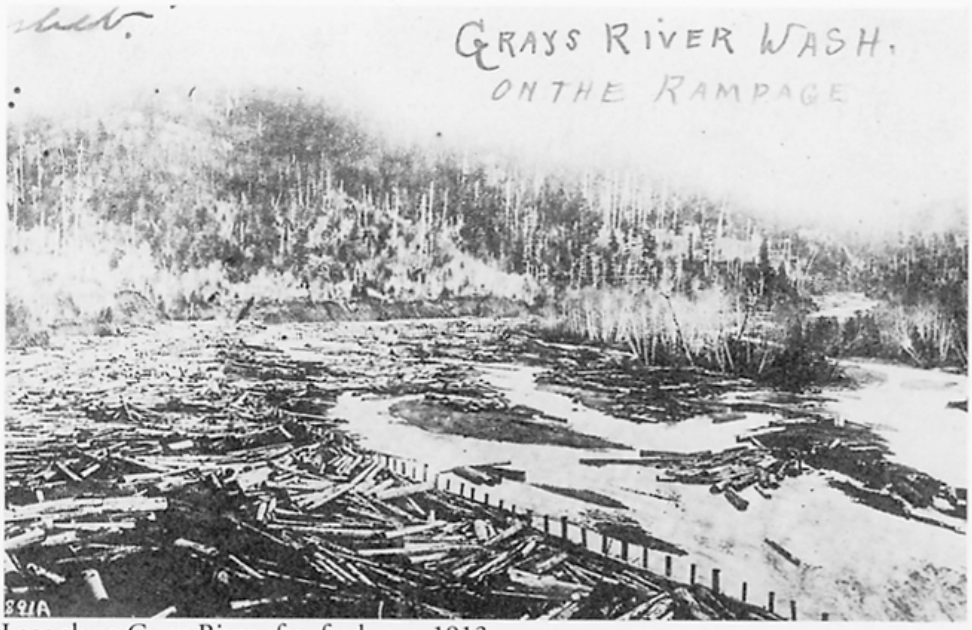
Figure A.31. Logs on the Grays River, Circa 1910

Figure A.32. Logs on the Grays River, Circa 1910

Figure A.33. Splash Dam on the West Fork of the Grays River, Circa 1900

Figure A.34. Logging Railroad in the Grays River Watershed, Circa 1910

Figure A.35. Grays River Valley, Circa 1910



Logs along Grays River after freshet, *c.* 1913.
Courtesy Friends of Skamokawa.

Figure A.36. Logs on the Grays River, Circa 1913

A.3 References

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Appendix B

Upper Grays River
Geomorphic Assessment Photographic Log

Appendix B

Upper Grays River
Geomorphic Assessment Photographic Log

Photographs presented in this appendix were provided by Herrera Environmental Consultants, Inc., Seattle, Washington.

Photo Number	Photo Description
MW-2-1	Large rotational landslide above Rd. 7050; note people in center for scale.
MW-2-2	Gulley erosion below large rotational landslide shown in photo MW-2-1.
MW-2-3	Gulley erosion below large rotational landslide shown in photo MW-2-1.
MW-2-4	1996 aerial photograph of large rotational landslide in photo MW-2-1 and debris flow.
MW-2-5	2003 aerial photograph of large rotational landslide in photo MW-2-1 and debris flow.
MW-3-1	Debris-flow deposit at Rd. 6200.
MW-3-3	2003 aerial photograph of debris flow in photo MW-3-1.
MW-4-1	Shallow translational landslides.
MW-4-2	2003 aerial photograph of shallow translational landslides shown in photo MW-4-1.
RS-1-1	Example of mainline road conditions (Rd. 7000).
RS-2-1	Example of mainline road conditions (Rd. 7000).
RS-3-1	Example of non-active road conditions (Rd. 6200).
RS-4-1	Example of decommissioned road conditions (Rd. 6275).
CC-1-1	Colluvial channel scoured to bedrock from historical debris flow.
CC-2-1	Boulder cascade channel.
HTC-1-1	Confined boulder dominated step-pool channel segment.
HTC-1-2	Relic LWD providing grade control in confined step-pool channel.
HTC-2-1	Confined boulder step-pool channel.
TRC-1-1	Plane-bed of Cabin Creek.
TRC-1-2	Plane-bed reach of Cabin Creek with boulder steps.
TRC-2-1	Moderately confined plane-bed reach of Alder Creek.
TRC-3-1	Moderately confined plane-bed channel of Mitchell Creek.
TRC-4-1	Confined plane-bed reach.
EGR-1-1	Plane-bed morphology in the East Fork Grays River.
EGR-1-2	Looking downstream at plane-bed morphology in the East Fork Grays River.
SGR-1-1	Bifurcated cobble bedded plane-bed channel at the upstream end of the survey reach.
SGR-1-2	Debris dam with entrained LWD.
SGR-1-3	Foreset beds of sand are evidence of a prograding depositional environment.

Photo Number	Photo Description
SGR-1-4	Coarse grained alluvial sediment visible in the terrace deposit.
SGR-2-1	Valley jam on the South Fork Grays River.
SGR-2-2	Looking downstream over a gravel and cobble bar from the upstream end of the survey reach.
WGR-1-1	Gravel and cobbled bedded plan bed reach of the West Fork Grays River.
WGR-2-1	Gravel bedded pool riffle reach of the West Fork Grays River.
WGR-2-2	Bifurcated gravel bedded pool-riffle reach in the West Fork Grays River.
WGR-2-3	Erosional scar in the weak marine sedimentary rocks at the left edge of the West Fork Grays River floodplain.
CJ-1-1	Beaver dam integrated into LWD in gravel and sand bedded low-gradient floodplain channel of Crazy Johnson Creek.
GRR-1-1	Cobble bedded Plan-bed channel of the main stem Grays Rivet.
GRR-1-2	Trees buried in growth position indicate variability in vertical channel elevation.
GRR-1-3	Buried LWD jam exposed in right bank of exiting channel alignment.
GRR-1-4	Looking downstream, the main channel and avulsion path is located to the left of the photo.
GRR-1-5a	Eroding hillslope near the head of the avulsion channel alignment.
GRR-1-5b	Eroding hillslope at the left margin of the floodplain valley.
GRR-1-6	Looking downstream within the straightened and levied portion of the main stem Grays River.

Photo MW-2-1. Survey Location MW-2 (October 26, 2004). Looking up from Rd. 7050 (above Blaney Creek) at headscarp of large rotational landslide in basalt. Note people in center for scale (arrow).

Photo MW-2-2. Survey Location MW-2 (October 26, 2004). Looking down westernmost gulley (one of three) below large rotational landslide shown in photo MW-2-1. Gully begins at outlet of culvert beneath Rd. 7050 and has eroded an estimated 3,500 m^3 of sediment.

Photo MW-2-3. Survey Location MW-2 (October 26, 2004). Looking across the western gulley shown in photo MW-2-2 from below Rd. 7050. Erosion results from concentrated surface water directed by the forest road drainage network.

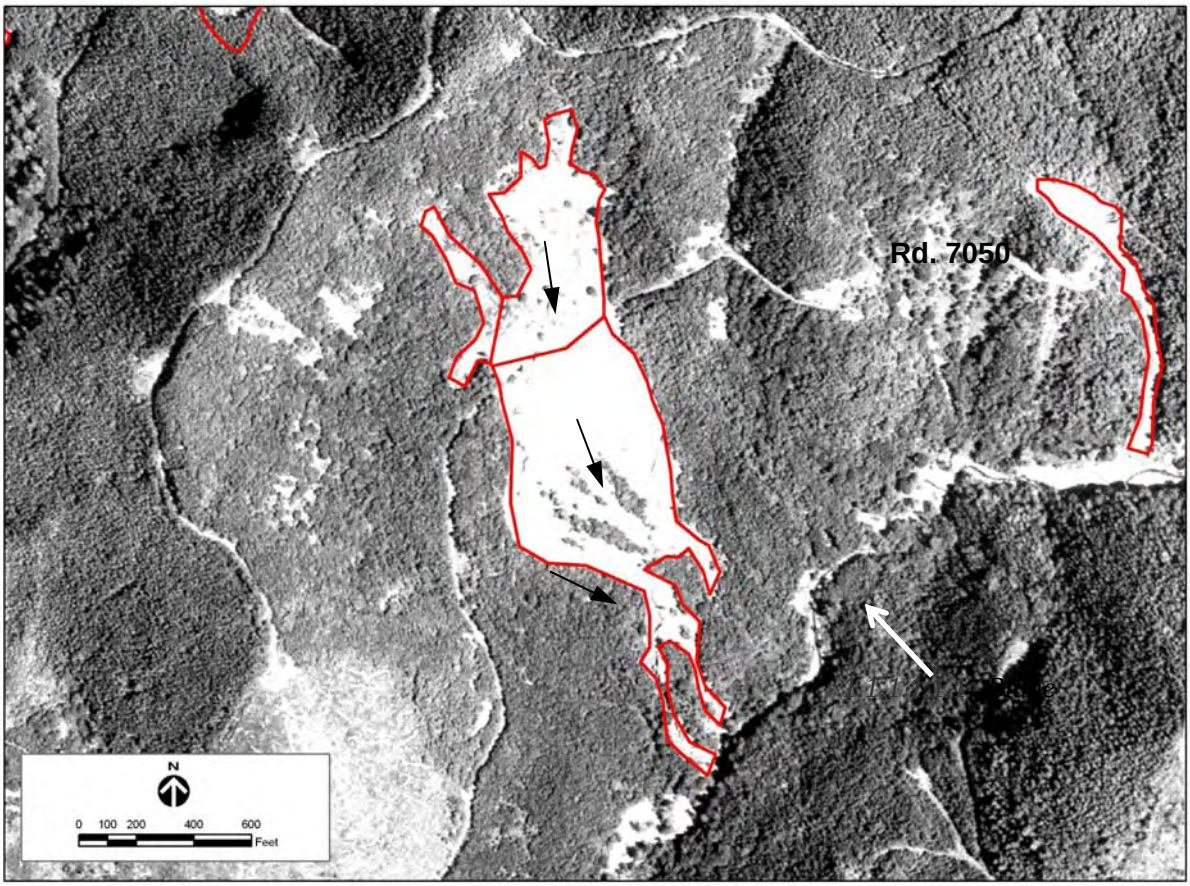

Photo MW-2-4. Survey Location MW-2 (1996 aerial photograph). Large rotational landslide and sediment delivery to Blaney Creek. Lack of vegetation indicates failure occurred between 1994 and 1996. Photos MW-2-1 through 3 were taken from Rd. 7050, where the road crosses the landslide. Outline created during landslide inventory.

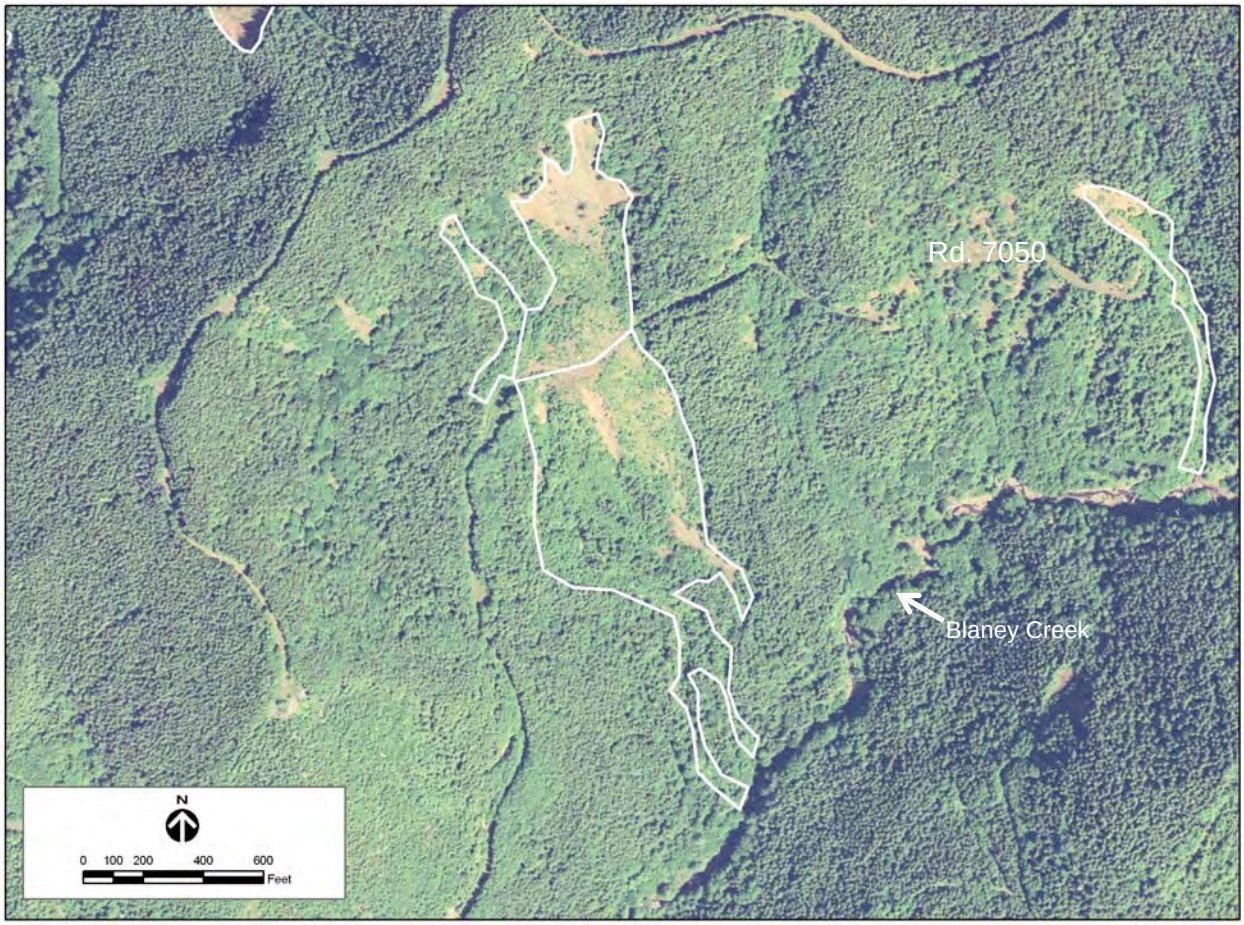

Photo MW-2-5. Survey Location MW-2 (2003 aerial photograph). Large rotational landslide showing partial revegetation, except on exposed bedrock headwall and in actively eroding gullies below Rd. 7050.

Photo MW-3-1. Survey Location MW-3 (October 27, 2004). Sediment and wood accumulation from debris flow in colluvial hollow above Rd. 6200.

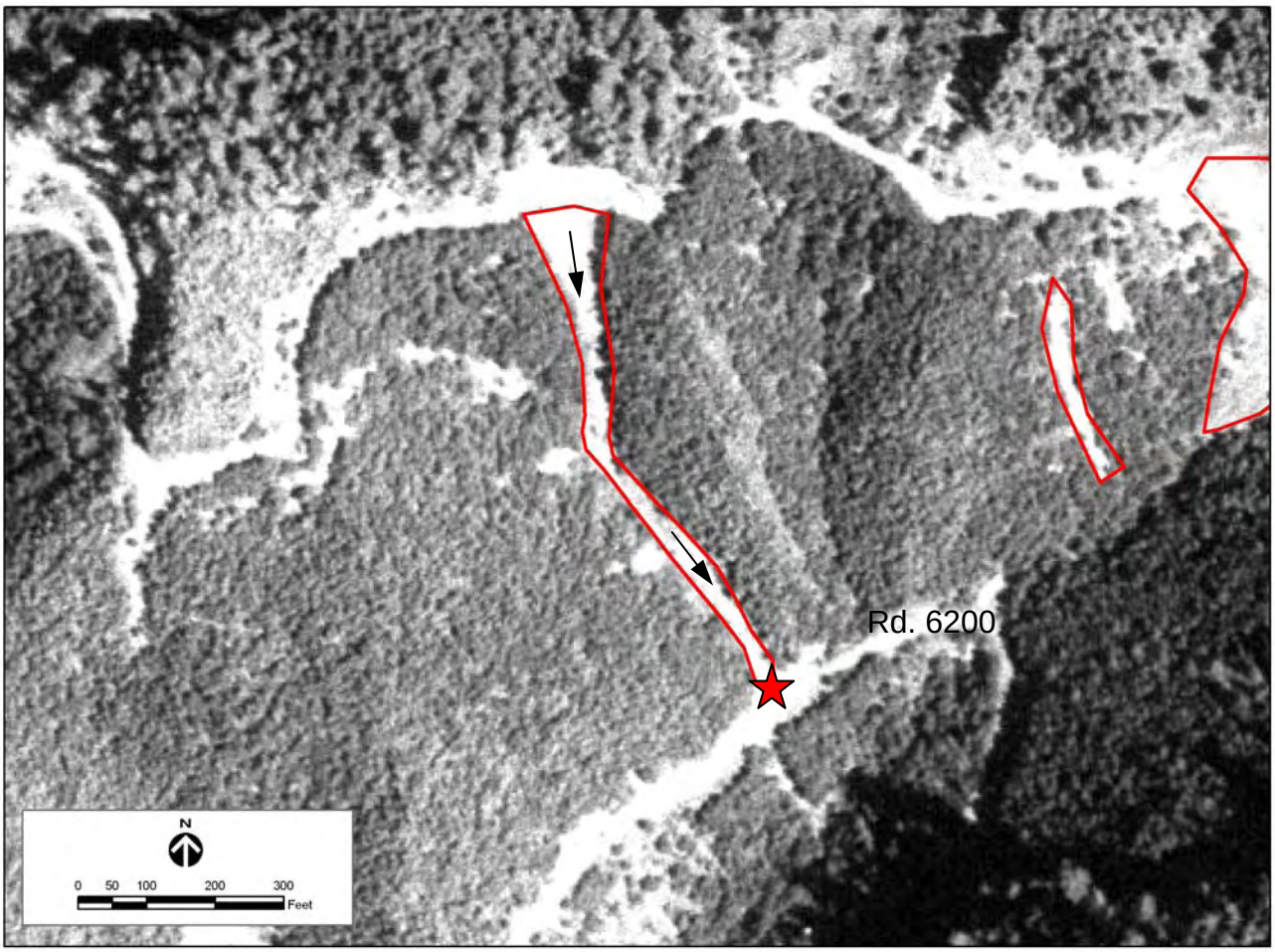

Photo MW-3-3. Survey Location MW-3 (1996 aerial photograph). Debris-flow initiation and runout above Rd. 6200. Star denotes location of photos MW-3-1.

Photo MW-4-1. Survey Location MW-4 (October 28, 2004). Translational landslides on steep slopes within the headwaters of the West Fork Grays River. Landsliding has removed the thin ~1-meter soil horizon down to bedrock of the Crescent Formation (basalt). The visible vertical relief is approximately 500 meters. View is toward the east.

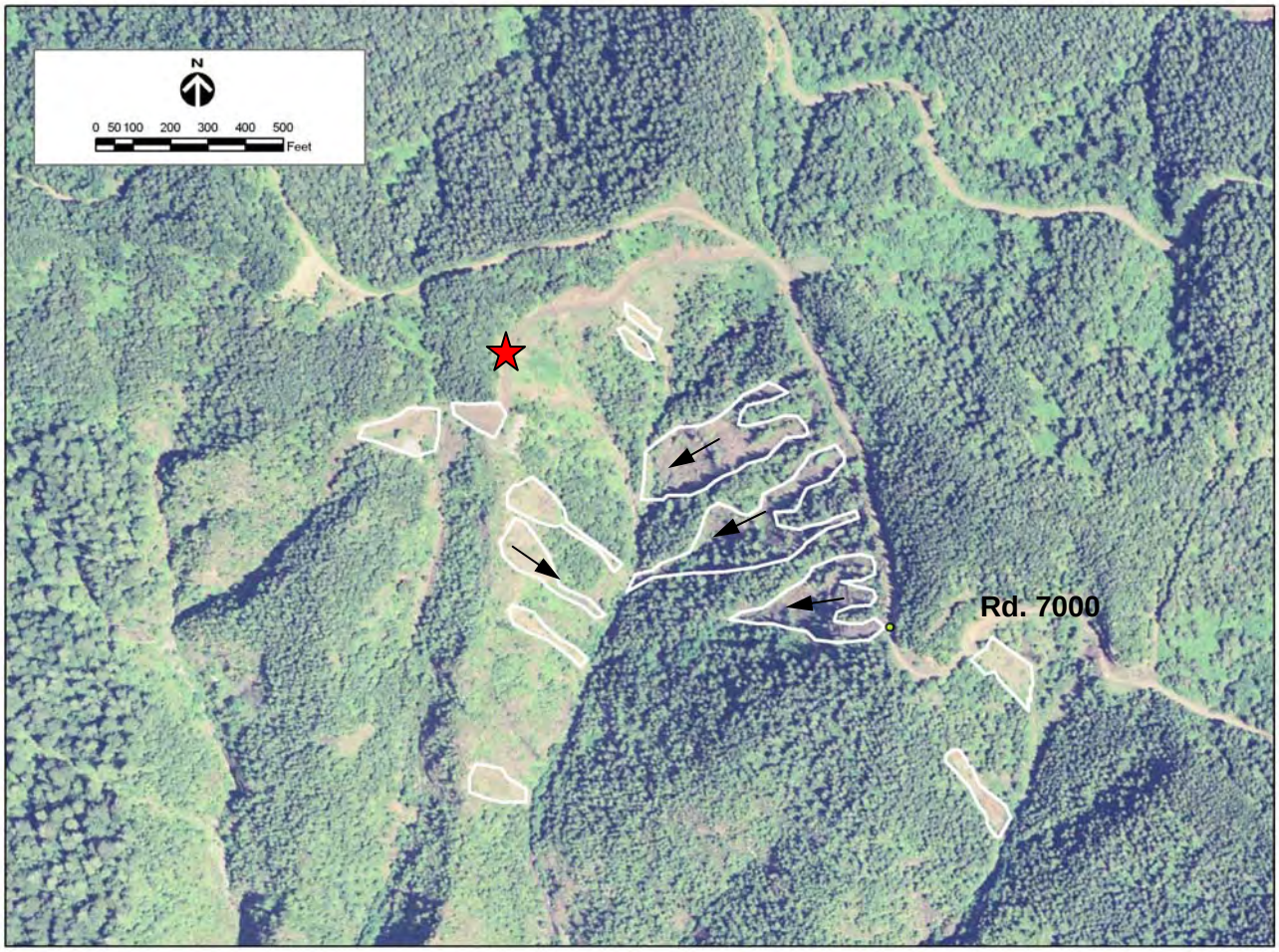

Photo MW-4-2. Survey Location MW-4 (2003 aerial photograph). Translational landslides in the headwaters of the West Fork Grays River. Star denotes approximate location of photo-point MW-4-1.

Photo RS-1-1. Survey Location RS-1 (October 28, 2004). Example of mainline road conditions (Rd. 7000).

Photo RS-2-1. Survey Location RS-2 (October 28, 2004). Example of mainline road conditions (Rd. 7000).

Photo RS-3-1. Survey Location RS-3 (October 27, 2004). Example of non-active road conditions (Rd. 6200).

Photo RS-4-1. Survey Location RS-4 (October 27, 2004). Example of decommissioned road conditions (Rd. 6275).

Photo CC-1-1. Survey Reach CC-1 (October 27, 2004). Colluvial channel scoured to bedrock from historical debris flow. Angular cobbles form small fan deposit in the lower right part of photo.

Photo CC-2-1. Survey Reach CC-2 (October 27, 2004). Boulder cascade channel is incising into debris flow deposits from upstream. Channel substrate is mixed cobble and gravel between boulders. Stream banks are poorly sorted but little fine sediment is in the active channel indicating this fraction of the sediment load is rapidly transported downstream.

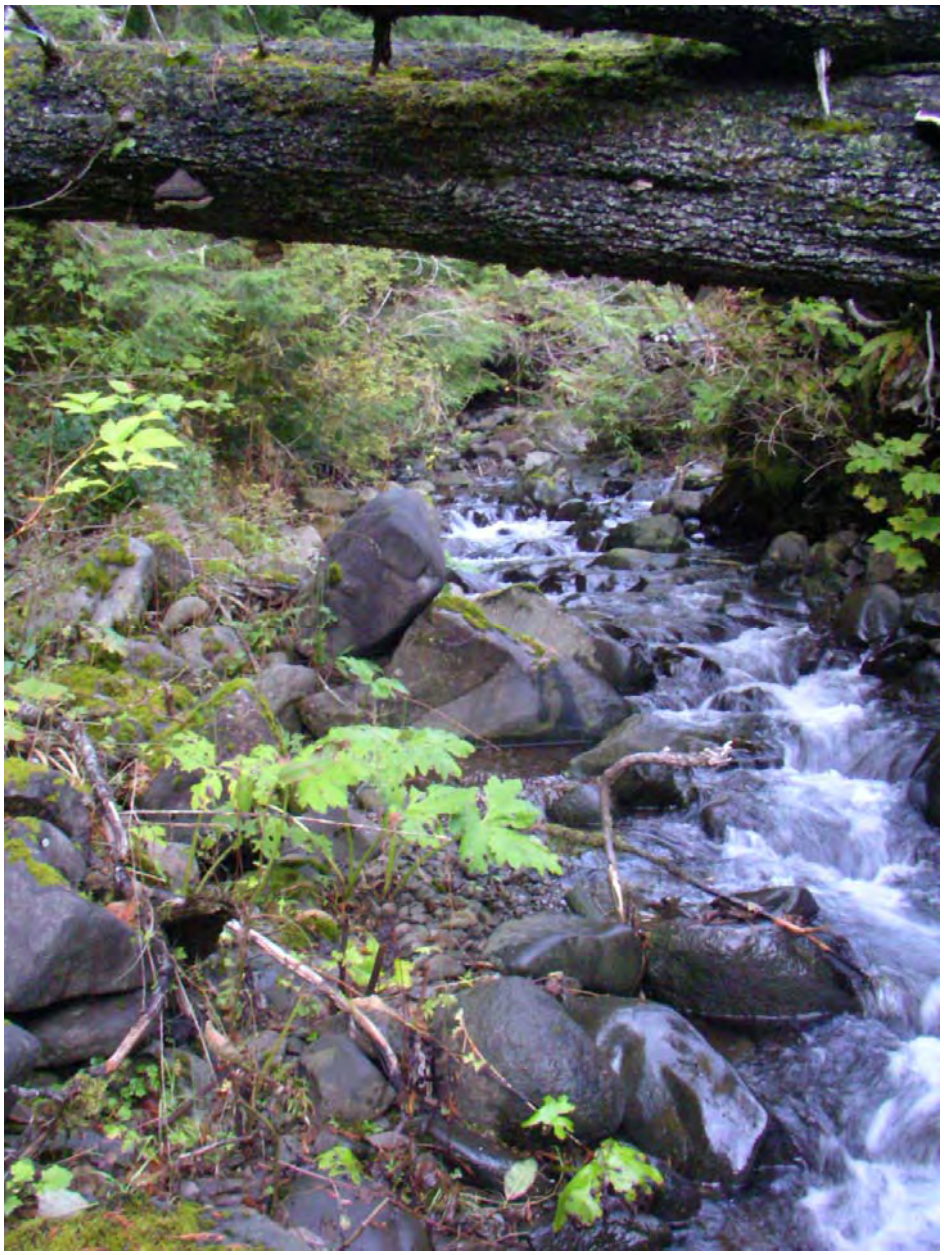

Photo HTC-1-1. Survey Reach HTC-1 (October 27, 2004). Confined boulder-dominated, step-pool channel segment.

Photo HTC-1-2. Survey Reach HTC-1 (October 27, 2004). Relic LWD providing grade control and sediment storage in confined step-pool channel. Extensive lateral bar deposits of cobble, sand and gravel in transport-dominated channel indicate high sediment loading.

Photo HTC-2-1. Survey Reach HTC-2 (October 27, 2004). Confined boulder step-pool channel. Channel substrate is predominantly cobble and boulders. Gravel substrate sediment covers approximately 10% of the active channel and is visible in a lateral bar deposit behind boulders at the right side of the photo (river left). Functional LWD is lacking in the reach, and there is poor recruitment potential from the adjacent riparian forest.

Photo TRC-1-1. Survey Reach TRC-1 (October 28, 2004). Plane-bed channel of Cabin Creek. Channel is lacking functional LWD and adjacent riparian community of alders offers poor future recruitment potential. Significant fine sediments (silt to fine gravel) composed of marine siltstone blanket the substrate in this reach and are visible on the channel bed in the front center portion of the photograph.

Figure TRC-1-2. Survey Reach TRC-1 (October 28, 2004). Plane-bed reach of Cabin Creek with boulder steps. Deposits of loose fine sediments (silt to fine gravel) composed of marine siltstone are visible in lee areas behind boulders. These deposits covered approximately 40% of the active channel bed and were up to 0.5 m deep.

Photo TRC-2-1. Survey Reach TRC-2 (October 28, 2004). Moderately confined plane-bed reach of Alder Creek. Channel instability is indicated by extensive vertical eroding banks and frequent recruitment of alders growing adjacent to the channel. The reach contains significant loose deposits of fine-grained marine siltstone sediment, which forms lateral and mid-channel bars and covers approximately 25% of the active channel bed.

Photo TRC-3-1. Survey Reach TRC-3 (October 28, 2004). Moderately confined plane-bed channel of Mitchell Creek. Channel is primarily gravel and cobble bedded. Banks generally appeared stable, but undercut alders at the channel margins indicate local erosion and widening. Channel contains little hydraulic complexity, and there are few opportunities for sediment storage in the active channel. There was no functional LWD observed in the surveyed reach, and the alder-dominated riparian forest provides poor recruitment potential.

Photo TRC-4-1. Survey Reach TRC-4 (October 28, 2004). Confined plane-bed reach. Flow is toward camera. Channel bed is predominantly cobble bedded with gravel and boulders. Inset floodplain surfaces indicate a previous period of channel widening and aggradation. Person at right is standing on inset floodplain surface. Eroding banks and undercut and locally recruited riparian vegetation indicate continued lateral instability. Vertical instability and further downcutting are suggested by exposure of buried relic LWD at the base of the inset floodplain surface. The channel contains large bars, and moderate fine sediment loading was apparent in low-energy depositional zones.

Photo EGR-1-1. Survey Reach EGR-1 (October 27, 2004). Plane-bed morphology in the East Fork Grays River. Channel substrate is predominantly gravel and cobble. No functional LWD was encountered in the 500-m survey reach, and there is limited recruitment potential from the adjacent riparian forest.

Photo EGR-1-2. Survey Reach EGR-1 (October 27, 2004). Looking downstream at plane-bed morphology in the East Fork Grays River. This photo illustrates the lack of channel complexity and limited recruitment potential from the adjacent riparian forest.

Photo SGR-1-1. Survey Reach SGR 1 (March 24, 2004). Bifurcated cobble-bedded plane-bed channel at the upstream end of the survey reach. Little LWD was observed in the reach, and the adjacent riparian forest dominated by alders provides little recruitment potential.

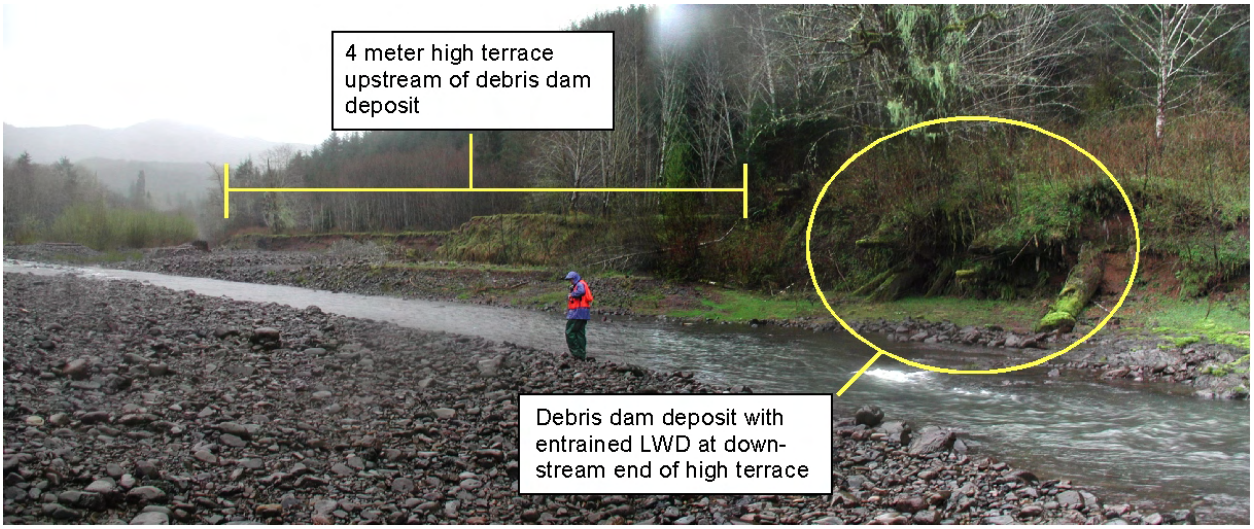

Photo SGR-1-2. Survey Reach SGR 1 (March 24, 2004). Remnant of a debris dam with entrained LWD is visible in the lower right portion of the photo. A 4-m-high terrace formed behind the debris dam. Radiocarbon dating of LWD from the debris dam deposit and from wood deposited 1 m below the top of the terrace surface indicates that the terrace began forming between 215 and 335 years ago (275 years before present ±60 years) and deposition continued through at least 145 and 265 years ago (205 years before present ±60 years).

Photo SGR-1-3. Survey Reach SGR 1 (March 24, 2004). Foreset beds of sand are evidence of a prograding depositional environment, and silt layers indicate periods of low-energy deposition.

Photo SGR-1-4. Survey Reach SGR 1 (March 24, 2004). Coarse-grained alluvial sediment visible in the terrace deposit at the right edge of the photo occurs within the upstream portion of the terrace, representing a higher-energy depositional environment.

Photo SGR-2-1. Survey Reach SGR 2 (March 24, 2004). Valley jam on the South Fork Grays River. Jam is approximately 50 m wide and creates a 1.5-m step in the channel profile. Jam provides significant fine and coarse sediment storage and generates a steep hydraulic gradient between mainstem channels.

Photo SGR-2-2. Survey Reach SGR 2 (March 24, 2004). Looking downstream over a gravel and cobble bar from the upstream end of the survey reach. The gradient of the channel at left is more moderate than the right channel, creating a hydraulic gradient from left to right across the bar between the channels.

Photo WGR-1-1. Survey Reach WGR 1: (September 22, 2004). Gravel and cobble-bedded plane-bed reach of the West Fork Grays River. The lower 1.25 km of the West Fork contains low wood loading and little channel complexity. The riparian community adjacent to the West Fork is typically alder-dominated and provides poor recruitment potential. The channel is confined against marine sedimentary bedrock at the floodplain margin through much of the survey reach. Recruitment potential is locally moderate, with occasional mature conifers adjacent to the current channel alignment. The alder-dominated forest visible at photo left is typical of the adjacent riparian community.

Photo WGR-2-1. Survey Reach WGR 2: GPS 44-59 (September 22, 2004). Gravel-bedded pool riffle reach of the West Fork Grays River. Recruitment potential is locally moderate with occasional mature conifers adjacent to the current channel alignment. The alder-dominated forest visible in the back ground is typical of the adjacent riparian community.

Photo WGR-2-2. Survey Reach WGR 2: GPS 44-59 (September 22, 2004). Bifurcated gravel-bedded pool-riffle reach in the West Fork Grays River. Moderate LWD loading provides local hydraulic complexity. Riparian forests of alders generally provide low LWD recruitment potential. Visible large mid-channel bars and fine sediment deposition in low-energy zones are indicative of high sediment loading that was observed.

Photo WGR-2-3. Survey Reach WFGR 2: GPS 44-59. (September 22, 2004). Erosional scar in weak marine sedimentary rocks at the left edge of the West Fork Grays River floodplain. The lack of mass wasting deposits at the base of the feature indicates total delivery from the sediment source to the channel network. Bedrock erosion at the floodplain margin was observed frequently in the West Fork Grays River.

Photo CJ-1-1. Survey Reach CJ-1 (March 23, 2004). Beaver dam integrated into LWD in gravel and sand-bedded low-gradient floodplain channel of Crazy Johnson Creek. Upwelling was observed in the floodplain channel downstream of similar beaver ponds.

Photo GRR-1-1. Survey Reach GRR-1 (March 23, 2004). Cobble-bedded plane-bed channel of the main-stem Grays River. Flow is toward the camera. The limited LWD and riparian recruitment potential apparent in the photo are typical of the response reach.

Photo GRR-1-2. Survey Reach GRR-1 (March 23, 2004). Trees buried in growth position indicate variability in channel elevations. Alders growing on the floodplain surface are approximately 25 years old. The existing floodplain surface is approximately 1.2 m above the historical surface.

Photo GRR-1-3. Survey Reach GRR-1 (March 23, 2004). Buried LWD jam exposed in the right bank of existing channel alignment. Buried jams observed in multiple locations suggest that historic LWD loading conditions were greater in the past.

Photo GRR-1-4. Survey Reach GRR-1 (September 20, 2004). Looking downstream, the main channel and avulsion path are located to the left of the photo. The depositional surface at right is associated with the avulsion event in December 1999. The vegetated floodplain surface was located behind the levee prior to the avulsion. The depositional surface is approximately 1 m above the floodplain surface.

Photo GRR-1-5a and 5b. Survey Reach GRR-1 (September 21, 2004). Eroding hillslope (a) and (b) at the left margin of the floodplain valley. Erosion in (a) is occurring near the head of the lower avulsion channel alignment; the height of the exposure is approximately 15 m. Erosion throughout the meander occupied by the mainstem alignment in 1996 is visible in (b). The unvegetated channel has been at the base of this slope since at least 1939. Landsliding along this slope is visible in the 1996 aerials photos and indicates a recent increase in the local sediment delivery from this source over the last decade.

Photo GRR-1-6. Survey Reach GRR-1 (March 23, 2004). Looking downstream within the straightened and levied portion of the Gorley subreach in the Grays River response reach. This reach is located downstream of the pre-avulsion (December 1999) confluence of the West Fork and mainstem Grays River and is now occupied solely by the West Fork. Future aggradation in this reach could increase the probability of an avulsion through the downstream meander (to the photo right), where levees have limited channel migration and isolated floodplain surfaces were converted to agriculture.

Appendix C

Historical Sediment Yield

Appendix C

Historical Sediment Yield

1942

Subbasin Group	Sediment Yield (tons/year)				Sediment Yield (tons/km^2/year)			
	Mass Wasting	Roads	Creep	Total	Mass Wasting	Roads	Creep	Total
Upper Main Stem	1,148	0.6	1,048	2,196	30	0	27	57
East Fork	1,338	178	1,209	2,724	30	4	27	61
Middle Main Stem	613	61	513	1,186	30	3	25	58
South Fork	1,560	213	1,244	3,017	30	4	24	58
Lower Main Stem	461	5	336	802	30	0	22	52
West Fork	1,278	31	1,258	2,566	30	1	30	60
Fossil Creek	535	68	375	977	30	4	21	55
Total Basin	6,932	556	5,982	13,469	30	2	26	58

1953

Subbasin Group	Sediment Yield (tons/year)				Sediment Yield (tons/km^2/year)			
	Mass Wasting	Roads	Creep	Total	Mass Wasting	Roads	Creep	Total
Upper Main Stem	1,148	123.1	1,048	2,319	30	3	27	61
East Fork	1,338	113	1,209	2,660	30	3	27	60
Middle Main Stem	613	49	513	1,175	30	2	25	57
South Fork	1,701	623	1,244	3,568	33	12	24	69
Lower Main Stem	996	515	336	1,847	65	33	22	120
West Fork	1,278	198	1,258	2,734	30	5	30	64
Fossil Creek	570	117	375	1,061	32	7	21	59
Total Basin	7,643	1,739	5,982	15,363	33	8	26	66

1964

Subbasin Group	Sediment Yield (tons/year)				Sediment Yield (tons/km^2/year)			
	Mass Wasting	Roads	Creep	Total	Mass Wasting	Roads	Creep	Total
Upper Main Stem	1,148	259.0	1,048	2,455	30	7	27	64
East Fork	1,669	779	1,209	3,656	37	17	27	82
Middle Main Stem	613	735	513	1,860	30	36	25	91
South Fork	1,701	1,185	1,244	4,130	33	23	24	79
Lower Main Stem	1,029	597	336	1,963	67	39	22	128
West Fork	1,367	503	1,258	3,128	32	12	30	73
Fossil Creek	578	128	375	1,081	32	7	21	61
Total Basin	8,105	4,186	5,982	18,273	35	18	26	79

1976

Subbasin Group	Sediment Yield (tons/year)				Sediment Yield (tons/km^2/year)			
	Mass Wasting	Roads	Creep	Total	Mass Wasting	Roads	Creep	Total
Upper Main Stem	25,222	939.5	1,048	27,210	659	25	27	711
East Fork	14,269	1,735	1,209	17,212	320	39	27	386
Middle Main Stem	5,635	964	513	7,111	276	47	25	348
South Fork	11,050	1,656	1,244	13,950	212	32	24	268
Lower Main Stem	1,658	640	336	2,635	108	42	22	171
West Fork	14,499	809	1,258	16,565	340	19	30	389
Fossil Creek	1,367	244	375	1,985	77	14	21	111
Total Basin	73,699	6,986	5,982	86,668	319	30	26	375

1983

Subbasin Group	Sediment Yield (tons/year)				Sediment Yield (tons/km^2/year)			
	Mass Wasting	Roads	Creep	Total	Mass Wasting	Roads	Creep	Total
Upper Main Stem	45,139	1,379.3	1,048	47,566	1,179	36	27	1,242
East Fork	15,605	1,811	1,209	18,625	350	41	27	417
Middle Main Stem	11,176	1,133	513	12,822	547	55	25	627
South Fork	17,154	1,814	1,244	20,213	330	35	24	389
Lower Main Stem	3,884	782	336	5,002	252	51	22	325
West Fork	41,595	1,328	1,258	44,180	976	31	30	1,037
Fossil Creek	1,849	311	375	2,535	104	17	21	142
Total Basin	136,403	8,559	5,982	150,943	590	37	26	653

1990

Subbasin Group	Sediment Yield (tons/year)				Sediment Yield (tons/km^2/year)			
	Mass Wasting	Roads	Creep	Total	Mass Wasting	Roads	Creep	Total
Upper Main Stem	48,656	1,428.8	1,048	51,133	1,271	37	27	1,336
East Fork	15,605	1,811	1,209	18,625	350	41	27	417
Middle Main Stem	12,889	1,242	513	14,643	631	61	25	717
South Fork	18,096	1,846	1,244	21,186	348	35	24	407
Lower Main Stem	4,655	842	336	5,833	303	55	22	379
West Fork	47,561	1,409	1,258	50,227	1,116	33	30	1,179
Fossil Creek	3,240	459	375	4,073	182	26	21	228
Total Basin	150,700	9,038	5,982	165,720	652	39	26	717

1996

Subbasin Group	Sediment Yield (tons/year)				Sediment Yield (tons/km^2/year)			
	Mass Wasting	Roads	Creep	Total	Mass Wasting	Roads	Creep	Total
Upper Main Stem	50,722	1,449.4	1,048	53,220	1,325	38	27	1,390
East Fork	15,605	1,811	1,209	18,625	350	41	27	417
Middle Main Stem	13,727	1,256	513	15,496	672	61	25	758
South Fork	23,945	1,914	1,244	27,103	460	37	24	521
Lower Main Stem	6,289	875	336	7,501	409	57	22	488
West Fork	56,220	1,472	1,258	58,950	1,319	35	30	1,383
Fossil Creek	6,683	495	375	7,553	375	28	21	424
Total Basin	173,192	9,274	5,982	188,447	749	40	26	815

2003

Subbasin Group	Sediment Yield (tons/year)				Sediment Yield (tons/km^2/year)			
	Mass Wasting	Roads	Creep	Total	Mass Wasting	Roads	Creep	Total
Upper Main Stem	40,722	1,471.5	1,048	43,241	1,064	38	27	1,129
East Fork	17,259	1,820	1,209	20,288	387	41	27	455
Middle Main Stem	9,929	1,273	513	11,715	486	62	25	573
South Fork	54,591	1,975	1,244	57,811	1,049	38	24	1,111
Lower Main Stem	25,814	902	336	27,053	1,678	59	22	1,759
West Fork	86,388	1,569	1,258	89,215	2,027	37	30	2,093
Fossil Creek	3,782	503	375	4,660	212	28	21	261
Total Basin	238,487	9,515	5,982	253,983	1,032	41	26	1,099

Total Sediment Yield (tons/year)

Subbasin Group	1942	1953	1964	1976	1983	1990	1996	2003
Upper Main Stem	2,196	2,319	2,455	27,210	47,566	51,133	53,220	43,241
East Fork	2,724	2,660	3,656	17,212	18,625	18,625	18,625	20,288
Middle Main Stem	1,186	1,175	1,860	7,111	12,822	14,643	15,496	11,715
South Fork	3,017	3,568	4,130	13,950	20,213	21,186	27,103	57,811
Lower Main Stem	802	1,847	1,963	2,635	5,002	5,833	7,501	27,053
West Fork	2,566	2,734	3,128	16,565	44,180	50,227	58,950	89,215
Fossil Creek	977	1,061	1,081	1,985	2,535	4,073	7,553	4,660
Total Basin	13,469	15,363	18,273	86,668	150,943	165,720	188,447	253,983

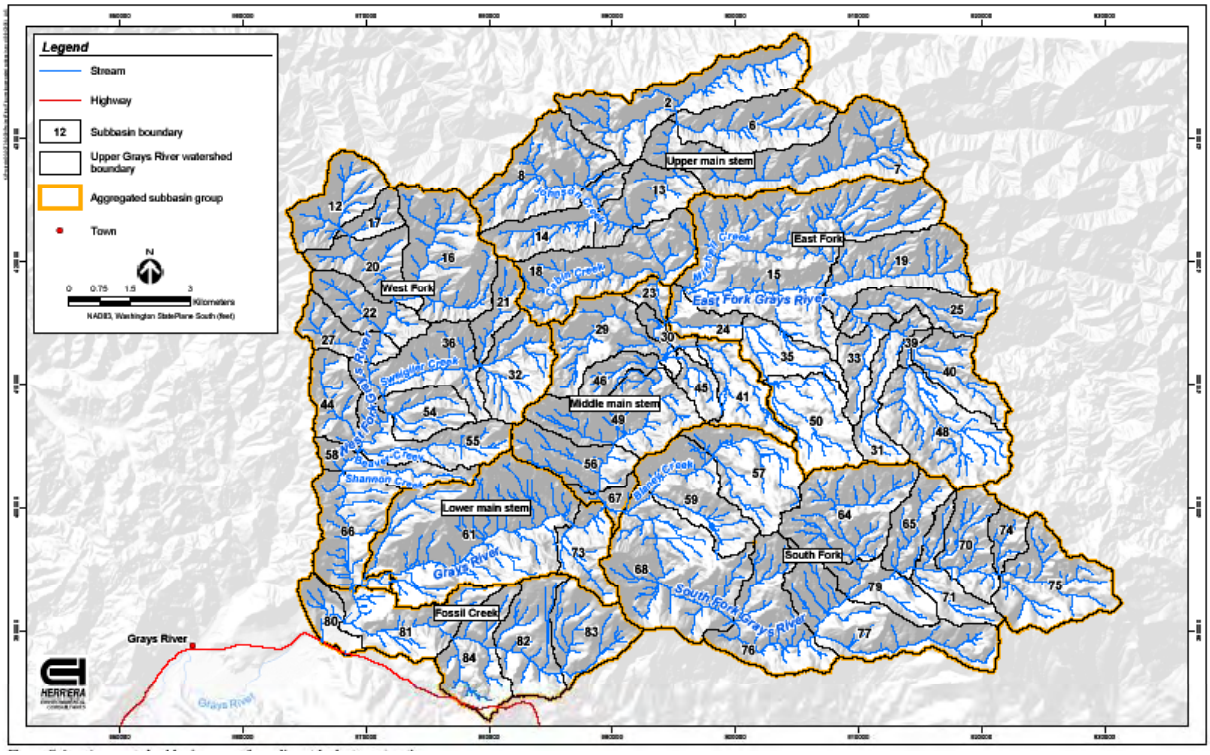

Figure C.1. Aggregated Subbasin Groups for Sediment Budget Construction

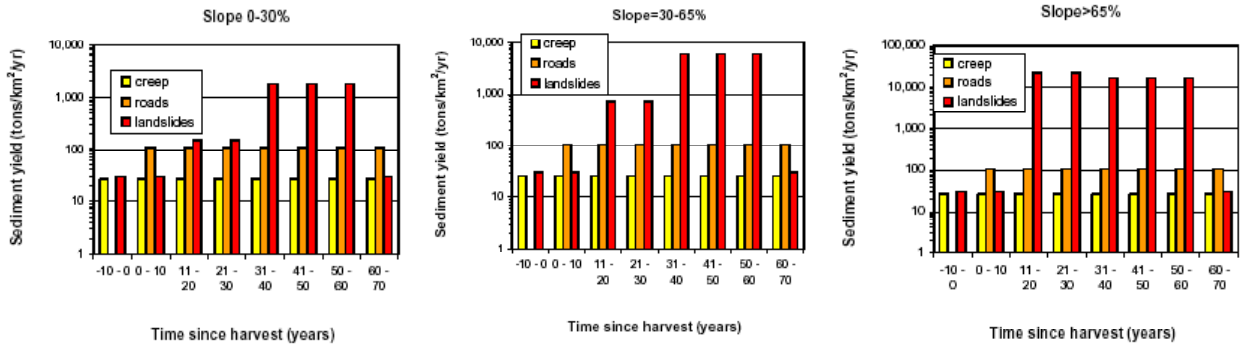

Figure C.2. Reconstructed Sediment Yield from Soil Creep, Roads, and Landslides as a Function of Time Since Last Harvest. The three plots illustrate the range in sediment yield for the three slope categories.

Appendix D

Landslide Inventory

Appendix D

Landslide Inventory

The table and figure presented in this appendix were provided by Herrera Environmental Consultants, Inc., Seattle, Washington.

Landslide ID	Type	Subbasin	Area (m^2)	Estimated Volume (m^3)	Road Intersection	Year Last Harvested	Mass Wasting Map Unit
0	rot	12	9,705	902	N	1976	223
1	df	12	22,118	2,231	Y	1990	223
2	t	12	7,651	900	N	1990	223
3	t	12	12,631	1,592	Y	1990	223
4	t	12	15,554	1,681	Y	1990	223
5	df	12	60	270	N	1990	223
6	t	12	26,484	3,081	Y	1983	223
7	t	12	23,233	2,158	Y	1983	223
8	rot	12	4,883	454	N	1990	223
9	t	12	24,941	2,366	Y	1976	223
10	df	12	6,317	587	Y	1976	223
11	df	12	14,073	1,625	Y	1976	223
12	df	12	13,033	1,211	Y	1976	223
13	t	12	19,572	1,900	Y	1976	223
14	t	12	41,022	6,174	Y	1976	223
15	df	17	9,811	1,024	Y	1983	223
16	t	20	29,117	2,779	Y	1976	223
17	rot	20	2,877	267	N	1983	223
18	t	66	6,508	616	Y	1990	122
19	t	61	5,436	2,378	N	1983	121
20	t	16	21,260	5,427	Y	1983	123
21	t	16	18,418	2,532	Y	1000	143
22	t	16	7,947	738	Y	1000	143
23	t	16	9,586	1,794	Y	1990	123
24	t	16	3,192	301	Y	1990	123
25	t	20	13,102	1,444	Y	1976	223
26	t	22	468	2,034	N	1983	223
27	df	61	1,570	1,480	N	1990	121
28	t	81	2,001	768	N	1983	121
29	t	81	6	1,255	N	1976	121
30	t	81	1,311	231	N	1983	122
31	df	16	4,414	790	Y	1976	123
32	t	16	9,191	1,688	N	1976	123
33	t	16	7,519	699	N	1983	123

df = Debris flow.
rot = Rotational slide.
t = Translational slide.

Landslide ID	Type	Subbasin	Area (m^2)	Estimated Volume (m^3)	Road Intersection	Year Last Harvested	Mass Wasting Map Unit
34	t	16	12,647	1,177	N	1983	123
35	t	16	7,145	744	N	1983	123
36	t	16	8,708	809	N	1983	223
37	t	16	5,832	542	N	1983	223
38	df	16	9,003	879	N	1983	223
39	t	16	23,103	2,242	N	1983	223
40	t	16	10,636	992	N	1983	223
41	df	16	32	735	N	1976	223
42	t	16	17,982	1,672	N	1976	223
43	df	16	15,345	1,434	N	1976	223
44	t	16	3,237	325	Y	1976	223
45	df	16	1,685	287	Y	1976	223
46	t	16	54,143	5,745	Y	1976	223
47	df	16	61,745	6,050	Y	1976	223
48	df	16	39,090	4,348	Y	1976	223
49	df	16	11,234	1,044	N	1976	223
50	t	16	5,011	1,301	Y	1976	223
51	t	16	21,070	1,957	N	1983	223
52	t	16	3,861	359	N	1983	223
53	df	16	12,036	1,189	Y	1983	223
54	t	16	90,533	13,110	Y	1983	223
55	df	36	2,203	1,042	N	1983	222
56	df	54	25,300	2,482	Y	1976	223
61	df	70	3,515	4,623	N	1976	222
62	t	70	57,056	6,269	N	1976	222
63	df	7	2,173	5,458	Y	1976	122
64	t	7	31,373	3,019	Y	1983	123
65	df	7	46,610	4,517	Y	1983	123
66	df	7	43,780	4,554	Y	1983	123
67	t	7	10,841	1,008	N	1983	123
68	t	7	8,783	816	N	1976	223
69	t	7	15,467	1,479	N	1983	223
70	t	7	35,109	3,262	N	1983	223
71	t	7	47,229	4,388	Y	1983	223
72	t	7	35,086	3,260	Y	1983	223
73	t	7	5,292	566	Y	1983	223
74	df	6	21,322	2,679	Y	1976	123
75	df	18	19,830	1,879	Y	1983	123
76	t	12	4,003	603	N	1983	223

df = Debris flow.
rot = Rotational slide.
t = Translational slide.

Landslide ID	Type	Subbasin	Area (m^2)	Estimated Volume (m^3)	Road Intersection	Year Last Harvested	Mass Wasting Map Unit
77	df	12	9,930	923	N	1983	223
78	df	12	24,351	2,338	N	1983	223
79	t	12	18,949	2,449	N	1983	222
80	df	22	9,212	2,967	Y	1983	223
81	rot	44	24,169	2,426	N	1942	132
82	df	44	806	4,956	Y	1964	132
83	df	66	331	1,631	N	1964	132
84	t	66	16,219	3,084	N	1942	132
85	t	12	221	3,339	Y	1976	223
86	t	17	7,799	1,368	Y	1983	223
87	df	17	568	1,099	N	1990	223
88	df	20	21,748	3,259	Y	1983	223
89	df	20	6,353	920	Y	1983	223
90	t	20	27,517	4,683	Y	1983	223
91	df	36	13,923	3,683	Y	1983	222
92	rot	54	9,073	90,385	N	1942	131
93	rot	16	25,518	2,604	N	1983	223
94	rot	16	51,899	5,140	N	1983	223
95	t	16	27,104	3,327	Y	1990	223
96	df	16	15,091	1,859	Y	1990	223
97	t	16	9,483	1,896	Y	1990	223
98	rot	16	60,784	5,647	N	1983	223
99	df	16	30,134	4,335	Y	1976	223
100	df	16	54,431	5,995	Y	1983	223
101	rot	61	69,791	34,276	N	1953	131
102	df	16	47,245	4,389	Y	1976	223
103	df	16	22,455	2,086	N	1976	223
107	df	16	9,942	1,051	N	1983	223
108	df	16	174	3,326	Y	1983	222
111	df	16	12,681	2,349	Y	1983	223
112	df	16	12,088	1,688	Y	1983	223
113	df	16	1,894	5,010	Y	1983	223
114	df	16	8,517	2,895	Y	1976	223
115	t	16	12,294	1,451	Y	1983	223
119	t	61	6,445	9,057	Y	1983	222
123	t	84	105	1,647	N	1976	122
124	rot	8	45	1,344	Y	1953	132
125	t	8	21,061	2,907	Y	1976	123
126	rot	14	20,296	1,886	N	1976	223

df = Debris flow.
rot = Rotational slide.
t = Translational slide.

Landslide ID	Type	Subbasin	Area (m^2)	Estimated Volume (m^3)	Road Intersection	Year Last Harvested	Mass Wasting Map Unit
127	t	14	9,316	928	N	1976	223
128	t	32	12,371	1,205	N	1964	132
136	df	61	62,176	12,415	N	1964	232
137	t	61	61,030	5,741	N	1964	233
142	df	82	18,270	2,429	N	1990	122
145	df	8	20,087	3,502	Y	1976	222
146	df	8	7,313	1,741	Y	1976	222
147	t	18	4,765	3,939	Y	1976	122
148	df	18	27,167	3,490	Y	1976	123
149	df	18	12,567	1,167	Y	1976	123
150	df	18	8,563	1,214	N	1983	123
151	df	18	36,204	5,270	Y	1976	122
152	df	83	3,590	1,109	Y	1990	121
153	df	82	19,350	3,200	Y	1990	121
154	t	18	39,976	4,125	Y	1983	123
155	t	18	20,080	1,867	Y	1983	222
161	t	49	26,919	2,743	N	1976	223
162	df	49	24,208	12,194	Y	1976	221
163	df	49	1,470	3,801	Y	1983	222
164	rot	49	27,782	5,950	Y	1983	222
165	df	56	8,680	5,022	N	1964	231
166	t	56	227	90,793	N	1964	231
167	rot	67	12,815	3,765	N	1983	222
168	t	67	40	744	N	1942	231
170	t	67	212	300	N	1983	223
174	t	83	38,429	3,570	Y	1990	121
175	t	83	17,283	2,952	Y	1990	121
176	t	2	7,679	3,826	Y	1976	222
178	df	2	163	1,451	Y	1976	222
179	df	2	13,568	1,964	Y	1976	222
180	df	7	18,391	2,092	N	1976	223
181	df	7	15,783	1,816	Y	1976	223
182	t	7	20,464	1,901	N	1983	223
183	df	7	7,912	742	N	1983	223
184	df	13	958	5,340	N	1976	122
185	rot	13	17,558	2,024	N	1990	223
186	df	13	7,038	1,744	Y	1990	223
187	df	18	9,213	1,045	Y	1983	223
188	df	18	10,402	2,422	Y	1983	222

df = Debris flow.
rot = Rotational slide.
t = Translational slide.

Landslide ID	Type	Subbasin	Area (m^2)	Estimated Volume (m^3)	Road Intersection	Year Last Harvested	Mass Wasting Map Unit
191	t	30	11,170	1,346	N	1983	122
198	t	68	7,488	7,466	N	1983	222
199	t	68	91,842	24,982	Y	1983	222
203	t	2	7,532	3,681	Y	1976	222
204	t	2	45,575	4,411	Y	1976	223
205	df	2	24,877	3,787	Y	1976	223
206	t	7	16,039	2,646	Y	1976	223
208	rot	45	15,005	2,638	N	1976	122
211	df	59	26,118	4,758	Y	1983	222
212	df	59	21,565	3,285	Y	1983	222
213	df	6	8,773	2,089	Y	1983	123
214	df	6	14,358	1,874	Y	1983	123
215	df	6	8,119	1,646	Y	1983	123
216	df	6	19,209	1,976	Y	1983	122
217	df	6	548	2,158	N	1976	122
218	df	6	1,698	1,398	Y	1983	122
219	t	7	9,691	1,195	Y	1976	223
220	df	7	37,410	4,102	Y	1983	223
221	df	7	27,341	2,901	Y	1983	223
222	df	7	4,800	458	Y	1983	223
223	df	7	7,191	773	Y	1983	223
224	df	7	6,361	658	Y	1983	223
225	t	15	3,337	1,423	Y	1976	122
226	t	15	4,668	967	Y	1976	122
227	rot	15	6,415	1,931	N	1976	122
228	rot	15	1,653	867	N	1976	121
229	t	15	2,762	1,479	Y	1976	121
230	df	15	515	1,267	Y	1976	121
231	df	15	12,747	1,907	Y	1976	122
233	df	76	30,920	2,934	Y	1964	231
234	df	6	13,592	2,613	Y	1976	122
235	rot	6	7,717	1,182	N	1976	223
236	df	6	10,705	1,170	Y	1983	222
237	df	6	4,903	5,868	Y	1976	222
238	df	6	5,878	1,463	Y	1983	222
239	rot	6	3,812	3,032	N	1983	222
240	df	6	18,981	3,853	Y	1983	222
241	t	7	9,339	1,093	N	1983	223
242	t	7	1,256	3,154	Y	1976	222

df = Debris flow.
rot = Rotational slide
t = Translational slide.

Landslide ID	Type	Subbasin	Area (m^2)	Estimated Volume (m^3)	Road Intersection	Year Last Harvested	Mass Wasting Map Unit
244	df	15	401	3,320	Y	1976	121
245	df	15	24,007	2,357	Y	1976	221
246	rot	15	2,310	1,449	N	1976	221
248	rot	7	27,057	2,516	Y	1983	223
249	rot	7	2,951	1,786	Y	1976	223
250	t	7	28,240	4,844	Y	1976	223
251	rot	7	6,244	2,327	Y	1976	222
252	t	15	3,125	1,150	Y	1976	222
253	df	15	655	1,229	N	1976	222
254	df	15	632	2,029	Y	1976	222
256	df	15	33,794	6,153	Y	1976	221
257	df	19	10,367	1,174	Y	1976	121
258	rot	19	4,632	2,750	Y	1976	122
260	rot	64	15,303	1,620	N	1976	222
261	df	7	10,776	2,107	Y	1983	222
262	t	15	25,040	4,374	N	1976	222
263	df	33	19,040	2,318	Y	1964	232
264	df	33	17,178	2,629	Y	1964	232
265	t	15	4,809	1,759	Y	1976	222
266	t	15	26,446	10,423	Y	1976	222
267	df	19	81,239	14,036	Y	1964	232
268	df	19	19,605	3,292	Y	1976	222
269	t	25	7,103	1,329	Y	1976	123
272	rot	61	91	10,315	N	1942	231
273	df	20	13,386	2,351	Y	1976	223
274	t	59	126,670	98,435	Y	1976	222

df = Debris flow.
rot = Rotational slide.
t = Translational slide.

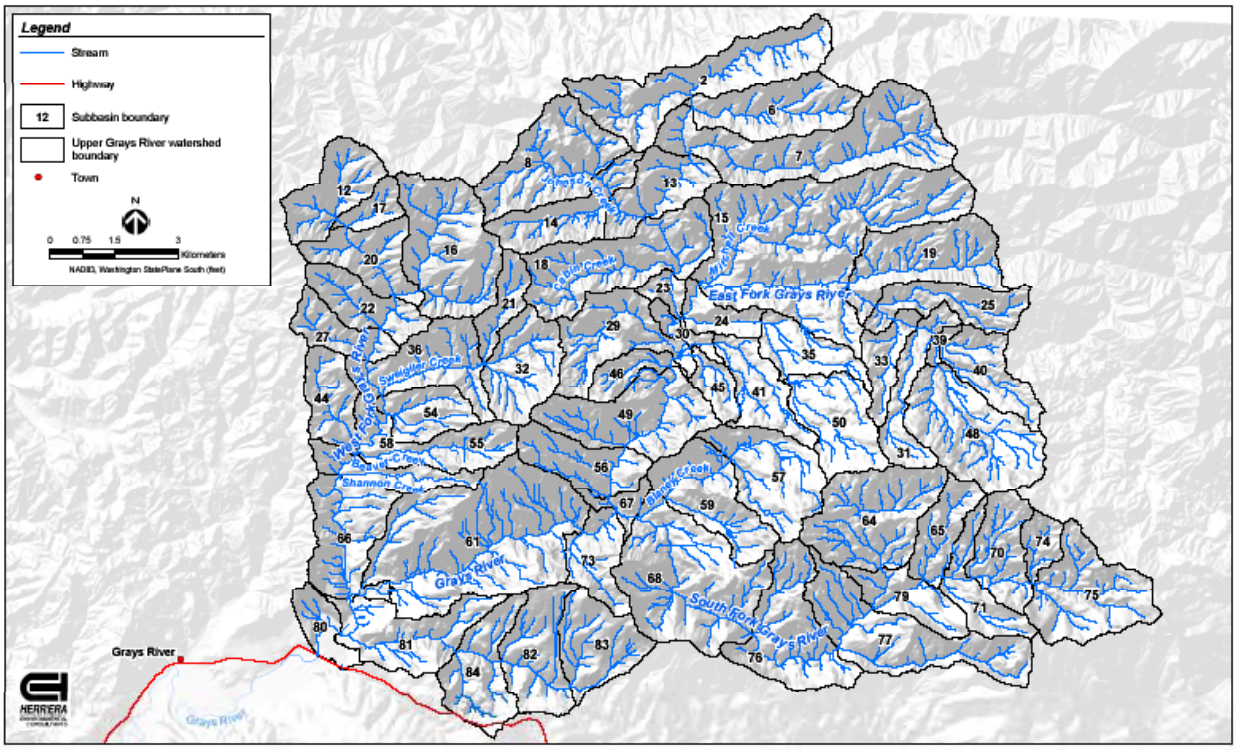

Figure D.1. Subbasins Used for Analysis of Mass Wasting and Road Surface Erosion Sediment Production

Appendix E

Status of Fish Populations and Habitat in the Grays River

Appendix E

Status of Fish Populations and Habitat in the Grays River

The watershed management plan for the Grays/Chinook River subbasin (NPCC 2004) includes all essential information recommended by the Joint Natural Resources Cabinet (2001) for such assessments. The plan details the distribution, life history, habitats, and population trends for focal species and other species of interest in the Grays River subbasin.

Salmonids identified as focal species occurring in the Grays River subbasin include chum, fall Chinook, and coho salmon and winter steelhead (Table E.1). Under the Endangered Species Act (ESA), focal species are those listed or proposed for listing or that are of concern because of a high probability of their extinction. These species have been identified as priority species for recovery during the Columbia River basin planning process (NPCC 2004). Coastal cutthroat trout and Pacific lamprey are also included in the Grays River subbasin plan because habitat degradation affecting focal species will likely also affect these species, and coastal cutthroat trout were proposed for ESA listing as threatened in 1999 (Roler et al. 2002).

Table E.1. Status of Focal Salmonid and Steelhead Populations in the Grays River Subbasin

Focal Species	ESA Status	Hatchery Component[(a)]	Historical Numbers[(b)]	Recent Numbers[(c)]	Current Viability[(d)]	Extinction Risk[(e)]
Fall Chinook	Threatened	No	1,500 – 10,000	100-300	Low+	~40%
Winter steelhead	Not Listed	Yes	4,500	400-600	Low+	~30%
Chum	Threatened	Yes	8,000 – 14,000	500-10,000	Low+	~30%
Coho	Proposed	Yes	5,000 – 40,000	Unknown	Low	~70%

(a) Significant numbers of hatchery fish are released in the subbasin.
(b) Historical population size inferred from presumed habitat conditions using Ecosystem Diagnosis and Treatment Model and NOAA rough calculations.
(c) Approximate current annual range in number of naturally produced fish returning to the subbasin.
(d) Prospects for long-term persistence based on criteria developed by the NOAA Technical Recovery Team.
(e) Probability of extinction within 100 years corresponding to estimated viability.
Source: NPPC (2004).

The Grays River subbasin plan also presents detailed information regarding the condition of stream channels and aquatic habitats. Habitat assessments summarized in the plan include the Integrated Watershed Assessment (IWA) for Water Resource Inventory Area (WRIA) 25, Ecosystem Diagnosis and Treatment (EDT) modeling of the Grays River subbasin, and habitat factors limiting analyses. Current conditions of nine specific habitat features are discussed: watershed hydrology, passage obstructions, water quality, key habitat availability, substrate and sediment, woody debris, channel stability, riparian function, and floodplain function. The IWA is a screening and prioritization tool based on geographic information systems (GIS) that identifies areas for preservation and restoration of key habitats. Analyses for the IWA are focused on the three primary aquatic habitat-forming processes—hydrology, sedimentation, and large woody debris recruitment potential. EDT model inputs are basic reach-scale stream condition and fish life history data. The EDT model evaluates aquatic habitat condition and outputs fish population information including stage-specific productivity and capacity and diversity estimates. The

subbasin plan presents three EDT analyses—a population analysis, a reach analysis, and a habitat factor analysis—that are broadest to finest in scope, respectively. Comparison of historic, current, and intermediate "properly functioning condition" model runs allows for reach-scale prioritization of restoration and management efforts based on extent of degradation and potential production contribution to the population. Reaches are ranked as high, medium, or low priority based on importance of restoration/preservation actions to population recovery. Reaches are categorized as preservation, restoration, or both, based on the nature of actions producing the greatest population increase. The IWA and EDT processes are conceptually linked in that IWA outputs and EDT inputs are estimates of stream condition. Consistency between the two processes is analyzed in the subbasin plan to evaluate correlation between them. Findings from these assessments presented in the Grays River subbasin plan are summarized here, with the exception of IWA findings for the upper Grays watershed. Other sources of information are acknowledged as appropriate.

EDT modeling has been completed for chum, fall Chinook, and coho salmon and winter steelhead in the Grays River. Fewer than 20% of stream condition ratings in the Grays River were based on empirical data; the remainder were estimated from extrapolation from other watersheds, expansion of Grays River data from another reach, expert opinion, or hypothetical information. Data limitations or absences identified during the EDT process for the Grays River include riparian condition, sediment delivery, bed scour, habitat surveys, macroinvertebrate sampling, and nature of obstructions.

E.1 Status of Salmonids in the Grays River Subbasin

E.1.1 Chum Salmon

Distribution – Spawning occurs in the Grays River from RM 9.5 to RM 13.0, from the Grays River Hatchery (RM 1.4) downstream to the confluence with the Grays River, and the lower 0.5 mile of Crazy Johnson Creek (Figure E.1).

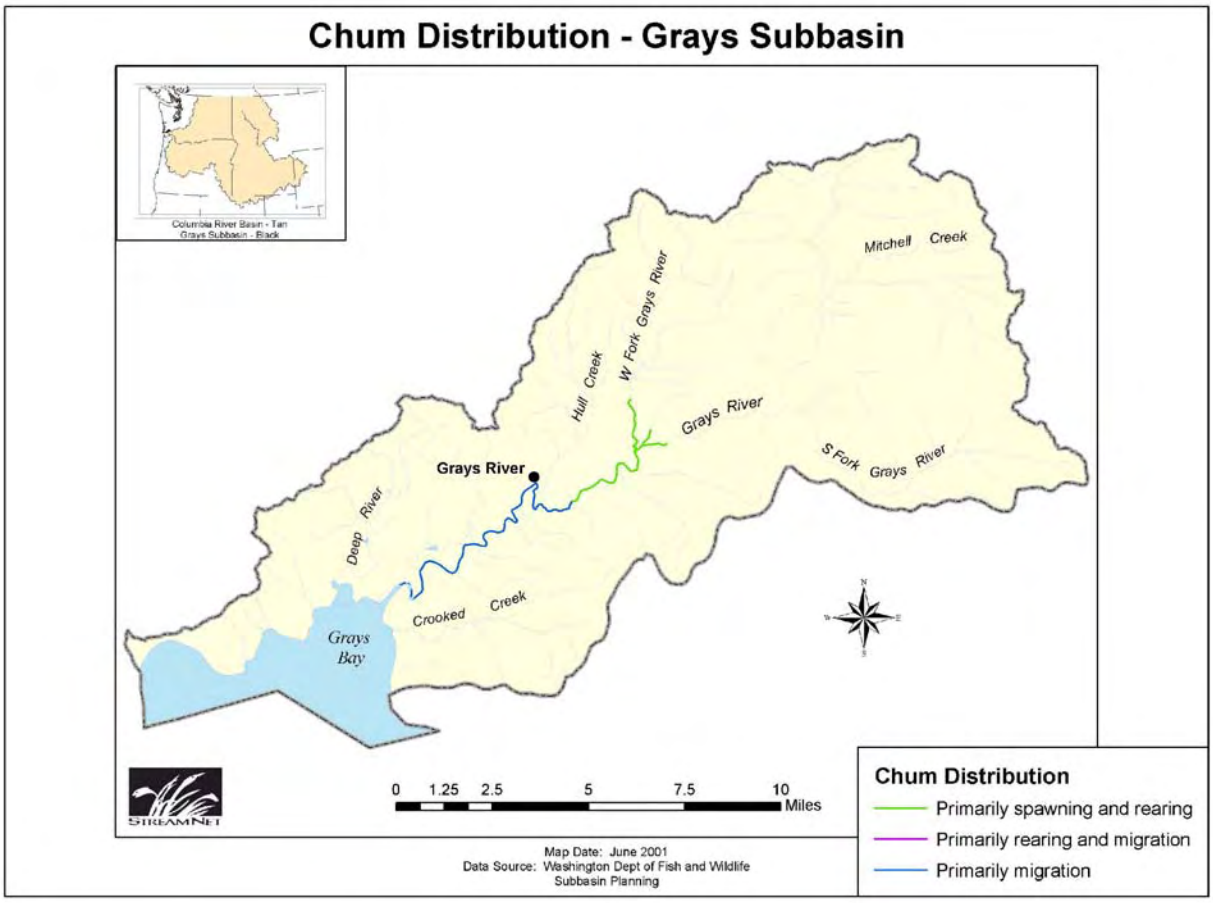

Figure E.1. Distribution of Chum Salmon in the Grays River and Tributaries (StreamNet 2001)

Life History and Habitats – Adults enter the Grays River from mid October through November. Spawning occurs through mid December. Juveniles emerge in early spring and migrate soon after emergence (March through May).

Population Trend – Historic run size in the Grays/Chinook River population ranged from 8,000 to 14,000 fish, with most spawning occurring in the Grays River (NPCC 2004). Most production is from natural spawning. Supplementation from the Grays River Hatchery contributes a small portion of production annually. The Washington Department of Fish and Wildlife (WDFW) salmonid stock inventory (SaSI) process categorized chum salmon in the Grays River as depressed (WDFW 2002). Recent population estimates indicate a growing population during 1999–2004, although return numbers in 2005 were low.

E.1.2 Fall Chinook Salmon

Distribution – Spawning occurs in the West Fork from the Grays River Hatchery (RM 1.4) downstream to the confluence with the Grays River, and in the Grays River above tidal influence to the confluence with the West Fork (RM 8-14; Figure E.2). Fall Chinook salmon occur in the mainstem Grays River to approximately the confluence of the East Fork Grays River (Figure E.2).

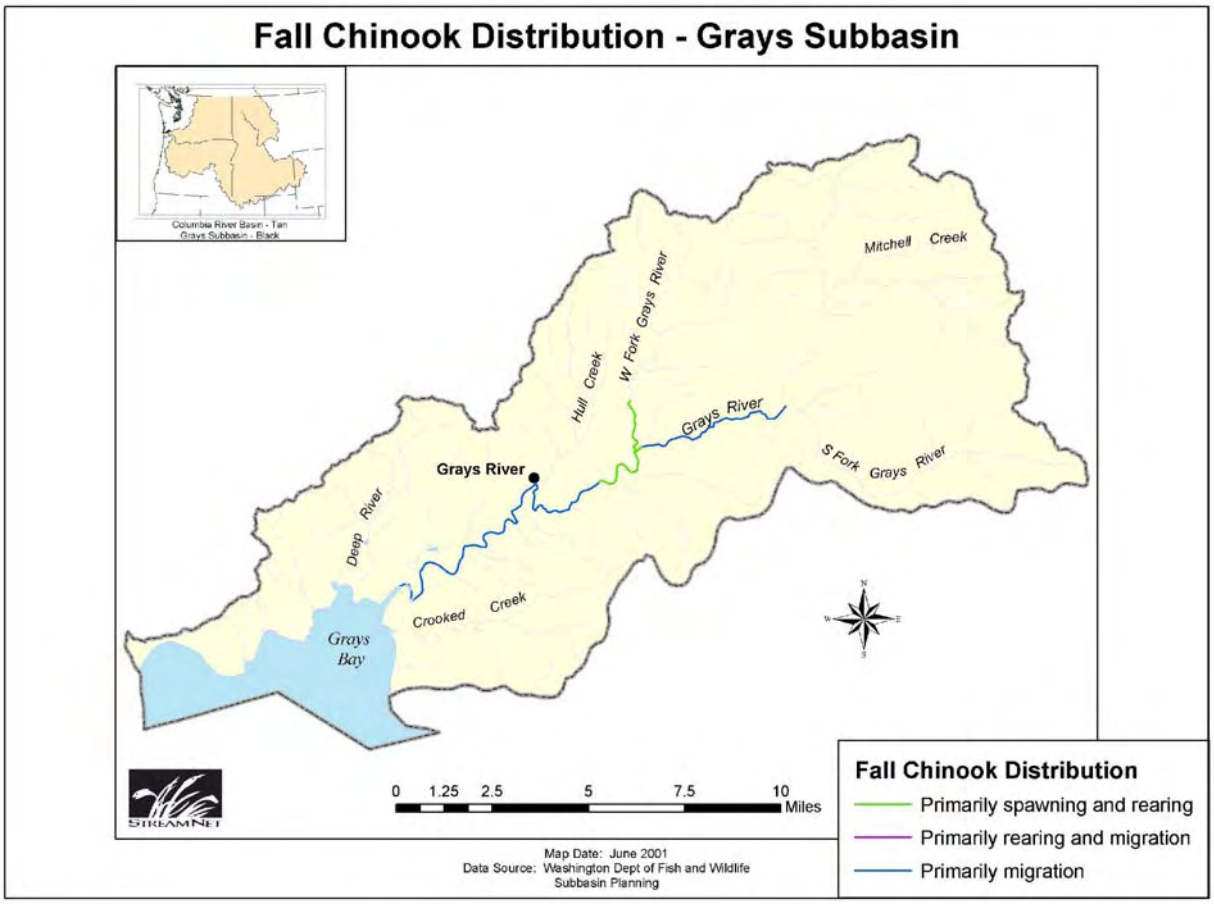

Figure E.2. Distribution of Fall Chinook Salmon in the Grays River and Tributaries (StreamNet 2001)

Life History and Habitats – Spawning individuals return to the Grays River and West Fork in mid August to mid September. Spawning occurs during September and October. Juveniles emerge in early spring, rear in shallow-water habitats adjacent to and downstream from spawning areas, and migrate to the ocean in late spring to summer of their first year.

Population Trend – The population has declined from a historic level of 1,500 to 10,000 fish to 100 to 300 current spawning individuals. The WDFW SaSI process categorized this population as depressed (WDFW 2002). Few natural fall Chinook salmon juveniles are thought to be produced annually. Hatchery supplementation was discontinued in 1998.

E.1.3 Coho Salmon

Distribution – Potential spawning areas include the upper Grays, South Fork, West Fork, Crazy Johnson Creek, and Hull Creek (Figure E.3).

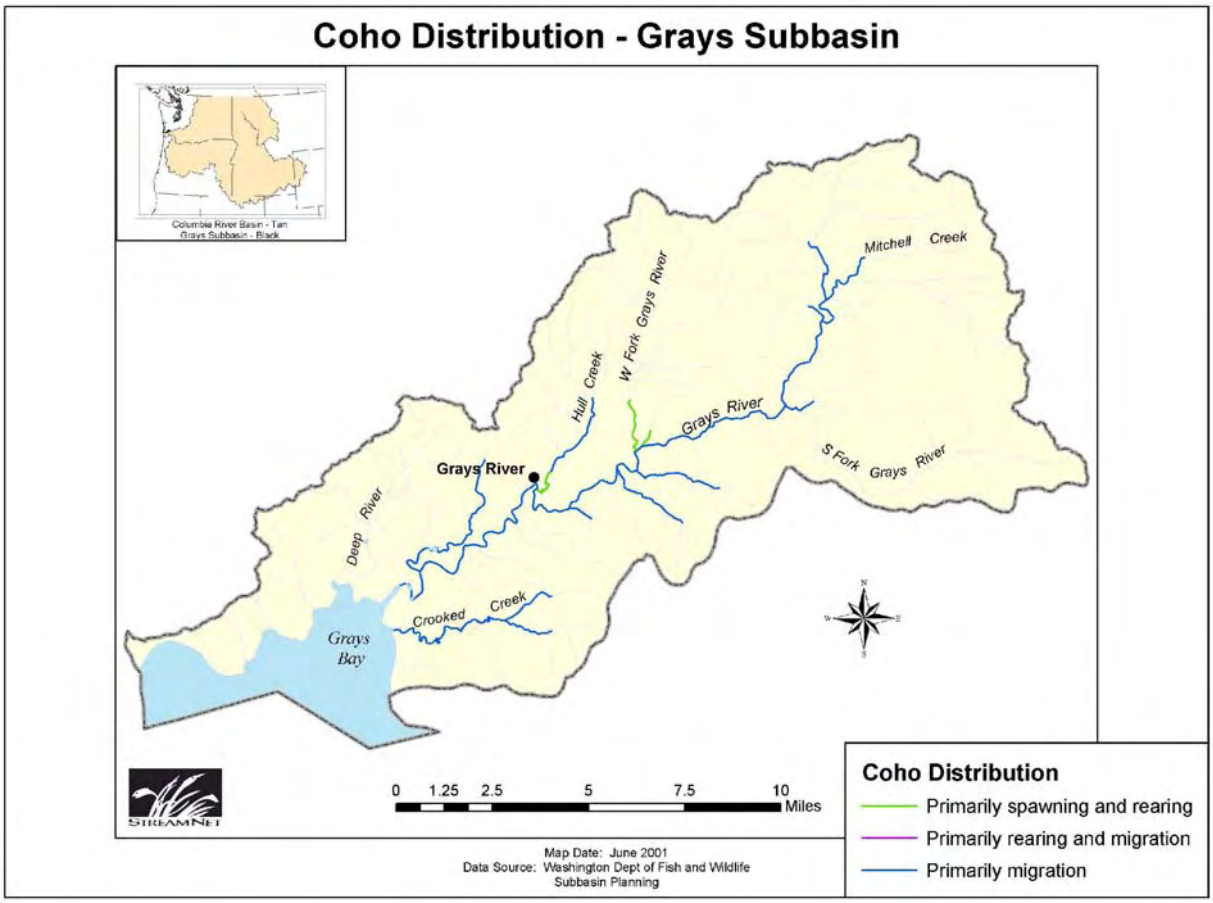

Figure E.3. Distribution of Coho Salmon in the Grays River and Tributaries (StreamNet 2001)

Life History and Habitats – Two stocks, late (type N) and early (type S) occur in the subbasin. Spawning occurs primarily in the upper mainstem and large tributaries throughout the upper watershed. Spawners enter the Grays River from mid August through February, and spawning occurs from late October through late November. Juvenile rearing occurs upstream and downstream of spawning areas. Juveniles rear for one year before migrating to the ocean in the spring.

Population Trend – The current coho run is a fraction of its historic run size; current returns are unknown but presumed to be low. Some hatchery-produced fish spawn naturally. The Grays River Hatchery releases 150,000 coho annually; hatchery production accounts for most of the coho returning to the basin. The WDFW SaSI process categorized this population as unknown (WDFW 2002).

E.1.4 Winter Steelhead

Distribution – Spawning occurs in accessible streams throughout the Grays River subbasin above tidal influence, including the East, West, and South Forks (Figure E.4).

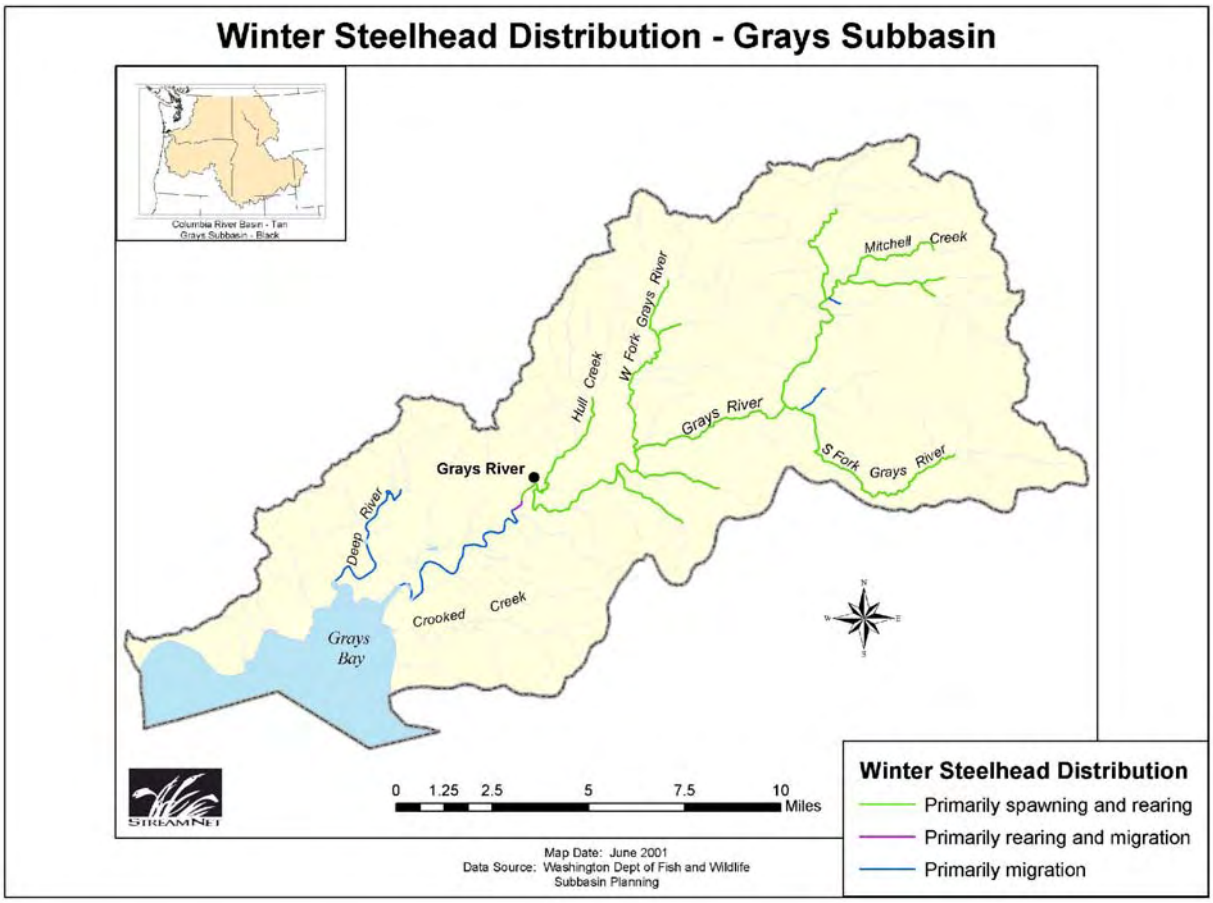

Figure E.4. Distribution of Winter Steelhead in the Grays River and Tributaries (StreamNet 2001)

Life History and Habitats – Spawners enter the Grays River from December through April. Spawning occurs from March through early June. Juveniles emerge in June and July and rear upstream and downstream from spawning areas. Juveniles rear for one to two years before migrant juveniles depart for the ocean in spring.

Population Trend – The population has declined from 2000 fish annually in the 1920s and 1930s to an average of 658 fish during 1991–2001. Although hatchery supplementation has continued since 1957, hatchery fish contribute little to natural winter steelhead production in the Grays River. The WDFW SaSI process categorized this population as depressed (WDFW 2002).

E.1.5 Coastal Cutthroat Trout

Range – Coastal cutthroat trout are present throughout the subbasin, including both anadromous and resident life histories (Figure E.5).

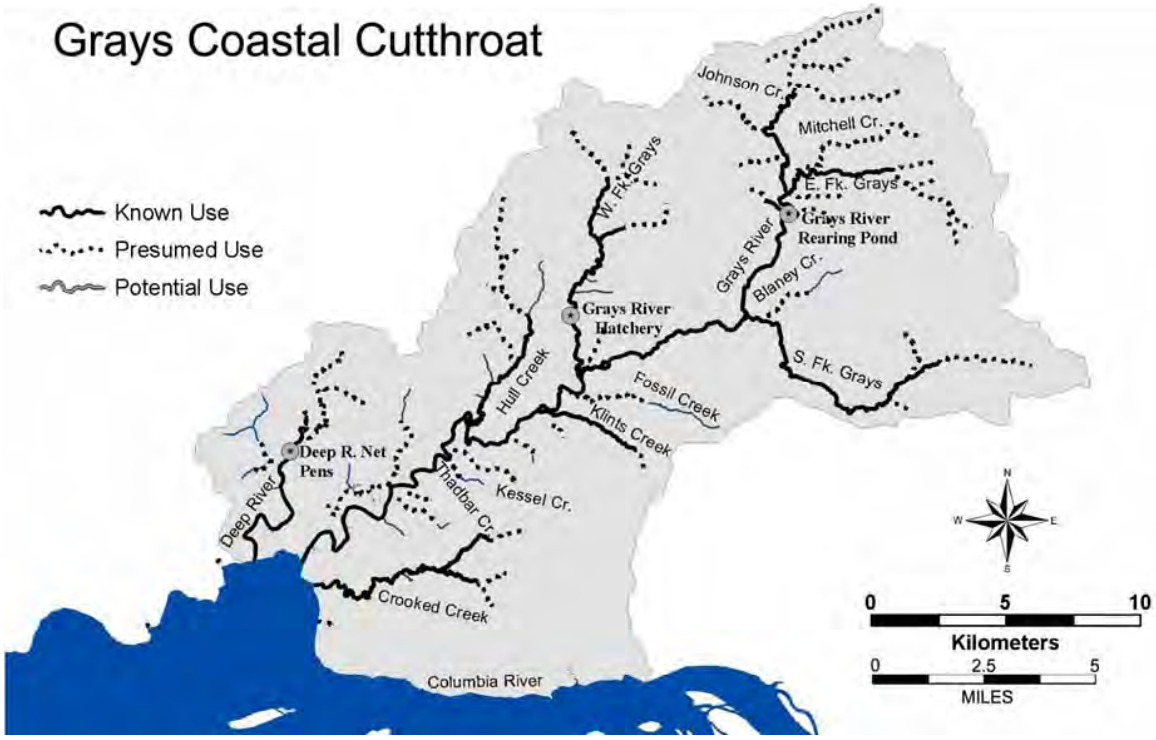

Figure E.5. Distribution of Coastal Cutthroat Trout in the Grays River and Tributaries (NPCC 2004)

Life History and Habitats – Resident, fluvial, and anadromous life histories are present in the subbasin. Migratory fish reside lower in the basin and are not thought to use steep upper tributary reaches. Anadromous spawners enter the Grays River in late July through mid April and spawn during January through April. Spawning timing of fluvial and resident fish is undocumented but believed to be similar to that of the anadromous spawning. Juveniles rear two or three years near spawning areas before migrating from their natal stream.

Population Trend – The coastal cutthroat trout population in the Grays River has not been quantified, but the WDFW SaSI process categorized this population as depressed (WDFW 2000). All production is from natural spawning.

E.2 Status of Salmonid Habitat in the Grays River

E.2.1 Chum and Fall Chinook Salmon Spawning Areas

Chum and fall Chinook salmon spawning areas include the Grays River between the canyon reach and the State Route 4 bridge, the lower West Fork downstream from the Grays River Hatchery, and Crazy Johnson Creek. These reaches are the primary chum and fall Chinook salmon spawning areas within the study area and are used for spawning also by coho salmon. These reaches include hydrologic unit codes

(HUCs) 17080006030403 (Grays River and Crazy Johnson) and 17080006030201 (lower West Fork) and include ETD reaches Grays 2, Gorley Creek 1 and 2, West Fork 1 and 2, and Crazy Johnson Creek. As a result of the 1999 avulsion, the Grays River abandoned Grays River 2C and 2D and shifted into the Gorley 1 and 2 reaches. All reaches in the study area used by chum and/or fall Chinook salmon for spawning are Tier 1 reaches (Grays 2A–2D, West Fork 1, and Crazy Johnson), except Gorley 1 (Tier 2) and Gorley 2 (Tier 4). Tier 1 reaches are those that would yield the greatest benefits toward accomplishing the plan's objectives. The Grays River subbasin plan prioritized EDT reaches based on the watershed plan biological objectives, fish distribution, critical life history stages, current habitat conditions, and potential for fish population recovery (NPCC 2004).

Logging and resultant landslides, erosion, and channel instability in the upper watershed and agricultural modifications of the lower channel have damaged key salmon spawning areas in these reaches (Roler 2002). Roler identified the lack of stable spawning habitat as the primary physical limitation on current chum production. Low seasonal water flows limit access of chum and fall Chinook salmon to spawning grounds in the Grays River and West Fork, confining chum salmon to less stable mainstem areas (Roler 2002). The lower Grays River largely lacks a vegetated riparian corridor. All reaches currently have insufficient riparian and inchannel large woody debris (LWD), contributing to relatively homogeneous aquatic habitat and channel and bank instability (Roler 2002). Few farms adjacent to the lower Grays River have streamside fencing to restrict cattle access from the river. Elevated nitrates and phosphates are presumed, but effects are unknown. The lower watershed experiences elevated water temperatures from June through September (Roler 2002). Lack of key habitats such as side channels and quality pools are the result of agricultural diking and channel straightening. Substrates contain excessive sediments. Large woody debris is all but absent in the Grays River spawning reach and inadequate in the lower West Fork. Areas of bank instability occur in the Grays River spawning reaches associated with cattle access and the 1999 avulsion. Riparian areas are in poor shape associated with cattle access and agricultural practices. Floodplain function has been impaired throughout the Grays River spawning reach from agricultural diking and bank armoring. The 1999 avulsion restored floodplain width within the upper spawning reach.

The Integrated Watershed Analysis (IWA) of Grays River spawning reaches indicates that these reaches are moderately impaired or impaired by local and upstream sediment delivery (Table E.2). In particular, the lower West Fork is impaired by local sediment delivery. Impairment reflects the volume of sediment delivered to stream channels as a function of land use, estimated from road density, slope, and natural erodability data sources, relative to that expected from natural processes alone. Local ratings reflect these data within the immediate watershed; watershed ratings reflect conditions in the immediate watershed as well as all watersheds draining to it. Both the West Fork and Grays River spawning reaches are hydrologically impaired at both the local and watershed levels (Table E.2). Hydrologic function ratings in the IWA were derived from road density, area of impervious surface, subwatershed attributes, and land cover (vegetation type). Local riparian function of both reaches was moderately impaired. Local riparian function is a measure of the functionality of riparian condition and large woody debris recruitment, and affects instream habitat conditions such as channel morphology, substrate, nutrient cycling, temperature, and habitat structural diversity. The local riparian function rating was developed primarily from land cover data within a stream corridor buffer. The forest cover rating (percentage of mature timber) was low/poor in the lower Grays River and moderately low/fair in the lower West Fork. The road density rating was moderately low/fair condition (moderately high density) in both reaches.

Table E.2. Habitat-Limiting Factors for Salmonids in the Grays River Watershed (WCC 2002)

Limiting Factor	Functions and Effects	Location
Access – culverts and low flows	Restrict access to some salmonid habitats	Culverts – various Low flows - Lower West Fork, Grays between covered bridge and the canyon, Grays Bay bar
Floodplain connectivity	Diking, channel hardening, historic splash damming have disconnected main channel from off-channel habitats	Throughout
Side channel availability	See floodplain connectivity. Overall reduction in floodplain complexity. Most channels currently single-thread.	Throughout
Bank erosion/stability	Associated with natural soil instability and historic and current agricultural practices resulting in loss of vegetated riparian corridor	Throughout but in particular West Fork, South Fork
Riparian conditions	Loss of riparian function affects water quality, erosion rates, streambank stability, instream habitat	Riparian conditions considered poor throughout subbasin with isolated areas fair or good.
Large woody debris	LWD critical to pool development, retention of spawning gravels, habitat diversity, salmonid cover	LWD abundance below habitat standards throughout most of subbasin
Pool frequency	Pools serve as critical holding habitat for pre-spawning salmonids, temperature refuge from elevated summer temperatures	Pool frequency below habitat standards through most of subbasin with isolated areas fair or good. Good pool habitat generally associated with good LWD abundance
Water quality	High temperatures are of concern for rearing juveniles and resident fish, may also affect migrating fish in early fall. Turbidity elevated due to mass wasting and bank instability.	Temperature of concern particularly lower in basin. Turbidity of concern in West Fork, South Fork, Muddy Trib
Water quantity	Low flows limit access to habitat. May be associated with accumulated bedload in West Fork and mainstem Grays River.	Mainstem Grays River between covered bridge and canyon, West Fork. The majority of the subbasin is impaired hydrologically, particularly the South Fork.

EDT reaches in the study area that are chum salmon spawning reaches include Grays 2A–D, WF Grays 1, Crazy Johnson, and Gorley Creek 1 and 2. All of these reaches are classified as high priority except Gorley 1 and 2 and the upper portion of West Fork 1. Recovery emphasis for most reaches includes both restoration and preservation. Recovery of Grays 2B–D, lower West Fork 1, and Crazy Johnson produce the largest increases in the chum salmon abundance and productivity in the Grays subbasin. In terms of fall Chinook salmon recovery, Grays 2A and 2C are medium-priority reaches and Grays 2B and the lower West Fork are low-priority reaches. The reach immediately downstream from the study area is the only high-priority reach in the Grays River. Recovery emphasis is on preservation for all reaches, with most changes in abundance and productivity associated with maintaining current levels of habitat quality and avoiding further degradation of high-and medium-priority reaches. Study area reaches in the lower mainstem Grays River and lower West Fork (including Crazy Johnson and Gorley 1 and 2)

are all high-priority recovery reaches for coho salmon. Recovery of Grays 2A–2C and Grays 2 just downstream from the study area produces the largest increase in coho salmon abundance and productivity primarily associated with restoration.

E.2.2 Upper Watershed

The upper watershed includes the Grays River watershed above the lower end of the canyon and the West Fork above the Grays River Hatchery. Effects of habitat degradation in the upper watershed would directly affect winter steelhead and cutthroat trout and indirectly affect chum and fall Chinook salmon through downstream effects of altered hydrology and sediment delivery. Pacific lamprey also may be affected in the upper watershed, although the ability of this species to pass falls on the mainstem is unknown.

The upper Grays River is unstable, prone to mass wasting, and turbid during flood events. Fine sediment is released into flood waters throughout the watershed due to the native mudstone soils, logging activity, and high road density. The South Fork in particular carries an excessive fine sediment load with impacts on the Grays River downstream from the South Fork (Roler 2002). The subbasin plan cites elevated peak flows and inadequate summer low flows as sources of impairment and suggests that excess groundwater withdrawal may be a concern in the future. Passage obstruction due to low summer flows and culverts exists (NPCC 2004). High water temperatures are a concern throughout the subbasin, with the West Fork a Washington State 303(d) listed-stream for impairment due to temperature. Excessive turbidity is also a water quality problem in some reaches of the upper watershed. Nutrient limitation may be a problem due to lower salmonid escapement than historically. Lower reaches of most major Grays River tributaries lack quality pool habitat. Much of the subbasin's channel substrates contain excessive fine sediments associated with high road and road crossing densities and mass failures as well as naturally erosive soils. Large woody debris availability is inadequate in 75% of lower basin reaches surveyed and throughout the upper basin as well (WCC 2002). Where LWD is present, it is often in large logjams or consisting of small-diameter material. Areas of bank instability occur in upper West, East, and South Fork subwatersheds associated with mass failures. Riparian areas are in poor shape throughout the upper basin and in particular in the upper West Fork associated with timber harvest practices.

Fourteen EDT reaches in the study area are high priority for winter steelhead. Recovery emphasis for most is restoration, although for some, restoration and preservation are equally important. High-priority reaches occur in all of the major upper forks (West, East, South) as well as the upper mainstem Grays River. Habitat factor analysis indicates that primary factors affecting winter steelhead include sediment (incubation), temperature (age 0 rearing), and flow and habitat diversity (age 1). Secondary and tertiary factors for these life stages include flow and key habitat, pathogens, predation, and competition, respectively. The relative importance of habitat factors to winter steelhead recovery is available in the subbasin plan by ETD reach. Limiting habitat factors in priority reaches include habitat diversity, temperature, flow, and sediment. There are eleven Tier 1 EDT reaches in the upper watershed, several in each of the main Grays River tributaries.

E.2.3 Limiting Factors

Aquatic habitat evaluations have been conducted by Washington State to identify limiting factors for salmonids (WCC 2002) and as part of the Grays River subbasin planning process (NPCC 2004). The nine limiting factors identified by Washington State (WCC 2002) are summarized in Table E.2. The Grays River subbasin plan identifies eight habitat limiting factor categories (Table E.3) and three threat categories (Table E.4) associated with four priority areas (mainstem and tributaries, headwaters/East Fork, South Fork, West Fork). Within each category, specific limiting factors and threats are identified. Habitat factors identified in the EDT process that limit chum, fall Chinook, and coho salmon are generally similar. Sediment is a primary factor limiting the incubation stage of all three species. Channel instability also limits the fall Chinook salmon incubation stage. Key habitat limits the prespawning holding stage of chum and fall Chinook salmon and the age 0 winter rearing stage of coho salmon. Habitat diversity limits the spawning stage of chum salmon. Temperature limits the spawning stage of fall Chinook salmon and the age 0 rearing stage of coho salmon. The relative importance of habitat factors to recovery of these species is available in the subbasin plan by ETD reach. Habitat limiting factors in priority reaches for chum salmon are habitat diversity and sediment. Habitat limiting factors in priority reaches for fall Chinook salmon are temperature and sediment. Habitat limiting factors in priority reaches for coho salmon are channel stability, temperature, sediment, and key habitat.

Washington's list of limiting factors is oriented toward habitat condition and process outcome, whereas the subbasin plan list is more mechanistic and process-oriented. However, the subbasin plan list reflects that of the earlier Washington list, and the two analyses are largely consistent in their conclusions. Disrupted habitat-forming processes are primarily sedimentation from both the upper watershed and the Grays River spawning areas and absence of inchannel and riparian large woody debris. Hydrologic processes are relatively intact within the watershed, although surface summer low flows may have declined associated with excess bedload. Alterations in these processes have produced unstable channels and banks, excess suspended and subsurface sediments, excess bedload, and altered and homogenized bedform. These collective changes in function and structure have degraded salmonid habitat in multiple ways, including lack of habitat diversity (including quality pools and off-channel and side-channel rearing and spawning habitats), elevated turbidity and temperature, unstable incubation environments, and altered hyporheic processes.

Table E.3. Salmonid Habitat Limiting Factors Identified in the Grays River Subbasin Plan by Priority Area (NPCC 2004)

Limiting Factor	Priority Area[a]			
	MM	HW	SF	WF
Habitat connectivity				
Blockages to off-channel habitats	✓			
Habitat diversity				
Lack of stable instream woody debris	✓	✓	✓	✓
Altered habitat unit composition	✓	✓	✓	✓
Loss of off-channel and/or side-channel habitats	✓			
Channel stability				
Bed and bank erosion	✓	✓	✓	✓
Channel down-cutting (incision)	✓		✓	
Mass wasting		✓	✓	
Riparian function				
Reduced stream canopy cover	✓	✓	✓	✓
Reduced bank/soil stability	✓	✓	✓	✓
Exotic and/or noxious species	✓			
Reduced wood recruitment	✓	✓	✓	✓
Floodplain function				
Altered nutrient exchange processes	✓			
Reduced flood flow dampening	✓			
Restricted channel migration	✓			
Disrupted hyporheic processes	✓			
Stream flow				
Altered magnitude, duration, or rate of change	✓	✓	✓	✓
Water quality				
Altered stream temperature regime	✓	✓	✓	✓
Excessive turbidity	✓		✓	✓
Bacteria	✓			
Substrate and sediment				
Excessive fine sediment	✓	✓	✓	✓
Embedded substrates	✓	✓	✓	✓

(a) MM=middle mainstem and tributaries, HW=headwaters/East Fork, SF=South Fork, WF=West Fork.

Table E.4. Salmonid Threats Identified in the Grays River Subbasin Plan by Priority Area (NPCC 2004)

Threat	Priority Area[(a)]			
	MM	HW	SF	WF
Agriculture/grazing				
Clearing of vegetation	✓			
Riparian grazing	✓			
Floodplain filling	✓			
Forest practices				
Timber harvests –sediment supply	✓	✓	✓	✓
Timber harvests – impacts to runoff	✓	✓	✓	✓
Riparian harvests (historical)		✓	✓	✓
Forest roads – sediment supply	✓	✓	✓	✓
Forest roads – runoff	✓	✓	✓	✓
Forest roads – riparian/floodplain		✓		
Splash-dam logging (historical)	✓	✓	✓	
Channel manipulations				
Bank hardening	✓			
Channel straightening	✓			
Artificial confinement	✓			
Dredge and fill activities	✓			
(a) MM=middle mainstem and tributaries, HW=headwaters/East Fork, SF=South Fork, WF=West Fork.				

E.3 References

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Appendix F

Methods for Bathymetry and Stage Inputs to the MASS2 Model

Appendix F

Methods for Bathymetry and Stage Inputs to the MASS2 Model

The Modular Aquatic Simulation System Two-Dimensional (MASS2) hydrodynamic model (Perkins and Richmond 2004a, 2004b) used to estimate velocity and depth required a bathymetric input and stage:discharge calibration. These data were collected in the chum and fall Chinook spawning areas in Crazy Johnson, the West Fork from the West Fork Hatchery to the Grays River, and the Grays River from the State Route 4 (SR 4) bridge upstream to the canyon section upstream from the Gorley reach. Detailed methods on each of these processes are described in this appendix.

F.1 Bathymetry and Global Positioning System Surveys

Topographic and hydrographic surveys were conducted in the Grays River watershed between the outlet of the canyon section (north of the Gorley property) and the SR 4 bridge. The topographic survey was performed primarily with a Trimble 5700/5800 real-time kinematic global positioning system (RTK GPS). This survey was complemented by a theodolite and GPS-assisted laser range finder where overhead canopy was too dense to receive a GPS signal. The hydrographic survey was achieved with a survey-grade, shallow-water, single-beam echo sounder coupled with RTK GPS.

F.1.1 Global Positioning System Base Station and Survey Initialization

The Grays River surveys were established from a permanent Washington State Department of Transportation (WSDOT) benchmark, Fossil/Monument ID 5045,[(a)] where a GPS base station was established, allowing a starting point for accurate and repeatable measurements throughout the project (see Figure F.1). The Fossil benchmark is about 50 mi east of the Grays River Bridge of SR 4 and has a relative horizontal accuracy of 20 mm and a relative vertical accuracy of 10 mm. The vertical accuracy of the established Fossil RTK GPS base station was verified with WSDOT benchmark H 203/Monument ID 5683[(b)]) located on the southeast corner of the Grays River bridge. This monument has a vertical accuracy of 10 mm. Because the horizontal accuracy of the H 203 benchmark exceeded an acceptable tolerance, the GPS base station was back-referenced by establishing a new benchmark on the Gorley property (Figure F.2) and using the Fossil benchmark for verification. The newly established benchmark was further used as primary base station for the surveys in the Grays River watershed while the Fossil benchmark was used as verification after GPS base station setup. Differences between horizontal and vertical verification measurements and true benchmark position ranged from 1 to 5 cm.

All surveys were collected in the following projection:

- projection: stateplane Washington south, Zone 4602
- horizontal units: meters
- vertical units: meters
- horizontal datum: North American Datum of 1983 (NAD83)
- vertical datum: North American Vertical Datum of 1988 (NAVD88)
- spheroid: Geodetic Reference System, 1980 (GRS80).

(a) http://www.wsdot.wa.gov/Monument/report.cfm?monumentid=5045.
(b) http://www.wsdot.wa.gov/Monument/report.cfm?monumentid=5683.

Figure F.1. GPS Base Station Setup at Washington State Department of Transportation Fossil Benchmark

Figure F.2. New Benchmark Established on Gorley Property Provided Centralized GPS Coverage for Study Area

F.1.2 Topographic Survey

An RTK GPS survey was conducted in different stages between April and August 2004 to supplement areas too shallow for hydrographic data collection. These areas typically included the active stream channel less than 50 cm in depth to a bankfull elevation or higher as deemed appropriate. The survey began on the mainstem Grays River at the outlet of the canyon section and extended to the SR 4 bridge. Additional areas of channel survey included 1) the West Fork Grays River from the Grays River Hatchery to the mainstem Grays River confluence; 2) Fossil Creek from its confluence with the mainstem Grays River 1.5 km upstream; and 3) Crazy Johnson Creek from its confluence with the West Fork Grays River to its channel initiation. The survey was conducted systematically to capture topological features such as old and dry channels or exposed bars. Shallow waters were surveyed perpendicular to flow direction with transect distances of less than 40 m. Figure F.3 illustrates results of the GPS survey from the canyon section to the SR 4 bridge. The GPS points in Figure F.3 are color-coded to represent elevation.

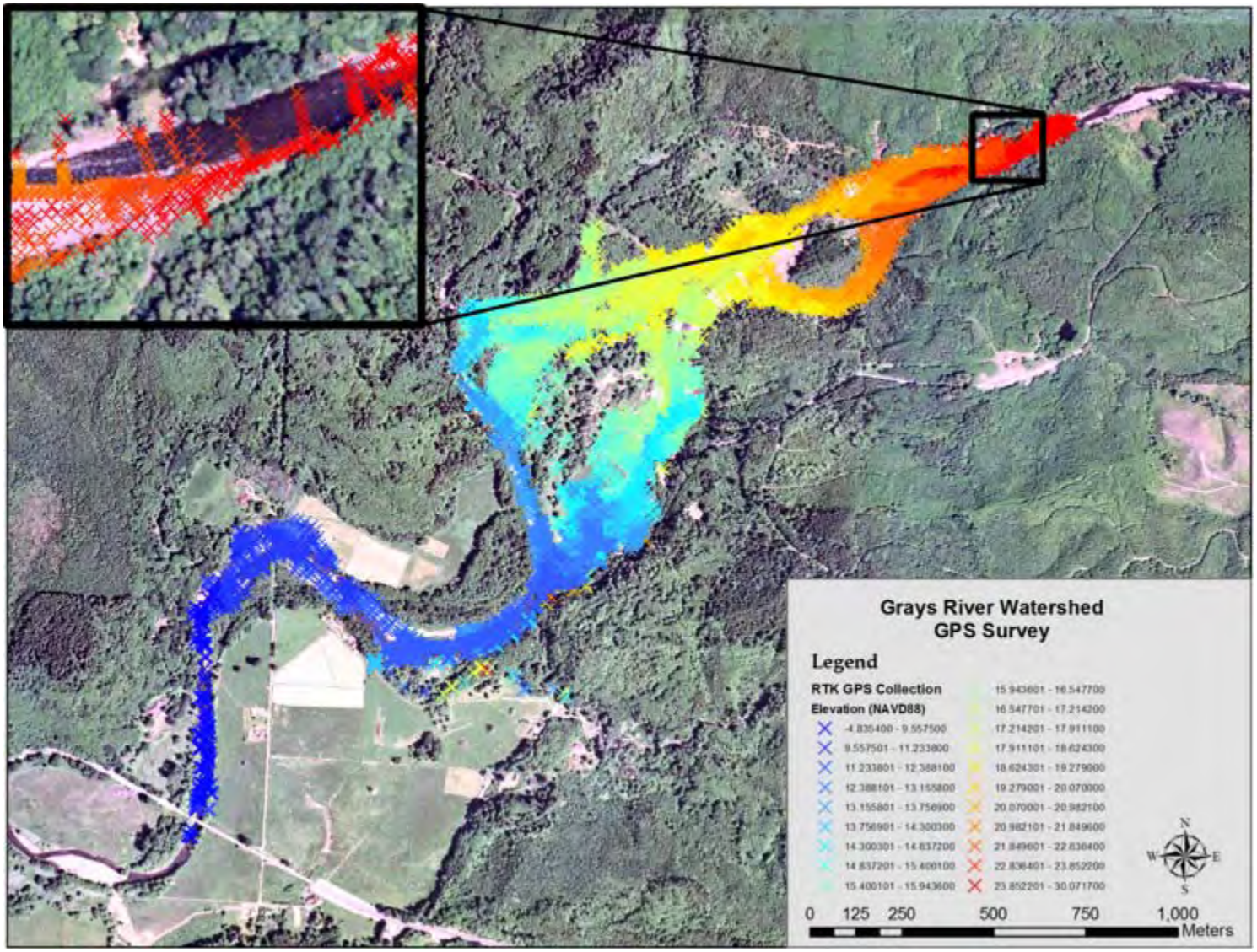

Figure F.3. RTK GPS Topographic Survey for the Grays River Project

At some areas in the watershed, the tree coverage or steep canyon walls did not permit sufficient GPS signal and/or triangulation for accurate GPS point measurements. In these cases, we used a Litz Set 2 theodolite or an Impulse laser rangefinder (ILRF) in combination with a MapStar Angle Encoder Module to survey traditional perpendicular to channel cross sections. The theodolite has an accuracy of 5 to 10 cm, the Angle Encoder is 0.3 degree, and the ILRF distance measurement varies between 3 and 5 cm. Both instruments require positioning calibration by being established on a known GPS location and defining the bearing to another known GPS location. The instruments' rotation to the desired point is internally computed, as are vertical angle and distance to the measured point. Although the theodolite needs a prism reflector for a measurement, the ILRF can use the reflective surface of most objects. For measurements over longer stretches, both instruments were set on the newly measured points and a backsight measurement to the last point defined the new bearing. The topographic survey with these instruments is significantly slower than a GPS survey because every new instrument setup starts with a measurements and calculation of the bearing between two known points.

The theodolite was used in the West Fork Grays River from its confluence with the mainstem Grays River to the confluence of Crazy Johnson Creek and the West Fork Grays River. Parts of the old dike east of the lower West Fork Grays River and berm to the west of the lower West Fork Grays River also were measured with the theodolite. The ILRF was used in the West Fork Grays River from the Grays River Hatchery to the confluence with Crazy Johnson Creek and in Fossil Creek. GPS measurements were taken to diminish an increasing measuring error whenever conditions permitted.

F.1.3 Bathymetric Survey

Bathymetric data were collected using an Innerspace 455 survey-grade, shallow water, single-beam echo sounder with an 8-degree transducer, operating at 208 kHz, and a manufacturer's stated vertical accuracy of 3.05 cm. Positioning and depth data were collected and saved at a rate of one measurement per second. Horizontal and vertical position of the echo sounder was derived using an RTK GPS, providing the most efficient and accurate data possible for the survey. The RTK GPS antenna and integrated receiver were mounted on a fixed-length survey pole directly above the echo sounder transducer. To calculate a true bottom elevation, the depth and survey pole length reported by the echo sounder were subtracted on-the-fly from the synchronized RTK GPS elevations.

The equipment, including a laptop computer for real-time data capture, was mounted on a 9-ft modified fishing cataraft platform. Figure F.4 displays the hydrographic survey cataraft with the established hardware setup. Figure F.5 illustrates the general process to capture RTK GPS data with an echo sounder. The survey cataraft positioning was operated by two people on opposite banks of the channel using guide ropes fixed to the bow of the boat, resulting in a pattern of lateral transects (perpendicular to channel flow). The distance between transects was generally less than 20 m in areas with similar channel characteristics, and significantly more transects were collected through channel bends and sections with significant bed gradient change. To test for the data quality of the hydrographic survey, the bathymetric data were overlapped with the GPS topographic survey.

F.1.4 Data Compilation

The bathymetric and topographic data were processed using Golden Software's Surfer and the Environmental Software Research Institute (ESRI) ArcGIS 9.1 software. The data set was divided into sections with similar flow orientations and was processed using an anisotropic kriging technique, which generates a stream-wise surface grid from an irregular set of point data. This procedure ultimately

provided a set of contour lines that would then be used to generate a raster-based bathymetric surface. The use of anisotropic kriging has been found to minimize typical sinkhole and hillock effects commonly found when processing transect data, often leaving regions between data points with higher elevations than actually observed (e.g., bullseye effect) or other miscellaneous data interpolation anomalies. Each stream section that was processed was verified for conformity with adjacent sections, and, if necessary, suspect data points were eliminated and data were reprocessed. The individual stream-sections were assembled into one surface. Digital elevation models (DEMs) from the U.S. Geological Survey (USGS) were processed into vector spot elevations at 10-m intervals. Individual DEM tiles were assembled to cover the extents of the project area and additional processing was performed to seam mosaic gaps and filter out data anomalies. The final surface with a resolution of 1 m × 1 m was created using an inverse distance weighting (IDW) technique that combined the anisotropic kriging-generated contour lines, topographic and bathymetric survey point data, and USGS 10-m DEMs to represent both the active channel surface and surrounding terrestrial topography. IDW is an interpolation technique that estimates cell values in the raster that have been weighted so that the farther a point is from the cell being evaluated, the less important it is in calculating the cell's value. The final bathymetric surface grids were used for the MASS2 model. Figures F.6 through F.16 detail the resulting bathymetry.

Figure F.4. Fully Equipped Hydrographic Survey Cataraft

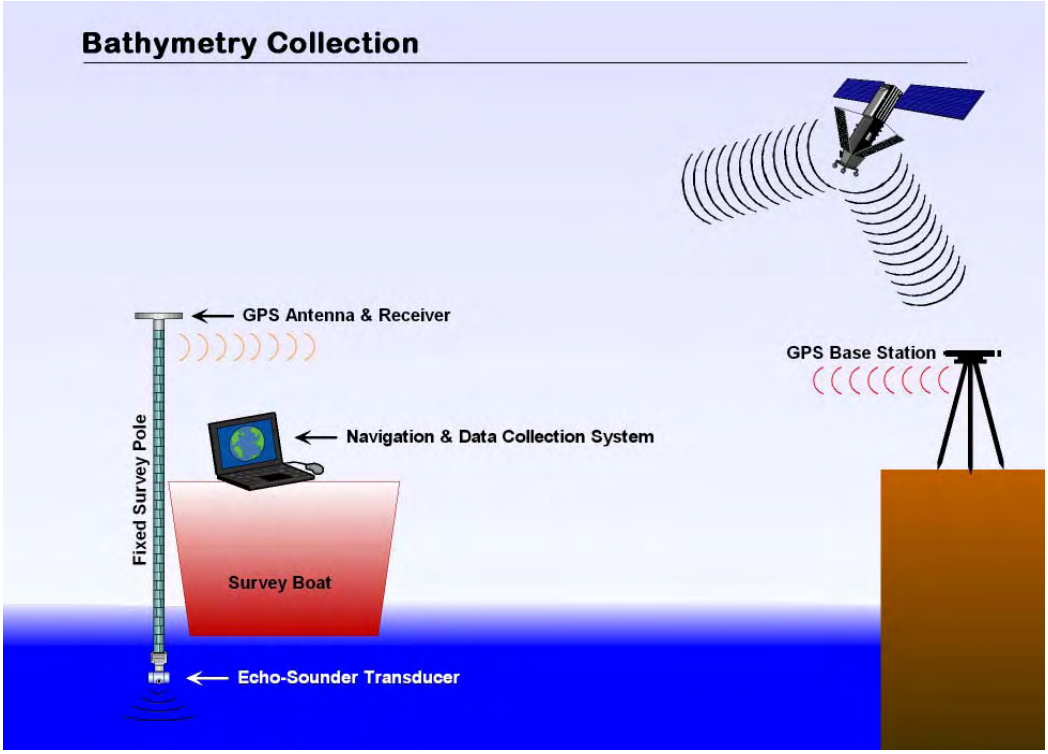

Figure F.5. Collection Process Using Integrated Group of Components to Enable Real-Time Data Collection

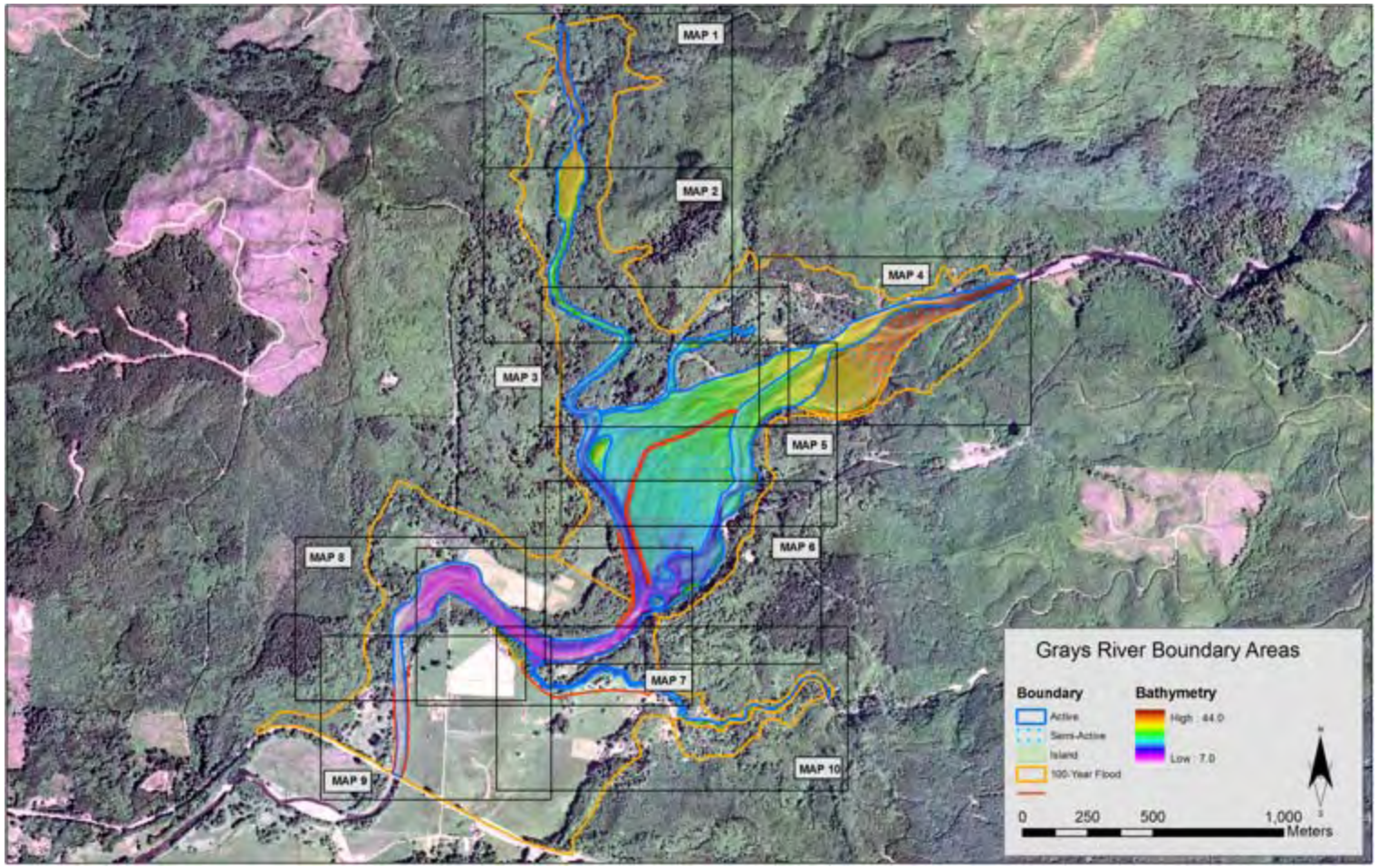

Figure F.6. Overview of Bathymetry and Index Map of the Grays River Project Area

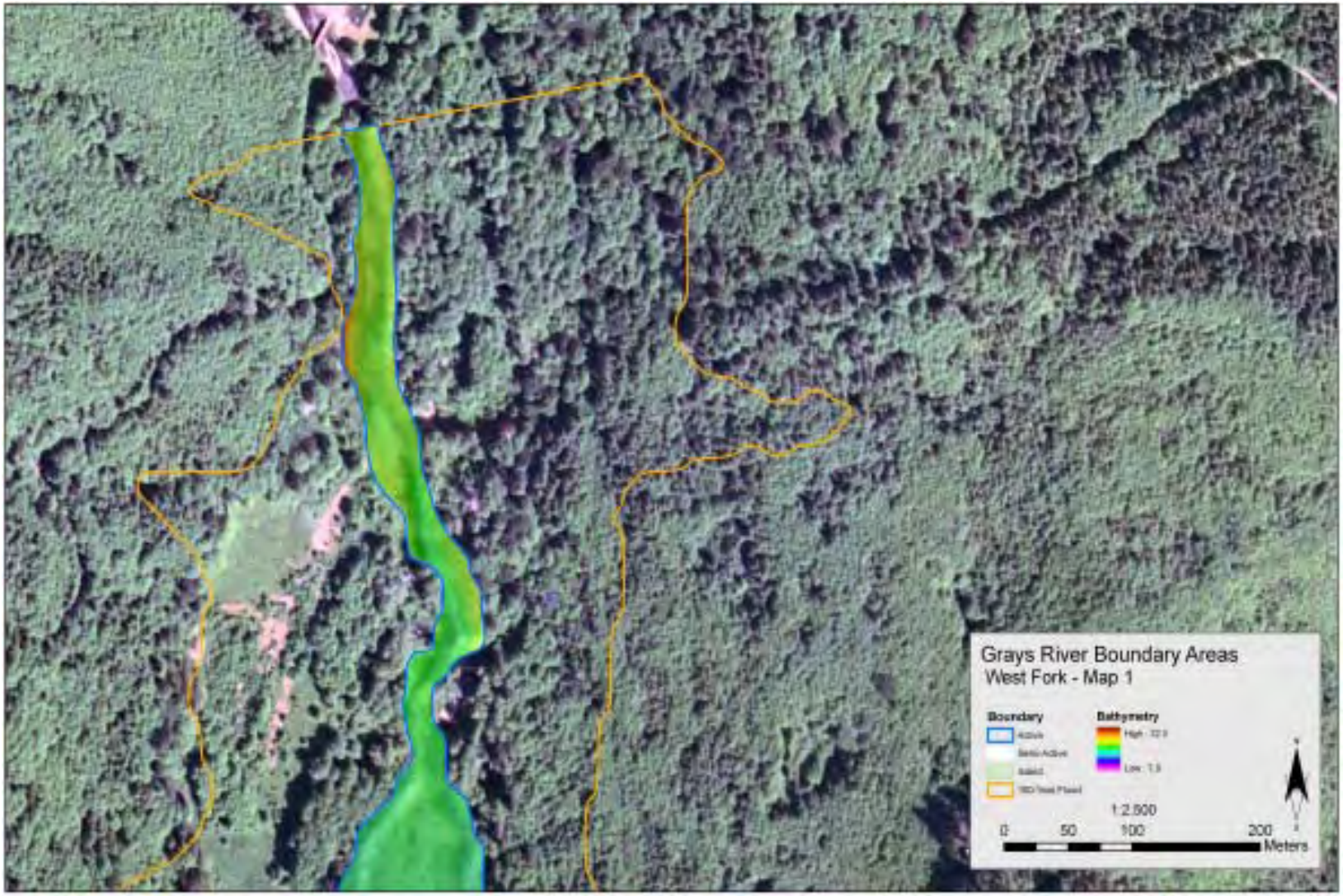

Figure F.7. Bathymetry of the Grays River Project Area – Upper Section West Fork Grays River

Figure F.8. Bathymetry of the Grays River Project Area – Middle Section West Fork Grays River

Figure F.9. Bathymetry of the Grays River Project Area – West Fork Grays River and Crazy Johnson Creek

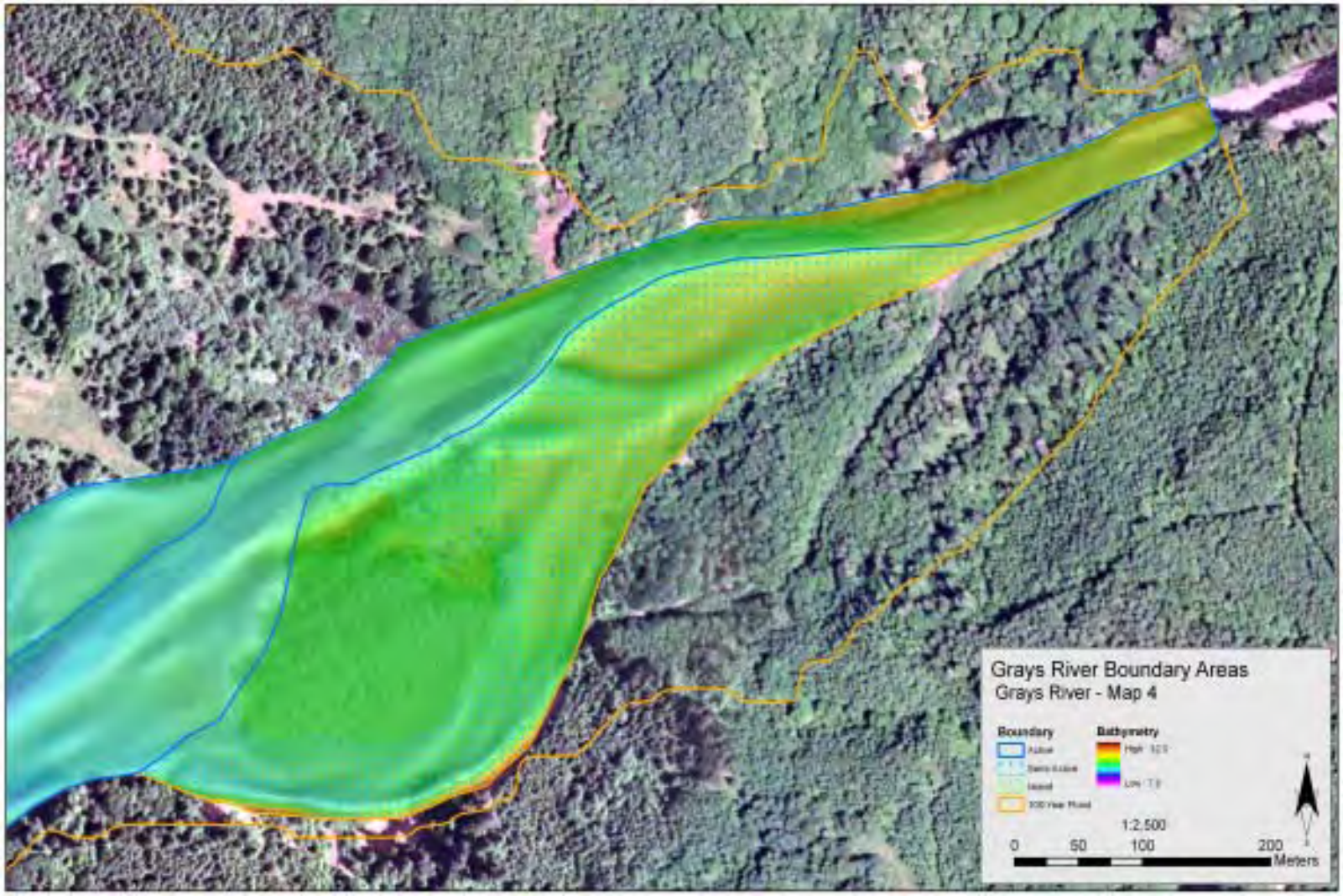

Figure F.10. Bathymetry of the Grays River Project Area – Upper Section Grays River

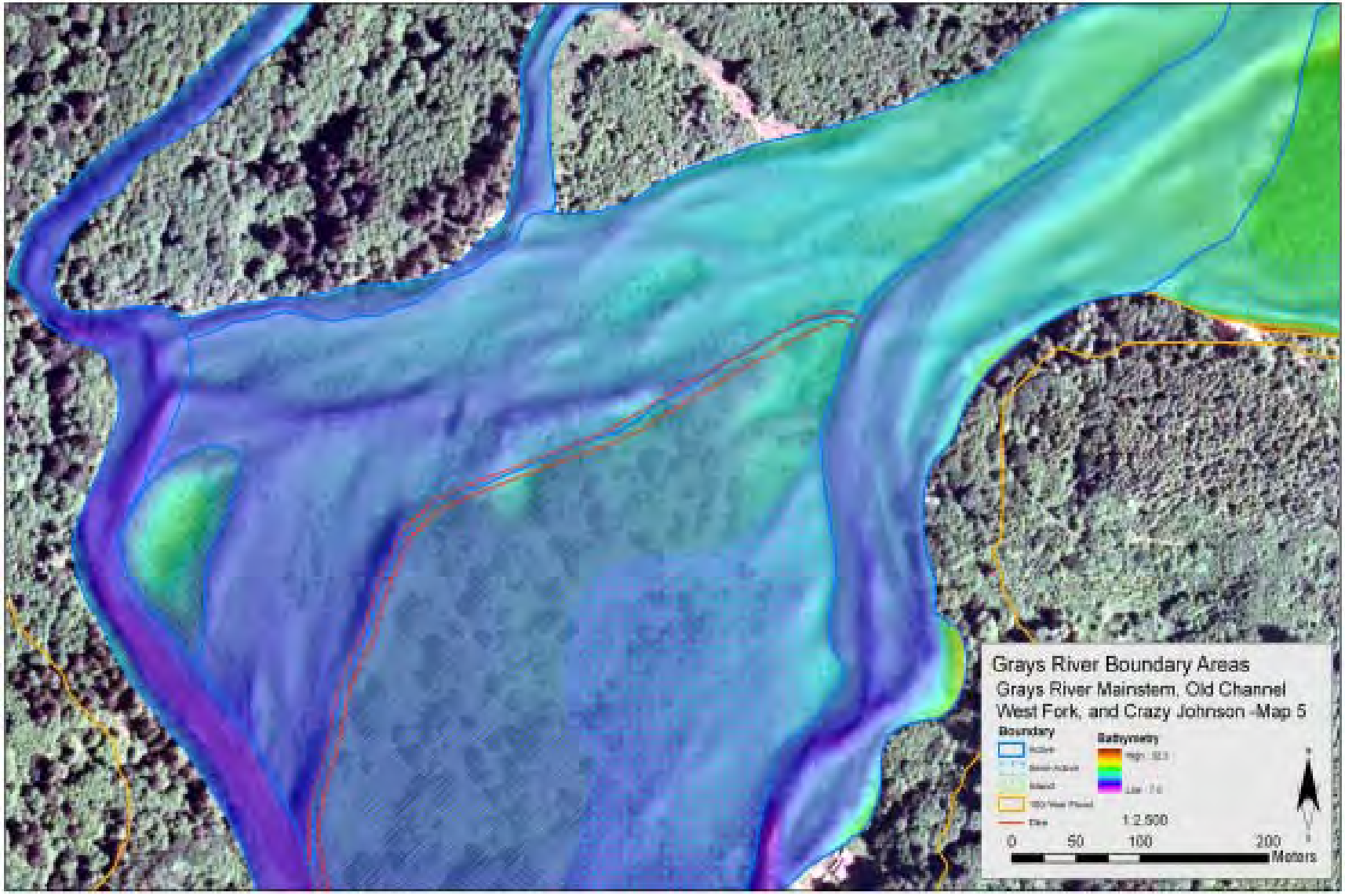

Figure F.11. Bathymetry of the Grays River Project Area – Old Grays River Channel

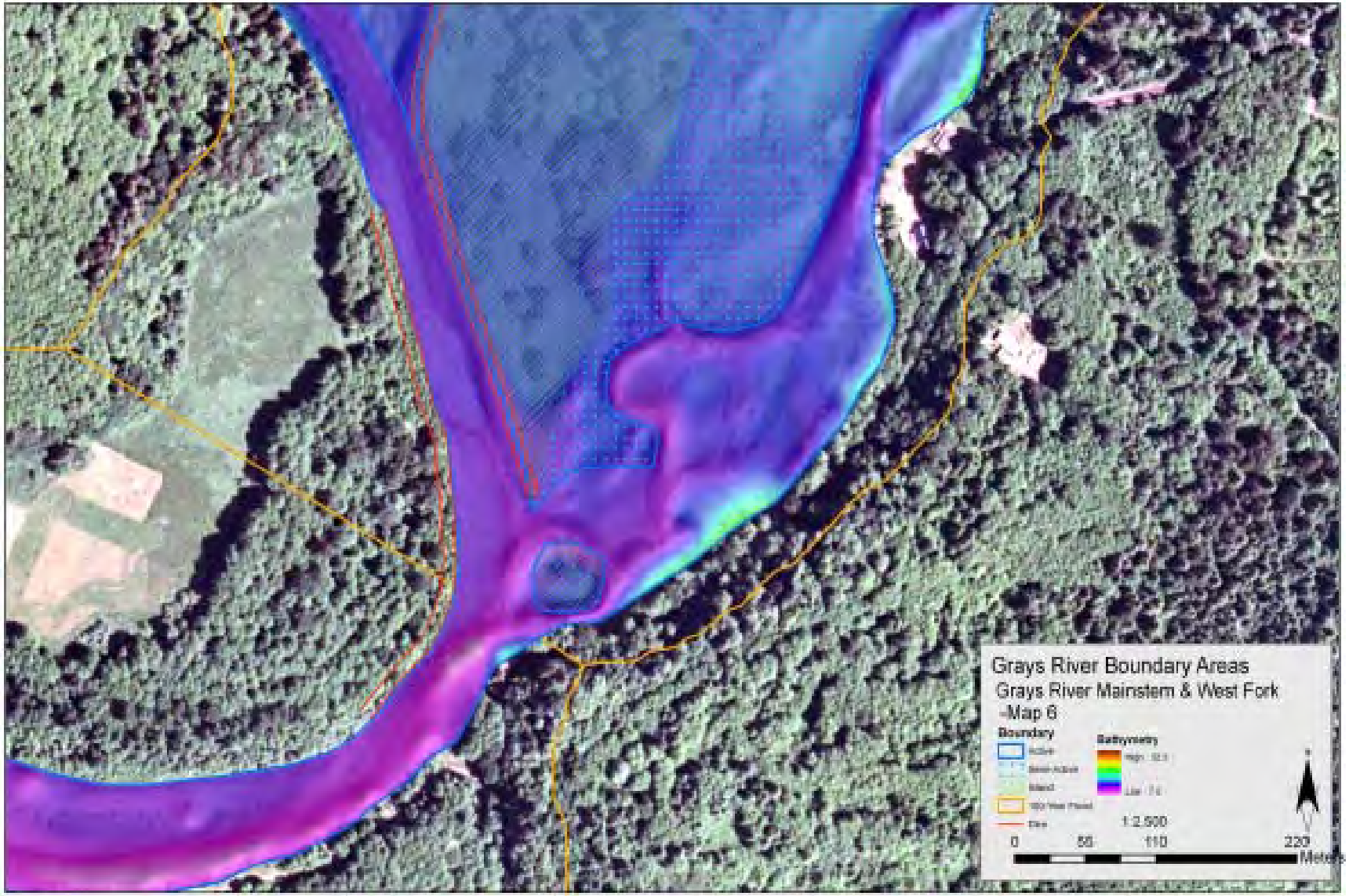

Figure F.12. Bathymetry of the Grays River Project Area – Grays River/West Fork Grays River Confluence

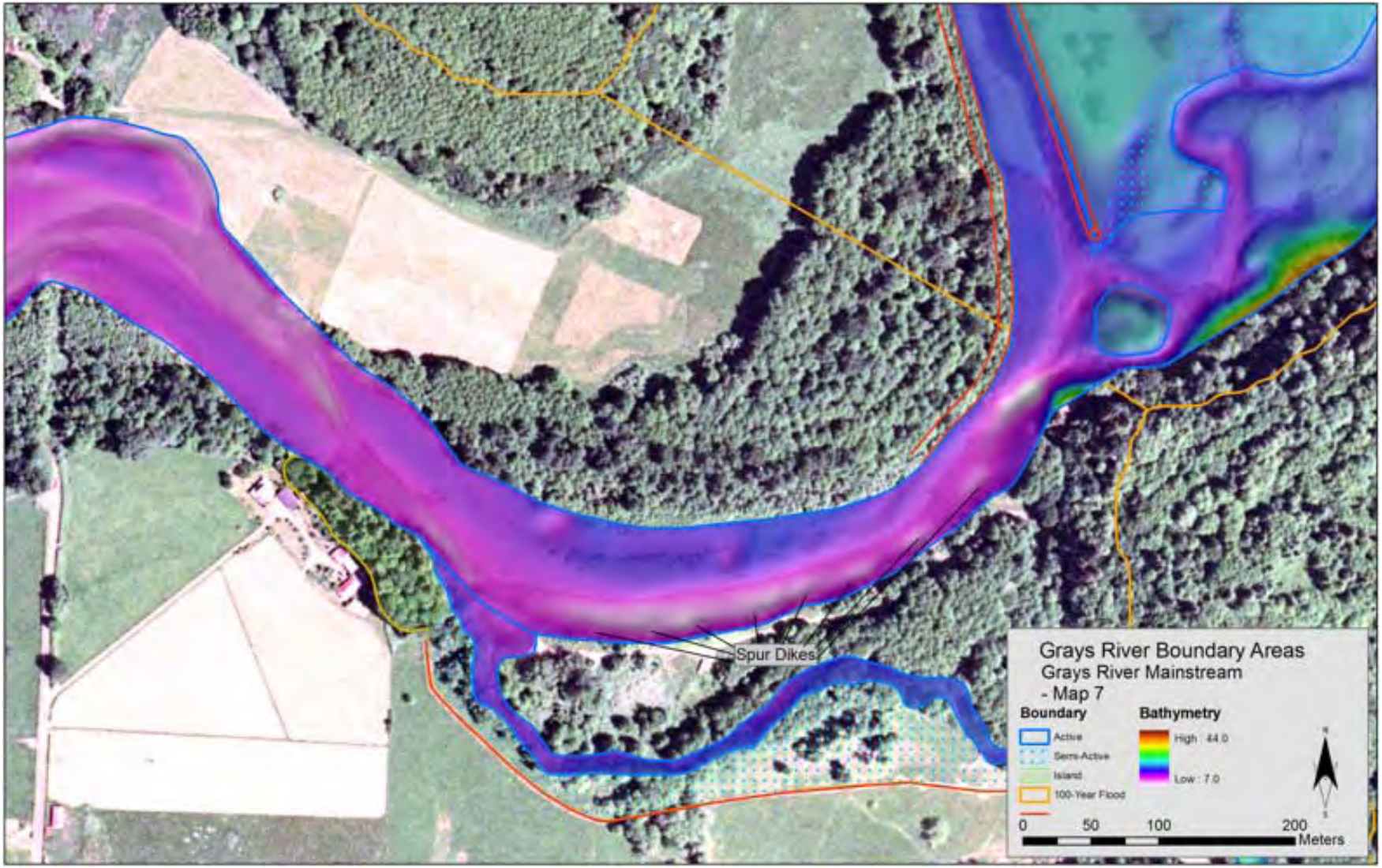

Figure F.13. Bathymetry of the Grays River Project Area – Grays River / Fossil Creek Confluence

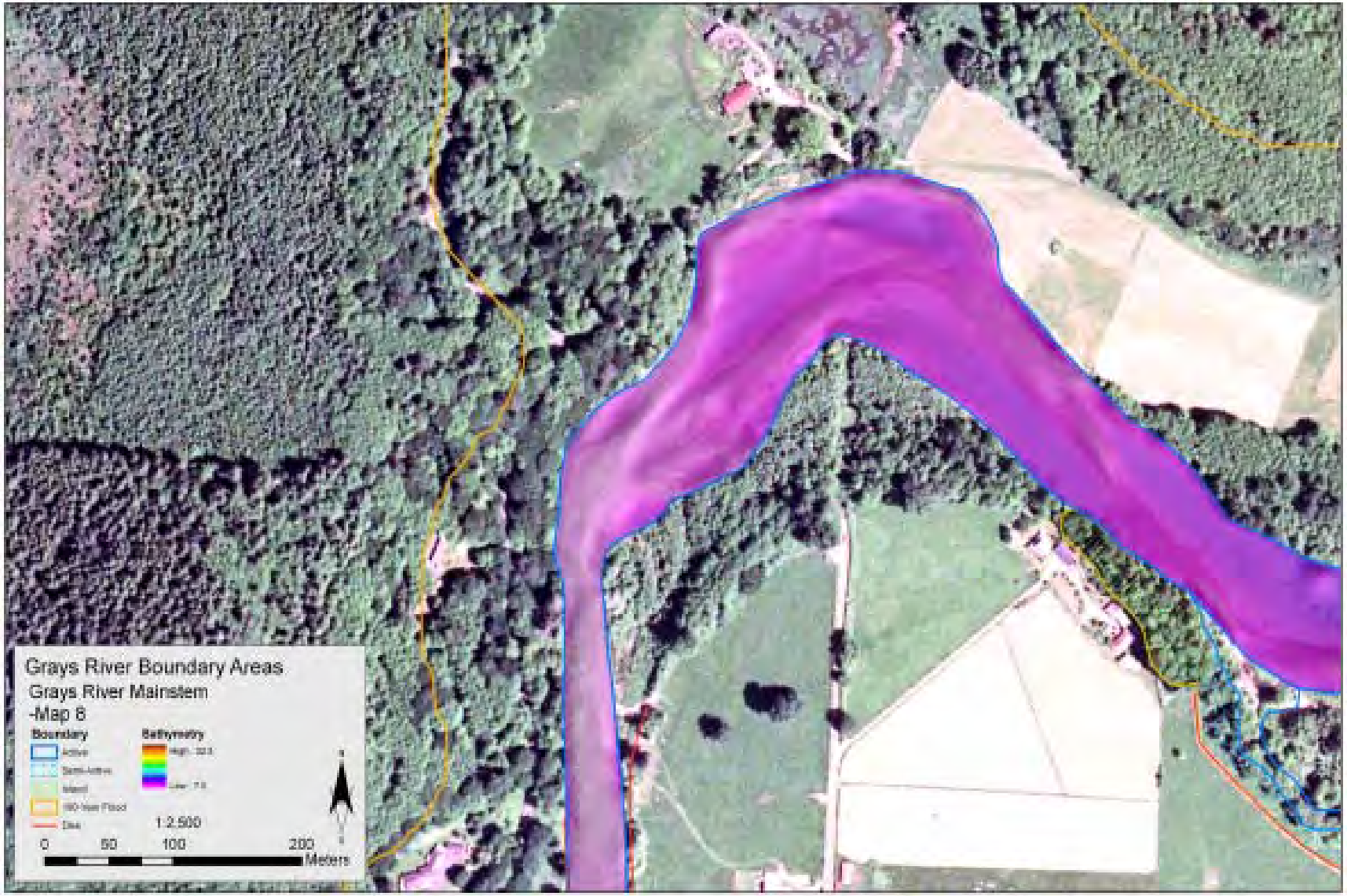

Figure F.14. Bathymetry of the Grays River Project Area – Grays River Mainstem

Figure F.15. Bathymetry of the Grays River Project Area – Grays River at Highway 4 Bridge

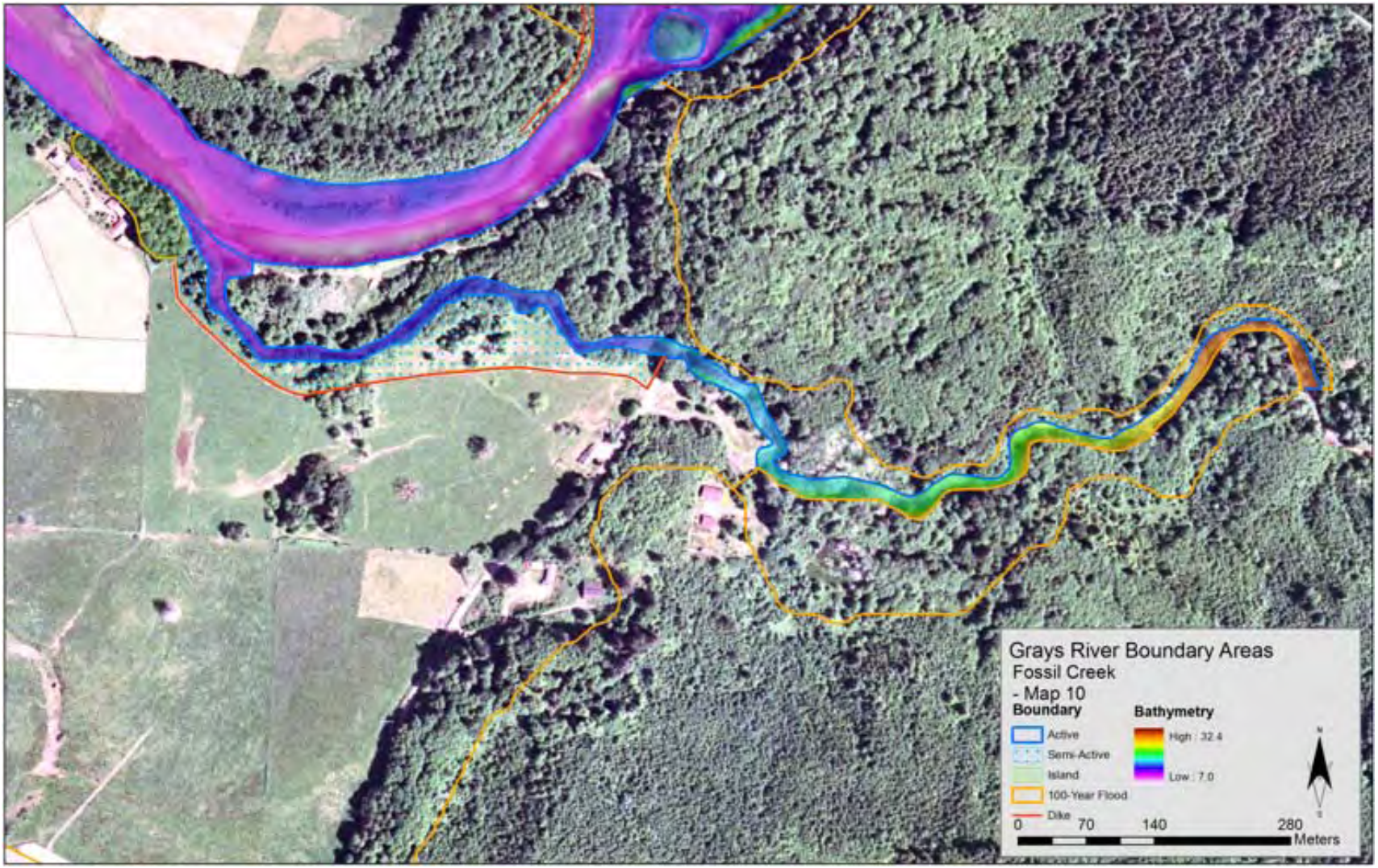

Figure F.16. Bathymetry of the Grays River Project Area – Fossil Creek

F.1.5 Stream Stage Data Collection Methods

During April 18 through 20, 2004, 10 Solinst pressure transducers were deployed at each hydrologic input to the MASS2 model spatial domain within the Grays River basin to record stream stage (Figure F.17). Pressure transducers were deployed inside stainless steel piezometers installed in the riverbed at each location. During piezometer installation, a 2.54-cm-diameter steel drive rod was inserted into the piezometer and used to pound the piezometer into the sediment (Geist et al. 1998). The piezometers had an inside diameter of 3.2 cm and were installed such that the 30-cm-long screens were situated from 0 to 30 cm above the riverbed. Pressure transducers were attached to piezometer caps using stainless steel wire and placed with their sensors near the bottom of the piezometer screens. At each location, the distance from the top of the piezometer to the sensor tip and the distance from the top of the piezometer to the adjacent riverbed were recorded. The elevations of the piezometer tops and adjacent riverbed were later surveyed using either RTK GPS or laser transit, allowing pressure readings to be converted to stream stage using a known datum. At each location, channel cross sections were measured and current-meter–based discharge estimates made so that stream stage could be related to stream discharge.

Each pressure transducer recorded total pressure (barometric pressure that is not corrected to sea level plus pressure due to the overlying water when a logger is deployed underwater). Sensors at locations 1, 2, 3, 4, 5, and 6 (Figure F.17) also recorded temperature. The sensor at location 4 was installed in a metal housing screened to the atmosphere, located near the Grays River fish hatchery. This sensor was never submerged and was used to correct data from all other sensors for pressure changes due to barometric pressure fluctuations. To correct for atmospheric pressure fluctuations, barometric pressure was subtracted from total pressure readings collected by the other sensors, to obtain the river stage.

Pressure sensor accuracy is a function of the pressure range of the sensor, effects due to temperature changes, and sensor drift. Instruments recording only pressure are capable of measuring a pressure range equivalent to 5 m of water and have an accuracy of ±2.5 cm water. These instruments do not compensate for changes in temperature. Instruments recording both pressure and temperature are capable of measuring a pressure range of 10 m of water and have an accuracy of ±1.0 cm water because they compensate for changes in temperature.

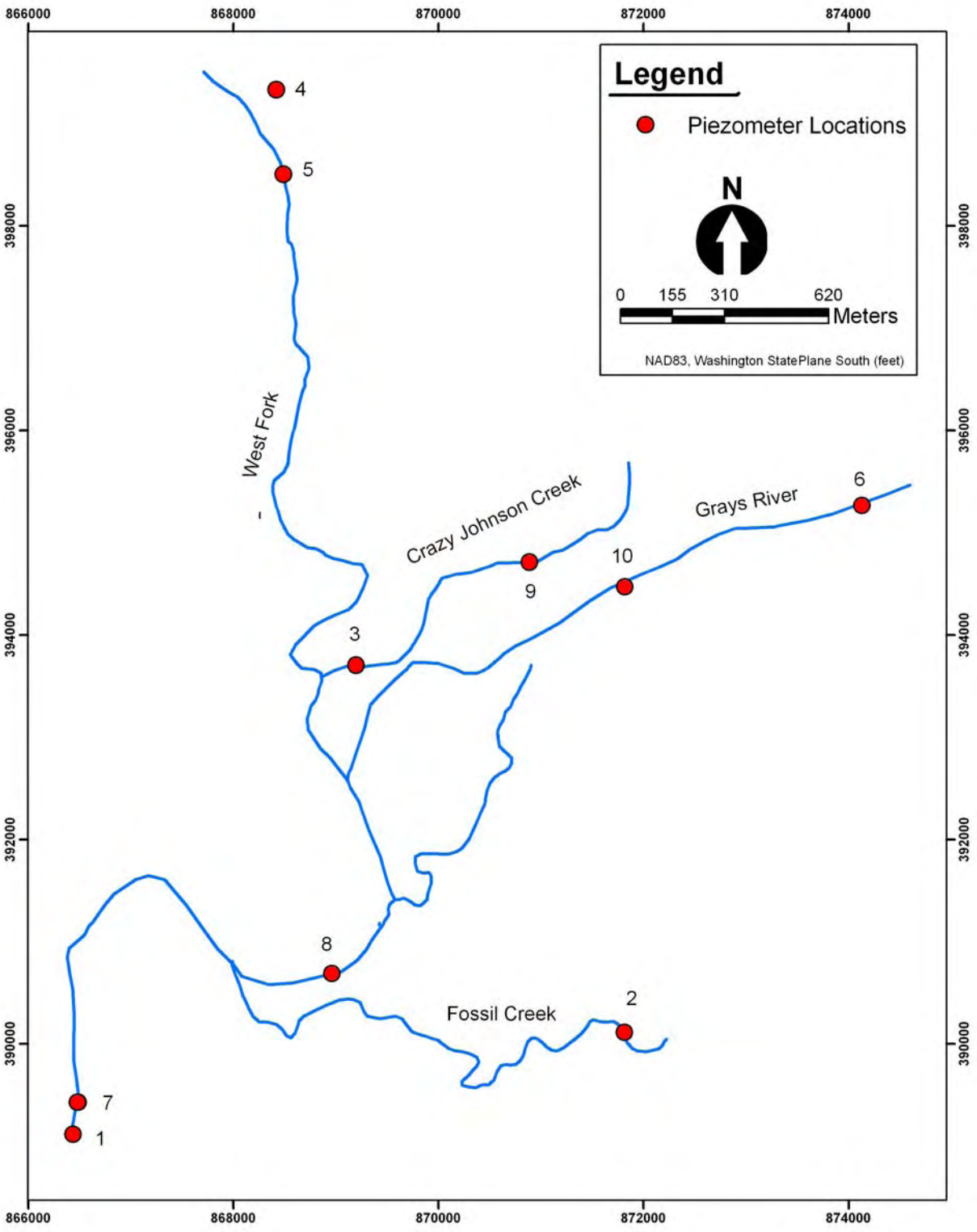

Figure F.17. Location of Piezometers for Collecting Atmospheric and Barometric Pressure for MASS2 Model Calibration

F.2 References

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Appendix G

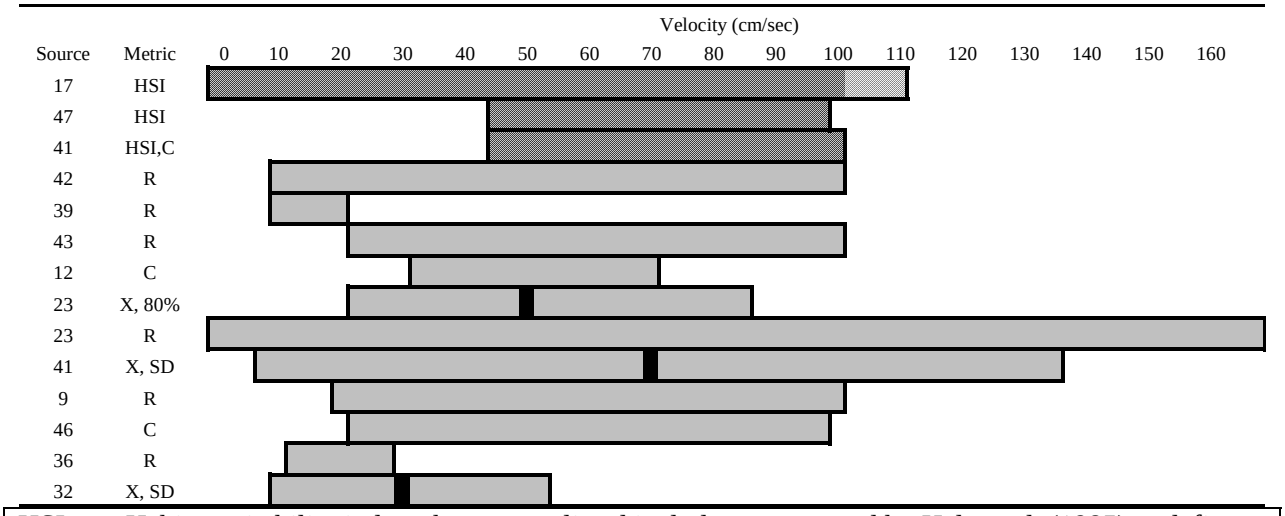
Values for Chum and Fall Chinook Spawning Velocity, Depth, and Substrate Conditions

Appendix G

Values for Chum and Fall Chinook Spawning Velocity, Depth, and Substrate Conditions

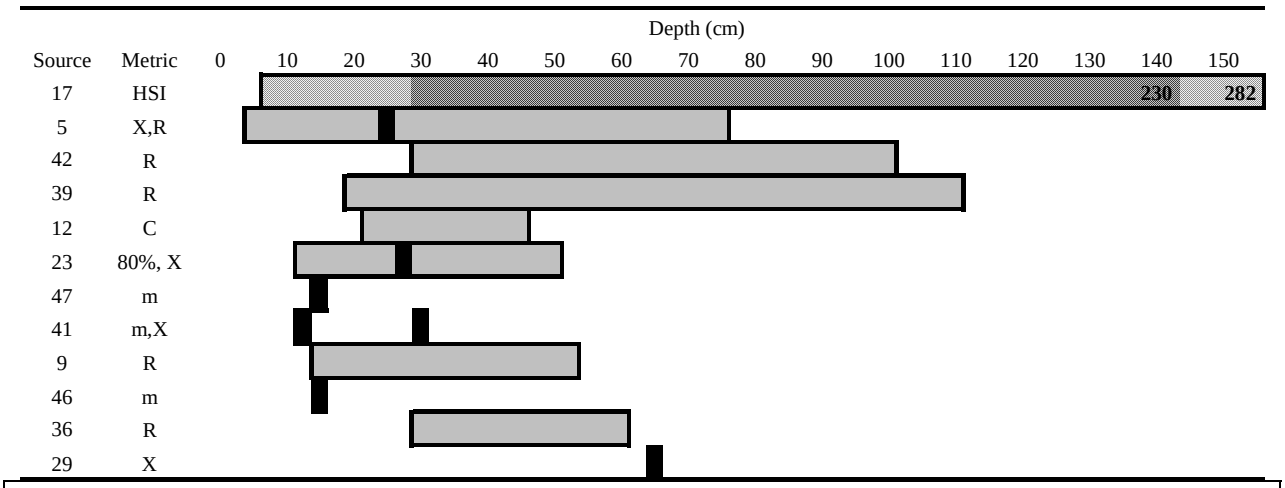
We conducted an extensive review of the literature to identify empirical estimates of velocity, depth, and substrate spawning limits for chum and fall Chinook salmon. In Section 5.2, we used these limits to estimate total potential spawning habitat in the study area. Habitat suitability indices (HSI) for chum and fall Chinook salmon for all three variables have been developed for use in instream flow studies (Hale et al. 1985; Raleigh et al. 1986) and have been used successfully for estimating fall Chinook salmon spawning habitat availability (Hanrahan et al. 2004). However, considerably more information is now available for both species than was used to develop the HSI limits, and most of the available data including those used to develop the HSI are from large river systems and may not apply to the Grays River. We disregarded empirical evidence of spawning use in rare instances if the habitat conditions appeared to represent extreme outliers. Summary tables of spawning velocity, depth, and substrate use by chum and fall Chinook salmon are presented below. In each table, literature sources are identified by number; sources are listed at the end of this appendix.

Table G.1. Values for Chum Salmon Spawning Velocity



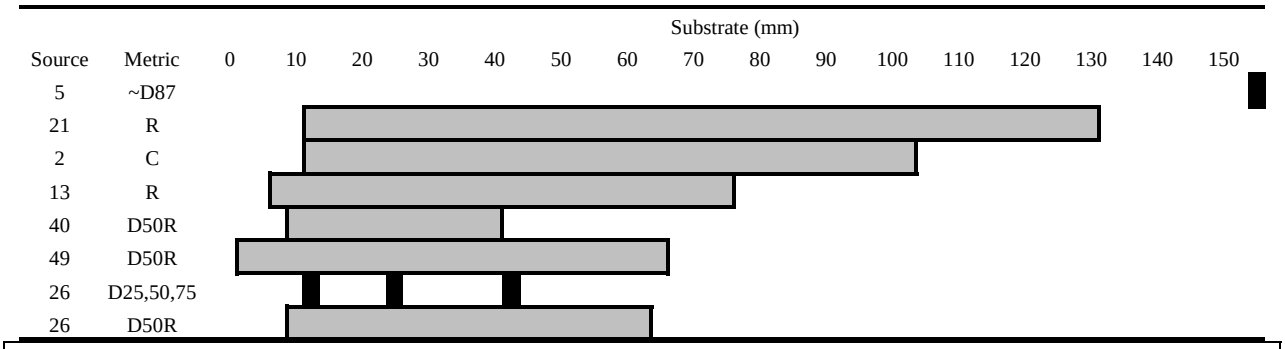
HSI =	Habitat suitability index; the sources listed include or were used by Hale et al. (1985) to define the HSI. Values are SI = 0.1 (light hashed) and 0.4 (dark hashed).
Gray =	R = Range.
Black =	X = Mean unless stated otherwise.
SD =	Standard deviation.
C =	Recommended criteria.

Table G.2. Values for Chum Salmon Spawning Depth



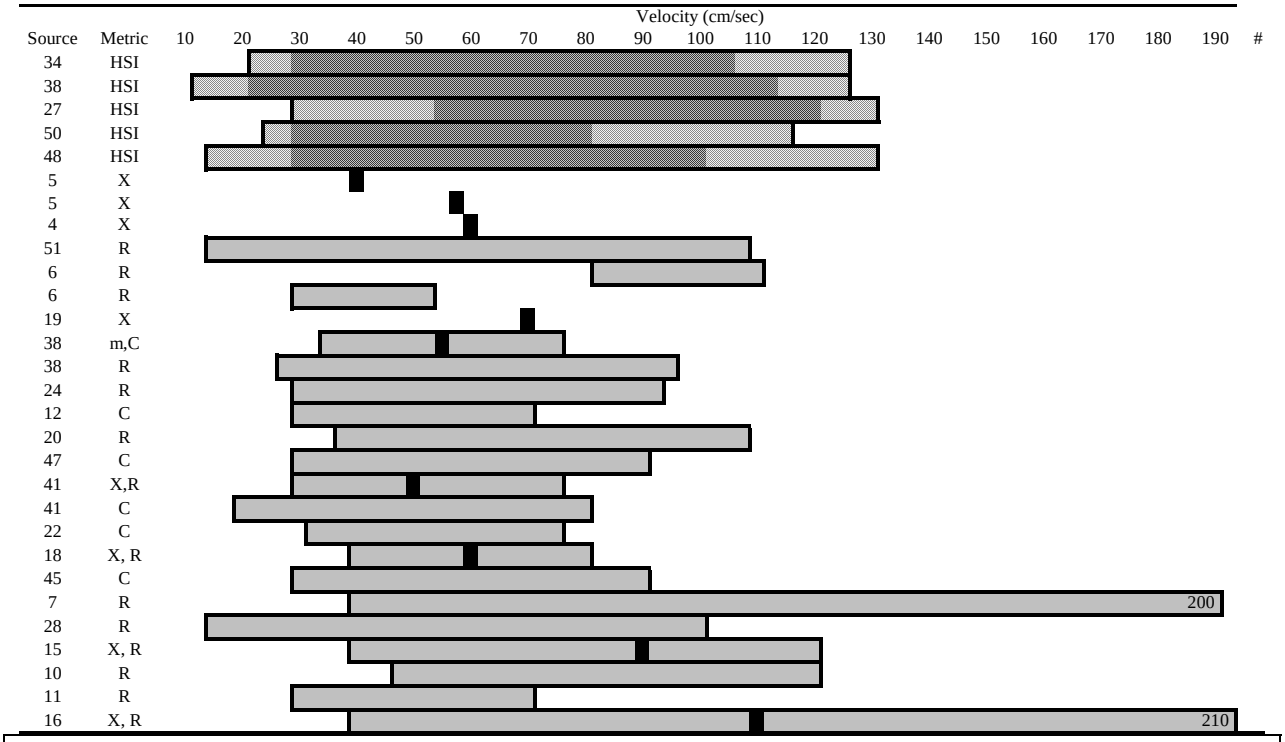
HSI =	Habitat suitability index; the sources listed include or were used by Hale et al. (1985) to define the HSI. Values are SI = 0.1 (light hashed) and 0.4 (dark hashed).
Black =	X = Mean unless stated otherwise.
Gray =	R = Range.
m =	Minimum.
C =	Recommended criteria.
# =	Maximum value studied.

Table G.3. Values for Chum Salmon Spawning Substrate



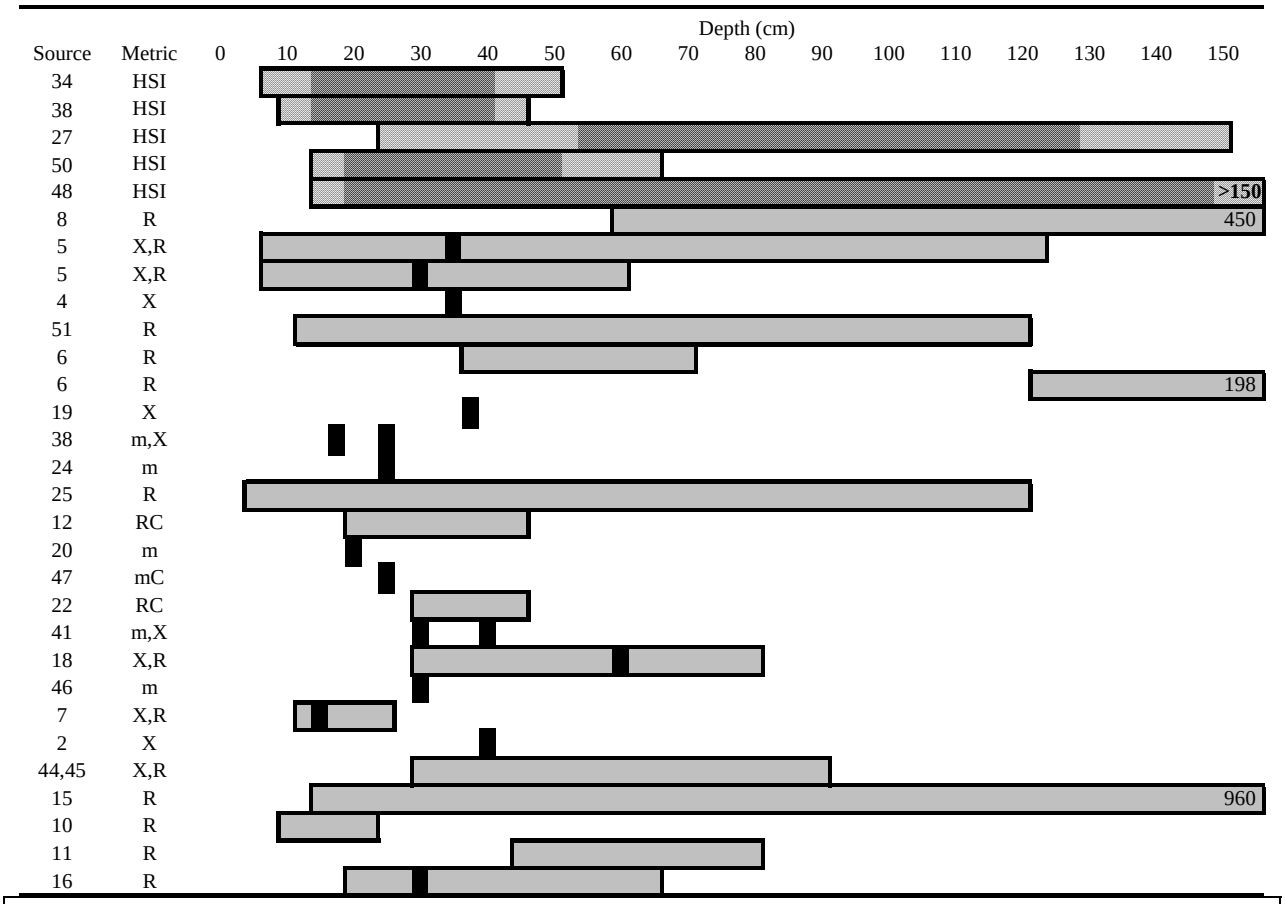
Gray = R = range.
C = Recommended criteria.
D# = Substrate percentile. When a percentage metric is given, that percentage was within range shown.

Table G.4. Values for Fall Chinook Salmon Spawning Velocity



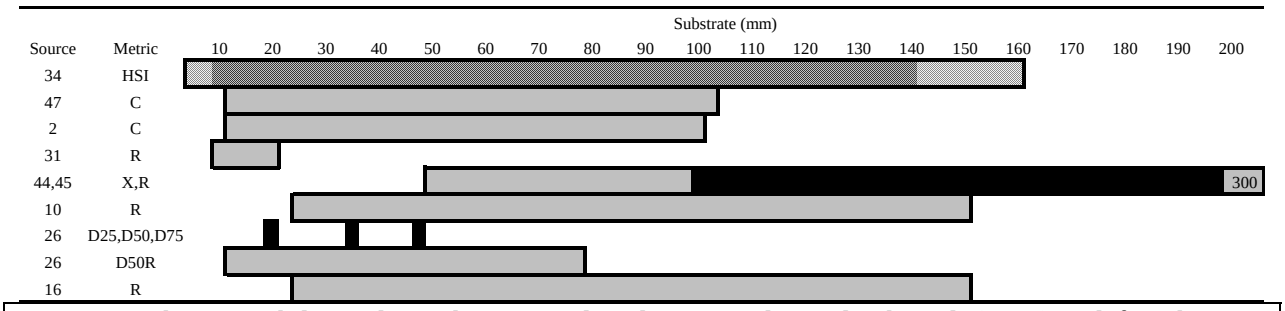
HSI = Habitat suitability index. The sources listed were used in Raleigh et al. (1986) to define the HSI; values are SI = 0.1 (gray hashed) and 0.4 (black hashed).
Gray = R = Range.
Black = X = Mean unless stated otherwise.
m = Minimum.
C = Recommended criteria.
# = Maximum value studied.

Table G.5. Values for Fall Chinook Salmon Spawning Depth



HSI =	Habitat suitability index. The sources listed were used in Raleigh et al. (1986) to define the HSI; values are SI = 0.1 (gray hashed) and 0.4 (black hashed).
Gray =	R = Range.
Black =	X = Mean unless stated otherwise.
m =	Minimum.
C =	Recommended criteria.
# =	Maximum value studied.

Table G.6. Values for Fall Chinook Salmon Spawning Substrate



HSI =	Habitat suitability index. The sources listed were used in Raleigh et al. (1986) to define the HSI; values are SI = 0.1 (gray hashed) and 0.4 (black hashed).
Gray =	R = Range.
Black =	X = Mean unless stated otherwise.
C =	Recommended criteria.
D# =	Substrate percentile. When a percentage metric is given, that percentage was within range shown.
#	Maximum value studied.

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Appendix H

Chum and Fall Chinook Salmon Spawning Habitat in the Grays River Watershed

Appendix H

Chum and Fall Chinook Salmon Spawning Habitat in the Grays River Watershed

This appendix provides additional detail regarding methods and the availability and quality of chum and fall Chinook salmon spawning habitat in the Grays River study area, presented in Section 5.2. The order of Appendix H sections is consistent with that of Section 5.2.

H.1 Upper Grays River Channel Instability 2003-2005

The upper Grays River channel is currently highly unstable following the 1999 channel avulsion. In this reach, the channel changed location several times during our bathymetry, substrate, and temperature data collection in 2004–2005 (Figure H.1). Current spawning habitat conditions in this reach likely differ somewhat from the conditions in which our data were collected.

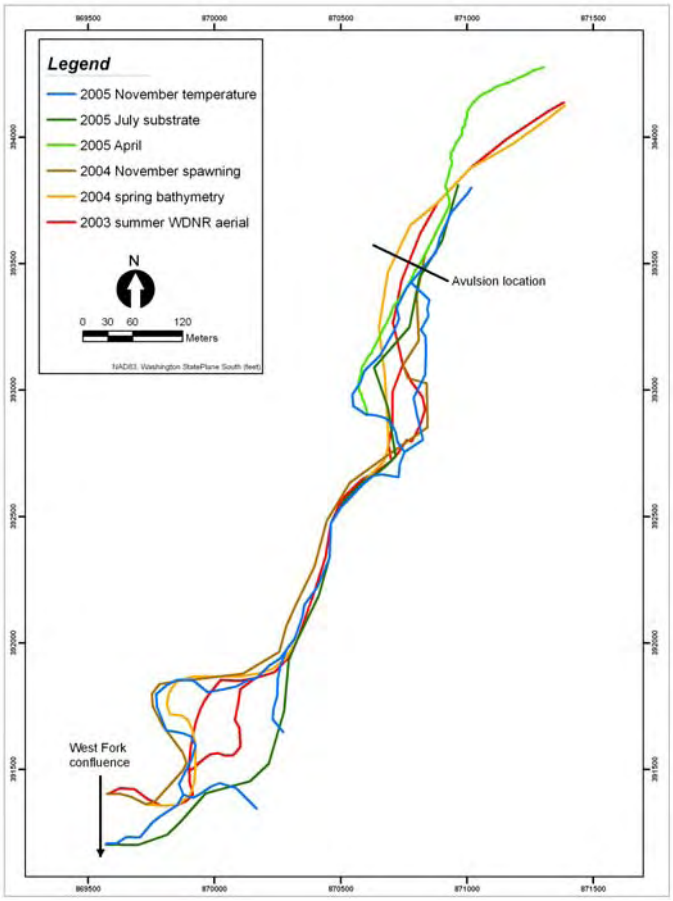

Figure H.1. Upper Grays River Reach Channel Location, 2003–2005, Showing Extreme Instability of Upper and Lower Sections

H.2 Spawning Habitat Availability

H.2.1 Chum Salmon

Most of the Grays River is within chum salmon spawning limits for velocity and depth, and the entire area is within chum salmon spawning substrate limits (Figures H.2 through H.4). Velocity is slightly more limiting than depth (6.3, 4.4%, respectively). Velocity is within spawning limits for chum salmon in 84.1% (upper Grays) to 100.0% (Crazy Johnson) of the total area available. Of the area within velocity limits, most is within the preferred range for chum salmon except in Crazy Johnson Creek. In this creek, 78.5% of the total area is slower than preferred. When all reaches are combined, 83.6% is within the preferred range.

Depth is within chum salmon spawning limits in 84.6% (Crazy Johnson) to 97.5% (upper Grays) of the total area available. Distribution of habitat among limit categories varies among the stream reaches, with the majority in the preferred range in Crazy Johnson and both reaches of the West Fork and the majority deeper than preferred in both Grays River reaches. When all reaches are combined, the majority (55.6%) of area available is deeper than preferred, with 33.8% in the preferred range, 6.3% shallower than preferred, and 4.4% beyond the limits range (too shallow). Although 95.6% of the area available in the Grays River system is within chum salmon spawning limits, only 33.8% is within the preferred range for depth.

All of the substrate in the Grays River study area is within spawning limits for chum salmon. However, almost half (43.7%) is outside the preferred range, with 37.8% finer than preferred. More than half of the area in both reaches of the West Fork and the lower Grays are in the preferred substrate range. In contrast, 83.6 % of Crazy Johnson is finer than preferred.

H.2.2 Fall Chinook Salmon

Area within fall Chinook salmon spawning velocity limit ranges from 18.8% (Crazy Johnson) to 95.3% (lower West Fork), as indicated in Figures H.3, H.4, and H.5. Velocities are too slow for fall Chinook salmon in most of Crazy Johnson. Of the relatively little area in other reaches that is not in the probable use range, more is too slow in the lower Grays and both reaches of the West Fork and too fast in the upper Grays. When all reaches are combined, most (88.4%) of the Grays River is within fall Chinook salmon spawning velocity limits, with 71.7% in the probable use range.

Area within fall Chinook salmon spawning depth limits ranges from 76.7% (Crazy Johnson) to 95.2% (lower Grays). When all reaches are combined, most (92.4%) of the Grays River is within fall Chinook salmon spawning depth limits, with 47.4% in the probable use range. In all reaches, relatively little area is shallower than the probable use range, and in all reaches except the lower Grays, most of the area is within the probable use depth range. In the lower Grays, 34.4 and 57.2% are the probable use range, and deeper than the probable use range, respectively.

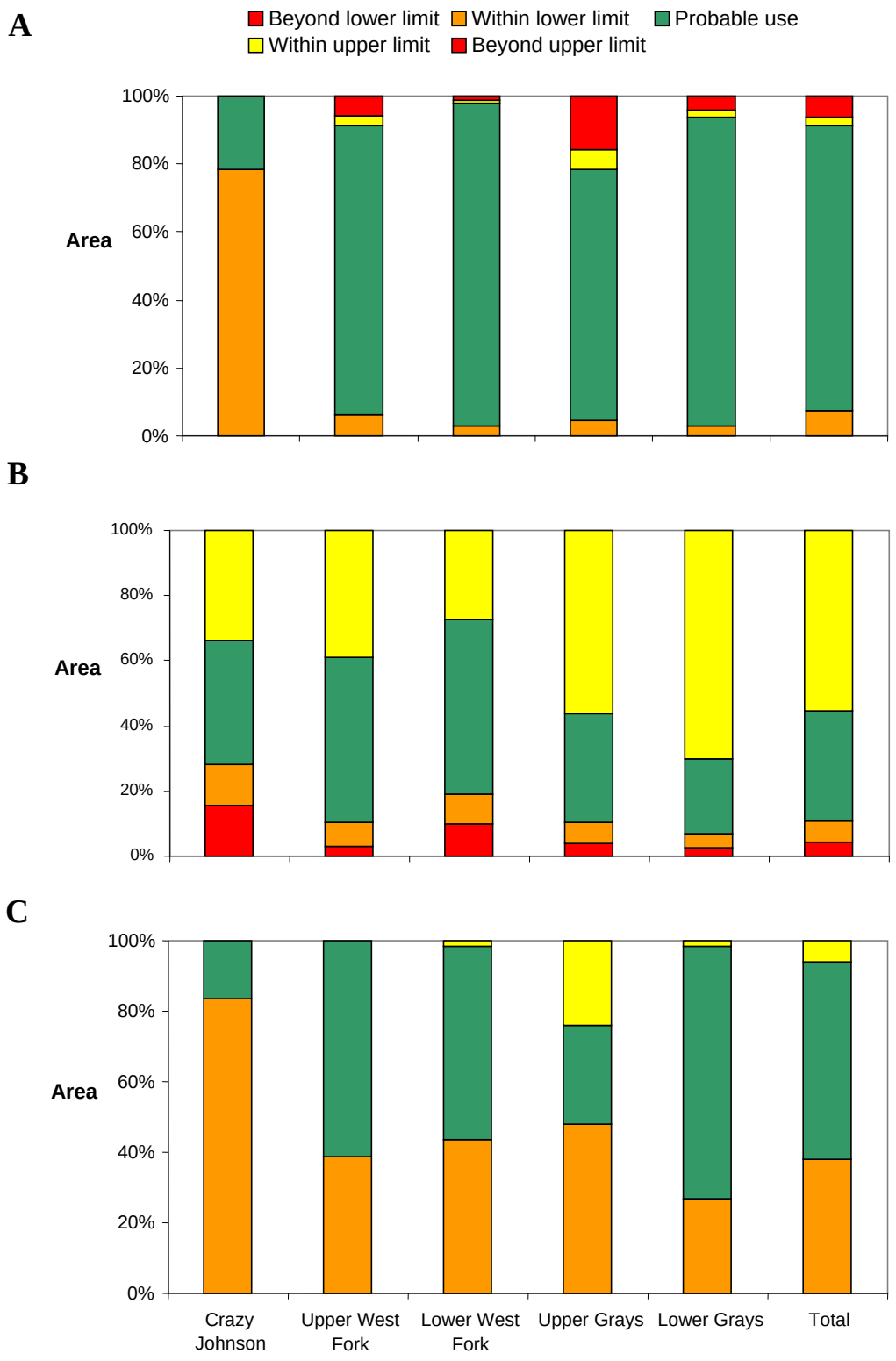

Figure H.2. Chum Salmon Spawning Habitat Limitation by A) Velocity, B) Depth, and C) Substrate

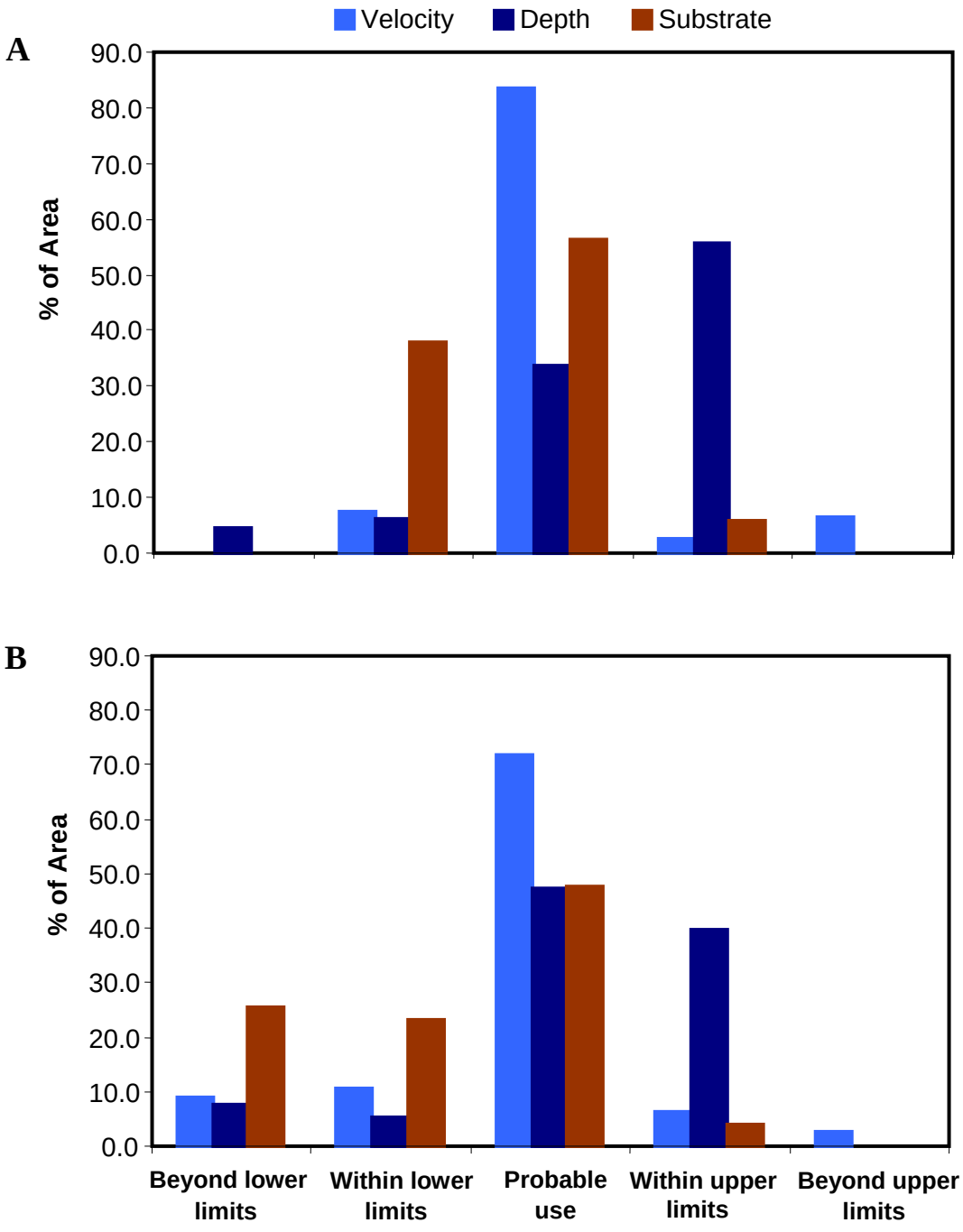

Figure H.3. Distribution of Wetted Area Among Velocity, Depth, and Substrate Spawning Use Categories for Chum (A) and Fall Chinook (B) Salmon

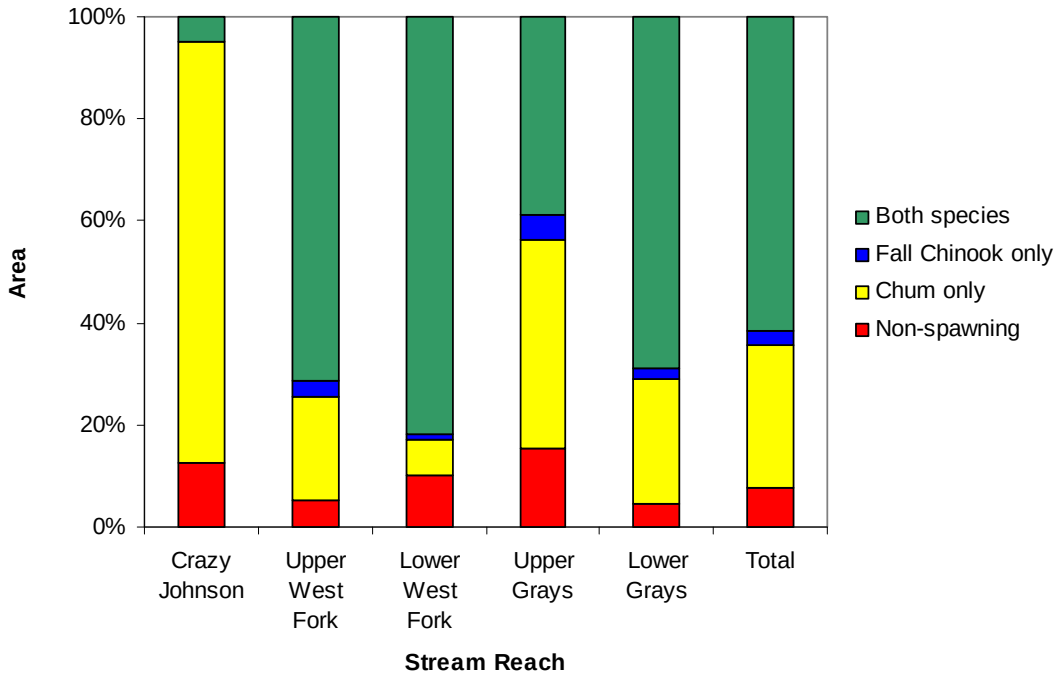

Figure H.4. Spawning Area Availability for Chum and Fall Chinook Salmon by Stream Reach Based on Velocity, Depth, and Substrate

Unlike chum salmon, spawning habitat for fall Chinook salmon is most limited by substrate. Area within fall Chinook salmon spawning substrate limits ranges from 16.4% (Crazy Johnson) to 100.0% (lower West Fork). When all reaches are combined, most (74.6%) of the Grays River is within fall Chinook salmon spawning substrate limits, with 47.6% in the probable use range. Variation among reaches includes Crazy Johnson, in which only 12.0% is in the probable use range and 83.6% is beyond the lower limit (too small), to the lower West Fork in which 100% is within spawning limits and 48.7% is in the probable use range, to the lower Grays, in which 79.2% is within spawning limits with 66.6% in the probable use range. Of the total area within spawning limits but not the probable use range, most substrate is finer than the probable use range; only the upper Grays has substrate coarser than the probable use range (19.1%). A slight majority of area in the upper and lower West Fork is finer than the probable use range (53.1, 51.3%, respectively).

H.3 Spawning Habitat Quality

Data on the quality of both chum and fall Chinook salmon spawning areas in the Grays River watershed are summarized in Tables H.1 through H.3. Graphical representation of habitat quality for both species is presented in Figure H.6.

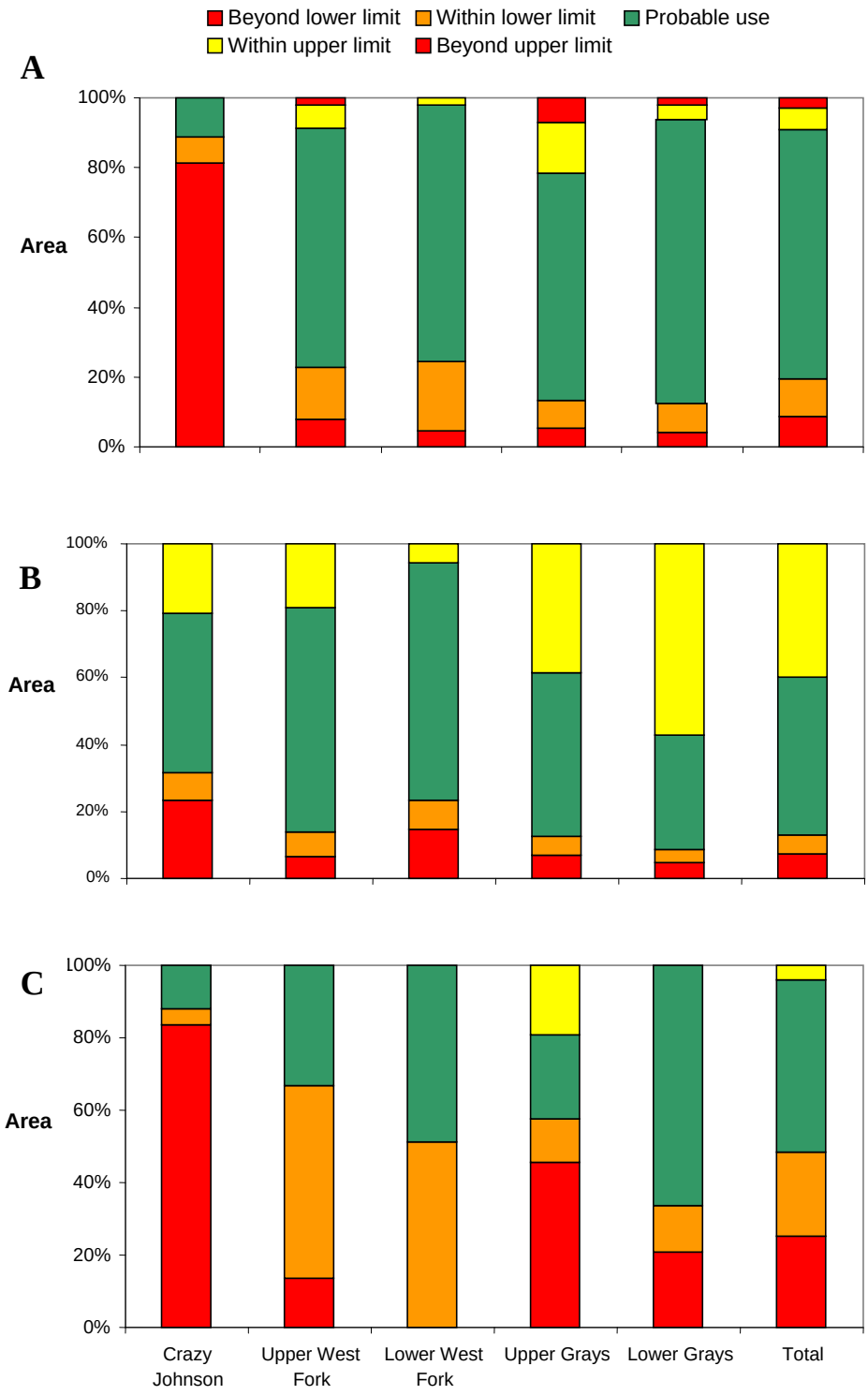

Figure H.5. Fall Chinook Salmon Spawning Habitat Limitation by A) Velocity, B) Depth, and C) Substrate

Table H.1. Quality of Spawning Habitat Based on Velocity, Depth, and Substrate for Chum Salmon by Stream Reach. See Section 5.2 for explanation of quality coding system.

Stream Reach	Habitat Quality Score							
	1.0		1.3		1.6		2.0	
	ha	%	ha	%	ha	%	ha	%
Total	0.65	3.6	5.21	29.3	8.65	48.7	3.26	18.4
Crazy Johnson	0.38	47.5	0.31	37.9	0.07	8.9	0.05	5.7
Upper West Fork	0.06	1.9	0.77	26.4	1.20	41.2	0.89	30.5
Lower West Fork	0.01	0.4	0.42	20.8	1.07	52.5	0.53	26.3
Upper Grays	0.15	4.6	1.38	42.2	1.42	43.2	0.33	10.0
Lower Grays	0.05	0.5	2.33	26.6	4.90	56.1	1.47	16.8

Table H.2. Quality of Spawning Habitat Based on Velocity, Depth, and Substrate for Fall Chinook Salmon by Stream Reach. See Section 5.2 for explanation of quality coding system.

Stream Reach	Habitat Quality Score							
	1.0		1.3		1.6		2.0	
	ha	%	ha	%	ha	%	ha	%
Total	0.29	2.3	2.58	20.2	6.55	51.4	3.33	26.1
Crazy Johnson	0.00	0.2	0.03	66.2	0.01	28.8	0.00	4.8
Upper West Fork	0.15	6.2	0.60	25.2	0.98	41.6	0.64	27.0
Lower West Fork	0.04	2.2	0.31	16.3	1.02	54.0	0.52	27.5
Upper Grays	0.06	3.6	0.63	35.4	0.74	41.6	0.35	19.4
Lower Grays	0.03	0.5	1.01	15.2	3.78	56.9	1.83	27.4

Table H.3. Chum Salmon Spawning Habitat Quality by Stream Reach Limited by Velocity, Depth, Substrate, and Hyporheic Temperature

	Low Quality (score = 1.0)		Moderate Quality (score = 1.3, 1.6)		Highest Quality (score = 2.0)		Total Spawning	
	ha	%	ha	%	ha	%	ha	%
Total	1.99	10.3	3.14	16.3	0.73	3.7	5.88	29.6
Crazy Johnson	0.41	46.5	0.01	0.7	0.00	0.0	0.43	46.9
Upper West Fork	0.26	8.1	0.23	7.4	0.11	3.6	0.60	18.9
Lower West Fork	0.31	13.9	0.39	17.3	0.13	5.7	0.83	36.2
Upper Grays	0.32	8.8	0.29	8.0	0.05	1.4	0.67	16.3
Lower Grays	0.69	7.4	2.22	23.8	0.44	4.7	3.35	35.8

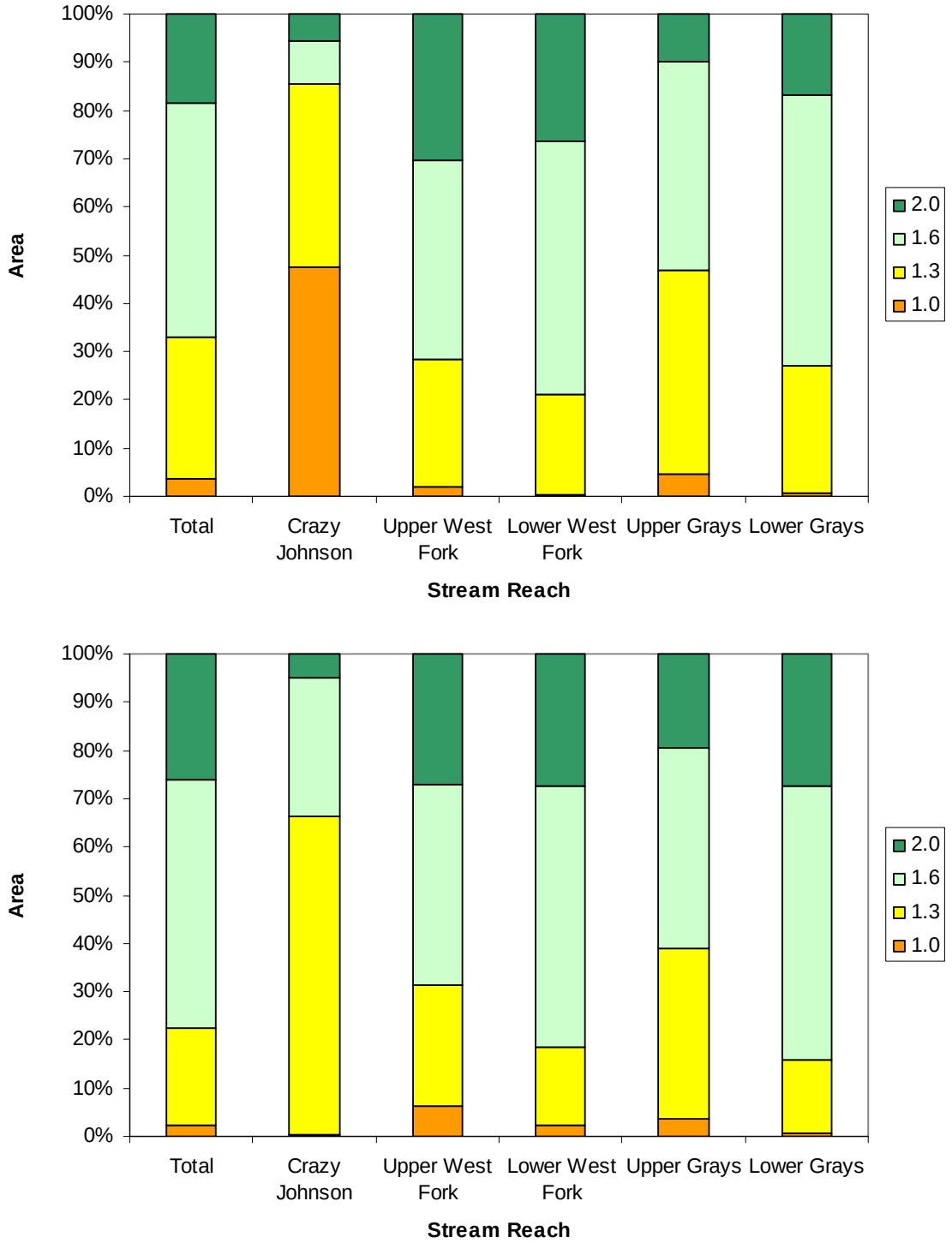

Figure H.6. Quality of Spawning Habitat Based on Velocity, Depth, and Substrate for A) Chum and B) Fall Chinook Salmon by Stream Reach. Quality score ranges from 0.0 (non-spawning, not pictured) to 2.0 (highest quality). See text for additional explanation of quality scoring.

H.4 Chum Salmon Spawning Area Use, 2002–2003

Table H.4. Chum Salmon Spawning Areas During 2002 and 2003. See Section 5.1 for additional methods description regarding delineation of spawning areas.

	2002					2003			
	Velocity (cm/sec)								
	Spawning		Non-Spawning			Spawning		Non-Spawning	
	Mean	Range	Mean	Range		Mean	Range	Mean	Range
Total	18.7	0.0 - 96.0	38.1	0.0 - 194.2		40.7	0.0 - 185.0	58.1	0.0 - 233.3
Crazy Johnson	3.4	0.0 - 42.1	4.0	0.0 - 60.2		10.2	0.0 - 81.9	9.5	0.0 - 93.4
Upper West Fork	23.6	2.1 - 82.5	33.1	0.0 - 148.5		55.1	11.9 - 110.2	53.0	0.0 - 194.0
Lower West Fork	65.5	28.6 - 96.0	25.3	0.0 - 117.7		-	-	42.4	1.0 - 145.5
Upper Grays	-	-	53.3	0.0 - 194.2		-	-	72.6	0.0 - 233.3
Lower Grays	21.3	0.0 - 90.7	41.2	0.0 - 177.4		53.4	0.7 - 185.0	61.2	0.0 - 201.3
	Depth (cm)								
	Spawning		Non-Spawning			Spawning		Non-Spawning	
	Mean	Range	Mean	Range		Mean	Range	Mean	Range
Total	32.1	3.6 - 161.0	49.0	3.8 - 315.1		75.9	4.4 - 281.9	69.2	3.5 - 374.7
Crazy Johnson	40.7	4.0 - 107.2	43.8	3.8 - 152.1		35.8	4.4 - 153.4	45.9	4.5 - 152.9
Upper West Fork	28.5	3.7 - 60.9	33.4	4.0 - 135.5		47.0	5.6 - 102.4	48.3	4.8 - 155.3
Lower West Fork	11.5	6.7 - 17.9	25.9	4.0 - 93.3		-	-	37.7	3.6 - 107.2
Upper Grays	-	-	52.6	3.9 - 219.2		-	-	64.3	4.7 - 255.6
Lower Grays	30.8	3.6 - 161.0	59.7	3.9 - 318.9		99.2	7.8 - 281.9	88.8	3.5 - 374.7
	d_{50} (mm)								
	Spawning		Non-Spawning			Spawning		Non-Spawning	
	Mean	Range	Mean	Range		Mean	Range	Mean	Range
Total	22.5	2.0 - 38.0	20.0	2.0 - 51.5		17.2	2.0 - 38.0	22.0	2.0 - 51.5
Crazy Johnson	7.8	2.0 - 38.0	6.6	2.0 - 38.0		13.1	2.0 - 38.0	5.8	2.0 - 38.0
Upper West Fork	14.2	8.7 - 22.5	17.8	2.0 - 39.9		13.4	4.6 - 18.2	17.8	2.0 - 39.9
Lower West Fork	23.3	2.0 - 33.0	20.1	12.0 - 42.8		-	-	20.1	2.0 - 42.8
Upper Grays	-	-	34.4	2.0 - 51.5		-	-	34.4	2.0 - 51.5
Lower Grays	27.6	7.3 - 34.9	21.8	4.4 - 43.8		20.0	7.3 - 34.9	22.0	4.4 - 43.8
	ΔT (°C)								
	Spawning		Non-Spawning			Spawning		Non-Spawning	
	Mean	Range	Mean	Range		Mean	Range	Mean	Range
Total	1.9	-0.1 -4.2	1.0	-1.0 - 6.3		1.9	-0.3 - 5.0	1.0	-1.0 - 6.3
Crazy Johnson	0.4	-0.1 - 1.3	0.4	-0.3 - 1.4		0.4	-0.3 - 1.4	0.5	-0.3 - 1.4
Upper West Fork	1.7	-0.1 - 4.1	0.5	-1.0 - 4.6		1.8	0.1 - 4.1	0.5	-1.0 - 4.6
Lower West Fork	3.6	3.2 - 4.0	1.0	-0.1 - 4.5		-	-	1.0	-0.1 - 4.5
Upper Grays	-	-	0.9	-0.7 - 6.3		-	-	0.9	-0.7 - 6.3
Lower Grays	2.2	-0.1 - 4.2	1.2	-0.1 - 5.3		2.6	0.0 - 5.0	1.2	-0.1 - 5.3

d_{50} = substrate particle size, ΔT = the difference between hyporheic and river temperatures

H.5 Effects of Flow

H.5.1 Wetted Area and Habitat Availability

Table H.5. Effect of Flow on Wetted Area (ha) in the Grays River Study Area

Stream Reach	% Exceedance Flow		
	90	50	10
Total[(a)]	14.74	19.80	28.20
Crazy Johnson	0.84	0.89	1.30
Upper West Fork	2.56	3.15	4.60
Lower West Fork	1.75	2.28	3.09
Upper Grays	2.53	4.09	6.50
Upper Grays abandoned channel	No flow	No flow	1.96
Upper Grays floodplain	No flow	No flow	11.00
Lower Grays	7.06	9.36	12.71
(a) Does not include upper Grays abandoned channel or floodplain.			

H.5.2 Limiting Factors

Habitat factors limiting spawning area availability for chum salmon changed with flow. Over the study area, 98.4, 93.7, and 73.0 % of the total wetted area was within velocity limits for chum salmon spawning at 90, 50, and 10 % exceedance flows, respectively, and 91.1, 95.6, and 97.1% of total area was within depth limits for these flows (Table H.6). Thus, at the highest flow velocity was the most limiting variable, particularly in the Upper Grays but also in the Lower West Fork and lower Grays. In contrast, at the lowest flow depth became the most limiting factor, particularly in Crazy Johnson and the lower West Fork. Depth was also limiting at the 10 % exceedance flow in the upper Grays floodplain area inundated only at this flow, in which depths were sufficient for use by chum salmon in only 45.5% of the wetted area. Because substrate data were representative of habitat conditions at all flows and substrate was not limiting anywhere in the study area, spawning habitat availability for chum salmon did not differ among flows due to substrate conditions.

Over the study area, substrate was the most limiting factor for fall Chinook salmon at all flows, followed by velocity (Table H.7). In most reaches at most flows, more than 70% of the wetted area was within spawning limits for each variable. Exceptions were Crazy Johnson, in which 8.8 to 41.1% and 11.9 to 21.6% of the wetted area at all flows was within spawning limits for velocity and substrate, respectively, and the upper Grays, in which 45.7 to 54.6% were within limits for substrate. Only 32.6% of the wetted upper Grays floodplain area was within limits for depth. The limiting effect of these variables on spawning habitat area tended to decrease with increasing flow.

Table H.6. Chum Salmon Habitat Availability Within Spawning Velocity and Depth Limits at Three Flows

Stream Reach	% Exceedance Flow					
	90		50		10	
	ha	%	ha	%	ha	%
	Velocity					
Total	14.51	98.4	18.86	93.7	20.60	73.0
Crazy Johnson	0.84	100.0	0.98	100.0	1.27	97.4
Upper West Fork	2.55	99.5	3.03	94.0	3.54	77.0
Lower West Fork	1.75	100.0	2.27	98.8	2.89	93.4
Upper Grays	2.43	96.2	3.50	84.1	3.88	59.6
Upper Grays old channel	No flow		No flow		1.95	99.7
Upper Grays floodplain	No flow		No flow		10.98	99.8
Lower Grays	6.93	98.2	9.07	95.8	9.03	71.0
	Depth					
Total	13.42	91.1	19.24	95.6	27.39	97.1
Crazy Johnson	0.70	83.4	0.83	84.6	1.24	95.6
Upper West Fork	2.26	88.2	3.13	97.0	4.37	95.0
Lower West Fork	1.45	83.0	2.07	90.0	3.06	99.0
Upper Grays	2.41	95.4	3.98	95.9	6.08	93.5
Upper Grays old channel	No flow		No flow		1.62	82.7
Upper Grays floodplain	No flow		No flow		5.00	45.5
Lower Grays	6.60	93.5	9.23	97.5	12.64	99.4

Table H.7. Fall Chinook Salmon Habitat Availability Within Spawning Velocity, Depth, and Substrate Limits at Three Flows

Stream Reach	% Exceedance Flow					
	90		50		10	
	ha	%	ha	%	ha	%
	Velocity					
Total	11.76	79.8	17.79	88.4	22.99	81.5
Crazy Johnson	0.07	8.8	0.18	18.8	0.53	41.1
Upper West Fork	1.91	74.6	2.89	89.8	3.62	78.8
Lower West Fork	1.35	76.9	2.19	95.3	3.00	97.1
Upper Grays	2.29	90.8	3.63	87.3	4.55	69.9
Upper Grays old channel	No flow		No flow		1.69	86.5
Upper Grays floodplain	No flow		No flow		7.70	70.0
Lower Grays	6.13	86.9	8.90	94.0	11.28	88.8
	Depth					
Total	12.74	86.4	18.60	92.4	27.09	96.1
Crazy Johnson	0.66	78.4	0.76	76.7	1.20	92.7
Upper West Fork	2.06	80.2	3.01	93.4	4.30	93.5
Lower West Fork	1.30	74.3	1.96	85.3	3.04	98.3
Upper Grays	2.34	92.6	3.86	93.0	5.94	91.5
Upper Grays old channel	No flow		No flow		1.46	74.8
Upper Grays floodplain	No flow		No flow		3.58	32.6
Lower Grays	6.38	90.4	9.02	95.2	12.60	99.1
	Substrate					
Total	10.82	73.4	14.89	74.6	19.98	70.9
Crazy Johnson	0.10	11.9	0.16	16.4	0.28	21.6
Upper West Fork	2.21	88.2	2.74	86.3	3.94	85.8
Lower West Fork	1.75	99.3	2.30	100.0	3.09	100.0
Upper Grays	1.24	48.7	2.25	54.6	2.96	45.7
Upper Grays old channel	No flow		No flow		0.77	39.5
Lower Grays	5.52	77.7	7.46	79.2	9.70	76.3

Appendix I

Analysis of Macroinvertebrate Community Integrity

Appendix I

Analysis of Macroinvertebrate Community Integrity

Measurements of chemical and physical components of natural waters do not typically provide sufficient information to detect or resolve water resource impairment problems. Biological evaluation of surface waters provides a broader approach because degradation of sensitive ecosystem processes is more frequently identified by the response of native biota to anthropogenic influences. Assessments of biotic integrity directly measure the most sensitive resources at risk to aquatic resource degradation; they measure an ecological component that integrates and reflects the cumulative effects of human influence over the long term. Integrated measures of macroinvertebrate integrity synthesize chemical, physical, and biological perturbations within the aquatic ecosystem. Inclusion of multiple levels of biological measures (e.g., community trophic structure, functional feeding groups, reproductive strategy) enhances the ability to accurately diagnose the source of degradation.

Over the last century, biological monitoring tools needed to establish water quality criteria have taken a number of different approaches and examined a wide range of biota—most commonly, fish, macro-invertebrates, and algae. Biological assessment (measuring and evaluating biota directly) has ranged from simple diversity indexes (Hilsenhoff 1982) multi-metric indexes (Karr and Chu 1999; Morley 2000; Wiseman 2003). The multi-metric approach evaluates biological condition by integrating measures of an empirically tested set of biological attributes. This approach was used first with stream fish and has since been modified for invertebrates and periphyton (Barbour et al. 1999). An advantage of the multi-metric approach is its relative simplicity; results are easily communicated and understood by nonscientists.

We used the multi-metric index of integrity approach to evaluate aquatic habitat conditions within the Grays River watershed. The benthic index of biological integrity (BIBI) is a multi-metric index based on attributes of the benthic invertebrate community (Karr and Chu 1999; Morley 2000; Wiseman 2003). Since the mid 1990s, the BIBI has been used to evaluate the biological condition of Pacific Northwest streams and watersheds (Kleindl 1995; Fore et al. 1996; May et al. 1997; Karr and Chu 1999; Morley 2000; Wiseman 2003). Benthic-dwelling macroinvertebrates are effective indicators of stream condition because they are relatively long-lived, sedentary, diverse, and abundant. They are also key components of the aquatic foodweb, and many are sensitive to human disturbance (Vannote et al. 1980; Karr 1991; Rosenberg and Resh 1993). The BIBI comprises specific community metrics that predictably and reliably respond to human impact (Karr and Chu 1999).

I.1 Methods

Invertebrates were collected from each site in late September and early October when flows are typically stable, taxa richness high, and field crews have easy access to stream sites without too much conflict with spawning salmon (Fore et al. 1996). One riffle was sampled at each site. Three locations within each riffle were sampled and combined to provide one composite sample per site. Sampling locations within each riffle were selected to provide longitudinal representation (riffle head, middle, and tail), as well as lateral representation (midchannel, near bank, and intermediate between midchannel and bank).

At each site, a Surber sampler (500-µm mesh, 0.1-m^2 frame) was used to collect samples. A disturbance-removal technique was used to collect macroinvertebrates. Larger pieces of the substrate were scrubbed to dislodge organisms. The remaining substrate was then disturbed to an approximate depth of 10 cm for 1 minute using a garden claw. In the field, each sample was strained through a 500-µm sieve, mineral material picked through and discarded, and the remaining sample preserved in a solution of 70% ethanol. Water-quality data were recorded at each site to detect any significant differences between sites in physical and chemical properties that can affect macroinvertebrate populations. The location of each monitoring site was mapped with a global positioning system (GPS) unit. The GPS data were transferred digitally to the geographic information systems (GIS). Each site also was characterized with digital pictures. Grays River watershed sample sites are listed in Table I.1.

Table I.1. 2004 Grays River Watershed Biomonitoring Sample Sites

1	Lower mainstem upstream of SR 4 highway bridge
2	Middle mainstem downstream of Fossil Creek confluence (Puddicombe)
3	Lower Fossil Creek upstream of Fossil Creek Road
4	Middle mainstem at Gorley (main channel)
5	Middle mainstem at Gorley (side channel)
6	Middle mainstem upstream Gorley Springs (canyon mouth)
7	Lower West Fork downstream of WDFW hatchery
8	Lower South Fork (just upstream of confluence)
9	Upper mainstem upstream of South Fork confluence
10	Upper mainstem downstream of South Fork confluence
11	Lower East Fork at 7200 Road bridge
12	Upper mainstem (North Fork) just upstream of East Fork confluence
13	Lower Blaney Creek upstream of South Fork confluence
14	Lower Alder Creek downstream of old bridge and dam site
15	7220 Road tributary at RM 21 (MP 8.5 on 7200 Road)
16	7230 Road tributary to East Fork

The samples were then shipped to a certified biological laboratory (Aquatic Biology Associates, Corvallis, Oregon) for processing. Under microscopy, invertebrates were separated from remaining mineral and organic debris, identified, and counted. Insect nymphs and larvae, the bulk of benthic samples, were identified to genus where practical; non-insect taxonomic identification varied from family to phylum. Nonbenthic invertebrates, pupae, and terrestrial adults were excluded from sample analysis. A complete reference set of invertebrates identified throughout the study was also assembled and verified by independent analysis.

The BIBI was applied to the macroinvertebrate sample results. The standard Pacific Northwest BIBI is composed of ten metrics of taxa richness and diversity, population attributes, disturbance tolerance, and feeding and other habits (see Table I.2). When values from the ten metrics are combined, BIBI ranges from a minimum of 10 to a maximum of 50 and can detect five categories of resource condition. Each metric is assigned scoring criteria that represent the degree of human disturbance present (see Table I.3). The BIBI commonly used in the Pacific Northwest has generally been developed and tested in small, wadable streams. This BIBI consistently yields reliable results in small streams across different ecoregions and types of human impact (Barbour et al. 1999). However, the Pacific Northwest stream BIBI does not appear to be effective on larger rivers (Jessup and Gerritsen 2002; Royer and Mebane 2002). Therefore, we also applied a BIBI under development for Pacific Northwest rivers (Celedonia 2004). The BIBI scoring criteria for the proposed river BIBI are shown in Table I.4.

Table I.2. Metrics of the Pacific Northwest Benthic Index of Biological Integrity and Their Predicted and Observed Response to Watershed Development (Karr and Chu 1999)

Metric	Category	Response	Scoring Criteria		
			1	3	5
Total Taxa (#)	Richness	Overall biodiversity decreases as aquatic ecosystem is altered.	0-19	20-40	>40
Mayfly (*Ephemeroptera*)	Richness	Diversity of mayflies generally declines with human influences. Particularly sensitive to chemical pollutants and changes in nutrients or food sources.	0-4	5-8	>8
Stonefly (*Pleoptera*)	Richness	Some of the most sensitive organisms. Very sensitive to sedimentation of substrata and to higher stream temperature.	0-3	4-7	>7
Caddis-fly (*Tricroptera*)	Richness	Diversity declines steadily with human influences, especially hydrologic changes.	0-4	4-9	>=10
Long-Lived Taxa	Richness	Live in stream for more than 1 year. Sensitive to human influences that change annual cycles such as hydrologic regime.	0-2	3-4	>4
Dominance of the Three Most Common Taxa (%)	Relative Abundance	As biodiversity declines with human influence, a few taxa tend to dominate the macroinvertebrate assemblage. Opportunistic species tend to increase.	>75%	50-75%	<50%
Sensitive Taxa	Richness	Intolerant taxa are the first to disappear with human influence.	0-2	3	>3
Tolerant Taxa (%)	Relative Abundance	Tolerant taxa are always present, but as human disturbance increases, these organisms begin to dominate the macroinvertebrate assemblage.	>50%	20-50%	<20%
Clinger Taxa	Richness	These organisms live on the streambed substrata. Very sensitive to siltation and flow increases resulting from human land-use activities.	0-10	11-20	>20
Predators (%)	Relative Abundance	Represent the top of the benthic macroinvertebrate food web. Depend on abundance and diversity of other macroinvertebrate organisms. Less disturbed sites tend to support a greater diversity of prey and thus have more predators.	0-10%	10-20%	>20%

Table I.3. Five Classes of Biological Condition (Morley 2000)

Biological Condition	BIBI Score	Description
Excellent	46-50	Comparable to least disturbed reference condition; overall high taxa diversity, particularly of mayflies, stoneflies, caddis-flies, long-lived, clinger, and intolerant taxa. Relative abundance of predators high.
Good	38-44	Slightly divergent from least disturbed condition; absence of some long-lived and intolerant taxa; slight decline in richness of mayflies, stoneflies, and caddis-flies; proportion of tolerant taxa increases.
Fair	28-36	Total taxa richness reduced - particularly intolerant, long-lived, stoneflies, and clinger taxa. Relative abundance of predators declines; proportion of tolerant taxa continues to increase.
Poor	18-26	Overall taxa diversity depressed; proportion of predators greatly reduced as is long-lived taxa richness; few stoneflies or intolerant taxa present; dominance by three most abundant taxa often very high.
Very Poor	10-16	Overall taxa diversity very low and dominated by a few highly tolerant taxa; mayfly, stonefly, caddis-fly, clinger, long-lived, and intolerant taxa largely absent. Relative abundance of predators very low.

Table I.4. Metrics of Pacific Northwest River BIBI and Scoring Criteria (Celedonia 2004)

		Scoring criteria		
Metric	Response	0	5	10
Richness				
Number of Ephemeroptera taxa	Decrease	≤ 5	6 - 8	≥ 9
Number of Plecoptera taxa	Decrease	≤ 2	3 - 5	≥ 6
Composition				
Percent Hydropsychidae	Increase	> 20	5 - 20	< 5
Tolerance				
Number of intolerant taxa	Decrease	≤ 5	6 - 11	≥ 12
Percent tolerant individuals	Increase	> 25	10 - 25	< 10
Feeding Ecology				
Percent gatherers	Decrease	< 35	35 - 55	> 55
Number of detritus shredder taxa	Decrease	≤ 1	2	≥ 3
Habit				
Number of clinger taxa	Decrease	≤ 2	3 - 5	≥ 6
Number of sprawler taxa	Decrease	≤ 2	3 - 5	≥ 6

I.2 Results

The results of the bio-monitoring efforts are shown in Table I.5. Figures I.1 and I.2 show the stream and river BIBI scores separately. Figure I.3 shows a comparison of the river and stream BIBI scores.

Stream and river BIBI scores are summarized in Table I.5. Using the stream-based BIBI, very few sites had a *good* level of biological integrity and no sites could be considered *excellent* or unimpacted (Figure I.1). Most sites were *fair* or *poor*. Similarly, applying the river BIBI, no sites were found to be *excellent* (natural or unimpacted) or *good*, and only a few sites were rated *fair* (Figure I.2). Most sites were classified as having *poor* biological integrity, with one site rated as *very poor*. The two classification systems appear to be inconsistent in their ratings of stream condition (Figure I.3).

Table I.5. Sample Date and Stream and River BIBI Scores for the Grays River Watershed

Sample Site	Sample Date	Stream BIBI Score	River BIBI Score
Alder Creek	9/29/2004	40	55
Blaney Creek	9/30/2004	44	65
Fossil Creek	9/30/2004	26	60
7220 Tributary Creek	9/29/2004	34	55
7230 Tributary Creek	9/29/2004	34	40
East Fork Grays River	9/30/2004	42	55
North Fork Grays River	10/1/2004	38	50
South Fork Grays River	10/2/2004	36	50
West Fork Grays River (at WDFW hatchery)	9/30/2004	32	60
Grays River Upper Mainstem (upstream of South Fork confluence)	9/29/2004	34	55
Grays River Upper Mainstem (downstream of South Fork confluence)	9/30/2004	22	70
Grays River Middle Mainstem (canyon mouth)	9/30/2004	26	60
Grays River Middle Mainstem (Gorley)	9/30/2004	28	60
Grays River Side-Channel (Gorley)	9/30/2004	24	50
Grays River Middle Mainstem (Puddicombe)	9/29/2004	28	60
Grays River Lower Mainstem (SR 4)	9/30/2004	24	50

I.3 Discussion

Because only a single year of biological monitoring was conducted in 2004, it is difficult to draw definitive conclusions from the data available. However, these limited observations suggest that the cumulative effects of human land-use activities in the watershed have compromised the biological integrity of the Grays River ecosystem. In addition, these data provide a *baseline* for future monitoring and can be used to evaluate trends in biological integrity. This information would be useful in evaluating the effectiveness of watershed management policies, timber-harvest mitigation practices, and any restoration activities implemented as a result of the recommendations of this study.

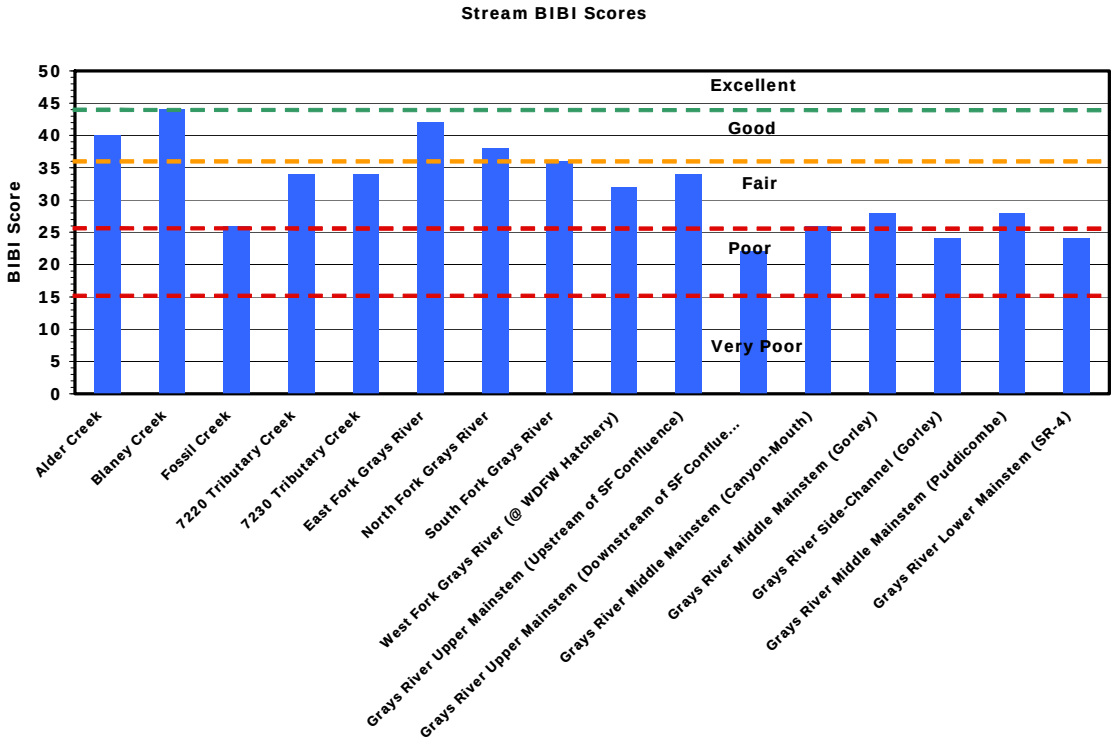

Figure I.1. 2004 Stream BIBI Scores for Grays River

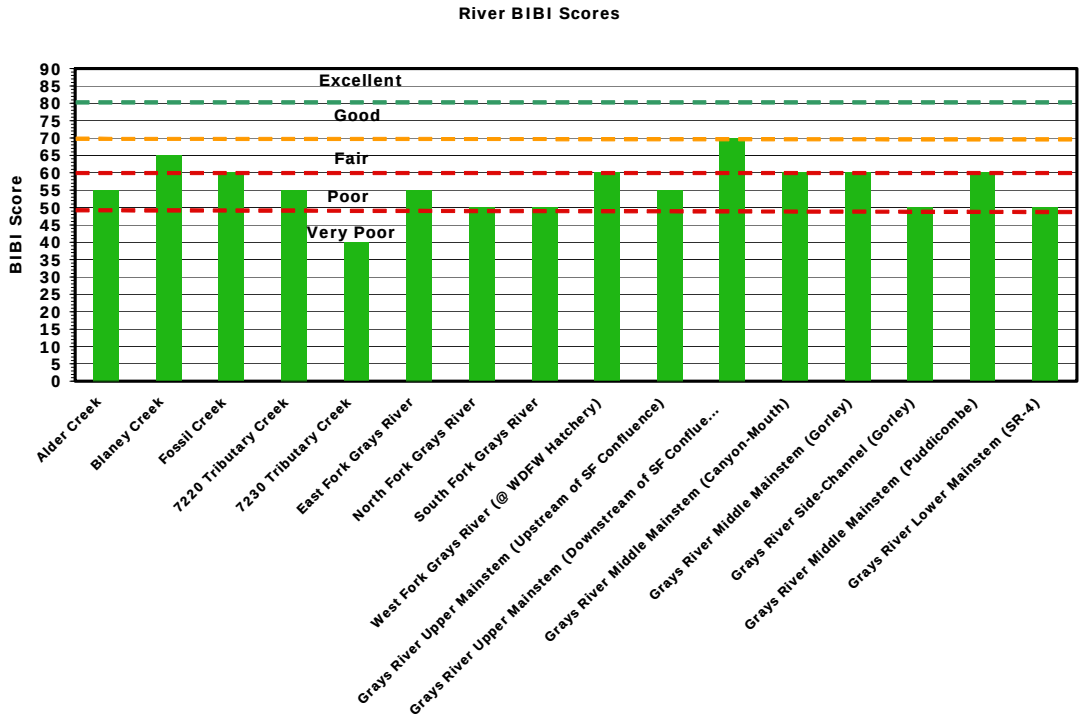

Figure I.2. 2004 River BIBI Scores for the Grays River Watershed

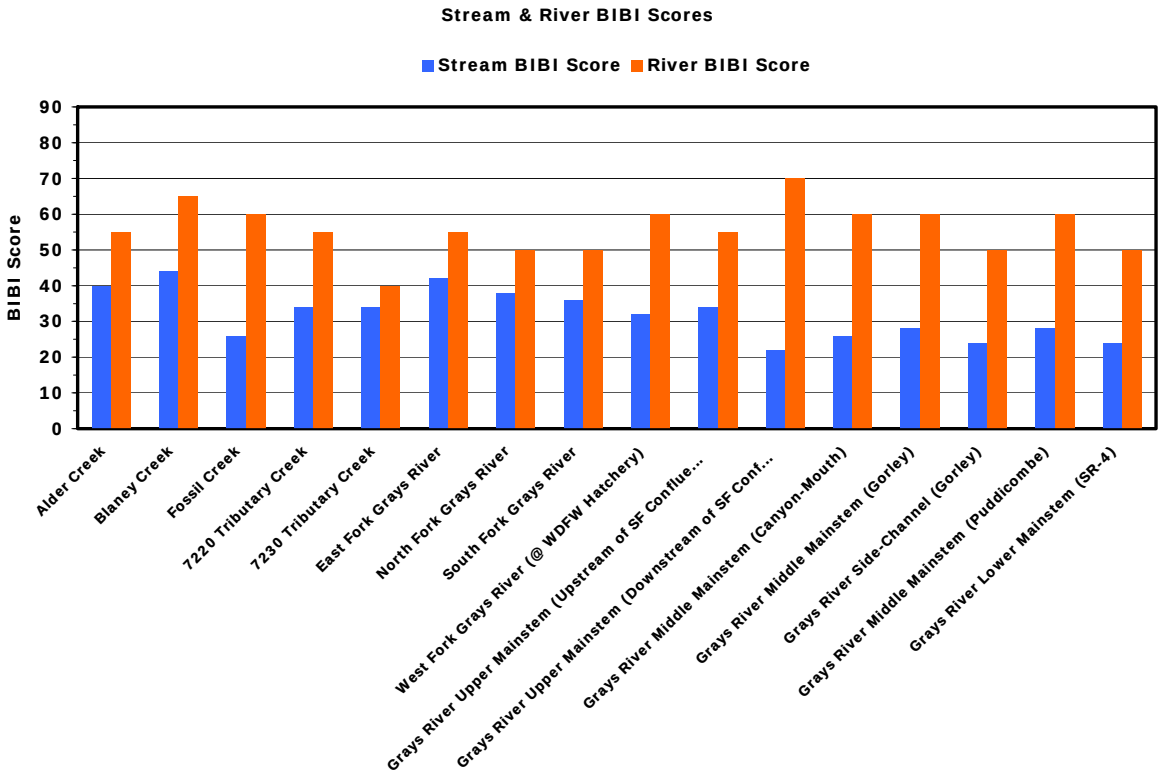

Figure I.3. Stream and River BIBI Scores

I.4 References

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Appendix J

Photographic Logs of the Grays River Watershed

J.1 Grays River Historic Flooding

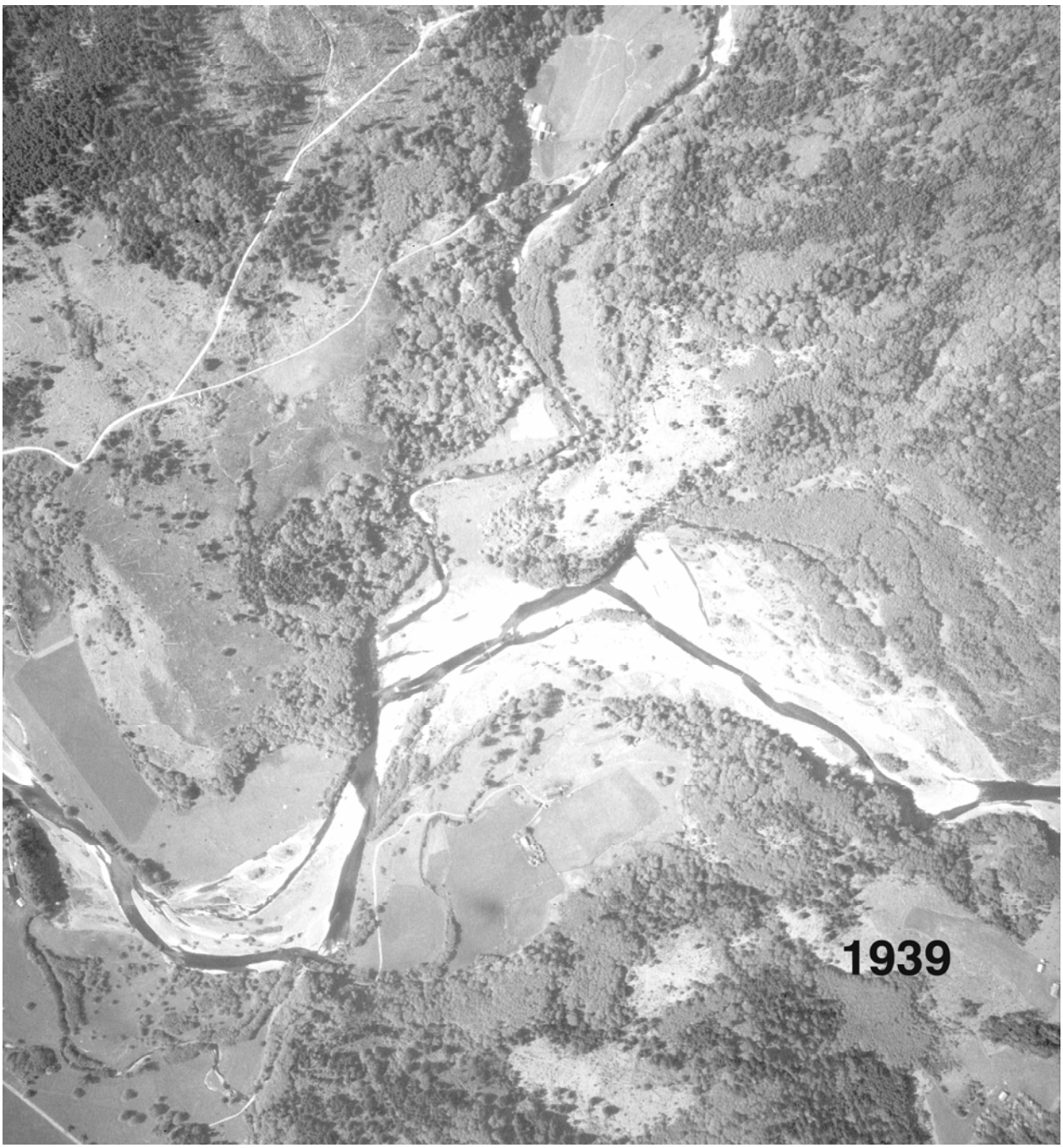
Photographs courtesy of Marie Fauver, Grays River, Washington.

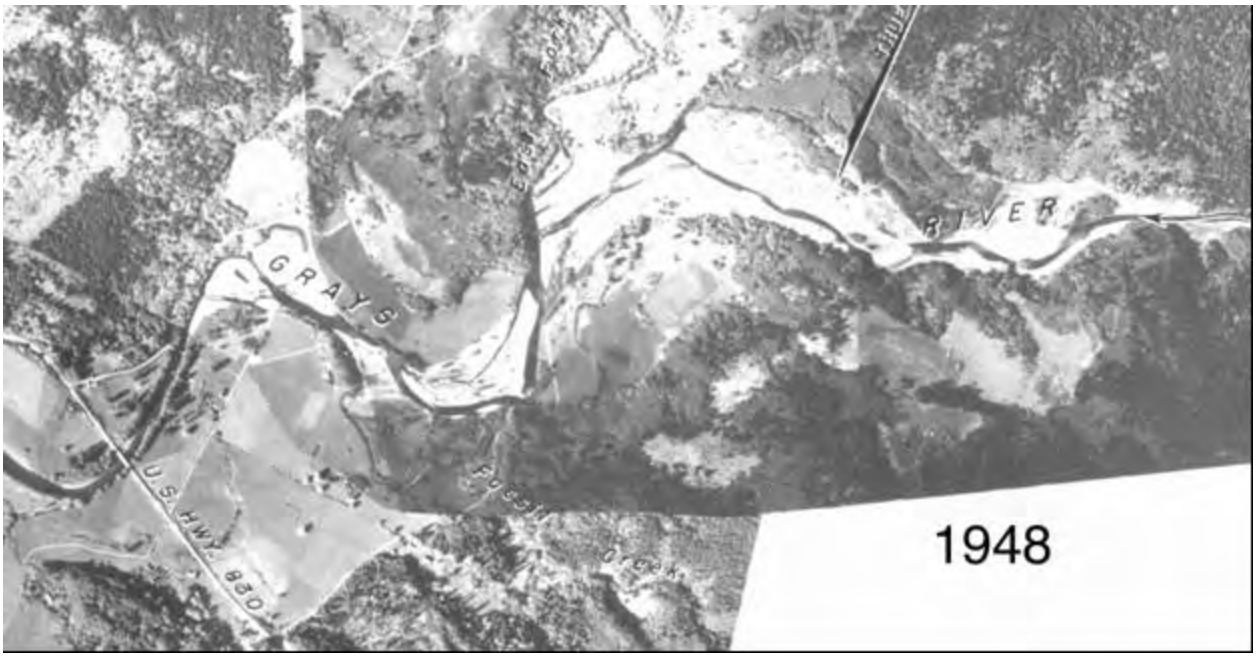
Photographs courtesy of Marie Fauver, Grays River, Washington.

Photographs courtesy of Marie Fauver, Grays River, Washington.

J.2 Grays River Response Reach Aerial Photos

This section provides a history of the Grays River response reach as viewed from the air (photos courtesy of the Washington Department of Fish and Wildlife).

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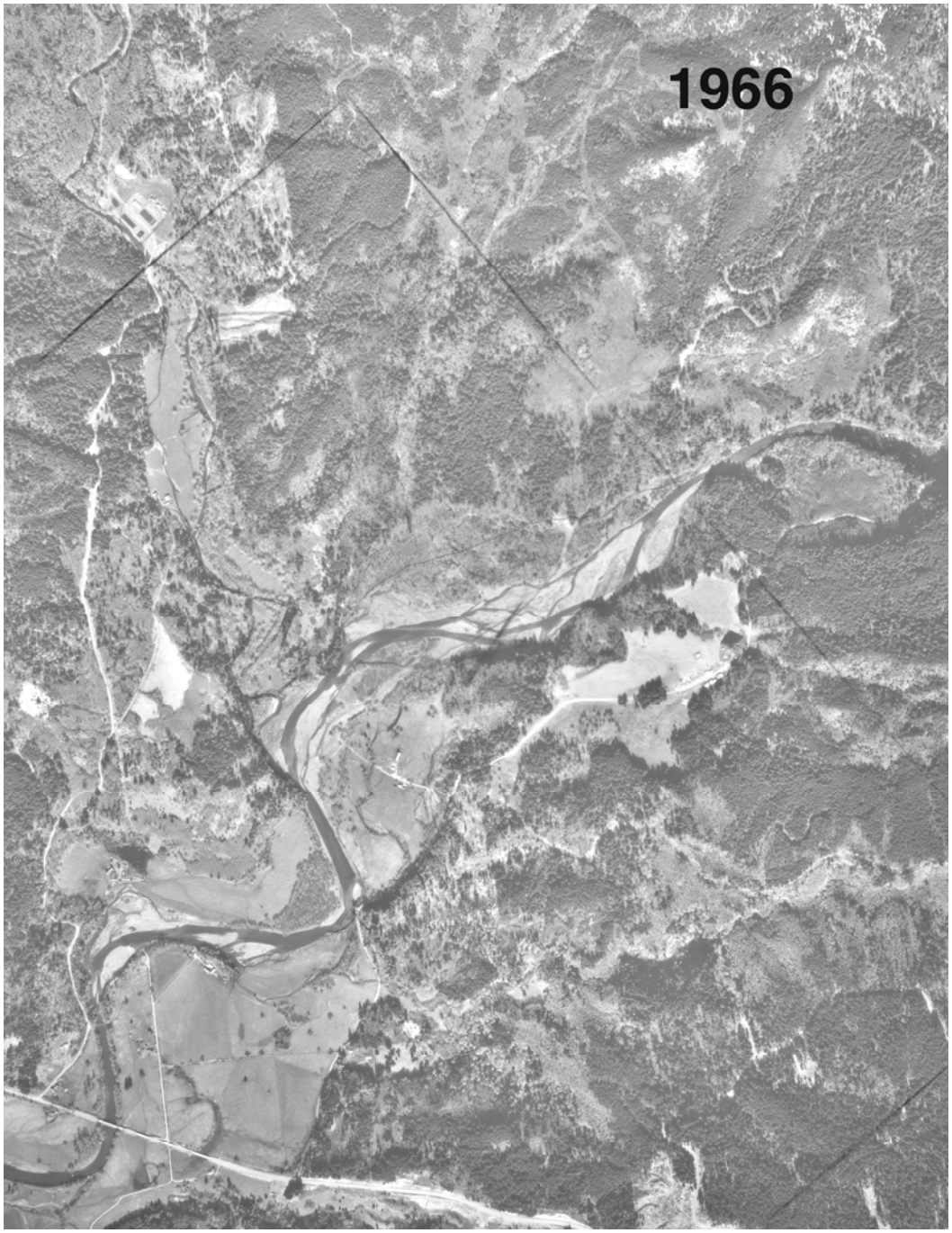
1948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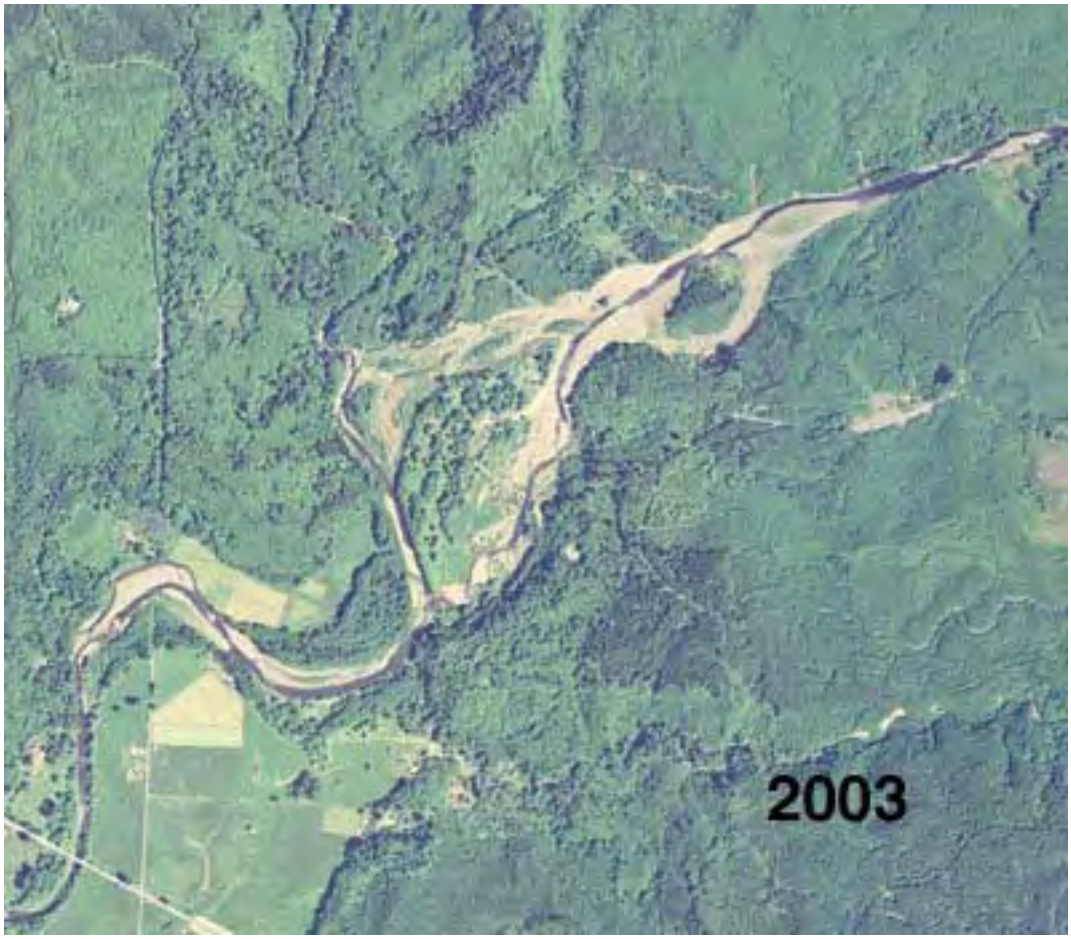
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J.3 Grays River Field

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