**Spawning Habitat Studies
of
Hanford Reach Fall Chinook Salmon (*Oncorhynchus tshawytscha*)**

Final Report

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Summary

The Pacific Northwest National Laboratory conducted this study for the Bonneville Power Administration (BPA) with funding provided through the Northwest Power and Conservation Council^(a) and the BPA Fish and Wildlife Program. The study was conducted in the Hanford Reach of the Columbia River. The goal of study was to determine the physical habitat factors necessary to define the redd capacity of fall Chinook salmon that spawn in large mainstem rivers like the Hanford Reach and Snake River. The study was originally commissioned in FY 1994 and then recommissioned in FY 2000 through the Fish and Wildlife Program rolling review of the Columbia River Basin projects. The work described in this report covers the period from 1994 through 2004; however, the majority of the information comes from the last four years of the study (2000 through 2004). Results from the work conducted from 1994 to 2000 were covered in an earlier report.

More than any other stock of Pacific salmon, fall Chinook salmon (*Oncorhynchus tshawytscha*) have suffered severe impacts from the hydroelectric development in the Columbia River Basin. Fall Chinook salmon rely heavily on mainstem habitats for all phases of their life cycle, and mainstem hydroelectric dams have inundated or blocked areas that were historically used for spawning and rearing. The natural flow pattern that existed in the historic period has been altered by the dams, which in turn have affected the physical and biological template upon which fall Chinook salmon depend upon for successful reproduction. Operation of the dams to produce power to meet short-term needs in electricity (termed power peaking) produces unnatural fluctuations in flow over a 24-hour cycle. These flow fluctuations alter the physical habitat and disrupt the cues that salmon use to select spawning sites, as well as strand fish in near-shore habitat that becomes dewatered. The quality of spawning gravels has been affected by dam construction, flood protection, and agricultural and industrial development. In some cases, the riverbed is armored such that it is more difficult for spawners to move, while in other cases the intrusion of fine sediment into spawning gravels has reduced water flow to sensitive eggs and young fry.

Recovery of fall Chinook salmon populations may involve habitat restoration through such actions as dam removal and reservoir drawdown. In addition, habitat protection will be accomplished through set-asides of existing high-quality habitat. A key component to evaluating these actions is quantifying the salmon spawning habitat potential of a given river reach so that realistic recovery goals for salmon abundance can be developed. Quantifying salmon spawning habitat potential requires an understanding of the spawning behavior of Chinook salmon, as well as an understanding of the physical habitat where these fish spawn. Increasingly, fish biologists are recognizing that assessing the physical habitat of riverine systems where salmon spawn goes beyond measuring microhabitat like water depth, velocity, and substrate size. Geomorphic features of the river measured over a range of spatial scales set up the physical template upon which the microhabitat develops, and successful assessments of spawning habitat potential incorporate these geomorphic features.

What are the geomorphic features of rivers that control the amount of fall Chinook salmon spawning habitat? Ideally we could go to historic fall Chinook salmon production areas (e.g., in the upper Snake River above Hells Canyon Dam) and study the geomorphic features of spawning habitat in its pre-impacted condition. This information could then be used to develop a model of productive spawning

(a) Formerly known as the Northwest Power Planning Council; the Council name was changed in 2003 to emphasize the conservation aspect of its energy and fish and wildlife responsibilities.

habitat, and the model could then be applied to either current production areas for purposes of long-term protection, or used to make restoration decisions in impacted habitat. However, much of the historic fall Chinook salmon production areas (especially in the upper Snake River above Hells Canyon Dam) are now inundated by reservoirs and not available for study.

Although we cannot make direct measurements of the geomorphic features in these historic production areas, we have been able to use reach-scale (1–10 km) data to propose a conceptual spawning habitat model based on geomorphic features across a range of spatial scales. The geomorphic template upon which fall Chinook salmon historically spawned consisted of low-gradient reaches that contained unconsolidated river sediment where groundwater and river water interacted within well-developed hyporheic habitats. These hyporheic habitats were present in historic spawning areas because there was an abundance of channel bed features that promoted surface water exchange with the riverbed. While we have a general model of the reach-scale geomorphic features in historic fall Chinook salmon production areas, we do not have specific, quantifiable metrics that link these geomorphic features to the micro-habitat characteristics to which fall Chinook salmon respond when they select redd sites. For this reason, we initiated this study in the Hanford Reach.

The Hanford Reach of the Columbia River is as close in characteristics to the historic production areas as any that currently exist in the Columbia River Basin. The Hanford Reach contains the appropriate mix of habitat features that fall Chinook salmon select during spawning, including unconsolidated sediment, low water surface gradient, and an abundance of islands and bars that promote the interaction of surface and ground waters within hyporheic habitats. These geomorphic features on the river landscape create a mosaic of habitats with different hydrological, physical, chemical, and biotic properties often found in healthy river ecosystems. The Hanford Reach fall Chinook salmon population was selected as a surrogate population for studying other fall Chinook salmon stocks, including Snake River stocks that spawn in the Snake River below Hells Canyon Dam. The Reach is an appropriate study area because adult fall Chinook salmon from the Hanford Reach and Snake River exhibit similar life history characteristics, and the two river reaches are large, mainstem alluvial rivers that exist in an unimpounded state. The Hanford Reach population is one of a few salmon populations in the Pacific Northwest that are healthy, while Snake River fall Chinook salmon are now listed as threatened under the Endangered Species Act.

We had three primary objectives for this study. The first objective was to determine the relationship between physical habitats at different spatial scales and fall Chinook salmon spawning locations. The second objective was to estimate the fall Chinook salmon redd capacity for the Reach. The third objective was to suggest a protocol for determining preferable spawning reaches of fall Chinook salmon. To ensure that we collected physical data within habitat that was representative of the full range of potential spawning habitat, the study area was stratified based on geomorphic features of the river using a two-dimensional river channel index that classified the river cross section into one of four shapes based on channel symmetry, depth, and width. We found that this river channel classification system was a good predictor at the scale of a river reach (~1 km) of where fall Chinook salmon would spawn. Using this two-dimensional river channel index, we selected study areas that were representative of the geomorphic classes. A total of nine study sites distributed throughout the middle 27 km of the Reach (study area) were investigated. Four of the study sites were located between river kilometer 575 and 580 in a section of the river where fall Chinook salmon have not spawned since aerial surveys were initiated in the 1940s; four sites were located in the spawning reach (river kilometer [rkm] 590 to 603); and one site was located upstream of the spawning reach (rkm 605).

One of the tasks necessary to accomplish the first objective of the study was to confirm that fall Chinook salmon spawned in similar locations as in past years and that the spatial pattern of the redds was clustered. Spawning in clusters suggests that adults are seeking specific features of the habitat in the Reach. To accomplish this task, we used aerial photography from 1994 and 1995 to document weekly spawning patterns. Our hypothesis was that the weekly pattern of redd placement would be clustered at the spatial scale of the Hanford Reach but randomly distributed within the two spawning areas. Two spatial analysis methods were used to study the distribution of salmon redds. These methods included the calculation of the area within a convex hull bounding the salmon redds for each consecutive week and the use of nearest-neighbor analysis. The convex hull technique suggested that fall Chinook salmon spawned in similar locations each year, and they selected the boundary of the spawning area early in the spawning period. In other words, there appear to be specific features within areas of the Reach that adults select when they return to spawn, and the boundaries of these areas do not change much from one year to the next. It would appear from the results that the features they are selecting are fairly stable over spawning seasons. The nearest-neighbor analysis suggested that, for the most part, the redds within the clusters are not random but instead uniformly distributed. While there were some subtle differences between the two sites and two years, the results suggested that as the spawning areas filled in over time, the maximum distances between redds went from about 30 m to about 13 m while the shorter distances changed only slightly, staying around 3 to 5 m. This uniform distribution within the redd clusters suggests that females are occupying and defending redd sites, maintaining a buffer around their individual redd sites.

Once we verified that redds were clustered, we began looking at the physical spawning habitat within the nine study sites. Our first approach was to expand upon an earlier study where we developed logistic models of spawning habitat using water depth, velocity, substrate, and slope. Separate models were constructed for 1994, 1995, and 2001 using both static variables as well as hydraulic variables that changed over time as discharge changed. Results were similar to those of the previous study; that is, water velocity and water depth were important discriminators of spawning habitat at all sites and in all years. Models of habitat selection using water velocity and water depth correctly predicted about 70% of the individual spawning areas. Although there were differences in habitat predicted to be used and the persistence of the correct depth and velocity, addition of variables that represented these dynamic conditions in a revised logistic regression model did not improve predictions. The results suggest that there are some characteristics of the physical habitat within spawning areas that are different between spawning and non-spawning locations that were not included in these simple models.

These findings led to us investigate other geomorphic features within the nine study sites, including riverbed substrate grainsize characteristics and groundwater–surface water interactions. Substrate grain-size was assessed for both surface and subsurface material. In addition to characterizing grainsize, we also measured substrate permeability. Our results suggested that hydraulic and substrate characteristics among and within sites were too variable to be useful in predicting whether individual sites would be used for spawning. However, geomorphic variables at the scale of river reaches were different between spawning and non-spawning reaches (i.e., the four sites in spawning areas versus the four sites downstream of spawning areas). Specifically, surface substrate in the non-spawning reach was composed of large cobble and boulders as compared to the spawning reach. The content of fine silt and coarse sand was about twice as high in non-spawning sites as in spawning sites. The permeability, specific discharge, and vertical hydraulic gradient were all higher in spawning reach than in non-spawning reach, with permeability nearly twice as high and specific discharge more than three times as high. As the amount of coarse sand and silt increased, the permeability and specific discharge declined. There are several

possible explanations for the differences in substrate quality between the spawning and non-spawning reach, including the influence of erosion from the White Bluffs; low bed turnover from a lack of historic spawning; closer proximity of the fine-grained clay-like Ringold Formation to the riverbed surface; and intrusion of fine material into interstitial spaces due to surface water fluctuations.

In addition to assessing substrate grainsize and permeability, we also looked at the interaction of groundwater and surface water within hyporheic habitats. Our approach for this part of the study was to employ the use of data loggers that recorded the temperature and water levels of both the river and the hyporheic zone over long time periods (e.g., the entire spawning period). There was no difference in the temperature and vertical hydraulic gradient of spawning and non-spawning sites when measured over continuous periods that included the entire and partial segments of the spawning period. Temperature of the river fluctuated approximately 1–2°C over a 24-hour cycle. The temperature fluctuations were concomitant with fluctuations of water levels caused from power peaking operations at Priest Rapids Dam. Temperature of the hyporheic zone mimicked that of the river, and differences were normally less than 1°C. Consequently, there was no difference in predicted hatch and emergence dates using river and hyporheic temperatures.

While flow fluctuations did not affect the difference in temperature between the hyporheic zone and the river, the vertical hydraulic gradient between the river and the hyporheic zone did respond to fluctuations in flow. At sites where riverbed permeability was relatively low, water level of the hyporheic zone showed a lag in response to changes in river stage, whereas in sites where the riverbed permeability was relatively high, there was no lag. When a lag occurred, the pressure differential between the river head and the hyporheic head resulted in a hysteretic response. This hysteretic response was more pronounced as riverbed permeability decreased, showing that the low permeability inhibited movement of river water into the riverbed. In general, the pronounced hysteretic responses occurred in the non-spawning locations rather than in the spawning locations. Lower interstitial velocities within riverbed sediment of the non-spawning areas may affect survival to emergence of fall Chinook salmon embryos; however, we did not measure this directly.

Our second objective of the study was to assess the redd capacity of the Hanford Reach. While previous investigations had established a qualitative relationships between geomorphic features and fall Chinook salmon spawning locations, there was no quantitative link between geomorphic features and redd capacity. We accomplished this objective by first classifying existing habitat by the geomorphic form of the river channel and then used this information to make an estimate on quantity of habitat that would potentially fall within these geomorphic characteristics. The redd capacity for the Hanford Reach study area (i.e., the middle 27 km) was estimated to be 8,989 to 23,794 redds. This estimate is for the existing and “expanded” habitat—habitat that meets the geomorphic model but is not currently being used. In the resulting estimate is a redd density of approximately 34 redds/hectare, which is a conservative estimate compared to previous investigations. We extrapolated this redd capacity to the entire Hanford Reach based on the proportional spawning inside and outside the study area and then applied an adult-to-redd ratio to estimate escapement. Our results indicate that the Hanford Reach could support an average of approximately 92,885 adults (range 50,937 to 134,833) if all potential habitat were used. For comparison, average escapement from 1990 to 2005 was 51,062 (range 27,012 to 89,312).

This estimate needs to be considered just that—an estimate. The estimate is based on a relatively simple geomorphic model of channel morphology in areas where existing fall Chinook salmon spawn. Our results from existing spawning areas showed very good correlation between spawning locations and

geomorphic features of the river channel, including longitudinal and lateral slope of the riverbed. The geomorphic model predicted about one third of the expanded habitat area would occur downstream of rkm 580, which is the lower extent of current spawning within our study area. Results from our investigation of sediment grainsize and permeability determined that this reach was lower in permeability and possessed more fine silts and coarse sands than the spawning reach. Our simple geomorphic model of channel morphology did not take into consideration substrate quality. We do not know if substrate quality within this section of the river would support successful egg incubation. We did not measure embryo survival as part of this study, and inclusion of substrate quality results into simple models of embryo survival was inconclusive. However, it is unlikely that sediment permeability within this lower segment is sufficiently high to support egg incubation. Thus, we believe that the expanded habitat estimate for the area downstream of rkm 580 is likely overly optimistic, and in reality, the redd capacity we present for the Hanford Reach is higher than what the Reach could actually support.

Our last objective was to develop a protocol that could be applied to other river reaches. The protocol that we envision for evaluating fall Chinook salmon spawning habitat involves an approach that begins at the watershed scale and becomes increasingly detailed at progressively finer scales, culminating in estimates of habitat quality at the scale of individual redds. The specific variables measured for habitat quantity and quality will be dictated by the management and research questions specific to a given area. At all scales of inquiry, the physical characteristics of a given area act as controlling factors that set the template within which any potential habitat is ultimately realized. It is imperative that conclusive statements regarding the potential fall Chinook salmon spawning habitat quality and quantity of a river reach be developed in the context of the template provided by the reach. This protocol for evaluating fall Chinook salmon spawning habitat provides a framework for research and management actions concerning the recovery of fall Chinook salmon in the Columbia River basin and western North America.

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Abbreviations

ANOVA	analysis of variance (ANOVA)
ATU	accumulated thermal unit(s)
BPA	Bonneville Power Administration
χ^2	chi-square
cm	centimeter
DEM	digital elevation model
ESRI	Environmental Systems Research Institute
GIS	geographic information system
GPS	global positioning system
ha	hectare(s)
kfcs, KFCS	thousand feet per second
km	kilometer
LB	left bank
LIDAR	light detection and ranging
m	meter
m ²	square meter
m ³	cubic meter
MASS2	Modular Aquatic Simulation System 2-D
mm	millimeter
PNNL	Pacific Northwest National Laboratory
RB	right bank
<i>RPI</i>	riffle proximity index
rkm	river kilometer
s	second
SHOALS	Scanning Hydrographic Operational Airborne Lidar Survey
SI	sand index
<i>TDI</i>	thalweg distance index
TIN	triangulated irregular network
USACE	U.S. Army Corps of Engineers
USGS	U.S. Geological Survey
VHG	vertical hydraulic gradient

Contents

Summary	iii
Acknowledgments.....	ix
Abbreviations.....	xi
1.0 Introduction.....	1.1
2.0 Fall Chinook Salmon Life History and Study Sites	2.1
2.1 General Characteristics	2.1
2.2 Fall Chinook Salmon Life History	2.2
2.3 Selection of Study Sites.....	2.5
3.0 Spatial and Temporal Pattern Analysis of Fall Chinook Salmon Spawning.....	3.1
3.1 Methods.....	3.2
3.2 Results	3.4
3.3 Discussion	3.16
4.0 Assessment of Fall Chinook Salmon Spawning Habitat Using Standard Habitat Characteristics.....	4.1
4.1 Methods.....	4.1
4.1.1 Study Site Selection.....	4.1
4.1.2 Data Collection and Summary	4.1
4.1.3 Logistic Regression Analysis.....	4.3
4.1.4 Dynamic Analysis and Additional Logistic Regression Modeling.....	4.6
4.2 Results	4.8
4.2.1 General Results	4.8
4.2.2 Logistic Regression Analysis with Only Static Variables	4.13
4.2.3 Dynamic Analysis.....	4.16
4.3 Discussion	4.25
4.4 Conclusion.....	4.28
5.0 Assessment of Substrate Quality of Fall Chinook Salmon Spawning Habitat in the Hanford Reach	5.1
5.1 Methods.....	5.3
5.1.1 Sediment Grainsize	5.3
5.1.2 Riverbed Permeability	5.6
5.1.3 Relationship Between Grainsize Distribution and Survival to Emergence	5.10
5.2 Results	5.11
5.2.1 Sediment Grainsize	5.11

5.2.2	Riverbed Permeability	5.20
5.2.3	Relationship Between Grainsize Distribution and Survival to Emergence	5.24
5.3	Discussion	5.25
5.3.1	Sediment Grainsize	5.25
5.3.2	Riverbed Permeability	5.28
5.3.3	Grainsize Versus Spawning and Survival to Emergence	5.29
5.4	Conclusions	5.31
6.0	Groundwater and Surface Water Interactions Within Fall Chinook Salmon Spawning Sites in the Hanford Reach.....	6.1
6.1	Methods.....	6.2
6.1.1	Measurements of Vertical Hydraulic Gradient, Temperature, and Flux.....	6.2
6.2	Data Analysis	6.5
6.2.1	Objective 1 - Site-to-Site Comparisons of Temperature, Vertical Hydraulic Gradient, and Flux	6.5
6.2.2	Objective 2 – Effect of Hydropower Operations on Groundwater–Surface Water Interactions.....	6.6
6.2.3	Objective 3 – Hatching and Emergence Dates Using Bed and River Temperatures	6.6
6.3	Results	6.8
6.3.1	Objective 1 – Site-to-Site Comparisons of Temperature, Vertical Hydraulic Gradient, and Flux	6.8
6.3.2	Objective 2 – Effect of Hydropower Operations on Groundwater–Surface Water Interactions.....	6.18
6.3.3	Objective 3 – Hatching and Emergence Dates Using Bed and River Temperatures	6.25
6.4	Discussion	6.29
6.4.1	Site to Site Comparisons of Temperature, Vertical Hydraulic Gradient, and Flux	6.29
6.4.2	Effect of Hydropower Operations on Groundwater–Surface Water Interactions.....	6.31
6.4.3	Hatching and Emergence Dates Using Bed and River Temperatures.....	6.32
7.0	Fall Chinook Salmon Redd Capacity in the Hanford Reach.....	7.1
7.1	Methods.....	7.2
7.1.1	Potential Spawning Habitat	7.2
7.1.2	Redd Capacity.....	7.4
7.2	Results	7.5
7.2.1	Existing Spawning Habitat	7.5

7.2.2 Expansion Spawning Habitat.....	7.5
7.3 Discussion	7.10
8.0 Protocol for Evaluating Fall Chinook Salmon Spawning Habitat	8.1
8.1 Protocol	8.1
8.1.1 Watershed Scale.....	8.2
8.1.2 Reach Scale.....	8.2
8.1.3 Site Scale.....	8.3
8.2 Conclusion.....	8.4
9.0 Conclusions	9.1
10.0 References	10.1
Appendix A – Individual Study Site Maps	A.1
Appendix B – Grainsize Data	B.1
Appendix C – Hydraulic Conductivity Results.....	C.1

Figures

2.1	Hanford Reach Area and Locations of Data Collection Sites	2.1
2.2	Ten Fall Chinook Salmon Spawning Areas	2.4
2.3	Representative Channel Cross Sections	2.7
2.4	Individual Study Sites.....	2.9
2.5	Three-Year Average Redds per 0.1 km from the Hanford Reach of the Columbia River	2.10
3.1	Location of Areas for Analysis of Spatial and Temporal Patterns of Redd Distribution	3.2
3.2	Redd Distribution by Week for 1994	3.5
3.3	Redd Distribution by Week for 1995	3.6
3.4	TINs Based on Redd Locations in Area 7 for Each Week in 1994	3.7
3.5	TINs Based on Redd Locations in Area 7 for Each Week in 1995	3.8
3.6	TINs Based on Redd Locations in Upstream Segment of Area 4 for Each Week in 1994.....	3.9
3.7	TINs Based on Redd Locations in Upstream Segment of Area 4 for Each Week in 1995.....	3.10
3.8	TINs Based on Redd Locations in Downstream Segment of Area 4 for Each Week in 1994.....	3.11
3.9	TINs Based on Redd Locations in Downstream Segment of Area 4 for Each Week in 1995.....	3.12
3.10	Percentage of Total Area Occupied for Area 7	3.13
3.11	Percentage of Total Area Occupied for Upstream and Downstream Area 4.....	3.13
3.12	Spatial Point Pattern Analysis of Fall Chinook Salmon for Area 7 in 1994, by Week	3.14
3.13	Spatial Point Pattern Analysis of Fall Chinook Salmon for Area 7 in 1995, by Week	3.15
3.14	Spatial Point Pattern Analysis of Fall Chinook Salmon for Area 4 in 1994, by Week	3.16
3.15	Spatial Point Pattern Analysis of Fall Chinook Salmon for Area 4 in 1995, by Week	3.17
4.1	Distribution of the Shortest Distance Between Nearest Spawning Habitat Cells.....	4.4
4.2	Hourly Discharge Rates of Priest Rapids Dam During October and November 1994, 1995, and 2001	4.7
4.3	Sample Map Showing Consistency of Spawning for Site 64-RB	4.9
4.4	Static Variables for Spawning and Non-Spawning Locations, by Year.....	4.12
4.5	McFadden's ρ^2 as a Function of the Number of Terms in the Logistic Regression Models for the Three Years.....	4.14
4.6	Maps for Study Area 64-RB Showing Locations of Predicted Spawning Cells and the Actual Spawning Cells for 1994, 1995, and 2001	4.17
4.7	Mean Frequency Plots for Velocity and Depth in 1994, 1995, and 2001 for Habitat Cells Predicted To Be Spawning Cells	4.20

4.8	Box Plots of Velocity and Depth Values for Spawning and Non-Spawning Locations for 1994.....	4.21
4.9	Dynamic Variables <i>VEL1</i> , <i>VEL2</i> , and <i>VEL3</i> by Spawning and Non-Spawning Sites in Year 1994	4.23
4.10	McFadden's ρ^2 Versus Number of Terms Added in Logistic Regression	4.25
5.1	Freeze Core Process	5.3
5.2	Study Areas Where Surface and Subsurface Samples Were Collected.....	5.5
5.3	Piezometer Screens.....	5.8
5.4	Piezometer Installation	5.8
5.5	Slug Test Process.....	5.9
5.6	Grainsize Distribution of 43 Individual Freeze Cores Collected at Nine Sites in the Hanford Reach of the Columbia River	5.11
5.7	Freeze Core 64-2 Sampled in 2001 from the Riverbed at Site 64	5.12
5.8	Percent Finer Than Curve for Composite Subsurface Grainsize Data Collected by Freeze Coring at Nine Sites in the Hanford Reach in 2000 and 2001	5.13
5.9	Grainsize Distribution for Composite Subsurface Sediment Sampled in 2000 and 2001 at Nine Sites in the Hanford Reach	5.13
5.10	Percentage of Total Subsurface Sample Weight Categorized into Sediment Types Using the Wentworth Scale	5.14
5.11	Percent Finer Than Curves for Surface Grainsize Distributions	5.15
5.12	Grainsize Distribution for Surface Sediment Sampled in 2000 and 2002 Using Videography from Nine Study Sites in the Hanford Reach	5.16
5.13	Percent Finer Than Curves for Subsurface Samples of the Riverbed at Non-Spawning and Spawning Sites.....	5.17
5.14	Percentage of the Total Subsurface Sample of the Riverbed Collected from Spawning and Non-Spawning Sites in the Hanford Reach	5.18
5.15	Percentage of Total Surface Sample of the Riverbed Classified Using the Wentworth Scale	5.19
5.16	Percent Finer Than Curves for Surface Grainsize Distributions in Non-Spawning and Spawning Sites.....	5.19
5.17	Hydraulic Conductivity, Log Hydraulic Conductivity, Vertical Hydraulic Gradient, and Specific Discharge Measured at Nine Study Sites in 2000, 2001, and 2003	5.21
5.18	Hydraulic Conductivity, Specific Discharge, and Vertical Hydraulic Gradient Measured at Spawning and Non-Spawning Sites in the Hanford Reach in 2001, 2002, and 2003	5.23
5.19	Relationship Between the Percentage of the Total Sample Classified as Either Silt or Coarse Sand Versus the Hydraulic Conductivity of Riverbed Sediments at Eight Sites.....	5.24
6.1	Monthly Average Hyporheic Temperatures at Eight Sites.....	6.9

6.2	Monthly Average Temperature Differences Between River and Hyporheic Zone at Eight Sites from 2001–2004	6.9
6.3	River Temperature	6.11
6.4	Continuous River Temperatures	6.12
6.5	Hyporheic Temperature	6.13
6.6	Continuous Hyporheic Temperatures	6.14
6.7	Temperature Differences	6.15
6.8	Monthly Mean Vertical Hydraulic Gradient Between River and Hyporheic Zone at Nine Sites in the Hanford Reach from October 2001 through October 2004.....	6.18
6.9	Vertical Hydraulic Gradient	6.19
6.10	Temperature of Hyporheic Zone and River Compared to River Stage at Site 64 During the Spawning Period in 2003	6.21
6.11	Hyporheic and River Temperature as Compared to River Stage During the Spawning Period in 2001 and 2004 at Site 90	6.22
6.12	An Example of the Times-Series Response of Delta VHG and Delta River Stage During a 5-Day Period in 2004 at a Site Characterized as Highly Variable	6.24
6.13	An Example of the Time-Series Response of Delta VHG and Delta River Stage During a 5-Day Period in 2004 at a Site Characterized as Relatively Stable	6.24
6.14	The Relationship Between VHG and River Stage at Sites 81, 90, and 96 During a 25-Hour Period in 2004	6.26
6.15	Days to Emergence Calculated Based on 1,000 ATU for all Three Years for River and Hyporheic.....	6.29
7.1	Fall Chinook Salmon Spawning Habitat Within the Study Area	7.6
7.2	Riffle Proximity Index <i>RPI</i> for Existing Fall Chinook Salmon Spawning and Non-Spawning Locations	7.7
7.3	Estimates of the Mean Riffle Proximity Index <i>RPI</i> Were Significantly Larger in Existing Fall Chinook Salmon Spawning Locations Than in Non-Spawning Locations	7.8
7.4	The Mean Thalweg Distance Index <i>TDI</i> of Existing Fall Chinook Salmon Spawning Areas Located Along Riffle Cross Sections Was Not Significantly Correlated with the Riffle Proximity Index <i>RPI</i>	7.9
7.5	The Mean Thalweg Distance Index <i>TDI</i> of Existing Fall Chinook Salmon Spawning Areas Located Along Pool Cross Sections Was Significantly Correlated with the Riffle Proximity Index <i>RPI</i>	7.9

Tables

2.1	Fall Chinook Salmon Escapement to the Hanford Reach, 1994 Through 2004.....	2.4
2.2	Criteria for Categorizing Cross Sections Based on F and d^*	2.7
2.3	Summary of the Geomorphic Classification of Sampled Cross Sections.....	2.8
2.4	Nine Study Sites	2.8
3.1	Dates of Aerial Photography	3.3
3.2	Area and Redd Densities for Study Areas	3.4
4.1	Summary of Habitat Cell Types After Reclassifying Spawning Cells.....	4.4
4.2	Habitat Suitability Criteria for the Hanford Reach.....	4.8
4.3	Consistency of Spawning Locations for 1994, 1995, and 2001	4.10
4.4	Univariate Summaries of Static Variables for Spawning and Non-Spawning Locations	4.11
4.5	Results of Sequential Logistic Regression Showing Order of Variable Selection for Each Year	4.13
4.6	Regression Coefficients of the Full and Simplified Models for the Sequential Logistic Regression for Each Year	4.15
4.7	Prediction Success Rates for the Simplified Logistic Regression Model.....	4.16
4.8	Area Belonging to Habitat Classes in Each Study Area.....	4.18
4.9	Classified Spawning Cells for Each Study Site for Each of the Three Years Using the Logistic Regression Model and a Probability Cutoff of 0.5	4.19
4.10	Sequentially Selected Variables with Priority on the First-Order Terms for Years 1994, 1995, and 2001	4.24
4.11	Coefficients for Full and Simplified Models for Years 1994, 1995, and 2001	4.26
4.12	Prediction Success for Revised Logistic Regression Model.	4.27
4.13	Coefficients of Correlation Between the Static Velocity, $VEL0$, and $VEL1$, $VEL2$, $VEL3$ for Years 1994, 1995, and 2001	4.27
5.1	Freeze Core Sediment Samples Collected from Nine Study Sites in the Hanford Reach in 2000 and 2001	5.4
5.2	Piezometers Where Slug Tests Were Conducted in 2000, 2001, and 2003.....	5.7
5.3	Wentworth Particle Size Classification Used to Group Sediment Classes.....	5.10
5.4	Summary Statistics for Composite Subsurface Grainsize Data Collected Using Freeze Cores at Nine Sites in the Hanford Reach in 2000 and 2001	5.14
5.5	Percentage of Coarse Sand, Fine Sand, and Silt, and Sand Index for Subsurface Sediment Sampled from Nine Sites in the Hanford Reach in 2000 and 2001	5.15
5.6	Summary Statistics for Riverbed Surface Grainsize Data Collected Using Underwater Video in the Hanford Reach in 2000 and 2002	5.16
5.7	Sorting Indices for Subsurface Samples of the Riverbed at Non-Spawning and Spawning Sites.....	5.17

5.8	Percentage of the Entire Sample That Are Silts, Fine Sand, and Coarse Sand Within Subsurface Samples of the Riverbed at Spawning and Non-Spawning Sites	5.17
5.9	Surface Sediment Statistics for Non-Spawning and Spawning Sites	5.20
5.10	Permeability, Vertical Hydraulic Gradient, and Specific Discharge Data for All Sites	5.20
5.11	Summary Percentages for “Fines” Based on Riverbed Grainsize Data Collected Using Freeze Cores at Nine Sites in the Hanford Reach in 2000 and 2001	5.24
5.12	Survival to Emergence Estimates for Chinook Salmon Using Various Relationships Reported in the Literature	5.25
5.13	Hydraulic Conductivity Reported in Alluvial Environments Similar to the Hanford Reach	5.28
6.1	Piezometers Where Onset Temperature Loggers Were Deployed from 21 November 1999 to 3 April 2000	6.3
6.2	Distance from Surface of Riverbed to Sensor in the Hyporheic Zone Piezometers at Nine Sites Monitored from 18 October 2001 to 27 October 2004	6.4
6.3	Two Spawning Periods Used to Compare Spawning and Non-Spawning Sites in 2001, 2002, and 2003	6.5
6.4	Summary of Data Availability for Objective 3 for the Period from 15 November Through 15 May	6.7
6.5	Group Mean for Spawning and Non-Spawning Sites from Various Time Periods	6.10
6.6	Mean Hyporheic Temperature by Site During the Spawning Period 15–22 November	6.16
6.7	Mean Temperature Difference During the Spawning Period 15–22 November	6.17
6.8	Mean Survey Vertical Hydraulic Gradient by Site for 15–22 November Time Period	6.20
6.9	Flux Calculations	6.20
6.10	Summary Statistics of River Stage During Five-Day Periods in the Fall During the Years 2001 through 2004	6.23
6.11	The Time Lag Identifies the Shift Required for the Time Series of Delta River Stage Relative to Delta VHG To Achieve Maximum Correlation	6.25
6.12	Dates of Accumulated Temperature Units First Exceeding 500 from 15 November Through 15 May	6.27
6.13	Dates of Accumulated Temperature Units First Exceeding 1,000 from 15 November Through 15 May	6.27
6.14	Difference in Days Between Hatch Calculated Using Temperatures from the River Versus the Riverbed	6.28
6.1	Difference in Days Between Emergence Calculated Using Temperatures from the River Versus the Riverbed	6.28
7.1	Thalweg Distance Index Criteria from Stations Located Along Fall Chinook Salmon Spawning Cross Sections in Pools and Riffles	7.4
7.2	Redd Capacity of Existing Fall Chinook Salmon Spawning Areas Based on Redd Sizes and Inter-Redd Spacing	7.7

7.3	Summary Frequency of Four Bed Form Types in Existing Fall Chinook Salmon Spawning and Non-Spawning Areas	7.8
7.4	Redd Capacity of Expanded Fall Chinook Salmon Spawning Areas Based on Redd Sizes and Inter-Redd Spacing	7.10

1.0 Introduction

Fall Chinook salmon (*Oncorhynchus tshawytscha*) have been affected negatively by the development and operation of the mainstem Columbia River hydroelectric system (Dauble et al. 2003). Many areas where fall Chinook historically spawned within the mainstem Snake and Columbia rivers are now a series of low-velocity impoundments, and access to other habitats is blocked by impassable barriers. Natural flow regimes have been altered due to flood protection and irrigation storage so that now the seasonal maximum flows are not as great and occur later in the spring. Power peaking (i.e., operation of dams to maximize power production that occurs at peak times during the 24-hour cycle) results in rapid flow fluctuations that alter the natural exchange of surface water and groundwater within hyporheic habitats. Flow fluctuations result in stranding of fish in shallow, near-shore habitats. All of these changes have reduced the production capacity for fall Chinook salmon within the Columbia River Basin.

Most fall Chinook salmon carry out their entire freshwater life cycle within the mainstem of the Columbia and Snake rivers rather than in tributary streams. As such, the development and operation of the hydroelectric system has affected them more than the other species of Pacific salmon that use the Columbia River watershed. Restoration of fall Chinook salmon is centered on the protection and creation of mainstem riverine habitat (USACE et al. 1992; USACE 1994). Restoration may occur through a variety of techniques, including dam removal to restore riverine habitats, drawdown of reservoirs to promote tailrace habitat, and/or protection of existing fall Chinook salmon spawning habitat to enable colonization of spawners into unused areas. Predicting the quantity and quality of fall Chinook salmon spawning habitat requires an adequate understanding of the habitat requirements of adult fall Chinook salmon through time and space.

Fisheries managers have a general understanding of where fall Chinook salmon spawned in the Columbia and Snake rivers prior to major hydropower development (Dauble et al. 2003). Fall Chinook salmon spawned over a distance of approximately 600 km in the Columbia River drainage, and over 800 km in the Snake River drainage. Spawning was located in low gradient river reaches that contained unconsolidated river sediment where groundwater and surface water interacted within well developed hyporheic habitats. These hyporheic habitats were present in those reaches because of an abundance of bars, islands, and other geomorphic features that promoted surface water exchange with the riverbed. Present day fall Chinook salmon spawning areas are restricted to a 158 km reach of the Snake River below Hells Canyon Dam (rkm 397) (Groves and Chandler 1999); the 90 km long Hanford Reach of the Columbia River between Priest Rapids Dam (rkm 639) and McNary Dam pool (Dauble and Watson 1997); and the tailraces of some dams in the Snake and Columbia rivers including a fairly significant spawning population (~10,000 annually) in the tailrace of Wanapum Dam (rkm 669) in the mid-Columbia River (McMichael et al. 2004, 2005).

The Hanford Reach population of fall Chinook salmon is one of a few salmon stocks in the U.S. that are as healthy now as in the early 19th century (Huntington et al. 1996). Up to 90% of the fall Chinook salmon that pass McNary Dam (rkm 470) will return to the Hanford Reach to spawn. Reasons for the success of the Hanford Reach population are many, including significant out-planting of sub-yearling fall Chinook salmon juveniles from Priest Rapids salmon hatchery; flow control at Priest Rapids Dam during the spawning period as required by the Vernita Bar Settlement Agreement between Grant County Public Utility District (owners of Priest Rapids Dam) and the regulatory agencies and tribes; and a geomorphic template of the river that still resembles historic fall Chinook salmon production areas (Dauble and

Watson 1997; Dauble and Geist 2000; Dauble et al. 2003). It is this latter feature that we are studying. The Hanford Reach contains the appropriate mix of habitat features that fall Chinook salmon select during spawning, including unconsolidated sediment, low water surface gradient, and an abundance of islands and bars that promote the interaction of surface and ground waters within hyporheic habitats. These geomorphic features on the river landscape create a mosaic of habitats with different hydrological, physical, chemical, and biotic properties often found in healthy biological communities (Stanford et al. 1996, 2006).

Our assessment of spawning habitat requirements of Hanford fall Chinook has been done over several years and at several spatial scales. We have measured habitat at scales that range from alluvial river segment (1 km) to the head of mid-channel bars (100 m) to individual redds (m). Linking coarse-scale measurements of habitat within historic production areas to finer-scale features from current production areas has provided us an opportunity to identify scale-dependent habitat features that are important in predicting fall Chinook salmon spawning locations. Ultimately this approach will result in the development of an evaluation method (i.e., protocol) for determining preferable spawning reaches for fall Chinook salmon.

To make inference to other river reaches, we use the Hanford Reach as a surrogate for optimal fall Chinook salmon spawning habitat. In this report, we describe the spawning habitat features of Hanford fall Chinook salmon using both standard habitat characteristics (i.e., depth, substrate, velocity) and geomorphic characteristics previously found to be important in redd site selection (hyporheic zone water quality, permeability, and grainsize distribution of riverbed sediments). Using this information, we calculate the redd capacity of the Reach under current operations. Understanding the capacity of the Reach will help protect the Hanford Reach fall Chinook salmon core population, which will be important for seeding satellite populations and for making assessments of fall Chinook salmon redd capacity in other rivers throughout the Columbia River Basin.

The objectives of this study were to

- Determine the relationship between physical habitats at different spatial scales and fall Chinook salmon spawning locations.
- Estimate the fall Chinook salmon redd capacity for the Hanford Reach.
- Suggest a protocol for determining preferable spawning reaches for fall Chinook salmon.

This report describes how we accomplished these objectives. The organization of the report is based upon how we implemented the study. Our first task was to select appropriate study sites within the Hanford Reach. To ensure that we collected physical data within habitat that was representative of the full range of potential spawning habitat, the study area was stratified based on geomorphic features of the riverbed channel. Nine specific locations were then selected for study. This process is described in Section 2, which also provides a description of the Hanford Reach and the life history of fall Chinook salmon that spawn in the Hanford Reach.

One of the tasks necessary to accomplish the first objective of the study was to confirm that fall Chinook salmon spawned in locations similar to those in past years and that the spatial pattern of the

redds was clustered. Spawning in clusters suggests that adults are seeking specific features of the habitat in the Reach. To accomplish this task, we used aerial photography from 1994 and 1995 to document weekly spawning patterns. Section 3 describes this effort.

Once we verified that redds were clustered, we began looking at the physical spawning habitat within the nine study sites. Our first approach was to expand upon an earlier study in which we developed logistic models of spawning habitat using water depth, velocity, substrate, and slope. Separate models were constructed for 1994, 1995, and 2001 using both static variables as well as hydraulic variables that changed over time as discharge changed. These models allowed us to identify areas within the Hanford Reach that would not be spawning habitat-limited due to constraints imposed by depth, substrate, lateral slope, and/or velocity (i.e., standard spawning habitat characteristics). Section 4 describes model construction and the results from this effort.

The findings from the modeling of standard habitat characteristics led to us investigate other, non-traditional features of the physical habitat within the nine study sites. The objective of Section 5 and of Section 6 was to explore whether substrate quality (Section 5) and the interaction of groundwater and surface water (Section 6) may offer alternative explanations to the standard spawning habitat features. For assessing substrate quality, we looked at sediment permeability and sediment grainsize. For groundwater–surface water interactions, we looked at how water level fluctuations from hydropower operation would affect cues that adult salmon may use for selecting spawning sites and the hatching and emergence of fry.

The second objective of the study was to determine the fall Chinook salmon redd capacity for the Hanford Reach. We propose a redd capacity that is based on a combination of standard and geomorphic features. This model is described in Section 7 of the report.

Our last objective was to develop a protocol that could be used to evaluate fall Chinook salmon spawning habitat in other river reaches. The protocol that we envision for evaluating fall Chinook salmon spawning habitat involves an approach that begins at the watershed scale and becomes increasingly detailed at progressively finer scales, culminating in estimates of habitat quality at the scale of individual redds. Section 8 is a description of this protocol.

The findings from this study were numerous. We summarize the key findings in a series of conclusions presented in Section 9. Section 10 lists the references cited in this report. Additional detailed supporting information is provided in a series of appendixes.

2.0 Fall Chinook Salmon Life History and Study Sites

The Hanford Reach of the Columbia River provided the area and sites for this study.

2.1 General Characteristics

The Hanford Reach is located in the Columbia Plateau (Figure 2.1). The river channel is mostly unconfined, and much of the riverbed consists of a relatively uniform layer of fluvial sediments (Reidel et al. 1994). Quaternary gravel bars in the Hanford Reach are surrounded by a more extensive alluvial flood plain composed of Pleistocene Hanford formation flood gravels and Pliocene fluvial deposits of the Ringold formation (Jenkins 1922). The Ringold and Hanford formations contain an unconfined aquifer through which the Hanford Reach flows. The unconfined aquifer overlies bedrock composed of Miocene

Figure 2.1. Hanford Reach Area and Locations of Data Collection Sites

Columbia River basalts (Reidel et al. 1994). The Hanford Reach is generally a gaining stream. Short-term changes in the direction of water flow within the riverbed are frequently observed (Geist 2000; Arntzen et al. 2006). As the Columbia flows through the Reach, the bed material varies from boulders and coarse gravel to fine silt and clay. The large range in sediment size is caused in part by the difference in composition between the geologic formations through which the river flows and the way alluvium originating from those formations has been reworked by the river. The Hanford formation is composed largely of coarse gravel, while the upper Ringold Formation exposed in the bed of the Columbia River comprises largely silt and mud-sized sediment (Cannon et al. 2005). The ability of water to flow through the bed is affected largely by which of these types of sediment dominates (Arntzen et al. 2006).

The Hanford Reach appears to have had no major changes in river planform and channel cross-sectional characteristics over the past century (Dauble and Geist 2000). Only small differences in bank migration, river bar shifting, and channel aggradation or degradation have been observed. Riverbed materials in the Reach are thought to be sufficiently coarse to resist movement by flows as high as the regulated 100-year frequency discharge (i.e., 12,500 m³/s; Dauble and Geist 2000). Lack of bed material mobility here has resulted in an extremely stable channel thalweg with little lateral migration (Hall 1988).

The Hanford Reach is a seventh-order stream bounded by Priest Rapids Dam and McNary pool on the upstream and downstream portion, respectively. The Reach contains no major tributary streams. The Federal Energy Regulatory Commission has established minimum licensed flows of 1,086 m³/s through the project, and these flows are regulated by releases at Priest Rapids Dam. At this scale, daily fluctuations in Reach surface elevation due to power peaking operations (i.e., relatively rapid fluctuations in flow over a 24-hour cycle designed to follow electrical load requirements; also referred to as load-following) can be significant (e.g., up to 2 m/day).

Average annual discharge through the Hanford Reach has not changed appreciably during the 20th century. For example, mean annual discharge for the Columbia River at Trinidad, Washington (rkm 710) was 3,270 m³/s from 1912 to 1946 (U.S. Army 1952). Average annual discharge at Priest Rapids Dam, (rkm 639) from 1959–2004 (after dam construction) increased by less than 4% (USGS 2005). Annual average discharge for the Reach during 1959–2004 ranged from 2,143 m³/s in 2001 to 4,786 m³/s in 1997. Mean annual discharge was 3,384 m³/s (1959–2004).

Over its 90-km distance, the Hanford Reach has a fairly uniform slope of about 0.2 m/km (Dauble and Geist 2000). The Columbia River changes course (i.e., thalweg shift sense; Dauble and Geist 2000) 25 times within the Reach. The only major change in thalweg bearing for the Reach occurs between rkm 586 and rkm 605. The width of the Reach ranges from about 240 to 1,200 m. The Reach widens with distance downstream, providing room for increased channel complexity or braiding. This is particularly evident in the middle portion of the Reach.

2.2 Fall Chinook Salmon Life History

Hanford Reach fall Chinook salmon (*Oncorhynchus tshawytscha*) exhibit an ocean-type life history (Healy 1991). Ocean-type Chinook salmon migrate to the ocean their first year of life, normally within three months after emerging from the gravel, spend most of their life in the ocean, and return to fresh water spawning areas a few weeks before spawning. Not all Columbia Basin fall Chinook salmon strictly adhere to an ocean-type life history. Connor et al. (2005) describe an alternative juvenile life history for

fall Chinook salmon in the Snake River; fall Chinook salmon that are delayed in development, due to cooler incubation temperatures, appear to hold over in freshwater up to a year before entering salt water. This alternative life history is likely an adaptation to uncertainties in juvenile survival and productivity in freshwater and estuarine environments resulting from a delayed emergence that places the migrants within reservoirs that are near the species' thermal tolerances (Connor et al. 2003). However, no evidence currently exists that would suggest Hanford Reach fall Chinook salmon hold over before salt water entry. Therefore, our assumption is that Hanford Reach fall Chinook salmon adhere to the ocean-type life history.

Fall Chinook salmon fry emerge from gravel nests (i.e., redds) in the Reach beginning in March and reside in the Reach until late June; the average date of median passage of sub-yearling fall Chinook salmon at McNary Dam (rkm 470) ranged from 27 June to 2 July during the five-year period from 2000 to 2005 (Columbia Basin Research 2006). Newly emerged fry inhabit near-shore areas feeding on aquatic insect larvae and zooplankton (Dauble et al. 1980; Tiffan et al. 2006). As they grow in size, they begin moving offshore into deeper and swifter water, migrating downstream as they rear (Dauble et al. 1989). This migration has been described as a spiraling migration behavior whereby fish alternate between stationary rearing and offshore downstream movement between suitable habitat (Coutant and Whitney 2006). By the end of July, about 90% of the sub-yearling Chinook salmon have passed below Bonneville Dam (rkm 230) into the estuary. Rearing time in the estuary before moving into the ocean can range from a few days to weeks, depending on stock characteristics and environmental conditions (Healy 1991). Once in the ocean, fall Chinook salmon tend to not disperse more than 1,000 km from their natal river, although information on ocean migrations is sketchy (Healy 1991). Adult fall Chinook salmon spend approximately one to five years in the ocean before returning to the Columbia River to begin their upstream migration to spawning areas.

Adult fall Chinook salmon bound for spawning areas in the Columbia Basin begin their upstream migration of the Columbia River in late July. The date of peak passage of adult fall Chinook salmon at McNary Dam is approximately the last week of September. About 90% of the fall Chinook salmon that migrate above McNary Dam will end up spawning in the Hanford Reach (Dauble et al. 2003). Escapement of adult fall Chinook salmon to the Hanford Reach averaged 45,995 from 1991 to 2005 (range 27,012 to 89,312). Escapement during the period from 1994 through 2004 is presented in Table 2.1. Pre-spawning adult salmon begin moving into and holding near spawning areas in earnest around the first week of October, with spawning initiated around a median date of 16 October (range 28 September to 26 October; Dauble and Watson 1997). During the period 1948 to 1992 the median date of peak spawning (i.e., the date of the highest total redd count) was 11 November with the range 26 October to 26 November (Dauble and Watson 1997). Dauble and Watson (1997) identified ten major fall Chinook salmon spawning areas distributed throughout the Hanford Reach (Figure 2.2). Vernita Bar and upper Locke Island were the two most important spawning areas in the Hanford Reach during the period 1948–1992, with about 60% of the total spawning occurring in those areas alone. Spawning activity within these ten major spawning areas has been consistent even during years of high escapement. Recent aerial surveys by PNNL (R. Mueller, PNNL, unpublished data) and aerial photography in 2004 by the U.S. Fish and Wildlife Service (Anglin et al. 2005) confirm that spawning continues to occur primarily within the ten major areas identified by Dauble and Watson (1997), with Vernita and Locke Island the two most used areas (Anglin et al. 2005).

Table 2.1. Fall Chinook Salmon Escapement to the Hanford Reach, 1994 Through 2004

Year	Adult Escapement
2004	79,464
2003	89,312
2002	69,342
2001	44,140
2000	36,027
1999	27,012
1998	29,410
1997	34,007
1996	37,548
1995	38,381
1994	48,857

Source: Hoffarth, P. 2005, pp. 55–56.

Figure 2.2. Ten Fall Chinook Salmon Spawning Areas

2.3 Selection of Study Sites

Although the Reach is approximately 90 km long, only 27 km (rkm 574 to 602) had been surveyed for high-resolution bathymetry at the time of our study (note: since the completion of our study, additional survey work has been completed over the entire 90 km). Detailed bathymetric data were needed to characterize the morphology of the channelbed and to run the hydraulic modeling. Thus, we restricted our study to this 27-km section, hereinafter referred to as the study area.

To ensure that study sites were representative of the full range of potential spawning habitat, the study area was stratified based on geomorphic features of the river channel. The two-dimensional form of the channelbed surface was quantified through the analysis of cross-sectional hydraulic geometry resulting from hydrodynamic modeling. The bathymetry surface required for the hydraulic models was constructed from a combination of surveyed cross sections, light detection and ranging (LIDAR) surveys, and hydrographic surveys with a single-beam echosounder. The U.S. Army Corps of Engineers (USACE) surveyed channelbed elevations at 55 cross sections throughout the study area, with an average spacing of 500 m between cross sections and 25 m between stations on each cross-section. A LIDAR survey of both right and left shorelines was performed by the USACE Scanning Hydrographic Operational Airborne Lidar Survey (SHOALS) system. The LIDAR survey encompassed the active channel up to a depth of approximately 5 m as well as near-shore topography (Nugent et al. 2001). Depth data in the remainder of the study area were collected using a 200-kHz single-beam echosounder deployed from a boat. The data were collected along transects spaced 20–300 m apart (depending on channel complexity) oriented perpendicular to flow, as well as along longitudinal transects oriented parallel to flow. Depth data from the bathymetry surveys were subtracted from corresponding water surface elevations, resulting in a comprehensive coverage of bathymetry point elevations. The elevation data from all these sources were combined into a geographic information system (GIS) database to create a triangulated irregular network (TIN) model representing the interpolated bathymetry surface for the study area.

Depth-averaged water velocities and water surface elevations throughout the study area were simulated using the hydrodynamic and water quality model MASS2 (Modular Aquatic Simulation System 2-D). MASS2 (Perkins and Richmond 2004a, 2004b) is an unsteady, two-dimensional model that simulates hydrodynamics and water quality in rivers and estuaries for subcritical and supercritical flow regimes. The model uses a structured multi-block, curvilinear computational mesh to represent the river geometry. Finite-volume methods (Patankar 1980) are used to discretize and solve the conservation equations for mass, momentum, and water quality constituents. The model is computationally efficient; it has been used to simulate flow conditions over long reaches (25 to 125 km) at high spatial resolution (cells sizes are typically 5 to 50 m) and high temporal resolution (on the order of 30 seconds) for time periods encompassing several months. MASS2 has been used in several applications on the Columbia and Snake River systems. These studies include simulating dissolved gas transport (Richmond et al. 2000), time-varying juvenile salmon stranding (Perkins et al. 2004), and radionuclide transport (Kincaid et al. 2000).

The computational grid for MASS2 was developed from the bathymetry surface and contained an average computational cell dimension of 10 m and 15 m laterally and longitudinally, respectively. The boundary conditions required for MASS2 were river discharge at the upstream boundary and water surface elevation (stage) at the downstream boundary. Downstream stage boundary conditions were computed using the 1-D MASS1 model (Waichler et al. 2005). MASS2 was calibrated by adjusting

Manning roughness coefficients until the simulated stage closely matched empirical stage recordings from the seven water-level loggers. The average hourly difference between simulated and empirical stage estimates ranged from 0.06–0.22 m (G. A. McMichael, PNNL, private communication). Velocities simulated by MASS2 were validated by comparison with velocities measured by acoustic Doppler current profiler at several locations and varying conditions in the Hanford Reach (G. A. McMichael, PNNL, private communication).

The MASS2 model output was imported into a GIS database for analysis of hydraulic geometry at closely spaced cross sections. Model results were extracted from 331 cross sections spaced 50–100 m apart throughout the 27-km study area. The hydraulic geometry at these cross sections was estimated with a steady discharge of $5,094 \text{ m}^3/\text{s}^{-1}$. This discharge represents the annual maximum flow with a 1-year return period, based on the annual maximum flood data series for 85 years of record (U.S. Geological Survey gage 12472800).

At each cross section, the model results were used to calculate the top width, mean depth, and depth at individual stations spaced 3 m apart along a cross section. These data were used to calculate the width to mean depth ratio F (an index of channel shape) and the maximum depth to mean depth ratio d^* (an index of cross-section asymmetry) for each cross section. When considered simultaneously, these two indices summarize a continuum of cross-sectional channel form that ranges from narrow and deep triangular channels to wide and shallow rectangular channels (Figure 2.3). To categorize each cross section within this continuum, we assigned each cross section into one of four categories based on the combined F and d^* values (Table 2.2).

Once classified, each transect was designated as a fall Chinook salmon spawning area or non-spawning area using information from Dauble and Watson (1997) and recent surveys. All spawning areas observable from the aircraft were mapped and incorporated into the GIS database as unique polygons. During an especially high escapement year (1994), individual redds were digitized from georeferenced aerial photographs and included in the GIS database. Each of the 331 cross sections was labeled spawning or non-spawning depending on whether or not it intersected a spawning area or cluster of redds.

Within the 27 km, only 10 (3%) of the transects were representative of the geomorphic class Fd^*3 (Table 2.3). Therefore, this class was removed from further consideration. All of the fall Chinook salmon spawning areas within the study area were located at cross sections with large width-to-depth ratios (Fd^*1 and Fd^*2 , Table 2.3). The cross sections containing spawning areas had much larger width-to-depth ratios than the Fd^*1 and Fd^*2 cross sections identified as non-spawning. The majority (70%) of the cross sections containing spawning areas were symmetrical from bank to bank (Fd^*2), while the remaining 30% were more asymmetrical (Fd^*1) (Table 2.3). Transects classified as Fd^*1 represented river cross sections that were typically asymmetrical and wide, with part of the transect containing a shallow bar that deepened into a V-shaped deep thalweg.

Figure 2.3. Representative Channel Cross Sections

Table 2.2. Criteria for Categorizing Cross Sections Based on F and d^*

Fd^* Category	Definition	Description
1	$F \geq 100$ and $d^* \geq 2.0$	Cross section is typically asymmetrical, wide, with part shallow bar deepening into a V-shaped deep thalweg
2	$F \geq 100$ and $d^* < 2.0$	Cross section is symmetrical, wide, and with relatively uniform shallow depths
3	$F < 100$ and $d^* \geq 2.0$	Cross section is asymmetrical, narrow, with part shallow bar deepening into a V-shaped thalweg
4	$F < 100$ and $d^* < 2.0$	Cross section is symmetrical, narrow, and relatively uniformly deep

Table 2.3. Summary of the Geomorphic Classification of Sampled Cross Sections

<i>Fd*</i> Category	Count	Relative Frequency (%)	% Spawning
1	54	16	30
2	132	40	70
3	10	3	0
4	135	41	0

Nine study sites were randomly selected from the pool of 44 classified transects (Table 2.4). The location of each transect within the Reach (upper, middle, or lower) was used to stratify the selection process so that we ended up with three sites in each river section (Table 2.4). Each study site was about 1.2 to 1.5 km in length. A detailed individual map of each of the nine study sites can be found in Appendix A. Note that at study sites where transects that were used to classify the site ran through an island (sites 59, 61, and 64), we established study sites on each side of the island and labeled these as either right bank (RB) or left bank (LB) (Figure 2.4). In addition, a distinction was made between some data collected at site 94, with one site in the main channel (94-Channel) and a second at an exposed river bar (94-Bar). In some cases, data were collected in specific sub-sites and combined to represent the nine study sites. In other cases, the data were kept separate and presented for all 12 study sites. This will be made clear in subsequent sections.

Table 2.4. Nine Study Sites

Site	Section Location	<i>Fd*</i> Value	Shoreline Distance (km)	Median (Range) Spawning Density (redds/0.1 km)
54	Upper	4	1.2	0
59	Upper	1	1.2	13 (4–25)
61	Upper	2	1.4	4 (0–11)
64	Middle	2	1.2	113 (99–148)
81	Middle	2	1.4	25 (25–109)
90	Middle	4	1.4	0
94	Lower	4	1.5	0
96	Lower	2	1.4	0
101	Lower	4	1.3	0

Aerial photographs taken in 1994, 1995, and 2001 were used to calculate spawning density over lineal river distance; these results are plotted in Figure 2.5 along with the nine study sites. The data showed that the spawning locations tend to be clustered within the section of the Reach where islands and bars are located; four of the nine study sites were located in this section while five were in sections upstream or downstream that were not used for spawning (Figure 2.5). The four lower sites (90, 94, 96, and 101) and the one uppermost site (54) were not used for spawning in any of the three years of photographs. Further, redd surveys since 1948 also suggest these areas were consistently not used for spawning. Therefore, sites 54, 90, 94, 96, and 101 were determined to be representative of non-spawning areas, while sites 59,

Figure 2.4. Individual Study Sites. There were nine sites with some sites broken into more than one group.

61, 64, and 81 were classified as spawning sites (Table 2.4). This classification was consistent with the spawning classification of 331 transects previously described in this section. The reader may want to refer back to this table in subsequent sections where we analyze differences in physical habitat data between spawning and non-spawning sites.

Figure 2.5. Three-Year (1994, 1995, and 2001) Average Redds per 0.1 km from the Hanford Reach of the Columbia River. The nine study sites are shown as black bars with labels.

3.0 Spatial and Temporal Pattern Analysis of Fall Chinook Salmon Spawning

The study of the spatial patterns of plants and animals by scientists has a long history in forestry and ecology (Goodall 1952, 1970; Pielou 1959, 1960; Ripley 1987). Plant ecologists commonly use pattern analysis to evaluate intra- and inter-specific relationships, seed dispersal, disturbance, herbivory, and environmental heterogeneity (Moeur 1993). Wildlife biologists use pattern analysis to understand animal home range, habitat use, species interactions, and migration patterns (White and Gorroh 1990). Pattern recognition and the processes that control these patterns is a fundamental step in resource management (Levin 1992).

Point pattern analysis is one technique used by ecologists to evaluate spatial pattern. Point pattern analysis involves measuring the relationships among point locations within a study area. A number of models have been used to identify point patterns, including Ripley's K and nearest-neighbor analysis (Ripley 1981; Cressie 1993). In both these models, the pattern of points can be assessed as following one of three outcomes—randomly distributed, clustered, or regularly distributed. Random patterns result from completely undifferentiated processes. Clustered patterns are those in which the points are significantly more grouped in the study area than they are in a completely random distribution, whereas regular patterns (sometimes also called uniform) are those in which the points are more spread out over the environment than they would be in a random distribution. The spatial pattern of the redds of Pacific salmon within a spawning area may be indicative of the physical habitat features of the site and the biological cues between fish. There are generally three hypotheses that can be stated for redd site construction (from McNeil 1967): 1) hypothesis of uniform distribution, i.e., once a redd has been dug, there is a decreased likelihood that subsequent spawners will dig in this area; 2) hypothesis of random distribution, i.e., redds are dug at random locations without regard to habitat or other individuals; and 3) hypothesis of clustered distribution, i.e., once a redd has been dug, there is an increased likelihood that other fish will spawn at this location.

Geist et al. (2000) used point pattern analysis to show that fall Chinook salmon (*Oncorhynchus tshawytscha*) spawning in the Hanford Reach of the Columbia River was highly clustered at spatial scales of a river reach (10 km) but uniformly distributed within shorter distances of 2 to 5 m. Their analysis determined that fall Chinook salmon select redd sites that would appear to correspond to an underlying physical habitat feature unique to those areas where clustering was prevalent. However, inter-female competition for redd sites indicated by uniform distribution appeared to override habitat features at some point during the spawning period. Understanding the evolution through time of the pattern is needed to determine whether there are specific habitat features fish used to select sites or to understand the interaction between females. The objective of this study was to evaluate the weekly spawning patterns in two spawning areas in the Hanford Reach. Our hypothesis was that the weekly pattern of redd placement would be clustered between the two areas (i.e., at the scale of the Hanford Reach) but randomly distributed within the two spawning areas.

3.1 Methods

The spatial distribution of fall Chinook salmon redds was analyzed for two areas (Area 4 and Area 7) in the Hanford Reach of the Columbia River. These were two of the ten areas where fall Chinook salmon consistently spawn in the Hanford Reach (Dauble and Watson 1997). Area 4 was adjacent to Locke Island and downstream of Site 64-RB (Figure 3.1). Area 7 was located near an island between Site 54 and Site 59 (Figure 3.1). Both areas are major fall Chinook salmon spawning areas, accounting for over 20% of the fall Chinook redd locations in the Hanford Reach (R. P. Mueller, PNNL, unpublished data). Each area contains suitable habitat for fall Chinook salmon spawning based on gravel size, water velocity, lateral slope, and depth (Geist et al. 2000; PNNL, unpublished data).

Figure 3.1. Location of Areas for Analysis of Spatial and Temporal Patterns of Redd Distribution

For both areas, fall Chinook salmon redds were photographed from a fixed-wing airplane flying approximately 600 m above the river (photographic scale of ~1:2,400). Photographic flights occurred in 4 consecutive weeks during the peak spawning intervals in 1994 and 1995 (Table 3.1). Each redd was digitized into an ArcInfo[®] geographic information system (GIS) coverage. Aerial photographs of redd distributions are limited by water depth. However, supplemental redd surveys conducted in 1995 using an underwater video camera showed that all redds present in the study area could be seen on the aerial photographs. Additional details on the flights and processing of photographs can be found in Geist et al. (2000) and Visser et al. (2002).

Table 3.1. Dates of Aerial Photography

	1994	1995
Week 1	29 October	29 October
Week 2	3 November	6 November
Week 3	9 November	12 November
Week 4	12 November	19 November

Adult escapement to the Hanford Reach in 1994 and 1995 was estimated to be 48,857 and 38,381, respectively; this compares with an average escapement over the past 15 years of about 51,000 adults (Hoffarth 2006). River flows ranged from 1,360 to 1,580 m³/s on days when photographs were taken. This compares with daily average discharge at Priest Rapids Dam (rkm 639) of 1,200–3,800 m³/s during peaking fall Chinook salmon spawning over the period 1959–1985 (Dauble and Watson 1997).

We used two spatial analysis methods to study the distribution of salmon redds. These methods included the calculation of the area within a convex hull bounding the salmon redds for each consecutive week and the use of refined nearest-neighbor analysis. The convex hull is the smallest convex region enclosing a planar set of points (Graham and Yao 1983) and can intuitively be thought of as the shape that a rubber band would take if it were stretched around a set of push-pins occupying the redd locations on a map. We compared the area within the convex hull defined for the redds for each week in each year to the total area of the convex hull defined in the fourth week, to identify the proportion of the total area ever covered by redds in a year with the area covered in earlier weeks. The convex hull was created using the CREATETIN command in ArcInfo because the triangulated irregular network (TIN) created by that command from a set of points includes the convex hull of those points as one of its properties (ESRI 1994). The area within the convex hull for each week was retrieved using the VOLUME command within ArcInfo (ESRI 1994). Because the Columbia River in the Locke Island area has a northwest-southeast orientation, we rotated the coordinates of the redd point locations so that the longitudinal boundaries of the rectangular study area were parallel to the axis of the Columbia River. Because of the length of the Locke Island study area and the presence of a large gap in the middle of the study area in which there are very few redds (see Figure 3.1), the Locke Island study area was subdivided into two areas, an upstream and a downstream area, for creation of the TINs.

We used refined nearest-neighbor analysis (Boots and Getis 1988) to examine the spatial arrangement of the redds for each week and determine if they were distributed randomly or if instead they were clustered or uniform. Refined nearest-neighbor analysis uses the cumulative distribution function $F(d)$ to describe the probability that the nearest neighbor to a redd is within a given distance, d . A clustered pattern would be associated with a tendency for greater clumping or clustering of redds than would be expected for a random distribution, while a uniform pattern would have the redds more evenly spaced than would be expected for a random distribution. This technique was used by Geist et al. (2000) to show that the redds in the Locke Island area in the fourth week of 1994 and 1995 were clustered at larger scales but tended to be uniformly distributed within the clusters. We used software discussed by Moeur (1993) to perform the refined nearest-neighbor analysis. The software generates a 95% confidence envelope around the expected value for each distance d using Monte Carlo simulation. If the proportion of the nearest neighbors less than distance d falls outside the confidence envelope, then the hypothesis that the spatial pattern of the redds resulted from a random process is rejected at the 95% confidence level. The direction of the deviation above or below the confidence envelope indicates whether the non-random

pattern tends toward a uniform or clustered distribution, respectively. The nearest-neighbor analysis is based on the use of a rectangular area encompassing the points of interest, so we used the rotated coordinates that placed the long axis of the study area parallel to the axis of the Columbia River in the Locke Island area (Area 4).

3.2 Results

There were fewer redds in 1995 than in 1994 at both areas (Table 3.2), but fall Chinook salmon spawning locations within the study areas were consistent between the two years (Figures 3.2 and 3.3). When the TINs were plotted, we see that by the first week the outline of the redds is not that different from the outline of the redds in week 4 in Area 7 (Figures 3.4 and 3.5), as well as in the upstream (Figures 3.6 and 3.7) and downstream (Figures 3.8 and 3.9) segments at Area 4. More specifically, in the first week of spawning, the redds occupied about 75% and 68% of the total area in 1994 and 1995, respectively (Table 3.2). This rose sharply to 95% and 84% in week 2 during 1994 and 1995, respectively (Figure 3.10).

Table 3.2. Area and Redd Densities for Study Areas

Year	Week	Location	Number of Redds	Convex Area (m ²)	Proportion of Area (%)	Density (redds/ha)
1994	1	Area 4 Upstream	86	13.6	56.5	6.3
	2		309	21.8	90.9	14.2
	3		531	23.0	95.8	23.1
	4		722	24.0	100.0	30.1
1995	1	Area 4 Upstream	109	9.2	98.1	11.9
	2		199	9.3	100.0	21.3
	3		239	9.3	100.0	25.6
	4		259	9.3	100.0	27.7
1994	1	Area 4 Downstream	79	14.5	49.5	5.4
	2		576	27.6	94.1	20.9
	3		965	27.7	94.7	34.8
	4		1,160	29.3	100.0	39.6
1995	1	Area 4 Downstream	154	21.8	82.8	7.1
	2		548	25.0	95.1	21.9
	3		668	25.8	98.4	25.8
	4		737	26.3	100.0	28.0
1994	1	Area 7	94	9.3	75.4	10.1
	2		300	11.7	95.1	25.6
	3		598	12.1	98.1	49.4
	4		743	12.3	100.0	60.3
1995	1	Area 7	150	7.3	68.6	20.6
	2		353	9.0	85.4	39.0
	3		515	10.1	95.3	50.9
	4		587	10.6	100.0	55.3

Figure 3.2. Redd Distribution by Week for 1994

In Area 4, the redds in week 1 occupied a smaller proportion of the total area than was observed at Area 7 (Figure 3.11; Table 3.2). For example, in 1994 only 50 to 56% of the total area was occupied in the first week at Area 4. In 1995, the occupation percentages were more similar to Area 7 at 82.8 and 98% (Table 3.2). The two areas are similar in that the proportion of the area occupied in week 2 exceeds 90%.

One interesting observation for the upstream segment of Area 4 is that the number of redds and the total area occupied in 1994 is much larger than it is for the same segment in 1995 (Table 3.2). The relative drop in the number of redds (722 to 259) and in the area (24 to 9.3 ha) from 1994 to 1995 is much larger for the upstream segment than it is for the downstream segment or for Area 7. However, the redd density in the fourth week was similar across years at Area 4 (range 27.7 to 39.6 redds/ha), although densities at Area 7 at 55 to 60 redds/ha were about twice as high.

Significant changes in the nearest neighbor distances occurred with time in 1994 (Figure 3.12) and 1995 (Figure 3.13) at Area 7. In week 1 for 1994, we see that several percent of the nearest-neighbor

Figure 3.3. Redd Distribution by Week for 1995

distances are greater than 30 m. By week 2, this has decreased to about 20 m, and in the third and fourth weeks, about 99% of the nearest-neighbor distances are less than 13 m. At short distances, on the other hand, only slight changes are seen in the nearest-neighbor distributions. During the first week of 1994, the closest that any redds occur to one another is 7 m. Over the succeeding weeks as the redds infill the spawning area, the minimum distance gradually decreases to 3 m. This infilling of the cluster is not random. By the second week of 1994, the spatial distribution of redds within the cluster is a more uniform distribution of redds found for short distances than would be expected if the distribution of redds were random. The difference between the empirical curve and the lower bound of the Monte Carlo confidence envelope increases with time (Figure 3.12), indicating a greater deviation from randomness as the redd cluster develops and the spacing becomes more uniform. Similar results were observed in 1995 except that there was a slightly higher probability for greater distances between redds than was seen in 1994 for the fourth week (13 to 15 m; Figure 3.13). Another minor difference is that the maximum distances associated with the first week are somewhat shorter in 1995 than 1994.

Figure 3.4. TINs Based on Redd Locations in Area 7 for Each Week in 1994

Figure 3.5. TINs Based on Redd Locations in Area 7 for Each Week in 1995

Figure 3.6. TINs Based on Redd Locations in Upstream Segment of Area 4 for Each Week in 1994

Figure 3.7. TINs Based on Redd Locations in Upstream Segment of Area 4 for Each Week in 1995

Figure 3.8. TINs Based on Redd Locations in Downstream Segment of Area 4 for Each Week in 1994

Figure 3.9. TINs Based on Redd Locations in Downstream Segment of Area 4 for Each Week in 1995

Figure 3.10. Percentage of Total Area Occupied for Area 7

Upstream

Downstream

Figure 3.11. Percentage of Total Area Occupied for Upstream and Downstream Area 4

Figure 3.12. Spatial Point Pattern Analysis of Fall Chinook Salmon for Area 7 in 1994, by Week. The solid lines for $F(d)$ represent a cumulative histogram of the nearest-neighbor distances calculated from the data. The dashed lines in each panel represent the upper and lower boundaries of the 95% confidence envelope of 100 Monte Carlo simulations of a random point distribution. Deviation of the empirical distributions above or below the confidence envelope indicates that redd distributions are clustered or uniform, respectively. Random redd distribution is indicated when the solid line is contained within the confidence envelope.

The results for Area 4 differ slightly from the results shown for Area 7, with one difference being that the maximum distances between redds were greater in 1994 (Figure 3.14) and 1995 (Figure 3.15) for Area 4 than for Area 7 (note that the distance scales for Figures 3.14 and 3.15 are longer than those used for Figures 3.12 and 3.13). For Area 4, the 97th percentile of the empirical distribution is not reached until a distance of 20 m in 1994 versus 12 m for Area 7; the 97th percentile in 1995 does not occur until a distance of 30 m versus 14 m for Area 7. Another significant difference is that the redds in Area 4 do not appear to be as uniformly distributed as those in Area 7, especially in 1995. For 1994, the redd distribution does not become significantly uniform until the third week, and even then is just slightly outside the confidence envelope for distances from 3–5 m (Figure 3.14). This contrasts with the results for Area 7 (Figure 3.12) where the redd distribution is significantly uniform by the second week and shows a much greater deviation below the confidence envelope, with a uniform distribution indicated for distances

Figure 3.13. Spatial Point Pattern Analysis of Fall Chinook Salmon for Area 7 in 1995, by Week. The solid lines for $F(d)$ represent a cumulative histogram of the nearest-neighbor distances calculated from the data. The dashed lines in each panel represent the upper and lower boundaries of the 95% confidence envelope of 100 Monte Carlo simulations of a random point distribution. Deviation of the empirical distributions above or below the confidence envelope indicates that redd distributions are clustered or uniform, respectively. Random redd distribution is indicated when the solid line is contained within the confidence envelope.

ranging from 1–8 m in the fourth week. In 1995, which had a smaller number of redds present in the study areas, the contrast is even greater. Comparing Figures 3.15 and 3.13, we find that although there was a tendency for the empirical curve to get closer to the lower boundary of the confidence envelope for Area 4 in 1995, it never crossed it, so that the redd distribution for Area 4 in 1995 cannot be distinguished from a completely random spatial distribution at short distances (significant clustering of the redds is still indicated for larger distances of 7–34 m). For Area 7 in 1995, the distribution for short distances was significantly uniform even in the first week and was uniform for distances from 2–7 m in the fourth week.

Figure 3.14. Spatial Point Pattern Analysis of Fall Chinook Salmon for Area 4 in 1994, by Week. The solid lines for $F(d)$ represent a cumulative histogram of the nearest-neighbor distances calculated from the data. The dashed lines in each panel represent the upper and lower boundaries of the 95% confidence envelope of 100 Monte Carlo simulations of a random point distribution. Deviation of the empirical distributions above or below the confidence envelope indicates that redd distributions are clustered or uniform, respectively. Random redd distribution is indicated when the solid line is contained within the confidence envelope.

3.3 Discussion

As shown by Geist et al. (2000) for Area 4, fall Chinook spawning areas in the Locke Island area of the Hanford Reach are very consistent from year to year. Geist et al. (2000) reported that individual spawning sites in the Locke Island area were statistically similar between 1994 and 1995. This study found that the general outline and shape of the main spawning cluster is also very consistent from year to year in Area 7. This consistency in both areas suggests that the factors controlling salmon spawning are relatively stable. This stability is not surprising, given the relatively small differences in bank migration, river bar shifting, and channel aggradation or degradation in the Columbia River, as discussed in Section 2. As noted there, riverbed materials in the Reach are thought to be sufficiently coarse to resist movement by flows as high as the regulated 100-year frequency discharge (i.e., 12,500 m³/s). This

Figure 3.15. Spatial Point Pattern Analysis of Fall Chinook Salmon for Area 4 in 1995, by Week. The solid lines for $F(d)$ represent a cumulative histogram of the nearest-neighbor distances calculated from the data. The dashed lines in each panel represent the upper and lower boundaries of the 95% confidence envelope of 100 Monte Carlo simulations of a random point distribution. Deviation of the empirical distributions above or below the confidence envelope indicates that redd distributions are clustered or uniform, respectively. Random redd distribution is indicated when the solid line is contained within the confidence envelope.

leads to a stable configuration of the sedimentary bars and deposits that control the distribution of sediment properties and hyporheic properties (e.g., porosity, hydraulic conductivity, and groundwater flow) that appear to be important for salmon spawning in the Hanford Reach (Geist 2000) and elsewhere (Geist et al. 2002; Hanrahan 2006).

Analysis of redd locations in the study areas shows that the major changes in redd distribution during the spawning period are an increase in redd density and a decrease in nearest-neighbor distances between redds. For 1994, the maximum distance between redds decreased from more than 30 m to about 13 m. The minimum distance between redds also decreases as the redd density increases. However, the minimum distance appears to have a lower limit of at least 3 m, and the nearest-neighbor analysis indicates that by the second week, the redd spacing is more uniform than would be expected from a random distribution. These findings at Area 7 corroborate a separate analysis of spawning pattern done by Geist et al. (2000) and are also consistent with findings of Dauble and Watson (1997), who reported fall

Chinook salmon spawning in definitive clusters throughout the Reach. Collectively the results support our hypothesis that fall Chinook salmon select spawning areas in a non-random behavior at the scale of the Hanford Reach but behave in a more uniform manner within redd clusters. These results are partially consistent with a study conducted on pink salmon (*O. gorbuscha*), which were found to be clustered in their spawning locations (McNeil 1967). However, it is not clear from McNeil's study where there was a shift in pattern from clustered to uniform as distances between redds was decreased.

The results for Area 4 show a uniform distribution of redds within redd clusters in 1994 but not in 1995, when the numbers of redds in Areas 4 and 7 were lower. Also, there is a higher probability for redds to be separated by longer minimum distances in Area 4 than Area 7, especially in 1995. Given those differences, it is interesting to note that the redd densities within the convex hull areas in week 4 are roughly similar for Area 4 in both years (~28 to 40 redds/ha). The redd densities in Area 7 are higher than those in Area 4 by about a factor of two (55–60 redds/ha), which may be associated with the greater tendency for uniform redd distribution in Area 7. Visser et al. (2002) reported redd densities of 73 redds/ha in 1994 and 54 redds/ha in 1995 within seven spawning areas in the Hanford Reach (two of which are included in our analysis). Consistent with our results, Visser et al. (2002) reported that spawning clusters were about 21% smaller in 1995 than in 1994. These year-to-year differences in density and spawning cluster size were statistically different, which the authors attributed to the lower adult escapement in 1995 as compared to 1994.

The results for the convex hull areas derived from the TIN analyses (Figures 3.10 and 3.11) show that most of the area that is eventually occupied by redds is occupied early in the spawning season. In all cases, at least 50% of the eventual total area is occupied in the first week for which spawning data is available. In all but one case (Area 7 in 1995), more than 90% of the area is occupied by the second week. The tendency for the entire area to be occupied relatively early suggests that the entire spawning area has roughly equally acceptable characteristics for spawning and that the returning salmon quickly outline the limits of the acceptable spawning area, then infill it at a much greater density. If this were not the case, then one might expect to find early spawning concentrated in a few "sweet spots" that would grow principally around their edges as the salmon occupied less favorable spawning areas.

Point pattern analysis of salmon spawning in the Hanford Reach showed that the redds were clustered at a spatial scale of the entire Reach—that is, there were specific spawning areas that adults selected for spawning. This suggests that habitat features used by adults to select redd sites are not ubiquitous throughout the entire Reach. That the distribution of redds within the clusters was uniform in most areas suggests that females establish and defend sites so that redd superimposition is reduced. We did not observe redd superimposition during the two years of the study, and redd superimposition is normally observed only when spawning escapements are very high (D. Dauble, PNNL, personal communication). The lack of superimposition suggests that spawning habitat is not limited in the Hanford Reach (Dauble and Watson 1997). Identification of the specific habitat features that are associated with spawning clusters, as well as understanding the capacity of this habitat, will assist managers in setting management goals for the Hanford Reach.

4.0 Assessment of Fall Chinook Salmon Spawning Habitat Using Standard Habitat Characteristics

The identification of the effect of varying streamflows on ecological habitat is a widely recognized need, and a number of methods for providing that identification are described by Behrouz Ahmadi-Nedushan et al. (2006). Previous work for this project, documented in Geist et al. (2000), discussed the application of measurements at the scale of redd clusters and logistic regression for identification of habitat suitable for fall Chinook salmon (*Oncorhynchus tshawytscha*) spawning in the Hanford Reach of the Columbia River. That study indicated that cluster-scale measurements significantly improved predictions of habitat use relative to the more traditional PHABSIM method employed by Geist et al. (1997). The PHABSIM method, which employed a non-local but river-specific set of suitability criteria, is described in Bovee (1995). The use of the river-specific set of criteria resulted in predictions that over 50% of the river bottom in two areas of the Hanford Reach should have been used for spawning, when in fact, the actual use was less than 20%. The cluster-scale approach provided a significant improvement in decreasing the amount of area expected to be useable habitat, but it was applied to a limited area of the Hanford Reach, in the vicinity of Locke Island and Wooded Island. In addition, although the habitat predictions were significantly better than those obtained by Geist et al. in 1997, there were still significant errors of commission, with approximately 30 to 40% of the area near Locke Island predicted to have redds but in which spawning did not actually occur (Geist et al. 2000).

The purpose of the present study was two-fold. First, the study attempted to confirm the preliminary habitat modeling results for the Hanford Reach shown in Geist et al. (2000) and extend them to a wider area of the Reach through the examination of a wider range of study areas. Second, because of the continued occurrence of relatively high errors of commission for the approach used by Geist et al. (2000), a plan was developed to examine the effect of dynamic changes in flow rates on habitat modeling and, in particular, to look at the connection between the variability and persistence of habitat variables due to changes in flow rates and error rates in habitat modeling.

4.1 Methods

4.1.1 Study Site Selection

Twelve study sites were selected from the middle 27.4 km of the Hanford Reach (Figure 2.4), and data were collected from each of the sites for an analysis of fall Chinook salmon habitat. A complete description of the approach used for study site selection can be found in Section 2.3 of this report.

4.1.2 Data Collection and Summary

Once the study sites were selected, we began to assess whether standard habitat characteristics (i.e., substrate, depth, velocity, and slope) were associated with the locations of fall Chinook salmon spawning. Substrate size classification data were collected by underwater videography in 2000 and 2002. Because of the challenges inherent in sampling the bed of large rivers (i.e., deep and fast water), the substrate data were collected using a boat-deployed, video-based adaptation of the pebble count method (Wolman 1954). The video system consisted of a high-resolution monochrome camera with wide-angle lens connected to an 8-mm camera recorder located on the boat. The camera was placed inside a waterproof housing and mounted on a weighted platform containing two downward-pointing lasers, providing reference scale

within each video image. Positional data were recorded with a real-time differential global positioning system (GPS). Images were processed in the laboratory using Optimas, an image-based software.

The pebble count method typically dictates sampling 100 stones selected without bias from specific geomorphic features (Wolman 1954; Church et al. 1987; Kondolf 2000). The study area was delineated into 13 distinct geomorphic units based on the location of the study sites, channel morphology, and hydraulic characteristics. A total of 100 substrate images were collected by underwater videography within each geomorphic unit. Within each unit, samples were collected in a spatially distributed manner so that a fairly uniform coverage of each unit was attained. This sampling scheme was appropriate for obtaining overall sediment size characteristics of a distinct geomorphic unit, or even for an entire reach (Wolcott and Church 1991; Crowder and Diplas 1997).

The substrate images were treated as pebble counts, in that the processing software automatically selected the grain at the center of each image. The grain was then measured by the software, resulting in an estimate of the length of the apparent long (a) and intermediate (b) axes for each grain. In addition to the boat surveys using the video camera, shallow shoreline areas that could be waded were sampled by the traditional pebble count method (Wolman 1954). All pebble count sampling was performed by the same observer to eliminate multiple-observer sampling error (Wohl et al. 1996). To reduce measurement error and obtain samples directly comparable with sieve analysis, all grainsize measurements were made with an aluminum template containing square openings in $\frac{1}{2}$ -phi size classes from 128 mm (-7 phi) to 2 mm (-1 phi) (Hey and Thorne 1983). The grainsize estimates derived from all substrate sampling were used to develop cumulative grainsize distributions for each geomorphic unit. Substrate grainsize maps were generated by first calculating variograms (Isaaks and Srivastava 1989) of the empirical substrate measurements. The substrate measurements were the size in millimeters of the median grain diameter (i.e., d_{50}). Each geomorphic unit was separately mapped using standard geostatistical software (Deutsch and Journel 1998) to create a spatial map, and ordinary kriging (Isaaks and Srivastava 1989) was used to estimate the spatial distribution of the substrate data within the defined mapping areas. This substrate surface was subsequently converted to an ArcInfo grid coverage for use in assigning substrate values to habitat cells.

The bathymetry and MASS2 model described in Section 2.3 were used in steady-state mode to generate velocities and depths over a range of discharges from 40,000 to 180,000 cfs in 10,000-cfs increments. The results from each model run were imported into a GIS database, providing hydraulic data (i.e., depth and velocity) for each node in the study area. A continuous surface for each hydraulic variable was created in the GIS using an inverse distance weighting interpolation between nodes. The interpolated value within 3-m $\times$ 3-m cells was determined by a linearly weighted average of the cell's three nearest nodes. The weight was a function of inverse distance, such that nearby sampling points had more influence on the interpolated value.

We then created time series of depth and velocity for each 3-m $\times$ 3-m grid cell in the study area. These time series were generated using the hourly discharge values at Priest Rapids Dam for the period between 15 October and 15 November (fall Chinook salmon spawning period; Dauble and Watson 1997) for each of the three years (1994, 1995, and 2001). For each node, the depth and velocity values corresponding to the nearest 10,000-cfs model run were selected for each hourly discharge value. For example, if the hourly flow at Priest Rapids Dam was 97,000 cfs, then the depth and velocity values for the 100,000-cfs model run would be selected for each node. It is recognized that this procedure does not fully account for the dispersion of the flow as it moves downstream from the dam. However, computing the

unsteady velocity and depth for the discharge time series for each year and then interpolating the MASS2 output for each hour to all 3-m $\times$ 3-m cells was not computationally feasible.

Riverbed slope was estimated in the GIS via an algorithm that fits a plane to the bathymetric elevation values of a 3-cell by 3-cell neighborhood around the cell being processed. The slope for a given cell was calculated as a unitless gradient using the SLOPE function in ArcInfo, which uses the average maximum technique based on the surrounding 3 $\times$ 3 neighborhood (Burrough 1986).

Aerial photography was used in 2001 to determine where fall Chinook salmon located their redds within the study area. Stereo color aerial photography at a nominal negative scale of 1:7,200 was obtained by a contractor (Walker and Associates, Seattle, Washington) on 8 November 2001 using a Zeiss precision aerial camera equipped with a gyro-stabilized mount and an image-motion-compensation magazine. This camera was also equipped with an event-marking feature that permitted shutter-opening events to be recorded in a GPS data format collected during the flight mission. A GPS was used to navigate flight lines and fire the camera. Pre-marks were established prior to the flight and were used in combination with a digital elevation model (DEM) to orthorectify the images during photograph processing. The resulting accuracy of the images was approximately ± 8 cm. Individual salmon redds were digitized from the images into an ArcInfo GIS coverage. In addition to the aerial photography obtained in 2001, aerial photography from 1994 and 1995 was available for this analysis (Geist et al. 2000).

Once the GIS coverages were created for each of the habitat variables, individual “habitat cells” (Payne and Lapointe 1997) were created that contained a value for substrate, slope, depth, and velocity at a given discharge level. Habitat values were assigned to habitat cells in the GIS by treating the center of each 3-m $\times$ 3-m habitat cell as a sampling point at which the data were extracted. A cell was identified as a spawning cell if the cell contained the center of a spawning location digitized from aerial photographs. However, from previous studies, it was known that salmon spawning is clustered (Geist et al. 2000). The distance of an observed spawning cell from its nearest neighbor was calculated, and it was found that the average distance was about 10 m, as shown in Figure 4.1. For the analysis, all 3-m $\times$ 3-m cells that were within 10 m of a cell that contained the center of a redd were considered to be within a spawning cluster and therefore were identified as spawning cells during the analyses. Table 4.1 shows the distribution of spawning cells for the 12 study sites. Note that the largest number of spawning cells for each study site occurred in 1994, although 1994 did not have the largest number of redds throughout the Hanford Reach of the three years (R. P. Mueller, PNNL, unpublished data) or the highest number of adults passing McNary Dam (Fish Passage Center, www.fpc.org/adultsalmon/adulthistory/YTD-MCN.html).

4.1.3 Logistic Regression Analysis

Logistic regression, based on negative log likelihood (Hosmer and Lemeshow 1989), was used to estimate the probability of a binary response (suitable or unsuitable spawning site selection) given a set of explanatory variables (substrate, depth, velocity, and slope, denoted as *SUB*, *DEP*, *VEL*, and *SLOPE*, respectively). Site-specific models were created for 1994, 1995, and 2001 using the median discharge values for each of those years during the fall Chinook salmon spawning period. The daily 50% exceedence values of hourly flows during the spawning periods were approximately 50,000 cfs for 1994 and 2001 and 70,000 cfs for 1995. The velocity and depth of the single flow discharge rate closest to the daily 50% exceedence value were used in the data analysis.

Figure 4.1. Distribution of the Shortest Distance (m) Between Nearest Spawning Habitat Cells

Table 4.1. Summary of Habitat Cell Types After Reclassifying Spawning Cells

Study Area	Number of Cells	Number of Spawning Cells					
		Year 1994		Year 1995		Year 2001	
		Original	New Assign	Original	New Assign	Original	New Assign
54	19,443	0	0	0	0	0	0
59-RB	10,760	48	824	48	795	40	862
59-LB	2,476	22	582	3	108	1	20
61-RB	15,934	43	616	29	509	4	84
61-LB	7,617	22	484	14	421	0	0
64-RB	6,225	615	8,091	385	6,432	360	6,280
64-LB	11,486	0	0	0	0	0	0
81	11,545	690	8,242	446	6,459	195	3,918
90	21,058	0	0	0	0	0	0
94	45,874	0	0	0	0	0	0
96	29,218	0	0	0	0	0	0
101	21,385	0	0	0	0	0	0
TOTAL	203,021	1,440	18,839	925	14,724	600	11,164

The logistic regression model and parameters were derived using a training data set that was a subsample of the data from 1994, 1995, and 2001 (random samples of 5,000 cells from spawning and non-spawning cells for each year). This subsampling was necessary because of the highly unbalanced nature of the data from the Hanford Reach that included many more non-spawning cells than spawning cells. The resulting logistic regression models were then tested against data from the cells that were not included in the training data set. These data is referred to as the test data set.

Logistic regression models (including first-order, second-order, and cross terms) were constructed using the training data. Because of the lack of knowledge of the functional form of a variable on the spawning behavior, second-order (i.e., squared) terms of the variables were included in the logistic

regression to introduce some nonlinearity, a practice that is common in the analysis of multivariate data (e.g., Wold et al. 2001). A variety of criteria were used for evaluating the logistic regression. A log-likelihood statistic was calculated based on summing the probabilities associated with the predicted and actual outcomes for each case:

$$\text{log likelihood} = \sum_{i=1}^N [Y_i \ln(\hat{Y}_i) + (1 - Y_i) \ln(1 - \hat{Y}_i)] \quad (4.1)$$

For a perfect model, the log-likelihood could theoretically reach 0. Two nested models can be compared by computing the difference in their log-likelihoods and using the chi-square (χ^2) test:

$$\chi^2 = 2[\text{log likelihood}(\text{model with } N \text{ variables}) - \text{log likelihood}(\text{model with } N - 1 \text{ variables})] \quad (4.2)$$

For simple models based on a single term of the variables, the model was compared to one based on only a constant term. The larger the χ^2 , the more improvement there is compared to the constant model and the more influence the variable is expected to have on the spawning site selection. The χ^2 test showed that all single-term models were significantly different from the model with only the constant, which indicated that multivariate logistic regression models would be needed to model spawning site selection of Hanford Reach fall Chinook salmon and evaluate the relative value of the different variables.

The relative improvement in sequentially developed multivariate models was evaluated using the McFadden's ρ^2 statistic. McFadden's ρ^2 is a measure based on transformation of the likelihood ratio statistic that is intended to mimic an R^2 for multiple regression (Tabachnick and Fidell 2001); it is calculated as follows:

$$\text{McFadden's } \rho^2 = 1 - \frac{LL(B)}{LL(0)} \quad (4.3)$$

where $LL(B)$ is the log-likelihood of the candidate model and $LL(0)$ is the log-likelihood of the constant-only model. Usually McFadden's ρ^2 is much lower than R^2 for multiple regressions, with values in the range of 0.2 to 0.4 considered to be high (Tabachnick and Fidell 2001).

A sequential procedure was used to determine the most influential variables to include in the logistic regression. Starting from the single-variable model, the most important variable was selected based on the ranking of their relative importance in explaining the spawning site selection, measured using either $LL(N)$ or $2 \times (LL(N) - LL(0))$. In the next step, all possible two-variable models were evaluated, and the best two-variable model was determined. Variables were added into the model one at a time, and the variable was retained that contributed most to the existing model without causing the regression coefficients of the existing variables to become insignificant. In the situation when multiple variables present similar improvements on the existing model, main effects were considered first, followed by the squared terms and then the cross terms. In each step, the differences between the two consecutive models were tested using McFadden's ρ^2 to see if any significant improvement resulted from the inclusion of the additional variable.

Performance of the logistic regression model was evaluated using the prediction success for the test data set. In prediction success tables, cases are "partially" assigned to each outcome based on the

modeled probability of belonging to that outcome (Tabachnick and Fidell 2001). This evaluation approach differs slightly from that used by Geist et al. (2000), which was based on the classification success. The classification success is determined by using a probability cutoff, usually 0.5, to classify each location as either spawning or non-spawning and then comparing the predicted and actual classifications. The prediction success rate is recommended over the classification success rate because the prediction success rate is not sensitive to the choice of probability cutoff used to classify each case (Tabachnick and Fidell 2001).

4.1.4 Dynamic Analysis and Additional Logistic Regression Modeling

Although high percentages of correct prediction rates were obtained for both actual spawning and non-spawning locations by Geist et al. (2000) and by this study, the large number of actual non-spawning cells still led to very large errors for the predicted spawning cells, even when combined with the low prediction error rate for non-spawning cells. For example, approximately 63% to 77% of the predicted spawning cells were actually not used by salmon, depending on the year in which the observations were made. Further study was therefore conducted attempting to gain insight into the cause of that phenomenon.

All cells classified as spawning by the logistic regression models developed using the methods described above (and a probability cutoff of 0.5) were extracted and identified as belonging to one of two groups—a group that consists of cells classified as spawning, which occurred within an observed redd cluster, and a group consisting of cells classified as spawning that did not occur within an observed redd cluster. To determine if the variability of the flow rate and the resulting variations in velocity and depth could be used to understand the difference between locations classified as spawning where spawning occurred and those locations where it did not, several new variables were generated that reflected the variability in velocity and depth at each location as well as the persistence of velocity and depth conditions that appeared to be favorable or unfavorable. This supplemental analysis is referred to as the dynamic analysis.

Complete records of hourly discharge rates at the Priest Rapids Dam were available for the three years. Figure 4.2 shows the hourly discharge rates for Priest Rapids Dam for 1994, 1995, and 2001. There were a total of 1464 hourly records of flow discharge rate for each year starting on 1 October and ending on 30 November. Those hourly discharge rates were used as input to MASS2, which produced time series of velocity and depth for each cell in the study. For the logistic regression analysis described above, the values of velocity and depth used for each cell were static values based on the 50% exceedance daily flow discharge rates for each year. For the dynamic analysis, a numeric simulation of the velocity and the water depth were performed for 15 discharge rates ranging from 40,000 to 180,000 cfs with a 10,000-cfs increment, which covers the discharge rate range for 1994, 2001, and most of 1995. These simulations were performed using MASS2. Based on these simulations, hourly time series of velocity and depth were obtained for all cells by taking the simulated velocity and depth at the discharge rate closest to the actual discharge rate from the Priest Rapids Dam at that hour.

These time series of velocity and depth were analyzed to identify differences between spawning and non-spawning locations. In the analysis we prepared frequency plots comparing the values in the time series to published habitat suitability criteria (Table 4.2). The results of that analysis, which are reported later in this section, showed that there were differences in the velocity time series for the two classes of locations.

Figure 4.2. Hourly Discharge Rates of Priest Rapids Dam During October and November 1994, 1995, and 2001. KCFS = thousand cubic feet per second.

Table 4.2. Habitat Suitability Criteria for the Hanford Reach (Hanrahan et al. 2004)

Depth (m)	Suitability Index (Hanford Reach only)	Velocity (m/s)	Suitability Index (Hanford Reach only)
0.3–0.5	0.04	0.10–0.25	0.03
0.5–1.0	0.04	0.25–0.50	0.35
1.0–1.5	0.21	0.50–0.75	0.52
1.5–2.0	0.61	0.75–1.00	1
2.0–2.5	1	1.00–1.25	0.73
2.5–3.0	0.99	1.25–1.50	0.26
3.0–3.5	0.65	1.50–1.75	0.1
3.5–4.0	0.63	1.75–2.00	0.1
4.0–4.5	0.19		
4.5–5.0	0.15		
5.0–5.5	0.08		
5.5–6.0	0.04		
6.0–6.5	0.02		
6.5–7.0	0.02		
7.0–8.0	0.02		
8.0–9.0	0.02		

Three new variables were retained to examine the influence of summary variables that attempt to capture the variability and persistence of the velocity time series:

- the proportion of velocity observations at each cell that were less than 1.0 m/s
- the proportion of velocity observations that fell between 1.0 and 2.5 m/s
- the standard deviation of the velocity values in each time series.

The first two variables were measures of the persistence of unfavorable and favorable velocities, while the third measured the variability in the velocity time series. The sequential logistic regression procedure described above was then repeated for each year, with the only change being the addition of the three dynamic velocity variables. The goal of that analysis was to see if adding variables that accounted for the persistence and variability in the velocity time series would improve the accuracy of the resulting logistic regression models, which would be seen by a significant decrease in the errors of commission for the prediction success rates.

4.2 Results

4.2.1 General Results

Spawning in the Hanford Reach consistently occurs in the same areas. This was briefly discussed in Geist et al. (2000) and in Section 3 of this report, which showed the reoccupation of habitat cells in 1994 and 1995 for the Locke Island area. Examination of the data collected for this study showed that this phenomenon is widespread in the Hanford Reach and occurs for all years studied. Figure 4.3 provides a

Figure 4.3. Sample Map Showing Consistency of Spawning for Site 64-RB. Red indicates spawning in all 3 years; green cells were used 2 of the years; blue cells were used 1 year; and gray cells were never occupied.

sample map for study site showing the distribution of spawning during the three years of the study. Table 4.3 shows the distribution of spawning for all study sites where spawning occurred. Spawning distributions in both 1995 and 2001 are compared with 1994, the year with the largest number of redds in the study sites. The χ^2 results for the individual tables and for all data taken together indicate that spawning is highly non-random, with large numbers of cells consistently occupied in all three years or never occupied. Note that there is a greater tendency for sites to be unoccupied in 1995 and 2001 than were occupied in 1994 than the reverse, due to the larger number of redds present in the study sites in 1994.

Tables 4.4A and 4.4B present univariate summaries of the static variables incorporated in the primary logistic regression for each of the three years. In Table 4.4A, the statistics for the non-spawning cells are presented for each year. Table 4.4B provides the same statistics for the spawning cells. Figure 4.4 shows the box plots of four variables—grainsize of the substrate, bed slope, and the average velocity and depth at the appropriate discharge rate. Because the grainsize of the substrate and the bed slope have wide value ranges, the box plots for those variables are shown on a logarithmic scale for better visualization.

It can be seen from Table 4.4 and Figure 4.4 that the values of all four variables present some extent of separation between spawning and non-spawning cells. Velocity appears to provide the greatest separation between the two categories, followed by depth, then substrate and bed slope. The results indicate that salmon habitat cells tend to have higher velocity, shallower depth, finer grainsize of the substrate, and lower slope. Spawning habitat cells also tend to have relatively small variation in each of the variables. Analysis of variance (ANOVA) confirms the significant difference in the river variables between the two categories (not shown).

Table 4.3. Consistency of Spawning Locations for 1994, 1995, and 2001. Part A compares spawning results from 1994 and 1995, while Part B compares 1994 and 2001. Comparisons were tested using a series of cross tabulations and χ^2 tests.

Part A

Study Area 59-RB			Study Area 59-LB			Study Area 61-RB			Study Area 61-LB			Study Area 64-RB			Study Area 81			All Data		
	Year 1995			Year 1995			Year 1995			Year 1995			Year 1995			Year 1995			Year 1995	
Year 1994	Yes*	No	Year 1994	Yes	No	Year 1994	Yes	No	Year 1994	Yes	No	Year 1994	Yes	No	Year 1994	Yes	No	Year 1994	Yes	No
Yes	664	208	Yes	91	513	Yes	403	256	Yes	150	327	Yes	6,041	2,665	Yes	6,307	2,625	Yes	13,656	6,594
No	179	1,425	No	20	10,136	No	135	6,639	No	244	14,872	No	776	2,004	No	598	2,015	No	1,952	175,428
χ^2		1,062	χ^2		1,235	χ^2		3,131	χ^2		1,671	χ^2		1,502	χ^2		1,916	χ^2		109,956
p-value		0	p-value		0	p-value		0	p-value		0	p-value		0	p-value		0	p-value		0

Part B

Study Area 59-RB			Study Area 59-LB			Study Area 61-RB			Study Area 61-LB			Study Area 64-RB			Study Area 81			All Data		
	Year 2001			Year 2001			Year 2001			Year 2001			Year 2001			Year 2001			Year 2001	
Year 1994	Yes	No	Year 1994	Yes	No	Year 1994	Yes	No	Year 1994	Yes	No	Year 1994	Yes	No	Year 1994	Yes	No	Year 1994	Yes	No
Yes	592	280	Yes	21	583	Yes	67	592	Yes	0	477	Yes	5,830	2,876	Yes	3,608	5,324	Yes	10,118	10,132
No	310	1,294	No	0	10,156	No	20	6,754	No	0	15,116	No	810	1,970	No	505	2,108	No	1,645	175,735
χ^2		575.3	χ^2		353.8	χ^2		505.9	χ^2			χ^2		1,236	χ^2		391.2	χ^2		7,807,709
p-value		0	p-value		0	p-value		0	p-value			p-value		0	p-value		0	p-value		0

*Yes/No denotes presence or absence of spawning in habitat cells for years being compared.

Table 4.4. Univariate Summaries of Static Variables for Spawning and Non-Spawning Locations

Part A

Non-Spawning - 1994					Non-Spawning - 1995					Non-Spawning - 2001				
	Substrate	Slope	V050 ¹	D050 ²		Substrate	Slope	V070 ³	D070 ⁴		Substrate	Slope	V050 ¹	D050 ²
Num	177,380	177,380	177,380	177,380	Num	182,022	182,022	182,022	182,022	Num	185,867	185,867	185,867	185,867
Min	0.1	0.00768	0	0.103	Min	0.1	0.00768	0	0.777	Min	0.1	0.00768	0	0.103
Max	1,460.5	101.123	2.095	14.973	Max	1,460.5	101.123	2.204	15.757	Max	1,460.5	101.123	2.095	14.973
Med	82.4	3.97508	0.742	3.849	Med	81.5	3.91578	1.14	4.659	Med	81	3.87543	0.759	3.667
Mean	118.017	5.4367	0.73906	4.22761	Mean	116.725	5.37416	1.07421	5.09243	Mean	115.715	5.33666	0.76067	4.14601
Stdev	114.43	5.40011	0.40531	2.7696	Stdev	113.412	5.37199	0.43695	2.71415	Stdev	112.642	5.35278	0.41335	2.73448

Part B

Spawning - 1994					Spawning - 1995					Spawning - 2001				
	Substrate	Slope	V050 ¹	D050 ²		Substrate	Slope	V070 ³	D070 ⁴		Substrate	Slope	V050 ¹	D050 ²
Num	20,250	20,250	20,250	20,250	Num	15,608	15,608	15,608	15,608	Num	11,763	11,763	11,763	11,763
Min	9.3	0.01175	0.01	0.31	Min	9.3	0.02302	0.112	1.105	Min	9.3	0.01175	0.154	0.346
Max	504.6	88.0891	2.073	4.403	Max	504.6	60.1348	2.221	5.619	Max	185.9	23.2518	2.073	4.171
Med	59.6	2.5786	1.227	2.13	Med	58.6	2.66414	1.492	2.808	Med	60.2	2.76049	1.204	1.884
Mean	63.7969	3.25866	1.22929	2.0975	Mean	62.7371	3.3403	1.49703	2.78011	Mean	61.0406	3.26797	1.24156	1.85
Stdev	30.8926	2.96098	0.32276	0.75497	Stdev	28.0417	2.81936	0.28823	0.72864	Stdev	18.4645	2.32369	0.33253	0.74452

¹Velocity associated with 50th percentile exceedance flow of 50 kcfs.
²Depth associated with 50th percentile exceedance flow of 50 kcfs.
³Velocity associated with 50th percentile exceedance flow of 70 kcfs.
⁴Depth associated with 50th percentile exceedance flow of 70 kcfs.

Figure 4.4. Static Variables for Spawning and Non-Spawning Locations, by Year. The medians of the distributions are shown as the center of the notches, and the lower and upper quartiles as the hinges of the box plots. Asterisks and circles represent outlier data points at 1.5 and 3.0 times the interquartile range, respectively. The notches represent an approximate 95% confidence interval around the median (McGill et al. 1978).

4.2.2 Logistic Regression Analysis with Only Static Variables

As shown in Table 4.5, the number of significant terms introduced by the sequential logistic regression process varies from 9 to 13 for the three different years. It is interesting to note that the velocity and the depth are the two variables selected first for all three years. Table 4.5 also shows that large

Table 4.5. Results of Sequential Logistic Regression Showing Order of Variable Selection for Each Year. Evaluation criteria in the table are defined in Section 4.2.2.

Year94

Variable	LL(N)	2*(LL(N-1)- LL(N))	p	McFadden's ρ^2
VEL	-5,080.64	3,701.66	0.00	0.267
DEP	-3,038.52	4,084.24	0.00	0.562
SUB	-2,651.53	773.98	0.00	0.617
DEP ²	-2,240.06	822.95	0.00	0.677
SUB ²	-2,192.02	96.08	0.00	0.684
VEL ²	-2,144.10	95.83	0.00	0.691
SUB_VEL	-2,120.12	47.96	0.00	0.694
SUB_DEP	-2,107.56	25.13	0.00	0.696
SLOPE ²	-2,100.35	14.42	0.00	0.697
SUB_SLOPE	-2,082.66	35.39	0.00	0.700

Year95

Variable	LL(N)	2*(LL(N-1)- LL(N))	p	McFadden's ρ^2
DEP	-5,167.36	3,361.25	0.00	0.242
VEL	-3,142.62	4,049.49	0.00	0.547
SUB	-2,758.29	768.67	0.00	0.602
DEP ²	-2,642.88	230.81	0.00	0.619
SUB_VEL	-2,609.89	65.98	0.00	0.623
SUB_DEP	-2,577.91	63.95	0.00	0.628
SLOPE	-2,567.30	21.22	0.00	0.630
SLOPE_VEL	-2,522.08	90.43	0.00	0.636
VEL ²	-2,486.86	70.43	0.00	0.641
SLOPE ²	-2,453.52	66.69	0.00	0.646
SUB ²	-2,447.99	11.07	0.00	0.647
SLOPE_DEP	-2,443.83	8.31	0.00	0.647
SUB_SLOPE	-2,435.23	17.21	0.00	0.649

Year01

Variable	LL(N)	2*(LL(N-1)- LL(N))	p	McFadden's ρ^2
DEP	-5,138.50	3,361.25	0.00	0.259
VEL	-2,567.52	5,141.95	0.00	0.630
SUB	-2,154.94	825.16	0.00	0.689
VEL ²	-1,934.72	440.45	0.00	0.721
SUB ²	-1,895.60	78.29	0.00	0.727
VEL_DEP	-1,877.75	35.60	0.00	0.729
DEP ²	-1,844.73	66.1	0.00	0.734
SUB_VEL	-1,842.66	4.1	0.00	0.734
SUB_DEP	-1,833.69	18	0.00	0.735

improvements in the logistic regression statistics are seen only when the first three or four terms are introduced into the model. After that, although each additional variable brings statistically significant new information into the existing model, the size of the increase in the model evaluation criteria (e.g., χ^2 and ρ^2) decreases (see Figure 4.5). This suggests that the first three or four variables are sufficient for a regression model. Therefore, a simplified model was built with the first three or four variables in the sequence as an alternative to the full model. All the regression coefficients of the full model and the simplified model (Table 4.6) are statistically significant.

The performance of the simplified logistic regression model was evaluated using the prediction success rates for the testing data that were not included in training the models. Table 4.7 presents prediction success tables obtained by applying the simplified logistic regression models for each year on the testing data.

Figure 4.5. McFadden's ρ^2 as a Function of the Number of Terms in the Logistic Regression Models for the Three Years

From Tables 4.6 and 4.7, the following observations can be made. First, the variables in the simplified models are usually led by either the depth or the velocity terms in the first-order and followed by terms associated with grainsize of the substrate or squared terms of velocity, depth, grainsize of the substrate and cross terms between them. Second, the percentages of correct predictions are very high, ranging from 85% up to 98%—that is, from 85% up to 98% of spawning or non-spawning cells are predicted correctly by the logistic regression model. Third, the habitat cells predicted by the models as not suitable for spawning are rarely used by salmon for spawning; that is, the error of omission is very low. However, large percentages of predicted spawning habitat cells are not actually used by salmon (a high error of commission). For the training data themselves, the omission error rates are between 2% and 7%, and the commission error rates are around 10%, which are both acceptable (not shown). However, for the testing data, the percentages of the predicted spawning habitat cells not actually used for spawning are nearly 63% for 1994, 76% for 1995, and 77% for 2001 (Table 4.7). The high error of commission on the test data and the tendency for salmon to occupy the same spawning areas each year both indicate that there may be some influential parameters not included in the logistic regression model and that further

Table 4.6. Regression Coefficients of the Full and Simplified Models for the Sequential Logistic Regression for Each Year

1994			1995			2001		
Variable	Full	Simplified	Variable	Full	Simplified	Variable	Full	Simplified
CONSTANT	-6.208593	-4.837146	CONSTANT	-10.877573	-0.2535637	CONSTANT	-6.602791	-5.4395003
SUB	-0.040963	-0.023298	SUB	-0.037354	-0.0266335	SUB	0.032018	-0.0364240
SLOPE	0	0	SLOPE	0.778151	0	SLOPE	0	0
VEL	11.636048	6.2146894	VEL	12.818703	5.5540572	VEL	16.027081	19.6561532
DEP	2.761529	2.9151838	DEP	1.931214	-1.4977077	DEP	-1.692511	-2.2910001
SUB ²	0.000049	0	SUB ²	0.000037	0	SUB ²	-0.000531	0
SLOPE ²	0.002165	0	SLOPE ²	-0.006081	0	SLOPE ²	0	0
VEL ²	-3.611217	0	VEL ²	-3.146576	0	VEL ²	-7.958878	-5.8423473
DEP ²	-0.726855	-0.8956617	DEP ²	-0.370344	0	DEP ²	-0.654628	0
SUB_SLOPE	-0.001119	0	SUB_SLOPE	-0.001908	0	SUB_SLOPE	0	0
SUB_VEL	0.023569	0	SUB_VEL	0.029883	0	SUB_VEL	0.03987	0
SUB_DEP	-0.007992	0	SUB_DEP	-0.011071	0	SUB_DEP	-0.014061	0
SLOPE_VEL	0	0	SLOPE_VEL	-0.29039	0	SLOPE_VEL	0	0
SLOPE_DEP	0	0	SLOPE_DEP	-0.057797	0	SLOPE_DEP	0	0
VEL_DEP	0	0	VEL_DEP	0	0	VEL_DEP	2.870737	0

investigation of the factors controlling spawning is needed. These might include the assessment of other variables and/or different spatial or temporal approaches to incorporating the current variables into the analysis. Figure 4.6 shows the predicted and actual results for one of the study areas, 64-RB. As illustrated, most of the areas surrounding spawning areas were predicted also to be spawning areas although they were not used for spawning by salmon.

The prediction performance presented in Table 4.7 was recalculated in terms of the amount of area belonging to each outcome (e.g., predicted spawning, actual spawning) and presented in Table 4.8. Instead of showing the sum of the probability values for each cell, the area was calculated based on the fact that each cell represents an area of about 9 m². For each study site, the probability that each cell within a study site would be predicted to be a spawning cell first was multiplied by the area of the cell; then the areas for all cells in a study area were added. The percentages of the acreage within each study area belonging to each habitat type are also included. The results in Table 4.8 indicate that there are several locations where spawning did not occur during the three seasons studied, including study areas 54, 64-LB, 90, 94, 96, and 101. For most of those areas, the portion of the area predicted to be non-spawning was large, and the overall error rates were very low. For example, at least 94% of study areas 54, 64-LB, 94, and 101 were predicted to be non-spawning in all three years of the study. Study areas 90 and 96 were an exception to that general trend, with anywhere from 4% to 17% of study area 90 predicted to be spawning and between 26% and 40% of study area 96 predicted to be spawning, although no spawning was observed in either of those areas. In addition, several study areas where spawning occurred had much less productive acreage than the model predicted. For example, study areas 59-RB, 64-RB, and 81 all have substantial amounts of acreage predicted to be spawning habitat in which spawning actually occurred. However, those areas also had from 30% to 55% of their area predicted to be spawning by the

Table 4.7. Prediction Success Rates for the Simplified Logistic Regression Model. Total number of habitat cells in the test data set was 187,630.

1994		
	Predicted Spawning	Predicted Non-Spawning
Actual Spawning	13,216.13	2,033.87
Actual Non-Spawning	22,657.66	149,722.34
Correct	86.66%	86.86%
Error	63.16%	1.34%
1995		
	Predicted Spawning	Predicted Spawning
Actual Spawning	8,924.73	1,683.27
Actual Non-Spawning	28,447.87	14,8574.13
Correct	84.13%	83.93%
Error	76.12%	1.12%
2001		
	Predicted Spawning	Predicted Spawning
Actual Spawning	6,029.65	733.35
Actual Non-Spawning	20,305.63	16,0561.37
Correct	89.16%	88.77%
Error	77.10%	0.45%

logistic regression model in which no spawning occurred. The smallest errors of overprediction occurred in 1994 (Table 4.8), when the escapement numbers were largest, suggesting that at least a portion of the overprediction errors in 1995 and 2001 are associated with the smaller numbers of salmon available to occupy the suitable spawning habitat.

The tendency for the models to overpredict the proportion of areas that should be useable for spawning led to an effort to identify other factors that might contribute to that tendency. As discussed in Methods (Section 4.3), all cells throughout the study areas were classified as spawning or non-spawning based on a probability cutoff of 0.5 for each year of the study. The cells that were predicted to be spawning by the model were then examined to look for additional factors that might allow generation of a better predictive model. The results of that additional analysis follow.

4.2.3 Dynamic Analysis

The results presented in Section 4.2.2 show that a significant portion of the cells classified as spawning cells by the logistic regression model were not actually used by salmon. The distribution of the classified spawning cells in terms of where they occur in the study areas and the number of actual spawning and non-spawning cells is presented in Table 4.9.

Figure 4.6. Maps for Study Area 64-RB Showing Locations of Predicted Spawning Cells and the Actual Spawning Cells for 1994, 1995, and 2001. The results for the simplified models from the sequential process were used to construct the maps. Color coding maps the consistency and inconsistency between the predicted and actual spawning. Green indicates cells predicted to be used for spawning and actually used for spawning; grey denotes cells predicted as unsuitable and actually not used for spawning; red indicates the cells predicted to be used but actually not used for spawning (commission error), and blue denotes the cells predicted not to be used but actually used for spawning (omission error).

Figure 4.7 shows the frequency in the time series of velocity and depth calculated using MASS2. Each point on the plot represents the mean frequency with which the velocity (m/s) or depth (m) values in the hourly time series fell into the suitability criteria bins identified in Table 4.2. The frequency values are plotted at the midpoints of the suitability criteria bins. Comparing the plots for velocity, we found that the major differences in the mean occurrence frequencies between spawning and non-spawning cells are that spawning cells have higher frequency for velocities between 1 and 2.5 m/s and lower frequencies below 1m/s. Non-spawning cells tend to have higher frequencies of values falling in the range less than 1 m/s. The results in Figure 4.7 show that there was not as large a difference between the depth frequency curves for spawning and non-spawning cells as occurs for the velocity frequency plots.

Table 4.8. Area (m²) Belonging to Habitat Classes in Each Study Area

Y1994	P-S,A-S*		P-S,A-NS*		P-NS,A-S*		P-NS,A-NS*		Total
54	0	(0%)	171.27	(0%)	0	(0%)	174,788.7	(99.9%)	174,960
59-RB	7,210.53	(32.3%)	11,376	(51%)	637.47	(2.8%)	3,060	(13.7%)	22,284
59-LB	3,270.42	(3.3%)	6,638.31	(6.8%)	2,165.58	(2.2%)	84,765.69	(87.5%)	96,840
61-RB	4,467.69	(6.6%)	9,905.58	(14.8%)	1,463.31	(2.1%)	51,060.42	(76.3%)	66,897
61-LB	756.27	(0.5%)	6,424.47	(4.5%)	3,536.73	(2.5%)	129,619.5	(92.3%)	140,337
64-RB	74,474.37	(72%)	21,682.8	(20.9%)	3,879.63	(3.7%)	3,337.2	(3.2%)	103,374
64-LB	0	(0%)	2,800.53	(4.9%)	0	(0%)	53,224.47	(95%)	56,025
81	67,761	(65.2%)	14,802.66	(14.2%)	12,627	(12.1%)	8,714.34	(8.3%)	103,905
90	0	(0%)	19,308.06	(10.1%)	0	(0%)	170,060.9	(89.8%)	189,369
94	0	(0%)	22,824.72	(6%)	0	(0%)	355,103.3	(93.9%)	377,928
96	0	(0%)	92,960.91	(36.4%)	0	(0%)	161,928.1	(63.5%)	254,889
101	0	(0%)	1,028.52	(0.5%)	0	(0%)	190,833.5	(99.4%)	191,862
Total	157,940.2	(8.8%)	209,923.9	(11.8%)	24,309.81	(1.3%)	1,386,496	(77.9%)	1,778,670

Y1995	P-S,A-S*		P-S,A-NS*		P-NS,A-S*		P-NS,A-NS*		Total
54	0	(0%)	61.2	(0%)	0	(0%)	174,898.8	(99.9%)	174,960
59-RB	7,098.48	(31.8%)	12,165.39	(54.5%)	488.52	(2.1%)	2,531.61	(11.3%)	22,284
59-LB	854.01	(0.8%)	10,438.83	(10.7%)	144.99	(0.1%)	85,402.17	(88.1%)	96,840
61-RB	2,633.13	(3.9%)	6,726.51	(10%)	2,208.87	(3.3%)	55,328.49	(82.7%)	66,897
61-LB	433.26	(0.3%)	9,063.72	(6.4%)	3,112.74	(2.2%)	127,727.3	(91%)	140,337
64-RB	52,923.33	(51.1%)	34,549.2	(33.4%)	8,429.67	(8.1%)	7,471.8	(7.2%)	103,374
64-LB	0	(0%)	1,294.92	(2.3%)	0	(0%)	54,730.08	(97.6%)	56,025
81	54,076.14	(52%)	30,896.55	(29.7%)	8,068.86	(7.7%)	10,863.45	(10.4%)	103,905
90	0	(0%)	31,946.04	(16.8%)	0	(0%)	157,423	(83.1%)	189,369
94	0	(0%)	22,634.55	(5.9%)	0	(0%)	355,293.5	(94%)	377,928
96	0	(0%)	102,110.2	(40%)	0	(0%)	152,778.8	(59.9%)	254,889
101	0	(0%)	1,448.1	(0.7%)	0	(0%)	190,413.9	(99.2%)	191,862
Total	118,018.3	(6.6%)	263,335.2	(14.8%)	22,453.74	(1.2%)	1,374,863	(77.2%)	1,778,670

Y2001	P-S,A-S*		P-S,A-NS*		P-NS,A-S*		P-NS,A-NS*		Total
54	0	(0%)	0.27	(0%)	0	(0%)	174,959.7	(99.9%)	174,960
59-RB	7,257.78	(32.5%)	11,207.97	(50.2%)	860.22	(3.8%)	2,958.03	(13.2%)	22,284
59-LB	106.02	(0.1%)	5,460.66	(5.6%)	82.98	(0%)	91,190.34	(94.1%)	96,840
61-RB	653.49	(0.9%)	6,275.97	(9.3%)	129.51	(0.1%)	59,838.03	(89.4%)	66,897
61-LB	0	(0%)	1,412.19	(1%)	0	(0%)	138,924.8	(98.9%)	140,337
64-RB	54,462.96	(52.6%)	35,621.28	(34.4%)	5,297.04	(5.1%)	7,992.72	(7.7%)	103,374
64-LB	0	(0%)	38.07	(0%)	0	(0%)	55,986.93	(99.9%)	56,025
81	31,814.82	(30.6%)	47,019.51	(45.2%)	5,202.18	(5%)	19,868.49	(19.1%)	103,905
90	0	(0%)	6,802.74	(3.5%)	0	(0%)	182,566.3	(96.4%)	189,369
94	0	(0%)	6,584.31	(1.7%)	0	(0%)	371,343.7	(98.2%)	377,928
96	0	(0%)	67,179.96	(26.3%)	0	(0%)	187,709	(73.6%)	254,889
101	0	(0%)	119.52	(0%)	0	(0%)	191,742.5	(99.9%)	191,862
Total	94,294.98	(5.3%)	187,722.5	(10.5%)	11,572.02	(0.6%)	1,485,080	(83.4%)	1,778,670
* P = predicted, A = actual, S = spawning, NS = non-spawning.									

Table 4.9. Classified Spawning Cells for Each Study Site for Each of the Three Years Using the Logistic Regression Model and a Probability Cutoff of 0.5. Classified spawning cells are identified as to whether spawning actually occurred or not.

Year 1994			
Study Area	Classified Spawning, Actual Spawning	Classified Spawning, Actual Non-Spawning	Total
54	0	0	0
59-RB	852	1,327	2,179
59-LB	448	677	1,125
61-RB	547	633	1,180
61-LB	0	244	244
64-RB	8,644	2,509	11,153
64-LB	0	0	0
81	8,324	1,754	10,078
90	0	835	835
94	0	562	562
96	0	9,935	9,935
101	0	0	0
Overall	18,815	18,476	37,291

Year 1995			
Study Area	Classified Spawning, Actual Spawning	Classified Spawning, Actual Non-Spawning	Total
54	0	0	0
59-RB	841	1,515	2,356
59-LB	111	798	909
61-RB	257	311	568
61-LB	8	211	219
64-RB	6,692	4,117	10,809
64-LB	0	0	0
81	6,838	4,100	10,938
90	0	1,160	1,160
94	0	547	547
96	0	11,577	11,577
101	0	5	5
All	14,747	24,341	39,088

Year 2001			
Study Area	Classified Spawning, Actual Spawning	Classified Spawning, Actual Non-Spawning	Total
54	0	0	0
59-RB	847	1,382	2,229
59-LB	14	411	425
61-RB	79	547	626
61-LB	0	10	10
64-RB	6,560	4,393	10,953
64-LB	0	0	0
81	3,936	5,620	9,556
90	0	74	74
94	0	20	20
96	0	6,926	6,926
101	0	0	0
All	11,436	19,383	30,819

Figure 4.7. Mean Frequency Plots for Velocity and Depth in 1994, 1995, and 2001 for Habitat Cells Predicted To Be Spawning Cells. Red curves are for cells where spawning did not occur, while blue curves are the frequencies for cells where spawning actually occurred.

Figure 4.7 suggests that there are differences in the velocity time series for cells where spawning did not occur and those where it did, even though all of the cells examined for the dynamic analysis were predicted to be spawning locations based on the logistic regression performed for the 50% exceedance flow conditions. These differences were found in information gleaned from the full time series, which were not included in the logistic regression. To further examine the differences between velocity and depth time series for the two groups of cells, we prepared box plots of the velocity and depth. Examining the frequency plots presented in Figure 4.7, it appears that spawning cells tend to be associated with velocities between 1.0 and 2.5 m/s and depths between 1.5 and 4.0 m. For each spawning cell, we calculated the proportion of values within the velocity time series for each cell that fell into one of three classes: less than 1.0 m/s, between 1.0 and 2.5 m/s, and greater than 2.5 m/s. We then summarized the proportion of cells meeting those criteria for spawning and non-spawning cells in a series of box plots, shown in the top half of Figure 4.8. Similar box plots were prepared for the proportion of values in the depth time series for each cell that fell into one of three classes: less than 1.5 m, between 1.5 and 4.0 m, and greater than 4.0 m. The box plots for depth are presented in the bottom half of Figure 4.8.

Figure 4.8. Box Plots of Velocity and Depth Values for Spawning and Non-Spawning Locations for 1994. The medians of the distributions are shown as the center of the notches, and the lower and upper quartiles as the hinges of the box plots. Asterisks and circles represent outlier data points at 1.5 and 3.0 times the interquartile range, respectively. The notches represent an approximate 95% confidence interval around the median (McGill et al. 1978).

The velocity box plots in the top half of Figure 4.8 indicate that there was a significant difference in 1994 in the proportion of values in the velocity time series for each cell that fell within the preferred range (1.0 and 2.5 m/s) for spawning and non-spawning cells. There were only a few spawning cells where the proportion of the velocity values outside the preferred range was less than 100%, while a substantial number of non-spawning cells had high proportions of velocities that fell outside the preferred range. For example, about 25% of the non-spawning cells had less than 60% of their velocity values that fell within the preferred range. Comparison of the left and right figures in the top half of Figure 4.8 indicates that most of the non-spawning cells that had velocity values that fell outside the preferred range had velocities lower than those in the preferred range.

A similar effect can be seen in the box plots of depth for 1994 in the lower panel of Figure 4.8, although the effect is not as strongly developed. In the middle box plot, there appears to be a significant difference between the proportion of depth values in the time series for a cell that fall within the preferred depth range of 1.5 and 4.0 m, with spawning cells tending to have a higher proportion of cells that fall in the preferred range. Examining the figures to the left and right, it appears that spawning cells are far less likely than non-spawning cells to have a measurable proportion of depth values less than 1.5 m, while spawning and non-spawning cells are equally likely to have a significant proportion of depth values that exceed 4.0 m. Similar results can be seen for 1995 and 2001 (not shown). Taking all three years together, we see that there is a major difference between the velocity time series for spawning and non-spawning cells. Non-spawning cells are far more likely to have a major proportion of velocity values outside the preferred range, with more than half of the non-spawning cells having at least 10% of the velocity values falling below the 1.0 m/s threshold. There does appear to be a shift in depth values between spawning and non-spawning cells, with spawning cells tending to have a higher proportion of depth values within the preferred threshold, although the relationship for depth is not nearly as strong as it is for velocity.

Based on the results of the previous analysis, a decision was made to retain three new variables, termed dynamic variables, that would serve as summary measures for the time series of the modeled velocity data and use them to run an expanded logistic regression analysis. *VEL1* measures the proportion of velocity values at a location that are less than 1.0 m/s; it provides a summary value of the tendency for unfavorable velocity conditions to persist. The second dynamic variable, *VEL2*, identifies the proportion of velocity values that fall in the range from 1.0 to 2.5 m/s and thus provides a measure of the persistence of favorable velocity conditions in the time series. The third dynamic variable, *VEL3*, is the standard deviation of the velocity time series at each location and provides a direct measure of the variability of the velocity. Figure 4.9 shows box plots for the three dynamic variables for 1994. The box plots suggest that there is excellent separation between spawning and non-spawning locations for *VEL1* and *VEL2*. The variability of the velocity time series is higher for non-spawning locations; however, the degree of separation for *VEL3* between the spawning and non-spawning locations is not as good as it is for *VEL1* and *VEL2*. Similar results were found for 1995 and 2001.

Table 4.10 shows the results from the revised sequential logistic regression analysis that includes the three dynamic variables. In Table 4.10 and the discussion below, *VELO* is the static velocity value that represents the velocity in each cell associated with the 50% exceedance flow from Priest Rapids Dam. *VELO* is equal to the velocity variable used in the original logistic regression.

Figure 4.9. Dynamic Variables *VEL1*, *VEL2*, and *VEL3* by Spawning and Non-Spawning Sites (Labeled as 1 and 0) in Year 1994. The medians of the distributions are shown as the center of the notches, and the lower and upper quartiles as the hinges of the box plots. Asterisks and circles represent outlier data points at 1.5 and 3.0 times the interquartile range, respectively. The notches represent an approximate 95% confidence interval around the median (McGill et al. 1978).

The velocity and depth again show up as important variables in the sequential selection process. In all three years, the dynamic variable *VEL2*, which accounts for the proportion of favorable values in the velocity time series at a location, was chosen early in the selection process. For example, in 1994, *VELO* was chosen as the first variable with *VEL2* chosen fourth, while in 1995 and 2001, *VEL2* was chosen second (after the depth) while *VELO* was chosen fourth. Table 4.10 and Figure 4.10 show that after the first three to four variables, there is not much improvement in McFadden's ρ^2 or other model selection criteria. Table 4.11 provides the coefficients of the logistic regression model for the full and simplified models for the sequential logistic regression that include the dynamic velocity variables.

The performance of the revised logistic regression model that included the dynamic variables (Table 4.12) was only slightly better than that of the original model. For example, comparison of the data in Tables 4.7 and 4.12 shows that in 1994 the number of cells predicted to be spawning that were actually non-spawning decreased from 22,658 to 22,506. This results in an extremely small decrease in the error of commission rate from 63.2 to 63.0, which would not be significant, given the potential uncertainty in

Table 4.10. Sequentially Selected Variables with Priority on the First-Order Terms for Years 1994, 1995, and 2001. Evaluation criteria are defined in Section 4.2.3. *VELO* and *DEP0* are the static velocity and depth variables used in the original logistic regression. All parameters are statistically significant with $P < 0.005$.

Year	Variables	LL(N)	McFadden's ρ^2	2*(LL(N)-LL (N-1))	% ρ^2
1994	VELO	-5,080.642	0.267	3,701.659	37.67%
	DEP0	-3,038.523	0.562	4,084.239	79.23%
	SUB	-2,651.533	0.618	773.979	87.12%
	VEL2	-2,550.270	0.632	202.526	89.18%
	DEP0 ²	-2,215.959	0.680	668.622	95.98%
	SUB ²	-2,162.725	0.688	106.467	97.07%
	VELO ²	-2,137.614	0.692	50.222	97.57%
	VEL1 ²	-2,114.096	0.695	47.036	98.05%
	VEL2 ²	-2,092.894	0.698	42.405	98.49%
	SLOPE ²	-2,086.438	0.699	12.912	98.62%
	SUB_SLOPE	-2,056.732	0.703	59.413	99.22%
	SUB_VELO	-2,044.377	0.705	24.709	99.48%
	DEP0_VELO3	-2,036.385	0.706	15.985	99.63%
	SUB_VELO3	-2,023.376	0.708	26.018	99.90%
	SUB_VELO2	-2,018.393	0.709	9.965	100.00%
1995	DEP0	-5,167.359	0.255	3,528.226	38.14%
	VEL2	-3,073.440	0.557	4,187.837	83.42%
	SUB	-2,761.311	0.602	624.259	90.17%
	VELO	-2,659.595	0.616	203.432	92.37%
	SLOPE	-2,643.218	0.619	32.754	92.73%
	DEP0 ²	-2,561.253	0.631	163.931	94.50%
	VEL3	-2,495.215	0.640	132.075	95.92%
	SUB ²	-2,466.727	0.644	56.976	96.54%
	VEL1 ²	-2,438.651	0.648	56.152	97.15%
	SLOPE_VELO	-2,398.742	0.654	79.818	98.01%
	SUB_VELO	-2,374.168	0.658	49.149	98.55%
	SLOPE ²	-2,351.061	0.661	46.214	99.04%
	SLOPE_VELO3	-2,337.780	0.663	26.561	99.33%
	SUB_SLOPE	-2,324.402	0.665	26.756	99.63%
	DEP0_VELO2	-2,313.016	0.666	22.772	99.87%
	SLOPE_DEP0	-2,307.140	0.667	11.753	100.00%
2001	DEP0	-5,138.496	0.259	3,585.951	35.09%
	VEL2	-2,483.295	0.642	5,310.403	87.03%
	SUB	-2,165.164	0.688	636.262	93.26%
	VELO	-2,023.474	0.708	283.379	96.04%
	VEL3	-1,994.137	0.712	58.675	96.61%
	VELO ²	-1,880.484	0.729	227.304	98.83%
	VEL1 ²	-1,854.075	0.733	52.820	99.35%
	VELO_VELO3	-1,837.176	0.735	33.798	99.69%
	VEL3 ²	-1,832.940	0.736	8.470	99.77%
	SLOPE_DEP0	-1,827.988	0.736	9.904	99.86%
	SLOPE	-1,821.097	0.737	13.782	100.00%

Figure 4.10. McFadden's ρ^2 Versus Number of Terms Added in Logistic Regression

the data used to train the model. The results for other comparisons of the models provide similar results, indicating that the incorporation of the dynamic variables that account for persistence of favorable velocities does not lead to a significant improvement in the prediction capability of the logistic regression model. Analysis of the correlation coefficients between the dynamic velocity variables and the static velocity variable (Table 4.13) indicated that there was a high correlation, about 0.9, between *VELO* and *VEL2*.

4.3 Discussion

Application of the logistic regression approach developed for Locke Island by Geist et al. (2000) produced similar results when applied over the entire Hanford Reach. The error rates for identification of areas that are not suitable spawning habitat were low, less than 2%, so that the models perform very well in prediction of areas that are not suitable habitat. Predictions of areas that are spawning habitat were not as successful, with between 63% and 77% of the areas predicted to be spawning habitat not being used (Table 4.7) for the test data. Note that this comparison is more stringent than that employed by Geist et al. (2000) because no test data were employed in training the logistic regression. In most cases, the error rates for training data sets are much lower than for test data sets that are not used in training multivariate statistical methods (Tabachnick and Fidell 2001). The lowest error rates in this study occurred for 1994, the year with the highest number of redds within the study sites. Overall, the results of the current study support those reported by Geist et al. (2000), indicating that an approach using characteristics measured within local spawning areas performs better for prediction of spawning habitat than standard PHABSIM approaches based on published redd selection criteria (Bovee 1995) or river-specific suitability criteria (Shirvell 1989; Bovee 1995).

In that regard, the habitat suitability criteria for velocity identified by published criteria for the Hanford Reach (Hanrahan et al. 2004) and shown in Table 4.2 do not appear to fit the data observed in this study. For example, the mean occurrence frequency plots (Figure 4.7) have a peak between 1 to 2.5 m/s for all three years, but the peak in the suitability criteria listed in Table 4.2 is between 0.25 and

Table 4.11. Coefficients for Full and Simplified Models for Years 1994, 1995, and 2001

Variables	Year 1994		Year 1995		Year 2001	
	Full	Simple	Full	Simple	Full	Simple
CONSTANT	-40,931.1504	-4.6151	-9.3262	-2.1086	-26.9080	0.4448
SUB	-0.0213	-0.0228	-0.0433	-0.0249	-0.0363	-0.0358
SLOPE	0	0	0.4945	0	0.1488	0
VEL0	12.4020	4.3050	1.5919	2.9405	37.8249	3.9971
DEP0	1.1985	2.6444	1.9072	-1.4842	-2.2344	-2.3906
VEL1	0	0	0	0	0	0
VEL2	81,848.2718	2.4459	13.9538	5.7963	7.9026	4.7026
VEL3	0	0	-7.3266	0	33.5909	0
SUB ²	0.00004	0	0.00004	0	0	0
SLOPE ²	0.0024	0	-0.0065	0	0	0
VEL0 ²	-4.2180	0	0	0	-11.3295	0
DEP0 ²	-0.7156	-0.8251	-0.2965	0	0	0
VEL1 ²	40,926.6991	0	5.4106	0	13.6150	0
VEL2 ²	-40,920.8887	0	0	0	0	0
VEL3 ²	0	0	0	0	-24.4526	0
SUB_SLOPE	-0.0012	0	-0.0026	0	0	0
SUB_VEL0	0.0338	0	0.0135	0	0	0
SUB_DEP0	0	0	0	0	0	0
SUB_VEL1	0	0	0	0	0	0
SUB_VEL2	-0.0355	0	0	0	0	0
SUB_VEL3	-0.0645	0	0	0	0	0
SLOPE_VEL0	0	0	-0.1473	0	0	0
SLOPE_DEP0	0	0	-0.0524	0	-0.0872	0
SLOPE_VEL1	0	0	0	0	0	0
SLOPE_VEL2	0	0	0	0	0	0
SLOPE_VEL3	0	0	0.5265	0	0	0
VEL0_DEP0	0	0	0	0	0	0
VEL0_VEL1	0	0	0	0	0	0
VEL0_VEL2	0	0	0	0	0	0
VEL0_VEL3	0	0	0	0	-21.9410	0
DEP0_VEL1	0	0	0	0	0	0
DEP0_VEL2	0	0	-1.5270	0	0	0
DEP0_VEL3	2.8029	0	0	0	0	0
VEL1_VEL2	0	0	0	0	0	0
VEL1_VEL3	0	0	0	0	0	0
VEL2_VEL3	0	0	0	0	0	0

1.5 m/s. Thus, the most suitable range in the suitability criteria table is not quite consistent with the highest occurrence frequency range of the spawning cells identified from the aerial photography.

Additional work done for this study examined the importance of the time series of flows in the Hanford Reach in predicting the location of spawning habitat and especially the variability and persistence of velocity and depth values associated with the flows. Dynamic variables based on hourly time series were identified that accounted for the variability in velocity and depth for each cell in the study areas in the Hanford Reach. The additional variables identified the proportion of velocity and depth

Table 4.12. Prediction Success for Revised Logistic Regression Model. Total number of habitat cells in the model was 187,630.

1994		
	Predicted Spawning	Predicted Non-Spawning
Actual Spawning	13,228.68	2,021.32
Actual Non-Spawning	22,505.85	14,9874.2
Correct	86.75%	86.94%
Error	62.98%	1.33%
1995		
	Predicted Spawning	Predicted Non-Spawning
Actual Spawning	9,006.61	1,601.39
Actual Non-Spawning	26,907.06	150,114.9
Correct	84.90%	84.80%
Error	74.92%	1.06%
2001		
	Predicted Spawning	Predicted Non-Spawning
Actual Spawning	5,992.61	770.39
Actual Non-Spawning	21,643.09	159,223.9
Correct	88.61%	88.03%
Error	78.32%	0.48%

Table 4.13. Coefficients of Correlation Between the Static Velocity, *VELO*, and *VEL1*, *VEL2*, *VEL3* for Years 1994, 1995, and 2001

Year	1994	1995	2001
VELO vs. VEL1	-0.9218	-0.9243	-0.9235
VELO vs. VEL2	0.9217	0.9234	0.9234
VELO vs. VEL3	-0.1568	-0.0515	-0.1792
VEL1 vs. VEL2	-0.99998	-0.99970	-0.99997

measurements that fell within specific ranges that appeared to be selected either for or against by spawning salmon. Analysis showed that several of those variables, and especially the proportion of velocities that fell within the range of 1.0 to 2.5 m/s (*VEL2*), differed significantly between locations that were predicted to be spawning by the logistic regression model where spawning actually occurred and locations that were predicted to be spawning where spawning did not occur. In addition, sequential logistic regression that included the original variables plus the additional dynamic variables for velocity indicated that *VEL2* was one of the most significant variables for prediction of spawning. However, the resulting sequential logistic regression model that incorporated the dynamic variables did not provide significant improvement in the percentage of errors for areas predicted to be spawning. The high

correlation between *VEL2* and *VELO* suggests that while *VEL2* is an important variable, it did not provide enough information that was significantly different from *VELO* to impact the error rates.

The continued high error rates for prediction of spawning habitat suggest that there are other factors that lead salmon to reject areas that appear to be suitable spawning habitat based on locally defined criteria for velocity, depth, and substrate. In many cases, the errors tend to occur near the edges of redd clusters, surrounding the areas that are predicted to be used and are actually used (e.g., see Figure 4.6a). However, there are also areas, shown in Table 4.8, where substantial acreage is predicted to be spawning where no spawning has been observed. As noted by Geist et al. (2000), the additional factors that might explain the errors in those areas are the presence of geomorphic features, including sedimentary structures on the river bottom, and hyporheic upwelling. For example, substrate quality as defined by riverbed permeability and riverbed grainsize is higher in spawning areas than non-spawning areas (see Section 5).

4.4 Conclusion

In conclusion, the analysis performed for this study confirms that the approach of using sequential logistic regression to determine local factors that influence the choice of salmon habitat used by Geist et al. (2000) can be applied to study areas throughout the Hanford Reach, not just the Locke Island and Wooded Island areas. The dynamic analysis and associated supplemental logistic regression model suggest that the persistence of favorable velocity conditions over the course of a spawning season is an important variable for predicting salmon habitat suitability but does not do an appreciably better job of prediction than the velocity associated with the 50% exceedance flow rate.

5.0 Assessment of Substrate Quality of Fall Chinook Salmon Spawning Habitat in the Hanford Reach

The quality of salmonid spawning gravel is an important indicator of anthropogenic impacts to rivers. Management actions such as dam construction, bank armoring, and other flood protection prescriptions reduce the supply of smaller, mobile gravels to rivers by capturing sediment before it can be transported downstream (e.g., Parfitt and Buer 1980). Water that passes over riverbed deposits eventually winnows away the mobile particles, resulting in a coarsening of the grains on the riverbed's surface; this process is referred to as armoring and results in larger particles comprising the bed's framework (the structure of the deposit) relative to the particles comprising the pore spaces of the framework (i.e., the bed matrix). Timber harvesting, road building, and agricultural encroachment into riparian habitats often result in an increase in the concentration of fine sediment (generally <1 mm) within the bed matrix (Montgomery and Buffington 1997; Stover and Montgomery 2001; Micheli et al. 2004). As the proportion of fine sediment within the bed matrix increases, sediment permeability decreases owing to the reduction in the amount of open pore spaces through which water can move (Chapman 1988; Shepherd 1989).

Spawning gravel requirements of salmonids change depending on their life stage (Kondolf 2000). During spawning, adult salmonids must be able to move the particles on the riverbed surface in order to construct redds where eggs will be deposited. Bed coarsening results in a bed framework that may be too large for adult salmon to move during redd construction. Thus, the size distribution of the bed framework sets an upper size limit to suitable spawning gravels (Kondolf 2000); this upper size limit will vary with the size of the fish because larger fish are capable of moving larger particles (Kondolf and Wolman 1993). Sediment intrusion into the river bed where salmonid eggs are placed clogs the pore spaces within the bed framework. An increase in the fine material within the bed matrix reduces intragravel water velocity, which in turn lowers incubation success by reducing the amount of oxygen delivered to eggs (Chapman 1988). Although results vary, intragravel dissolved oxygen levels from 2 to 8 mg/L are required for successful incubation of salmonid eggs (Coble 1961; Sowden and Power 1985; Bjornn and Reiser 1991). Successful incubation, therefore, depends not on the structure of the bed framework but rather on the amount of fine material comprising the bed matrix.

Water movement from the riverbed into the river also depends on the potential energy gradient between the bed and the river, which is termed the vertical hydraulic gradient (VHG). Adult salmon preferentially select sites of upwelling or downwelling to deposit their eggs (Healy 1991; Geist et al. 2000, 2002). Fall Chinook salmon in the Hanford Reach spawned in areas where river water upwelled from the riverbed into redd sites (Geist 2000). However, at Ives Island in the Columbia River where chum salmon (*Oncorhynchus keta*) and Chinook salmon (*O. tshawytscha*) spawn in sympatry, chum salmon selected upwelling sites to spawn while Chinook spawned in downwelling sites (Geist et al. 2002). The geomorphic features of the riverbed often dictate the amount of upwelling or downwelling that will occur in salmonid spawning areas (Geist and Dauble 1998; Dauble and Geist 2000). Thus, the riverbed features that control hydraulic gradients and sediment permeability will influence redd site selection and incubation success of salmonids.

The ability of fry to emerge from riverbed gravels following hatching is dependent upon open pore spaces from which they can exit the egg pocket. It is conceivable that eggs can incubate successfully but for fry to become trapped as they attempt to emerge. Laboratory studies have demonstrated that sediment in the size range between 1 and 10 mm block emergence (Bjornn 1969; Phillips et al. 1975; Harshbarger

and Porter 1982) while those responsible for reducing permeability are less than 1 mm (McNeil and Ahnell 1964; Cederholm and Salo 1979; Tagart 1984). Thus there is an additional limiting sediment size class that is too coarse to have an effect on incubation but that prevents emergence.

Many laboratory and field studies have been conducted to relate the amount of fine sediment to incubation and emergence success (see comprehensive review by Chapman 1988). Many of these studies attempted to relate single indices to incubation and emergence success. For example, the geometric mean (d_g) provides a measure of central tendency while emphasizing the extremes of the distribution rather than the median (Kondolf 2000). The geometric sorting coefficient (s_g) is an indication of the sorting (or grouping) of similarly sized particles (Kondolf 2000). The Fredle index (F_i), proposed by Lotspeich and Everest (1981), is a concise measure of d_g and s_g , is common in the literature (Kondolf 2000), and is thus a useful tool for comparing results to the literature. While it would be convenient to have a single index that relates substrate quality to impacts to salmon populations, it is unrealistic, given the heterogeneity in sediment grainsize distributions and the fact that gravel requirements of salmonids differ depending on the species and the life stage (e.g., during redd construction, incubation, and emergence). A more realistic approach would use a multivariate approach considering various grainsize indices, measures of permeability, percentages of fines, and direct measures of dissolved oxygen (Chapman 1988). Further, this approach should consider how these variables compare to other locations where fall Chinook salmon spawned and consider how conditions change over time (e.g., due to sediment intrusion), armoring, or the extent of upwelling and downwelling (Kondolf 2000).

There has not been a comprehensive investigation of substrate quality in the Hanford Reach of the Columbia River. Geist et al. (2000) found that the grainsize of surface sediments did not influence redd site selection of adult fall Chinook salmon, but they did not look specifically at the size of the particles relative to female size. Geist (2000) documented that river water upwelled into redd sites near Locke Island, a heavily used spawning area as compared to a non-spawning site located downstream at Wooded Island. However, only one spawning and one non-spawning site were selected, and no information was collected on the grainsize distribution.

This study was initiated to investigate the spawning and incubation habitat of fall Chinook salmon in the Hanford Reach. The goal of the study was to determine the relationship between fall Chinook salmon redd site selection and the physical habitat at different spatial scales. Previous sections of this report documented that fall Chinook salmon spawning locations are clustered (Section 3) and are dependent upon the water velocity and water depth (Section 4). However, some areas are predicted to be spawning habitat based on their velocity and depth that are not actually used (Section 4). Therefore, the objective of this section and of Section 6 is to explore whether riverbed grainsize and permeability (this section) and the interaction of groundwater and surface water (Section 6) may offer alternative explanations to the standard spawning habitat features. In this section, we are specifically interested in 1) documenting differences in riverbed particle size among the individual *sites* versus spawning and non-spawning *reaches*; 2) measuring differences in riverbed permeability among the *sites*; 3) relating riverbed grainsize to permeability; and 4) comparing substrate quality at sites across all life stages of fall Chinook salmon.

5.1 Methods

5.1.1 Sediment Grainsize

5.1.1.1 Data Collection

Sediment grainsize was characterized within the riverbed as well as on the surface of the riverbed. To characterize riverbed particle size distribution, we used a modified tri-tube freeze core technique (Everest et al. 1980; Rood and Church 1994). Freeze coring has proven to be a useful technique to describe both the framework and matrix. A series of hollow stainless steel tubes was driven into the riverbed to a depth of approximately 50 cm below the bed surface. Each tube was 152 cm in length with a 2.4-cm inside diameter and a 3.3-cm outside diameter. The tubes were separated a distance of 15 cm on center by two triangular 0.6-cm-thick aluminum plates. Approximately 15 to 20 L of liquid nitrogen were applied into the tubes over a 30- to 45-minute time span for each core, freezing the sediment around the tubes. The frozen sample mass was extracted from the bed using a 1-ton tripod and chain hoist and thawed over an aluminum collection box (Figure 5.1). The frozen sample was then photographed, placed in a vinyl bag, and transported to the U.S. Geological Survey Cascades Volcano Research Laboratory for drying. The dried samples were sieved into $\frac{1}{2}$ -phi size classes from 256 mm (-8 phi) to 0.0625 mm (4 phi). For each

a) Pouring Liquid Nitrogen

b) Extracted Core

Figure 5.1. Freeze Core Process

sample, the weight of the substrate in each size class was determined, yielding a percent-by-weight value for each size class. In 2001 only, the percent finer for each size class from 0.032 mm (5 phi) to 0.001 mm (10 phi) were determined using a Sedigraph Model 5001 (Weaver and Grobler 1981; Singer et al. 1988). For laboratory sample handling and quality assurance/quality control, researchers followed the guidelines of Guy (1969).

A total of 43 freeze cores were collected in 2000 and 2001 (Table 5.1; see Appendix A for locations of individual cores within each study site). In Section 2 we described how the study area was delineated into 13 geomorphic units based on channel morphology and channel hydraulics; freeze core samples were taken from 9 of the 13 geomorphic units (Figure 5.2). We attempted to collect at least four freeze cores and at least 100 kg of sediment from each site to ensure representative data (Shirazi et al. 1981; Church et al. 1987).

Substrate on the riverbed surface was characterized by underwater videography in 2000 and 2002. This modified pebble count method was described in detail in Section 4 and is briefly reviewed again here. The pebble count method typically dictates sampling 100 stones selected

Table 5.1. Freeze Core Sediment Samples Collected from Nine Study Sites in the Hanford Reach in 2000 and 2001

Site	Number of Cores	Dry Weight (kg) of All Cores Combined	Greatest Distance Between Cores
54	5	181.4	18 m
59	4	133.5	9 m
61	6	258.9	10 m (4 cores) 80 m (1 core) 225 m (1 core)
64	5	229.1	6 m
81	4	97.6	17 m
90	5	159.0	9 m (3 cores) 70 m (2 cores)
94	4	142.2	65 m
96	5	186.4	40 m
101	5	178.0	20 m (4 cores) 90 m (1 core)

without bias from specific geomorphic features (Wolman 1954; Church et al. 1987; Kondolf 2000). Pebble counts were collected from all 13 geomorphic units but only the surface substrate in the same 9 sites as the subsurface cores is presented in this section (Figure 5.2). In areas where the site was split into a right bank and left bank (59, 61, and 64), 100 images were taken for the whole unit, not each bank. The right bank data was used in the spawning analysis in this section. The number of images collected on the right bank for 59, 61, and 64 ranged from 38 to 62. Within each unit, samples were collected in a spatially distributed manner so that a fairly uniform coverage of each unit was attained. This sampling scheme was appropriate for obtaining overall sediment size characteristics of a distinct geomorphic unit or for an entire reach (Wolcott and Church 1991; Crowder and Diplas 1997).

5.1.1.2 Data Analysis

For subsurface samples, the weights for each size class were used to evaluate the grainsize distribution. For surface grainsize samples, the substrate images were treated as pebble counts, in that the processing software automatically selected the grain at the center of each image. The grain was then measured by the software, resulting in an estimate of the length of the apparent long (*a*) and intermediate (*b*) axes for each grain. The grainsize estimates derived from all substrate sampling were used to develop cumulative grainsize distributions. Although percent finer than curves provide a more complete representation of the overall grainsize distribution, they are difficult to read when more than four to five distributions are represented. In contrast, a box-and-whisker plot enables the reader to see specific percentiles of the grainsize distribution (i.e., 10th, 25th, 50th, 75th, and 90th percentiles) for the individual cores and is more easily read when there are many cores. For the site-to-site comparison where there were fewer curves, both percent finer than curves and box-and-whisker plots were used. In addition to total grainsize distributions, several indices of substrate composition provide means of evaluating the quality of spawning gravels.

Figure 5.2. Study Areas Where Surface and Subsurface Samples Were Collected

The geometric mean (d_g) provides a measure of central tendency while emphasizing the extremes of the distribution rather than the median (Kondolf 2000). The geometric mean (d_g) is determined by

$$d_g = (d_{84} \times d_{16})^{0.5} \quad (5.1)$$

The symbols d_{84} and d_{16} represent the grainsize (mm) at which 84% and 16% of the sampled grains were finer than.

The geometric sorting coefficient (s_g) is an indication of the sorting (or grouping) of similarly sized particles (Kondolf 2000). When particles of all sizes are well mixed together (also known as dispersion), s_g values increase. Conversely, when particles of the same size are grouped together (i.e., a deposit is well-sorted by particle size classes), s_g values decrease. The geometric sorting coefficient is determined by

$$s_g = (d_{84} / d_{16})^{0.5} \quad (5.2)$$

The Fredle index (F_i) combines central tendency (d_g) with a different sorting coefficient (s_t). The Fredle sorting coefficient (s_t) is similar to s_g , except uses d_{25} and d_{75} instead of d_{16} and d_{84} . The Fredle index is a concise measure of d_g and s_g , is common in the literature (Kondolf 2000), and is thus a useful tool for comparing results to the literature. The Fredle index is determined by

$$F_i = d_g / s_t = [(d_{84} \times d_{16})^{0.5}] / [(d_{75}/d_{25})^{0.5}]. \quad (5.3)$$

5.1.2 Riverbed Permeability

5.1.2.1 Data Collection

Hydraulic conductivity (a measure of the ability of water to flow through sediment) was determined using a modified slug test approach on piezometers emplaced into the riverbed in 1999–2001 and 2003 at locations throughout the Reach (Table 5.2). Appendix A should be consulted to see locations of piezometer placement within individual study sites. In 1999, we installed 24 piezometers on nine transects throughout the study area. Piezometers were installed on both sides of the river channel and on both sides of mid-channel islands. Slug test data were subsequently collected in 2000 from 19 of these piezometers (Table 5.2). During 2000 and 2001, we installed and conducted slug tests in 87 additional piezometers at locations within each of the nine study sites (Table 5.2). These piezometers were installed in clusters of three, with approximately three clusters per site. In 2003, we conducted slug tests on nine piezometers that were used to house level loggers (see Section 6). Individual piezometers were labeled according to their location. For example, piezometers installed in 1999 (and sampled in 2000) were labeled as being on either the right or left bank of the river channel (RB or LB) or right or left side of an island (IR or IL). Piezometers installed in 2000 and 2001 were differentiated based on which cluster they were in. For example, the three piezometers installed at site 54 within the first cluster were designed as F1,2,3 (Table 5.2).

Piezometer screens were constructed of either slotted stainless steel Johnson screen (0.038-cm slot size) or brass wire mesh Mass-Midwest screen with an equivalent size opening (Figure 5.3). Both had a 31.0-cm screened interval and a 3.2-cm inside diameter. The screen was welded on one end to a 12-cm

Table 5.2. Piezometers Where Slug Tests Were Conducted in 2000, 2001, and 2003

Site	Individual Piezometers (2000)	Piezometer Clusters (2000 and 2001)	Logger Sites (2003)
54	54RB 54LB	54RBF1,2,3 54RBF4,5,6 54RBF7,8,9	54RB
59	59RB 59IR 59IL	59IRF1,2,3 59IRF4,5,6 59IRF7,8,9 59IRF10,11,12	59IR
61	61IR 61IL 61IL_2	61IRF1,2,3 61IRF7,8,9 61IRF10,11,12 61IRF13,14,15	61IR
64	64RB 64IR 64ILB	64RBF1,2,3 64RBF4,5,6 64RBF7,8,9	64RB
81	81IL	81RBF1,2,3 81RBF4,5,6 81ILF7,8,9	81RB
90	90RB 90LB	90LBF1,2,3 90LBF4,5,6 90LBF7,8,9	90LB
94	94RB ^(a)	94RBF1,2,3 94RBF4,5,6 94RBF7,8,9	94BAR
96	96RB 96LB	96RBF4,5,6 ^(b) 96RBF8,9	96_1 ^(c)
101	101LB 101RB	101LBF1,2,3 101LBF4,5,6 101LBF7,8,9	101LB
(a) 94LB was slug-tested but later determined to be broken so data were not used. (b) 96RBF1,2,3 and 7 were determined to be either broken or not sealed properly as values were equivalent to PVC screen placed in the water. (c) 96_2 tested but not used because it had not been in place a similar amount of time as other eight piezometers tested in 2003.			

drive point and on the other end to a variable-length section of galvanized steel pipe (3.2 cm inner diameter) threaded on top. Piezometers were installed by inserting a solid steel drive-rod into the piezometer and either pounding the piezometer with the rod or with a post-pounder (Geist et al. 1998 [Figure 5.4]). The depth of the screen below the riverbed averaged approximately 63 cm in the piezometers installed in 1999 (range 37.5–71.5 cm) as compared to approximately 31 cm in all other years (range 22–37 cm).

Hydraulic conductivity was measured using a slug test method that was developed for this project (Arntzen et al. 2006). To perform the test, an airtight pressure-regulating wellhead assembly was

a) Johnson Screen b) Mass-Midwest

Figure 5.3. Piezometer Screens

Figure 5.4. Piezometer Installation

threaded to the top of the piezometer (Figure 5.5). The assembly consisted of a 5-cm ball valve coupled to a 20-cm section of schedule 40 polyvinyl chloride pipe containing a small valve stem for pressurizing. A pressure transducer (Instruments NW Model 9800) was lowered into the piezometer to measure changes in hydraulic head during the test. A modified rubber stopper was used to seal the transducer cable entry into the well assembly. The system was pressurized with a portable battery-powered air compressor (Black & Decker VersaPak cordless inflator), causing the water level in the piezometer to be depressed downward. After the water level in the well was depressed, the air compressor was shut off and the ball valve simultaneously opened, marking the beginning of the slug test. The change in water level was measured by the transducer and recorded by an electronic data logger (Campbell cr10x). When the pressure was released, the data logger recorded the pressure response (rising water level) with respect to time. When possible, the slug tests were repeated three to five times in each piezometer to ensure precision (Butler et al. 1996).

5.1.2.2 Data Analysis

Methods for estimating hydraulic conductivity from slug test data have been summarized by Butler (1998). For the case in which overdamped slug test response data are collected from a partially penetrating well in an unconfined aquifer (e.g., riverbed conditions), the Bouwer and Rice method is a reasonable data analysis technique (Bouwer and Rice 1976; Bouwer 1989; Butler 1998; Weight and Wittman 1999). The method involves plotting the natural logarithm of the head response against time. The slope of these data together with data regarding piezometer geometry, were used to solve for hydraulic conductivity. Hydraulic conductivity was determined as

$$K = \frac{r_c^2 \ln(R_e/R)}{2L_e} \frac{1}{t} \ln\left(\frac{H_0}{H_t}\right) \quad (5.4)$$

where K = hydraulic conductivity (cm/s^{-1})
 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_0 = drawdown at time $t = 0$
- H_t = drawdown at time $t = t$
- t = time from H_0 .

Slug tests were summarized using the geometric mean hydraulic conductivity (Butler 1998). Uncertainty was evaluated using the standard error associated with linear regressions using the Bouwer and Rice method and the standard deviation of replicate samples.

Figure 5.5. Slug Test Process

The relationship between riverbed grainsize distribution and permeability was assessed by comparing riverbed hydraulic conductivity to summary statistics of the grainsize distribution. We began by comparing the percentage of the entire grainsize distribution retained on each sieve to the geometric mean hydraulic conductivity value for each site. Acknowledging that heterogeneity existed within the grainsize distribution at each of the sites, we explored whether there were better combinations of sediment sample locations and permeability sample locations that should be used to assess which grainsize controlled the permeability. Many of the individual piezometers were not installed or sampled near the freeze core sites. For example, riverbed grainsize data was only characterized on the right bank sides of the river channels at study sites 59, 61, and 64 whereas slug tests were conducted in piezometers on both sides of the islands. Therefore, we decided to use the geometric mean hydraulic conductivity value from either the piezometer cluster or the level logger piezometers (see Table 5.2). This decision was based on how close the individual cores were to the sites where the slug tests were conducted. The level logger piezometers were closest to the freeze cores at sites 54, 64, 81, 90, 94, and 101 while the cluster piezometers were closest to the freeze cores at sites 59 (cluster 1) and 61 (cluster 1). At site 96, there were two cores very close to the level logger sites and three cores within 20 m of the cluster sites. Because the two cores were very close to the level logger sites, the level logger piezometer was selected for comparison of permeability to sediment.

We used four spawning sites and four non-spawning sites for the comparison of grainsize data to permeability data. The designation of spawning and non-spawning was based on whether fall Chinook

salmon spawned within the study site. This designation was described in Section 2. For purposes of this analysis, we excluded site 54 because while it was classified as a non-spawning site, it was not contiguous to the other non-spawning sites.

To determine the particle sizes that were controlling the permeability, the geometric mean permeability values were regressed against the percentage of silts, fine sand, coarse sand, and sand index. Particle size groupings of cobble, gravel, sand, and silt were based on the Wentworth particle scale (Table 5.3). We calculated the sand index (SI) developed by Peterson and Metcalf (1981), which is

$$SI = (S_c/16 + S_f/8) \quad (5.5)$$

where S_c is the percentage of coarse sand (0.5–2 mm) and S_f is the percentage of fine to medium sand (0.06–0.5 mm).

Table 5.3. Wentworth Particle Size Classification Used to Group Sediment Classes

Wentworth Category	Range of Particle Sizes
Large cobble	>128 to ≤256 mm
Cobble	>64 to ≤128 mm
Coarse gravel	>16 to ≤64 mm
Fine gravel	>2 to ≤16 mm
Coarse sand	>0.5 to ≤2 mm
Fine sand	>0.062 to ≤0.5 mm
Silt	≤0.062 mm

5.1.3 Relationship Between Grainsize Distribution and Survival to Emergence

Subsurface grainsize data collected from the Hanford Reach were compared to previous studies relating grainsize data to survival to emergence. Tappel and Bjornn (1983) used the percentage of substrate smaller than 0.85 and 9.50 mm to derive the following relationship predicting survival to emergence of Chinook and steelhead:

$$\text{Percent survival} = 93.4 - 0.171S_{9.5}S_{0.85} + 3.87S_{0.85} \quad (5.6)$$

where $S_{9.5}$ is the percentage of substrate finer than 9.5 mm and $S_{0.85}$ is the percentage of substrate fine than 0.85 mm. The authors found that these two size classes best approximated the size composition of stream bed material smaller than 25.4 mm. Approximately 90 to 93% of the variability in salmonid embryo survival was correlated with the composition of substrate of these size classes. The data from the individual cores collected at the nine sites in the Hanford Reach were used to compare to Tappel and Bjornn's data. Our sieve sizes did not correspond to Tappel and Bjornn's so we used 1 mm as a surrogate for the 0.85 mm and 11.3 mm as a surrogate for 9.5 mm. Using the regression equation in Tappel and Bjornn (1983) for Chinook salmon, the survival to emergence was predicted for the nine sites. Consistent with the Tappel and Bjornn paper, if the survival predictions were more than 100% or less than 0%, they were rounded down to 100% and up to 0%, respectively.

Similar to Tappel and Bjornn (1983), Bennett et al. (2003) conducted an analysis on Snake River fall Chinook salmon but used green eggs instead of eyed eggs. They determined that the percentages finer than 6.4 mm and 0.85 mm were more representative of Snake River substrate than the 0.85 mm and 9.5 mm values used by Tappel and Bjornn. Our sieve sizes did not correspond to these values so we used 1 mm as a surrogate for the 0.85 mm and 8 mm as a surrogate for 6.4 mm. Similar to Tappel and Bjornn's results, Bennett et al. (2003) reported that the best second-order equation for relating substrate composition to emergence success was

$$\text{emergence success} = 46.643 + 1.194 S_{0.85} - 0.111 S_{6.4} S_{0.85} \quad (R = 0.97, P < 0.001). \quad (5.7)$$

where $S_{6.4}$ is the percentage of substrate finer than 6.4 mm. We used Bennett's equation and predicted emergence success at each of the nine sites in the Hanford Reach.

5.2 Results

5.2.1 Sediment Grainsize

The median particle size (D_{50}) values of the subsurface samples based on individual cores were most variable at sites 54, 64, 96, and 101 and least variable at sites 81 and 90 (Figure 5.6). At site 64 there was significant variability within individual cores. For example, within one core at site 64 (core 2) the upper 36 cm of the core was composed of gravel within a sand matrix while below this depth it was composed of gravel within a predominantly clay matrix (Figure 5.7). In the grainsize distribution for core 2, the influence of these finer-sized particles is evident in that 25% of the material was less than 0.325 mm (Figure 5.6). Comprehensive subsurface grainsize distribution results are included in Appendix B.

Figure 5.6. Grainsize Distribution of 43 Individual Freeze Cores Collected at Nine Sites in the Hanford Reach of the Columbia River. Cores are labeled as to the site and their number. The middle of the box is the median, with the lower and upper limits of the box the 25th and 75th percentiles, respectively. The 10th and 90th percentiles are shown as the lower and upper ends of the whiskers.

Figure 5.7. Freeze Core 64-2 Sampled in 2001 from the Riverbed at Site 64

Individual freeze cores were combined and summarized for each site (Figures 5.8 and 5.9). The median particle size of the three lower-reach sites (94, 96, and 101) ranged from 17.4 to 25.5 mm (Table 5.4). The median particle size for the other sites ranged from 27.5 to 57.7 mm. The geometric means from the nine sites ranged from 4.5 mm at site 94 to 30 mm at site 54 (Table 5.4). The sorting coefficients ranged from 3.6 at site 90 to 18.5 at site 64. From a relative standpoint, it would appear that the sediment deposits at sites 54, 61, and 90 would be very well sorted (i.e., homogenous) as compared to sites 64 and 94 that were poorly sorted (i.e., heterogeneous). Sites 59, 81, 96, and 101 would appear to be intermediate between these extremes. Fredle index values ranged from 0.4 at site 94 to 12.3 at site 54.

The combined percentage silt (<0.062 mm) and fine sand (0.062 to 0.5 mm) within the subsurface samples of the riverbed was highest at sites 64 and 94 and was lowest at site 54 (Figure 5.10). In general, the percentage of the entire particle size distribution that was silt (<0.062 mm) was notable at sites 64 (2.3%), 94 (2.1%), 96 (1.5%), and 101 (3.7%) (Table 5.5). At all other sites, the percentage silt was less than 0.5%. Three of the four sites with relatively high percentages of silt (94, 96, and 101) are downstream of the spawning reach, but site 64 is a significant spawning site.

Underwater video data of the riverbed surface samples were summarized for each of the 9 study sites (Figures 5.11 and 5.12). The median size of surface particles at the three lower-reach sites (94, 96, and 101) ranged from 90.5 to 99.4 mm (Table 5.6). The median size of surface particles for the other sites ranged from 45.0 to 105.7 mm. The geometric means ranged from 45.5 mm at site 64 to 149.7 mm at site 101 (Table 5.6). The sediments on the riverbed surface were generally well sorted, which is to be expected given that they were armored and that there was a lack of fine sediments there. The sorting coefficients ranged from 1.7 at site 81 to 6.0 at site 101. The sorting coefficients at each site were less than or equal to 3.0 except for site 101, where s_g equaled 6.0. The difference in sorting at site 101 is likely due to the increased number of boulders on the riverbed there. Fredle index values ranged from 26.3 at site 64 to 56.8 at site 59. Comprehensive data on riverbed surface grainsize distribution are included in Appendix B.

Figure 5.8. Percent Finer Than Curve for Composite Subsurface Grainsize Data Collected by Freeze Coring at Nine Sites in the Hanford Reach in 2000 and 2001

Figure 5.9. Grainsize Distribution for Composite Subsurface Sediment Sampled in 2000 and 2001 at Nine Sites in the Hanford Reach. The middle of the box is the median, with the lower and upper limits of the box the 25th and 75th percentiles, respectively. The 10th and 90th percentiles are shown as the lower and upper ends of the whiskers.

Table 5.4. Summary Statistics for Composite Subsurface Grainsize Data Collected Using Freeze Cores at Nine Sites in the Hanford Reach in 2000 and 2001

Site	N	D ₅₀ (mm)	D ₈₄ (mm)	Geometric Mean (mm)	Sorting Coefficient (s _g)	Fredle Index (f _i)
54	5	57.7	113.8	30.0	3.8	12.3
59	4	32.0	62.3	6.2	10.0	3.1
61	6	50.6	104.7	20.4	5.1	6.6
64	5	40.8	125.4	6.8	18.5	1.5
81	4	27.5	56.9	5.2	11.0	2.2
90	5	35.5	64.4	17.9	3.6	10.0
94	4	17.4	75.6	4.5	16.8	0.4
96	5	22.3	93.1	8.7	10.7	2.5
101	5	25.5	132.5	12.1	10.9	3.0

Figure 5.10. Percentage of Total Subsurface Sample Weight Categorized into Sediment Types Using the Wentworth Scale

Table 5.5. Percentage of Coarse Sand, Fine Sand, and Silt, and Sand Index for Subsurface Sediment Sampled from Nine Sites in the Hanford Reach in 2000 and 2001

Site	Silt	Coarse Sand	Fine Sand	Sand Index
54	0.36	2.19	7.90	1.12
59	0.57	3.25	14.55	2.02
61	0.20	2.68	9.60	1.37
64	2.34	3.96	17.18	2.39
81	0.53	1.65	16.20	2.13
90	0.28	3.14	11.29	1.61
94	2.11	3.79	25.81	3.46
96	1.54	4.87	11.60	1.76
101	3.69	4.79	10.04	1.55

Figure 5.11. Percent Finer Than Curves for Surface Grainsize Distributions

Figure 5.12. Grainsize Distribution for Surface Sediment Sampled in 2000 and 2002 Using Videography from Nine Study Sites in the Hanford Reach. The middle of the box is the median, with the lower and upper limits of the box the 25th and 75th percentiles, respectively. The 10th and 90th percentiles are shown as the lower and upper ends of the whiskers.

Table 5.6. Summary Statistics for Riverbed Surface Grainsize Data Collected Using Underwater Video in the Hanford Reach in 2000 and 2002

Site	D ₅₀ (mm)	D ₈₄ (mm)	Geometric Mean (mm)	Sorting Coefficient (s _g)	Fredle Index (f _i)
54	69.8	178.9	67.8	2.6	34.4
59	105.7	175.3	84.5	2.1	56.8
61	101.5	135.1	75.4	1.8	47.3
64	45.0	91.7	45.5	2.0	26.3
81	63.8	101.8	58.4	1.7	38.0
90	70.5	129.1	57.2	2.3	30.5
94	93.0	155.2	69.1	2.2	46.7
96	90.5	150.6	74.8	2.0	46.7
101	99.35	895.8	149.7	6.0	37.2

The spawning reach sites (59, 61, 64, and 81) had 16.6% sediment finer than 1.0 mm in the subsurface samples compared to 18.1% for the non-spawning reach sites (90, 94, 96, and 101; Figure 5.13). The D₅₀ of the subsurface sediment at sites in the spawning reach was 36.6 mm compared to the D₅₀ of sites in the non-spawning reach of 27.2 mm (Table 5.7). The percentage of silt in the subsurface sample of the riverbed in the non-spawning reach was about twice as high as in the spawning areas (Table 5.8 and Figure 5.14), although the significance of the difference was marginal (chi-square 2.92, df = 1, $P = 0.087$). The percentage of coarse sand within subsurface samples was significantly higher in the non-spawning reach (chi-square 5.8, df = 1, $P = 0.016$). The percentage of fine sand was not different.

Figure 5.13. Percent Finer Than Curves for Subsurface Samples of the Riverbed at Non-Spawning (closed circles; 90, 94, 96, and 101) and Spawning (open circles; 59, 61, 64, and 81) Sites

Table 5.7. Sorting Indices for Subsurface Samples of the Riverbed at Non-Spawning (90, 94, 96, and 101) and Spawning Sites (59, 61, 64 and 81)

Site	D ₅₀ (mm)	D ₈₄ (mm)	Geometric Mean (mm)	Sorting Coefficient (s _g)	Fredle Index (f _i)
Non-Spawning	27.2	91.2	6.8	13.5	1.9
Spawning	36.6	96.4	7.9	12.2	2.7

Table 5.8. Percentage of the Entire Sample That Are Silts, Fine Sand, and Coarse Sand Within Subsurface Samples of the Riverbed at Spawning and Non-Spawning Sites

	Spawning (percent)	Non-Spawning (percent)
Silts	0.8	2.1
Fine sand	13.1	14.9
Coarse sand	2.8	4.6
Sum	17.7	21.6

Figure 5.14. Percentage of the Total Subsurface Sample of the Riverbed Collected from Spawning and Non-Spawning Sites in the Hanford Reach. Error bars show 95% confidence limit of the mean.

The combined percentage of boulders (256 to 724 mm) within the surface sediments of the riverbed was higher at non-spawning locations (i.e., 54, 94, 96, and 101). There were also higher percentages of large cobble (128 to 180 mm) at non-spawning sites relative to spawning sites, with the exceptions of sites 59 and 61, which also had high percentages of large cobble. Cobble-sized particles (64 to 90 mm) were fairly evenly distributed between spawning and non-spawning sites. Coarse gravel (16 to 45 mm) was more abundant at spawning sites, with the exceptions of sites 59 and 61 where they were similar to non-spawning locations. Surface gravel smaller than 16 mm was scarce and comprised only 7.1% of the sample at the site where it was most abundant (Figure 5.15).

The combined mean size of surface particles was 77.4 mm for spawning sites and 86.3 mm at non-spawning sites (Figure 5.16, Table 5.9). Geometric mean particle size was 65.5 mm at spawning sites and 70.2 mm at non-spawning sites. Sorting coefficients were similar (2.0 for spawning sites, 2.5 for non-spawning sites). The Fredle index also was similar (39.8 for spawning, 38.2 for non-spawning).

Figure 5.15. Percentage of Total Surface Sample of the Riverbed Classified Using the Wentworth Scale

Figure 5.16. Percent Finer Than Curves for Surface Grainsize Distributions in Non-Spawning (closed circles; 90, 94, 96, and 101) and Spawning (open circles; 59, 61, 64 and 81) Sites

Table 5.9. Surface Sediment Statistics for Non-Spawning (90, 94, 96, and 101) and Spawning (59, 61, 64, and 81) Sites

Site	D ₅₀ (mm)	D ₈₄ (mm)	Geometric Mean (mm)	Sorting Coefficient (s _g)	Fredle Index (f _i)
Non-Spawning	86.3	173.4	70.2	2.5	38.2
Spawning	77.4	130.7	65.5	2.0	39.8

5.2.2 Riverbed Permeability

A total of 111 piezometers were tested for hydraulic conductivity (K), vertical hydraulic gradient (VHG), and specific discharge (q) during 2000, 2001, and 2003 at individual piezometer, clustered piezometer, and level logger sites (Table 5.10). Data for each piezometer tested are listed in Appendix C.

Table 5.10. Permeability (K), Vertical Hydraulic Gradient (VHG), and Specific Discharge (q) Data for All Sites

Site	N	Geometric Mean K (SE) (cm/s)	Mean Log (K) (SE)	Mean VHG (SE)	Mean q (SE) (cm/s)
54	12	0.0049 (0.0025)	-2.3062 (0.0997)	0.0530 (0.0394)	0.0003 (0.0002)
59	16	0.0061 (0.0008)	-2.2156 (0.0473)	0.0479 (0.0097)	0.0003 (0.0001)
61	16	0.0085 (0.0004)	-2.0701 (0.0206)	0.0410 (0.0067)	0.0004 (0.0001)
64	13	0.0081 (0.0014)	-2.0921 (0.0747)	0.0639 (0.0222)	0.0004 (0.0001)
81	11	0.0112 (0.0014)	-1.9521 (0.0633)	0.0683 (0.0130)	0.0009 (0.0003)
90	12	0.0083 (0.0004)	-2.0801 (0.0219)	0.0124 (0.0102)	0.0001 (0.0001)
94	11	0.0039 (0.0021)	-2.4069 (0.1633)	0.0058 (0.0134)	0.0002 (0.0001)
96	8	0.0006 (0.0004)	-3.2061 (0.1622)	0.1390 (0.0347)	0.0002 (0.0001)
101	12	0.0013 (0.0009)	-2.9005 (0.1432)	0.0280 (0.0152)	0.0001 (0.0001)

The highest geometric mean hydraulic conductivity was at site 81 (1.11×10^{-2} cm/s) and lowest was at site 96 (6.22×10^{-4} cm/s [Figure 5.17]). The largest range in hydraulic conductivity within a site was at site 54 (range = 0.03 cm/s), whereas site 96 had very little variability in the data (range = 0.003 cm/s). The hydraulic conductivity values were log transformed because of the large range in the untransformed data (2.01×10^{-4} cm/s to 3.20×10^{-2} cm/s). Median log-transformed hydraulic conductivity values were lowest at sites 54, 94, 96, and 101 (Table 5.10, and Figure 5.17). Sites 94, 96, and 101 were notably lower in average values (-2.4, -3.2, and -2.9) than the other six sites (-1.9 to -2.3).

Figure 5.17. Hydraulic Conductivity, Log Hydraulic Conductivity, Vertical Hydraulic Gradient, and Specific Discharge Measured at Nine Study Sites in 2000, 2001, and 2003. The line through the middle of the box represents the median value, the lower and upper edge of the box the 25th and 75th percentiles of the data, and the lower and upper whiskers the 10th and 90th percentiles. Dots indicate outliers. Box and whiskers based on geometric mean K values for multiple readings taken in individual piezometers.

A one-way-analysis of geometric mean hydraulic conductivity by site (Tukey-Kramer procedure) showed that the hydraulic conductivity at sites 61, 64, and 81 was significantly ($\alpha = 0.05$) higher than the conductivity at sites 96 and 101, with the greatest difference between site 81 and site 96 (range = 0.011 cm/s). The hydraulic conductivity at site 90 was also significantly higher than at site 96 by 0.007 cm/s. A one-way-analysis of log hydraulic conductivity by site (Tukey-Kramer procedure) showed that the log hydraulic conductivity values at sites 96 and 101 were significantly lower than log hydraulic conductivity values at all other sites. In addition, log conductivity values at site 94 were significantly less than log hydraulic conductivity values at site 81.

The highest average vertical hydraulic gradient (VHG) was at site 96 (0.139) versus the lowest average VHG at site 94 (0.006) (Table 5.10). A one-way-analysis of the average VHG by site (Tukey-Kramer procedure) showed that the average VHG at site 96 was significantly higher than at sites 61, 90, 94, and 101. The median VHG was highest at site 96 (0.127) and lowest at site 90 (0.007), although the median VHG at sites 54, 94, and 101 were nearly as low (Figure 5.17). The VHG ranged the most at site 54 (−0.259 to 0.270) and the least at sites 61 (0.0 to 0.107) and 90 (−0.033 to 0.076). Sites 59, 61, 81, and 96 were always upwelling (i.e., positive VHG) whereas sites 54, 64, 90, 94, and 101 exhibited some downwelling (i.e., negative VHG) (Figure 5.17).

The highest average specific discharge was found at site 81 (9.01×10^{-4} cm/s), which was significantly higher than all other sites except site 64 (based on the Tukey-Kramer procedure). The lowest average specific discharge was found at site 90 (8.91×10^{-5} cm/s), but there was no significant difference between site 90 and the other sites (except 81) based on a Tukey-Kramer one-way analysis of means. Median specific discharge values were highest at site 81 (5.22×10^{-4} cm/s) and lowest at site 96 (3.87×10^{-5} cm/s), although sites 90, 94, and 101 were also low (Figure 5.17) with median specific discharge values of 6.99×10^{-5} cm/s, 5.13×10^{-5} cm/s, and 3.91×10^{-5} cm/s, respectively. In contrast to these low values, the median specific discharges at the other sites (54, 59, 61, and 64) ranged from 1.03×10^{-4} cm/s to 2.94×10^{-4} cm/s. Specific discharge ranged the most at sites 54 (range = 2.4×10^{-3} cm/s) and 81 (range 2.68×10^{-3} cm/s), and the least at site 101 (5.06×10^{-4} cm/s).

The hydraulic conductivity, specific discharge, and vertical hydraulic gradient were all higher in the spawning reach when compared to the non-spawning reach (Figure 5.18). The geometric mean hydraulic conductivity value was significantly higher at spawning sites than at non-spawning sites (chi-square 18.06, df = 1, $P < 0.001$). The average geometric mean at non-spawning sites was 5.0×10^{-3} cm/s (95% confidence interval 3.7×10^{-3} to 6.4×10^{-3} cm/s). This compares to the average geometric mean hydraulic conductivity at spawning sites of 9.0×10^{-3} cm/s (95% confidence interval 7.8×10^{-3} to 1.0×10^{-2} cm/s). The vertical hydraulic gradient was significantly higher in spawning sites than non-spawning sites (chi-square 4.85, df = 1, $P = 0.028$). The average VHG at spawning sites was 0.053 (95% confidence interval 0.037 to 0.070) while at non-spawning sites the average VHG was 0.039 (95% confidence interval 0.020 to 0.058). Note that analysis of variance (ANOVA) did not show the VHG different between spawning and non-spawning sites. The specific discharge at spawning sites was significantly higher than non-spawning sites (chi-square 17.3, df = 1, $P < 0.0001$). Average specific discharge at spawning sites was 4.50×10^{-4} cm/s (95% confidence interval 3.4×10^{-4} to 5.6×10^{-4} cm/s) as compared to 1.37×10^{-4} cm/s (95% confidence interval 1.0×10^{-5} to 2.6×10^{-4} cm/s) at non-spawning sites.

Figure 5.18. Hydraulic Conductivity, Specific Discharge, and Vertical Hydraulic Gradient Measured at Spawning (59, 61, 64, and 81) and Non-Spawning (90, 94, 96, 101) Sites in the Hanford Reach in 2001, 2002, and 2003. Error bars represent 95% confidence interval of the mean.

The amount of coarse sand (0.5 mm to 2.0 mm) was significantly related to the hydraulic conductivity values for the eight sites that were used in the reach analysis ($r = -0.895$, $P = 0.001$). As the percentage of coarse sand increased, the permeability decreased (Figure 5.19). Increased concentration of silt also reduced permeability, but the relationship was not statistically significant ($r = -0.479$, $P = 0.192$). Fine sand and the sand index were not related to hydraulic conductivity.

Figure 5.19. Relationship Between the Percentage of the Total Sample Classified as Either Silt or Coarse Sand Versus the Hydraulic Conductivity of Riverbed Sediments at Eight Sites (excludes site 54)

5.2.3 Relationship Between Grainsize Distribution and Survival to Emergence

The percentage of subsurface substrate smaller than 1 mm (averaged for all the cores at a given site) ranged from 9.4% at site 54 to 30.2% at site 94 (Table 5.11). The percentage of subsurface substrate finer than 11.3 mm ranged from 19.5% at site 90 to 42.4% at site 94, while the percentage of subsurface substrate finer than 8 mm ranged from 16.1 to 38.8% (Table 5.11).

Table 5.11. Summary Percentages for “Fines” Based on Riverbed Grainsize Data Collected Using Freeze Cores at Nine Sites in the Hanford Reach in 2000 and 2001

Site	N	Percentage Finer Than			
		0.0625 mm	1 mm	8 mm	11.3 mm
54	5	0.36	9.4	16.1	20.0
59	4	0.57	17.0	20.8	24.1
61	6	0.20	11.0	22.3	27.4
64	5	2.34	22.1	26.9	29.0
81	4	0.53	17.8	23.8	27.7
90	5	0.28	13.5	17.4	19.5
94	4	2.11	30.2	38.8	42.4
96	5	1.54	15.4	27.2	32.0
101	5	3.69	15.7	30.1	35.0

Using the regression equation in Tappel and Bjornn (1983) and Bennett et al. (2003) for Chinook salmon, the survival to emergence was predicted for the nine sites (Table 5.12). Using the Tappel and Bjornn equations, the three lower sites had the lowest survival rates while site 90 (a non-spawning site) had the highest survival estimates at 100%. The survival estimates using Bennett's equation were lower than those made using the Tappel and Bjornn (1983) equation. Bennett's model suggested that survival to emergence at site 64 (a spawning site) would be only about 7%. Both authors' models showed that site 94 had the lowest survival to emergence.

Table 5.12. Survival to Emergence Estimates (percentages) for Chinook Salmon Using Various Relationships Reported in the Literature

Site	Tappel and Bjornn (1983)	Bennett et al. (2003)
54	97.6	41.1
59	89.3	27.7
61	84.4	32.6
64	69.5	6.9
81	77.9	20.8
90	100.0	36.6
94	0.0	0.0
96	68.8	18.5
101	60.2	13.0

5.3 Discussion

5.3.1 Sediment Grainsize

There was a large amount of variability in the sediment grainsize from both subsurface and surface samples. This variability was evident within sites and may have been due to the methodology used to collect the surface sample. Surface samples were collected using a modified pebble count method (Wolman 1954). At least 100 gravel samples were measured from common geomorphic units by the same observer, thereby reducing heterogeneity across channel features and also reducing observer to observer error. However, at three sites surface samples were subdivided according to either right bank or left bank. In order to compare more directly to subsurface freeze core results, only surface data from the bank nearest the subsurface samples were used. Consequently less than 100 samples (40 to 50) were considered on the right banks at three sites, and thus may have contributed to some of the variability in sediment characteristics on the riverbed surface.

Subsurface samples of sediment were collected using a modified tri-tube freeze core method (Lotspeich and Reid 1980; Rood and Church 1994). Variability in sediment sizes among or within sites may have been also due to the freeze core methodology. Freeze cores often result in a "ragged" edge and under-represent the smaller size fraction on the core's margins (Kondolf 2000). Further, the freeze core technique can under-represent the fine grain material from the upper part of the riverbed core when surface water temperatures are relatively warm (Hanrahan et al. 2001; Arntzen et al. 2001). Poor freezing occurs in this layer because the liquid nitrogen is not as efficient in freezing the sediment material that is in direct contact with the surface water. However, all but one of our sites was represented by sediment samples that exceeded 100 kg, which is well in excess of the minimum recommended sample weight for

sediment sizes we encountered (Church et al. 1987). Thus, it is not expected that issues with the pebble count method or the freeze core method can fully explain all the variability in the surface and subsurface sediment samples.

The variability in grainsize data was more likely due to the sediment heterogeneity that exists within individual study sites. The results at site 64 demonstrate how the heterogeneity in subsurface sediment samples can vary at spatial scales measured in meters both vertically and horizontally. Core 2 sampled at site 64 shows an interfacial layer at approximately 36 cm below the bed of the river (Figure 5.7). The upper 36 cm appears to be mostly gravel in a fine sand matrix, while the core section below 36 cm is composed largely of gravel in a clay matrix, likely from a fine-grained lens within the Ringold Formation. Adjacent cores at site 64 (all were within 6 m of one another) showed slightly different results. For example, core 3 was much smaller than core 2 and did not contain clay. This core was sampled in deeper, faster water than core 2, and the grainsize curve for this core depicts it as a coarser representation of the riverbed than core 2. In contrast, core 4, sampled in shallow water within 2 m of core 2 and 3, was composed mostly of the suspected Ringold Formation deposit but did not show a clear demarcation between sands and clays as evidenced in core 2. Thus, it appears that the riverbed sediment is heterogeneous at spatial scales of meters at some of the sites, which is not surprising for fluvial deposits like the Ringold Formation.

Site-to-site variability in sediment deposition also occurred and was evident in the results from the sediment core sampling. The geometric sorting coefficient is an indication of how well-mixed sediments are. Sorting values that are low would tend to show that the sample is homogeneous. In contrast, sediment samples that were a mix of a range of size classes would be represented by a fairly high sorting coefficient. The sorting coefficients for subsurface samples ranged from 3.6 at site 90 to 18.5 at site 64. From a relative standpoint, it would appear that the sediment deposits at sites 54, 61, and 90 would be very well sorted (i.e., all the same) as compared to sites 64 and 94 that were poorly sorted. Sites 59, 81, 96, and 101 would appear to be intermediate between these extremes.

When the substrate samples from individual cores were combined together to represent spawning and non-spawning sections, there were small but measurable differences in sediment grainsize. The percentage of silt and coarse sand within subsurface samples collected from study sites throughout the Reach was about twice as high in the non-spawning sites than it was in spawning sites. However, site 64, which is an important fall Chinook salmon spawning area, had some of the highest amounts of fine material. In contrast to the non-spawning sites, the silt and clay material at site 64 was confined to the sediment layer greater than 30 cm below the bed surface. This contrasts with the other three sites where the silt was distributed throughout the entire profile. Fall Chinook salmon that spawn at site 64 are likely successful because they deposit their eggs in this gravelly material near the riverbed surface and not in the clay-rich material located deeper in the riverbed.

The riverbed at all nine sites would be described as armored. This assessment is based on the fact that the sorting coefficients for the subsurface samples at all sites were above 3.0 and the D_{84}/D_{50} ratios were greater than or equal to 2 (Sutherland 1987; Froehlich 1995). The sorting coefficients ranged from 3.6 to 18.5 and were, on average, higher at non-spawning sites than at spawning sites (13.5 versus 12.2). The D_{84}/D_{50} ratios ranged from 1.8 to 5.2 with the ratio higher in non-spawning sites than spawning sites (3.9 versus 2.3). In addition, the relationship of the riverbed surface sediment to the subsurface (D_{50s}/D_{50B}) showed that spawning and non-spawning sites were both greater than 1, an indication of armoring (Lisle et al. 2000).

A full assessment of whether the bed is armored is beyond the scope of this study. A detailed analysis of shear stresses, hydraulic capacity, flow fluctuations, and ultimately bed transport would be needed to evaluate whether the riverbed in the Hanford Reach is armored (Andrews and Nankervis 1995; Emmett and Wolman 2001). However, the Hanford Reach is subject to several anthropogenic impacts that would tend to increase its susceptibility to armoring. Smaller and more mobile sediment grains are captured by upstream hydroelectric dams. The hydraulic capacity is reduced owing to extensive flood control measures that are now in place since the development of the Columbia River hydropower system. Flow fluctuations from load following operations create a pumping mechanism whereby finer grained material is entrained into the bed matrix and winnowed from the bed surface. All these factors have probably increased the incidence of armoring in the Hanford Reach.

Despite the armored nature of the bed, it is obvious that fall Chinook salmon can still use the Reach for spawning, and the empirical results of the analysis of riverbed surface grain size shows that the particles are well within the range potentially moveable by adult fall Chinook salmon. Fall Chinook salmon in the Hanford Reach have fork lengths ranging from 62 cm to 120.7 cm, with an average fork length of 98.2 cm (D. R. Geist, PNNL, unpublished data). Using the relationship between median size of spawning gravel and fish length ($D_{50} \text{ [mm]} = 0.4 \text{ fork length [cm]}$) from Kondolf and Wolman (1993), Hanford Reach fall Chinook should be able to spawn in gravels with median particle sizes ranging from 24.9 mm to 48.3 mm, with an average median particle size of 39.3 mm. In addition to average median particle sizes, Kondolf and Wolman (1993) also suggest maximum median particle sizes of spawning gravel relative to fish length. Based on fork lengths typical of Hanford Reach fall Chinook salmon, the maximum median grain sizes for Hanford Reach fall Chinook should range from approximately 65 mm to 110 mm. In fact, the median surface particle size ranged from 45.0 mm at site 64 to 105.7 mm at site 59. The combined median surface particle size was 77.4 mm for spawning areas compared to 86.3 mm for non-spawning areas. The median particle size for subsurface samples ranged from 17.4 mm at site 94 to 57.7 mm at site 54. The combined median subsurface particle size was 36.6 mm for spawning areas compared to 27.2 mm for non-spawning areas.

The median gravel size on the riverbed surface was within the range that could be moved by a fall Chinook salmon at all of the study sites. This assertion is supported in general by Kondolf and Wolman (1993) and specifically by fall Chinook salmon research conducted in the Snake River, which identified spawning in areas with dominant substrates up to 150 mm (Groves and Chandler 1999). Groves and Chandler defined dominant substrate as that which filled greater than 55% of the video screen in underwater video surveys. Although not equivalent, this is comparable to the median particle size determined using our underwater video method. The largest median particle size on the river bed surface recorded this way in the Hanford Reach was 105.7 mm, well within the Snake River size limit. Median particle sizes predicted for smaller Hanford Reach fall Chinook using the relationships published by Kondolf and Wolman (1993) are comparable to results they presented for other locations and are comparable to smaller median particle sizes for some of the Hanford Reach locations. Given that the median gravel size was always within that which could be moved by a spawning fall Chinook salmon, the variation in gravel size used for spawning is probably largely a function of habitat availability. Thus, it would appear both surface and subsurface grain sizes are within the range useable by fall Chinook salmon.

5.3.2 Riverbed Permeability

Hydraulic conductivity was variable among the study sites but was within the range observed in other alluvial environments similar to the Hanford Reach (Table 5.13). There was a slight declining trend in hydraulic conductivity with the two sites that were furthest downstream in the study area (sites 96 and 101) an order of magnitude lower in hydraulic conductivity than the three sites where fall Chinook salmon consistently spawn (59, 64, and 81). The specific discharge (product of hydraulic gradient and hydraulic conductivity) was lowest at the four lower sites, but variability was high enough to mask any significant differences among all but one of the sites.

Table 5.13. Hydraulic Conductivity Reported in Alluvial Environments Similar to the Hanford Reach

Reference	Location	Sediment Type	Hydraulic Conductivity (cm/s)
Fetter 1994	General	Sands	1.0×10^{-5} to 1.0×10^{-3}
Weight and Wittman 1999	Alluvial valley, Dillon Montana	Alluvial silt, sand, and gravel	7.1×10^{-5} to 8.6×10^{-2}
Springer et al. 1999	Colorado River	Sand Bars	7.0×10^{-4} to 4.9×10^{-2}
Morrice et al. 1997	Headwater Streams	Sand and cobble sized gravel	1.3×10^{-4} to 4.1×10^{-3}
Barrash et al. 1997	Boise River	Alluvial sands	1.0×10^{-2} to 1.0×10^{-1}
Arntzen et al. 2001	Hells Canyon, Snake River	Alluvial silt, sand, and gravel	2.0×10^{-3} to 3.8×10^{-1}
This study	Hanford Reach, Columbia River	Alluvial silt, sand, and gravel	6.0×10^{-4} to 1.1×10^{-2}

Hydraulic conductivity was about twice as high in spawning sites as compared to non-spawning sites, while specific discharge values were nearly three times as high in the spawning reach. The amount of coarse sand and silt was related to the declining hydraulic conductivity in non-spawning sites. The larger amounts of coarse sand and silt observed at the non-spawning sites would increase the amount of matrix between the gravel clasts, thus lowering the hydraulic conductivity and specific discharge of the gravels.

Although the general relationship between hydraulic conductivity and the movement of surface water into the riverbed is well established, surface water movement and hyporheic flow are also complicated by additional factors, including channel morphology, depth of alluvium, location of groundwater discharge zones, and the magnitude and duration of surface water head changes (Vaux 1962; Stanford and Ward 1993; Wondzell and Swanson 1996; Packman and Bencala 2000; Alexander and Caissie 2003). These additional factors complicate comparisons of hyporheic flow between different locations, especially when sampling must occur across large spatial scales. Our study sites were distributed within a 25-km reach of the Columbia River. Although we grouped our study sites into locations with similar channel morphology, water depths, channel velocities, and surface water fluctuations, little is known about the depth of alluvium and precise locations of zones of groundwater discharge into the river within most of our study site (Lee et al. 1997; Peterson et al. 1998; Geist 2000). These additional complicating factors influence the apparent specific discharge of hyporheic water and thus the survival to emergence of salmonids.

The hydraulic conductivity and the vertical hydraulic gradient together determine the magnitude and direction of hyporheic water flow, thereby controlling the delivery of dissolved oxygen to and removal of metabolic waste from salmon redds (Chapman 1988). We estimated survival to emergence using specific discharge results and found survival rates that were very low. For example, using relationships published by Coble (1961), the survival estimates in spawning areas ranged from less than 1% to 10%. In non-spawning locations, the highest estimated survival to emergence was less than 1%. Combined results for

all spawning locations resulted in a survival to emergence estimate of less than 1%. Using the approach of Sowden and Power (1985), survival to emergence estimates varied, but all remained below 5%. It is likely that hyporheic specific discharge estimates are low because we sampled gravels that were not within salmon redds and thus still included fine sediment that would be removed during the spawning process, thus increasing hydraulic conductivity and the hyporheic specific discharge.

5.3.3 Grainsize Versus Spawning and Survival to Emergence

Survival to emergence was also estimated based on subsurface grainsize data using equations of Tappel and Bjornn (1983) and Bennett et al. (2003). Survival estimates using Bennett's equation were lower than those made using the Tappel and Bjornn (1983) equation. This finding is consistent with the empirical results from the two studies and is most likely because Bennett used green eggs and Tappel used eyed eggs. Mortality from newly fertilized eggs to the eyed stage in fall Chinook salmon embryos are approximately 10% based on laboratory studies (PNNL, unpublished data).

Tappel and Bjornn (1983) predicted survival of Chinook and steelhead embryos from the South Fork Salmon River, Idaho and Clearwater River system, Washington using the percentage of grains smaller than 0.85 mm and 9.5 mm. The authors argued that most grainsize data from alluvial rivers will exhibit a log-normal particle size distribution; however some sediment samples from the Salmon River had substantial deviations from lognormality. In general the greatest deviations occurred at the upper end of the distribution (>25.4 mm). Thus, they reasoned that within the distribution finer than 25.4 mm, the distribution was lognormal and could accurately be represented by the percent finer than values that corresponded to grainsizes of 0.85 and 9.50 mm. The survival to emergence of Chinook salmon was impacted by gravel material of 1.70 to 4.76 mm in diameter. Survival isolines showed that the percentage of sediment smaller than 9.5 mm was not as critical in determining survival as the percentage of sediment smaller than 0.85 mm. The authors constructed a regression relationship to estimate percent survival. Hanford Reach survival estimates using Tappel and Bjornn's method were significantly lower than the author's results for Chinook survival. For Hanford Reach survival estimates, we used sieve sizes that most closely approximated those used by Tappel and Bjornn (1983), including sediments finer than 1 mm and 11.3 mm. The percentage of substrate finer than 1 mm and finer than 11.3 mm in the Hanford Reach was slightly higher than the percentage of substrate finer than 0.85 mm and 9.5 mm from the Salmon River and Clearwater River systems, which lowered Hanford Reach survival estimates. Tappel and Bjornn (1983) reasoned that embryo survival in the field should be lower than survival rates obtained in their lab study due to the relatively high concentration of organic matter that should be present in natural streams. Similar to our study, Tappel and Bjornn (1983) did not attempt to account for changes in grainsize distribution caused by fine removal during spawning. The percentage of substrate finer than 1 mm and 8 mm were substantially higher than Bennett's values from the Snake River.

Bennett's study was based on the potential habitat in the mainstem Snake River. The mean percent fines <6.4 mm from seven sites in the Snake River ranged from 6.7% to 19.8% with the percentages increasing at sites lower in the study reach. The mean percentage of fines less than 0.85 mm ranged from 1.2% to 8.5% with a similar finding that upstream spawning sites had lower values than downstream spawning sites. The authors did not report a significant relationship between the percent fines less than 0.85 mm and 6.4 mm. We found percent fines (less than 1 mm) within the nine study sites in the Hanford Reach that ranged from 9.4% to 30.2%. That the percent fines in the Hanford Reach were much higher than the Snake River is not surprising. The most likely explanation for the differences between the two rivers is that the geomorphic templates through which the two rivers flow are quite different. The

template most often associated with fall Chinook salmon production areas is one situated within broad alluvial floodplains possessing a complex channel composed of unconsolidated alluvium organized through dynamic equilibrium into bars and islands (Battelle and USGS 2000). This template results in large amounts of highly permeable substrate conducive to high survival-to-emergence ratios. The templates for the Hanford Reach production area and the Snake River production areas are very different from one another and represent a continuum ranging from the “normative” (Stanford et al. 1996) template (e.g., the Hanford Reach) to one dominated by a bedrock canyon (i.e., Hells Canyon of the Snake River). The physical controlling factors set the limits (i.e., the template) within which any potential habitat is ultimately realized. The variation in substrate quality is reflective of the differences in their templates, including the geomorphic features of the river channel, position in the watershed, and longitudinal slope of the river through the three reaches. For example, fine sediment content was highest in the Hanford Reach primarily because of the alluvial unconsolidated geologic formations through which it flows and because of the relatively low energy gradient imposed on the river by the low longitudinal slope of the channel bed. In other words, the fine sediment content in the Hanford Reach was what would be expected, given the template provided. Similarly, the bedrock landforms of the Hells Canyon control the local channel morphology, and the lower percent fines found within Hells Canyon riverbed sediments is consistent with the template provided.

Various indicators have been presented in the literature to assess survival to emergence. Platts et al. (1979) suggested the geometric mean of the subsurface grainsize sample as a companion to the percentage fines because d_g is commonly used in other disciplines. A number of studies have suggested that survival of embryos declines at geometric means less than 15 mm. The geometric means of the subsurface samples from the nine sites in the Hanford Reach ranged from 4.5 mm at site 94 to 30.0 mm at site 54. However, the geometric mean may not be a good metric against which to compare survival (Kondolf 2000). Tappel and Bjornn (1983) showed that two sediment samples with equal geometric means could have very different particle size distributions and survival to emergence. In gravels with equal geometric means, Tappel and Bjornn (1983) found higher survivals than previously reported by Shirazi and Seim (1979). Kondolf (2000) suggests there is no reason why the geometric mean should be a unifying metric for use in salmon spawning survival studies.

The Fredle index of subsurface samples has also been used as an indicator of survival to emergence. Based on the survival-to-emergence estimation method of Lotspeich and Everest (1981), survival in spawning areas ranged from 25% at site 64 to 90% at site 61. Survival in non-spawning sites ranged from 0% at 94 to 100% at 90. Survival for the combined f_i for all spawning sites is 50%; for non-spawning sites, it is 35%. Sowden and Power (1985) also related survival to emergence to the Fredle index. Using Sowden and Powers relationship with Hanford Reach f_i values, spawning survival would range from 10% at site 64 to 80% at site 61. Survival in non-spawning sites would range from 0% at site 94 to 100% at site 90. Survival for combined spawning sites would be 25%; for non-spawning sites, it would be 13%.

The survival to emergence predicted by the equations is probably lower than the actual survival in the Hanford Reach. This is because spawning salmon reduce the percentage of fines in the substrate during redd construction. Kondolf (1993) quantified the relationship between percent fines less than 1 mm in unused gravel compared to post-spawning gravel and noted that more fine sediment is usually removed during the spawning process when initial gravels have higher fine sediment content. It is thus likely that Hanford Reach sediment permeability increases significantly during the spawning process; however, the process of fine sediment intrusion during incubation is equally likely to occur (Lisle 1989). Healy (1991) reviewed survival to emergence of all Chinook salmon stocks and concluded it is probably around 30%.

Lapointe et al. (2004) determined that Atlantic salmon (*Salmo salar*) embryo survival declined with large silt contents, but if sand content (coarse and fine sand combined) was very low (<5%), then increasing silt did not have as much of a reduction in survival to emergence. For sand contents over 10%, an increment of 1% silt had over three times the effect on survival as a 1% increment in sand content. Variations in only a few percentage points in silt had deleterious effects on survival to emergence if sand content was elevated. “For example, at 15% sand (SI of 1.25) mean survival decreases from 60% to 20% as silt content increases over the range 0%–4%” (Lapointe et al. 2004, p. 2276).

5.4 Conclusions

Hanford Reach riverbed sediments were composed of alluvial sand, silt, and gravel in a heterogeneous mixture within the surface and the subsurface. At all sites evaluated, the riverbed was armored. The armored condition of riverbed sediments is likely confounded by anthropogenic activities, including sediment entrainment behind dams, reduced hydraulic capacity for flood control, and fluctuations in river discharge for load-following hydroelectric operations. Despite the armored nature of the riverbed, particle sizes at all locations evaluated were within the range that could be excavated by fall Chinook salmon during spawning. It is likely that particle size variation between and within spawning and non-spawning areas is a function of habitat availability rather than the inability of fall Chinook salmon to displace sediment during spawning. The silt and coarse sand content, which decreases sediment permeability and prevents hyporheic flow through salmon redds, was twice as high in non-spawning areas as in spawning areas.

The hydraulic conductivity of Hanford Reach riverbed sediments ranged from 6.0×10^{-4} to 1.1×10^{-2} cm/s, a similar result to those obtained in other research in alluvial environments. The hydraulic conductivity was twice as high in spawning areas as in non-spawning areas. Lower hydraulic conductivity in non-spawning sites was proportional to the increased coarse sand and silt content in those locations. Hyporheic water flow, which provides dissolved oxygen to eggs and removes metabolic waste during incubation, is an important influence on survival to emergence. The flow of hyporheic water through spawning gravels was three times higher in spawning locations than in non-spawning areas. Survival-to-emergence estimates using specific discharge of hyporheic water were higher in spawning areas than in non-spawning areas but low overall. Estimates ranged from 1% to 10% in spawning areas and were less than 1% in non-spawning locations.

Survival to emergence was evaluated also using the grainsize distribution, primarily the percentage of substrate finer than the 1-mm, 8-mm, and 11.3-mm particle size classes so results could be compared to survival estimates from studies conducted on Snake River fall Chinook salmon populations. Hanford Reach survival estimates made from comparisons to results from eyed egg studies ranged from 0–100% in non-spawning areas to 69–89% in spawning areas. Estimates made from comparisons to results from green egg studies ranged from 0–41% in non-spawning areas to 7–33% in spawning areas. Of the nine study sites evaluated in the Hanford Reach, survival-to-emergence estimates were generally higher in the six upstream sites than in the three downstream locations.

The relatively high percentage of fines in the Hanford Reach and associated low survival estimates are not surprising. The most likely explanation is related to the geomorphic template through which the Hanford Reach flows, which includes broad alluvial floodplains possessing a complex channel comprising heterogeneous unconsolidated alluvium. Despite the propensity of Hanford Reach sediments toward fine particles, anthropogenic influences are likely removing fine particles from the riverbed surface (thus increasing riverbed armoring) while simultaneously increasing the percentage of fine sediments in the subsurface at egg pocket depth. Although we found important differences suggesting substrate quality is better in spawning areas than in non-spawning areas, it is likely that anthropogenic influences are having a negative impact on processes affecting substrate quality of Hanford Reach riverbed sediments.

6.0 Groundwater and Surface Water Interactions Within Fall Chinook Salmon Spawning Sites in the Hanford Reach

Standard spawning habitat characteristics that were measured in fall Chinook salmon (*Oncorhynchus tshawytscha*) spawning sites were able to correctly predict spawning site selection approximately 70% of the time (see Section 4 of this report). Very little improvement in predictive power was attainable, even after accounting for the variability in physical habitat characteristics that resulted from changes in river flow in response to load-following operations (i.e., producing electric power in response to short-term electrical demand) at Priest Rapids Dam (at the upstream terminus of the Hanford Reach). Previous investigations into fall Chinook salmon spawning site selection suggested that redds were found in areas where upwelling river water was present (Geist 2000). Investigators have suggested that the interaction of groundwater and surface water within hyporheic habitats may offer an explanation for some of the error associated with models that use standard habitat characteristics to predict redd site selection (Geist and Dauble 1998; Dauble and Geist 2000).

The hyporheic zone is the subsurface region of streams and rivers where groundwater and surface water interact (Valett et al. 1993). Water flow into and out of the hyporheic zone is influenced largely by advective exchange with the river, a process generally controlled by channel morphology, pressure head of overlying surface water, and the permeability of riverbed sediments (Landon et al. 2001; Wörman et al. 2002; Cardenas and Zlotnik 2003; Rose 2003). Consequently, the magnitude of river flow and the underlying geology can significantly affect chemical, physical, and biological gradients within the hyporheic zone, which ultimately will affect the structure and function of aquatic ecosystems (Stanford and Ward 1993; Curry et al. 1994; Wroblicky et al. 1998; Soulsby et al. 2001; Alexander and Caissie 2003).

The interaction of groundwater and surface water within hyporheic zones is now recognized as important in redd site selection by salmonids. Examples of redd site selection being influenced by groundwater–surface water interactions in hyporheic habitats have been presented for chum salmon (*O. keta*; Geist et al. 2002), sockeye salmon (*O. nerka*; Lorenz and Eiler 1989), Chinook salmon (Geist 2000), Atlantic salmon (*Salmo salar*; Alexander and Caissie 2003), brook trout (*Salvelinus fontinalis*; Webster and Eiríksdóttir 1976; Witzel and MacCrimmon 1983), and bull trout (*S. confluentus*; Baxter 1997). Adult salmonids selecting redd sites likely use physical and chemical cues originating from hyporheic zones in choosing optimal sites to deposit eggs. Intergravel movement of water within the egg pocket optimizes incubation conditions by moderating temperature, removing waste from eggs and delivering oxygen to eggs. Emergence of offspring from the gravel is controlled by temperature and egg pocket environment; optimal emergence increases probability of survival (Brannon 1987).

Higher river flows result in hydraulic gradient reversals during which the direction of water flow in the hyporheic zone is altered (Fraser et al. 1996; Wroblicky et al. 1998). Physicochemical changes during gradient shifts when stream discharge is rising often differ from those when it is declining, and the data can exhibit a noticeable hysteresis (Soulsby et al. 2001; Alexander and Caissie 2003; Arntzen et al. 2006). In river systems influenced by hydroelectric dams operated in a load-following mode, physiochemical gradient shifts occur many times per day. This diurnal fluctuation in physiochemical characteristics

affects the hyporheic zone structure and function (Williams 1993; Curry et al. 1994) and likely alters the cues that adult salmon use to select redd sites. Water quality changes within the hyporheic zone will also affect the timing of fry emergence.

The objectives of this study were to 1) document and describe the physiochemical characteristics of the hyporheic zone in fall Chinook salmon spawning and non-spawning sites, 2) evaluate whether operations of Priest Rapids Dam affect the interaction of groundwater and surface water in spawning and non-spawning sites and determine whether sediment permeability affects this interaction, and 3) calculate hatch and emergence dates of fall Chinook salmon using river and hyporheic temperatures to determine whether there are differences among the study sites.

6.1 Methods

6.1.1 Measurements of Vertical Hydraulic Gradient, Temperature, and Flux

Beginning in November 1999 and ending in November 2004, temperature and hydraulic interactions between groundwater and surface water at each of the nine study sites (see Section 2) were monitored using self-contained temperature and water-level sensors/data loggers suspended inside piezometers. With the exception of November 1999 through March 2000 (during which only temperature of the hyporheic zone was monitored), each site contained one piezometer emplaced into the riverbed and one river standpipe (i.e., no river standpipe was installed in 1999). Piezometer screens were constructed of slotted stainless steel Johnson screen (0.038-cm slot size) with a 31.0-cm screened interval and a 3.2-cm inside diameter. The screen was welded on one end to a 12-cm drive point and on the other end to a variable-length section of galvanized steel pipe (3.2 cm inner diameter) threaded on top. Piezometers were installed by inserting a solid steel drive-rod into the piezometer and either pounding the piezometer with the rod or with a post-pounder (Geist et al. 1998). Once piezometers were in place, the internal drive-rod was removed and the piezometer was developed by removing fine sediment (smaller than 0.038 cm) with a hand pump. The river standpipe was constructed from an unscreened section of galvanized pipe perforated with holes (0.64 cm diameter) and welded to a drive point. The same drive-rod used to install the piezometers was used to install the river standpipe. The standpipe was pounded into the riverbed until the top of the perforated section of pipe was approximately 20 cm above the bed of the river. The elevations (meters above sea level, National Geodetic Vertical Datum) of the piezometer and the river standpipe were surveyed by a licensed surveyor (Rogers Surveying, Richland, Washington).

Data loggers were suspended inside the piezometer and river standpipe by nonstretch stainless steel cable attached to a watertight cap at the top of each pipe. The temperature and pressure sensor end of the data loggers was placed near the top of the piezometer screen and just above the riverbed surface inside the river standpipe. In 1999, temperature loggers (Onset Optic Stowaway, accuracy $\pm 0.2^{\circ}\text{C}$) were deployed in the piezometers (none in the river). The depth below the riverbed to the temperature sensor ranged from 37.5 cm at site 101 to 72.5 cm at site 81 (Table 6.1). The Onset loggers were set to record data once every hour.

Table 6.1. Piezometers Where Onset (stowaway) Temperature Loggers Were Deployed from 21 November 1999 to 3 April 2000

Piezometer Label	Transect (river kilometer)	DL ^(a) (cm)
T54LB04	54 (605.4)	61.0
T59IL04A	59 (603.2)	67.5
T59IR04	59 (603.2)	67.5
T61IR04	61 (602.1)	67.5
T64IR04	64 (600.0)	71.5
T81RB04	81 (590.4)	72.5
T90LB04	90 (585.2)	70.5
T94LB04	94 (581.9)	62.5
T96RB04	96 (580.5)	69.5
T101LB04	101 (577.4)	37.5
Mean		64.5 (w/o 101 = 67.8)
(a) DL is the distance below the riverbed to the top of the screen.		

In subsequent years, two different models of temperature-pressure loggers manufactured by Solinst were used. For nearly all deployments, the Solinst Model 3001 LT temperature-pressure logger was used. In 2003, we deployed Solinst Model 3000 L pressure loggers (no temperature) in the piezometer and river standpipe at site 101. According to calibration certificates provided by the data logger manufacturer, the instruments were accurate to $\pm 0.1^{\circ}\text{C}$ and ± 0.7 cm of water. The temperature of each logger was checked in our laboratory by comparing it to the temperature recorded by a thermometer certified by the National Institute of Standards and Technology. Solinst data loggers were programmed to record temperature and absolute pressure (centimeters of water) every 15 min during the 2001–2002 deployments; in subsequent years, recordings were made every 30 min. One additional Solinst Model LT data logger was deployed near transect 54 to record atmospheric pressure at the same intervals. Atmospheric pressure was subtracted from the absolute pressure readings in the piezometer and standpipe to determine the gage pressure (centimeters of water). All data collected using temperature and pressure data loggers were stored in an access database. The database was copied to a data compact disk and provided directly to the Bonneville Power Administration.

The average distance below the riverbed to the sensor for the loggers deployed from 2001–2004 ranged from 26.5 cm at site 81 to 41.0 cm at site 96 (Table 6.2). At six of the nine sites, the distance below the riverbed over the entire deployment did not vary more than 5 cm (sites 54, 59, 61, 81, 90, and 94—see Table 6.2). At site 64, the sensor was moved up 8 cm from 39 to 31 cm from November 2002 to November 2004. Thus, throughout most of the study, the sensor was 31 cm below the surface of the riverbed. At site 101, the sensor was approximately 24.0 cm below the bed from October 2001 to February 2003 (the temperature data from October 2002 to February 2003 were not collected because of sensor failure). In November, 2003, the piezometer was reinstalled and a Model L logger was installed. The distance to the sensor was deepened by 9 cm to 42 cm. The largest difference in distance to sensor occurred at site 96 (Table 6.2). From October 2001 to June 2002, the distance to the sensor was 32.0 cm; from November 2002 to November 2003, the distance was 36.1 cm; and from November 2003 to November 2004, the distance was 45.0 cm.

Table 6.2. Distance from Surface of Riverbed to Sensor in the Hyporheic Zone Piezometers at Nine Sites Monitored from 18 October 2001 to 27 October 2004

Site	Average $dl^{(a)}$ (cm)	Maximum dl (cm)	Minimum dl (cm)	Range (cm)
54	34.5	35.0	33.0	2.0
59	26.8	27.0	26.0	1.0
61	30.0	30.0	30.0	0.0
64	32.5	39.0	31.0	8.0
81	26.5	26.5	26.5	0.0
90	30.4	35.0	30.0	5.0
94	27.3	31.0	26.5	4.5
96	41.0	45.0	32.0	13.0
101	29.2	32.0	23.0	9.0
Overall	30.9	33.4	28.7	4.7
(a) Distance from surface of riverbed to sensor.				

The vertical hydraulic gradient (VHG) was determined as the ratio of the difference in head pressure between river and hyporheic zone to the distance from the riverbed to the top of the piezometer screen:

$$\text{VHG} = \frac{dh}{dl} = \frac{[\text{head inside piezometer} - \text{river head}]}{\text{distance from riverbed to top of piezometer screen}} \quad (6.1)$$

Surveyed elevation data for the elevation of the logger in the river and the hyporheic zone were used to adjust head level readings and determine survey-corrected VHG. If survey data were not available, VHG values were expressed as un-corrected VHG (also referred to as “raw” VHG). Unless otherwise indicated, VHG values used in the remainder of the report are survey-corrected VHG values.

Flux of water into and out of the riverbed was determined for the 15–22 November spawning period during each of the three years using the Darcy equation

$$Q = KIA \quad (6.2)$$

where Q = flux ($\text{cm}^3/\text{s}^{-1}$)
 K = hydraulic conductivity (cm/s)
 I = vertical hydraulic gradient (unitless)
 A = area (arbitrarily set at 100 cm^2).

Flux was expressed in terms of a net movement of water (converted to units of $\text{cm}^3/\text{h}^{-1}$) by observing the direction of movement (based on positive or negative VHG), as well as by an absolute magnitude (nondirectional). VHG was calculated as described above, while hydraulic conductivity values were estimated as described in Section 5. We decided that at least 90% of the hourly data needed to be present to make the calculation of either net or absolute flux over the one-week spawning period.

6.2 Data Analysis

6.2.1 Objective 1 - Site-to-Site Comparisons of Temperature, Vertical Hydraulic Gradient, and Flux

Differences in temperature and VHG during the spawning period were compared among spawning and non-spawning sites. Spawning site designation was based on an analysis of aerial photographs of redds (Visser et al. 2002) as well as on annual spawning surveys conducted since 1948 for the U.S. Department of Energy (Dauble and Watson 1997; PNNL, unpublished data). The process used to designate study sites as spawning and non-spawning was described in Section 2. The spawning period was defined as 18 October through 22 November (peak spawning is 11 November based on Dauble and Watson [1990]). A one-week subset of the entire spawning period was used to analyze the continuous temperature data. This one-week period enabled us to make more comparisons of data than would have been available otherwise, had we used the entire period from 18 October through 22 November (Table 6.3).

Table 6.3. Two Spawning Periods Used to Compare Spawning and Non-Spawning Sites in 2001, 2002, and 2003

Period	Year	Spawning	Non-Spawning
18 October through 22 November	2001	64	54, 90, 96, 101
	2003	59, 64, 81	61, 90, 94, 96
15 November through 22 November	2001	64	54, 90, 96, 101
	2002	59, ^(a) 64, 81 ^(b)	54, 61, 90, 94, 96, 101
	2003	59, 64, 81	61, 90, 94, 96
(a) Hyporheic temperature only. (b) River temperature and VHG.			

Comparisons were made of hyporheic temperature, river temperature, and the difference in temperature between the hyporheic zone and the river. Comparisons between spawning and non-spawning sites were performed using box plots, two-sample t-tests, and nonparametric Kolmogorov-Smirnov tests. Note that for two-sample t-tests, the reported probability is the Bonferroni adjusted probability with separate variance t value. If the probability is less than 0.05, the tested variable is significantly different for spawning and non-spawning sites.

Analyses of hydraulic gradients between the river and riverbed were primarily based on dh values. The dh values were used so that hydraulic gradients could be evaluated relative to the uncertainty error of the instruments (± 1.4 cm), which does not vary over the range of depths for which they were used in this study. Differences in mean dh among sites and spawning period were evaluated through one-way analysis of variance. Differences in mean dh between individual sites were identified using Tukey's HSD post-hoc multiple comparison test.

6.2.2 Objective 2 – Effect of Hydropower Operations on Groundwater–Surface Water Interactions

The discharge and rate of flow fluctuation at Priest Rapids Dam was examined to determine if dam operations affected hyporheic conditions. Water surface elevations from the river sensors were converted to river stage using the vertical survey results. These river stage values were compared to the temperature during two time periods—the spawning period from 18 October through 22 November and a period of time in the winter when hyporheic temperatures tend to stabilize at the lowest temperatures of the year (1 January through 15 March).

In addition to temperature, we also looked at the relationship between VHG and river stage. To overcome limitations on the availability of survey VHG, we used the difference in VHG from one hour to the next; we refer to this as delta VHG. For the first comparison, we looked at the same time periods that were used when looking at temperature, namely the spawning period (18 October through 22 November) and the winter period (1 January through 15 March). Two conclusions were reached in this initial examination of the data. First, the delta river stage (i.e., the change in river stage from one hour to the next hour) should be used rather than raw river stage. Second, shorter time periods need to be examined. The relationships were difficult to visualize over the long spawning and winter time periods. Therefore, we focused on five-day time periods from the fall for each of the four years (5–11 November 2001; 15–19 November 2002; 23–27 October 2003; and 19–23 October 2004). These five-day time periods were selected because they fall within the spawning period and they have similar characteristics for the four-year period. Each five-day period is from Monday through Friday and included five major excursions in river stage. To be sure that the five-day period was representative of the longer spawning period, we also compared the same values for the entire spawning period and found little difference.

Cross-correlations were calculated to determine the correlation between delta VHG and delta river stage. The correlations were based on the original data in 15- or 30-minute increments. Correlation plots were constructed that showed the strength of the relationship, the direction of the relationship (positive or negative), and the lag time between changes in river stage and changes in VHG.

We were interested in whether riverbed permeability affected the relationship between dam operations and hyporheic response (see Arntzen et al. 2006). A subset of the sites was examined in more detail by plotting the relationship between river stage and VHG over a 25-hour time period in October 2004 at three sites (sites 81, 90, and 96). These three sites were assumed representative of other sites of similar permeability. Riverbed permeability at sites 81, 90, and 96 was measured at 1.54×10^{-2} cm/s, 6.03×10^{-3} cm/s, and 4.3×10^{-4} cm/s, respectively (Section 5 of this report). Other time periods were not examined, but it is assumed they would be similar.

6.2.3 Objective 3 – Hatching and Emergence Dates Using Bed and River Temperatures

The accumulated thermal units (ATU) were calculated from river and riverbed temperature at each of the nine sites for each of the deployment years (river temperatures not available in 1999). The spawning and incubation period for the calculation of ATU was 15 November through 15 May. We had data from various sites for all three years (2001–2002, 2002–2003, and 2003–2004; Table 6.4). We assumed that hatching would occur at 500 ATU and emergence would occur at 1,000 ATU (Piper et al. 1982).

Table 6.4. Summary of Data Availability for Objective 3 for the Period from 15 November Through 15 May. Gray cells mean no daily average data available.

Site	15 November 2001 to 15 May 2002 (182 days)		15 November 2002 to 15 May 2003 (182 days)		15 November 2003 to 15 May 2004 (183 days)	
	River	Hyporheic	River	Hyporheic	River	Hyporheic
54		x	x	x		
59				x	x	x
61			x	x	Gaps between 21 November 03 to 22 February 04: too many data points missing	x
64	x	x	x	x	No data from 24 February 04 to 15 May 04	x
81			x		x	
90	x	x	x	x	x	x
94			Gaps between 2 January 03 to 4 March 03: linear interpolation	x	Gaps between 28 November 03 to 30 November 03 and 21 February 04 to 12 April 04: linear interpolation	x
96	x	x	No data from 4 February 03 to 15 May 03	x	No data from 15 November 03 to 21 November 03: linear interpolation to first hyporheic temperature	x
101	x	x	No data from 4 February 03 to 15 May 03	No data from 4 February 03 to 15 May 03		

One ATU is equal to the number of degrees above 0°C summed for each day. Thus, if on day 1 and day 2 the average temperature was 5°C for each day, the ATU would be 5 on day 1 and 10 on day 2. Snake River fall Chinook reached 50% emergence at about $1,006 \pm 9$ ATU (Connor et al. 2003). Geist et al. (in press) showed that 50% hatch of Snake River fall Chinook salmon occurred around 535 (range 502–587) ATU and that 50% emergence occurred around 944 (range 868–1,001) ATU. Unpublished work by Battelle in 2003 with Priest Rapids Hatchery fall Chinook salmon showed that 50% emergence occurred around 1,020 ATU (range 999 to 1,054 ATU).

6.3 Results

6.3.1 Objective 1 – Site-to-Site Comparisons of Temperature, Vertical Hydraulic Gradient, and Flux

6.3.1.1 Temperature

Hyporheic temperatures were very consistent from site to site, and the seasonal cycle was smoothly reproduced (Figure 6.1). The seasonal low temperatures always occurred in February, with the monthly average 4.8°C in 2002 (range 4.4 to 5.1°C), 5.7°C in 2003 (range 5.3 to 6.4°C), and 4.1°C in 2004 (range 3.8 to 4.9°C). In two of the three years, average hyporheic temperatures in February were lowest at site 90. The warmest hyporheic temperatures consistently occurred in August, with the monthly average 19.5°C in 2003 (range 18.9 to 20.4°C) and 19.6°C in 2004 (range 18.9 to 20.6°C). The warmest hyporheic temperatures consistently occurred at site 64.

The river was usually cooler than the hyporheic zone from November through April, with the largest differences (0.7 to 1.2°C) occurring in January and February (Figure 6.2); average river temperature in January and February during the study ranged from about 3°C in 2004 to around 6°C in 2002. Around May in each year, the river began heating up faster than the hyporheic zone and stayed warmer than the hyporheic zone until November. The largest differences during this period averaged 1.0 to 1.2°C and occurred in July (Figure 6.2), when the river was around 19°C and the hyporheic zone around 17.5 to 18.5°C. The largest single-month difference in temperature between the hyporheic zone and the river in the winter period occurred at site 59 in February 2004, when the hyporheic zone was about 2°C warmer than the river. In contrast, the river reached an average of 1.7°C warmer than the hyporheic zone in that same year in July at site 96.

Average river temperatures over the entire spawning period (18 October through 22 November) in spawning and non-spawning areas in 2001 and 2003 were within 0.4°C of one another, averaging around 13.0°C in both years (Table 6.5). There were few site-to-site differences in river temperature (Figure 6.3). As shown in Figure 6.4, at the start of the spawning period in mid-October, river temperatures were slightly less than 16°C in 2001 and well over 16°C in 2003 (no data are available for 2002). By 11 November (peak spawning), river temperatures in both years were around 12°C (Figure 6.4). River temperatures during the last week of the spawning period for all three years were much less variable than for the entire period (Figure 6.3). Average river temperatures during this one-week period were around

Figure 6.1. Monthly Average Hyporheic Temperatures at Eight Sites. Site 81 not shown because of logger malfunction.

Figure 6.2. Monthly Average Temperature Differences Between River and Hyporheic Zone at Eight Sites from 2001–2004. Temperature difference is the river temperature minus the hyporheic temperature. Site 81 not shown because of logger malfunction.

Table 6.5. Group Mean for Spawning and Non-Spawning Sites from Various Time Periods

Variable	Site	18 October Through 22 November		15 November Through 22 November		
		2001	2003	2001	2002	2003
River Temperature	Non-Spawning	12.70	12.98 ^(a)	11.98	11.82	10.85
	Spawning	13.10	13.00 ^(a)	12.21	11.96	11.07
Hyporheic Temperature	Non-Spawning	12.93	13.31 ^(a)	12.21	12.11	11.32
	Spawning	13.28	13.44 ^(a)	12.41	12.60	11.72
Temperature Difference	Non-Spawning	-0.15 ^(b)	-0.42	-0.20 ^(a,b)	-0.30	-0.46
	Spawning	-0.18 ^(b)	-0.47	-0.20 ^(a,b)	-0.84	-0.63
River Stage	Non-Spawning	103.24	160.77	130.63	169.41	171.06 ^(a,b)
	Spawning	NA	151.41	NA	149.88	159.39 ^(a,b)
Vertical Hydraulic Gradient	Non-Spawning	0.03	0.10	0.03	-0.12	0.11
	Spawning	0.97	-0.13	0.97	-0.0001	-0.13
(a) No significant difference in t-test (95% confidence). (b) No significant difference in Kruskal-Wallis test. (c) No significant difference in Kolmogorov-Smirnov test.						

12°C in 2001 and 2002 and 11°C in 2003 in both spawning and non-spawning areas (Table 6.5). The differences between spawning and non-spawning sites were less than 0.4°C and, although statistically significant, had little biological significance.

Hyporheic temperatures over the entire spawning period averaged about 13°C in 2001 and 2003 in both spawning and non-spawning areas (Table 6.5). The hyporheic temperature of spawning areas was slightly warmer than non-spawning areas (about 0.4°C in 2001 and 0.1°C in 2003) but were near the expected variability range of the instrumentation, suggesting these differences were not significant. In general, hyporheic temperatures in 2001 were more variable at the lower three sites (sites 90, 96, and 101) than at the upper two sites (sites 54 and 64; Figure 6.5). With the exception of site 94, hyporheic temperatures were equally variable at all sites in 2003 (Figure 6.5). In both 2001 and 2003 over the entire spawning period, the warmest hyporheic temperatures were always observed at site 64.

In general, the temperature of the hyporheic zone mimicked the river temperature (compare Figure 6.4 to Figure 6.6). The average temperature of the hyporheic zone did not differ much from the river temperature, with less than 0.2°C and 0.4°C separating them in 2001 and 2003, respectively (Table 6.5). Over the entire spawning period in 2001, there were few differences among sites in the temperature between the river and hyporheic zone (Figure 6.7). This contrasts with 2003, when there was considerable variability among the sites in temperature differences between the hyporheic zone and the river (Figure 6.7). In 2003, the largest differences in temperature were observed at site 64, where the hyporheic zone was nearly 1°C warmer than the river (Figure 6.7).

As with river temperature, there was considerably less individual site variability in hyporheic temperatures during the one-week spawning period than observed over the entire 5-week spawning period (Figure 6.5). In all years, hyporheic temperatures during the one-week spawning period were warmer in

18 October through 22 November

2001

15 November through 22 November

2001

2002

No data

2002

2003

2003

Figure 6.3. River Temperature. The medians of the distributions are shown as the center of the notches, and the lower and upper quartiles as the hinges of the box plots. Asterisks and circles represent outlier data points at 1.5 and 3.0 times the interquartile range, respectively. The notches represent an approximate 95% confidence interval around the median (McGill et al. 1978).

Figure 6.4. Continuous River Temperatures

18 October through 22 November

2001

2002
No data

2003

15 November through 22 November

2001

2002

2003

Figure 6.5. Hyporheic Temperature. The medians of the distributions are shown as the center of the notches, and the lower and upper quartiles as the hinges of the box plots. Asterisks and circles represent outlier data points at 1.5 and 3.0 times the interquartile range, respectively. The notches represent an approximate 95% confidence interval around the median (McGill et al. 1978).

Figure 6.6. Continuous Hyporheic Temperatures

18 October through 22 November

15 November through 22 November

2001

2001

2002

No data

2002

2003

2003

Figure 6.7. Temperature Differences (river minus hyporheic). The medians of the distributions are shown as the center of the notches, and the lower and upper quartiles as the hinges of the box plots. Asterisks and circles represent outlier data points at 1.5 and 3.0 times the interquartile range, respectively. The notches represent an approximate 95% confidence interval around the median (McGill et al. 1978).

spawning areas than they were in non-spawning areas, however, differences were only 0.2 to 0.5°C (Table 6.5). During the one-week spawning period, site 64 had relatively warm hyporheic temperature relative to hyporheic temperatures at most other sites (Table 6.6). The average hyporheic temperature at site 64 over the three year period during the spawning interval was 12.4°C (11.9 to 12.8°C). The 3-year average hyporheic temperature at other sites ranged from 11.5°C (site 94) to 12.3°C (site 54). Whether looking at differences in spawning and non-spawning among individual sites, or between the spawning and non-spawning reach, the spawning areas were about 0.3°C warmer than the non-spawning areas. For example, the average hyporheic temperature of the spawning sites during the one week spawning period averaged over the three years was 12.2°C (sites 59 and 64) versus 11.9°C at non-spawning sites (sites 54, 61, 90, 94, 96, and 101). The average hyporheic temperature in the spawning reach (sites 59, 61, 64, and 81) was 12.1°C as compared to 11.8°C in the non-spawning reach (sites 90, 94, 96, and 101). These differences between spawning and non-spawning sites are close to the accuracy of the instruments. Thus, one would conclude that there are no biologically significant differences in the hyporheic temperatures during a one-week period (15–22 November) in 2001–2003 at spawning and non-spawning sites.

During the one-week spawning period in each of the three years we studied, the difference in temperature of the hyporheic zone and the river was consistently greatest at site 64 (Figure 6.7; Table 6.7). The hyporheic zone at site 64 was about 0.64°C warmer than the river when averaged across the three years (range 0.20 to 0.87°C; Table 6.7). The hyporheic zone at all sites was, on average, warmer than the river, with greater differences found at spawning sites than at non-spawning sites. For example, the hyporheic zone was about 0.58°C warmer than the river at the spawning sites when assessed over the three-year period during the one-week spawning period (Table 6.7). For comparison, the hyporheic zone at non-spawning sites averaged 0.34°C warmer than the river when assessed over this same time period.

Table 6.6. Mean Hyporheic Temperature by Site During the Spawning Period 15–22 November. Standard error is in parentheses.

Site	2001	2002	2003	Overall Mean
54	12.32 (0.022)	12.35 (0.011)	ND ^(a)	12.34 (0.012)
59	ND	12.37 (0.008)	11.54 (0.015)	11.96 (0.023)
61	ND	12.31 (0.009)	11.55 (0.019)	11.93 (0.022)
64	12.41 (0.018)	12.83 (0.011)	11.91 (0.019)	12.39 (0.018)
81	ND	ND	ND	ND
90	12.42 (0.021)	12.08 (0.007)	11.47 (0.018)	11.99 (0.019)
94	ND	11.98 (0.013)	11.00 (0.036)	11.49 (0.031)
96	12.07 (0.023)	11.90 (0.007)	11.24 (0.030)	11.74 (0.020)
101	12.02 (0.025)	12.01 (0.008)	ND	12.02 (0.013)
(a) ND = no data.				

Table 6.7. Mean Temperature Difference (river-hyporheic) During the Spawning Period 15–22 November. Standard errors in parentheses.

Site	2001	2002	2003	Overall Mean
54	ND ^(a)	−0.55 (0.006)	ND	−0.55 (0.006)
59	ND	ND	−0.40 (0.007)	−0.40 (0.007)
61	ND	−0.32 (0.003)	−0.43 (0.003)	−0.35 (0.004)
64	−0.20 (0.004)	−0.84 (0.006)	−0.87 (0.003)	−0.64 (0.013)
81	ND	ND	ND	ND
90	−0.34 (0.007)	−0.20 (0.007)	−0.64 (0.010)	−0.39 (0.009)
94	ND	−0.15 (0.011)	−0.16 (0.016)	−0.15 (0.010)
96	−0.10 (0.010)	−0.20 (0.006)	−0.80 (0.074)	−0.20 (0.012)
101	−0.15 (0.008)	−0.33 (0.007)	ND	−0.24 (0.007)
(a) ND = no data.				

The hyporheic zone within the spawning reach (upper four sites not including site 54) was, on average, 0.47°C warmer than the river; this compares to 0.29°C for the non-spawning reach (lower four sites). The differences in temperature between the hyporheic zone and the river were the most variable at sites 90, 94, and 96 (Figure 6.7).

6.3.1.2 Vertical Hydraulic Gradient

In general for all the data, the survey-corrected VHG at most of the sites fluctuated around 0 (Figure 6.8). The exception was at sites 96 and 59, where the values fluctuated around a value of −0.4. Raw VHG values were the only data available in 2001 and fluctuated around 1 and −1. These values are not directly comparable to survey-corrected VHG values in subsequent years.

Values for VHG were very similar between the two spawning periods (Figure 6.9). For example, in 2001 when we had VHG values for the entire spawning period (18 October through 22 November), the VHG values were the same for each time period (Table 6.5). Therefore, we used the VHG values from the one-week spawning period to assess whether there were differences in VHG between spawning and non-spawning areas. Our analysis did not find an obvious pattern in VHG values between spawning and non-spawning sites. Values in spawning sites ranged from negative 0.13 to positive 0.97 whereas non-spawning sites the range was from negative 0.12 to positive 0.11 (Table 6.5). Sites for which we have VHG data for all three years were few. The mean (standard error) VHG values at site 64 were 0.970 (0.001) in 2001, 0.0 (0.002) in 2002, and −0.015 (0.003) in 2003 (Table 6.8). Site 90 averaged (standard error) 0.836 (0.003) in 2001, −0.071 (0.002) in 2002, and 0.090 (0.002) in 2003. The most variable site was site 96 where the average (standard error) was −1.25 (0.037) in 2001, −0.722 (0.034) in 2002, and 0.116 (0.071) in 2003. The standard errors at site 96 were considerably larger than standard errors at other sites (Table 6.8).

Figure 6.8. Monthly Mean Vertical Hydraulic Gradient (VHG) Between River and Hyporheic Zone at Nine Sites in the Hanford Reach from October 2001 through October 2004. The sites plotted with raw VHG values are marked with an asterisk; these are not comparable to survey-corrected VHG values.

6.3.1.3 Flux

There was considerable variation from one year to the next in the amount of water moving through the riverbed (Table 6.9). For example, absolute flux at site 64 (a spawning site) ranged from $2.43 \times 10^5 \text{ cm}^3/\text{week}$ in 2001 to around $8 \times 10^3 \text{ cm}^3/\text{week}$ in 2002 and 2003. There was no apparent difference in the amount of flux in spawning and non-spawning sites; in fact, site 90 consistently had some of the highest flux values of all the sites.

6.3.2 Objective 2 – Effect of Hydropower Operations on Groundwater–Surface Water Interactions

6.3.2.1 Temperature

River stage fluctuated significantly during the spawning period in all years of the study. For example, in 2003 the river level fluctuated daily over a 2–3 m range at site 64, which was representative of other sites (Figure 6.10). In general, there was very little relationship between river stage and hyporheic temperature during the spawning period. At some sites, there were small fluctuations in river temperature ($\sim 0.5^\circ\text{C}$) as river stage fluctuated (see example in Figure 6.10). In the case of site 64 in 2003, river temperature was more influential of hyporheic temperature than was river stage (Figure 6.10). This was true of most sites in all years. Similar results were observed during the winter months—that is, hyporheic temperature did not respond to fluctuations in river stage but was influenced primarily by river temperature.

18 October through 22 November

2001

15 November through 22 November

2001

2002

No data

2002

2003

2003

Figure 6.9. Vertical Hydraulic Gradient (note raw VHG is plotted in 2001). The medians of the distributions are shown as the center of the notches, and the lower and upper quartiles as the hinges of the box plots. Asterisks and circles represent outlier data points at 1.5 and 3.0 times the interquartile range, respectively. The notches represent an approximate 95% confidence interval around the median (McGill et al. 1978).

Table 6.8. Mean Survey Vertical Hydraulic Gradient by Site for 15–22 November Time Period (standard error in parentheses)

Site	2001	2002	2003	Overall Mean
54	0.67481 (0.01545)	0.01815 (0.01254)	ND ^(a)	0.34648 (0.01950)
59	ND	ND	−0.36632 (0.00483)	−0.36632 (0.00483)
61	ND	−0.04579 (0.00353)	0.04837 (0.00601)	−0.01834 (0.00401)
64	0.97070 (0.00113)	−0.00033 (0.00295)	−0.01545 (0.00294)	0.31831 (0.01929)
81	ND	0.00004 (0.00315)	−0.00602 (0.00409)	−0.00299 (0.00258)
90	0.83599 (0.00317)	−0.07110 (0.00241)	0.08984 (0.00226)	0.28677 (0.01688)
94	ND	0.09124 (0.00332)	0.17727 (0.00385)	0.13393 (0.00366)
96	−1.25563 (0.03665)	−0.72236 (0.03398)	0.11565 (0.07117)	−0.90564 (0.03040)
101	−0.14669 (0.00147)	0.04669 (0.00140)	ND	−0.05000 (0.00504)
(a) ND = no data.				

Table 6.9. Flux Calculations. These values are the cubic centimeters of water moving through the bed per week (192 hours or adjusted if entire period not available—site 90 all three years and site 96 in 2001).

Total Flux (cc)	Site	2001		2002		2003	
		Net	Abs.	Net	Abs.	Net	Abs.
Spawning	59					-1.66×10^5	1.66×10^5
	64	2.43×10^5	2.43×10^5	-8.22×10^1	8.37×10^3	-3.87×10^3	8.82×10^3
	81			4.22×10^1	3.45×10^4	-6.41×10^3	4.83×10^4
Non-Spawning	54	1.81×10^5	1.81×10^5	4.86×10^3	4.22×10^4		
	61			-2.91×10^4	3.34×10^4		
	90	3.48×10^5	3.48×10^5	-2.96×10^4	2.96×10^4	3.74×10^4	3.74×10^4
	96	3.73×10^4	3.73×10^4	-2.15×10^4	2.15×10^4		
	101	-2.25×10^4	2.25×10^4	7.16×10^3	7.18×10^3		
(a) ND = no data. Net is the volume of water moving into or out of the bed after the net effect of upwelling and downwelling is considered. Absolute movement ignores the direction of movement.

Figure 6.10. Temperature of Hyporheic Zone (red) and River (blue) Compared to River Stage (black) at Site 64 During the Spawning Period in 2003

Some of the largest fluctuations in hyporheic temperature occurred during periods of low river stage. For example, consider site 90 during the spawning period in 2001 (Figure 6.11, top panel) and 2004 (Figure 6.11, bottom panel). Total river discharge was lower in 2001 than in 2004, and hyporheic temperature tended to show a greater amount of variability. It is likely that during periods of low river stage, the sensors in the bed of the river at site 90 became dewatered or at least became more influenced by effects of solar input. Similar findings were observed at site 101 in those same years. Further, in 2004 when flows were high relative to the other three years, the largest fluctuations in temperature occurred at site 94, which had the lowest recorded water levels (due to piezometer placement). In fact, it is at site 94 where a number of the observations are truncated because of the low water.

6.3.2.2 Vertical Hydraulic Gradient

River stage fluctuated over a range of 2 m during the 5-day data analysis period at most sites in all years (Table 6.10). In some years at some sites (e.g., site 90 in 2002, 2003, and 2004; site 96 in 2004), the range in river stage fluctuations exceeded 3 m.

Site 96 always showed the greatest variability in delta VHG in response to changes in river stage (Figure 6.12), with site 54 showing the second largest range. Other sites, especially sites 61, 64, 90, and 101, showed only slight responses to changes in river stage (Figure 6.13). Sites 59 and 81 were more variable from year to year and were intermediate in their response, showing more response to changes in VHG than some sites (e.g., site 90) but still far less responsiveness than was seen at sites 54 and 96. Site 94 cannot be characterized because there were so many time periods when data were missing or incomplete.

Figure 6.11. Hyporheic (red) and River Temperature (blue) as Compared to River Stage (black) During the Spawning Period in 2001 (top panel) and 2004 (bottom panel) at Site 90

Sites 54 and 96 were the only two sites that consistently showed a strong cross-correlation between delta VHG and delta stage; average correlation coefficients for these two sites were approximately 0.80 and 0.94, respectively (Table 6.11). A few other sites (e.g., 61, 81, and 90) showed a very slight, but recognizable, response of delta VHG to changes in stage. The lags that showed the maximum correlation for site 54 and site 96 ranged from 0 to 60 minutes. For those values greater than zero, they indicated a tendency for the delta VHG to peak 30 to 60 minutes sooner than the peak in the delta stage curve. Detailed examination of the curves suggests that the response of the delta VHG to changes in stage was very rapid, within the temporal resolution of the data. As delta stage increased for site 54 and site 96, the delta VHG became increasingly negative; this led to the negative sign on the cross-correlation coefficients. For an increase in stage, the delta VHG increased only as long as the rate of increase in delta stage was positive. Once the inflection point of the delta stage curve passed (and the rate of increase of delta stage therefore started to decrease), the delta VHG curve reached its minimum and started to increase.

Table 6.10. Summary Statistics of River Stage During Five-Day Periods in the Fall During the Years 2001 through 2004. The five-day periods are described in the text.

Year	Site	Minimum (cm)	Maximum (cm)	Range (cm)
2001	54	27.475	230.625	203.15
	90	20	274.125	254.125
	96	11.465	254.24	242.775
	101	21.976	253.476	231.5
2002	54	85.329	289.25	203.92
	61	80.623	287.773	207.15
	64	91.727	320.927	229.2
	81	62.434	250.234	187.8
	90	87.358	388.286	300.927
	96	78.774	361.156	282.382
	101	92.453	352.904	260.451
2003	59	82.484	312.819	230.335
	61	76.123	288.923	212.8
	64	92.277	327.077	234.8
	81	60.434	253.384	192.95
	90	87.486	392.042	304.556
2004	59	73.731	316.223	242.492
	61	71.88	299.597	227.717
	81	55.239	261.164	205.925
	90	73.051	395.065	322.014
	96	76.185	380.22	304.035

River stage and VHG exhibited a hysteretic relationship at sites 81, 90, and 96 during a 25-hour period in October 2004 (see Figure 6.14). At midnight on 22 October (solid circle in each panel), the VHG values at sites 81 and 90 were slightly positive while at site 96 the VHG was very negative. A rising stage at sites 81 (200 to 240 cm) and 90 (250 to 360 cm) did not alter the VHG much, but rising stage at site 90 (240 to 360 cm) caused the VHG to become increasingly negative. When the stage reached the maximum height at the three sites, the VHG was the most negative at site 96. As the stage started to decrease, the VHG became increasingly positive at site 96 and changed little at sites 90 and 81.

Figure 6.12. An Example of the Times-Series Response of Delta VHG and Delta River Stage During a 5-Day Period in 2004 at a Site Characterized as Highly Variable (site 96)

Figure 6.13. An Example of the Time-Series Response of Delta VHG and Delta River Stage During a 5-Day Period in 2004 at a Site Characterized as Relatively Stable (site 90)

Table 6.11. The Time Lag Identifies the Shift Required for the Time Series of Delta River Stage Relative to Delta VHG To Achieve Maximum Correlation. For example, a 30-minute lag means the maximum correlation occurs for delta river stage values that are 30 minutes later than the time value for delta VHG. Statistical significance ($\alpha = 0.05$) indicated by italicized coefficient.

Year	Site	Lag (min)	Coefficient
2001	54	30	<i>-0.777</i>
	90	105	-0.471
	96	0	<i>-0.936</i>
	101	90	-0.110
2002	54	30	<i>-0.812</i>
	61	0	-0.467
	64	0	-0.153
	81	30	0.151
	90	0	-0.216
	96	0	<i>-0.936</i>
	101	-30	0.091
2003	59	60	0.292
	61	30	-0.433
	64	0	-0.086
	81	30	0.167
	90	0	-0.181
2004	81	120	0.359
		-90	-0.336
	90	-90	-0.375
	96	60	<i>-0.948</i>

6.3.3 Objective 3 – Hatching and Emergence Dates Using Bed and River Temperatures

Using river temperatures, the accumulation of 500 ATU (i.e., hatch) occurred 9–12 January 2002; 11–16 January 2003; and 1–11 February 2004 (Table 6.12). Using riverbed temperatures, hatch occurred 5–10 January 2002; 4–10 January 2003; and 15–23 January 2004 (Table 6.12). Hatch usually occurred earliest at Site 59 or 64 and latest at sites 96 or 101. The only significant differences in river temperature between sites within a given year occurred in 2003–2004 when there was approximately a 10-day difference in hatch just between sites. If riverbed temperatures were used instead of river temperatures, the range of hatch dates moved up 3 days in 2001–2002; 6 days in 2002–2003; and 18 days in 2003–2004 (Table 6.13).

Figure 6.14. The Relationship Between VHG and River Stage at Sites 81, 90, and 96 During a 25-Hour Period in 2004 (2400 hours on 22 October through 0100 hours on 23 October). The nature of the response appeared to be related to the hydraulic conductivity of the riverbed, which was highest at site 81 ($1.54 \times 10^{-2} \text{ cm/s}^{-1}$), intermediate at site 90 ($6.0 \times 10^{-3} \text{ cm/s}^{-1}$), and lowest at site 96 ($4.3 \times 10^{-4} \text{ cm/s}^{-1}$). Open circles indicate rising river stage; pink circles indicate falling stage. The solid black circle indicates the first reading (i.e., midnight on 22 October) while arrows indicate the passing of time.

Table 6.12. Dates of Accumulated Temperature Units First Exceeding 500 from 15 November Through 15 May (hatch)

Site	2001-2002 (182 days)		2002-2003 (182 days)		2003-2004 (183 days)	
	River	Riverbed	River	Riverbed	River	Riverbed
54		6 January 02	15 January 03	6 January 03		
59				4 January 03	1 February 04	15 January 04
61			11 January 03	7 January 03		19 January 04
64	9 January 02	6 January 02	12 January 03	4 January 03	3 February 04	15 January 04
81			13 January 03		6 February 04	
90	10 January 02	5 January 02	14 January 03	8 January 03	7 February 04	18 January 04
94			13 January 03	8 January 03	4 February 04	23 January 04
96	11 January 02	8 January 02	16 January 03	10 January 03	11 February 04	18 January 04
101	12 January 02	10 January 02	16 January 03	10 January 03		

Table 6.13. Dates of Accumulated Temperature Units First Exceeding 1,000 from 15 November Through 15 May (emergence)

Site	2001-2002 (182 days)		2002-2003 (182 days)		2003-2004 (183 days)	
	River	Riverbed	River	Riverbed	River	Riverbed
54		5 April 02	16 April 03	30 March 03		
59				21 March 03	30 April 04	10 April 04
61			14 April 03	1 April 03		18 April 04
64	18 April 02	5 April 02	13 April 03	3/29/03		16 April 04
81			14 April 03		2 May 04	
90	16 April 02	3 April 02	13 April 03	5 April 03	1 May 04	21 April 04
94			13 April 03	3 April 03	30 April 04	22 April 04
96	17 April 02	9 April 02		6 April 03	3 May 04	19 April 04
101	19 April 02	13 April 02				

The accumulation of 1,000 ATU (i.e., emergence) using river temperature occurred 16–19 April 2002; 13–16 April 2003; and 30 April through 3 May 2004 (Table 6.14). Using riverbed temperatures, emergence occurred 3–13 April 2002; 21 March to 6 April 2003; and 10–22 April 2004 (Table 6.14). Emergence normally occurred first at sites 64 or 90 and last at sites 96 or 101 (Figure 6.13). Emergence occurred an average of 10 days earlier using riverbed temperature instead of river temperature in 2002, and 13 days in 2003 and 2004 (Table 6.15). The greatest differences occurred at site 64 and 90 in 2002 (13 days earlier); 54 in 2003 (17 days earlier); and 59 (20 days earlier). The least amount of difference in emergence dates was at site 101 in 2002 (6 days); site 90 in 2003 (8 days); and site 94 in 2004 (8 days).

Table 6.14. Difference in Days Between Hatch (500 ATU) Calculated Using Temperatures from the River Versus the Riverbed

Site	2001-2002	2002-2003	2003-2004
54		9	
59			17
61		4	
64	3	8	19
81			
90	5	6	20
94		5	12
96	3	6	24
101	2	6	
Average	3	6	18

Table 6.15. Difference in Days Between Emergence (1,000 ATU) Calculated Using Temperatures from the River Versus the Riverbed

Site	2001-2002	2002-2003	2003-2004
54		17	
59			20
61		13	
64	13	15	
81			
90	13	8	10
94		10	8
96	8		14
101	6		
Average	10	13	13

Figure 6.15. Days to Emergence Calculated Based on 1,000 ATU for all Three Years for River and Hyporheic. The number of days to emergence (i.e., 1,000 ATU) was calculated using 15 November of each year as the spawn date. The number of days were calculated using the river temperatures as well as the hyporheic temperatures.

6.4 Discussion

6.4.1 Site to Site Comparisons of Temperature, Vertical Hydraulic Gradient, and Flux

Seasonal and annual variation in hyporheic temperatures was consistent; that is, seasonal lows occurred during the winter each year, and seasonal highs were in the summer. There was little difference between hyporheic and river temperature, with the hyporheic temperatures mimicking the river temperature at most sites. The average temperature difference during the study period was on the order of about 1°C and the largest differences were in July–August and February when the hyporheic zone was about 2°C cooler and warmer than the river, respectively. There were excursions in temperature differences between the hyporheic zone and the river at a few sites—the most notable differences were at site 59 and site 96. One explanation for these excursions in temperature is that the piezometers where these temperature sensors were placed may have been dewatered during periods of low river flow. As a result of this dewatering, humidity, wind speed, solar radiation and air temperature appeared to have a significant effect on temperature of the hyporheic zone and river.

There was no indication that vertical hydraulic gradient, an indication of potential upwelling, or flux of water into and out of the hyporheic zone differed between spawning and non-spawning sites. This was true when evaluated among individual sites, as well as when evaluated between the spawning and non-spawning reaches. Annual variation among sites was high, with no consistent pattern observed. Site 64 (a spawning area) consistently had the highest hyporheic temperature during the spawning period, but there was no indication that hyporheic temperatures in spawning and non-spawning sites were different, either during the entire spawning period or the one-week interval. Around the peak of fall Chinook salmon spawning activity in 2002 and 2003, hyporheic temperatures were 0.6 to 0.8°C warmer than the river at spawning sites as compared to about 0.3 to 0.5°C at non-spawning sites. These differences were statistically significant. In 2001, the differences between the hyporheic zone and the river were about 0.2°C for both spawning and non-spawning sites; these differences were not statistically significant.

The spawning period analysis of temperature and VHG was hindered because we did not have one consistent set of locations where we measured data year after year. There was only one spawning site (site 64) and two non-spawning sites (sites 90 and 96) where we had data from each of the three years. In

addition, there were only two years (2001 and 2003) in which data were available for the entire spawning period of 18 October through 22 November. Using the one week spawning period in November increased the amount of data available to 3 years. Average river and hyporheic temperatures were cooler in the one-week period at the end of the overall spawning period and there were wider variations in temperature early in the spawning period that were not captured by the one-week spawning period. However, median temperature patterns in spawning and non-spawning areas did not differ. When assessed over the entire spawning period, there were substantially more occurrences where the river temperature was warmer than the hyporheic zone than was observed during the one-week spawning period. Overall there was, however, little difference between the two periods in the pattern of temperature changes between the entire spawning period and the one-week period. Temperature differences between the river and hyporheic zone were, in general, the same direction and magnitude in both periods. VHG values were nearly identical. Therefore, using just the one-week spawning period where we have three seasons of data was justified.

Although statistical significance was found for temperature differences between the hyporheic zone and the river in 2002 and 2003, it is not known if these temperature differences are biologically meaningful. We defined biological significance as large enough to elicit a behavioral response in pre-spawning adult salmon that were searching for redd sites or in influencing hatch and emergence dates (see below). It would appear that temperature gradients more than 1°C would be necessary to elicit a behavior response. Chum salmon selected spawning sites in the Ives Island area of the Columbia River that had hyporheic temperatures that were 7 to 11°C warmer than the river (Geist et al. 2002). Bull trout spawning sites are often influenced by hyporheic or groundwater upwelling (Graham et al. 1981; Baxter and Hauer 2000), although groundwater temperatures in sites selected and not selected by bull trout at the time of spawning were no different in the Chowade River, British Columbia (Baxter and McPhail 1999).

There is little evidence that Chinook salmon use temperature cues originating in the hyporheic zone to select redd sites. Fall Chinook salmon in the Hanford Reach were previously found to select upwelling sites to spawn, but there were no temperature differences between the river and the hyporheic zone at these upwelling sites (Geist 2000). The large volume of the river, the turbulence along the riverbed that promotes mixing, and the relatively highly permeable sediments were suspected to be reasons why temperature gradients were not more substantial in the Hanford Reach. Geist (2000) hypothesized that perhaps there were olfactory cues in upwelling water that attracted adults to spawning sites, but found that temperatures of the hyporheic zone and the river were essentially the same. The lack of a temperature gradient does not preclude the diffusive flux through the bed sediments of other constituents that might provide an olfactory (or other) cue that salmon might respond to. Fall Chinook salmon selected spawning sites in Ives Island area of the Columbia where the hyporheic and river temperatures did not differ (Geist et al. 2002). Chum salmon also spawn at the Ives Island site, and did select areas where temperature differences were significant. The authors speculated that perhaps one reason for the species-specific differences in redd site selection was to avoid overlap in spawning habitat use. This does not explain our findings in the Hanford Reach as there are no other species that potentially compete for spawning sites with fall Chinook salmon. There was little difference in temperatures of the hyporheic zone at egg pocket depth (~30 cm) and the water column at fall Chinook salmon spawning sites in the Snake River although differences did increase with increasing depth into the riverbed (Hanrahan 2006).

It is likely that the high permeability of riverbed sediments in the Hanford Reach and other fall Chinook salmon spawning locations is the reason for the lower temperature and hydraulic gradients (see Section 5 for permeability information). River water is able to easily penetrate the substrate, which

immediately reduces the temperature gradients by equalizing temperature and hydraulic gradients by equalizing pressure. In riverbeds where permeability is reduced, these gradients form more easily and are expressed when measured. For example, no temperature gradient between the hyporheic zone and the river was found in an Atlantic salmon spawning stream where permeability was high, whereas in a degraded stream that no longer supports Atlantic salmon spawning and permeability was reduced, there was a temperature gradient of 2 to 3°C (Malcolm et al. 2002). Similar findings were reported in fall Chinook salmon spawning areas in the tailrace of Wanapum Dam (Geist et al. 2005). In areas where permeability was reduced (i.e., non-spawning areas), the river water was not able to penetrate as easily into the bed which resulted in short-term gradients in temperature and VHG. These gradients were not observed in spawning areas. Our assessment of changes in VHG in response to power peaking operations supports the finding that permeability of riverbed sediments is affecting the exchange of surface water into the bed (see Section 6.4.2).

6.4.2 Effect of Hydropower Operations on Groundwater–Surface Water Interactions

River stage fluctuated significantly during the spawning period with some daily excursions in river stage upwards of 3 m. These fluctuations in river stage were cyclical and a function of power peaking operations at Priest Rapids Dam. In Section 4 we described how accounting for these fluctuations did not improve our ability to predict fall Chinook salmon spawning habitat use.

The river stage fluctuations did not result in changes to the temperature of the hyporheic zone. In fact, as previously documented, water temperature of the hyporheic zone was more influenced by the river temperature. That river temperature responded to fluctuations in water level is noteworthy. It would appear that the change in water depth associated with changes in river flow are altering the relative input of solar radiation to the near-shore riparian environment and resulting in diurnal fluctuations in river temperature of ~1-2°C at locations where we had sensors. We do not know if this fluctuation in river temperature affects the biological productivity of the littoral habitats of the Hanford Reach. It may be a topic that deserves further study. However, the temperature fluctuations in both the river and the hyporheic zone were relative small. Therefore, our conclusion from analyzing the relationship between river stage and temperature is that there are too many extraneous factors other than riverbed permeability that have the potential to affect temperature of the hyporheic zone (e.g., air temperature, river stage, river temperature). This makes it difficult to use riverbed temperature as an indirect measure of sediment permeability in the Hanford Reach.

However, vertical hydraulic gradient did respond to changes in water level with the response dependent upon not only the water stage but on the riverbed permeability. At sites where permeability was relatively low, water level of the hyporheic zone showed a lag in response to changes in river stage. For example, at site 96 there was a highly significant negative correlation between delta river stage and delta VHG which had a lag time of zero (Figure 6.12, Table 6.11). This suggests that when the river stage increased, there was an immediate increase in the potential energy gradient between the river and the hyporheic zone because the river stage increased but the hyporheic zone did not. The response of the survey VHG to changes in stage at this site also showed this rapid increase in the rate of change. This response in survey VHG exhibited a hysteretic relationship when plotted over a 25-hour time period in 2004 (Figure 6.14). The high, negative correlation coefficient and the lagged, hysteretic response both suggest that the low sediment permeability at site 96 inhibited the movement of water into the bed of the river which set up an energy gradient between the bed and the river. In contrast to site 96, water level of

the hyporheic zone at sites where permeability was higher changed almost immediately in response to changes in river stage. With the rapid change in water level came a zeroing of the VHG because an energy gradient was never created, i.e., the hyporheic and river water levels were the same thus no energy gradient. At these sites, there were low correlations between delta river stage and delta VHG, and the hysteretic relationship was not as pronounced.

Arntzen et al. (2006) also found that permeability of riverbed sediments in the Hanford Reach affected the response of the VHG to changes in river stage. At sites where the permeability was high, the response of VHG to changes in river stage occurred very quickly. As such, when the VHG was plotted against river stage, the hysteric loop was flat (e.g., Figure 4A from Arntzen et al. [2006]). In contrast, when the riverbed sediments were lower in permeability, there was a delay in the response of VHG to changes in river stage. In this case, when the river stage was plotted against the VHG the hysteric loop was wide (e.g., Figure 4C). Arntzen et al. (2006) postulated that the hysteresis “was caused by a time lag between change in river stage and the pressure response of the hyporheic zone, and that the duration of the lag was a function of the magnitude of the river stage change and the hydraulic conductivity of the hyporheic zone.” Thus, the observations from this study during one week of the spawning period appear to corroborate the findings reported by Arntzen et al. (2006) for a single 25-hour time period. It would appear from the logger data that sediment permeability and river stage affect the relative flux of water into and out of the sediments; the flux in the potential energy gradient between the river and the hyporheic zone exhibits a hysteretic relationship in the response in VHG at sites where riverbed permeability is low; and there is a lag between river fluctuation and response in the riverbed.

6.4.3 Hatching and Emergence Dates Using Bed and River Temperatures

Hyporheic temperatures only slightly influenced the date of hatch or emergence over what would occur if river temperatures were used. The date of hatch was 3 to 18 days sooner if hyporheic temperatures were used rather than river temperatures. At emergence, the difference was about 10 to 13 days. These differences in timing are within the range of differences noted in an upstream-downstream direction just using river temperatures. Thus, there is no evidence to suggest that using hyporheic temperatures to estimate hatch and emergence of Hanford Reach fall Chinook salmon is any better than using river temperatures.

Other studies of salmonid spawning have noted similar results. Estimates of fall Chinook salmon hatch and emergence using temperatures of surface water and shallow (30 cm) hyporheic zone were within a few days of one another in a study on the Snake River (Hanrahan 2006). Predictions of brown trout (*Salmo trutta*) fry emergence were about 17 days earlier using hyporheic temperatures instead of river temperatures with large variability between sites using riverbed temperatures (Acornley 1999). Brown trout hatched about 10 to 11 days earlier using hyporheic temperatures instead of surface water temperatures; emergence timing was no different (Crisp 1990). Other studies of Atlantic salmon and brown trout have only found differences of a few days when using hyporheic and river temperatures (Clark et al. 1999; Malcolm et al. 2002).

7.0 Fall Chinook Salmon Redd Capacity in the Hanford Reach

Development of hydroelectric dams in the Columbia River Basin of the United States has contributed to the declining abundance of fall Chinook salmon (*Oncorhynchus tshawytscha*) through conversion of rivers to reservoirs and blocked access to historic spawning areas (Dauble et al. 2003). The decline of fall Chinook salmon has prompted management and regulatory agencies to consider actions directed at recovering lost salmon spawning areas, including dam removal, reservoir drawdown, and reintroduction into blocked historic habitat (Dauble et al. 2003; Hanrahan et al. 2004; Groves and Chandler 2005; Hanrahan et al. 2005). A key component to evaluating these actions is quantifying the salmon spawning habitat potential of a given river reach so that realistic recovery goals for salmon abundance can be developed (Independent Scientific Group 2000).

The ability to accurately predict the spawning habitat potential of current and blocked historic habitat is limited due to the prediction methods used and the lack of availability of historic data. The typical approach to salmon spawning habitat modeling uses depth, velocity, substrate, and channelbed slope characteristics to predict habitat potential, an approach that often overestimates the amount of available habitat (Geist and Dauble 1998). Historic records of spawning habitat use are limited in availability and are not of sufficient detail for predicting where salmon may spawn, should they be reintroduced into blocked habitat. Moreover, much of the historic salmon spawning habitat is now inundated by reservoirs, further limiting our ability to predict future habitat use based on current conditions. These limitations suggest the need for a means of predicting the quantity and location of salmon spawning areas based on riverine characteristics that can be easily quantified.

Fish biologists are increasingly recognizing the importance of geomorphic characteristics of rivers to salmon spawning habitat. However, little effort has been directed at incorporating quantitative measures of these characteristics into habitat modeling. Channel morphology influences the depth, velocity, substrate, channelbed slope, and the hydrologic exchange between the river and riverbed, and previous researchers have investigated the association of salmon spawning with specific geomorphic characteristics (Payne and Lapointe 1997; Geist and Dauble 1998; Montgomery et al. 1999; Dauble and Geist 2000; Moir et al. 2004). What these studies lacked was a quantification of the geomorphic characteristics of the river bed form at the pool-riffle scale to which salmon spawning was associated. For example, salmon spawning was identified to have occurred near geomorphic features such as islands and alternate bars and in 'response' type reaches, but identification of these features and reaches was qualitative (Geist and Dauble 1998; Moir et al. 2004).

The objective of this study was to estimate the fall Chinook salmon redd capacity in the Hanford Reach of the Columbia River using quantitative features of the river's geomorphic characteristics. In this section, we describe an approach for quantifying the channel morphology in the Hanford Reach and correlating the channel morphology with fall Chinook salmon spawning locations. Our intent is to elucidate the characteristics of geomorphically suitable fall Chinook salmon spawning habitat for purposes of restoration planning in the Columbia River and other large gravel-bed rivers.

7.1 Methods

7.1.1 Potential Spawning Habitat

Estimates of production potential were based on the number of fall Chinook salmon redds that could be constructed in potentially suitable spawning habitat. Two different estimates of spawning habitat were used in the analysis. The first estimate was based on the existing spawning habitat used within the study area. Fall Chinook salmon spawning areas have been monitored annually since 1948 using aerial observation of spawning activity (Dauble and Watson 1997; also see Section 2 of this report) as well as through the use of aerial photography in 1994 and 1995 (Visser et al. 2000, 2002). All spawning areas were mapped and incorporated into a GIS database as unique polygons (hereafter termed “existing habitat”). Spawning activity within these areas has remained consistent since 1948 (Dauble and Watson 1997; R. P. Mueller, PNNL, unpublished data).

The second estimate of potential fall Chinook salmon spawning habitat within the study area was based on an analysis of the channel morphology associated with existing spawning areas in the Hanford Reach. This estimate of potential spawning habitat (hereafter termed *expansion habitat*) is based on the assumption that the existing spawning areas do not represent the entire potential habitat available—that is, all potential habitat in the Reach is not fully seeded. The potential expansion habitat was identified by 1) quantifying the channel morphology of existing spawning areas and 2) locating additional areas that contained geomorphic characteristics similar to those of the existing spawning areas. The expansion habitat is considered geomorphically suitable habitat.

Quantifying the channel morphology of existing spawning areas involved identifying the longitudinal and lateral bed forms where fall Chinook salmon spawning occurs. The analysis of the longitudinal bed form profile was completed using sequential bed elevation data from the thalweg of the river based on bathymetric data collected using LIDAR and SHOALS surveys (see Section 2 of this report). Thalweg points from each of the 331 cross sections were used in a bed form differencing technique to identify pools and riffles (O’Neill and Abrahams 1984). Application of the technique resulted in identification of the thalweg points that were either riffle crests or pool bottoms. After riffle crests and pool bottoms were identified, two additional analyses were completed to classify where the remaining thalweg points were located relative to the riffle crests and pool bottoms. First, the thalweg points were determined to be in riffles or pools based on their riffle proximity index (*RPI*):

$$RPI = 1 - \frac{rcelev - tpelev}{rcelev - pbelev} \quad (7.1)$$

where *rcelev* = nearest riffle crest elevation, *tpelev* = thalweg point elevation, and *pbelev* = nearest pool bottom elevation (Hanrahan 2006). The *RPI* ranges from 0.0 to 1.0, where thalweg points with a *RPI* greater than 0.50 were categorized as being in riffles, while the remaining points were categorized as being in pools. Second, all thalweg points were categorized as being located in one of four areas along the longitudinal profile: 1) upstream side of riffle crests, 2) downstream side of riffle crests, 3) upstream side of pool bottoms, and 4) downstream side of pool bottoms.

An analysis of the lateral channelbed profile of each cross section within fall Chinook salmon spawning areas was completed to identify where along a cross section spawning occurred relative to the channel thalweg. The analysis was based on the channelbed elevation of individual stations spaced 3 m apart along each spawning cross section (see Section 2 for description of bathymetric data). The relationship between individual stations and the thalweg point on each cross section was determined by the thalweg distance index (*TDI*) for each station:

$$TDI = 1 - \frac{maxelev - selev}{maxelev - tpelev} \quad (7.2)$$

where *maxelev* = maximum bed elevation at a discharge of 1982 m³ s⁻¹ (approximate 50% hourly exceedence discharge during the spawning interval 15 October to 15 November, based on data from USGS gage 12472800 for the years 1994 through 2001); *selev* = station bed elevation; and *tpelev* = thalweg point elevation. For each station, the *TDI* ranges from 0.0 (nearest in elevation to the thalweg) to 1.0 (farthest from the thalweg in elevation). Station *TDI* values were analyzed separately for points located on riffle cross sections (*RPI* > 0.5) and pool cross sections (*RPI* ≤ 0.5). The analyses included calculation of summary statistics and ordinary least-squares regression of station *TDI* as a function of cross-section *RPI*.

Expansion habitat was identified by locating regions of the study area that contained geomorphic characteristics similar to those of the existing spawning areas. Quantification of the expansion habitat was based on 92 non-spawning cross sections characterized as *Fd*1* or *Fd*2* (see Section 2 of this report). These cross sections likely have the greatest potential to contain spawning habitat, as all of the existing spawning areas occur in *Fd*1* or *Fd*2* cross sections. Most of the cross sections (n = 61) considered non-spawning were those that did not intersect a spawning area across the entire width of the channel, including the width of any channels branching from the main channel. Where the main channel splits and each branch channel conveys approximately the same discharge (river kilometer 600–605; see Figure 2.4), portions of cross sections (n = 31) were considered non-spawning if they did not intersect a spawning area in one of the branch channels.

Applying *TDI* criteria from the spawning areas refined locations of potential expansion habitat within each of the 92 non-spawning cross sections. The *TDI* for all stations on the 92 non-spawning cross sections was calculated. Stations were considered potential expansion habitat if their *TDI* values were within the *TDI* criteria range from spawning areas (Table 7.1). The *TDI* criteria from spawning areas differed for stations located along pool (*RPI* ≤ 0.5) and riffle (*RPI* > 0.5) cross sections.

Stations meeting the *TDI* criteria for potential expansion habitat were imported as a point dataset into the GIS database. The potential expansion habitat points on each cross section were used as a background for depicting the spatial extent of the expansion habitat. The spatial extent of the expansion habitat was then digitized onscreen as a set of polygons. The estimated lateral extent of the expansion habitat polygons was limited to the lateral extent of the expansion habitat points. The estimated longitudinal extent of the expansion habitat polygons was the contiguous area between cross sections containing expansion habitat points plus the area one-half the distance to the nearest cross section not containing expansion habitat points.

Table 7.1. Thalweg Distance Index (*TDI*) Criteria from Stations Located Along Fall Chinook Salmon Spawning Cross Sections in Pools and Riffles. Stations located on non-spawning cross sections were considered potential expansion habitat if their *TDI* values were within the *TDI* criteria range. Different criteria were used for pools and riffles because *TDI* and *RPI* were significantly correlated ($P = 0.0001\text{--}0.0006$, $r^2 = 0.44\text{--}0.58$) for the former but not the latter ($P = 0.37\text{--}0.74$, $r^2 = 0.001\text{--}0.01$).

Bed Form Type	TDI Criteria
Pools	$TDI = (0.80122 - 0.9773(RPI)) \pm 0.125^{(a)}$
	$TDI = (0.79368 - 1.128(RPI)) \pm 0.125^{(b)}$
Riffles	0.1–0.8
(a) Applied to all stations in the main channel and to stations in the left bank channel at rkm 600–605.	
(b) Applied to stations in the right bank channel at rkm 600–605.	

7.1.2 Redd Capacity

Observations from studies in the Columbia River indicate that spawning fall Chinook salmon use a relatively small proportion of seemingly suitable habitat within a contiguous area (Swan 1989; Dauble and Watson 1990; Dauble and Watson 1997; Geist and Dauble 1998; Geist et al. 2000; Visser 2000). For example, a study of deep-water spawning in the Hanford Reach of the Columbia River indicated that spawning fall Chinook salmon used 13–34% (mean = 22%) of suitable habitat within the study areas (Swan 1989). At one primary spawning area in the Hanford Reach, Geist et al. (1997) reported 3–10% utilization of suitable habitat for two years of study. Habitat modeling using depth, velocity, and substrate variables for two spawning areas in the Hanford Reach predicted that 50% of the areas were suitable spawning habitat, while actual use was 0% in one area and 20% in the other (Geist et al. 2000). Finally, a study of nine concentrated spawning areas (redd clusters) in the Hanford Reach during our study resulted in an average use of 15.4% one year and 12.6% the following year (Visser 2000). Indeed, research from the Hanford Reach of the Columbia River indicates that fall Chinook salmon spawn in distinct clusters within areas predicted by habitat models or other methods to be suitable (Dauble and Watson 1997; Geist et al. 2000; Visser 2000; Visser et al. 2002). Without accounting for proportional use of potentially suitable habitat, it is likely that available habitat will be overestimated, which may lead to unattainable production goals (Geist and Dauble 1998). Therefore, the amount of potentially suitable habitat described above (i.e., existing spawning habitat used and expansion habitat) was adjusted by a range of percentages, providing lower and upper estimates of reduced suitable area based on proportional use. Specifically, the reduced suitable area was calculated as 5–30% (in increments of 5%) of the original potential suitable area, reflecting the range of proportional use reported in the studies discussed above.

Redd capacity was defined as the total number of redds required to fill a given area of suitable spawning habitat. Two different methods were used to estimate redd capacity. The first method used the average size of a fall Chinook salmon redd divided into the amount of suitable spawning habitat. The average size of a Hanford Reach fall Chinook salmon averaged 17 m² in a study at Vernita Bar done by Chapman et al. (1986) and is the value we used in our analysis. This method of calculating redd capacity results in estimates on the upper end of the range as it does not account for inter-redd spacing. The second method for estimating redd capacity was based on the average area used by a redd, including inter-redd spacing, divided into the amount of suitable spawning habitat. Inter-redd distances for Hanford

Reach fall Chinook salmon averaged 2.8 m in 1994 (Visser 2000). This inter-redd distance was applied to the average redd sizes, resulting in an average inter-redd area of approximately 45 m².

Both methods of estimating redd capacity were applied to the existing and expansion habitat scenarios. The total redd capacity estimated for the study area was calculated based on the reduced suitable spawning area (calculated as 5–30% [in increments of 5%] of the original potential suitable area), reflecting the range of proportional use reported in the studies discussed above.

7.2 Results

7.2.1 Existing Spawning Habitat

The quantity of existing fall Chinook salmon spawning habitat within the study area is approximately 164 hectares (ha) (Figure 7.1). This is the total area that encompasses the existing spawning habitat. The analysis of redd capacity within the existing spawning habitat resulted in a large range of redd numbers, depending on the calculation method used. Applying only the average redd size of 17 m² (i.e., no inter-redd spacing, high concentrated spawning) to the reduced spawning habitat area (i.e., the habitat reduced 5–30% based on proportional use) resulted in estimates ranging from 4,824 to 28,941 redds required to fill the given habitat area (Table 7.2). This estimate is reduced to 1,822–10,933 redds when accounting for the average area used for a redd, including inter-redd spacing (Table 7.2).

Results from the analyses of all thalweg points indicated that existing spawning areas were more likely to occur near riffle crests than elsewhere along the longitudinal profile. The median riffle proximity index (*RPI*) for spawning areas was larger (0.80) and less variable (interquartile range [IQR] = 0.31) than the *RPI* for non-spawning areas (median = 0.55, IQR = 0.36; Figure 7.2). Results from the Mann-Whitney *U* test indicated that the mean *RPI* in spawning areas was significantly larger than the mean *RPI* in non-spawning areas ($z = 6.64$, $P < 0.001$; Figure 7.3).

Results from the cross-tabulation analysis indicated that existing spawning habitat use was dependent on bed form type ($\chi^2 = 21.7 = 3$, $P < 0.001$). The upstream and downstream sides of riffle crests contained 26% and 55% of the spawning habitat use, respectively (Table 7.3). The remaining spawning habitat use occurred on the upstream (10%) and downstream (9%) ends of pools.

For existing fall Chinook salmon spawning areas located along riffle cross sections ($RPI > 0.5$), the mean thalweg distance index (*TDI*) was not correlated with *RPI* ($P = 0.37 - 0.74$, $r^2 = 0.001 - 0.01$; Figure 7.4). The mean *TDI* for fall Chinook salmon spawning areas located along pool cross sections ($RPI \leq 0.5$) was significantly correlated with *RPI* ($P = 0.0001 - 0.0006$, $r^2 = 0.44 - 0.58$; Figure 7.5). These data suggest that fall Chinook salmon spawning areas associated with deep pool bed forms along the longitudinal profile (*RPI* approaching 0.0) are located in the upper elevation bar features within those pools (*TDI* approaching 1.0).

7.2.2 Expansion Spawning Habitat

The quantity of geomorphically suitable potential expansion fall Chinook salmon spawning habitat within the study area is approximately 317 ha more habitat than the existing spawning habitat (Figure 7.1). The analysis of redd capacity based on the potential expansion spawning habitat resulted in a large range of redd numbers similar to that found for the existing spawning habitat. Applying the 17-m²

Figure 7.1. Fall Chinook Salmon Spawning Habitat Within the Study Area. Existing habitat encompasses approximately 164 ha while potential expansion habitat includes approximately 317 ha.

Table 7.2. Redd Capacity of Existing Fall Chinook Salmon Spawning Areas Based on Redd Sizes and Inter-Redd Spacing. Redd capacity is defined as the total number of redds that could fill the existing spawning habitat area (164 ha), reduced by 5–30% to reflect proportional use.

	Percentage Use of Total Suitable Area (ha)					
	5% (8.2)	10% (16.4)	15% (24.5)	20% (32.7)	25% (40.9)	30% (49.1)
Redd Size (m²)						
17	4,824	9,647	14,471	19,294	24,118	28,941
Redd Size Including Inter-Redd Spacing (m²)						
45	1,822	3,644	5,467	7,289	9,111	10,933

Figure 7.2. Riffle Proximity Index *RPI* for Existing Fall Chinook Salmon Spawning and Non-Spawning Locations. The boxplots indicate the median (small box), surrounded by the 25th and 75th percentiles (large box) and extend to the 10th and 90th percentiles (whiskers) of observed values.

Figure 7.3. Estimates of the Mean Riffle Proximity Index *RPI* Were Significantly Larger in Existing Fall Chinook Salmon Spawning Locations Than in Non-Spawning Locations ($P < 0.001$). The boxplots indicate the mean (small box), surrounded by the standard error of the mean (large box), and extend to the 95% confidence interval (whiskers).

Table 7.3. Summary Frequency Table of Four Bed Form Types in Existing Fall Chinook Salmon Spawning and Non-Spawning Areas. Points along the longitudinal profile were categorized as being on the upstream (proximal) or downstream (distal) side of a riffle crest or pool bottom. The data included all thalweg points in spawning areas ($n = 107$) and a random sample of thalweg points in non-spawning areas ($n = 110$).

	Riffle Proximal	Riffle Distal	Pool Proximal	Pool Distal
Spawning				
Count	28	59	11	9
Frequency	26%	55%	10%	9%
Non-Spawning				
Count	15	42	27	26
Frequency	14%	38%	24%	24%

Figure 7.4. The Mean Thalweg Distance Index TDI of Existing Fall Chinook Salmon Spawning Areas Located Along Riffle Cross Sections ($RPI > 0.5$) Was Not Significantly Correlated with the Riffle Proximity Index RPI . The solid line represents a linear fit between RPI and TDI while the dashed line indicates the 95% confidence interval.

Figure 7.5. The Mean Thalweg Distance Index TDI of Existing Fall Chinook Salmon Spawning Areas Located Along Pool Cross Sections ($RPI \leq 0.5$) Was Significantly Correlated with the Riffle Proximity Index RPI . The solid line represents a linear fit between RPI and TDI while the dashed line indicates the 95% confidence interval.

average redd size (i.e., no inter-redd spacing, high-concentrated spawning) to the reduced expansion spawning habitat area resulted in estimates ranging from 9,324 to 55,941 redds required to fill the given habitat area (Table 7.4). That estimate is reduced to 3,522–21,133 redds when accounting for the average area used for a redd, including inter-redd spacing (Table 7.4).

Table 7.4. Redd Capacity of Expanded Fall Chinook Salmon Spawning Areas Based on Redd Sizes and Inter-Redd Spacing. Redd capacity is defined as the total number of redds that could fill a given area of spawning habitat, reduced by 5–30% to reflect proportional use.

	Percentage Use of Total Suitable Area (ha)					
	5% (15.8)	10% (31.7)	15% (47.5)	20% (63.4)	25% (79.2)	30% (95.0)
Redd Size (m²)						
17	9,324	18,647	27,971	37,294	46,618	55,941
Redd Size Including Inter-Redd Spacing (m²)						
45	3,522	7,044	10,567	14,089	17,611	21,133

7.3 Discussion

The results of fall Chinook salmon redd capacity estimates for existing habitat used within the study area provide a realistic metric of actual spawning activity. Based on aerial visual surveys for the years 1996 through 2005, annual redd counts within the study area have averaged 4,289, ranging from 3,178 to 5,396 (R. P. Mueller, PNNL, unpublished data). Visser et al. (2002) found that aerial photography improved the accuracy of aerial redd counts in the Hanford Reach, and estimates of redd abundance based on aerial photographs were about twice as high as estimates based on visual surveys. Thus, a more accurate estimate of annual average redd abundance within our study area during the years 1996 through 2005 would be 8,577 (range 6,356 to 10,792). These redd counts fall within the range of the redd capacity estimate results (1,822–10,933 redds) when accounting for the average area used for a redd plus the inter-redd spacing.

Our estimate of redd capacity within existing habitat was highly variable and depended upon both the value we selected for the percentage of the total potential area that was used and the average size of the redds. Based on a review of previous studies conducted in the Hanford Reach, we selected proportional use values that ranged between 5 and 30% (Geist et al. 1997, 2000; Visser 2000; Visser et al. 2002). The midpoint of the proportional use values from these studies was about 15% and was closest to a study by Visser (2000) in the same river reach where we conducted this study. Visser found that between about 16 and 18% of the area within spawning polygons were actually disturbed by fall Chinook salmon. The area required for pairs of spawners also highly influenced the redd capacity estimates, with small differences in redd area resulting in large differences in the results. We used 17 m² as our estimate of redd size based on empirical data collected from fall Chinook salmon redds in the Hanford Reach (Chapman et al. 1986). This decision was based on the fact that Chapman's work was on the same stock of fish we were studying, and they included the tailspill in the estimate of redd size. However, we acknowledge that fall Chinook salmon likely require a larger area in which to spawn than simply the individual size of their redd. Burner (1951) suggested that individual pairs of spawners may require up to four times the size of individual redds. This seems excessive and not consistent with our observations of fall Chinook salmon spawning in the Reach. For example, we noted in a previous Hanford Reach study that fall Chinook

salmon exhibited a uniform distribution at inter-redd distances of 2 to 5 m, suggesting this is the space required between individual spawners (Geist et al. 2000). Visser (2000) reported inter-redd distances of Hanford Reach fall Chinook of 2.8 m, which is the value we used to calculate a redd area of 45 m² that accounts for inter-redd space requirements. Space requirements for individual spawners likely change based on escapement and habitat conditions, so the use of a range of redd sizes would appear to be a prudent choice to define redd capacity.

Estimates of geomorphically suitable fall Chinook salmon spawning habitat within the study area suggest that there could be nearly twice as much potential expansion spawning habitat than the habitat currently used. Much of the predicted geomorphically suitable spawning habitat is close in spatial proximity to existing spawning areas, suggesting a high probability that the expansion areas could be used, especially during years of high escapement. Indeed, during recent high escapement years of 1994 (48,857) and 2001 (44,140), aerial photographs documented that fall Chinook salmon redds were located in the geomorphically suitable expansion habitat. However, about one third of the expanded habitat was downstream of rkm 580, which is the lower extent of current spawning within our study area. Results from our investigation of sediment grainsize and permeability determined that this reach was lower in permeability and possessed more fine silts and coarse sands than the spawning reach. The fine-grained sediments affecting the riverbed in this reach derive from near-channel geologic features, some of which are recent and active landslides (Triangle Associates 2003). Because our simple geomorphic model of channel morphology did not take into consideration substrate quality, it is likely that fine-grained sediments significantly affect the habitat quality of this reach. This is why we used an adjustment factor to correct downward the amount of expanded habitat within the Reach (i.e., 5% of total, or 3,522 redds). Even with this adjustment factor, it is likely that not all of the expanded habitat is useable because the analysis did not consider all the factors that affect habitat suitability, including microhabitat conditions that limit where fall Chinook salmon spawn (see Sections 4 through 6 of this report).

The total redd capacity for the study area (existing and expanded habitat) using a proportional reduction of 15% for existing habitat (24.6 ha) and 5% for expanded habitat (15.9 ha) and accounting for a range in redd sizes (17 to 45 m²) would result in an average redd capacity estimate of 16,392 redds (range 8,989–23,794). This results in a redd density of approximately 34 redds/ha (range 19 to 49 redds/ha), which is a conservative estimate compared to previous studies in the Reach. For example, mean fall Chinook redd density in a study in 1994 and 1995 within the same study area as ours ranged from 54 to 73 redds/ha (Visser et al. 2002). Not surprisingly, redd density was related to escapement, with low escapement resulting in low redd densities. It is likely that our use of a wider range of escapement values reduced the density below that reported by Visser.

Dauble and Watson (1990) suggested that, on average, visual surveys detected one redd for every nine adult fall Chinook salmon that returned to the entire Hanford Reach. Visser et al. (2000) suggested that by using aerial photography, this ratio would be about one redd for every 3.4 adults, which is likely a more reasonable value for our study. Therefore, using a ratio of 3.4 adults per one redd results in an average fall Chinook salmon escapement estimate within our study area of about 55,731 adults (range 30,562–80,900). Approximately 40% of the spawning occurs outside of our study area (Dauble and Watson 1997; R. P. Mueller, PNNL, unpublished data). Therefore, an estimate of adult escapement for the entire Hanford Reach, using redd capacity estimates for the existing and expanded habitat, would be an average of approximately 92,885 (range 50,937–134,833). For comparison, average escapement to the Reach from 1990–2005 was 51,062 (range 27,012–89,312; Hoffarth 2006). Thus, our estimate of adult escapement using redd capacity appears to be a reasonable estimate.

The results indicate that most fall Chinook salmon spawning occurs in riffles and at a vertical location within 80% of the riffle crest elevation. This location corresponds to the transitional area between pools and riffles, an important habitat area where many salmon species tend to prefer to spawn (Bjornn and Reiser 1991). Owing to localized variations in sediment transport rates, these transitional areas are depositional zones that are often associated with the formation of bars (lateral, midchannel, point) and islands. Previous researchers have observed salmon spawning locations to be associated with depositional 'response' reaches (Montgomery et al. 1999; Moir et al. 2004), especially near the upstream end of bars and islands (Dauble and Watson 1997; Geist and Dauble 1998; Coulombre-Pontbriand and Lapointe 2004). The results reported here corroborate these earlier findings and quantify the location along the longitudinal profile where Chinook salmon spawning occurs. By quantifying the two-dimensional location (vertical and longitudinal) of salmon spawning areas, these findings complement previous planform mapping of fall Chinook salmon spawning areas. Dauble and Geist (2000) reported that Chinook salmon spawning in the Hanford Reach of the Columbia River and the Hells Canyon Reach of the Snake River was associated with the presence of bars that were mapped along 1.6-km river segments. Based on planform mapping of morphological channel types (bar, fan, glide, pool, rapid, riffle), Groves and Chandler (2002) observed that most fall Chinook salmon spawning in the Snake River occurred in riffles. Hanrahan (2006) quantified the relationship between Snake River fall Chinook salmon spawning and channel morphology, reporting that 84% of the spawning occurred in riffles and at a vertical location within 80% of the riffle crest elevation.

Our approach to estimating fall Chinook salmon redd capacity in the Hanford Reach used information on the geomorphic characteristics of river channels to estimate redd capacity in a large gravel-bedded river. This approach differs from previous geomorphic and salmon habitat investigations in the Hanford Reach in that it quantifies the relationship between redd numbers and geomorphic features. The use of relatively simple metrics enabled us to see how channel form affects spawning habitat and ultimately redd densities and adult escapement. A similar approach in other gravel-bedded rivers where fall Chinook salmon spawn may shed light on the factors limiting their successful recovery.

About a third of the expanded habitat area is in the non-spawning segment downstream of the 100-F slough. Our simple geomorphic model of channel morphology did not take into consideration substrate quality, although as we reported in Section 5 of the report, the substrate quality in this section of the river is of lower quality than within the spawning segment. It is unlikely that sediment permeability within this lower segment is sufficiently high enough to support egg incubation, although we have not directly measured this. Literature based estimates of salmon egg survival to emergence were inconclusive and highly variable. Results from Swan Falls suggest that elevated silt content would reduce incubation success. Thus, we believe that the expanded habitat estimate for the area below 100-F is likely overly optimistic which is why we reduced our area estimate by 95%. In reality, the redd capacity estimate is lower than what we found.

8.0 Protocol for Evaluating Fall Chinook Salmon Spawning Habitat

Rivers can be thought of as hierarchically organized geomorphic features arranged predictably within a watershed (Frissell et al. 1986; Schlosser and Angermeier 1995; Imhof et al. 1996). At progressively higher levels of organization, large rivers incorporate a variety of spatial scales. Although most habitat assessments focus on channel or site characteristics, it is well known that habitat hierarchies are spatially nested—that is, a geomorphic feature at one level affects the form and function of features at a lower level (Frissell et al. 1986; Grant et al. 1990; Gregory et al. 1991). Because rivers can be viewed as hierarchically organized geomorphic features (watershed- to site-scale), assessments of fish habitat quantity and quality should be similarly evaluated in a hierarchical fashion. Failure to do so may culminate in misleading results when extrapolating data from one scale to estimate habitat quantity or quality at another scale. For example, lateral bars studied at the site scale in the Hells Canyon Reach of the Snake River contained high-quality spawning habitat for fall Chinook salmon (*Oncorhynchus tshawytscha*) (Arntzen et al. 2001; Hanrahan 2006). However, the size, number, and spatial connectivity of these bars are very limited because of the geomorphology of Hells Canyon. In the absence of the larger geomorphic context, conclusions regarding fall Chinook salmon spawning habitat in the entire Hells Canyon Reach may be misinterpreted.

In a recent study of the impacts of hydroelectric development in the Columbia River Basin on main-stem salmonid habitats and riverine processes, Dauble et al. (2003) completed an assessment ranging from the watershed scale to the reach scale. Spawning locations for fall Chinook salmon have been well documented during most of the hydroelectric development period on the Columbia River. This information, in addition to surveys conducted after all mainstem hydroelectric projects were completed, provided a means to evaluate the relationship of various geomorphic features to spawning locations (Dauble et al. 2003). Their spawning habitat model was based on longitudinal gradient ($<0.05\%$), geology ($>50\%$ unconsolidated), and the presence of bars and islands. These features were found in spawning reaches that contained spawning densities greater than 10 redds per kilometer (Dauble et al. 2003). When the geomorphic model was applied to the entire Columbia River Basin, it predicted that there were approximately 283 and 311 km of suitable spawning habitat for fall Chinook salmon in the mainstem Columbia and Snake rivers, respectively, during the pre-development period. This distance is about 40 to 50% less than what was documented as historic spawning habitat. When the model was applied to specific river reaches containing the highest known redd densities, it predicted that only 20 to 66% of the total habitat available for spawning was suitable spawning habitat. This analysis suggests that only a portion of river reaches where spawning was thought to occur were actually used.

8.1 Protocol

Fall Chinook salmon spawning habitat quantity and quality should be evaluated in a hierarchical fashion to minimize errors associated with scale dependence. The level of detail required for each spatial scale of inquiry should be dictated by the management and research questions being addressed. As a protocol for evaluating fall Chinook salmon spawning habitat, we envision an approach that begins at the watershed scale and becomes increasingly detailed at progressively finer scales, culminating in estimates of habitat quality at the scale of individual redds.

8.1.1 Watershed Scale

At the watershed scale, the most important physical features that ultimately affect the spawning habitat of fall Chinook salmon are the longitudinal slope and flow regimes (Dauble and Geist 2000). This is because the longitudinal slope of a river reflects the long-term geological development of a watershed, and under conditions of uniform discharge, a direct relationship exists between slope and bed material particle size (Richards 1982). In unconstrained reaches of large gravel- and cobble-bed rivers, the longitudinal slope is reduced and alluvium is deposited (Knighton 1998). This alluvium is highly porous, allowing river water to penetrate into the bed material (hyporheic habitats), creating conditions beneficial to spawning and egg incubation. Thus, geologic formations affect longitudinal slope, which along with the flow regime, affect the quality of spawning habitat (i.e., substrate, velocity profiles, and other features) available at the site scale.

Assessments of fall Chinook salmon spawning habitat potential at the watershed scale should be based on the controlling factors (e.g., longitudinal slope, flow regime, geology, and channel morphology) associated with historic and/or contemporary spawning habitat. A good example of this type of approach is that of Dauble et al. (2003), which included a geomorphic model of fall Chinook salmon spawning habitat. The result of such an approach is the identification and quantification of large river segments possessing the greatest potential for containing fall Chinook salmon spawning habitat.

8.1.2 Reach Scale

At the reach scale, the most important physical features that ultimately affect the spawning habitat of fall Chinook salmon are channel morphology and associated hydraulic features. This is because channels that are capable of bedload sediment transport result in the development of lateral and point bars (Church and Jones 1982). Salmonid spawning usually occurs at the transition between pools to riffles (Bjornn and Reiser 1991), which are areas often associated with a lateral bar deposition area (Church and Jones 1982). Downwelling and upwelling of hyporheic flow occurs at the upstream and downstream portions of a channel bar or island creating interstitial flow pathways through the bed material (Brunke and Gonser 1997). The extent of this upwelling and downwelling is often a function of the channel morphology and flow characteristics (Harvey and Bencala 1993).

Examples of reach features in large, alluvial rivers include gravel bars and islands that are longer than one channel width in length. The location and morphology of these features affect specific hydraulic features of the spawning habitat at the channel unit scale or pool/riffle system (i.e., distinct hydraulic and geomorphic structures with characteristic bed topography, water surface slope, depth, and velocity patterns) and those at the subunit or microhabitat scale (i.e., transitory hydraulic features within a channel unit that have homogeneous substrate type, water depth, and velocity).

Assessments of fall Chinook salmon spawning habitat potential at the reach scale should be based on the channel morphology associated with historic and/or contemporary spawning habitat. Ideally this evaluation would involve quantifying the morphological features of the spawning habitat. Channel morphology can be quantified in numerous ways, including cross-section hydraulic geometry, longitudinal riverbed elevation profiles, mapping of morphological units (e.g., pools, riffles), and classification schemes based on channel shape, slope, patterns, material, and other physical characteristics.

The objective of assessing fall Chinook salmon spawning habitat potential at the reach scale is to refine the estimate of potential habitat derived from the watershed-scale analysis. Examples of this type of approach include the geomorphic classification of fall Chinook salmon spawning areas described in Section 2 of this report and the quantification of bed forms associated with fall Chinook salmon spawning areas as described in Section 7 of this report and in Hanrahan (2006). The quantitative association between bed forms and fall Chinook salmon spawning (Section 7; Hanrahan 2006) provides an important context for estimating the spawning habitat potential of Hanford Reach and Snake River fall Chinook salmon populations. The objective identification and quantification of stage-independent pool-riffle bed forms will result in more accurate estimates of salmon spawning habitat potential, as microhabitat characteristics (e.g., depth, velocity, substrate size) in salmon spawning areas encompass such a large range that their predictive potential is often limited. For example, incorporating the quantitative association between salmon spawning habitat and riffles into habitat modeling will improve the predictive potential of such models by providing a quantitative means of identifying the suitable microhabitat located in riffles and not elsewhere along the longitudinal profile. Such an approach is not possible through visual planform mapping of pools and riffles, as the identified geomorphic units are stage-dependent, subject to individual interpretation, and provide no quantitative metric that can be incorporated into habitat models.

Results from previous research have suggested that geomorphic characteristics at the pool-riffle scale may also be important controlling factors of other microhabitat variables (e.g., substrate quality, hyporheic exchange, hydraulic habitat complexity) important to spawning salmon and other fish (Geist and Dauble 1998; Baxter and Hauer 2000; Geist 2000; Moir et al. 2002; Coulombre-Pontbriand and Lapointe 2004; Crowder and Diplas 2006). Incorporating the quantitative linkages between bed forms and these other microhabitat variables when predicting habitat potential will also result in more accurate estimates of suitable habitat. Improvements to the predictive potential of habitat models will help management and regulatory agencies in the Columbia River Basin and elsewhere to develop realistic recovery goals for salmon abundance.

8.1.3 Site Scale

The objective of assessing fall Chinook salmon spawning habitat potential at the site scale is to refine the estimate of potential habitat derived from the reach-scale analysis. The refinement is necessary to improve predictions of habitat quantity and quality. For example, previous sections of this report documented that fall Chinook salmon spawning locations in the Hanford Reach are dependent on the water velocity and depth (Section 4). However, some areas predicted to be spawning habitat based on their velocity and depth are not actually used (Section 4). Therefore, it was necessary to evaluate other physical variables concerning habitat quality, such as substrate grainsize and riverbed permeability, that may limit the quantity of fall Chinook salmon spawning habitat (Section 5).

Assessments of fall Chinook salmon spawning habitat potential at the site scale (including individual redds) should include standard habitat characteristics used to describe salmon spawning habitat. Those standard habitat characteristics include water depth, substrate size, and velocity (Swan 1989; Chapman et al. 1986; Geist et al. 1997; Groves and Chandler 1999). Often these characteristics differ considerably both between and within major spawning areas of similar stocks or races of Chinook salmon. For example, water depth over fall Chinook salmon redds in the Hanford Reach of the Columbia River has been reported to be from 0.3 to 9.0 m (Chapman et al. 1986; Swan 1989); substrate particle size ranges from 5 to 30.5 cm (Swan 1989); and near-bed velocity ranges from 0.4 to 2 m/s⁻¹ (Chapman et al. 1986).

Similar variability in physical habitat characteristics has been noted for Chinook salmon spawning sites in the Snake River, Idaho (Connor et al. 1993; Groves and Chandler 1999).

Because of the large variability in site-scale standard habitat characteristics of fall Chinook salmon spawning habitat, evaluations of spawning habitat potential at this scale should also include measurements of other microhabitat variables. Examples of important microhabitat variables important to spawning salmon and other fish include substrate quality, water quality, hyporheic exchange, and hydraulic habitat complexity (Geist and Dauble 1998; Baxter and Hauer 2000; Geist 2000; Moir et al. 2002; Coulombre-Pontbriand and Lapointe 2004; Crowder and Diplas 2006).

The specific microhabitat variables measured for habitat quality will be dictated by the management and research questions specific to a given area. Site-scale physical characteristics limiting habitat potential at one site may not be relevant at another site. Thus, it is important to develop working hypotheses of multiple site-scale limiting factors from which testable hypotheses can be fully developed. Testing multiple hypotheses during the same period of study is often desirable, as conflicting conclusions of habitat quality may be developed depending on the microhabitat variables being measured. For example, evaluation of substrate quality in Snake River fall Chinook salmon spawning areas suggested high-quality habitat (Hanrahan et al. 2005), while a companion study of water quality indicated poor conditions for incubating fall Chinook salmon embryos (Groves and Chandler 2005).

8.2 Conclusion

Fall Chinook salmon production areas throughout the Columbia River Basin are physically very different, and their respective production capacities reflect these differences. The physical characteristics of these areas act as controlling factors that shape the quantity and quality of potential habitat for various life-history stages of fall Chinook salmon. The physical controlling factors set the limits (i.e., the template) within which any potential habitat is ultimately realized. The template most often associated with fall Chinook salmon production areas is one situated within broad alluvial floodplains possessing a complex channel composed of unconsolidated alluvium organized through dynamic equilibrium into bars and islands (Dauble et al. 2003). However, the templates for each production area in the Columbia River Basin are very different from one another and represent a continuum ranging from that of the Hanford Reach described in this report to one dominated by a bedrock canyon such as that of the Snake River in Hells Canyon.

It is imperative that conclusive statements regarding the *quality* and *quantity* of a river reach relative to fall Chinook salmon spawning be developed in the context of the template provided by the reach. The variation in habitat *quality* among river reaches is reflective of the differences in their templates, including the geomorphic features of the river channel, position in the watershed, and longitudinal slope of the river through the given reach. For example, fine sediment content in a given reach may be expected to be relatively high because of the alluvial, unconsolidated geologic formation through which it flows and because of the relatively low energy gradient imposed on the river by the low longitudinal slope of the channel bed. Assuming that fine sediment aggradation is not a problem, the fine sediment content in such a reach should not be cause for concern, as the quality of that substrate is what would be expected given the template provided.

Similar conclusions apply to habitat *quantity* as well. A specific example is provided by comparison of fall Chinook salmon spawning areas in the Hells Canyon Reach of the Snake River and the Hanford

Reach of the Columbia River. Although substrate permeability has been estimated to be higher in the Hells Canyon Reach (Arntzen et al. 2001), it would be misleading to interpret this to mean that the Hells Canyon Reach contained the best fall Chinook salmon spawning habitat of the reaches compared. The bedrock landforms of Hells Canyon control the local channel morphology at all of the study sites. The river possesses the characteristics of passive meandering, where the planform pattern is imposed by the local landform (Richards 1982; Thorne 1997). While all the study sites are located along lateral bars, these bars are largely remnant deposits of the Pleistocene Bonneville Floods (O'Connor 1993) rather than features resulting from contemporary geomorphic processes. The confining nature of the Hells Canyon valley walls and the limited alluvium recruitment within the canyon restricts the number, size, and substrate composition of depositional features within the Hells Canyon Reach. As a result, the lateral bars that are present at each of the study sites become that much more important in the context of providing spawning sites for fall Chinook salmon. However, the physical size, quantity, and longitudinal position of the bars relative to one another result in limited availability of potential production areas as dictated by the physical template within Hells Canyon. As such, the quantity of fall Chinook spawning habitat within Hells Canyon is less than what is available in the Hanford Reach.

The protocol that we envision for evaluating fall Chinook salmon spawning habitat involves an approach that begins at the watershed scale and becomes increasingly detailed at progressively finer scales, culminating in estimates of habitat quality at the scale of individual redds. The specific variables measured for habitat quantity and quality will be dictated by the management and research questions specific to a given area. At all scales of inquiry, the physical characteristics of a given area act as controlling factors that set the template within which any potential habitat is ultimately realized. Conclusive statements regarding the potential fall Chinook salmon spawning habitat *quality* and *quantity* of a river reach must be made in the context of the template provided by the reach. This protocol for evaluating fall Chinook salmon spawning habitat provides a framework for research and management actions concerning the recovery of fall Chinook salmon in the Columbia River Basin and western North America.

9.0 Conclusions

Our assessment of spawning habitat requirements of Hanford Reach fall Chinook salmon (*Oncorhynchus tshawytscha*) was done over several years and at several spatial scales. We had three primary objectives for this study. The first objective was to determine the relationship between physical habitats at different spatial scales and fall Chinook salmon spawning locations. This objective was accomplished in a series of investigations that we have described in this report. The second objective was to determine what the redd capacity could be in the Reach. The final objective was to recommend a protocol for identifying river reaches that could be protected or restored, to increase the amount of fall Chinook salmon production in the Columbia River Basin. The key conclusions from the study are presented in this section.

Conclusion 1 – Fall Chinook salmon in the Hanford Reach select specific sites to spawn, and females defend redd sites so as to maintain spacing between redds.

As we described in Section 3, the analysis of aerial photographs of two spawning areas showed that fall Chinook salmon select specific sites at which to construct redds. The boundaries of these spawning areas are established early in the spawning period, usually by the first week. The spawning areas are consistent from one year to the next. The pattern analysis of these spawning areas shows that redd site selection at the scale of the Hanford Reach is clustered, which supports our conclusion that there are specific and unique features of the physical habitat that fall Chinook salmon use to select spawning areas. Aerial surveys since the 1940s support this conclusion. The distribution of redds within the boundaries of the spawning areas appears to be uniform. However, we saw a tendency toward less uniform distributions in areas with lower densities (i.e., Area 4—Locke Island), especially in 1995 when the number of redds was reduced. These findings support our conclusion that females maintain spacing between redds likely as a mechanism to prevent redd superimposition. However, we know from years when fall Chinook salmon escapement to the Reach approached 80,000 adults that redd superimposition occurs. In those high escapement years, however, we have not noticed much spreading of spawning into new uncolonized areas. Thus, our conclusion needs to be framed within the context of the escapements we observed during the two years of the study (1994 and 1995) in which we analyzed spawning patterns.

Conclusion 2 – There are differences in the physical habitat within spawning and non-spawning areas that can be represented by microhabitat characteristics of water depth, water velocity, substrate size, and lateral slope.

All four of the microhabitat variables we measured and/or modeled showed some separation between spawning and non-spawning areas. Water velocity appears to provide the greatest separation, followed by depth, then substrate, and bed slope. The results indicated that spawning areas tended to have higher velocity, shallower depth, finer grainsize of the substrate, and lower slope. The logistic regression models that were constructed for each of the nine sites for three separate study years showed that water velocity and water depth were the only two variables included in all models and in all years.

Conclusion 3 – Fluctuation in river discharge due to operation of Priest Rapids Dam resulted in less stable velocity and water depth in non-spawning sites that were predicted to be used for spawning. However, incorporation of these dynamic variables into the predictive models did not improve model performance.

A significant portion (55 to 75%) of the predicted spawning habitat cells (from the models discussed in Conclusion 2) were not actually used for spawning. We used a two-dimensional hydrodynamic model to analyze the time series of velocity and depth within cells predicted to be used for spawning and actually used, compared to those that were not used. We found that cells used for spawning showed a higher frequency for velocities between 1 and 2.5 m/s and lower frequencies below 1 m/s. Depths within spawning cells were normally between 1.5 and 4.0 m. Non-spawning cells (but predicted to be spawning) tended to have higher frequencies of velocity values falling in the range less than 1 m/s and depths in the range of 1.5 to 4.0 m. While there appeared to be differences in the persistence of velocity and depth within the spawning and non-spawning cells, we were not able to improve model predictions (i.e., reduce our overprediction errors) when we incorporated these variables into the logistic regression analysis.

Conclusion 4 – Differences in substrate quality provide one explanation for the clustered pattern of fall Chinook salmon spawning in the Hanford Reach.

Substrate quality was assessed by investigating sediment grainsize and measuring sediment permeability. Our results suggested that hydraulic and sediment characteristics among and within sites were too variable to be useful in predicting whether individual sites would be used for spawning. However, substrate quality in the segment of the river where fall Chinook salmon did not spawn (i.e., the four sites downstream of the 100-F slough) was of significantly lower quality than within the segment where fall Chinook salmon did spawn (i.e., the four sites between the 100-F slough and upstream end of White Bluffs). Specifically, surface substrate in the non-spawning reach was composed of large cobble and boulders as compared to the spawning reach, which was more likely to have gravelly cobble substrate. In addition, the content of fine silt and coarse sand was about twice as high in non-spawning sites than in spawning sites. The permeability, specific discharge, and vertical hydraulic gradient were all higher in spawning reach than in non-spawning reach, with permeability nearly twice as high and specific discharge over three times as high. As the amount of coarse sand and silt increased, the permeability and specific discharge declined. There are several possible explanations for the differences in substrate quality between the spawning and non-spawning reach, including the influence of erosion from the White Bluffs; low bed turnover from a lack of historic spawning; closer proximity of the fine-grained clay-like Ringold Formation to the riverbed surface; and intrusion of fine material into interstitial spaces due to surface water fluctuations.

Conclusion 5 – There was little evidence that fall Chinook salmon use temperature differences or the vertical hydraulic gradient between the riverbed and the river to select redd sites.

We used continuous temperature and water level recorders to examine the interaction of groundwater and surface water within hyporheic habitats. The period of data collection was usually as long as a spawning period (i.e., 6 weeks in October and November) and data were collected over multiple years. There was no difference in the temperature and vertical hydraulic gradient of spawning and non-spawning sites when measured over continuous periods that included the entire and partial segments of the spawning period. River temperature fluctuated approximately 1–2°C over a 24-hour cycle. These fluctuations were concomitant with fluctuations in surface water caused from power peaking operations at Priest Rapids Dam. Temperature of the hyporheic zone mimicked that of the river, and differences with the river were normally less than 1°C. Consequently, there was no difference in predicted hatch and emergence dates using river and hyporheic temperatures.

Conclusion 6 – Fluctuations in the river due to operations of Priest Rapids Dam affected the hydraulic gradients between the river and the riverbed differently, depending on the permeability of the sediment.

The continuous information gathered from the temperature and water level recorders was used to evaluate whether there were differences in the response time of water level changes and upwelling or downwelling. At sites where riverbed permeability was relatively low, water level of the hyporheic zone showed a lag in response to changes in river stage, whereas in sites where the riverbed permeability was relatively high, there was no lag. When a lag occurred, the pressure differential between the river head and the hyporheic head resulted in a hysteretic response. This hysteretic response was more pronounced as riverbed permeability decreased, showing that the low permeability inhibited movement of river water into the riverbed bed. In general, the pronounced hysteretic responses occurred in the non-spawning locations. Although it is not likely that salmon use this response in vertical hydraulic gradient to select redd sites (we found no difference between spawning and non-spawning sites), it is possible that the variability in flux rates could affect the development rate of embryos.

Conclusion 7 – The redd capacity of the Hanford Reach is dependent upon the geomorphic features of the river channel that control site-specific habitat.

Although previous investigations, including this one, had established a qualitative relationship between geomorphic features and fall Chinook salmon spawning locations, there was no quantitative link between geomorphic features and redd capacity. Our findings suggested that the Hanford Reach could support about 90,000 adult spawners if all potential habitat were used. This is slightly above the highest escapement recorded in the Reach since 1990. While our geomorphic model was useful in predicting potentially available habitat, we acknowledge that it does not contain all of the variables that influence spawning habitat abundance, such as substrate quality. For example, about one-third of the expanded habitat estimate is within a segment of the Reach in which fall Chinook salmon have not spawned historically. Our investigation into the substrate quality in this segment showed that riverbed permeability was lower and the percentage of fine silts and coarse sands was higher than found in the spawning reach. Thus, our estimate should be used with caution.

Conclusion 8 – A scale-based approach should be used to evaluate fall Chinook salmon spawning habitat.

The protocol that we envision for evaluating fall Chinook salmon spawning habitat involves an approach that begins at the watershed scale and becomes increasingly detailed at progressively finer scales, culminating in estimates of habitat quality at the scale of individual redds. The specific variables measured for habitat quantity and quality will be dictated by the management and research questions specific to a given area. At all scales of inquiry, the physical characteristics of a given area act as controlling factors that set the template within which any potential habitat is ultimately realized. It is imperative that conclusive statements regarding the potential fall Chinook salmon spawning habitat quality and quantity of a river reach be made in the context of the template provided by the reach. This protocol for evaluating fall Chinook salmon spawning habitat provides a framework for research and management actions concerning the recovery of fall Chinook salmon in the Columbia River Basin and western North America.

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Appendix A

Individual Study Site Maps

Appendix B

Grainsize Data

Appendix B

Grainsize Data

Subsurface

The subsurface grainsize data were collected using sediment freeze cores. For each core collected, the weight (grams) of sediment retained on each sieve was included.

Table B.1. Subsurface Grainsize Data

Freeze core ID	Sieve Size (180 to 16 mm)							
	180	128	90.5	64	45.2	32	22.6	16
T54FC1	0.0	0.0	8570.4	5449.5	4675.7	2890.5	2460.7	2001.4
T54FC2	0.0	3599.7	9240.7	8819.2	7579.2	3184.0	1876.4	974.3
T54FC3	0.0	3860.2	9637.4	5647.2	1116.0	2000.7	2028.2	1742.9
T54FC4	0.0	0.0	7735.7	6196.7	3907.8	3660.3	3183.9	1869.9
T54FC5	0.0	8200.0	5240.5	2731.7	1473.6	2191.6	1765.0	1523.8
T59FC1	0.0	0.0	1380.2	4773.4	7008.3	6925.3	5490.2	2673.2
T59FC2	0.0	0.0	0.0	6291.3	5545.8	6836.7	4732.5	3565.4
T59FC3	0.0	0.0	0.0	3061.6	3455.6	5016.4	4206.4	1358.1
T59FC4	0.0	0.0	0.0	4314.6	4379.6	7997.2	5358.1	1665.4
T61FC1	0.0	6099.1	1654.2	6980.7	1299.4	1101.7	1526.9	1381.6
T61FC2	0.0	0.0	13667.8	8641.9	1706.4	3270.7	3150.1	2668.5
T61FC3	0.0	0.0	14404.9	6119.2	2803.2	836.9	1464.4	1533.7
T61FC4	0.0	0.0	17431.7	11903.5	5378.7	3694.8	2892.8	4162.3
T61FC10	0.0	0.0	12824.7	7001.5	2291.5	1357.9	1736.7	1908.4
T61FC11	0.0	0.0	0.0	10607.9	4339.1	2426.5	2123.5	2042.2
T64FC1	0.0	8372.6	9657.3	3694.3	2364.8	4008.4	2982.7	1456.4
T64FC2	0.0	10744.4	1366.8	9691.4	7661.8	7536.5	5703.6	3425.2
T64FC3	0.0	4911.1	10468.8	1890.4	2795.9	2991.5	1882.3	1743.8
T64FC4	0.0	0.0	2513.6	3527.7	4226.6	3348.5	2521.3	2030.2
T64FC5	0.0	10677.8	6594.4	2495.2	4754.7	2749.9	3085.9	1920.0
T81FC1	0.0	0.0	0.0	2754.7	3632.5	2366.4	2503.5	2195.3
T81FC2	0.0	0.0	0.0	3229.3	6034.0	4267.5	3748.4	3040.9
T81FC3	0.0	0.0	0.0	1218.5	5006.9	3162.6	2053.9	1707.0
T81FC4	0.0	0.0	0.0	1254.8	6548.9	3987.5	3996.8	2164.1
T90FC1	0.0	0.0	0.0	3002.1	4477.2	7919.9	4722.7	3625.8
T90FC2	0.0	0.0	3872.9	0.0	7462.4	4357.5	3157.1	1811.9
T90FC3	0.0	0.0	0.0	6612.9	4935.2	3569.3	3002.5	1517.2
T90FC4	0.0	0.0	1693.4	7343.5	8140.4	4678.8	4926.1	3127.2
T90FC5	0.0	0.0	0.0	3383.1	8256.9	8593.5	4730.8	3051.0
T94FC1	0.0	0.0	9215.5	3882.8	3159.9	6082.5	6353.7	4583.6
T94FC2	0.0	0.0	1864.5	655.4	793.5	1120.7	1016.8	2027.6
T94FC3	0.0	0.0	1760.0	1167.3	1320.3	2477.8	3077.8	2859.3
T94FC4	0.0	0.0	3023.9	7641.6	3882.1	2247.0	1763.9	1744.0
T96FC1	0.0	0.0	6008.4	2137.3	1378.8	1750.7	1658.6	1238.9
T96FC2	17555.0	0.0	2956.8	1548.1	2491.5	4497.5	5145.8	4840.8
T96FC3	0.0	0.0	0.0	4355.9	2079.4	2610.4	3136.7	1990.5
T96FC4	0.0	5435.2	3765.4	3963.7	3016.3	4870.8	4737.8	4663.5
T96FC5	0.0	0.0	14639.2	4120.4	2480.5	2354.5	3118.3	3394.0
T101FC1	0.0	13129.7	3374.9	10966.6	4905.2	2805.3	1745.3	1592.8
T101FC2	0.0	0.0	0.0	0.0	1209.1	408.6	1094.4	811.4
T101FC3	0.0	13800.4	6057.1	6016.8	2291.6	2140.7	1573.1	1770.6
T101FC4	0.0	3135.7	0.0	2322.1	1046.2	1934.5	3777.3	3949.7
T101FC5	0.0	0.0	0.0	444.9	834.4	3500.4	5064.5	3051.3

Table B.1. (contd)

Freeze core ID	Sieve Size (11.3 to 0.7 mm)								
	11.3	8	5.6	4	2.8	2	1.4	1	0.7
T54FC1	1578.1	1178.5	722.5	498.2	302.7	314.1	278.0	121.8	110.4
T54FC2	828.0	576.9	376.8	222.7	176.5	157.9	147.7	93.4	137.5
T54FC3	1939.3	1209.9	583.1	345.5	204.3	151.7	123.2	57.9	66.4
T54FC4	2249.1	2678.6	1459.2	836.9	639.0	568.0	480.4	224.8	208.3
T54FC5	1421.4	1520.8	1145.4	748.6	373.5	381.9	291.2	127.7	143.2
T59FC1	1981.6	2072.5	553.9	229.6	371.4	366.1	806.4	311.2	307.7
T59FC2	2064.5	1539.6	424.1	180.5	170.2	167.0	282.5	136.5	150.9
T59FC3	214.7	104.6	46.1	56.6	71.0	66.1	89.0	38.5	48.1
T59FC4	1102.2	580.7	203.5	112.3	139.3	128.5	170.1	66.9	77.8
T61FC1	1521.3	1622.9	1169.2	804.3	628.3	412.1	334.4	74.1	47.8
T61FC2	3234.0	2720.2	1926.5	1463.7	1118.5	787.7	600.8	138.7	94.5
T61FC3	1843.6	1833.4	1358.5	910.8	749.0	545.1	467.9	350.0	74.1
T61FC4	4264.7	4380.4	3098.3	2315.6	1904.7	1259.2	1004.6	213.6	151.4
T61FC10	1423.6	1604.1	1155.7	814.2	661.0	543.2	361.3	89.7	53.8
T61FC11	1281.6	1083.9	740.1	434.4	314.0	263.0	170.9	43.3	44.4
T64FC1	902.1	397.1	190.2	140.0	75.7	87.1	114.6	84.8	146.7
T64FC2	2621.8	1893.8	1466.8	1032.3	546.1	707.4	930.8	695.0	1005.3
T64FC3	1164.7	818.9	527.2	387.8	170.0	191.9	265.5	200.9	285.4
T64FC4	1228.9	971.6	656.5	494.0	175.2	276.3	289.8	195.4	266.2
T64FC5	1073.2	566.7	341.7	274.6	76.4	112.1	163.1	122.3	224.2
T81FC1	1188.5	944.0	452.8	311.6	189.3	123.1	86.4	22.1	29.4
T81FC2	1915.5	1129.7	664.0	399.2	315.0	208.0	141.7	34.7	37.7
T81FC3	1097.6	556.6	333.6	167.7	119.1	81.1	62.1	25.0	31.0
T81FC4	1457.2	1203.9	767.0	493.6	385.6	282.4	142.5	35.8	32.4
T90FC1	1450.5	535.0	310.3	172.7	113.3	121.8	192.6	158.6	245.0
T90FC2	1226.8	652.6	443.0	292.4	189.2	203.0	251.8	184.2	286.9
T90FC3	1048.4	830.5	442.1	198.2	121.5	109.8	104.0	67.7	87.6
T90FC4	1383.2	755.5	360.9	202.9	136.1	146.7	229.7	170.1	246.7
T90FC5	969.0	500.5	315.8	193.8	107.4	161.1	270.7	208.0	310.9
T94FC1	1887.8	531.0	275.0	195.8	129.1	119.0	219.8	203.7	215.8
T94FC2	1809.9	1272.0	891.4	703.2	496.8	230.1	240.6	198.7	392.2
T94FC3	2563.4	1582.7	941.0	771.6	478.4	244.3	402.1	249.4	341.0
T94FC4	1907.0	1682.8	1502.1	1282.3	1245.0	630.5	435.8	180.4	355.4
T96FC1	700.5	410.4	274.2	202.2	205.3	219.5	261.1	166.5	139.1
T96FC2	4497.6	3024.9	2178.5	1719.5	1032.3	892.2	830.0	534.3	721.0
T96FC3	1476.9	981.3	659.6	520.8	432.8	612.7	836.4	641.9	685.6
T96FC4	4620.4	4027.5	3299.8	2396.5	401.0	538.1	656.6	511.1	648.3
T96FC5	1946.0	1462.9	1308.2	1068.6	512.9	591.8	654.9	339.3	394.5
T101FC1	1460.3	1316.9	1092.5	825.6	666.2	339.1	138.5	32.1	28.1
T101FC2	1621.0	1076.9	1272.1	1265.2	671.7	948.7	914.0	491.6	477.8
T101FC3	1279.9	1177.2	914.9	589.2	377.4	195.9	94.3	36.3	36.3
T101FC4	4107.7	3396.1	2694.6	2367.1	889.7	1595.0	1372.6	689.5	641.8
T101FC5	2491.3	1827.7	1546.8	1128.6	277.2	872.2	774.1	459.8	507.1

Table B.1. (contd)

Freeze core ID	Sieve Size (0.5 to pan mm)							
	0.5	0.355	0.25	0.18	0.125	0.09	0.0625	pan
T54FC1	213.2	632.1	1041.4	889.1	462.6	295.1	140.9	142.8
T54FC2	251.2	694.3	1115.2	589.0	219.0	157.9	79.8	118.8
T54FC3	147.5	465.6	792.1	735.2	404.5	291.8	147.5	162.2
T54FC4	383.4	892.2	804.6	534.8	284.0	203.5	101.8	120.7
T54FC5	370.0	790.0	668.3	428.4	222.0	161.1	84.7	115.8
T59FC1	840.0	2576.5	2822.3	1317.4	557.0	224.6	153.8	235.2
T59FC2	377.3	1268.3	1643.9	813.9	359.6	154.1	112.4	167.0
T59FC3	300.7	1025.9	1036.7	477.5	216.5	93.8	68.6	176.8
T59FC4	338.3	1310.0	1680.9	857.6	378.2	155.6	128.5	188.2
T61FC1	256.8	887.4	799.1	311.7	118.2	49.0	35.8	58.5
T61FC2	515.6	1891.8	1973.0	908.9	469.4	220.8	163.3	188.0
T61FC3	405.6	1318.8	1131.9	406.1	148.2	58.7	39.6	51.5
T61FC4	871.9	3146.3	3085.3	1208.8	468.2	167.8	115.0	131.4
T61FC10	294.6	1168.3	1071.0	412.5	138.4	74.3	51.2	58.9
T61FC11	268.5	1137.4	1097.4	382.8	103.2	76.6	31.1	38.8
T64FC1	797.9	928.5	492.9	369.1	201.8	144.4	87.1	135.3
T64FC2	1787.2	4629.3	6441.3	4865.1	2755.2	3065.5	2395.3	3748.1
T64FC3	384.8	622.5	663.3	349.0	146.2	104.4	66.6	106.4
T64FC4	542.5	1519.8	1607.4	923.3	498.7	529.1	478.5	930.1
T64FC5	560.5	1146.5	1457.4	1090.5	922.3	529.9	331.2	433.1
T81FC1	249.9	1113.6	1815.6	1264.3	795.7	253.6	132.3	180.1
T81FC2	275.3	1111.3	1456.9	822.8	507.7	150.9	103.0	136.6
T81FC3	180.2	697.7	832.8	509.5	310.3	97.1	62.1	91.1
T81FC4	223.5	1169.6	1396.5	680.8	362.6	95.5	69.1	113.5
T90FC1	423.5	1296.0	1308.7	490.1	181.3	96.3	69.4	75.1
T90FC2	392.1	1093.7	1254.0	571.3	213.0	169.1	90.2	140.3
T90FC3	244.1	1051.2	841.0	254.6	92.3	58.4	32.7	45.6
T90FC4	508.3	1920.5	1982.1	676.3	216.9	153.1	65.9	100.0
T90FC5	407.1	1268.4	1514.5	597.3	210.3	127.5	55.9	80.5
T94FC1	391.2	1195.8	3097.3	2532.7	1474.0	683.6	435.6	607.0
T94FC2	334.7	2039.6	3603.2	1997.7	1270.8	549.1	439.3	784.4
T94FC3	346.1	2402.2	5002.9	3018.1	1964.5	809.2	615.8	1414.9
T94FC4	877.0	987.7	1277.1	700.2	319.7	160.8	117.9	198.3
T96FC1	65.3	68.1	292.4	384.1	195.9	120.2	64.3	71.0
T96FC2	555.0	658.8	824.8	814.4	430.5	409.8	285.3	1431.7
T96FC3	291.8	286.9	1609.5	2518.8	1361.5	967.7	476.5	622.4
T96FC4	521.5	401.0	330.4	268.0	145.4	151.7	101.8	180.8
T96FC5	662.8	1088.9	2943.3	3298.4	1712.3	1223.1	647.1	883.8
T101FC1	184.6	1131.8	1366.5	642.1	244.8	210.7	116.4	224.7
T101FC2	533.2	962.5	990.2	512.4	540.1	588.6	491.6	1114.8
T101FC3	150.0	945.8	1412.7	645.9	188.7	94.3	75.0	166.9
T101FC4	498.8	562.4	670.4	603.7	387.6	374.9	622.8	2176.5
T101FC5	473.3	567.9	693.0	622.0	368.5	456.4	774.1	2893.7

Surface

The surface grainsize data were collected using underwater video equipment. For each sampling location, the intermediate axis (b-axis) of sediment grains on the riverbed were measured (cm). The results for each grain measured are presented in columns below the sampling location.

Table B.2. Surface Grainsize Data

T54	T59RB	T59LB	T61RB	T61LB	T64RB	T64LB	T81	T90	T94bar	T94chan	T96	T101
5.23	1.97	2.55	10.81	3.8	9.18	6.05	9.22	7.28	6.27	5.8	8.64	14.4
1.98	15.74	8.56	13.09	2.86	1.12	9.87	3.14	1.74	7.49	76.84	13.56	1.84
20.57	4.47	4.19	12.84	3.72	6.14	8.49	13.78	4.01	14.71	7.6	19.69	21.24
12.83	4.09	6.4	3.03	3.96	7.59	4.28	4.23	7.32	6.04	14.29	9.05	12.01
2.74	7.94	6.26	5.33	47.41	9.56	6.83	6.34	12.91	10.25	1.14	16.95	1.84
71.36	15.53	8.51	11.03	3.52	1.89	12.2	10.49	5.57	14.97	7.61	3.64	27.32
27.02	7.98	9.2	10.15	7.06	7.65	6.86	4.47	9.54	18.73	9.41	9.09	142.98
12.38	11.21	3.78	7.46	2	3.28	3.39	9.31	9.12	16.98	13.19	10.49	28.11
11.37	9.35	9.78	2.37	4.99	2.91	2.97	1.56	7.34	11.61	3.62	3.97	91
3.37	3.46	3.25	10.31	2.17	2.43	5.24	3.61	8.95	17.48	7.7	7.34	2.11
4.88	13.07	3.82	17.2	1.93	2	16.05	2.27	1.58	4.79	2.55	3.69	8.46
6.42	13.58	4.53	18.89	3.51	3.05	11.52	9.43	7.61	6.4	1.8	2.77	97.65
9.8	5.45	2.88	17.1	7.65	11.64	0.5	5.67	9.18	2.18	1.41	2.86	77.65
7.09	1.53	3.13	11.11	1.7	2.51	9.74	12.17	2.13	8.31	2.31	15.25	1.11
26.54	2.85	4.38	4.45	4.34	4.86	7.04	12.01	2.63	5.51	9.19	14.52	19.94
7.63	9.92	1.99	8.46	6.71	2.06	2.88	7.81	12.44	8.91	6.86	12.37	7.13
10.85	0.55	1.71	12.01	5.13	1.58	123.48	1.9	1.99	16.68	3.52	4.51	7.77
4.45	3.4	3.21	18.59	9.66	8.95	6.49	2.31	2.05	10.37	4.33	12.28	9.9
12.87	16.31	2.76	11.22	3.01	4.85	3.97	8.62	1.7	9.35	2.04	14.81	4.23
8.12	4.03	1.57	13.68	9.06	8.34	8.53	1.52	9.84	9.35	14.13	6.01	1.93
4.84	17.95	5.36	4.94	1.6	9.17	10.77	0.97	3.95	4.97	36.56	7.82	33.24
3.63	16.72	5.52	15.55	1.34	3.84	8.4	4.09	1.8	9.13	45.04	29.77	2.44
7.16	14.96	2.56	13.42	2.3	7.26	5.44	4.03	7.26	3.25	16.63	10.05	1.49
11.11	3.8	3.45	9.76	4.45	13.01	8.44	4.52	5.25	18.51	24.21	4.17	102.86
9.28	8.2	7.03	12.08	4.84	12.8	6.68	7.47	16.02	0.5	30.77	14.94	117.12
17.76	9.57	5.98	2.51	2.54	8.53	7.53	11.9	13.76	11.7	2.9	8.06	26.1
5.78	3.93	5.36	8.04	2.75	4.28	2.95	2.67	15.28	2.12	9.96	11.06	6.09
3.79	9.16	2.83	16.73	40.7	11.58	3.8	7.27	5.68	9.64	11.44	5.43	6.28
3.23	9.94	3.16	3.42	8.83	2.26	12.87	63.67	1.99	13.14	9.01	8.43	114.62
2.5	10.21	3.31	13.34	9.66	3.88	12.39	2.21	3.58	16.79	16	8.11	115.86
31.14	13.87	8.45	4.45	6.52	2.77	7.85	11.41	3.16	14.18	14.25	10.57	2.58
3.52	17.05	5.99	10.36	3.37	5.53	8.47	8.5	14.89	6.73	8.76	8.53	7.6
15.47	10.88	2.03	6.85	1.68	2.81	6.88	10.14	3.61	6.15	0.5	6.35	68.45
2.99	4.76	11.08	10.81	7.41	2.19	3.37	6.68	5.43	4.07	1.58	3.42	8.24
11.18	19.68	8.27	2.42	7.08	3.89	5.98	3.82	2.83	2.58	20.49	4.12	68.59
11.68	10.94	164.98	3.75	3.67	4.71	3.13	8.53	6.5	3.86	1.31	11.72	79.31
12.9	14.07	7.28	12.43	3.71	2.77	4.06	8.28	5.14	2.97	12.17	3.03	101.6
2.3	7.21	8.33	4.4	11.81	8.19	4.73	7.48	12.27	8.36	10.96	14.01	38.06
18.86	18.53	4.82	10.68	9.78		9.65	4.95	10.11	1.42	15.08	10.18	32.69
4.21	2.71	6.62	13.8	4.03		3.34	9.27	2.04	17.65	2.57	28.63	5.46
8.31	11.69	9.11	11.05	9.4		9.33	7.25	6.12	8.8	20.29	18.82	97.39
6.12	6.22	3.44	7.22	4.44		4.52	6.19	2.99	9.6	32.52	18.48	10.86
28.38	19.63	5.67	9.49	8.17		2.71	0.5	2.56	7.13	6.08	11.58	95.01
1.88	12.17	3.32	10.14	3.27			3.19	2.72	17.49	5.43	11.34	91.33
2.28	9.39	1.89	12	3.7			13.74	16.6	15.52	30.75	10.31	39.16
5.13	23.29	5.05	3.85	5.87			3.4	2.5	2.68	7.19	3.09	8.65
2.65	9.46	12.93	5.73	4.14			6.17	5.67	3.13	2.22	1.54	20.15
5.48	9.97	6.59	2.94	6.22			3.65	4.71	8.27	11.52	6.04	1.8
15.18	17.94	5.72	2.02	4.29			3.38	3.38	3.08	2.16	1.95	2.01
3.46	16.18	3.37	4.71	4.92			6.13	3.84	9.83	11.45	26.33	1.5
11.99	18.54	8.33	5.37	3.79			5.29	4.09	2.74	3.99	4.5	82.23
18.59	7.84	5.62	7.74	6.62			2.8	13.13	10.51	14.57	5.3	24.63
5.08	17.12	9.26	10.37	2.72			7.98	9.02	12.08	2.62	35.19	8.18
3.01	13.69	10.77	4.51	10.84			1.88	11.03	11.05	3.05	8.61	4.28
7	18.76		13.82	8.34			3.58	6.42	1.65	11.36	3.04	58.37
2.46	17.4						3.85	10.75	8.09	17.67	11	5.76
8.5	16.34						1.99	8.59	8.36	14.71	6.02	3.45
1.32	16.98						5.8	0.5	14.72	3.85	6.58	4.41

Table B.2. (contd)

T54	T59RB	T59LB	T61RB	T61LB	T64RB	T64LB	T81	T90	T94bar	T94chan	T96	T101
1.69	23.58						4.11	6.85	8.96	1.76	19.8	2.78
4.6	10.25						6.2	8.49	6.44	1.97	9.73	131.09
8.12	22.43						10.83	9.78	11.86	23.7	9.04	8.44
18.35	10.06						9.58	1.31	12.04	24.97	27.34	103.11
4.44							7.97	1.59	11.87	26.2	7.51	11.99
5.12							9.4	2.79	8.26	19.38	8.41	2.06
1.4							7.73	8.46	5.92	36.73	13.96	9.83
2.78							3.46	12.45	10.8	5.56	8.56	4.57
2.06							7.36	11.38	21.98	30.81	13.94	78.01
9.42							11.84	1.55	2.96	16.94	8.05	2.39
12.75							8.98	16.93	0.5	32.81	4.74	8.51
10.36							13.36	5.67	9.3	7.19	2.62	99.25
2.09							4.14	10.18	21.62	4.43	2.24	14.85
15.95							13.16	14.41	2.19	8.72	33.24	12.26
33.92							1.41	51.9	10.61	4.76	15.36	92.19
13.82							5.41	11.51	11.63	11.71	4.32	5.41
3.45							8.48	2.14	5.26	3.97	6.52	1.99
4.71							6.6	3.14	12.89	6.87	5.16	118.58
2.91							5.61	11.65	16.62	3.68	13.64	22.53
1.96							5.16	8.83	10.86	11.01	11.28	3.92
6.96							6.88	3.84	15.17	3.15	2.88	87.77
1.99							8.06	2.72	3.39	10	30.53	1.58
18							10.19	4.51	18.73	12.55	3.93	4.92
43.65							3.54	3.35	16.14	20.32	19.32	4.93
7.66							3.34	3.51	1.4	2.23	14.7	9.97
6.14							1.74	4.23	16.45	17.94	11.19	73.86
5.19							6.38	3.92	11.08	1.92	7.26	82.72
11.75							11.44	5.65	9.48	1.23	9.64	17.57
2.36							6.78	12.55	8.96	12.83	11.11	121.81
9.99							3.62	17.64	9.77	13.38	4.87	2.33
4.47							7.37	24.44	0.5	2.02	10.96	7.37
17.3							10.05	12.5	83	46.55	8.74	72.24
4.84							5.97	9.04	5.69	10.76	11.28	8.09
31.88							6.9	13.07	15.93	2.26	11.63	10.92
12.42							9.54	16.93	0.85	6.98	18.89	7.07
8.24							9.95	10.44	5.86	2.89	12.72	3.03
2.93							10.22	2.73	9.4	10.9	13.03	15.31
3.21							4.13	3.35	10.3		15.53	13.47
2.33							5.55	8.07	2.1		15.25	3.35
4.55							13.57	0.5	12.33		10.73	1.32
1.99							5.36	1.42	11.88		3.73	1.32
38.42							9.32	0.5	7.83		5.29	5.29
2.34							8.35	7.25	5.8		10.5	5.4
6.99							4.38	8.4			7.19	4.02
9.33							4.08	7.67			12.06	13.43
1.68							6.38	8.79			3.32	11.37
3.23							12.11	2.54			1.79	129.7
137.57							8.77	12.93			12.51	2.77
20.17							1.05	16.81			1.77	2.97
21.26							8.56	44.96			1.64	4.01
23.06							6.34	12.33			10.67	10.81
9.35							4.14	19.31			2.39	2.42
							12.42	27.97			7.96	
							9.87	10.07				
							2.91					
							6.73					
							11.65					

Appendix C

Hydraulic Conductivity Results

Appendix C

Hydraulic Conductivity Results

Report results presented geometric mean hydraulic conductivity values (K-cm/s) for replicate slug tests conducted in Hanford Reach piezometers. This appendix includes the individual hydraulic conductivity values (K-cm/s) estimated from each slug test.

Year	Piezometer	K (cm/s)	Year	Piezometer	K (cm/s)	Year	Piezometer	K (cm/s)
2000	T54RB04	0.002204	2000	T59IRF2	0.005429	2000	T81IRBF1	0.013049
2000	T54RB04	0.00127	2000	T59IRF2	0.005168	2000	T81IRBF1	0.01104
2000	T54RB04	0.001227	2000	T59IRF2	0.005384	2000	T81IRBF2	0.009582
2000	T54LB04	0.004624	2000	T59IRF3	0.004869	2000	T81IRBF2	0.008659
2000	T54LB04	0.004758	2000	T59IRF3	0.005073	2000	T81IRBF2	0.008468
2000	T54LB04	0.005666	2000	T59IRF3	0.004944	2000	T81IRBF2	0.013053
2000	T54LB04	0.005774	2000	T59IRF4	0.007742	2000	T81IRBF3	0.012583
2000	T59RB04	0.007679	2000	T59IRF4	0.007823	2000	T81IRBF3	0.021937
2000	T59RB04	0.006984	2000	T59IRF4	0.007536	2000	T81IRBF3	0.012157
2000	T59RB04	0.006746	2000	T59IRF5	0.00642	2000	T81IRBF4	0.014126
2000	T59IR04	0.011644	2000	T59IRF5	0.006381	2000	T81IRBF4	0.012713
2000	T59IR04	0.01116	2000	T59IRF5	0.00648	2000	T81IRBF4	0.013374
2000	T59IR04	0.00852	2000	T59IRF6	0.005588	2000	T81IRBF5	0.01553
2000	T61IR04	0.010296	2000	T59IRF6	0.005536	2000	T81IRBF5	0.01488
2000	T61IR04	0.009718	2000	T59IRF6	0.005985	2000	T81IRBF5	0.014772
2000	T61IR04	0.008625	2000	T59IRF7	0.002077	2000	T81IRBF6	0.018939
2000	T64RB04	0.003574	2000	T59IRF7	0.002295	2000	T81IRBF6	0.01834
2000	T64RB04	0.003015	2000	T59IRF7	0.002176	2000	T81IRBF6	0.018475
2000	T64IR04	0.01004	2000	T59IRF8	0.005222	2000	T81ILF7	0.018041
2000	T64IR04	0.00934	2000	T59IRF8	0.005297	2000	T81ILF7	0.015162
2000	T64IR04	0.009539	2000	T59IRF8	0.005269	2000	T81ILF7	0.014771
2000	T64IR04	0.008924	2000	T59IRF8	0.005819	2000	T81ILF8	0.005271
2000	T59IL	0.016054	2000	T59IRF9	0.003141	2000	T81ILF8	0.005559
2000	T59IL	0.017742	2000	T59IRF9	0.003478	2000	T81ILF8	0.005337
2000	T59IL	0.016939	2000	T59IRF9	0.004159	2000	T81ILF8	0.005226
2000	T61ILA	0.007166	2000	T59IRF9	0.00405	2000	T81ILF9	0.004607
2000	T61ILA	0.004676	2000	T61IRF1	0.00834	2000	T81ILF9	0.004581
2000	T61ILA	0.004817	2000	T61IRF1	0.009501	2000	T81ILF9	0.004696
2000	T61ILB	0.008214	2000	T61IRF2	0.008656	2000	T94RBF1	0.00509
2000	T61ILB	0.008313	2000	T61IRF2	0.008627	2000	T94RBF1	0.004487
2000	T61ILB	0.008468	2000	T61IRF2	0.007751	2000	T94RBF1	0.004231
2000	T64ILB	0.01092	2000	T61IRF3	0.006843	2000	T94RBF2	0.006117
2000	T64ILB	0.010959	2000	T61IRF3	0.007714	2000	T94RBF2	0.005341
2000	T64ILB	0.011179	2000	T61IRF3	0.008027	2000	T94RBF2	0.005027
2000	T81ILC	0.007549	2000	T59IRF10	0.005393	2000	T94RBF3	0.001186
2000	T81ILC	0.006867	2000	T59IRF10	0.005094	2000	T94RBF3	0.001445
2000	T81ILC	0.00638	2000	T59IRF10	0.005201	2000	T94RBF4	0.016265
2000	T90RB04	0.006381	2000	T59IRF11	0.006785	2000	T94RBF4	0.013865
2000	T90RB04	0.005744	2000	T59IRF11	0.006609	2000	T94RBF4	0.014332
2000	T90RB04	0.00647	2000	T59IRF11	0.006565	2000	T94RBF4	0.016407
2000	T90LB04	0.009965	2000	T59IRF11	0.006438	2000	T94RBF5	0.020895
2000	T90LB04	0.010275	2000	T59IRF11	0.00662	2000	T94RBF5	0.018174
2000	T90LB04	0.009409	2000	T59IRF12	0.006652	2000	T94RBF5	0.025777
2000	T90LB04	0.009664	2000	T59IRF12	0.007079	2000	T94RBF6	0.014234
2000	T96RB04	0.000657	2000	T59IRF12	0.006793	2000	T94RBF6	0.014021
2000	T96RB04	0.00102	2000	T61IRF7	0.007361	2000	T94RBF6	0.014452
2000	T96RB04	0.001506	2000	T61IRF7	0.007178	2000	T94RBF7	0.002017
2000	T96RB04	0.000641	2000	T61IRF7	0.007106	2000	T94RBF7	0.001557
2000	T101RB04	0.000292	2000	T61IRF8	0.006472	2000	T94RBF7	0.001479
2000	T101RB04	0.000295	2000	T61IRF8	0.006552	2000	T94RBF8	0.001765
2000	T94RB04	0.000498	2000	T61IRF8	0.00637	2000	T94RBF8	0.001204
2000	T94RB04	0.000343	2000	T61IRF9	0.007263	2000	T94RBF8	0.001318
2000	T96LB04	0.000965	2000	T61IRF9	0.007154	2000	T94RBF9	0.002544
2000	T96LB04	0.000766	2000	T61IRF9	0.007228	2000	T94RBF9	0.002268
2000	T101LB04	0.000518	2000	T61IRF9	0.007441	2000	T94RBF9	0.002668
2000	T59IRF1	0.005993	2000	T81IRBF1	0.02186	2000	T96RB04B	0.002842
2000	T59IRF1	0.005473	2000	T81IRBF1	0.014385	2000	T96RB04B	0.002613
2000	T59IRF1	0.005596						

Year	Piezometer	K (cm/s)
2001	T54RBF1	0.004462
2001	T54RBF1	0.003683
2001	T54RBF1	0.003226
2001	T54RBF1	0.003509
2001	T54RBF2	0.004445
2001	T54RBF2	0.004334
2001	T54RBF2	0.004748
2001	T54RBF2	0.004443
2001	T54RBF3	0.004099
2001	T54RBF3	0.004427
2001	T54RBF3	0.004204
2001	T54RBF3	0.004404
2001	T54RBF4	0.003241
2001	T54RBF4	0.003321
2001	T54RBF4	0.003356
2001	T54RBF4	0.003525
2001	T54RBF5	0.00484
2001	T54RBF5	0.00491
2001	T54RBF5	0.005272
2001	T54RBF5	0.005537
2001	T54RBF6	0.002897
2001	T54RBF6	0.002873
2001	T54RBF6	0.00261
2001	T54RBF6	0.002775
2001	T54RBF7	0.015743
2001	T54RBF7	0.015838
2001	T54RBF7	0.016459
2001	T54RBF7	0.015364
2001	T54RBF8	0.004229
2001	T54RBF8	0.003784
2001	T54RBF8	0.003816
2001	T54RBF8	0.004433
2001	T54RBF9	0.031595
2001	T54RBF9	0.03173
2001	T54RBF9	0.032594
2001	T54RBF9	0.032117
2001	T61IRF10	0.010646
2001	T61IRF10	0.010324
2001	T61IRF10	0.009714
2001	T61IRF10	0.010275
2001	T61IRF11	0.010218
2001	T61IRF11	0.009829
2001	T61IRF11	0.009836
2001	T61IRF11	0.010471
2001	T61IRF12	0.011099
2001	T61IRF12	0.010758
2001	T61IRF12	0.010406
2001	T61IRF12	0.010451
2001	T61IRF13	0.01045
2001	T61IRF13	0.00966
2001	T61IRF13	0.009447
2001	T61IRF13	0.009115
2001	T61IRF14	0.008589
2001	T61IRF14	0.00935
2001	T61IRF14	0.009898
2001	T61IRF14	0.010127
2001	T61IRF15	0.009528
2001	T61IRF15	0.009894

Year	Piezometer	K (cm/s)
2001	T61IRF15	0.010307
2001	T61IRF15	0.011214
2001	T64RBF1	0.009628
2001	T64RBF1	0.010026
2001	T64RBF1	0.010673
2001	T64RBF1	0.011559
2001	T64RBF2	0.010036
2001	T64RBF2	0.010328
2001	T64RBF2	0.010669
2001	T64RBF2	0.010638
2001	T64RBF3	0.011133
2001	T64RBF3	0.011381
2001	T64RBF3	0.011658
2001	T64RBF3	0.012184
2001	T64RBF4	0.004677
2001	T64RBF4	0.004329
2001	T64RBF4	0.004664
2001	T64RBF4	0.004791
2001	T64RBF5	0.002846
2001	T64RBF5	0.002583
2001	T64RBF5	0.002882
2001	T64RBF5	0.004034
2001	T64RBF6	0.006533
2001	T64RBF6	0.006865
2001	T64RBF6	0.00701
2001	T64RBF7	0.017451
2001	T64RBF7	0.017529
2001	T64RBF7	0.017755
2001	T64RBF7	0.016771
2001	T64RBF8	0.01517
2001	T64RBF8	0.015955
2001	T64RBF8	0.016375
2001	T64RBF8	0.014321
2001	T64RBF9	0.0153
2001	T64RBF9	0.015313
2001	T64RBF9	0.016662
2001	T64RBF9	0.016999
2001	T90LBF1	0.008872
2001	T90LBF1	0.008794
2001	T90LBF1	0.009943
2001	T90LBF1	0.008777
2001	T90LBF2	0.007377
2001	T90LBF2	0.007593
2001	T90LBF2	0.008022
2001	T90LBF2	0.00811
2001	T90LBF3	0.006456
2001	T90LBF3	0.00714
2001	T90LBF3	0.007688
2001	T90LBF3	0.007668
2001	T90LBF4	0.009944
2001	T90LBF4	0.010089
2001	T90LBF4	0.010151
2001	T90LBF4	0.010279
2001	T90LBF5	0.00822
2001	T90LBF5	0.008498
2001	T90LBF5	0.008831
2001	T90LBF5	0.009842

Year	Piezometer	K (cm/s)
2001	T90LBF6	0.008921
2001	T90LBF6	0.008748
2001	T90LBF6	0.009434
2001	T90LBF7	0.007724
2001	T90LBF7	0.007837
2001	T90LBF7	0.008439
2001	T90LBF7	0.00848
2001	T90LBF8	0.009533
2001	T90LBF8	0.009472
2001	T90LBF8	0.009544
2001	T90LBF8	0.010158
2001	T90LBF9	0.008952
2001	T90LBF9	0.009622
2001	T90LBF9	0.00955
2001	T90LBF9	0.009934
2001	T96RBF4	0.000345
2001	T96RBF5	0.000362
2001	T96RBF6	0.000201
2001	T96RBF8	0.000203
2001	T96RBF8	0.000361
2001	T96RBF9	0.002842
2001	T96RBF9	0.003067
2001	T96RBF9	0.004078
2001	T101LBF1	0.01059
2001	T101LBF1	0.005546
2001	T101LBF1	0.004201
2001	T101LBF2	0.000232
2001	T101LBF2	0.000363
2001	T101LBF2	0.000309
2001	T101LBF3	0.001604
2001	T101LBF3	0.001712
2001	T101LBF3	0.001941
2001	T101LBF4	0.000544
2001	T101LBF4	0.000307
2001	T101LBF5	0.00583
2001	T101LBF5	0.007989
2001	T101LBF5	0.025089
2001	T101LBF5	0.010101
2001	T101LBF6	0.001177
2001	T101LBF6	0.000679
2001	T101LBF7	0.001637
2001	T101LBF7	0.001378
2001	T101LBF7	0.0013
2001	T101LBF8	0.002304
2001	T101LBF8	0.002343
2001	T101LBF8	0.002761
2001	T101LBF9	0.001083
2001	T101LBF9	0.001141
2001	T101LBF9	0.000985

Year	Piezometer	K (cm/s)
2003	T54RBHYP	0.00347
2003	T54RBHYP	0.003675
2003	T54RBHYP	0.004528
2003	T59IRHYP	0.006672
2003	T59IRHYP	0.007035
2003	T59IRHYP	0.007118
2003	T59IRHYP	0.005565
2003	T61IRHYP	0.009391
2003	T61IRHYP	0.007843
2003	T61IRHYP	0.009335
2003	T61IRHYP	0.010356
2003	T64RBHYP	0.003407
2003	T64RBHYP	0.003471
2003	T64RBHYP	0.004302
2003	T64RBHYP	0.003394
2003	T81RBHYP	0.015144
2003	T81RBHYP	0.015598
2003	T81RBHYP	0.015496
2003	T90LBHYP	0.005576
2003	T90LBHYP	0.006261
2003	T90LBHYP	0.006276
2003	T94BARHYP	0.008064
2003	T94BARHYP	0.009845
2003	T94BARHYP	0.010067
2003	T96RBHYP_1	0.00043
2003	T96RBHYP_2	0.001885
2003	T96RBHYP_2	0.002203
2003	T96RBHYP_2	0.00191
2003	T101LBHYP	0.001962
2003	T101LBHYP	0.002428
2003	T101LBHYP	0.002286