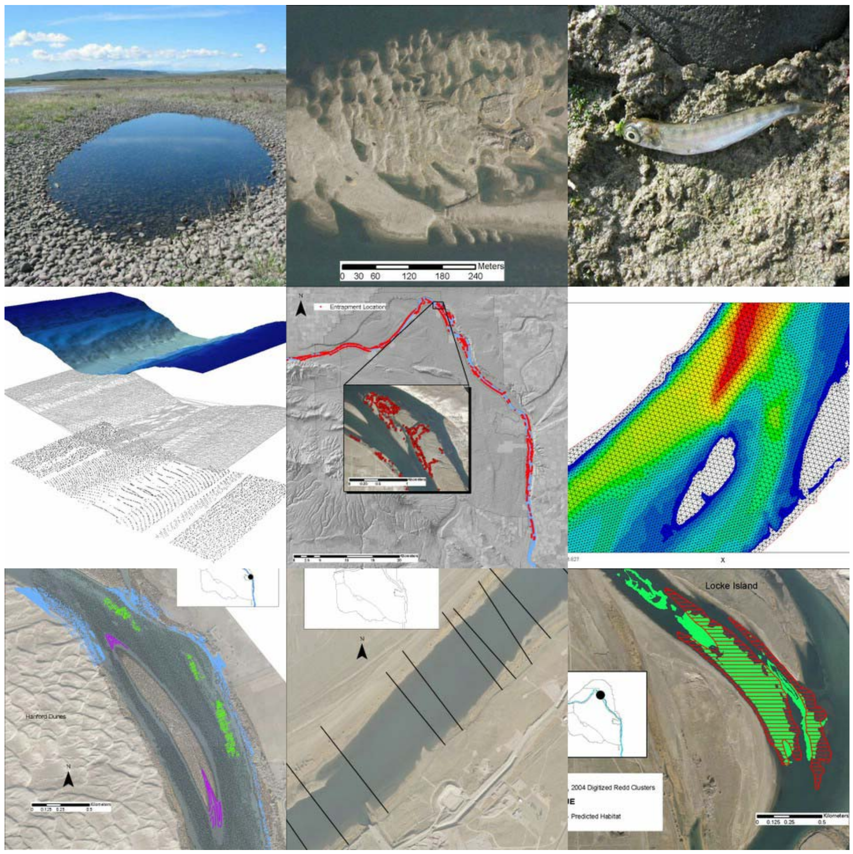
Effects of Hydropower Operations on Spawning Habitat, Rearing Habitat, and Stranding/Entrapment Mortality of Fall Chinook Salmon in the Hanford Reach of the Columbia River

Final Report



THE FISH PASSAGE CENTER   NUGENT GIS AND ENVIRONMENTAL SERVICES

Effects of Hydropower Operations on Spawning Habitat, Rearing Habitat, and Stranding/Entrapment Mortality of Fall Chinook Salmon in the Hanford Reach of the Columbia River

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Final Report

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EXECUTIVE SUMMARY

This report describes research conducted primarily in 2003 and 2004 to evaluate the effects of upstream dam operations on spawning and rearing conditions for fall Chinook salmon, *Oncorhynchus tshawytscha,* in the Hanford Reach of the Columbia River. Results from habitat modeling tasks which continued in 2005 and 2006 are also included in this report. This study is focused on the effects of streamflows and streamflow fluctuations on 1) entrapment and entrapment mortality of juveniles, 2) adult spawning habitat, and 3) juvenile rearing habitat. An independent peer review was conducted on the draft version of this report utilizing three reviewers, each with different areas of expertise and different levels of knowledge regarding hydrodynamic modeling, fall Chinook biology, life history, and habitat requirements, and fishery issues relating to hydropower development and operations. Peer review comments have been incorporated into this final version.

A foundational element of this study was the development and application of hydrodynamic models to characterize the hydrology of the Hanford Reach. The hydrodynamic modeling component of this study built upon earlier work characterizing the physical conditions in the Hanford Reach and provided a basis for quantitatively assessing the effects of streamflow and streamflow fluctuation on entrapment mortality of juvenile fish, adult spawning habitat, and juvenile rearing habitat for fall Chinook. Our assessment capitalized on recent advances in habitat mapping, remote sensing technology, hydrodynamic simulation models, statistical sampling methods and GIS technology to characterize habitat conditions and evaluate the effects of streamflow and flow fluctuations on Hanford Reach fall Chinook. The hydrodynamic models included River2D, which was used to estimate water velocities, depths, and water surface elevations for habitat modeling purposes, and MASS1, which was used to produce steady and unsteady-state streamflow simulations for the entire Hanford Reach for determining entrapment flow bands and entrapment event histories.

Using empirical physical and biological data integrated with hydrodynamic model output and GIS analyses, we mechanistically estimated the impact of fluctuating streamflows on juvenile fall Chinook salmon for the entire Hanford Reach in 2003. We estimated that 1,602,891 juvenile fall Chinook were entrapped, and of those, 1,297,104 were mortalities. No estimates of stranded fish were made. Our results confirm that the impact from entrapment and mortality of juvenile fall Chinook due to flow fluctuations associated with hydropower operations is higher than previously estimated. These impacts appear to be significantly greater than those previously estimated for both entrapped and stranded fish. Our observations of significant entrapment in spite of the current protection measures highlight the importance of developing hydro operational strategies that are more effective at reducing juvenile mortality.

In our evaluation of the physical factors related to entrapment, we found little quantitative basis for the assumption that streamflow fluctuations at low streamflows are more harmful than the same fluctuations at high streamflow levels, or vice versa. Using simulations of alternative hydrographs, we found that entrapment impacts could be reduced by controlling the size and frequency of streamflow fluctuations, and the results of our field work suggest that the timing of fluctuations (early vs. late rearing season) could also be used to reduce the impacts. The simulation results suggest that reducing flow fluctuations has considerable potential for reducing mortality levels if fluctuation magnitudes are kept at or below 10 thousand cubic-feet-per-second (kcfs). The number of entrapped Chinook dramatically increased with increasing streamflow fluctuations of 20, 30 and 40 kcfs. Based on the non-linear relationship between the fluctuation magnitude and frequency, and fish entrapped, it appears fluctuations above 10 kcfs produce dramatic increases in the number of fish entrapped. The size and frequency of flow fluctuations were directly related to the number of entrapments affected.

We conducted a re-regulation analysis to evaluate the physical capability to reduce flow fluctuation magnitude and frequency during the spring rearing period for juvenile fall Chinook. We determined that the physical ability exists to control flow fluctuation magnitude and frequency to the extent required to greatly reduce expected juvenile Chinook entrapment mortality impacts. Our analysis of the capability of the Priest Rapids and Wanapum projects to re-regulate and reduce flow fluctuations indicated that flexibility may exist within "normal" operating limits to both limit the number of flow reductions and their magnitude.

We used three approaches to place the estimates of juvenile entrapment mortality into a population context. Using smolt-to-adult survival rate data, we estimated that 4,100-8,200 adults may have been lost due to entrapment mortality in 2003. Using fry production modeling, the 2003 entrapment impact likely constituted a 12% reduction in the fry population. In other years under the Protection Plan, the impacts to the population caused by operations at Priest Rapids Dam could have ranged between 31% and 90%. The scale of these impacts imposed upon the fry population could reduce harvest numbers of all adult Chinook populations in ocean and in-river fisheries by 9,000-170,000 fish. These potential impacts represent large reductions in the allowable harvest of fall Chinook by commercial and sport ocean fisheries, and commercial, sport, and tribal treaty in-river fisheries.

In our analysis of spawning habitat, we found that fall Chinook redds are distributed throughout the Hanford Reach, but the highest concentration was found in the middle segment near White Bluffs. In consideration of this distribution, our spawning distribution results indicate that the highest management priority area should be the White Bluffs/Locke Island area, followed by Vernita Bar. Managing streamflows to provide spawning habitat at White Bluffs may require a different strategy than the current management protocol for Vernita Bar. Depending on the goals of a spawning season streamflow management plan, the hourly hydrograph at Priest Rapids Dam may need to be structured to produce the desired effect at White Bluffs. Adjustments could then be made in consideration of the effects at Vernita Bar.

We developed a comprehensive, Reach-wide assessment of the effects of streamflows and flow fluctuations on spawning habitat. During the course of development of our spawning habitat model, we found that the nature of physical conditions along the Reach in terms of river channel geomorphology and hydrograph characteristics is quite variable. In addition, the distribution and magnitude of the physical habitat parameters varied between the segments. Fish responses to these variable conditions were also different among the three segments we examined. These factors indicated that segment-specific spawning habitat models may be required for each of the three segments. However, our first steps were to assemble and evaluate various groups of habitat metrics as candidate models, produce a single model that could be used to predict spawning habitat Reach-wide, and assist managers in planning the desired strategy to achieve management goals for fall Chinook production.

We conducted spawning habitat model investigations in order to understand the effects of the current altered hydrograph on spawning habitat availability and persistence, and to develop a tool to evaluate the effect of alternative hydrographs on spawning habitat. We made significant progress in terms of understanding the effect of spatial and temporal variation in physical conditions along the Reach, and we were able to use that knowledge to develop a spawning habitat model that can be applied Reach-wide. We developed metrics quantifying the persistence of suitable conditions across the variable hourly hydrograph and found that these metrics were accurate predictors of redd locations. Based on several metrics that characterized habitat persistence, we found that Chinook were more likely to spawn at locations where hydraulic conditions were persistently suitable rather than at locations where hydraulic conditions were more variable. These results suggest that the quality and quantity of spawning habitat would increase if managed streamflows were stabilized. We also identified other variables that are important for predicting fall Chinook spawning habitat, and gained insight into the level of contrast required to produce reasonable habitat simulations. These accomplishments have provided the foundation for the next steps in our research to refine our current spawning habitat model and/or develop site specific models for each of the important spawning areas in the Hanford Reach.

Results of our fall Chinook rearing habitat studies and modeling were consistent with the detailed studies that have been conducted previously in the Hanford Reach. We found that although rearing habitat varies with streamflow, stability appears to be more important to juvenile Chinook than the absolute flow level. Stable streamflows and habitat conditions require less movement and less energy expenditure than constantly fluctuating streamflows and spatially variable habitat conditions. Stable streamflows would also help to reduce the potential for stranding or entrapment of juveniles.

INTRODUCTION

This report describes studies conducted in 2003-2005 to evaluate the effects of upstream dam operations on spawning and rearing conditions for fall Chinook salmon *Oncorhynchus tshawytscha* in the Hanford Reach of the Columbia River. Specifically, the goals of this study are focused on the effects of actual and alternative streamflows and streamflow variation on 1) adult spawning habitat, 2) juvenile rearing habitat, and 3) entrapment/stranding mortality of juvenile fall Chinook. The study was conducted cooperatively by the U.S. Fish and Wildlife Service (FWS), U.S. Geological Survey (USGS), Washington Department of Fish and Wildlife (WDFW), Nugent GIS and Environmental Services, Columbia River Inter-Tribal Fish Commission (CRITFC), the Fish Passage Center (FPC), Alaska Department of Fish and Game (ADFG), and the Yakama Nation (YN).

An independent peer review was conducted on the draft version of this report utilizing three eminent scientists with extensive experience in quantitative analyses of hydrodynamic and fisheries modeling for river management. Each reviewer provided a slightly different perspective and level of expertise regarding hydrodynamic modeling, fall Chinook biology, life history, and habitat requirements, and fishery issues relating to hydropower development and operations. The first reviewer, Dr. Paul Higgins, is an industry representative from B.C. Hydro who has extensive experience in all aspects of the work conducted for this study. The second reviewer, Dr. William Miller is a consultant with extensive experience in hydrodynamic and habitat modeling. The third reviewer, Dr. Peter Steffler, is a professor at the University of Alberta with extensive expertise in hydrodynamic modeling, and one of the primary developers of the River2D hydrodynamic model. Our goal was to obtain a comprehensive, thorough technical review of all aspects of our work, incorporate the peer review comments into our final report, and obtain a final set of comments from the reviewers on the end product. The peer review process was coordinated by ESSA Technologies. Their report which includes the specific comments from each reviewer as well as our responses and disposition of the comments, is included as Appendix A.

The Hanford Reach near Richland, Washington, is the last significant unimpounded portion of the mainstem Columbia River still accessible to anadromous fish, and it supports the largest and most productive population of wild salmon remaining in the Pacific Northwest (Huntington et al. 1996; Dauble and Watson 1997). These large, mainstem-spawning fall Chinook are a cornerstone in efforts to preserve and restore widely depleted and at-risk Columbia Basin salmon stocks. Hanford fall Chinook are one of the few remaining Columbia River populations that have not warranted listing under the Endangered Species Act. This population is considered a critical “core population” of fall Chinook salmon that may re-colonize nearby tributaries and mainstem areas (ISAB 2000). Hanford fall Chinook are also the primary stock supporting Columbia River Treaty Indian subsistence and commercial fisheries as well as non-Indian sport and commercial fisheries. This stock makes significant economic contributions throughout the Pacific Northwest with ocean, sport and commercial fisheries through Canada and as far north as southeast Alaska.

The Hanford fall Chinook population remains strong because critical spawning and rearing habitats in the unimpounded Hanford Reach are largely intact (ISAB 2000). However, construction and operation of the Columbia River hydropower system, including Priest Rapids Dam, has reduced population productivity. Of particular concern are the effects of seasonal and daily streamflow regulation and operation of upstream dams on the quantity, quality and persistence of habitat available for spawning and rearing fall Chinook. Daily flow fluctuations have also artificially increased mortality by dewatering redds during the fall and winter, and stranding or entrapping juvenile salmon during spring and summer.

In accomplishing the goals of this study, we will make progress towards a better definition of the Hanford Reach's production potential and limitations, and help identify effective protection, restoration, and management alternatives for fall Chinook. This information has application in the Federal Energy Regulatory Commission (FERC) relicensing process for the Priest Rapids hydroelectric project. Study results will help guide development of sustainable escapement goals and fisheries by the Pacific Salmon Commission, Pacific Fisheries Management Council, and Columbia River Fish Management Plan under *U.S. v. Oregon*. Stock productivity improvements that may result from effective application of study results will ultimately lead to significant conservation and fishery benefits including improvements to the aquatic community as a whole.

Study Area

The Hanford Reach is located on the mid-Columbia River in south-central Washington State. The Reach extends from Priest Rapids Dam at river kilometer (rkm) 639 downstream for 90 km to the head of McNary Pool near Richland, Washington (Figure 1). On June 9, 2000, Presidential Proclamation 7319 established the 78,900 hectare Hanford Reach National Monument (Monument) which includes the Columbia River. This designation continues the protection of the Hanford Site and Reach that began during World War II when the Hanford Nuclear Reservation was established for the production of weapons-grade nuclear materials. The FWS co-manages the Monument under existing agreements with the United States Department of Energy.

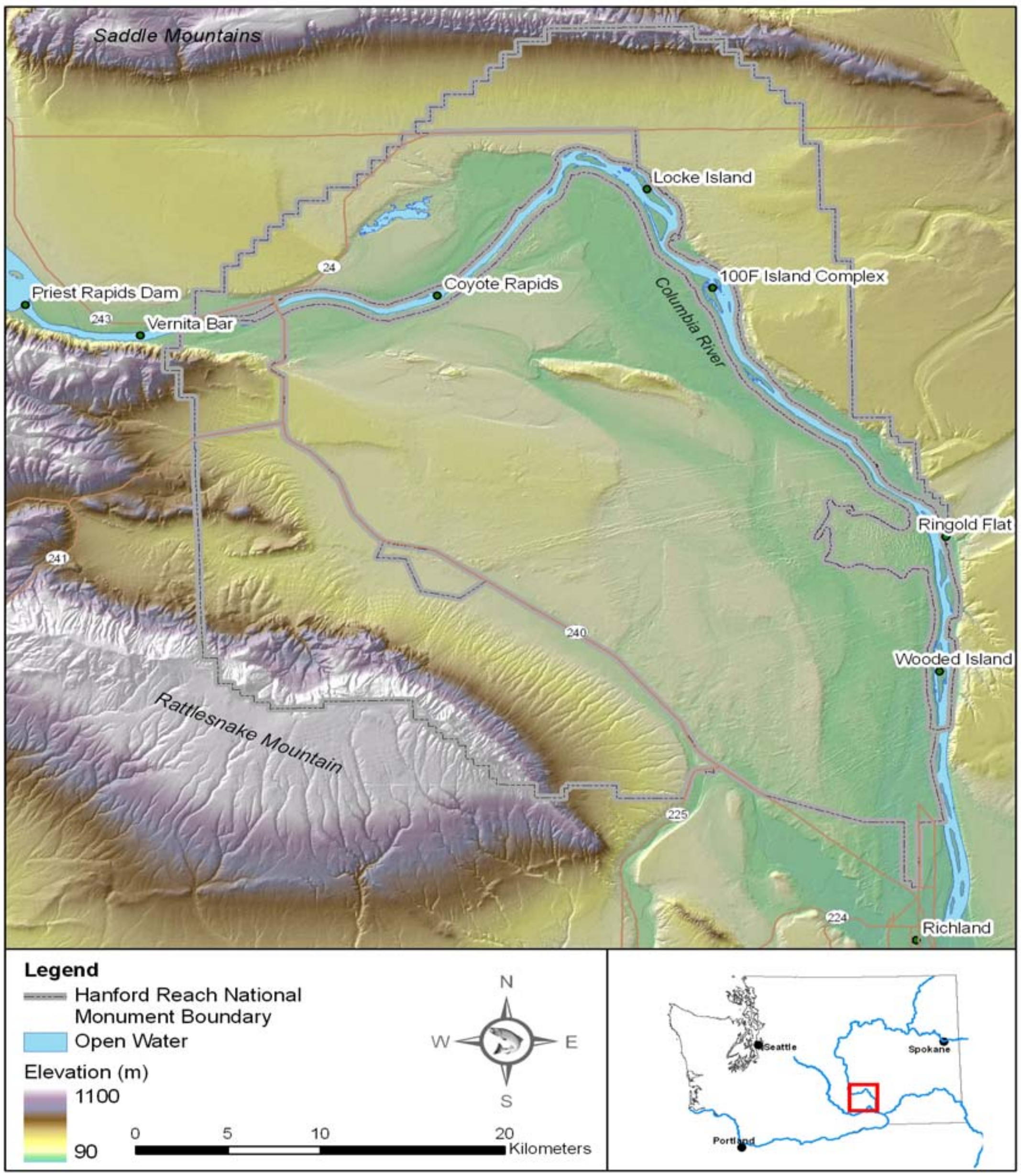


Figure 1. General location of the Hanford Reach of the Columbia River.

The Hanford Reach lies within the Columbia Basin, the hottest, driest part of Washington State (Franklin and Dyrness 1973). The Columbia Basin is a product of Miocene basalt volcanism that occurred over the past 17 million years. Present day geomorphology of the Columbia Basin, however, is largely a result of the repetitive cataclysmic flooding from the Bonneville and Missoula floods (O'Conner 1993). Within the Hanford Reach, the Columbia River runs unconfined by bedrock and flows over uniform layers of fluvial sediments (Reidel et al. 1994). A relatively small portion of the upper segment is confined by bedrock (4 km or 4.4 % of the Reach), but no spawning has been documented there and this area has been excluded from our habitat modeling due to the complicated basalt bathymetry and lack of biological significance for fall Chinook. Fluvial sediments, which are the youngest and most abundant in the Hanford Reach, lay on top of the older basalt layer and the lake deposits known as the Ringold formation (Jenkins 1922). In fact, substrates in the Hanford Reach are believed to be sufficiently coarse so as to resist movement by streamflows as high as the regulated 100-year discharge of 440 kcfs (Dauble & Geist 2000). As such, this lack of bed mobility has resulted in an extremely stable thalweg with little lateral migration (Hall 1988).

The Hanford Site was established in 1943 for the Manhattan Project of the United States Department of Defense. Since that time, very little development along the riverine corridor has occurred with the exception of several water intake stations built in the 1940s for reactor cooling. No levees or other manmade structures have affected the river channel to any degree. However, irrigation of agricultural lands just to the north of the Columbia River near White Bluffs has resulted in significant slumping into the river between rkm 597 and 600, as well as other areas to a lesser degree.

The river channel itself runs east, northeast, southeast and then south past Richland, Washington where the river is backwatered by McNary Dam and Reservoir. The river flows unimpounded for approximately 90 rkm and with a longitudinal slope of 0.2 m/km (Dauble & Geist 2000). Bed slope, Reach-wide, averages 2.4 %. At a steady discharge of 90 kcfs, the average depth is just over 4 m and average velocity is about 1 m/s. The typical average daily flow during the spawning season is approximately 90 kcfs. This flow is in the lower range of flows that occur during the rearing season. For a range of discharges from 30 to 400 kcfs, the average change in water surface elevation for every 10 kcfs increment, is approximately 0.2 m. Comparatively, the Columbia River within the Hanford Reach is very wide, flat, and shallow (Figure 2). Channel widths vary from 220 m in single, simple channels, to 1400 m over the widest island complex.

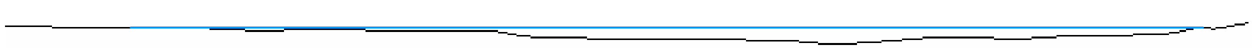


Figure 2. Plot of the river channel depicting the typically wide flat channel configuration with equally proportioned axes. This channel section is approximately 600 m wide and would have an average depth of approximately 4 m.

Hanford Reach Fall Chinook

The mainstem Columbia River historically supported at least eight major fall Chinook spawning areas, extending from rkm 235 – 1124 (Dauble et al. 2003). Today, the Hanford Reach is the only significant vestige of what once was a highly-productive natural system for fall Chinook populations. At the turn of the 20th century, mainstem fall Chinook populations were a major contributor to the 10 to 16 million salmon returning annually to the Columbia River (ISAB 2000). The present annual return of salmon to the river is approximately 1 million fish, the majority (>80%) of which are produced artificially in hatcheries (ISAB 2000). Development of the hydroelectric system was estimated to have decreased historical abundance by 5 to 11 million fish annually (ISAB 2000), and today wild salmon abundance is approximately 1% of predevelopment abundance (National Research Council [NRC] 1986).

The population of fall Chinook salmon that spawns in the Hanford Reach is one of several mainstem-spawning fall Chinook populations referred to as upriver brights (URB) by fishery managers (ODFW and WDFW 2003). A small population of URBs spawns in the Snake River between Lower Granite and Hells Canyon dams. URB fall Chinook are released from Priest Rapids and Ringold Hatcheries on the Columbia River and Lyons Ferry Hatchery on the Snake River. The Hanford Reach population typically represents between 80 and 90% of combined hatchery and wild numbers of Columbia River's URB fall Chinook salmon counted at McNary Dam (Dauble and Watson 1997). These wild fish are remnants of much larger populations which spawned in Columbia and Snake River habitats flooded or blocked by dams (ISAB 2000). NOAA-Fisheries has classified the Snake River population as a separate ESU and listed it as Threatened under the Endangered Species Act (ESA) in 1992. Hanford fall Chinook are part of an upper Columbia summer and fall Chinook ESU for which listing was deemed to be not warranted in 1998.

Returns of adult fall Chinook past McNary Dam were relatively stable at 20 to 50 thousand fish per year from 1960 through the early 1980s as fisheries were managed to harvest additional numbers (Figure 3). Harvest rates on bright fall Chinook declined in the late 1970s with reductions in ocean and inriver fisheries to protect weak stocks including Snake River bright fall Chinook. The combined effects of reduced fisheries and excellent ocean survival in years following the 1982-83 El Niño, increased McNary Dam counts to over 150,000 and Hanford spawning escapements to nearly 90,000 adults in the late 1980s. Throughout much of the 1990's, spawning escapement was depressed, ranging between 25,000 and 50,000 adults as a result of poor ocean conditions. In 2000, indices of ocean conditions (upwelling and Pacific Decadal Oscillation) were good, indicating favorable conditions for salmonid productivity. These favorable ocean conditions along with continued harvest restrictions are likely responsible for the high escapement to the Hanford Reach (89,000 adults) observed in 2003. Spawning escapements during recent years have produced between 8 and 28 million juvenile fall Chinook salmon which rear in the Reach during spring and early summer (Nugent et al. 2002).

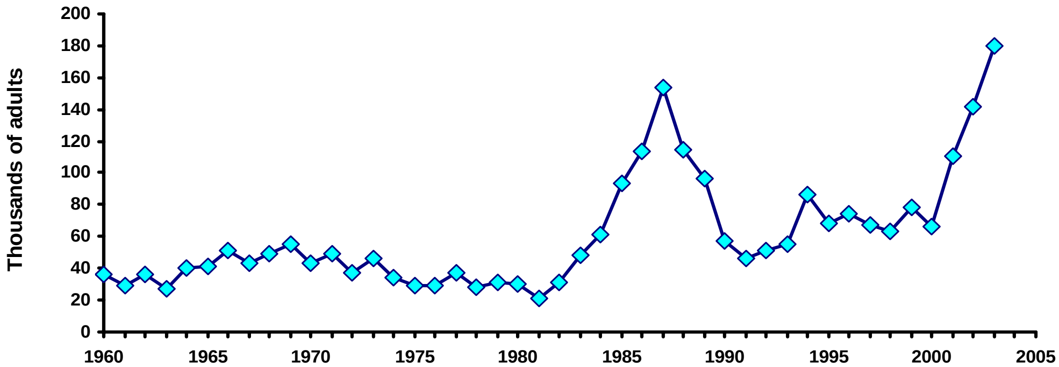


Figure 3. Counts of upriver bright fall Chinook at McNary Dam, 1960-2004. (Includes wild and hatchery Columbia and Snake river fish.)

Hanford fall Chinook have a major impact on fisheries in the Columbia River and Pacific Ocean. The URB stock of fall Chinook is a far north migrating stock and provides significant harvest in ocean fisheries off Southeast Alaska and British Columbia. Between 1985 and 2001, URBs comprised about 16% of the Chinook salmon catch in Southeast Alaska, 9% of the North British Columbia catch, 7% of the Central British Columbia harvest, and 10% of the West Coast Vancouver Island catch (Pacific Salmon Commission 2002). Under the jurisdiction of the Pacific Salmon Treaty, these fisheries are regulated in part, to limit catches on upriver bright fall Chinook which are an indicator stock for the Pacific Salmon Commission.

The URB stock is the backbone of the non-tribal fall season sport and commercial fisheries, and ceremonial, subsistence and commercial treaty Indian fisheries of four Native American tribes in the mainstem Columbia River. Tribal and non-tribal fisheries are regulated to meet guidelines and limits of the Columbia River Fish Management Plan (CRFMP), the Endangered Species Act (ESA), and management agreements negotiated by Parties to the *U.S. v. Oregon* court case (ODFW and WDFW 2000). In 1999, a management goal of 46,000 adult fall Chinook upstream of McNary Dam was set to provide for a sport fishery in the Hanford Reach, hatchery broodstock collection above McNary Dam, and an interim escapement goal of 40,000 naturally spawning fall Chinook (ODFW and WDFW 2000).

It is recognized by fishery managers that the stock-recruitment-based interim escapement goal of 40,000 naturally spawning adult fall Chinook was developed with limited data during a time frame when the Hanford Reach URB population was adjusting to significant hydrosystem management events (e.g., construction of John Day Dam and filling of the reservoir in 1968 displaced a portion of this substantial, naturally spawning population from the John Day reservoir area to the Hanford Reach and other areas). In addition, the historic escapement range of naturally spawning fall Chinook used in the stock-recruitment analyses for the interim goal was relatively narrow compared to recent higher escapements, which may have biased the interim escapement goal low. It should also be noted that the interim escapement goal was developed during a period when the hydrosystem was managed primarily for hydropower production, and these conditions may not have been conducive for optimum productivity of the URB population during the freshwater life stage. In fact, in a recent study conducted by Pacific Northwest National Laboratory (PNNL), the carrying capacity of the Hanford Reach in terms of spawners was estimated to be between 74,000 and 90,000 using spatial metrics obtained from aerial photography and analyzed in a Geographic Information System (GIS) (Visser 2000). Data collection and analysis conducted for this study will provide significant new information to evaluate URB fall Chinook productivity and the Reach's carrying capacity relative to various hydrosystem management options. This information will play a critical role in determining whether an update of the current interim URB escapement goal is warranted.

Other Fishery Resources

The Reach provides productive spawning and rearing areas, and serves as a migratory corridor for many species of anadromous and resident fishes. Anadromous fishes include spring, summer, and fall Chinook, coho, and sockeye salmon; steelhead; and Pacific lamprey. White sturgeon, formerly an anadromous species, has suffered from passage problems at Columbia River dams and is now present in the Reach as an isolated, fragmented population. Native resident fish species that have been observed in, or use the Reach include rainbow trout, bull trout, cutthroat trout, mountain whitefish, northern pikeminnow, sand roller, other minnows, suckers, and sculpins.

Hydro Operations Affecting the Hanford Reach

Priest Rapids Dam at the head of the Hanford Reach is the most downstream of a seven-dam hydroelectric complex on the mid-Columbia River that includes Wanapum, Rock Island, Rocky Reach, Wells, Chief Joseph, and Grand Coulee dams (Figure 4). This complex is operated under a power-peaking or load-following mode to meet electrical demand in the Pacific Northwest. Hydropower generation through these projects largely governs streamflow in the Hanford Reach. The mid-Columbia projects are part of the larger Columbia River hydropower system and are operated under an international treaty and other agreements that affect river flows and fish resources. These include the Columbia River Treaty between the United States and Canada, the Pacific Northwest Coordination Agreement, Mid-Columbia Hourly Coordination Agreement (HCA), and the Vernita Bar Agreement. The HCA and Vernita Bar Agreement, established as FERC license conditions for the Priest Rapids Project, have the most direct effect on river flows and fluctuations in the Reach.

Before the construction of major dams and water storage projects, Columbia River streamflows near the site of the Priest Rapids Project were lowest during the winter (December-March). Snowmelt gradually increased streamflows in the spring and early summer and peak flows normally occurred in June (Figure 5, 1918 – 1940). Flows then gradually decreased in the fall and returned to low winter flows. Little daily or hourly fluctuation in streamflow occurred under pre-dam conditions.

Completion of the Columbia River hydropower and flood control system occurred in several stages, and eventually altered the seasonal flow pattern across the annual hydrograph. Grand Coulee was the first, relatively large storage project, constructed in 1942. The operation of the project for both power and flood control did not substantially alter the seasonal hydrograph (Figure 5, 1945 – 1970). With the addition of large storage projects over the next 30 years, including the massive Mica Project in Canada in 1973 (storage – 20 million acre-feet), peak spring flows were reduced, average minimum flows were increased, and the period of lowest flows was shifted from winter to autumn (Figure 5, 1979 – 2005). Before completion of the Columbia River dam and reservoir system, peak June streamflows averaged 328 kcfs and lowest flows in February averaged 41 kcfs. Peak flows (mean monthly) in June have now been reduced to an average of 165 kcfs while the lowest average monthly flow, which now occurs in September, has increased to 84 kcfs (Figure 5).

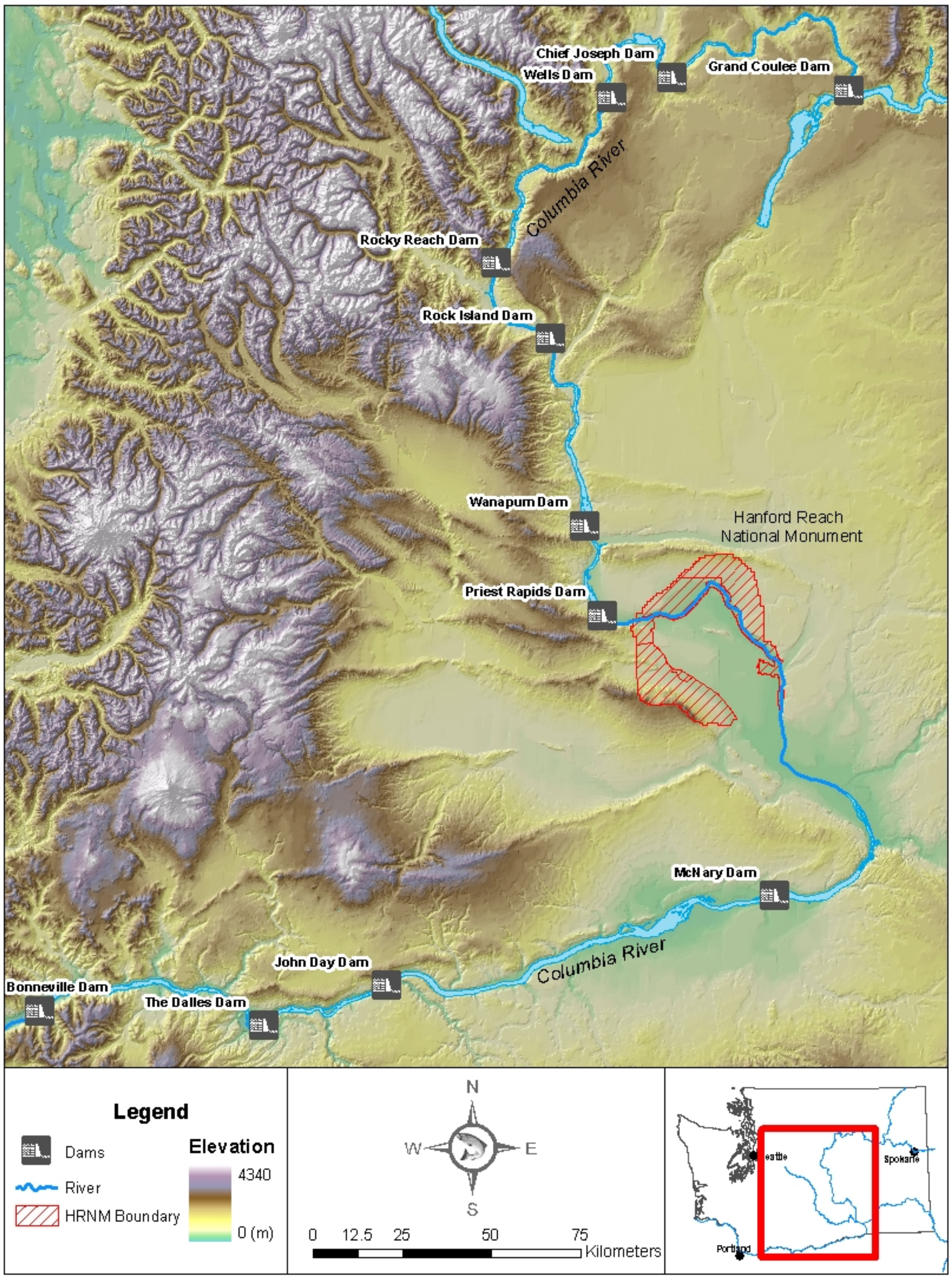


Figure 4. Major dams of the lower and mid Columbia River.

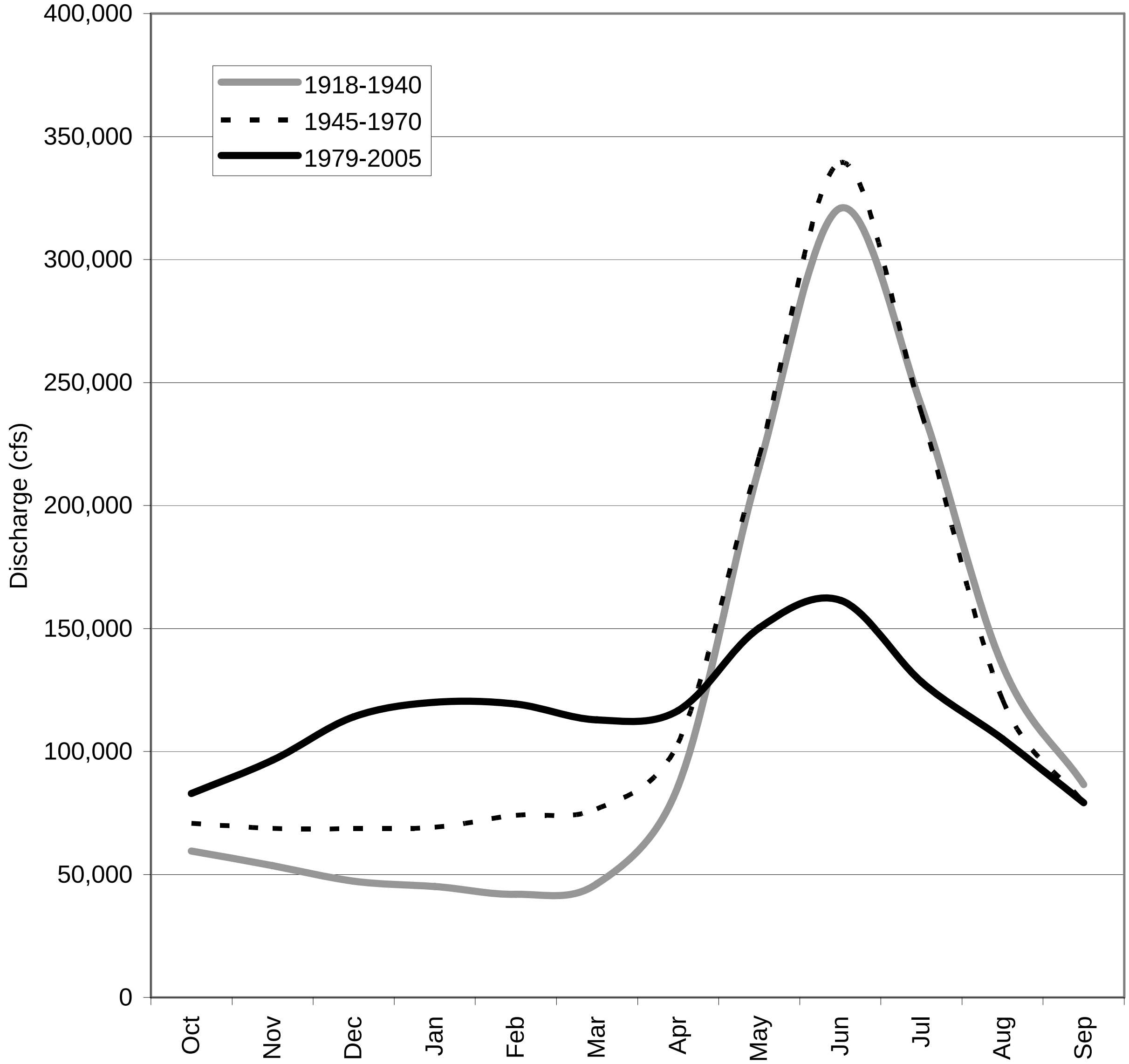


Figure 5. Historic and current hydrographs as measured at the USGS Priest Rapids Gage #12472800 located downstream from Priest Rapids Dam in the Hanford Reach.

A significant amount of inter-annual variation in the hydrograph still occurs (Figure 6), although it is much reduced compared to the variation that was present before the large storage projects developed the capacity to capture and re-distribute high spring flows to other parts of the year. Much of the variation occurs from May through July and is associated with characteristics of the winter snow pack. Both water supply and weather conditions (power demand) contribute to hydrograph variation during the winter months of December through early March.

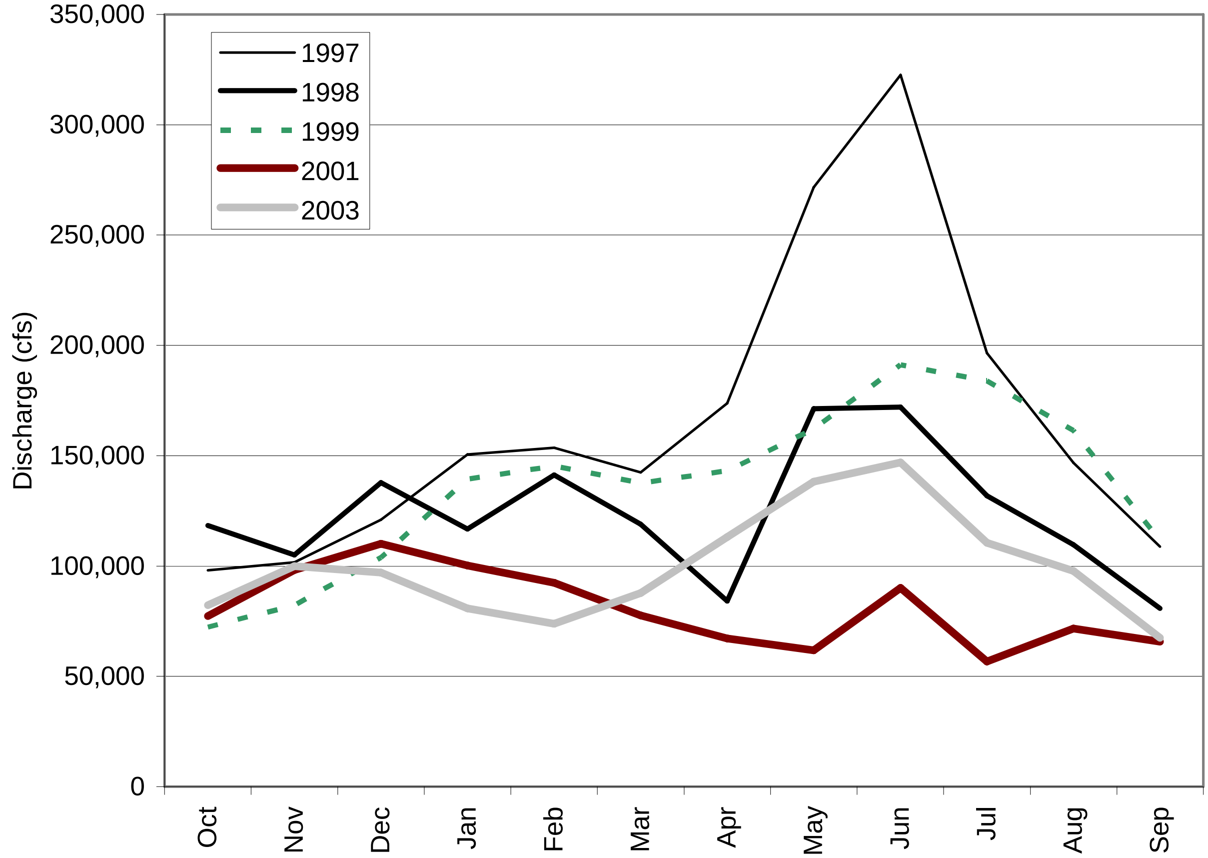


Figure 6. Annual mean monthly hydrographs as measured at the USGS Priest Rapids Gage #12472800 located downstream from Priest Rapids Dam in the Hanford Reach for a range of water years from the maximum runoff year (1997) to the 98% exceedance probability year (2001) from the period of record between 1970 and 2005.

Operation of the mid-Columbia River projects to meet power demand (load-following) has resulted in both seasonal and diel modifications and variation to the Columbia River hydrograph. The primary seasonal modifications that have occurred consist of storing, and thus reducing spring flows to avoid flooding, and then releasing the stored water during the winter for power production and increasing flows when power demand is highest (Figure 5, Figure 6). Diel modifications that have occurred reflect the hourly manipulations that are conducted to follow the daily cycle of power demand. This cycle generally consists of releasing water during daylight hours when power demand is highest (high flows), then storing water at night when power demand is the lowest (low flows). This type of load-following operation results in hourly and daily fluctuations in discharge from Priest Rapids Dam throughout most of the year, including during the spawning (Figure 9), incubation, emergence, and rearing periods (Figure 7) for fall Chinook salmon. Typical project operations result in fluctuations as great as 2.1 meters/hour and four meters in a 24-hour period in the Priest Rapids Dam tailrace during the fall Chinook salmon emergence and rearing period (Nugent et al. 2002).

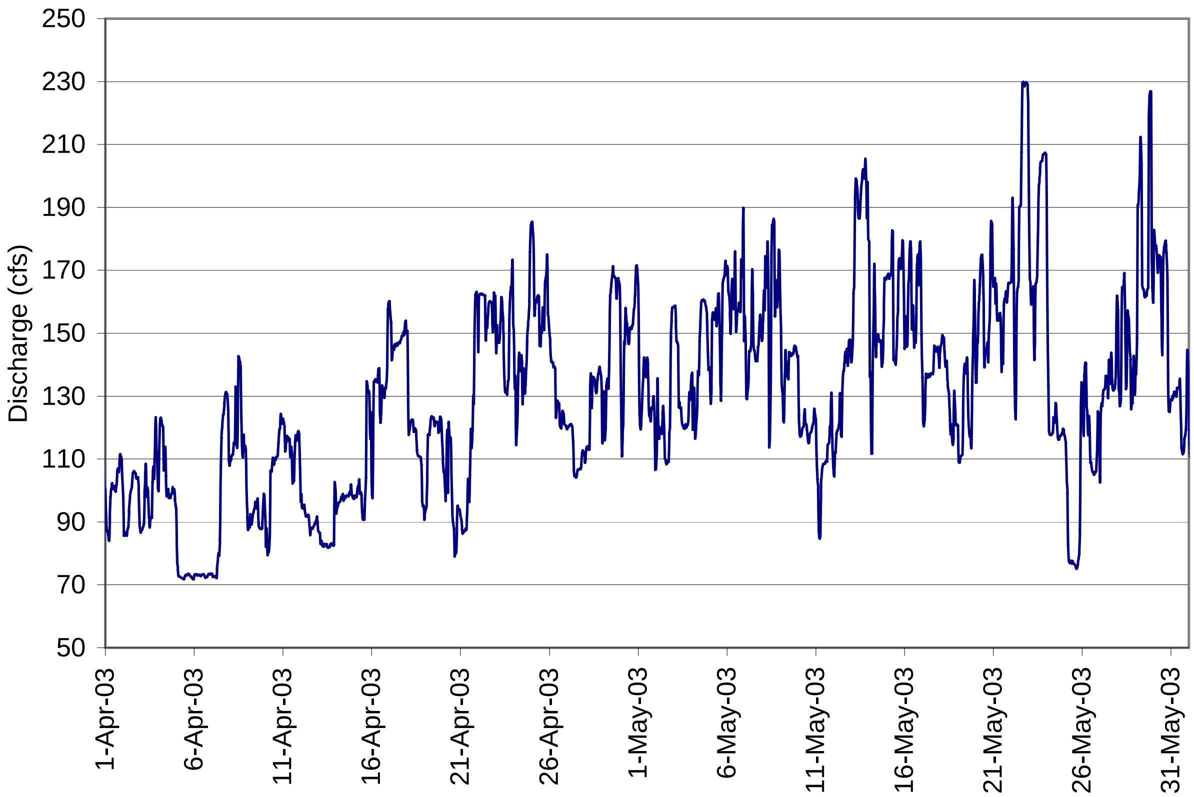


Figure 7. Hourly discharge values as measured at the USGS Priest Rapids Gage #12472800 located downstream from Priest Rapids Dam in the Hanford Reach for a typical spring rearing season when juvenile fall Chinook are present and susceptible to stranding and entrapment. This hydrograph is an example of the diel effect of load following during the spring of 2003.

The major impacts of streamflow fluctuations on fall Chinook salmon in the Reach include reducing the amount of suitable spawning habitat, dewatering of established redds, stranding and entrapment of juveniles, and a reduction in primary and secondary productivity in nearshore rearing areas. Following development of the hydrosystem, load-following operations created variable and sporadic fluctuating streamflows when fall Chinook salmon were spawning in October and November. Subsequently, at night and on weekends when electrical demand was low, flows dropped and salmon redds were dewatered causing mortality of incubating eggs and alevins as well as disturbance to spawning fish. During spring, after emergence had occurred, fluctuating streamflows (Figure 7) caused the stranding and entrapment of rearing juveniles. Stranding occurs when fish are trapped on or under streambed substrates as water elevation drops. Fish are entrapped when they stay in pools that become isolated as river levels decline. Fish mortality occurs primarily when fish are stranded, and when entrapments drain, or shallow water in entrapments heats up causing thermal mortality (Figure 8). Fish are also lost to predators in small, shallow entrapments (Nugent et al. 2002).

Figure 8. Top left - Typical entrapment as seen from the ground; Top right - Series of entrapments visible from Orthophotography; Middle left and right - Stranded Chinook; Bottom left and right – Entrapped Chinook, thermal mortalities.

Adult Spawning Measures

With the construction of Priest Rapids dam in 1959, discharges in the Hanford Reach began fluctuating daily during the spawning period in October and November. Repeated observations of dewatered salmon redds led to efforts to develop an operating agreement to reduce the impacts of streamflow fluctuations on Chinook salmon spawning and egg incubation. In 1988, the Vernita Bar Settlement Agreement was signed by the power-producing entities, fishery agencies and Indian tribes. This Agreement was the major formal operation to protect fall Chinook salmon that spawn in the Vernita Bar area of the Hanford Reach. The Agreement expired on October 31, 2005 when the existing FERC license for the Priest Rapids Project expired. The Agreement included flow manipulations to minimize fall Chinook salmon spawning above the water elevation that would occur at a streamflow of 70 kcfs at Vernita Bar. Streamflow is manipulated by using the Mid-Columbia Hourly Coordination Agreement and reverse load following at the Priest Rapids Project. The intent of reverse load following is to limit Chinook salmon spawning (which was thought to occur mainly during daylight hours) to lower elevations on Vernita Bar by reversing the normal load following pattern and providing low flows during the day, and higher flows at night. The Vernita Bar Agreement of 1988 was recently replaced by the Hanford Reach Fall Chinook Protection Program (HRFCPP) which maintains the same general reverse load following operation during the spawning season, but has included additional operational measures during the spring rearing season.

While the Agreement was an important step in protecting Chinook salmon redds on Vernita Bar from dewatering, several other important factors associated with fall Chinook spawning in the Hanford Reach were not included:

- There was no provision in the Agreement to consider the effect of streamflow management at Vernita Bar on the other significant spawning areas in the Reach;

- There was no provision in the Agreement to monitor spawning at the other significant spawning areas (initiation and end of spawning specifically);

- There was no provision in the Agreement to consider the effect of the resulting streamflows on the amount of spawning habitat available;

- There was no provision in the Agreement to adjust streamflows and habitat availability to accommodate different levels of expected spawning escapement;

- There was no provision to limit the amplitude of streamflow fluctuations during nighttime hours;

The Vernita Bar Agreement and current HRFCPP are not based on a quantitative assessment of streamflow requirements for spawning sites throughout the Reach. Rather, the sole purpose of these plans was to protect salmon redds on Vernita Bar from being dewatered by reductions in streamflow during the incubation and emergence periods. The Agreement was based in part on research conducted by Chapman et al. (1986) solely at the Vernita Bar spawning area which covers approximately three km of the 90 km-long Hanford Reach. The majority of spawning occurs elsewhere in the Reach, near areas such as Locke Island at rkm 596-600 (Dauble and Watson 1990) where considerable streamflow fluctuations still occur 40 rkm downstream from Priest Rapids Dam (Figure 9). In 2004, Vernita Bar only accounted for 27% of the peak redd counts within the Reach. Prior to the current study, no quantitative research had been conducted to assess the effects of streamflows structured under the Agreement on downstream spawning areas.

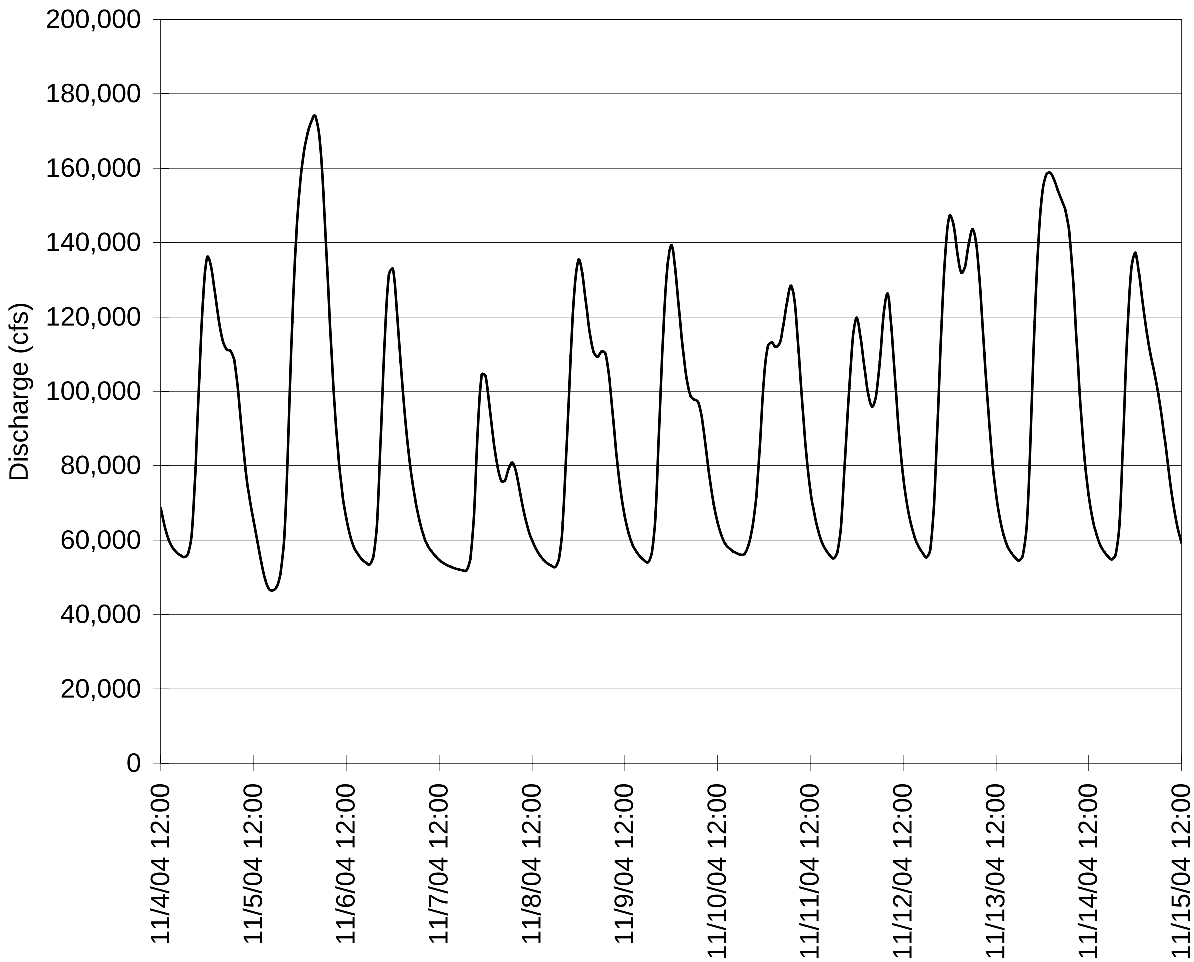


Figure 9. Hourly discharge values for a fall spawning season at Locke Island (White Bluffs) in the Hanford Reach of the Columbia River. This hydrograph is representative of the reverse load following operation that is implemented each year under the Hanford Reach Fall Chinook Protection Program.

The former hydro operations were based on spawning habitat assessments at Vernita Bar at a time when redd counts were about half of current levels. As a result, those operations may have artificially constrained fall Chinook production, future population sizes, and numbers of fish available for harvest. Recent limitations on fishery exploitation rates to protect weak ESA-listed stocks including Snake River bright fall Chinook have substantially increased escapement to the Hanford Reach. Increased numbers of spawners may not continue to result in increased numbers of fish for harvest if spawning habitat availability is limited by current discharges and discharge patterns. Hanford Reach fall Chinook have become particularly important to fisheries in the face of widespread reductions in harvest opportunities on other fish stocks resulting from hydro-related impacts and subsequent listings under the ESA.

Hydro operations in the Reach have been based in part on the assumption that fall Chinook only spawn during the day, and as such, only stable daytime flows were provided. Resent research, conversely, has documented spawning 24 hours a day (McMichael 2003). Typical spawning period operations consist of providing nine or ten hours of stable discharge between 50 – 70 kcfs during daylight hours only. At night, streamflows are then increased (reverse load-following) to evacuate Priest Rapids and Wanapum pools. It is assumed that spawning will occur within the daytime portion of this operation. Chapman (1986) determined that although some redds on Vernita Bar could be completed in less than 24 hours, fall Chinook salmon often took 5-7 days to complete redds and spawn. In addition, the Vernita Bar Agreement and HRFCPP do not limit the amplitude or shaping of nighttime flows and these flows can often reach levels as high as 190 kcfs. As this highly-modified and unnatural hydrograph propagates downstream, there is a time delay and a change in the wave shape, dissipating proportionally with the distance traveled downstream. The result is a variable sequence of hourly streamflows for all downstream locations. For instance in 2004, daily streamflows averaged 91 kcfs but fluctuated regularly from 55 kcfs up to 173 kcfs following streamflow management protocols established in the HRFCPP. Dauble (2003) concluded the primary actions required for enhancement of fall Chinook salmon in the Columbia River basin are the establishment of natural flow regimes and the maintenance of geomorphic features common to alluvial floodplains.

Juvenile Protection Measures

Observations of the stranding and entrapment of large numbers of juvenile fall Chinook salmon caused by water level fluctuations in the Reach (Page 1976, duri9 et al. 1981, DeVore 1988, Geist 1989, Wagner 1995, Ocker 1996) led to a plan for operations during the spring rearing period to reduce the observed impacts. While the fishery managers and the Independent Scientific Advisory Board to the Northwest Power Planning Council recommended that streamflows remain stable during the juvenile Chinook stranding susceptibility period (ISAB 1998), the hydro operators found this operation too costly. As an alternative, the Hanford Reach Juvenile Fall Chinook Protection Program was developed in 1999 to "limit" streamflow fluctuations during the spring rearing period. Studies were conducted from 1998-2003 to quantify the impact of flow fluctuations prior to implementation of the program (1998), and to quantify and monitor the impacts of flow fluctuations following implementation of the program (1999-2003). These studies and the associated loss estimates led to the creation of the 2004 Vernita Bar Plus Agreement (officially the Hanford Reach Fall Chinook Protection Program) that allows for reverse load-following when fall Chinook are spawning, and flow fluctuations within specified flow limits during the fall Chinook rearing period. However, there are no hard constraints requiring the operator to stay within the flow limits during the rearing period, and they are frequently exceeded.

Limitations of previous stranding/entrapment study designs may have led to underestimates of stranding and entrapment impacts and inadequate protection measures for rearing habitat and juvenile fish. In part these limitations resulted from inherent difficulties in quantifying stranding and entrapment due to the large size of the affected area and the frequent and irregular flow fluctuations typical in the Reach. Previous estimates of fish losses were considered to be minimal since they were only calculated for roughly one third of the Hanford Reach. Sampling efficiency was assumed to be 100% even though it was not possible to account for losses to predators and scavengers or to account for fish that were not collected by researchers (Nugent et al. 2002). Daylight sampling protocols overlooked some dewatered areas that had already been re-inundated. The sampling design also inadequately represented the larger entrapment pools. Detection probabilities were low in areas where sampling was problematic (e.g., rock cobble where fry are difficult to detect, even when present). Finally, loss estimates were based on a random, area-based sample design that produced highly uncertain estimates, in part because of the expansion effects of high sample variance. This high sample variance was due to the wide variation of fish densities observed by habitat type. Entrapment pools had relatively high densities of fish, whereas areas of potential stranding had zeros or very low densities of fish.

Study Approach

This study builds upon earlier work to address information gaps and other limitations on our understanding of the effects of streamflow and streamflow variation on entrapment/stranding mortality of juveniles, adult spawning habitat, and juvenile rearing habitat. Our study does not attempt to account for or enumerate true stranding. However, we do believe true stranding is a significant factor contributing appreciably to mortality and should be examined in detail with future assessments.

Effects of fluctuating flows on entrapment and stranding are based on a temporal and habitat-stratified sampling approach that focuses on entrapments throughout the entire Reach and across the rearing period. Previous studies indicated that the majority of the problem was associated with entrapment rather than stranding but the random, area-based study design caused most sampling in previous studies to occur in stranding rather than entrapment habitats. Our entrapment-based approach is a significant improvement over past designs which were hampered by low fish sampling probabilities and also limited to only a portion of the Reach. Our approach provides an assessment of the effect of streamflow fluctuations on rearing fall Chinook salmon throughout the entire Reach with a higher degree of confidence, particularly in those sections that were not previously studied.

Effects of steady flows and flow fluctuations on adult spawning habitat are evaluated based on the distribution of spawning habitat throughout the Reach and the variation in physical characteristics including parameters related to hourly streamflow levels and streamflow fluctuations. This information is used to build a multivariate logistic model which could explain patterns of Chinook spawning habitat selection and includes variables that explicitly account for the dynamic nature of the physical conditions. The model is then used to evaluate the effects of streamflow and flow variability on spawning habitat within the Hanford Reach. The model will provide quantitative information on the relationship between river flows and flow fluctuations, and spawning habitat. While previous studies have provided valuable information for management of the Reach's fall Chinook salmon, they have been limited for the most part to Vernita Bar for spawning flow management, and to the middle third of the Reach for rearing flow management and stranding/entrapment mortality estimates.

Effects on juvenile rearing habitat are evaluated based on juvenile distribution, behavior, habitat preferences and comparisons of usable habitat at different streamflows. Seasonal changes in the relative abundance, distribution, and length-composition of fall Chinook in nearshore habitats are based on seining surveys. Underwater video surveys are used to examine fall Chinook diel behaviors that may lead to stranding or entrapment. Rearing habitat use was described with an existing model developed by Tiffan et al. (2002). Effects of streamflow on rearing habitat availability are based on simulations with a combination of streamflow, habitat, and habitat use models.

A diagram describing the analytical approaches that were used, including inputs, tools, outputs, and synthesis of results is provided in (Figure 10). These assessments capitalize on recent advances in habitat mapping, computer modeling technology and remote sensing to describe habitat conditions in the Reach and to evaluate the physical and biological effects of streamflow and streamflow fluctuations. The entrapment/stranding evaluation and spawning and rearing habitat assessments relied on two hydrodynamic models. MASS1 (Modular Aquatic Simulation System 1D) is a one-dimensional flow model developed by the Department of Energy's (DOE) Pacific Northwest National Laboratory (PNNL) to estimate hourly water surface elevation and streamflow profiles throughout the Hanford Reach based on Priest Rapids discharge and other inputs. River2D is a depth-averaged flow model developed by the University of Alberta to estimate water surface elevations, depths, and two-dimensional velocities. Hydrodynamic modeling methods and analyses are presented in the first chapter of this report followed by chapters on the entrapment/stranding evaluation, spawning habitat assessment, and rearing habitat assessment.

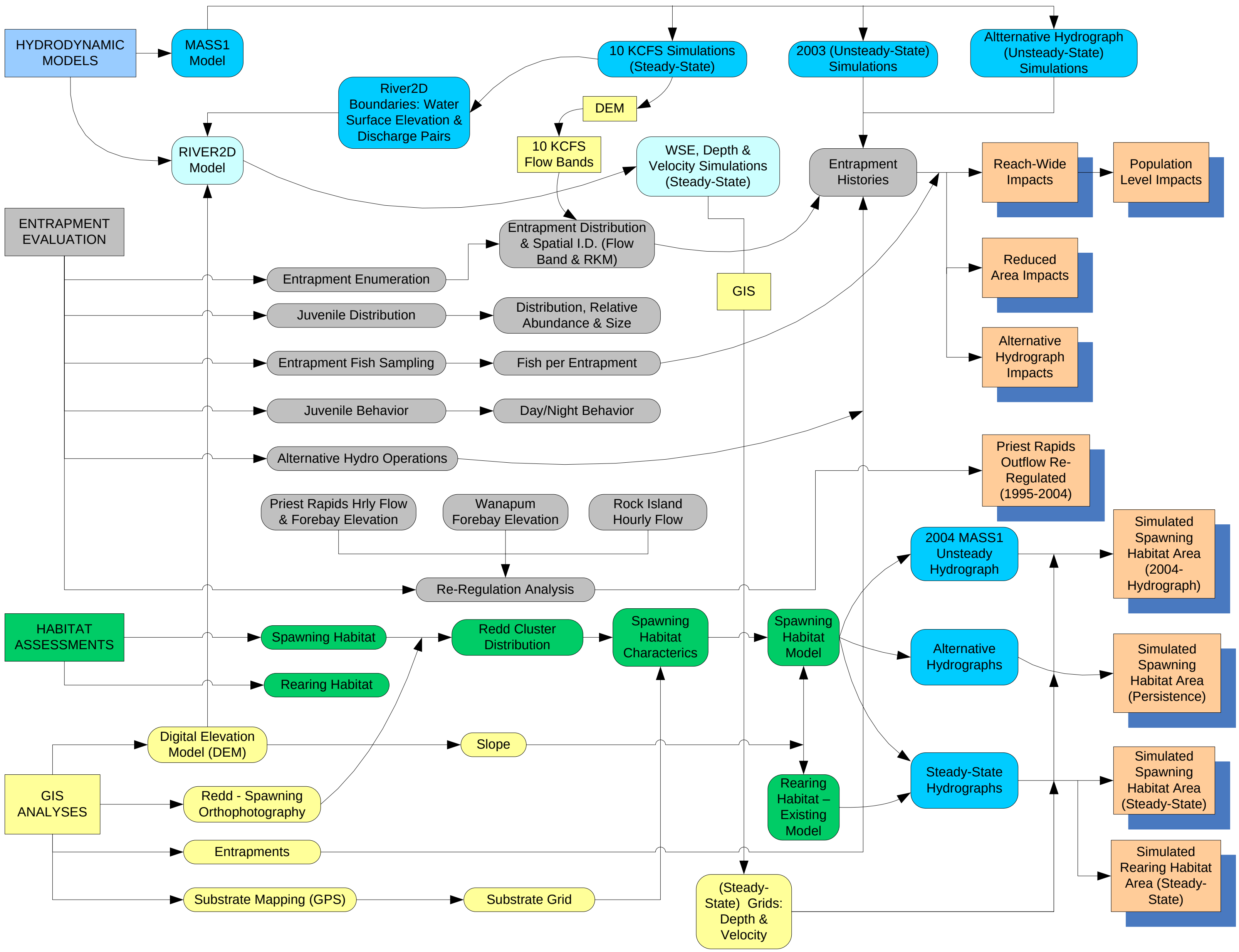


Figure 10. Flow chart of analytical approaches used for Hanford Reach Studies.

HYDRODYNAMIC MODELING

Note to the reader: All measurement units with the exception of discharge are metric.

Streamflows in the Hanford Reach are highly regulated and reflect manipulation of the hydrograph for flood control and power production. Both seasonal and daily streamflows are managed to produce an economical power supply for the Pacific Northwest and provide some measure of protection for fish and wildlife resources (NPPC 2003). In order to evaluate the effects of this type of streamflow manipulation on fall Chinook salmon in the Hanford Reach, a quantitative, spatial description of the physical and hydraulic conditions, including variation, associated with the range of streamflows that can occur in the Reach was required.

At the most resolute scale our models (primarily River2D) simulate spawning habitat metrics (depth and velocity) near the maximum reported fall Chinook redd size, approximately 100 m^2 or 10 X 10 m cell. While our bathymetry is twice as resolute, 100 m^2 is a biologically justified goal and a realistic goal considering the scale of the entire Hanford Reach (90 km). The area (ha) of individual redd clusters represents the actual target scale for which our models will be required to simulate spawning habitat. Redd clusters are aggregations of individual redds (Geist and Dauble 1998) and are typically composed of hundreds of redds. In 2004, we calculated a mean redd cluster size of 6.38 ha or 63,800 m^2 using aerial photos of redd clusters digitized in our GIS.

For our evaluation of the effects of streamflow variation on juvenile Chinook, it was important to capture the rise and fall of the river both as a function of time, and also as a function of distance downstream from Priest Rapids Dam. We used an existing model (MASS1) in an unsteady mode on a one-half hour time step to produce output at 0.6 km intervals throughout the Reach.

The objective of our hydrodynamic modeling tasks was to describe the physical and hydraulic conditions (depths, velocities, water surface elevations, slopes and shoreline location and area) for a wide range of steady and unsteady streamflows throughout the 90 km Hanford Reach. Results of this hydrodynamic modeling effort were then used to conduct an evaluation of fluctuating streamflows and entrapment of juvenile fall Chinook salmon, and for an assessment of spawning and rearing habitat variation.

Although a total of seven hydroelectric and flood control projects regulate streamflows through the mid-Columbia and Hanford Reach (Figure 4, above), the Priest Rapids Project at the upstream end of the Reach determines the actual, short-term streamflow patterns that influence physical conditions downstream. Longer term seasonal streamflow patterns are a function of flood control manipulations that result from reservoir rule curves, water storage agreements between the United States and Canada under the Columbia River Treaty and several Non-Treaty Storage Agreements, and management of annual runoff and water supply for both power production and to benefit fish species listed under the ESA. Short term streamflow patterns, those occurring within a 24 hour period, are managed primarily to meet power demand (load following), and can result in tailwater elevations below Priest Rapids Dam that fluctuate as much as four vertical meters in four hours, and two vertical meters per hour (USGS, gaging station

12472800, unpublished data). Streamflows through the Reach can range from as low as 36 kcfs (FERC license minimum) during low flow time periods, to higher than 450 kcfs during the spring freshet. A spatial and temporal, quantitative description of depths, velocities, and water surface elevations associated with this range of streamflows was required to conduct the entrapment evaluation and habitat assessment for fall Chinook salmon in the Hanford Reach and serve as a foundation for future habitat assessments.

General Approach

We used two types of hydrodynamic models to produce the detailed physical output required for the entrapment evaluation and habitat assessments. MASS1, a one-dimensional flow model, was used to produce both steady and unsteady flow profiles throughout the entire Reach. Our general approach consisted of using MASS1 to simulate Reach-wide water surface elevations and the respective shoreline (flow band) locations for a range of steady state streamflows for the entrapment analysis. We also used MASS1 to simulate hourly unsteady water surface elevations throughout the Hanford Reach for the spring 2003 hydrograph to re-create the effect of streamflow fluctuations on the repeated inundation and subsequent dewatering of river bank depressions known as entrapments. Water surface elevations from MASS1 as well as entrapment locations were integrated into our GIS to determine the flow band of each entrapment from spatial queries. This allowed us to build a record or event history for the entire season for the number of times the water surface increased through each depression, and then subsequently decreased, resulting in the formation of the pooled entrapment. Each occurrence represented a single and unique entrapment event. Shoreline locations from River2D could have been used for the entrapment analysis, but since MASS1 was used to simulate the actual, unsteady flow conditions in 2003, we chose to conduct the two modeling tasks with the same model – MASS1.

River2D, a two-dimensional, finite element hydrodynamic model (Ghanem et al. 1996) was used to produce detailed coverages of depth, velocity, and water surface elevation for a spatially explicit GIS analysis of spawning and rearing habitats. Table 1 provides an overview of what models and metrics were used for the different analyses. Several types of data were required to build the hydrodynamic model. A digital elevation model (DEM) was first required for building a digital river channel and routing streamflows through the Reach. Substrate coverages were required to determine roughness values for use with the model. Hydraulic boundary conditions were required as inputs/targets for model solution at each flow simulated and were derived from the MASS1 model. Additionally, empirical depth, velocity, and water surface elevation data were required for validation of hydrodynamic model output. River2D had previously been constructed and validated for 33 km of the Hanford Reach between rkm 572 and 605 by Tiffan et al. (2002) and simulations were made at 10 kcfs intervals between 40 and 400 kcfs. The spatial extent of our hydrodynamic modeling tasks consisted of building and validating similar River2D models for the upper (rkm 605 to 639) and lower (rkm 546 to 573) segments which would provide full coverage for the entire Reach. The simulations from the previous effort in the middle segment were acquired and incorporated into our GIS for subsequent analyses.

Table 1. Summary of models, simulations and analysis. Q = Discharge and WSE = Water Surface Elevation.

Model/Tool	Metrics Derived	Total Simulations	Analysis
River2D – Steady State	Depth, Velocity, WSE	76	Spawning and Rearing Habitat
MASS1 – Steady State	Q and WSE	145	Entrapment - Shorelines
MASS1 – Unsteady State	Q and WSE	145	Entrapment - Event History
GIS/DEM	Elevation Model & Slope	2	ALL, except MASS1 unsteady

Field Methods and Results

The field work conducted for our hydrodynamic modeling tasks consisted of several efforts to develop the hydrodynamic models. The MASS1 model had already been configured, calibrated, and validated for the Hanford Reach and did not require any additional field data (McMichael et al. 2003).

Digital Elevation Model Data Sources

Our hydraulic and habitat models required a continuous and seamless, high-resolution digital elevation model (DEM) of the Columbia River channel up to the 400 kcfs shoreline boundary. Prior to our efforts, existing bathymetric data for the upper and lower segments was limited to dated, coarse, transect-based descriptions of the channel. We contracted two separate and complimentary bathymetric surveys for the upper and lower study segments and supplemented with other available data as required. It should be noted that the vast majority of the data were acquired from a Compact Hydrographic Airborne Rapid Total Survey (CHARTS) conducted by the U.S. Army Corps of Engineers (USACE). The final DEM, included data from six additional data sets varying in resolution and spatial extent. All data were maintained and re-projected into:

Projection: *Lambert Conformal Conic*
Coordinate system: *Washington State-plane*
Zone: *4602 (south zone)*
Horizontal Datum: *North American Datum of 1983 (NAD83)*
Horizontal Units: *Meters*
Vertical Datum: *North American Vertical Datum of 1988 (NAVD88)*
Vertical Units: *Meters*

Data sets that were used to construct the DEM are described below.

CHARTS Survey

We contracted a CHARTS survey (Heslin and Lillycrop 2003) for the Hanford Reach in 2003 which was conducted by the USACE. A Scanning Hydrographic Operational Airborne Lidar Survey (SHOALS) was originally scheduled for this mapping effort. The SHOALS survey was postponed from the original planned date in fall of 2002 for a number of reasons including weather, high river discharges, the USACE schedule, and equipment problems. During 2002 and 2003, the next generation of lidar technology was developed, made available, and used for this project. CHARTS is a unique system that consists of an airborne laser transmitter/receiver capable of measuring 1,000 soundings per second and a topographic laser transmitter/receiver capable of measuring 9,000 soundings per second. The system operates from a King-Air 200 aircraft flying at altitudes between 200 and 700 meters with ground speeds between 125 and 175 knots. CHARTS provides a fully operational compact airborne lidar (Light Detection And Ranging) hydrographic and topographic mapping and charting system using infrared 532 nm laser technology. The CHARTS system also includes a ground-based data processing system for calculating accurate horizontal position, water depth, and elevation. However, the maximum depth of detection is three times the Secchi depth while the minimum is 0.0 – 0.1 m. The survey area was comprised of upper and lower study segments, each adjoining a 1998 SHOALS survey contracted by the USGS, Biological Resources Division, Columbia River Research Laboratory (Tiffan et al. 2002). The upper segment of our survey was between rkm 605 and 639, while the lower segment was between rkm 548 and 573 (Figure 11). The data sets described below were used to fill in areas where CHARTS data were not available.

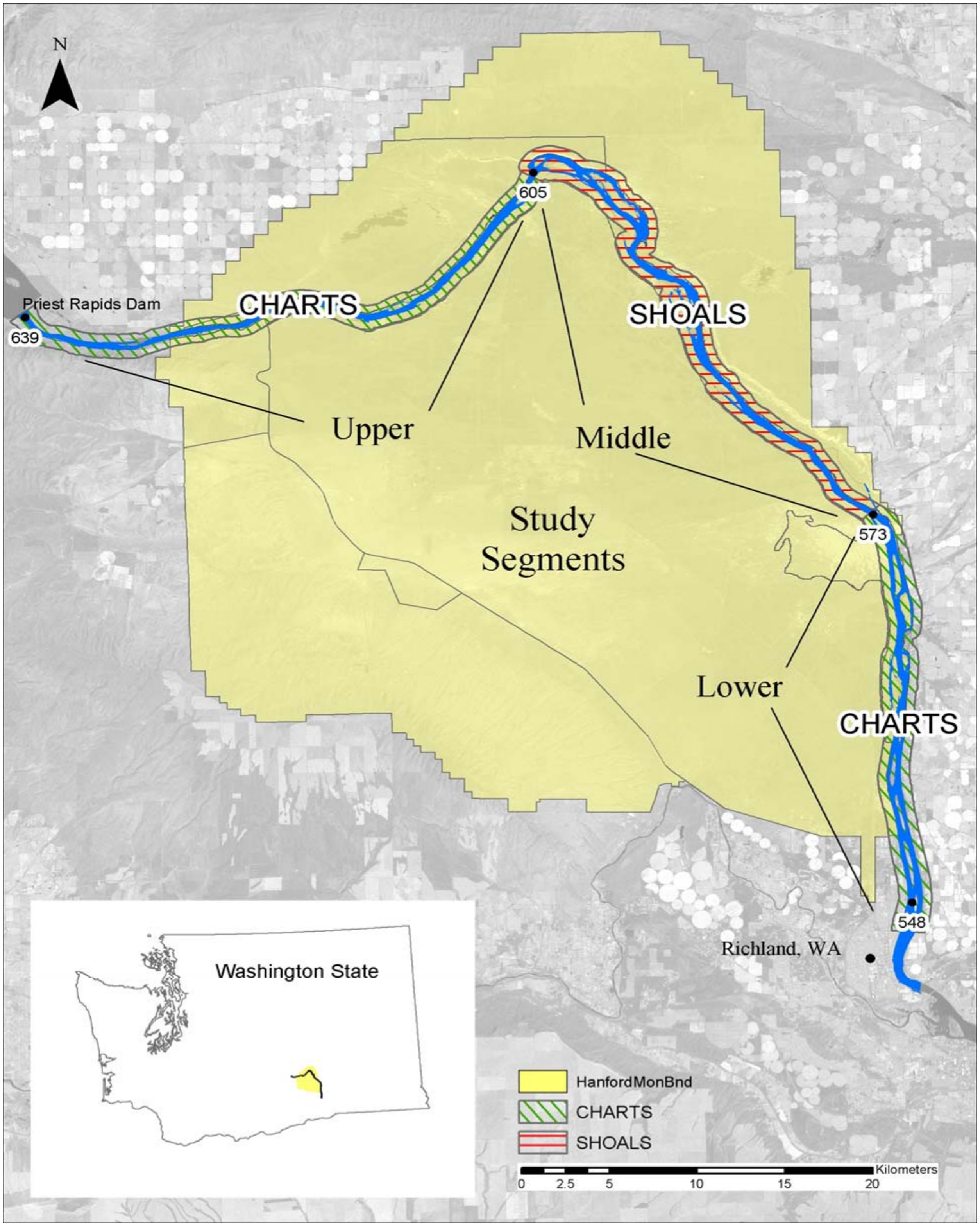


Figure 11. Location of the Upper, Middle and Lower study segments with CHARTS and SHOALS area of geographic coverage.

SHOALS Survey

SHOALS Lidar data were collected by the USACE in July 1998 (Figure 11) for 33 km of the Hanford Reach between rkm 572 and 605 (Tiffan et al. 2002). The SHOALS system is the predecessor to CHARTS with the same functional capabilities of collecting data both on land and below the water surface (Guenther et al. 1996; Lillycrop et al. 1996; Parson et al. 1996). This data set was used to supplement overlapping portions of the new study sites lacking sufficient data; typically, the portions of the CHARTS survey that were less than 0.1 m in depth.

Deep-Water Bathymetric Survey

We contracted a hydrographic survey in December 2003 to supplement the CHARTS data for the subset of locations where depth exceeded the range of the CHARTS survey and precluded data acquisition. Bathymetric data were collected using an Innerspace-448 survey grade echo sounder with a 3-degree single beam transducer at 3db, operating at 208 kHz and a ping rate of 15 to 20 Hz, with a manufacturer's stated vertical accuracy of 3.05 cm. Horizontal positioning was derived from an Ashtech BRG2 differential Global Positioning System (GPS) receiver. The manufacturer's stated horizontal accuracy is 0.9 meters at a 95% confidence level. Water surface elevations were obtained from Real Time Kinematic (RTK) GPS surveys and entered into the echo sounder data collection software. Elevations were shot adjacent to each bathymetric survey and at various times to account for fluctuations in river elevation.

USGS 10 m DEMs

USGS DEMs were acquired and used to a limited extent for areas where the CHARTS survey did not provide data; primarily, up to the 400 kcfs shorelines. These requirements were few and usually above the 350 kcfs shoreline.

USACE Surveyed Cross Sections

USACE, Seattle District, surveyed cross-sections from Priest Rapids Dam at rkm 639, to the mouth of the Yakima River at rkm 542 as part of a reconnaissance effort for what would have been Ben Franklin Dam. The cross-sections were spaced at approximately 0.4 km intervals. These data were used infrequently to fill in data gaps left by the 0.0 to 0.1 m-depth limitation of the CHARTS System.

White Sturgeon Project Cross Sections

River cross sections were established in 1994-1996 in various locations in the Hanford Reach for hydraulic data collection under the BPA-funded White Sturgeon Project (Anglin 1996, Anglin et al. 1997, Anglin et al. 1998). Cross section profiles consisted of horizontal distance and relative elevation for a variable number of points from the high water mark on one side of the river across to the high water mark on the opposite side. Subsets of these cross sections were geo-referenced to provide bottom profile data points for the DEM (Figure 12). RTK-GPS equipment was used to establish horizontal and vertical locations for cross section headpins accurate to within $\pm$ 5 cm. These data allowed previously surveyed bottom profile locations and elevations to be transformed into the State Plane Washington South coordinate system referenced to the NAVD88 vertical datum and the NAD83 horizontal datum.

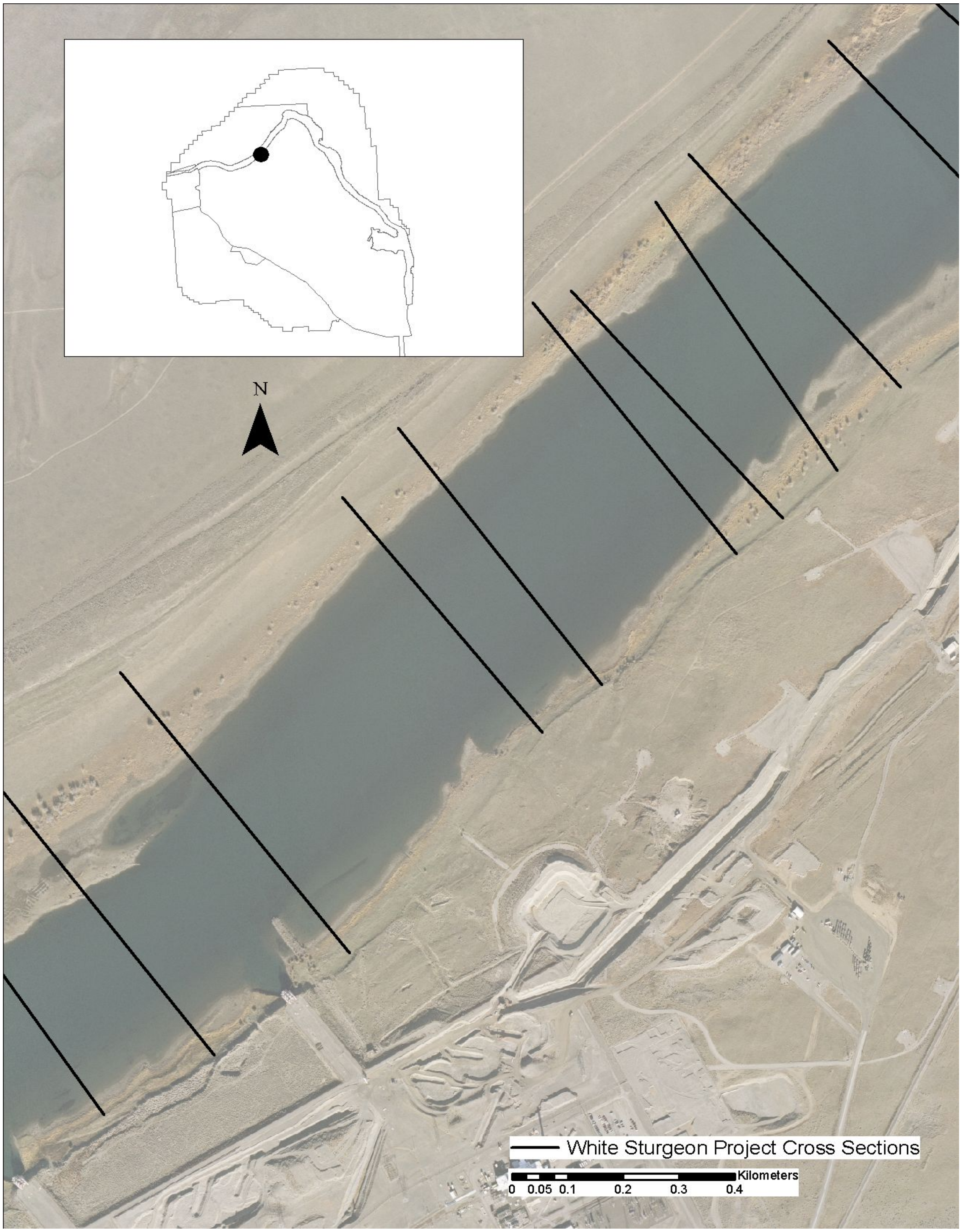


Figure 12. Example of hydraulic data collection cross sections for the white sturgeon project. Only a small subset of the data was required for DEM generation.

Digital Elevation Model Generation

We generated a triangulated irregular network (TIN) with the 2003 CHARTS data (Figure 13) for the upper and lower study segments using the GIS software package, ArcGIS®, from Environmental Systems Research Institute, Inc. (ESRI). This TIN was used to identify erroneous points and areas requiring data supplementation. The raw data set used to construct the TIN consisted of approximately 4.8 and 1.5 million data points with the same density (15.4 m^2/data point) for the upper and lower segments, respectively. This level of resolution was adequate for compiling a DEM suitable for hydrodynamic modeling at this scale. The point density could also be expressed as 127 bed measurements for a linear cross section 500 m wide. Additional data were required to complete the DEM for the small subset of locations lacking suitable channel description. Since the CHARTS survey did not collect data between 0.0 – 0.1 m in depth, flat shallow areas, particularly around islands and sloughs that were present at the time of the survey flight required additional data. For the areas requiring supplementation we used data from the original SHOALS survey, the USACE cross section surveys, the White Sturgeon Project, and shoreline files generated from orthophotography. Secondly, the CHARTS survey did not collect data if water depth exceeded 8 m. We used data from the deepwater hydrographic survey to supplement these areas. Lastly, the CHARTS survey did not collect data in a subset of areas equating to the highest discharge we would model (400 kcfs). We used USGS 10 m DEM data for supplementing these locations.

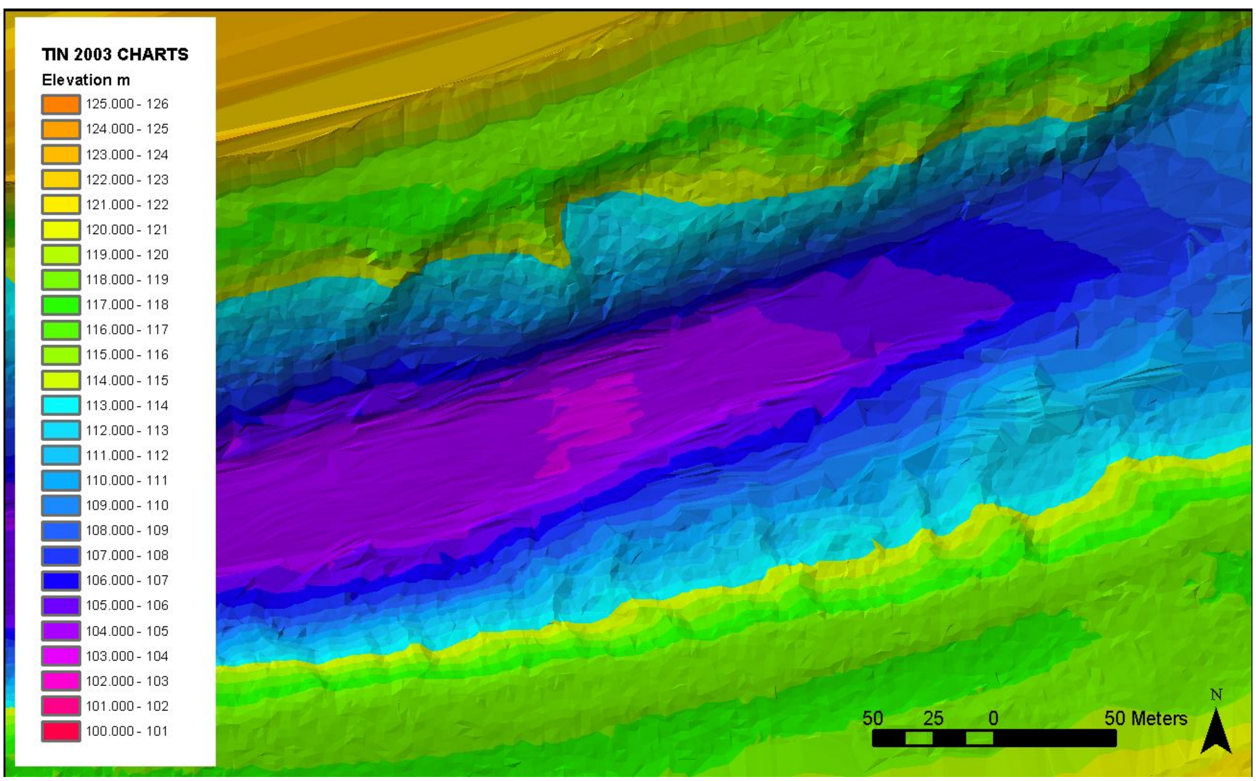


Figure 13. A representative Triangulated Irregular Network (TIN) generated from the original 2003 CHARTS survey data for a section of the upper segment of the Hanford Reach 1.5 km above Vernita Bridge.

The SHOALS survey constituted the majority of points used to supplement the CHARTS data. Prior to integration, we compared elevations for 48,384 SHOALS points overlapping with respective CHARTS data in the GIS. We computed differences between the two data sets and calculated a mean delta of 0.14 m. This is comparable to the stated accuracy of both data sets of 0.15 m. For the sub-set of areas requiring supplementation, we clipped the data to match the no-data section and/or inserted breaklines and populated with topo points. Breaklines were digitized in ArcMap using point cross-sectional data, orthophotography, and in some areas, personal knowledge to guide placement. We used an editing tool in ArcMap to fill each breakline with a defined number of points, typically at 5 m intervals. Points were then assigned known elevations from one of the supplemental data sets and the process was repeated as necessary. We created and plotted TINs at regular intervals in the ArcGIS 3D viewer, ArcScene. With this tool, reconstructed shorelines could be viewed in 3D and from a number of angles for a visual check of continuity and relative validity. The breakline process was also used with the supplementary USGS 10 m DEM data in some areas missing CHARTS coverage up to the 400 kcfs shoreline. When the editing process was complete, we exported all of the vector points that were used to generate the final TIN as a text file suitable for import into River 2D.

The final three-dimensional surfaces (DEMs) created in our GIS for the upper and lower segments were composed of 2,650,326 and 1,064,857 points, respectively. The relative DEM density was similar for the two segments but the lateral extent (width) was greater for the upper segment accounting for the difference in total points. Horizontal data from the CHARTS and SHOALS surveys were accurate to within ± 3 m, and elevations to within ± 15 cm. The deep water bathymetric survey resulted in 26,528 data points with horizontal locations accurate to within ± 0.9 m, and elevations to within ± 3 cm. Empirical cross section data from the white sturgeon project and the USACE cross sections were not incorporated into the DEMs, but rather used to guide the placement and set elevations for the supplementary breaklines that were generated. Additionally, no 10 m USGS DEM points were required for our DEM in the upper segment. The total distribution of points used is presented in (Table 2). Additionally, Figure 14 graphically depicts the data sources used and Figure 15 demonstrates the GIS process we used to build our DEMs for a small sub-section of the river. In Figure 15, the bottom array represents the point elevation data collected in the field and the middle layer represents a triangulated irregular network (TIN) modeled from the array. The top layer is a representation of the data prior to importing into River2D.

Table 2. Distribution of points used in the final DEMs.

	Upper Segment		Lower Segment	
Data Set	n	%	n	%
CHARTS 2003	**2,591,612**	**97.78%**	**1,033,148**	**97.02%**
SHOALS 1998	45,055	1.70%	8,235	0.77%
Deep Water Survey	8,788	0.33%	17,740	1.67%
USGS DEM	0	0.00%	1,321	0.12%
Breakline	4,871	0.18%	4,413	0.41%
	2,650,326	100.00%	1,064,857	100.00%

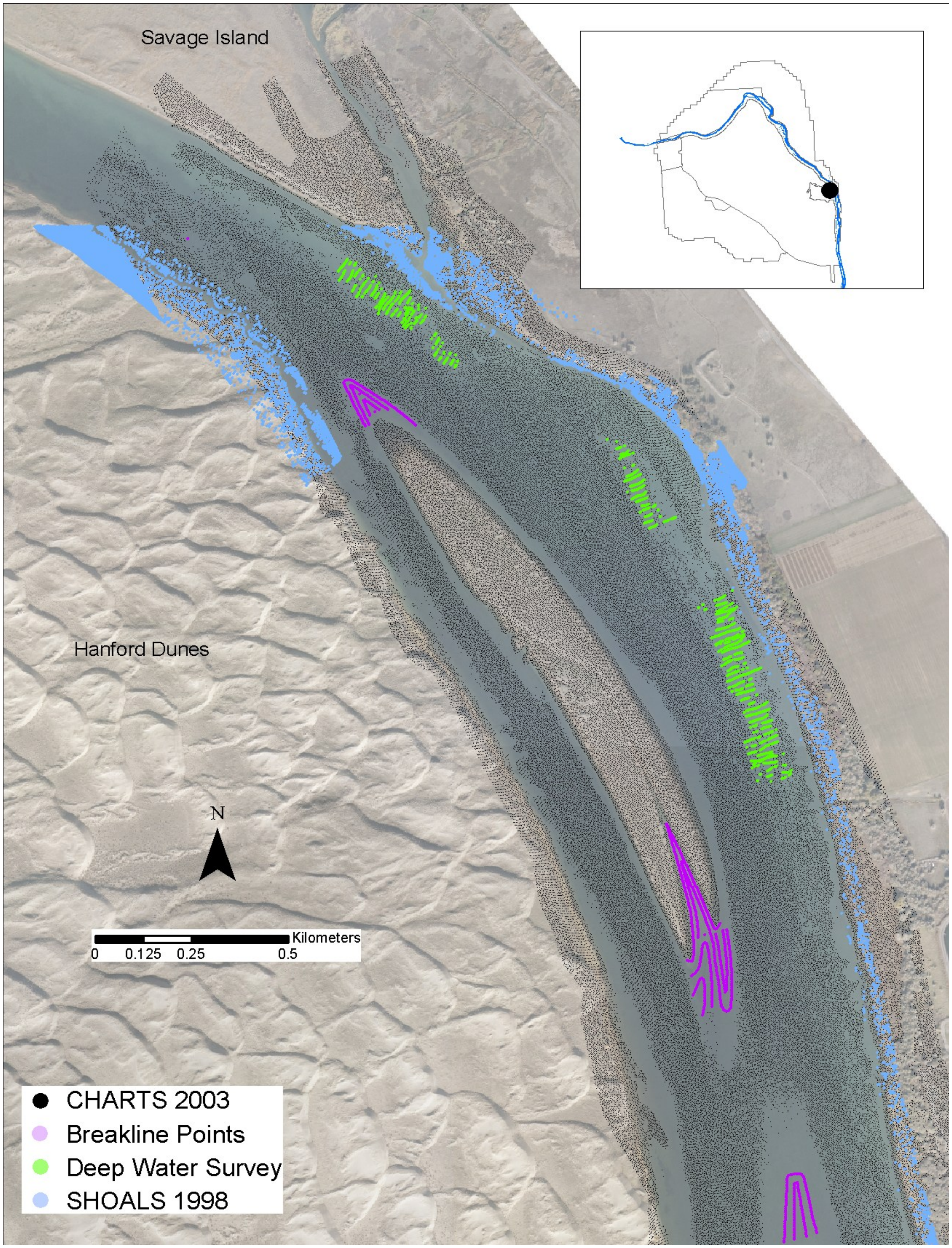


Figure 14. Schematic example of the different data sources used to generate our final DEM.

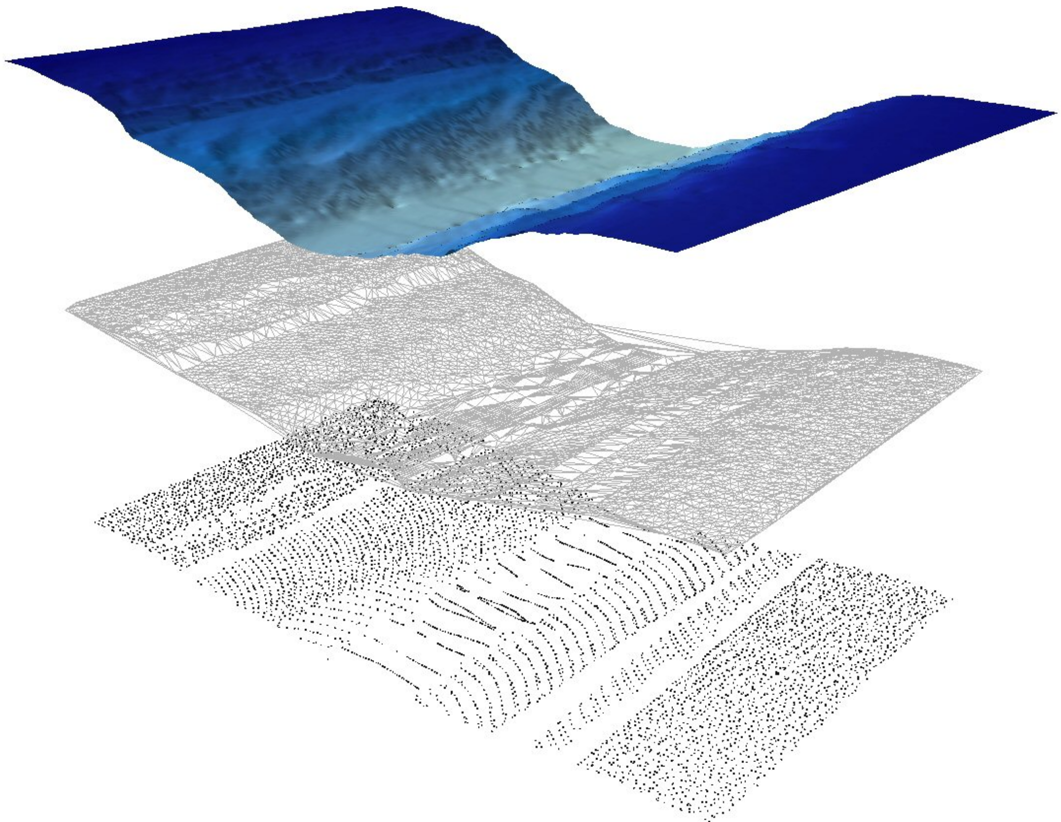


Figure 15. GIS representation of the process used to generate a DEM for hydrodynamic modeling. This sub-section is located between Vernita Bridge and Vernita Bar and is representative of bathymetry for the upper segment. Note that the Z axis in the top plot has been exaggerated 3X.

Substrate Coverages-River2D

Both our hydraulic and habitat models required substrate information for the upper and lower segments. Prior to our efforts, only limited substrate coverages existed (Ward 2000). We systematically collected substrate data for each of the two study sites modeled. Surveys were conducted using a point survey methodology along transects parallel to the river. We used a visual classification method modified from Bovee (1982) to collect dominant, subdominant and percent fine substrate data. A limited, transect-based substrate data set does exist for the middle segment. Insufficient project resources precluded us from collecting additional substrates in this area.

The number of longitudinal survey transects and point density was predetermined in our GIS based on several factors. We used our available resources, the number of available surveyors, channel width and complexity to determine transect and point density. Resources included the amount of time and funding available. The target distance between cross sections and survey points was set at 60 m based on a GIS assessment and the available resources. Wide, flat areas such as those opposite Coyote Rapids (Figure 16) required up to eight cross sections per shoreline, and narrow and steep shorelines (opposite shoreline, Figure 16) required as few as one.

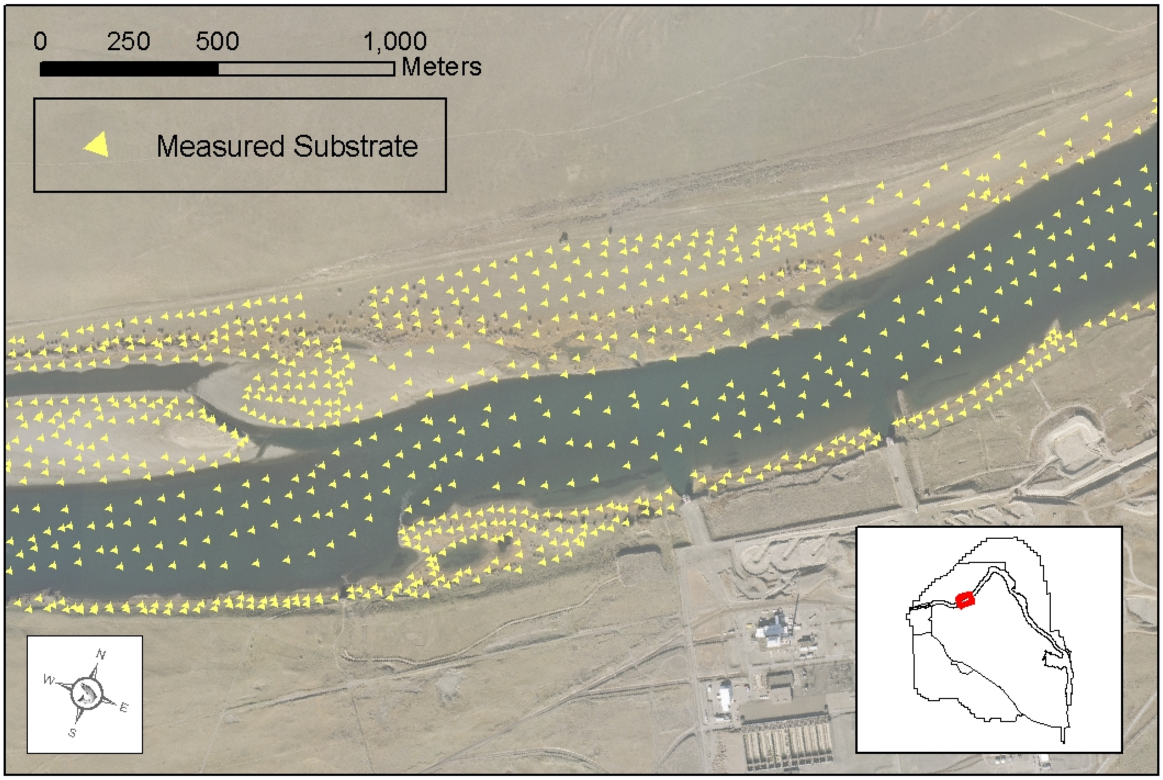


Figure 16. Substrate data points collected with underwater videography and shoreline surveys.

In the field, actual point location and intensity was increased if substrate complexity increased. Substrate complexity is the amount of variation in the three substrate measures. At low complexity, a given area had uniform substrates and percentage fines over a large area which required few survey points. Areas with high complexity required more intense surveying to reflect variations in substrates and percent fines. Areas with discrete breaks in the dominant substrates had survey points equally spaced on each side of the break so that interpolation with Euclidean allocation was correct. Survey intensity was somewhat subjective, however some criteria were set. High intensity surveying, with the exception of discrete class breaks, had survey points no closer than 10 meters apart and the furthest distance between survey points did not exceed 60 meters. Surveys were conducted up to the highest observable high water mark or 400 kcfs (referenced on field maps) and to a wading depth of 0.5 m.

For depths greater than 0.5 m, we used a single boat-mounted videography system. The sampling method required: an underwater video sled modified from Groves (1998) with reference lasers added for size scaling, a bow mounted 24-V hoist system, and onboard video monitors and DGPS (Differentially-corrected GPS). The reference lasers were mounted in parallel and set at a distance matching the transition size between small and large cobble substrates. The in-river substrate classification and sampling density was identical to the bank protocol, but the sample area was limited by: depths greater than 18 m, boat draft and on a few occasions, by high velocity. Surveys were conducted from upstream to downstream working with the current following depth contours. The boat operator used a GPS to identify, in real-time, 60 m sample points along each transect. All transects were parallel to the river channel. At each sample point, the boat operator maintained position over the substrate while it was being classified. The sled operator used the 24-volt electric hoist to position the sled 0.5 – 1.0 m above the riverbed and then classified the substrate. The sled operator recorded the substrates in real-time as codes on the DGPS unit.

We collected dominant substrate, subdominant substrate, and percent fines for all data points. Two coding schemes were used to capture substrate qualities important to both rearing and spawning fall Chinook (Table 3). Dominant and subdominant substrates were classified using a modified Brusven Index coding system (Delong and Brusven 1991). The two class sets used were based on separate studies in the Hanford Reach, each with a unique set of substrate classes and codes. For a juvenile fall Chinook rearing study, Tiffan et al. (2002) used codes that more finely discriminated between the small particles and more coarsely categorized larger particles up to 4 m. Geist (2000) used codes that uniformly described substrates at equal intervals up to 152 mm for an analysis describing Chinook spawning habitat.

At each individual point, a one m^2 area (Figure 17) was visually assessed and assigned a dominant, subdominant and percent fines code (Orth 1983). Dominant substrate is classified as the substrate covering up to at least 50% of the one m^2 area in the survey point. Subdominant substrate is the next most abundant substrate size. Percent fines were visually assessed by the degree that larger particles (rearing codes 5 to 7, spawning codes B to F) were surrounded or covered by fine sediment (Platts et al. 1983; Table 4).

Figure 17. Example of individual bank and in-river substrate data collection points. Ruler in photo at left is 30.5 cm (1 foot) in length, and red dots in photo at right are scaling lasers, 150 mm apart.

Table 3. Substrate codes, particle sizes, and descriptions used to classify dominant and subdominant substrates for rearing and spawning fall Chinook.

Rearing Code	Particle size (mm)	Particle size (inch)	Description
1			Live, Dense Organic Material
2	<0.004	<0.00016	Clay/Soft
3	0.004 - 0.062	0.00016 - 0.0024	Silt
4	0.062 - 2.0	0.0024 - 0.079	Sand
5	2 - 64	0.079 - 2.52	Gravel
6	64 - 250	2.52 - 9.84	Cobble
7	250 - 4000	9.84 - 13 feet	Boulder
8	>4000	>13 feet	Bedrock
9	0.004		Clay/Hard

Spawning Code	Particle size (mm)	Particle size (inch)
A	<6.3	<.25
B	6.30 - 25.4	0.25 - 1.0
C	25.4 - 50.8	1.0 - 2.0
D	50.9 - 76.2	2.0 - 3.0
E	76.3 - 152.0	3.0 - 6.0
F	>152.0	>6.0

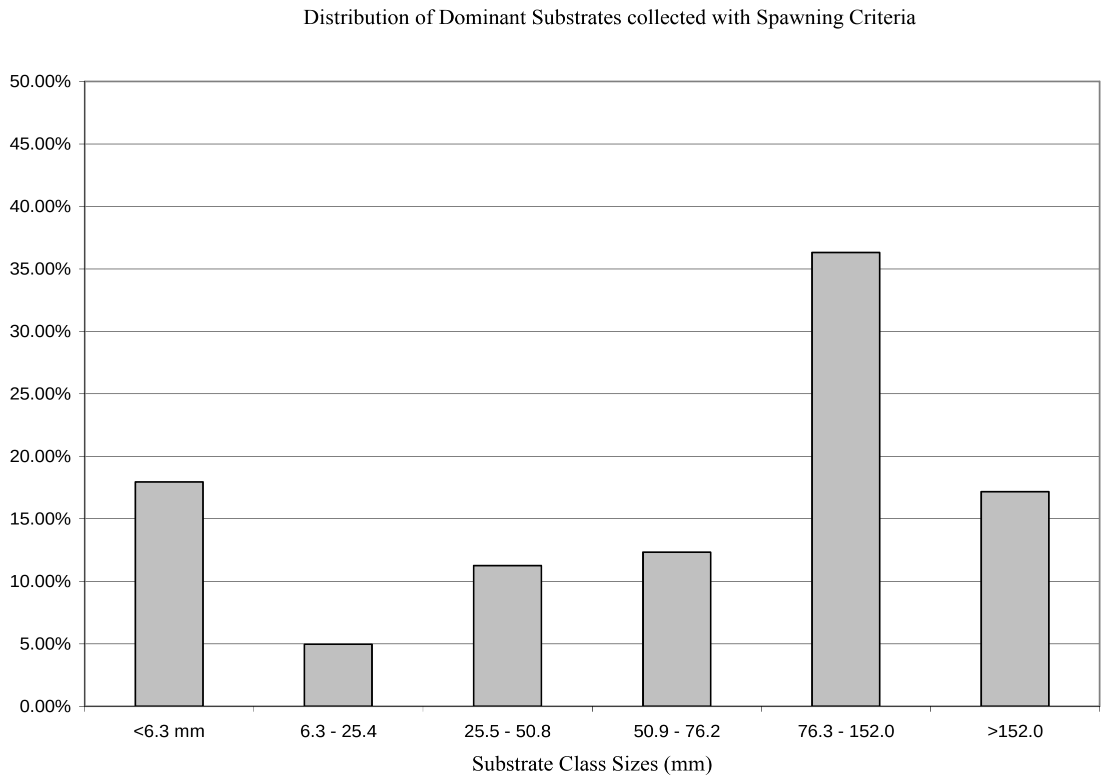
Table 4. Percent fines codes and classification descriptions.

Code	Description
1	0 to 25 percent of substrate belongs to Rearing codes 3 or 4.
2	25 to 50 percent of substrate belongs to Rearing codes 3 or 4.
3	50 to 75 percent of substrate belongs to Rearing codes 3 or 4.
4	75 to 100 percent of substrate belongs to Rearing codes 3 or 4.

We used point substrate data collected in the field to model a continuous raster substrate surface for the two study segments. Point vector data were converted in the GIS to grids (rasters) with a 4 m cell size. Cells between the field-measured values were assigned a NODATA value. To interpolate and assign the NODATA cells a value, we used a Euclidean allocation process.

In addition to using substrate characteristics for our habitat assessments, we used the general substrate characteristics for the Hanford Reach to determine a uniform roughness height for calibration of River2D.

We collected a total of 11,790 substrate data points for the upper and lower segments of the Hanford Reach. Substrate data collected along exposed shorelines up to the 400 kcfs flow level accounted for 7,869 data points and the remaining 3,921 were collected in-river with underwater videography (66.7 and 33.3%, respectively). This effort covered a total distance of 59 rkm. The dominant substrates and total combined distribution in the upper and lower segments are presented in (Figure 18). For all substrates within the 90 kcfs wetted perimeter, we considered sizes to the extent that they would limit spawning based on suitability. Fines, bedrock and boulders accounted for 2.24, 2.13 and 4.39% of the river channel respectively, and summed to 8.76%. Substrates over the remaining 91.24% of the channel fell into size categories that were suitable for spawning. We did not analyze the longitudinal spatial variation of surficial substrates within the channel any further. Also, no sub-surface measurements or assessments of substrates were undertaken.



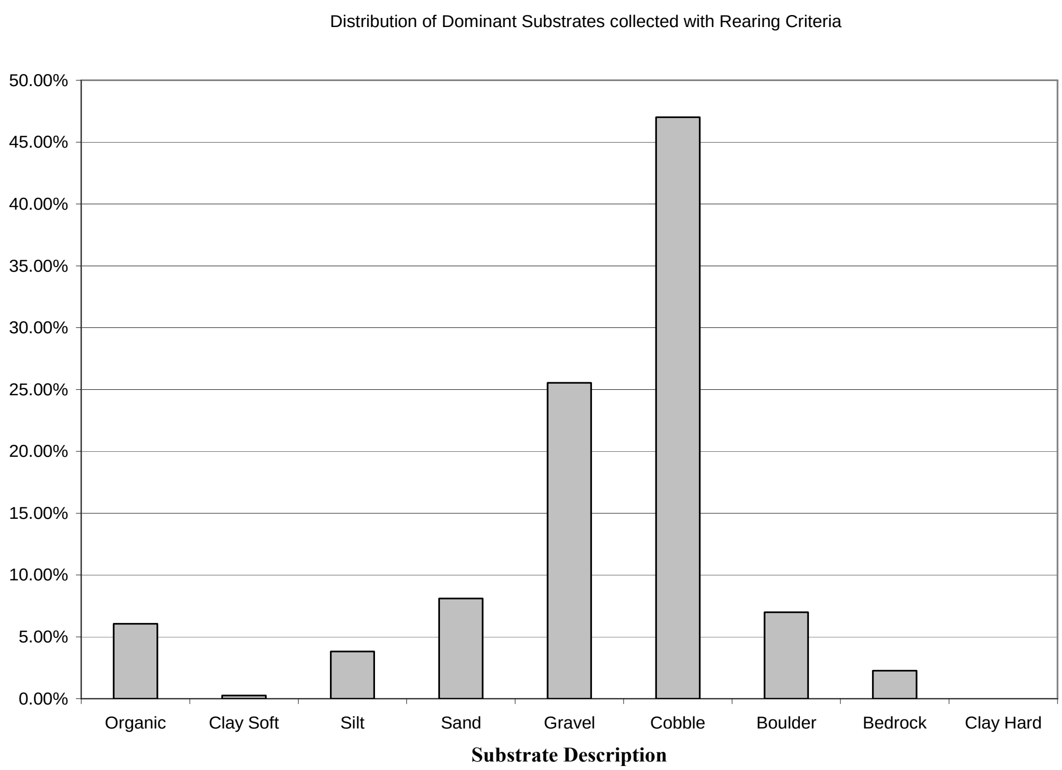


Figure 18. Relative frequency of dominant substrates collected for each coding scheme in the upper and lower segments.

Euclidean allocation

We modeled three new raster data layers with the Euclidean allocation process in our GIS using the 11,790 vector-based substrate data points. Each raster is a continuous cell based model of substrate for each of the three criteria used. The Euclidean algorithm records the identity of the closest source cell for each NODATA cell. A distance is then calculated from the center of the source cells to the center of each of the surrounding cells without values. The algorithm proceeds as follows. For each cell, the distance is calculated to each source cell by calculating the hypotenuse with the x-max and y-max as the other two legs of the triangle. This calculation derives the true Euclidean, not cell, distance. The shortest distance to a source is determined and the value is assigned to the cell location on the output grid. That is to say, the NODATA cells were assigned substrate characteristics of the nearest cell with empirical data. At the end of the process, a continuous and complete substrate surface is produced. This process was completed for each of the substrate coding systems (rearing and spawning) and for the percent fines layer, resulting in three separate substrate grids. An example of the output is presented in Figure 19 for the rearing codes. The yellow triangles represent the location of the original substrate data used in the allocation process.

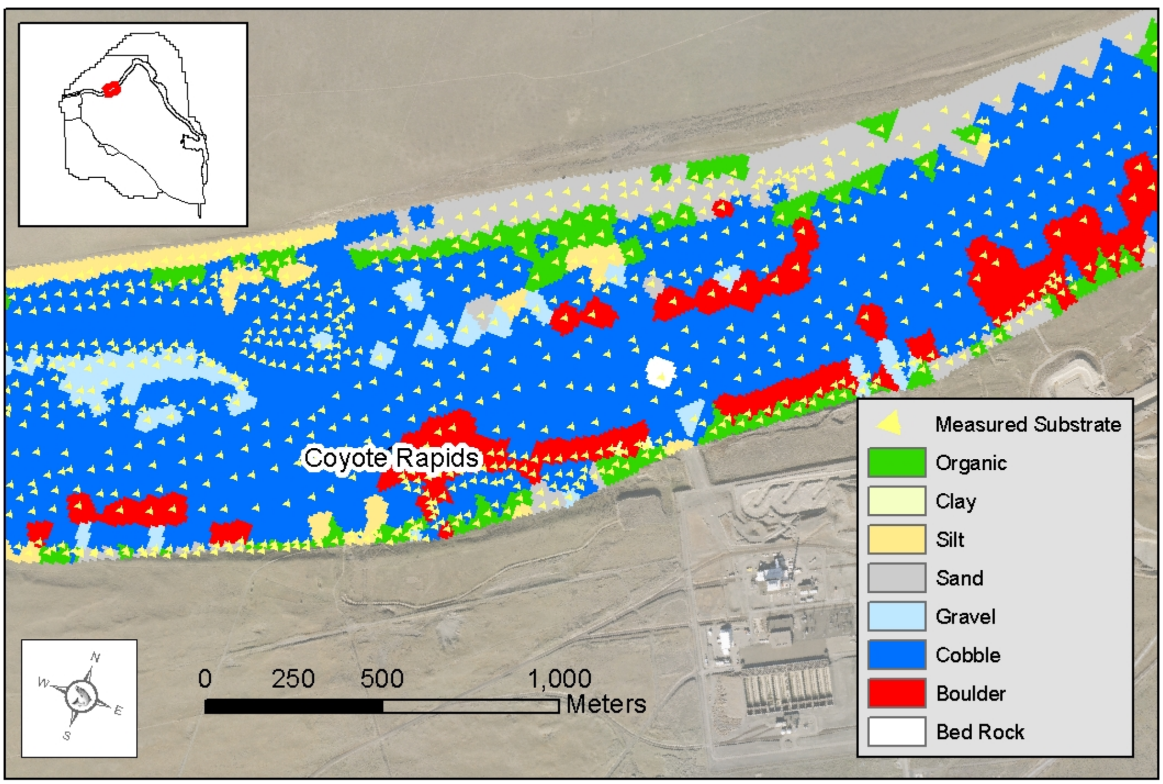


Figure 19. Results of the Euclidean allocation process (interpolation) for the Coyote Rapids area located at rkm 616.

River2D Modeling Methods, Results, and Discussion

Overview

We estimated steady state, depth-averaged velocities, water surface elevations, and depths for the upper and lower segments of our study site using the River2D hydrodynamic model (Ghanem et al. 1996). Each of the segments was modeled independently. The River2D model is a two-dimensional, depth-averaged, finite element hydrodynamic model developed for use in streams and rivers. It has been verified through a number of comparisons with theoretical, experimental, and field results (Ghanem et al. 1995; Waddle et al. 1996). It is basically a transient model but provides for an accelerated convergence to steady-state conditions (Steffler and Blackburn 2002). The River2D model is based on the two-dimensional, depth-averaged St. Venant equations (Steffler and Blackburn 2002). These equations represent the conservation of water mass and of the two components of the momentum vector. Model inputs included inflow discharges and outflow water surface elevations for the respective model boundaries, riverbed bathymetry (DEM), and representative riverbed roughness values (substrate height). In the middle segment, River2D was successfully used by Tiffan et al. (2002) to model a range of discharges similar to those modeled for this study, and as such, no additional modeling was conducted in that area. River2D output from that study however, was incorporated into our GIS for habitat assessments. The streamflows we modeled ranged from 30 to 400 kcfs and were simulated at 10 kcfs intervals. This range of streamflows includes all conditions that are likely to occur in the Hanford Reach under current hydrosystem configuration.

We chose to model at 10 kcfs flow intervals based on several factors. First, we knew that River2D could model water surface elevations to within 0.1 m of empirical data (Garland et al. 2004, and Hanrahan et al. 2004). Second, we knew from MASS1 that the average change in water surface elevation in the Hanford Reach was about 0.2 m between each 10 kcfs flow band. Thus, if each simulated WSE from River2D had a maximum error of 0.1 m (0.2 m between two successive runs), we could successfully discriminate between the two simulations. Third, 10 kcfs flow bands had been modeled previously in the middle segment. By modeling the upper and lower segments at the same intervals, output from all model runs could be combined to develop Reach-wide coverage for depths, velocities, and water surface elevations at 10 kcfs streamflow intervals. Lastly, available resources precluded additional modeling at finer streamflow intervals.

Mesh Construction

We used the River2D Bed module to import our final raster data set which had been developed in a GIS and exported as a text file. We used the River2D mesh module to create a finite element mesh. Most of the editing and point refinement was completed in a GIS and little work was actually required from these programs besides general formatting. The River2D Bed module was used to import the raw data into the modeling program and define the computational and flow boundaries for the range of discharges to be modeled. We used the mesh program to triangulate the original point data (nodes) and set the inflow and outflow boundary conditions. For the initial output mesh, we produced a uniform 20-m resolution mesh for each 10 kcfs flow interval. Actual meshes used for modeling were dual meshes composed of 10- and 20-m node densities with 10 m spacing comprising the dominant node spacing for each simulation. With this method, node densities for near-shore areas and all depths up to 6 m were doubled to 10 m (Figure 20). Fall Chinook redds are not visible on aerial photos at depths greater than six meters which limits the need to produce model output at the higher resolution density. By using the higher density mesh in all depths less than 6 m, we could more precisely simulate depths and velocities in areas of increased complexity which were in most of the primary spawning and rearing habitats. Given the large size of our study site, a 10 m mesh was the best compromise considering the computation limitations associated with the desired resolution of 4 m. A 10 m mesh size also approximates the scale at which fall Chinook construct redds. The entrapment analysis uses the original DEM to derive shorelines, which would approximate a mesh size of about 4 m.

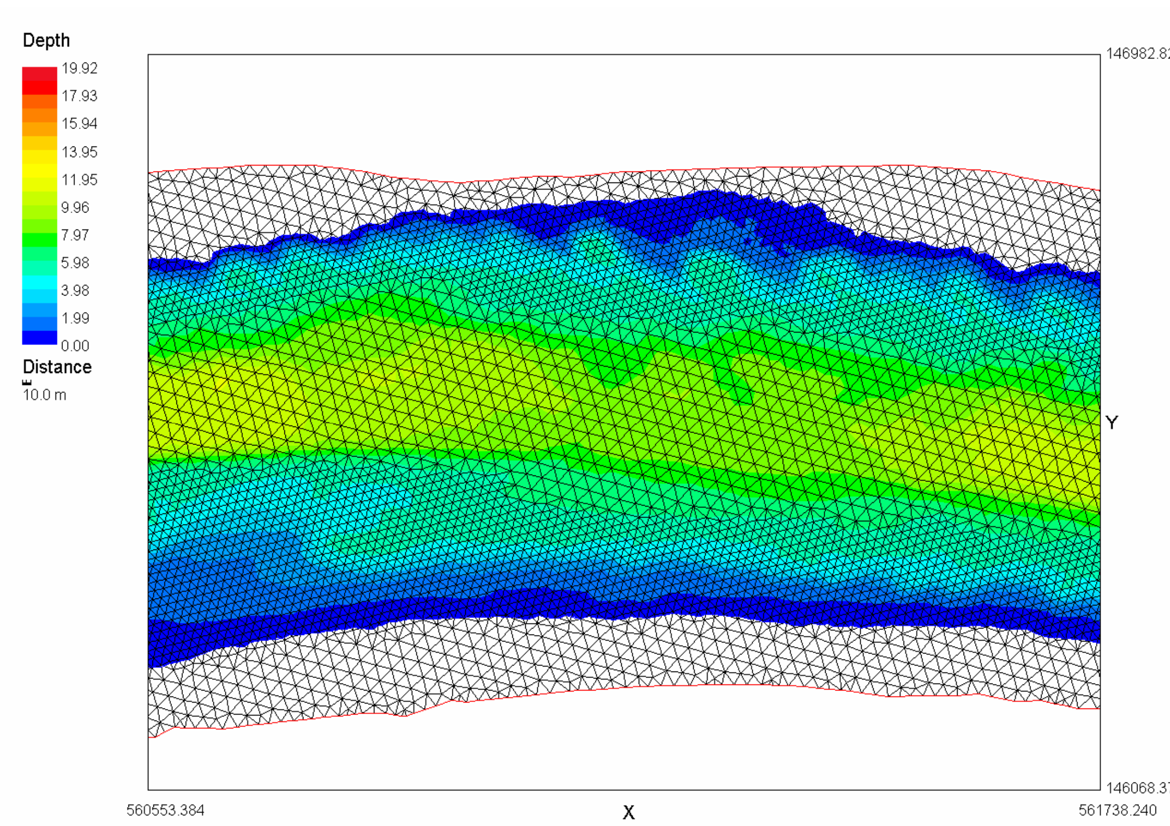


Figure 20. An example of a dual computational mesh, with a typical 10- and 20-m node density. The area represented in this figure is located in the middle of the upper segment.

Model boundary locations for both the upper and lower study segments were carefully identified and positioned (Figure 21). Boundary locations were selected with a uniform longitudinal aspect, uncomplicated channel morphology and distant from islands and areas of biological interest. The River2D model requires a water surface elevation for the downstream boundary and a discharge at the upstream boundary. The conditions are linked in that a steady state simulation requires the upstream discharge to be the condition resulting in the downstream water surface elevation at a steady-state streamflow. Locations with rating tables (e.g. USGS gages) are typically selected for these boundaries. We used the MASS1 streamflow model described previously for water surface elevations at model boundaries, which provided more flexibility in boundary location selection. In addition, MASS1 was successfully used by Tiffan et al. (2002) in the middle segment of the Hanford Reach.

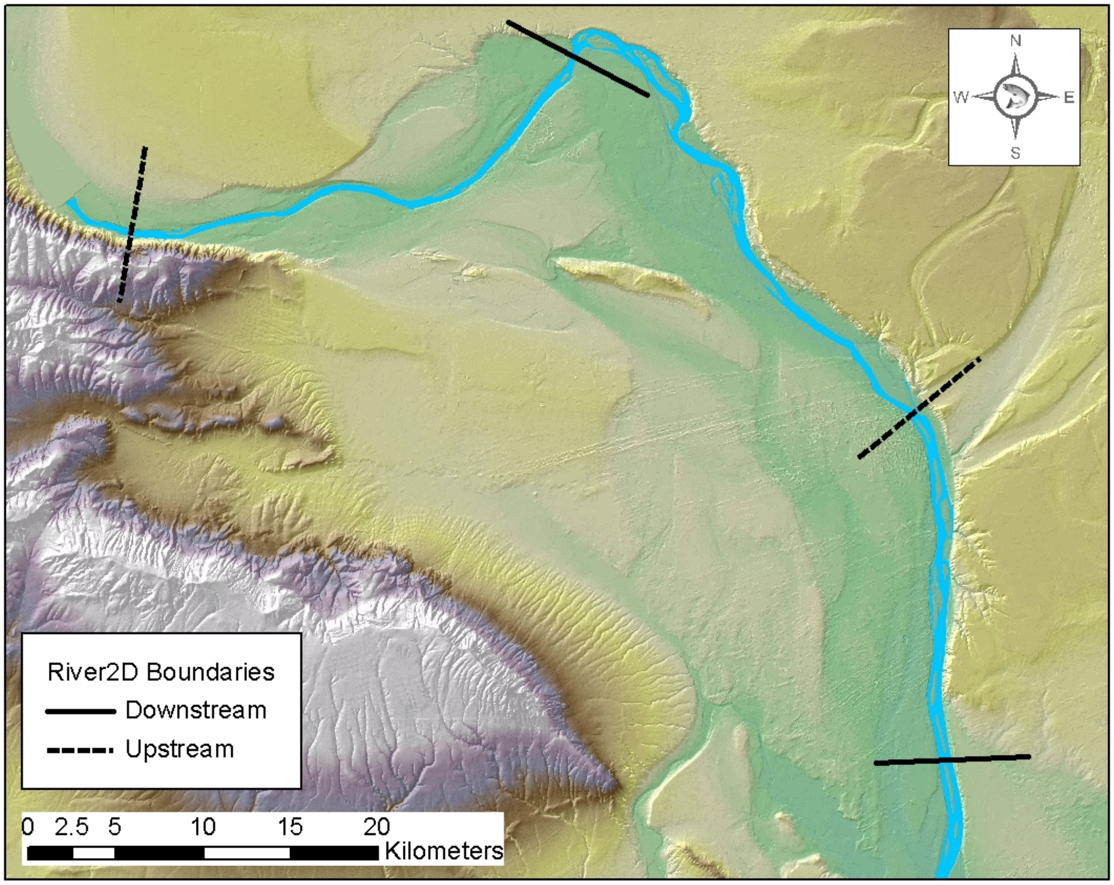


Figure 21. Specific locations of River2D model boundaries for the upper and lower segments.

River2D Streamflow Simulation Details

We divided the total range of streamflow simulations into 100 kcfs blocks. Each parent block, (400, 300, 200, 100 kcfs) was run to solution on a 20-m uniform mesh with the respective boundary conditions set. Final solution was defined as having a very small final solution change (x 10^{-5}), near equal inflow and outflow discharges (+/- 1%), and having run at the maximum time step. This initial run was then verified for further simulation and the node spacing was decreased to 10 m for near-shore areas and all locations down to 6 m in depth (Figure 20). We used the River2D "Auto Refine" option for refining the mesh. We attempted to conduct simulations with the entire mesh at a 10 m resolution but encountered "Run Time" and memory errors. River2D has a 2 GB memory limit when run on a 32 bit operating system, and the memory required to run our simulations with an overall 10-m mesh resolution exceeded this limit. Thus, our decision to proceed with the 10 m mesh in complex areas with biological relevance, and a 20 m mesh in all other areas. Once the new (combination 10/20 m mesh) parent simulation was run to final solution and archived, boundary conditions were modified for each subsequent 10 kcfs discharge interval under the parent run. These subsequent discharges were run to solution and the process repeated to simulate flow conditions for each of the 10 kcfs intervals under the parent discharge.

River2D Calibration

An initial step for calibration of the River2D model is to adjust the bed roughness heights in the bed file until the predicted water surface elevations approximate empirical measurements for a given discharge. Model calibration is typically done along a longitudinal transect of the river and requires near steady-state conditions so water surface elevation measurements can be matched to a measured discharge upstream. This allows one to compare predicted water surface elevations at a given discharge, as produced by River2D, to empirical measurements collected in the field. Increasing or decreasing the roughness heights results in the estimated water surface elevations increasing or decreasing. When close agreement between simulated and empirical water surface elevations is achieved, the first step in validation of River2D output is conducted with a comparison of predicted depths with empirical depths. Where significant disagreement occurs, the likely cause is errors in the bathymetry, specifically elevations. Ideally, any errors can be corrected before proceeding to an evaluation of velocity simulations and a comparison with empirical data. The second calibration step is then conducted, which consists of a comparison of simulated velocities to observed. Roughness heights are adjusted if needed for velocity calibration with the goal of achieving adequate agreement of both simulated water surface elevations and velocities with empirical data. Typically this process is conducted for a low, intermediate, and high discharge with physical measurements (transects) spatially representative across the study site.

The large size of the Hanford Reach (90 km long, 0.3 to 1.3 km wide) would have made the complete model calibration process, as described above, a huge task. In addition, the constantly changing streamflows made it very difficult to collect water surface elevations, spatially, for a common streamflow. As a result, we did not have the time or resources to conduct a comprehensive, robust calibration exercise. Instead, we selected a uniform roughness height of 0.1 m based on substrate characteristics as described previously, to approximate the relatively uniform cobble/gravel substrate of the Hanford Reach. This same roughness height was successfully used in the previous assessment for the middle segment. Additionally, we converted the Manning's n values that were used by McMichael et al. (2003) for calibration of the MASS1 model in the Hanford Reach to roughness heights. Those conversions resulted in an average roughness height of 0.118 m, and a range from 0.096 to 0.147 m. Small differences in bed roughness, such as between sand, gravel, or cobble, were insignificant at the scale of the Columbia River. Boulder fields, which are few, were typically impacted with subdominant gravels and cobbles decreasing potential frictional forces. We rationalized that our selection of a uniform roughness height was adequate if the measured velocities and depths agreed reasonably well with the validation transects we placed throughout the Hanford Reach. Although not ideal, our approach was dictated by available resources, and was consistent with a previous modeling exercise that was conducted and validated in the middle segment of the Hanford Reach (Tiffan et al. 2002). Following the designation of 0.1 m as the uniform roughness height to be used, we proceeded with the validation process.

River2D Validation

Our validation goal was to compare simulated River2D water depths and velocity magnitudes to empirically measured water depths and velocity magnitudes for a range of streamflows. The most significant output from River2D from the standpoint of fall Chinook habitat is velocity. Our objective for velocity accuracy was for simulated values to be within 20% or better of empirical values. To evaluate our accuracy, we created an acoustic Doppler current profiler (ADCP) validation dataset that was comprised of 16 transects (cross sections) collected between 1996 and 2005 (Figure 22). Streamflows for the different transects ranged from 134 to 193 kcfs (Table 5). Prior to this study, ADCP data had been collected at 5 transects in the upper section in 1996 (Anglin 1996, Anglin et al. 1997, Anglin et al. 1998) and 1 transect in the middle section in 1999 (Tiffan et al. 2002). To fill in the gaps, we collected an additional 4 transects in 2004 (3 in the lower section and 1 in the middle section), and another six transects in the middle section in 2005. Differences in equipment, software, and study objectives over the 10-year period resulted in different techniques being employed in the collection and processing of the ADCP data. Dependant on study objectives and resources at hand, ADCP data were processed in one of three ways: (1) ADCP data were averaged from multiple transect passes, (2) 1 ADCP pass was selected from a set of transect passes based upon the best match between ADCP and River2D, or (3) only a single ADCP pass was available.

Each ADCP collection effort utilized a RTK GPS receiver or a total station to accurately measure water surface elevations, and an ADCP to measure depths, water column velocities, and discharges at individual cross sections. An ADCP measures water velocity based on the Doppler effect (the apparent change in the frequency of a wave resulting from relative motion of the source and the receiver) to characterize the motion, direction, and depth of water from the returning echoes of four acoustic beams (RD Instruments 1989). Water velocities were measured along cross sections at approximately 1-m intervals. At each horizontal sampling point, the ADCP measured velocity magnitude and direction in bins that were either 0.25 m or 0.5 m in size, from approximately 0.9 m below the water surface to approximately 0.5 m above the river bottom. The locations of the depth and velocity bins were georeferenced with the GPS receiver.

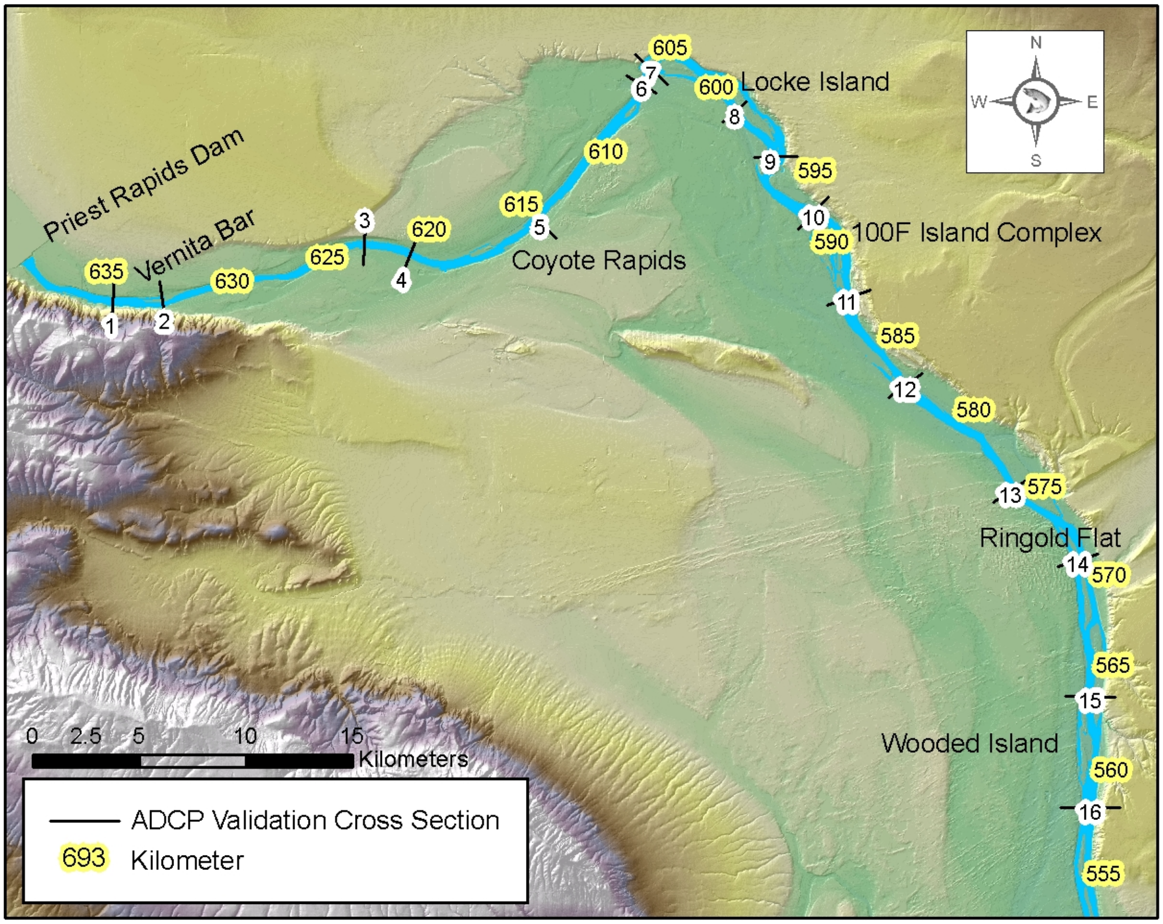


Figure 22. Locations of cross-sections used for River2D validation.

The XY locations of the depth-averaged ADCP bins were imported into a GIS and attributed with their predicted velocity magnitudes, water surface elevations, and depths. To calculate model error (i.e., differences between ADCP and River2D estimates) we calculated the root mean squared error (RMSE [Willmott 1981]) for each transect. To compare the magnitude of errors between the 16 transects, we standardized the data by dividing RMSE by the mean depth or velocity across each transect. Thus, the standardized error represents the percent difference between the estimated and empirical datasets for each transect after adjusting for the mean depth or velocity at that transect. We also created XY graphs to visually assess whether the errors in water velocity and depth estimates were consistent laterally across the validation transects (i.e., shallow edge areas versus deeper water habitats).

River2D Validation Results

Estimated water velocity errors (RMSE) at the 16 transects ranged between 0.08 and 0.57 m/sec ($\bar{X}$ = 0.21 m/sec, *SD* = 0.12), while the standardized velocity errors ranged from 8% to 61% ($\bar{X}$ = 20%, *SD* = 15%) (Table 5). This level of accuracy was consistent with our original objective. Transect #2, located at rkm 632, was an outlier compared to the other transects with an error rate over 2 times greater than most transects. At transects where multiple passes were averaged, simulated velocities were usually within one standard deviation of ADCP mean velocities (Figure 23). Estimated water depth errors (RMSE) at the 16 transects ranged between 0.06 m and 1.7 m (X = 0.73 m, SD = 0.48). The standardized depth errors ranged from 4% to 49% (X = 18%, SD = 12%). Similar to the velocity analysis, transect #2 was an outlier compared to the other transects. At transects where multiple passes were averaged, simulated depths were usually within one standard deviation of ADCP mean depths (Figure 24). The error rates were different among the three Hanford Reach sections; section 1 had the highest standardized errors for velocity and depth (33% and 27%, respectively), section 2 had 15% and 18% error, respectively, and section 3 had the lowest errors (9% and 3% respectively).

Table 5. Summary of comparative metrics of water velocity and depth collected empirically with an ADCP and simulated with River2D at 16 cross sections (see Figure 22).

[a]ID	Cross section location (RM)	Cross section location (RKM)	Section	[b]Q (kcfs)	[c]Velocity *RMSE* (m/s)	[d]Standardized velocity deviation (%)	[c]Depth *RMSE* (m)	[d]Standardized depth deviation (%)	[e]Year
1	394	634	Upper	193	0.24	26	0.84	18	1996
2	393	632	Upper	160	0.57	61	1.70	49	1996
3	389	626	Upper	158	0.17	18	0.67	19	1996
4	385	619	Upper	141	0.16	20	0.50	12	1996
5	382	615	Upper	141	0.25	40	1.20	35	1996
6	376	605	Middle	162	0.15	10	1.0	16	2005
7	376	605	Middle	162	0.20	12	0.88	22	2005
8	372	599	Middle	134	0.26	14	0.64	19	2005
9	371	597	Middle	165	0.20	13	0.67	21	2005
10	368	592	Middle	162	0.23	11	0.48	16	2005
11	365	587	Middle	157	0.21	14	0.72	19	2005
12	362	582	Middle	145	0.09	8	0.38	8	1999[f]
13	358	576	Middle	147	0.33	38	1.64	22	2004
14	355	571	Lower	155	0.08	7	0.06	2	2004
15	351	565	Lower	158	0.10	8	0.19	4	2004
16	347	558	Lower	157	0.16	12	0.18	3	2004
X					0.21	20	0.73	18	
SD					0.12	15	0.48	12	

[a]Transect ID is displayed on Figure 22
[b]Discharge at time of validation
[c]*RMSE* = root mean squared error
[d]*RMSE* / *X* velocity or depth for the transect
[e]Year ADCP data was collected
[f]from Tiffan et al. 2002

The plots in Figure 23 and Figure 24 help to illustrate the effects of water surface elevation, velocity, and depth on validation comparisons. In Figure 24, plots for cross sections at RM 347, 351, and 355 show good agreement between empirical and River2D depths. The slight differences that can be observed are consistent across the cross section and likely a result of small water surface elevation errors. The velocity plots for these cross sections in Figure 23 also show reasonable agreement, however additional evidence of the water surface elevation error is apparent. The predicted water surface elevation for these cross sections is generally higher than the empirical, and the result is predicted velocities that are generally, slightly lower. The plot for the cross section at RM 362 in Figure 24 shows a more substantial difference between empirical and River2D depths, and the differences are not consistent across the cross section. These are likely a result of bed elevation errors. In fact, this cross section is in an area where SHOALS data were collected in 1998, and bathymetry data are sparse. The corresponding velocity plot in Figure 23 shows the effect of this bed elevation error. Agreement between the empirical and River2D velocities is reasonable, although the locations where predicted velocities are higher or lower, correspond to predicted depths that were lower or higher, respectively, at those locations.

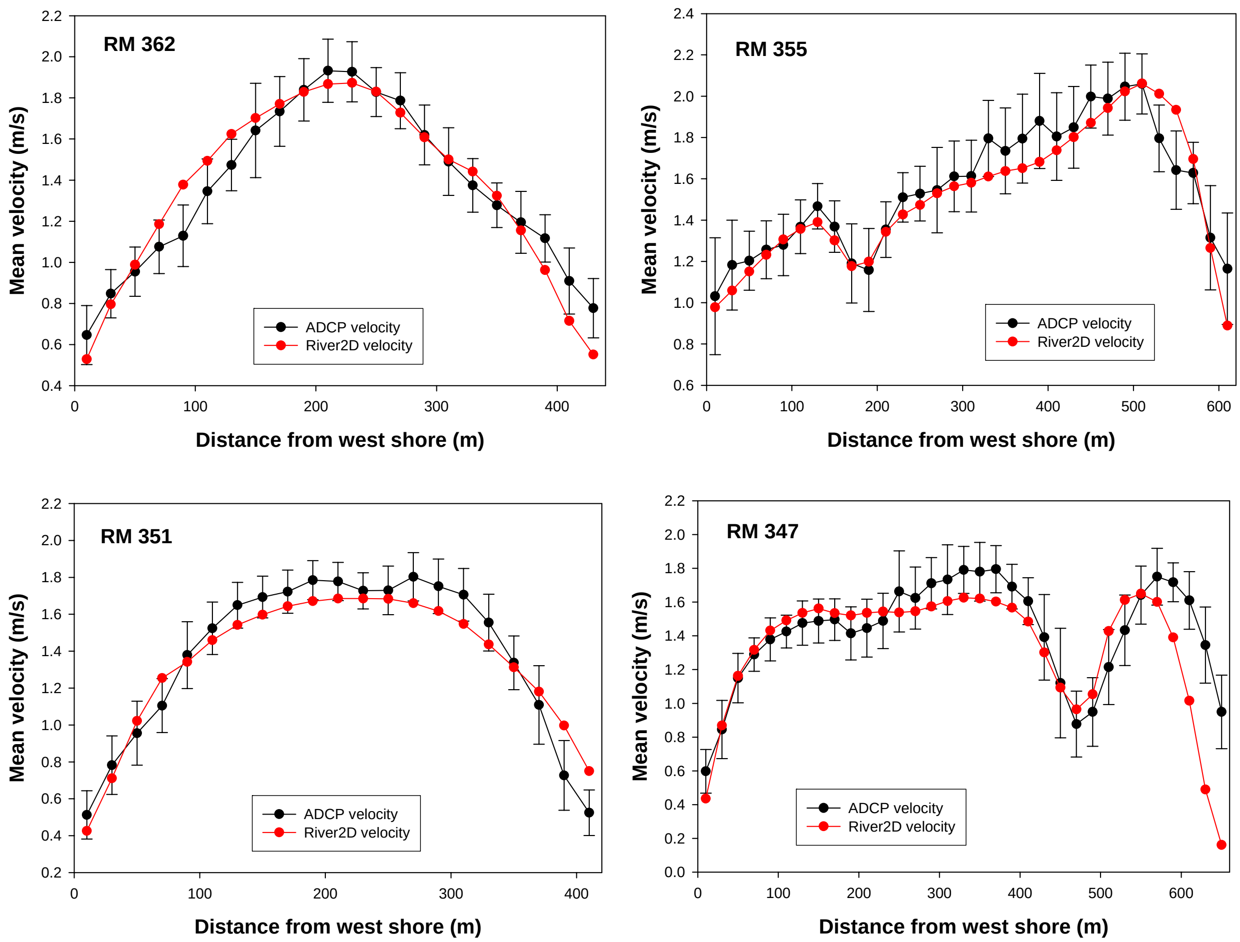


Figure 23. Comparisons of water velocities measured with an ADCP and simulated with River2D at four cross sections in the lower and middle segments of the Hanford Reach. Error bars represent ±1 SD about mean ADCP-measured velocities.

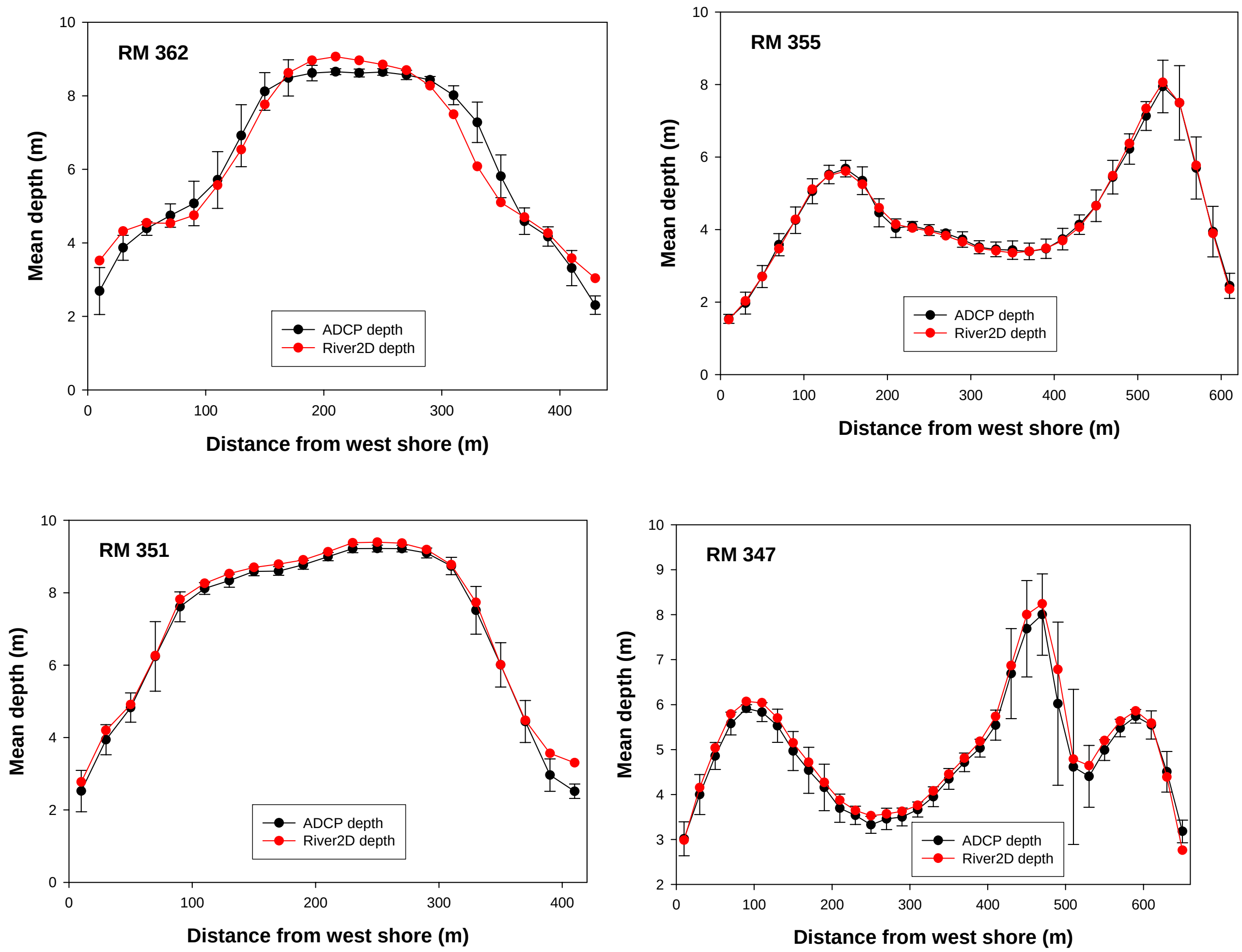


Figure 24. Comparisons of water depths measured with an ADCP and simulated with River2D at four cross sections in the lower and middle segments of the Hanford Reach. Error bars represent ±1 SD about mean ADCP-measured depths.

River2D Modeling Results

Model simulations were considered converged when the difference between the inflow and outflow rate was less than 1% of the input discharge (Waddle et al. 2000). All final model simulations converged within this guideline for both the upper and lower segments. From these results, we compiled estimates of depth, velocity, and water surface elevation for a total of 76 streamflow simulations, 38 for the upper and lower segments respectively. Typical simulations constituted about 115,000 nodes and 225,000 elements. We wrote a script to convert each final flow simulation from its native River2D format, into database (dbf) files suitable for import into our GIS in GRID (ESRI 1992) format. The script produced tables for all metrics produced at each node within the River2d computational mesh. However, we only imported node values of depth, velocity and water surface elevation. We combined depth, velocity, and water surface elevation GRIDs from this effort with the existing GRIDs from the middle segment modeled by Tiffan et al. (2002) to produce a Reach-wide set of GRIDs for habitat modeling.

River 2D Discussion

The objective of our hydrodynamic modeling was to spatially and temporally describe the seasonal and hourly physical conditions for a range of streamflows throughout the Hanford Reach as they relate to habitat for fall Chinook salmon at various life stages. A few years ago, modeling at this level of cell resolution and scale would not have been possible in the Columbia River. In recent studies, River2D has proven useful for estimating the physical metrics required for predictions of fall Chinook habitats in the mainstem Columbia River (Geist et al. 2003, Hanrahan et al. 2004, Tiffan et al. 2002, and Garland et al. 2003). In addition, two-dimensional hydrodynamic models (e.g. River2D) are generally considered to result in more realistic estimates of hydraulic conditions than one-dimensional hydrodynamic models (e.g. PHABSIM type; cross section average) (Hardy 1998; Lane 1998). Arguably the most important input data for River2D is the bathymetry, and we think that the comprehensive bathymetry collected with the CHARTS system provided the most detailed and accurate bathymetry for the Hanford Reach to date.

There are several factors that may have influenced the performance of the River2D model. Calibration of the model usually consists of iterative adjustments of bottom roughness heights to achieve agreement between simulated and empirical WSEs and velocities. This involves making localized adjustments where channel geomorphology may cause simulated WSE to deviate from reality, particularly at different streamflows. Given the size of our study area (90 km) and the number of discharges we modeled, it was not possible with our current limitations to make iterative adjustments to bottom roughness. As a result, we applied a uniform bottom roughness height (0.1 m) to approximate that of the cobble/gravel substrate type that was prevalent in the Reach. This value was similar to the average roughness height value determined from conversions of Manning's n values used for MASS1 modeling in the Reach. We believe this approach was reasonable. A qualitative comparison of five measured WSEs with River2D simulated WSEs showed an error rate ranging from 2% to 6%. More work is planned to verify and enhance the calibration of our Hanford Reach model. Another factor that may have influenced River2D output was error associated with MASS1 estimates of water surface elevation used at the downstream River2D boundaries. The average mean absolute error for MASS1 water surface elevation estimates was 0.11 m. Since River2D matched the MASS1 elevation at the downstream boundary, the error at that location was entirely from MASS1. However, simulated conditions upstream would include both the effect of River2D modeling error, and the error associated with the target water surface elevation at the downstream boundary (additive). At some distance upstream, the effect of a 0.11 m error at the boundary would become negligible. The only way to characterize and quantify these errors is with a comprehensive and rigorous validation exercise. Such an exercise is planned for the next iteration of our habitat modeling.

Overall, the estimated water velocities and depths output by River2D looked reasonable for our study objectives. And for the most important parameter, velocity, we met our objective of within 20% of empirical values, although with additional calibration and validation, we think we could achieve better results. The standardized error rates observed in section 1 were higher than sections 2 or 3, but this appears to be largely because of transect #2 (Figure 22) which had large errors associated with depth (49%) and velocity (61%) estimates. Since the error rates for the other 4 transects in section 1 were similar to error rates observed elsewhere, we think the larger error rate associated with this transect was artificial, and a result of errors associated with the single-pass ADCP data. However, there were some patterns in the error that warrant further discussion. In our analysis we observed that nearshore areas oftentimes produced more error between ADCP and River2D estimates. This may have been a result of the difficulty associated with accurately measuring depths and velocities in shallow water areas with an ADCP. Acoustic backscattering and noise can occur when the ADCP transducers are too close to the river bottom for the frequency and mode being used to collect data. Secondly, nearshore areas oftentimes have rapid transition zones between deep and shallow water areas (e.g. along a thalweg), that make both ADCP measurements and River2D simulations more problematic. A third possibility is the effect of roughness along the channel margins. If the actual roughness is higher along the banks (e.g. boulders), and we used a uniform roughness value, this would cause River2D to over-predict velocities compared to empirical data. If the actual roughness is lower than our uniform roughness value (e.g. sand, fines), this would cause River2D to under-predict velocities compared to empirical data. More thorough calibration and validation will be required to improve the match between empirical and simulated conditions in near-shore areas.

For our spawning habitat model, we needed sufficient cell resolution along with relatively accurate water depth and velocity estimates to predict the occurrence of known spawning sites at the scale of redd clusters, not individual redds. Redd clusters are groups of individual redds typically numbering in the hundreds, and in 2004 mean redd cluster size was 6.38 ha. Since our computational mesh resolution was 10 to 20 m, we had excellent resolution to discern redd clusters. The River2D model produced reasonable estimates of water velocities and depths in the Hanford Reach as evidenced by the Reach-wide standardized velocity and depth errors (20% and 18%, respectively) and small variation (15% and 12%, respectively). This level of accuracy was sufficient to characterize depths and velocities in the Hanford Reach under different streamflows at a scale consistent with the scale of redd clusters. We plan to continue work on calibration and validation of our River2D model for the Reach along with our ongoing spawning habitat modeling efforts.

MASS1 Modeling Methods, Results, and Discussion

Overview

The MASS1 model (Modular Aquatic Simulation System 1D) developed at the DOE Pacific Northwest National Laboratory is a one-dimensional, steady and unsteady hydrodynamic and water quality model for river systems (Richmond and Perkins 1998). This model has previously been applied to the mid-Columbia River for water temperature simulations (Perkins et al. 2002). Steady and unsteady streamflow parameters are simulated by MASS1 by solving the one dimensional equations for conservation of mass and momentum. These equations are also referred to as the St. Venant equations. A brief description of the MASS1 model is provided in

Appendix B. The MASS1 model is only able to calculate cross-sectional average estimates of water surface elevation and water quality conditions in a river system. Thus, only single values of hydraulic parameters are computed at each point or cross section in the model. The primary parameters of interest for our work were water surface elevation and discharge. Other model outputs were not used.

As part of several studies conducted for the relicensing of the Priest Rapids Project, the MASS1 model was configured to simulate the Columbia River between Priest Rapids Dam (rkm 639) and McNary Dam (rkm 471), and the Snake River from Ice Harbor Dam (rkm 15) to the mouth. The structure of the model for the Hanford Reach is based on a cross section survey conducted by the USACE (Seattle District). Figure 25 depicts the locations of the 145 cross sections surveyed throughout the Reach. The frequency and location of these surveyed cross sections provide a reasonably detailed template of river bathymetry for the Reach.

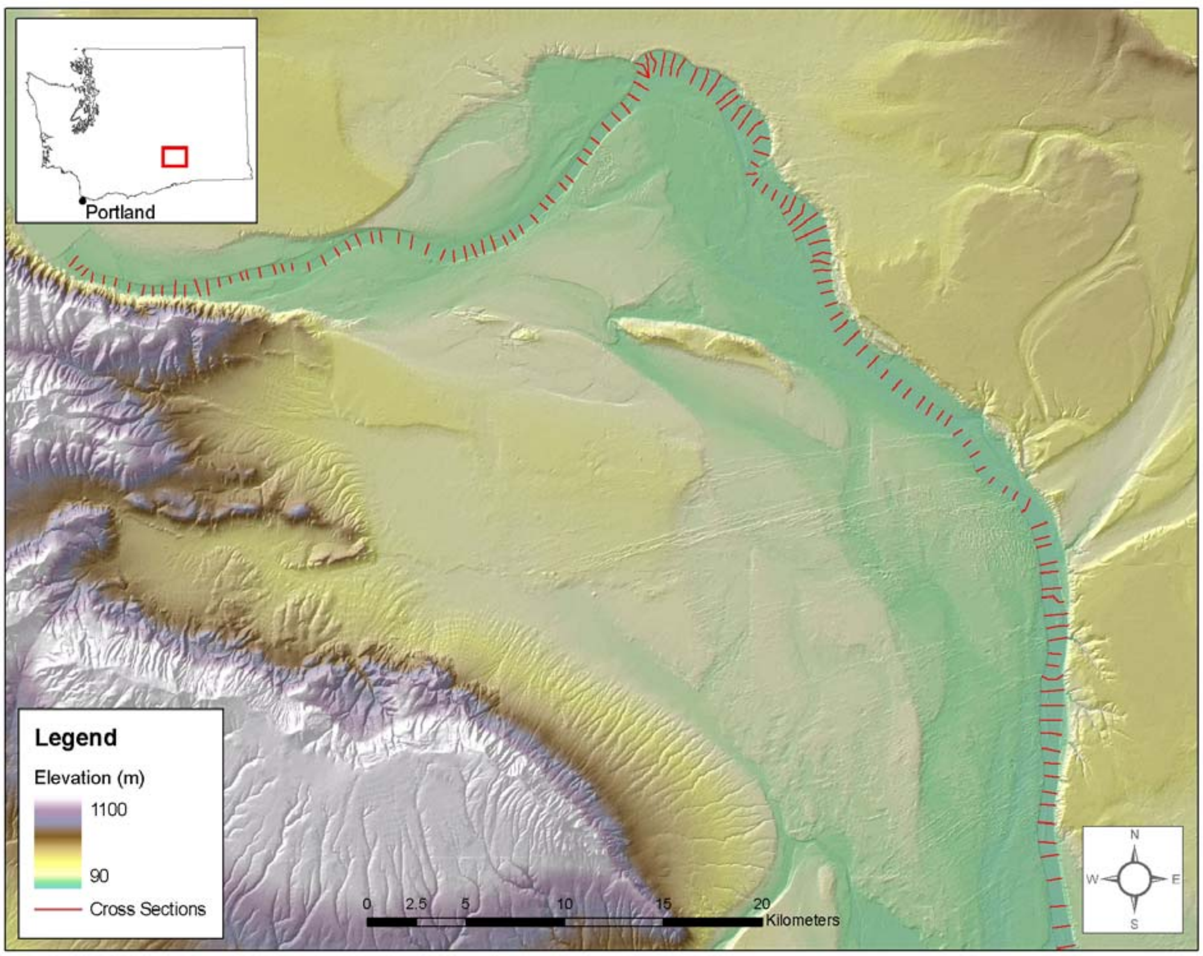


Figure 25. USACE cross section (MASS1) locations.

The MASS1 model was calibrated and verified for the Hanford Reach using empirical stage data from water level recorders maintained by the Department of Energy (DOE). The calibration and validation methods and results are discussed in detail in McMichael et al. (2003). Following is a brief summary of those methods and results.

MASS1 was calibrated and validated by adjusting values of Manning's *n* to minimize the difference between simulated stage and empirical stage observations from the DOE water level recorders and other stage monitors. Values of Manning's *n* were chosen so that the absolute value of the bias was less than 0.03 m when possible. Resulting calibration *n* values generally ranged from 0.022 to 0.029, with a single value of 0.0352 being required for a short, 6 km section of the Reach.

Validation of MASS1 was conducted by comparing simulated stages, and observed stages for three years at eight locations. Strong linear relationships characterized all comparisons, with all but one R^2 value ranging between 0.90 and 0.99, and an average R^2 value of 0.96. Mean absolute error (MAE) ranged from 0.05 to 0.21 m, with an average MAE of 0.11 m.

MASS1 Model Inputs and Simulation Details

The primary data required from MASS1 for our analyses was water surface elevation and discharge at the 145 cross section locations throughout the Hanford Reach. To generate these data, MASS1 required stage and discharge data from several sources. Hourly forebay elevations at McNary Dam on the Columbia River were used for the downstream boundary condition. Hourly discharge data were used for inflow boundaries at Priest Rapids Dam and Ice Harbor Dam on the lower Snake River. The Yakima River and the Walla Walla River were included as tributaries to the Columbia River, and observed hourly discharge was used as input data.

Water surface elevations and discharge at cross sections in the lower 27 km of the Hanford Reach are influenced by inflows from the Snake, Yakima, and Walla Walla rivers, and the backwater effect from McNary Dam and reservoir. The elevation of McNary reservoir has the largest influence on MASS1 calculated outputs for the lower cross sections. Water surface elevations and streamflows in the upper 55 km of the Reach are unimpeded except for local backwater effects from hydraulic controls.

Real and simulated streamflows and forebay elevations were used with MASS1 to characterize Hanford Reach water surface elevations for various scenarios. Steady state model runs were conducted to produce water surface elevations for flow band identification so entrapments could be coded to their respective flow band. These model runs also provided water surface elevations and discharges to be used as boundary conditions for 2-dimensional flow modeling using River2D. Unsteady state model runs were conducted with the actual 2003 streamflow data from Priest Rapids Dam to reproduce water surface elevations at half-hour intervals for the 145 cross sections throughout the Reach associated with the constantly changing streamflows characteristic of hydropower operations at the Dam. Unsteady state model runs were also conducted using alternative hydrographs at Priest Rapids Dam for evaluation of a range of operational scenarios and the effect on flow fluctuations throughout the Reach. Development of these alternative hydrographs is discussed later under the Entrapment/Stranding section of the report. Data sets resulting from MASS1 runs of the actual 2003 hydrograph and alternative hydrographs were then used to build entrapment histories for the entrapment analysis.

We ran MASS1 steady state in increments of 10 kcfs at streamflows ranging from 30 to 400 kcfs to identify water surface elevations for each flow band for entrapment coding. Average April and May inflows for the Snake (80 kcfs), Yakima (8 kcfs), and Walla Walla (1 kcfs) rivers, and average forebay elevations for McNary reservoir (103.2 m) were used as inputs to simulate typical springtime flows and water surface elevations. Streamflows used to run MASS1 unsteady for the 2003 hydrograph at Priest Rapids Dam ranged from 60 kcfs to 261 kcfs.

Triangulated irregular network (TIN) data models were developed using the water surface elevations at each transect generated by steady state MASS1 model runs. Water surface elevation maps were created from the TINs for each 10 kcfs increment. These water surface elevation maps were superimposed and intersected with the final digital elevation model (DEM) for the Hanford Reach to generate shoreline maps (flow bands) for each streamflow increment (Figure 26). The intersection of the TIN and the DEM represents the new modeled shoreline. We used the original high resolution DEM for this analysis with a cell resolution of approximately 4 m. The shoreline maps were combined in our GIS to establish the area of shoreline exposed by each 10 kcfs streamflow reduction. Entrapments located through field observations were plotted on these maps to determine the streamflow band at which the entrapments formed. Entrapments are defined as shallow depressions along the river bank that result in isolated pools. This event occurs in the simulations when flows cover an entrapment on the GIS layer and then recede to a level where the entrapment is isolated from the main river channel. The total entrapment events for every entrapment were tabulated in the simulation, building a complete entrapment history.

Figure 26. Graphical representation depicting the intersection of the MASS1 modeled water surface elevation as a TIN, and the DEM of the stream channel for a subsection of the Hanford Reach.

MASS1 Steady-State Streamflow Simulation Results

Steady-state flow model simulations from MASS1 produced the data that allowed us to construct and locate 10 kcfs flow bands throughout the entire Hanford Reach on both river channel shorelines and all islands. We were then able to calculate the area for each flow band (Figure 27) and determine the flow-related locations of entrapments mapped during field surveys.

We chose to model at 10 kcfs flow increments for several reasons, including modeling error. As discussed previously, mean absolute error from validation of MASS1 water surface simulations ranged from 0.05 to 0.21 m, and averaged 0.11 m. From the MASS1 modeling we conducted for the entrapment evaluation, we calculated an average change in water surface elevation of about 0.2 m between each 10 kcfs flow band (range 30 – 400 kcfs). We did not want to risk the occurrence of "overlapping" flow bands by using increments that were smaller than the potential modeling error.

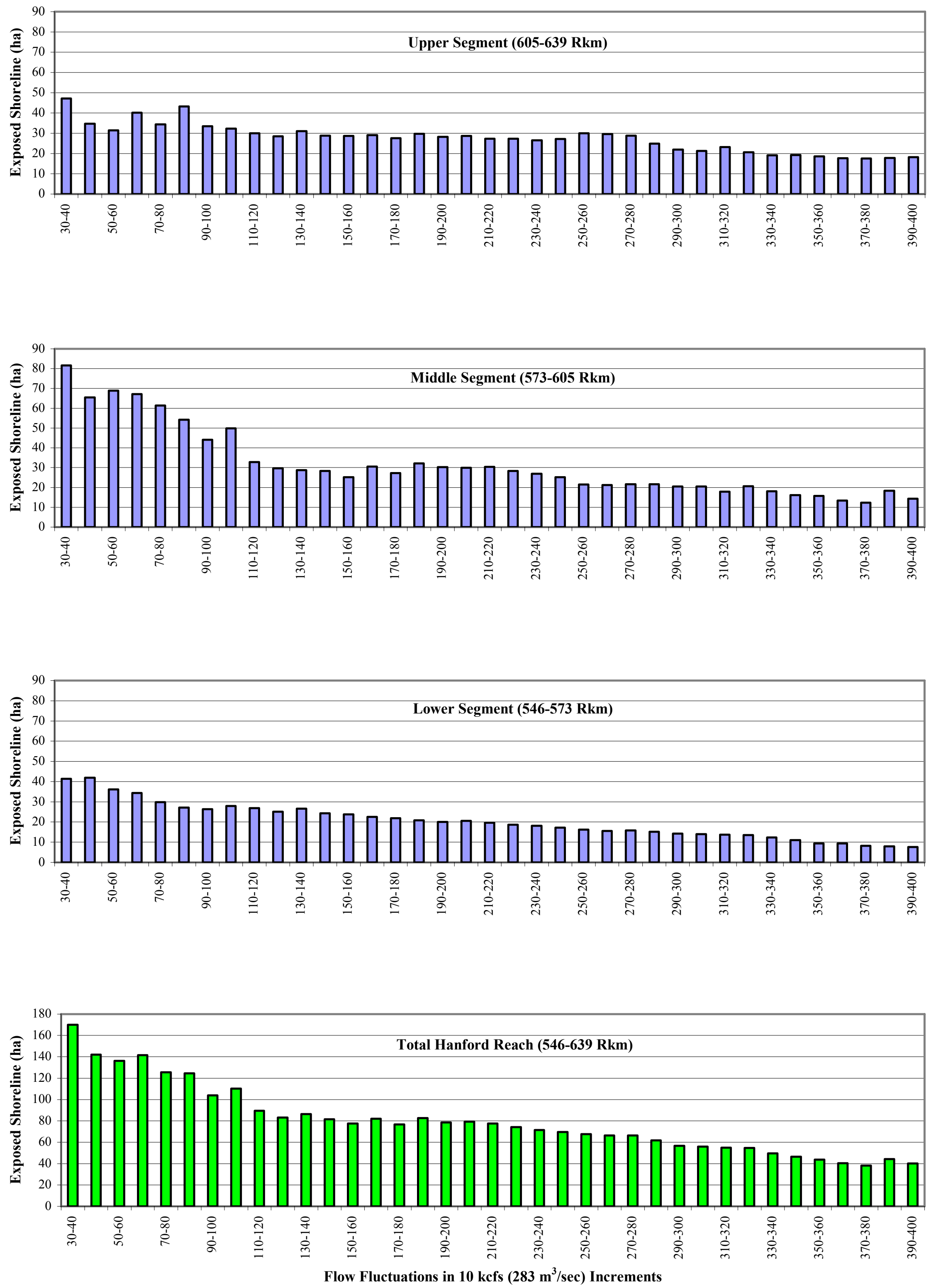


Figure 27. Relative area of shoreline exposed within each 10 kcfs flow band.

These steady-state flow model runs also provided stage-discharge data for boundary conditions for River2D modeling. For the upper segment, target water surface elevations ranging from 114.78-121.43 m were simulated at rkm 605 near the upstream end of White Bluffs to correspond with discharges ranging from 30 kcfs to 400 kcfs. For the same range of discharges in the lower segment, target water surface elevations ranging from 104.46-108.67 m were simulated at rkm 552 near the Port of Benton, upstream from Richland, Washington. Additional steady state simulation details for five cross sections at various locations throughout the Reach can be found in the form of rating curves in Appendix C.

Mass1 Unsteady-State Streamflow Simulation Results

We used MASS1 to model a total of 912,050 half-hour unsteady discharge and water surface elevation data points for simulation of the spring 2003 hydrograph for the entire Hanford Reach. Discharges and water surface elevations were modeled at each of the USACE cross sections from February 20 to June 30, 2003 to encompass the entire rearing period. Figure 28 illustrates the seasonal variability of the hydrograph during the rearing season, and shows average daily streamflows for comparison. These simulations are real-time in that for all times and locations throughout the Hanford Reach, they provide accurate hydraulic simulations (discharge and water surface elevation). This process was required for quantification of the distribution of entrapment events as they were created for all segments of the Reach, and for capturing wave dissipation (Figure 29).

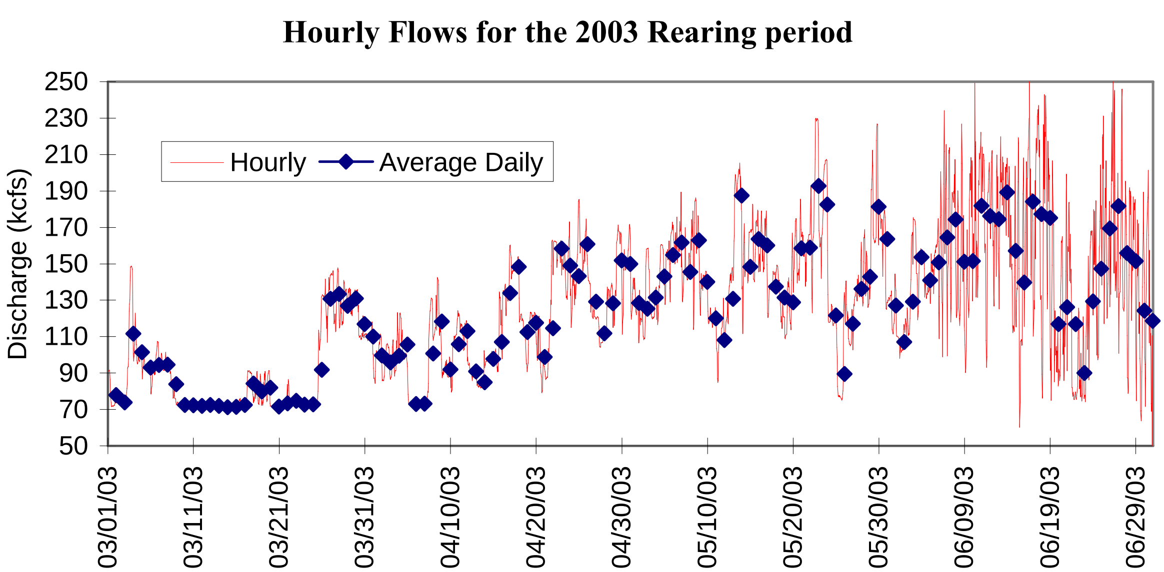


Figure 28. Hourly and average daily discharge values for the 2003 rearing season.

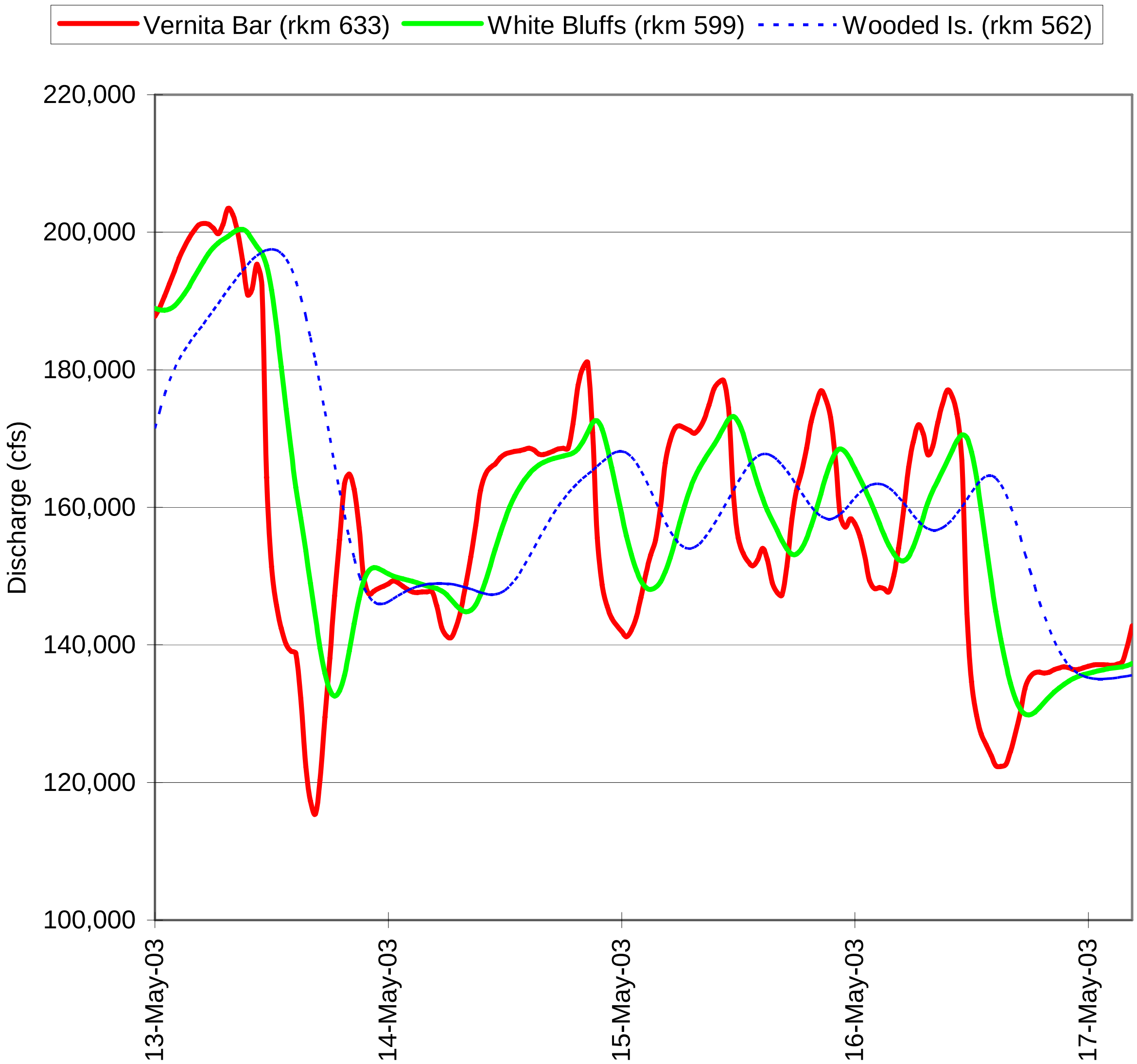


Figure 29. Example of MASS1 unsteady flow output and discharge wave-dissipation from locations in each of the three study segments in the Hanford Reach.

MASS1 Discussion

The results of the MASS1 simulations, produced estimates of discharge and water surface elevation that provided the necessary boundary conditions at the required locations for our Rived2D simulations. The availability of MASS1 saved time and resources that would have been spent collecting or attempting to model similar data. As such, our resources were focused on collecting habitat and biological data. The capability of MASS1 to produce unsteady-state streamflow simulations for the entire Hanford Reach increased the efficiency of our hydrodynamic modeling tasks in that we did not have to run River2D in an unsteady state. The simulations from MASS1 provided the necessary detail and accuracy in streamflow simulations to estimate entrapment histories for the GIS analysis. We could have conducted unsteady-state simulations with River2D, but the additional time required would have reduced our efficiency, the simulations would have been complicated by segmentation of the Reach (many more model simulations), and conducting simulations for the entire Reach without segmentation would have likely exceeded the computational limits of River2D for the scale we chose.

Conclusions

The hydrodynamic modeling component of this study built upon earlier work characterizing the physical conditions in the Hanford Reach, and provides a foundation for quantitatively assessing the effects of streamflow and streamflow variation on entrapment mortality of juvenile Chinook, adult Chinook spawning habitat, and juvenile Chinook rearing habitat. Our assessment capitalizes on recent advances in habitat mapping, remote sensing technology, simulation models of hydrodynamics, statistical sampling methods, and GIS technology to characterize habitat conditions and evaluate the effects of flow and flow fluctuations on Hanford Reach fall Chinook. The hydrodynamic modeling and remote data collection techniques that we used in this study have dramatically increased our ability to accurately describe the dynamics of aquatic habitats in this large, mainstem river segment.

River2D simulations generally did a satisfactory job of estimating water velocities, depths, and water surface elevations for habitat modeling purposes in the Hanford Reach. Both the size of the Columbia River in the Hanford Reach, and the availability of existing data, presented some unique modeling challenges. We believe that the performance of River2D was, in general, a fair representation of the hydraulic conditions in the Hanford Reach. In spite of some limitations, River2D proved to be a useful tool for predicting hydraulic conditions at the scale we evaluated fall Chinook salmon spawning habitat. We believe that collecting additional bathymetry for the DEM that was used for the middle segment would improve the accuracy of River2D, especially in the areas where the original SHOALS data were sparse. We also believe that additional calibration and validation of River2D could improve the accuracy of simulated conditions. These work tasks are planned as part of our continued Hanford Reach studies.

The availability of MASS1 saved time and resources that would have been spent collecting or attempting to model similar data. The capability of MASS1 to produce unsteady-state streamflow simulations for the entire Hanford Reach increased the efficiency of our hydrodynamic modeling tasks in that we did not have to run River2D in an unsteady state. MASS1 simulations provided the necessary detail and accuracy in streamflow simulations to estimate entrapment histories for the GIS analysis, and it also provided boundary conditions for our River2D modeling which saved the time that would have been required to develop rating curves at those locations.

Recommendations

- Bathymetric work should continue until an acceptable level of data has been acquired to improve the DEM used for hydrodynamic modeling, particularly in the middle section of the Reach.

- Substrate characterization should continue until coverage is complete for all areas to provide the relevant level of detail for both calibration of River2D and to improve spawning habitat simulations.

- Thorough, comprehensive calibration of River2D should be completed as the first step towards increasing the accuracy of hydraulic simulations.

- Reach-wide River2D validation should be continued and completed in conjunction with calibration.

- Work should continue towards running the River2D model in transient or unsteady mode to enhance future spawning, entrapment, and stranding studies.

ENTRAPMENT EVALUATION

Overview and Objectives

Significant stranding and entrapment losses of juvenile fall Chinook salmon in the Hanford Reach due to load-following operations at Priest Rapids Dam have been previously estimated (McMichael et al. 2003). While the previous studies funded by the Bonneville Power Administration (BPA) and Grant PUD were the first attempts to quantify the magnitude of fall Chinook mortality caused by dam operations in the Reach, several issues confound and limit the utility of the resulting estimates. First, the sampling for these studies was conducted on only a portion of the Reach, leaving the remaining portions unassessed. Second, the sampling plan specified the random selection of sites within areas defined by 40 kcfs flow bands without regard to the magnitude of the fluctuation. The result was the inclusion of samples where dewatering may not have occurred. Third, the sampling plan did not explicitly incorporate the spatial or temporal dynamics of stranding and entrapment. Stranding and entrapment impacts are highly variable over the rearing season and along the Reach, and thus the assessment plan needs to account for these seasonal and spatial patterns. Fourth and most important, the sampling approach had problems with detecting stranded fish. Fish stranded on substrates (Figure 30) within the Hanford Reach are inherently difficult to find (i.e., detectability is low, even when fish are present). On larger substrates fish tend to migrate downwards as the water recedes, requiring excavation of the site to locate dead fish. On finer substrates, fish are exposed to predators and are often quickly removed. Because of the problems with detection of stranded fish, these previous estimates of stranding and entrapment impacts are likely biased low.

Figure 30. Stranded fall Chinook salmon found on substrates impacted with fines. Most substrates within the Hanford Reach are not impacted making it very difficult to locate any stranded Chinook.

To address these issues, we developed a Reach-wide assessment plan which focuses on entrapment mortality to index the impacts of Priest Rapids Dam operations on Hanford Reach juvenile fall Chinook. The plan accounts for the spatial and temporal patterns of entrapment Reach-wide and utilizes fine-scale modeling to determine the effects of dam operations on the creation of entrapments. Field experience has demonstrated that there is a greater chance of detecting at least a portion of the fish isolated in entrapments compared to fish stranded on river substrates. Entrapments have the advantage of being well-defined, temporally stable, geographic locations. Flow fluctuations can create hundreds and even thousands of entrapments in the Reach during a single fluctuation event. Although fall Chinook mortality is caused by both stranding and entrapment, focusing on entrapments provides a more tractable index for assessing the minimum impacts due to flow fluctuations downstream from Priest Rapids Dam.

We initiated the following research to assess and quantify these entrapment losses in order to provide information on juvenile fall Chinook impacts, to examine operational alternatives for use in the FERC relicensing proceedings for the Priest Rapids Hydroelectric Project, and for discussions with the other Columbia Basin fishery and hydrosystem co-managers. The objectives of our entrapment studies were to: 1) develop a quantitative estimate of number of fish entrapped and the number of mortalities in the entire Hanford Reach in the spring of 2003, 2) place the 2003 estimate in context with previous estimates and population-level effects and determine the potential impacts to ocean and in-river fisheries, 3) identify the relative abundance, distribution, and growth (i.e. entrapment susceptibility) of juveniles rearing throughout the Reach in 2003, 4) explore day- and night-time behavior related to habitat use and the potential for entrapment, 5) evaluate the impacts of alternative hydro operations on entrapment, and 6) determine if it is within the physical capabilities of the Priest Rapids and Wanapum Projects to re-regulate streamflows and dampen flow fluctuations downstream in the Hanford Reach. To account for observed differences in fish impacts along the length of the Reach and over time, our approach incorporates both spatial and temporal stratification.

Methods

We pursued several lines of research in our effort to quantify and understand the factors affecting fall Chinook entrapment in the Hanford Reach in spring 2003:

- To quantify the number of fall Chinook that were entrapped in 2003, we sampled entrapments to determine fish density (i.e., the number of Chinook per entrapment), and we determined the number of entrapments that were created through a combination of field surveys and modeling. Entrapments are defined as isolated pools, separated from the main river with a minimum wetted surface area of one square meter, that result from streamflow reductions. Once entrapment locations were identified, hydrodynamic models were used to recreate entrapment histories which describe the event time series of entrapment flooding and subsequent isolation from the river as a result of flow fluctuations. By integrating results of field sampling to determine the number of fish per entrapment, with the entrapment histories to determine the number of entrapment events created, we produced a Reach-wide estimate of the number of fish entrapped and the number of mortalities. This 2003 impact estimate was placed into a population-level context based on application of recent smolt-to-adult survival rates for wild and hatchery fall Chinook and estimates of fry production within the Reach. The effects of various levels of juvenile mortality on ocean and in-river fisheries were evaluated using the Pacific Salmon Commission model for Chinook.

- To assess seasonal changes in the relative abundance, distribution, and length-composition of fall Chinook in nearshore habitats, we conducted seining surveys. To examine fall Chinook diel behaviors that may lead to stranding or entrapment, we conducted underwater video surveys on rearing juvenile fish.

- To quantify the number of fall Chinook that would have been entrapped under alternative operations, we developed and analyzed simulated hydrographs using methods consistent with our entrapment-based impact estimate. As part of this process, we generated predicted entrapment histories corresponding to the simulated alternative hydrographs which covered a range of alternative operations at Priest Rapids Dam. By combining these predicted entrapment histories with the empirical data from 2003 on fish per entrapment, we examined the relative impacts of alternative operational scenarios. These impacts for alternative operating scenarios were evaluated relative to the loss of potential harvest for ocean and in-river fisheries.

- Finally, to reduce entrapment formation and impacts, we examined the physical storage capacity at the Priest Rapids and Wanapum Projects to determine the potential for dampening flow fluctuations in the Hanford Reach.

Entrapment Enumeration

Because the individual entrapment (Figure 31) was the sampling unit selected for the impact analysis and evaluation in 2003, we needed to 1) survey and identify the population of entrapments present in the Hanford Reach, and 2) estimate the number of times that these identified entrapments were created over the rearing period. The total population of entrapment events and their geographic locations were required both for expansion of the entrapment fish sampling results to the entire Reach for the impact estimate, and for simulation modeling to evaluate alternative hydro operations and their expected impact on juvenile Chinook.

Figure 31. Typical entrapments mapped in the Hanford Reach.

Total Numbers and Distribution of Entrapments

We define an entrapment as an isolated pool separated from the main river with a minimum wetted surface area of one square meter. Several different efforts were used to quantify the total population of entrapments in the Hanford Reach. These efforts included: entrapment enumeration during entrapment fish surveys, aerial counts of entrapments, and extensive shoreline surveys during fall 2003. Because the aerial surveys were not geo-referenced and there were problems with differentiating and identifying individual entrapments, the population of entrapments consisted of only those identified during the entrapment fish sampling and the 2003 shoreline surveys.

Entrapment counts during fish surveys. - Entrapment fish sampling sites were randomly selected within each river segment as part of the fish field sampling protocol in 2003 (described below). All entrapments within the recently-dewatered zone of each selected entrapment fish sampling site were enumerated and mapped by field crews. Crews recorded the GPS coordinates at the center of all isolated entrapment pools, took measurements to determine surface area of the pool, and numbered and flagged each pool. We also estimated the initial size of the pool when it became isolated from the river and before any drainage would have occurred.

Aerial entrapment counts. - We conducted weekly flights over the Hanford Reach to determine if total counts of entrapments could be made more efficiently from a fixed-wing aircraft. Flights were scheduled weekly, on Saturdays, corresponding to expected weekend reductions in discharge from Priest Rapids Dam. The flights were conducted to determine the feasibility of obtaining both real-time counts and a video record of the total number of entrapments that typically form during weekend decreases in river discharge, and to identify critical locations where large numbers of entrapments form. During each flight, real-time counts of entrapments isolated from the river were made, and each shoreline was videotaped for enumeration of entrapments later, from the video record. Flights were conducted at 0900 hours on April 12, 19, 26, May 10, 17, and 24, 2003. Results from aerial counts of entrapments provided us with qualitative data on the distribution, relative numbers, and locations where entrapment densities were high.

Shoreline surveys during fall 2003. - To further identify the population of entrapments in the Reach, walking shoreline surveys were conducted during fall 2003. Field crews identified and mapped entrapments in the Hanford Reach from rkm 546 near Richland, WA upstream to rkm 639 at Priest Rapids Dam. Daily survey locations were determined based on the changes in discharge during the previous day at Priest Rapids Dam and the expected locations of dewatered areas along the Reach. Load-following was in effect so it was crucial to know where and when sampling segments along the Reach would be the least inundated with water to conduct a comprehensive survey. Load-following actually helped facilitate the shoreline surveys by impounding many of the entrapments overnight and leaving pools the next morning. However, in the lower half of the Hanford Reach some sampling segments were surveyed only at high discharge levels, which limited the lateral extent of the survey to high riverbank elevations and reduced the number of entrapments that could be mapped.

Shoreline surveys were conducted on successive days from October 20 - 23, 2003 and again on October 28 - 29, 2003. Entrapments were mapped from the water's edge at the time of the survey and extended up laterally to an approximate high water mark or an estimated shoreline of 200 kcfs. We conducted these surveys along both riverbanks and on all islands for the entire Hanford Reach. At each entrapment, a surveyor recorded a GPS position and attributes on a GPS data logger. Attributes included: size class (1 to 19.63 m^2, 19.64 to 176.7 m^2, and greater than 176.7 m^2), date, time, and the presence or absence of water. Entrapments of area less than one meter were not mapped. Field crews worked in teams of 2-4 depending on the width of the riverbank mapped. Surveyors recorded a code of "0" to denote that no entrapments were present in that immediate area (approximately 200 m of riverbank). The effort included 11-14 field surveyors from the following organizations: FWS, USGS, WDFW, YN, and Nugent GIS and Environmental Services. Each surveyor mapped entrapments using Trimble GPS capable of differential correction.

<u>Determination of Entrapment Event Histories</u>

Following the identification of the population of entrapments along the Hanford Reach, we developed methods for re-creating the 2003 entrapment event history, which represents the timing and location of entrapment events in the Reach during the fall Chinook rearing period. Our approach for developing the entrapment event history included the following steps:

1) dividing the Reach into 145 locations centered on the USACE transects every 0.6 rkm,
2) spatially associating the population of entrapment locations with individual transects,
3) spatially assigning the population of entrapments to 10 kcfs flow bands,
4) applying MASS1 to propagate unsteady-state hourly flows at the transect locations,
5) based on the MASS1 output, determining the full or partial crossing of flow band boundaries at each of the transect locations,
6) and determining the expected number of entrapment events that were created.

As discussed in the Hydrodynamic Modeling chapter, the USACE surveyed 145 transects throughout the Hanford Reach approximately every 0.6 rkm. At each transect location we identified the entrapments within 0.3 rkm upstream and downstream of each transect. We then determined the 10 kcfs flow band associated with each entrapment. The specific details of the hydrodynamic modeling methods used to identify 10 kcfs flow bands were discussed previously. Identification of these flow bands was necessary to code the location of the individual entrapments.

Unsteady-state modeling using MASS1 propagated the hourly changes in streamflow during spring of 2003 from Priest Rapids Dam downstream, throughout the Hanford Reach. This modeling effort provided hourly water surface elevations and associated discharges at each of the 145 transects through the Reach. We used the results from this modeling to re-create the unsteady flow profile (i.e., the rise and fall of the water surface) spatially and temporally throughout the Reach, and the histories of flooding and subsequent de-watering of entrapments during the juvenile fall Chinook salmon emergence and rearing period (late February through

June). We then calculated the history of each entrapment by examining the number of times the flow band associated with the entrapment was flooded and subsequently dewatered.

Our next task was to query the MASS1 output at the transect locations with respect to the entrapments associated with each transect and flow band. Entrapments can be created under three situations relative to flows and flow band boundaries (Figure 32–Figure 34): flows can begin above the upper boundary and drop to below the lower boundary (a "full drop"), flows can begin above the upper boundary and partially drop into the band and subsequently rise to above the upper boundary (a "partial drop"), or flows can begin below the lower boundary and partially rise into the band and subsequently drop below the lower boundary (a "partial rise").

To calculate the total number of times that entrapments were created, we calculated the expected number of entrapments as,

$$N_{i,j,p} = \sum_{t=1}^{14} \left(F_{i,j,t} \cdot E_{i,j} + D_{i,j,t} \cdot E_{i,j} + R_{i,j,t} \cdot E_{i,j} \right) \qquad \textbf{(Equation 1)}$$

where $N_{i,j,p}$ is the expected number of entrapments created at transect *i*, within flow band *j*, during 14-day sampling period *p*, *F* is the number of full drops during day *t*, *D* is the proportion that a partial drop intrudes into flow band *j* during day *t*, *R* is the proportion by which a partial rise intrudes into flow band *j* during day *t*, and $E_{i,j}$ is the number of entrapments associated with transect *i* and flow band *j*. The $N_{i,j,p}$ were summed over flow bands *j* and transects *i* within sampling periods *p* (*p* = 1, 2, …,6) and within river segments (upper, middle, and lower)

$$T_{segment,p} = \sum_{j=30}^{400} \sum_{i} N_{i,j,p} \, . \qquad \textbf{(Equation 2)}$$

The resulting $T_{segment,p}$ summarizes the expected number of entrapments that were created in each combination of river segment and 14-day sampling period.

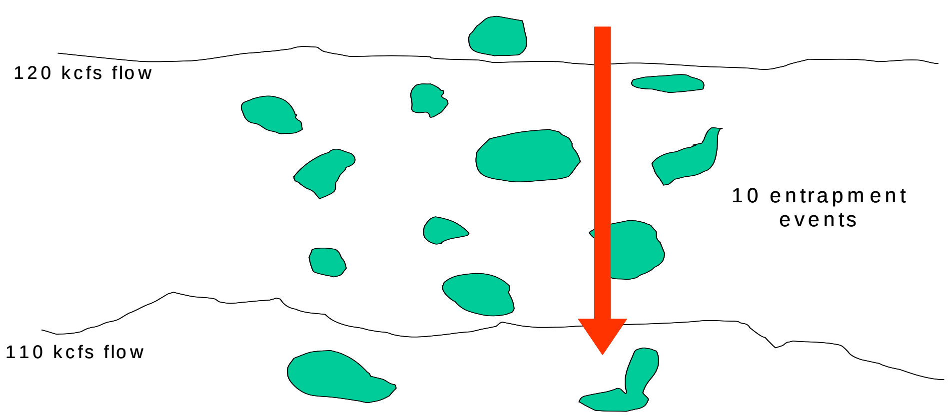


Figure 32. An illustration of a full drop through the 10 kcfs flow band boundaries. In this case, flows began above 120 kcfs and dropped to below 110 kcfs, resulting in 10 entrapment events at this location from the 110-120 kcfs flow band.

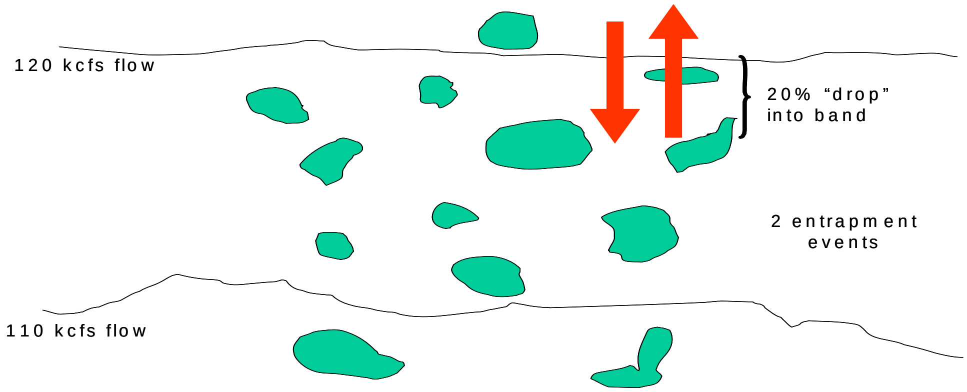


Figure 33. An illustration of a partial drop through the 10 kcfs flow band boundaries. In this case, flows began above 120 kcfs, dropped to 118 kcfs, and rose to above 120 kcfs, resulting in a 20% intrusion into the flow band. The expected number of entrapments is 20% times the number of entrapments in the flow band, or 2 entrapment events in this example.

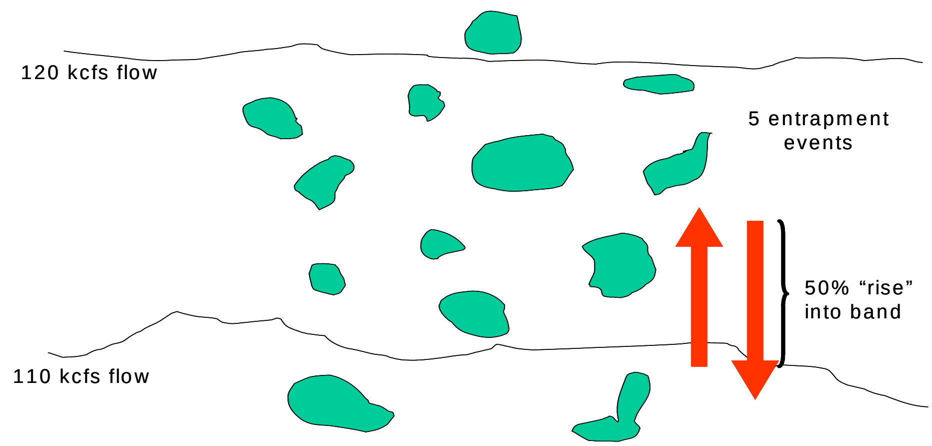


Figure 34. An illustration of a partial rise through the 10 kcfs flow band boundaries. In this case, flows began below 110 kcfs, rose to 115 kcfs, and then dropped to below 110 kcfs, resulting in a 50% intrusion into the flow band. The expected number of entrapments is 50% times the number of entrapments in the flow band, or 5 entrapment events in this example.

Juvenile Distribution

Juvenile fall Chinook salmon were seined from 15 nearshore sampling sites in the Hanford Reach once a week during the emergence and rearing period to assess relative abundance, distribution, and fish length. The 15 sites were dispersed throughout the study area from Howard Amon Park in Richland (rkm 544.0) to Vernita Bar (rkm 635.7) (Table 6). To help account for differences in relative abundance and fish length along the Reach, we divided the Reach into three river segments for sampling: upper (Priest Rapids Dam to rkm 605), middle (rkm 605 to rkm 573), and lower (rkm 573 to rkm 544).

We initiated seining surveys on February 19, 2003, one day prior to the estimated start of emergence (Nugent et al. 2002d), and continued through June 23. We sampled six nearshore locations within the middle segment of the Hanford Reach from Locke Island (rkm 600.2) to the 100F area (rkm 589.0) from February 19 through March 24, following the standard protocol from evaluations in prior years. We then expanded the sampling to 15 sites from Vernita Bar (rkm 632.5) downstream to Howard Amon Park (rkm 544.0) near Richland, Washington from March 31 through June 23 at weekly intervals.

Table 6. Nearshore sites by river segment used to determine relative abundance and length composition of juvenile fall Chinook salmon in the Hanford Reach.

River Segment	Seining site	Location
Upper	1	Below Vernita Bar
Upper	2	China Bar
Upper	3	Coyote Rapids
Upper	4	Island #1
Middle	5	Island #2
Middle	6	Locke Island
Middle	7	DOE ferry landing
Middle	8	100 F Area
Middle	9	Upstream of Hanford Slough
Middle	10	Hanford Slough
Middle	11	Lower end of Savage Island
Lower	12	Homestead Island
Lower	13	Wooded Island
Lower	14	North Richland
Lower	15	Howard Amon Park

Seining techniques were similar to methods described by Key et al. (1994). A beach seine, 21.3 m x 1.8 m with a 1.8 m^2 bag, 4.8 mm diamond mesh, and 15.2 m leads, was used to collect juvenile fall Chinook salmon and other fish species from the designated nearshore sampling sites. One lead of the seine was attached to the bow of a 5.5 m boat, the seine was folded and laid on the bow, and the other lead was held by a person on shore. The boat was then backed perpendicular to shore to a distance of 15.2 m and then backed upstream allowing the seine to be fed out parallel to shore. Once the seine was deployed, the boat was maneuvered back into shore. Both ends of the seine were then simultaneously hauled to shore. The surface area sampled in this manner was approximately 320 m^2.

When samples contained less than 100 juvenile fall Chinook salmon, we anesthetized all fish with tricaine methanesulfonate (MS-222), obtained a total count, and measured and recorded fork lengths. If samples were larger than 100 juvenile Chinook but less than 1,000, we obtained a total count, anesthetized a subsample of 100 Chinook, and measured and recorded fork lengths. When samples were larger than 1,000 juvenile Chinook, we subsampled to estimate total numbers and obtain length-frequency data. Subsampling was necessary to reduce holding time and stress. Our subsampling protocol consisted of releasing two nets of Chinook from the holding tank to the river and counting the Chinook in one net. The single net count from the retained Chinook was multiplied by three to estimate the total number of fish sampled at the site.

All fish were released back into the river after sampling. River temperature, relative velocity, dominant and subdominant substrate size (modified Wentworth code; Platts et al.

1983), substrate embeddedness (Platts et al. 1983), and vegetation density (absent, sparse, medium, or dense) were recorded for each site.

We compiled field data from the seining surveys to examine patterns in relative abundance, distribution, and fish size both temporally and spatially. We graphically summarized the patterns in juvenile Chinook abundance in each segment of the Reach over time to determine if entrapment results (described below) were related to the relative abundance or distribution of rearing fish in nearshore areas. We compiled length-frequency data and used the minimum length as an indicator of ongoing emergence. We also calculated mean fork lengths to help determine the length at which juvenile Chinook susceptibility to entrapment declines.

Entrapment Fish Sampling

Fish per Entrapment

We implemented an entrapment-based approach for quantifying impacts of flow fluctuations on juvenile fall Chinook across the Hanford Reach in 2003. The entrapment-based approach facilitated a wider distribution of sampling effort because field crews were able to concentrate on tangible and readily-identifiable entrapments rather than having to conduct area-based sampling at streambank sites that may not contribute to the problem but require time-consuming sampling efforts (e.g., the previous area-based approach described in McMichael et al. 2003). The field sampling protocol for quantifying the density of entrapped fish (i.e., number of Chinook per entrapment) was based on a stratified, random sampling design that included designation of river segments (upper, middle, and lower) for entrapment sampling, as well as entrapment sampling sites within each river segment. Similar to the seining surveys, segments for entrapment sampling included: upper (Priest Rapids Dam to rkm 605), middle (rkm 605 to rkm 573), and lower (rkm 573 to rkm 544). This spatial stratification was adopted to help account for potential differences in the number of fish per entrapment along the length of the Reach.

We identified a total of 81 sampling sites among the three sampling reaches prior to the start of the field season. Entrapment fish sampling took place at 74 of those sites. The upper river segment contained 15 sampling sites, the middle river segment contained 31 sampling sites, and the lower river segment contained 28 sampling sites. River segments to be sampled on any given day were dependent on the magnitude and duration of the previous flow fluctuation and the resulting expected response in downstream areas. Sampling was not always conducted in all river segments because the combinations of flow fluctuation amplitude and wave dissipation did not always result in observable flow fluctuations in downstream areas. Thus, we concentrated sampling in upstream river segments when fluctuations were too small to affect downstream areas, and we conducted sampling in all river segments when events were large. Figure 35 provides an example of the wave dissipation that occurs in the Hanford Reach. Flow drops that occurred at the end of April 29th and 30th, and during May 3rd were relatively large. The resulting flow reductions that occurred downstream in the Hanford and Richland reaches were also large, although they were dampened and displayed a temporal lag. All three river segments were sampled during these types of events. Smaller flow drops that occurred on May 1st and 2nd

were too small and gradual to warrant a sampling effort in the downstream river segments. Only the upper segment was sampled during these types of events.

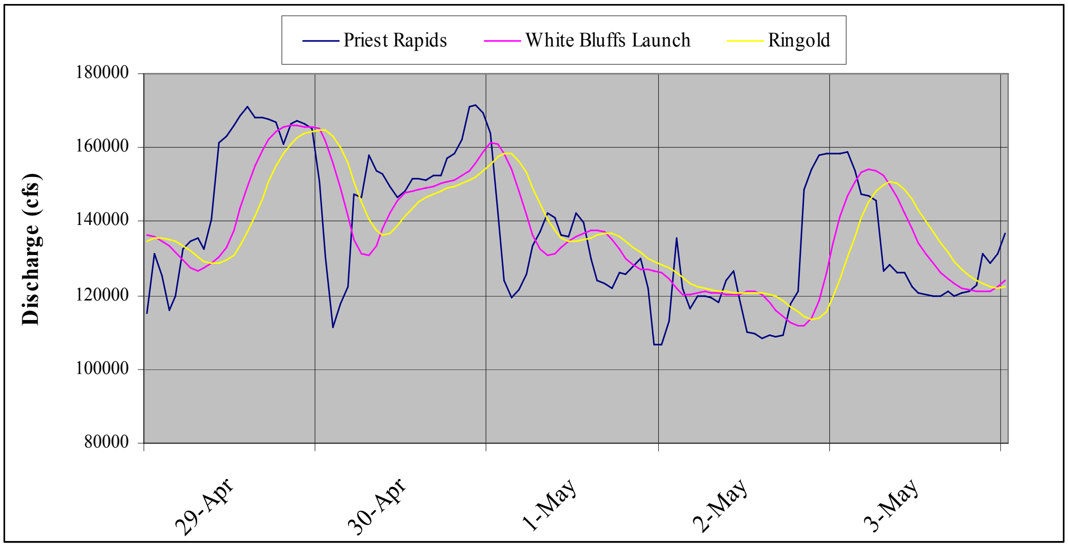


Figure 35. Hourly flows in the Hanford Reach below Priest Rapids Dam, and in the White Bluffs and Ringold areas, April 29 – May 3, 2003.

We determined the river segments to be sampled on a given day based on a review of the hourly hydrograph for the previous 48 hours and the size of the flow drop(s) that occurred (using data similar to those described in Figure 35). Sample sites within each river segment were selected randomly without replacement. Three, two-person crews were scheduled to work seven days a week from April 1 through June 21, 2003 in order to conduct sampling over the entire 90 km Hanford Reach. The unpredictable occurrence of flow fluctuation events required sampling every day of the week. We implemented temporally-staggered work shifts that encompassed all daylight hours to enable the three crews to sample sites within river segments where flow fluctuations were most likely to have produced entrapments based on the magnitude and duration of the reductions in discharge from Priest Rapids Dam.

Upon arrival at a sample site, we counted all entrapments within the zone that was recently dewatered. If no entrapments were present at the designated site, crews moved to the closest adjacent site. Field crews recorded GPS coordinates at the center of all isolated entrapment pools, took measurements to determine the surface area of the pool, and numbered and flagged each pool. We also recorded qualitative, visual observations of fish presence, active drainage, or re-inundation of entrapments by the river. After completing the initial survey identifying all entrapments in the area, crews either surveyed all entrapments at the site or subsampled entrapments for fish presence and abundance. Determining the number of entrapments to sample was based on the time remaining in the work shift. When all entrapments could not be sampled with the time remaining, crews sampled every n^{th} entrapment. For example, if only 3 of a total of 22 entrapments could be sampled in the time remaining, every 7^{th} entrapment was sampled. A coin toss or roll of dice was used to select the first entrapment. In this example, if entrapment number 2 was selected as the starting point, entrapments 2, 9, and 16 were sampled.

We conducted detailed sampling of each selected entrapment. Data collected included an estimate of surface area, measurements of depth , maximum water temperature, drainage rate, substrate type (modified Wentworth code; Platts et al. 1983), substrate embeddedness (Platts et al. 1983), vegetation type and density (absent, sparse, medium, or dense). We used beach seines or backpack electrofishing to collect fish from entrapments and recorded the fish species present, their abundance, and their fork lengths.

In total, 935 entrapments were sampled in detail. We sampled 253 entrapments in the upper river segment, 456 entrapments in the middle river segment, and 226 entrapments in the lower river segment. To account for seasonal differences in the number of fish per entrapment, we divided the entrapment sampling season into six, 14-day sampling periods: March 30-April 12, April 13-April 26, April 27-May 10, May 11-May 24, May 25-June 7, and June 8-June 21. To account for spatial and temporal patterns in our Reach-wide estimate of entrapped fish, we calculated the mean number of Chinook per entrapment by river segment and 14-day sampling period:

$$\overline{X}_{segment,\,p} = \frac{\sum_{k=1}^{n} x_k}{n} \qquad \textbf{(Equation 3)}$$

where x_k is the number of Chinook in the k^{th} entrapment sampled for fish, and n is the number of entrapments sampled for fish, within each segment and 14-day sampling period p.

Entrapment Fates

We also attempted to determine the fate of these entrapments prior to departure. Entrapment fates were classified as follows:

- Reached lethal water temperature for fall Chinook (25°C);
- Drained;
- Large entrapment of sufficient size and depth that drainage or lethal water temperature was unlikely;
- Undetermined for entrapments that did not fit any of the previous criteria by the end of the work shift.

We left the numbered flags at entrapments that were classified as undetermined at the end of the work shift, and field crews revisited the entrapment the following day and attempted to determine a fate. All entrapments where fates could not be determined during sampling were initially listed as unknown. Fates for these unknown entrapments were assigned post-field season based on water temperature, depth, and flow history. There were 198 (21.2%) entrapments categorized as unknowns by the end of the field season.

The criteria for assigning fates to unknown entrapments post-field season were:

- All entrapments with water temperatures at or above 23°C were listed as thermally lethal;
- All entrapments with mean depth of less than 5 cm were listed as drained;
- All entrapments with mean depth greater than 5 cm were listed as either drained or reflooded based on a drainage rate of 0.019 cm per minute and the flow history for the closest transect to the entrapment.

A drainage rate of 0.019 cm per minute was the median drainage rate for monitored entrapments in 2003. Median drainage rate was used to determine entrapment drainage instead of mean as it was the more conservative rate. Mean drainage rate was higher than median at 0.03 cm per minute. Flow history (river elevation and discharge) for the closest transect was calculated by hourly discharge from Priest Rapids Dam and use of the MASS1 flow model.

Estimating the Number of Fish Entrapped

<u>Reach-wide Estimate - Stratified Approach</u>

Once the number of expected entrapment events in each segment and sampling period ($T_{segment,p}$, Equation 2) had been calculated, we estimated the number of entrapped fish as the product of the expected entrapment events and the mean number of fish per entrapment ($\overline{X}_{segment,p}$, Equation 3) within each segment and sampling period, summed across river segments and sampling periods

$$Number\ of\ Chinook\ entrapped = \sum_{segments}\sum_{p=1}^{6}(T_{segment,p}\cdot\overline{X}_{segment,p}). \qquad \textbf{(Equation 4)}$$

Following the estimation of the number of Chinook entrapped, we needed to calculate an appropriate scalar representing the mortality rate for entrapped fish to derive an estimate of the number of Chinook mortalities resulting from entrapment. We considered two approaches for estimating an appropriate mortality rate scalar. One approach was to divide the number of entrapments that went lethal by the total number of entrapments sampled (i.e., entrapment lethality). The other approach was to divide the number of fish found in entrapments that had died or were expected to die due to lethal entrapment conditions by the total number of fish sampled (i.e., fish lethality). We discovered several problems with the fish lethality approach. First, following capture and enumeration, live fish were returned to the river and therefore did not have the “opportunity” to die during the fish entrapment sampling time frame. Second, fish which burrowed into the substrate as entrapments drained and subsequently died were extremely difficult to enumerate, and not accounting for these losses would have biased our fish lethality estimate low. Third, the high variability in the number of fish in individual entrapments caused high imprecision in the fish lethality estimate. To compare the precision of these two approaches, we bootstrapped (Manly 1998) the fish lethality data and the entrapment lethality data. We found that across the bootstrap samples, the fish lethality estimates had a coefficient of variation (CV) of 46.9% while the entrapment lethality estimates had a CV of 1.7%. For these reasons, we adopted the entrapment lethality approach, calculating segment- and sampling-period-specific entrapment lethality as follows:

$$L_{segment,\,p} = \frac{\text{number of in-season lethal entrapments (drained or thermal) + number of post-season lethal entrapments (drained or thermal)}}{\text{total number of entrapments sampled}}$$

(Equation 5)

where entrapment lethality ($L_{segment,p}$) is defined as the sum of the number of entrapments that were determined to be lethal during the entrapment fish sampling season and the number of entrapments that were determined to have become lethal after the entrapment fish sampling season, divided by the number of entrapments sampled during the entrapment fish sampling season, by river segment and 14-day sampling period (p).

We combined the entrapment lethality estimates with the number of Chinook entrapped estimates to arrive at our estimate of the number of Chinook mortalities resulting from entrapment (Equation 6).

$$\text{Number of Chinook entrapment mortalities} = \sum_{segments} \sum_{p=1}^{6} (T_{segment,\,p} \cdot \overline{X}_{segment,\,p} \cdot L_{segment,\,p})$$ **(Equation 6)**

We used bootstrapping to obtain confidence bounds on our estimate of the number of Chinook entrapped and the number of Chinook mortalities resulting from entrapment (Manly 1998). The bootstrapping procedure consisted of generating 5000 bootstrap samples (with replacement) of the mean number of fish per entrapment within each segment and sampling period and the mean entrapment lethality rate (in the case of determining the number of mortalities) within each segment and sampling period. The bootstrap samples were substituted into Equation 4 and Equation 6 to generate 5000 bootstrap estimates of the number of Chinook entrapped and the number of Chinook entrapment mortalities in the Hanford Reach in 2003. The 125th and 4875th ordered estimates were used to represent the 95% bootstrap confidence bounds.

Reduced-area Estimate

Evaluations of juvenile fall Chinook salmon stranding in the Hanford Reach were conducted from 1998 through 2001 for the middle segment of the Reach under the Bonneville Power Administration's Fish and Wildlife Program and with additional funding from Public Utility District Number 2 of Grant County (GCPUD) (Wagner et al. 1999; Nugent et al. 2001, 2002a, 2002b, 2002c). Funding for these evaluations was terminated after the 2001 study. Beginning in 2002, a monitoring effort was implemented with a study area that was reduced in size to approximately half (15.7 km) that of the area originally studied during 1999-2001 (Nugent et al. 2002d). This monitoring effort was continued by WDFW and GCPUD in spring 2003 along with the new, Reach-wide entrapment evaluation that was conducted as part of our work.

To facilitate comparisons between the methodology we used, and that used by Grant PUD for their reduced-area estimate (Murray 2003), we calculated the number of entrapped Chinook using our methodology in the 15.7 km reduced-area sampled in 2003. The reduced-area was composed of that portion of the Reach between rkm 584.5 and rkm 600.2. Using the methods described above, we estimated the number of expected entrapment events in the reduced-area by sampling period ($T_{reduced-area,p}$) and the mean number of Chinook per entrapment in the reduced-area by sampling period ($\overline{X}_{reduced-area,p}$). Our estimate of the number of Chinook entrapped in the reduced-area consisted of summing the products of the $T_{reduced-area,p}$ and $\overline{X}_{reduced-area,p}$ over the sampling periods. We also calculated the entrapment lethality rate based on samples collected in the reduced area by sampling period ($L_{reduced\text{-}area,p}$) to estimate the number of entrapment mortalities in the reduced-area. Similar to the methods described above, we generated 5000 bootstrap estimates of the number of Chinook entrapped and the number of entrapment mortalities in the reduced area to determine the 95% bootstrap confidence bounds.

Determining population-level impacts

Following our estimation of the number of mortalities due to entrapment, our next objective was to place the mortality estimates into a population-level context. We used three approaches to accomplish this objective. Our first approach was to use coded-wire tagging estimates of smolt-to-adult survival rates to quantify the number of adults that may have been lost due to entrapment mortality in 2003. Our second approach was to estimate impacts relative to the number of juveniles produced in the Reach using estimates of escapement and fecundity, and published estimates of egg-to-fry survival. Our third approach was to examine how various levels of population mortality would translate into reductions in ocean fisheries.

Smolt-to-adult survival - We summarized estimates of smolt-to-adult survival rates for wild and hatchery fall Chinook in the Hanford Reach based on coded-wire tagging. The estimates of these survival rates are possible because of long term index marking (1986-present) of both wild and hatchery Chinook for the Pacific Salmon Commission coast-wide harvest assessments. The adult portion of this analysis was measured at various adult life stages to capture impacts of mortality rates on ocean and in-river fisheries, and escapement. We collected recent smolt-to-adult equivalent survival rates and applied these to the mortality estimates to quantify the number of fish which may have been lost to fisheries and escapement due to hydro operations. By examining these rates for both hatchery and wild fish (both of which are subject to entrapment), we provide some bounds on the population-level effects of our entrapment mortality estimates. However, we primarily focused on the wild fish estimates because they are most directly relevant to the Hanford Reach fall Chinook population.

Coded-wire tagging survival rate data were available for fall Chinook released from Priest Rapids Hatchery (brood years 1975-1997) and for wild fall Chinook tagged and released in the Hanford Reach (brood years 1986-1997) (Pacific Salmon Commission 2005). We summarized the data and estimated the combined survival of each of these two marked groups to ocean fisheries, in-river fisheries, and escapement back to the Hanford Reach.

Juvenile production - Previous efforts to quantify the magnitude of the impacts relative to the Hanford Reach fall Chinook population (McMichael et al. 2003) have focused on the percent reduction in the population size at the juvenile life stage (e.g., the number of juveniles killed relative to an estimate of the total juvenile population size). To estimate juvenile production, precise and accurate estimates of the number of spawners, proportion female, female fecundity, and egg-to-fry survival are required. The estimates of spawner escapement, proportion female and average fecundity are known with reasonable certainty. However, the fry production estimates are particularly sensitive to the estimates of egg-to-fry survival, which are highly uncertain.

We conducted a literature review on egg-to-fry survival rates for ocean-type fall Chinook and report the published values. Using these estimates, along with the estimates of escapement, proportion female and average fecundity determined by WDFW, we calculated estimates of recent fry production in the Reach. We compare our estimate of entrapment mortality in 2003 to estimates of fry production to quantify the proportion of the fry population that may have been lost due to entrapment. To quantify the range of potential impacts to the population while under the Juvenile Fall Chinook Protection Program, we also compared the Reach-wide mortality estimate for 2001 reported by McMichael et al. (2003) to estimates of the fry population in that year.

Impacts to ocean fisheries - In 1999, the United States and Canada reached a comprehensive management agreement under the Pacific Salmon Treaty of 1985 to implement a coast-wide, abundance-based management approach for ocean fisheries. Under this approach, allowable harvest levels for Chinook ocean fisheries are based upon the aggregate abundance of stocks contributing to each fishery. Under this management system, harvest levels can increase as ocean survival and in-river salmon production and productivity increase. More importantly, as the survival of stocks contributing to a fishery declines, the allowable harvest level will decline as well.

Under the treaty's harvest management approach, the more that a stock or stock group contributes to the stock aggregate in any fishery, the greater the impact of changes in that stock's productivity (survival) will be reflected upon that fishery. Stocks that contribute heavily to any fishery are commonly called "driver" stocks. Hanford Reach Upriver Bright Fall Chinook (URB) salmon stocks tend to be far-north migrating and contribute heavily to ocean fisheries in Southeast Alaska (SEAK) and Northern British Columbia (NBC), making them driver stocks for those fisheries.

To evaluate the effects of potential reductions in the juvenile population due to stranding and entrapment, we conducted an analysis using the Pacific Salmon Commission's Chinook model, which was developed and is maintained by the bilateral Chinook Technical Committee (CTC). The analysis was completed based upon input on survival changes for the URB stock using a hind-casting approach; that is, assuming that the juvenile salmon losses from entrapment started in 1999 and then the Chinook model is run forward from that year and losses to fisheries and escapement back to the Hanford Reach are estimated. The runs modeled use the following procedures and assumptions:

- Assume a 25% loss in URB outmigrations in the years 1999, 2000, 2001, and 2002.
- Go into the Chinook model and adjust the EV (error term in the spawner-recruit curve) for the URB stock to show a 25% reduction in age 2 cohort for the 1998 to 2001 escapements.

- Run the Chinook model and project the 2002, 2003, and 2004 abundances. Only the URB stock will show a decline due to the 25% reduction in the age 2 cohort abundances for the 3 brood years. All other stocks abundances remain unchanged.
- Calculate the abundance indices. Look up the total allowable harvest according to the treaty (Table 1 of the PSC annexes). The difference in total allowable harvest with and without the 25% reduction in age-2 cohort represents the impact of the 25% entrapment and stranding mortality rate to the ocean fisheries.
- Repeat the simulation with high (50%) and low (5%) population impact estimates.

Juvenile Behavior

We conducted underwater video surveys on subyearling fall Chinook in the Hanford Reach to assess their diel behavior and search for factors that may lead to stranding or entrapment. We hypothesized that subyearlings may be at greater risk of stranding and entrapment at night because they are less active nocturnally (Venditti and Garland 1996). We used underwater videography to observe movement patterns along shallow-water shoreline areas and to collect behavioral information during the daytime and nighttime in 2003 to determine if diel behavior might explain the relative risk of subyearlings to water level fluctuations.

Field work was conducted near the 100-F island complex (rkm 590) of the Hanford Reach from April 25 to May 10, 2003. Our underwater video system consisted of four black and white underwater cameras, which were deployed along a line extending out from the shore and perpendicular to the flow. Cameras were deployed in pairs and oriented toward each other and separated by a distance of about 1 m. One camera pair was set near shore in water at least 0.2 m deep, and the other pair was set immediately adjacent to the near-shore pair so that the middle two cameras in the array shared a common attachment point (Figure 36). This arrangement provided a minimum linear coverage of 2 m. Nighttime illumination was provided by six infrared LEDs surrounding the lens in each camera (SeaView Video Technology, St. Petersburg, FL) and by two underwater white lights fitted with infrared filter lenses (Optical Instruments Laboratory, Inc, Houston, TX). These lights were suspended between each camera pair just under the water's surface to provide overhead illumination of the center on each camera pair's field of view. Video images from the four cameras were recorded to VHS tapes using a multi-plexer, video cassette recorder, and LCD monitor. Time and date information was saved concurrently with video information.

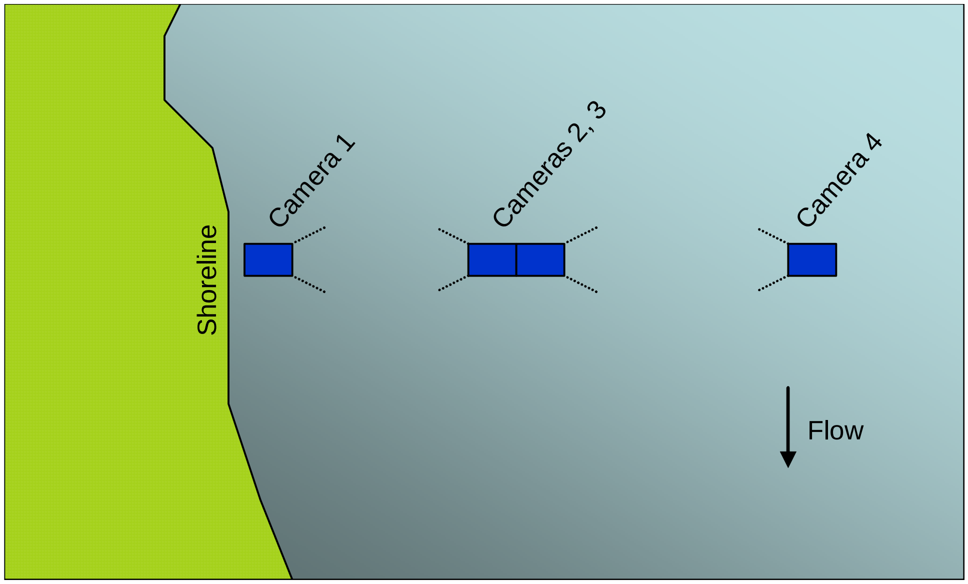


Figure 36. Overhead diagram of underwater video camera set-up during fieldwork conducted in the Hanford Reach of the Columbia River, 2003.

Water level fluctuations in the Hanford Reach required us to occasionally move the camera array into deeper water to ensure they remained under water. Video deployment sites were selected based on observations of fish presence and the likelihood of successfully monitoring fish behavior. Individual sites were monitored for periods ranging from 2 to 24 h based on the number of hours that conditions were suitable for monitoring. We deployed our video system at nine different locations and times in 2003.

Video tapes were analyzed for various aspects of subyearling Chinook salmon diel behavior and habitat associations. We pooled all video tapes then randomly selected 100 1-min clips from both daytime and nighttime periods. Nighttime was defined as the period from 0.5 h after sunset to 0.5 h before sunrise. Daytime was defined as the period from 0.5 h after sunrise to 0.5 h before sunset. Crepuscular periods included the 1-h intervals surrounding sunrise and sunset, and were included in the “daytime” dataset.

Each video clip was reviewed and values were recorded for the following categorical behavioral and habitat variables when fish were present: shoreline proximity (1=nearshore, 2=offshore); rheotactic orientation (1=upstream, 2=downstream); activity (1=feeding, 2=swimming, 3=holding); and water column position (1=top one-third, 2=middle third, 3=bottom one-third). Feeding events were defined as fish striking the surface to capture food items (Venditti and Garland 1996), while swimming was defined as any fish that entered and exited the field of view during a 1-min video clip. Fish that displayed holding behavior were those that spent the majority of their time in the field of view at a particular location. Student’s *t*-tests were used to determine whether differences existed in habitats and behaviors between daytime and nighttime periods.

Effects of Alternative Hydro Operations

To evaluate the impacts on fall Chinook entrapment associated with alternative hydro operations at Priest Rapids Dam, we developed simulated hydrographs with various levels of peaking operations and examined their effects in terms of the expected number of fall Chinook that would be entrapped under those operations.

As is evident on Figure 37, there is considerable variability in the magnitude, frequency, and duration of peaking operations at Priest Rapids Dam during the juvenile rearing period. Upon inspection of the hourly discharge data, it was evident that there was not a consistent pattern in the shape of the flow fluctuations. However, our objective was to evaluate how the number and magnitude of flow fluctuations would affect the expected number of fall Chinook that would be entrapped. To accomplish this objective we analyzed the hourly discharge data for patterns in the general shape of flow fluctuations to characterize what an "average" flow fluctuation looked like.

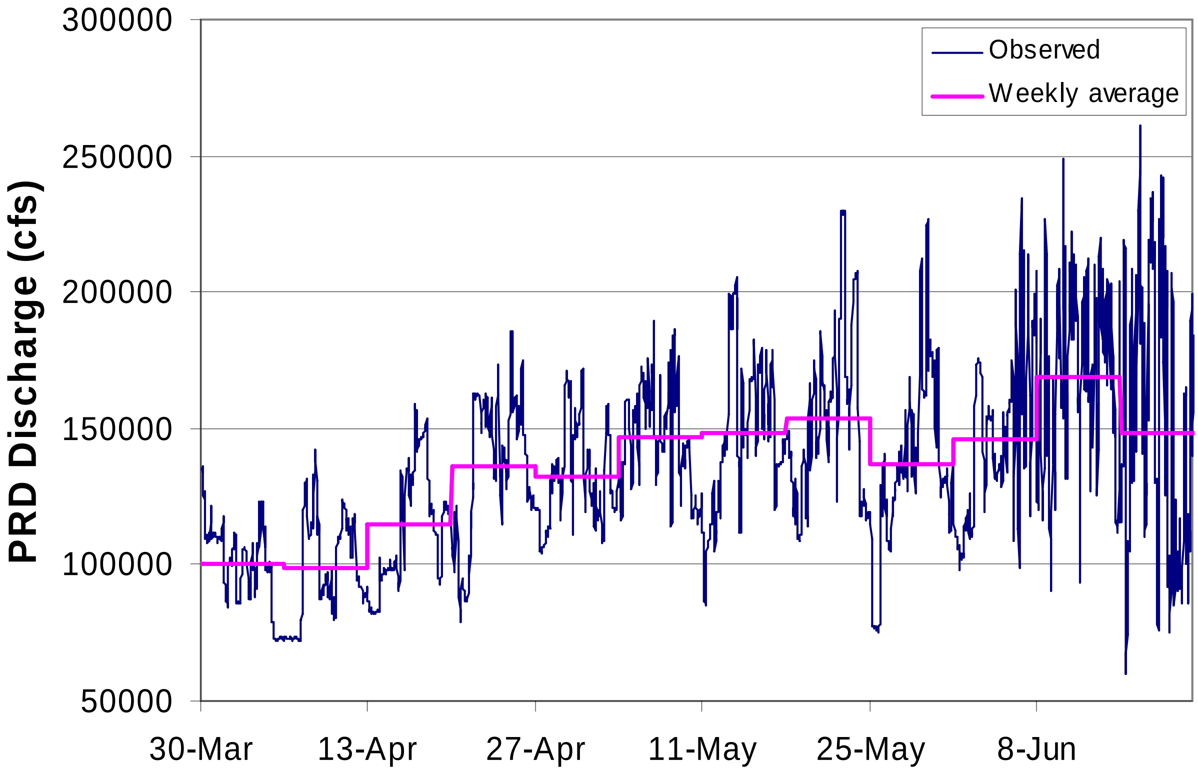


Figure 37. Observed discharges from Priest Rapids Dam during March 30 though June 21, 2003. Also plotted is the weekly average flow over the same time period.

Using the hourly discharge data collected from March 31 through June 15, 2003, we calculated the mean up-ramping and down-ramping rates (in cfs/hour) and the mean number of hours that flows were within 1% of local peak and trough flows. We found that discharges increased at a mean rate of 5800 cfs/hour and decreased at a mean rate of 6500 cfs/hour. Flows remained within 1% of the local peaks and troughs for a mean of 2 hours. Given these results, we chose to define a typical fluctuation shape as one which rose at a rate of 5000 cfs/hour, remained at the peak for 2 hours, decreased at a rate of 5000 cfs/hour, remained at the local trough for 2 hours, and then rose to the starting point at a rate of 5000 cfs/hour.

Once our typical fluctuation shape had been defined, we then developed simulated hydrographs with various numbers of fluctuations per week and various fluctuation magnitudes. These simulated hydrographs were imposed upon the weekly average flow levels to reflect the observed seasonal changes in flow volumes. We developed eight simulated hydrographs. These included either five or ten fluctuations per week, and flow fluctuation magnitudes of 10 kcfs, 20 kcfs, 30 kcfs, and 40 kcfs. Each of the fluctuations followed the typical fluctuation shape defined above. Figure 38 provides an example of two of the simulated hydrographs along with the observed discharges during May 4-10, 2003.

Once the simulated hydrographs had been developed, we used the same methods described above to estimate the expected number of entrapments that would have been created had the operations at Priest Rapids Dam followed our simulated hydrograph. Once the number of expected entrapments had been calculated by river segment and 14-day period, we applied the mean number of Chinook per entrapment estimates to arrive at an expected number of fish that would have been entrapped given the simulated hydrographs. We summarize these results to provide information on the relative impact that alternative operations would be expected to have on the number of fall Chinook that would have been entrapped in 2003.

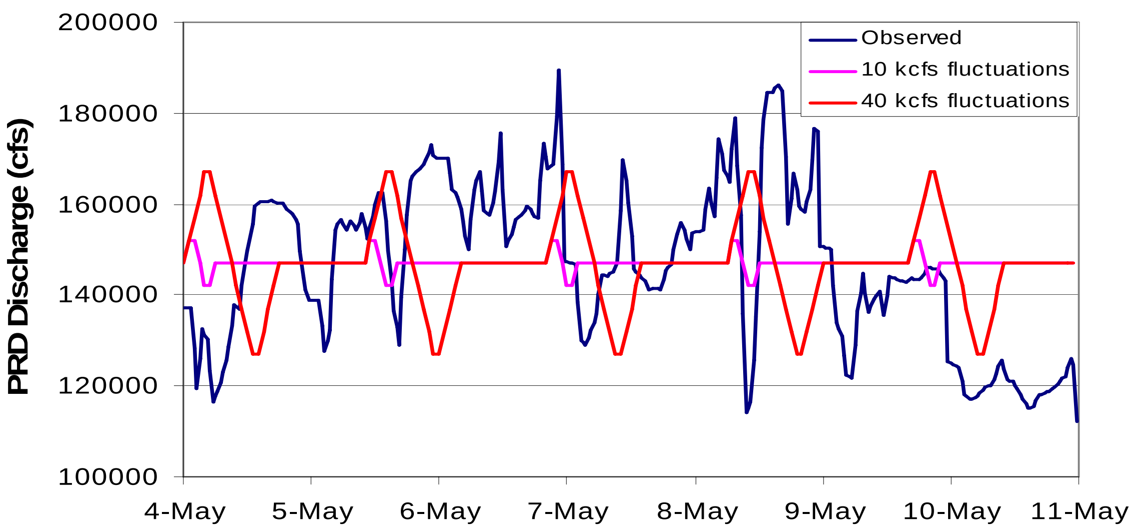


Figure 38. Observed discharges from Priest Rapids Dam during May 4 through May 11, 2003 and two of the simulated hydrographs depicting five 10 kcfs fluctuations and five 40 kcfs fluctuations per week.

Capacity for Dampened Flow Fluctuations

In addition to our analysis of the actual 2003 spring hydrograph and alternative hydrographs, we conducted an evaluation of the ability of the Priest Rapids Project to re-regulate or dampen flow fluctuations during the spring rearing period when juvenile Chinook are present. The basic components of this evaluation included using empirical data to determine the forebay volumes for the Priest Rapids Project (PRD) and Wanapum Project (WAN) that are used in the course of "normal" operations, and then using that volume to re-regulate streamflows coming into the two projects from the Rock Island hydroelectric project immediately upstream. For this analysis, operational changes were made only at the PRD and WAN projects. All of the upstream mid-Columbia projects were unaffected. Our goal was to evaluate the physical flexibility available to reduce streamflow fluctuations during the March through May rearing period for juvenile fall Chinook salmon in the Hanford Reach. We used a simple mass balance approach for this analysis. We did not have access to the optimization model used under the Hourly Coordination Agreement to incorporate the effects of re-regulation on power production in the mid-Columbia. This analysis consisted of the following steps:

- Determination of the forebay volumes in both Priest Rapids (PRD) and Wanapum (WAN) reservoirs that are available to modify incoming streamflows;
- Determination of hourly streamflows coming downstream, into PRD and WAN from the next upstream hydroelectric project, Rock Island;
- Determination of the target streamflows downstream from PRD into the Hanford Reach;
- Either supplementation of incoming flows that were lower than the target flow with storage from PRD and WAN, or storage of a portion of the incoming flows that were higher than the target flow.

Determination of Forebay Volumes in PRD and WAN

We compiled forebay elevation data for both projects from the October/November time period when a reverse-load-following operation is conducted under the Vernita Bar Agreement to control fall Chinook spawning locations. We used the October/November time period for this analysis because the projects use more forebay volume during this reverse-load-following operation than at most other times of the year. Implicit in this approach is the assumption that similar forebay volumes could be used during the spring rearing period to re-shape incoming flows and reduce flow fluctuations. The "normal" load following cycle (higher flows during the day when electrical demand is high, lower flows at night when demand is low) is reversed (lower flows during the day, higher flows at night) during the fall Chinook spawning period in an attempt to limit spawning locations to areas lower within the river channel that can be maintained throughout the winter during the low flow portion of the power generation cycle each day. This operation is based on the idea that Chinook select their redd sites and initiate redd construction during the day. During this time period, the upstream hydroelectric projects are conducting normal load following operations. Thus, the PRD and WAN projects must reverse this cycle to accomplish the reverse loading objective. Forebay fluctuations and volumes used to accomplish the operation are greater than those that occur during other times of the year when "normal" load following is conducted. We used these empirical forebay data for this analysis as a more realistic scenario than using total active storage. Total active storage is rarely, if ever used on a regular basis.

We analyzed forebay elevations and fluctuations over a ten-year period (1995-2004) when the reverse-load-following operation was being implemented (Table 7). Hourly outflows from PRD were used to determine the specific time periods when reverse-loading occurred each year. These time periods are typically characterized by steady daytime flows of between 50 to 70 kcfs. Additionally, weekend forebay data were eliminated from this data set, considering weekend operations are often different from weekday operations. We assumed it was fully within the capability of the PRD and WAN projects to operate in a similar manner on weekends. For each day of the reverse load following operation, the differences in maximum and minimum forebay elevations were calculated for both the PRD and WAN reservoirs. We then conducted an exceedance analysis using the daily delta values calculated from the observed forebay fluctuations from all periods of the reverse load following operation (minus weekends) over the ten-year period of record. We also conducted this same analysis using the differences in maximum and minimum forebay elevations on a weekly basis rather than a daily basis over the ten year record. Each week of forebay fluctuations consisted of four or five days. The exceedance analysis was repeated using these observed weekly delta values.

Table 7. Time periods from 1995 - 2004 that were used to calculate storage volumes in PRD and WAN forebays.

Year	Start Date	End Date
1995	October 15	November 19
1996	October 15	November 24
1997	October 18	November 23
1998	October 17	November 22
1999	October 16	November 19
2000	October 15	November 19
2001	October 18	November 18
2002	October 15	November 24
2003	October 18	November 24
2004	October 15	November 21

We used the 50% exceedance values for daily and weekly forebay fluctuations to estimate the associated storage volumes using the reservoir "Capacity Curves" (obtained from GCPUD) for the PRD and WAN reservoirs. Specifically, we calculated storage at each project from the top of the normal operating range (148.7 m for PRD and 174.0 m for WAN) downward to a depth equal to the 50% exceedance value. Normal operating ranges for the projects were obtained from the "1998 Columbia River Water Management Report, Appendix C" (this document can be found at http://www.nwd-wc.usace.army.mil/crwmg/crwmg_reports.htm). It is important to note that the capacity curves show that storage at each reservoir level changes with streamflow. For the purpose of our evaluation, we used a streamflow level of 100 kcfs as a reasonable value for the relevant time period; however, from inspection of the curves it does not appear that storage volumes between two elevations would be significantly different at any flow between 100 and 300 kcfs. Results of this analysis provided us with volumes for both pools based on both daily and weekly analyses that would be available to re-regulate incoming streamflows.

Determination of Hourly Streamflows into PRD and WAN from the Rock Island Project

We compiled hourly streamflows for the Rock Island project from March through May for the ten-year period from 1995 to 2004. Historic hourly streamflow data are available for all Columbia River hydroelectric projects on the USACE web site (http://www.nwd-wc.usace.army.mil). These hourly streamflows represented the starting point for the re-regulation analysis.

Determination of the Target Streamflows Downstream from PRD into the Hanford Reach

Considering that the goal of this analysis is to re-regulate streamflows, or dampen flow fluctuations, we calculated the target outflows from PRD as the weekly average flow from the historical data for March through May for the years, 1995 to 2004 for the Rock Island Project. We used a weekly average target in an attempt to dampen not only daily fluctuations within a 24-hour period, but also differences between average daily flows.

Re-regulation of Incoming Streamflows from Rock Island to Target Flows out of PRD

The final step of our analysis consisted of calculating the amount of storage needed daily at PRD and WAN to dampen flows from Rock Island to the weekly average target outflow from PRD downstream into the Hanford Reach. When incoming flows from Rock Island were higher than the target flow, the additional water was stored in either PRD or WAN reservoir, and the result was a positive volume. When incoming flows from Rock Island were lower than the target flow, water was withdrawn from either PRD or WAN reservoir, and the result was a negative volume.

We used hourly flows from Rock Island for the years 1995 through 2004 as a starting point, and we used the volumes calculated from the empirical forebay data to attempt to operate to the target flow each hour. Positive and negative volumes were accumulated as the target flow was met each hour. If we ran out of either storage volume, or supplementation water, incoming flows from Rock Island were passed through the PRD and WAN reservoirs, and a flow fluctuation occurred downstream into the Hanford Reach. As a final step, we determined the number of days for March through May each year for the ten-year time period when forebay volumes were not exceeded. These values represented the proportion of time that PRD and WAN reservoirs could have re-regulated incoming flows from Rock Island to the flat, target flow through the Hanford Reach.

Results

Entrapment Enumeration

Total numbers and distribution

We identified a total of 1,257 entrapments during entrapment fish sampling surveys in 2003. An additional 7,341 individual entrapments were identified during the intensive ground survey conducted in October 2003. After accounting for overlap between the two surveys, the total number of entrapment sites mapped and entered into the GIS was 7,932. We also counted 5,758 entrapments during six aerial flights in 2003. Aerial counts provided guidance for our October 2003 intensive ground survey, but they were not directly incorporated into the dataset because there was no way to assign specific geographic coordinates. The distribution of Reach-wide entrapments is presented in Figure 39.

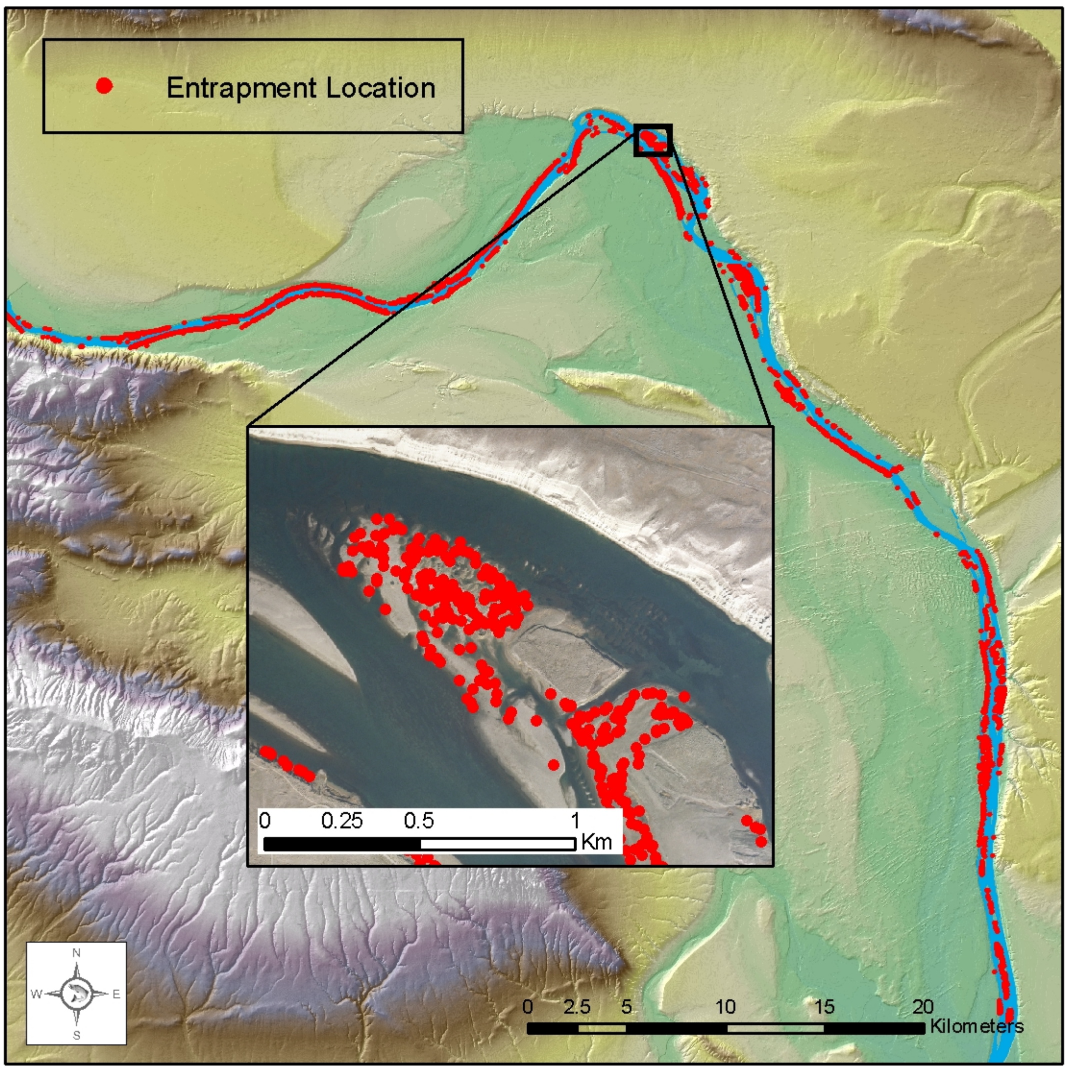


Figure 39. Distribution of entrapment locations throughout the Hanford Reach with an inset of Locke Island.

Large numbers of entrapments were identified in all three segments. The upper segment contained the most entrapments (36%), followed by the middle (33%) and lower segments (31%) (Figure 40). The number of entrapments in each flow band generally declined with flows greater than 110 kcfs (Figure 41). There were relatively larger numbers of entrapments in the upper and middle segments at lower flows (<120 kcfs), and similar numbers in all three segments at higher flows (>150 kcfs) (Figure 42).

Our data on the distribution of entrapments between segments and among flow bands reflected streamflow conditions that were present during the various surveys. Streamflows that occurred during the intensive ground survey in October 2003 did not allow us to comprehensively map entrapments in the lower flow levels (i.e., those below 80 kcfs). Streamflows at Priest Rapids Dam ranged from 38 to 117 kcfs early in the survey week and 38 to 180 kcfs later in the week. These wide-ranging flows produced variable conditions throughout the Reach, which resulted in incomplete entrapment mapping within the lower flow bands, especially for the lower segment of the Reach. The reduced frequency of entrapments below the 80-90 kcfs flow band (Figure 41) may reflect the omission of entrapments at lower flow levels which were under water during the time of the surveys. In addition, the pattern of lower numbers of entrapments in the lower and middle segments relative to the upper segment (Figure 42) may reflect the generally higher flow levels that occurred during surveys in the downstream segments. We believe that our enumeration of entrapments at the higher flow bands (i.e., those between 90 and 200 kcfs) is comprehensive.

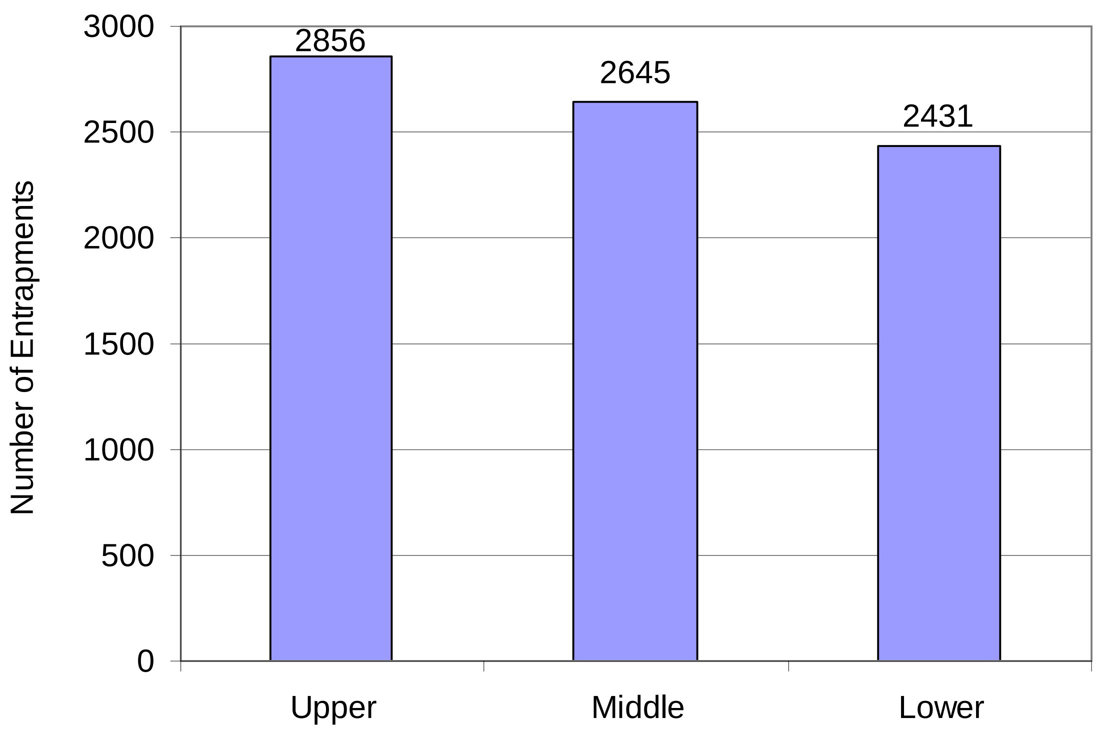


Figure 40. Entrapment distribution by river segment for the Hanford Reach.

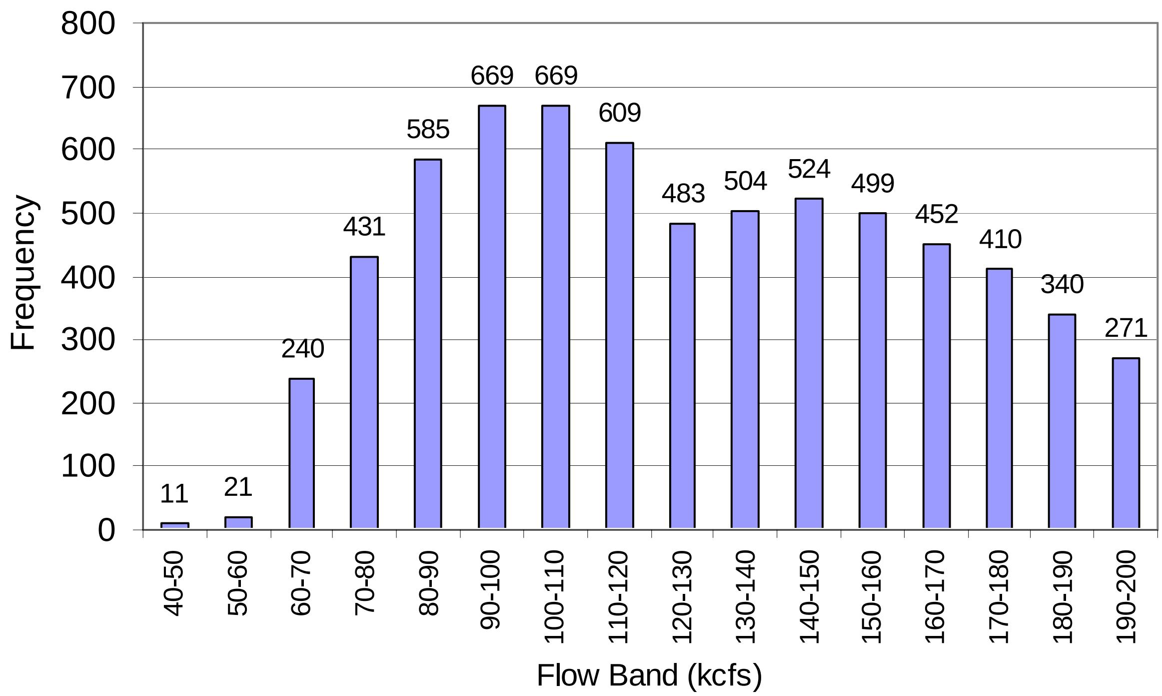


Figure 41. Entrapment distribution by flow band for the Hanford Reach.

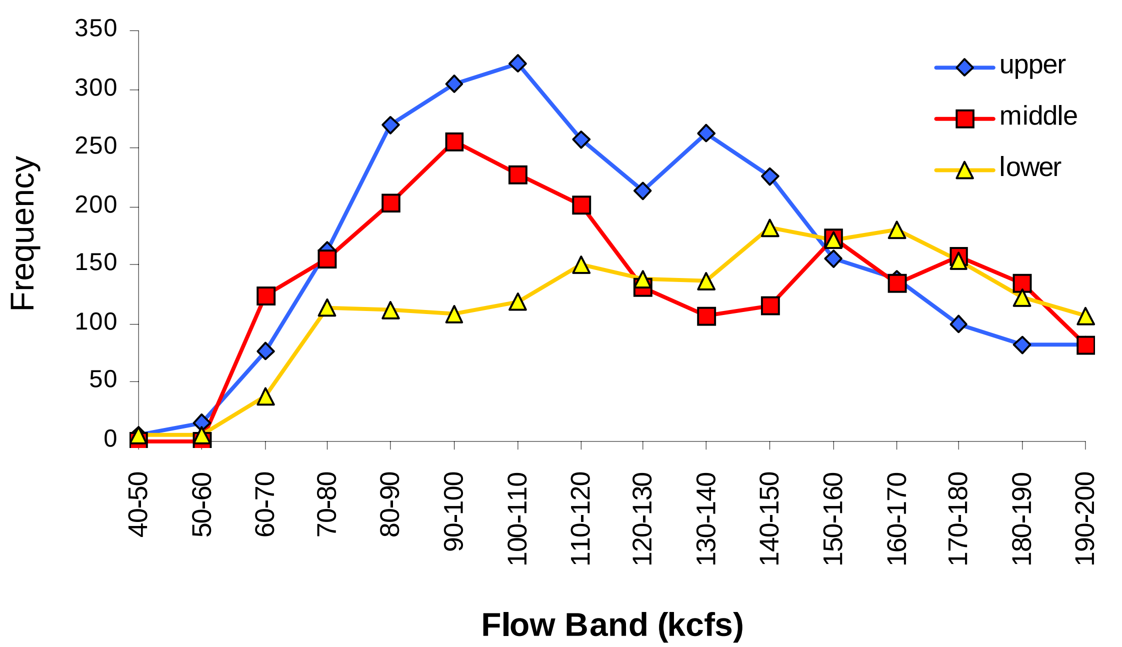


Figure 42. Entrapment distribution by river segment and flow band for the Hanford Reach.

Numbers of Entrapment Events

We estimated a total of 126,226 entrapment events in the spring of 2003 for an average of 1,503 events per day (Table 8). The total number of entrapment events ranged from 11,346 in the first sampling period to 47,581 during the last sampling period. Events generally increased over the sampling season, with a sharp increase during the June 8 – June 21 sampling period (Figure 43). A high number of entrapment events in the upper segment confirmed our expectations that wave dampening and dispersion would tend to reduce the number of entrapments created in downstream areas compared to upstream areas near Priest Rapids Dam. The incomplete mapping of entrapments in the lower segments discussed earlier may have also contributed to the lower number of estimated entrapment events in those areas.

Table 8. Entrapment events by river segment and sampling period, the total number of entrapments by sampling period and by river segment, and the average number of entrapment events per day (E/day) by sampling period, during March 30, 2003 to June 21, 2003.

	River segment				
Sampling period	Upper	Middle	Lower	Total	E/day
Mar-30 to Apr-12	7,826	1,259	2,261	11,346	810
Apr-13 to Apr-26	7,473	1,938	2,480	11,891	849
Apr-27 to May-10	9,757	3,047	3,460	16,264	1,162
May-11 to May-24	10,274	3,997	4,865	19,136	1,367
May-25 to Jun-7	11,704	3,870	4,434	20,008	1,429
Jun-8 to Jun-21	27,931	9,553	10,097	47,581	3,399
Total	74,965	23,664	27,597	126,226	1,503

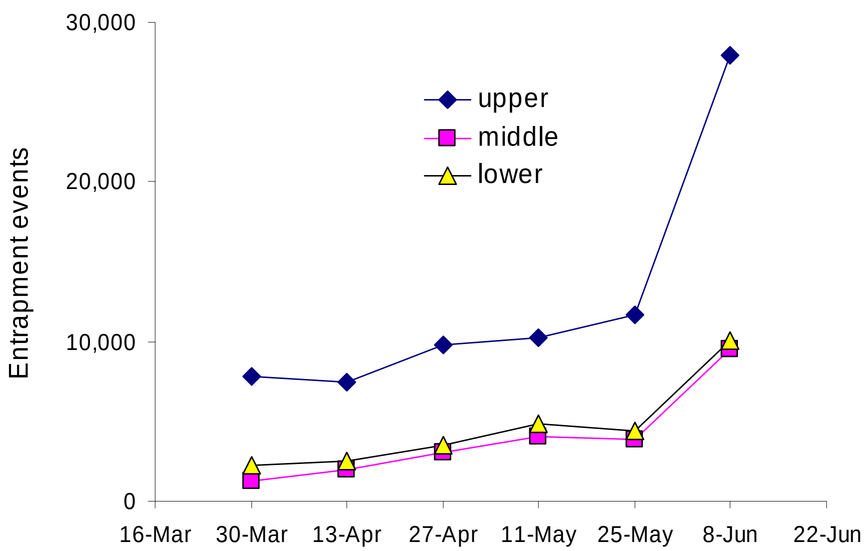


Figure 43. Entrapment events by river segment and sampling period.

Corroboration of Event Estimates with Aerial Entrapment Counts

Aerial counts provided a systematic basis for testing our model-derived estimates of entrapment numbers created by any given flow event. The average numbers of entrapments formed per day were within the general range of the number of entrapments observed during the aerial surveys across the Reach (Table 8 and Table 9). Flights were scheduled at 9:00 am on Saturdays from April 12 through May 24. Flights anticipated decreases in discharge that typically occur on weekends due to decreased power demands. Entrapments counted should be considered minimum estimates as a single aerial flight can only capture a portion of the isolated pools formed during a given event. Many pools drained prior to the scheduled flight times, and others continued to form as the river elevations decreased in downstream areas as illustrated in Figure 44.

Table 9 . Summary of aerial video counts of entrapments in the Hanford Reach, 2003.

Date	Apr 12	Apr 19	Apr 26	May 10	May 17	May 24
Discharge (kcfs)						
Start	115	120	170	145	175	207
End	90	95	123	120	135	120
Change	25	25	47	25	40	87
Total Entrapments	**1,036**	**420**[1]	**2,019**	**753**	**795**	**735**
Priest Rapids Dam to Vernita Bridge						
Franklin shore	28	1	27	9	2	6
Benton shore	111	27	104	37	33	18
Vernita Bar	3	1	28	1	9	0
China Bar	12	2	133	70	76	100
Vernita Bridge to Locke Island						
Franklin shore.	162	51	203	95	53	76
Benton shore	116	37	238	120	146	159
Coyote Rapids	11	0	5	3	0	5
Island #1	2	0	12		0	1
Island #2	0	0	11	1	2	2
Skull Island	4	3	51	13	6	49
Long Island	0	0	3	0	0	0
Vernita (total)	**449**	**122**	**815**	**349**	**327**	**416**
Locke Island to Ferry Landing						
Franklin shore.	30	0	0	0	0	0
Benton shore	6	1	48	6	3	14
Locke Island (Upper)	23	7	49	19	16	8
Locke Island (Lower)	70	15	43	26	12	8
White Bluffs Slough	17	9	35	51	18	16
Ferry Landing to Wooden Power Lines						
Franklin shore.	20	7	55	15	10	20
Benton shore	99	13	45	11	17	19
F-Islands	87	28	32	48	22	44
Hanford Slough	10	5	33	23	32	53
Hanford (total)	**362**	**85**	**340**	**199**	**130**	**182**
Wooden Power Lines to Ringold (canal)						
Franklin shore.	2	7	153	7	0	17
Benton shore	42	19	2	15	7	11
Savage	6	3	159	13	27	0
Ringold to Wooded Island (bottom)						
Franklin shore.	13	8	86	0	45	0
Benton shore	37	32	18	15	22	25
Island at Ringold	18	0	3	20	0	9
Homestead Island	17	21	24	22	40	24
Lower Homestead Island	21	66	360	57	145	17
Fir Island	8	3	3	0	3	0
Wooded Island	36	23	48	34	40	12
Wooded Island to Howard Amon Park						
Franklin shore.	0	23	0	10	9	4
Benton shore	18	8	5	5	0	16
Johnson Island	7	0	0	7	0	0
Refuge Island #1	0	0	0	0	0	0
Refuge Island #2	0	0	0	0	0	0
Refuge Island #3	0	0	0	0	0	2
Nelson Island	0	0	3	0	0	0
Richland (total)	**225**	**213**	**864**	**205**	**338**	**137**

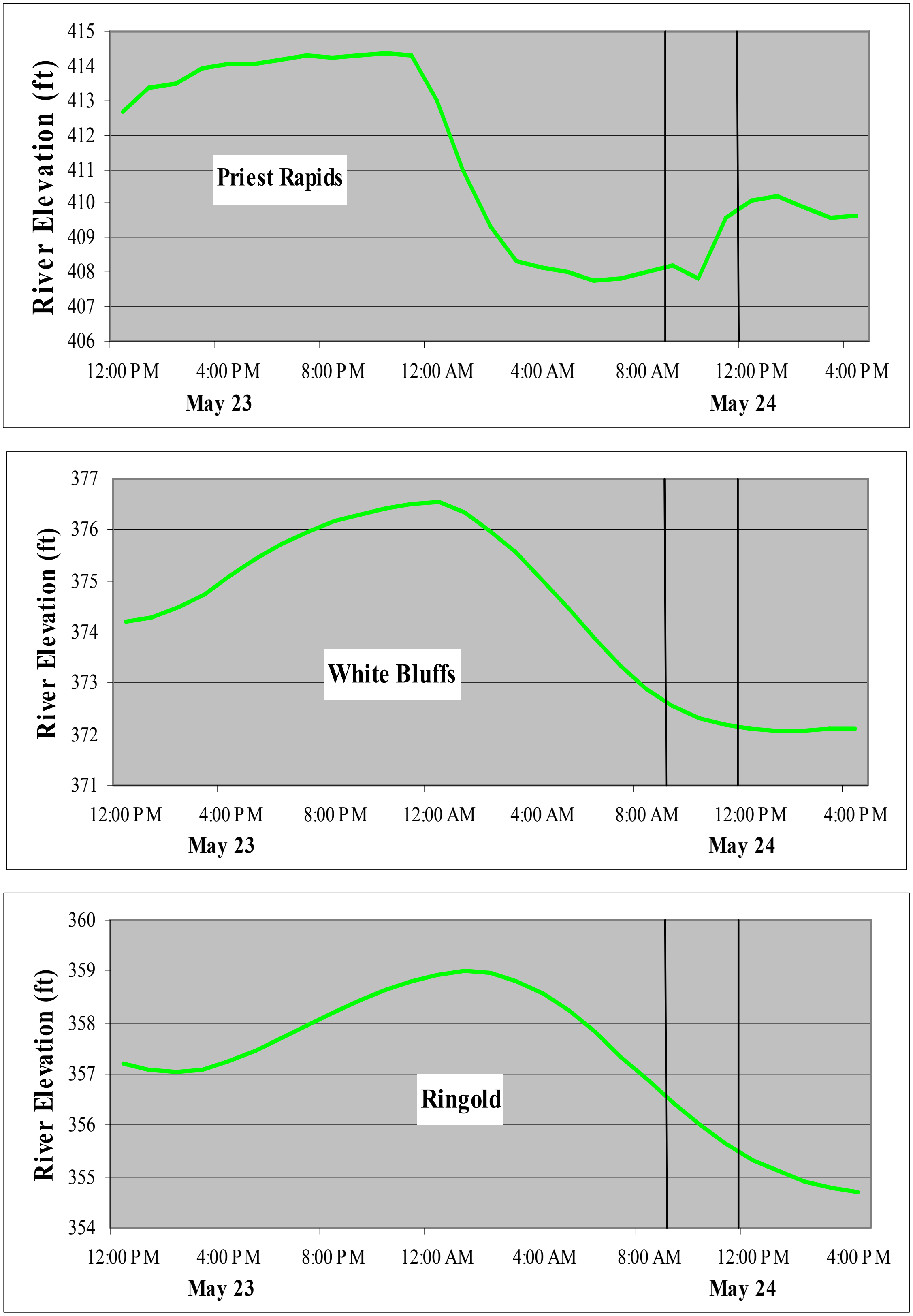


Figure 44. River elevation at three locations, May 23-24, 2003 (flight 0900 – 1200, May 24).

Juvenile Distribution

We collected a total of 42,588 juvenile fall Chinook salmon from nearshore sites using seining over the sampling season. Collections of juvenile Chinook increased the first week of April, and the mean number of Chinook per seine haul was relatively high (>150 fish/haul) throughout the period from April 7 through June 2 (Figure 45). We observed peak relative abundance on May 5, with a mean of 389 juvenile Chinook collected per seine haul. The mean number of Chinook per seine haul declined to less than 50 after June 16.

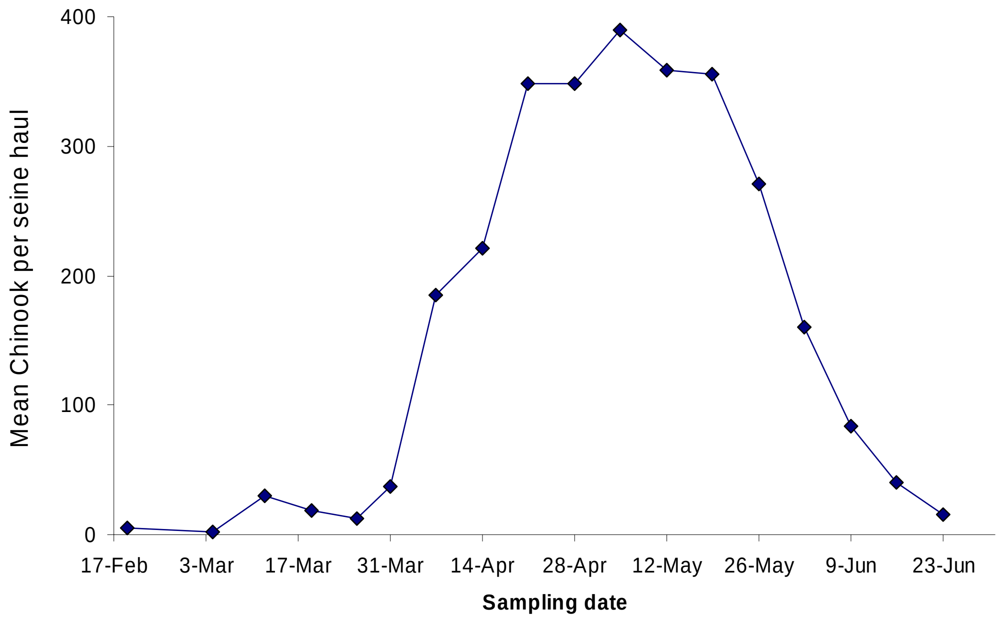


Figure 45. Mean number of Chinook per seine haul across the Hanford Reach in 2003.

Seasonal relative abundance of juveniles in nearshore sampling areas increased with downstream location. Mean catches were 157, 232, and 252 Chinook/seine haul in the upper, middle and lower segments, respectively (Figure 46).

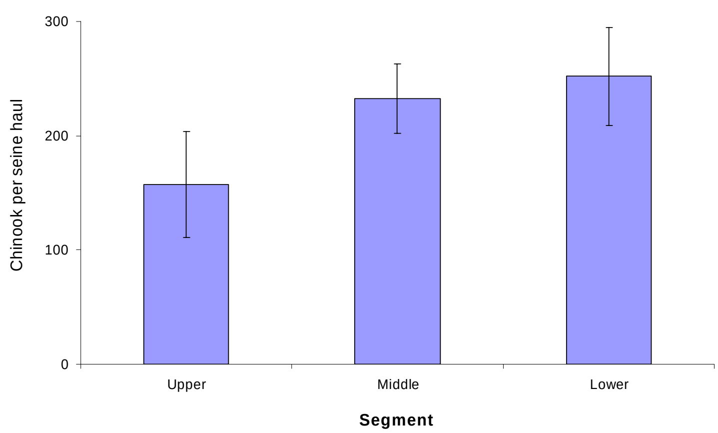


Figure 46. Mean number of Chinook per seine haul (+/- 1 SE) by river segment.

To facilitate eventual comparisons between the seining data and the entrapment fish sampling data, we examined the mean number of Chinook per seine haul across the three river segments and during the six, 14-day entrapment sampling periods (Figure 47). Each river segment displayed a general pattern of increasing and then decreasing mean catches over the sampling periods. The timing of peak relative abundance occurred first in the upper segment (April 27 period) followed in succession by the middle (April 27, May 11 periods) and lower (May 11 period) segments. Although relative abundance was variable among segments earlier in the season, mean catches generally increased with location in a downstream direction within each of the last three sampling periods. These results suggest that while emergent Chinook are present throughout the Reach during the rearing period, gradual movement downstream over the rearing period tends to result in increased relative abundance in downstream areas. Because fall Chinook tend to move downstream during the rearing season, and not upstream, the spatial distribution of spawning locations may also be contributing to the observed patterns in relative abundance.

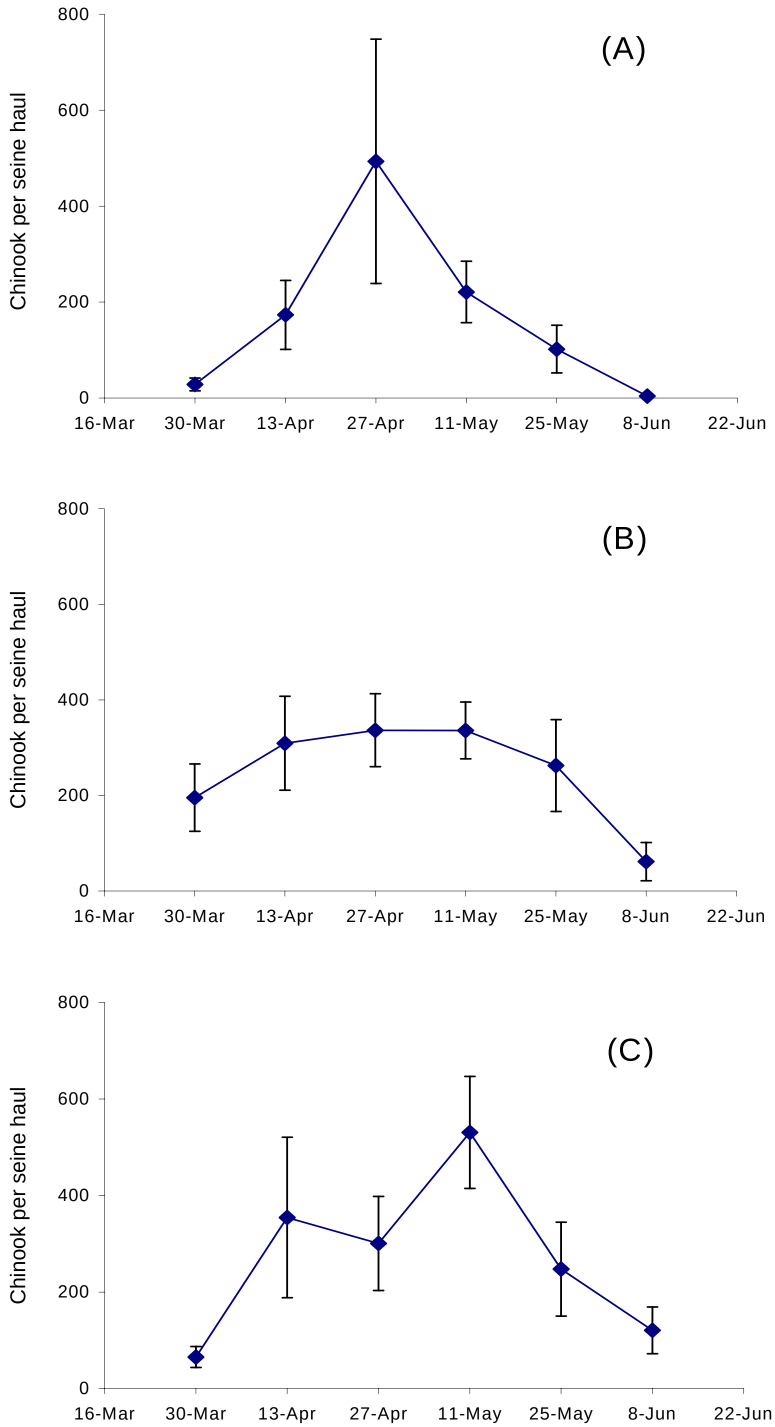


Figure 47. Mean number of Chinook per seine haul (+/- 1 S.E.) in the upper (A), middle (B), and lower (C) river segments over the sampling season. The means are reported by the first day of the 14-day entrapment sampling periods.

Juvenile Chinook mean and maximum fork lengths generally increased with each successive survey over the seine sampling season (Figure 48). We collected Chinook with minimum fork lengths less than 40 mm through the final survey on June 23. However, the proportion of newly emergent fry (<42 mm) in the sample decreased sharply by June 9 (7.2% of sample).

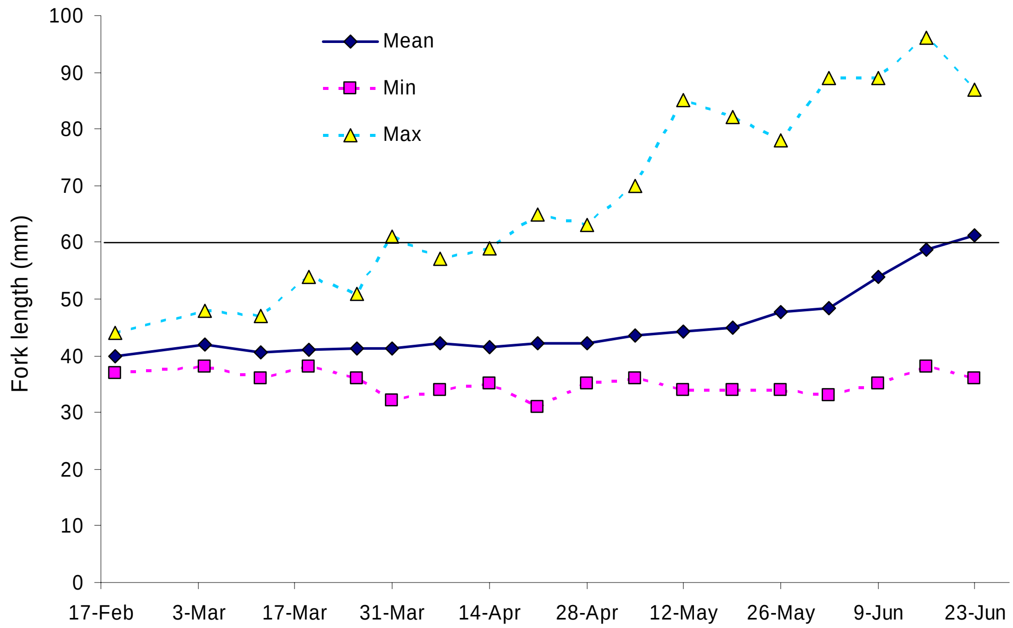


Figure 48. Mean, minimum, and maximum Chinook fork lengths from 2003 seine sampling. The horizontal line at 60 mm denotes the size at which susceptibility to entrapment is thought to decline.

Entrapment Fish Sampling

<u>Fish per Entrapment</u>

We identified a total of 1,257 entrapments formed by reductions in discharge from Priest Rapids Dam during our surveys between April 1 and June 21, 2003. We sampled 935 (74%) of these entrapments for fish numbers and lengths, and for detailed information related to the impacts of physical conditions on juvenile fall Chinook survival. Of the 935 entrapments sampled in detail, 179 contained fish (19.2%) and 164 contained juvenile fall Chinook (17.6%). We observed fish in an additional 46 entrapments (4.9%), but none were recovered during those entrapment seining efforts. We collected a total of 33,177 juvenile Chinook from the entrapments. The median size of the entrapments was 31 m^2 and 95% of the entrapments were less than 750 m^2. The mean depth of entrapments at the time of sampling was 8 cm.

Entrapment sampling began on March 30, well after the February 20 estimated start of emergence in 2003. During the first entrapment sampling period (March 30 to April 12), the mean numbers of Chinook per entrapment were relatively high at 2.6, 204.0, and 13.7 Chinook per entrapment for the upper, middle, and lower segments (Table 10, Figure 49). Because we did not sample entrapments prior to March 30, we could not determine whether, or to what degree, juvenile Chinook were entrapped prior to March 30. The seining data indicated that overall abundance prior to March 30 was not particularly high (Figure 45), however abundance in the middle segment was relatively high (Figure 47), indicating significant entrapment potential existed. The absence of fish per entrapment data for this time period has likely biased our entrapped and mortality estimates low.

For each entrapment sampling period and overall, the middle segment showed the highest values for mean number of Chinook per entrapment (Table 10). The middle segment also showed values of Chinook per entrapment that were approximately one to two orders of magnitude greater than the other segments during the first and third sampling periods. Across the segments, the highest mean number of Chinook per entrapment occurred during the second period in the upper segment, during the third period in the middle segment, and during the first period in the lower segment. The proportions of entrapments with Chinook showed some correspondence with the Chinook per entrapment results (Table 10, Figure 50). Chinook presence in entrapments was high (generally greater than 20%) in the upper and middle segments during the first three periods and during the first period in the lower segment. The highest proportion of entrapments with Chinook occurred during the first period in the middle segment (64%). After May 11, less than 15% of the entrapments contained Chinook in all three segments. However, despite the low frequency of occurrence in entrapments after May 11, the middle segment still showed 5.0 to 12.1 Chinook per entrapment.

Table 10. Mean number of Chinook per entrapment (C/E), the associated standard error of the mean (SE), and the proportion of entrapments with Chinook (PC) by river segment and 14-day entrapment sampling period.

	Upper segment			Middle segment			Lower segment		
Sampling period	C/E	SE	PC	C/E	SE	PC	C/E	SE	PC
Mar-30 to Apr-12	2.6	1.2	0.28	204.0	123.0	0.64	13.7	6.6	0.34
Apr-13 to Apr-26	9.2	8.3	0.19	12.5	6.2	0.33	0.8	0.7	0.09
Apr-27 to May-10	8.2	3.1	0.41	312.5	272.6	0.32	4.5	3.1	0.14
May-11 to May-24	0.1	0.1	0.09	12.1	8.4	0.14	0.4	0.4	0.01
May-25 to Jun-7	0.0	0.0	0.00	11.8	8.4	0.09	0.2	0.2	0.04
Jun-8 to Jun-21	0.0	0.0	0.00	5.0	4.9	0.03	0.6	0.6	0.05
Overall	3.2	1.2	0.20	68.8	36.4	0.22	4.7	2.0	0.10

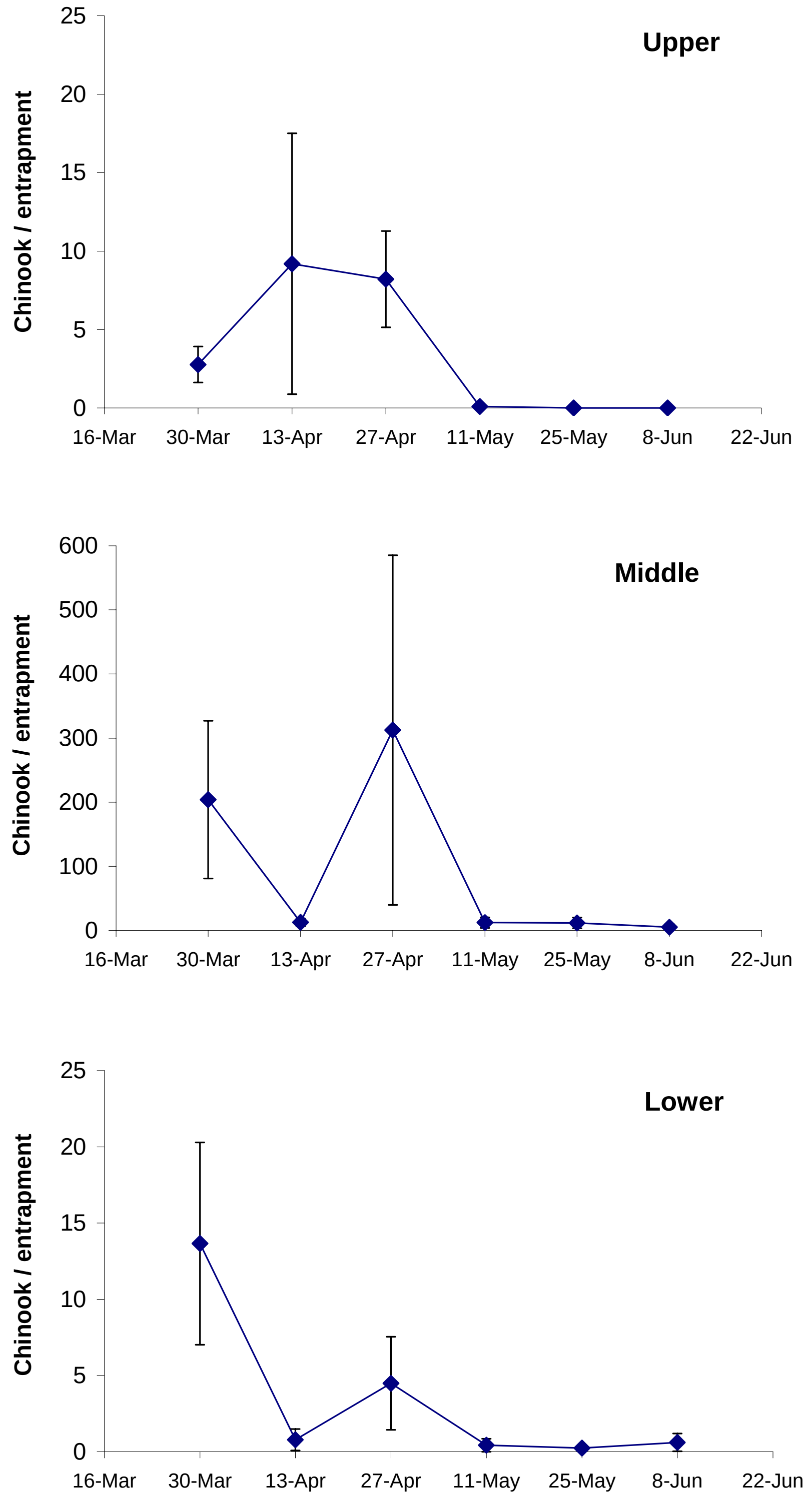


Figure 49. Mean number of Chinook per entrapment (+/- 1 S.E.) in the upper, middle, and lower river segments over the sampling season. The means are reported by the first day of the 14-day entrapment sampling periods. Note the scale for the middle segment is up to 600.

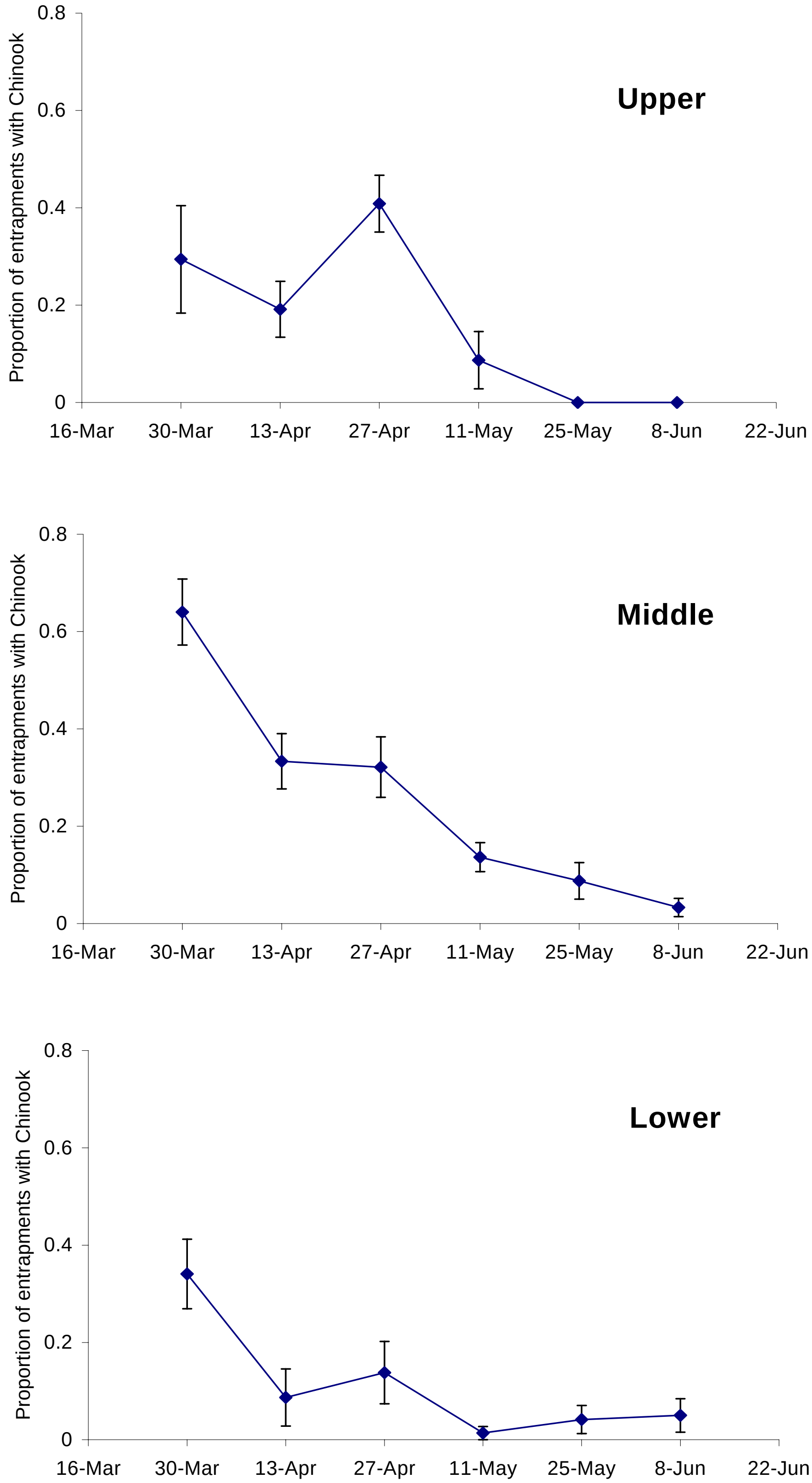


Figure 50. The proportion of entrapments with Chinook (+/- 1 S.E.) in the upper, middle, and lower river segments over the sampling season. The proportions are reported by the first day of the 14-day entrapment sampling periods.

Factors Related to Entrapment

We performed several analyses to investigate potential factors that could have contributed to the mean abundance of Chinook in entrapments. It has been hypothesized that susceptibility of juvenile fall Chinook to entrapment and stranding decreases as fork length reaches 60 mm (Nugent et al. 2002c). In past studies, occurrence of Chinook greater than 60 mm in entrapment samples has been rare (Nugent et al. 2001, 2002a, 2002b, 2002c). However, mean fork length of Chinook sampled along nearshore areas in the Reach in 2003 did not exceed 60 mm until June 23 (Figure 48). Although the number of fish captured in entrapments was relatively low during mid-June, the seining data suggest that nearly half of the near-shore population was still at or below the size thought to be vulnerable to entrapment during this time.

To evaluate the degree of correspondence between the incidence of entrapment and general fish distribution patterns throughout the Reach, we plotted the mean Chinook per seine haul versus the mean Chinook per entrapment for the six, 14-day sampling periods in the three river segments (Figure 51). We found little correspondence between the two. The high numbers of fish per entrapment in the middle segment corresponded to intermediate relative abundances based on the seining data. When the seining data showed around 500 Chinook/haul, the entrapment data showed 0.4 to 8.2 Chinook/entrapment. Even when we excluded the observations with >200 Chinook/entrapment, there was not a significant relationship between the entrapment data and the seining data (p = 0.66).

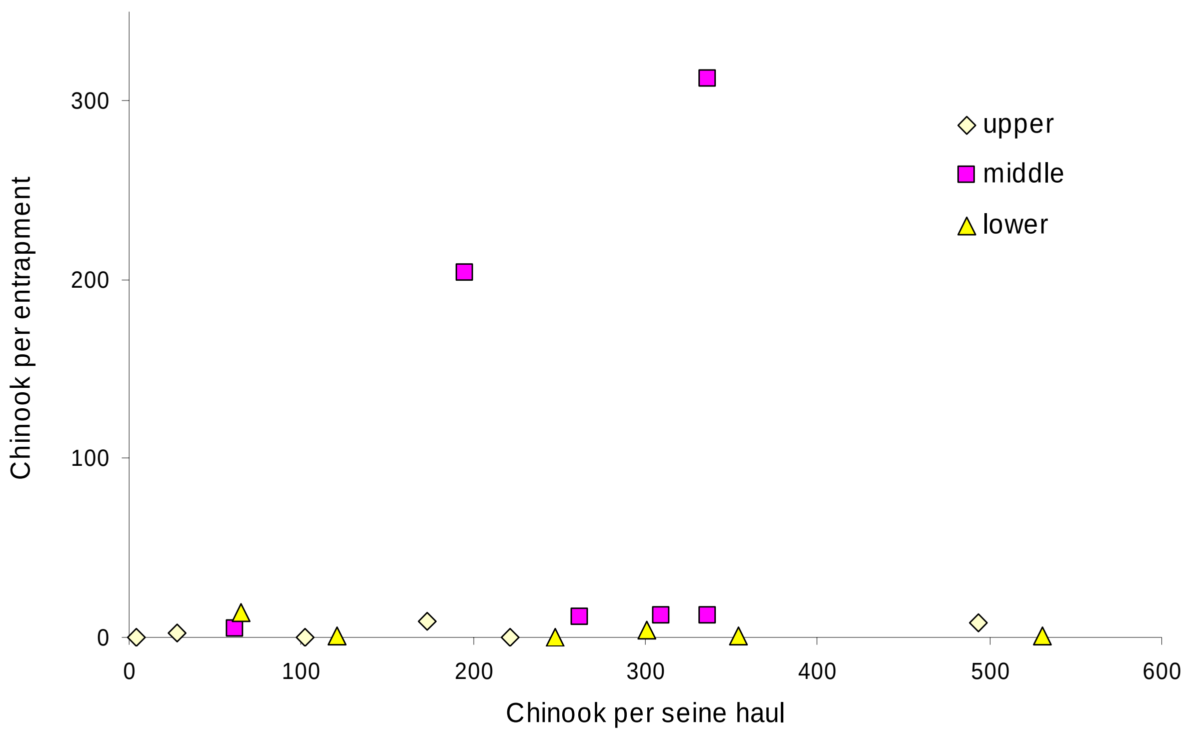


Figure 51. Chinook per entrapment versus Chinook per seine haul for the upper, middle, and lower segments during the six, 14-day sampling periods.

We hypothesized that Chinook/entrapment would decrease as flow increased because it has been suggested that flow fluctuations at low flows are likely to affect more juvenile Chinook in the Hanford Reach than fluctuations at high flows. To evaluate this hypothesis, we calculated the weekly mean number of Chinook per entrapment and the weekly average streamflow (discharge from Priest Rapids Dam) and conducted a regression analysis (Figure 52). We found no significant relationship between the weekly mean number of Chinook per entrapment and the weekly average flow ($p = 0.18$). We also conducted a regression analysis on the effects of flow variability, expressed as the weekly coefficient of variation (CV) of hourly flows, and the weekly mean number of Chinook per entrapment, and found no significant relationship ($p = 0.27$) (Figure 53).

In the analyses above, we attempted to characterize potential relationships between streamflow variables and Chinook per entrapment Reach-wide. To account for the spatial (upper, middle, and lower segments) and temporal (the six, 14-day sampling periods) effects that may have obscured the effects of streamflow levels on Chinook per entrapment, we conducted an analysis of covariance (ANCOVA) to determine whether Chinook per entrapment was related to the estimated 10 kcfs flow band within which it occurred. The ANCOVA results showed that Chinook per entrapment was not significantly related to entrapment flow band levels ($p = 0.79$) after accounting for spatial and temporal effects.

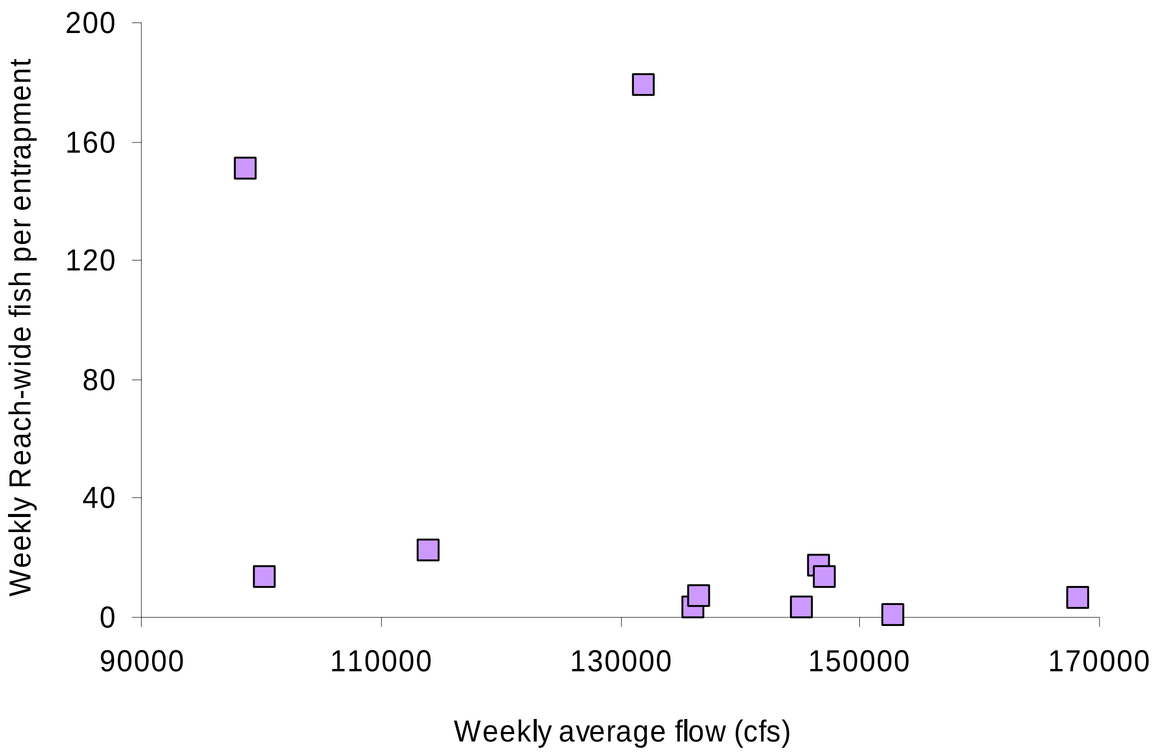


Figure 52. Weekly Reach-wide mean numbers of Chinook per entrapment versus weekly average flow (cfs).

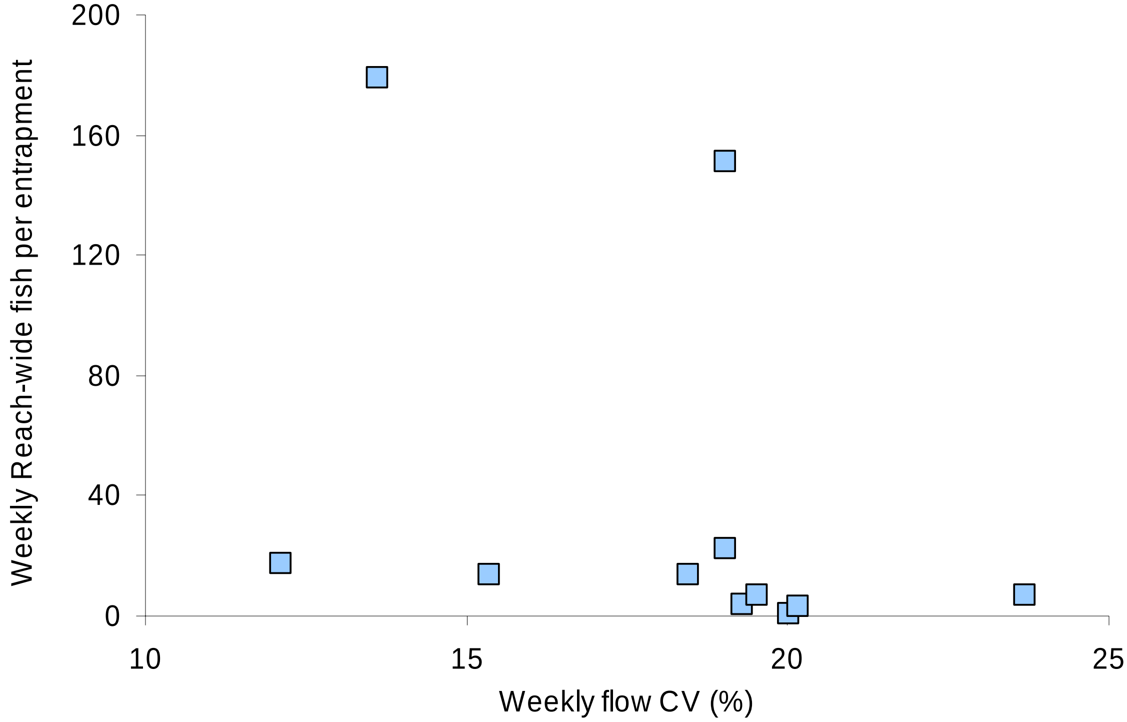


Figure 53. Weekly Reach-wide mean numbers of Chinook per entrapment versus weekly flow coefficient of variation.

We conducted additional analyses that compared the size of entrapments to flow levels, to the probability of containing Chinook, and to the number of Chinook in individual entrapments. Using linear regression on the entrapment fish sampling data, we found that entrapment size was not related to flow levels ($p = 0.28$). That is, the size of individual entrapments is similar across flow levels. Using logistic regression, we examined the relationship between entrapment size and the probability of an entrapment containing Chinook. We found that entrapment size was positively associated with the probability of containing juvenile Chinook ($p = 0.04$). However, across the range of typical entrapment sizes, changes in the probability of containing Chinook were not large (Figure 54). For example, we estimated that a 30 m^2 entrapment has an 18% chance of containing Chinook, while a 750 m^2 entrapment has a 20% chance of containing Chinook. Using linear regression, we found that statistically, entrapment size was positively associated with the number of Chinook in entrapments ($p < 0.05$). However, across the range of typical entrapment sizes, the difference in Chinook abundance was not large, and was of questionable biological significance. Across the season, a 30 m^2 entrapment was predicted to contain 1.3 Chinook, while a 750 m^2 entrapment was predicted to contain 1.5 Chinook.

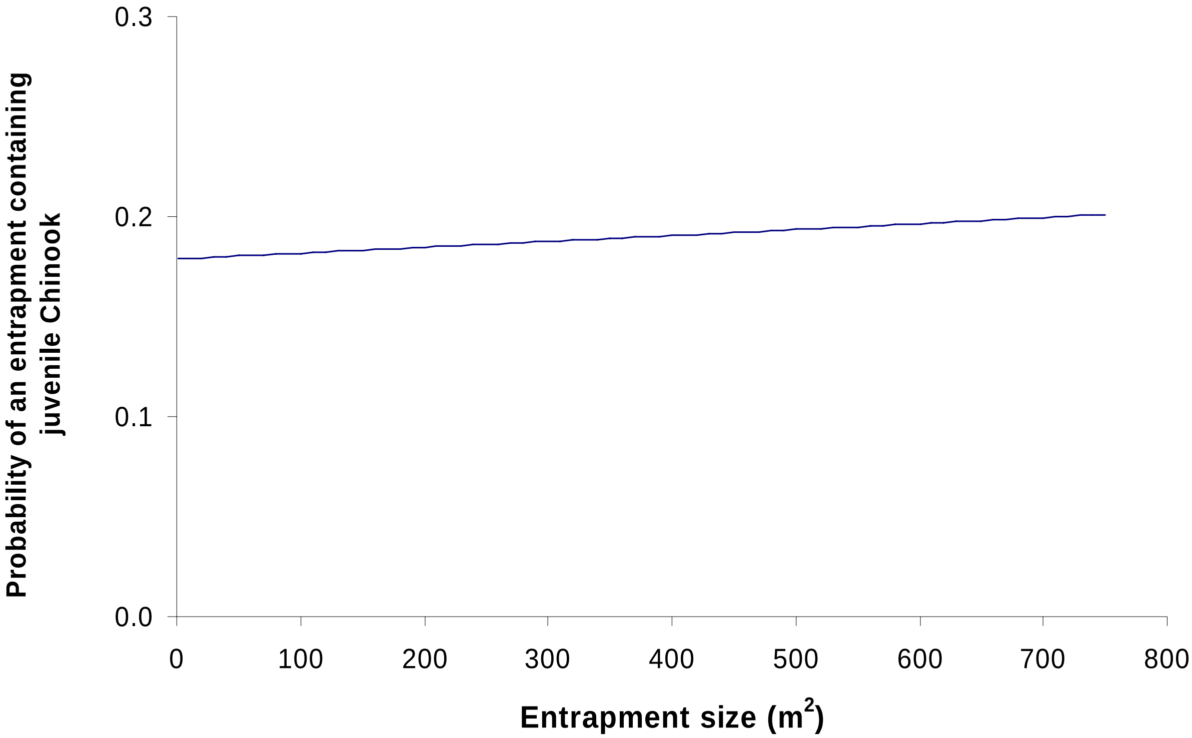


Figure 54. Logistic regression fit of the probability that an entrapment will contain juvenile Chinook as a function of entrapment size. Ninety-five percent of the entrapments were less than 750 m^2.

Entrapment Fates

Physical attributes were measured at all 935 entrapments for detailed information that could be used to determine the fate of the entrapments and to relate those attributes to the likelihood that entrapped juvenile fall Chinook would have survived. Our data indicated the following outcomes for these entrapments:

- Re-flooded with river water, no mortality (20.9%);
- Large entrapments, no mortality (0.4%);
- Drained through the substrate, mortality (46.3%);
- Reached lethal water temperature, mortality (32.4%).

Overall, we estimated an entrapment lethality rate of 78.7% in 2003 (Equation 5 in Methods, *Entrapment Fish Sampling*). Across the sampling periods, entrapment lethality was lower in the upper segment (71.7%) than in the middle (80.7%) and lower (81.4%) segments (Table 11). Within each segment, entrapment lethality was generally lower during the last two sampling periods compared to earlier periods. These results are consistent with observations in the Hanford Reach from studies in 2000, when 85.2% of the entrapments became lethal (Nugent et al. 2002b).

Table 11. Percent entrapment lethality (*L*) by river segment and sampling period.

	Upper segment	Middle segment	Lower segment
Sampling period	*L*	*L*	*L*
Mar-30 to Apr-12	88.9	90.0	79.5
Apr-13 to Apr-26	38.3	81.4	95.7
Apr-27 to May-10	80.3	82.1	79.3
May-11 to May-24	91.3	87.1	86.3
May-25 to Jun-7	76.9	73.6	73.9
Jun-8 to Jun-21	71.4	69.2	76.3
Overall	71.7	80.7	81.4

Estimating Numbers of Chinook Entrapped and Entrapment Mortalities

Reach-wide Estimate - Stratified Approach

Combining the results from the entrapment event histories and the numbers of Chinook per entrapment by river segment and sampling period, our estimate of the number of Chinook entrapped in 2003 is 1,602,891 with a 95% bootstrapped confidence interval of 504,177 to 3,513,510 (Table 12). Incorporating the segment- and sampling period-specific entrapment lethality rates (Table 11) resulted in an estimate of 1,297,104 Chinook entrapment mortalities with a 95% bootstrapped confidence interval of 395,387 to 2,913,791. Spatially, the middle segment accounted for 86% of the total number of Chinook entrapped (Table 12). Temporally, the sampling period of April 27 – May 10 accounted for 65% of the entrapped fish. Substantial numbers of Chinook were also entrapped during the March 30 – April 12 period, with the majority of those fish coming from the middle segment.

We compared our Chinook ***entrapment*** mortality estimate for 2003, to the ***entrapment and stranding*** mortality estimate reported in McMichael et al. (2003) for the entire Hanford Reach (Figure 55). Our estimate, which only accounts for mortality of entrapped fish (1,297,104 Chinook), was approximately 2.5 times the mortality estimate for entrapped and stranded fish (527,922 Chinook) reported in McMichael et al. (2003).

Reduced-area Estimate

To facilitate comparisons with the index area estimates based on the sampling protocol reported in McMichael et al. (2003), we estimated the number of entrapment events and number of Chinook per entrapment by sampling period for that portion of the Reach between rkm 584.5 and rkm 600.2. Our estimate of the number of Chinook entrapped in this reduced-area is 1,061,180 Chinook with a 95% bootstrapped confidence interval of 201,603 to 2,567,240. The McMichael et al. (2003) estimate of fish "at-risk" of stranding and entrapment mortality in this reduced-area was 164,643 Chinook. Our estimate for entrapment only, is 6.4 times the McMichael et al. (2003) estimate for Chinook "at-risk" of entrapment or stranding in the reduced area.

Applying the reduced-area entrapment lethality rates resulted in an estimate of 875,412 Chinook entrapment mortalities with a 95% bootstrapped confidence interval of 161,289 to 2,178,687. A combined stranding and entrapment mortality estimate of 154,853 Chinook for 2003 was presented for this reduced-area in McMichael et al. (2003). Our reduced-area mortality estimate for entrapment-only in 2003 was over 5.6 times the stranding and entrapment mortality estimate generated by McMichael et al. (2003) (Figure 56). The difference between the two estimates was statistically significant ($p < 0.05$, two-sample Z-test with unequal variances).

To place the estimates from our entrapment-based sampling approach into context with previous impact estimates based on an area-based sampling approach, we compared our estimate of the number of Chinook entrapped in the reduced-area to previous estimates of the number of Chinook "at-risk" of stranding or entrapment mortality in the reduced-area (McMichael et al. 2003). Historical estimates of impacts with confidence intervals are only available for the reduced-area. Figure 57 displays the difference between the FWS estimate of entrapment-only in the reduced-area in 2003 in comparison to GPUD estimates of fish "at-risk" of stranding and entrapment in the reduced area during 1999-2003.

Table 12. Estimates of the number of Chinook entrapped by river segment and sampling period, the total number of Chinook entrapped by river segment and sampling period, and the total number of Chinook entrapped in 2003.

	River segment			
Sampling period	Upper	Middle	Lower	Total
Mar-30 to Apr-12	20,434	256,892	30,883	308,210
Apr-13 to Apr-26	68,686	24,166	1,941	94,792
Apr-27 to May-10	80,118	952,387	15,511	1,048,016
May-11 to May-24	893	48,421	2,066	51,380
May-25 to Jun-7	-	45,557	1,060	46,618
Jun-8 to Jun-21	-	47,764	6,111	53,875
Total	170,132	1,375,188	57,571	1,602,891

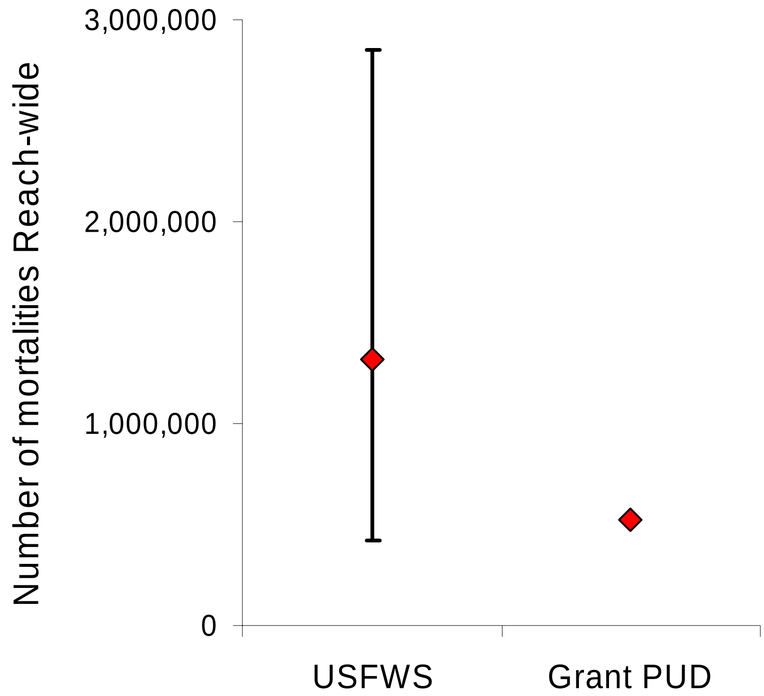


Figure 55. Estimates of the number of mortalities due to entrapment with a 95% confidence interval (USFWS estimate) and the number of mortalities due to stranding or entrapment (Grant PUD estimate) for the entire Hanford Reach in 2003.

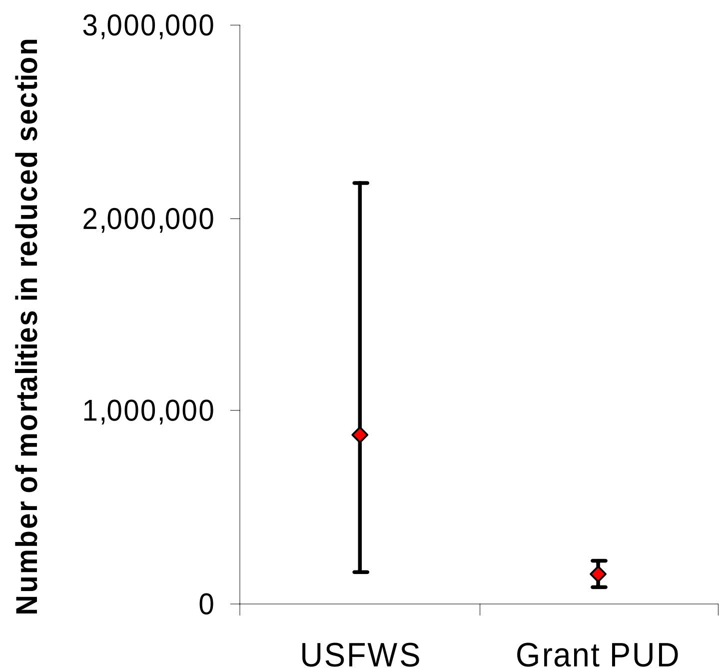


Figure 56. Estimates of the number of mortalities due to entrapment (USFWS estimate) and the number of mortalities due to stranding plus entrapment (Grant PUD estimate) with their respective 95% confidence intervals for the reduced-area section of the Hanford Reach in 2003.

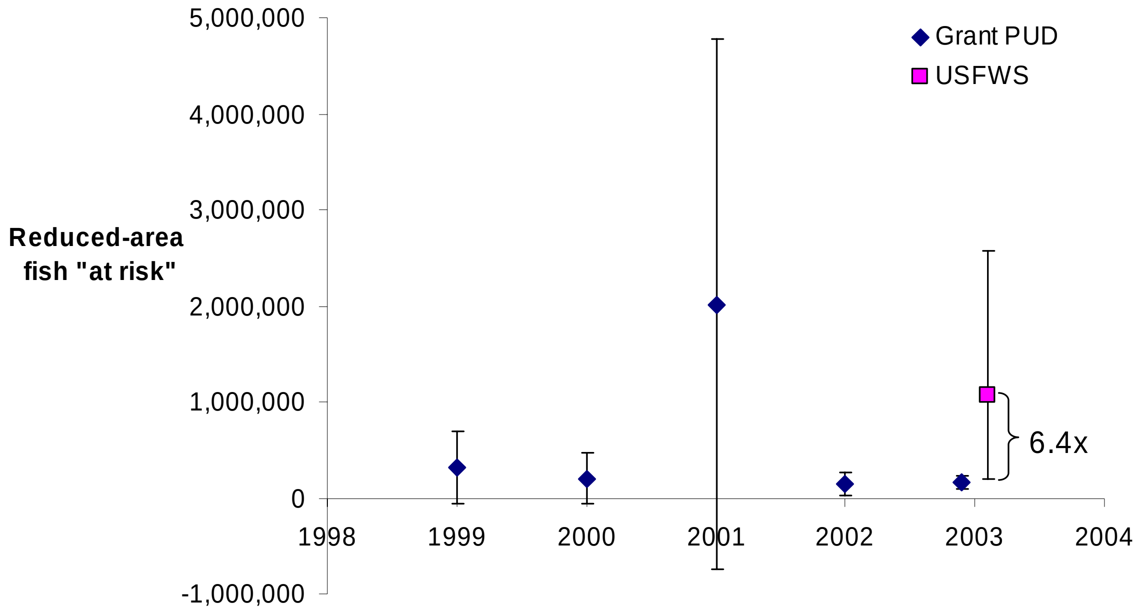


Figure 57. USFWS estimate of the number of Chinook entrapped in the reduced-area in 2003 versus Grant PUD estimates of the number of Chinook "at-risk" of entrapment and stranding mortality during 1999-2003 (data from McMichael et al. 2003). All estimates are presented with their 95% confidence intervals.

Sample Size Effects

We were also interested in evaluating the effects of increasing the sampling effort for fish per entrapment surveys to increase the precision of our impact estimates. We simulated sampling entrapments at rates of half, two-, three-, four-, and five-times the original effort of 935 entrapments over the season (Figure 58). While increasing the sampling effort would help reduce the confidence interval widths, the large amount of natural variability observed in the numbers of Chinook per entrapment may limit the degree that enhanced sampling can substantially improve precision.

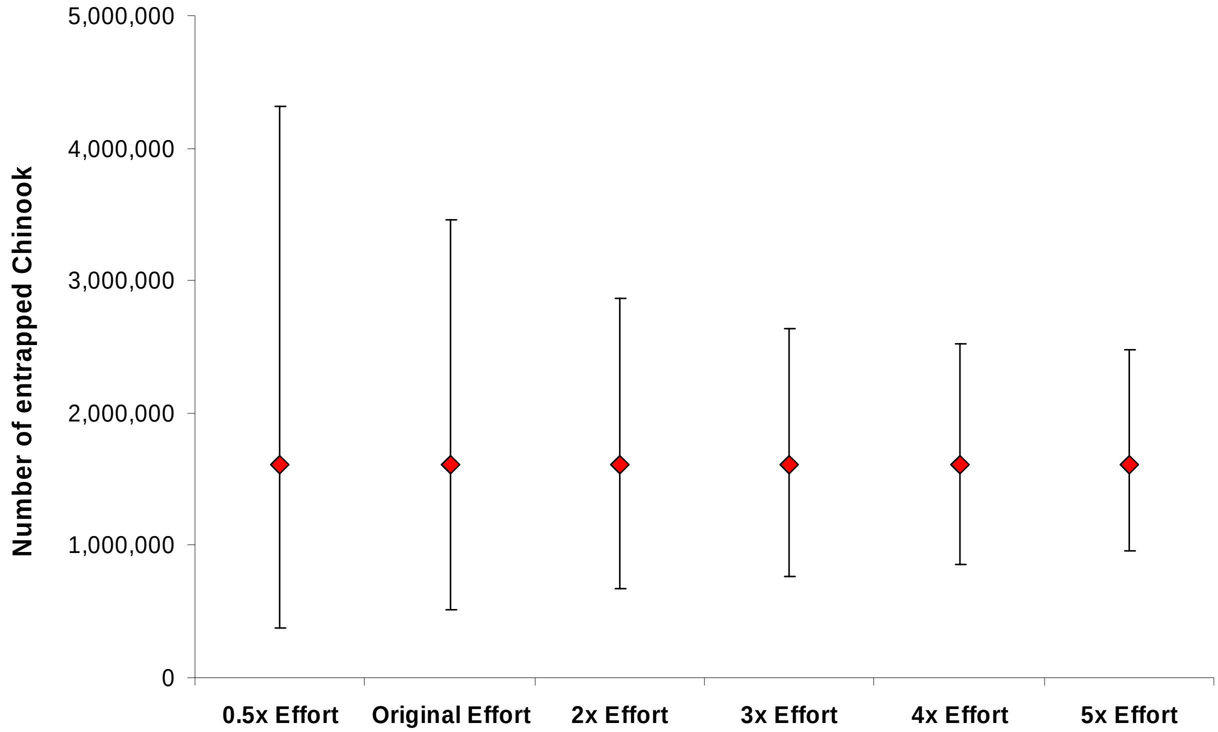


Figure 58. Evaluation of the effects of reduced and enhanced sampling effort on the precision of the entrapped Chinook estimate. The vertical lines represent the boundaries of the 95% bootstrapped confidence intervals.

Population-level Impacts

The mean smolt-to-adult (SAR) survival rate of the Priest Rapids Hatchery stock was 0.98% and the mean SAR of the Hanford Reach wild stock was 0.32%. Applying these SARs to the estimated 1,297,104 fall Chinook mortalities in 2003 results in estimates of 12,712 adults (using the hatchery SAR) and 4,151 adults (using the Hanford wild SAR) that would have been available to ocean fisheries, in-river fisheries, and escapement back to the Hanford Reach had the entrapment mortality not occurred.

It should be noted that these SARs were developed during time periods when ocean survival conditions were not favorable for Columbia River fall Chinook. Ocean survival conditions have improved considerably since 1999, which may result in much higher SARs for the recent fall Chinook brood years, including those that outmigrated in 2003. The Hanford wild SAR has historically ranged up to 0.63%, which would have translated into 8,172 adults, but recent SARs may be even higher than this survival rate estimate.

The second method we used to evaluate the population level impacts was to examine fry production using demographic modeling. The most critical parameter influencing fry production is the egg-to-fry survival rate. We conducted a literature review of published estimates of egg-to-fry survival rates to help quantify and bound likely values for this parameter. Bradford (1995) reviewed salmon survival rates and reported ocean-type Chinook geometric mean egg-to-fry survival rates of 12% from Fall Creek, California (Wales and Coots 1954) and 6% from the Big Qualicum River, British Columbia (Fraser et al. 1983). Pahlke (1995) reported geometric mean egg-to-fry survival rates of 2.9% and 2.8% for the Unuk and Chickamin Rivers, Alaska based on five years of data. Seiler et al. (2002) reported geometric mean survivals of 9.7% in the Skagit River based on eleven years of data. Healey (1980) reported that egg-to-fry survivals were 15-20% in the Nanaimo River, British Columbia based on two years of data. Each of these studies determined survival by dividing the estimated fry population by the estimated number of eggs deposited over a number of years. The geometric mean of the estimates from these six populations is 10% with a range of 2.8-24%. McMichael et al. (2003) calculated an arithmetic mean survival rate of 27.8% based on a redd-capping study of seven redds in Wanapum tailrace in one year. We calculated a geometric mean survival rate of 24.2% for the same data set. Table 13 presents recent estimates of fry production based on WDFW data on escapement, proportion female, egg retention, average fecundity, and our literature-derived values for egg-to-fry survival.

Table 13. Recent estimates of fry production based on WDFW data on escapement, proportion female, egg retention, average fecundity, and our literature-derived values for egg-to-fry survival.

Emergence Year	Adults	Proportion Female	Fecundity	Egg Retention	Deposited Eggs	Egg-to-fry Survival 2.8%	10%	24%
2005	79,464	0.45	4,224	0.005	151,291,878	4,236,173	15,129,188	36,310,051
2004	89,312	0.51	4,422	0.005	200,018,155	5,600,508	20,001,815	48,004,357
2003	69,117	0.40	4,003	0.005	111,217,958	3,114,103	11,121,796	26,692,310
2002	44,140	0.37	4,418	0.005	70,822,946	1,983,042	7,082,295	16,997,507
2001	36,027	0.54	4,794	0.005	92,798,930	2,598,370	9,279,893	22,271,743
2000	27,012	0.46	4,371	0.005	54,040,388	1,513,131	5,404,039	12,969,693
1999	29,410	0.46	4,200	0.005	56,536,019	1,583,009	5,653,602	13,568,645
1998	34,007	0.46	4,420	0.005	68,797,317	1,926,325	6,879,732	16,511,356

Using the geometric mean egg-to-fry survival (10%) of the published studies that we reviewed, we estimate that 12% of the fry production was killed in 2003 due to entrapment alone (Figure 59). Using the published ranges of egg-to-fry survival rates (2.8-24%), the 2003 mortalities due to entrapment could have constituted 5% to 42% of the fry production in the Reach.

We also evaluated Grant PUD's estimate of the number of Chinook that were killed in 2001 (6,864,851 Chinook) in context with the estimated fry production for that year (Figure 60). Using the geometric mean published value for egg-to-fry survival rate (10%) results in an estimate of 74% of the fry production may have been killed due to stranding and entrapment in 2001. Using the published ranges of egg-to-fry survival rates (2.8-24%), 31% to over 90% of the fry production may have been lost due to stranding and entrapment in 2001.

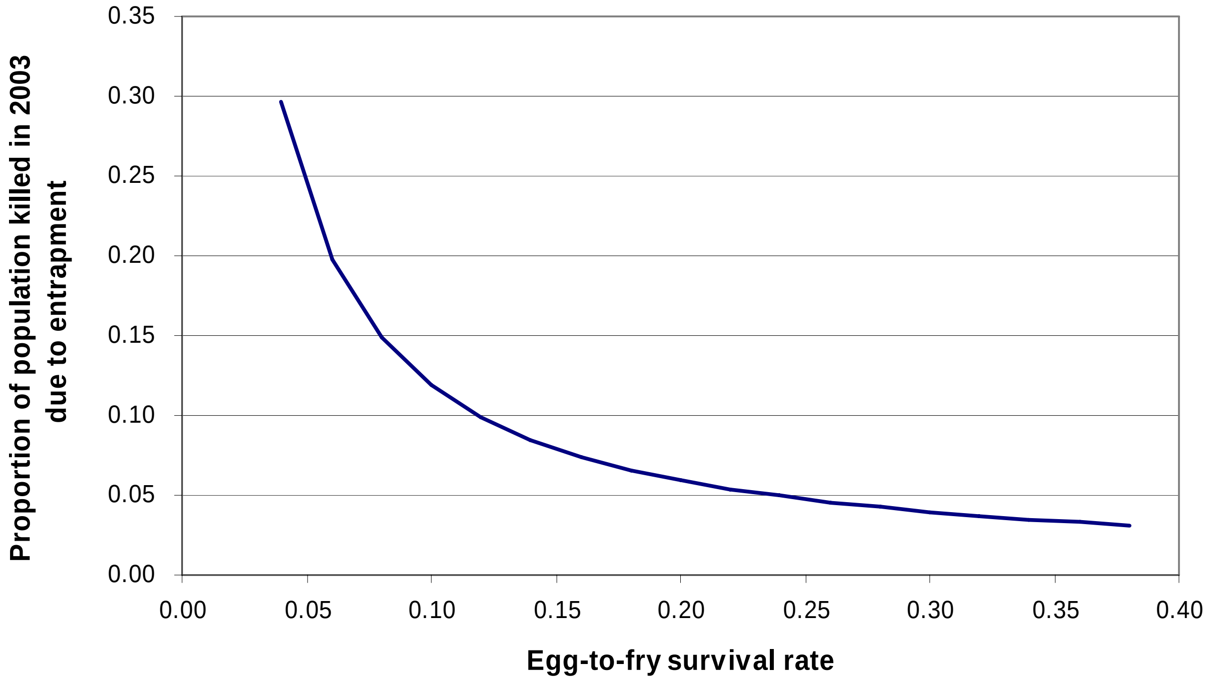


Figure 59. Estimated proportion of the fry population that was killed in 2003 due to entrapment as a function of egg-to-fry survival rates.

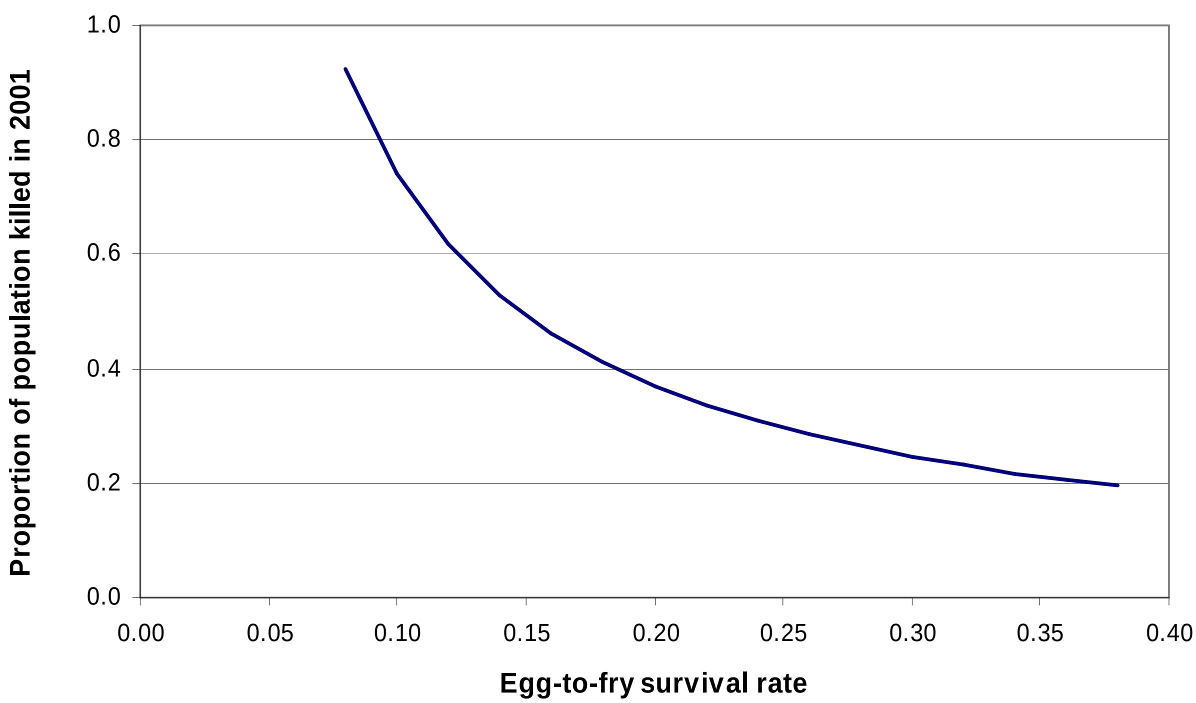


Figure 60. Estimated proportion of the fry population that was killed in 2001 as a function of egg-to-fry survival rates.

Our third method for determining population-level impacts of juvenile mortality in the Reach was to examine how various percent-reductions in the fry population would translate into reduced ocean and in-river catches. Based on the results of our demographic modeling above, percent-reductions in the fry population due to stranding and entrapment may range from 5% to 50% or more. Therefore we evaluated how 5%, 25%, and 50% reductions in fry production from the Hanford Reach would affect catch of all Chinook in ocean and in-river fisheries. We estimated that catches in ocean and in-river fisheries would decline by 9,000 to 170,000 Chinook when stranding and entrapment mortality reduced fry production in the Reach by the specified amounts (Table 14).

Table 14. The 2002-2004 return year average number of Chinook that would be lost as catch in ocean and in-river fisheries corresponding to various levels of percent-reduction in the fry population due to stranding and entrapment in the Hanford Reach.

Percent Reduction in Fry Population	Catch Reduction
5%	9,073
25%	42,402
50%	169,754

Juvenile Behavior

Underwater video cameras for collecting diel behavioral information on juvenile fall Chinook salmon were deployed in depths ranging from 0.1 to 0.6 m and in velocities ranging from 3 to 20 cm/s. Mean fork length of juvenile Chinook during the study period was 44 mm (range = 39-50 mm) and mean weight was 0.7 g (range = 0.3-1.2 g). A total of 86 hours of video data was collected in 2003. Review of 100 daytime and 100 nighttime video clips selected randomly resulted in observations of 172 and 96 fish, respectively. However, 29 of the fish observed during nighttime hours were positioned very close to the overhead infrared lights located within camera arrays. These fish were excluded from the data analyses to remove potential bias associated with the presence of this light source.

We observed significant diel differences in activity levels and water column positions (Table 15). During daytime hours, 90 % of the fish observed were either feeding or swimming and 87% were located in the middle or upper portion of the water column. However, during nighttime hours 61% of the fish were inactive (holding) and located in the lower portion of the water column. Fish did not show any preferential use of nearshore (within 1 m) versus offshore (1-2 m) habitats between the daytime and nighttime (Table 15). We observed fish moving both upstream and downstream during the daytime and nighttime. During the daytime, the percentage of fish moving upstream and downstream was nearly equal (53% and 47%, respectively), but at night fish generally moved upstream (73%). The diel comparison of movement direction was not significant, but only by a slim margin (Table 15). During the daytime, 41% of the fish observed were alone, 37% were in groups of 2-5 fish, and 22% were in groups of 6 or more individuals. Fish observed at night were always alone.

Table 15. Results of *t*-tests comparing habit use variables during day and nighttime hours for juvenile fall Chinook salmon in the Hanford Reach of the Columbia River, 2003. Categorical values were assigned for proximity to shore (1 = nearshore, 2 = offshore), movement direction (1 = upstream, 2 = downstream), activity level (1 = feeding, 2 = swimming, 3 = holding) and position in water column (1 = upper 1/3, 2 = middle 1/3, 3 = lower 1/3).

Variable	Mean day	Mean night	*t*-value	df	*P*
Activity level	2.03	2.61	-9.11	237	<.0001
Position in water column	2.02	2.60	-7.23	237	<.0001
Proximity to shore	1.46	1.43	0.37	237	0.71
Movement direction	1.47	1.27	1.95	178	0.052

Effects of Alternative Flow Patterns

Simulations clearly illustrated the relationship between the frequency and magnitude of fluctuations and the anticipated impact, and they also demonstrated the sensitivity of entrapment impacts to flow fluctuations (Figure 61, Table 16–Table 18). Our evaluation considered eight alternative hydro operations: five or ten fluctuations per week with fluctuation magnitudes of 10, 20, 30, or 40 kcfs. The fluctuation scenarios were built upon the observed weekly average flows for the period. Fluctuation magnitudes of 10 kcfs resulted in estimates of 82,739 and 99,373 entrapped Chinook for the five and ten fluctuations per week scenarios. Scenarios with 30 and 40 kcfs fluctuations resulted in the greatest predicted numbers of entrapped Chinook. As fluctuation magnitudes increased from 10 kcfs to 20 kcfs, predicted impacts increased by a factor of 4.6 for the five fluctuation scenario, and by a factor of 7.3 for the 10 fluctuation scenario. As fluctuation magnitudes increased from 20 kcfs to 30 kcfs, predicted impacts doubled for both the five and 10 fluctuations per week scenarios. As fluctuation magnitudes increased from 30 kcfs to 40 kcfs, predicted impacts increased by about 50% for both the five and 10 fluctuations per week scenarios.

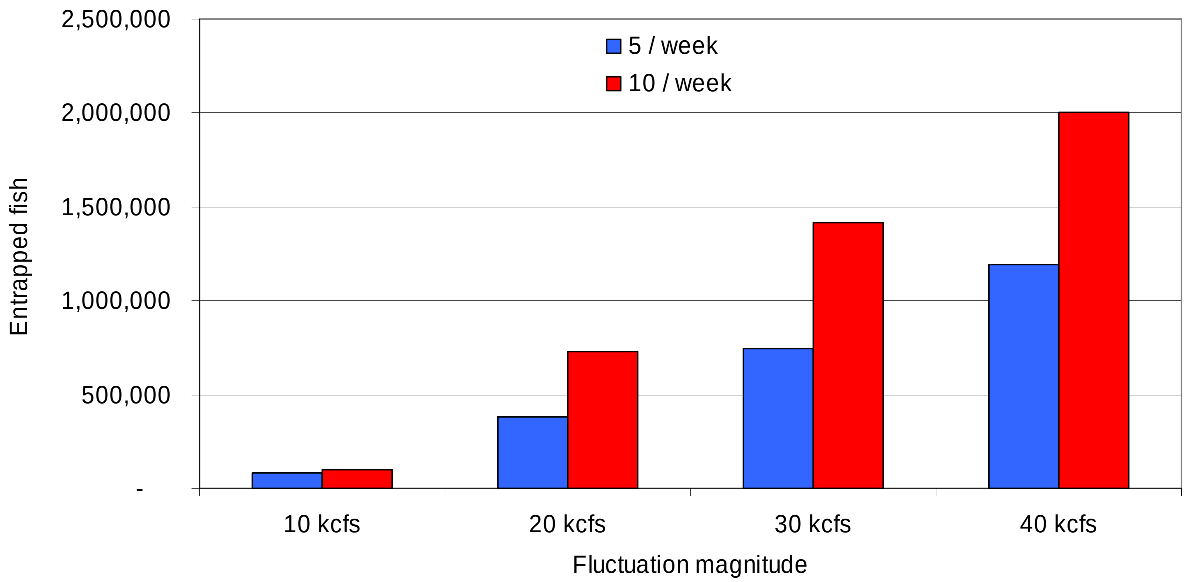


Figure 61. Simulated number of Chinook entrapped for alternative hydro operations of 5 or 10 fluctuations per week and fluctuation magnitudes of 10, 20, 30, or 40 kcfs.

Table 16. Simulated number of Chinook entrapped associated with alternative hydro operations of 5 or 10 fluctuations per week and fluctuation magnitudes of 10, 20, 30, or 40 kcfs.

	Fluctuation magnitude			
Fluctuations	10 kcfs	20 kcfs	30 kcfs	40 kcfs
5 / week	82,365	383,416	748,359	1,192,709
10 / week	98,742	724,427	1,417,154	2,006,750

Table 17. Numbers of entrapments created Reach-wide associated with alternative hydro operations of 5 or 10 fluctuations per week and fluctuation magnitudes of 10, 20, 30, or 40 kcfs.

	Fluctuation magnitude			
Fluctuations	10 kcfs	20 kcfs	30 kcfs	40 kcfs
5 / week	7,444	25,660	49,484	74,133
10 / week	13,076	50,555	97,416	128,557

Table 18. Numbers of entrapments created in the middle segment associated with alternative hydro operations of 5 or 10 fluctuations per week and fluctuation magnitudes of 10, 20, 30, or 40 kcfs.

	Fluctuation magnitude			
Fluctuations	10 kcfs	20 kcfs	30 kcfs	40 kcfs
5 / week	525	3,371	8,217	12,544
10 / week	631	6,727	15,998	21,682

The fluctuations represented in the 10-per-week at 40-kcfs fluctuation scenario, fell within the range of fluctuation magnitudes and frequencies observed in 2003 (Figure 62) and might be considered a baseline representing current operations. The number of entrapments created in the middle segment for the 10-per-week at 40-kcfs scenario (21,682 Table 17) was similar to the number of entrapments created in the middle segment based on the 2003 empirical hydrograph (23,664 Table 8). This result suggests that the fluctuation characterization for this scenario had a similar effect on entrapment creation in the middle segment as the 2003 empirical hydrograph did. Our empirical estimate of the number of Chinook entrapped in 2003 (1.6 million) fell within the range of impacts predicted for the 10-per-week at 30-kcfs and 10-per-week at 40-kcfs scenarios, which is consistent with the similarities between simulated and observed fluctuations presented. Subsequently, we analyzed the 2003 operations and found that during the period of March 30 through June 5, there was an average of 6.7 fluctuations per week with an average magnitude of 42 kcfs. Fluctuations of less than 10 kcfs were not counted. This result further supports the 10-per-week at 40-kcfs scenario was a close match to the 2003 empirical hydrograph.

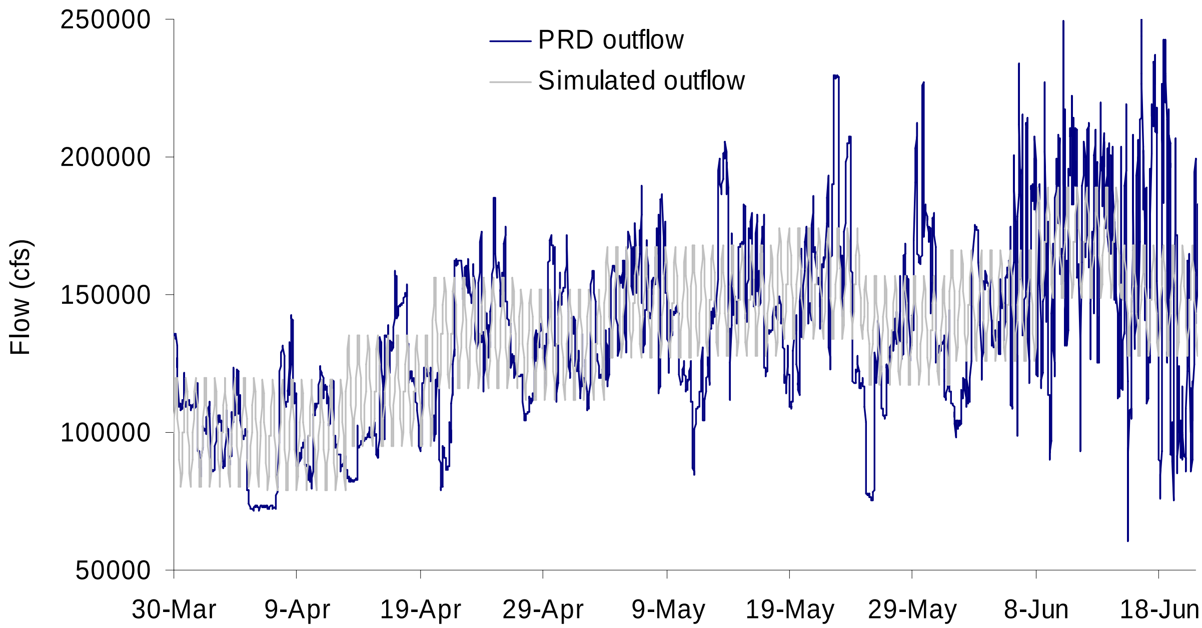


Figure 62. Actual 2003 outflows from (PRD) Priest Rapids Dam and the simulated outflow scenario of ten, 40-kcfs fluctuations per week.

We found strong associations between the simulated number of fish entrapped and the number of entrapments which were created both Reach-wide and within the middle segment (Figure 63), which is not surprising given that the number of entrapments created was used to generate the impact estimates. Accordingly, we found the strongest association to be between the simulated impact estimates and the number of entrapments which were created in the middle segment. While the effects of fluctuation frequency and magnitude on the impact estimates are nonlinear (i.e. neither additive nor multiplicative across the range of operations simulated), there appears to be a strong linear relationship between the number of entrapments created in the middle segment and the overall impact estimates. Because the middle segment had the highest densities of Chinook per entrapment (Table 10), operations which minimize the number of entrapments created in the middle segment would also likely minimize the number of Chinook entrapped across the Reach.

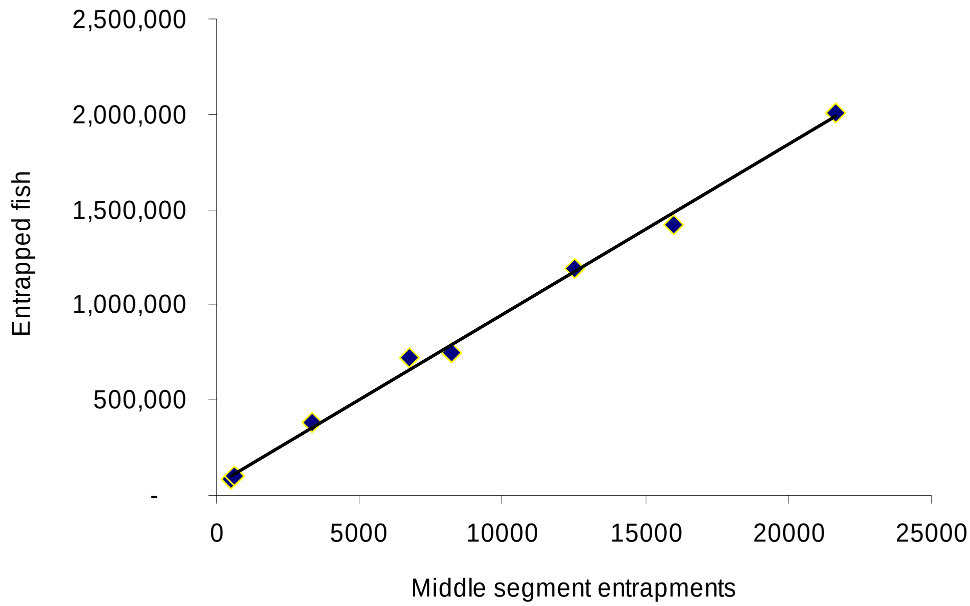


Figure 63. Relationship between the simulated number of entrapments created in the middle segment versus the number of entrapped Chinook Reach-wide in the evaluation of alternative hydro operations.

Capacity for Dampened Flow Fluctuations

PRD and WAN Forebay Storage Analysis

The forebay storage analysis provided reservoir volumes for the subsequent re-regulation analysis. The first forebay analysis conducted on a daily time step showed that PRD used 3.7 feet, and WAN used 3.0 feet of volume at the 50% exceedance level to re-regulate incoming flows for reverse load following during October/ November (Table 19). The results of the analysis on a weekly time step showed that PRD used 5.0 feet and WAN used 6.5 feet of volume at the 50% exceedance level for re-regulation. Using the 50% daily and weekly forebay fluctuation exceedance values for both projects along with the capacity curves for the two reservoirs, we calculated 25,000 acre-ft and 34,000 acre-feet, respectively, was used for re-regulation from PRD, and 45,000 acre-feet and 93,000 acre-feet, respectively, was used for re-regulation from WAN (Table 20) during this time period. As discussed previously, this analysis was based on forebay volumes used during October/ November for reverse load following, and we assumed similar volumes could be used during the spring for reducing flow fluctuations.

Table 19. Exceedance values (%) for forebay fluctuations (ft), 1995 - 2004.

	Priest Rapids		Wanapum	
Exceedance	Daily	Weekly	Daily	Weekly
20%	4.6	5.4	4.0	7.8
50%	3.7	5.0	3.0	6.5
80%	2.7	4.2	2.2	4.8

Table 20. Calculated storage volumes (acre-feet) for PRD and WAN forebays using the 50% daily and weekly exceedance values for observed forebay fluctuations.

Interval	PRD	WAN	Total
Using Daily 50% Exceedance Value	25,000	45,000	70,000
Using Weekly 50% Exceedance Value	34,000	93,000	127,000

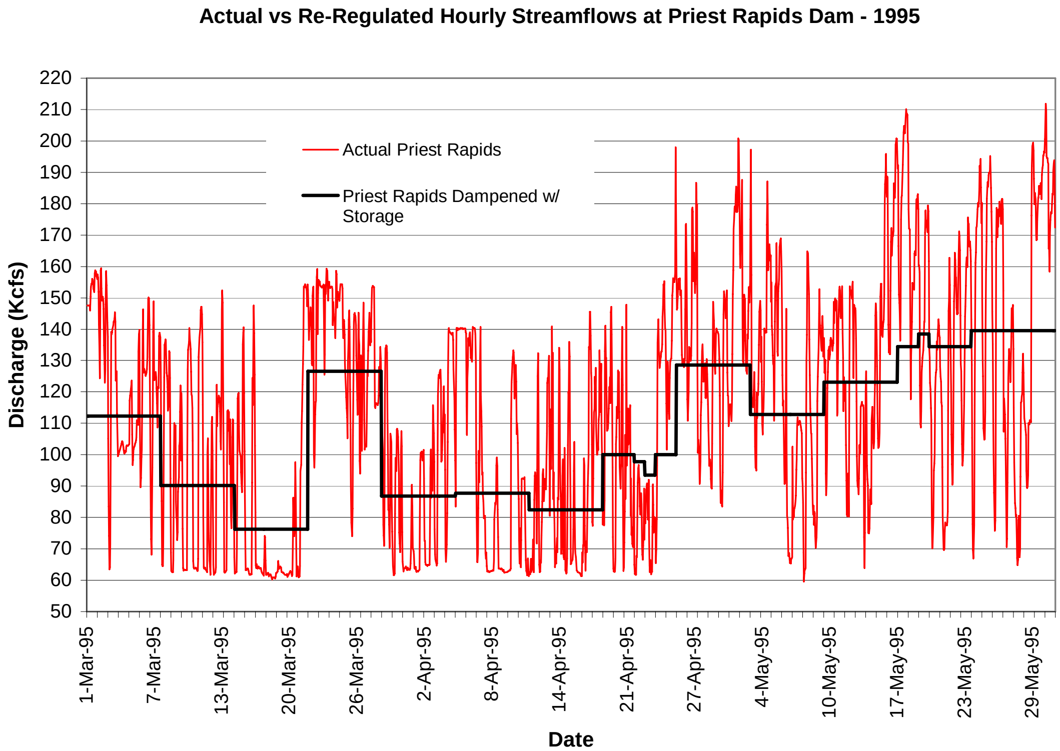
We used the total weekly storage volume (127,000 acre-feet) to adjust hourly streamflows coming into the PRD and WAN projects from the Rock Island project to the weekly average outflow target for PRD that was calculated from Rock Island hourly flows. Actual hourly streamflows at PRD were adjusted to the target, and the resulting volume was accumulated. Higher incoming flows required storage (positive volume), and lower incoming flows required supplementation (negative volume). When the available storage for the week had been used, incoming streamflows from Rock Island were passed through PRD with no modification until storage again, became sufficient to resume the operation. The number of days that incoming flows were passed from Rock Island through WAN and PRD with no modification (storage capacity exceeded), along with the percentage of days when successful adjustment to the target weekly flow occurred are shown in (Table 21).

Table 21. Daily success rates for re-regulation of Rock Island streamflows to the weekly average streamflow target at PRD.

Year	Number of Days Storage Capacity Exceeded	% of Days Average Weekly Flow Target Met
1995	3	96.7
1996	1	98.9
1997	7	92.4
1998	13	85.9
1999	2	97.8
2000	0	100.0
2001	1	98.9
2002	4	95.7
2003	5	94.6
2004	0	100.0

The available storage volumes of Priest Rapids and Wanapum reservoirs based on our exceedance analysis provide significant potential for stabilizing streamflows on an hourly basis downstream from Priest Rapids Dam into the Hanford Reach. Figure 64 through Figure 68 show plots of actual hourly March through May flows versus hourly flows re-regulated to weekly targets based on the calculated storage capacity from the forebay analysis, with deviations when the storage capacity was exceeded. For instance, the re-regulation analysis for 1996 (Figure 64) limited the number of flow reductions to a total of eight. Five of these drops were less than or equal to about 10 kcfs, and the remaining three were about 15, 25, and 35 kcfs. Similarly, the re-regulation analysis for 2003 limited the number of flow reductions during the three month rearing season to a total of eight. Five of these drops were less than or equal to about 10 kcfs, and the remaining three were between 10 and 20 kcfs. For both of these years, the re-regulated flows show a substantial reduction in fluctuations over flows that actually occurred.

Transitions between weekly flow targets were rather abrupt and a function of our target flow calculation process. For real-time implementation of a similar operation, streamflow forecasts could be used, and a rolling average flow target could be calculated rather than independent weekly average targets.



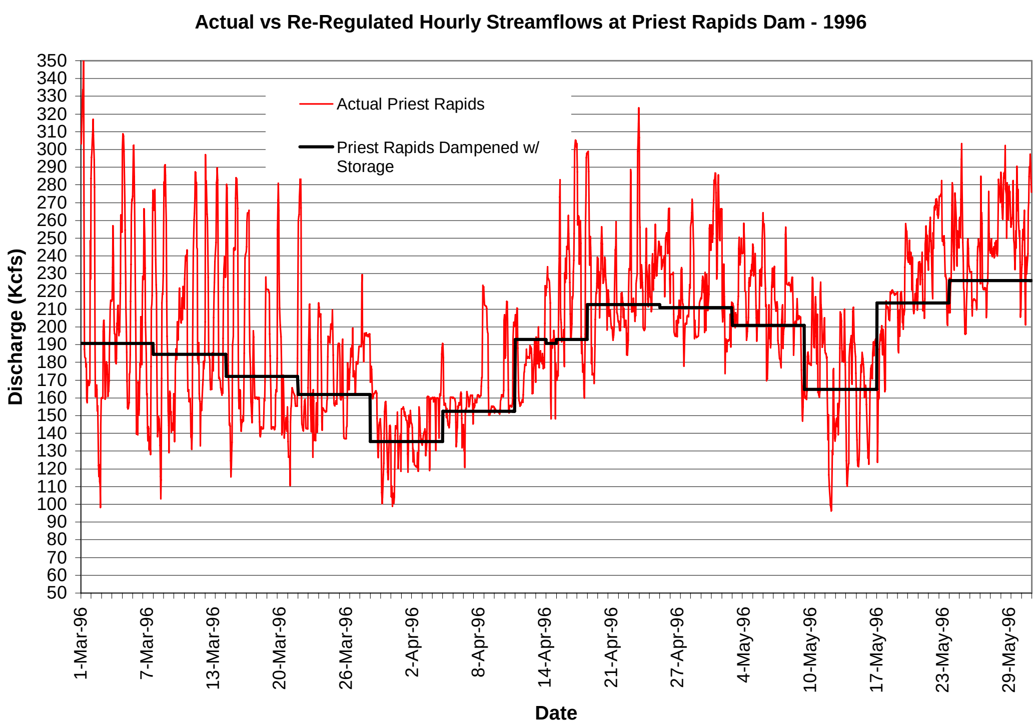
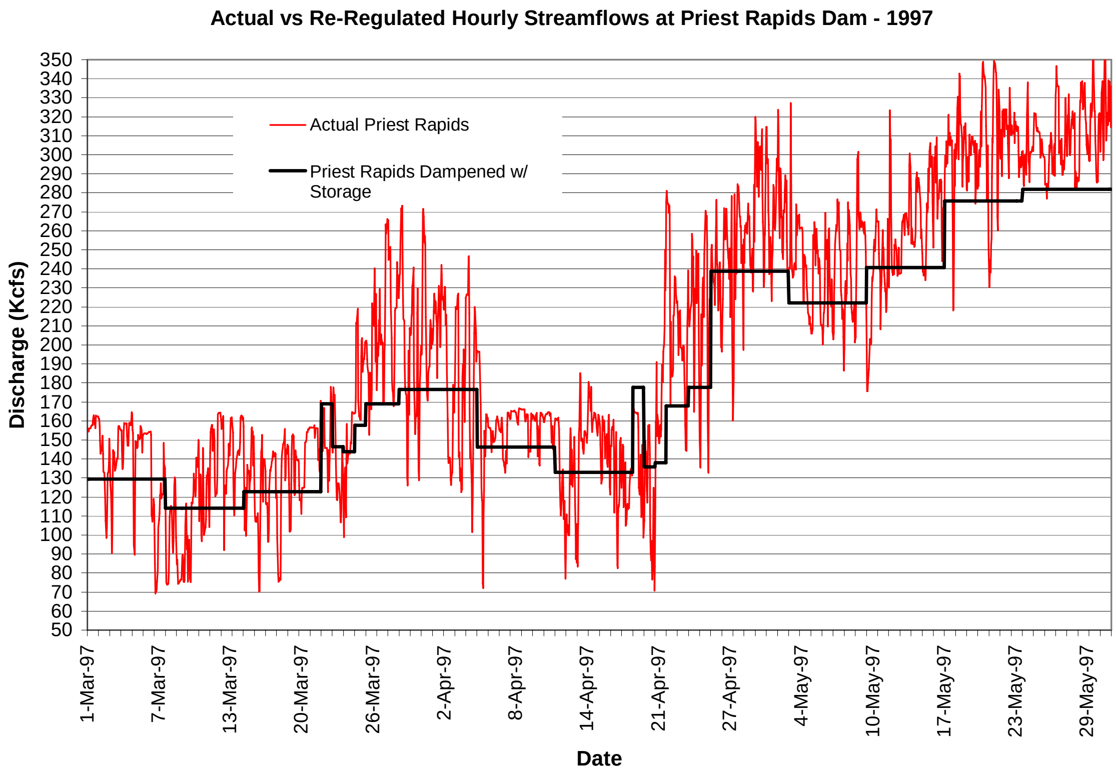


Figure 64. Re-regulation analysis for March through May, 1995 and 1996.



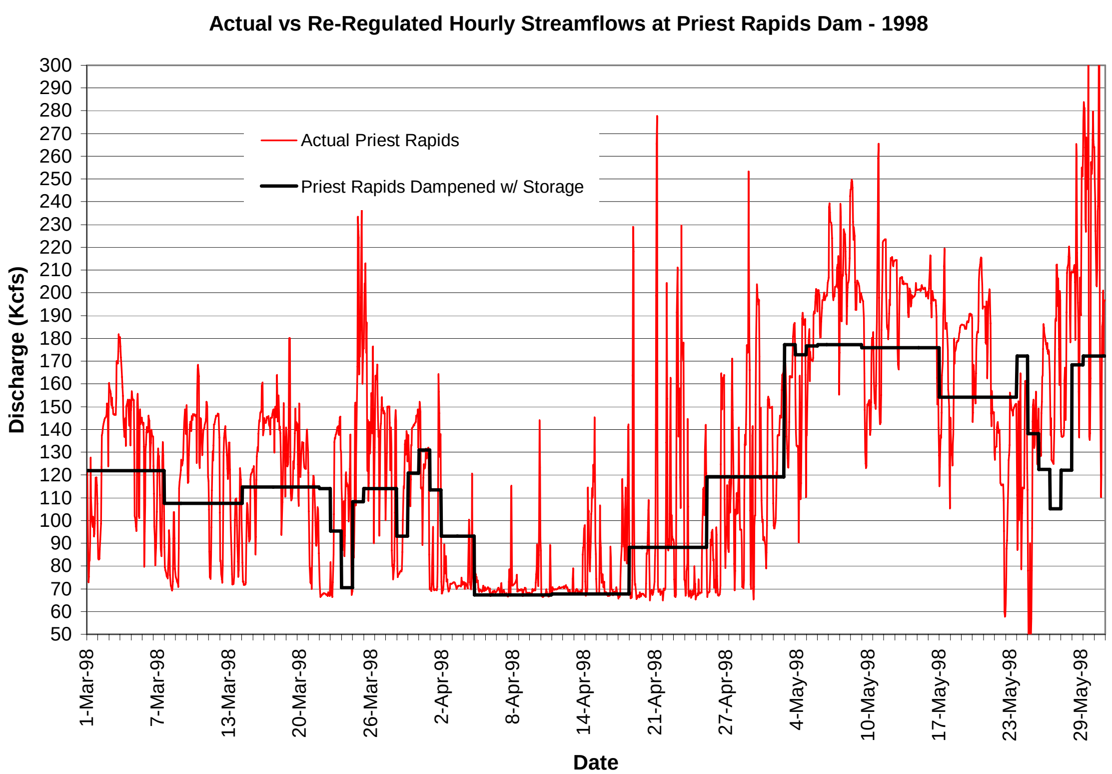
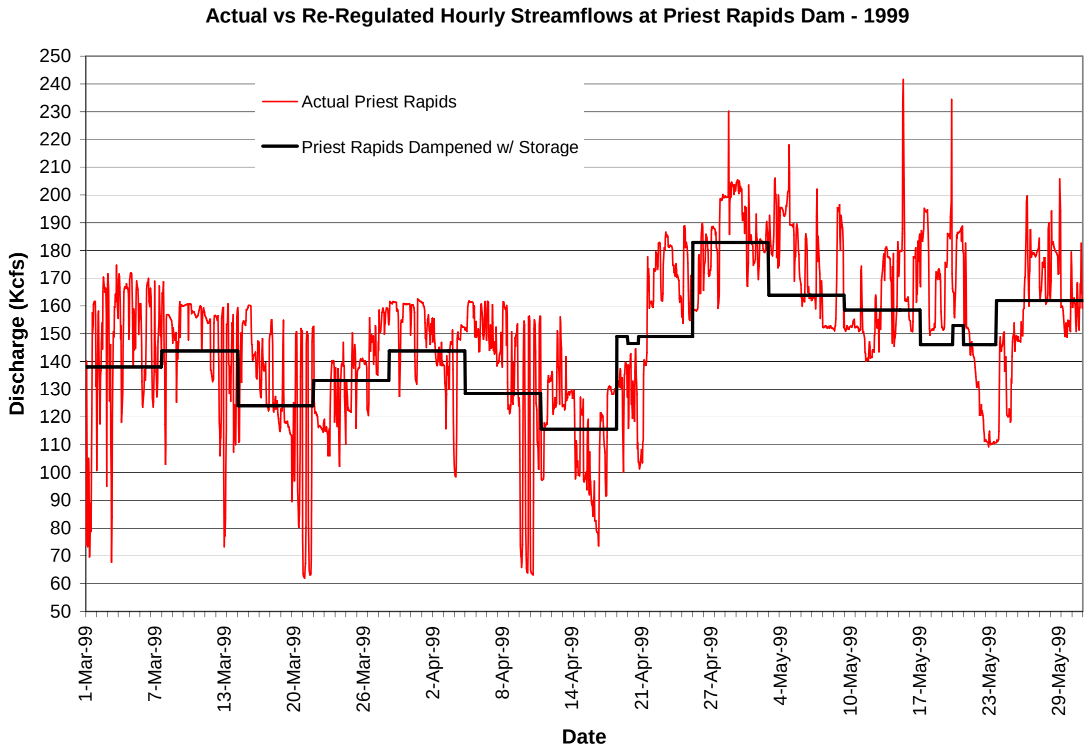


Figure 65. Re-regulation analysis for March through May, 1997 and 1998.



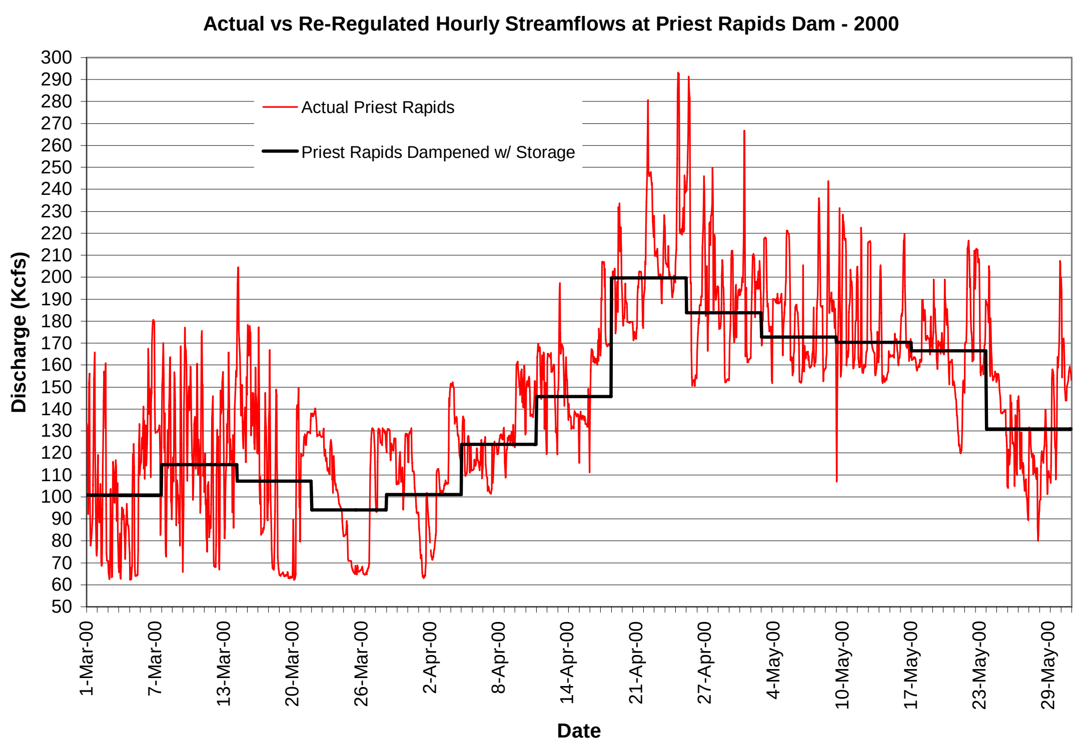
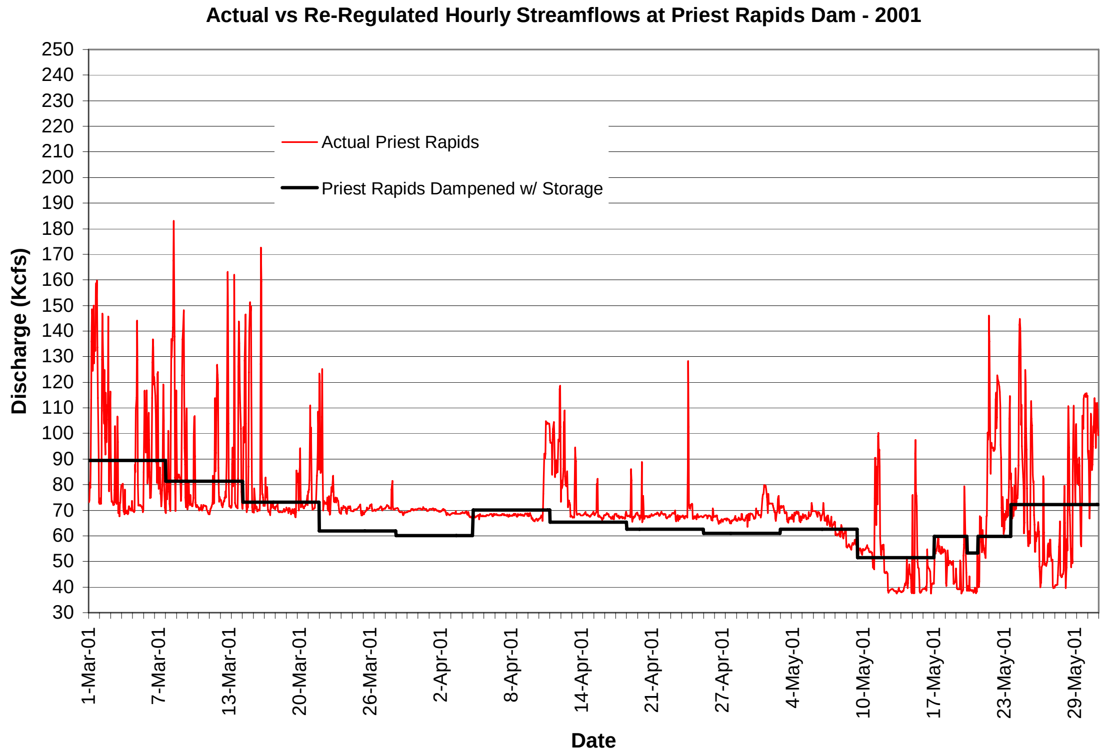


Figure 66. Re-regulation analysis for March through May, 1999 and 2000.



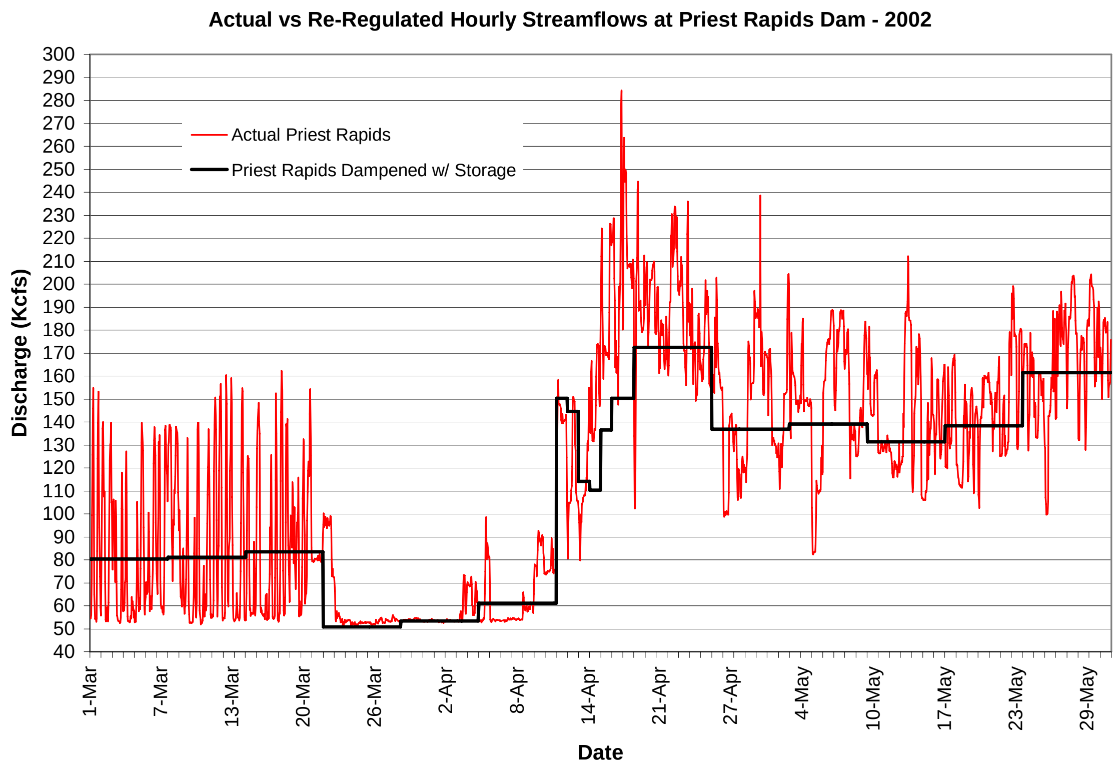
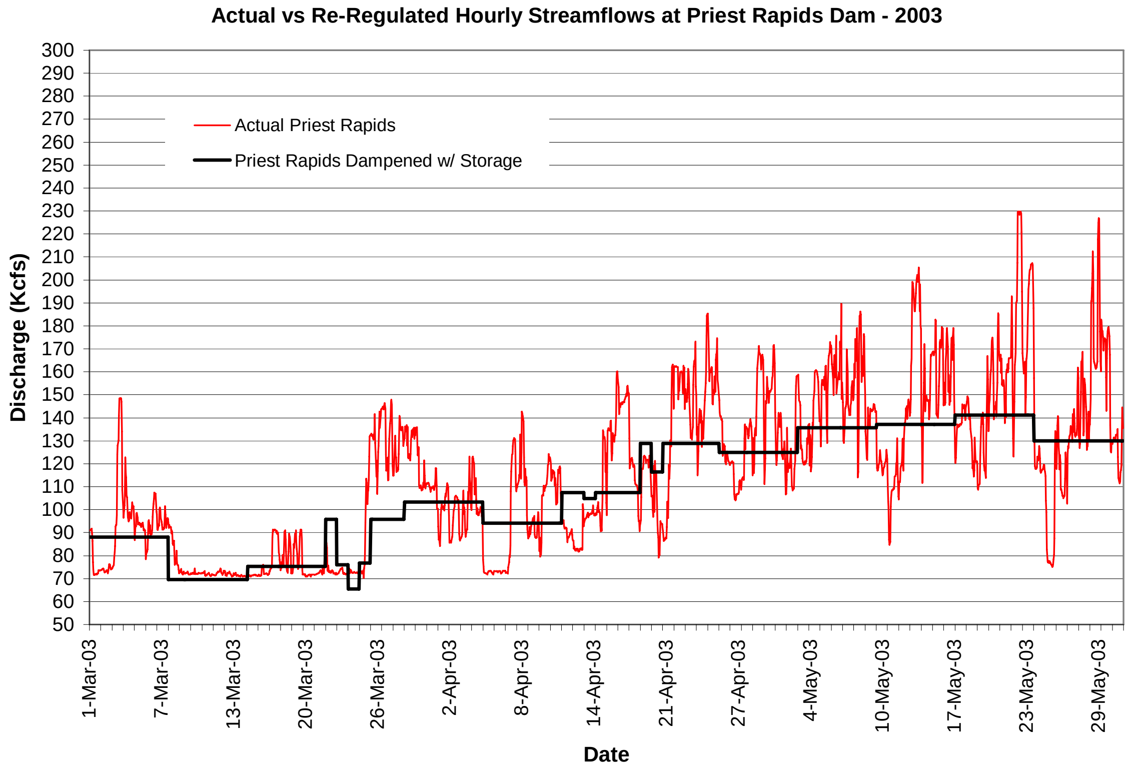


Figure 67. Re-regulation analysis for March through May, 2001 and 2002.



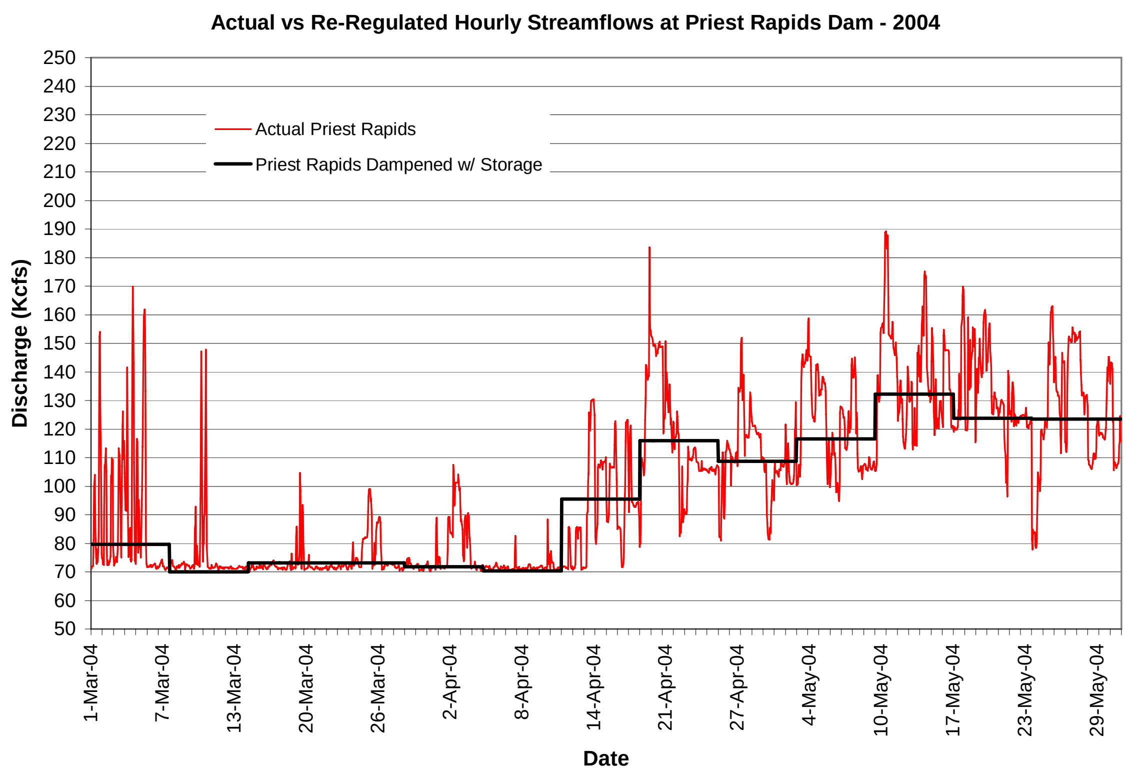


Figure 68. Re-regulation analysis for March through May, 2003 and 2004.

Discussion

The large scale of the Hanford Reach, the ephemeral and dynamic nature of streamflow fluctuations, and the challenges of empirically estimating the effects on fall Chinook required an improved and innovative approach for accomplishing our objectives. We used a combination of field sampling efforts and hydrodynamic modeling. Field sampling provided reference information on fish entrapment under existing operations. Hydrodynamic models allowed us to relate fish impacts along the length of the Reach and over time to specific operations. Empirical physical and biological data integrated with hydrodynamic model output and GIS analysis then allowed us to mechanistically estimate the impact of fluctuating flows on juvenile fall Chinook.

We focused our sampling efforts on entrapments as our primary sampling unit. Although fall Chinook mortality is caused by both stranding and entrapment, focusing on entrapments provided a more tractable index for assessing the minimum impacts due to flow fluctuations downstream of Priest Rapids Dam. Entrapments have the advantage of being well-defined, temporally-stable, geographic locations. Juvenile fall Chinook salmon favor shoreline habitats (Tiffan et al. 2002), thus they are susceptible to entrapment when streamflows drop sharply leaving isolated pools on the river bank. Stranding and entrapment studies conducted in the Hanford Reach by Wagner et al. (1999) found that 99% of the juvenile fall Chinook were found in entrapments rather than on exposed substrate (stranding) following a flow drop. Focusing on entrapments as our sampling unit increased our confidence that if fish were impacted (entrapped), they would be observed. Once these locations were identified, we were able to use hydrodynamic models to recreate entrapment histories. By integrating results of field sampling to determine the spatial and temporal number of fish per entrapment, with the entrapment histories of the number of entrapment events, we produced Reach-wide and section-specific estimates of the number of fish entrapped and the fate of these fish.

Our results confirm that flow fluctuations due to hydropower operations cause significant mortality of juvenile fall Chinook that rear in the Hanford Reach. These impacts appear to be significantly greater than previously estimated. We determined that roughly 1.6 million fall Chinook had been entrapped in 2003. Large numbers of juvenile fall Chinook were affected by flow fluctuations despite operational restrictions designed to reduce entrapment and stranding mortality. Current operational plans specify limits on the magnitude of flow fluctuations during the spring rearing period. Our observations of significant entrapment in spite of the current protection measures highlight the importance of developing operational alternatives that are more effective at minimizing juvenile mortality.

Our estimates of the number of entrapped fish were three to six times the McMichael et al. (2003) estimates using the area-based methodology for the number of fall Chinook entrapped or stranded in the same areas. There are several potential reasons for this disparity. One reason is that the area-based assessment methodology likely suffers from low detection probability for stranded fish and underestimated entrapment effects. The substrates typically found on the Reach make locating and enumerating juvenile fall Chinook extremely difficult. Even when fish are present in an area, extensive excavations would be required to ensure that all or most of the stranded fish had been counted. Low detection probabilities would result in negatively-biased density estimates and could explain the disparity between our estimates and the estimates generated using the area-based methodology. Another potential reason for the disparity is the inefficient coverage of entrapments in the area-based methodology. With the area-based methodology, there is no differentiation between the densities of fish in entrapment areas versus stranding areas, and entrapments were infrequently encountered in the sample plots. However, in 2003 we found a density of 0.4 fish/m^2 in our entrapment sampling, while the area-based methodology found 0.003 fish/m^2 in their sample plots. These two-orders-of-magnitude differences in the densities of fish in entrapments versus the area-based sample plots could also explain the differences between the two estimates.

While our work provides a useful tool for estimating the relative effects of alternative operations on entrapment, our results should not be construed to represent the total mortality effects of operations on fall Chinook. The magnitude of stranding mortality relative to entrapment mortality is a significant uncertainty which limits our ability to quantify the cumulative effects of Priest Rapids operations on fall Chinook mortality in the Reach. If full quantification of mortality is deemed necessary, then studies would need to be conducted to reliably estimate the effect of true stranding along with entrapment evaluations. Alternatively, various assumptions about the relationship between entrapment effects and stranding effects could be incorporated into the impact estimates and the decision-making process. However, our methodology provides a reliable index of entrapment mortality which can be used to judge the relative performance of alternative hydrosystem operations on the expected number of entrapped fish.

While our methodology provides a robust index of entrapment effects, there are several reasons to suspect that our estimates are conservative. Because our 2003 field sampling program started well after emergence was underway and our entrapment enumeration was not complete, particularly at lower flows, there is reason to believe that the number of entrapments created at low flows was under-estimated. Flows ranged from 60 to 261 kcfs during spring 2003, and because entrapment mapping was incomplete across this entire flow range, fluctuation effects on entrapment histories would have been under-estimated. In addition, the large number of fish per entrapment during the first period in the middle segment corresponded to a period when the seining surveys suggested low abundance. If entrapment fish sampling had been conducted in the previous two-week period when seining abundance was also low, similar high numbers of entrapped fish may have been present.

To a lesser degree than previous studies, incomplete detection of entrapped fish was also an issue for our entrapment work, although the mechanism is somewhat different. Fish were observed in an additional 46 entrapments (4.9% of the pools sampled in detail) during our field sampling effort, but were not recovered during seining. Observations of fish moving vertically, down into the interstitial spaces between cobble substrates were common. In addition, fish were observed moving upwards, out of the substrate. This is presumably a predator avoidance behavior. Additional work is needed to identify and quantify the effect of this behavior on sampling efficiency. This effect has likely biased our impact estimate low. An additional factor that likely affected our estimates was predation of entrapped fish. Chinook mortality in entrapment pools due to predation was not quantified. Our inability to account for this source of mortality has likely biased our impact estimate low. Predation studies could be conducted in the future to determine the significance of this source of mortality.

We used three approaches to place our impact estimate into a population context. Using the SAR data, we estimated the 4,100-8,200 adults may have been lost due to entrapment mortality in 2003. Using the fry production modeling data, the 2003 entrapment impact alone likely constituted a 12% reduction in the fry population. In other years under the Protection Plan the impact to the population caused by operations at Priest Rapids Dam may have been 31% to over 90%. The scale of these impacts to the fry population would translate into losses of 9,000-170,000 adults in ocean and in-river fisheries. These foregone adults represent large reductions in the number of adults available to commercial and sport ocean fisheries, commercial, sport, and tribal treaty in-river fisheries. These impacts directly correspond to significant lost harvest opportunities and the associated economic benefits, agency responsibilities for management of public and tribal trust resources, and preservation of the adult population that escapes to spawn. Improving freshwater rearing survival conditions becomes particularly important for reducing the risk of lower adult returns in years of poor ocean survival and poor freshwater rearing and outmigration conditions. The actions for reducing mortalities related to entrapment events appear to have a high degree of certainty relative to the other actions and uncertainties that impact Columbia River fall Chinook including but not limited to: downstream mainstem migration improvement measures; transportation of fall Chinook; avian, pisciverous, and pinniped predation control measures; mainstem adult passage improvements; and variable ocean/climatic conditions.

We found little correspondence between the seining data and the number of fish per entrapment data. This was due both to the high variability in our seine catches as well as variability in the number of Chinook found in entrapments. Juvenile fall Chinook salmon generally travel in schools and routinely move up and downstream in shoreline habitats making their capture unpredictable. We believe that this result suggests that while seining may be useful as a coarse indicator of fish presence or size, it would not serve as a useful index by itself of potential for entrapment impacts. We also found no evidence that flow levels, flow bands, or variability in flow could explain the observations of the number of fish per entrapment. While the simulation results demonstrated the strong effects of flow fluctuations on overall entrapment impacts, the factors that determine fish densities in entrapments remain unclear. These results are consistent with those reported in McMichael et al. (2003) on the current state of uncertainty regarding the mechanisms which influence entrapment and stranding.

Based on the observations in these two studies, there is little quantitative basis for assuming that flow fluctuations at low flows are more harmful than the same fluctuations at high flow levels, or vice versa. However, the seining efforts and flow comparisons may not have been sensitive enough to detect meaningful effects, even if they may have been present (i.e., high potential for Type-II errors). The lack of correspondence between fish per entrapment and the gross measures of hydraulic conditions in the Reach may support further detailed mechanistic evaluations and planned experimental flow manipulation studies using the simulation model that we developed to evaluate alternative hydro operations.

Our collection of diel behavioral data on juvenile fall Chinook salmon was limited both in seasonal and geographic scope, and represented only a brief portion of all of the potential rearing areas and the rearing period. Our study area was chosen because it is a major rearing area for juvenile fall Chinook salmon and we believed we had the greatest likelihood of observing fish there. Although we did not observe fish early in the rearing period, the size of the fish we observed (<50 mm) was in the range of the greatest susceptibility to stranding and entrapment. We did not attempt to determine if there were size-related differences in behavior. The past investigation of Venditti and Garland (1996) at different locations in the Hanford Reach lead us to believe that the behaviors we observed also occur elsewhere in the Reach.

The diel behavior of juvenile Chinook salmon has been shown to be highly variable from river to river, impacted by water temperature, food availability and presence of predators. In most conditions, Chinook salmon typically are quiescent at night and rest in low velocity microhabitats. Our finding that juvenile fall Chinook salmon are typically inactive at night and associated with the river bottom supports this finding and agrees with similar observations made by Venditti and Garland (1996), Tabor and Piaskowski (2002), and McMichael et al. (2003). This nighttime behavior may make them more susceptible to stranding and entrapment events resulting from nighttime flow reductions. At night, fish may also move to the bottom and remain inactive to minimize their risk of predation. Alternatively, Bradford and Higgins (2001) showed that juvenile Chinook often hide in the substrate during the day to presumably avoid predators and emerge in the evening to feed. If this were true in the Hanford Reach, then our video sampling would not have detected fish hiding in the substrate and would have biased our conclusions on diel fish activity. The substrate in many of the shoreline habitats we examined was embedded to the point that fish likely could not use the interstitial spaces. Additional studies to examine use of the entire range of rearing habitats available including subsurface habitats employing a wider range of techniques including video and snorkel surveys throughout the study area would be insightful. However, our observations over numerous years support the notion that juvenile fall Chinook salmon in the Hanford Reach do not hide during the day, but are actively feeding and moving. We believe that the very shallow water that fish inhabit during the day may actually serve as a refuge from larger piscine predators. Based on these observations, we conclude that the critical period in which juvenile fall Chinook salmon are most susceptible to nighttime water elevation decreases extends from the start of emergence to the time that the population is of sufficient size to have moved into offshore habitats. During this time, fish would benefit from limiting water level fluctuations at night. Additional studies to determine the portion of stranding and entrapment that is attributable to daytime and nighttime flow reductions would be helpful to further elucidate the effects of diel behaviors on stranding and entrapment.

We found that there is considerable potential for reducing the impacts of flow fluctuations on fall Chinook if fluctuation magnitudes are kept below 10 kcfs through the use of our hydro operation simulation model. The number of entrapped Chinook dramatically increased with increasing flow fluctuations of 20, 30 and 40 kcfs. Based on the non-linear relationship between the fluctuation magnitude and frequency, and fish entrapped, it appears fluctuations above 10 kcfs produce dramatic increases in the number of fish entrapped. Flow fluctuations create variable numbers of entrapments in the Reach depending on the size of the fluctuation and the number and location of entrapments in the affected areas. The model we developed allows us to explicitly consider wave dampening and dissipation downstream in the middle segment, where the majority of the entrapment occurred. If other operational alternatives are proposed, our model could serve as a useful tool for assessing the potential impacts. Given the flexible capabilities of our evaluation tool, there is also the potential to develop and evaluate adaptive, in-season operational guidelines. These guidelines could allow for more fluctuations during periods when the numbers of fish per entrapment are low, and more stable flows during critical periods when the numbers of fish per entrapment are high. The next phase of our work should be to use this tool in an exploratory mode for ongoing evaluations of operational alternatives. We believe the mechanistic basis of this evaluation tool improves the assessment capabilities for managers to explore the impact of alternative hydrosystem operations on the Chinook population of the Hanford Reach.

Our analysis of the capability of the Priest Rapids and Wanapum projects to re-regulate and reduce flow fluctuations indicated that the flexibility exists within "normal" operating limits to both limit the number of flow reductions and the magnitude of the reductions. We examined empirical forebay volumes used to implement the reverse loading operation that is conducted under the Vernita Bar Agreement in October and November and discovered that if those same volumes could be used during the March through May rearing period, flow fluctuations could be substantially reduced. We would expect a corresponding reduction in entrapment mortality, although it is not possible to quantify the expected reduction without additional analysis. Our re-regulation analysis was an elementary, physical analysis using a mass balance approach, and we did not consider other fishery issues that may be relevant during the spring time period, nor did we attempt to evaluate the effect of our analysis on power production. The Hourly Coordination Agreement power optimization software should be configured with the goal of reducing or eliminating flow fluctuations from Priest Rapids Dam into the Hanford Reach to determine how such an operation during the spring would affect power output from the mid-Columbia. In addition, an evaluation should be conducted to determine if these operations would affect the ability of the system to meet Biological Opinion flow targets during the spring.

Our evaluation was conducted in spring 2003 when the *2003 Hanford Reach Juvenile Fall Chinook Protection Program,* now termed the *Hanford Reach Fall Chinook Protection Program,* was in effect. This program is described in McMichael et al. 2003. The intent of this program was to control flow fluctuations in a way that was thought to limit juvenile Chinook mortality. Our results indicated that the actual level of impact was significantly higher than measured by the monitoring program (area-based methodology) that accompanied the *Protection Program.* The results of our evaluation of alternative hydrographs indicated that the impact could be reduced by controlling the size and frequency of flow fluctuations, and the results of our field work suggest that the timing of fluctuations (early vs. late rearing season) can also be used to reduce the impact. The size and frequency of flow fluctuations are directly related to the number of entrapments affected. In other words, larger fluctuations affect more flow bands, and the more flow bands that are affected, the more entrapment events that are created. This effectively increases the potential to impact fish. Considering the results of our re-regulation analysis, the physical ability exists to control flow fluctuation magnitude and frequency to the extent required to reduce expected juvenile Chinook impacts. Connor and Pflug (2004) also found that reducing the frequency of flow fluctuations and slowing the rates of change during downramping reduced stranding rates for salmon fry in the Skagit River.

While our work focused on the impacts of fluctuating flows on juvenile Chinook salmon, it is also important to consider the impacts on the rest of the Hanford Reach aquatic community.

The other fish species that reside in the Hanford Reach are subject to similar impacts. For example, Nugent et al. (2002d) discussed the stranding or entrapment of 18 other fish species in the Hanford Reach during studies from 1997 to 2002. In addition, the University of Idaho conducted studies on the effects of flow fluctuations on benthic macroinvertebrates (in Nugent et al. 2001, 2002a). They found that density and biomass of invertebrates within the river fluctuation zone were severely limited compared to the communities on continually inundated areas. Fluctuation zone impacts on both primary and secondary productivity, in turn, may have impacts on juvenile Chinook and other fish species in terms of the food supply in nearshore rearing areas. Invertebrates contribute to the secondary productivity of the aquatic ecosystem, and are an important food supply for most fish species including juvenile Chinook salmon. Indirect effects of flow fluctuations on fall Chinook productivity through ecosystem effects were not evaluated in this assessment.

Conclusions

Using empirical physical and biological data integrated with hydrodynamic model output and GIS analysis, we mechanistically estimated the impact of fluctuating flows on juvenile fall Chinook salmon for the entire Hanford Reach. Our results confirm that flow fluctuations due to hydropower operations cause significant mortality of juvenile fall Chinook that rear in the Hanford Reach. These impacts appear to be significantly greater than those previously estimated. Our observations of significant entrapment in spite of the current protection measures highlight the importance of developing operational strategies that are more effective at minimizing juvenile mortality.

While our work provides a useful tool for estimating the relative effects of current and alternative operations on entrapment, our results do not represent the total mortality effects of operations on juvenile fall Chinook. Our impact estimate was conservative for several important reasons; 1) sampling started well after emergence was underway, 2) entrapment enumeration was not complete, likely resulting in under-estimates of entrapment histories, 3) detection and enumeration of entrapped fish was incomplete, 4) predation on entrapped fish was not accounted for, and 5) we did not attempt to quantify the potentially significant level of mortality due to stranding. Considering these factors along with our impact estimate, we believe that previous efforts may have greatly under-estimated the actual mortality level impacts that occurred.

Based on the results of our studies, there is little quantitative basis for the assumption that a flow fluctuation at low flows is more harmful than the same fluctuation at high flows, or vice versa. In addition, our evaluation of alternative hydrographs indicated that the impact could be reduced by controlling the size and frequency of flow fluctuations, and the results of our field work suggest that the timing of fluctuations (early vs. late rearing season) could also be used to reduce the impacts. The simulation results suggest that reducing flow fluctuations has considerable potential for reducing mortality levels if fluctuation magnitudes are kept below 10 kcfs. The number of entrapped Chinook dramatically increased with increasing flow fluctuations of 20, 30 and 40 kcfs. Based on the non-linear relationship between the fluctuation magnitude and frequency, and fish entrapped, it appears fluctuations above 10 kcfs produce dramatic increases in the number of fish entrapped. The size and frequency of flow fluctuations were directly related to the number of entrapments affected. Considering the results of our re-

regulation analysis, we believe the physical ability exists to control flow fluctuation magnitude and frequency to the extent required to reduce expected juvenile Chinook mortality impacts. Our analysis of the capability of the Priest Rapids and Wanapum projects to re-regulate and reduce flow fluctuations indicated that the flexibility exists within "normal" operating limits to both limit the number of flow fluctuations and the magnitude of the fluctuations.

We used three approaches to place our impact estimate into a population context. Using the SAR data, we estimated the 4,100-8,200 adults may have been lost due to entrapment mortality in 2003. Using the fry production modeling data, the 2003 entrapment impact alone likely constituted a 12% reduction in the fry population. In other years under the Protection Plan the impact to the population caused by operations at Priest Rapids Dam may have been 31% to over 90%. The scale of these impacts to the Hanford Reach fry population would translate into potential harvest reductions for all adult Chinook of 9,000-170,000 fish in ocean and in-river fisheries. These foregone adults represent large reductions in the number of adults available to commercial and sport ocean fisheries, commercial, sport, and tribal treaty in-river fisheries, and spawning escapement to the Hanford Reach.

Recommendations

- Continue the fish sampling program during the emergence and rearing periods to estimate the total number of fish entrapped using our modeling framework. The sampling effort should be kept at a level similar to the 2003 effort, or increased to reduce the resulting uncertainty in the estimate.
- Develop and implement a study plan to evaluate entrapment fish sampling efficiency.
- Complete the enumeration of entrapments, especially at the lower flow levels.
- Implement additional stratification of entrapments for the field sampling program by size, location (flow band), and other physical features to help reduce uncertainty around impact estimates.
- Develop and implement planned flow manipulation experiments to quantify the diel impact on fish per entrapment.
- Develop and implement a plan to estimate the effect of fluctuating flows on ***stranding*** of juvenile Chinook, and design a statistically rigorous sampling program to survey stranding areas.
- When stranding field studies are complete, incorporate an evaluation component for stranding into our modeling evaluation system.
- Investigate the role of water temperature as it relates to stranding and entrapment susceptibility.
- Continue with the development and evaluation of alternative operations and their effect on Chinook stranding and entrapment.
- Conduct a focused study on the impact of various ramping rates on entrapment and stranding.

- Conduct a more rigorous abundance index seining program to determine if such a program can be linked to subsequent entrapment and stranding locations and magnitudes. The index seining could then potentially be used as a monitoring tool.
- Conduct a focused study on predation rates on stranded and entrapped fish.
- Conduct similar studies on resident fish species that are also susceptible to entrapment.
- Evaluate the effect of re-regulating flows from the Priest Rapids Project into the Hanford Reach on power production from the mid Columbia using the Hourly Coordination Agreement power optimization software.
- Develop analytical approaches to improve estimates of the effect of entrapment and stranding mortality rates on adult productivity of Hanford Reach fall Chinook.

SPAWNING HABITAT ASSESSMENT

Overview

Fall Chinook salmon require unique physical habitat conditions for spawning which historically were abundant in the mainstem Columbia River. Prior to hydrosystem development, fall Chinook selected spawning locations under the relatively stable flow conditions that were available during the fall period. Following development of the hydrosystem, many of these spawning habitats were lost and the remaining habitats continue to be compromised by hydrosystem operations. It has been estimated that only 10-13% of the historical spawning habitat remains (Dauble et al. 2003, Dauble and Watson 1990). Due to the dynamic operations of the hydrosystem, the characteristics that define suitable habitat conditions change temporally and spatially, often on timescales of less than an hour. These operations constitute neither normative river flows nor do they provide the stable spawning conditions that were historically available.

Escapements as high as 90,000 fall Chinook have returned to spawn in the Hanford Reach. Fall Chinook spawning occurs in a number of specific areas scattered throughout the upper, middle, and lower segments of the Hanford Reach (Figure 69). Spawning habitat in the middle and lower segments of the Reach has been associated with water depths of 2-4 m, water velocities of 1.4-2.0 m/s, and in areas with lateral slopes of less than 4% (Geist et al. 2000). In addition to the observations of shallow-water spawning, spawning activity has also been documented in deep waters (up to 9 m), but the extent of deep-water spawning has not been quantified (Chapman et al. 1986, Swan 1989). Mathematical descriptions of the relationship between physical variables and the presence or absence of spawning have been developed previously by other researchers for the Hanford Reach. An existing model (Geist et al. 2000) provides insight into the factors which may be important to spawning site selection by fall Chinook.

The availability and suitability of spawning habitat is believed to be a function of substrate, streamflow, and channel morphology, and these vary both along the Reach and over time under current hydrosystem operational strategies. Fluctuating flows resulting from hydropower generation increase the frequency of flooding and dewatering of spawning habitats, and produce a high level of variation in water velocities and depths. Understanding the effects of hydrosystem operational decisions on the quantity and location of spawning habitat is a critical need for resolving biological and hydrosystem management questions. Previous evaluations and models have not effectively captured the effects of flow fluctuations from Priest Rapids Dam on the distribution and amount of spawning habitat in the entire Hanford Reach. Existing spawning habitat models use average daytime conditions to predict spawning habitat and do not explicitly consider the effects of fluctuating streamflows. Furthermore, there has been no investigation of the effects of alternative operations (e.g., different flow volumes or levels of flow fluctuations) on the relative quantity of spawning habitat. While it is biologically reasonable to expect that fall Chinook spawning in the Reach would benefit from less variable flow conditions, quantitative data on this topic do not exist. To conserve and better manage the Hanford Reach fall Chinook population, it is necessary to understand not only the relationship between steady flow levels and the amount of spawning habitat, but also the effects of fluctuating or unsteady flows on spawning habitat. This type of insight would allow fishery and hydrosystem managers to work together to design flow regimes that minimize the impacts of hydropower generation and fluctuating flows on potential spawning habitat.

The objectives of this spawning habitat assessment were to: 1) quantify the distribution of spawning habitat throughout the Hanford Reach, 2) characterize the physical characteristics associated with spawning and non-spawning sites, including parameters related to streamflow and streamflow fluctuations, 3) build a model which could retrospectively describe patterns of Chinook spawning habitat selection and which included variables that explicitly accounted for the dynamic nature of the physical conditions, 4) prospectively quantify the relative changes in spawning habitat area that may occur under alternative hydrosystem management strategies, and 5) begin development of a tool that can be used by fishery and hydrosystem managers for in-season operational planning. Given that fall Chinook historically spawned under more stable streamflows, we hypothesized that they were more likely to spawn at locations where suitable spawning conditions were persistently available rather than at locations where suitable spawning conditions were only intermittently available. In our analysis we attempted to confirm or reject this hypothesis.

We employed a suite of analytical and statistical tools within a GIS framework to accomplish our objectives. First, we digitized fall Chinook spawning locations throughout the 90-km Hanford Reach using digital orthophotography. Second, we characterized the hydraulic conditions that occurred during the spawning season at spawning and non-spawning sites with a two-dimensional, depth-averaged hydrodynamic model. Third, we created a database for exploratory analysis by attributing a set of spawning and non-spawning point locations with hydraulic and geomorphic data. Forth, we built, calibrated, and tested several GIS models of fall Chinook spawning habitat with multivariate logistic regression and cell-based modeling. We also evaluated an existing spawning habitat model developed by Geist et al. (2000) in the Hanford Reach. Fifth, we estimated relative changes in fall Chinook spawning habitat at different simulated streamflows (40 – 180 kcfs) and flow regimes (steady and fluctuating) with the GIS habitat model. We used this process through several iterations of model-building and testing for specific areas within the Reach, as well as for the Reach as a whole. Details presented here regarding methods and results discuss the latest iteration of our habitat modeling, and reflect the insight we have gained from our previous work.

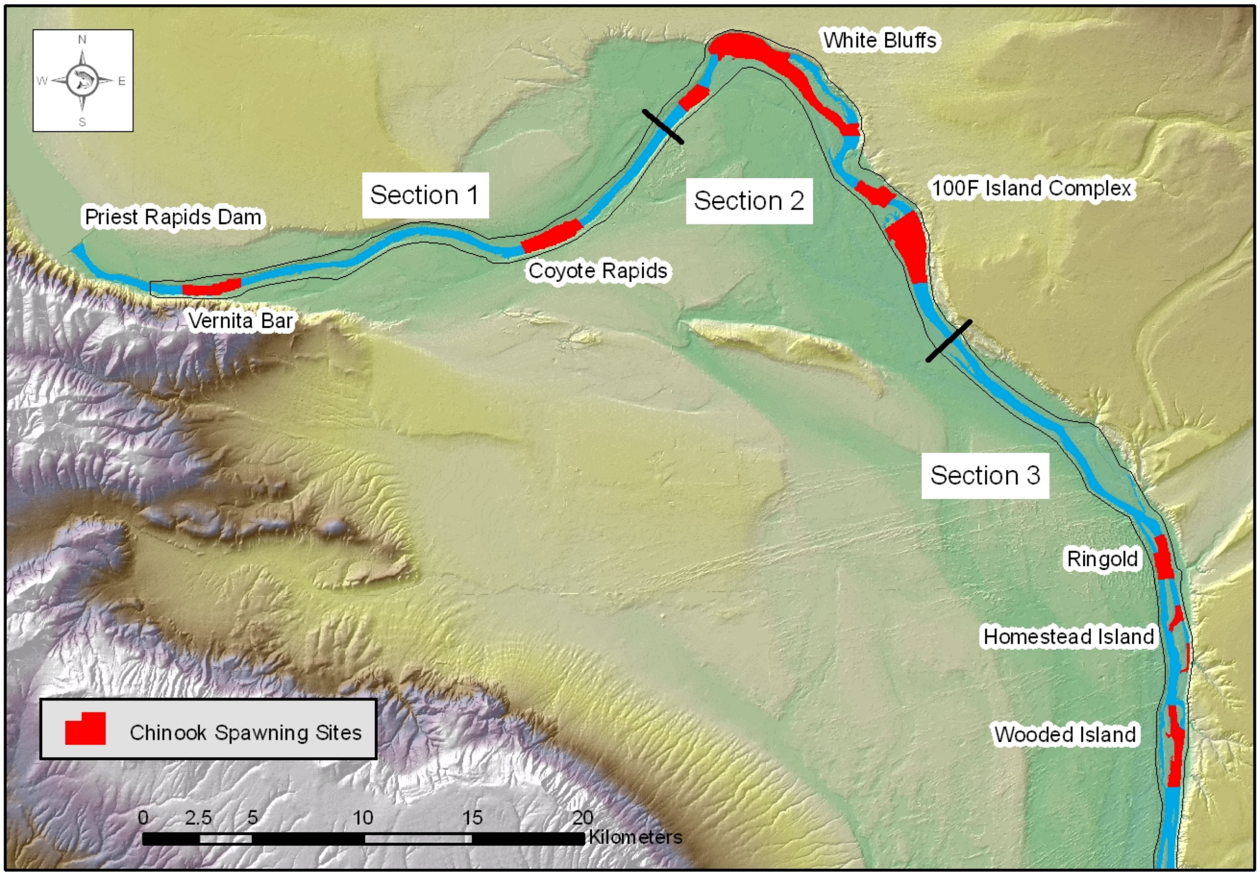


Figure 69. General fall Chinook spawning locations (red blocks) in the Hanford Reach along with section locations.

Methods

Spawning Distribution

Spawning distribution throughout the Reach was identified based on aerial photos of fall Chinook nests or redds. Aerial photography is practical in the Hanford Reach because water clarity is high and redds are large and distinct (Visser et al. 2002). Secchi disc measurements, which are an index of water clarity, made in the summer of 2003 were 4 m or greater (FWS unpublished data). In addition, identification of redds from aerial photos is possible because of their large size. Chapman et al. (1986) reported average redd sizes in the Hanford Reach were 17 m^2 and our studies have identified redd sizes up to 50 m^2. Fall Chinook typically build their redds in clusters (Geist and Dauble 1998). Newly excavated redds appear as light-colored ovals that contrast with the darker undisturbed substrate. Redds contrast with adjacent substrates because periphyton is removed from the substrate during redd construction. Redds usually remain visible in the Columbia River for about 6 weeks, at which time periphyton begin to recolonize the substrates (Dauble and Watson 1990). Visibility of redds from an airplane can be affected by water surface turbulence from wind or water currents, atmospheric haze or low level clouds, flight altitude, and localized turbidity from eroding banks of the Ringold formation. Flights are carefully planned to optimize climatic conditions while photos are being taken.

In 2004, we contracted a 0.3 m cell resolution orthophotography flight for the entire Hanford Reach to image, enumerate, and map fall Chinook redd clusters. Orthophotography provides both high resolution images of redds and redd clusters and geographic locations for all features in the images. Thus they are suitable for further analysis in a GIS. The survey was conducted two days after the timing of peak redd counts, which had been determined by aerial surveys conducted by PNNL. Orthophotos were taken between 10:53 AM and 11:37 AM. Because the majority of redds were excavated from November 4 through November 15, and our photography was conducted on November 17, a quantitative assessment of redd clusters was possible. Flights were made when streamflows and water depths were low (54 kcfs at Vernita Bar). However, as the flight progressed downstream, discharge and depth increased due to the reverse load-following operations the previous night. These conditions made digitizing some redd clusters more difficult, particularly for clusters near the thalweg and in the lower segment of the river.

Redd cluster locations were identified from 103 digital tiff images photographed in 2004 for the entire Reach (rkm 639 to 548). All images were scanned in detail for the presence of redds. Redds within 30 m of one another were considered clusters and the perimeter of each cluster was digitized and stored in the GIS. Individual redds observed further than 30 m from the cluster were not digitized. A second set of Reach-wide orthophotos was obtained in 2005. The resulting images were processed as described above.

Geist et al. (2000) identified spawning and non-spawning areas using aerial photography and underwater video. They collected physical and hydraulic data along transects using an ADCP and an electronic total station. Lateral slope and substrate characteristics were also determined.

Hydrodynamic Modeling

We estimated steady-state, depth-averaged water velocities, water surface elevations and depths for the upper, middle, and lower segments of the Hanford Reach with the River2D hydrodynamic model (Ghanem et al. 1996). The modeling methods we used for Sections 1 and 3 are described in detail in the first section of this report. The last step of the hydrodynamic modeling process consisted of exporting the depths, velocities, and water surface elevations along with their locations to be formatted in grids for further use in ARC/INFO. Because Tiffan et al. (2002) had already modeled a range of discharges with River2D for Section 2, we incorporated their results into our GIS database to conduct the habitat assessments. Due to the enormous size of the Hanford Reach, we modeled the upper and lower segments independently (separate boundary conditions) and then assembled the three segments (sections) into one seamless GIS database. Hydrodynamic modeling conducted by Geist et al. (2000) to develop their habitat model was accomplished using the suite of tools referred to as the Physical Habitat Simulation System (PHABSIM; Milhous 1979; Stalnaker 1979). Results from their simulations included depth and minimum, maximum, and average velocity for the hourly flows that occurred during daylight hours within the peak spawning period at Locke Island.

Spawning Habitat Database Development

To develop our database of spawning conditions, we determined the physical conditions that were present when fall Chinook selected and constructed their redds during fall of 2004. First, we used MASS1 and the 2004 Priest Rapids Dam hydrograph to reconstruct the hourly time series of streamflows that occurred at each spawning site over the 11 days (264 hours) that corresponded to peak spawning. Then we converted depths, velocities, and water surface elevations from River2D flow simulations of these hourly streamflows (40 – 180 kcfs, in 10 kcfs increments) into 4 m resolution grids with regularly spaced cells (16 m2), by linear interpolation to develop a GIS database for habitat modeling. This resulted in 45 unique grids that were composed of the full range of depths, velocities, and water surface elevations that Chinook salmon experienced during the 264-hour peak spawning period. Through a series of spatial queries using these grids, we assembled the time series of depths, velocities, and water surface elevations at each of the analysis cells corresponding to the hourly streamflows at each location.

We generated 16 predictor variables (as grids) from the 30 depth and velocity grids to characterize the hydraulic conditions in the Hanford Reach,. Six predictor variables corresponded to depth and velocity at the minimum, median, and maximum flow conditions during the spawning period. The remaining 10 variables were used to characterize the time series of hydraulic conditions over the full 264 hours (24-hours a day), or 132 hours (daylight hours only). We characterized the range in velocity and depth as well as the variability in the time series' using the coefficient of variation (CV) for depth and velocity. One of our objectives was to develop a metric which could capture the relative stability or persistence of suitable spawning habitat over time, and to evaluate whether fall Chinook spawning site selection is associated with persistent habitat. We developed persistence metrics which were the proportion of hours that velocities were within various ranges that might be suitable for spawning Chinook salmon. Initial exploratory analyses suggested that redd locations were associated with locations where the velocities were frequently greater than 1.0 m/s over the 264-hour time series'. Therefore we developed variables which were the proportion of hours that water velocities exceeded 1.0 m/s. Any dewatering events would also be captured via hourly velocity values of "0" within the time series. The range variables, the CV variables and the velocity proportion variables all characterize the degree of habitat persistence.

We characterized geomorphic features of the Hanford Reach with five predictor variables: bed slope, distance-to-islands, substrate, river section, and river-mile. We calculated riverbed slope with the GRID and SLOPE functions in ARC/INFO, and distance-to-islands with the DISTANCE function (ESRI 1992). The distance-to-islands variable was treated as a static geomorphic variable since we computed it at a 100-kcfs steady-state flow. The DISTANCE variable was intended to be a surrogate for upwelling or downwelling. Hydraulic pressure differentials work to both force water into the substrate (downwelling), and allow it to re-surface as upwelling. These differentials are the result of geomorphic features such as riffles, gravel bars, and islands. In preliminary analyses, we found a consistent association between actual redd locations and the distance (relatively short) to a bar or island. We hypothesized that this association might be a result of downwelling and upwelling associated with the geomorphic feature. The river section and substrate variables were categorical, with three classes each. For river section, values 1 – 3 referred to the upper, middle, and lower sections of the Reach, respectively (Figure 69). For substrate, values 1 – 3 referred to gravel-dominant, cobble-dominant, or other (boulders, sand, silt, etc.), respectively. For both categorical variables, we set the reference category to class 3 so the odds ratio would compute the relative probability of spawning occurrence for classes 1 and 2. At a finer scale than section, the river-mile variable identified the closest river mile location for each cell in the GIS.

Following the development of the 21 candidate predictor variables, we classified each as to whether the variable captured the persistence of the habitat characteristic, the hydraulic conditions, or the geomorphic features of potential spawning habitat. Our classification was used to identify the types of variables (persistence, hydraulic, or geomorphic) that were most strongly associated with differentiating between spawning and non-spawning sites.

Spawning Habitat Model Development

For model development and hypothesis testing, we generated 5,000 random presence/absence points throughout the Hanford Reach with ARC/INFO. We defined a presence site as any randomly selected point that was within a redd cluster, and we defined an absence site as any random point outside a redd cluster. The relative scarcity of fall Chinook redds in the Hanford Reach resulted in 224 of the 5,000 random points falling inside a redd cluster, and 4,776 outside. Our choice of 5,000 random points minimized the possibility of pseudo-replication (sampling a redd more than once) because it produced a nearest neighbor distance of 49 m, which was substantially larger than the size of the redds (17 m2 to 50 m2). We also found that using 5,000 points provided reasonable levels of contrast for each of the explanatory variables considered.

We used a randomized sampling design instead of a case-control design because it allowed us to calculate the probability that a redd would occur at a given location (Keating and Cherry 2004). We attributed the 5,000 random points with the 21 predictor grids using a GIS identity operation. For validation purposes, we generated a complimentary set of 5,000 random points that were not used in model development and attributed each with a presence (n = 216) or absence (n = 4,784) value as described above.

We used cell-based modeling and logistic regression (Hosmer and Lemeshow 2000) to build and test a GIS spawning habitat model for the Hanford Reach. We used logistic regression because it is well suited for the examination of the relationship between a binary response (i.e., the presence or absence of redds) and various explanatory variables. We used the results from the logistic regression model to predict the probability of spawning habitat with the following equation:

$$\mathrm{P}_i = \frac{e^{g(x)}}{1+e^{g(x)}} \qquad \textbf{(Equation 7)}$$

where (P_i) is the probability that a cell location will contain redds and g(x) is the linear combination of parameter estimates obtained from the logistic regression equation. The logistic regression equation can produce a range of probabilities from 0 - 99%, with higher values indicating an increased likelihood of suitable spawning habitat.

We evaluated the strength of the associations between 21 predictor variables and the 5,000 randomly-selected presence/absence sites, one at a time, with the likelihood ratio test statistic (*G*). Larger *G* values indicate larger changes in the log-likelihood and a stronger association between the predictor and the response variable. Following this univariate analysis, we evaluated the predictive capability of different combinations of covariates and spawning occurrence with multivariate logistic regression. Geist et al. (2000) used a similar process to develop their spawning habitat model using multivariate logistic regression. We examined multivariate model performance with three types of criteria: model fit statistics, classification accuracy, and biological plausibility. Model fit statistics included the Akaike's Information Criterion (AIC), the *G* statistic, Nagelkerke's pseudo R^2, and the Hosmer-Lemeshow goodness-of-fit test ($\hat{C}$). The AIC statistic measures the relative degree that the explanatory variables accounted for the variability in redd presence and absence and includes adjustments for the number of parameters that are estimated so as to avoid models with too many parameters or overfitting data (Burnham and Anderson 2002). Lower values of AIC represent better fitting models. The $\hat{C}$ statistic is considered to be a better measure of fit than R^2 because it measures the expected versus actual observations based upon deciles of risk (Hosmer and Lemeshow 2000). The closer $\hat{C}$ is to 1, the better the overall fit of the model. We checked for linearity between the logit and the continuous variables with the Box-Tidwell test (Box and Tidwell 1962). If nonlinearity was observed, we examined model fit by transforming the covariate (e.g., squared, categorical, exponential). Our classification accuracy criteria were the percent of correctly classified presence sites, absence sites and sites overall. These statistics were calculated by estimating the model parameters for a candidate model, applying the fitted model to the estimation data, and summing the number of correct classifications. Finally, to decide between models with similar degrees of fit, we qualitatively assessed the biological plausibility of each model. For this criterion, models with a more understandable biological mechanism were deemed preferable to models without clear biological mechanisms.

GIS Habitat Model Application and Validation

To generate a spatially explicit fall Chinook habitat map of the Hanford Reach for the 2004 peak spawning season, we input our best model into the GRID function in ARC/INFO and populated it with the appropriate GIS layers (hydraulic and geomorphic). This resulted in a probability grid comprised of 16-m^2 cells, with each cell containing a probability of spawning (0 - 77%). The next step was to convert the continuous probability grid into a binary format so that we could explicitly display potential spawning habitat. We overlaid the probability grid and validation data (i.e., the 5,000 randomly-selected points held out of the model fitting exercises) and examined model classification accuracy at three probability cutpoints (5%, 10%, and 15%). These cutpoints were chosen to find the best balance between omission error (finding a redd where no suitable spawning habitat was predicted) and commission error (not finding a redd in predicted suitable habitat) rates. The random point selection method we used to build our model did not necessitate using a 0.5 probability cutpoint that is commonly used in case-control studies. The accuracy statistics we focused upon were overall model accuracy (number of correct predictions / total number of predictions), model sensitivity (percent of presence sites correctly classified), and model specificity (percent of absence sites correctly classified). Once a cutpoint was selected, we reclassified the continuous probability grid into a binary format and calculated the amount of potential spawning habitat. Lastly, to assess the accuracy of the spawning habitat model temporally, we overlaid predicted spawning habitat from the 2004 peak spawning season and 2005 redd clusters obtained from digital orthophotography the following year.

Prospective Evaluation of Alternative Hydrosystem Operations

We conducted prospective spawning habitat simulations for steady- and unsteady-state hydrographs to gain insight into the relative pattern of habitat change under alternative hydrosystem operations. To characterize the relative amount of potential spawning habitat at different flow fluctuations and magnitudes, we compared spawning habitat simulations for three scenarios: (1) the actual 2004 hydrograph for the 11-day peak spawning period, (2) steady-state flows from 60 – 110 kcfs, and (3) unsteady-state hydrographs consisting ±10-, ±20-, and ±30-kcfs sine-wave-shaped flow fluctuations about median flows of 60 – 110 kcfs, in 10 kcfs increments. Flow scenario 1 was achieved when we built and tested the GIS habitat model because we used the actual 2004 hydrograph for the peak spawning period. Our simulated flow fluctuations were simplified approximations in terms of their shape compared to the actual 2004 hydrograph (Figure 70), but we were primarily interested in the relative magnitude of change versus the absolute change in habitat.

Estimating spawning habitat under steady-state flows required that we make six separate GIS model runs, each time resetting the maximum depth variable (DEPTHMAX) to the steady-state flow (e.g., 60, 70 kcfs). In all six steady-state flow simulations, we set the velocity CV variable (VELCV24) to 0 since there were no velocity fluctuations. Similarly, we set the velocity persistence variable (VELPR24) to 1 if the velocity was >1.0 m/s and 0 if the velocity was ≤1.0 m/s since the sites would have experienced those flows 100% of the time.

Estimating spawning habitat under unsteady-state flows required that we run the GIS model 18 times, because we modeled six median flows (60 – 110 kcfs) at 10-kcfs intervals, with three distinct flow fluctuations about each median flow. For example, to estimate the amount of potential spawning habitat at an 80-kcfs median flow, with ±10-, ±20-, and ±30-kcfs flow fluctuations, required we run the GIS habitat model three times, changing the input grids for each flow simulation and then calculating potential spawning habitat. Since we lacked the finer resolution flows (i.e., we only modeled in 10-kcfs increments), we used the maximum, minimum, and median flow grids to approximate the velocity CV (VELCV24) and velocity persistence (VELPR24) variables.

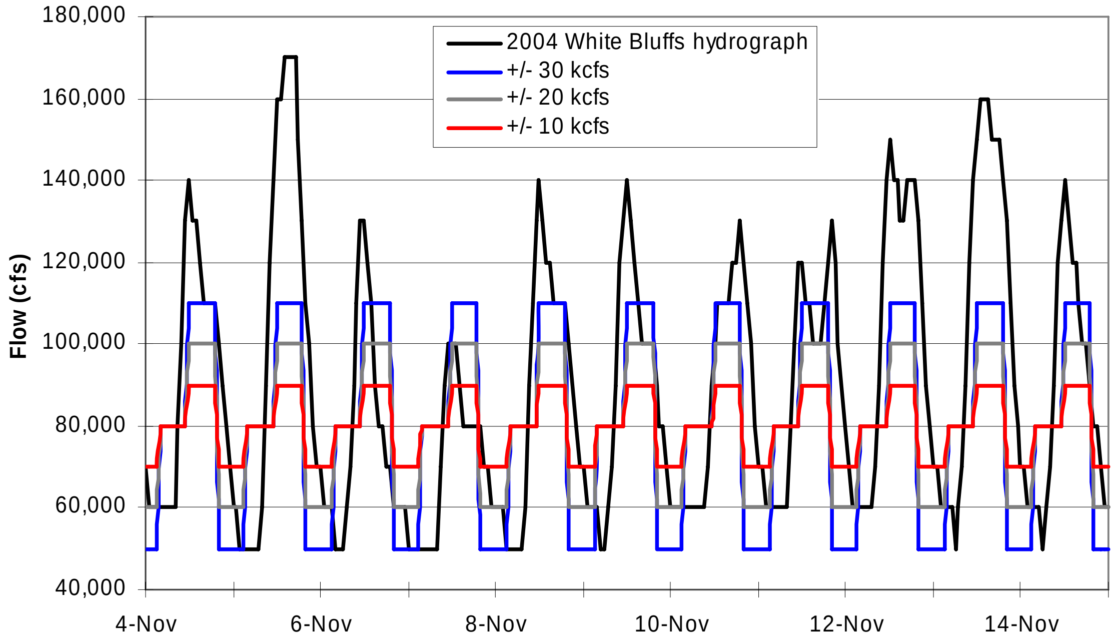


Figure 70. The 2004 White Bluffs hydrograph during the spawning period, along with three simulated unsteady-state hydrographs used in prospective spawning habitat modeling. The simulated hydrographs depicted here represent +/- 10 kcfs, +/- 20 kcfs, and +/- 30 kcfs fluctuations about a median flow of 80 kcfs.

Results

Spawning Distribution

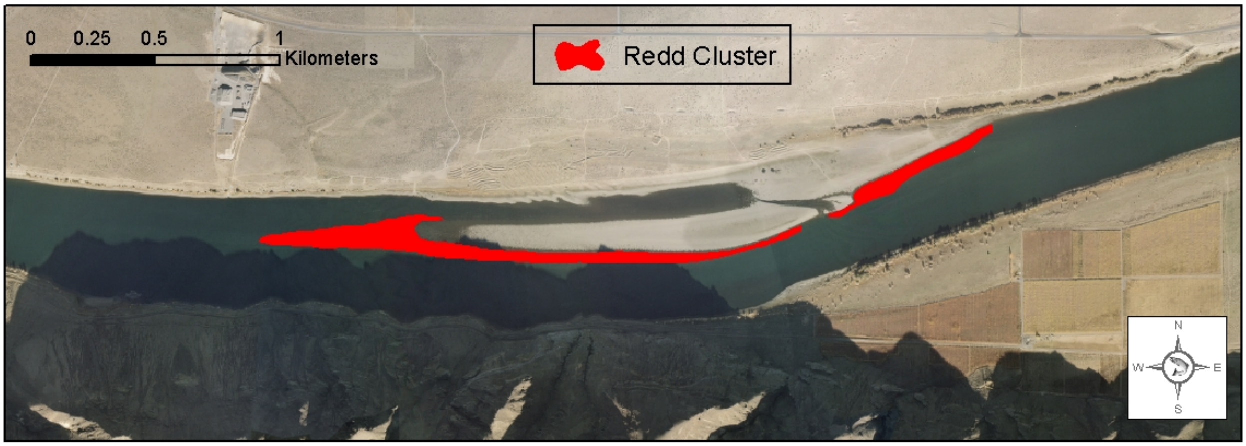
Fall Chinook spawning is concentrated in nine areas scattered throughout the Hanford Reach (Figure 69). In terms of area, the majority of known spawning (redd clusters) in the Reach occurred in the middle segment in 2004 (158.2 ha) from White Bluffs to 100F slough (Table 22). A smaller area of redd clusters occurred in the upper segment (20.6 ha), primarily near Coyote Rapids and at Vernita Bar, and the lower segment near Ringold, Homestead Island, and Wooded Island contained the smallest area of redd clusters (12.7 ha). We identified a total of 30 individual redd clusters for the entire Hanford Reach from 2004 images. Cluster sizes ranged from 0.3 to 52.0 ha and averaged 6.4 ha. No redd clusters were identified outside of the 10 known spawning areas previously identified by Dauble and Watson (1990). Aerial surveys conducted to determine peak redd counts in 2004 indicated that 29% of the redds were located in the upper segment, 67% were located in the middle segment, and 4% were located in the lower segment (Table 22) (see Mueller 2004 for additional information). This distribution was similar to the 1948-1992 averages of 39%, 59%, and 2%, respectively for the same areas as reported by Dauble and Watson (1997).

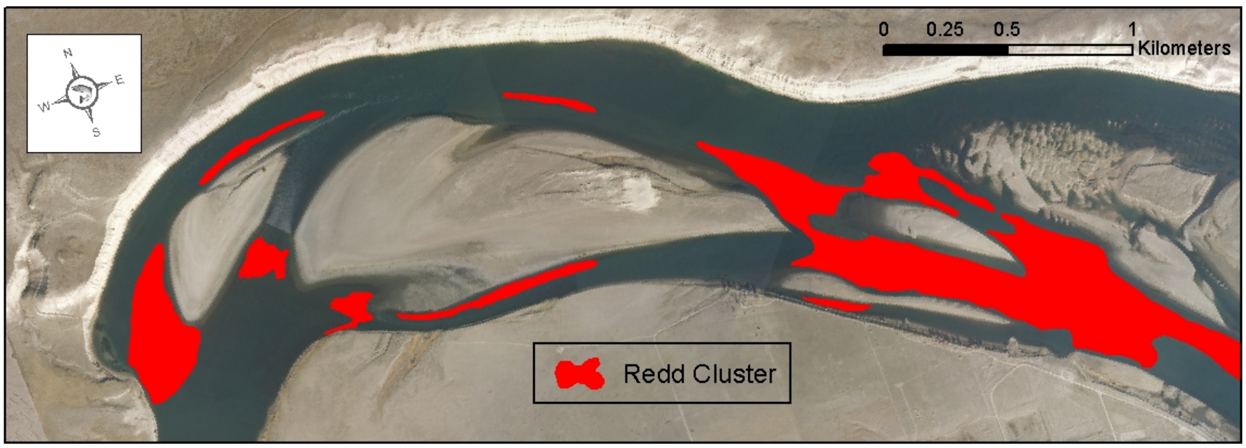
The distribution of spawning area (clusters) among the three segments in 2005 was similar to 2004 (Table 22). However, based on aerial surveys during the spawning season (Mueller 2005) a slightly smaller proportion of redds occurred in the upper segment near Vernita Bar, and a larger proportion of redds occurred in the lower segment.

Table 22. Relative distribution of redd clusters (area) and the number of redds across the three segments of the Hanford Reach in 2004 and 2005.

Segment	Percent of total redd area (clusters) from Orthophotography		Percent of total number of redds from aerial surveys	
	2004	2005	2004	2005
Upper	11%	11%	29%	22%
Middle	82.5%	79%	67%	69%
Lower	6.5%	10%	4%	9%

Each of the key spawning areas is characterized by a complex channel structure (Figure 71) although channel configurations vary in complexity, slope profile, depth, and velocity distributions for a given flow. Channel form in the upper segment near Vernita Bar is narrow and constrained on the south shore by bedrock cliffs, with a single lateral gravel bar adjacent to the north shore. In contrast, the river takes a 90 degree turn in the middle segment where it is constrained on the north shore by the White Bluffs. The energy dissipation effect of this feature over geologic time resulted in the deposition of alluvial material across a wide, shallow channel and the formation of the gravel-bar islands around the White Bluffs corner. In the lower segment, the combination of a somewhat flatter gradient along with the backwater effect from McNary reservoir results in a relatively wide river channel with numerous islands formed from alluvial material and over-topped with fine materials and vegetation.





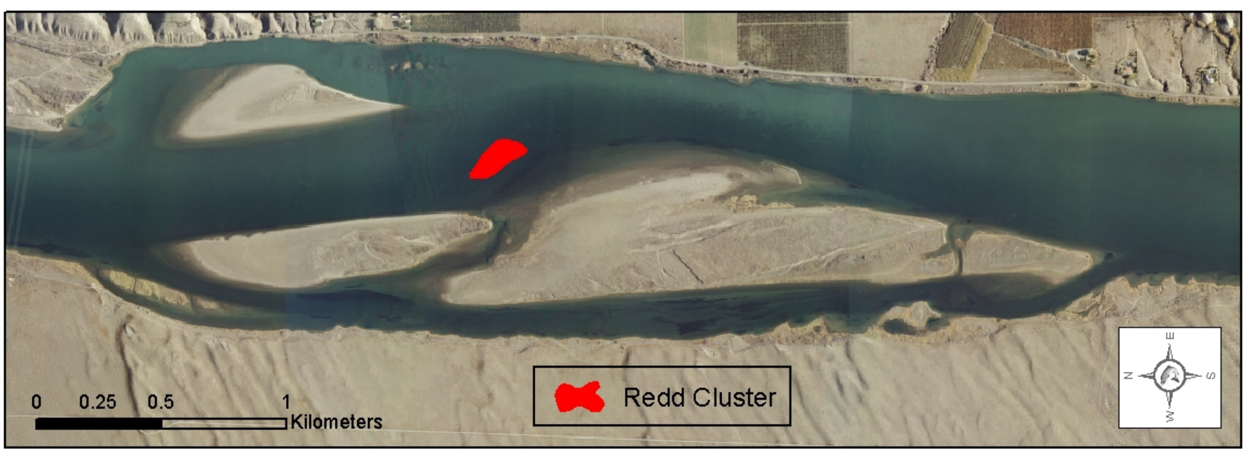


Figure 71. Variation in river channel morphology in the Hanford Reach between the lower segment at Wooded Island (bottom plate), the middle segment at White Bluffs (middle plate), and the upper segment at Vernita Bar (upper plate).

Different channel morphologies resulted in site-specific differences in redd distribution patterns based on aerial photos from 2004. Spawning areas in the upper segment near Vernita Bar were limited to narrow bands, while spawning areas in the middle segment near White Bluffs were characterized by wide expanses of redd locations (Figure 72). Redds were more closely spaced and individual redds (including inter-redd spacing) were smaller (50-60 m^2), at Vernita Bar. Redd density was lower and redds were larger in the middle segment near White Bluffs. Redd sizes including inter-redd spacing averaged 118 m^2, within the middle segment, which is similar to the redd size (including inter-redd spacing) range cited by Hanrahan et al. 2004 from Hanford Reach data (83 to 117 m^2). A comparison of the distribution of redd (cluster) area among segments, and the total number of redds counted in each segment (Figure 72) also suggests a higher density per unit area in the upper segment relative to the middle segment. These patterns indicate that suitable, persistent spawning conditions occur over a relatively smaller area in the upper segment as compared to the middle segment and that some degree of redd superimposition may be occurring at Vernita Bar.

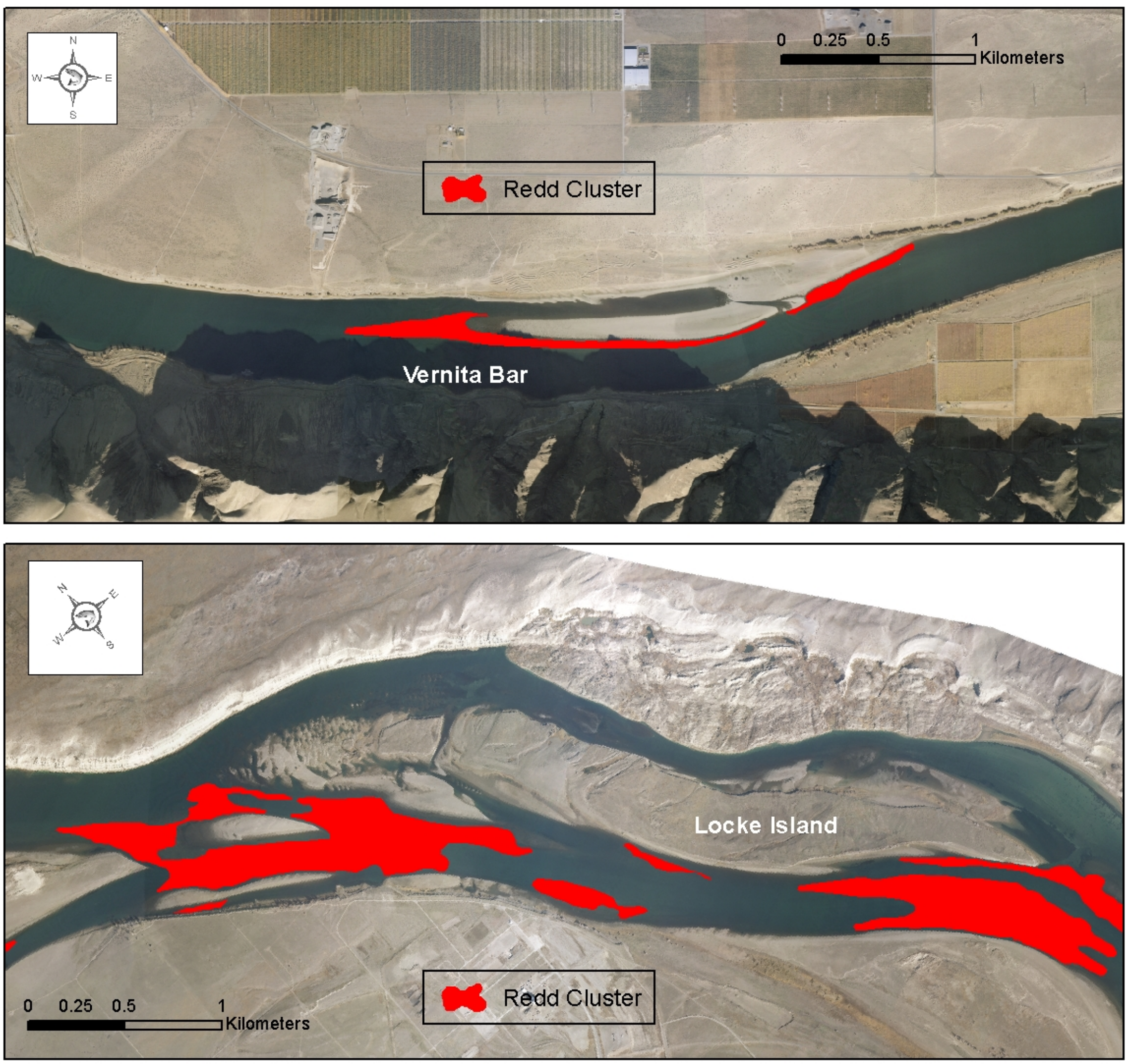


Figure 72. Locations and spatial distribution of spawning areas (red shading) near Locke Island and Vernita Bar digitized from orthophotos taken in 2004.

Physical Variables Associated with Spawning Habitats

Using univariate logistic regression analyses, we determined which variables describing physical conditions in the Reach were most strongly associated with spawning habitats (Table 23). Seven of the top eight predictor variables we examined in our analysis were related to water velocity. Of those variables, our persistence variable (VELPR24), which was the proportion of hours that water velocities at a site were >1.0 m/s on a 24-hour basis, had the largest *G* statistic and a positive coefficient followed by the same variable on a 12-hour basis (VELPR12). The positive signs of the estimated coefficients for these two variables indicate that the probability of spawning increased as the proportion of time that velocities were > 1.0 m/s increased. The median velocity (VEL80), which we classified as a hydraulic variable, was almost as significant as VELPR12.

The variables which characterized the range and CV of depths and velocities measured the instability, or lack of persistent conditions, at redd presence and absence sites. All eight of these persistence variables had negative estimated coefficients. The negative signs indicate that as the range or CV of depth and velocity increases, the likelihood of spawning decreases, suggesting that a site's suitability for spawning diminished as flow variability increased. Of these variables, the velocity CV on a 24-hour basis (VELCV24) had the largest *G* statistic. The increase in depth and velocity variation indicated by these metrics is directly related to the increase in streamflow variation as the hourly flow pattern moves from constant under steady flow conditions to highly variable under a typical load-following hydrograph.

Of the five pairs of variables that we used to characterized 24-hour (day/night) versus 12-hour (daytime) hydraulic conditions, four had *G* statistics which indicated that 24-hour flows were more influential for determining site suitability than daytime flows alone. All five of the geomorphic variables contained negative coefficients. Thus, site suitability decreased as the slope increased, distance to an island increased, or as the dominant substrate class increased from gravel to cobble, and cobble to boulder. Since all the univariate tests resulted in *p*-values <0.25, all 21 variables were tested in the multivariate models (Hosmer and Lemeshow 2000).

Table 23. Univariate logistic regression results for 21 predictor variables in the Hanford Reach (n = 5,000). β is the estimated coefficient. The likelihood ratio test statistic (*G*) is calculated as -2(change in log likelihood) for the constant-only model versus the full model (constant and predictor variable). Larger *G* values indicate a stronger association between the predictor and response variable (presence of redds). Class represents our assessment of whether the variable captures the persistence of the habitat characteristic, hydraulic conditions, or geomorphic features of potential spawning habitat.

Variable	β	*G*	*p-value*	Description	Class
VELPR24	2.722	178.554	<0.001	proportion of day-night hours > 1m/sec	persistence
VELPR12	2.292	167.102	<0.001	proportion of daytime hours > 1m/sec	persistence
VEL80	1.327	166.403	<0.001	velocity - median	hydraulic
VELCV24	-4.8	156.938	<0.001	CV of velocity (24 hrs)	persistence
VEL50	1.227	136.796	<0.001	velocity - minimum	hydraulic
SUBSTRATE	-1.483	122.7	<0.001	substrate - (3 size classes)	geomorphic
VELCV12	-3.359	125.014	<0.001	CV of velocity (daytime)	persistence
VEL170	0.971	81.369	<0.001	velocity - maximum	hydraulic
DISTANCE	-0.001	74.81	<0.001	distance from Islands	geomorphic
DEPTH50	-0.198	39.874	<0.001	depth - minimum	hydraulic
DEPTHMAX	0.175	39.388	<0.001	depth - maximum	hydraulic
DEPTH80	-0.171	35.096	<0.001	depth - median	hydraulic
DEPCV12	-0.897	24.742	<0.001	CV of depth (daytime)	persistence
RIVERMILE	-0.022	18.625	<0.001	location within reach	geomorphic
DEPCV24	-0.92	15.493	<0.001	CV of depth (24 hrs)	persistence
SLOPE	-5.19	11.845	0.003	lateral slope (percent)	geomorphic
VELRANGE24	-0.543	10.933	0.001	velocity range (24 hours)	persistence
DEPRANGE24	-0.212	4.069	0.039	depth range (24 hours)	persistence
VELRANGE12	-0.351	3.847	0.05	velocity range (daytime)	persistence
SECTION	-0.169	3.661	0.056	river section - reach	geomorphic
DEPRANGE12	-0.191	2.821	0.087	depth range (daytime)	persistence
CONSTANT	-4.511		<0.001		

Multivariate Logistic Regression

We developed and evaluated the fit of 10 multivariate logistic regression models (Table 24). We refer to model 1 as the full model because it contained the greatest number of predictor variables, and the remaining models had one or more variables removed from the full model (with the exception of model 10 developed by Geist et al. 2000). Results presented in Table 24 represent the effect of removing one or more geomorphic variables from the full model. Models 1 and 4 had the highest Nagelkerke psuedo R^2 statistic (0.48). In terms of the R^2 statistic, there was little difference between models 1, 4, 5, and 7, as all had R^2 values between 0.45 and 0.48. A second group of models was not far behind, with R^2 values between 0.4 – 0.45 (models 2, 3, 8, and 9). Only models 6 and 10 had R^2 values <0.4; model 6 had all geomorphic variables removed (Table 24), while model 10 contained a velocity, slope, and depth variable. According to the *AIC* statistic, model 1 achieved the best fit (i.e., lowest *AIC* value), though models 4, 5, and 7 also performed well. The correct classification results did not aide in model selection, as all of the models had overall classification accuracies between 87.5 – 90.4%.

The Hosmer-Lemeshow goodness-of-fit statistic ($\hat{C}$) was more useful for model evaluation because values ranged from 0.02 - 1.0 (Table 24). The difference in the $\hat{C}$ statistic was especially apparent between models 1 and 2 (0.97 versus 0.11, respectively), both of which had similar R^2 values (0.48 and 0.44, respectively). Model 10 had the lowest $\hat{C}$ statistic (0.02) and model 7 the highest (1.0). However, when we used the velocity at the median flow of 70 kcfs, instead of the velocity at the mean daytime flow of 80 kcfs in model 10, the $\hat{C}$ statistic increased to 0.82, demonstrating the sensitivity of this statistic. Models 1, 5, and 7 demonstrated similar performance in terms of the $\hat{C}$ statistic (0.97, 0.98, and 1.00, respectively).

The effects of removing different geomorphic variables from the full model are indicated by the model fit statistics for models 2-9 in (Table 24). These results suggest that the SECTION and SUBSTRATE geomorphic variables are important for achieving good model fit to the data. Removal of the DISTANCE variable did not appreciably affect model fit by itself, and removal of the SLOPE variable reduced model fit somewhat, although not as significantly as removal of the SECTION and SUBSTRATE variables. Models 4, 5, and 7 illustrate the effect of removing the SLOPE or DISTANCE variables individually, or together. Model 4 without SLOPE still fit the data reasonably well, although the $\hat{C}$ statistic was reduced, and the difference between models 5 and 7 with and without SLOPE was minimal.

Table 24. Results of a multivariate logistic regression analysis in the Hanford Reach (*n* = 5,000). We compared accuracy and fit of a full model (Model 1) with eight nested models (models 2-9). In contrast, Model 10 was built by fitting the set of covariates that were found to be important in a previous study at Locke Island (Geist et al. 2000). To calculate accuracy statistics, we set the probability cutpoint for all models at 10% (cells with a probability value ≤10% were considered unsuitable for spawning, values >10% were considered suitable).

Model[a]	R^{2b}	*AIC*	$\hat{C}^c$	Present[d]	Absent[e]	Overall[f]	Description[g]
1	0.48	1062.2	0.97	77.4	90.8	90.2	
2	0.44	1138.8	0.11	72.6	89.8	89.0	section
3	0.43	1152.5	0.88	77.0	89.9	89.3	substrate
4	0.48	1067.6	0.66	77.9	90.5	89.9	slope
5	0.47	1077.3	0.98	77.0	91.0	90.4	distance
6	0.35	1271.7	0.70	72.1	88.2	87.5	slope, substrate, section, distance
7	0.46	1086.5	1.00	77.0	90.7	90.1	slope, distance
8	0.42	1163.6	0.63	70.4	89.4	88.6	slope, distance, section
9	0.41	1175.7	0.69	76.1	89.8	89.2	slope, distance, substrate
10[h]	0.32	1327.6	0.02	73.9	88.5	87.8	Geist et al. (2000)

Notes:

[a] Model 1 is a full model that was comprised of 4 geomorphic, 3 hydraulic, and 2 persistence variables (see Table 25)

[b]Nagelkerke pseudo R^2 statistic

[c]Hosmer-Lemeshow goodness-of-fit statistic

[d]Percent of presence sites correctly classified (sensitivity)

[e]Percent of absence sites correctly classified (specificity)

[f]Overall model accuracy (#correct predictions / total # of predictions)

[g]Variable removed from Model 1

[h]Battelle model (Geist et al., 2000). Velocity at the mean daytime flow was used for Locke Island (80 kcfs). The $\hat{C}$ statistic increased to 0.82 when the velocity at the median flow was used (70 kcfs).

Based on these measures of fit, we concluded that the best models were models 1, 5, and 7. These models had high $\hat{C}$ statistics, low *AIC* statistics, and high Nagelkerke psuedo R^2 statistics. However, when we examined the slope variables in model 1 (Table 25), we found that the estimated coefficients were biologically implausible. Model 1 had two slope terms with opposing signs, however, the magnitude of the coefficients indicated that the probability of finding a redd increased until slope exceeded about 15%. In other words, the coefficients suggested that the probability of spawning increased with increasing slope up to about 15%, which we felt was biologically implausible and inconsistent with past studies and our observations of redd site slope in the field. Since model 7 (10 parameters) was more parsimonious than either model 1 (13 parameters) or model 5 (12 parameters), we used it for all subsequent spawning habitat modeling. Model details are presented in Table 25 for the full model 1 and the selected model 7.

Model 7 contained two geomorphic variables (SECTION and SUBSTRATE), two velocity persistence variables (VELPR24 and VELCV24), and two depth hydraulic variables (DEPTHMAX and DEPTHMAX^2). In contrast to the univariate analysis, in the multivariate models the geomorphic variables affected the log-likelihood more than the velocity variables. Backward stepping revealed that SUBSTRATE had the largest *G* statistic (93.2), followed by SECTION (81.2), VELPR24 (67.5), DEPTHMAX^2 (63.9), DEPTHMAX (34.7), and VELCV24 (8.2). Similar to the univariate analysis, VELPR24 had a positive coefficient and VELCV24 a negative coefficent. There was evidence for nonlinearity in the 2 velocity variables, but we did not achieve noticeable improvement in model fit with various transformations. A relative interpretation of the odds ratios (*exp(B)*) for the 2 categorical variables (SECTION and SUBSTRATE) found section 1 was more likely to contain a redd than section 3, but not as likely as section 2. Also, redds were most likely to occur in a gravel-dominant substrate class, as compared to the reference category (e.g., boulders, silt, sand, etc.), and next most likely in a cobble-dominant substrate class. The two depth variables had opposing signs, indicating that water depth (DEPTHMAX) increased the likelihood of spawning activity until it became too deep to be suitable. In no case did interaction terms improve the $\hat{C}$ statistic, so we did not include them in the model.

Table 25. Two of the multivariate models we developed and tested for the Hanford Reach (n = 5,000). Model 1 is the full model and model 7 was judged to have the best fit, and was subsequently used for habitat modeling. The SECTION and SUBSTRATE variables were categorical with the 3rd class set as the reference category.

Model	Covariate	*B*	S.E.	Wald	df	*p*-value	*Exp(B)*	95.0% C.I.for EXP(B)	
								Lower	Upper
1	SECTION (3)			49.745	2	0			
1	SECTION (1)	1.754	0.43	16.658	1	0	5.776	2.488	13.409
1	SECTION (2)	2.426	0.357	46.157	1	0	11.314	5.619	22.781
1	DISTANCE	-0.001	0	14.582	1	0	0.999	0.999	1
1	SLOPE	13.108	4.416	8.81	1	0.003	492706	85.8	282831703
1	SLOPE^2	-46.78	19.535	5.734	1	0.017	0	0	0
1	DEPTHMAX	3.537	0.648	29.792	1	0	34.379	9.652	122.447
1	DEPTHMAX^2	-0.473	0.072	43.575	1	0	0.623	0.541	0.717
1	VELPR24	2.887	0.394	53.759	1	0	17.941	8.292	38.817
1	VELCV24	-2.021	0.713	8.035	1	0.005	0.132	0.033	0.536
1	SUBSTRATE (3)			74.107	2	0			
1	SUBSTRATE (1)	3.221	0.458	49.456	1	0	25.059	10.211	61.497
1	SUBSTRATE (2)	1.928	0.442	19.042	1	0	6.874	2.892	16.339
1	CONSTANT	-13.92	1.656	70.656	1	0	0		
7	SECTION			52.546	2	0			
7	SECTION (1)	1.447	0.422	11.767	1	0	4.249	1.859	9.710
7	SECTION (2)	2.365	0.354	44.44	1	0	10.644	5.310	21.334
7	SUBSTRATE (3)			76.491	2	0			
7	SUBSTRATE (1)	3.106	0.453	47.006	1	0	22.335	9.191	54.279
7	SUBSTRATE (2)	1.783	0.438	16.551	1	0	5.947	2.519	14.039
7	DEPTHMAX	3.257	0.644	25.569	1	0	25.965	7.348	91.752
7	VELPR24	2.788	0.386	52.022	1	0	16.245	7.616	34.651
7	VELCV24	-1.904	0.716	7.081	1	0.008	0.149	0.037	0.605
7	DEPTHMAX^2	-0.445	0.071	39.375	1	0	0.641	0.557	0.736
7	CONSTANT	- 12.96	1.646	61.954	1	0	0		

GIS Habitat Model Validation and Application

After running model 7 in a cell-based modeler (ARC/GRID), we examined model accuracy at three probability cutpoints (5%, 10%, and 15%; [Figure 73]) and with a receiver operating characteristic (ROC) curve (Figure 74). We selected a 5% cutpoint for the calculation of spawning habitat Reach-wide because it balanced model specificity and sensitivity (Figure 73), and produced 85.1% overall accuracy with our independent validation dataset (n = 5,000). The ROC curve generated from model 7 shows the tradeoff between model sensitivity and specificity at all possible cutpoints (Figure 74). Effectively, the lower we dropped the probability cutpoint, the fewer redds were missed by the model (omission), but the greater the model over-estimated habitat (commission). However, the high ROC score (0.94) produced by the GIS-based model at all possible cutpoints demonstrates that it had a phenominal ability to differentiate between use and nonuse sites in the Hanford Reach (Hosmer and Lemeshow 2000). At a 5% cutpoint, the GIS-based model classified 15.4% (712.5 ha) of the Hanford Reach (4,614 ha) as potential fall Chinook spawning habitat in 2004 (Figure 75). The distribution of simulated spawning habitat closely matched the distribution of observed spawning habitat (redd clusters) across the Reach, with 13%, 83% and 4% of the predicted habitat located in the upper, middle, and lower segments, respectively (Figure 76, Table 22).

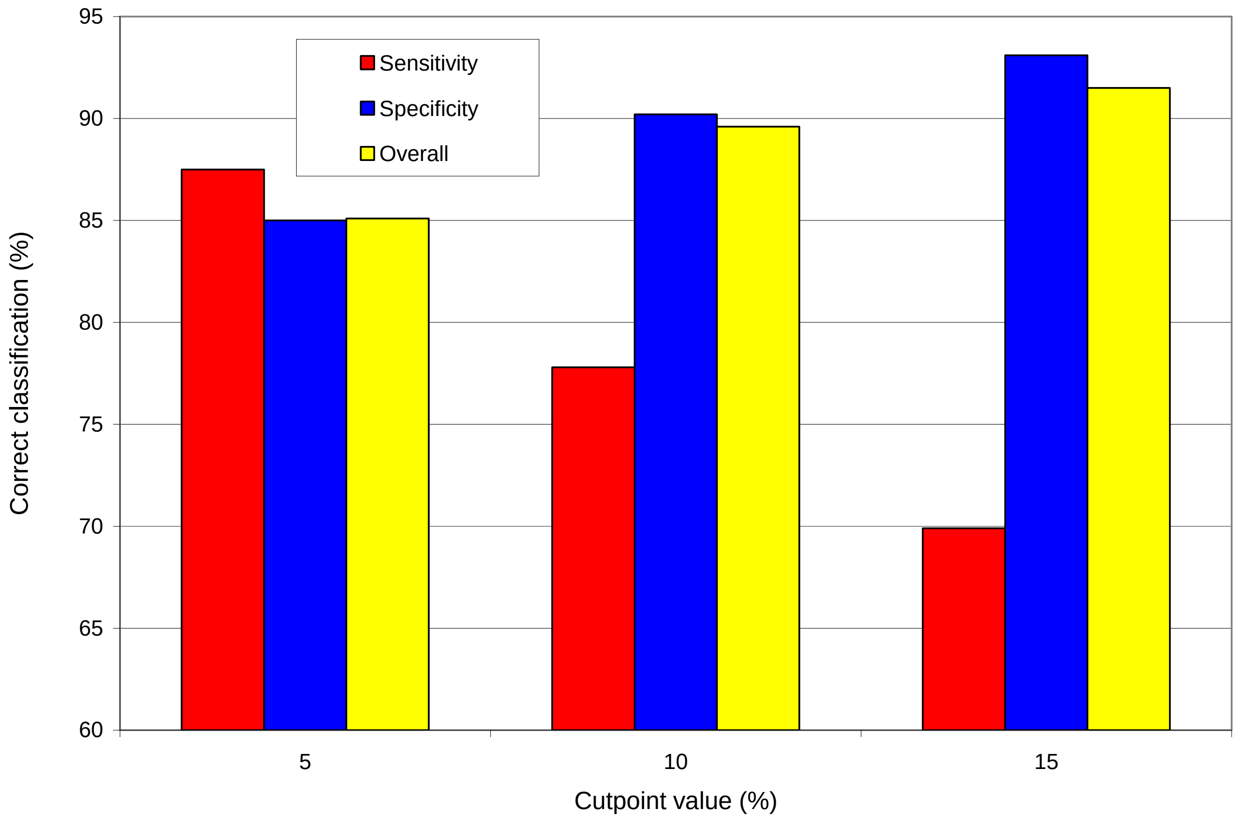


Figure 73. Accuracy of the GIS spawning habitat model over the entire Hanford Reach at 5%, 10%, and 15% probability cutpoints. We determined model accuracy by overlaying 5,000 randomly selected points (use/nonuse) not used in model development and binary grids output by the model at different probability cutpoints. Sensitivity = % redds correctly classified; Specificity = % nonuse sites correctly classified; Overall accuracy = #correct / n.

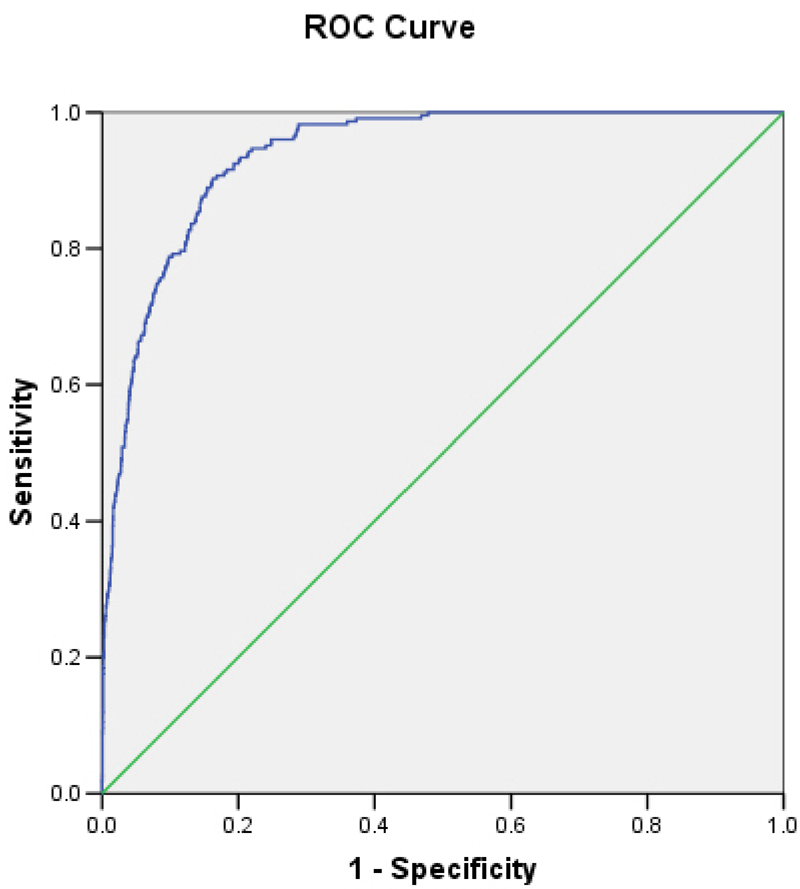


Figure 74. A receiver operating characteristic (ROC) plot calculated for the entire Hanford Reach (n = 5,000) by application of model 7 (see Table 25**). The null hypothesis is that the area under the curve is 0.5, but the actual area was 0.94 (P<0.001, SE = 0.006, 95% CI = [0.93 - 0.95]). Sensitivity is the percentage of presence sites (contained a redd) that were correctly classified, and 1 - specificity is commission error, or the percentage of absence sites that the GIS habitat model falsely predicted to be suitable.**

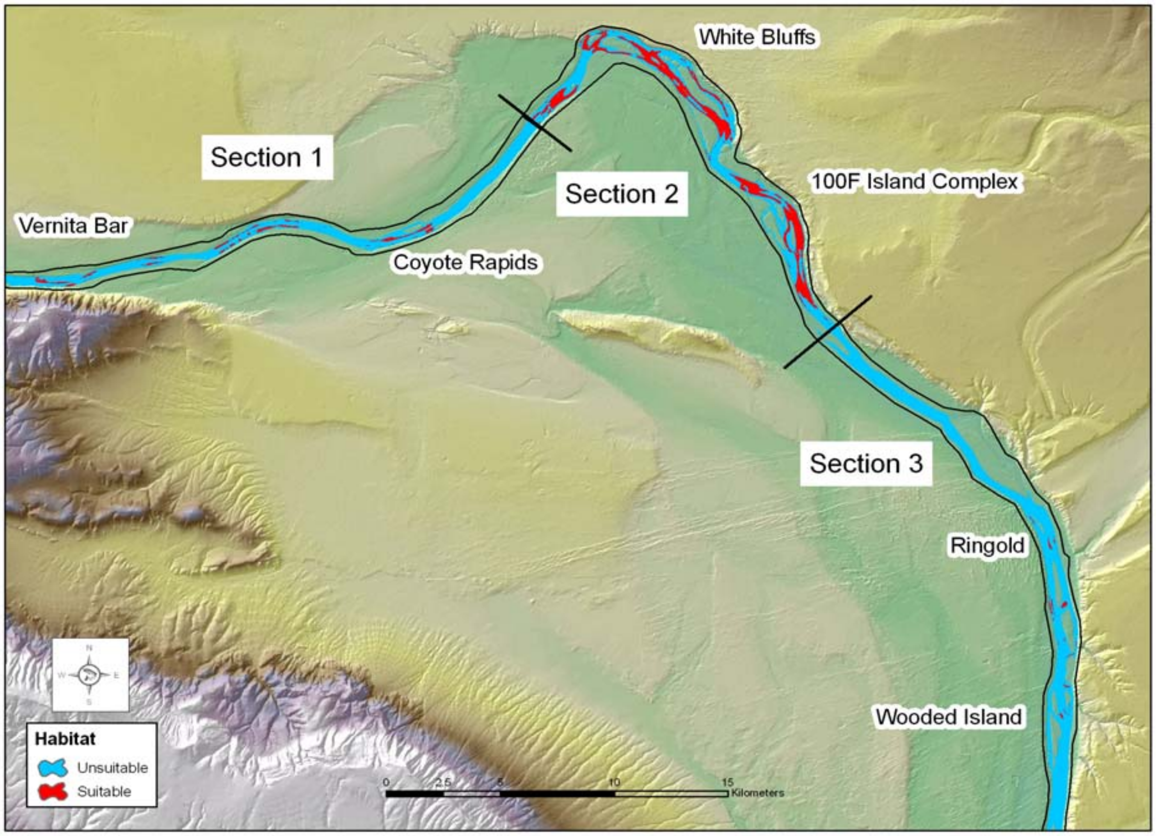


Figure 75. Distribution of predicted spawning habitat in 2004 using a 5% cutpoint and applying model 7. The model achieved an overall accuracy of 85%.

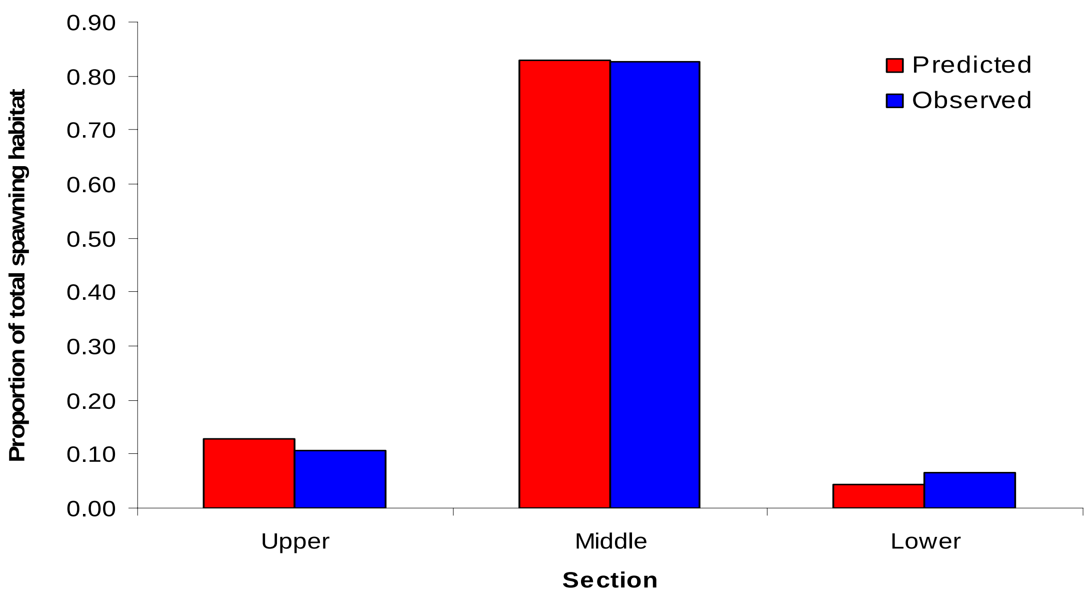


Figure 76. Predicted and observed proportion of spawning habitat area in the upper, middle, and lower section of the Hanford Reach in 2004.

We used orthophotos taken during peak spawning in 2005 to conduct a qualitative evaluation of our spawning habitat model across years. Although we did not model the actual 2005 hydrograph, operations leading up to peak spawning over each 24-hour period were very similar between years, and we assumed that the predicted habitat from 2004 would also be similar to habitat predicted using the actual 2005 hydrograph. In 2005, there were 242.4 ha of fall Chinook redd clusters in the Hanford Reach, a 26.6% increase in area used over 2004. When we overlaid the 2004 predicted spawning habitat and the 2005 redd clusters, 79.8% of them were correctly classified, with 20.2% omission, 11.9% commission, and 86.2% overall accuracy. This accuracy level was very similar to that achieved in 2004 (85.1% overall). Considering that the flow volumes and the degree of fluctuations leading up to peak spawning were similar between 2004 and 2005, it was not surprising that the fall Chinook spawned in similar locations, and that the relative distribution of redd clusters was similar between years (Figure 77, Table 22). There were, however, several areas that were used in 2005, that were not used in 2004. Figure 78 and Figure 79 show areas in the lower and middle sections of the Reach that were predicted as suitable during 2004, but no redd clusters were observed. It is significant to note that these "new" 2005 redd clusters overlay the areas that our model predicted to be suitable habitat in 2004. Because of the similarities between the 2004 and 2005 hydrographs, we expect this same area would have been predicted as suitable using hydraulic and persistence conditions from 2005.

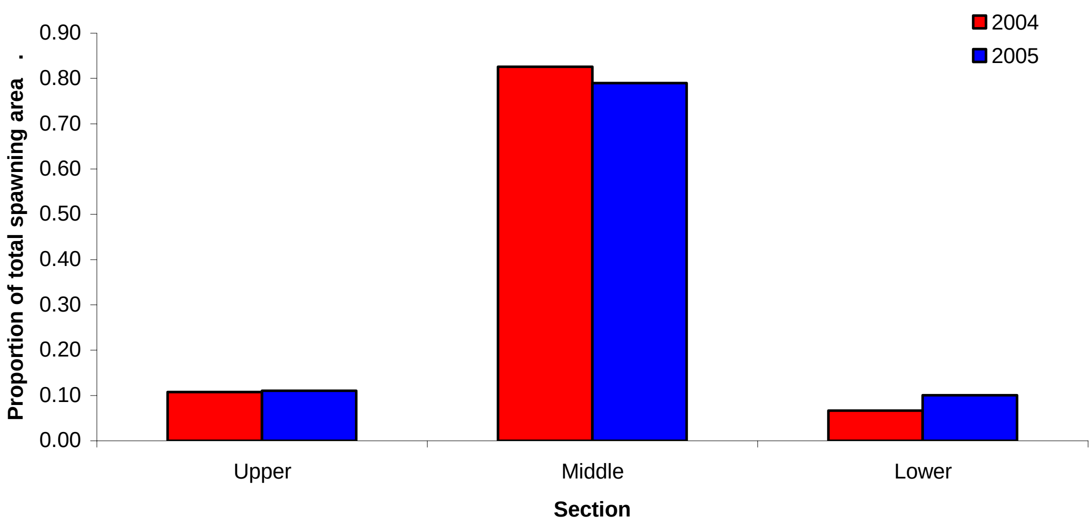


Figure 77. Observed proportions of total spawning area (redd clusters) by section for 2004 and 2005.

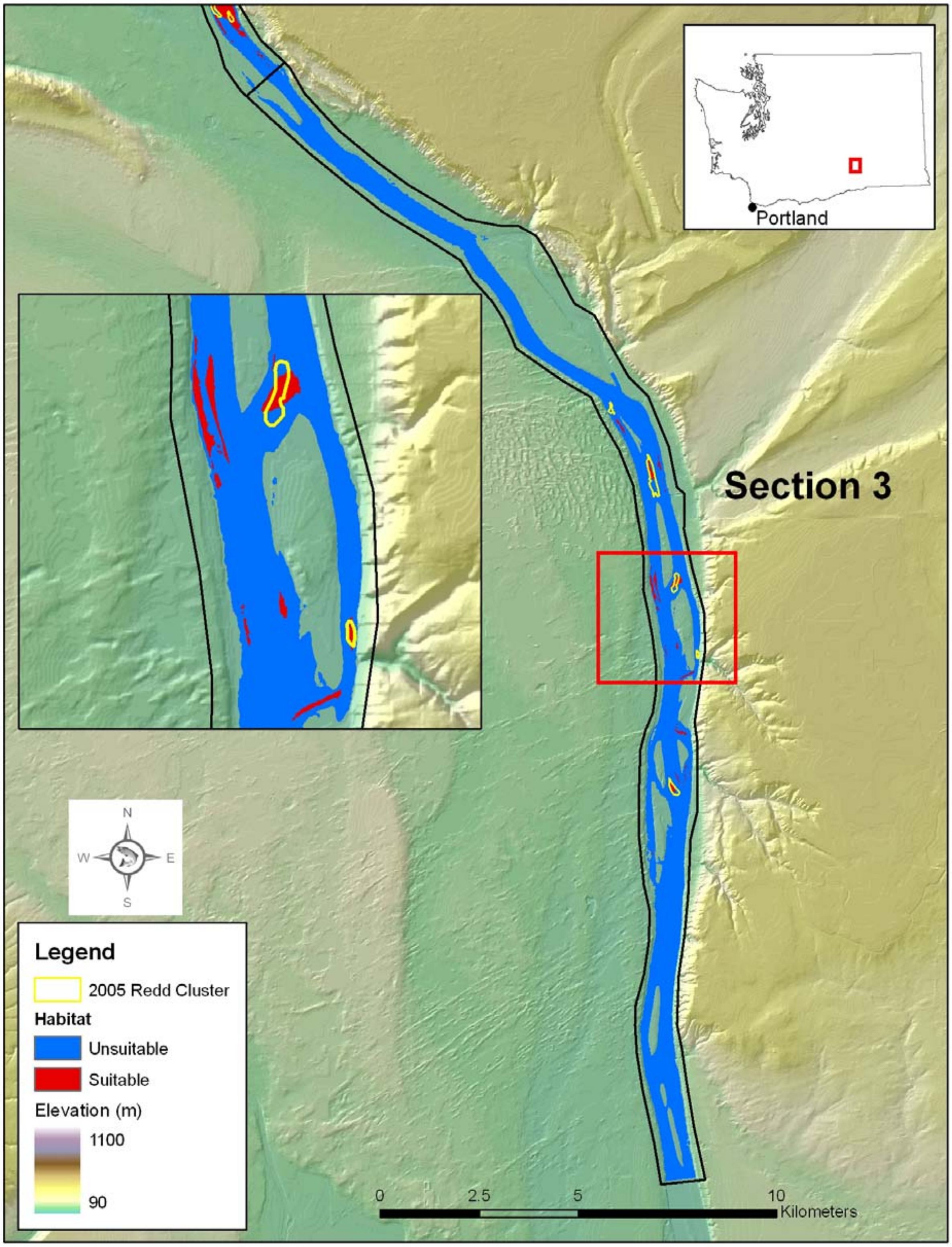


Figure 78. Lower section of the Hanford Reach showing 2005 redd clusters in an area predicted as suitable habitat in 2004, but not actually used during 2004.

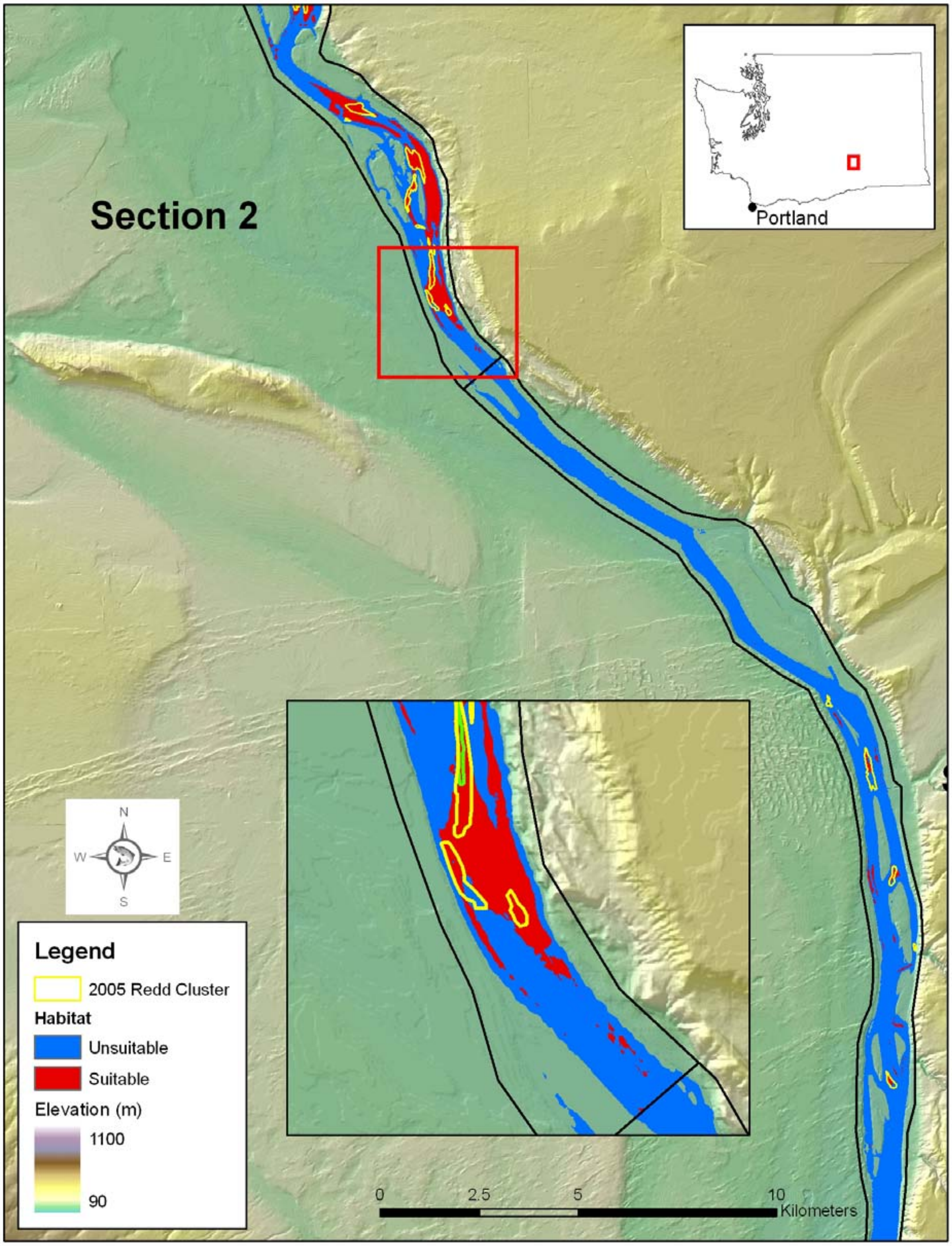


Figure 79. Middle section of the Hanford Reach near 100-F slough showing 2005 redd clusters in an area predicted as suitable habitat in 2004. The small green polygon in the inset was a 2004 redd cluster.

Simulated Alternative Operations

We applied model 7 to 23 hypothetical hydrographs to simulate the relative change in spawning habitat that would occur under alternative operational scenarios. The scenarios included steady-state flows ranging from 60 to 110 kcfs in 10 kcfs increments, and unsteady-state flows of ±10, ±20, and ±30 kcfs (fluctuation ranges of 20, 40, and 60 kcfs, respectively), around each of the steady flows (Figure 80). The 2004 observed hydrograph had a median flow of approximately 80 kcfs, with fluctuations down to 50 kcfs and up to 170 kcfs (a fluctuation range of 120 kcfs).

Under steady flows, predicted spawning habitat increased with increasing flows until leveling off between 100 and 110 kcfs. This pattern indicated that channel morphology may limit the rate of further development of potential spawning habitat within the Hanford Reach at flows beyond 100 or 110 kcfs. For all of the hydrographs we modeled, predicted spawning habitat increased when the amplitude of flow fluctuations decreased (Figure 80). For example, if the fluctuations around the median flow that occurred in 2004 (80 kcfs) had been limited to a range of 60 kcfs (i.e. 50-110 kcfs), rather than the range of 120 kcfs that actually occurred, our model results suggest that spawning habitat area would have increased by 36%. These patterns were similar within each of the three sections of the Hanford Reach. Our model also suggests that spawning habitat area would increase by 21% over the amount produced by the 2004 hydrograph if the fluctuation range had been limited to 40 kcfs around an average flow of 60 kcfs (i.e. 40-80 kcfs). In addition, the 21% increase in habitat over the 2004 hydrograph using an average flow of 60 kcfs would have required substantially less water. The following calculations illustrate the water savings that would have resulted if the 60 kcfs average flow operation had been implemented rather than the 80 kcfs average flow operation that was actually implemented in 2004 for the two weeks prior to peak spawning:

1 kcfs/day = 1,983 acre-ft/day
80 kcfs average flow - 60 kcfs average flow = 20 kcfs
20 kcfs/day X 1,983 acre-ft/day = 39,660 acre-ft/day
14 days X 39,660 acre-ft/day = 555,240 acre-ft

It is interesting to note that this volume of water is equivalent to 11% of the total storage capacity of the Grand Coulee Reservoir.

All 17 of the unsteady-state hydrographs we evaluated with our GIS habitat model produced more habitat than the actual 2004 hydrograph, but less than what the model produced for each of the six steady-state hydrographs. The larger the flow fluctuation we modeled around each steady-state flow, the less predicted spawning habitat the GIS model produced. The inverse relationship between the magnitude of the flow fluctuation and the predicted spawning habitat was consistent with the negative coefficient of the VELCV24 variable, and the positive coefficient of the VELPR24 variable (Table 25). Steady-state flows produced the most predicted spawning habitat because there were no flow fluctuations, which neutralized the negative impact of the VELCV24 coefficient and increased the persistence effect captured by the VELPR24 variable.

Figure 81 shows spawning habitat coverages near Vernita Bar comparing habitat predicted using the 2004 hydrograph with an 80 kcfs steady flow, 80 kcfs with a ±30 kcfs fluctuation (50-110 kcfs), 60 kcfs steady flow, and 60 kcfs with a ±20 kcfs fluctuation (40-80 kcfs). Relative increases in habitat area reflect the values from the histogram in Figure 80 that result from the reduced level of flow fluctuations.

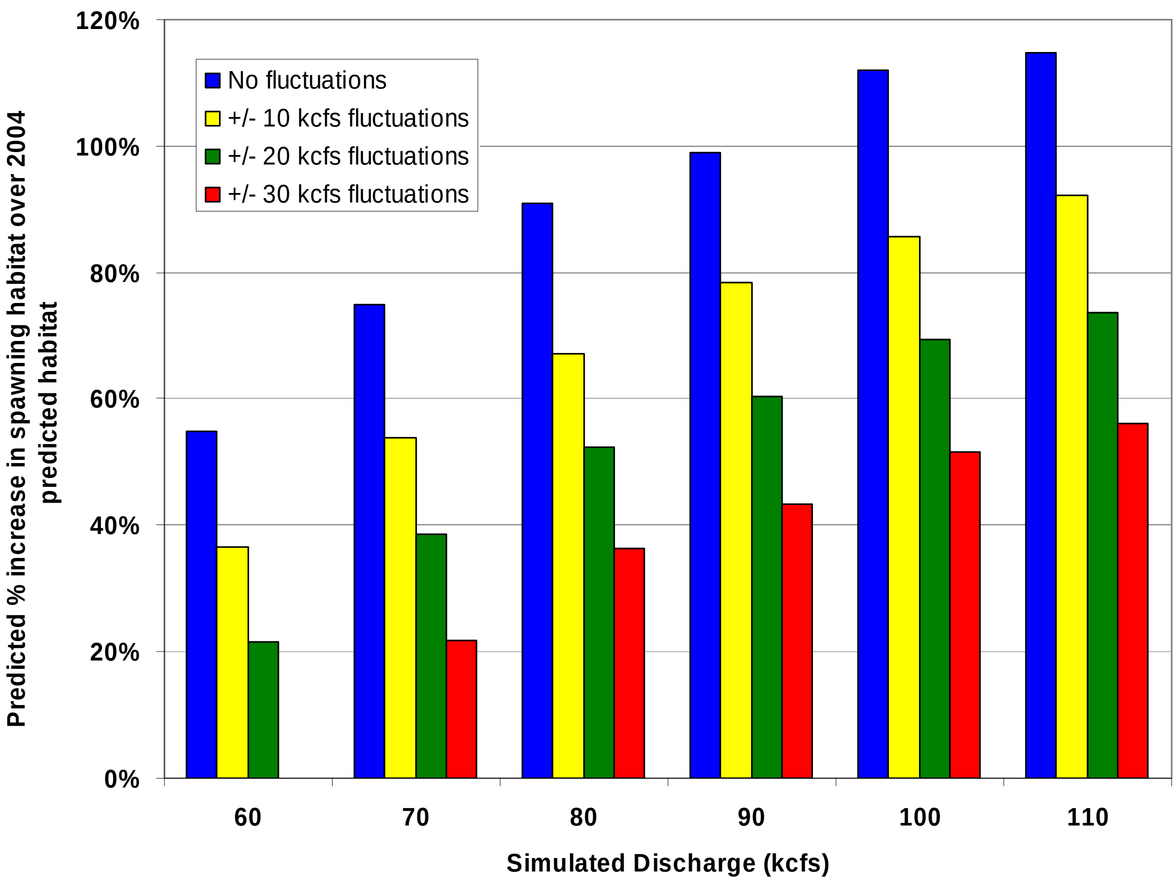
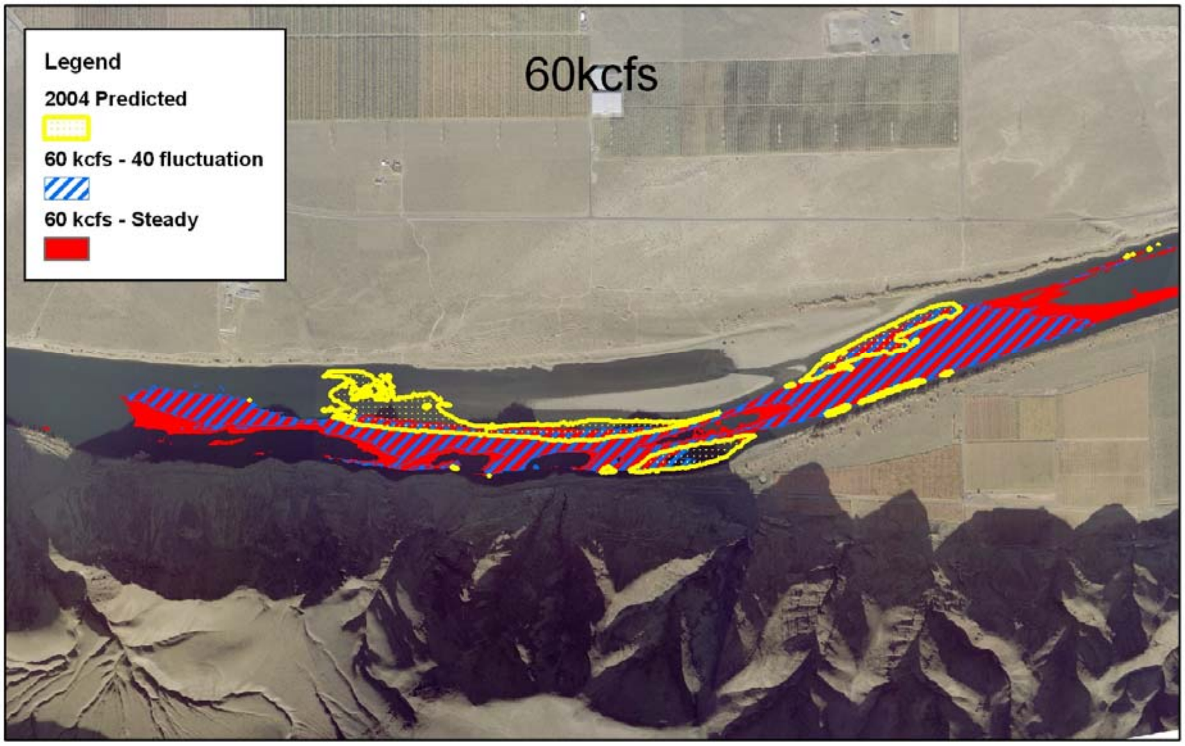


Figure 80. Predicted relative increase in spawning habitat area over the 2004 predicted spawning habitat area for 23 simulated alternative operations. The X-axis represents spawning habitat under the 2004 hydrograph. Fluctuations are structured around the average flow. For example, fluctuations of ±10 kcfs at a discharge of 80 kcfs imply that flows varied between 70 kcfs and 90 kcfs.



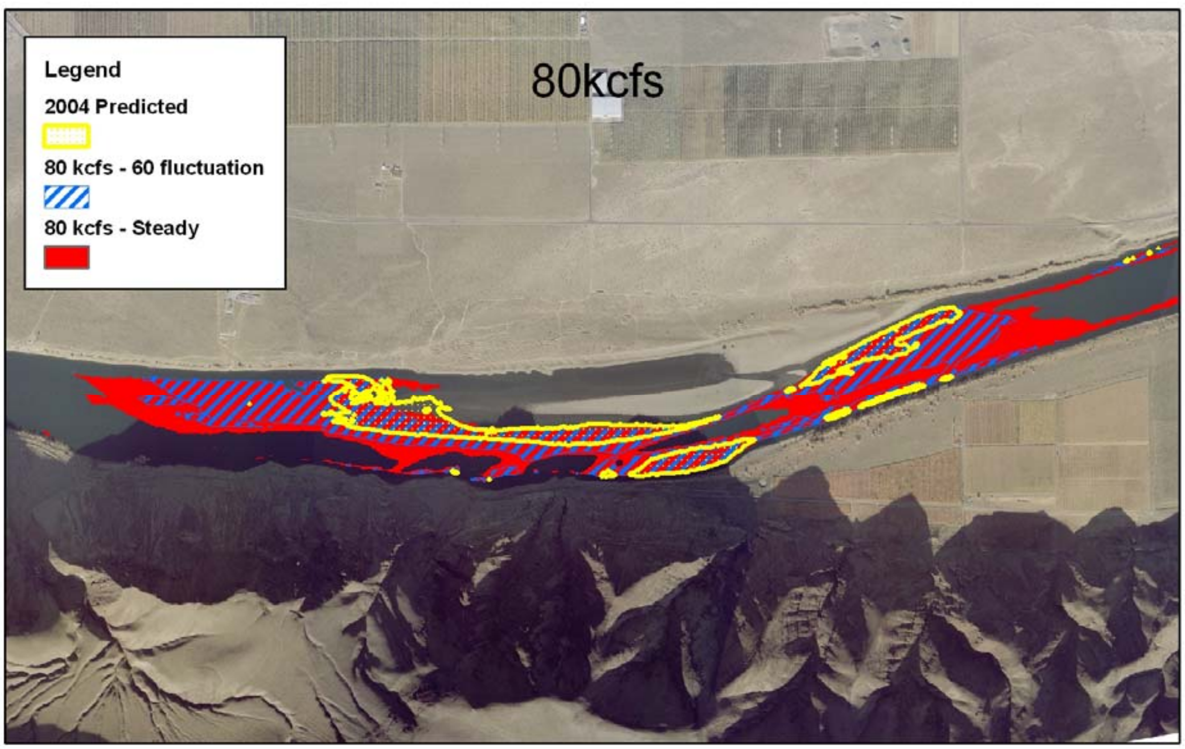


Figure 81. Comparison of the spawning habitat predicted for the 2004 actual hydrograph with the same average flow (80 kcfs) configured as steady-state, and with reduced fluctuations ±30 kcfs (bottom plate), as well as with a reduced average steady-state flow of 60 kcfs, and 60 kcfs with a ±20 kcfs fluctuation (top plate).

Discussion

The results of our spawning habitat model development and validation, as well as our habitat simulations reflect the influence of stream channel geomorphology, hydrograph variation, and fish responses to these conditions in terms of predicted suitable habitat. Results confirm that the variability and complexity associated with natural geomorphic features as well as a relatively natural flow regime are important factors for restoring or maintaining fall Chinook spawning habitat and the resulting productivity. The river channel and geomorphic features in the Hanford Reach are in excellent condition for the most part. However, the flow regime is highly variable and unnatural. Fall Chinook salmon did not evolve under this type of flow regime, and therefore it is critical to understand the effects of the current altered hydrographs on spawning habitat availability and persistence, especially since these effects have not been previously studied in detail.

From the onset of this work we hypothesized that fall Chinook were more likely to spawn at locations where suitable spawning conditions were persistently available rather than at locations where suitable spawning conditions were only intermittently available. Our results support and were consistent with this hypothesis. As further evidence, the orthophotography suggests fall Chinook would rather spawn in areas with persistently suitable conditions and lower levels of variability than in less persistence or less suitable habitat. This indicates that the result of providing similar amounts of spawning habitat by providing similar operations every year for a range of different escapement sizes will, at some point, result in redd superimposition.

We developed novel variables which captured the variation in physical conditions, and the persistence of suitable conditions over the time series of streamflows that Chinook experienced, and these variables demonstrated excellent discriminatory performance. We found that fall Chinook were more likely to spawn at locations where velocities were persistently greater than 1.0 m/s. While this lower limit of 1.0 m/s is consistent with observations made by other researchers, we are unaware of research that has attempted to quantify the persistence of suitable habitat conditions in a manner similar to ours. In future work we hope to examine the performance of different thresholds, as well as section-specific thresholds for the various spawning areas in the Hanford Reach. We also found that fall Chinook were more likely to spawn at locations where the variability in velocity or depth, as measured by either the CV or the range, was low. These results are consistent with our hypothesis that a more stable hydrograph, as was present prior to hydrosystem development, would increase the quantity and quality of fall Chinook spawning habitat. Considering that our model is limited to depths <6 m because of our limited ability to identify redds from orthophotos in deeper water, it is likely we underestimated habitat in areas where deep-water spawning occurs. Our future plans include conducting redd surveys in deep-water areas to determine the importance of these areas to fall Chinook salmon and incorporating these data into section-specific models.

Differences in fish response to physical conditions as evidenced by redd distribution patterns from site to site may have been associated with the relative persistence of suitable conditions among the different areas. Changes in the hydrograph and associated physical conditions in the upper segment were more abrupt over each 24-hour period than conditions that occurred in the middle and lower segments where wave dissipation resulted in a smoother, less variable set of conditions. This resulted in physical conditions that were more persistent at locations in the middle and lower segments. The primary spawning site in the upper segment is Vernita Bar which is only 6 rkm downstream from Priest Rapids Dam. Thus, this spawning area essentially reflects the hourly hydrograph as it is discharged from the Dam. The primary spawning sites in the middle segment are roughly 40 rkm downstream, and in the lower segment, 71 rkm downstream. These river lengths allow the time step of flow changes that occur at the dam to increase with increasing distance downstream, resulting in less variable, somewhat more stable conditions. In addition, changes in the physical conditions associated with the changing hydrograph were distributed over a larger area with more complex channel morphology in the middle and lower segments, further buffering variation. These observations indicate that when the temporal and spatial distribution of depths and velocities is sufficiently different among areas, along with differences in stream channel geomorphology, separate models would likely perform better than the Reach-wide model we developed as far as capturing fish response to each of the unique sets of conditions. In future work, we plan to explore section-specific models to further elucidate how hydrosystem operations and channel morphology interact, and how fish respond to those interactions in each area.

Our spawning habitat analysis was based only on physical variables to examine flow-related changes in habitat and did not incorporate the potential effects of these changes on fish behavior. It is not known whether flow fluctuations influence spawning behavior or success. In the future, we hope to focus on the diel aspects of redd site selection and the consequences of different diel hydrograph patterns at spawning areas other than Vernita Bar.

The tools we developed to conduct our spawning habitat assessment for the 90-km Hanford Reach represent significant progress towards evaluation of the effects of current operations, as well as alternative operational scenarios at Priest Rapids Dam on spawning habitat conditions throughout the Reach. Development of the DEM and the hydrodynamic model for the Reach has provided the framework to determine relevant physical metrics associated with specific streamflows throughout the Reach. These metrics provide the basis for determining suitable physical conditions at all sites. Our acquisition of a complete series of orthophotos for 2004 provided the basis for both model building and testing to link actual habitat use with the conditions that were present. Our acquisition of a complete series of orthophotos for 2005 provided further evidence for corroboration of our habitat simulations. Our model building and testing process provided the first step in evaluating the relationship between persistence and variability of conditions across widely fluctuating flows, and the associated fish responses. We have started development of a tool utilizing our predictive model(s) that will enable fishery and hydrosystem managers to evaluate a range of operations that will provide sufficient spawning habitat for the expected annual escapement to assist with in-season operational planning. We believe that by using these tools and our results, we can continue to make progress towards understanding the effects of hydrosystem operational decisions on spawning habitat and productivity of Hanford Reach fall Chinook. In addition, these tools can provide fishery managers with information to make informed decisions regarding operations that will preserve overall stock productivity and increase freshwater productivity during periods of poor ocean and climate conditions for this internationally important stock of salmon. These tools can also provide useful information to assist fishery managers and hydrosystem operators with evaluating operations that not only provide adequate spawning habitat for fall Chinook salmon, but also accommodate power production in the mid-Columbia River.

Conclusions

Although fall Chinook redds are distributed throughout the Hanford Reach, the highest concentration was found in the middle segment near White Bluffs. Our 2004 and 2005 orthophotography confirmed the redd distribution observed over the last 40 years during aerial surveys. Since fluctuating flows from Priest Rapids Dam produce variable hydrographs throughout the Reach as a result of wave attenuation and dissipation, specific areas need to be prioritized for management actions. Redd distribution results indicate that the highest priority spawning area should be the White Bluffs/Locke Island area, followed by Vernita Bar. Managing streamflows to provide spawning habitat at White Bluffs requires a different strategy than the current management protocol for Vernita Bar. Streamflows are temporally offset by 6 to 12 hours, and the load following cycle is dampened. Depending on the goals of a spawning season streamflow management plan, the hourly hydrograph at Priest Rapids Dam should be structured to produce the desired effect at White Bluffs. Adjustments should then be made considering the effects at Vernita Bar.

The nature of physical conditions along the Reach in terms of river channel geomorphology and hydrograph characteristics is quite variable. Fish responses to these variable conditions were somewhat different among the three sections in the Reach. A qualitative comparison between habitat predicted by applying our spawning habitat model developed with Reach-wide observations, to habitat used (redd clusters) in each of the three sections in the Hanford Reach, indicated fish response was different among areas. This was at least partially a function of differences in the distribution of physical habitat parameters and variation in channel morphology between the segments. The “general purpose”, Reach-wide model we developed performs exceptionally well for predicting spawning habitat use, and will be useful for fishery managers and hydrosystem operators to evaluate the effects of a range of hydrographs on both available spawning habitat and power production. However, our results indicate that we may be able to build section-specific spawning habitat models that will more precisely capture and describe fish response among the three areas.

In order to understand the effects of the current altered hydrographs on spawning habitat availability and persistence, we plan to continue our current spawning habitat model investigations. We have made significant progress thus far in terms of understanding the effect of spatial and temporal variation in physical conditions along the Reach, and the spawning habitat model we developed provides a sound basis for the next steps to complete model building work within each specific spawning area. We identified and applied the important concept of persistence of suitable spawning habitat conditions to the variable hourly hydrograph. We also identified variables that are important for predicting fall Chinook spawning habitat, and gained insight into the level of contrast required to increase confidence in our habitat simulations. These accomplishments have resulted in a useful tool for both pre-season and in-season planning that fishery managers and hydrosystem operators can use to evaluate options for river operations to balance management goals for fall Chinook production, with goals for power production. They have also provided the foundation for the next steps in our research to build spawning habitat models that are site-specific and tailored to the specific conditions that occur among spawning areas in the Hanford Reach.

Recommendations

- A monitoring program should be implemented to collect comprehensive biological data on adult fall Chinook responses to hourly flow fluctuations during the spawning season. This effort should be applied to all three of the segments in the Hanford Reach considering the variation in conditions and fish response we observed in our work. It should include an assessment of the energetic costs of fluctuating flows and the impact on completion of successful spawning.
- Because the majority of fall Chinook spawning occurs in the middle segment, operations at Priest Rapids Dam should be structured for the desired effect at the White Bluffs area.
- Ramped operations should be investigated to determine the utility of providing spawning habitat at different flow levels and locations throughout the season to avoid redd super-imposition.
- Studies should be conducted to determine the extent of redd superimposition for various levels of returning adult abundance, the effect on production, and what operational scenarios minimize redd superimposition.
- Studies should be conducted to determine whether any part of the spawning process including redd site selection, redd-building, spawning, and defending the redd is affected by day or night time periods.
- Studies should be conducted to determine the extent, location, and physical characteristics associated with deep-water spawning.
- Additional work should be conducted to determine other geomorphic features that might be influential in spawning site selection by fall Chinook, including an investigation into the feasibility of predicting hyporheic flow or upwelling using geomorphic models.
- Aerial orthophotography should be conducted on a regular basis to determine patterns in timing and location of spawning activity throughout the spawning season. This becomes particularly important if plans are made to test alternative operational scenarios.
- Work should continue to determine the carrying capacity for various flow levels and hydrographs so operations can be crafted to accommodate expected escapement levels.
- Work should be conducted with the regional fishery agencies and tribes to evaluate spawning escapement goals using habitat models and a monitoring program for various operational alternatives.

REARING HABITAT ASSESSMENT

Overview

A previous study in the middle segment of the Hanford Reach showed rearing habitat is uniformly distributed along shorelines and that habitats vary with fluctuating water levels (Tiffan et al. 2002). Because juvenile fish can move in response to flow and habitat changes, quantifying the spatial distribution of rearing habitat for a range of flows will provide a picture of habitat connectivity for the entire Hanford Reach. This is important to rearing fish, particularly if movement between habitats increases predation risk, energy consumption, or results in some other form of reduced performance. Rearing habitat use may also have important implications to risks of stranding or entrapment as a result of flow fluctuations.

The objectives of this rearing habitat assessment were to: 1) quantify the amount of rearing habitat for the entire Hanford Reach for a range of streamflows, 2) examine the flow-related distribution of that habitat, and 3) examine the patch size and connectivity of the rearing habitat.

Methods

The rearing habitat model we used was derived by Tiffan et al. (2002). We summarize their methods here to provide context for our analyses. Subyearling fall Chinook salmon were collected from shoreline habitats in the Hanford Reach using point electrofishing during April through May in 1994 and 1995. Sampling was stratified to include habitats with different combinations of velocity, depth, and substrate. The velocities sampled ranged from 0 to 0.4 m/s, depths ranged from 0 to 3.3 m, and substrates were grouped into five size categories (<1 mm, 1-32 mm, 32-64 mm, 64-256 mm, and >256 mm) that were most common in the Hanford Reach.

Data were collected using a 5.5-m electrofishing boat (35 cm draft) with two 1.0-m umbrella anode arrays and an electrical output of 2 amps at 60 pulses/s DC. A sample was collected by driving directly towards the shoreline, abruptly stopping the boat about 1.5-5 m from shore depending on depth, and shocking a localized area for at least 8 s. At the end of sampling, a buoy was set to mark the area where fish were collected, or the center of the shocked area if fish were absent. Physical characteristics were measured at each site to describe habitat and included water velocity, flow direction, water depth, distance to the shoreline, substrate size, and presence of cover and vegetation.

Data were analyzed using logistic regression techniques to determine the probability of a habitat cell being suitable for rearing subyearling fall Chinook salmon. The final model developed by Tiffan et al. (2002) included lateral bed slope and water velocity. The correct classification of fish presence and absence in rearing habitats using their model was 76% using a probability level of 0.5. The correct prediction rate of fish presence was 78%, whereas fish were absent in the remaining 22% of the habitats predicted to contain fish (error of commission). Conversely, fish were present in 31% of the habitats where their model predicted them to be absent (error of omission).

Rearing Habitat Use

We used the logistic regression model derived by Tiffan et al. (2002) to predict the amount of subyearling rearing habitat at different streamflows. This model predicts the probability of subyearling rearing presence or absence using slope and water velocity and is expressed as

$$g(x) = -3.19 + 2.23V_1 + 2.45V_2 + 1.96V_3 + 2.66S_1 + 2.42S_2 + 2.28S_3 + 1.04S_4 \quad \textbf{(Equation 8)}$$

where $V_{1\text{-}3}$ represent different categories of water velocity, and $S_{1\text{-}4}$ represent different categories of slope (Table 26). Both water velocity and slope were modeled as design variables where an individual variable assumed a value of 1 when its category contained a measure for a given habitat cell; otherwise the value was 0. Expression of the probability P_i uses the equation

$$P_i = \frac{e^{g(x)}}{1 + e^{g(x)}} \quad \textbf{(Equation 9)}$$

as previously described. These equations provided the foundation for a grid-based assignment of the probability of presence of rearing subyearlings in habitat cells near the shoreline. Habitat attributes of each GIS cell were used in the logistic regression model to obtain the probability of fish presence in each cell. We created probability coverages in the GIS, and habitat cells with probabilities greater than or equal to 0.5 were considered suitable for rearing fall Chinook salmon. In addition, Tiffan et al. (2002) observed non-use of habitats to occur where water velocities were greater than 0.71 m/s and water depths were greater than 1.5 m. We applied these same criteria in our analyses. Coverages of suitable rearing habitat were created for each simulated discharge and probability level. These binary rearing grids were then converted to rearing polygons in the GIS, and their areas were summed for each segment, discharge, and probability level. Finally, we calculated descriptive statistics such as the sum of all rearing areas and the average size of rearing areas. We did not calculate errors of omission and commission for the rearing model because they were calculated previously by Tiffan et al. (2002).

Table 26. Summary of the variable categories used by Tiffan et al. (2002) to predict the presence of rearing subyearling fall Chinook salmon in shoreline habitats of the Hanford Reach. Water velocities greater than 0.4 m/s and slopes greater than 40% serve as reference categories for these design variables.

Variable	Variable category
Velocity (V_1)	0-0.1 m/s
Velocity (V_2)	0.1-0.2 m/s
Velocity (V_3)	0.2-0.3 m/s
Slope (S_1)	0-10%
Slope (S_2)	10-20%
Slope (S_3)	20-30%
Slope (S_4)	30-40%

Rearing Habitat Availability

The physical habitat variables known to be important to rearing fall Chinook salmon were estimated for the entire Hanford Reach using outputs from River2D. Physical variables were estimated for 37 discharges from 40 to 400 kcfs in 10 kcfs increments. For each stream flow modeled, we used GRID (ESRI 1992) to create velocity and depth TINs from the River2D mesh. To ensure consistency with the original bathymetric data, we resampled TINs into 16-m^2 resolution grids by linear interpolation and then calculated the slope (%) of each cell with GRID. Physical parameters for each cell in the GIS were then combined with a logistic regression model to calculate probabilities for each cell, with higher values indicating an increased likelihood of suitable habitat in that cell. Coverages were created for a range of steady state streamflows and for several probability levels. The area of all cells with a particular probability coverage were then summed for each streamflow to determine the total amount of rearing habitat available.

Results

Rearing Habitat Use and Availability

The amount of juvenile fall Chinook salmon rearing habitat in the Hanford Reach generally decreased as flows increased (Figure 82). The amount of rearing habitat ranged from a maximum of 713 ha at 40 kcfs to a minimum of 376 ha at 390 kcfs. The greatest net decrease in habitat amount per unit of flow increase (mean = 36.2 ha/10 kcfs) occurred between flows of 40 to 100 kcfs, whereas decreases in habitat were ten times lower (mean = 3.7 ha/10 kcfs) between flows of 110 to 400 kcfs.

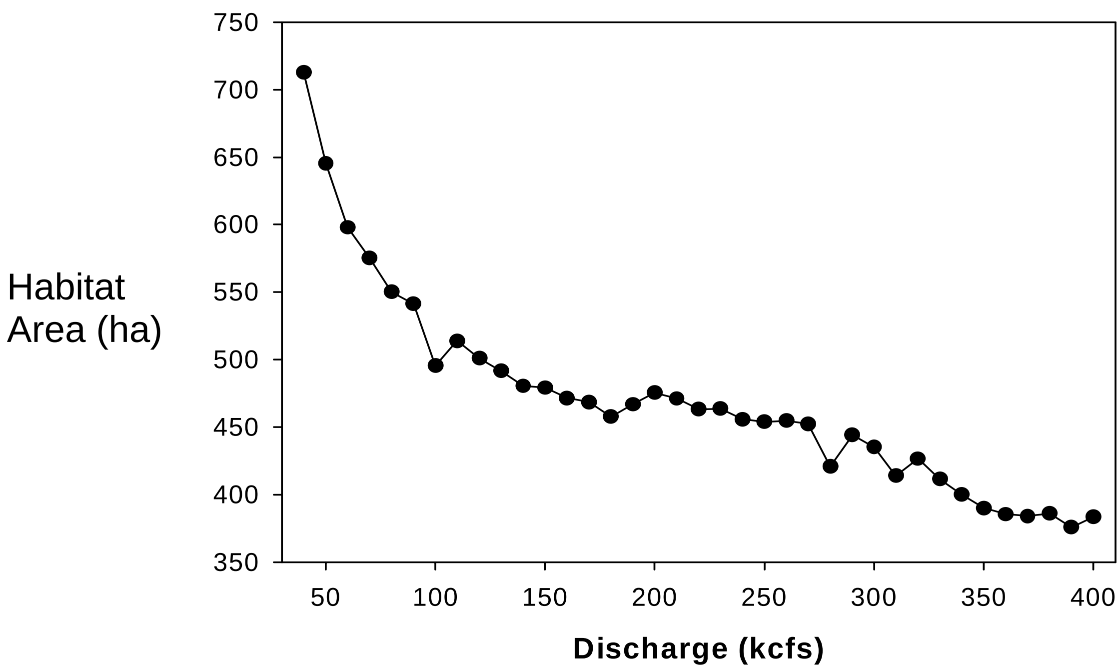


Figure 82. The relationship between juvenile fall Chinook salmon rearing habitat and modeled steady-state flows in the Hanford Reach of the Columbia River.

The number and size of distinct rearing habitat patches were inversely related and varied with flow (Figure 83). At lower flows, there were fewer rearing patches than at higher flows, but their size was much greater at lower flows than at higher flows. A similar trend was apparent for the number and size of islands present in the Hanford Reach at different flows (Figure 84). At lower flows, there were fewer but larger islands than at higher flows where islands were smaller but more numerous.

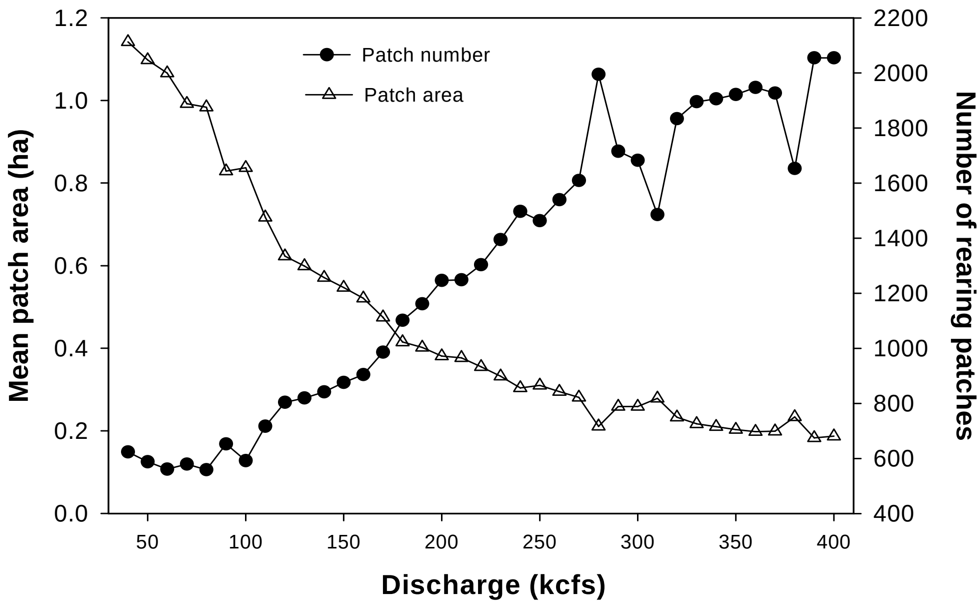


Figure 83. The relationship between juvenile fall Chinook salmon rearing habitat patch size and number, and discharge in the Hanford Reach of the Columbia River.

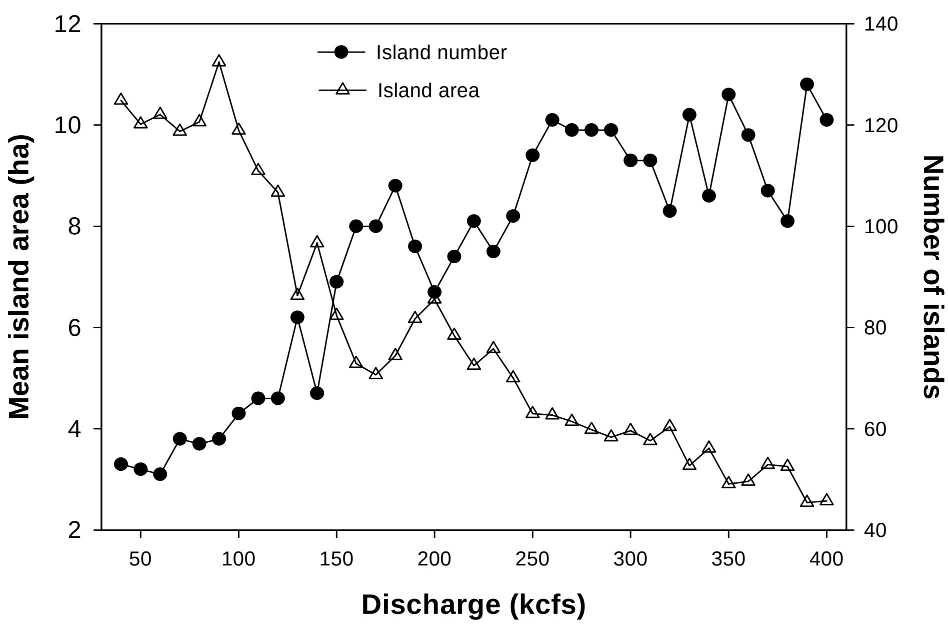


Figure 84 The relationship between island area and island number and discharge in the Hanford Reach of the Columbia River.

Each year it is estimated that 8-28 million fall Chinook salmon fry are naturally produced in the Hanford Reach (Nugent et al. 2002). Using these estimates in conjunction with estimates of rearing habitat allowed us to determine potential fish densities in rearing habitats at different river flows. At the lowest flow simulated (40 kcfs), fish density would be 3.9 fish/m^2 if the fry population was 28 million, and 1.1 fish/m^2 if the population was 8 million fish. At the highest flow simulated (400 kcfs), densities would equate to 7.3 fish/m^2 and 2.1 fish/m^2 for the aforementioned population sizes. In 2003, river flows averaged 116 kcfs for the period from March 1 through May 31. Using the closest simulated flow of 120 kcfs and a fry estimate of 23 million (Paul Hoffarth, Washington Department of Fish and Wildlife, personal communication) yields a density estimate of 4.6 fish/m^2 in 2003.

Because rearing fish are stranded and entrapped in rearing areas with low slopes, we calculated the amount of area in different slope categories for some sample flow bands (Figure 85). Most 10-kcfs flow bands are characterized by slopes less than 5%, which are also important in defining juvenile fall Chinook salmon rearing habitat. Therefore, most juvenile fall Chinook salmon rearing habitats can potentially strand and entrap fish if dewatered during a flow reduction.

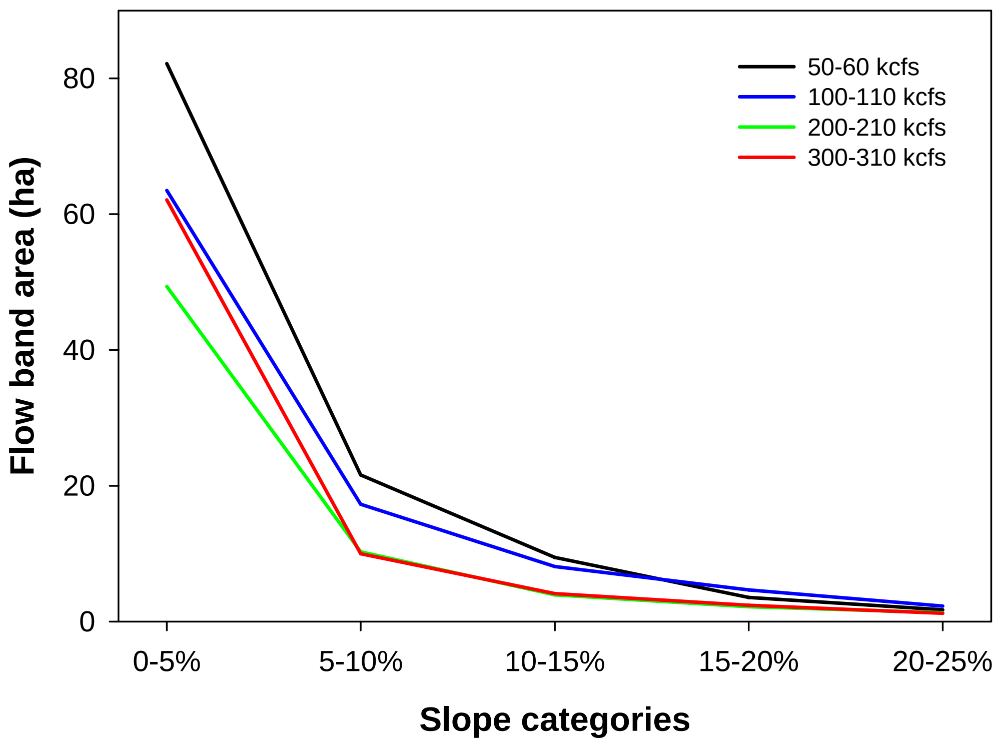


Figure 85. An example of the amount of area by slope category in four different flow bands in the Hanford Reach of the Columbia River.

Discussion

The amount of fall Chinook salmon rearing area in the Hanford Reach increased as flow decreased because of flatter near-shore slopes and reduced water velocities. In contrast, at higher flows water velocities were generally greater and the shorelines were located on steeper banks (higher lateral slopes) due to fuller river channels. These findings are similar to those of Tiffan et al. (2002) for the middle segment of the Hanford Reach. Garland et al. (2004) also reported similar findings for juvenile fall Chinook salmon rearing habitat in the mainstem Columbia River below Bonneville Dam. In that study, the authors found islands provided more rearing area per unit of shoreline, but that was due somewhat to the geomorphology of their study area. Similarly, the larger islands that existed in the Hanford Reach at low flows provided extensive shorelines for rearing fall Chinook salmon. However, the rearing area provided by the greater number of islands at higher flows was offset by their smaller sizes.

Tiffan et al. (2002) found that between 77-97% of the shorelines in the middle segment of the Hanford Reach provided suitable rearing habitat for fall Chinook salmon. We found that at lower flows, rearing habitat is characterized by fewer numbers of larger habitat patches indicating that rearing habitat is more connected than at higher flows. This may be important to rearing fall Chinook salmon if flow fluctuations elicit up or downstream movements. Movements within larger habitat patches may pose less risk in terms of predation or energetic costs than movement between patches of suitable habitat. Currently, little information exists on the biological consequences to juvenile fall Chinook salmon fitness and survival of temporally and spatially changing rearing habitats

Conclusions

Results of our fall Chinook rearing habitat studies and modeling were consistent with the detailed studies that have been conducted previously in the Hanford Reach. Although rearing habitat varies with streamflow, stability is likely more important to juvenile Chinook than the absolute flow level. Stable flows and habitat conditions require less movement and less energy expenditure than constantly fluctuating flows and spatially variable habitat conditions. Stable flows also reduce the potential for stranding or entrapment of juveniles.

Recommendations

- Studies should be conducted to determine the effect of fluctuating flows on the suitability of rearing habitats.
- Studies should be conducted to identify the relationship between rearing habitat and entrapment and stranding locations.
- Juvenile behavior studies should be conducted to determine spatial variation and size dependent variation in behavior. Studies should include sampling across the full range of habitat types that are present as well as an evaluation of the use of interstitial spaces in the substrate by rearing juveniles.
- Studies should be conducted to determine if there are differences in diel habitat use.

CONCLUSIONS

Conclusions and recommendations for each section of the report have been compiled and repeated here for the ease of reference.

Hydrodynamic modeling

The hydrodynamic modeling component of this study built upon earlier work characterizing the physical conditions in the Hanford Reach and provides a foundation for quantitatively assessing the effects of streamflow and streamflow variation on entrapment mortality of juvenile Chinook, adult Chinook spawning habitat, and juvenile Chinook rearing habitat. Our assessment capitalizes on recent advances in habitat mapping, remote sensing technology, simulation models of hydrodynamics, statistical sampling methods and GIS technology to characterize habitat conditions and evaluate the effects of flow and flow fluctuations on Hanford Reach fall Chinook. The hydrodynamic modeling and remote data collection techniques that we used in this study have dramatically increased our ability to accurately describe the dynamics of aquatic habitats in this large, mainstem river segment.

River2D simulations generally did a satisfactory job of estimating water velocities, depths, and water surface elevations for habitat modeling purposes in the Hanford Reach. Both the size of the Columbia River in the Hanford Reach, and the availability of existing data, presented some unique modeling challenges. We believe that the performance of River2D was, in general, a fair representation of the hydraulic conditions in the Hanford Reach. In spite of some limitations, River2D proved to be a useful tool for predicting hydraulic conditions at the scale we evaluated fall Chinook salmon spawning habitat. We believe that collecting additional bathymetry for the DEM that was used for the middle segment would improve the accuracy of River2D, especially in the areas where the original SHOALS data were sparse. We also believe that additional calibration and validation of River2D could improve the accuracy of simulated conditions. These work tasks are planned as part of our continued Hanford Reach studies.

The availability of MASS1 saved time and resources that would have been spent collecting or attempting to model similar data. The capability of MASS1 to produce unsteady-state streamflow simulations for the entire Hanford Reach increased the efficiency of our hydrodynamic modeling tasks in that we did not have to run River2D in an unsteady state. MASS1 simulations provided the necessary detail and accuracy in streamflow simulations to estimate entrapment histories for the GIS analysis, and it also provided boundary conditions for our River2D modeling which saved the time that would have been required to develop rating curves at those locations.

Entrapment Studies

Using empirical physical and biological data integrated with hydrodynamic model output and GIS analysis, we mechanistically estimated the impact of fluctuating flows on juvenile fall Chinook salmon for the entire Hanford Reach. Our results confirm that flow fluctuations due to hydropower operations cause significant mortality of juvenile fall Chinook that rear in the Hanford Reach. These impacts appear to be significantly greater than those previously estimated. Our observations of significant entrapment in spite of the current protection measures highlight the importance of developing operational strategies that are more effective at minimizing juvenile mortality.

While our work provides a useful tool for estimating the relative effects of current and alternative operations on entrapment, our results do not represent the total mortality effects of operations on juvenile fall Chinook. Our impact estimate was conservative for several important reasons; 1) sampling started well after emergence was underway, 2) entrapment enumeration was not complete, likely resulting in under-estimates of entrapment histories, 3) detection and enumeration of entrapped fish was incomplete, 4) predation on entrapped fish was not accounted for, and 5) we did not attempt to quantify the potentially significant level of mortality due to stranding. Considering these factors along with our impact estimate, we believe that previous efforts may have greatly under-estimated the actual mortality-level impacts that occurred.

Based on the results of our studies, there is little quantitative basis for assuming that a flow fluctuation at low flows is more harmful than the same fluctuation at high flows, or vice versa. In addition, our evaluation of alternative hydrographs indicated that the impact could be reduced by controlling the size and frequency of flow fluctuations, and the results of our field work suggest that the timing of fluctuations (early vs. late rearing season) could also be used to reduce the impacts. The simulation results suggest that reducing flow fluctuations has considerable potential for reducing mortality levels if fluctuation magnitudes are kept below 10 kcfs. The number of entrapped Chinook dramatically increased with increasing flow fluctuations of 20, 30 and 40 kcfs. Based on the non-linear relationship between the fluctuation magnitude and frequency, and fish entrapped, it appears fluctuations above 10 kcfs produce dramatic increases in the number of fish entrapped. The size and frequency of flow fluctuations were directly related to the number of entrapments affected. Considering the results of our re-regulation analysis, we believe the physical ability exists to control flow fluctuation magnitude and frequency to the extent required to reduce expected juvenile Chinook mortality impacts. Our analysis of the capability of the Priest Rapids and Wanapum projects to re-regulate and reduce flow fluctuations indicated that the flexibility exists within "normal" operating limits to both limit the number of flow fluctuations and the magnitude of the fluctuations.

We used three approaches to place our impact estimate into a population context. Using the SAR data, we estimated the 4,300-8,300 adults may have been lost due to entrapment mortality in 2003. Using the fry production modeling data, the 2003 entrapment impact alone likely constituted a 12% reduction in the fry population. In other years under the Protection Plan the impact to the population caused by operations at Priest Rapids Dam may have been 31% to over 90%. The scale of these impacts to the Hanford Reach fry population would translate into potential harvest reductions for all adult Chinook of 9,000-170,000 fish in ocean and in-river fisheries. These foregone adults represent large reductions in the number of adults available to commercial and sport ocean fisheries, commercial, sport, and tribal treaty in-river fisheries, and spawning escapement to the Hanford Reach.

Spawning Habitat Studies

Although fall Chinook redds are distributed throughout the Hanford Reach, the highest concentration was found in the middle segment near White Bluffs. Our 2004 and 2005 orthophotography confirmed the redd distribution observed over the last 40 years during aerial surveys. Since fluctuating flows from Priest Rapids Dam produce variable hydrographs throughout the Reach as a result of wave attenuation and dissipation, specific areas need to be prioritized for management actions. Redd distribution results indicate that the highest priority spawning area should be the White Bluffs/Locke Island area, followed by Vernita Bar. Managing streamflows to provide spawning habitat at White Bluffs requires a different strategy than the current management protocol for Vernita Bar. Streamflows are temporally offset by 6 to 12 hours, and the load following cycle is dampened. Depending on the goals of a spawning season streamflow management plan, the hourly hydrograph at Priest Rapids Dam should be structured to produce the desired effect at White Bluffs. Adjustments should then be made considering the effects at Vernita Bar.

The nature of physical conditions along the Reach in terms of river channel geomorphology and hydrograph characteristics is quite variable. Fish responses to these variable conditions were somewhat different among the three sections in the Reach. A qualitative comparison between habitat predicted by applying our spawning habitat model developed with Reach-wide observations, to habitat used (redd clusters) in each of the three sections in the Hanford Reach, indicated fish response was different among areas. This was at least partially a function of differences in the distribution of physical habitat parameters and variation in channel morphology between the segments. The "general purpose", Reach-wide model we developed performs exceptionally well for predicting spawning habitat use and will be useful for fishery managers and hydrosystem operators to evaluate the effects of a range of hydrographs on available spawning habitat. However, our results indicate that we may be able to build section-specific spawning habitat models that will more precisely capture and describe fish response among the three areas.

In order to understand the effects of the current altered hydrographs on spawning habitat availability and persistence, we plan to continue our current spawning habitat model investigations. We have made significant progress thus far in terms of understanding the effect of spatial and temporal variation in physical conditions along the Reach, and the spawning habitat model we developed provides a sound basis for the next steps to complete model building work within each specific spawning area. We identified and applied the important concept of persistence of suitable spawning habitat conditions to the variable hourly hydrograph. We also identified variables that are important for predicting fall Chinook spawning habitat, and gained insight into the level of contrast required to increase confidence in our habitat simulations. These accomplishments have resulted in a useful tool for both pre-season and in-season planning that fishery managers and hydrosystem operators can use to evaluate options for river operations to balance management goals for fall Chinook production, with goals for power production. They have also provided the foundation for the next steps in our research to build spawning habitat models that are site-specific and tailored to the specific conditions that occur among spawning areas in the Hanford Reach.

Rearing Habitat Studies

Results of our fall Chinook rearing habitat studies and modeling were consistent with the detailed studies that have been conducted previously in the Hanford Reach. Although rearing habitat varies with streamflow, stability is likely more important to juvenile Chinook than the absolute flow level. Stable flows and habitat conditions require less movement and less energy expenditure than constantly fluctuating flows and spatially variable habitat conditions. Stable flows also reduce the potential for stranding or entrapment of juveniles.

RECOMMENDATIONS

These recommendations represent study tasks and future analyses that we believe would help to reduce uncertainty and address issues identified from this work. We hope these recommendations will help to guide future research and management of Hanford Reach fall Chinook.

Hydrodynamic Modeling

- Bathymetric work should continue until an acceptable level of data has been acquired to improve the DEM used for hydrodynamic modeling, particularly in the middle section of the Reach.

- Substrate characterization should continue until coverage is complete for all areas to provide the relevant level of detail for both calibration of River2D and to improve spawning habitat simulations.

- Thorough, comprehensive calibration of River2D should be completed as the first step towards increasing the accuracy of hydraulic simulations.

- Reach-wide River2D validation should be continued and completed in conjunction with calibration.

- Work should continue towards running the River2D model in transient or unsteady mode to enhance future spawning, entrapment and stranding studies.

Entrapment Studies

- Continue the fish sampling program during the emergence and rearing periods to estimate the total number of fish entrapped using our modeling framework. The sampling effort should be kept at a level similar to the 2003 effort, or increased to reduce the resulting uncertainty in the estimate.
- Develop and implement a study plan to evaluate entrapment fish sampling efficiency.
- Complete the enumeration of entrapments, especially at the lower flow levels.
- Implement additional stratification of entrapments for the field sampling program by size, location (flow band), and other physical features to help reduce uncertainty around impact estimates.
- Develop and implement planned flow manipulation experiments to quantify the diel impact on fish per entrapment.
- Develop and implement a plan to estimate the effect of fluctuating flows on ***stranding*** of juvenile Chinook, and design a statistically rigorous sampling program to survey stranding areas.

- When stranding field studies are complete, incorporate an evaluation component for stranding into our modeling evaluation system.
- Investigate the role of water temperature as it relates to stranding and entrapment susceptibility.
- Continue with the development and evaluation of alternative operations and their effect on Chinook stranding and entrapment.
- Conduct a focused study on the impact of various ramping rates on entrapment and stranding.
- Conduct a more rigorous abundance index seining program to determine if such a program can be linked to subsequent entrapment and stranding locations and magnitudes. The index seining could then potentially be used as a monitoring tool.
- Conduct a focused study on predation rates on stranded and entrapped fish.
- Conduct similar studies on resident fish species that are also susceptible to entrapment.
- Evaluate the effect of re-regulating flows from the Priest Rapids Project into the Hanford Reach on power production from the mid Columbia using the Hourly Coordination Agreement power optimization software.
- Develop analytical approaches to improve estimates of the effect of entrapment and stranding mortality rates on adult productivity of Hanford Reach fall Chinook.

Spawning Habitat

- A monitoring program should be implemented to collect comprehensive biological data on adult fall Chinook responses to hourly flow fluctuations during the spawning season. This effort should be applied to all three of the segments in the Hanford Reach considering the variation in conditions and fish response we observed in our work. It should include an assessment of the energetic costs of fluctuating flows and the impact on completion of successful spawning.
- Because the majority of fall Chinook spawning occurs in the middle segment, operations at Priest Rapids Dam should be structured for the desired effect at the White Bluffs area.
- Ramped operations should be investigated to determine the utility of providing spawning habitat at different flow levels and locations throughout the season to avoid redd super-imposition.
- Studies should be conducted to determine the extent of redd superimposition for various levels of returning adult abundance, the effect on production, and what operational scenarios minimize redd superimposition.
- Studies should be conducted to determine whether any part of the spawning process including redd site selection, redd-building, spawning, and defending the redd is affected by day or night time periods.

- Studies should be conducted to determine the extent, location, and physical characteristics associated with deep-water spawning.
- Additional work should be conducted to determine other geomorphic features that might be influential in spawning site selection by fall Chinook, including an investigation into the feasibility of predicting hyporheic flow or upwelling using geomorphic models.
- Aerial orthophotography should be conducted on a regular basis to determine patterns in timing and location of spawning activity throughout the spawning season. This becomes particularly important if plans are made to test alternative operational scenarios.
- Work should continue to determine the carrying capacity for various flow levels and hydrographs so operations can be crafted to accommodate expected escapement levels.
- Work should be conducted with the regional fishery agencies and tribes to evaluate spawning escapement goals using habitat models and a monitoring program for various operational alternatives.

Rearing Habitat

- Studies should be conducted to determine the suitability and relationship between persistent and non persistent rearing habitats with flow fluctuations.
- Studies should be conducted to identify the relationship between rearing habitat and entrapment and stranding locations.
- Juvenile behavior studies should be conducted to determine spatial variation and size dependent variation in behavior. Studies should include sampling across the full range of habitat types that are present as well as an evaluation of the use of interstitial spaces in the substrate by rearing juveniles.
- Studies should be conducted to determine if there are differences in diel habitat use.

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APPENDIX A

INDEPENDENT PEER REVIEW DOCUMENT

The authors of this report contracted an independent scientific peer review coordinated by FWS and conducted by ESSA Technologies (ESSA). FWS worked with ESSA to select three independent, qualified scientists for the review. FWS developed a list of 20 specific questions for the reviewers to consider, and also requested any other relevant comments they might have had regarding the methods, results, conclusions, or discussion presented in the report. Three eminent scientists with extensive experience in quantitative analyses of hydrodynamic and fisheries modeling for river management were selected to conduct the review on the draft report. Each reviewer provided a slightly different perspective and level of expertise regarding hydrodynamic modeling, fall Chinook biology, life history, and habitat requirements, and fishery issues relating to hydropower development and operations. The first reviewer, Dr. Paul Higgins, is an industry representative from B.C. Hydro who has extensive experience in all aspects of the work conducted for this study. The second reviewer, Dr. William Miller is a consultant with extensive experience in hydrodynamic and habitat modeling. The third reviewer, Dr. Peter Steffler, is a professor at the University of Alberta with extensive expertise in hydrodynamic modeling, and one of the primary developers of the River2D hydrodynamic model. Our goals were to obtain a comprehensive, thorough technical review of all aspects of our work, incorporate the peer review comments into our final report, and obtain a final set of comments from the reviewers on the end product. ESSA provided the draft report materials to the peer reviewers, coordinated the review, and compiled the reviewers' individual comments into a coherent report. The authors then responded to each of the peer reviewers' comments within the report with a discussion of the specific comment and a description of any report modifications that resulted. The authors' responses to each comment have been inserted within the peer review report in black italic text. Page numbers cited by the reviewers were associated with the original draft version of the report and no longer correspond to this final version. This final version of the report has been given back to the peer reviewers for a summary evaluation of the final product. The final evaluation will be made available when it is received.

Fall Chinook Salmon Habitat Analysis and Stranding/Entrapment Evaluation in the Hanford Reach of the Columbia River Report

Peer Review Comments

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Feb 28, 2005

Fall Chinook Salmon Habitat Analysis and Entrapment Evaluation In the Hanford Reach of the Columbia River

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Table of Contents

Background

Streamflows downstream from the Priest Rapids Hydroelectric Project can affect the amount of available spawning and rearing habitat and cause stranding and entrapment losses of juvenile fall Chinook in the 82-km long Hanford Reach, the last un-impounded section of the Columbia River upstream from Bonneville Dam. The U.S. Fish and Wildlife Service (FWS) in cooperation with the Columbia River Inter-Tribal Fish Commission (CRITFC), the Alaska Department of Fish and Game (ADFG), the U.S. Geological Survey (USGS), the Washington Department of Fish and Wildlife (WDFW), the Yakama Nation (YN), and Nugent GIS conducted studies in the Hanford Reach to evaluate the effects of streamflow regulation on spawning/rearing habitat of fall Chinook salmon, and to quantify associated stranding and entrapment losses.

The FWS-led studies were designed to examine the relationships between streamflow, hydrosystem load-following and resultant flow fluctuations on fall Chinook salmon in the Hanford Reach. Specific focus was dedicated to assessing the impacts of fluctuating flows on juvenile Chinook salmon mortality and on the amount and distribution of spawning and rearing habitat. The objectives of the spawning and rearing habitat components of the study were to 1) quantify fall Chinook spawning and rearing habitat as a function of streamflow and streamflow fluctuations for the entire Reach, 2) examine the distribution of spawning habitat throughout the Reach, 3) estimate the amount of spawning habitat required to accommodate various escapement levels, and 4) examine the relationship between rearing habitat distribution and entrapment results. The information provided is intended to better define the Hanford Reach's fall Chinook production potential and help identify streamflow requirements for both spawning and protection of rearing juvenile fall Chinook. The objectives of the associated stranding and entrapment studies were to 1) provide a quantitative estimate of fall Chinook entrapment mortality caused by fluctuating river flows (load following) in spring of 2003, 2) provide insight into the factors that lead to stranding and entrapment, and 3) to provide mainstem hydro project operators with operational alternatives that will help to reduce the impacts of flow fluctuations on fall Chinook.

A foundational element of the study was a characterization of the hydrodynamics in the Hanford Reach. The objective of the hydrodynamic modeling was to provide the physical characteristics associated with a range of streamflows throughout the Hanford Reach. These physical characteristics are required for both the evaluation of fluctuating flows and entrapment of juvenile fall Chinook, and for spawning and rearing habitat modeling.

In January/February of 2005, a draft "Fall Chinook Salmon Habitat Analysis and Stranding/Entrapment Evaluation In the Hanford Reach of the Columbia River" report presenting all preliminary results and discussion was prepared. This report was sent for independent peer review by three eminent scientists with extensive experience in quantitative analyses of hydrodynamic and fisheries modeling for river management:

- Dr. Paul Higgins, B.C. Hydro, Burnaby B.C.
- Dr. William Miller, Miller Ecological Consultants, Inc., Fort Collins Colorado
- Dr Peter Steffler, Depart. of Civil & Environ. Engineering, University of Alberta

Comments from all three reviewers were received by ESSA Technologies by Feb. 23rd, 2005 and compiled into this review report.

Summary of Reviewer Comments

General Comments

The reviewers generally endorsed the modeling approaches undertaken for the Hanford Reach assessments and found the exercise interesting and worthwhile. However, although there seemed agreement that the foundation for a useful evaluation approach has been developed, the general view was that a great deal of work will still be required to fully refine and validate the integrated models. Additionally, the reviewers generally found the current report structuring very cumbersome and difficult to follow. All were in agreement that a major reorganization is required to improve context and readability.

Paul Higgins

- The report provides a very detailed description of field and data analysis methods. The descriptions are well done and inspire confidence as to the competence of the investigators to complete the work.
- Overall I believe a good job has been done on this very difficult investigation. My impression is that the work done to date provides an excellent basic foundation for the assessment procedures and with refinement and testing of some of the assumptions over time this evaluation approach will be very useful.

Bill Miller

- The report would benefit by some significant reorganization to clarify and help the flow of the document. As written the report is hard follow with the many section changes throughout the document.
- I recommend that the authors consider splitting the current document into three separate reports. The smaller reports would be more comprehensible and better guide the reader through the objectives, methods and results. With the multiple topics the report is very difficult to follow and I did not see a cohesive link between the physical modeling, results and conclusions.
- Additional graphics of River2D and GIS output would better illustrate results for the reader.
- I recommend a major rewrite of the report prior to finalization. The current organization of the report does not provide the reader with a cohesive document.
- I strongly recommend consideration of three separate reports, one for each study. This would shorten and simplify the presentation for the reader.
- The report also would benefit from an integration that summarizes across all the studies. If it remains a single report, a final chapter that combines all the studies at the population level is needed. The individual conclusions from each study, without relating it to the various study components, do not provide the reader with enough material to gain the conclusions that the authors reached.

Peter Steffler

- The organization of the report could be improved by concentrating the hydrodynamic analyses in the Methods section. The true purpose of the report and the useful results are the habitat and stranding/entrapment analyses. The hydrodynamic analyses are a means to that end and therefore part of the methods.
- The report would benefit from a paragraph or page with more information on the general hydrologic, hydraulic and geomorphologic characteristics of the Hanford Reach of the Columbia. This information would help in framing initial expectations of what is going on.
- One editorial suggestion to improve clarity is to adopt either imperial or metric units for measurements and stick with them

Section Specific Comments

I) Hydrodynamic Modeling:

Paul Higgins

- The application of different data sources to produce a DEM can sometimes be problematic, but not a critical limitation or flaw, if done carefully. It appears this is the case here.
- I think that the approach taken here is sufficient to provide an acceptable DEM for the study area and for the intended use of providing the 'backbone' for hydrodynamic modelling.
- My only concern is that there is a failure to acknowledge that the validation data are incomplete. Ten cross-sections were completed to conduct the validation/model tuning but this represents a very small part of the study area.
- The presentation of the substrate information was minimal and did not provide a good summary of the spatial variation of surficial sediment in the channel.

Bill Miller

- The reliance on secondary reports of previous hydraulic modeling do not allow the reader to independently judge the accuracy of the modeling in this present study. Those reports, since they are one of the key factors in interpreting the flow fluctuations, should be attached as appendices for the reader.

Peter Steffler

- The DEM work looks to have been very thorough and effective.
- A major issue in the hydrodynamic part of the study is the fact that the 2D model was not calibrated. In the end, I agree with the authors that the results are satisfactory despite the lack of calibration. However, the justification is weak.
- The choice of mixed 20m/10m mesh spacing seems reasonable but should be better justified.
- The 2D model validation section is presented in something of a vacuum. This discussion should consider both what accuracy is required for subsequent analysis and what accuracy is realistically obtainable from the literature or similar studies.
- No hydrodynamic validation appears to have been done at the Locke islands spawning area. The Locke islands area appears to be much more hydrodynamically complex than the validation sections and so the overall validation may be questionable.
- In the end, one wonders why the 2D model was used for the entire Hanford reach. Would it not have been more efficient and productive to concentrate the modeling and validation only at the important areas? The data collected represents an enormous effort and will be valuable for some time to come, but it seems to have come at the expense of focusing the study.

II) Entrapment/Stranding Evaluation:

Paul Higgins

- The entrapment evaluation is a technically challenging exercise. I think that the general approach to the investigation is reasonable and technically adequate.
- The methods were clearly described and contained enough detail to understand what was observed during the entrapment studies, known limitations of the assessment procedure and the basic spatial (reach scale) and temporal patterns of entrapment during Spring 2003.
- Overall I think this work is novel and of very high quality but it is clear that several years of study will be required to refine the assumptions of the evaluation and to collect sufficient follow-up monitoring data to confirm that it is providing defensible conclusions.

Bill Miller

- The discussion section for entrapment and also for the spawning habitat was generally lacking in ecological background and also in linking the physical changes to ecological consequences. There was a major effort in the report to summarize the statistics but very little interpretation of those statistics to real world biological consequences.

III) Habitat Assessment:

Paul Higgins

- The methodologies are technically accepted and described in a clear and concise fashion. It provides a very robust and defensible approach to habitat assessment and will provide an excellent tool for evaluation of the effects of PRD operations on chinook spawning habitat.
- Overall, I was very impressed with this component of the work. It is clearly presented and very interesting. I am impressed with the work in that it applies a novel state-of the art approach to habitat assessment but is quite realistic about how far inferences can be pushed.

Bill Miller

- The use of River2D for modeling habitat is appropriate. However, it appears from the methodology stated for spawning, rearing and for entrapment that those habitats are considerably smaller than the mesh size employed in the River2D simulations. Therefore, the simulations for the hydrodynamic model may not be accurately representing the habitats as they exist within the river.

Individual Reviewer Comments

Dr. Paul Higgins
Dr. William Miller
Dr. Peter Steffler

Review of:

Fall Chinook Salmon Habitat Analysis and Stranding/Entrapment Evaluation in the Hanford Reach of the Columbia River Report

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February 2005

"I would like to compliment the investigators on the work. It is a momentous task and they have really done the 'full pull'. Recognizing that parts of the work are incomplete (i.e., stranding omitted) I think the USFWS is well along in getting a defensible approach for evaluating the relative impacts of alternative flow regimens d/s PRD. There are some loose (i.e., uncertain) parts in the evaluation but from the discussion I can see they recognize that and I expect that the plan is to keep the effort up to refine and test the evaluation approach and its key assumptions." (P. Higgins)

Response: *We appreciate the comment.*

INTRODUCTION

The Introductory Section of the report provides readers with the general information about the study area, fishery resources, and hydrosystem layout. The section is clearly written and provides sufficient detail for a reader unfamiliar (i.e., me) with the Hanford Reach to grasp the context of the flow-habitat management problem.

Response: *No response needed.*

Some information that I thought was missing from the Introductory Section was: a) plots of annual flow regimes in the Hanford Reach to show temporal patterns of streamflow at interannual, seasonal, and diel time scales, and, b) more complete information on how the investigations are integrated to support the overarching program for flow and fish population management.

Response: *a) We agree and have added plots/discussion of annual flow regimes demonstrating general patterns at interannual, seasonal and diel time scales to the introduction.*

Response: *b) We added a flow chart that was not yet completed in the draft report that demonstrates how the investigations are related and integrated.*

General flow statistics provided in the report are useful for describing general seasonal flow changes due to flow regulation, but do not convey information about hydraulic conditions or changes in stage at relevant time scales relevant to entrapment and fish habitat use (i.e., hours, days, months) in the regulated state. Typical hydrographs showing diel patterns of dam/discharge or total river discharge for annual periods is helpful for understanding the range of flows under considerations, the expected seasonal changes and interannual variation in river flow. These flow data are also needed for judging limitations of alternative approaches and methods used in the investigations for data collection.

Response: *We agree and have added hydrographs in the Introduction as described above, as well as to other relevant sections of the report.*

The final "Goals of this Evaluation' section could be also enhanced. Clear and useful information is provided in previous sections focusing on hydro-operations and spawning habitat /entrapment issues that help illustrate the overall strategy for management of regulated flows in Hanford Reach for Chinook salmon. However, it remained unclear how the three studies will be integrated (Figure 3 would probably have helped) and how they fit into the overall flow-habitat management strategy (such that they interface with things like escapement goals developed by the Pacific Salmon Commission, as noted in the report).

Response: *The report has been rearranged into three logical sections as each of the reviewers had suggested and the goals for each section have been clarified. The flow chart described above has been completed and added to the report providing a graphical depiction of our field and analytical processes. We have added discussion in the Introduction regarding our work, Chinook productivity, and escapement goals.*

The objectives of the Hydrodynamic Modelling component of the work are clearly stated as: 1) to describe physical and hydraulic conditions in Hanford Reach associated with the range of streamflow variation resulting from the operation of Priest Rapids Dam, 2) to provide information needed to support Entrapment/Stranding evaluations.

Response: *No response needed.*

The objectives for the Entrapment/Stranding evaluation are clearly stated as: 1) to estimate mortality of load following during Spring 2003, 2) to develop a better understanding of the factors affecting stranding and entrapment, and, 3) to provide assessment entrapment/stranding impacts of alternative flow operations for flow management purposes.

Response: *No response needed.*

The objectives of the Habitat Assessment are clearly stated as: 1) to quantify chinook spawning and rearing habitat as a function of streamflow and flow fluctuations; 2) to examine distribution of spawning and rearing habitat throughout Hanford Reach, 3) to estimate the flow required to accommodate different levels of escapement, and 4) to examine the relationship between rearing habitat and entrapment results.

Response: *No response needed.*

METHODS

Hydrodynamic Modelling

The report provides a very detailed description of field and data analysis methods. The descriptions are well done and inspire confidence as to the competence of the investigators to complete the work.

***Response**: No response needed.*

The digital elevation model (DEM) for the Hanford Reach was developed using two LIDAR based survey techniques. CHARTS and SHOALS are both reliable approaches for topographic data collection and have similar resolution and functional limitations. Barring that significant morphological or surficial bed form changes have occurred in the middle study segment (rkm 572-605) in the six years of elapsed time between the development of the SHOALS and CHART field survey then it appears that the data are sufficient to accurately describe the moderate depths portions of the study segments. Additional data were required to deal with missing data in deep water sections, thalweg profiles, and edge/upland portions of the study segments (i.e., depths of <0.1, flows greater than 9,911 m^3s^{-1}). The use of the USACE 10 m DEM for very shallow and upland areas are acceptable for hydrodynamic purposes as the River 2D was run at a 20 m mesh resolution. The application of the BPA white sturgeon cross sections for bottom profiling and to tie to the Washington State plane coordinate system is acceptable under the assumption that the channel has not significantly changed since the data were collected approximately 10 years ago. The application of different data sources to produce a DEM can sometimes be problematic, but not a critical limitation or flaw, if done carefully. It appears this is the case here.

***Response**: We appreciate the comment and have added text comparing the difference between the SHOALS and CHARTS data for a portion of the reach where overlap occurs. Error was found to be within the error of the original data collection specifications. We will continue to improve our DEM as new data are collected for the few areas requiring supplementation (almost entirely within the middle segment). These new data will also be used to validate the supplementary data used in the original DEM.*

I was also looking for some discussion of the issues of scale in the report. In this investigation the spatial extent of the River2D model is relatively large (>50 km^2), however the scale of habitat use is quite small (1-100 m^2). Functional operation of the data intensive 2-d model requires a trade-off between spatial extent and model resolution (i.e., uniform mesh grid size), and this may impact how accurately the model projects local water surface elevation, wetted edges, and water velocity distributions needed for habitat analyses. Recognizing that development of an accurate digital elevation model (DEM) for a 80 km long and ~0.5 km wide reach is a daunting and difficult task, I think that the approach taken here is sufficient to provide an acceptable DEM for the study area and for the intended use of providing the 'backbone' for hydrodynamic modelling.

***Response**: We agree that the discussion of scale is an important issue and have added descriptions of scale in the report in terms of modeling and predicting habitat. Specifically we discuss the scale at which the models are predicting habitat metrics and what scale we can appropriately predict habitat based on the model scale.*

Overall the River2D validation process applied during this investigation was reasonable but has some limitations. Strengths and weaknesses are discussed below.

Strengths of the validation process and data are:

a) Similar methods were used in each of the surveys implemented for validation data, producing similar levels of resolution and data quality (i.e., accuracy and precision).
b) Replicate transects were conducted to allow quantification of measurement precision
c) The hydrometric and water surface elevation data from each of the three surveys are collected over similar flow ranges.
d) The validation data are available over the whole study area.

Response*: No response needed.*

Weaknesses of the model validation approach include:

e) The hydrometric and water surface elevation survey validation data are limited to the lower end of the range of flows simulated. Validation data were collected at flows ranging from 3,681 to 6,230 m^3s^{-1} (~130 kcfs to 222 kcfs), where as River2D flow simulations were conducted from 2,832 to 11,327 m^3s^{-1} (~100 to 400 kcfs). Thus, there is uncertainty about the capability of the model to produce accurate representations of the locations of the wetted edge of the river and velocity distributions above 222 m^3s^{-1}. I suspect there are practical reasons for the lack of validation data above 222m^3s^{-1} (i.e., safety considerations for field staff or low frequency of occurrence), but without it reliability of the results diminishes as simulated streamflows flow exceed maximum validation flows.

Response*: We agree that more representative validation data should have been collected. Limited time and funding restricted our work in this area. As resources become available, we plan to collect additional validation data both geographically and across hydrographic scales. As you point out, we can not verify model output at flows less than 134 kcfs or greater than 193 kcfs. It is important to recognize however, that even though this range does not encompass the entire range of flows we modeled, it does include much of the range of flows that occurred during the spawning and rearing periods for which most of our habitat simulations are made.*

f) The locations of the validation cross-sections appear to be opportunistic rather than strategically placed. There are data available for each of the three river segments, however several of the more complicated bed forms don't seem to be assessed. For example, there is an absence of validation cross sections adjacent to Locke Island which contains a relatively complex bed form (sharp bend, island etc.). One would expect this river segment to strongly influence downstream hydrodynamics. Further, since a goal of the evaluation is to better represent flow-habitat impacts downstream of Vernita Bar the representation of a location where apparently 25% of the chinook population spawn should be considered a priority. Also since the Locke Island site was used for evaluation of the new DOI spawning model, it would be expected that there would be more focus on validation at that important spawning habitat complex.

Response*: We have collected depth and water velocity data at six additional cross sections in the middle segment including adjacent to Locke Island. You will see a map and the validation comparisons in the final report. We still plan to collect additional validation data in the future as discussed previously.*

g) It is stated (p.29, para. 3) that where water surface elevation produced by River2D closely matches the measured water surface elevation in the validation data the model was considered calibrated. This may be useful for modelling depth, however, the parameters of interest are location of the wetted edge of the river (i.e., for determining whether there is flow access to entrapment pools and when they are isolated from continuous flows) and velocity distribution (i.e., for input into spawning and rearing habitat preference modelling). It should not be assumed that when you match water surface elevation the model is producing accurate velocity distributions. Further elaboration as to why water surface elevation was the only criteria may be required here.

Response*: We agree that water surface elevation adjustment using roughness heights is not the only calibration step to conduct. We have added clarification regarding additional calibration steps that should be taken, the steps we took, and the rationale for those steps.*

The application of the 1-d MASS1 model is appropriate and defensible for producing hydrodynamic predictions of water surface elevation in the Hanford Reach. In theory, the steady state application of the model should do a reasonable job of identify the water surface elevation at the 283 m^3s^{-1} flow increments, and for counting entrapment locations within the flow bands. The unsteady application is also appropriate for providing realistic downstream water surface elevation histories for proposed dam operating alternatives.

Response*: No response needed.*

Questions I had about its steady MASS1 calibration were:

a) Over what flow range was the model calibrated? McMichael et al. (2003) was cited for this but I was unable to obtain it. I assume/expect that calibration was conducted over the approximate range of ~1,000 to 11,327 m^3s^{-1}.

Response*: Streamflow data were not presented in McMichael et al. (2003). Only the stage range was cited. The sample sizes cited for calibration of MASS1 indicated that empirical hourly stage measurements were used from eight sites during 1999. Hourly streamflows varied from about 50 kcfs (1416 m^3/s) to 250 kcfs (7079 m^3/s) during 1999. This range does not encompass the entire range of flows we modeled but it does include the entire range of flows present during the fall spawning season and most of the spring rearing season, including during the time period when most of the stranding and entrapment occurs.*

b) Since the simulated flow range was large, was it assumed that Mannings n varied with flow increment, that is, did calibration involve setting Mannings n for each of the 283 m^3s^{-1} flow increments or was a single value selected that minimized WSE prediction errors across all flows?

Response*: The authors did not indicate that Manning's n was adjusted across flows. MASS1 was calibrated by adjusting roughness values "so that the absolute value of the bias [predicted vs observed stage] was less than 0.03 m (0.1 ft) when possible" across the range of flows.*

c) Since roughness is depth dependent and the river channel simulated was very wide, was there any attempt to capture lateral variation in roughness, say between thalweg, marginal or overbank areas?

Response*: As far as we can tell, this was not conducted for the MASS1 calibration. In addition, as far as we know, it is not possible to use multiple n values for an individual cross section in MASS1.*

Questions I had about the application of the steady MASS1 results to produce entrapment histories were:

d) Are the flow increments small enough? Should they be uniform increments of flow or stage? I'm uncertain about how the magnitude of stage change that results from each 283 m^3s^{-1} increment impacts river stage over the range of simulated flows. If the flows are 1,133 m^3s^{-1}, a 283 m^3s^{-1} increment is 25% of total discharge, but only 2.5% of the 11,327 m^3s^{-1}. Would finer increments at lower discharges and larger increments at high discharges provide better resolution of estimation of entrapment risks?

Response*: We chose to model at 10 kcfs flow increments based on several factors. As discussed in the report, mean absolute error from validation of MASS1 water surface simulations ranged from 0.05 to 0.21 m, and averaged 0.11 m. From the MASS1 modeling we conducted for the entrapment evaluation we calculated an average change in water surface elevation of about 0.2 m between each 10 kcfs flow band* *(range 30 – 400 kcfs). We did not want to risk the occurrence of "overlapping" flow bands by using increments that were smaller than the potential modeling error. Secondly, 10 kcfs increments were successfully modeled in the middle segment. By modeling the upper and lower segments at the same increment (10 kcfs), all model runs could be combined for Reach-wide coverage for 10 kcfs flow bands. Lastly, available resources precluded additional modeling at finer increments.*

e) Development of entrapment history requires good hydrodynamic information as well as high resolution topographic data (i.e., the digital elevation model). Entrapment sites likely vary in size (from m^2 to km^2) and morphology (i.e., round depressions, back channel, side channels, alcoves, etc.) and it is unclear that the resolution of the digital elevation model is accurate enough to sufficiently capture variation in bed topographic features associated with entrapment locations. Is it? If so, how was this determined?

***Response**: We did not use the DEM to model entrapments although we recognize some potential to do so. We knew that the DEM did not have a resolution sufficient to model entrapments less than about 16 m^2. We also knew from previous field work that many entrapments were smaller than 16 m^2, so we chose to manually map the population of entrapments with empirically collected field data using a team of 14 surveyors each equipped with a GPS.*

Entrapment/Stranding Evaluation –
The entrapment evaluation is a technically challenging exercise. I think that the general approach to the investigation is reasonable and technically adequate. It appears that the current studies have built upon site-specific experiences and refined entrapment evaluation/stranding assessment procedures. A suitable amount of effort is applied in a strategic manner to sample entrapment locations at the widest possible range of conditions that are experienced and several useful auxiliary studies have been implemented. Below I discuss two general concerns associated with the approach:

a) Failure to consider bar stranding is a significant concern: I am unfamiliar with past assessments of beach stranding in the Hanford Reach so I have some concern about 'placing all evaluation eggs in one basket'. The report indicates that past work has suggested that it is difficult to quantify bar stranding because of a variety of reasons such as predation or difficulty in recovering fish from interstitial spaces. All of these reasons are valid, however even if you cannot find stranded fish on gravel bars it does not mean it's not a problem. Even if the rate of bar stranding is low, the total area where bar stranding could occur is probably much greater than the entrapment locations. I understand the rationale, and agree that 'pothole'entrapment is easier to quantify, but bar stranding should not be rejected as a key mortality cause because you can not measure it. Although the process of bar stranding and entrapment are similar, entrapment should not be thought of as a proxy for bar stranding until there is solid evidence to support that assumption. Does that exist?

***Response**: We acknowledge that stranding is a source of mortality in the Overview of the Entrapment/Stranding Evaluation chapter as follows; "Although fall Chinook mortality is caused by both stranding and entrapment, focusing on entrapments provides a more tractable index for assessing the **minimum** impacts due to flow fluctuations downstream from Priest Rapids Dam." You are correct when you say that our report indicates quantification of stranding would be extremely difficult, however the primary reasons we did not attempt to include a stranding evaluation were time and funding limitations. We did not, and would not propose entrapment as a proxy for stranding. We could not support the assumption that the mechanisms underlying both entrapment and stranding are similar enough to use one as a surrogate for the other. In our conclusions, we identify that "we did not attempt to quantify the potentially significant level of mortality due to stranding." And in our recommendations, we propose to "Develop and implement a plan to estimate the effect of fluctuating flows on **stranding** of juvenile Chinook, and design a statistically rigorous sampling program to survey stranding areas", and "When* *stranding field studies are complete, incorporate an evaluation component for stranding into our modeling evaluation system."*

b) Needs more clarity on how entrapment is defined: I continue to be uncertain how entrapments areas are defined. In the modeling component of the work they appear to be driven by the stage predictions and the digital elevation model of the river channel and floodplain. However, field assessments empirically identify, georeference, and describe the entrapment sites. Are these field surveys a comprehensive inventory of the sites throughout the study area? Are field surveys used at all to improve on the digital elevation model prediction of entrapment locations? Do they include data on size and morphology?

Response: *We have enhanced our detailed description of entrapments. In the report we define entrapments initially in the* ***MASS1 Model Inputs and Simulation Details*** *section as "Entrapments are defined as shallow depressions along the river bank that result in isolated pools. This event occurs in the simulations when flows cover an entrapment on the GIS layer and then recede to a level where the entrapment is isolated from the main river channel". We also define entrapments several times in the* ***"Entrapment/Stranding Evaluation"*** *section under* ***"Total Numbers and Distribution of Entrapments"****. In addition, we have added numerous photos of entrapments. You are correct that in the modeling of entrapment histories, entrapment events are driven by stage predictions (MASS1) and the digital elevation model. The initial result at this step defines the geographic extent of the river bank that was flooded and subsequently dewatered. The geographic locations of the entrapments are then used to create the entrapment history. Locations are identified on the DEM, but the DEM is not used to define the X, Y, and Z of the entrapment.*

The field surveys conducted for our work were intended to be a comprehensive inventory for the entire study area. However, this could only be true if perfect conditions had occurred throughout the entire Hanford Reach for a long enough time period to complete entrapment mapping across the range of all possible flows. These perfect conditions never occurred. Thus, our conclusion, "entrapment enumeration was not complete, likely resulting in under-estimates of entrapment histories."

We did not use the DEM to predict entrapment locations (see response above). We assumed at the time, that the field effort would be the most comprehensive and defensible, so we invested all of our effort and time in that direction. When additional funding becomes available, we plan to use both the DEM and aerial Orthophotography collected in 2004 and 2005 to identify additional entrapments.

All mapped entrapments were placed in size categories, and detailed data on morphology was collected during fish surveys on a subset of entrapments.

The report provides a reasonably thorough description of the methods for each aspect of the entrapment work. However, the following items listed below are worthy of some discussion and possible refinement.

a) Surveys of Diel Behavior: The evaluation of the diel behavior of juvenile chinook is an important component of the work. I'm unclear about the utility of the video surveys. The video cameras are used to make observations that provide basic information about diel behavior and the relative stranding/entrapment risk during different parts of the day. One could argue the assumption that stranding risk is greatest at night (p. 32 para 3). The diel behavior of juvenile chinook has been shown to be highly variable from river to river, impacted by water temperature, food availability and presence of predators. In most conditions chinook salmon typical are quiescent at night in resting locations in low velocity microhabitats. However, it has been observed that they also utilize interstitial spaces within the river channel bottom during both day and night time periods (see Bradford and Higgins 2001; CJFAS 58:365-374). The use of the video cameras can only demonstrate relative time spent in the water column during day and night - not when the fish are concealed in the substrate or other forms of cover when they are most susceptible. One would expect in the larger river situation that there are many predators capable of consuming juvenile chinook and so there may be a predisposition to hide during the day. The fact that, in most cases, juvenile chinook emerge from cover at night and occupy these resting locations suggests that they could be less vulnerable then their counter parts hiding in cover during the day. The point is that the diel surveys need to try to find fish hiding in the substrate as well as those in the water column.

In addition, there should be consideration of sampling at a broad range of sites to explore the possibility of spatial variation in seasonal patterns of diel habitat use.

Response: *We did not have much experience or knowledge regarding juvenile Chinook diel behavior, and funding for this effort was minimal. As a result, we decided to take a very basic, reconnaissance level approach to begin the work. It consisted of observing day/night behavior to see what the fish were doing, and some quantification of basic activities (activity level, position in water column, proximity to shore, movement direction). The goal was to gain enough insight into juvenile behavior so that we could describe relative stranding/entrapment susceptibility, and suggest operational modifications that would help avoid large impacts. The tendency of juveniles to hide, either in the interstitial spaces or in other types of cover are likely driven more by predation avoidance than anything else. The rearing habitats used by juvenile Chinook in the Hanford Reach may not be highly suitable habitats for the dominant predators (Northern pikeminnow, smallmouth bass). Thus, the higher activity level during the day. We did detect significant differences between day/night water column positions and activity levels which suggest more activity during the day, and less at night. You raise several valid issues, primarily variation among rivers and spatial variation within a river, and the level of use of interstitial spaces in the substrate. When we resume these studies, we will incorporate these potential effects into the study design.*

b) Seining Surveys: The objective of the seining surveys should be made more explicit. I expect the objective of this component of work is not to estimate abundance but rather provide a measure of relative abundance, distribution, emergence timing, and fish growth patterns. This information is useful for developing the seasonal and spatial risk profile. My experience with seining is that beach seine catch rate in a large river it is highly variable and that n=15 sites may be too few to adequately capture temporal and spatial changes in relative abundance of juvenile chinook over the ~80 km long Hanford Reach. The use of index sites makes practical sense, however, more extensive sampling (i.e., increase the number of sites sampled, random selection of sites) would be helpful to make sure that patterns of relative abundance and growth are captured with the n=15 site index sampling.

Response: *We have improved the description of the objective for this work. We are in the process of discussing the details for another evaluation in spring of 2007, and we will consider your comments when we develop the study plan for these surveys.*

c) Entrapment Fish Sampling: I think that the approach used in this assessment is quite good and has a very high probability of providing useful information. My only concern was how representative sampled areas were of the Hanford Reach as a whole.

It is expected that entrapment surveys will produce highly variable estimates of entrapment rates and there is some expected uncertainty in estimating the number of entrapment events. Together this suggests that entrapment histories will be highly uncertain, thus further description of how the entrapment estimates (and their confidence intervals) will be derived is warranted. In addition, what apriori analyses are planned for the entrapment survey data?

Response: *In terms of representing the Hanford Reach as a whole, both the entrapment enumeration surveys and the randomized sampling of entrapments for fish took place along the entire 80-km of the Reach. In addition, sampling of entrapments for fish took place throughout the 12-week period of this study, with an average of 52 entrapments sampled for fish for each section/time period combination. We incorporated the uncertainty associated with the number of fish per entrapment and the lethality of entrapments through bootstrapping the data collected during the entrapment fish surveys. We have added language to clarify how our confidence intervals were calculated. While admittedly there is some uncertainty in the number of entrapment events that occurred, we believe that the magnitude of this uncertainty is very low, and therefore we did not attempt to quantify or incorporate the uncertainty in the number of entrapment events into our estimates.*

d) Entrapment Enumeration: The report outlines how entrapments were identified through field surveys, aerial surveys and shoreline surveys. This is obviously a difficult job because of the spatial extent of the Hanford Reach, the need to enumerate entrapments at a wide range of elevations/flow levels (which need to be dewatered to identify) and uncertain dynamic patterns of flow. I think the approach of using aerial surveys at the expected weekly minimum flow levels is probably the best approach. However, the sufficiency of this depends on how low the flows actually were during the Spring 2003 survey period. I'm unclear how the video record of aerial surveys will be collected and analyzed. The use of shoreline surveys helps to ground truth aerial estimates but are limited by the range of flows where entrapment site observations are made.

Response: *Our entrapment enumeration results reflect the extensive spatial scale and dynamic streamflow patterns. The results of our work, including the incomplete enumeration of entrapments are a function of these factors. Aerial surveys are economic and a time-efficient way to estimate entrapment numbers, but extracting data and conducting any analysis is extremely limited without adding some other components to the surveys that would also significantly increase the cost. As a result, entrapment locations were all derived from on-the-ground surveys, and total counts were corroborated with counts from the aerial surveys. We plan to complete the enumeration in the future using a combination of the DEM, aerial Orthophotography, and additional ground surveys.*

e) Estimating the number of fish entrapped: I think this estimation could be possibly be improved by stratification in the estimation process. Field survey data currently serve to parameterize the number of fish that are trapped in depressions on the floodplain per flow reduction event (i.e., mean and bootstrapped variance of fish per entrapment event). The mean number of fish per unit entrapment event trapped is then used to extrapolate a total reach wide loss entrapment estimate for a given flow reduction event. This assumes that size of entrapment area is not important. I strongly suspect that the size/layout of entrapment locations influence the entrapment loss (i.e., number of fish trapped per event). The field data could be examined and, if appropriate, stratified to refine the parameter used to extrapolate reach wide loss. This approach could make an implicit linear assumption (i.e., slope=1, intercept =0) that bigger entrapment areas trap proportionally more fish. However, the field data should help to test or better understand how to formulate extrapolation relationships. Has there been an attempt to assess this linear assumption and estimate slope/intercept to adjust for size of area? Further, the morphology of the channel is important as round depressions in the floodplain would likely have different entrapment rates (and size vs. entrapment relationships) than back channels, side channels and alcoves. In addition, some consideration of the statistical approach for developing confidence intervals for the estimate should be conducted, as this will ultimately be required to support management decisions.

It also was unclear to me how or why the Reduced Area estimate is going to be used in the overall analysis.

Response: *Your suggestion to stratify entrapments based on their size is well-taken. However, it does not appear that substantial improvements in precision would result, and some limitations in the data preclude us from implementing this suggestion with the 2003 data.*

Your suspicion that the size of an entrapment influences the number of fish in entrapments is supported by the field data, but only to a limited degree. Using logistic regression, we found that entrapment size is positively associated with the probability of juvenile Chinook presence in an entrapment, though the relationship is not very biologically significant. For example, the parameter estimates for the logistic equation predict that while a 100 m^2 entrapment had an 18% chance of containing Chinook, increasing the entrapment size to 500 m^2 only increased the probability to 19%. Using linear regression, we found that entrapment sizes were positively associated with the number of Chinook in entrapment, though again the relationship was not very biologically significant. Across the sampling periods, a 10 m^2 entrapment was expected to have 1.29 fish, while a 300 m^2 entrapment was expected to have 1.34 fish. Although these results suggest that some increase in precision may be attained by stratifying by entrapment size, we do not believe that these improvements would be substantial.

Limitations in the data also prevent us from implementing this suggestion. During our comprehensive location and enumeration of entrapments, the sizes of the entrapments were classified into four size categories: 1-5m, 5-10m, 10-15m and greater than 15m diameter. We could have modified our entrapment event history calculations to determine entrapment histories for each of these size classes. However, if we were to apply the fish sampling data to these histories, problems would arise due to the lack of samples from all entrapment size classes in each of the section/time period combinations.

We are looking into ways to stratify the sampling and analytical approach in future assessments. We are continuing to examine approaches where entrapment size is accounted for, as well as stratifying by smaller river sections where entrapment impacts are thought to be higher than surrounding areas.

Our calculation of the reduced-area estimates was intended for comparison to those historically produced for this subsection of the Reach by Grant PUD. While the Grant PUD estimates attempted to account for both stranding and entrapment and our estimates accounted for entrapment only, we believed it was important to put our estimates into context with the alternative sampling approach historically implemented by Grant PUD.

f. Determining Population Level Impacts: While I think it is necessary to attempt to place the stranding losses into a population context, it is a very difficult task. Given the geographic size of the study area, expected abundance of the juvenile chinook population (++ millions), and non-linear compensatory processes regulating survival it is unclear whether this is even feasible. The report correctly acknowledges this (at least some of it) and suggests application of an approach that uses expected smolt-to-adult survival and regional harvest rates estimated through CWT and PIT programs for hatchery and wild chinook to recursively estimate fry production from escapement. More information is required to justify this approach because it is not completely described and this alternative approach is also likely to be fraught with uncertainty. A key concern is that factors influencing escapement (i.e., harvest, upstream passage conditions/difficulties) and smolt-adult survival (outmigrant mortality at downstream dams, ocean conditions, abundance of predators in early marine phases, etc.) outside of the zone of direct influence of Priest Rapids Dam cannot be properly accounted for.

Response: *While we agree that there is considerable difficulty and uncertainty in placing entrapment losses into a population-level context, we believe that our characterizations of the population-level effects have captured a wide range of alternative assumptions on potential impacts. We characterized* *uncertainty in juvenile Chinook abundance by examining alternative assumptions of egg-to-fry survival rates. To clarify, we did not attempt to recursively estimate fry production from escapement. We characterized uncertainty in the potential losses of adults by examining the effects of alternative assumptions of smolt-to-adult survival rates (SARs). These historical data on past SARs incorporate a range of various in-river survival conditions downstream of the Reach, as well as various ocean survival conditions, harvest levels, and upstream passage conditions. Finally, our use of the Chinook Technical Committee (CTC) model for Chinook quantified a range of potential impacts on ocean and in-river fisheries given alternative assumptions on the magnitude of juvenile mortality. The CTC model accounts for recent estimates of in-river and early-ocean survival, and our analysis examined how various reductions in the number of juvenile outmigrants would have translated into losses of adults harvested in ocean and in-river fisheries as well as for escapement.*

g. Evaluating Impacts of Alternative Hydro Operations: The approach for developing simulated hydrographs for flow management used here is a very useful and powerful assessment approach. The approach, however, seems limited by the scope of retrospective assessment. A fuller understanding of the impacts of fluctuations on stage history and entrapment formation should come from examination of many different years and over the flow history. Retrospective analysis based on historical flow data provide a reference point or reference points for evaluation of the possible benefits of future operating scenarios. This reference point is useful because the entrapment assessment approach is not providing absolute estimates of mortality/risk but rather relative assessments of expected mortality/risk. The historical database may be stratified around when significant differences in hydro-operations have been initiated (i.e.,Vernita Bar Agreement, Vernita Plus Agreement, etc.) to track historical performance.

Response: *We agree that retrospective analyses would certainly be informative. While we did not do so in the report, we will be implementing the suggestion to retrospectively estimate the number of entrapments that were created historically in future work.*

h. Evaluating Capacity for Dampened Flow Fluctuations: Given the explicit interdependency and coordinated operation of the mid Columbia Dams it is unclear how this re-regulation analysis was accomplished. Typically, an operational power model is required to route flows using historical inflows patterns, known storage parameters for the key reservoirs, and typical operating rules. From the report it is unclear how the different elements of the analysis are integrated: Was this a simple mass balance approach or was a more formal optimization model applied? Typically an optimization model is required but apparently was not applied here? Why?

Response: *This was a simple mass balance approach. We were not given access to the operational model that was developed pursuant to the Mid-Columbia Hourly Coordination Agreement. This operational model is actually an optimization model as you mentioned. Apparently, if there was a desire to incorporate constraints on outflows (and flow fluctuations) from Priest Rapids Dam, the optimization software that runs the mid-Columbia would have to be modified to include those constraints, then optimize power production given the new constraints. Ideally, we would have been working with Grant County PUD on this project (they are the control point for mid-Columbia hourly coordination) and they would have helped to describe the range of operational configurations that could still meet load while minimizing flow fluctuations. This did not happen, and we do not have sufficient expertise to accomplish this on our own. So we were left with an elementary mass balance approach using outflows from Rock Island Dam, empirical reservoir storage volumes for Wanapum and Priest Rapids pools (for re-regulation), and target outflows (weekly) from Priest Rapids Dam with the goal of eliminating or reducing the frequency and magnitude of the flow fluctuations that currently occur. We have clarified this in the report, and added a recommendation to conduct this exercise using the Hourly Coordination software.*

Habitat Assessment

Overall, I was impressed with the general approach to habitat assessment. It is built upon successful 2-d numerical habitat models for the chinook of the Hanford Reach that employ a logistic regression approach to model habitat preference (i.e., Geist et al. 2000; Tiffan et al. 2002,) The methodologies are technically accepted and described in a clear and concise fashion.

The methods section for the spawning and rearing habitat assessments are very clear and concise, with three exceptions:

One exception is that it is uncertain how the habitat assessments will be applied to assess the spawning habitat required to accommodate various escapement levels (an objective of habitat assessment). The spawning habitat models will provide estimates of persistent spawning habitat available under different flow regimes but additional information on spawning habitat area requirements is required to translate this into supported spawning population abundance. Spawning area requirements are also likely to be density dependent. The model can not provide this information, so: Are there empirical data from the Hanford Reach available to estimate spawning area requirements as is required to make the explicit translation from flow->spawning habitat ->escapement. Clarification of how this objective will be met is required.

Response: *Although the general approach we used for the habitat assessments did not change, the analysis results in the current final draft are substantially different than the draft version you reviewed. We realized that with each round of model-building and evaluation, we learned enough to proceed to the next round and improve results. You will see the current results in your final review of the report. The draft version did not include the methods or protocol for conducting the habitat vs. escapement exercise. The idea was to provide managers with a tool for pre-season planning for spawning operations (flows) in the Hanford Reach. Escapement estimates are available during the summer from the Joint Staff report of Oregon and Washington. Following that estimate, more than 90% of the upriver brights pass Bonneville Dam by October 1. This information would be used to estimate the total number of adults expected back to the Hanford Reach. Historic sex ratio data from Hanford Reach surveys would be used to estimate the number of females expected. For planning purposes, we are assuming that every female will build a redd. We derived a range of values for redd size (including inter-redd spacing) from the literature, and from limited empirical data from Orthophotography in the Reach. The result is the calculated total area required to accommodate all of the female spawners. We then applied our spawning habitat model to a number of different operational scenarios ranging from steady flows to highly fluctuating flows and produced a matrix of habitat areas. From the matrix, any cell with an area equal to or greater than the required area could be selected for implementation. This was originally proposed as the first step towards implementation of the concept of planning for various escapement sizes and configuring the Priest Rapids hydrograph accordingly, while "reserving" the remaining flexibility for power production.*

We have since changed this objective to "begin development of a tool that can be used by fishery and hydrosystem managers for in-season operational planning". Our anticipation is that we would start with a very simple approach as described above. As our empirical database grows and our understanding of how physical conditions relate to habitat (e.g. section-specific models) and productivity improves, we plan to move closer to a process that will allow us to improve the conditions we provide for each years escapement while maintaining an adequate level of flexibility for power production.

Another exception is the computation of the habitat persistence metric. I think that the concept of persistence is an excellent one and the approach for computation is simple and robust. Persistence, however, is regulated not only by hydraulic suitability but also by dewatering from fluctuating flows. The approach explained in the report uses the preference model to quantify the time a given cell exceeds a

50% probability of occupation without correction for dewatering loss. Is dewatering considered in the computation of the persistence metric?

Response: *We have conducted many iterations of modeling towards the goal of producing a biologically plausible spawning habitat model that includes some type of persistence metric given the hourly variation in streamflows. The approach described in the report has been essentially the same, but we have tested many different persistence metrics as well as other measures of velocity and geomorphic variables. You will see the results of this effort and the results from the current habitat modeling iteration in our final report.*

Although our approach is similar, the current Reach-wide model we have developed includes a velocity persistence metric that is the proportion of time velocities are >1 m/s for each cell in the GIS. To determine the proportion of time, the GIS looks at the hourly time series of velocities for each cell and determines the proportion that fits our criteria. It may be possible for velocities to meet our threshold for persistence at a given cell that is dewatered some small portion of the time, but in fact, none of the actual spawning sites, or our predicted habitat included areas that were ever dewatered. Consideration of dewatering is implicit in our computation of the persistence metric because the hourly time series would include that condition if it occurred.

The last exception is associated with the new DOI spawning model. I had two questions: a) How were the n=300 sampling points determined for development of the new habitat preference model? Was this to approximately match the number of redds observed in the orthophotography? or to attempt to sample a fixed proportion of the wetted channel area?, and b) How will the predictive capacity of the two models (Geist et al. vs. DOI) be compared? It seems relevant to apply the models side by side at the Locke Island complex to evaluate overall performance? How will the results be treated?

Response: *The spawning habitat model that you will see in the final report is the result of a subsequent iteration of our model-building process. It is a Reach-wide model (not just Locke Island) and a significant step towards building the more specific site models. The sample size we used was n=5,000.*

Our choice of 5,000 random points was selected to minimize the possibility of pseudo-replication (sampling a redd or use data point more than once) because it produced a nearest neighbor distance of 49 m, which was substantially larger than the size of the redds (17 m^2 to 50 m^2). We also found that using 5,000 points provided reasonable levels of contrast for each of the explanatory variables considered. Details are described in the ***Spawning Habitat Model Development*** *section of the SPAWNING HABITAT ASSESSMENT chapter in the report.*

We evaluated the predictive capability of different combinations of covariates including the Geist et al. 2000 model and spawning occurrence with multivariate logistic regression. We examined multivariate model performance with three types of criteria: model fit statistics, classification accuracy, and biological plausibility. Model fit statistics included the Akaike's Information Criterion (AIC), the G statistic, Nagelkerke's pseudo R^2, and the Hosmer-Lemeshow goodness-of-fit test ($\hat{C}$). The AIC statistic measures the relative degree that the explanatory variables accounted for the variability in redd presence and absence and includes adjustments for the number of parameters that are estimated so as to avoid models with too many parameters or overfitting data (Burnham and Anderson 2002). Lower values of AIC represent better fitting models. The $\hat{C}$ statistic is considered to be a better measure of fit than R^2 because it measures the expected versus actual observations based upon deciles of risk (Hosmer and Lemeshow 2000). The closer $\hat{C}$ is to 1, the better the overall fit of the model. Our classification accuracy criteria were the percent of correctly classified presence sites, absence sites and sites overall. These statistics were calculated by estimating the model parameters for a candidate model, applying the fitted model to the estimation data, and summing the number of correct classifications. Finally, to decide *between models with similar degrees of fit, we qualitatively assessed the biological plausibility of each model. For this criterion, models with a more understandable biological mechanism were deemed preferable to models without clear biological mechanisms. Results of these comparisons are presented in the* ***Multivariate Logistic Regression section of*** *the SPAWNING HABITAT ASSESSMENT chapter in the report.*

Spawning and rearing habitat models were developed in previous investigations so it is not possible within the context of this review to assess whether the biological and physical data were sufficient to meet the study objectives. Both models have undergone significant peer review, however, so it is expected the approaches are technically sound and widely accepted. It may be useful to provide a summary of the field methods used to develop the habitat preference models developed by Geist et al. (2000) and Tiffan et al. (2002).

Response: *We have provided summary information on the methods used to develop these models.*

RESULTS

Digital Elevation Model (DEM)

The results of the DEM work are presented clearly and logically. Well done.

Response: *No response needed.*

Since the DEM is a fundamental component of the overall assessment it would be helpful to present information to demonstrate similarities in the data sources and resolution between the upper, middle, and lower reaches. For example, Table 6 suggests that the density of points collected in the upper reach is ~48% (i.e., ratio of points per Rkm) of that in the upper reach. Providing similar data for the middle reach (and explanations for differences) would be useful to show how we should expect or not expect the resolution of the component parts of the DEM to vary.

Response: *We agree and have provided a comparison of data between the CHARTS and SHOALS data sets. For the upper and lower segments data point densities were similar with approximately 1 point per every 4 X 4 meter cell. The density difference the reviewer comments on is explained by channel width variation between the upper and lower segments. We did not use the original DEM in the middle segment for our analysis, but rather the results, specifically the resultant GIS layers were integrated into our GIS. 10 kcfs shorelines had previously been simulated and as such were not estimated again.*

This DEM is a massive one and a huge technical challenge. I expected a bit more difficulty in the integration of the middle reach DEM to the upper and lower ones, as well as greater need for filling using auxiliary data sources. This was not mentioned, were there other issues associated with integrating SHOALS and CHARTS derived data?

Response: *As mentioned in the previous comment we only integrated the results, depth velocity and WSE grids into our GIS. As the hydrodynamic modeling stands now, three independent simulations were conducted for: 1) the upper, 2) middle and 3) lower segments, respectively.*

Substrate

The presentation of the substrate information was minimal and did not provide a good summary of the spatial variation of surficial sediment in the channel. The presentation of upper and lower reach sediment data leads to an implicit assumption that there was no difference in sediment composition between reaches. Based on first principles and experience in dammed rivers, I expected that without the sediment recruitment from tributaries or upstream there would be some form of reach scale gradient where, on average, large substrates classes would be more dominant in the upper reach and less dominant in middle and lower reaches. Comments in the report suggest that the sediment characteristics are homogenous (i.e., associated with setting roughness). A table presenting % composition by size category (for spawning) or substrate description (for rearing) at the reach level would be useful to better understand the spatial variability or lack thereof.

Response: *We did not intend to provide a comprehensive assessment of surficial sediment throughout the reach. The objective of our substrate tasks were to develop roughness estimates for hydrodynamic modeling and for describing substrates adequate to predict spawning and rearing habitat for fall chinook. As such we added a discussion of substrates relevant to spawning habitat. For all substrates within the 90 kcfs wetted area we looked at the extent of substrates that would limit spawning for the upper and lower segments. Fines, bedrock and boulders accounted for 2.24, 2.13 and 4.39% of the substrates respectively, and summed to 8.76%. Based on this assessment, we did not analyze the spatial variation of surficial substrates in the channel. Also, no sub-surface measurements or assessments of substrates were undertaken. Since no new substrates surveys were made for this research effort in the middle segment, a comprehensive assessment of reach wide substrates would not have been possible.*

HYDRODYNAMIC MODELLING

<u>Hydrodynamic Model (River 2D Validation Results)</u>

This component of the study is presented clearly and comprehensively.

Response: *No response needed.*

The correspondence of model results with ACDP estimated water depths and velocity is encouraging. Further elaboration about why the RM 358 site grossly underestimates local water depths is required. It may be useful to examine (as I looked but did not find) a comparative plot for velocity for the RM 358 site to help pin down this problem. Since there are relative few cross-sections used in the validation analysis at a single flow level, and we must extrapolate our confidence to a large spatial area, it seems worth it to chase that down. It may be something as simple as how/when (i.e., not at steady state?) the data were collected at that site or it might uncover deeper issues problems (i.e., accurately representing local channel bed roughness).

Response: *The most likely cause of the large discrepancy at this site is bathymetry error. The SHOALS system was used in 1998 to collect bathymetry information in the middle segment, and in this particular area, data coverage was lacking for the middle of the river because of depth and turbidity. Thus, bathymetry was interpolated from the adjacent USACE cross sections. The effect of this error can also be seen in the comparison of ADCP velocity data with River2D simulated velocities. We have inserted a brief explanation of this error in the validation results section of the report. This is an example of an area where additional bathymetric surveys are already being planned.*

One editorial suggestion to improve clarity is to adopt either imperial or metric units for measurements and stick with them. The two that are switched back and forth most frequently are: discharge and river location metrics.

Response: *We partially agree and have decided to use imperial units for streamflow (kcfs) and metric units for all other measures. Fishery and hydrosystem managers in the Columbia Basin have always dealt with streamflows in units of cfs or kcfs, and they continue to do so, universally. Since they are one* *of the primary audiences of this report, we want the material to be easily understood and intuitive without having to convert measurements.*

MASS1 Steady-State Flow Simulations

The results for the steady 1d simulation are clearly presented. It may also be helpful to present reach or segment scale rating curves produced by the model. Presenting the general relationship between steady state river flow and water surface elevation should be helpful in interpretation of the observed exposed shoreline analysis. I suggest this to provide a stronger foundation of understanding to meet the objective of providing better information about downstream impacts of regulated flows on fish habitat. Fully acknowledging that the habitat models themselves will provide that assessment, it is always helpful and instructive to understand basic reach level differences in hydraulic characteristics to help explain and support inferences made though the more complicated habitat models.

Response: *We agree that rating curves for the steady state simulations will provide useful information to the reader and will present them in an appendix.*

MASS1 Unsteady-State Flow Simulations

The results for the unsteady 1d simulation are clearly presented and clearly written. It is instructive because it documents what we can expect for an attenuation response. The only improvement I could think of would to present the results of the unsteady simulation in relation to some measured stage data during the 2003 simulation (i.e., add it to Figure 33).

Response: *We agree it would be interesting to present empirical stage data, but the value would primarily be to show how well the simulation results tracked the empirical data at the three locations displayed in the current graphic. Since time is becoming a factor for finalizing the report, we chose not to include empirical data.*

Hydrodynamic Modelling - (River2D)

The results from this section are brief but clear. This section could be improved by, as suggested above for the MASS1 model results, showing the results of a simulation of known flows and measured water surface elevations.

Response: *We agree with your comment. However, there are several factors that make this a difficult task with respect to River2D. River2D was run for steady state discharges, unlike MASS1 and your previous comment. Thus, comparing simulated River2D results with empirical stage data would be difficult. Since the streamflows and corresponding stages are constantly changing in the Hanford Reach, we would have to collect empirical stage and discharge data, then simulate the specific discharge(s) with River2D and compare the output by location with the empirical data. This would be a useful exercise for calibration of River2D, but there is not sufficient time to conduct the work and include the results in this report.*

ENTRAPMENT RESULTS

Video Surveys

The results from the video surveys are presented in a clear and understandable manner. The information does provides a sketch of diel behaviour of young fish, however there are some limitations. These include:

a) Spatial variation in patterns of diel behaviour: What is the influence of the site on the results? The video surveys were conducted at one location (100-F island complex). In smaller rivers, we observe variation in patterns of habitat use associated with sites due to meso-scale habitat configuration and other geomorphic characteristics (i.e., no islands). This may or may not be the case for the large river situation. This warrants further documentation or discussion.

Response: *Time, logistics and funding did not allow us to conduct these studies at multiple sites. Thus it is not known how much spatial variation there might be in behaviour patterns. We have documented this in the report and cited the need under recommendations.*

b) Size dependent changes in diel behaviour: There are some assumptions made in the report about the influence of fish body size on entrapment. We expect that patterns of diel behaviour will be, at least for some part, dependent on fish size. The study was conducted over a relatively narrow window of time during the progression of juvenile chinook from emergent fry to outmigrant life stage. They increased in size rapidly (i.e., 40 mm to up to 90 mm) and the expected shift in habitat use could influence entrapment/stranding risk. Measurements of fish size from seining surveys and from entrapments help sketch the pattern of change in temporal risk, but independent behavioural observations may be useful to further support those indirect inferences.

Response: *This is another aspect of the behaviour work that could have been included with additional time and funding. In addition to multiple sites, we could include temporal stratification that would address effects of fish size. We have documented this in the report and cited the need under recommendations.*

c) Concealment behaviour: In large rivers where other larger piscivorous fish reside, salmonids commonly seek cover in the channel bed substrate. Video surveys capture behaviour of the fish in the water column but cannot infer what proportion of the fish are hiding, and thus at the highest risk of stranding during rapid flow reductions. This is difficult to quantify of course, but is helpful for accurately portraying risk and supporting or refuting inferences about when the fish are at greatest risk of stranding.

Response: *This is another aspect of the behaviour work that could have been included with additional time and funding. We have discussed options for quantifying this, and may be able to incorporate it in future studies. We discuss this in the report and cite the need under recommendations.*

d) Limited range of physical conditions observed limit inferences: The fixed nature of the video gear is a possible limitation. The cameras are excellent for sampling key habitats at a small scale. However, I expect they only give a partial picture of temporal/spatial patterns of habitat use. Juvenile Chinook likely also use habitats (in terms of depths and velocities) outside that which the video system sampled. Typically, this work is done with snorkellers so that the full range of habitat use (micro and meso spatial scales; diel and weekly temporal scales) can be documented for the period that juvenile chinook are present in Hanford Reach. From a practical perspective, explicitly linking habitat use to entrapment or stranding risk requires estimating proportions of fish present undertaking behaviours that increase risk of entrapment or stranding (e.g., shallow depths, substrate concealment). Deriving this proportion requires sampling the full range of habitats used.

Response: *This is another aspect of juvenile behaviour studies that would be enlightening. We have documented this in the report and cited the need under recommendations.*

<u>Seining Surveys</u>

The results from this component of the work are clearly described and well documented. The results of the work appear robust and variation is reasonable given the nature of the sampling protocols and intensity. The trend is clear.

My only question was: Was there any attempt to quantify intra-site variation (i.e., replicate sets across a homogenous gravel bar). I would expect that in large juvenile fish distribution is patchy (likely cover dependent) and it is conceivable that this could contribute to more variation than was observed in the catch data. How much this would increase total variation - I do not know, but don't think this would alter the inferences about relative abundance through time.

Response: *There was no replicate sampling. These surveys were intended to provide qualitative distribution data through time, and quantitative fish size data through time.*

Entrapment Fish Sampling

This section was clearly written and contained enough detail to understand what was observed during the entrapment studies, known limitations of the assessment procedure and the basic spatial (reach scale) and temporal patterns of entrapment during Spring 2003.

A hounding question in my mind is: Are all entrapment features created equal? This assumption is a fundamental one in the assessment process. The large sample size of entrapment locations should promote a comprehensive database of observations of entrapment rates and physical characteristics that would likely be useful in helping to better understand the entrapment process and the physical factors that are strongly correlated to it. For example, Figure 38 shows mean/S.E. for chinook entrapment by date and river reach. This gives some impression that in some cases (i.e., Middle Reach on March 30 and on April 27) episodic increases (in the order of magnitude) in entrapment rate occurs; this is also accompanied by increased variability in mean entrapment. Examining whether this is correlated to some physical attribute (area; pothole vs. sidechannel vs. back channel; cover vs. no cover; etc.) should help develop better understanding of what is driving entrapment or whether these are anomalies due to contagious fish distribution. I expect the average C/E (Chinook/Entrapment) parameter for extrapolating results in a general way, such as choosing what operating scenarios, will likely minimize entrapment mortality. However, I suspect there is value in some stratification that can be employed to provide more accurate and precise projections of number of chinook entrapped. The two apparent outliers on Figures 40, 41 and 42 may the first place to go looking.

Response: *We have added analyses examining the relationship between entrapment size and the probability of containing Chinook and the expected number of Chinook in an entrapment. While these analyses found statistically-significant associations between entrapment size and the probability of an entrapment containing Chinook and the number of Chinook in an entrapment, we did not interpret them as particularly biologically significant. Nevertheless, in future work we will be considering stratification of entrapments based on their size, type, and physical characteristics as well as finer-scale spatial stratification, in an effort to maximize the precision of the impact estimates. We have cited this need in the recommendations.*

Estimating the number entrapped and mortalities

The results from this assessment are clear and well done.

The very high estimate of entrapment for Apr 27 to May 10 sticks out, and it drives conclusions. This underscores the need to closely understand what went on during that period in that area. It may point to significant entrapment risk features, as suggested above. More explanation to confirm this assumption of the estimation procedure and why this occurred would be helpful.

The mortality rate assumption is a critical part of the entrapment evaluation approach. I am not clear how the 82.3% at risk mortality (cited from McMichael et al. 2003) was developed. Decimal place accuracy implies that this was very rigorous. Did it include the effects of release from entrapments associated with upramping? Load following will cause the river flow to increase and decrease. The water surface level increases will release entrapped fish before they face thermal, predatory or desiccation mortality factors. This time lag effect will differentially apply across flow bands (i.e., expect lower mortality in lower elevation flow bands) and river segments. If the 82.3% estimate is a mean across flow bands, it will overestimate mortality at lower flow bands that see more water/dewater events and likely have greater 'releases rates'. Given that assumption, is that 82.3% mortality rate more appropriately considered a maximum mortality rate?

Response: *While the estimate of mean Chinook per entrapment during the Apr 27 – May 10 period was high, the estimate was also high for the Mar 30 – Apr 12 period. We have added analyses describing the associations between entrapment size and the probability of containing Chinook and the number of Chinook in an entrapment. While we found statistically-significant positive associations, the biological significance of these associations is questionable. Our interpretation of the data is that the processes which affect entrapment rates are highly variable, both spatially and temporally, and that the physical characteristics of individual entrapments contribute only a small amount to this variability. However, we are looking into finer-scale spatial stratification as a means to help account for the observed variability in future work.*

We have clarified our approach for estimating entrapment lethality. Our approach does account for fish escaping from entrapments following upramping, as entrapments which reflooded were not counted as being lethal. Also, the average effects of entrapment locations on reflooding across the flow bands were represented in the entrapment lethality estimates. Our approach also accounted for spatial (separate estimates for the upper, middle, and lower segments) and temporal (by sampling period) changes in entrapment lethality.

<u>Estimating the impacts associated with the alternative hydro operations</u>

The results from this work are clearly presented. The results are useful in providing general direction about the impacts of a broad range of operation scenarios on expected entrapment.

The logical and useful extension of the analysis is to develop a retrospective history of entrapment and estimated losses based on historical flow data from Priest Rapid Dam. Examination of the historical range of entrapment moralities (in comparison with chinook stock assessment information) would provide a reference point from which future flow planning scenarios could be compared. It would also be interesting to compare that to the URB chinook population stock assessment data to test whether the population varies coherently with the entrapment mortality estimates.

Response: *The reviewer's suggestion is well-taken and we are planning on conducting this research in the future. We are planning on running historical flow data from Priest Rapids Dam and calculating the number of entrapment events that occurred each year during the rearing season. The resulting data could provide a useful index for explaining residual variability in the stock-recruit relationship for this population. However, the lack of historical data on the number of Chinook per entrapment may limit the* *amount of residual variability that can be accounted for. While we are confident that the field data captured the patterns in Chinook per entrapment in 2003, we are not confident that other years would display the same patterns, and therefore applying the 2003 biological data to entrapment histories in previous years may not be appropriate. If we are allowed to conduct future evaluations, we may be able to describe general patterns in Chinook per entrapment across the Reach and over the season, which would help in indexing the effects of historical operations.*

In addition, using run reconstruction of URB catch, escapement, and age data, we have developed a time series of parents and progeny for this population. We are evaluating best-fit recruit/spawner (R/S) models, and we plan to examine the pattern of residuals from these models relative to the retrospective history of entrapment and estimated losses of juveniles.

Evaluating capacity for dampened flow fluctuations

I am uncertain about this part of the work. It is unclear why, when the objective of the analysis is (p109, para 2 "evaluate the physical flexibility available to reduce streamflow fluctuations during the March through May rearing period..."), that the Oct-Nov period is used. Is this an error or did I miss something? It seems to me to be the wrong time of year.

In general, this is not my area of expertise but I also wonder whether the 1995-2004 dataset provide a broad enough set on inflow and power demand conditions to provide reliable results about operational capacity? Ten years is not likely sufficient to describe hydrological variation or variation in patterns of system operation, and I'd be also looking in detail at those operations to makes sure some other upstream or downstream component of system operation did not artificially constrain PRD or WAN operations.

Response: *Also see our response to your comments on the "Methods" section.*

Your question on time period: Obviously, in this case, the evaluation of physical flexibility has to do with reducing flow fluctuations during the spring rearing period. Part of doing this analysis was determining the storage volumes available for re-regulating incoming flows. We chose to use empirical data for this rather than making assumptions regarding availability or usage of active storage. We chose to use the October/November time period to determine storage volumes because of the voluntary operation that is conducted every year for spawning Chinook. Grant PUD turns around the normal load following hydrograph (high during the day, low at night) and implements reverse load following (low during the day, high at night) to limit Chinook spawning to lower elevations in the channel. In order to reverse the incoming hydrograph, they have to catch and hold higher daytime flows, then release them at night. This requires much more volume in the pools than operations during any other time of year. So we used this time period and the associated empirical data to calculate re-regulation volumes (or storage) that might be available, and assumed that the same pool manipulations were possible during the spring rearing period.

*Normally we would agree that 10 years is not usually sufficient to characterize hydrologic variation. However, the 1995-2004 time period included both drought and flood years (98% and <1% exceedance probabilities based on the 1970-2005 post-Mica time period), and everything in between with a rather typical distribution. So it may not have been perfect, but we think it was very reasonable as far as capturing variation. As we discussed earlier, our expertise on the details of operational and power demand issues is very limited. As far as we could tell, the only constraints on PRD and WAN were meeting load. We attempt to be very explicit regarding the description of our analysis, in that it is **only** a **physical** (mass balance) analysis, and does not attempt to consider operational issues. We have added clarification to the report.*

FALL CHINOOK SPAWNING ANG REARING HABITAT ASSESSMENT

Chinook Salmon Spawning Habitat Modelling

Overall, I was very impressed with this component of the work. It is clearly presented and very interesting. It provides a very robust and defensible approach to habitat assessment and will provide an excellent tool for evaluation of the effects of PRD operations on chinook spawning habitat. Well done! Below I transcribe some of my marginal comments.

a) Acknowledge seasonal changes in habitat selection processes – Using the 1 week (i.e., 169 h) time series to develop/tune the revised model may (or may not!) have limitations. The model is tuned to the period just approaching the peak (just prior to Nov 5) and thus may be better suited for describing higher density situations rather than early or late components of the run. If suitable and persistent habitat is limited the presence of high densities of spawners could influence habitat selection. The point is that we should expect, in addition to the dynamic nature of the flows [on a temporal (diurnal, season) and spatial scale], habitat selection processes are also dynamic and likely change over the season based on what is available and not occupied.

Response: *We have completely re-done the habitat analysis, although the approach was very similar. Your main point here seems to be the seasonal aspect of the hydrograph and the habitat selection process. We did not have sufficient time and funding to conduct this analysis for different portions of the spawning season. The hydrograph over each 24-hour period is extremely repeatable, with almost no variation in pattern, particularly downstream near White Bluffs. We would not anticipate much of a difference in the spatial distribution of physical conditions between early, mid, and late season time periods. As far as habitat selection, you make a good point. We were not able to discriminate between early, mid, and late season redds with only one set of orthophotos, so it would be hard to detect changes in habitat selection as density in spawning areas increases. We have made several observations over the course of three years of photos that are related to the subject. The redd polygons are relatively consistent in size and location each year. And the density of redds within those polygons has varied with escapement. In other words, during the low escapement year the redd polygons were similar and there were larger areas of inter-redd spacing, and during the high escapement year, the polygons were similar and there was almost no inter-redd spacing. We have cited the need for additional Orthophotography in the recommendations section to examine within season variation in spawning activity.*

b) How is dewatering treated in the persistence metric? Notwithstanding reverse block load operation to prevent redd dewatering, I expect it still occurs. Is dewatering considered in the persistence metric? Does a dewatering event set the habitat polygon p(use)=0? This is further complicated by the fact that subsequent operations could make this habitat polygon useful persistent habitat. Some explanation is required to clarify.

Response: *See our earlier response to your comment on persistence and dewatering. We have tested several different measures as persistence metrics. At this point, the persistence metric with the best predictive capability is the proportion of hours (over all 24-hour periods) that a cell velocity is > 1 m/sec. The 24-hour periods are comprised of the 11 days leading up to peak spawning. The calculation of this metric requires examination of all hourly velocities for all cells, and if any are 0 (dewatered), they are included in the computation of the proportion. As we said previously, we have found that most areas that become dewatered at some point, for some amount of time, are not used for spawning. In addition, when we examine water surface coverages for the lowest flow that occurred in the time series and compare it to the redd polygons in the GIS, it becomes obvious that very few, if any, of the redd locations were subjected to dewatering. We have provided additional clarification on this issue in the report.*

Chinook Salmon Rearing Habitat Modelling

This component of the work is clear and concise. Almost too concise. However, since this is based on the well accepted model that has been previously peer reviewed it is acceptable to make this a minor component of the work.

Response: *None required.*

Do the results correspond to the original state objectives?

Yes, with the exception of the re-regulation analyses (discussed immediately above) the results represent correspond to the stated objectives of the work.

DISCUSSION

Hydrodynamic Modelling

The discussion provides a realistic dialogue on the applicability of the results in relation to the overall study objectives.

My only (picky) concern is that there is a failure to acknowledge that the validation data are incomplete. Ten cross-sections were completed to conduct the validation/model tuning but this represents a very small part of the study area and in some respects is not directly compatible for the 2d results (i.e., cross sections measure perpendicular flow vector strength but 2d by definition allows for deviation from that by incorporating lateral vectors). Furthermore, this was not done over the full range of flows that the RIV 2d model was used for. I still think the work that was done is very good, but this could help improve the model and avoid simplifications such as assumptions of spatially homogenous or flow independent bed roughness effects.

Response: *We agree with the reviewer and have disclosed in the conclusions that a lack of comprehensive validation and subsequent calibration is a potential limitation of this study. We plan to collect a spatially comprehensive validation data set both geographically and across the hydrographic scale (50 – to 250 kcfs).*

Entrapment and Stranding Evaluation Discussion

The discussion provides a good evaluation of the strengths and weaknesses of the entrapment evaluation, and produces logical conclusions. With respect to recommendations, all of the recommended actions are useful and will improve the evaluation approach. Overall I think this work is novel and of very high quality but it is clear that several years of study will be required to refine the assumptions of the evaluation and to collect sufficient follow-up monitoring data to confirm that it is providing defensible conclusions.

Response: *No response required.*

Spawning and Rearing Habitat Evaluation

The discussion provides a good evaluation of the strengths and weaknesses of the spawning and rearing habitat evaluation. With respect to recommendations, all of the recommended actions are useful and will improve the evaluation approach.

I am impressed with the work in that it applies a novel state-of the art approach to habitat assessment but is quite realistic about how far inferences can be pushed. This is not always the case! Too many times this type of approach is used with out full acknowledgement of other factors that drive habitat selection and how that should be interpreted in a biological sense. To that end I believe there is a need to conduct more behaviour studies of spawning chinook salmon. Two important uncertainties identified were: 1) to confirm or refute assumptions about diurnal spawning behavior; and 2) examine behavioral energetics of spawning chinook in fluctuating flows. Better understanding of diurnal activity patterns and energetic requirements will help to better define the dynamics of habitat selection

Response: *These issues are included in the recommendations section..*

<u>Does this work provide an analytical set of robust tools to evaluate operational alternatives for the dams in question, to minimize entrapment impacts, and to optimize spawning and rearing habitats?</u>

While the assessment approach may have some uncertainties it is presented in an explicit way and provides the foundation for effective evaluation of alternative operating plans on relative entrapment impacts.

Overall I believe a good job has been done on this very difficult investigation. My impression is that the work done to date provides an excellent basic foundation for the assessment procedures and with refinement and testing of some of the assumptions over time this evaluation approach will be very useful. Based on that I suggest the priorities are:

a) Complete the entrapment evaluation model:
   - Complete data collection for all flow levels and spatial area to complete the physical models for entrapment evaluation.
   - Assess the most critical assumptions and develop a plan for systematic refinement and testing of those assumptions.
   - Conduct follow-up monitoring to validate predictions

As an example of critical assumptions: The two main assumptions in the entrapment evaluation that I had the most trouble accepting were: 1) fixed C/E irrespective of entrapment size or structure, and 2) mortality rates are independent of entrapment location (i.e., elevation flow band). These are critical driving parameters for the entrapment evaluation.

Response: *We agree with your suggestions, and have included them in our recommendations. We are planning on evaluating critical assumption 1) by stratifying on entrapment size and other physical characteristics. We are also planning on examining time to re-flood to address critical assumption 2). Drainage and lethal temperatures are the primary sources of mortality, and these are affected by the time required to re-flood at various locations within the flow band that was originally dewatered. The size of the flow band also determines whether there is a significant difference in mortality rates among individual entrapments.*

b) Develop a parallel approach for assessing stranding losses.

This will be more challenging to do well. However, I agree with the report in that it is plausible that stranding losses could be as high as entrapment losses. Also that it is not clear whether the processes that cause each type of loss are similar and that stranding or entrapment losses are correlated in anyway. Until that is clarified I don't think it will be easy to develop flow regimes below PRD.

Response: *We agree that this is extremely important and likely difficult, but at least some progress needs to be made on the issue of stranding. This issue is also included in our recommendations.*

c) Develop or integrate with a more realistic system operation model to permit more informed gaming with 'the system':

I acknowledge that the simple regulation analysis is the first step in examining the flexibility within the system to mitigate flow fluctuation and associated impacts. However, it is very difficult to be very realistic without some consideration of power demand, system capacity/response characteristics, and full accounting of up and downstream constraints. I simply assumed that this will occur at some point in the near future, if not available already.

Response: *We agree that it would be useful, enlightening, and a required integration of reality to be able to evaluate a range of scenarios using the Hourly Coordination Agreement optimization software. Since it is beyond our expertise, we hope the mid-Columbia operators will see the value in this and help us move forward.*

GENERAL FORMATTING

In general the use of tables and figures in the report was adequate and effective. The only minor problem was the omission of Figure 3. Judging by its caption, it appears to be a key piece of information to help reviewers integrate the analytical approach to the habitat evaluations.

Response: *Figure 3 is now Figure 10 and it is complete, and additional figures and tables have been added throughout the report for clarification.*

Review of:

Fall Chinook Salmon Habitat Analysis and Stranding/Entrapment Evaluation in the Hanford Reach of the Columbia River Report

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"I felt the approach was sound, however, I am probably somewhat biased towards 2D models. The combination of 2D hydraulics with a GIS seems like a natural progression from the standard PHABSIM. The approach provides the tools to graphically display habitat and river hydraulics in a manner that is readily understood by those familiar with river hydraulics and those who are not. A flow chart to show the conceptual approach and the sequence of analysis would be helpful to those not familiar with the approach. We have applied 2D hydraulics and GIS habitat analysis in several different river systems. We have developed a similar GIS tool to quantify habitat based on habitat suitability criteria (depth and velocity). The model components used in the Hanford Reach are very similar to our approach on the Flathead River for BPA. Since the main objective of the entrapment study was to determine how often and at what flows the entrapments formed, it seemed like there should have been more comparison between the measured water surface data and simulated water surface data at the flows where water surface was measured. That comparison would strengthen the link between formation of entrapments and chinook mortality. A similar comparison could be made for the spawning evaluation to show redd dewatering (i.e. non-persistence) after construction."(B. Miller)

Response: *We appreciate the reviewer's comments and agree that combining 2D hydraulics with a GIS is a natural progression from the PHABSIM approach. The reviewer comments on the lack of "a flow chart to show the conceptual approach and sequence of analysis". We have added a flow chart to show the conceptual and analytical approaches employed in this study.*

The reviewer discusses the need for more comparison between measured and simulated water surface elevations for the entrapment study. The entrapment analysis uses MASS1 to determine water surface elevations for development of the subsequent entrapment histories . MASS1 was previously calibrated and validated for the Hanford Reach with empirical stage data from several locations. The calibration and validation process and results are discussed in the report along with the citation where the reader can access all of the detailed discussion.

The reviewer also discusses comparing simulated and measured water surface elevations with regard to redd dewatering after construction. Dewatering of established redds has not been a problem for almost 20 years following implementation of the Vernita Bar Agreement. However, the concept of persistence has proven to be a functional interpretation of the effect of highly variable hourly flows during redd site selection. One observation that can be made from our work is that although persistence may not mean suitable conditions 100% of the time, it also does not include dewatering of spawning sites any of the time. We determined this from the hourly record of water depths at actual redd sites.

The report would benefit by some significant reorganization to clarify and help the flow of the document. There is a mixture throughout the introduction and methods sections of narrative that includes methods, discussion and results. Separation of those items to the various sections as appropriate would benefit the report. The report should include numbering of section headings and subsection headings to better inform the reader of location within the document. As written the report is hard follow with the many section changes throughout the document. The results and discussion sections should relate the objectives to the results found in the study period, as it is not clear as written whether those objectives were met.

The report would benefit from a concise well-written executive summary that summarizes all parts of the study. I recommend that the authors consider splitting the current document into three separate reports. The smaller reports would be more comprehensible and better guide the reader through the objectives, methods and results. With the multiple topics the report is very difficult to follow and I did not see a cohesive link between the physical modeling, results and conclusions.

Response: *The reviewer makes a legitimate comment and we have chosen to rearrange the document into three logical chapters along with an executive summary, conclusions, and recommendations.*

INTRODUCTION

The introduction section would benefit by some reorganization to quickly orient the reader as to why the study was undertaken, and to provide better presentation of background material. The objectives are not explicitly stated in the introduction. There are goals stated but no specific objectives. It is difficult to understand the overall intent of the study. I assume that it was conducted to look at some existing baseline condition and possibly propose flow manipulations for future operations of the Priest Rapids Dam and other dams on the river.

Response: *The Introduction section has been enhanced and details regarding why the study was undertaken are discussed. We have listed the three goals of the study in the Introduction, and listed the specific objectives for each of the goals in the relevant sections.*

METHODS

Hydrodynamic Modelling

The hydrodynamic modeling section needs a statement of specific objectives. In addition, this section and the other sections of the report would benefit by using numerical headings to orient the reader to subsections contained within each major section of the report. A report of this length requires a better road map for the reader, especially since there are multiple sections with somewhat similar headings.

Response: *We have re-organized the report into three "chapters" each with it's own explicit set of objectives in the opening section of the chapter. We have also added new headings/subheadings for each section, along with listing them within the header of each page for orientation. In addition, all sections are hyper-linked from the Table of Contents.*

The hydrodynamic section would benefit by a summary table of dates, discharge and data collected for each of the various methods. In addition, a discussion of the accuracy or error of each of the measurement methodologies would help the reader to understand modeling accuracy. The different spatial scales associated with each of the acquisition methods for the topography and bathymetry affect the overall modeling. There is no discussion of expected model error in the methods section. The hydrodynamic section should include a discussion of acceptable model error.

Response: *We believe that we have provided a sufficient level of detail regarding the various methods used to collect data for the DEM. We do not think additional detail will add to the reader's understanding, although we have added discussion of the details and accuracy for the dominant data sources (CHARTS, SHOALS, deep water survey) used to build the DEM. We have also added discussion of expected model error.*

Since hydrodynamic modeling requires several interpolations to get the final mesh for the model, each of the measurement accuracies will impact generation of mesh. The discussion of accuracy is important when small changes in water surface may be within the error of the interpolation and measurement and therefore may not be accurately reflected as a real change in water surface.

Response: *We believe that the CHARTS survey which accounts for over 97% of the data used in the DEM has provided a very comprehensive and accurate DEM, particularly considering the scale of the Columbia River. As such, mesh error for the upper and lower segments is not expected to mask real changes in parameters such as water surface elevation, particularly following calibration and validation measures. We could conduct a sensitivity analysis to determine the real effect of DEM data measurement accuracy, but not within the timeframe of this report.*

The 1-dimensional hydrodynamic model creates flow bands for flow fluctuations. The use of that information for the entrapment study may be influenced by local physical conditions that have an impact on those entrapment sites that are not accurately reflected by the 1-dimensional model. The use of 2-dimensional models for local hydraulic changes has been documented in studies on spawning habitat in other rivers. Studies by Mussetter and others in 1993 showed that the hydrodynamic model provided information on a spatially small area that a 1-dimensional model was incapable of predicting. Consideration should be given to adaptation of the River2D model to localized areas of entrapments, especially since the stranding of juvenile and young Chinook in those areas is of importance. The local hydraulics in those sections could be modeled more explicitly with River2D.

Response: *We agree that a 2-dimensional hydrodynamic model such as River2D produces more accurate simulations of the physical and hydraulic environment than a 1-dimensional hydrodynamic model given similar calibration quality. However, the time and computational resources required to run River2D in an unsteady state for a river the size of the Columbia, for a three-month period on an hourly time step were prohibitive or not available. We believe that the 1 dimensional MASS1 model, in combination with the highly accurate DEM we created, was sufficiently accurate and suitable for generating flow bands.*

The use of River2D for modeling habitat is appropriate. However, the mesh size generated in River2D is either 10m or 20m mesh. That spatial element size should be determined from the spatial accuracy needed for the habitats under evaluation. It appears from the methodology stated for spawning, rearing and for entrapment that those habitats are considerably smaller than the mesh size employed in the River2D simulations. Therefore, the simulations for the hydrodynamic model may not be accurately representing the habitats as they exist within the river. A test of mesh size against model output and habitat predictions on some very small river reaches could be used to test whether the mesh size is suitable for the study.

Response: *The reviewer brings up a good point. We added clarification to our habitat assessments that supports the use of dual 10 and 20 m meshes. At the most resolute scale our models simulate habitat metrics (depth and velocity) near the maximum redd size, approximately 10 m in diameter. While our bathymetry is twice as resolute, 10 m is a biologically justified goal and a realistic goal considering the overall scale of the entire Hanford Reach (90 km). In fact, redd clusters approximate the true scale for which our habitat models will be predicting habitat. Redd clusters are groups of individual redds typically numbering in the hundreds and in 2004 mean redd cluster size was 6.38 ha, much larger than a 10 or 20 m mesh size. With respect to the entrapment evaluation, we did not use River2D to model entrapments or entrapment characteristics. We used MASS1 and the DEM to identify shorelines for flow bands.*

The reliance on secondary reports of previous hydraulic modeling do not allow the reader to independently judge the accuracy of the modeling in this present study. Any previous results relied on in this study should be summarized in tabular format if possible and included in the methodology or results section.

Response: *We have cited Tiffan et al. 2002 and included their validation information in the accuracy table in the* ***River2D Validation Results*** *section of the report.*

Entrapment/Stranding Evaluation

The entrapment and stranding evaluation had specific objectives stated within it and the approach appeared to be appropriate. Impacts to other biota, which may in turn have an impact on the rearing Chinook, within the study reach seemed to be overlooked. There is no discussion given to the dewatering or river fluctuation impacts to primary and secondary productivity (i.e., potential food sources for young salmonids). It appears that impacts to these factors were not considered or possibly overlooked. I recommend that these factors be covered in the discussion section of the report.

Response: *We agree with your comments. We identify primary and secondary productivity as an issue in the Introduction, and we provide a short discussion in the Discussion section of the report that identifies resident fish species, secondary productivity, and food resources for juvenile Chinook as also being impacted by flow fluctuations and the flooding/dewatering of the littoral zone. We acknowledge that we did not have the time or resources to conduct quantitative analyses for these impacts.*

The entrapment study methods are somewhat clearer than the hydrodynamics section; however, the use of tables to summarize sampling dates and locations could reduce the amount of narrative and provide the reader with a better understanding of study methodology. A photograph of a typical entrapment site would be useful to orient the reader.

Response: *We have added a flow chart (Figure 10) to illustrate the components of the entrapment study as well as the other study components. We have added additional photos of entrapment sites throughout the report.*

Since there are different flow regimes at night and day, was night sampling considered? Is the study reach too dangerous for work after dark? Diel changes in habitat use have been documented in other studies. It appeared that all sampling occurred during daytime with some mention of problems resulting from the delay between when actual sampling occurred and when the entrapment had changed in water level. Were the data segregated temporally to test for differences caused by sampling time? It may be appropriate to consider nighttime sampling or at least a test of that to see if it results in a different evaluation of entrapment than the daytime study.

Response: *Your basic point of day/night differences is a good one. Flows are generally low in the early morning, followed by a sharp increase at daylight to mid-morning, then increasing throughout the day into darkness, with a sharp drop usually around midnight. The result is that entrapment formation begins just below Priest Rapids Dam shortly after midnight, then proceeds downstream with entrapment formation starting in the early morning in the middle of the Reach, and about daylight in the lower Reach. Navigation in the Hanford Reach during darkness is dangerous, particularly if personnel are not extremely familiar with the Reach. That was the overwhelming factor behind the absence of night sampling. It would be interesting to investigate diel changes in habitat use, although most of the changes we are aware of were associated with much smaller rivers. We have added this investigation to our recommendations section. We did not segregate data temporally as a function of sampling time because all sampling was conducted during daylight hours. Your comment on nighttime sampling is relevant, and we are planning on conducting this type of test with experienced personnel and advanced navigation equipment when we conduct the next entrapment/stranding evaluation in spring 2007.*

The discussion on evaluating the impacts associated with alternative hydro operations needs revision. There is a lengthy discussion of how a simulated discharge or a simulated hydrology for the reach was derived. It is unclear from that discussion how it was to be used. Is there an evaluation of the current baseline being conducted? If so, that should be made clear in the document.

Response: *We have provided additional clarification regarding these alternative hydro operations and how they relate to the baseline. We assume that your reference to the "current baseline" refers to actual conditions that occurred in 2003. There is no other current "baseline". The primary focus of this work was to evaluate the actual conditions that occurred in 2003. The lengthy discussion on development of alternative hydrographs was necessary for the reader to follow the process from actual to alternative hydrograph. The main point behind this exercise was to evaluate the effect of a similar load following pattern on entrapment of juvenile Chinook only with less severe fluctuations, and compare it to the 2003 actual operations which represent the baseline.*

On page 48 there is a discussion of using 50% exceedance values for daily and weekly forebay fluctuations. I caution the use of hydrology exceedance values for biological evaluations. The biota will experience the hydrology in real time; therefore, discussion of how certain flows occur by exceedance level may not be directly related to the response to the biota. That should be taken into consideration when viewing results. Due to selection of certain exceedance levels, while acceptable from a flow management standpoint, made it convenient but difficult to interpret the impact to biota.

Response: *In this application, the 50% exceedance value was selected to determine the "average" forebay volume that was used to re-regulate streamflows during the reverse load following operation conducted by the PUD in October/November. More forebay volume is used during this time of year than in any other time of year. We assumed this capability also existed for use during spring operations during the rearing period. The reduced flow fluctuations that resulted from this analysis represent the relative (reduced) biological impact associated with that level of fluctuations. This relative impact can be inferred from Figure 60. Our analysis was meant to be an evaluation of the potential re-regulation capability that exists, and the relative, reduced biological impact. Thus, the exceedance analysis was totally a "water" exercise with the goal of reducing flow fluctuations generated by Priest Rapids Dam. The analysis was a physical one with implicit connections to a reduced biological effect on juvenile Chinook.*

Habitat Assessment

The spawning and rearing habitat assessment has several places where objectives are stated. Those should be combined into one section. There are objectives stated in the first paragraph and in the overview paragraph of the spawning habitat modeling, and in the rearing habitat assessment. I suggest listing them all in one area.

Response: *Your suggestion is relevant. We have re-organized the report into four chapters with the Spawning Habitat Assessment and Rearing Habitat Assessment as separate chapters. We have consolidated the objectives for each chapter in the respective Overviews.*

The spawning habitat model and the rearing habitat model use the River2D hydrodynamic results in a GIS framework. Conceptually that approach is technically adequate. The determination needs to be made on whether the mesh size simulated in River2D appropriately represents the habitats evaluated. The accuracy of the field measurements impact overall accuracy of the DEM used to generate the mesh for River2D. A mesh size of 10m may result in a much larger area than specific habitats of interest, especially if localized hydrodynamic conditions would influence placement of a redd for spawning. Mesh size may need to be further refined in some areas, especially in known spawning areas, to better reflect the local hydrodynamic conditions. A test similar to the one mentioned above for the River2D mesh could be used to determine impacts of spatial scale to output for the habitat assessment.

Response: *We encountered computational limitations on mesh resolution at the scale of the segments we modeled in the Hanford Reach. We used the dual 10/20 m mesh size because River2D would not come to convergence with a 10 m resolution throughout the segment. The two specific habitats of interest were spawning habitat and rearing habitat. Biologically relevant units for spawning habitat were redd clusters, and these approximate the functional scale for which our models will be predicting habitat. Redd clusters are groups of individual redds typically numbering in the hundreds and in 2004 mean redd cluster size was 6.38 ha, much larger than a 10 or 20 m mesh size. Biologically relevant units for rearing habitat were shoreline and off-channel "strips" that were continuous areas of shallow, slow velocity water. These areas were also much larger than the 10 or 20 m mesh size. Local hydrodynamic conditions at the scale of the Hanford Reach are distributed across hectares, particularly in spawning and rearing areas. In other words, physical conditions were relatively homogenous across these areas. Conversely, based on our experience in smaller rivers and streams, the scale is reduced and local hydrodynamic conditions are much more variable, being relatively homogenous on the scale of meters rather than hectares. A sensitivity test on the effects of mesh resolution on habitat predictions would certainly be interesting, but is beyond the scope of our report.*

There is a discussion of determination for spawning habitat criteria using slope, depth and velocity. It is unclear if the slope was water surface slope or energy slope. It was also unclear in the model variables test whether either interstitial flow or a bed permeability value was considered. It has been well documented in the published literature that the interstitial flow is important for salmon to determine redd locations. A simple look at surface depth and velocity may not appropriately incorporate all selection factors. In addition, it is not stated whether any correlation was completed between the redd locations observed and hatching success to determine if there was a correlation with redd persistence, hydrodynamic conditions and hatching success throughout the study reach. Consideration should be given to those factors and included in the introduction or discussion.

Response: *Development of a spawning habitat model for the Hanford Reach has been on ongoing effort with many iterations of analytical approach, variable testing, simulations, and accuracy assessments. The details presented in the version of the report that you reviewed were associated with one of the initial iterations. The analytical approach that we are currently using is the same, but the variables we have tested are different, and we have tested both Reach-wide models and area-specific models. We will provide you the current material for review and comment when the report is finalized.*

Any slope variable used in the spawning habitat model refers to bed slope. We have clarified this. Bed slope is used to reflect the affinity of a salmon for flatter rather than steeper bottom configurations.

We are familiar with the apparent significance of interstitial water movement, hyporheic flow, or upwelling to spawning site selection in salmonids. Again, the problem is scale. To develop a spatial coverage that would characterize upwelling, installation of piezometers or a similar type of equipment would be required. On the scale of the Columbia River, this would be very difficult, and definitely beyond the scope of this study. As a surrogate, we examined a geomorphic variable, proximity to a gravel bar or island. In the absence of groundwater influence, hydraulic pressure differentials work to both force water into the substrate, and allow it to re-surface as upwelling. These differentials are the result of geomorphic features such as riffles, gravel bars, and islands. We found a consistent association between actual redd locations and the distance (relatively short) to a bar or island. This became our surrogate variable for upwelling. We have described this in the final report.

We did not conduct any assessment of hatching success or incubation survival, so we were not able to develop any correlations with hydrodynamic conditions.

RESULTS

The use of the hydraulic model River2D was relied on for both the habitat analysis and the entrapment evaluation. However the report results did not show River2D output in a manner that would demonstrate how the entrapments were dewatered or isolated as flows progressed from high flow to low flow. Additional graphics of River2D and GIS output would better illustrate this result for the reader.

Response: *River2D was not used to any degree for the entrapment evaluation. We added a table and additional text to more clearly relate which models were used for the different assessments.*

The model error was discussed in the results but it was not clear how the error was applied or factored into interpretation of results. The model error was shown to be as high as 0.2 meters but there is no mention of what the change in water surface was between flow simulations. For example if the change from one 10,000 CFS flow to the next was 0.1 meter across a section of the river it would affect the interpretation of the model since model error could exceed predicted change. Model simulations that show water surface changes greater than the error could be interpreted as a real effect seen on the system.

Response: *We chose to model at 10 kcfs flow increments for several reasons, including modeling error. As discussed in the report, mean absolute error from validation of MASS1 water surface simulations ranged from 0.05 to 0.21 m, and averaged 0.11 m. From the MASS1 modeling we conducted for the entrapment evaluation, we calculated an average change in water surface elevation of about 0.2 m between each 10 kcfs flow band (range 30 – 400 kcfs). We did not want to risk the occurrence of "overlapping" flow bands by using increments that were smaller than the potential modeling error. We have inserted this additional explanation in the report.*

The illustration of water surface change with discharge should be included as water surface elevation versus discharge at the same cross-sections that depth and velocity were displayed. This could be used to visually represent how the model performed compared with model error. Model error bars could be displayed on the graph of water surface versus discharge.

Response: *The reviewer makes a valid point. We did not display water surface elevations because of uncertainties associated with some of the empirical data. Until we can clear up these uncertainties, depth comparisons will serve as a surrogate for water surface elevations. In addition, we are planning additional validation data surveys before we continue future studies and modeling efforts.*

The report relies on previous modeling for the middle reach but does not report those results other than through citations. Those reports, since they are one of the key factors in interpreting the flow fluctuations, should be attached as appendices for the reader.

Response: *We have cited Tiffan et al. 2002 and included their validation information in the accuracy table in the **River2D Validation Results** section of the report.*

DISCUSSION

The discussion section for entrapment and also for the spawning habitat was generally lacking in ecological background and also in linking the physical changes to ecological consequences. There was a major effort in the report to summarize the statistics but very little interpretation of those statistics to real world biological consequences.

Response: *We were required to focus this effort on fall Chinook salmon. Ecological implications are at least as important, and directly related to conditions fall Chinook are subjected to. The range and magnitude of what we are trying to accomplish for fall Chinook required us to focus narrowly on that specific subject. We have provided some perspective on the "real world biological consequences" of the factors we have analyzed.*

The River2D model was stated to have 10 meter grid size for the area near shore and 20 meter grid size for the open water areas of the river channel. It is unclear how this grid size was used when the redds were much smaller than that grid pattern. A discussion of model mesh size as compared to redd size should be included. This should include graphic presentations to demonstrate model applicability of selected River2D sites compared with the redd locations.

Response: *At the most resolute scale our models simulate habitat metrics (depth and velocity) near the maximum redd size, approximately 10 m. While our bathymetry is twice as resolute, 10 m is a biologically justified goal and a realistic goal considering the overall scale of the entire Hanford Reach (90 km). In fact, redd clusters approximate the true scale for which our habitat models will be predicting habitat. Redd clusters are groups of individual redds typically numbering in the hundreds and in 2004 mean redd cluster size was 6.38 ha. Also see our response to your similar comment earlier.*

The spawning section should include a discussion of how long the fish generally are in the area when they construct redds. This discussion would add to the interpretation of flow fluctuations, as it is unclear whether daily flow fluctuations during spawning influence redd location. This is important for interpretation of the persistence of spawning habitat especially in near shore areas around the islands. A presentation of areas watered and dewatered with fluctuation in flow is important to display so the reader can interpret the consequences of redds being formed in shoreline areas.

Response: *We agree and have added discussion and references to spawning duration. Chapman (1986) determined that fall Chinook salmon could complete redds on Vernita Bar in less than 24 hours, but oftentimes take 5-7 days to complete spawning. We did not monitor this ourselves. It is not currently known whether or how daily flow fluctuations during spawning influence redd location. We have identified the need to study this issue further in the Recommendations under the spawning habitat assessment.*

Since very few redds were established in areas that were dewatered, this was not an issue that was explicitly addressed in our analysis. It is implicitly accounted for via our persistence metric. Dewatering would result in velocities of 0 for some number of hourly observations. Thus, dewatering events were captured. These events were almost non-existent within known spawning areas, primarily as a function of the reverse load following that is conducted during the spawning season..

The spawning model relies strongly on the velocity to predict persistence of the redd locations. River2D can be used to assess shear stress on bottom substrates. The use of the model for this type of assessment could determine the near bed velocity which may be an important factor in redd location. This type of approach has been applied to spawning evaluations for other species.

Response: *The reviewer makes a valid point and we agree that bed velocity may be an appropriate metric to consider for spawning fall Chinook. We plan to continually upgrade our spawning model as new data become available. We hope to look at shear velocity as a potential surrogate for bed velocity in the near future. We also plan to either empirically or mathematically determine near bed velocities to determine whether they are correlated with spawning locations.*

Specific comments on Results and Discussion

Page 6, second full paragraph. There's a discussion of location of redds based on mean column water velocities. The velocity at redd construction may be the factor that is most important. Salmon are known to select locations that are in areas with down welling velocities and downward flow through the surface substrates. Local hydraulics at these locations are important to the redd selection process. Measurements taken after redd construction may not be a good indicator of whether the location would have permanence. Therefore the measurement of velocities during the actual redd construction period is probably most important.

Response: *Your comment is correct. The spawning habitat model that is presented in the current version of the report is based on spawning locations and conditions from the fall of 2004. The velocities that were associated with redd sites (determined from Orthophotography) were produced from River2D modeling runs. Without being able to determine exactly when individual redds were established, we re-constructed hourly velocities based on the hourly hydrograph for the 264 hours (11 days) prior to peak spawning (based on redd counts) in 2004, and associated those data with redd clusters. In the Hanford Reach, the 24-hour hydrograph is managed such that the flow pattern is very consistent from one day to the next during the spawning season. We developed various metrics from this data set including our persistence metric to determine which were significant predictor metrics.*

Figure 25 should be re-labelled so that the values for the rearing codes are descriptors rather than numeric code.

Response: *We agree and have labelled the graphic with descriptive codes.*

Figure 31 needs to be re-labelled so that the labels on the figures are more readable.

Response: *This figure has been enhanced.*

Page 83, River2D: The discussion mentions that the model simulations generally did an adequate job of simulating water surface and water velocities. A specific definition of “adequate” should be provided. Specifically how closely did water surface elevations match simulated flows and what was the change between simulated discharge and model error compared with actual water surface change.

Response: *We agree that in this context, the term adequate is vague and have chosen the term "reasonable" with additional explanation based on the validation results..*

Page 84, second paragraph discussing factors that influence performance of River2D. The roughness coefficients are one means to calibrate water surface elevations. One also should consider the channel geometry error associated with sources of that geometry that could affect construction of the mesh topography and therefore water surface elevations.

Response: *Your comment is correct. The process that we planned to use started with adjusting roughness heights to match empirical water surface elevations. Then a comparison of River2D simulated depths with empirical depths would have been used to identify errors in channel geometry or bathymetry. We do not however, believe that channel geometry error or bed error was a problem in the upper and lower segments that we modeled. Both of these sections were surveyed with the CHARTS system and the result was excellent bathymetric coverage. However, there is still a need to complete a more robust calibration/validation exercise to fine tune and evaluate the performance of our Hanford Reach model. We are planning on completing this before we move forward with the next iteration of habitat modeling.*

Page 84, third paragraph. Model error is mentioned for Mass1 but it is unclear whether the Mass1 error is additive with the River2D error for combined error in water surface elevations, which may be greater than the single error of each model. A discussion of additive error for both gathering topography and simulations of each hydraulic model should be included in the results.

Response: *The reviewer makes a valid point. Since River2D matched the MASS1 elevation at the downstream boundary where it was used, the error at that location was entirely from MASS1. However, simulated conditions upstream would include both the effect of River2D modeling error, and the error associated with the target water surface elevation at the downstream boundary (additive). At some distance upstream, the effect of MASS1 error at the boundary would become negligible. For that reason, we placed the boundaries distant from areas of biological interest. The best way to characterize and quantify these errors along with bathymetry error, is with a comprehensive and rigorous validation exercise. Such an exercise is planned before the next iteration of our habitat modeling. We have added additional discussion on this issue in the* ***River 2D Discussion*** *section of the report.*

Page 89, first full paragraph. The conclusion stated here is that Chinook tend to move downstream during the rearing season, not upstream. However the presentation on page 86 for video data shows that most of the observed movement was upstream. There seems to be a conflict here between the data presented and the conclusions.

Response: *The temporal and spatial scales for the two pieces of information you cited from the report are completely different. The first conclusion refers to the pattern of movement throughout the entire Hanford Reach during the entire rearing period. The discussion describes the abundance of juvenile fish among the three Hanford Reach segments from late March through early June. The spatial scale of the video surveys was several hundred meters or less, and the temporal scale was two weeks. In addition, the upstream movement you refer to was only at night (equal upstream and downstream movement during the day), and only by individual fish.*

Page 106, evaluating impacts associated with alternative hydraulic rations. The section should include a discussion of how water surface elevation changes compare with model error to convince the reader that the fluctuations predicted per week and the number of entrapments reach wide can be supported by the simulations. The inclusion of this discussion would strengthen the report to convince the reader that the simulation model is accurately predicting the number of entrapments in the study reach.

Response: *We discussed this issue previously. The previous discussion identified the mean absolute error from validation of MASS1 water surface simulations as ranging from 0.05 to 0.21 m, and averaging 0.11 m. The average change in water surface elevation was about 0.2 m between each 10 kcfs flow band (range 30 – 400 kcfs). And the entrapment locations were fixed in space as specific geographic locations based on measured empirical data with accuracy of about 1 m. We felt this was sufficient evidence to convince the reader that our predictions of entrapment events were "accurate", particularly considering the scale of this work.*

Page 109, first paragraph. This paragraph discusses the association between simulated number of fish entrapped the number of entrapments. Again, the discussion would benefit from the previously recommended discussion of how well the model is predicting water surface elevations and therefore the construction or the creation of entrapments along the reach. The discussion of error bands of the simulations would provide a range of the estimate and strengthen the discussion.

Response: *See previous response. The error values required to construct and display "error bands" are not distributed equally among streamflows or segments of the Hanford Reach. It is beyond the scope of our work to retrieve this level of detail from previous efforts (MASS1 calibration and validation) and incorporate it into our analysis.*

The graph on page 111, second paragraph following Table 13. The use of exceedance values for daily fluctuations is shown in the table. It is unclear from the discussion how these exceedence values relate to real-time data. Since the biological affects are in real-time the consequences of exceedence may not be a direct cause and effect. More interpretation of the ecological consequences rather than the statistical presentation should be included.

Response: *We have clarified the description of the process we used to derive these exceedance values from empirical data. As we discussed previously, this was totally a "water" exercise with the goal of reducing flow fluctuations generated by Priest Rapids Dam. The analysis was a physical one (i.e. reduced flow fluctuations) with implicit connections to a biological effect (reduced entrapment) downstream.*

Page 113, Table 17. There is a fairly short period of record used for the number of days of storage capacity exceeded. It is unclear how representative this time period is of the full range of conditions that may be seen at Priest Rapids Dam. Generally for hydrologic studies longer periods of record are used to encompass a wide range of hydrological conditions and therefore allow a wider range of ecological interpretation.

Response: *We would agree that 10 years is not usually sufficient to characterize hydrologic variation. However, this analysis was not intended to be a comprehensive evaluation of long term re-regulation capability. Rather it was meant to provide insight on the relative rate of re-regulation success during high flow years, low flow years, and intermediate flow years. The 1995-2004 time period included both drought and flood years (98% and <1% exceedance probabilities based on the 1970-2005 time period), and everything in between with a rather typical distribution. The hydrograph was permanently changed following construction of the large Canadian storage projects (e.g. Mica) around 1970, thus the exclusion of earlier years. Our period of record may not have been perfect, but we think it was very reasonable as far as capturing variation.*

Page 116, in the Discussion section. Throughout this page there are conclusion statements but no data presented in the report that support the statements. Supportive data is needed to show why those conclusions are reached. For example, in the first paragraph the statement is made that sampling started well after emergence was underway and therefore the actual impacts could exceed the estimates by a large margin. However, the numbers per seine haul in entrapment are shown to peak well after sampling began. This data does not seem to support the statement that the magnitude of loss would be much higher than presented in the data set. The third paragraph on this page discusses the inability to account for predation mortality and that therefore the estimate of the impact was low. Since no data was developed for predation mortality the statement should be neutral, rather than whether the bias was high or low, until studies are completed.

Response: *The statements you refer to are meant to be discussion points and qualitative conclusions that characterize (not quantify) the nature of our impact estimate. Addressing any of the factors discussed, would increase our impact estimate. The quantitative question of "how much?" cannot be answered without additional studies or data. Our sampling did, in fact, start well after emergence was underway. As a result, we think it is safe to say that our impact estimate would have been higher if our sampling program had started in early March rather than early April. It is not possible to say how much higher. We have clarified this in the text and removed the implication that this effect could have been large. The same logic applies to your comment on predation mortality. It obviously occurs, and we have observed it on many occasions, particularly bird predation. Thus, if we were able to account for it, our impact estimate would have been higher. Since we did not conduct any focused predation studies we cannot quantify the impact. As a result, we can say our impact estimate is biased low without accounting for predation, we just cannot say how much. We have re-written this entire section and clarified our statements.*

Page 118. It was concluded that the model was a useful tool for assessing potential impacts. However there is no discussion of how model error compares with actual changes in water surface elevation. The exclusion of this discussion of change in water surface per 10kcfs would greatly add to the confidence in the simulation model. For example if the model error is plus or minus 21 cm and the predicted change from 10,000 CFS flow to the next is plus or minus 10 cm, then the changes are within model error. Therefore the conclusion that there is an actual change in water surface that would affect stranding based on model results is not supported in the modeling effort. Much more detailed discussion with graphical comparison of modeled water surface elevations to measured water surface elevations would provide a basis for the conclusion stated on the page.

Response: *We discussed this issue previously. The previous discussion identified the mean absolute error from validation of MASS1 water surface simulations as ranging from 0.05 to 0.21 m, and averaging 0.11 m. The average change in water surface elevation between each 10 kcfs flow band (range 30 – 400 kcfs) was about 0.2 m. Thus, the actual changes in water surface elevation are larger than model error. In addition, the majority of flow drops throughout the rearing season were greater than 10 kcfs, so changes in water surface elevation for determination of entrapment events were more commonly 0.4 m or greater. This further reduces the effect of modeling error. We have clarified this in the report, although we did not have time to add graphical comparisons.*

GENERAL FORMATTING

The report would benefit by adding tables to summarize data collection and also by the use of additional figures. One such figure is a decision tree to visually portray the development of alternative hydrographs. The current figures that are in the report are unreadable in some cases because of font size.

Response: *We have reorganized the report into separate chapters and added many new tables and graphs. Graphics quality has been improved and formatting has been standardized. We have included a flow chart that shows the components of the various studies, data collected and generated, analyses conducted, and tools developed.*

GENERAL IMPRESSIONS

In conclusion, I recommend a major rewrite of the report prior to finalization. The current organization of the report does not provide the reader with a cohesive document. The report should be rewritten so that there is a connected flow in the material from the introduction and objectives through methods onto results for each study. I strongly recommend consideration of three separate reports, one for each study. This would shorten and simplify the presentation for the reader. While there may be some repetition of material especially for the methodology it would be less confusing to read and would allow better integration across the studies.

Response: *We have conducted a major rewrite of the entire report. The four major components; Hydrodynamic Modeling, Entrapment/Stranding Evaluation, Spawning Habitat Assessment, and Rearing Habitat Assessment have been presented as separate chapters with references to results from other chapters where relevant. Considering the potential regulatory application of this work, we opted to retain the work as a single document.*

The report also would benefit from an integration that summarizes across all the studies. If it remains a single report, a final chapter that combines all the studies at the population level is needed. The individual conclusions from each study, without relating it to the various study components, do not provide the reader with enough material to gain the conclusions that the authors reached.

Response: *We agree that a final chapter focused on the range of implications to population productivity would be useful. We have expanded the section on population level impacts (within the Entrapment Evaluation section) with regards to juvenile mortality. And we are developing plans to evaluate the implications of all of our work on longer term productivity. However, the work and modeling that would be required to produce a comprehensive evaluation of the range of implications of these issues on population productivity was beyond the scope (both time and funding) of our original effort. We have clarified our discussions and conclusions and the basis for them in a way that we hope the reader will understand.*

Review of:

Fall Chinook Salmon Habitat Analysis and Stranding/Entrapment Evaluation in the Hanford Reach of the Columbia River Report

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"I found the study to be very interesting and an excellent example of the application of hydrodynamic modeling to a very important problem. Thank you for the opportunity to participate." (P. Steffler)

As my expertise is primarily in the area of hydrodynamic modeling, this review will focus mostly on the hydrodynamic modeling methods and validation. I considered the habitat and stranding/entrapment sections also primarily from the point of view of the impact of the assumptions and limitations entailed by the hydrodynamic modeling. Specific comments are listed below and a general summary is provided at the end of this review. The numbering of the specific comments is for reference only and is not based on any ordering of importance or sequence.

Response: *We appreciated the comments.*

1. The organization of the report could be improved by concentrating the hydrodynamic analyses in the Methods section. The true purpose of the report and the useful results are the habitat and stranding/entrapment analyses. The hydrodynamic analyses are a means to that end and therefore part of the methods. The hydrodynamic results and discussion presented are for the purpose of validation of the models and to establish confidence in them. Again, this is exactly the purpose of the methods section of the report. Further, the two analyses should be treated separately, rather than intermingled by stages. Practically, this means that all of the information for each hydrodynamic modeling activity is concentrated together rather than scattered and is thus easier to assimilate and evaluate.

Response: *We agree with your comments. However, we have chosen to continue the Hydrodynamic Modeling work as a separate chapter. The work conducted, the associated costs, and the data, analyses, and results were so extensive that we found it to be cumbersome to include in the Methods sections for the other sections (Entrapment/Stranding Evaluation, Spawning Habitat Assessment, Rearing Habitat Assessment). As you suggested, we have presented the two hydrodynamic modeling activities (MASS1, River2D) separately within a common chapter.*

Within the revised organization, the field data collection activities and methods could be discussed as appropriate within the modeling framework. That is, data collected for model input discussed separately from the data collected for model validation. This would increase clarity and reduce redundancy, as the data would be tied more directly to its utility.

Response: *We agree and have clarified the text accordingly.*

In fact, I cut and pasted the provided Word files in order to approximate the above suggested organization for the hydrodynamic modeling section. This was primarily to aid my review as I was having trouble keeping track of what information was where. The modified (mangled?) file is available if the Authors wish to see it or use it as a starting point for revision. Other than the rearrangement, none of the text or figures has been changed, although some headings might have been lost.

Response: *We appreciate your work and have in fact, incorporated your revisions into the report.*

2. The Introduction of the report could include a paragraph or page with more information on the general hydrologic, hydraulic and geomorphologic characteristics of the Hanford Reach of the Columbia. Typical widths, depths, slopes, Froude Numbers, sediment loads, bedform regime, recorded channel morphology (horizontal and vertical), manmade “improvements” such as dikes, spurs, bridges and associated training works are all of interest. The investigators (and likely the primary users of the report) probably take this information for granted and most of it is available in great detail in the final results. For external readers, this information helps in framing initial expectations of what is going on. The discharge regime, as it is dominated by dam releases, is well covered.

Response: *We agree and have added material describing the general hydrologic and geomorphologic characteristics of the Hanford Reach.*

3. The DEM work looks to have been very thorough and effective. The one thing that might have been included is some discussion on the correspondence of the various techniques where there was overlapping coverage. This would help shed light on relative accuracy and also on the channel changes over time. This issue arises (see below) in the discussion of the model validation.

Response: *We appreciate the comment and have added a comparison of the two dominant data sources for the DEM as proposed by the reviewer.*

4. The substrate data collection and analysis also looks good.

Response: *No response needed.*

5. Clearly, a major issue in the hydrodynamic part of the study is the fact that the 2D model was not calibrated. In the end, I agree with the authors that the results are satisfactory despite the lack of calibration. However, the justification is weak. There is a good deal of information available that could have been used. At this stage it is obviously too late to do anything except try to strengthen the justification. The bottom line, of course, is the validation. As noted in a subsequent point, most of the errors in velocity predictions are likely to be due to other causes. The following points highlight weaknesses in the arguments made in the report that the Authors might address.

Response: *We agree that the hydrodynamic modeling results are reasonable despite the lack of a rigorous calibration process. As suggested, we have clarified and strengthened the justification for the process (lack of) that we used.*

a. The report implies that if calibration were to be undertaken, water surface elevation data would have to have been taken along the entire reach for every discharge considered. This is not true. Ideally, one would have a low, intermediate, and high flow profile for calibration. Practically, one profile is often all that is used. Sometimes, only a few scattered points are available, but this still better than nothing.

Response: *Considering the constantly changing streamflows in the Hanford Reach, collecting good quality empirical data under "stable" conditions is much more difficult than it appears. In the* ***River 2D Discussion*** *section of the report, we present some error data for water surface elevations along with the calibration discussion. Since we have extensive future work planned in the Hanford Reach, the obvious solution to existing questions regarding calibration/validation of River2D is to conduct a comprehensive, rigorous calibration/validation exercise. This is currently being planned.*

b. The 1D model was used to set boundary condition elevations for the 2D model with an expected error of < 0.1m. Why couldn't steady state 1D model water surface profiles be used for calibration of the 2D model? 0.1 m accuracy is not great but is a significant improvement on 0.3m.

Response: *Using MASS1 for validation was proposed, but we assumed the logic to be circular, i.e. validation of a model with a model, and did not consider it any further. At the time, we had plans to collect more empirical validation data, but with time and funding running short, it did not happen. We plan to continue with additional validation work in the near future, hopefully with empirical data, but with MASS1 simulations if necessary.*

c. The variation of the calibrated Manning's n in the 1D model could have been used by converting the n values to a roughness height (see River2D documentation or the n->ks converter function in R2D_Bed). This might still be used to strengthen the justification for the overall mean roughness height. Also, the longitudinal variation in the 1D model n values might be correlated with substrate variations.

Response: *We agree with the reviewer and have conducted the Manning's n value conversions as you recommended. The results are now described in the report as a range of 0.096 m to 0.147 m, with an average of 0.118 m. This value is similar to our chosen uniform roughness height of 0.1 m.*

d. A common approach (e.g., Waddle et al., 2000, where no calibration was performed) is to assign differing roughness heights to different substrate classes based on sediment size. In this report, collection of substrate data was partially justified for 2D model input, but not used.

Response: *We collected substrate data for both our spawning habitat analysis, and to determine roughness heights for River2D. The dominant substrate size was 7.6 cm to 15.2 cm with an average of 11.4 cm, or 0.114 m. This value along with other reasons led to our uniform roughness height choice of 0.1 m.*

e. The choice of a roughness height of 0.1 m seems to have been made arbitrarily. The usual rule of thumb is that the roughness height is some multiple (3-5) of the mean or dominant sediment size. This would suggest that if the mean cobble size of 10 cm was desired, a roughness height of about 30 cm should have been used.

Response: *We based our choice of roughness height on the dominant substrate size and on the successful previous use of 0.1 m in the middle segment of the Hanford Reach (Tiffan et al. 2002). The River2D users* *manual (Steffler and Blackburn 2002) provides guidance as "For resistance due primarily to bed material roughness, a starting estimate of k_s can be taken as 1-3 times the largest grain diameter." We erred by using the dominant substrate size rather than the largest grain diameter, but planned on adjusting for calibration. This was not done, and is planned for future refinement of model results.*

f. The argument is made that variations in roughness height are insignificant at the scale of the Columbia River. There is merit in this, but it could be elaborated by giving example calculations, showing small changes in the non-dimensional Chezy coefficient (C*) from relatively large changes in the roughness height.

Response: *We would not have had to concern ourselves with whether variations in roughness height were significant if we had the time and resources to conduct a more comprehensive calibration exercise. Completing calibration work has already been planned.*

g. In the same paragraph, it is argued that the detailed bathymetry and model capability accounts for extra resistance due to boulder fields. This is only true if: the boulders occupy most of the depth or protrude from the water surface; the computational mesh is refined to a scale at a fraction (~1/6) of the typical boulder size; and each boulder is resolved with 20 or more points to describe it's shape. A more likely explanation for the apparent turbulence observed at these locations is numerical oscillations caused by bed level fluctuations with a relatively coarse grid.

Response: *We agree and have removed this section from our methods.*

6. Specific locations of 2D model boundary conditions should be provided.

Response: *We agree and have produced a new explanatory figure.*

7. The choice of mixed 20m/10m mesh spacing seems reasonable but should be better justified. Running one or two cases with finer or coarser meshes to see the difference in predicted result would have been interesting and hopefully reassuring. An argument for grid spacing on the basis of depth can be made. The shallow water equations that the model is based on are valid for wavelengths greater than about 10 depths. To accurately model these wavelengths, there should be at least 5 or 6 nodes (think of how many points would define a sine function over one wavelength). Therefore a mesh spacing of about 1 or 2 times a typical depth is about optimum. It also makes sense to have a finer mesh where it is shallower and where more velocity variation is expected.

Response: *You make a valid point, however we did not have time to run any additional simulations. We did attempt to run each segment at a 10 m resolution but realized that with 32 bit operating systems, only 2 GB of memory can be allocated with River2D even if the system has more, which ours did at 4 GB. Future assessments with finer meshes may be possible when 64 bit systems become available. We may also choose to break our segments into shorter segments if a finer mesh is deemed necessary.*

8. Some details of the computational effort of running the 2D model could be given. There are several oblique references to this in terms of resulting study limitations but it should be explicit as the essential choice in any computational study is the three-way tradeoff between range of coverage, accuracy, and time and effort. The typical number of nodes and/or elements, iterations/time steps to convergence, and clock times for a run would be of interest.

Response: *We agree and most of the suggested details have been added.*

9. The 2D model validation section is important and of particular interest to me. I have a number of comments listed below.

   a. The validation should start with a discussion of the objective accuracies required or desired. The present validation section is presented in something of a vacuum. This discussion should consider both what accuracy is required for subsequent analysis and what accuracy is realistically obtainable from the literature or similar studies.

Response: *The suggested discussion has been added.*

   b. One criticism that might be levelled at the validation data is that some of the key areas noted in the habitat study were not represented with validation sections.

Response: *You make a valid point about the absence of validation data for the Locke Island area, when the spawning model was developed from that area. We have collected additional empirical data in this area and are including it in the report to address this concern and to support continued spawning habitat analyses.*

   c. The order of validation comparisons should be changed to water surface elevation first, depth second and velocity last. The reason is that the sources of the elevation errors are easier to identify, and they contribute directly to the depth errors. Depth error, in turn, makes a contribution to velocity error. Identification of the discrepancy sources is useful in assessing the model validity and in providing guidance for future studies. The statistical error analysis provided in the report is useful but does not provide much guidance for improvement.

Response: *We have inserted additional discussion and clarification of the calibration/validation process.*

   d. The water surface elevation discrepancy is directly attributable to the roughness choice and thus to the calibration of the model. It was noted that measured elevations were higher than the model for 3 sections and 7 were lower. It would be worth looking at each section and comparing the calibrated 1D model Manning's n values for those sections to see if there is a consistent pattern in the sign and magnitude of the discrepancy. Quite often, the most useful comparison is a complete water surface profile over the entire study reach. The reason is that roughness actually correlates with water surface slope and depth. Sometimes the depth is different at a particular location not on account of a local roughness, but because of downstream errors causing a backwater effect.

Response: *Your comments and recommendations are relevant. In addition to having very limited empirical water surface elevation data to work with, the data had various problems. One obvious solution is to step back and acquire comprehensive, good quality data to complete the validation/calibration process. We currently have plans to do this prior to our next modeling exercise. Now that the majority of the work has been done to develop the hydrodynamic model (bathymetry, DEM, mesh, first round of flow modeling), we will be able to focus our future efforts on the comprehensive, calibration/validation of model output.*

   e. Depth is derived directly from water surface elevation minus bed elevation. Since water surface elevation is very flat (in actuality and in the model) across a typical section, the only additional error, compared to water surface elevation, is bed elevation error. In normal cases, where the water surface error is reduced to a few cm by roughness calibration, the depth error is almost entirely bed elevation error. Note that this means that there is really no, or very little, depth error due to the 2D model itself. Therefore, one should look to discrepancies in the bed level provided to the model and the bed level used for validation.

Response: *We look forward to collecting comprehensive water surface elevation data as well as additional depth and velocity data so we can do a more thorough job of calibrating the model, then identifying bed elevation errors, and finally validating velocity output thus, realizing the benefits of more accurate simulations. An example of the concept you discuss in your comment can be seen in the depth comparison plot for the validation cross section at RM 358. The large discrepancy between ADCP depths and River2D depths has been caused by bathymetry errors. An examination of the SHOALS data for this area revealed an absence of data for the middle two-thirds of the river channel. Bathymetry was interpolated from adjacent USACE cross sections and did not represent the actual bathymetry at this site as evidenced by ADCP data.*

On Figure 27, the validation sections at RM 347, 351, 355 indicate almost purely water surface elevation error, as the depth error is small and consistent across the entire channel. RM 358 shows very large depth error that is certainly bed error. The error is large enough that one wonders if two different sections are being compared. RM 362 shows much less discrepancy than RM 358 but much more than the first 3 sections. The discrepancy is also not consistent across the channel. This suggests that there is significant bed elevation error. Both these two sections should be looked at more closely.

Response: *Your observation is almost certainly correct. The cross sections at RM 347, 351, and 355 were in the lower section covered by the more recent CHARTS survey which provided very thorough bathymetry. Thus your comment that the error is mostly water surface elevation error is likely correct, particularly considering the lack of a robust calibration process. You are also correct regarding the cross sections at RM 358 and 362. These cross sections were in the middle section covered by the SHOALS survey in 1998. We are aware that we have bathymetry problems related to interpolation in this section, and as a result, bed elevations in some areas are suspect. We have plans to correct both the calibration issues and the bathymetry problems in the future.*

On figure 29 (for the upper sections), we also see a combination of water surface and bed elevation errors. In this case some of the bed elevation discrepancies could be explained by morphological changes over the time interval between the DEM construction and the section data collection. For example RM 390.6 clearly shows a lateral shift of the channel. RM 382.8 shows a smaller shift and an extensive area that has scoured out by about 3 m. By the way, what is “relative distance” as plotted? It would be best if all plot axes were consistent.

Response: *As you point out, the data suggest that the bed may have shifted or changed over time. While this is possible, we believe that it is more likely that the validation data itself is suspect. That data was collected about 10 years ago with different equipment and methods and should be scrutinized. In short a Reach-wide, comprehensive validation data set along with some additional bathymetry work and calibration would be the best way to address the range of issues you have identified, and we have plans conduct the work.*

f. After considering depth error, the velocity error makes more sense. It is no surprise that velocities at RM 358 differ greatly. Really, until the bed discrepancy there is worked out, there is no point in presenting the velocity comparison here, other than to show that if a different bathymetry is used, a different velocity distribution results. To a lesser extent, the same conclusion can be reached at RM 362. In general, one should look first to the depth comparisons to explain the discrepancies in the velocity comparisons.

Response: *We have added additional discussion regarding the effects of water surface elevation, depth, and velocity errors, and we have eliminated the plot for the cross section at RM 358.*

In cases where there is a consistent variation in the velocity profiles without a corresponding variation in depth, we may look to the model for other explanations. For example, at RM 351, the depth comparison is very close, indicating a good bed profile match. The modeled depth is slightly larger, indicating the model water surface elevation is a bit high. On the corresponding velocity comparison, we see that the average model velocity is a bit low, which is consistent with a higher depth for the same discharge. However, we also see that the model bank velocities are higher than measured. This suggests that the roughness distribution across the channel may be non-uniform in reality, with larger roughness material near the banks. It would be interesting to compare this speculation (hypothesis?) with the substrate map for this section.

Response: *You make a valid point. From our experience, near-shore ADCP measurements can be problematic if one is not careful. Acoustic backscatter and noise can affect data in shallow water areas near the river bank. More conscientious empirical data collection can solve this problem. Then the focus would turn to refining our roughness values on a spatial basis including near-shore areas, and re-evaluating the WSE match. Then correct bed elevation errors if needed, and finally look at velocity validation differences to determine if local adjustments of roughness values are needed and could improve the fit without resulting in elevated "pillows" of water, or unrealistic local backwater effects.*

g. The discussion of the 2D model validation is mostly acceptable, but should be revised in the light of the above points.

Response: *The suggested revisions have been added.*

10. Figures 4 and 5 are redundant. On figure 8, it would useful to indicate which sections were obtained by what means at what time and also provide the river km (mile) reference.

Response: *The redundant figure has been removed and Figure 8 (now 22) has been changed.*

11. The seemingly arbitrary mix and interchange of imperial and SI units is distracting and annoying. Any consistent theme would be preferable. Choose one and have the other in brackets would probably be best.

Response: *We partially agree and have decided to stick with imperial units for streamflow (kcfs) so that managers and regulators can interpret the document intuitively, without converting measurements. In the Columbia Basin, both fisheries and hydrosystem managers view the movement of water in thousands of cubic feet per second, while being satisfied with SI units for most other applications. All other units in the report will be SI.*

12. The MASS1 1D modeling seems to have been carried out in a straightforward manner. The calibration and validation exercises are described briefly and some statistics are provided. As 1D modeling, steady or unsteady, is a well developed and used procedure, this is probably adequate. The only thing I would suggest is that since the MASS1 model is perhaps not widely known, a few details of its numerical formulation would be interesting.

Response: *We have provided a brief description of the MASS1 model in an Appendix.*

13. With respect to the habitat analysis, I have only a few comments, mostly related to use of the hydrodynamic model.

    a. The preferred velocity range band is fairly broad (1.0 – 2.0 m/s) which suggests that velocity prediction accuracy should be some fraction of this. It would be interesting for the Authors to discuss velocity accuracy requirements in this light.

Response: *The current spawning habitat model uses two velocity metrics; velocities >1 m/sec as a threshold for persistence, and the velocity coefficient of variation (CV). The persistence metric looks at the proportion of the time that velocities exceed 1 m/sec. We have found that this is a good predictor variable for spawning locations; higher proportion of the time at or above this threshold, more spawning activity. We have also found that the CV is a consistently good predictor; lower CV more spawning activity. We have identified 20% as our error threshold in the report. Accepting this level of error was based primarily on field experience and professional judgement. We have plans to create section-specific models in the near future that will perform much better than the Reach-wide model described in the current report. We hope by then, we will have completed adequate calibration/validation for River2D and our accuracy will be such that it will not be an issue from the standpoint of habitat modeling.*

    b. As noted above, no hydrodynamic validation appears to have been done at the Locke islands spawning area. I'm not sure where the other validation sections land with respect to the other important sites. The Locke islands area appears to be much more hydrodynamically complex than the validation sections and so the overall validation may be questionable. In a sense, the habitat validation may be providing hydrodynamic validation, but the quality of the habitat model is limited by the reduced confidence in the velocity predictions.

Response: *You make a valid point about the absence of validation data for the Locke Island area. We have collected additional validation data in the Locke Island area and elsewhere in the middle of the Reach, and you will see the results in the final report. We still plan to conduct more validation in the future, in all areas of the Reach.*

    c. Within the Locke Island or Vernita bar sites, the substrate distribution may affect the velocity distribution significantly enough that the predicted habitat selection areas could be affected. It would be interesting to compare substrate maps for these areas. It would also be interesting to produce maps that code the reason that the habitat model is not selecting areas that are in fact being used.

Response: *We agree with your observations. We have plans to conduct a number of sensitivity analyses and investigate the distributions of various measures of depth, velocity, nose velocity, substrate, and geomorphic characteristics to gain additional insight into what is driving habitat selection. These investigations are planned for our next iteration of spawning habitat modeling for section specific models in the Reach.*

    d. In the end, one wonders why the 2D model was used for the entire Hanford reach. Would it not have been more efficient and productive to concentrate the modeling and validation only at the important areas? The data collected represents an enormous effort and will be valuable for some time to come, but it seems to have come at the expense of focusing the study.

Response: *We chose to model the entire Hanford Reach because our original plan was to use River2D simulations for all of our analyses, including entrapment modeling. As we discovered later, we did not have sufficient experience with River2D to run it in an unsteady mode to reproduce hourly flow fluctuations for entrapment modeling. We will be using Reach-wide River2D output to establish 10 kcfs shorelines for new entrapment modeling during spring of 2007. In addition, escapements to the Hanford Reach have likely not been at carrying capacity, and we wanted to know about areas Reach-wide that had the characteristics of spawning habitat, even if they had not been identified as "important" areas.*

Did the substrate data end up getting used at all? It would have been useful for varying the hydrodynamic resistance. It was noted that it would be used in the habitat model, but does not appear even in the list of potential variables, Table 1 of the habitat chapter.

Response: *In the draft report you reviewed, substrate was only used to approximate and support the use of a 0.1 m uniform roughness height as described previously. The final report will include an entirely new, more recent iteration of our spawning habitat modeling exercise. For this iteration, substrate was evaluated and used in the most recent version of the Reach-wide spawning model. Future work with River2D in the Hanford Reach will incorporate additional calibration/validation using roughness heights based on field measured substrate sizes.*

Appendix

Fall Chinook Salmon Habitat Analysis and Entrapment/Stranding Evaluation
in the Hanford Reach of the Columbia River

Peer Reviewer Questions

Following is a set of suggested questions for consideration by peer reviewers of the Hanford Reach report. Reviewer comments can be organized either as direct responses to the specific questions, or as general responses to the individual report sections (keeping in mind specific issues identified in the questions). The first stage of review will focus on the Introduction and Methods sections of the report, and the second stage of review will include the Results and Discussion sections with conclusions. Questions 1-12 will be relevant for the first stage of review, questions 13-18 will be relevant for the second stage of review, while questions 19 and 20 apply to both review stages.

INTRODUCTION

1) Does the Introduction provide sufficient detail on the issues and frame the overall study goals clearly enough for readers not familiar with the Columbia River?

2) Were the objectives clearly stated for the three main portions of the study: Hydrodynamic Modeling, Entrapment/Stranding Evaluation, and Habitat Assessment?

METHODS

I) Hydrodynamic Modeling

3) Were the methods used for hydrodynamic modeling described in sufficient detail?

4) Were the data used to develop the River2D modeling framework (DEM) sufficient for the study's objectives?

5) Were the input data and validation process for River2D sufficient to expect realistic results from the model simulations, given the level of accuracy and precision required for the study's objectives?

6) Was the application of the 1-dimensional hydrodynamic model (MASS1) appropriate for steady state and unsteady flow simulations used to create flow bands and flow fluctuations for the entrapment histories, given the level of accuracy and precision required for the study's objectives?

II) Entrapment/Stranding Evaluation

7) Was the approach used for the entrapment evaluation logical and technically adequate?

8) Were the methods used to enumerate entrapments, create entrapment histories, and develop the impact estimate for the entrapment evaluation described in sufficient detail?

9) Do you have any suggestions for improving the field or analytical approach to address the stranding or entrapment problem?

III) Habitat Assessment

10) Was the approach used for the habitat assessment logical and technically adequate?

11) Were the methods used for the habitat assessment described in sufficient detail?

12) Were the physical and biological data used for the habitat assessment sufficient to meet the study's objectives?

RESULTS

13) Was the presentation of the results for each section clear and understandable?

14) Do the results correspond to the original stated objectives?

DISCUSSION

15) Does the discussion present a sufficiently clear evaluation of the results that leads to logical and defensible conclusions?

16) Does this work provide an analytical set of robust tools to evaluate operational alternatives for the dams in question, to minimize entrapment impacts and to optimize spawning and rearing habitat?

17) Do you have any suggestions on how to improve the suite of predictive tools for analyzing entrapment impacts and optimizing spawning and rearing habitat?

18) How would you prioritize future work on these issues given the stated goals and objectives, and your evaluation of the strengths and weaknesses of the study to date?

GENERAL

19) Were the tables and figures used in each section adequate?

20) Provide any additional clarifying, explanatory, or editorial comments on each section as needed.

Summary of responses to each question in the Peer Reviewer template. (•) within a table cell indicates that an attempt was made by the reviewer to address this specific review question.

	REVIEWERS		
REVIEW QUESTIONS	**Paul Higgins**	**Bill Miller**	**Peter Steffler***
INTRODUCTION			
1.	•	•	
2.	•	•	
METHODS			
I) Hydrodynamic Modeling			
3.	•	•	•
4.	•	•	•
5.	•	•	•
6.	•	•	•
II) Entrapment/Stranding Evaluation			
7.	•	•	
8.	•	•	
9.	•	•	
III) Habitat Assessment			
10.	•	•	
11.	•	•	
12.	•	•	
RESULTS			
13.	•	•	•
14.	•	•	•
DISCUSSION			
15.	•	•	
16.	•	•	•
17.	•	•	•
18.	•		
GENERAL			
19.	•	•	
20.	•	•	•

* Note that Peter Steffler's responses for Results and Discussion questions are applicable only to the Hydrodynamic Modeling section

APPENDIX B

DESCRIPTION OF THE MASS1 HYDRODYNAMIC MODEL

3.0 Description of MASS1

The Modular Aquatic Simulation System 1D (MASS1) is a one-dimensional, unsteady, cross-section averaged flow and water quality model. A single value of water surface elevation, discharge, velocity, concentration, and temperature is computed at each point in the model at each time interval. Lateral and vertical variations of these quantities are not simulated. Bathymetric data are a primary requirement of any surface water hydrodynamic and transport model. MASS1 requires bathymetry (bottom elevations) input as a series of cross sections. A cross section is a series of elevations along a line (not necessarily straight) extending laterally across the river. Cross sections can be either prismatic (rectangular, trapezoidal, etc.) or natural sections defined from topographic or bathymetric surveys.

The other primary data requirements of MASS1 are river inflows, which can include main stem flow at the upstream end of the network as well as distributed lateral inflows and tributary inflows, and river water surface elevation at the control section located at the downstream end of the network. MASS1 can be used only for subcritical flow (flow having a Froude number less than 1).

3.1 Hydrodynamics

Unsteady flow in rivers and canals is simulated in MASS1 by solving the one-dimensional equations of mass (Equation 3.1) and momentum (Equation 3.2) conservation. These equations are often referred to as the St. Venant equations:

$$\frac{\partial A}{\partial t} + \frac{\partial Q}{\partial t} = 0 \tag{3.1}$$

$$\frac{\partial Q}{\partial t} + \frac{\partial}{\partial x}(\alpha\frac{Q^2}{A}) + gA\frac{\partial y}{\partial x} + gAS_f = 0 \tag{3.2}$$

where:

A = river cross-sectional area, ft^2
Q = water discharge, ft^3/sec
y = water surface elevation, ft
S_f = friction slope, ft/ft, as defined in (3.3)
α = momentum friction correction factor
t = time, s
x = coordinate along the channel, ft.

The friction slope term can be computed using either the Manning or Chezy equations (see Chow 1959). In MASS1, the friction slope is expressed in terms of the discharge and channel conveyance (***K***) as:

$$S_f = \frac{Q\,|\,Q\,|}{K^2} \tag{3.3}$$

and the conveyance is computed using the Manning equation:

$$K = \frac{C_0}{n} A R^{2/3} \tag{3.4}$$

where:

C_0 = 1.49 for English units and 1.0 for metric units
n = Manning channel roughness coefficient
R = hydraulic radius, ft
= ***A/P***
P = channel wetted perimeter, ft.

Equations 3.3 and 3.4 represent the combined effects of variable channel geometry and resistance to flow (roughness) on the hydrodynamic simulation.

The average shear stress acting on the channel bottom can be computed from:

$$\tau = \gamma R S_f \tag{3.5}$$

where:

τ = bed shear stress, lb/ft2
γ = unit weight of water, lb/ft^3

MASS1 can simulate transport of general species and thermal energy (temperature), but these quantities were not included in this study.

3.2 Solution Methods

The foregoing equations are a coupled system of nonlinear partial differential equations. In general, analytical solutions to these equations can only be obtained for simplified channel geometries and boundary conditions. Therefore, numerical methods must be used to solve these equations for most practical situations. Finite difference methods are used in MASS1. The hydrodynamic Equations 3.1 and 3.2 are discretized using the Preissmann four-point implicit finite difference scheme, and the resulting system of nonlinear algebraic equations is solved using the double sweep method, as described in Cunge et al. (1980).

3.3 Model Topology

The first step in developing the numerical solution procedures implemented in MASS1 is to define the topology of the river systems that can be simulated. Here the topology defines how the channel system is connected, as well as the location and type of hydraulic control structures. The topology of the channel system is represented by dividing the river system into a series of links that are then further divided into a series of computational points along that link (Figure 3.1). Nodes occur at upstream or downstream boundary points and at the junction of two or more links.

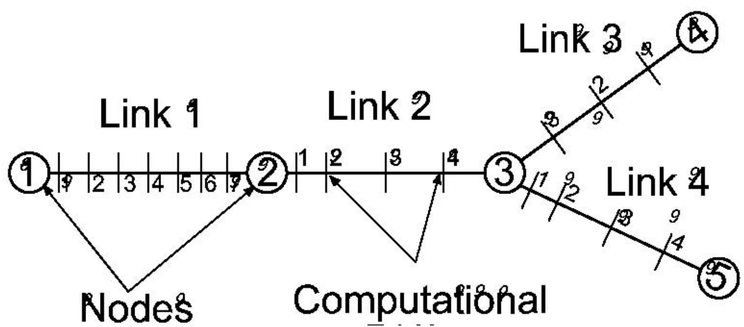


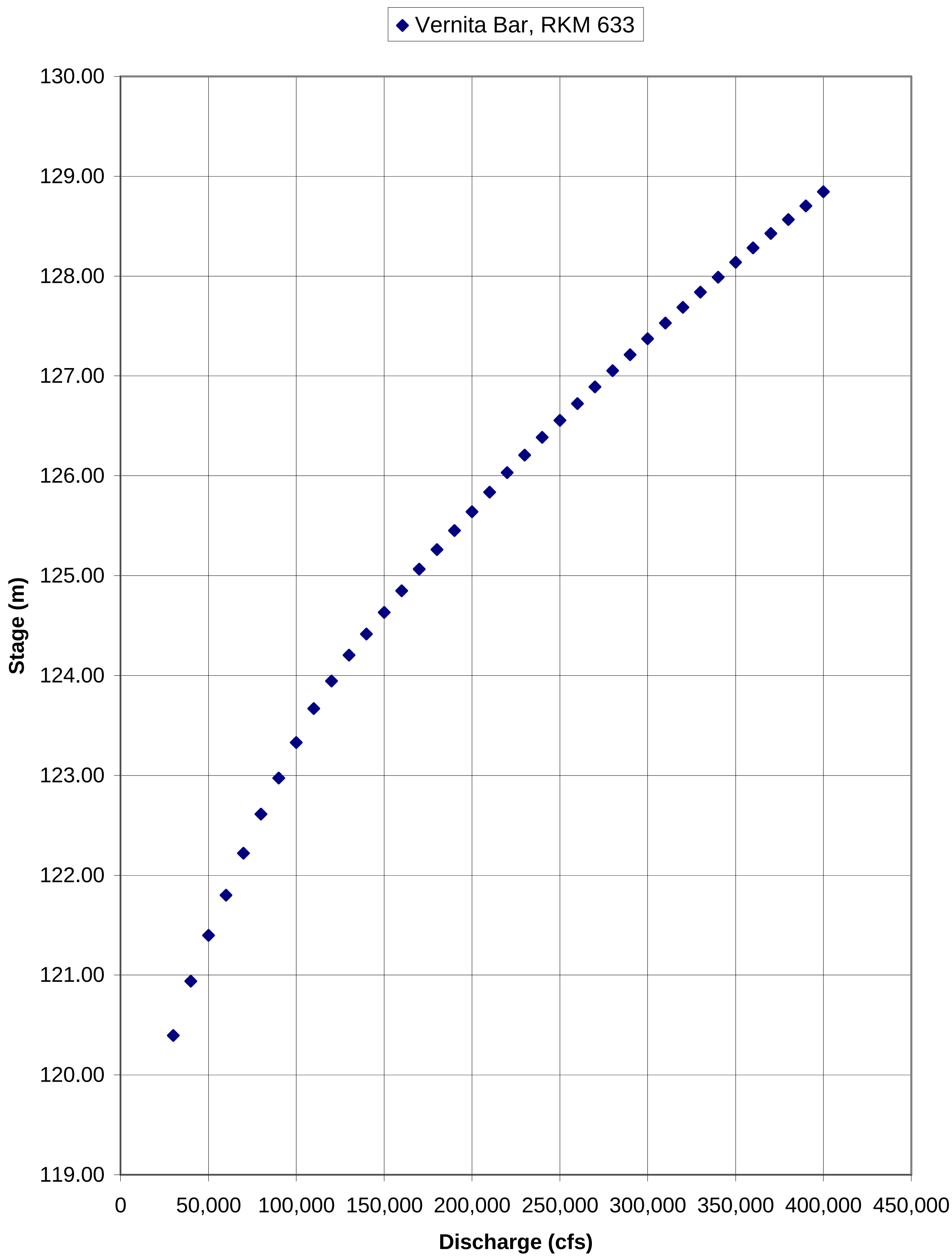
Figure 3.1. Typical MASS1 topological scheme.

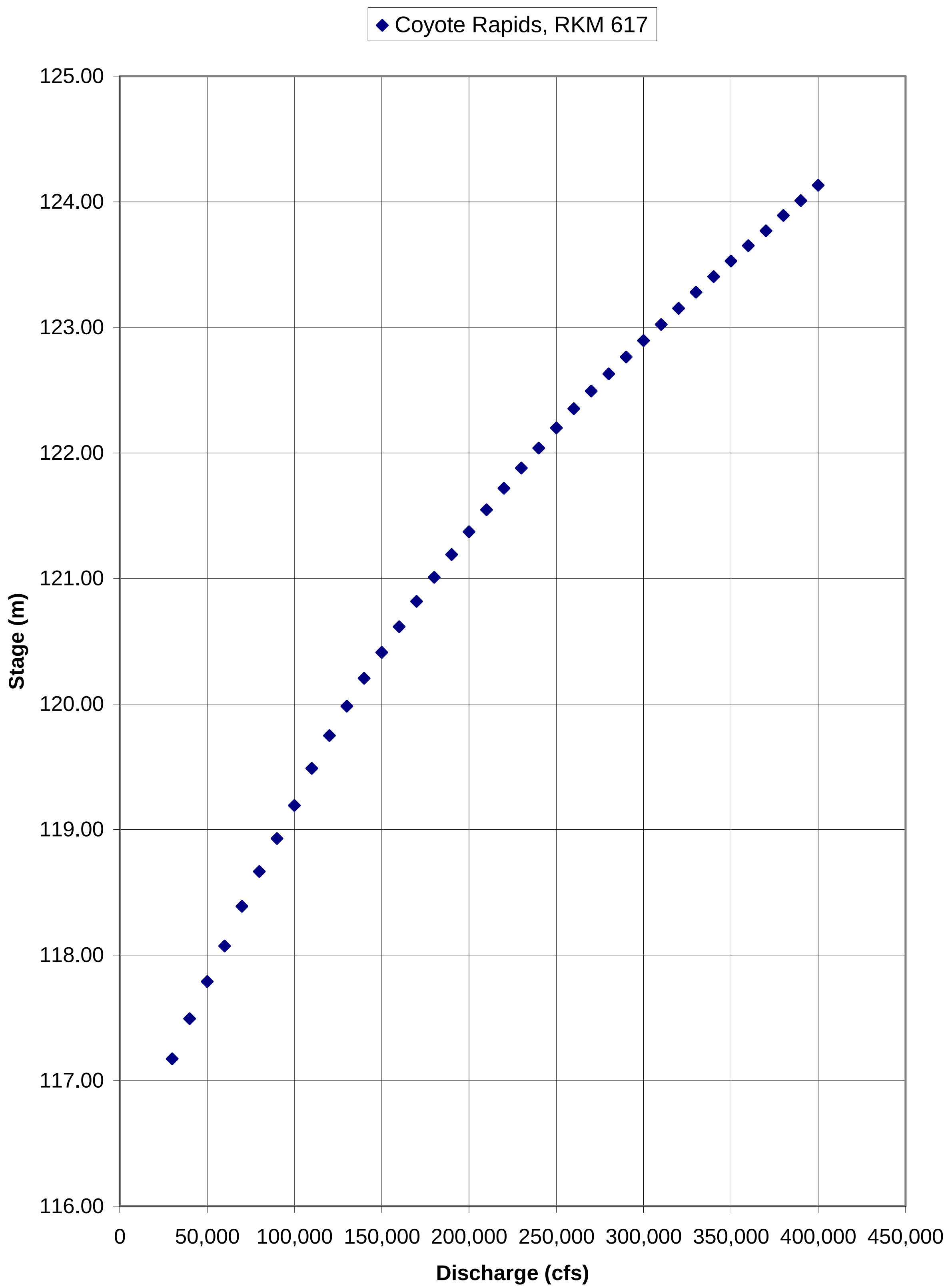
This material was excerpted from the following document:

Hydrodynamic Simulation of the Columbia River, Hanford Reach, 1940 - 2004
S. R. Waichler, W. A. Perkins, M. C. Richmond
June 2005
Prepared for the U.S. Department of Energy, Contract DE-AC05-76RL01830
Pacific Northwest National Laboratory
Richland, Washington 99352

APPENDIX C

MASS1 RATING CURVES


Vernita Bar, RKM 633
130.00
129.00
128.00
127.00
126.00
125.00
124.00
123.00
122.00
121.00
120.00
119.00
Stage (m)
0
50,000
100,000
150,000
200,000
250,000
300,000
350,000
400,000
450,000
Discharge (cfs)



Coyote Rapids, RKM 617
Stage (m)
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123.00
122.00
121.00
120.00
119.00
118.00
117.00
116.00
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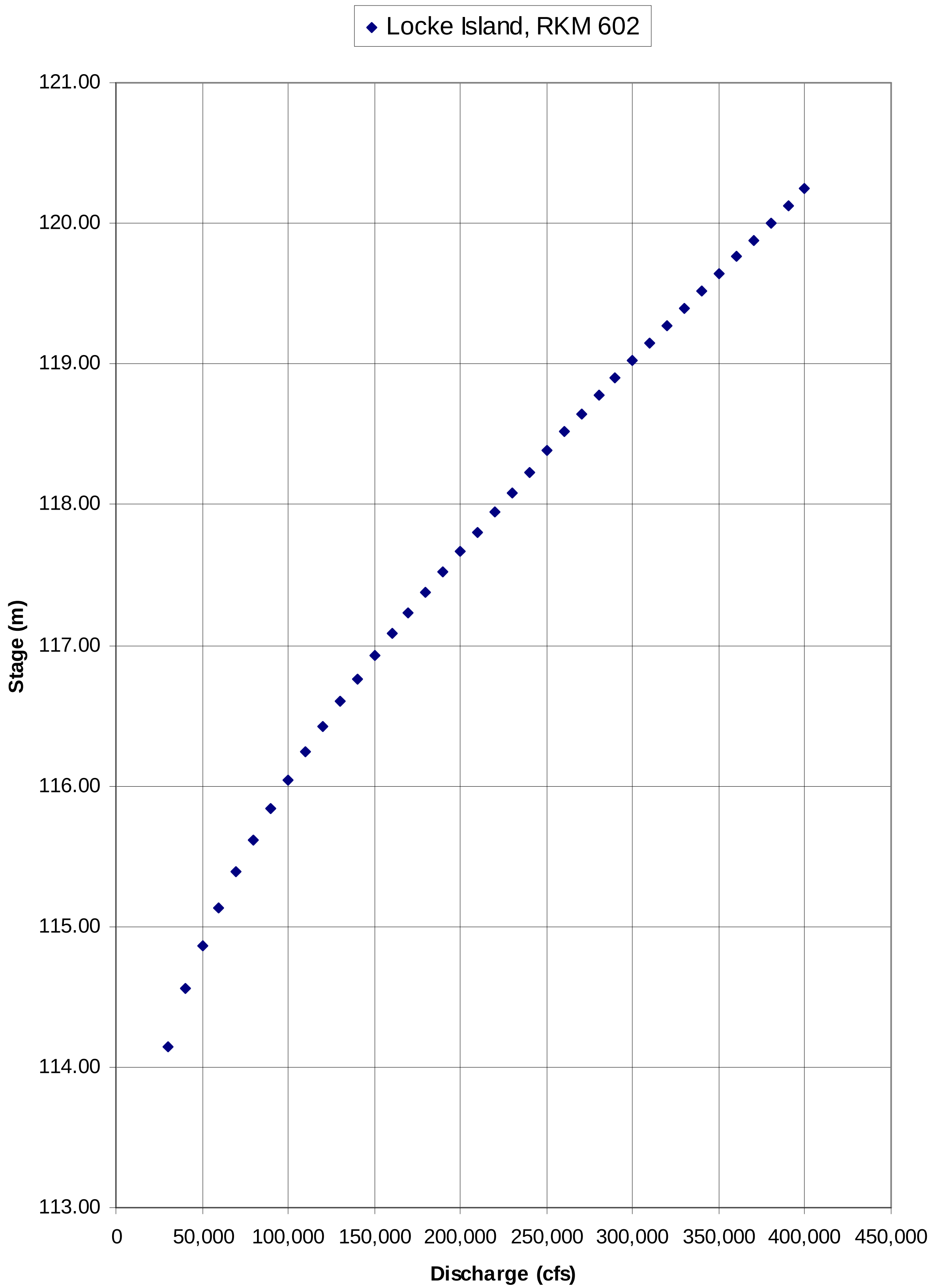
150,000
200,000
250,000
300,000
350,000
400,000
450,000
Discharge (cfs)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118.00
117.00
116.00
115.00
114.00
113.00
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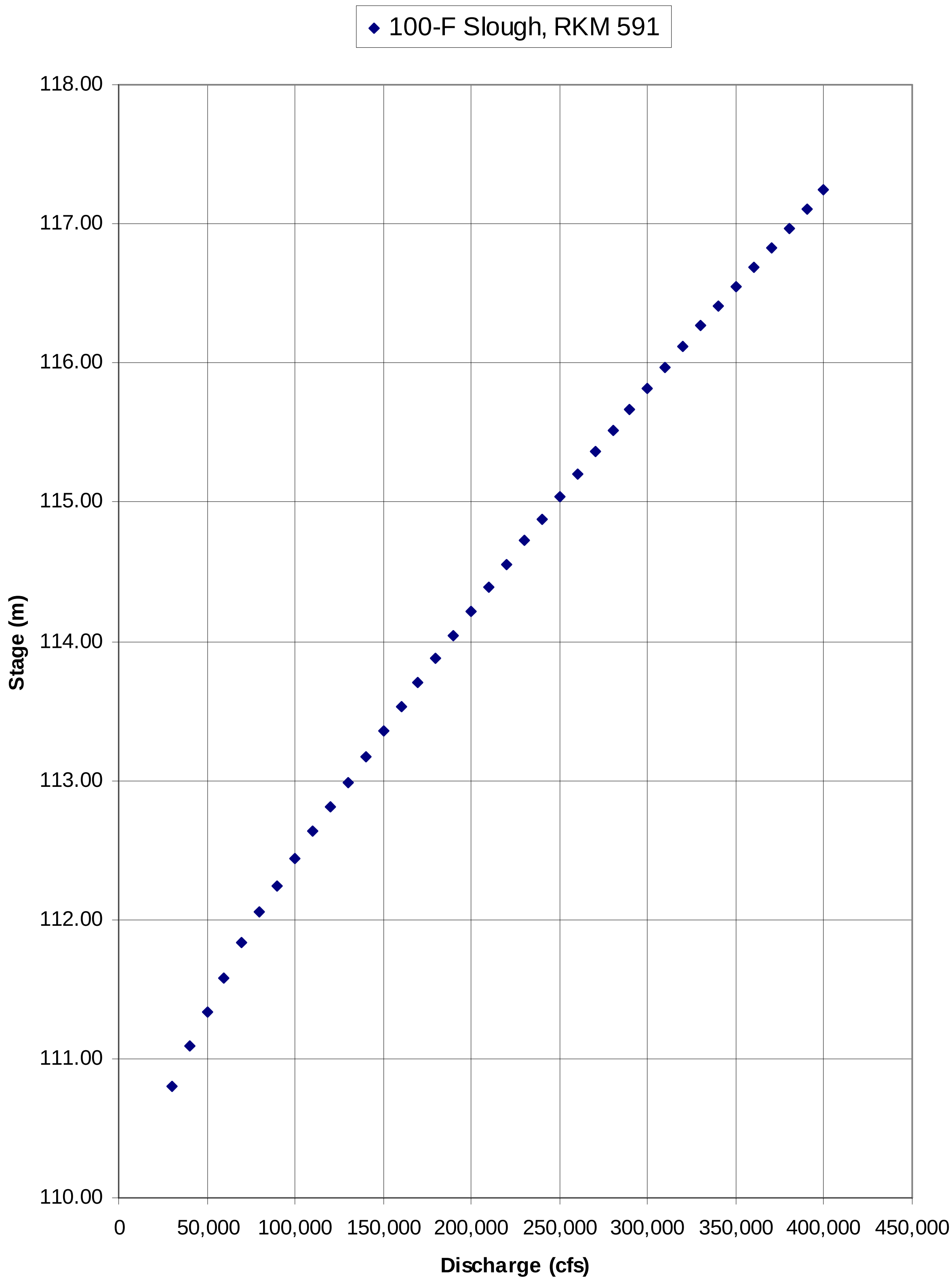
100,000
150,000
200,000
250,000
300,000
350,000
400,000
450,000
Discharge (cfs)


100-F Slough, RKM 591
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117.00
116.00
115.00
114.00
113.00
112.00
111.00
110.00
Stage (m)
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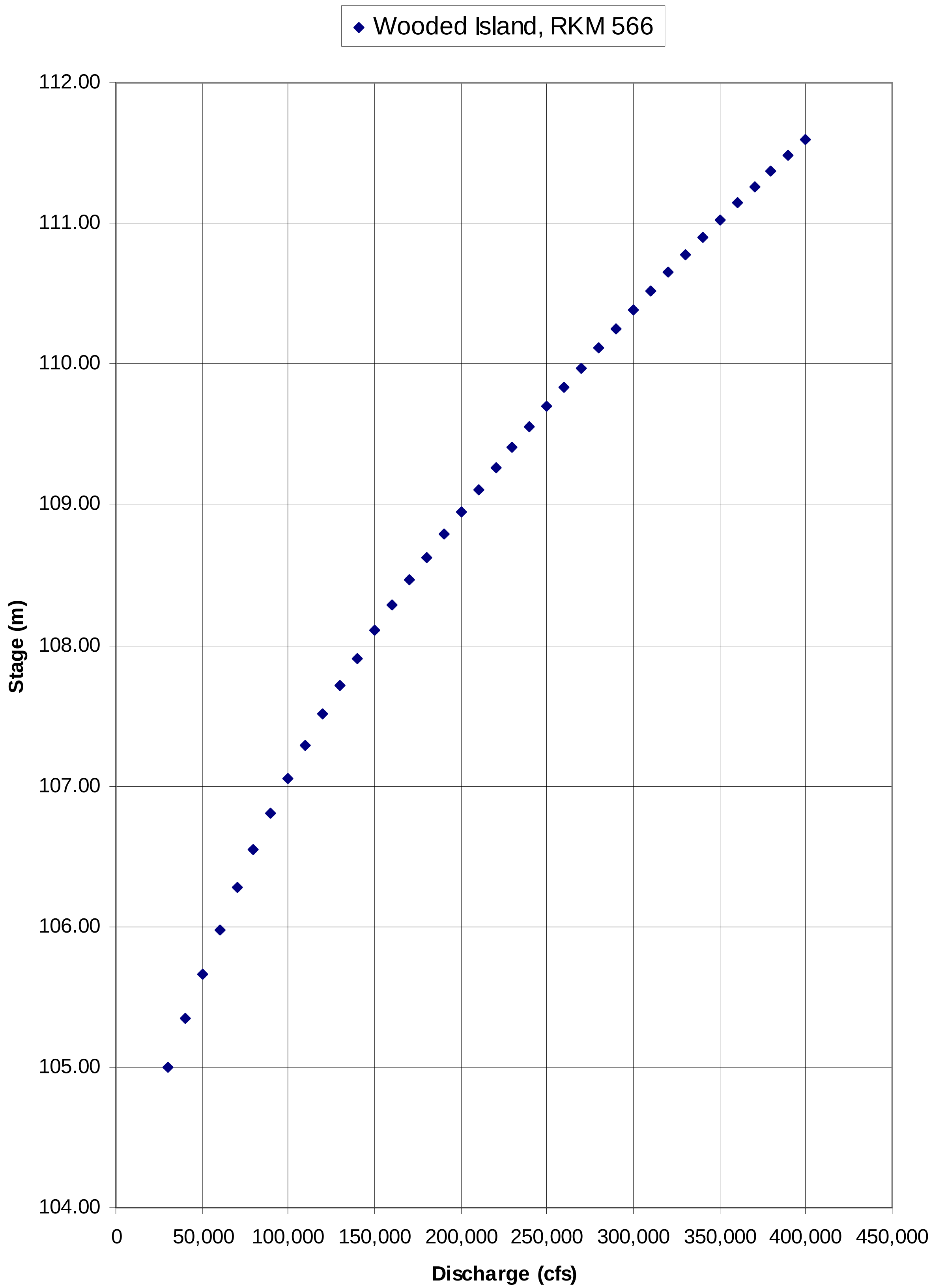
100,000
150,000
200,000
250,000
300,000
350,000
400,000
450,000
Discharge (cfs)


Wooded Island, RKM 566
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111.00
110.00
109.00
108.00
107.00
106.00
105.00
104.00
Stage (m)
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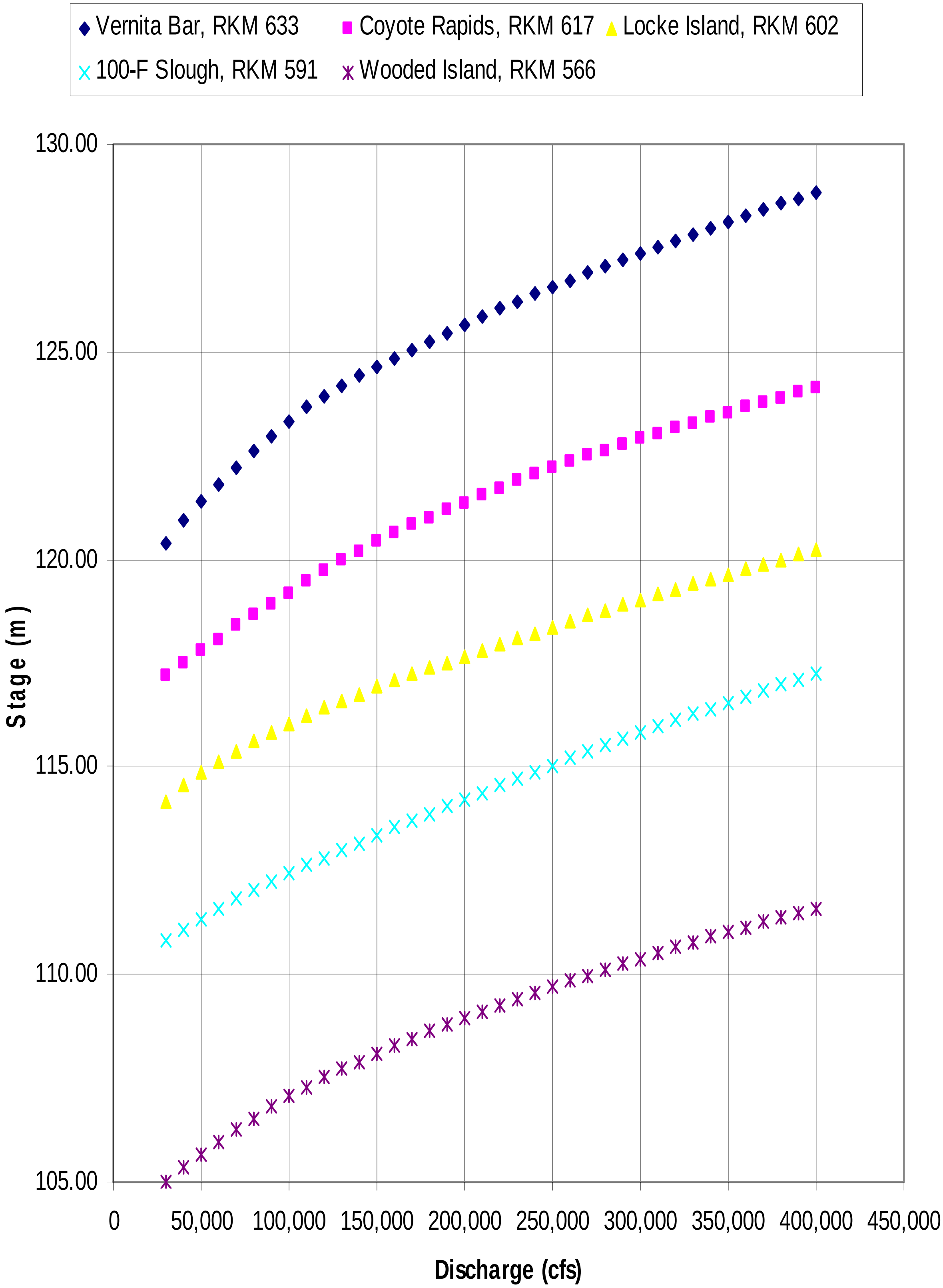
50,000
100,000
150,000
200,000
250,000
300,000
350,000
400,000
450,000
Discharge (cfs)


Vernita Bar, RKM 633
Coyote Rapids, RKM 617
Locke Island, RKM 602
100-F Slough, RKM 591
Wooded Island, RKM 566
130.00
125.00
120.00
115.00
110.00
105.00
Stage (m)
0
50,000
100,000
150,000
200,000
250,000
300,000
350,000
400,000
450,000
Discharge (cfs)