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1998 EVALUATION OF JUVENILE FALL CHINOOK SALMON STRANDING ON THE HANFORD REACH OF THE COLUMBIA RIVER

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1998 Evaluation of Juvenile Fall Chinook Salmon Stranding on the Hanford Reach of the Columbia River

Prepared for
The Bonneville Power Administration
The Public Utility District Number 2 of Grant County

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Executive Summary

The Washington Department of Fish and Wildlife (WDFW) in cooperation with the Bonneville Power Administration (BPA), Grant County Public Utility District (GCPUD), Pacific Northwest National Laboratory (PNNL), University of Idaho (U of I), Streamside Programs Consultation (SPC), and United States Geological Survey Biological Resources Division (USGS/BRD) performed the 1998 Evaluation of Juvenile Fall Chinook Salmon (*Oncorhynchus tshawytscha*) Stranding on the Hanford Reach of the Columbia River. The 1998 evaluation, in the second year of a multi-year study, assessed the impacts of water fluctuations from Priest Rapids Dam on rearing juvenile fall chinook salmon, other fish species, and benthic macroinvertebrates. The field effort was performed from March 12 through October 5. The study area encompassed the 82 km of the Hanford Reach from Priest Rapids Dam to Richland Washington.

The objectives of the 1998 evaluation were to collect basic information on the physical parameters of the Hanford Reach, evaluate the extent of stranding and entrapment of juvenile fall chinook salmon and other fish species, and identify critical habitat zones. The 1998 evaluation was exploratory. The data will be used to generate a sampling design for 1999. The information also will be used in the development of a model to determine susceptibility of juvenile fall chinook salmon to stranding and entrapment due to river elevation fluctuations. WDFW subcontracted U of I and SPC to assess the effects of river fluctuations on the benthic macroinvertebrate communities and USGS/BRD to study the effects of heat stress on the survival, predator avoidance ability, and physiology of juvenile fall chinook salmon. The United States Army Corps of Engineers (COE) was subcontracted to collect detailed bathymetry data on the Hanford Reach using the Scanning Hydrographic Operational Airborne Lidar Survey (SHOALS) system.

River conditions on the Hanford Reach in 1998 were characterized by below average river flows, above normal ambient air temperatures, near normal precipitation, and near average solar radiation levels.

Juvenile fall chinook salmon first appeared in both nearshore and entrapment sites on March 19 and were last encountered in entrapments on June 24 and last sampled in nearshore sites on June 27. Peak numbers of individuals and mortalities were observed between early April and early May. Juvenile fall chinook salmon distribution and mortality was highest at island complex areas. Individuals less than 59 mm in length were most susceptible to entrapment. Juvenile fall chinook salmon appeared to be most vulnerable to reductions in flow at night, in the first 0.9 m of vertical flow reduction, and when reductions occurred between 100 kcfs and 140 kcfs. Juvenile fall chinook salmon were found to be stranded/entrapped by the smallest flow reductions measurable. The majority of stranding mortalities (94.4%) occurred within 24 hours of the entrapment creation time while most thermal mortalities (99.8%) took place within three days. Stranding mortality occurred more often over coarse unembedded substrates while thermal mortality took place more frequently over fine embedded substrates. Juvenile fall chinook salmon were found most regularly in areas absent of vegetation.

Other fish species found stranded and entrapped in 1998 included northern pikeminnow (*Ptychocheilus oregonensis*), redside shiner (*Richardsonius balteatus*), sucker (*Catostomus* spp.), peamouth (*Mylocheilus caurinus*), threespine stickleback (*Gasterosteus aculeatus*), sculpin (*Cottus* spp.), smallmouth bass (*Micropterus dolomieui*), mountain whitefish (*Prosopium williamsoni*), dace (*Rhinichthys* spp.), common carp (*Cyprinus carpio*), lamprey (*Lampetra* spp.), bullhead (*Ameiurus* spp.), and yellow perch (*Perca flavescens*). These species were affected at different times and lifestages.

Artificial substrate type, colonization variability, and sample sites were determined by U of I and SPC in 1998 and used to develop the study plan for the 1999 benthic macroinvertebrate evaluation. USGS/BRD found that juvenile fall chinook salmon exposed to thermal stressors similar to those found on the Hanford Reach had no increased vulnerability to predation. COE collected detailed bathymetry on 35.1 km^2 of the Hanford Reach from Rkm 571.3 to Rkm 606.9.

WDFW and the joint fish managers recommended that operations at Priest Rapids Dam create no fluctuations and/or steadily increasing flows on the Hanford Reach of the Columbia River throughout the fall chinook salmon emergence and rearing period. This recommendation was provided to the power managers who subsquentially proposed a protection plan to meet the follow criteria: 1) substantially more protection for juvenile fall chinook fry than occurred in 1998, 2) preservation of opportunity for load-following/power

peaking operations, 3) allow system coordinated river operations, 4) provide ability to monitor and evaluate in-season and adaptively manage operations to reduce stranding and entrapment. The proposed plan set forth the following operating scenarios: 1) limit daily fluctuations to a range of +/-20 kcfs (a range of 40 kcfs) when weekly average flows are less than 170 kcfs, 2) limit daily fluctuations to a range of +/-30 kcfs (a range of 60 kcfs) when weekly average flows are above 170 kcfs, and 3) rewetting of entrapment zones. Further development of the interim protection plan should be continued in 1999.

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Introduction

The Washington Department of Fish and Wildlife (WDFW) has been contracted through the Bonneville Power Administration (BPA) and the Grant County Public Utility District (GCPUD) to perform an evaluation of juvenile fall chinook salmon (*Oncorhynchus tshawytscha*) stranding on the Hanford Reach. The evaluation, in the second year of a multi-year study, has been developed to assess the impacts of water fluctuations from Priest Rapids Dam on rearing juvenile fall chinook salmon, other fish species, and benthic macroinvertebrates of the Hanford Reach. This document provides the results of the 1998 field season.

Background

The background section for this document includes, the impetus for the evaluation, a description of the environmental conditions that exist on the Hanford Reach, and a summary of the 1997 report.

Impetus for the Evaluation

The BPA has been directed by the National Marine Fisheries Service (NMFS) under the Endangered Species Act - Section 7 - Biological Opinion on the Reinitiating of Consultation on 1994-1998 Operation of The Federal Columbia River Power System and Juvenile Transportation Program to perform the following:

> "Beginning in 1995, BPA will evaluate the affect of power peaking operations on juvenile and adult salmon passage and on the river ecology downstream of Bonneville Dam and on the Hanford Reach, downstream of Priest Rapids Dam. Contingent on the results of these evaluations BPA will develop a plan to decrease power peaking operations from mid-March through mid-December on the lower Snake and Columbia Rivers." (page 162, #11)

In addition, as an objective of the 1994 Columbia River Basin Fish and Wildlife Program, BPA has been directed to perform the following:

> "Beginning in 1995, evaluate alternative ramping rates for flow fluctuations at mainstem Snake and Columbia River dams to constrain reductions and increases in total flow per 24-hour period at these projects." (Page 5-20, 5.1D.4)

This evaluation of juvenile fall chinook salmon stranding on the Hanford Reach is consistent with both of these objectives.

Description of Stranding and Entrapment Conditions on the Hanford Reach

The Hanford Reach supports the larger of the only two remaining healthy naturally spawning fall chinook salmon populations in the Columbia River System (Huntington et al.1996). This population is a primary source of ocean and freshwater sport, commercial, and in-river tribal fisheries (Dauble and Watson 1997) and is a primary component of the Pacific Salmon Treaty between the United States and Canada. River flows for this section of the Columbia River are manipulated by discharge from Priest Rapids Dam. Flow fluctuations from Priest Rapids Dam occur rapidly due to changes in hydroelectric power generation (power peaking), irrigation, water storage, flood control, and other causes. These fluctuations have been observed to cause stranding and entrapment of juvenile fall chinook salmon on gently sloped banks, gravel bars, and in pothole depressions on the Hanford Reach (Page 1976, Becker et al. 1981, DeVore 1988, Geist 1989, Wagner 1995, Ocker 1996).

Stranding of juvenile fall chinook salmon occurs when the fish are trapped on or beneath the unwatered substrate as the river level recedes. Entrapment occurs when the fish are separated from the main river channel in depressions as the river level recedes. Entrapped fish may become stranded when depressions drain completely. Fish mortality occurs from stranding, warming of water in entrapments (thermal stress), and by possible piscivorous and avian predation in small shallow entrapments.

The impact of river fluctuations due to operation of hydroelectric facilities on rearing salmonids has been assessed on numerous Columbia River tributaries and other river systems (Thompson 1970, Witty and Thompson 1974, Phinney 1974a and 1974b, Bauersfeld 1978, Tipping et al. 1978 and 1979, Becker et al. 1981, Woodin 1984, and Beck 1989) but limited research has been conducted on the Hanford Reach (Page 1976, Becker et al. 1981). The 1998 evaluation has been performed to provide baseline information on the stranding and entrapment of juvenile fall chinook salmon on the Hanford Reach and for directing the future management of flows from Priest Rapids Dam.

Description of the Hanford Reach

The Hanford Reach stretches from Priest Rapids Dam 82 km downstream to Richland Washington. The physiography, river dynamics, and climate of the area create a unique habitat for wildlife and fish populations.

Physiography

Lands along the Hanford Reach are owned and/or managed by the United States Department of Energy (DOE) Hanford Site, the United States Fish and Wildlife Service (USFWS) Saddle Mountain National Wildlife Refuge, and the WDFW Waluke Wildlife Management Area (**Figure 1**). In 1943, the United States Atomic Energy Commission requisitioned these lands surrounding the Hanford Reach for the siting of facilities to produce plutonium for the first atomic weapons. Due to the secure nature of the facilities, the Hanford Reach and the surrounding lands have remained protected and only limited development has occurred in small intensely disturbed areas surrounding the facilities. The undeveloped areas contain one of the largest remnant sections of shrub-steppe ecosystem in the Columbia River Basin.

For descriptive purposes, the Hanford Reach can be broken down into five distinct river sections. These sections are Priest Rapids Dam (Rkm 639.0) to Coyote Rapids (Rkm 615.6), from Coyote Rapids to the beginning of the White Bluffs (Rkm 605.1), from the beginning of the White Bluffs to Hanford Slough (Rkm 582.6), Hanford Slough to Savage Island (Rkm 572.9), and from Savage Island to the McNary Pool (Rkm 545.6) in Richland. Detailed plan views of the Hanford Reach are provided in **Figures 2, 3, and 4**.

The first segment of river from Priest Rapids Dam to Coyote Rapids flows to the east. This section of river consists of a series of gentle meanders. The meanders are characterized by cutbanks on the outside of the meanders and point bars on the inside downstream portion of the meanders. The cutbanks in this section are typified by steep embankments or to a lesser extent rock walls. The cutbank from Rkm 637.3 to Rkm 632.4 is an outcropping of basalt associated with the terminus of Umtanum Ridge. Gentle embankments, flats and downstream gravel bars distinguish the point bars in this section. Notable downstream gravel bars critical to fall chinook salmon spawning are Vernita Bar (Rkm 637.3) and a gravel bar immediately upstream of Coyote Rapids at Rkm 616.4.

At Coyote Rapids the river turns and flows to the northeast. The next section of river from Coyote Rapids to the beginning of the White Bluffs is straight and channelized with relatively steep embankments. Some fall chinook salmon spawning occurs at the top of the island at Rkm 606.7.

At the beginning of the White Bluffs, the river makes an abrupt turn to the southeast. Unconsolidated bluffs on the northeast bank and island complexes dominate this next section of river from the beginning of the White Bluffs to the bottom of Hanford Slough. The river becomes braided through this segment and the bluffs rise to greater than 150 m above the surface of the river. The island complexes with associated islands, gravel bars, and backwater sloughs provide extensive critical spawning and rearing habitat for fall chinook salmon.

Below Hanford Slough the river continues to flow to the southeast to the bottom of Savage Island. This section of the river from the bottom of Hanford Slough to the bottom of Savage Island is straight and channelized with relatively steep embankments. No observed fall chinook salmon spawning occurs in this section of the river.

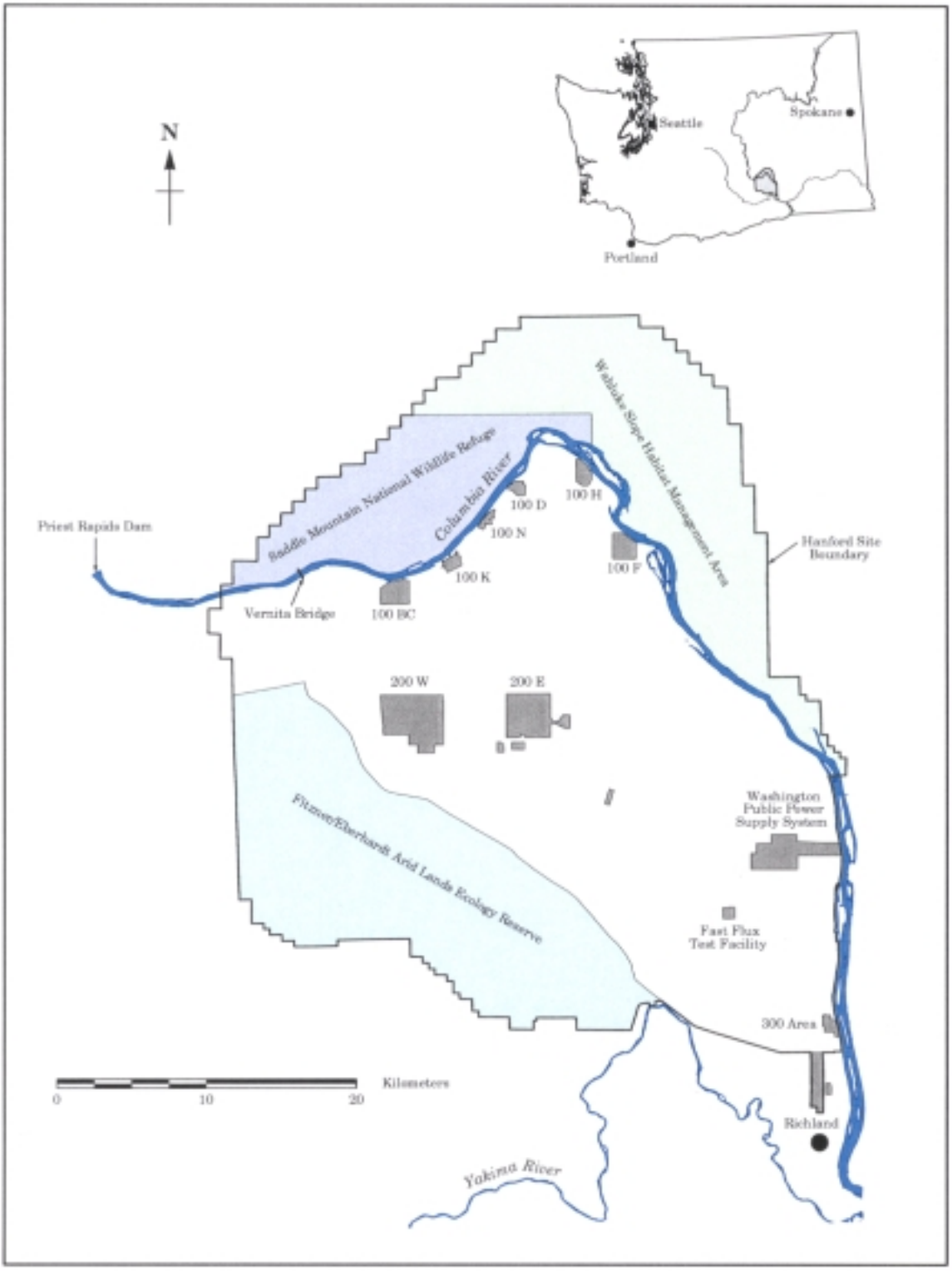


Figure 1. The Hanford Reach of the Columbia River, Washington.

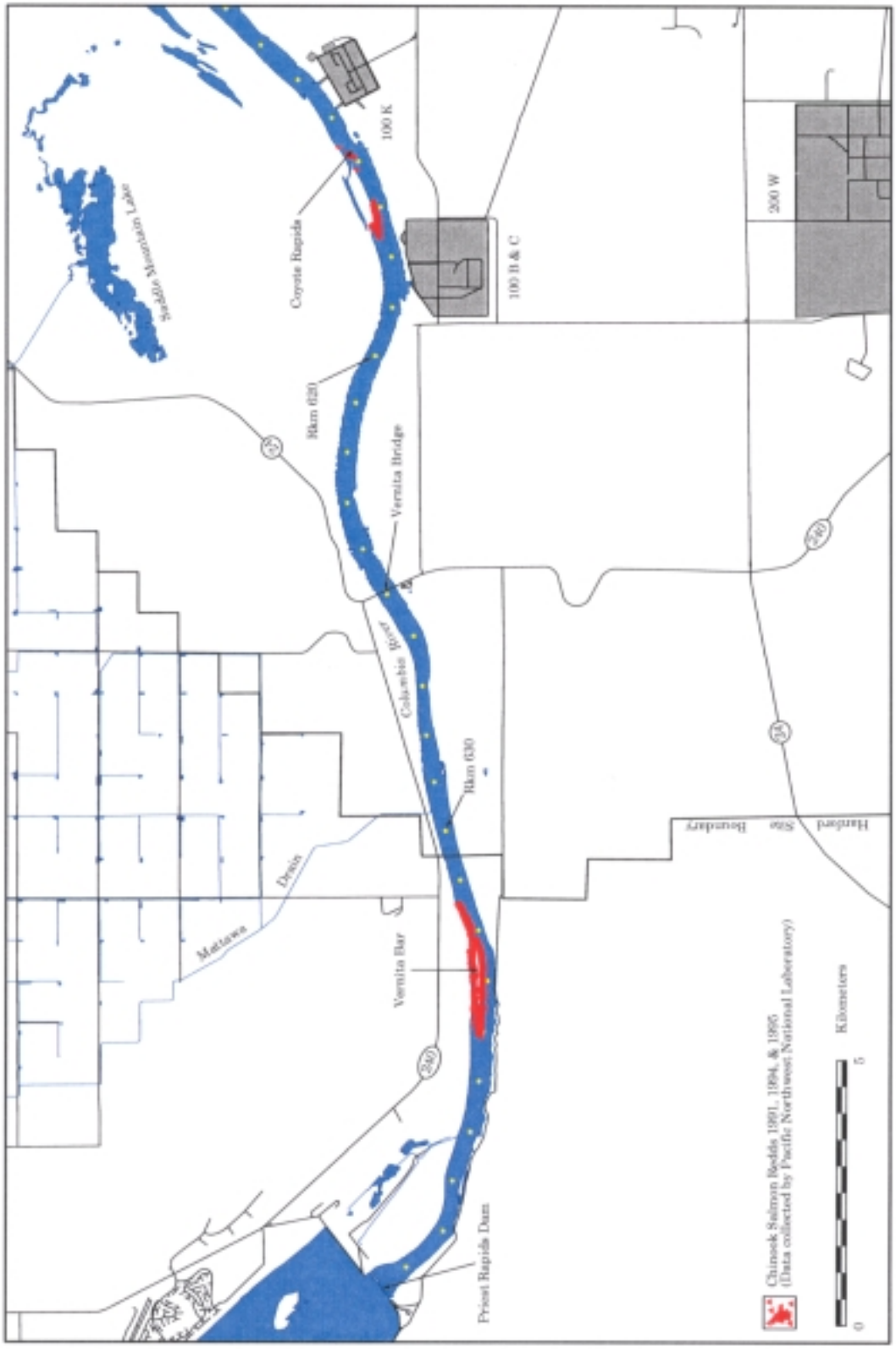


Figure 2. Plan view of the Hanford Reach of the Columbia River from Priest Rapids Dam to Coyote Rapids.

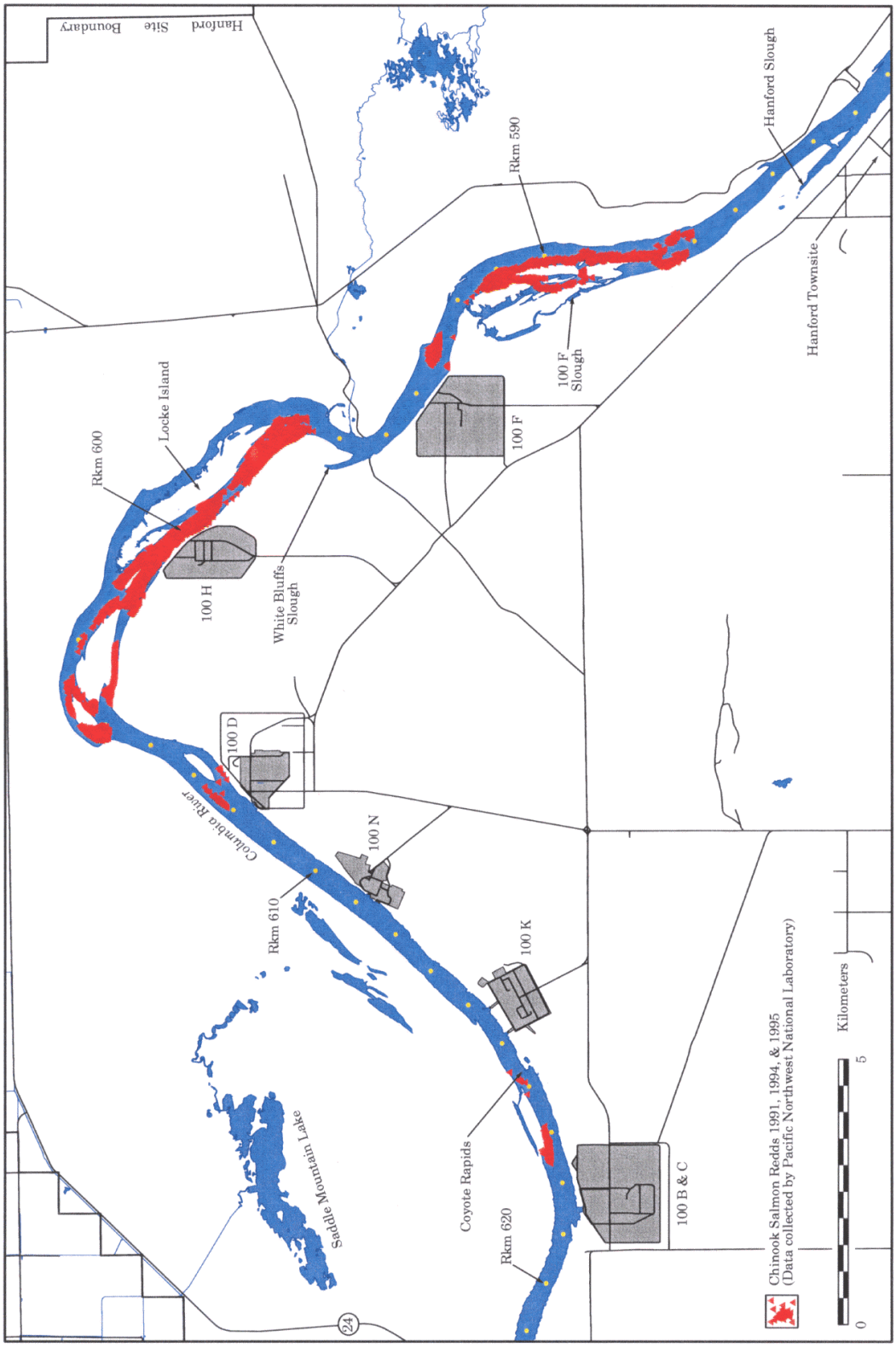


Figure 3. Plan view of Hanford Reach of the Columbia River from Coyote Rapids to Hanford Slough.

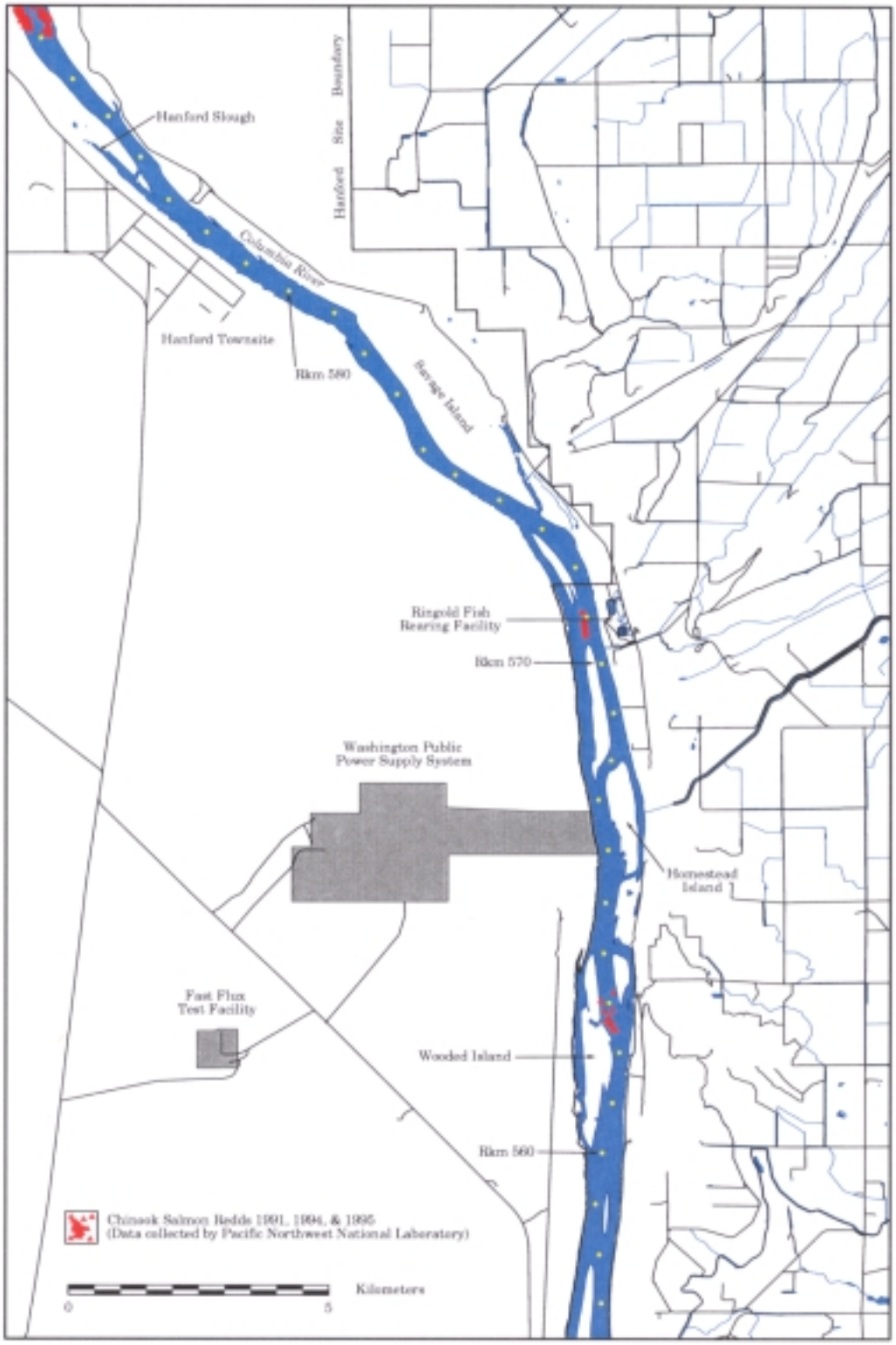


Figure 4. Plan view of Hanford Reach of the Columbia River from Hanford Slough to Richland.

Below Savage Island the river turns to the south. Unconsolidated bluffs on the eastern bank and sand dunes and steep embankments on the western bank dominate this final section of the Hanford Reach, from the bottom of Savage Island to the top of the McNary Pool in Richland. The river channel is incised and straight and island formation appears restricted by the river channel. Braiding is less pronounced than in upper stretches of the river providing less gravel bar and backwater areas. Fall chinook salmon spawning occurs at the top of the main channel island adjacent to the Ringold fish hatchery (Rkm 570.5) and near Wooded Island (Rkm 561.6).

Climate

The Hanford Reach, situated in the rain shadow of the Cascade Mountain Range, receives an annual mean precipitation of 16.1 cm and is considered mid-latitude semi-arid (Glantz et al. 1990). Most of the precipitation falls between October and May (Rickard 1988). Summers are warm and dry with temperatures often exceeding 38.0^{o}C (Glantz et al.1990). Winters are cool with occasional precipitation and outbreaks of cold artic air that can drop temperatures below –18.0^{o}C (Glantz et al.1990).

During the fall chinook salmon emergence and rearing period (March – June) average maximum temperatures range from 14.1^{o}C in March to 28.8^{o}C in June. Average minimum temperatures range from 1.1^{o}C in March to 12.9^{o}C in June. Precipitation averages 4.5 cm during the fall chinook salmon emergence and rearing period. Large diurnal temperature contrasts can occur during this time period due to low relative humidity in combination with intense solar radiation during the day and radiational cooling at night (Hanford Meteorological Station, PNNL 1998).

River Dynamics

The Hanford Reach is the only un-impounded and last free flowing section of the Columbia River above Bonneville Dam. Priest Rapids Dam, built in 1959, regulates flow discharges and is the major influence on river dynamics of the Hanford Reach.

Daily fluctuations in river elevation on the Hanford Reach are the result of power generation from Priest Rapids Dam and can vary significantly on an hourly basis. Historically, under normal project operations, tailwater reductions in excess of seven vertical feet per hour (2.1 m/hr) and 13 vertical feet (4.0 m) within a 24-hour period have occurred during the fall chinook salmon emergence and rearing period.

Seasonal daily average discharges from Priest Rapids Dam range from about 40 kcfs to 250 kcfs (Dauble and Watson 1997). Average seasonal flows from 1988 to 1997 show that spring runoff reaches a peak during mid-June and decreases significantly to a low in September **(Figure 5**). The Federal Energy Regulatory Commission (FERC) has established 36 kcfs as a minimum flow from Priest Rapids Dam (Dauble and Watson 1997).

Downstream fluctuations are dampened by channel configuration and bank storage. Translation time of fluctuations downstream is determined by a variety of factors that may include river configuration, bank storage, and magnitude and duration of the fluctuation. **Figure 6** illustrates the entire flow regime from below Priest Rapids Dam (Rkm 639.0) to the bottom of Wooded Island (Rkm560.6) over a one-week period during the 1998 fall chinook salmon emergence and rearing period. Observations of stranding and entrapment events that occurred during this time period and the number of juvenile fall chinook salmon sampled with each event are also represented.

The Hanford Reach has no natural tributaries and receives little additional influent from other sources. Other minimal sources of influent include irrigation runoff and groundwater discharge.

Summary of 1997 Report

In 1997, WDFW performed pilot fieldwork from May 7 through July 28. The work was performed to aid in the development of a work plan for the 1998 evaluation of juvenile fall chinook salmon stranding on the Hanford Reach.

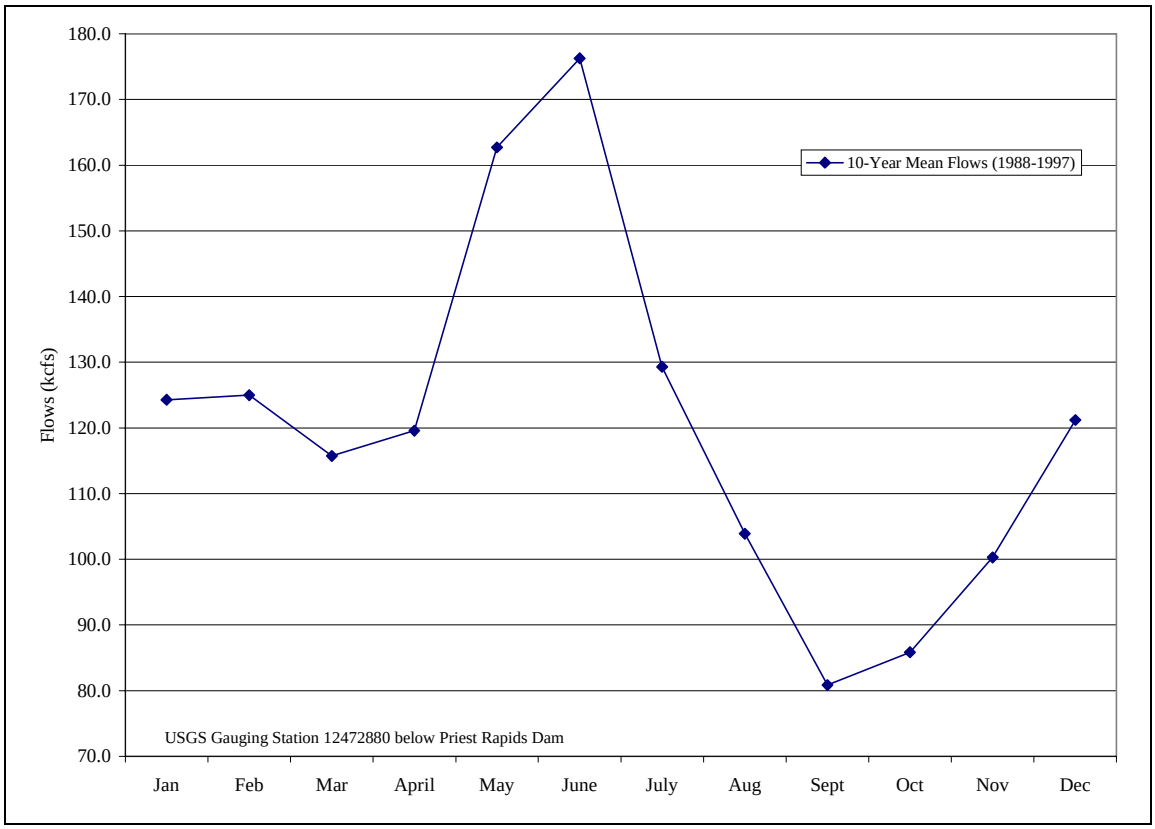


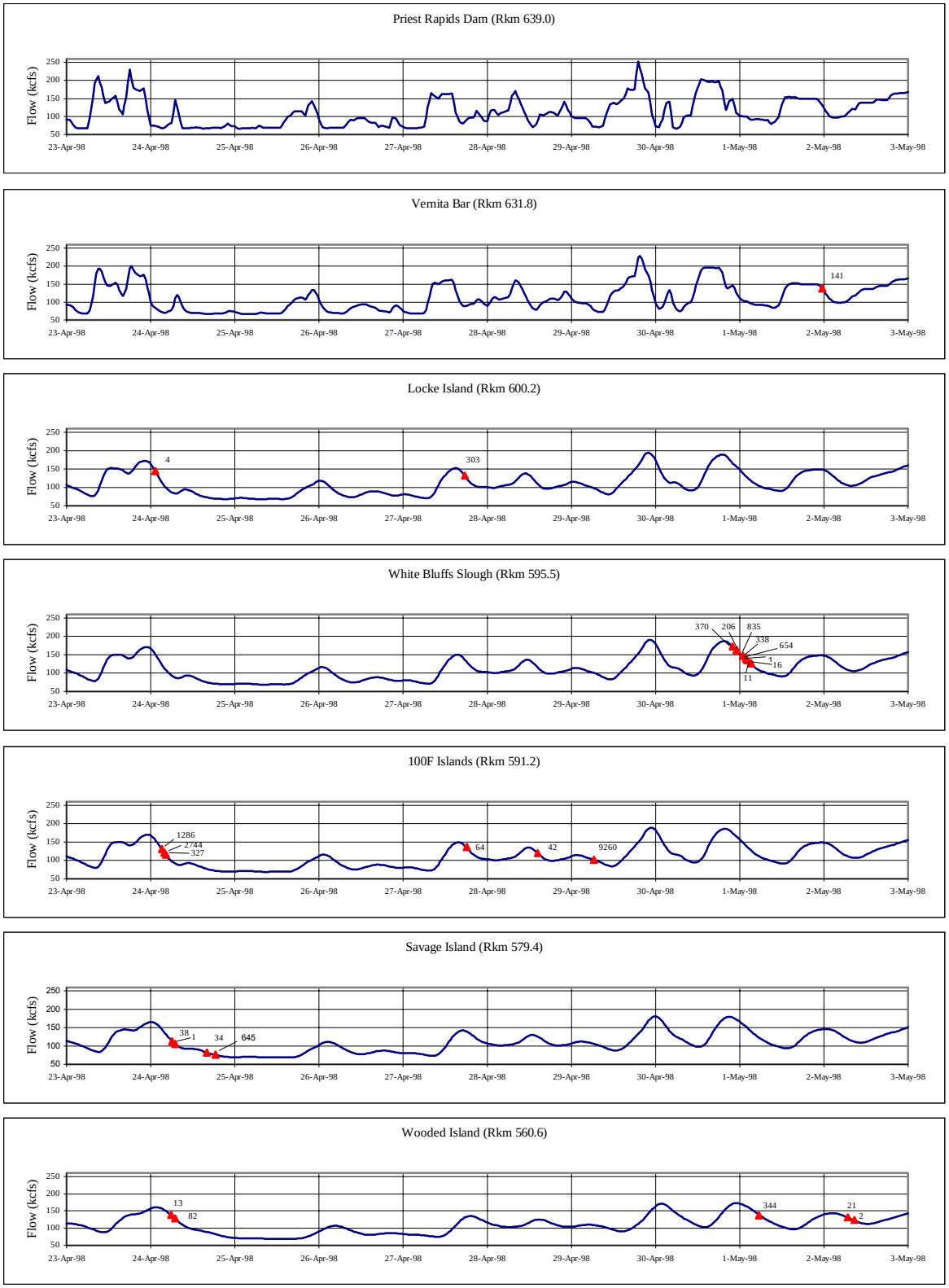
Figure 5. Mean 10-year flows for the Columbia River below Priest Rapids Dam (1988 – 1997).

The Hanford Reach was exposed to exceptionally high river flows in 1997. For example, annual flows in 1997 averaged 169 kcfs compared to only 114 kcfs (range 91 kcfs – 161 kcfs) during the previous ten years (1987 – 1996). In addition, June flows in 1997 averaged 323 kcfs compared to 156 kcfs (range 111 kcfs – 237 kcfs) for the previous ten years.

High spring river flows in 1997 hampered field activities. Field operations that could not be completed included two controlled river elevation reduction tests and the assessment of stranding in cobble substrate. Investigation work completed included the identification of the primary juvenile fall chinook salmon production areas and the determined feasibility of a benthic macroinvertebrate evaluation. In addition, Pacific Northwest National Laboratory (PNNL) completed work on the Modular Aquatic Simulation System 1D (MASS1), a one-dimensional unsteady flow model for the Hanford Reach.

Results of the field investigations indicated that despite the high flow year, juvenile fall chinook salmon as well as other fish species were observed stranded and entrapped. Other fish species found stranded and entrapped included sucker (*Catostomus* spp.), northern pikeminnow (*Ptychocheilus oregonensis*), redside shiner (*Richardsonius balteatus*), peamouth (*Mylocheilus caurinus*), smallmouth bass (*Micropterus dolomieui*), threespine stickleback (*Gasterosteus aculeatus*), sculpin (*Cottus* spp.), common carp (*Cyprinus carpio*), and largemouth bass (*Micropterus salmoides*).

Field observations indicated stranding and entrapment susceptibility of juvenile fall chinook salmon appeared to decrease with increasing fish size. A size threshold of 81 mm was identified as the end of juvenile fall chinook salmon stranding and entrapment susceptibility. Thermal stress and thermal shock appeared to be the primary source of juvenile fall chinook salmon mortality in entrapment areas. Juvenile fall chinook salmon and other fish species demonstrated evidence of specific habitat preference and appeared to be somewhat segregated. Some species of fish appeared to be more susceptible as spawning adults while others were most susceptible as newly hatched fry. These observations were used to formulate objectives for the 1998 work.



▲ Stranding or Entrapment Event (number indicates stranded or entrapped juvenile fall chinook salmon)

Figure 6. Columbia River flow regime from Priest Rapids Dam to Wooded Island (April 23 – May 3, 1998).

Objectives

Seven objectives were developed for the 1998 evaluation. These objectives are as follows:

1) Collect basic information on physical parameters of the Hanford Reach.

2) Evaluate the extent of stranding and entrapment on juvenile fall chinook salmon on the Hanford Reach due to river elevation reductions.

3) Collect information on the effect of river elevation reductions on other adult and juvenile fishes on the Hanford Reach.

4) Identify critical habitat zones where juvenile fall chinook salmon are abundant and are more susceptible to stranding and entrapment as a result of river elevation reductions.

5) Develop a model to determine the primary period of vulnerability of juvenile fall chinook salmon to stranding and entrapment due to river elevation fluctuations.

6) Determine the effect of water fluctuations on the benthic macroinvertebrate community.

7) Determine the effect on juvenile fall chinook salmon of exposure to warm water and/or rapid water temperature change.

Methods

The seven methods used to achieve the established objectives included stranding and entrapment surveys, nearshore river surveys, coarse substrate surveys, a Scanning Hydrographic Operational Airborne Lidar Survey (SHOALS), a benthic macroinvertebrate survey, temperature surveys and thermal stress investigations, and entrapment modeling.

Stranding and Entrapment Surveys

WDFW and GCPUD personnel collected stranding and entrapment data from early spring through early fall. This data was collected to provide basic information on the physical parameters of the Hanford Reach and to evaluate and collect information on the extent of stranding and entrapment of juvenile fall chinook salmon and other adult and juvenile fishes. Data was also collected to provide basic information on the effects of river fluctuations on critical juvenile fall chinook salmon rearing habitat.

Stranding and entrapment surveys were performed during two sampling periods, 1) spring and early summer and 2) summer and early fall. The first sampling period was designed to target the emerging and rearing wild fall chinook salmon while the second period was designed to target other fish species.

Spring and Early Summer

During the spring and early summer, shoreline areas within the 24-hour river fluctuation zone were surveyed for stranding and entrapment sites. Entrapments were defined as depressions within the fluctuation zone separated from the river and containing water. Unwatered depressions within the fluctuation zone were considered potential stranding sites. Sites were selected from a range of river elevations and habitat types throughout the Hanford Reach.

Selected sites were given a specific identification code, a sample time, and a location to the nearest 0.2 Rkm. Each site was sampled only once during the season with the exception of 22 sites used to reference the same location under varying conditions. Physical measurements were collected at each site and water quality measurements were collected from entrapments.

Recorded physical measurements of stranding and entrapment sites included distance to river, elevation of the re-inundation point of the depression, dominant and subdominant substrate, substrate embeddedness, and vegetation density. Physical measurements of entrapments also included size, depth, and drainage rate from the beginning to the end of the sample interval, and drainage or re-inundation times if occurring during the sample interval.

Distance to river was measured as the closest distance from the re-inundation point of the depression to the river along the most probable watercourse. The elevation of the re-inundation point of the depression was determined by surveying the point in relationship to the elevation of the river surface. The river elevation was determined using the sample time and MASS1. Dominant and subdominant substrates were classified according to a modified Wentworth code (Platts et al. 1983); substrate embeddedness was classified according to Platts et al. (1983); and vegetation density was recorded as absent, sparse, medium, or dense (**Appendix A**). The area of the entrapments was estimated by recording one length and three width measurements. Depths were measured at quarter length intervals along the width measurements. Drainage rates were measured by placing a staff gauge in the entrapment and taking water level measurements at the beginning and end of the sample interval.

Water quality measurements recorded from each entrapment and the nearshore area adjacent to the entrapment included dissolved oxygen, percent oxygen saturation, and water temperature. Dissolved oxygen and percent oxygen saturation were recorded at the beginning of the sample interval and water temperatures were recorded at the beginning and end of the sample interval.

All sites were surveyed for juvenile fall chinook salmon and other fish species. Fish found were either classified as at risk if alive or as mortalities. Mortality events were recorded as thermal or stranding. Occurrences in which juvenile fall chinook salmon were found in entrapments with water temperature exceeding 26.0^{o}C were classified and recorded as thermal mortality events. Occurrences in which fish were found unwatered or in entrapments that drained during sampling were classified and recorded as stranding mortality events.

Fish in entrapments were caught using dip nets or a stick seine (9.1 m x 1.8 m with a 1.8 m^2 bag with 4.8 mm diamond mesh). An effort was made to collect all fish in the entrapments. Live fish were anesthetized with tricaine methanesulfonate (MS-222), identified to species (or lowest possible taxa), counted, and measured (fork length). When large samples were encountered, a minimum of 500 fork lengths was measured per species. The fish were allowed to recover and were released into the river. All mortalities were hand collected, identified to species (or lowest possible taxa), counted, and measured.

Summer and Early Fall

Summer and early fall sampling activities for other fish species were performed in collaboration with a United States Geological Survey Biological Resources Division (USGS/BRD) Hanford Reach predation study. A modified sampling method was used to sample fish from entrapments. The 4.8 mm mesh stick seine used during the spring and early summer was replaced by a 15.2 m x 1.2 m with a 1.2 m^2 bag and a 0.8 mm mesh beach seine to capture both larval and larger fish. Large fish captured were anesthetized with MS-222, identified to species (or lowest possible taxa), counted, measured (fork and standard length), allowed to recover, and released back into the river. All larval fish captured were preserved in 10% buffered formalin and transported to the USGS/BRD Columbia River Research Laboratory for identification. All larval fish sent to the laboratory were identified to the lowest possible taxa and enumerated. All physical and water quality measurements for stranding and entrapment were performed using the same methods described for the spring and early summer sampling period.

Nearshore River Surveys

WDFW and GCPUD personnel performed nearshore river surveys from spring through early fall to determine size susceptibility ranges for stranding and entrapment of juvenile fall chinook salmon and other fish species. Spring and early summer sampling was performed to identify critical juvenile fall chinook salmon habitat areas and to obtain a nearshore baseline size structure for comparison with stranded and entrapped juvenile fall

chinook salmon and other fish species. Summer and early fall sampling was performed to allow for a comparison of size and species of fish found in entrapments to nearshore samples.

Spring and Early Summer

Nearshore river surveys were performed concurrent with spring and early summer stranding and entrapment surveys. Nearshore samples were collected adjacent to stranding and entrapment sites.

A large beach seine (21.3 m x 1.8 m with a 1.8 m^2 bag and 4.8 mm diamond mesh) was used to collect juvenile fall chinook salmon and other fish species in the nearshore river area. The seine was pulled perpendicular to the shore to collect fish. All fish collected were anesthetized with MS-222, identified to species, and counted. When large samples were encountered, a minimum of 200 fork lengths of juvenile fall chinook salmon were measured and recorded at each site. Other fish were identified to species (or lowest possible taxa), measured, and recorded. All fish were released back into the river. Water quality parameters including dissolved oxygen, percent oxygen saturation, and water temperature were recorded at mid-depth within the sample area.

Summer and Early Fall

Nearshore surveys were performed weekly at nine fixed index sites during the summer and early fall sampling period. The nine index sites consisted of six main channel and three slough locations. The six main channel sites included three at Locke Island, two at 100 F Islands, and one at the Hanford Townsite. The three slough sites included one in White Bluffs Slough and two in Hanford Slough. Sampling locations were shallow ($\leq$ 1.0 m) with little or no water velocity and substrates were a mixture of sand, fine sediment, and gravel/cobble. Vegetation at the sites varied seasonally and with water level fluctuations, and consisted primarily of algal accumulations in shoreline margins and aquatic macrophytes, predominately *Myriophyllum* spp.

Index sites were sampled using a small beach seine (15.2 m x 1.2 m with a 1.2 m^2 bag with 0.8 mm mesh), following the methods of Barfoot et al. (1999). To collect fish, the seine was pulled perpendicular to the shore from a depth of one meter or from a distance of 15 meters from the shore whichever was attained first.

Fish collected that were large enough to be identified in the field were anesthetized with MS-222, measured, recorded, and released. Larval fish and smaller juvenile fish collected were preserved in 10% buffered formalin and transported to the USGS/BRD laboratory for analysis. In the situation where large numbers of larval fish were collected volumetric sub-sampling was performed and excess fish were released back into the river.

At the laboratory, preserved samples of larval and juvenile fish were sorted, identified to the lowest possible taxa, and counted. Larval fish numbers by taxa were estimated using simple extrapolation of the sub-samples. A maximum of 50 specimens of each taxa were randomly selected and measured to the nearest 0.1 mm standard length.

Water quality parameters were collected at all index sites during each sample event. Water temperatures were recorded at mid-depth offshore at the most distant point of the seine haul and at a 20 cm depth near shore. Dissolved oxygen concentrations were measured at the most distant point of the seine haul.

Coarse Substrate Surveys

WDFW and GCPUD personnel performed coarse substrate surveys during the fall chinook salmon emergence and rearing period. Data was collected to evaluate possible relationships between substrates and the stranding of juvenile fall chinook salmon and other fish species.

Although entrapment areas were the main focus of the early spring and summer sampling activities, some effort was extended to assess the stranding of juvenile fall chinook salmon and other fish species in the interstitial spaces of coarse (16 mm diameter or greater) substrate within the 24-hour fluctuation zone and outside of obvious depressions. Random 4 m^2 plots were selected in areas of coarse substrate within the fluctuation zone. These plots were investigated for stranded fish in the interstitial spaces. Substrate was moved by hand or rakes. Parameters recorded for each plot included substrate composition according to the modified Wentworth code

(Platts et al. 1983), substrate embeddedness according to Platts et al. (1983), vegetation density (absent, sparse, medium, and dense), distance to river, and number of juvenile fall chinook salmon and other fish species found in the plot. Fork length measurements were recorded for all fish found.

Scanning Hydrographic Operational Airborne Lidar Survey (SHOALS)

The SHOALS survey was performed to provide detailed bathymetry of the Hanford Reach and to aid in development of a juvenile fall chinook salmon susceptibility model. The United States Army Corps of Engineers (COE) collected detailed bathymetry data for the Hanford Reach using the SHOALS system. The SHOALS system consisted of an airborne laser transmitter/receiver capable of measuring 200 soundings per second. The system was operated from a Bell 212 helicopter, flying at altitudes between 200 and 400 m with ground speed of 60 knots. The SHOALS system also included a ground-base data processing system for calculating accurate horizontal position and water depth. Depending on the complex interaction of radiance of bottom material, incident sun angle and intensity, and type and quantity of organics or sediments in the water column, the SHOALS system was capable of sensing bottom to depths equal to two or three times the Secchi depth (measure of transparency of the water). The horizontal positional accuracy of the data was +/-3 m and the vertical positional accuracy was +/-15 cm.

Benthic Macroinvertebrate Surveys

A two-year evaluation to assess the effects of water fluctuations on the benthic community of the Hanford Reach was initiated by the University of Idaho (U of I) and Streamside Programs Consultation (SPC) in 1998. A preliminary survey was conducted on artificial substrates in 1998 to determine sufficient sample size, colonization period, substrate type, and sample locations for evaluating effects of diel (24-hour) water level fluctuations on benthic macroinvertebrates of the Hanford Reach. Detailed methods are included in **Appendix B**.

Temperature Surveys and Thermal Stress Investigations

WDFW performed water temperature surveys to provide a comparison between river and entrapment temperatures and to assist USGS/BRD in developing laboratory temperature tolerance experiments. USGS/BRD performed laboratory temperature tolerance experiments to determine effects of thermal stressors on performance of Hanford Reach juvenile fall chinook salmon.

Temperature data was collected with Optic Stowaway dataloggers in selected entrapments and nearshore sites from April 22 through June 29. A total of 11 units were placed at four selected entrapment locations including eight at 100 F Islands, one at White Bluffs Slough, one at Wooded Island, and one at Locke Island. Two additional units were employed at 100 F Islands to record nearshore river temperatures. Dataloggers were set to record temperature every five minutes and were downloaded every one to four weeks. Monitored entrapments were selected to represent the range of filling, warming, and draining scenarios observed by the field crews.

The USGS/BRD assessed effects of thermal stressors on performance of Hanford Reach juvenile fall chinook salmon by exposing fish to heat stress scenarios similar to monitored entrapments on the Hanford Reach. Specific objectives of the laboratory work were to assess the extent of direct mortality, vulnerability to predation, and physiological effects attributable to various sublethal heat stress scenarios. Detailed methods are provided in **Appendix C**.

Entrapment Modeling

WDFW used MASS1 to perform entrapment modeling. PNNL created MASS1 prior to this study and modified the hydraulic module for use on this project. The entrapment modeling was used to determine water elevation histories of entrapments encountered in 1998. The entrapment histories were then used to determine the details of juvenile fall chinook salmon and other fish species vulnerability to stranding and entrapment due to river elevation fluctuations.

River elevations/discharges at specific time intervals for fixed cross-sectional transects were computed using 1998 hourly discharge data from Priest Rapids Dam and MASS1. MASS1 uses cross-sectional transects that are spaced approximately 0.4 km apart throughout the Hanford Reach. Entrapment histories were back-calculated using the entrapment flooding elevation, survey time, and elevation/discharge data from the nearest cross-sectional transect. For clarification, an illustration of the typical components of an entrapment history is provided in **Figure 7**. Similarly, entrapment discharge histories can also be illustrated.

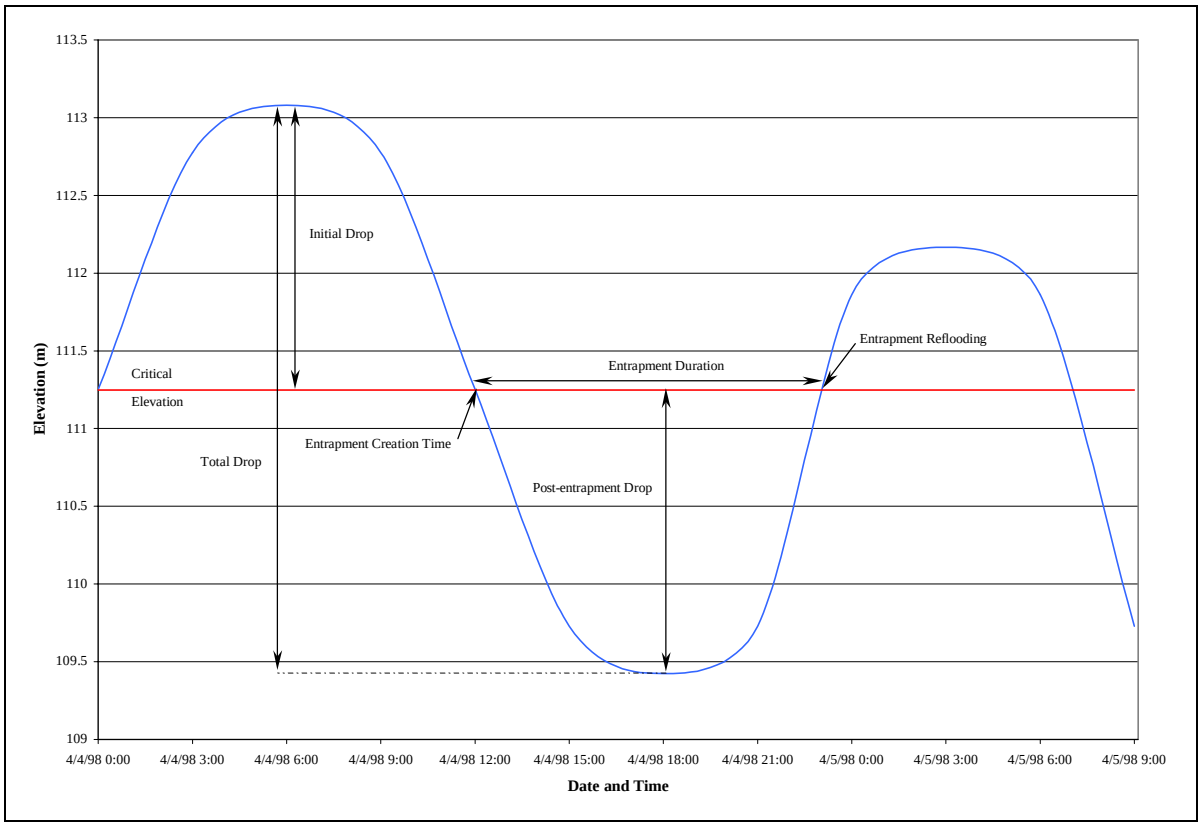


Figure 7. Typical components of an entrapment history.

Component Definitions
Critical Elevation/Discharge - the specific river elevation/discharge at which entrapment isolation occurs.
Entrapment Creation Time - the specific time an entrapment becomes isolated.
Initial Drop - the river elevation/discharge reduction that creates an entrapment.
Post-entrapment Drop – the elevation/discharge reduction that occurs after entrapment isolation.
Total Drop – the overall river elevation/discharge reduction (Initial Drop + Post-entrapment Drop).
Entrapment Duration - the duration of time that occurs from entrapment isolation to re-inundation.

Components of entrapment histories were used to determine various aspects of juvenile fall chinook salmon and other fish species vulnerability to stranding and entrapment. Critical elevation/discharge data were employed to establish vulnerability of fish to entrapment at different river stages. Entrapment creation times were examined to determine the most vulnerable time of day for fish to be stranded and entrapped. Initial drop data were used to calculate the minimum elevation/discharge reduction necessary to entrap fish. Post-entrapment drop data were used to evaluate fish mortality in relation to the magnitude of elevation/discharge reduction. Entrapment duration data were used to assess fish mortality in relation to the duration of entrapment isolation. Time of day data were analyzed using circular or directional statistics (Fisher 1993). Chi-square analysis was used to determine if entrapments with juvenile fall chinook salmon occurred in flow bands in proportion to the number of times the water fluctuated within those bands. The expected number of entrapments with juvenile fall chinook salmon was based on the number of observed flow fluctuations at transect 81 (Rkm 366.9), the middle

cross-section on the Hanford Reach. Chi-square analysis was also used to ascertain if the number of juvenile fall chinook salmon found entrapped occurred in flow bands in proportion to the number of times the water fluctuated within those bands. Bonferroni confidence intervals were calculated for each 40 kcfs flow band to determine the propensity or avoidance of juvenile fall chinook salmon to entrapment in each flow band (Neu et al. 1974; Byers et al. 1984).

Results

This section of the document provides a description of the 1998 river conditions and a compilation of the results of the 1998 evaluation. Although information on the relationships of juvenile fall chinook salmon and other fish species to the habitat parameters of substrate composition, substrate embeddedness, and vegetation are presented, total availability of each parameter on the Hanford Reach was unknown and, therefore, preference or avoidance of each habitat parameter by juvenile fall chinook salmon and other fish species could not be statistically assessed.

1998 Hanford Reach Conditions

River conditions on the Hanford Reach in 1998 were marked by below average river flows, above normal ambient air temperatures, near normal precipitation, and near average solar radiation levels (**Table 1**).

Table 1. Comparison of 1998 monthly average river flows, air temperatures, precipitation, and solar radiation levels to past years on the Hanford Reach of the Columbia River.

River Flows (kcfs)[1]												
	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.
1998	116.8	141.3	119.0	84.1	171.3	172.0	131.8	109.6	80.8	72.2	82.2	103.9
Mean (1988-1997)	124.3	125.0	115.7	119.6	162.7	176.3	129.3	103.9	80.9	85.8	100.3	121.2
Departure	-7.5	+16.3	+3.3	-35.5	+8.6	-4.3	+2.5	+5.7	-0.1	-13.6	-18.1	-17.3
Air Temperature (^oC)[2]												
	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.
1998	2.3	5.7	9.1	12.4	16.9	21.7	27.8	25.5	21.7	11.3	7.6	0.6
Normal (1961-1990)	-0.4	3.3	7.6	11.5	16.3	20.9	24.6	23.9	18.7	11.6	4.6	-0.3
Departure	+2.7	+2.3	+1.6	+0.9	+0.6	+0.7	+3.2	+1.6	+2.9	-0.3	+3.0	+0.9
Precipitation (cm)[2]												
	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.
1998	3.1	2.9	1.3	0.2	1.3	1.2	0.9	0.1	0.3	0.7	3.3	1.1
Normal (1961-1990)	2.0	1.6	1.2	1.0	1.3	1.0	0.5	0.7	0.8	1.0	2.3	2.6
Departure	+1.1	+1.3	+0.1	-0.9	0.0	+0.3	+0.4	-0.6	-0.5	-0.3	+1.0	-1.5
Solar Radiation (Langleys)[2]												
	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.
1998	91.0	160.4	303.0	445.0	452.9	607.8	590.8	556.6	394.3	255.6	109.4	87.4
Mean (1980-1997)	96.8	170.9	300.2	423.4	520.1	573.6	598.7	521.8	387.6	241.2	116.9	75.9
Departure	-5.7	-10.6	+2.9	+21.6	-67.2	+34.2	-7.9	+34.8	+6.7	+14.4	-7.5	+11.5

[1]Data from USGS Gauging Station 12472880 below Priest Rapids Dam

[2] Data from Hanford Meteorological Station, PNNL

Comparison of 1998 flows to the 10-year mean flows (1988-1997) indicates that 1998 was a low flow year. Low flows in 1998 were associated with the peak and waning of El Nino, which brought below normal snow pack and rapid April snowmelt (Columbia River Water Management Report Water Year 1998). Flows during the fall chinook salmon emergence and rearing period were lowest in April before the spring freshet.

Air temperatures in 1998 were above normal for all months except October. July had the largest departure with air temperatures at 3.2^{o}C above normal. During the fall chinook salmon emergence and rearing period, air temperature departures above normal ranged from 1.6^{o}C in March to 0.6^{o}C in May.

Precipitation during the 1998 fall chinook salmon emergence and rearing period was near normal, with the exception of April, in which precipitation was 0.9 cm below normal. Precipitation for the summer and early fall portion of the season was below normal with the exception of July, which was 0.4 cm above normal.

Comparison of 1998 mean monthly solar radiations levels to an 18-year mean (1980-1997) indicates that 1998 was near average. Exceptions include April, June and August, which were above average, and May, which was below average. Solar radiation levels provide a good indication of cloud cover. Cloud cover is the predominant atmospheric condition that reduces the amount of solar radiation reaching the surface of earth.

Stranding and Entrapment, Coarse Substrate, and Nearshore Surveys

In 1998, 50 strandings, 151 entrapments, 159 nearshore, and 28 coarse substrate sites were investigated on the Hanford Reach. The spring and early summer sampling period occurred from March 16 through June 28. The summer and early fall sampling period extended from June 29 through October 5. Coarse substrate surveys were conducted from March 22 through June 15.

Spring and Early Summer

During the spring and early summer, 44 strandings, 122 entrapments, 49 nearshore, and 28 coarse substrate sites were examined on the Hanford Reach. A total of 45,934 fish encompassing 12 genera and at least 13 species were collected at these sites (**Table 2**). Juvenile fall chinook salmon comprised the majority (86.1%) of fish sampled during this time period, followed by northern pikeminnow (5.3%), redside shiner (5.3%), peamouth (1.5%), and sucker (1.4%). An assortment of other species were found in low numbers including threespine stickleback, sculpin, smallmouth bass, mountain whitefish (*Prosopium williamsoni*), dace (*Rhinichthys* spp.), common carp, lamprey (*Lampetra* spp.), and coho salmon (*Oncorhynchus kisutch*). Juvenile fall chinook salmon also made up the bulk (87.3%) of fish observed stranded and entrapped during this time period. Other species found stranded and entrapped in sizeable numbers included northern pikeminnow (6.1%), redside shiner (3.3%), sucker (1.7%), and peamouth (1.2%). All species collected except coho salmon (only 1 fish sampled) were observed stranded or entrapped.

Table 2. Total number of fish sampled during the spring and early summer sampling period (March 16 – June 28) on the Hanford Reach of the Columbia River in 1998.

Common Name	Scientific Name	Nearshore	Stranded[1]	Entrapped[2]	Total Fish
Chinook Salmon	*Oncorhynchus tshawytscha*	8,075	7,868	23,627 (2,943)	39,570
Coho Salmon	*Oncorhynchus kisutch*	1			1
Common Carp	*Cyprinus carpio*			7	7
Dace	*Rhinichthys* spp.	5	1	5	11
Mountain Whitefish	*Prosopium williamsoni*	6		6 (4)	12
Northern Pikeminnow	*Ptychocheilus oregonensis*	222	6	2,190 (1)	2,418
Lamprey	*Lampetra* spp.		4		4
Peamouth	*Mylocheilus caurinus*	281	46	369	696
Redside Shiner	*Richardsonius balteatus*	1,238	13	1,184 (5)	2,435
Sculpin	*Cottus* spp.	5	20	22 (1)	47
Smallmouth Bass	*Micropterus dolomieui*		1	22	23
Sucker	*Catostomus* spp.	31	7	596 (3)	634
Threespine Stickleback	*Gasterosteus aculeatus*	2	23	51 (1)	76
Total		9,866	7,989	28,079 (2,958)	45,934

[1] All stranded fish were counted as mortalities.

[2] Numbers in ( ) represent thermal mortalities.

Juvenile Fall Chinook Salmon

A total of 39,570 juvenile fall chinook salmon were sampled on the Hanford Reach in 1998. This total consisted of 8,075 individuals collected in nearshore sites, 7,868 in stranding sites, and 23,627 in entrapment

sites. Thirty-four percent (10,811) of the 31,495 stranded and entrapped juvenile fall chinook salmon were recorded as direct mortalities. Direct mortalities included 7,868 stranded and 2,943 thermal induced fatalities.

Seasonal Trends

Estimated date of emergence of fall chinook salmon on the Hanford Reach was March 12 in 1998 (Carlson 1998). Sampling began on March 16 and the first juvenile fall chinook salmon were observed in both nearshore and entrapment sites on March 19. Peak numbers of individuals were observed between early April and early May (**Table 3**). Peak numbers of mortalities also were found during this time period. End of emergence of fall chinook salmon was calculated at May 4 (Carlson 1998). Juvenile fall chinook salmon were last encountered in entrapments on June 24 and last sampled in nearshore sites on June 27.

Table 3. Weekly sampling results of juvenile fall chinook salmon on the Hanford Reach of the Columbia River in 1998.

Week	Nearshore	Stranded[1]	Entrapped[2]	Total Mortalities (Stranded + Thermal)	Total Stranded/Entrapped	Total Chinook
March 16-22	41	48	30	48	78	119
March 23-29	30	28	85	28	113	143
March 30-April 5	72	17	62	17	79	151
April 6-12	1,041	11	2,686	11	2,697	3,738
April 13-19	940	5,180	952 (54)	5,234	6,132	7,072
April 20-26	992	666	6,374 (1,775)	2,441	7,040	8,032
April 27-May 3	434	1,781	11,258 (1,042)	2,823	13,039	13,473
May 4-10	1,118	54	266 (27)	81	320	1,438
May 11-17	647	0	645	0	645	1,292
May 18-24	522	59	265	59	324	846
May 25-31	660	21	549 (5)	26	570	1,230
June 1-7	86	0	306	0	306	392
June 8-14	499	0	6	0	6	505
June 15-21	374	3	138 (40)	43	141	515
June 22-28	619	0	5	0	5	624
Total	8,075	7,868	23,627 (2,943)	10,811	31,495	39,570

[1] All stranded fish were counted as mortalities.

[2] Numbers in ( ) represent thermal mortalities.

Distribution

Juvenile fall chinook salmon were observed stranded and entrapped throughout the 82 km of the Hanford Reach, however, fish were found most often at the island complex areas of Locke Island/White Bluff Slough, 100 F Islands, and Wooded Island (**Figure 8**). These island complex areas appear to provide excellent rearing habitat as well as high stranding and entrapment potential.

Size Susceptibility

Mean fork length of juvenile fall chinook salmon was approximately 41 mm in samples collected nearshore and in entrapments for the week ending March 22. Mean fork length remained similar for both nearshore and entrapments through May 24 with fish increasing in size to 45 mm. After May 24, mean fork lengths increased considerably and leveled at approximately 59 mm in entrapments but continued to increase in nearshore areas before leveling at 68 mm (**Figure 9).**

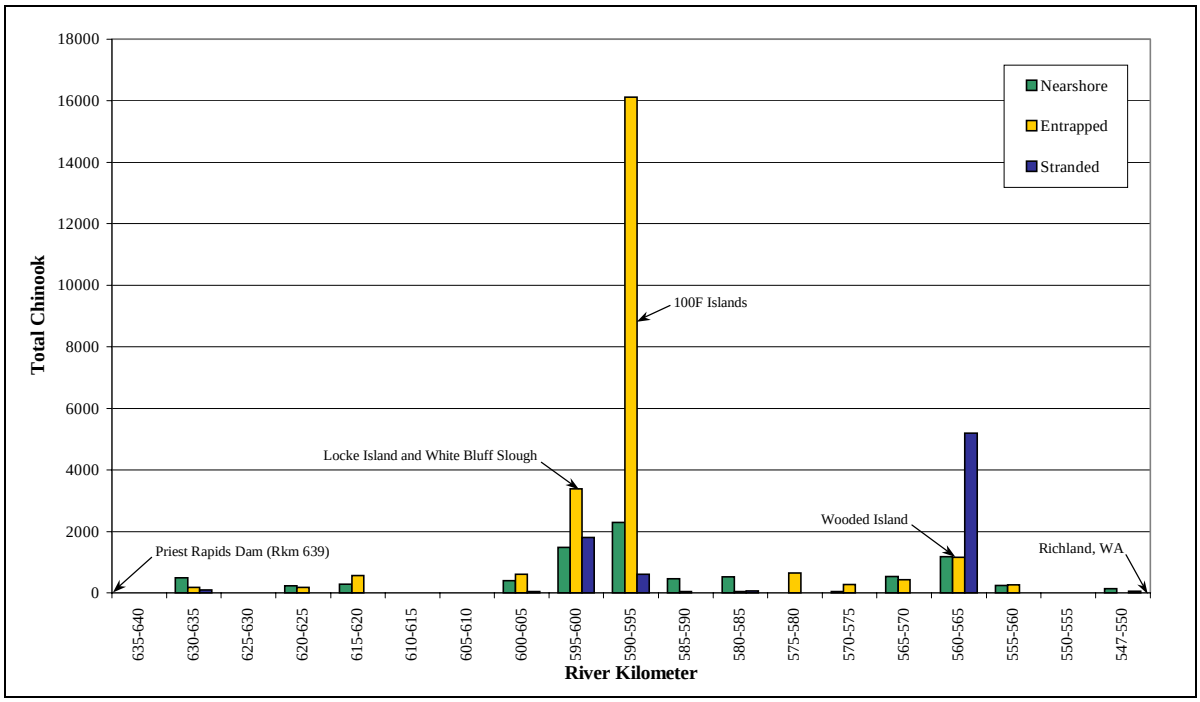


Figure 8. Distribution of juvenile fall chinook salmon observed on the Hanford Reach of the Columbia River in 1998.

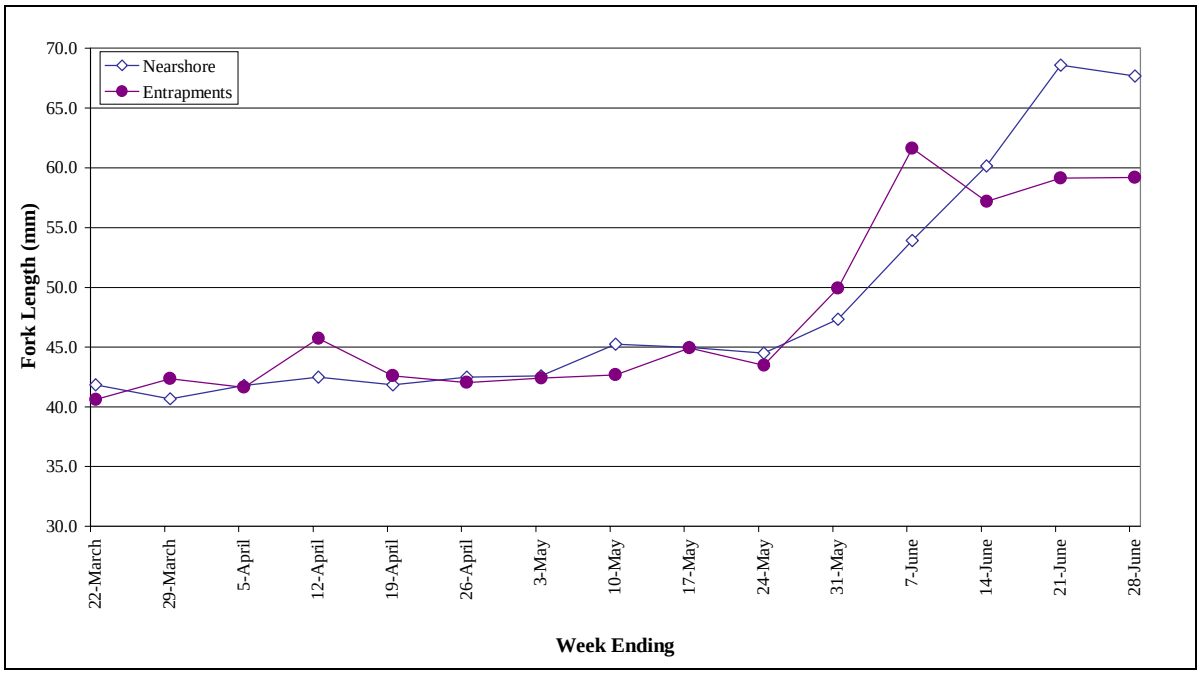


Figure 9. Mean fork length of juvenile fall chinook salmon found in nearshore and entrapment sites on the Hanford Reach of the Columbia River in 1998.

Maximum and minimum fork lengths for each week were plotted and compared for stranding, entrapment, and nearshore samples (**Figure 10**). Although the end of emergence for fall chinook salmon on the Hanford Reach was calculated at May 4 (Carlson1998), minimum fork lengths for all samples remained at or below 39 mm through June 7. Typical fork length of emerging fall chinook salmon on the Hanford Reach ranges from 35 to 40 mm.

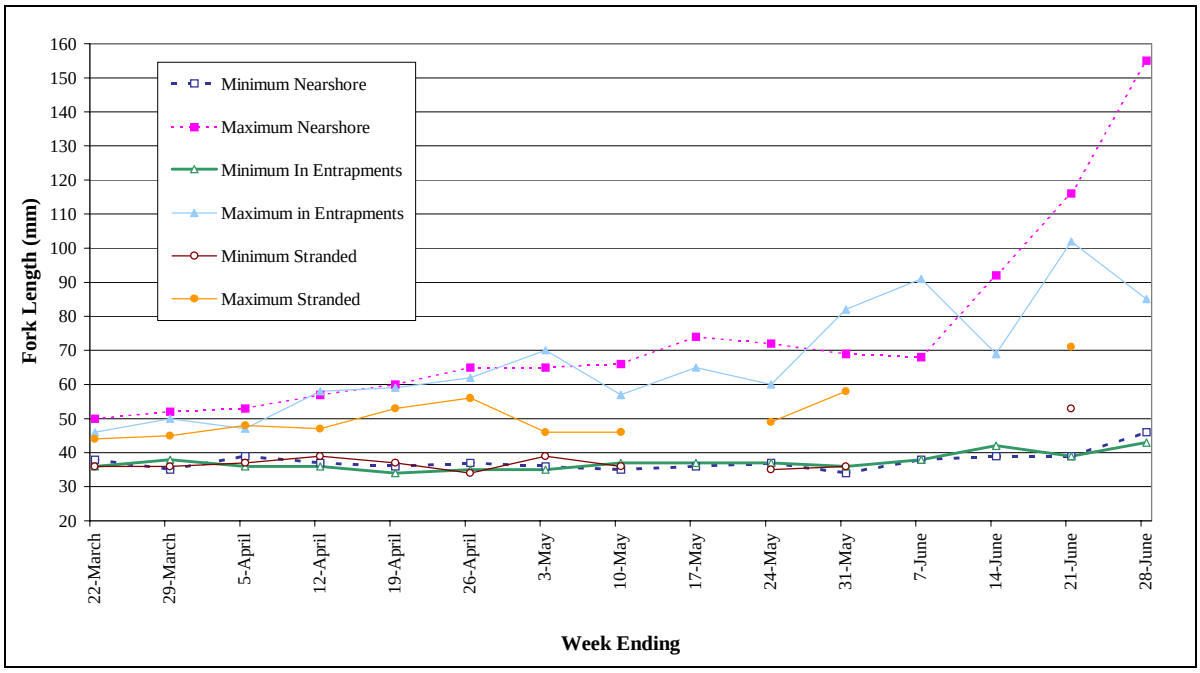


Figure 10. Maximum and minimum lengths of juvenile fall chinook salmon collected in stranding, entrapment, and nearshore sites on the Hanford Reach of the Columbia River in 1998.

Thermal Mortality, Maximum Air Temperatures, and Solar Radiation

Thermal mortality of juvenile fall chinook salmon occurs when water temperatures exceed 25.1°C, the upper lethal limit (Brett 1952; Templeton and Coutant 1971). Water temperature in entrapments on the Hanford Reach can attain lethal levels due to high ambient air temperatures and/or solar radiation. Thermal mortality of entrapped juvenile fall chinook salmon on the Hanford Reach was observed on 11 days in 1998 from mid-April through mid-June (**Table 4**). Thermal mortality events took place on four occasions when daily maximum air temperatures were below 25.1°C. This type of event occurs when solar radiation heats substrate and entrapment water above ambient air temperature.

Table 4. Juvenile fall chinook salmon thermal mortalities observed on the Hanford Reach of the Columbia River in 1998 and daily maximum air temperatures and solar radiation levels.

Date	Chinook Thermal Mortalities	Daily Maximum Air Temperature (°C)[1]	Daily Solar Radiation (Langleys)[1]
17-April	54	20.0	556
21-April	489	27.2	559
26-April	1286	21.7	564
29-April	106	30.6	564
1-May	934	33.9	577
2-May	2	31.7	484
4-May	14	30.6	607
10-May	13	24.4	511
28-May	5	25.0	683
15-June	1	26.1	672
16-June	39	27.2	574
Total	2943		

[1] Data from Hanford Meteorological Station, PNNL

Substrate Size

Juvenile fall chinook salmon were observed stranded and entrapped over all dominant substrate sizes sampled except very coarse sand (code 2) (**Figure 11**). The highest mean number of juvenile fall chinook salmon per site was observed over dominant substrates of coarse gravels to small pebbles (codes 5 and 6).

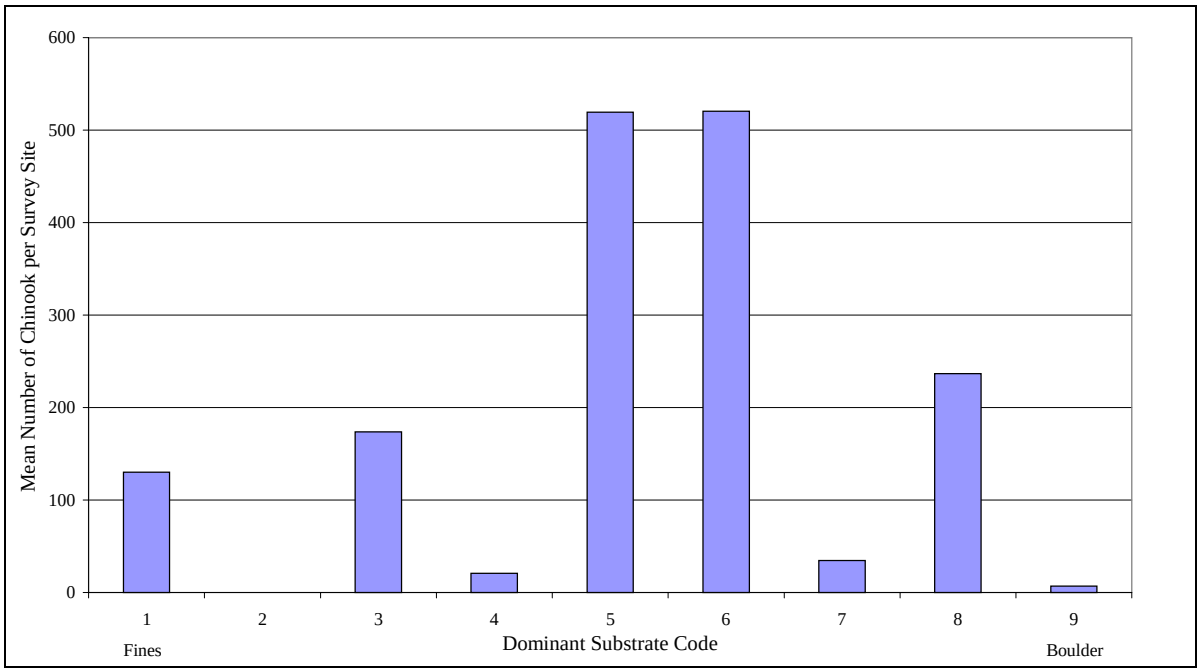


Figure 11. Dominant substrates and mean number of juvenile fall chinook salmon found in stranding and entrapment sites on the Hanford Reach of the Columbia River in 1998.

Mortalities were observed over all dominant substrate sizes except for coarse sand (code 2) and fine gravel (code 3) (**Figure 12**). Mean number of juvenile fall chinook salmon stranding mortalities per survey site was higher over the larger dominant substrates of small pebbles (code 6) to cobble (code 8) and medium gravel (code 4). The highest mean number of juvenile fall chinook salmon thermal mortalities per survey site was observed over fine dominant substrates (code 1).

Substrate Embeddedness

Juvenile fall chinook salmon were observed stranded and entrapped over substrates with all degrees of embeddedness (**Figure 13**). Mean number of stranding mortalities was highest over the least embedded substrate (code 1) in which drainage rates were expected to be highest while thermal mortalities were the highest over the most embedded substrate (code 4) in which drainage rates were expected to be lowest, providing time for heating (**Figure 14**).

Coarse Substrate Surveys

A total of 172 coarse substrate sites encompassing 688 m^2 were investigated on the Hanford Reach during the fall chinook salmon emergence and rearing period. Six juvenile fall chinook salmon and one dace (*Rhinichthys* spp.) were observed. Results indicated that 87.2 juvenile fall chinook salmon and 14.5 dace were stranded per hectare of coarse substrate.

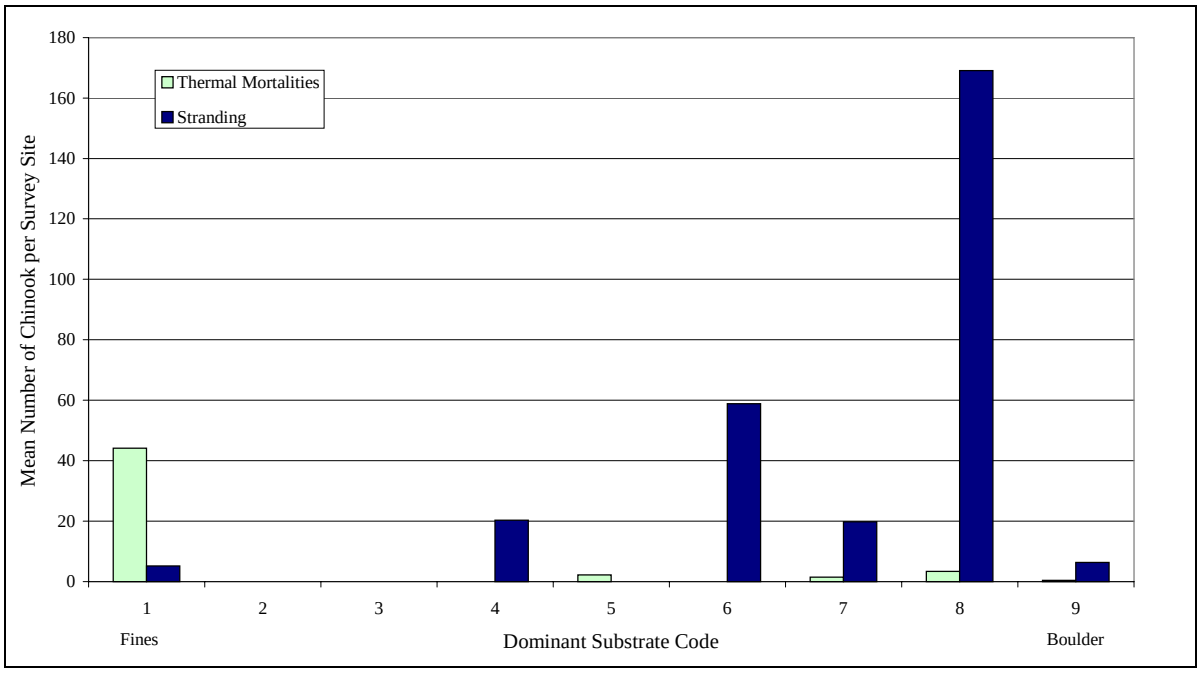


Figure 12. Dominant substrates and mean number of juvenile fall chinook salmon mortalities found in stranding and entrapment sites on the Hanford Reach of the Columbia River in 1998.

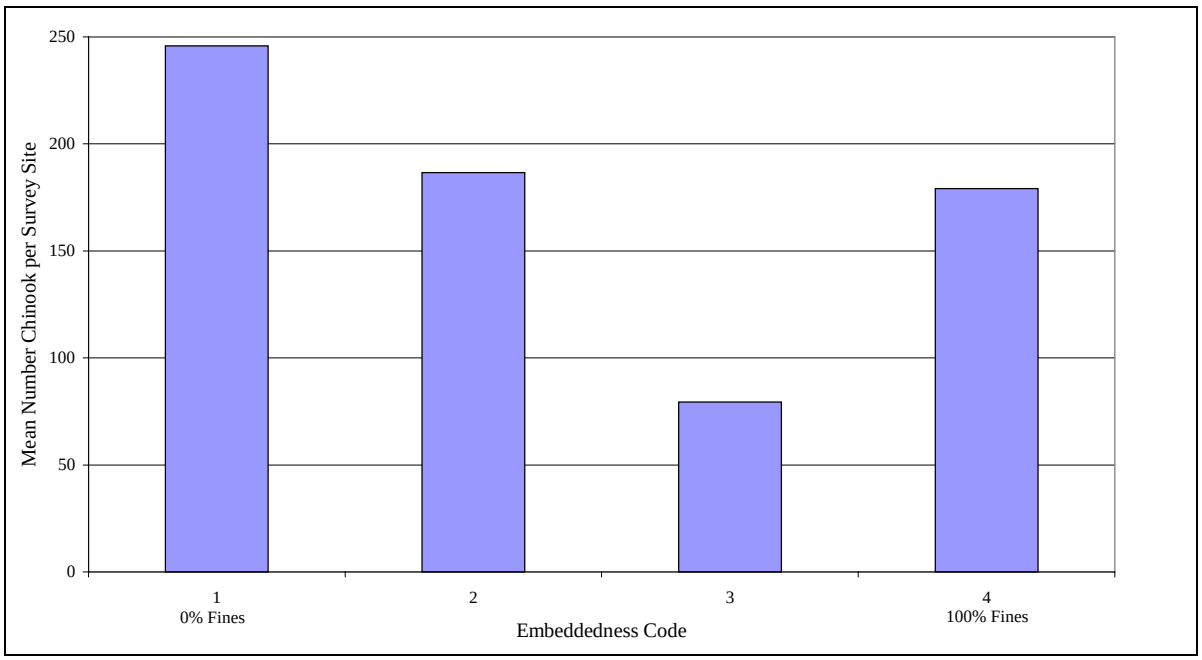


Figure 13. Substrate embeddedness and mean number of juvenile fall chinook salmon found in stranding and entrapment sites on the Hanford Reach of the Columbia River in 1998.

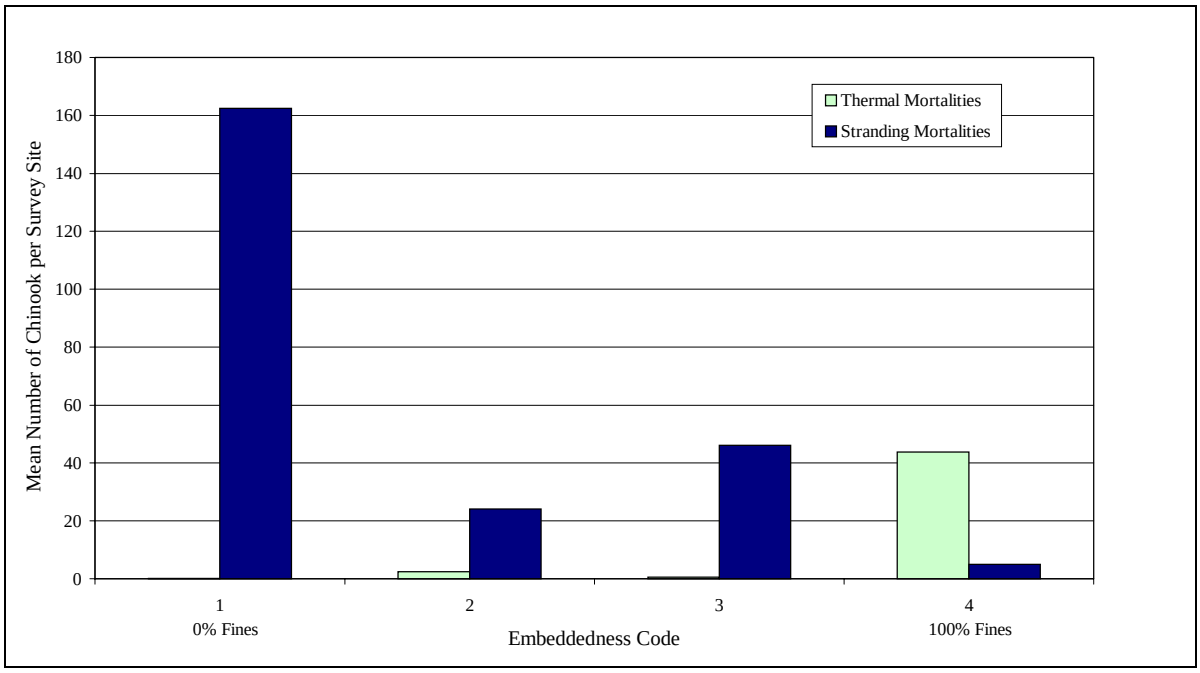


Figure 14. Substrate embeddedness and mean number of juvenile fall chinook salmon mortalities found in stranding and entrapment sites on the Hanford Reach of the Columbia River in 1998.

Vegetation

Juvenile fall chinook salmon were observed stranded and entrapped in all vegetation densities (**Figure 15**). Mean number of juvenile fall chinook salmon sampled per site was highest when vegetation was absent and lowest when vegetation was dense. The highest mean number of stranding and thermal mortalities occurred where vegetation was absent and decreased with increasing vegetation (**Figure 16**). Thermal mortalities were not observed in entrapments with dense vegetation.

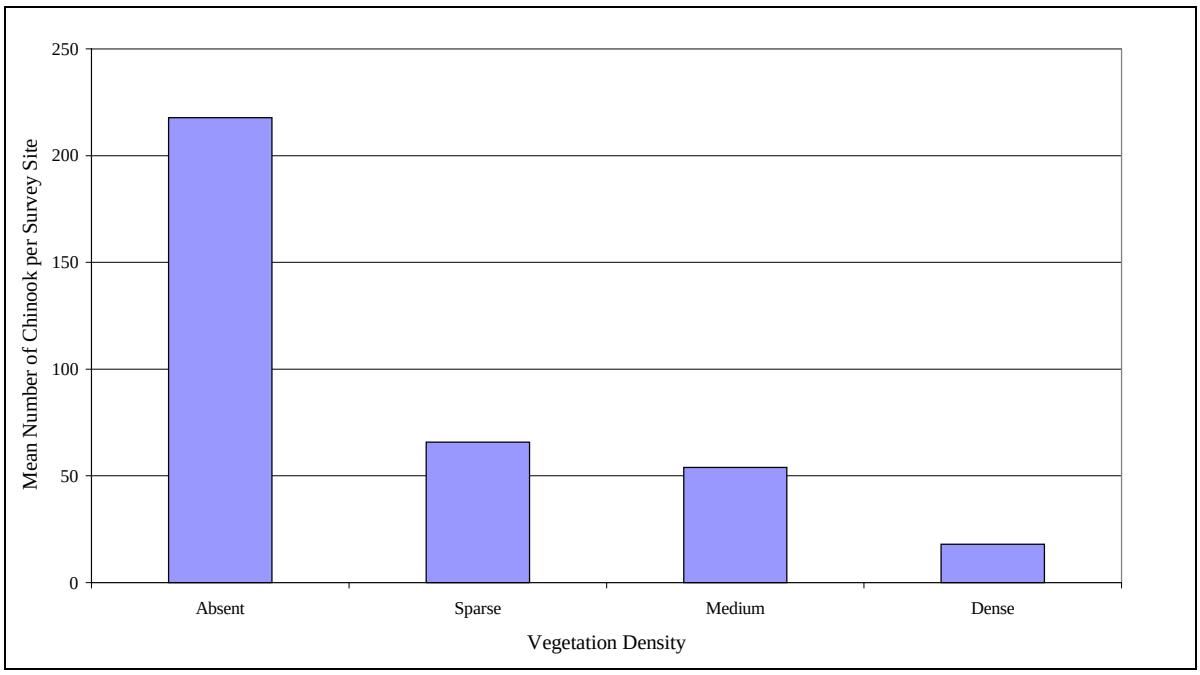


Figure 15. Vegetation density and mean number of juvenile fall chinook salmon found in stranding and entrapment sites on the Hanford Reach of the Columbia River in 1998.

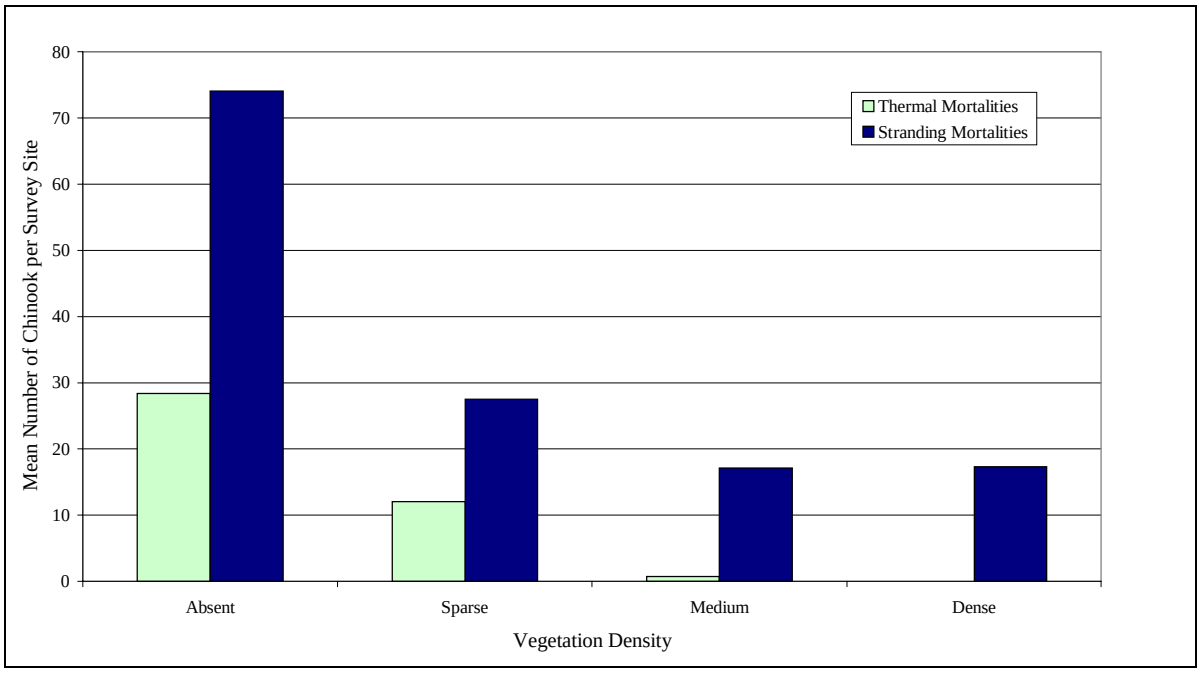


Figure 16. Vegetation density and mean number of juvenile fall chinook salmon mortalities found in stranding and entrapment sites on the Hanford Reach of the Columbia River in 1998.

Entrapment Creation Times

Entrapment creation times were modeled for 106 entrapments containing juvenile fall chinook salmon. Rayleigh test of uniformity revealed that entrapments did not occur randomly throughout the day (p=0.00). Mean entrapment creation time was at 0338 hours with a 95% confidence interval from 0206 to 0510 hours (**Figure 17**). The mean resultant length (r=0.33) and the von Mises concentration parameter (k=0.69) were small, however, the distribution of entrapment creation times for entrapments holding juvenile fall chinook salmon appeared to be bimodal with most entrapments occurring in the early morning and, to a lesser extent, in the late afternoon/early evening.

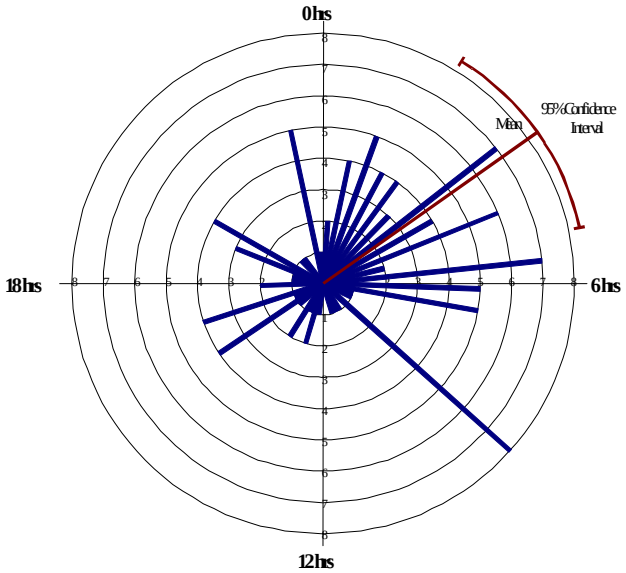


Figure 17. Entrapment creation times for 106 entrapments found on the Hanford Reach of the Columbia River in 1998 containing juvenile fall chinook salmon displayed on a 24-hour clock.

Entrapment times for 31,262 juvenile fall chinook salmon were also analyzed to determine the time of day when most individuals became entrapped. A Rayleigh test of uniformity showed individuals were not randomly entrapped throughout the day (p=0.00). Mean entrapment time was at 0359 hours with a narrow 95% confidence interval from 0356 to 0402 hours (**Figure 18**). The mean resultant length (r=0.54) and the von Mises concentration parameter (k=1.29) were again small but the distribution of entrapment times appeared to be bimodal with most individuals entrapped in the early morning and a smaller number of individuals entrapped in the late afternoon/early evening. Entrapments with larger numbers of entrapped juvenile fall chinook salmon seemed to occur most frequently at night, followed by dawn and dusk, with fewest occurring during daylight.

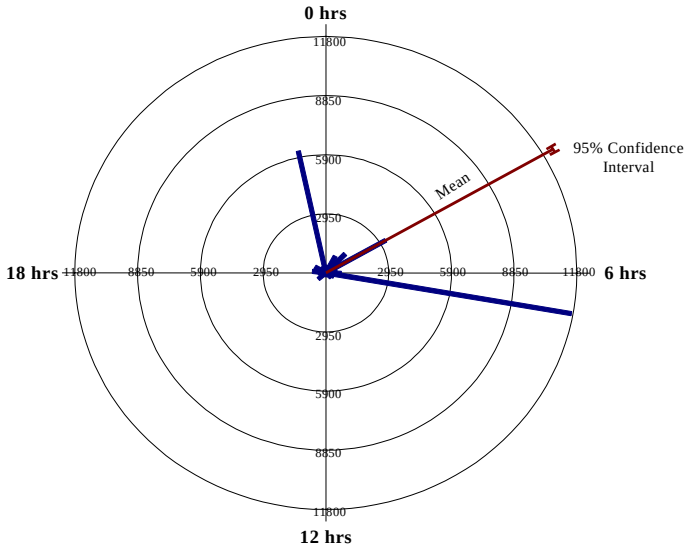


Figure 18. Entrapment time of juvenile fall chinook salmon found on the Hanford Reach of the Columbia River in 1998 displayed on a 24-hour clock.

Although data showed a pattern for the time of day in which juvenile fall chinook salmon were entrapped, no pattern of flow fluctuations at Priest Rapids Dam during the fall chinook salmon emergence and rearing period was observed. From March 24 to June 24, a total of 185 flow drops, 10 kcfs or greater, were recorded at Priest Rapids Dam. Start times of these flow drops were examined to determine if flow fluctuations occurred uniformly throughout the day (**Figure 19**). A Rayleigh test of uniformity showed fluctuations at Priest Rapids Dam were randomly spaced throughout the day (p=0.51). The mean resultant length (r=0.06) and the von Mises concentration parameter (k=0.12) also suggested uniformity.

Initial Drops in River Elevation

Juvenile fall chinook salmon were stranded and entrapped by a range of river elevation reductions (**Table 5**). However, most stranding and entrapment of individuals (94.2%) was the consequence of initial drops in river elevation of less than 0.9 m. Juvenile fall chinook salmon were found to be stranded/entrapped by the smallest flow reductions measurable.

Downramp Rates

Juvenile fall chinook salmon were stranded and entrapped on the Hanford Reach by downramp rates as great as 0.86 m/hr but rates of 0.15 m/hr or less captured 95.6% of the fish (**Table 6)**.

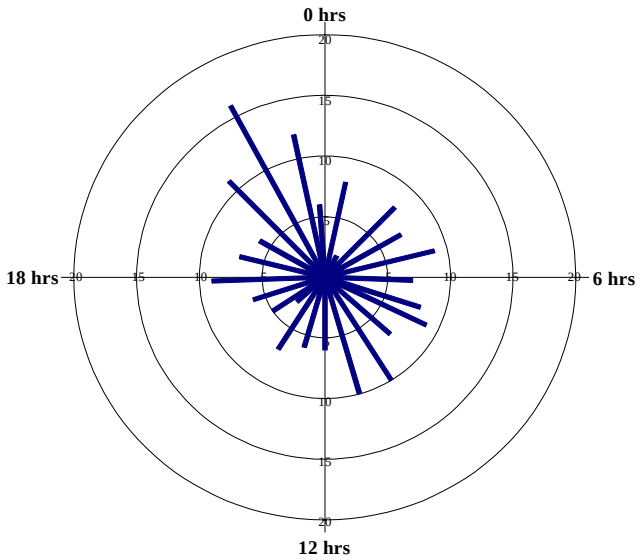


Figure 19. Start times of flow drops, 10 kcfs or greater, at Priest Rapids Dam on the Columbia River from March 24 to June 24, 1998 displayed on a 24-hour clock.

Table 5. Initial drops in river elevation and juvenile fall chinook salmon stranding and entrapment on the Hanford Reach of the Columbia River in 1998.

Initial Drop		Number of	Stranded/Entrapped
(m)	(ft)	Sites	Chinook
0.0 to 0.3	0.0 to 1.0	44	14436 (46.2%)
>0.3 to 0.6	>1.0 to 2.0	22	3354 (10.7%)
>0.6 to 0.9	>2.0 to 3.0	18	11659 (37.3%)
>0.9 to 1.2	>3.0 to 4.0	13	299 (1.0%)
>1.2 to 1.5	>4.0 to 5.0	3	266 (0.9%)
>1.5 to 1.8	>5.0 to 6.0	0	0 (0.0%)
>1.8 to 2.1	>6.0 to 7.0	3	683 (2.2%)
>2.1 to 2.4	>7.0 to 8.0	2	2 (0.0%)
>2.4 to 2.7	>8.0 to 9.0	1	563 (1.8%)
Total		106	31262

Table 6. Downramp rates and juvenile fall chinook salmon stranding and entrapment on the Hanford Reach of the Columbia River in 1998.

Downramp Rate		Number of	Stranded/Entrapped
(m/hr)	(ft/hr)	Sites	Chinook
0.00 to 0.03	0.00 to 0.10	13	902 (2.9%)
>0.03 to 0.06	>0.10 to 0.20	31	16312 (52.2%)
>0.06 to 0.09	>0.20 to 0.30	27	2192 (7.0%)
>0.09 to 0.12	>0.30 to 0.40	13	3667 (11.7%)
>0.12 to 0.15	>0.40 to 0.50	11	6856 (21.9%)
>0.15 to 0.18	>0.50 to 0.60	5	437 (1.4%)
>0.18 to 0.21	>0.60 to 0.70	5	333 (1.1%)
>0.21	>0.70	1	563 (1.8%)
Total		106	31262

Entrapment Duration

Modeling of entrapment creation times showed most stranding and entrapment sites on the Hanford Reach were sampled early in their development (**Table 7**). The majority of stranding mortalities (94.4%) occurred within 24 hours of the entrapment creation time while most thermal mortalities (99.8%) took place within three days. Mortality rates of observed stranded and entrapped juvenile fall chinook salmon increased from 11.0% to 49.9% after 12 hours of isolation from the river. Modeling of entrapment duration revealed that 59.7% of stranded and entrapped juvenile fall chinook salmon were located in entrapments isolated from the river for durations of 24 hours or more (**Table 8**).

Table 7. Entrapment creation time to sample time, number of stranded and entrapped juvenile fall chinook salmon, and mortalities on the Hanford Reach of the Columbia River in 1998.

Entrapment to Sample Time (Hours)	Number of Sites	Total Stranded/Entrapped Chinook	Chinook Stranding Mortalities	Chinook Thermal Mortalities	Total Chinook Mortalities
0-12	49	12,717	441	964	1,405
12-24	22	8,919	6,851	96	6,947
24-36	6	1,067	39	39	78
36-48	7	673	31	489	520
48-60	4	3,282	327	64	391
60-72	2	1,504	0	1,286	1,286
72-84	6	193	7	0	7
84-96	1	199	0	0	0
96-108	1	4	4	0	4
108-120	1	76	0	0	0
>120	7	2,628	21	5	26
Total	106	31,262	7,721	2,943	10,664

Table 8. Entrapment duration and juvenile fall chinook salmon stranding and entrapment on the Hanford Reach of the Columbia River in 1998.

Duration of Entrapment (Hours)	Number of Sites	Stranded/Entrapped Chinook
0-12	23	10,418 (33.3%)
12-24	18	2,198 (7.0%)
24-36	14	1,243 (4.0%)
36-48	14	6,954 (22.2%)
48-60	4	134 (0.4%)
60-72	4	1,322 (4.2%)
72-84	6	4,766 (15.2%)
84-96	3	404 (1.3%)
96-108	6	190 (0.6%)
108-120	1	76 (0.2%)
>120	13	3,557 (11.4%)
Total	106	31,262

Critical Flow Fluctuation Zones

MASS1 and hourly discharge data from Priest Rapids Dam were used to model flow fluctuations at transect 81 (Rkm 590.5), the middle transect between the upper and lowermost locations along the river where stranded and/or entrapped juvenile fall chinook salmon were found. The numbers of flow fluctuation reductions at transect 81 for each 40 kcfs band (60-100, 100-140, 140-180, 180-220, 220-260, 260-300) from March 24 to June 24 were counted. The expected number of entrapments and stranded and entrapped juvenile fall chinook salmon for each 40 kcfs flow band were calculated and compared to the observed (**Table 9**). Chi-square goodness-of-fit comparisons showed that the expected and observed number of entrapments and stranded and

entrapped juvenile fall chinook salmon differed significantly (entrapments, P<0.001, X^2=39.14, df=5; entrapped juvenile fall chinook salmon, P<0.001, X^2=30321.25, df=5). Bonferroni 95% confidence intervals suggest that entrapments were created more than expected within the 100-140 kcfs flow band and juvenile fall chinook salmon were stranded and entrapped more than expect within the 60-100 and 100-140 kcfs flow bands. Bonferroni 95% confidence interval also indicate that entrapments were formed less than expected within flow bands 180-220, 220-260, and 260-300 kcfs and juvenile fall chinook salmon were stranded and entrapped less frequently within flows bands 140-180, 180-220, 220-260, and 260-300 kcfs. These results are likely affected by the amount of area exposed at each flow band that could not be calculated at this time. The SHOALS data will provide this information in 1999.

Table 9. Critical flow fluctuation zones, entrapment creation, and juvenile fall chinook salmon stranding and entrapment on the Hanford Reach of Columbia River in 1998.

Entrapments

Flow Band (kcfs)	Flow Drops at Transect 81	Proportion of Total Flow Drops	Observed	Expected	Bonferroni 95% CI
60-100	14.50	0.160	16	16.95	0.060 ≤ P ≤ 0.241
100-140	22.11	0.244	50	25.84	0.345 ≤ P ≤ 0.598**
140-180	24.48	0.270	29	28.61	0.161 ≤ P ≤ 0.386
180-220	19.89	0.219	9	23.25	0.014 ≤ P ≤ 0.155*
220-260	8.42	0.093	2	9.83	-0.016 ≤ P ≤ 0.053*
260-300	1.31	0.014	0	1.53	0.000 ≤ P ≤ 0.000*
Totals	90.70	1.000	106	106.00	

Number of Stranded and Entrapped Juvenile Fall Chinook Salmon

Flow Band (kcfs)	Flow Drops at Transect 81	Proportion of Total Flow Drops	Observed	Expected	Bonferroni 95% CI
60-100	14.50	0.160	9,432	4,998.65	0.295 ≤ P ≤ 0.308**
100-140	22.11	0.244	18,059	7,620.76	0.570 ≤ P ≤ 0.585**
140-180	24.48	0.270	2,596	8,436.78	0.079 ≤ P ≤ 0.087*
180-220	19.89	0.219	1,148	6,855.58	0.034 ≤ P ≤ 0.039*
220-260	8.42	0.093	27	2,900.44	0.000 ≤ P ≤ 0.001*
260-300	1.31	0.014	0	449.80	0.000 ≤ P ≤ 0.000*
Totals	90.70	1.000	31,262	31,262.00	

*Observed less than expected.
**Observed more than expected.

Other Fish Species

At least twelve species of fish other than fall chinook salmon were collected in nearshore, stranding, and entrapment sites during the spring and early summer sampling period (**Table 2**). These species included at least two anadromous (coho salmon and lamprey) and ten resident species. A total of 6,364 individuals of these species were sampled, of which 4,573 were found stranded or entrapped. Thermal and stranding mortalities of these species during the spring and early summer were minimal with 136 individuals collected. Northern pikeminnow, redside shiner, peamouth, and sucker comprised the majority of individuals sampled and were the only species to provide enough information for analysis.

Substrate Size

The highest mean numbers of stranded and entrapped northern pikeminnow, redside shiner, peamouth, and sucker per site were observed in locations associated with dominant substrates of coarse gravel (code 5) and small pebble (code 6) (**Table 10**). These species also were observed in low numbers over dominant substrates of fines (code 1), large pebble (code 7), and cobble or rubble (code 8). Stranding and thermal mortalities were

minor with most stranding mortalities occurring over larger dominant substrates of small pebble (code 6) to boulder (code 9) and thermal mortalities taking place over dominant substrates of clay to coarse sand (code 1).

Table 10. Dominant substrates and number of northern pikeminnow, redside shiner, peamouth, and sucker observed during the spring and early summer sampling period (March 16 – June 28) on the Hanford Reach of the Columbia River in 1998.

Total Number of Fish

Substrate	Number	Stranded/Entrapped				Stranding Mortalities				Thermal Mortalities				Total Mortalities			
Code	of Sites	NPM	RSS	PM	SUC	NPM	RSS	PM	SUC	NPM	RSS	PM	SUC	NPM	RSS	PM	SUC
1	63	84	70	35	57					1	3		2	1	3		2
2	0																
3	2		1														
4	3																
5	5	296	33	14	71												
6	24	1,623	1,041	350	410	3	5	46	1				1	3	5	46	2
7	29	184	38	9	20	1	6		1		2			1	8		1
8	33	9	13	7	45	2	1		5					2	1		5
9	7		1				1								1		
Total	166	2,196	1,197	415	603	6	13	46	7	1	5	0	3	7	18	46	10

Mean Number of Fish per Site

Substrate	Number	Stranded/Entrapped				Stranding Mortalities				Thermal Mortalities				Total Mortalities			
Code	of Sites	NPM	RSS	PM	SUC	NPM	RSS	PM	SUC	NPM	RSS	PM	SUC	NPM	RSS	PM	SUC
1	63	1.33	1.11	0.56	0.90					0.02	0.05		0.03	0.02	0.05		0.03
2	0																
3	2		0.50														
4	3																
5	5	59.20	6.60	2.80	14.20												
6	24	67.63	43.38	14.58	17.08	0.13	0.21	1.92	0.04				0.04	0.13	0.21	1.92	0.08
7	29	6.34	1.31	0.31	0.69	0.03	0.21		0.03		0.07			0.03	0.28		0.03
8	33	0.27	0.39	0.21	1.36	0.06	0.03		0.15					0.06	0.03		0.15
9	7		0.14				0.14								0.14		

NPM - Northern Pikeminnow (*Ptychocheilus oregonensis*)
RSS - Redside Shiner (*Richardsoninus balteatus*)
PM - Peamouth (*Mylocheilus caurinus*)
SUC - Sucker Spp. (*Catostomus* spp.)

Substrate Embeddedness

Stranded and entrapped northern pikeminnow, redside shiner, peamouth, and sucker were observed over all degrees of substrate embeddedness (**Table 11**). The highest mean numbers of fish per site occurred over substrates with interstices of 50 to 75% fines (code 3). The least embedded substrates (code 1) had the highest overall observed mortality. Stranding and thermal mortalities were again minor with most stranding mortalities occurring over the least embedded substrates (code 1 and 2) and thermal mortalities taking place over the most embedded substrates (code 4).

Vegetation

The highest mean numbers of stranded and entrapped northern pikeminnow, redside shiner, peamouth, and sucker per site were observed in locations with sparse to absent vegetation (**Table 12**). Most stranding and thermal mortalities occurred over sites with absent vegetation. No stranding or entrapment took place over sites with dense vegetation.

Table 11. Substrate embeddedness and number of northern pikeminnow, redside shiner, peamouth, and sucker observed during the spring and early summer sampling period (March 16 – June 28) on the Hanford Reach of the Columbia River in 1998.

Total Number of Fish

Embeddedness	Number	Stranded/Entrapped				Stranding Mortalities				Thermal Mortalities				Total Mortalities			
Code	of Sites	NPM	RSS	PM	SUC	NPM	RSS	PM	SUC	NPM	RSS	PM	SUC	NPM	RSS	PM	SUC
1	36	950	674	69	60	5	7	46	6		2			5	9	46	6
2	34	382	60	14	72	1	5							1	5		
3	31	780	382	290	374		1		1				1		1		2
4	65	84	81	42	97					1	3		2	1	3		2
Total	166	2,196	1,197	415	603	6	13	46	7	1	5	0	3	7	18	46	10

Mean Number of Fish per Site

Embeddedness	Number	Stranded/Entrapped				Stranding Mortalities				Thermal Mortalities				Total Mortalities			
Code	of Sites	NPM	RSS	PM	SUC	NPM	RSS	PM	SUC	NPM	RSS	PM	SUC	NPM	RSS	PM	SUC
1	36	26.39	18.72	1.92	1.67	0.14	0.19	1.28	0.17		0.06			0.14	0.25	1.28	0.17
2	34	11.24	1.76	0.41	2.12	0.03	0.15							0.03	0.15		
3	31	25.16	12.32	9.35	12.06		0.03		0.03				0.03		0.03		0.06
4	65	1.29	1.25	0.65	1.49					0.02	0.05		0.03	0.02	0.05		0.03

NPM - Northern Pikeminnow (*Ptychocheilus oregonensis*)
RSS - Redside Shiner (*Richardsoninus balteatus*)
PM - Peamouth (*Mylocheilus caurinus*)
SUC - Sucker Spp. (*Catostomus* spp.)

Table 12. Vegetation and number of northern pikeminnow, redside shiner, peamouth, and sucker observed during the spring and early summer sampling period (March 16 – June 28) on the Hanford Reach of the Columbia River in 1998.

Total Number of Fish

Vegetation	Number	Stranded/Entrapped				Stranding Mortalities				Thermal Mortalities				Total Mortalities			
Code	of Sites	NPM	RSS	PM	SUC	NPM	RSS	PM	SUC	NPM	RSS	PM	SUC	NPM	RSS	PM	SUC
Absent	79	647	160	104	107	6	7	46	5	1	3		3	7	10	46	8
Sparse	60	1,489	1,032	310	445		6		1						6		1
Medium	24	60	5	1	51				1		2				2		1
Dense	3																
Total	166	2,196	1,197	415	603	6	13	46	7	1	5	0	3	7	18	46	10

Mean Number of Fish per Site

Vegetation	Number	Stranded/Entrapped				Stranding Mortalities				Thermal Mortalities				Total Mortalities			
Code	of Sites	NPM	RSS	PM	SUC	NPM	RSS	PM	SUC	NPM	RSS	PM	SUC	NPM	RSS	PM	SUC
Absent	79	8.19	2.03	1.32	1.35	0.08	0.09	0.58	0.06	0.01	0.04		0.04	0.09	0.13	0.58	0.10
Sparse	60	24.82	17.20	5.17	7.42		0.10		0.02						0.10		0.02
Medium	24	2.50	0.21	0.04	2.13				0.04		0.08				0.08		0.04
Dense	3																

NPM - Northern Pikeminnow (*Ptychocheilus oregonensis*)
RSS - Redside Shiner (*Richardsoninus balteatus*)
PM - Peamouth (*Mylocheilus caurinus*)
SUC - Sucker Spp. (*Catostomus* spp.)

Summer and Early Fall

A total of 145 sites were investigated throughout the Hanford Reach during the summer and early fall. These sites consisted of 6 strandings, 29 entrapments, and 110 nearshore sites. Fifteen genera with a minimum sixteen species were observed including American shad (*Alosa sapidissima*), bluegill (*Lepomis macrochirus*), bullhead (*Ameiurus* spp.), common carp, chiselmouth (*Acrocheilus alutaceus*), dace, largemouth bass, northern pikeminnow, lamprey, peamouth, redside shiner, sculpin, smallmouth bass, sucker, threespine stickleback, and

yellow perch (*Perca flavescens*) (**Table 13**). All species except American shad, bluegill, chiselmouth, and largemouth bass were represented in stranding or entrapment sites. All species sampled were affected at different lifestages.

Table 13. Total number of fish sampled during the summer and early fall sampling period (June 29 – October 5) on the Hanford Reach of the Columbia River in 1998.

			Nearshore			Entrapped			Thermal Mortalities			Stranding Mortalities		
Common Name	**Scientific Name**	**Dates Observed in Larval Form**	**L**	**J**	**A**	**L**	**J**	**A**	**L**	**J**	**A**	**L**	**J**	**A**
American Shad	*Alosa sapidissima*	July 22 - August 18	X	X										
Bluegill	*Lepomis macrochirus*	August 4	X	X										
Bullhead	*Ameiurus* spp.	August 30					X							
Chiselmouth	*Acrocheilus alutaceus*	July 21 - September 22	X	X										
Common Carp	*Cyprinus carpio*	July 15 - August 2	X	X				X				X		
Dace	*Rhinichthys* spp.	July 14 - August 12	X	X		X			X			X		
Largemouth Bass	*Micropterus salmoides*	-		X										
Northern Pikeminnow	*Ptychocheilus oregonensis*	July 13 - September 22	X	X		X	X	X	X			X	X	
Lamprey	*Lampetra* spp.	-					X			X				
Peamouth	*Mylocheilus caurinus*	July 14 - August 12	X	X		X							X	
Redside Shiner	*Richardsonius balteatus*	July 12 - September 1	X	X		X			X			X		
Sculpin	*Cottus* spp.	-		X			X			X				
Smallmouth Bass	*Micropterus dolomieui*	July 8 - 22	X	X		X	X	X					X	
Sucker	*Catostomus* spp.	July 7 - August 26	X	X		X	X		X	X			X	
Threespine Stickleback	*Gasterosteus aculeatus*	-		X	X			X						
Yellow Perch	*Perca flavescens*	-					X							

L - Larvae
J - Juvenile
A - Adult

Impacts to fish larvae were difficult to evaluate due to their abundance and fragile nature. Larvae were first collected on July 7 and were last seen in samples on September 22. Fish larvae including common carp, dace, northern pikeminnow, peamouth, redside shiner, smallmouth bass, and sucker, were identified as stranded or entrapped. Common carp, dace, northern pikeminnow, redside shiner, and sucker, were found as mortalities.

Juvenile fish continued to be found in samples throughout the summer and early fall period. Juvenile fishes including bullhead, northern pikeminnow, lamprey, sculpin, smallmouth bass, and yellow perch, were observed stranded or entrapped. Northern pikeminnow, lamprey, peamouth, sculpin, smallmouth bass, and sucker were found as thermal or stranding mortalities.

Four species of adult fish were collected in entrapments during the summer and early fall sampling period including common carp, northern pikeminnow, smallmouth bass, and threespine stickleback. No adult fish were observed as mortalities. Northern pikeminnow and smallmouth bass were the only species seen entrapped in all lifestages.

Habitat trends were difficult to assess due to variability in the data and low number of sample sites. No strong relationships were detected between fish species gathered during the late season and the habitat parameters of substrates, embeddedness, and vegetation.

Scanning Hydrographic Operational Airborne Lidar Survey (SHOALS)

Detailed bathymetry was collected on 35.13 km^2 of the Hanford Reach from Rkm 571.3 to Rkm 606.9 from August 1 through August 4 (**Figure 20**). This area was projected to cover areas impacted by river flows from approximately 30 kcfs to 400 kcfs. Seechi depth measurements in the forebay of Priest Rapids Dam during the survey dates ranged from 2.3 to 3.2 meters.

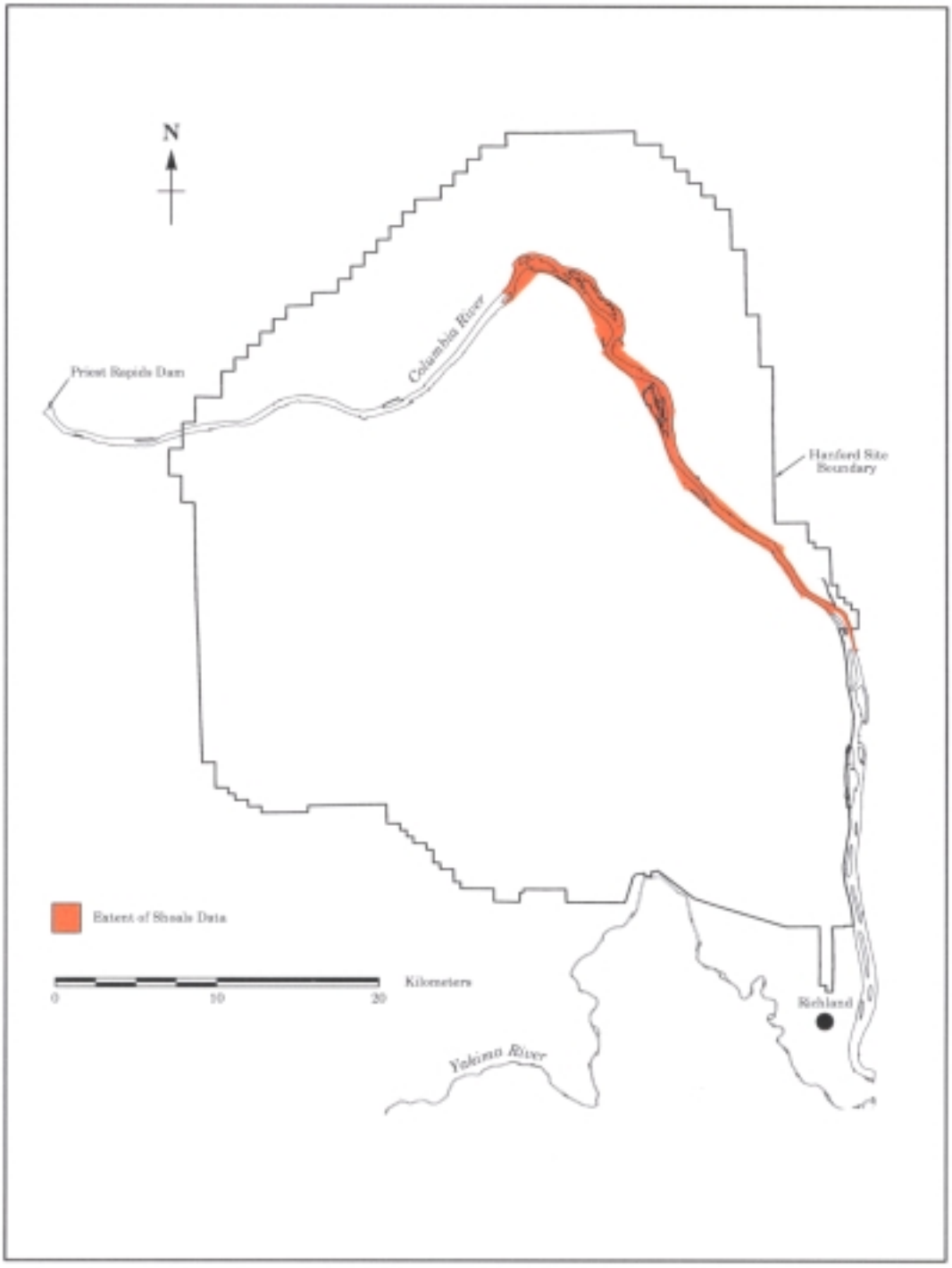


Figure 20. Extent of SHOALS coverage on the Hanford Reach of Columbia River.

Benthic Macroinvertebrate Survey

Based on preliminary results, the U of I and SPC determined an optimal combination of parameters for the 2-year benthic macroinvertebrate evaluation to be conducted in 1999 and 2000. U of I and SPC found that brick artificial substrates with a colonization period of 4 weeks at site location number 2 (Rkm 598 on the top of

Rosseau Island across from Locke Island) is the optimal combination to assess the effects of water level changes during the 1999 benthic macroinvertebrate study. Further details are provided in **Appendix B**.

Temperature Monitoring and Thermal Stress Investigations

Optic Stowaway temperature dataloggers were positioned in entrapments and stranding sites that repeatedly contained juvenile fall chinook salmon. Temperature dataloggers were also placed in the river to provide baseline river temperature data. A repeat entrapment, located at 100 F Islands, containing two optic stowaway units was modeled from April 21 through June 28 using MASS1 (**Figure 21)**. The figure presents temperatures in the river, in the entrapment, and the lethal juvenile fall chinook salmon temperature threshold. The figure also shows the river fluctuations and the critical elevation of the entrapment. Temperatures ranged from a daytime high of 37.0°C to a nighttime low of 6.0°C. River temperatures remained relatively uniform, with small daytime increases, small nighttime decreases, and a cumulative seasonal increase. Extreme entrapment temperatures may indicate complete entrapment drainage.

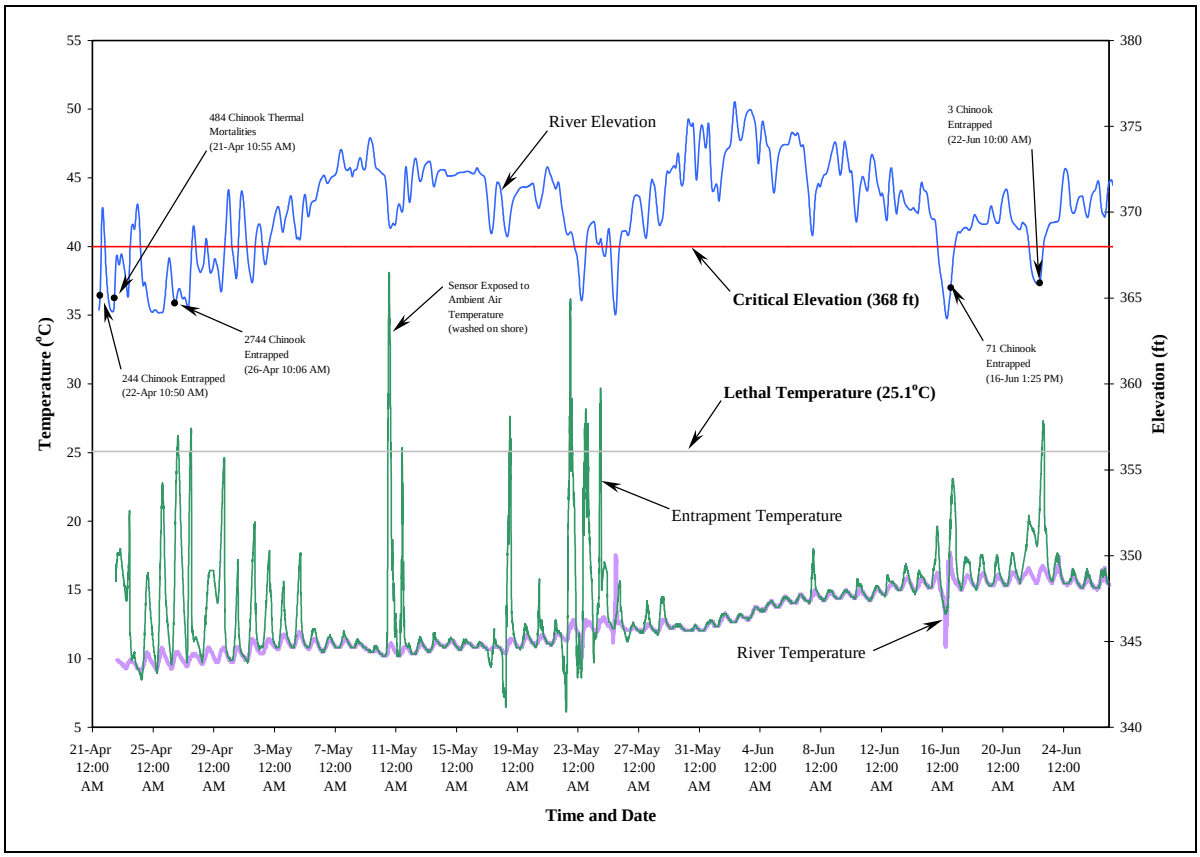


Figure 21. History of an entrapment (water temperature, river elevation history and stranding and entrapment events of juvenile fall chinook salmon) located at 100 F islands on the Hanford Reach of Columbia River.

Results of the USGS/BRD laboratory thermal stress investigations for juvenile fall chinook salmon exposed to thermal stressors similar to those on the Hanford Reach indicated no increased vulnerability to predation. There were no variations in behavioral and performance factors observed when juvenile fall chinook salmon were exposed to two temperature fluctuation scenarios between 26.0°C and 12.0°C. Complete results are provided in **Appendix C**.

Conclusions

The 1998 evaluation assessed the impacts of water fluctuations from Priest Rapids Dam on rearing juvenile fall chinook salmon, other fish species, and the benthic macroinvertebrate communities of the Hanford Reach. The impacts were identified as stranding, entrapment, and thermal mortality of fish species and desiccation of benthic macroinvertebrates. Results of the evaluation provided preliminary information on fall chinook salmon emergence timing, rearing, distribution, size susceptibility, and stranding and thermal mortality. Data also provided information on juvenile fall chinook salmon susceptibility and mortality in relation to substrate size, embeddedness, vegetation, time of day, size of flow fluctuations, downramp rates, duration of entrapment, and flow fluctuation zones.

Juvenile Fall Chinook Salmon

Geist and Dauble (1998) noted Hanford Reach fall chinook salmon tend to spawn in areas with complex channel pattern, rather than where the channel is straight and simple. Redd counts, performed by PNNL in 1991, 1994, and 1995, showed that primary fall chinook salmon spawning occurred at Vernita Bar, Locke Island/White Bluffs, and 100 F Islands (**Figures 2, 3, & 4**). According to Healy (1998), upon emergence, fry swim or are displaced downstream. Vernita Bar, Locke Island/White Bluffs, and 100 F Islands provide rearing habitats that are immediately adjacent to or downstream of spawning areas. Island complexes such as Locke Island and 100 F Islands provide irregular shorelines with finer substrates (<32 mm) and lower water velocities. These early rearing areas are generally in close proximity to deeper higher velocity currents containing larger substrates used later in the juvenile fall chinook salmon rearing period. In the Hanford Reach, Dauble et al. (1989) found juvenile fall chinook salmon primarily in shoreline areas of reduced current velocity but were present throughout the river cross section during their early rearing and outmigration period. In a comparable river environment such as the Snake River, Chapman and Bjornn (1969) and Everest and Chapman (1972), reported that juvenile fall chinook salmon were most abundant where substrate particle size was small, velocity was low, and depth was shallow, but were additionally found, at least in small numbers, in virtually every habitat investigated (Healy 1998). During our investigation, juvenile fall chinook salmon were located throughout the Hanford Reach but highest concentrations were found at Locke Island/White Bluff Slough and 100 F Island. These areas contain good juvenile fall chinook salmon rearing habitat but also provide the highest potential for stranding and entrapment.

Juvenile fall chinook salmon began appearing in nearshore and entrapment sites on the Hanford Reach on March 19, one week after the estimated date of emergence (March 12). Stranding and entrapment of juvenile fall chinook salmon and its related mortality peaked at about the calculated end of emergence (May 4). Although juvenile fall chinook salmon continued to be stranded and entrapped through June 24, they appear most vulnerable in the early lifestages when their size and poor mobility keep them close to shore. Fish using the shallow shorelines are highly susceptible to any fluctuation. Our data suggest that most stranding and entrapment takes place in the first 0.9 m of vertical flow reduction. Juvenile fall chinook salmon must attain a mean fork length of approximately 59 mm before they are large and mobile enough to move into deeper habitats and actively avoid stranding and entrapment.

Our results show juvenile fall chinook salmon become stranded and entrapped over a variety of substrate sizes but are found most often over coarse gravels and small pebbles (codes 5 and 6). Stranding mortality, however, occurs more readily in larger substrates with minimal substrate embeddedness usually within 24 hours of the entrapment creation time. Thermal mortalities in entrapments take place over highly embedded fine substrates where drainage rates are low. Most thermal mortalities typically happen within three days of the entrapment creation time. Thermal mortalities occur when entrapment water temperatures exceed 25.1^{o}C, the upper lethal limit of juvenile fall chinook salmon. These events normally arise on days when ambient air temperatures are above 25.1^{o}C but can occur on cooler days when solar radiation is high.

Juvenile fall chinook salmon seem to avoid vegetation. Information collected on the Hanford Reach showed juvenile fall chinook salmon numbers decreased as vegetation density increased. Mortalities were highest over sites absent of vegetation. Juvenile fall chinook salmon were observed traveling over flooded vegetation but were not observed moving through vegetation. Less observed mortality in dense vegetation may also be attributed to sampling bias as these sites are more difficult to sample.

Our data indicate that juvenile fall chinook salmon are entrapped more frequently during nighttime hours and during the twilight hours of dawn and dusk. This behavior has been noted in earlier studies. Edmundson et al. (1968) and Don Chapman Consultants (1989) found that at night juvenile fall chinook salmon moved inshore to quiet water over sandy substrates or into pools and that most individuals settled to the bottom. Juvenile fall chinook salmon are most active in the water column during the day and become torpid on the river bottom at night (Key et al. 1994; Vendetti et al. 1996).

Juvenile fall chinook salmon on the Hanford Reach appeared to be most susceptible to stranding and entrapment when reductions in flow occurred between 100 kcfs and 140 kcfs in 1998. This susceptibility is likely due to the timing of these flows and to the channel configuration of the areas impacted. Flows between 100 kcfs and 140 kcfs typically occur during the early emergence and rearing period. These flow levels also impact areas of the channel marked by large gravel flats associated with a lower flood terrace.

Other Fish Species

Seventeen genera of fish other than chinook salmon with a minimum of 18 species were sampled on the Hanford Reach in nearshore, stranding and entrapment sites in 1998. Fishes sampled from nearshore sites included coho salmon, common carp, dace, mountain whitefish, northern pikeminnow, peamouth, redside shiner, sculpin, sucker, threespine stickleback, American shad, bluegill, chislemouth, largemouth bass, and smallmouth bass. Fishes found stranded and entrapped included northern pikeminnow, redside shiner, sucker, peamouth, threespine stickleback, sculpin, smallmouth bass, mountain whitefish, dace, common carp, lamprey, bullhead, and yellow perch. Some species known to occur on the Hanford Reach but absent from sample sites in 1998 included white sturgeon (*Acipenser transmontanus*), pumpkinseed (*Lepomis gibbosus*), white crappie (*Pomoxis annularis*), black crappie (*Pomoxis nigromaculatus*), channel catfish (*Ictalurus punctatus*), walleye (*Stizostedion vitreum*), sand roller (*Percoosis transmontana*), and sockeye salmon (*Oncorhynchus nerka*).

Seven species of fish known to reside or migrate through the Hanford Reach hold designations on federal and/or state endangered or threatened species lists and have the potential for impacts from stranding and entrapment. These species include Upper Columbia River spring chinook salmon, Upper Columbia River steelhead (*Oncorhynchus mykiss*), Columbia River bull trout (*Salvelinus confluentus*), Pacific lamprey (*Lampetra tridentatus*), river lamprey (*Lampetra ayresi*), mountain sucker (*Catostomus platyrhynchus*), and leopard dace (*Rhinichthys falcatus*).

Upper Columbia River spring chinook salmon were classified as a federal candidate species (elevated to federal endangered status March 1999) by the USFWS in 1998. Some juvenile spring chinook salmon found on the Hanford Reach may originate from WDFW Ringold Fish Hatchery (Rkm 570.5) but most individuals are downstream migrants from Upper Columbia River stocks. Although these juvenile spring chinook salmon tend to be large (>100 mm) and less susceptible to stranding and entrapment, three fish were seined from an entrapment at 100 F Islands on April 29, 1998. These individuals were not adipose fin clipped and may have been from wild stocks. Flow fluctuations from Priest Rapids Dam during juvenile spring chinook salmon migration may impact this species.

Based on determinations by the NMFS, Upper Columbia River steelhead were listed as a federal endangered species in 1997. The listing encompasses the Wells Hatchery stock and all naturally spawned populations of steelhead (and their progeny) in streams in the Columbia River Basin upstream from the Yakima River to the United States/Canada Border. This listing includes the Hanford Reach where steelhead redds have been reported in the past (Eldred 1970; Watson 1973; Becker 1985). Steelhead on the Hanford Reach would most likely spawn between February and early June, with peak spawning in mid-May (Mueller and Geist 1999). Depending on water temperatures, fry would emerge in mid-June to mid-August. No juvenile steelhead were collected in any sample sites in 1998.

The USFWS determined threatened status for Columbia River bull trout in 1998. Historically, bull trout occurred on the Hanford Reach. Gray and Dauble (1977) reported capturing bull trout (described as Dolly Varden *Salvelinus malma*) near Wooded Island (Rkm 557 – 566) and Coyote Rapids (Rkm 605 – 613). It is now believed, however, that bull trout has been extirpated from this area. Most bull trout in the mid-Columbia

River geographic area have been isolated by dams or unsuitable habitat created by water diversions. (63 FR 31651).

Pacific and river lamprey are listed as federal species of concern and river lamprey are also designated as a Washington State candidate species. Both species have been observed on the Hanford Reach (Gray and Dauble 1977). Five juvenile lamprey were sampled as mortalities in stranding and entrapment sites in 1998. Since lamprey ammocoetes spend 4-6 years in mud as filter feeders before migrating to the ocean, it is difficult to assess the impacts of flow fluctuations on the Hanford Reach to these species.

Mountain sucker and leopard dace are considered Washington State candidate species. These species have also been noted on the Hanford Reach by Gray and Dauble (1977). In 1998, 8,021 sucker and 38 dace were found stranded or entrapped on the Hanford Reach. These fish were not differentiated to species and may have been mountain sucker or leopard dace.

Macroinvertebrates

Desiccation of the macroinvertebrate community was observed during the 1998 investigation. The U of I and SPC determined sampling parameters and protocols for the 1999 full-scale evaluation. The evaluation on the impacts of desiccation of the macroinvertebrate community will be performed during the summer and early fall of 1999.

Recommendations

The joint fish managers, consisting of WDFW, the Columbia River Intertribal Fish Commission, the Tribes of the Columbia River Basin, the Oregon Department of Fish and Wildlife, NMFS, and USFWS, recommended, based on the results the 1998 evaluation, that operations at Priest Rapids Dam create no fluctuations (flat loading) and/or steadily increasing flows on the Hanford Reach of the Columbia River throughout the fall chinook salmon emergence and rearing period. This recommendation was provided to the power managers, a group consisting GCPUD, BPA, the United States Bureau of Reclamation, and the Mid-Columbia Public Utility Districts (Chelan and Douglas Counties). System models were developed by GCPUD to determine the feasibility of implementing this reccomendation. Through this process it was determined that neither flat loading or steadily increasing project discharge could be accomplished throughout the fall chinook salmon emergence and rearing period. Subsquentially, on February 5, 1999, at a Policy Group meeting between the joint fish and power managers, the framework for a 1999 interim protection plan was proposed by the power managers. This was designed to meet the following criteria:

1) Substantially more protection for juvenile fall chinook fry than occurred in 1998.

2) Preservation of opportunity for load-following/power peaking operations.

3) System coordinated river operations.

4) Provide ability to monitor and evaluate in-season and adaptively manage operations to reduce stranding and entrapment.

The power managers proposed the following operating scenarios to incorporate in the 1999 interim protection plan:

1) Limit daily fluctuations to a range of +/-20 kcfs (a range of 40 kcfs) when weekly average flows are less than 170 kcfs.

2) Limit daily fluctuations to a range of +/-30 kcfs (a range of 60 kcfs) when weekly average flows are above 170 kcfs.

3) Rewetting of entrapment zones.

Further development of the interim protection plan should be continued in 1999.

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Appendix A

Substrate Size, Substrate Embeddedness, and Vegetation Codes

Substrate Codes

Dominant substrate is most common to the sample area and subdominant is the next most common substrate class.

Code	Substrate class
1	Fines (clay to coarse sand (<1 mm))
2	Very coarse sand (1-2 mm)
3	Fine gravel (2-4 mm)
4	Medium gravel (4-8 mm)
5	Coarse gravel (8-16 mm)
6	Small pebble (16-32 mm)
7	Large pebble (32-64 mm)
8	Cobble or rubble (64-256 mm)
9	Boulder (>256 mm)

Substrate Embeddedness Codes

The substrate embeddedness is estimated visually. Substrate embeddedness refers to the degree that the interstices between the larger particles are filled by sand, silt or clay.

Code	% Fines	Description
1	0-25	Openings between dominant sized particles are 1/3 to 1/2 the size of the particles. Few fines in between. Edges are clearly discernable.
2	25-50	Openings are apparent but <1/4 the size of the particles. Edges are discernable but up to half obscured.
3	50-75	Openings are completely filled but half of edges are still discernable.
4	75-100	All openings are obscured. Only one or two edges discernable and size cannot be determined without removal.

Vegetation Codes

Vegetation is assessed visually to estimate the percent of ground coverage.

Code	Description
1	No vegetation present.
2	Sparse vegetation, substrate is completely evident.
3	Medium vegetation, substrate is only partially obscured.
4	Dense vegetation, substrate is nearly or completely obscured by the vegetation.

Appendix B

Effects of Water Level Fluctuations on Macroinvertebrates in the Hanford Reach, Columbia River: 1998 Evaluation of Benthic Macroinvertebrate Colonization on Artificial Substrata (Year 1)

Appendix B

Effects of Water Level Fluctuations on Macroinvertebrates in the Hanford Reach, Columbia River: 1998 Evaluation of Benthic Macroinvertebrate Colonization on Artificial Substrata (Year 1)

Prepared for
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ABSTRACT

The Hanford Reach, the last free-flowing section of the Columbia River, provides critical habitat for wild fall chinook salmon *Oncorhynchus tshawytscha*. Although free flowing, the Hanford Reach is affected by rapid changes in discharge from hydroelectric operations of Priest Rapids Dam, immediately upstream. Fluctuations in discharge result in wide changes in water levels that can strand biota in shallow areas. We conducted a preliminary survey of benthic macroinvertebrates on artificial substrata in 1998 to determine sufficient sample size, colonization period, substrate type, and sampling locations to evaluate effects of diel water level fluctuations on benthic macroinvertebrates in the Hanford Reach during Year-2. Samples were collected with construction bricks and barbecue baskets containing concrete cones as artificial substrata to compare their effectiveness. Five of each substrate were collectively attached to a strand, with six replicate strands deployed at each of three sites, and half retrieved from each site at 4 or 6 weeks, respectively. Total invertebrate densities ranged from 500/m^2 to 120,140/m^2. Benthic macroinvertebrate fauna was low in diversity and dominated primarily by Chironomidae (midges) and Trichoptera (caddisfly) larvae. Collectively, both taxa accounted for approximately 87% of the macroinvertebrate fauna by density. Barbecue baskets colonized significantly higher densities and standing crops of benthic macroinvertebrates than brick substrates. However, we selected bricks as our substrate for Year-2 studies and a shorter colonization duration because of lower variances and concern for sorting time.

INTRODUCTION

Aquatic invertebrates play an important role in the structure and function of river ecosystems. Benthic macroinvertebrates represent an intermediate link in the plant-to-fish food chain, and their role is integral in the support of viable fish populations (Brusven et al. 1974). Benthic macroinvertebrates serve as an important food source to both downstream migrating and rearing salmonids. Curet (1994) reported that subyearling chinook salmon *Oncorhynchus tshawytscha* in Lower Granite Reservoir, Snake River, were opportunistic feeders, consuming mainly dipteran and ephemeropteran insects. In the Hanford Reach, Columbia River, Nietzel (1996) reported a close association between organisms in the stomachs of nearly all juvenile and adult fishes and the benthic and drift invertebrate communities. A reduction in benthic macroinvertebrates as a result of environmental changes, such as water level fluctuations, could potentially influence the productivity of the associated fish community.

The purpose of our study is to evaluate effects of diel water level fluctuations on the benthic macroinvertebrate community downstream of Priest Rapids Dam in the Hanford Reach, Columbia River (Figure 1). This section of the river provides critical habitat for the fall chinook salmon stock in the Columbia River. Determining the ecological impacts of fluctuations in water level will help in developing plans to adjust hydropower operations to lessen effects on aquatic species and protect critical habitat.

Currently, the largest of only two remaining wild stocks of fall chinook salmon in the Columbia River spawns in the Hanford Reach. This reach is the last free-flowing section of the Columbia River and is known as the last “strong-hold” for wild chinook salmon. Although free flowing, the Hanford Reach is affected by changes in discharge from the

hydroelectric Priest Rapids Dam at the upstream end of the reach (Figure 2). Water level fluctuations of over 1.7 vertical meters (5ft) commonly occur along the reach (Nietzel 1996) as a result of hydropeaking operations at the dam that strand rearing chinook salmon and their food items (Bauersfeld 1978, Becker et al. 1981). Concern has been expressed about the impacts of these water level fluctuations on macroinvertebrate food production and populations.

The damming of major rivers for hydroelectric power generation throughout the world has revealed several obvious patterns of effects on their downstream riverine ecosystems. No use of water, with the possible exception of gross waste disposal, can have as massive an effect on the original ecology of a river as large-scale hydroelectric development (Ruggles and Watt 1975). The physical variables of thermal and flow regimes are most often observed to change downstream of hydroelectric dams, ultimately affecting the river biota. Our primary interest is with respect to the implications of changes in flow regimen in the Hanford Reach, Columbia River.

Changes in flow regimen downstream of dams can be described in terms of four general categories. The receiving stream may be affected by seasonal flow constancy, reduced flow, increased flow, and short-term flow fluctuation (Ward 1976). These scenarios occur or are often associated with each other depending on the operational characteristics of the particular dam. Each of these types of changes in flow regimen is likely to affect the downstream aquatic biota in different ways. Our study examines effects of short- and long-term fluctuations in flow resulting from power peaking operations of the Priest Rapids Dam on benthic macroinvertebrates.

Prior investigations have specified and quantified flow fluctuations by different means, yet each has documented a general pattern of detrimental effects of rapidly varying flows on benthic macroinvertebrates. Flow fluctuation has been described by different researchers on the basis of changes in flow, riffle area, velocity, depth, or wetted substrate. Invertebrate communities have demonstrated reduced species diversity, density, biomass, and mean individual weight as a result of changing flows (Cushman 1985).

Little information has been collected on the benthic macroinvertebrate fauna in the Hanford Reach of the Columbia River, despite its importance as a chinook salmon spawning and rearing area. Robeck et al. (1954) studied the Columbia River and its tributaries from spring 1951 to spring 1953 and their findings suggested that the Columbia River had "clean water" forms of macroinvertebrates such as mayflies, midges, caddisflies, and mollusks at that time. Page and Neitzel (1978) used artificial substrata and found similar results to the earlier studies in the Columbia River; the fauna was low in diversity and dominated primarily by midges and caddisfly larvae, which comprised more than 90% of the fauna. In addition, no site differences in the abundance of aquatic macroinvertebrates were observed. This body of literature suggests that three categories of benthic macroinvertebrates - midges, caddisfly larvae, and others (flatworms, mayflies, and other benthic invertebrates) may be appropriate organisms for assessing effects of water level fluctuations in the Hanford Reach, Columbia River. Therefore our study determines the impacts of water level fluctuations on abundance of these organisms.

PURPOSE AND SCOPE OF STUDY:

Diel water level fluctuations downstream of electrical power generation facilities can adversely affect shallow water organisms by exposing the shoreline to desiccation. Benthic macroinvertebrates can be affected directly by loss of habitat and desiccation and indirectly by loss of periphyton that is food. Numerous authors have reported that the littoral zone of aquatic ecosystems are highly productive for benthic macroinvertebrates and important foraging areas for fishes. However, fluctuating water levels can expose these areas to desiccation and create conditions only inhabitable by more drought tolerant communities. Such water level fluctuations occur downstream of Priest Rapids Dam in the Hanford Reach, Columbia River. Because of the importance of this area to rearing chinook salmon, this study has the following objectives:

OBJECTIVES:

1. To determine the best colonization duration, location, and artificial substrata to produce adequate numbers of benthic macroinvertebrates for Year-2 manipulative studies in the Hanford Reach, Columbia River.
   (Year-1/1998)

2. To evaluate the long- term effects of diel fluctuations of water levels on the benthic macroinvertebrate community in the Hanford Reach, Columbia River.
   (Year-2/1999)

3. Evaluate the short-term effects of diel water level fluctuations on the benthic macroinvertebrate community in the Hanford Reach, Columbia River.
   (Year-2/1999)

We examined the colonization of benthic macroinvertebrates on two different artificial substrata at three locations in the Hanford Reach. To effectively evaluate effects of diel water level fluctuations, sufficient numbers of benthic macroinvertebrates are needed to

accurately represent the invertebrate community that experience these changes. Thus, satisfactory completion of objective I provides data on the best location for colonization by midges, caddisfly larvae, and others (flatworms, mayflies, and other benthic invertebrates) and the best substratum to be able to test effects for Year-2 sampling.

STUDY AREA

The Hanford Reach of the Columbia River is a 81.6 km (51 mile) free-flowing section extending from RM (river mile) 396 to RM 345 (Figure 1). The Priest Rapids Dam, at the upstream end of the reach (RM 396), has been in full operation since September 1961 and since then has largely influenced the daily flow regimen of the Hanford Reach. Daily average flows range from 700 to 1008m^3/s. Monthly mean flows typically peak from April through June during spring runoff from snow melt, and are lowest from September through October (Nietzel 1996). Sampling locations are as follows:

Three sampling locations were selected. The first location is approximately at RM 368 while the second and third sites are at RM 370.5 and 379, respectively (Figure 1). The third location (RM 379) is considered the reference site since artificial substrata have been successfully colonized by target organisms at this location (C. Cushing, Retired, Battelle Northwest Laboratory, Personal Communication). The three sampling sites were also chosen because they consist of similar macro-habitats.

METHODS and MATERIALS

The purpose of Year-1 sampling was to identify the best duration and location for colonization of benthic macroinvertebrates and the best artificial substratum. Attached organisms were sampled with artificial substrata primarily because of the difficulty in sampling by other means under the extreme conditions of fast current, large, hard substrate, and variable water levels in the Hanford Reach. In addition, artificial substrata can standardize the sampling so as to avoid confounding effects of differences in substrate size and variability (Rosenberg and Resh 1982). Artificial substrata do not necessarily reflect the entire community dynamics present because they are selective for certain organisms; however, they do provide a useful means of comparing population trends among locations (Brusven and Trihey 1978).

Benthic macroinvertebrates were sampled by deploying two types of artificial substrata. The first type of substratum, described by Benfield et al. (1974), consists of 10 concrete cones, each with a surface area of 171.8 cm^2, placed in a wire barbecue basket measuring 25.4 cm long X 16.5 cm in diameter. The second method used construction bricks, each with a surface area of 763.1cm^2. Five barbeque baskets and five bricks were attached to a strand of polypropylene rope, and six replicate strands were deployed at each of the three sites, for a total of 180 samples (Figure 3). Each strand with its attached substrata were placed at sufficient depths to avoid exposure to air and anchored at both ends to avoid loss. Barbecue baskets and bricks were positioned at each location for 4 or 6 weeks to assess the necessary time for colonization, following predicted colonization durations established on the Snake River, Idaho (Haber and Brusven 1982, Nightengale 1999).

After each period of colonization, the bricks and the baskets were retrieved, placed in individual plastic buckets with water, disconnected from each strand, and brushed and rinsed of all attached organisms into their respective sample bucket. Contents of the buckets were then poured through a 0.595 mm sieve bucket (#30), rinsed into sample containers, and preserved in 10% formalin solution with rose bengal dye. All samples were then transported to the laboratory for sorting, identification, enumeration, and weighing.

Organisms were sorted from detritus into the three major taxonomic groups of interest (midges, caddisfly larvae, and all other invertebrates), then enumerated, and weighed collectively by taxonomic group. Taxa composition by biomass was only determined for the three broad taxa groups because of sample processing time constraints, particularly weighing all taxa separately for each sample. Dry weights were measured after oven drying at 60°C for 36 hours and weighed at two time intervals to assure complete dryness. This produced data on the number and dry weight of each taxonomic group of interest per basket or brick.

Statistical comparisons of mean density and biomass estimates were made for each substrata, colonization period, and site for the three broad taxonomic groups using a Kruskal-Wallis rank sum test. Based on the variability in colonization of barbecue baskets and bricks, the number of samples necessary to obtain significant statistical power for year-2 studies was calculated for each combination of site, colonization, and substratum. Sample size determination was calculated using the following equation:

$$\textit{sample size} = (\textit{t-value})^2 \times (\textit{sample variance}) / (\textit{accuracy} = 0.25) \times (\textit{sample mean})^2$$

RESULTS

We collected a total of 148 artificial substrata; 32 substrata were not utilized because of unexpected exposures (dewatering), suspected vandalism, or movement and subsequent loss. All three sampling sites colonized more than adequate numbers of benthic macroinvertebrates for year-2 tests of water level fluctuations. Mean benthic macroinvertebrate density across all samples was 27,621/m^2, and ranged from 500/m^2 to 134,652/m^2 (Table 1). Mean macroinvertebrate standing crop was 2.26g/m^2, and ranged from 0.03g/m^2 to 11.76g/m^2 (Table 2). Total macroinvertebrate fauna was low in diversity and dominated primarily by Chionomidae (midges) and Trichoptera (caddisfly) larvae (Figure 4). Midge and caddis collectively accounted for 87.43% of the total macroinvertebrate abundance, and 92.88% of the biomass, with midges contributing 51.49% of the abundance and caddis 67.89% of the biomass (Figure 4). Most of the macroinvertebrate taxa occurred in a large percentage of the samples collected (Table 3), although few taxa besides midge and caddis comprised a large portion of any individual sample.

Substrata

Mean density was significantly higher on barbecue basket substratum than brick substratum for midge ($p < 0.043$), and caddis ($p < 0.0035$) but not for ‘other’ taxa ($p > 0.6275$; Figure 5). Mean biomass was significantly higher on barbecue basket substrata for midge ($p < 0.0418$) and caddis ($p < 0.0382$), and no significant difference existed between substrata for the ’other’ taxa ($p > 0.4498$; Figure 5).

Comparisons of proportional abundance of benthic macroinvertebrate taxa between substrata, revealed few differences (Figure 6). Barbecue basket samples were colonized by higher percentages of midges and Hydropyschidae caddis, but smaller proportions of other caddis (Hydroptilidae, Leptoceridae, and Psychomyiidae) and most taxa of the'other' category. Taxa composition by weight was similar to composition by abundance, with only slightly higher proportions of midges and caddis found on barbecue basket substratum than on brick substratum (Figure 7).

Colonization Period

The average midge density estimate (18,882/m^2) was significantly higher for 4 week colonization samples than 6 week (9,172/m^2) colonizations ($p < 0.0004$; Figure 8). However, caddis taxa density was significantly lower (8,503/m^2) on 4 week colonization ($p < 0.0046$) substrata than 6 week (11,428/m^2). No significant difference was found in average density for the 'other' taxa group between colonization periods ($p > 0.1085$). Mean standing crops were not significantly different for any of the taxa groups, midge ($p > 0.1804$), caddis ($p > 0.1489$), and 'other' ($p > 0.3622$; Figure 8) for 4 or 6 week colonizations.

Comparisons of benthic macroinvertebrate taxa composition by density, between colonization durations reveal several substantial differences (Figure 6). Six week colonization samples had higher proportions of caddis but smaller proportions of midge than 4 week colonizations. Taxa composition based on biomass revealed little difference among groups between colonization periods (Figure 7).

Sites

Mean density was significantly different among all three sites for midges ($p < 0.0001$), caddis ($p < 0.0001$), and 'other' ($p < 0.0001$) taxa groups (Figure 9). Substrata at

site 2 (RM 370) were colonized with highest densities of all three sites (x = 41,906/m^2). Mean standing crops were significantly different among sites for midge ($p < 0.0140$), caddis ($p < 0.0001$), and 'other' ($p < 0.0049$) taxa groups (Figure 9).

Proportional abundance of benthic macroinvertebrate taxa by density and standing crop revealed significant differences (Figures 6, 7). Substrata at site 1 (RM 368) had a much higher proportion of midge and smaller proportion of caddis, and slightly smaller proportion of the 'other' category. Substrata taxa composition by weight at site 2 (RM 370) was noticeably different from both site 1 (RM 368) and site 3 (RM 379), with a higher proportion of caddis and lower proportions of midge and 'other' taxa categories (Figure 7).

DISCUSSION

Our findings revealed that the Hanford Reach benthic macroinvertebrate community on artificial substrata was comprised of mainly chironomids and several caddisfly taxa, mainly Hydropsychidae. These findings corroborate those found previously in the Hanford Reach (Page and Nietzel 1977a, 1977b, 1978, 1979; Page et.al. 1979; Nietzel 1996), and community composition was also similar to samples collected prior to the construction of Priest Rapids Dam (Figure 10). Robeck et al. (1954) found a benthic macroinvertebrate community dominated by midges and caddisfly, and with a larger component of molluscs. *Psychomyia* spp. caddisfly larvae were the only benthic macroinvertebrate we collected which had not previously been reported in the Hanford Reach.

Although the community composition was similar, our average total density (27,621/m^2) was nearly four times higher than the average total density (7,473/m^2) reported

by Robeck et al. (1954). However, both the environmental conditions and collection method were different, and therefore our high densities may in part be due to great differences in flow regime and using artificial substrata. Although, our mean benthic macroinvertebrate density is also high relative to more recent studies in other medium to large rivers in the Pacific Northwest. Gislason (1985) reported maximum densities of 16,763 macroinvertebrates/m^2 for the Skagit River, Washington, whereas Brusven et. al. (1974) found 6,494 macroinvertebrates/m^2 in the Snake River, Idaho. Gore (1977) found 5,350 macroinvertebrates/m^2 in the Tongue River, Montana. Our findings demonstrated the high abundance of benthic macroinvertebrates in the Hanford Reach, Columbia River.

Colonization on our artificial substrata was highly variable both between and within substrata, colonization periods, and among sites. Variation in densities plus variation induced possibly by water level fluctuations may have influenced colonization density, biomass, and community composition, especially among sampling sites. We attempted to place substrata at depths below predicted lowest water levels during the colonization periods; unfortunately, some substrata were unexpectedly exposed. However, our large sample sizes allowed us to discard substrata that appeared desiccated. Even though substrata included in this analysis were not exposed, microhabitat differences in depth and velocity likely varied as a result of the fluctuations in discharge at Priest Rapids Dam (Figure 11).

We determined the combination of site, colonization, and substratum which required the least number of samples to maintain sufficient precision in estimating the true mean density of all three macroinvertebrate taxa groups (Figure 12). The combination of brick artificial substrata colonized for 4 weeks at site 2 was selected to assess effects of water level changes during year-2 tests (Table 4). Although the combination of barbecue basket

substrata colonized for 4 weeks at Site 2 required fewer samples, we selected brick substrata because they were colonized at significantly lower densities and would provide the same precision for less effort to process samples.

We believe the use of artificial substrata will provide the most representative results of effects of diel water level fluctuations on benthic macroinvertebrates. Advantages of using pre-colonized substrata are numerous as periodic water level fluctuation tests commence with a known faunal abundance and composition. Precolonized substrata minimize the variation associated with previous seasonal and diel water level fluctuations, minimize sampling variation, and focus on effects of current water level changes.

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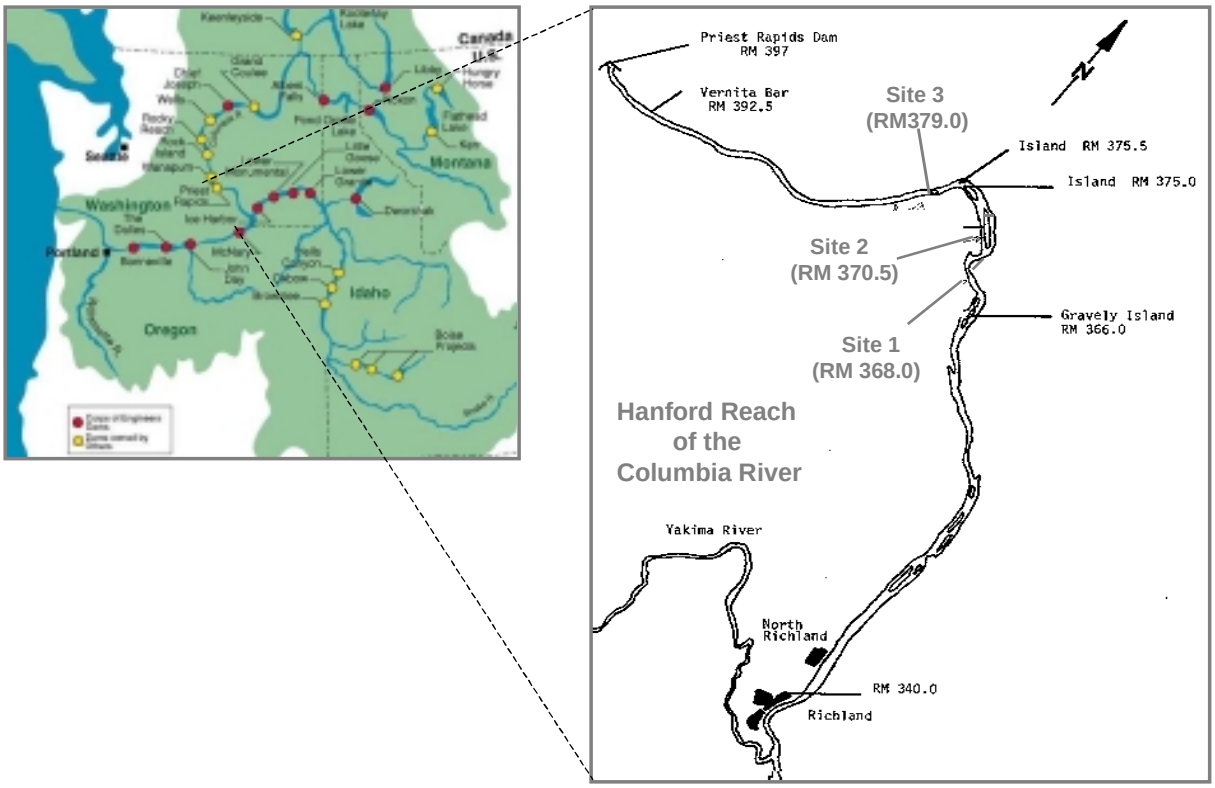


Figure 1. The Hanford Reach, Columbia River, Washington and location of invertebrate sampling sites.

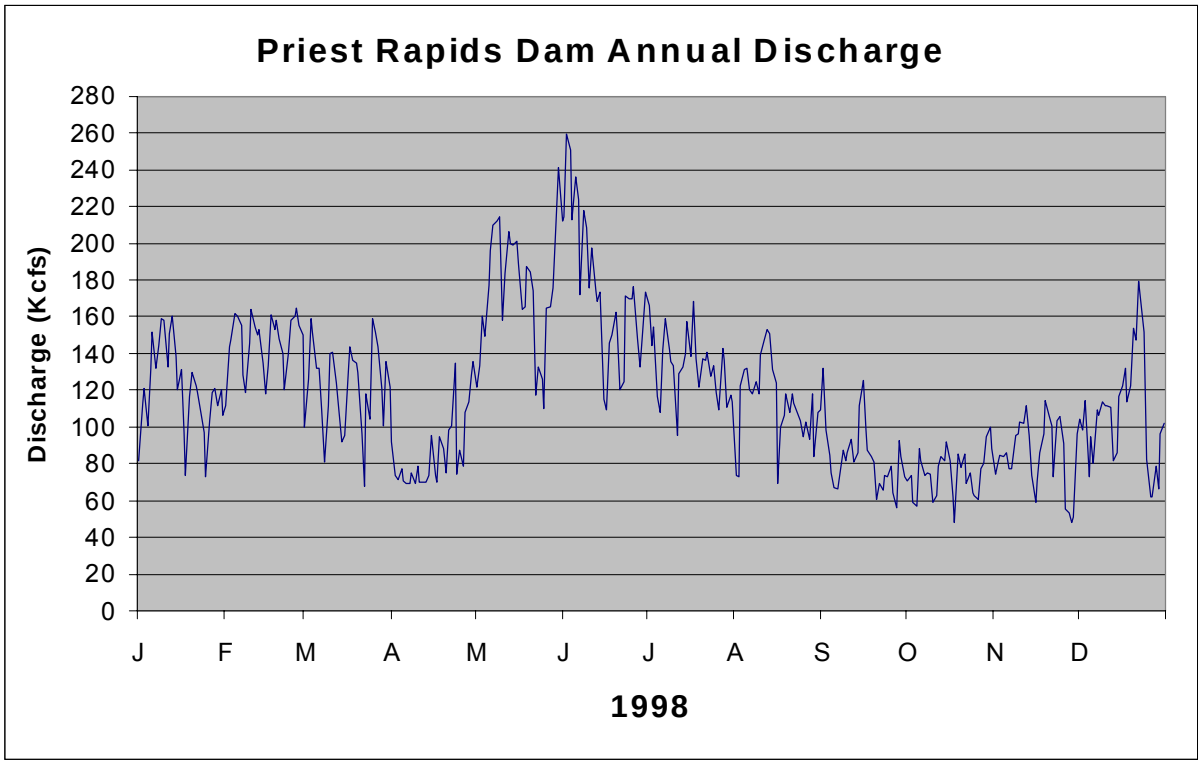
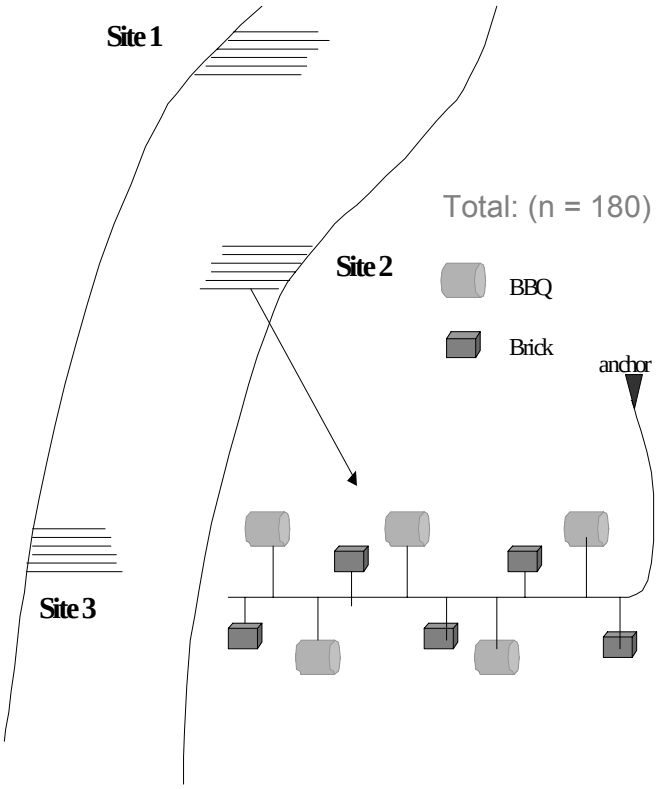


Figure 2. Annual discharge from Priest Rapids Dam in 1998 into the Hanford Reach, Columbia River, Washington.



BBQ Baskets:

- 5/strand
  (L-25.4cm X diam-6.5cm)
- 10 concrete cones / basket
  (SA=171cm^2 / BBQ)$^{)}$
- 30 baskets at each of 3 sites

Bricks:

- 5/strand (SA=763.1cm^2)
- 30 bricks at each of 3 sites

Retrieved half of both types of substrata from all sites at 4 weeks & remaining half at 6 weeks.

Figure 3. Artificial substrate sampling design for preliminary sampling to determine optimal substrata, colonization period, sites, and sample size (Objective I) for year-2 tests of water level fluctuation effects.

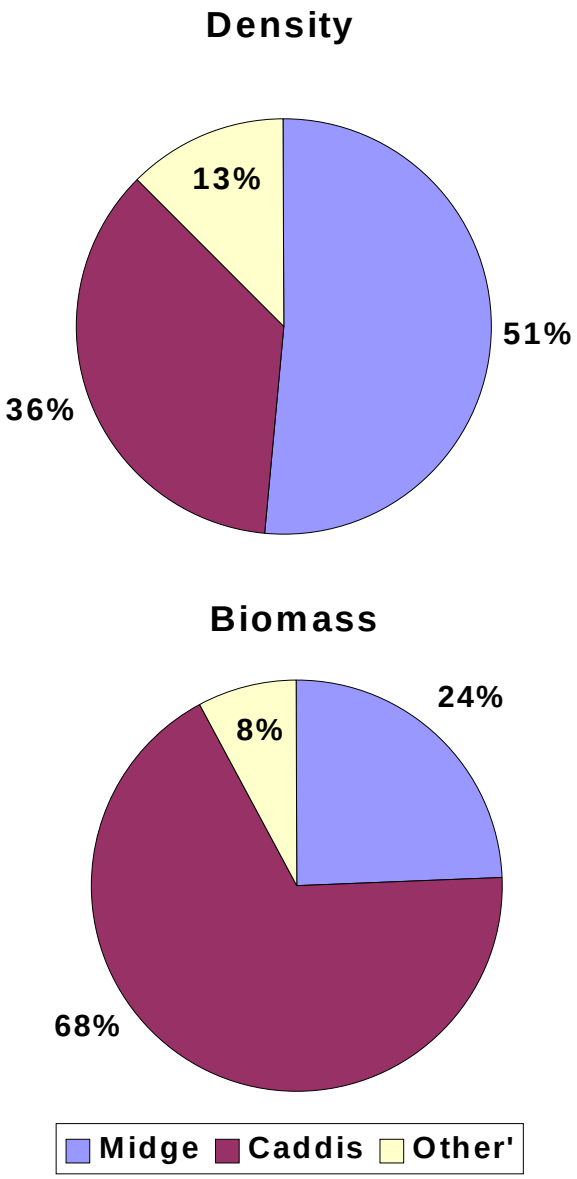


Figure 4. Mean proportional density and biomass of benthic macroinvertebrates collected on artificial substrata in the Hanford Reach, Columbia River, Washington, during 1998.

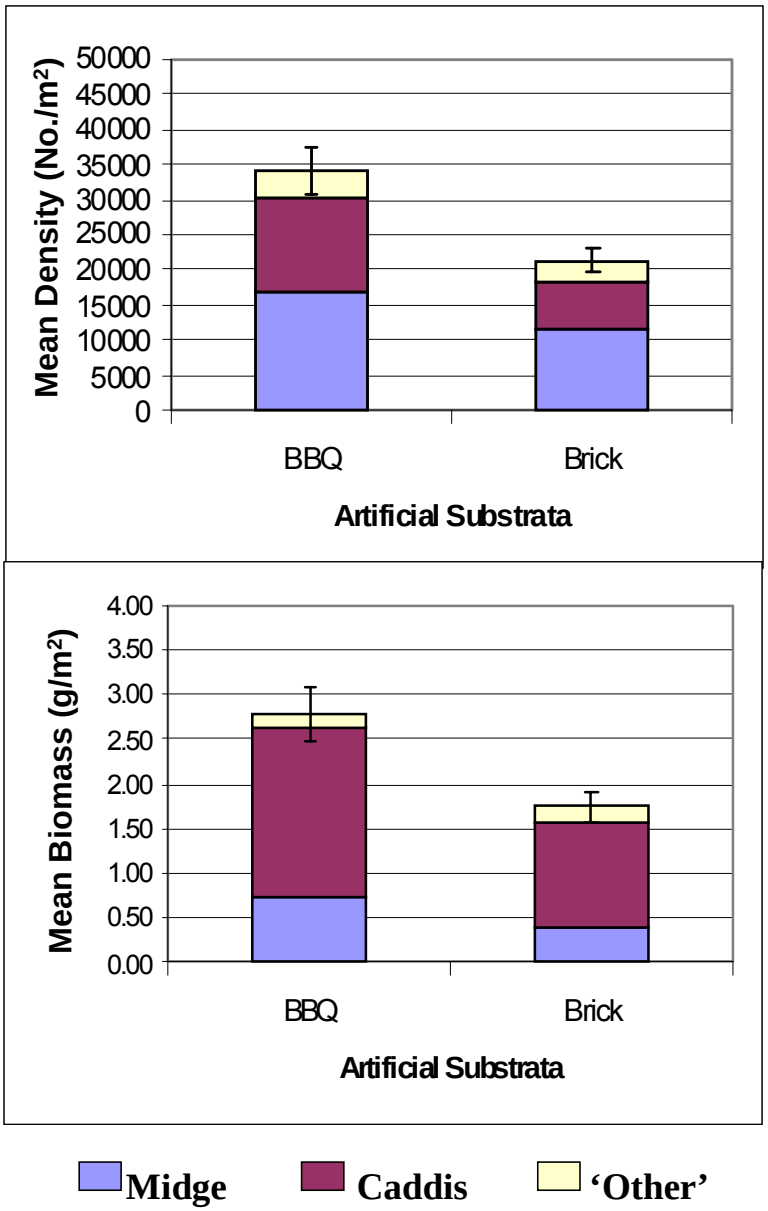


Figure 5. Mean density and biomass of midge, caddis, and other taxa categories of benthic macroinvertebrates collected on barbecue basket and brick artificial substrata in the Hanford Reach, Columbia River, Washington, during 1998. Vertical error bars represent (+/-) 1 Standard Error of the total density and biomass.

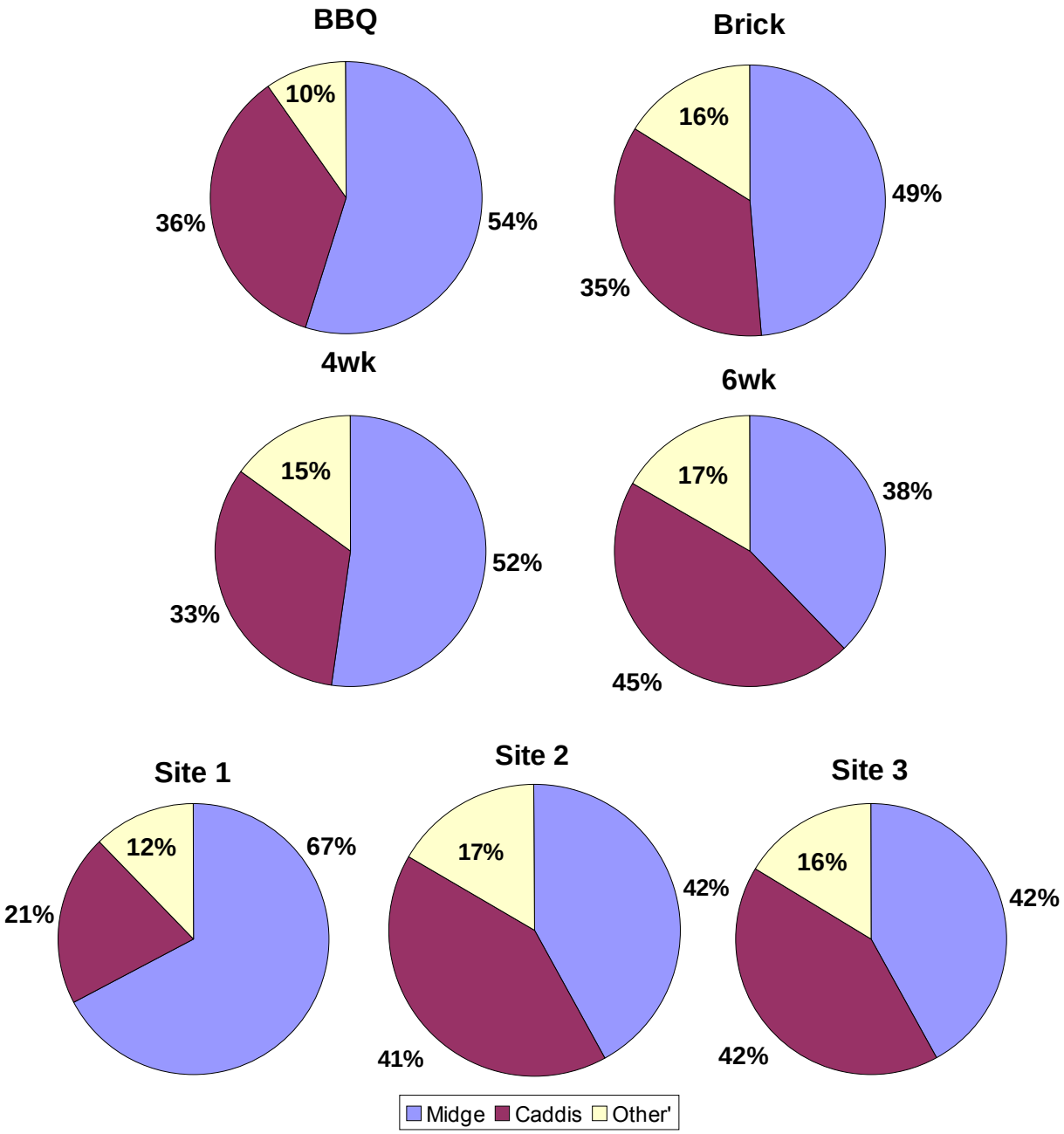


Figure 6. Mean proportional density of benthic macroinvertebrates collected from barbecue basket and brick artificial substrata colonized for 4 or 6 weeks at River Mile (RM) 360.5 (site 1), RM 370.5 (site 2), and RM 375.0 (site 3) in the Hanford Reach, Columbia River, Washington, during 1998.

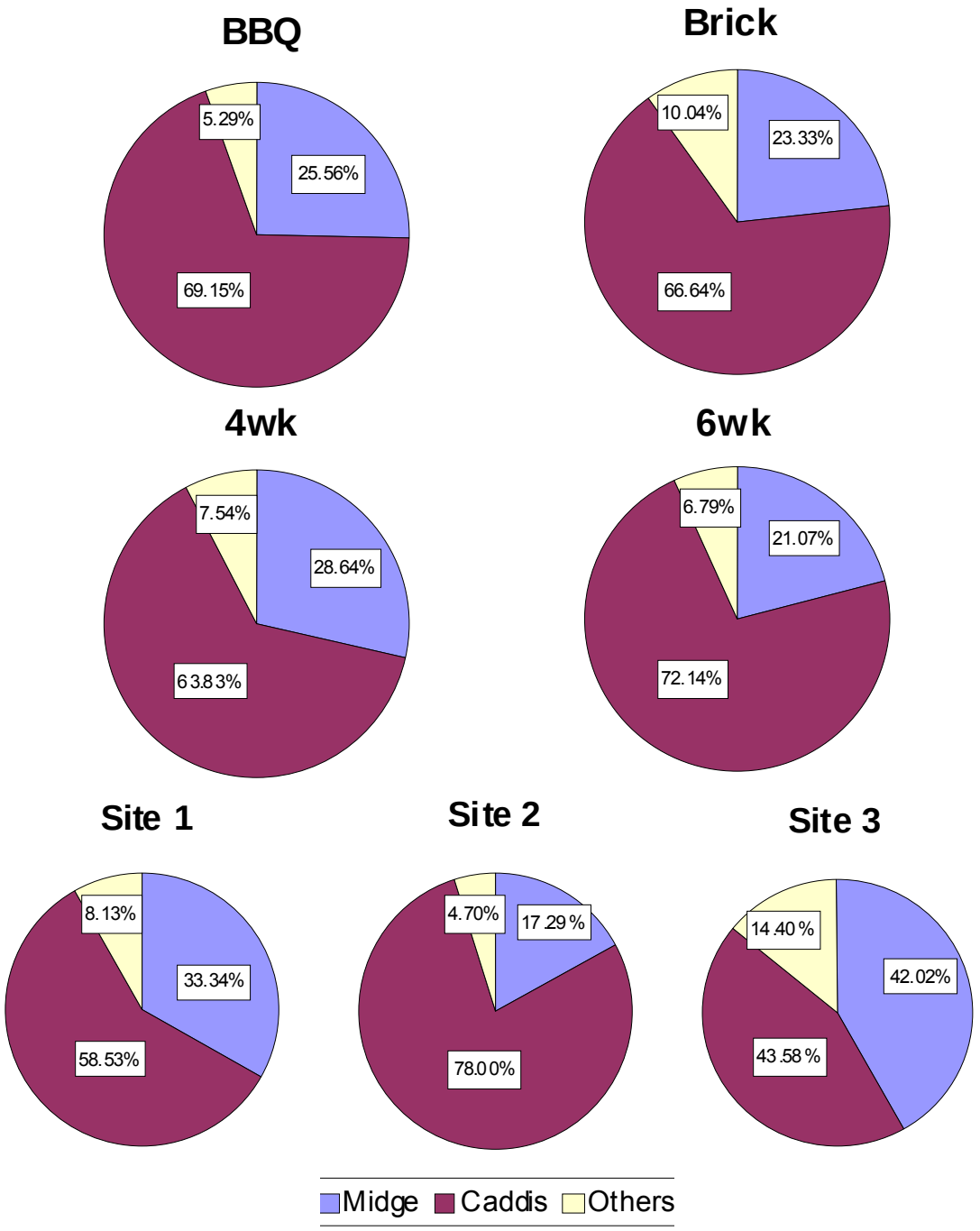


Figure 7. Mean proportional biomass of benthic macroinvertebrates (Mollusc weights excluded) collected from barbecue basket and brick artificial substrata colonized for 4 or 6 weeks at River Mile (RM) 360.5 (site 1), RM 370.5 (site 2), and RM 375.0 (site 3) in the Hanford Reach, Columbia River, Washington, during 1998.

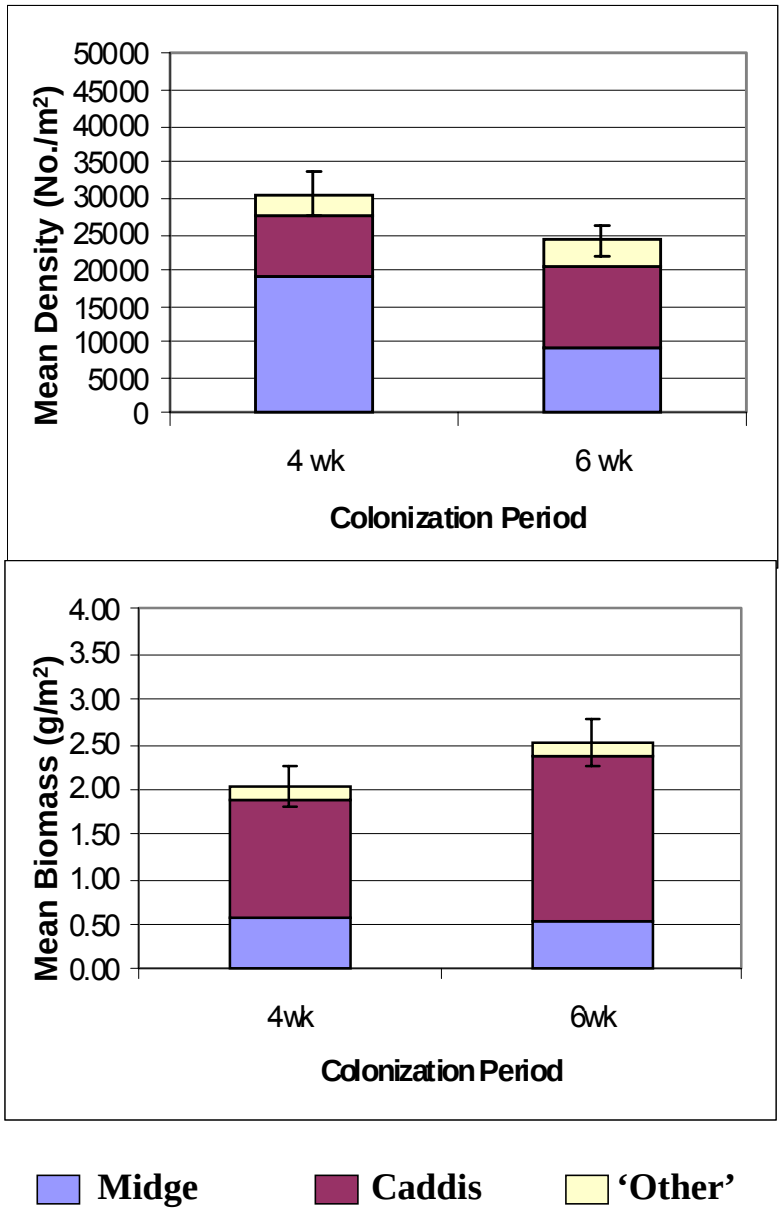


Figure 8. Mean density and biomass of midge, caddis, and other taxa categories of benthic macroinvertebrates collected on artificial substrata following 4 week or 6 week colonization periods in the Hanford Reach, Columbia River, Washington, during 1998. Vertical error bars represent (+/-) 1 standard error of the total density and biomass.

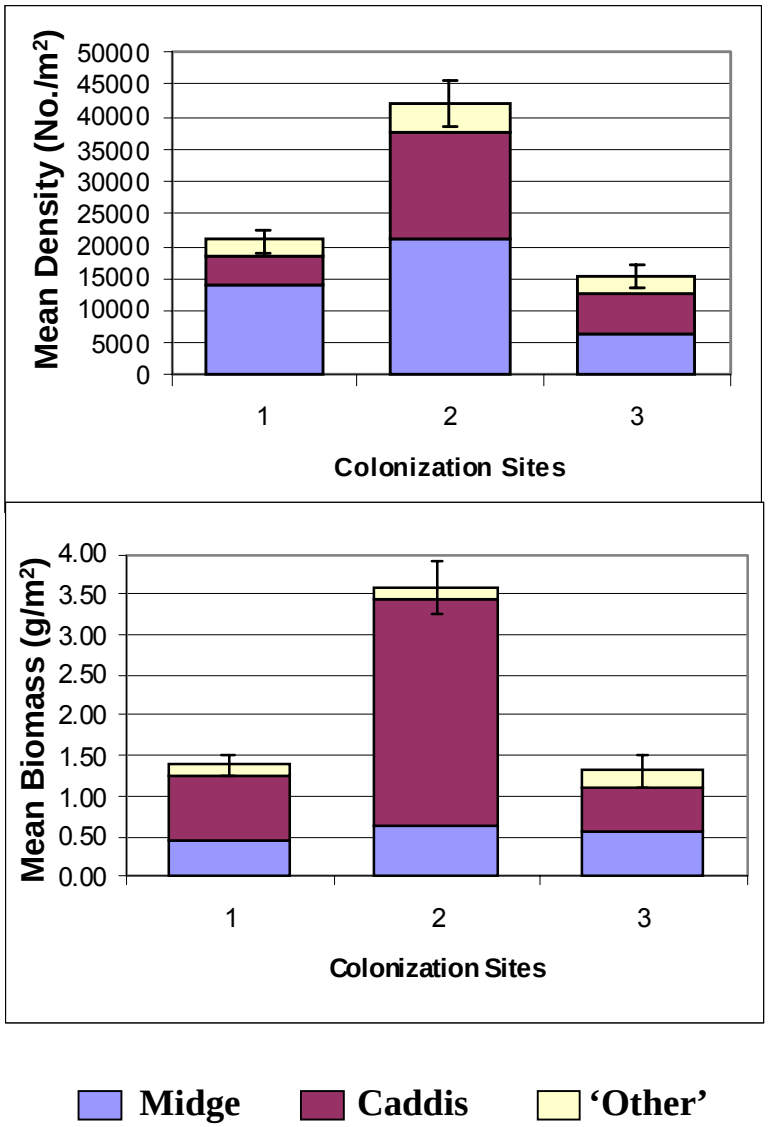


Figure 9. Mean density and biomass of midge, caddis, and other taxa categories of benthic macroinvertebrates collected at River Mile (RM) 360.5 (site 1), RM 370.5 (site 2), and RM 375.0 (site 3) of the Hanford Reach, Columbia River, Washington, during 1998. Vertical error bars represent (+/-) 1 standard error of the total density and biomass.

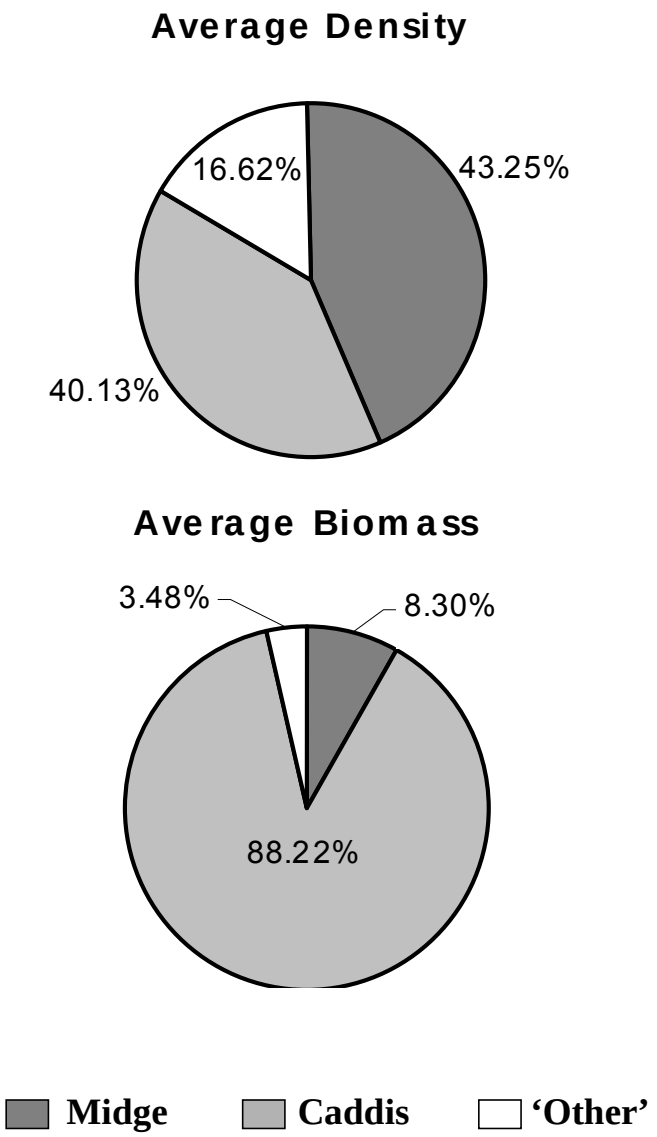


Figure 10. Mean proportional abundance and biomass (mollusc wts. excluded) of benthic macroinvertebrates from Surber samples collected from the Hanford Reach, Columbia River, Washington, 1952 (Robeck et al. 1954).

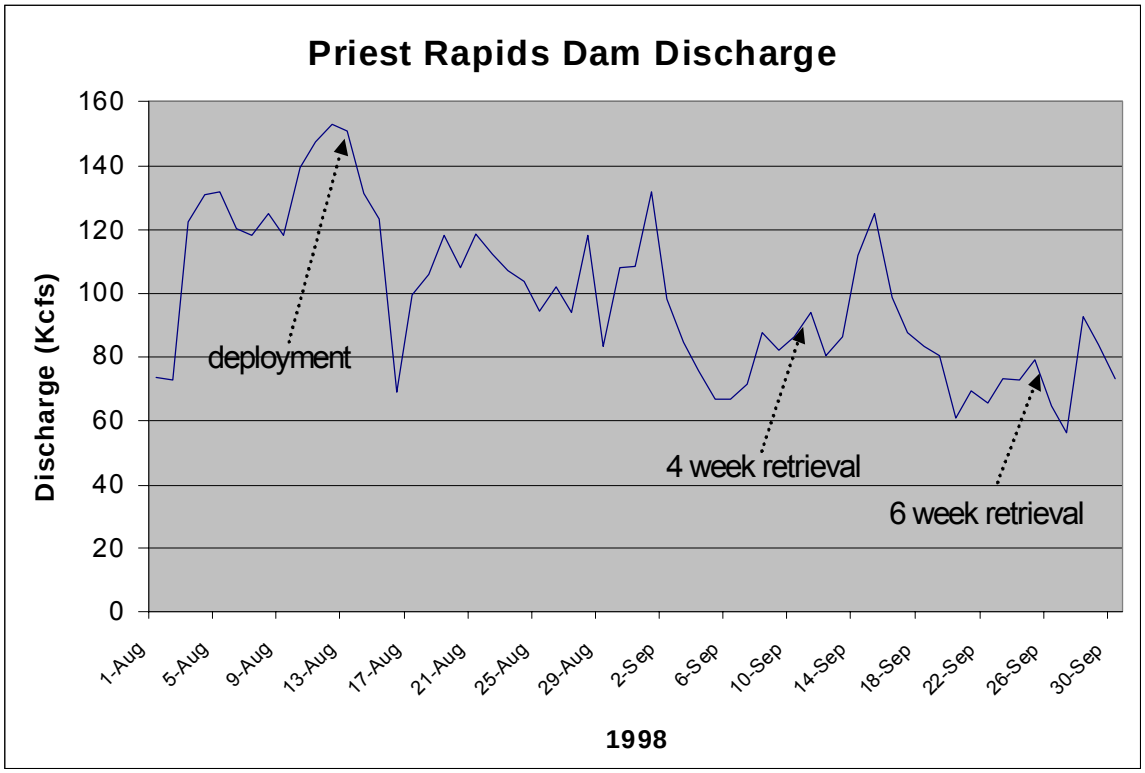
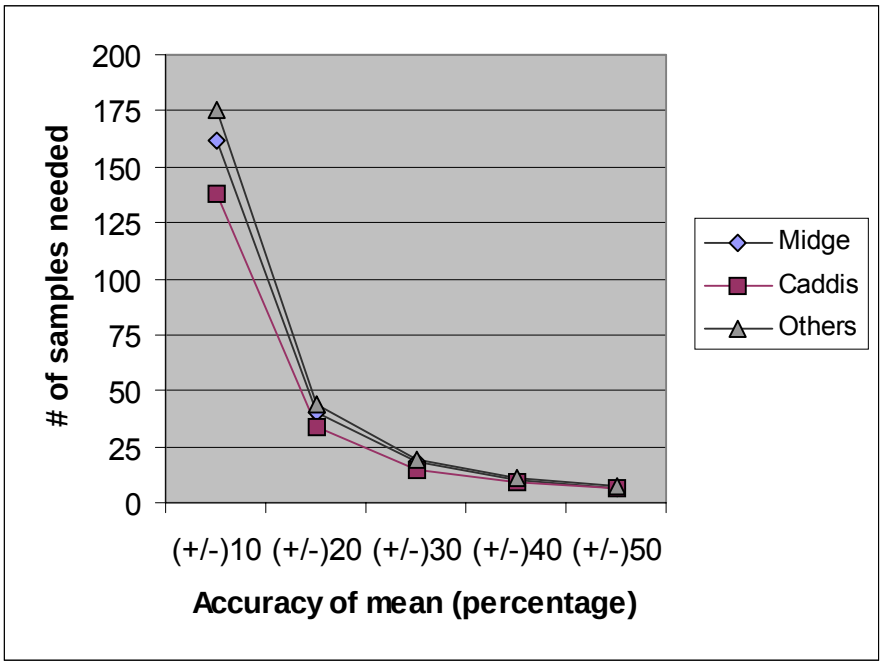


Figure 11. Mean daily discharge from Priest Rapids Dam, Columbia River from 1 August through 30 September, 1998. Dates of deployment and 4 or 6 week retrievals of artificial substrata are indicated.

$$\text{sample size} = \frac{(\text{t-value})^2\ (\text{sample variance})}{(\text{accuracy} \times \text{sample mean})^2}$$



Example: n=15, t-value = 1.761, μ = 5539
σ^2 = 16025784, accuracy = 0.10

$$\frac{(1.761)^2\ (16025784)}{(0.10 \times 5539)^2} = 162 \text{ samples for Midges}$$

Figure 12. Benthic macroinvertebrae sample size determination with example calculations using brick artificial substrata samples colonized for 6 weeks at site 2.

Table 1. Mean benthic macroinvertebrate density (#/m^2), standard error (SE), sample sizes and ranges, on artificial substrata in the Hanford Reach, Columbia River. Site 1 was at river mile 368.0, site 2 (RM 370.5), and site 3 (RM 379.0). Colonization periods (4wk and 6wk) and sites (1, 2,3) show the combined density of both bricks and baskets

	Mean (#/m^2)	**(+/-) SE**	**Min**	**Max**	**(n)**
Brick	21388	1672	589	60936	75
Basket	33854	3319	500	134652	73
4 wk	30457	3049	500	134652	79
6 wk	24193	2087	917	78233	69
Site 1	20836	1806	5180	53205	38
Site 2	41906	3550	917	134652	60
Site 3	15387	1862	500	54132	50

Table 2. Mean benthic macroinvertebrate biomass (g/m^2), standard error (SE), sample sizes and ranges, on artificial substrata in the Hanford Reach, Columbia River. Site 1 was at river mile 368.0, site 2 (RM 370.5), and site 3 (RM 379.0). Colonization periods (4wk and 6wk) and sites (1, 2,3) show the combined density of both bricks and baskets

	Mean (g./m^2)	**(+/-) SE**	**Min**	**Max**	**(n)**
Brick	1.74	0.16	0.06	6.81	75
Basket	2.78	0.30	0.03	11.76	73
4 wk	2.02	0.22	0.03	9.42	79
6 wk	2.52	0.27	0.11	11.76	69
Site 1	1.39	0.12	0.38	3.62	38
Site 2	3.59	0.32	0.18	11.76	60
Site 3	1.31	0.19	0.03	7.29	50

Table 3. Checklist of benthic macroinvertebrate taxa collected on artificial substrata in the Hanford Reach, Columbia River, categorized by percent occurence in samples.

Taxa	Brick	BBQ	Site 1	Site 2	Site 3
Cnidaria					
Hydrozoa	R	U	0	R	C
Hydra spp.					
Platyhelminthes					
Turbellaria (flatworms)	C	A	U	A	A
Dugesia spp.					
Nematoda (roundworms)	A	A	A	A	A
Annelida					
Oligochaeta (earthworms)	A	A	A	A	A
Arthropoda					
Crustacea					
Amphipoda (scuds)	R	0	0	0	R
Gammaridae					
Gammarus spp.					
Insecta					
Collembola (springtails)	0	R	0	0	R
Ephemeroptera (mayflies)	A	A	A	A	A
Ephemerellidae					
Caenidae					
Heptageniidae					
Trichoptera (caddisflies)					
Hydropsychidae	A	A	A	A	A
Hydropsyche spp.					
Cheumatopsyche spp.					
Hydroptilidae	A	A	C	A	A
Hydroptila spp.					
Leptoceridae	C	C	C	C	C
Oecetis spp.					
Psychomyiidae	C	C	U	A	C
Psychomyia spp.					
Lepidoptera					
Pyralidae (butterfly larvae)	C	C	U	C	C
Petrophila spp.					

Table 3. Continued

Taxa	Brick	BBQ	Site 1	Site 2	Site 3
Diptera					
Chironomidae (midges)	A	A	A	A	A
Simulidae (craneflies)	0	R	0	R	0
Arachnida					
Hydracarina (water mites)	A	A	A	A	A
Mollusca					
Gastropoda (snails)	C	U	R	C	C
Pelecypoda (mussels)	R	R	0	R	R

a (0) = absent from samples
b (R) = rare, found in less than 10 % of samples
c (U) = uncommon, found in 10 to 30 % of samples
d (C) = common, found in 30 to 60 % of samples
e (A) = abundant, found in 60 % or more of samples

Table 4. Required sample size (n) to estimate benthic macroinvertebrate mean density to within the specified accuracy of the true mean, for each combination of substrata, colonization, and site.

	Specified	**Sample Size (n) Needed: (# of samples)**			
	% Accuracy	**Basket**		**Brick**	
Site	**of Mean**	**4 wk**	**6 wk**	**4 wk**	**6 wk**
1	+/- 10	84	431	141	91
	+/- 20	21	108	35	23
	+/- 30	9	48	16	10
2	+/- 10	57	102	105	175
	+/- 20	14	26	26	44
	+/- 30	6	11	12	19
3	+/- 10	782	253	786	231
	+/- 20	196	63	197	58
	+/- 30	87	28	87	26

Appendix C

Effects of heat stress on the survival, predator avoidance ability, and physiology of juvenile fall chinook salmon

Annual Report of Research 1998

Effects of heat stress on the survival, predator avoidance ability, and physiology of juvenile fall chinook salmon

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Introduction

The Hanford Reach is the last free flowing section of the Columbia River and supports a large and important population of wild fall chinook salmon (*Oncorhynchus tshawytscha*). Recently, substantial concerns have been raised about potential losses of these fish due to river level fluctuations resulting from power peaking operations at hydroelectric facilities. Rapid fluctuations in river level can cause stranding (i.e., the trapping of fish on or beneath the unwatered substrate as a result of receding river level) of juvenile fall chinook salmon and lead to direct, rapid mortality. Indeed, such stranding has been observed in the Hanford Reach and has also been documented on other rivers during controlled drawdown experiments (Thompson 1970; Phinney 1974; Tipping et al. 1978; Woodin 1984).

Besides stranding, fish can also become entrapped in isolated potholes and backwaters that form when water rapidly recedes. Although entrapment (defined as separation of fish from the main river channel in enclosed backwater areas as a result of receding river level) may not cause direct, rapid mortality, it can lead to delayed mortality from two indirect sources--namely predation and alterations in performance caused by sublethal heat stress. Predation by birds and fish on juvenile salmon entrapped in pools has been observed on several occasions in the Hanford Reach. Such predation could be substantial in these shallow, isolated backwaters since prey have few options with which to counter the intense predation pressure. To date, however, there have been no quantitative studies addressing the magnitude of avian and piscine predation on juvenile salmon in these temporary backwaters.

Alterations in performance of fish caused by sublethal heat stress is a more insidious, yet potentially devastating, effect of entrapment. Juvenile fall chinook salmon entrapped in

isolated backwaters can be subjected to rapid (e.g., during the course of daylight hours) and extreme (e.g., >10 C above ambient temperatures) increases in temperature and then experience a relatively abrupt decrease in temperature as they are liberated back into the main river when water levels subsequently rise. The behavioral and physiological effects and potential for delayed, but direct mortality of this type of stressor on fall chinook salmon are unknown. Sublethal heat stress is known to increase the vulnerability of fish to predation (Sylvester 1972; Coutant 1973; Yocom and Edsall 1974; Deacutis 1978), but these results are likely not applicable to the situation experienced by fall chinook salmon because of species, age, and procedural differences. Sublethal heat stress also causes a variety of physiological disturbances in fish, including general stress responses (Wedemeyer 1973; Strange et al. 1977), and altered acid-base balance, enzyme activities, ionic and osmotic relations, nervous system function, and metabolic rate (Crawshaw 1977; Crawshaw 1979; Bailey et al. 1991; Soncini and Glass 1997). In addition, elevated temperatures can also elicit the rapid synthesis of a suite of proteins called stress proteins that play a role in repair and protection from environmentally induced cellular damage and may be useful as biomarkers of natural and anthropogenic environmental stress (Sanders 1993). Clearly, the heat stress experienced by juvenile fall chinook salmon during entrapment has the potential to alter behavioral and physiological mechanisms that could lead to delayed, direct or indirect mortality. An understanding of the effects of heat stress on juvenile fall chinook salmon--at several levels of organization--is necessary to fully assess the impacts of entrapment on populations of these fish in the Hanford Reach.

For the present research, we exposed juvenile fall chinook salmon to heat stress scenarios similar to those being experienced by fish in the Hanford Reach and assessed the

effects of such thermal stressors on their performance. Specifically, our objectives were to assess: (1) the extent of direct mortality attributable to various heat stress scenarios on juvenile fall chinook salmon; (2) the vulnerability to predation by smallmouth bass (*Micropterus dolomieui*) of juvenile fall chinook salmon subjected to sublethal heat stressors; and (3) some physiological effects of sublethal heat stress on juvenile fall chinook salmon, including some general stress responses, indicators of osmoregulatory function, and accumulation of stress proteins**.** In this paper, we report on our first year of research, which focused primarily on objectives 2 and 3.

Methods

Test fish.— Subyearling fall chinook salmon were collected by seine from the Handford Reach during April and transported to our laboratory in a truck equipped with a large aluminum tank and aerated water. At the time of transport, fish averaged (± SE) 38.25 ± 0.57 mm in length and 0.68 ± 0.05 g in weight. The fish were placed in 1400 L flow-through circular tanks, reared under ambient photoperiod, and were fed twice daily (at a ration of 3-4 % body weight per day) with blood worms for the first month and a commercial starter feed thereafter. Fish used in predation trials and the physiological experiment, which were conducted from July through September, averaged 76.97 ± 0.44 mm in length and 4.7 ± 0.1 g in weight.

Smallmouth bass (mean length ± SE = 311.7 ± 4.3 mm; mean weight = 505 ± 24 g) were used as predators in this study and were collected by angling from the lower Yakima River, Washington, and transported to our laboratory as previously described. They were placed in each of two 3.75-m-diameter, 1-m-deep circular tanks (N = 4 fish per tank) which were lined with gravel and cobble substrates and four pieces of 0.25-m-long, 15-cm-diameter

PVC pipe randomly scattered along the bottom to serve as cover. The predation tanks were surrounded with curtains to minimize outside disturbance and fish were held under a low light ambient photoperiod. Predators were fed a maintenance diet of juvenile chinook salmon. All prey and predator tanks received 12°C well water in which the excess dissolved gas generated by heating the water was dissipated by a packed column.

Predation trials.— Four groups of about 15-20 juvenile fall chinook salmon were serially removed by netting from their stock tanks and placed into a 19 L bucket containing 50 mg/L tricaine methanesulfonate (MS-222) and buffered with an equal amount of sodium bicarbonate. Two groups were designated as control fish and the other two groups were designated as treatment fish. We randomly selected and marked two groups (i.e., either controls or treatments) by clipping the adipose fin with surgical scissors, whereas the other groups were sham marked by simply holding them out of the water for a sufficient period. After marking, 14 fish were placed in each of two control and treatment tanks. These 0.61-m-diameter circular tanks contained 104 L of water with a 1 L/min inflow of the same water as the prey stock tanks. Prey were held in these tanks for 2-3 d before being subjected to a thermal stress.

On the day of a trial, treatment fish were subject to one of two thermal stress scenarios. The first scenario included heating the water from ambient temperature (12°C) at a rate of 2°C / h up to 26°C, whereas the second scenario included heating the water at a rate 4°C / h up to 26°C (Figure 1). Fish remained at 26°C for about 10-15 min. A 2000-Watt immersion heater placed in each tank and a temperature controller were used to achieve the heating. At completion of the heating, the immersion heaters were turned off and a 6-7°C inflow of water was used to cool water temperature down from 26°C to the starting

temperature of 12°C in 1h. Control fish did not receive any thermal stress and remained at 12°C.

Immediately after fish were exposed to the thermal stress scenarios, water level in all tanks was dropped to approximately 16 L. Fish were then released into a 19 L bucket below the tanks by removing stand pipes and opening knife gates. Fourteen each of control and treatment fish were mixed in one bucket and poured into a predator tank. Smallmouth bass were allowed to consume juvenile salmon until 50% of the prey were eaten or 2 h had past, whichever occurred first. Observations of fish behavior and estimates of the number of prey eaten were made from overhead using concealed platforms. At completion of the predation trial, the remaining prey were removed from the predator tanks by seining and placed in a lethal dose of MS-222. Lengths and weights were recorded and fish were identified as treatment or control based on the presence of an adipose fin.

Physiology experiment.— We conducted a preliminary experiment to assess some physiological responses of juvenile fall chinook salmon to the Hanford Reach-type thermal stress scenarios. At the time, we did not have enough fish to conduct our preferred experimental design and our population of test fish was experiencing what we believe to be smoltification-related mortality. Nevertheless, to obtain some initial data, we conducted the following.

Forty fish were stocked in each of the two treatment tanks, and 16 fish were stocked in each of the two control tanks. Water volumes were adjusted in the tanks to maintain similar loading densities and fish were held in these tanks for 2 d prior to applying the treatment. At 0700 on the third day, fish in treatment tanks were subjected to the thermal stressor depicted in Figure 1A; control fish remained at ambient (12°C) temperature during

the experiment. Just prior to the stressor, we removed 4 fish from each treatment tank (time = 0) and also removed a similar number of fish at the following time intervals after the start: 3.5 h, which corresponded to the 50th percentile of the temperature increase; 5 h, which corresponded to the 75th percentile of the temperature increase; 7 h, which was the time of maximum temperature; 8 h, which was the point immediately after the rapid temperature decrease; and 24 h. We removed control fish only at time = 0 and at 24 h and staggered these times by 0.5 h from the sampling times for treatment fish. All fish were placed in a lethal dose of MS-222, weighed and measured, and blood samples collected into ammonium-heparinized capillary tubes after severance of the caudal peduncle. Plasma was obtained by centrifugation and stored at -80°C for future analysis. Plasma lactate was assayed using a commercial kit (Sigma Diagnostics, St. Louis, MO) modified by us for use with microplates, cortisol was measured using an enzyme-linked immunosorbent assay (ELISA) modified from procedures described by Munro and Stabenfeldt (1984), and ions (Na^+ and Cl^-) were measured using flame photometry. After collecting blood, we excised the liver (and weighed it), a small piece of the caudal peduncle, and a whole gill arch and placed them in liquid nitrogen. These samples were transferred to an ultra-low freezer and stored at -80°C for future analysis of stress protein 70 (SP-70) expression as described by Forsyth et al. 1997.

Data analysis.— Predation data were analyzed in a manner identical to that of Mesa (1994). We first subjected data to a heterogeneity X^2 analysis to determine if the individual tests were homogenous (Sokal and Rohlf 1980). Chi-squared goodness-of-fit tests were then used on pooled data to determine if predation was random (i.e., 50:50) on treated versus

control fish. Tests where less than 30% or more than 70% of the prey were eaten were excluded from the analysis for reasons discussed in Mesa and Warren (1997).

For the physiology data, we calculated means and SE's for all plasma constituents at each sample period and plotted them over time. Within each sample period, we compared the mean of treatment fish to the overall mean of control fish using two-sample *t*-tests. The SP-70 content of liver samples were expressed in relative units as a percentage of a positive control and analyzed as just described.

Results

Predation trials

We conducted a total of 8 trials using fish exposed to the first temperature scenario (Figure 1A) with an average of 48% of fish released eaten per trial. A total of 53 control and 54 treatment fish were eaten during the trials, which did not differ significantly from random (Table 1; $P = 0.919$). Statistical power (to detect a 20% difference in predation rates) of the pooled X^2 test was 0.66. Duration of predation trials ranged from 5 min to 2 h, however the number of control and treatment fish consumed was independent of trial duration (X^2 test of independence, $P = 0.501$). There were no significant differences in length or weight of control and treatment fish during the trials (*t*-tests, $P > 0.05$).

We conducted a total of 12 trials using fish exposed to the second temperature scenario (Figure 1B) with an average of 47.5% of fish released eaten per trial. There were a total of 67 control and 82 treatment fish eaten, which again was not significantly different from random (Table 2; $P = 0.217$). Statistical power of the pooled X^2 test was 0.79. Duration of predation trials ranged from 10 min to 2 h, and again the number of control and treatment fish consumed was independent of trial duration (X^2, $P = 0.556$). There was one

trial with a significant (*t* test, $P < 0.05$), but minor, difference in length of control and treatment fish.

Physiology experiment

At the time of this writing, we had completed the analysis for plasma lactate only. Lactate levels in control fish were similar at time 0 and 24 h (Figure 2). In treatment fish, lactate levels were variable and generally increased as temperature increased. Plasma lactate concentrations differed significantly between treatment and control fish only at the 7 h sample, which was the time of maximum temperature. We will report on the other physiological variables in future reports.

Discussion

Our results indicate that juvenile fall chinook salmon exposed to our thermal stressors showed no increase in vulnerability to predation, which may at first seem a bit surprising. The thermal stressors we exposed our fish to, which were derived from field data and included a 14°C temperature increase over several hours followed by a rapid (1 h) decrease to ambient temperature, are unique stressors that, to our knowledge, have not been previously studied. These thermal stress scenarios apparently did not sufficiently compromise the many behavioral and performance factors that affect the vulnerability of prey fish to predation (see Bams 1967; Coutant 1973; Mesa et al. 1994). We noted no apparent behavioral differences in our treatment and control fish upon release to predators, and all fish responded to predation pressure by forming schools soon after attacks had started. In contrast, other research (see references in the introduction) has shown that fish exposed to acute thermal shocks were significantly more vulnerable to predation than un-shocked fish and often times displayed aberrant behavior. Thus, aside from species, age, or size differences, there appear

to be differences in the nature and severity of acute thermal shocks and the type of thermal stressor we exposed our fish to, at least with regards to predator avoidance ability. Further research is necessary to elucidate the mechanisms underlying the predator avoidance effects (or lack of) in thermally challenged fish.

There are several factors to consider in the interpretation of our predation results. First, we only tested two of potentially dozens of thermal scenarios experienced by fish in the Hanford Reach. Several factors associated with this type of thermal stressor, including the rate of temperature changes, the initial and maximum temperatures, and the time spent by fish at the maximum temperature, could affect the responses of fish. For example, the rate of temperature increase is known to affect the critical thermal maxima of fish (e.g., Elliott and Elliott 1995). Second, our results using the thermal stress scenario depicted in Figure 1A had only moderate statistical power, which reflects the probability in these tests of correctly rejecting the null hypothesis. Finally, the size of fish we used in our study was larger than that typically seen in fall chinook salmon rearing in the Hanford Reach. We purposely grew our fish to a size almost twice that of fish in the Hanford Reach to facilitate our experimental protocol. Size is known to influence aspects of the thermal biology (Baroudy and Elliott 1994), stress response physiology (Barton 1997), and predator avoidance ability (Fuiman and Magurran 1994) of fishes. Clearly, because of the considerations discussed here, more work is needed before we have a more complete understanding of the predator avoidance effects of the thermal stressors experienced by fish in the Hanford Reach.

Plasma lactate concentrations were higher (and more variable) in treatment fish relative to controls during exposure to the stressor, significantly so at the time of maximum temperature. Such elevations in plasma lactate indicate some degree of anaerobic

metabolism was occurring in fish exposed to the thermal stressor, but explanatory mechanisms are unclear. Exposure of fish to temperature change results in a variety of physiological and behavioral responses (see the Introduction), some of which could influence lactate kinetics. For example, at temperatures approaching the lethal limits for animals there could be problems with oxygen delivery to the tissues due to the low affinity of hemoglobin for oxygen at high temperatures (Schmidt-Nielsen 1991). The maximum temperature our fish were exposed to (26°C) is very close to reported lethal temperatures for chinook salmon (Brett 1952; Templeton and Coutant 1971). The addition of large amounts of lactic acid could severely disturb the acid-base balance of blood, particularly if pathways for oxidation of lactate are compromised due to extreme temperatures. In future reports, we will have more information on the physiological responses of fall chinook salmon to the Hanford Reach-type thermal stressors and will more fully discuss the effects and any possible consequences.

Our objectives for FY 1999, which will be our second year of research, include the following: (1) assess the influence of the Hanford Reach-type thermal stressors on direct and delayed mortality; (2) continue examining the predator avoidance effects of these thermal stressors with a focus on using fish of a size similar to that of fish rearing in the Hanford Reach; and (3) finish the analysis of our 1998 physiological data and continue with experiments aimed at assessing the physiological consequences of exposing fish to these thermal stressors.

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Table 1. Predation on juvenile fall chinook salmon subject to a heating of 2°C/h up to 26°C and temperature decrease within 1 h to the starting temperature of 12°C.

Replicate or statistic	Number eaten		Percent eaten	Statistics	
	Control	Treatment		df	X^2
1	7	5	43	1	0.333
2	4	6	36	1	0.400
3	11	8	68	1	0.474
4	4	7	46	1	0.818
5	10	5	54	1	1.667
6	6	8	50	1	0.286
7	5	8	46	1	0.692
8	6	7	46	1	0.077
Total	53	54		8	4.747
Pooled				1	0.009
Heterogeneity				7	4.738

Note: No predation rates differed significantly ($P \leq 0.10$) from random (50:50, treatment:control).

Table 2. Predation on juvenile fall chinook salmon subjected to a heating of 4°C/h up to 26°C and a temperature decrease within 1 h to the starting temperature of 12°C.

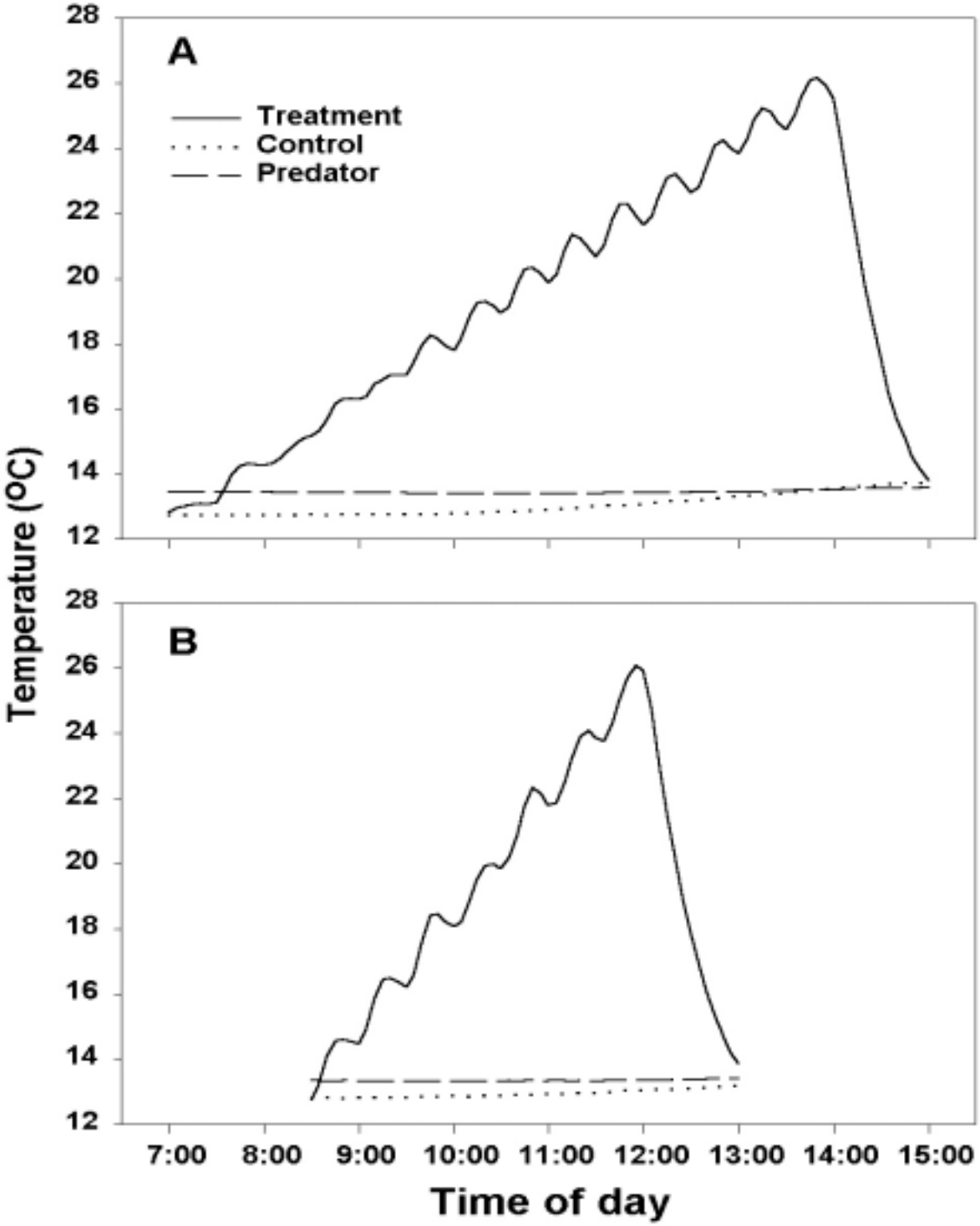
Replicate or statistic	Number eaten		Percent eaten	Statistics	
	Control	Treatment		df	X^2
1	5	4	56	1	0.111
2	5	9	50	1	1.143
3	8	6	52	1	0.286
4	8	7	53	1	0.067
5	4	5	32	1	0.111
6	5	8	46	1	0.692
7	7	10	65	1	0.529
8	5	9	50	1	1.143
9	7	4	39	1	0.818
10	6	8	58	1	0.286
11	3	6	32	1	1.000
12	4	6	38	1	0.400
Total	67	82		12	6.586
Pooled				1	1.510
Heterogeneity				11	5.076

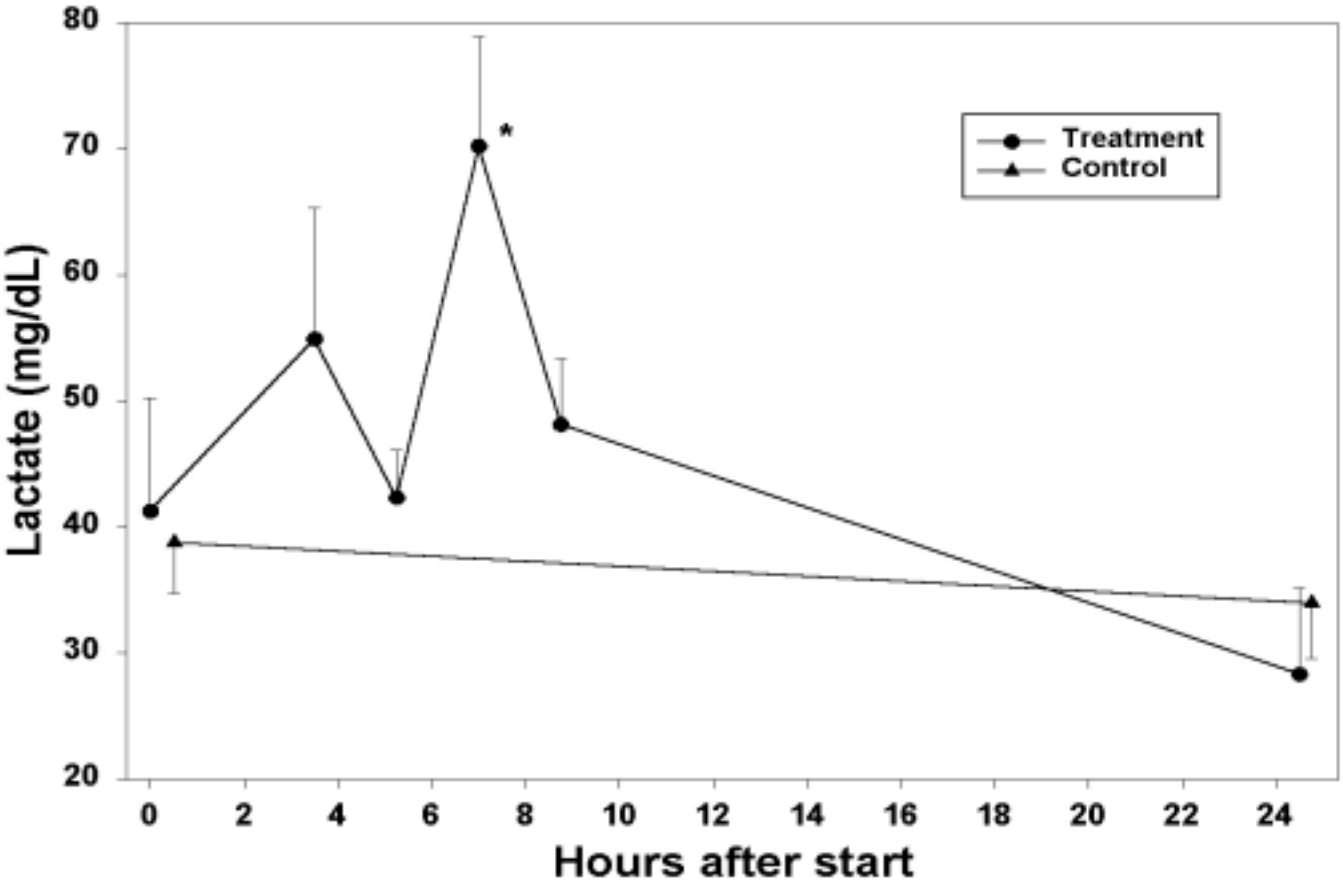
Note: No predation rates differed significantly ($P \leqq 0.10$) from random (50:50, treatment:control).

Figure Captions

Figure 1. Temperature scenarios used in our laboratory predation experiments. Treatment fish in panel A were exposed to a 2°C/h rate of temperature increase up to a maximum of 26° C followed by a decrease over 1 h to 12° C. Fish in panel B were exposed to the same scenario except at a 4°C/h rate of temperature increase. Predators and control prey remained at 12°C.

Figure 2. Mean (and SE) plasma lactate concentrations for fish at selected time intervals after they had been exposed to the temperature scenario depicted in Figure 1A and for unstressed controls. Asterisk denotes a significant difference between treatment and control fish at that time interval.


A
Treatment
Control
Predator
B
Temperature (°C)
28
26
24
22
20
18
16
14
12
7:00
8:00
9:00
10:00
11:00
12:00
13:00
14:00
15:00
Time of day


80
70
60
50
40
30
20
Lactate (mg/dL)
0
2
4
6
8
10
12
14
16
18
20
22
24
Hours after start
*
Treatment
Control