



Predation by northern pikeminnow on juvenile salmonids in The Dalles Dam tailrace: field, laboratory, and habitat modeling studies (FY2000)

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Executive Summary

Predation by resident fish is known to be a substantial cause of juvenile salmonid mortality, especially in dam tailraces and outfall locations. Conditions in The Dalles Dam tailrace are unique compared to other projects on the Columbia or Snake rivers, having a complex basin with a series of downriver islands where predators are known to reside. In May-June of 1999, northern pikeminnow and smallmouth bass were sampled in the tailrace of The Dalles Dam during periods immediately following the release of PIT-tagged juvenile salmonids for survival studies. Over twice as many smallmouth bass ($N = 101$) were collected as northern pikeminnow ($N = 40$), but none of the predators had PIT tags within their gut. A laboratory study was conducted to estimate the time required for PIT tags in juvenile salmonids to be evacuated from the gut of northern pikeminnow after consuming a tagged preyfish. Evacuation rate was sensitive to temperature, with median evacuation time being 21 h at 18 °C and 30 h at 14 °C. These results suggest that field studies to estimate predator population sizes, feeding rates, or predation on specific release groups would require considerably more effort than we allocated during 1999.

A preliminary habitat model was developed for northern pikeminnow in The Dalles Dam tailrace using radio-telemetry data collected in 1993 and 1994 (97 tagged fish). Logistic regression techniques were used in model building. Prior to modeling, independent variables, local water velocity, depth, distance to shore, and bottom substrate, were developed for each contact position of a tagged predator. Water velocity and depth for fish positions were derived from hydraulic model simulations in the tailrace at 100, 150, 200, and 250 kcfs. Model building was based on 389 observations collected over the two years. We built separate models for total river discharges of 150, 200, and 250 kcfs. The models developed here were based on observations collected when there was no spill at The Dalles Dam, since fish could not be tracked in the restricted zone of the tailrace when there was significant spill. This limitation might be removed in future model revisions.

Univariate and preliminary multivariate tests suggested that all independent variables could be used in the logistic regression models. Log-transformed water velocity, depth, and distance to shore provided the best model fits. The frequency of occurrence of northern pikeminnow generally increased with lower water velocity,

shallower depth, nearness to shore, and when bedrock was the bottom substrate. Models of habitat use generally reflected these patterns although there may be some non-linear responses to physical variables. For example, the frequency of occurrence of tagged northern pikeminnow was highest at water depths of 5-15 m, and lower at shallow (0-5 m) and deeper depths (>15 m).

Fitted equations were used in a geographic information system to predict the relative quality of northern pikeminnow habitat throughout The Dalles Dam tailrace for three flow conditions. These types of habitat models should be useful in evaluating various spill patterns (north vs south, e.g.), locating sites for bypass outfalls, or estimating the total amount of high-quality predator habitat and its response to discharge and operational parameters. Such models can be linked to hydraulic simulation models to provide 2-dimensional summaries of predator habitat. Future work will attempt to improve the northern pikeminnow models by testing some further assumptions, develop habitat models for juvenile salmonids in The Dalles Dam tailrace using recent telemetry studies, and consider the potential need for a smallmouth bass habitat model.

Introduction

Predation by resident fish is known to be a substantial cause of juvenile salmonid mortality, especially in dam tailraces and outfall locations (Rieman et al. 1991; Ward et al. 1995). In some situations, loss of juvenile salmonids to predators is thought to approach or surpass mortality from dam passage (Rieman et al. 1991), although the effects of impoundment and dam passage may contribute to some of the observed high predation rates. Behavior and habitat use by northern pikeminnow *Ptychocheilus oregonensis*, the primary predator on juvenile salmonids, may be influenced by changes in facility operations (Faler et al. 1988; Isaak and Bjorn 1996); less is known, however, about smallmouth bass *Micropterus dolomieu* behavior in near-dam areas. Predators are opportunistic and respond to changes in the availability and abundance of juvenile salmonids as prey items (Petersen and DeAngelis 1992; Shively et al. 1994). Thus, if dam operations concentrate or direct juvenile salmonids into eddies, low velocity areas, or near structures occupied by predators (see Martinelli and Shively 1997; Shively et al. 1996; Shively et al. 1996), mortality may be elevated beyond that incurred through other dam passage routes.

To more effectively pass juvenile salmonids at dams and thereby increase survival through the hydroelectric system, the NMFS Biological Opinion mandated increased proportions of total river discharge be passed as spill. However, recent juvenile salmonid survival studies conducted at The Dalles Dam (**Figure 1**) suggested that juvenile salmonid survival at this dam may be lower at higher levels (64%) of spill than at lower levels (30%) of spill (Dawley et al. 1998). Relative survival in that study was evaluated through the release of PIT-tagged study fish (treatment and reference groups) at The Dalles Dam and their later detection at Bonneville Dam. Reference groups were released 0.7 km downstream of the dam in a high velocity area away from turbulent flow and areas thought to harbor large numbers of aquatic predators (Dawley et al. 1998). Thus, mortality of tagged fish in the treatment groups may be directly due to physical trauma incurred in the immediate tailrace area or from other sources, such as predation in the near-dam environment (tailrace or downstream island areas).

Conditions in The Dalles Dam tailrace and the area immediately downstream of the tailrace are relatively unique compared to other projects in the basin. The river

environment in the tailrace area is relatively shallow with armored bedrock substrate and frequently has extremely turbulent flow patterns, while downstream of the project there are a number of bedrock islands and an adjacent slough-like habitat on the south side (Oregon shore) of the river. At high flows or high percentages of spill, juvenile salmonids may be redirected into complex hydraulic areas in the tailrace, and also into island habitats on the Oregon shoreline (Shively et al. 1996) where their vulnerability to predators such as northern pikeminnow and smallmouth bass may be higher. Because of this, we hypothesized that predation may be an important component of the lower relative survival observed at higher levels of spill by Dawley et al. (1998) at The Dalles Dam.

In this report, we conducted two studies to begin further exploration of predation on juvenile salmonids in The Dalles Dam tailrace. First, we sampled predators in the tailrace and tested for the presence of PIT tags following NMFS releases of test groups. The objective of this work was to see if predators collected in different areas of the tailrace might have ingested different proportions of control versus release groups of tagged salmonids. As part of this field study, we conducted a laboratory experiment on the evacuation rates of PIT tags from northern pikeminnow. Knowledge of the rate of PIT tag evacuation from predators would be necessary to assure that our field sampling effort had not missed a significant number of tags that were eaten and evacuated prior to our collection of predators. Knowing the evacuation rate, and some of the factors that control this rate, would also be useful in future studies.

In the second component of this study, we used historical radio-telemetry data from northern pikeminnow in The Dalles Dam tailrace to begin development of a habitat model. The habitat model for northern pikeminnow distribution should be useful in predicting how predators respond to varying levels of spill, spill from north versus south routes, and what areas have potential value for bypass outfall location. Model results may also be used to begin development of mechanistic relationships between predator behavior and the distribution of physical characteristics in the tailrace, which are produced by project operations.

The three components of this work (field collection of predators, laboratory PIT tag study, and habitat modeling) are summarized separately below, followed by a general discussion.

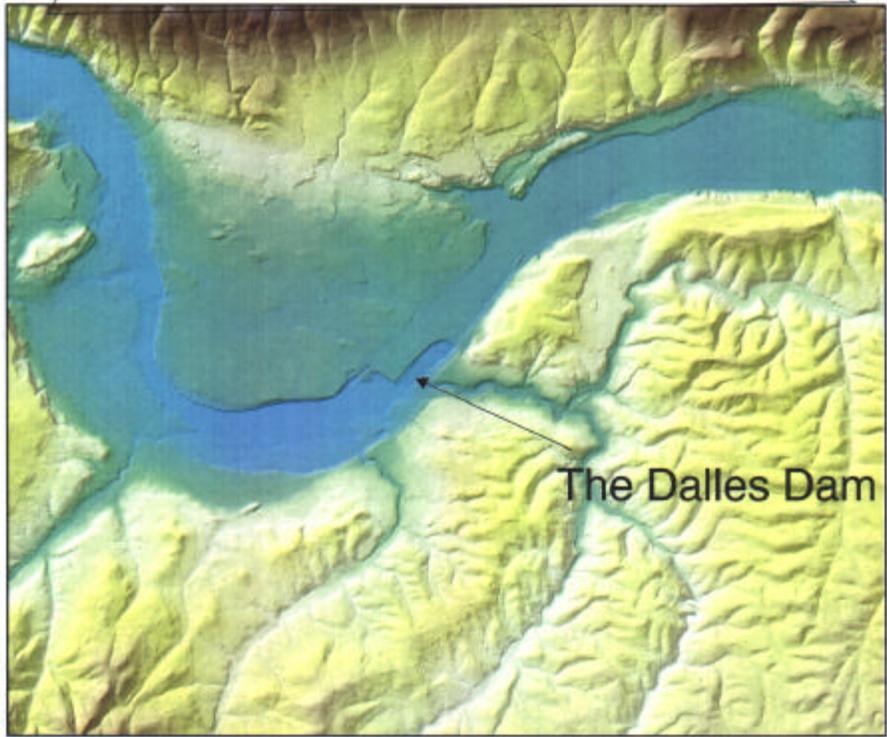
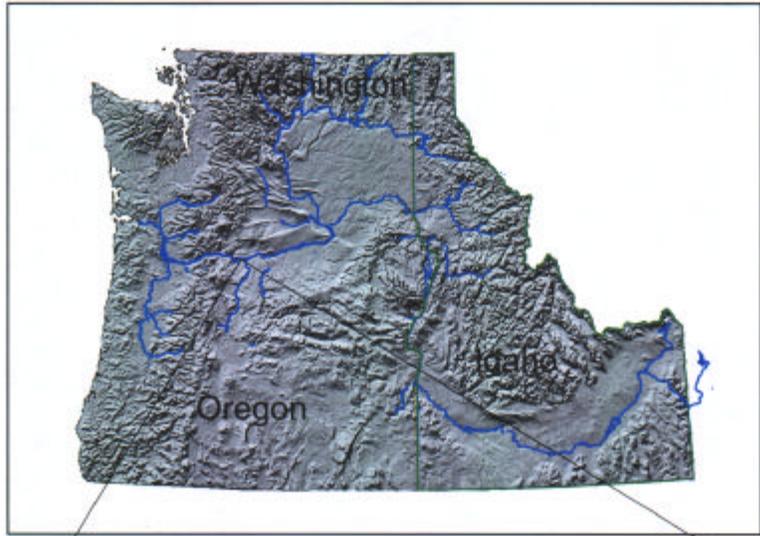


Figure 1. Location of The Dalles Dam.

Field sampling of predators

In this portion of the study, we collected data on the relative abundance of predators (northern pikeminnow and smallmouth bass) in specific areas of The Dalles Dam tailrace, and the number of PIT-tagged juvenile salmon consumed by predators.

Methods

We used boat electrofishing in the bridge and basin islands area below The Dalles Dam to collect predators (smallmouth bass and northern pikeminnow). Sampling was conducted primarily in May and June on days or nights following PIT-tag releases (Earl Dawley, NMFS). Sampling was conducted along transects that matched earlier efforts in The Dalles Dam tailrace (see Appendix 1). Sampling included nearshore zones along the north shoreline, south shoreline, outer island area, and inner island area. For each period of effort (day or night; ~8 hrs), we anticipated completing about six to ten 15-min electrofishing runs (standard effort; Ward et al. 1995, e.g).

Transects start and stop locations were assigned prior to the field work. Transects were haphazardly selected prior to beginning a sampling effort. For each transect, electroshocking began at the upriver end of the transect and the boat was moved downriver along the shoreline for approximately 15 minutes. This duration of effort coincides with other predator sampling that has been conducted in the Columbia and Snake rivers (e.g., Ward et al. 1995). Predator catches were standardized to 15-min effort (CPUE).

Northern pikeminnow and smallmouth bass were netted and placed in a live-well until processed. Each fish was scanned for the presence of a PIT tag in the predator's gut. Preliminary work with PIT-tagged salmonids indicated that PIT tags could be effectively detected in predator guts when gain on the reader was increased. Predators were weighed (nearest 10 g), fork length FL was measured (nearest mm), and marked with a floy tag to allow mark-recapture abundance estimates. Tagging was coordinated with ODFW personnel, who were tagging northern pikeminnow in Bonneville Reservoir (D. Ward, ODFW). Tagged predators were returned to the river near the location where they had been collected.

Results

Equipment problems with the electroshocking boat in early June prevented us from sampling in July and August. We sampled 41 transects in May and 8 transects in June. Seven transect samples yielded no predators, northern pikeminnow were collected in 13 transects, and smallmouth bass were collected along 29 transects. Total sampling effort was about 735 minutes. Effort was about evenly divided between the Washington shore transects ($N = 22$) and the Oregon shore transects ($N = 27$).

Over twice as many smallmouth bass ($N = 101$) were collected as northern pikeminnow ($N = 40$). Over 76% of northern pikeminnow were collected as single individuals in a transect, whereas smallmouth bass catches ranged from 1 per transect to 19 in a transect (Figure 2). Northern pikeminnow were slightly larger on average than smallmouth bass in both mass and length (Table 1), although these differences were not significant (t -tests; $P > 0.4$).

A two-way ANOVA suggested that catch-per-unit of effort (CPUE; #/15-min) was significantly higher for smallmouth bass than for northern pikeminnow (Table 2; $P < 0.01$; \log_e transformed CPUE), but there was only a marginal difference between the Oregon and Washington shores (Table 2; $P = 0.07$). There was no obvious diel pattern to the CPUE of either species; smallmouth bass were captured during 13 different hourly periods and northern pikeminnow were collected in 8 different hourly periods.

Table 1 Size of northern pikeminnow and smallmouth bass collected in The Dalles Dam tailrace, 2000.

Species	Weight (g)			Length (FL, mm)		
	Average	SD	N	Average	SD	N
Northern pikeminnow	564	595	20	313	105	20
Smallmouth bass	479	363	101	293	73	101

Table 2 Average CPUE of northern pikeminnow and smallmouth bass collected in The Dalles Dam tailrace, 2000.

Species	All transects		Oregon shore transects		Washington shore transects	
	Average (SE)	N	Average (SE)	N	Average (SE)	N
Northern pikeminnow	1.5 (0.3)	13	1.3 (0.3)	4	1.7 (0.5)	9
Smallmouth bass	3.6 (0.7)	29	3.2 (1.1)	16	4.1 (0.7)	13

Fifteen of the 20 captured northern pikeminnow were captured along the Washington shore in four transects (230070, 230080, 230090, 230100). Four pikeminnow were collected at transect 230300 along the edge of a downstream island. Smallmouth bass were collected in almost equal numbers along the Washington (50/101) and Oregon shores (51/101). Twenty bass were collected at each of two transects – 230090 on the Washington shore, and 230200, which is at the upper end of the basin islands (Figure 2).

Nine of the tagged smallmouth bass were recaptured, all but one by anglers. With 101 marked smallmouth bass available, a Petersen estimate of the local smallmouth bass population was 1,040. A Schnabel estimate of the smallmouth bass population was 1,995 fish. No northern pikeminnow were recaptured, although we marked only 20 individuals.

All predators collected were scanned multiple times with a PIT tag detector, but no PIT tags were detected in the predators. Sampling generally began about 5 h after release of PIT tag groups and continued for 6-8 h.

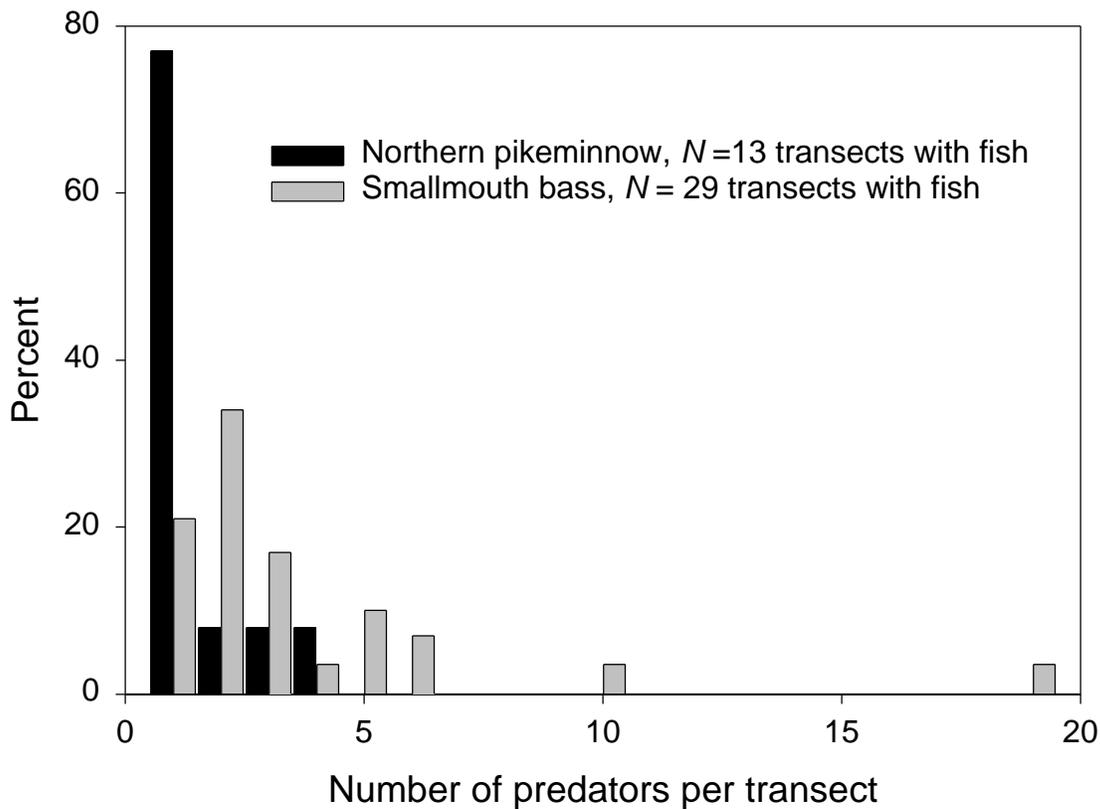


Figure 2. Frequency distributions of the number of northern pikeminnow and smallmouth bass collected per transect sample in The Dalles Dam tailrace, 2000.

Evacuation of PIT tags from northern pikeminnow

Passive Integrated Transponder (PIT) tags consist of an integrated circuit chip, capacitor, and antenna coil encased in glass (Prentice et al. 1990). These tags are implanted within the peritoneal cavity of fish and transmit a unique signal to a reader. PIT tags have become increasingly common during the last decade and are used in studies of fish movement, habitat use, and predation risk (e.g., Utne et al. 1997; MacKenzie and Greenberg 1998; Skalski et al. 1998). Because of their small size (0.07 g; 10 x 2 mm) and internal placement in fish, PIT tags may be especially useful to mark prey in predation experiments since predators are often attracted to an external tag.

PIT tags have been used in various streams and rivers to estimate reach survival, migration rates, adult return patterns, and predation losses in various streams and rivers

(e.g., Faengstam et al. 1993; MacKenzie and Greenberg 1998; Zabel et al. 1998). In the Columbia River Basin (USA), PIT tags are used extensively, and over 6.6 million salmonids have been tagged and released since 1987 (PITAGIS website; <http://www.psmfc.org/pittag/>). Since 1998, over 1 million salmonids were tagged yearly, with a peak of 1.5 million tagged fish in 1999.

Northern pikeminnow prey on juvenile salmonids in western North American, and are a primary source of mortality for downstream migrating salmonids in the Columbia River (Ward et al. 1995). The occurrence of PIT-tagged prey within the gut of northern pikeminnow could be used to identify the species, release location, original size of a preyfish, and the passage history through dams with PIT tag detectors (Skalski et al. 1998). Over 1 million northern pikeminnow have been collected in a predator management program (Friesen and Ward 1999), but predators were not checked for the presence of PIT tags.

We conducted a laboratory experiment to determine how PIT tags in juvenile salmonids that are consumed by northern pikeminnow are evacuated in relation to predator size, temperature, prey size, and number of preyfish eaten in a meal. We compared evacuation times among test groups and developed simple models that can be used to predict the time it takes for a tag to be evacuated and the probability of tags remaining in predator guts over time. Information on the evacuation rates of PIT tags from predators can be used in designing field or laboratory experiments, or in evaluating the presence of PIT tags in captured predators.

The time necessary to evacuate a PIT tag from a predator could be used to determine whether tagged fish (NMFS studies) might be recaptured in The Dalles Dam tailrace. If PIT tags are evacuated fairly slowly, for example, then recovery of PIT tags might in predator guts might be used to demonstrate the source of salmonids eaten, and thus their relative vulnerability through different passage routes. On the other hand, if PIT tags are rapidly evacuated from predators, then only those predators preying on salmonids very near the dam would contain tags, and it would be less likely that passage vulnerability could be determined.

Methods

Predators were captured by electrofishing (400 Volts pulsed DC, 4 amps) on 12 July below Bonneville Dam and placed in five-foot circular tanks with an inflow rate of 7.5 liters/min at 18 °C. Each predator was weighed (g), measured (fork length, mm), and tagged with a uniquely marked dart tag. Predators also received fin clips in case of tag loss during handling and force feeding. Northern pikeminnow were fed a maintenance diet of juvenile chinook salmon and acclimated to laboratory conditions for a minimum of 2 weeks.

We conducted trials at two target temperatures, 18 and 14 °C (Table 3). Temperatures were measured daily and inflow and water quality parameters were monitored weekly. Temperatures varied slightly over a diel period, but were always within $\pm 0.5^\circ\text{C}$, and usually were within $\pm 0.2^\circ\text{C}$ of the target temperature (Table 3).

Juvenile chinook salmon used as prey were measured (mm; FL) and weighed (0.1 g). All prey were individually marked with PIT tags, which are approximately 12 x 2 mm.

After a 72 h period of starvation (Beyer et al. 1988), six predators in each of two tanks were force-fed a meal consisting of one or two PIT-tagged juvenile salmonids (Table 3). Predators were netted and held in a surgical cradle using foam moistened with artificial fish slime (Stress Coat) to reduce injury to the fish. Prey were killed by physical injury, to avoid any possible effects of chemicals on northern pikeminnow digestion rates. The salmonid meal was force fed to predators with a 60 CC or smaller (30 CC) syringe (Vondracek 1987). Predators were usually out of water less than 2 minutes for feeding. Time of feeding was recorded and predators were placed back into the tank.

We estimated the approximate time when PIT tags might be expected to occur in the tank, and started hourly sweeps of the tank at that time. Sweeps were conducted with a magnet moved slowly around the floor of the tank to minimize disturbing the fish. PIT-tag recovery time was the mid point of the interval during which the PIT tag was recovered. For example if a tag were recovered at 11:00 then the tag recovery time for analytical purposes was 10:30 if the scanning interval were one hour.

Experiments were conducted between August 2 and October 11, 2000. Predators were replaced in one tank on August 23 due to a fungal infection. Time between trials

was about 7 d. For trials at the second temperature (14 °C), we acclimated predators at a rate of 1 °C per day down to the desired temperature.

Data analysis – We first examined frequency distributions of tag evacuation times and compared two- versus one-prey results. Second, we fit a simple regression model for predicting evacuation times of PIT tags. Finally, we used a power function (dos Santos and Jobling 1991) to fit a model of the probability of tag evacuation at a given time after ingestion.

Table 3. Dates and conditions for evacuation experiments with northern pikeminnow.

Start date (2000)	Run	Temperature (°C)	Prey per predator
2 August	1	17.9	1
7 August	2	18.2	1
16 August	3	18.1	2
29 August	4	18.1	2
27 September	5	14.1	1
3 October	6	14.0	1
10 October	7	14.0	1

Results

Predators used in experiments ranged in size from 246 to 1260 g (mean 737 g; SD 283 g; N = 25), and were from 288 to 485 mm FL (mean 399 mm; SD 53 mm; N=25). Of 153 preyfish tagged and fed to predators, 3 were regurgitated, 13 tags were evacuated prior to our sampling, and 1 was retained past our sampling interval. We could not assign an accurate time to these observations where tags were not collected during hourly monitoring so these observations were excluded from most analyses below. To test how these missed observations might have influenced model parameters, we assigned uniform random times for recovery between 2 and 10 h, our minimum observed recovery time.

Of the 13 tags that were evacuated very rapidly, 7 were in particularly small prey used during the two-prey trials (Runs 3 and 4; Table 3). One tag from Run 1 was retained for about 5 days, well past out 60-h sample period. Juvenile salmonids used as prey ranged in size from 2.9 to 19.4 g (mean 10.4 g; SD 5.0 g; N=153) and were from 67 to 130 mm FL (mean 99 mm ; SD 18; N=153).

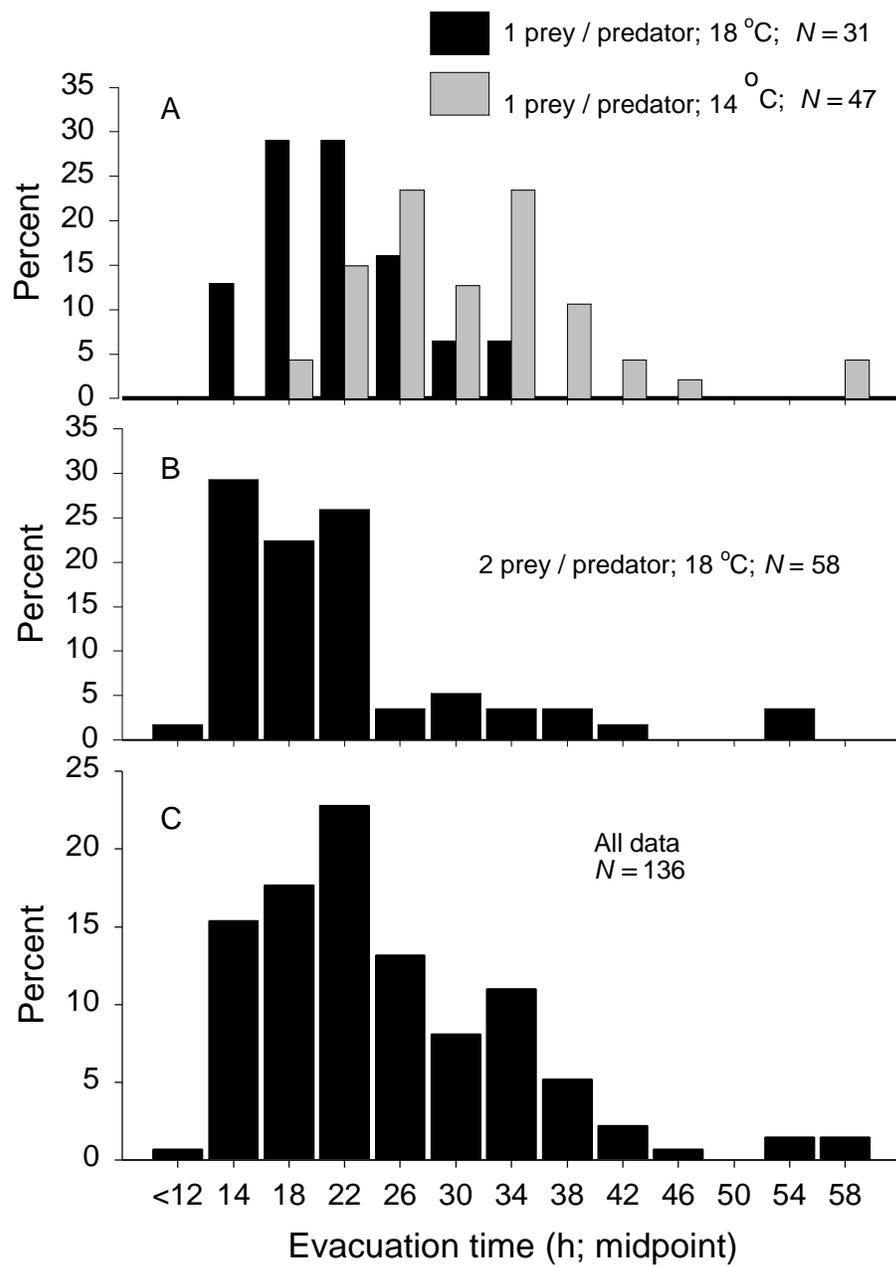


Figure 3. Evacuation times of PIT tags from northern pikeminnow in the laboratory. Predators were fed either one prey (A) or two prey (B), and experiments were run at 14 and 18 °C (A). Panel C shows all data pooled.

Two- versus one-prey meals

We compared the effects of number of prey on evacuation rates in runs 1-4, all conducted at 18 °C (Table 3). In these trials, we were comparing number of prey eaten and not truly variations in total meal mass (see e.g. Beyer et al. 1988). Since the preyfish we used in the two-prey trials averaged only 5.7 g, the mass of prey in the one-prey trials was significantly greater (~5 g; Table 4) than the mass in the two-prey meal (*t*-test; $P < 0.01$).

The frequency distributions of evacuation times for two- versus one-prey meals were similar (Figure 3). Median evacuation times differed by 0.3 h (1-prey median = 22.2 h; 2-prey median= 21.9 h) and mean evacuation times differed by less than one h (Table 4). Mean evacuation times for one- versus two-prey meals at 18 °C were not significantly different (\log_{10} transformed times; *t*-test; $P > 0.50$). Due to this lack of a difference in evacuation times, we pooled the one- and two-prey data in analyses below.

Table 4. Individual salmonids size, meal size, and evacuation times in experiments with a one-prey meal ($N = 31$) versus a two-prey meal ($N = 58$). All results below were from experiments at 18 °C.

	One-prey meal	Two-prey meal
	Average (SD)	Average (SD)
Salmonid mass (g)	16.1 (1.7)	5.7 (1.5)
Meal mass (g)	16.1 (1.7)	11.3 (2.9)
Evacuation time (h)	22.2 (5.4)	21.9 (9.2)

General patterns

All frequency distributions of tag evacuation times were well-fit by lognormal distributions ($P > 0.15$ for 14 °C data, 18 °C data, and all observations combined; Kolmogorov-Smirnov D).

Evacuation time was negatively correlated with predator size (Pearson correlation coefficient $r = -0.19$; $P < 0.03$; $N = 136$), with smaller predators having somewhat shorter evacuation times. Temperature seemed to have a strong effect on the frequency distributions of evacuation times (Figure 3). Median evacuation time was 21 h for all prey at 18 °C and 30 h at 14 °C (Figure 3). Mean evacuation times were significantly different for the two temperature groups (t -test; $P < 0.01$). Evacuation time was weakly correlated with prey size ($r = 0.19$; $P < 0.03$; $N = 136$), suggesting tags from larger prey took longer to pass through the predators.

There appeared to be no simple relationship between 90% evacuation times (ET90) predicted by the Beyer et al. (1988) equation and the evacuation time for a PIT tag. Linear regressions between ET90 and evacuation time for the two temperature groups were not significant ($P > 0.2$), and plots of these data suggested no obvious patterns.

Predicting PIT tag evacuation time and probabilities

Based on the above results, we first developed an empirical regression model for predicting the evacuation time of PIT tags using temperature, predator size, and prey size as independent variables. Because evacuation time distributions were skewed, we fit independent variables to \log_e transformed times. Predator weight and prey weight were also transformed to improve normality. Residual plots were examined to evaluate the fit for each independent variable.

The fitted model was highly significant ($P < 0.001$) and parameters are given in Table 5. The model for PIT tag evacuation time t was:

$$t = \exp(a + B * \text{temperature} + C * \log_e(\text{predator mass}) + D * \log_e(\text{prey mass})) \quad r^2 = 32\%.$$

Table 5. Model for predicting the evacuation time ($\log_e(\text{hours})$) of PIT tags from northern pikeminnow as a function of temperature, predator size, and prey size.

Parameter	Parameter Estimate	SE	<i>P</i>
Intercept (a)	2.290	0.197	<.0001
B(Temperature)	-0.035	0.007	<.0001
C($\log_e(\text{prey mass})$)	0.112	0.057	.05
D($\log_e(\text{predator mass})$)	-0.154	0.065	.02

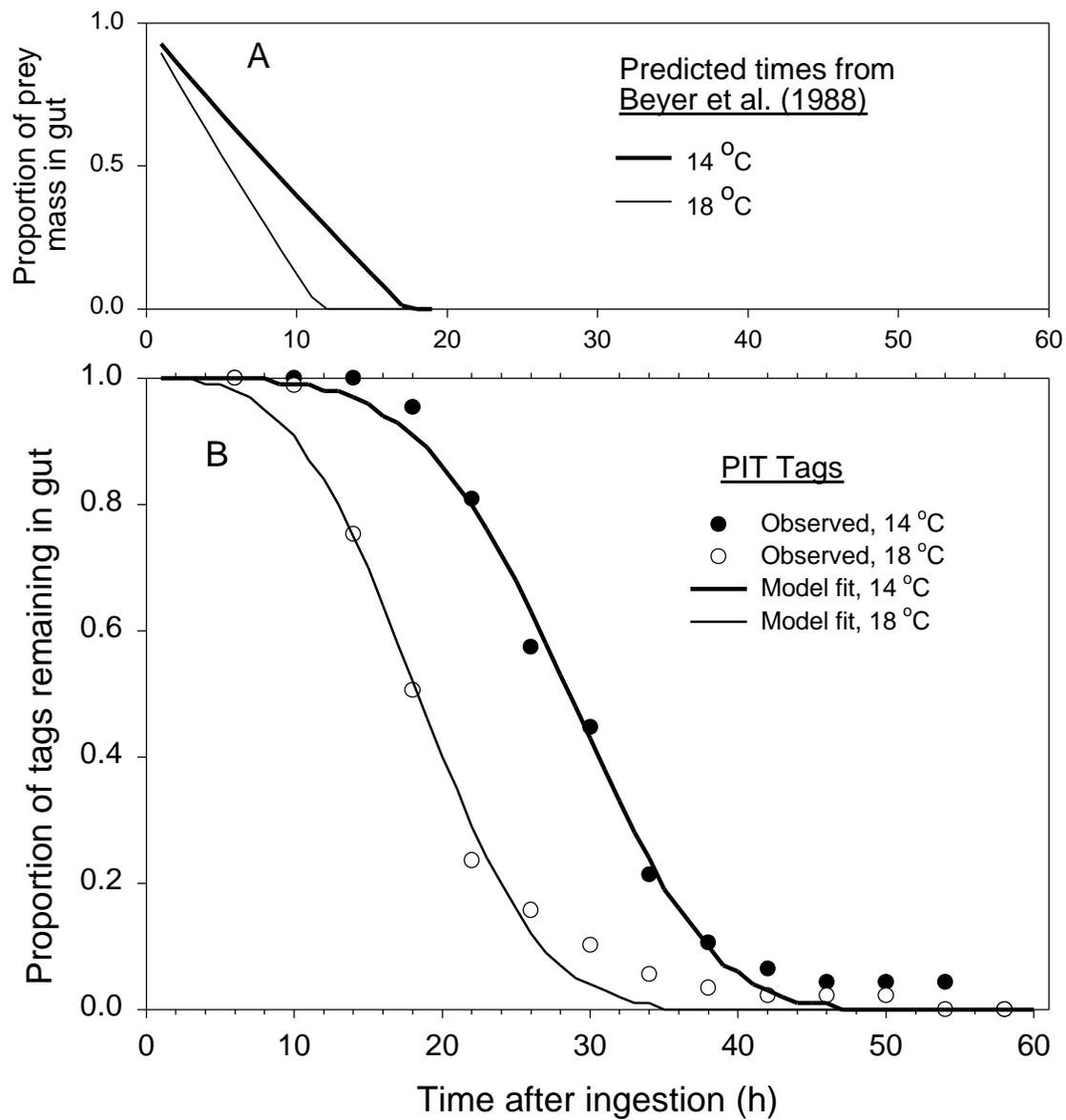


Figure 4. Timing of prey digestion and PIT tag evacuation in northern pikeminnow. Panel A shows predicted evacuation of a 16 g salmonid prey from a 737 g predator at two temperatures using the model of Beyer et al. (1988). Panel B shows the observed and predicted probabilities of PIT tags remaining in a northern pikeminnow gut at a given time at two temperatures. Model prediction in panel B are based on equation 2 in the text.

We also modeled the probability of PIT tags remaining within the gut of a northern pikeminnow. The laboratory observations suggested that tag evacuation may occur in two, somewhat distinct, phases: a delay/digestion phase followed by a tag evacuation phase. During the delay period, prey fish are being digested and the embedded tags are not highly susceptible to evacuation. This is suggested by the relatively long period before tags begin to show up in significant number (Figure 3), and also by studies by dos Santos and Jobling (1991). The differences between the 14 and 18 C experiments suggest higher temperatures may decrease this delay period (Figure a1) since digestion rates are temperature dependent (Beyer et al. 1988). Following the digestion period, the majority of tags were evacuated in about 10-15 h, however, some tags being retained for fairly long periods (Figure 3).

To model the probability of tag evacuation we used a power function. dos Santos and Jobling (1991) modeled the mass W of prey in cod *Gadus morhua* evacuated at time t using a power function as:

$$W_t = W_0 \cdot 2^{-(t/H)^S} \quad (1)$$

where W_0 is the initial prey mass, S controls the shape of the function, and H is a measure of the half-life (h) of a meal. This is a very flexible form and can describe a variety of types of evacuation depending upon the value of S . For our case in particular, if $S > 1$, the function can describe a process with an initial delay in tag evacuation (dos Santos and Jobling 1991).

The probability that a PIT tag would remain in a predator gut at time t P_{tag_t} was assumed to be analogous to W_t/W_0 in the dos Santos and Jobling formulation:

$$P_{tag_t} = 2^{-(t/H)^S} \quad (2)$$

We use this form because it has been shown to describe gut evacuation processes in a variety of cases (Elashoff et al. 1982; dos Santos and Jobling 1991).

We fit equation 2 separately to the 14 and 18 °C laboratory data. P_{tag_t} was the proportion of all PIT tags in an experiment that remained in predator guts at 4-h intervals. Model parameters were fit with a nonlinear, least squares program (SAS PROC NLIN).

The modified power function provided a good fit the data, for both the 14 and 18 °C experiments (Figure 4). The shape parameter S was much greater than 1 (Table a4), suggesting a delay period followed by an exponential rate of tag evacuation (exponential decay). Parameter S was similar between the two temperature groups (Table 6). The overlap of S estimates and confidence intervals suggested tag evacuation is largely independent of temperature following the delay period. The half-life of tag evacuation H was different between temperature groups, which was mainly a difference in the initial delay (Figure 4). It is interesting to note that the difference between the half-life of tags between temperature groups (~10 h; Table 6) is almost identical to the difference between predicted ET90 time for these groups (~9 h; Figure 4). This ET90 is a function of prey size, predator size, meal size, and temperature (Beyer et al. 1988).

The probability (or proportion in Figure 4) of tag retention was accurately predicted except for some tags that were retained somewhat longer than was predicted by the model (Figure 4).

Table 6. Model parameters for predicting the probability that a PIT tag remains in a northern pikeminnow gut within a number of hours (t , time in the model). Models were fit to 14 °C and 18 °C results separately. The models were fit to 15 data points (Figure 4).

Parameter	14 °C		18 °C	
	Parameter estimate	?95% Confidence interval	Parameter Estimate	?95% Confidence interval
S	4.3	3.7, 4.9	3.2	2.6, 3.9
H	28.6	27.8, 29.3	18.4	17.5, 19.3

Discussion

PIT tags ingested by northern pikeminnow were retained for relatively long periods compared to estimated digestion times of preyfish mass, and the process of tag evacuation can be predicted using predator size, prey size, and temperature. The relatively long delay before significant numbers of tags were observed in the tanks, followed by the exponential evacuation of tags, suggests that the preyfish mass, or its body wall, had to be largely digested before tags were especially susceptible to evacuation. The power function that we used for modeling the probability of tag retention worked well and also provided clues about how evacuation occurs.

dos Santos and Jobling (1991) studied the rates of evacuation of small pieces of wire, plastic, or glass from cod, and observed patterns similar to ours. For cod fed a single meal, large plastic beads (5 mm) showed a relatively slow rate of evacuation compared to evacuation of the herring meal. Dos Santos and Jobling (1991) also fit a power function to their data. The half-life (H) of plastic beads was >300 h and the shape parameter (S) was >1 (1.2) suggesting a delay period. Two-mm beads passed through the gut more rapidly than the large beads, but still slower than digested herring. Cod experiments were conducted at 5.0 – 7.2 °C compared to 14 – 18 °C for our experiments with northern pikeminnow, which would partially explain the longer retention times for particles in cod compared to PIT tags in northern pikeminnow.

Petersen and DeAngelis (1992) used the presence of diagnostic bones, prey mass, and digestion rates to develop a model of feeding behavior for northern pikeminnow eating salmonids. They concluded that predators were likely eating salmonids during brief feeding bouts. An assumption of their model was that diagnostic bones, mainly dentaries and cleithra of preyfish, were retained in the predator's gut for a period of time beyond the time required for 90% digestion of the prey mass. PIT tags have about the same mass as these diagnostic bones, so our results could be used to corroborate their assumption and improve this type of model.

About 8% of the tags in our experiments were evacuated prior to the start of our sampling effort (~6 h after feeding), and we could not use these untimed tag evacuations in our analyses. Thus, the frequency distributions and model parameter estimates are

slightly biased, however, these biases are not large. For example, the mean tag retention time for the 18 °C experiments was 22.0 h ($N = 89$), and this mean would have been 20.7 h ($N = 98$) if we included the untimed tag observation and assigned them random evacuation times between 1 and 6 h. Inclusion of the untimed observations in the regression and probability models, and assuming random time distributions for the untimed observations, caused parameter estimates to vary less than 15% (results not presented). The only parameter that seemed especially sensitive to these untimed observations was the prey weight parameter in the regression model, which increased ~50% with the added observations. Tags evacuated early came from somewhat smaller prey than for the data used to fit the models (8.3 g versus 10.6 g, respectively;), but this size difference was not significant (t -test; $P = 0.1$).

An application of these results could be to predict the times that PIT tags might be expected to be in predators in a specific area, which could guide sampling efforts for predators. For example, with a predator population that averages 1000 g feeding on 10-g salmonids at 18 °C, the half-life of PIT tags in predator guts would be ~18 h, and 90% of the tags would be evacuated within ~27 h. As the temperature declines, predator size increases, or prey size increases, tag retention time increases since it requires longer for complete digestion of the prey body. The equations given here can be used to make approximate estimates of these times.

Habitat modeling of northern pikeminnow

Field methods

We used radio-tag data of northern pikeminnow tracked during 1993 and 1994 in The Dalles Dam tailrace to develop habitat models. Predators were tracked between May-September (1994) or May-December (1993), but we limited our dataset to observations from May-July for each year since we were primarily interested in predator behavior during the juvenile salmonid outmigration. Predator tagging, tracking, and positional methods are summarized in (Martinelli and Shively 1997). Fish were located by boat tracking in both the boat-restricted zone (BRZ) and further downriver in the

tailrace, which we will call the non-restricted zone (NRZ). Fish were generally located to within ~5 m (Martinelli and Shively 1997).

A preliminary analysis of the 1993 and 1994 telemetry dataset (May-July) showed that 54% (461 of 853) observations were collected during periods when spill was <2.0 kcfs at The Dalles Dam. During periods with little spill, boat tracking can be conducted in both the BRZ and the NRZ, increasing the spatial coverage and sample size. For those observations when spill was ≥ 2.0 kcfs (392 observations), spill ranged from 10.5 kcfs to 217 kcfs and averaged 72.9 kcfs. For modeling reported here, we used only observations collected when there was <2.0 kcfs spill. During 1993 and 1994, water was released through the northern-most spill bays when spill <2.0 kcfs, as an adult attractant flow.

Fish were tracked across a wide variety of flows at The Dalles Dam, all of which could not be simulated. Hydraulic simulations of flow in The Dalles Dam tailrace were conducted by PNNL (Rakowski and Richmond 2000) at 100, 150, 200, and 250 kcfs. We pre-selected these model runs based on the frequency of occurrence of fish observations for 1993 and 1994.

Model Theory and Development Protocol

We used logistic regression techniques to develop resource selection models for northern pikeminnow in The Dalles Dam tailrace (Manly et al. 1993; Hosmer and Lemeshow 2000). This approach uses independent physical variables in the tailrace environment to model the probability of occurrence of northern pikeminnow under certain conditions. Probabilities are predicted with fitted logistic regression models. The basic form of the regression model is:

$$P_x = \frac{\exp(\beta_0 + \beta_1 x_1 + \dots + \beta_n x_n)}{1 + \exp(\beta_0 + \beta_1 x_1 + \dots + \beta_n x_n)} \quad \text{(Equation 1)}$$

where P_x is the probability that a predator will use a particular location with x_1, \dots, x_n physical properties, called independent variables. Fitting these resource selection models requires data on the types of habitats that are used and unused during the sample period (Manly et al. 1993).

For model building, we followed explanatory variable selection protocols and model assessment procedures detailed in Hosmer and Lemeshow (1989, 2000). Although it is desirable to model resource selection for each individual fish (B. Manly, personal communication), we pooled habitat measures across individuals for analysis because of the relatively low number of data points per individual. Model building was conducted in three phases (Hosmer and Lemeshow 2000): 1) univariate data analysis; 2) multivariate modeling with all candidate variables (from the univariate analysis); and 3) final model building. Only variables that differed significantly between used and random habitat points were included in resource selection models. We first assessed the importance of each continuous explanatory variable by building univariate logistic regression models using conservative (< 0.25) P values as suggested by Hosmer and Lemeshow (2000). Discrete variables (substrate type) was evaluated for initial inclusion in the resource selection analysis based on the likelihood ratio chi-square tests. We then began development of multivariate models, striving for the simplest (fewest parameters), biologically plausible models possible. After independent variables were removed ($P > 0.10$) from the initial full multivariate models, we compared the fit of each successive model to the former model using likelihood ratio tests. We also examined variable coefficients and standard errors to determine if there were large changes in estimates between the full models and subsequent subset models (Hosmer and Lemeshow 2000). Odds ratios were computed and used to assist in model interpretation (Hosmer and Lemeshow 2000). Based on preliminary testing, continuous independent variables were log transformed [$\log_{10}(x + 1)$] prior to model building.

Independent Variables

From earlier studies in The Dalles Dam tailrace and elsewhere (Shively et al. 1996; Martinelli and Shively 1997; Isaak and Bjornn 1996; Petersen et al. 2000), we hypothesized that water velocity, water depth, distance to the nearest shore or structure, and the type of bottom substrate would likely be important to northern pikeminnow. For each location record of a radio-tagged fish, these independent variables were estimated indirectly using hydraulic model simulations (Rakowski and Richmond 2000) or bottom substrate maps (USGS, unpublished data).

To estimate the conditions for each predator contact, fish locations were plotted with GIS (ArcView) on substrate coverages or coverages derived from hydraulic simulations. A general substrate map has been developed with USGS data and includes categories from mud to bedrock. The Battelle Pacific Northwest National Laboratory (PNNL) provided us with data from 2-dimensional hydraulic model simulations of The Dalles Dam tailrace (C. Rawkowski; Rawkoski and Richmond 2000). PNNL conducted four model runs and provided us with four GIS point coverages for our analyses. Each run represented a different total flow through the turbines only (no spill; see Rakowski and Richmond 2000). We requested four simulations only since this was considered a preliminary effort at model building; additional simulations may be added at a later date. Point coverages were converted to grids of velocity, depth, and distance to shore for each discharge scenario. Grid cell size was 6.1 x 6.1 m, which was approximately the range of accuracy for detecting tagged northern pikeminnow (Martinelli and Shively 1997), and matched the range of points in the runs provided. Average velocity in a cell was computed using directional vectors. Distance to shore was computed using a function in ArcView GRID, after some improvements to the island coverage files.

For model development, we selected fish observations collected on days when total flow recorded for a tracking observation was within 10% of one of the simulation runs (100, 150 200, 250 kcfs); this is referred to as the “10% criteria”. We also deleted observations where a fish was contacted more than once within a 1-h period, to increase the independence of observations. Limiting data by the 10% criteria, no consecutive contacts within 1 hour, and to the four hydraulic simulations, we were able to use ~79% (366/461) of the available observation for model-building.

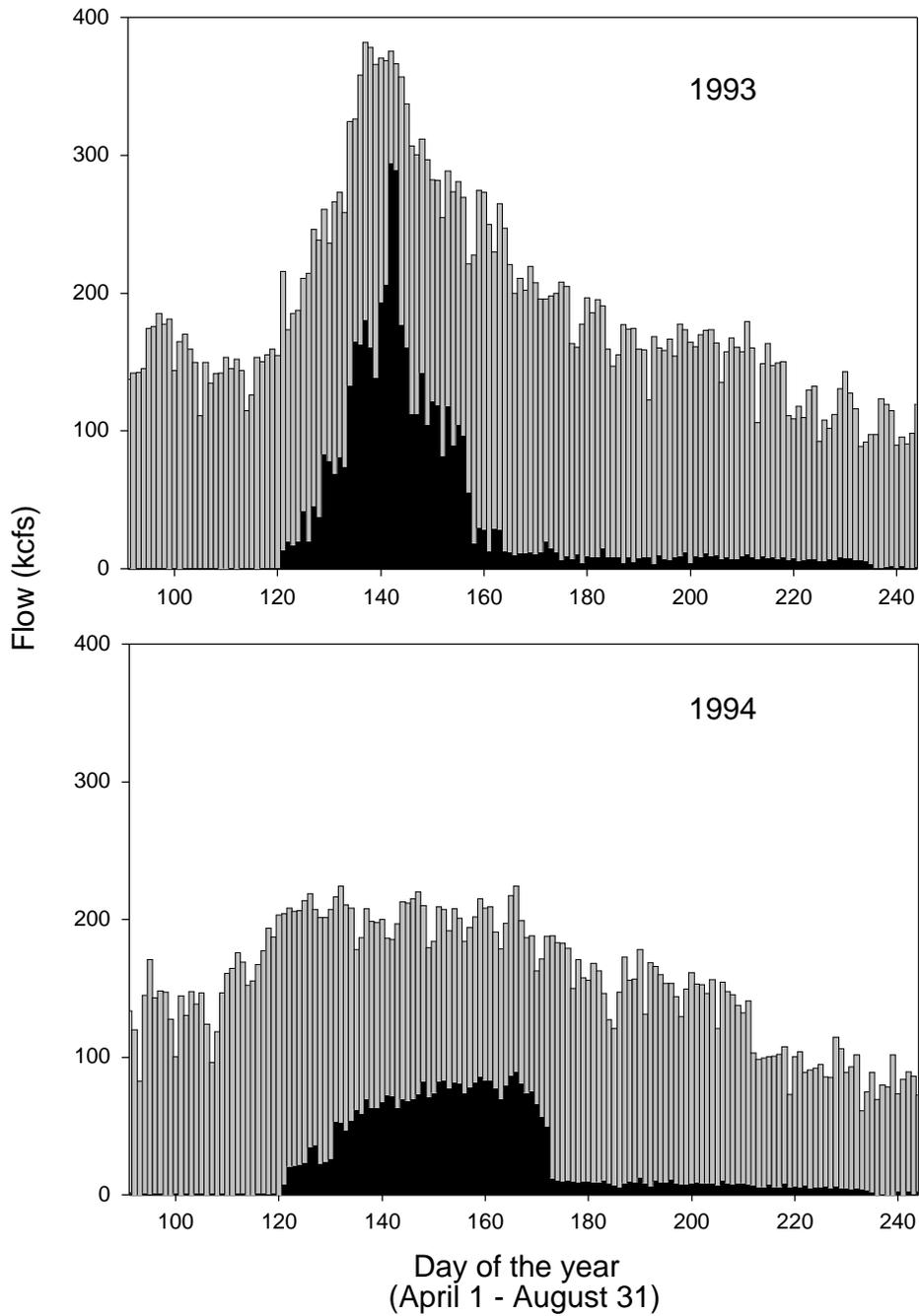


Figure 5. Spill (black bar) and total flow (gray) at The Dalles Dam in 1993 and 1994.

Results

For spring and summer 1993 and 1994, average daily flow and spill at The Dalles Dam are shown in Figure 5. For our study period in 1993 (May – July; day of year 121 to 212), total flows ranged from ~215 kcfs at the start of May, peaked at ~380 kcfs, and declined to ~160 kcfs by the end of July (Figure 5). In 1994, flow was ~205 kcfs in early May, rose little during spring and summer, and was about 103 kcfs at the end of July (Figure 5). Spill generally followed total flow in each year, but maximum spill was much greater in 1993 than in 1994 (Figure 5).

For May through July, 853 fish contacts were recorded and 461 of these were for periods when spill was less than 2.0 kcfs, from 97 different tagged northern pikeminnow. Over 30% of all observations used were from fish contacted only once and the median number of contacts was 5 (Figure 6). One individual was contacted 12 times.

A basic assumption of resource selection modeling was that predators were capable of moving to and selecting different areas (habitats) in a relatively short period of time. Short-term movement rates of all northern pikeminnow tracked during 1993 and 1994 were as high as 5 km/h (Martinelli and Shively 1997), suggesting predators could easily transit this area within an hour or two, if they chose to. Average movement rates were of course much lower. The average movement distance for northern pikeminnow in The Dalles Dam tailrace was 0.8 km (Martinelli and Shively 1997), suggesting predators did not commonly range from one end of the tailrace to the other. Barfoot and Petersen (unpublished MS) also noted that northern pikeminnow had fairly restricted “home ranges” of <2 km, for fish tagged in the free-flowing Hanford Reach and Snake River.

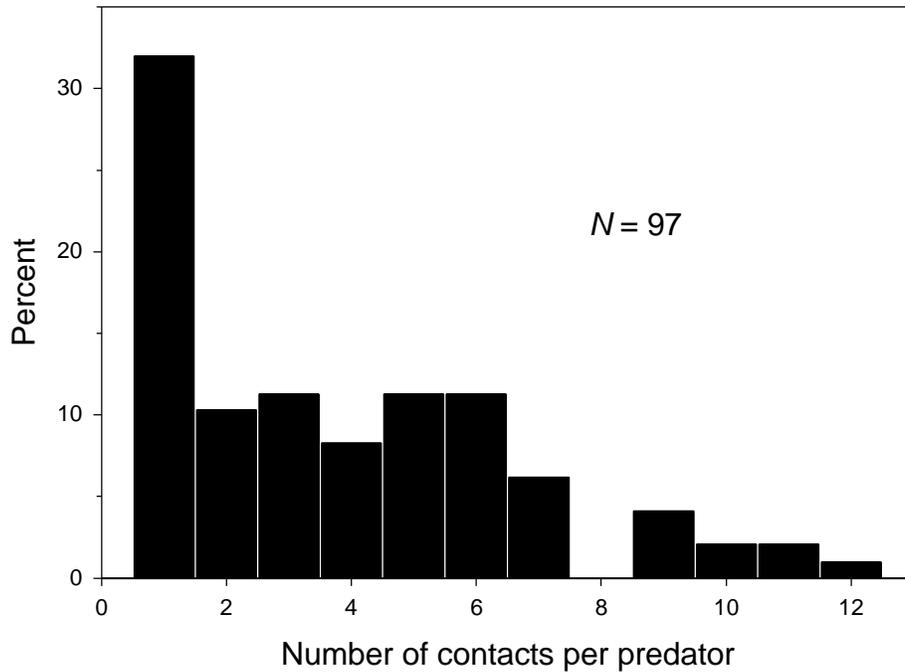


Figure 6. Number of observations per predator for the 97 radio-tagged northern pikeminnow in The Dalles Dam tailrace used in modeling. Fish were tagged and tracked during 1993 and 1994.

The frequency of contacts during May-July by flow levels are shown in Figure 7. Contacts each year were quite similar, with most contacts occurring around 175-225 kcfs (Figure 7). Some contacts of tagged fish in 1994 were made above the daily average flow (compare Figure 6 and Figure 7), which was caused by diel variation in flow and the fact that we used hourly flows.

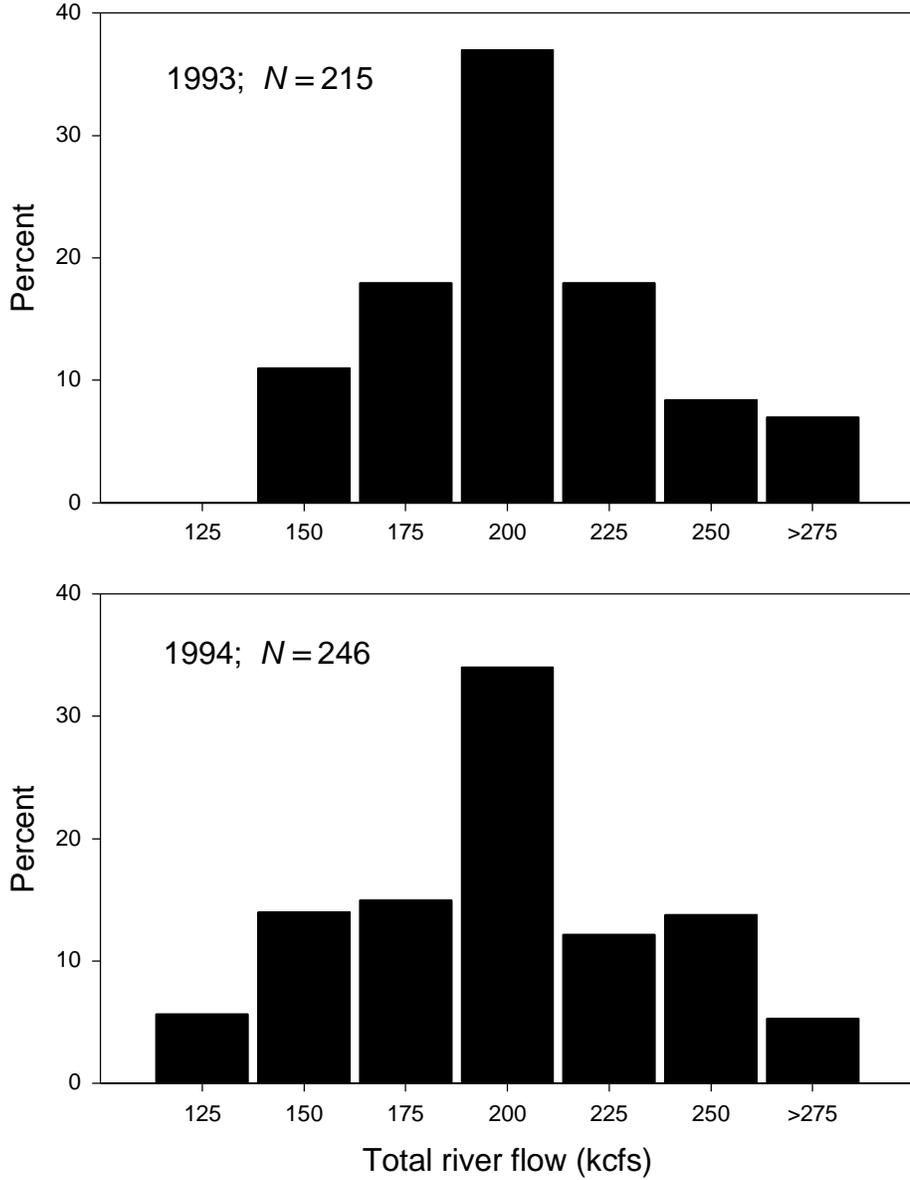


Figure 7. Frequency distribution of tagged northern pikeminnow by flow in the tailrace of The Dalles Dam during 1993 and 1994 (25 kcfs categories; spill < 2.0 kcfs).

Independent variables

Figure 8 indicates the general distribution of substrate types in The Dalles Dam tailrace. About 37% of the study area, where we had substrate data, is bedrock, 19% is boulder (or boulder and cobble), 23% is cobble and gravel, 11% is sand or mud. No data

were available for bottom substrate in 9% of the study area. Figure 9 shows the distribution of the water velocity, water depth, and distance from shore. Water velocity, depth, and distance from shore patterns are representative of a moderate flow, being derived from hydraulic simulations conducted at 200 kcfs (Rakowski and Richmond 2000).

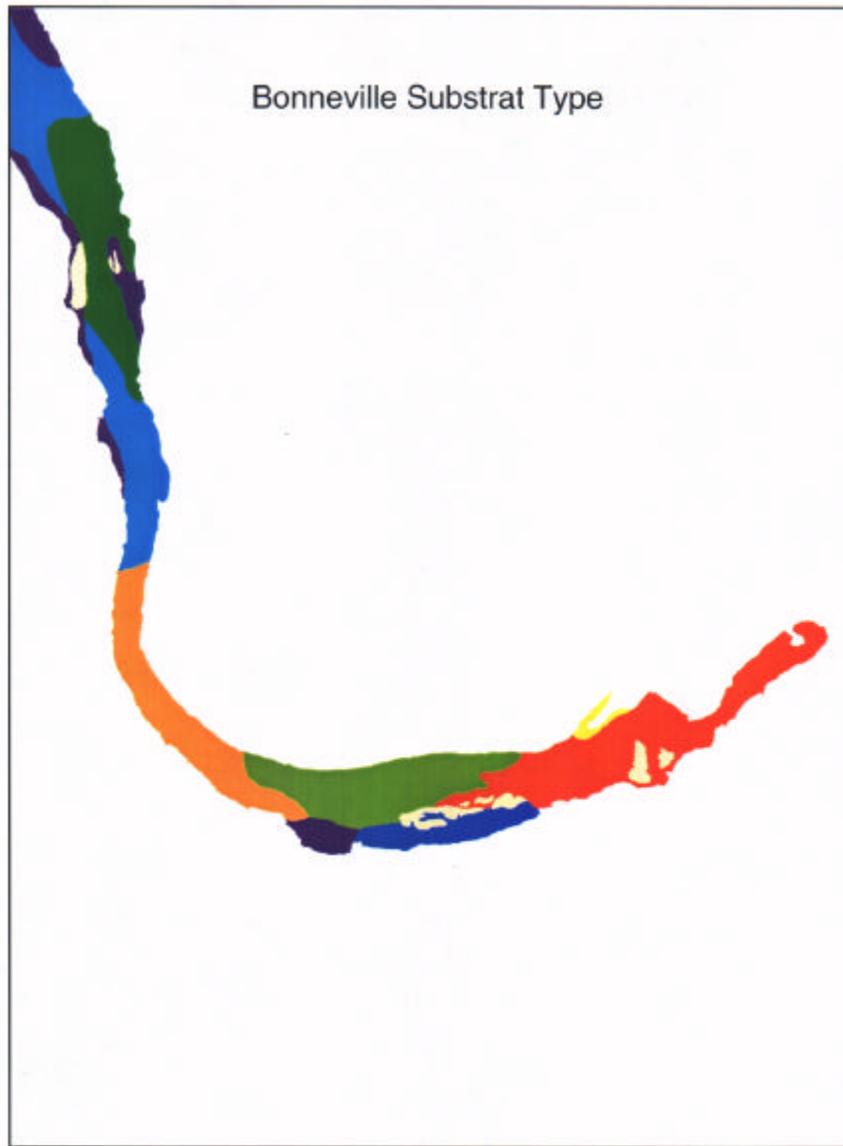


Figure 8. Distribution of substrate types in The Dalles Dam tailrace and the upper portion of the Bonneville Reservoir. Red = bedrock, yellow = boulder, blue= sand, green = cobble.

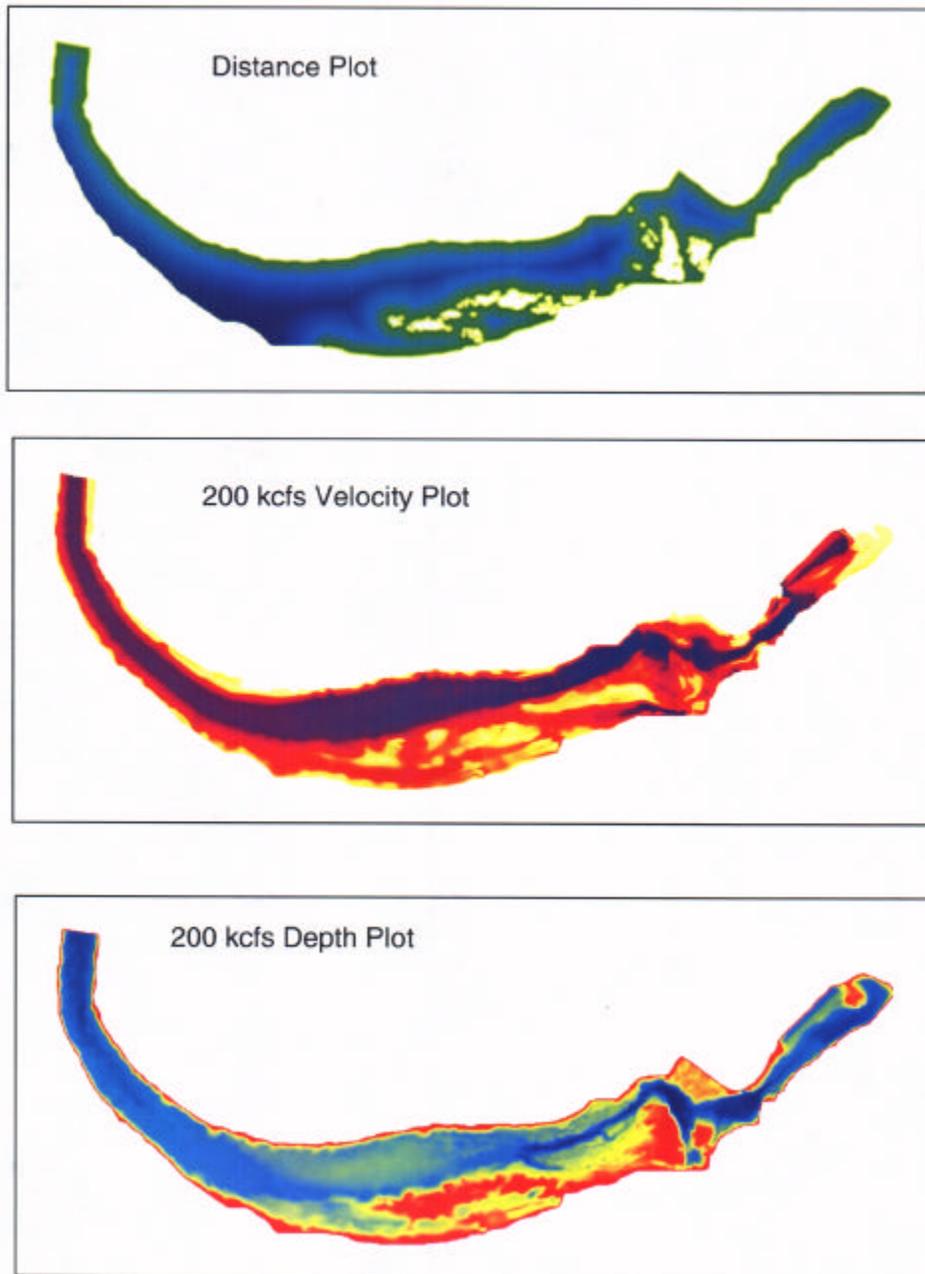


Figure 9. Distance from shore, water velocity, and water depth in The Dalles Dam tailrace, based on hydraulic simulations for 200 kcfs.

Northern pikeminnow contacts were predominantly above bedrock for those within the 10% flow criteria (Table 7). Because of this strong pattern and to prevent zero counts in some categories that can cause model overfitting or computational errors (Hosmer and Lemeshow 2000), we re-coded substrate as a dichotomous variable called *sub1*. Variable *sub1* was assigned either a value of 0 (bottom substrates of boulder, cobble, gravel, and sand) or was assigned a value of 1 (bottom substrate was bedrock). Log-likelihood tests of the re-coded substrate variable (*sub1*) were significant for all flow levels, thus we retained this variable for modeling.

Table 7. Frequency of northern pikeminnow contacts above different categories of bottom substrates by flow category (May-July, 1993-1994). Contacts were made within 10% of the flow (10% flow criteria; see text).

Substrate type	Flow category (kcfs)			
	100	150	200	250
Sand and mud	0	6	12	4
Cobble and gravel	0	3	1	3
Boulder	0	1	4	0
Bedrock	13	103	162	53

Velocity varied from near zero to about 9 ft/s in the simulation model for the tailrace (Figure 9). The distribution of available velocities shifted toward higher velocities as flow increased in the simulations, as expected (Figure 9). At the highest flows simulated, 200 and 250 kcfs, the median velocity was about 1 ft/s, while for other flows, the median velocity was <0.5 ft/s (Figure 9).

Frequency distributions of use versus available velocity suggested that pikeminnow use of velocity may be complex and vary with flow through the tailrace

(Figure 10). Little can be concluded about the lowest flow, 100 kcfs, since we had only 13 northern pikeminnow observations. At 150 kcfs, there was much higher use than availability at 1-2 ft/s, while use was less than available habitat at other velocities (Figure 10). At 200 and 250 kcfs, a fairly high proportion of northern pikeminnow were observed in areas with quite low water velocities, <0.5 ft/s. For these higher flows, pikeminnow use of areas with water velocities between about 3 and 5 ft/s was generally lower than the observed frequency of these habitats. At the highest water velocities, pikeminnow occurred roughly in proportion to the velocities frequencies (Figure 10).

At the three higher simulated flows, northern pikeminnow were strongly distributed in shallow water, generally less than 5 m (~15 ft; Figure 11). Some pikeminnow were observed at all depths. There appeared to be a slightly lower use of very shallow depths (<5 ft; Figure 11) compared to this available habitat. The relatively high proportion of radio-tag contacts in deep water (> 60 ft; Figure 11) in the low flow case (100 kcfs) is likely an artifact of having few total contacts in this sample.

Northern pikeminnow tended to be within about 70 m (~200 ft) of shores or structures in The Dalles Dam tailrace (Figure 12). Occurrence of tagged fish near shorelines was higher than the available frequency of cells at a given distance. Very few northern pikeminnow were detected at relatively large distances from shore (Figure 12), suggesting fish did not commonly occur in mid-channel.

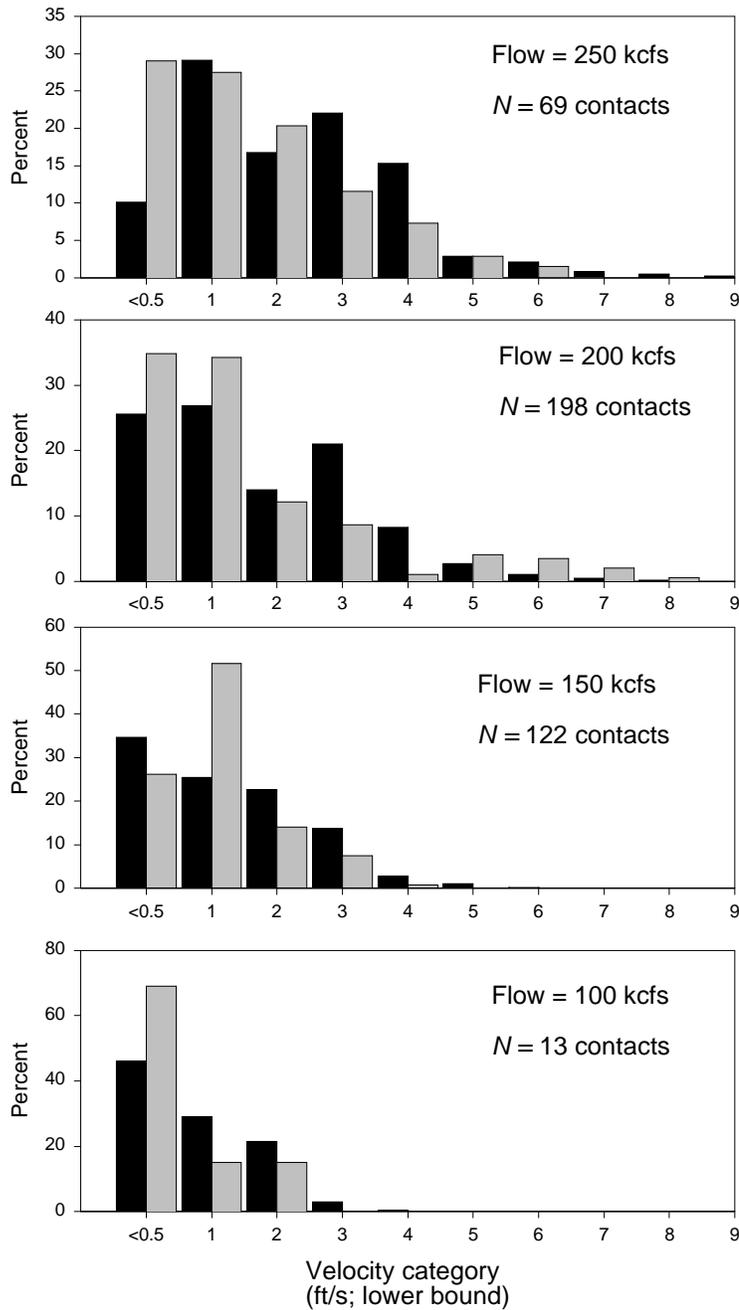


Figure 10. Available (black bars) and Used (gray bars) water velocities within The Dalles Dam tailrace for four flows. The Available category is from hydraulic simulations, while the Used category is from radio-tagged northern pikeminnow tracked in the tailrace. *N* is the number of radio-tag contacts in the Used frequency distribution.

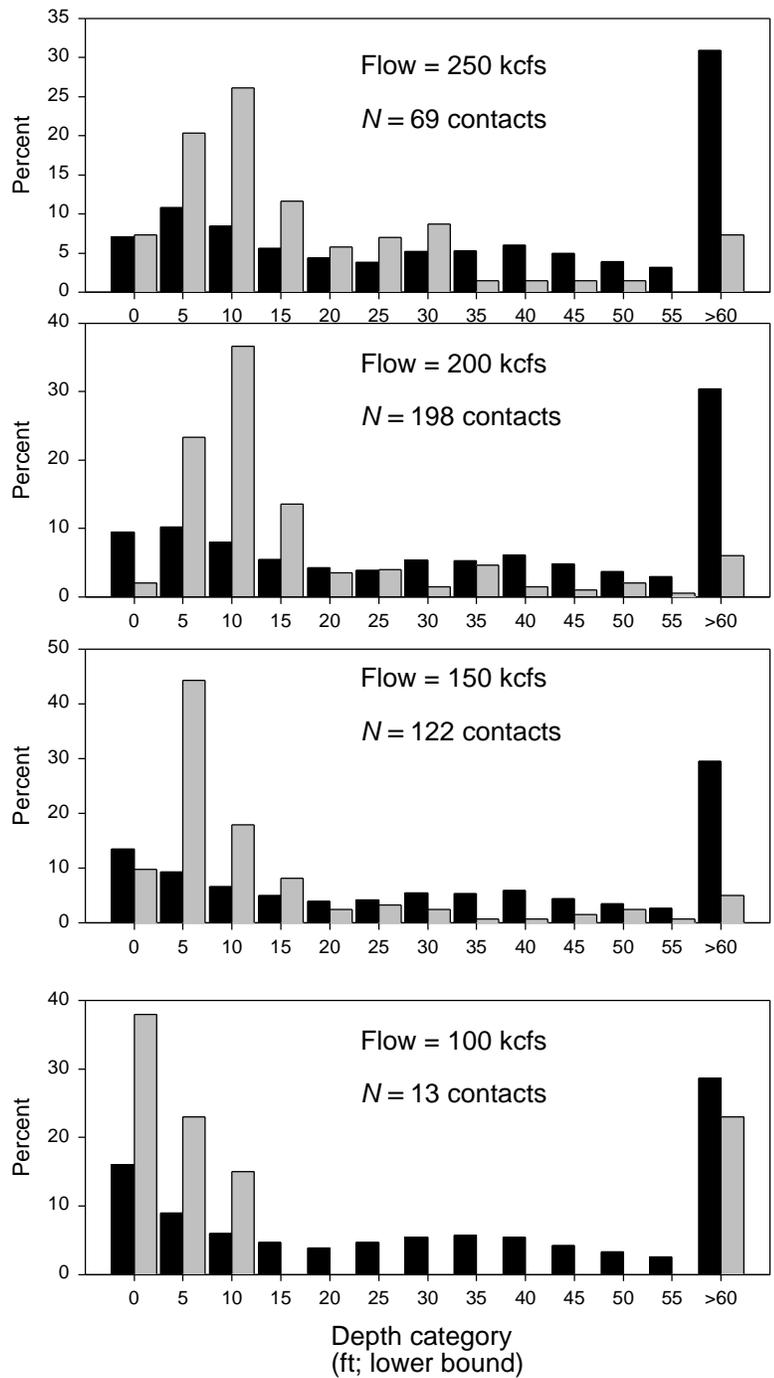


Figure 11. Available (black bars) and Used (gray bars) water depths within The Dalles Dam tailrace for four flows. The Available category is from hydraulic simulations, while the Used category is from radio-tagged northern pikeminnow tracked in the tailrace. N is the number of radio-tag contacts in the Used frequency distribution.

Univariate tests

Logistic regressions on the four independent parameters produced significant ($P < 0.05$) fits in 11 of 12 cases (Table 8), with only the test for water velocity at the 150 kcfs flow being marginally significant. Models for the 100 kcfs flow were not fit because of the low number of available observations ($N = 13$).

Table 8. Univariate test results for logistic regression. Cell values are the probability that the variable (\log_{10} transformed) was significant in a univariate model of habitat use for three different flows. Substrate probabilities were computed with a likelihood ratio chi-square test.

Flow (kcfs)	Variable			
	Velocity	Depth	Distance to shore	Substrate
150	0.066	<0.001	<0.001	<0.001
200	<0.001	<0.001	<0.001	<0.001
250	<0.001	<0.001	<0.001	<0.001

Preliminary multivariate model

We fit separate resource selection models to three flows, 150, 200, and 250 kcfs; there were insufficient radio-telemetry data to fit a model to the 100 kcfs flow (Figure 7). The univariate analyses suggested that all of the candidate variables should be used in the multivariate model. This preliminary multivariate model thus included water velocity, depth, distance to shore or structure, and the re-coded substrate variable (bedrock and non-bedrock). Statistical output is summarized for each flow in Appendix 2.

For the 150 kcfs case, all variables were significant at the 0.10 test level (Hosmer and Lemeshow 2000), except for water velocity which had a probability of 0.103. The Hosmer and Lemeshow goodness-of-fit test suggests that we do not reject the null hypothesis that these data fit the model. For the 200 kcfs case, the results were similar to

the 150 kcfs model just discussed (Appendix 2). The Hosmer and Lemeshow goodness-of-fit test would reject the null hypothesis in this case. For data at 250 kcfs, all variables were significant at the 0.05 level except for the distance variable, which was not significant. The Hosmer and Lemeshow goodness-of-fit test indicates the model cannot be rejected.

The sign of the parameter for velocity was negative in all univariate models, but was positive for one (150 kcfs) of the three multivariate models. In the multivariate models, the velocity parameter increased with increasing flow. The magnitude and sign of the depth parameter was consistent across the univariate and the multivariate models. The distance parameter was negative in all three of the univariate models, but was positive in the three multivariate models. The substrate coefficients were consistent across all multivariate models.

Based upon these comparisons of univariate and multivariate models, we removed distance from the multivariate model and refit with only substrate, velocity, and depth. Log-likelihood tests at each flow ($G = -2 * [\text{difference in log-likelihood}]$) were all < 0.001 suggesting that distance should be retained in the overall model.

We used the models for each flow in Appendix 2 for the remainder of this report, recognizing that improvements can likely be made and further model testing is necessary (see the Discussion). The relative probability of use by northern pikeminnow for the three different flows are given by:

$$P_{150} = \{ \exp(-8.01 + 0.35 * V - 0.81 * Dp + 0.31 * Dt + 2.65 * sub) \} /$$

$$[1 + \{ \exp(-8.01 + 0.35 * V - 0.81 * Dp + 0.31 * Dt + 2.65 * sub) \}] \quad \text{Equation 2}$$

$$P_{200} = \{ \exp(-6.82 - 0.21 * V - 0.61 * Dp + 0.16 * Dt + 2.61 * sub) \} /$$

$$[1 + \{ \exp(-6.82 - 0.21 * V - 0.61 * Dp + 0.16 * Dt + 2.61 * sub) \}] \quad \text{Equation 3}$$

$$P_{250} = \{ \exp(-6.42 - 0.52 * V - 0.63 * Dp + 0.03 * Dt + 2.14 * sub) \} /$$

$$[1 + \{ \exp(-6.42-0.52*V-0.63*Dp+0.03*Dt+2.14*sub) \}] \quad \text{Equation 4}$$

where P_{nnn} is the probability at flow nnn , V is log water velocity, Dp is log water depth, Dt is log distance, and sub is the re-coded substrate for bedrock (1) and non-bedrock (0).

Preliminary model evaluation

The coefficients for the independent variables were generally consistent for the three models although there were some results that may require further testing. The water velocity coefficient was positive for the 150 kcfs model, but negative for the other two models. Inspection of the use versus available data (Figure 10) suggested that the response to water velocity was similar at different river flows, except for an unusual peak at 1 ft/s in the 150 kcfs case. The coefficients for water depth were relatively large and negative in all cases (Equations 2-4), however the use data suggested preferred depths of about 5-15 ft (Figure 11). The coefficients for distance to shore were consistently small and positive, which suggests that probabilities increase with distance from shore, something that was unexpected (Martinelli and Shively 1997). The pattern for this variable was similar to depth – perhaps an avoidance of locations very near shore and in very shallow water. Non-linear response equations may fit some of these data better (Hosmer and Lemeshow 2000).

We used Equations 2-4 to predict the probability of northern pikeminnow occurrence in the tailrace of The Dalles Dam for three different flows (Figure 13). For each cell having a velocity, depth, and distance from shore (derived from the simulations of Rakowski and Richmond 2000), and having substrate classification, we could compute a probability. Initial probabilities were re-scaled to a range of 0-1 for ease of interpretation.

Increased total discharge in the three simulations caused some shifts in predicted habitat that would be generally expected. For example, the higher discharge (250 kcfs) caused a decrease in the quality of predator habitat in the middle of the channel where water velocities would increase considerably (Figure 13). The higher discharge also caused the quality of the predator habitat in the area just down from the spillway to decrease, although there was no difference in percent spill in these simulations. As would

be expected, high predicted probabilities occurred in areas where there was a high frequency of radio-tag contacts (Figure 14). For example, there was a relatively high frequency of radio-tag contacts in the lower island zone (Oregon shore) and this area consistently shows as having “high” quality habitat for northern pikeminnow (Figure 14).

Finer-scale analysis of habitat can also be investigated with this spatial model. Increasing the river flow and turbine discharge from 150 to 250 kcfs, for example, causes some fairly distinct changes in predator habitat near the navigation lock (Figure 15). In this area, the higher flow appears to cause a narrowing of the low-probability predator habitat, compared to the lower flow simulation. This result is likely a consequence of velocity and depth patterns through this area at different flows.

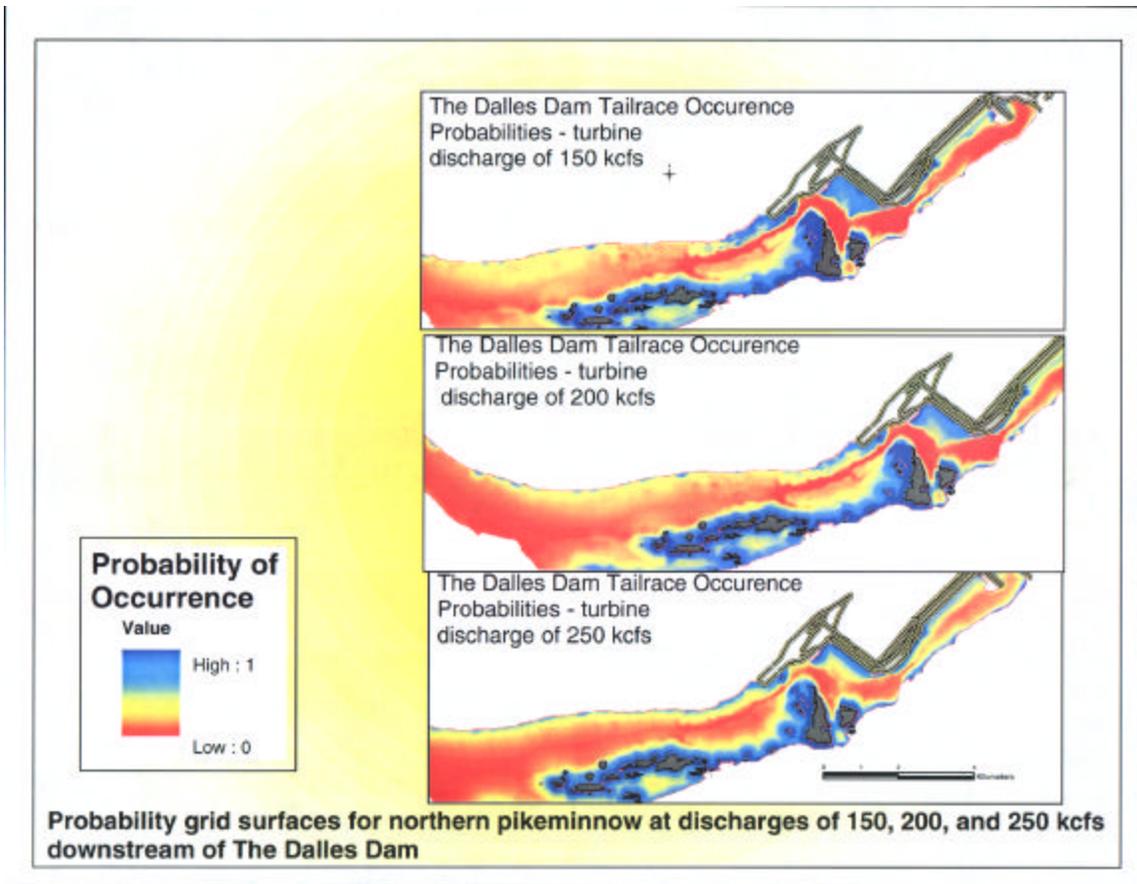


Figure 13. Distribution of northern pikeminnow habitat for three flows using logistic regression models. Habitat is scaled as a probability (0-1) of occurrence where each predicted value was divided by the maximum predicted value. There was no spill for these scenarios.

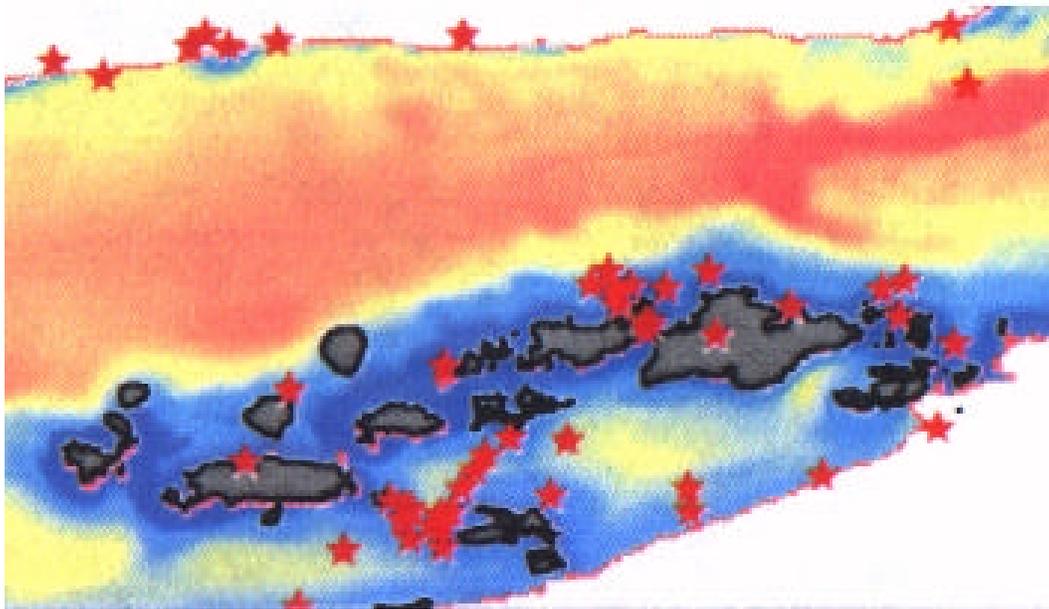


Figure 14. Predicted habitat quality for northern pikeminnow in a section of The Dalles Dam tailrace at 200 kcfs. Red stars indicate positions of radio-tagged fish used in development of the model. See Figure 13 for a legend of habitat quality.

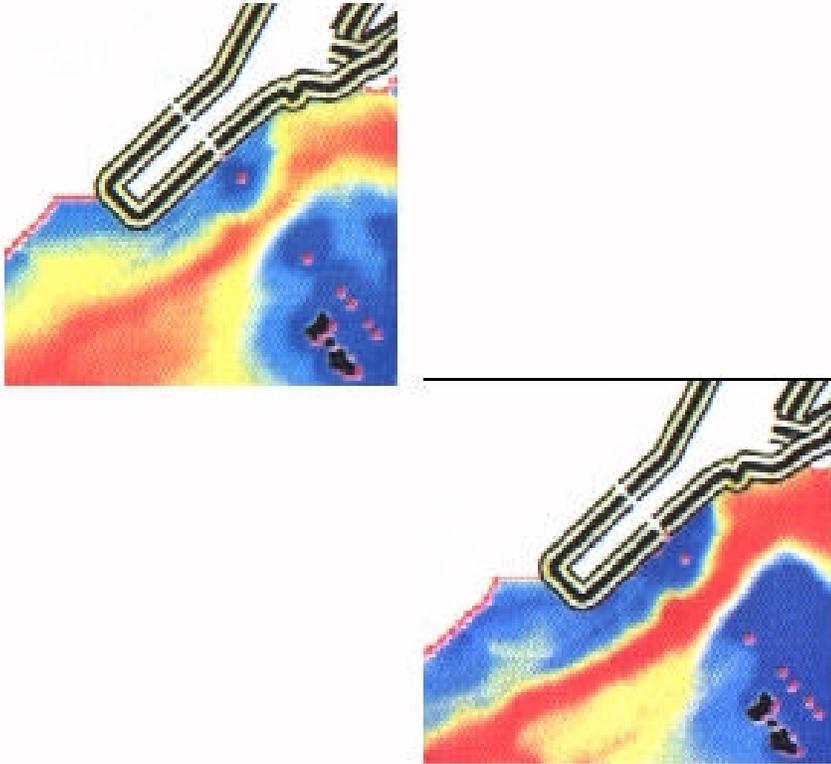


Figure 15. Fine-scale predictions of northern pikeminnow habitat near The Dalles Dam navigation lock. Upper image represents a turbine discharge of 250 kcfs and the lower image is for a turbine discharge of 150 kcfs. See Figure 13 for the legend for habitat probabilities.

Discussion

Results from 2001

Consistent high wind conditions (often >25 mph) and equipment failure limited our field sampling efforts during the spring and summer. Electroshock sampling is ineffective when winds create high waves and probes cannot be kept in the water. With the limited data collected, we observed a relatively high abundance of smallmouth bass in The Dalles Dam tailrace. About twice as many bass were collected as northern pikeminnow and the density of bass was roughly twice that of northern pikeminnow. Along the riprap shore of the Washington shore, smallmouth bass had a density of over 4 per 15-minute transect compared to less than 2 for northern pikeminnow. This relatively high number of smallmouth bass at The Dalles Dam tailrace compared to northern pikeminnow may be influenced by the ongoing Northern Pikeminnow Management Program, or by an increase in the bass population. We estimated 1,000 to 2,000 smallmouth bass in The Dalles Dam tailrace although this is based on relatively few marked and recaptured fish. We could not make a population estimate for northern pikeminnow.

We did not detect any PIT tags within northern pikeminnow or smallmouth bass digestive tracts although our sampling should have coincided with the time when tags were present. The experiment that we conducted in the laboratory provided good evidence that PIT tags should be retained in northern pikeminnow guts for at least 20 h after ingestion. Our sampling effort started within a few hours of release of the PIT-tagged fish and continued for 6-8 h, so it is unlikely that juvenile salmon were consumed and tags were evacuated before we collected predators.

Failure to detect PIT tags in predators is perhaps not surprising considering that we collected relatively few predators, and the rate of survival through the tailrace is generally >90% (Dawley et al. 1998). Petersen et al. (1994) collected a large number of coded-wire tags from the guts of northern pikeminnow in the Bonneville Dam tailrace following the release of batches of 5,000 – 10,000 tagged salmonids. Much of their sampling, however, was conducted closer to the site of prey release than we could

achieve in the current study at The Dalles Dam. Future experiments using PIT-tagged released fish should probably concentrate sampling nearer to the release site, and can use our tag evacuation model to estimate a sample period.

The laboratory experiment that we conducted using tagged juvenile salmon and northern pikeminnow showed that PIT tags were evacuated from predators at a relatively low rate, and evacuation was especially sensitive to temperature. PIT tags in preyfish consumed at 14 °C were retained within the gut of the predator for about twice as long as tags from prey consumed at 18 °C. This result is consistent with evacuation of glass beads from cod (dos Santos and Jobling 1991).

The different magnitude of coefficients for independent variables in the habitat models causes probabilities to change at different rates. For example, changing distance to shore from 1 to 100 m (x100), while holding other independent variables constant, causes P to increase ~5% . A change of similar magnitude in water velocity (0.1 to 10 m/s; x100) in the model caused P to ~40% change. More sensitivity analyses such as these will be needed before applying these models.

The habitat models do not completely explain the occurrence of northern pikeminnow in the tailrace of The Dalles Dam since a moderate number of predators were observed in what might be called “poor” habitat. For example, there were quite a few predator occurrences in habitat with low probabilities in the 200 and 250 kcfs simulations of predator habitat . Fish observed in these areas might be moving across the river from a good site to another good site, or they could be feeding on juvenile salmonids that are moving down the river along the thalweg. Northern pikeminnow in the McNary Dam tailrace were shown to feed during short periods, or “bouts” (Petersen and DeAngelis 1992). Predators might thus be capturing juvenile salmonids where they are traveling and dense, and then returning to low velocity, shallow areas for digestion. If this is the case, then the habitat models that we’ve begun developing here may be more representative of resting and digesting areas than they are of areas with a high likelihood of actual predation events. Future work (see below) will compare the overlap of predator and prey habitats, and will also consider some specific tagging experiments to examine such questions. Note that there are some non-intuitive results in habitat distributions.

For example, at higher flows there might be a “bottleneck” effect on low quality predator habitat in the mid channel below the navigation lock.

Potential analyses or studies

Several questions should be explored before the predator habitat model can be used for management decisions. Some of the more important questions are:

- ?? Are there threshold effects (Hosmer and Lemeshow 2000) as predators respond to variables such as water velocity?
- ?? Is the log transformation most appropriate for modeling, or would other transformations produce models that fit the data better?
- ?? Are there interactions between the main effects variables?
- ?? Should we include total river discharge in models, probably as a design variable? Discharge influences velocity in particular, but there may be some advantages to including it as a separate variable, such as a simpler model that could apply to a broader range of flows. The similarity in coefficients of the three fit models suggest some combination is possible.
- ?? Is the physical scale of analysis and prediction appropriate for specific decisions, or should we strive for finer or rougher scales?
- ?? Can data at certain spill levels be used to improve the models? This would require additional runs of PNNL model at specific discharge and spill levels.

A couple specific applications of corroborated habitat models will be mentioned. First, future model runs might include different spill patterns and examine how predator habitat predictions change. North spill versus south spill is often discussed in management forums, and these models could be used to add some analyses of predator habitat to the debates. Second, location of the future smolt bypass outfall might be explored with these models. PNNL has various runs from a 2-dimensional hydraulic simulation model for outfall locations (C. Rakowski, personal communication), and these might be combined with resource use models to predict predator habitat. Spatial analyses of the distribution of habitat types and the total amount of habitat available to predators could be estimated.

For the immediate future (FY02), efforts will be focused on improving and corroborating the northern pikeminnow model, and beginning development of spatial habitat models for juvenile salmonids that transit The Dalles Dam tailrace. The relatively high density of smallmouth bass in The Dalles Dam tailrace (this study; Ward et al. 1995), and their likely predation on juvenile salmonids (e.g. Tabor et al. 1993), suggests that more attention should be given to understanding their distribution, abundance, and effect. We will consider whether data and models for smallmouth bass could be applied in The Dalles Dam tailrace to predict distributions or habitat quality. Such data might be available from other studies in the Columbia or Snake rivers (e.g. Petersen et al. 2000).

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Appendicies

Appendix 1. Starting locations and waypoints for field sampling of predators within The Dalles Dam tailrace, 1999. Most transects were laid out in sequence so the end point of a transect is generally the start point of the next transect. OR = Oregon, WA = Washington.

Transect	Shore	Latitude	Longitude	Waypoint
230-170	OR	45 ⁰ 36' 07.82"	121 ⁰ 10' 22.49"	1
230-180	OR	45 ⁰ 36' 03.00"	121 ⁰ 09' 54.90"	2
230-190	OR	45 ⁰ 36' 06.55"	121 ⁰ 09' 28.22"	3
230-200	OR	45 ⁰ 36' 10.70"	121 ⁰ 09' 05.39"	4
230-210	OR	45 ⁰ 36' 17.14"	121 ⁰ 08' 52.58"	5
230-220	OR	45 ⁰ 36' 20.87"	121 ⁰ 08' 33.29"	6
230-230	OR	45 ⁰ 36' 24.24"	121 ⁰ 08' 19.07"	7
230-230 (end)	OR	45 ⁰ 36' 32.06"	121 ⁰ 08' 16.78"	8
230-080	WA	45 ⁰ 36' 28.07"	121 ⁰ 10' 25.75"	9
230-070	WA	45 ⁰ 36' 31.19"	121 ⁰ 10' 50.17"	10
230-090	WA	45 ⁰ 36' 30.73"	121 ⁰ 09' 56.26"	11
230-100	WA	45 ⁰ 36' 33.10"	121 ⁰ 09' 29.43"	12
230-110	WA	45 ⁰ 36' 33.58"	121 ⁰ 09' 06.24"	13
230-110 (end)	WA	45 ⁰ 36' 45.48"	121 ⁰ 08' 35.73"	14
230-300	WA	45 ⁰ 36' 19.58"	121 ⁰ 09' 21.49"	15
230-310	WA	45 ⁰ 36' 18.99"	121 ⁰ 09' 02.57"	16
230-310	WA	45 ⁰ 36' 18.10"	121 ⁰ 09' 21.68"	17

Appendix 2. Summary statistics for the preliminary model fits, by flow level. Output is from the SAS procedure PROC LOGISTIC.

150 Kcfs

Model Fit Statistics

Criterion	Intercept Only	Intercept and Covariates
AIC	1727.272	1545.898
SC	1736.754	1593.306
-2 Log L	1725.272	1535.898

Testing Global Null Hypothesis: BETA=0

Test	Chi-Square	DF	Pr > Chi Sq
Likelihood Ratio	189.3741	4	<.0001
Score	198.9250	4	<.0001
Wald	121.8492	4	<.0001

The LOGISTIC Procedure

Analysis of Maximum Likelihood Estimates

Parameter	DF	Estimate	Standard Error	Chi-Square	Pr > Chi Sq
Intercept	1	-8.0144	0.6419	155.8865	<.0001
logv	1	0.3488	0.2138	2.6626	0.1027
logdepth	1	-0.8122	0.1164	48.7029	<.0001
logdist	1	0.3051	0.1087	7.8867	0.0050
sub1	1	2.6497	0.3444	59.1783	<.0001

Odds Ratio Estimates

Effect	Point Estimate	95% Wald Confidence Limits
logv	1.417	0.932 2.155
logdepth	0.444	0.353 0.558
logdist	1.357	1.097 1.679
sub1	14.150	7.204 27.793

Hosmer and Lemeshow Goodness-of-Fit Test

Chi-Square	DF	Pr > Chi Sq
6.7644	5	0.2388

200 Kcfs

Model Fit Statistics

Criterion	Intercept Only	Intercept and Covariates
AIC	2600.407	2334.242
SC	2609.889	2381.650
-2 Log L	2598.407	2324.242

Testing Global Null Hypothesis: BETA=0

Test	Chi-Square	DF	Pr > Chi Sq
Likelihood Ratio	274.1659	4	<.0001
Score	277.4579	4	<.0001
Wald	168.0779	4	<.0001

The LOGISTIC Procedure

Analysis of Maximum Likelihood Estimates

Parameter	DF	Estimate	Standard Error	Chi-Square	Pr > Chi Sq
Intercept	1	-6.8200	0.4844	198.2334	<.0001
logv	1	-0.2097	0.1542	1.8481	0.1740
logdepth	1	-0.6115	0.0962	40.4025	<.0001
logdist	1	0.1611	0.0801	4.0434	0.0443
sub1	1	2.6122	0.2698	93.7370	<.0001

Odds Ratio Estimates

Effect	Point Estimate	95% Wald Confidence Limits	
logv	0.811	0.599	1.097
logdepth	0.543	0.449	0.655
logdist	1.175	1.004	1.375
sub1	13.629	8.032	23.127

Hosmer and Lemeshow Goodness-of-Fit Test

Chi-Square	DF	Pr > Chi Sq
41.8576	6	<.0001

Model Fit Statistics

Criterion	Intercept Only	Intercept and Covariates
AIC	1008.433	924.211
SC	1017.915	971.619
-2 Log L	1006.433	914.211

Testing Global Null Hypothesis: BETA=0

Test	Chi-Square	DF	Pr > Chi Sq
Likelihood Ratio	92.2224	4	<.0001
Score	96.0556	4	<.0001
Wald	63.0914	4	<.0001

The LOGISTIC Procedure

Analysis of Maximum Likelihood Estimates

Parameter	DF	Estimate	Standard Error	Chi-Square	Pr > Chi Sq
Intercept	1	-6.4229	0.7692	69.7325	<.0001
logv	1	-0.5165	0.2604	3.9327	0.0474
logdepth	1	-0.6286	0.1682	13.9649	0.0002
logdist	1	0.0287	0.1271	0.0512	0.8210
sub1	1	2.1442	0.4162	26.5454	<.0001

Odds Ratio Estimates

Effect	Point Estimate	95% Wald Confidence Limits	
logv	0.597	0.358	0.994
logdepth	0.533	0.384	0.742
logdist	1.029	0.802	1.320
sub1	8.535	3.775	19.294

Hosmer and Lemeshow Goodness-of-Fit Test

Chi-Square	DF	Pr > Chi Sq
7.2827	4	0.1217