PALLID STURGEON MANAGEMENT AND RECOVERY IN THE YELLOWSTONE RIVER

Annual Report for 2008

Matthew Jaeger, Alan Ankrum, and Trevor Watson
Montana Fish, Wildlife & Parks

Gillian Hadley and Jay Rotella
Montana State University

George Jordan and Ryan Wilson
U.S. Fish and Wildlife Service

Sue Camp
U.S. Bureau of Reclamation

Tony Thatcher
DTM Consulting, Inc.

Karin Boyd
Applied Geomorphology, Inc.

ABSTRACT

Pallid sturgeon spawn in the Yellowstone River downstream of Intake Diversion although long drift distances following hatching may preclude recruitment; larval pallid sturgeon likely drift into Sakakawea Reservoir and die. Reestablishing demographically adequate populations that spawn far upstream of reservoirs is necessary for natural recruitment and pallid sturgeon recovery to occur. Accordingly, management options that are likely to prevent extinction and result in recovery of self-sustaining pallid sturgeon populations include a combination of 1) habitat restoration and conservation and 2) population augmentation. To provide information pursuant to making the best management decisions for pallid sturgeon recovery in the lower Yellowstone River our specific objectives were to 1) determine seasonal movements and habitat selection of juvenile pallid sturgeon in the lower Yellowstone River, 2) develop monitoring strategies to most effectively estimate survival of hatchery-reared pallid sturgeon in the lower Yellowstone River, and 3) assess the status and demographics of adult pallid sturgeon in RPMA 2. Median daily movement rates of juvenile pallid sturgeon were higher during spring and runoff than winter. Both up and downstream movements occurred during spring, runoff, and summer and predominately downstream movements occurred during winter. Most habitats were used in proportion to their availability during most seasons; however, pallid sturgeon selected unarmored pools at the valley margin (bluff pools) during summer and winter but avoided these habitats if they were armored (valley margin rip-rap pools) during all seasons other than winter when they were used in proportion to availability. Targeted sampling of bluff pool habitats from 2006 to 2008 in summer resulted in catch rates (5.8 fish per hour or 1.2 fish per trammel net drift) of hatchery-reared pallid sturgeon that were 20 to 90 times greater than those of previous randomized sampling designs. Annual survival probability following release was highest for 2-year-olds, followed by summer and spring yearlings, which had similar initial survival probabilities, and fingerlings, which had the lowest survival probabilities. Increased capture probabilities and number of recaptures, reduced sampling intervals, and the ability to evaluate survival model assumptions resulted from integrating telemetry into our monitoring approach. Wild adult pallid sturgeon in RPMA 2 have declined by about 76% since 1988 and present abundance was 125 (100, 150). The population is skewing slowly towards male fish; male-to-female gender ratio changed from 1.2:1 in 1988 to 2:1 in 2008. Estimated annual probability of survival for adult pallid sturgeon was higher (0.986) than described in planning documents. Our findings suggest that stream bank armoring may degrade the quality of habitat for pallid sturgeon and that a targeted monitoring approach that includes a telemetry component will result in more effective and informed management decisions. Habitat restoration projects intended to benefit extant wild pallid sturgeon in RPMA 2 should be implemented immediately given observed gender ratios and declines in abundance.

INTRODUCTION

Pallid sturgeon, a species native to the Yellowstone River, was listed as endangered in 1990. Declines in pallid sturgeon distribution and abundances are attributed to alteration of a natural flow regime and habitat degradation caused by impoundments and channelization throughout the upper Missouri River (Kallemeyn 1983). No recruitment has occurred in Recovery Priority Management Area (RPMA) 2 in at least 30 years. Accordingly, recovery efforts have focused on preserving the pallid sturgeon genetic pool through a captive breeding and conservation stocking program until habitat restoration permits the re-establishment of self-sustaining populations (U.S Fish and Wildlife Service 2008). Because limited time remains before extant populations senesce, identification of management strategies that provide the best opportunity for survival to maturity and successful spawning and recruitment by wild or hatchery-reared pallid sturgeon is essential for continued existence of this species.

Because of its relatively pristine character, including a near-natural hydrograph and associated temperature and sediment regimes, the Yellowstone River provides an excellent opportunity for pallid sturgeon recovery. The importance of natural riverine function is emphasized by the movements and behavior of extant pallid sturgeon; most pallid sturgeon ascend the lower Yellowstone River each spring and the Yellowstone River may be the only location in RPMA 2 that is used for and supports successful spawning (Bramblett and White 2001; Fuller et al. 2007). However, inadequate larval drift distances between putative spawning areas downstream of Intake Diversion and the headwaters of Sakakawea Reservoir preclude recruitment; larval pallid sturgeon likely drift into the reservoir and die (Kynard et al. 2007; Braaten et al. 2008). Recovery options that would restore sufficient larval drift distances in the upper Missouri River basin include reestablishment of demographically adequate populations that spawn far upstream of reservoirs or dewatering or removal of mainstem reservoirs. Efforts to provide fish passage at Intake Diversion, which would allow pallid sturgeon to access an additional 164 miles of spawning and larval drift habitat, are currently underway. Because the population to realize benefit from this restoration action will be comprised of both wild extant and hatchery-reared pallid sturgeon an understanding of the demographics of both groups is important. As extant fish senesce and recovery is increasingly reliant on natural reproduction of hatchery-reared pallid sturgeon it is essential that managers have adequate information to create a suitable "replacement" population through the conservation stocking program. This requires an understanding of both demographics and behavioral ecology of juvenile and adult pallid sturgeon. Considerable work on behavioral ecology of adult pallid sturgeon has occurred in RPMA 2 (e.g., Bramblett and White 2001; Fuller et al. 2007) but formal assessment of demographic rates using existing mark-recapture data has been somewhat limited. Although pallid sturgeon were stocked in the Yellowstone River beginning in 1998, stocking practices and monitoring programs that resulted in low numbers of recaptures and violated survival model assumptions have limited inferences regarding the conservation stocking program to date (Hadley and Rotella 2009).

In summary, management options that are likely to prevent extinction and result in recovery of self-sustaining pallid sturgeon populations include a combination of 1) habitat restoration and conservation and 2) population augmentation. Our goal is to provide information pursuant to making the best management decisions for pallid sturgeon recovery in the lower Yellowstone River. Our specific objectives were to:

1) Determine seasonal movements and habitat selection of juvenile pallid sturgeon in the lower Yellowstone River.
2) Develop monitoring strategies to most effectively estimate survival of hatchery-reared pallid sturgeon in the lower Yellowstone River.
3) Assess the status and demographics of adult pallid sturgeon in RPMA 2.

Knowledge of seasonal movements and habitat selection of juvenile pallid sturgeon will allow prioritization of habitat conservation or restoration actions. This information will also result in formulation of the most effective monitoring strategies and allow assessment of survival model assumption violations. Development of monitoring strategies to estimate survival of hatchery-reared pallid sturgeon will yield information required for effective management of the conservation stocking program. Similarly, assessment of the status and demographics of adult pallid sturgeon will provide information needed to make the best management decisions to prevent pallid sturgeon extirpation and achieve natural recruitment within RPMA 2.

STUDY AREA

The study area consists of the 470 km of the Yellowstone River downstream of Rancher Diversion (Figure 1). Mean annual discharge at the USGS gauging station in Miles City, Montana, is 323 m^3/s and mean annual peak discharge is 1480 m^3/s. River geomorphology varies throughout the study area in direct response to valley geology; straight, sinuous, braided, and irregular-meander channel patterns occur (Silverman and

Tomlinsen 1984). The channel is often braided or split and long side channels are common. Islands and bars range from large vegetated islands to unvegetated point and mid-channel bars (White and Bramblett 1993). Substrate is primarily gravel and cobble upstream of river kilometer 50 and is primarily fines and sand below (Bramblett and White 2001). The fish assemblage is comprised of 49 species from 15 families, including eight state-listed Species of Special Concern and one federally listed endangered species (White and Bramblett 1993; Carlson 2003). The primary deleterious anthropogenic effect on the fish assemblage is water withdrawal for agriculture (White and Bramblett 1993). About 90% of all water use on the Yellowstone River is for irrigation, which corresponds to annual use of 1.5 million acre-feet (White and Bramblett 1993). Four mainstem low-head irrigation diversions dams occur in the study area. The largest and downstream-most of these, Intake Diversion, diverts about 38 m^3/s during the mid-May to mid-September irrigation season (Hiebert et al. 2000). Intake Diversion has likely served as a barrier to upstream movements of pallid sturgeon since its construction in 1905. About 576,629 fish of 36 species are annually entrained in Intake Canal, of which as many as 8% are sturgeon (Hiebert et al. 2000).

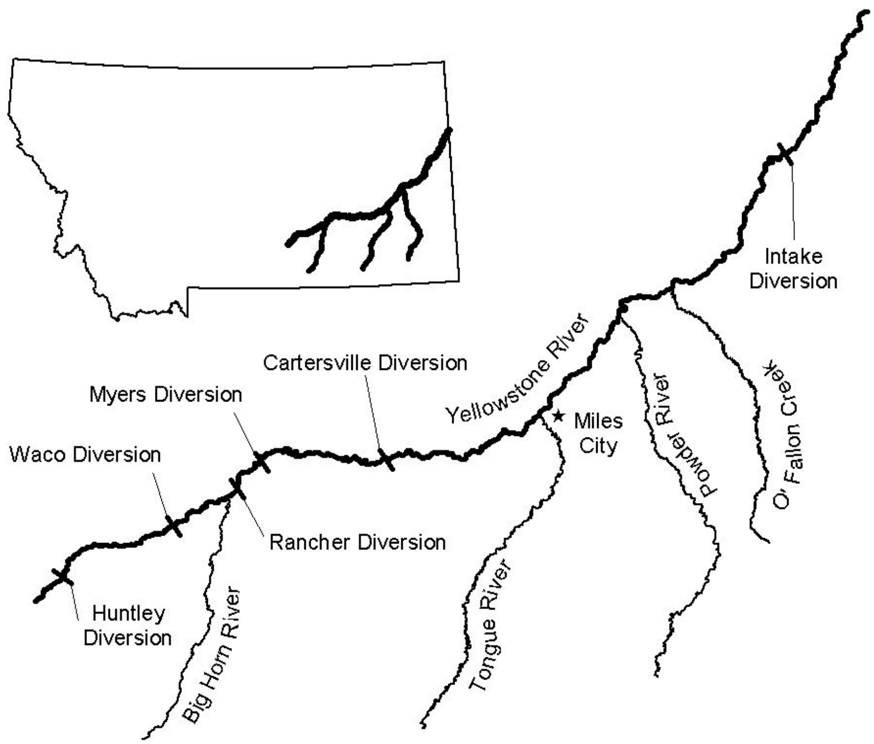

Figure 1. The lower Yellowstone River, its major tributaries, and diversion dams.

METHODS

Movements and habitat selection: Movements and habitat selection of juvenile pallid sturgeon were assessed using two groups of telemetered fish. Group 1 included fifty-nine 2004 year-class fish telemetered at the Miles City State Fish Hatchery prior to release and stocked at three locations (Rancher, Cartersville, and Intake diversions) on 20 October

2006. Group 2 was comprised of 30 fish captured and telemetered in the Yellowstone River downstream of Intake Diversion during summer 2007. Group 2 fish were originally stocked at various locations in RPMA 2 between 18 August 2002 and 24 April 2007. Transmitters were surgically implanted using procedures modified from Hart and Summerfelt (1975). Incisions were closed using surgical staples. The 450-mm long whip antennae trailed externally (Ross and Kleiner 1982). Each transmitter emitted a unique code detectable with radio antennae at 148.800 MHz.

Fish were relocated by boat about every other week during ice-free months (April to November) and by aircraft during winter months (November to March) about every two to three weeks. Following detection, coordinates of each pallid sturgeon location were determined using a hand-held global positioning unit. Location was converted to river kilometer using geographic information system (GIS) software. Fixed receiving stations were placed near Cartersville and Intake diversions and the confluence with the Big Horn, Tongue, Powder, and Missouri rivers to assess movements over diversion dams, into tributaries, and out of the Yellowstone River. To minimize bias related to acclimation to a riverine environment following stocking only data collected beginning 180 days post-stocking for group 1 and 150 to 1850 days post-stocking for group 2 were included in analyses. Between 48 and 69 individual telemetered pallid sturgeon contributed data to analyses during each season.

Each year was divided into seasons based on the hydrograph and broad changes in water temperature (Figure 2). Spring was defined as the period from April 1 to the date at which discharge at Miles City increased to above 15,000 cfs and encompassed lowland runoff. Runoff occurred when discharge was greater than 15,000 cfs at Miles City and encompassed mountain runoff. Summer ranged from the date when discharge decreased below 15,000 cfs to September 30. Winter was defined as the period from October 1 to March 30. Water temperatures typically range from $5^{o}C$ to $20^{o}C$ during spring, from $15^{o}C$ to $30^{o}C$ during runoff and summer, and are less than $10^{o}C$ during winter.

Total and net movement rates (km/d) were calculated for each telemetered juvenile pallid sturgeon. Total movement rate during each season was calculated by dividing the distance in river kilometers between successive relocations for a given fish by the number of days that had elapsed between successive relocation (White and Garrot 1990). Net movement rate during each season was calculated by dividing the change in river kilometer between successive relocations by the number of days that had elapsed between relocations such that a positive rate indicates upstream movement and a negative rate indicates downstream movement (Bramblett 1996). Because additional movement may occur between relocations, calculated movement rates represent the minimum movement for the time period between relocations. Median monthly movement rates were compared using a Kruskal-Wallis test (Zar 1999). When significant differences were detected, Dunn’s multiple comparisons test was used to determine which seasonal rates differed (Zar 1999). Seasonal directionality of net movements were assessed with a Wilcoxon signed rank test (Zar 1999).

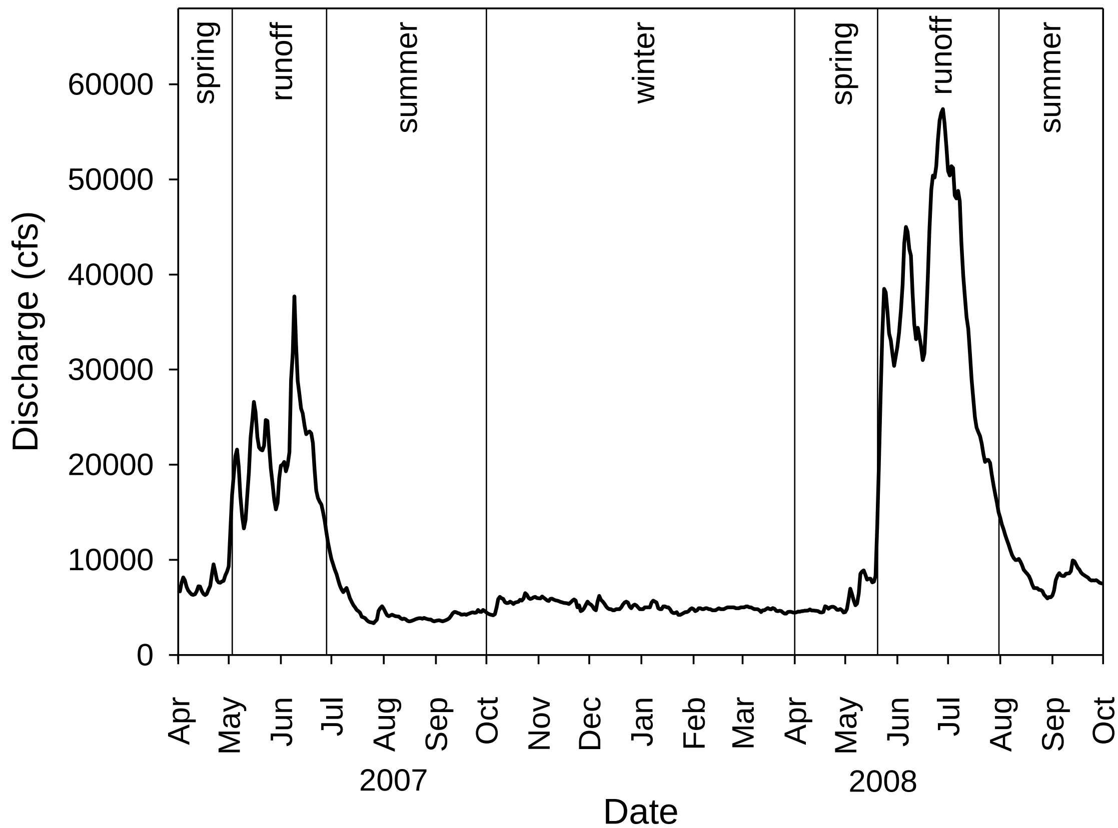

Figure 2. Discharge and delineation of seasons for the lower Yellowstone River.

Habitat types were delineated using low-level 1:24,000 scale color infrared aerial photographs and physical features inventory (Natural Resources Conservation Service 2002; DTM and AGI 2008a), geologic maps (Montana Bureau of Mines and Geology 1979-2001b), and GIS software. Habitat type classification was predicated on geomorphic function (i.e. pool, crossover, side channel) and bank material (i.e. bedrock, alluvium, rip-rap) as described by Jaeger et al. (2005). Habitats units were delineated under conditions corresponding to 1) base flow and 2) runoff time periods. Habitats units were delineated during base flow considering all areas inundated by water in aerial photographs taken in August 2001 at discharges between 3500 and 4200 cfs (Natural Resources Conservation Service 2002). Habitats delineated during runoff include all areas within the active channel that were comprised of unvegetated substrate and bounded by woody vegetation stands, terrace margins, or bedrock valley wall (DTM and AGI 2008b). Total acreage of each habitat type available during base flow and runoff periods was quantified using GIS software. Physical characteristics of each habitat type were determined using a stratified random sampling design. Five to 10 habitat units of each habitat type were randomly selected for physical characterization. Within each selected habitat unit, velocity, depth, and substrate were measured at 100 randomly selected points.

Habitat use by individual fish was calculated for each season as the proportion of relocations that were made within each habitat type (Manly et al. 2002). Use was determined by projecting coordinates of each fish relocation on a map of delineated habitats using GIS software. Base flow habitat delineations were used for relocations collected during spring, summer, and winter and runoff delineations were used for relocations collected during runoff. Chi-square tests with log-likelihood test statistics (Manly et al. 2002) were used to test the null hypothesis of seasonal selection in proportion to availability for different habitat types. Selection ratios and simultaneous 95% Bonferroni confidence intervals (Manly et al. 2002) were used to determine level of selection for specific resource categories. Selection ratios for the population were obtained by averaging selection ratios calculated for individual telemetered fish (Manly et al. 2002).

Hatchery-reared juvenile monitoring and survival estimation: A targeted sampling approach based on observed patterns of habitat selection was used to maximize the number of pallid sturgeon captured. Five bluff pools and one terrace pool between the confluence with the Missouri River and Intake Diversion were sampled with drifting trammel nets during August and September 2006 to 2008. Trammel nets were 6 feet tall by 125 feet long with 1-inch inner mesh and 8-inch outer mesh.

Survival probabilities for hatchery-reared juvenile pallid sturgeon in the Yellowstone River were estimated using data from both mark-recapture and telemetry efforts. These data were not suitable for analysis using techniques that combine mark-recapture data with known-fate data because the detection probabilities for telemetered fish were not expected to equal 1. The dataset consisted of recaptures and telemetry relocations for juvenile pallid sturgeon in the lower Yellowstone River from 1998 through 2008. Releases occurred beginning in August 1998, and the first recaptures occurred in spring 1999 with first telemetry ‘recaptures’ occurring in Spring 2007. Data were analyzed for 308 recaptured fish and 57 telemetered fish with complete covariate data, as well as 34,656 individuals that were never recaptured after release. Only individuals released at Intake Diversion or below were included in analysis as sampling only occurred in this reach. Fish were classified by stocking category (fingerling, spring yearling, summer yearling, or 2-year-old), recapture type (recapture or telemetry), and age. Fourteen sampling occasions were established as shown in Table 1. For all fish, capture probability was fixed to 0 on occasions 4 and 9 as no sampling occurred then. For telemetered fish, capture probability was fixed to 0 on occasions 2 through 10 because telemetry sampling did not begin until occasion 11.

Apparent annual survival and capture probabilities were estimated using Cormack-Jolly-Seber capture-recapture models (Pollock et al. 1990, Lebreton et al. 1992, Williams et al. 2002) and Program MARK (White and Burnham 1999). All models estimated different detection probabilities for fish that were physically recaptured versus those ‘recaptured’ using telemetry as telemetry detection probabilities were known to be higher. All possible combinations of age, season, and stocking category effects on survival probability were included as candidate models. For each of these combinations, capture probability was allowed to vary by sampling occasion or by season. Models that allowed variation by sampling occasion were constructed with either 1) occasion-specific variation for recaptures only, or 2) occasion-specific variation for both recaptures and telemetry relocations. This resulted in an initial model set of 21 models. The top-ranked models (which all included the effect of age on survival rate) were additionally run with a logarithmic age function in place of the linear age function to determine whether the logarithmic function resulted in a better fit of the data.

Table 1. Sampling occasions for mark-recapture and telemetry sampling of juvenile pallid sturgeon in the Lower Yellowstone River.

Occasion	Begin	End	Duration (days)	Interval (Months)
1	8/11/1998	8/11/1998	0	0
2	5/4/1999	7/7/1999	64	9.93
3	8/17/2000	10/17/2000	61	15.65
4	7/18/2002	9/18/2002	62	23.35
5	8/7/2003	8/28/2003	21	12.15
6	8/9/2004	11/18/2004	101	13.6
7	4/12/2005	5/18/2005	36	7.12
8	7/19/2005	11/2/2005	106	4.43
9	3/28/2006	5/1/2006	34	7.2
10	7/13/2006	11/18/2006	128	5.13
11	4/3/2007	5/16/2007	43	7.38
12	8/9/2007	10/4/2007	56	4.48
13	3/26/2008	5/7/2008	42	7.43
14	7/30/2008	8/28/2008	29	3.98

Goodness-of-fit (GOF) could only be evaluated for models that did not use individual covariates (i.e., age). A GOF test of the most complex model without the age covariate ($\phi_{stock\_category+season}\, p_{recap\_type+t}$) resulted in an estimated variance inflation factor ($\hat{c}$) of 3.97. Age was known to be a very important covariate for survival probability because models without age were approximately 90 AIC units removed from equivalent models that included the age covariate. Thus, the model used to test GOF was known to be lacking an important explanatory variable and hence, poor fit was expected. We investigated the robustness of the top-ranked model to changes in $\hat{c}$, and found that it remained the top model for values of $\hat{c}$ up to approximately 2.5.

Monthly survival rates generated by the top-ranked model were converted to annual rates by exponentiating the monthly rates according to the number of months in the appropriate intervals and recalculating standard errors using the delta method and program R (R Development Core Team 2008). Separate capture probabilities were estimated for each sampling occasion for recaptures and telemetry.

<u>Extant adult abundance and survival estimation</u>: Abundance and survival of wild adult pallid sturgeon within RPMA 2 was estimated using existing mark-recapture records for

PIT tagged wild adult sturgeon that have been captured for various reasons (primarily as broodstock for hatchery-rearing efforts) since 1988. Two analyses were conducted for different sets of adult mark-recapture data.

Analysis 1: The first analysis involved working with as complete a dataset as possible. The original recaptures file contained 677 recapture records for 255 individuals. We established 40 sampling occasions comprising each spring and fall from Fall 1988 through Fall 2008, excluding Fall 2000 when there were no captures (Table 2). These sampling occasions resulted in an input file with 651 captures for 250 individuals. The general framework used to designate sampling occasions was to assign April and May captures to a Spring sampling occasion, and August through November captures to a Fall sampling occasion. Exceptions occurred in 1989 and 1990 when February and March captures were included as Spring sampling because no captures occurred in April and May. Abundance was estimated using the Jolly-Seber model type (Jolly 1965, Seber 1965) in program MARK (White and Burnham 1999) with the POPAN formulation (Schwarz and Arnason 1996). Models varied according to whether apparent survival probability ($\hat{\phi}$) and capture probability (p) were constant, depended on season, or depended on sampling occasion. The 'probability of entry' (*p-ent)* was fixed to 0 because no new fish were being born or immigrating to this population.

Analysis 2: We created a subset dataset for individuals with known sex and weight. Sex was used as a group covariate and weight was an individual covariate. Weight was calculated as the average of all weights measured for an individual at various sampling occasions. The subset dataset included 542 captures for 159 individuals. Thirty-six sampling occasions were used (occasions 2, 4, 6, and 8 from Analysis 1 were eliminated because no captures occurred). Jolly-Seber analysis was again used with the POPAN formulation in program MARK. Models in which $\hat{\phi}$ varied by season (winter versus summer), group (males versus females), and weight (included as an individual covariate) were included. Capture probability (p) varied by sampling occasion in all models. As with the previous analysis, the 'probability of entry' (*p-ent)* was fixed to 0 because no new fish were being born or immigrating to this population.

Table 2. Dates included in each sampling occasion, length of sampling occasions, and length of intervals between sampling occasions.

Occasion	**Begin Date**	**End Date**	**Length (days)**	**Midpoint**	**Interval (months)**
1	8/3/1988	8/4/1988	1	8/3/1988	
2	3/10/1989	3/21/1989	11	3/15/1989	7.47
3	9/13/1989	9/13/1989	0	9/13/1989	6.05
4	2/10/1990	2/10/1990	0	2/10/1990	5.00
5	9/13/1990	9/17/1990	4	9/15/1990	7.23
6	5/31/1991	5/31/1991	0	5/31/1991	8.60
7	9/22/1991	10/10/1991	18	10/1/1991	4.10
8	4/10/1992	4/10/1992	0	4/10/1992	6.40
9	9/15/1992	10/29/1992	44	10/7/1992	6.00
10	4/15/1993	5/30/1993	45	5/7/1993	7.08
11	9/9/1993	9/29/1993	20	9/19/1993	4.48
12	4/30/1994	5/18/1994	18	5/9/1994	7.73
13	9/7/1994	11/16/1994	70	10/12/1994	5.20
14	4/24/1995	5/31/1995	37	5/12/1995	7.08
15	8/24/1995	10/12/1995	49	9/17/1995	4.27
16	4/27/1996	5/15/1996	18	5/6/1996	7.72
17	8/28/1996	10/16/1996	49	9/21/1996	4.62
18	4/22/1997	5/30/1997	38	5/11/1997	7.72
19	9/16/1997	10/21/1997	35	10/3/1997	4.85
20	4/14/1998	4/28/1998	14	4/21/1998	6.65
21	8/11/1998	10/6/1998	56	9/8/1998	4.67
22	4/12/1999	5/19/1999	37	4/30/1999	7.82
23	8/4/1999	8/26/1999	22	8/15/1999	3.55
24	4/11/2000	5/3/2000	22	4/22/2000	8.37
25	4/24/2001	5/9/2001	15	5/1/2001	12.48
26	8/13/2001	8/13/2001	0	8/13/2001	3.45
27	4/22/2002	5/16/2002	24	5/4/2002	8.80
28	8/7/2002	9/25/2002	49	8/31/2002	3.98
29	4/22/2003	5/14/2003	22	5/3/2003	8.15
30	9/5/2003	11/19/2003	75	10/12/2003	5.42
31	4/20/2004	4/29/2004	9	4/24/2004	6.50
32	9/14/2004	11/9/2004	56	10/12/2004	5.68
33	4/7/2005	4/29/2005	22	4/18/2005	6.27
34	8/11/2005	10/24/2005	74	9/17/2005	5.07
35	4/17/2006	5/11/2006	24	4/29/2006	7.47
36	8/15/2006	11/14/2006	91	9/29/2006	5.12
37	4/5/2007	5/9/2007	34	4/22/2007	6.82
38	9/20/2007	9/20/2007	0	9/20/2007	5.03
39	4/8/2008	5/29/2008	51	5/3/2008	7.55
40	8/12/2008	10/15/2008	64	9/13/2008	4.42

RESULTS

Movements and habitat selection: Total movement rates of juvenile pallid sturgeon varied among seasons ($P < 0.001$; Figure 3). Median daily movement rate was highest during runoff followed by spring, summer, and winter. Movement rates among runoff, spring, and summer were not significantly different although movement rates in winter were significantly lower than during runoff or spring. Net movements did not vary among seasons ($P = 0.083$) and there was no significant directionality of movement during spring, runoff, and summer ($P > 0.05$; Figure 4); both up and downstream movements occurred. Predominately downstream movements occurred during winter ($P = 0.001$).

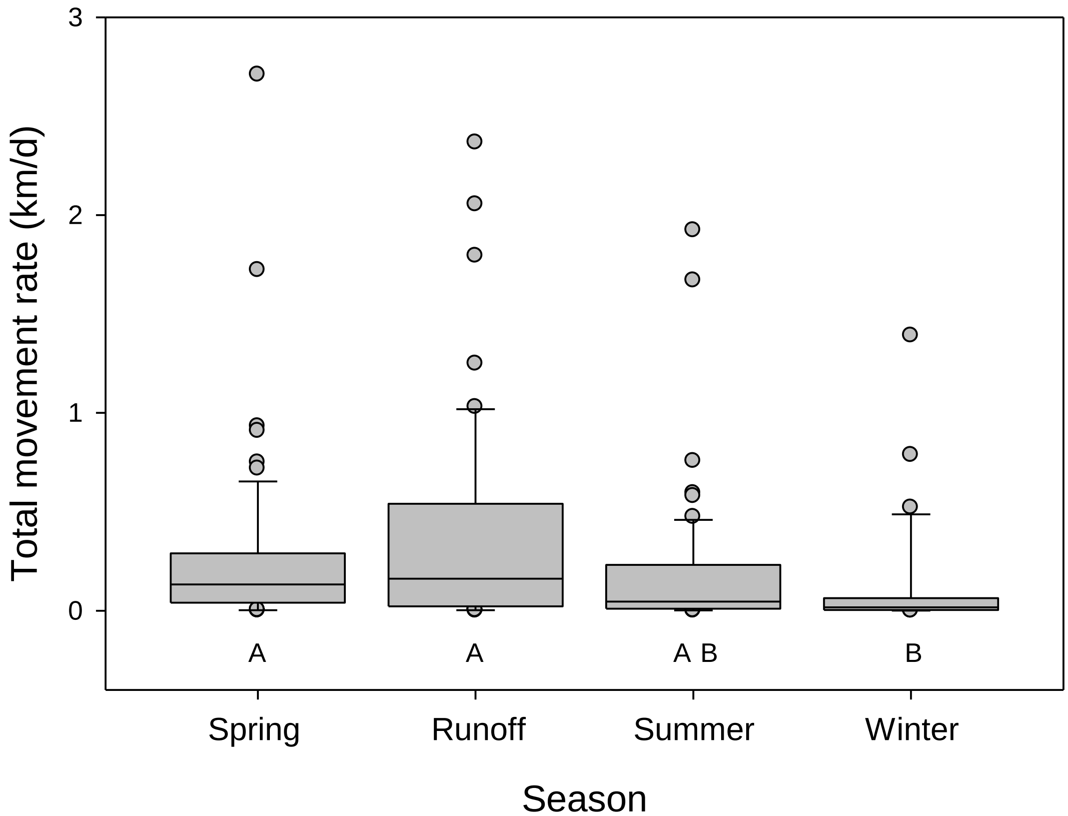

Figure 3. Total movement rates by season of juvenile telemetered pallid sturgeon in the Yellowstone River. Lines within boxes represent medians, boxes represent 25th and 75th percentiles, whiskers represent 10th and 90th percentiles, and circles represent outliers beyond the 10th and 90th percentiles. Movement rates in seasons with the same letters are not significantly different ($P \leq 0.05$).

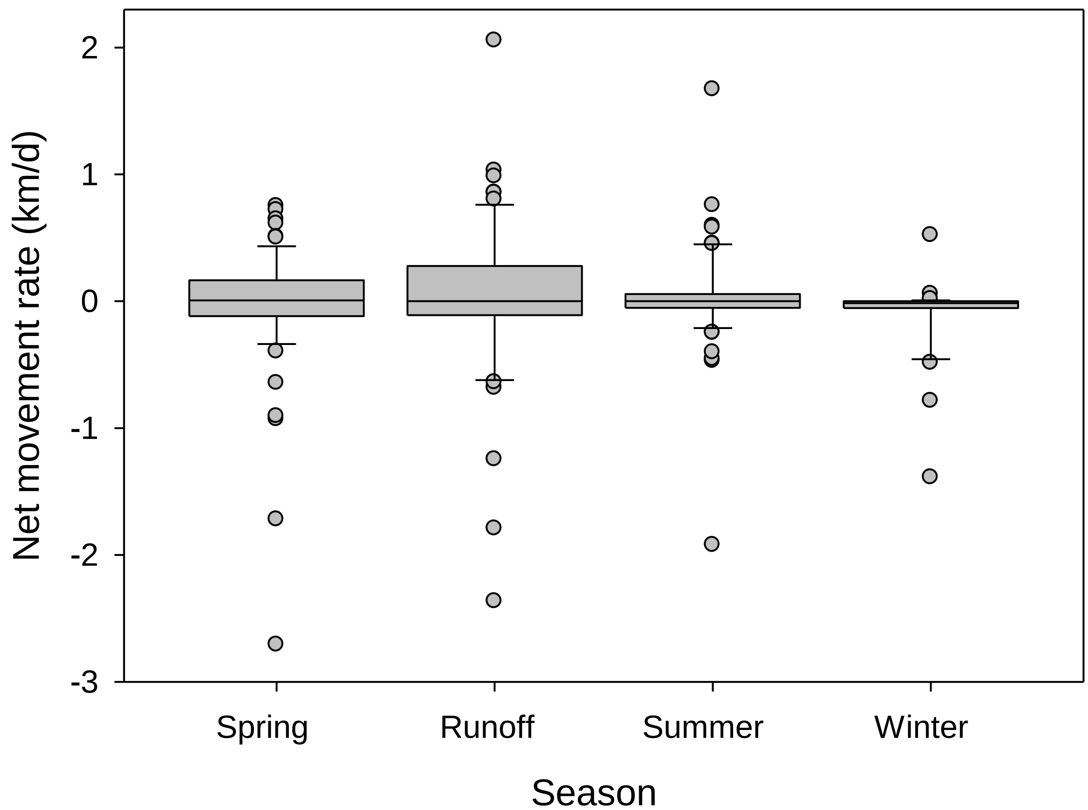

Figure 4. Total movement rates by season of juvenile telemetered pallid sturgeon in the Yellowstone River. Lines within boxes represent medians, boxes represent 25^{th} and 75^{th} percentiles, whiskers represent 10^{th} and 90^{th} percentiles, and circles represent outliers beyond the 10^{th} and 90^{th} percentiles. Negative values indicate predominately downstream movements, positive values indicate predominately upstream movements, and values near zero indicate no predominate directionality of movement.

Habitat selection of juvenile pallid sturgeon varied among and within seasons (Figure 5). Pallid sturgeon did not use habitat types in proportion to their availability during any season ($P < 0.001$); however, most habitats were used in proportion to their availability during most seasons. Bluff pools were positively selected during summer and winter although their armored equivalents, valley margin rip-rap pools, were avoided during all seasons other than winter when they were used in proportion to availability. Side channels were used in proportion to availability during runoff when discharges are highest and they are most abundant but were avoided during all other seasons.

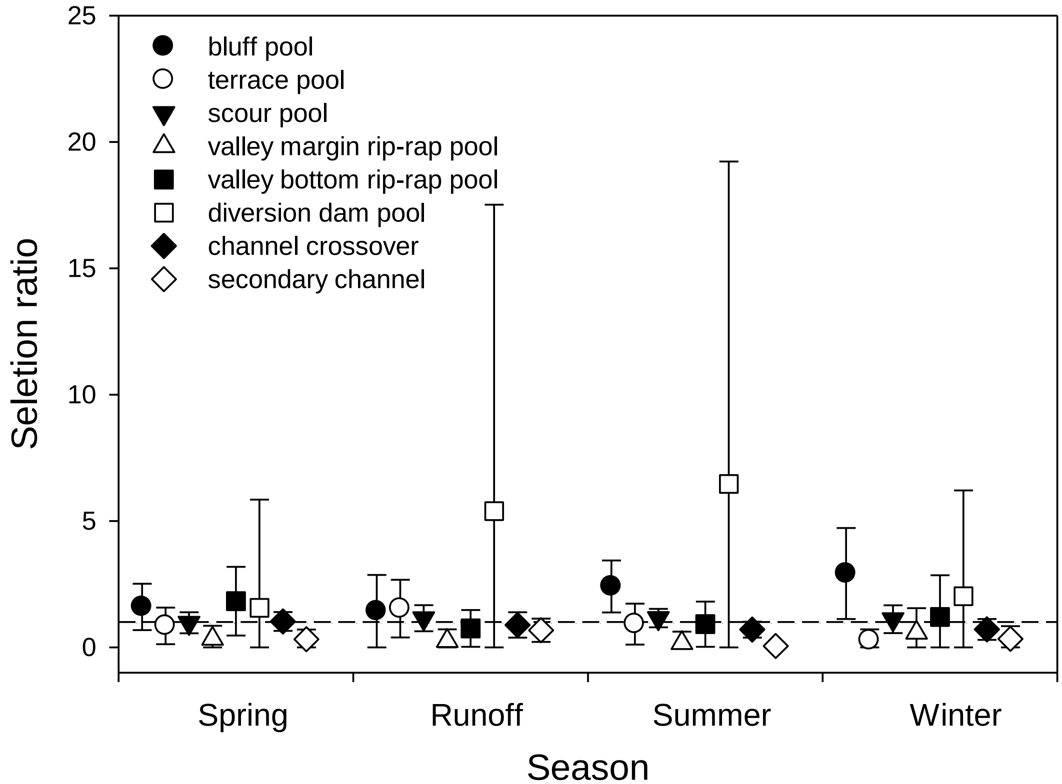

Figure 5. Seasonal selection ratios and simultaneous 95% Bonferroni confidence intervals of habitat types by telemetered juvenile pallid sturgeon in the Yellowstone River. Values larger than 1 indicate positive selection, values less than 1 indicate negative selection, and values equal to 1 indicate use in proportion to availability.

Differences in width, average and maximum depth, average and bottom velocity, and percentage of boulder and bedrock substrate occur among habitat types (ANOVA; $P < 0.001$, Table 3). Mainstem pools at the valley margin (bluff, valley margin rip-rap) were generally longer and had lower average and bottom velocities than pools in the valley bottom (scour, valley bottom rip-rap). Armored pools (valley margin rip-rap, valley bottom rip-rap) generally had higher maximum and average depths, greater variability of depths, and a higher percentage of boulder and bedrock substrates than their unarmored equivalents (bluff, scour pools). Terrace pools had characteristics of pools at the valley margin and pools in the valley bottom. Channel crossovers were shorter, shallower, had higher velocities, and a lower percentage of boulder and bedrock substrates than other mainstem habitat types. Secondary channels were generally narrower and shallower than mainstem habitats. Average bottom (0.61 m/s) and mean (0.78 m/s) velocities in secondary channels were lower than those (0.69 and 0.86 m/s, respectively) in mainstem habitats (t-test, $P < 0.001$).

Table 3. Characteristics of Yellowstone River habitats. Standard errors are displayed in parentheses.

Habitat	Mean length (km)	Mean width (m)	Mean depth (m)	Mean max. depth (m)	Mean velocity (m/s)	Mean bottom velocity (m/s)	Mean % Bldr./ Bdrck.
Bluff pool	1.3	147 (1.71)	1.48 (0.04)	3.36 (0.31)	0.69 (0.02)	0.53 (0.01)	19.4 (6.43)
Terrace pool	0.9	138 (2.98)	1.52 (0.03)	3.11 (0.35)	0.92 (0.02)	0.72 (0.02)	16.6 (2.60)
Scour pool	0.7	115 (1.09)	1.27 (0.23)	2.35 (0.22)	0.92 (0.02)	0.73 (0.01)	3.2 (3.17)
Valley margin rip-rap pool	1.3	156 (0.56)	1.83 (0.46)	4.25 (0.48)	0.68 (0.01)	0.52 (0.01)	27.3 (7.68)
Valley bottom rip-rap pool	0.9	156 (1.00)	1.67 (0.41)	3.68 (0.56)	0.81 (0.02)	0.65 (0.02)	18.8 (9.82)
Channel Crossover	0.4	145 (1.64)	0.96 (0.02)	1.96 (0.20)	1.16 (0.02)	0.95 (0.02)	1.0 (0.67)
Secondary Channel	0.8	82 (0.84)	0.64 (0.01)	1.51 (0.13)	0.78 (0.01)	0.61 (0.01)	3.5 (2.30)

Hatchery-reared juvenile monitoring and survival estimation: Hatchery-reared pallid sturgeon from most release groups were recaptured in the Yellowstone River each year (Figure 6). A total of 105 hatchery-reared pallid sturgeon were captured in 12.5 hours during 2006, 179 fish were captured in 20.5 hours during 2007, and 140 fish were captured in 39.5 hours during 2008. Catch rates were similar in 2006 (8.4 fish per hour) and 2007 (8.7 fish per hour) but were lower in 2008 (3.5 fish per hour). Catch rate appeared to be negatively correlated with discharge, although data were sparse and the relationship was not statistically significant (P = 0.076; Figure 7).

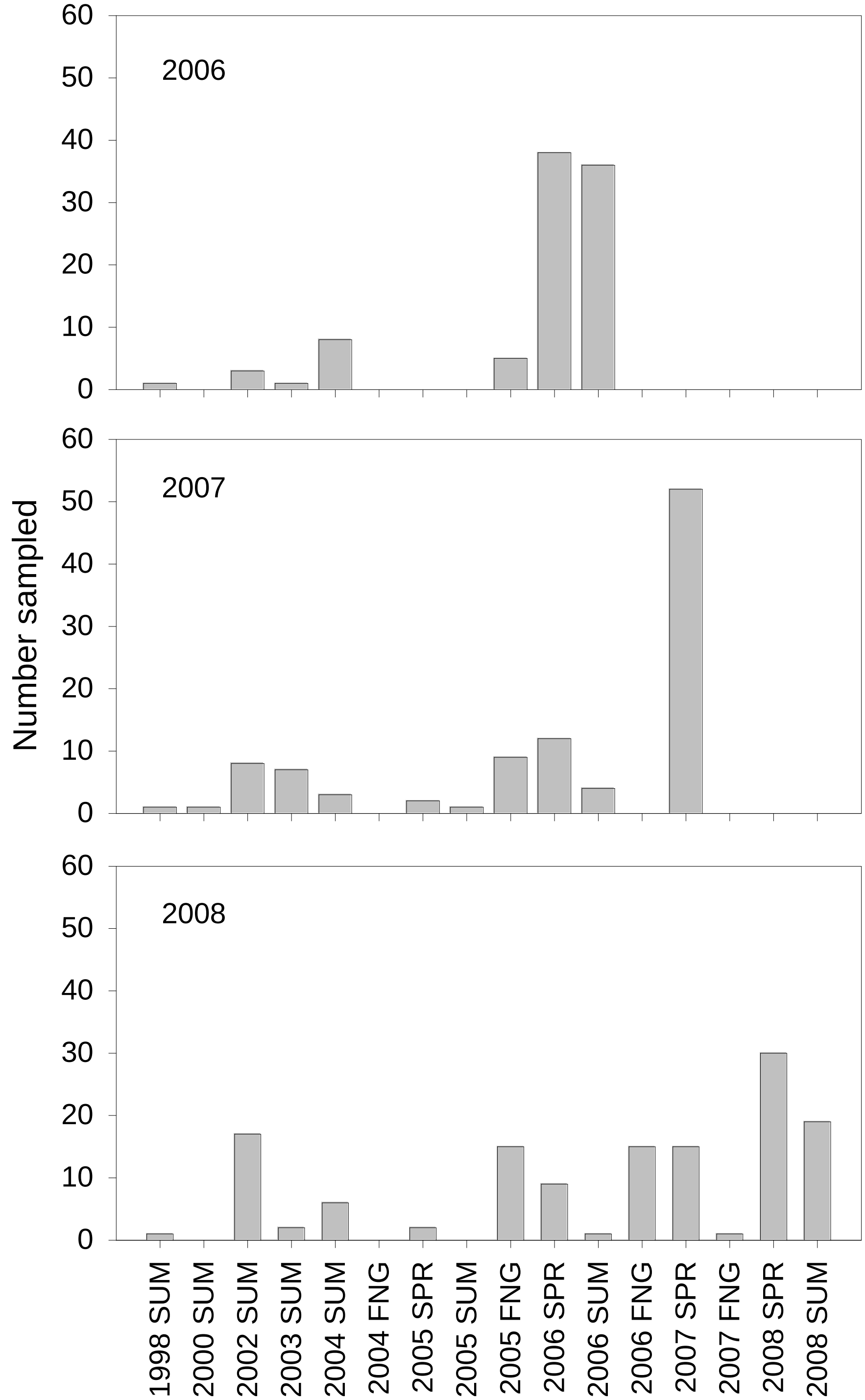

Figure 6. Recaptures of hatchery-reared pallid sturgeon release groups in 2006, 2007, and 2008.

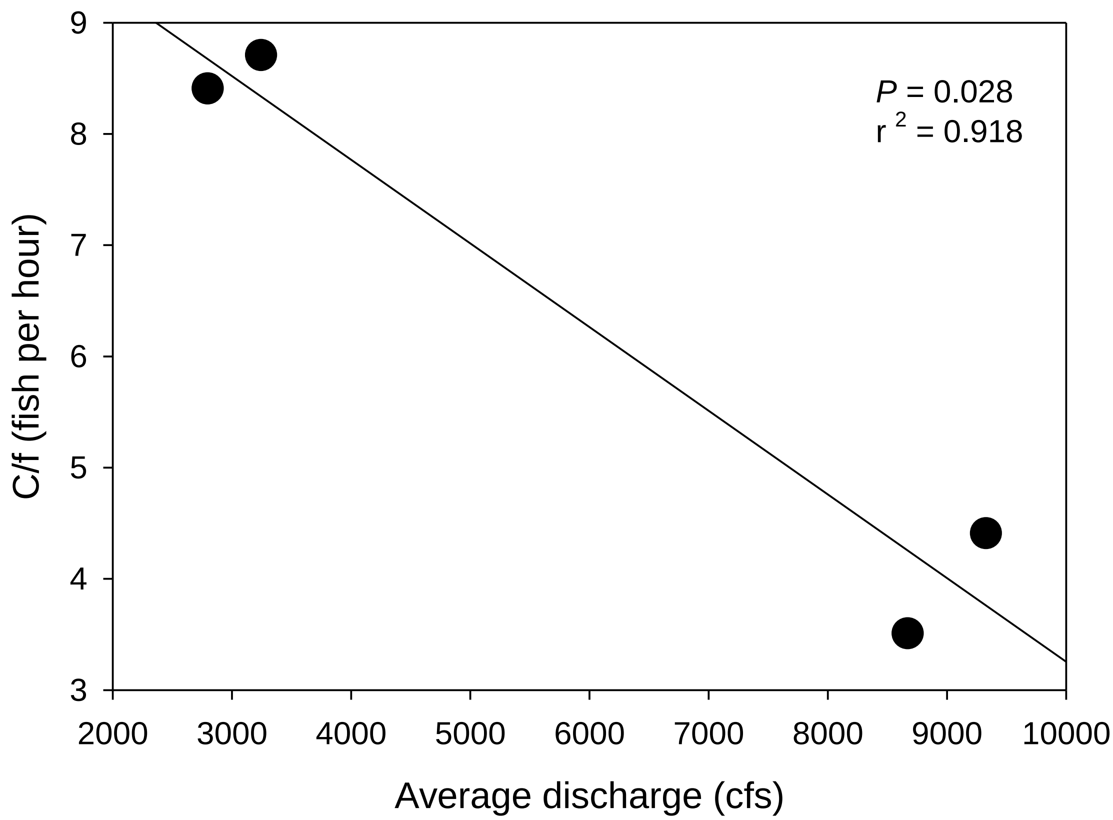

Figure 7. Relationship between average pallid sturgeon catch rate and average Yellowstone River discharge during the sampling period in 2006, 2007, and 2008.

The top-ranked survival model ($\phi_{stock\_category+\ln age+season}\, p_{recap\_type+t}$) had an AIC_c weight of 0.78 and was the only model that was strongly supported by the data. Estimates of effect size for all model covariates are provided in Table 4 and monthly survival estimates generated from this model for each of the four stocking categories are described in Tables 5 to 8. Annual survival probability following release was highest for 2-year-olds, followed by spring and summer yearlings, which had similar initial survival probabilities, and fingerlings, which had the lowest survival probabilities (Table 9). Survival rate increased with age following a logarithmic function; survival sharply increased then approached a threshold at ages greater than 4 years. Survival probabilities were lower in winter (β_{winter} = -1.045) and the 95% confidence interval for the effect size did not overlap zero (-2.029, -0.061). Detection probabilities were higher for telemetry 'captures' and ranged from 0.583 to 0.941 (Table 10). Detection probability also varied among sampling occasions and models using season (spring vs. fall) to explain temporal variation in detection were not as strongly supported as models allowing non-seasonal variation.

Table 4. Estimated effect sizes for covariates in the top-ranked model for juvenile pallid sturgeon in the Lower Yellowstone River, 1998-2008.

Parameter	*Estimate*	*SE*	*95% LCL*	*95% UCL*
Survival probability (ϕ) portion of top-ranked model				
Intercept (2-year-olds)	-5.964	1.233	-8.381	-3.548
Adjustment for fingerlings	1.066	0.397	0.288	1.844
Adjustment for spring yearlings	0.236	0.334	-0.418	0.890
Adjustment for summer yearlings	0.262	0.272	-0.271	0.795
Slope of logarithmic age effect	1.411	0.171	1.076	1.745
Season effect (winter)	-1.045	0.502	-2.029	-0.061
Detection probability (*p*) portion of top-ranked model				
Intercept (Sampling occasion 14)	-3.522	0.188	-3.890	-3.155
Adjustment for telemetry detections	4.504	0.490	3.543	5.465
Adjustment for occasion 2 - Recaptures	-0.264	0.536	-1.314	0.786
Adjustment for occasion 3- Recaptures	0.113	0.617	-1.097	1.323
Adjustment for occasion 4- Recaptures	0.000	0.000	0.000	0.000
Adjustment for occasion 5- Recaptures	-2.844	1.015	-4.834	-0.854
Adjustment for occasion 6- Recaptures	-0.975	0.363	-1.686	-0.263
Adjustment for occasion 7- Recaptures	-1.604	0.387	-2.362	-0.846
Adjustment for occasion 8- Recaptures	-2.116	0.468	-3.034	-1.198
Adjustment for occasion 9- Recaptures	0.000	0.000	0.000	0.000
Adjustment for occasion 10- Recaptures	-0.527	0.196	-0.911	-0.143
Adjustment for occasion 11- Recaptures	-2.960	0.592	-4.120	-1.800
Adjustment for occasion 12- Recaptures	0.364	0.154	0.063	0.666
Adjustment for occasion 13- Recaptures	-2.385	0.467	-3.299	-1.470
Adjustment for occasion 11 - Telemetry	-0.645	0.551	-1.726	0.435
Adjustment for occasion 12 – Telemetry	1.783	1.063	-0.301	3.866
Adjustment for occasion 13 - Telemetry	1.282	0.794	-0.275	2.840

Table 5. Estimated monthly survival probabilities for sturgeon released as fingerlings in the Lower Yellowstone River. Estimates are for fish released in Fall 2004 at an age of approximately 81 days.

Interval	**Length (months)**	**Monthly ϕ**	**SE (ϕ)**	**95% LCL**	**95% UCL**
Fall 2004 to Spring 2005	7.13	0.599	0.033	0.534	0.661
Spring 2005 to Fall 2005	4.43	0.958	0.018	0.906	0.982
Fall 2005 to Spring 2006	7.20	0.940	0.011	0.916	0.958
Spring 2006 to Fall 2006	5.13	0.986	0.007	0.963	0.994
Fall 2006 to Spring 2007	7.37	0.975	0.007	0.957	0.986
Spring 2007 to Fall 2007	4.50	0.992	0.004	0.978	0.997
Fall 2007 to Spring 2008	7.43	0.985	0.005	0.971	0.992
Spring 2008 to Summer 2008	3.97	0.995	0.003	0.985	0.998

Table 6. Estimated monthly survival probabilities for sturgeon released as spring yearlings in the Lower Yellowstone River. Estimates are for fish released in Spring 2005 at an age of approximately 287 days.

Interval	**Length (months)**	**Monthly ϕ**	**SE (ϕ)**	**95% LCL**	**95% UCL**
Spring 2005 to Fall 2005	4.43	0.905	0.026	0.841	0.945
Fall 2005 to Spring 2006	7.20	0.870	0.025	0.814	0.911
Spring 2006 to Fall 2006	5.13	0.967	0.010	0.941	0.982
Fall 2006 to Spring 2007	7.37	0.943	0.012	0.914	0.963
Spring 2007 to Fall 2007	4.50	0.983	0.006	0.967	0.991
Fall 2007 to Spring 2008	7.43	0.966	0.009	0.945	0.979
Spring 2008 to Summer 2008	3.97	0.989	0.004	0.977	0.994

Table 7. Estimated monthly survival probabilities for sturgeon released as summer yearlings in the Lower Yellowstone River. Estimates are for fish released in Summer 1998 at an age of approximately 420 days.

Interval	**Length (months)**	**Monthly ϕ**	**SE (ϕ)**	**95% LCL**	**95% UCL**
Summer 1998 to Spring 1999	9.93	0.898	0.011	0.875	0.918
Spring 1999 to Fall 2000	15.63	0.960	0.006	0.946	0.970
Fall 2000 to Summer 2002	23.37	0.977	0.003	0.969	0.983
Summer 2002 to Summer 2003	12.13	0.988	0.003	0.982	0.992
Summer 2003 to Fall 2004	13.60	0.991	0.002	0.986	0.995
Fall 2004 to Spring 2005	7.13	0.989	0.003	0.982	0.994
Spring 2005 to Fall 2005	4.43	0.996	0.002	0.991	0.998
Fall 2005 to Spring 2006	7.20	0.991	0.003	0.984	0.995
Spring 2006 to Fall 2006	5.13	0.997	0.001	0.992	0.999
Fall 2006 to Spring 2007	7.37	0.993	0.002	0.987	0.996
Spring 2007 to Fall 2007	4.50	0.997	0.001	0.993	0.999
Fall 2007 to Spring 2008	7.43	0.994	0.002	0.988	0.997
Spring 2008 to Summer 2008	3.97	0.997	0.001	0.994	0.999

Table 8. Estimated monthly survival probabilities for sturgeon released as 2-year-olds in the Lower Yellowstone River. Estimates are for fish released in Fall 2006 at an age of approximately 850 days.

Interval	**Length (months)**	**Monthly ϕ**	**SE (ϕ)**	**95% LCL**	**95% UCL**
Fall 2006 to Spring 2007	7.37	0.936	0.015	0.900	0.959
Spring 2007 to Fall 2007	4.50	0.980	0.009	0.951	0.992
Fall 2007 to Spring 2008	7.43	0.960	0.009	0.937	0.975
Spring 2008 to Summer 2008	3.97	0.986	0.006	0.967	0.995

Table 9. Comparison of estimated annual survival rates and standard errors (SE) for hatchery-reared juvenile pallid sturgeon released as fingerlings (release age of 81 days for this example), spring yearlings (release age of 299 days), summer yearlings (release age of 399 days), and 2-yr-olds (release age of 832 days). Rates are for as many years post-release as data were available.

Yrs. post-release	**Annual survival estimate**	**SE**	**95% LCL**	**95% UCL**
Fingerlings				
0-1	0.02	0.01	0.01	0.04
1-2	0.61	0.06	0.49	0.73
2-3	0.80	0.05	0.70	0.91
Spring Yearlings				
0-1	0.23	0.05	0.14	0.33
1-2	0.56	0.04	0.48	0.64
2-3	0.71	0.05	0.61	0.81
Summer Yearlings				
0-1	0.32	0.04	0.24	0.40
1-2	0.61	0.05	0.52	0.70
2-3	0.73	0.03	0.67	0.80
3-4	0.75	0.03	0.69	0.82
4-5	0.86	0.03	0.80	0.91
5-6	0.90	0.02	0.85	0.94
6-7	0.90	0.02	0.85	0.94
7-8	0.92	0.02	0.88	0.96
8-9	0.93	0.02	0.90	0.97
9-10	0.94	0.02	0.91	0.98
2-year-olds				
0-1	0.56	0.08	0.40	0.71
1-2	0.70	0.06	0.57	0.82

Table 10. Estimated detection probabilities for juvenile pallid sturgeon in the Lower Yellowstone River, 1998 – 2008.

Sampling occasion	***p***	**SE(*p*)**	***p***	**SE(*p*)**
	Recapture		Telemetry	
Spring 1999	0.022	0.011	-	-
Fall 2000	0.032	0.019	-	-
Summer 2002	-	-	-	-
Summer 2003	0.002	0.002	-	-
Fall 2004	0.011	0.004	-	-
Spring 2005	0.006	0.002	-	-
Fall 2005	0.004	0.002	-	-
Spring 2006	-	-	-	-
Fall 2006	0.017	0.003	-	-
Spring 2007	0.002	0.001	0.583	0.071
Fall 2007	0.041	0.007	0.941	0.056
Spring 2008	0.003	0.001	0.906	0.063
Summer 2008	0.029	0.005	0.727	0.093

Extant adult abundance and survival estimation:

Analysis 1: The top-ranked model (w = 0.95) indicated important seasonal variation in $\hat{\phi}$ and temporal variation in p. This model estimated a negative adjustment for winter survival relative to summer survival ($\hat{\beta}_{winter}$ = -1.91), but the 95% CI on that estimate (-10.49, 6.68) included zero, indicating low precision. Based on the top-ranked model, $\hat{\phi}$ varied from 0.988 (0.944, 0.999) for intervals falling over winter months (1 October – 1 April) to 0.998 (0.981, 0.998) for intervals only including summer months (2 April – 30 September). Capture probability varied by sampling occasion from 0.002 (<0.001, 0.015) to 0.418 (0.309, 0.535). The 'super-population' size (number of individuals that were ever part of the population; see Schwarz and Arnason 1996) was estimated to be 531 (442, 663). According to derived estimates of $\hat{N}$ for each sampling occasion, estimated population size declined from 531 at start of sampling in 1988 to 125 (100, 150) by the most recent sampling occasion in Fall 2008 (Table 11).

Table 11. Estimates of population size ($\hat{N}$) at each sampling occasion, derived from model parameters estimated for the top-ranked model for wild adult pallid sturgeon in RPMA 2, 1988-2008.

$\hat{N}$	**SE ($\hat{N}$)**	**95% LCL**	**95% UCL**
531	56	422	640
502	48	408	595
495	53	391	599
471	41	390	551
461	46	371	551
434	40	355	513
431	46	341	520
401	29	343	459
396	30	337	455
370	26	319	422
367	24	320	415
345	23	300	390
341	21	300	382
321	20	282	360
319	18	283	354
299	18	264	334
296	16	265	328
278	16	246	310
275	14	247	303
257	20	218	297
255	16	225	286
239	18	203	276
238	15	208	268
224	16	192	255
211	12	188	234
210	13	184	235
197	13	172	222
196	15	167	225
184	13	158	210
182	17	150	214
171	12	147	194
169	14	141	196
158	12	134	182
157	12	133	180
147	12	122	171
145	12	122	169
136	14	108	163
134	13	110	159
126	14	98	154
125	13	100	150

Analysis 2: When only individuals of know sex and weight were included, $\hat{\phi}$ in the top-ranked model (w = 0.45) varied by sex. The model without sex as a covariate was approximately half as likely, given the data (w = 0.25). The effect of sex on $\hat{\phi}$ was positive: $\hat{\beta}_{male}$ = 0.79, indicating higher survival for male pallid sturgeon, but the confidence interval included zero (95% CI ($\hat{\beta}_{male}$) = -0.05, 1.64), indicating that the effect of sex was not estimated with high precision. Estimated male survival probability was 0.998 (0.996, 0.999) and female survival probability was 0.996 (0.994, 0.998).

Capture probability (p) varied with sampling occasion, as in Analysis 1. Estimates of p ranged from 0.005 (<0.001, 0.032) to 0.414 (0.316, 0.519). The super-population size (number of individuals that were ever part of the population) was estimated to be approximately 123 (108, 158) for males and 101 (88, 144) for females. According to derived estimates of $\hat{N}$ for each sampling occasion, estimated male population size declined from 123 (100, 147) at start of sampling in 1988 to 81 (66, 96) by the most recent sampling occasion in Fall 2008 (Table 12). For females, estimated population size declined from 101 (71, 131) in 1988 to 40 (28, 52) in Fall 2008. During this period male-to-female gender ratio changed from 1.2:1 to 2:1.

Table 12. Estimates of population size ($\hat{N}$) at each sampling occasion for males and females, derived from the model estimates of the top-ranked model for the subset of data containing sex and weight covariates.

Occ	$\hat{N}$	SE($\hat{N}$)	95% LCL	95% UCL	$\hat{N}$	SE($\hat{N}$)	95% LCL	95% UCL
	Males				Females			
1	123	12	100	147	101	15	71	131
2	121	11	100	142	96	13	70	123
3	118	10	99	137	92	12	69	115
4	115	9	98	133	88	10	67	108
5	113	8	98	128	84	9	66	102
6	112	7	97	126	81	9	65	98
7	111	7	97	125	80	8	64	96
8	109	7	96	122	78	8	63	93
9	108	6	96	121	76	7	62	90
10	107	6	95	119	74	7	61	87
11	106	6	95	118	73	7	60	86
12	105	6	94	116	71	6	59	83
13	104	5	94	115	70	6	58	81
14	103	5	93	113	68	6	57	79
15	102	5	92	112	66	5	56	77
16	101	5	91	110	65	5	54	75
17	100	5	90	109	64	5	53	74
18	99	5	89	108	62	5	52	72
19	98	5	89	107	61	5	51	71
20	97	5	87	106	59	5	49	69
21	94	5	85	104	56	5	46	66
22	94	5	84	104	55	5	46	65
23	92	5	82	103	54	5	44	64
24	92	5	81	102	53	5	43	63
25	91	6	80	101	51	5	41	61
26	90	6	79	101	50	5	40	61
27	89	6	77	100	49	5	38	59
28	88	6	76	100	48	5	37	59
29	87	6	75	99	47	6	36	58
30	86	6	74	99	46	6	35	57
31	85	7	72	98	45	6	33	56
32	84	7	71	98	44	6	32	55
33	83	7	70	97	43	6	31	54
34	83	7	69	97	42	6	30	53
35	82	7	67	96	41	6	29	52
36	81	8	66	96	40	6	28	52

DISCUSSION

Pallid sturgeon moved up and downstream to use a variety of habitats throughout the year. Movement rates were generally correlated with the relative discharge regime within each season; fish moved more during seasons with higher discharges. Adult pallid sturgeon in the Yellowstone River also moved more during seasons with higher discharges, although adult movement rates were higher than those of juveniles during these periods (Bramblett and White 2001). In areas where natural seasonal pattern of discharge were eliminated by impoundments no seasonal differences in movement rates of juvenile pallid sturgeon were observed (Jordan et al. 2006). Pallid sturgeon moved among and used most habitats in proportion to availability (i.e., random habitat use) during seasons when movement rates were highest (spring, runoff). Random habitat use during these periods was likely dictated by movement pattern; transient fish passed through most habitats as they moved up and downstream rather spending appreciable time in any one habitat. During summer and winter, when movement rates declined, selection of bluff pool habitats occurred. Bluff pools were generally longer and had lower average and bottom velocities than pools in the valley bottom. Although pallid sturgeon selected habitats with relatively low average velocities, the resolution of our data do not allow us to determine what velocities fish were using within bluff pools; fish may have used low, high, or average velocities. Juvenile pallid sturgeon were reported to use relatively swift velocities (Jordan et al. 2006; Gerrity et al. 2008), although limited quantification of large-scale availability of velocities and possible interaction with selection of the deepest part of the channel (i.e., the thalweg), which is also the swiftest, make findings related to velocity somewhat ambiguous. Adult (Bramblett and White 2001) and juvenile (Jordan et al. 2006; Gerrity et al. 2008) pallid sturgeon occupied relatively deep parts of the channel in previous studies. The bluff pool habitats that pallid sturgeon selected for most of the year (summer, winter) had significantly ($P < 0.05$) shallower average and maximum depths than armored valley margin and bottom rip-rap pools, which were avoided or used in proportion to availability during these periods. However, depths in bluff pools were more uniform than in armored habitats. Although the specific factors driving positive selection of unarmored pools at the valley margin (bluff pools) and avoidance of these habitats once they are armored (valley margin rip-rap pools) remains unclear, these findings suggest that stream bank armoring may degrade the quality of habitat for pallid sturgeon. Depth may dictate use of side channel habitats. Side channels were avoided during seasons with low discharges but used in proportion to availability during high discharges when they are likely deeper. Juvenile pallid sturgeon avoided side channel habitats in areas with proportionally lower or no seasonal increases in discharge (Jordan et al. 2006; Gerrity et al. 2008).

Telemetry is a useful to tool to aid in the development of monitoring programs designed to estimate survival. Monitoring programs designed to address crucial information needs (e.g., survival estimation) in a targeted approach rather than focusing on broad omnibus surveillance monitoring will result in more effective and informed decision-making and conservation of rare species (Nichols and Williams 2006). When estimating survival using mark recapture analysis it is recommended that sampling intervals are kept as short as possible and that capture probabilities are as high as possible (Hadley and Rotella

2009). Accordingly, development of a sampling approach that results in capture of many individuals in a relatively short period is necessary. Investigation of seasonal habitat selection resulted in identification of sampling locations (bluff pools) and time periods (summer) to best meet this requirement. Targeted sampling of seasonally selected bluff pool habitats resulted in catch rates 20 to 90 times higher than those of previous randomized sampling designs on the Yellowstone River and capture probabilities of fish stocked in the Yellowstone River were higher than in other parts of the Upper Basin where randomized sampling designs are employed (Hadley and Rotella 2009). Using this targeted approach, sampling was reduced to intervals ranging from 17 to 56 days each year compared to continuous sampling over a six-month period by previous randomized designs. Integration of telemetry also allows inferences to be made about survival model assumptions. For example, seasonal patterns of movement and habitat selection indicate that fish likely move randomly among sampled and unsampled habitats over the course of the year thereby reducing possible bias resulting from stratified or targeted sampling. This study and previous work (Jaeger et al. 2008) suggest that emigration of stocked fish from the Yellowstone River is likely low and that most fish that emigrate return during the time when sampling occurs (summer), which reduces another possible source of bias. Encounter histories from telemetered fish can also be directly integrated into survival estimation, which increases capture probabilities and improves precision of survival estimates.

There are notable differences in estimated survival probabilities between this analysis and a prior comprehensive analysis for RPMA 2, which did not include data from telemetered fish (Hadley and Rotella 2009). Standard errors for monthly survival estimates in the Yellowstone-only analysis using telemetry captures are roughly 1/2 to 1/3 the magnitude of standard errors for Yellowstone-stocked fish in the previous analysis. The inclusion of telemetry data appears to improve the precision of survival rate estimates, which was recommended of pallid sturgeon monitoring programs (Hadley and Rotella 2009). Monthly survival rates for fish released as fingerlings were similar between the two analyses, but rates for fish released as spring and summer yearlings were estimated to be higher in the Yellowstone-only analysis. Reasons for this disparity between analyses are not clear. One possibility is that the inclusion of telemetered fish may result in bias relative to the recapture-only dataset used in the main RPMA 2 analysis. Fish that are selected for telemetry implants 1) have already survived to an age at which implants may be used, and 2) may be selected due to their larger-than-average size and ability to carry a radio of a certain weight. Survival probabilities for other sturgeon species are positively correlated with size at release (Justice et al. 2009). Thus, telemetry fish may be higher-quality individuals with an inherently better chance of survival. This would explain why differences between the analyses were negligible for fingerlings (no telemetry data included), but larger for yearlings (more telemetered individuals contributing to survival estimates). Conversely, survival for spring and summer yearlings may have been underestimated by the previous analysis.

Future work should examine results from separate analyses of mark-recapture and telemetry data to determine the level of confidence in results from the combined analysis. In general, there are numerous advantages to using data from multiple sources in a joint analysis. Precision of survival estimates should increase (Burnham 1993), potential radio effects may be evaluated to determine whether telemetry-based survival estimates could be biased, and true rather than apparent survival rates may be estimated by using telemetry to evaluate emigration rates. Model structures have been developed for estimating survival and recapture rates using mark-recapture and telemetry data (Powell et al. 2000). However, these methods require reasonable probabilities of capture (i.e., $p > 0.20$) of individuals in the study area and perfect or near-perfect detection of telemetered individuals. Neither of these conditions is met for pallid sturgeon in the Lower Yellowstone River. Unless these conditions are met with future datasets, it is recommended that future analyses use the Cormack-Jolly-Seber model type with separately estimated detection probabilities for telemetered versus PIT-tagged-only individuals, as was done in this analysis.

Survival estimates and population sizes for adult pallid sturgeon in RPMA 2 were similar to but slightly different than those reported elsewhere. Survival probabilities and population sizes may have been underestimated by theoretically derived or extrapolated survival estimates. Current planning documents use a theoretically determined annual adult survival probability of 0.9 (U.S. Fish and Wildlife Service 2008). Bratten et al. (2009) estimated annual survival probability to be 0.95 using catch curve methods based on three previous population estimates. Our more data intensive approach resulted in an estimated annual survival probability (0.986) that had confidence intervals overlapping those of Bratten et al. (2009) but that did not include the estimate currently being used in planning documents (U.S. Fish and Wildlife Service 2008). Previous abundance estimates reported year-specific population sizes about 14% smaller than those reported here. Abundance was estimated as 250 in 1995 (Krentz 1996), 178 in 2001 (Kapuscinski 2002), and 158 in 2004 (Klugle and Baxter 2005) whereas our analysis estimated abundances of 320 in 1995, 210 in 2001, and 170 in 2004. Bratten et al. (2009) projected that there were 344 (281, 420) adult pallid sturgeon present in 1989 whereas we estimate a population size of 502 (408, 595) when using all available data. Although differences among point estimates of survival and abundance were relatively small they could result in development of disparate management goals when extrapolated over time periods consistent with pallid sturgeon life spans (30 to >50 years) to predict previous population sizes or project future stocking rates in planning documents. Population trajectories of other sturgeon species are highly sensitive to relatively small changes in adult survival rates (Quist 2001; Bruch 2008). Because of the variety in approach and statistical rigor of the methodologies used to obtain abundance and survival estimates for adult pallid sturgeon we recommend that managers carefully compare and evaluate all available parameter estimates before integrating them into planning documents or management decisions.

Changes in the extant population of pallid sturgeon in RPMA 2 over the past 20 years suggest that limited time remains to implement habitat restoration projects intended to benefit wild fish. The abundance of wild adult pallid sturgeon in RPMA 2 has declined by about 76% since 1988 and the population appears to be skewing towards male fish. It was estimated that the extant population is currently comprised of two males for every female pallid sturgeon. Thus, as populations continue to decline it is expected that female pallid sturgeon will likely be extirpated long before the last male. Even before eventual extirpation the effect of proportionally fewer female fish may be significant because most female pallid sturgeon only spawn every two to three years (Fuller et al. 2008). The presence of few reproductive individuals can eventually reduce the likelihood of natural recruitment through depensatory mechanisms irrespective of habitat conditions (Ricker 1975). Although survival rates of hatchery-reared pallid sturgeon suggest that stocked fish will survive to adulthood (Hadley and Rotella 2009) and realize benefits from habitat conservation and restoration actions, projects intended to benefit extant wild pallid sturgeon should be implemented immediately given the observed gender ratios and declines in abundance. Accordingly, we feel that implementation of the planned fish passage and entrainment protection project at Intake Diversion on the Yellowstone River must occur relatively soon to maximize the benefits of this action.

ACKNOWLEDGEMENTS

The Western Area Power Administration, U.S. Bureau of Reclamation, U.S. Army Corps of Engineers, U.S. Fish and Wildlife Service, and Montana Fish, Wildlife & Parks provided funding and resources necessary to complete this work. This work was made possible by the efforts and assistance of Michael Burt, George Landon, Rosa McIver, and Shane Schulze with sampling, telemetering, and relocating fish in the field and surveying habitats. Recapture data for extant adult pallid sturgeon in RPMA 2 and for juvenile hatchery-reared pallid sturgeon recaptured in the Yellowstone River was collected by various state, federal, and university biologists and researchers over the past 20 years and is housed in a database maintained by the U.S. Fish and Wildlife Service.

LITERATURE CITED

Bramblett, R. G. 1996. Habitats and movements of pallid and shovelnose sturgeon in the Yellowstone and Missouri Rivers, Montana and North Dakota. Ph.D. dissertation, Montana State University, Bozeman.

Bramblett, R. G., and R. G. White. 2001. Habitat use and movements of pallid and shovelnose sturgeon in the Yellowstone and Missouri Rivers in Montana and North Dakota. Transactions of the American Fisheries Society 130:1006-1025.

Braaten, P. J. and D. B. Fuller, L. D. Holte, R. D. Lott, W. Viste, T. F. Brandt, R. G. Legare. 2008. Drift dynamics of larval pallid Sturgeon in a natural side channel of the upper Missouri River, Montana. North American Journal of Fisheries Management 28:808-826.

Braaten, P. J. and D. B. Fuller, R. D. Lott, and G. R. Jordan. 2009. An estimate of the historic population size of adult pallid sturgeon in the upper Missouri River Basin, Montana and North Dakota. Journal of Applied Ichthyology 25:2-7.

Bruch, R. M. 2008. Modeling the population dynamics and sustainability of lake sturgeon in the Winnebago system, Wisconsin. Doctoral dissertation. University of Wisconsin-Milwaukee.

Carlson, J. 2003. Montana animal species of special concern. Montana Natural Heritage Program and Montana Fish, Wildlife and Parks Report, Helena.

DTM and AGI. 2008a. Yellowstone River Human Impacts Timeline: Report prepared for Custer County Conservation District and Yellowstone River Conservation District Council, 52p.

DTM and AGI. 2008b. Yellowstone River Channel Migration Zone Mapping: Report prepared for Custer County Conservation District and Yellowstone River Conservation District Council, 34p.

Fuller, D. B., M. E. Jaeger, and M. Webb. 2008. Spawning and Associated Movement Patterns of Pallid Sturgeon in the Lower Yellowstone River. Upper Basin Pallid Sturgeon Recovery Workgroup 2007 Annual Report. Upper Basin Workgroup, U.S. Fish and Wildlife Service, Bozeman, Montana.

Gerrity, P. C., C. S. Guy, and W. M. Gardner. 2008. Habitat use of juvenile pallid sturgeon and shovelnose sturgeon with implications for water-level management in a downstream reservoir. North American Journal of Fisheries Management 28:832-843.

Hadley, G. L. and J. J Rotella. 2009. Upper Basin pallid sturgeon survival estimation project. Upper Basin Pallid Sturgeon Recovery Workgroup 2008 Annual Report. Upper Basin Workgroup, Montana Fish, Wildlife & Parks, Helena.

Hart, L. G., and R. C. Summerfelt. 1975. Surgical procedures for implanting ultrasonic transmitters into flathead catfish (*Pylodictis olivaris*). Transactions of the American Fisheries Society 1:56-59.

Hiebert, S. D., R. Wydoski, and T. J. Parks. 2000. Fish entrainment at the lower Yellowstone diversion dam, Intake Canal, Montana, 1996-1998. USDI Bureau of Reclamation Report, Denver, Colorado.

Jaeger, M. E., A. V. Zale, T. E. McMahon, B. J. Schmitz. 2005. Seasonal movements, habitat use, aggregation, exploitation, and entrainment of saugers in the lower Yellowstone River: an empirical assessment of factors affecting population recovery. North American Journal of Fisheries Management 25:1550-1568.

Jaeger, M. E., and T. Watson, A. Ankrum, M. A. Nelson, J. Rotella, G. Jordan, S. Camp. 2008. Assessment of pallid sturgeon restoration efforts in the Yellowstone River annual report for 2007. Upper Basin Pallid Sturgeon Recovery Workgroup 2007

Annual Report. Upper Basin Workgroup, U.S. Fish and Wildlife Service, Bozeman, Montana.

Jordan, G. R., R. A. Klumb, G. A. Wanner, and W. J. Stancill. 2006. Poststocking Movements and Habitat Use of Hatchery-Reared Juvenile Pallid Sturgeon in the Missouri River below Fort Randall Dam, South Dakota and Nebraska. Transactions of the American Fisheries Society 135:1449-1511.

Justice, C., B. J. Pyper, R. C. P. Beamesderfer, V. L. Parmagamian, P. J. Rust, M. D. Neufeld, S. C. Ireland. 2009. Evidence of density- and size-dependent mortality in hatchery-reared juvenile white sturgeon (*Acipenser transmontanus*) in the Kootnai River. Canadian Journal of Fisheries and Aquatic Sciences 66:802-815.

Kallemeyn, L. W. 1983. Status of the pallid sturgeon, *Scaphirhynchus albus*. Fisheries 8:3-9.

Kapuscinski, K. L. 2002. Population abundance estimation of wild pallid sturgeon in recovery-priority management area #2 of the Missouri and Yellowstone river from 1991-2001. Montana Fish, Wildlife & Parks, Helena.

Klungle, M. M. and M. W. Baxter. 2005. Lower Missouri and Yellowstone rivers pallid sturgeon study 2004 report. Pages 69-91 *in* Upper Basin Pallid Sturgeon Recovery Workgroup 2004 Annual Report. Upper Basin Workgroup, Montana Department of Fish, Wildlife and Parks, Helena.

Krentz, S. 1996. 1995 Summary report, Missouri-Yellowstone rivers pallid sturgeon status. U. S. Fish and Wildlife Service, Missouri River Fish and Wildlife Management Assistance Office, Bismarck, North Dakota.

Kynard, B., E. Parker, D. Pugh, and T. Parker. 2007. Use of laboratory studies to develop a dispersal model for Missouri River pallid sturgeon early life intervals. Journal of Applied Ichthyology 23:365-374.

Lebreton, J-D., K. P. Burnham, J. Clobert, and D. R. Anderson. 1992. Modeling survival and testing biological hypotheses using marked animals: a unified approach with case studies. Ecological Monographs 62:67-118.

Manly, B. F. J., L. L. McDonald, D. L. Thomas, T. L. McDonald, and W. P. Erickson. 2002. Resource selection by animals. Kluwer Academic Press, Dordrecht.

Montana Bureau of Mines and Geology. 1979. Geologic map of the Hysham 30' x 60' quadrangle, Montana. 1:100,000. Open-file report no. 371. Butte.

Montana Bureau of Mines and Geology. 1980. Geologic map of the Terry 30' x 60' quadrangle, Montana. 1:100,000. Open-file report no. 284. Butte.

Montana Bureau of Mines and Geology. 1981a. Geologic map of the Sidney 30’ x 60’ quadrangle, Montana. 1:100,000. Open-file report no. 277. Butte.

Montana Bureau of Mines and Geology. 1981b. Geologic map of the Wibaux 30’ x 60’ quadrangle, Montana. 1:100,000. Open-file report no. 282. Butte.

Montana Bureau of Mines and Geology. 1998. Geologic map of the Glendive 30’ x 60’ quadrangle, Montana. 1:100,000. Open-file report no. 371. Butte.

Montana Bureau of Mines and Geology. 2000. Geologic map of the Billings 30’ x 60’ quadrangle, Montana. 1:100,000. Geologic map series no. 59. Butte.

Montana Bureau of Mines and Geology. 2001a. Geologic map of the Miles City 30’ x 60’ quadrangle, Montana. 1:100,000. Open-file report no. 426. Butte.

Montana Bureau of Mines and Geology. 2001b. Geologic map of the Forsyth 30’ x 60’ quadrangle, Montana. 1:100,000. Open-file report no. 425. Butte.

Natural Resources and Conservation Service. 2002. Lower Yellowstone River Corridor Color Infrared Digital Orthophotography, Montana. Natural Resources and Conservation Service, Bozeman. Available: www.nris.state.mt.us/yellowstone/LowerYel/LowerYelPhotos.html (April 2004).

Nichols, J. D. and B. K. Williams. 2006. Monitoring for conservation. Trends in ecology and evolution 21: 668-673.

Pollock, K. H., J. D. Nichols, C. Brownie, and J. E. Hines. 1990. Statistical inference for capture-recapture experiments. Wildlife Monographs 107.

Powell, L. A., M. J. Conroy, J. E. Hines, J. D. Nichols, and D. G. Krementz. 2000. Simultaneous use of mark-recapture and radiotelemetry to estimate survival, movement, and capture rates. Journal of Wildlife Management 64:302-313.

Quist, M. C., Guy, C. S., Pegg, M. A., Braaten, P. J., Pierce, C. L., and Travnichek, V. H. 2002. Potential influence of harvest on shovelnose sturgeon populations in the Missouri River system. North American Journal of Fisheries Management 22:537-549.

R Development Core Team 2008. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, URL: http://www.R-project.org.

Ricker, W. E. 1975. Computation and interpretation of biological statistics of fish populations. Bulletin of the Fisheries Research Board of Canada 191.

Ross, M. J., and C. F. Kleiner. 1982. Shielded-needle technique for surgically implanting radio-frequency transmitters in fish. Progressive Fish-Culturist 44:41-43.

Silverman, A. J., and W. D. Tomlinsen. 1984. Biohydrology of mountain fluvial systems: the Yellowstone (part I). U. S. Geologic Survey, Completion Report G-853-02, Reston.

U.S. Fish and Wildlife Service. 2008. Pallid Sturgeon (*Scaphirhynchus albus*) range-wide stocking and augmentation plan. Billings, Montana.

White, R. G., and R. G. Bramblett. 1993. The Yellowstone River: its fish and fisheries. Pages 396-414 *in* L. W. Hesse, C. B. Stalnaker, N. G. Benson, J. R. Zuboy, editors. Restoration planning for the rivers of the Mississippi River ecosystem. Biological Report 19, National Biological Survey, Washington, D.C.

White, G. C., and K. P. Burnham. 1999. Program MARK: survival estimation from populations of marked animals. Bird Study 46 (suppl.): S120-139.

White, G. C., and R. A. Garrot. 1990. Analysis of wildlife radio-tracking data. Academic Press, San Diego.

Williams, B. K., J. D. Nichols, & M. J. Conroy. 2002. Analysis and management of animal populations: modeling, estimation, and decision making. Academic Press, New York.

Zar, J. H. 1999. Biostatistical Analysis. Prentice Hall, Upper Saddle River.