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ARTICLE

Population Size of Hatchery-Reared and Wild Pallid Sturgeon in the Lower Missouri River

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Abstract

The population size of pallid sturgeon *Scaphirhynchus albus* is currently unknown throughout much of the Missouri River. Listed as federally endangered in 1990, the pallid sturgeon remains one of the rarest fishes in the Missouri and Mississippi River basins, and little to no natural recruitment occurs. Artificial population supplementation via a hatchery propagation program was initiated, necessitating the collection of sexually mature pallid sturgeon. Therefore, the Nebraska Game and Parks Commission maintained an intensive broodstock collection and mark-recapture effort from 2008 to 2010 to capture reproductively ready adults for the propagation program. Coordinated crews fished baited trotlines from the confluence of the Platte and Missouri rivers at river kilometer (rkm) 957.6 to a point about 80.5 rkm downstream. A total of 438 pallid sturgeon were captured, which amounts to a 7.8% recapture rate. The objectives of the study were to (1) use these data to estimate the annual population sizes of wild-origin and hatchery-reared pallid sturgeon within the above-mentioned 80.5-rkm reach of the lower Missouri River and (2) compare current population levels with the Pallid Sturgeon Recovery Team's population objective. We used the mark-recapture data in a robust-design analysis to derive population estimates and annual survival, capture, and temporary emigration rates. The annual population estimate for wild pallid sturgeon varied from 5.4 to 8.9 fish/rkm, whereas the estimate for known hatchery-reared fish varied from 28.6 to 32.3 fish/rkm. The robust-design approach to our analysis resulted in useful estimates of population size and other variables important to quantifying species recovery and management targets; the approach may be suitable for other fisheries management data sets.

The population size of the pallid sturgeon *Scaphirhynchus albus* in the Missouri River is unknown throughout much of the river, including the lower portion from Gavins Point Dam at river kilometer (rkm) 1,305.2 downstream to the confluence of the Missouri and Mississippi rivers (rkm 0.0). Listed as federally endangered (USFWS 1990) on September 6, 1990, pallid sturgeon remain one of the rarest fish species in the Missouri and Mississippi river basins (Dryer and Sandvol 1993). Initial declines appear to be correlated with commercial harvest but subsequent recruitment failures are probably related to extensive modification of river corridors by dam construction, reservoir development, and river channelization (Webb et al. 2004). These

modifications have blocked fish movement, destroyed or altered spawning areas, reduced food sources, altered water temperatures, reduced turbidity, and changed the hydrograph (Keenlyne and Evenson 1989; Dryer and Sandvol 1993; Pegg et al. 2003). Today, natural recruitment is minimal to nonexistent across the lower reach of the Missouri River (Snyder 2000; Hrabik et al. 2007; USFWS 2007).

Modifications and fragmentation of pallid sturgeon habitat along the entire Missouri River have been extensive. Approximately 28% of the river has been impounded by main-stem hydropower dams; 51% has been channelized for barge traffic; and an accumulated 22% of the river is downstream of dams

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where temperature, flows, and turbidity have been greatly altered (Keenlyne 1989). The lower Missouri River encompasses a 1,305.2-rkm reach of largely altered riverine habitat from Gavins Point Dam to the confluence of the Missouri and Mississippi rivers. Only 93 rkm remains as an unchannelized meandering river, but water temperatures and turbidity in this reach are influenced by Gavins Point Dam. The remaining 1,211 rkm have been channelized for navigation, which has reduced slow, shallow-water habitats believed important to pallid sturgeon life cycles (USFWS 2000, 2003).

Preventing pallid sturgeon extirpation may depend largely on the success of the artificial propagation program (USFWS 2008b). The Pallid Sturgeon Recovery Plan (Dryer and Sandvol 1993) outlined several criteria for effective propagation and stocking programs. The first recommendation was the development of a propagation plan that used existing endangered and rare species propagation knowledge to maintain a genetically sound program (Bollig 1993; UBPSWPC 2005). The propagation plan also detailed the number of broodstock needed, including brood source capture locations and protocols for spawning and rearing. The second recommendation called for development of a stocking plan to guide production-goal targets and recommendations on release size and timing that increases likelihood of survival (USFWS 2008b). The third recommendation called for development of a tagging plan that mandated that tagging of all hatchery-reared pallid sturgeon released (Steffensen et al. 2008). Plan implementation called for following the guidelines for artificially supplementing an imperiled wild population (Miller and Kapuscinski 2003) and experiences from other sturgeon research (Conte et al. 1988; Collins et al. 1994; Rein et al. 1994; UBPSWTC 2007).

The artificial propagation program was initiated at the Missouri Department of Conservation's Blind Pony State Fish Hatchery in 1992 (USFWS 2008b) and has since continued at multiple hatcheries (Smith and Brannen 2010). Approximately 100,000 hatchery-reared pallid sturgeon have been stocked into the lower Missouri River at a variety of sizes, year-classes, and locations since 1994 (Smith and Brannen 2010; Steffensen et al. 2010). The population objective for the lower Missouri River set by the Pallid Sturgeon Recovery Team (Recovery Team) is 23 sexually mature fish per river kilometer (USFWS 2008b). Stocking rates to meet this objective are based on survival of stocked hatchery-reared pallid sturgeon. To date, however, it is unknown if current stocking rates are adequate to achieve this population level. Therefore, the objectives of the study were to (1) produce population and survival estimates for naturally reproduced (wild origin) and hatchery-reared pallid sturgeon, and (2) compare the current population level to the population objective (i.e., number of pallid sturgeon/rkm) identified by the Recovery Team for the entire lower Missouri River.

METHODS

The study area included an 80.5-rkm reach of the lower Missouri River from the confluence of the Platte and Missouri rivers

at rkm 957.6 downstream to rkm 877.1 (Figure 1). This upper channelized reach is characterized by uniform channel morphology. All outside bends are revetted by large limestone rock, and the inside bends have a series of dike structures.

Data used for this analysis were acquired from the Nebraska Game and Parks Commission (NGPC) intensive broodstock collection effort. This effort was directed at collecting sexually mature pallid sturgeon and started in 2008 with yearly efforts continuing through 2010. Annual sampling was conducted during early April when each crew randomly deployed 8–10 trot lines daily. Throughout the annual effort, crews covered the 80.5-rkm reach of the lower Missouri River. Depths and velocities on outside bends prevented sampling in those habitats, so our sampling occurred on the inside bend between and around the wing dikes structures (Figure 1). Sampling occurred in the same areas across all years.

Baited trot lines were used following protocols outlined for the Pallid Sturgeon Population Assessment Program (Welker and Drobish 2010). Trot lines were 61 m long with 40 3/0-circle hooks per line and baited with earthworms *Lumbricus terrestris*. Hooks were tied to a 38-cm leader and fastened to the main line using trot line snaps. Hooks were spaced every 1.5 m to avoid hook and fish entanglement. All trot lines were deployed parallel to the rivers current on the inside bend in channel-border habitats. Channel-border habitat is defined as the area downstream of the pool habitat formed by wing dikes and is between the bank and thalweg where depths are greater than 1.2 m. Trot lines were deployed in the early afternoon and pulled the following morning; they were fished overnight for a maximum of 24 h (USFWS 2008a). Our effective sampling area was ambiguous, but we hypothesized that a trot line placed between wing dikes would create the effective sampling area shown in Figure 1. Our subsequent approach to analyses of our data were based on our concern that the outer portion of the river bend was outside our effective sampling area.

To determine their origin (i.e., naturally produced or hatchery-reared), all pallid sturgeon collected were examined for passive integrated transponder (PIT) tags, coded wire tags (CWTs), elastomer marks, and scute removal (USFWS 2008a). If no tags were present, a genetic sample was collected from the caudal fin and analyzed to confirm origin (DeHaan et al. 2008). All fish collected were then categorized by origin for this analysis.

The mark-recapture data collected during this 3-year sampling effort were suited for the robust design (Kendall et al. 1997) used to derive population estimates, survival estimates, capture rates, and emigration rates. Robust sampling designs include primary sampling periods with repeated secondary sampling periods that occur during a short time interval within the primary periods. Assumptions under the robust design are a combination of assumptions for closed-population methods, related to the secondary samples, and the Jolly-Seber methods for open populations, related to the primary periods (Kendall et al. 1997). These assumptions are (1) the population is assumed

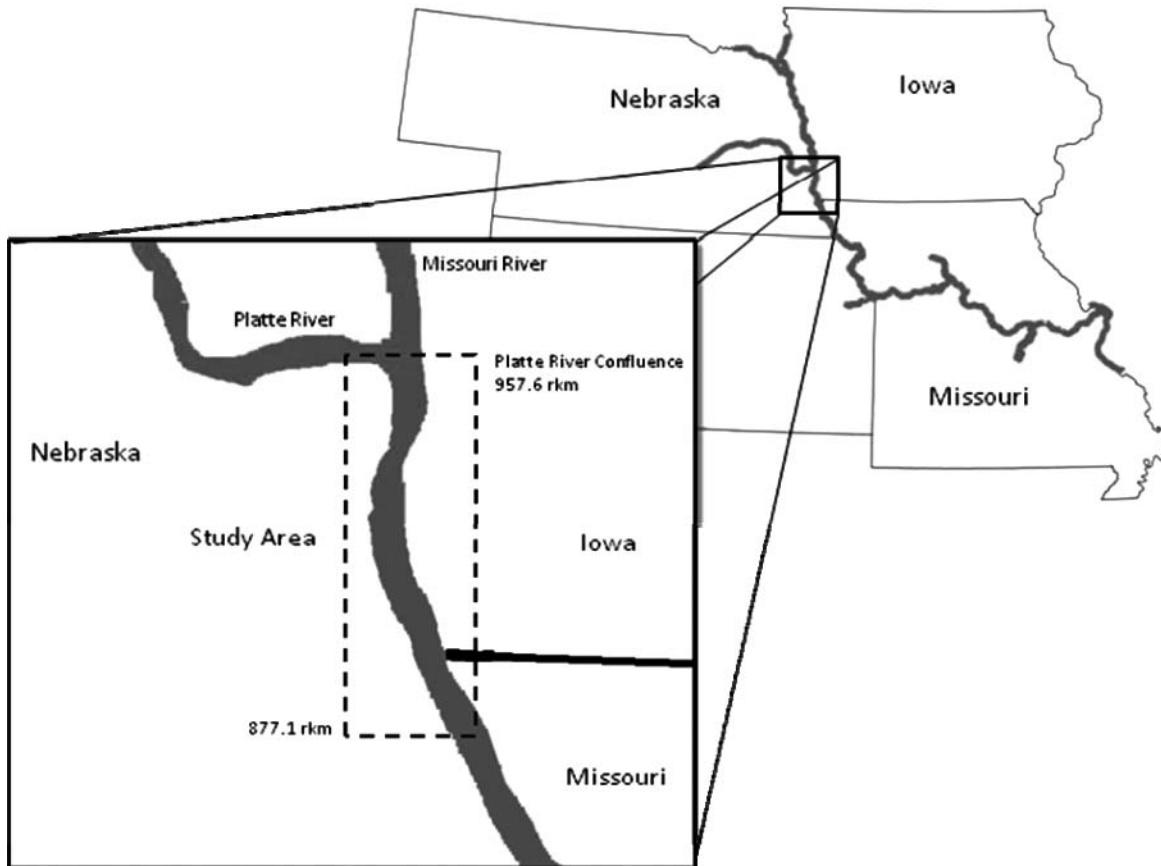


FIGURE 1. Pallid sturgeon study area encompassing the 80.5-river-kilometer (rkm) sampling reach from the confluence of the Missouri and Platte rivers at rkm 957.6 (from the mouth of the Missouri River) downstream to rkm 877.1.

closed to additions and deletions across all secondary sampling sessions within a primary period, (2) temporary emigration is assumed to be either completely random, Markovian, or based on a temporary response to first capture, and (3) survival rates are assumed to be the same for all animals in the population, regardless of availability for capture (Cooch and White 2010). Three primary periods (2008, 2009, and 2010) occurred, and the number of secondary sampling occasions varied by year. The number of days fished varied by weather and crew availability, but the daily effort was relatively constant.

The robust design was uniquely suited to our sampling scheme because it allowed us to explore the possibility that fish were temporarily (between years) not available for capture. And, it allowed us to use the movement information to adjust estimates of population size. The robust-design estimator uses closed periods (within years) to estimate capture rate (p), recapture rate (c), and population size (N) while using open periods (between years) to estimate annual survival (ϕ), and emigration parameters (γ' and γ'').

We established four hypotheses, which represented research questions relevant to management regarding pallid sturgeon in our sample. These hypotheses were to determine whether (1)

hatchery and wild fish have different population sizes, (2) annual survival varies by origin of fish (hatchery or wild), (3) hatchery and wild fish have different emigration patterns, and (4) hatchery and wild fish have different capture probabilities.

A set of 40 biologically reasonable models were constructed to investigate annual population changes between groups; the models represented possible combinations of our four hypotheses. We used Akaike's information criterion (AIC) to compare models (Burnham and Anderson 2002), and conducted analyses with the robust-design module of program MARK (White and Burnham 1999). We report AIC scores with a second-order correction for small sample sizes (AIC_c), which asymptotically become equal to AIC scores as sample size increases; AIC_c is the default value provided by program MARK. We used 95% confidence intervals (CIs) to assess differences between demographic rate estimates for groups.

We knew a priori that our annual sample sizes could limit the number of parameters estimable, and the robust design is parameter-rich (Kendall et al. 1997). Thus, we decided to keep p , c , ϕ , and the emigration parameters (γ' and γ'') time-constant in our models; population size (N) was allowed to vary by year. In the set of 40 models, we included scenarios in which p and

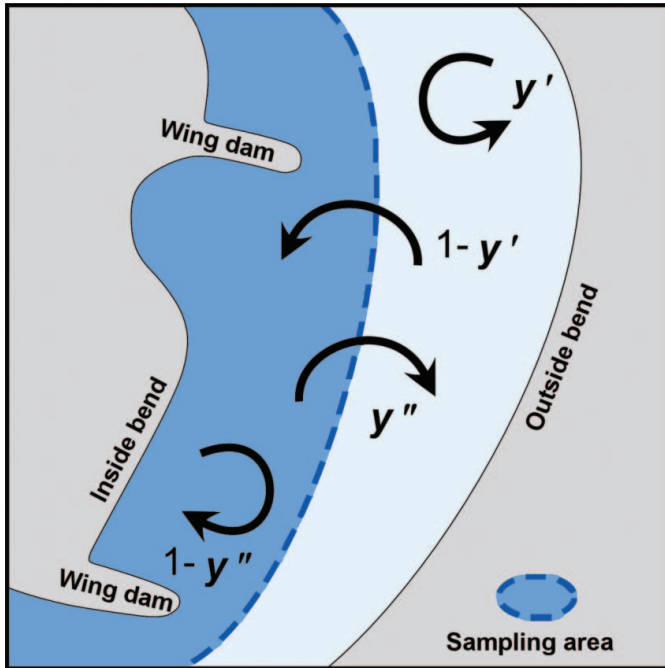


FIGURE 2. Depiction of the robust design's application to pallid sturgeon sampling at a typical wing dike field within the channelized Missouri River. The darken region of the river shows the areas where trot lines were deployed; the arrows show the temporary emigration patterns (γ' and γ'' ; see Table 1) between primary sampling periods (years, in our study). Accounting for temporary emigration of pallid sturgeon enables post hoc population estimates for the entire river reach (super population). [Figure available in color online.]

c probabilities were equal or unequal to allow for effects of capture on recapture probability to be assessed.

The robust design provides estimates of temporary emigration via two parameters: γ' and γ'' (Figure 2). The first, γ' , represents the probability that an individual that is away from the study area remains away from the study area in the next period, given that the individual survives to the next period (essentially, the probability that an animal that has emigrated away from the study area does not immigrate back such that immigration rate could be defined as $1 - \gamma'$). The second, γ'' , represents the probability that an animal within the study area emigrates from the study area in the next period, given that it survives (essentially, the emigration rate). We allowed temporary emigration (γ' and γ'') to vary in three specific ways (Kendall et al. 1997) within the set of 40 models. First, we constructed models in which $\gamma' = \gamma'' = 0$; this represented a no-emigration scenario in which emigration and immigration did not occur and marked animals remained in the study area during the entire study. Second, we allowed γ' and γ'' to vary independently, which represented a scenario in which the probability of an individual being away from the study area could depend on its position (at the study area or away) in the previous period (referred to as “Markovian emigration” by Kendall et al. 1997). Last, we included a group of models in which $\gamma' = \gamma''$; this represented a scenario

in which the probability that an individual was away from the study area was the same, regardless of its position in the previous period (referred to as “random emigration” by Kendall et al. 1997). After reviewing our model results, we removed models that lacked signs of converging on suitable parameter values or produced erroneous confidence intervals around the estimate. We were prepared to use model averaging to estimate our parameters of interest (\emptyset and N) if a single model did not carry a majority of the AIC_c weight; standard errors reported with model-averaged parameters are unconditional standard errors (Burnham and Anderson 2002).

The robust design provides estimates of population size (N) for the study area (defined as the area from which fish could have been captured by our methods). However, an estimate of the population size of the super population (N_{sp} ; the area in which fish could immigrate to and emigrate from—in our case the 80.5-rkm stretch of the Missouri River) is obtainable because the estimates of temporary emigration (γ' and γ'') allow a larger inference. Our best model indicated that $\gamma' = \gamma''$; thus we were able to estimate N_{sp} as

$$\hat{N}_{sp} = \frac{\hat{N}}{1 - \hat{\gamma}}.$$

We approximated the variance of N_{sp} using the delta method (after methods of Powell 2007) because the variance of N_{sp} is a function of the variance of our estimate of N and the variance of our estimate of γ (ignoring covariance), that is,

$$\hat{\text{var}}(\hat{N}_{sp}) = \hat{\text{var}}(\hat{N}) \left(-\frac{1}{\hat{\gamma} - 1} \right)^2 + \hat{\text{var}}(\hat{\gamma}) \left(\frac{\hat{N}}{\hat{\gamma}^2 - 2\hat{\gamma} + 1} \right)^2.$$

We calculated 95% CIs as $\pm 1.96 \times \text{SE}(N_{sp})$.

We report our estimates using subscripts and superscripts to qualify parameter symbols. The superscript “origin” indicates that the model provided origin-specific estimates for wild and hatchery-reared individuals, and “.” indicates an estimate that was pooled by origin. The subscript “ t ” indicates a parameter allowed to vary by time (year), and “.” indicates the parameter was constant across time (year).

RESULTS

During 2008, crews deployed 303 trotlines in 11 consecutive days and collected 165 pallid sturgeon, 42 of which were genetically determined to be wild origin and 123 were hatchery origin. In 2009, crews deployed 324 trotlines in 9 d and collected 160 pallid sturgeon, of which 23 were determined to be wild origin and 137 were hatchery stock. During the 2010, crews deployed 346 trotlines in 11 consecutive days and collected 167 pallid sturgeon, of which 28 were determined to be wild origin and the remaining 139 were hatchery origin. Overall, 492 individuals were collected during the sampling efforts, of which 38 or 7.8% were recaptures, of which 1 fish was recaptured four times, 4

TABLE 1. Comparison of competing models used to describe the variation in annual pallid sturgeon population estimates in the lower Missouri and Platte rivers from 2008 to 2010, including survival (\emptyset), temporary emigration ($\gamma'' =$ the probability of being off the study area given that the animal was on the study area during the previous primary trapping session; $\gamma' =$ the probability of being off the study area given that the animal was not on the study area during the previous primary trapping session), capture (p) and recapture (c), and population size (N). Superscripts include “origin” (wild and hatchery-reared individuals were allowed to have distinct parameter estimates) and “.” (estimates were pooled by origin). Subscripts include “ t ” (the parameter was allowed to vary by time—i.e., year), and “.” (the parameter was constant across time). Models are ranked by corrected Akaike’s information criterion (AIC_c ; the first row shows the highest-ranking model), where k is the number of parameters, ΔAIC_c is the difference between a model’s AIC_c value and that of the highest-ranked model, and W_{AIC_c} is the Akaike weight (sum of all weights = 1.00). The AIC_c statistics are not shown for models with $W_{AIC_c} < 0.01^a$.

Model	k	AIC_c	ΔAIC_c	W_{AIC_c}
$\emptyset.(\gamma''.) = (\gamma'.)(p^{origin})(c^{origin})(N_t)$	12	-591.5	0.0	0.28
$\emptyset^{origin}(\gamma''^{origin}) = (\gamma'^{origin})(p^{origin})(c^{origin})(N_t)$	14	-590.6	0.9	0.18
$\emptyset.(\gamma''^{origin}) = (\gamma'^{origin})(p^{origin})(c^{origin})(N_t)$	13	-590.6	0.9	0.18
$\emptyset^{origin}(\gamma''.) = (\gamma'.)(p^{origin})(c^{origin})(N_t)$	13	-589.4	2.1	0.10
$\emptyset.(\gamma''.)(\gamma''.)(p.)(c.)(N_t)$	10	-587.5	4.0	0.04
$\emptyset.(\gamma''.) = (\gamma''.)(p.)(c.)(N_t)$	10	-587.5	4.0	0.04
$\emptyset.(\gamma''^{origin})(\gamma'^{origin})(p.)(c.)(N_t)$	12	-587.1	4.4	0.03
$\emptyset^{origin}(\gamma''.) = (\gamma'.)(p.)(c.)(N_t)$	11	-586.3	5.2	0.02
$\emptyset^{origin}(\gamma''.)(\gamma''.)(p.)(c.)(N_t)$	11	-586.3	5.2	0.02
$\emptyset^{origin}(\gamma''.) = 0\gamma''. = 0p^{origin} = (c^{origin})(N_t)$	10	-586.2	5.3	0.02
$\emptyset^{origin}(\gamma''^{origin}) = (\gamma'^{origin})(p.)(c.)(N_t)$	12	-585.5	6.0	0.01
$\emptyset.(\gamma''^{origin}) = (\gamma'^{origin})(p.)(c.)(N_t)$	11	-585.5	6.0	0.01
$\emptyset^{origin}(\gamma''.) = 0\gamma''. = (0p^{origin})(c^{origin})(N_t)$	12	-584.6	6.9	0.01
$\emptyset.(\gamma''.) = 0\gamma''. = (0p^{origin})(c^{origin})(N_t)$	11	-584.5	6.9	0.01
$\emptyset.(\gamma''.) = 0\gamma''. = 0p. = (c.)(N_t)$	8	-584.4	7.1	0.01
$\emptyset^{origin}(\gamma''.) = (\gamma'.)(p^{origin}) = (c^{origin})(N_t)$	11	-584.1	7.4	0.01
$\emptyset^{origin}(\gamma''.)(\gamma''.)(p^{origin}) = (c^{origin})(N_t)$	11	-584.1	7.4	0.01

^aAdditional models ($W_{AIC_c} < 0.01$):

1. $\emptyset. \gamma''. = 0\gamma''. = 0p^{origin} = (c^{origin})(N_t)$,
2. $\emptyset^{origin}(\gamma''.) = 0\gamma''. = 0p. = (c.)(N_t)$,
3. $\emptyset.(\gamma''.) = 0\gamma''. = (0p.)(c.)(N_t)$,
4. $\emptyset.(\gamma''.)(\gamma''.)(p.) = (c.)(N_t)$,
5. $\emptyset.(\gamma''.) = (\gamma''.)(p.) = (c.)(N_t)$,
6. $\emptyset^{origin}(\gamma''^{origin}) = (\gamma'^{origin})(p^{origin}) = (c^{origin})(N_t)$,
7. $\emptyset^{origin}(\gamma''.) = 0\gamma''. = (0p.)(c.)(N_t)$,
8. $\emptyset.(\gamma''^{origin}) = (\gamma'^{origin})(p^{origin}) = (c^{origin})(N_t)$,
9. $\emptyset.(\gamma''.) = (\gamma''.)(p^{origin}) = (c^{origin})(N_t)$,
10. $\emptyset.(\gamma''.)(\gamma''.)(p^{origin}) = (c^{origin})(N_t)$,
11. $\emptyset^{origin}(\gamma''.) = (\gamma'.)(p.) = (c.)(N_t)$,
12. $\emptyset^{origin}(\gamma''^{origin}) = (\gamma'^{origin})(p.) = (c.)(N_t)$,
13. $\emptyset^{origin}(\gamma''^{origin}) = (\gamma'^{origin})(p.) = (c.)(N_t)$,
14. $\emptyset^{origin}(\gamma''^{origin})(\gamma'^{origin})(p.) = (c.)(N_t)$

fish were recaptured three times, and 33 were recaptured only once. The fork lengths for wild pallid sturgeon varied from 589 to 1,094 mm (mean = 892.7 mm), whereas hatchery-reared pallid sturgeon fork lengths varied from 300 to 1,001 mm (mean of 556.8).

Survival estimates (\emptyset) and temporary emigration probabilities (γ'' and γ') were fixed time-constant and did not vary by origin for the highest ranking model: $\emptyset.(\gamma''.) = (\gamma'.)(p^{origin})(c^{origin})(N^{origin}_t)$, where $w_{AIC_c} = 0.28$ (Table 1). Capture probabilities (p) and recapture probabilities (c) were also fixed time-constants but the top model did vary by origin. Similarly, we only estimated population size estimates (\hat{N}) as year- and origin-specific in every model. Population estimates (\hat{N}) for the super population of wild pallid sturgeon within the 80.5-rkm study area were 715 (SE = 35) during the 2008 sam-

pling period, 437 (SE = 23) in 2009, and 459 (SE = 23) in 2010, whereas estimates of the super population of known hatchery-reared pallid sturgeon were 2,304 (SE = 43) during the 2008 sampling period, 2,305 (SE = 45) in 2009, and 2,600 (SE = 48) in 2010. Therefore, the population estimates for pallid sturgeon per river kilometer varied from 5.4 (SE = 0.45) to 8.9 (0.85) for wild origin and from 28.6 (SE = 0.89) to 32.3 (0.96) for hatchery-reared (Figure 3).

The top model estimated an overall annual survival rate for all pallid sturgeon of 0.78 (SE = 0.31). Model-averaging provided survival estimates by origin, although unconditional variances, as with most model-averaging exercises, were quite large for both the wild ($\emptyset = 0.77$, SE = 0.37) and hatchery ($\emptyset = 0.69$, SE = 0.32) fish. The temporary emigration probability ($\gamma'' = \gamma'$) was 0.84 (SE = 0.09). Wild pallid sturgeon had a higher

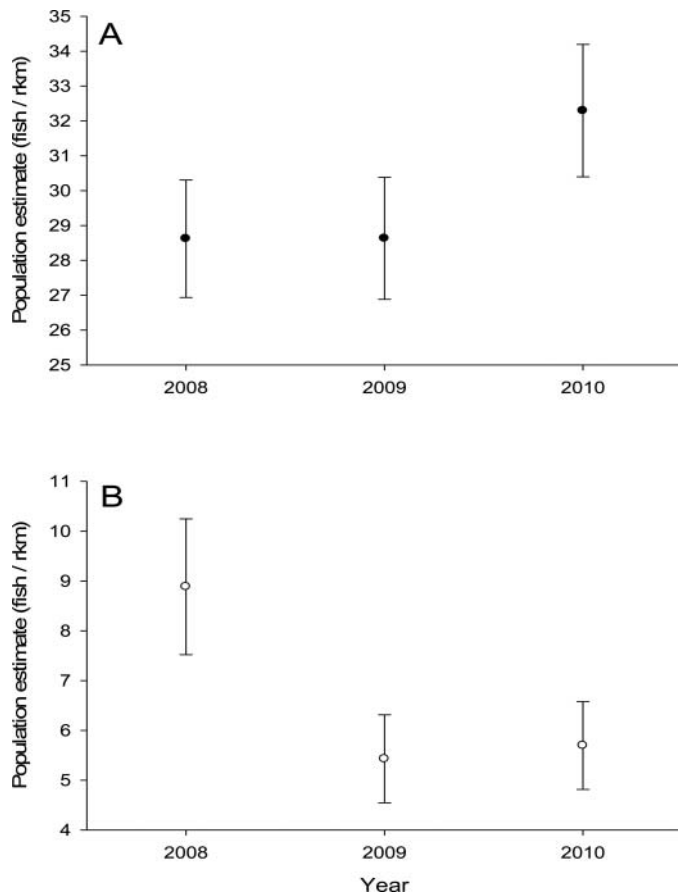


FIGURE 3. Population estimates (2008–2010) for known (A) hatchery-reared and (B) wild-origin pallid sturgeon within the 80.5-river-kilometer (rkm) study reach of the lower Missouri River. Estimates were derived from the best model (Table 1). Error bars represent 95% confidence intervals.

capture (p) probability ($p = 0.09$, $SE = 0.03$) than did hatchery fish ($p = 0.04$; $SE = 0.01$); in contrast, recapture rates (c) showed the opposite relationship, wild fish ($c = 0.002$; $SE = 0.001$) being less than hatchery fish ($c = 0.01$; $SE = 0.002$).

DISCUSSION

Our data provide the first population estimates for pallid sturgeon in this reach of the Missouri River. Population size estimates for wild pallid sturgeon were variable during the study period. Our results indicate a higher wild pallid sturgeon population estimate in 2008 than in 2009 and 2010, whereas the population estimate of hatchery-reared pallid sturgeon was consistent in 2008 and 2009 and increased in 2010. Trot lines collected a wide range of length distributions, including the largest fish known throughout this reach of the Missouri River. The smallest genetically confirmed wild pallid sturgeon collected was 589 mm. This indicates that unless natural reproduction occurs, wild pallid sturgeon are fully recruited to trot lines, and all estimates are not size biased. However, small hatchery-reared pallid sturgeon are not fully recruited to this gear. Based on

recaptures from the stocking program, it takes approximately 2 years for a pallid sturgeon to reach 300 mm (Shuman et al. 2011); therefore, parameter estimates throughout this paper do not include hatchery-reared fish less than 300 mm or less than age 1.

Our population size estimates are considerably higher than published estimates for pallid sturgeon elsewhere in the Missouri River. For example, the USFWS (2007) reported as few as 45 wild pallid sturgeon remain in the upper Missouri River from the confluence of the Marias River (3,302 rkm) to the headwaters of Fort Peck (3,004 rkm), which equates to about 0.15 fish/rkm. Klungle and Baxter (2005) estimated 158 (95% CI, 129–193) wild pallid sturgeon remain in the reach of the Missouri River between Fort Peck Dam (2,850 rkm) and the headwaters of Lake Sakakawea (2,524 rkm), including the lower 300 rkm of the Yellowstone River, upstream to the confluence with the Tongue River. Therefore, there are approximately 0.25 wild fish/rkm in this reach. No population estimates are available for reaches of the Missouri River (below Gavins Point at rkm 1,305) outside of this study area or the Mississippi River.

The robust design, because it accounts for temporary emigration, was well-suited to provide an estimate of the entire population within the 80.5-rkm reach; our emigration estimates indicated that a small proportion of the population (0.16) was in the vicinity of our capture hooks during each period. Using estimates of temporary emigration, we were able to estimate the size of the super population (Kendall et al. 1997), which was an objective of our study. Previous estimates of population size on the Missouri River had no mechanism to estimate the population beyond the subpopulation immediately in the vicinity of their sampling gear. We encourage biologists to consider the robust design as they plan sampling strategies because the methods used to capture fish during annual sampling efforts may fit nicely into this design.

The long-term objective for pallid sturgeon in the lower Missouri River is to establish a naturally reproducing and self-sustaining population with an estimated target of 30,000 adults (Dryer and Sandvol 1993; USFWS 2008b). This population objective is only one criteria, which by itself and may or may not contribute to a naturally reproducing and self-sustaining population. Assuming that population objective is nearly achieved, it should result in a density of 23 adults/rkm, which is similar to our estimate for hatchery-reared fish. The majority of fish identified in the hatchery-reared group for this analysis were immature; however, most of these fish will reach spawning age because survival for hatchery stocked pallid sturgeon in the lower Missouri River is high (0.78 for this study; 0.92 in Steffensen et al. 2010 for fish >age 1). Stocked fish from the 1992, 1997, and 2002 year-classes have been documented in reproductively ready condition (A. Delonay, U.S. Geological Survey, personal communication), indicating that the number of spawning-age pallid sturgeon could be greatly influenced by the stocking program in the near future, provided hatchery-reared individuals begin contributing to the population. Also, the

population estimate for hatchery-reared fish will probably continue to increase as recently stocked hatchery-reared progeny fully recruit to our sampling gears.

Our targeted sampling location and the historical importance of the Platte River to pallid sturgeon may disproportionately influence our population estimates compared with other reaches of the channelized Missouri River. Wild and hatchery-reared pallid sturgeon have been collected in the lower Platte River (Snook 2001; Swigle 2003; M. J. Hamel and M. A. Pegg, University of Nebraska–Lincoln, unpublished data), and the Platte River confluence may be an area of concentration for pallid sturgeon at certain times of the year, which could falsely increase population sizes in our study reach. However, similar numbers of pallid sturgeon were captured throughout our 80.5-rkm study area, suggesting there is an even distribution of pallid sturgeon throughout the study area.

An age-specific survival analysis of pallid sturgeon based on age at stocking had previously been completed using a larger, more spatially extensive data set of hatchery-reared fish (Steffensen et al. 2010). Survival estimates for hatchery-reared pallid sturgeon increased from 0.05 (SE = 0.01) for age-0 fish to 0.68 (SE = 0.11) for age-1 fish and 0.92 (SE = 0.02) for older (> age 1) fish; our pooled survival estimate for this study, combining all ages of wild and hatchery-reared fish was 0.78 (SE = 0.31). The Steffensen et al. (2010) estimates were based solely on hatchery-reared fish stocked at a variety of ages (most at ages 0 and 1). Our estimate of annual survival (0.78) from our top model (Table 1) did not provide evidence of origin-specific survival, but our confidence in the top model was not overwhelming (AIC $w = 0.28$). The second-best model ($w = 0.18$) incorporated origin-specific survival, which led us to model average annual survival across models. The model-averaged estimate for wild fish was only 8% higher than for hatchery-reared fish, and confidence intervals overlapped considerably. Thus, our analysis, at this time, does not provide evidence that survival rates vary between wild and hatchery pallid sturgeon. It is possible that evidence for such differences may be provided from continued sampling, which may improve the variances around those estimates.

Field crews do not have the ability to determine sexual maturity. Therefore, all presumed wild fish collected are transported to the hatchery to potentially be used in the propagation program. This means that some fish are unavailable for recapture within that year, and this is reflected in the low ($c = 0.002$) recapture rates for wild fish in our sample. However, fish are restocked after being assessed for maturity or used in the propagation program and can be recaptured in subsequent years. Therefore, the survival estimate derived from this analysis should not be affected by the hatchery process.

As hatchery-reared pallid sturgeon continue to survive and presumably supplement the reproductive population, stocking strategies in the lower Missouri River will need to be reviewed. Our study provides the first measurement of the parameters necessary to evaluate stocking rates and population objectives for pallid sturgeon in the lower Missouri River. Our analysis

shows that valuable information and estimates can be gathered from the current data set by using modern estimation methods. However, our estimates should be viewed as initial estimates, and managers and policy makers should evaluate the confidence intervals and other measures of precision we provide. Our targeted collection of reproductively ready pallid sturgeon will allow continued monitoring of the wild and hatchery sturgeon populations in the lower Missouri River and continue to provide insight into how well stocking rates are meeting the population objective.

Recovery plans for endangered species are often quantified as a target number of individuals within a given area (USFWS 1998, 2008b) or a minimum population size to maintain genetic diversity and be self-sustaining (USFWS 1995; NMFS 1998). However, given the rarity of an endangered species and its difficulty to be captured and recaptured, it is often difficult to assess populations using traditional approaches like relative abundance. The robust design used for this analysis provides estimable population estimates along with other parameter estimates (i.e., survival rates, immigration rates, and capture/recapture probabilities) that are beneficial and often necessary components of fisheries management and should be fully considered for recovery and management efforts.

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