Freshwater Mollusk Biology and Conservation

Modeling densities, apparent survival, and population size of Louisiana Pigtoe (Pleurobema riddellii) at three different sites in the Neches River, Texas

--Manuscript Draft--

Manuscript Number:	FMBC-D-21-00003R1
Full Title:	Modeling densities, apparent survival, and population size of Louisiana Pigtoe (Pleurobema riddellii) at three different sites in the Neches River, Texas
Short Title:	Densities, apparent survival and population size of Louisiana Pigtoe
Order of Authors:	David Ford
	Edith D. Plants-Paris
	Neil B. Ford
Corresponding Author:	David Ford Edge Engineering and Science, LLC Houston, Texas UNITED STATES
Article Type:	Regular Article
Keywords:	Recapture rates; mark-recapture; MARK; population dynamics; vital rates; long-term
Abstract:	A large majority of North American unionids are imperiled to some degree. This includes several Texas species, such as the Louisiana Pigtoe, Pleurobema riddellii, which has been proposed for federal protections and is currently under review by the U. S. Fish and Wildlife Service. Understanding this species' population dynamics, such as apparent survival, densities, and population size, is vital to their conservation; however, much of this information is currently lacking. We used previous survey efforts within Texas' Neches River Basin to identify three sites with locally high P. riddellii abundances. At each site a 25 m2 area with the greatest P. riddellii density was selected and excavated. All P. riddellii were collected and given a unique identifying PIT tag. We returned once each summer from 2014 to 2019, excluding 2018, excavated the area and collected all marked individuals. Any new P. riddellii were also given a unique tag. The computer program MARK was used to create POPAN models to compare the apparent survival, population size, and density of P. riddellii at each site by year. A total of 392 P. riddellii were collected and marked over the five-year period. The most parsimonious model found apparent survival and recapture probability to vary over time and probability of entry to vary by survey location. Apparent survival averaged 78.3 ± 3.6%, and the mean density for the sites was 1.7/m2 in 2014 and 5.2/m2 in 2019. The combined population size was 429 ± 14 individuals. Recapture rates were highest at HWY 79 and lowest at HWY 294. Rates were significantly different between years (2016 vs 2019) but not between sites. These data offer insights into the current status of P. riddellii within the Neches River Basin, and some of the first population dynamic information for this species, which is crucial for its conservation.

DOI:10.31931/fmbc-d-21-00003

Running head: Density, survival, and population size of Louisiana Pigtoe

DENSITY, APPARENT SURVIVAL, AND LOCAL POPULATION SIZE OF LOUISIANA PIGTOE (*PLEUROBEMA RIDDELLII*) IN THE NECHES RIVER, TEXAS

David F. Ford[1*], Edith D. Plants-Paris[2], and Neil B. Ford[2]

[1]*Edge Engineering and Science, 16285 Park Ten Place #400, Houston, TX 77084 USA*

[2]*Department of Biology, University of Texas at Tyler, Tyler, TX, 75799 USA*

*Corresponding Author: dfford@edge-es.com

ABSTRACT

Most North American unionids are imperiled to some degree, including the Louisiana Pigtoe, *Pleurobema riddellii*, which is currently under review for listing under the U.S. Endangered Species Act. Understanding a species' population dynamics, including spatial and temporal variation in survival, density, recruitment and population size, is vital for conservation, but this information is lacking for *P. riddellii*. We conducted a mark-recapture study to estimate apparent survival, density, recruitment, and population size of *P. riddellii* within a 25 m^2 area at three sites (75 m^2 total) in the Neches River, Texas, from 2014 to 2019. We used the program MARK to evaluate POPAN models for estimating population parameters. We collected a total of 392 unique individuals of *P. riddellii* over the five-year period and the observed recapture rate averaged 55.6%. The most parsimonious POPAN model indicated that apparent survival varied temporally, the recapture rate varied temporally and spatially, and both the entry probability (recruitment) and population size varied spatially. Apparent survival averaged 83.3% ± 3.4 (SE)/year, overall population size across the three sites was 429 individuals (overall density = 5.7/m^2), and recruitment averaged 6.3%/year. High survival, relatively high density, the presence of recruitment, and the lack of temporal changes in population size suggest that these populations are stable. The presence of *P. riddellii* throughout a long section of the river, including localized patches of higher abundance, suggests that the total population size in the Neches River is relatively large and the river is a global stronghold for the species.

KEY WORDS - recapture rates, mark-recapture, MARK, population dynamics, vital rates, long-term

INTRODUCTION

Estimates of population vital rates and population size are important for effective species conservation (Matter et al. 2013). Vital rates, such as survival and recruitment, are the main determinants of a population's growth rate and ultimately, its viability (Akçakaya et al. 2004; Bonnot et al. 2011; Connette and Semlitsch 2015; Newton et al. 2020). Population size can influence viability primarily because small populations can be more vulnerable to Allee effects or biotic and abiotic factors (Kramer et al. 2009; Nystrand et al. 2010). Population models incorporating vital rates and population size can inform conservation efforts by making predictions about the resilience of a species to environmental impacts (Fonnesbeck and Dodd 2003; Connette and Semlitsch 2015).

North America's freshwater mussels (Unionoidae) are one of the most highly imperiled faunal groups on the continent (Williams et al. 1993; Bogan 2008; Haag 2012). Information about mussel population dynamics is especially important for evaluating population viability and responses to various environmental and anthropogenic factors. Annual survival and recruitment differ widely among mussel species, and these patterns can have a large influence on population growth and stability (e. g. Payne and Miller 2000; Villella et al. 2004; Haag 2012). However, vital rates remain unknown for numerous mussel species, and the long lifespan of many species requires multi-year sampling to estimate those factors (Villella et al. 2004; Newton et al. 2011; Newton et al. 2020). Mark-recapture studies can provide relatively unbiased estimates of population size and survival rates, which can be difficult to estimate directly (Daura-Jorge and Simões-Lopes 2014; Pace et al. 2017; Schachat et al. 2019).

We used a mark-recapture study to estimate apparent survival, recruitment, and population size for the Louisiana Pigtoe, *Pleurobema riddellii*, at three sites in the Neches River of eastern Texas from 2014 to 2019. This species in currently under review by the U. S. Fish and Wildlife Service (2009) for listing under the U. S. Endangered Species Act. Little life history and population information is available for this species, and these data will be valuable to future conservation efforts.

METHODS

Study Species

Pleurobema riddellii was known historically from portions of western Louisiana, eastern Texas, and Red River tributaries in Arkansas (Vidrine 1993; Howells et al. 1996; Howells 2010, 2014). The species has experienced a large range constriction over the past decades, and sizable populations in Texas are currently known only from the upper Neches River basin (Burlakova et al. 2011; Ford et al. 2014, D. F. Ford et al. 2016). In the Neches River, *P. riddellii* occurs in riffles and shallow to moderately deep runs in stable gravel-and cobble-substrates (N. B. Ford et al. 2016; Glen 2017) and is a host specialist on drift-feeding minnows (*Pimephales vigilax*, *Cyprinella venusta* and *Cyprinella lutrensis*; Hinkle 2018; Marshall et al. 2018). Estimates of individual growth are available for the species and maximum life span is likely over 40 years (Ford et al. 2020). However, estimates of population vital rates and population size are lacking.

Study Area

The Neches River is a sixth-order stream and drains approximately 26,000 km^2 (Texas Parks and Wildlife Department 1974; Horizon System Corporation 2015). Seasonal stream flow patterns were similar among all years of our study (2014-2019; United States Geological Survey gage 0803200 Neches River near Neches, TX, https://waterdata.usgs.gov/tx/nwis/nwis, accessed February 10, 2021), except for the winter of 2015 and most of 2016 during which flow was consistently high (>30 m^3/second, maximum = 134 m^3/second).

We selected three study sites that supported the highest abundance of *P. riddellii* observed over multiple years of mussel surveys in the Neches River basin (Walters et al. 2017; Ford et al. 2020). The most upstream site (HWY 79) was 8.6 km downstream of the Highway 79 bridge (Anderson County), the next site (CHC) was 22.2 km downstream of the HWY 79 site near Cherokee Hunting Club Road (Cherokee County), and the most downstream site (HWY 294) was 11.3 km downstream of CHC, upstream from the Highway 294 bridge (Cherokee County). We established a 150-m study reach at each site.

In 2014, we conducted initial site sampling by dividing each 150-m study reach into three 50-m segments and excavating 27 – 0.25 m^2 quadrats in each segment (total of 81 quadrats in each 150-m reach). In each 50-m segment, we distributed the 27 quadrats across the stream by placing nine quadrats at randomly chosen locations in the center of the stream and nine quadrats at randomly chosen locations along each bank. We calculated an estimate of mean density of *P. riddellii* in each of the three 150-m reaches as the mean density among the 81 total quadrats (three sets of 27 quadrats per reach). In each 150-m reach, we identified the quadrat with the highest number of *P. riddellii* and established a 5 m x 5 m grid (25 m^2) centered on that quadrat for the mark-recapture study. No *P. riddellii* were collected in initial site sampling at HWY 79; we conducted a qualitative search at this site and located the 5 x 5 m grid where the first specimen was found.

Sampling Methods

We sampled the 25-m^2 grids at each site once/year in late summer or early fall during low water conditions from 2014 to 2019, but we did not sample in 2018. We sampled each grid by placing a 1-m^2 quadrat at one corner of the grid, searching it for mussels by excavating the substrate, and then flipping the quadrat over to the adjacent 1-m^2 location until the entire 25-m^2 area was searched. We affixed a passive integrated transponder (PIT tag, Biomark, Boise, ID, USA) to the shell and a numbered bee tag (Betterbee, Greenwich, NY, USA) to the opposite valve. We measured shell length of each *P. riddellii* encountered and then returned all individuals to the substrate in the grid. After 2014, we made an initial pass over the grid with a PIT tag receiver to locate previously tagged individuals and then excavated the grid as described above to ensure that all individuals were collected. On each sampling occasion, we recorded the tag numbers and measured all recaptured *P. riddellii* and tagged and measured newly encountered individuals. We also recorded dead individuals encountered in the grid. Loss of tags was rare, and no individuals lost both tags, which allowed us to identify all recaptured mussels.

Mark-recapture Analysis

We calculated recapture rates of *P. riddellii* for each sampling event as:

$R_c = T_r/T_m$,

where R_c is the recapture rate for the sampling event, T_r is the number of marked *P. riddellii* recovered during the sampling event, and T_m is the total number of *P. riddellii* marked prior to the sampling event.

We used the POPAN model in the program MARK (White and Burnham 1999) for our mark-recapture analysis. This model has the following assumptions: 1) marks are not lost and can be read correctly, 2) sampling is instantaneous and animals are released immediately after sampling, 3) the study area remains constant, and its size does not change, 4) all animals (marked and unmarked) have an equal probability of survival between each sampling event, and 5) all animals (marked and unmarked) have an equal probability of being captured between each pair of sampling events. Generally, these assumptions were met by our study. PIT tags may have allowed somewhat greater capture rates of previously marked individuals, but our thorough excavation of the grids likely effectively detected untagged individuals (see Discussion).

The POPAN model calculates four statistics, apparent survival (ϕ), the recapture probability during the sampling event (*p*), the probability of a new individual entering or being located within the sample area from the total population (entry probability, *pent*), and super-population size (*N*). Apparent survival is the probability of an individual surviving between sampling events, given that the organism is still present within the site, while the recapture probability is the probability of an individual being captured during a sampling event assuming it is alive. Entry probability (*pent*) is the probability of entry from the population (the population in the 25-m^2 grid) into the study area, as a result of immigration or birth (i.e., recruitment). We interpreted estimates of *pent* derived from the POPAN model to represent annual recruitment. Adult mussels are relatively sedentary, but it is possible that some individuals moved into or out of a sampling grid. However, given the large size of the grid, this is unlikely except along the edges (Schwalb and Pusch 2007), and the number of immigrating or emigrating adults is expected to be low (Newton et al. 2015; 2020). Juveniles that recruited to a grid by dropping off host fishes initially are too small to be detected by our sampling but are detectable after about three years, at which time they average >20 mm length (Ford et al. 2020). The super-population size (hereafter referred to as population) is considered the number of individuals ever present in the sampling area. We calculated *N* for each of the three 25-m^2 grids. Both *N* and recruitment were rounded to the nearest whole individual. We calculated all parameters using a one-year time interval between successive samples, except for 2017 to 2019, where we used a two-year time interval to account for the lack of sampling in 2018.

We included a group effect (sampling site) and a time effect (year) in the POPAN models to evaluate spatial and temporal variation in model parameters. We used corrected Akaike's information criterion to rank candidate models. We used $\hat{C}$-adjusted quasi-AIC ($QAIC_c$) values to select the most parsimonious model from the list of candidate models, and we used a goodness-of-fit test in the program RELEASE in MARK to determine the fit of a chosen model. The most parsimonious model is the one with the smallest $QAIC_c$ value, which explains most of the variation in the data, while using the least number of model parameters.

RESULTS

Between 2014 and 2019 we captured a total of 392 *P. riddellii* individuals from all three sites and found eight (2.0%) dead individuals (Table 1). All dead individuals were recovered from HWY 79 in 2017 (three individuals) and 2019 (five individuals). Of the 392 *P. riddellii* individuals, we had a total of 944 captures, including 138 individuals (35.2%) that were captured once and not recaptured, 69 (17.6%) that were recaptured once (initial capture + 1 recapture), 94 (24.0%) that were recaptured twice, 69 (17.6%) that were recaptured three times, and 22 (5.7%) that were recaptured in all sampling events after 2014. Recapture rate averaged 55.6% across all sites and years. Recapture rates did not differ between sites (ANOVA, $F_{9,11} = 4.26$, $P = 0.480$) but were significantly different between sampling years ($F_{12,15} = 3.49$, $P = 0.001$). Recapture rates differed only between 2016 and 2019 (Tukey HSD, $P < 0.001$). At all sites, initial captures of untagged individuals declined from 2014 to 2017, but initial captures increased in 2019

(Figure 1). Conversely, recapture rates generally increased during the first three years then remained relatively steady after 2016, except in 2019, when recapture rates appeared to decrease substantially, particularly at HWY 294. Mean *Pleurobema riddellii* density across all three 25-m^2 grids was 1.7/m^2 in 2014 and 2.7/m^2 in 2019 (mean = 2.5/m^2; Table 1).

The most parsimonious POPAN model included apparent survival (ϕ), which varied temporally; recapture probability (*p*), which varied spatially and temporally; and entry probability (*pent*) and population size (*N*), which both varied spatially ($\chi^2_{(21)}$, $P = 0.002$; Table 2). Mean survival across sites was 83.3% ± 3.4 (SE). Apparent survival was >80% in all years, except between 2017 to 2019 when it was 73.8% (Table 3). Recapture probability (*p*) averaged 67.9% (range = [38.5–95.3%]) across all sites and years. Both the lowest (2015) and highest (2017) recapture probabilities were found at CHC. Recapture probability was lowest for all sites in 2015 and highest in 2017, except at HWY 79, where recapture probability was highest in 2019. Entry probability (*pent*) across all three sites ranged from 2.8% ± 1.4 (SE) at HWY 79 to 10.8% ± 1.3 at HWY 294 (mean across sites = 6.3% ± 1.5). Assuming all individuals entering the populations originated from recruitment, values of *pent* represent the addition of 2 ± 2 (SE) to 27 ± 3 individuals/year. The estimated total population size across all three sites was 429 ± 8 (SE) individuals (Table 3).

Of the total 79 *P. riddellii* captures at HWY 79, 14 (17.7%) individuals were captured once, 10 (12.7%) were recaptured once, 32 (40.5%) were recaptured twice, 17 (21.5%) were recaptured three times, and six (7.9%) were recaptured in all sampling events after 2014. The recapture rate at HWY 79 averaged 63.6%. We found no *P. riddelli* at HWY 79 during the initial site sampling in 2014. Within the 25-m^2 grid, densities were 1.3/m^2 and 1.6/m^2 in 2014 and 2019, respectively (Table 1). *Pent* was the lowest of the three sites, and an estimated 2 ± 2 (SE) new individuals recruited to the site each year. The estimated population size at HWY 79 (87 ± 4 [SE] individuals; Table 3) was the lowest of any site.

Of the total 87 *P. riddellii* captures at CHC, 27 (31.0%) were captured once, 18 (20.7%) were recaptured once, 16 (18.4%) were recaptured twice, 20 (23.0%) were recaptured three times, and six (6.9%) were recaptured in all sampling events after 2014. The recapture rate at CHC averaged 58.1%. Density of *P. riddellii* was 2.8/m^2 during the initial site sampling in 2014 (Table 1). Within the 25-m^2 grid, densities were 1.2/m^2 in 2014 and 1.7/m^2 in 2019 (Table 1). *Pent* indicated that an estimated 5 ± 2 (SE) new individuals recruited to the site each year. The estimated population size at CHC (96 ± 4 [SE] individuals) was similar to HWY 79 but much lower than HWY 294 (Table 3).

Of the total 226 *P. riddellii* captures at HWY 294, 97 (42.9%) were captured once, 41 (18.1%) were recaptured once, 46 (20.4%) were recaptured twice, 32 (14.2%) were recaptured three times, and 10 (4.4%) were recaptured during all sampling events after 2014. Recapture rate at HWY 294 averaged 50.5%. Density of *P. riddellii* was 0.2/m^2 during the initial site sampling. Within the 25-m^2 grid, densities were 2.6/m^2 in 2014 and 4.7/m^2 in 2019 (Table 1). *Pent* was highest at this site, and an estimated 27 ± 3 (SE) new individuals recruited to the site each year. The estimated population size at HWY 294 (246 ± 6 [SE] individuals) was the highest observed at any site (Table 3).

Density estimates differed substantially among sampling approaches and analytical methods. Based on the area of the 25-m^2 grid and estimates of population size from the POPAN model, estimated densities were 3.5/m^2 at HWY 79, 3.8/m^2 at CHC, and 9.8/m^2 at HWY 294 (overall = 5.7/m^2; see Table 3). These estimates were very similar to estimates based on area sampled and the total number of unique individuals captured across all five years of sampling in the 25-m^2 grid (3.2/m^2 at HWY 79, 3.5/m^2 at CHC, 9.0.m^2 at HWY 294, 5.2/m^2 overall; Table 1). However, density estimates from quadrat sampling in individual years were about 50% lower than estimates made by the previous two methods (Table 1).

DISCUSSION

Although density and population size varied among sites, all three of our study sites in the Neches River appear to support relatively large populations of *P. riddellii*, with densities of about 3–9 individuals/m^2. Other parameters suggest that these populations are stable, particularly the lack of temporal variation in population size. Survival varied across time but not by location, suggesting that annual, river-wide variation in environmental factors was a more important determinant of survival than local variation among sites. However, apparent survival was generally high (usually >80%), similar to values reported for several other mussel species from stable populations (e.g., 87–>97%, Hart et al. 2001; Villela et al. 2004; Meador et al. 2011; Reátegui-Zirena et al. 2013; Wisniewski et al. 2014; Hyde et al. 2017). Values of *pent* were relatively low, but they represent recruitment strength similar to that seen in other stable mussel populations (e.g., 1–45%, Villella et al. 2004; Haag and Warren 2010; Matter et al. 2013); in contrast, declining populations are often characterized by a near absence of recruitment (Haag 2012).

It is important to note that our density and population size estimates for these local populations do not reflect the overall abundance of *P. riddellii* throughout the Neches River. Our estimates were obtained from small areas of high *P. riddellii* density at sites identified by previous surveys as having the highest density of the species in the river. Indeed, previous surveys at other sites in the Neches River basin found lower densities of *P. riddellii* at most sites

(0.39–0.79/m^2; Andrew Glen, personal communication; Ford et al. 2014; D. F. Ford 2016). Nevertheless, the presence of *P. riddellii* throughout a long section of the river, including localized patches of higher abundance, suggests that the total population size in the Neches River is relatively large.

Variation in habitat characteristics among sites (see Ford et al. 2020) may partially explain the higher density and population size observed at HWY 294. The HWY 294 site had an extensive shallow riffle with gravel-and-cobble substrate. The CHC site had a deeper riffle with more mud and silt, and HWY 79 did not have a riffle but instead consisted of deeper, pooled habitat. *Pleurobema riddellii* is thought to prefer gravel-and-cobble substrates (Glen 2017; Ford et al. 2020), and the known hosts for *P. riddellii* are riffle-dwelling minnows (Hinkle 2018; Marshall et al. 2018). The greater abundance of riffle habitat at HWY 294 may have provided more habitat for *P. riddellii* and its host fishes.

Actual survival is difficult to estimate but is often higher than apparent survival because permanent emigration from the study area results in biased estimates of apparent survival (Gilroy et al. 2012; Hyde et al. 2017). Permanent emigration is considered less of an issue for estimating survival of mussels because of their sedentary nature (Balfour and Smock 1995; Villela et al. 2004; Newton et al. 2020), but mussels can move substantial distances in some cases (Haag 2012; Daniel and Brown 2014; Newton et al. 2015). Because we did not sample for missing *P. riddellii* outside of the mark-recapture area, we could have missed individuals that emigrated out of the 25-m^2 grid or that were displaced by the 2016 flood. Temporary emigration, such as burrowing deeper into the substrate during colder months or higher flows, also could have biased our survival estimates, but we sampled in late summer and early fall when the water was warm and the flow was low.

The use of PIT tags may have introduced some bias into our parameter estimates by increasing the likelihood of recapturing tagged individuals compared with previously uncaptured individuals (see Kurth et al. 2007). However, even by using a PIT tag reader we missed a substantial proportion of tagged individuals in any given year. This fact, combined with our extensive excavation of the substrate to find unmarked individuals, probably minimized any bias associated with the use of PIT tags.

Our estimates of density varied markedly between the initial site sampling of the 150-m reach in 2014 and later sampling of the 25-m^2 grids, even in 2014. Higher density in the 25-m^2 grids, as observed at two of the sampling sites, was expected because the areas with the highest density in the 150-m reach were selected for the sampling grids. However, our density estimates were substantially lower in the 25-m^2 grids than over the 150-m reach at CHC. The variation in density estimates between our 150-m reach and the 25-m^2 grids illustrates the characteristically patchy nature of mussel distribution (Strayer 1999; Strayer et al. 2004) and the effects of scale on sample estimates. Based on the results of our broader-scale initial site sampling at HWY 79 and HWY 294, *P. riddellii* might have been considered absent or rare, respectively, at those sites, but our more focused sampling of the 25-m^2 grids revealed that both sites supported substantial populations. Conversely, our initial site sampling at CHC indicated a higher density than revealed by our sampling of the 25-m^2 grids.

The large difference in density estimates between our annual samples and longer-term sampling illustrate other sample design issues. Our estimates from quadrat sampling in individual years were about 50% lower than estimates from the mark-recapture model or from the combined five-year quadrat sampling dataset. This discrepancy is probably explained by our overall observed annual recapture rate (55.6%) and our estimate of overall recapture probability from the POPAN model (67.9%). Detectability is rarely 100%, but we appear to have missed a substantial proportion of the population in any given year despite our focused sampling in a small area and extensive excavation of the substrate. Multi-year sampling is often impractical to implement on a large scale. Our sampling methods were broadly similar to adaptive sampling, in which additional sampling effort is allocated in areas where the target species is found (Strayer and Smith 2003). A more formalized application of adaptive sampling may be appropriate when the goal of a study is to provide accurate density estimates in a single sampling effort for a patchily distributed species. In addition, accounting for detectability may help provide more accurate density estimates in single sampling events (e.g., Smith et al. 2000; Bailey et al. 2004; Wisniewski et al. 2014).

Our multi-year sampling of *P. riddellii* populations provided estimates of density, survival, population size, and recruitment that are important for conservation efforts. These estimates provide baseline data for monitoring of the species' status over time. The lack of temporal variation in population size, high survival, and apparent levels of recruitment we document suggest that these local populations are stable. Our population parameter estimates can be coupled with other demographic information to construct population models, which can provide a quantitative assessment of the current trajectory and viability of *P. riddellii* populations (e.g., increasing, stable, decreasing). The occurrence of *P. riddellii* at relatively high density throughout a long, interconnected reach of the Neches River indicates that the river is a global stronghold for this species. The decline of *P. riddellii* and other mussel species across Texas (Howells et al. 1997; Randklev et al. 2010) highlights the imperative for protection of the Neches River basin.

LITERATURE CITED

Akçakaya, H. R., V. C. Radeloff, D. J. Mladenoff, and H. S. He. 2004. Integrating landscape and metapopulation modeling approaches: viability of the sharp-tailed grouse in a dynamic landscape. Conservation Biology 18:526–537.

Bailey, L. L., T. R. Simons, and K. H. Pollock. 2004. Estimating site occupancy and species detection probability parameters for terrestrial salamanders. Ecological Applications 14:692–702.

Balfour, D. L., and L. A. Smock. 1995. Distribution, age, and movements of the freshwater mussel *Elliptio complanata* (Mollusca: Unionidae) in a headwater stream. Journal of Freshwater Ecology 10:255–268.

Bogan, A. E. 2008. Global diversity of freshwater mussels (Mollusca, Bivalvia) in freshwater. Hydrobiologia 595:139–147.

Bonnot, T. W., F. R. Thompson III, and J. J. Millspaugh. 2011. Extension of landscape-based population viability models to ecoregional scales for conservation planning. Biological Conservation 144:2041–2053.

Burlakova, L. E., A. Y. Karatayev, V. A. Karatayev, M. E. May, D. L. Bennett, and M. J. Cook. 2011. Biogeography and conservation of freshwater mussels (Bivalvia: Unionidae) in Texas: Patterns of diversity and threats. Diversity and Distributions 17:393–407.

Connette, G. M., and R. D. Semlitsch. 2015. A multistate mark-recapture approach to estimating survival of PIT-tagged salamanders following timber harvest. Journal of Applied Ecology 52:1316–1324.

Daniel, W. M., and K. M. Brown. 2014. The role of life history and behavior in explaining unionid mussel distributions. Hydrobiologia 734:57–68.

Daura-Jorge, F. G., and P. C. Simões-Lopes. 2014. Mark-recapture vs. line-transect abundance estimates of a coastal dolphin population: a case study of *Tursiops truncates* from Laguna, southern Brazil. Latin American Journal of Aquatic Mammals 11:133–143.

Fonnesbeck, C. J., and C. K. Dodd Jr. 2003. Estimation of flatted musk turtle (*Sternotherus depressus*) survival, recapture, and recovery rate during and after a disease outbreak. Journal of Herpetology 37:602–607.

Ford, D. F., A. D. Walters, L. R. Williams, M. G. Williams, and N. B. Ford. 2016. Mussel assemblages in streams of different sizes in the Neches River Basin of Texas. Southeastern Naturalist 15:26–40.

Ford, D. F., E. D. Plants-Paris, and N. B. Ford. 2020. Comparison of Louisiana Pigtoe (*Pleurobema riddellii*, Mollusca, Unionidae) growth at three different locations in the Neches River Basin of east Texas. Hydrobiologia 847:2021–2033.

Ford, N. B., K. Heffentrager, D. F. Ford, A. D. Walters, and N. Marshall. 2014. Significant recent records of unionid mussels in northeast Texas rivers. Walkerana 17:8–15.

Ford, N. B., L. R. Williams, M. G. Williams, J. Banta, J. Placyk, and H. Hawley. 2016. Final report: endangered species research projects for freshwater mussels, Region 2, East Texas. Texas Comptroller of Public Accounts, Austin, Texas. Available at https://comptroller.texas.gov/programs/natural-resources/docs/reports/ComptrollerFinalReport2016reduced.pdf (accessed April 25, 2022).

Gilroy, J. J., T. Virzi, R. L. Boulton, and J. L. Lockwood. 2012. A new approach to the "apparent survival" problem: estimating true survival rates from mark–recapture studies. Ecology 93:1509–1516.

Glen, A. R. 2017. Examining the relationship between mesohabitats and freshwater mussels in an east Texas River. Masters' Thesis. University of Texas at Tyler, Tyler.

Haag, W. R. 2012. North American freshwater mussels: natural history, ecology, and conservation. Cambridge University Press, New York.

Haag, W. R., and M. L. Warren. 2010. Diversity, abundance, and size structure of bivalve assemblages in the Sipsey River, Alabama. Aquatic Conservation: Marine and Freshwater Ecosystems 20:655–667.

Hart, R. A., J. W. Grier, A. C. Miller, and M. Davis. 2001. Empirically derived survival rates of a native mussel, *Amblema plicata*, in the Mississippi and Otter Tail Rivers, Minnesota. American Midland Naturalist 146:254–263.

Hinkle, E., 2018. Suitable host fish, population structure, and life-history characteristics for the state-listed, Louisiana pigtoe, *Pleurobema riddellii*. Master's thesis. University of Texas at Tyler, Tyler.

Horizon Systems Corporation. 2015. NHDPlus Version 2. Available at http://www.horizon-systems.com/nhdplus/NHDPlusV2_home.php (accessed January 15, 2015).

Howells, R. G. 2010. Louisiana Pigtoe (*Pleurobema riddellii*): summary of biological and ecological data for Texas. Prepared for the Save Our Springs Alliance, Austin, Texas. Biostudies, Kerrville, Texas.

Howells, R. G. 2014. Field guide to Texas freshwater mussels (2nd Edition). Biostudies, Kerrville, Texas.

Howells, R. G., R. W. Neck, and H. D. Murray. 1996. Freshwater mussels of Texas. University of Texas Press, Austin.

Howells, R. G., C. M. Mather, and J. A. M. Bergmann. 1997. Conservation status of selected freshwater mussels in Texas. Pages 117–128 in K. S. Cummings, A. C. Buchanan, C. A. Mayer, and T. J. Naimo (editors). UMRCC Symposium Proceedings. Upper Mississippi River Conservation Committee Symposium, Rock Island.

Hyde, J. M., B. B. Niraula, J. M. Miller, J. T. Garner, and P. M. Stewart. 2017. Estimation of apparent survival, detectability, and density of three federally threatened mussel species in a small watershed. Freshwater Mollusk Biology and Conservation 20:20–31.

Kramer, A. M., B. Dennis, A. M. Liebhold, and J. M. Drake. 2009. The evidence for Allee effects. Population Ecology 51:341–354.

Kurth, J., C. Loftin, J. D. Zydlewski, and J. Rhymer. 2007. PIT tags increase effectiveness of freshwater mussel recaptures. Journal of the North American Benthological Society 26:253–260.

Marshall, N. T., J. A. Banta, L. R. Williams, M. G. Williams, and J. S. Placyk Jr. 2018. DNA barcoding permits identification of potential fish hosts of unionid freshwater mussels. American Malacological Bulletin 36:42–56.

Matter, S. F., F. Borrero, and C. Fleece. 2013. Modeling the survival and population growth of the freshwater mussel, *Lampsilis radiata luteola*. American Midland Naturalist 169:122–136.

Meador, J. R., J. T. Peterson, and J. M. Wisniewski. 2011. An evaluation of the factors influencing freshwater mussel capture probability, survival and temporary emigration in a large lowland river. Freshwater Science 30:507–521.

Newton, T. J., S. J. Zigler, J. T. Rogala, B. R. Gray, and M. Davis. 2011. Population assessment and potential functional roles of native mussels in the Upper Mississippi River. Aquatic Conservation: Marine and Freshwater Ecosystems 21:122–131.

Newton, T. J., S. J. Zigler, and B. R. Gray. 2015. Mortality, movement, and behaviour of native mussels during a planned water-level drawdown in the Upper Mississippi River. Freshwater Biology 60:1–15.

Newton, T. J., S. J. Zigler, P. R. Schrank, M. Davis, and D. R. Smith. 2020. Estimation of vital population rates to assess the relative health of mussel assemblages in the Upper Mississippi River. Freshwater Biology 65:1726–1739.

Nystrand, M., M. Griesser, S. Eggers, and J. Ekman. 2010. Habitat-specific demography and source-sink dynamics in a population of Siberian jays. Journal of Animal Ecology 79:266–274.

Pace III, R. M., P. J. Corkeron, and S. D. Kraus. 2017. State-space mark-recapture estimates reveal a recent decline in abundance of North Atlantic right whales. Ecology and Evolution 7:8730–8741.

Payne, B. S., and A. C. Miller. 2000. Recruitment of *Fusconaia ebena* (Bivalvia: Unionidae) in relation to discharge of the lower Ohio River. American Midland Naturalist 144:328–341.

Randklev, C. R., S. Wolverton, B. Lundeen, and J. H. Kennedy. 2010. A paleozoological perspective on unionid (Mollusca: Unionidae) zoogeography in the upper Trinity River basin, Texas. Ecological Applications 20:2359–2368.

Reátegui-Zirena, E. G., P. M. Stewart, and J. M. Miller. 2013. Growth rates and age estimations of the fuzzy pigtoe, *Pleurobema strodeanum*: a species proposed for listing under the Endangered Species Act. Southeastern Naturalist 12:161–170.

Schachat, S. R., C. C. Labandeira, M. E. Clapham, and J. L. Payne. 2019. A Cretaceous peak in family-level insect diversity estimated with mark-recapture methodology. Proceedings of the Royal Society B: Biological Sciences 286:20192054.

Schwalb, A. N., and M. T. Pusch. 2007. Horizontal and vertical movements of unionid mussels in a lowland river. Journal of the North American Benthological Society 26:261–272.

Smith, D. R., R. F. Villela, D. P. Lemarié, and S. von Oettingen. 2000. How much excavation is needed to monitor freshwater mussels? Pages 203–218 in R. A. Tankersley, D. I. Warmolts, G. T. Watters, B. J. Armitage, P. D. Johnson, and R. S. Butler (editors). Freshwater Mollusk Symposium Proceedings. Ohio Biological Survey, Columbus.

Strayer, D. L. 1999. Use of flow refuges by unionid mussels in rivers. Journal of the North American Benthological Society 18:468–476.

Strayer, D. L., and L. C. Smith. 2003. A guide to sampling freshwater mussel populations. Monograph, 8. American Fisheries Society, Bethesda, Maryland.

Strayer, D. L., J. A. Downing, W. R. Haag, T. L. King, J. B. Layzer, T. J. Newton, and S. J. Nichols. 2004. Changing perspectives on pearly mussels, North American's most imperiled animals. Bioscience 54:429–439.

Texas Parks and Wildlife Department. 1974. An analysis of Texas waterways: a report on the physical characteristics of rivers, streams, and bayous in Texas. Texas Parks and Wildlife Press, Austin, Texas.

United States Fish and Wildlife Service. 2009. Endangered and threatened wildlife and plants: partial 90-day finding on a petition to list 475 species in the southwestern United States as threatened or endangered with critical habitat. Federal Register 74:66865–66905.

Vidrine, M. F. 1993. The historical distributions of freshwater mussels in Louisiana. Gail Q. Vidrine Collectibles, Eunice, Louisiana.

Villella, R. F., D. R. Smith, and D. P. Lemarié. 2004. Estimating survival and recruitment in a freshwater mussel population using mark-recapture techniques. American Midland Naturalist 151:114–133.

Walters, A. D., D. Ford, E. T. Chong, M. G. Williams, N. B. Ford, L. R. Williams, and J. A. Banta. 2017. High- resolution ecological niche modelling of threatened freshwater mussels in east Texas, USA. Aquatic Conservation: Marine and Freshwater Ecosystems 27:1251–1260.

White, G. C., and K. P. Burnham. 1999. Program MARK: survival estimation from populations of marked animals. Bird Study 46:120–139.

Williams, J. D., M. L. Warren Jr., K. S. Cummings, J. L. Harris, and R. J. Neves. 1993. Conservation status of freshwater mussels of the United States and Canada. Fisheries 18:6–22.

Wisniewski, J. M., N. M. Rankin, D. A. Weiler, B. A. Strickland, and H. C. Chandler. 2014. Use of occupancy modeling to assess the status and habitat relationships of freshwater mussels in the lower Flint River, Georgia, USA. Walkerana 17:24–40.

Table 1. Densities (number/m^2) of *Pleurobema riddellii* estimated from initial site sampling and later sampling of the 25-m^2 grids at three sites in the Neches River, Texas, from 2014 to 2019. Numbers in parentheses are the number of unique individuals located during each sampling event. The column "Mean" represents mean values across all five years. The column "Totals" represents density estimates based on the total number of unique individuals encountered across all five years and the sample area (25 m^2, or 75 m^2 for "Overall").

		25-m^2 mark-recapture grid						
Site	Initial site sampling (2014)	2014	2015	2016	2017	2019	Mean	Totals
HWY 79	0.0	1.3 (32)	2.2 (54)	2.2 (54)	2.2 (56)	1.6 (40)	1.9 (47)	3.2 (79)
CHC	2.8	1.2 (29)	1.9 (48)	2.6 (65)	1.5 (37)	1.7 (42)	1.8 (44)	3.5 (87)
HWY 294	0.2	2.6 (64)	4.2 (104)	4.9 (123)	3.4 (86)	4.7 (118)	4.0 (99)	9.0 (226)
Overall	1.0	1.7 (125)	2.8 (206)	3.2 (242)	2.3 (172)	2.7 (200)	2.5 (189)	5.2 (392)

Table 2. AIC Ranking of POPAN models for estimating mark-recapture parameters for *Pleurobema riddellii* at three sites in the Neches River, Texas, from 2014 to 2019. Model parameters are apparent survival (ϕ), recapture probability (*p*), probability of entry (*pent*), and population size (*N*). Parameters denoted with (t) indicate variance by survey year, (g) indicates variance by sampling site, and (.) indicates no variance by sampling time or site. Parameters that are a function of year and site simultaneously are denoted by the interaction term (g*t). NP is the number of parameters used in the model.

Model	**QAICc**	**$QAIC_c$ Weight**	**NP**
Φ(t) *p*(g*t) *pent*(g) *N*(g)	1403.565	0.506	25
Φ(g*t) *p*(t) *pent*(.) *N*(g)	1405.074	0.238	21
Φ(g*t) *p*(t) *pent*(g) *N*(t)	1405.452	0.197	21
Φ(.) *p*(g*t) *pent*(g) *N*(g*t)	1409.072	0.032	22
Φ(g*t) *p*(g*t) *pent*(.) *N*(g*t)	1410.979	0.012	31
Φ(g) *p*(t) *pent*(g) *N*(g)	1411.691	0.009	14
Φ(g*t) *p*(g*t) *pent*(g) *N*(t)	1413.145	0.004	31
Φ(t) *p*(t) *pent*(.) *N*(g*t)	1415.697	0.001	13
Φ(g) *p*(t) *pent*(g) *N*(.)	1418.395	0.000	12
Φ(g*t) *p*(t) *pent*(.) *N*(.)	1418.772	0.000	19
Φ(t) *p*(t) *pent*(.) *N*(.)	1418.909	0.000	11
Φ(.) *p*(g*t) *pent*(.) *N*(.)	1418.931	0.000	20

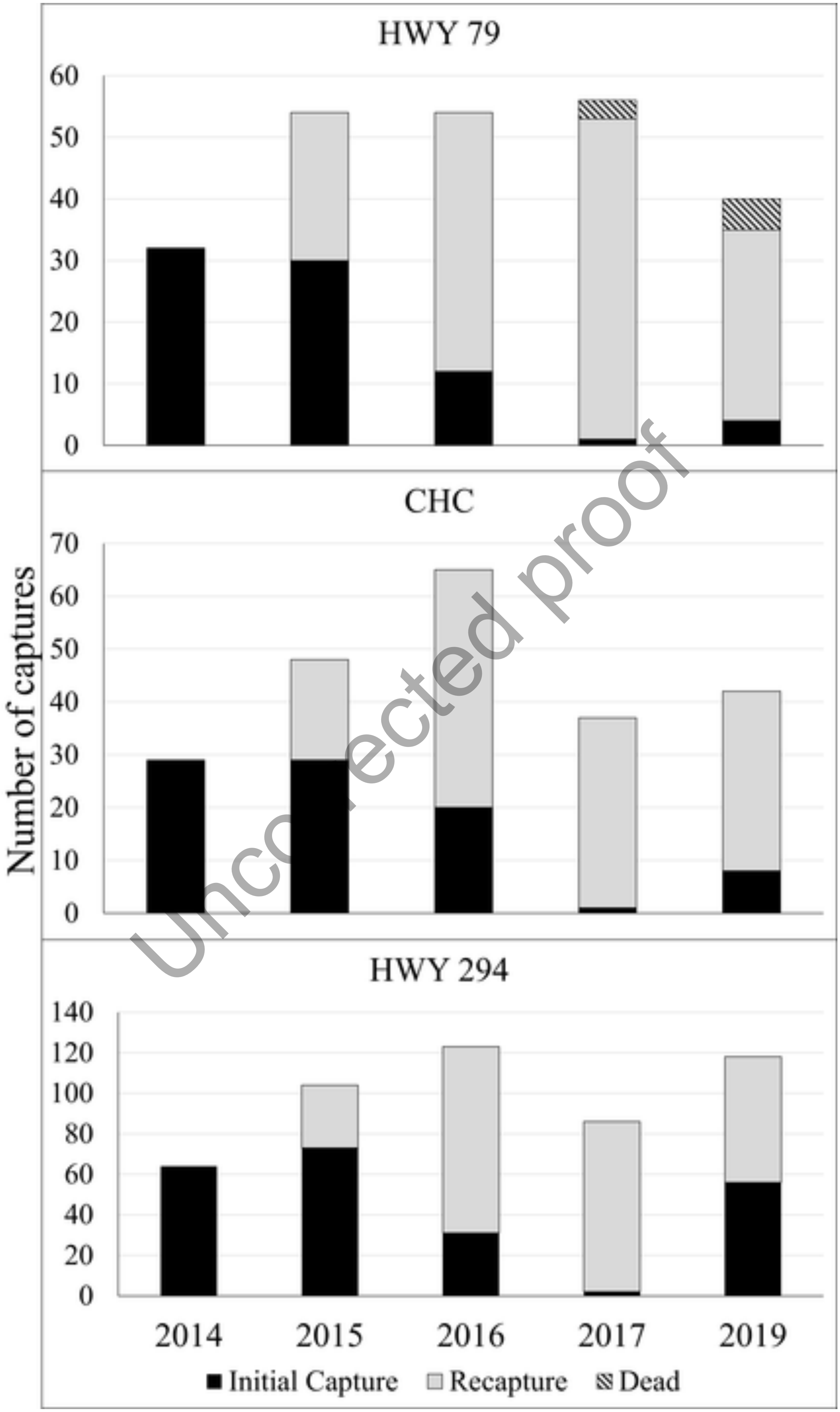
Table 3. POPAN model mark-recapture parameter estimates for *Pleurobema riddellii* at three sites in the Neches River, Texas, from 2014 to 2019. Parameters are apparent survival (ϕ), recapture probability (*p*), probability of entry into the sampling area from the overall population in the area (*pent*), population size (*N*), and recruitment (*r*) from the most parsimonious model (see Table 2). Error for each estimate is SE. Values of apparent survival, *pent* and *N*, are the same across sites or years, respectively, following the most parsimonious POPAN model, which indicated only temporal variation for apparent survival and only spatial variation for *pent* and *N*; the POPAN model indicated both temporal and spatial variation for *p*. Recruitment is the estimated number of recruits in each year and was estimated as (*pent*/100) × *N*. Population size and recruitment are rounded to the nearest whole individual.

			HWY 79		
Year	Φ	*p*	*pent*	*N*	*r*
2015	85.0 ± 3.7	41.3 ± 6.3	2.8 ± 1.4	87 ± 4	2 ± 2
2016	89.1 ± 2.6	74.3 ± 5.9	2.8 ± 1.4	87 ± 4	2 ± 2
2017	85.3 ± 4.3	80.3 ± 5.4	2.8 ± 1.4	87 ± 4	2 ± 2
2019	73.8 ± 3.1	88.2 ± 5.3	2.8 ± 1.4	87 ± 4	2 ± 2
			CHC		
2015	85.0 ± 3.7	38.5 ± 6.7	5.4 ± 1.8	96 ± 4	5 ± 2
2016	89.1 ± 2.6	68.8 ± 6.8	5.4 ± 1.8	96 ± 4	5 ± 2
2017	85.3 ± 4.3	95.3 ± 3.2	5.4 ± 1.8	96 ± 4	5 ± 2
2019	73.8 ± 3.1	62.4 ± 7.9	5.4 ± 1.8	96 ± 4	5 ± 2
			HWY 294		
2015	85.0 ± 3.7	45.9 ± 5.6	10.8 ± 1.3	246 ± 6	27 ± 3
2016	89.1 ± 2.6	74.0 ± 4.8	10.8 ± 1.3	246 ± 6	27 ± 3
2017	85.3 ± 4.3	84.6 ± 3.8	10.8 ± 1.3	246 ± 6	27 ± 3
2019	73.8 ± 3.1	61.6 ± 5.5	10.8 ± 1.3	246 ± 6	27 ± 3

FIGURE LEGENDS

Figure 1. Captures of *Pleurobema riddellii* at three sites in the Neches River, Texas, from 2014 to 2019.

Figure 1


HWY 79
60
50
40
30
20
10
0
CHC
70
60
50
40
30
20
10
0
HWY 294
140
120
100
80
60
40
20
0
Number of captures
2014
2015
2016
2017
2019
Initial Capture
Recapture
Dead