MAXIMIZING LEARNING OPPORTUNITIES IN CONSERVATION: GULF STURGEON STOCK STATUS AND TRENDS

By

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A DISSERTATION PRESENTED TO THE GRADUATE SCHOOL  
OF THE UNIVERSITY OF FLORIDA IN PARTIAL FULFILLMENT  
OF THE REQUIREMENTS FOR THE DEGREE OF  
DOCTOR OF PHILOSOPHY

UNIVERSITY OF FLORIDA

2023

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To Kara and Elijah

ACKNOWLEDGMENTS

First, I must thank my Lord and Savior, Jesus Christ, for His abundant grace in my life. I’d like to thank my advisor, Bill Pine, and my co-advisor, Andrew Carlson, for their mentorship, guidance, and encouragement during my graduate program. I’d also like to send a massive ‘thank you’ to the rest of the “database team”, Andrè Breton and Krystan Wilkinson, for their friendship and collaboration. Thank you to Jim Hines and Jennifer Moore. Your generous mentorship and support reduced learning curves and gave me access to expertise not found in textbooks or online message boards. I’d like to thank the rest of my supervisory committee, Lew Coggins, Greg Kiker, and Conor McGowan, for their feedback and encouragement throughout this process. I’d like to thank Chris Carroll and Big Fin Scientific for their fantastic data collection and management tools. To Tyler Coleman and Jamie Casteel, thank you for your unconditional friendship. You enriched my time at the University of Florida, and I love you both. I also want to thank all of the Gulf Sturgeon researchers from across the Gulf Coast who have shared their data and sturgeon knowledge with me: Melissa Price, Mike Randall (USGS), Joe Heublein (NMFS), Dewayne Fox (DSU), Adam Kaeser, Frank Parauka, Kayla Kimmel, Ashley Baer (USFWS), Adam Fox (UGA), Michael Andres (USM), and John Knight (FWC).

I’d like to thank my parents, Dave and Amy, for their relentless encouragement. To my wife, Kara, I could not have done this without your love and support. To my son, Elijah, you are a miracle. I eagerly await the opportunity to share my appreciation of God’s creation with you on a boat or in a duck blind. I love you both.

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Abstract of Dissertation Presented to the Graduate School  
of the University of Florida in Partial Fulfillment of the  
Requirements for the Degree of Doctor of Philosophy

MAXIMIZING LEARNING OPPORTUNITIES IN CONSERVATION:  
 GULF STURGEON STOCK STATUS AND TRENDS

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August 2023

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Gulf of Mexico Sturgeon *Acipenser oxyrinchus desotoi* (“Gulf Sturgeon”) are large, long-lived, anadromous fish designated as a threatened species under the Endangered Species Act. Hypothesized factors that contributed to declines in Gulf Sturgeon populations include overexploitation, loss of spawning habitat, and recent episodic mortality events such as the Deepwater Horizon oil spill. Like many wide-ranging migratory species, Gulf Sturgeon management decisions are often made at the Gulf-wide level, whereas data are primarily collected from seven river systems where spawning occurs. This creates a mismatch between the data available (discrete population units) and data needed for decision-making (range-wide population status). Threats to species recovery (e.g., recent oil spills, changing climate, and hydrologic modifications) occur at both spatial levels. Successful recovery planning necessitates integrating threat assessments, population status, and decision-making at the scale of the species. We addressed key information needs identified by NOAA and USFWS to inform recovery planning by modernizing data workflow and advancing analyses. This was achieved by (1) building a database of 19,000+ capture-recapture records over 30+ years of fieldwork, (2) implementing an electronic logbook program to reduce data entry errors, and (3) developing a tagging-based assessment framework, informed and evaluated via simulation, to understand spatially explicit patterns in demography and movement. Our results suggest consistently high site fidelity range-wide, but spatiotemporal differences in survival. We observed lower survival in the western Gulf (Pearl and Pascagoula rivers). This suggests that tuning threat assessments, restoration efforts, and management actions to target specific population segments may be more effective in promoting recovery than broad-scale, range-wide efforts. The underlying drivers of these differences in regional survival represent an important area for future research as lower survival rates may impede progress toward stated management recovery criteria. We also observed lower survival over short time periods that may represent episodic mortality events. Future monitoring efforts must utilize high-resolution data if managers are interested in measuring the effects of episodic mortality events, such as the Deepwater Horizon oil spill. Despite lower survival in the western Gulf, our analyses suggest that there is little concern for the future viability of the species.

CHAPTER 1

introduction

Gulf Sturgeon *Acipenser oxyrhynchus desotoi* are a long-lived anadromous species that was listed as threatened under the Endangered Species Act (ESA) in 1991 (NOAA and USFWS [National Oceanic Atmospheric Administration and U.S. Fish and Wildlife Service] 1991). Reasons for Gulf Sturgeon decline include range-wide in-river and nearshore habitat modification and unsustainable harvest (NOAA and USFWS 1991; Pine and Martell 2009; Ahrens and Pine 2014). While the closure of the commercial fishery was an important management action to promote Gulf Sturgeon recovery (Flowers et al. 2020) threats to Gulf Sturgeon recovery and downlisting from the ESA are possibly expanding including impacts from extractive oil and gas operations (PDARP) and more frequent severe weather events influenced by changing climate including frequency of extreme drought and storm events (Dula et al. 2022). Despite ESA listing for over 30 years and management actions including fishery closures and critical habitat designation and protection, Gulf Sturgeon population status, trends, and the best recovery path remains uncertain (Flowers et al. 2020).

Gulf Sturgeon were listed under the US Endangered Species Act in 1991 (56FR 49653; NOAA and USFWS 1991). Current management units for Gulf sturgeon include seven river systems and adjacent estuarine and marine habitats across the northern Gulf of Mexico from the Pearl River in Louisiana to the Suwannee River in Florida. The current Gulf Sturgeon Recovery Plan (GSRP) outlines a variety of criteria before populations can be considered recovered and delisting proposed (U.S. Fish and Wildlife Service [USFWS] 1995). A short-term goal of the GSRP is to ensure that wild stocks are not currently declining while the primary long-term goal of the GSRP is to establish self-sustaining populations that could allow delisting of the species, with a secondary goal of population recovery to a point at which directed fishing could be sustained (USFWS 1995). The primary factors that have been identified as potentially contributing to declines in Gulf sturgeon populations historically throughout their native range include overfishing, loss of spawning habitat, alteration of riverine habitat, or a combination of these and other factors (Clugston et al. 1995; Zehfuss et al. 1999). Since about 2003 “other factors” that are associated with known Gulf sturgeon mortality events include fish kills associated with paper plant effluent, major hurricanes (i.e., Katrina, Rita, Michael), recurring red tide events, and the 2010 Deepwater Horizon oil spill. Natural events such as hurricanes represent a type of episodic mortality occurring throughout the evolutionary history of the species. Anthropogenic disturbances such as oil or effluent spills represent acute mortality events that were not experienced during the evolutionary history of the species, particularly at the temporal frequency that these events have occurred in the last 15 years. Changing climate may also modify the frequency or severity of some of these events. Following the 2010 Deepwater Horizon oil spill, the Final Programmatic Damage Assessment and Restoration Plan and Final Programmatic Environmental Impact Statement (“PDARP”; section 5.5.7) identified that large numbers of Gulf sturgeon were exposed to oil, and these fish were affected by exposure (Deepwater Horizon Natural Resource Damage Assessment Trustees 2016). During this same time, other events, which may have affected Gulf sturgeon populations, including prolonged extreme drought, were occurring.

The Gulf Sturgeon Recovery Plan (GSRP) suggests the use of population models to assess restoration and management options for Gulf sturgeon, identify future research needs, and forecast time to population recovery (Section 1.3.2; USFWS 1995). Progress has been made on this task in a variety of ways. Capture-recapture models have been used to assess population status for Gulf sturgeon in several individual rivers (5-year status review) and one stock assessment has been conducted to estimate trends in abundance, mortality, and recruitment (Pine and Martell 2009). Other efforts have estimated carrying capacity using stock-reduction methods (Ahrens and Pine 2014) and efforts continue to estimate natural mortality from large-scale telemetry efforts (Rudd et al. 2014).

In 2009 a stock assessment report was presented to the Gulf sturgeon partnership. This report mimicked the NOAA-SEDAR process of compiling and reviewing available data (up to 2005), completing a series of assessment and modeling efforts to determine stock status, and then assembling an independent review panel to assess overall effort and make recommendations in terms of future work (Pine and Martell 2009). The key outcome from this assessment was the identification that long-term demographic data for completing assessment efforts were only available for the Apalachicola and Suwannee rivers. Analyses of these data (mostly from passive tagging programs) revealed divergent trends in future Gulf sturgeon stock status depending on model structure used because of confounding parameter estimates between capture probability and survival in the tagging data. This led to a joint NOAA-USFWS effort to estimate mortality using telemetry tags over a 5-year period (2010-2015). Using data from 2010-2011 Rudd et al. (2014) presented estimates of adult mortality for, and transition probabilities between, each of the key Gulf sturgeon river systems. However, following the Deepwater Horizon oil spill, data from other years were no longer available for use as tag detections of telemetered fish were sequestered during litigation. These data, along with data collected as part of the NRDA response across the Gulf of Mexico, are now available for analyses. At the same time, limited work has been done to integrate this information to develop models to forecast Gulf sturgeon populations in the future and assess basic concerns related to risk of extinction for each population to inform decision making related to restoration actions. These pieces of information are critical management needs, and this dissertation was designed to address these needs.

CHAPTER 2

Range-wide multistate mark-recapture estimates of state-specific survival and movement rates for threatened Gulf Sturgeon

Background

Gulf Sturgeon *Acipenser oxyrinchus desotoi* populations experienced a brief, intense period of commercial exploitation (Ahrens and Pine 2014) and large-scale habitat alterations that blocked access to spawning grounds and changed in-river habitats during the late 19th and early 20th centuries (Clugston et al. 1995; USFWS and GSMFC (U.S. Fish and Wildlife Service and Gulf States Marine Fisheries Commission) 1995; Zehfuss et al. 1999). These factors contributed to the range-wide collapse and federal listing of Gulf Sturgeon as threatened under the Endangered Species Act (ESA) in 1991 (NOAA [National Oceanic Atmospheric Administration] and USFWS 1991). This listing considers Gulf Sturgeon, and species recovery, as a single stock (i.e., a single range-wide management unit). As such, Gulf Sturgeon are not likely to recover at a rate to be delisted by 2023 (Flowers et al. 2020), the year initially proposed by the current Gulf Sturgeon Recovery Plan (USFWS and GSMFC 1995).

Gulf Sturgeon are anadromous. They require riverine spawning habitat, estuarine feeding areas, and movement corridors between these habitats to complete their life cycle (NOAA and USFWS 1991). Because Gulf Sturgeon need multiple habitat types to persist, and these habitats each have unique and shared threats, it is necessary to understand and characterize threats to each life stage and examine how these threats may put population recovery at risk while recognizing new challenges to species recovery, including changing climate.

Gulf Sturgeon mortality events in recent decades have occurred at various spatial scales with impacts spanning single rivers (e.g., Temple-Inland paper mill spill; Louisiana Department of Wildlife and Fisheries 2011) to potentially range-wide harm (*Deepwater Horizon* oil spill; USFWS 2015; Deepwater Horizon Natural Resource Damage Assessment Trustees 2016). Emerging threats to Gulf Sturgeon populations associated with climate change (e.g., changes to flow regimes, critical habitat, intensity and frequency of severe weather events) may also affect range-wide population recovery (Dale et al. 2021). While these threats are identifiable at specific spatial scales, information on how the spatial scale of the threat matches the spatial scale of Gulf Sturgeon habitat use is lacking. It is difficult to rank threats to the species to inform conservation actions without considering how these sturgeon move among, between, and through identified critical habitats.

A greater understanding of Gulf Sturgeon movement may also improve insight into population demography over time and space. Previous research on Gulf Sturgeon geographic distribution and dispersal demonstrated high fidelity to natal rivers (Wooley and Crateau 1985; Rudd et al. 2014). Rudd et al. (2014) found that fidelity rates to river of tagging were higher among rivers with greater genetic relatedness than among individual rivers. Rivers with greater genetic relatedness also appear to represent geographically distinct regions in the Gulf of Mexico (Stabile et al. 1996). If Gulf Sturgeon exhibit high fidelity to individual rivers, other differences in population demographic rates, such as survival, may provide evidence of differences in threats across the same spatial scales.

Distinct Population Segments

Differences in population demography and threats affecting populations of vertebrate and invertebrate species have warranted listing, delisting, and reclassification under the ESA as distinct population segments (DPSs). In 1996, the U.S. Fish and Wildlife Service (USFWS) and the National Oceanic Atmospheric Administration (NOAA) adopted a policy (61 FR 4722) describing what constitutes a DPS of a taxonomic species (USFWS and NOAA 1996). This joint DPS policy identified discreteness and significance as two necessary elements for DPS status under the ESA. The policy states that

a population of a vertebrate species may be considered discrete if it satisfies either of the following conditions: (1) It is markedly separated from other populations of the same taxon as a consequence of physical, physiological, ecological, or behavioral factors (quantitative measures of genetic or morphological discontinuity may provide evidence of this separation) or (2) it is delimited by international governmental boundaries within which differences in control of exploitation, management of habitat, conservation status, or regulatory mechanisms exist that are significant in light of Section 4(a)(1)(D) of the ESA (USFWS and NOAA 1996).

The significance of each potential Gulf Sturgeon DPS to the species as a whole must also be evaluated. Examples of possible indicators of significance provided by the DPS policy include:

1. persistence of the DPS in an unusual or unique ecological setting for the species;
2. evidence that loss of the DPS would result in a significant gap in the species’ range;
3. evidence that the DPS represents the only surviving natural occurrence of a species that may be more abundant elsewhere as an introduced population outside its historical range; or
4. evidence that the DPS differs markedly from other populations of the species in its genetic characteristics.

The main advantage of listing a population as a DPS, as opposed to listing as a single stock, is that it allows for population management and conservation efforts at a finer scale designed to meet a population’s specific needs, which can be especially important for populations with unique characteristics or distinct threats. For a species with a large extant range and potentially different population trajectories and population threats across this range, assessing the potential for including Gulf Sturgeon within a DPS context as part of future conservation efforts could be an important exercise for recovery planning.

Objectives

In this paper, we assess whether Gulf Sturgeon populations exhibit spatially explicit survival and transition rates that support their management under the current ESA listing as a single stock. We (1) estimate regional- and river-specific survival rates of adult Gulf Sturgeon while accounting for incomplete detection; (2) estimate adult Gulf Sturgeon river fidelity and straying rates (i.e., transitions out of the river associated with the initial tagging event); and (3) assess whether these data support alternative management strategies such as single stock or DPS’s. This work was motivated by observed differences in episodic mortality events within Gulf Sturgeon critical habitat in recent decades. These events include the Deepwater Horizon oil spill and river-specific impacts from hurricanes or red tide events.

Materials and Methods

Acoustic Telemetry Tagging and Monitoring

In 2010 a standardized acoustic telemetry monitoring program for adult (≥1350 mm total length [TL]) Gulf Sturgeon was established by NOAA for the seven rivers within the critical habitat designation to reduce uncertainty in estimates of survival and movement (Pine and Martell 2009). To tag fish for this program, Gulf Sturgeon were captured using drifted or anchored gill nets, tagged using surgically implanted acoustic transmitters (Table 2-1; methods following Kahn and Mohead 2010), and virtually recaptured using a network of VEMCO VR2W acoustic receivers (Vemco-Amirix Systems, Halifax, Nova Scotia). The receiver network captured the movement of Gulf Sturgeon to and from one of seven river systems included in the ESA critical habitat designation: the Pearl, Pascagoula, Escambia, Blackwater/Yellow, Choctawhatchee, Apalachicola, and Suwannee rivers (USFWS and NOAA 2003). Due to their proximity, we considered all Blackwater River sampling, including physical captures and virtual recaptures, part of the Yellow River. Receiver location maximized the likelihood of Gulf Sturgeon detection during the in- and out-migration periods by monitoring areas of the river that were geographically restricted.

We assumed acoustic transmitter tag longevity based on manufacturer specifications, and we censored telemetry tags from analysis after this tag expiration date (Table 2-2). When longevity estimates from the manufacturer were unavailable, we assumed that transmitter longevity was five years from the initial deployment.

Survival and Movement Rate Estimation

We utilized a maximum likelihood framework to estimate river- and region-specific survival and movement rates from virtual recaptures (Hightower et al. 2001). We converted virtual recaptures to standard capture history format and multistate capture-recapture models (Williams et al. 2002) in Program MARK (White and Burnham 1999) through RMark (Laake 2013) in Program R (R Core Team 2022). We organized the data based on the genetic relatedness or river of capture. We developed *a priori* models as different hypotheses related to how Gulf Sturgeon survival and transition probabilities may be best represented (similar to Rudd et al. 2014). Models fit to these data assessed temporal and spatial trends in survival and transition probability. Our multistate model estimates three parameters:

1. true survival — the probability of surviving to the next time step
2. capture probability — the probability of virtual recapture during each time interval; and
3. transition rate — the probability that a Gulf Sturgeon moves between rivers or regional areas in a given model.

As an extension of the Cormack–Jolly–Seber (CJS) model, the multistate model shares the same basic assumptions (Williams et al. 2002). The traditional CJS framework estimates apparent survival as the joint probability that the animal is alive (true survival) and remains in the study area (Lebreton et al. 1992). Because our receiver array network included all rivers classified as Gulf Sturgeon critical habitat, we assume all individuals stay in the study area and that by estimating the transition probability, the survival estimates in this framework represent true survival.

Data Characteristics and Capture History Formatting

Data on virtual recaptures were compiled, normalized, and converted to capture-history format from a standardized database. We condensed virtual recaptures from the receiver array into single daily contacts (i.e., daily contacts represented by a single acoustic detection) and removed false detections. We then generated individual capture histories on annual time steps for Gulf Sturgeon with initial capture events in one of the seven rivers in the critical habitat designation between 2010 and 2022. Standardized sampling across the Gulf began in 2010, but fish were acoustically tagged before 2010 in a few rivers. For this study, 73 individuals tagged between 2005 and 2009 informed capture histories within the 2010–2022 timeframe. Some of these Gulf Sturgeon, as well as other Gulf Sturgeon telemetered after 2010, received multiple acoustic tags between 2010 and 2022 because the tag life of a single tag did not cover the time period of interest (Table 2-1). Virtual recaptures of Gulf Sturgeon with multiple acoustic tags were combined into a single capture history for that animal.

Of the 1,031 total Gulf Sturgeon used to inform our analyses, 992 were tagged as adults, and 54 were initially tagged as sub-adults (i.e., <1350-mm total length TL). Individuals initially telemetered as sub-adults did not contribute to capture histories used in the multistate estimation until they were physically recaptured, measured, and determined to meet the adult length threshold.

Other approaches to compiling data to create capture histories exist. For example, only Gulf Sturgeon that were adults at the initial capture size could be included in the analyses, and fish that were observed to grow into the adult size class could have been excluded. Incorporating all fish once attended as adult sizes led to a larger pool of marked fish across the entire time period. Tables 2-3–2-5, respectively, summarize these sub-adult data in numbers of acoustic tags deployed in sub-adults, telemetered sub-adult totals, and the timing of when these telemetered sub-adults recruited into the adult fish pool.

For Gulf Sturgeon contacts with only fork length (FL) measurements, we generated a linear model informed by over 42,500 Gulf Sturgeon contacts to predict TL measurements from FL measurements using the following equation:

|  |  |
| --- | --- |
|  | (3-1) |

For future studies that may wish to convert Gulf Sturgeon TL to FL, the inverse equation is:

|  |  |
| --- | --- |
|  | (3-2) |

Capture histories for individual Gulf Sturgeon detected within a given year were denoted with a letter code to represent the state (i.e., river unit in the multistate model) where the detection occurred (Rudd et al. 2014). If we detected an individual Gulf Sturgeon in multiple states within a given year, it received the state associated with the most detection days for that occasion. Because our capture histories represented annual time steps, we used insight from Gulf Sturgeon life history and assumed that the state occupied was the river a Gulf Sturgeon entered during the spring migration and over-summering period. We also assumed these adult fish left river habitats during fall or early winter (Wooley and Crateau 1985). For group assignment within multistate models, fish received the state associated with the river of initial capture.

Multistate Model Structures

For our primary analyses, we fit three models to estimate survival, capture, and transition probabilities over multiple spatial scales:

1. A null model with a constant rate of survival and detection of Gulf Sturgeon across their range and individual estimates of all river-to-river transition probabilities. This model represents Gulf Sturgeon management under the current ESA listing and incorporates knowledge from previous research that Gulf Sturgeon movement is non-random.
2. The “regional-relatedness” model that collapsed the seven rivers with known spawning sites into four genetically distinct units by allele frequency: (1) the western Gulf: Pearl and Pascagoula rivers; (2) Escambia Bay: Escambia and Yellow rivers; (3) Choctawhatchee River; and (4) the eastern Gulf: Apalachicola and Suwannee rivers (Stabile et al. 1996). We estimated survival, capture, and transition probabilities for all regional units. This model represents the hypothesis that Gulf Sturgeon populations with higher relatedness may exhibit similar population dynamics and allowed us to determine if survival differed among these related regional units.
3. The “river-specific” model estimated each river’s survival, capture, and transition probabilities. This model represents the hypothesis that Gulf Sturgeon population dynamics are so variable among river populations that management should occur at the river level to identify and meet population-specific needs.

We used the delta method to calculate normal-approximated 95% confidence intervals (CI) for all region and river fidelity estimates. We compared models using Akaike Information Criterion corrected for small sample size (AICc; Burnham and Anderson 2004) and AIC weights. Based on the results of model 3 (river-specific analysis) a secondary analysis explored how river-specific patterns in survival varied over time.

Results

Tagging Summary

The number of acoustic transmitters deployed annually ranged from 0 to 63 for a total of 58–283 tags in each river between 2010 and 2022 (Table 2-1). The annual number of tagged Gulf Sturgeon was lower for rivers in the western Gulf of Mexico (Pearl and Pascagoula) because of lower sturgeon catch rates (Table 2-1). Because tag life was less than the study length for some tags (Table 2-2), the number of active acoustic tags (i.e., deployed acoustic tags that are potentially available for virtual recapture) was 2–148, with the mean number of active tags each year between 16.6 and 117.2 (Table 2-6). The percentage of acoustic tags virtually recaptured in their river of deployment ranged between 47.5% (Pearl River) and 95.1% (Suwannee River; Table 2-7).

Survival Estimation

Because of biological realities and management interest in all three models, we present results of all models first and then evaluate statistical fit of models from an AIC framework. Our null model estimated a constant, range-wide survival rate of 0.88 and capture probability of 0.94 (Table 2-8).

Regional-Relatedness Grouping

Estimated capture probabilities for genetic groups ranged from 0.78 to >0.98, with standard errors ≤0.02 (Table 2-8). Estimated survival rate by genetic grouping was lowest for the western region (0.82) and highest for the Choctawhatchee River (0.93), with most areas exceeding 0.84 (Table 2-8). Standard errors for all survival rate estimates were low (~0.02) for all genetic groups (Table 2-8). The Choctawhatchee River survival rate 95% CI did not overlap with any of the other confidence intervals suggesting the highest survival for adult Gulf Sturgeon from the Choctawhatchee River compared to other regions (Table 2-8).

River-Specific Grouping

River-specific capture probability estimates exceeded 0.91 for all rivers except the Yellow River (Table 2-8), and standard error estimates were low for all rivers (about 0.01) except the Pearl River (Table 2-8). Riverine survival rates ranged from 0.85 to 0.93 for all rivers but the Pearl River, where survival was estimated to be 0.67, and the 95% confidence intervals for the Pearl River did not overlap with the other rivers (Table 2-8). Overall, survival estimates were precise (smaller than point estimate), with relatively low 95% CI ranges (Table 2-8).

Annual Estimates for River-Specific Grouping

Yearly survival rates ranged from 0.28 for the Pearl River in 2015 to 1.0, which was observed in most systems for at least one annual estimate (Figure 2-1). Precision around these estimates was fairly consistent within rivers, but varied between rivers (Figure 2-1). Pearl River annual survival estimates were the lowest and the most uncertain, exceeding 0.74 only once between 2010 and 2017 (Figure 2-1). In contrast, Choctawhatchee River survival rates never fell below 0.8 and remained consistently high between 0.88 and 1.0 in most years (Figure 2-1). Survival estimates of 1.0 can be an indicator of poor model parameter fit. However, careful assessment of these estimates suggests that the survival of telemetered animals in these years did approach 1 because of year-over-year observation of the same group of marked animals.

Fidelity and Exchange Between Rivers and Regions

River-specific fidelity to the river of tagging ranged from 0.66 (Escambia River) to 0.99 (Suwannee River; Table 2-9). Fidelity was 0.88 or higher for five of the seven rivers (Pearl, Pascagoula, Choctawhatchee, Apalachicola, and Suwannee), and the precision around these estimates was high (Table 2-9). Grouping the Escambia and Yellow rivers together estimated the regional fidelity of Escambia Bay to be similarly high at 0.87 (Table 2-10). The exchange between rivers ranged from movement from the river of tagging to one other river (observed in the Pearl River; Table 2-9) to movement to four other rivers (observed in the Pascagoula, Escambia, Yellow, Choctawhatchee, and Apalachicola rivers). Overall, the exchange rate was generally low (1–4%) for most rivers. The highest estimated exchange rate between regions was estimated as 0.13 for Gulf Sturgeon transitioning from the Escambia Bay rivers to the Choctawhatchee River (Table 2-10). No other regional transition exceeded 0.03, the reciprocal rate of Gulf Sturgeon moving from the Choctawhatchee River to the Escambia Bay rivers. There was a <2% annual probability of Gulf Sturgeon leaving the eastern or western regions (Table 2-10).

All observed transitions between rivers are summarized in Figure 2-2. For example, we observed 938 instances of Choctawhatchee River-tagged fish being detected in the Choctawhatchee River in two consecutive years. If a particular fish was detected in four consecutive years in the Choctawhatchee River, that would represent three observed transitions back to the Choctawhatchee River. In contrast, only one Choctawhatchee River-sourced fish was observed in the Choctawhatchee River one year and in the Pascagoula River the next year (Figure 2-2).

AIC Model Comparison

Akaike Information Criteria support was highest for the river-specific model (lowest AIC value, highest AIC weights), a model with unique survival parameters for each river and year (Table 2-11). The river-specific model received an AIC weight of 1, and no other models had ΔAICc <250 (Table 2-11). Despite the AIC penalty associated with additional model parameters, the model with the most parameters (57 parameters; river-specific model) received the most support, and the model with the fewest parameters (20; regional-relatedness model) received the least support (Table 2-11).

Discussion

We found adult Gulf Sturgeon survival rates in the western U.S. Gulf of Mexico were lower than in the eastern Gulf. Annual river-specific survival estimates (Figure 1) indicate lower survival rates in the Pearl River (a western Gulf of Mexico River) than in the other six rivers where survival was estimated. This suggests that the lower survival in the Pearl River is likely the driver for lower survival for the pooled western Gulf of Mexico estimate overall. These persistently lower annual survival rates in the western Gulf of Mexico (first identified by Morrow et al. 1998) likely reflect higher annual adult mortality in this part of the Gulf Sturgeon range from unknown sources. It is unclear whether the low survival estimate of 0.28 in 2015 represents higher episodic mortality effects in this region or the sensitivity of these estimates given the low sample size in this river. Because our model accounts for both incomplete detection and emigration (accounted for through the transition [Psi] parameter) these estimates of survival are likely not influenced by these two common sources of bias. We observed one of the lowest Apalachicola River survival rates of the past five years in 2018 (0.67), the same year Hurricane Michael made landfall as a category 5 hurricane. Our findings are concordant with the empirical mortality calculation of undetected fish by Dula et al. (2022) in their evaluation of the impacts of Hurricane Michael on Gulf Sturgeon in the Apalachicola River.

We documented high adult Gulf Sturgeon fidelity rates (generally >90% to the river of tagging), which supports the hypothesis that the differential survival we observed could be a result of some distinct groups of Gulf Sturgeon having lower survival rates. This distinctiveness can be defined either based on genetic relatedness or by the river of initial capture. If fidelity was low and widespread exchange among populations was occurring, the adult survival rates would likely be similar across population segments because all fish had the same probability of survival. Our estimates of differential survival highlight the need to consider whether these differences in survival are a function of different risks the population segments are experiencing or reflect natural variation in the life history of Gulf Sturgeon across its range.

We also conducted this analysis on a more conservative dataset which only included fish acoustically telemetered as adults. This dataset, which did not allow telemetered sub-adults to ‘grow into inclusion,’ displayed the same trends and similar estimates across all models. The primary difference between these datasets was a marginal increase in precision for rivers that had more tagged sub-adults that were included in later capture history occasions as adults.

Comparison With Previous Study

The multistate model estimates in our study expand on earlier efforts to estimate these parameters before the Deepwater Horizon oil spill and subsequent data embargo related to the associated litigation. Both this study and Rudd et al. (2014) estimated capture probability within 5% of one another for the Pascagoula and Suwannee rivers. However, our study estimated much higher capture probabilities in the Pearl, Escambia, Yellow, Choctawhatchee, and Apalachicola rivers (generally 18–30% higher) likely because of an increase in the number of receivers deployed in each river in the last 10 years. Despite our 22% higher estimate, both studies estimated the Yellow River (which includes data from the Blackwater River) to have a lower detection probability than all other rivers. Estimated capture probabilities from virtual recaptures were high in both studies compared to passive tagging estimates for Gulf Sturgeon (generally ~0.10; Zehfuss et al. 1999; Sulak et al. 2014).

Our survival estimates fell within the 95% CI’s associated with the survival estimates in Rudd et al. (2014) for all rivers except the Pascagoula River; we estimated survival to be 88%, and they estimated it as 51%. In contrast, none of the Rudd et al. (2014) river-specific survival estimates fell within our 95% CI’s for survival (unless their estimate of Blackwater survival is considered a Yellow River estimate). The Rudd et al. (2014) survival estimates were higher than those in this study for all rivers except the Pascagoula River.

Despite differences in the numbers of total tags deployed, tag longevity, and active transmitters available for detection each year in each river, we generated precise survival estimates (SE 0.01–0.02) for rivers that averaged at least five acoustic tag deployments per year. Rudd et al. (2014) simulated the bias and precision of the same multistate capture-recapture model under a variety of scenarios involving different numbers of tags deployed each year and the number of years the tags were monitored. Results from this work suggest that both the precision and accuracy of mean parameter estimates increased with a higher sample size of transmitters deployed and longer monitoring time. When assessing the tradeoffs between increased sample sizes or monitoring program length, their simulation results suggested that longer monitoring periods led to more precise parameter estimates (Rudd et al. 2014).

Recommendations for Monitoring Programs

In the first Gulf Sturgeon stock assessment, Pine and Martell (2009) recommended minimizing variation in sampling programs to limit the introduction of additional uncertainty in understanding the status and trends in Gulf Sturgeon. Changes in Gulf Sturgeon monitoring programs have occurred since 2009. For example, the introduction of side-scan sonar as a tool to both enumerate sturgeon (Flowers and Hightower 2013) and to assess a sampling site for the presence of Gulf Sturgeon before sampling with nets has led to large increases in Gulf Sturgeon catch per unit of effort. These changes in catchability (*q*) may introduce bias in model parameter estimates for models conditioned on catch or, in tagging-based assessments such as this analysis, non-random sampling of Gulf Sturgeon could introduce bias because the Gulf Sturgeon population sampling is no longer random. This increase in *q* has also led to increased variations in the number of fish captured and tagged with telemetry tags each year in some rivers leading to increases in numbers of telemetered fish each year. For example, 34 acoustic tags were implanted in the Yellow River in Gulf Sturgeon over the last ten years of the study. However, because most Gulf Sturgeon were tagged early in the study with long-lived 5– to 10–year tags and Gulf Sturgeon survival rates are high relative to shorter-lived species, the pool of marked Gulf Sturgeon in the Yellow River persisted throughout the study resulting in the third-most precise riverine survival estimate (Table 8). In contrast, annual acoustic transmitter deployments were consistent in the Pascagoula River where smaller numbers of shorter-lived tags were deployed each year. Despite the Pascagoula River representing the second lowest average number of active transmitters available for detection, the difference in precision for the Pascagoula River and Yellow River survival estimates was <0.01 and the survival estimates in both systems cover the same number of years, even with the use of different types of tags. This suggests that the staggered entry design (Pollock et al. 1989) employed in the Pascagoula River or the alternate approach of deploying a larger number of long-lived tags in the first two years in the Yellow River both resulted in maintaining adequate sample sizes throughout the study.

Critically, we found that extended monitoring time, represented by the total years of array deployment and the pool of active tags in each river, had a greater effect on precision than variability in sampling effort and capture rates. Rudd (2014) found similar results through simulation. We recommend that if survival or transition probabilities are the primary parameters of interest, then field efforts should focus on supporting the continued monitoring of existing telemetered animals through array maintenance and modern, living-data workflow management (Yenni et al. 2019) instead of continued tagging to increase sample sizes of telemetered Gulf Sturgeon. Given continued improvements in tagging technology related to battery life and tag longevity, pools of tagged Gulf Sturgeon can be more easily maintained than with the tags deployed 2010–2012 used in Rudd et al. (2014). To increase efficiency, minimize cost, and maximize inference gained from survival analyses informed by acoustic telemetered fish, monitoring programs should: 1) define population parameters of interest, 2) determine acceptable levels of precision for parameter estimates, and 3) determine the minimum numbers of telemetered fish to achieve this precision via simulation.

A Framework for DPS Application for Gulf Sturgeon: Atlantic Sturgeon

The congeneric Atlantic Sturgeon *Acipenser oxyrinchus oxyrinchus* has been managed through the use of five DPSs since 2012. The effort to transition Atlantic Sturgeon to DPS from single stock management began in 2007 following a federal agency-led species status review recommended dividing the species into five DPSs (Atlantic Sturgeon Status Review Team [ASSRT] 2007). In 2009, the Natural Resources Defense Council filed a petition requesting an Atlantic Sturgeon listing status change which NMFS determined included substantial information indicating that the petitioned actions may be warranted (NOAA 2012a, 2012b). Based on the ecological separation of populations during spawning from tagging data and genetic analyses, NMFS determined that these populations were discrete (NOAA 2012a, 2012b). NMFS also concluded that these DPSs were significant since the spawning habitat of each population grouping was found in separate and distinct ecoregions, as identified by The Nature Conservancy (The Nature Conservancy 2004; NOAA 2012a, 2012b).

In the case of Atlantic Sturgeon, the federal government did not independently identify DPSs. A petition instigated their review of a set of DPSs identified by a status review team which resulted in NMFS recognizing these DPSs as sufficiently discrete and significant for designation under the ESA. The studies that informed this status review played a critical role in DPS identification by supplying the analyses and the motivation for ESA reclassification. Research dedicated to listed species should intentionally be structured to inform ESA decisions so that when a preponderance of evidence is accumulated, federal agencies can do their part to make a ruling. When the limiting factor in this process is the federal agency tasked with making these listing decisions, petitions to initiate these reviews can be effective.

Could DPS be Considered for Gulf Sturgeon?

Rudd et al. (2014) and this work confirm high Gulf Sturgeon fidelity using between two (Rudd et al. 2014) and twelve (this study) years of data. The high fidelity rates and the precision around the fidelity estimates in this study provide the most substantial evidence yet of spatial separation of Gulf Sturgeon populations. Estimated fidelity between Rudd et al. (2014) and this study were very similar (within 1–5%) for most rivers. However, our estimate of Apalachicola River fidelity was ~19% higher than Rudd et al. (2014) for reasons that are unknown. Our efforts also further built on Rudd et al. (2014) by generating fidelity and transition estimates for the Pearl, Pascagoula, and Suwannee rivers, which were not possible earlier because of data deficiencies. Future reference to degrees of population mixing among all rivers, except between the Escambia Bay rivers, may better be referred to as straying rates considering the low probabilities associated with such transitions.

In addition to the observed population separation demonstrated by the fidelity estimated from tagging,Stabile et al. (1996) reported extensive genetic structuring of Gulf Sturgeon populations and low gene flow estimates indicating strong maternal homing fidelity on a river-specific basis. The authors identified 4–5 genetically distinct Gulf Sturgeon stocks depending on the technique used, and recommended nearly three decades ago that Gulf Sturgeon throughout the Gulf of Mexico should not be managed as a ‘single panmictic unit’ (Stabile et al. 1996).

Despite high genetic relatedness within the western and eastern regions, respectively (Stabile et al. 1996), these rivers may be more geographically isolated than shared genetics suggest. Of the 150 telemetered Gulf Sturgeon in the western region, only eleven (7.3%) were observed in the two rivers (Pearl and Pascagoula) which we used to define western Gulf of Mexico rivers. Similarly, of the 259 telemetered Gulf Sturgeon in the eastern Gulf (Apalachicola and Suwannee rivers), only seven individuals (2.7%) were observed in both eastern rivers. In contrast, 55 (18%) of the 306 telemetered Gulf Sturgeon in the Choctawhatchee River were observed in the Escambia Bay rivers and 41 (14.4%) of the 284 telemetered Gulf Sturgeon in the Escambia Bay rivers were observed in the Choctawhatchee River. However, these central Gulf rivers are reported to have different population structure based on published genetic information (Stabile et al. 1996). Genetic relatedness is a valuable measurement of population discreteness, but this result suggests that genetics studies alone may not fully capture the observed movements and transitions between these systems. However, from a population perspective, the genetic information would clarify reproductive separation between these systems, whereas the movement and exchange between systems observed by telemetry may represent a more diverse range of movements. This includes possible behaviors related to expanding foraging arenas, range recolonization or expansion, or perhaps alternative spawning behaviors such as fall spawning which may not have been captured in Stabile et al. (1996). The integration of observed movement patterns and genetic information could provide new insight into Gulf Sturgeon life history and behavior.

Waldman et al. (2002) reviewed gene flow rates as a measure of reproductive interchange among populations of several sturgeon species. This study was used by NOAA to justify a high degree of reproductive isolation for Atlantic Sturgeon as a measure of DPS discreteness and is uniquely useful as it independently evaluates and compares the discrete characteristics of multiple sturgeon species. The authors observed strong homing fidelity in Gulf Sturgeon populations through their strong stock structuring and low gene flow rates for Gulf Sturgeon, which they suggested were comparatively lower than Atlantic Sturgeon. Waldman et al. (2002) also observed that Gulf Sturgeon had lower haplotypic diversity than southern populations of Atlantic Sturgeon of similar latitude, concluding that Gulf Sturgeon have extremely low evolutionary effective population sizes (*N*fe). Bowen and Avise (1990) supported this finding by comparing mtDNA variation among Atlantic and Gulf Sturgeon and estimating *N*fe for female Gulf Sturgeon populations to be 50 compared to 8,500 for Atlantic Sturgeon. Given the genetic distinctness measured by Stabile et al. (1996) and Waldman et al. (2002) and the evidence for low *N*fe, there appears to be sufficient evidence that should any of the river populations become extinct, this would represent a significant loss to Gulf Sturgeon genetic diversity.

Management Recommendations

Based on high resolution original analyses of Gulf Sturgeon fidelity and survival patterns range-wide, plus published genetic information, my work suggests a minimum of five DPSs could be considered for Gulf Sturgeon: the western Gulf (Pearl and Pascagoula), the Escambia Bay rivers (Escambia and Yellow), the Choctawhatchee River, the Apalachicola River, and the Suwannee River. Grouping rivers in this framework would allow resource managers to address recurring uncertainties observed regionally, including lower survival in the western Gulf of Mexico systems. This approach can also allow better assessment of potential benefits from management actions related to fish passage issues by evaluating populations that may share some spatial redundancy in required habitats (such as spawning sites in multiple shared rivers such as the populations in the Escambia Bay rivers) versus management actions which may benefit a single discrete population with less exchange (fish passage within the Apalachicola River). These are the critical types of decisions and problem framing which must be undertaken to help define “what is recovery” in the context of Gulf Sturgeon in the present, and future, northern Gulf of Mexico river basins.

Table 2-1. The annual number of acoustic transmitters surgically implanted in adult (≥1350 mm TL) Gulf Sturgeon by river and year of deployment. Tag models and performance characteristics vary.

|  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
|  | Year | | | | | | | | | | | | | |
| River | before  2010 | 2010 | 2011 | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 | 2020 | 2021 | 2022 |
| Pearl | 0 | 23 | 1 | 0 | 6 | 0 | 1 | 1 | 2 | 0 | 0 | 0 | 9 | 15 |
| Pascagoula | 0 | 2 | 3 | 3 | 5 | 5 | 8 | 9 | 2 | 7 | 13 | 8 | 4 | 23 |
| Escambia | 9 | 27 | 16 | 6 | 0 | 0 | 17 | 5 | 5 | 6 | 5 | 0 | 0 | 1 |
| Yellow | 27 | 63 | 41 | 21 | 0 | 0 | 17 | 10 | 9 | 6 | 1 | 0 | 0 | 0 |
| Choctawhatchee | 35 | 47 | 33 | 29 | 30 | 0 | 0 | 11 | 5 | 60 | 60 | 6 | 2 | 0 |
| Apalachicola | 1 | 21 | 15 | 18 | 0 | 0 | 8 | 13 | 14 | 16 | 3 | 9 | 0 | 0 |
| Suwannee | 2 | 20 | 29 | 23 | 0 | 0 | 0 | 16 | 11 | 15 | 5 | 23 | 0 | 0 |

Table 2-2. Acoustictag longevity implanted in adult (≥1350 mm TL) Gulf Sturgeon summarized by river of deployment.

|  |  |  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
|  | Acoustic tag battery life | | | | | | | | | |
| River | 1-year | 2-year | 3-year | 4-year | 5-year | 6-year | 7-year | 8-year | 9-year | 10-year |
| Pearl | 2 | 13 | 0 | 0 | 12 | 17 | 0 | 0 | 5 | 15 |
| Pascagoula | 0 | 6 | 4 | 0 | 32 | 15 | 0 | 0 | 32 | 50 |
| Escambia | 3 | 1 | 0 | 6 | 35 | 30 | 0 | 0 | 0 | 23 |
| Yellow | 4 | 0 | 1 | 22 | 46 | 97 | 0 | 0 | 0 | 26 |
| Choctawhatchee | 23 | 26 | 20 | 13 | 61 | 60 | 0 | 0 | 0 | 117 |
| Apalachicola | 8 | 0 | 1 | 0 | 0 | 54 | 0 | 0 | 9 | 48 |
| Suwannee | 0 | 8 | 0 | 0 | 0 | 60 | 0 | 0 | 0 | 76 |

Table 2-3.  The annual number of acoustic transmitters surgically implanted in sub-adult Gulf Sturgeon (<1350 mm TL) by river and year of deployment. Individual fish implanted with an acoustic tag in multiple years contributed to more than one annual total. Each sub-adult Gulf Sturgeon later contributed to analyses once it was recaptured and determined to be an adult (i.e., ≥1350 mm TL).

|  |  |  |  |  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
|  | Year | | | | | | | | | | | |
| River | 2010 | 2011 | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 | 2020 | 2021 |
| Pearl | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 |
| Pascagoula | 6 | 3 | 1 | 1 | 4 | 2 | 8 | 5 | 6 | 1 | 1 | 1 |
| Escambia | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Yellow | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Choctawhatchee | 1 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Apalachicola | 0 | 1 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 |
| Suwannee | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

Table 2-4. Annual totals of telemetered sub-adult Gulf Sturgeon (i.e., <1350 mm TL) by river and initial year of tagging. Each sub-adult fish only contributed to the total associated with its initial capture. Each sub-adult Gulf Sturgeon contributed to analyses once it was recaptured and determined to be an adult (i.e., ≥1350 mm TL).

|  |  |  |  |  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
|  | Year | | | | | | | | | | | |
| River | 2010 | 2011 | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 | 2020 | 2021 |
| Pearl | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 |
| Pascagoula | 5 | 1 | 1 | 0 | 3 | 2 | 7 | 5 | 4 | 1 | 0 | 1 |
| Escambia | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Yellow | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Choctawhatchee | 1 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Apalachicola | 0 | 1 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Suwannee | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

Table 2-5. Annual totals representing the first year that Gulf Sturgeon tagged as sub-adults (<1350 mm TL) were recruited into our adult dataset (≥1350 mm TL). Each sub-adult was initially telemetered during a year prior to the totals summarized here. These totals do not reflect all years these fish contributed to analyses, but rather the first year they recruited into our adult dataset.

|  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
|  | Year | | | | | | | | | | | |  |
| River | 2010 | 2011 | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 | 2020 | 2021 | 2022 |
| Pearl | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 1 | 0 |
| Pascagoula | 0 | 0 | 0 | 1 | 0 | 1 | 2 | 1 | 3 | 3 | 8 | 4 | 7 |
| Escambia | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Yellow | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Choctawhatchee | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 |
| Apalachicola | 0 | 0 | 0 | 0 | 2 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Suwannee | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

Table 2-6. Annual totals of Gulf Sturgeon with active acoustic transmitters available for detection by river.

|  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
|  | Year | | | | | | | | | | | | | |
| River | 2010 | 2011 | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 | | 2020 | 2021 | 2022 |
| Pearl | 23 | 24 | 23 | 16 | 16 | 17 | 18 | 11 | 10 | | 10 | 4 | 14 | 30 |
| Pascagoula | 2 | 5 | 8 | 14 | 19 | 28 | 37 | 37 | 45 | | 60 | 70 | 70 | 97 |
| Escambia | 26 | 49 | 54 | 54 | 54 | 65 | 55 | 45 | 39 | | 40 | 40 | 23 | 24 |
| Yellow | 62 | 123 | 146 | 148 | 147 | 143 | 127 | 96 | 63 | | 44 | 44 | 27 | 27 |
| Choctawhatchee | 67 | 109 | 118 | 123 | 114 | 113 | 101 | 85 | 125 | | 136 | 142 | 144 | 144 |
| Apalachicola | 20 | 34 | 51 | 51 | 53 | 60 | 73 | 57 | 59 | | 44 | 53 | 57 | 55 |
| Suwannee | 22 | 45 | 68 | 68 | 68 | 66 | 82 | 73 | 68 | | 53 | 76 | 76 | 70 |

Table 2-7. The percentage of acoustically tagged adult (≥1350 mm TL) Gulf Sturgeon virtually recaptured in each river. Columns indicate the river associated with the initial tagging event and rows indicate the river of detection. Each Gulf Sturgeon may contribute to percentages in multiple rivers of detection.

|  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- |
|  | Natal River | | | | | | |
| River of Detection | Pearl | Pascagoula | Escambia | Yellow | Choctawhatchee | Apalachicola | Suwannee |
| Pearl | 47.5% | 4.2% | × | × | × | × | × |
| Pascagoula | 4.9% | 68.6% | 2.2% | 1.0% | 0.3% | × | × |
| Escambia | × | 1.7% | 70.7% | 27.6% | 9.8% | 0.9% | × |
| Yellow | × | 0.8% | 37.0% | 69.8% | 10.5% | 0.9% | 0.7% |
| Choctawhatchee | × | 0.8% | 21.7% | 10.9% | 93.8% | 7.7% | × |
| Apalachicola | × | × | 2.2% | 1.6% | 2.0% | 81.2% | 2.1% |
| Suwannee | × | × | × | × | × | 3.4% | 95.1% |

Table 2-8. Area-specific survival and capture probabilities for adult Gulf Sturgeon (≥1350 mm TL) using the null, regional-relatedness, and river-specific models, all with upper (UCL) and lower (LCL) 95% confidence limits.

|  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- |
|  | Survival probability | | |  | Capture probability | | |
| River or area | Estimate | LCL | UCL |  | Estimate | LCL | UCL |
| Null Model |  |  |  |  |  |  |  |
| Range-wide | 0.88 | 0.87 | 0.89 |  | 0.94 | 0.93 | 0.95 |
|  |  |  |  |  |  |  |  |
| Regional-relatedness method |  |  |  |  |  |  |  |
| West | 0.82 | 0.78 | 0.86 |  | 0.98 | 0.94 | 0.99 |
| Escambia Bay | 0.85 | 0.82 | 0.88 |  | 0.78 | 0.73 | 0.81 |
| Choctawhatchee | 0.93 | 0.91 | 0.94 |  | 0.98 | 0.97 | 0.99 |
| East | 0.88 | 0.85 | 0.90 |  | 0.98 | 0.97 | 0.99 |
|  |  |  |  |  |  |  |  |
| River-specific method |  |  |  |  |  |  |  |
| Pearl | 0.67 | 0.57 | 0.76 |  | 0.92 | 0.77 | 0.97 |
| Pascagoula | 0.88 | 0.84 | 0.91 |  | 0.98 | 0.95 | 0.99 |
| Escambia | 0.86 | 0.82 | 0.89 |  | 0.97 | 0.94 | 0.98 |
| Yellow | 0.86 | 0.83 | 0.89 |  | 0.77 | 0.73 | 0.81 |
| Choctawhatchee | 0.93 | 0.91 | 0.94 |  | 0.98 | 0.96 | 0.98 |
| Apalachicola | 0.85 | 0.81 | 0.88 |  | 0.97 | 0.94 | 0.98 |
| Suwannee | 0.90 | 0.87 | 0.92 |  | 0.99 | 0.98 | 1.00 |

Table 2-9. Transition probabilities of adult Gulf Sturgeon (≥1350 mm TL) movement between rivers with 95% confidence intervals (CI) in parentheses. Columns indicate the river occupied in a given sampling occasion, and rows denote possible destinations in the following sampling occasion. Estimates along the diagonal represent river fidelity rates. An “×” represents an unobserved transition during the study.

|  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- |
|  | Pearl | Pascagoula | Escambia | Yellow | Choctawhatchee | Apalachicola | Suwannee |
| Pearl | 0.88  (0.81, 0.96) | 0.02  (0.01, 0.05) | × | × | × | × | × |
| Pascagoula | 0.12  (0.06, 0.21) | 0.95  (0.93, 0.98) | 0.01  (0.00, 0.04) | 0.00  (0.00, 0.02) | 0.00  (0.00, 0.01) | × | × |
| Escambia | × | 0.02  (0.01, 0.04) | 0.66  (0.61, 0.72) | 0.13  (0.11, 0.16) | 0.03  (0.03, 0.05) | 0.01  (0.00, 0.02) | × |
| Yellow | × | 0.00  (0.00, 0.03) | 0.21  (0.17, 0.26) | 0.77  (0.74, 0.81) | 0.03  (0.02, 0.05) | 0.01  (0.00, 0.03) | 0.00  (0.00, 0.01) |
| Choctawhatchee | × | 0.00  (0.00, 0.03) | 0.10  (0.07, 0.14) | 0.09  (0.07, 0.11) | 0.92  (0.90, 0.94) | 0.03  (0.02, 0.06) | × |
| Apalachicola | × | × | 0.01  (0.00, 0.04) | 0.00  (0.00, 0.01) | 0.01  (0.01, 0.02) | 0.94  (0.91, 0.96) | 0.01  (0.00, 0.02) |
| Suwannee | × | × | × | × | × | 0.01  (0.01, 0.03) | 0.99  (0.98, 1.00) |

Table 2-10. Transition probabilities of adult Gulf Sturgeon (≥1350 mm TL) movement between related regions with 95% confidence intervals (CI) in parentheses. Columns indicate the region occupied in a given sampling occasion, and rows denote possible destinations in the following sampling occasion. Estimates along the diagonal represent fidelity estimates. An “×” represents an unobserved transition during the study.

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
|  | West | Escambia Bay | Choctawhatchee | East |
| West | 0.99  (0.98, 1.00) | 0.00  (0.00, 0.01) | 0.00  (0.00, 0.00) | × |
| Escambia Bay | 0.01  (0.00, 0.02) | 0.87  (0.84, 0.89) | 0.03  (0.03, 0.04) | 0.00  (0.00, 0.01) |
| Choctawhatchee | 0.00  (0.00, 0.02) | 0.13  (0.10, 0.16) | 0.96  (0.95, 0.97) | 0.01  (0.01, 0.02) |
| East | × | 0.00  (0.00, 0.01) | 0.01  (0.00, 0.01) | 0.98  (0.97, 0.99) |

Table 2-11. Model selection table describing Akaike Information Criterion corrected for small sample size (AICc), including the difference in each model’s AICc score and the top model’s AICc score (ΔAICc), the number of parameters (K), the negative log-likelihood, and the weight of each model for three competing models characterizing adult Gulf Sturgeon survival (S), capture probability (p), and transition probability (Psi).

|  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- |
| Model | Model parameterization | ΔAICc | K | nll | AIC weight |
| River-specific | S(river) p(river) Psi(river to river) | 0.00 | 57 | 6132.46 | 1.00 |
| Null | S(constant) p(constant) Psi(river to river) | 254.78 | 45 | 6411.93 | <0.01 |
| Regional-Relatedness | S(region) p(region) Psi(region to region) | 137388.57 | 20 | 143596.62 | <0.01 |

A diagram of different types of lines

Description automatically generated with medium confidence

Figure 2-1. Annual area-specific survival probabilities for adult Gulf Sturgeon (≥1350 mm TL) using a model which assumed constant detection probabilities by river. Results are for the "river-specific" grouping of adult Gulf Sturgeon described in text.

A picture containing text, diagram

Description automatically generated

Figure 2-2. Transition frequencies between observable states represented by rivers of detection. Totals summarize each observed transition between rivers and are grouped by the river occupied in the first year. Abbreviated rivers represent transition destinations: PE–Pearl, PR–Pascagoula, ER–Escambia, YR–Yellow, CR–Choctawhatchee, AR–Apalachicola, SR–Suwannee.

Chapter 3

Unifying three decades of disparate tagging data to estimate Gulf Sturgeon survival using the Barker mark-recapture model

Background

Gulf of Mexico Sturgeon *Acipenser oxyrinchus desotoi* “Gulf Sturgeon” were listed under the US Endangered Species Act in 1991 (56 FR 49653; National Oceanic Atmospheric Administration [NOAA] and U.S. Fish and Wildlife Service [USFWS] 1991). Current management units for Gulf Sturgeon include seven river systems and adjacent estuarine and marine habitats across the northern Gulf of Mexico from the Pearl River in Louisiana to the Suwannee River in Florida. The Gulf Sturgeon Recovery Plan (GSRP) identifies population models as a tool to assess restoration and management options for Gulf Sturgeon, identify future research needs, and forecast time to population recovery (Section 1.3.2; USFWS and GSMFC 1995). Cooperative Gulf Sturgeon research among agency, academic, and NGO partners has been ongoing throughout the species’ range since the 1970s. Most research efforts have focused on basic ecology and life-history questions (e.g., growth, food habits, movement; Huff 1975; summary provided in Sulak et al. 2016) or short-term (<5 year) assessments of population status and trends (e.g., Morrow et al. 1998; Sulak and Clugston 1999; Zehfuss et al. 1999). Gulf Sturgeon are long-lived (assumed maximum age of ≥50 years; Pine and Martell 2009), and these discrete population assessment studies were all conducted over much shorter periods of time than the maximum age for the species. Because Gulf Sturgeon have low catch rates, low capture probabilities, and may demonstrate skip-spawning behavior, uncertainty persists as to how well short-term snapshots of population status may reflect overall trends in Gulf Sturgeon demography at the generation time scale.

Gulf Sturgeon population assessment studies from the 1980s until the early 2000s used a variety of tag types to identify previously captured fish to inform study objectives. Gulf Sturgeon research, like most fisheries programs of the 1970s and 1980s, widely implemented external tags that were more prone to shedding and misidentification (e.g., anchor tags), especially in long-lived species (Nielsen 1992; Pine et al. 2012). In the mid-1980s, passive integrated transponder (PIT) represented a major advance in tagging technology by providing an individually identifiable tag with a long tag life (Pine et al. 2001). By the 1990s, advances in radio telemetry and sonic tag technology were incorporated into multiple Gulf Sturgeon studies (Wooley and Crateau 1985; Odenkirk 1989; Carr et al. 1996), providing breakthrough advancement in our understanding of Gulf Sturgeon ecology and life history. Expanded use of telemetry tags as part of Gulf Sturgeon research allowed the presence and absence of Gulf Sturgeon to be inferred passively (virtually) through detections of the telemetry tag, which did not necessitate a physical capture of the animal. Due to the higher price of telemetry tags, PIT tags were the most common (but not exclusive) tag type used for Gulf Sturgeon (Chapter 3).

The 2009 Gulf Sturgeon Stock Assessment (Pine and Martell 2009) synthesized available PIT tagging data and fit these data to a variety of age-structured models in an attempt to assess the status and trends in Gulf Sturgeon populations over a multi-decade period. This effort had limited success (Pine and Martell 2009), and a recommendation from the stock assessment review panel was to directly estimate natural mortality (M) in a range-wide telemetry study of adult Gulf Sturgeon using virtual recaptures of fish via an array of autonomous receivers. This effort was designed to take advantage of Gulf Sturgeon migratory behavior and detect adults as they seasonally transition between ocean, estuarine, and riverine habitats by deploying the receiver array in the mouths of the seven rivers considered in the Gulf Sturgeon recovery plan. These detections were then used to estimate the survival of the telemetered fish as virtual recaptures (Hightower et al. 2001; Rudd et al. 2014; Chapter 3).

The analysis of these telemetry recaptures to directly estimate survival and fidelity to the river of tagging is the focus of Chapter 3. However, while the post-2009 epoch of Gulf Sturgeon research efforts focused on telemetry recaptures, substantial information to inform demography is also contained in the PIT tag capture histories of these animals. Critically, the lifespan of Gulf Sturgeon exceeds the life of all known non-natural tags (e.g., close kinship mark-recapture). Even the use of the longest-lived standard telemetry tag in Gulf Sturgeon (approximate tag life of 10 years) limits our ability to capture information not only on basic life-history such as longevity but more importantly from a recovery perspective, inform trends in survival over a time period which begins to approach the generational time of Gulf Sturgeon. This is important for several reasons. Flowers et al. (2020) identified that the closure of the Gulf Sturgeon fishery was the most likely management action to reduce mortality and promote population recovery among the different management actions they evaluated. However, Flowers et al. (2020) identified that because of the long life of Gulf Sturgeon and the erosion of the population age structure by the fishery, recovery of the age structure would occur on decadal time scales regardless of management actions. Therefore, the integration of these two tagging data streams, which represented contrasting prevalence, longevity, and resolution, was of paramount importance to measuring Gulf Sturgeon survival and evaluating species recovery over multiple decades.

The current Gulf Sturgeon Recovery Plan (GSRP) outlines a variety of criteria before populations can be considered recovered and delisting proposed (USFWS and Gulf States Marine Fisheries Commission [GSMFC] 1995). A short-term goal of the GSRP is to ensure that wild stocks are not currently declining, while the primary long-term goal of the GSRP is to establish self-sustaining populations that could allow delisting of the species, with a secondary goal of population recovery to a point at which directed fishing could be sustained (USFWS and GSMFC 1995). The primary factors that have been identified as potentially contributing to declines in Gulf Sturgeon populations historically throughout their native range include overfishing, loss of spawning habitat, alteration of riverine habitat, or a combination of these and other factors (Clugston et al. 1995; Zehfuss et al. 1999). Since about 2003, known Gulf Sturgeon mortality events associated with paper plant effluent, major hurricanes (i.e., Katrina, Rita, Michael), recurring red tide events (USFWS and NMFS [National Marine Fisheries Service] 2022), and the 2010 Deepwater Horizon oil spill (DWH; Deepwater Horizon Natural Resource Damage Assessment Trustees 2016) have occurred. Natural events such as hurricanes represent episodic mortality that has occurred throughout the evolutionary history of the species (Dula et al. 2022). Anthropogenic disturbances such as oil or effluent spills represent acute mortality events that were not experienced during the evolutionary history of the species, particularly at the temporal frequency that these events have occurred in the last 15 years. A changing climate may also modify the frequency or severity of some of these events with ecosystem effects such as drought (Pine et al. 2015; Leitman et al. 2016) or red tide events (USFWS and NMFS 2022). Following the 2010 DWH oil spill, the Final Programmatic Damage Assessment and Restoration Plan and Final Programmatic Environmental Impact Statement (“PDARP”; section 5.5.7) identified that large numbers of Gulf Sturgeon were exposed to oil, and these fish were affected by exposure. How this event, and others since the last Gulf Sturgeon stock assessment was completed in 2009 (Pine and Martell 2009), have impacted Gulf Sturgeon recovery trends is unknown.

As part of the DWH oil spill damage assessment process substantial field sampling efforts have occurred since 2010. Sampling has focused primarily on juvenile life stages (*Deepwater Horizon* Open Ocean Trustee Implementation Group 2019; <https://tinyurl.com/3dx97zvn>), continued tagging of adult Gulf Sturgeon with long-lived telemetry tags (3–10 year tag life), and maintenance of an autonomous array of receivers to monitor specific locations for the presence/absence of telemetered sturgeon. We take advantage of more than two decades of Gulf Sturgeon marking efforts prior to the DWH oil spill and a decade of sampling efforts post-DWH to update our collective understanding of Gulf Sturgeon demography by estimating trends in survival for this long-lived species. Understanding trends in survival can inform recovery planning in multiple ways, including bench-mark comparisons of field-based survival estimates in time periods with and without events that could be managed (e.g., oil spills) and those that cannot (e.g., hurricanes). Trends in survival can also be used to infer trends in population status. For example, declining trends in adult Gulf Sturgeon survival over time could cause concern for declines in population growth rate because any increase in the rate of death would have to be offset by an increase in recruitment otherwise the population would decline.

In this study we unify all available Gulf Sturgeon tagging data within a common analytical framework to assess collective trends in Gulf Sturgeon survival. Specifically, we 1) analyze Gulf Sturgeon as a single stock to estimate the temporal dynamics of range-wide survival; 2) consider the effect of individual episodic mortality events on range-wide survival by measuring the impact of the DWH oil spill – a known episodic mortality event (Deepwater Horizon Natural Resource Damage Assessment Trustees 2016) which may have altered survival patterns in Gulf Sturgeon potentially changing recovery trajectory; 3) evaluate spatial trends in survival among river populations; 4) compare and contrast spatiotemporal survival trends among river populations and what they might mean for Gulf Sturgeon management; and 5) establish context around these survival estimates to determine a range of possible values to be considered by managers.

Materials and Methods

Data Overview

Field collection efforts for Gulf Sturgeon used in this chapter are described in Chapters 1 and 2. Briefly, Gulf Sturgeon were captured by agency and academic cooperators over three decades as part of various field studies primarily focused on assessing Gulf Sturgeon life history. As part of these studies, Gulf Sturgeon were implanted with a variety of active and passive tags. Since the 1990’s, PIT tags have been the most common tag type used to mark Gulf Sturgeon. Beginning in about 2010, changes in technology and the development of autonomous receivers allowed the use of acoustic tags and passive receivers to create networks of listening stations to detect the presence of Gulf Sturgeon tagged with telemetry tags. Because Gulf Sturgeon life spans likely exceed the functional life of any one tag type, many Gulf Sturgeon have been tagged and “known” as different animals over the last three decades depending on the type of tag they are marked with or the type of tag read or detected by researchers. Beginning in 2018 efforts to consolidate tagging information in a common database were started. In this framework every fish collected was assigned a unique fish identification number and then all subsequent real (handled physically by biologists) and virtual (detected on a receiver array) recaptures were recorded, along with biological information (for physical recaptures) such as size.

From the capture information from these unique fish identification numbers, we generated two sets of annual capture histories: one informed exclusively by PIT tag capture and recaptures and one informed exclusively by acoustic telemetry virtual recaptures. Capture histories for PIT tag data spanned from 1990 to 2021, while telemetry data availability began in 2010. We added leading zeroes to the telemetry capture histories to ensure the two types had the same number of annual occasions. Our data included adult Gulf Sturgeon (i.e., individuals with a minimum recorded total length of at least 1350 mm) implanted with PIT tags or acoustic telemetry tags at some point (i.e., not necessarily during the same occasion; Table 1).

Capture History Format

The formatting structure of traditional Barker models with “LDLDLD” capture histories is as follows: a “1” in the “L” portion signifies a live capture, a “1” in the “D” portion represents a recovery (i.e., dead individual), and a “2” in the “D” portion represents a resight event (i.e., the individual was reported alive during the interval; Barker 1997). To use the “D” portion for telemetry contacts, and thereby convert a traditional Barker model into a joint live-recapture/live resight/tag-recovery parameterization of this model (Barker 1997), we represented all telemetry contacts with a “2” and combined the PIT tag and telemetry capture histories such that subsequent “L” and “D” occasions corresponded to PIT tag and telemetry contacts for the same sampling interval. Treating acoustic telemetry virtual recaptures as resight events allowed us to estimate the probabilities of detection associated with each tag type.

Barker Model Parameterization

We inferred years without sampling (sampling effort is not consistently recorded in the field by Gulf Sturgeon cooperators) for each river from the total number of annual contacts for each tag type (i.e., PIT tags and acoustic transmitters). If there were no contacts in a given year for a specific river and tag type, it was fixed to zero and the corresponding detection probability for that river was therefore not estimated for that year. Using the Barker model parameterization summarized in Table 2, we estimated the following parameters: survival probability (*)*; recapture probability (i.e., PIT tag detection probability) (); and the probability that an individual survived and was resighted alive between sampling occasions (i.e., acoustic telemetry detection probability) ().Several Barker model parameters were not estimated given their inapplicability to our data. Since our capture histories were only informed by live-recaptures (i.e., fish caught in gill nets and virtually), the probability of dead recovery () was fixed to zero. To establish that acoustic tags were able to be detected (i.e., “resighted”) in each sampling occasion, we fixed the probability that a fish died but was resighted alive between capture events before dying,, to 1. Because our sampling area encompassed all seven rivers designated as Gulf Sturgeon critical habitat (U.S. Office of the Federal Register 2003), we assumed all individuals were at risk of capture in each sampling occasion and therefore fixed the probability of temporary site fidelity () to 1 and the probability of temporary emigration () to zero.

Barker Models

We fit a variety of models to comprehensively characterize spatiotemporal dynamics for Gulf Sturgeon over the last three decades. Secondarily, these models were designed to allow us to better understand the effects of sampling program changes on survival estimation by allowing to vary with . Because of complicated changes in sampling programs over time (e.g., there is no standardized monitoring effort, only sampling programs for specific, often short-term life history studies), estimates had to be averaged over blocks of years. These blocks were informed by captures and recaptures from individual sampling years, but the parameter estimate was generated as an average across those years. For example, a five-year group could include sampling from years 2000–2004, but only a single parameter estimate for that grouping of years would be estimated, and this estimate would represent the average over the years 2000–2004. Model parameterizations are summarized in Table 3. A general, uncertain understanding of spatiotemporal variations in sampling effort motivated us to allow to vary with our temporal parameterization of survival in these models (Table 3). We did not have reason to believe there were meaningful changes in over time because a core group of autonomous receivers was consistently maintained. Therefore, we estimated a constant rate of in each model (Table 3). We used the Akaike information criterion corrected for small sample sizes (AICc) model selection procedure (Burnham and Anderson 2004) to assess the statistical fit of each individual model relative to other models. However, we were primarily interested in trends over time in parameters, not the performance of one model compared to another.

Results

Data Summary

Analyses characterized tagging data for 7,188 Gulf Sturgeon tagged between 1990 and 2021 that were available for recapture between 1991 and 2022. The total number of Gulf Sturgeon that only received PIT tags (6,222) was much higher than the number of Gulf Sturgeon that received both PIT and acoustic telemetry tags (966) due to the lag in implementation associated with this newer technology and its higher associated cost. The first 44 Gulf Sturgeon in this dataset were telemetered between 2005 and 2007, and full telemetry implementation across rivers was not achieved until 2010. River-specific totals of Gulf Sturgeon that exclusively received PIT tags ranged between 141 and 2,681 (Table 2). In contrast, between 30 and 351 Gulf Sturgeon were implanted with both PIT and acoustic tags in each river (Table 2). The number of tagged Gulf Sturgeon was lower for rivers in the western Gulf of Mexico (Pearl and Pascagoula) because of lower sturgeon catch rates.

Gulf-Wide Capture Probability

Model 1 estimated a constant, Gulf-wide of 0.04 and of 0.80 (Table 4). Time-varying Gulf-wide estimates of ranged from 0.02 to 0.08 and generally decreased over time with the lowest estimates representing more recent sampling years (Tables 5–11). All estimates of capture probability had high precision (SE ≤0.01), including annual estimates in Model 5 (Tables 4–11).

Gulf-Wide Survival

Despite the lack of high-resolution acoustic telemetry data before 2010, we estimated for five-year groupings between 1995 and 2009 with relatively high precision with an SE range of 0.01–0.02 (Models 2, 3, and 5; Tables 5, 6, and 8). This high precision was also observed in our 1990–2009 estimate in Model 4 (Table 7).

Across models, Gulf-wide Gulf Sturgeon was consistent over time with estimates generally ranging between 0.84 and 0.97 (Figures 2–5; Tables 4–8). Model 1 estimated a constant survival rate across all years and rivers of 0.88 and a 95% CI of 0.87–0.89 (Table 4). This null model 95% CI overlapped with most time-varying CI’s from Model 2 and Model 3 (Figures 2–3; Tables 5–6). This trend of relatively constant Gulf-wide over time was most evident in Model 4 (Figure 4; Table 7) as before the DWH oil spill (1990–2009 = 0.89; 95% CI 0.88 to 0.90) was nearly identical to the post-spill estimate (2010–2021 = 0.88; 95% CI 0.88 to 0.89).

Of the ten annual estimates between 2012 and 2021 in Model 5 (i.e., estimates representing individual years instead of block of years), seven had overlapping 95% CI’s with Model 1, which assumed constant across all years (Figure 5; Table 8). Model 5 annual estimates for 2014, 2016, and 2017 were the only estimates after 2010 that were lower than 0.90 (Table 8; Figure 5). Data deficiencies and large changes in capture probability likely contributed to estimation failure in 2010 and 2011 (Table 8; Figure 5).

Gulf-wide estimates appeared to be more dynamic when examined at a finer temporal resolution. For grouped estimates between 2010 and 2021, Models 2 and 3 estimated as 0.89–0.90 and 0.84–0.97, respectively (Tables 5–6; Figures 2–3). When 2010–2021 estimates were modeled annually, ranged between 0.72 and 0.96. This trend of greater variability in estimates associated with shorter time periods was also observed in Model 3, where a relatively large increase in survival coincided with a reduction of years represented by the time-varying estimate from five years (as for the 1990–2019 data) to two years, 2020–2021 (Figure 3; Table 6). When this final time-varying estimate represented average survival over seven years in Model 2, Gulf-wide survival appeared constant, with estimates ranging from 0.89 to 0.90 between 2000 and 2021.

River-Specific Capture Probability

Estimates of *p* were low range-wide (generally 0.02–0.07) with a minor decreasing trend from western-Gulf rivers to eastern-Gulf rivers (Tables 9–11, Figures 6, 9, and 10). There was also an inverse trend of increasing precision over time (Tables 10–11; Figures 9–10). Precision for most *p* estimates was high (SE ≤0.01). The Pascagoula, Escambia, and Yellow rivers each had a time-varying estimate of between 0.12 and 0.13 with SE ≤0.02 (Tables 10–11; Figures 9–10). All estimates ≥0.13 were much less precise (SE ≥0.06; Tables 10–11; Figures 9–10).

River-Specific Survival

Each river displayed increasingly precise survival estimates over time (Tables 10–11; Figures 7–8). Model 6 exhibited relatively high (≥0.87), precise (SE = 0.01) survival rates for six of the seven rivers (Table 9; Figure 6). In contrast, the Pearl River had a lower of 0.76 and relatively wide 95% CI of 0.70–0.81 (Table 9; Figure 6). Similar to the Gulf-wide models, most Gulf Sturgeon river populations exhibited constant survival rates over time (Tables 10–11; Figures 7–8). Only three time-varying estimates were lower than 0.81, and there was high uncertainty around these estimates (SE 0.08–0.18; Tables 10–11). As we observed in the final estimates of Models 2 and 3, the seven-year estimates in Model 7 (Table 10; Figure 7) did not show the range-wide increase in survival that we observed in the two-year estimates in Model 8 (Table 11; Figure 8).

AIC Model Comparison

Akaike Information Criteria support was highest for Model 8 (lowest AICc value, highest AIC weights), a model with river-specific estimates of and representing five-year periods between 1990 and 2019 and a single two-year period from 2020 to 2021 (Table 12). Model 8 received an AIC weight of 1, and no other models had ΔAICc <99 (Table 12). Despite the AIC penalty associated with additional model parameters, the model with the most parameters (K = 149; Model 8) received the most support, and the model with the fewest parameters (K = 5; Model 1) received the least support (Table 12). The model that received the second most support, Model 7, was the only other model that estimated river-specific and over time.

Discussion

Gulf-Wide Status and Trends

The multidecadal Gulf Sturgeon capture-recapture data used in this chapter are the longest-term data assimilation available to assess trends in survival at a time scale that approaches the maximum age of the species. A major advantage of these long-term data over short-term assessments is that decadal time scales allow us to begin to assess survival and mortality patterns both as chronic mortality (i.e., mortality from natural processes including senescence) and episodic mortality from specific events which may or may not be anthropogenically influenced (i.e., DWH oil spill, Hurricane Michael). Chronic mortality would represent all mortality inputs that act on Gulf Sturgeon every year. Despite the consistent temporal nature of chronic mortality, we could see changes in the magnitude of these processes, which may be identified as a changing baseline survival rate over time.

In contrast, episodic mortality would characterize discrete events that may be represented in model results as declines, but then return, to the long-term average. These distinctions can be further separated into thinking about events that could be regulated through management, and those that can’t. For example, if higher mortality is estimated in years and in areas where dredging operations are ongoing, and dredging operations identify Gulf Sturgeon presence through mitigation efforts (as currently done for beach renourishment projects) then management regulations could be designed to focus dredging efforts in a time and place when and where Gulf Sturgeon are absent. Other episodic events, such as an oil spill, could cause both episodic mortality associated with the event or the response, as well as longer-term chronic mortality associated with ecosystem contaminants or other factors.

In this analysis, when Gulf Sturgeon are considered a single stock, and models fit to the available data are grouped in 5-year time steps, the estimated annual 𝑆 was about 0.90 across all time periods (Figures 2 and 3). The absence of increasing or decreasing trends over time suggests survival is relatively stable in five-year time steps, and when compared to a life history-derived survival reference point (0.905; Ahrens and Pine 2014), there do not appear to be chronic trends in Gulf Sturgeon survival that suggest recovery could be limited by adult survival. However, this does not suggest that episodic mortality events are not occurring, and the role of these episodic patterns in determining the overall status of Gulf Sturgeon populations is likely important.

On smaller, yearly time steps, the estimated Gulf Sturgeon annual survival in 2014, 2016, and 2017 was less than the 0.9 estimated for five-year time steps, with annual estimates ranging from 0.72 to 0.77. In each case, the years immediately before and after the years with lower survival were similar to the longer-term rate, suggesting temporally isolated fluctuations in survival likely due to episodic mortality events. Such fluctuations in vital rates are often attributed to model sensitivity in these rates due to small sample sizes (Chapter 3). However, a careful assessment of model performance and parameter uncertainty suggests these years with lower estimated survival are likely accurately representing temporal variation in the parameter of interest as the upper 95% CI for survival in 2014, 2016, and 2017 are still below the long-term estimated 𝑆 of 0.9. These results document both the magnitude of episodic mortality that can be experienced by adult Gulf Sturgeon and also the resilience of these populations to return rapidly to the baseline mortality following episodic events.

Most episodic mortality events for Gulf Sturgeon are not well documented because the mortalities are observed after the event happens, and baseline survival rates from before the event may not be available. For example, following the Temple-Inland papermill spill (Louisiana Department of Wildlife and Fisheries 2011), dead Gulf Sturgeon were recovered. Other than observing a dead sturgeon, it was impossible to determine whether these mortalities, which were likely due to low dissolved oxygen, had significant impacts on the Gulf Sturgeon population overall. In contrast, following the DWH oil spill, a large response effort by federal management agencies documented the extent of the oil spill, and the potential impacts to Gulf Sturgeon, and created dedicated funding to assess the recovery of Gulf Sturgeon to these impacts. For example, USFWS (2015) estimated between 1,100 and 3,600 Gulf Sturgeon were potentially exposed to the DWH oil in the nearshore areas of the northern Gulf of Mexico (USFWS 2015; Deepwater Horizon Natural Resource Damage Assessment Trustees 2016), and this exposure caused injury to the species (USFWS 2015; Deepwater Horizon Natural Resource Damage Assessment Trustees 2016).

This documented impact motivated our efforts to assess whether changes in adult Gulf Sturgeon survival were apparent in the long-term Gulf Sturgeon tagging data. Specifically, we were motivated to assess whether there was evidence for either an episodic mortality signal related to the DWH oil spill, or chronic trends in survival, as observed in the post-DWH injury to Bottlenose Dolphins (*Tursiops truncates*), which included reproductive failure (Lane et al. 2015, Kellar et al. 2017) and long-lasting mortality consequences (Litz et al. 2014; Schwacke et al. 2017). Our 𝑆 estimates for the time periods before and after the oil spill are effectively the same (Figure 4). This result does not necessarily mean that the DWH oil spill did not have an effect on adult Gulf Sturgeon survival, but that its effects likely impacted Gulf Sturgeon over a shorter period of time. This conclusion is supported by the sharp, isolated decreases in survival that were only evident when survival was modeled annually. Unfortunately, high-resolution telemetry data were not well-established at the time of this event, as indicated by the model failure for the 2010 and 2011 𝑆 estimates. Therefore, if quantifying the effects of specific episodic events is of management interest, high resolution monitoring programs should be implemented before these events and designed to track their potential subsequent effects.

River Population Status and Trends

Understanding trends in Gulf Sturgeon survival by river is important to informing where within the range of Gulf Sturgeon lower survival could be occurring. Building on the documented lower Gulf-wide estimates of survival in 2014, 2016, and 2017, we examined annual survival estimates over five-year time periods in each of the seven river basins. From a river-specific survival perspective, lower survival was estimated in the Yellow and Choctawhatchee rivers in 2014, and lower survival in the Pearl, Escambia, Yellow, Apalachicola and Suwannee rivers in 2016 and 2017. Gulf Sturgeon demonstrate high fidelity to the river of tagging (Chapter 3) which would suggest that any geographically isolated threats to Gulf Sturgeon population health would also be isolated to the population(s) within that geographic area. While possible causes for these geographic events are described in the most recent five-year status review for the species (USFWS and NMFS 2022), and Hurricane Michael is documented to have impacted Apalachicola Gulf Sturgeon in 2018 (Dula et al. 2022), it is not possible to identify specific causes of episodic mortality events in each river and year.

The lowest estimated annual adult Gulf Sturgeon survival rates for individual river systems were found in the Pearl and Pascagoula rivers in the 1990s. Reported survival rates in these rivers have been lower than other river systems previously (Morrow et al. 1998; Rudd et al. 2014; Chapter 3). While the lower rates for the Pascagoula River are from years when only PIT tag data are available, these results over the full range of available data suggest that adult Gulf Sturgeon survival has increased since the early 1990s, which is encouraging from a population recovery perspective. The lower baseline survival observed for the Pearl River across multiple model parameterizations may indicate a stronger influence of chronic threats facing this area or this may reflect greater susceptibility to episodic mortality events, like hurricanes, due to its location in the Gulf. These spatial trends in 𝑆 provide another line of inference that the threats facing Gulf Sturgeon are likely not the same across the range of the species.

Placing Gulf Sturgeon Survival Estimates in Context – Comparisons to Life History-Based Estimates

In this study we estimated true survival, 𝑆, an analog of total mortality (𝑍), from thirty years of tagging data. While these tagging data provide information on *S* from the last thirty years, to place the estimates in greater biological context, we used standard fisheries approaches to estimate total mortality (Z) as an analog for natural mortality (𝑀 *=* 1 ̶ 𝑆). Since much of the fisheries literature is reported in terms of mortality, a key element in most population models used to assess stock status (Walters and Martell 2005; Flowers et al. 2020), we will shift to interpreting these results as mortality estimates. Total mortality is commonly calculated as the sum of natural mortality (𝑀) and fishing mortality (*F*). Because 𝑀 is almost always unobserved and fisheries management actions are most directed at *F*, quantifying population losses through 𝑀 estimation is a necessary first step to better understand sources of mortality that can be managed (e.g., bycatch, ship strikes). Therefore, an accurate estimate of *M* becomes a cornerstone of nearly all population models and in the planning and interpretation of the subsequent management actions that the population models were designed to inform. For Gulf Sturgeon, directed fishing has been closed for more than two decades. Therefore, *F* = 0 and 𝑍 ≈ 𝑀, which will allow us to use 1 ̶ 𝑆 to evaluate mortality estimators for 𝑍 or 𝑀 interchangeably.

Estimates of mortality are often difficult to measure directly, so information-limited fisheries often assume constant values or use practical estimators informed by life history parameter estimates such as the Brody growth coefficient (𝐾) and maximum age (𝑡𝑚𝑎𝑥). We used the available tagging data and information on time at liberty to calculate 𝑡𝑚𝑎𝑥  for use in estimating *M* to compare to estimates derived from capture-recapture models. Alternative estimates of 𝑡𝑚𝑎𝑥 from aging hard parts are not available for adult Gulf Sturgeon.

Two fish within the Standardized Gulf Sturgeon Capture-Recapture Database were at liberty within our study (i.e., the maximum number of years between capture and recapture events) for 26 years, one year greater than the maximum life expectancy of Suwannee River Gulf Sturgeon by Sulak and Clugston (1999). The first (hereafter Fish 1) was captured in 1992, measuring 1403 FL/1600 TL in the Suwannee River, and was recaptured in the Suwannee River in 2018. The second (hereafter Fish 2) was captured in 1993, measuring 1765 TL (1586 FL was interpolated using the equation in Chapter 3) in the Apalachicola River and was recaptured in the Apalachicola River in 2019. Using the river-specific direct aging von Bertalanffy parameter estimates in Flowers et al. (2010) for asymptotic length (𝐿∞: Suwannee = 1697 mm; Apalachicola = 2168 mm), 𝐾 (Suwannee = 0.21; Apalachicola = 0.13), and the theoretical age at zero-length (𝑡o: Suwannee = -0.63; Apalachicola = -0.83), we determined the ages of Fish 1 as 33.7 years old in 2018 and Fish 2 as 35.3 in 2019. Only Fish 2 was released alive. These empirical observations of fish at liberty for 26 years appear to be limited by the total number of years these data represent (32 years, 1990–2022). In their critique of empirical length-at-age curves for Gulf Sturgeon older than age 8–10, Sulak et al. (2016) suggest mature males tend to slow down in increasing in length such that a 1500–1600 mm TL male could be 10, 15, 20, or 30 years old. If this is true and Fishes 1 or 2 were males, the range of possible ages would be 36–56. Therefore, a maximum age for Gulf Sturgeon of 50 years, as estimated from life history characteristics by Ahrens and Pine (2014), is supported by these data as well as other Atlantic Sturgeon *A. o. oxyrinchus* studies (Smith 1985).

With this empirical range of 𝑡𝑚𝑎𝑥 values (35–50 years) and two possible 𝐾 values (0.13 and 0.21; Flowers et al. 2010), we can now use these values to evaluate a few mortality estimators given the direct vital rate estimates in this study. The widely used Jensen (1996) estimator of 𝑀 = 1.5𝐾 provides a range of 𝑀 estimates between 0.2 and 0.35. As Kenchington (2014) pointed out in a review of natural mortality estimators for information-limited fisheries, this estimator performs best when applied to archetypal exploited teleosts, and therefore would perform poorly in our application to Gulf Sturgeon. Then et al. (2015) evaluated the predictive performance of empirical estimators of 𝑀 on over 200 fish species, determined that 𝑡𝑚𝑎𝑥-based estimators performed better than methods that used 𝑡𝑚𝑎𝑥 and 𝐾 (e.g., Alverson-Carney method; Alverson and Carney 1975) or just 𝐾 (e.g., Beverton-Holt method; Beverton and Holt 1959). They recommend using the updated Hoenignls estimator of when a 𝑡𝑚𝑎𝑥 estimate is available (Then et al. 2015). Using the Hoenignls estimator, a 𝑡𝑚𝑎𝑥 of 50 provides a 𝑍 of 0.14, which is close to the Model 1 constant 𝑀 estimate of 0.12 and is well within the range of time-varying, Gulf-wide direct estimates of 𝑀 in this study ranging from 0.03 to 0.28. A 𝑡𝑚𝑎𝑥 of 35 provides an 𝑀 of 0.19, which appears high but reasonable given our data. If a pre-exploitation 𝑡𝑚𝑎𝑥 can be observed, Kenchington (2014) recommends the estimator 4.3/𝑡𝑚𝑎𝑥, which estimates 𝑍 = 0.09 for a 𝑡𝑚𝑎𝑥 of 50 and 𝑍 = 0.12 for a 𝑡𝑚𝑎𝑥 of 35. This estimator appears to suitably approximate the baseline Gulf-wide mortality rates in this study.

For a fish population recovering from fishery removals and age-structure erosion, life history-based mortality estimates, such as those from longevity or catch-curve type methods, would likely be positively biased due to absent older age classes. But as a reference point to compare tagging-based mortality estimates these 𝑡𝑚𝑎𝑥-based estimators are a suitable starting point for comparison and planning. These indirect/empirical estimators also provide additional lines of inference to give us greater confidence that our direct estimates of Gulf Sturgeon survival sufficiently represent the true population(s) we modeled. Additional work is needed to determine how the range of direct and indirect mortality estimates presented here influence Gulf Sturgeon population viability.

Management Recommendations

Unlike estimates of abundance which require additional context for interpretation, such as carrying capacity, mortality estimates can be interpreted, and sometimes estimated, with respect to life history expectations for the species. These models display stability in the stock range-wide that is conducive to the recovery of the species. The evolutionary history of this species suggests that Gulf Sturgeon are resilient to the threats they have faced thus far, and these results support this conclusion. The population viability consequences of the lower survival in the western Gulf are uncertain and are an area of concern. As new and changing anthropogenic threats arise, monitoring programs designed to capture these phenomena must be in place if we are to measure their effects on Gulf Sturgeon population health.

When coupled with the high-resolution telemetry data within the Barker model framework, the PIT tag data generated new retrospective insight into multi-decade mortality trends for the range-wide stock and the individual river populations. The Barker model is not the only framework suited to integrating these multiple streams of tagging data. However, this well-established, readily accessible method provided a faster means of analyzing these data than a conventional stock assessment. Continued opportunistic adaptations of long-standing tools may allow protected species research to inform managers at a faster rate commensurate with the needs of a species in recovery.

Table 3-1. Summary of the number of Gulf Sturgeon in each river marked with PIT tags or both PIT and acoustic telemetry tags between 1990 and 2022.

|  |  |  |
| --- | --- | --- |
| River | PIT tags | PIT and telemetry tags |
| Pearl | 141 | 30 |
| Pascagoula | 178 | 83 |
| Escambia | 417 | 102 |
| Yellow | 805 | 167 |
| Choctawhatchee | 1,200 | 351 |
| Apalachicola | 800 | 95 |
| Suwannee | 2,681 | 138 |

Table 3-2. Barker model parameter definitions within the context of this study and whether or not each parameter was estimated or fixed to a specific value.

|  |  |  |
| --- | --- | --- |
| Parameter | Study Definition | Value |
| *S* | Gulf Sturgeon survival probability | Estimated |
| *p* | PIT tag capture probability | Estimated |
| *R* | Acoustic tag capture probability | Estimated |
| *r* | The probability of a dead recovery | Fixed to 0 |
| *R’* | The probability that a fish that died was first resighted alive | Fixed to 0 |
| *F* | Temporary site fidelity | Fixed to 1 |
| *F’* | Temporary emigration | Fixed to 0 |

Table 3-3. Barker model summaries including model-specific hypotheses and spatiotemporal parameterization of survival (*S*), PIT tag capture probability (*p*), and rate of acoustic tag detection (*R*).

|  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- |
| Model No. | *S* and *p* | | *R* | | Model Hypotheses |
| Spatial groups | Temporal groups | Spatial groups | Temporal groups |
| 1 | Gulf-wide | Constant | Gulf-wide | Constant | Survival did not differ across space or time. |
| 2 | Gulf-wide | 1990–1994, 1995–1999, 2000–2004, 2005–2009, 2010–2014, 2015–2021 | Gulf-wide | Constant | Survival changed over time and is best represented in 5–7 year periods. |
| 3 | Gulf-wide | 1990–1994, 1995–1999, 2000–2004, 2005–2009, 2010–2014, 2015–2019, 2020–2021 | Gulf-wide | Constant | Survival changed over time and is best represented in 2–5 year periods. |
| 4 | Gulf-wide | 1990–2009, 2010–2021 | Gulf-wide | Constant | Gulf-wide survival declined between 1990–2009 and 2010–2021 due to a large-scale episodic mortality event, *Deepwater Horizon*. |
| 5 | Gulf-wide | 1990–1994, 1995–1999, 2000–2004, 2005–2009, 2010, 2011, 2012, 2013, 2014, 2015, 2016, 2017, 2018, 2019, 2020, 2021 | Gulf-wide | Constant | Survival changed over time and should be modeled in time groups based on the resolution of the data. |
| 6 | Rivers | Constant | Rivers | Constant | Survival differs spatially among river populations. |
| 7 | Rivers | 1990–1994, 1995–1999, 2000–2004, 2005–2009, 2010–2014, 2015–2021 | Rivers | Constant | Survival differs over time and space and should be modeled in 5–7 year intervals. |
| 8 | Rivers | 1990–1994, 1995–1999, 2000–2004, 2005–2009, 2010–2014, 2015–2019, 2020–2021 | Rivers | Constant | Survival differs over time and space and should be modeled in 2–5 year intervals. |

Table 3-4. Gulf-wideGulf Sturgeon survival estimates, standard errors (SE), and associated 95% confidence intervals (lower confidence limits [LCL] and upper confidence limits [UCL]) from a Barker model (Model 1) estimating constant survival (), constant PIT tag capture probability (*p)*, and a constant rate of acoustic tag detection (). Fixed parameters are defined in Table 3-1.

|  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- |
| Parameter | Years | Estimate | SE | LCL | UCL |
| *S* | Constant | 0.88 | <0.01 | 0.87 | 0.89 |
| *p* | Constant | 0.04 | <0.01 | 0.04 | 0.05 |
| *R* | Constant | 0.80 | <0.01 | 0.78 | 0.81 |
| *r* | Fixed | 0.00 | – | – | – |
| *R’* | Fixed | 0.00 | – | – | – |
| *F* | Fixed | 1.00 | – | – | – |
| *F’* | Fixed | 0.00 | – | – | – |

|  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- |
| Parameter | Years | Estimate | SE | LCL | UCL |
| *S* | 1990–1994 | 0.97 | 0.03 | 0.85 | 0.99 |
| *S* | 1995–1999 | 0.80 | 0.02 | 0.76 | 0.84 |
| *S* | 2000–2004 | 0.90 | 0.01 | 0.87 | 0.93 |
| *S* | 2005–2009 | 0.90 | 0.01 | 0.87 | 0.93 |
| *S* | 2010–2014 | 0.89 | 0.01 | 0.88 | 0.91 |
| *S* | 2015–2021 | 0.89 | 0.01 | 0.88 | 0.90 |
| *p* | 1990–1994 | 0.05 | 0.01 | 0.04 | 0.07 |
| *p* | 1995–1999 | 0.07 | 0.01 | 0.06 | 0.09 |
| *p* | 2000–2004 | 0.06 | <0.01 | 0.05 | 0.07 |
| *p* | 2005–2009 | 0.04 | <0.01 | 0.04 | 0.05 |
| *p* | 2010–2014 | 0.04 | <0.01 | 0.04 | 0.05 |
| *p* | 2015–2021 | 0.02 | <0.01 | 0.02 | 0.03 |
| *R* | Constant | 0.79 | 0.01 | 0.78 | 0.81 |
| *r* | Fixed | 0.00 | – | – | – |
| *R’* | Fixed | 0.00 | – | – | – |
| *F* | Fixed | 1.00 | – | – | – |
| *F’* | Fixed | 0.00 | – | – | – |

Table 3-5. Five-year survival estimates, standard errors (SE), and associated 95% confidence intervals (lower confidence limits [LCL] and upper confidence limits [UCL]) for Gulf Sturgeon 1990–2014 and a single seven-year survival estimate representing 2015–2021 from a Barker model (Model 2) estimating river-specific survival (), river-specific PIT tag detection probability (*p)*, and a constant rate of acoustic tag detection (). Fixed parameters are defined in Table 3-2.

Table 3-6. Five-year survival estimates, standard errors (SE), and associated 95% confidence intervals (lower confidence limits [LCL] and upper confidence limits [UCL]) for Gulf Sturgeon 1990–2014 and a single two-year time period representing 2020–2021 from a Barker model (Model 3) estimating river-specific survival (), river-specific PIT tag detection probability (), and a constant rate of acoustic tag detection (). Fixed parameters are defined in Table 3-2.

|  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- |
| Parameter | Years | Estimate | SE | LCL | UCL |
|  | 1990–1994 | 0.97 | 0.03 | 0.86 | 0.99 |
|  | 1995–1999 | 0.80 | 0.02 | 0.76 | 0.84 |
|  | 2000–2004 | 0.90 | 0.01 | 0.87 | 0.93 |
|  | 2005–2009 | 0.90 | 0.01 | 0.87 | 0.93 |
|  | 2010–2014 | 0.90 | 0.01 | 0.89 | 0.91 |
|  | 2015–2019 | 0.84 | 0.01 | 0.82 | 0.86 |
|  | 2020–2021 | 0.97 | 0.01 | 0.96 | 0.98 |
| *p* | 1990–1994 | 0.05 | 0.01 | 0.04 | 0.07 |
| *p* | 1995–1999 | 0.07 | 0.01 | 0.06 | 0.09 |
| *p* | 2000–2004 | 0.06 | <0.01 | 0.05 | 0.07 |
| *p* | 2005–2009 | 0.04 | <0.01 | 0.04 | 0.05 |
| *p* | 2010–2014 | 0.04 | <0.01 | 0.04 | 0.05 |
| *p* | 2015–2019 | 0.02 | <0.01 | 0.02 | 0.03 |
| *p* | 2020–2021 | 0.02 | <0.01 | 0.02 | 0.03 |
|  | Constant | 0.79 | 0.01 | 0.78 | 0.80 |
|  | Fixed | 0.00 | – | – | – |
|  | Fixed | 0.00 | – | – | – |
|  | Fixed | 1.00 | – | – | – |
|  | Fixed | 0.00 | – | – | – |

Table 3-7. Gulf-wide Gulf Sturgeon survival estimates, standard errors (SE), and associated 95% confidence intervals (lower confidence limits [LCL] and upper confidence limits [UCL]) representing 1990–2009 and 2010–2021 from a Barker model (Model 4) estimating time-varying survival (), time-varying PIT tag detection probability (*p)*, and a constant rate of acoustic tag detection (). Fixed parameters are defined in Table 3-2.

|  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- |
| Parameter | Years | Estimate | SE | LCL | UCL |
| *S* | 1990–2009 | 0.89 | 0.01 | 0.88 | 0.90 |
| *S* | 2010–2021 | 0.88 | <0.01 | 0.88 | 0.89 |
| *p* | 1990–2009 | 0.05 | <0.01 | 0.05 | 0.06 |
| *p* | 2010–2021 | 0.03 | <0.01 | 0.03 | 0.03 |
| *R* | Constant | 0.80 | 0.01 | 0.78 | 0.81 |
| *r* | Fixed | 0.00 | – | – | – |
| *R’* | Fixed | 0.00 | – | – | – |
| *F* | Fixed | 1.00 | – | – | – |
| *F’* | Fixed | 0.00 | – | – | – |

Table 3-8. Gulf-wide Gulf Sturgeon survival estimates, standard errors (SE), and associated 95% confidence intervals (lower confidence limits [LCL] and upper confidence limits [UCL]) representing 1990–2009 in five-year intervals and annual survival estimates 2012–2021 from a Barker model (Model 5) estimating time-varying survival (), time-varying PIT tag detection probability (), and a constant rate of acoustic tag detection (). Fixed parameters are defined in Table 3-2. Data-deficient estimates were removed.

|  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- |
| Parameter | Years | Estimate | SE | LCL | UCL |
| *S* | 1990–1994 | 0.96 | 0.03 | 0.84 | 0.99 |
| *S* | 1995–1999 | 0.81 | 0.02 | 0.76 | 0.85 |
| *S* | 2000–2004 | 0.90 | 0.02 | 0.86 | 0.93 |
| *S* | 2005–2009 | 0.90 | 0.02 | 0.86 | 0.93 |
| *S* | 2010 | – | – | – | – |
| *S* | 2011 | – | – | – | – |
| *S* | 2012 | 0.92 | 0.02 | 0.88 | 0.94 |
| *S* | 2013 | 0.90 | 0.02 | 0.85 | 0.93 |
| *S* | 2014 | 0.72 | 0.03 | 0.67 | 0.77 |
| *S* | 2015 | 0.93 | 0.02 | 0.88 | 0.97 |
| *S* | 2016 | 0.74 | 0.03 | 0.68 | 0.79 |
| *S* | 2017 | 0.77 | 0.03 | 0.71 | 0.82 |
| *S* | 2018 | 0.91 | 0.02 | 0.87 | 0.94 |
| *S* | 2019 | 0.95 | 0.01 | 0.91 | 0.97 |
| *S* | 2020 | 0.94 | 0.02 | 0.90 | 0.96 |
| *S* | 2021 | 0.96 | 0.01 | 0.92 | 0.98 |
| *p* | 1990–1994 | 0.05 | 0.01 | 0.04 | 0.07 |
| *p* | 1995–1999 | 0.08 | 0.01 | 0.06 | 0.10 |
| *p* | 2000–2004 | 0.07 | <0.01 | 0.06 | 0.07 |
| *p* | 2005–2009 | 0.04 | <0.01 | 0.03 | 0.05 |
| *p* | 2010 | 0.05 | 0.01 | 0.04 | 0.06 |
| *p* | 2011 | 0.04 | <0.01 | 0.03 | 0.05 |
| *p* | 2012 | 0.03 | <0.01 | 0.03 | 0.04 |
| *p* | 2013 | 0.04 | 0.01 | 0.03 | 0.05 |
| *p* | 2014 | 0.02 | <0.01 | 0.01 | 0.03 |
| *p* | 2015 | 0.03 | <0.01 | 0.02 | 0.04 |
| *p* | 2016 | 0.02 | <0.01 | 0.02 | 0.03 |
| *p* | 2017 | 0.01 | <0.01 | 0.01 | 0.02 |
| *p* | 2018 | 0.03 | <0.01 | 0.02 | 0.04 |
| *p* | 2019 | 0.03 | 0.01 | 0.02 | 0.05 |
| *p* | 2020 | 0.02 | <0.01 | 0.01 | 0.02 |
| *p* | 2021 | 0.03 | 0.01 | 0.02 | 0.04 |
| *p* | 2022 | 0.02 | <0.01 | 0.01 | 0.03 |
| *R* | Constant | 0.79 | 0.01 | 0.78 | 0.80 |
| *r* | Fixed | 0.00 | – | – | – |
| *R’* | Fixed | 0.00 | – | – | – |
| *F* | Fixed | 1.00 | – | – | – |
| *F’* | Fixed | 0.00 | – | – | – |

Table 3-9. Gulf Sturgeon survival estimates, standard errors (SE) and associated 95% confidence intervals (lower confidence limits [LCL] and upper confidence limits [UCL] from a Barker model (Model 6) estimating river-specific survival (), river-specific PIT tag capture probability (), and a constant rate of acoustic tag detection (). Fixed parameters are defined in Table 3-2.

|  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- |
| Parameter | River | Estimate | SE | LCL | UCL |
| *S* | Pearl | 0.76 | 0.03 | 0.70 | 0.81 |
| *S* | Pascagoula | 0.87 | 0.01 | 0.84 | 0.89 |
| *S* | Escambia | 0.87 | 0.01 | 0.85 | 0.89 |
| *S* | Yellow | 0.87 | 0.01 | 0.86 | 0.89 |
| *S* | Choctawhatchee | 0.89 | 0.01 | 0.88 | 0.91 |
| *S* | Apalachicola | 0.89 | 0.01 | 0.87 | 0.90 |
| *S* | Suwannee | 0.89 | 0.01 | 0.88 | 0.90 |
| *p* | Pearl | 0.10 | 0.02 | 0.07 | 0.14 |
| *p* | Pascagoula | 0.07 | 0.01 | 0.05 | 0.09 |
| *p* | Escambia | 0.05 | 0.01 | 0.04 | 0.07 |
| *p* | Yellow | 0.07 | <0.01 | 0.06 | 0.08 |
| *p* | Choctawhatchee | 0.04 | <0.01 | 0.04 | 0.05 |
| *p* | Apalachicola | 0.05 | <0.01 | 0.05 | 0.06 |
| *p* | Suwannee | 0.03 | <0.01 | 0.02 | 0.03 |
| *R* | Constant | 0.80 | 0.01 | 0.78 | 0.81 |
| *r* | Fixed | 0.00 | – | – | – |
| *R’* | Fixed | 0.00 | – | – | – |
| *F* | Fixed | 1.00 | – | – | – |
| *F’* | Fixed | 0.00 | – | – | – |

Table 3-10. Five-year survival estimates, standard errors (SE) and associated 95% confidence intervals (lower confidence limits [LCL] and upper confidence limits [UCL]) for Gulf Sturgeon 1990–2014 and a single seven-year survival period representing 2015–2021 from a Barker model (Model 7) estimating river-specific survival (), river-specific PIT tag capture probability (), and a constant rate of acoustic tag detection (). Fixed parameters are defined in Table 3-2. Data-deficient estimates, as identified by years without river-specific sampling or data that did not result in model convergence, were removed. Estimates of 1.0 do not have 95% confidence intervals.

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| Parameter | River | Years | Estimate | SE | LCL | UCL |
| *S* | Pearl | 1990–1994 | – | – | – | – |
| *S* | Pearl | 1995–1999 | – | – | – | – |
| *S* | Pearl | 2000–2004 | 0.83 | 0.09 | 0.59 | 0.94 |
| *S* | Pearl | 2005–2009 | 0.82 | 0.13 | 0.44 | 0.96 |
| *S* | Pearl | 2010–2014 | 0.81 | 0.05 | 0.68 | 0.89 |
| *S* | Pearl | 2015–2021 | 0.84 | 0.05 | 0.72 | 0.91 |
| *S* | Pascagoula | 1990–1994 | – | – | – | – |
| *S* | Pascagoula | 1995–1999 | 1.00 | 0.00 | – | – |
| *S* | Pascagoula | 2000–2004 | 0.71 | 0.08 | 0.52 | 0.84 |
| *S* | Pascagoula | 2005–2009 | 0.51 | 0.16 | 0.23 | 0.78 |
| *S* | Pascagoula | 2010–2014 | 0.93 | 0.04 | 0.79 | 0.98 |
| *S* | Pascagoula | 2015–2021 | 0.94 | 0.01 | 0.91 | 0.96 |
| *S* | Escambia | 1990–1994 | – | – | – | – |
| *S* | Escambia | 1995–1999 | 0.83 | 0.16 | 0.36 | 0.98 |
| *S* | Escambia | 2000–2004 | 0.84 | 0.07 | 0.67 | 0.93 |
| *S* | Escambia | 2005–2009 | 0.89 | 0.04 | 0.79 | 0.94 |
| *S* | Escambia | 2010–2014 | 0.91 | 0.02 | 0.86 | 0.94 |
| *S* | Escambia | 2015–2021 | 0.85 | 0.02 | 0.81 | 0.89 |
| *S* | Yellow | 1990–1994 | 0.68 | 0.18 | 0.29 | 0.92 |
| *S* | Yellow | 1995–1999 | 1.00 | 0.00 | – | – |
| *S* | Yellow | 2000–2004 | 0.85 | 0.04 | 0.77 | 0.91 |
| *S* | Yellow | 2005–2009 | 0.98 | 0.03 | 0.80 | 1.00 |
| *S* | Yellow | 2010–2014 | 0.88 | 0.02 | 0.85 | 0.91 |
| *S* | Yellow | 2015–2021 | 0.86 | 0.02 | 0.82 | 0.88 |
| *S* | Choctawhatchee | 1990–1994 | – | – | – | – |
| *S* | Choctawhatchee | 1995–1999 | 0.84 | 0.06 | 0.69 | 0.92 |
| *S* | Choctawhatchee | 2000–2004 | 0.86 | 0.02 | 0.81 | 0.90 |
| *S* | Choctawhatchee | 2005–2009 | 0.98 | 0.03 | 0.79 | 1.00 |
| *S* | Choctawhatchee | 2010–2014 | 0.87 | 0.01 | 0.84 | 0.90 |
| *S* | Choctawhatchee | 2015–2021 | 0.87 | 0.01 | 0.90 | 0.93 |
| *S* | Apalachicola | 1990–1994 | 0.92 | 0.09 | 0.57 | 0.94 |
| *S* | Apalachicola | 1995–1999 | 0.94 | 0.07 | 0.57 | 0.99 |
| *S* | Apalachicola | 2000–2004 | 0.97 | 0.05 | 0.60 | 1.00 |
| *S* | Apalachicola | 2005–2009 | 0.82 | 0.03 | 0.75 | 0.88 |
| *S* | Apalachicola | 2010–2014 | 0.95 | 0.02 | 0.90 | 0.98 |
| *S* | Apalachicola | 2015–2021 | 0.85 | 0.02 | 0.81 | 0.88 |

Table 3-10 Continued

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| Parameter | River | Years | Estimate | SE | LCL | UCL |
| *S* | Suwannee | 1990–1994 | 0.99 | 0.03 | 0.46 | 1.00 |
| *S* | Suwannee | 1995–1999 | 0.86 | 0.03 | 0.80 | 0.91 |
| *S* | Suwannee | 2000–2004 | 0.94 | 0.03 | 0.85 | 0.98 |
| *S* | Suwannee | 2005–2009 | 0.97 | 0.03 | 0.85 | 0.99 |
| *S* | Suwannee | 2010–2014 | 0.95 | 0.01 | 0.92 | 0.97 |
| *S* | Suwannee | 2015–2021 | 0.89 | 0.01 | 0.86 | 0.91 |
| *p* | Pearl | 1990–1994 |  |  |  |  |
| *p* | Pearl | 1995–1999 |  |  |  |  |
| *p* | Pearl | 2000–2004 | 0.26 | 0.06 | 0.16 | 0.39 |
| *p* | Pearl | 2005–2009 | 0.01 | 0.01 | 0.00 | 0.04 |
| *p* | Pearl | 2010–2014 | 0.02 | 0.01 | 0.01 | 0.07 |
| *p* | Pearl | 2015–2021 | 0.02 | 0.01 | 0.01 | 0.07 |
| *p* | Pascagoula | 1990–1994 | – | – | – | – |
| *p* | Pascagoula | 1995–1999 | 0.27 | 0.07 | 0.15 | 0.43 |
| *p* | Pascagoula | 2000–2004 | 0.12 | 0.03 | 0.07 | 0.18 |
| *p* | Pascagoula | 2005–2009 | 0.25 | 0.26 | 0.02 | 0.83 |
| *p* | Pascagoula | 2010–2014 | 0.00 | 0.00 | 0.00 | 0.00 |
| *p* | Pascagoula | 2015–2021 | 0.06 | 0.01 | 0.04 | 0.09 |
| *p* | Escambia | 1990–1994 | – | – | – | – |
| *p* | Escambia | 1995–1999 | 0.00 | 0.00 | 0.00 | 0.00 |
| *p* | Escambia | 2000–2004 | 0.12 | 0.02 | 0.08 | 0.18 |
| *p* | Escambia | 2005–2009 | 0.05 | 0.01 | 0.04 | 0.08 |
| *p* | Escambia | 2010–2014 | 0.04 | 0.01 | 0.02 | 0.07 |
| *p* | Escambia | 2015–2021 | 0.05 | 0.01 | 0.04 | 0.07 |
| *p* | Yellow | 1990–1994 | 0.35 | 0.00 | 0.35 | 0.35 |
| *p* | Yellow | 1995–1999 | 0.05 | 0.05 | 0.01 | 0.31 |
| *p* | Yellow | 2000–2004 | 0.13 | 0.02 | 0.10 | 0.17 |
| *p* | Yellow | 2005–2009 | 0.03 | 0.01 | 0.02 | 0.04 |
| *p* | Yellow | 2010–2014 | 0.10 | 0.01 | 0.08 | 0.12 |
| *p* | Yellow | 2015–2021 | 0.02 | <0.01 | 0.02 | 0.03 |
| *p* | Choctawhatchee | 1990–1994 | – | – | – | – |
| *p* | Choctawhatchee | 1995–1999 | 0.07 | 0.03 | 0.04 | 0.14 |
| *p* | Choctawhatchee | 2000–2004 | 0.05 | 0.01 | 0.04 | 0.06 |
| *p* | Choctawhatchee | 2005–2009 | 0.06 | 0.01 | 0.05 | 0.08 |
| *p* | Choctawhatchee | 2010–2014 | 0.03 | <0.01 | 0.02 | 0.04 |
| *p* | Choctawhatchee | 2015–2021 | 0.02 | <0.01 | 0.02 | 0.03 |
| *p* | Apalachicola | 1990–1994 | 0.37 | 0.16 | 0.13 | 0.70 |
| *p* | Apalachicola | 1995–1999 | 0.13 | 0.04 | 0.07 | 0.22 |
| *p* | Apalachicola | 2000–2004 | 0.07 | 0.01 | 0.05 | 0.10 |
| *p* | Apalachicola | 2005–2009 | 0.05 | 0.01 | 0.03 | 0.07 |
| *p* | Apalachicola | 2010–2014 | 0.06 | 0.01 | 0.04 | 0.07 |
| *p* | Apalachicola | 2015–2021 | 0.04 | 0.01 | 0.03 | 0.05 |
| *p* | Suwannee | 1990–1994 | 0.05 | 0.01 | 0.03 | 0.06 |
| *p* | Suwannee | 1995–1999 | 0.06 | 0.01 | 0.05 | 0.07 |

Table 3-10 Continued

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| Parameter | River | Years | Estimate | SE | LCL | UCL |
| *p* | Suwannee | 2000–2004 | 0.02 | <0.01 | 0.02 | 0.03 |
| *p* | Apalachicola | 2010–2014 | 0.06 | 0.01 | 0.04 | 0.07 |
| *p* | Apalachicola | 2015–2021 | 0.04 | 0.01 | 0.03 | 0.05 |
| *p* | Suwannee | 1990–1994 | 0.05 | 0.01 | 0.03 | 0.06 |
| *p* | Suwannee | 1995–1999 | 0.06 | 0.01 | 0.05 | 0.07 |
| *p* | Suwannee | 2000–2004 | 0.02 | <0.01 | 0.02 | 0.03 |
| *p* | Apalachicola | 2010–2014 | 0.06 | 0.01 | 0.04 | 0.07 |
| *p* | Apalachicola | 2015–2021 | 0.04 | 0.01 | 0.03 | 0.05 |
| *p* | Suwannee | 1990–1994 | 0.05 | 0.01 | 0.03 | 0.06 |

Table 3-11.Five-year survival estimates, standard errors (SE) and associated 95% confidence intervals (lower confidence limits [LCL] and upper confidence limits [UCL]) for Gulf Sturgeon 1990–2019 and a single two-year survival period representing 2020–2021 from a Barker model (Model 8) estimating river-specific survival (), river-specific PIT tag capture probability (), and a constant rate of acoustic tag detection (). Fixed parameters are defined in Table 3-2. Data-deficient estimates are represented by dashed lines. Estimates of 1.0 do not have 95% confidence intervals.

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| Parameter | River | Years | Estimate | SE | LCL | UCL |
| *S* | Pearl | 1990–1994 | – | – | – | – |
| *S* | Pearl | 1995–1999 | – | – | – | – |
| *S* | Pearl | 2000–2004 | 0.83 | 0.09 | 0.59 | 0.94 |
| *S* | Pearl | 2005–2009 | 0.80 | 0.13 | 0.44 | 0.95 |
| *S* | Pearl | 2010–2014 | 0.81 | 0.06 | 0.68 | 0.90 |
| *S* | Pearl | 2015–2019 | 0.71 | 0.09 | 0.51 | 0.85 |
| *S* | Pearl | 2020–2021 | 0.95 | 0.04 | 0.79 | 0.99 |
| *S* | Pascagoula | 1990–1994 | – | – | – | – |
| *S* | Pascagoula | 1995–1999 | 1.00 | 0.00 | – | – |
| *S* | Pascagoula | 2000–2004 | 0.71 | 0.08 | 0.52 | 0.85 |
| *S* | Pascagoula | 2005–2009 | 0.50 | 0.15 | 0.23 | 0.76 |
| *S* | Pascagoula | 2010–2014 | 0.93 | 0.04 | 0.79 | 0.98 |
| *S* | Pascagoula | 2015–2019 | 0.92 | 0.02 | 0.86 | 0.95 |
| *S* | Pascagoula | 2020–2021 | 0.97 | 0.02 | 0.91 | 0.99 |
| *S* | Escambia | 1990–1994 | – | – | – | – |
| *S* | Escambia | 1995–1999 | 0.83 | 0.16 | 0.36 | 0.98 |
| *S* | Escambia | 2000–2004 | 0.84 | 0.07 | 0.67 | 0.93 |
| *S* | Escambia | 2005–2009 | 0.88 | 0.04 | 0.79 | 0.94 |
| *S* | Escambia | 2010–2014 | 0.91 | 0.02 | 0.87 | 0.95 |
| *S* | Escambia | 2015–2019 | 0.82 | 0.03 | 0.76 | 0.87 |
| *S* | Escambia | 2020–2021 | 0.94 | 0.03 | 0.85 | 0.98 |
| *S* | Yellow | 1990–1994 | 0.68 | 0.18 | 0.29 | 0.92 |

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| Parameter | River | Years | Estimate | SE | LCL | UCL |
| *S* | Yellow | 1995–1999 | 1.00 | 0.00 | – | – |
| *S* | Yellow | 2000–2004 | 0.85 | 0.04 | 0.77 | 0.91 |
| *S* | Yellow | 2005–2009 | 0.98 | 0.03 | 0.80 | 1.00 |
| *S* | Yellow | 2010–2014 | 0.89 | 0.02 | 0.85 | 0.91 |
| *S* | Yellow | 2015–2019 | 0.84 | 0.02 | 0.79 | 0.87 |
| *S* | Yellow | 2020–2021 | 0.89 | 0.03 | 0.81 | 0.94 |
| *S* | Choctawhatchee | 1990–1994 | – | – | – | – |
| *S* | Choctawhatchee | 1995–1999 | 0.84 | 0.06 | 0.69 | 0.92 |
| *S* | Choctawhatchee | 2000–2004 | 0.86 | 0.02 | 0.81 | 0.90 |
| *S* | Choctawhatchee | 2005–2009 | 0.98 | 0.03 | 0.76 | 1.00 |
| *S* | Choctawhatchee | 2010–2014 | 0.88 | 0.01 | 0.85 | 0.90 |
| *S* | Choctawhatchee | 2015–2019 | 0.87 | 0.01 | 0.84 | 0.89 |
| *S* | Choctawhatchee | 2020–2021 | 0.98 | 0.01 | 0.96 | 0.99 |
| *S* | Apalachicola | 1990–1994 | 0.82 | 0.09 | 0.57 | 0.94 |
| *S* | Apalachicola | 1995–1999 | 0.94 | 0.07 | 0.57 | 0.99 |
| *S* | Apalachicola | 2000–2004 | 0.97 | 0.05 | 0.60 | 1.00 |
| *S* | Apalachicola | 2005–2009 | 0.82 | 0.03 | 0.75 | 0.88 |
| *S* | Apalachicola | 2010–2014 | 0.96 | 0.02 | 0.91 | 0.98 |
| *S* | Apalachicola | 2015–2019 | 0.81 | 0.02 | 0.77 | 0.85 |
| *S* | Apalachicola | 2020–2021 | 0.96 | 0.02 | 0.88 | 0.99 |
| *S* | Suwannee | 1990–1994 | 0.99 | 0.03 | 0.46 | 1.00 |
| *S* | Suwannee | 1995–1999 | 0.86 | 0.03 | 0.80 | 0.91 |
| *S* | Suwannee | 2000–2004 | 0.94 | 0.03 | 0.85 | 0.98 |
| *S* | Suwannee | 2005–2009 | 0.97 | 0.03 | 0.85 | 0.99 |
| *S* | Suwannee | 2010–2014 | 0.96 | 0.01 | 0.92 | 0.98 |
| *S* | Suwannee | 2015–2019 | 0.81 | 0.02 | 0.76 | 0.86 |
| *S* | Suwannee | 2020–2021 | 1.00 | 0.00 | – | – |
| *p* | Pearl | 1990–1994 | – | – | – | – |
| *p* | Pearl | 1995–1999 | – | – | – | – |
| *p* | Pearl | 2000–2004 | 0.26 | 0.06 | 0.16 | 0.39 |
| *p* | Pearl | 2005–2009 | 0.01 | 0.01 | 0.00 | 0.05 |
| *p* | Pearl | 2010–2014 | 0.02 | 0.01 | 0.01 | 0.07 |
| *p* | Pearl | 2015–2019 | 0.04 | 0.03 | 0.01 | 0.13 |
| *p* | Pearl | 2020–2021 | 0.00 | 0.00 | – | – |
| *p* | Pascagoula | 1990–1994 | – | – | – | – |
| *p* | Pascagoula | 1995–1999 | 0.27 | 0.07 | 0.15 | 0.43 |
| *p* | Pascagoula | 2000–2004 | 0.12 | 0.03 | 0.07 | 0.18 |
| *p* | Pascagoula | 2005–2009 | 0.26 | 0.26 | 0.02 | 0.83 |
| *p* | Pascagoula | 2010–2014 | 0.00 | 0.00 | 0.00 | 0.00 |
| *p* | Pascagoula | 2015–2019 | 0.09 | 0.02 | 0.05 | 0.15 |
| *p* | Pascagoula | 2020–2021 | 0.04 | 0.01 | 0.02 | 0.08 |
| *p* | Escambia | 1990–1994 | – | – | – | – |

Table 3-11 Continued

Table 3-11 Continued

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| Parameter | River | Years | Estimate | SE | LCL | UCL |
| *p* | Escambia | 1995–1999 | 0.00 | 0.00 | 0.00 | 0.00 |
| *p* | Escambia | 2000–2004 | 0.12 | 0.02 | 0.08 | 0.18 |
| *p* | Escambia | 2005–2009 | 0.06 | 0.01 | 0.04 | 0.08 |
| *p* | Escambia | 2010–2014 | 0.04 | 0.01 | 0.02 | 0.07 |
| *p* | Escambia | 2015–2019 | 0.05 | 0.01 | 0.04 | 0.07 |
| *p* | Escambia | 2020–2021 | 0.04 | 0.01 | 0.02 | 0.07 |
| *p* | Yellow | 1990–1994 | – | – | – | – |
| *p* | Yellow | 1995–1999 | 0.05 | 0.05 | 0.01 | 0.31 |
| *p* | Yellow | 2000–2004 | 0.13 | 0.02 | 0.10 | 0.17 |
| *p* | Yellow | 2005–2009 | 0.03 | 0.01 | 0.02 | 0.04 |
| *p* | Yellow | 2010–2014 | 0.10 | 0.01 | 0.08 | 0.12 |
| *p* | Yellow | 2015–2019 | 0.02 | <0.01 | 0.01 | 0.02 |
| *p* | Yellow | 2020–2021 | 0.03 | 0.01 | 0.02 | 0.05 |
| *p* | Choctawhatchee | 1990–1994 | – | – | – | – |
| *p* | Choctawhatchee | 1995–1999 | 0.07 | 0.03 | 0.04 | 0.14 |
| *p* | Choctawhatchee | 2000–2004 | 0.05 | 0.01 | 0.04 | 0.06 |
| *p* | Choctawhatchee | 2005–2009 | 0.06 | 0.01 | 0.05 | 0.08 |
| *p* | Choctawhatchee | 2010–2014 | 0.03 | <0.01 | 0.02 | 0.04 |
| *p* | Choctawhatchee | 2015–2019 | 0.02 | <0.01 | 0.01 | 0.03 |
| *p* | Choctawhatchee | 2020–2021 | 0.02 | <0.01 | 0.02 | 0.04 |
| *p* | Apalachicola | 1990–1994 | 0.37 | 0.16 | 0.13 | 0.70 |
| *p* | Apalachicola | 1995–1999 | 0.13 | 0.04 | 0.07 | 0.22 |
| *p* | Apalachicola | 2000–2004 | 0.07 | 0.01 | 0.05 | 0.10 |
| *p* | Apalachicola | 2005–2009 | 0.05 | 0.01 | 0.03 | 0.07 |
| *p* | Apalachicola | 2010–2014 | 0.05 | 0.01 | 0.04 | 0.07 |
| *p* | Apalachicola | 2015–2019 | 0.05 | 0.01 | 0.04 | 0.06 |
| *p* | Apalachicola | 2020–2021 | 0.05 | 0.01 | 0.03 | 0.07 |
| *p* | Suwannee | 1990–1994 | 0.05 | 0.01 | 0.03 | 0.06 |
| *p* | Suwannee | 1995–1999 | 0.06 | 0.01 | 0.05 | 0.07 |
| *p* | Suwannee | 2000–2004 | 0.02 | <0.01 | 0.02 | 0.03 |
| *p* | Suwannee | 2005–2009 | 0.02 | <0.01 | 0.02 | 0.03 |
| *p* | Suwannee | 2010–2014 | 0.01 | <0.01 | 0.01 | 0.02 |
| *p* | Suwannee | 2015–2019 | 0.01 | <0.01 | 0.00 | 0.01 |
| *p* | Suwannee | 2020–2021 | 0.01 | <0.01 | 0.00 | 0.01 |
| *R* | Constant | 1990–2021 | 0.79 | 0.01 | 0.78 | 0.80 |
| *r* | Fixed | Fixed | 0.00 | – | – | – |
| *R’* | Fixed | Fixed | 0.00 | – | – | – |
| *F* | Fixed | Fixed | 1.00 | – | – | – |
| *F’* | Fixed | Fixed | 0.00 | – | – | – |

Table 3-12. Model selection table describing Akaike Information Criterion corrected for small sample size (AICc), including the difference in each model’s AICc score and the top model’s AICc score (ΔAICc), the number of parameters (K), the negative log-likelihood, and the weight of each model for eight competing Barker models characterizing adult Gulf Sturgeon survival (), capture probability (). Each model estimates a constant rate of acoustic detection (). Fixed parameters are summarized in Table 3-2.

|  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- |
| Model No. | Model parameterization | ΔAICc | K | nll | AIC weight |
| 8 |  | 0.00 | 149 | 22688.95 | 1.00 |
| 7 |  | 99.80 | 128 | 22831.40 | <0.01 |
| 5 |  | 204.04 | 53 | 23087.16 | <0.01 |
| 3 |  | 444.20 | 23 | 23387.58 | <0.01 |
| 2 |  | 558.92 | 20 | 23508.31 | <0.01 |
| 6 |  | 611.65 | 23 | 23555.03 | <0.01 |
| 4 |  | 660.86 | 8 | 23634.29 | <0.01 |
| 1 |  | 758.95 | 5 | 23738.38 | <0.01 |

Diagram, text

Description automatically generated

Figure 3-1**.** Conceptual diagram outlining the differences in Gulf Sturgeon monitoring data between 1990 and 2022 used to inform Barker model survival estimates.

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Figure 3-2. Five-year survival estimates for Gulf Sturgeon 1990–2014 and a single seven-year survival estimate representing 2015–2021 from a Barker model (Model 2) estimating river-specific survival (), river-specific PIT tag capture probability (), and a constant rate of acoustic tag detection (). Fixed parameters are summarized in Table 3-2. The errors bars provided represent 95% confidence intervals.

**A picture containing diagram, text, line, plot

Description automatically generated**

Figure 3-3. Five-year survival estimates for Gulf Sturgeon 1990–2014 and a single seven-year survival estimate representing 2015–2021 from a Barker model (Model 3) estimating river-specific survival (, river-specific PIT tag detection probability(), and a constant rate of acoustic tag detection (). The errors bars provided represent 95% confidence intervals.

**A picture containing text, screenshot, line, diagram

Description automatically generated**

Figure 3-4. Gulf-wide Gulf Sturgeon survival estimates and associated 95% confidence intervals representing 1990–2009 and 2010–2021 from a Barker model (Model 4) estimating time-varying survival (), time-varying PIT tag detection probability (), and a constant rate of acoustic tag detection (). Fixed parameters are summarized in Table 3-2.

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Description automatically generated**

Figure 3-5. Gulf-wide Gulf Sturgeon survival estimates and associated 95% confidence intervals representing 1990–2009 in five-year intervals and annual survival estimates 2012–2021 from a Barker model (Model 5) estimating time-varying survival (), time-varying PIT tag capture probability (), and a constant rate of acoustic tag detection (). Fixed parameters are defined in Table 3-2. Data-deficient estimates, as identified by years without river-specific sampling or data that did not result in model convergence, were removed. Five-year estimates and annual estimates occupy the same amount of space on x-axis.

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Figure 3-6.Gulf Sturgeon survival estimates and associated 95% confidence intervals from a Barker model (Model 6) estimating river-specific survival (), river-specific PIT tag capture probability (), and a constant rate of acoustic tag detection (). Fixed parameters are summarized in Table 3-2. Estimates of are plotted on the top and estimates are plotted on the bottom. Rivers plotted on the x-axis are arranged from west to east based on their location in the Gulf of Mexico.

**A picture containing text, diagram, line, plan

Description automatically generated**

Figure 3-7. Five-year survival estimates and associated 95% confidence intervals for Gulf Sturgeon 1990–2014 and a single seven-year survival period representing 2015–2021 from a Barker model (Model 7) estimating river-specific survival (), river-specific PIT tag capture probability (), and a constant rate of acoustic tag detection (). Fixed parameters are summarized in Table 3-2. Data-deficient estimates, as identified by years without river-specific sampling or data that did not result in model convergence, were removed. Estimates of 1.0 do not have 95% confidence intervals.

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Description automatically generated**

Figure 4-8. Five-year survival estimates and associated 95% confidence intervals for Gulf Sturgeon 1990–2019 and a single two-year survival period representing 2020–2021 from a Barker model (Model 8) estimating river-specific survival (), river-specific PIT tag capture probability (), and a constant rate of acoustic tag detection (). Fixed parameters are summarized in Table 3-2. Data-deficient estimates, as identified by years without river-specific sampling or data that did not result in model convergence, were removed. Estimates of 1.0 do not have 95% confidence intervals.

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Figure 3-9. Passive Integrated Transponder (PIT) tag capture probability () estimates and associated 95% confidence intervals estimated in five-year intervals for 1990–2014 tagging data and a single seven-year time period associated with 2015–2021 tagging from a Barker model (Model 7) estimating river-specific survival (), river-specific , and a constant rate of acoustic tag detection (). Fixed parameters are summarized in Table 3-2. The recapture process occurred in 1991–2022 and recaptured Gulf Sturgeon were initially tagged between 1990 and 2021, the range of years associated with survival estimates. Data-deficient estimates were removed. Estimates of 1.0 do not have 95% confidence intervals.

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Figure 3-10. Passive Integrated Transponder (PIT) tag capture probability () estimates and associated 95% confidence intervals estimated in five-year intervals for 1990–2014 tagging data and a single two-year time period associated with 2020–2021 tagging from a Barker model (Model 8) estimating river-specific survival (), river-specific , and a constant rate of acoustic tag detection (). Fixed parameters are summarized in Table 3-2. The recapture process occurred in 1991–2022 and recaptured Gulf Sturgeon were initially tagged between 1990 and 2021, the range of years associated with survival estimates. Data-deficient estimates were removed. Estimates of 1.0 do not have 95% confidence intervals.

chapter 4

GULF STURGEON POPULATION VIABILITY MAY BE LIMITED BY CREEPING CHRONIC RATES OF ADULT MORTALITY

Background

Gulf Sturgeon *Acipenser oxyrinchus desotoi* stocks were significantly reduced in the mid-to-late 20th century due to a combination of factors (i.e., overfishing, dam construction, and habitat degradation), which led to listing as a threatened species under the US Endangered Species Act (ESA) in 1991 (56 FR 49653; National Oceanic Atmospheric Administration [NOAA] and U.S. Fish and Wildlife Service [USFWS] 1991). Gulf Sturgeon have been managed as a single stock (i.e., range-wide as a species) under the ESA, as the species is considered threatened over most, if not all, of its range (NOAA and USFWS 1991). Despite decades of ESA protections, Gulf Sturgeon are not recovering at a rate to be delisted by the 2023 target (Flowers et al. 2020), in the Gulf Sturgeon Recovery Plan (GSRP; USFWS and Gulf States Marine Fisheries Commission [GSMFC] 1995).

In 1996, USFWS and NOAA adopted a joint policy (61 FR 4722) that outlined classification criteria for distinct population segments (DPSs), which they generally defined as discrete and significant populations of vertebrate species and subspecies that may be managed as individual units under the ESA (USFWS and National Marine Fisheries Service [NMFS] 1996). Within this DPS framework, managers could meet the varying individual needs of populations of listed species and potentially reclassify specific populations to extend or remove ESA protections. Rudd et al. (2014) and Chapter 2 suggested that Gulf Sturgeon may warrant classification as DPS’s. Gulf Sturgeon population characteristics that support DPS classification include genetic distinctness, high river fidelity, and differential survival across their range, specifically lower survival in the western Gulf of Mexico (Morrow et al. 1998; Rudd et al. 2014; Chapters 2 and 3). If extirpation risk also differs among river populations, this may further justify the reclassification of this listed species into DPSs to facilitate the direction of recovery actions to the population(s) that may need it most.

Population Viability Analysis (PVA) is a wide-ranging suite of quantitative analytical methods for assessing threats to fish and wildlife populations that commonly uses simulation-based methods of evaluating population persistence in the context of differing population dynamic rates, given some initial population size (Beissinger and McCullough, 2002; Morris & Doak, 2002). Therefore, PVA is built upon a synthesis of knowledge about a species’ life history, threats, environment, and management objectives. Most imperiled species lack important life history information needed to inform PVA (Coulson et al. 2001; Schultz and Hammond 2003). Fortunately, Gulf Sturgeon research dates back several decades (Huff 1975), and recent efforts to synthesize and model these multi-decadal data (Ahrens and Pine 2014; Flowers et al. 2020; Chapter 3) have generated the population dynamic estimates needed to forecast Gulf Sturgeon population fates to better understand potential conservation needs that may vary among these populations that are currently managed uniformly as a single species.

In this study, our objective was to assess the extirpation risks for Gulf Sturgeon populations with a range of starting sizes and select mortality rates estimated in Chapters 2 and 3. To evaluate the effects of various possible mortality sources on these populations, we simulated the following threats in the context of a range of population sizes and estimated the probability of extirpation along various time horizons: 1) chronic increases in baseline mortality, 2) varying episodic mortality event frequency, and 3) varying recruitment failure frequency.

Materials and Methods

We used a PVA simulator to transparently reflect uncertainty in the fate of Gulf Sturgeon populations under a variety of scenarios (i.e., threat-population size combinations). In our simulations, we separated populations into three life stages: pre-recruits, juveniles/sub-adults, and adults. Pre-recruits were subject to density-dependent mortality and were separated into multiple stanzas, where the strength of density dependence was specific to each stanza.

Model Description

We modified a PVA model originally detailed in Pine et al. (2013) for assessing directed take of Humpback Chub *Gila cypha* in Arizona and later used to assess risks to populations of two native fish species in New Mexico from habitat modifications (Pine et al. 2017). The original model was coded in Microsoft Visual Basic for Applications (VBA) by Carl Walters (University of British Columbia) and was migrated to program R (R Core Team 2022) and updated to its current form by Brett van Poorten (Simon Fraser University). I updated model inputs based on results from chapters 2 and 3 in this dissertation.

A summary of the equations used to inform the PVA model is included in Table 1. This model is an individual-based PVA model that simulates the dynamics of female fish only. Each individual PVA simulation was initialized by creating a list of N0 (*i*=1, 2, …,N0) individual female fish which represents the number of female fish which were informed by a range of recent population estimates for Gulf Sturgeon rivers identified as critical habitat (NOAA and USFWS 2003; USFWS and NMFS 2022). An age was assigned to each individual fish based on an assumed initial stable age distribution based on age-specific survival rates *Sa*. For each subsequent year of simulation t = 1, 2, 3,,…, *T* the age of each individual fish increases by 1. Fish die and are removed from the population (list of individual female fish) with the probability of 1-*S(ai)* and the sum of the survival fish is the total surviving population at time *Nt*. Random stochasticity is introduced into each individual simulation based on the distribution of the initial age structure and survival and the N0 for each simulation. For age-classes age 1+ relative survival rates are used based on a Lorenzen function (Lorenzen 2000). We also assumed that older fish had a lower variance in survival than young fish. Random effects on survival were included to mimic environmental effects (Pine et al. 2013) by varying the maximum survival As (in the Beverton-Holt stock recruitment formulation, Walters and Martell 2004) for each life history stanza included in the model (pre-recruit, juvenile/sub-adult, and adult). Life stanzas were used to screen specific scenarios of interest such as recruitment failure (juvenile stanza) or changes to adult mortality (adult stanza). Each stanza and year were independent and assumed a maximum survival followed a normal distribution with a specified mean and standard deviation. This survival rate was informed by estimates from Chapters 2 and 3. Recruitment for each year was a function of the number of individual fry (*Et*) computed for that year (from the list of all female fish) and the survival of these fry through each of the life stanzas (stanza-specific survival rate). The number of recruits was then added to the number of live fish at age 1.

Model input parameters are summarized in Table 2. Where possible, inputs were informed by results from Chapters 2 or 3. Fish were recruited to the vulnerable population at age 4. Published estimates of Gulf Sturgeon recruitment compensation ratio (*recK*), the improvement in juvenile survival at low population levels relative to carrying capacity (Goodyear 1977, 1980), range from 3.9 to 5 (Flowers 2008; Flowers et al. 2009, 2020; Ahrens and Pine 2014). Higher *recK* values imply that populations have larger increases in juvenile survival at low population sizes than at carrying capacity, which implies that such populations are more resilient to population declines due to this compensatory response to depletion (Walters and Martell 2004). Given the lack of available data on juvenile Gulf Sturgeon survival rates at different abundance levels, we used a highly conservative value of *recK* = 2 which is lower than observed values for many species (Goodwin et al. 2006). If the actual value of *recK* is >2, the resilience of these populations would be greater than these models suggest (e.g., lower extirpation probabilities, faster recovery).

Model Scenarios

I compared extirpation risk among the following three threats to better understand how extirpation risk may change if threats to different life stages were realized: Threat 1 – increases in chronic mortality rates (“chronic creep”); Threat 2 – increases in episodic mortality frequency; and Threat 3 – increases in recruitment failure frequency. These comparisons, informed by possible population threats discussed in the Gulf Sturgeon community in the last twenty years (USFWS and NMFS 2022; Dula et al. 2022), were also selected to provide insight into the relative importance of juvenile mortality (recruitment failure) and adult mortality (chronic and episodic mortality) to population viability. The simulated scenarios are summarized in Table 3.

Initial vulnerable abundance (i.e., number of adult females; ) in these simulations ranged between 100 and 10,000 individuals to represent the wide range of Gulf Sturgeon population sizes across the Gulf of Mexico (see summary in USFWS and NMFS 2022). I varied PVA model input parameters such as and adult mortality which informed various other equations related to survival at age, which then informed fecundity equations to determine recruitment. This recurring annual update of population size from these starting values occurred for 200 years.

The frequency of events (recruitment failure or mortality) represents mean frequencies of occurrence over the maximum 200-year time horizons. Extirpation probabilities represent the percentage of the 1,000 trials of each scenario that resulted in population collapse over 50-year, 100-year, and 200-year time horizons. A simulation trial that resulted in extirpation in 50 years was considered a population that went extinct over the 100 and 200 years as well.

Results

Creeping Chronic Mortality Rates

Baseline mortality rates of 0.11 resulted in no extirpation risk across all population sizes and time horizons (Scenarios 1–4; Tables 3–4; Figure 1). However, when adult mortality rates increased to 0.13, 200-year extinction probabilities ranged between 27.4% and 90.5% (Scenarios 5–8; Tables 3–4). A further increase in chronic mortality to 0.15 resulted in an 11.3% 50-year extirpation probability for populations starting with 100 fish, 100-year extirpation probabilities >26% for all populations starting with ≤1000 individuals, and full extirpation for all population sizes after 200 years (Scenarios 9–12; Tables 3–4).

Increasing Episodic Mortality Event Frequency

When the average occurrence of episodic mortality events was 1/50 years, we observed a 15.9% 200-year extirpation probability for populations starting with 100 individuals (Scenario 13; Tables 3–4). There was effectively no extirpation risk for populations starting with ≥500 fish (Scenarios 14-16; Tables 3–4). If mean event frequency increased to 1/25 years, the range of 200-year extirpation probabilities was 6.4–64.1% across all initial population sizes (Scenarios 17–20; Tables 3–4). The maximum simulated mean episodic event frequency of 1/10 years resulted in an 11.8% 50-year extirpation probability for populations starting with 100 fish, 100-year extirpation probabilities >25% for all populations starting with ≤1000 individuals and full extirpation for all population sizes after 200 years (Scenarios 21–24; Tables 3–4).

Increasing Recruitment Failure Frequency

Within the context of a baseline adult mortality rate of 0.11, recruitment failure 1/10 years resulted in effectively no extirpation risk for populations ≥500 initial individuals (Scenarios 30–32; Tables 3–4). When the mean frequency of recruitment failure increased to 1/5 years, extirpation probabilities ranged between 5.3% and 21.2% for these same ≥500 initial fish populations (Scenarios 30–32; Tables 3–4). Across both the five- and ten-year mean frequencies, there was a small probability (<5%) of extirpation for the smallest initial population size (Scenarios 25 and 29; Tables 3–4). Across a 200-year time horizon, these small populations had the greatest extirpation probabilities; (10.2% and 64.6%; Scenarios 25 and 29; Tables 3–4).

Discussion

Overall, this work suggests that increases in adult Gulf Sturgeon mortality are a greater risk to viability than recruitment failure. Flowers et al. (2020) developed an age-structured population model for Gulf Sturgeon to examine their population recovery characteristics and determined that any additional adult Gulf Sturgeon mortality beyond their simulated levels of 0.095 would likely slow population recovery. We observed that increases in both chronic and episodic mortality frequency both led to rapid increases in extinction risk across all time horizons (50, 100, 200 years). Because Gulf Sturgeon maximum age is likely at least 50 years (Chapter 3), these time horizons represent relatively few generations of Gulf Sturgeon within a population recovery, or extinction, context. Despite the difficulty of imagining a world 200 years in the future, this time horizon only represents about four generations for this long-lived species. Although this may present management challenges, it is important to interpret these scenarios through this extended, foggy lens to better understand the long-term ramifications of increases in mortality.

In the absence of quantitative standards for interpreting ESA status using extinction risk estimates, some studies have suggested their own thresholds. According to Hamilton and Moller (1995), a population with a ≤5% extirpation risk over 100 years is not at risk of extinction. Therefore, this applied definition of an imperiled population within the context of our PVA results suggests the following populations are at risk of extinction: (1) populations of 100 initial individuals facing a 0.13 chronic adult mortality rate; (2) populations of ≤1,000 initial individuals facing a 0.15 chronic adult mortality rate; (3) populations of 100 initial individuals facing a significant episodic mortality event every 25 years on average; and (4) populations of ≤10,000 initial individuals facing a significant episodic mortality event every 10 years on average. According to this definition, recruitment failure as often as every 5 years on average would not pose a significant extinction threat to any Gulf Sturgeon population.

The mortality rates used in the different PVA simulations are informed by recent empirical evidence. For example, a chronic mortality rate of 0.15 is similar to a population experiencing a Hurricane Michael-level episodic event (Dula et al. 2022) about once every ten years. The general equivalence of these two phenomena has implications on how we interpret mortality rate estimates and the consequences of major episodic mortality events. For the Pearl River, the mortality rate associated with the upper 95% confidence limit in Chapter 2 and the estimate of Chapter 3 was 0.24, 9% greater than the highest simulated adult mortality rate in this study. No other river had an estimated baseline mortality rate exceeding 0.15 in Chapters 2 or 3. With a mortality rate of this level, the PVA model would suggest ≥50% 100-year extirpation probability for populations starting with 500 individuals. A very conservative approach to applying this PVA model, by using the upper 95% CI on survival, suggests the Pearl River may have a higher extirpation risk than other river populations. Given the reports of higher mortality in this river since the 1990’s (Morrow et al. 1998), the persistence of this population may suggest one or both of the following: (1) the mortality rate on average is lower than 0.24; (2) other compensatory responses are occurring at adult or other life stages. This is because with a chronic mortality rate of 0.24 (Model 6; Chapter 3) and an initial population of 500 the PVA model would suggest a 97.6% 50-year extirpation probability. If recruitment compensation is higher in the Pearl River, or in any population, than what was assumed here, this would result in higher adult mortality thresholds. Recruitment compensation ratio estimation is a common issue facing fishery stock assessments, as it often relies on informed guesses and fixed parameter values for processes that may be dynamic (Martell et al. 2008; Flowers et al. 2020). Because population regulation assumptions (i.e., density dependence) have significant effects on population viability estimates, LaMontagne et al. (2002) suggests incorporating spatial heterogeneity when modeling these regulatory mechanisms. The prospect of such spatial heterogeneity is supported by previous estimates of regional differences in survival among these populations and their respective high fidelity rates, which suggests these populations are likely facing different types or levels of threats (Chapters 2 and 3). Although differing extinction risks among populations is not among the criteria for DPS classification, reclassification of Gulf Sturgeon as DPS’s may allow managers to better account for potential differences in viability through restoration actions and potentially delisting specific DPS’s that no longer warrant protection under the ESA. Future studies of Gulf Sturgeon population viability should consider assessing the impact of different population regulation assumptions on extirpation risk, particularly for the western Gulf populations.

The decline in Gulf Sturgeon populations was initially due to overfishing and other factors (NOAA and USFWS 1991). Despite a harvest moratorium Gulf Sturgeon populations have not rebounded to previous levels (Odenkirk 1989; USFWS and GSMFC 1995; Ahrens and Pine 2014). Flowers et al. (2020) assessed the utility of different management actions to promote Gulf Sturgeon recovery and identified fishery closure as likely the single most effective recovery measure. However, these authors highlight how the full benefits of this closure are not likely to be realized for decades because of the severe depletion of older, fecund fish before closure resulted in an erosion of the age structure (Walters et al. 2008; Flowers et al. 2020). Our results suggest chronic increases in baseline adult mortality represent the greatest threat to Gulf Sturgeon population viability. If age structure erosion is delaying recovery by limiting recruitment, it would be expected that additional adult mortality would exacerbate this issue.

The relative contributions of demographic stochasticity (creeping chronic mortality rates and recruitment failure) and environmental stochasticity (e.g., hurricanes) to extinction risk, depend on the magnitude and frequency of episodic mortality events and the mean and variance of the long-term population growth rate beneath carrying capacity () (Lande 1993). We simulated different frequencies of a realistically sized mortality event to better understand how episodic events of this severity increase extinction risks to populations of different sizes. We found similar increases between increasing frequency of episodic mortality and a chronic increasing of mortality over time. This is because when events occur with high frequency the impact to the population is the same as if the population was incrementally increasing. We found that increases in the frequency of episodic mortality events increased extirpation risk such that increases in frequency had a pseudo-chronic effect of increasing total mortality over time. Predicting the frequency and severity of episodic threats is a critical element of population extinction risk assessments (Menges 1990; Lande 1993; Mangel and Tier 1994). Despite the inability to manage the occurrence of these episodic mortality events, better estimates of the frequencies and magnitudes of such events (e.g., oil spills, hurricanes) will allow future PVA efforts to better represent the effects of these threats on Gulf Sturgeon population viability.

Lande (1993) modeled density-dependent population growth with respect to different demographic risks and determined that a population of modest size can persist for a long time in the face of episodic mortality events if  is substantially positive. High variability is commonly associated with lower population viability over time as this variability tends to slow long-term population growth leading to higher extirpation risk than populations with less variable (Morris and Doak 2002). Additional work to assess how Gulf Sturgeon population growth rates may differ among populations will provide additional insight into the resilience of these populations.

Lande (1993) also suggested that in sufficiently large populations, environmental stochasticity poses a greater extinction risk than demographic stochasticity. Events associated with individual fish tend to average out in larger populations, but contribute disproportionately to the persistence of smaller populations (Lande 1993). Therefore, higher estimates of baseline mortality in smaller populations are cause for greater concern than in larger populations. While the viability of larger populations may be less vulnerable to individual fates, they are still susceptible to mass die-offs associated with phenomena such as oil spills (Yaghmour et al. 2022) or pathogens (Fereidouni et al. 2019). This suggests episodic mortality likely poses a greater threat to population viability for larger Gulf Sturgeon populations (e.g., Suwannee River) than creeping chronic mortality or recruitment failure.

All plant and animal species may be at risk in coming decades from increasing threats from climate change (Bellard et al. 2012). Recovery planning efforts for listed species are slowly evolving to recognize climate change and incorporate future climate conditions into long-range planning documents (Malcom and Li 2018). In 2022, a proposed revision of section 10(j) regulations of the ESA would allow reintroductions of species of plants and animals into habitats outside of their historic range to adapt to a changing environment (US Department of the Interior 2022). NOAA Fisheries identified multiple threats to ongoing management efforts from changing climate (Karp et al. 2018) with an emphasis on risk planning. These types of scenario-based efforts combined with this PVA tool could be included in future Gulf Sturgeon recovery planning efforts to promote resilience to current and future threats to the species.

Simulation is invaluable to assessments of protected species as managers are limited by the potential for collapse from actual population manipulation. However, predictions of future population dynamics using PVA can only be accurate if the data are reliable and the future vital rate distributions are stationary (Coulson et al. 2001). The long-term, range-wide monitoring data (Chapters 2 and 3) and recent estimate of episodic mortality (Dula et al. 2022) used to inform our simulations of adult mortality give us confidence in the reliability of these estimates. In the face of the present and changing threats to Gulf Sturgeon (Waldman and Quinn 2022) the stationarity of these population dynamic rates is uncertain and perhaps unlikely. Also, episodic mortality events may have facilitated many extinctions, but estimates of their frequency are unreliable (Coulson et al. 2001). Nevertheless, we must continue to make predictions and take conservation actions in the face of this uncertainty, possibly by utilizing an Adaptive Management framework (Walters 1986; Runge 2011), to make use of the available data in our decision-making lest we regress to “faith-based” management (Hilborn 2006; Lacy 2019). Sustained investment into, and consistent use of, monitoring programs and modeling efforts that build detailed population dynamic models for Gulf Sturgeon is the best way to improve our understanding of the underlying processes continually shaping these populations and how we might facilitate their ongoing recovery.

Table 4-1. A summary of the equations used to inform the population viability analysis simulations.

|  |  |  |
| --- | --- | --- |
| Equation No. | Study Definition | Equation |
| 1 | Length-at-age |  |
| 2 | Weight-at-age |  |
| 3 | Fecundity-at-age |  |
| 4 | Size-based survival for recruited age-classes |  |
| 5 | Equilibrium egg density per recruit |  |
| 6 | Maximum survival of Beverton-Holt recruitment function |  |
| 7 | Carrying capacity parameter of Beverton-Holt recruitment function |  |
| 8 | Maximum survival of Beverton-Holt recruitment function |  |
| 9 | Carrying capacity parameter of Beverton-Holt recruitment function |  |
| 10 | The number of recruits in the first simulated year |  |
| 11 | Gulf Sturgeon abundance in the first year |  |

Table 4-2. Population viability analysis parameter estimates and sources

|  |  |  |
| --- | --- | --- |
| Model Parameter | Estimate | Comment and Source |
| Initial number of vulnerable fish () | 100–10,000 | Current population sizes are unknown. See USFWS and NMFS (2022) for a summary of river population abundance estimates. |
| Average long-term age 1 recruitment () | 67 |  |
| Compensation ratio in recruitment (*)* | 2.8 | Assumed value. More conservative compensation than estimates from Flowers (2008) or Ahrens and Pine (2014) |
| Metabolic rate parameter of von Bertalanffy function (𝐾*)* | 0.13 | Estimated from direct length-at-age and tagging data from 1978-2007 for the Apalachicola River by Flowers et al. (2010). |
| Minimum adult natural mortality rate () | 0.0627 |  |
| Length at 50% selectivity () | 0.27 |  |
| Selectivity shape parameter () | 0.045 |  |
| Weight at maturity relative to asymptotic weight () | 0.15 | Proportion of body weight lost to spawning estimated in Flowers et al. (2010). |
| Standard deviation of environmental effect on age 0 survival () | 0.6 | Assumed to be high to reflect highly variable natural environment. |

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| Scenario No. | Threat Definition | Adult Mortality | Vulnerable Abundance | Mean Freq. |
| 1 | Chronic mortality – baseline conditions | 0.11 | 100 | – |
| 2 | Chronic mortality – baseline conditions | 0.11 | 500 | – |
| 3 | Chronic mortality – baseline conditions | 0.11 | 1,000 | – |
| 4 | Chronic mortality – baseline conditions | 0.11 | 10,000 | – |
| 5 | Chronic mortality – creeping baseline | 0.13 | 100 | – |
| 6 | Chronic mortality – creeping baseline | 0.13 | 500 | – |
| 7 | Chronic mortality – creeping baseline | 0.13 | 1,000 | – |
| 8 | Chronic mortality – creeping baseline | 0.13 | 10,000 | – |
| 9 | Chronic mortality – creeping baseline | 0.15 | 100 | – |
| 10 | Chronic mortality – creeping baseline | 0.15 | 500 | – |
| 11 | Chronic mortality – creeping baseline | 0.15 | 1,000 | – |
| 12 | Chronic mortality – creeping baseline | 0.15 | 10,000 | – |
| 13 | Additional 35% episodic mortality | 0.11 | 100 | 1/50 years |
| 14 | Additional 35% episodic mortality | 0.11 | 500 | 1/50 years |
| 15 | Additional 35% episodic mortality | 0.11 | 1,000 | 1/50 years |
| 16 | Additional 35% episodic mortality | 0.11 | 10,000 | 1/50 years |
| 17 | Additional 35% episodic mortality | 0.11 | 100 | 1/25 years |
| 18 | Additional 35% episodic mortality | 0.11 | 500 | 1/25 years |
| 19 | Additional 35% episodic mortality | 0.11 | 1,000 | 1/25 years |
| 20 | Additional 35% episodic mortality | 0.11 | 10,000 | 1/25 years |
| 21 | Additional 35% episodic mortality | 0.11 | 100 | 1/10 years |
| 22 | Additional 35% episodic mortality | 0.11 | 500 | 1/10 years |
| 23 | Additional 35% episodic mortality | 0.11 | 1,000 | 1/10 years |
| 24 | Additional 35% episodic mortality | 0.11 | 10,000 | 1/10 years |
| 25 | Recruitment failure | 0.11 | 100 | 1/10 years |
| 26 | Recruitment failure | 0.11 | 500 | 1/10 years |
| 27 | Recruitment failure | 0.11 | 1,000 | 1/10 years |
| 28 | Recruitment failure | 0.11 | 10,000 | 1/10 years |
| 29 | Recruitment failure | 0.11 | 100 | 1/5 years |
| 30 | Recruitment failure | 0.11 | 500 | 1/5 years |
| 31 | Recruitment failure | 0.11 | 1,000 | 1/5 years |
| 32 | Recruitment failure | 0.11 | 10,000 | 1/5 years |

Table 4-3. A summary of the various mortality scenarios we evaluated using population viability analysis simulations including the average frequency of occurrence for episodic events.

Table 4-4. Extirpation probabilities associated with 50-year, 100-year, and 200-year time horizons for all 32 simulated population viability scenarios.

|  |  |  |  |
| --- | --- | --- | --- |
| Scenario No. | 50-year Probability | 100-year Probability | 200-year Probability |
| 1 | 0% | 0% | 0.5% |
| 2 | 0% | 0% | 0% |
| 3 | 0% | 0% | 0% |
| 4 | 0% | 0% | 0% |
| 5 | 0.1% | 18.5% | 90.5% |
| 6 | 0% | 0% | 58.2% |
| 7 | 0% | 0% | 46.2% |
| 8 | 0% | 0% | 27.4% |
| 9 | 11.3% | 90.8% | 100% |
| 10 | 0% | 47.2% | 100% |
| 11 | 0% | 26.2% | 100% |
| 12 | 0% | 3.8% | 99.6% |
| 13 | 0.1% | 1.3% | 15.9% |
| 14 | 0% | 0% | 0.6% |
| 15 | 0% | 0% | 0.3% |
| 16 | 0% | 0% | 0% |
| 17 | 0% | 9% | 64.1% |
| 18 | 0% | 0.1% | 20.4% |
| 19 | 0% | 0% | 14.1% |
| 20 | 0% | 0% | 6.4% |
| 21 | 11.8% | 78.9% | 100% |
| 22 | 0.5% | 38.5% | 99.9% |
| 23 | 0% | 25.7% | 99.7% |
| 24 | 0% | 5.3% | 99.1% |
| 25 | 0% | 0.3% | 10.2% |
| 26 | 0% | 0% | 0.1% |
| 27 | 0% | 0% | 0.1% |
| 28 | 0% | 0% | 0% |
| 29 | 0% | 4.4% | 64.6% |
| 30 | 0% | 0% | 21.2% |
| 31 | 0% | 0% | 13.9% |
| 32 | 0% | 0% | 5.3% |

![A graph of a number of numbers

Description automatically generated with medium confidence]()

Figure 4-1. Population projections from 1,000 simulations of Scenario 1, in which a chronic mortality rate of 0.11 was applied to an initial population of 100 adult female Gulf Sturgeon.

chapter 5

conclusion

Based on high-resolution original analyses of Gulf Sturgeon fidelity and survival patterns range-wide, plus published genetic information, my work suggests a minimum of five DPSs could be considered for Gulf Sturgeon: the western Gulf (Pearl and Pascagoula), the Escambia Bay rivers (Escambia and Yellow), the Choctawhatchee River, the Apalachicola River, and the Suwannee River. Grouping rivers in this framework would allow resource managers to address recurring uncertainties observed regionally, including lower survival in the western Gulf of Mexico systems. This approach can also allow better assessment of potential benefits from management actions related to fish passage issues by evaluating populations that may share some spatial redundancy in required habitats (such as spawning sites in multiple shared rivers such as the populations in the Escambia Bay rivers) versus management actions which may benefit a single discrete population with less exchange (fish passage within the Apalachicola River). These are the critical types of decisions and problem framing which must be undertaken to help define “what is recovery” in the context of Gulf Sturgeon in the present and future northern Gulf of Mexico river basins.

These insights would not have been possible without high-resolution acoustic telemetry data. The utility gained through the incorporation of the multi-decadal PIT tag data was the ability to retroactively take a glimpse into the past to estimate survival rates. These range-wide survival rates, representing largely unpublished spatiotemporal estimates, provided insight into the dynamics of Gulf Sturgeon survival rates within the context of a time period that more closely represented the generation time of the species. This approach utilized the greatest amount of Gulf Sturgeon information (i.e., number of rivers, fish, tags, and years of data) and determined that adult Gulf Sturgeon survival is high in most years. Therefore, when annual estimates are averaged out over 5+ year periods, Gulf-wide survival rates appear consistent. However, fine-scale temporal estimates (e.g., annual) reveal large decreases in survival (high mortality) during certain years. These large, isolated decreases in survival seem to represent episodic mortality events rather than small increases in baseline, chronic mortality that would be less dynamic.

The PVA results suggested that increases in adult Gulf Sturgeon mortality are a greater risk to viability than recruitment failure. Simulations of various scenarios representing chronic and episodic mortality effects on Gulf Sturgeon population viability revealed thresholds of resilience, suggesting both represented threats to Gulf Sturgeon population persistence. With an assumed interpretation that a population with a ≥5% extirpation probability over 100 years is at risk, the following theoretical populations would be imperiled: (1) populations of 100 initial individuals facing a 0.13 chronic adult mortality rate; (2) populations of ≤1,000 initial individuals facing a 0.15 chronic adult mortality rate; (3) populations of 100 initial individuals facing a significant episodic mortality event every 25 years on average; and (4) populations of ≤10,000 initial individuals facing a significant episodic mortality event every 10 years on average.

Coupling the high mortality estimates associated with the western Gulf across these analyses and others with the simulated results of such high chronic mortality rates, the western Gulf populations are likely at greater risk of extirpation than the river populations in the central and eastern Gulf of Mexico. Even if the assumptions undergirding this PVA related to critical population dynamic rates (e.g., recruitment compensation) are overly conservative and therefore inaccurate, the size of these western Gulf populations is likely relatively small (100–1,000 adults; USFWS and NMFS 2022), making these populations more vulnerable. Differing viability among populations may not be among the criteria to justify DPS reclassification, but this information may help maximize restoration actions by directing resources to the river populations with the greatest need.

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Biographical Sketch

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