DETERMINING DYNAMIC RATE FUNCTIONS AND ASSESSING THE INFLUENCE OF RIVER ALTERATION ON THE SHORTNOSE STURGEON IN GEORGIA

by

HUNTER JAMES RIDER

(Under the direction of Adam G. Fox and Martin J. Hamel)

ABSTRACT

The federally endangered Shortnose Sturgeon (*Acipenser brevirostrum*) inhabits Atlantic coastal rivers of North America. Overharvesting and habitat alteration resulted in population declines across the species range. Critical knowledge gaps persist on river-specific demographics and the influences of river modifications on southern populations. Prior studies indicate that age information from sturgeon fin spines is erroneous and can produce inaccurate population parameters. We used mark-recapture data to estimate dynamic rate functions in Georgia coastal rivers and estimate abundance through time in relation to dredging in the Savannah River. Mark-recapture revealed that the Shortnose Sturgeon grew slowly in Georgia coastal rivers and attained maximum ages that exceeded previously reported life spans from fin spine derived age data. Juvenile abundance significantly declined following a major dredging project. Comparative assessments of abundance in a control river were similar over the same time, emphasizing the influence that habitat alterations may have on Shortnose Sturgeon populations.

INDEX WORDS: Shortnose sturgeon, Endangered species, Population dynamics, Mark-recapture, Fin spines, Age and growth, Population modeling, Dredging

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DEDICATION

I would like to dedicate this thesis to my mother, father, and sister. I am forever thankful for the unwavering support and encouragement that you all have provided me throughout my graduate school journey. Your belief in me has been a continuous source of strength, and you have undoubtedly helped instill a passion for the sciences in me and helped me grow into the person I am today.

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CHAPTER 1

INTRODUCTION AND LITERATURE REVIEW

Species Description

The Shortnose Sturgeon (*Acipenser brevirostrum*) is a benthic fish that inhabits river systems, estuaries, and the ocean of the Atlantic coast of North America (Vladykov and Greeley 1963, Dadswell et al. 1984). Historically, populations of the Shortnose Sturgeon were distributed across major coastal river systems, from the Saint John River, New Brunswick, Canada, to the St. Johns River, Florida, United States (Vladykov and Greeley 1963, Kynard 1997). The Shortnose Sturgeon morphologically exhibits countershading, with a copper coloration on the dorsal side and a white coloration on the ventral side. In addition to coloration, a wide mouth (>60% interorbital width) and the absence of plates between the anal fin and lateral scutes are morphological distinctions between the Shortnose Sturgeon and juveniles of the Atlantic Sturgeon (*A. oxyrinchus oxyrinchus*) that are sympatric in inhabited riverine systems (Dadswell et al. 1984, Kynard 1997). Reaching maximum lengths of 1.2m, there is little sexual dimorphism between males and females of the Shortnose Sturgeon; however, females tend to be larger than males at the same age (Dadswell 1979, Dadswell et al. 1984).

Life History

The Shortnose Sturgeon is described as amphidromous- that is, a species that completes its life cycle in freshwater, but occasionally makes movements into saline environments (Kieffer and Kynard 1993, SSSRT 2010). The Shortnose Sturgeon uses multiple habitats of the river with complex migratory strategies that directly influence its life history (SSSRT 2010). For most of the year, adults inhabit the freshwater-saltwater interface of their natal rivers, making seasonal movements into brackish waters for foraging opportunities (Hall et al. 1991, Devries 2006, Ingram and Peterson 2018). The estuarine environment is used as foraging habitat by adults and juveniles and is characterized as having sand and mud substrates, as well as high variation in salinity (Kynard 1997).

Although the Shortnose Sturgeon is a long-lived fish, characterized with slow growth relative to other fishes, latitudinal variation in growth has been observed (Dadswell et al. 1984, SSSRT 2010). Southern populations (i.e., south of the mid-Atlantic Bight) of the Shortnose Sturgeon grow much more quickly than their northern counterparts, leading to sexual maturation at a younger age. In southern populations, males mature in 2−5 years and females mature in 4−7 years (Dadswell et al. 1984). In contrast, males from northern populations mature in 10−11 years and females mature in 7−18 years, depending on the river system (Dadswell et al. 1984, Kynard 1997). The rapid growth of the Shortnose Sturgeon in southern rivers also leads to smaller maximum sizes, which may be accompanied with a reduced longevity (i.e., shorter lived) relative to fish in northern populations (Dadswell et al. 1984). The oldest reported individual from the Saint John River, Canada, was 67 years old, compared to the oldest reported individual from the Altamaha River, GA, a 10− year−old fish (Heidt and Gilbert 1978, Dadswell et al. 1984). Shorter life spans can equate to fewer spawning runs and subsequently, less lifetime reproductive output in southern rivers (Hamel et al. 2020).

Depending on latitude, adults migrate upstream to spawn between January and May (Dadswell et al. 1984, Hall et al. 1991, Rogers and Weber 1994). In southern rivers, the Shortnose Sturgeon begins making spawning migrations between January-March (Hall et al.

1991, Rogers and Weber 1994, Devries 2006, Ingram and Peterson 2018). Water temperature is thought to be the major environmental factor influencing spawning migrations of the Shortnose Sturgeon (Dadswell et al. 1984), as fish generally begin moving upstream at temperatures of 9-16°C (Taubert 1980, Buckley and Kynard 1981, Kynard 1997, Ingram and Peterson 2018). Males are thought to spawn every 1-2 years, while females are thought to spawn less frequently, or every 3-5 years (Dadswell 1979, Kieffer and Kynard 1996, Kieffer and Kynard 2012, Ingram and Peterson 2018). Spawning habitat across the species range is characterized as the most upstream habitat used by the Shortnose Sturgeon, generally having a hard bottom and moderate bottom velocities (Kynard 1997, Kynard et al. 2016). Shortly after spawning, adults migrate quickly downstream to the saltwater-freshwater interface (Dadswell et al. 1984).

Eggs of the Shortnose Sturgeon are demersal and adhere to suitable bottom material for eight to thirteen days after fertilization, with hatching dependent on surrounding water temperatures (Meehan 1910, Buckley and Kynard 1981, Dadswell et al. 1984). Newly hatched individuals begin dispersing downriver for hundreds of kilometers (Kynard et al. 2016). After downstream dispersal, young-of-year (YOY) individuals inhabit freshwater reaches above the river’s salt wedge for approximately one year, where they opportunistically forage on drifting food (Kynard et al. 2016). YOY grow in body size during this time and develop a salinity tolerance (Dadswell et al. 1984, Jenkins et al. 1993), which increases with both age and size (Jenkins et al. 1993). Therefore, older juveniles (i.e., age 1+) can move into their natal river’s estuary, and their habitat use follows spatiotemporal patterns similar to adults (Dadswell et al. 1984, Kynard 1997).

Status and Threats

The fishing industry for sturgeon in North America began in the late 19th century, as a commercial fishery harvested both the Shortnose Sturgeon and the Atlantic Sturgeon for smoked meat and caviar, with caviar being processed from eggs of adult females (Cobb 1900, Gilbert 1989). Large numbers of harvested sturgeon were reported through the end of the late 19th century, with harvests of over 2.2 million kg in some years for all sturgeon species combined. By 1956, however, that harvest had drastically declined to 326,000 kg in the United States (Cobb 1900, Vladykov and Greeley 1963). Until the 1970s, this fishery referred to both the Atlantic Sturgeon and the Shortnose Sturgeon as "common sturgeon," harvesting both species despite the Shortnose Sturgeon having less economic importance due its smaller size and scarcity relative to the Atlantic Sturgeon (Vladykov and Greeley 1963, Murawski and Pacheco 1977). Population declines across the species' range from overfishing, along with habitat alteration and fragmentation from anthropogenic development in the 20th century, warranted the listing of the Shortnose Sturgeon on the U.S. Endangered Species Conservation Act in 1967 (NMFS 1998).

Despite its endangered status, a multitude of anthropogenic factors have impeded the recovery of the Shortnose Sturgeon across its range. Development of major rivers for industrial use in the 20th century diminished water quality, potentially causing further declines in populations across the species range (NMFS 1998). The development of dams in some river systems has obstructed the migratory routes of the Shortnose Sturgeon, preventing fish from reaching suitable spawning habitat and reproducing (Kynard 1997, Cooke et al. 2004, Kynard et al. 2016). Although a moratorium was placed on harvesting the Shortnose Sturgeon after its endangered species listing, these fish were susceptible to bycatch from other commercial fisheries, increasing mortality rates of incidentally caught adult fish (Collins et al. 2000, Kynard et al. 2016). More recently, climate change has posed a serious threat to population persistence, as sea level rise from global warming may result in an upstream shift in the freshwater: saltwater interface of rivers, possibly reducing survivability of larval and YOY sturgeon. Additionally, rising temperatures will decrease dissolved oxygen concentrations and intensify water quality problems, which could degrade thermal regimes that the Shortnose Sturgeon relies on during critical nursery and summer refuge periods (Collins et al. 2000, SSSRT 2010, Kynard et al. 2016).

Currently, populations of the Shortnose Sturgeon are presumably natal to 19 river systems. The lack of recovery of the Shortnose Sturgeon in many natal rivers runs the risk of permanent losses of individual populations that are critical for the overall survival of the species (NMFS 1998, SSSRT 2010). Therefore, the National Marine Fisheries Service (NMFS) recommended that the species be managed on a river-by-river basis as 19 distinct population segments to conserve genetic diversity (NMFS 1998). However, the most recent Shortnose Sturgeon Recovery Plan (SSSRT 2010) recommended conservation and management of the Shortnose Sturgeon at both a regional and a riverine population level, based on genetic clustering of geographically proximate populations. In addition to genetics, differences in life history characteristics and ecology also support this grouping (SSSRT 2010). A key objective for recovery of the Shortnose Sturgeon across its range is to develop an understanding of its population dynamics at the riverine level by establishing long-term, reliable recruitment and abundance estimates for assessing trends in populations over time (NMFS 1998). Furthermore, filling in knowledge gaps regarding the life history characteristics and demographics of southern populations of the Shortnose Sturgeon will provide managers with additional information to better manage the species for riverine-level recovery efforts (SSSRT 2010).

Age Data

Age data are a foundational aspect of understanding population dynamics in animal studies and are essential for the determination of life history characteristics and population models (Campana 2001). In fisheries science, fish ages are determined from calcified structures (Campana 2001). Periodic growth increments form on calcified structures through varying deposition rates that are related to the seasonal, somatic growth of the fish (Nakamoto et al. 1995). These growth increments are interpreted as being deposited on calcified structures annually and can be counted to produce age estimates (Campana and Neilson 1985, Francis 1990). Otoliths, or the inner ear bone of the fish, are commonly considered to be the most accurate and precise structures for determining age estimates (Secor et al. 1995, Campana 2001, Begg et al. 2005). However, otoliths require sacrifice of the individual for collection, so alternative, non-lethal aging methods – like fin spines or rays – are preferred for threatened and endangered species (Stevenson and Secor 1999, Murie et al. 2009, Baremore and Rosati 2014, Ulaski and Quist 2021, Martin et al. 2023).

Fish age data derived from calcified structures of known age fish can be validated by verifying growth band formations across the age structure on a species-by-species basis (Beamish and McFarlane 1983, Campana 2001). A known age fish refers to an individual that was initially tagged (e.g., at age 1) and caught a known period of time later (Campana 2001). A lack of validation studies has resulted in some ambiguity on fish age in some species because not all structures form consistent growth bands throughout the fish's life. Therefore, the bands on the structure do not equate to the fish's actual age (Beamish 1979, Campana 2001), which may result in erroneous age estimates (typically underestimation). The use of erroneous age estimates in studies on population dynamics can lead to inaccurate estimates in parameters, such as growth and mortality that are used to inform management actions, especially for long-lived species (Campana 2001). For example, aging inaccuracies resulted in overexploitation is the New Zealand stock of the Orange Roughy (*Hoplostethus atlanticus*). Original estimates of 30-year longevity in this species resulted in intensive harvest and near-population collapse. Subsequent research revealed that the Orange Roughy can live to ages exceeding 100 years and take ~25 years to reach sexual maturity (van den Broek 1983, Smith et al. 1995, Branch 2001, Campana 2001). Had the age and growth of the Orange Roughy been better understood initially, harvest limits may have been set lower, averting overharvest and population crash. Despite the ramifications of erroneous age estimates, validation of aging methods has frequently been overlooked before using age estimates in analyses of population dynamics (Beamish and McFarlane 1983, Hamel et al. 2016).

In studies on sturgeons, the pectoral fin spine (also known as the first ray) is most often used when determining age estimates (Brennan and Cailliet 1989, Woodland and Secor 2007, Hamel et al. 2014, Ruddle 2018, Ulaski and Quist 2021, Gragson and Fox 2022), as it has been considered the most reliable non-lethal aging structure for these long-lived fishes (Brennan and Cailliet 1989, Baremore and Rosati 2014). Few studies have validated fin spine aging in sturgeons, despite potential for mismanagement of these long-lived fishes (Baremore and Rosati 2014). Bruch et al. (2009) reported that pectoral fin spines from individuals of the Lake Sturgeon (*Acipenser fulvescens*) could be accurately aged up to 14 years before error increased, advising against the use of fin spines for collecting aging data on sturgeons. In pectoral fin spines collected from individuals of the White Sturgeon (*Acipenser transmontanus*), aging errors increased with fish age because individuals experiencing slow growth did not form a detectable growth band every year (Rien and Beamesderfer 1994). Similarly, age estimates derived from pectoral fin spines of the Shortnose Sturgeon increased in error with fish age, with ages of known-age fish being underestimated at six years and older (Rider et al. 2025). Although concerns of using fin spine derived age estimates have been revealed with multiple age-validation studies on sturgeons, pectoral fin spines have been used historically to estimate ages and develop population dynamic parameters for managing the Shortnose Sturgeon (e.g., Fleming et al. 2003, Devries 2006, Woodland and Secor 2007, Ruddle 2018). The concern of producing erroneous age estimates from pectoral fin spines warrants the use of alternative methods for determining population parameters of the Shortnose Sturgeon.

Mark-recapture data can be used in a diverse number of analyses to fill critical gaps in knowledge regarding the population dynamics of a species, including population abundances and survival rates (Lettink and Armstrong 2003). Furthermore, mark-recapture data is a useful tool in validating growth increment deposition periodicity in aging structures and in determining absolute age in long-lived species (Campana 2001, Hamel et al. 2014). Absolute age validation can be completed when known age fish are marked and subsequently recaptured (Campana 2001). Limitations of these methods stem from the reduced probability rates of recapturing older individuals over time (Campana 2001) and the associated effort and costs of collecting fish in general (Hamel et al. 2014). However, the associated errors and biases of estimating fish age from calcified structures justify the use of mark-recapture methods (Campana 2001), especially for long-lived fishes. For the Pallid Sturgeon (*Scapirhynchus albus*), mark-recapture data were more accurate for determining general characteristics of growth and estimating age, because fin spines demonstrated inconsistent growth band deposition and spacing (Hamel et al 2014). These findings corroborate those of Paragamian and Beamesderfer (2003), who used mark-recapture data from the White Sturgeon to determine that actual ages of fish were 1.5–2.0 times the ages estimated from fin spines. Mark-recapture methods are a practicable tool for determining growth and estimates of age for species that lack aging validation studies or have calcified structures that are difficult to interpret (Hamel et al. 2014, Hamel et al. 2015). Thus, mark-recapture can result in more accurate population parameter estimates that are used by managers to make decisions regarding conservation and management actions of long-lived species.

Population Demographics - Abundance Estimates

Developing long-term population trends of the Shortnose Sturgeon in each population segment is essential for implementing effective recovery and management plans of the species (NMFS 1998). Determining abundance estimates across demographic groups (i.e., juveniles, adults) of a population can inform scientists of temporal trends and help determine if recruitment is sufficient for increasing population sizes or replacing spawners (i.e., a stable population) (NMFS 1998, SSSRT 2010). Although highlighted as a key objective for assessing recovery actions of the species, few long-term monitoring programs persist across the species range (NMFS 1998, SSSRT 2010). Long-term (i.e., 20+ years) population monitoring of the Shortnose Sturgeon in the Hudson River displayed a presumed healthy population with an estimated abundance of ~60,000 individuals; the largest documented population across the species range (Bain et al. 2007). More importantly, the long-term monitoring program described in Bain et al. (2007), effectively justified the Hudson River population’s federal delisting due to its trends in increased population size and stability over decades of assessments. Northern populations of the Shortnose Sturgeon, in general, have been well-studied (NMFS 1998), leading to other documented reports of population recovery. This includes the population in the Saint John River, New Brunswick, with the total population estimated at 18,000 individuals (Dadswell 1979), and the Delaware River, estimated at >12,000 adults (ERC 2006).

At the current southern extent of the range of the Shortnose Sturgeon, data are deficient such that limited knowledge exists about population demographics (NMFS 1998, SSSRT 2010). The few existing assessments indicate that much smaller populations persist in southern rivers compared to northern rivers. In the Altamaha River, total annual abundances were variable, with annual point estimates ranging from a low of 1,206 fish (95% CI = 566−2,759) in 2009 to a high of 5,551 fish (95% CI = 2,804−11,304) in 2006; the Altamaha River supports the largest documented population south of the Delaware River (Peterson and Bednarski 2013). Similar abundances were later reported in the Savannah River, Georgia, by Bahr and Peterson (2017), where the maximum point estimate of abundance was 2,432 fish (95% CI = 1,025−6,102) in 2013. Although these studies were important for deriving baseline population abundances, long-term data on population abundances are still lacking across the species range, including in other Georgia rivers (i.e., Ogeechee River). There is a need for long-term, quantified assessments at the riverine population level to determine river-specific recovery and management strategies for the Shortnose Sturgeon (SSSRT 2010).

Within the South Atlantic Bight (North Carolina to Florida), the Savannah River is an industrialized system that contains the second largest quantified population of the Shortnose Sturgeon in the southern extent of its current distribution (NMFS 1998, Bahr and Peterson 2017). The Savannah River estuary is anthropogenically developed and contains the second largest container port on the US Atlantic coast (USACE 2012). The Shortnose Sturgeon uses the Savannah River estuary as nursery and foraging habitat to complete its life cycle (Hall et al. 1991, Collins et al. 2002). Despite its importance for the Shortnose Sturgeon, the Savannah River estuary has been severely altered by point-source pollution, development, and dredging (Pearlstine et al. 1989, Collins et al. 2000). Dredging is the deepening of navigational channels by excavating large volumes of sediments from aquatic environments (Carse and Lewis 2020). Although coastal development (e.g., navigable waterways) is essential to the economic growth and expansion of coastal regions (Engler et al. 1991, Carse and Lewis 2020), modifications by dredging can have negative impacts on aquatic environments and their biodiversity, particularly fish (Wenger et al. 2017). Dredging can affect the persistence of fish species that use coastally developed areas to complete their life cycles, especially for early life history stages. Dredging procedures result in the direct removal of habitat (i.e., sediment, structure) for benthic fishes (Fischer et al. 2012, Freedman et al. 2013, Soinski et al. 2022), as well as individual fish that persist in benthic habitat through entrainment (Eleftheriou and Robertson 1992, Drabble 2012, Barletta et al. 2016). Dredging operations can cause perturbations in sediments, which can increase sediment concentrations in the water column. Increased exposure to sediments can result in higher mortality of fish, particularly among early life stages, and can lower recruitment and cause a potential decline in long-term population persistence (Wenger et al. 2018). Abiotic factors are also susceptible to changes from dredging. In general, the removal of sediment in estuaries through dredging can increase salinity through farther upstream penetration of the salt wedge in increased depths, which can reduce dissolved oxygen (Johnston Jr 1981, Barletta et al. 2016). Thus, removal of sediments can influence surrounding abiotic factors and lead to additional changes in habitat or fish assemblages (Barletta et al. 2016).

The United States Army Corps of Engineers (USACE) completed its most recent harbor modification to the Savannah River estuary in 2022 – the Savannah Harbor Expansion Plan (i.e., SHEP) (USACE 2025). The SHEP was a multi-year channel deepening modification to 60 river kilometers (rkm) of the Savannah River estuary, deepening the channel by approximately 1.5 m to allow for larger cargo ships to access the river’s container port (USACE 2012). Before the

SHEP was initiated in 2015, environmental impact assessment (i.e., EIA) models predicted that the channel deepening would result in increased salinities and decreased dissolved oxygen concentrations in the entire estuary, leading to a loss of habitat for both juvenile and adult Shortnose Sturgeon. To mitigate predicted habitat loss, the USACE installed oxygen injection systems in the Savannah River estuary to combat decreases in dissolved oxygen levels (USACE 2012, USACE 2025); these oxygen injection systems, however, are unlikely to fully mitigate the habitat degradation caused by the SHEP because they do not address the increase in salinity (USACE 2012). Increases in salinity will likely be more detrimental to juvenile Shortnose Sturgeon survival than to adults (Jenkins et al. 1993) and may limit the future population's size and stability (Bahr and Peterson 2017).

The persistence of the populations of the Shortnose Sturgeon is highly sensitive to anthropogenic disturbances that can alter recruitment success and survival (Peterson and Bednarkski 2013). Although the ecological effects of the SHEP are uncertain, the predicted habitat degradation from EIA models warrant the long-term assessment and monitoring of both recruitment and abundance of the Shortnose Sturgeon in the Savannah River (USACE 2012). The effects of the SHEP on the Savannah River Shortnose Sturgeon population have not yet been assessed. However, mark-recapture efforts for the Shortnose Sturgeon have been employed in the Savannah River in the summer periods from 2013 – 2024, providing researchers with a multi-year dataset to develop interannual population parameters such as demographic based abundance estimates (Bahr and Peterson 2017, Cummins 2018). Long-term monitoring efforts to assess the Savannah River Shortnose Sturgeon population is key to understanding how trends in recruitment and population abundance change in relation to management actions or anthropogenic alteration over time. Thus, a long-term mark-recapture dataset can be used to assess the potential temporal change in demographic-based abundances of the Savannah River Shortnose Sturgeon as a result of SHEP dredging efforts.

Rationale and Objectives

Despite the Shortnose Sturgeon being federally listed as endangered, populations of the Shortnose Sturgeon are not recovering in the southern extent of the species' current range (Peterson and Bednarski 2013, Bahr and Peterson 2017, Kleinhans and Fox 2024). A better understanding of population dynamics is key to proper management, but many knowledge gaps have been identified – particularly in southern populations – including those related to age and population structure, abundance, and recruitment (NMFS 1998, SSSRT 2010). Therefore, the first goal of my study is to determine growth characteristics and predicted ages of the Shortnose Sturgeon with aging methods alternative to pectoral fin spines. In chapter 2, I use a long-term mark-recapture dataset collected from populations of the Shortnose Sturgeon among Georgia coastal rivers to quantify river-specific characteristics of growth and predicted ages. Furthermore, I compare age estimates derived from mark-recapture data to those derived from fin spines to further highlight any potential aging error associated with pectoral fin spines. The second goal of my study is to determine interannual demographic based abundances of the Savannah River Shortnose Sturgeon population over time in relation to dredging from the SHEP. In chapter 3, I use Huggins closed population capture-recapture models to quantify interannual adult and juvenile abundances from 2013 – 2024 with mark-recapture data collected from individuals of the Shortnose Sturgeon in the Savannah River. These estimates provide me with a 12-year dataset that spans the pre- and post- dredging periods of SHEP. I use long-term estimates of demographics-based point abundances from the Savannah River and a reference river in a Before-After, Control-Impact (BACI) model approach to determine potential changes in abundances of Savannah River Shortnose Sturgeon over time in conjunction to SHEP. In my BACI model approach, the Savannah River serves as the impacted system, while the Altamaha River, a geographically proximate and relatively unaltered river, serves as the control system. Therefore, point abundance estimates from the Altamaha River can account for potential natural variability in changes in abundance over time.

References

Bahr, D. L. and D. L. Peterson. 2017. Status of the Shortnose Sturgeon population in the Savannah River, Georgia. Transactions of the American Fisheries Society 146(1):92-98.

Bain, M. B., N. Haley, D. L. Peterson, K. K. Arend, K. E. Mills, and P. J. Sullivan. 2007. Recovery of a US endangered fish. PLoS One 2(1):e168.

Baremore, I. E. and J. D. Rosati. 2014. A validated, minimally deleterious method for aging sturgeon. Fishery Bulletin 112(4):274-282.

Barletta, M., F. J. Cysneiros, and A. R. Lima. 2016. Effects of dredging operations on the demersal fish fauna of a South American tropical-subtropical transition estuary. Journal of Fish Biology 89(1):890-920.

Beamish, R. J. 1979. New information on the longevity of Pacific ocean perch (*Sebastes alutus*). Journal of the Fisheries Board of Canada 36(11):1395-1400.

Beamish, R., and G. McFarlane. 1983. The forgotten requirement for age validation in fisheries biology. Transactions of the American Fisheries Society 112(6):735-743.

Begg, G. A., S. E. Campana, A. J. Fowler, and I. M. Suthers. 2005. Otolith research and application: current directions in innovation and implementation. Marine and freshwater research 56(5):477-483.

Branch, T. A. 2001. A review of orange roughy *Hoplostethus atlanticus* fisheries, estimation methods, biology and stock structure. South African Journal of Marine Science 23(1):181-203.

Brennan, J. S., and G. M. Cailliet. 1989. Comparative Age-Determination Techniques for White Sturgeon in California. Transactions of the American Fisheries Society 118(3):296-310.

Bruch, R. M., S. E. Campana, S. L. Davis-Foust, M. J. Hansen, and J. Janssen. 2009. Lake sturgeon age validation using bomb radiocarbon and known-age fish. Transactions of the American Fisheries Society 138(2):361-372.

Buckley, J., and B. Kynard. 1981. Spawning and rearing of shortnose sturgeon from the Connecticut River. The Progressive Fish-Culturist 43(2):74-76.

Campana, S. E., and J. D. Neilson. 1985. Microstructure of fish otoliths. Canadian Journal of Fisheries and Aquatic Sciences 42(5):1014-1032.

Campana, S. E. 2001. Accuracy, precision and quality control in age determination, including a review of the use and abuse of age validation methods. Journal of Fish Biology 59(2):197-242.

Carse, A., and J. A. Lewis. 2020. New horizons for dredging research: The ecology and politics of harbor deepening in the southeastern United States. Wiley Interdisciplinary Reviews: Water 7(6):e1485.

Cobb, J. N. 1900. The sturgeon fishery of Delaware River and Bay. Report of Commissioner of Fish and Fisheries 25:369-380.

Collins, M. R., S. G. Rogers, T. I. Smith, and M. L. Moser. 2000. Primary factors affecting sturgeon populations in the southeastern United States: fishing mortality and degradation of essential habitats. Bulletin of Marine Science 66(3):917-928.

Collins, M. R., W. C. Post, D. C. Russ, and T. I. J. Smith. 2002. Habitat use and movements of juvenile Shortnose Sturgeon in the Savannah River, Georgia-South Carolina. Transactions of the American Fisheries Society 131(5):975-979.

Cooke, D. W., J. P. Kirk, J. V. Morrow Jr, and S. D. Leach. 2004. Population dynamics of a migration limited shortnose sturgeon population. Proceedings of Annual Conference of Southeastern Association for Fish and Wildlife Agencies 58:82-91.

Cummins, A. J. 2018. Recruitment of Atlantic Sturgeon and Shortnose Sturgeon within the Savannah River, Georgia and South Carolina. Master's thesis. University of Georgia, Athens, USA.

Dadswell, M. 1979. Biology and population characteristics of the shortnose sturgeon, *Acipenser brevirostrum* LeSueur 1818 (Osteichthyes: Acipenseridae), in the Saint John River estuary, New Brunswick, Canada. Canadian Journal of Zoology 57(11):2186-2210.

Dadswell, M. J., B. D. Taubert, T. S. Squiers, D. Marchette, and J. Buckley. 1984. Synopsis of biological data on Shortnose Sturgeon, *Acipenser brevirostrum* LeSueur 1818. NOAA Technical Report NMFS 14.

DeVries, R. J. 2006. Population dynamics, movements, and spawning habitat of the shortnose sturgeon, *Acipenser brevirostrum*, in the Altamaha River system, Georgia. Master's thesis. University of Georgia, Athens, USA.

Drabble, R. 2012. Monitoring of East Channel dredge areas benthic fish population and its implications. Marine Pollution Bulletin 64(2):363-372.

Eleftheriou, A., and M. R. Robertson. 1992. The effects of experimental scallop dredging on the fauna and physical environment of a shallow sandy community. Netherlands Journal of Sea Research 30:289-299.

Environmental Research and Consulting (ERC). 2006. Final report of Shortnose Sturgeon population studies in the Delaware River, January 1999 through March 2003. Prepared for NOAA Fisheries, Protected Resources Division, Gloucester, Massachusetts and New Jersey Division of Fish and Wildlife, Bureau of Freshwater Fisheries, Lebanon.

Fischer, J., C. Paukert, and M. Daniels. 2012. Fish community response to habitat alteration: impacts of sand dredging in the Kansas River. Transactions of the American Fisheries Society 141(6):1532-1544.

Fleming, J., T. Bryce, and J. Kirk. 2003. Age, growth, and status of shortnose sturgeon in the lower Ogeechee River, Georgia. Proceedings of the Annual Conference of the Southeast Association of Fish and Wildlife Agencies 57:80-91.

Francis, R. I. C. C. 1990. Back-calculation of fish length: a critical review. Journal of Fish Biology 36(6):883-902.

Freedman, J. A., R. F. Carline, and J. R. Stauffer. 2013. Gravel dredging alters diversity and structure of riverine fish assemblages. Freshwater Biology 58(2):261-274.

Gilbert, C. 1989. Atlantic and shortnose sturgeons. Fish and Wildlife Service. US Dept. of Interior, Biol. Report, 82 (122), TR EL-82 (11-122).

Gragson, D. L., and A. G. Fox. 2022. Comparison of first and second pectoral fin rays for use in aging Shortnose Sturgeon (*Acipenser brevirostrum* LeSueur, 1818). Southeastern Naturalist 21(1):1-10.

Hall, J. W., T. I. Smith, and S. D. Lamprecht. 1991. Movements and habitats of shortnose sturgeon, *Acipenser brevirostrum* in the Savannah River. Copeia 695-702.

Hamel, M. J., J. D. Koch, K. D. Steffensen, M. A. Pegg, J. J. Hammen, M. L. Rugg, and J. M. Jech. 2014. Using mark–recapture information to validate and assess age and growth of long-lived fish species. Canadian Journal of Fisheries and Aquatic Sciences 71(4):559-566.

Hamel, M. J., M. A. Pegg, R. R. Goforth, Q. E. Phelps, K. D. Steffensen, J. J. Hammen, M. L. Rugg, and B. Jonsson. 2015. Range-wide age and growth characteristics of Shovelnose Sturgeon from mark–recapture data: implications for conservation and management. Canadian Journal of Fisheries and Aquatic Sciences 72(1):71-82.

Hamel, M. J., J. J. Spurgeon, C. J. Chizinski, K. D. Steffensen, and M. A. Pegg. 2016. Variability in age estimation results in ambiguity and false understanding of population persistence. North American Journal of Fisheries Management 36(3):514-522.

Hamel, M. J., J. J. Spurgeon, K. D. Steffensen, and M. A. Pegg. 2020. Uncovering unique plasticity in life history of an endangered centenarian fish. Sci Rep 10(1):12866.

Heidt, A. R., and R. J. Gilbert. 1978. The shortnose sturgeon in the Altamaha River drainage, Georgia. Pages 54-60 in R.R. Odum and L. Landers, editors. Proceedings of the rare and endangered wildlife symposium. Georgia Department of Natural Resources, Game and Fish Division, Technical Bulletin WL 4, Athens, Georgia.

Ingram, E. C., and D. L. Peterson. 2018. Seasonal movements of shortnose sturgeon (*Acipenser brevirostrum)* in the Altamaha River, Georgia. River Research and Applications 34(7):873-882.

Jenkins, W. E., T. I. Smith, L. D. Heyward, and D. M. Knott. 1993. Tolerance of shortnose sturgeon, *Acipenser brevirostrum*, juveniles to different salinity and dissolved oxygen concentrations. Proceedings of the Annual Conference of the Southeastern Association of Fish and Wildlife Agencies 47:476-484.

Johnston Jr, S. A. 1981. Estuarine dredge and fill activities: a review of impacts. Environmental management 5(5):427-440.

Kieffer, M., and K. B. Kynard. 1993. Annual Movements of Shortnose and Atlantic Sturgeons in the Merrimack River, Massachusetts. Transactions of the American Fisheries Society (122):1088-1103.

Kieffer, M. C., and B. Kynard. 1996. Spawning of the Shortnose Sturgeon in the Merrimack River, Massachusetts. Transactions of the American Fisheries Society 125(2):179-186.

Kieffer, M., and B. Kynard. 2012. Spawning and non-spawning migrations, spawning, and the effect of river regulation on spawning success of Connecticut River Shortnose Sturgeon. Life history and behaviour of Connecticut River shortnose and other sturgeons. B. Kynard, P. Bronzi and H. Rosenthal (Eds.). WSCS. Demand GmbH, Norderstedt, Spec. Publ 4:73-113.

Kleinhans, M., and A. G. Fox. 2024. Long-term trends in abundance and recruitment of Shortnose Sturgeon in the Altamaha River, Georgia. Marine and Coastal Fisheries 16(3):e10294.

Kynard, B. 1997. Life history, latitudinal patterns, and status of the shortnose sturgeon, *Acipenser brevirostrum*. Environmental Biology of Fishes 48:319-334.

Kynard, B., S. Bolden, M. Kieffer, M. Collins, H. Brundage, E. J. Hilton, M. Litvak, M. T. Kinnison, T. King, and D. Peterson. 2016. Life history and status of Shortnose Sturgeon (*Acipenser brevirostrum* LeSueur, 1818). Journal of Applied Ichthyology 32:208-248.

Lettink, M., and D. P. Armstrong. 2003. Mark-recapture analysis for monitoring threatened species: Introduction and case study. Department of Conservation Technical Series 28.

Martin, B. A., S. M. Burdick, R. K. Paul-Wilson, and R. J. Bart. 2023. Validating a Nonlethal Method of Aging Endangered Juvenile Lost River and Shortnose Suckers. Journal of Fish and Wildlife Management 14(1):121-134.

Meehan, W. E. 1910. Experiments in Sturgeon Culture. Transactions of the American Fisheries Society 39(1):85-91.

Murawski, S. A., and A. L. Pacheco. 1977. Biological and Fisheries Data on Atlantic Sturgeon, *Acipenser oxyrhynchus* (Mitchill). National Marine Fisheries Service Technical Series Report 10:69.

Murie, D. J., D. C. Parkyn, C. C. Koenig, F. C. Coleman, J. Schull, and S. Frias-Torres. 2009. Evaluation of finrays as a non-lethal ageing method for protected goliath grouper *Epinephelus itajara.* Endangered Species Research 7:213-220.

Nakamoto, R. J., T. T. Kisanuki, and G. H. Goldsmith. 1995. Age and growth of Klamath River green sturgeon (*Acipenser medirostris*). U.S. Forest Service and U.S. Fish and Wildlife Service, funded by U.S. Fish and Wildlife Service. Project Number 93-FP-13.

National Marine Fisheries Service (NMFS). 1998. Final recovery plan for the Shortnose Sturgeon, *Acipenser brevirostrum*. National Marine Fisheries Service.

Paragamian, V. L., and R. C. P. Beamesderfer. 2003. Growth estimates from tagged White Sturgeon suggest that ages from fin rays underestimate true age in the Kootenai River, USA and Canada. Transactions of the American Fisheries Society 132(5):895-903.

Pearlstine, L. R., R. Bartleson, W. Kitchens, and P. Latham. 1989. Lower Savannah River hydrological characterization. Florida Cooperative Fish and Wildlife Research Unit, Technical Report 35, University of Florida, Gainesville.

Peris-Mora, E., J. M. Diez Orejas, A. Subirats, S. Ibanez, and P. Alvarez. 2005. Development of a system of indicators for sustainable port management. Marine Pollution Bulletin 50(12):1649-1660.

Peterson, D. L., and M. S. Bednarski. 2013. Abundance and Size Structure of Shortnose Sturgeon in the Altamaha River, Georgia. Transactions of the American Fisheries Society 142(5):1444-1452.

Rider, H. R., M. K. Morgan, A. T. Bond, J. D. Nolan, A. G. Fox, M. J. Hamel. 2025. Validating Age Estimates from Pectoral Fin Spines and Length-Frequency Analysis of Known-Age Shortnose Sturgeon. Journal of the Southeastern Association of Fish and Wildlife Agencies 12:52-60.

Rien, T. A., and R. C. Beamesderfer. 1994. Accuracy and Precision of White Sturgeon Age Estimates from Pectoral Fin Rays. Transactions of the American Fisheries Society 123(2):255-265.

Rogers, S., and W. Weber. 1994. Movements of shortnose sturgeon in the Altamaha River system, Georgia. Georgia Department of Natural Resources, Coastal Resources Division, Marine Fisheries Section.

Ruddle, V. K. 2018. Age structure, reproduction, and recruitment of Atlantic Sturgeon (*Acipenser oxyrinchus)* and Shortnose Sturgeon (*Acipenser brevirostrum*) in the Cooper River, South Carolina. College of Charleston, Charleston, South Carolina, USA.

Secor, D. H., Dean, J. M. & Campana, S. E. (Eds) 1995. Recent Developments in Fish Otolith Research. University of South Carolina Press, Columbia, South Carolina, USA.

Shortnose Sturgeon Status Review Team (SSSRT). 2010. A biological assessment of shortnose sturgeon (Acipenser brevirostrum). Report to National Marine Fisheries Service, Northeast Regional Office.

Smith, D. C., S. G. Robertson, G. E. Fenton, and S. A. Short. 1995. Age determination and growth of orange roughy (*Hoplostethus atlanticus*): a comparison of annulus counts with radiometric ageing. Canadian Journal of Fisheries and Aquatic Sciences 52(2):391-401.

Soinski, T. A., C. d. S. Gato, A. A. Matsumoto, M. L. Brazão, and W. S. Smith. 2022. Dredging on water bodies and margin interventions: effects on fish. Boletim do Instituto de Pesca, 48:e641.

Stevenson, J. T., and D. H. Secor. 2000. Age determination and growth of Hudson River Atlantic sturgeon, *Acipenser oxyrinchus*. Fishery Bulletin 98(1):153-166.

Taubert, B. D. 1980. Reproduction of shortnose sturgeon (*Acipenser brevirostrum*) in Holyoke Pool, Connecticut River, Massachusetts. Copeia 1980(1):114-117.

Ulaski, M. E., and M. C. Quist. 2021. Filling knowledge gaps for a threatened species: age and growth of Green Sturgeon of the southern distinct population segment. Journal of Fish and Wildlife Management 12(1):234-240.

United States Army Corps of Engineers (USACE). 2012. Savannah harbor expansion project. Final General Re-evaluation Report, Savannah District, Savannah, Georgia.

United States Army Corps of Engineers (USACE). 2025. SHEP Progression. Accessed March 26, 2025, from https://www.sas.usace.army.mil/Missions/Civil-Works/Savannah-Harbor-Expansion/.

van den Broek, W. 1983. Information on orange roughy [*Hoplostethus atlanticus*, Challenger Plateau, New Zealand]. Catch'83.

Vladykov, V. D., and J. R. Greeley. 1963. Order Acipenseroidei. Fishes of the Western North Atlantic Part III. Memoir Sears Foundation for Marine Research 1, Copenhagen, Denmark. 630 pp.

Wenger, A. S., E. Harvey, S. Wilson, C. Rawson, S. J. Newman, D. Clarke, B. J. Saunders, N. Browne, M. J. Travers, J. L. McIlwain, P. L. A. Erftemeijer, J. P. A. Hobbs, D. McLean, M. Depczynski, and R. D. Evans. 2017. A critical analysis of the direct effects of dredging on fish. Fish and Fisheries 18(5):967-985.

Wenger, A. S., C. A. Rawson, S. Wilson, S. J. Newman, M. J. Travers, S. Atkinson, N. Browne, D. Clarke, M. Depczynski, P. L. Erftemeijer, and R. D. Evans. 2018. Management strategies to minimize the dredging impacts of coastal development on fish and fisheries. Conservation Letters 11(5):e12572.

Woodland, R. J., and D. H. Secor. 2007. Year-class strength and recovery of endangered Shortnose Sturgeon in the Hudson River, New York. Transactions of the American Fisheries Society 136(1):72-81.

CHAPTER 2

AGE AND GROWTH CHARACTERISTICS OF SHORTNOSE STURGEON IN GEORGIA COASTAL RIVERS REVEALED WITH MARK-RECAPTURE DATA[1]

[1] H. J. Rider, M. K. Morgan, A. T. Bond, J. D. Nolan, A. G. Fox, M. J. Hamel. To be submitted to *Transactions of the American Fisheries Society*.

Abstract

The Shortnose Sturgeon (*Acipenser brevirostrum*) is an endangered fish for which population declines are attributed to anthropogenic effects. One metric for evaluating recovery is assessing population demographics using age data. The pectoral fin spine is the most accepted structure for use in obtaining age estimates in sturgeons. However, fin spine age assignments in sturgeons are often associated with low accuracy and imprecision. Therefore, we used 771 capture-recapture events from Shortnose Sturgeon collected in the Savannah, Ogeechee, and Altamaha Rivers, Georgia, USA, from 2004−2024 to determine river-specific growth characteristics and age estimates. Predicted ages derived from mark-recapture data were compared with ages derived from fin spines to assess associated errors. Mark-recapture data showed that the Shortnose Sturgeon grew slowly and achieved life spans exceeding 20 years in Georgia rivers. Juvenile growth was similar among populations; however, asymptotic lengths and maximum ages differed among populations, with Ogeechee River fish reaching the largest average maximum sizes and greatest predicted longevity. Age estimates derived from fin spines were associated with low precision between readers. Estimates of mean length-at-age based on fin spines were lower than estimated ages at similar fork lengths based on captured-recaptured individuals. Thus, fin spine derived age estimates may lead to inaccuracies in modeling populations, whereas mark-recapture data may be a suitable alternative in producing estimates of age and growth. Continuing to monitor growth characteristics and predicted ages will be important for planning and assessing recovery efforts of the endangered Shortnose Sturgeon.

Introduction

The Shortnose Sturgeon (*Acipenser brevirostrum*) is a large, long-lived, amphidromous fish that inhabits river systems on the North American Atlantic coast (SSSRT 2010). Historically, populations of the Shortnose Sturgeon were distributed among 41 coastal river systems, from the Saint John River, Canada, to the St. Johns River, Florida, USA (Dadswell et al. 1984). The Shortnose Sturgeon uses both the estuary and freshwater reaches of its natal river system to complete its life cycle. Adult fish periodically spawns in upriver reaches and occasionally migrate to saline environments and neighboring rivers outside of the spawning window (Dadswell et al. 1984, Kynard et al. 2016). Populations of the Shortnose Sturgeon experienced declines from habitat alteration and overexploitation by commercial fisheries in the 20th century, and the species was the first fish listed under the U.S. Endangered Species Conservation Act in 1967 (SSSRT 2010). Despite federal protection, recovery of the Shortnose Sturgeon has been hindered by continued anthropogenic influences on extant populations (Kynard et al. 2016), including incidental bycatch from commercial fisheries, continued habitat alteration, degradation in water quality, and climate change (Collins et al. 2000, Cooke and Leach 2004, Kynard et al. 2016). Populations of the Shortnose Sturgeon currently inhabit 19 river systems, and the species is managed at the riverine-population level (SSSRT 2010, Wirgin et al. 2010).

The understanding of river-specific population demographics of the Shortnose Sturgeon is crucial for producing effective recovery plans for the species across its range. Across its broad distribution, there are latitudinal differences in the Shortnose Sturgeon's population demographics. Individuals in southern populations grow faster than their northern counterparts, resulting in sexual maturation at younger ages, smaller maximum body sizes, and reduced longevity (Dadswell et al. 1984). In southern populations (i.e., Georgia and South Carolina, USA), the Shortnose Sturgeon likely matures at 2 – 5 years - depending on sex - and lives to a maximum age of 20 years (Marchette and Smiley 1982, Dadswell et al. 1984). In contrast, the Shortnose Sturgeon in northern latitude populations (i.e., Bay of Fundy, Gulf of Maine, and northeastern USA) attains maturation at 5 – 12 years and reaches a maximum age exceeding 60 years (Dadswell et al. 1984). Age at maturity and longevity are integral for understanding population dynamics, and understanding differences among populations can lead to more realistic expectations for species recovery (Kynard et al. 2016).

In studies on sturgeons, the pectoral fin spine (also known as the first ray) is most often used to estimate age (Brennan and Cailliet 1989, Woodland and Secor 2007, Hamel et al. 2014, Ruddle 2018, Ulaski and Quist 2021, Gragson and Fox 2022), as it has been considered the most reliable, non-lethal aging structure for these long-lived fishes (Brennan and Cailliet 1989, Baremore and Rosati 2014). However, age validation studies focused on pectoral fin spines of sturgeons have reported inaccuracies in age estimates and low precision from this structure, often indicating that errors in age estimates increase with fish age (Rien and Beamesderfer 1994, Bruch et al. 2009, Baremore and Rosati 2014, Hamel et al. 2014, Izzo et al. 2021). Similarly, Rider et al. (2025) found that fin spine-based age estimates underestimated ages of the Shortnose Sturgeon as early as age- 6, with errors attributed to growth increment crowding in older fish that exhibit minimal somatic growth. Published population dynamics for long-lived fish species derived from fin spine age assignments are subject to inaccuracies that can potentially lead to mismanagement (Hamel et al. 2016), as may be the case for the Shortnose Sturgeon. The high level of error associated with age assignments from pectoral fin spines (Rider et al. 2025) warrants the use of alternative methods for deriving more accurate population dynamics of the Shortnose Sturgeon.

Mark-recapture data can be used in a diverse array of analyses to fill knowledge gaps about the population dynamics of a species, including population abundances and survival rates (Lettink and Armstrong 2003). Additionally, differences in individual size between capture and recapture events have been used to develop models investigating animal growth. In Pallid Sturgeon (*Scaphirhynchus albus*), mark-recapture data were more accurate than fin spines for determining growth and estimating age, because fin spines demonstrated inconsistent growth increment deposition and spacing (Hamel et al. 2014). These findings corroborate Paragamian and Beamesderfer (2003), who used mark-recapture of White Sturgeon (*Acipenser transmontanus*) to determine that actual ages were 1.5 – 2.0 times the ages estimated from fin spines. Mark-recapture methods are a practical tool for determining growth and estimates of age for species that lack aging validation studies across their age structure or have calcified structures that are difficult to interpret (Hamel et al. 2014, Hamel et al. 2015). Although mark-recapture techniques can be logistically constraining and time consuming (Campana 2001), they can produce age estimates with reduced error compared to age estimates derived from fin spines. More accurate fish ages result in more accurate estimates of the population parameters used in demographic models, resulting in more appropriate conservation and management actions. Therefore, the objectives of this study were to (1) use a long-term mark-recapture dataset to quantify characteristics of age, growth, and associated life history traits of the Shortnose Sturgeon in three Georgia coastal river systems, and (2) compare mean fork length-at-age estimates derived from mark-recapture data and fin spines.

Methods

We used a long-term mark-recapture dataset collected in Georgia coastal river systems by researchers from the University of Georgia. Data were collected from three rivers: the Savannah, Ogeechee, and Altamaha (Figure 2.1). Bottom anchored trammel and gill nets were used for sampling sturgeon during May – August of 2004 – 2024 in the tidally influenced portions of our study systems. Biological data including fork length were collected, and each fish was tagged with a passive integrated transponder (PIT) tag so that recaptured individuals could be recognized. These mark-recapture data have been used in prior studies to characterize long-term trends in population sizes and recruitment of sturgeons among Georgia coastal rivers (Peterson and Farrae 2011, Peterson and Bednarski 2013, Bahr and Peterson 2017, Cummins 2018, Fox et al. 2023, Kleinhans and Fox 2024). Additionally, fin spine sections were collected in the 2023–2024 sampling seasons. Using diagonal cutters, a ~1 cm portion of the anterior marginal pectoral fin spine was obtained from a sub-sample of fish.

Deriving growth from mark-recapture

Our analysis dataset was limited to fish with at least 365 days between capture and recapture dates to minimize the influence of potential measurement errors within shorter capture/recapture windows. If a fish was recaptured multiple times, we used the information only from the initial and most recent encounter (i.e., the longest ‘at-large’ time duration for each fish). Each individual was assigned to the river in which the most recent recapture event occurred for river-specific comparisons. To determine trends in growth and quantify potential differences in growth among river systems, we assessed each fish’s somatic growth by using the change in fork length between initial capture and latest recapture. We used the equation below to estimate annual growth increments of marked individuals:

$$G_i = \frac{(L_r - L_c)}{Y_i},$$

where G_i is growth of the *i*th fish, L_r is fork length at recapture, L_c is fork length at initial capture, and Y_i is the number of years between capture events for the *i*th fish. We plotted annual growth increments by fork length at first capture to observe potential differences in somatic growth of Shortnose Sturgeon among Georgia coastal rivers. We compared the slopes of the ensuing regression of annual growth on size of juvenile fish by river system with an analysis of covariance (i.e., ANCOVA) to determine if potential differences in growth persisted among Georgia river systems. We classified fish as juveniles if they were measured with a fork length < 500 mm, the published length at which the Shortnose Sturgeon reaches sexual maturity across its range (Dadswell et al. 1984). We focused on comparing juvenile growth among rivers, as adults exhibit minimal growth in some sturgeons (Hamel et al. 2015).

We estimated fork length-at-age of Georgia Shortnose Sturgeon with our mark-recapture data fitted to mixed-sex von Bertalanffy curves based on Fabens' (1965) modification method:

$$\Delta L = (L_\infty - L_c)(1 - e^{-kT}),$$

where ΔL is the change in length between capture events, L_∞ is the asymptotic length, L_c is the length at initial capture, *k* is the growth rate coefficient, and *T* is the number of years at large between initial capture and recapture. We used a non-linear regression approach in program R (R Core Team 2021) to estimate parameters L_∞ and *k* for each river. We then coupled the von Bertalanffy parameters of interest with Kirkwood's (1983) modified von Bertalanffy equation to estimate ages (t) for the Shortnose Sturgeon at any given fork length (L_t):

$$t = t_0 - \log_e[\frac{1-\frac{L_t}{L_\infty}}{k}].$$

An estimate of time at zero (t_0) could not be determined with Kirkwood's equation, therefore, we set t_0 equal to zero. Using the calculated Fabens' parameter of interests, predicted ages, and fork lengths of Shortnose Sturgeon, we visualized von Bertalanffy growth curves by river system. Longevity of the Shortnose Sturgeon in Georgia could be estimated from the growth curves and was characterized as the predicted age at the fork length corresponding to the von Bertalanffy parameter L_∞. Additionally, the 95% confidence intervals (CI) corresponding to the von Bertalanffy growth parameters of L_∞ and k were compared among river systems to determine if potential differences in characteristics of growth persisted.

Fin spines

We processed Shortnose Sturgeon fin spine sections following methods described by Rider et al. (2025). Ages were independently assigned to each sectioned fin spine structure by two readers. After independent age assignments were completed, a concert reading was held between readers to resolve any discrepancies in age assignments and finalize a consensus age for each structure. Fin spines were excluded from further aging analyses if a consensus age could not be finalized between readers or if structures were deemed to be too damaged for age assignment. For each fin spine where a consensus age was reached, we used the coefficient of variation equation (i.e., CV; Chang 1982):

$$\mathrm{CV} = 100\% \text{ x } \frac{\sqrt{\sum_{\mathrm{i}=1}^{\mathrm{R}} \frac{\left(\mathrm{X}_{ij} - \mathrm{X}_{j}\right)^2}{\mathrm{R} - 1}}}{\mathrm{X}_{j}},$$

and average percent error (i.e., APE; Beamish and Fournier 1981) equation:

$$\mathrm{APE} = 100\% \text{ x } \frac{1}{\mathrm{R}}\sum_{i=1}^{\mathrm{R}} \frac{\left|\mathrm{X}_{ij} - \mathrm{X}_j\right|}{\mathrm{X}_j}$$

to characterize precision between reader age estimates from fin spines, where X_{ij} corresponds to the *i*th age estimate of the *j*th individual, X_j is the average estimated age of the *j*th individual, and R is the number of reads per structure. We also used age assignments from independent readers to develop an age bias plot to determine if biases persisted between reader age estimates of the same structures (Campana et al. 1995). To complete our objective in comparing mean length-at-ages between aging methods, we used ImageJ software (Schneider et al. 2012) to produce back-calculated measurements from aged fin spine sections taken from Shortnose Sturgeon. Those measurements were combined with the Fraser-Lee method (Francis 1990) to estimate back-calculated fork length-at-ages:

$$\mathrm{F_i} = \mathrm{c} + (\mathrm{F}_c - \mathrm{c})\left(\frac{\mathrm{S}_i}{\mathrm{S}_c}\right),$$

where F_i is the length of the fish at the *i*th time, F_c is the length of the fish at time of capture, S_i is the length of the spine structure at the *i*th time, S_c is the length of the spine structure at capture, and c corresponds to the intercept of the regression of spine radii (i.e., S_c) and fork lengths (i.e., F_c). We used the Simple Fisheries Stock Assessment Methods (FSA) package (Ogle et al. 2023) in program R to perform fin spine-based data analyses (R Core Team 2021).

Results

The Shortnose Sturgeon mark-recapture dataset included 7,193 captures and had 1,528 recaptures, including 771 recaptures more than 365 days after initial capture. The overall mean period between capture-recapture events was 4 years (SE ± 0.12), and the maximum period between capture-recapture events was 19 years (Table 2.1). The maximum time at large between capture-recapture events were similar among the Savannah, Ogeechee, and Altamaha River populations – all had recaptures ≥ 17 years after initial capture (Table 2.1). Annual growth increments from mark-recapture data revealed that somatic growth was the greatest in smaller, juvenile fish (≤ 400 mm fork length); this trend was observed among all three populations (Figure 2.2). As initial fork length increased, a decrease in annual somatic growth was observed until an asymptotic fork length was attained. Annual growth increments suggested that minimal somatic growth – representative of an asymptotic fork length – was reached at ~700 mm fork length (Figure 2.2). However, both the Altamaha and Ogeechee Rivers had fish that exceeded 950 mm fork length at initial capture, whereas the largest individual in the Savannah River was 933 mm fork length (Figure 2.2).

The slopes of the regression of annual growth on size of juvenile fish were similar among populations ($DF = 2$, $F = 0.088$, $P = 0.9162$) (Figure 2.3). However, for any given fork length at initial capture, Ogeechee River juveniles were larger than their counterparts in both the Altamaha and Savannah Rivers (Figure 2.3). This was consistent with the von Bertalanffy equation results, in that the L_∞ parameter from Ogeechee River fish (892 mm fork length) was the largest of any river (Figure 2.4a). However, the *k* growth coefficient parameter appeared to be similar among populations (Figure 2.4b). Ogeechee River fish also reached the greatest predicted maximum age of ~28 years based on the average maximum body size derived from the river's population (i.e., representative of L_∞), although all study rivers featured predicted maximum ages exceeding 20 years (Figure 2.5). Altamaha River fish attained the smallest average maximum sizes (L_∞ = 760 mm fork length; Figure 2.4a) among populations, and juveniles were smallest at similar initial fork lengths (Figure 2.3).

A total of 268 fin spine sections were aged by independent readers. Twenty-seven structures (10%) were excluded due to poor readability. Consensus ages ranged from 1 to 17 years (mean ± SE = 6.41 ± 3.34), with most (89%) of the structures assigned an age of ≤ 10 years (Table 2.2). The CV estimate of total between reader age assignments from fin spines was 12.26, and the APE estimate was 8.67. Discrepancies between independent reader age estimates increased as growth increment counts increased on fin spine sections (Figure 2.6). When compared to mean fork lengths-at-ages derived from mark-recapture data, fin spine based aging methods underestimated fork lengths at similar ages by up to 11 years. However, fork lengths-at-ages from fin spine based aging methods exceeded those from the mark-recapture analysis for fish ≥ 12 years of age (Figure 2.7).

Discussion

Long-term mark-recapture data allowed us to describe river-specific growth characteristics and estimate age data from multiple populations of the Shortnose Sturgeon without the use of potentially erroneous age assignments derived from calcified structures. Predicted ages from mark-recapture data revealed that the Shortnose Sturgeon can attain older ages than was previously described using fin spine derived age assignments. We found that the Shortnose Sturgeon can live at least 28 years in southern rivers – almost a decade longer than the previously reported maximum of 20 years, which was derived from fin spines by Marchette and Smiley (1982). Paragamian and Beamesderfer (2003) similarly found that predicted ages derived from White Sturgeon mark-recapture data were 30 – 60% greater than fin spine derived ages. Furthermore, mean fork lengths-at-ages derived from fin spine-based methods in this study were generally less than those derived from mark-recapture data through age 12 (Figure 2.7). Additionally, the low precision calculated between reader age estimates from fin spines in this study supports the overall difficulty in assigning ages to these calcified structures.

Growth rate of juvenile fish was similar among the study systems, despite variation observed in juvenile body size at similar fork lengths. Differences in juvenile body sizes could be explained by interannual variation in body sizes of individuals within juvenile cohorts. Prior studies on the Shortnose Sturgeon in Georgia coastal rivers have used modal distributions from length frequency analysis to parse out length-based age groups for recruitment and abundance estimates. Variation in body size of individuals within presumed age-1 juvenile cohorts have been observed among years in Georgia coastal rivers (Cummins 2018, Kleinhans and Fox 2024); this may be explained by differences in hatch timing or other environmental factors. Data used in this study from the Altamaha and Savannah Rivers span multiple years and recruitment cohorts. Therefore, our analysis of juvenile growth in these systems encompasses interannual variations in growth, producing an overall average body size of juveniles over a long period of time (Figure 2.3). Our juvenile growth results for the Ogeechee River population of Shortnose Sturgeon were derived from a mark-recapture dataset that included one recruitment cohort, which may not accurately represent the average body size of an Ogeechee River juvenile over time. Future work is needed to understand the river-specific processes that influence somatic growth of juveniles, as juvenile growth influences population size if larger bodied juveniles experience greater survival and therefore recruit to the population at greater rates (Hubert and Quist 2010).

Intraspecific competition for available resources is a major factor that governs individual growth rates of fish, particularly when similar environmental conditions persist over time (Ross 1997). The Ogeechee River has been described as containing a small population of a few hundred individuals of the Shortnose Sturgeon, dominated by larger, adult fish (Farrae 2010, Kynard et al. 2016). Prior research through genetic analysis indicates that the Ogeechee River is inhabited both by Ogeechee-natal fish and by non-natal fish originating from surrounding rivers that immigrated for foraging opportunities and habitat resources (Wirgin et al. 2010, Peterson and Farrae 2011). The Altamaha River contains the largest population of the Shortnose Sturgeon south of the Delaware River, followed by the Savannah River population (Peterson and Bednarski 2013, Bahr and Peterson 2017, Kleinhans and Fox 2024). Both the Altamaha and Savannah Rivers have populations estimated to contain thousands of fish. Therefore, the Ogeechee River system's smaller population may provide non-natal, migrating individuals with greater foraging and resource opportunities relative to their natal river systems. Less competition for resources might explain why Ogeechee River fish attained the larger average maximum body sizes observed in this study than those in the other, more densely populated rivers. Fecundity typically increases with female body size (Trippel 1998, Scott et al. 1999), so fish that inhabit the Ogeechee River may achieve greater reproductive potential than individuals from the other two study rivers. A similar pattern has been observed in the Shortnose Sturgeon populations of the Gulf of Maine, where individuals from the abundant Kennebec River spawning population are observed to inhabit the Penobscot River, where spawning does not occur (Fernandes et al. 2010, Dionne et al. 2013, Altenritter et al. 2018). Altenritter et al. (2018) found that Kennebec River fish that inhabited the Penobscot River exhibited larger body sizes, faster growth, and greater relative health than those that remained in their natal river. Our study assigned each fish to the river population of its most recent capture, both because there is currently a reliable way to genetically determine natal origin of an individual Shortnose Sturgeon, and because density-dependent effects in the river of residence may be more important to growth than river of origin. Regular inter-riverine movement, which is poorly understood in the Shortnose Sturgeon, may be a factor that complicates our river-specific comparisons of growth and age estimates in this study. Therefore, future research on comparing demographics of partially migrating contingents to river-resident fish in Georgia coastal rivers may further clarify the differences observed in growth characteristics among populations of the Shortnose Sturgeon.

Characteristics of growth in this study can serve as baselines for future mark-recapture derived age and growth studies to assess recovery efforts of the Shortnose Sturgeon. The point at which an individual's growth trajectory reaches an asymptotic length can be used as a response variable to changes resulting from management actions, environmental change, or climatological events. Faster growth of individuals in early life stages through environmental or anthropogenic stressors can lead to earlier onset sexual maturation at smaller body sizes and younger ages. Earlier maturation may be beneficial as it allows for quicker reproductive contribution to increase the population abundance during potential population declines (Trippel 1995, Hutchings and Reynolds 2004). However, an earlier onset of maturation could reduce the longevity and asymptotic length of individuals within a population (Trippel 1995, Hamel et al. 2015). This could reduce lifetime reproductive output in two ways for the Shortnose Sturgeon. First, smaller females produce fewer eggs (Trippel 1998, Bennet et al. 2016). Second, decreased longevity would potentially result in fewer spawning runs over a shorter lifespan. Therefore, the continuation of long-term mark-recapture efforts for the Shortnose Sturgeon is essential for evaluating trends in riverine-level population demographics. Continuing to collect mark-recapture information from populations in the southern extent of the Shortnose Sturgeon's range will refine river-specific population characteristics and aid managers in developing conservation priorities for population persistence. However, because of the latitudinal differences in population characteristics of the Shortnose Sturgeon, these methods would also need to be employed to refine population parameters previously described in higher latitude rivers.

Using ages derived from mark-recapture, we found that the Shortnose Sturgeon in Georgia coastal rivers can result in maximum ages almost a decade older than the maximum ages that were previously reported based on fin spines. Using fin spines to age sturgeons can underestimate age because of the difficulty in reading these structures due to inconsistent annual growth increment deposition and/or crowding of annual growth increments (Bruch et al. 2009, Koch et al. 2011, Baremore and Rosati 2014, Hamel et al. 2014, Izzo et al. 2021). The use of erroneous age estimates can lead to inaccurate calculations of various dynamic rate functions (i.e., growth, recruitment, and mortality; Hamel et al. 2016). Collectively, these inaccuracies lead to a false understanding of life history that can lead to potential mismanagement of the Shortnose Sturgeon. By estimating longevity based on mark-recapture, we eliminated errors related to reading fin spines. Our results can be used to provide updated, more accurate information on population characteristics of the Shortnose Sturgeon in the southern extent of its extant range. For instance, the Shortnose Sturgeon exhibits a periodic life history strategy in which individuals intermittently spawn multiple times in a life span. This strategy is successful when recruitment is highly variable but limits the species from quickly increasing population abundances over short periods of time (Dadswell et al. 1984, Winemiller 2005). Because individuals have longer lifespans than previously thought, there is potential for more spawning runs and a higher lifetime reproductive output. Additional opportunities to spawn are crucial for the Shortnose Southern in southern rivers because of continuous threats from habitat degradation and climate change (e.g., loss of habitat from warming waters and drought conditions) that can impede the species from effectively recovering (SSSRT 2010, Kynard et al. 2016). Having accurate information on these life history traits is an important attribute for projecting populations under varying management scenarios, further supporting mark-recapture as an alternative aging tool to potentially erroneous fin spine derived age assignments.

References

Altenritter, M. E., G. B. Zydlewski, M. T. Kinnison, J. D. Zydlewski, and G. S. Wippelhauser. 2018. Understanding the basis of Shortnose Sturgeon (*Acipenser brevirostrum*) partial migration in the Gulf of Maine. Canadian Journal of Fisheries and Aquatic Sciences 75(3):464-473.

Bahr, D. L. and D. L. Peterson. 2017. Status of the Shortnose Sturgeon population in the Savannah River, Georgia. Transactions of the American Fisheries Society 146(1):92-98.

Baremore, I. E. and J. D. Rosati. 2014. A validated, minimally deleterious method for aging sturgeon. Fishery Bulletin 112(4):274-282.

Beamish, R. J. and D. A. Fournier. 1981. A method for comparing the precision of a set of age determinations. Canadian Journal of Fisheries and Aquatic Sciences 38:982–983.

Bennett, M. G., M. R. Whiles, and G. W. Whitledge. 2016. Population-level responses of life history traits to flow regime in three common stream fish species. Ecohydrology 9(7):1388-1399.

Brennan, J. S. and G. M. Cailliet. 1989. Comparative Age-Determination Techniques for White Sturgeon in California. Transactions of the American Fisheries Society 118(3):296-310.

Bruch, R. M., S. E. Campana, S. L. Davis-Foust, M. J. Hansen, and J. Janssen. 2009. Lake sturgeon age validation using bomb radiocarbon and known-age fish. Transactions of the American Fisheries Society 138(2):361-372.

Campana, S. 2001. Accuracy, precision and quality control in age determination, including a review of the use and abuse of age validation methods. Journal of Fish Biology 59(2):197-242.

Campana, S. E., M. C. Annand, and J. I. McMillan. 1995. Graphical and statistical methods for determining the consistency of age determinations. Transactions of the American Fisheries Society 124(1):131–138.

Chang, W. Y. 1982. A statistical method for evaluating the reproducibility of age determination. Canadian Journal of Fisheries and Aquatic Sciences 39(8):1208-1210.

Collins, M. R., S. G. Rogers, T. I. Smith, and M. L. Moser. 2000. Primary factors affecting sturgeon populations in the southeastern United States: fishing mortality and degradation of essential habitats. Bulletin of Marine Science 66(3):917-928.

Cooke, D. W., and S. D. Leach. 2004. Implications of a migration impediment on Shortnose Sturgeon spawning. North American Journal of Fisheries Management 24:1460–1468.

Cummins, A. J. 2018. Recruitment of Atlantic Sturgeon and Shortnose Sturgeon within the Savannah River, Georgia and South Carolina. Master's thesis. University of Georgia, Athens, Georgia, USA.

Dadswell, M. J., B. D. Taubert, T. S. Squiers, D. Marchette, and J. Buckley. 1984. Synopsis of biological data on Shortnose Sturgeon, Acipenser brevirostrum LeSueur 1818. NOAA Technical Report NMFS 14.

Dionne, P. E., G. B. Zydlewski, M. T. Kinnison, J. Zydlewski, and G. S. Wippelhauser. 2013. Reconsidering residency: characterization and conservation implications of complex migratory patterns of shortnose sturgeon (*Acispenser brevirostrum*). Canadian Journal of Fisheries and Aquatic Sciences 70(1):119-127.

Fabens, A. J. 1965. Properties and fitting of the Von Bertalanffy growth curve. Growth 29(3):265-89.

Farrae, D. J. 2010. Population dynamics and movements of Shortnose Sturgeon in the Ogeechee River, Georgia. Master's thesis. University of Georgia, Athens, Georgia, USA.

Fernandes, S. J., G. B. Zydlewski, J. D. Zydlewski, G. S. Wippelhauser, and M. T. Kinnison. 2011. Seasonal Distribution and Movements of Shortnose Sturgeon and Atlantic Sturgeon in the Penobscot River Estuary, Maine. Transactions of the American Fisheries Society 139(5):1436-1449.

Fox, A. G., M. A. Baker, A. J. Cummins, H. S. Evans Jr, K. L. Cummins, N. Q. Hancock, and D. L. Higginbotham. 2023. Recruitment of juvenile Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*) in the Savannah, Ogeechee, and Satilla Rivers in Georgia. Fishery Bulletin 121(3):129-140.

Francis, R. I. C. C. 1990. Back-calculation of fish length: a critical review. Journal of Fish Biology 36(6):883–902.

Gragson, D. L., and A. G. Fox. 2022. Comparison of first and second pectoral fin rays for use in aging Shortnose Sturgeon (*Acipenser brevirostrum* LeSueur, 1818). Southeastern Naturalist 21(1):1-10.

Hamel, M. J., J. D. Koch, K. D. Steffensen, M. A. Pegg, J. J. Hammen, M. L. Rugg, and J. M. Jech. 2014. Using mark–recapture information to validate and assess age and growth of long-lived fish species. Canadian Journal of Fisheries and Aquatic Sciences 71(4):559-566.

Hamel, M. J., M. A. Pegg, R. R. Goforth, Q. E. Phelps, K. D. Steffensen, J. J. Hammen, M. L. Rugg, and B. Jonsson. 2015. Range-wide age and growth characteristics of Shovelnose Sturgeon from mark–recapture data: implications for conservation and management. Canadian Journal of Fisheries and Aquatic Sciences 72(1):71-82.

Hamel, M. J., J. J. Spurgeon, C. J. Chizinski, K. D. Steffensen, and M. A. Pegg. 2016. Variability in age estimation results in ambiguity and false understanding of population persistence. North American Journal of Fisheries Management 36(3):514-522.

Hubert, W. A., and C. Q. Quist, editors. 2010. Inland fisheries management in North America, third edition. American Fisheries Society, Bethesda, Maryland.

Hutchings, J. A., and J. D. Reynolds. 2004. Marine fish population collapses: consequences for recovery and extinction risk. BioScience 54(4):297-309.

Izzo, L. K., D. L. Parrish, G. B. Zydlewski, and R. Koenigs. 2021. Second fin ray shows promise for estimating ages of juvenile but not adult Lake Sturgeon. North American Journal of Fisheries Management 41(1):217–228.

Kirkwood, G. P. 1983. Estimation of von Bertalanffy growth curve parameters using both length increment and age–length data. Canadian Journal of Fisheries and Aquatic Science 40(9):1405–1411.

Kleinhans, M., and A. G. Fox. 2024. Long-term trends in abundance and recruitment of Shortnose Sturgeon in the Altamaha River, Georgia. Marine and Coastal Fisheries 16(3):e10294.

Koch, J. D., K. D. Steffensen, and M. A. Pegg. 2011. Validation of age estimates obtained from juvenile Pallid Sturgeon *Scaphirhynchus albus* pectoral fin spines. Journal of Applied Ichthyology 27(2):209–212.

Kynard, B., S. Bolden, M. Kieffer, M. Collins, H. Brundage, E. J. Hilton, M. Litvak, M. T. Kinnison, T. King, and D. Peterson. 2016. Life history and status of Shortnose Sturgeon (*Acipenser brevirostrum* LeSueur, 1818). Journal of Applied Ichthyology 32:208-248.

Lettink, M., and D. P. Armstrong. 2003. Mark-recapture analysis for monitoring threatened species: Introduction and case study. Department of Conservation Technical Series 28.

Marchette, D. E. and R. Smiley. 1982. Biology and life history of incidentally captured Shortnose Sturgeon, *Acipenser brevirostrum*, in South Carolina. South Carolina Wildlife and Marine Resources Research Institute, Charleston, South Carolina, United States.

Ogle D. H., J. C. Doll, A. P. Wheeler A. P., and A. Dinno. 2023. FSA: Simple Fisheries Stock Assessment Methods. R package version 0.9.5, https://CRAN.R-project.org/package=FSA.

Paragamian, V. L., and R. C. P. Beamesderfer. 2003. Growth estimates from tagged White Sturgeon suggest that ages from fin rays underestimate true age in the Kootenai River, USA and Canada. Transactions of the American Fisheries Society 132(5):895-903.

Peterson, D. L. and M. S. Bednarski. 2013. Abundance and size structure of Shortnose Sturgeon in the Altamaha River, Georgia. Transactions of the American Fisheries Society 142(5):1444-1452.

Peterson D. L., and D. J. Farrae. 2011. Evidence of metapopulation dynamics in Shortnose Sturgeon in the southern part of their range. Transactions of the American Fisheries Society 140(6):1540–1546.

R Core Team. 2021. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Rider, H. R., M. K. Morgan, A. T. Bond, J. D. Nolan, A. G. Fox, M. J. Hamel. 2025. Validating Age Estimates from Pectoral Fin Spines and Length-Frequency Analysis of Known-Age Shortnose Sturgeon. Journal of the Southeastern Association of Fish and Wildlife Agencies 12:52-60.

Rien, T. A. and R. C. Beamesderfer. 1994. Accuracy and precision of White Sturgeon age estimates from pectoral fin rays. Transactions of the American Fisheries Society 123(2):255-265.

Ross, M. R. 1997. Fisheries conservation and management. Prentice Hall, Upper Saddle River, New Jersey.

Ruddle, V. K. 2018. Age structure, reproduction, and recruitment of Atlantic sturgeon (*Acipenser oxyrinchus*) and Shortnose Sturgeon (*Acipenser brevirostrum*) in the Cooper River, South Carolina. Master's thesis. College of Charleston, Charleston, South Carolina.

Schneider, C. A., W. S. Rasband, and K. W. Eliceiri. 2012. NIH Image to ImageJ: 25 years of image analysis. Nature Methods 9(7):671–675.

Scott, B., G. Marteinsdottir, and P. Wright. 1999. Potential effects of maternal factors on spawning stock–recruitment relationships under varying fishing pressure. Canadian Journal of Fisheries and Aquatic Sciences 56(10):1882-1890.

Shortnose Sturgeon Status Review Team (SSSRT). 2010. A biological assessment of shortnose sturgeon (*Acipenser brevirostrum*). Report to National Marine Fisheries Service, Northeast Regional Office.

Trippel, E. A. 1995. Age at maturity as a stress indicator in fisheries. Bioscience 45(11):759-771.

Trippel, E. A. 1998. Egg size and viability and seasonal offspring production of young Atlantic Cod. Transactions of the American Fisheries Society 127(3):339-359.

Ulaski, M. E., and M. C. Quist. 2021. Filling knowledge gaps for a threatened species: age and growth of Green Sturgeon of the southern distinct population segment. Journal of Fish and Wildlife Management 12(1):234-240.

Winemiller, K. O. 2005. Life history strategies, population regulation, and implications for fisheries management. Canadian Journal of Fisheries and Aquatic Sciences 62(4):872-885.

Wirgin I., C. Grunwald, J. Stabile, and J.R. Waldman. 2010. Delineation of discrete population segments of Shortnose Sturgeon, *Acipenser brevirostrum*, based on mitochondrial DNA control region sequence analysis. Conservation Genetics 11:689–708.

Woodland, R. J., and D. H. Secor. 2007. Year-class strength and recovery of endangered Shortnose Sturgeon in the Hudson River, New York. Transactions of the American Fisheries Society 136(1):72-81.

Tables and Figures

Table 2.1. Sample sizes for captures of the Shortnose Sturgeon in Georgia coastal rivers from 2004 – 2024. Sample size is indicative of the number of individuals per study system included in our analysis. We present the mean time at large in years, as well as the standard error (SE) and maximum (max) time at large between capture-recapture events in each river.

		Time at large (years)		
River	Sample size	Mean	SE	Max
Altamaha	455	4.48	0.16	18.01
Ogeechee	47	4.05	0.61	17.05
Savannah	269	4.15	0.18	18.96

Table 2.2. Summary statistics of consensus age assignments derived from pectoral fin spines of the Shortnose Sturgeon. Fin spines were collected from fish in Georgia coastal rivers from 2004 – 2024. We present the sample size for each consensus age, as well as fork length mean, fork length standard error (SE), and fork length range for each consensus age.

		Fork length (mm)		
Consensus age	Sample size	Mean	SE	Range
1	24	378	20	332 – 415
2	14	438	32	398 – 498
3	12	476	36	416 – 548
4	13	539	66	463 – 735
5	31	567	64	456 – 702
6	28	630	94	479 – 805
7	31	671	79	524 – 817
8	33	685	83	521 – 934
9	15	733	71	606 – 897
10	13	756	58	686 – 872
11	9	700	84	537 – 788
12	9	796	47	723 – 865
13	2	794	44	763 – 825
14	4	838	74	755 – 934
15	1	877	-	-
16	1	880	-	-
17	1	853	-	-
Total	*241*	*616*	*141*	*332 – 934*

Figure 2.1. Study site in Georgia, USA, from which mark-recapture data were collected from individuals of the Shortnose Sturgeon in 2004 – 2024. The ovals over each estuary indicate the approximate sampling area in each river.

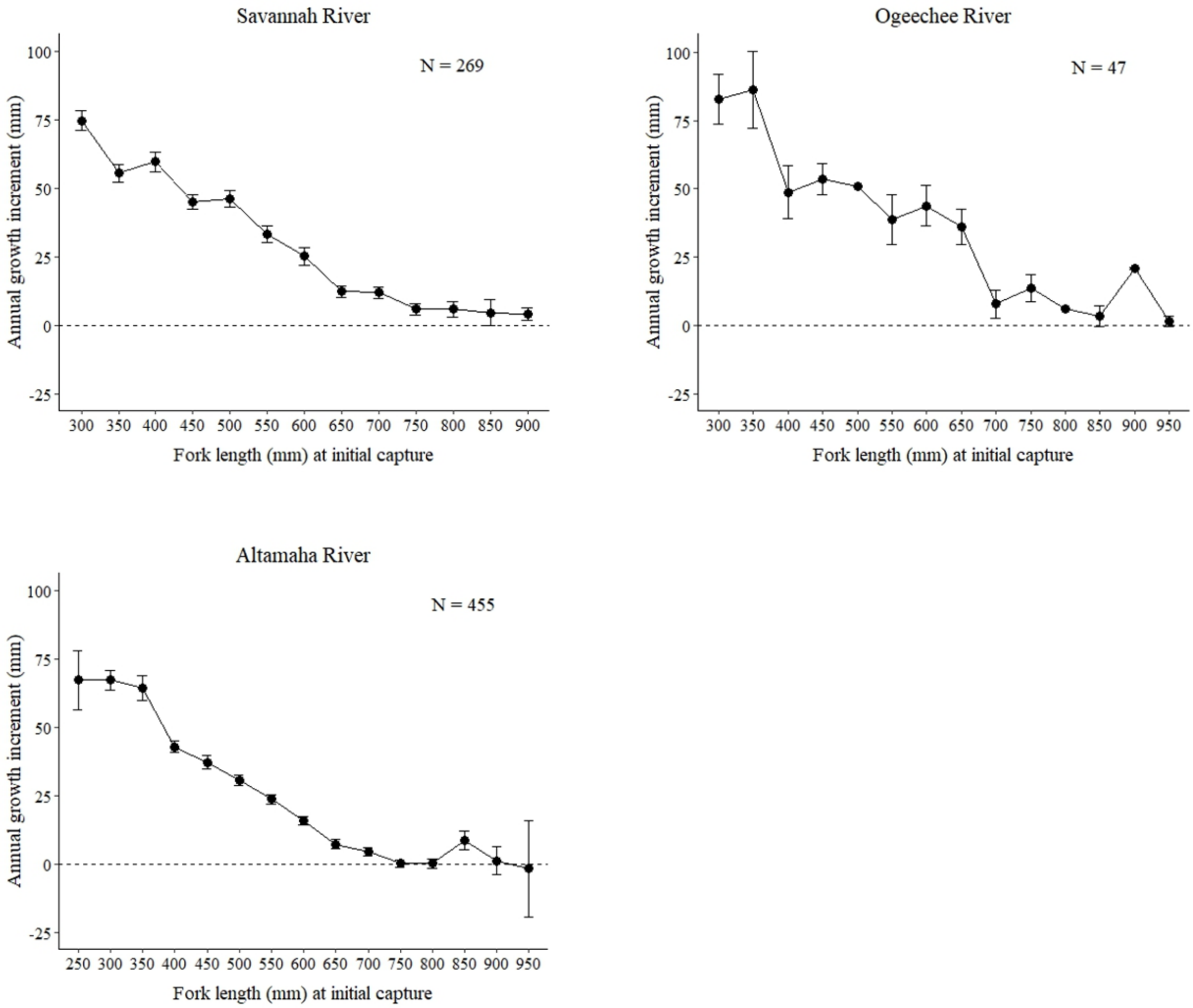

Figure 2.2. Annual growth increments derived from mark-recapture data of the Shortnose Sturgeon. Mark-recapture data were collected from populations of the Shortnose Sturgeon in Georgia coastal rivers in 2004 – 2024. The dotted horizontal line corresponds to zero annual somatic growth. Fork length at initial capture corresponds to the measured fork length at tagging, and growth is derived from the most recent recapture of each individual.

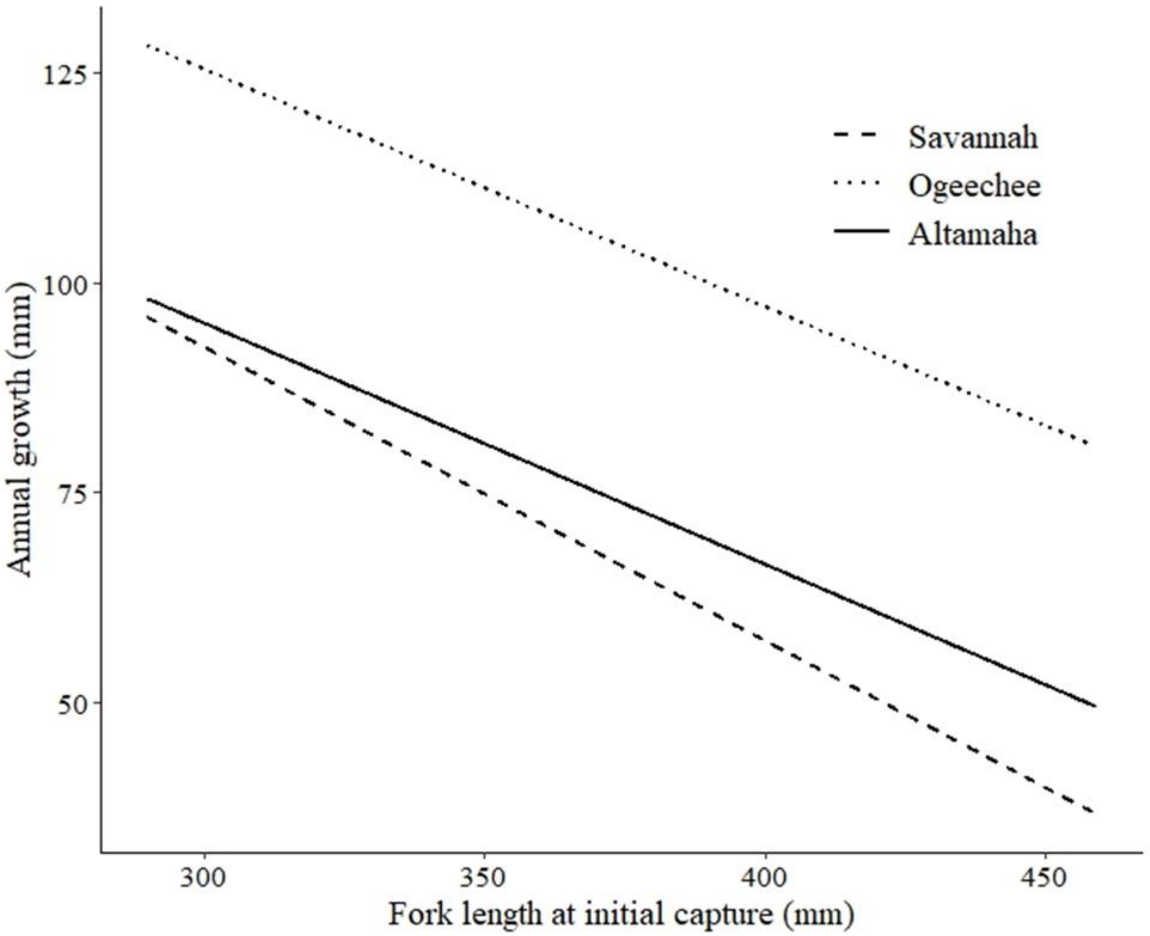

Figure 2.3. Regressions of annual growth increments of juvenile Shortnose Sturgeon (i.e., fish <500 mm fork length) derived from mark-recapture data obtained from fish in Georgia coastal river systems.

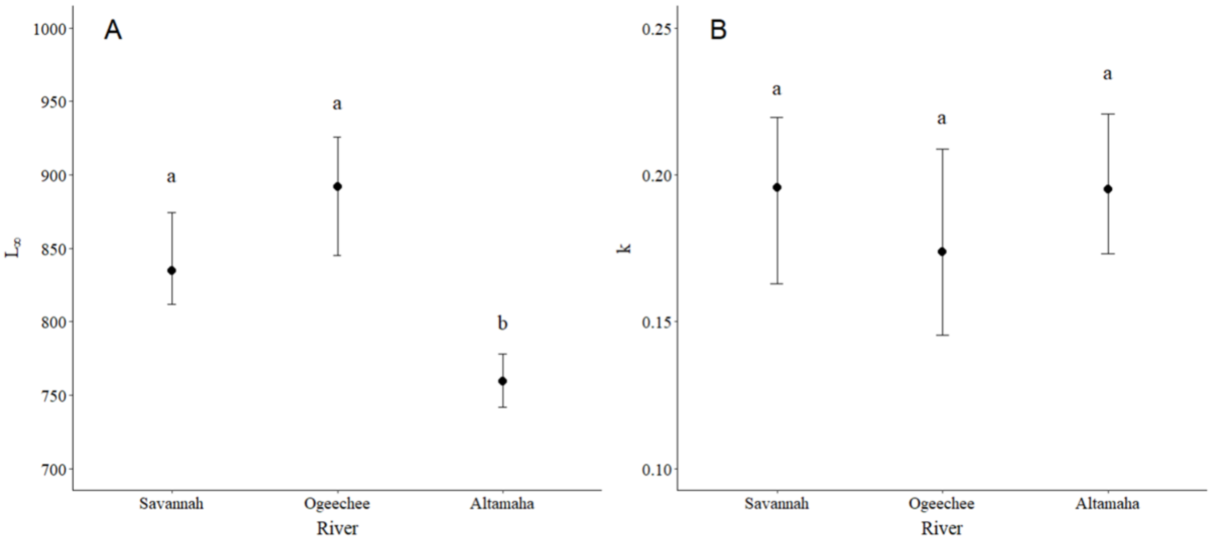

Figure 2.4. Growth parameters derived from the Fabens modification to the von Bertalanffy equation with mark-recapture data. Mark-recapture data were collected from individuals of the Shortnose Sturgeon in the Savannah, Ogeechee, and Altamaha Rivers, Georgia, from 2004 – 2024. (A) Comparison of the L_∞ von Bertalanffy parameter among the Savannah, Ogeechee, and Altamaha populations. The L_∞ parameter is indicative of the asymptotic fork length that individuals can attain within each population. (B) Comparison of the *k* parameter among the Savannah, Ogeechee, and Altamaha populations. The *k* parameter is indicative of the growth rate coefficient in the von Bertalanffy equation. Error bars correspond to 95% confidence intervals surrounding mean point estimates. Letters above each error bar are indicative of significant differences among populations, based on 95% confidence intervals.

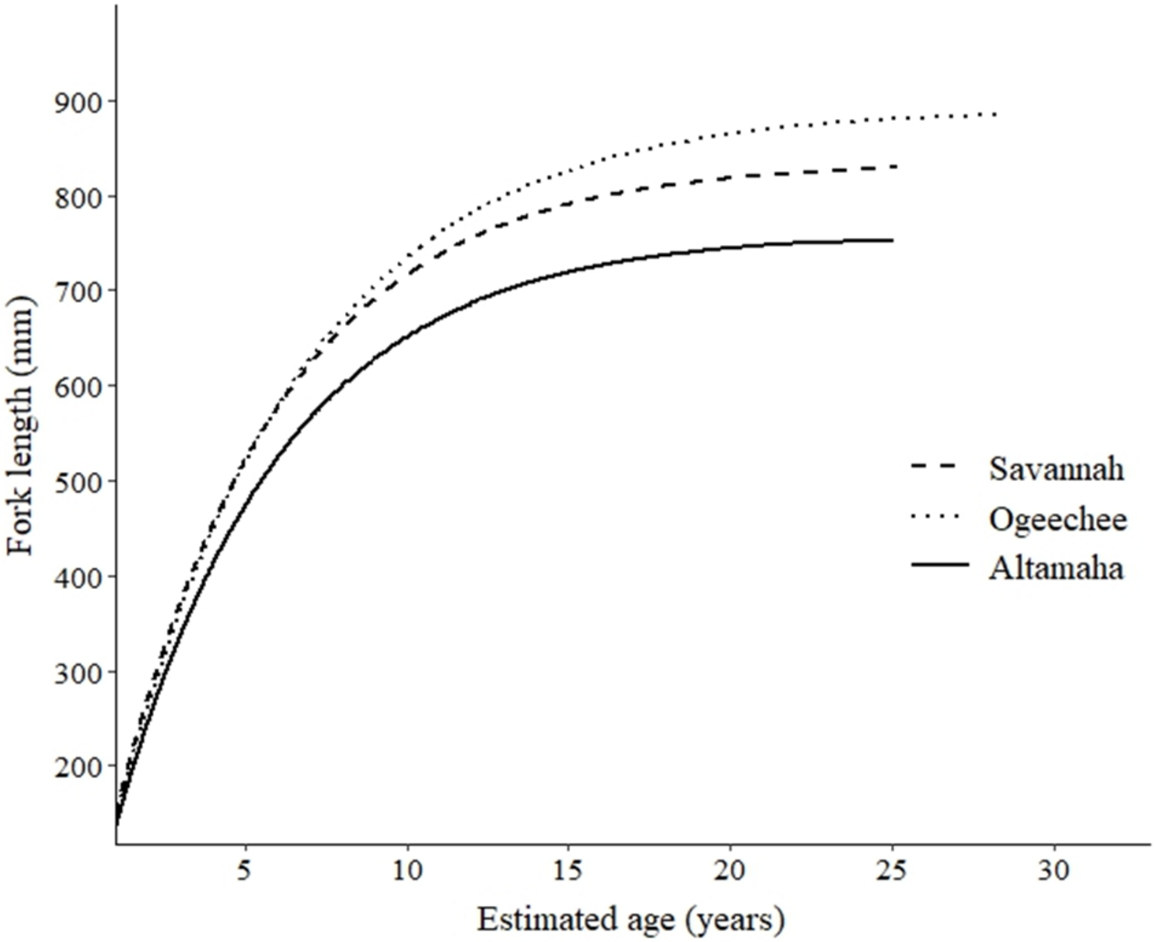

Figure 2.5. Mixed-sex growth curves displaying length-at-age relationships of the Shortnose Sturgeon developed from mark-recapture data obtained from fish in the Savannah, Ogeechee, and Altamaha Rivers from 2004 – 2024. Annual growth increment data were supplied to a modified von Bertalanffy growth curve that used the difference in growth between capture events and the time at large between capture events to derive parameters L_∞ and k. Estimated parameters were coupled with mark-recapture data to predict ages of Shortnose Sturgeon at any given fork length.

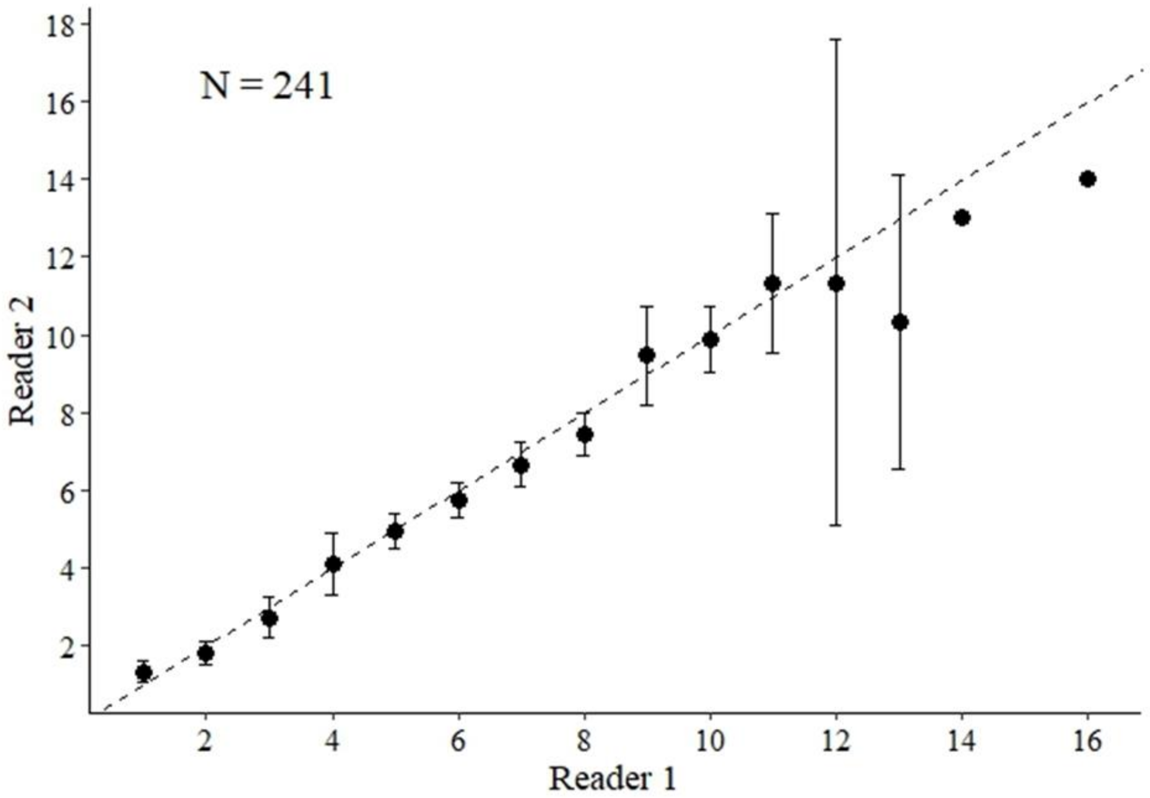

Figure 2.6. Age bias plot comparing independent reader age assignments estimated from Shortnose Sturgeon pectoral fin spines, collected from Georgia, USA coastal rivers from 2004 – 2024. The dotted line corresponds to a 1:1 equivalence line between reader estimates. The error bars correspond to 95% confidence intervals surrounding the mean age estimates of reader 2 for age estimates of reader 1.

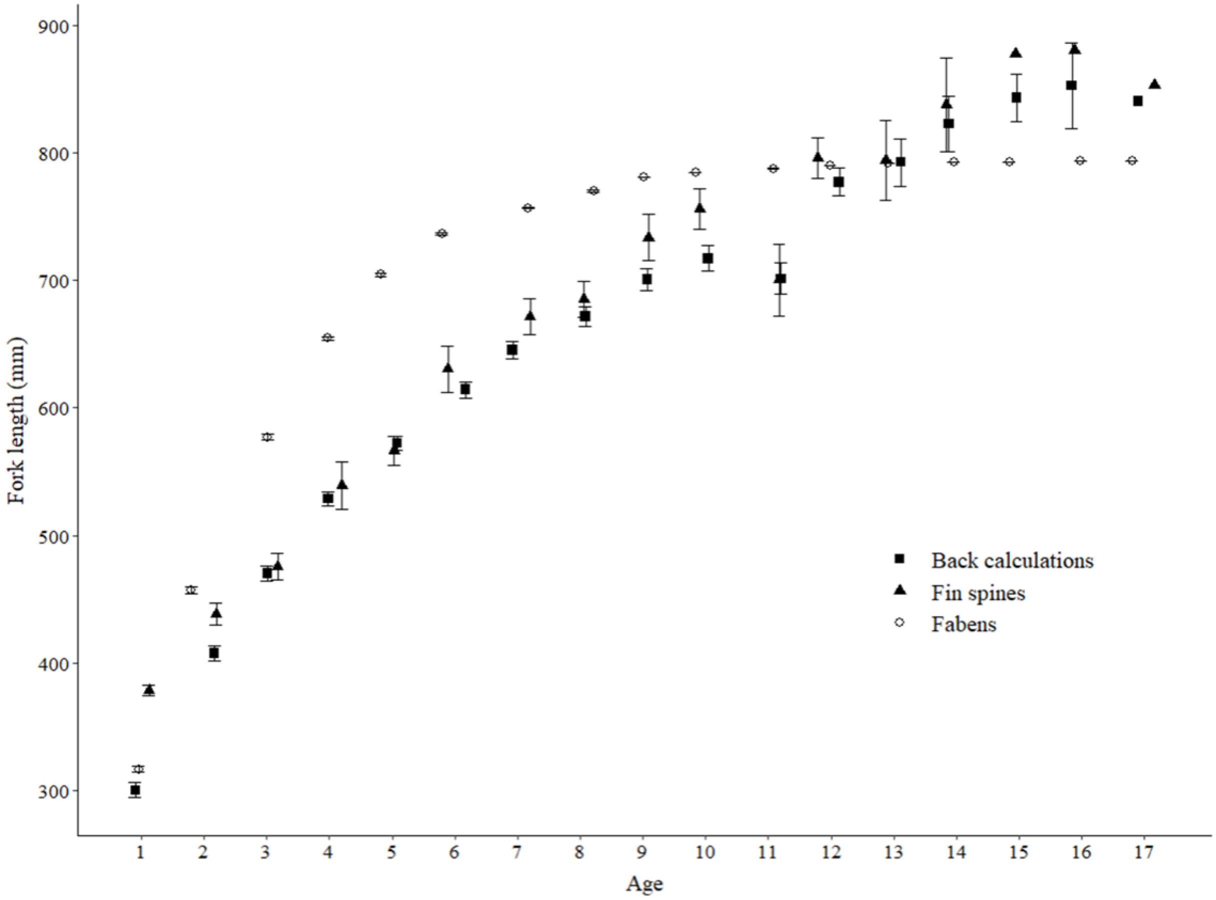

Figure 2.7. Comparison of mean fork length-at-ages, estimated from back-calculation methods of aged fin spines, consensus ages assigned to fin spines through a concert reading, and from mark-recapture data. Fin spines were collected from individuals of the Shortnose Sturgeon in the Savannah, Ogeechee, and Altamaha Rivers, Georgia, from 2023 – 2024. Mark-recapture data were collected from fish in the Savannah, Ogeechee, and Altamaha Rivers, Georgia, from 2004 – 2024. The error bars surrounding each estimate correspond to ± standard error of the mean. Points are jittered on the x-axis to improve figure readability.

CHAPTER 3

ASSESSMENT OF THE SHORTNOSE STURGEON POPULATION IN THE SAVANNAH RIVER, IN RELATION TO ESTUARINE DREDGING USING A BEFORE-AFTER CONTROL-IMPACT MODEL APPROACH[1]

[1] H. J. Rider, M. K. Morgan, A. T. Bond, J. D. Nolan, A. G. Fox, M. J. Hamel. To be submitted to *Environmental Biology of Fishes*.

Abstract

Estuaries are productive ecosystems that support diverse aquatic species and human economies through fisheries, recreation, and navigation. However, coastal development and dredging have altered the ecological processes of estuarine habitats. In the Savannah River estuary, the Savannah Harbor Expansion Project (SHEP) deepened the river's navigational channel by dredging to amplify vessel commerce. The endangered Shortnose Sturgeon *Acipenser brevirostrum* relies on estuarine habitat in the Savannah River to complete its life cycle; the SHEP dredging was predicted to decrease estuarine habitat availability for the local population of the Shortnose Sturgeon. To assess population trends in relation to the SHEP, we conducted mark-recapture sampling for the Shortnose Sturgeon in the Savannah River in the summers of 2013–2024 and used Huggins closed population capture-recapture models to estimate annual juvenile and adult abundances. We applied a before-after control-impact (BACI) model approach to compare changes in demographic-based abundances over time in the Savannah River to abundances in the geographically proximate, relatively unaltered Altamaha River. The BACI model indicated a significant decline in juvenile abundance in the Savannah River during and after the SHEP, while adult abundance remained unchanged. The timeframe of this decline, and the absence of a decline in the reference river suggest it may be associated with dredging via the SHEP. Juvenile declines may further threaten recovery of the Savannah River population. Continued population-level monitoring efforts and protection of the remaining estuarine habitat are critical for the long-term persistence of the Savannah River Shortnose Sturgeon population.

Introduction

Estuaries are physically complex environments made up of various habitats that support high biodiversity (Day Jr et al. 2013). For centuries, humans have also settled around coastal areas – including estuaries – because of access to abundant food sources, recreational activities, and commerce. Today, many estuaries are occupied by large ports, which are major drivers of economic growth (Crain et al. 2009, Day Jr et al. 2013). However, alteration of estuarine habitat by anthropogenic influences including urban development, agricultural processes, shipping ports, and dredging, has led to the degradation of estuarine habitats and subsequent losses of biodiversity (Goldberg 1995, Kennish 2002, Lotze et al. 2006, Le Pape et al. 2007).

Dredging is the deepening of navigational channels by excavating large volumes of sediments from aquatic environments (Carse & Lewis 2020). Although navigable waterways are essential to the economic growth of coastal regions (Carse & Lewis 2020), modifications by dredging can have concerning impacts on aquatic environments and their biodiversity, particularly fish (Wenger et al. 2017); dredging is deemed as one of the greatest threats to estuarine environments (Peris-Mora et al. 2005). Dredging procedures result in the direct removal of habitat (i.e., sediment, structure) for benthic fishes (Fischer et al. 2012, Freedman et al. 2013, Soinski et al. 2022), but can also remove individual fish through entrainment (Eleftheriou and Robertson 1992, Drabble 2012, Barletta et al. 2016). As a dredge removes estuarine substrate, it can also create sediment plumes in the water column (Wenger et al. 2017). Exposure to suspended sediments can result in higher mortality of fishes, particularly among early life stages (Auld and Schubel 1978, Griffin et al. 2009, Lowe et al. 2015), and the subsequent decreased recruitment to the adult population can potentially lead to long-term population declines (Wenger et al. 2017). Dredging and exposure to suspended sediment can also alter behavior, such as foraging patterns and success (Johnston and Wildish 1982, De Robertis et al. 2003, Wenger et al. 2012). The removal of sediment in estuaries through dredging can also allow farther penetration of the salt wedge in increased depths, increasing salinity and reducing dissolved oxygen (Johnston Jr 1981, Barletta et al. 2016). Thus, removal of sediments can influence surrounding abiotic factors and lead to additional changes in habitat and fish assemblages (Barletta et al. 2016).

The Savannah River of Georgia and South Carolina, USA, features a heavily industrialized estuary containing the second busiest container port on the U.S. Atlantic Coast, and the fourth busiest port in the nation (USACE 2012). The Savannah River estuary has been subject to anthropogenic alterations since the construction of a navigational channel in the late 19th century; continuous dredging has deepened the estuary by almost two-fold its original depth, from 6.6 m to 12.8 m (USACE 2012). The most recent harbor modification was the Savannah Harbor Expansion Plan (SHEP), in which dredging was conducted by the United States Army Corps of Engineers (USACE) in the estuary from September 2019–March 2022 (USACE 2025). The SHEP was a multi-year channel deepening modification to 60 river kilometers (rkm) of the estuary, which deepened the channel by approximately 1.5 m to allow larger cargo ships to access the container port (USACE 2012). Before the SHEP was initiated in 2015, environmental impact assessment models predicted that channel deepening would result in increased salinities and decreased dissolved oxygen concentrations in the entire estuary. These abiotic changes to the estuarine environment from the SHEP were expected to have harmful effects on fish populations that inhabit the estuary, particularly the endangered Shortnose Sturgeon *Acipenser brevirostrum* (USACE 2012).

The Shortnose Sturgeon is a long-lived, late-maturing fish that is found in coastal river systems on the U.S. Atlantic Coast (SSSRT 2010). The Shortnose Sturgeon exhibits an amphidromous life history strategy and uses freshwater and estuarine environments within its natal river system to complete its life cycle (Dadswell et al. 1984). Historically, populations occurred from the Saint Johns River, Canada, to the St. Johns River, Florida, USA (Dadswell et al. 1984); however, many populations were depleted or extirpated by habitat degradation and overexploitation, leading to the species being listed as endangered in 1967 (SSSRT 2010). The Shortnose Sturgeon is managed on a riverine-population level (SSSRT 2010). Many extant riverine populations have demonstrated little or no recovery since the species' listing because of continued habitat alteration, incidental bycatch, and other anthropogenic factors (Collins et al. 2000, SSSRT 2010).

The Savannah River is presumed to contain the second largest Shortnose Sturgeon population in the southern extent of the species' current range, with a 2013 estimate putting total population abundance as large as 2,432 individuals (Bahr and Peterson 2017). However, this population size is relatively small compared to populations inhabiting rivers at northern latitudes, as northern populations attain total population sizes in the tens of thousands (Kynard et al. 2016). The upper Savannah River estuary serves as nursery habitat for juveniles that cannot yet tolerate much salinity (Hall et al. 1991, Jenkins et al. 1993, Collins et al. 2002). Salinity tolerance is size- and temperature based; Once juveniles are large enough, they join adults in foraging in the estuary in fall and winter (Hall et al. 1991, Jenkins et al. 1993). The estuary also serves as a corridor for adults' amphidromous migrations into marine coastal environments and neighboring river systems (Hall et al. 1991, Kynard et al. 2016). In warmer months, both juveniles and adults will move upriver and aggregate in areas with little to no salinity in the freshwater/saltwater interface of the Savannah River (Hall et al. 1991, Ingram and Peterson 2018). The use of estuaries by the Shortnose Sturgeon makes the species a model organism for determining the potential influences that dredging can have on fish species that rely on estuarine environments.

The Savannah River estuary provides essential habitat for the completion of the Shortnose Sturgeon's life cycle, even though it may have already been degraded and reduced by previous dredging projects in the late 20th century. That dredging likely altered physical habitat characteristics and degraded water quality in the Savannah River estuary (Collins et al. 2002). The SHEP was anticipated to have similar adverse effects on the estuary, including increasing salinities and decreasing dissolved oxygen concentrations, which would likely result in additional habitat loss for both adults and juveniles of the Shortnose Sturgeon (USACE 2012). To mitigate one potential adverse effect, the USACE installed oxygen injection systems in the estuary to combat decreases in dissolved oxygen levels (USACE 2012, USACE 2025). However, this mitigation could not address the predicted SHEP-related increase in estuarine salinity (USACE 2012). Increases in salinity are likely to differentially impact juveniles in comparison to adults, as the Shortnose Sturgeon's tolerance to salinity increases with age and body size (Jenkins et al. 1993).

An important research objective for assessing recovery of the Shortnose Sturgeon is to understand interannual recruitment and abundance in extant populations (NMFS 1998). To quantify these population characteristics, University of Georgia researchers have been conducting a decade-long mark-recapture study on the Shortnose Sturgeon in coastal Georgia Estuaries, including the Savannah River and the nearby Altamaha River, Georgia (Peterson and Farrae 2011, Peterson and Bednarski 2013, Bahr and Peterson 2017, Cummins 2018, Fox et al. 2023, Kleinhans and Fox 2024). The objectives of this study were to 1) quantify interannual abundance of juveniles and adults in the Savannah River's population of the Shortnose Sturgeon over a 12-year period, 2) quantify those same population metrics in the Altamaha River (reference system), and 3) use a before-after, control-impact (BACI) modeling approach to determine potential changes in demographic-based abundances of the Savannah River Shortnose Sturgeon population over time in relation to the SHEP, using the Altamaha River as a reference.

Study Area

The Savannah River is an anthropogenically altered system that flows 484 km from its headwaters in the Blue Ridge Mountains to the Atlantic Ocean, forming much of the Georgia-South Carolina state border. In addition to the estuarine Port of Savannah, the Savannah River contains three large impoundments, with its lowest dam located at rkm 300 in Augusta, Georgia. This New Savannah Bluff Lock and Dam impedes sturgeon from reaching much of their historic spawning habitat in the Savannah River (SSSRT 2010). The lower Savannah River is tidally influenced up to 80 rkm upstream of the river's mouth, and the environment typically becomes brackish between rkm 33–38 (Hall et al. 1991) (Figure 3.1).

The Altamaha River estuary lies approximately 90 km south of the Savannah River estuary (Figure 3.1). It is formed by the confluence of its major tributaries, the Ocmulgee and Oconee Rivers, flowing 212 rkm from the Georgia Piedmont to the Atlantic Ocean. Although both tributaries are impounded above the fall line, the mainstem Altamaha River is undammed and the Shortnose Sturgeon population has access to nearly all its historic habitat (SSSRT 2010). The estuary of the Altamaha River is a tidally flooded salt marsh with little anthropogenic development, with the freshwater-saltwater interface occurring up to rkm 54 during normal flows

(Rogers & Weber 1994). The Altamaha River contains the largest quantified population of the Shortnose Sturgeon in the southern extent of the species' current range (Peterson and Bednarski 2013). We use this estuary as a reference in our study because we expect that both systems experience similar natural conditions (i.e., weather, seasonal patterns, environmental perturbations) due to their geographic proximity.

Methods

Sampling Sites

Sampling for this study was part of a long-term mark-recapture study of the Shortnose Sturgeon populations in Georgia conducted by researchers from the University of Georgia. That study has used consistent methods and relatively consistent sampling sites for well over a decade. Sampling of the Savannah River (Figure 3.1) occurred near the freshwater-saltwater interface (~35–50 rkm from the mouth) every summer from 2013–2024, providing a 12-year mark-recapture dataset. Comparatively, sampling of the Altamaha River estuary for the Shortnose Sturgeon also occurred every summer from 2013–2024. Sampling occurred approximately 10–30 rkm from the mouth; additional details about Altamaha River sampling locations can be found most recently in Kleinhans and Fox (2024). In both rivers, fish collection targeted low salinity reaches below the head of tide, as the Shortnose Sturgeon tends to aggregate there during the summer period (Hall et al. 1992, Collins et al. 2002, Ingram and Peterson 2018).

Data Collection

We deployed nets for sturgeon collection in the months of May–August. We sampled 3–5 days per week in each river, depending on weather and tide conditions. Sampling gear included anchored monofilament gill and trammel nets across the channel at slack high and low tides. Typically, we set one trammel net and two gill nets at each location; nets were deployed at sites with obstruction free bottoms within sturgeon aggregation areas. Nets soaked for 30–60 minutes. These methods have successfully captured Shortnose Sturgeon in Georgia rivers ((Peterson and Farrae 2011, Peterson and Bednarski 2013, Bahr and Peterson 2017, Ingram et al. 2020, Kleinhans and Fox 2024). We placed captured fish in floating net pens upon picking up nets to allow for recovery. Once all nets were retrieved, we collected morphometrics from each fish, including length (mm) and weight (g), and marked it with a passive integrated transponder tag (PIT tag) if a tag was not detected. We immediately released sturgeon at the location of their capture after the data collection process.

Data Analysis

Prior studies on populations of the Shortnose Sturgeon in Georgia have typically classified fish age by using either a fork length threshold of 500 mm to distinguish between juveniles and adults (Peterson and Bednarski 2013, Ingram et al. 2020) or a length frequency analysis for delineating age-based groups (Bahr and Peterson 2017, Cummins 2018, Kleinhans and Fox 2024). Length frequency analysis is useful for assigning individual fish to age-1 cohorts in the Shortnose Sturgeon, but this method breaks down for individuals older than age-1, as the distribution of length modes – representative of presumed age-classes – overlap due to differences in individual growth (Rider et al. 2025). Therefore, we used fork length to assign each fish as a juvenile (<450 mm fork length) or an adult (≥450 mm fork length). This 450 mm fork length allowed us to conservatively assign fish to a demographic group because fork length at sexual maturity across the species range is 450–550 mm, depending on sex (Dadswell et al. 1984, NMFS 1998).

After assigning individual fish to demographic groups, we used Huggins closed-population capture-recapture models (Huggins 1989, 1991) to separately estimate abundances of juveniles and adults in each river and study year. These Huggins models have previously been used to produce estimates of abundance and recruitment of populations of the Shortnose Sturgeon in Georgia (Peterson and Bednarski 2013, Bahr and Peterson 2017, Cummins 2018, Ingram et al. 2020, Kleinhans and Fox 2024). Although some estimates of abundance have been published for the Savannah River and Altamaha River, the length cutoffs between demographic groups for those prior studies differed from the 450 mm threshold between juveniles and adults that we used in this study (Savannah River: Bahr and Peterson 2017, Cummins 2018; Altamaha River: Kleinhans and Fox 2024). Therefore, the abundances estimated in this paper may differ slightly from those reported previously. Huggins models assume population closure, and prior studies have shown that populations of the Shortnose Sturgeon meet this assumption during summer months in southern rivers, when movement of fish is limited (Hall et al. 1991, Peterson and Bednarski 2013, Bahr and Peterson 2017, Ingram and Peterson 2018). We constructed capture histories for each individual fish using 1-week sampling periods to permit fish to randomly mix with unmarked fish before potentially being recaptured in later sampling events. A set of candidate models was developed to account for variation in capture probability: constant capture probability among individuals and sampling periods (M_0), capture probability that varied with sampling period (M_t), capture probability that varied with group (juvenile/adult) (M_g), capture probability that varied with sampling period and group (M_{t+g}), and capture probability that varied with sampling period, group, and the interaction between period and group (M_{txg}). Candidate models were fit in the software package RMark (Laake 2013) and evaluated with Akaike's information criterion corrected for small sample sizes (AICc) to analyze and compare model performances within years (Akaike 1973, Hurvich and Tsai 1989). The candidate model with the highest Akaike weight (W) for a given year was selected for evaluating juvenile and abundance estimates.

To evaluate the effects of the SHEP on the Savannah River population of the Shortnose Sturgeon, we used BACI models, which use concurrent data from a reference (control) that has not been subject to disruption as a comparison to data from a treatment (impacted) that has experienced a disruption to assess how the treatment has potentially responded to the disruption over the same time (Underwood 1991, Smith 2002). The reference accounts for natural variability that may influence the dependent variable of both the reference and treatment over the duration of the study (Smith 2002). This model also accounts for initial differences in the dependent variable between the reference and the treatment before the disruption occurs. Therefore, if the disruption causes a change in the dependent variable of the treatment, the model will detect a difference in the dependent variable between the reference and treatment after the disruption has occurred compared to the initial differences before the disruption (Underwood 1991, Shoup and Carl 2023). To effectively detect a temporal difference in the dependent variable, a BACI requires multiple samples in the “after” period from both the treatment and control systems (Underwood 1991). In our study, the treatment was the Savannah River, which had been affected by dredging, and the un-dredged Altamaha River served as the reference system. The dependent variables in our study were the annual point estimates of juvenile and adult abundance of the Shortnose Sturgeon in each river. Our analysis featured 7 years of mark-recapture data representative before the SHEP (the summers of 2013–2019) and 5 years of data during and after the SHEP (the summers of 2020–2024). Juvenile and adult abundance estimates were evaluated in separate linear models, and abundance estimates were log-transformed to meet model assumptions of normality and homoscedasticity. The linear model included time (before-after), river (Altamaha-Savannah), and the time × river interaction as fixed effects. Potential significance ($P < 0.05$) detected in the BACI interaction suggests that the abundance of the impacted population is changing in a different way than the reference population. Because both rivers are likely experiencing the same general environmental conditions, such a difference may be a result of the SHEP. All data analyses and visualizations were performed in Program R version 4.3.1 (R Core Team 2023).

Results

Over the 12-year period of study, we set a total of 2,701 nets (mean = 209 nets deployed/year) in the Savannah River, yielding a total of 1,796 hours of netting effort (range = 53–238 h of effort/year; mean = 150 h) (Table 3.1). We captured a total of 2,084 Shortnose Sturgeon in the Savannah River across all years of the study (range = 79–280 total captures/year; mean = 174) of which 99 of those captured fish were recaptured (5%) (Table 3.1). In the Altamaha River, we deployed a total of 2,013 nets (mean = 181 nets deployed/year) over the duration of the study, and the total netting effort was 1,663 hours (range = 44–284 h of effort/year; mean = 151 h). Nets were not set for targeting the Shortnose Sturgeon in 2013 in the Altamaha River (Table 3.1). We captured a total of 1,601 Shortnose Sturgeon in the Altamaha River (range = 43–298 total captures/year; mean = 146), of which 55 of those captured fish were recaptured (3%) (Table 3.1).

The top Huggins closed capture model, based on Akaike weight, varied among years for each study river, (Table 3.2; Table 3.3). In the Savannah River, the time covariate was included in the top model in every year for which an estimate of abundance could be calculated. The time-varying model (M_t) best fit the data in 7 years, and the time and group additive model (M_{t+g}) or interactive model ($M_{t\times g}$) best fit the data in the other 4 years for the Savannah River (Table 3.2). There was 1 year (i.e., 2020) for which we were unable to derive estimates of abundance in the Savannah River. In the Altamaha River, the time covariate was included in the top model in every year for which an estimate could be produced. Model M_t had the highest Akaike weight and was selected for 6 years, model $M_{t\times g}$ was selected for 2 years, and model M_{t+g} was selected for 1 year for the Altamaha River (Table 3). There were 2 years (i.e., 2013, 2014) for which we were unable to derive estimates for the Altamaha River.

Over the study period in the Savannah River, point estimates of juvenile abundance varied from a minimum of 15 juveniles in both 2023 and 2024 (2023 95% CI = 8–49; 2024 = 5–58) to a maximum of 603 juveniles in 2018 (95% CI = 347–1,084) (Figure 3.2; Table 3.4). Adult abundance estimates in the Savannah River varied from a minimum of 634 adults in 2021 (95% CI = 357 – 1,217) to a maximum of 3,886 adults in 2019 (95% CI = 1,184–13,511) (Figure 3.3; Table 3.5). We could not calculate reasonable estimates of juvenile abundance in 2021 in the Savannah River because not enough juveniles were recaptured in that sampling season (Figure 3.2; Table 3.1). For the Altamaha River, estimates of juvenile abundance varied from a minimum of 31 juveniles (95% CI = 8–132) in 2018 to a maximum of 1,000 juveniles in 2020 (95% CI = 406–2,576) (Figure 3.2; Table 3.4). Adult abundance estimates in the Altamaha River varied from a minimum of 249 adults (95% CI = 63–1,267) in 2015 to a maximum of 3,575 adults (95% CI = 1,821–7,271) in 2017 (Figure 3.3; Table 3.5). We could not calculate abundance estimates in the Altamaha River in 2013 and 2014 because there were either no capture events or recapture events in those sampling seasons (Figure 3.2, 3.3; Table 3.1). Furthermore, we could not calculate reasonable estimates of juvenile abundance in 2021 or adult abundance in 2024 for the Altamaha River because there were not enough recaptures.

The BACI model results noted a significant interaction ($P < 0.01$) between time (before-after) and river (Altamaha-Savannah) when evaluating juvenile abundance estimates (Table 3.6). This interaction indicated that a significant decline occurred in juvenile abundance from the before period to the after period in the Savannah River, whereas significant temporal change in juvenile abundance was not observed in the Altamaha River (Table 3.6). The results of the BACI model presented below indicate log-transformed estimates, where "Altamaha before" represents the intercept (i.e., baseline juvenile abundance in the Altamaha River from 2013 – 2019), and other noted effects represent the difference in juvenile abundance from the intercept. Before the SHEP, juvenile abundance was not significantly different between rivers (Savannah before = 5.79, SE = 0.47; Altamaha before = 5.13, SE = 0.36; $P = 0.18$; Table 3.6), as estimated abundance was slightly higher in the Savannah River (Figure 3.4). In the period of the SHEP's progress and following completion of dredging, juvenile abundance significantly declined in the Savannah River (interaction = 1.58, SE = 0.77, $P < 0.01$), while the Altamaha River showed a non-significant increase (Altamaha after = 5.84, SE = 0.54, $P = 0.21$; Table 3.6) in juvenile abundance (Figure 3.4). Similar temporal estimates of juvenile abundance in the Altamaha River indicate that factors other than natural variability contributed to the decline in juvenile abundance over time in the Savannah River.

The BACI model results did not indicate a significant interaction ($P = 0.38$) between the model effects of time and river when evaluating adult abundance estimates (Table 3.6). Before the SHEP, adult abundances were not significantly different between the Altamaha and Savannah Rivers (Altamaha before = 7.43, SE = 0.32; Savannah before = 7.54, SE = 0.42; $P = 0.8$; Table

3.6), with adult abundance in the Savannah River being slightly greater (Figure 3.5). In the period of the SHEP's progression and following completion of dredging, a non-significant temporal change in adult abundance was observed in both the Altamaha River (Altamaha after = 7.40, SE = 0.48, $P = 0.95$) and the Savannah River (Interaction = 6.84, SE = 0.66, $P = 0.38$; Table 3.6). Although the model did not indicate a significant change in adult abundance over time, in both study rivers adult abundance declined from the 'before' period to the 'after' period (Figure 3.5).

Discussion

Our study used a long-term mark-recapture dataset of the Shortnose Sturgeon to produce interannual abundance estimates for juveniles and adults for two extant populations in Georgia coastal rivers. Using the BACI model approach with our point abundance estimates for juveniles and adults, we documented that, compared to the nearby Altamaha River, there was a significant decline in juvenile abundance of the Savannah River Shortnose Sturgeon population that occurred in conjunction with and after the channel deepening via the SHEP. Although we did not investigate mechanisms for the decline of juveniles in the Savannah River, the fact that there was not a similar decline in the Altamaha River suggests that this is not a regional phenomenon. The SHEP-related major habitat alterations to the Savannah River are a parsimonious explanation for the observed juvenile population in that estuary. Adult abundance did not significantly change in the Savannah River over time in comparison to the adult abundance in the reference Altamaha River. However, adult abundance in the Savannah River declined during and after the SHEP; this could potentially be attributed to fewer juvenile fish surviving to maturity because of perturbations caused by the SHEP. However, adult abundance in the Altamaha River also declined over the same time frame. Therefore, there may be other unknown mechanisms influencing survival of adult contingents of the Shortnose Sturgeon at the regional level.

Although dredging-related mitigation efforts were implemented in the Savannah River to decrease potential habitat loss of the Shortnose Sturgeon, additional environmental stressors persist that potentially amplify threats to the Savannah River Shortnose Sturgeon population. Although oxygen injectors were implemented in the estuary to mitigate predicted water quality degradation from the SHEP, suitable estuarine habitat was still predicted to decrease for adult and juvenile Shortnose Sturgeon because of increased salinity levels in the estuary (USACE 2012). Our study did not measure the environmental variables of salinity and dissolved oxygen concentrations from the SHEP that could influence abundance of the Savannah River Shortnose Sturgeon population. However, salinity tolerance in the Shortnose Sturgeon increases with size and age (Jenkins et al. 1993). Therefore, smaller juvenile fish (i.e., young of year) that are exposed to higher salinity levels over time will likely experience increased mortality relative to larger juveniles (i.e., age-1+) and adults. Older juvenile fish may not experience a direct increase in mortality from increased salinity alone relative to young-of-year fish; however, prior research on the movement and habitat use of juvenile fish in the Savannah River suggested that previous dredging projects in the system's estuary caused juveniles to abandon estuarine habitat that they used before dredging (Collins et al. 2002). In abandoning previous nursery habitat, juvenile fish moved upriver into potentially less suitable habitat that experienced water quality degradation from habitat alteration, exposing individuals to additional environmental stressors with a decrease in availability for foraging and environmental refuge (Collins et al. 2002). Additionally, less suitable habitat can increase intraspecific competition (Dadswell et al. 1984, Ross 1997), potentially increasing density-dependent mortality in juvenile fish (Hixon and Jones 2005, Lorenzen and Camp 2019). Furthermore, a potential loss in refuge habitat may limit juvenile fish survival during periods of environmental stress, as southern populations experience warm waters (i.e., >30°C) for multiple months in the summer (SSSRT 2010). Because tolerance to water temperature increases with body size in the Shortnose Sturgeon (Ziegeweid et al. 2008), increasing water temperatures from climate change will be more detrimental to juvenile survival than adults (Collins et al. 2000, Kynard et al. 2016). Future research identifying current nursery habitat in the Savannah River is crucial for mitigating any further habitat loss that could arise from future anthropogenic influences or a warming climate (SSSRT 2010). The determination of preferred physical habitat characteristics and resource availability in current nursery habitat can aid in projecting how further habitat alteration may influence juvenile characteristics such as behavior and survival in the Savannah River moving forwards. Furthermore, such information can be used to implement effective mitigation plans in wake of further anthropogenic alteration. Our results suggest that further loss of nursery habitat used by juvenile fish may pose a major threat to juvenile survival, which could cause bottlenecks in recruitment and threaten the long-term persistence of the Savannah River Shortnose Sturgeon population.

The benthic nature of the Shortnose Sturgeon increases the likelihood for individuals to experience direct mortality from dredging operations while inhabiting the estuary of the Savannah River (Dadswell et al. 1984, Collins et al. 2002). Adult and juvenile fish can potentially be entrained by dredging equipment when inhabiting the channelized portion of the Savannah River estuary, or areas that are dredged in the estuary (Reine et al. 2014, Chapman et al. 2019). Entrainment is the direct uptake of an aquatic organism by a suction field that occurs from dredging machinery during operation (Reine et al. 2014), and sturgeon can be potentially entrained if individuals cannot exhibit the capacity to swim out of the dredging machinery's suction field (Boysen and Hoover 2009, Hoover et al. 2011). In addition to dredging operations, potential vessel interactions are another poseable threat to the Shortnose Sturgeon. Fish using the navigational channel in the Savannah River estuary will be vulnerable to vessel strikes, and vessel strikes can result in injury or mortality to fish inhabiting navigational channels, with larger, adult individuals being more susceptible to strikes (Brown and Murphy 2010, Balazik et al. 2012, Hondorp et al. 2017, Demetras et al. 2020). Although information is currently lacking on the movement rate and habitat that the Shortnose Sturgeon may use in the Savannah River estuary, these mortality related threats that fish may be exposed to in the Savannah River can further impede recovery of southern populations of the Shortnose Sturgeon (SSSRT 2010).

We used the Altamaha River as a control in this study because of its geographic proximity to the Savannah River, and because the Altamaha River is anthropogenically unaltered relative to the Savannah River. Therefore, variations in interannual abundance in the Altamaha River are likely due to potential "natural" changes (e.g., climate and hydrology) rather than an anthropogenic perturbation. Although the Altamaha River Shortnose Sturgeon population may be experiencing potential environmental stressors (e.g., climate change; SSSRT 2010), its geographic proximity to the Savannah River would suggest that both populations would experience similar effects. Future studies quantifying specific environmental, hydrological, and limnological attributes in each system may help to identify river-specific attributes that may contribute to changes in abundances of the Shortnose Sturgeon.

There were a few years in our study in which abundance estimates could not be quantified for juveniles or adults in either study river with the Huggins closed population capture-recapture models, as those years lacked captures or recaptures to produce viable estimates. This includes two years in the Savannah River during the 'after' time frame that included the SHEP's progression and completion. Thus, our BACI model approach sometimes produced differing results if those missing years of juvenile abundance estimates could have been quantified, considering the number of juveniles that were initially captured in the Savannah River in 2020 and 2021. Furthermore, in our BACI model approach, we used point estimates of abundance as the response variable, despite potentially large uncertainties (i.e., 95% confidence intervals) (Figures 3.2, 3.3). Regardless of this potential uncertainty, we consistently saw lower Savannah River juvenile abundances every year after the SHEP, suggesting that a consistent disturbance is likely disrupting the river's population of the Shortnose Sturgeon (Figure 3.2; Table 3.1).

The Huggins closed population capture recapture model that we used to quantify abundance relies on an assumption that the study area is closed to immigration and emigration during sampling. Movement of the Shortnose Sturgeon during our sampling period is limited in southern rivers, as fish aggregate in tidally influenced holding areas with little to no saltwater intrusion to alleviate environmental stressors that occur in summer months (Hall et al. 1991, Collins et al. 2002, Ingram and Peterson 2018). Furthermore, prior studies assessing population abundances in coastal rivers of Georgia featured effort that was evenly distributed throughout suspected summer holding areas as well as habitat above and below these areas. Efforts outside of the suspected summer holding areas did not yield captures during the summer sampling period (Altamaha River: Peterson and Bednarski 2013; Savannah River: Bahr and Peterson 2017), providing additional evidence that our targeted populations of the Shortnose Sturgeon are closed during summer months. However, a recent telemetry study found that adults of the Shortnose Sturgeon may make frequent inter-river migrations – but that those migrations occurred outside our summer sampling window (Nolan et al. *In Review*). Therefore, we acknowledge that the adult abundances we quantified may include some non-natal individuals and may exclude some natal individuals. Thus, more research is needed to better identify meta-population dynamics by quantifying the proportion of migratory adult fish relative to river-resident individuals among southern populations of the Shortnose Sturgeon. Because of these inter-river movements, anthropogenic threats within the Savannah River may pose threats to other populations of the Shortnose Sturgeon.

Establishing multi-year abundance estimates of the Shortnose Sturgeon in each riverine population is an essential research objective in the species recovery plan to identify how populations at the riverine level respond to stressors or potential recovery actions over time (NMFS 1998). Our results demonstrate the value of long-term monitoring among multiple river systems in evaluating changes to systems that could negatively affect fish communities. In our assessment of how the Savannah River population of the Shortnose Sturgeon population changed during the SHEP, we needed not only long-term data from Savannah River, but also similar data from the Altamaha River for comparison. For the Shortnose Sturgeon, continuing assessments among Georgia's coastal rivers will serve as an important tool for evaluating perturbations to systems, but also for implementing and evaluating management actions.

References

Akaike, H. 1973. Information theory and an extension of the maximum likelihood principle. Pages 267–281 in B. N. Petrov and F. Csaki, editors. Second international symposium on information theory. Akademiai Kiado, Budapest.

Auld, A. H., and J. R. Schubel. 1978. Effects of suspended sediment on fish eggs and larvae: a laboratory assessment. Estuarine and Coastal Marine Science 6(2):153-164.

Bahr, D. L. and D. L. Peterson. 2017. Status of the Shortnose Sturgeon population in the Savannah River, Georgia. Transactions of the American Fisheries Society 146(1):92-98.

Balazik, M. T., G. C. Garman, J. P. Van Eenennaam, J. Mohler, and L. C. Woods III. 2012. Empirical evidence of fall spawning by Atlantic Sturgeon in the James River, Virginia. Transactions of the American Fisheries Society 141(6):1465-1471.

Barletta, M., F. J. Cysneiros, and A. R. Lima. 2016. Effects of dredging operations on the demersal fish fauna of a South American tropical-subtropical transition estuary. Journal of Fish Biology 89(1):890-920.

Boysen, K. A., and J. J. Hoover. 2009. Swimming performance of juvenile white sturgeon (*Acipenser transmontanus*): training and the probability of entrainment due to dredging. Journal of Applied Ichthyology 25:54-59.

Brown, J. J., and G. W. Murphy. 2010. Atlantic Sturgeon Vessel-Strike Mortalities in the Delaware Estuary. Fisheries 35(2):72-83.

Carse, A., and J. A. Lewis. 2020. New horizons for dredging research: The ecology and politics of harbor deepening in the southeastern United States. Wiley Interdisciplinary Reviews: Water 7(6):e1485.

Chapman, E. D., E. A. Miller, G. P. Singer, A. R. Hearn, M. J. Thomas, W. N. Brostoff, P. E. LaCivita, and A. P. Klimley. 2019. Spatiotemporal occurrence of green sturgeon at dredging and placement sites in the San Francisco estuary. Environmental Biology of Fishes 102(1):27-40.

Collins, M. R., S. G. Rogers, T. I. Smith, and M. L. Moser. 2000. Primary factors affecting sturgeon populations in the southeastern United States: fishing mortality and degradation of essential habitats. Bulletin of Marine Science 66(3):917-928.

Collins, M. R., W. C. Post, D. C. Russ, and T. I. J. Smith. 2002. Habitat use and movements of juvenile Shortnose Sturgeon in the Savannah River, Georgia-South Carolina. Transactions of the American Fisheries Society 131(5):975-979.

Crain, C. M., B. S. Halpern, M. W. Beck, and C. V. Kappel. 2009. Understanding and managing human threats to the coastal marine environment. Annals of the New York Academy of Science 1162:39-62.

Cummins, A. J. 2018. Recruitment of Atlantic Sturgeon and Shortnose Sturgeon within the Savannah River, Georgia and South Carolina. Master's thesis. University of Georgia, Athens, USA.

Dadswell, M. J., B. D. Taubert, T. S. Squiers, D. Marchette, and J. Buckley. 1984. Synopsis of biological data on Shortnose Sturgeon, *Acipenser brevirostrum* LeSueur 1818. NOAA Technical Report NMFS 14.

Day Jr, J. W., W. M. Kemp, A. Yáñez-Arancibia, and B.C. Crump. editors. 2013. Estuarine ecology, 2nd edition. John Wiley and Sons, Hoboken, New Jersey.

Demetras, N. J., B. A. Helwig, and A. S. McHuron. 2020. Reported vessel strike as a source of mortality of White Sturgeon in San Francisco Bay. California Fish and Wildlife 106(1):59-65.

De Robertis, A., C. H. Ryer, A. Veloza, and R. D. Brodeur. 2003. Differential effects of turbidity on prey consumption of piscivorous and planktivorous fish. Canadian Journal of Fisheries and Aquatic Sciences 60(12):1517-1526.

Drabble, R. 2012. Monitoring of East Channel dredge areas benthic fish population and its implications. Marine Pollution Bulletin 64(2):363-372.

Eleftheriou, A., and M. R. Robertson. 1992. The effects of experimental scallop dredging on the fauna and physical environment of a shallow sandy community. Netherlands Journal of Sea Research 30:289-299.

Fischer, J., C. Paukert, and M. Daniels. 2012. Fish community response to habitat alteration: impacts of sand dredging in the Kansas River. Transactions of the American Fisheries Society 141(6):1532-1544.

Fox, A. G., M. A. Baker, A. J. Cummins, H. S. Evans Jr, K. L. Cummins, N. Q. Hancock, and D. L. Higginbotham. 2023. Recruitment of juvenile Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*) in the Savannah, Ogeechee, and Satilla Rivers in Georgia. Fishery Bulletin 121(3):129-140.

Freedman, J. A., R. F. Carline, and J. R. Stauffer. 2013. Gravel dredging alters diversity and structure of riverine fish assemblages. Freshwater Biology 58(2):261-274.

Goldberg, E. D. 1995. Emerging problems in the coastal zone for the twenty-first century. Marine Pollution Bulletin 31(4-12):152-158.

Griffin, F. J., E. H. Smith, C. A. Vines, and G. N. Cherr. 2009. Impacts of suspended sediments on fertilization, embryonic development, and early larval life stages of the Pacific herring, *Clupea pallasi*. The Biological Bulletin 216(2):175-187.

Hall, J. W., T. I. Smith, and S. D. Lamprecht. 1991. Movements and habitats of shortnose sturgeon, *Acipenser brevirostrum* in the Savannah River. Copeia 695-702.

Hixon, M. A., and G. P. Jones. 2005. Competition, Predation, and Density-Dependent Mortality in Demersal Marine Fishes. Ecology 86(11):2847-2859.

Hondorp, D. W., D. H. Bennion, E. F. Roseman, C. M. Holbrook, J. C. Boase, J. A. Chiotti, M. V. Thomas, T. C. Wills, R. G. Drouin, S. T. Kessel, and C. C. Krueger. 2017. Use of navigation channels by Lake Sturgeon: Does channelization increase vulnerability of fish to ship strikes? PLoS One, 12(7):e0179791.

Hoover, J. J., K. A. Boysen, J. A. Beard, and H. Smith. 2011. Assessing the risk of entrainment by cutterhead dredges to juvenile lake sturgeon (*Acipenser fulvescens*) and juvenile pallid sturgeon (*Scaphirhynchus albus*). Journal of Applied Ichthyology 27(2):369-375.

Huggins, R. M. 1989. On the statistical analysis of capture experiments. Biometrika 76(1):133–140.

Huggins, R. M. 1991. Some practical aspects of a conditional likelihood approach to capture experiments. Biometrics 47(2):725– 732.

Hurvich, C. M., and C. Tsai. 1989. Regression and time series selection in small samples. Biometrika 76:297–307.

Ingram, E. C., and D. L. Peterson. 2018. Seasonal movements of shortnose sturgeon (*Acipenser brevirostrum*) in the Altamaha River, Georgia. River Research and Applications 34(7):873-882.

Ingram, E. C., D. L. Peterson, and A. G. Fox. 2020. Abundance of endangered shortnose sturgeon (*Acipenser brevirostrum*) in the Altamaha River in Georgia. Fishery Bulletin 118(2):198-204.

Jenkins, W. E., T. I. Smith, L. D. Heyward, and D. M. Knott. 1993. Tolerance of shortnose sturgeon, *Acipenser brevirostrum*, juveniles to different salinity and dissolved oxygen concentrations. In Proceedings of the Annual Conference of the Southeastern Association of Fish and Wildlife Agencies 47:476-484.

Johnston Jr, S. A. 1981. Estuarine dredge and fill activities: a review of impacts. Environmental management 5(5):427-440.

Johnston, D. D., and D. J. Wildish. 1982. Effect of suspended sediment on feeding by larval herring (*Clupea harengus harengus* L.). Bulletin of Environmental Contamination and Toxicology 29:261-267.

Kennish, M. J. 2002. Environmental threats and environmental future of estuaries. Environmental Conservation 29(1):78-107.

Kleinhans, M., and A. G. Fox. 2024. Long-term trends in abundance and recruitment of Shortnose Sturgeon in the Altamaha River, Georgia. Marine and Coastal Fisheries 16(3):e10294.

Kynard, B., S. Bolden, M. Kieffer, M. Collins, H. Brundage, E. J. Hilton, M. Litvak, M. T. Kinnison, T. King, and D. Peterson. 2016. Life history and status of Shortnose Sturgeon (*Acipenser brevirostrum* LeSueur, 1818). Journal of Applied Ichthyology 32:208-248.

Laake, J. L. 2013. RMark: An R interface for analysis of capture-recapture data with MARK (Processed report 2013-01). National Marine Fisheries Service, Alaska Fisheries Science Center.

Le Pape, O., C. Gilliers, P. Riou, J. Morin, R. Amara, and Y. Désaunay. 2007. Convergent signs of degradation in both the capacity and the quality of an essential fish habitat: state of the Seine estuary (France) flatfish nurseries. Hydrobiologia 588(1):225-229.

Lorenzen, K., and E. V. Camp. 2019. Density-dependence in the life history of fishes: When is a fish recruited? Fisheries Research 217:5-10.

Lotze, H. K., H. S. Lenihan, B. J. Bourque, R. H. Bradbury, R. G. Cooke, M. C. Kay, S. M. Kidwell, M. X. Kirby, C. H. Peterson, and J. B. C. Jackson. 2006. Depletion, degradation, and recovery potential of estuaries and coastal seas. Science 312(5781):1806-1809.

Lowe, M. L., M. A. Morrison, and R. B. Taylor. 2015. Harmful effects of sediment-induced turbidity on juvenile fish in estuaries. Marine Ecology Progress Series 539:241-254.

National Marine Fisheries Service (NMFS). 1998. Final recovery plan for the Shortnose Sturgeon, *Acipenser brevirostrum*. National Marine Fisheries Service.

Nolan, J. D., W. C. Post, E. Waldrop, J. J. Facendola, M. S. Loeffler, W. Collier, E. C. Ingram, M. J. Hamel, A. G. Fox. *In Review*. Novel coastal migration capabilities of an endangered amphidromous fish, the Shortnose Sturgeon (*Acipenser brevirostrum*). Journal of Fish Biology.

Peris-Mora, E., J. M. Diez Orejas, A. Subirats, S. Ibanez, and P. Alvarez. 2005. Development of a system of indicators for sustainable port management. Marine Pollution Bulletin 50(12):1649-1660.

Peterson D. L., and D. J. Farrae. 2011. Evidence of metapopulation dynamics in Shortnose Sturgeon in the southern part of their range. Transactions of the American Fisheries Society 140(6):1540–1546.

Peterson, D. L., and M. S. Bednarski. 2013. Abundance and Size Structure of Shortnose Sturgeon in the Altamaha River, Georgia. Transactions of the American Fisheries Society 142(5):1444-1452.

R Core Team. 2023. R: A language and environment for statistical computing. Version 4.3.1. R Foundation for Statistical Computing, Vienna, Austria.

Reine K., D. Clarke, M. Balazik, S. O'Haire, C. Dickerson, C. Frederickson, G. Garman, C. Hager, A. Spells, and C. Turner. 2014. Assessing impacts of navigation dredging on

Atlantic sturgeon (*Acipenser oxyrinchus*). U.S. Army Corps of Engineers, Washington, D.C. Report # ERDC/EL TR-14–12.

Rider, H. R., M. K. Morgan, A. T. Bond, J. D. Nolan, A. G. Fox, M. J. Hamel. 2025. Validating Age Estimates from Pectoral Fin Spines and Length-Frequency Analysis of Known-Age Shortnose Sturgeon. Journal of the Southeastern Association of Fish and Wildlife Agencies 12:52-60.

Rogers, S., and W. Weber. 1994. Movements of shortnose sturgeon in the Altamaha River system, Georgia. Georgia Department of Natural Resources, Coastal Resources Division, Marine Fisheries Section.

Ross, M. R. 1997. Fisheries conservation and management. Prentice Hall, Upper Saddle River, New Jersey.

Shortnose Sturgeon Status Review Team (SSSRT). 2010. A biological assessment of shortnose sturgeon (*Acipenser brevirostrum*). Report to National Marine Fisheries Service, Northeast Regional Office.

Shoup, D. E., and D. D. Carl. 2023. Long-Term Standardized Sampling as a Tool for Evaluating Management Actions: BACI Analysis to Reevaluate Predatory Control of Stunted Crappie with Saugeye. Fisheries 48(9):377-386.

Smith, E. P. 2002. BACI design. Encyclopedia of environmetrics 1:141-148.

Soinski, T. A., C. d. S. Gato, A. A. Matsumoto, M. L. Brazão, and W. S. Smith. 2022. Dredging on water bodies and margin interventions: effects on fish. Boletim do Instituto de Pesca, 48:e641.

Underwood, A. J. 1991. Beyond BACI: experimental designs for detecting human environmental impacts on temporal variations in natural populations. Marine and Freshwater Research 42(5):569-587.

United States Army Corps of Engineers (USACE). 2012. Savannah harbor expansion project. Final General Re-evaluation Report, Savannah District, Savannah, Georgia.

United States Army Corps of Engineers (USACE). 2025. SHEP Progression. Accessed March 26, 2025, from https://www.sas.usace.army.mil/Missions/Civil-Works/Savannah-Harbor-Expansion/.

Wenger, A. S., J. L. Johansen, and G. P. Jones. 2012. Increasing suspended sediment reduces foraging, growth and condition of a planktivorous damselfish. Journal of Experimental Marine Biology and Ecology 428:43-48.

Wenger, A. S., E. Harvey, S. Wilson, C. Rawson, S. J. Newman, D. Clarke, B. J. Saunders, N. Browne, M. J. Travers, J. L. McIlwain, P. L. A. Erftemeijer, J. P. A. Hobbs, D. McLean, M. Depczynski, and R. D. Evans. 2017. A critical analysis of the direct effects of dredging on fish. Fish and Fisheries 18(5):967-985.

Ziegeweid, J. R., C. A. Jennings, D. L. Peterson, and M. C. Black. 2008. Effects of salinity, temperature, and weight on the survival of young-of-year Shortnose Sturgeon. Transactions of the American Fisheries Society 137(5):1490–1499.

Tables and Figures

Table 3.1. Summary details of annual sampling efforts for the Shortnose Sturgeon *Acipenser brevirostrum* in both the Savannah River estuary, Georgia and South Carolina, and the Altamaha River estuary, Georgia, using trammel and gill nets from 2013–2024. We present effort of net deployment in hours (h). We also present juvenile capture data and adult capture data. In our study, a juvenile was defined as a fish <450 mm fork length and an adult as a fish ≥450 mm fork length. Recaptures of marked individuals are separate captures. NA is indicative of no sampling effort deployed in a given year for a study river.

			Juvenile captures		Adult captures	
Year	Sampling Period	Effort (h)	Marked	Recaptured	Marked	Recaptured
			Savannah River			
2013	May 15 – July 3	174	14	0	136	4
2014	May 12 – July 18	187	39	4	215	12
2015	May 28 – July 29	142	62	4	142	9
2016	May 13 – August 2	205	40	2	185	9
2017	May 8 – July 27	154	79	5	143	7
2018	May 2 – July 27	238	57	5	207	11
2019	May 7 – July 31	194	3	0	129	2
2020	May 19 – August 5	122	16	0	63	0
2021	June 21 – August 2	94	17	0	108	8
2022	May 9 – August 2	53	2	0	80	2
2023	May 16 – August 2	135	7	2	142	6
2024	May 23 – July 18	98	2	0	97	7
			Altamaha River			
2013	NA	0	0	0	0	0
2014	May 7 – June 27	85	38	0	22	0
2015	May 7 – June 1	44	18	1	22	2
2016	May 4 – July 11	109	15	0	153	3
2017	May 2 – July 28	183	43	8	237	10
2018	May 1 – July 27	193	2	0	129	4
2019	May 6 – August 2	222	20	0	88	2
2020	May 18 – August 7	284	48	2	131	5
2021	May 14 – August 8	219	48	0	233	9
2022	May 6 – July 27	65	15	1	62	2
2023	May 8 – July 14	150	26	1	82	3
2024	May 13 – June 26	109	23	2	91	0

Table 3.2. Model selection table to evaluate Huggins closed-population capture-recapture models that were used to quantify interannual juvenile and adult abundances of the Shortnose Sturgeon population in the Savannah River estuary, Georgia and South Carolina, from 2013–2024. Each model had an associated Akaike's Information Criterion corrected for small sample sizes (AIC$_c$). Each model's ΔAICc is indicative of the difference between the AICc of that specified model and the AICc of the model with the smallest AICc value. Each model also had an associated Akaike's weight (W_i) and the number of parameters (K) used to estimate that model. The selected model to represent abundance estimates for a given year was the model with the highest W_i. We denote time varying capture probability as *t*, group varying capture probability as *g*, and constant capture probability as *0*. Model M_{t+g} is indicative of the additive effect between *t* and *g*, while M_{txg} is indicative of the interactive effect between *t* and *g*.

Year	Model	AICc	ΔAICc	W_i	K
2013	M_t	630.01	0.00	0.65	8
	M_{t+g}	631.25	1.25	0.35	9
	$M_{t\times g}$	641.42	11.41	0.00	16
	M_0	672.87	42.86	0.00	1
	M_g	674.09	44.09	0.00	2
2014	M_t	1333.48	0.00	0.40	10
	$M_{t\times g}$	1333.72	0.25	0.35	20
	M_{t+g}	1334.51	1.03	0.24	11
	M_0	1340.50	7.02	0.01	1
	M_g	1341.52	8.04	0.00	2
2015	$M_{t\times g}$	1060.05	0.00	0.99	20
	M_t	1068.63	8.58	0.01	10
	M_{t+g}	1070.65	10.60	0.00	11
	M_0	1078.29	18.25	0.00	1
	M_g	1080.30	20.25	0.00	2
2016	M_t	1188.59	0.00	0.67	12
	M_{t+g}	1190.02	1.43	0.33	13
	$M_{t\times g}$	1202.88	14.30	0.00	24
	M_0	1241.16	52.57	0.00	1
	M_g	1242.58	53.99	0.00	2
2017	$M_{t\times g}$	1179.68	0.00	1.00	24
	M_t	1198.62	18.95	0.00	12
	M_{t+g}	1200.47	20.79	0.00	13
	M_0	1235.30	55.62	0.00	1
	M_g	1237.12	57.45	0.00	2
2018	M_t	1445.87	0.00	0.71	13
	M_{t+g}	1447.70	1.84	0.29	14
	$M_{t\times g}$	1459.70	13.83	0.00	26
	M_0	1497.60	51.73	0.00	1
	M_g	1499.42	53.55	0.00	2

2019	M_t	665.88	0.00	0.73	12
	M_{t+g}	667.82	1.94	0.27	13
	$M_{t\times g}$	683.16	17.28	0.00	24
	M_0	685.60	19.72	0.00	1
	M_g	687.52	21.64	0.00	2
2021	M_{t+g}	521.61	0.00	0.48	10
	M_t	521.91	0.30	0.41	9
	$M_{t\times g}$	524.51	2.90	0.10	18
	M_g	633.24	111.63	0.00	2
	M_0	633.55	111.95	0.00	1
2022	M_t	321.54	0.00	0.73	9
	M_{t+g}	323.50	1.96	0.27	10
	$M_{t\times g}$	331.21	9.67	0.00	18
	M_0	386.76	65.22	0.00	1
	M_g	388.67	67.13	0.00	2
2023	M_{t+g}	691.61	0.00	0.61	11
	$M_{t\times g}$	693.84	2.23	0.20	20
	M_t	693.89	2.28	0.19	10
	M_g	763.83	72.22	0.00	2
	M_0	766.10	74.49	0.00	1
2024	M_t	445.10	0.00	0.71	9
	M_{t+g}	446.90	1.80	0.29	10
	$M_{t\times g}$	458.39	13.29	0.00	18
	M_0	499.39	54.29	0.00	1
	M_g	501.15	56.05	0.00	2

Table 3.3. Model selection table to evaluate Huggins closed-population capture-recapture models that were used to quantify interannual juvenile and adult abundances of the Shortnose Sturgeon population in the Altamaha River estuary, Georgia, from 2013–2024. Each model had an associated Akaike's information criterion corrected for small sample sizes (AICc). Each model's ΔAICc is indicative of the difference between the AICc of that specified model and the AICc of the model with the smallest AICc value. Each model also had an associated Akaike's weight (W_i) and the number of parameters (K) used to estimate that model. The selected model to represent abundance estimates for a given year was the model with the highest W_i. We denote time varying capture probability as *t*, group varying capture probability as *g*, and constant capture probability as *0*. Model M_{t+g} is indicative of the additive effect between *t* and *g*, while M_{txg} is indicative of the interactive effect between *t* and *g*.

Year	Model	AICc	ΔAICc	W_i	K
2015	M_t	97.95	0.00	0.60	4
	M_{t+g}	98.88	0.93	0.38	5
	$M_{t\times g}$	105.26	7.31	0.02	8
	M_0	123.11	25.16	0.00	1
	M_g	123.97	26.01	0.00	2
2016	M_t	665.81	0.00	0.67	9
	M_{t+g}	667.27	1.47	0.32	10
	$M_{t\times g}$	674.80	8.99	0.01	18
	M_0	778.75	112.94	0.00	1
	M_g	780.20	114.39	0.00	2
2017	$M_{t\times g}$	1369.76	0.00	0.76	22
	M_{t+g}	1372.12	2.36	0.24	12
	M_t	1381.58	11.82	0.00	11
	M_g	1501.64	131.88	0.00	2
	M_0	1511.07	141.31	0.00	1
2018	M_t	720.75	0.00	0.62	13
	M_{t+g}	722.66	1.91	0.24	14
	M_0	724.29	3.54	0.14	1
	M_g	726.17	5.42	0.00	2
	$M_{t\times g}$	738.44	17.69	0.00	26
2019	M_t	537.42	0.00	0.57	11
	M_{t+g}	538.65	1.22	0.31	12
	$M_{t\times g}$	540.73	3.31	0.12	22
	M_0	546.35	8.93	0.00	1
	M_g	547.54	10.12	0.00	2
2020	M_t	929.14	0.00	0.73	12
	M_{t+g}	931.16	2.02	0.27	13
	$M_{t\times g}$	939.60	10.45	0.00	24
	M_0	943.66	14.52	0.00	1
	M_g	945.66	16.52	0.00	2

2021	M_{t+g}	1403.89	0.00	0.61	13
	M_t	1404.86	0.97	0.37	12
	$M_{t\times g}$	1410.72	6.82	0.02	24
	M_g	1497.80	93.91	0.00	2
	M_0	1498.79	94.89	0.00	1
2022	M_t	303.53	0.00	0.70	7
	M_{t+g}	305.27	1.74	0.30	8
	$M_{t\times g}$	312.36	8.82	0.00	14
	M_0	333.75	30.22	0.00	1
	M_g	335.45	31.92	0.00	2
2023	$M_{t\times g}$	455.24	0.00	0.47	20
	M_t	456.17	0.93	0.29	10
	M_{t+g}	456.56	1.32	0.24	11
	M_0	535.90	80.67	0.00	1
	M_g	536.26	81.03	0.00	2

Table 3.4. Interannual juvenile abundance estimates of the Shortnose Sturgeon populations in the Savannah River estuary, Georgia and South Carolina, and the Altamaha River estuary, Georgia, from 2013–2024. The years 2013 – 2019 are representative of the period before the SHEP occurred, and the years 2020 – 2024 are representative of the years in conjunction with and following completion of the SHEP. Juvenile abundance estimates were derived from Huggins closed population capture-recapture models. NA indicates years in which juvenile abundance could not be estimated.

	Savannah River		Altamaha River	
Year	Abundance	95% CI	Abundance	95% CI
2013	227	88 – 628	NA	NA
2014	300	182 – 517	NA	NA
2015	461	208 – 1149	203	52 – 1043
2016	388	216 – 725	346	119 – 1066
2017	591	286 – 1348	127	79 – 241
2018	603	347 – 1084	31	8 – 132
2019	90	22 – 412	490	146 – 1772
2020	NA	NA	1000	406 – 2576
2021	NA	NA	NA	NA
2022	31	8 – 157	160	60 – 485
2023	15	8 – 49	395	145 – 1166
2024	15	5 – 58	219	58 – 1108

Table 3.5. Interannual adult abundance estimates of the Shortnose Sturgeon populations in the Savannah River estuary, Georgia and South Carolina, and the Altamaha River estuary, Georgia, from 2013 – 2024. The years 2013 – 2019 are representative of the period before the SHEP occurred, and the years 2020 – 2024 are representative of the years in conjunction with and following completion of the SHEP. Adult abundance estimates were derived from Huggins closed population capture-recapture models. NA indicates years in which adult abundance could not be estimated.

	Savannah River		Altamaha River	
Year	Abundance	95% CI	Abundance	95% CI
2013	2205	928 – 5539	NA	NA
2014	1656	1065 – 2659	NA	NA
2015	1070	607 – 1996	249	63 – 1267
2016	1783	1044 – 3158	3531	1304 – 10069
2017	1365	712 – 2768	3575	1821 – 7271
2018	2189	1303 – 3791	1999	844 – 5021
2019	3886	1184 – 13511	2156	665 – 7507
2020	NA	NA	2728	1130 – 6884
2021	634	357 – 1217	3142	1672 – 6115
2022	1253	403 – 4337	661	259 – 1880
2023	1770	820 – 4056	1244	471 – 3551
2024	717	366 – 1526	NA	NA

Table 3.6. Results of the before-after control-impact (BACI) model used with abundance point estimates as response variables. Abundance point estimates were representative of interannual demographic based abundances of the Shortnose Sturgeon populations in the Savannah River estuary, Georgia and South Carolina, and the Altamaha River estuary, Georgia, from 2013 – 2024. Abundance point estimates were derived from Huggins closed-population capture recapture models. We present each dependent variable with their associated effect. Each effect has a BACI model estimate, a standard error (SE) of that estimate, and a p-value (*P*) to assess statistical significance ($P < 0.05$). Estimate values are presented on the log-transformed scale. A *P* with an asterick (*) indicates that statistical significance was detected for a given effect of a dependent variable.

Dependent variable	Effect	Estimate	SE	*P*
Juvenile abundance	Altamaha before (intercept)	5.13	0.36	<0.01*
	Savannah before	0.66	0.47	0.18
	Altamaha after	0.71	0.54	0.21
	Interaction (time × river)	-3.55	0.77	<0.01*
Adult abundance	Altamaha before (intercept)	7.43	0.32	<0.01*
	Savannah before	0.11	0.42	0.80
	Altamaha after	-0.03	0.48	0.95
	Interaction (time × river)	-0.59	0.66	0.38

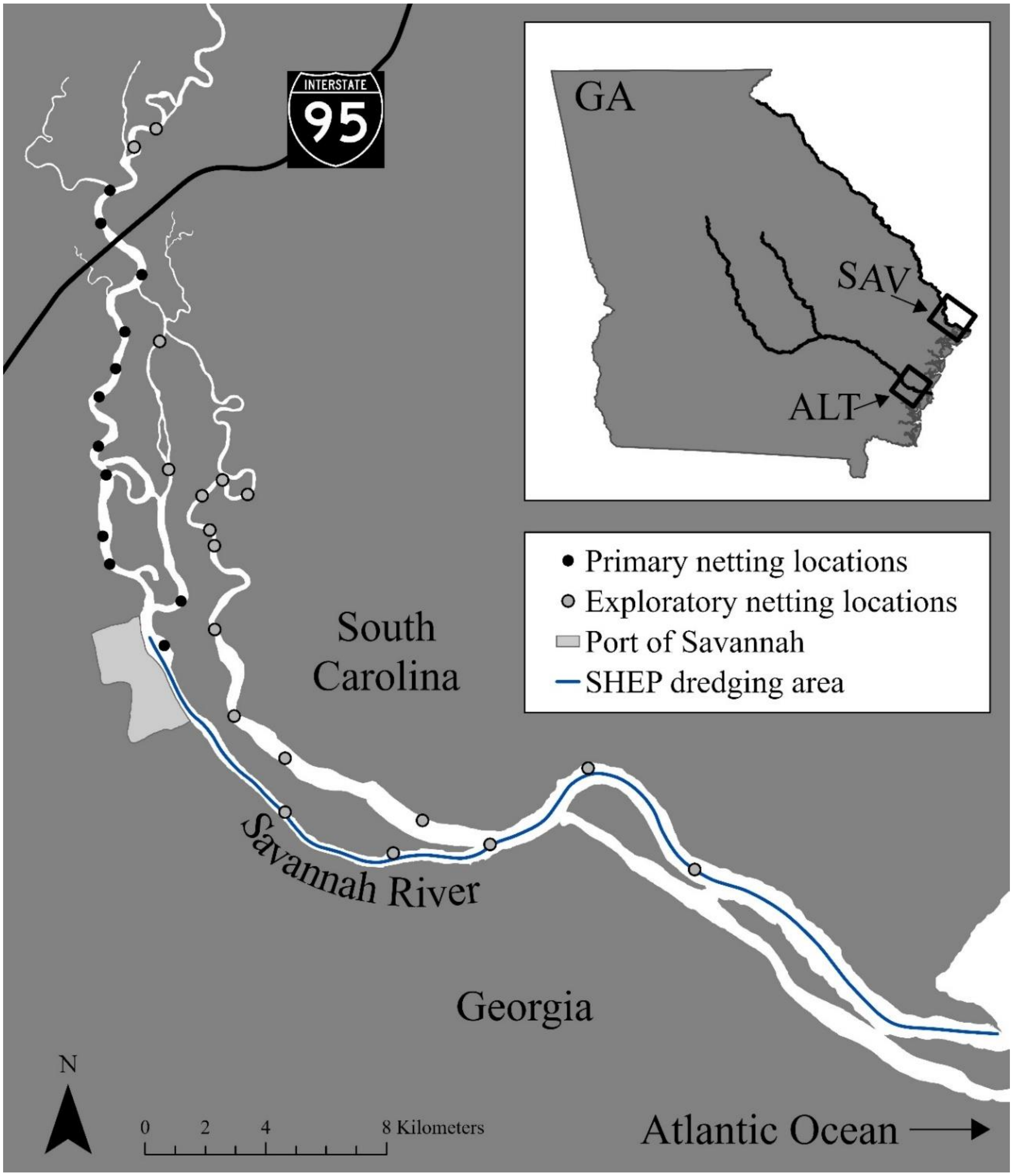

Figure 3.1. Study area in which mark-recapture efforts were deployed for the Shortnose Sturgeon in the Savannah River estuary, Georgia and South Carolina in the summer periods of 2013–2024. Primary netting locations are sites with repeated sampling in each year of the study. Exploratory netting locations are sites that were sampled to determine the potential spatial occurrence of the Shortnose Sturgeon in summer months; these sites received relatively little effort because they rarely yielded fish capture. In the inset map, boxes over each river are indicative of the general summer sampling areas for the Shortnose Sturgeon in the Savannah River (SAV) and the geographically proximate Altamaha River (ALT), Georgia.

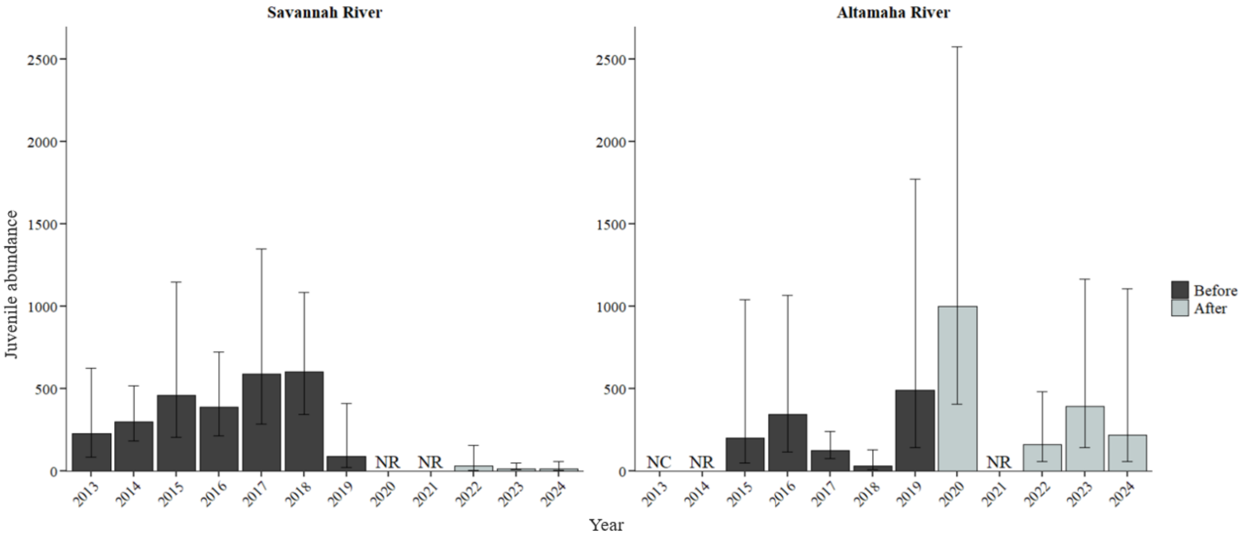

Figure 3.2. Juvenile abundance estimates for the Shortnose Sturgeon populations in the Savannah River estuary, Georgia and South Carolina, and the Altamaha River estuary, Georgia, from 2013–2024. The top of each bar is indicative of the abundance point estimate for that given year. Each bar has associated error bars that are representative of 95% confidence intervals. Dark gray bars represent the time period before the Savannah Harbor Expansion Project (SHEP) occurred in the Savannah River estuary. Light gray bars represent juvenile abundances in each river during and after the SHEP. Abundance estimates could not be calculated for years in which no captures (NC) or no recaptures (NR) of juveniles occurred.

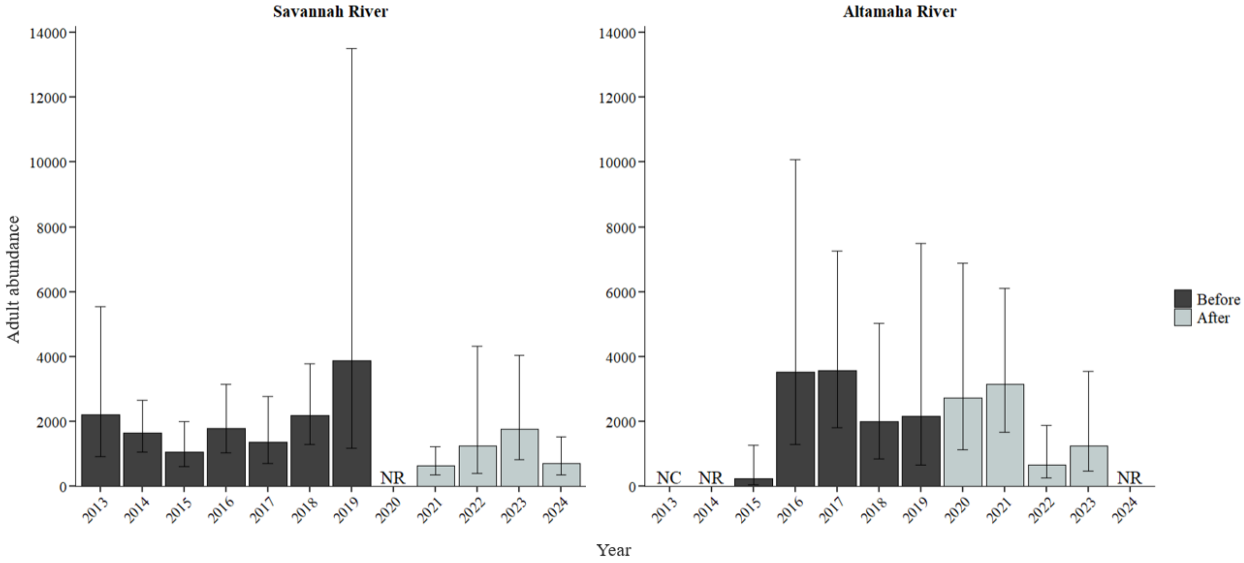

Figure 3.3. Adult abundance estimates for the Shortnose Sturgeon populations in the Savannah River estuary, Georgia and South Carolina, and the Altmaha River estuary, Georgia, from 2013 – 2024. The top of each bar is indicative of the abundance point estimate for that given year. Each bar has associated error bars that are representative of 95% confidence intervals. Dark gray bars are representaitve of interannual adult abundances in each river before the Savannah Harbor Expansion Project (SHEP) occurred in the Savannah River estuary. Light gray bars represent adult abundances in each river during and after the SHEP. Abundance estimates could not be calculated for years in which no captures (NC) or no recaptures (NR) of adults occurred.

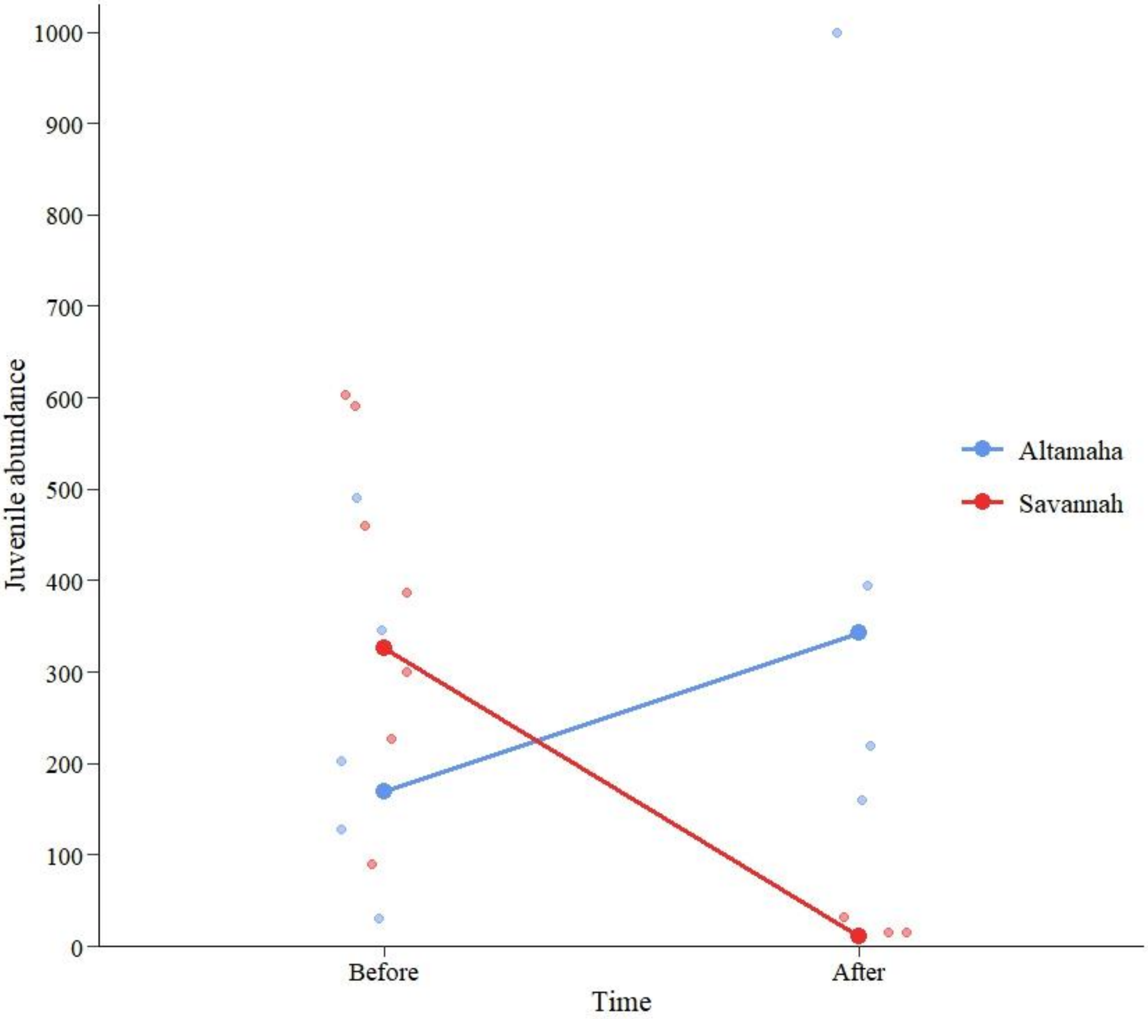

Figure 3.4. Time-specific estimates of juvenile abundances derived from the before-after control-impact (BACI) model for the Shortnose Sturgeon populations in the Savannah River estuary, Georgia and South Carolina, and the Altamaha River estuary, Georgia, from 2013 – 2024. In this model, the Savannah River was the impact system, and the Altamaha River was the control system. The 'before' period encompassed 2013–2019 (before the Savannah Harbor Expansion Project [SHEP]), and the 'after' period encompassed 2020–2024 (during and after SHEP). Larger points connected to lines represent the back-transformed estimates of juvenile abundance produced from the BACI model for each study river. Log-transformed estimates of abundance are presented in table 6. Smaller, transparent points are representative of the yearly point abundance estimates used in the BACI model approach.

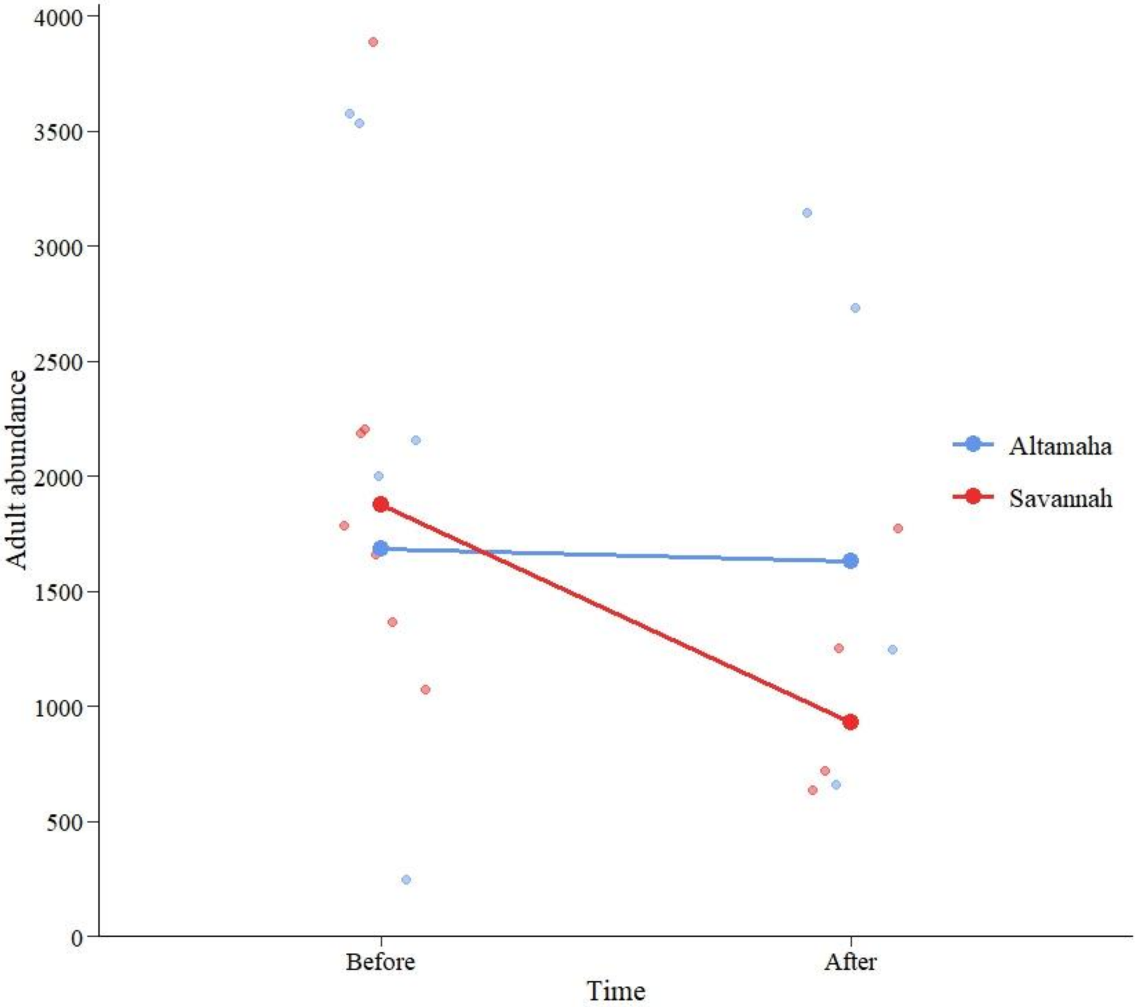

Figure 3.5. Time-specific estimates of adult abundances derived from the before-after control-impact (BACI) model for the Shortnose Sturgeon populations in the Savannah River estuary, Georgia and South Carolina, and the Altamaha River estuary, Georgia, from 2013 – 2024. In this model, the Savannah River was the impact system, and the Altamaha River was the control system. The 'before' period encompassed 2013–2019 (before the Savannah Harbor Expansion Project [SHEP]), and the 'after' period encompassed 2020–2024 (during and after SHEP). Larger points connected to lines represent the back-transformed estimates of adult abundance produced from the BACI model for each study river. Log-transformed estimates of abundance are presented in table 6. Smaller, transparent points are representative of the yearly point abundance estimates used in the BACI model approach.

CHAPTER 4

CONCLUSIONS

The Shortnose Sturgeon was listed as federally endangered in 1967 after populations across the species' range experienced declines from overexploitation and habitat degradation. However, since its listing, the Shortnose Sturgeon endangered status has been unchanged because population sizes have been impeded from recovery due to continued habitat alteration in natal river systems, increased mortality from bycatch in adjacent commercial fisheries efforts, as well as the persistence of knowledge gaps in the population dynamics of the Shortnose Sturgeon at the riverine level (SSSRT 2010). A thorough understanding of population dynamics at the riverine-population level is critical for determining the recovery status and trajectory of the species (NMFS 1998, SSSRT 2010). Fisheries managers rely on accurate data about parameters of population dynamics, including abundance, recruitment, mortality, growth rates, and movement when implementing effective recovery actions for the Shortnose Sturgeon.

To address knowledge gaps about population age structure of the Shortnose Sturgeon, we developed an alternative approach to determining age and growth data for populations of the Shortnose Sturgeon in coastal rivers of Georgia, USA, using a long-term mark-recapture dataset. We also used a long-term set of interannual mark-recapture data to examine trends in demographic-based abundances of the Savannah River Shortnose Sturgeon population to assess how the population has changed in conjunction with major estuarine dredging over time.

Known ages of fish are important for validating ageing techniques that are used to accurately produce population parameters like growth and mortality rates. The Shortnose Sturgeon Status Review Team (2010) recommended that techniques for determining age information in the Shortnose Sturgeon be validated to improve the quality of age data for populations of the Shortnose Sturgeon. Historically, the pectoral fin spine has been the most accepted structure for use in attaining age estimates from sturgeons (Baremore and Rosati 2014). However, Rider et al. (2025) identified errors in assigning ages to Shortnose Sturgeon individuals with pectoral fin spines; fin spine aging of known-age fish resulted in underestimated ages for fish six years and older. The use of erroneous age estimates in population models can result in inaccurate findings regarding population parameters, especially in long-lived species (Hamel et al. 2016). Because of the inaccuracies in fin-spine age data used by previous studies to develop and present population dynamics and life history characteristics of the Shortnose Sturgeon across its range, we used an alternate method to assess river-specific characteristics of age and growth in several Georgia rivers. We determined that the Shortnose Sturgeon in southern populations can attain maximum ages [28 years] almost a decade longer than what was previously described with pectoral fin spine age information (Marchette and Smiley 1982). Our mark-recapture analysis revealed that maximum body sizes and predicted longevity differed among geographically proximate river systems in Georgia, supporting the necessity of riverine-level recovery assessments of the Shortnose Sturgeon. Given the inaccuracies we observed in ages derived from pectoral fin spines, we suggest that ages assessed via mark-recapture are a valid alternative approach that can be used in other populations with sufficient data. Because there are known latitudinal differences in the population dynamics of the Shortnose Sturgeon, we recommend that ages and growth rates of mid-Atlantic and northern populations of the species be re-evaluated using this approach. A better understanding of age and growth will contribute to more accurate and effective riverine-level recovery plans across the species' range.

Understanding abundance and recruitment are necessary for assessing populations and planning species recovery (NMFS 1998). Long-term abundance datasets can reveal trends in these demographics, as well as identify how populations of the Shortnose Sturgeon respond to management actions or anthropogenic activities. We quantified interannual demographic-based abundances of the Shortnose Sturgeon in the Savannah River from 2013–2024 and compared those abundances before and after a major habitat change. The Savannah Harbor Expansion Project (SHEP) was a multi-year dredging operation to deepen the Savannah River estuary for access by larger cargo ships. This project was predicted to decrease estuarine habitat that is used by the Shortnose Sturgeon to complete its life cycle (USACE 2012). The estuarine environment in the Savannah River serves as a nursery for juveniles, as well as foraging habitat and a potential migratory corridor for adults (Hall et al. 1991, Collins et al. 2002, Kynard et al. 2016). We compared demographic-based point abundance estimates for the Savannah River Shortnose Sturgeon population using a before-after control-impact (BACI) statistical design to determine trends in juvenile abundance and adult abundance over time in conjunction with the completion of the SHEP. Our results indicate that there was a significant decline in juvenile abundance after the completion of the SHEP, whereas adult abundance was similar over the same time frame. A decline in juvenile abundance, along with the potential for a future decrease in the number of sexually mature adults, may threaten the long-term persistence of the Savannah River Shortnose Sturgeon population. Future research is needed to better investigate the mechanism of this decline and to determine how suitable nursery habitat has changed in the Savanna River following the SHEP. Understanding the spatial-temporal use of estuarine habitat by juveniles is crucial to mitigating further potential degradation of the Savannah River estuary by future anthropogenic development and aiding in recovery efforts of the Savannah River Shortnose Sturgeon population.

We recommend mark-recapture efforts continue to be employed for data collection of the Shortnose Sturgeon in coastal rivers of Georgia and throughout the species' range. Although mark-recapture is a resource- and time-intensive data gathering method, it has great value as a tool for producing more accurate riverine-specific demographics of the endangered Shortnose Sturgeon than the traditional fin-ray based approaches. Those data are needed for population viability assessments and in determining recovery actions for the species. Furthermore, many populations of the Shortnose Sturgeon will continue to be threatened by anthropogenic actions including development, bycatch, pollution, dredging, and changes in climate (SSSRT 2010, Kynard et al. 2016). Continued monitoring efforts – like mark-recapture studies –will be critical for assessing how populations of the Shortnose Sturgeon respond to those anthropogenic perturbations in the future.

References

Baremore, I. E. and J. D. Rosati. 2014. A validated, minimally deleterious method for aging sturgeon. Fishery Bulletin 112(4):274-282.

Collins, M. R., W. C. Post, D. C. Russ, and T. I. J. Smith. 2002. Habitat use and movements of juvenile Shortnose Sturgeon in the Savannah River, Georgia-South Carolina. Transactions of the American Fisheries Society 131(5):975-979.

Hall, J. W., T. I. Smith, and S. D. Lamprecht. 1991. Movements and habitats of shortnose sturgeon, *Acipenser brevirostrum* in the Savannah River. Copeia 695-702.

Hamel, M. J., J. J. Spurgeon, C. J. Chizinski, K. D. Steffensen, and M. A. Pegg. 2016. Variability in age estimation results in ambiguity and false understanding of population persistence. North American Journal of Fisheries Management 36(3):514-522.

Kynard, B., S. Bolden, M. Kieffer, M. Collins, H. Brundage, E. J. Hilton, M. Litvak, M. T. Kinnison, T. King, and D. Peterson. 2016. Life history and status of Shortnose Sturgeon (*Acipenser brevirostrum* LeSueur, 1818). Journal of Applied Ichthyology 32:208-248.

Marchette, D. E. and R. Smiley. 1982. Biology and life history of incidentally captured Shortnose Sturgeon, *Acipenser brevirostrum*, in South Carolina. South Carolina Wildlife and Marine Resources Research Institute, Charleston, South Carolina, United States.

National Marine Fisheries Service (NMFS). 1998. Final recovery plan for the Shortnose Sturgeon, *Acipenser brevirostrum*. National Marine Fisheries Service.

Rider, H. R., M. K. Morgan, A. T. Bond, J. D. Nolan, A. G. Fox, M. J. Hamel. 2025. Validating Age Estimates from Pectoral Fin Spines and Length-Frequency Analysis of Known-Age Shortnose Sturgeon. Journal of the Southeastern Association of Fish and Wildlife Agencies 12:52-60.

Shortnose Sturgeon Status Review Team (SSSRT). 2010. A biological assessment of shortnose sturgeon (*Acipenser brevirostrum*). Report to National Marine Fisheries Service, Northeast Regional Office.

United States Army Corps of Engineers (USACE). 2012. Savannah harbor expansion project. Final General Re-evaluation Report, Savannah District, Savannah, Georgia.