Texas Coast-wide Characterization of the Saltmarsh Topminnow (Fundulus jenkinsi)

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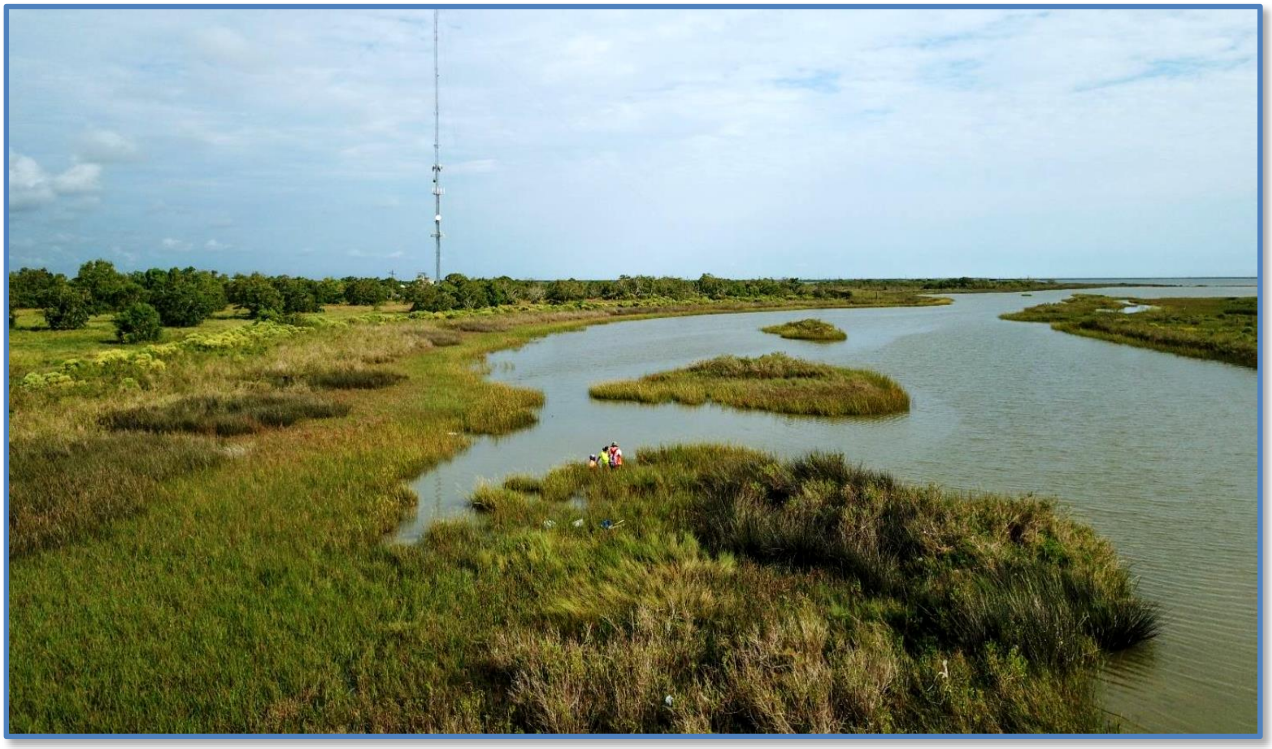

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Environmental Institute of Houston
University of Houston - Clear Lake

Texas Coast-wide Characterization of the Saltmarsh Topminnow (Fundulus jenkinsi)

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Executive Summary

The Saltmarsh Topminnow (*Fundulus jenkinsi*) occurs sporadically in tidal marsh habitat along the U.S. coast of the Gulf of Mexico, from Florida to Texas. Little has been known about the exact extent of their range, distribution, or abundance in Texas but previous studies have shown a link between *Spartina alterniflora* marsh habitat and *F. jenkinsi* occurrences. Historically, the Galveston Bay population represents the western most extent of this species' range.

The Saltmarsh Topminnow has been previously found in low to moderate salinities and appears to utilize the edge of the salt marshes and have been primarily found within small intertidal creeks connected to large salt marsh channels. The Texas Parks and Wildlife Department (TPWD) coastal fisheries monitoring program has been unsuccessful in detecting *F. jenkinsi* in Texas estuaries during the period from April 1976 to August 2018 (71,135 coast-wide bag-seine samples) using their standardized monitoring program.

The Saltmarsh Topminnow has been listed as a species of concern by the National Oceanic & Atmospheric Administration (NOAA) and in the states of Louisiana, Mississippi, Alabama, and Florida since 2004 (Federal Register 2004a). Collectively, NOAA and the U.S. Fish and Wildlife Service (USFWS) determined that the petition presented substantial scientific information indicating that the petitioned action may be warranted and published a joint 90-day finding in the Federal Register (USFWS 2011). The USFWS agreed to assume jurisdiction of the species and responsibility for determining whether listing the Saltmarsh Topminnow as threatened or endangered is warranted. The Saltmarsh Topminnow is listed as a species of greatest conservation need for the Gulf Coast Prairies and Marshes ecoregion in the state of Texas (TPWD 2011).

Due to their apparent rarity, there is an urgent need to determine the current population status of *F. jenkinsi* within Texas and across its historical range. Comprehensive data on both the species range, habitat requirements, and demographics is currently lacking within Texas. These data are needed by resource agencies to support ongoing management and conservation of this species and related habitat. Given this species' restricted range in Texas and the recent projections of coastal development, ongoing land subsidence, and projected sea level rise, it is important that the occurrence and habitat associations of this species are carefully delineated as these and other threats can cumulatively degrade saltmarsh habitat and consequently threaten this species of fish.

The primary goal of the study was to conduct a thorough assessment of the status of Saltmarsh Topminnow populations along the entire Texas coast. Historical and new data collected within all major estuaries over a four-year period was used to assess the population viability and extirpation risk of this species. This project builds upon the findings of Guillen et al. (2015) by extending the study temporally and spatially into additional estuaries.

The major objectives of this study that support the primary goal included:

1. Developing estimates of population densities of Saltmarsh topminnow in major estuaries of Texas

2. Determining habitat associations and preferences of Saltmarsh Topminnow using statistical models and GIS
3. Estimating various demographic and population parameters of F. jenkinsi within Texas

The objectives outlined above were accomplished by: 1) synthesizing and reviewing past literature including agency reports and previous studies conducted by EIH-UHCL, 2) conducting new surveys using passive and active collection techniques including Breeder traps and seine hauls of areas where Saltmarsh Topminnow have historically been captured, and 3) executing additional surveys in areas where they have not been reported, but meet the basic known habitat requirements for the species. During each survey, physicochemical, habitat and biological community data were collected to assess potential interactions between *F. jenkinsi*, co-occurring species of fish, and habitats.

A total 501 *F. jenkinsi* were collected and a total of 162 sites were sampled along the upper to middle Texas coast from February 7, 2017 to March 7, 2018. This data was combined with information archived by Guillen et al. (2015) which included data collected from February 17, 2014 to November 20, 2014 when a total of 215 *F. jenkinsi* were captured and a total of 135 sites were sampled along the upper Texas coast (Galveston Bay and Sabine Lake). The range of *F. jenkinsi* captured per collection, when present, ranged from 3 to 17 specimens per event depending on the estuary and time period.

An east to west gradient of *F. jenkinsi* density based on geographic location was documented during the current study. Highest densities were detected in estuaries found in the eastern part of the state including Sabine Lake, Galveston Bay, and Matagorda Bay. San Antonio Bay represented the most western estuary that yielded positive catches of *F. jenkinsi*. The current study failed to capture *F. jenkinsi* in tributaries of Aransas Bay at locations where other investigators had previously detected them. This east to west trend of declining catch rates appears to be correlated with increasing salinities. When *F. jenkinsi* was captured in the more western estuaries (Matagorda and San Antonio) always occurred at salinities less than 19 ‰. Currently the most western reported record of *F. jenkinsi* is from Oso Creek at FM 763, where 9 specimens were captured in 1999-2000 (Nicolau 2001). Although the exact date of collection(s) is/are not recorded the reported salinity ranged from 0.9 to 4.1 ‰ and averaged 2.6 ‰. In addition to these coastal bend estuaries, we failed to capture *F. jenkinsi* within the San Bernard-Brazos and East Matagorda Bay systems. This disjointed distribution between estuaries may be due to a lack of suitable tidal wetland creek habitat within these systems or insufficient spatial and temporal sampling effort.

A gradient of declining *F. jenkinsi* frequency and density downstream within individual estuaries was observed during the current study. The highest densities of *F. jenkinsi* were generally found upstream in less saline wetlands within Sabine Lake, Galveston Bay, Matagorda Bay, and San Antonio Bay. Although the current study failed to capture Salt Marsh Topminnow in former upstream locations within the Aransas Bay estuary in the Mission and Aransas Rivers, these areas did yield specimens during the survey conducted by Tolan et al. (2011). These upstream tidal creeks and wetlands typically exhibit comparatively lower salinities within their respective estuary.

Our study supports previous literature which states that *F. jenkinsi* seem to prefer lower to mid-salinity ranges. The difference in distribution of *F. jenkinsi* across salinity gradients between bays is most likely due to the degree of freshwater inflow and resulting salinity in each system. These data suggests that while *F. jenkinsi* is able to inhabit a wide salinity range (0.1-19.2 ‰) it is equally important to have corridors of appropriate marsh habitat available throughout the salinity gradient within an estuary. This is important because large fluctuations in either fresh or saltwater input may trigger *F. jenkinsi* to actively moving to other locations within the estuary to stay within their preferred salinity range, which is defined as 0.1-12.4 ‰ based on the 75% percentile of positive trap, and seine catches.

During the current study, tidal regime had a major influence on *F. jenkinsi* CPUE. Lowest minimum and median water levels generally occurred during the winter months when strong cold fronts would reduce water levels across the Gulf coast. During the winter, median water levels were lower than other seasons and months. At the same time median seasonal seine CPUE was highest during the winter. Highest average seine CPUE occurred during the months of November, December, January and February. Conversely, most of the collections containing zero or low catches of Saltmarsh Topminnow occurred at sites sampled during the summer when water levels were highest, which allows small nekton to seek refuge in inundated marsh vegetation therefore evading our standard sampling gear. It is therefore advisable to revisit sites during winter months to collect additional samples and increase the likelihood of capturing *F. jenkinsi* if present.

The Saltmarsh Topminnow was found in marshes of various dominant plant species. However, *F. jenkinsi* was significantly more likely to be caught at sites where *Spartina alterniflora* was the dominant vegetation. Previous studies have also stressed the linkage of *F. jenkinsi* presence and *S. alterniflora* (Peterson & Turner 1994). Our data suggests that they are also found in association with other marsh vegetation including oligohaline and mesohaline species and not strictly *S. alterniflora*. Analysis of the fish assemblages caught over the course of this study show that *F. jenkinsi* have a strong association with certain fish species. In this study *F. jenkinsi* captured in seines were frequently found in association with at least one other species from the family Fundulidae and often in combination with other marsh edge estuarine fish species (e.g. *Poecilia latipinna*, *Gambusia affinis*, and *Cyprinodon variegatus*). Similarly, the assemblages *F. jenkinsi* did not usually occur in were dominated by more open water species (*Anchoa mitchilli*, *Brevoortia patronus* and *Menidia beryllina*). This pattern of community association provides additional evidence that open water habitat favored by these species is not selected for by *F. jenkinsi*. Significantly higher fish species diversity were associated with the presence of *F. jenkinsi* suggesting that it may prefer saltmarsh habitat that is also able to support a highly diverse array of fish species.

The overall length distribution for *F. jenkinsi* exhibited two modal peaks, one at the 20-24 mm (SL) and the second at the 32-34 mm during the winter, spring and fall. This data supports and provides additional evidence that the two modes of standard length represents at least two separate age classes. A preliminary estimate of the von Bertanlanfy growth model and instantaneous natural mortality was developed for this species. Estimates of instantaneous natural mortality M = 2.61 (92.6% annual mortality) indicate few *F. jenkinsi* live beyond a year.

Based on a comprehensive review of recent literature and the findings of this study we conclude that *F. jenkinsi* is likely not as rare as previously thought. The paucity of historical data reflects the inherent bias of past monitoring programs. The Texas Parks and Wildlife Department (TPWD) coastal fisheries monitoring program utilizes a large bag seine randomly deployed within each major bay system. We hypothesize that the inability to capture *F. jenkinsi* by the TPWD standardized sampling program was due to 1) the inability to deploy bag seine gear in small intertidal creeks and marshes, 2) sampling protocol that targets mostly open shorelines in secondary and primary bays, and 3) inherent bias in the sampling method, which utilizes relatively large mesh (1.3 cm stretched mesh in the bag and 1.9 cm stretched mesh on the wings). Analysis of the *F. jenkinsi* head dimensions during this study illustrated that most of the *F. jenkinsi* were small enough to swim through the bag seine mesh.

Recent historical (Nicolau 2001; Tolan et al. 2011) records indicate that viable local populations of *F. jenkinsi* may be expected to be found further west extending to Corpus Christi Bay. For now, the best estimate of the western extent of *F. jenkinsi* is Oso Creek within the Corpus Christi estuary. Further surveys are critically needed in these areas and the upper Laguna Madre to evaluate the long-term viability of these populations and to define the western extent of the species. In addition, resurveys of many of the index sites visited during this study should be conducted at a minimum in the winter when the species is most vulnerable to sampling gear during low tides. Initiation of mark-recapture studies at index sites in each major estuary would also help establish a more reliable method to determine and validate age and growth estimates for this species. Such methods would also facilitate tracking the movement of *F. jenkinsi* individuals and give better estimates of how much inter-marsh migration occurs and the fidelity to certain habitat types.

Focus areas of future research should include better documentation of oocyte development and fecundity estimates during the spawning season, and the examination of the influence of high tide stages on gonad development and egg production (Lang et al. 2012). Better documentation of these processes would lead to more accurate predictions of spawning times and estimation of offspring production. Further diet and trophic ecology studies of *F. jenkinsi* are needed along with supporting studies on prey availability and abundance would help define any potential limiting factors associated with diet. Such information would help researchers and managers understand the entire spectrum of habitat and biological needs of this species.

Data from this study provides critical information that USFWS can use to evaluate the need for further research and/or the listing of this species for protection under the ESA. Based on the results of our study, this species is likely more extensive than previously thought and has likely been underestimated due to the selectivity of existing agency fisheries monitoring programs which do not cover the preferred habitat of this species. Based on our findings and previous literature there are many management actions that can be taken currently to promote the conservation of this species. For example, future efforts to restore or create new wetlands should incorporate a network of small interconnecting tidal creeks to maximize edge habitat. These wetlands should also be located in the estuary where there is some degree of fresh water inflow to maintain the preferred salinity range of *F. jenkinsi*. Efforts to prevent large sections of currently established tidal creeks from becoming channelized as well as restoring fish passage should also be goals of future conservation projects.

Introduction

Historic Distribution and Range

The Saltmarsh Topminnow (*Fundulus jenkinsi*) occurs sporadically in tidal marshes along the U.S. coast of the Gulf of Mexico, from Florida to Texas (Peterson et al. 2003, Peterson et al 2016, Martin et al. 2012). Little is known about the exact extent of their range, distribution, or abundance but previous studies have documented a positive association between *Spartina alterniflora* marshes and *F. jenkinsi* occurrences (Peterson and Turner 1994). Historically, the Galveston Bay population represented the western most extent of this species' range (Simpson and Gunter 1956 cited in NatureServe Explorer 2014; Patrick et al. 1998; Hoese and Moore 1998; Jordan and Evermann 1896, Guillen et al. 2015; Martin et al. 2012). Additional sporadic collections of *this species* have been made in Cedar Lakes Creek (Guillen 1996)[1], Matagorda Bay (Akin et al. 2003), the Mission and Aransas Rivers (Tolan et al. 2011), and Sabine Lake (Patrick et al. 1998). Based on recent data, the Galveston Bay population appears to be well established, and their presence (while more sporadic) is expected further south and west along the Texas coast (Guillen et al. 2015, Nicolau 2001). Currently the most western reported record of *F. jenkinsi* is from Oso Creek at FM 763, where nine specimens were captured in 1999-2000 (Nicolau 2001). Although the exact date of collection(s) is/are not recorded the reported salinity ranged from 0.9 to 4.1 ppt and averaged 2.6 ppt. There are no reported occurrences of *F. jenkinsi* from the Atlantic coast of Mexico (Castro-Aguirre et al. 1999).

Past studies within Galveston Bay and Sabine Lake have detected *F. jenkinsi* infrequently (Hoese and Moore 1998; USGS 2011; Guillen 1996, Patrick et al. 1998) but when a direct effort was established in 2014 and 2015 to evaluate the distribution and abundance of *F. jenkinsi* in these bay systems a patchy, but wide-spread distribution was documented (Guillen et al. 2015). The Texas Parks and Wildlife Department (TPWD) coastal fisheries monitoring program has been unsuccessful in detecting *F. jenkinsi* in Texas estuaries during the period from April 1976 to August 2018 (71,135 coast-wide bag-seine samples) using their standardized monitoring program. It is highly likely that the current TPWD coastal fisheries monitoring program design which utilizes relatively large mesh bag seines and trawls is not an effective method to capture this species. The bag seines are 18.3m wide with 1.3 cm stretched mesh in the bag and 1.9 cm stretched mesh on the wings and are typically used for shoreline sampling, but because of their size are mostly used to sample open water with minimal vegetation (Martinez-Andrade 2015). The trawls are used in the open bay and are 5.69 m wide with 3.8cm stretched mesh throughout the net (Martinez-Andrade 2015). The lack of catch of *F. jenkinsi* is likely attributed to the species relatively small size which would allow it to pass through the net mesh and its observed affinity for moderate to low salinity wetland edge and tidal creek habitats which are typically not included in the current TPWD sampling frame which focuses on open bay margin sites.

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[1] *F. jenkinsi* data collected by Guillen (1996) but not included in report. Detailed data available from senior author of this report.

Habitat, Ecology, and Life History Characteristics

The Saltmarsh Topminnow is considered an estuarine species and has been previously found in low to moderate salinities (Peterson & Ross 1991, Lopez et al. 2010, Griffith 1974, Guillen et al. 2015). Past research suggests that *F. jenkinsi* have been shown to utilize the edge of the salt marshes (Peterson et al. 2003, Lang et al. 2012, and Lopez et al. 2010) and are primarily found within small intertidal creeks connected to larger channels within the saltmarsh. Access to these small, interconnected tidal creeks appears to be an important contributing factor in the Saltmarsh Topminnow's diet and reproduction. High water levels allow *F. jenkinsi* access to larger foraging areas in the inundated marsh and provides refuge from aquatic predators found in deeper water. While little is known about the diet of *F. jenkinsi,* Lopez et al. (2010) found that its diet consists of both small aquatic and terrestrial invertebrates such as Amphipoda, Gastropoda, Copepoda, Diptera, and Hemiptera, and varied according to the age of the individual and season. An additional diet study by Roberston (2016) identified Nematocera, Collembola, and Amphipods as the most common food items in Texas *F. jenkinsi* in terms of occurrence and percent composition and confirmed that there was a significant difference in prey type by season.

The Saltmarsh Topminnow, like other members of the Fundulidae, are batch spawners and capable of spawning more than once during a single spawning season (Lopez et al. 2010). Monthly gonadosomatic indices (GSI) and ovarian histological analysis of female *F. jenkinsi* indicate the spawning season occurs from March through August (Lang et al. 2012), but slight extensions of the spawning season were shown more recently in the upper Texas Coast (Robertson 2016). Many fish and invertebrates species synchronize their spawning with specific positions of the moon and associated tides (Ricardo et al. 2014; Takemura et al. 2010). Spawning intensity for *F. jenkinsi* appears to increase with the timing of spring tides, when tidal heights are at their greatest, and decrease during neap tides (Lang et al. 2012). The higher water levels allow greater access into the inundated marsh to deposit their eggs on more protected, interior and higher ground. Saltmarsh Topminnow reproduction and diet are strongly linked to inundated salt marsh access. Therefore intertidal creeks embedded within salt marshes are thought to be essential to this species survival as these creeks act as access points for cover, reproduction, and foraging.

Conservation Status

The Saltmarsh Topminnow has been listed as a species of concern by the National Oceanic & Atmospheric Administration (NOAA) and in the states of Louisiana, Mississippi, Alabama, and Florida since 2004 (Federal Register 2004a). Additionally, it is listed as “vulnerable” by the IUCN Red list of threatened species (Collette et al. 2015). These designations were due in part to sparse populations, lack of information regarding its biology and ecology, and the threat that human activities pose to their essential habitat (Peterson et al. 2003, NOAA 2009, Collette et al. 2015). In 2010, the WildEarth Guardians and Sarah Felsen petitioned NOAA and the U.S. Fish and Wildlife Service (USFWS) to list *F. jenkinsi* under the U.S. Endangered Species Act (Felson, 2010). Collectively, NOAA and USFWS determined that the petition presented substantial scientific information indicating that the petitioned action may be warranted and published a joint 90-day finding in the Federal Register (Crabtree 2011; Federal Register 2004b). The USFWS agreed to assume jurisdiction of the species and responsibility for determining whether listing the Saltmarsh Topminnow as threatened or endangered is warranted (Crabtree 2011). The Saltmarsh Topminnow is listed as a species of greatest conservation need for the Gulf Coast Prairies and Marshes ecoregion in the state of Texas. Therefore, responsibility falls to the Texas Parks and Wildlife Department (TPWD) to coordinate with their conservation partners and develop initiatives and goals that will monitor and address the needs of *F. jenkinsi* and their related essential habitats within the state (TPWD 2005, TPWD 2011a, TPWD 2011b, TPWD 2012).

Problem Statement

Due to their apparent rarity there was an urgent need to determine the current population status of *F. jenkinsi* within Texas and across its historical range. Comprehensive data on habitat requirements and demographics of the species were needed across its entire potential range within Texas. This data was needed to inform resource agencies to support ongoing management and conservation of this species and associated essential habitat. Research conducted in other States along the Gulf of Mexico confirm that there is a direct link between *F. jenkinsi* abundance and coastal saltmarsh habitat including specific salinity regimes (Lopez et al., 2010). The link between *F. jenkinsi* abundance and specific habitat requirements needs to be better described and quantified to develop meaningful management recommendations for the long-term conservation of this species. Given this species' expected restricted range in Texas and the recent projections of coastal development, ongoing land subsidence, and projected sea level rise (Warren Pinnacle Inc., 2011; Montagna et al. 2011), it is important that the distribution and habitat associations for this species are carefully delineated. These and other threats could cumulatively reduce the geographic extent of the essential saltmarsh habitat of *F. jenkinsi*.

Study Goals and Objectives

The primary goal of the study was to conduct a thorough assessment of the status of *F. jenkins*i populations along the entire Texas coast. Historical and new data collected within all major estuaries over a four year period was used to assess the extirpation risk of this species. This project builds upon the findings of Guillen et al. (2015) by extending the study temporally and spatially into additional estuaries.

The major objectives of this study that support the primary goal included:

1. Developing estimates of population densities of Saltmarsh topminnow in major estuaries of Texas
2. Determining habitat associations and preferences of Saltmarsh Topminnow using statistical models and GIS
3. Estimating various demographic and population parameters of F. jenkinsi within Texas

Post-Hurricane Recovery Special Study

During the study period, Hurricane Harvey hit the Texas coast and caused catastrophic widespread flooding throughout Southeast Texas, with major impacts to the Galveston Bay area during August-September 2017. Hurricane Harvey was an extremely rare event, exceeding other tropical cyclone rainfall totals of record in the contiguous United States (Oldenborgh et al. 2017). The Texas Water Development Board (TWDB) estimated the total freshwater inflow to the Galveston Bay system caused by Hurricane Harvey (August 25 – September 20, 2017) was 11.5 million acre-feet, which was greater than the average annual freshwater inflow to the system based on the 65-year period of record from 1941-2005 (Schoenbaechler 2018; Guthrie et al. 2012). The record influx of fresh water to Galveston Bay inundated saltmarshes and flushed out the salt water, bringing with it suspended sediment, debris, and potential pollutants.

Saltmarshes are dynamic communities subject to variable amounts of both tidal saltwater exchange and freshwater inflow. Saltmarshes represent ecotones between terrestrial and aquatic systems and are therefore particularly susceptible to extreme physical disturbances that can cause significant changes to associated biotic communities (Piazza and Peyre 2009). Storm surges and rain-induced flooding generated from tropical storms are examples of climatically induced extreme pulse events that can cause intense disturbance (Yang et al. 2008). The persistence of saltmarsh communities subjected to these types of disturbances can serve as indicators of overall estuarine ecosystem resilience. The resilience of coastal ecosystems has become a subject of increasing interest as it relates to global climate change and the expected increase in the number and severity of tropical storms (Switzer et al. 2006; Webster et al. 2005; Trenberth et al. 2018). Hurricane Harvey provided a unique opportunity to evaluate the resiliency and recovery of saltmarsh nekton communities and Saltmarsh Topminnow populations. As a result, a special study was conducted to monitor two sites in Galveston Bay containing *F. jenkinsi* to better understand how saltmarsh nekton communities recover from major disturbance events. Although limited in scope, this provided a unique opportunity to address this question.

Methods

Literature Review and GIS Database

Past records on the occurrence of Saltmarsh Topminnow were obtained from the 1) Fishes of Texas project (Hendrickson & Cohen 2014), 2) VertNet (http://vertnet.org/) , 3) Smithsonian Shore fishes of the Greater Caribbean (https://biogeodb.stri.si.edu/caribbean/en/pages), 4) previous targeted Texas-based studies (Guillen et al. 2015), 5) the TPWD coastal fisheries independent bag seine and trawl data, 5) the Texas Species of Greatest Conservation Need (SGCN) database; 6) NOAA, National Marine Fisheries Service historic studies, 7) published agency and peer reviewed literature, and 8) theses and dissertations. The location and dates of historic occurrences as well as this study's collections were incorporated into an ArcGIS geodatabase and are depicted on a sampling site map (ESRI 2012; Figure 1). A GIS layer containing the most recent land-use data (year 2010) archived by the National Coastal Change Analysis Program (C-CAP) digital land-use atlas was used to depict the distribution of estuarine and palustrine emergent wetland habitats (NOAA 2014 and NOAA 2018). The C-CAP wetland layer was overlaid on the *F. jenkinsi* occurrence maps to depict the distribution of available preferred habitat.

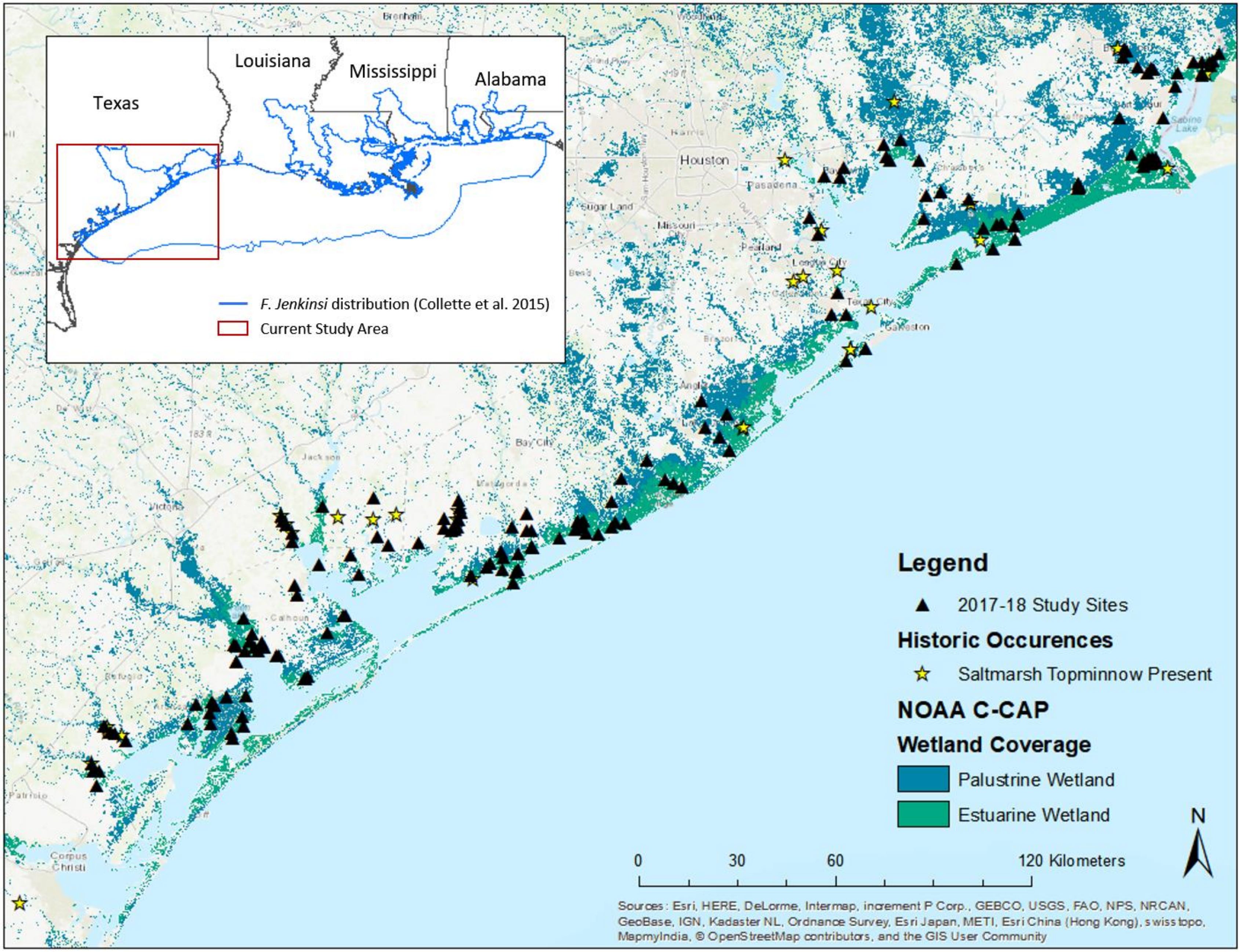
Study Area and Sampling Frequency

During the current study period (October 2016 through April 2018), up to 6-10 sites per each of the 5 major bay systems listed were sampled every 3 months (quarterly). Two sites, with the highest likelihood of consistently supporting *F. Jenkinsi* per major estuary were identified as index sites and sampled during every sampling trip. The remaining 4-8 sites changed each quarterly sampling trip with the goal of assessing the spatial distribution of *F. jenkinsi*. Due to the extreme disturbance that Hurricane Harvey caused to the entire study area, regular monitoring for *F. jenkinsi* was suspended during the three months immediately following the hurricane (September – November 2017).

Sampling sites within each estuary were selected along an upstream-downstream axis of previously documented salinity gradients to include sites that will likely yield *F. jenkinsi* , as well as locations that would likely not meet the published salinity preferences of this species but serve as controls to help define the habitat niche of the species within Texas. Additionally, areas where the species have been detected historically were visited during this study to confirm the continued existence of this species. For example, an attempt was made to sample oligohaline tidal sites in the Mission and Aransas Rivers where *F. jenkinsi* were historically captured (Tolan et al. 2011). Based on the higher salinities found in southwestern Texas estuaries and the limited records of *F. jenkinsi,* it was expected that the occurrence of this species in these estuaries would to be rare at best, and likely represent the western extent of the range of this species.

Post-Hurricane Recovery Special Study

Two sites with historic monitoring data in Galveston Bay were chosen to monitor how saltmarsh nekton communities, including *F. jenkinsi* recovered from a disturbance event (Figure 2). Site 21 is located at an unnamed tidal tributary to the southern arm of Moses Lake (TCEQ Segment 2431C) at 25th Ave N. near Texas City, Galveston County, TX (29.408970, -94.951680) (Figure 2). Site 113 is located in an unnamed tidal tributary to Lone Oak Bayou at the FM 562 Bridge Crossing in Chambers County, TX (29.611300, -94.677060) (Figure 2). Both sites are located on relatively small tidal creeks with small contributing watersheds. While they have direct tidal access, they are both located more than 2 km upstream from the open waters of Galveston Bay. All sampling protocols remained consistent with the monitoring study. Prior to hurricane landfall, the sites were sampled every other month from February to August 2017. Post-disturbance, sites were sampled every other week through December 2017.



Sources: Esri, HERE, DeLorme, Intermap, increment P Corp., GEBCO, USGS, FAO, NPS, NRCAN, GeoBase, IGN, Kadaster NL, Ordnance Survey, Esri Japan, METI, Esri China (Hong Kong), swisstopo, MapmyIndia, © OpenStreetMap contributors, and the GIS User Community

Figure 1 Map of sampling sites (black triangles) and docmented historical occurrences (yellow star). Wetlands based on 2010 NOAA Coastal Change Analysis Program (NOAA C-CAP 2010).

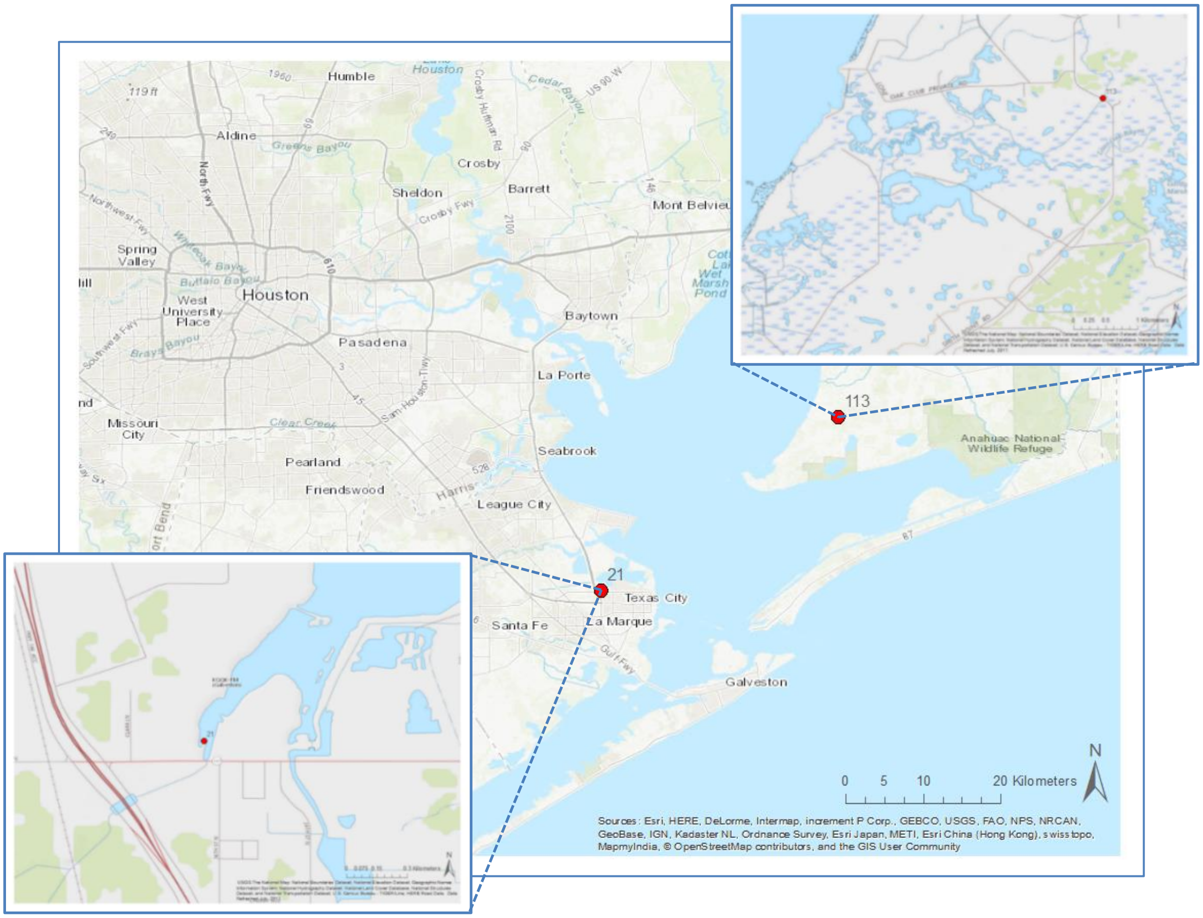


Figure 2. Map of saltmarsh nekton community post-hurricane Harvey special study sites, Galveston Bay, Texas. Coordinates of site 21 (29.408970, -94.951680) and site 113 (29.611300, -94.677060).

Sampling Methods

During each sampling event, fish collections were conducted using a straight seine (15’ x 4’,1/8 inch bar mesh) and Breder traps (Breder 1960). Three replicate seine hauls, 10 meters each, were conducted at each site. The Breder traps were constructed with clear Plexiglas (0.08’’ thickness) using the same dimensions (12'' x 6'') as Lopez et al. (2011). Three replicate traps were set facing the marsh edge at each site at deployed overnight. All Saltmarsh Topminnow and associated nekton were counted and effort was recorded. Scientific and common names followed conventions set forth by the American Fisheries Society (McLaughlin et al. 2005; Page et al. 2013). The standard length of up to five individuals from each species including Saltmarsh Topminnow were measured from each replicate. When possible, all fish except Saltmarsh Topminnow were identified in the field and released. Up to 20 Saltmarsh Topminnow from each site/event were retained for detailed demographic analysis (sex, age, weight, and morphometric) and gonad development. All retained specimens were administered a lethal dose of MS-222, fixed in a 10% buffered formalin solution, and then brought back to the laboratory where they were transferred to a 70% ethanol solution, identified to species, counted, and standard length measurements (mm) taken. A subset of specimens were placed immediately in 70% ethanol for genetic archival.

Figure 3. Example of a Breder trap set facing the marsh edge.

Additional environmental data including water depth (m), sediment type, and dominant vegetation species and density were measured for each replicate of each gear. Tide stage (flood, high slack, ebb, low slack) were recorded. Water level (ft) referenced to mean lower low water (MLLW) was obtained from the closest NOAA tide gage site. Water quality measurements including temperature (°C), dissolved oxygen (mg/L and %), conductivity (µS/cm), pH, and salinity (ppt) were collected using an YSI 600 XLM or ProDSS sonde before seining and upon retrieving traps using procedures outlined in TCEQ (2012). Tidal creek channel width in meters (if applicable) was measured with a tape measure or laser rangefinder. If the creek was too large recent Google Earth aerial imagery was used to estimate stream width. Water transparency also referred to as Secchi-disk transparency (or Secchi depth), was measured at each site using a Secchi tube during trap deployment and retrieval following procedures outlined in TCEQ (2012).

Laboratory Processing

Up to 20 *F. jenkinsi* collected per sampling event (from one or more replicates) received a unique specimen identifier. For these 20 specimens a suite of additional morphometric measurements were taken for each individual including snout length, pre-orbital length, head length, snout to dorsal fin origin, snout to anal fin origin, standard length, total length, eye diameter, body depth, caudal peduncle depth, inter-orbital space, head width, and caudal peduncle width (Figure 4). Anal fin ray count and dorsal fin ray count were recorded as well as noting the presence or absence of lateral spots and an anal sheath.

To assess the condition and reproductive condition of *F. jenkinsi*, the total weight (TW, mg) was taken for each individual. The sex of each individual was determined using a combination of the dimorphic characteristics described by Lopez et al. (2010) and observation of the extracted gonads (Figure 5 and Figure 6). Once extracted, the gonads were weighed (GW, mg) and the gonadosomatic index (GSI) calculated: [(GW / TW)*100]. Reproductive activity of female *F. jenkinsi* were further assessed by classifying the ovarian stage (Figure 5) using methods described by Brown-Peterson et al. (2011), Lopez et al. (2010), and Robertson (2016). Any anomalies (eg. parasites, cycts, tumors, etc.) observed before and during dissection were noted.

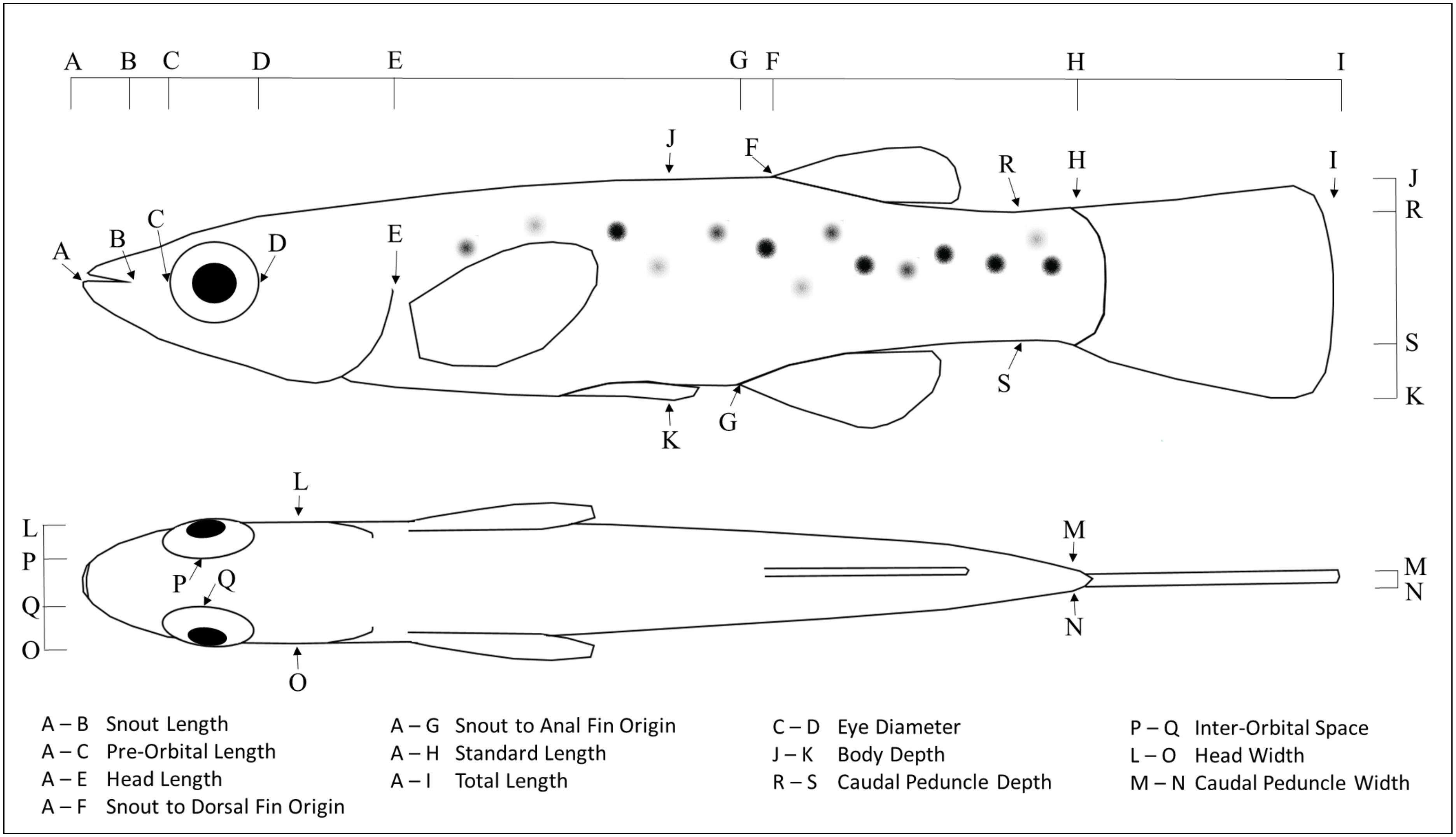


Figure 4. Morphometric measurements taken for each *F. jenkinsi* collected (up to 20 individuals per sampling event).

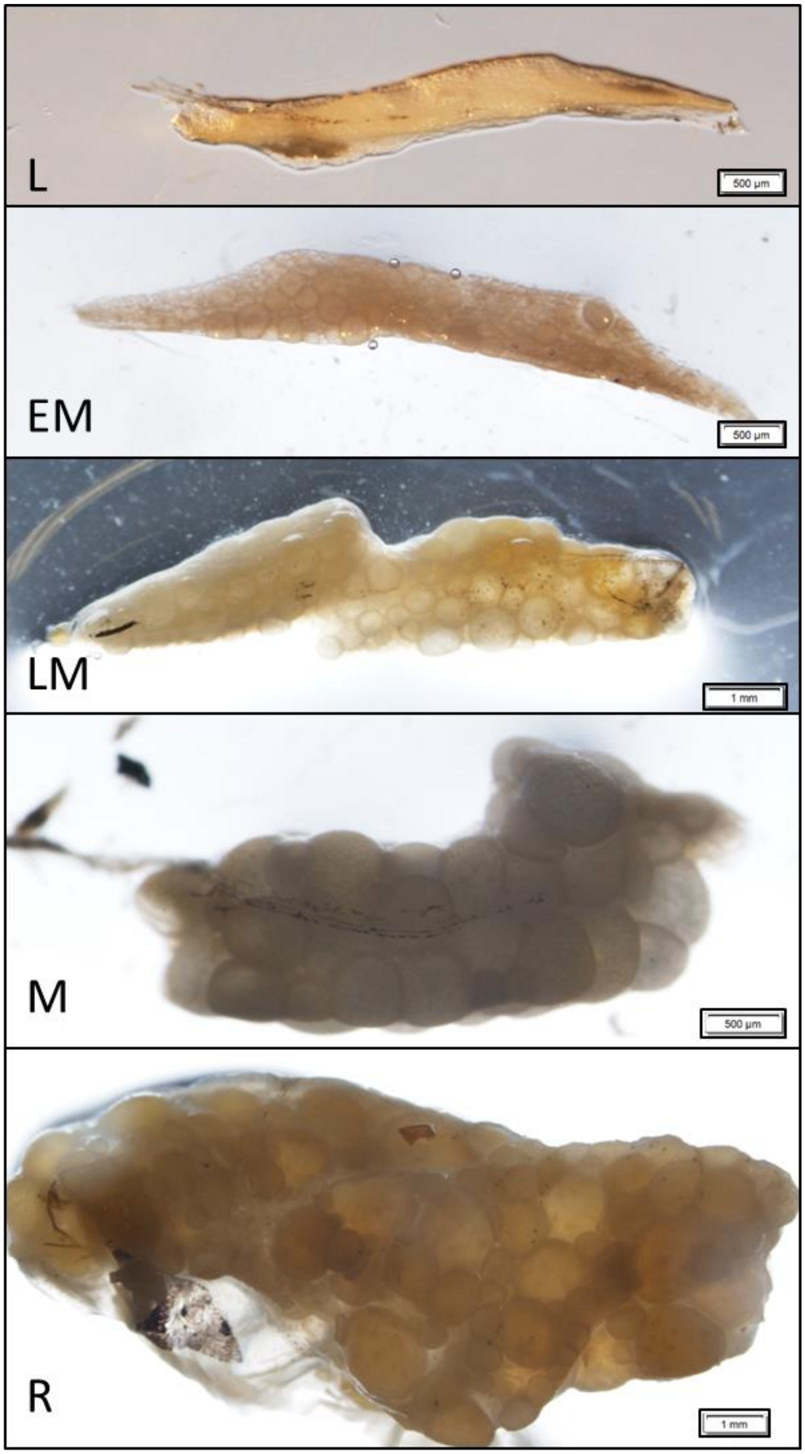


Figure 5. Example ovary stages of female *F. jenkinsi* (L) Latent (EM) Early maturing (LM) Late maturing (M) Mature and (R) Ripe.

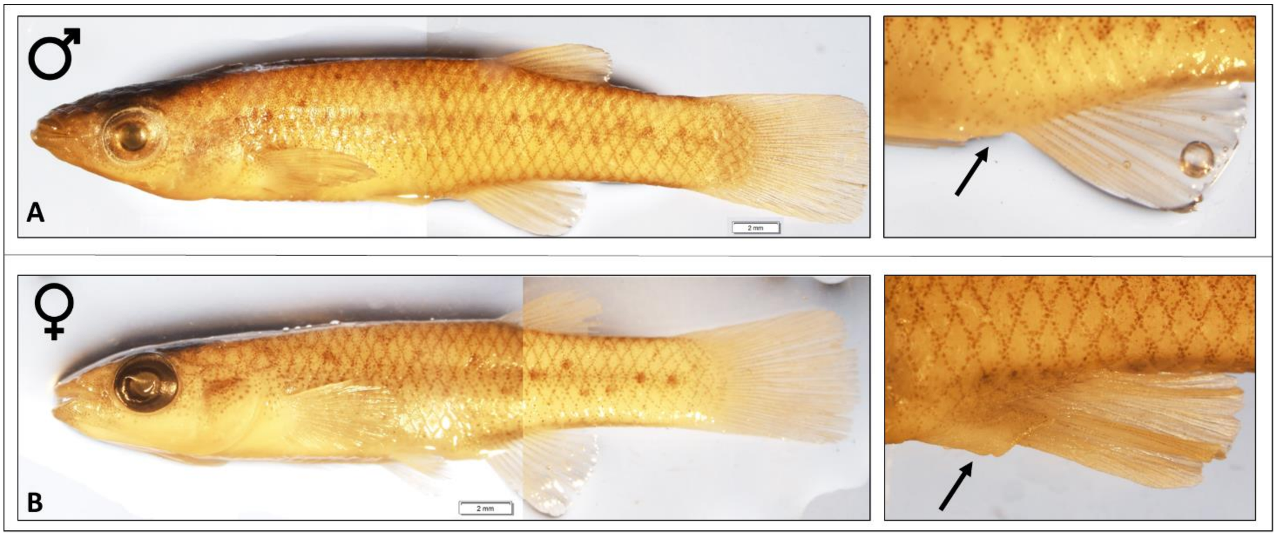


Figure 6. Example *F. jenkinsi* (A) Male genital region showing the anal fin with exposed papilla (arrow). (B) Female genital region showing the anal sheath (arrow).

Data Analysis

Nekton community structure was characterized by calculating total taxa abundance (N), relative abundance (%), taxa richness (S), Shannon-Wiener diversity (H') and Pielou's evenness index (J') (Magurran 2013) and catch per unit effort (CPUE). For the current (2017-18) CPUE was calculated for traps as catch/hr. soak time, and for seines it was calculated as catch/m^2. However, due to the lack of detailed effort data collected during 2014, CPUE was also calculated as catch/replicate for both traps and seines to facilitate comparison between both study periods (current study and Guillen et al. 2015).. Additionally, because catch data from the Guillen et al. (2015) included only fish (not invertebrates or amphibians) only data on fish collected during 2017-2018 were used for inter-study community comparisons and analyses. When catch data are presented for just the current study period (2017-18), all nekton (fish, invertebrate and amphibians) catch was included in analyses. Prior to statistical analysis, water chemistry, transparency, tide, and depth data associated with trap catch was averaged between the values measured during deployment and retrieval.

Seine and trap nekton data were analyzed separately. Nekton assemblage data were log+1 transformed and a Bray-Curtis similarity index was generated between sample collections using the PRIMER 7 statistical software package (Bray and Curtis 1957; Clarke and Warwick 2001). Site groupings based on similar species assemblages were further investigated using an analysis of similarity (ANOSIM) to test for significant patterns in community structure when *F. jenkinsi* was present. One-way ANOSIMs were used to test for significant differences in community structures. Non-metric multidimensional scaling (nMDS) plots of assemblages were also constructed in PRIMER 7 to visualize assemblage similarities.

All physicochemical and habitat variables were tested for normality prior to statistical analysis (Shapiro and Wilk 1965). If non-normal, nonparametric statistical methods were used. Statistical analyses was conducted using R 3.5.1, or Minitab 18 (R Core Team 2018, Newton 2014). The relationship between catch per unit effort (CPUE) of *F. jenkinsi* and multiple variables was evaluated using either zero-inflated binomial or Poisson linear models (R, package pscl). The relationship between the presence/absence of *F. jenkinsi* and categorical variables was evaluated using the Kruskal-Wallis one way analysis of variance on ranks with subsequent post-hoc Dunn's or Mann-Whitney tests adjusted to reduce false discovery rates when applicable (Benjamini and Hochberg 1995; Daniel 1990; Orlich 2010; Dunn 1964).

The standard length of all *F. jenkinsi* specimens collected were measured and grouped into length frequency histograms to visually assess density and age structure by month and season. We attempted to conduct modal progression analysis on monthly modal lengths using the NORMSEP routine in FISAT II (Gayanilo 2005) to estimate short-term mortality and growth. The NORMSEP routine requires sufficient replication of length data per sampling period (e.g. month) to provide reliable estimates of growth and mortality. However, we were unable to compute the model progression analysis due to low sample size. Quarterly and monthly mean GSI was calculated for both males and females and plotted to assess reproductive condition by quarter.

Ovary phases were coded (1 Latent = L, 2 Early maturing =EM, 3 Late maturing = LM, 4 Mature = M, and 5 Ripe = R) and linear regression analysis on these ranked scores were conducted using Minitab 18 to evaluate potential relationships between standard length and total weight factors versus female and male GSI values. Linear regression analysis was also used to investigate the relationship between season, standard length, and GSI factors versus ovary development. Statistical significance was determined at the α-level of 0.05 for methods used.

Simple univariate correlation analysis was conducted between external morphometric variables including the presence (coded 1) and absence (coded 0) of an anal sheath, and the dependent variable sex (coded 1 = male, 0 = female). The strength of the relationship was measured using Pearson's coefficient (r). Candidate variables exhibiting a strong (high r) and significant ($p < 0.05$) linear relationship with sex were evaluated further using stepwise logistic regression.

Stepwise logistic regression was conducted using the Minitab 18 statistical software package in an effort to find one or more external morphometric traits that may be useful in differentiating the sex of individual fish. Logistic regression analysis is used to investigate the association between a categorical dependent variable and a set of independent variables, which can be continuous, count, or categorical data. The name logistic regression refers to the fact that the dependent variable has only two values, such as 0 = female and 1 = male. (Hosmer et al. 2013; Peng and So 2002). The generic form of the logistic regression that relates the probability of an event occurring (π) and significant predictors (x_i) is shown below:

$$\pi = \frac{1}{1+e^{-(\alpha+\beta_1 x_1+.....\beta_i x_i)}}$$

We sought to define the probability (π) an individual fish being a male versus the probability ($1- \pi$) of it being a female. Another way of expressing this is the odds of a fish being a female versus being a male.

Odds = $\dfrac{\pi}{1-\pi}$ Example: 0.8 ÷ (1.0-0.8 = 0.2) = 4 to 1 odds of specimen being a male versus female.

The logistic model can be rearranged to predict the logit of Y (e.g. Odds of different sex) from X_i (one or more independent variables). The logit is the natural logarithm (ln) of odds of each sex. The simple logistic model in this form is displayed below:

$\ln\left(\dfrac{\pi}{(1-\pi)}\right)$ = log (odds) = logit = 👌 + βX, where $\beta X = \beta_1 x_1 + \ldots \beta_i x_i$

Hence, π is equal to the probability of (Y = outcome of interest (e.g. sex = male) |X = x_i) = $\dfrac{e^{a+\beta X}}{(1+e^{a+\beta X})}$

Where π is the probability of the outcome of interest (e.g. sex classification), under variable Y, 👌 is the Y intercept and β is the slope parameter. X can be categorical or continuous, whereas Y is always categorical. For logistic regression an “analysis of deviance table” is generated that provides a summary of significance tests analogous to an analysis of variance table for linear regression (Hosmer et al. 2013; Newton 2014). Individual scatterplots and histograms were also constructed to support logistic analysis.

Estimation of Life History Parameters

The Food and Agricultural Organization (FAO) has provided an online tool to estimate key life history population parameters on their FishBase web site. The FAO created a life history “Key Facts” page that provides tools to generate estimates with margins of error (95% confidence intervals) for multiple life-history parameters of fishes, which contains about 2/3 of the range of the observed values. As a default the tools use the best available data in FishBase as input for various computational equations. Users can replace these default values with their own estimates and recalculate the life history parameters.

The FishBase tool was used to estimates the various parameters of the von Bertanlanfy growth equation and instantaneous natural mortality for *F. jenkinsi.* Estimation of these parameters was aided by Excel spreadsheets programmed with the necessary equations that are available at the web site. The web links are provided below.

https://www.fishbase.se/summary/Fundulus-jenkinsi.html

https://www.fishbase.se/popdyn/KeyfactsSummary_2v2.php?ID=3193&GenusName=Fundulus&SpeciesName=jenkinsi&vStockCode=3389&fc=570

Due to the paucity of data FAO provided default values for *F. jenkinsi* based on family wide (e.g. Fundulidae) information or from a limited studies.

The primary objective of the current study was to provide preliminary estimates of the parameters of the von-Bertalanffy growth equation and other demographic parameters such as instantaneous rate of natural mortality (M) (von Bertalanffy 1938; Haddon 2011). The widely used von-Bertanlanfy equation is defined in equation 1 below.

Equation 1. The von Bertanlanffy growth model $L_t = L_\infty(1-e^{-K(t-t_0)})$

Normally length at age (dependent variable) is plotted against age (independent variable) and the parameters are solved by fitting the line using computer intensive non-linear curve maximum likelihood methods which minimizes the sum of square deviations between actual and predicted lengths at age. This is easily accomplished using tools like Solver in Excel (Haddon 2011). In the von Bertanlanffy equation L_t = equal to the length at age t; t = age; L_∞ = asymptotic average body length estimated from fitting of the data to the equation, K = growth rate coefficient estimated from fitting the equation to the data, and t_0 = is the theoretical age at which specimen has zero length (used to fix horizontal position on graph; usually not zero). Since we did not directly age any of the specimens collected during the study it is impossible to directly fit the equation. Instead we utilized a variety of indirect methods based on empirical relationships to obtain preliminary estimates of the parameters of the von Bertanlanffy equation and subsequently used those parameters to estimate natural mortality.

The maximum length of an organism is a strong predictor for many life history parameters (Blueweiss et al. 1978). The default value used here is the maximum length (L_{max}) ever reported for the species. If no other data are available, this value is used to estimate asymptotic length (L_∞) and length at first maturity (L_m). The point estimate of asymptotic length, L_∞ and the corresponding 95% confidence interval are estimated from maximum length (L_{max}) using an empirical relationship between the two parameters (Froese and Binohlan 2000). Equation 2 illustrated below was empirically derived by Froese and Binohlan (2000) to estimate L_∞ from L_{max}.

Equation 2. $\mathbf{Log_{10} L_\infty = 0.044+0.9841+Log_{10} (L_{max})}$

The length of maturity (L_m) was calculated using the empirical formula provided by Froese and Binohlan (2000) and incorporated into FishBase online calculator.

Equation 3. $\mathbf{Log_{10} L_m = 0.8979 * Log_{10} (L_\infty)}$

Time (age) of maturity was estimated upon visual examination of length frequency distributions. Using this as input for t_m we estimated the likely maximum lifespan t_{max} using the following formula:

Equation 4. $\mathbf{Log_{10} t_{max} = 0.5496+0.957 * log_{10} t_m.}$

The parameter t_{max} was used to estimate the growth parameter K using the following equation.

Equation 5. $\mathbf{K = 3/t_{max}}$

The final parameter of the von Bertalanffy growth function which needed to be estimated is t_0 which is defined as the hypothetical age (in years) the fish would have had a zero length, had their early life stages grown in the manner described by the growth equation which for most fishes is not the case. It is a constant inserted to improve the model fit, similar to intercept in a linear regression model, and its effect is to move the origin of the growth curve sideways (shift it) along the X-axis without affecting either K or L_∞. We used an empirical equation (Pauly 1979) to estimate a default value for t_0 from L_∞ and K. The formula is listed below in Equation 6.

Equation 6. $\log(-t_0) = -0.3922 - 0.2752 \log L_\infty - 1.038 \log K.$

A commonly used indirect method used to estimate instantaneous natural mortality (M) was described by Pauly (1980). He estimated instantaneous natural mortality from the parameters of the von Bertalanffy growth equation parameters (L_∞ and K) and mean environmental temperature (T, in degrees Celsius).

Equation 7: $\log_{10} M = -0.0066 - 0.279 \log_{10} L_\infty + 0.6543 \log_{10} K + 0.4634 \log_{10} T$

This method assumes that there is a relationship between size and natural mortality. This relationship is usually relatively weak on its own, but the inclusion of average annual environmental temperature improves the model fit as poikilotherms living in warmer water will usually have higher mortality rates than an equivalent organism living in colder water (Pauly 1980). The relationship was developed based on independent estimates of natural mortality and ambient temperature data for 175 fish stocks of fish (Pauly 1980).

Results

Distribution and Abundance

A total of 162 sites were sampled along the upper to middle Texas coast from February 7, 2017 to March 7, 2018. The distribution of historical sightings along with the locations and relative abundance where *F. jenkinsi* were captured during this study and the previous 2014-15 surveys are depicted in (Table 1-2, and Figure 7).

A total of 24 sites were sampled in Sabine Lake (Table 1 and Figure 8). Saltmarsh Topminnow were captured at 9 sites (37.5% of the sites) (Figure 8). When present, the average density of *F. jenkinsi* was nine individuals per sampling event. The detection rate of *F. jenkinsi* between the Guillen (2015) study and the current study were similar with a total of 53 sites sampled in 2014-15, and *F. jenkinsi* captured at 14 of them (26.4%). For the 2015 study, when present, the average density of *F. jenkinsi* was twelve individuals per sampling event. Within the Sabine Lake system, *F. jenkinsi* were present along the Sabine and Neches Rivers and tributaries, as well as a few occurrences near Keith Lake, which connects to the Port Arthur Ship Channel. Additionally, four sites were sampled in Spindletop Gully in Jefferson County, which drains into the Gulf Intercostal Waterway, of which *F. jenkinsi* were detected at two of the sites.

Saltmarsh Topminnow were captured at 10 out of 31 sites (32.3% of the sites) within Galveston Bay (Figure 7 and 9). The detection rate of *F. jenkinsi* documented by the Guillen (2015) study was lower than the current study with a total of 81 sites sampled in 2014-15, and *F. jenkinsi* captured at only 11

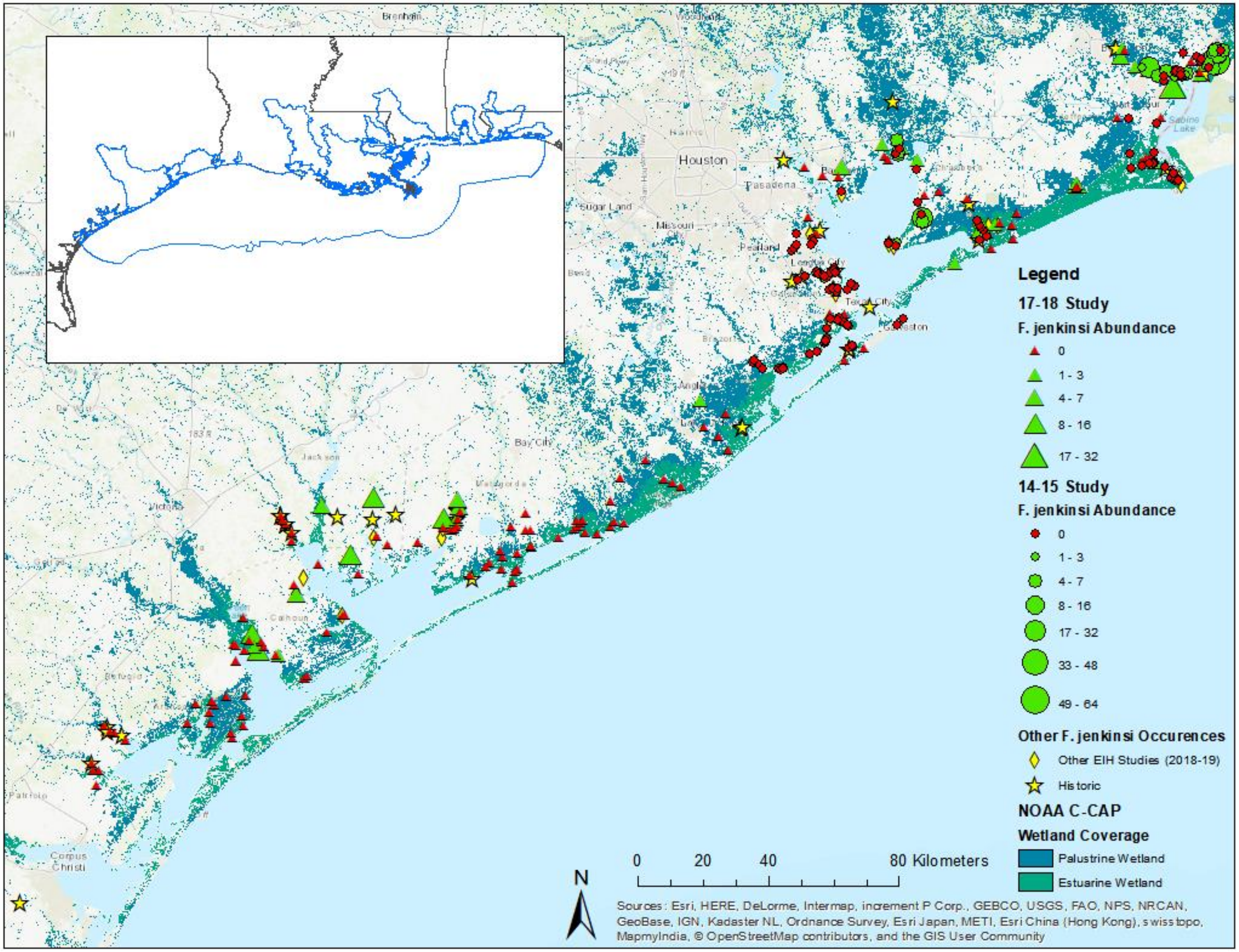
(13.5%) of them. When present, the average abundance of *F. jenkinsi* was 15 individuals per sampling event during 2017-18 in contrast it was five individuals per sampling event in 2014-15. Within the Galveston Bay system, *F. jenkinsi* was not detected in West Bay and Clear Lake, however one individual was collected in Bastrop Bayou, which is a tributary to Bastrop Bay (Figure 9). Attempts were made to locate *F. jenkinsi* in the San Bernard and Brazos River Watersheds, but none were captured at the six sites sampled (Figure 7). A total of 21 sites were sampled in the East Matagorda Bay System, and no *F. jenkinsi* were captured (Figure 7).

There have been a number of historic observations in the Matagorda Bay system (Figure 10). A total of 34 sites were surveyed, and at 9 of those, *F. jenkinsi* were captured (26.5%) (Table 1 and Figure 10). When present, the average abundance of *F. jenkinsi* was six individuals per sampling event. Within the Matagorda Bay system *F. jenkinsi* were generally detected a considerable distance upstream from the open bay.

Although there were no historical records of *F. jenkinsi* in the San Antonio Bay system, *F. jenkinsi* were captured at 4 of the 30 (13.3%) sites surveyed during the current study (Table 1 and Figure 11). The average abundance of *F. jenkinsi* was seven individuals per sampling event at sites where it was captured. A total of 8 sites were sampled in the Copano Bay (Mission and Aransas Rivers) watershed focusing on areas of historic reports of *F. jenkinsi* (Figure 12). However, this effort failed to capture any new specimens of *F. jenkinsi*.

Table 1. Summary of all sites sampled from February 2017 to March 2018 representing bay systems where *Fundulus jenkinsi* was captured as a proportion of the total number of sites (% Sites w/ *F. jenkinsi*) and of the number of individuals captured at sites where it was present in each Texas estuary. Data does not include the 15 specimens captured during the Hurricane Harvey special study.

Spatial Occurrence			***F. jenkinsi* Density**		
Estuary	% Sites w/ *F. jenkinsi*	Total # of Sites	Min-Max	Average per Event	Number of *F. jenkinsi*
Sabine Lake	37.5	24	1 - 84	17	200
Chambers-Jefferson County	25.0	8	1 - 4	3	5
Galveston Bay	32.3	31	1 - 110	15	190
San Bernard/Brazos	0.0	6	-	-	0
East Matagorda Bay	0.0	21	-	-	0
Matagorda Bay	26.5	34	1 - 16	8	77
San Antonio Bay	13.3	30	1 - 14	7	29
Aransas Bay	0.0	8	-	-	0
Grand Total		**162**			**501**



Sources: Esri, HERE, DeLorme, Intermap, increment P Corp., GEBCO, USGS, FAO, NPS, NRCAN, GeoBase, IGN, Kadaster NL, Ordnance Survey, Esri Japan, METI, Esri China (Hong Kong), swisstopo, MapmyIndia, © OpenStreetMap contributors, and the GIS User Community

Figure 7. Study sites and *F. jenkinsi* abundance during the current study (triangles), and Guillen (2015) (circles), historic occurrences (yellow stars), and more recent occurrences from other EIH studies (yellow diamonds). Wetland coverage depicting all estuarine and palustrine wetland areas (Source: NOAA C-CAP 2010).

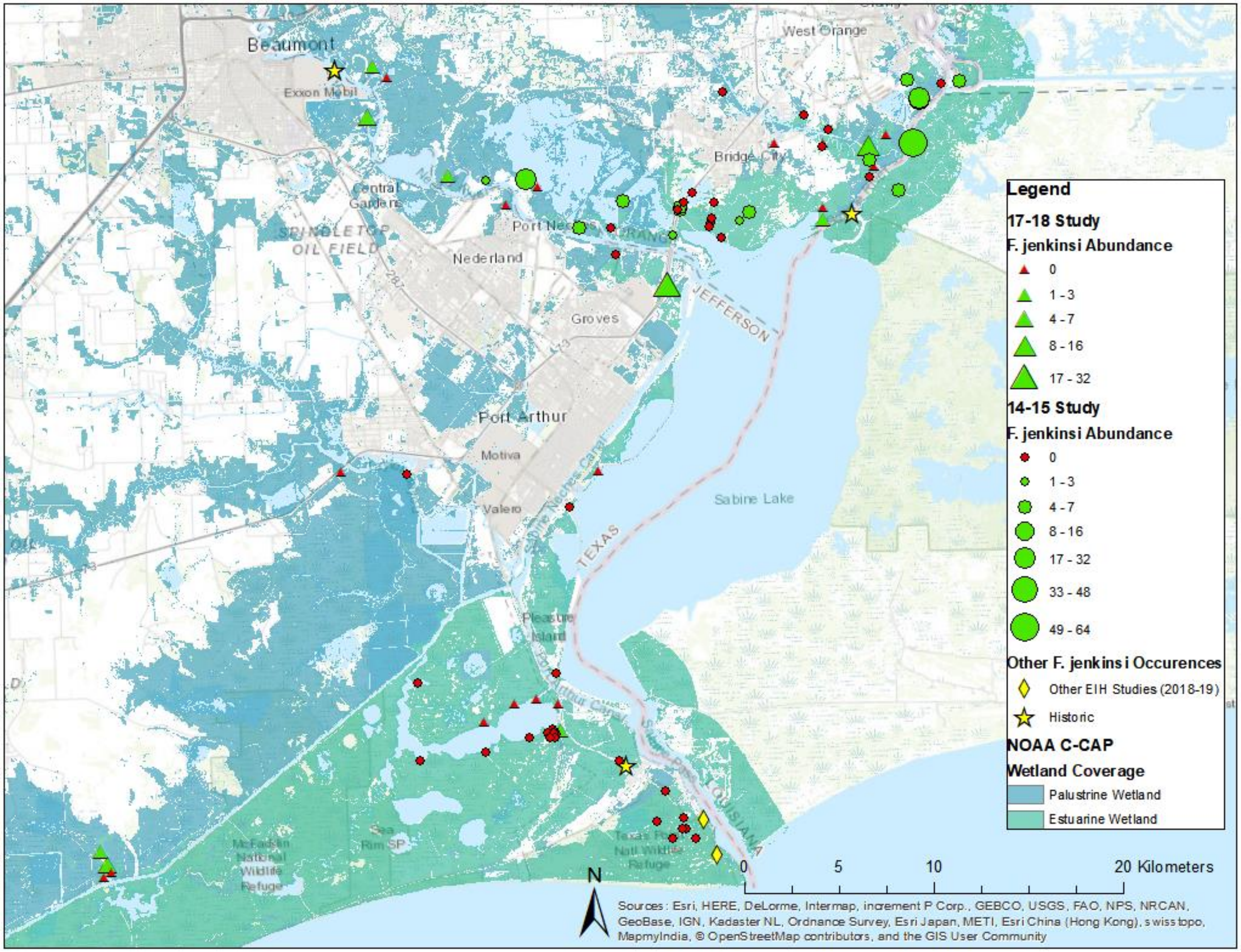


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Figure 8. Study sites and *F. jenkinsi* abundance within the Sabine Lake estuary. Current study (triangles), and Guillen (2015) (circles), historic occurrences (yellow stars), and more recent occurrences from other EIH studies (yellow diamonds). Wetland coverage depicting all estuarine and palustrine wetland areas (Source: NOAA C-CAP 2010).

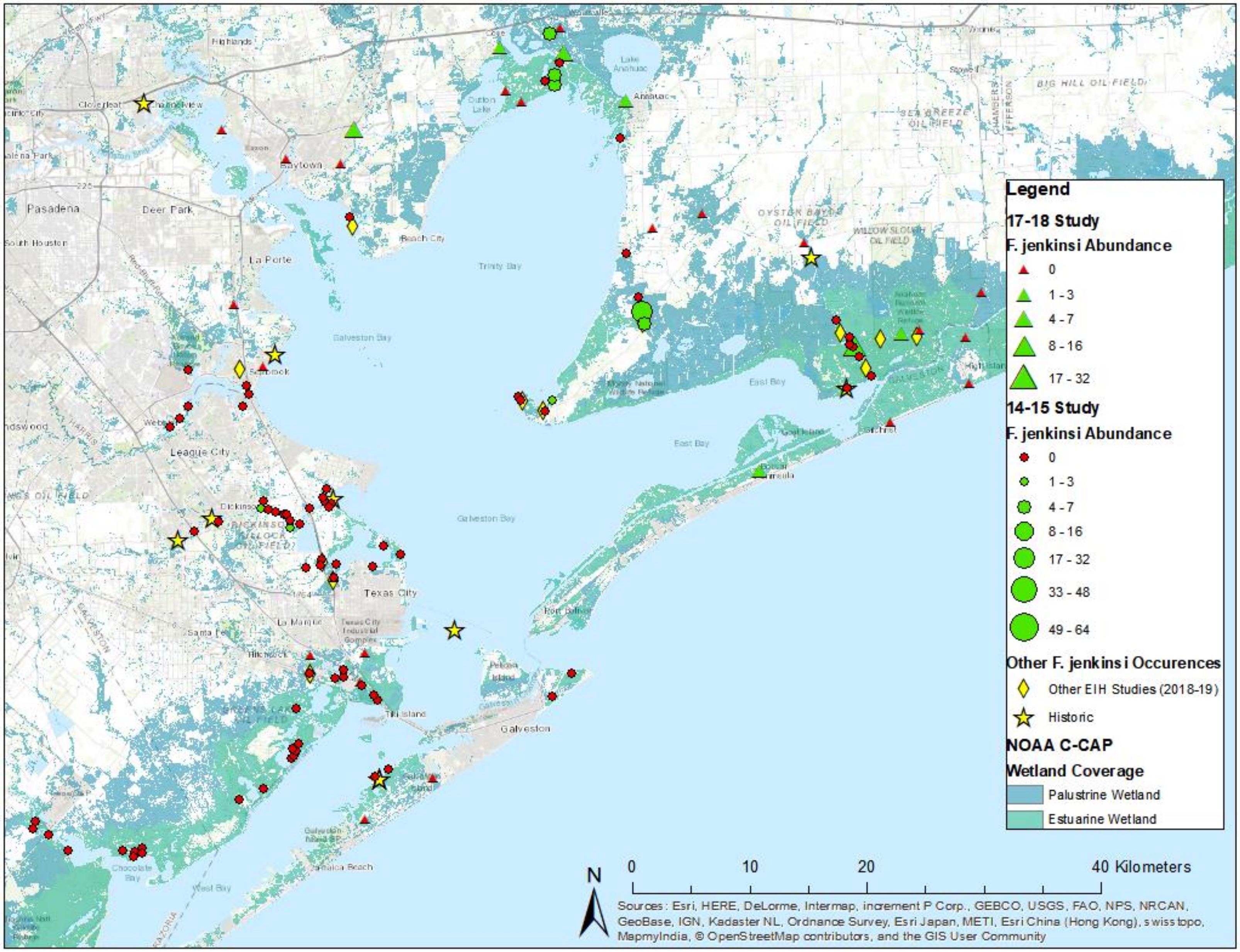

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Figure 9. Study sites and *F. jenkinsi* abundance within the Galveston Bay estuary. Current study (triangles), and Guillen (2015) (circles), historic occurrences (yellow stars), and more recent occurrences from other EIH studies (yellow diamonds). Wetland coverage depicting all estuarine and palustrine wetland areas (Source: NOAA C-CAP 2010).

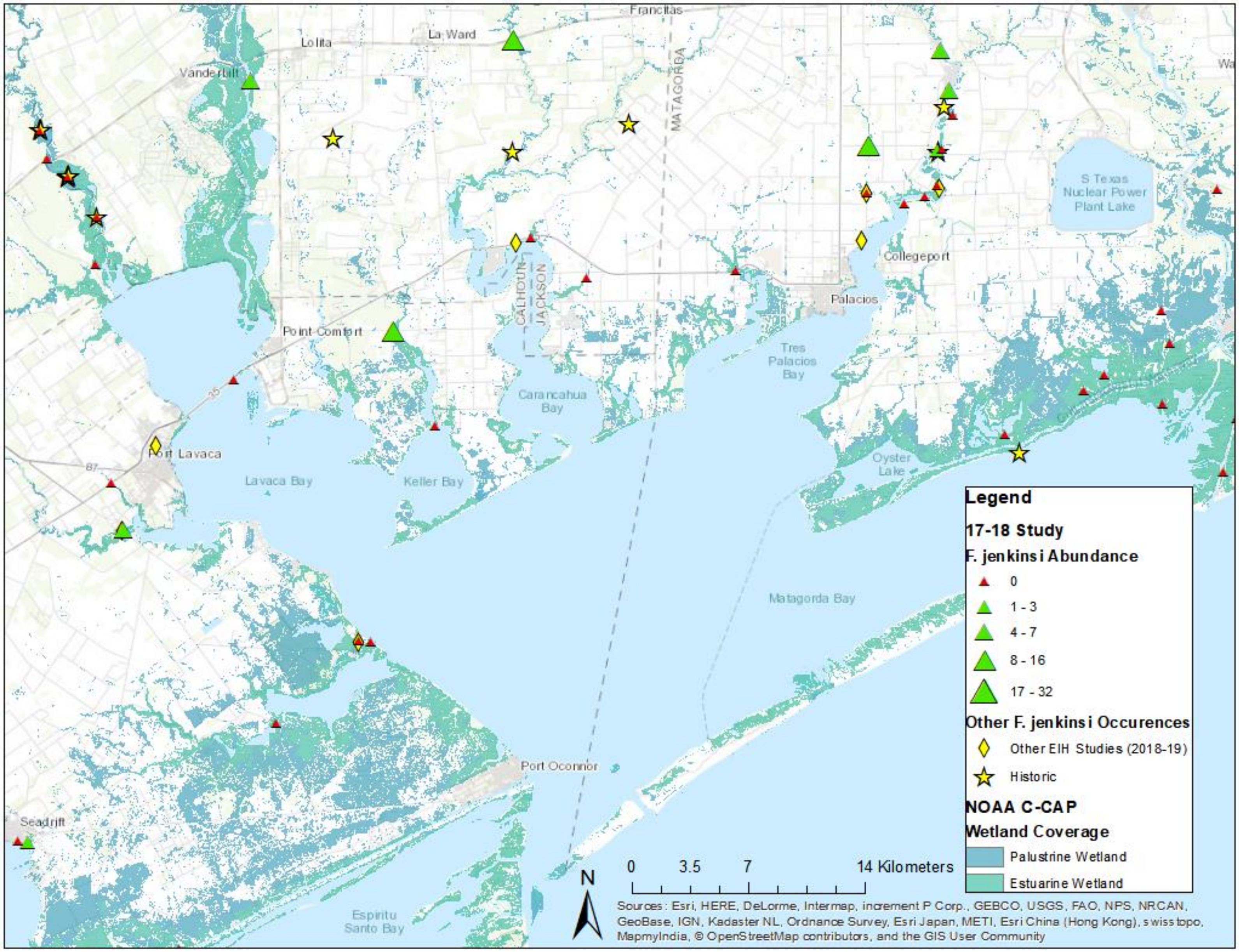

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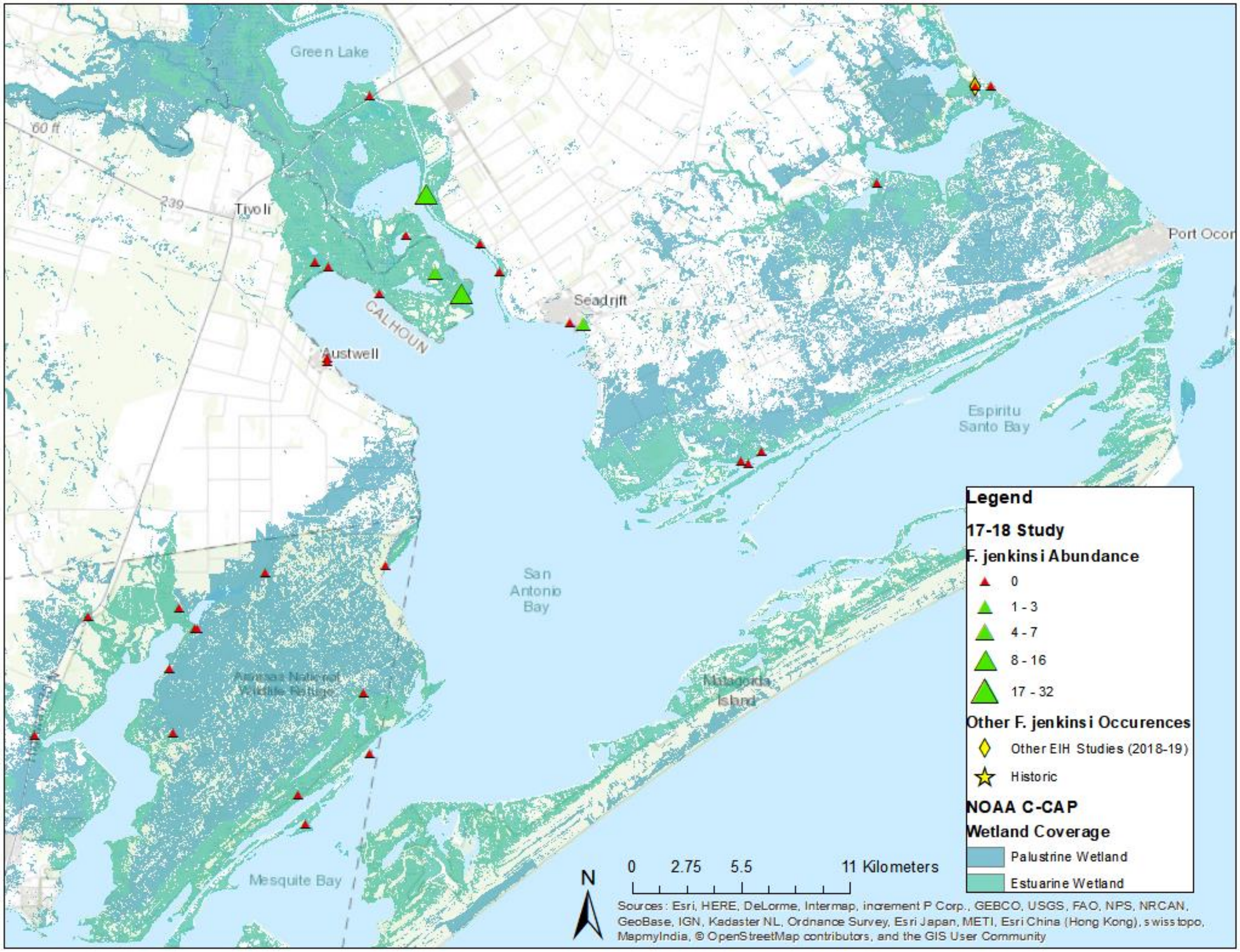
Figure 10. Study sites and *F. jenkinsi* abundance within the Matagorda Bay estuary. Current study (triangles), historic occurrences (yellow stars), and more recent occurrences from other EIH studies (yellow diamonds). Wetland coverage depicting all estuarine and palustrine wetland areas (Source: NOAA C-CAP 2010).



Sources: Esri, HERE, DeLorme, Intermap, increment P Corp., GEBCO, USGS, FAO, NPS, NRCAN, GeoBase, IGN, Kadaster NL, Ordnance Survey, Esri Japan, METI, Esri China (Hong Kong), swisstopo, MapmyIndia, © OpenStreetMap contributors, and the GIS User Community

Figure 11. Study sites and *F. jenkinsi* abundance within the San Antonio Bay estuary. Current study (triangles), historic occurrences (yellow stars), and more recent occurrences from other EIH studies (yellow diamonds). Wetland coverage depicting all estuarine and palustrine wetland areas (Source: NOAA C-CAP 2010).

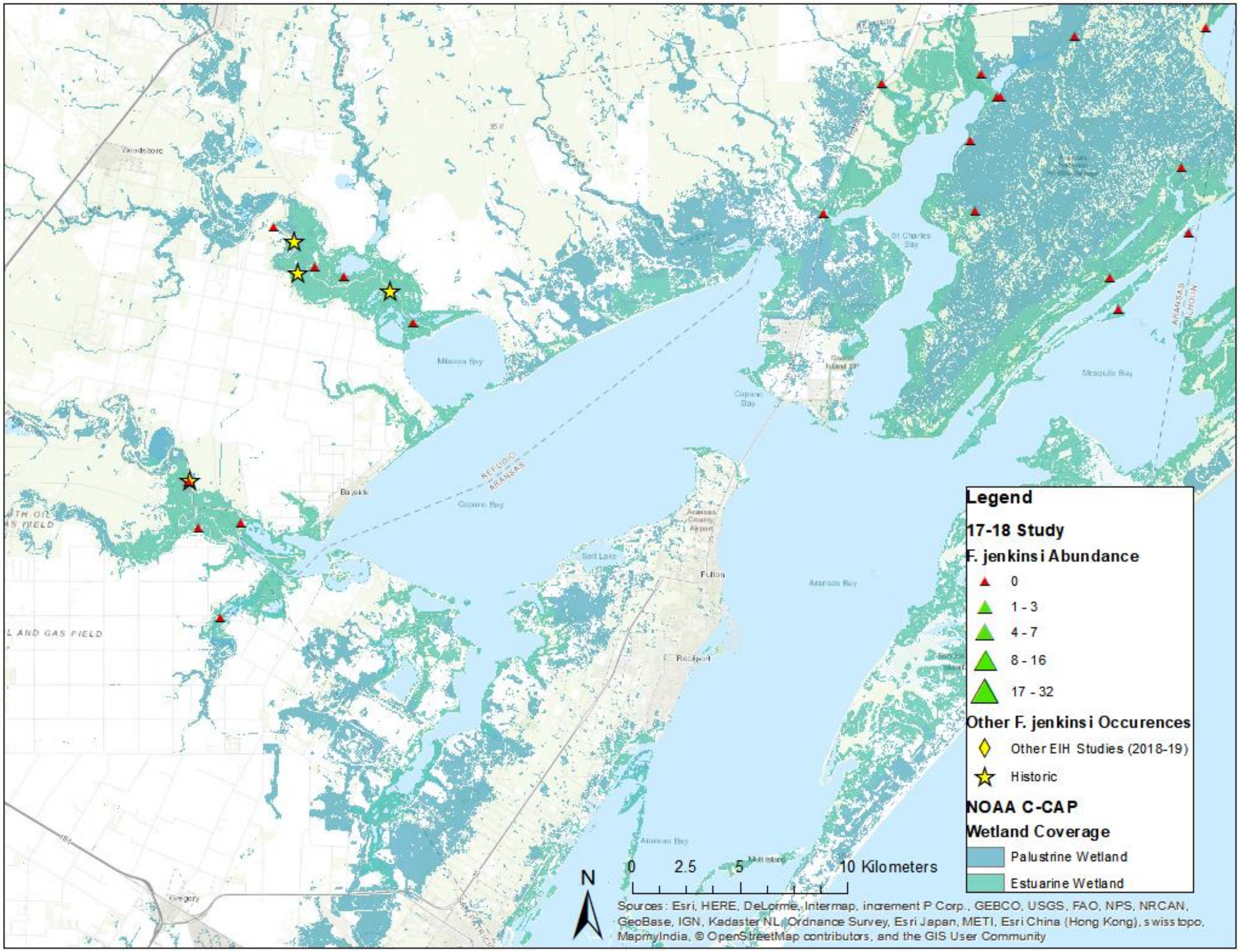


Figure 12. Study sites and *F. jenkinsi* abundance within the Aransas Bay Area. Current study (triangles), historic occurrences (yellow stars), and more recent occurrences from other EIH studies (yellow diamonds). Wetland coverage depicting all estuarine and palustrine wetland areas (Source: NOAA C-CAP 2010).

Table 2. Summary of all sites sampled from February 2014 to March 2015 representing bay systems where *F. jenkinsi* was captured as a proportion of the total number of sites (% sites w/ *F. jenkinsi*) and of the number of individuals captured at sites where *F. jenkinsi* was present (Source: based on subsequent analysis of data collected by Guillen et al. 2015).

Spatial Occurrence			***F. jenkinsi* Density**		
Estuary	**% Sites w/ *F. jenkinsi***	**Total # of Sites**	**Min.-Max.**	**Average per Event**	**No. of *F. jenkinsi***
Sabine Lake	26.4%	53	2-64	12	161
Galveston Bay	13.5%	81	1-22	5	54
Grand Total		**134**			**215**

A Kruskal Wallis test of CPUE of *F. jenkinsi* from all estuaries where *F. jenkinsi* were present (Sabine, Galveston, Matagorda, and San Antonio) failed to detect any significant difference in seine catch rates (p = 0.1803) or trap (p = 0.134) data. However visually, there appeared to be a general linear decrease in *F. jenkinsi* CPUE from east to west (Figure 13). Furthermore, this east to west trend of decreasing abundance was significant (p = 0.0014) when *F. jenkinsi* density was modeled by longitude using a zero-inflated Poisson (ZIP) model (Figure 14). This same trend was also documented using a binomial general linear model (GLM) (seine p = 0.00169; trap p = 0.0139) where the probability of catching *F. jenkinsi* increased eastward within the study area (Figure 14). While not depicted here, a similar trend was observed with trap data.

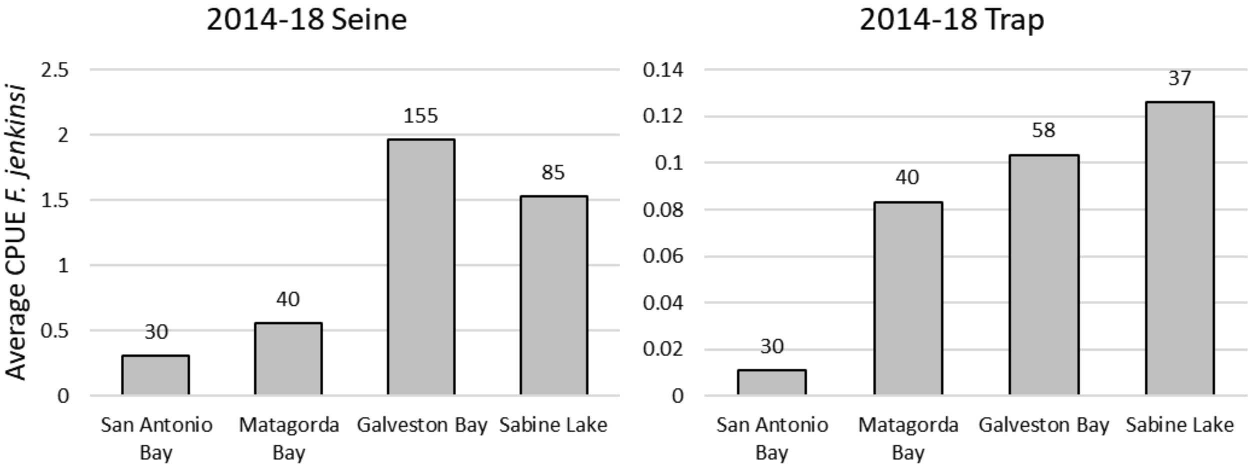


Figure 13. Average catch per unit effort (CPUE) of *F. jenkinsi* by estuary for all 2014 – 2018 data by seine and trap gear types. The total number of replicates (n) used to estimate the average estuary CPUE is depicted as a number above each bar.

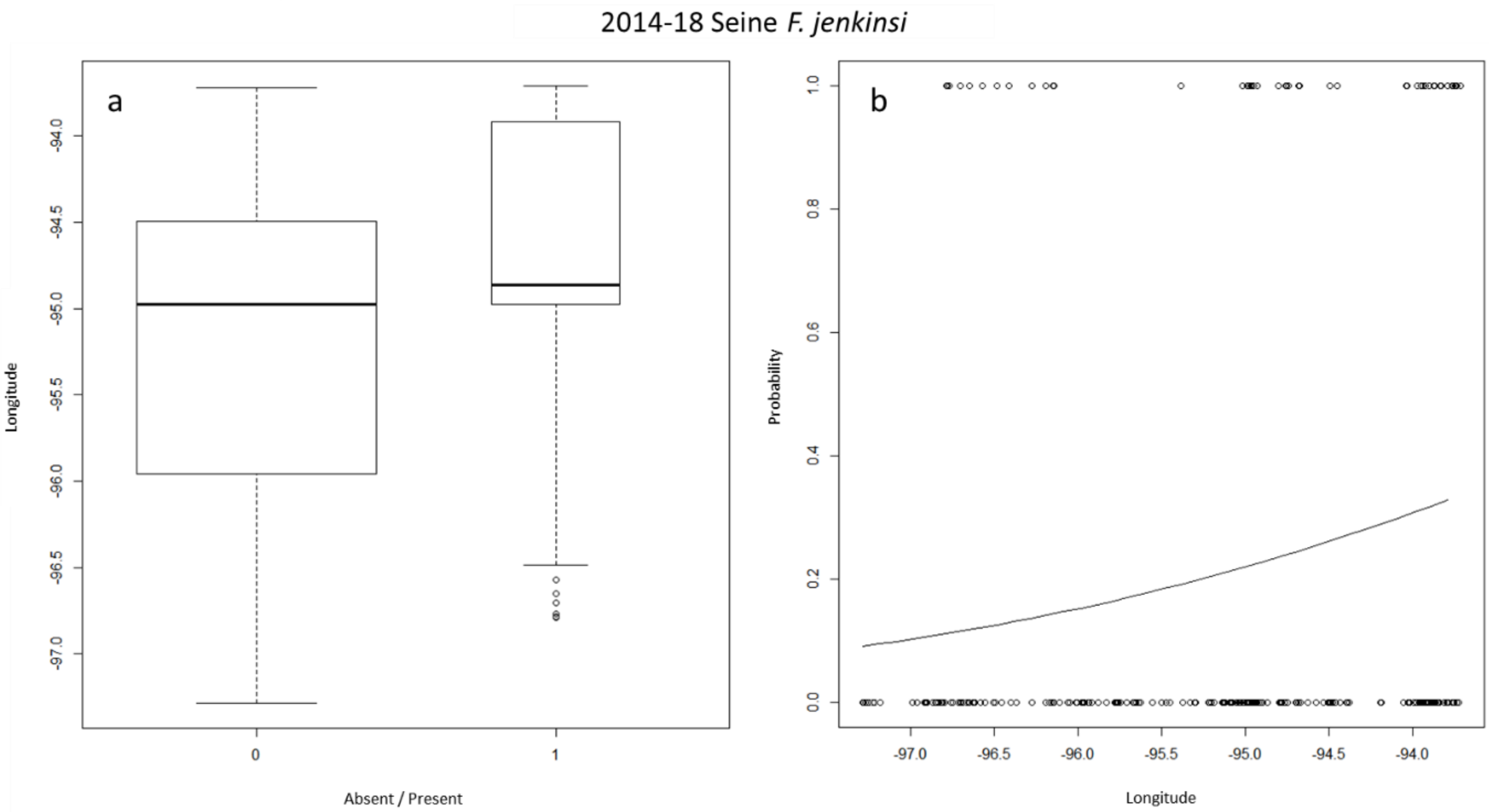


Figure 14. a) Boxplot of the longitude of sites where *F. jenkinsi* was present versus not captured in 2014-18 seine samples and b) fitted binomial GLM applied to the presence/absence of *F. jenkinsi* by longitude. A similar trend was observed with trap data.

Fish Community Composition

A total of 343,663 specimens consisting of 37 families and 88 species of fish were collected during all sampling events in 2014-2018. The five most abundant families and the corresponding species captured during all sampling events are summarized in Table 3. The families Poeciliidae, Clupeidae, Engraulidae, Atherinopsidae, and Fundulidae cumulatively represented nearly 84% of the total catch. Four species, including *Brevoortia patronus*, *Anchoa mitchilli*, *Gambusia affinis,* and *Poecilia latipinna,* represented the top 66% of all species collected numerically (Table 3).

Table 3 List of relative abundances of the five most abundant fish families and their species composition across all fish data (N_{Fish}) at all quarterly sites and relative abundances of the two most abundant invertebrate families and their species composition observed across all catch data (N_{Total}) at all quarterly sites.

Fish Abundance			
Family	% of N_{Fish}	Species	% of N_{Fish}
Poeciliidae	27.6	*Gambusia affinis*	14.1
		Poecilia latipinna	13.1
		Peocilia formosa	0.2
		Heterandria formosa	0.2
Clupeidae	25.0	*Brevoortia patronus*	24.7
		Dorosoma petenense	0.2
		Dorosoma cepedianum	<0.1
Engraulidae	14.2	*Anchoa mitchilli*	14.2
Atherinopsidae	9.6	*Menidia beryllina*	9.6
Fundulidae	7.4	*Adinia xenica*	3.3
		Lucania parva	1.6
		Fundulus pulvereus	1.0
		Fundulus grandis	0.9
		Fundulus jenkinsi	0.5
		Fundulus chrysotus	<0.1
		Fundulus similis	<0.1
		Fundulus notatus	<0.1
		Lucania goodei	<0.1
All Other Families	16.3	All Other Species	16.4
Invertebrate Abundance			
Family	% of N_{Total}	Species	% of N_{Total}
Palaemonidae	63	*Palaemonetes pugio*	63
Penaeidae	4	*Farfantepenaeus aztecus*	2
		Litopenaeus setiferus	2
Portunidae	1	*Callinectes sapidus*	1
All Other Families	32	All Other Species	32

Seasonality

A total of 511 *F. jenkinsi* individuals were caught over the course of this study from all Texas estuaries. *Fundulus jenkinsi* were collected during every seasonal sampling event however catch was highest during the Dec – Feb quarter (Table 4). Seasonal variation of CPUE of *F. jenkinsi* catch was highly significant (Kruskal-Wallis H1 = 17.46, *p* = 0.001).

To minimize between site variation in order to focus on and better visualize seasonal changes in fish community assemblage a one-way ANOSIM on site assemblages using only index sites (sites sampled seasonally throughout the study) was conducted. The assemblages found at index sites differed significantly between seasons (Global R= 0.22; *p*= 0.001). Non-metric MDS plot of fish assemblages for index sites across seasons show a clear cyclical seasonal pattern in fish assemblage similarity within Texas estuaries (Figure 15). Since there was no significant difference in *F. jenkinsi* CPUE among index sites (ANOVA p = 0.914) all index sites were pooled, and the average *F. jenkinsi* CPUE was plotted by month showing that *F. jenkinsi* were captured in greatest numbers in the winter (Figure 16). The *F. jenkinsi* caught with seines exhibited highly significant (Kruskal-Wallis p = 3.042e-07) seasonal differences in catch per unit effort. Subsequent multiple comparison tests (Mann Whitney test with BH adjustment for false discovery rates) found significant differences between all seasonal pair groups except for Fall and Spring, and Fall and Winter. We failed to detect any significant seasonal differences in *F. jenkinsi* CPUE for trap samples.

Table 4. Total number (N) of *F. jenkinsi* individuals captured during quarterly sampling events from December 2016 – March 2018. * Data represents Post-Hurricane Harvey recovery period sampling for *F. jenkinsi* at two Galveston Bay index sites.

***F. jenkinsi* Quarterly Catch**							
Quarter	**Sabine Lake**	**Chambers and Jefferson**	**Galveston Bay**	**East Matagorda**	**Matagorda Bay**	**San Antonio Bay**	**Total N**
Dec16-Feb17	98	--	55	--	--	--	153
Mar-May17	2	--	10	0	49	0	61
Jun-Aug17	7	--	3	0	0	12	22
Sep-Nov17*	--	--	15	--	--	--	15
Dec17-Mar18	93	5	122	0	28	17	265
Grand Total	**200**	**5**	**205**	**0**	**77**	**29**	**516**

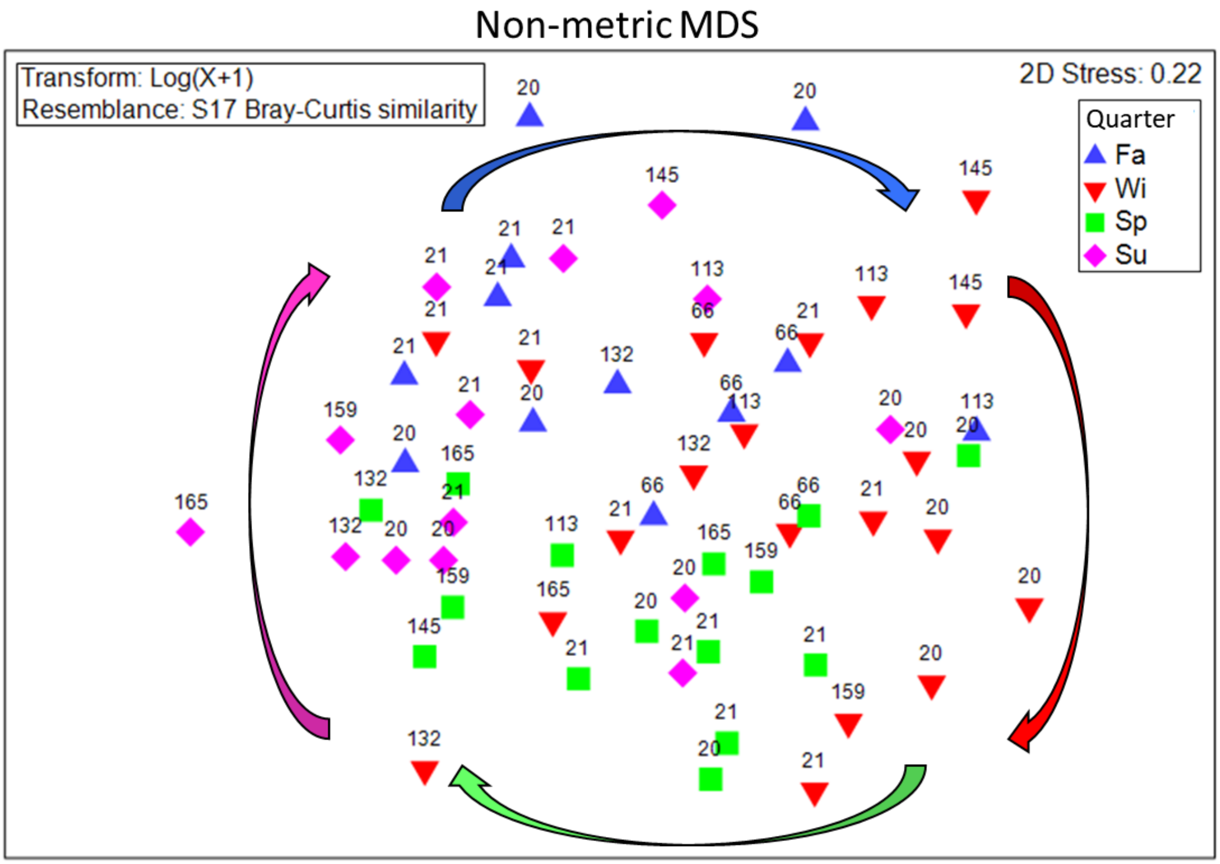


Figure 15. nMDS ordination plot of fish assemblage data illustrating the similarity of fish assemblages collected with seines at index sites by seasons, fall (blue triangles), winter (red upside-down triangles), spring (green squares), and summer (pink diamonds). Site number reported above each event. (Primer 7).

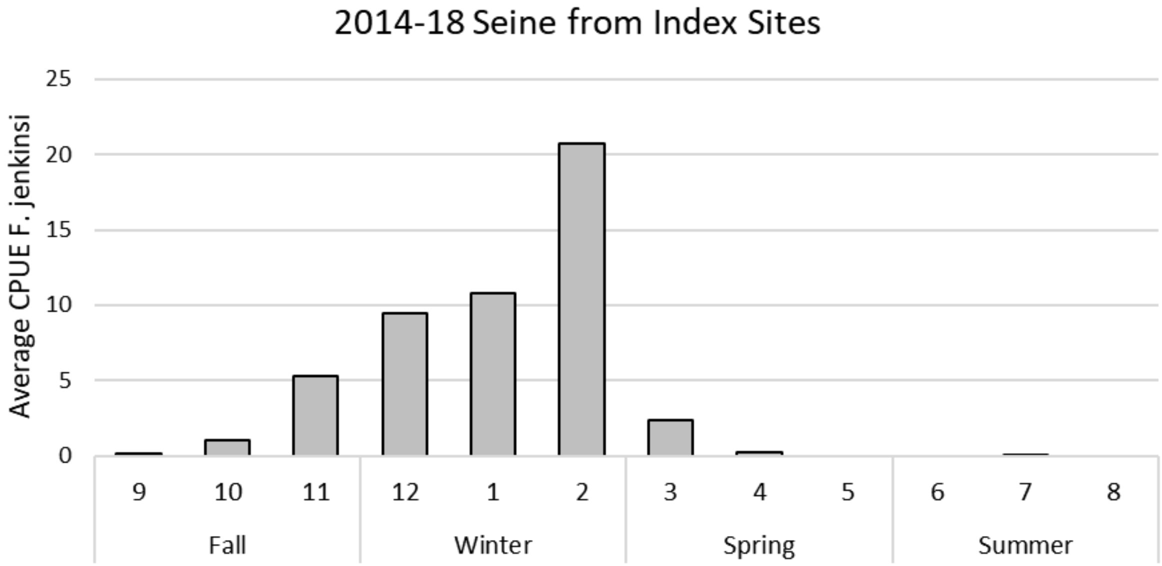


Figure 16. Average catch per unit effort (CPUE) of *F. jenkinsi* collected in seine samples at index sites from 2014-18 by month/season.

Assemblage Associations

The mean fish species diversity (H’) of sites containing *F. jenkinsi* was significantly greater than the mean species diversity of sites lacking *F. jenkinsi* for both seine and trap catch (Zero-inflated Poisson Seine *p* = <0.001, trap *p* = 0.01228) (Figure 17). Additionally for seine samples, the Evenness (p = >0.001) and Richness (p = >0.001) of sites where *F. jenkinsi* was caught was higher than sites where it was not, while there was no difference in catch per unit effort (*p* = 0.821).

Fish community assemblage did not significantly differ between index and non-index sites (one-way ANOSIM, R = -0.041, *p* = 0.925). Therefore all data were pooled and analyzed by gear type for further community analyses. The fish community assemblage sampled with seines differed significantly between sites where *F. jenkinsi* was present versus locations where it was not captured (one-way ANOSIM, R = 0.073, *p* = 0.006). Species most often found at sites where *F. jenkinsi* where present versus locations where it was not captured are displayed in Table 5 and 6. The Mosquitofish *Poecilia latipinna* was observed at nearly 82 percent of the sites that *F. jenkinsi* was captured in seines. *Fundulus grandis*, *Gambusia affinis*, and *Cyprinodon variegatus* were all observed at higher frequencies in seine collections when *F. jenkinsi* were present compared to when it was not collected. Alternatively, *Menidia beryllina*, *Anchoa mitchilli*, and *Brevoortia patronus* exhibited a higher affinity to sites were F. jenkinsi was not captured. Species associations were lower for the trap samples, but *G. affinis* and *Fundulus pulvereus* were more highly associated with traps were *F. jenkinsi* was collected. The fish community assemblage did not show a significant difference between traps from sites where *F. jenkinsi* were present versus assemblages were F. jenkinsi were not caught (one-way ANOSIM, R = -0.006, p = 0.51). The non-metric MDS plot of seine fish assemblages show a grouping along the MDS1 ordination for some of the same species that appear to have an association with the presence of *F. jenkinsi* (Figure 18).

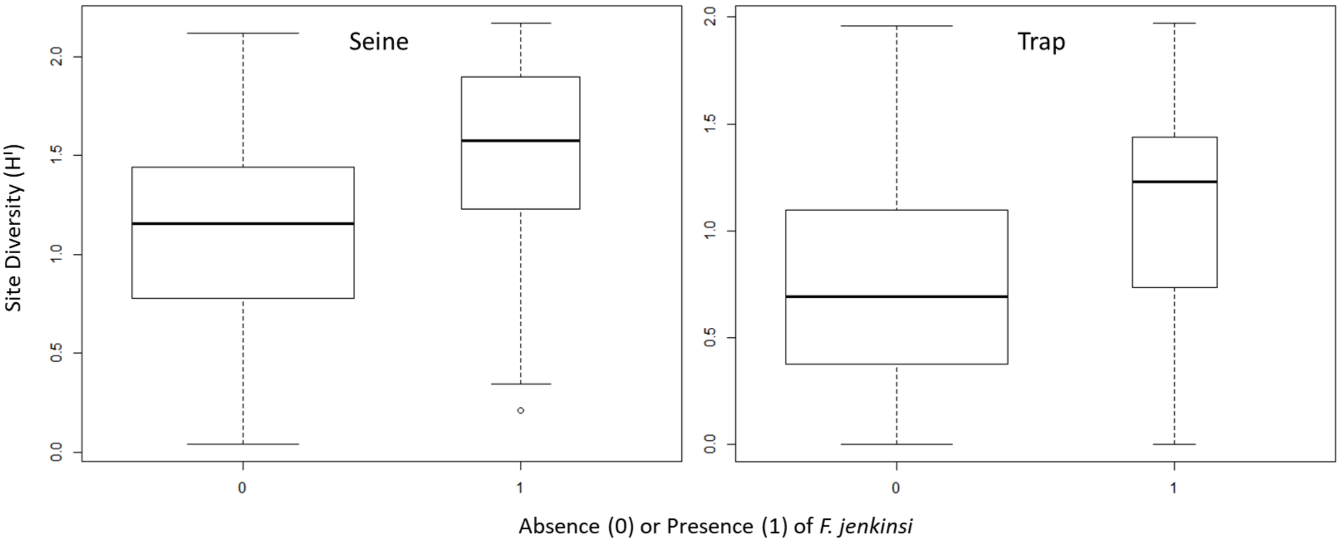


Figure 17. Boxplot of fish species diversity (H’) between sites where *F. jenkinsi* were collected (1) and not collected (0) in by gear type.

Table 5. The percent occurrence of the most prevalent species (percent occurrence > 15%) of fish captured in seine collections during 2014 and 2017-18 containing and lacking *F. jenkinsi*. Pearson correlations associated with the nMDS plot (Figure 18) contributing to the side to side ordination (MDS1).

Species	Percent Occurrence		Pearson Correlation MDS1
	F. jenkinsi Present	*F. jenkinsi* Absent	
Poecilia latipinna	81.9	40.9	-26.4
Fundulus grandis	75.0	46.0	-24.2
Gambusia affinis	72.2	39.4	-18.7
Cyprinodon variegatus	70.8	44.9	-39.5
Menidia beryllina	61.1	85.8	-7.0
Micropogonias undulatus	61.1	34.7	10.8
Fundulus pulvereus	52.8	20.8	-28.4
Anchoa mitchilli	51.4	59.9	25.6
Brevoortia patronus	48.6	52.2	11.0
Adinia xenica	47.2	27.7	-23.8
Mugil cephalus	45.8	53.6	< 0.01
Lucania parva	44.4	24.5	-13.9
Leiostomus xanthurus	36.1	44.2	-1.1
Gobiosoma bosc	33.3	12.4	-2.1
Lagodon rhomboides	23.6	31.4	6.5
Sciaenops ocellatus	23.6	7.7	-3.5

Table 6. The percent occurrence of the most prevalent species (percent occurrence > 15%) of fish found in trap collections during 2014 and 2017-18 containing and lacking *F. jenkinsi*.

Species	Percent Occurrence	
	F. jenkinsi Present	*F. jenkinsi* Absent
Gambusia affinis	54.2	29.1
Fundulus pulvereus	41.7	23.1
Poecilia latipinna	41.7	46.2
Cyprinodon variegatus	29.2	32.4
Adinia xenica	16.7	19.8
Fundulus grandis	16.7	26.9
Micropogonias undulatus	16.7	6.0

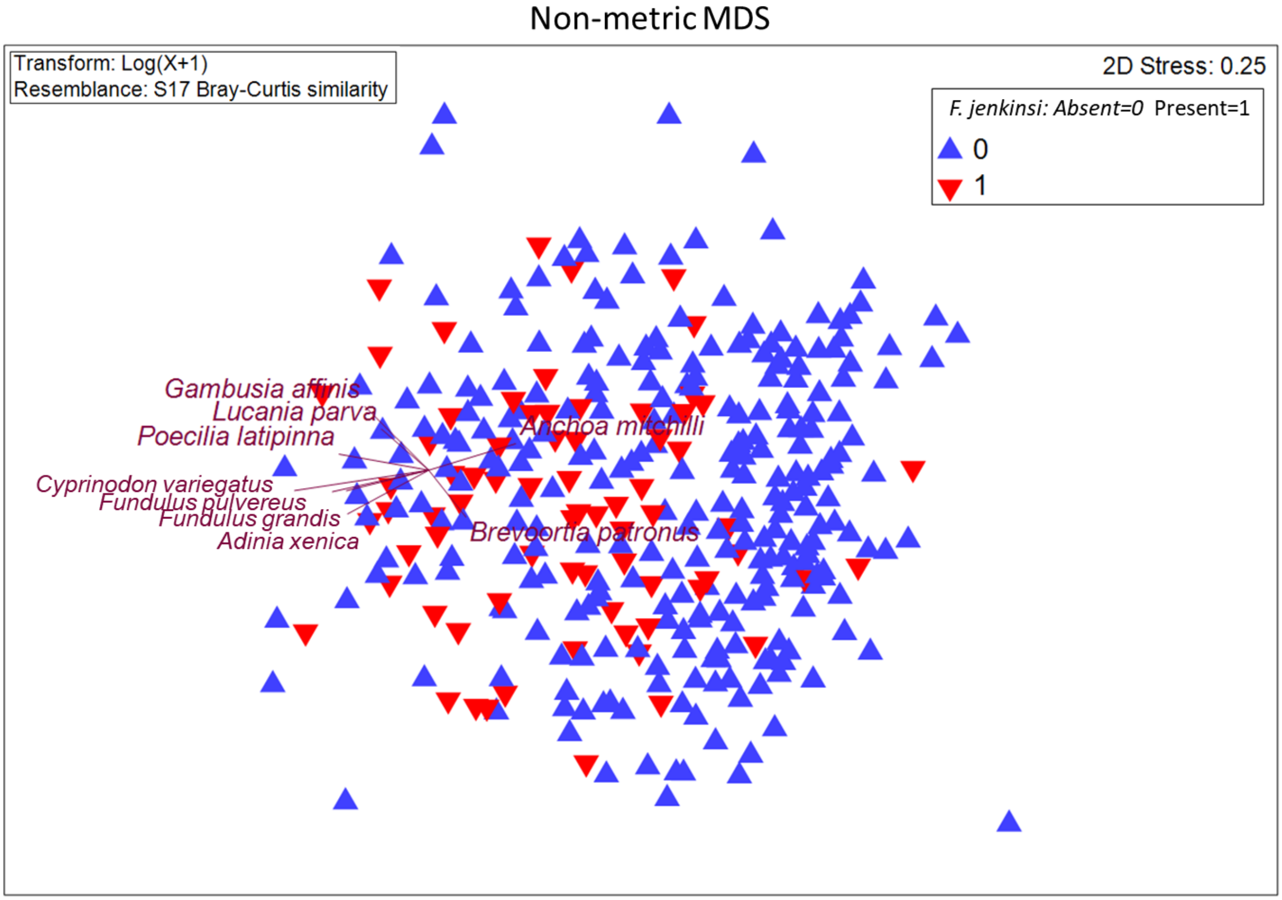


Figure 18. nMDS plot of fish assemblage data illustrating the similarity of fish assemblages at sites where *F. jenkinsi* were collected (blue triangles) versus not collected (red inverted triangles) in seines

Gear Selectivity

Fundulus jenkinsi were found in fish assemblages sampled by both seines and Breder traps. Out of the sites where *F. jenkinsi* was caught, 19.5 percent of the sampling events caught *F. jenkinsi* using the trap gear type only, while 53.7 percent caught *F. jenkinsi* using the seine gear type only. At the remaining 26.8% of the sites where *F. jenkinsi* were caught, they were collected in both gear types.

The catch rates of *F. jenkinsi* were compared between gear types to evaluate the relative ability of each gear to capture the target species (Figure 19). A total of 208 collections where both seines and traps were used were compared to evaluate the ability of each method to detect *F. jenkinsi*. A total of 52 collections contained *F. jenkinsi* captured with one or both types of sampling gear. The trap and seine CPUE exhibited a significant but weak correlation ($r = 0.206$, $p = 0.003$). Similarly the occurrence (presence/absence) *F. jenkinsi* in trap and seine collections exhibited a significant but weak rank correlation ($r_s = 0.356$, p = <0.001). There was also a statistically significant association between the occurrence of *F. jenkinsi* in traps and seines ($x^2 = 26.358$, $p < 0.001$; Table 7). Seines were more effective at capturing *F. jenkinsi* (40/208 or 19.2% of the collections). In contrast traps only captured *F. jenkinsi* at

27/208 or 13% of the collections. When combined together both methods detected *F. jenkinsi* at 52/208 or 25% of the collections. Although the catch rates of both methods were correlated both methods together increased the probability of detecting *F. jenkinsi*. Traps captured F. jenkinsi in 12 collections where seines failed to detect them (Table 7). In contrast, *F. jenkinsi* were detected in 25 collections where traps failed to detect them.

The average total length (mm) of *F. jenkinsi* caught via seine (n = 420) was 32.53 mm (σ = 7.417) with a range of 19-58 mm and the average total length of *F. jenkinsi* caught via trap was 36.23 mm (σ = 8.98) with a range of 16-50 mm (Figure 20). The total length of *F. jenkinsi* was significantly larger for individuals caught in the trap gear compared to the seine (ANOVA, $p < 0.0016$). This difference was also mirrored using total weight (mg). The proportion of male to female *F. jenkinsi* did not differ by gear type. *F. jenkinsi* head width frequency distribution shows that the mesh size of the TPWD 60' bag seine typically used to monitor nekton populations in Texas Bays is too coarse to effectively sample this small fish (Figure 21). This may partially explain the lack of *F. jenkinsi* catch in TPWD bag seine collections.

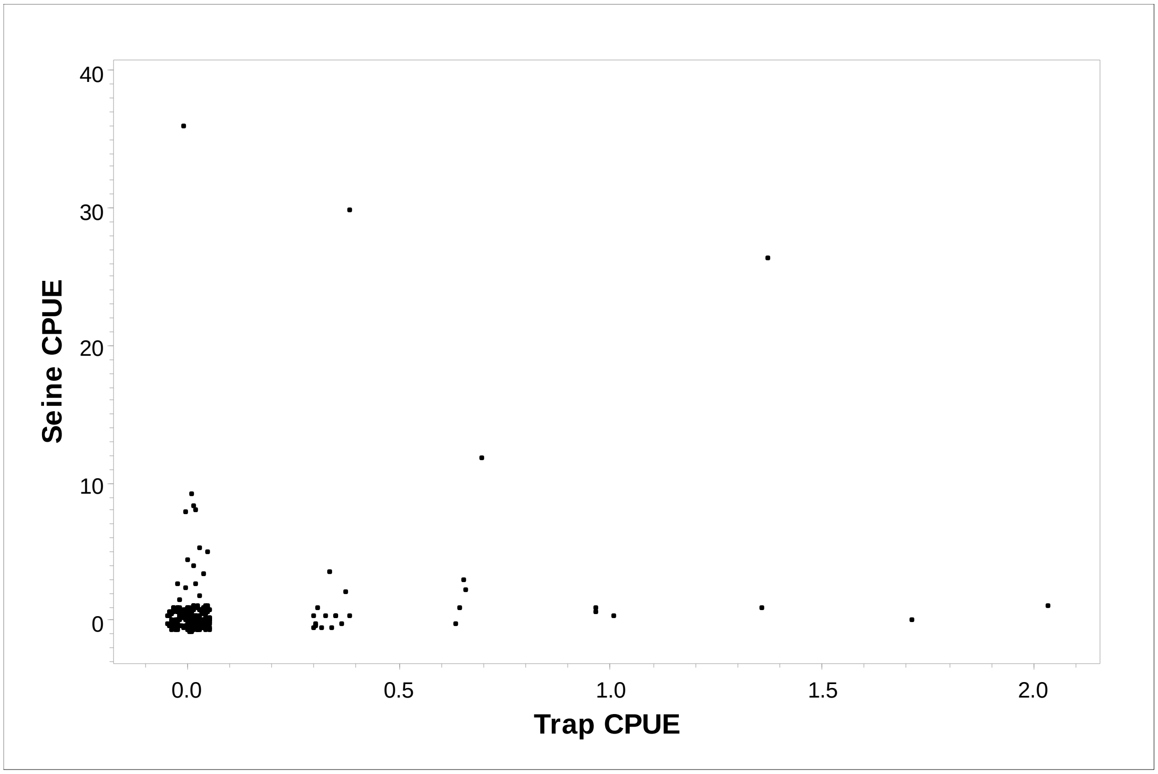


Figure 19. Comparison of *F. jenkinsi* seine and trap CPUE. N = 208.

Table 7. Cross tabulation and chi-square test of the occurrence of *F. jenkinsi* in trap and seine collections ($x^2 = 26.358$, $p < 0.001$).

Seines Absence/Presence → Traps Absence/Presence ↓	F. jenkinsi Absent	F. jenkinsi Present	Total
F. jenkinsi Absent – Count	156	25	181
F. jenkinsi Absent – Expected Count	146.19	34.81	
F. jenkinsi Present – Count	12	15	27
Present – Expected Count	21.81	5.19	
Total	**168**	**40**	**208**

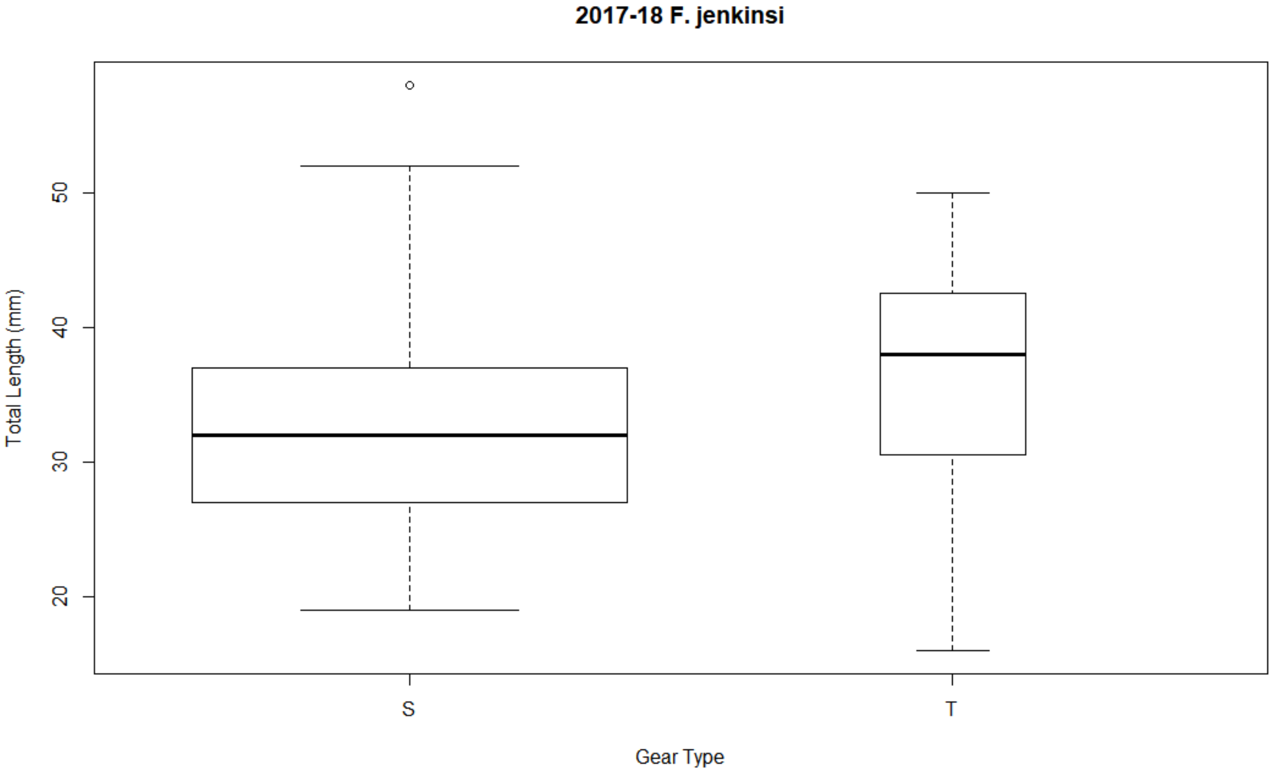


Figure 20. Boxplot of the total length of *F. jenkinsi* caught by gear type in 2017-18 collection.

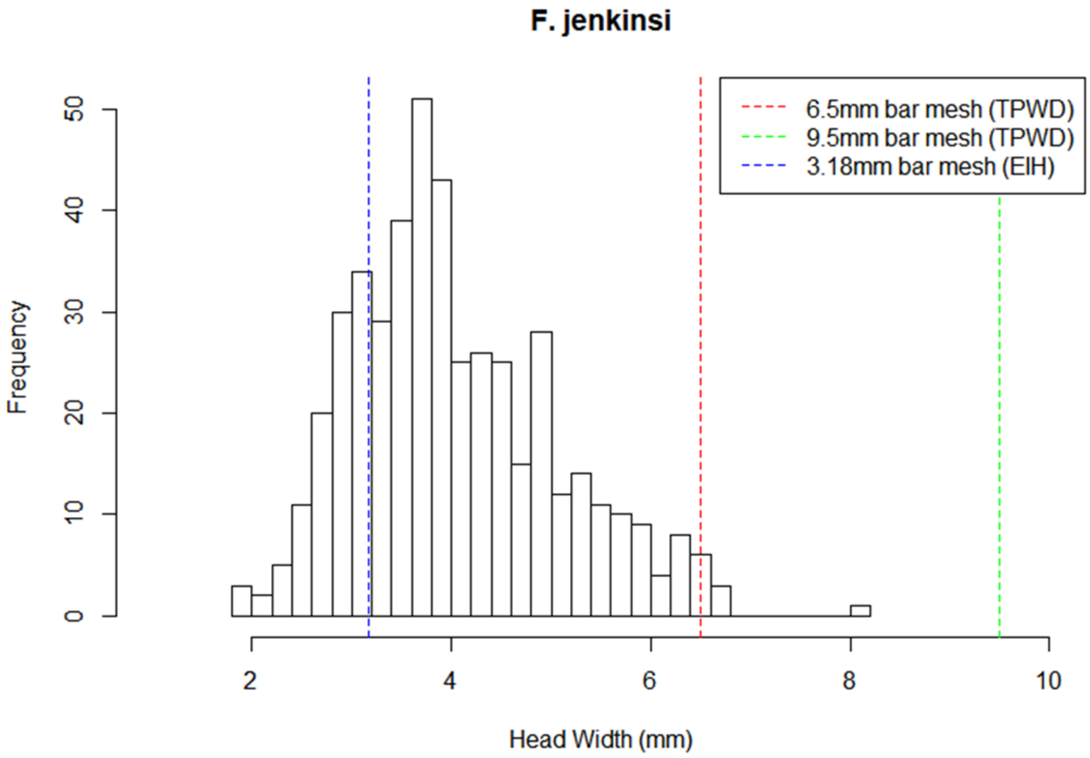


Figure 21. Frequency distribution of *F. jenkinsi* head widths (mm). Red dashed line = 6.5mm bar mesh of TPWD bag seine in bag, green dashed line = 9.5mm bar mesh of TPWD Bag Seine on wings, and blue dashed line = 3.18 bar mesh of EIH straight seine (Martinez-Andrade 2015).

Habitat Characteristics

Water Chemistry and Clarity

During the extended study period (2014-2018) surface water temperature averaged 23.6 °C and varied between 5.8 and 38.4 °C (Table 8 and Figure 22). Median water temperatures varied significantly between estuaries (Kruskal-Wallis H= 22.80; *p* = 0.002). Median water temperature in San Antonio Bay was significantly higher compared to Sabine Lake and Aransas Bay, whereas the median water temperature at Matagorda Bay was significantly higher than Sabine Lake (Figure 23 and Table 9). The generally lower and less variable water temperatures observed in the Aransas Bay estuary during this study reflects the bias of collecting samples only during March 2018. As expected the median surface water temperature varied significantly between seasons (Kruskal-Wallis H = 182.43; *p* <0.001; Figure 23 and Table 10).

Table 8. Summary statistics for selected water quality variables measured during 2014-2018. N* = number of missing observations; Q1 – first quartile; Q3 – third quartile.

Variable	N	N*	Mean	Minimum	Q1	Median	Q3	Maximum
Temp (°C)	365	0	23.6	5.8	18.3	24.8	29.4	38.4
Salinity (ppt)	365	0	11.5	0.1	3.6	10.8	17.7	51.3
DO (mg/L)	362	3	8.3	0.9	6.4	8.1	10.2	21.2
Secchi-disk Transparency (m)	365	0	0.245	0.018	0.142	0.212	0.304	0.938
Depth (m)	192	173	0.200	0.020	0.102	0.159	0.283	0.784

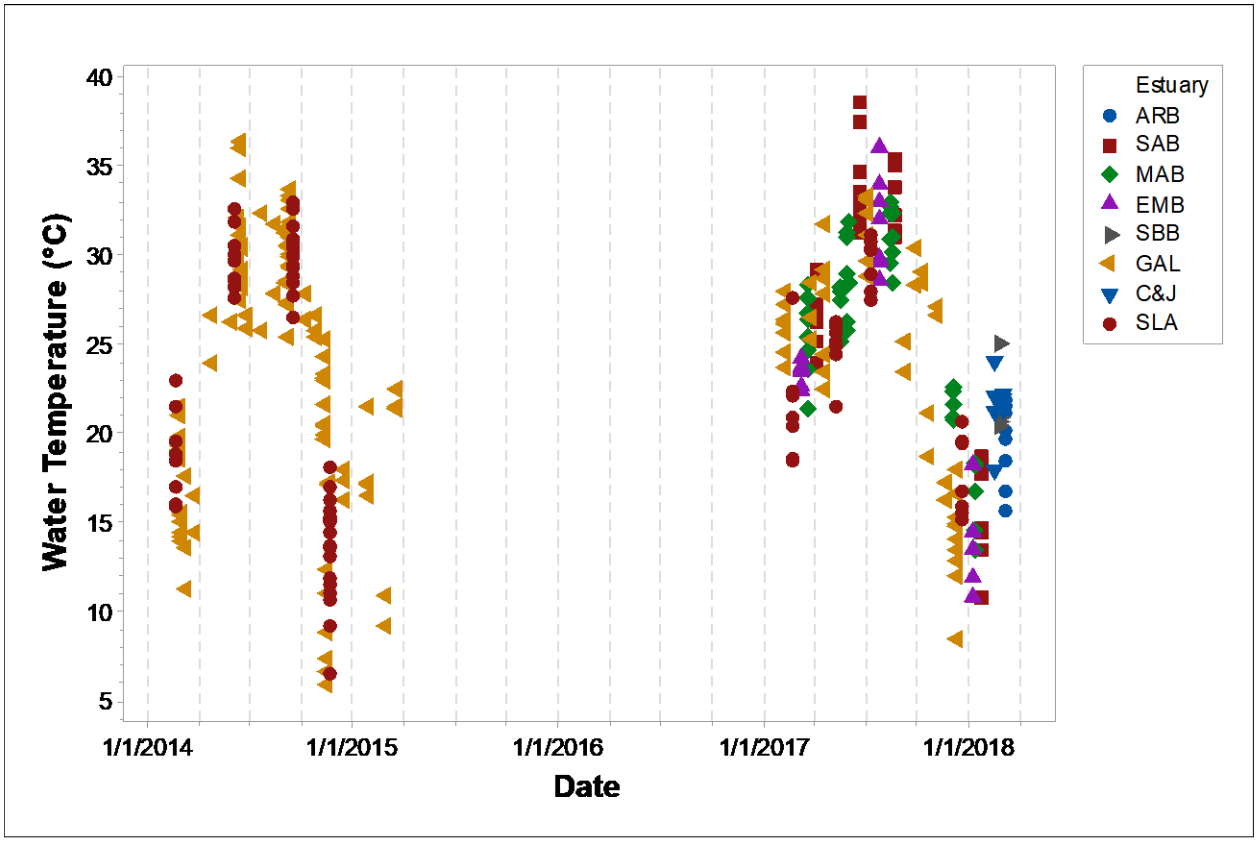


Figure 22. Water temperature measurements collected within each estuary during the study period 2014-2018. ARB –Aransas Bay; SAB – San Antonio Bay, MAB – Matagorda Bay; EMB – East Matagorda Bay; SBB – San Bernard and Brazos Rivers; GAL – Galveston Bay; C&J – Chambers and Jefferson County coastal waterways; SLA – Sabine Lake.

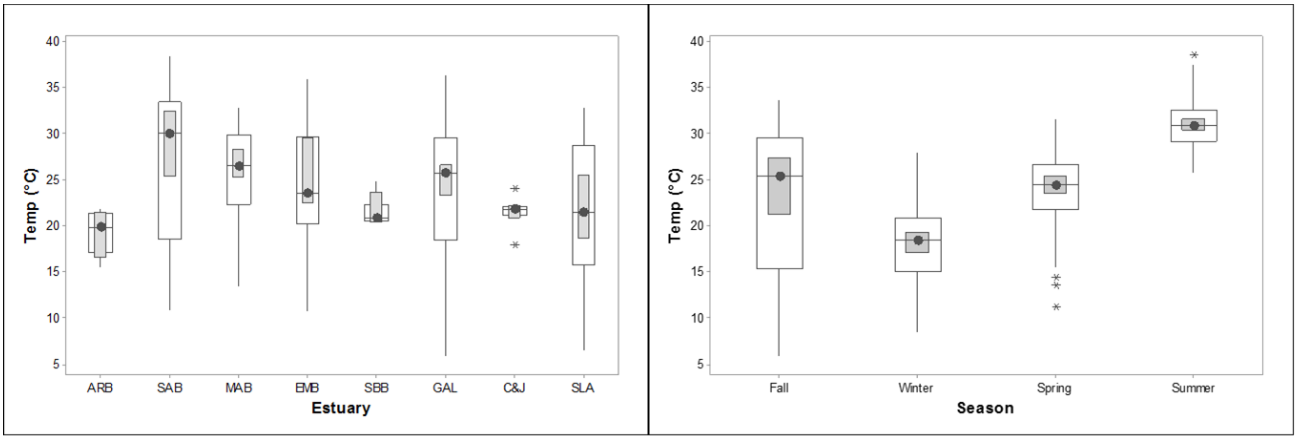


Figure 23. Boxplots of water temperatures recorded at collection sites within each estuary and across all seasons during the study period in 2014-2018. ARB –Aransas Bay; SAB – San Antonio Bay, MAB – Matagorda Bay; EMB – East Matagorda Bay; SBB – San Bernard and Brazos Rivers; GAL – Galveston Bay; C&J – Chambers and Jefferson County coastal waterways; SLA – Sabine Lake. The gray inner box represents the 95% confidence interval of the median.

Table 9. Estuaries exhibiting significant differences in median water temperature using Dunn's multiple comparison test.

Estuary Comparisons	Z vs. Critical value	P-value
Sabine Lake vs. San Antonio Bay	3.49462 >= 2.69	0.0005
Aransas Bay vs. San Antonio Bay	3.04265 >= 2.69	0.0023
Sabine Lake vs. Matagorda Bay	2.80319 >= 2.69	0.0051

Table 10. Seasons exhibiting significant differences in median water temperature using Dunn's multiple comparison test.

Seasonal Comparisons	Z vs. Critical value	P-value
Winter vs. Summer	13.4447 >= 2.128	<0.001
Fall vs. Summer	8.2333 >= 2.128	<0.001
Summer vs. Spring	7.4674 >= 2.128	<0.001
Winter vs. Spring	5.2853 >= 2.128	<0.001
Winter vs. Fall	5.2443 >= 2.128	<0.001

Salinity averaged 11.5 ‰ and ranged between 0.1 to 51.3 ‰ during 2014-18 (Table 8 and Figure 24). The majority of salinity measurements were below 17.7 ‰ (75th percentile). The maximum salinity reading was taken during the summer within a tidal creek in the San Antonio - Nueces Bay watershed in an area historically contaminated by oilfield brine, limited streamflow and tidal connectivity, and subject to high levels of evaporation (Figure 25). The median salinity varied significantly between estuaries (Kruskal-Wallis H = 45.25; p < 0.001; Table 11). Salinities were generally lower in eastern estuaries state, although there was considerable variation within each estuary. The generally lower and less variable salinity found in the Aransas Bay system reflects the bias of collecting samples only during March 2018 when precipitation is typically higher in Texas. The lowest (< 5 ‰) salinities were generally recorded at tidal creek wetlands within Chambers and Jefferson counties (Figure 25). The median salinity varied significantly between seasons (Kruskal-Wallis H= 35.67; p < 0.001; Table 12). Winter months exhibited the lowest median salinities, whereas summer and fall exhibited the highest median salinities (Figure 24).

Dissolved oxygen averaged 8.3 mg/L and ranged between 0.9 to 21.2 mg/L during 2014-2018 (Table 8 and Figure 26). The median dissolved oxygen level varied significantly between estuaries (Kruskal-Wallis H = 25.85 p = 0.001; Figure 27). The median dissolved oxygen level in San Antonio Bay was higher compared to most estuaries (Table 13). Dissolved oxygen seldom fell below 2-3 mg/L (hypoxia) except at some Galveston Bay sites. The median dissolved oxygen level varied significantly between seasons (Kruskal-Wallis H = 35.37 p = < 0.001; Table 14 and Figure 27). The fall season exhibited significantly lower median dissolved oxygen levels in comparison to the summer and winter, most likely due to the higher frequency of hypoxic (< 2.0 mg/L) events.

Water transparency (i.e. Secchi-disk transparency) as measured with the Secchi tube exhibited an average value of 0.245 m and ranged between 0.018 and 0.938 m (Table 8 and Figure 28). Median Secchi-disk transparency was generally higher east Texas estuaries including Galveston Bay, Sabine Lake, and Chambers and Liberty counties in comparison to San Antonio and Matagorda Bay (Kruskal-Wallis H = 42.29; p < 0.001) (Figure 29 and Table 15). Median water transparency was significantly lower (Kruskal-Wallis H = 58.2; p <0.001) during the summer months in comparison to other seasons (Figure 29 and Table 16). Median Secchi-disk transparency was also higher during the fall in comparison to the spring.

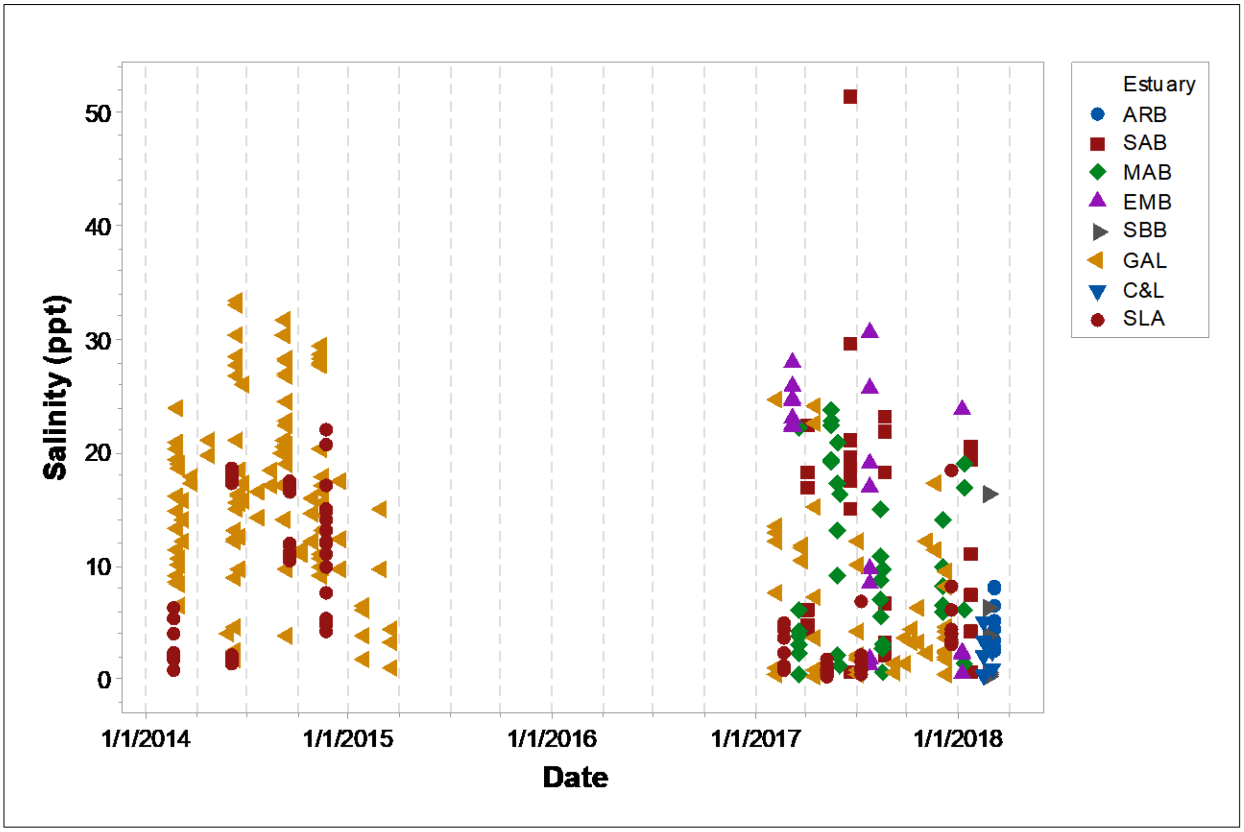


Figure 24. Salinity measurements collected within each estuary during the study period 2014-2018. ARB – Aransas Bay; SAB – San Antonio Bay, MAB – Matagorda Bay; EMB – East Matagorda Bay; SBB – San Bernard and Brazos Rivers; GAL – Galveston Bay; C&J – Chambers and Jefferson County coastal waterways; SLA – Sabine Lake.

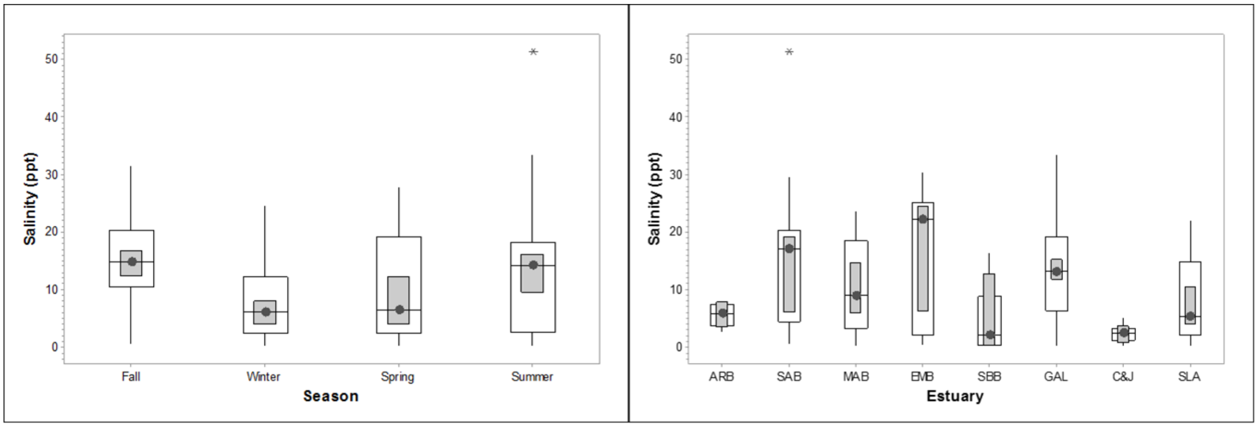


Figure 25. Boxplots of salinity recorded at collection sites within each estuary and across all seasons during the study period in 2014-2018. ARB –Aransas Bay; SAB – San Antonio Bay, MAB – Matagorda Bay; EMB – East Matagorda Bay; SBB – San Bernard and Brazos Rivers; GAL – Galveston Bay; C&J – Chambers and Jefferson County coastal waterways; SLA – Sabine Lake. The gray inner box represents the 95% confidence interval of the median.

Table 11. Estuaries exhibiting significant differences in salinity using Dunn's multiple comparison test.

Estuary Comparisons	Z vs. Critical value	P-value
Galveston Bay vs. Sabine Lake	4.50216 >= 2.69	<0.0001
Chambers-Jefferson vs. East Matagorda Bay	3.74671 >= 2.69	0.0002
Galveston Bay vs. Chambers-Jefferson	3.70562 >= 2.69	0.0002
Chambers-Jefferson vs. San Antonio Bay	3.55117 >= 2.69	0.0004
Sabine Lake vs. East Matagorda Bay	3.34599 >= 2.69	0.0008
Sabine Lake vs. San Antonio Bay	3.16329 >= 2.69	0.0016
East Matagorda Bay vs. San Bernard/Brazos	2.95926 >= 2.69	0.0031
Galveston Bay vs. San Bernard/Brazos	2.77812 >= 2.69	0.0055
San Antonio Bay vs. San Bernard/Brazos	2.74200 >= 2.69	0.0061

Table 12. Seasons exhibiting significant differences (adjusted for ties) in salinity using Dunn's multiple comparison test.

Seasonal Comparisons	Z vs. Critical value	P-value
Winter vs. Fall	5.74475 >= 2.128	<0.0001
Fall vs. Spring	3.80952 >= 2.128	0.0001
Winter vs. Summer	3.32715 >= 2.128	0.0009
Fall vs. Summer	2.22349 >= 2.128	0.0262

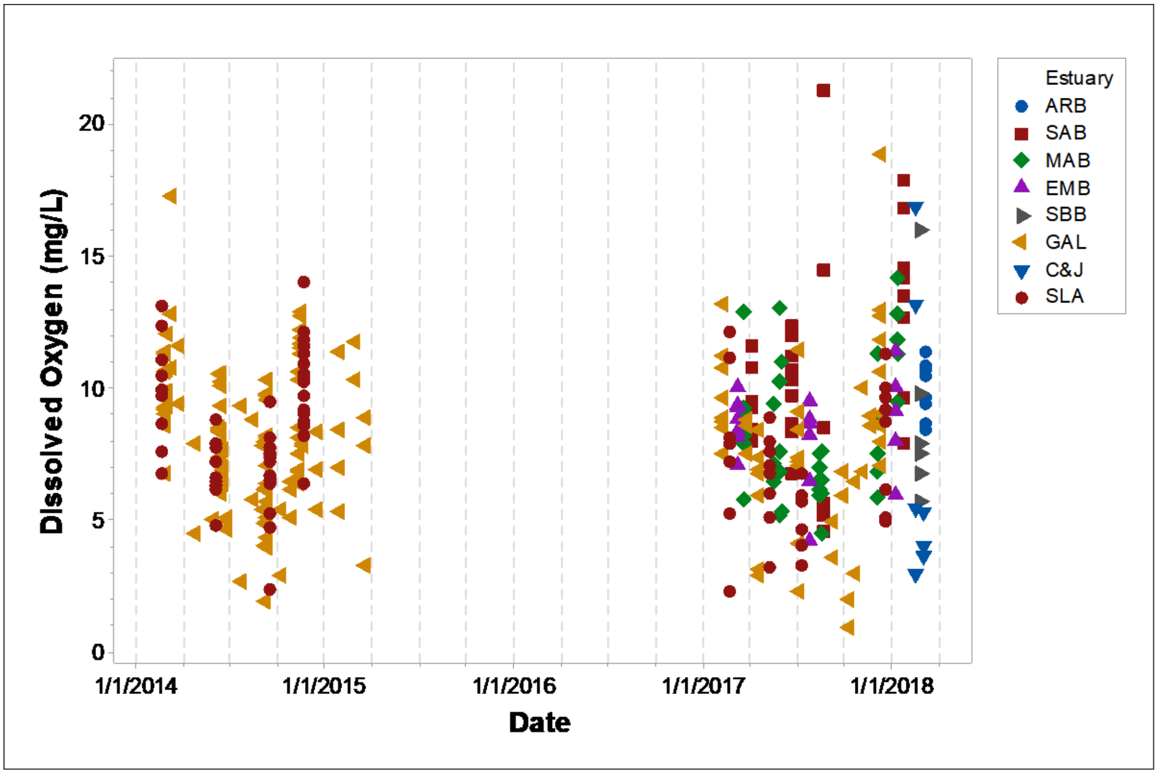


Figure 26. Dissolved oxygen measurements collected within each estuary during the study period 2014-2018. ARB –Aransas Bay; SAB – San Antonio Bay, MAB – Matagorda Bay; EMB – East Matagorda Bay; SBB – San Bernard and Brazos Rivers; GAL – Galveston Bay; C&J – Chambers and Jefferson County coastal waterways; SLA – Sabine Lake.

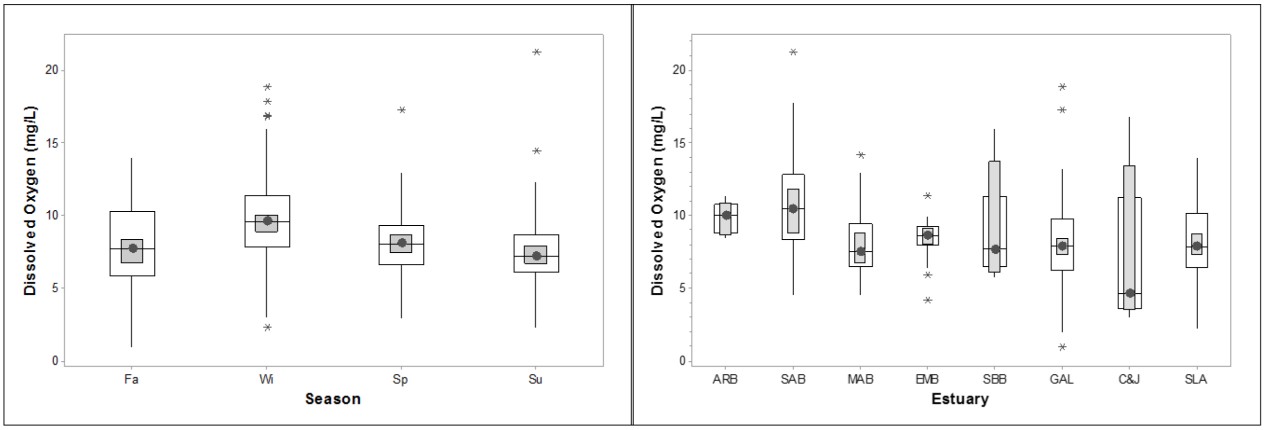


Figure 27. Boxplots of dissolved oxygen recorded during all collections at all sites within each estuary and across all seasons during the study period in 2014-2018. ARB –Aransas Bay; SAB – San Antonio Bay, MAB – Matagorda Bay; EMB – East Matagorda Bay; SBB – San Bernard and Brazos Rivers; GAL – Galveston Bay; C&J – Chambers and Jefferson County coastal waterways; SLA – Sabine Lake. The gray inner box represents the 95% confidence interval of the median.

Table 13. Estuaries exhibiting significant differences (adjusted for ties) in dissolved oxygen using Dunn's multiple comparison test.

Estuary Comparisons	Z vs. Critical value	P-value
Sabine Lake vs. San Antonio Bay	5.45092 >= 2.69	<0.0001
Galveston Bay vs. San Antonio Bay	5.07011 >= 2.69	<0.0001
Chambers-Jefferson vs. San Antonio Bay	4.23730 >= 2.69	<0.0001
Matagorda Bay vs. San Antonio Bay	3.73853 >= 2.69	0.0002

Table 14. Seasons exhibiting significant differences (adjusted for ties) in dissolved oxygen using Dunn's multiple comparison test.

Seasonal Comparisons	Z vs. Critical value	P-value
Winter vs. Fall	2.51812 >= 2.128	0.0118
Fall vs. Summer	2.15324 >= 2.128	0.0313

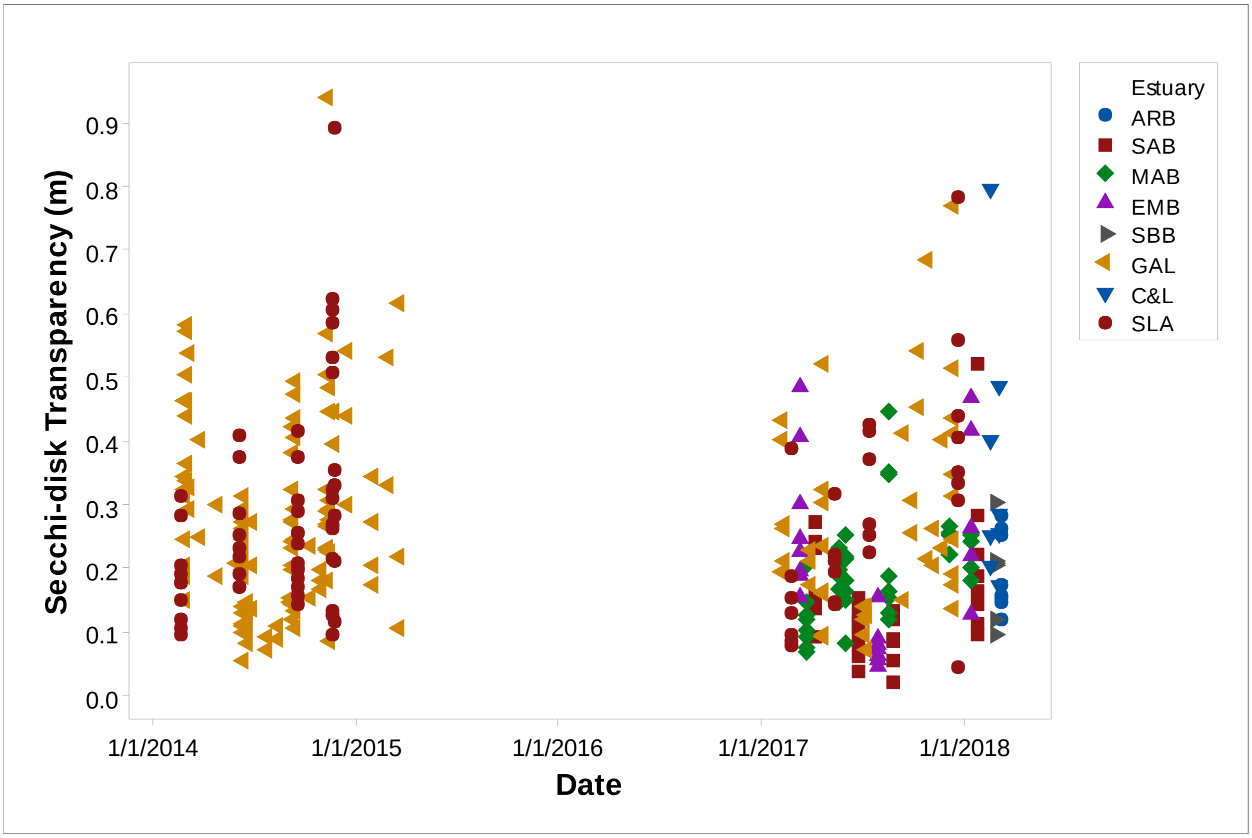


Figure 28. Secchi-disk transparency measurements collected within each estuary during the study period 2014-2018. ARB –Aransas Bay; SAB – San Antonio Bay, MAB – Matagorda Bay; EMB – East Matagorda Bay; SBB – San Bernard and Brazos Rivers; GAL – Galveston Bay; C&J – Chambers and Jefferson County coastal waterways; SLA – Sabine Lake.

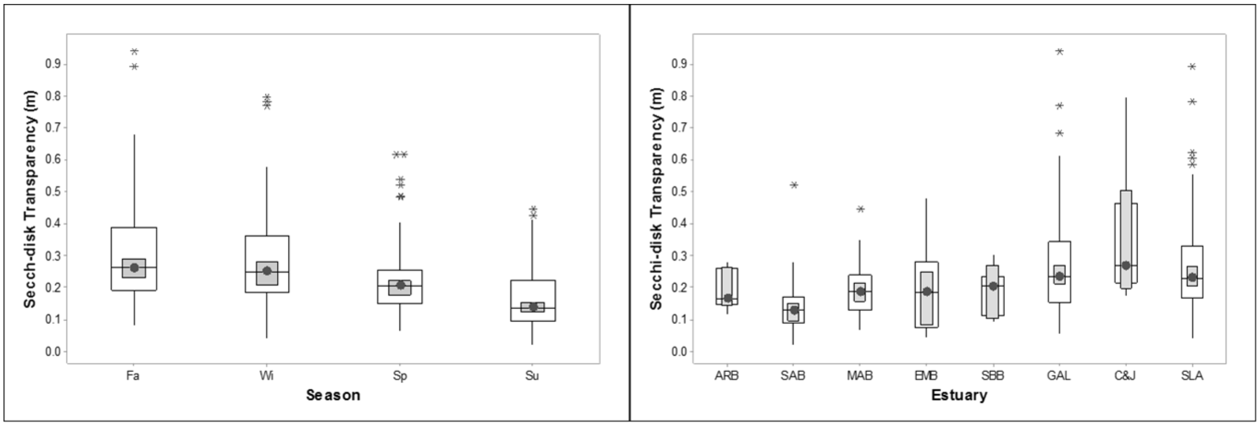


Figure 29. Boxplots of Secchi-disk transparency (m) recorded at collection sites within each estuary and across all seasons during the study period in 2014-2018. ARB –Aransas Bay; SAB – San Antonio Bay, MAB – Matagorda Bay; EMB – East Matagorda Bay; SBB – San Bernard and Brazos Rivers; GAL – Galveston Bay; C&J – Chambers and Jefferson County coastal waterways; SLA – Sabine Lake. The gray inner box represents the 95% confidence interval of the median.

Table 15. Estuaries exhibiting significant differences (adjusted for ties) in Secchi-disk transparency using Dunn's multiple comparison test.

Estuary Comparisons	Z vs. Critical value	P-value
Galveston Bay vs. San Antonio Bay	5.22523 >= 2.69	0.0000
Sabine Lake vs. San Antonio Bay	4.78382 >= 2.69	0.0000
Chambers-Jefferson vs. San Antonio Bay	3.77138 >= 2.69	0.0002
Galveston Bay vs. Matagorda Bay	3.07303 >= 2.69	0.0021
Sabine Lake vs. Matagorda Bay	2.71576 >= 2.69	0.0066

Table 16. Seasons exhibiting significant differences (adjusted for ties) in Secchi-disk transparency using Dunn's multiple comparison test.

Seasonal Comparisons	Z vs. Critical value	P-value
Fall vs. Summer	6.87983 >= 2.128	0.0000
Winter vs. Summer	6.38233 >= 2.128	0.0000
Summer vs. Spring	3.40528 >= 2.128	0.0007
Fall vs. Spring	3.18825 >= 2.128	0.0014

Water depth was not recorded during 2014-15 (Table 8 and Figure 30). Recorded water levels were obtained from nearby NOAA tide gages in an attempt to use those values as a surrogate for local measurements. Due to the distance to the closest gage and the complex bay bathymetry, these two measures of depth exhibited a weak ($r = 0.270$) but significant ($p < 0.001$) positive correlation. Although the Kruskal-Wallis detected significant ($H = 13.82$; $p = 0.054$) differences between estuaries, Dunn's test failed to detect any significant pairwise difference (Figure 29 and Table 15). Significant differences in sampling depth were detected between seasons (Kruskal-Wallis H=15.40; *p* = 0.002). Many of the observed water levels in the estuaries were generally lower (< 0.2 m) in the winter or spring which likely reflects the effects of the passage of major northerly cold fronts and/or spring tides (Figure 31 Table 17).

Analysis of water level measurements collected from NOAA tide gages concurred with patterns of actual water depths at each site (Figure 32). Water levels ranged between -0.961 and 2.621 ft. Statistically significant differences in water levels were detected between estuaries (Kruskal-Wallis H=17.37; *p* = 0.015; Table 18). Water levels in Galveston Bay and Sabine Lake were significantly lower than the San Bernard/Brazos estuary (Table 18). Statistically significant differences in water levels were detected between seasons (Kruskal-Wallis H=59.11; *p* = <0.001) with winter exhibiting the lowest median water level compared to all seasons, and fall having significantly lower median levels compared to spring and summer (Figure 33 and Table 19).

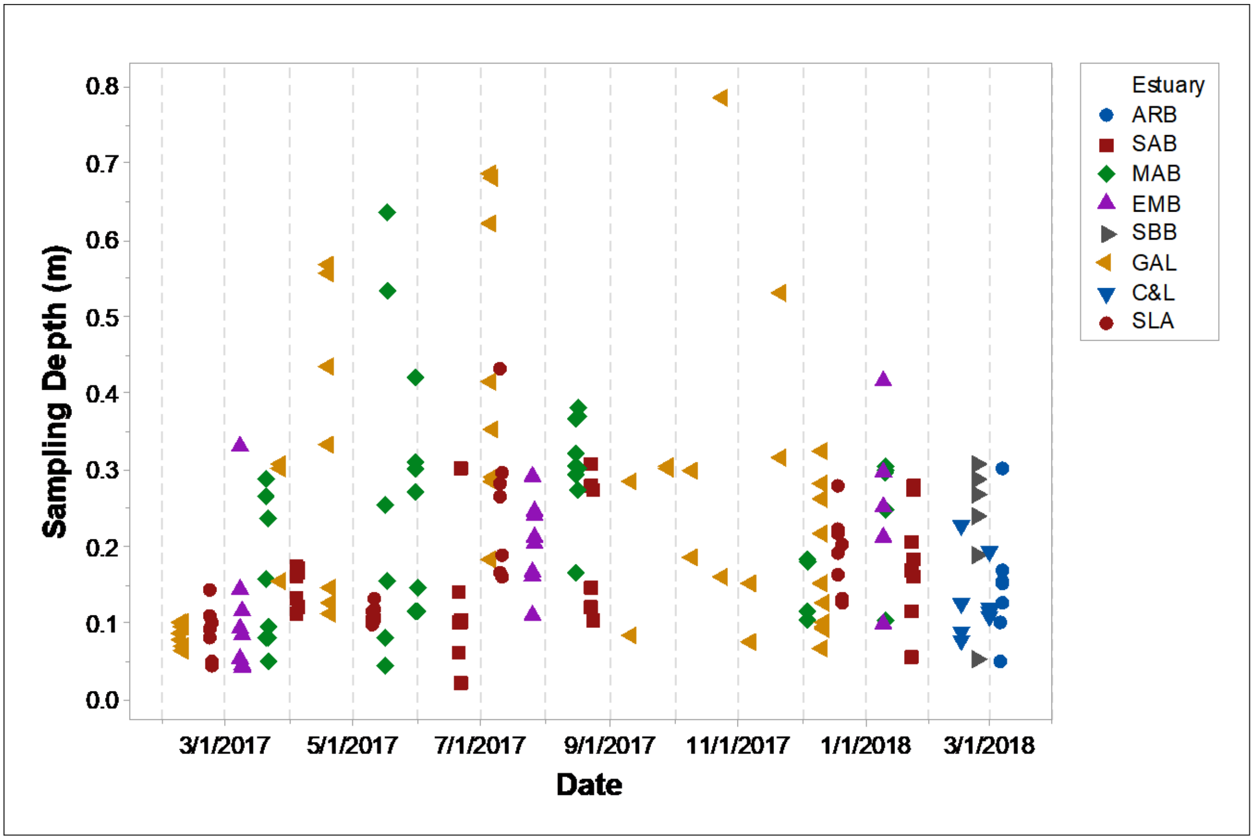


Figure 30. Sampling depth of individual collections within each estuary during the study period 2014-2018. ARB –Aransas Bay; SAB – San Antonio Bay, MAB – Matagorda Bay; EMB – East Matagorda Bay; SBB – San Bernard and Brazos Rivers; GAL – Galveston Bay; C&J – Chambers and Jefferson County coastal waterways; SLA – Sabine Lake.

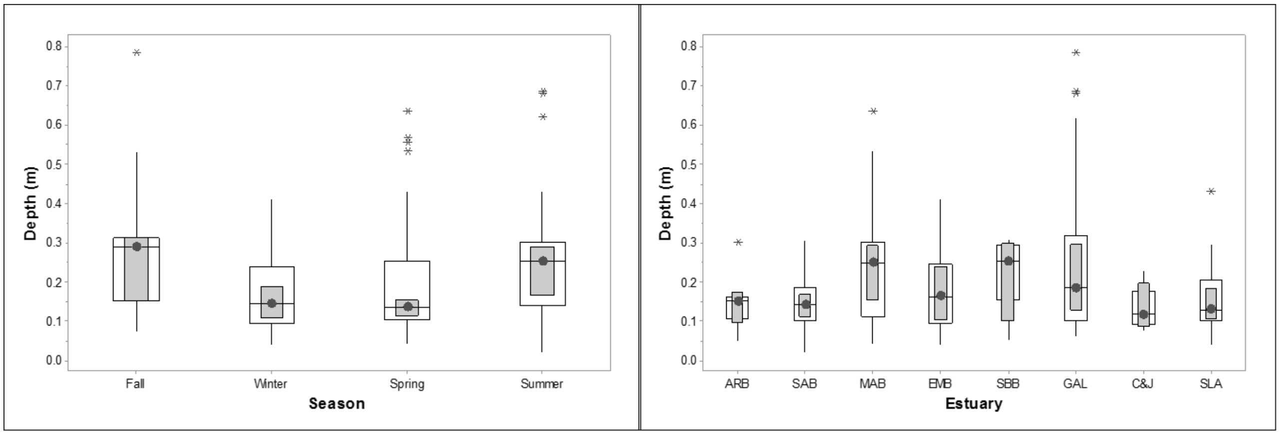


Figure 31. Boxplots of water depth (m) recorded at collection sites within each estuary and across all seasons during the study period in 2014-2018. ARB –Aransas Bay; SAB – San Antonio Bay, MAB – Matagorda Bay; EMB – East Matagorda Bay; SBB – San Bernard and Brazos Rivers; GAL – Galveston Bay; C&J – Chambers and Jefferson County coastal waterways; SLA – Sabine Lake. The gray inner box represents the 95% confidence interval of the median.

Table 17. Seasons exhibiting significant differences in sampling depth using Dunn's multiple comparison test.

Seasonal Comparisons	Z vs. Critical value	P-value
Winter vs. Summer	3.21205 > = 2.128	0.0013
Summer vs. Spring	2.85342 > = 2.128	0.0043
Winter vs. Fall	2.39916 > = 2.128	0.0164
Fall vs. Spring	2.18237 > = 2.128	0.0291

Table 18. Estuaries exhibiting significant differences in tide gage water level using Dunn's multiple comparison test.

Estuary Comparisons	Z vs. Critical value	P-value
Galveston Bay vs. San Bernard/Brazos	3.30282 >= 2.69	0.0010
Sabine Lake vs. San Bernard/Brazos	3.22250 >= 2.69	0.0013

Table 19. Seasons exhibiting significant differences in tide gage water level using Dunn's multiple comparison test.

Seasonal Comparisons	Z vs. Critical value	P-value
Winter vs. Summer	6.49146 >= 2.128	<0.0001
Winter vs. Spring	6.45496 >= 2.128	<0.0001
Fall vs. Spring	3.30867 >= 2.128	0.0009
Winter vs. Fall	3.29134 >= 2.128	0.0010
Fall vs. Summer	3.24774 >= 2.128	0.0012

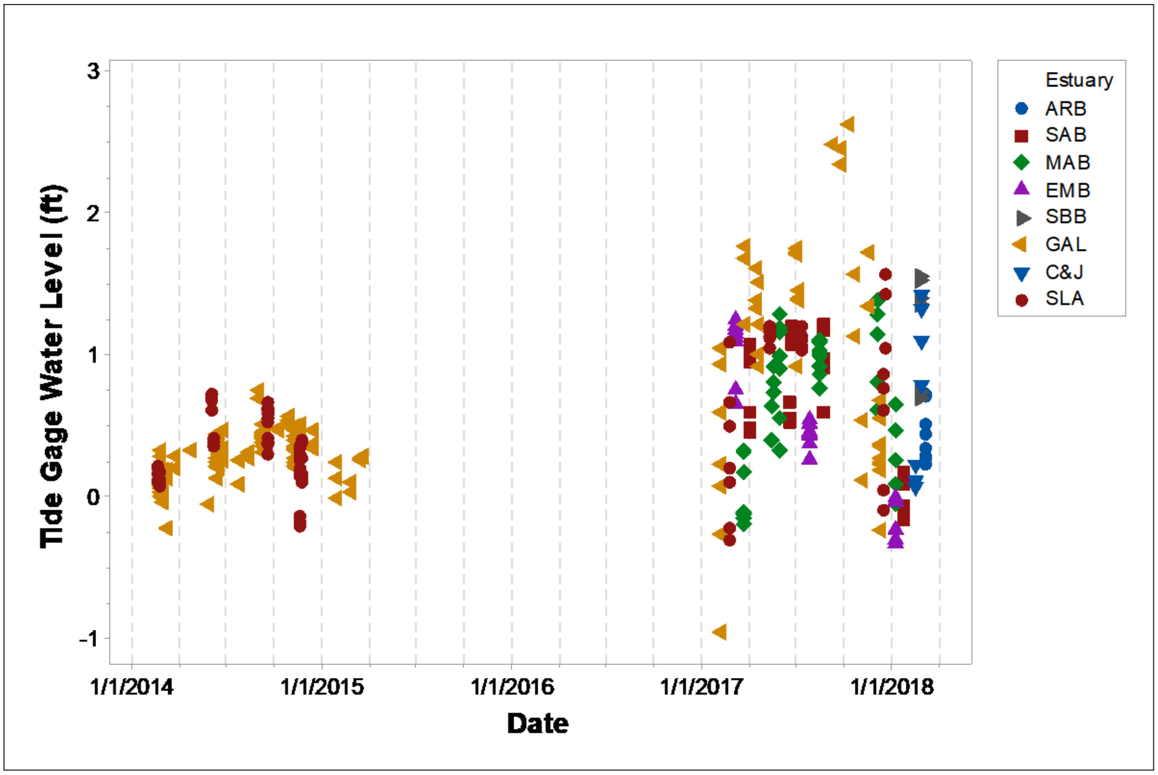


Figure 32. Water level measured at NOAA tide gages (referenced to mean lower low water datum) associated with individual collections within each estuary during the study period 2014-2018. ARB –Aransas Bay; SAB – San Antonio Bay, MAB – Matagorda Bay; EMB – East Matagorda Bay; SBB – San Bernard and Brazos Rivers; GAL – Galveston Bay; C&J – Chambers and Jefferson County coastal waterways; SLA – Sabine Lake.

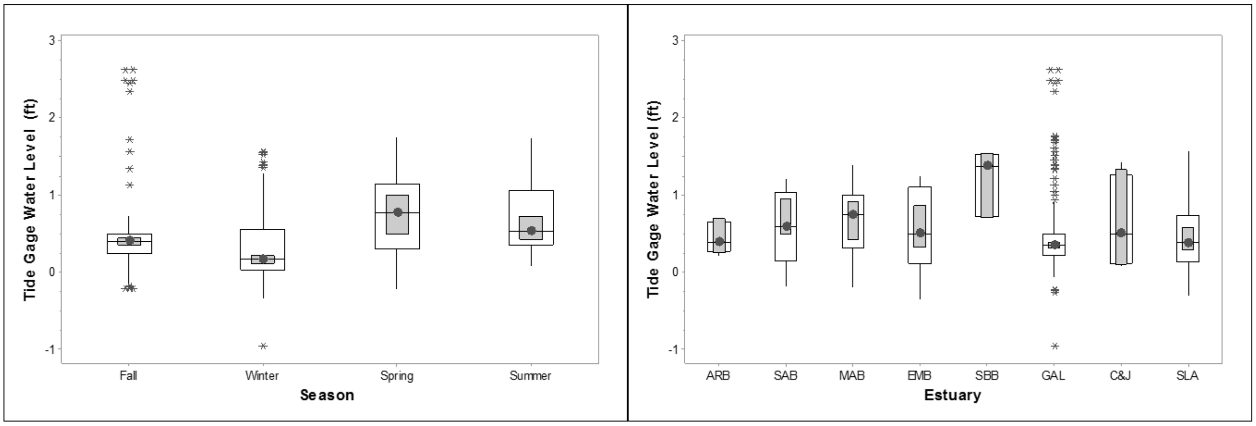


Figure 33. Boxplots of tide gage water level recorded in association with collection sites within each estuary and across all seasons during the study period in 2014-2018. ARB –Aransas Bay; SAB – San Antonio Bay, MAB – Matagorda Bay; EMB – East Matagorda Bay; SBB – San Bernard and Brazos Rivers; GAL – Galveston Bay; C&J – Chambers and Jefferson County coastal waterways; SLA – Sabine Lake. The gray inner box represents the 95% confidence interval of the median.

Catch per unit effort by gear type of *F. jenkinsi* using 2014-18 data, was compared to each water chemistry variable. Linear models were constructed to test the possible influence of these variables on *F. jenkinsi* CPUE. During the study period *F. jenkinsi* seine CPUE increased as salinity declined (zero-inflated Poisson, p< 0.001). Salinity was significantly lower at sites were *F. jenkinsi* were caught (p< 0.001) (Figure 34a). A predictive binomial GLM generated a probability catch curve for seine captured *F. jenkinsi* based on site salinity (Figure 34b). At sites where the salinity was equal or less than 10 ‰, the probability of catching *F. jenkinsi* in seines was greater than 20%. We failed to detect any significant difference in the salinity at sites where *F. jenkinsi* was captured using traps, versus where the species was not collected (*p* = 0.2515).

The catch of *F. jenkinsi* catch increased as water temperature declined (zero-inflated Poisson p < 0.001). In addition, water temperature was significantly lower at sites were *F. jenkinsi* were caught in seines (p< 0.001) (Figure 35a). The predictive binomial GLM generated a seine probability catch curve for *F. jenkinsi* based on site temperature (Figure 35b). At sites where the temperature was equal or less than 15 °C, the probability of catching *F. jenkinsi* in seines was greater than 40%. Temperature was also significantly lower at sites where *F. jenkinsi* was caught in traps (*p*= 0.001517). This likely represents a spurious correlation associated with seasonality when cold fronts generated lower water temperatures and shallower depths. Water levels measured at tide gages in the bay, were negatively correlated (r = -0.161, *p* = 0.002) with *F. jenkinsi* catch rates.

The catch of *F. jenkinsi* increased as dissolved oxygen levels increased (zero-inflated Poisson *p* = 0.0185). Dissolved oxygen was significantly higher at sites were *F. jenkinsi* were caught (p = 0.0038) (Figure 36a). A predictive binomial GLM (p = 0.0185) provided a probability catch curve for *F. jenkinsi* based on site dissolved oxygen (Figure 36b). There was no significant difference in dissolved oxygen between sites where *F. jenkinsi* was collected with traps versus where it was not captured (*p* = 0. 7088).

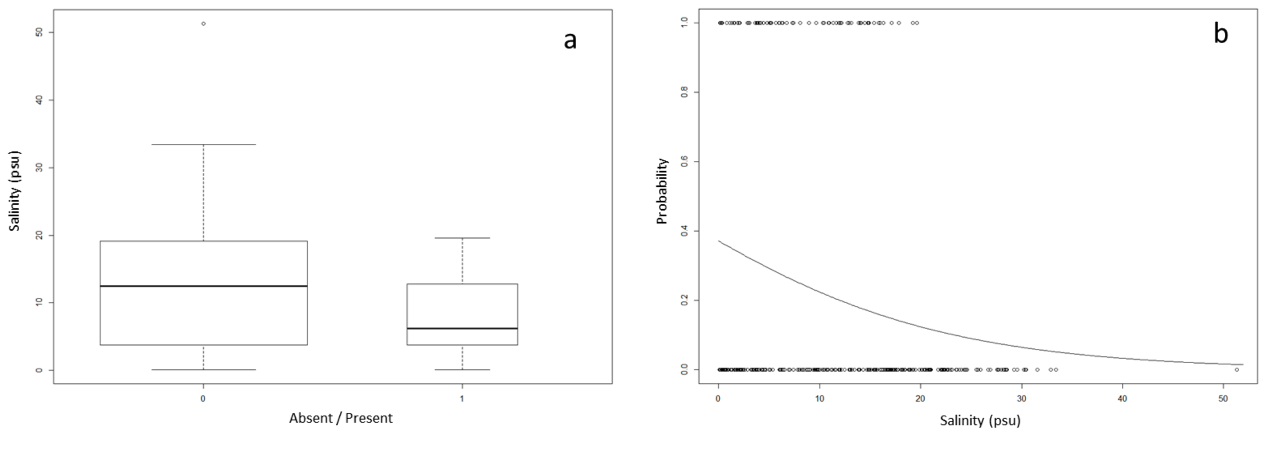

2014-18 Seine *F. jenkinsi*

Figure 34. a) Boxplot of the salinity of sites where *F. jenkinsi* was present versus not captured in 2014-18 seine samples and b) fitted binomial GLM applied to the presence/absence of *F. jenkinsi* by salinity.

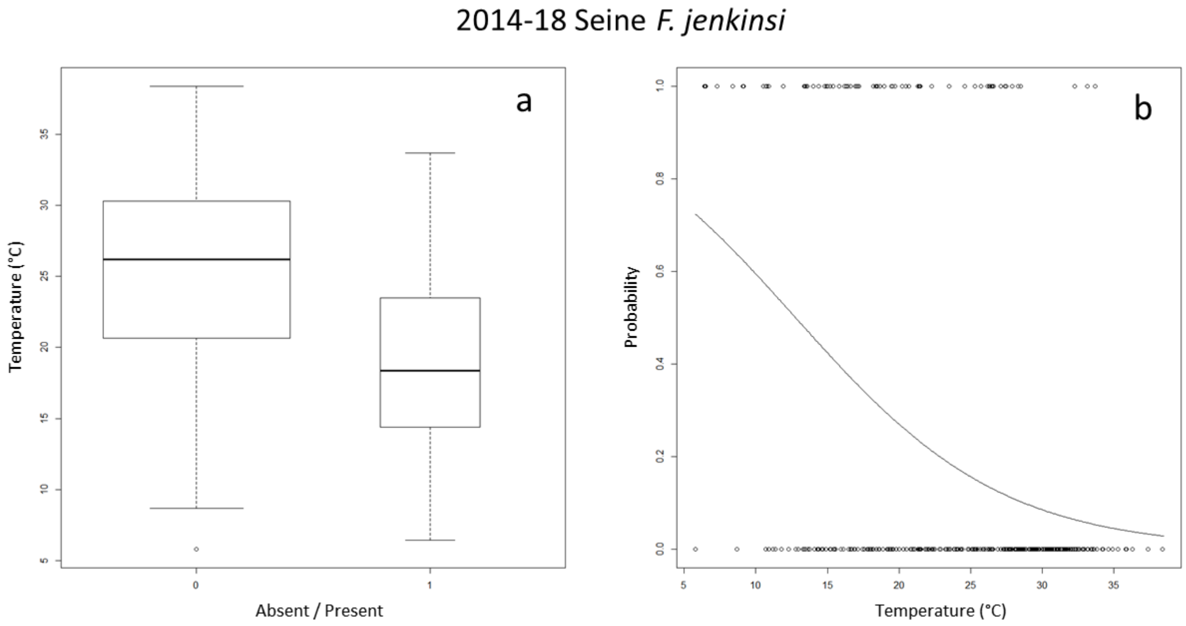


Figure 35 a) Boxplot of the surface water temperature (°C) of sites where *F. jenkinsi* was present versus not captured in 2014-18 seine samples and b) fitted binomial GLM applied to the presence/absence of *F. jenkinsi* by Temperature (°C).

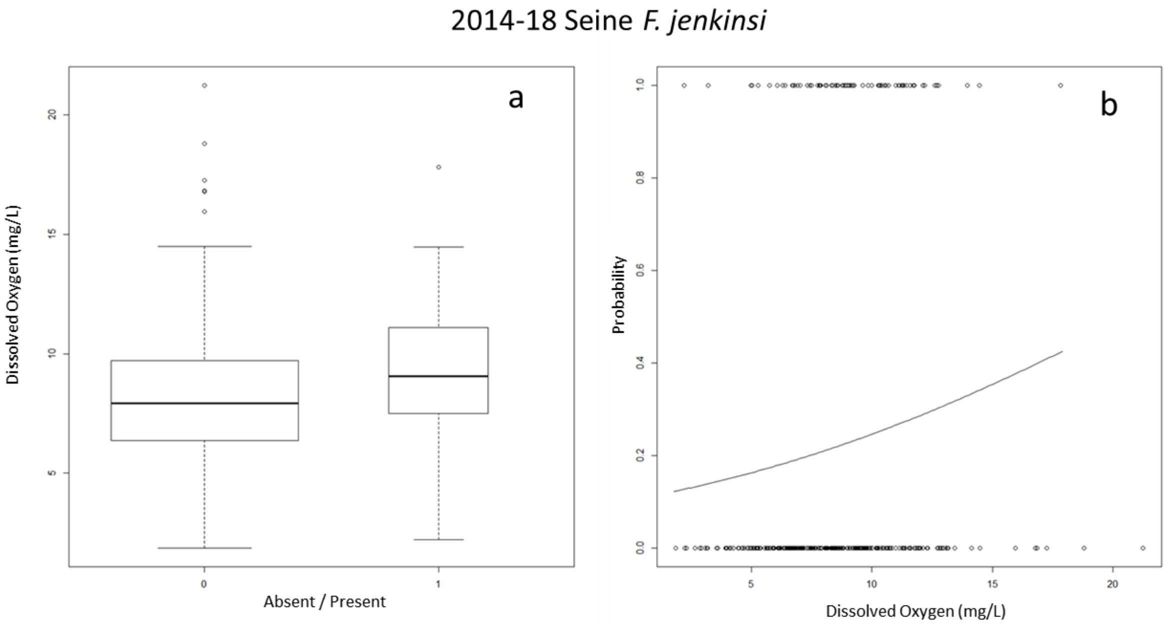


Figure 36. a) Boxplot of the Dissolved Oxygen (mg/L) of sites where *F. jenkinsi* was present or not caught in 2014-18 seine samples and b) fitted binomial GLM applied to the presence/absence of *F. jenkinsi* by Dissolved Oxygen (mg/L) .

Physical Habitat

F. jenkinsi catch increased as water (tide) level referenced to MLLW declined (zero-inflated Poisson $p < 0.001$). Tide level was significantly lower at sites were *F. jenkinsi* were caught ($p < 0.001$) (Figure 36a). A predictive binomial GLM ($p < 0.001$) provided a probability catch curve for *F. jenkinsi* based on site tide level (Figure 37 and 34b). At tide levels of 0 to -1 ft., the probability of capture ranged between 40 to 80%. There was no significant difference in tide level between sites where *F. jenkinsi* was captured versus not caught in traps ($p = 0.1596$).

There was no significant relationship between *F. jenkinsi* catch and sediment type for either seine or trap data ($p = 0.389$, and $p = 0.0826$ respectively). The majority (63%) of sites sampled had mud (clay) sediment present. Muck (silt) was the second most common sediment type present at 28% of the sites sampled. Few sites had other sediment types of sand, detritus, or shell hash. No *F. jenkinsi* were caught at sites with detritus or shell has as the dominant sediment type. The average CPUE of F. jenkinsi was similar for sites with sediment type of mud and muck (0.0239 and 0.0205 respectively), while catch was lower at sites with sand (0.005). Sediment is likely not a highly influential factor for *F. jenkinsi* distribution or abundance.

F. jenkinsi are significantly more likely to be caught at sites where *Spartina alterniflora* is the dominant vegetation compared to *Phragmites australis*, and *Poaceae sp*. (pairwise Wilcox test $p < 0.001$ and $p = 0.0185$ respectively). While *S. alterniflora* was the most common dominant vegetation species when *F. jenkinsi* was present (in both seines and traps); at sites where they were not caught, *S. alterniflora* was also the dominant vegetation (Table 20). Other dominant vegetation species that were present *when F. jenkinsi* were present were: *P. australis, Rupia maritima, Spartina patens*, and *Typha sp*. We were significantly more likely to catch *F. jenkinsi* when the percent cover of vegetation was lower (<20% and <40% median cover) for both seines and traps ($p=0.0136$, and $p=0.0469$) respectively (Figure 38). However, the predictive binomial GLM was not found to be useful as the maximum predictive value was less than 20% for both seine and trap data. Furthermore it is likely that the significant relationship between percent cover of vegetation and catch was a result of gear efficiency, and may not reflect true habitat association.

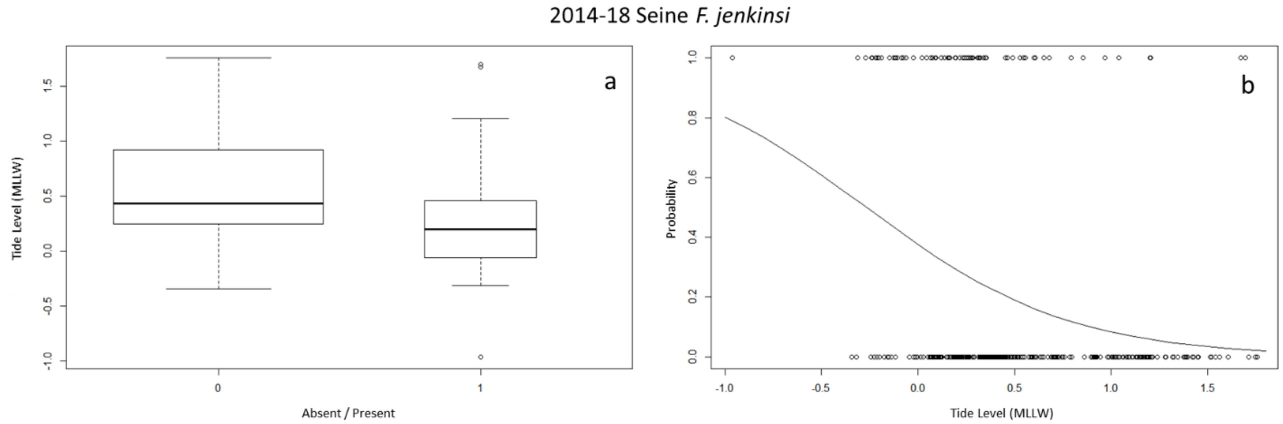


Figure 37 a) Boxplot of the tide level (ft. datum: MLLW) of sites where *F. jenkinsi* was present or not caught in 2014-18 seine samples and b) fitted binomial GLM applied to the presence/absence of *F. jenkinsi* by tide level.

Table 20 Percent occurrence of the most common dominant plant species at either seine or trap replicates where *F. jenkinsi* were captured and not captured.

Seine 2014-18

	Percent Occurrence	
Dominant Species	F. jenkinsi Present	*F. jenkinsi Absent*
Spartina alterniflora	41.1	60.0
Phragmites australis	23.4	6.4
Rupia maritima	5.7	0.4
Spartina patens	5.7	5.3
Typha sp.	5.0	2.3
No Vegetation	4.3	3.9
Alternanthera philoxeroides	3.5	2.0
Distichlis spicata	2.8	2.9
Spartina spartinae	2.1	1.2
Halodule wrightii	1.4	0.1
Schoenoplectus californicus	1.4	0.2
Iva frutescens	0.7	1.3
Juncus roemarianus	0.7	0.3

Trap 2014-18

	Percent Occurrence	
Dominant Species	F. jenkinsi Present	*F. jenkinsi Absent*
Spartina alterniflora	32.4	47.3
Phragmites australis	11.8	5.2
Typha sp.	11.8	5.3
Schoenoplectus sp.	8.8	2.9
Alternanthera philoxeroides	5.9	3.7
Distichlis spicata	5.9	3.9
No Vegetation	5.9	3.4
Isolepis cernia	2.9	0.5
Juncus roemarianus	2.9	0.3
Scirpus americanus	2.9	2.7
Spartina patens	2.9	5.8
Zizaniopsis miliacea	2.9	0.2

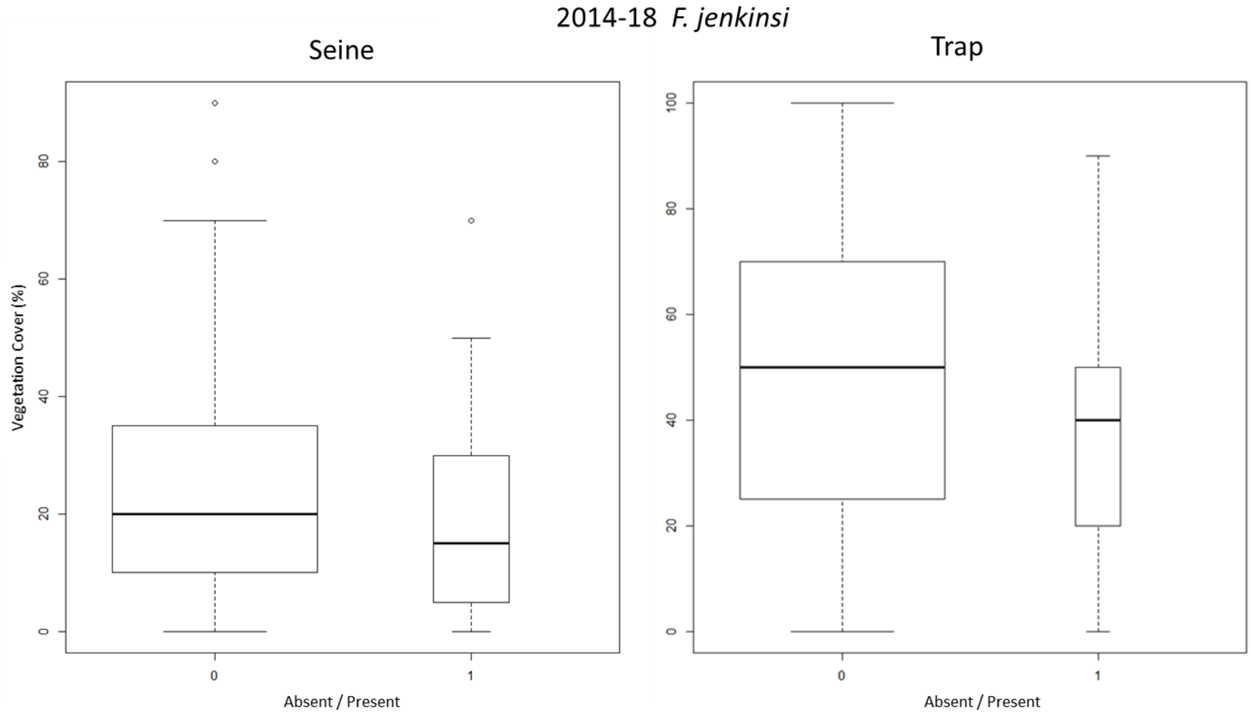


Figure 38. Boxplots of the vegetation cover (%) at sites comparing where *F. jenkinsi* was present versus not captured during 2014-18 by gear type.

Life History Characteristics

Size Distribution

The standard lengths of *F. jenkinsi* individuals caught throughout the study were not normally distributed. The standard lengths of *F. jenkinsi* captured ranged from 13.1-50.0 mm with an average length of 26.70 mm (Figure 39). Examination of the length frequency histogram suggests a multimodal distribution of multiple age groups. Monthly standard length histograms further documents the multimodal distribution, which is most evident during the month of March (Figure 40). Unfortunately, due to the low catch rates during April through October it was not possible to utilize length frequency modal analysis tools (e.g. NORMSEP) available through the FISAT II software package or other statistical packages (Figure 41). The lower proportion of smaller (< 30 mm) size individuals collected during November and the dominance of individuals 32-48 mm SL indicates that few individuals likely live past this size range, and most individuals likely do not live more than 1 year of age (Figure 39). The standard length of *F. jenkinsi* was significantly different between seasons ($p<0.001$) with individuals captured in summer being significantly smaller than individuals caught during other seasons (Figure 42 and 43). However, catch was low in the summer months, and highest in the winter.

Individuals selected for GSI analysis were also used to assess standard length distribution by gender (Figure 44 and 45). On average, females were 28.5 mm (± 7.65 mm SD) with a range of 13.1-50.0 mm and males were an average of 26.9 mm (± 6.34 mm SD) with a range of 14.3-46.0 mm. Mean standard length of male *F. jenkinsi* was statistically (p = 0.01673) shorter, however this may be attributed to the higher number of females collected (total n females = 365, males = 276) and is likely not biologically significant due to the large overlap in length by sex.

The mean total body weight of *F. jenkinsi* did not significantly differ between genders (p=0.058). Data was therefore pooled from both genders to plot the association of *F. jenkinsi* length and total weight (Figure 46). The best-fit model for the relationship between length and weight in our data series was a power equation ($r^2 = 0.9708$) where for every unit increase in standard length (x) total body weight would also increase by $6.024x^{0.297}$ milligrams.

Reproduction

A total of 152 individuals (365 females and 276 males) were collected for GSI analysis with the largest female measuring 50 mm and the largest male measuring 46 mm (Figure 45). Elevated GSI values were observed during April, May, and August and September for females and March - August for males (Figure 47).

The sample sizes of each ovarian phase were underdeveloped (n = 6), latent (n= 139), early maturing (n= 154), late maturing (n= 46), mature (n= 13, and ripe (n=5). However, both female standard length and female GSI values showed a relationship to ovary phase (Figure 48 and 49). Ovary maturation phase increased with larger standard length values as well as increased with greater GSI values.

Ovarian phases were pooled by season in order to assess temporal shifts in the reproductive activity of *F. jenkinsi* (Figure 50). Ovarian development showed signs of seasonal progression. Spring and summer showed the greatest percentages of late maturing or greater gonads.

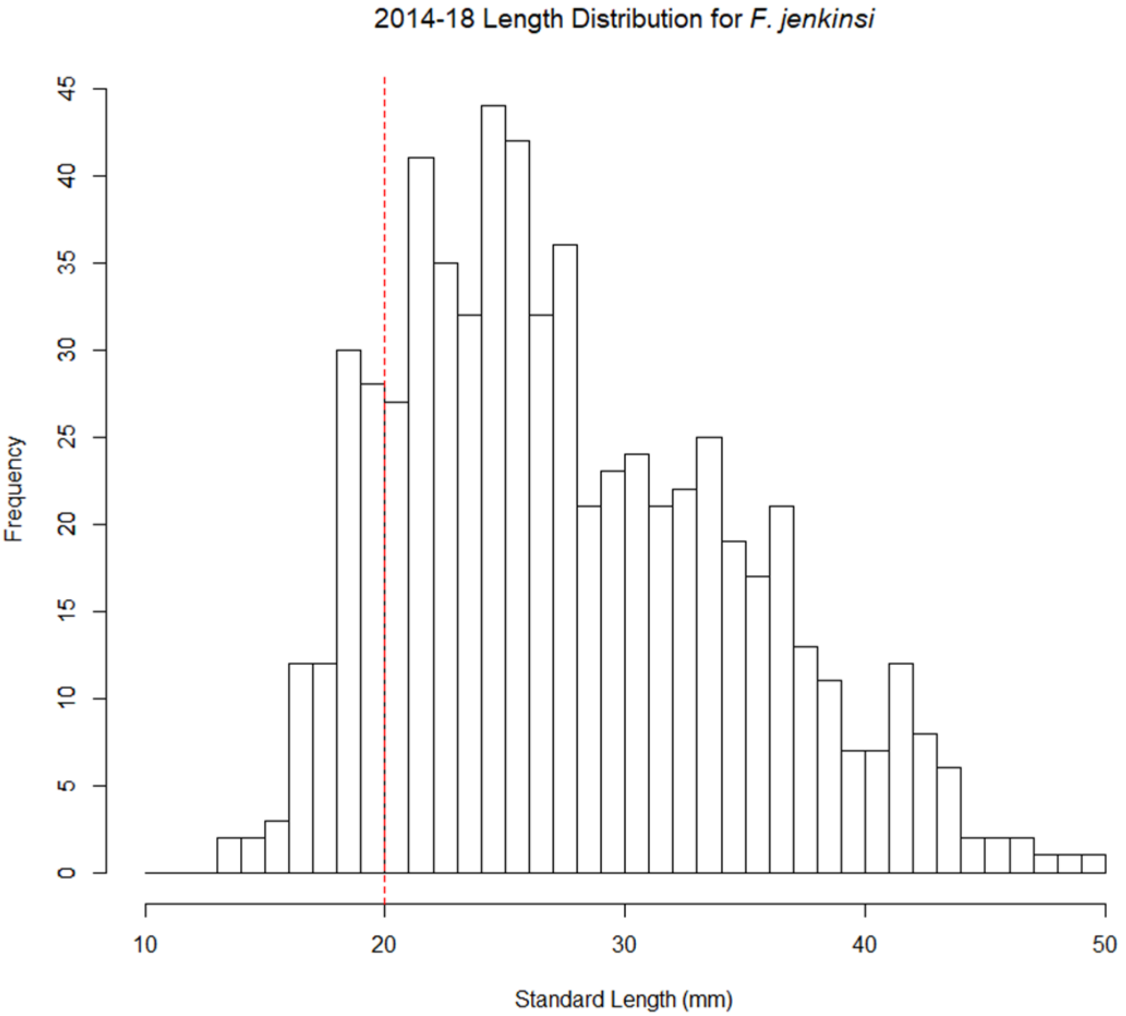


Figure 39. Length distribution of *F. jenkinsi* across all 2014-2018 sampling events and all seasons. Red vertical line represents break between juveniles and adults at 20mm Standard Length (Lopez et al. 2010).

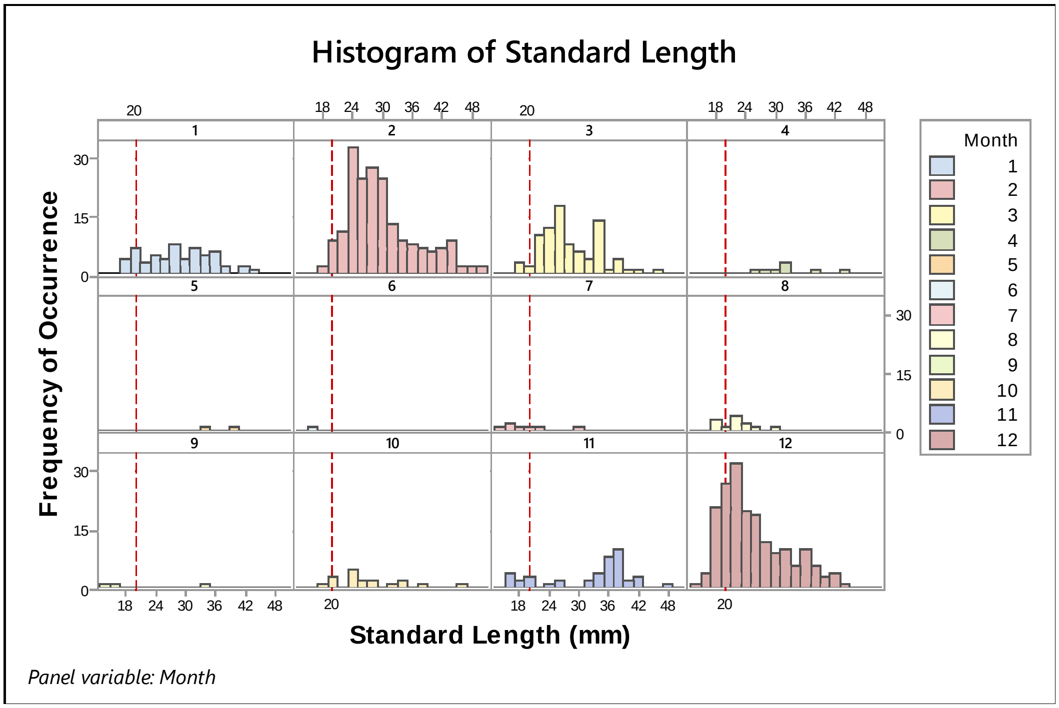


Figure 40. Monthly length distribution of *F. jenkinsi* pooled across all 2014-2018 sampling events. Red vertical line represents break between juveniles and adults at 20mm Standard Length (Lopez et al. 2010).

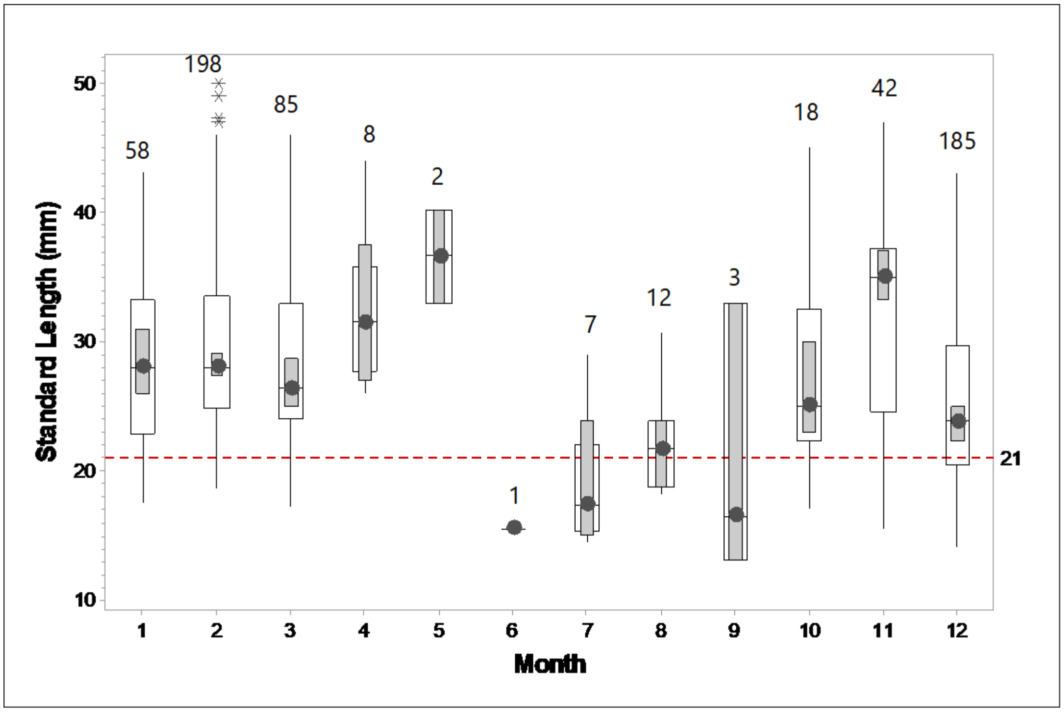


Figure 41. Boxplot of standard length of *F. jenkinsi* pooled by month for all 2014-2018 sampling events. Red line represents delineation between juveniles and adults (Lopez et al. 2010).

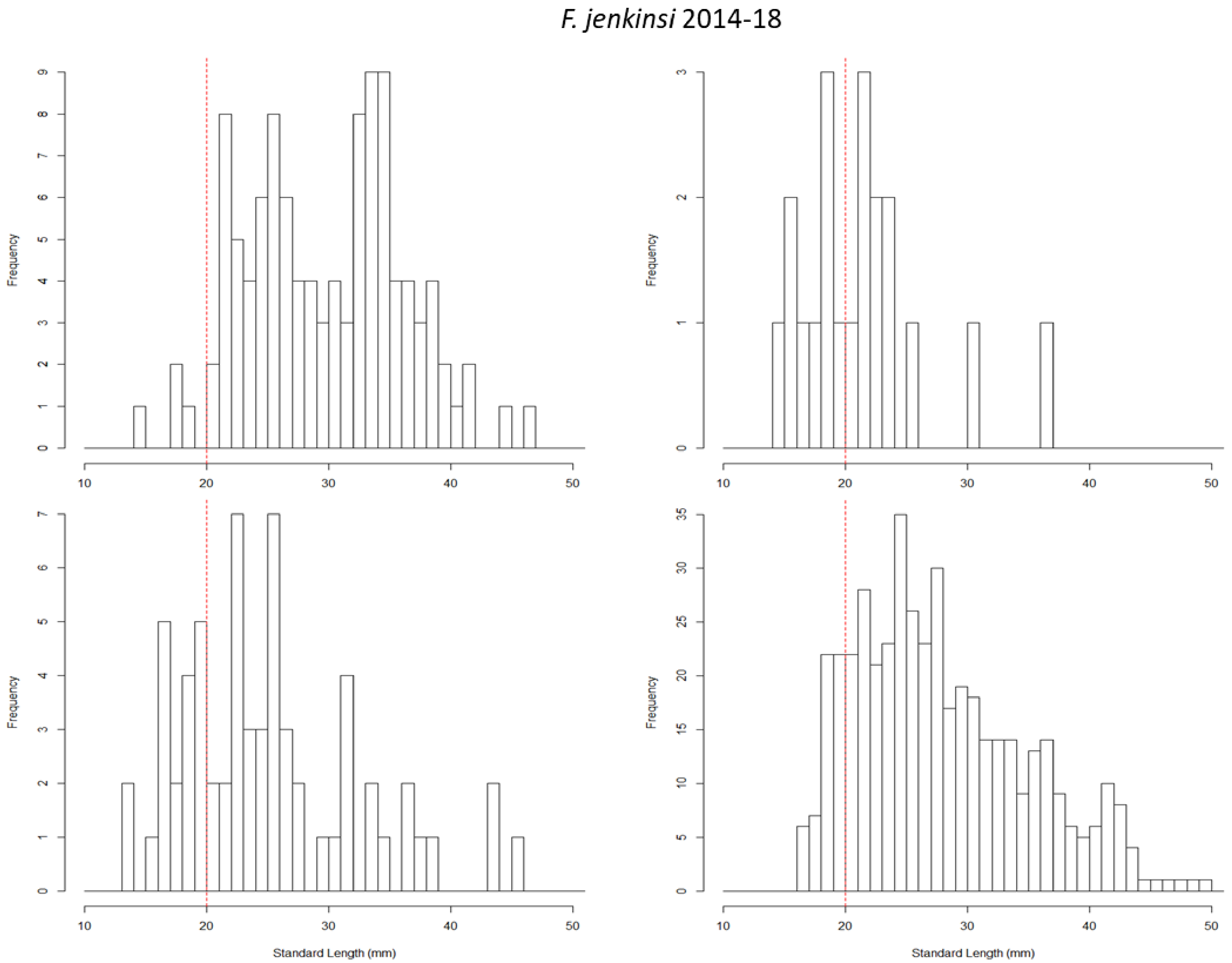


Figure 42. Length distribution of *F. jenkinsi* by season for all 2014-2018 sampling events. Red vertical line represents delineation between juveniles and adults (Lopez et al. 2010).

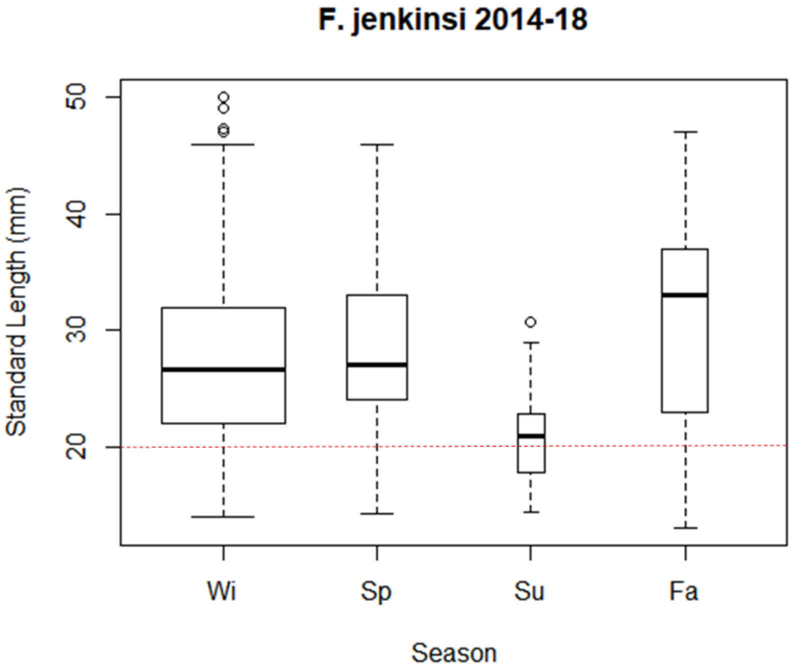


Figure 43. Boxplot of standard length of *F. jenkinsi* by season for all 2014-2018 sampling events. Red line represents delineation between juveniles and adults (Lopez et al. 2010).

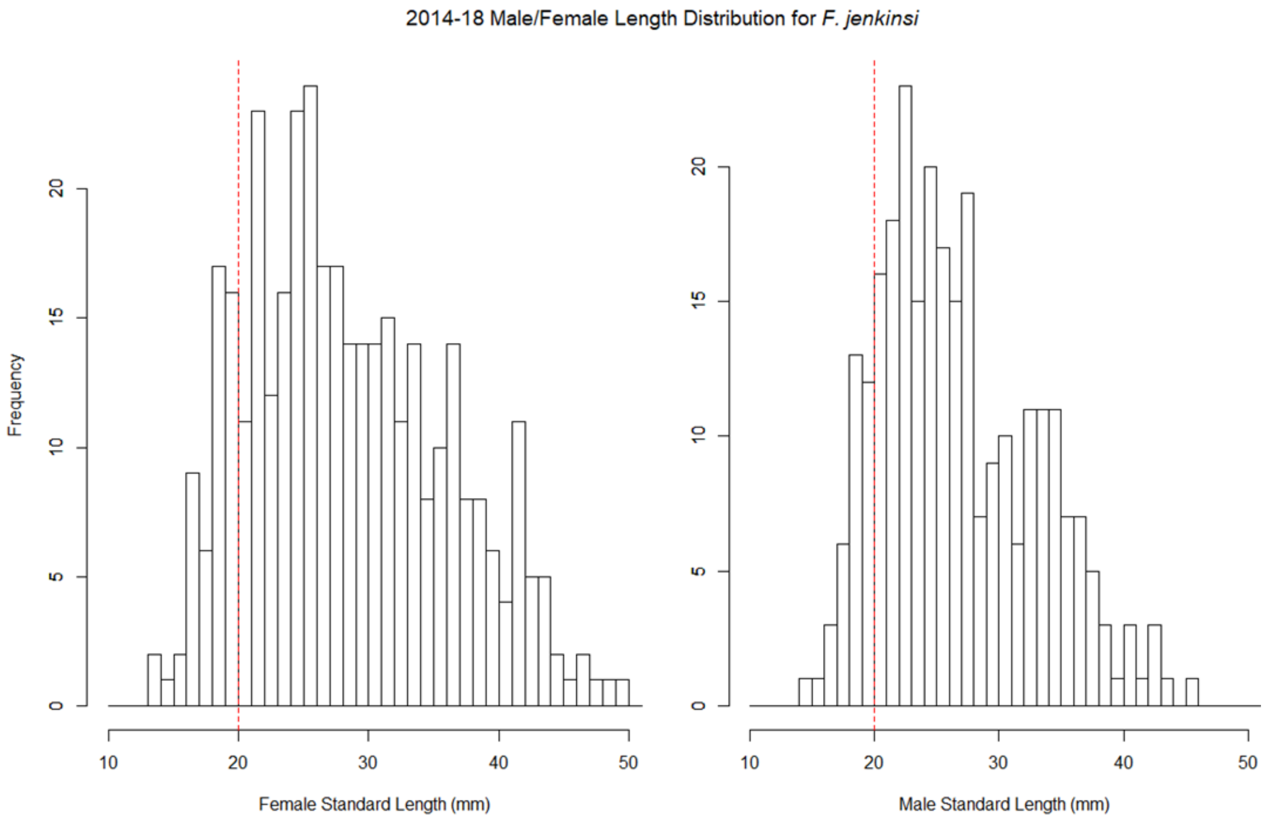


Figure 44 Length distribution of *F. jenkinsi* by sex for all 2014-2018 sampling events. Red vertical line represents break between juveniles and adults (Lopez et al. 2010).

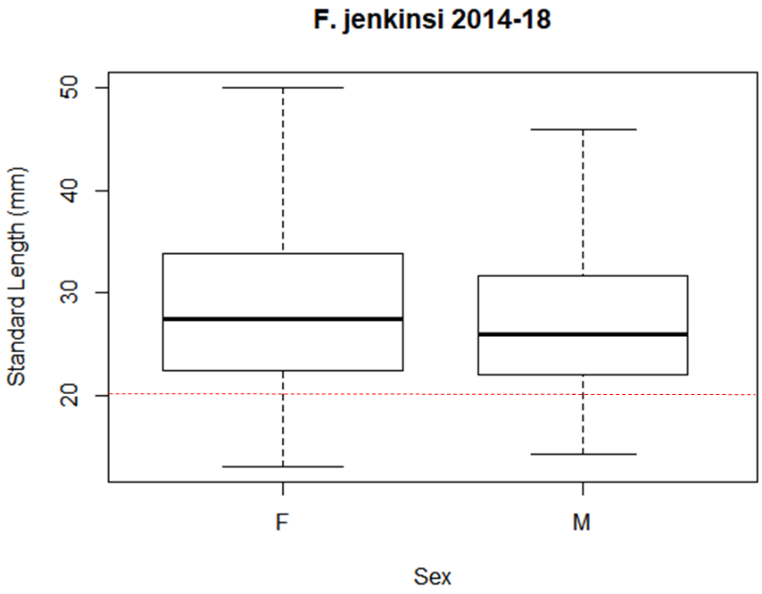


Figure 45. Boxplot of standard length (mm) of *F. jenkinsi* by sex for all 2014-2018 sampling events. Red line represents break between juveniles and adults (Lopez et al. 2010).

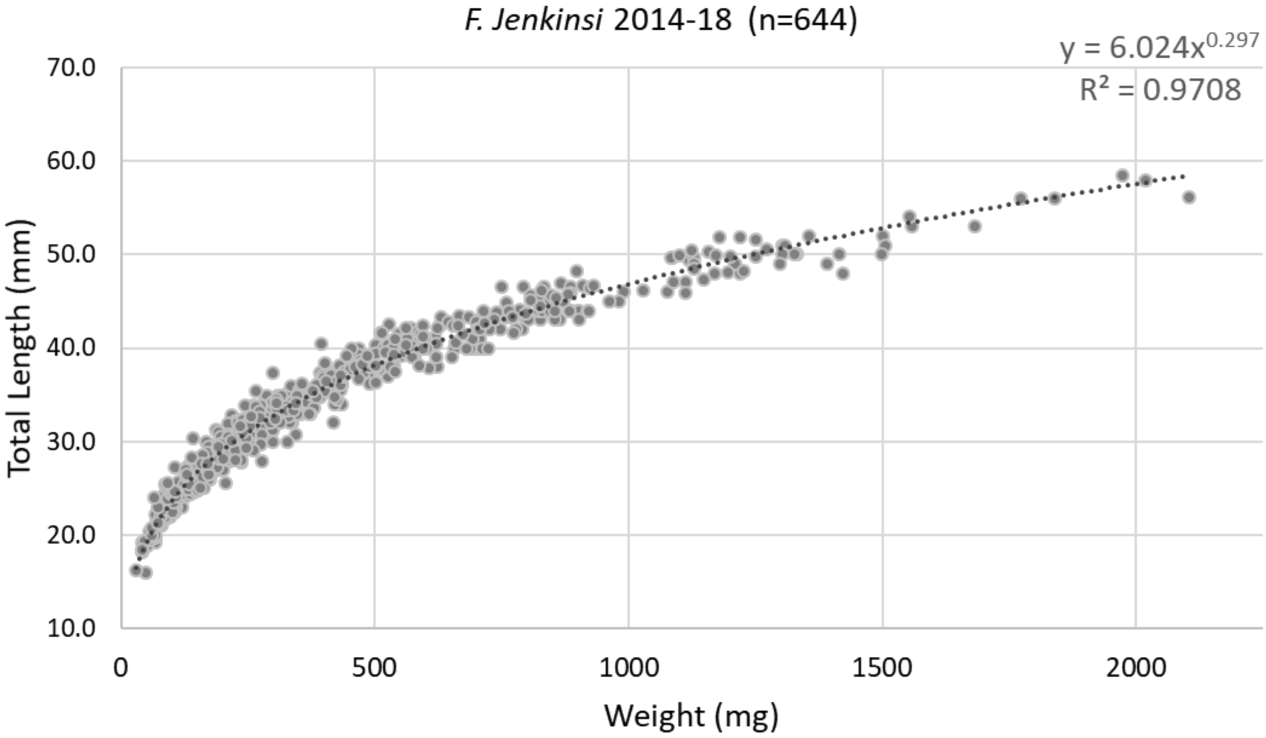


Figure 46. Scatter plot of *F. jenkinsi* total length (mm) versus total body weight (mg).

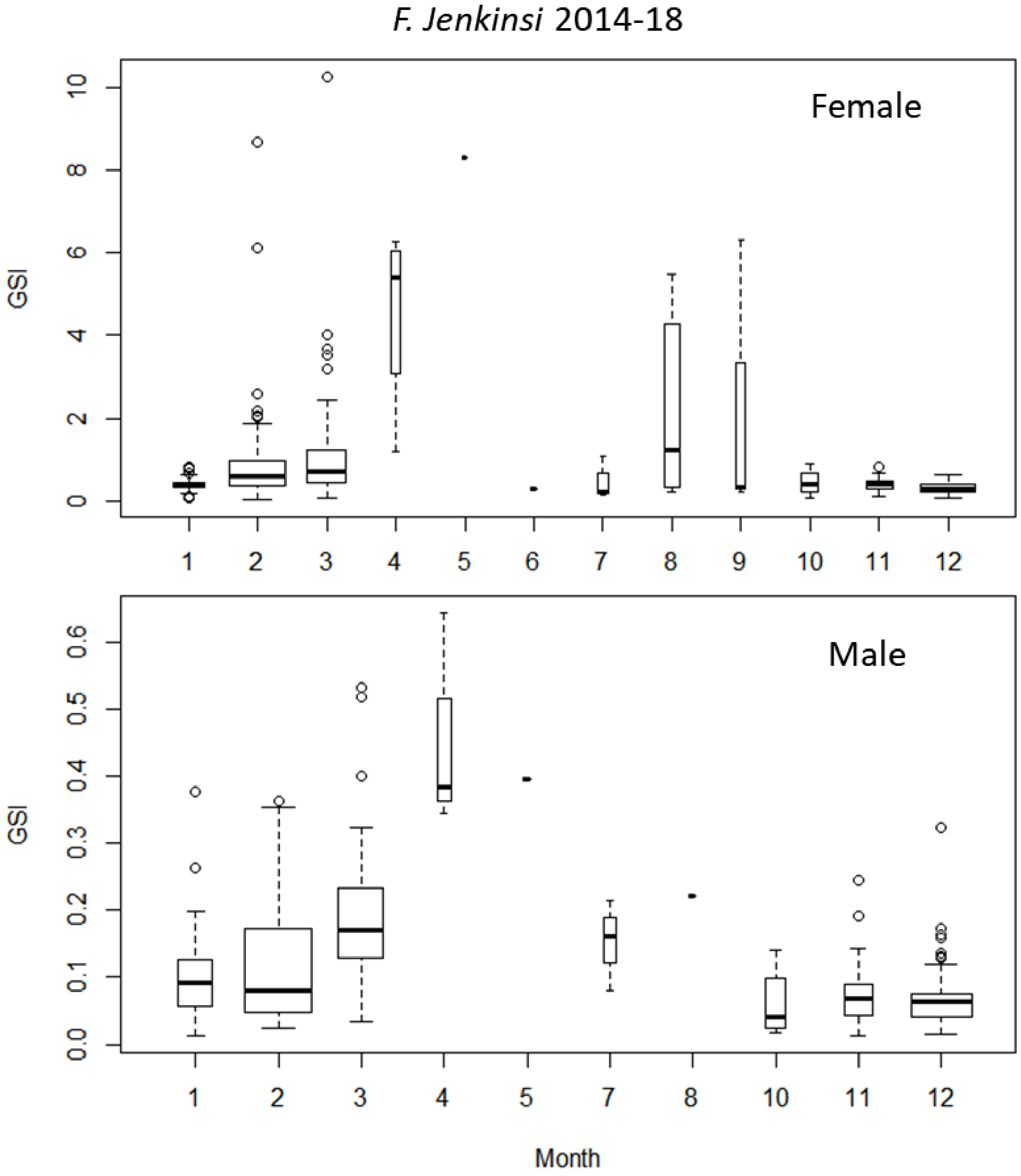


Figure 47. Boxplot of female (above) and male (below) gonadosomatic index (GSI) by month for *F. jenkinsi*.

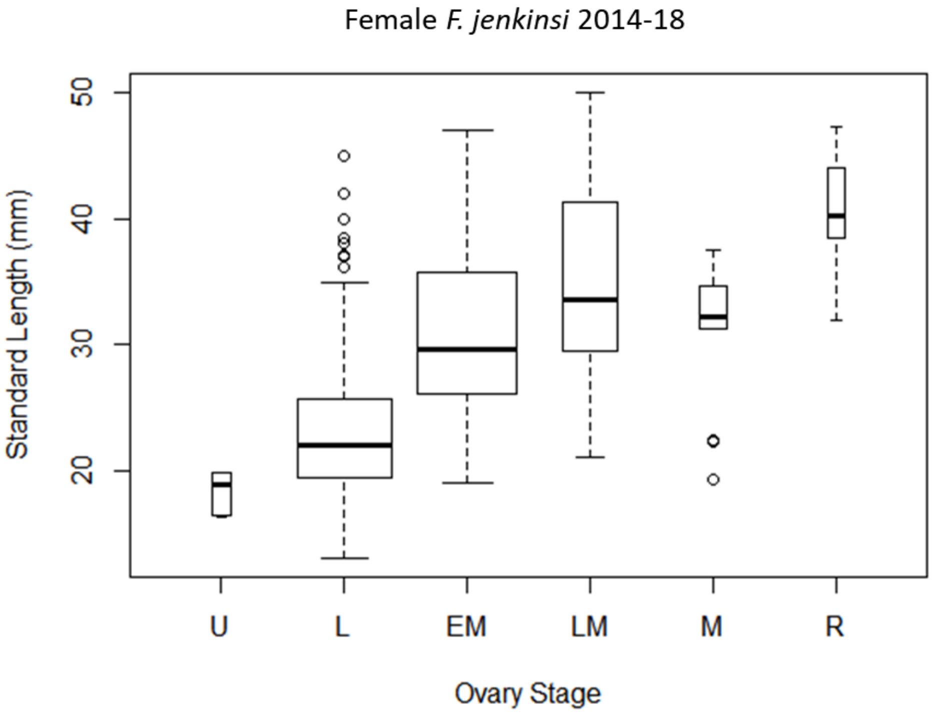


Figure 48. Boxplot of female F. jenkinsi ovarian phase by standard length (mm). (U = undeveloped, L= latent, EM= early maturing, LM= late maturing, M= mature, R= ripe).

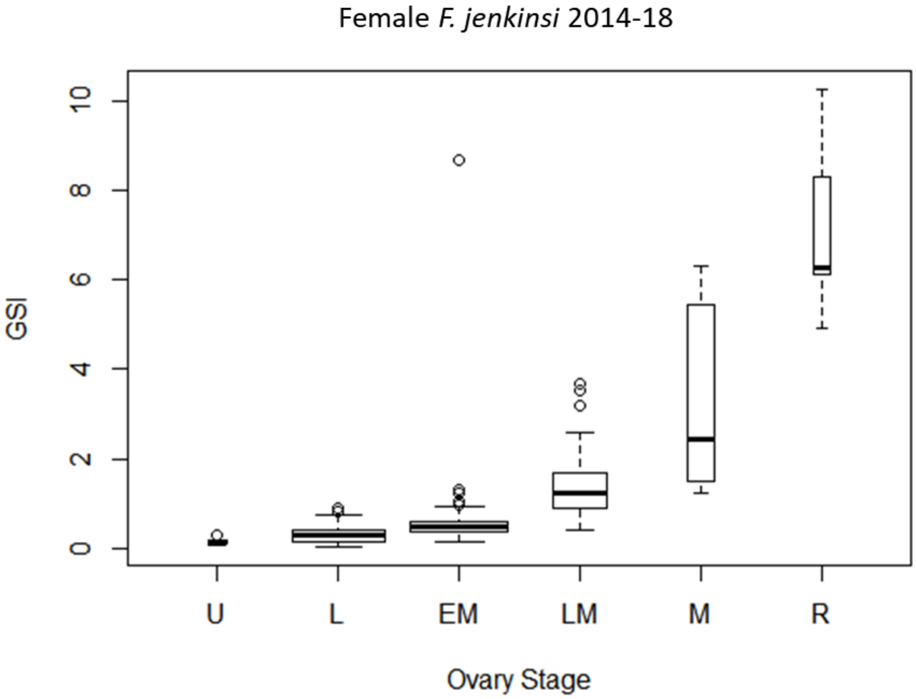


Figure 49. Boxplot of female *F. jenkinsi* ovarian phase distribution by GSI values (U = undeveloped, L= latent, EM= early maturing, LM= late maturing, M= mature, R= ripe).

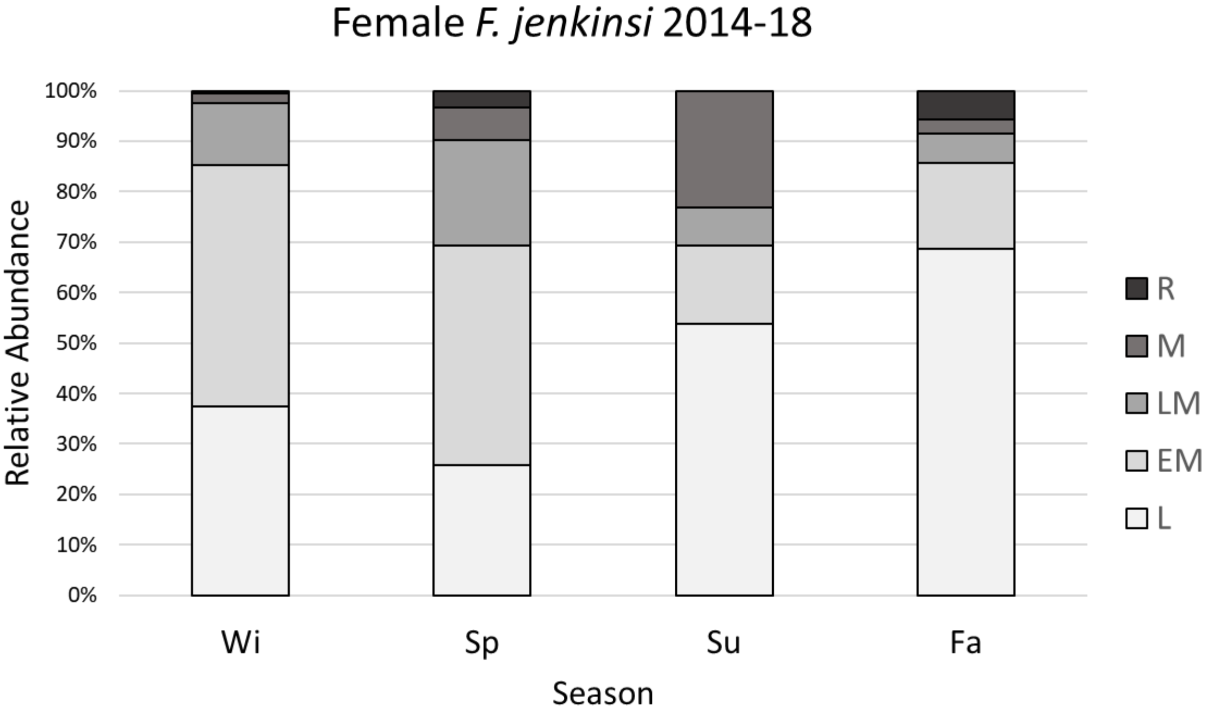


Figure 50. Seasonal (spring, summer, fall, winter) percentages by ovarian phases of female *F. jenkinsi*.

Estimation of Life History Parameters

The primary objective of the current study was to provide preliminary estimates of the parameters of the von-Bertalanffy growth equation and other demographic parameters such as instantaneous rate of natural mortality (M) (von Bertalanffy 1938; Haddon 2011). We utilized a variety of indirect methods based on empirical relationships to obtain preliminary estimates of the parameters of the von Bertanlanffy equation and subsequently used those parameters to estimate natural mortality.

The largest *F. jenkinsi* collected during the 2014-2018 study period in Texas was 50 mm SL. This is similar to maximum lengths of 42.6, 50.0, 55.0, 60, and 65 mm SL reported by other investigators in the eastern Gulf of Mexico (Thompson, B.A. 1999; Hoese and Moore 1998, Lopez et al. 2011; NOAA 2009; McEachran and Fechhelm 1998). Using 50 mm SL as the best estimate of L_{max} for Texas L_∞ was estimated to be 54.0 mm using equation 2 referenced in the methods section. The 95% confidence interval of L_∞ was estimated to be between 45 and 64 mm SL.

Using 54 mm SL as the best estimate of L_∞ for Texas the L_m was calculated using equation 3 in the methods section to be 38.0 mm SL. The 95% confidence interval of L_m was 28 to 51 mm SL. The lower limit of 28 mm SL is close to the < 20 mm SL threshold used to designate immature life stages of *F. jenkinsi* as reported by Lopez et al. (2010).

Time (age) of maturity was estimated upon visual examination of length frequency distributions. Based on this limited data *F. jenkinsi* probably reached maturity within 6 months (0.5 years = t_m). Using this as input for t_m we estimated the likely maximum lifespan t_{max} using equation 4 to be 1.8 years with a 95%

confidence interval of 1.2 to 2.0 years. The parameter t_{max} was used to estimate the growth parameter K using equation 5 described in the methods. When t_{max} was set to a value of 1.8 years, a point estimate of K = 1.67/year was generated.

The final parameter of the von Bertalanffy growth function which needed to be estimated was t_0. Using the empirical equation 6 described by Pauly (1979) and our estimates of L_∞ and K we estimated t_0 to be -0.17. The final form of the von Bertanlanffy growth equation for *F. jenkinsi* with all three estimated parameters is illustrated below in equation 8 and graphically depicted in Figure 51.

Equation 8. von Bertanlanfy growth model for *F. jenkinsi* $L_t = 54(1 - e^{-1.67(t+0.17)})$

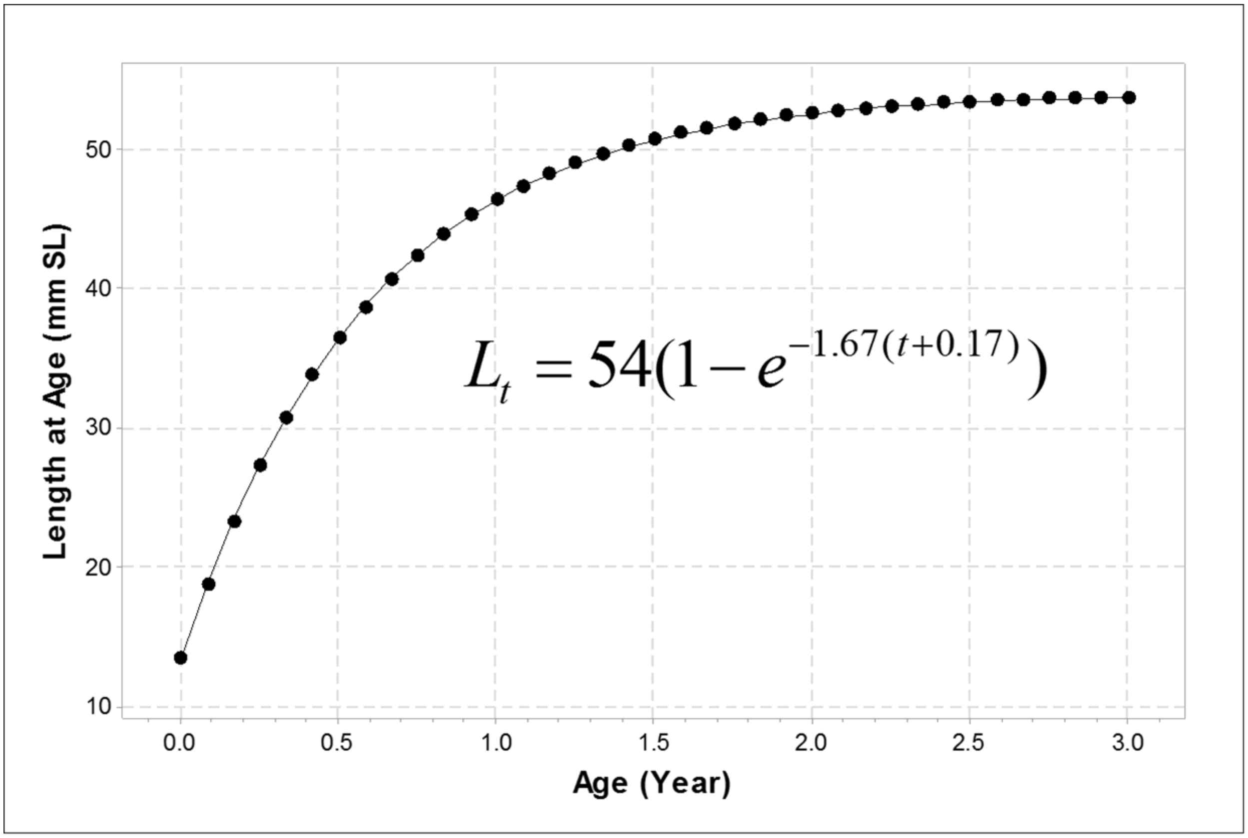


Figure 51. Graphical representation of the estimated von Bertanlanfy growth function of length at age versus age for *F. jenkinsi.*

Equation 7 in the methods section, which was developed by Pauly (1980) was used to estimate instantaneous natural mortality (M) from the parameters of the von Bertalanffy growth equation parameters (L_∞ and K) and mean environmental temperature (T, in degrees Celsius).

For *F. jenkinsi* the following assumptions and input were used to estimate M:

T = average water temperature estimate from the study period which was 23.6 ºC.

L_∞ = asymptotic length = 54 mm SL

K = growth rate coefficient = 1.67

Based on these input parameters the predictive model of M for *F. jenkinsi* based on estimates of L_∞ and K derived from the von Bertanlanfy equation and average annual temperature T is illustrated below in equation 10 (Pauly 1980).

Equation 9: $\mathbf{Log_{10}\ M = -0.0066 - 0.279\ log_{10}\ (54) + 0.6543\ log_{10}\ (1.67) + 0.4634\ log_{10}\ (23.6)}$

Using these values as input the point estimate of instantaneous natural mortality (M) for *F. jenkinsi* was 2.61/year with a 95% confidence interval of 1.72 to 3.95/year. The instantaneous rate of natural mortality (M; 1/year) refers to the late juvenile and adult phases of a population. Using the following formula you can convert M to annual rates (*v*). Therefore annual natural mortality *(v)* = $1\text{-}e^{-M} = 1\text{-}e^{-2.61}$ = 92.6% annual mortality. The 95% confidence interval would range between 82.0 to 98.1% annual mortality. This suggests that very few *F. jenkinsi* live beyond 1 year old.

Relationship of External Morphometric Characteristics and Sex

A total of 462 specimens collected during 2017 and 2018 were selected for analysis of external morphometric traits that might be used to identify the sex of Saltmarsh Topminnow. Only positively identified male and female fish based on internal examination of gonads were used. Since the full suite of external morphometric and meristic measurements were not measured during 2014-2015 that data set was not used. Specific external traits that were examined are listed on Figure 4. The specimens included in this analysis ranged between 13.1 and 47.3 mm SL with median and average size of 25.7 and 26.2 mm SL respectively. Although there was considerable overlap in the distribution of the length of male and female specimens, a higher frequency of larger (> 21 mm SL) individuals were female.

All morphometric variables exhibited significant ($p < 0.05$) negative correlations with the presence of male fish. Anal ray fin count (ARCt) and dorsal ray fin count (DRCt) were the only measurements that did not exhibit a significant correlation with sex and where not considered for further statistical analysis. The presence of an anal sheath exhibited the most significant ($p < 0.001$) and largest ($r = -0.659$) absolute correlation with the occurrence of male specimens suggesting an inverse relationship between the occurrence of the anal sheath and the likelihood of the specimen being male. We also found highly significant ($p < 0.001$, $r > 0.732$) positive correlations between standard length (SL) and most of the other morphometric measurements with the exception of ARCt and DRCt. Standard length exhibited only a weak ($r = 0.1950$) significant positive correlation with ARCt and failed to exhibit any significant correlation with DRCt. Standard length exhibited weak but significant correlations with sex ($r = -0.1440$, $p = 0.0020$) and sheath presence ($r = 0.3280$, $p < 0.0001$). This suggests that larger specimens were more likely to be females and possess an anal sheath.

Stepwise logistic regression analysis yielded the following equation which produced the highest deviance adjusted r^2 value (43.78%) and lowest AIC (359.41).

Y' = 1.034 + 0.356 StoDorLen - 0.372 SL + 1.077 Depth + 1.303 Cpdepth - 1.565 Hwidth

- 0.0 Sheath (code 0 - absent) - 6.088 Sheath (code 1 - present).

In this model the categorical dichotomous predictor variable sheath presence was recoded and split into two categorical variables, sheath 0 = absent and sheath 1 = present.

Under this model the probability of a specimen being male (code 1) is = π, where $\pi = \frac{e^{Y'}}{(1+e^{Y'})}$.

The largest coefficient value (in absolute terms) was the sheath (code 1) variable, -6.088. The sheath 0 and 1 variables combined accounted for 40.07% of the total adjusted Deviance R^2. Based on the Deviance Goodness of Fit (GOF) test the model appears to fit the data well (p value = 1). However, the Pearson and Hosmer-Lemeshow GOF tests indicated a poor fit (p values low). It should be noted however that all of the GOF tests may perform erratically under conditions where multiple continuous predictors are used. Furthermore when the sample size is large it is often not possible to find any reasonably parsimonious model with a p-value greater than 0.05 (Allison 2014; Hosmer and Hjort 2000). Since the presence of the anal sheath appeared to be the variable most strongly associated with the sex of the fish, we also re-ran the logistic regression with sheath status as the only predictor variable. Logistic regression yielded the following equation with a deviance adjusted r^2 value (40.07%) and an AIC (377.61)

Y' = 0.720 – 5.231 Sheath (1 or 0).

And the probability of a specimen being a male (code 1) is equal to $\pi = \frac{e^{Y'}}{(1+e^{Y'})}$.

Deviance and Pearson GOF tests indicated the model fit the data well. The Hosmer and Lemeshow GOF test was not conducted due to lack of degree of freedom. A chi-square test was also conducted to test for correlation between the categorical variables sex and sheath presence. The chi-square tests indicated that there was a significant positive relationship between the presence of an anal sheath and the probability of a specimen being female. However, this relationship was not consistent since there were still many (33.3%) females lacking an anal sheath (Figure 52 and Figure 53). The size of the specimens also affects the likelihood of expression of the anal sheath characteristic in females (Figure 54). As noted earlier standard length was identified as one of the continuous variables identified as being associated with the probability of being male or female. All male specimens larger than the 1st decile (18.8 mm SL) lacked an anal sheath (Figure 54). Therefore the presence of an anal sheath in any specimen larger than 18.8 mm SL would definitely indicate that the organism is not a male. However, the false negative rate (females assigned as males) based on lack of a sheath was relatively high (70%) in smaller specimens declining to zero percent in the upper decile (> 35.80 mm SL) of individuals examined (Figure 55 and 56). Female specimens measuring ≥32.82 to 35.80 possessed an anal sheath in 85% of the cases. Therefore we conclude that the probability that a female possesses an anal sheath is highly probable (≥ 85% chance) in larger (>32.82 mm SL) specimens of Saltmarsh Topminnow. This probability declines to 81.0% when specimens that are ≥29.46 to 32.82 mm SL are evaluated. The likelihood of a female have an anal sheath declines below 73% in smaller fish (Figure 54 and 55).

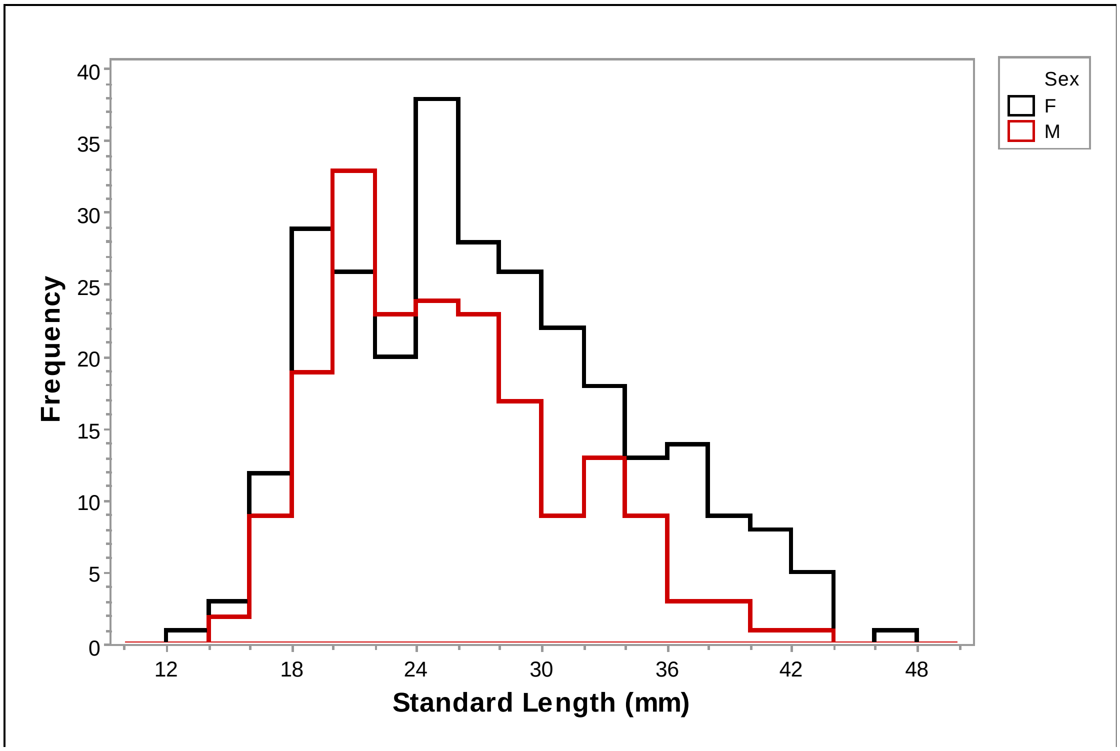


Figure 52. Length frequency of male and female fishes (n = 462) collected during 2017-18 that were used to evaluate external characteristics for identifying the sex of Saltmarsh Topminnow.

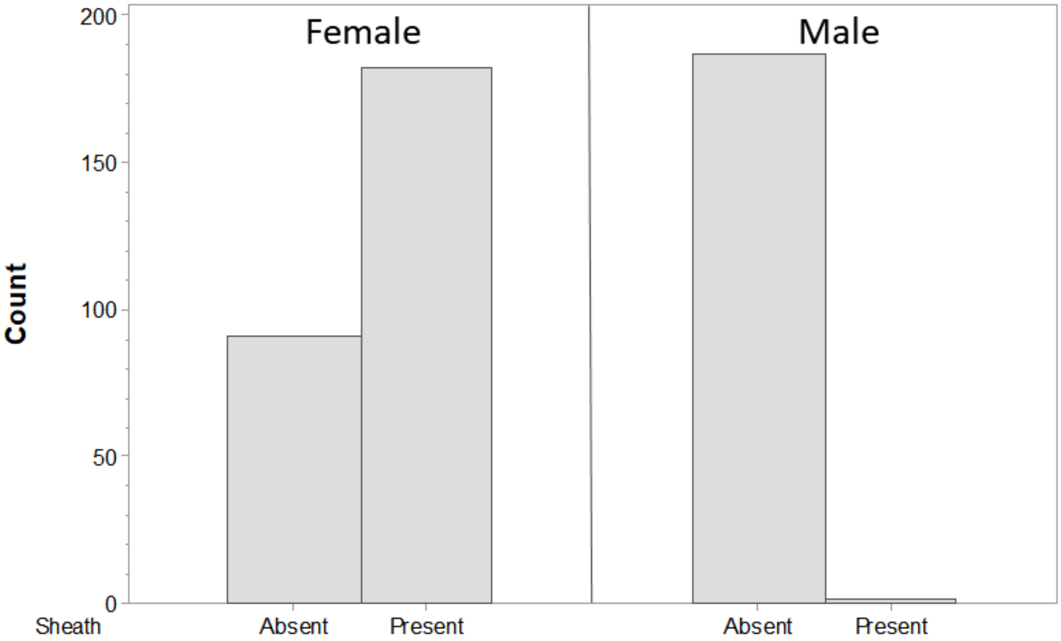


Figure 53. Frequency of individual male and female fishes (n = 462) possessing an anal sheath during 2017-18 that were used to evaluate external characteristics for identifying the sex of Saltmarsh Topminnow.

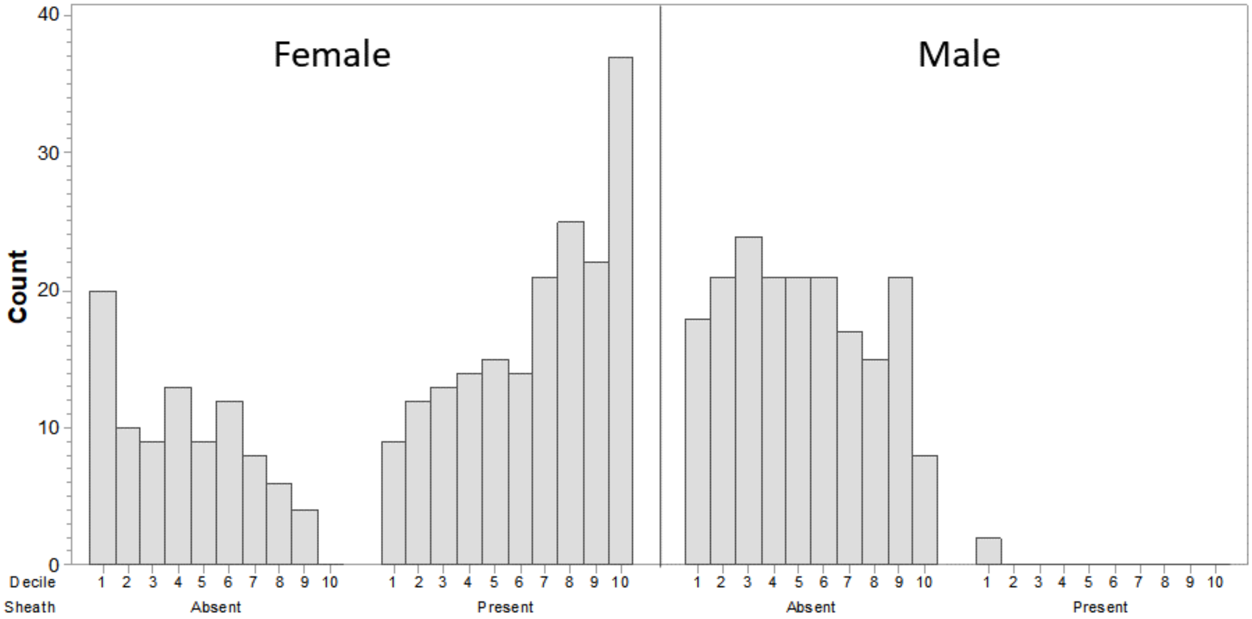


Figure 54. Frequency of individual male and female Saltmarsh Topminnow (n = 462) by length deciles possessing an anal sheath during 2017-18.

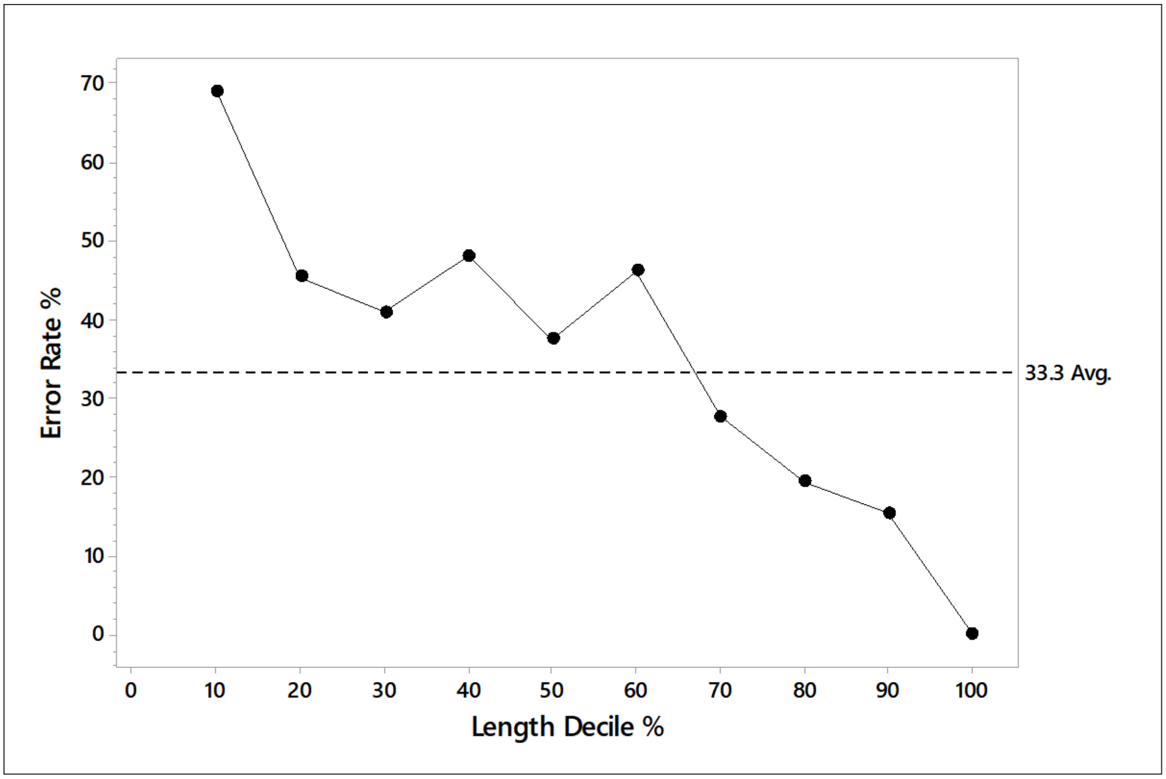


Figure 55. Error rate of sex classification of female Saltmarsh Topminnow based on the absence of the anal sheath versus length deciles. The overall average error rate for all female specimens is also depicted.

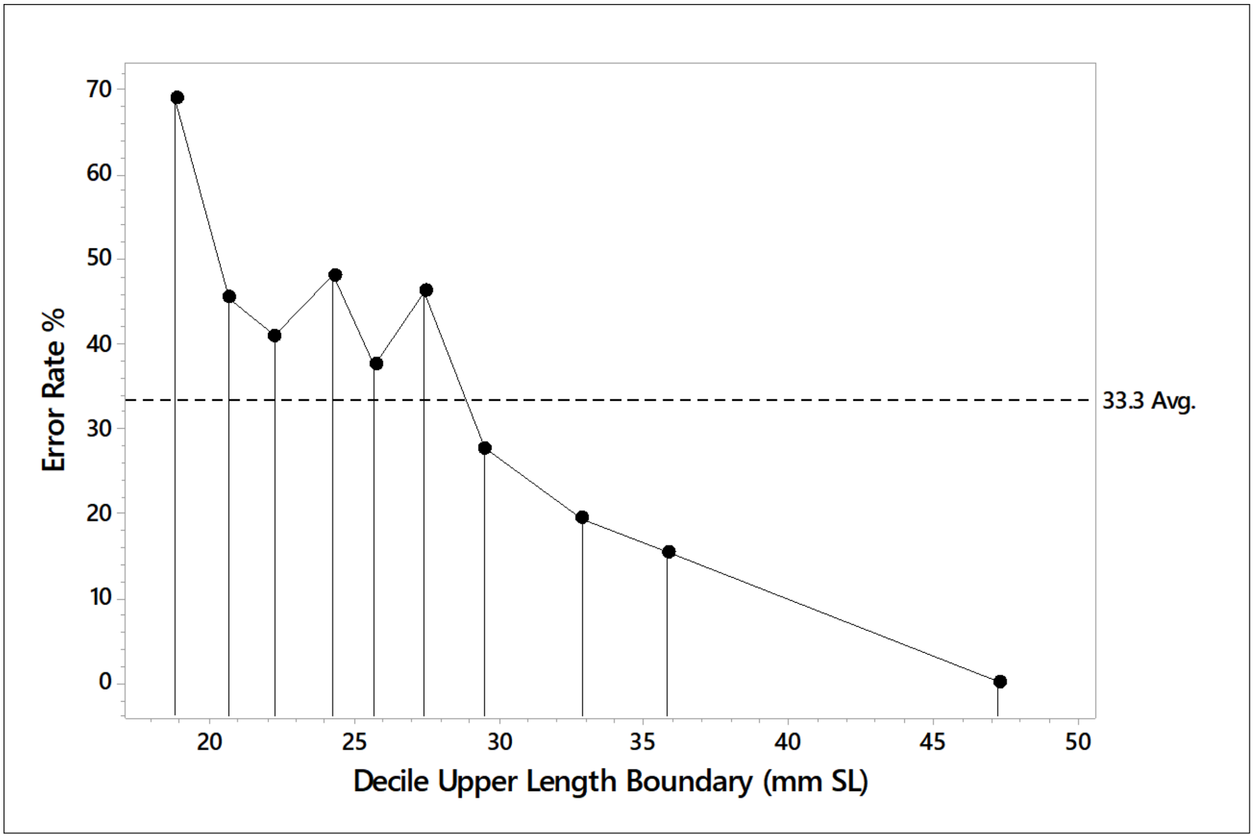


Figure 56. Error rate of sex classification of female Saltmarsh Topminnow based on the absence of the anal sheath versus upper length boundaries of length deciles. The overall average error rate for all female specimens is also depicted.

Post-Hurricane Recovery Special Study

Hurricane Harvey made landfall on August 25, 2017 in Rockport, Texas and then slowly moved over southeast Texas where torrential rains fell within the Galveston Bay watershed. Water levels at the two study sites peaked on August 29, 2017 (Figure 57). Immediately following the Hurricane flood event (September 11, 2017, 17 days after landfall), salinities were very low at sites 21 (1.16 ‰) and 113 (0.47 ‰) (Figure 58). Salinity at Site 21 had recovered to levels within a standard deviation (σ) of the average salinity recorded at the site from all pre-disturbance historical data approximately 64 days after the Hurricane (Figure 58). While no historic data are available to compare Site 113 to, nekton community structure and dominant marsh plants suggest that the average salinity at the site prior to the hurricane was likely lower but comparable to Site 21. Both sites experienced increases in water level during the disturbance with complete inundation of saltmarsh habitat however there was no significant physical damage observed (from scouring) to the saltmarsh plant structure after the disturbance event at either of the two study sites.

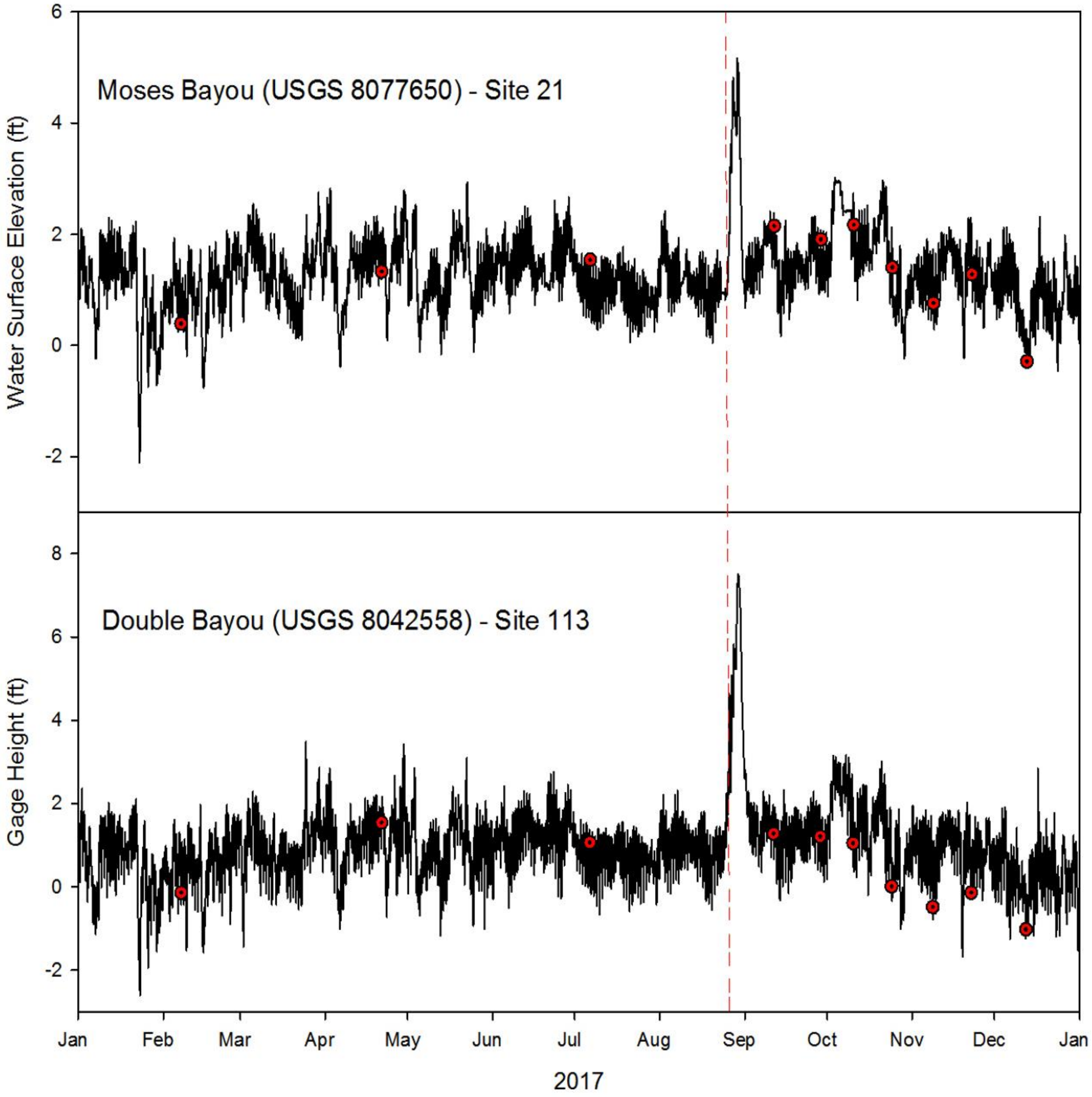


Figure 57. USGS water level data for 2017, illustrating the flood event following Hurricane Harvey's Landfall on August 25, 2017 (red dashed line) with saltmarsh nekton community sampling events plotted as red circles.

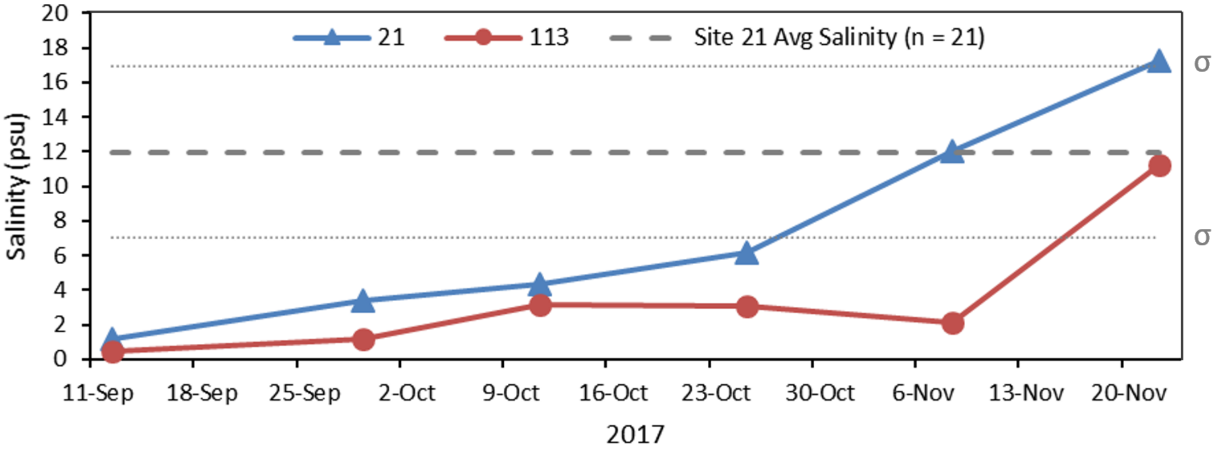


Figure 58. Salinity (‰ or psu) recorded at each sampling event, (Site 21 blue triangles, Site 113 red circles) and the average salinity (dashed line) from all pre-disturbance and historic sampling events at site 21 with standard deviation intervals (dotted lines marked with σ) Excel 2016

A total of 61,134 individuals representing 44 different nekton species were captured during the study. Total community abundance and taxa richness declined immediately following the flood disturbance (Figure 59a). Community diversity and evenness increased immediately following the disturbance (Figure 59b). Decreased total community abundance during the post-disturbance period was primarily driven by the decline in daggerblade grass shrimp (*Palaemonetes pugio*). At site 21, the fish community differences observed immediately after the disturbance were primarily driven by the increased abundance of Bay Anchovy, *Anchoa mitchilli*. Other species that had occurred with regularity at site 21 that were not captured immediately following the disturbance were Pinfish, *Lagodon rhomboides* and the Gulf Pipefish, *Syngnathus scovelli*. Site 113 experienced a reduction in some of the historically observed species immediately following the disturbance, including the Atlantic Croaker *Micropogonias undulatus,* Naked Goby *Gobiosoma bosc,* and the Striped Mullet *Mugil cephalus*.

Standard length was compared pre- versus post-disturbance for the most abundant species (*P. pugio*) and the two most abundant fish species observed at each site (inland silverside, *Menidia beryllina* and *A. mitchilli*). No significant difference in *P. pugio* and *M. berrylina* lengths were observed between sites within each disturbance time period. Consequently, all length data for each of these species were pooled for both sites to evaluate differences in lengths between pre and post disturbance periods. In contrast, site 21 had significantly larger *A. mitchilli* than site 113 ($p = < 0.001$) so each site's data were analyzed separately. Specimens of *P. pugio* were significantly larger pre-disturbance compared with post-disturbance ($p = < 0.001$). Similarly, *M. beryllina* specimen lengths were significantly larger pre versus post-disturbance ($p = 0.0437$). *A. mitchilli* lengths showed no significant difference between pre and post-disturbance periods for either site (Site 21 $p = 0.204$, Site 113 $p = 0.285$).

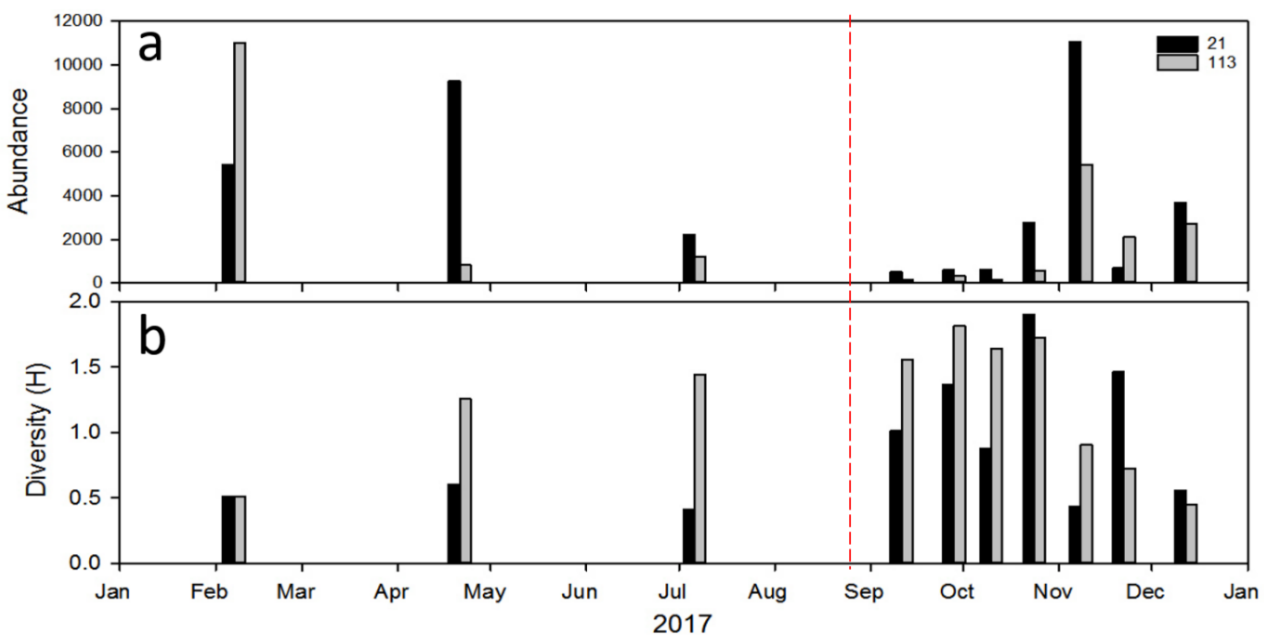


Figure 59 a) Abundance and b) Shannon Weiner diversity of seine catch by sampling event at sites 21 (black bars) and 113 (grey bars). The red dashed line denotes the landfall date of Hurricane Harvey.

During our study, we observed seasonal shifts in community structure similar to historic (2014-15) data collected from site 21 (Guillen et al. 2015). However, the seasonal shift in species composition from the post-disturbance period deviated from patterns observed prior to Hurricane Harvey which depict seasonal trends along the second ordination (MDS2) from the bottom right corner of the figure to the top left (see historic data-blue triangles, Figure 60). The community samples collected post-disturbance (see green squares, Figure 60 were instead dispersed on the opposite MDS1 ordination, falling out on the bottom left of the figure. The species with the highest absolute correlation with the MDS1 ordination, which depicts the shift in community composition from historic conditions to post-disturbance time periods were Spot *Leiostomus xanthurus (r = 0.876), M. cephalus (r = 0.800), A. mitchilli (r = -0.841), and M. beryllina (r = -0.742*) and are displayed as vector overlays on (Figure 60). Historic data collected by (Guillen et al. 2015) did not include invertebrate species, therefore *P. pugio* and other invertebrate species that also likely show high correlation with the MDS1 ordination axis for Figure 60 are not depicted.

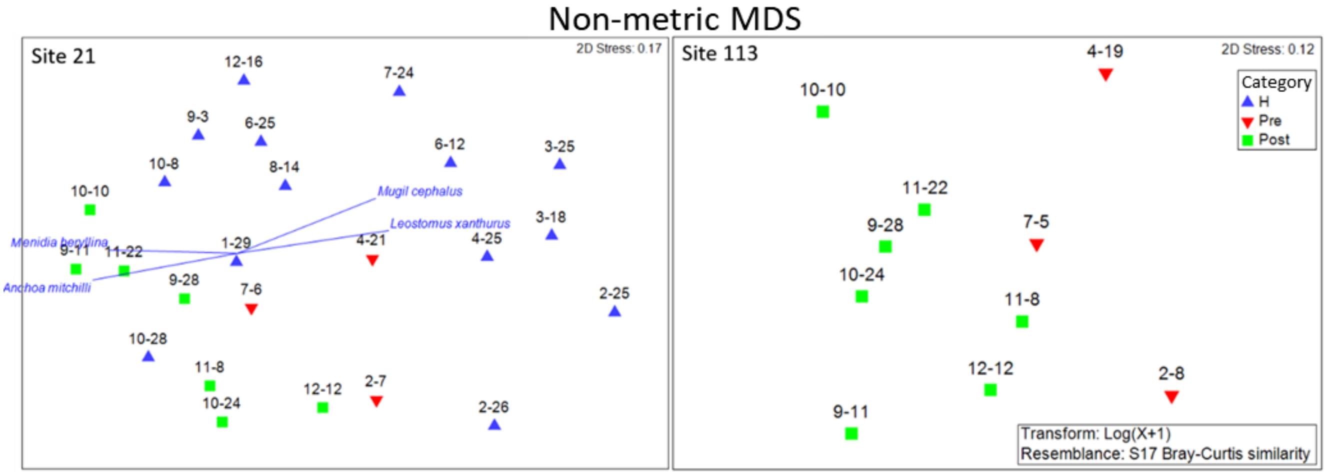


Figure 60 nMDS plots of nekton communities by site. Events labeled by month-day sampled, historic data "H" (Guillen et al. 2015) as blue triangle, 2017 data prior to flood "Pre" (red upside-down triangles) and post-flood data "Post" (green squares). Ordinations on Site 21 for 4 species with highest correlation to MDS1.

Discussion

Distribution and Abundance

Prior to the current extended study (2014-2018) there was little organized data collection on the distribution and density of *F. jenkinsi* in Texas. Limited data indicated that *F. jenkinsi* occurred sporadically from Galveston Bay, Texas to Escambia Bay, Florida (Thompson 1999). Past collections of this species in Texas extend back to 1891 when the original type specimens were collected within Dickinson Bayou in Galveston Bay (Evermann 1892). Periodic collections of small numbers of *F. jenkinsi* had been reported from east Texas estuaries including Galveston Bay and Sabine Lake during 1932, 1940, 1945, 1951, 1953, 1969, and 1996 (Patrick et al. 1998; Simpson and Gunter 1956; Jordan and Evernmann 1896; Guillen 1996 and Martin et al. 2012). Early reports of Saltmarsh Topminnow from the Rio Grande River delta by Fowler (1945) have been subsequently discounted (Simpson and Gunter 1956; NatureServe Explorer 2014; Miller 1955; Thompson 1999). During 1998 Akin et al. (2002) collected *F. jenkinsi* in Mad Island marsh in Matagorda Bay. Additional sporadic collections of this *F. jenkinsi* were made in Cedar Lakes Creek located west of the Brazos River and within Oyster Bayou in Galveston Bay (Guillen 1996)[2]. Sneck-Fahrer and East (2007) captured F. jenkinsi in Mustang Bayou near Alvin, Texas in the Galveston Bay system. During 2003-2004, *F. jenkinsi* were collected in Garcitas Creek, Tres Palacios River, and West Carancahua Creek in the Matagorda Bay system using standard 60 ft. TPWD bag seines (Tolan and Nelson 2009; Tolan et al. 2007). Saltmarsh Topminnow were also captured in the Mission and Aransas Rivers during 2008 and 2009 using 60 ft. TPWD bag seines (Tolan et al. 2011). During 1999-2000 *F. jenkinsi* were captured in Oso Creek in the Corpus Christi Bay estuary using benthic nekton sled

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[2] *F. jenkinsi* data collected by Guillen (1996) but not included in report. Detailed data available from senior author of this report.

(Nicolau 2001). Currently the most western reported record of *F. jenkinsi* is from Oso Creek at FM 763, where 9 specimens were captured in 1999-2000 (Nicolau 2001). Although the exact date of collection(s) is/are not recorded the reported salinity ranged from 0.9 to 4.1 ‰ and averaged 2.6 ‰. There are no reported occurrences of *F. jenkinsi* from the Gulf coast of Mexico (Castro-Aguirre et al. 1999).

The TPWD coastal fisheries monitoring program has not detected *F. jenkinsi* over the last 43 years (April 1976 to August 2018) in spite of extensive (71,135 coast-wide bag-seine) sampling effort. The sampling design utilizes a large bag seine randomly deployed within each major bay system. Although not presented, the spatial and temporal intensity of the sample coverage is sufficient to visibly outline most of the inner estuary coastline when depicted on a map. We hypothesize that the inability to capture *F. jenkinsi* by TPWD standardized sampling programs is due to 1) inability to deploy bag seine gear in small intertidal creeks and marshes, 2) sampling that targets mostly open shorelines in secondary and primary bays, and 3) inherent bias in the sampling method, which utilizes relatively large mesh (1.3 cm stretched mesh in the bag and 1.9 cm stretched mesh on the wings) (Martinez-Andrade 2015). Analysis of the head dimensions during this study illustrated that most of the *F. jenkinsi* were small enough to swim through the bag seine mesh. However, recent studies conducted by TPWD in tidal streams using their bag seine have been successful in capturing *F. jenkinsi* (Tolan et al. 2007; Tolan and Nelson 2009; and Tolan et al. 2011). These creeks would not normally be sampled by TPWD during routine long-term monitoring. We conclude that the primary mechanism contributing to the zero catch rates is likely the sampling bias associated with current stratified sampling, which is biased again selecting small oligohaline tidal creeks and wetlands versus escapement through the large mesh. Additional studies are needed to determine the selectivity of the current sampling gear and how it affects CPUE and estimates of *F. jenkinsi* densities.

Examination of data collected during this study (2014-2018) documented the existence of a gradient of in *F. jenkinsi* density between and within estuaries. This conclusion is based on several lines of evidence. First, a greater percentage of the sites sampled that possessed *F. jenkinsi* were located in eastern Texas estuaries including Sabine Lake, Chambers and Jefferson County coastal wetlands, Galveston Bay and Matagorda Bay. The occurrence of *F. jenkinsi* in these estuaries greatly extends the westward range of *F. jenkinsi* in Texas in comparison to historical accounts that suggested that Galveston Bay was the western boundary (NOAA 2009). Although *F. jenkinsi* were captured in other estuaries the frequency of occurrence and density declined west of Matagorda Bay. During the study we failed to capture any *F. jenkinsi* in Aransas Bay tributaries in spite of successful past collections in 2008-2009 by Tolan et al. (2011). Furthermore, Nicolau (2001) collected *F. jenkinsi* even further west in Oso Creek, a tributary of Oso Bay and Corpus Christi Bay. In addition to these coastal bend estuaries, we failed to capture *F. jenkinsi* within the San Bernard-Brazos and East Matagorda Bay systems. This disjointed distribution between estuaries may be due to a lack of suitable tidal wetland creek habitat within these systems or insufficient spatial and temporal sampling effort.

A gradient of decreasing *F. jenkinsi* frequency and density downstream within individual estuaries was observed during the current study. The highest densities of F. jenkinsi were generally found upstream in less saline wetlands within Sabine Lake, Galveston Bay, Matagorda Bay, and San Antonio Bay. Although the current study failed to capture *F. jenkinsi* in former upstream locations within the Aransas Bay estuary in the Mission and Aransas Rivers, these areas did yield specimens during the survey conducted by Tolan et al. (2011). These upstream tidal creeks and wetlands typically exhibit comparatively lower salinities within their respective estuary.

Data from this study confirms, contrary to historical accounts, that Saltmarsh Topminnow are found in fragmented patchy populations within multiple estuaries and watersheds along the Texas coast west of Galveston Bay. The infrequent sightings of *F. jenkinsi* prior to 1996 most likely reflects a lack of efficient monitoring effort in suitable habitat (e.g. vegetation, salinity) using appropriate effective sampling gear, rather than major fluctuations in populations of this species. Both the lack of an efficient monitoring design and biased sampling gear lowers the detection probability of an organism. Sampling in small tidal creeks and associated wetlands is challenging and frequently not included in any routine monitoring program conducted by state fisheries agencies. These agencies have until recently focused their limited resources on monitoring game fish in open bay habitat, larger tributary rivers, and inland reservoirs. Nongame “bait” fish populations and ecological monitoring of nekton communities has been limited across all Gulf States and is usually not executed using a long-term temporal sampling design.

Accuracy in estimating presence of a threatened species is important for conservation and determination of geographic extent and population viability. However false absences bias many monitoring programs (Wedderburn 2018). Many factors influence detection probability, including the size of an organism and abundance, habitat characteristics and sampling method. The Saltmarsh Topminnow is one of the smallest topminnows inhabiting coastal wetlands along the Gulf coast (Hoese and Moore 1998). Imperfect detection can be addressed by accounting for probability of detection when estimating occupancy by modelling detection/non-detection data collected in replicate surveys. This can be improved by using contrasting sampling approaches. Wedderburn (2018) found that large variations in estimated probability of detection of wetland fishes can occur between sampling devices. He concluded that multi-species wetland habitat monitoring programs using a single sampling device may wrongly estimate the occupancy of a target fish. During the current study we used two sampling methods that have been used by past studies of this species (Peterson et al. 2003) to increase the detection probability for this species. Comparison of methods showed that similar geographic patterns in occurrence were detected with both gear types. While the seine gear type caught the highest number of *F. jenkinsi* and was the gear type that detected the presence most often, there were nearly 20 % of the sites where F. jenkinsi was caught in the trap gear type alone. Furthermore, additional sampling conducted since the conclusion of this study using 4.12m wide, 1/16 inch mesh, fyke net, have detected F. jenkinsi in areas that our previous efforts were unsuccessful. It is recommended that multiple gear types are used to better describe the abundance and distribution of a sporadically distributed species such as *F. jenkinsi*.

During the current study tidal regime had a major influence on CPUE. Lowest minimum and median water levels generally occurred during the winter months when strong cold fronts would reduce water levels across the Gulf coast. During the winter, median water levels were lower than other seasons and months. At the same time median seasonal seine CPUE was highest during the winter. The highest average seine CPUE occurred during the months of November, December, January and February. One of the highest individual catch rates of *F. jenkinsi* occurred during monthly sampling in the winter

(February 2015) when 293 specimens were captured during one seine replicate. Extremely low water levels at the time of sampling had forced fish and other marsh nekton into a single disconnected pool. Conversely, most of the collections containing zero or lowest catches of Saltmarsh Topminnow occurred at sites sampled during the summer when water levels were highest, which allows small nekton to seek refuge in inundated marsh vegetation therefore evading our standard sampling gear. This effectively results in a high false negative (zero catch; absence) rate since even though the target species may be present; it is not vulnerable to our sampling gear. It is therefore advisable to revisit sites during winter months to collect additional samples and increase the likelihood of capturing *F. jenkinsi* if present. In addition, the use of throw traps or high marsh net pens might aid in capturing organisms during high water events.

Fish Assemblages

Analysis of the fish assemblages caught over the course of this study show that *F. jenkinsi* have a strong association with certain fish species. In this study *F. jenkinsi* captured in seines were frequently found in association with at least one other species from the family Fundulidae and often in combination with other marsh edge estuarine fish species (e.g. *P. latipinna*, *G. affinis*, and *C. variegatus*). It is not surprising considering that these species share very similar habitat niches (Baltz et al. 1993; Peterson & Ross 1991). Similarly, the assemblages *F. jenkinsi* did not usually occur in were dominated by more open water species (*A. mitchilli*, *B. patronus* and *M. beryllina*). This pattern of community association indicates that open water habitat favored by those species is not selected for by *F. jenkinsi*.

Significantly higher fish species diversity were associated with the presence of *F. jenkinsi* suggesting that *F. jenkinsi* may prefer saltmarsh habitat that is also able to support a highly diverse array of fish species. Low fish diversity is therefore associated with a lower likelihood of finding *F. jenkinsi* in coastal saltmarshes. Based on the results of our MDS analysis it appears that seasonality is a major factor associated with fish community composition. However, *F. jenkinsi* is considered to be a year round resident marsh species (Neill & Turner 1987) and therefore the differences in abundance and occurrence of this species is likely due to other underlying factors. Based on the data collected during this study we conclude that one of the most likely factors is water level, which directly affects our ability to efficiently capture this species. While, statistically, there was no strong evidence in this study to show that tide levels influence fish assemblage composition or *F. jenkinsi* presence this does not rule out water level as an influencing factor in *F. jenkinsi* capture. Tidal stages were not segregated by season and that is mostly likely why no significant difference was discovered. Apparent tidal levels vary greatly by season (Turner 1991) and while the Gulf coast may not experience a large lunar induced tidal fluctuation as some parts of country they can exhibit large fluctuations due to meteorological forcing (Ward 1980) which would influence the amount of saltmarsh inundation, and thus ability to accurately capture *F. jenkinsi*. Extreme high water levels usually occur during warmer months when winds are from the southeast. In contrast, extreme low water levels are more commonly encountered during the colder months of the year when cold fronts with strong northerly winds cause water levels to drop precipitously along Gulf coast estuaries. During these periods, water levels recede to deeper tidal creeks and expose most of the marsh surface. Fish would be much more vulnerable to capture during these periods.

Due to varying water levels, the ability to accurately capture *F. jenkinsi* depends not only on the extent of marsh inundation but also the type of gear used. During our study we captured *F. jenkinsi* using both seines and Breder traps. The traps were better at catching the target assemblage of fish during high water levels when the marsh was inundated while seining was better for sites during lower levels of inundation. During high inundation periods seining failed to capture the fish species that are closely associated with habitat edges and tended to capture more open water species. In contrast, the traps are specifically designed to catch edge species, like *F. jenkinsi*, that reside in inundated marsh vegetation. During this study the use of multiple gear types (traps and seine) increased the likelihood of capturing *F. jenkinsi* and should be incorporated in all future sampling efforts focusing on cryptic wetland species.

Habitat

Our study supports previous literature, which states (Peterson & Ross 1991; Lopez et al. 2010; and Griffith 1974) that *F. jenkinsi* seem to prefer lower to mid salinity ranges. During our study *F. jenkinsi* were collected within similar salinity ranges (0.12 -19.2 ‰). The majority (75 cumulative percentile) of *F. jenkinsi* were captured when salinity was less than 12.5 ‰. The difference in distribution of F. jenkinsi across salinity gradients between bays is most likely due to the degree of freshwater inflow and resulting salinity in each system. Estuaries in the eastern portion of the state (Sabine Lake to Matagorda Bay) have a higher average freshwater inflow and lower average salinity compared to those in the western (San Antonio Bay, Aransas Bay, Corpus Christi Bay and Laguna Madre) estuaries (Orlando et al 1993). Data collected during this study and literature suggests that while *F. jenkinsi* is able to inhabit a wide salinity range (0-19 ‰) it is equally important to have appropriate marsh habitat available along the existing salinity gradients. For example, Sabine Lake contained numerous sites where *F. jenkinsi* was collected because of favorable salinity regimes and extensive wetlands found from the mouth of the estuary upstream to the upper tributary river mouths.

Mean water temperatures differed significantly at sites where *F. jenkinsi* were collected versus not collected. Higher numbers of *F. jenkinsi* were generally collected in traps and seines at lower temperatures (median 18.5-20.1 ℃) versus higher temperatures (median 24.0- 38.4 ℃). This pattern reflects the seasonal influence of lower water levels in the winter associated with a higher frequency of positive catches of *F. jenkinsi*. As noted earlier, low water levels resulted in easier and more efficient seine collections of the target species. However, the association of higher catches and occurrence with lower temperatures may also be due to the interaction and influence of water temperature with *F. jenkinsi* physiology. Cold weather typically slows down the metabolism of poikilothermic temperate estuarine species (Clarke & Johnson 1999) and the higher catch rate could be exacerbated as a result of the decrease in swimming activity and gear avoidance caused by the lower seasonal temperatures.

The Saltmarsh Topminnow was found in marshes of various dominant plant species. However, *F. jenkinsi* was significantly more likely to be caught at sites where *S. alterniflora* was the dominant vegetation. Previous studies have also stressed the linkage of *F. jenkinsi* presence and *S. alterniflora* (Peterson & Turner 1994). Our data suggests that they are also found in association with other marsh vegetation including oligohaline and mesohaline species and not strictly *S. alterniflora*. Past studies have found that several estuarine fish species selectively use microhabitats characterized by varying degrees of *S. spartina* stem density (Baltz et al. 1993) and that higher abundances of *F. jenkinsi* were captured in low to moderate stem densities (<25 stems/ $0.25\mathrm{m}^{-2}$) (Lopez et al. 2011). As *F. jenkinsi* are a species that is heavily associated with inundated marsh habitats our higher catches in low vegetative cover is most likely due to the fact that low vegetation caused less interference with sampling equipment.

Life History

The overall length distribution for *F. jenkinsi* exhibited two modal peaks, one at the 20-24 mm (SL) and the second at the 32-34 mm during the winter, spring and fall. The lack of discernable peaks in the summer months was due to low catch rates previously discussed. These peaks align with the values produced from the limited analysis conducted using the NORSEP algorithm in FiSAT II by Robertson (2016). The average length of *F. jenkinsi* did not vary by season but they did follow a similar pattern of distribution with peaks appearing at around the 22 and 35 mm SL mark for both fall and winter seasons. The length frequencies of the males and females used for GSI analysis also reflected this trend and possessed peaks at standard length values of approximately 22 mm and 36 mm SL. This data provides further support and evidence that the two modes of standard length values represents at least two separate age classes. This data also reinforces the conclusion that few specimens live longer than a year.

There exists very limited data on the growth rates of this species and estimating an individual's age is difficult to do. This is in part due to the difficulty in aging individuals and defining class groups. Since *F. jenkinsi* are considered batch spawners (Lopez, Peterson, Lang, & Charbonnet, 2010) and spawn over a relatively long season (Lang et al. 2012) the sizes of age classes have the potential to vary greatly as individuals in the same class can hatch months apart from each other. Future analysis of growth utilizing methods such as mark and recapture would provide additional data and a method of validating age and growth estimates for this species. Such methods would also have the benefit of allowing the tracking of movement of *F. jenkinsi* individuals and potential residency time in specific habitat types.

Past studies have documented that *F. jenkinsi* are most likely batch spawners, whose spawning season extends from March through August (Lopez et al. 2011; Lang et al. 2012). The GSI analysis conducted during this study supports these estimates. We found that there is a significant rise in the GSI values for both male and females during the spring and early summer months. Due to the low catch of individuals captured during the summer there is a sizable gap for GSI values from the months of May through September. Only a few individuals were caught during the months of May, June and July. However, the few individuals collected during those months and August and September indicate GSI levels were declining. The limited data we compiled documents a sharp and steady rise in male and female GSI values from January to April. There was a dip in the female GSI values during the months of July through January. Data from the current study agrees with and supports previously documented patterns and assumptions regarding *F. jenkinsi* reproduction.

Both GSI value and standard length showed a significant linear relationship with the ovary phase of female *F. jenkinsi*. A greater degree of ovary maturation is expected to be seen more frequently in the older (larger SL) individuals. We would also expect that ovary phase and GSI values to be directly related as the size of the ovary and its maturation are dependent on each other. Along with the GSI analysis, the data gathered from the ovarian phase analysis in this study supports past assertions (Lopez et al.

2010; Lopez et al. 2011; Lang et al. 2012) that there is a seasonal trend in *F. jenkinsi* reproductive organ growth and development. Ovarian development showed signs of seasonal progression with greater percentages of more developed ovaries being found in spring and summer and more latent and early developing ovaries being found in fall and winter. Further analysis of oocyte stage and development during spring and summer spawning months is warranted. If additional research is conducted, the relationship of tidal stages and gonad development should be evaluated (Lang et al. 2012), to more accurately map the specific spawning dates of this species.

A preliminary estimate of the von Bertanlanfy growth model and instantaneous natural mortality lays the groundwork for future population studies and modeling of this species. The parameter estimates of the von Bertanlanffy equation are similar to other studies of members of the Family Fundulidae (Vastano 2016). Estimates of instantaneous natural mortality M = 2.61 (92.6% annual mortality) indicate few *F. jenkinsi* live beyond a year. This high mortality rate is offset by a life history that incorporates batch spawning over several months (Lang et al. 2012). Currently there is no data available on the annual potential fecundity of this species.

We examined multiple external morphometrics traits using a combination of correlation, logistic regression and chi-square tests to evaluate the ability of these traits in determining the sex of *F.jenkinsi*. We found that the presence of an anal sheath was useful in differentiating the sex of *F.jenkinsi*. However, we also found that length was a significant co-factor that influenced the accuracy of sex classification of specimens. We concluded that the presence of an anal sheath in any specimen larger than 18.8 mm SL would indicate that the specimen is not a male. However, the converse is not true since the false negative rate (females assigned as males) based on lack of a sheath was relatively high, especially in smaller specimens. Based on these observations we would discourage the use of the presence of the anal sheath alone for determining the sex of smaller specimens (< 29.5 mm SL) *F.jenkinsi*.

Post-Hurricane Recovery Special Study

Overall, total saltmarsh nekton abundance observed during sampling events immediately following Hurricane Harvey disturbance was lower than before, while community diversity increased following the disturbance. These differences can be largely attributed to a reduced catch in the numerically dominant *P. pugio* following the event, leading to higher diversity and evenness, despite a reduced number of taxa (Magurran 2004). The most numerically dominant species (*P. Pugi*o) and the most abundant fish species (*M. berrylina*) at site 21 experienced a reduction in size following the disturbance. This suggests that re-population of the saltmarshes following the disturbance was more influenced by new recruits (juveniles) than the return of displaced adults (Reese et al. 2008).

Due to the large geographic scale and long duration of the event, it is highly unlikely that any saline water refugia for marine species remained near the study areas, while freshwater species were most likely displaced downstream into estuarine saltmarsh habitats. The occurrence of primarily freshwater and oligohaline species such as *C. carpio, L. oculatus*, *C. boleosoma*, and, *D. petenense*, captured post-disturbance indicate a large-scale displacement of species from upstream or adjacent drainages into atypical habitats. Furthermore, when the marsh was inundated with deeper water, the efficiency of sampling with seines was reduced. Many habitat-associated species could retreat to the inundated marsh, which could have shifted the bias of the gear to open-water species such as *A. mitchelli* and *M. beryllina*.

While this study illustrated a dramatic reduction in *P. pugio* abundance immediately following a large freshwater flooding event, Piazza and Peyre (2009) observed a dramatic increase in eastern grass shrimp *Palaemonetes paludosus* abundance at a tidal marsh community immediately following Hurricane Katrina, which caused a significant tidal storm surge in their study area. Large disturbance events caused by hurricanes, including tidal surge and wind damage, and overland flooding caused by excessive rain have been shown to impact saltmarsh nekton communities inversely, but in both cases fluctuations in *Palaemonetes spp.* have been major drivers of these community changes (Piazza and Peyre 2009). Although *P. pugio* are present in tidal saltmarshes in Galveston Bay year-round, peak abundance usually occurs in late-summer, when juvenile *Litopenaeus setiferus* abundance is low due to seasonal spawning cycles (Kneib 1995). The timing of the disturbance event may have caused an exaggerated impact to the *P. pugio* population, resulting in a reduced optimal growth and spawning potential.

Natural seasonal cycles in species composition make it difficult to discern cause and effect of disturbance to saltmarsh nekton community structure (Akin et al. 2003). Temporal variation in assemblage structure is often driven by large-scale migrations and seasonal spawning patterns of many species that use the estuary as nursery habitat (Hall et al. 2016). It is possible that this large-scale flood event caused high outflow velocities at tidal passes, and very low, prolonged salinities in Galveston Bay that could have disrupted larval recruitment to Galveston Bay. Continued monitoring of estuary nekton communities is recommended to identify any long-term impacts to the 2017 year-cohorts that were associated with this flood disturbance. Longer-term surveillance may be especially important for species known to recruit in late-summer/early-fall such *Sciaenops ocellatus, Callinectes sapidus,* and shrimp, *Peneaid sp.* (Holt et al. 1983; Pile et al. 1996; Reese et al. 2008).

Post-disturbance sampling concluded before the community structure showed signs of complete recovery and return to seasonal composition comparable to historic data. However, estuarine environments have been shown to be highly resilient to short-term, natural disturbance events (Paperno et al. 2004; Waide 1991). In similar systems that have experienced significant habitat damage following a disturbance event, the recovery period extended up to 18 months (Piazza et al. 2009). Because neither of the study sites sustained significant damage to structural habitat from Hurricane Harvey, it is likely that the nekton community structure continued to recover as background levels of water quality and salinity returned and uninterrupted seasonal recruitment of immature organisms continued.

Future Research and Management

Recent historical (Nicolau 2001; Tolan et al. 2011) records indicate that viable local populations of *F. jenkinsi* may be expected to be found further west extending to Corpus Christi Bay. For now the best estimate of the western extent of *F. jenkinsi* is Oso Creek within the Corpus Christi estuary. Further surveys are critically needed in these areas and the upper Laguna Madre to evaluate the long term viability of these populations and to define the western extent of the species. In addition, resurveys of many of the index sites visited during this study should be conducted at a minimum in the winter when the species is most vulnerable to sampling gear during low tides. Initialization of mark-recapture studies would also help establish a more reliable method to determine and validate age and growth estimates for this species. Such methods would also facilitate tracking the movement of *F. jenkinsi* individuals and give better estimates of how much inter-marsh migration occurs and the fidelity to certain habitat types.

Focus areas of future research should include better documentation of oocyte development and fecundity estimates during the spawning season, and the examination of the influence of high tide stages on gonad development and egg production (Lang et al. 2012). Better documentation of these processes would lead to more accurate predictions of spawning times and estimation of offspring production. Further diet and trophic ecology studies of *F. jenkinsi* are needed along with supporting studies on prey availability and abundance would help define any potential limiting factors associated with diet. Such information would help researchers and managers understand the entire spectrum of habitat and biological needs of this species.

Data from this study provides critical information that USFWS can use to evaluate the need for further research and/or the listing of this species for protection under the ESA. Based on the results of our study, this species is likely more extensive than previously thought and has likely been underestimated due to the selectivity of existing agency fisheries monitoring programs which do not cover the preferred habitat of this species. Based on our findings and previous literature there are many management actions that can be taken currently to promote the conservation of this species. Currently, many of Florida’s and Mississippi’s management plans concerning the habitat of *F. jenkinsi* can be incorporated into future conservation plans or mitigation within the state of Texas. For example, restored or newly created wetlands should include a network of small interconnecting tidal creeks to maximize edge habitat (Baltz et al. 1993; Peterson et al. 2003; Lopez et al. 2010). These wetlands should also be located in the estuary where there is some degree of fresh water inflow to maintain the preferred salinity range of the Saltmarsh Topminnow (Peterson & Ross 1991; Lopez et al. 2010; and Griffith 1974) as well as other estuarine species. Efforts to prevent large sections of currently established tidal creeks from becoming channelized as well as restoring fish passage should also be goals of future conservation projects, especially in the areas where *F. jenkinsi* have been found to be present.

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