

Ecological Function and Recovery of Biological Communities within Dredged Ridge-Swale Habitats in the South-Atlantic Bight

Volume 3: Final Report on the Fishes and Ecosystems of the Sand Shoals



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March 2024

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Prepared under Cooperative Ecological Studies Unit Agreement
M13AC00012

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U.S. Department of the Interior
Bureau of Ocean Energy Management
Sterling, VA



DISCLAIMER

Study collaboration and funding were provided by the U.S. Department of the Interior, Bureau of Ocean Energy Management (BOEM), Environmental Studies Program, Washington, DC, under Agreement Number M13AC00012. This report has been technically reviewed by BOEM, and it has been approved for publication. The views and conclusions contained in this document are those of the authors and should not be interpreted as representing the opinions or policies of BOEM, nor does mention of trade names or commercial products constitute endorsement or recommendation for use.

REPORT AVAILABILITY

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CITATION

Murie D, Bucatari J, Hansen D, editors. 2024. Ecological function and recovery of biological communities within dredged ridge-swale habitats in the South-Atlantic Bight. Volume 3: final report on the fishes and ecosystems of the sand shoals. Sterling (VA): U.S. Department of the Interior, Bureau of Ocean Energy Management. 201 p. Report No.: OCS Study BOEM 2024-016. Contract No.: M13AC00012.

Chapter Citations:

Murie D, Smith, G, Jr. 2024. Chapter 13: fish assemblages on offshore sand shoals and potential impacts due to dredging events. In: Murie D, Bucatari J, Hansen D, editors. Ecological function and recovery of biological communities within dredged ridge-swale habitats in the South-Atlantic Bight. Volume 3: final report on the fishes and ecosystems of the sand shoals. Sterling (VA): U.S. Department of the Interior, Bureau of Ocean Energy Management. Report No.: OCS Study BOEM 2024-016. Contract No.: M13AC00012. p. 1-77.

Parkyn D, Murie D. 202-43. Chapter 14: fish movement and shoal habitat use. In: Murie D, Bucatari J, Hansen D, editors. Ecological function and recovery of biological communities within dredged ridge-swale habitats in the South-Atlantic Bight. Volume 3: final report on the fishes and ecosystems of the sand shoals. Sterling (VA): U.S. Department of the Interior, Bureau of Ocean Energy Management. Report No.: OCS Study BOEM 2024-016. Contract No.: M13AC00012. p. 78-118.

Murie D, Smith, G, Jr., Croteau A 2024. Chapter 15: isotopic pathways on sand shoals in relation to sand dredging. In: Murie D, Bucatari J, Hansen D, editors. Ecological function and recovery of biological communities within dredged ridge-swale habitats in the South-Atlantic Bight. Volume 3: final report on the fishes and ecosystems of the sand shoals. Sterling (VA): U.S. Department of the Interior, Bureau of Ocean Energy Management. Report No.: OCS Study BOEM 2024-016. Contract No.: M13AC00012. p. 119–150.

Vilas D, Chagaris D, Ahrens R, Murie DJ, Smith Jr. G, Behringer D, Jennings L, Baker P, Frank C, Philips E. 2024. Chapter 16: ecopath modeling of sand dredging events. In: Murie D, Bucatari J,

Hansen D, editors. Ecological function and recovery of biological communities within dredged ridge-swale habitats in the South-Atlantic Bight. Volume 3: final report on the fishes and ecosystems of the sand shoals. Sterling (VA): U.S. Department of the Interior, Bureau of Ocean Energy Management. Report No.: OCS Study BOEM 2024-016. Contract No.: M13AC00012. p. 151–166.

Murie D. 2024. Chapter 17: summary and overview. In: Murie D, Bucatari J, Hansen D, editors. Ecological function and recovery of biological communities within dredged ridge-swale habitats in the South-Atlantic Bight. Volume 3: final report on the fishes and ecosystems of the sand shoals. Sterling (VA): U.S. Department of the Interior, Bureau of Ocean Energy Management. Report No.: OCS Study BOEM 2024-016. Contract No.: M13AC00012. p. 167–176.

ABOUT THE COVER

Atlantic Croaker, Leopard Searobin, Spotted Whiff, and Smooth Butterfly Ray (Murie Lab, University of Florida).

ACKNOWLEDGMENTS

As a research team, we would like to sincerely thank Jim Stringer of the Florida Research and Recovery Group (FRRG), including all the captains and mates of the *Laffin' Place*. The FRRG gave us priority use of the vessel and the flexibility to work schedules around inclement weather and crew availability, greatly facilitating this research. We are also greatly appreciative of Michael Dickson and Geoffrey Smith Jr., the two primary research program biologists during this project; they provided critical planning and organization of all the various research teams involved in this study. As a cooperative agreement, this research would not have been possible without the collaboration of BOEM personnel. We especially thank Jennifer Bucatari for all of her constructive guidance and review of all aspects of the project throughout this long-term monitoring project, including the final report and deliverables. Deena Hansen and Geoff Wikel provided specific guidance on various aspects of the project and we greatly appreciated their attention to detail and constructive feedback in the review process. We also thank BOEM personnel that contributed to the overall review of specific chapters, including Geoff Wikel, Paulina Chen, Lora Turner, Kerby Dobbs, Doug Piatkowski, Paul Knorr, Jake Levenson, and Michael Rasser. Further specific acknowledgments are provided within each chapter.

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List of Abbreviations and Acronyms

2D	two-dimensional
3D	three-dimensional
ACT	Atlantic Cooperative Telemetry
ANCOVA	Analysis of Covariance
BACI	Before-After-Control-Impact
BOEM	Bureau of Ocean Energy Management
CSII	Canaveral Shoal II
CSII-BA	Canaveral Shoal II Borrow Area
DD	days detected
DL	days at liberty
DSCF	Dwass, Steel, Critchlow-Fligner
DW	disk width
EBM	Ecosystem Based Management
EE	ecotrophic efficiency
EFH	Essential Fish Habitat
EwE	Ecopath with Ecosim
FACT	Florida Acoustic Cooperative Telemetry
FCI	Finn's Cycling Index
FG	functional group
FL	Fork Length
FRRG	Florida Research and Recovery Group
GLATOS	Great Lakes Acoustic Telemetry Observing System
IACUC	Institutional Animal Care and Use Committee
IC	Impact-Control
IQR	interquartile range
IRL	Indian River Lagoon
KSC	Kenney Space Center
LOA	Letter of Acknowledgment
LGW	long gravity wave
MDS	Multidimensional Scaling
mMDS	Metric-Multidimensional Scaling
MPL	mean path length
NE	Northeast
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NW	Northwest
NWS	National Weather Service
OCS	Outer Continental Shelf
OTN	Ocean Telemetry Network
PERMANOVA	Permutational Multivariate Analysis of Variance
PPD	Proportion of Posterior Draws

SAFMC	South Atlantic Fisheries Management Council
SD	standard deviation
SE	Southeast
SEA	standard ellipse area
SGW	short gravity wave
SIBER	Stable Isotope Bayesian Ellipses (in R)
SIMPER	similarity percentage
SL	standard length
SW	Southwest
TDR	temperature-depth recorder
TED	turtle excluder device
TL	total length
TST	total system throughput
UF	University of Florida
USGS	US Geological Service

13 Fish Assemblages on Offshore Sand Shoals and Potential Impacts Due to Dredging Events

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Key Points

- Sand shoals off the east coast of Florida have diverse fish assemblages that are dynamic in all aspects of space and time.
- Fish species richness varied among shoals: it was highest on Bull Shoal (150 species), followed by Chester Shoal (134 species), and lastly CSII-BA and CSII (118 and 117 species, respectively).
- Both the Shannon Diversity Index and Simpson's Index of Diversity were moderate for all seasons except most winters, which was associated with low diversity but high abundance of juvenile sciaenids (drums and croakers).
- Spatially, all shoals had fish assemblages that were relatively different in multivariate space, varying in either direction or size based on both abundance and biomass (standardized on a per area basis). This finding indicated that all shoals had some individualistic features and natural variation that could make it difficult to generalize to other shoals in the area that were not sampled.
- Swale habitat on the shoals had significantly greater diversity, abundance, and biomass compared to ridge habitat. In addition, fish assemblages were significantly different between swale and ridge habitats of the shoals.
- Temporally, fish abundance, biomass, and assemblages varied significantly on an annual (year to year), seasonal, and diel basis, with seasonal and diel factors contributing more to the temporal differences than year.
- Seasonal changes in abundance and biomass were highly significant, with fall having the greatest abundance and biomass, followed by winter, summer, and spring.
- Though the fish assemblages were dynamic from season to season, there was a core assemblage of many fish species that clearly associated with the sand shoals year-round and were not transient.
- Abundance and biomass of fishes were 2.5X and 2.8X higher at night compared to day, and fish assemblages were also significantly different between day and night.
- For two dredging events (winter 2013/14 to spring 2014 and then again in spring 2018), there was no significant impact on the fish assemblages of CSII-BA (the dredged shoal), relative to the natural variability observed in the fish assemblages of the reference shoals (CSII, Chester Shoal, and Bull Shoal).

13.1 Introduction

Sand shoals are relatively common features along the eastern U.S. seaboard that can provide vertical structure in a landscape of little to no physical relief. Typically, each shoal has a prominent ridge that is surrounded by lower-lying, relatively flat areas (i.e., swales), referred to as a ridge-swale complexes

(Michel et al. 2013). For fishes, these ridge and swale mesoscale habitats can provide physical structure that potentially enhances foraging, protection from predators, spawning, and potential nursery areas for recruitment (Michel et al. 2013)—all functions that are associated with the definition of Essential Fish Habitat (EFH) (SAFMC 1998). Understanding the variability in the use of sand shoals by fish assemblages on a spatiotemporal basis is necessary for both defining the scope of potential EFH and to monitor and ameliorate any impacts to potential EFH by anthropogenic activities.

One primary use of sand shoals is dredging sand for beach renourishment following major hurricane activity. In October 2012, Hurricane Sandy traveled up the eastern US seaboard, paralleling the coast from Florida to Maine. Although it did not make landfall in Florida, the sheer size (diameter) of the hurricane was historic, and the waves generated, which some estimated to be 20 ft, stripped the sand from coastal shorelines on the east coast of Florida (NWS 2012). To restore the depleted shores, sand was dredged from an Outer Continental Shelf (OCS) site, the Canaveral Shoal borrow area (CSII-BA), which is in federal waters off of Cape Canaveral on the east coast of Florida (Figure 13-1). CSII-BA has been dredged seven times since 2000, with a varying portion of the entire area subject to dredging (Table 1-1). Included in this timeline are the more recent dredging events of 2013/14 and 2018 that occurred in the timeframe of this study.

Although potential dredging impacts on EFH and fish assemblages have been studied previously (see review by Michel et al. 2013), there have been no studies directly assessing potential dredging impacts on fish assemblages of cape-associated shoals and, more specifically, of the Canaveral Shoals off the east coast of Florida. Previous biological studies of shoals off the East Coast of the USA have focused on fish species composition, abundance, diversity, and assemblage differences based on microhabitats present for juveniles on shoals (Diaz et al. 2003a), cross-shelf distances relative to a specific sand ridge (Vasslides and Able 2008), and ridges versus flat-bottomed areas (Slacum et al. 2010). These studies have provided many insights into the importance of shoal habitats to fishes on the East Coast, but they were not able to directly assess the impact to fish assemblages during contemporaneous sand dredging activities.

In assessing the impact of sand dredging on fish assemblages it is also necessary to consider spatial and temporal aspects because abundance and diversity of fish assemblages can vary based on micro- and mesoscale habitat differences, seasons, diurnal periods, and over years. When assessing the impact of dredging on one shoal, it is also important to consider the natural variability in fish assemblages among similar shoals since spatiotemporal changes can potentially occur irrespective of dredging activities. These shoals can be thought of as reference or control shoals relative to the dredged shoal and incorporated into powerful Beyond-BACI (Before-After-Control-Impact with multiple sampling times) analyses (Underwood 1992; Anderson et al. 2008; Smokorowski and Randall 2017), depending on the timeline between the start of sampling and the dredging event.

The current study provided an opportunity to employ a robust Beyond-BACI sampling design (detailed in Chapter 1), which allowed for comparisons of fish assemblages before and after dredging on CSII-BA relative to the shoal portion that was not dredged (referred to as CSII), as well as two other reference shoals, Chester and Bull Shoals; these latter shoals provided comparisons based on the natural biological variability of shoals in the area.

As extensively reviewed by Michel et al. (2013) and Wenger et al. (2017), sand dredging is perceived to have a mostly negative impact on biological communities, including fishes and their habitat. Some of these effects may be felt directly, such as through the removal of small fishes when using suction dredges (Palmer et al. 2008) or through increased turbidity temporarily irritating fish gills (Michel et al. 2013). Redeposition of any suspended sediment also can directly impact fish reproduction by burying or smothering benthic eggs (Michel et al. 2013). Other effects may have an indirect impact on fishes, for example, by removing benthic invertebrates that form an important prey base for fishes in the area or through increased short-term water turbidity potentially resulting in a decreased ability for visual

predatory fishes to forage effectively (Utne-Palm 2002; Michel et al. 2013). Negative impacts can therefore potentially reduce the abundance, biomass, and diversity of fish assemblages. Few positive impacts of dredging have been proposed but include enhanced production of crustaceans in the dredged area that ultimately provide a food resource for fishes (Diaz et al. 2004) and a potential increase in the complexity (i.e., rugosity or increased vertical relief) of the bottom (Michel et al. 2013).

13.1.1 Goals and Objectives

The overall goal of this study was to describe and quantify the fish diversity, abundance, biomass, and assemblages associated with offshore sand shoals on a spatiotemporal basis in relation to sand dredging events. Our specific objectives included:

1. Spatially, to compare fish diversity, abundance, biomass, and assemblages among reference shoals and the dredged shoal, as well as in association with specific shoal habitats (ridge versus swale habitats);
2. Temporally, to compare fish diversity, abundance, biomass, and assemblages among years (annual), among seasons, and between diurnal (day versus night) periods;
3. To determine if sand dredging events result in changes in the fish assemblages associated with sand shoals, including:
 - a) Compare fish assemblages among dredged versus non-dredged (reference) shoals
 - b) Determine if potential impacts to fish assemblages from dredging events can be distinguished from natural annual, seasonal, and diurnal variability in the fish assemblages observed on reference shoals.

13.2 Methods

13.2.1 Study Area and Spatiotemporal Sampling

Fish assemblages were sampled on three offshore shoals: Canaveral Shoal II (CSII), Chester Shoal, and Bull Shoal (Figure 13-1). Each shoal was divided into ridge versus swale habitats, where ridge habitat was represented by the top of shoal and the swale habitat was the relatively flat-bottom area surrounding each shoal (see Section 1.2.1). Each shoal was also divided into quadrants for sampling site selection to ensure representative sampling over the entire shoal in each season. Canaveral Shoal II was further divided by quadrants into those that were dredged for sand (referred to hereafter as CS II-BA, representing the borrow area) and quadrants that were not dredged (referred to hereafter as CSII) (Figure 13-1).

Each shoal and shoal habitat types were sampled on an annual, seasonal, and diurnal basis. Annual sampling occurred for 6 years from November 2013 to July 2019. Seasonal sampling occurred spring from March–May, summer was June–September, fall was October–November, and winter was December to the following February. Sampling delays due to weather and vessel operations offshore sometimes pushed seasonal sampling 2 weeks into the next season, especially with inclement weather in the fall and winter sampling trips; when this occurred, the next seasonal sampling did not occur for at least 2–4 weeks after the previous sampling. Diurnal sampling during the day occurred anytime between an hour after sunrise to an hour before sunset. Night sampling occurred between approximately an hour after sunset to an hour before sunrise (i.e., excluded periods of civil twilight).

For each seasonal and diurnal sampling event, 12 randomly stratified, standardized tows were conducted on each shoal, with 4 and 9 sites in ridge versus swale habitats, respectively, using quadrant strata to ensure representation across each of the shoals for each sampling event (Figure 13-1). The exception to this was CSII and CSII-BA where the NW quadrant was not sampled due to its shallow location and sampling sites were therefore assigned to the swale area of CSII-BA that was to be dredged. Sampling

sites were chosen using an ArcGIS random-stratified sampling model. Daytime trawls were conducted in all seasons unless weather or vessel interruptions occurred. Nocturnal samples were initially collected only in winter and summer seasons to determine if nighttime assemblages were different than daytime assemblages. Since the assemblages and abundance of fishes appeared to be different between the daytime and nighttime trawl samples, based on the preliminary winter/summer sampling, nocturnal samples were taken in all seasons starting in spring 2017.

Water temperature (surface and bottom) (± 0.8 °C) and water depth (± 30 cm) were recorded at each site using a Sensus Ultra Temperature-Depth-Recorder (TDR) (ReefNet, Inc.) attached to the trawl. Some TDRs were detached from the trawl, whereby surface-water temperature and water depth was measured from the vessel's depth sounder.

13.2.2 Sampling Relative to Dredging Events

13.2.2.1 First Dredging Event (Dredge-1)

Originally, this study was initiated to assess the impact of dredging on fish assemblages for a dredging event that was to take place on CSII-BA in November 2013 through April 2014. The research study was implemented in October 2013 and it was therefore possible to only carry out limited sampling prior to the start of the actual dredging. In order to work around the dredging timeline, sampling was started first on CSII-BA in fall 2013 with sampling on 13, 19, and 23 November (8 stations total) just prior to the start of dredging on CSII-BA on the 27 November. Complimentary sampling in CSII (non-dredged shoal) was done on the 23 November (5 stations total) and Bull Shoal, one of the reference shoals, was sampled as soon as possible after CSII and CSII-BA on 7 December 2013 (6 stations total). Chester was not able to be sampled in this timeframe due to inclement weather. To keep the three “pre-dredge” sampling events on CSII, CSII-BA, and Bull uniquely identified from the next season of sampling in winter 2013/14, the samples from Bull were identified internally as also belonging to fall 2013 (although the samples were physically collected on the 7 December). This was deemed acceptable because the sampling was done within 2 weeks of the start of the winter season (December–February), which could regularly happen due to inclement weather, and also because all shoals were sampled for the full winter 2013/14 season starting in late January 2014. This first dredging event (referred to as Dredge-1) was therefore comprised of one pre-dredge season of sampling on CSII-BA, CSII, and Bull Shoals (fall 2013), followed by the main dredging in winter 2013/14 through spring 2014 (dredging ended in April 2014).

13.2.2.2 Second Dredging Event (Dredge-2)

CSII-BA was dredged a second time during this study during winter 2017/18 through spring 2018, referred to as Dredge-2. Although not a component of the original study, this second dredging event allowed for a more extensive suite of pre-dredge samples to be compared to post-dredge samples for all shoals.

13.2.3 Fish Sampling

Fish samples were collected using a semi-balloon bottom (otter) trawl (Figure 12-1) with a 6.1 m (20 ft) headrope, a body of 3.8 cm (1.5 in) stretched mesh, and a cod-end of 2.54 cm (1 in) stretched mesh fitted with an inner liner of 0.3175 mm (1/8 in) knotless netting; trawl doors were 18 in by 36 in. The trawl was also fit with a required 81.3 cm (32 in) trawl-exclusion device (TED) to allow sea turtles to escape the trawl. The trawl was towed at 1.5–2.0 kts for 10 minutes (on-bottom to off-bottom time). Trawling time was permitted to be relatively short to negate any potential endangerment of sea turtles incidentally captured; no sea turtles were captured during the study. Trawls were towed by the *Laffin' Place*, a 29-ft Island Hopper vessel with twin 240-hp Yanmar diesels, owned by the FRRG working out of Port Canaveral, FL.

Additionally, beginning in spring 2014, a set of tows were done using an identical trawl but with the TED closed (under a National Marine Fisheries Service (NMFS) Letter of Acknowledgment (LOA) permit) and without the cod-end liner (referred to as closed-TED trawls). This was done because it was noticeable that the fish sampled using the standardized trawl were relatively small (i.e., majority too small to tag for the acoustic tracking study, see Chapter 14). The TED was closed (and liner removed) to check that larger fishes were not being excluded from the day or night standardized tow samples by exiting through the TED.

On retrieval of each trawl, the contents of the cod-end were emptied into a large tub for sorting. The cod-end liner was opened fully and rinsed down with seawater via a hose and nozzle to recover small organisms caught in the liner. Larger fish (generally ≥ 25 mm total length) were immediately placed in an 18.9 L (5 gal) plastic bucket filled with sea water and lined with a 5-gal bucket paint strainer bag with 250 μ m mesh. Once all larger fish were placed in the bucket, the strainer bag was removed, tied closed, and placed in a 400 mg/L eugenol-seawater solution for euthanasia. Fish remained in the eugenol-seawater solution for 10 minutes after opercular movements had ceased. After euthanasia, the fish in the strainer bag were rinsed thoroughly in seawater and then placed in labeled plastic bags and stored on ice until frozen on return to the lab. After all larger fish were removed, the rinse water in the mixing tub was poured through fine mesh aquarium nets to collect any very small juvenile and larval fish. These were placed in labeled plastic bags and stored on ice until being frozen on return to the lab.

13.2.4 Processing of Fish Samples

Labeled bags with fish samples were thawed in a refrigerator. Fish were then identified to the lowest possible taxon based on a variety of dichotomous keys, including Robins and Ray (1986), Lieske and Myers (2002), Richards (2006a,b), and Kells and Carpenter (2011); as well as various primary literature. Some larval and juvenile fish were difficult to identify to species-level due to lack of identification keys and damage to their bodies during the trawling, wash-down, and sorting. However, the vast majority of larvae and juveniles were identified to the family level.

Each sample was separated by species and any obvious cohorts or approximate size classes of larval/small juvenile, larger juvenile, subadult/small adults, adult/larger adult based on each species' maximum size and life history. The lengths and weights of up to 20 individuals of each size class from each trawl sample were measured and weighed. For fish $> \sim 25$ mm, lengths were recorded to the nearest mm using a fish measuring board and, for smaller fish, lengths were recorded to the nearest 0.01 mm with digital calipers. Fish were measured for maximum total length (TL), fork length (FL), or standard length (SL) as appropriate for the fish morphology. For fish weighing more than 2 g, weights were recorded to the nearest 0.01 g with a benchtop balance; for smaller fish, weights were recorded to the nearest 0.0001 g with an analytical balance. After 20 individuals of each species and size class were measured and weighed, the remainder were counted and a batch weight was recorded. White muscle samples were collected for stable isotope analysis from at least two individuals of each species and size class from each trawl collected through the end of 2016. Due to the high species diversity of fishes and limited time and funds to process the resulting extensive number of isotopes samples, from 2017 onward the collection of isotope samples was limited to a set of targeted species representing different trophic levels (see Chapter 15).

Permits for sampling of fish occurred under a LOA issued by National Oceanic and Atmospheric Administration (NOAA)/NMFS for both the standardized trawling and the trawling with the TED closed. Field sampling and laboratory processing of fish samples occurred under the Institutional Animal Care and Use Committee (IACUC) protocols #201709892 and #2020009892 at the University of Florida.

13.2.5 Data Analyses

13.2.5.1 Overall Species Composition and Diversity

Initially, size distributions of fish from the standardized tows (cod-end liner and open-TED) were compared to the size distributions of fish collected from the closed-TED (with no cod-end liner) tows to test whether larger fish were being selectively excluded by the presence of the TED in the trawl net of the standardized tows.

Overall fish species composition among the shoals, ridge-swale habitats, seasons, and day-night periods was summarized by percent numerical abundance and percent biomass. Diversity of fish species by shoal, ridge-swale habitat, season, and day-night was determined using total number of species i.e., species richness), Simpson's Diversity Index and Shannon Diversity Index estimated using PRIMER v7 (Plymouth Routines in Multivariate Ecological Research, version 7) DIVERSE program (Clarke and Gorley 2015). The complement of Simpson's Diversity Index ($1-\lambda$, sometimes called the Gini-Simpson Index) was chosen because it takes into account both the number of species present and their relative abundance (richness and evenness):

$$1 - \lambda = 1 - (\sum p_i^2) \quad (13-1)$$

where p_i is the proportion of individuals of species i in the community. Equation 13-1 applies to conditions where there are reasonably large sample sizes, otherwise the Simpson's Diversity Index is modified for small sample sizes as $1-\lambda'$, with $\lambda' = [\sum_i N_i(N_i - 1)]/[N(N - 1)]$, where N_i is the number of individuals of i species (Clarke et al. 2014). The index ranges from 0 to 1, with values close to 0 indicating low diversity and values close to 1 indicating high diversity. Simpson's Diversity Index is less sensitive to the occurrence of rare species compared to the Shannon Diversity Index (H'), which is sensitive to the number of species (including very rare species) but also takes into account their abundance (Krebs 1989; Clarke et al. 2014). The Shannon Index is:

$$H' = -\sum_i p_i \log_e(p_i) \quad (13-2)$$

where p_i is the proportion of the total count from the i th species. The Shannon Diversity Index is the most commonly used index and was therefore provided as a comparison; however, it is also sensitive to sampling effort because of species richness and should only be compared across studies with similar sampling effort and design (Clarke et al. 2014). After estimation of the indices, the values were normalized so that they could all be displayed on the same scale (Clarke and Gorley 2015).

13.2.5.2 Species Abundance and Biomass

Abundance of fish species for each tow was standardized to number per 10,000 m² swept area of the trawl because trawl paths differed slightly in length. Similarly, biomass was standardized to g per 10,000 m². Abundance and biomass were standardized to 10,000 m², instead of km² (i.e., 1,000,000 m²) for easier interpretation of data analyses and better visualization of the data. The total length of the trawl path was determined by GPS start (trawl on bottom) and GPS end (trawl off bottom) vessel positions. The swept area of the trawl was calculated as the towed distance multiplied by the trawl headrope size. In the few cases where either the start or end GPS position was inadvertently not immediately recorded or recorded with an obvious error then the trawl path distance was estimated by taking the average of all other trawls towed on the shoal during the same sampling day given that all trawls were explicitly timed.

Abundance and biomass were compared among shoals, between ridge-swale habitats, among seasons, and between day-night. Data were initially checked for non-normality using a Shapiro-Wilk test and for homogeneity of variances using Levene's test (SAS 2012) and found to significantly vary from those

assumptions. Data were therefore log-transformed (i.e., $\log_e(x + 1)$) but were observed to still be non-normal and heteroscedastic. A nonparametric Kruskal-Wallis test (NPAR1WAY, SAS Institute 2012) was therefore applied to the untransformed data and, if significant, was followed by pairwise two-sided multiple comparison analysis using the Dwass, Steel, Critchlow-Fligner (DSCF) method (SAS Institute 2012). All tests were statistical significance at $P \leq 0.05$.

Due to their asymmetric distributions, abundance and biomass were plotted using notched boxplots derived from *geom_boxplot* in R (Krzywinski and Altman 2014), where the waist was the median value (50th percentile), the lower box hinge was the 25th percentile (Q1, or the first quartile), the upper box hinge was the 75th percentile (Q3, or third quartile). The lower whisker represented data within 1.5*IQR of the lower hinge, where IQR was the interquartile range (Q1 to Q3); the upper whisker was data within 1.5*IQR of the upper hinge. Data falling below or above the lower and upper whisker boundaries, respectively, were considered to be outliers. For notched boxplots, the notches represent approximately the 95% confidence limits around the median (McGill et al. 1978).

13.2.5.3 Species Assemblages

Species assemblages based on abundance and biomass were compared on a spatial scale (shoals, ridge-swale) and a temporal scale (years, seasons, day-night) using PRIMER v7 with the added Permutational Multivariate Analysis of Variance (PERMANOVA+) package (Anderson et al. 2008; Anderson 2017). PERMANOVA is a semiparametric method based on a distribution-free permutation technique and therefore has no requirement of multivariate normality (Anderson 2017). Abundance was averaged to the quadrant level for all analyses because quadrant was used to randomly stratify the sampling sites but was not a factor of interest in the study. Species abundance data were first subjected to dispersion weighting because some of the species were highly aggregated schooling species (e.g., anchovies). Based on shade plots in PRIMER these data were fourth-root transformed to ensure that all species were represented in the analysis. A zero-adjusted Bray-Curtis resemblance matrix was used due to zeroes in the data matrix. The Bray-Curtis resemblance matrix was then used in PERMANOVA+ to test for differences in the position of the centroids of the data for each factor being examined (e.g., shoals, ridge-swale habitats). If the PERMANOVA was significant then a pairwise PERMANOVA was run to determine where the differences occurred. Confidence regions (95%) were calculated using 150 bootstraps per group and the smooth, nominal 95% bootstrapped regions were projected on to a mMDS (metric-multidimensional scaling), either two-dimensional (2D) or three-dimensional (3D), to visualize the positions of the various factor levels in multidimensional space. A stress value associated with each mMDS provided a measure of how well the 2D or 3D ordination represented the data (Clarke et al. 2014). Stress values of a MDS < 0.05 give excellent (ideal) representations of the sample relationships, values of < 0.1 are still good and likely would not give misleading representation, values of < 0.2 give useful relationship representations but towards the higher end should be relied upon for general trends versus details in the MDS plot, and stress values > 0.3 indicate that the pattern on the MDS is near to random and are therefore not generally useful (Clarke et al. 2014).

For PERMANOVA tests that were significant, SIMPER (**S**imilarity **P**ercentage) (PRIMER v7) was used to interpret the average % dissimilarity between or among all pairs of groups of the individual factor levels (Anderson et al. 2008). SIMPER was also used to identify the species contributing to the dissimilarity (%) between pairs of groups, with species that explained at least 50% of the accumulated dissimilarities provided for comparison.

Due to the multivariate sampling design incorporating both spatial and temporal factors, analyses of fish assemblages were first structured by univariate analysis for a general overview of the fish assemblages related to spatial (shoal, ridge-swale habitats) and temporal (year, season, day-night) comparisons. This was followed by multivariate analysis of factors to determine interactions among factors relevant to spatial and temporal comparisons using two-way or three-way PERMANOVAs.

13.2.5.4 Potential Impacts of Dredging on Fish Assemblages

Direct comparisons to elucidate the impact of dredging on fish assemblages on CSII-BA relative to the reference shoals were tested using PERMANOVA+ (Anderson et al. 2008).

Comparisons used in the Dredge-1 event initially included tests for differences in daytime fish assemblages based on abundance between CSII-BA, CSII, and Bull in fall 2013 samples, which were pre-dredge samples for the first dredging event. Only day assemblages were tested because night sampling was not inclusive of fall samples until fall 2017. Due to significant differences observed among seasonal sampling (see results associated with Section 13.2.5.3), fall 2013 fish assemblages were then compared to those from fall 2014, fall 2015, fall 2016, and fall 2017. These comparisons were done to examine potential short-term versus long-term changes in fish assemblages associated with the first dredging event on CSII-BA. Although only one season was sampled on CSII-BA, CSII, and Bull Shoals (i.e., not Chester Shoal) prior to the dredging event, this sampling nevertheless followed a Beyond-BACI design where multiple control sites were sampled along with the impact site, and all sites were sampled before and after the dredging event (multiple times in the case of after dredging) (Underwood 1992; Anderson et al. 2008). The impact of dredging on fish assemblages was therefore analyzed using an asymmetrical PERMANOVA, where a significant interaction between (Impact/Control) X (Before/After) indicates an impact (change in direction or size in the multivariate effect) (Anderson et al. 2008).

Comparisons used in the Dredge-2 event included a pre-dredge year of sampling (winter 2016/17, spring 2017, summer 2017, fall 2017) and a dredge/post-dredge year of sampling (winter 2017/18, spring 2018, summer 2018, and fall 2018). Only day assemblages were tested because night sampling was not inclusive of all years until fall 2017. This sampling followed a complete Beyond-BACI design where multiple control sites were sampled along with the impact site, and all sites were sampled four seasons before and four seasons after the dredging (Underwood 1992; Anderson et al. 2008). The impact of dredging on fish assemblages was analyzed using an asymmetrical PERMANOVA, where a significant interaction between (Impact/Control) X (Before/After) indicates an impact (change in direction or size in the multivariate effect) (Anderson et al. 2008).

13.3 Results

13.3.1 Overall Species Composition and Diversity

Length frequency distributions of fish sampled using the open-TED and closed-TED trawls had similar maximum length distributions, which indicated that the closed-TED trawl was not sampling larger fish overall compared to the standardized open-TED trawl (Figure 13-2). In general, both open- and closed-TED trawls collected fish mostly < 200 mm TL. Most notably, however, the closed-TED trawls had a reduced capacity to retain smaller fish (i.e., larvae and juveniles, mostly < 100 mm TL) that passed through the mesh of the cod-end in the absence of the liner. Therefore, all analyses forward were based on the open-TED standardized trawls.

In total, 159,791 fish were caught in the standardized trawls on CSII-BA, CSII, Bull and Chester Shoals, representing 91 distinct genera and 137 distinct species (Appendix G). A further 44 fish groups could not be identified down to the species-level, primarily because they were larvae or juveniles. For example, juvenile sciaenids (drums and croakers) that were mostly < 30 mm TL were classified as Sciaenidae (juveniles). For the purposes of comparisons, however, these groups were included in the analyses because larvae/juvenile fishes were distinct from adults of the same Genus or Family. The maximum number of species/species groups observed was therefore 181. The following comparisons highlight general observations, not statistical significance; Sections 13.3.2–13.3.6 present statistical analysis for assemblages.

13.3.1.1 Species Composition Based on Numerical Abundance

13.3.1.1.1 Among Shoals

Overall, the most numerically abundant fish species on CSII-BA were juvenile sciaenids (drums and croakers) (49.5%), followed by Striped Anchovies (11.0%), unidentified anchovy species (*Anchoa* spp.) (5.3%), juvenile clupeids (herring-like species) (5.2%), juvenile cusk-eels (4.7%), unidentified fish larvae (4.3%), and Leopard Searobins (2.5%). All other fish species represented < 2% numerical abundance (Table 13-1).

Juvenile sciaenids (52.4%) were also the most numerical dominant fish species/group on CSII, with anchovies (*Anchoa* sp.) (10.2%), Striped Anchovy specifically (8.3%), juvenile clupeiform/clupeids (7.1%), Silver Seatrout (3.3%), Star Drum (2.4%) and Atlantic Croaker (2.3%) contributing to the numerical abundance. All other fish species were < 2% numerical abundance (Table 13-1).

The most numerically abundant fish species on Chester Shoal were anchovies (*Anchoa* sp.) (36.8%), and Striped Anchovy specifically (24.7%). Juvenile sciaenids were also common (8.5%), as were juvenile cusk-eels (3.7%) and juvenile clupeids (3.7%). Leopard Searobins represented 3.0% of the numerical abundance and Silver Seatrout contributed a further 2.6%. All other fish species represented < 2% of the numerical abundance on Chester Shoal (Table 13-1).

Overall, the most numerically abundant fish species on Bull Shoal were juvenile sciaenids (25.5%), followed by anchovy species (*Anchoa* sp.) (20.6%), Striped Anchovy and Dusky Anchovy specifically (8.9% and 4.3%, respectively), Leopard Searobins (4.5%), juvenile cusk-eels (4.3%), and juvenile Clupeiformes (herring-like fishes) and unidentified fish larvae (both 2.5%). All other fish species were represented by < 2% numerical abundance (Table 13-1).

13.3.1.1.2 Between Ridge and Swale Habitats

Sciaenids, including Atlantic Croaker, Banded Drum, Silver Seatrout and Star Drum, were two to nine times more common in the swales compared to the ridges (Table 13-2). In total, juvenile and larval fishes comprised ~40% of fish collected on both the ridges and the swales. Anchovies, including Dusky Anchovy, Striped Anchovy, and unidentified *Anchoa* spp., were marginally more numerically abundant in the ridge sampling compared to the swale sampling (41.9% versus 36.6% in total). Fringed Flounder was more numerically abundant in swales compared to ridges (1.0% versus 0.1%, respectively), but Spotted Whiff were equally represented in both the ridge and swale sampling (1.3% and 1.3%, respectively) (Table 13-2). Inshore Lizardfish and Leopard Searobin were more numerically abundant on ridges compared to swale abundance.

13.3.1.1.3 Among Seasons

In the fall, the most numerical abundant species was juvenile sciaenids (34.8%), followed by Striped Anchovy (26.2%), Silver Seatrout (9.1%), and unidentified *Anchoa* spp. (5.3%); all other species represented < 5% of the numerical abundance during the fall (Table 13-3).

In spring, Striped Anchovy and unidentified *Anchoa* spp. were numerically dominant (total of 47.9%) with unidentified juvenile clupeids contributing a further 22.1% and juvenile scienids another 6.8%; all other species contributed < 5% (Table 13-3).

Anchovies were also numerically dominant in the summer samples, with unidentified *Anchoa* spp. contributing 44.8% and Striped Anchovy 15.2% (Table 13-3). Juvenile Cusk-eel were also numerically abundant in the summer samples (6.8%); all other species contributed < 5%.

Winter samples were overwhelmingly dominated by juvenile sciaenids (76.0%) and all other species represented < 3% (Table 13-3).

13.3.1.1.4 Between Day and Night

Numerically, juvenile sciaenids dominated the day samples (46.7%) followed by anchovies, both Striped Anchovy (11.3%) specifically as well as unidentified *Anchoa* spp. (9.9%) (Table 13-4). Twelve other species were sampled at > 1% numerical abundance, including juvenile clupeids, Ladyfish/Malacho leptocephalus larvae, and unidentified fish larvae (total of 10.30%), Dusky Anchovy (2.6%), three sciaenids, one flatfish species, as well as others (Table 13-4).

At night, Striped Anchovy (17.1%) and unidentified *Anchoa* spp. (30.8%) dominated the catches, along with juvenile sciaenids (16.7%) and juvenile cusk-eels (5.6%) (Table 13-4). Eight other species contributed > 1% of the numerical abundance of the catch, including juvenile clupeids (total of 4.4%), Dusky Anchovy (1.6%), three sciaenid species, one flatfish, as well as others (Table 13-4).

13.3.1.2 Species Composition Based on Biomass

13.3.1.2.1 Among Shoals

Among all shoals, 30 species made up between 87% and 93% of the total biomass.

On CSII-BA, ~62% of the total biomass was from Atlantic Croaker (26.4%), Leopard Searobin (16.9%), Banded Drum (7.8%), Silver Seatrout (5.6%), and Spotted Whiff (5.3%) (Table 13-5). Twelve other species represented 1% to 4.7% of the total biomass, including one flatfish species, three ray species, two sciaenids, and Inshore Lizardfish plus others (Table 13-5).

Similarly, ~58% of the total biomass on CSII was comprised of Atlantic Croaker (34.2%), Silver Seatrout (7.6%), Banded Drum (6.3%), Star Drum (6.0%), and Leopard Searobin (5.5%), in addition to Smooth Butterfly Ray (5.3%) (Table 13-5). A further nine species represented between 1.1% and 4.7% of the biomass, including two flatfishes, two sciaenids, two rays, plus others (Table 13-5).

Biomass on Chester Shoal (~65%) was predominantly Leopard Searobin (19.8%), Atlantic Croaker (18.9%), Inshore Lizardfish (11.4%), Banded Drum (9.8%), and Silver Seatrout (5.6%) (Table 13-5). Eleven other species contributed between 1.1% and 4.4% biomass, including three flatfishes, one sciaenid, anchovies, cusk-eels, and others (Table 13-5).

On Bull Shoal, ~56% of the total biomass was comprised of Atlantic Croaker (19.7%), followed by Banded Drum (14.7%) and Leopard Searobin (14.2%), with Inshore Lizardfish contributing 7.2%. A further 12 fish species contributed 1–4% each to the total biomass of Bull Shoal, including four flatfish species and three sciaenids (Table 13-5).

13.3.1.2.2 Between Ridge and Swale Habitats

Sciaenids, including Atlantic Croaker, Banded Drum, Silver Seatrout, Southern Kingfish, Spot, and Star Drum, collectively represented ~54% of the biomass collected in swales compared to ~23% of the biomass collected on the ridges (Table 13-6). Blotched Cusk-eel, Striped Cusk-eel, and Cusk-eel juveniles represented 4.4% of the biomass on ridges and 1.77% of the biomass in swales. Flatfishes, in general, had greater relative biomass on ridges than in swales (10.4% versus 4.24%, respectively), with the exception of Fringed Flounder (0.68% and 3.30% on ridge versus swale, respectively) (Table 13-6). Inshore Lizardfish (11.7% versus 5.3%, for ridge versus swale) and Leopard Searobins (28.0% versus 9.1%) also had greater relative biomass on ridges compared with swales.

13.3.1.2.3 Among Seasons

The majority of biomass in each season was represented by Atlantic Croaker, Banded Drum, Inshore Lizardfish, and Leopard Searobin (Table 13-7). These four species represented 63.41% of the biomass in spring, 49.17% in summer, 54.06% in fall, and 57.72% in winter. Relative biomass of flatfishes ranged from 10.01% in spring to 16.65% in summer, 4.09% in fall, and 7.94% in winter.

13.3.1.2.4 Between Day and Night

Approximately 57% of the daytime biomass was comprised of Atlantic Croaker (24.8%), Inshore Lizardfish (10.6%), Leopard Searobin (10.3%), Silver Seatrout (6.7%), and Banded Drum (5.0%) (Table 13-8). During night, the biomass was also dominated by Atlantic Croaker (23.0%), Leopard Searobin (16.4%), and Banded Drum (15.2%), with Silver Seatrout contributing a further 4.5% but Inshore Lizardfish contributed lower biomass (3.76%) compared to the daytime biomass. Cownose Ray biomass was only observed during the day, and the biomass of Atlantic Cutlassfish, Smooth Butterfly Ray and Spot were higher in the day than in the night samples (Table 13-8). Banded Drum, on the other hand, had much higher biomass during the night (15.21%) compared to the day (5.04%), and Blotched Cusk-eel and juvenile cusk-eels were only caught during the night.

13.3.2 Diversity

In general, all of the indices of diversity, including species richness (S), Shannon Diversity Index (H'), and Simpson's Diversity Index ($1-\lambda$), provided similar information when tracked by ordered year and season samples among the study shoals over the duration of the study period (Figure 13-3). Species diversity for Bull Shoal was consistently higher than Chester that was in turn higher than CSII and CSII-BA. The Shannon Diversity Index and Simpson Diversity Index were moderate to high for most seasons and year except they both showed relatively low diversity during the fall and winter of each year, in particular on CSII and CSII-BA (Figure 13-3).

On a shoal-wide basis, the three diversity indices were consistent within each shoal but differed among the shoals (Figure 13-4). Bull Shoal had the highest diversity, with a total of 150 species encountered, a high Simpson's Diversity Index (0.875), and a relatively high Shannon Diversity Index (2.841) over the duration of the study (Table 13-9). This was in contrast to CSII and CSII-BA, which had lower but similar species richness (117 and 118 species, respectively), as well as Simpson Diversity Indices of 0.702 and 0.730, respectively (Figure 13-4, Table 13-9). Chester Shoal had moderate diversity relative to Bull Shoal, with 134 species encountered and moderate Simpson and Shannon Diversity Indices (0.790 and 2.255, respectively).

Overall, swale habitats showed consistently higher diversity compared to ridge habitats (Figure 13-5). However, whereas absolute values of diversity were greater for species richness at swale habitats (168 species) compared to ridge habitats (140 species), Simpson and Shannon Diversity Indices were not notably different between the habitats (Table 13-9).

Seasonally, species richness peaked in the summer with 149 species encountered and was lower and relatively consistent among spring, fall and winter (105–111 species) (Figure 13-6, Table 13-9). Simpson and Shannon Diversity Indices were consistently highest in spring but relatively consistent with summer and fall as well (Figure 13-6). Both indices, however, dropped markedly in winter (Figure 13-6), with both the Simpson and Shannon Diversity Indices reaching low values (0.420 and 1.327, respectively).

Diversity based on numerical abundance was consistently greater during the night compared to the day (Figure 13-7). However, the diversity of both time periods was relatively high, with a total species richness of 143 during the day compared to 158 during the night (Table 13-9). Simpson and Shannon Diversity Indices were also similar and relatively high (Table 13-9).

13.3.2.1 Species Abundance and Biomass

13.3.2.2 Among Shoals

Fish median abundance (number/10,000m²) differed among shoals (Kruskall-Wallis: $\chi^2 = 29.19$, $P < 0.0001$) (Figure 13-8). Median abundance was not significantly different between CSII and CSII-BA (188 vs 110 fish/10,000m²) (DSCF: $P = 0.0903$) (Figure 13-8), but CSII had greater abundance than Bull Shoal (89 fish/10,000m²) or Chester Shoal (78 fish/10,000m²) (DSCF: both $P < 0.0001$) (Table 13-10). Fish abundance on CSII-BA, however, did not differ significantly from fish abundance on Bull or Chester Shoals (Table 13-10).

Median fish biomass (g/10,000 m²) was significantly different among shoals ($\chi^2 = 18.3574$, $P = 0.0004$). Comparisons among the shoals indicated that CSII (1,142 g/10,000m²) and Bull (1,093 g/10,000m²) were not different from each other but had significantly greater median biomass than either Chester (808 g/10,000m²) or CSII-BA (667 g/10,000m²), which were not different from one another (DSCF: Table 13-11) (Figure 13-9).

13.3.2.3 Between Ridge and Swale Habitats

Overall, median fish abundance on swales (120 fish/10,000m²) was significantly greater than fish abundance on ridges (54 fish/10,000m²) ($\chi^2 = 68.72$, $P < 0.0001$) (Figure 13-10). Similarly, median fish biomass was greater in swales (1,262 g/10,000m²) than in ridges (489 g/10,000m²) ($\chi^2 = 132.97$, $P < 0.0001$) (Figure 13-11).

13.3.2.4 Among Seasons

Fish abundance differed among seasons (Kruskall-Wallis: $\chi^2 = 44.58$, $P < 0.0001$) (Figure 13-12). Fall had the highest median fish abundance (143 fish/10,000m²), which was significantly greater than the abundance in summer and winter (96 and 98 fish/10,000m², respectively), which was in turn significantly greater than the fish abundance in the spring (56 fish/10,000m²) (Table 13-10).

Median fish biomass also differed among seasons (Kruskall-Wallis: $\chi^2 = 32.89$, $P < 0.0001$) (Figure 13-13). Fall had the greatest median fish biomass (1,200 g/10,000m²) but was not significantly different than the winter median biomass (956 g/10,000m²). Winter biomass was not different than the median biomass in the summer (908 g/10,000m²), and spring had the lowest median fish biomass (623 g/10,000m²).

13.3.2.5 Between Night and Day

Median fish abundance was greater during the night (153 fish/10,000m²) compared to the day (61 fish/10,000m²) (Kruskall-Wallis: $\chi^2 = 129.53$, $P < 0.0001$) (Figure 13-14). Similarly, median fish biomass was greater during the night (1,563 g/10,000m²) than during the day (558 g/10,000m²) (Kruskall-Wallis: $\chi^2 = 147.35$, $P < 0.0001$) (Figure 13-15).

13.3.3 Univariate Analysis of Fish Assemblages Based on Spatial Comparisons Among Shoals and Ridge-Swale Habitats

13.3.3.1 Comparison Among Shoals

Based on species abundance, fish assemblages among the shoals occupied different dimensional space (PERMANOVA: $F_{\text{pseudo}} = 8.4486$, $P = 0.001$). Pairwise comparisons indicated that the fish assemblages among all shoals were significantly different from one another (Table 13-12). A mMDS based on the distances among centroids for the shoals and 95% bootstrapped confidence regions had low stress (0.10) and showed all shoals separated in relative space from one another (Figure 13-16). Average dissimilarities

among the shoal pairs varied between a high of 84.82% between Bull and CSII to 77.84% dissimilarity between Chester and CSII-BA (SIMPER: Table 13-13), with between 17 and 20 fish species contributing to 50% of the dissimilarities observed for each shoal-pair (Table 13-13).

Based on species biomass, fish assemblages among the shoals differed significantly (PERMANOVA: $F_{\text{pseudo}} = 8.4796$, $P = 0.001$). PERMANOVA pairwise comparisons indicated that the fish assemblages among the shoals were all significantly different from one another (Table 13-14). A mMDS showed differences in the distances among the centroids, with a relatively low stress value (0.10) and bootstrapped confidence regions that did not overlap (Figure 13-17). As with assemblages based on abundance, dissimilarities were greatest between Bull and CSII (84.90%) and lowest between Chester and CSII-BA (77.48%) (SIMPER: Table 13-15). Between 16 and 24 species contributed to ~50% of the dissimilarities between shoals (Table 13-15).

Fish assemblages based on both species abundance and biomass showed similar results with all shoals having assemblages that differed from one another significantly. In addition, bootstrapped confidence regions around Bull and Chester Shoals were more compact than CSII, which was smaller again compared to CSII-BA (Figures 13-16, 13-17).

13.3.3.2 Comparison Between Ridge-Swale Habitats

Based on abundance, fish assemblages differed between ridge and swale habitats (PERMANOVA: $F_{\text{pseudo}} = 63.336$, $P = 0.001$). Within-group average similarity for ridge assemblages was 36.5% and for swale assemblages was 32.3% but between-group average similarity was only 29.0% (PERMANOVA: pairwise comparisons). A mMDS based on the distances among centroids for the ridge-swale habitats and 95% bootstrapped confidence regions had low stress (0.04) and showed the ridge and swale habitats separated in dimensional space from one another (Figure 13-18). Ridge and swale assemblages had an average 83.65% dissimilarity, with 19 species contributing to ~50% of the dissimilarity (Table 13-16).

Based on species biomass, fish assemblages also differed between ridge and swale habitats (PERMANOVA: $F_{\text{pseudo}} = 62.527$, $P = 0.001$). Within-group similarity for ridge assemblages was 37.5% and for swale assemblages was 32.7%, with between-group similarity 29.5% (PERMANOVA: pairwise comparisons). A mMDS based on fish assemblage biomass had low stress (0.04) and clear separation of the centroids and the 95% bootstrapped confidence regions (Figure 13-19). SIMPER indicated that ridge and swale assemblages were 83.59% dissimilar, with 17 species contributing to ~50% of the dissimilarity (Table 13-17).

13.3.4 Univariate Analysis of Temporal Comparisons Among Years, Seasons, and Day-Night Fish Assemblages

13.3.4.1 Comparison Among Years

Based on fish species abundance, fish assemblages were different among the years of 2014–2018 (PERMANOVA: $F_{\text{pseudo}} = 9.0538$, $P = 0.001$). Years excluded from the analysis were 2013 and 2019 because they only sampled a portion of the year and not all seasons were represented in the annual data. For 2014–2018, all years were significantly different from one another in dimensional space (Figure 13-20) (PERMANOVA: pairwise comparisons) (Table 13-18). Pairwise similarities within and between years indicated that 2018 shared less similarity with 2014–2017 (range of 27.65%–30.03% similarity) than all the other years (31.45%–34.56%) (Table 13-19). This was also observed in the mMDS of the centroids with 95% confidence regions where 2018 was separated further from all other years in dimensional space (Figure 13-20).

Based on fish species biomass, fish assemblages were different among the years of 2014 to 2018 (PERMANOVA: $F_{\text{pseudo}} = 9.081$, $P = 0.001$). PERMANOVA pairwise comparisons between years

indicated that fish assemblages in all years differed from one another (Table 13-20). Similar to differences among years based on species abundance, fish assemblages based on biomass in 2018 shared less similarities with the other years (range of 28.52%–30.77%) compared to the similarities among the years of 2014–2017 (range of 32.10% to 35.55%) (Table 13-21). A 2D mMDS showed the separation of all of the fish assemblages by year, with 2018 furthest away from the other years (Figure 13-21).

13.3.4.2 Comparisons Among Seasons

Based on species abundances, fish assemblages differed among seasons (PERMANOVA: $F_{\text{pseudo}} = 22.015$, $P = 0.0001$). Pairwise PERMANOVA tests indicated that all seasons were different from one another (Table 13-22). A 2D mMDS had low stress (0.10) and showed separation of all seasons without any overlap in the 95% bootstrapped confidence regions (Figure 13-22). Fall and summer seasons had the greatest average dissimilarity in assemblages based on abundance (85.85%) (SIMPER: Table 13-23), which was also notable in the 2D mMDS (Figure 13-22). The majority of difference between fall and summer assemblages was attributed to a greater abundance of most species during the summer (Table 13-23). Between 14 and 19 species accounted for at least 50% of the dissimilarities between seasons (Table 13-23).

Fish assemblages based on species biomass also differed by season (PERMANOVA: $F_{\text{pseudo}} = 19.113$, $P = 0.001$). As with fish assemblages based on species abundance, pairwise PERMANOVA tests indicated that fish assemblages based on species biomass for all seasons were different from one another (Table 13-24). Fall versus summer fish assemblages had the highest average dissimilarity (85.56%), with a direction of increased abundance of most fishes during the summer except sciaenids that had increased abundance in the fall (SIMPER: Table 13-25). At least 50% of the dissimilarities between season-pairs were based on 14–18 species (Table 13-25).

13.3.4.3 Comparison Between Day-Night

Based on fish species abundance, fish assemblages were significantly different between day and night (PERMANOVA: $F_{\text{pseudo}} = 51.43$, $P = 0.0001$). A 2D mMDS had low stress (0.05) and showed centroids with non-overlapping 95% bootstrapped confidence regions (Figure 13-24). On average, within-day collections were 34.55% similar and within-night collections were 32.70% similar, and between day and night collections were 28.90% similar (PERMANOVA: pairwise comparisons). SIMPER results showed that day and night assemblages based on abundance had an average dissimilarity of 82.70% and was attributed to the majority of fish species being more abundant during the night, with juvenile cusk-eels, and Blotched and Striped Cusk-eels, only present during the night (Table 13-26).

Based on species biomass, there was a significant difference in fish assemblages during day versus night (PERMANOVA: $F_{\text{pseudo}} = 52.571$, $P = 0.001$). Average similarity within-day fish assemblages was 35.18% and within-night assemblages was 33.35%, with average similarity between day and night assemblages 29.52% (PERMANOVA: pairwise comparisons). As for abundance, a mMDS based on biomass had low stress (0.05) and clearly showed separation between the 95% bootstrapped confidence regions for day versus night assemblages (Figure 13-25). Based on biomass, species assemblages during the day and night had an average dissimilarity of 82.53%, with 18 species/groups contributing ~50% to the dissimilarity (Table 13-27). As with species abundance assemblages, cusk-eels (unidentified juveniles, Blotched and Striped) were only captured during night, and the direction of greater abundance for the other species/groups was also towards night (Table 13-27).

13.3.5 Multivariate Analysis of Fish Assemblages

Given the similarity in fish assemblages based on species abundance and species biomass responses to univariate comparisons on the basis of both spatial (Shoal, Ridge_Swale) and temporal (Year, Season, Day_Night) factors, multivariate analyses of fish assemblages were based on species abundance metrics.

Shoal, Season, and ShoalxSeason interaction were all significant when comparing fish assemblages based on species abundances (PERMANOVA: Table 13-28). Partitioning of the components of variation in the PERMANOVA indicated that Season accounted for the greatest amount of variation (218.24) after residual variation (Table 13-28). The importance of Season, more so than Shoal, was also indicated in a 3D mMDS plot (stress = 0.12) that showed clusters based on Season (type of symbol) more so than Shoal (color of symbols) (Figure 13-26).

Furthermore, inclusion of the factor of Day_Night into the analysis with Shoal and Season fish assemblages based on abundance indicated that all the main effects as well as their interactions were significant (PERMANOVA: Table 13-29). Partitioning of the components of variation in the PERMANOVA indicated that Day_Night and then Season accounted for the greatest amount of variation, followed by Day_NightxSeason interaction (Table 13-29). This was visualized with a 3D mMDS with stress of 0.13 where the separation in dimensional space was greatest between day versus night fish assemblages (open versus solid symbols on left and right sides of plot, respectively) and secondarily among Seasons (type of symbol) (Figure 13-27). Among Seasons, fall and summer were the most dissimilar as indicated by the spread in their distance in the mMDS plot (i.e., top versus bottom of plot, respectively). Many pairwise shoal differences were observed based on combinations of Season and Day_Night (PERMANOVA pairwise comparisons) (Table 13-30), as expected based on the significant interaction term among Shoal, Season, and Day_Night (Table 13-29).

Fish assemblages based on abundance had a significant interaction when both shoal and ridge-swale factors were considered simultaneously (PERMANOVA: Table 13-31). Partitioning of the variation showed that the ridge-swale factor accounted for the greatest amount of variation (361.24) following the residuals (Table 13-31). This was clearly visualized in a 2D mMDS plot with low stress (0.08) where the centroids for day were clustered separately from the centroids for night on the first MDS axis (MDS1), but individual shoals were different in direction or abundance on MDS2 axis (Figure 13-28).

Based on abundance, fish assemblages for Season and Ridge_Swale habitats had a significant interaction (PERMANOVA: Table 13-32). A 2D mMDS with moderate stress (0.12) showed separation primarily in the dimensional space for ridge versus swale habitats along MDS1 axis and separation of seasons along the MDS2 axis (Figure 13-29), although the size and direction varied as expected given the significant interaction term.

Combining all three factors of Shoal, Season, and Ridge_Swale resulted in a nonsignificant interaction term for ShoalxSeasonxRidge_Swale (PERMANOVA: $F_{\text{pseudo}} = 0.9284$, $P = 0.7104$). The PERMANOVA was therefore rerun after removing the nonsignificant interaction term and resulted in significant main effects and two-way interactions in the main effects (PERMANOVA: Table 13-33). Partitioning of the variation showed that most was due to Ridge_Swale (373.21) and secondarily by Season (215.86), both following residual variation. MDS plots for two-way main effects were previously generated for ShoalxSeason (Figure 13-26), ShoalxRidge_Swale (Figure 13-28), and SeasonxRidge_Swale (Figure 13-29). MDS plots for one-way main effects were also previously generated under the univariate analysis (Section 13.3.3), including for Shoal (Figure 13-16), Season (13-18), and Ridge_Swale (13-22).

13.3.6 Dredging Events and Potential Impacts on Fish Assemblages

13.3.6.1 Dredge-1 Event for Fish Assemblages

Based on abundance, day fish assemblages did not differ between CSII and CSII-BA in the pre-dredge period of fall 2013 (PERMANOVA: $F_{\text{pseudo}} = 1.247$, $P = 0.424$). However, with the inclusion of Bull Shoal in the pre-dredge fall 2013 period then fish assemblages were seen to differ among CSII, CSII-BA, and Bull Shoals (PERMANOVA: $F_{\text{pseudo}} = 2.631$, $P = 0.012$). PERMANOVA pairwise tests consistently indicated no difference between CSII and CSII-BA ($P = 0.403$). Although differences between CSII and Bull ($P = 0.059$) and CSII-BA and Bull ($P = 0.063$) were marginally nonsignificant at $P < 0.05$, they still indicated that CSII and CSII-BA were both more different from Bull compared to any differences in the day fish assemblages between CSII and CSII-BA.

When comparing fall 2013 daytime fish assemblages on the shoals in comparison to daytime fish assemblages in fall 2014, 2015, 2016, and 2017, there was no significant interaction between shoal and fall season (PERMANOVA: $F_{\text{pseudo}} = 1.190$, $P = 0.152$) but both the main effects of shoal ($F_{\text{pseudo}} = 2.224$, $P = 0.007$) and fall-yr ($F_{\text{pseudo}} = 2.334$, $P < 0.001$) were significant. A 3D nonmetric-MDS of shoal and year had a relatively low stress value (0.1) and indicated that the fish assemblages on all the shoals changed from fall to fall (Figure 13-30), including the reference shoal (Bull Shoal) and CSII (reference portion of CSII). PERMANOVA pairwise tests for each fall from 2013 to 2017 indicated that fish assemblages started out very similar on all three shoals, changed in similar directions and sizes over the ensuing fall seasons (pairwise comparisons nonsignificant), and by fall 2017 were more similar to one another than in the pre-dredge season of fall 2013 (Table 13-33). This indicated that the changes that were observed on CSII-BA were not due to the dredging event itself and were similar to changes that occurred on the other shoals during this same time period.

Fish assemblages for CSII-BA, CSII, and Bull Shoals for one season (fall 2013) before and four seasons (fall 2014, fall 2015, fall 2016, and fall 2017) after the Dredge-1 event on the Impact shoal and the control/reference shoals (i.e., a Beyond-BACI design) had no significant interaction between Before/After and Impact/Control (Table 13-36). This indicated that any changes to CSII-BA could not be ascribed to a dredging impact and was not discernible from the natural variability observed in the fish assemblages of CSII and Bull Shoals during the same period of time.

13.3.6.2 Dredge-2 Event for Fish Assemblages Based on Abundance

Based on daytime species abundance, there was a significant interaction between Shoal (CSII, CSII-BA, Chester, and Bull) and Season-Year (winter 2016/17, spring 2017, summer 2017, and fall 2017) in the year prior to the second dredging event (PERMANOVA: $F_{\text{pseudo}} = 1.638$, $P = 0.0046$). PERMANOVA pairwise tests indicated that the fish assemblages on the shoals were similar in winter 2016/17, differed significantly in spring 2017 and summer 2017, and then most pairwise shoal comparisons indicated more similarities in fish assemblages in the fall 2017 (Table 13-34). This was visualized using a 2D nMDS with stress of 0.13, which showed a similar pattern of seasonal change in fish assemblages with averages for all shoals in winter 2016/17 and fall 2017 closer together in dimensional space than spring 2017 or summer 2017 (Figure 13-31).

Comparing all four seasons prior to dredging (winter 2016/17, spring 2017, summer 2017, fall 2017) along with the subsequent four seasons during (winter 2017/18, spring 2018) or after dredging (summer 2018, fall 2018) yielded a significant interaction among Shoal and Season-Year (PERMANOVA: $F_{\text{pseudo}} = 1.623$, $P = 0.0001$). PERMANOVA pairwise tests showed no significant differences between fish assemblages on the shoals in winter 2017/18 (Table 13-35). Most observed differences among shoals were in spring 2017 and summer 2017 (pre-dredge Season-Year, as indicated previously). Bull and Chester Shoals, as well as Chester and CSII Shoals, differed in spring 2018 and Chester Shoal fish

assemblages differed from CSII and CSII-BA in summer 2018, as well as from CSII in fall 2018 (Table 13-35). Fish assemblages on CSII and CSII-BA were not significantly different for any of the seasons pre-dredge or during/post dredge (PERMANOVA pairwise test all $P > 0.05$) (Table 13-35). A 2D nMDS plot (stress = 0.16) was used to try to visualize the differences among Season-Year combinations of fish assemblages (Figure 13-32). Most notably, shoals in the season in which Dredge-2 started, winter 2017/18, clustered together (triangle symbols and red dashed line) as expected from nonsignificant differences in their fish assemblages. Fish assemblages in both pre-dredge (spring 2017, summer 2017) and post-dredge (summer 2018, fall 2018) periods showed considerable variability in fish assemblages as indicated by their position at a distance from the cluster of Season-Year pairs where shoals were more similar (i.e., winter 2016/17, winter 2017/18) (Figure 13-32).

Fish assemblages four seasons before and four seasons after the Dredge-2 event on the Impact shoal and the control/reference shoals (i.e., a Beyond-BACI design) had no significant interaction between Before/After and Impact/Control (Table 13-37). Again, this indicated that any changes to CSII-BA due to dredging was not discernible from the natural variability observed in the control/reference shoals and therefore could not be ascribed to dredging.

13.4 Discussion

Overall, the sand shoals off the east coast of Florida have diverse fish assemblages that are dynamic in all aspects of space and time. Spatially, fish species richness varied among the shoals and the total number of species was highest on Bull Shoal (150 species) compared to Chester Shoal (134 species) and CSII and CSII-BA Shoal (117–118 species, respectively) (Table 13-9). On average over the duration of the study, Bull Shoal showed species richness ranging from ~30 to 60 species per sampling season (Figure 13-3), Chester was similar in pattern but had slightly lower diversity (~20–50 species per sampling season), and CSII and CSII-BA had the lowest species richness at ~15–40 species per sampling season. Both the Shannon Diversity Index and the Simpson Diversity Index were moderate for all seasons except most winters (Figure 13-3). These indices were mostly likely influenced in the winter by the relatively low diversity but high abundance of one group in particular, the juvenile sciaenids (drums and croakers). Juvenile sciaenids represented ~75% of the abundance of all fishes sampled in the winter (Table 13-3).

The number of fish species documented in this study was high compared to species richness encountered in previous shoal studies. Vasslides and Able (2008) noted an overall total of 61 species in their study of fish communities over a 23-km transect that crossed a shoreface sand ridge off southern New Jersey. Although their study used two different trawl types that provide some relative comparison (beam trawl and otter trawl) over multiple years each (1991–95 and 1997–2006, respectively), the species richness could have been limited to some degree because their sampling was restricted to mid- to late- summer only (July to September). Similarly, fish diversity was comparatively low in a study using multiple gears sampling for 2 years on four shoals (sampling only the top of the shoal, hence the ridges) and four flat-bottom sites off the coast of Maryland and Delaware (Slacum et al. 2010). Small trawl sampling resulted in a total of 25 fish species whereas the larger commercial trawl sampled 43 fish species (Slacum et al. 2010). Comparing species richness among studies was complicated, however, due to varying mesh sizes in the trawls and the varying duration and total number of tows completed in each study (i.e., sampling effort). Comparatively, however, a species accumulation curve as a function of sample size (number of tows) for the present study yielded ~150 species for 256 trawls (Figure 13-33), which is the total number of trawl samples collected by Slacum et al. (2010) (i.e., both trawl types but not inclusive of their gill net samples).

It may be expected, however, that species richness on shoals off the east coast of Florida would be higher than shoals in more northerly regions on the basis of biogeographic provinces and corresponding sources of fish species. The area around Cape Canaveral on the east coast of Florida is known as a transition zone

between the Carolina and the Caribbean biogeographic provinces and is characterized as having tropical to warm temperate waters (Toonen et al. 2016). It is also directly offshore of the Indian River Lagoon (Figure 13-1), which is an estuary that spans ~250 km (156 miles) on a north-south axis (SJRWMD 2023). The Indian River Lagoon has one of the highest fish diversities in North America (397 fish species out of a total of 782 fish species found on the east central coast of Florida); this diversity can act as a source for fish species found further offshore (Gilmore 1995). Gilmore (1995) credits the high diversity primarily to the extensive north-south axis of the estuary and the steep north-south isothermal gradient formed, in particular, in the winter by the northward-flowing Florida Current/Gulf Stream. Both of these physical factors contribute to a heterogeneous thermal environment in the Indian River Lagoon and adjacent coastal shelf areas (Gilmore 1995). In addition, the Florida Current originating in the Florida Straits through a convergence of currents flowing north from the Caribbean Sea and the northeastern Antilles (Fautin et al. 2010), as well as the Loop Current coming from the Gulf of Mexico, brings tropical/subtropical waters and therefore fish eggs and larvae from tropical/subtropical fish species up the east coast of Florida (Gilmore 1995).

Spatially, fish abundance and biomass (as number or grams per 10,000 m², respectively) varied among the shoals, with CSII having the greatest abundance but similar biomass to Bull Shoal (Figures 13-8 and 13-9). On an overall basis, all the shoals had fish assemblages that were relatively different in multivariate space, varying in either direction or size based on both abundance and biomass (Figures 13-16 and 13-17). This indicated that the shoals all had some individualistic features and natural variation that could make it difficult to generalize to others shoals in the area that were not sampled. The similarity between results based on numerical abundance and biomass that was observed throughout the analyses was probably due to the relatively small size of all the fishes collected, with most fish sampled < 200 mm (i.e., relatively consistent biomass per individual).

Habitats within each shoal consistently had significantly greater diversity of fishes in swales compared to ridges (Figure 13-5). In addition, both fish abundance and fish biomass were significantly higher in swales than on ridges (Figures 13-10, 13-11). Similarly, Slacum et al. (2010) found abundance, species richness, and species diversity to be higher on flat-bottom habitats compared to shoals, the latter being comparable to ridges since their sampling was restricted to the tops of the shoals. Vasslides and Able (2008) also documented lower fish abundance on top of the shoal ridge compared to either side of the ridge in their study off southern New Jersey. Fish assemblages in the present study also differed significantly between ridge and swales (Figures 13-18, 13-19), with 19 and 17 species, respectively, contributing to ~50% of the dissimilarities. Inshore Lizardfish, Fringed Flounder, Leopard Searobin, Spotted Whiff, and Atlantic Croaker contributed the most to the dissimilarity between swales and ridges, with all except Leopard Searobin being in greater abundance and biomass in swales (Table 13-16, 13-17). Slacum et al. (2010) also noted differences in fish assemblages between shoals (i.e., ridges) and flat-bottom areas, although they found only 4 to 7 species responsible for ~50% of the dissimilarity. This lower number of species showing dissimilarity could be expected based on the overall lower fish diversity shown in their study but also demonstrates how different the northern shoal fish assemblages (Maryland, Delaware, New Jersey) are compared to the southern shoals off the east coast of Florida.

Slacum et al. (2010) attributed some of the differences in abundance and fish assemblages between shoals (ridges) and flat-bottom habitats to higher productivity and hence food resources in flat-bottomed, typically muddier areas. This was based on Cutter et al. (2000) that showed benthic invertebrate densities were higher in the troughs (i.e., swales) adjacent to Fenwick and Weaver Shoals off the coast of Delaware, which were the same two shoals (out of four) used in Slacum et al. (2010). Similarly, most epibenthic and infaunal invertebrates (Chapter 11) had higher abundance, and demersal invertebrates (Chapter 12) had higher biomass, in swales compared to ridges, with few exceptions.

Temporally, fish abundance, biomass, and assemblages varied on an annual, seasonal, and diel basis. Although fish assemblages also varied overall from year to year of the study (Figures 13-20, 13-21), seasonal and diel factors contributed more to the temporal differences. Seasonal changes in abundance and biomass were highly significant, with fall having the greatest abundance and biomass, followed by winter, summer, and spring (Figures 13-12, 13-13). Overall, fish assemblages also varied by season with all seasons significantly different from one another (Figures 13-22, 13-23), and with fall and summer having the greatest dissimilarity. Diversity indices also differed from spring through fall and, as mentioned previously, decreased markedly during winter (Figure 13-6) most likely due to fewer species being present but in larger numbers and biomass (i.e., the sciaenids). Although there was a significant Shoal x Season interaction, visualizing this interaction clearly demonstrated that the seasonal factor was dominant (Figure 13-26). Slacum et al. (2010) also found a strong effect on fish assemblages based on season and furthermore detected differences between ridge and flat-bottom habitats in some, but not all, seasons. The differences in ridge and swale fish assemblages as a function of season in the present study was also evident when visualized using a 2D metric-MDS (Figure 13-29).

Seasonal differences in fish assemblages were primarily driven by dissimilarities (size and direction of the multivariate effect) in Inshore Lizardfish, Spotted Whiff, Leopard Searobin, and Atlantic Croaker abundance and biomass, and with Silver Seatrout, Fringed Flounder, and juvenile sciaenids depending on the season (Tables 13-23, 13-25). Overall, Inshore Lizardfish and Leopard Searobin were more abundant in the spring and summer, Spotted Whiff in the summer, and Atlantic Croaker, Silver Seatrout and juvenile sciaenids in the fall and winter. Although there were clear seasons of abundance for most species, there were also 21 species that were present on all of the shoals in every season and represented at least > 1% of the species composition on any individual shoal during a season (Table 13-3). These species could be considered to be “resident” and were primarily in the Sciaenidae (croakers and drums), Clupeidae (anchovies and herrings), Triglidae (searobins), and Paralichthyidae (flatfishes). Many other species were present in all or most seasons but occurred in low abundance or biomass. Therefore, while the fish assemblages were dynamic from season to season, there was a core assemblage of many fish species that clearly associated with the sand shoals and were not transient.

Very few previous studies have considered diel sampling of fish assemblages in relation to shoals but our study indicated that day-night differences were significant. The number of species (143 versus 158 for day and night, respectively) and diversity (0.753 and 0.841, respectively) were marginally greater during the night. However, abundance (2.5X) and biomass (2.8X) were significantly higher at night, and fish assemblages were also significantly different between day and night. While some fish species were only captured at night, such as Blotched and Striped Cusk-eels and juvenile cusk-eels, other species were captured during day and night but in much higher abundance and biomass during the night (Table 13-26). In studying juvenile fishes in habitats with differing degrees of biogenic habitat, Diaz et al. (2003b) noted an increase in the density of fishes at night in bare sandy habitats compared to complex habitats, whereas during the day the more complex habitats had approximately twice as many fish as the bare sandy habitats. The shoals off the east coast of Florida are comprised of primarily sand (fine to coarse) and shell hash (fine and coarse) (Chapter 4), with little “complex” habitat other than small patchy distributions of sand dollars and *Diopatra* tube worms (pers. obs.). However, many of the decapod crustaceans, including shrimps and amphipods, scavenge at night and bury in the sand during the day. The sergestid shrimps (e.g., *Acetes americanus carolinae*) vertically migrate from the deeper waters of the shoals into the water column during the night, returning to the bottom again at dawn. Night on the shoals could therefore be an active period of time for fishes foraging on small invertebrates (and other fishes).

Sampling during the night on offshore shoals should be a priority for any monitoring program based on the abundance, biomass, and fish assemblages observed in this study. However, these coastal shoals are relatively shallow, have large waves generated by offshore winds and storms, and have currents

influenced by tides and coastal bathymetry, which all contribute to the difficulty in safely trawling at night consistently throughout a long-term study.

The ultimate objective of this study was to determine if dredging events that took place during the study resulted in a measurable impact to the fish assemblages of the dredged shoal, CSII-BA. Due to the temporal and spatial extent of sampling, we were able to use a powerful Beyond-BACI analysis to assess the impact of dredging on CSII. Beyond-BACI analysis requires the fish assemblages to be sampled multiple times before and after the impact event (dredging), and it also requires more than one reference shoal to be sampled along with the potentially impacted shoal (Underwood 1992; Anderson et al. 2008). To our knowledge, our study is the first that has been able to use this approach for assessing dredging impacts to sand shoals in the USA. Based on this analysis for two dredging events, one in winter 2013/14 through spring 2014 and a second dredging event in winter 2017/18 through spring 2018, there was no detectable impact on the fish assemblages of CSII-BA (the dredged shoal) relative to the natural variability observed in fish assemblages of the reference shoals (CSII, Bull, and Chester Shoals). In general, the shoals off the east coast of Florida are dynamic with fish assemblages that are spatially and temporally variable, and any impacts from dredging the shoals are encompassed within the natural variability of the shoals as a whole.

13.5 Acknowledgments

This research would not have been possible without the collaboration of Jim Stringer of the FRRG, including all the captains and mates of the *Laffin' Place*. FRRG gave us exclusive contract use of the research vessel for all 6 years of the study, which allowed us the flexibility to plan and schedule field sampling around inclement weather and personnel availability. Mike Dickson, Monica Ditch, Bethan Gillett, Amanda Croteau, Daryl Parkyn, Fabio Caltabellotta, Alex Linares, and Kerrin Toner contributed to the field work and sample processing for this project, and we are sincerely thankful for their dedicated work. In addition, over the span of the 6 years of this project there have been dozens of biologists, undergraduate biological technicians, and undergraduate student volunteers that have participated and assisted in field and lab work, which was greatly appreciated and collectively provided substantial assistance.

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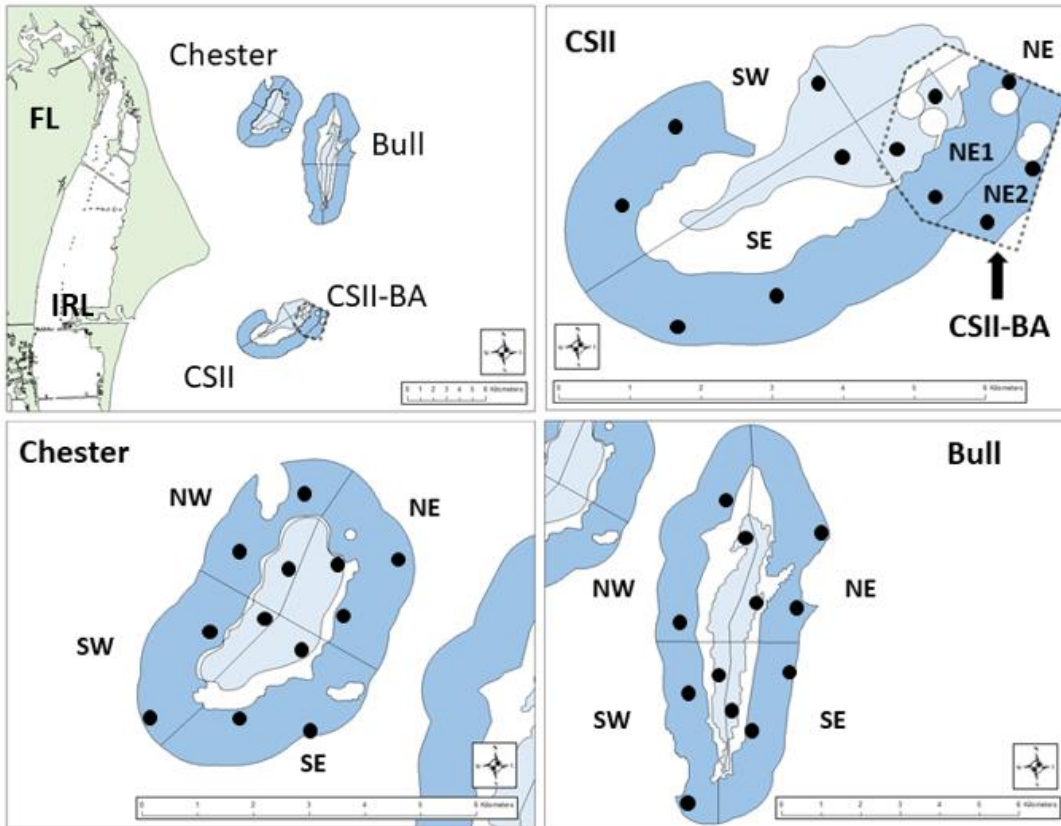


Figure 13-1. Location of study shoals off Cape Canaveral on the east coast of Florida.

This figure shows the ridge (light blue) and swale (dark blue) regions and potential stratified random sampling stations (solid black circles) in each quadrant identified by NW, NE, SW, and SE, for CSII, CSII-BA, Chester, and Bull Shoals. IRL denotes the Indian River Lagoon. The CSII-BA (dashed outline) was subdivided into a ridge portion and a swale portion, with the latter further subdivided to ensure that pre-dredge sampling was conducted in at least one portion of the swale where active dredging occurred.

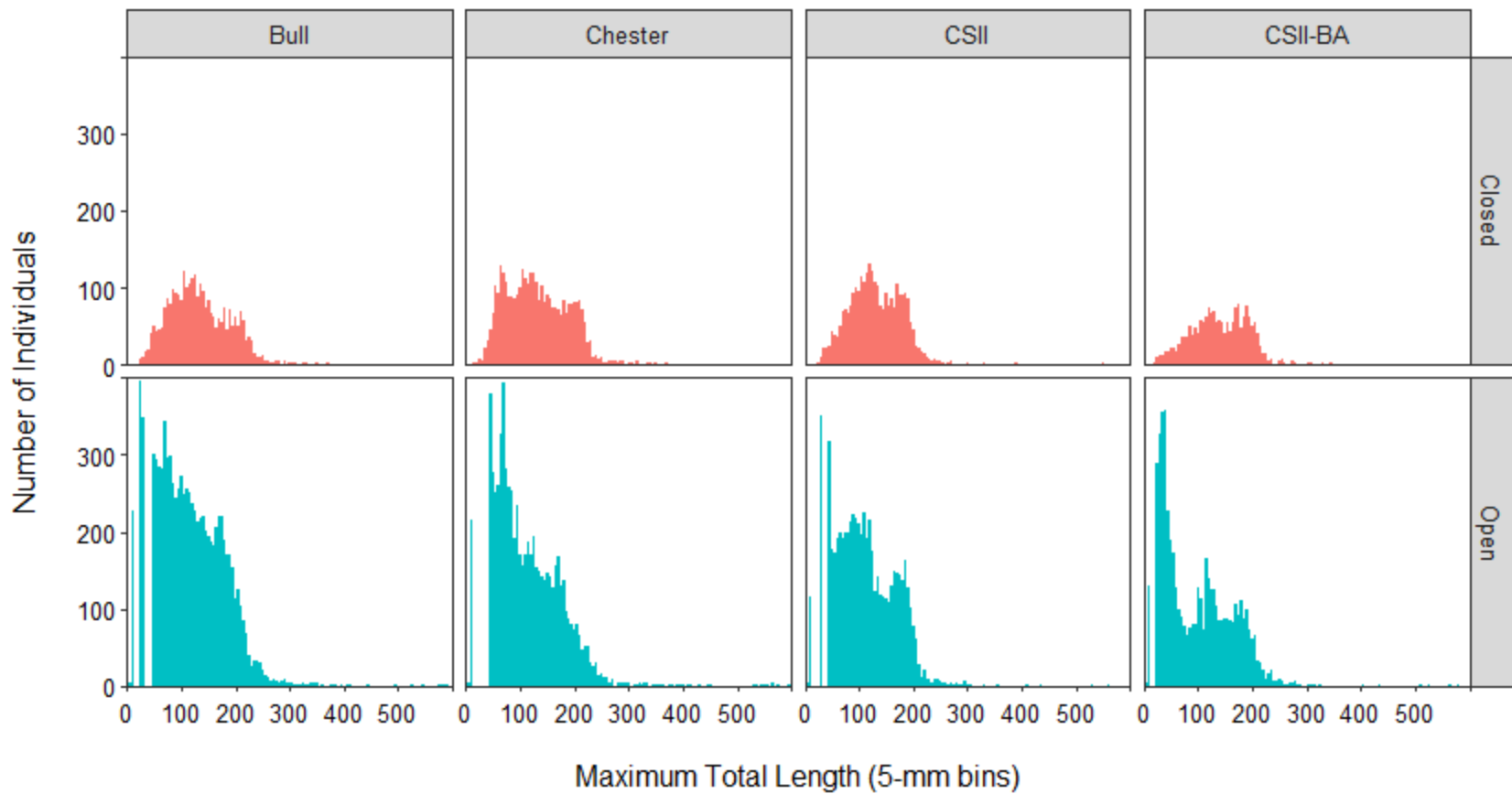


Figure 13-2 Length distributions of fish collected on offshore sand shoals using the standardized open-TED trawl with a cod-end liner (blue) and closed-TED trawl without a cod-end liner (salmon).

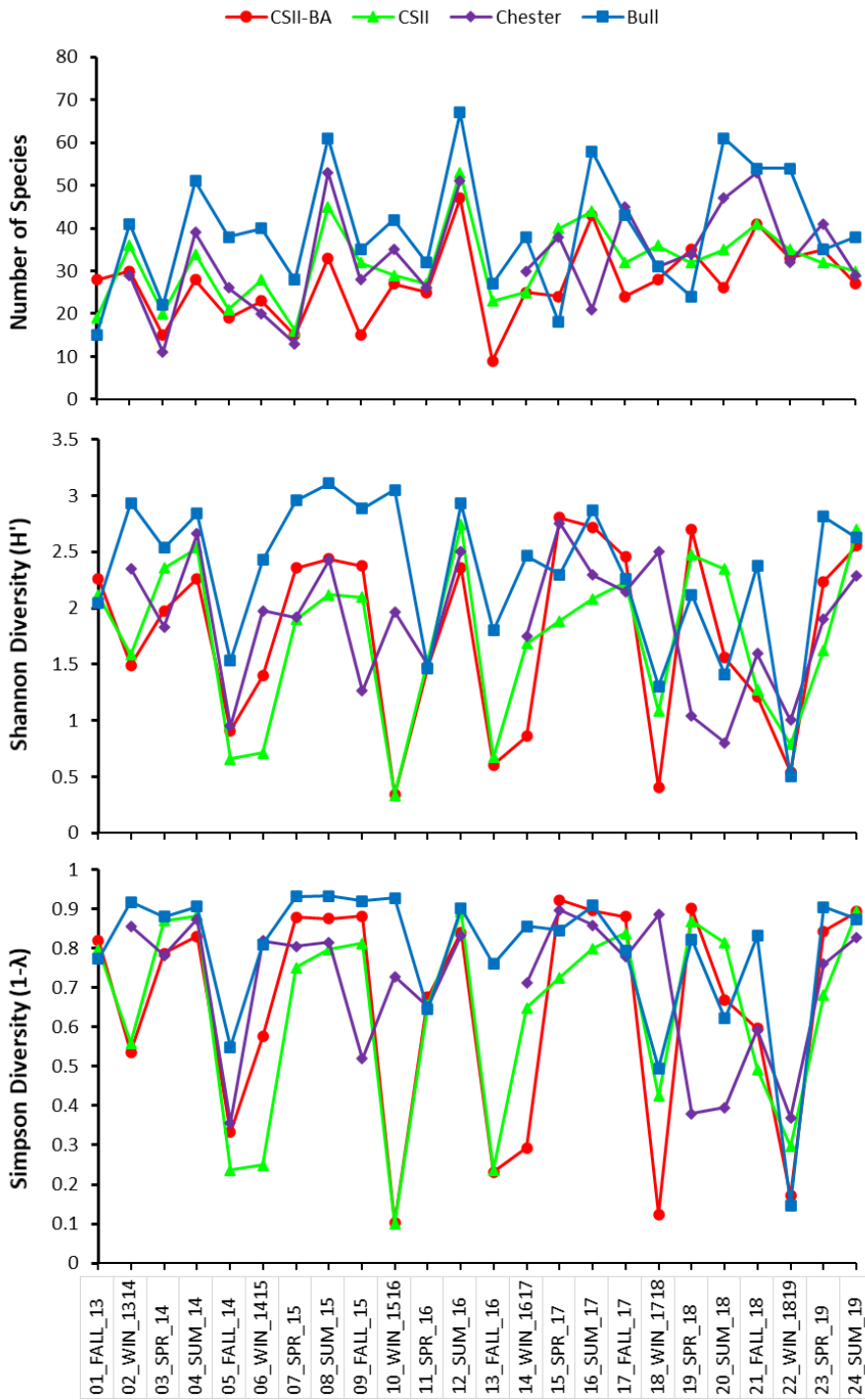


Figure 13-3. Overall indices of diversity based on numerical abundance for all study shoals by ordered season-year over the duration of the study.

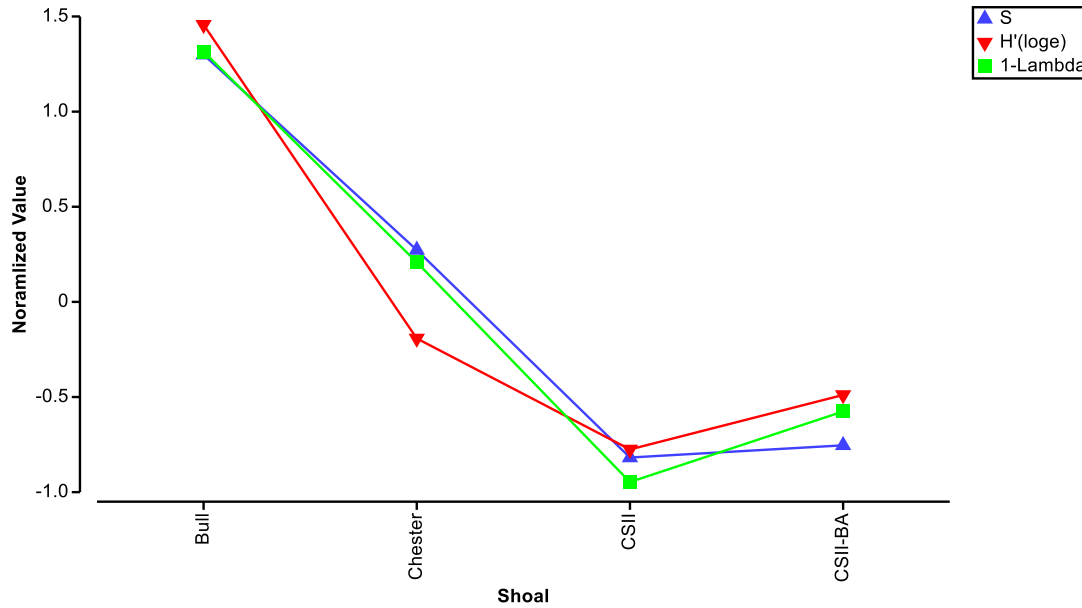


Figure 13-4. Overall indices of diversity based on numerical abundance among shoals.

S is the total number of species, H' is the Shannon Index of Diversity, and 1-Lambda is Simpson's Index of Diversity. Values have been normalized to allow simultaneous display of multiple diversity indices. Absolute values are given in Table 13-9.

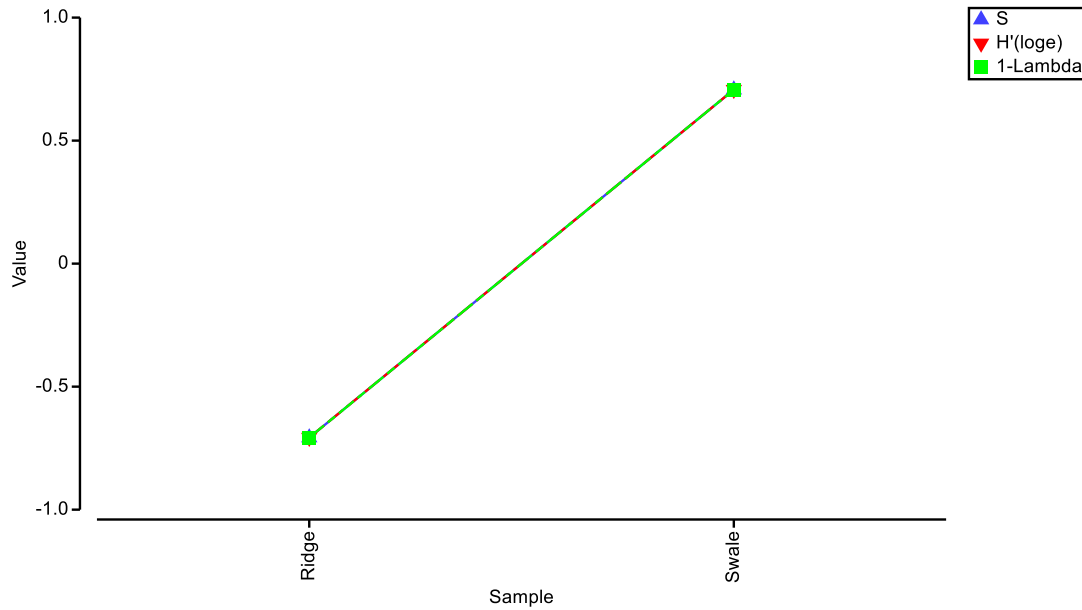


Figure 13-5. Overall indices of diversity for ridges versus swales based on numerical abundance.

S is the total number of species, H' is the Shannon Index of Diversity, and 1-Lambda is Simpson's Index of Diversity. Values have been normalized to allow simultaneous display of multiple diversity indices. Absolute values are given in Table 13-9.

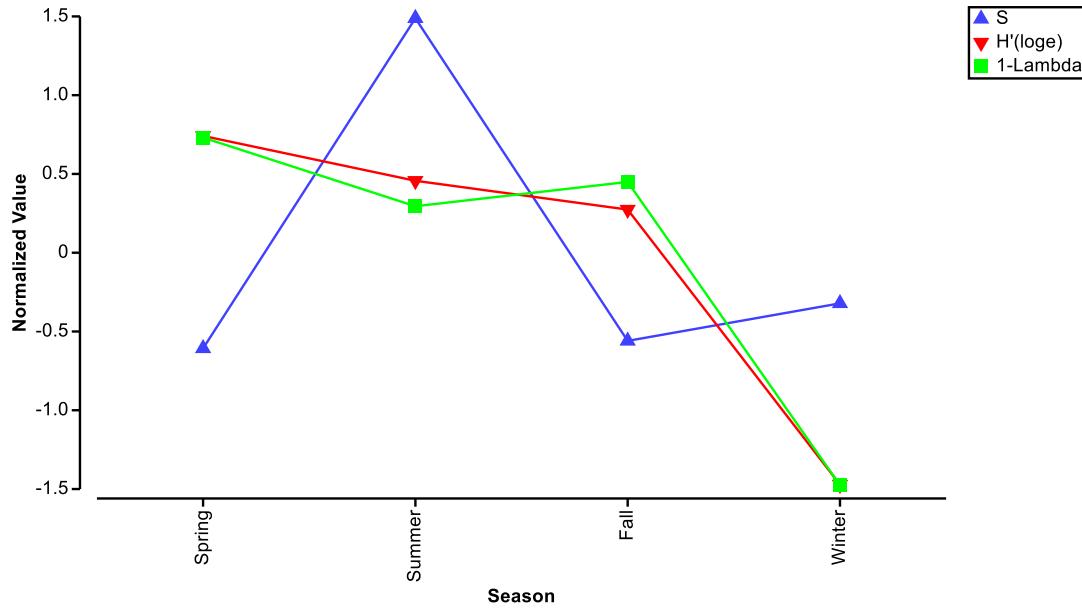


Figure 13-6. Overall indices of diversity among seasons based on numerical abundance.

S is the total number of species, H' is the Shannon Index of Diversity, and 1-Lambda is Simpson's Index of Diversity. Values have been normalized to allow simultaneous display of multiple diversity indices. Absolute values are given in Table 13-9.

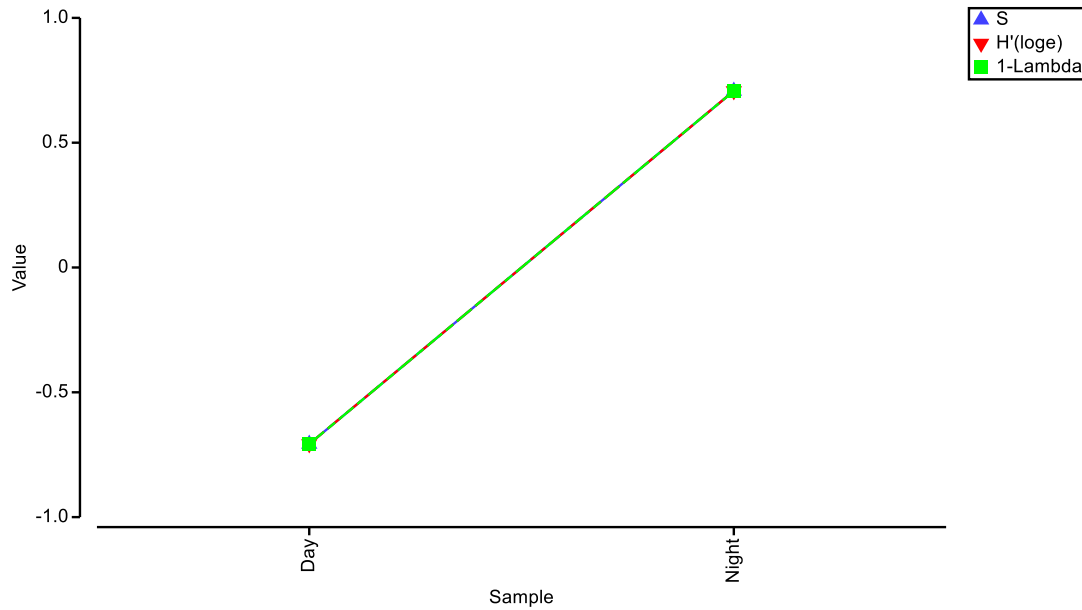


Figure 13-7. Overall indices of diversity for day versus night sampling based on numerical abundance.

S is the total number of species, H' is the Shannon Index of Diversity, and 1-Lambda is Simpson's Index of Diversity. Values have been normalized to allow simultaneous display of multiple diversity indices. Absolute values are given in Table 13-9.

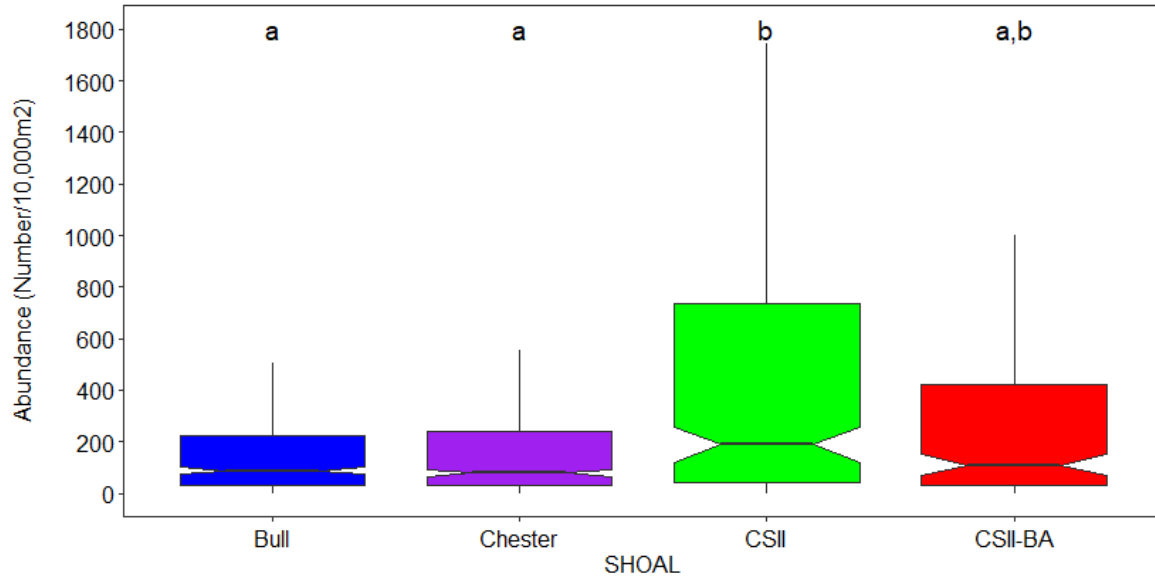


Figure 13-8. Fish abundance (number per 10,000 m²) among shoals.

Boxplot notch is the median value, box ends are the 25th and 75th percentiles, and the whiskers represent the lower and upper bounds of the first and fourth quartiles; individual outliers are not plotted. Shoals with the same letter were not significantly different ($P > 0.05$), and unlike letters indicate significant difference at $P \leq 0.05$.

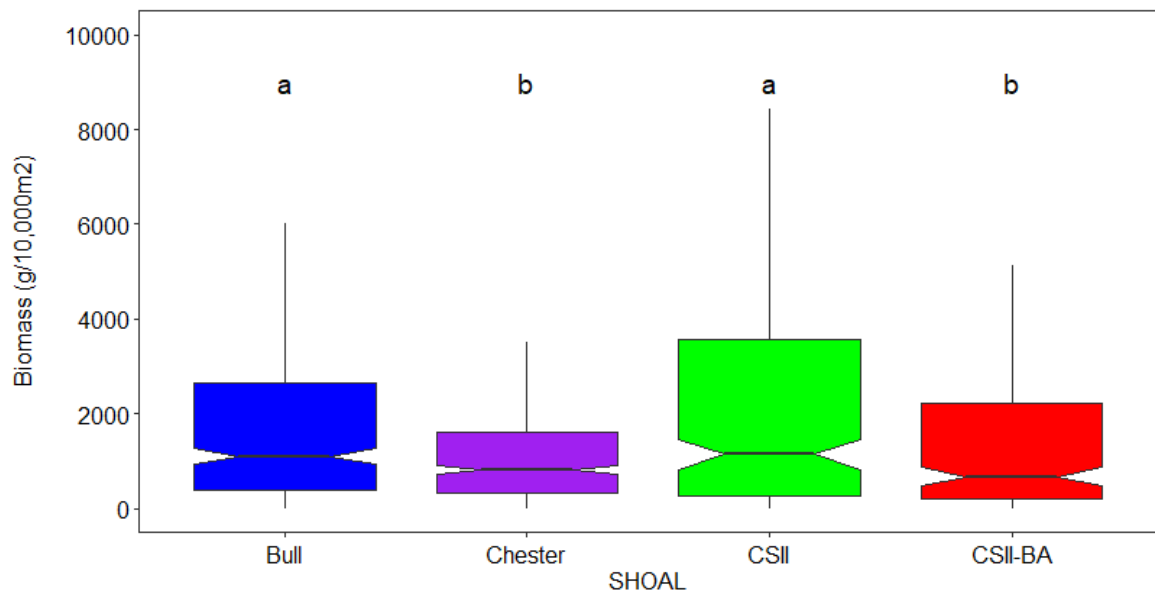


Figure 13-9. Fish biomass (g per 10,000 m²) among shoals.

Boxplot notch is the median value, box ends are the 25th and 75th percentiles, and the whiskers represent the lower and upper bounds of the first and fourth quartiles; individual outliers are not plotted. Shoals with the same letter were not significantly different ($P > 0.05$), and unlike letters indicate significant difference at $P \leq 0.05$.

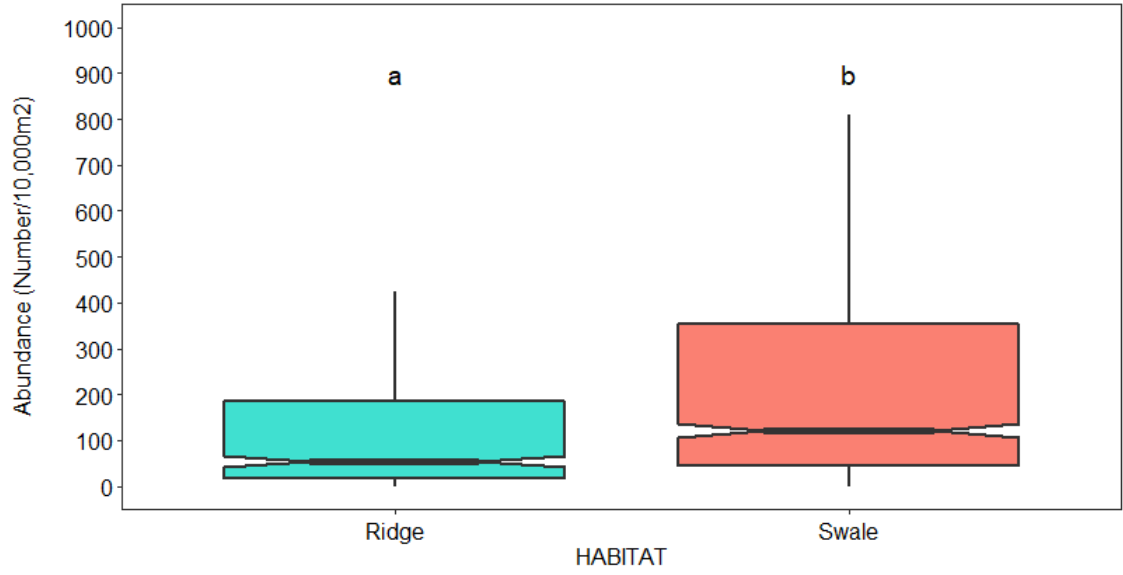


Figure 13-10. Fish abundance (number per 10,000 m²) between ridge and swale habitats. Boxplot notch is the median value, box ends are the 25th and 75th percentiles, and the whiskers represent the lower and upper bounds of the first and fourth quartiles; individual outliers are not plotted. Shoals with the same letter were not significantly different ($P > 0.05$), and unlike letters indicate significant difference at $P \leq 0.05$.

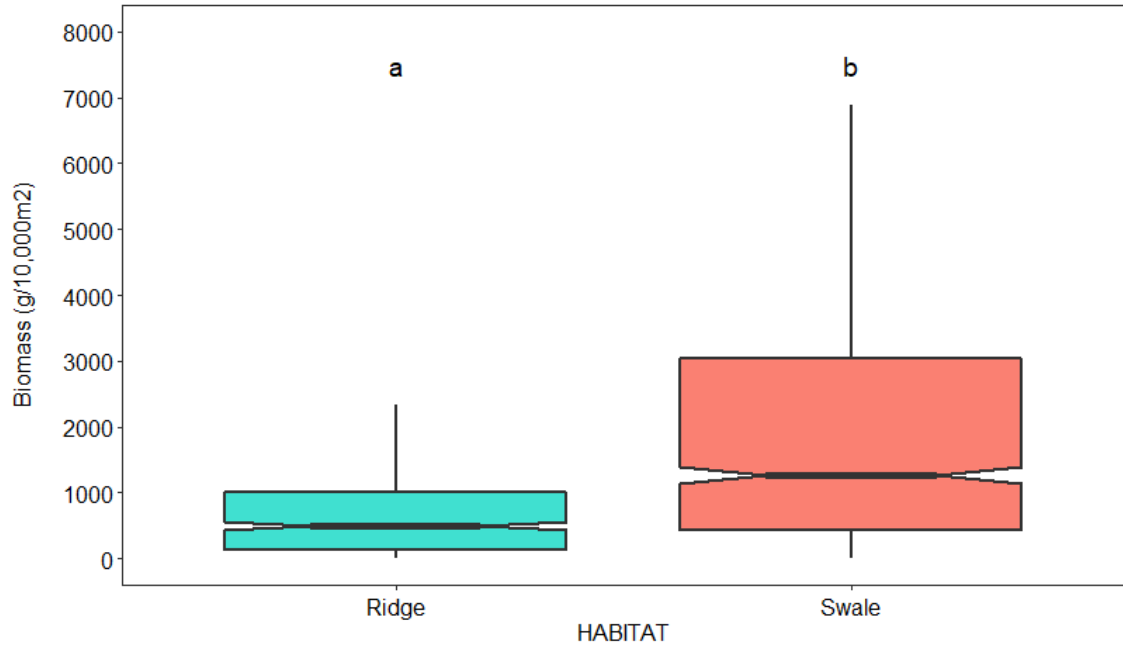


Figure 13-11. Fish biomass (g per 10,000 m²) between ridge and swale habitats. Boxplot notch is the median value, box ends are the 25th and 75th percentiles, and the whiskers represent the lower and upper bounds of the first and fourth quartiles; individual outliers are not plotted. Shoals with the same letter were not significantly different ($P > 0.05$), and unlike letters indicate significant difference at $P \leq 0.05$.

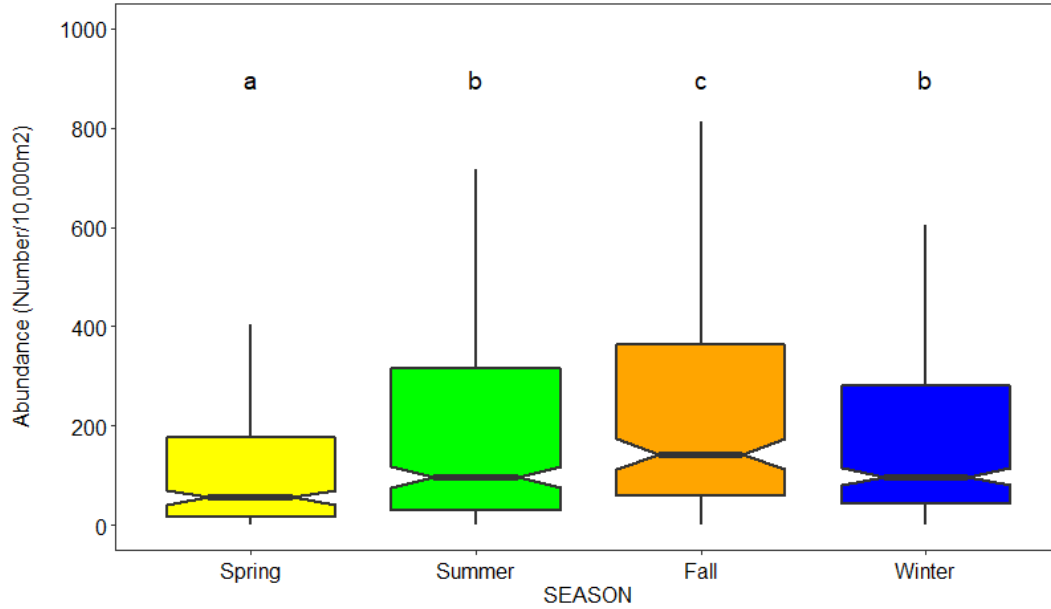


Figure 13-12. Fish abundance (number per 10,000 m²) among seasons.

Boxplot notch is the median value, box ends are the 25th and 75th percentiles, and the whiskers represent the lower and upper bounds of the first and fourth quartiles; individual outliers are not plotted. Shoals with the same letter were not significantly different ($P > 0.05$), and unlike letters indicate significant difference at $P \leq 0.05$.

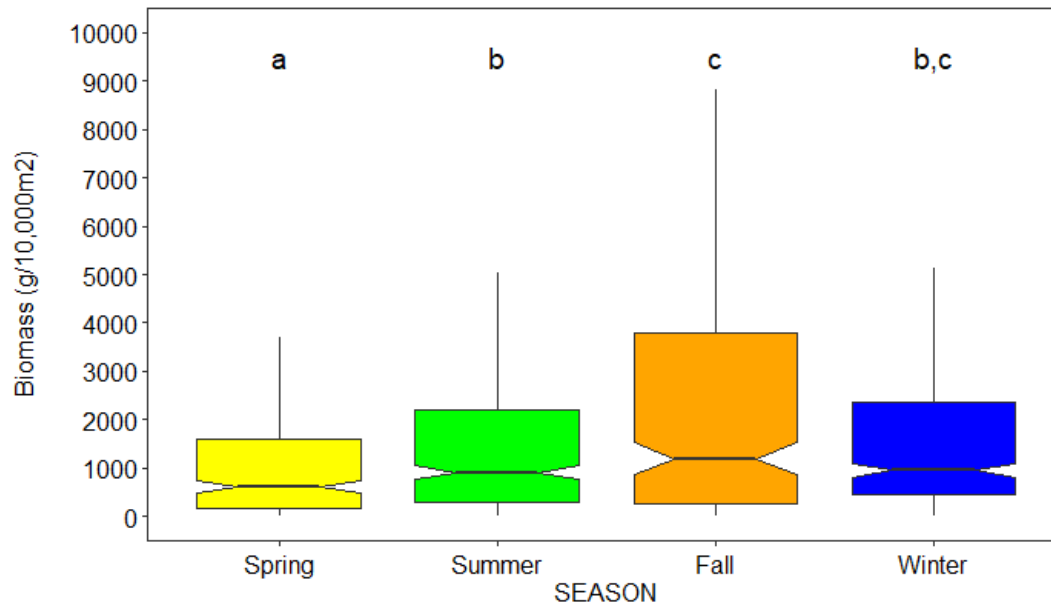


Figure 13-13. Fish biomass (g per 10,000 m²) among seasons.

Boxplot notch is the median value, box ends are the 25th and 75th percentiles, and the whiskers represent the lower and upper bounds of the first and fourth quartiles; individual outliers are not plotted. Shoals with the same letter were not significantly different ($P > 0.05$), and unlike letters indicate significant difference at $P \leq 0.05$.

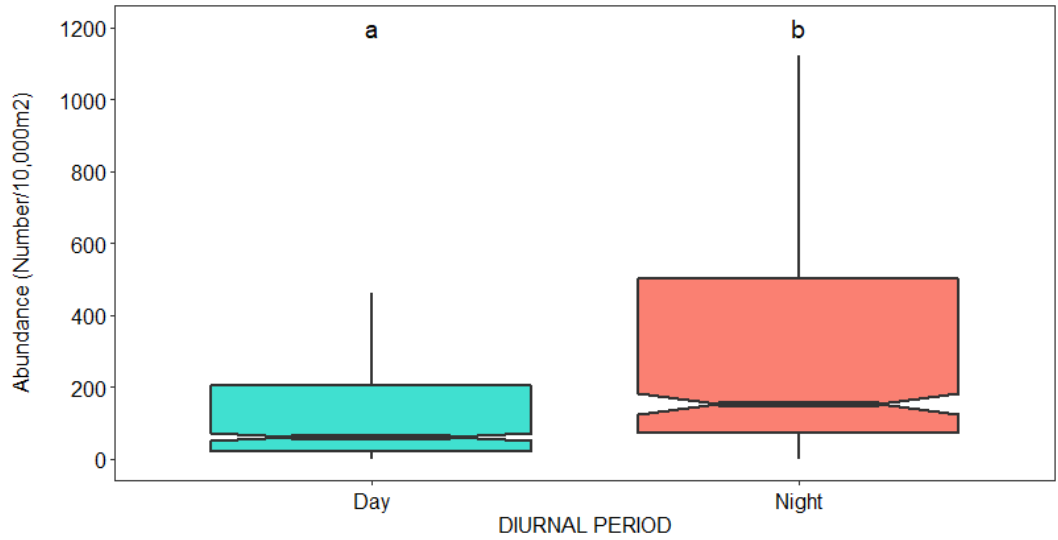


Figure 13-14. Fish abundance (number per 10,000 m²) between day and night periods.

Boxplot notch is the median value, box ends are the 25th and 75th percentiles, and the whiskers represent the lower and upper bounds of the first and fourth quartiles; individual outliers are not plotted. Shoals with the same letter were not significantly different ($P > 0.05$), and unlike letters indicate significant difference at $P \leq 0.05$.

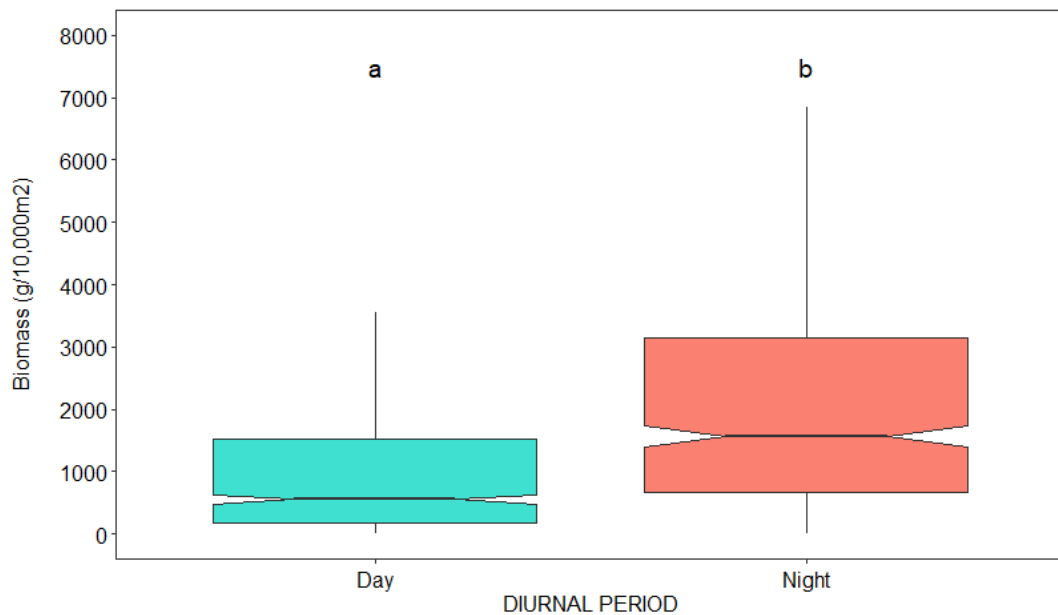


Figure 13-15. Fish biomass (g per 10,000 m²) between day and night periods.

Boxplot notch is the median value, box ends are the 25th and 75th percentiles, and the whiskers represent the lower and upper bounds of the first and fourth quartiles; individual outliers are not plotted. Shoals with the same letter were not significantly different ($P > 0.05$), and unlike letters indicate significant difference at $P \leq 0.05$.

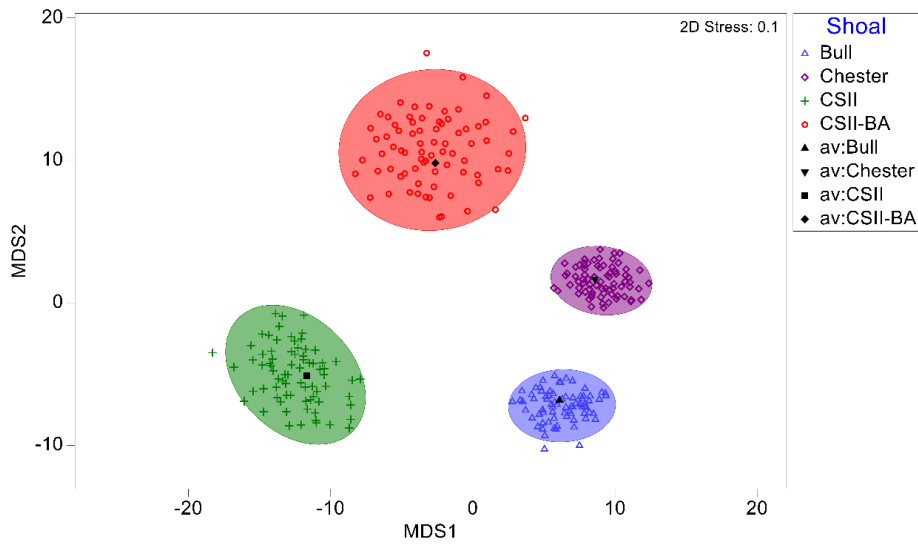


Figure 13-16. Metric-MDS showing position of shoals in multidimensional space based on species abundance of fish assemblages.

Bootstrapped averages with centroids (black symbols) and 95% confidence regions (ovals) are given for each shoal. Non-overlapping confidence regions indicate significant differences among shoals.

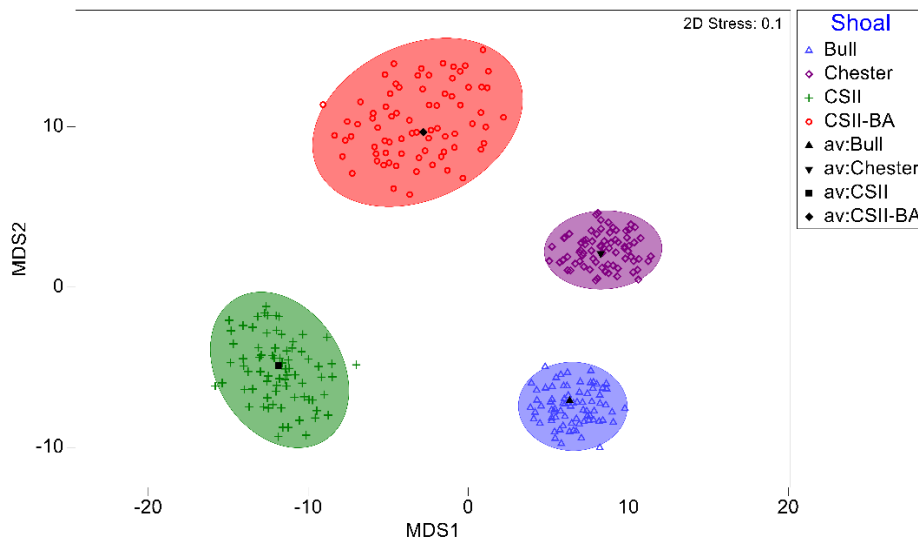


Figure 13-17. Metric-MDS showing position of shoals in multidimensional space based on species biomass of fish assemblages.

Bootstrapped averages with centroids (black symbols) and 95% confidence regions (ovals) are given for each shoal. Non-overlapping confidence regions indicate significant differences among shoals.

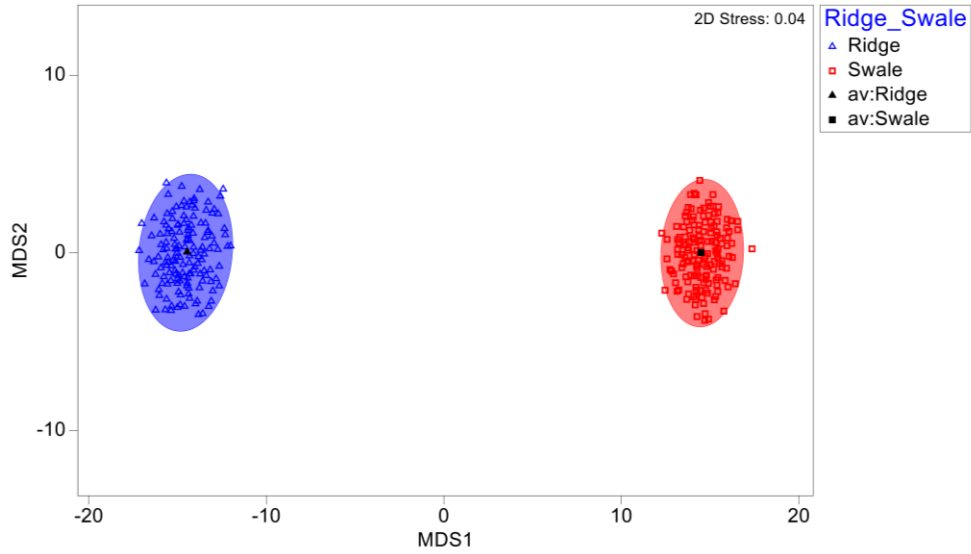


Figure 13-18. Metric-MDS showing position of ridge versus swale habitats in multidimensional space based on species abundance of fish assemblages.

Bootstrapped averages with centroids (black symbols) and 95% confidence regions (ovals) are given for each habitat. Non-overlapping confidence regions indicate significant differences between ridges and swales.

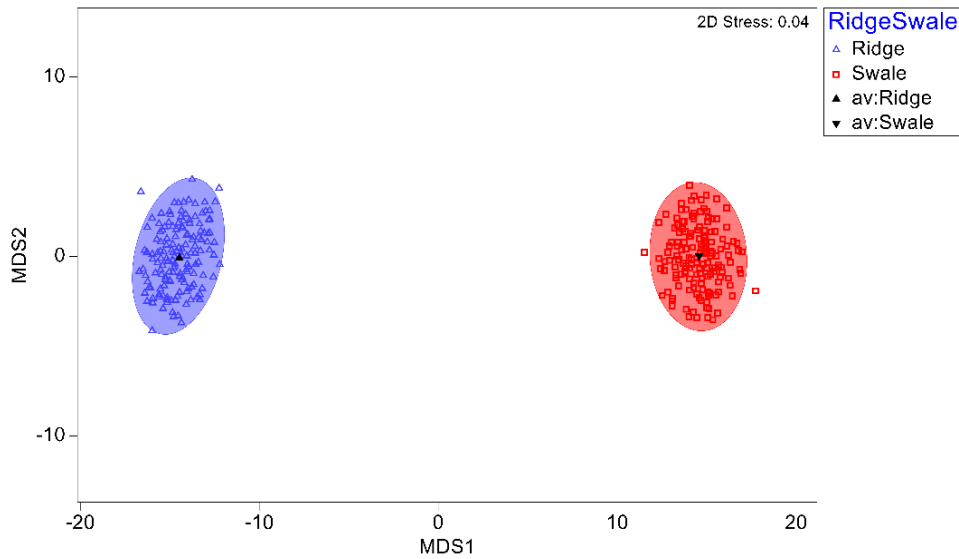


Figure 13-19. Metric-MDS showing position of ridge versus swale habitats in multidimensional space based on species biomass of fish assemblages.

Bootstrapped averages with centroids (black symbols) and 95% confidence regions (ovals) are given for each habitat. Non-overlapping confidence regions indicate significant differences between ridges and swales.

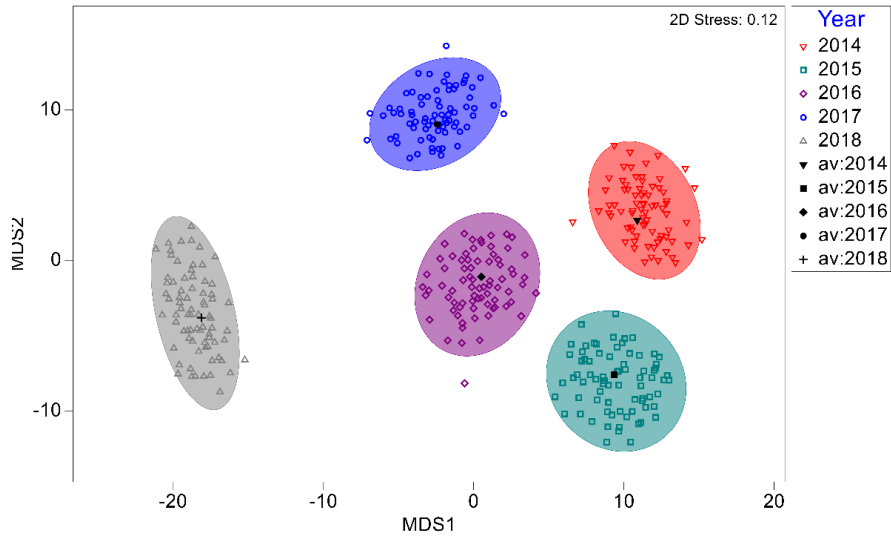


Figure 13-20. Metric-MDS showing position of sampling years in multidimensional space based on species abundance of fish assemblages.

Bootstrapped averages with centroids (black symbols) and 95% confidence regions (ovals) are given for each year. Non-overlapping confidence regions indicate significant differences among years.

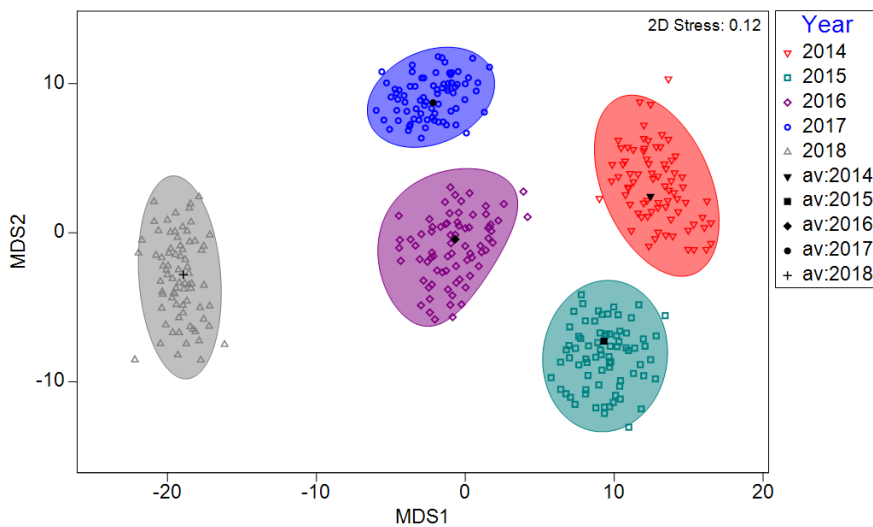


Figure 13-21. Metric-MDS showing position of sampling years in multidimensional space based on species biomass of fish assemblages.

Bootstrapped averages with centroids (black symbols) and 95% confidence regions (ovals) are given for each year. Non-overlapping confidence regions indicate significant differences among years.

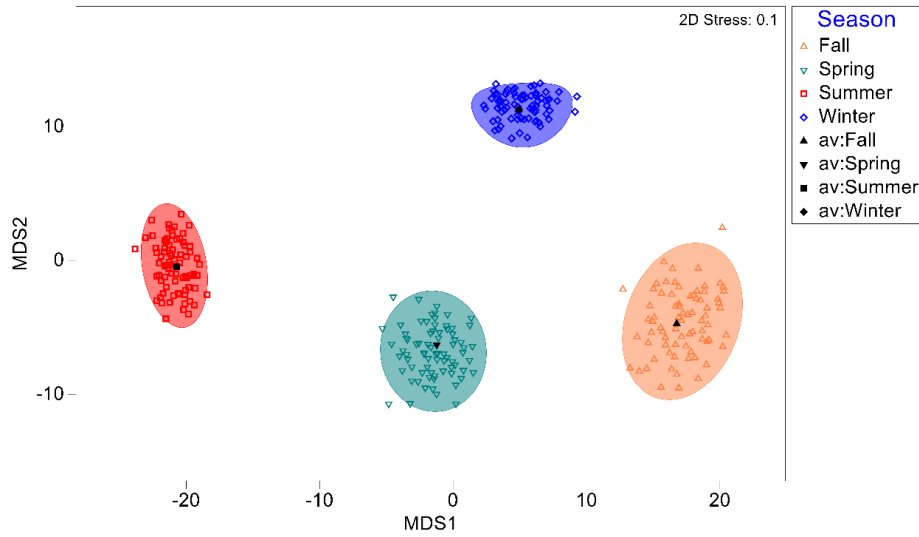


Figure 13-22. Metric-MDS showing positions among seasons in multidimensional space based on species abundance of fish assemblages.

Bootstrapped averages with centroids (black symbols) and 95% confidence regions (ovals) are given for each season. Non-overlapping confidence regions indicate significant differences among seasons.

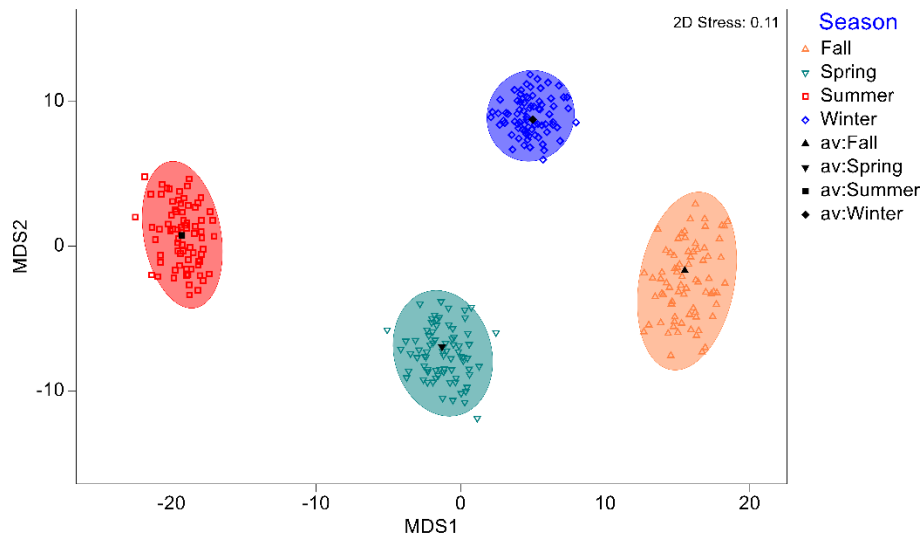


Figure 13-23. Metric-MDS showing positions among seasons in multidimensional space based on species biomass of fish assemblages.

Bootstrapped averages with centroids (black symbols) and 95% confidence regions (ovals) are given for each season. Non-overlapping confidence regions indicate significant differences among seasons.

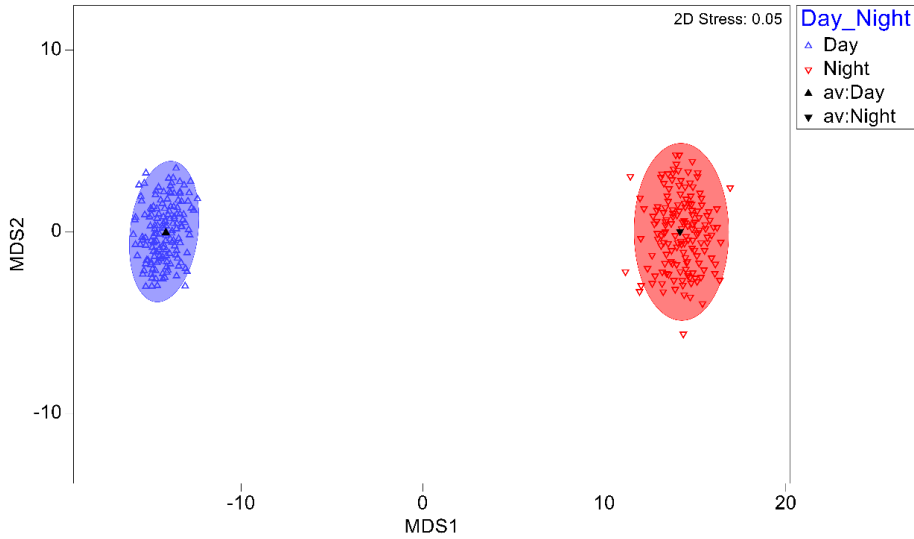


Figure 13-24. Metric-MDS showing positions of day versus night fish assemblages in multidimensional space based on species abundance.

Bootstrapped averages with centroids (black symbols) and 95% confidence regions (ovals) are given for day and night. Non-overlapping confidence regions indicate significant differences between day and night.

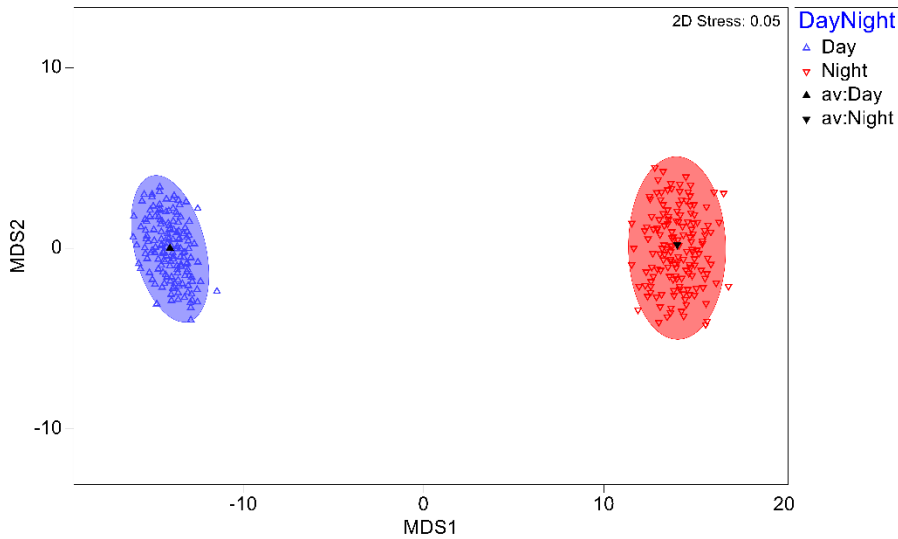


Figure 13-25. Metric-MDS showing position of day versus night fish assemblages in multidimensional space based on species biomass.

Bootstrapped averages with centroids (black symbols) and 95% confidence regions (ovals) are given for day and night. Non-overlapping confidence regions indicate significant differences between day and night.

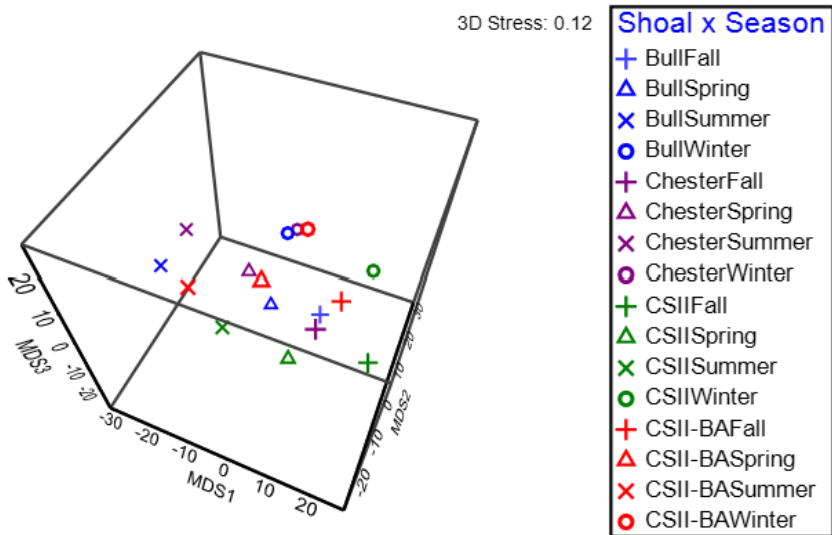


Figure 13-26. 3D metric-MDS showing relational differences in centroids among fish assemblages for combinations of Shoal and Season based on abundance.

Clusters of symbols relate to Season (type of symbol) versus Shoal (color of symbol).

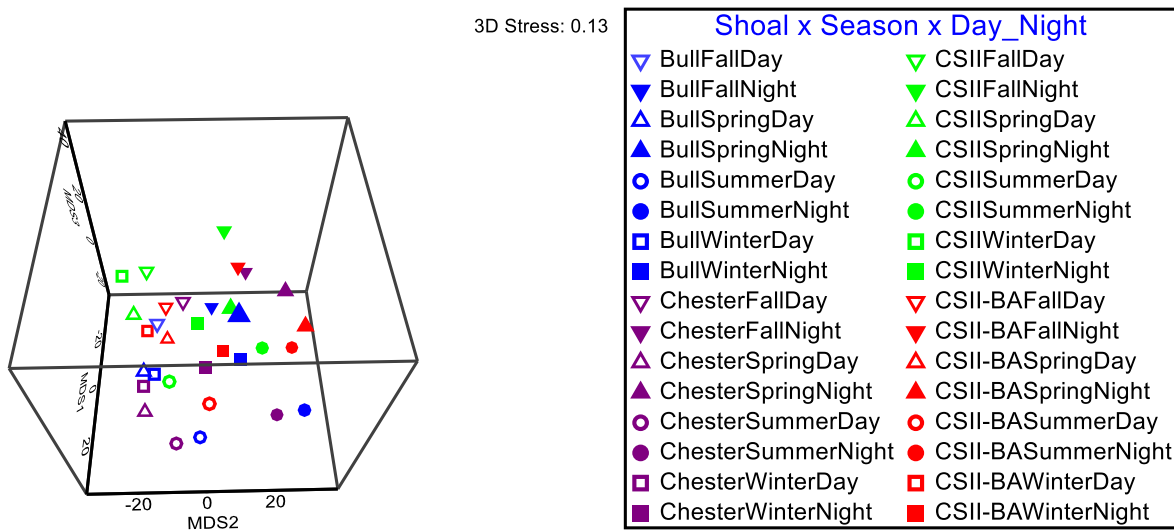


Figure 13-27. 3D metric-MDS showing relational differences in centroids among fish assemblages for combinations of Shoal, Season, and Day_Night based on abundance.

Clusters of symbols relate to Day_Night (open vs closed symbols), Season (type of symbol), and Shoal (color of symbol).

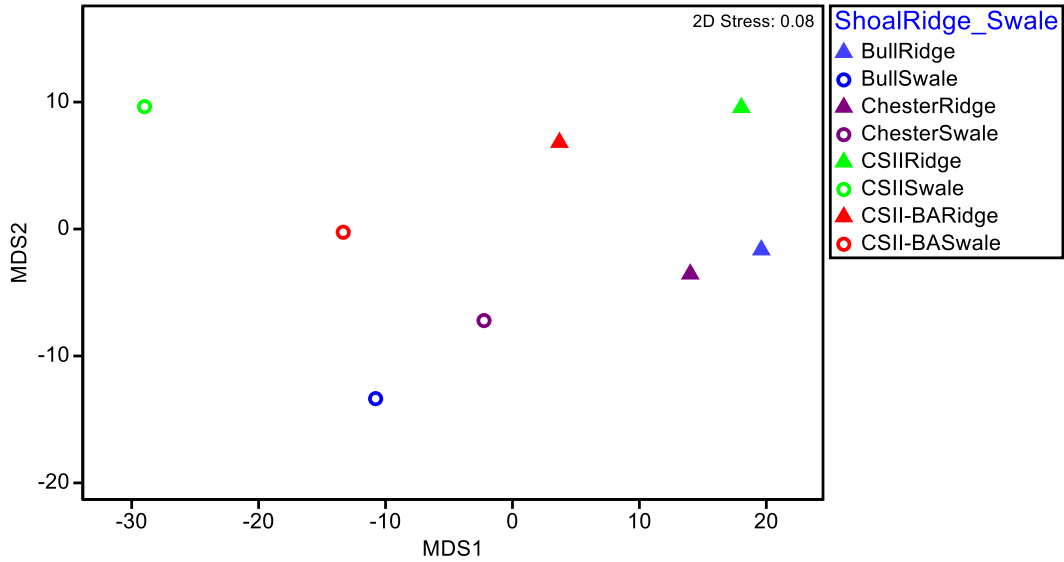


Figure 13-28. 2D metric-MDS showing relational differences in centroids among combinations of Shoal and Ridge-Swale habitats based on species abundance.

Clusters of symbols relate to Ridge_Swale (solid versus open symbols) and Shoal (symbol color).

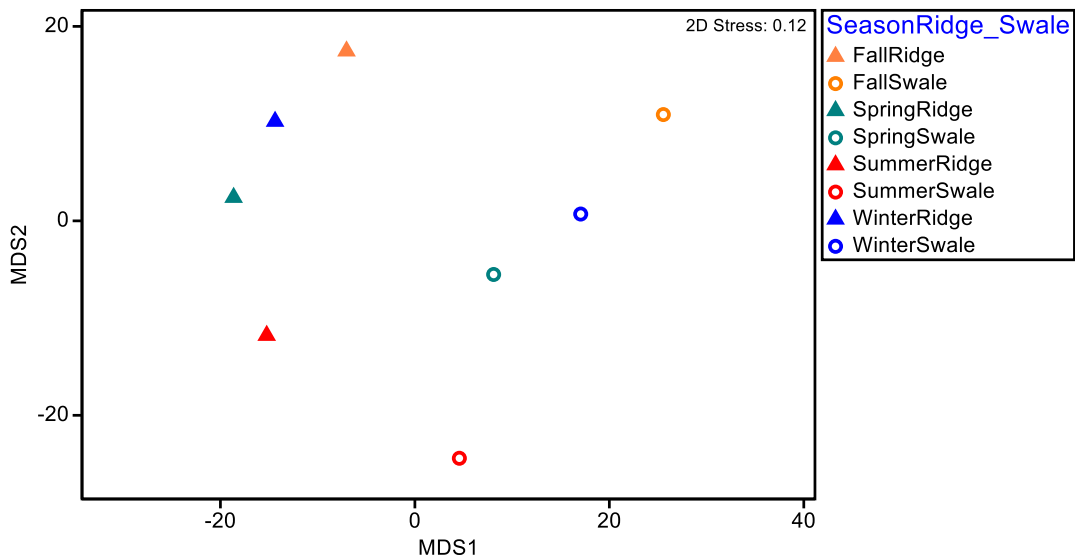


Figure 13-29. 2D metric-MDS showing relational differences in centroids among combinations of Season and Ridge-Swale habitats based on species abundance.

Clusters of symbols relate to Ridge_Swale (solid versus open symbols) and Season (symbol color).

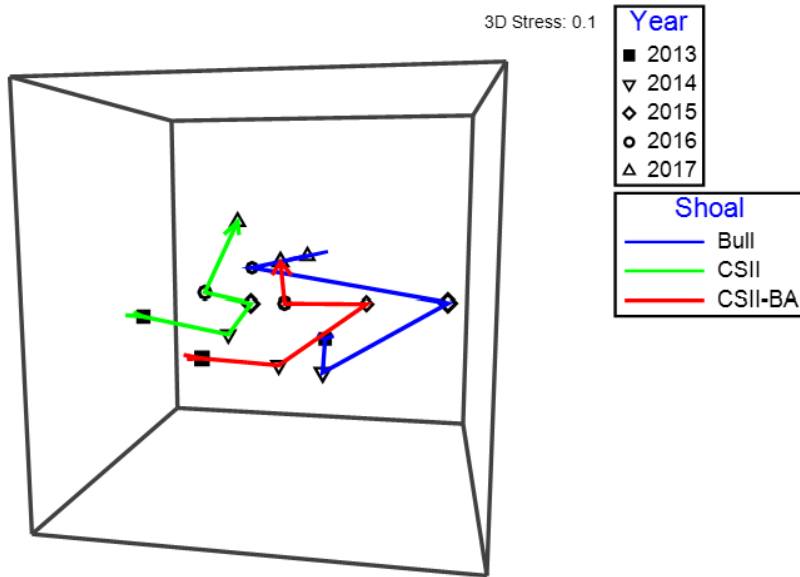


Figure 13-30. 3D nMDS showing relational differences in daytime fish assemblages during fall 2013 (solid square, pre-dredge period) relative to fall 2014–2017 (open symbols, post-dredge periods) for CSII, CSII-BA, and Bull Shoals.

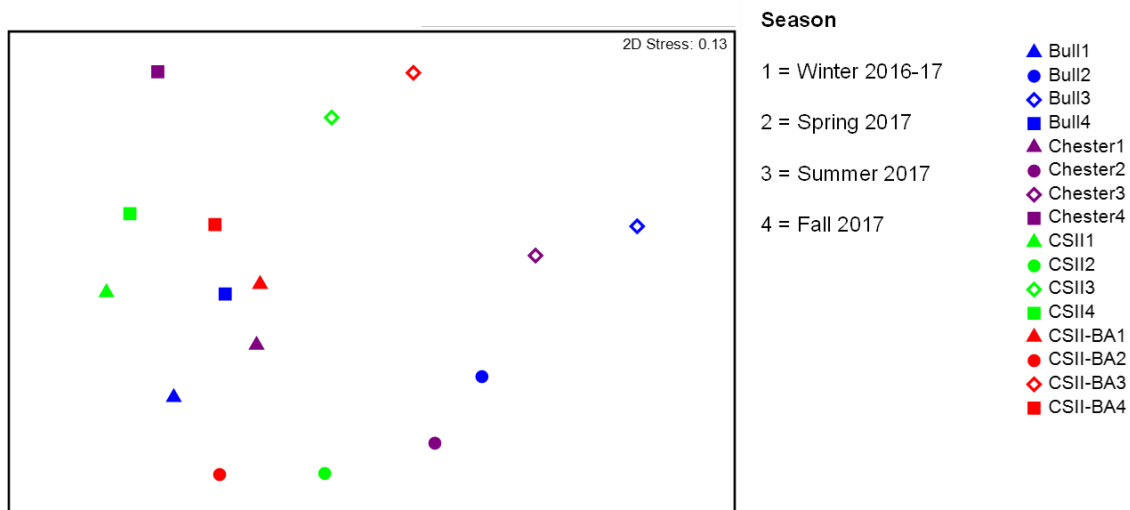


Figure 13-31. 2D nMDS showing relational differences in daytime fish assemblages for four seasons (winter 2016/17, spring 2017, summer 2017, and fall 2017) in the year preceding the second dredging event.

Clusters of symbols relate to Season-Year (symbol type) more so than Shoal (symbol color).

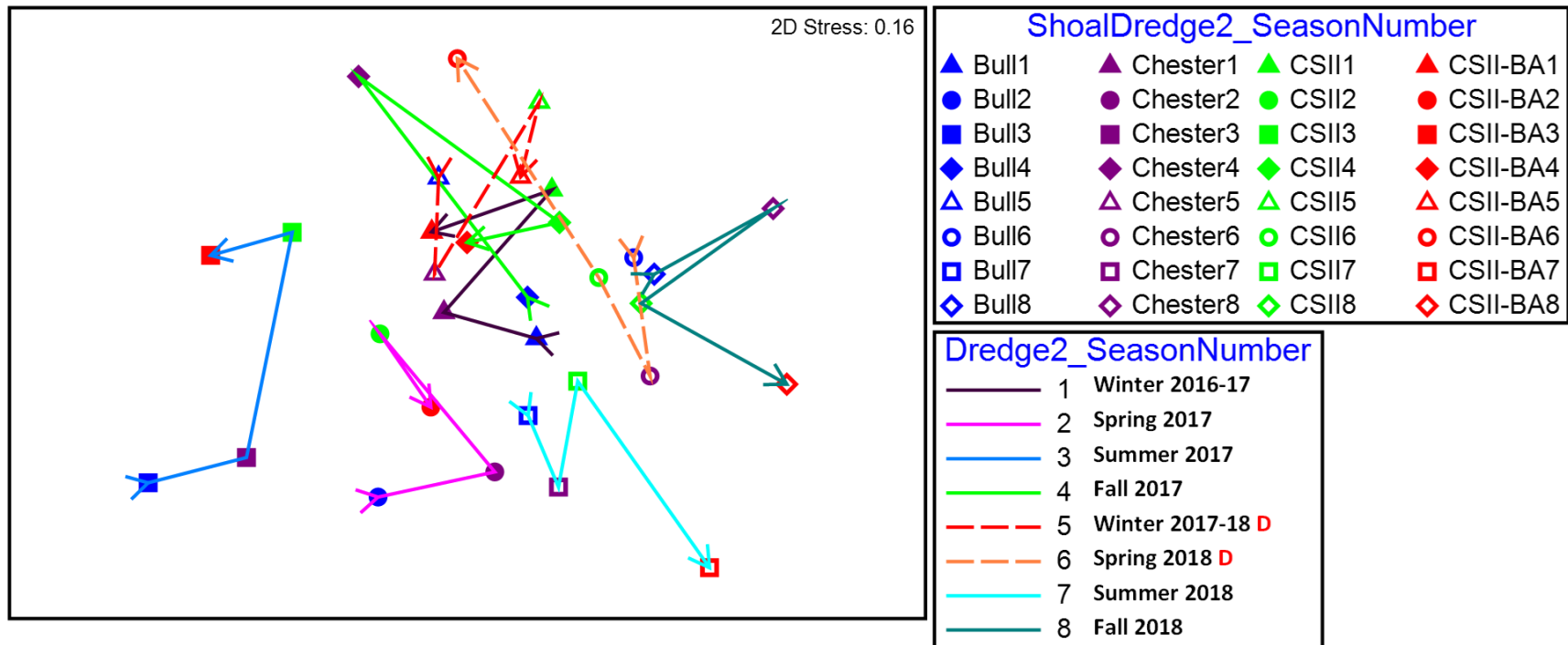


Figure 13-32. 2D nMDS showing relational differences in daytime fish assemblages for four seasons (winter 2016/17, spring 2017, summer 2017, and fall 2017) in the year preceding the second dredging event, and four seasons (winter 2017/18, spring 2018, summer 2018, and fall 2018) during (D) or after the Dredge-2 event.

Pre-dredge Season-Years are denoted by solid symbols and dredge/post dredge by open symbols. Similar line colors join similar Season-Years for all four shoals but should not be used to imply direction of differences. Season-Years where dredging occurred are denoted by dashed lines.

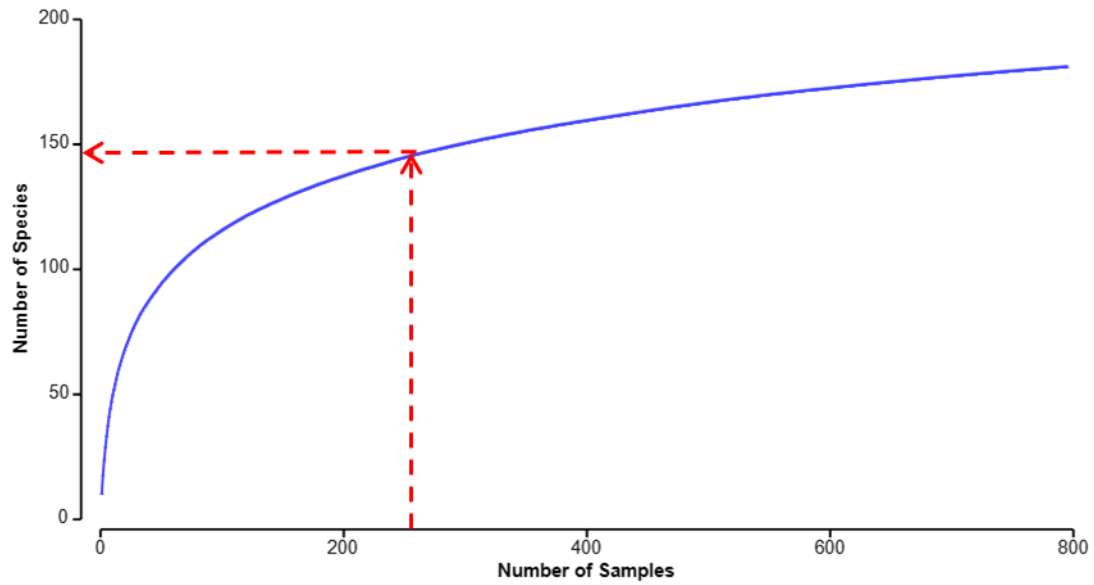


Figure 13-33. Species accumulation curve as a function of number of samples (trawls) for the present study.

Red dashed arrows indicate a sample size of 256 trawl tows for comparison to Slacum et al. (2010).

Table 13-1. Fish species representing > 1% of the numerical abundance on CSII-BA, CSII, Chester, and Bull Shoals.

Species	CSII-BA	CSII	Chester	Bull
Anchovy (<i>Anchoa</i> spp.)	5.35	10.17	36.79	20.61
Atlantic Croaker	1.41	2.33	0.71	1.65
Banded Drum	0.89	0.96	0.90	1.65
Clupeidae (juvenile)	1.65	3.77	0.72	0.56
Clupeiformes (juvenile)	5.25	3.35	3.69	2.55
Cusk-eel (juvenile)	4.72	0.90	3.72	4.28
Dusky Anchovy	1.09	0.88	1.66	4.35
Fringed Flounder	0.33	1.31	0.34	0.81
Inshore Lizardfish	0.68	0.28	0.93	1.14
Leopard Searobin	2.52	0.85	3.04	4.48
Sciaenidae (juvenile)	49.55	52.45	8.55	25.46
Silver Seatrout	1.67	3.30	2.64	1.31
Spotted Whiff	1.78	0.77	1.24	1.77
Star Drum	0.54	2.39	0.53	0.29
Striped Anchovy	11.04	8.31	24.76	8.92
Twospot Flounder	0.03	0.04	0.35	1.62
Unidentified fish (larvae/juvenile)	4.23	0.61	1.51	2.51

Table 13-2. Fish species representing > 1% of the numerical abundance on ridge versus swale habitats.

Species	Ridge (%)	Swale (%)
Anchovy (<i>Anchoa</i> spp.)	23.51	20.83
Atlantic Croaker	0.33	2.05
Banded Drum	0.64	1.33
Clupeidae (juvenile)	2.13	1.41
Clupeiformes (juvenile)	3.48	3.48
Cusk-eel (juvenile)	3.94	2.80
Dusky Anchovy	2.53	1.82
Fringed Flounder	0.12	1.04
Inshore Lizardfish	1.07	0.62
Ladyfish/Malacho (<i>leptocephalus</i>)	1.14	0.47
Leopard Searobin	3.70	2.24
Sciaenidae (juvenile)	27.84	30.70
Silver Seatrout	0.86	3.13
Spotted Whiff	1.26	1.32
Star Drum	0.16	1.41
Striped Anchovy	15.87	13.97
Unidentified fish (larvae/juvenile)	2.32	1.55

Table 13-3. Fish species representing > 1% of the numerical abundance during fall, spring, summer and winter.

Species	Fall (%)	Spring (%)	Summer (%)	Winter (%)
Anchovy (<i>Anchoa</i> spp.)	5.32	23.73	44.79	0.03
Atlantic Croaker	1.83	1.96	0.65	2.14
Atlantic Cutlassfish	2.09	1.13	0.00	0.34
Banded Drum	2.96	1.77	0.42	0.49
Clupeidae (juvenile)	0.01	6.96	0.79	1.03
Clupeiformes (juvenile)	0.62	15.17	2.82	0.02
Cusk-eel (juvenile)	0.53	0.10	6.83	1.56
Dusky Anchovy	1.68	0.72	4.20	0.12
Inshore Lizardfish	0.18	1.13	1.05	0.58
Kingfish (<i>Menticirrhus</i> spp. juvenile/unidentified)	0.41	0.12	0.49	1.06
Kingfish spp.	1.00	1.03	0.33	0.62
Leopard Searobin	2.01	1.97	3.82	2.11
Sciaenidae (juvenile)	34.75	6.77	2.08	75.96
Searobin (<i>Prionotus</i> spp. juvenile)	0.29	1.46	0.54	0.30
Seminole Goby	0.01	0.46	1.51	0.00
Silver Seatrout	9.07	0.98	0.16	1.85
Spotted Whiff	0.65	0.99	2.14	0.75
Star Drum	3.95	0.10	0.21	0.65
Striped Anchovy	26.18	24.15	15.25	1.29
Twospot Flounder	0.12	0.01	1.32	0.05
Unidentified fish (larvae/juvenile)	0.56	2.33	1.54	2.68

Table 13-4. Fish species representing > 1% of the numerical abundance of day versus night trawling on all shoals.

Common Name	Day (%)	Night (%)
Anchovy (<i>Anchoa</i> spp.)	9.89	30.80
Atlantic Croaker	1.67	1.34
Atlantic Cutlassfish	1.46	0.02
Banded Drum	0.76	1.37
Clupeidae (juvenile)	1.86	1.48
Clupeiformes (juvenile)	4.19	2.94
Cusk-eel (juvenile)	0.01	5.60
Dusky Anchovy	2.57	1.66
Inshore Lizardfish	1.06	0.55
Ladyfish/Malacho (<i>leptocephalus</i>)	1.24	0.27
Leopard Searobin	1.89	3.37
Sciaenidae (juvenile)	46.71	16.73
Silver Seatrout	3.56	1.48
Spotted Whiff	1.05	1.49
Star Drum	1.13	0.90
Striped Anchovy	11.34	17.10
Unidentified fish (larvae/juvenile)	3.02	0.88

Table 13-5. Fish species representing > 1% of the total biomass on CSII-BA, CSII, Chester, and Bull Shoals.

Species	CSII-BA	CSII	Chester	Bull
Anchovy (<i>Anchoa</i> spp.)	0.15	0.20	1.58	0.40
Atlantic Bumper	1.05	0.43	0.06	0.08
Atlantic Croaker	26.44	34.24	18.91	19.69
Atlantic Cutlassfish	0.69	0.19	1.73	0.57
Atlantic Spadefish	1.13	0.42	1.77	0.88
Banded Drum	7.85	6.27	9.85	14.75
Blackcheek Tonguefish	0.08	1.13	0.47	0.59
Blotched Cusk-eel	0.64	0.22	1.62	1.95
Bullnose Eagle Ray	1.66	0.00	0.00	0.00
Cownose Ray	1.17	1.53	0.00	0.00
Cusk-eel (juvenile)	0.53	0.15	1.16	0.50
Dusky Flounder	0.50	0.05	1.87	1.88
Fringed Flounder	1.67	4.52	1.74	2.28
Hardhead Catfish	1.44	0.76	0.00	0.35
Inshore Lizardfish	4.71	2.21	11.40	7.18
Leopard Searobin	16.93	5.46	19.85	14.19
Lesser Electric Ray	0.52	1.87	0.11	0.18
Pigfish	0.90	0.59	0.14	1.46
Rock Sea Bass	0.39	1.24	0.77	1.71
Sand Perch	0.29	0.21	1.17	2.17
Silver Seatrout	5.59	7.61	5.59	4.02
Smooth Butterfly Ray	1.22	5.32	0.14	1.94
Southern Kingfish	0.36	4.74	0.41	0.69
Spot	3.96	3.88	1.95	2.12
Spotted Whiff	5.32	2.02	4.39	3.86
Star Drum	3.44	6.01	0.88	0.36
Striped Anchovy	1.06	0.92	1.41	0.36
Striped Cusk-eel	1.39	0.36	0.70	0.31
Twospot Flounder	0.05	0.06	0.50	1.56
Weakfish	0.66	0.32	0.64	1.10

Table 13-6. Fish species representing > 1% of the total biomass in ridge versus swale sampling.

Common Name	Ridge (%)	Swale (%)
Atlantic Croaker	9.20	28.52
Atlantic Spadefish	1.36	0.91
Banded Drum	9.61	11.00
Blotched Cusk-eel	2.13	1.04
Cusk-eel (juvenile)	1.24	0.38
Dusky Flounder	2.09	1.03
Fringed Flounder	0.68	3.30
Inshore Lizardfish	11.70	5.22
Leopard Searobin	28.05	9.06
Northern Searobin	1.67	0.24
Rock Sea Bass	0.43	1.48
Sand Perch	0.61	1.44
Silver Seatrout	2.80	6.36
Smallmouth Flounder	1.37	0.12
Southern Kingfish	0.22	2.09
Spot	0.83	3.29
Spotted Whiff	6.01	2.90
Star Drum	0.79	2.68
Striped Anchovy	1.33	0.69
Striped Cusk-eel	1.05	0.35
Twospot Flounder	2.54	0.19

Table 13-7. Fish species representing > 1% of the total biomass in fall, spring, summer, and winter sampling.

Common Name	Spring (%)	Summer (%)	Fall (%)	Winter (%)
Anchovy (<i>Anchoa</i> spp.)	0.44	2.05	0.07	0.00
Atlantic Croaker	32.83	16.83	20.11	29.67
Atlantic Cutlassfish	0.76	0.00	2.16	0.13
Atlantic Spadefish	0.31	0.02	1.78	1.52
Banded Drum	6.74	2.23	25.55	5.14
Blackcheek Tonguefish	0.61	0.37	0.39	1.21
Blackwing Searobin	0.15	1.09	0.15	0.30
Blotched Cusk-eel	0.20	2.99	0.65	0.96
Bluntnose Stingray	0.00	0.09	1.05	0.66
Cownose Ray	0.00	0.00	0.34	1.36
Cusk-eel (juvenile)	0.03	1.86	0.04	0.26
Dusky Flounder	1.45	2.39	0.73	0.75
Fringed Flounder	3.60	3.69	1.08	2.87
Gulf Flounder	1.54	0.27	0.10	0.45
Hardhead Catfish	0.00	1.58	0.11	0.00
Inshore Lizardfish	10.04	10.12	1.89	7.16
Leopard Searobin	13.80	19.99	6.51	15.13
Northern Searobin	0.29	1.71	0.10	0.18
Rock Sea Bass	0.93	2.05	1.15	0.69
Sand Perch	0.43	2.40	1.10	0.70
Shelf Flounder	0.35	1.83	0.10	0.14
Silver Seatrout	5.85	0.95	10.04	4.87
Smooth Butterfly Ray	0.63	1.18	5.50	0.78
Southern Kingfish	0.50	0.22	1.59	3.54
Southern Stingray	1.26	0.00	0.25	0.00
Spot	2.17	1.56	1.42	5.30
Spotted Whiff	3.06	6.11	1.87	3.51
Star Drum	0.63	1.57	3.74	2.06
Striped Anchovy	0.96	1.27	0.82	0.43
Striped Cusk-eel	0.62	1.16	0.10	0.32
Striped Searobin	0.14	0.02	0.29	0.14
Twospot Flounder	0.01	2.36	0.21	0.22
Weakfish	0.00	0.00	1.90	0.58

Table 13-8. Fish species representing > 1% of the total biomass in day versus night samples.

Common Name	% Day	% Night
Atlantic Croaker	24.82	23.02
Atlantic Cutlassfish	1.70	0.06
Atlantic Spadefish	0.89	1.13
Banded Drum	5.04	15.21
Blotched Cusk-eel	0.00	2.35
Cownose Ray	1.11	0.00
Cusk-eel (juvenile)	0.00	1.07
Dusky Flounder	1.61	1.02
Fringed Flounder	3.17	2.25
Inshore Lizardfish	10.56	3.76
Leopard Seabob	10.33	16.38
Pigfish	1.22	0.52
Rock Sea Bass	1.17	1.27
Sand Perch	1.17	1.30
Silver Seatrout	6.72	4.51
Smooth Butterfly Ray	4.04	0.85
Southern Kingfish	2.26	1.12
Spot	3.57	1.98
Spotted Whiff	3.33	3.91
Star Drum	2.71	1.83
Striped Anchovy	0.60	1.04
Twospot Flounder	0.26	1.17

Table 13-9. Average indices of diversity metrics based on among shoals, among seasons, between ridge and swale habitats, and between day and night sampling periods.

Comparison	Item	Total Species (S)	Simpson (1 - λ)	Shannon (H')
Among Shoals	Bull	150	0.875	2.841
"	Chester	134	0.790	2.255
"	CSII	117	0.702	2.048
"	CSII-BA	118	0.730	2.150
Ridge-Swale	Ridge	140	0.835	2.430
"	Swale	168	0.837	2.518
Among Seasons	Spring	105	0.850	2.458
"	Summer	149	0.765	2.312
"	Fall	106	0.795	2.219
"	Winter	111	0.420	1.327
Day-Night	Day	143	0.753	2.213
"	Night	158	0.841	2.536

Table 13-10. Pairwise two-sided multiple comparison analysis using the Dwass, Steel, Critchlow-Fligner (DSCF) method for median abundance (individuals/10,000 m²) among shoals and among seasons.

Test	Comparison	Test Value	P
Among Shoals	Bull vs. CSII	6.572	< 0.0001
"	Bull vs. CSII_BA	2.628	0.2462
"	Bull vs. Chester	0.927	0.9136
"	CSII vs. CSII_BA	3.302	0.0903
"	CSII vs. Chester	6.905	< 0.0001
"	CSII_BA vs. Chester	3.075	0.1304
Among Seasons	Fall vs. Spring	8.816	< 0.0001
"	Fall vs. Summer	4.064	0.0211
"	Fall vs. Winter	3.658	0.0477
"	Spring vs. Summer	5.507	0.0006
"	Spring vs. Winter	7.276	< 0.0001
"	Summer vs. Winter	1.259	0.8099

Table 13-11. Pairwise two-sided multiple comparison analysis using the Dwass, Steel, Critchlow-Fligner (DSCF) method for median biomass (g/10,000 m²) among shoals and among seasons.

Test	Comparison	Test Value	P
Among Shoals	Bull vs. CSII	0.9896	0.8972
"	Bull vs. CSII-BA	4.1747	0.0167
"	Bull vs. Chester	4.6064	0.0062
"	CSII vs. CSII-BA	4.0164	0.0234
"	CSII vs. Chester	3.9462	0.0270
"	CSII-BA vs. Chester	1.0568	0.8779
Among Seasons	Fall vs. Spring	6.7265	< 0.0001
"	Fall vs. Summer	3.9138	0.0289
"	Fall vs. Winter	1.9776	0.5003
"	Spring vs. Summer	4.3057	0.0125
"	Spring vs. Winter	6.9345	< 0.0001
"	Summer vs. Winter	2.6872	0.2279

Table 13-12. PERMANOVA pairwise comparisons of fish assemblages based on abundance among shoals.

P(perm) is the permutation P-value.

Groups	t	P(perm)	Unique perms
Bull, Chester	2.3329	0.001	996
Bull, CSII	3.5643	0.001	999
Bull, CSII-BA	2.5245	0.001	999
Chester, CSII	3.9704	0.001	998
Chester, CSII-BA	2.256	0.001	999
CSII, CSII-BA	2.2939	0.001	998

Table 13-13. SIMPER results showing the fish species that explained at least 50% of the dissimilarity between pairs of shoals based on species abundances.

Shoal Comparisons	Average Dissimilarity (%)	Species	Contribution to Dissimilarity (%)
Bull and CSII	84.82	Leopard Searobin	4.78
"	"	Inshore Lizardfish	4.74
"	"	Fringed Flounder	4.26
"	"	Spotted Whiff	4.06
"	"	Atlantic Croaker	4.05
"	"	Sciaenidae (juvenile)	3.45
"	"	Banded Drum	2.43
"	"	Silver Seatrout	2.42
"	"	Unidentified fish (larvae/juvenile)	2.21
"	"	Spot	2.10
"	"	Striped Anchovy	2.03
"	"	Southern Kingfish	1.97
"	"	Kingfish spp.	1.96
"	"	Rock Sea Bass	1.87
"	"	Northern Searobin	1.85
"	"	Blackcheek Tonguefish	1.66
"	"	Searobin (<i>Prionotus</i> spp. juvenile)	1.60
"	"	Seminole Goby	1.57
"	"	Ladyfish/Malacho (leptocephalus)	1.56
Chester and CSII	83.18	Inshore Lizardfish	5.96
"	"	Leopard Searobin	5.40
"	"	Spotted Whiff	4.82
"	"	Fringed Flounder	4.36
"	"	Atlantic Croaker	4.13
"	"	Sciaenidae (juvenile)	3.54
"	"	Silver Seatrout	2.75
"	"	Unidentified fish (larvae/juvenile)	2.56
"	"	Banded Drum	2.54
"	"	Kingfish spp.	2.45
"	"	Striped Anchovy	2.42
"	"	Southern Kingfish	2.18
"	"	Spot	2.06
"	"	Ladyfish/Malacho (leptocephalus)	1.93
"	"	Cusk-eel (juvenile)	1.83
"	"	Blackcheek Tonguefish	1.59
Bull and Chester	80.91	Inshore Lizardfish	6.22
"	"	Leopard Searobin	5.24
"	"	Spotted Whiff	4.94

Shoal Comparisons	Average Dissimilarity (%)	Species	Contribution to Dissimilarity (%)
"	"	Atlantic Croaker	3.42
"	"	Fringed Flounder	3.40
"	"	Unidentified fish (larvae/juvenile)	2.74
"	"	Sciaenidae (juvenile)	2.52
"	"	Shelf Flounder	2.28
"	"	Banded Drum	2.24
"	"	Northern Searobin	2.10
"	"	Cusk-eel (juvenile)	2.05
"	"	Seminole Goby	1.99
"	"	Dusky Flounder	1.97
"	"	Striped Anchovy	1.97
"	"	Silver Seatrout	1.96
"	"	Smallmouth Flounder	1.92
"	"	Rock Sea Bass	1.76
"	"	Sand Perch	1.75
Bull and CSII-BA	80.78	Inshore Lizardfish	4.79
"	"	Spotted Whiff	4.43
"	"	Leopard Searobin	4.29
"	"	Atlantic Croaker	3.76
"	"	Fringed Flounder	3.18
"	"	Sciaenidae (juvenile)	3.13
"	"	Unidentified fish (larvae/juvenile)	3.01
"	"	Northern Searobin	2.33
"	"	Banded Drum	2.28
"	"	Silver Seatrout	2.22
"	"	Cusk-eel (juvenile)	2.19
"	"	Kingfish spp.	1.98
"	"	Striped Anchovy	1.86
"	"	Searobin (<i>Prionotus</i> spp. juvenile)	1.74
"	"	Seminole Goby	1.72
"	"	Rock Sea Bass	1.67
"	"	Striped Cusk-eel	1.63
"	"	Dusky Flounder	1.55
"	"	Spot	1.53
"	"	Smallmouth Flounder	1.49
CSII and CSII-BA	80.41	Spotted Whiff	4.62
"	"	Leopard Searobin	4.57
"	"	Atlantic Croaker	4.34
"	"	Inshore Lizardfish	4.34
"	"	Fringed Flounder	4.08
"	"	Sciaenidae (juvenile)	3.73

Shoal Comparisons	Average Dissimilarity (%)	Species	Contribution to Dissimilarity (%)
"	"	Unidentified fish (larvae/juvenile)	2.93
"	"	Silver Seatrout	2.78
"	"	Kingfish spp.	2.58
"	"	Banded Drum	2.53
"	"	Southern Kingfish	2.27
"	"	Striped Anchovy	2.21
"	"	Spot	2.15
"	"	Cusk-eel (juvenile)	2.07
"	"	Northern Searobin	1.90
"	"	Striped Cusk-eel	1.81
"	"	Star Drum	1.75
Chester and CSII-BA	77.84	Inshore Lizardfish	5.73
"	"	Spotted Whiff	4.65
"	"	Leopard Searobin	4.58
"	"	Atlantic Croaker	3.88
"	"	Unidentified fish (larvae/juvenile)	3.36
"	"	Sciaenidae (juvenile)	3.29
"	"	Fringed Flounder	3.11
"	"	Silver Seatrout	2.59
"	"	Cusk-eel (juvenile)	2.52
"	"	Kingfish spp.	2.51
"	"	Banded Drum	2.43
"	"	Striped Anchovy	2.30
"	"	Northern Searobin	2.13
"	"	Striped Cusk-eel	2.06
"	"	Ladyfish/Malacho (<i>leptocephalus</i>)	1.81
"	"	Searobin (<i>Prionotus</i> spp. juvenile)	1.69
"	"	Atlantic Spadefish	1.60

Table 13-14. PERMANOVA pairwise comparisons of fish assemblages based on species biomass among shoals.

P(perm) is the permutation P-value.

Groups	t	P(perm)	Unique perms
Bull, Chester	2.2381	0.0001	9913
Bull, CSII	3.4438	0.0001	9902
Bull, CSII-BA	2.4723	0.0001	9907
Chester, CSII	4.0053	0.0001	9931
Chester, CSII-BA	2.2648	0.0001	9929
CSII, CSII-BA	2.2965	0.0001	9919

Table 13-15. SIMPER results showing the fish species that explained at least 50% of the dissimilarity between pairs of shoals based on species biomass.

Shoal Comparisons	Average Dissimilarity (%)	Species	Contribution to Dissimilarity (%)
Bull and CSII	84.90	Leopard Searobin	5.72
"	"	Spotted Whiff	4.87
"	"	Inshore Lizardfish	4.61
"	"	Fringed Flounder	4.61
"	"	Atlantic Croaker	4.16
"	"	Sciaenidae (juvenile)	2.99
"	"	Striped Anchovy	2.88
"	"	Silver Seatrout	2.46
"	"	Spot	2.25
"	"	Unidentified fish (larvae/juvenile)	2.2
"	"	Rock Sea Bass	1.79
"	"	Seminole Goby	1.77
"	"	Northern Searobin	1.76
"	"	Ladyfish/Malacho (leptocephalus)	1.74
"	"	Banded Drum	1.74
"	"	Southern Kingfish	1.72
"	"	Blackcheek Tonguefish	1.69
"	"	Searobin (Prionotus spp. juvenile)	1.68
Chester and CSII	83.06	Leopard Searobin	5.31
"	"	Inshore Lizardfish	4.87
"	"	Spotted Whiff	4.7
"	"	Fringed Flounder	3.91
"	"	Atlantic Croaker	3.57
"	"	Striped Anchovy	2.86
"	"	Sciaenidae (juvenile)	2.52

Shoal Comparisons	Average Dissimilarity (%)	Species	Contribution to Dissimilarity (%)
"	"	Silver Seatrout	2.22
"	"	Unidentified fish (larvae/juvenile)	2.15
"	"	Spot	1.86
"	"	Ladyfish/Malacho (leptocephalus)	1.8
"	"	Kingfish (Menticirrhus spp. juvenile)	1.71
"	"	Southern Kingfish	1.54
"	"	Cusk-eel (juvenile)	1.49
"	"	Banded Drum	1.47
"	"	Star Drum	1.36
"	"	Blackcheek Tonguefish	1.36
"	"	Searobin (Prionotus spp. juvenile)	1.32
"	"	Blackwing Searobin	1.28
"	"	Shelf Flounder	1.22
"	"	Rock Sea Bass	1.19
"	"	Northern Searobin	1.14
Bull and Chester	80.75	Inshore Lizardfish	5.02
"	"	Leopard Searobin	4.94
"	"	Spotted Whiff	4.8
"	"	Fringed Flounder	3.05
"	"	Atlantic Croaker	2.99
"	"	Unidentified fish (larvae/juvenile)	2.17
"	"	Striped Anchovy	2.1
"	"	Shelf Flounder	1.88
"	"	Seminole Goby	1.8
"	"	Sciaenidae (juvenile)	1.71
"	"	Smallmouth Flounder	1.68
"	"	Cusk-eel (juvenile)	1.57
"	"	Dusky Flounder	1.56
"	"	Northern Searobin	1.56
"	"	Silver Seatrout	1.54
"	"	Ladyfish/Malacho (leptocephalus)	1.47
"	"	Sand Perch	1.46
"	"	Blotched Cusk-eel	1.46
"	"	Rock Sea Bass	1.43
"	"	Twospot Flounder	1.4
"	"	Searobin (Prionotus spp. juvenile)	1.33
"	"	Banded Drum	1.32
"	"	Blackwing Searobin	1.31
"	"	Spot	1.21
Bull and CSII-BA	80.70	Spotted Whiff	5.36
"	"	Leopard Searobin	5.1

Shoal Comparisons	Average Dissimilarity (%)	Species	Contribution to Dissimilarity (%)
"	"	Inshore Lizardfish	4.57
"	"	Atlantic Croaker	3.92
"	"	Fringed Flounder	3.55
"	"	Unidentified fish (larvae/juvenile)	2.97
"	"	Sciaenidae (juvenile)	2.7
"	"	Striped Anchovy	2.69
"	"	Silver Seatrout	2.38
"	"	Northern Searobin	2.25
"	"	Cusk-eel (juvenile)	2.03
"	"	Seminole Goby	1.95
"	"	Searobin (Prionotus spp. juvenile)	1.89
"	"	Striped Cusk-eel	1.79
"	"	Smallmouth Flounder	1.7
"	"	Banded Drum	1.7
"	"	Kingfish (Menticirrhus spp. juvenile)	1.67
"	"	Spot	1.6
CSII and CSII-BA	80.49	Leopard Searobin	5.5
"	"	Spotted Whiff	5.4
"	"	Atlantic Croaker	4.44
"	"	Fringed Flounder	4.41
"	"	Inshore Lizardfish	3.92
"	"	Striped Anchovy	3.32
"	"	Sciaenidae (juvenile)	3.25
"	"	Unidentified fish (larvae/juvenile)	2.97
"	"	Silver Seatrout	2.94
"	"	Spot	2.3
"	"	Star Drum	2.19
"	"	Kingfish (Menticirrhus spp. juvenile)	2.15
"	"	Striped Cusk-eel	1.98
"	"	Cusk-eel (juvenile)	1.97
"	"	Searobin (Prionotus spp. juvenile)	1.88
"	"	Northern Searobin	1.88
Chester and CSII-BA	77.48	Spotted Whiff	5.58
"	"	Inshore Lizardfish	5.57
"	"	Leopard Searobin	5.44
"	"	Atlantic Croaker	4.11
"	"	Fringed Flounder	3.44
"	"	Unidentified fish (larvae/juvenile)	3.35
"	"	Striped Anchovy	3.31
"	"	Sciaenidae (juvenile)	2.82
"	"	Silver Seatrout	2.63

Shoal Comparisons	Average Dissimilarity (%)	Species	Contribution to Dissimilarity (%)
"	"	Cusk-eel (juvenile)	2.42
"	"	Striped Cusk-eel	2.2
"	"	Kingfish (Menticirrhus spp. juvenile)	2.11
"	"	Northern Searobin	2.04
"	"	Searobin (Prionotus spp. juvenile)	1.88
"	"	Ladyfish/Malacho (leptocephalus)	1.87
"	"	Banded Drum	1.76

Table 13-16. SIMPER results showing the fish species that explained at least 50% of the dissimilarity between ridge and swale habitats based on abundance.

Direction of greater abundance is to indicate the general trend only and does not imply a significant difference between fish abundance (only general direction of difference). ND denotes no difference.

Comparison	Average Dissimilarity (%)	Species	Contribution to Dissimilarity (%)	Direction of Greater Abundance
Ridge and Swale	83.65	Inshore Lizardfish	5.26	Swale
"	"	Fringed Flounder	4.60	Swale
"	"	Leopard Seabrobin	4.54	ND
"	"	Spotted Whiff	4.48	Swale
"	"	Atlantic Croaker	4.45	Swale
"	"	Sciaenidae (juvenile)	2.80	Swale
"	"	Unidentified fish (larvae/juvenile)	2.63	Swale
"	"	Silver Seatrout	2.47	Swale
"	"	Banded Drum	2.45	Swale
"	"	Striped Anchovy	1.95	Swale
"	"	Kingfish spp.	1.92	Swale
"	"	Spot	1.90	Swale
"	"	Rock Sea Bass	1.85	Swale
"	"	Cusk-eel (juvenile)	1.83	ND
"	"	Northern Seabrobin	1.82	Ridge
"	"	Shelf Flounder	1.69	Swale
"	"	Seminole Goby	1.67	Swale
"	"	Southern Kingfish	1.61	Swale
"	"	Ladyfish/Malacho (leptocephalus)	1.58	ND

Table 13-17. SIMPER results showing the fish species that explained at least 50% of the dissimilarity between ridge and swale habitats based on biomass.

Direction of greater abundance is to indicate the general trend only and does not imply a significant difference between fish abundance (only general direction of difference). ND denotes no difference.

Comparison	Average Dissimilarity (%)	Species	Contribution to Dissimilarity (%)	Direction of Greater Biomass
Ridge and Swale	83.59	Leopard Seabrobin	5.35	ND
"	"	Spotted Whiff	5.34	Swale
"	"	Inshore Lizardfish	5.19	Swale
"	"	Fringed Flounder	5.03	Swale
"	"	Atlantic Croaker	4.69	Swale
"	"	Striped Anchovy	2.73	Swale
"	"	Unidentified fish (larvae/juvenile)	2.61	Swale
"	"	Silver Seatrout	2.48	Swale
"	"	Sciaenidae (juvenile)	2.39	Swale
"	"	Spot	2.02	Swale
"	"	Seminole Goby	1.87	Swale
"	"	Rock Sea Bass	1.8	Swale
"	"	Banded Drum	1.77	Swale
"	"	Cusk-eel (juvenile)	1.74	ND
"	"	Shelf Flounder	1.73	Swale
"	"	Northern Seabrobin	1.71	Ridge
"	"	Ladyfish/Malacho (leptocephalus)	1.7	ND

Table 13-18. PERMANOVA pairwise comparisons of fish assemblages based on abundance among years 2014–2018.

P(perm) is the permutation P-value.

Comparison	t	P(perm)	Unique perms
2014, 2015	1.9454	0.001	999
2014, 2016	1.9350	0.001	997
2014, 2017	2.4776	0.001	998
2014, 2018	4.1757	0.001	997
2015, 2016	2.1428	0.001	998
2015, 2017	2.9770	0.001	999
2015, 2018	4.2288	0.001	999
2016, 2017	1.9307	0.001	999
2016, 2018	3.1137	0.001	999
2017, 2018	3.2927	0.001	998

Table 13-19. Pairwise average similarities within (green diagonal) and between years 2014–2018 based on fish abundance.

YEAR	2014	2015	2016	2017	2018
2014	37.27	-	-	-	-
2015	34.56	33.65	-	-	-
2016	34.41	32.34	33.26	-	-
2017	33.98	31.45	32.68	33.74	-
2018	29.69	27.65	29.65	30.03	31.05

Table 13-20. PERMANOVA pairwise comparisons of fish assemblages based on biomass among years 2014–2018.

P(perm) is the permutation P-value.

Comparison	T	P(perm)	Unique perms
2014, 2015	1.7158	0.0017	9930
2014, 2016	2.0196	0.0001	9934
2014, 2017	2.4351	0.0001	9910
2014, 2018	4.1657	0.0001	9926
2015, 2016	2.0377	0.0001	9915
2015, 2017	2.8166	0.0001	9931
2015, 2018	4.2176	0.0001	9898
2016, 2017	2.0236	0.0001	9917
2016, 2018	3.2163	0.0001	9906
2017, 2018	3.3736	0.0001	9922

Table 13-21. Pairwise average similarities within (green diagonal) and between years 2014–2018 based on fish biomass.

YEAR	2014	2015	2016	2017	2018
2014	38.37	-	-	-	-
2015	35.55	33.98	-	-	-
2016	35.04	33.07	34.20	-	-
2017	34.38	32.10	33.22	34.14	-
2018	29.72	28.52	30.56	30.77	32.26

Table 13-22. PERMANOVA pairwise comparisons of fish assemblages based on abundance among seasons.

P(perm) is the permutation P-value.

Comparison	T	P(perm)	Unique Perms
Fall, Spring	3.629	0.0001	9917
Fall, Summer	6.095	0.0001	9895
Fall, Winter	3.680	0.0001	9928
Spring, Summer	4.281	0.0001	9912
Spring, Winter	3.465	0.0001	9928
Summer, Winter	5.814	0.0001	9925

Table 13-23. SIMPER results showing the average dissimilarity between season-pairs, fish species that explained at least 50% of the dissimilarity, and the season with the greater abundance for each fish species.

Direction of greater abundance is to indicate the general trend only and does not imply a significant difference between fish abundance (only general direction of difference).

Comparison	Average Dissimilarity (%)	Species	Contribution to Dissimilarity (%)	Direction of Greater Abundance
Fall and Summer	85.86	Inshore Lizardfish	5.45	Summer
"	"	Spotted Whiff	4.49	Summer
"	"	Leopard Searobin	4.46	Summer
"	"	Atlantic Croaker	3.21	Fall
"	"	Silver Seatrout	3.18	Fall
"	"	Fringed Flounder	3.01	Summer
"	"	Striped Anchovy	2.63	Fall
"	"	Banded Drum	2.43	Fall
"	"	Sciaenidae (juvenile)	2.37	Fall
"	"	Northern Searobin	2.21	Summer
"	"	Seminole Goby	2.19	Summer
"	"	Unidentified fish (larvae/juvenile)	2.14	Summer
"	"	Shelf Flounder	2.05	Summer
"	"	Kingfish spp.	1.92	Fall
"	"	Cusk-eel (juvenile)	1.83	Summer
"	"	Twospot Flounder	1.80	Summer
"	"	Rock Sea Bass	1.78	Same
"	"	Blackwing Searobin	1.70	Summer
"	"	Sand Perch	1.68	Summer
Fall and Spring	83.67	Inshore Lizardfish	5.48	Spring
"	"	Leopard Searobin	5.17	Fall
"	"	Atlantic Croaker	4.30	Fall

Comparison	Average Dissimilarity (%)	Species	Contribution to Dissimilarity (%)	Direction of Greater Abundance
"	"	Spotted Whiff	4.19	Spring
"	"	Silver Seatrout	3.87	Fall
"	"	Banded Drum	3.41	Fall
"	"	Fringed Flounder	3.35	Spring
"	"	Striped Anchovy	3.33	Fall
"	"	Sciaenidae (juvenile)	3.07	Fall
"	"	Kingfish spp.	2.80	Fall
"	"	Unidentified fish (larvae/juvenile)	2.49	Spring
"	"	Atlantic Cutlassfish	2.36	Fall
"	"	Southern Kingfish	2.17	Fall
"	"	Lined Seahorse	1.71	Fall
"	"	Rock Sea Bass	1.70	Fall
Spring and Summer	82.23	Inshore Lizardfish	6.06	Summer
"	"	Spotted Whiff	5.14	Summer
"	"	Leopard Searobin	5.03	Summer
"	"	Fringed Flounder	3.71	Summer
"	"	Unidentified fish (larvae/juvenile)	2.96	Summer
"	"	Seminole Goby	2.88	Summer
"	"	Atlantic Croaker	2.77	Spring
"	"	Northern Searobin	2.70	Summer
"	"	Shelf Flounder	2.43	Summer
"	"	Banded Drum	2.11	Spring
"	"	Dusky Flounder	1.95	Summer
"	"	Striped Anchovy	1.87	Summer
"	"	Blackwing Searobin	1.85	Summer
"	"	Twospot Flounder	1.77	Summer
"	"	Cusk-eel (juvenile)	1.74	Summer
"	"	Rock Sea Bass	1.68	Summer
"	"	Searobin (<i>Prionotus</i> spp. juvenile)	1.66	Summer
"	"	Anchovy (<i>Anchoa</i> spp.)	1.60	Summer
"	"	Flatfish (<i>Paralichthyidae</i> juvenile)	1.48	Summer
Summer and Winter	82.05	Inshore Lizardfish	5.74	Summer
"	"	Leopard Searobin	4.76	Summer
"	"	Spotted Whiff	4.70	Summer
"	"	Fringed Flounder	3.72	Summer
"	"	Atlantic Croaker	3.45	Winter
"	"	Sciaenidae (juvenile)	3.40	Winter
"	"	Unidentified fish (larvae/juvenile)	2.75	Summer
"	"	Northern Searobin	2.56	Summer
"	"	Cusk-eel (juvenile)	2.49	Summer

Comparison	Average Dissimilarity (%)	Species	Contribution to Dissimilarity (%)	Direction of Greater Abundance
"	"	Seminole Goby	2.35	Summer
"	"	Shelf Flounder	2.27	Summer
"	"	Ladyfish/Malacho (leptocephalus)	1.97	Winter
"	"	Twospot Flounder	1.87	Summer
"	"	Smallmouth Flounder	1.87	Summer
"	"	Blackwing Searobin	1.84	Summer
"	"	Spot	1.69	Winter
"	"	Rock Sea Bass	1.68	Summer
"	"	Sand Perch	1.64	Summer
"	"	Leopard Searobin	5.54	Winter
Spring and Winter	81.32	Inshore Lizardfish	6.37	Spring
"	"	Spotted Whiff	4.91	Winter
"	"	Atlantic Croaker	4.59	Winter
"	"	Fringed Flounder	4.22	Winter
"	"	Sciaenidae (juvenile)	4.17	Winter
"	"	Unidentified fish (larvae/juvenile)	3.23	Spring
"	"	Ladyfish/Malacho (leptocephalus)	2.75	Winter
"	"	Banded Drum	2.68	Spring
"	"	Kingfish spp.	2.32	Spring
"	"	Spot	2.23	Winter
"	"	Silver Seatrout	2.09	Winter
"	"	Cusk-eel (juvenile)	1.98	Winter
"	"	Searobin (<i>Prionotus</i> spp. juvenile)	1.88	Spring
"	"	Southern Kingfish	1.80	Winter
Fall and Winter	80.90	Leopard Searobin	5.06	Winter
"	"	Inshore Lizardfish	4.65	Winter
"	"	Atlantic Croaker	4.62	Fall
"	"	Spotted Whiff	4.33	Winter
"	"	Sciaenidae (juvenile)	4.19	Winter
"	"	Silver Seatrout	3.93	Fall
"	"	Fringed Flounder	3.56	Winter
"	"	Banded Drum	3.05	Fall
"	"	Striped Anchovy	2.93	Fall
"	"	Kingfish spp.	2.79	Fall
"	"	Ladyfish/Malacho (leptocephalus)	2.55	Winter
"	"	Southern Kingfish	2.38	Fall
"	"	Atlantic Spadefish	2.31	Fall
"	"	Spot	2.20	Winter
"	"	Unidentified fish (larvae/juvenile)	2.18	Winter

Table 13-24. PERMANOVA pairwise comparisons of fish assemblages based on biomass among seasons.

P(perm) is the permutation P-value.

Comparison	t	P(perm)	Unique Perms
Fall, Spring	3.4257	0.001	998
Fall, Summer	5.5892	0.001	999
Fall, Winter	3.4249	0.001	999
Spring, Summer	4.0216	0.001	997
Spring, Winter	3.3182	0.001	997
Summer, Winter	5.3991	0.001	998

Table 13-25. SIMPER results showing the average dissimilarity between season-pairs based on biomass for fish species that explained at least 50% of the dissimilarity, along with the season with the greater biomass for each fish species.

Direction of greater biomass is to indicate the general trend only and does not imply a significant difference between fish biomass (only general direction of difference). ND denotes no difference.

Comparison	Average Dissimilarity (%)	Species	Contribution to Dissimilarity (%)	Direction of Greater Biomass
<i>Fall vs Summer</i>	<i>85.56</i>	<i>Spotted Whiff</i>	<i>5.18</i>	<i>Summer</i>
"	"	Leopard Searobin	5.04	Summer
"	"	Inshore Lizardfish	5.01	Summer
"	"	Atlantic Croaker	3.57	Fall
"	"	Striped Anchovy	3.43	Fall
"	"	Fringed Flounder	3.30	Summer
"	"	Silver Seatrout	2.72	Fall
"	"	Seminole Goby	2.49	Summer
"	"	Unidentified fish (larvae/juvenile)	2.35	Summer
"	"	Northern Searobin	2.16	Summer
"	"	Shelf Flounder	2.13	Summer
"	"	Blackwing Searobin	2.05	Summer
"	"	Sciaenidae (juvenile)	2.04	Fall
"	"	Banded Drum	1.96	Fall
"	"	Rock Sea Bass	1.84	Fall
"	"	Twospot Flounder	1.77	Summer
"	"	Cusk-eel (juvenile)	1.76	Summer
"	"	Sand Perch	1.65	Summer
<i>Fall vs Spring</i>	<i>83.73</i>	Leopard Searobin	6.25	Spring
"	"	Inshore Lizardfish	5.37	Fall
"	"	Spotted Whiff	5.17	Spring
"	"	Atlantic Croaker	4.70	Spring
"	"	Striped Anchovy	4.19	Spring
"	"	Fringed Flounder	3.77	Fall
"	"	Silver Seatrout	3.60	Spring
"	"	Sciaenidae (juvenile)	2.63	Spring
"	"	Banded Drum	2.58	Spring
"	"	Unidentified fish (larvae/juvenile)	2.49	Fall
"	"	Kingfish (<i>Menticirrhus</i> spp. juvenile)	2.31	Spring
"	"	Rock Sea Bass	1.74	Spring
"	"	Spot	1.71	Spring
"	"	Atlantic Bumper	1.69	Spring
"	"	Star Drum	1.55	Spring
"	"	Clupeiformes (juvenile)	1.55	Fall

Comparison	Average Dissimilarity (%)	Species	Contribution to Dissimilarity (%)	Direction of Greater Biomass
Spring vs Summer	82.18	Inshore Lizardfish	5.92	Summer
"	"	Spotted Whiff	5.89	Summer
"	"	Leopard Searobin	5.82	Summer
"	"	Fringed Flounder	4.17	ND
"	"	Seminole Goby	3.26	Summer
"	"	Unidentified fish (larvae/juvenile)	3.14	Summer
"	"	Atlantic Croaker	3.05	Spring
"	"	Northern Searobin	2.58	Summer
"	"	Striped Anchovy	2.54	Summer
"	"	Shelf Flounder	2.52	Summer
"	"	Blackwing Searobin	2.15	Summer
"	"	Dusky Flounder	1.89	Summer
"	"	Searobin (Prionotus spp. juvenile)	1.85	Summer
"	"	Cusk-eel (juvenile)	1.73	Summer
"	"	Flatfish (Paralichthyidae juvenile)	1.70	Summer
"	"	Clupeidae (juvenile)	1.66	Spring
"	"	Twospot Flounder	1.65	Summer
Summer vs Winter	81.64	Leopard Searobin	5.56	Winter
"	"	Inshore Lizardfish	5.53	Summer
"	"	Spotted Whiff	5.46	Summer
"	"	Fringed Flounder	4.05	ND
"	"	Atlantic Croaker	3.57	Winter
"	"	Sciaenidae (juvenile)	2.90	Winter
"	"	Unidentified fish (larvae/juvenile)	2.74	Summer
"	"	Seminole Goby	2.61	Summer
"	"	Cusk-eel (juvenile)	2.41	Summer
"	"	Northern Searobin	2.41	Summer
"	"	Shelf Flounder	2.33	Summer
"	"	Blackwing Searobin	2.19	Summer
"	"	Ladyfish/Malacho (leptocephalus)	2.12	Winter
"	"	Striped Anchovy	2.09	Summer
"	"	Smallmouth Flounder	2.05	Summer
"	"	Blotched Cusk-eel	1.91	Summer
"	"	Twospot Flounder	1.81	Summer
Fall vs Winter	80.87	Leopard Searobin	6.12	Winter
"	"	Spotted Whiff	5.47	Winter
"	"	Inshore Lizardfish	4.86	Winter
"	"	Atlantic Croaker	4.84	Fall
"	"	Fringed Flounder	3.83	Winter
"	"	Striped Anchovy	3.82	Fall

Comparison	Average Dissimilarity (%)	Species	Contribution to Dissimilarity (%)	Direction of Greater Biomass
"	"	Sciaenidae (juvenile)	3.53	Winter
"	"	Silver Seatrout	3.48	Fall
"	"	Ladyfish/Malacho (leptocephalus)	2.72	Winter
"	"	Banded Drum	2.52	Fall
"	"	Kingfish (Menticirrus spp. juvenile)	2.33	Fall
"	"	Spot	2.30	Winter
"	"	Atlantic Spadefish	2.26	Fall
"	"	Unidentified fish (larvae/juvenile)	2.00	Winter
Spring vs Winter	80.82	Leopard Searobin	6.80	Winter
"	"	Inshore Lizardfish	6.40	Spring
"	"	Spotted Whiff	6.07	Winter
"	"	Atlantic Croaker	4.72	Winter
"	"	Fringed Flounder	4.67	ND
"	"	Sciaenidae (juvenile)	3.55	Winter
"	"	Ladyfish/Malacho (leptocephalus)	3.00	Winter
"	"	Unidentified fish (larvae/juvenile)	2.97	Spring
"	"	Striped Anchovy	2.56	Spring
"	"	Silver Seatrout	2.45	ND
"	"	Spot	2.35	Winter
"	"	Kingfish (Menticirrus spp. juvenile)	2.01	Spring
"	"	Cusk-eel (juvenile)	1.87	Winter
"	"	Searobin (Prionotus spp. juvenile)	1.82	Spring

Table 13-26. SIMPER results showing the average dissimilarity between fish assemblages based on abundance from day versus night, fish species that explained at least 50% of the dissimilarity, and the time period with the direction of greater abundance for each fish species.

Direction of greater abundance is to indicate the general trend only and does not imply a significant difference between fish abundance (only general direction of difference). *Indicates average abundance was zero or near-zero during day sampling.

Comparison	Average Dissimilarity (%)	Species	Contribution to Dissimilarity (%)	Direction of Greater Abundance
Day and Night	82.70	Inshore Lizardfish	4.66	Day
"	"	Leopard Searobin	4.62	Night
"	"	Spotted Whiff	4.43	Night
"	"	Atlantic Croaker	3.88	Night
"	"	Fringed Flounder	3.37	Night
"	"	Cusk-eel (juvenile)	3.19	Night*
"	"	Sciaenidae (juvenile)	2.54	Day
"	"	Banded Drum	2.39	Night
"	"	Unidentified fish (larvae/juvenile)	2.33	Night
"	"	Kingfish spp.	2.29	Night
"	"	Searobin (Prionotus spp. juvenile)	2.20	Night
"	"	Silver Seatrout	2.15	Night
"	"	Striped Anchovy	1.97	Night
"	"	Northern Searobin	1.94	Night
"	"	Blotched Cusk-eel	1.82	Night*
"	"	Striped Cusk-eel	1.73	Night*
"	"	Spot	1.61	Night
"	"	Rock Sea Bass	1.60	Night
"	"	Smallmouth Flounder	1.52	Night

Table 13-27 SIMPER results showing the average dissimilarity between fish assemblages based on biomass from day versus night collections, fish species that explained at least 50% of the dissimilarity, and the time period with the greater biomass for each fish species.

Direction of greater biomass is to indicate the general trend only and does not imply a significant difference between fish biomass (only general direction of difference). *Indicates average biomass was zero or near-zero during day sampling.

Comparison	Average Dissimilarity (%)	Species	% Contribution	Direction of Greater Biomass
Day vs Night	82.53	Leopard Searobin	5.40	Night
"	"	Spotted Whiff	5.23	Night
"	"	Inshore Lizardfish	4.52	Day
"	"	Atlantic Croaker	4.11	Night
"	"	Fringed Flounder	3.66	Night
"	"	Cusk-eel (juvenile)*	3.01	Night*
"	"	Striped Anchovy	2.81	Night
"	"	Searobin (<i>Prionotus</i> spp. juvenile)	2.35	Night
"	"	Unidentified fish (larvae/juvenile)	2.33	ND
"	"	Blotched Cusk-eel*	2.27	Night*
"	"	Silver Seatrout	2.22	Night
"	"	Sciaenidae (juvenile)	2.19	ND
"	"	Kingfish (<i>Menticirrhus</i> spp. juvenile)	1.96	Night
"	"	Banded Drum	1.85	Night
"	"	Northern Searobin	1.82	Night
"	"	Striped Cusk-eel*	1.76	Night*
"	"	Spot	1.72	Night
"	"	Seminole Goby	1.71	Night

Table 13-28. PERMANOVA partitioning and analysis of fish assemblages for Shoal and Season based on species abundances.

Analysis based on dispersion-weighted, fourth-root transformed abundances and zero-adjusted Bray-Curtis dissimilarities. Variation gives the estimated sizes of the components of variation. Table abbreviations include: df = degrees of freedom, SS = sums of squares, MS = mean square, Pseudo-F is the F test statistic, and P = significance value associated with the test.

Source	df	SS	MS	Pseudo-F	P	Variation
Shoal	3	54,318	18,106	8.402	0.0001	89.76
Season	3	106,090	35,364	16.411	0.0001	218.24
ShoalxSeason	9	38,924	4,325	2.007	0.0001	47.34
Residual	779	1,678,700	2,155	-	-	2154.90
Total	794	1,924,900	-	-	-	-

Table 13-29. PERMANOVA partitioning and analysis of fish assemblages for Shoal, Season, and Day_Night factors based on species abundances.

Analysis based on dispersion-weighted, fourth-root transformed abundances and zero-adjusted Bray-Curtis dissimilarities. Variation gives the estimated sizes of the components of variation. Table abbreviations include: df = degrees of freedom, SS = sums of squares, MS = mean square, Pseudo-F is the F test statistic, and P = significance value associated with the test.

Source	df	SS	MS	Pseudo-F	P	Variation
Shoal	3	47,640	15,880	8.185	0.0001	94.28
Season	3	106,930	35,644	18.372	0.0001	248.70
DayNight	1	69,925	69,925	36.041	0.0001	267.95
ShxSe	9	39,745	4,416	2.276	0.0001	61.07
ShxDA	3	15,433	5,145	2.652	0.0001	43.35
SexDa	3	36,009	12,003	6.187	0.0001	148.51
ShxSexDa	9	31,397	3,489	1.798	0.0001	76.38
Residual	763	1,480,300	1,940	-	-	1,940.10
Total	794	1,924,900	-	-	-	-

Table 13-30. PERMANOVA pairwise shoal comparisons of fish assemblages based on abundance among seasons and diurnal periods of day versus night.

Significant differences at $P \leq 0.05$ are in blue italics.

Season	Diurnal Period	Shoal Pairs	t	P
Spring	Day	Bull, Chester	1.9574	<i>0.0006</i>
"	"	Bull, CSII	1.6405	<i>0.0124</i>
"	"	Bull, CSII-BA	1.7333	<i>0.0036</i>
"	"	Chester, CSII	2.6729	<i>0.0001</i>
"	"	Chester, CSII-BA	2.1500	<i>0.0003</i>
"	"	CSII, CSII-BA	1.4503	<i>0.0393</i>
Spring	Night	Bull, Chester	2.6275	<i>0.0001</i>
"	"	Bull, CSII	1.7722	<i>0.0002</i>
"	"	Bull, CSII-BA	2.3107	<i>0.0005</i>
"	"	Chester, CSII	1.6636	<i>0.0062</i>
"	"	Chester, CSII-BA	1.1700	0.2060
"	"	CSII, CSII-BA	1.0656	0.3189
Summer	Day	Bull, Chester	1.2561	0.1179
"	"	Bull, CSII	1.9502	<i>0.0005</i>
"	"	Bull, CSII-BA	1.5422	<i>0.0113</i>
"	"	Chester, CSII	2.0205	<i>0.0002</i>
"	"	Chester, CSII-BA	1.6491	<i>0.0069</i>
"	"	CSII, CSII-BA	1.2281	0.1308
Summer	Night	Bull, Chester	1.4653	<i>0.0223</i>
"	"	Bull, CSII	2.4339	<i>0.0001</i>

Season	Diurnal Period	Shoal Pairs	t	P
"	"	Bull, CSII-BA	1.9246	0.0002
"	"	Chester, CSII	2.2968	0.0001
"	"	Chester, CSII-BA	1.8511	0.0010
"	"	CSII, CSII-BA	1.0670	0.3046
Fall	Day	Bull, Chester	1.3997	0.0498
"	"	Bull, CSII	1.5785	0.0196
"	"	Bull, CSII-BA	0.9998	0.3989
"	"	Chester, CSII	1.7921	0.0041
"	"	Chester, CSII-BA	1.1333	0.2245
"	"	CSII, CSII-BA	1.0333	0.3356
Fall	Night	Bull, Chester	1.6558	0.0004
"	"	Bull, CSII	1.7995	0.0005
"	"	Bull, CSII-BA	1.1381	0.2050
"	"	Chester, CSII	1.5836	0.0071
"	"	Chester, CSII-BA	0.9985	0.4425
"	"	CSII, CSII-BA	0.8121	0.6356
Winter	Day	Bull, Chester	1.1568	0.2057
"	"	Bull, CSII	2.4377	0.0001
"	"	Bull, CSII-BA	1.5593	0.0119
"	"	Chester, CSII	3.0456	0.0001
"	"	Chester, CSII-BA	1.9205	0.0018
"	"	CSII, CSII-BA	1.6512	0.0206
Winter	Night	Bull, Chester	1.9897	0.0003
"	"	Bull, CSII	1.6319	0.0033
"	"	Bull, CSII-BA	1.2277	0.0991
"	"	Chester, CSII	1.8696	0.0019
"	"	Chester, CSII-BA	1.0975	0.2772
"	"	CSII, CSII-BA	1.1534	0.1948

Table 13-31. PERMANOVA partitioning and analysis of fish assemblages for Shoal and Ridge_Swale based on species abundances.

Analysis based on dispersion-weighted, fourth-root transformed abundances and zero-adjusted Bray-Curtis dissimilarities. Variation gives the estimated sizes of the components of variation. Table abbreviations include: df = degrees of freedom, SS = sums of squares, MS = mean square, Pseudo-F is the F test statistic, and P = significance value associated with the test.

Source	df	SS	MS	Pseudo-F	P	Variation
Shoal	3	56,464	18,821	8.7576	0.0001	89.861
Ridge_Swale	1	113,310	113,310	52.723	0.0001	361.24
ShxRi	3	30,924	10,308	4.7964	0.0001	87.951
Residual	787	1,691,400	2,149	-	-	2,149.1
Total	794	1,924,900	-	-	-	-

Table 13-32. Three-way PERMANOVA partitioning and analysis of fish assemblages for Shoal, Season, and Ridge_Swale factors based on species abundances.

Analysis based on dispersion-weighted, fourth-root transformed abundances and zero-adjusted Bray-Curtis dissimilarities. Variation gives the estimated sizes of the components of variation. Table abbreviations include: df = degrees of freedom, SS = sums of squares, MS = mean square, Pseudo-F is the F test statistic, and P = significance value associated with the test.

Source	df	SS	MS	Pseudo-F	P	Variation
Season	3	103,970	34,657	17.9520	0.0001	215.86
Ridge_Swale	1	113,180	113,180	58.6230	0.0001	373.21
ShxSe	9	37,558	4,173	2.1616	0.0001	48.95
ShxRi	3	30,643	10,214	5.2908	0.0001	89.63
SexRi	3	15,345	5,115	2.6495	0.0001	32.75
Residual	794	1,924,900	1,931	-	-	1930.60
Total	794	1,924,900	-	-	-	-

Table 13-33. PERMANOVA pairwise comparisons of daytime fish assemblages based on abundance among CSII, CSII-BA, and Bull Shoals for fall 2013 to 2017.

Year	Groups	t	P
2013	Bull, CSII	1.763	0.059
"	Bull, CSII-BA	1.896	0.063
"	CSII, CSII-BA	1.117	0.403
2014	Bull, CSII	1.418	0.095
"	Bull, CSII-BA	1.072	0.257
"	CSII, CSII-BA	1.059	0.333
2015	Bull, CSII	1.191	0.223
"	Bull, CSII-BA	1.317	0.091
"	CSII, CSII-BA	0.914	0.600
2016	Bull, CSII	1.414	0.051
"	Bull, CSII-BA	1.084	0.274
"	CSII, CSII-BA	1.065	0.274
2017	Bull, CSII	0.636	0.853
"	Bull, CSII-BA	0.602	0.909
"	CSII, CSII-BA	0.562	0.605

Table 13-34. PERMANOVA pairwise comparisons of daytime fish assemblages on CSII, CSII-BA, Bull and Chester Shoals based on abundance among four seasons in the year preceding the second dredging event.

P-values were estimated using Monte Carlo runs (i.e., P(MC)); significant differences at $P \leq 0.05$ are in blue italics.

Year	Groups	t	P(MC)
Winter 2016-17	Bull, Chester	1.110	0.2976
"	Bull, CSII	1.063	0.3202
"	Bull, CSII-BA	1.230	0.2125
"	Chester, CSII	1.274	0.1835
"	Chester, CSII-BA	0.954	0.4697
"	CSII, CSII-BA	0.725	0.6475
Spring 2017	Bull, Chester	1.603	<i>0.0488</i>
"	Bull, CSII	1.640	<i>0.0428</i>
"	Bull, CSII-BA	1.718	<i>0.0344</i>
"	Chester, CSII	1.802	<i>0.0227</i>
"	Chester, CSII-BA	1.423	0.1207
"	CSII, CSII-BA	1.132	0.3358
Summer 2017	Bull, Chester	1.433	0.0860
"	Bull, CSII	2.006	<i>0.0148</i>
"	Bull, CSII-BA	1.695	<i>0.0483</i>
"	Chester, CSII	1.667	<i>0.0384</i>
"	Chester, CSII-BA	1.353	0.1298

Year	Groups	t	P(MC)
"	CSII, CSII-BA	0.755	0.6556
Fall 2017	Bull, Chester	1.860	<i>0.0221</i>
"	Bull, CSII	0.636	0.7984
"	Bull, CSII-BA	0.602	0.8197
"	Chester, CSII	1.585	0.0679
"	Chester, CSII-BA	1.030	0.3770
"	CSII, CSII-BA	0.562	0.7774

Table 13-35. PERMANOVA pairwise comparisons of daytime fish assemblages on CSII, CSII-BA, Bull and Chester Shoals based on abundance for four seasons (winter 2016/17, spring 2017, summer 2017, and fall 2017) in the year preceding the second dredging event, and four seasons (winter 2017/18, spring 2018, summer 2018, and fall 2018) during or after the Dredge-2 event.

P-values were estimated using Monte Carlo runs (i.e., P(MC)); significant differences at $P \leq 0.05$ are in blue italics.

Year	Groups	t	P(MC)
Winter 2016/17	Bull, Chester	1.110	0.2939
"	Bull, CSII	1.063	0.3273
"	Bull, CSII-BA	1.230	0.2140
"	Chester, CSII	1.274	0.1888
"	Chester, CSII-BA	0.954	0.4561
"	CSII, CSII-BA	0.725	0.6435
Spring 2017	Bull, Chester	1.603	<i>0.0483</i>
"	Bull, CSII	1.640	<i>0.0441</i>
"	Bull, CSII-BA	1.718	<i>0.0402</i>
"	Chester, CSII	1.802	<i>0.0219</i>
"	Chester, CSII-BA	1.423	0.1246
"	CSII, CSII-BA	1.132	0.3286
Summer 2017	Bull, Chester	1.433	0.0867
"	Bull, CSII	2.006	<i>0.0145</i>
"	Bull, CSII-BA	1.695	<i>0.0454</i>
"	Chester, CSII	1.667	<i>0.0390</i>
"	Chester, CSII-BA	1.353	0.1346
"	CSII, CSII-BA	0.755	0.6489
Fall 2017	Bull, Chester	1.860	<i>0.0230</i>
"	Bull, CSII	0.636	0.7886
"	Bull, CSII-BA	0.602	0.8180
"	Chester, CSII	1.585	0.0667
"	Chester, CSII-BA	1.030	0.3701
"	CSII, CSII-BA	0.562	0.7842
Winter 2017/18	Bull, Chester	1.380	0.1299
"	Bull, CSII	1.357	0.1498

Year	Groups	t	P(MC)
"	Bull, CSII-BA	1.166	0.2593
"	Chester, CSII	1.416	0.1243
"	Chester, CSII-BA	1.170	0.2621
"	CSII, CSII-BA	0.815	0.5962
Spring 2018	Bull, Chester	1.660	0.0365
"	Bull, CSII	1.001	0.3847
"	Bull, CSII-BA	1.444	0.1131
"	Chester, CSII	1.089	0.3087
"	Chester, CSII-BA	2.025	0.0167
"	CSII, CSII-BA	1.197	0.2685
Summer 2018	Bull, Chester	1.039	0.3651
"	Bull, CSII	1.182	0.2153
"	Bull, CSII-BA	1.144	0.2658
"	Chester, CSII	1.773	0.0151
"	Chester, CSII-BA	1.959	0.0148
"	CSII, CSII-BA	1.412	0.1476
Fall 2018	Bull, Chester	1.209	0.1961
"	Bull, CSII	0.859	0.5231
"	Bull, CSII-BA	0.786	0.6278
"	Chester, CSII	1.708	0.0355
"	Chester, CSII-BA	1.220	0.2061
"	CSII, CSII-BA	0.983	0.4233

Table 13-36. Asymmetrical four-way PERMANOVA analysis of fish assemblages based on species abundances to test for impact of the Dredge-1 event.

Analysis based on dispersion-weighted, fourth-root transformed abundances and zero-adjusted Bray-Curtis dissimilarities. The nonsignificant interaction between Before/After and Impact/Control factors is highlighted in bold and green. Table abbreviations include: df = degrees of freedom, SS = sums of squares, MS = mean square, Pseudo-F is the F test statistic, P(perm) and P(MC) are P-values derived by permutation versus Monte Carlo methods, and perms = number of permutations.

Source	df	SS	MS	Pseudo-F	P(perm)	perms	P(MC)
Before-After (BA)	1	2,861	2,861	0.7366	0.7057	9951	0.7104
Impact-Control (IC)	1	2,427	2,427	0.7448	0.6953	9943	0.7213
Time(BA)	3	12,628	4,210	1.7938	0.1215	9938	0.0479
Shoal(IC)	1	6,280	6,280	2.3619	0.1143	9948	0.0637
BAxIC	1	2,119	2,119	1.0436	0.4792	9946	0.4561
BAxShoal(IC)	1	3,043	3,043	1.1444	0.3948	9956	0.3464
ICxTime(BA)	3	5,268	1,756	0.7483	0.7185	9928	0.7528
Time(BA)xShoal(IC)	3	8,087	2,696	1.3786	0.1305	9898	0.1433
Res	48	93,850	1,955	-	-	-	-
Total	62	143,460	-	-	-	-	-

Table 13-37. Asymmetrical four-way PERMANOVA analysis of fish assemblages based on species abundances to test for impact of the Dredge-2 event.

Analysis based on dispersion-weighted, fourth-root transformed abundances and zero-adjusted Bray-Curtis dissimilarities. The nonsignificant interaction between Before/After and Impact/Control factors is highlighted in bold and green. Table abbreviations include: df = degrees of freedom, SS = sums of squares, MS = mean square, Pseudo-F is the F test statistic, P(perm) and P(MC) are P-values derived by permutation versus Monte Carlo methods, and perms = number of permutations.

Source	df	SS	MS	Pseudo-F	P(perm)	perms	P(MC)
Before-After (BA)	1	8,444.9	8,444.9	1.5894	0.1181	9922	0.1287
Impact-Control (IC)	1	2,219.6	2,219.6	0.77274	0.8126	9894	0.7175
Time(BA)	6	2,7851	4,641.8	2.188	0.017	9930	0.0004
Shoal(IC)	2	10485	5,242.5	1.8765	0.0263	9905	0.0409
BAxIC	1	1,268.6	1,268.6	0.67126	0.9152	9904	0.8719
BAxShoal(IC)	2	6869.8	3,434.9	1.2295	0.2462	9896	0.2473
ICxTime(BA)	6	13,179	2,196.5	1.0354	0.4298	9902	0.4045
Time(BA)xShoal(IC)	12	33,617	2,801.4	1.7171	0.0004	9816	0.0005
Res	135	2.20E+05	1,631.5	-	-	-	-
Total	166	3.82E+05	-	-	-	-	-

14 Spatiotemporal Fish Movement and Residency on Offshore Sand Shoals in Relation to Dredging Events

Daryl Parkyn and Debra Murie

Key Points

- Acoustically tagged, demersal fish species—including flatfishes, rays, and a skate—showed low residency over the sand shoals; on average, fish were detected over the sand shoals < 3% of the time that they were at large.
- Rays, in particular, had focused detections in the southern portion of the receiver array and specifically at the entrance to Port Canaveral.
- Detections for a Summer Flounder, a Smooth Butterfly Ray, a Bluntnose Stingray, and a Clearnose Skate all indicated that these fish were within the area of the dredged shoal (CSII-BA) either before or during, and then again after dredging activities, indicating that they were not directly affected by the second dredging event in spring 2018.

14.1 Introduction

Sand shoals offshore the East Coast of the U.S. Atlantic have been designated as EFH for a variety of fish species (SAFMC 1998), including those important in fisheries (e.g., sciaenids, flatfishes), as well as endangered/threatened species (e.g., Atlantic Sturgeon *Acipenser oxyrinchus oxyrinchus*). Shoals are expected to be important to fish species for foraging and spawning, as well as for nursery areas. Pickens and Taylor (2020) and Grothues et al. (2021) provide some preliminary results on the extent of use of sand shoal habitats by fishes. In addition to using trawl surveys to investigate the importance of shoals to fish species and assemblages (Chapter 13), acoustic telemetry can be used for more detailed information on the use and timing of shoal habitat by individual fish. Both active and passive acoustic telemetry (mobile and stationary receivers, respectively) have been used effectively for a wide variety of fishes in a wide range of environments (e.g., Harris et al. 2005; Hussey et al. 2015; Brownscombe et al. 2022), some specific to sand shoals (Iafrate et al. 2019, 2022).

Acoustic telemetry provides important information that is difficult to obtain using other methodologies since it can provide information 24 hours a day throughout an extended period of time (i.e., years). The deployment of an acoustic array to monitor the movement and distribution of tagged fishes allows us to understand how the sand shoal habitats are used by demersal fishes, including flatfishes, rays, and skates. Some of these demersal fishes are important in commercial and recreational fisheries (i.e., flatfishes), whereas all are ecologically important as demersal predators in the ecosystem and indicator-species for benthic perturbations.

Demersal fishes are ideally suited to monitor for potential impacts of perturbations, such as dredging of sand shoals, because of their reliance on the benthos for both their physical habitat and their prey resources. Using acoustic telemetry, their changes in occurrence and distribution can be monitored relative to dredging events. For instance, sand dredging may displace fish from the immediate area to adjacent areas or from the entire study area. Acoustic monitoring allows us to determine if displacement of fish is temporary (e.g., until dredging operations cease) or more long lasting (e.g., fish do not return to the dredged area for several months or years). Mobile demersal fish predators uniquely connect seafloor

habitats because they respond to local-scale changes and because they differentially select demersal habitats for feeding, which can be disrupted due to dredging events.

14.1.1 Goals and Objectives

The overall goal was to describe and quantify the movement and residency of demersal fishes associated with sand shoals off the east coast of Florida, with specific objectives to include:

- 1) Determine the spatial and seasonal extent of demersal fish movement and residency among sand shoals off the east coast of Florida;
- 2) Determine if any spatial or temporal movement patterns and residency are related specifically to the dredged shoal (CSII-BA) relative to the reference shoals (CSII, Chester, and Bull Shoals).

14.2 Methods

14.2.1 Study Area and Focal Species

The study area included the sand shoals and adjacent coastal areas off Cape Canaveral on the east coast of Florida. This area specifically included Canaveral Shoal II Borrow Area (CSII-BA) (the dredged shoal) and reference (non-dredged) shoals including Canaveral Shoal II (CSII), Chester Shoal, and Bull Shoal (Figure 1-1). Details of the sand shoals are given in Chapter 1.

The focal species for tagging were all demersal fishes that were known to be both associated with sandy bottoms of the shoals and also to forage on bottom-dwelling organisms. A brief summary of pertinent information for each species is provided.

Southern Flounder *Paralichthys lethostigma* ranges from North Carolina to the east coast of Florida, and then from southwest Florida through the Gulf of Mexico and into Mexico; it is not known to occur in south Florida or the Florida Keys (Robins and Ray 1986). Although a minor component of the total finfish commercial catch, it is the largest and most commercially valuable flounder off the east coast of Florida and a popular sports fish that is often gilled (i.e., speared) in estuaries. Juvenile Southern Flounder are found inshore in estuaries and inlets (Guindon and Miller 1995) where they eat small, active epifaunal prey, such as mysids, amphipods and calanoid copepods (Burke 1995). Burke (1995) also suggests that prey distribution (mysid gradients) may influence movement and guide Southern Flounder to nursery grounds. At about 10 cm in length, Southern Flounder undergo a dietary shift towards larger fish and shrimp. As adults the majority of their diet is fish, such as anchovies, mullet, menhaden, and Atlantic Croaker, along with penaeid shrimp (Enge and Mulholland 1985). Horst (2003) indicated that as flounders get larger they do not necessarily eat larger fish, they just eat more small ones. Southern Flounder migrate out of estuaries and inshore waters in October and November to deeper waters of the Gulf Stream and continental shelf to spawn and return to estuarine and nearshore waters by May (Enge and Mulholland 1985). Because of their seasonal spawning migration, Southern Flounder were expected to use the sand shoal habitat primarily between spring and late fall.

Summer Flounder *Paralichthys dentatus* are found in coastal and offshore waters from Nova Scotia, Canada, to Cape Canaveral, Florida, with the center of distribution from Delaware to Cape Hatteras, North Carolina (Gilbert 1986). The species prefers shallow nearshore shelf waters, coastal bays, lower to middle reaches of estuaries and harder sandy substrates. They typically burrow into sandy or muddy bottoms, as well as inhabit tidal creeks with submerged vegetation but are also around inlets, jetties, beaches, and nearshore reefs (Gilbert 1986; Burke 1995; South Carolina Department of Natural Resources (DNR) 2020). Although similar to Southern Flounder, diets of juveniles of the two species appear to diverge and Summer Flounder consume primarily infauna, including polychaetes, and other invertebrates. As adults they feed on small fishes, including anchovies, menhaden, bluefish, weakfish, mummichog, as well as penaeid shrimp, crabs, squid, and bivalve mollusks (Gilbert 1986; Burke 1995; SCDNR 2020). In

general, adult Summer Flounder migrate to nearshore waters to forage during spring through fall (Henderson and Fabrizio 2011), after which they move offshore into deeper waters of the continental shelf to spawn in late fall to early winter (Kraus and Musick 2001). Based on their seasonal spawning migration to deeper waters, Southern Flounder were expected to use the shallower sand shoal habitats primarily between spring and late fall.

Gulf Flounder *Paralichthys albigutta* are distributed in shallow inshore waters in the U.S. western Atlantic from North Carolina to southern Florida, the Gulf of Mexico, the northern Bahamas, and the western Caribbean to Colombia (Gilbert 1986). This species has similar habitat preferences to Summer Flounder and prefer hard and sandy substrates rather than the soft and muddy habitats preferred by Southern Flounder (Stokes 1977; Nall 1979). It forms a minor component of commercial and recreational fisheries. Larger Gulf Flounder, like Southern Flounder, feed primarily on fish (70%) with the main invertebrates being penaeid shrimps and Blue Crab (Stokes 1977; Gilbert 1986). Similar to Southern and Summer Flounders, Gulf Flounders migrate offshore in the late fall and early winter to spawn in deeper waters of the continental shelf (Enge and Mulholland 1985). Gulf Flounder were therefore expected to be present on the nearshore sand shoals on the east coast of Florida during spring through fall.

Smooth Butterfly Rays *Gymnura micrura* are distributed from Maryland to Brazil and the Gulf of Mexico, from inshore sandy and muddy bottoms to a depth of 40 m along the continental shelf (Robins and Ray 1986; Snyder and Burgess 2016). Butterfly rays can use both benthic and pelagic habitats, as they are capable of an oscillatory swimming mode (i.e., flapping their pectoral fins) (Rosenberger 2001). In northeastern Brazil, they are year-round residents in nearshore waters and do not make extension movements (Yokota et al. 2012). They appear to have a low-diversity diet dominated by teleosts. For example, in northeastern Brazil they primarily consumed a single species of grunt (Yokota et al. 2013). As piscivores they have a higher trophic level compared to other batoids (Parsons 2017). Based on limited information from Brazil, we would expect them to be year-round residents off the east coast of Florida.

Bluntnose Stingray *Hypanus say* is distributed throughout much of the Western Atlantic from New York and the northern Gulf of Mexico to Venezuela. It is also widespread in the West Indies, and throughout the Antilles (Carlson et al. 2020). The species inhabits coastal waters to depths of 10 m (Robins and Ray 1986). It is a relatively small stingray species, growing up to 78 cm disk width. Both adults and juveniles frequent estuaries and surf zones (Murphy et al. 1997). The species is associated with sandy or muddy substrate areas and preys on fish, clams, worms and shrimps (Collins 2015). Bluntnose Stingrays are migratory only in the northern part of their range, moving deeper in winter months, but in Florida they are year-round residents off the east coast of Florida (e.g., in the Indian River Lagoon) (Snelson et al. 1989).

The Bullnose Ray *Myliobatis freminvillei* is distributed widely in temperate coastal waters of the western Atlantic from Cape Cod to central Brazil, and is found typically in depths less than 10 m (Bigelow and Schroeder 1953). Their diet is primarily mollusks (> 80%), benthic crustaceans, and some polychaetes (Bowman et al. 2000; Szczepanski and Bengtson 2014). In its northern range (i.e., Delaware Bay), Bullnose Rays make seasonal migrations in the summer to coastal waters for feeding and reproduction (Bigelow and Schroeder 1953). Migratory movements of Bullnose Rays are not known in Florida.

The Cownose Ray *Rhinoptera bonasus* is a benthopelagic ray found in shallow water to 22 m in coastal areas from New England to Northern Brazil (Robins and Ray 1986). Based on tagging studies, this highly migratory species has been shown to leave Florida for Yucatan each fall, with some tagged fish caught as far south as Trinidad and Northern Brazil. The Cownose Ray enters bays and estuaries and can form huge schools inshore where it feeds mainly on benthic invertebrates, including crustaceans and mollusks. It is implicated in damaging seagrass beds due to its excavating activities (Murphy et al. 2013; Last et al. 2016). Based on its highly migratory behavior, Cownose Rays are only expected to be present off the east coast of Florida during the warmer months of the year.

The Southern Stingray *Hypanus americanus* is a large (up to 200 cm) abundant ray, distributed in coastal areas from New Jersey to Southern Brazil to depths of 53 m (Robins and Ray 1986). This species is associated with sandy bottoms, seagrass beds, lagoons, as well as the face of reefs (Michael 1993). Common in bays and estuaries (Robins and Ray 1986), the Southern Stingray buries under the sand during the day and forages at night, usually in seagrass beds (Michael 1993). The species feeds mainly on bivalves and worms, but also takes shrimps, crabs, and small fishes, often by excavating depressions in the sand to expose invertebrates and small fishes (Lieske and Myers 2002). Southern Stingrays are considered year-round residents with some site fidelity in the Bahamas (Schwanck et al. 2020) and residents of reef areas in the Caribbean (Tilley and Strindberg 2013). Based on reported site fidelity, it was expected that Southern Stingrays would be year-round residents on the east coast of Florida.

The Clearnose Skate *Rostroraja eglanteria* is distributed in inshore waters throughout much of the Western North Atlantic, from Massachusetts to southern Florida, as well as the northeastern Gulf of Mexico (Smith 1997). It occurs typically in waters less than 50 m but ranges from 0 to 330 m depth (Smith 1997). It has a diet that includes decapod crustaceans, such as shrimps, krill, and fiddler crabs, as well as bivalve mollusks, polychaetes, squid, and small fishes (Fitz and Daiber 1963; Szczepanski 2013). Clearnose Skates are known to migrate inshore to offshore in the Gulf of Mexico and from North Carolina to Delaware but in eastern Florida they are reported to stay in nearshore waters year-round (Bigelow and Schroeder 1953).

14.2.2 Fish Capture and Tagging

Fish for tagging with acoustic transmitters were captured during bottom trawling activities in association with monitoring fish assemblages over the study shoals (Chapters 13). On capture, fish were immediately immersed in water in large flat-bottomed trays. Flatfishes and skates were measured for TL whereas rays were measured for disk width (DW, mm). Fish size was converted into body weight using known weight-length regressions and only fish meeting a threshold size/weight were retained for tagging. For this threshold, the tag was not more than ~2% of the body weight of the fish (Table 14-1).

Initially, Sonotronics acoustic tags were used to actively track fish (i.e., triangulate on individual fish to obtain a specific location). However, this active tracking method proved ineffective because of the wide extent of fish movement and lack of detections of fish on the shoals despite extensive tracking effort using a towed receiver. Therefore, in summer 2015, the fish telemetry study changed over to passively tracking fish that had been acoustically tagged with InnovaSea® (Vemco) transmitters, which were then detected using fixed receiver arrays (detailed below). Subsequent information in this chapter is based on these latter fish only, which uses methods complimentary to Iafrate et al. (2019, 2022).

Smaller fish were tagged with InnovaSea (Vemco) V9 69 kHz coded transmitters that were 9 mm in diameter, 24 mm in length, weighed 2.0 g in water, and had an estimated tag life of 651 days. Larger fish were tagged with V13 69 kHz coded transmitters that were 13 mm in diameter, 30.5 mm in length, 5.1 g in water, and had an estimate tag life of 1316 days. Tags were modified for external attachment (Hunter et al. 2003; Henderson and Fabrizio 2011) by inserting the entire tag into Tygon tubing of appropriate diameter, puncturing the end of the tubing on either side of the tag, and inserting Monel stainless steel wire through the holes. The tag was then attached through the dorsal musculature of the fish using two hollow tagging needles embedded in a wooden handle and spaced apart depending on the tag size. After inserting the tagging needles through the dorsal musculature, the trailing ends of the Monel wire were inserted into the hollow tagging needles and were used to guide the wire back through the musculature of the fish. The wire was then fitted with a small plastic plate and securely tied/twisted to the underside of the fish, with the transmitter secured on the dorsal surface of the fish. We externally tagged the fish because the procedure was very quick (i.e., < 2 minutes), did not involve invasive surgery associated with internal implantation of transmitters (as per our IACUC protocol permit), and has been used effectively on flatfishes in sandy environments previously (Hunter et al. 2003; Cadrin and Moser 2006). Tag

retention rates for the various species were unknown, as none of the fish were held in captivity. Tags were labeled with a return phone number in case any fish were recaptured. However, no fish tags were returned during the study.

Tagged fish were placed in a fish trap that was fitted with a mechanism that would open the trap door once it hit the bottom so that fish were released directly on the bottom, decreasing their chances of predation risk compared to releasing them at the surface. All fish were released close to the midpoint of the trawl path, with their release location recorded using the boat's GPS.

14.2.3 Acoustic Arrays

Tracking of tagged fish in the area of the sand shoals relied on an acoustic array of 57 receivers maintained by Eric Reyier and colleagues at the Kennedy Space Center (KSC) (Figure 18 in Iafrate et al. 2019), which we augmented with an additional 5 receivers in 2015 and then another 23 receivers in May 2017 (Figure 14-2), for a total of ~85 receivers. These receivers were incorporated in the Florida Acoustic Cooperative Telemetry (FACT) network and were therefore used to detect species tagged by other researchers as well. In addition, several other acoustic arrays along the East Coast of the USA and Canada are part of either the FACT or the Atlantic Cooperative Telemetry (ACT) networks (Figure 14-1) and several arrays in the Gulf of Mexico are part of the iTag network, and all were available to detect fish moving further afield than just in the immediate study area. In addition, a Wave Glider (i.e., an unmanned surface vehicle) fit with two acoustic receivers to detect acoustically tagged fish was deployed off Cape Canaveral eight times between November 2017 and August 2019 and was also able to detect tagged fish. Details of the glider deployment is provided in Iafrate et al. (2022).

All of these networks are an informal association of Atlantic and Gulf coast marine researchers using InnovaSea (Vemco) transmitters and receivers and allow sharing of data among researchers. In this manner, these networks broaden the reach of local and regional studies by including detections from receivers deployed outside an individual researcher's study area. By agreement, however, researchers only report on the fish that they tag and deploy and so this report is limited to those fish captured and tagged as a direct result of this research study. A complementary study on larger, mobile fish, primarily sharks, was conducted concomitantly in the same area by Iafrate et al. (2019, 2022).

14.2.4 Deployment of Acoustic Receivers

In total, 23 Vemco VR2W acoustic receivers were initially deployed to form a "curtain" around the study shoals (Figure 14-2), integrating them into the receivers already positioned by the KSC. The positioning of the receivers was based on an assumed 200–400-m detection radius (Iafrate et al. 2019), with the intention of being able to detect fish either leaving the shoals area after tagging or entering the area at some point in time after previously emigrating from it. However, fishing trawlers actively working in the area, as well as major storm events, displaced or removed some of the receivers. In addition, during the COVID epidemic, our ability to find and replace displaced/lost receivers was severely impacted because of field and personnel restrictions in place. We therefore considered the curtain to be "leaky", with the possibility of tagged fish being able to pass through the curtained area undetected. Therefore, the focus was on determining the overall use and residency of tagged fish on the sand shoals, and their use of the shoals relative to dredging.

Each VR2W receiver had its surface painted with anti-fouling paint and was wrapped in "Press-n-Seal" material to reduce biofouling. Each receiver was attached to a sand anchor (auger) using 2 m length of rope. A 10-inch float sprayed fluorescent orange was attached 1 m above the receiver as a subsurface buoy to aid in finding and recovering the receiver. In high current areas, it was also necessary to attach another heavy weight (concrete blocks, etc.) via chain to the sand anchor to either prevent the receiver

from being dragged from its position by currents or to minimize complete loss of the receiver (i.e., dragged up by a trawler).

All receivers were deployed and retrieved using SCUBA divers (scientific divers at the University of Florida [UF] certified by the American Academy of Underwater Scientists) and support vessels. Receivers were retrieved and data downloaded every 6 months, when possible, to ensure its hydrophone was not impaired by fouling organisms, which may lead to erroneous or missed data. Each receiver was immediately replaced by another receiver that had been cleaned of any fouling organisms and prepared for re-deployment, including a new battery.

14.2.5 Telemetry Analysis

14.2.5.1 Data Acquisition

Acoustic data was downloaded from individual receivers, whose locations were geo-referenced using high-resolution GPS. Downloads occurred either immediately at sea using Bluetooth and a field computer (Getac Toughbook) if necessary, or as soon as was possible following retrieval. Receivers were cleaned of fouling organisms, their batteries replaced, and then were redeployed when needed. Upon return to the laboratory, the downloaded data was archived onto network drives maintained at UF. All tag and receiver data were entered in the VEMBU data system used by the FACT network to facilitate data sharing. Data files from the VEMBU system detailing tagged fish information, deployment and retrieval data for all receivers, and all detections, were then uploaded to the Ocean Telemetry Network (OTN) site (www.researchspace.com) to ensure data security and archiving. This is an online data workspace where the FACT Data Team processes the uploaded files from all the individual receivers/arrays registered to enable researchers to receive extracts of their matched tag detections. This expands the “listening” area far beyond any one individual project.

14.2.6 QA/QC of Acoustic Data

All tag, receiver, and detection data (from processed FACT/OTN extracts) were initially prepared and imported into the *glatos* package in R (<https://github.com/ocean-tracking-network/glatos>). The *glatos* package is a R toolkit under development and is used by the Great Lakes Acoustic Telemetry Observing System (GLATOS) to process and analyze their acoustic data but is generally useful in analyzing acoustic data overall and was available to use after registering with GLATOS.

All detection data was initially run through *glatos* to check for false detections, or detections that were suspect (i.e., overlapping signals or “collisions” from two or more transmitters, only one detection in a location far away), were filtered in the database following methods in Pincock (2012). False detections were not deleted during this filtering but instead were coded in an additional column as 1 or 0, depending on whether each individual detection passed the false detection filter or not. Typically, less than 1% of detections are filtered out of data due to false detections. Detections from transmitters that did not change position for long periods of time were also inspected for the potential to be a shed tag or a tag dropped through a mortality event, which could result in the tag resting on the bottom with continuing detections at the nearest receiver; these data were filtered out of the data and not used in analyses.

14.2.7 Statistical Analysis of Telemetry Data

Once filtered for false detections and dropped tags, data was summarized for each tagged fish for the following: 1) the number of unique receiver stations it visited, which gives an overall indication of its space use; 2) total number of detections prior to tag life expiring; 3) total unique days that the detections represented (days detected); and 3) days at liberty, which is the number of days between the release date and the date that the fish was last detected. Days at liberty was restricted to the maximum life (in days) of the tag, which was based on the size of the tag, power, and interval. For the V9 and V13 coded

transmitters used in this study, tag life was expected to be 651 and 1,316 days, respectively, which was set as the maximum days at liberty because after the tag expires the fish could be present but not able to be detected. In addition, no fish were known to have been directly removed from the detection area off Cape Canaveral through recreational or commercial fishing activities (i.e., this would shorten the days at liberty relative to the tag life span). Abacus plots were constructed to show the presence or absence of each individual at each receiver as a function of time since release (Binder et al. 2018).

An overall residency index (I_R) was then calculated as days detected divided by the total number of days that the fish was monitored in the study (i.e., days at large) ($I_R = D_d/D_t$) (Eqn. 1.2 in Kraft et al. 2023). This index is an approximation for the amount of time the fish spent in the study area. The I_R varies between 0 and 1, with zero indicating that the fish was never detected (i.e., 0 residency) and one indicating that the fish was detected every possible day that it was at large (i.e., fully resident; has remained in the study area for the duration of its tag life).

Temporally and spatially clustered detections for each fish were then condensed to distinct detection events. These events were based on a user-defined threshold of time that must occur before events were considered to be distinct because of multiple, sequential detections at the same receiver by the same fish. This threshold was set as 3,600 sec in the *'time_sep'* of the *'detection_event'* R code in the *glatos* package. These detection events were then used to calculate a second form of the residence index based on the number of days the fish was detected (D_d) divided by the detection interval (D_i), which was the number of days between its first and last detection (Eqn. 1.1 in Kraft et al. 2023); this index was referred to as the location index (I_L) to avoid confusion with the previous form of the residency index. The location index therefore indicated where a fish was spending most of its time when it was in the study area. The I_L was calculated using the *'residence_index'* in *glatos* using the Kessel et al. (2016) method (Nunes 2019). A boxplot of the I_L was plotted as a function of receiver locations, as well as a function of individual tagged fish, to visualize where fish were being detected overall in the study area and their use of specific sites.

Any seasonal distribution of tagged fish along the coast, and specifically in the study area of the shoals, was examined by comparing the presence/absence and duration of their occupancy on a seasonal basis, where spring was March–May, summer was June–September, fall was October–November, and winter was December through the following February.

14.2.8 Fish Distribution and Movement Relative to Dredging Events

Distribution and movement of tagged fish on CSII-BA, the dredged shoal, were compared relative to a dredging event and movements of the same species on reference shoals (CSII, Chester, and Bull Shoals). This dredging event was the second (referred to as Dredge-2) of two¹ dredging events that occurred during the study and took place from 3 February to 11 April 2018 (spring 2018).

14.2.9 Permits

Permits for sampling of fish occurred under LOAs with NOAA/NMFS for both the standardized trawling and the trawling with the TED closed. Field sampling and laboratory processing of fish samples occurred under IACUC protocol #201709892 and #202009892 at UF.

¹ The first dredging event (referred to as Dredge-1) occurred from 27 November 2013 to 22 April 2014, encompassing winter 2013/14 and spring 2014.

14.3 Results

14.3.1 Tagged Fish Metrics

In total, 57 fish were tagged between 3 August 2015 and 14 June 2019, including 4 Gulf Flounders, 8 Southern Flounders, 3 Summer Flounders, 7 Bluntnose Stingrays, 2 Bullnose Rays, 1 Cownose Ray, 1 Clearnose Skate, 1 Southern Stingray, and 30 Smooth Butterfly Rays (Table 14-2). Gulf Flounders were on average 310 mm TL (range 290–369 mm TL), Summer Flounders averaged 321 mm TL (range 295–343 mm TL), and Southern Flounders similarly averaged 327 mm TL (range 295–355 mm TL). Smooth Butterfly Rays averaged 418 mm DW (range 298–770 mm DW), Bullnose Rays were 340 and 372 mm DW, and the Cownose Ray was 455 mm DW. The single Southern Stingray tagged was 340 mm DW and the Bluntnose Stingrays averaged 388 mm DW (range 220–775 mm DW). The Clearnose Skate tagged was 455 mm DW.

14.3.2 Tag Detections and Residency Time

In total, there were 35,708 detections reported for the 57 fish tagged during this study, with detections occurring between 4 August 2015 and 12 January 2021. Of those detections, 288 were filtered out as false detections (0.81%). In addition, one Smooth Butterfly Ray was suspected of either dropping its tag or suffering mortality a few days after being tagged and released because of a lack of movement over an extended period of time with detections continuing at the same receiver station. This fish was therefore removed from the analysis, including 17,286 detections associated with this one tag (fish). This resulted in a total of 18,134 detections for 56 tagged fish used in the analysis.

Of the 56 tagged fish analyzed in the study, 45 fish were detected at 1 to 17 unique stations, and from 1 to 2,485 times (Figure 14-3A, Table 14-2). The majority of tagged fish were only detected at three or fewer unique receiver stations, with only rays and skates, including Smooth Butterfly Rays, detected at four or more stations. Fish were detected while in the receiver array from 1 to 31 days (Figure 14-3B, Table 14-2). The majority of all tagged fish were detected for three or fewer days in the array, with mostly rays and skates, including Smooth Butterfly Rays, detected over a greater number of days. The greatest number of days that a fish was detected in the study area was a Bluntnose Stingray that was detected in the study area for 17 days (Table 14-2). As with the previous indicators, the residency index (I_R) also showed that the majority of tagged fish did not remain in the study area, as evidenced by the extremely low values for I_R (i.e., most were < 0.02 on a scale of 0 to 1) (Table 14-2).

Of the 56 tagged fish in the study, 12 were never detected in the array after they were released. Fish tagged and not detected again included 5 of the 15 tagged flounders, and 7 of the 41 rays tagged, including 6 of the 29 Smooth Butterfly Rays tagged.

14.3.3 Location Index and Movements of Flatfishes

For all flatfishes combined, 1,599 detections were condensed down to 500 distinct detection events. The location index for flatfishes was difficult to interpret because of the overall low detection levels. Overall, the median location index for all the flatfishes was zero (Figure 14-4), indicating that most of the flatfishes were not detected over the majority of the stations that at least one flatfish was detected at. This was also borne out by 7 of the 10 flatfishes having an outlying location index of 1, which indicated that at least one flatfish was only detected at one specific receiver station while they were present in the array.

Overall, flatfishes were detected sporadically over the late fall to spring seasons, and less so in the summer season (Figure 14-5). Most flatfishes were detected in the northern portion of the array.

14.3.3.1 Location and Movements of Flatfishes Relative to Dredging Events

One Summer Flounder was only ever detected in CSII-BA and was present there during the second dredging event in early April 2018, was detected in CSII-BA again after dredging was complete, was detected again a few times up to early May 2018, and then returned to CSII-BA again in August of 2018 (Figure 14-6). No other flatfishes were detected in CSII-BA during the dredging event.

14.3.4 Location Index and Movements of Smooth Butterfly Rays

For all Smooth Butterfly Rays combined, 8,729 detections were condensed down to 241 distinct detection events. The location index for Smooth Butterfly Rays indicated that they were detected primarily at receiver stations CC6 and CC1 while they were in the study area (Figure 14-7). These two stations were located near the mouth of the channel leading into Port Canaveral (Figure 14-2). Smooth Butterfly Rays were also detected relatively often within the dredged area of CSII-BA (i.e., DRE7) relative to other stations (Figure 14-7).

Smooth Butterfly Rays were detected throughout the entire receiver array but primarily in the southern portion of the array during the summer season in July (Figures 14-8, 14-9). Both the abacus plot and the mapped detections indicated consistent use of the area at the mouth of Port Canaveral and the southern extent of the UF/KSC array (Figure 14-8). At least one Smooth Butterfly Ray was also detected traveling south of the main UF/KSC array in June (Figure 14-10).

14.3.4.1 Location and Movements of Smooth Butterfly Rays Relative to Dredging Events

A total of six Smooth Butterfly Rays were detected within the area of the dredged shoal (CSII-BA) over the study period. Most Smooth Butterfly Rays using CSII-BA were detected from August 2015 to January 2016, May 2016, and August–September 2017 (Figure 14-8). For example, Smooth Butterfly Ray 3 was tagged and released in CSII-BA in early August 2015 (16 months after the first dredging event), moved northward off the cape over the next 10 days, and then was detected again in CSII-BA in September 2015 (Figure 14-11).

However, Smooth Butterfly Rays were not detected in CSII-BA prior to or immediately after the second dredging event in spring 2018 (Figure 14-8), nor were any Smooth Butterfly Rays detected anywhere else within the study area in 2018 despite at least 11 rays still having active tags (estimated based on their tag life). Therefore, the absence of Smooth Butterfly Rays from CSII-BA in spring 2018 could not be ascribed to an impact from the second dredging event since they were also not present at non-dredged reference sites. Tagged Smooth Butterfly Rays were again detected in CSII-BA in spring 2019 (Figure 14-12), 10 months following the second dredging event.

14.3.5 Location Index and Movements of Other Rays and Skates

For Bluntnose Stingrays, 5,751 detections were condensed to 193 distinct detection events. The location index for Bluntnose Stingrays indicated that individuals were located at a variety of stations when present in the study area (Figure 14-13). Bluntnose Stingrays were present in the study area in the winter, spring, and summer seasons; they were not detected in the fall of either 2018 or 2019 (Figure 14-14). Most detections of Bluntnose Stingrays were concentrated towards the southern extent of the receiver array (Figures 14-15, 14-16).

For all other rays and skates, 1,969 detections were condensed to 116 distinct detection events. Bullnose Stingrays, a Cownose Ray, and a Clearnose Skate were all detected in the study area in winter 2018; a single Southern Stingray was detected in the study area in summer 2016 (Figure 14-17). The two Bullnose Stingrays were detected within CSII-BA only prior to the start of the second dredging event and then were not detected in the study area again (Figure 14-18). The one Southern Stingray was detected in

CSII-BA from April to August 2016, a time period between the two dredging events (Figure 14-19). The single Cownose Ray was detected primarily in the most southern extent of the receiver array, south of CSII-BA (Figure 14-20). The Clearnose Skate was detected within CSII-BA and just south of that during spring of 2018 (Figure 14-21).

14.3.5.1 Location and Movements of Other Rays and Skates Relative to Dredging Events

Bluntnose Stingrays were detected in CSII-BA prior to and during active dredging, as well as approximately 1 year later (Figure 14-15). They were not detected within CSII-BA immediately after dredging was completed but they were also not detected anywhere in the study area including non-dredged areas during the same time period. This indicated that the dredging event in spring 2018 could not be directly responsible for a lack of detections immediately following the dredging event. They were detected within CSII-BA, and in more northern portions of the study area, in fall 2018 following the dredging event. One Bluntnose Stingray was detected within CSII-BA a couple of months prior to the second dredging event and then during active dredging moved back-and-forth between CSII-BA and receiver stations within CSII, adjacent to CSII-BA (Figure 14-16).

Bullnose Stingrays and the Southern Stingray were detected within CSII-BA more than at other stations in the study area but at a time period only prior to the second dredging event (Figures 14-18, 14-19). Similarly, the Cownose Ray was only detected prior to the second dredging event but was not detected in CSII-BA at all (Figure 14-20). The effect of dredging activities on these three species could therefore not be determined.

The single Clearnose Skate was detected within CSII-BA and stations immediately adjacent to it during active dredging within CSII-BA in spring 2018 (Figure 14-21). This indicated that the skate was not directly impacted by the dredging activity and continued to use the area; it also did not avoid CSII-BA relative to adjacent stations. It was not detected at any other receiver stations, including any other FACT or ACT receivers.

14.4 Discussion

Overall, the demersal fish species used as focal species in this study showed low residency over the sand shoals. On average, the majority of the flatfishes, rays, and skate were detected over the sand shoals < 3% of the time that they were at large based on their tag life. This low residency indicated that the sand shoals and surrounding area were only a small portion of the total area off the east coast of Florida that they used over the time period of the study. This low residency rate may, in a small part, be due to the UF/KSC combined receiver array ultimately being a “leaky” curtain of receiver stations for our smaller tagged demersal fishes because of the range extent of their smaller acoustic tags. However, the array was considered to be “dense” during the study period with 70–90 receivers deployed over the sand shoals and in immediately adjacent areas depending on the specific time period. This density provided a high likelihood that, if the tagged demersal fishes were resident on the sand shoals, they would be detected adequately enough to reflect a higher residency index, which was not observed. The dynamic use of the area by fishes was also observed in the location index, an indication of where the fish were located when they were actually present in the study area, which determined that most fish were detected at only a few receiver stations and stayed a relatively short period of time (i.e., mostly days not months).

Iafrate et al. (2019, 2022) similarly found that acoustically tagged fish exhibited high mobility and low site fidelity. Smaller demersal fishes such as Spot *Leiostomus xanthurus* and Atlantic Croaker *Micropogonias undulatus* in the study area dispersed away from their original release sites, moving > 30 km away over 5–7 months. In addition, one tagged Roughtail Stingray *Bathytoshia centroura* was noted

as having wide-ranging movements as it remained in the study shoal area for only a day before moving south, then returning to Canaveral, and finally migrating north 1,400 km (Iafate et al. 2019).

The focused detections of Smooth Butterfly Rays at the entrance to Port Canaveral was interesting because these fish have been noted as being relatively rare but year-round residents of the Indian River Lagoon (IRL) (Snelson and Williams 1981). Roskar et al. (2021) also noted the presence of Smooth Butterfly Rays in the spring, summer, and fall in the lower IRL. These fish most likely move between the IRL and the nearby sand shoal areas, but this hypothesis was not confirmed during the study because no Smooth Butterfly Rays were detected on receivers in the IRL. Bluntnose Stingrays were also found to be year-round residents in the IRL (Snelson and Williams 1981; Roskar et al. 2021), with parturition occurring in lagoonal waters during May and June (Snelson and Williams 1981). Cownose Rays were also known to use the IRL during August to November but were relatively uncommon (Snelson and Williams 1981). The IRL is known to be an area used by a variety of elasmobranchs as a nursery, where most species occur over a restricted period of the year for parturition (Snelson and Williams 1981; Roskar et al. 2021). For rays, it would be beneficial to have a series of receivers monitoring the mouths of the inlets leading into the IRL, which would provide information on their passage into and out of the IRL relative to the adjacent coastal areas that include the sand shoals.

Tagged fish that were not detected at any of the stations in the extensive receiver array of the study area were assumed to have moved out of the study area immediately after being tagged and released. In addition, these fish were not detected at any of the stations associated with other FACT, ACT, iTag, or Wave Glider receivers. Alternatively, these fish could have shed their tags or expired in a location where the tag was not detected by any of the receivers. The one Smooth Butterfly Ray that either shed its tag or experienced mortality (i.e., tag was resting on the bottom in both cases) was notable due to the extended period of no movement by the tag, but it was the only fish that exhibited clear signs of a stationary tag.

14.4.1 Overall Residency and Movement Relative to Dredging Events

The overall expectation was that demersal fishes can be negatively affected by dredging activities, either through direct physical impact, feeding disturbance, increased turbidity, etc. (Wenger et al. 2017). However, based on the acoustically tagged, demersal fishes tracked in this study, that expectation appeared not to be the general case; however, our sample size was relatively low so the results should be interpreted conservatively.

The presence of a tagged Summer Flounder within the dredged shoal (CSII-BA) during, immediately after, and again months after the second dredging event indicated that the dredging activity did not directly cause Summer Flounder to leave the area while active dredging was occurring, nor prevent them from returning into the area after dredging was completed.

Detection data for Smooth Butterfly Rays was less conducive to examining an effect due to dredging because of the timing of their release. However, Smooth Butterfly Rays were detected within CSII-BA at a minimum of 10 months after the second dredging event, which indicated that any potential impact due to dredging was relatively short lived. Given their residency in this region, they serve as a metric to the degree of impact of the dredging activity.

Similarly, detection data was relatively sparse for Bluntnose Stingrays, although they also appeared to not be directly impacted by dredging since one was detected in CSII-BA prior to and during active dredging and returned to the dredged shoal again 1 year later. This finding should be considered preliminary, however, since only seven Bluntnose Stingrays were tagged in this study. In addition, the potential impact of dredging on Bullnose Ray, Cownose Ray, Southern Stingray, and Clearnose Skate could not be examined due to low sample sizes of acoustically tagged fish. Only one fish, the Clearnose Skate, was detected within CSII-BA during active dredging, indicating that it was not directly impacted. This

observation may relate to this fish's habit of burying in sand during the day and foraging at night (Packer et al. 2003).

Similarly, Iafate et al. (2019) concluded that based on acoustically tagged fish there was no evidence that the fish used the dredged shoal (CSII-BA) any differently than Chester Shoal, a control/reference shoal. This conclusion primarily was based on the number of unique tagged species, number of individuals detected, and amount of time spent being comparable between the dredged and control shoals.

14.5 Acknowledgments

This research would not have been possible without the collaboration of Jim Stringer of the Florida Research Group. The Florida Research Group gave us exclusive contract use of the research vessel for the duration of the field work, which allowed us the flexibility to plan and schedule field sampling around inclement weather and personnel availability. For the telemetry research, we specifically thank Eric Reyier and Bonnie Ahr of the KSC and Joe Iafate and Stephanie Watwood of the Naval Undersea Research Center for their collaboration in the placement of the receiver stations and their assistance in retrieving wayward receivers. We also thank all the biologists and UF-certified scientific divers involved in deploying and maintaining the receivers and tagging the fish, including Todd Van Natta, Alex Linares, Mike Dickson, Bethan Gillett, Monica Ditch, Doug Marcinek, Shea Husband, Larry Meyer, Alexandra Marshbanks, Kyle Cahill-Patray, Jessica Van Vaerenberg, Karen Trainer, Geoffrey Smith, and Cheryl Thacker.

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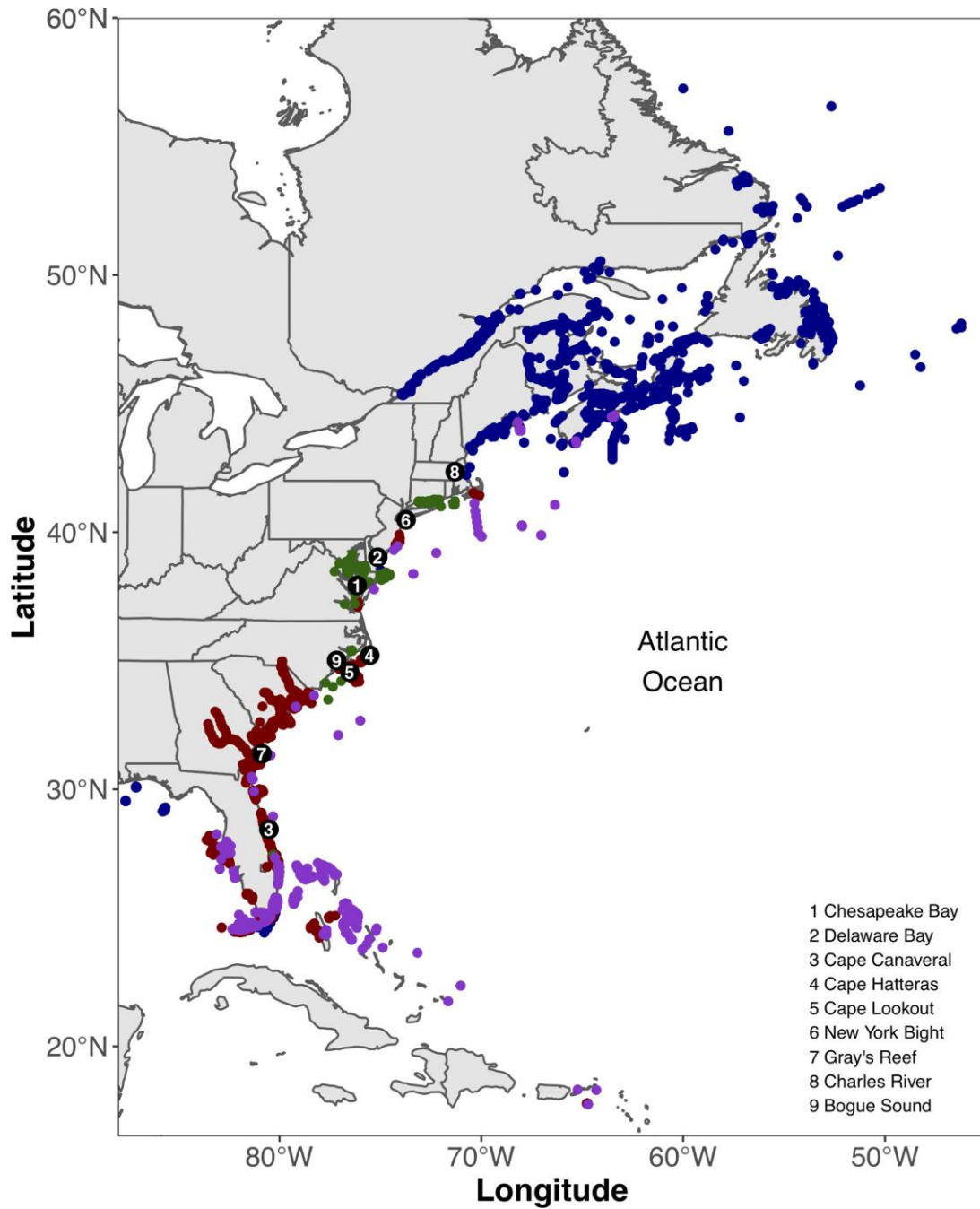


Figure 14-1. Location of receiver arrays in the Atlantic Cooperative Telemetry (ACT) and the Florida Acoustic Cooperative Telemetry (FACT) network along the U.S. East Coast.

The combined receiver array used in the study that was deployed and maintained by the Kennedy Space Center and the University of Florida is indicated by a "3" for Cape Canaveral (see Figure 14-2 for more details). Reprinted from Bangley et al. (2020) with permission.

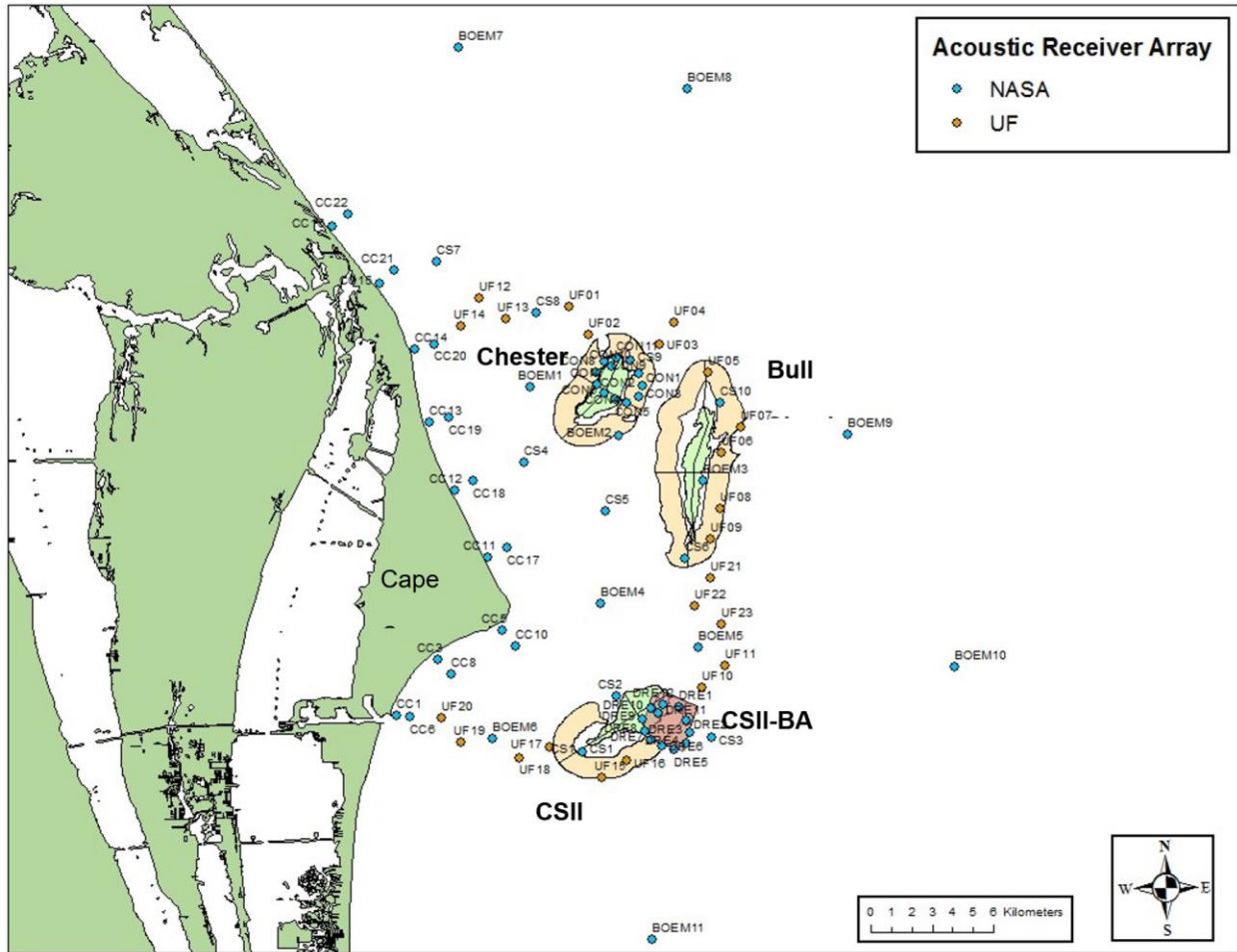


Figure 14-2. Cooperative receiver array used in the study area off the east coast of Cape Canaveral. Receivers deployed and maintained by the Kennedy Space Center (NASA) are blue and receivers deployed by the University of Florida (UF) are orange. Features of each shoal are demarcated by ridge (light green), swale (beige), and quadrants (thin black line bisecting shoal) (details provided in Chapter 1). The borrow area of Canaveral Shoal II (CSII-BA) is red.

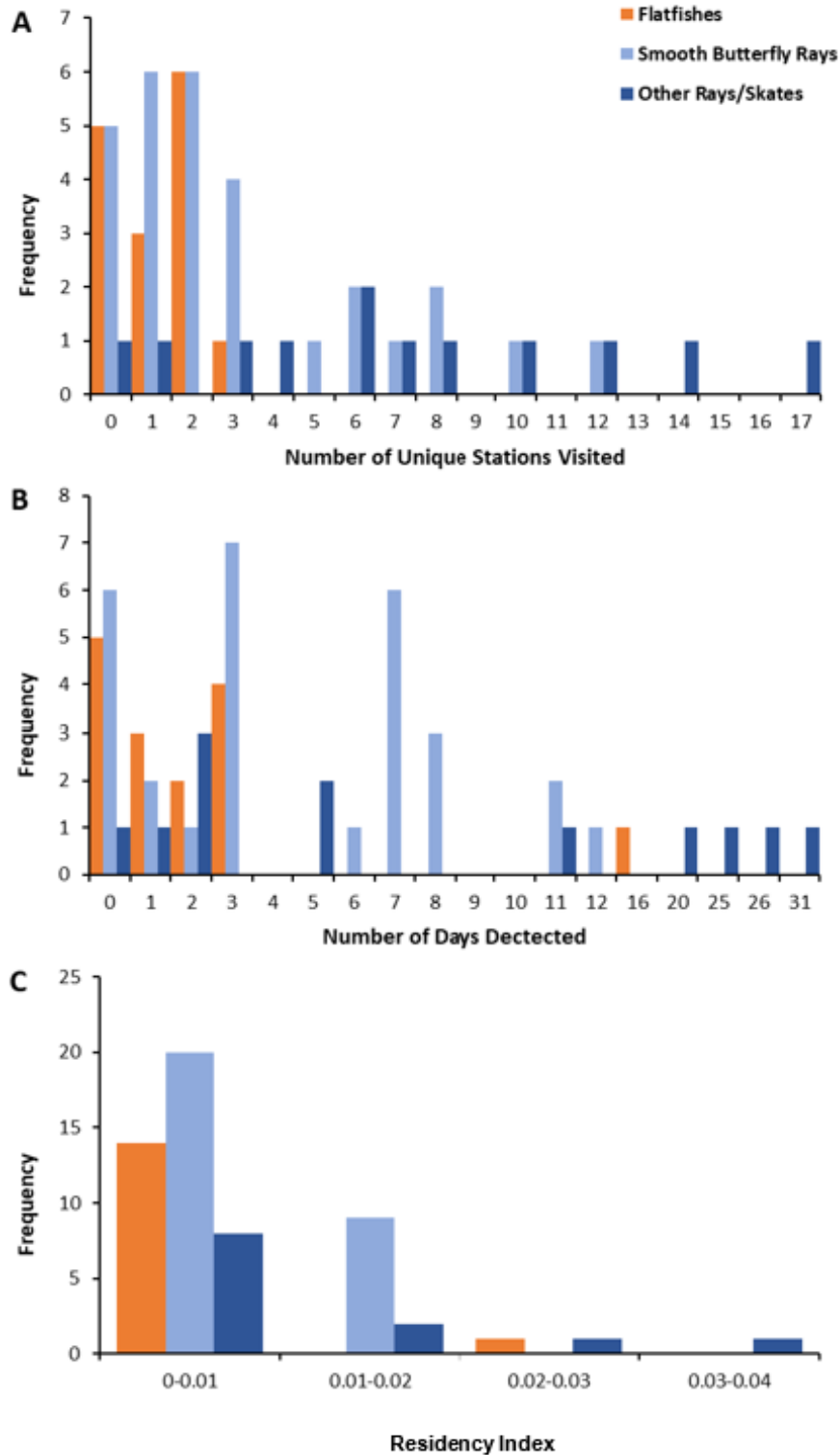


Figure 14-3. Detections of tagged fish in receiver array off of Cape Canaveral, Florida, for flatfishes, Smooth Butterfly Rays, and other rays/skates.

A) Number of unique receiver stations visited; B) number of days detected in the study area; and C) general distribution of residency index values across the study area and period.

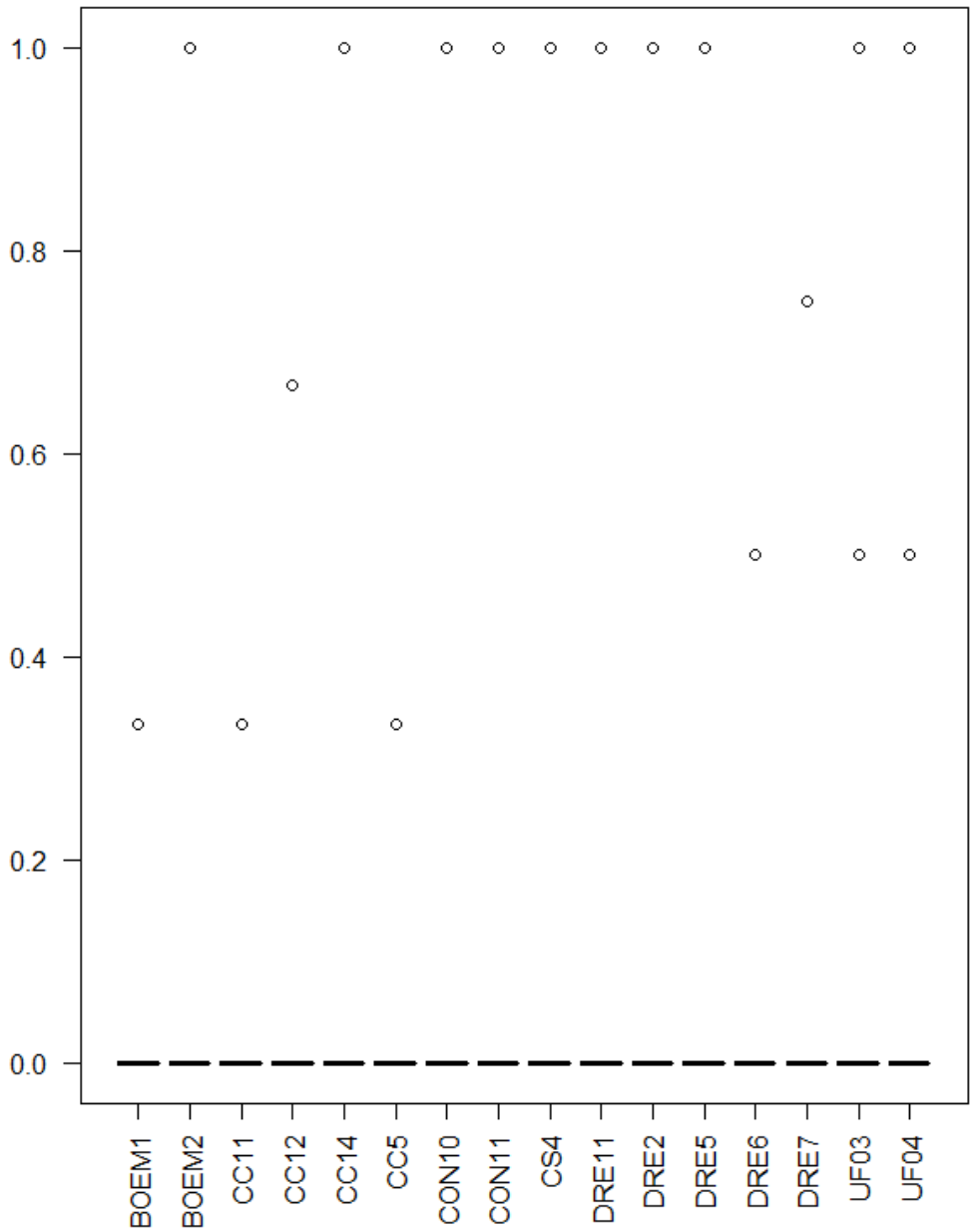


Figure 14-4. Location index for flatfishes tagged during the study period and detected within the receiver array off the east coast of Florida.

Boxplots show a solid horizontal line for the median location index and open symbols represent outliers.

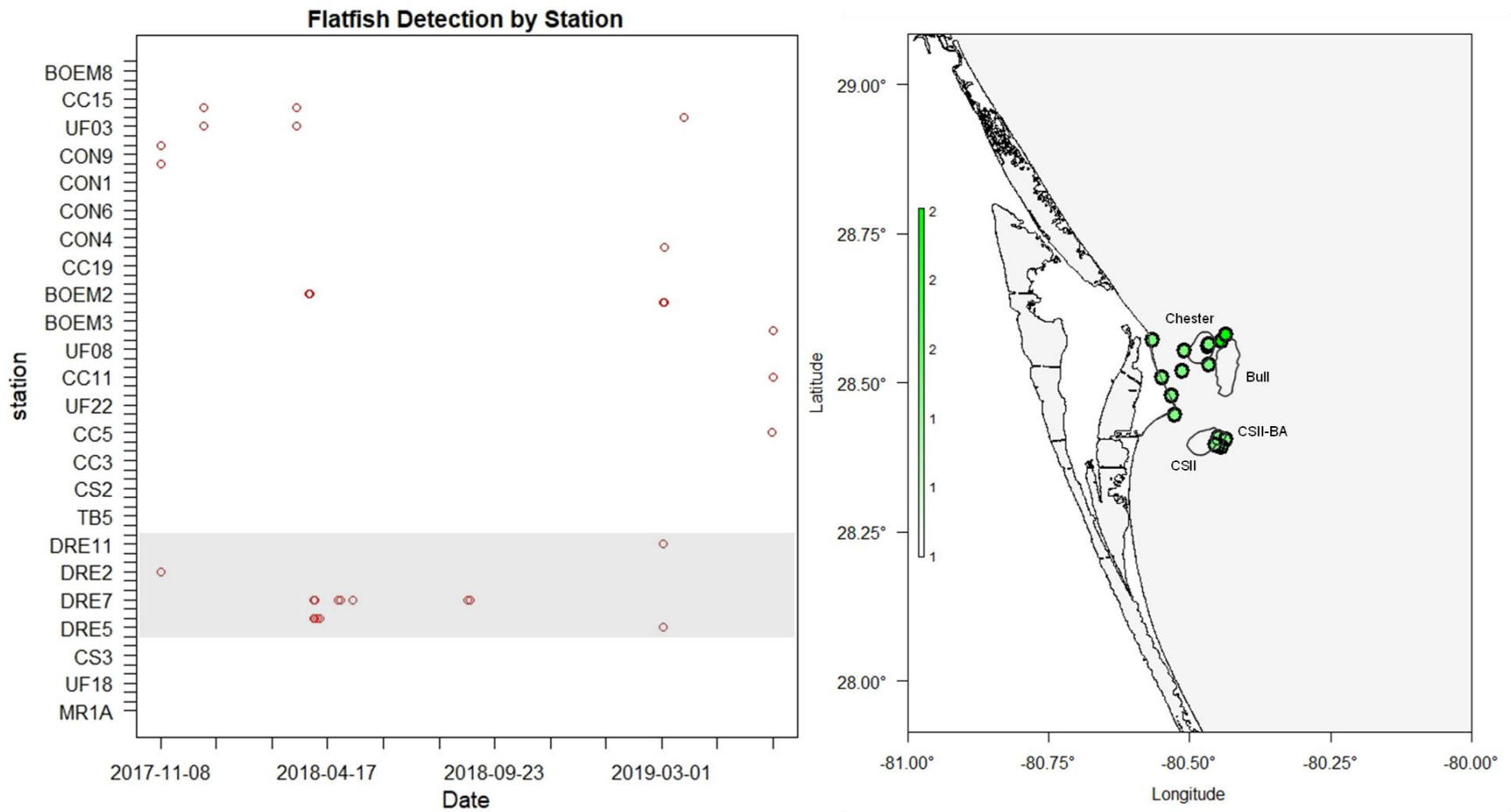


Figure 14-5. Abacus plot showing receiver stations where tagged flatfishes were detected over time and location of the detections off the east coast of Florida.

Receiver stations are, approximately, ordered from north (top) to south (bottom). Stations within CSII-BA are indicated by the shaded grey box.

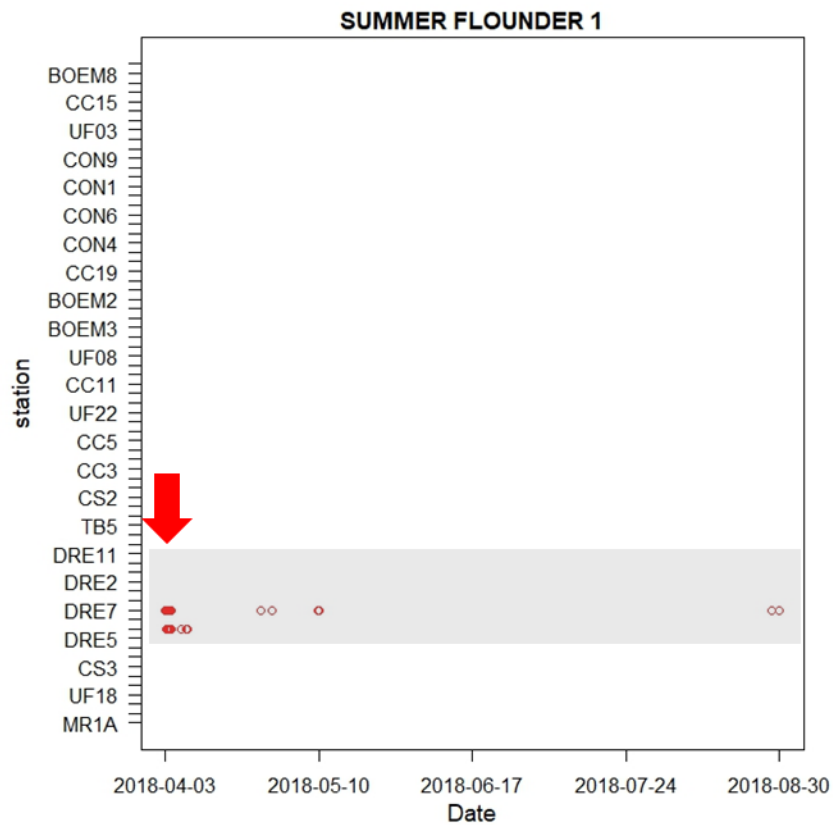


Figure 14-6. Abacus plot showing receiver stations where Summer Flounder 1 was detected within CSII-BA (dredged shoal) over a period of time that included active dredging, immediately post dredge, and months after dredging was completed.

Receiver stations within CSII-BA are indicated by the shaded grey box. Red arrow indicates period of active dredging within CSII-BA.

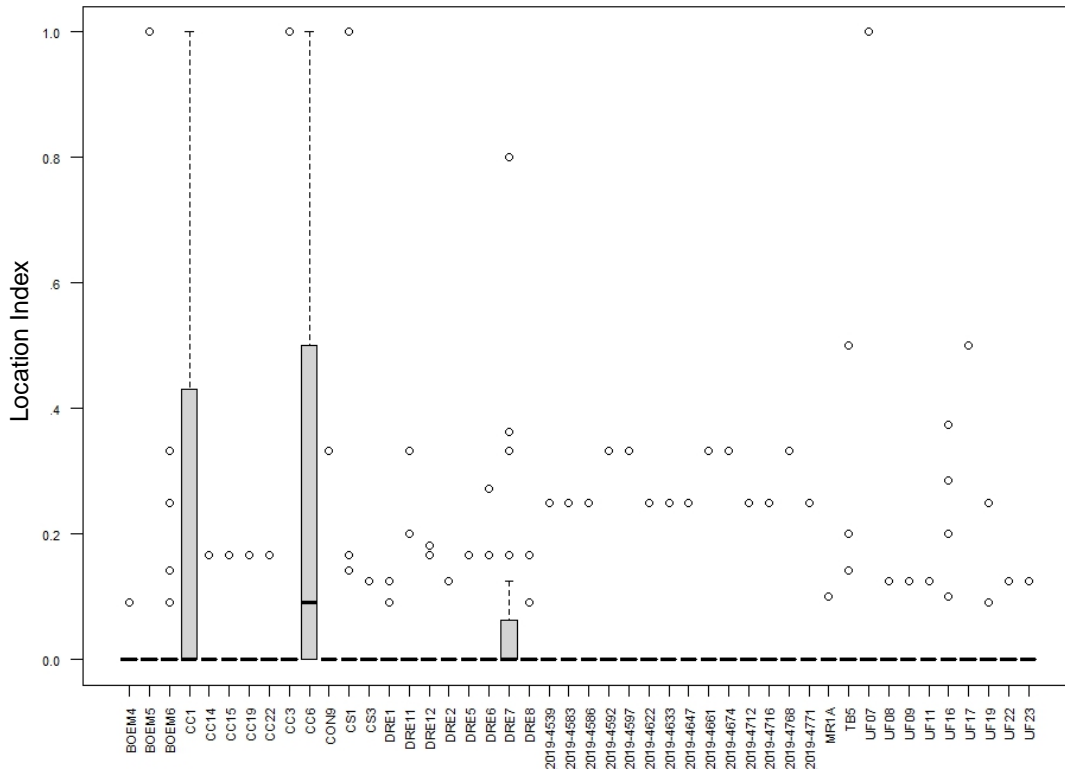


Figure 14-7. Location index for Smooth Butterfly Rays tagged during the study period and detected within the receiver array off the east coast of Florida.

Boxplots show a solid horizontal line for the median location index, grey box represents the 25th and 75th percentiles, whiskers represent 95th percentile, and open symbols represent outliers.

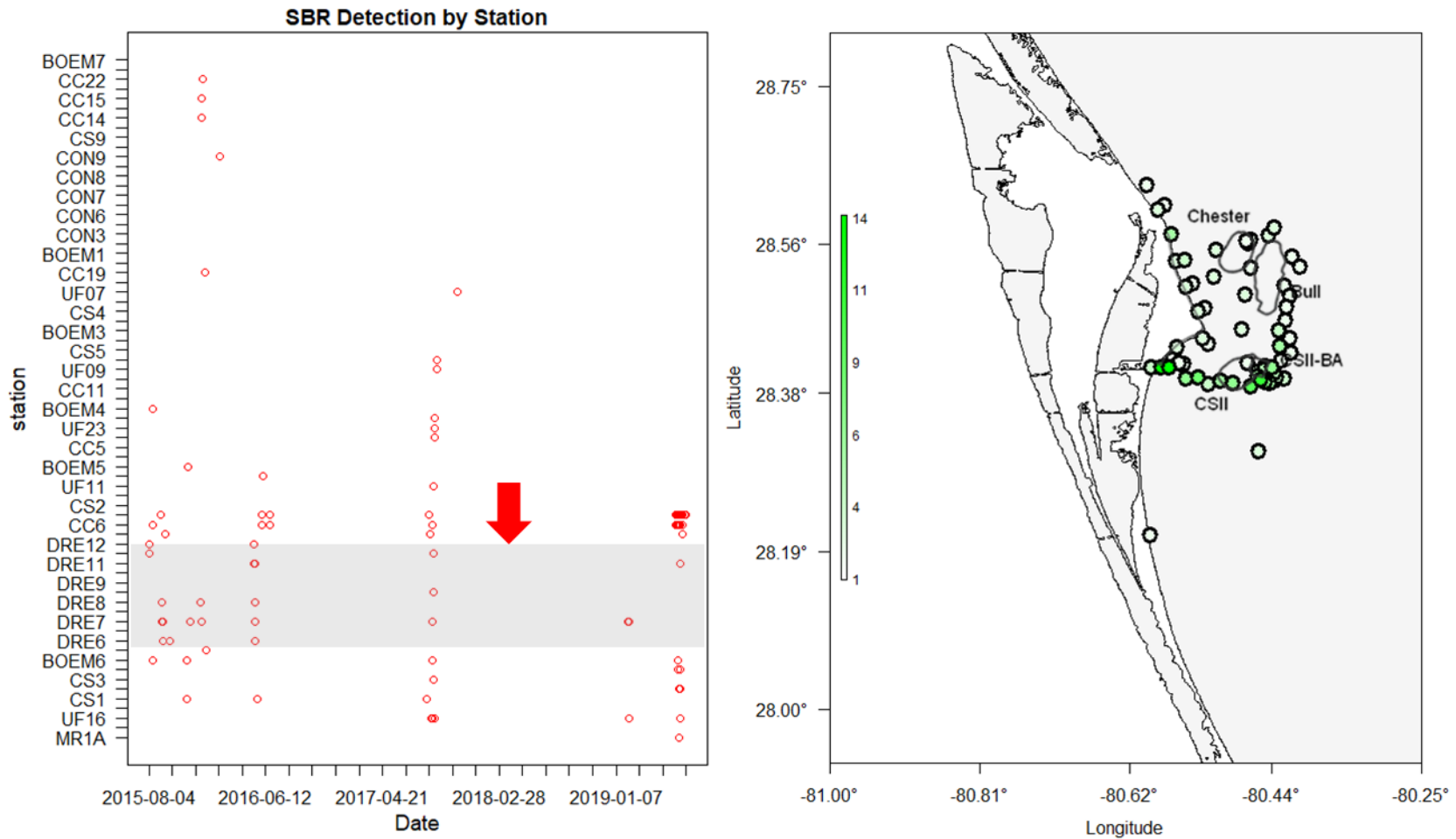


Figure 14-8. Abacus plot showing receiver stations where tagged Smooth Butterfly Rays were detected over time and location of the detections off the east coast of Florida.

Receiver stations are, approximately, ordered from north (top) to south (bottom). Stations within CSII-BA are indicated by the shaded grey box. Red arrow indicates period of active dredging within CSII-BA.

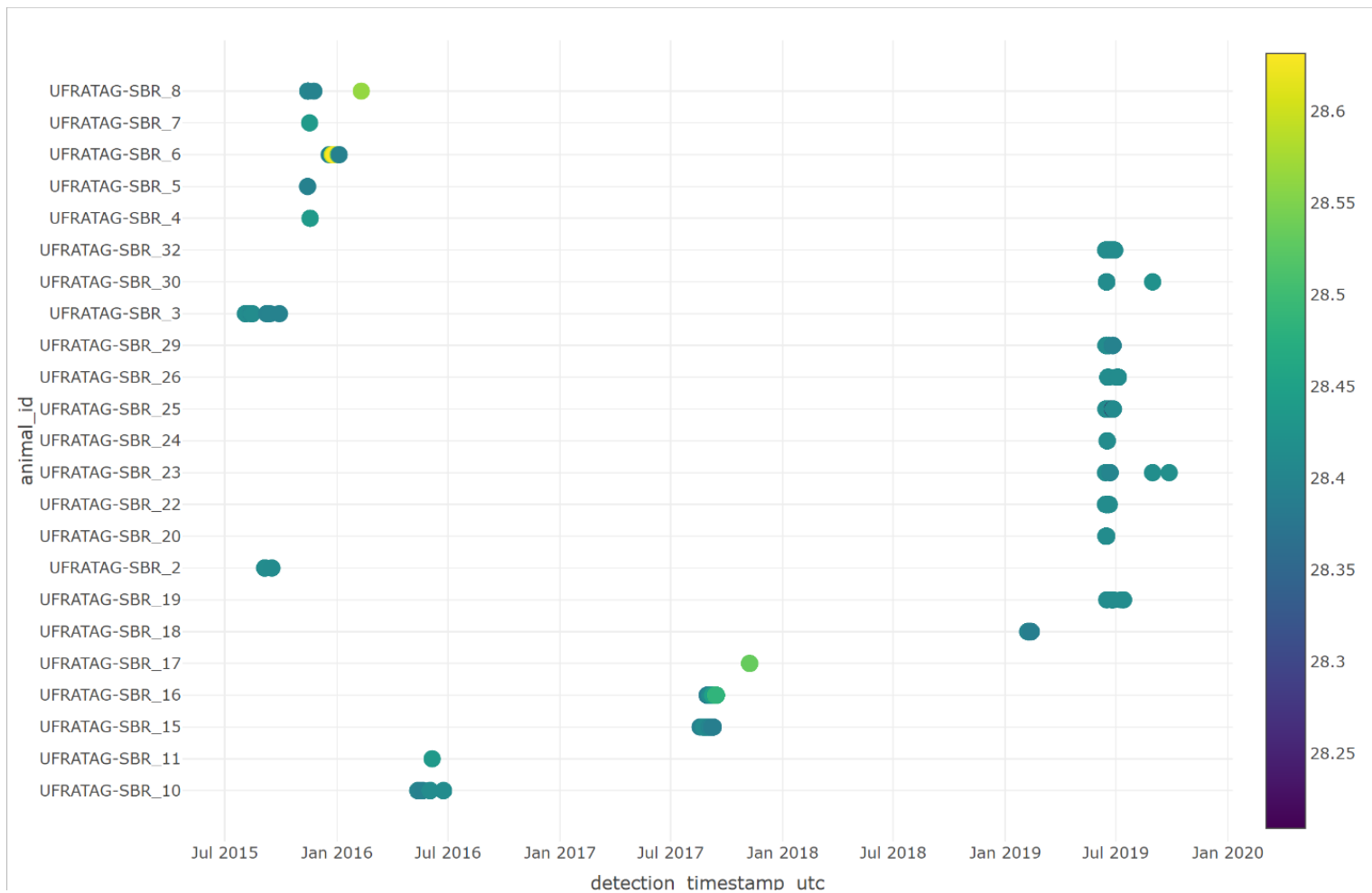


Figure 14-9. Abacus plot showing detections of tagged Smooth Butterfly Rays as a function of time and latitude off the east coast of Florida. Latitude ranges from yellow in the north to dark blue in the southern portion of the receiver array.

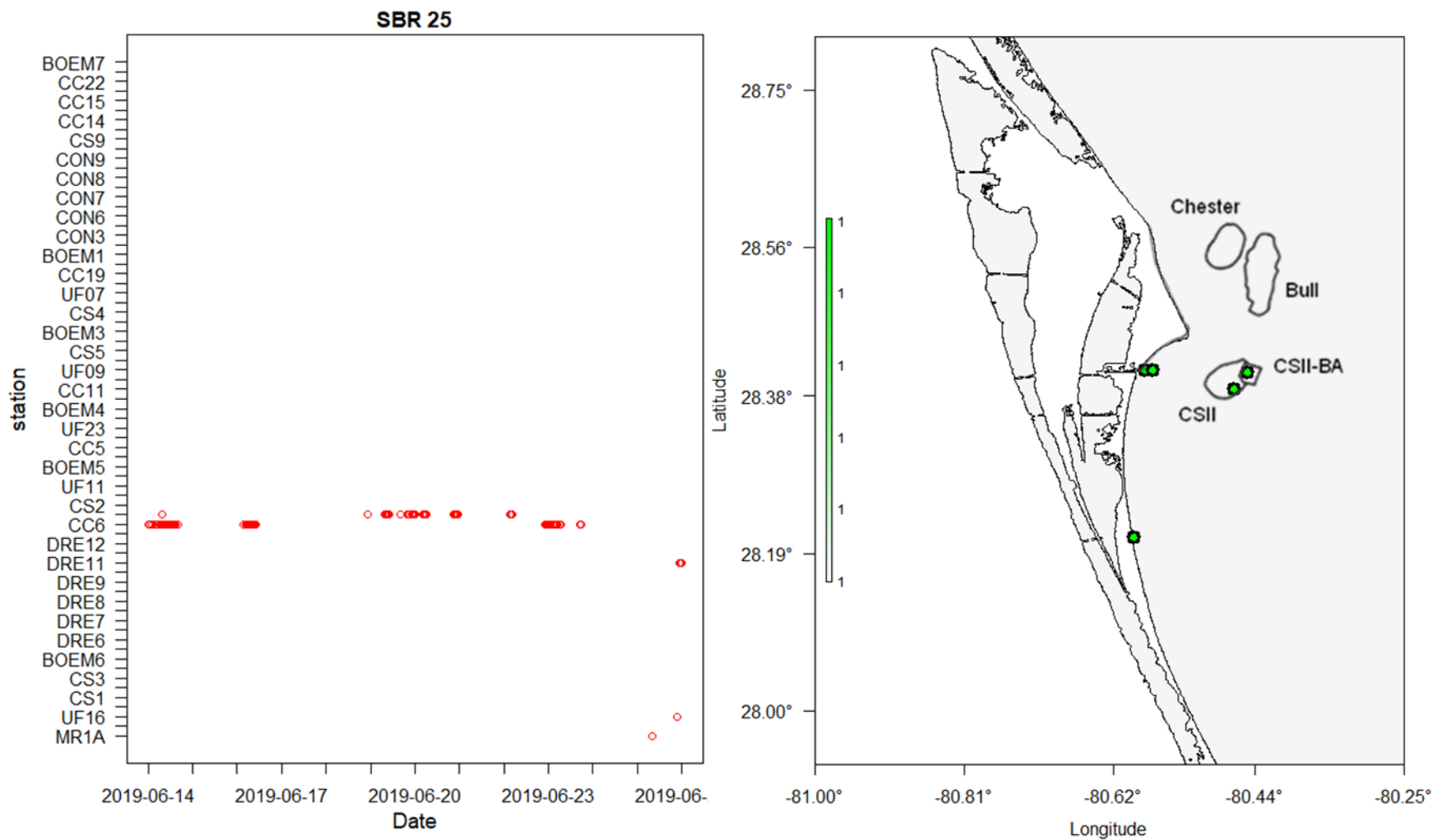


Figure 14-10. Abacus plot showing receiver stations where Smooth Butterfly Ray 25 was detected over time and location of the detections off the east coast of Florida.

Receiver stations are, approximately, ordered from north (top) to south (bottom).

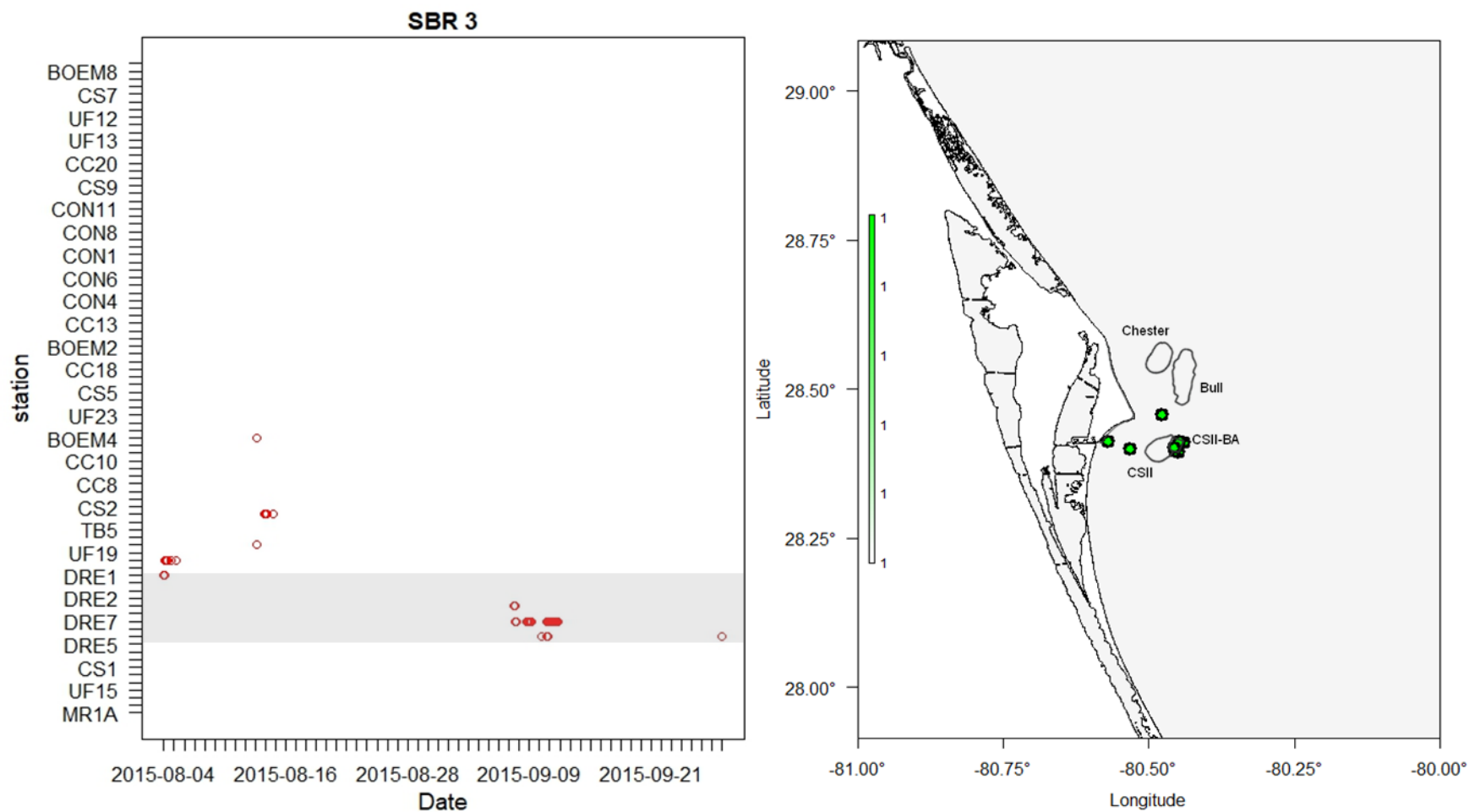


Figure 14-11. Abacus plot showing receiver stations where Smooth Butterfly Ray 3 was detected over time and location of the detections off the east coast of Florida.

Receiver stations within CSII-BA are indicated by the shaded grey box.

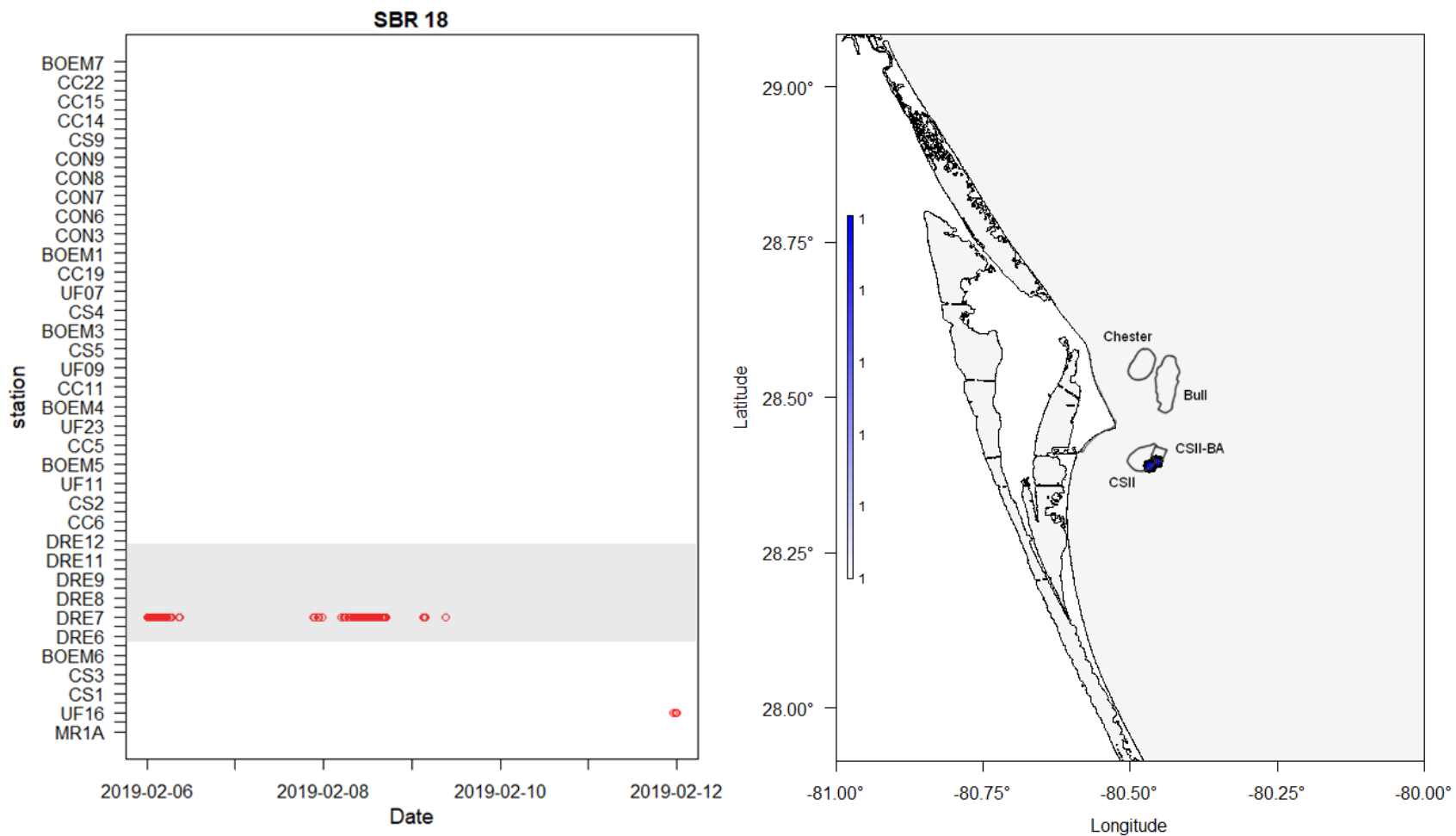


Figure 14-12. Abacus plot showing receiver stations where Smooth Butterfly Ray 18 was detected over time and location of the detections off the east coast of Florida.

Receiver stations within CSII-BA are indicated by the shaded grey box.

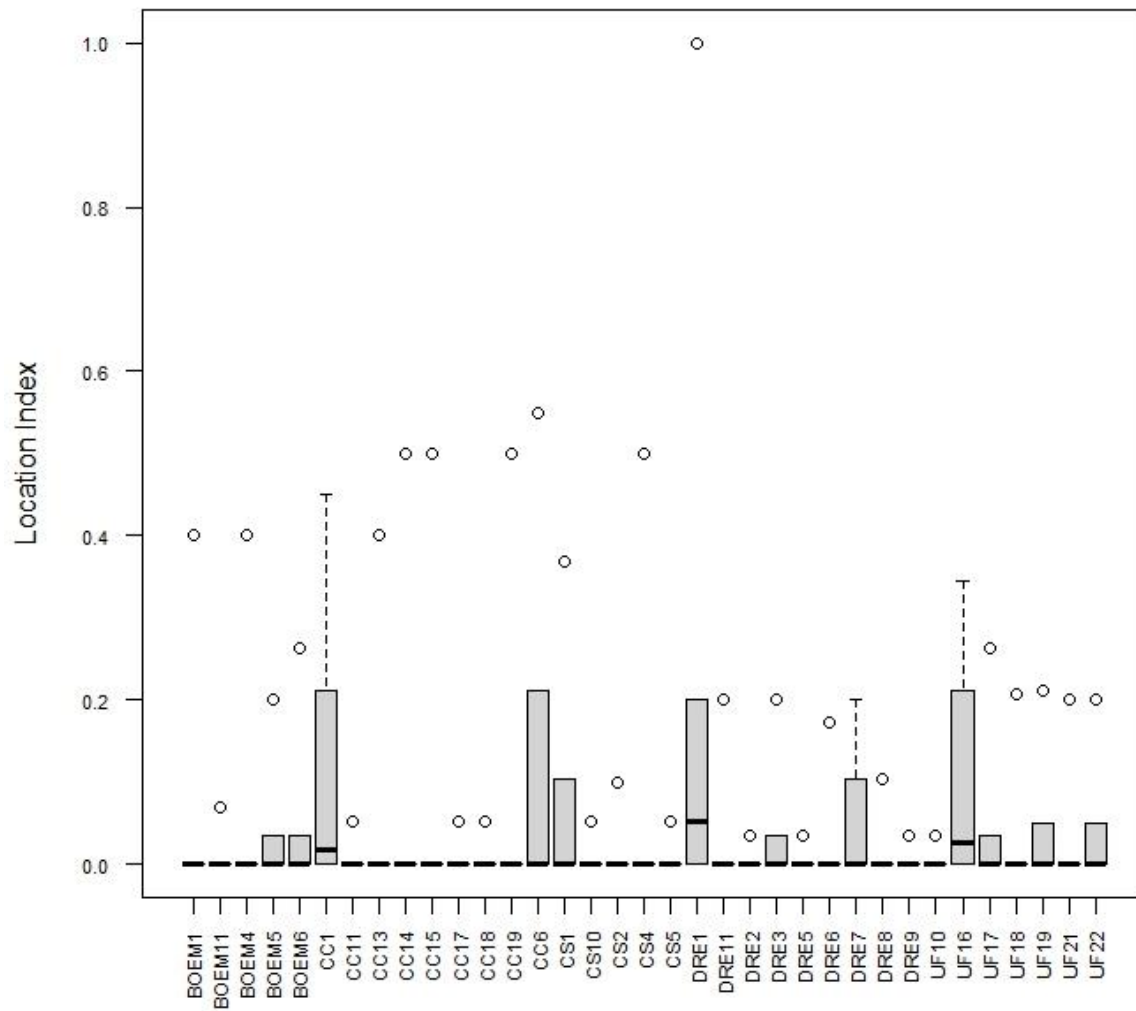


Figure 14-13. Location index for Bluntnose Rays tagged during the study period and detected within the receiver array off the east coast of Florida.

Boxplots show a solid horizontal line for the median location index, grey box represents the 25th and 75th percentiles, whiskers represent 95th percentile, and open symbols represent outliers.

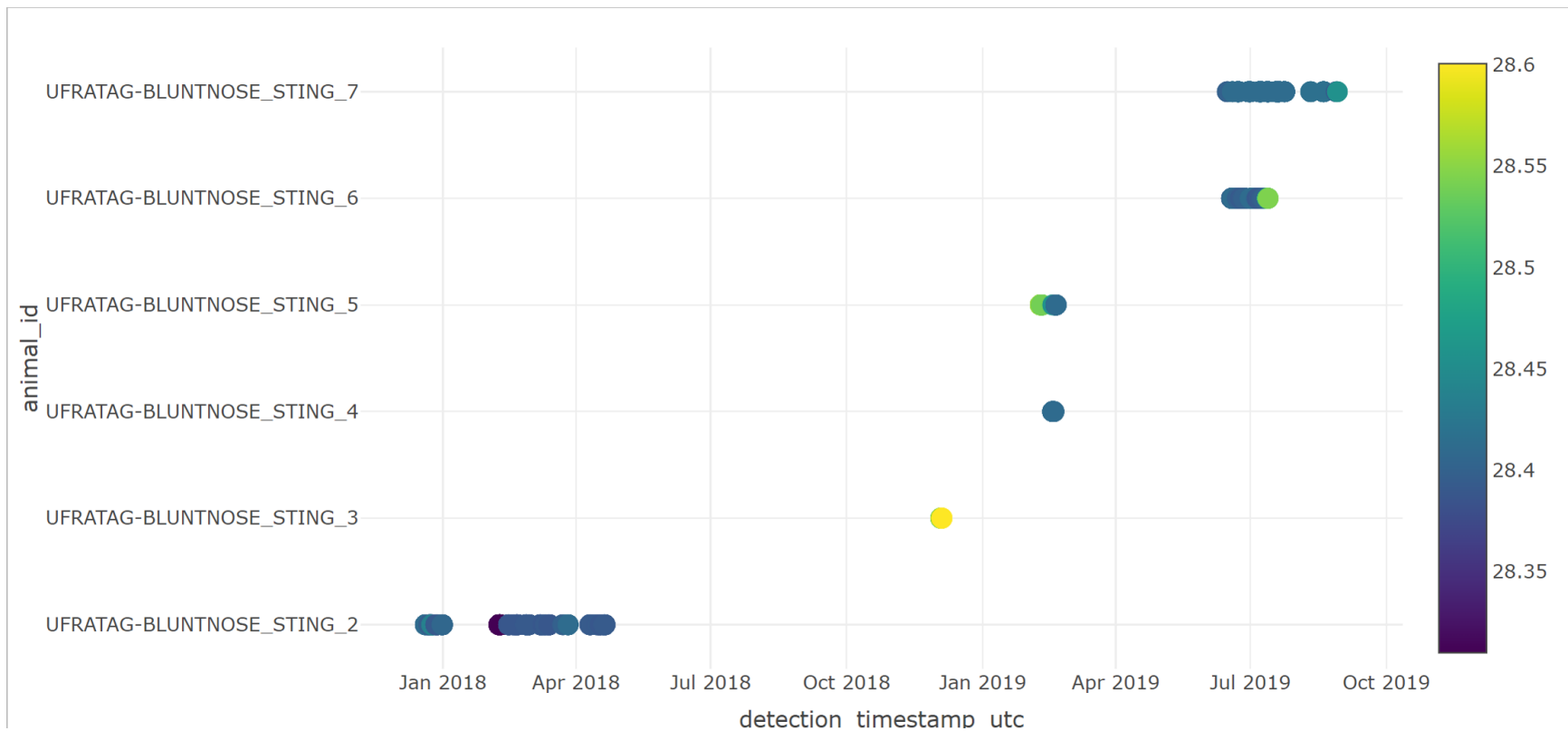


Figure 14-14. Abacus plot showing detections of tagged Bluntnose Stingrays as a function of time and latitude off the east coast of Florida. Latitude ranges from yellow in the north to dark blue in the southern portion of the receiver array.

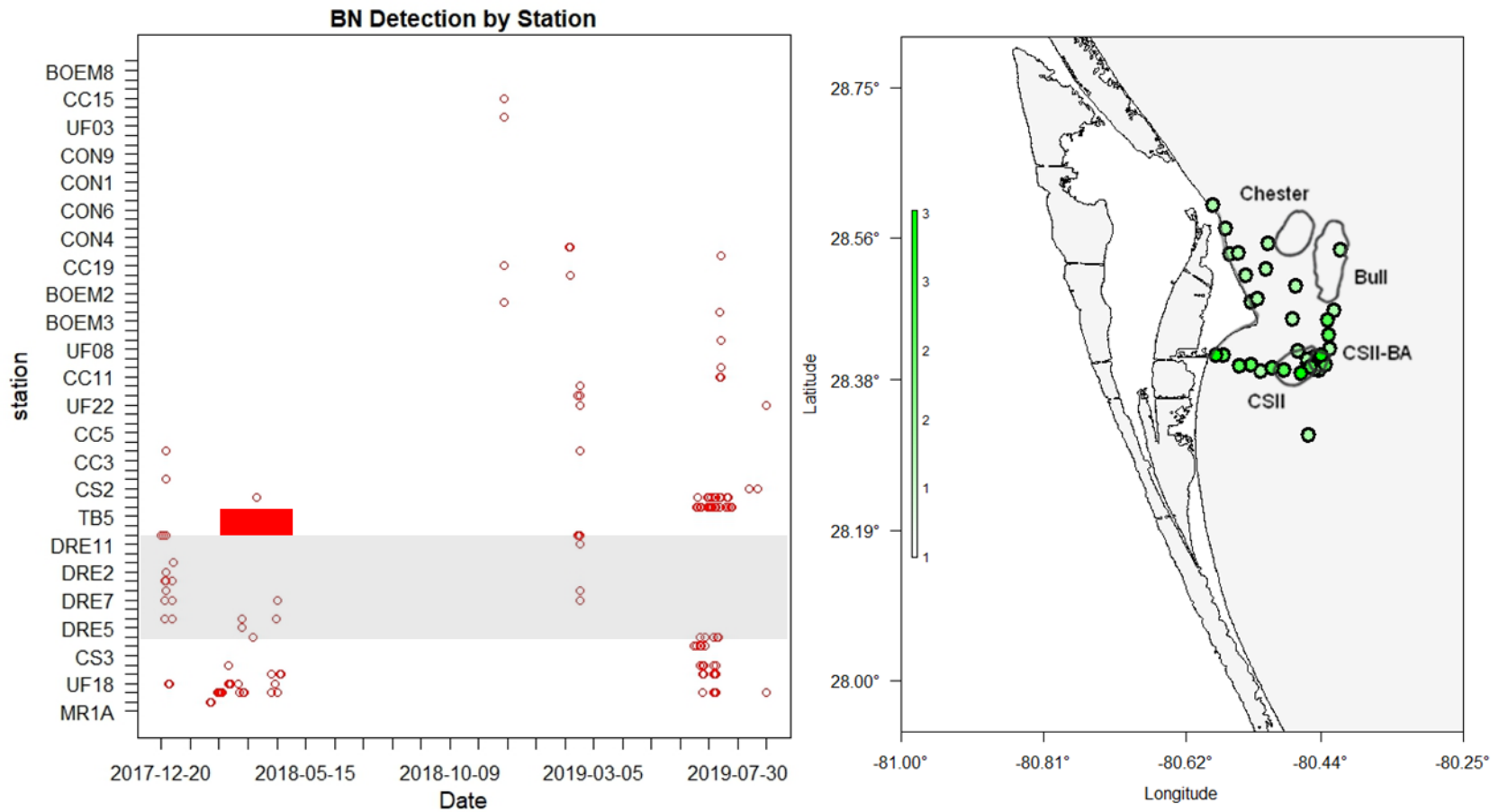


Figure 14-15. Abacus plot showing receiver stations where tagged Bluntnose Stingrays were detected over time and location of the detections off the east coast of Florida.

Receiver stations are, approximately, ordered from north (top) to south (bottom). Stations within CSII-BA are indicated by the shaded grey box. Red box indicates period of active dredging within CSII-BA.

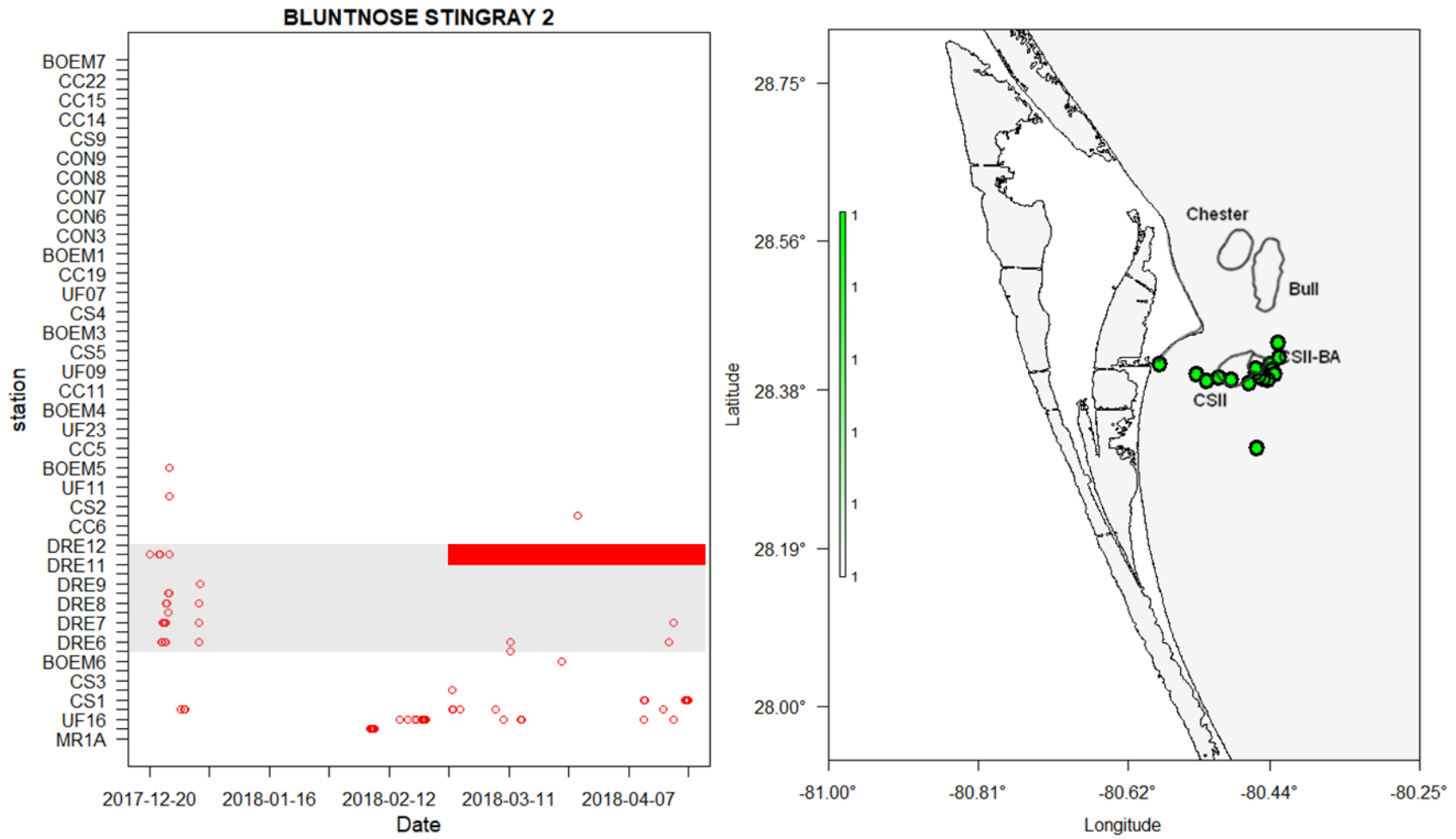


Figure 14-16. Abacus plot showing receiver stations where Bluntnose Stingray 2 was detected within CSII-BA (dredged shoal) over a period of time that included both prior to and during active dredging.

Receiver stations within CSII-BA are indicated by the shaded grey box. Red box indicates period of active dredging within CSII-BA.

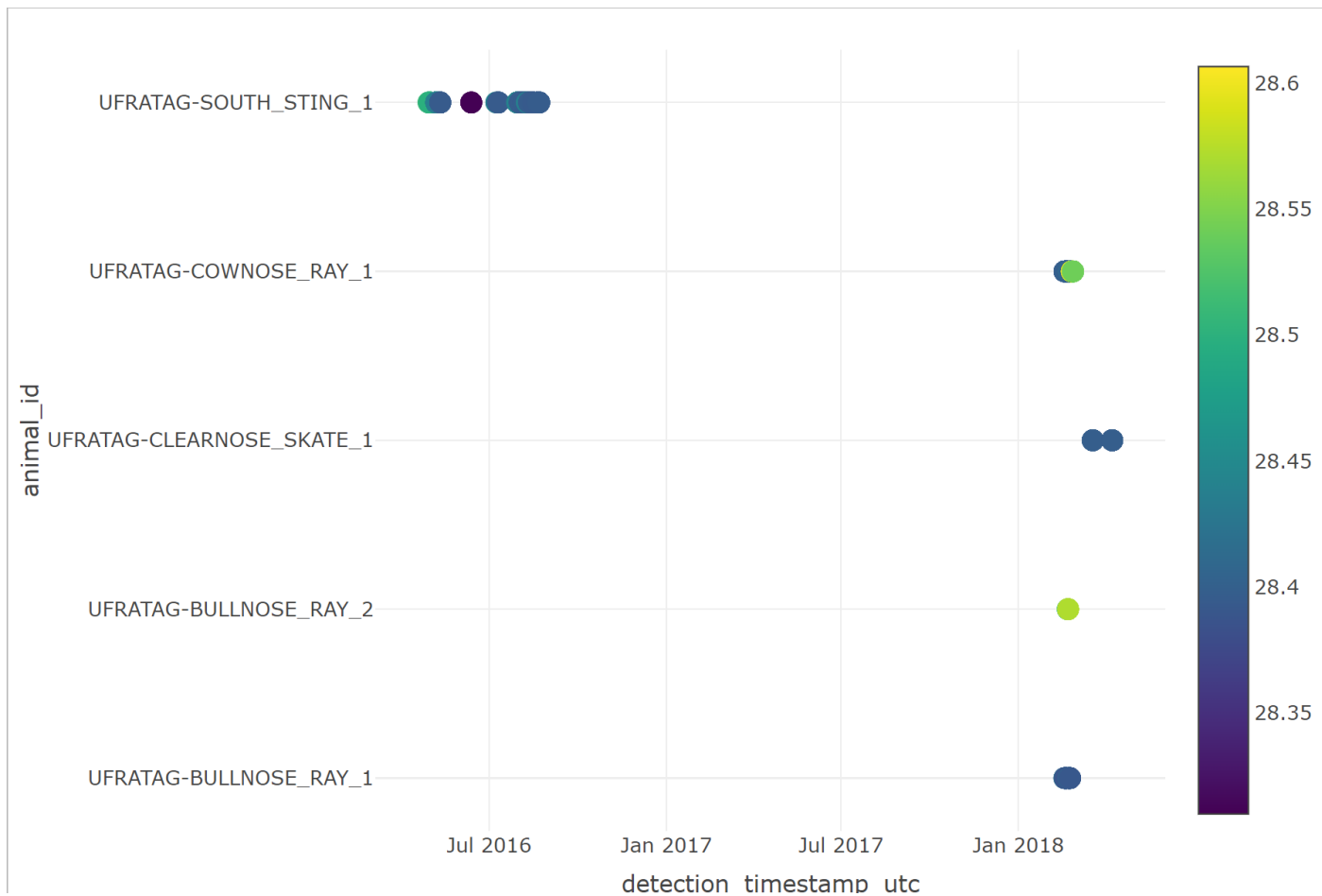


Figure 14-17. Abacus plot showing detections of tagged Bullnose Rays, Clearnose Skate, Cownose Ray, and Southern Stingray as a function of time and latitude off the east coast of Florida.

Latitude ranges from yellow in the north to dark blue in the southern portion of the receiver array.

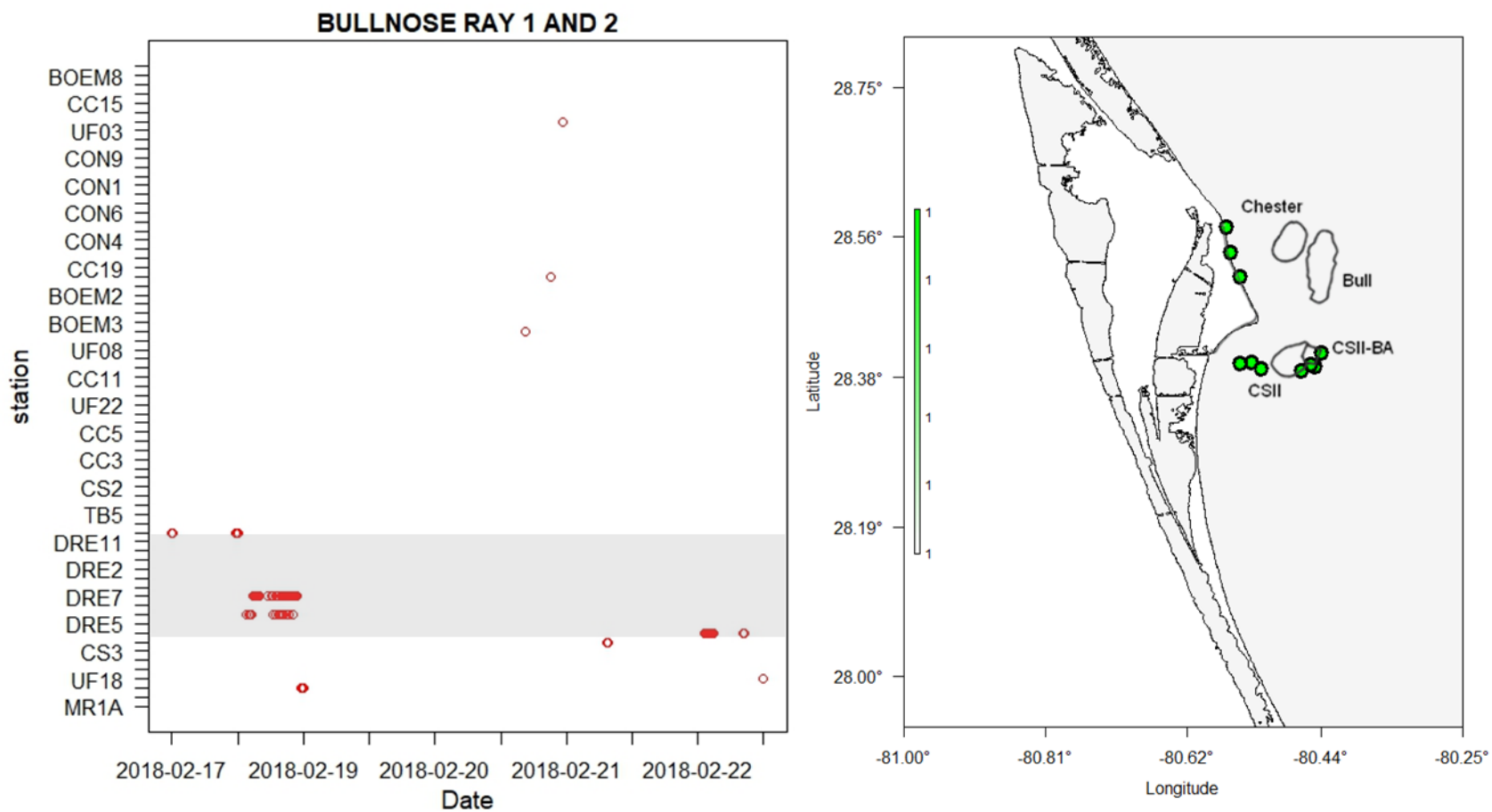


Figure 14-18. Abacus plot showing receiver stations where tagged Bullnose Rays were detected over time and location of the detections off the east coast of Florida.

Receiver stations are, approximately, ordered from north (top) to south (bottom). Stations within CSII-BA are indicated by the shaded grey box.

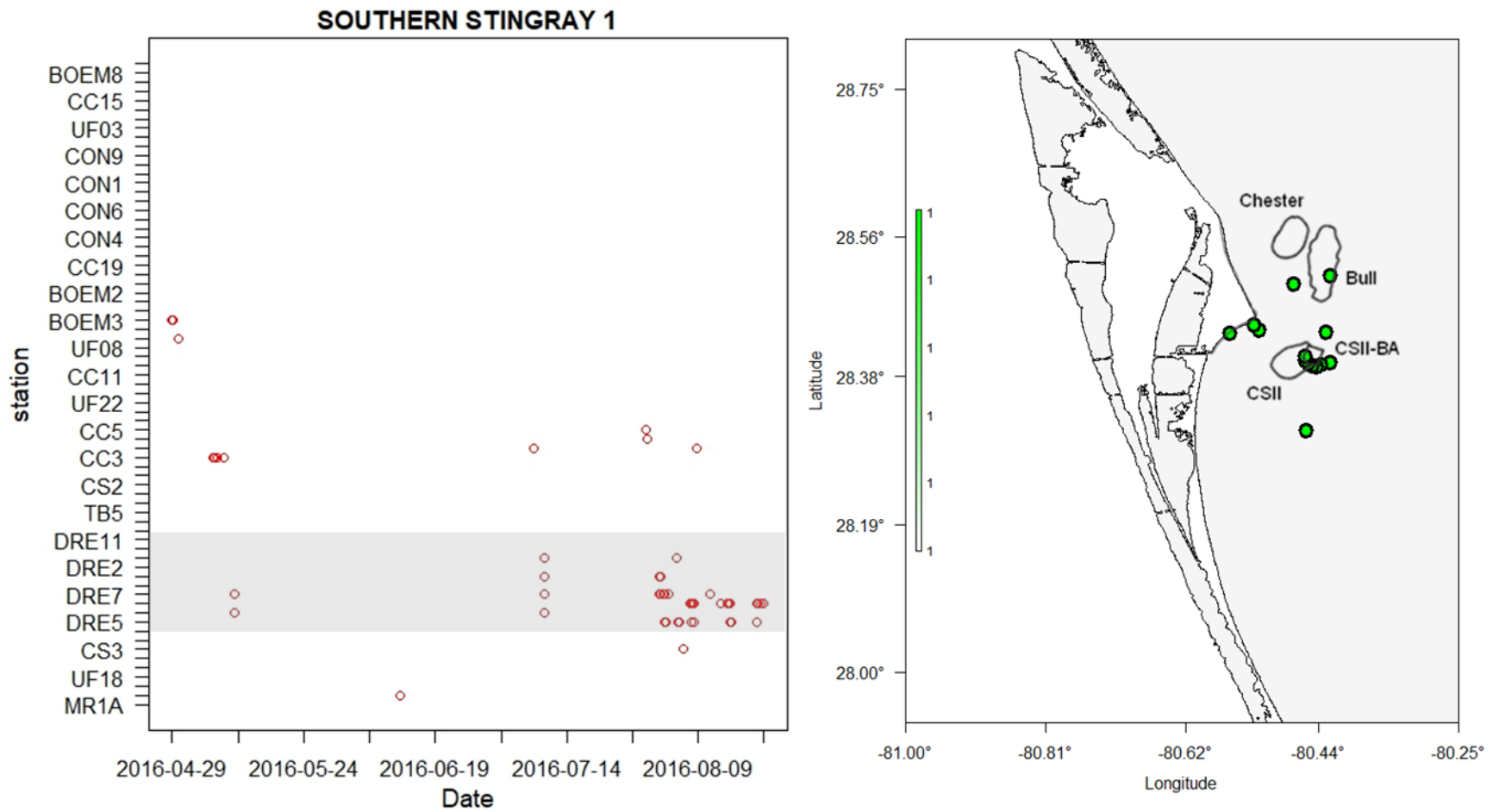


Figure 14-19. Abacus plot showing receiver stations where Southern Stingray 1 was detected within CSII-BA (dredged shoal) over a period of time and locations off the east coast of Florida.

Receiver stations within CSII-BA are indicated by the shaded grey box.

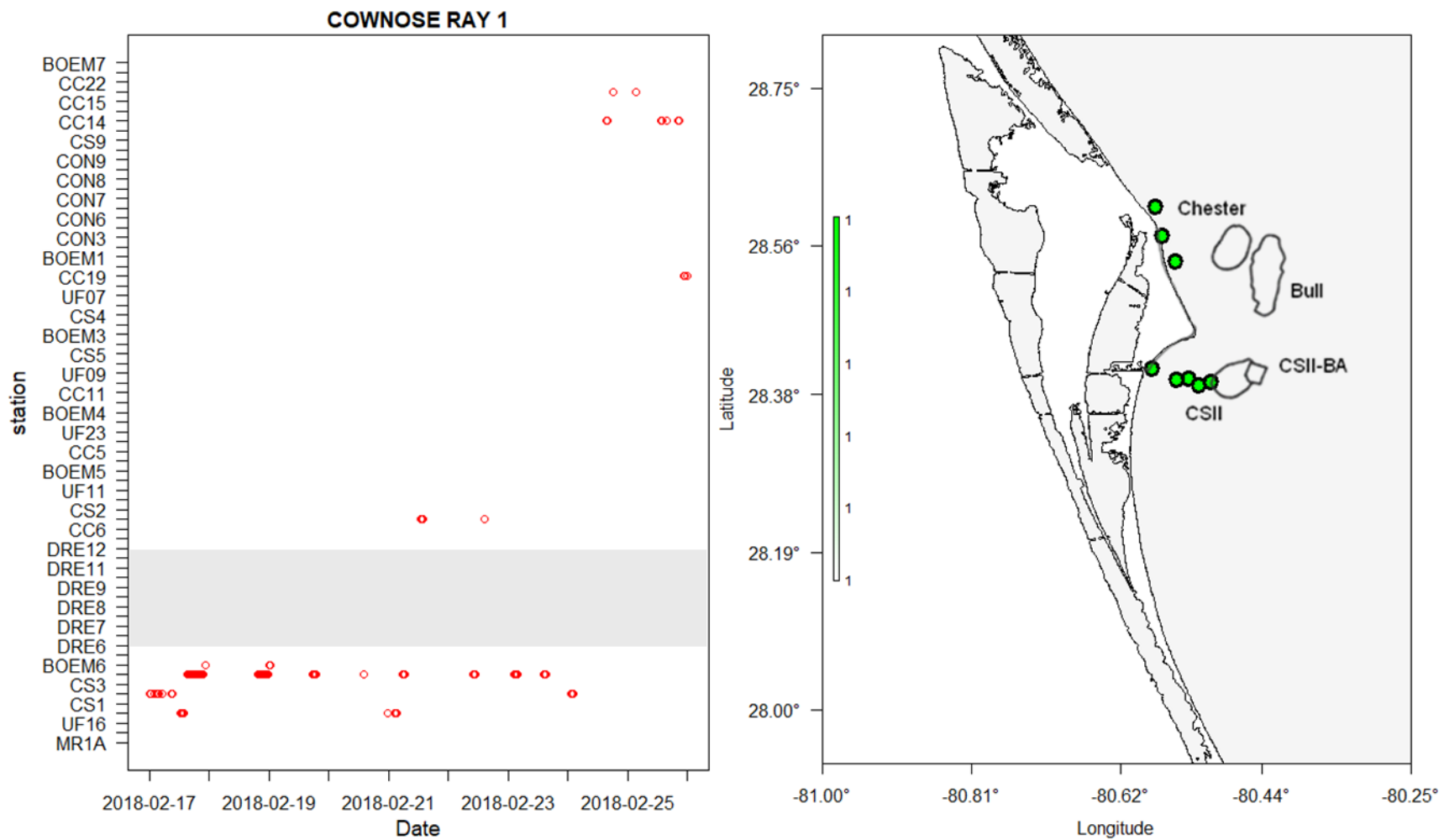


Figure 14-20. Abacus plot showing receiver stations where Cownose Ray 1 was detected over a period of time and locations off the east coast of Florida.

Receiver stations within CSII-BA are indicated by the shaded grey box.

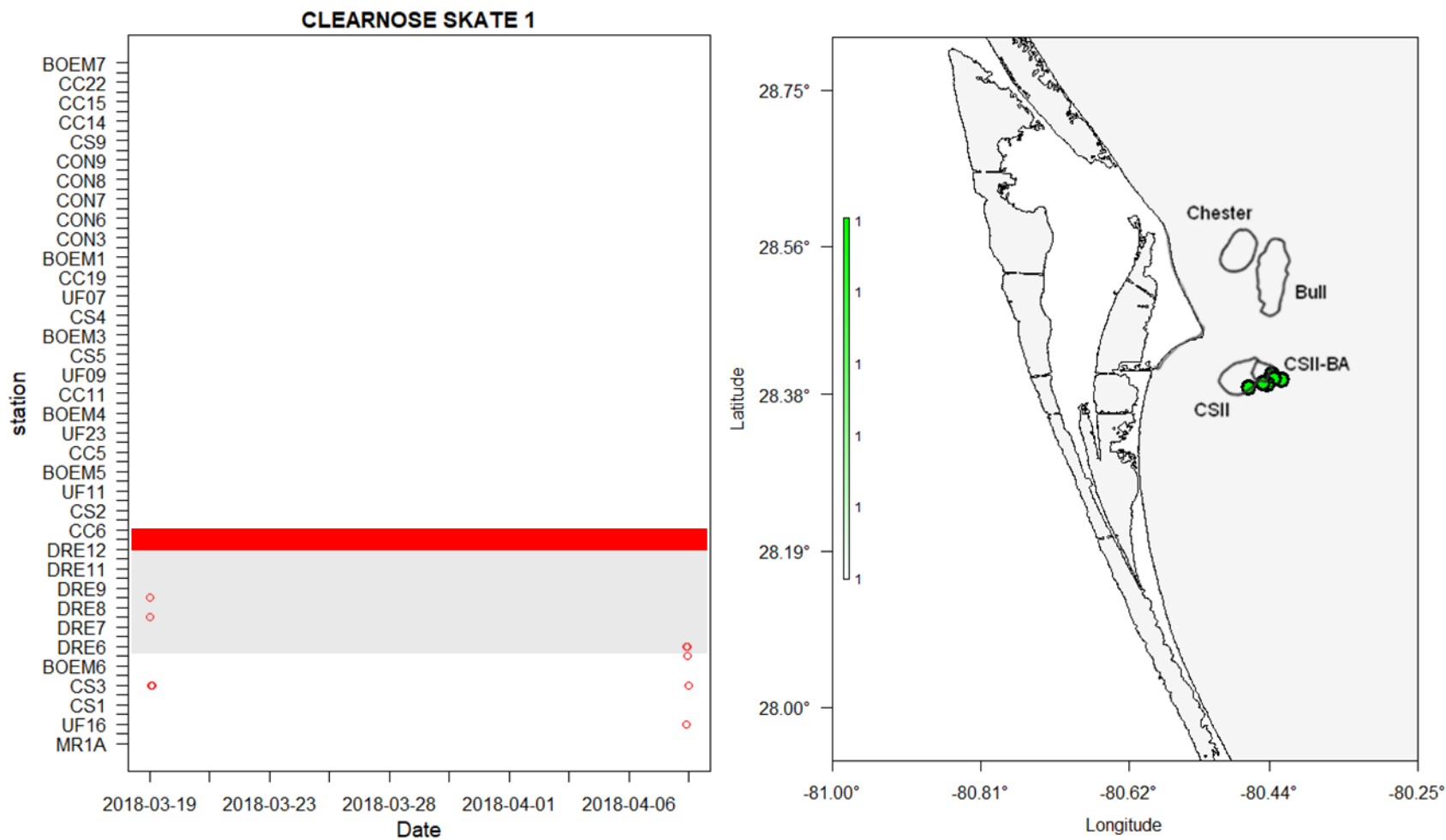


Figure 14-21. Abacus plot showing receiver stations where Clearnose Skate 1 was detected over a period of time and locations off the east coast of Florida.

Receiver stations within CSII-BA are indicated by the shaded grey box. Red box indicates period of active dredging within CSII-BA.

Table 14-1. Weight versus length regressions used to estimate weight of tagged fish to meet tag weight thresholds.

Common Name	Size Measurement	V9-Tag (Small) Size Threshold (mm)	V13-Tag (Large) Size Threshold (mm)	Weight-Length Regression	Source
Bluntnose Stingray	DW	> 155	> 210	$W = 0.000018924DW^{3.2275}$	Wigley et al. 2003
Bullnose Ray	DW	> 200	> 265	$W = 0.0000051290DW^{3.2885}$	Wigley et al. 2003
Clearnose Skate	TL	> 290	> 380	$W = 0.0000009486TL^{3.4235}$	Wigley et al. 2003
Cownose Ray	DW	> 190	> 255	$W = 0.0000070330DW^{3.2327}$	Wigley et al. 2003
Gulf Flounder	TL	> 250	> 300	$W = 0.000005715TL^{3.134}$	Safrit and Schwartz 1988
Smooth Butterfly Ray	DW	> 220	> 300	$W = 0.000007513DW^{3.0742}$	Wigley et al. 2003
Southern Flounder	TL	> 250	> 320	$W = 0.00000347TL^{3.21}$	Fischer and Thompson 2004
Southern Stingray	DW	> 155	> 210	$W = 0.00001773DW^{3.1428}$	Wigley et al. 2003
Summer Flounder	TL	> 250	> 300	$W = 0.0000046247TL^{3.2156}$	Wigley et al. 2003

Notes: DW = disk width; TL = total length

Table 14-2. Summary of all fish tagged and released in the study shoals off Cape Canaveral, Florida, with detection metrics.

Days at liberty reflect a maximum of the tag life (V9 or V13 tags) in days, after which the fish may be present but not detected because of a dead tag. No fish were known to be removed from the waters via fishing capture, etc., prior to the tag life expiring.

Fish Group	Common Name	Animal Identification Number	Number of Stations Detected	Number of Detections	Days Detected	Days at Liberty	Residency Index	Date of First Detection	Date of Last Detection
Flatfishes	Gulf Flounder	GULF_FL_1	0	0	0	651	0.0000		
Flatfishes	Gulf Flounder	GULF_FL_2	2	36	1	651	0.0015	2017-11-08	2017-11-08
Flatfishes	Gulf Flounder	GULF_FL_3	2	176	1	651	0.0015	2017-12-19	2017-12-19
Flatfishes	Gulf Flounder	GULF_FL_4	0	0	0	651	0.0000		
Flatfishes	Southern Flounder	SOUTH_FL_1	0	0	0	651	0.0000	2016-04-25	2016-04-25
Flatfishes	Southern Flounder	SOUTH_FL_2	1	7	1	651	0.0015	2017-11-09	2017-11-09
Flatfishes	Southern Flounder	SOUTH_FL_3	1	218	3	1,316	0.0023	2018-02-26	2018-03-31
Flatfishes	Southern Flounder	SOUTH_FL_4	2	6	2	651	0.0031	2018-03-17	2018-03-18
Flatfishes	Southern Flounder	SOUTH_FL_6	2	94	2	651	0.0031	2019-03-02	2019-03-04
Flatfishes	Southern Flounder	SOUTH_FL_7	1	9	3	651	0.0046	2019-03-22	2020-11-08
Flatfishes	Southern Flounder	SOUTH_FL_8	2	109	3	651	0.0046	2019-03-02	2019-03-16
Flatfishes	Southern Flounder	SOUTH_FL_9	3	60	3	651	0.0046	2019-06-13	2019-06-15
Flatfishes	Summer Flounder	SUMMER_FL_1	2	884	16	651	0.0246	2018-04-03	2018-09-01
Flatfishes	Summer Flounder	SUMMER_FL_2	0	0	0	651	0.0000		
Flatfishes	Summer Flounder	SUMMER_FL_3	0	0	0	651	0.0000		
Rays/Skates	Bluntnose Stingray	BLUNTNOSE_STING_1	0	0	0	651	0.0000		
Rays/Skates	Bluntnose Stingray	BLUNTNOSE_STING_2	17	2,302	31	1,316	0.0236	2017-12-20	2018-04-24
Rays/Skates	Bluntnose Stingray	BLUNTNOSE_STING_3	4	31	2	651	0.0031	2018-12-03	2018-12-04
Rays/Skates	Bluntnose Stingray	BLUNTNOSE_STING_4	1	43	2	651	0.0031	2019-02-17	2019-02-18
Rays/Skates	Bluntnose Stingray	BLUNTNOSE_STING_5	10	172	5	651	0.0077	2019-02-08	2019-02-19
Rays/Skates	Bluntnose Stingray	BLUNTNOSE_STING_6	12	1,684	20	1,316	0.0152	2019-06-18	2019-07-12
Rays/Skates	Bluntnose Stingray	BLUNTNOSE_STING_7	6	1,519	25	1,316	0.0190	2019-06-15	2021-02-21

Fish Group	Common Name	Animal Identification Number	Number of Stations Detected	Number of Detections	Days Detected	Days at Liberty	Residency Index	Date of First Detection	Date of Last Detection
Rays/Skates	Bullnose Ray	BULLNOSE_RAY_1	7	256	5	651	0.0077	2018-02-17	2018-02-23
Rays/Skates	Bullnose Ray	BULLNOSE_RAY_2	3	22	1	651	0.0015	2018-02-21	2018-02-21
Rays/Skates	Clearence Skate	CLEARNOSE_SKATE_1	6	70	2	651	0.0031	2018-03-19	2019-02-01
Rays/Skates	Cownose Ray	COWNOSE_RAY_1	8	775	11	1,316	0.0084	2018-02-17	2020-12-25
Rays/Skates	Smooth Butterfly Ray	SBR_1	0	0	0	651	0.0000		
Rays/Skates	Smooth Butterfly Ray	SBR_2	2	14	3	651	0.0046	2015-09-04	2015-09-16
Rays/Skates	Smooth Butterfly Ray	SBR_3	8	1,300	11	651	0.0169	2015-08-04	2015-09-28
Rays/Skates	Smooth Butterfly Ray	SBR_4	1	7	3	651	0.0046	2015-11-16	2015-11-17
Rays/Skates	Smooth Butterfly Ray	SBR_5	1	3	1	651	0.0015	2015-11-13	2015-11-13
Rays/Skates	Smooth Butterfly Ray	SBR_6	7	37	7	651	0.0108	2015-11-14	2016-01-04
Rays/Skates	Smooth Butterfly Ray	SBR_7	1	2	3	1,316	0.0023	2015-11-16	2015-12-04
Rays/Skates	Smooth Butterfly Ray	SBR_8	3	13	3	651	0.0046	2015-11-14	2016-02-09
Rays/Skates	Smooth Butterfly Ray	SBR_9	3	Dropped tag	-	-	-	2016-04-27	2016-05-02
Rays/Skates	Smooth Butterfly Ray	SBR_10	8	369	7	1,316	0.0053	2016-05-12	2016-06-23
Rays/Skates	Smooth Butterfly Ray	SBR_11	1	3	2	651	0.0031	2016-06-04	2016-06-05
Rays/Skates	Smooth Butterfly Ray	SBR_13	0	0	0	651	0.0000	2016-12-08	2016-12-08
Rays/Skates	Smooth Butterfly Ray	SBR_15	6	171	7	651	0.0108	2017-08-18	2017-09-08
Rays/Skates	Smooth Butterfly Ray	SBR_16	10	1,829	8	651	0.0123	2017-08-30	2017-09-14
Rays/Skates	Smooth Butterfly Ray	SBR_17	1	14	1	1,316	0.0008	2017-11-07	2017-11-07
Rays/Skates	Smooth Butterfly Ray	SBR_18	2	259	6	651	0.0092	2019-02-06	2020-10-15
Rays/Skates	Smooth Butterfly Ray	SBR_19	2	29	8	651	0.0123	2019-06-16	2019-07-13
Rays/Skates	Smooth Butterfly Ray	SBR_20	1	246	3	1,316	0.0023	2019-06-14	2019-06-16
Rays/Skates	Smooth Butterfly Ray	SBR_21	0	0	0	651	0.0000		
Rays/Skates	Smooth Butterfly Ray	SBR_22	2	582	7	651	0.0108	2019-06-14	2019-06-20
Rays/Skates	Smooth Butterfly Ray	SBR_23	12	61	8	739	0.0108	2019-06-14	2021-06-21
Rays/Skates	Smooth Butterfly Ray	SBR_24	2	20	3	651	0.0046	2019-06-17	2021-01-12

Fish Group	Common Name	Animal Identification Number	Number of Stations Detected	Number of Detections	Days Detected	Days at Liberty	Residency Index	Date of First Detection	Date of Last Detection
Rays/Skates	Smooth Butterfly Ray	SBR_25	5	467	11	651	0.0169	2019-06-14	2019-06-27
Rays/Skates	Smooth Butterfly Ray	SBR_26	3	148	7	1,316	0.0053	2019-06-18	2020-12-11
Rays/Skates	Smooth Butterfly Ray	SBR_27	0	0	0	651	0.0000		
Rays/Skates	Smooth Butterfly Ray	SBR_28	0	0	0	651	0.0000		
Rays/Skates	Smooth Butterfly Ray	SBR_29	2	461	7	651	0.0108	2019-06-15	2019-07-02
Rays/Skates	Smooth Butterfly Ray	SBR_30	6	209	3	1,316	0.0023	2019-06-15	2019-08-30
Rays/Skates	Smooth Butterfly Ray	SBR_31	0	0	0	1,316	0.0000		
Rays/Skates	Smooth Butterfly Ray	SBR_32	3	2,485	12	1,316	0.0091	2019-06-14	2021-08-08
Rays/Skates	Southern Stingray	SOUTH_STING_1	14	846	26	651	0.0399	2016-04-29	2016-08-22

15 Stable Isotope Composition and Niches of Focal Fish and Invertebrate Species on Shoals in Relation to Dredging Events

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Key Points

- Stable isotopes of $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ were used to trace potential trophic changes in basal carbon resources and trophic level, respectively, for focal invertebrate and fish species from the dredged shoal (CSII-BA) compared to the non-dredged reference/control shoals (CSII, Bull, and Chester Shoals).
- Overall, seasonal isotopic composition and isotope niche sizes of amphipods, Roughneck Shrimp, Aviu Shrimp, Leopard Searobin, Spotted Whiff, and Atlantic Croaker varied among the reference shoals as much or more than between the reference shoals and the dredged shoal.
- For all focal species except Aviu Shrimp, there were no differences in their trophic position based on their $\delta^{13}\text{C}$ or $\delta^{15}\text{N}$ isotope values in fall 2013 (i.e., immediately prior to the first dredging event in winter 2013/14 and spring 2014) compared to post-dredge seasons (fall 2014 and fall 2015) for CSII-BA, CSII, or Bull Shoals.
- Trophic positions of Aviu Shrimp changed between pre- and post-dredge fall periods associated with the first dredging event, but they did so simultaneously on all three shoals and therefore could not be ascribed to the dredging event itself.
- Isotope values of $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ for all focal species for all shoals were not different between the pre- and post-dredge seasons for the second dredging event in winter 2018, with few exceptions that were related to the natural variability among the non-dredged shoals.
- Overall, the stability in the trophic positions of the focal invertebrate and fish species during pre- versus post-dredge seasons, and within the dredged shoal relative to the reference shoals, indicated that dredging did not fundamentally change the trophic state and linkages of the food web of the shoals, as observed through focal species that spanned different trophic levels and feeding lifestyles.

15.1 Introduction

Changes to marine environments due to perturbations, such as dredging, can take many forms, including changes to species diversity, abundance, and trophic dynamics, to name only a few. Previous chapters have dealt specifically with the first two indices of change and the present chapter examines potential changes in trophic dynamics due to dredging. In environments with relatively low species diversity (i.e., cold temperate regions), a common approach to determining changes in trophic dynamics may include detailed stomach content analyses. Although this method is sampling intensive even in a low-diversity ecosystem, this method provides a direct look at predator-prey interactions in the ecosystem. However, stomach content analysis only provides a snapshot of what the animal consumed relatively recently, usually within the past 12–24 hr (Parrish and Margraf 1990; Berens and Murie 2008). In addition, variable rates in the digestion of prey can also introduce bias (DeNiro and Epstein 1978) and require sampling over extensive periods of the day and night (Berens and Murie 2008). Due to the necessary

sampling intensity of stomach content analysis studies, stable isotopes have become an alternative way of deciphering changes in trophic dynamics. Stable isotopes differ from diet analysis in that isotope values represent the assimilation of prey resources over a longer period of time, varying from months for muscle tissue to just days for blood samples (Hesslein et al. 1993; Perga and Gerdeaux 2005; Fry 2006; Guzzo et al. 2013).

Although stable isotopes do not provide direct information on a species-specific prey basis, they provide a means to differentiate changes in the source of carbon being assimilated by the organisms through the analysis of the stable isotopes of carbon, ^{13}C and ^{12}C , and the trophic level of the organisms through the analysis of the stable isotopes of nitrogen, ^{15}N and ^{14}N (Fry 2006). Carbon in tissues originates primarily through atmospheric CO_2 captured during photosynthesis and is propagated through the food chain. Plants discriminate against the heavier isotope of carbon (^{13}C) during photosynthesis and hence tissues of organisms are depleted in the heavier carbon isotope to varying degrees depending on the plant's specific type of photosynthetic pathway in which they fix carbon (C_3 versus C_4 pathways) (O'Leary 1988; Fry 2006). For example, the mangrove *Rhizophora* is a C_3 plant and has $\delta^{13}\text{C}$ values of -33 to -24 ‰ whereas the seagrass *Thalassia* is a C_4 plant and has less depleted values of $\delta^{13}\text{C}$ ranging from -16 to -7 ‰, with isotope compositions expressed in terms of delta values (δ) that are parts per thousand (‰) differences from a known standard (Anderson and Fourqurean 2003; Mancera-Pineda et al. 2009; Vaslet et al. 2012). Transfer of ^{13}C through trophic levels, on average, enriches tissues by only 0.4 ‰ $\delta^{13}\text{C}$ (0–1.5 ‰) (Post 2002), so carbon isotope values of predators reflect the carbon isotope values of their prey, which translates into “you are what you eat” (DeNiro and Epstein 1978). Therefore, large differences in $\delta^{13}\text{C}$ between organisms indicate that they have a different source of primary producers (e.g., mangroves, seagrasses, phytoplankton, etc.) or that they are consuming different prey sources that are, in turn, consuming different primary producers. Nitrogen in the tissues of an organism originates from its prey and therefore stable isotopes of nitrogen can be analyzed to differentiate changes in the trophic level of the organisms. The lighter isotope of nitrogen is used preferentially in metabolic reactions with the heavier isotope therefore accumulating in the predator's tissues, a process called enrichment (Zanden and Rasmussen 2001). On average, tissue is enriched by 3.4 ‰ $\delta^{15}\text{N}$ (1 SD = 1.0 ‰) per trophic level (Post 2002).

Carbon sources at the base of the food web could potentially shift with disturbance, for example due to dredging, because of the removal of the top portion of the sediments and the associated primary producers (e.g., microphytobenthos or algae), or by disturbing the sediments and resuspending nutrients into the water column and creating a larger base of planktonic primary producers. Comparing carbon isotopes between dredged and non-dredged shoals allows any shifts in basal carbon resources used by animals in the major trophic levels in the food web to be traced. Dredging may also potentially impact the trophic level structure of a food web. This can occur, for example, if the dredging activity removes a primary producer (e.g., microphytobenthos) consumed by an omnivore but not the animal prey it consumes, therefore shifting it from an omnivore to a strict carnivore only, which would shift it from a lower to a higher trophic level that would be reflected in the nitrogen isotopes. Overall changes in trophic structure of ecosystems can therefore be tracked over time in relation to a potential disturbance, such as dredging, especially if a BACI (Donázar-Aramendía et al. 2018) or a Beyond-BACI design is employed (this study).

15.1.1 Goals and Objectives

The overall goal was to describe the spatiotemporal trophic positions of fish and invertebrate species among the sand shoals off the east coast of Florida, and to determine if their trophic positions were impacted by dredging activities. Due to the diversity of organisms encountered on the shoals, a select group of species that occurred consistently in bottom trawls throughout the year on all of the shoals (see Chapter 13) were chosen as focal species that represented different trophic levels and feeding strategies.

These species included Atlantic Croaker *Micropogonias undulatus* (demersal mesocarnivore), Leopard Seabrook *Prionotus scitulus* (benthic mesocarnivore), Spotted Whiff *Citharichthys macrops* (benthic mesocarnivore), Roughneck Shrimp *Trachypanaeus constrictus* (benthic omnivore), Aviu Shrimp *Acetes americanus* (pelagic omnivore), and haustoriid/gammarid amphipods (*Acanthohaustorius* sp., *A. millsi*, *A. shoemaker*, *Protohaustorius wigleyi*, or *Ampelisca* sp.) (benthic/epibenthic omnivore). These fishes and invertebrates were used as model species to detect changes in both trophic level and basal carbon sources in the food web of dredged and non-dredged shoals.

The specific objectives were to 1) estimate isotopic niches of model species of fish and invertebrates on a spatial (shoal) and temporal (seasonal) basis; and 2) determine whether isotopic niches of the model species changed relative to pre- and post-dredging periods of time.

15.2 Methods

15.2.1 Sampling

Isotope samples for invertebrates and fishes were obtained as subsamples from those organisms identified, counted, and weighed as described in Chapters 11, 12 and 13. In brief, samples were collected from Canaveral Shoal II (CSII), Canaveral Shoal II Borrow Area (CSII-BA), Chester Shoal, and Bull Shoal (Figure 1-1). CS II-BA was the northeastern quadrant of CSII that was dredged, and CSII, Chester and Bull Shoals represented the reference shoals. Each shoal was sampled on an annual, seasonal, and diurnal basis. Annual sampling occurred for 6 years from November 2013 to July 2019. Seasonal sampling occurred with spring from March–May, summer was June–September, fall was October–November, and winter was December to the following February. Although sampling also occurred on a diurnal basis, isotope muscle samples were not differentiated between day and night sampling because they represent assimilation over a period of months rather than hours or days (Fry 2006). Further details of the sampling design are given in Chapter 1.

Fish and demersal invertebrate samples were collected using standardized tows of a 6.1 m (20 ft) semi-balloon bottom (otter) trawl deployed for 10 min (bottom time). Details of the trawl collections are given in Chapters 12 and 13. Amphipods were collected using standardized benthic sampling using a modified-Young grab. Details of the benthic grab collections are given in Chapter 11.

15.2.1.1 Sampling Relative to First Dredging Event (Dredge-1)

CSII, CSII-BA, and Bull Shoals were sampled in fall 2013, immediately prior to the first dredging event on CSII-BA, which took place between 27 November 2013 to 22 April 2014 (Table 1-1). Pre-dredge sampling was limited due to the initiation of the study in October 2013 with eight stations sampled on CSII-BA, five stations on CSII, and six stations sampled on Bull Shoal; Chester Shoal was not sampled in the pre-dredge period prior to the first dredging event. The first dredging event (referred to as Dredge-1) was therefore comprised of one pre-dredge season of sampling on CSII-BA, CSII, and Bull Shoals (fall 2013), followed by the main dredging in winter 2013/14 through spring 2014. Due to the significant seasonal effect observed on the assemblages of fishes and invertebrates on the shoals (see Chapters 11–13), comparisons relative to dredge effects were therefore limited to the fall, with one season prior to dredging (fall 2013) and two post-dredge fall seasons (fall 2014 and fall 2015).

15.2.1.2 Sampling Relative to Second Dredging Event (Dredge-2)

CSII-BA was dredged a second time during this study during winter 2017/18 through spring 2018 (3 Feb to 11 April 2018), referred to as Dredge-2 (Table 1-1). Although not a component of the original study plan, this second dredging event allowed for a more extensive suite of pre-dredge samples to be compared to post-dredge samples for all shoals. Pre-dredge sampling included four seasons prior to dredging,

including winter 2016/17, spring 2017, summer 2017 and fall 2017, whereas dredging and post-dredge sampling included winter 2017/18, spring 2018, summer 2018, and fall 2018.

15.2.2 Sample Processing

Fish and invertebrates identified, counted, and weighed for abundance and biomass analysis in conjunction with Chapters 11, 12, and 13 were then frozen until processed for isotopes. For each combination of year, shoal, and season, three isotope samples were taken for each focal species when available. For all fishes and Roughneck and Aviu Shrimps, a white muscle sample was taken (~ 5 mm cube) from the dorsal musculature of fishes or tail musculature of shrimps. For amphipods, whole animals were used because individuals were too small to dissect for muscle tissue. Multiple amphipods were pooled to provide enough weight for analysis by shoal within each season and year. All samples were stored in 1.5-ml microcentrifuge tubes in a -80 °C ultra-low freezer until processed for isotopes.

For isotope analysis, muscle samples were removed from the -80 °C freezer and inspected under a microscope and any extraneous scales, exoskeleton pieces, or sand particles removed. The sample was then superficially rinsed (surface only) using double-distilled water and placed in a pre-combusted, labeled glass vial. Glass vials were pre-combusted for 6 hrs at 500 °C to remove any residual carbon prior to use. For whole amphipods, all sand grains and detritus were removed while using a stereomicroscope and the whole body was then rinsed with double-distilled water and placed in a labeled glass vial. All samples were dissected/inspected while on a watch glass placed on an ice pack to keep the sample cold. All dissecting equipment was washed with ethanol and then rinsed with double-distilled water and air-dried between each sample. Samples in glass vials were then dried in a 60 °C oven for 48 hr (Jaschinski et al. 2008). Each sample was then removed, placed between a piece of pre-combusted, folded aluminum foil, and pulverized into a homogeneous powder using a pestle. Pulverized samples were then stored in capped pre-combusted glass vials in a -80 °C ultracold freezer until final analysis.

Prior to transport to the isotope lab for analysis, samples in capped glass vials were uncapped and redried in a 60 °C oven for 6 hr to remove any moisture incorporated during pulverization and storage, cooled in a desiccator, and recapped. Samples were then stored in a desiccator while transported to the Light Stable Isotope Mass Spec Lab in the Department of Geological Sciences at UF for analysis.

Muscle samples were analyzed for $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ using a Thermo Electron DeltaV Advantage isotope ratio mass spectrometer coupled with a ConFlo II interface linked to a Carlo Erba NA 1500 CNHS Elemental Analyzer. Samples (0.6–0.8 mg weight range, weighed exactly ± 0.001 mg) were loaded into tin capsules and placed in a 50-position automated Zero Blank sample carousel on a Carlo Erba NA1500 CNS elemental analyzer. After combustion in a quartz column at 1,020 °C in an oxygen-rich atmosphere, the sample gas was transported in a helium carrier stream and passed through a hot reduction column (650 °C) consisting of elemental copper to remove oxygen. The effluent stream then passed through a chemical (magnesium perchlorate) trap to remove water followed by a 0.7 m gas chromatography column at 120 °C to separate N_2 from CO_2 . The sample gas next passed into a ConFlo II interface and into the inlet of a Thermo Electron Delta V Advantage isotope ratio mass spectrometer running in continuous flow mode where the sample gas was measured relative to laboratory reference N_2 and CO_2 gases. Calibration was done using reference material USGS40 (L-glutamic acid) and USGS41 (L-glutamic acid enriched in ^{13}C and ^{15}N) (USGS 2019). Reference material (USGS40) was used at the beginning, end, and every fifteenth sample in each run. In addition, sample duplicates were run every fifth sample and an internal lab standard (Bovine) was run on every tenth sample; any inconsistent runs were resampled.

Stable isotope values were expressed in standard delta notation in ‰ as:

$$\delta^{13}\text{C or } \delta^{15}\text{N} = \left[\frac{R_{\text{sample}}}{R_{\text{standard}}} - 1 \right] \times 1,000 \quad (15-1)$$

where R is $^{13}\text{C}/^{12}\text{C}$ or $^{15}\text{N}/^{14}\text{N}$ for the sample or standard, and where the global standard for $\delta^{13}\text{C}$ is Vienna PeeDee Belemnite and for $\delta^{15}\text{N}$ is atmospheric nitrogen (N_2). Analytical precision estimated through replicate working standards were on average ± 0.10 ‰ and ± 0.09 ‰ for carbon and nitrogen, respectively. Percent C and percent N were also estimated to provide a C:N ratio, which was used as a proxy for lipid content (Post et al. 2007). Lipids can bias carbon isotope values because they are more depleted in ^{13}C relative to proteins and carbohydrates, and many organisms have varying lipid levels on a seasonal basis (Post et al. 2007). Most of the invertebrate samples and some fish samples had C:N ratios ≥ 3.5 , indicating variable lipid concentrations in the samples above the threshold given by Post et al. (2007), and all samples with C:N ≥ 3.5 were therefore mathematically normalized for lipids using the formula by Post et al. (2007):

$$\delta^{13}\text{C}_{\text{normalized}} = \delta^{13}\text{C}_{\text{untreated}} - 3.32 + (0.99 \times \text{C:N}) \quad (15-2)$$

where C:N is the carbon:nitrogen ratio of the original, untreated sample.

For amphipods, samples were analyzed for both $\delta^{15}\text{N}$ and $\delta^{13}\text{C}$ using the same protocol as for muscle samples because $\delta^{15}\text{N}$ and $\delta^{13}\text{C}$ were run simultaneously. In addition, however, a subsample of amphipods ($n = 107$) were acidified because the amphipods were processed whole and their exoskeletons contained inorganic carbon that needed to be removed in order to only have organic carbon remaining in the sample that was assimilated through their diet (Jaschinski et al. 2008). Pulverized, dried samples were acidified using 1 M HCl, added drop by drop to the sample, until there was no more effervescence, following the “champagne” method of acidification (Jaschinski et al. 2008). Acidified samples were not rinsed following the procedure, which has been shown to bias the carbon isotope values (Carabel et al. 2006). Acidified samples were only analyzed for $\delta^{13}\text{C}$ values. The subsample of acidified- $\delta^{13}\text{C}$ values were modeled as a function of their nonacidified- $\delta^{13}\text{C}$ values to obtain a regression formula that could be used to estimate the acidified- $\delta^{13}\text{C}$ values for the amphipod samples that were not directly acidified. Some shoals did not have enough haustorid amphipods on a seasonal basis and so gammarid amphipods were used as substitutes when necessary. Differences in the regression model between haustorid versus gammarid amphipods was tested using Analysis of Covariance (ANCOVA) (Zar 1996). In addition, a small subsample of haustorid amphipods from the same station sample were analyzed for $\delta^{13}\text{C}$ based on three different preparations (all pulverized): 1) whole, not acidified; 2) whole but acidified; and 3) muscle only (dissected from whole amphipods). This comparison was done to check on whether the acidification process was comparable to only sampling muscle tissue of the amphipods, similar to using muscle samples for all the other invertebrates and fishes. These samples were compared using paired t-tests (Zar 1996).

15.2.3 Shoal Isotope Analysis and Niche Metrics

Isotope data were initially visualized using bivariate plots (biplots) of $\delta^{15}\text{N}$ and $\delta^{13}\text{C}$ (Newsome et al. 2007) for each model species, with overall mean ± 1 standard error (SE) overlaid using SIBER in R (Jackson 2021). Isotopic niches of the focal species were then quantified to compare the size and extent of the isotopic niches among the species, shoals, and seasons, and then compared relative to the dredging events. Metrics used were based on the original community metrics described by Layman et al. (2007) but applied to subpopulations of focal species identified by shoal and season as per Quevedo et al. (2009) and Feiner et al. (2019). Isotopic niche quantification was modified based on the Stable Isotope Bayesian Ellipses package in R (SIBER) (Jackson et al. 2011) and applied to groups within the communities of the shoals (i.e., the focal species). The Layman metrics most relevant to this study included:

- 1) $\delta^{15}\text{N}$ range (NR): distance between the mean $\delta^{15}\text{N}$ of the group with the most enriched and most depleted mean $\delta^{15}\text{N}$ values (i.e., maximum mean $\delta^{15}\text{N}$ – minimum mean $\delta^{15}\text{N}$). A greater NR of

any group or groups would suggest that individuals in that group are, in general, feeding over a greater number of trophic levels.

- 2) $\delta^{13}\text{C}$ range (CR): distance between the mean $\delta^{13}\text{C}$ of the group with the most enriched and most depleted mean $\delta^{13}\text{C}$ values (i.e., maximum mean $\delta^{13}\text{C}$ – minimum mean $\delta^{13}\text{C}$). A greater CR of any group or groups would suggest that the individuals in that group are using prey resources reflecting more diverse carbon (basal) sources.
- 3) Mean distance to centroid (CD): average Euclidean distance of each individual in the group to the isotopic $\delta^{13}\text{C}$ - $\delta^{15}\text{N}$ centroid of the group. The isotopic $\delta^{13}\text{C}$ - $\delta^{15}\text{N}$ centroid of the group is the mean $\delta^{13}\text{C}$ and mean $\delta^{15}\text{N}$ for all the individuals in the group. This metric is a measure of the trophic diversity within each group.

Layman et al. (2007) also proposed the metric TA, which is the total area of the convex hull that encompasses all of the $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ data for a species or community. However, this metric is sensitive to sample size when applied to a single group community member where sample size is often < 50. In addition, it provides a biased metric when sample sizes differ among samples within the same study and when comparing across studies (Jackson et al. 2011). Alternatively, the standard ellipse area (SEA), which represents the 95% confidence ellipse of the mean, was developed to quantify the size of the isotopic niches for each focal species (Jackson et al. 2011; Jackson 2021). The SEA was also corrected for small sample size as SEA_c (Jackson et al. 2011).

Since these metrics provided only point estimates, uncertainty probabilities were estimated using the posterior distribution (100,000 posterior draws) of each estimate using Bayesian inference (SEA_B) in the SIBER package (Jackson et al. 2011; Jackson 2021). These results were visualized using density box plots showing the median, and the 50%, 75%, and 99% credible intervals of the SEA_B for each group. To test for differences between groups (i.e., is one group's SEA_B smaller than another group's), the proportion of the posterior draws (PPD) that were smaller in the first group compared to the second group was calculated, which was a direct proxy for the probability that the first group's SEA_B was significantly smaller than the second group (i.e., significant difference when $\text{PPD} > 95\%$) (Jackson et al. 2011). Using this method, differences in isotopic niche size for the focal species among shoals and seasons, and between pre- and post-dredge periods, could be statistically evaluated (Jackson et al. 2011; Jackson 2021).

15.2.4 Isotope Analysis of Dredging Events

For Dredge-1, isotope values for $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ (separately) were compared among the individual fall seasons in the pre-dredge and post-dredge periods for each shoal using a nonparametric Kruskal-Wallis test because Levene's test of homogeneity or Shapiro-Wilk's test of normality were significant, indicating that a parametric two-way analysis of variance (ANOVA) was not appropriate. Any differences ($P \leq 0.05$) within each shoal were determined using DSCF Pairwise Two-sided Multiple Comparisons (SAS Institute 2012).

For Dredge-2, four complete seasons were sampled on all four shoals prior to the dredging event and four seasons were sampled during/after the dredging event. It was therefore possible to compare isotope values for $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ (separately) between pre-dredge seasons versus post-dredge seasons for each shoal and focal species using a nonparametric Wilcoxon Rank Sum Test due to significant non-normality and heteroscedasticity, as tested using Shapiro-Wilk's tests and Levene's tests (SAS Institute 2012).

15.3 Results

15.3.1 Acidification of Amphipod Samples

There was no significant difference between $\delta^{13}\text{C}_{\text{acidified}}$ as a function of $\delta^{13}\text{C}_{\text{untreated}}$ for the two amphipod groups of haustorid versus gammarid (ANCOVA, slopes: $F = 0.41$, $P = 0.524$; groups: $F = 0.59$, $P = 0.443$) (Figure 15-1). The pooled regression equation was therefore used to correct the untreated $\delta^{13}\text{C}$ values for acidification as:

$$\delta^{13}\text{C}_{\text{acidified}} = (0.5275 \times \delta^{13}\text{C}_{\text{untreated}}) - 8.7902 \quad (r^2 = 0.586, P < 0.0001) \quad (15-3)$$

Whole amphipod tissue ($\delta^{13}\text{C}_{\text{untreated}}$) was significantly more enriched than acidified-whole amphipod tissue ($\delta^{13}\text{C}_{\text{acidified}}$) (Paired t-test: $t = 17.373$, $P < 0.0001$), as well as muscle-only amphipod tissue ($\delta^{13}\text{C}_{\text{muscle}}$) (Paired t-test: $t = 5.438$, $P < 0.006$). However, the muscle-only samples were not different from the acidified-whole amphipod samples (Paired t-test: $t = 1.122$, $P < 0.289$). Although the sample size was small for this paired comparison ($n = 5$), this indicated that the acidification of the amphipod samples appropriately corrected the whole ground amphipod tissue to muscle-only tissue values and therefore Equation 15-3 was applied to all unacidified amphipod samples.

15.3.2 Isotope Values and Isotopic Niches

15.3.2.1 Amphipods

A biplot of $\delta^{15}\text{N}$ and $\delta^{13}\text{C}$ for amphipods (haustorid and gammarid amphipods combined) indicated similar mean trophic levels (7 to 8 ‰ $\delta^{15}\text{N}$) and mean carbon sources (-17 to -16 ‰ $\delta^{13}\text{C}$) among all shoals (Figure 15-2A). This was also reflected in both the $\delta^{15}\text{N}$ range and the $\delta^{13}\text{C}$ range, which were both < 1 (Table 15-1), indicating that the amphipods on all shoals during all the seasons were feeding over the same number of trophic levels and on the same carbon (basal) resources. There was also low trophic diversity among individual amphipods from each shoal, as measured by the mean distance to the isotopic $\delta^{13}\text{C}$ - $\delta^{15}\text{N}$ centroid of the group ($CD = 0.224$ – 0.413) (Table 15-1).

Overall, isotopic niches as measured by SEA_C were relatively larger on CSII, especially in spring, summer and winter compared to the other shoals (Figure 15-2B, Table 15-2). However, within each season, the niches on CSII were only significantly larger than the niches on Chester Shoal in spring ($\text{PPD} = 0.976$), summer ($\text{PPD} = 0.999$), and winter ($\text{PPD} = 0.972$), and in summer on CSII-BA ($\text{PPD} = 0.968$) (Figure 15-2C). Otherwise, the size of the isotopic niches of amphipods from CSII-BA, the dredged shoal, were not significantly different from the isotopic niches of amphipods from all the other shoals (all $\text{PPD} < 0.95$).

15.3.2.2 Roughneck Shrimp

The Roughneck Shrimp biplot also showed $\delta^{15}\text{N}$ as a function of $\delta^{13}\text{C}$ to be similar among all the shoals with mean trophic levels between 10 to 10.5 ‰ $\delta^{15}\text{N}$ and mean carbon sources between -16 and -17 ‰ $\delta^{13}\text{C}$ (Figure 15-3A). The $\delta^{15}\text{N}$ range was greater than for amphipods but varied little among the shoals (1.21–1.55) (Table 15-1), which indicated that the shrimp were all feeding over the same number of trophic levels on all the shoals. Roughneck Shrimp from all the shoals were also feeding on similar carbon sources with the $\delta^{13}\text{C}$ range < 1 on all shoals except Chester Shoal, which was 1.4 (Table 15-1). This indicated that Roughneck Shrimp on Chester Shoal were consuming prey reflecting a higher diversity of carbon sources relative to the other shoals. Niche metrics of CD for Roughneck Shrimp (0.58–0.75) (Table 15-1) were greater than for amphipods (0.22–0.32), indicating that individual shrimp were feeding on a relatively more diverse prey base.

In general, isotopic niches of Roughneck Shrimp were relatively larger on Bull and Chester Shoals compared to CSII and CSII-BA (Figure 15-3B, Table 15-2). Based on SEA_B pairwise comparisons within seasons, Roughneck Shrimp from Bull Shoal and Chester Shoal had significantly larger isotopic niches than CSII in the spring (PPD = 0.994 and 0.993, respectively) (Figure 15-3C) and CSII-BA in the spring (PPD = 0.999 and 1.000, respectively), but CSII and CSII-BA were not significantly different (PPD = 0.144). Roughneck Shrimp on Chester Shoal also had a larger isotopic niche in the fall than shrimp on CSII (PPD = 0.999) but was not significantly larger than Bull Shoal and CSII-BA in the fall (PPD = 0.033 and 0.052, respectively). Isotopic niches of Roughneck Shrimp from CSII and CSII-BA were not different in the spring (PPD = 0.144) or winter (PPD = 0.857).

15.3.2.3 Aviu Shrimp

The biplot of $\delta^{15}N$ as a function of $\delta^{13}C$ for Aviu Shrimp showed a high degree of similarity among shoals in the mean trophic level ($\sim 10.5 \delta^{15}N$) and carbon source (-18 to $-18.5 \delta^{13}C$) (Figure 15-4A). The $\delta^{15}N$ range for Aviu Shrimp on all shoals was < 1 , indicating that they were feeding primarily at the same trophic levels (Table 15-1). The range observed in $\delta^{13}C$ among shoals, however, was the highest observed among all focal species (1.66–2.11) and indicated that Aviu Shrimp may be drawing upon more diverse carbon (basal) resources compared to the other focal species, especially on Chester Shoal (Table 15-1). This was supported by the relatively higher values for CD as well (0.60–0.75) that indicated that individual Aviu Shrimp had a relatively more diverse prey base compared to amphipods, but similar to Roughneck Shrimp.

Size of isotopic niches for Aviu Shrimp among the shoals within any one season were variable (Figure 15-4B, Table 15-2). In the spring, the isotopic niche of Aviu Shrimp from Chester Shoal was larger than the niche for Aviu Shrimp from Bull Shoal (PPD = 0.965) and CSII (PPD = 0.974) but was not different than CSII-BA (PPD = 0.123) (Figure 15-4C). In summer, isotopic niches of Aviu Shrimp were similar among all shoals (all PPD < 0.95) except for Chester Shoal, where they had a smaller niche than Bull Shoal (PPD = 0.964). In fall, Aviu Shrimp had similar-sized niches among all shoals (all PPD < 0.95). In winter, Aviu Shrimp on CSII-BA had a larger isotopic niche than all other shoals, primarily due to a couple of Aviu Shrimp that appeared to be feeding at a lower trophic level than the rest of the shrimp (Figures 15-4B, 15-4C) (PPD = 0.0.987, 0.995, and 0.997 for Bull, Chester, and CSII, respectively).

15.3.2.4 Spotted Whiff

The biplot for Spotted Whiff showed that all shoals had similar mean $\delta^{15}N$ and mean $\delta^{13}C$ ($\sim 11 \delta^{15}N$ and between -17 and $-17.5 \delta^{13}C$) (Figure 15-5A). As with Roughneck Shrimp, Spotted Whiff had a higher $\delta^{15}N$ -range among all shoals (1.16–1.80), indicating that they were feeding over relatively more diverse trophic levels (Table 15-1). The $\delta^{13}C$ -range was also relatively high (1.07–1.28) (Table 15-1), indicating that Spotted Whiff were also using a relatively wider base of carbon resources. Trophic diversity of Spotted Whiff within each shoal (CD = 0.61–0.86) was similar to that observed in Roughneck Shrimp and Aviu Shrimp, but higher than CD values observed for amphipods, Leopard Searobin, and Atlantic Croaker (Table 15-2).

Overall, isotopic niches of Spotted Whiff were markedly larger in the spring for all shoals and were relatively smaller on all shoals in fall in particular (Figure 15-5B, Table 15-2). Within each season, isotopic niches were only significantly larger in the spring for Spotted Whiff on Bull Shoal compared to Chester Shoal (PPD = 0.988) (Figure 15-5C). Isotopic niches for Spotted Whiff between CSII and CSII-BA were similar within each season (all PPD < 0.95).

15.3.2.5 Leopard Searobin

Leopard Searobin had similar isotope values for mean trophic level (~11 $\delta^{15}\text{N}$) and mean $\delta^{13}\text{C}$ (-16.5 to -17) among all the shoals (Figure 15-6A). The $\delta^{15}\text{N}$ range for Leopard Searobin on all shoals was < 1 (Table 15-1). The range observed in $\delta^{13}\text{C}$ among shoals, however, was the narrowest (0.20-0.33) observed among all the focal species, and the CD was the smallest (0.22–0.36) (Table 15-1). This indicated that Leopard Searobin on all shoals was feeding on prey from a narrow carbon (basal) source and had little trophic diversity among individuals.

Overall, Leopard Searobin isotopic niches were consistently larger in the summer for all shoals, showing more diversity in both the carbon and nitrogen sources utilized, with relatively smaller niches in the other seasons (Figure 15-6B, Table 15-2). Within each season, however, niche sizes were similar among all shoals (Figure 15-6C) (all PPD < 0.95).

15.3.2.6 Atlantic Croaker

A biplot of $\delta^{15}\text{N}$ as a function of $\delta^{13}\text{C}$ for Atlantic Croaker indicated similar mean trophic level (~12 $\delta^{15}\text{N}$) and mean carbon sources (-17 to -17.5 $\delta^{13}\text{C}$) among all shoals (Figure 15-7A). Within each shoal, the $\delta^{15}\text{N}$ -range was relatively narrow (0.29–0.86), as was the $\delta^{13}\text{C}$ -range (0.45–1.2), with Atlantic Croaker from Chester Shoal showing the largest range in carbon resource use (1.2) (Table 15-1). Trophic diversity among individuals from each shoal was also relatively low (0.23–0.46), indicating that Atlantic Croaker were most likely feeding over a relatively narrow trophic level and using prey resources that were feeding over less diverse carbon sources.

Isotopic niches of Croaker were of relatively similar size within each shoal among seasons except for Chester Shoal, which showed a marked change between summer and winter seasons with the niche size in the winter ~7X larger than the niche size in the summer (Figure 15-7B, Table 15-2). Isotopic niche sizes were similar for Croaker in the spring other than for Chester Shoal where the niche size was significantly smaller than the niche size on CSII (PPD = 0.969) (Figure 15-7C). In summer, the isotopic niche size for Croaker from Chester Shoal was smaller compared to all other shoals (PPD = 0.977, 1.000, and 0.978 for Bull, CSII, and CSII-BA, respectively). During fall, Croaker from all shoals had similar-sized isotopic niches (all PPD < 0.95). As noted previously, in winter, Croaker from Chester Shoal had a larger isotopic niche compared to all other shoals (PPD = 1.000, 0.995, 0.998 for Bull, CSII, and CSII-BA, respectively).

15.3.3 Dredging Events

15.3.3.1 Dredge-1 Event

Amphipod $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ did not differ between fall 2014, 2015, and 2016 on Bull Shoal (C: $\chi^2 = 3.806$, P = 0.149; N: $\chi^2 = 0.472$, P = 0.790) or CSII-BA (C: $\chi^2 = 2.489$, P = 0.288; N: $\chi^2 = 0.001$, P = 0.999) (Table 15-3). Amphipod sample size was too low to test for differences on CSII.

Similarly, Roughneck Shrimp $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ did not differ between fall 2014, 2015, and 2016 on Bull Shoal (C: $\chi^2 = 3.289$, P = 0.193; N: $\chi^2 = 5.422$, P = 0.066) or CSII-BA (C: $\chi^2 = 0.694$, P = 0.707; N: $\chi^2 = 4.250$, P = 0.119). Roughneck Shrimp sample size was too low on CSII to test for differences among years.

Aviu Shrimp differed significantly in $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ on all shoals between pre- and post-dredge fall periods (Figure 15-8, Table 15-3), except for $\delta^{15}\text{N}$ on Bull Shoal ($\chi^2 = 3.342$, P = 0.188). However, carbon isotope values were different on Bull Shoal ($\chi^2 = 6.414$, P = 0.040), with fall 2013 more enriched than fall 2014 but not different than fall 2015, and fall 2014 not different from fall 2015. Similarly, Aviu Shrimp carbon isotope values for CSII were different among fall 2013, 2014, and 2015 ($\chi^2 = 6.709$, P = 0.035), with fall 2013 more enriched than fall 2014 and fall 2015, but with no difference between fall 2014 and

fall 2015. Nitrogen isotopes for Aviu Shrimp from CSII were also different among fall 2013–15 ($\chi^2 = 5.982$, $P = 0.050$), with fall 2013 lower than fall 2014 and fall 2015, but with fall 2014 similar to fall 2015. For CSII-BA, $\delta^{13}\text{C}$ was different among fall 2013, 2014, and 2015 ($\chi^2 = 9.346$, $P = 0.009$), with fall 2013 enriched compared to fall 2014 and fall 2015, but with fall 2014 similar to fall 2015, similar to both Bull and CSII Shoals. Nitrogen isotope values were also different among the 3 years on CSII-BA ($\chi^2 = 8.628$, $P = 0.013$), with fall 2013 higher than fall 2014 and fall 2015 but fall 2014 not different than fall 2015.

As with amphipods and Roughneck Shrimp, $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ did not differ for Spotted Whiff between fall 2014, 2015, and 2016 on Bull Shoal (C: $\chi^2 = 5.833$, $P = 0.054$; N: $\chi^2 = 3.889$, $P = 0.143$) or CSII-BA (C: $\chi^2 = 1.746$, $P = 0.081$; N: $\chi^2 = 0.427$, $P = 0.513$). Spotted Whiff sample size was too low on CSII to test for differences among years.

Leopard Searobin also did not differ in $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ between fall 2014, 2015, and 2016 on Bull Shoal (C: $\chi^2 = 5.189$, $P = 0.075$; N: $\chi^2 = 1.869$, $P = 0.393$) or CSII-BA (C: $\chi^2 = 0.909$, $P = 0.635$; N: $\chi^2 = 5.576$, $P = 0.062$). Leopard Searobin sample size was too low on CSII to test for differences among years.

For Atlantic Croaker, the pre-dredge $\delta^{13}\text{C}$ isotope values in fall 2013 were not significantly different than post-dredge $\delta^{13}\text{C}$ values in fall 2014 and fall 2015 on CSII-BA ($\chi^2 = 1.487$, $P = 0.475$), CSII ($\chi^2 = 3.516$, $P = 0.172$), or Bull Shoal ($\chi^2 = 2.609$, $P = 0.272$). Nitrogen isotopes were also not different among fall 2013, 2014, and 2015 on CSII-BA ($\chi^2 = 1.445$, $P = 0.486$), CSII ($\chi^2 = 0.422$, $P = 0.810$), or Bull Shoal ($\chi^2 = 5.238$, $P = 0.073$).

15.3.3.2 Dredge-2 Event

Isotope values of $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ for all focal species for all shoals were not different between the pre- and post-dredge periods, with a few exceptions (Table 15-4). The $\delta^{13}\text{C}$ values for Amphipods on Chester Shoal were significantly more enriched in the pre-dredge period (median = -16.01) than in the post-dredge period (median = -16.67) but overall the difference (0.66) was small. All other differences were observed on Bull Shoal, where the $\delta^{15}\text{N}$ values gave a higher trophic level to Atlantic Croaker in the pre-dredge seasons (median = 11.75) compared to the post-dredge seasons (median = 12.15) ($P = 0.018$). The difference between these two values (0.40), however, was not equivalent to a full trophic level difference that would be expected to be ~ 3.4 ‰ $\delta^{15}\text{N}$. Bull Shoal also showed two differences in $\delta^{13}\text{C}$ values for Aviu Shrimp and Leopard Searobin with pre-dredge values (-18.28 and -16.55, respectively) more enriched than post-dredge values (-18.59 and -16.78, respectively) (Table 15-4). However, these differences in basal carbon resource isotope values were small (0.31 and 0.23, respectively) and likely not biologically meaningful. Pre- and post-dredge isotope values for $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ were not different on the dredged shoal (CSII-BA) for any of the focal species (Table 15-4).

15.4 Discussion

15.4.1 Trophic Similarities Among Shoals and Seasons Based on Isotopes

Isotopic composition and niches provide a synthesized view of the pathway of prey resources used by different predators, but also incorporate the variation in the prey resources themselves. The stable isotopes of nitrogen and carbon reflect both the trophic level of the predator and the basal carbon resources used by the prey it consumes, which in turn reflects the basal carbon resources of the food source that the prey feeds upon and assimilates (Post 2002; Fry 2006). For CSII-BA, CSII, Bull, and Chester Shoals, the $\delta^{13}\text{C}$ - $\delta^{15}\text{N}$ biplots within each focal species (Figures 15-2A to 15-7A) were similar among all shoals, indicating that these species were feeding at a similar trophic level and relying on similar basal carbon resources independent of the specific shoal. Overall, this was also reflected in the overall size of the isotopic niches for each species for all shoals over the duration of the study, which indicated a high degree of overlap for

each species on all four shoals (Figure 15-9). Based on these focal species, this indicated that all four of the shoals were operating on an overall similar trophic basis. When viewed on a seasonal basis (Figures 15-2B to 15-7B), there were some differences among the shoals and seasons for each focal species, but these differences were generally small and occurred on the non-dredged shoals (CSII, Bull, and Chester Shoals) as much or more than CSII-BA, the dredged shoal. Even though CSII-BA has been dredged multiple times since 2000 (Table 1-1), its overall trophic structure revealed through $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ isotopes of focal species was similar to the non-dredged shoals.

Overall, the size of the isotopic niches of focal species on a seasonal basis were also relatively similar among the shoals (Figures 15-2C to 15-7C) with few differences directly related to the dredged shoal, CSII-BA. Small, omnivorous amphipods (Chapter 11) had a larger isotopic niche on CSII compared to CSII-BA during the summer, but the isotopic niches on CSII were also larger for amphipods during the summer (and the spring and winter) compared to Chester Shoal, which was a reference shoal (Figure 15-2 B, C). This indicated that any differences between the isotopic niches of amphipods on CSII-BA could be ascribed to the natural variation observed among the shoals and not directly due to dredging events.

Roughneck Shrimp, a larger benthic omnivore (Chapter 12) with more mobility compared to amphipods, had larger isotopic niches on Bull and Chester Shoals in most seasons relative to CSII and CSII-BA (Figure 15-3B). However, CSII and CSII-BA had similar niche sizes for spring and winter and had reversed differences during summer and fall (e.g., CSII > CSII-BA in the summer but CSII-BA > CSII in the fall). This indicated that dredging events on CSII-BA had either not resulted in any differences, or no consistent difference, between the two shoals.

Aviu Shrimp of the Family Sergestidae are small pelagic omnivores (Chapter 12) that inhabit tropical to temperate oceans and various salinities from estuarine to coastal waters (Xiao and Greenwood 1993). In general, sergestid shrimps primarily feed on crustacean zooplankton, especially copepods (Flock and Hopkins 1992), but various species are also known to feed on a diverse array of prey, including phytoplankton, molluscan veligers, and amorphous particulate organic matter (e.g., combination of phytoplankton and protozoans) (Metillo 2011; Oh et al. 2011; Metillo et al. 2016). They therefore play an important role in coastal food webs through their intermediate trophic position whereby they consume dissolved particulate organic matter/detritus, phytoplankton, and zooplankton, as reflected in their isotope niche space, and in turn are preyed on by higher trophic levels, in particular fishes, squids, penaeid prawns, and baleen whales (Xiao and Greenwood 1993; Metillo 2011; Moura and Siciliano 2012; Metillo et al. 2016). The threatened Giant Manta Ray *Mobula birostris* that frequent the shoal areas off Cape Canaveral (Farmer et al. 2022) have also been observed actively feeding near the surface in the fall (October) and at night (pers. obs. Murie lab) and may also be taking advantage of aggregations of *Acetes* on the east coast of Florida. The large range in $\delta^{13}\text{C}$ values observed for Aviu Shrimp over all the shoals also attested to the diversity in the species' use of available carbon resources, further indicating its important trophic linkage between primary producers/consumers and higher trophic levels.

Metillo (2011) referred to *Acetes*, in particular, as the “Antarctic Krill” of the tropics since it aggregates in large coastal “swarms” (Metillo et al. 2016). Abundance of *Acetes* in many countries is vast, especially the Indo-Pacific, where it forms the basis for commercial fisheries that amount to ~15% of the global shrimp catch (Omori 1978). Off the east coast of Florida, *Acetes* contributes to the prey base of many commercially and recreationally important fish species [e.g., Weakfish *Cynoscion regalis*, Southern Kingfish *Menticirrhus americanus*, and Atlantic Croaker *Micropogonias undulatus* (Willis et al. 2015)]. Most sergestid shrimps vertically migrate on a daily basis, moving into the epi-pelagic zone (0-200 m depth) at night to feed on plankton and then returning to the upper meso-pelagic zone (200-600 m depth) at dawn (Omori and Gluck 1979; Flock and Hopkins 1992). Flock and Hopkins (1992) suggested, however, that vertical distribution of sergestids is regionally specific and that the shallower depth distribution of *Sergestes* in the eastern Gulf of Mexico compared to more oceanic regions (e.g., Bermuda,

Hawaii) may be due to temperature differences, as well as the depth of light penetration, with sergestid distribution shallower when light penetration is also shallower. Omori and Gluck (1979) also noted that sergestids have ontogenetic shifts in their vertical migration, with larvae off the southern California coast relatively abundant at night in 0–40 m depths. Although *Acetes* is known to feed primarily at night (Metillo 2011), the shoals off the east coast of Florida may be too shallow (< 50 m) for typical diel vertical migration. However, Aviu Shrimp may be available to foraging fish species throughout the day because of the overall shallow depth, but have increased availability at night, especially in high turbidity, when the shrimp are actively foraging nearer the surface.

Overall, the focal fish species, including Spotted Whiff, Leopard Searobin, and Atlantic Croaker, demonstrated similarities in isotopic composition and isotope niche size among the shoals and differences related specifically to the dredged shoal (CSII-BA) compared to the non-dredged shoals (Bull, Chester, and CSII) were not obvious. Notably, isotopic composition was similar among shoals for each fish species. In addition, seasonal changes in isotope niche size were similar across all shoals in most instances. For example, the increase in isotopic niche size for Leopard Searobin in the summer was observed across all shoals (Figure 15-6 B and C), as was the decrease in isotope niche size for Spotted Whiff in the fall (Figure 15-5 B and C). Differences that did arise mainly occurred between Bull Shoal and CSII showing differences compared to Chester Shoal and CSII-BA (i.e., isotope niche size for Spotted Whiff in the spring; Figure 15-5 B and C) or relative to Chester Shoal (i.e., isotope niche size markedly larger in the winter for Atlantic Croaker on Chester Shoal; Figure 15-7C). This latter observation was also supported by the large $\delta^{13}\text{C}$ -range observed for Atlantic Croaker from Chester Shoal, which indicated that the fish were feeding on prey using various basal carbon resources. Without direct observation of comparative stomach contents of Atlantic Croaker, however, it was not possible to know why their diet, as indicated by isotope metrics, changed so markedly during the winter on Chester Shoal.

Even in relative isotope niche comparisons, such as used in this study, it is desirable to have information on basal end members, or the source of the $\delta^{15}\text{N}$ and $\delta^{13}\text{C}$ at the base of the food web (Post 2002). Basal end members are preferably primary producers, including both benthic aquatic plants and pelagic phytoplankton, as well as detritus, that are ascribed as the original source of the carbon and nitrogen at the base of the food web. It is important to have some indication of any change in these basal end members among ecosystems being compared because if changes are observed in higher trophic levels it will be unclear whether the change has actually occurred at the observed trophic level without knowing if the basal end members were different among the ecosystems to start with (Post 2002).

Obtaining isotope samples from basal end members, however, is known to be one of the most difficult problems for studies that occur over extended time periods and multiple ecosystems (Post 2002). Among the shoals in this study, attached primary producers (e.g., benthic algae) and seagrasses were sparse due to the limited amount of hard or permanent substrate. Small red algae species were the most common primary producers that were primarily attached to benthic invertebrate shells, for example *Ceramium* and *Botryocladia*; these were very sparse and inconsistently available over all shoals and seasons. In addition, the isotope values for the red algae collected on the study shoals ranged between -30 and -35 ‰ $\delta^{13}\text{C}$, and as such the algae was most likely not providing any of the basal end members used by the focal species (i.e., all focal species had $\delta^{13}\text{C}$ values between approximately -14 and -21 ‰) (Figure 15-9). Similarly, although IRL is close by and is a potential source of both seagrass leaves (approx. -10 to -11 ‰ $\delta^{13}\text{C}$; Fourqurean and Schrlau 2003) and mangrove leaves (approx. -28 ‰ $\delta^{13}\text{C}$; Fourqurean and Schrlau 2003), both of these sources of primary production were observed rarely and in small quantities in the trawls while sampling the offshore shoals. Detritus from seagrasses and mangroves hold similar basal carbon end member isotope values as the leaves of the plants (Fourqurean and Schrlau 2003). It is therefore also unlikely that seagrasses or mangroves are the source of the basal primary productivity in the food web of the sand shoals.

Three other potential sources of primary producers in the shoal ecosystems were phytoplankton, microphytobenthos, and drifting/wracking *Sargassum*. Phytoplankton, in general, ranges from -19 to -24 ‰ $\delta^{13}\text{C}$ (average of -20) and 4-8 ‰ $\delta^{15}\text{N}$ (Fry 2006), similar to values by Rooker et al. (2006) where particulate organic matter that was used as a proxy for phytoplankton in the Gulf of Mexico averaged -21 ‰ $\delta^{13}\text{C}$ and 7.1 ‰ $\delta^{15}\text{N}$ (Figure 15-9). Zooplankton feeding on phytoplankton would be expected, on average, to be enriched by ~1 ‰ $\delta^{13}\text{C}$ and ~3.4 ‰ $\delta^{15}\text{N}$ (Figure 15-9). This indicated that phytoplankton via zooplankton was most likely the primary basal carbon resource being used by *Acetes* in particular, but also the majority of the focal species sampled over the shoals, with the exception of the amphipods.

Microphytobenthos was not measured for isotopes directly in the study. When basal end members are difficult to collect and analyze over all the relevant study sites and time periods, the next best option to obtain basal carbon isotope values is to collect species that prey directly on the basal end member (Post 2002). Post (2002), for example, used a filter-feeding mussel to estimate the basal $\delta^{13}\text{C}$ of a pelagic ecosystem [e.g., phytoplankton and particulate organic matter filtered out of the water column] and a surface-grazing snail to estimate the basal $\delta^{13}\text{C}$ of a littoral (shore) ecosystem (e.g., snails graze on periphyton and detritus). To obtain an overall proxy for a basal carbon resource for the sand shoals in this study, we consistently sampled amphipods from all shoals over all seasons. Amphipods are omnivores and feed on benthic/epibenthic detritus and microphytobenthos. Their overall mean $\delta^{13}\text{C}$ was -16.2 ‰ (Figure 15-9) and their isotopic niches among all shoal and seasons were similar, even though their isotopic niches on CSII were larger overall (Figure 15-2); on a study level, the isotopic niches of amphipods were consistent among all the shoals (Figure 15-9). Similarly, Moncreiff and Sullivan (2001) and Dauby (1995) reported that sand microflora had $\delta^{13}\text{C}$ values between -18 and -16 ‰.

Although not a focus of the current study, floating wracks of drifting *Sargassum* were present in large quantities throughout the study period. Wracks of this large brown algae would sink upon deterioration of its air bladders and was trawled up off the sea floor in a mostly decomposing condition; small quantities were observed attached to invertebrate shells. The carbon isotope of *Sargassum* and *Sargassum* detritus in the Gulf of Mexico ranged from -16 to -18 ‰ $\delta^{13}\text{C}$ and 2.5-2.8 ‰ $\delta^{15}\text{N}$ (Rooker et al. 2006) (Figure 15-9), similar to values of -16.6 to -16.8 ‰ $\delta^{13}\text{C}$ reported for *Sargassum* by Moncreiff and Sullivan (2001) (although their range for $\delta^{15}\text{N}$ values were enriched in comparison at 4.5 to 4.7 ‰). It was therefore possible that amphipods in particular were using *Sargassum* and *Sargassum* detritus as a basal carbon source in combination with microphytobenthos, but it was unlikely that *Sargassum* contributed substantially to the basal carbon resources of the other focal species directly. However, further studies employing both isotope and fatty acid signatures would be beneficial in elucidating the role of *Sargassum* in the trophic pathways of the shoal ecosystems (Rooker et al. 2006).

15.4.2 Isotopic Impacts of Dredge-1

For all focal invertebrate and fish species, except for Aviu Shrimp, there were no differences in the isotopic composition in pre-dredge $\delta^{13}\text{C}$ or $\delta^{15}\text{N}$ isotope values in fall 2013 compared to post-dredge $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values in fall 2014 and fall 2015 for CSII-BA, CSII, or Bull Shoals. Direct comparison of isotopic composition of these species before and after the potential impact to the dredged shoal (CSII-BA) relative to reference (control) shoals (CSII and Bull Shoals) (i.e., a BACI design) provided strong evidence that dredging CSII-BA did not impact the trophic structure of the shoal, as tracked through key focal species.

For Aviu Shrimp, $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values were different between the pre-dredge fall 2013 and the post-dredge periods of fall 2014 and fall 2015, but the differences were similar for all of the shoals, not just the dredged shoal of CSII-BA (Figure 15-8). Composition of $\delta^{13}\text{C}$ was consistently depleted by ~1‰ from fall 2013 to fall 2014 and 2015 on all shoals. The $\delta^{15}\text{N}$ composition between pre- and post-dredge fall periods was more variable and ranged from no difference among the fall periods on Bull Shoal to either increased or decreased enrichment between fall 2013 and falls 2014/2015 on CSII and CSII-BA,

respectively. However, the average change in the $\delta^{15}\text{N}$ composition values between the pre- and post-dredge fall periods was $\sim 0.5\text{‰}$ in both cases, which is much less than one trophic level change that would be expected to be $\sim 3.4\text{‰}$ (Post 2002). It was therefore clear that the Aviu Shrimp were still functioning at the same trophic level in the pre- versus post-dredge periods but the basal carbon resource(s) that they were feeding on, or that their prey were feeding on, had shifted slightly and was more depleted on all the shoals.

15.4.3 Isotopic Impacts of Dredge-2

Changes in the basal carbon resources used, as reflected in $\delta^{13}\text{C}$ values, did not differ in pre- versus post-dredge seasons among the shoals relative to the second dredging event in winter 2017/18 through spring 2018. A few differences in $\delta^{13}\text{C}$ values were observed on Bull Shoal for Aviu Shrimp and Leopard Searobin, and Atlantic Croaker on Chester Shoal, but these differences reflected small changes in the basal carbon resources used by the prey of these fish species. Most importantly, CSII-BA, the dredged shoal, did not show any differences in $\delta^{13}\text{C}$ values of the pre- and post-dredge periods for any of the focal species. This indicated that dredging did not fundamentally change the carbon resources at the base of the food web, as observed through focal species that spanned different trophic levels and feeding lifestyles. The natural variation in the $\delta^{13}\text{C}$ values observed on the non-dredged shoals (i.e., Bull and Chester Shoals for a few of the focal species) indicated that relatively small changes to the basal carbon resources can occur without any direct connection to a dredging event (i.e., Bull and Chester Shoals were not dredged). Shifts in basal carbon resources available can potentially occur due to a multitude of natural and anthropogenic causes (e.g., large-scale storm events moving sediment, intrusion of river water discharge during flood events).

Similarly, there were no significant changes in the trophic levels of the focal species, as indicated by the $\delta^{15}\text{N}$ values, in pre- versus post-dredge seasons among the shoals relative to the second dredging event; with the one exception of a small trophic enrichment (11.75 to 12.15 $\delta^{15}\text{N}$) in Atlantic Croaker on Bull Shoal (a non-dredged shoal). CSII-BA, the dredged shoal, showed no changes in the trophic level of any of the focal species. This indicated that dredging did not fundamentally change the prey base of these focal species. In addition, the one change that was observed with Atlantic Croaker on Bull Shoal was small (0.40 $\delta^{15}\text{N}$ enrichment in the post-dredge seasons) and was not equivalent to a full trophic level where the expectation would be a difference of $\sim 3.4\text{‰}$ $\delta^{15}\text{N}$ (Post 2002). This indicated that Atlantic Croaker most likely incorporated either another prey resource into their diet that had a slightly greater $\delta^{15}\text{N}$ itself, or that it changed the relative composition of its prey, that was then incorporated into the muscle of Atlantic Croaker, rather than Atlantic Croaker actually feeding at a higher trophic level itself. This is plausible because Atlantic Croaker feed primarily on fish, but also feed on crustaceans such as *Acetes*, as well as ophiuroids (brittle stars) (Willis et al. 2015; pers. obs., D. Murie). Brittle stars have diverse feeding habits ranging from being carnivores through either scavenging or trapping small animals like amphipods, to suspension feeding by trapping particles from the water column, to deposit feeding on detritus (Warner 1982). Increasing the amount of fish consumed relative to the amount of brittle stars consumed would therefore enrich the $\delta^{15}\text{N}$ composition of Atlantic Croaker without elevating it to a higher trophic level.

15.4.4 Conclusions

Overall, seasonal isotopic composition and isotope niche sizes of amphipods, Roughneck Shrimp, Aviu Shrimp, Leopard Searobin, Spotted Whiff, and Atlantic Croaker varied among the reference shoals as much or more than between the reference shoals and the dredged shoal. For all focal species except Aviu Shrimp, there were no differences in their trophic position based on their $\delta^{13}\text{C}$ or $\delta^{15}\text{N}$ isotope values in fall 2013 (i.e., immediately prior to the first dredging event in winter 2013/14 and spring 2014) compared to post-dredge seasons (fall 2014 and fall 2015) for CSII-BA, CSII, or Bull Shoals. The trophic positions of Aviu Shrimp, specifically, changed between pre- and post-dredge fall periods associated with the first

dredging event, but they did so simultaneously on all three shoals, and therefore the change could not be ascribed to the dredging event itself. In addition, the isotope values of $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ for all focal species for all shoals were not different between the pre- and post-dredge periods for the second dredging event in winter 2017/18 and spring 2018, with few exceptions that were related to the natural variability among the non-dredged shoals. Overall, the stability in the trophic positions of the focal invertebrate and fish species during pre- versus post-dredge seasons, and within the dredged shoal relative to the reference shoals, indicated that dredging did not fundamentally change the trophic state and linkages of the food web of the shoals, as observed through focal species that spanned different trophic levels and feeding lifestyles.

A caveat in this analysis is that isotope values from muscle samples used in this study can represent assimilation of resources over months; however, it was assumed that the values adequately captured seasonal differences within the season that the organisms were captured. To examine these similarities and differences on a more exact timescale, it would be most informative to use samples of blood from the organisms, which give information on prey resource assimilation within days of capture.

15.5 Acknowledgments

Mike Dickson, Monica Ditch, Bethan Gillett, Fabio Caltabellotta, Lucas Jennings, Colin Frank, Patrick Baker, Alex Linares, and Kerrin Toner contributed to the field work and sample processing for this project and we are sincerely thankful for their dedicated teamwork. In addition, over the span of this project there have been dozens of biologists, undergraduate biological technicians, and undergraduate student volunteers that have participated and assisted in field and lab work, which was greatly appreciated and collectively provided substantial assistance. All isotopes samples were analyzed in the Stable Isotope Mass Spec Lab at UF and we greatly appreciated the assistance of Dr. Jason Curtis in providing guidance on sample preparation and processing.

Permits for sampling of fish occurred under a LOA with NOAA/NMFS. Field sampling and laboratory processing of fish samples occurred under IACUC protocols #201709892 and #2020009892 at UF.

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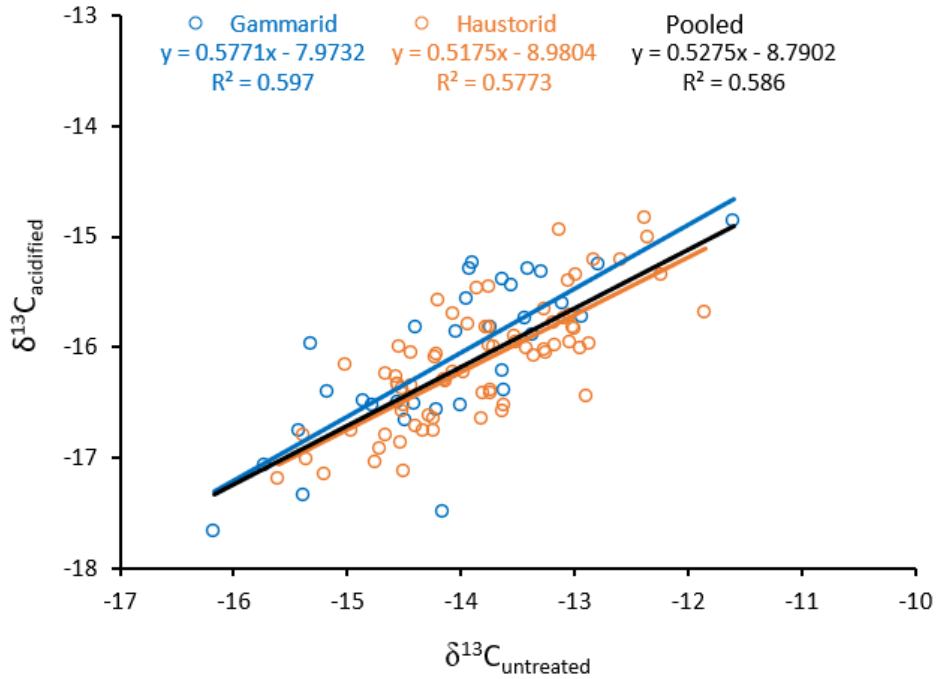


Figure 15-1. Relationship between $\delta^{13}\text{C}_{\text{acidified}}$ and $\delta^{13}\text{C}_{\text{untreated}}$ for gammarid and haustorid amphipods collected from the four shoals.

Differences between the two amphipod groups was not significant and the groups were therefore pooled to provide a predictive regression for estimating $\delta^{13}\text{C}_{\text{acidified}}$ values for those samples that only had $\delta^{13}\text{C}_{\text{untreated}}$ values (i.e., samples were not directly acidified).

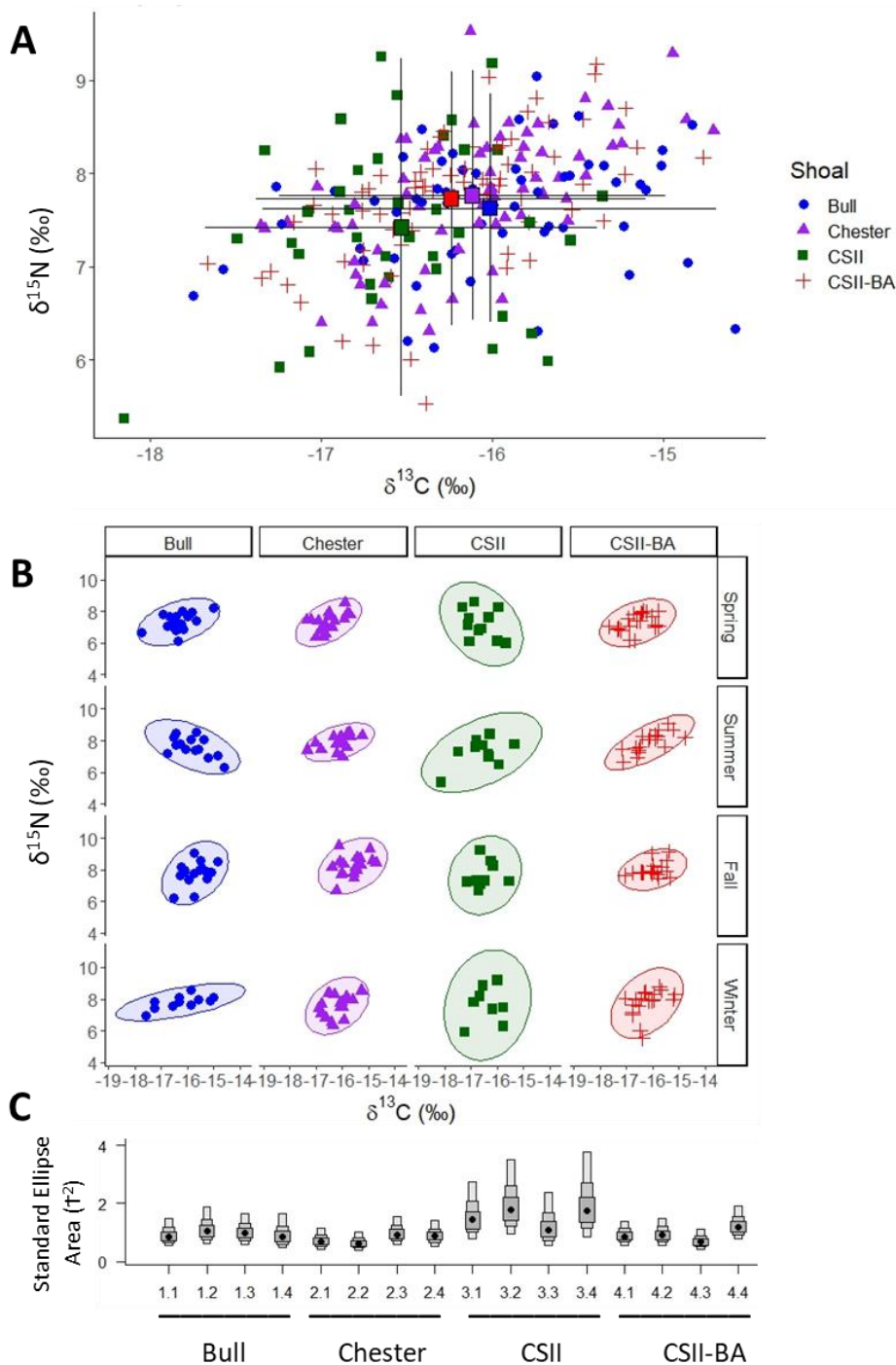


Figure 15-2. A) Biplot of $\delta^{15}\text{N}$ and $\delta^{13}\text{C}_{\text{corrected}}$ on an annual basis for amphipods; B) size-corrected standard ellipse areas (SEAC) as a function of shoal and season; and C) posterior Bayesian estimates of the standard ellipse areas (SEAB) as a function of shoal and season.

A) Solid symbols represent the overall mean with standard error bars for each shoal; B) ellipses represent the 95% confidence ellipse of the mean for the standard ellipse area corrected for small sample size (SEAC); and C) box plots represent the median (black dot), and the 50th, 75th, and 95th credible intervals (shaded greys) of the standard ellipse area for amphipods by shoal and season (season is denoted by the second digit in the x-axis categories, where 1 = spring, 2 = summer, 3 = fall, and 4 = winter).

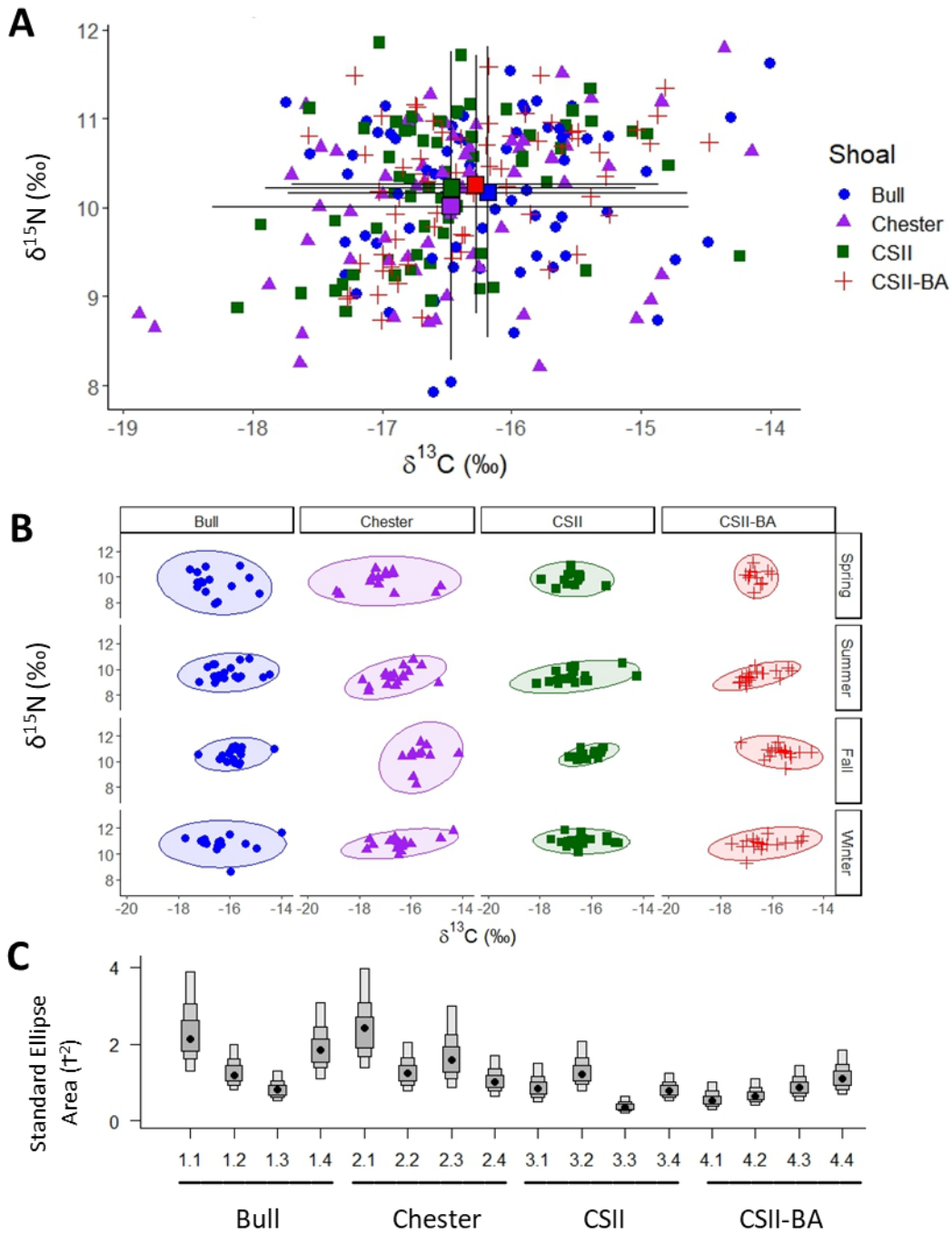


Figure 15-3. A) Biplot of $\delta^{15}\text{N}$ and $\delta^{13}\text{C}_{\text{corrected}}$ on an annual basis for Roughneck Shrimp; B) size-corrected standard ellipse areas (SEAc) as a function of shoal and season; and C) posterior Bayesian estimates of the standard ellipse areas (SEAB) as a function of shoal and season. Symbols are as given in Figure 15-2.

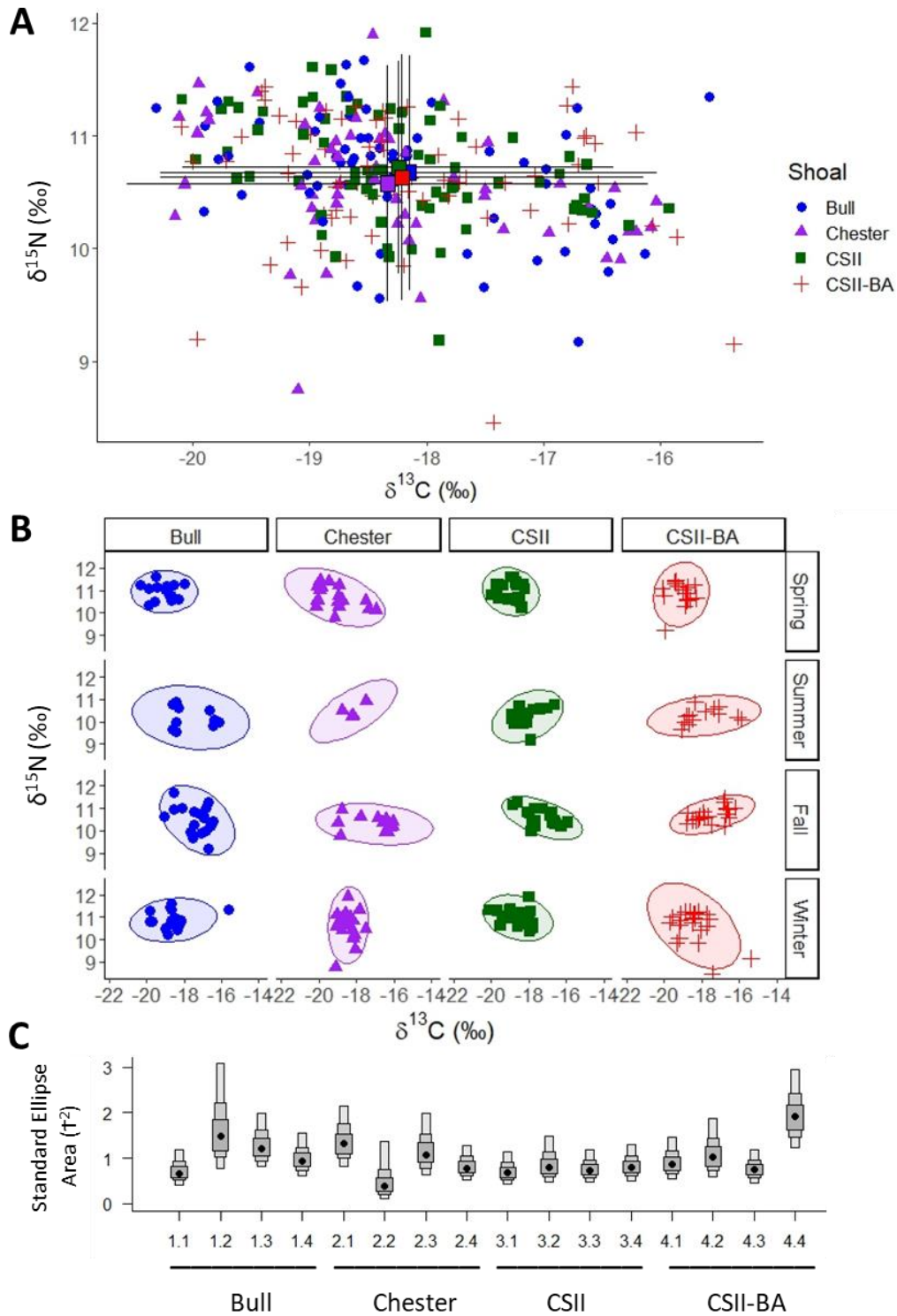


Figure 15-4. A) Biplot of $\delta^{15}\text{N}$ and $\delta^{13}\text{C}_{\text{corrected}}$ on an annual basis for Aviu Shrimp; B) size-corrected standard ellipse areas (SEAc) as a function of shoal and season; and C) posterior Bayesian estimates of the standard ellipse areas (SEAB) as a function of shoal and season.

Symbols are as given in Figure 15-2.

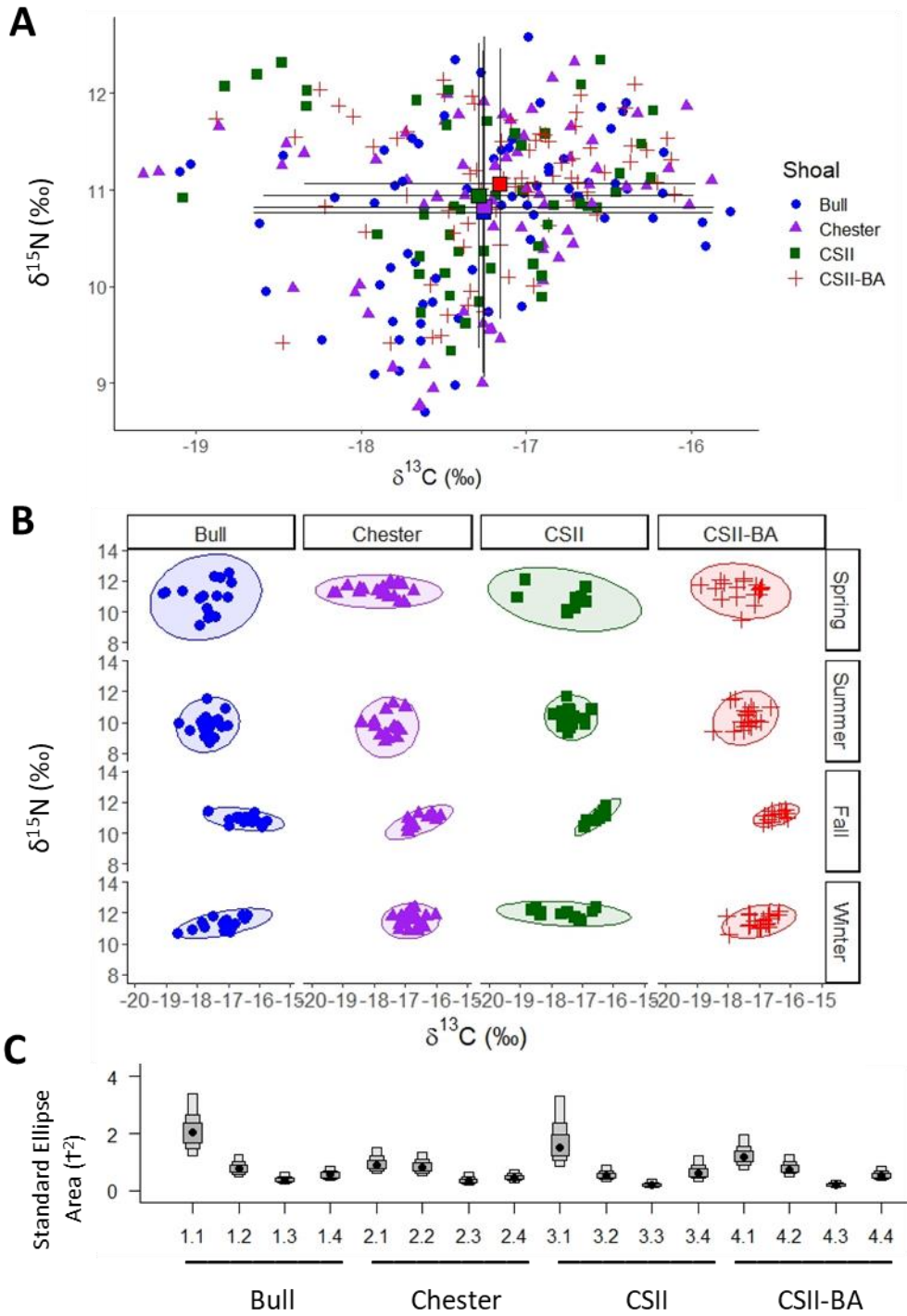


Figure 15-5. A) Biplot of $\delta^{15}\text{N}$ and $\delta^{13}\text{C}_{\text{corrected}}$ on an annual basis for Spotted Whiff; B) size-corrected standard ellipse areas (SEA_C) as a function of shoal and season; and C) posterior Bayesian estimates of the standard ellipse areas (SEA_B) as a function of shoal and season.

Symbols are as given in Figure 15-2.

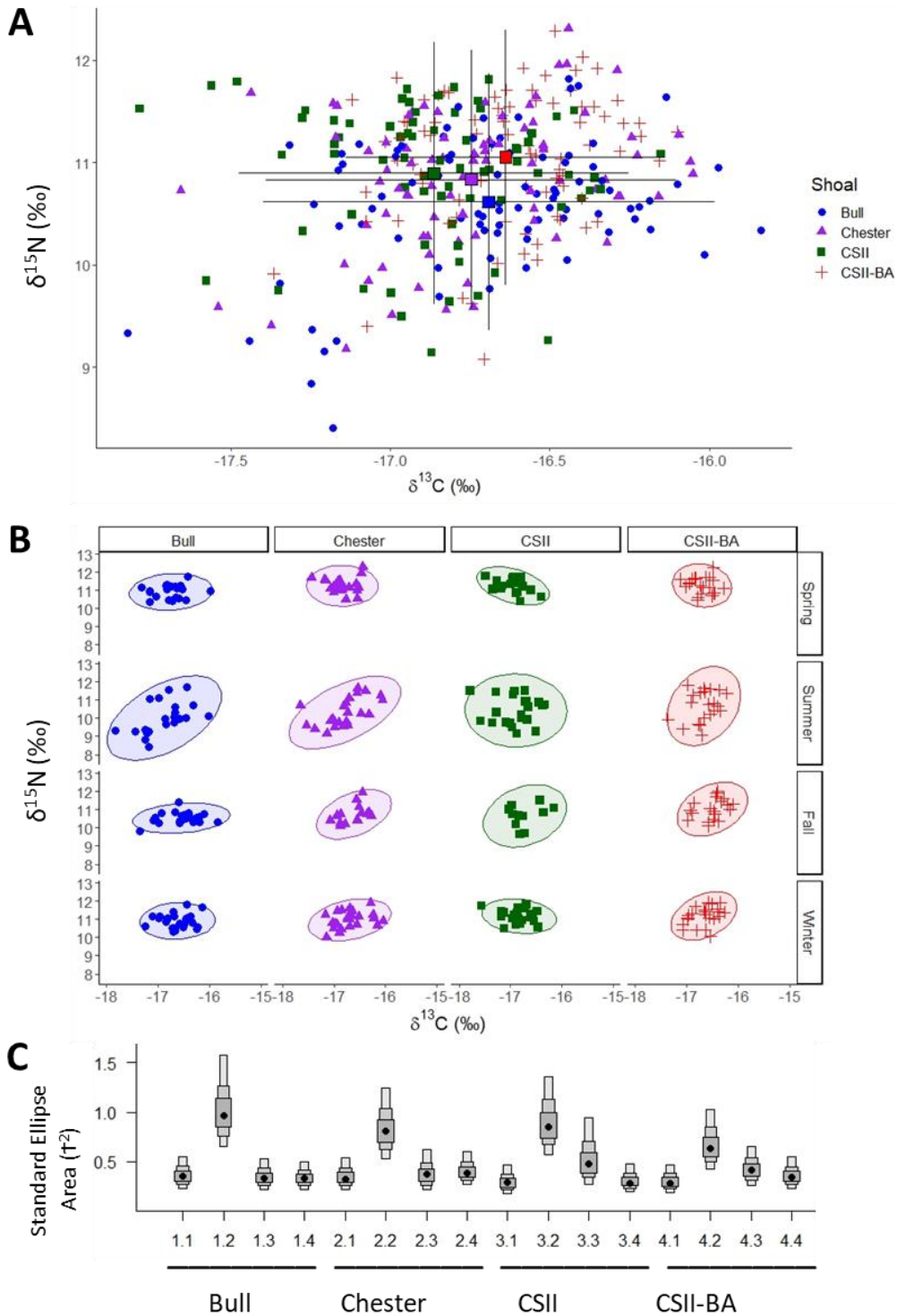


Figure 15-6. A) Biplot of $\delta^{15}\text{N}$ and $\delta^{13}\text{C}_{\text{corrected}}$ on an annual basis for Leopard Searobin; **B)** size-corrected standard ellipse areas (SEAc) as a function of shoal and season; and **C)** posterior Bayesian estimates of the standard ellipse areas (SEAB) as a function of shoal and season. Symbols are as given in Figure 15-2.

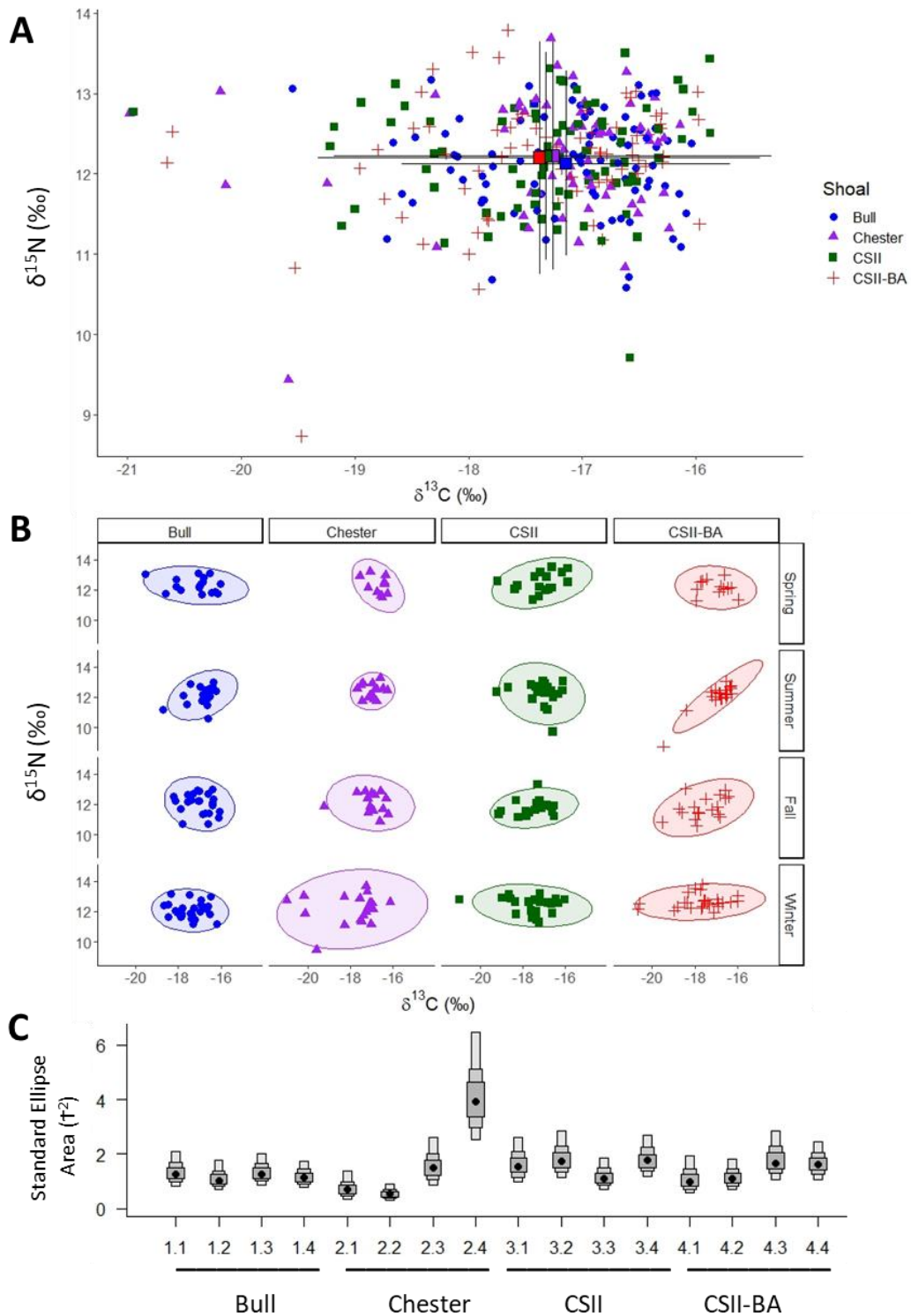


Figure 15-7. A) Biplot of $\delta^{15}\text{N}$ and $\delta^{13}\text{C}_{\text{corrected}}$ on an annual basis for Atlantic Croaker; B) size-corrected standard ellipse areas (SEAc) as a function of shoal and season; and C) posterior Bayesian estimates of the standard ellipse areas (SEAB) as a function of shoal and season.

Symbols are as given in Figure 15-2.

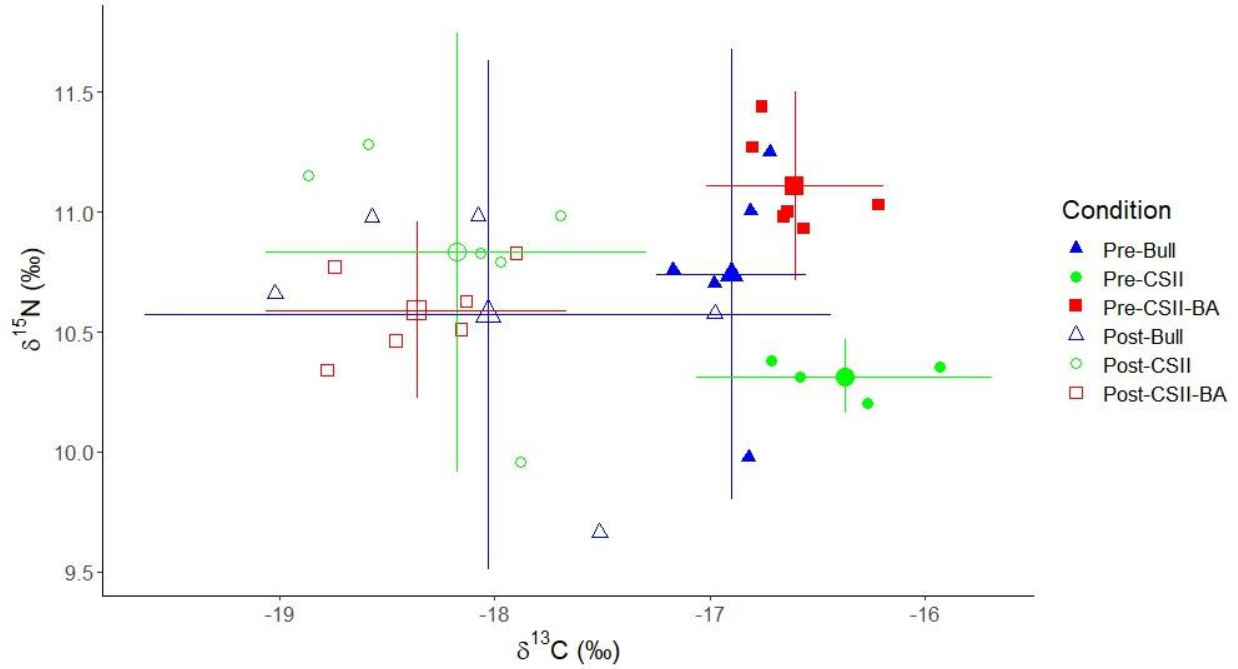


Figure 15-8. Biplot of $\delta^{15}\text{N}$ and $\delta^{13}\text{C}_{\text{corrected}}$ for Aviu Shrimp showing differences in the Dredge-1 Event between pre-dredge (fall 2013) and post-dredge fall periods (fall 2014 and fall 2015) for Bull, CSII, and CSII-BA shoals.

Small symbols represent individual Aviu Shrimp values and large symbols represent the means for the pre- and post-dredge fall periods.

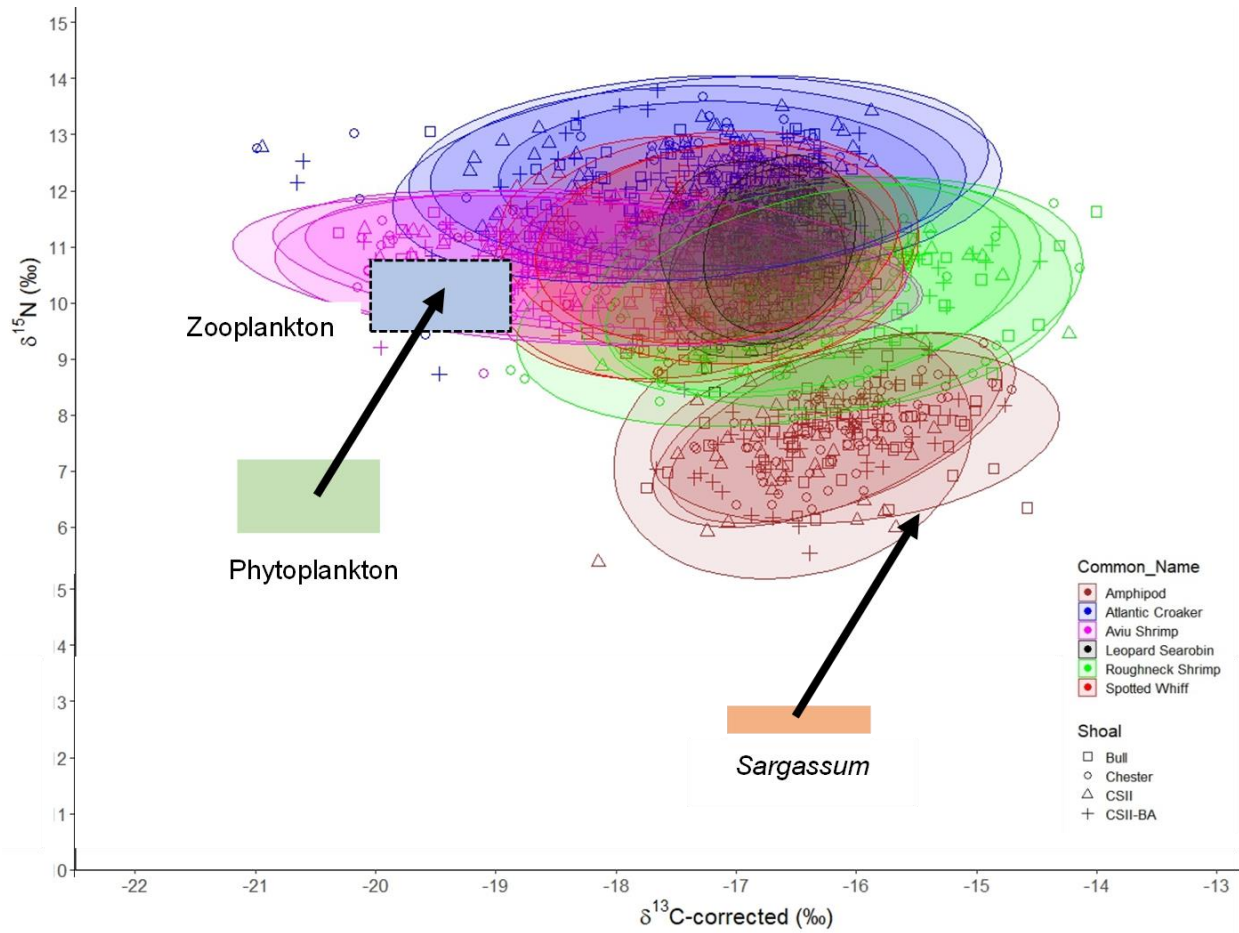


Figure 15-9. Standardized ellipse areas (SEAc) over the study period for focal species by shoal. Average and range of isotope values for phytoplankton, zooplankton, and *Sargassum* are given by the bottom of the arrow and the boxes, respectively. Arrows indicate the predicted direction and size of the trophic position as ~ 1 ‰ $\delta^{13}\text{C}_{\text{corrected}}$ and ~ 3.4 ‰ $\delta^{15}\text{N}$ (see text for specific details).

Table 15-1. Layman metrics for focal invertebrate and fish species among all shoals.

Focal Species	Shoal	$\delta^{15}\text{N}$ Range (NR)	$\delta^{13}\text{C}$ Range (CR)	Mean Distance to Centroid (CD)
Amphipods	Bull	0.471	0.628	0.321
Amphipods	Chester	0.918	0.929	0.413
Amphipods	CSII	0.457	0.228	0.224
Amphipods	CSII-BA	0.723	0.575	0.283
Atlantic Croaker	Bull	0.287	0.586	0.235
Atlantic Croaker	Chester	0.305	1.212	0.412
Atlantic Croaker	CSII	0.693	0.448	0.324
Atlantic Croaker	CSII-BA	0.857	0.833	0.456
Aviu Shrimp	Bull	0.766	1.767	0.751
Aviu Shrimp	Chester	0.405	2.114	0.621
Aviu Shrimp	CSII	0.737	1.665	0.664
Aviu Shrimp	CSII-BA	0.572	1.665	0.600
Leopard Searobin	Bull	0.905	0.334	0.356
Leopard Searobin	Chester	0.934	0.228	0.313
Leopard Searobin	CSII	0.824	0.234	0.348
Leopard Searobin	CSII-BA	0.612	0.203	0.226
Roughneck Shrimp	Bull	1.215	0.743	0.583
Roughneck Shrimp	Chester	1.430	1.410	0.753
Roughneck Shrimp	CSII	1.558	0.646	0.638
Roughneck Shrimp	CSII-BA	1.310	0.912	0.625
Spotted Whiff	Bull	1.480	1.227	0.713
Spotted Whiff	Chester	1.801	1.285	0.856
Spotted Whiff	CSII	1.630	1.074	0.703
Spotted Whiff	CSII-BA	1.158	1.139	0.615

Table 15-2. Standard Ellipse Area (SEA_c) for isotope niches of focal invertebrate and fish species for all shoals and seasons.

SEA was corrected for small sample sizes to give SEA_c; area is given in ‰ (per mil)².

Shoal	Season	Amphipods	Roughneck Shrimp	Aviu Shrimp	Spotted Whiff	Leopard Searobin	Atlantic Croaker
Bull	Spring	0.96	2.49	0.77	2.26	0.38	1.43
Bull	Summer	1.17	1.37	1.82	0.84	1.05	1.17
Bull	Fall	1.08	0.88	1.34	0.41	0.37	1.39
Bull	Winter	0.96	2.05	1.05	0.58	0.35	1.23
Chester	Spring	0.76	2.58	1.44	1.02	0.37	0.82
Chester	Summer	0.67	1.35	0.59	0.91	0.85	0.59
Chester	Fall	1.03	1.87	1.25	0.38	0.40	1.67
Chester	Winter	0.95	1.12	0.88	0.50	0.42	4.31
CSII	Spring	1.68	0.97	0.75	1.95	0.31	1.73
CSII	Summer	2.08	1.37	0.92	0.58	0.95	1.93
CSII	Fall	1.37	0.40	0.80	0.19	0.57	1.25
CSII	Winter	2.21	0.86	0.87	0.73	0.32	1.88
CSII-BA	Spring	0.94	0.63	0.98	1.31	0.32	1.19
CSII-BA	Summer	0.94	0.70	1.17	0.85	0.71	1.01
CSII-BA	Fall	0.74	0.96	0.79	0.22	0.45	1.89
CSII-BA	Winter	1.30	1.22	2.01	0.57	0.38	1.74

Table 15-3. Differences in carbon and nitrogen isotopes for Dredge-1 event, between pre-dredge fall 2013 and post-dredge fall periods (fall 2014 and fall 2015) for focal invertebrate and fish species sampled from Bull, CSII, and CSII-BA shoals.

Differences are indicated by Kruskal-Wallis χ^2 values with significance indicated by P-values in bold.

Focal Species	Shoal	Isotope	n	χ^2	P-value
Amphipods	Bull	C	8	3.806	0.149
Amphipods	Bull	N	8	0.472	0.790
Amphipods	CSII	C	3	NA	NA
Amphipods	CSII	N	3	NA	NA
Amphipods	CSII-BA	C	9	2.489	0.288
Amphipods	CSII-BA	N	9	0.001	0.999
Atlantic Croaker	Bull	C	14	2.610	0.271
Atlantic Croaker	Bull	N	14	5.238	0.073
Atlantic Croaker	CSII	C	13	3.517	0.172
Atlantic Croaker	CSII	N	13	0.422	0.810
Atlantic Croaker	CSII-BA	C	12	1.487	0.475
Atlantic Croaker	CSII-BA	N	12	1.445	0.486
Aviu Shrimp	Bull	C	10	6.415	0.041
Aviu Shrimp	Bull	N	10	3.342	0.188
Aviu Shrimp	CSII	C	10	6.709	0.035
Aviu Shrimp	CSII	N	10	5.982	0.050
Aviu Shrimp	CSII-BA	C	12	9.346	0.009
Aviu Shrimp	CSII-BA	N	12	8.628	0.013
Leopard Searobin	Bull	C	14	5.189	0.075
Leopard Searobin	Bull	N	14	1.869	0.393
Leopard Searobin	CSII	C	4	NA	NA
Leopard Searobin	CSII	N	4	NA	NA
Leopard Searobin	CSII-BA	C	11	0.909	0.635
Leopard Searobin	CSII-BA	N	11	5.576	0.062
Roughneck Shrimp	Bull	C	9	3.289	0.193
Roughneck Shrimp	Bull	N	9	5.422	0.067
Roughneck Shrimp	CSII	C	6	NA	NA
Roughneck Shrimp	CSII	N	1	NA	NA
Roughneck Shrimp	CSII-BA	C	8	0.694	0.707
Roughneck Shrimp	CSII-BA	N	8	4.250	0.119
Spotted Whiff	Bull	C	8	5.833	0.054
Spotted Whiff	Bull	N	8	3.889	0.143
Spotted Whiff	CSII	C	1	NA	NA
Spotted Whiff	CSII	N	1	NA	NA
Spotted Whiff	CSII-BA	C	6	1.746	0.081
Spotted Whiff	CSII-BA	N	6	0.429	0.513

Table 15-4. Differences in carbon and nitrogen isotopes for the Dredge-2 event for focal invertebrate and fish species sampled from Bull, Chester, CSII, and CSII-BA shoals.

Pre-dredge seasons included winter 2016/17, spring 2017, summer 2017, and fall 2017, and dredging/post-dredging seasons included winter 2017/18, spring 2018, summer 2018, and fall 2018. Differences are indicated by Wilcoxon Rank Sum Z scores with significance indicated by P-values in bold.

Focal Species	Shoal	Isotope	n	Z Score	P-value
Amphipods	Bull	C	18	-0.091	0.928
Amphipods	Bull	N	18	0.543	0.587
Amphipods	Chester	C	23	2.185	0.039
Amphipods	Chester	N	23	-0.277	0.782
Amphipods	CSII	C	18	-1.236	0.216
Amphipods	CSII	N	18	0.795	0.427
Amphipods	CSII-BA	C	26	-0.359	0.719
Amphipods	CSII-BA	N	26	0.513	0.608
Atlantic Croaker	Bull	C	23	0.893	0.372
Atlantic Croaker	Bull	N	23	-2.369	0.018
Atlantic Croaker	Chester	C	23	-1.446	0.148
Atlantic Croaker	Chester	N	23	-1.446	0.148
Atlantic Croaker	CSII	C	22	0.231	0.818
Atlantic Croaker	CSII	N	22	-1.286	0.198
Atlantic Croaker	CSII-BA	C	20	-0.880	0.379
Atlantic Croaker	CSII-BA	N	20	1.232	0.218
Aviu Shrimp	Bull	C	23	1.954	0.050
Aviu Shrimp	Bull	N	23	-0.651	0.515
Aviu Shrimp	Chester	C	22	1.002	0.316
Aviu Shrimp	Chester	N	22	-1.269	0.204
Aviu Shrimp	CSII	C	24	-0.375	0.708
Aviu Shrimp	CSII	N	24	0.491	0.624
Aviu Shrimp	CSII-BA	C	21	0.669	0.504
Aviu Shrimp	CSII-BA	N	21	-0.669	0.504
Leopard Searobin	Bull	C	24	-2.454	0.014
Leopard Searobin	Bull	N	24	-0.433	0.665
Leopard Searobin	Chester	C	24	-0.549	0.583
Leopard Searobin	Chester	N	24	0.433	0.665
Leopard Searobin	CSII	C	23	-0.523	0.601
Leopard Searobin	CSII	N	23	-1.569	0.117
Leopard Searobin	CSII-BA	C	24	-0.779	0.436
Leopard Searobin	CSII-BA	N	24	0.144	0.885

Focal Species	Shoal	Isotope	n	Z Score	P-value
Roughneck Shrimp	Bull	C	24	-1.761	0.078
Roughneck Shrimp	Bull	N	24	1.068	0.286
Roughneck Shrimp	Chester	C	24	-1.068	0.286
Roughneck Shrimp	Chester	N	24	1.126	0.260
Roughneck Shrimp	CSII	C	24	0.260	0.795
Roughneck Shrimp	CSII	N	24	0.664	0.507
Roughneck Shrimp	CSII-BA	C	23	0.708	0.479
Roughneck Shrimp	CSII-BA	N	23	0.000	1.000
Spotted Whiff	Bull	C	20	-1.119	0.263
Spotted Whiff	Bull	N	20	-0.116	0.908
Spotted Whiff	Chester	C	24	-1.184	0.237
Spotted Whiff	Chester	N	24	0.433	0.665
Spotted Whiff	CSII	C	22	0.394	0.694
Spotted Whiff	CSII	N	22	-0.591	0.554
Spotted Whiff	CSII-BA	C	25	0.027	0.978
Spotted Whiff	CSII-BA	N	25	0.027	0.978

16 Ecopath Modeling of Shoal Ecosystems in Relation to Dredging Events

Daniel Vilas, David Chagaris, Robert Ahrens, Debra Murie, Geoff Smith, Don Behringer, Lucas Jennings, Patrick Baker, Colin Frank, and Ed Philips

Key Points

- Data across multiple trophic levels were integrated into a modeling framework for a series of ecosystem models to evaluate the impacts of dredging activities in the borrow area of Canaveral Shoals II (CSII-BA) and reference (control) shoals (CSII and Chester Shoals).
- Ecological indicators, such as production flow rate (P), total system throughput (TST), and consumption flow rate (Q), were compared across shoals and seasons to determine whether dredging has impacted ecosystem structure and function.
- According to the modeled ecological indicators, there were no clear effects of dredging on the CSII-BA ecosystem based on seasonal sampling over a study period of 6 years. Most ecological indicators showed greater similarities between CSII-BA (dredged shoal) and Chester (reference shoal) over seasons than between CSII-BA and CSII, which indicated no impact on CSII-BA due to dredging events.
- Regarding ecosystem flows, in spring 2014 when CSII-BA was dredged, it had a production flow rate (P) in proportion to total system throughput (TST) (P/TST) and a consumption flow rate (Q) in proportion to TST (Q/TST) that was similar to CSII, which indicated that the rates were similar between dredged and non-dredged shoals.
- Relative to CSII, both CSII-BA and Chester Shoals had increases in P/TST and decreases in Q/TST during spring 2018 when CSII-BA was dredged. This finding suggested a potential change in the community structure and composition, but changes occurred on both shoals and therefore were unrelated to any dredging impact.
- Similar total biomass (excluding detritus) among the shoals in spring 2014 also indicated a lack of dredging impact on CSII-BA, and CSII-BA and Chester Shoals remained similar but relatively low compared to CSII. Although CSII-BA total biomass was lower in spring 2018 (second dredging event on CSII-BA) compared to both Chester and CSII, it was not significantly lower and had fully recovered by summer 2018, indicating that any impact from dredging was very short lived.
- In conclusion, the ecosystem model analysis suggested that the ecological indicators of the dredged shoal (CSII-BA) were mostly similar to the reference shoal (Chester), and any differences between the two shoals were recovered within one season following the dredging event. After two dredging events at CSII-BA, the scale and nature of any differences in the dredged ecosystem was comparable to the control shoal, suggesting that this difference or change may also be explained by the dynamic variability in the system.

16.1 Introduction

A marine ecosystem is composed of living and non-living elements that are connected to one another through food-web and habitat interactions. Marine ecosystems are increasingly being threatened by environmental (e.g., climate, meteorology) and anthropogenic factors such as pollution, habitat alteration, and overfishing (Brander et al. 2010). From a global perspective, 41% of marine ecosystems have been affected by multiple anthropogenic drivers and all marine ecosystems have been affected by at least some human activity (Halpern et al. 2008). Considering that impact, there is a special need to understand, evaluate, and track the potential effects that environmental and anthropogenic stressors have on each marine ecosystem.

Ecosystem based management (EBM) is an integrated approach to managing natural resources that recognizes the interconnectedness among ecosystem components and spans multiple use sectors (e.g., fishing, shipping, energy exploration, tourism). Dell'Apa et al. (2015) evaluated the status of EBM activities for 62 programs across 13 different U.S. federal agencies under the National Ocean Council. They found that all federal programs included in the study employ some level of EBM principles but there were substantial differences in how EBM is perceived and implemented across programs. Programs that focus on resource extraction, such as NOAA and BOEM, implement a greater number of EBM practices and principles than non-extractive programs. Ecosystem models are widely considered a powerful and necessary tool to support EBM practices. For example, Ecopath with Ecosim (EwE) is a popular software package for modeling aquatic and marine ecosystems and is able to address a wide variety of policy, management, and environmental questions (Christensen and Walters 2004; Colléter et al. 2015).

The potential effects of dredging on marine ecosystems are well documented (Tillan et al. 2011). In certain contexts and circumstances, dredging activities may severely impact biological communities and entire marine ecosystems (Cooper et al. 2008; Wenger et al. 2017). Prior to this study, no researchers have conducted an ecosystem modeling assessment to evaluate the potential impact or impact pathways of offshore shoal ecosystems in the southeast U.S. that are routinely dredged for beach renourishment projects. Here, we developed a series of ecosystem models to evaluate the impacts of dredging activities in the borrow area of Canaveral Shoals II (CSII-BA), a sand shoal off the east coast of Florida. The objectives were to integrate data across multiple trophic levels into a modeling framework for dredged (CSII-BA) and reference (control) shoals (CSII and Chester Shoals), and compare ecological indicators across shoals and seasons to determine whether dredging has impacted ecosystem structure and function.

16.2 Methods

16.2.1 Modeling Approach

To assess the dredging impact and evaluate the state of the ecosystem, multiple quantitative food-web models using an EwE (version 6.6) approach were developed (Christensen and Walters 2004; Christensen et al. 2008). Four seasonal Ecopath models (fall, winter, spring, and summer) for each year from fall 2013 to summer 2019 were built for the Canaveral Shoals II (CSII) and the borrow area (dredged) of CSII (CSII-BA), and Chester Shoal as a reference (control) site. A total of 72 Ecopath models were developed (6 years x 4 seasons x 3 shoals).

An Ecopath model represents a static mass-balanced snapshot of the ecosystem comprised of functional groups. A functional group is an individual species or aggregation of species that perform a similar function in the ecosystem, such as similar growth rates, consumption rates, diets, habitats and predators (Heymans et al. 2016). The basic inputs for functional groups in an Ecopath model are biomass density, consumption rates, mortality rates, and diet composition. The parameterization in Ecopath is based on two

master equations (Christensen et al. 2008) (Equations 1 and 2). The first Ecopath equation describes the energy balance within a group and maintains balance between assimilated consumption, production, and respiration:

$$\text{Consumption (Q)} = \text{production (P)} + \text{respiration (R)} + \text{unassimilated food (E)} \quad (1)$$

The second one describes how the production term for each functional group is partitioned into predation, fishing mortality, and migration processes:

$$\begin{aligned} \text{Production} = & \text{Fishing mortality} + \text{Predation mortality} + \text{Biomass accumulation} \\ & + \text{Net migration} + \text{Other mortality} \end{aligned} \quad (2)$$

or, more formally:

$$P_i = Y_i + B_i M2_i + E_i + BA_i + P_i(1 - EE_i) \quad (3)$$

where P_i is the total production rate of functional group i , Y_i is the total fishery catch rate of functional group i , $M2_i$ is the total predation rate for functional group i , B_i is the biomass of functional group i , E_i is the net migration rate, BA_i is the biomass accumulation rate for functional group i , and $P_i(1-EE_i)$ is the other natural mortality excluding predation (also known as $M0$). $M0$ represents a catch-all term including all mortality not accounted for by fishing or predation (e.g., mortality due to disease or old age). Ecotrophic efficiency (EE) is the proportion of production for functional group i that is used within the system or exported due to catches and/or migration (Christensen and Walters 2004; Christensen et al. 2008). Equation 2 can be re-expressed as:

$$\left(\frac{P}{B}\right)_i \cdot B_i = Y_i + \sum_j B_j \cdot \left(\frac{Q}{B}\right)_j \cdot DC_{ji} + E_i + BA_i + \left(\frac{P}{B}\right)_i \cdot B_i(1 - EE_i) \quad (4)$$

where $(P/B)_i$ is the production to biomass ratio for functional group i , $(Q/B)_j$ is the consumption to biomass ratio of functional group j , and DC_{ji} is the proportion of functional group i in the diet of functional group j . The energy balance within each functional group is ensured when consumption is equal to the production, respiration, and food that is unassimilated (Equation 1). An Ecopath model is considered ecologically and thermodynamically balanced when (1) $EE < 1.0$; (2) production consumption ratio (P/Q) values are between 0.05 and 0.35 with the exception of fast growing groups, such as bacteria; (3) R/A (respiration/food assimilation) < 1 ; (4) R/B (respiration/biomass) range from 1 to 10 for fishes and higher values for small organisms; (5) NE (net efficiency of food conversion) $> GE$ ($= P/Q$), where GE is gross efficiency; and (6) P/R (production/respiration) < 1 (Christensen et al. 2008; Heymans et al. 2016).

16.2.2 Model Parameterization

To represent Chester, CSII, and CSII-BA, we defined the functional groups based on biological and ecological features of species such as diets, commercial value and data availability. Ecopath models for all locations have the same functional groups in order to facilitate comparison among them. All species/groups were assigned into functional groups based on published diet studies, reference literature, and direct observation, as detailed in Table 16-1. These models all contain the same 39 functional groups and include 17 fish groups (FGs = 1–17, 34), 19 invertebrate groups (FGs = 18–33, 35), two primary producer (FGs = 36, 38), and one detritus group (FGs = 37) (Table 16-1).

Biomass estimates for the functional groups were determined from randomly stratified locations within ridge and swale habitats for each shoal (Chester, CSII, and CSII-BA) for each season (fall, winter, spring and summer) and each year (from 2013 to 2019). Various sampling methods were used in order to obtain

biomass estimates for the functional groups, which are detailed in Chapters 7–13. In summary, phytoplankton biomass estimates were obtained by determining the phytoplankton species composition from representative water samples taken using a Niskin water sampler, and then converting composition to biovolume. Bacterioplankton biomass was similarly estimated from the water samples using compositional data converted to biovolume. Sediment surface algae biomass estimates were obtained by chlorophyll extraction of sediment cores from benthic grabs. Detritus biomass estimates were obtained from drying and combusting sediment cores from benthic grabs to determine organic matter. Zooplankton biomass estimates were also obtained from the water samples, which were filtered over a 41- μm mesh. Benthic cores were used to sample benthic invertebrate infaunal and epifaunal species for biomass estimates. Biomass estimates for demersal invertebrates and fishes were obtained using bottom trawls. In cases where a functional group was not observed at a particular season or site, a negligible biomass value (10^{-5} t km^2) was input into the model. Production (P/B, year^{-1}) and consumption (Q/B, year^{-1}) rates were obtained from previously developed Ecopath models (Okey and Pugliese 2001; Chagaris 2013) (Table 16-2).

A binary (presence/absence) diet matrix (Table 16-3) was converted into proportion diet as required by Ecopath. For each Ecopath model, a diet matrix was built considering that the diet composition from prey i to predator j (DC_{ij}) is the product of prey biomass B_i and binary diet a_i (where a_i is 0 or 1) divided by the sum of the products of biomass and binary diet over all prey items (Equation 5). This procedure assumed that diet composition was proportional to the availability of prey in the ecosystem and electivity was equal for all prey items in each predator diet. The electivity describes a predator's preference for prey, where a value of 1 indicates a total preference for a prey, -1 indicates total avoidance, and 0 indicates that prey is consumed in proportion to their abundances. This method allowed us to be consistent during model development and mass-balance procedure and allowed diet proportions to change according to seasonal fluctuations in prey biomass as measured by the various surveys.

$$DC_{ij} = \frac{(B_i a_i)}{\sum_i (B_i a_i)} \quad (5)$$

In this study, a generic stepwise procedure was followed in order to maintain consistency across models and facilitate comparisons. Most of the models showed initial EE values higher than 1 for some functional groups. However, we considered a model acceptable if the EEs were all less than 5. Initially, three models showed “highly unbalanced” EE values caused by excess predation mortality. Therefore, we adjusted the diet matrix by assuming that some proportion of predator diet was “imported” from outside the system (see bottom of Table 16-2). The use of “diet import” is common in Ecopath models of small, open systems with transient species. Additionally, the balancing procedure identified functional groups with high biomass compared to prey functional groups (Lizardfishes & Snakefishes) and a diet comprised of just a single prey item, which consequently caused high predation mortalities (Polychaetes & Nemerteaans groups). In those cases, the binary diet matrix was expanded to include more potential prey items and therefore distribute the consumption more broadly across functional groups.

No formal calibration of the Ecopath models was conducted, as these represent snapshots with fixed inputs for each site and season. We did conduct “pre-bal” diagnostics to check that vital rates (production, consumption, and respiration ratios) were consistent with ecological theory. Accounting for the full uncertainty in the models would not be informative, because critical site-specific diet data are lacking altogether (i.e., the uncertainty in data is unquantified). It is expected that if the full uncertainty of the models (derived from the true uncertainty in the data) were represented, any signal dredging impacts would be obscured.

16.2.3 Ecological indicators

To evaluate the state and functioning of the ecosystem, several ecological indicators were extracted from the modeling approach. Considering ecosystem flows, the sum of all production and consumption flows were selected because of their information on the distribution of how energy flows through the system (Table 16-4). Regarding the size of the ecosystem, total system throughput (TST) and total biomass excluding detritus were selected. TST indicates the ecological size of the ecosystem and total biomass may reflect the impact of dredging on the ecosystem. Additionally, the Finn's Cycling Index (FCI, %) (Finn 1976) is the fraction of the ecosystem's throughput that is recycled and the Finn's mean path length (MPL) (Finn 1976) is the average number of groups that energy flows through and is an indicator of stress. Therefore, lower FCI and MPL are indicative of a stressed and less resilient ecosystem (Christensen 1995).

16.3 Results

The production and consumption flow ratio (P/TST and Q/TST) showed similar trends between 2013 and 2019 among ecosystems (Figure 16-1). P/TST oscillated over time between 0.33 and 0.47 except for spring 2015 in CSII where P/TST was 0.26. CSII was the non-dredged shoal and spring 2015 was 1 year post-dredging on CSII-BA (dredged shoal). Similarly, Q/TST fluctuated between 0.15 to 0.75 except for spring 2015 in CSII where it was 1.15. P/TST and Q/TST values on dredging years fell within the 95% confidence interval for individual shoals in spring, except for Chester in 2014 where P/TST and Q/TST were outside the 95% confidence interval (Figure 16-2). These values were lower for the P/TST and higher for the Q/TST for these years in spring.

The TST over time showed similarities among ecosystems from 2013 to 2016, after which Chester Shoal displayed higher TST in spring 2017 and 2018 and CSII-BA Shoal in fall 2017 and 2018 compared to CSII and Chester Shoals (Figure 16-1). In line with that, TST showed a wider distribution of values for spring at Chester Shoal (Figure 16-2), indicating high natural variability in the reference (control) shoal. Additionally, all three ecosystems obtained high values of TST in spring 2018 and fell outside their respective 95% confidence interval (Figure 16-2). For the total biomass (excluding detritus) indicator, Chester and CSII-BA ecosystems showed similar values over time while CSII was higher in fall 2014, spring 2015, and spring 2018 (Figure 16-1). Accordingly, spring 2018 values were higher for all three ecosystems and did not fall within its variability, similar to TST results (Figure 16-2). FCI oscillated over time between 5 and 45%, with the control shoal, Chester, with relatively low FCI from summer 2014 to spring 2015 and again from winter 2016 to summer 2017 (Figure 16-1). FCI for CSII was relatively high compared to CSII-BA in winter 2013, summer 2016, and winter 2017–spring 2018. Seasonally, FCI for Chester was high in spring 2014 and fell outside its 95% confidence interval (Figure 16-2). Overall, FCI was the most variable for all three shoals during the summer season (Figure 16-2). Finn's MPL was similar among ecosystems except for a quite high value for CSII in spring 2015 (Figure 16-1). Seasonally, MPL was high for spring 2014 in Chester and fell outside its 95% confidence interval (Figure 16-2).

16.4 Discussion

According to the modeled ecological indicators, there were no clear effects of dredging on the CSII-BA ecosystem based on seasonal sampling over a study period of 6 years. Most ecological indicators showed greater similarities between CSII-BA (dredged shoal) and Chester (reference shoal) than between CSII-BA and CSII, which may indicate no impact on CSII-BA due to dredging events.

Regarding ecosystem flows, in spring 2014 when CSII-BA was dredged it had a production flow rate in proportion to TST (P/TST) and a consumption flow rate in proportion to TST (Q/TST) that was similar to

CSII, which indicated that the rates were similar between dredged and non-dredged shoals. Relative to CSII, both CSII-BA and Chester Shoals had increases in P/TST and decreases in Q/TST during spring 2018 when CSII-BA was dredged. This suggested a potential change in the community structure and composition, but changes occurred on both dredged and reference shoals and therefore was unrelated to any dredging impact.

Similarly, total biomass (excluding detritus) among the shoals in spring 2014 also indicated a lack of dredging impact on CSII-BA, and CSII-BA and Chester Shoals remained similar but relatively low compared to CSII. Although CSII-BA total biomass was lower in spring 2018 (second dredging event on CSII-BA) compared to both Chester and CSII, it was not significantly lower than other spring estimates and was more similar to the reference shoals by summer 2018, indicating that any impact from dredging was very short lived.

FCI and MPL are indicators of relative ecosystem maturity. Higher values indicate mature (less disturbed) system states with more diversity of flows and greater recycling. In line with biomass indicator results, seasonal Finn indicators were similar among all the shoals, with the spring 2014 dredging of CSII-BA leading to low Finn indicators. Chester Shoal had slightly higher Finn indicators relative to CSII-BA and CSII. In spring 2018, when CSII-BA was dredged again, its FCI was marginally lower than Chester and CSII Shoals but again had recovered by summer 2018, indicating any change was short lived. The almost identical Finn's MPL between CSII-BA and Chester Shoal from spring 2018 on indicated that there was no impact due to dredging alone. CSII exhibited similar values to previous ecosystem models on the east Atlantic Coast (Monaco and Ulanowicz 1997).

Ecological indicators did not suggest that there were any notable impacts from dredging at an ecosystem level, and the estimated oscillation of ecological indicators may be due to the inherent variability in the system at a seasonal level (Brooks et al. 2006). This was similar to a seasonal effect that was observed in demersal invertebrate and fish assemblages (Chapters 12 and 13). Since high variability of ecological indicators were also exhibited at the reference shoal (Chester), indicators may suggest that dredging impact is not outside the bounds of natural variation in these systems. In addition, estimated values of ecological indicators were in the range of values of other ecosystems (Heymans et al. 2014).

We acknowledge a considerable amount of uncertainty in the trophic structure, since there was no quantitative site-specific dietary information. Models could have been largely unbalanced (high EE values) if diet compositions were not scaled to prey abundance. This consideration assumes that predators are consuming prey in proportion to their abundance in the environment (Murdoch 1969), which is a reasonable assumption for generalist consumers found on the study shoals. Nevertheless, the assumption that connectedness (i.e., number of pred-prey interactions) was equal among shoals and diet was always proportional to prey biomass may conceal true patterns of ecosystem indicators and obscure any differences between dredged and non-dredged shoals (Christensen and Walters 2004).

Here we used Ecopath static snapshots to describe the ecosystem configuration for each season-year and shoal rather than Ecosim, which is the temporal dynamic component inside EwE (Christensen et al. 2008). In theory, Ecosim may be a better tool to predict ecological indicators over time because it is not bound by mass-balance constraints and predator-prey populations can fluctuate according to trophic functional responses and environmental drivers. In this study, Ecopath was used because of the unavailability of timeseries data to fit the model, particularly the biomass of animals removed by dredging.

In conclusion, the ecosystem model analysis suggested that the ecological indicators of the dredged shoal (CSII-BA) were mostly similar to the reference shoal (Chester), and whenever any differences were noted they were recovered with one season following the dredging event. After two dredging events at CSII-BA, the scale and nature of any differences in the dredged ecosystem were comparable to the reference

shoal, suggesting that this difference or change may also be explained by the dynamic variability in the system.

16.5 References

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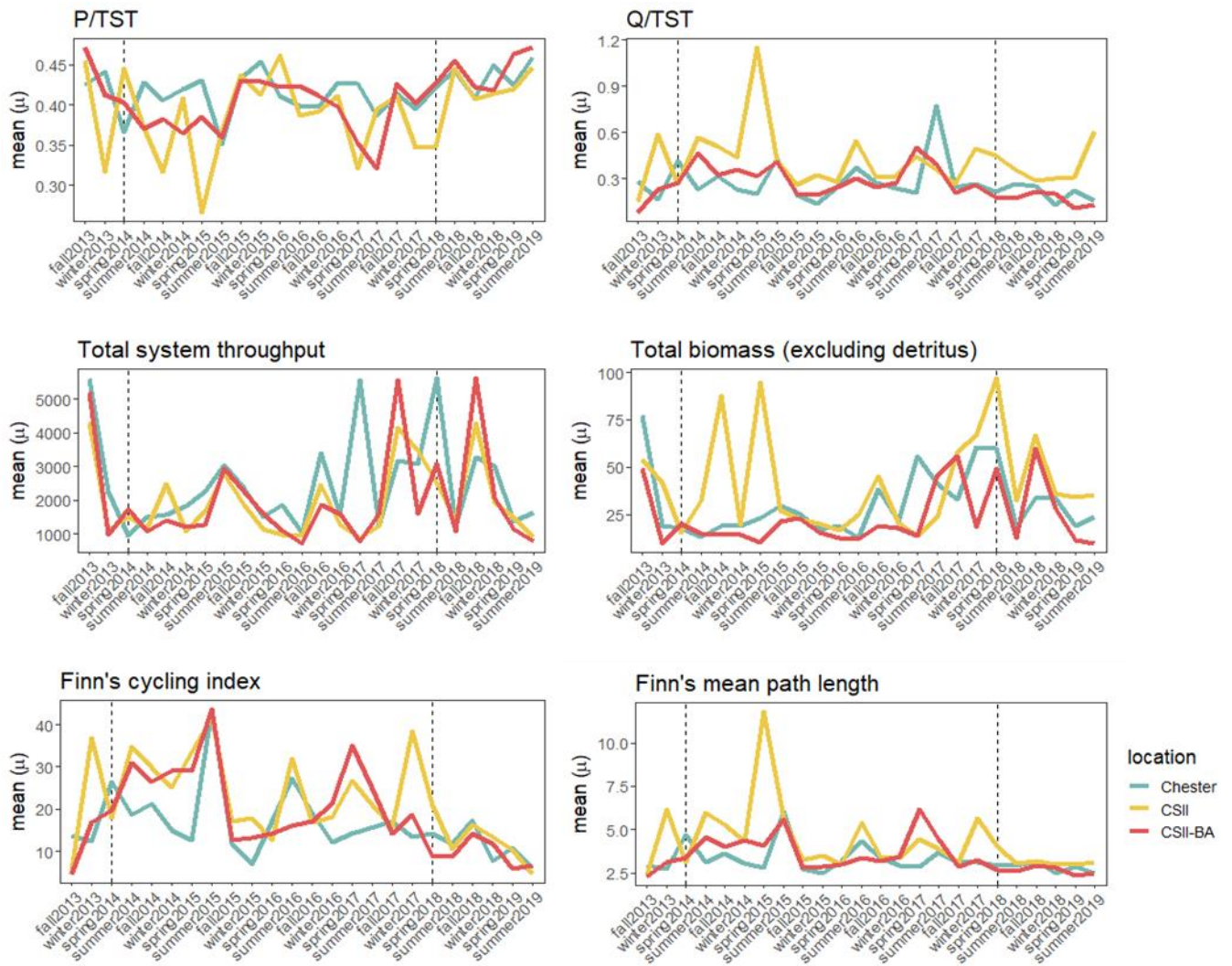


Figure 16-1. Production and consumption rate over total system throughput (P/TST and Q/TST), total system throughput (TST), total biomass (excluding detritus), Finn's Cycling Index and Finn's Mean Path Length of each season-year for the Chester, CSII and CSII-BA models.

Dashed lines indicate dredging events on CSII-BA in spring 2014 and 2018.

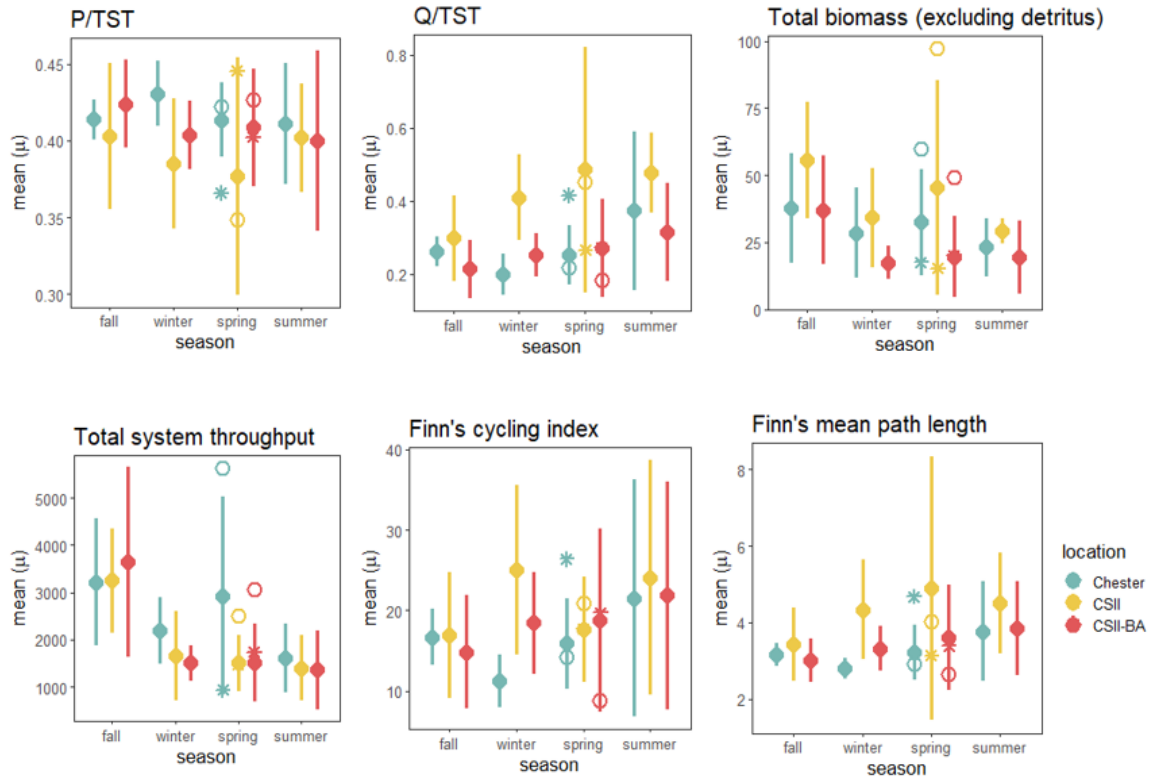


Figure 16-2. Seasonal mean (\pm 1SD) production and consumption rate over total system throughput (P/TST and Q/TST), total system throughput (TST), total biomass (excluding detritus), Finn's Cycling Index and Finn's Mean Path Length mean values and standard deviations of each season for the Chester, CSII, and CSII-BA models.

Values for spring 2014 (asterisks) and spring 2018 (open circles) when dredging events took place on CSII-BA are noted.

Table 16-1. Chester and Canaveral Shoals II (CSII and CSII-BA) functional groups and species composition of each functional group.

Functional Group No.	Functional Group	Species
1	Coastal Sharks (Small)	<i>Rhizoprionodon terraenovae</i> , <i>Sphyrna tiburo</i>
2	Skates & Rays	<i>Dasyatis americana</i> , <i>Dasyatis say</i> , <i>Gymnura micrura</i> , <i>Myliobatis freminvillei</i> , <i>Myliobatis goodei</i> , <i>Rhinoptera bonasus</i> , <i>Narcine bancroftii</i> , <i>Raja eglanteria</i> , <i>Rhinobatos lentiginosus</i>
3	Pelagic Carnivores (Medium)	<i>Carangidae</i> (juvenile), <i>Caranx crysos</i> , <i>Caranx hippos</i> , <i>Caranx latus</i> , <i>Caranx ruber</i> , <i>Caranx spp.</i> , <i>Echeneis naucrates</i> , <i>Pomatomus saltatrix</i> , <i>Scomberomorus maculatus</i> , <i>Sphyrna guachancho</i> , <i>Sphyrna spp.</i> , <i>Trichiurus lepturus</i>
4	Pelagic Carnivores (Small)	<i>Hypoatherina harringtonensis</i> , <i>Chloroscombrus chrysurus</i> , <i>Decapterus punctatus</i> , <i>Oligoplites saurus</i> , <i>Selene setapinnis</i> , <i>Selene vomer</i> , <i>Harengula jaguana</i> , <i>Opisthonema oglinum</i> , <i>Sardinella aurita</i> , <i>Hemiramphus brasiliensis</i> , <i>Mugil curema</i> , <i>Mugilidae</i> (juvenile), <i>Peprilus burti</i> , <i>Peprilus paru</i> , <i>Peprilus spp.</i> , <i>Peprilus triacanthus</i>
5	Pelagic Omnivores (Small)	<i>Brevoortia smithi</i> , <i>Brevoortia spp.</i> , <i>Brevoortia tyrannus</i> , <i>Clupeidae</i> (juvenile), <i>Clupeiformes</i> (juvenile)
6	Anchovies	<i>Anchoa hepsetus</i> , <i>Anchoa lyolepis</i> , <i>Anchoa spp.</i>
7	Atlantic Croaker	<i>Micropogonias undulatus</i>
8	Snappers	<i>Lutjanus campechanus</i> , <i>Lutjanus griseus</i> , <i>Lutjanus spp.</i> , <i>Lutjanus synagris</i>
9	SeatROUT & Weakfish	<i>Cynoscion nothus</i> , <i>Cynoscion regalis</i>
10	Other Drums & Croakers	<i>Bairdiella chrysoura</i> , <i>Larimus fasciatus</i> , <i>Leiostomus xanthurus</i> , <i>Menticirrhus americanus</i> , <i>Menticirrhus littoralis</i> , <i>Menticirrhus spp.</i> , <i>Pareques acuminatus</i> , <i>Pareques spp.</i> , <i>Sciaenidae</i> (juvenile), <i>Stellifer lanceolatus</i>
11	Sea Basses	<i>Centropristis ocyurus</i> , <i>Centropristis philadelphia</i> , <i>Centropristis spp.</i> , <i>Centropristis striata</i> , <i>Centropristis/Diplectrum spp.</i> , <i>Diplectrum bivittatum</i> , <i>Diplectrum formosum</i> , <i>Diplectrum spp.</i> , <i>Serraniculus pumilio</i> , <i>Serranus subligarius</i> , <i>Rypticus spp.</i>
12	Lizardfishes & Snakefishes	<i>Saurida brasiliensis</i> , <i>Synodontidae</i> (juvenile), <i>Synodus foetens</i> , <i>Synodus spp.</i> , <i>Trachinocephalus myops</i>
13	Searobins	<i>Prionotus carolinus</i> , <i>Prionotus evolans</i> , <i>Prionotus ophryas</i> , <i>Prionotus roseus</i> , <i>Prionotus rubio</i> , <i>Prionotus scitulus</i> , <i>Prionotus spp.</i> , <i>Prionotus tribulus</i>
14	Demersal Carnivores (Medium)	<i>Porichthys plectrodon</i> , <i>Ariopsis felis</i> , <i>Bagre marinus</i> , <i>Balistes capriscus</i> , <i>Trachinotus carolinus</i> , <i>Ariosoma balearicum</i> , <i>Dactylopterus volitans</i> , <i>Chilomycterus schoepfii</i> , <i>Chaetodipterus faber</i> , <i>Haemulon aurolineatum</i> , <i>Orthopristis chrysoptera</i> , <i>Halichoeres caudalis</i> , <i>Xyrichtys novacula</i> , <i>Aluterus heudelotii</i> , <i>Aluterus monoceros</i> , <i>Aluterus schoepfii</i> , <i>Ophichthus gomesii</i> , <i>Ophichthus puncticeps</i> , <i>Ophidion grayi</i> , <i>Ophidion holbrookii</i> , <i>Ophidion marginatum</i> , <i>Opistognathus robinsi</i> , <i>Acanthostracion quadricornis</i> , <i>Urophycis floridana</i> , <i>Urophycis regia</i> , <i>Scorpaena calcarata</i> , <i>Calamus arctifrons</i> , <i>Lagodon rhomboides</i> , <i>Sphoeroides spengleri</i>
15	Demersal Carnivores (Small)	<i>Antennarius spp.</i> , <i>Apogon spp.</i> , <i>Dactyloscopus crossotus</i> , <i>Dactyloscopus foraminosus</i> , <i>Dactyloscopus moorei</i> , <i>Dactyloscopus poeyi</i> , <i>Dactyloscopus spp.</i> , <i>Fistularia tabacaria</i> , <i>Eucinostomus</i>

Functional Group No.	Functional Group	Species
		<i>argenteus</i> , <i>Eucinostomus gula</i> , <i>Eucinostomus jonesii</i> , <i>Eucinostomus spp.</i> , <i>Gobiidae (unidentified)</i> , <i>Gobiosoma ginsburgi</i> , <i>Gobiosoma spp.</i> , <i>Microgobius carri</i> , <i>Monacanthidae (juvenile)</i> , <i>Stephanolepis hispidus</i> , <i>Stephanolepis setifer</i> , <i>Upeneus parvus</i> , <i>Halieutichthys aculeatus</i> , <i>Ogcocephalus cubifrons</i> , <i>Ogcocephalus parvus</i> , <i>Myrophis punctatus</i> , <i>Brotula barbata</i> , <i>Ophidiidae (juvenile)</i> , <i>Ostraciidae (juvenile)</i> , <i>Heteropriacanthus cruentatus</i> , <i>Pristigenys alta</i> , <i>Scorpaena spp.</i> , <i>Calamus spp.</i> , <i>Hippocampus erectus</i> , <i>Syngnathus floridae</i> , <i>Syngnathus fuscus</i> , <i>Syngnathus louisianae</i> , <i>Syngnathus pelagicus</i> , <i>Syngnathus scovelli</i> , <i>Syngnathus spp.</i> , <i>Syngnathus springeri</i> , <i>Sphoeroides spp.</i> , <i>Tetraodontidae (juvenile)</i> , <i>Astroscopus spp.</i>
16	Demersal Omnivores	<i>Acanthurus spp.</i> , <i>Blenniidae (juvenile)</i> , <i>Parablennius marmoreus</i>
17	Flatfishes	<i>Gymnachirus melas</i> , <i>Bothus ocellatus</i> , <i>Bothus robinsi</i> , <i>Bothus spp.</i> , <i>Symphurus civitatum</i> , <i>Symphurus plagiosa</i> , <i>Symphurus spp.</i> , <i>Ancylosetta ommata</i> , <i>Citharichthys macrops</i> , <i>Citharichthys spilopterus</i> , <i>Cyclosetta fimbriata</i> , <i>Etropus crossotus</i> , <i>Etropus cyclosquamus</i> , <i>Etropus microstomus</i> , <i>Etropus spp.</i> , <i>Paralichthyidae (juvenile)</i> , <i>Paralichthys albigutta</i> , <i>Paralichthys dentatus</i> , <i>Paralichthys lethostigma</i> , <i>Paralichthys spp.</i> , <i>Syacium guntieri</i> , <i>Syacium papillosum</i> , <i>Syacium spp.</i>
18	Squid [Carnivore]	<i>Loliginidae spp</i>
19	Large Crabs [omnivore]	
20	Large shrimp and stomatopods [omnivore]	<i>Penaeidae spp</i> , <i>Squillidae spp</i>
21	Pelagic Crustaceans	<i>Chlamydopleon dissimile</i> <i>Coifmaniella mexicana</i>
22	Swimming Crabs_Stomatopods [Carnivore]	<i>Polybiidae spp</i> , <i>Portunidae spp</i>
23	Small Crabs and Shrimp	<i>Diogenidae spp</i> , <i>Epialtidae spp</i> , <i>Paguridae spp</i> , <i>Pasiphaeidae spp</i> , <i>Pseudorhombilidae spp</i> , <i>Squillidae spp</i>
24	Amphipods and Others	<i>Ampeliscidae spp</i> , <i>Aoridae spp</i> , <i>Bateidae</i> <i>Bathyporeiidae spp</i> , <i>Colomastigidae spp</i> , <i>Haustoriidae spp</i> , <i>Ischyroceridae spp</i> , <i>Leucothoidae spp</i> , <i>Lilgeborgiidae spp</i> , <i>Gibberosus myersi</i> , <i>Melphidippidae spp</i> , <i>Metharpinia floridana</i> , <i>Eudevenopus honduranus</i> , <i>Diastylidae spp</i> , <i>Callianasidae spp</i> , <i>Ancinidae spp</i> , <i>Anthuridae spp</i> , <i>Chaetiliidae spp</i> , <i>Cirolanidae</i> <i>doteidae spp</i> , <i>Albuneidae spp</i> , <i>Emerita talpoida</i> , <i>Raninidae spp</i> , <i>Ostracoda spp</i> , <i>Pinnotheridae spp</i> , <i>Porcellanidae spp</i> , <i>Hippolytidae spp</i> , <i>Caprellidae spp</i> , <i>Apseudidae spp</i>
25	Bivalves and Brachiopods	<i>Bivalvia spp</i> , <i>Arcidae spp</i> , <i>Balanidae spp</i> , <i>Pectinidae spp</i> , <i>Lingulidae spp</i> , <i>Calyptraeidae spp</i> ,
26	Gastropods and Scaphopods	<i>Gastropoda spp</i> , <i>Nassariidae spp</i> , <i>Terebridae spp</i> , <i>Caecidae spp</i> , <i>Cerithiidae spp</i> , <i>Chaetopleuridae spp</i> , <i>Columbellidae spp</i> , <i>Olivellidae spp</i> , <i>Marginellidae spp</i> , <i>Tectonatica pusilla</i> , <i>Naticidae spp</i> , <i>Olividae spp</i> , <i>Pyramidellidae spp</i> , <i>Scaphopoda spp</i>
27	Echinoderms_Omnivores	<i>Ophiuroidea spp</i> , <i>Moira atropos</i> , <i>Rhynobrissus cuneus</i> , <i>Holothuroidea spp</i>

Functional Group No.	Functional Group	Species
28	Echinoderms_Sea Stars [carnivore]	<i>Luidiidae spp,</i>
29	Echinoderms_Urchins [herbivore]	<i>Arbacia punctulata spp, Lytechinus variegatus spp</i>
30	Polychaetes and Nemerteans	<i>Glyceridae spp, Amphinomidae spp, Platyhelminthes spp, Nemertea spp, Chrysopetalidae spp, Dorvilleidae spp, Eulepethidae spp, Goniadidae spp, Hesionidae spp, Lumbrineridae spp, Nephtyidae spp, Nereidae spp, Oeonidae spp, Onuphidae spp, Phyllodocidae spp, Pisionidae spp, Syllidae spp, Traviidae spp, Polynoidae spp, Aphroditidae spp,</i>
31	Suspension-Feeding Polychaetes	<i>Cistenides gouldii, Phoronis sp., Clitellata spp, Sipuncula spp, Acoetidae spp, Ampharetidae spp, Capitellidae spp, Chaetopteridae spp, Cirratulidae spp, Maldanidae spp, Opheliidae spp, Ophelinidae spp, Orbiniidae spp, Oweniidae spp, Paraonidae spp, Pilargidae spp, Polygordiidae spp, Questidae spp, Sabellidae spp, Sabellariidae spp, Spionidae spp, Terebellidae spp, Magelonidae spp</i>
32	Sessile Fauna and Bryozoans	<i>Bryozoa spp, Cupuladriidae spp, Selenariidae spp, Hydrozoa spp, Anthozoa spp, Anthozoa spp</i>
33	Lancelets	<i>Branchiostoma virginiae</i>
34	Fish Larvae	-
35	Zooplankton	-
36	Phytoplankton	-
37	Bacterioplankton (non_photosynthetic)	-
38	Surface_Sediment_Algae	-
39	Detritus	-

Table 16-2. Input production and consumption rate parameters (P/B and Q/B) and production consumption ratio estimate (P/Q) for the Chester and Canaveral Shoals II models (CSII and CSII-BA).

Functional Group	Functional Group Name	P/B (year ⁻¹)	Q/B (year ⁻¹)	P/Q (year ⁻¹)
1	Coastal Sharks (Small)	0.69	7.50	0.09
2	Skates & Rays	0.40	8.96	0.04
3	Pelagic Carnivores (Medium)	0.48	7.25	0.07
4	Pelagic Carnivores (Small)	1.05	8.79	0.12
5	Pelagic Omnivores (Small)	1.90	14.10	0.13
6	Anchovies	1.42	15.91	0.09
7	Atlantic Croaker	0.73	4.74	0.15
8	Snappers	0.38	4.04	0.09
9	Seatrout & Weakfish	0.83	5.61	0.15
10	Other Drums & Croakers	1.18	5.72	0.21
11	Sea Basses	0.52	4.67	0.11
12	Lizardfishes & Snakefishes	0.74	6.77	0.11
13	Searobins	0.86	10.11	0.09
14	Demersal Carnivores (Medium)	0.51	7.54	0.07
15	Demersal Carnivores (Small)	1.20	12.25	0.10
16	Demersal Omnivores	1.34	15.13	0.09
17	Flatfishes	1.57	10.39	0.15
18	Squid [Carnivore]	2.67	16.64	0.16
19	Large Crabs [omnivore]	1.69	9.35	0.18
20	Large shrimp and stomatopods [omnivore]	5.38	19.20	0.28
21	Pelagic Crustaceans	5.38	19.20	0.28
22	Swimming Crabs_Stomatopods [Carnivore]	1.34	7.43	0.18
23	Small Crabs and Shrimp	3.66	19.20	0.19
24	Amphipods and Others	9.00	36.50	0.25
25	Bivalves and Brachiopods	1.21	23.00	0.05
26	Gastropods and Scaphopods	3.00	12.00	0.25
27	Echinoderms_Omnivores	1.20	9.89	0.12
28	Echinoderms_Sea Stars [carnivore]	1.20	9.89	0.12
29	Echinoderms_Urchins [herbivore]	1.20	9.89	0.12
30	Polychaetes and Nemerteans	4.60	15.90	0.29
31	Suspension-Feeding Polychaetes	4.60	15.90	0.29
32	Sessile Fauna and Bryozoans	1.62	9.00	0.18
33	Lancelets	2.00	8.50	0.24
34	Fish Larvae	50.45	132.13	0.38
35	Zooplankton	15.05	43.00	0.35
36	Phytoplankton	55.57	-	-
37	Bacterioplankton (non_photosynthetic)	100.00	215.00	0.47

Table 16-3. Binary diet matrix used to develop proportional diet matrix based on seasonal and site-specific biomass of prey.

The numbers identify the functional groups of the model (Table 16-2). Columns represent predators and rows represent prey.

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33	34	35			
1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-		
2	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
3	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
4	1	1	1	-	-	-	-	-	1	-	-	1	-	-	-	-	1	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
5	1	1	1	-	-	-	-	-	1	-	1	1	-	-	-	-	1	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
6	1	1	1	1	-	-	-	1	1	1	1	1	-	-	1	-	1	1	-	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
7	1	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
8	1	1	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
9	1	1	-	-	-	-	-	-	1	-	1	1	-	-	-	-	1	-	-	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
10	1	1	-	-	-	-	-	-	1	-	1	1	-	-	-	-	1	-	-	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
11	1	1	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
12	1	1	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
13	1	1	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
14	1	1	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
15	1	1	-	-	-	1	-	1	-	-	1	-	1	-	1	-	1	-	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
16	1	1	-	-	-	-	-	1	-	-	1	-	1	-	1	-	1	-	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
17	1	1	-	-	-	-	-	-	-	-	-	1	1	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
18	1	1	1	1	-	-	1	1	1	1	1	1	-	1	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
19	1	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
20	1	1	1	-	-	-	-	-	1	1	1	1	1	1	1	-	1	-	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
21	-	1	1	1	-	-	1	1	1	1	1	1	1	1	1	1	1	1	-	1	-	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
22	1	1	-	-	-	-	1	-	-	1	1	-	-	1	-	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
23	-	1	-	-	-	-	1	1	1	1	1	1	1	1	1	1	1	1	1	1	-	1	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	
24	-	-	-	-	-	-	1	1	1	1	1	-	1	1	1	1	1	1	1	1	-	1	1	-	1	-	-	-	-	1	-	-	-	-	-	-	-	-
25	1	1	-	-	-	-	1	-	-	1	-	-	-	1	-	-	-	-	1	1	-	1	-	-	-	1	-	-	1	-	-	-	-	-	-	-	-	
26	-	1	-	-	-	-	1	-	-	1	-	-	-	1	-	-	1	-	1	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	
27	-	1	-	-	-	-	1	-	-	-	-	-	-	1	-	-	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
28	-	1	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
29	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
30	-	1	-	-	-	-	1	1	-	1	1	-	1	1	1	1	1	-	1	1	-	1	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
31	-	1	-	-	-	-	1	1	-	1	1	-	1	1	1	1	1	-	1	1	-	1	1	-	1	-	1	-	-	1	-	-	-	-	-	-	-	-
32	-	-	-	-	-	-	-	-	-	-	-	-	-	1	1	-	-	-	1	-	-	-	1	-	-	1	-	-	-	-	-	-	-	-	-	-	-	
33	-	-	-	-	-	-	1	-	-	1	-	-	1	1	1	-	-	-	1	1	-	1	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	-
34	-	-	1	1	-	-	1	1	1	1	1	1	1	1	1	1	1	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
35	-	-	-	1	1	1	-	-	-	1	-	-	-	-	1	1	-	1	-	1	1	-	1	1	-	-	-	-	-	-	1	-	1	1	1	1	1	
36	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	1	1	-	-	1	1	1	1	1	1	1	-	-	1	-	-	-	-	-	
37	-	-	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	1	-	1	1	1	1	-	1	-	-	1	1	1	1	1	1	1	
38	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	1	-	-	1	-	1	-	-	-	-	-	-	-	-	-	
39	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	1	-	-	1	1	-	1	-	1	1	-	-	-	-	-	-	-	1	
Import	-	-	-	-	-	-	-	-	-	-	-	0.8	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	0.8	-	-	-	-	-	-	-	

Table 16-4. Ecological indicators used in comparisons of Canaveral Shoals II (CSII and CSII-BA) and Chester Shoal models, based on Heymans et al. (2014) and Finn (1976).

Indicator	Acronym	Definition
Finn's Cycling Index	FCI	Fraction of the ecosystem's throughput that is recycled
Finn's Mean Path Length	MPL	Average number of groups that energy flows through
Primary Production/TST	P/TST	Primary production over the sum of all the flows through the ecosystem
Total Biomass (excluding first trophic level)	TBco	Total biomass of the community excluding detritus
Total Consumption/TST	Q/TST	Total consumption over the sum of all the flows through the ecosystem
Total Systems Throughput	TST	Sum of all flows in the model; considered to be an overall measure of "ecological size" of the ecosystem

17 Summary and Overview

Debra Murie

Key Points

- Sand shoals off the east coast of Cape Canaveral, Florida, are shallow, dynamic ridge-swale ecosystems. These environments are physically dominated by oceanographic processes and subject to frequent storm events.
- A Beyond-BACI sampling design provided a robust framework for assessing the impact of dredging on composition and abundance of plankton, invertebrates, and fish assemblages of the sand shoals, relative to their natural variability.
- Season was the primary driver of changes in abundance and biomass over time in all assemblages, with habitat (swale versus ridge), shoal, and year less so.
- With few exceptions, the impacts from dredging CSII-BA relative to reference shoals (CSII, Chester, and Bull Shoals) was not significant on the abundance and biomass of the various plankton, benthic and demersal invertebrates, and demersal fish assemblages.
- The few changes in abundance or density of invertebrate groups detected in the dredged shoal relative to the reference shoals were short lived (i.e., returned to pre-dredged levels within a season).
- The dynamic nature of the sand shoals and surrounding area was further evidenced by telemetry of acoustically tagged fishes, including flatfishes, rays, and a skate, which showed ephemeral use and low residency within specific shoal areas.
- The importance of a seasonal driving factor was also observed in the isotopic niches of representative fish and invertebrate species over time among the shoals, as indicated by the combined basal carbon resources assimilated and the trophic levels.
- At an ecosystem level, Ecopath models also showed the importance of a seasonal component in driving the trophic structure and productivity of the shoal ecosystems over time.

17.1 Introduction

In its entirety, this study spanned 6 years of sequential, seasonal sampling of the sand shoal ecosystems off the coast of Cape Canaveral, Florida. During that period of time, two dredging events occurred on CSII-BA, one in the winter–spring of 2013/14 and another in the spring of 2018 (see Table 1-1). To better understand the functioning of the shoals and assess the impact of dredging on their biological assemblages, sampling and analyses encompassed both the physical environment and biological components of the ecosystems. We used a Beyond-BACI sampling design to ensure that the natural variability in the shoal ecosystems, based on reference/control shoals monitored over time, could be used as a comparative measure of change (impact) observed due to the dredging events. Based on this extensive sampling, we conclude that the sand shoals off the east coast of Florida are dynamic ecosystems that are driven primarily by seasonal forces both in their physical and biological components. These environments are physically dominated by oceanographic processes and subject to frequent storm events

and resident biota have adapted to these conditions. With few exceptions, the natural variability of the biological assemblages of the shoals was similar or greater than any change observed during the dredging events. In addition, the very limited number of changes in species assemblages that occurred as a result of dredging were observed to recover, or return to a pre-dredged condition, within a single season, further supporting the dynamic functioning of the shoals.

The sections below provide the key points and observations from each chapter of the study, as summarized by the authors with specific details provided in each chapter referenced. Following the summary is a brief overview of dredging impacts and the concept of recovery applied to the sand shoals. The chapter ends with a brief discussion of lessons learned through this process and suggestions for future studies. A complete list of study products arising from this study is provided in Appendix H.

17.2 Physical Environment of Florida's East Coast Sand Shoals

17.2.1 Coastal Oceanography (Chapter 2)

Sand shoals off the east coast of Cape Canaveral, Florida, are shallow, dynamic ridge-swale ecosystems. These environments are physically dominated by oceanographic processes and subject to frequent storm events. Ridges and swales produce spatial variability in tidal and non-tidal (subinertial) flows over the inner continental shelf. Shoals with gentle slopes between ridges and swales, such as at Chester Shoal, showed Bernoulli-type hydrodynamics, i.e., flow enhancement over ridges, whereas shoals with steep slopes between ridges and swales, such as between CSII-BA (the dredged shoal) and Shoal E adjacent to it, displayed frictional hydrodynamics, i.e., flow enhancement over swales. The critical bed slope to switch from frictional to Bernoulli-type (inertia-dominated) dynamics was equal to the non-dimensional bottom drag coefficient (typically 0.0025 for sandy bottom). It was evident that any alteration to a swale or ridge can modify the hydrodynamics by influencing friction and inertia. Because erosion will develop where strongest flows appear, any dredging alteration will also shift the erosional processes accordingly.

Gulf Stream enhancements by southerly winds translate into strengthened northward inner-shelf flows, and vice versa. The dynamics across the shelf are mostly geostrophic², with occasional influence from wave stresses. Waves with periods between 20 and 200 s are modulated by semidiurnal tidal flows in the swales exposed to the ocean (seaward of a ridge), enhancing erosion. Shoals dissipate and transform waves with heights > 1 m, delaying erosion. Tropical storm winds produce currents that distort tides and enhance shelf erosion.

Comparisons of pre-dredging to after-dredging conditions at a moored acoustic doppler current profiler site suggested that changes can be grouped into three categories: 1) susceptibility for wave transformation caused by subinertial water-level variability; 2) infragravity (LGW) wave forcing by short waves (SGW); and 3) generation and dissipation of LGW over ridges. With respect to (1), susceptibility for wave transformations decreased after dredging. In reference to (2), there was decreased forcing of SGW after dredging and an increased proportion of free LGW. Related to (3), there was decreased generation and increased generation of LGW after dredging.

Therefore, any alterations to the seabed through dredging, either to ridge or to swale, will necessarily produce alterations to wave-related energy fluxes and the amount of erosional LGW energy that reaches the shore. These alterations will have effects on morphodynamic equilibrium and on erosional and depositional horizons.

² Relating to or denoting the component of a wind or current that arises from a balance between pressure gradients and Coriolis forces.

17.2.2 Bathymetry (Chapter 3)

Multibeam acoustic surveys of CSII-BA (the borrow area of Canaveral Shoals II) immediately following the first dredging event in winter 2013/14 to spring 2014 showed that the dredging activity was contained solely within the swale area of CSII-BA. This post-dredge survey in May 2014 showed clearly distinguishable draghead lines (furrows) from the trailing suction hopper dredge; furrows were less pronounced 1 year following dredging in surveys repeated in June 2015 (Figure 3-2). Of the surveyed area of CSII-BA, the overall average change in bathymetry was + 0.11 m in the year following the dredging event, showing some filling-in and smoothing of the dredged area of CSII-BA (Figure 3-7). Within the same time frame, Chester and Bull Shoals (the reference shoals) experienced an average change of -0.09 m, indicating very little change over the intervening year. Both CSII-BA and Chester Shoals had a slight but noticeable migration of their ridge crest to the southeast from May 2014 to June 2015, which is a natural occurring phenomenon with these dynamic sand shoals that has been reported previously (Olsen Associates 2013). These bed level changes indicate that these shallow shoal complexes are physically dominated.

17.2.3 Habitat Classification (Chapter 4)

The physical substrate of the shoals is primarily sand, shell sand, and sandy-mud with relatively low organic content, most likely a function of the high degree of water movement and resuspension of sand (see Chapter 2). Based on benthic grab cores, the majority of sediment grain size on CSII-BA (the dredged portion of Canaveral Shoals II), CSII (the non-dredged portion of Canaveral Shoals II as a reference/control shoal), and Chester and Bull Shoals (reference/control shoals) was 0.25–1.00 mm, and consisted of mostly fine, medium, and coarse sands, of both geologic and biogenic (i.e., crushed shell) origin. Canaveral Bight, which is shoreward of CSII and CSII-BA and southwest of Chester and Bull Shoals, had finer sediments mostly < 0.0625 mm that was comprised of muddy sand with ~2% organic content. Based on classifying the substrate using surface views of the benthic grabs, the majority of the substrate on the shoals was medium and coarse shell sand and medium shell hash of biogenic origin. These surface views of the substrate indicated a higher proportion of larger-sized biogenic shell hash, a product of extensive breaking and reshaping of mostly bivalve shells. Virtually none of the substrate types on CSII, CSII-BA, Chester, and Bull Shoals had vertical relief greater than a few centimeters, other than some complexity and vertical height provided by occasional patches of sand dollar beds and clumps of tube worms (e.g., *Diopatra*).

Overall, ridge sediments were significantly coarser and had lower organic content than swale sediments (see Chapter 11). Significant changes in sediment grain size and organic content at the dredged shoal (CSII-BA) were matched by similar changes at the reference shoals (CSII, Chester and Bull) at the same time, so could not be attributed to dredging.

17.2.4 Water Quality Characteristics and Chlorophyll *a* as a Proxy for Primary Productivity (Chapters 5, 6, and 8)

Water quality monitoring during the 6 years of the study (Chapter 6) determined that mean surface-water temperatures were greater than the bottom-water temperatures within each of the four shoals (CSII-BA, CSII, Chester and Bull Shoals) but did not differ among the four shoals over the study period and followed the same general temporal pattern; mean bottom-water temperature was higher at CSII compared to Bull Shoal, with CSII-BA and Chester Shoals overlapping with both of those shoals. Overall mean salinity, dissolved oxygen concentration, and pH for surface and bottom waters were not different within each shoal, nor were they different among the four shoals over the study period. Overall mean Secchi disk depths (i.e., light attenuation) did not differ among the four shoals over the study period. Mean turbidity levels over the study period were generally lower in the surface- than bottom-water samples for all shoals. Post-dredge seasons (i.e., spring and summer of 2014 and 2018) showed no major

differences among shoals, including the dredged shoal, CSII-BA, in the trends of mean surface and bottom turbidities, mean chlorophyll *a* in surface and bottom waters, and mean total phosphorus and total nitrogen in surface and bottom waters.

Based on satellite imagery (Chapter 5), chlorophyll *a* concentration, as a proxy for surface phytoplankton productivity, varied widely on CSII-BA, CSII, Chester, and Bull Shoals, and in the surrounding area, during pre-dredge and post-dredge periods of the first dredging event in winter 2013/14 to spring 2014. This variability encompassed a range of higher chlorophyll *a* levels than those observed during active dredging. Dredging activities on CSII-BA did not result in an increase in primary productivity in the surface waters of CSII-BA or any of the study shoals overall.

For microphytobenthos (Chapter 8), the range of chlorophyll *a* levels observed (i.e., 1–202 mg m⁻²) were in line with the range of values observed in other shallow ecosystems on the east and west coasts of Florida. Spatial and temporal differences in microphytobenthos chlorophyll *a* concentrations were observed in the study. The highest mean chlorophyll *a* concentrations were observed at CSII Shoal, followed by Bull Shoal, and the lowest concentrations were at Chester and CSII-BA Shoals. Seasonally, chlorophyll *a* concentrations generally peaked in the summer. A comparison of microphytobenthos chlorophyll *a* concentrations in the pre- to post-dredge sediment samples showed no major differences in concentrations, and patterns observed for the post-dredge period were generally similar at all the shoals.

17.3 Species Abundance, Biomass, and Assemblages of Florida's East Coast Sand Shoals

17.3.1 Phytoplankton (Chapter 7)

For phytoplankton (Chapter 7), there were no consistent significant differences in total mean phytoplankton biomass among shoals for either surface or bottom water. Diatoms, dinoflagellates, and cyanobacteria were regular major contributors to total phytoplankton biomass throughout the study. Dinoflagellates generally had higher mean biomass in surface-water than bottom-water samples, in part reflecting their ability to move up in the water column via flagellar motility. By contrast, diatoms generally had higher mean biomass in bottom-water than surface-water samples, in part reflecting a combination of sinking of cells in the water column and resuspension of sedimented cells from the benthos into the lower layers of the water column. Small-sized phytoplankton was found to be important in terms of both abundance and biomass as picoplanktonic cyanobacteria combined with nanoplanktonic eukaryotes often represented over 50% of total phytoplankton biomass. This observation highlights the importance of the microbial loop in the Cape Canaveral shelf.

Seasonally, the highest mean total phytoplankton biomass levels over the study period were observed in the fall, followed by winter, and lowest levels were observed in the spring and summer. Seasonal differences were in part attributable to shifts in predominant seasonal wind directions, which drive water along the coast from the north in the fall and winter, but from the south in the spring and summer, including eddies and upwelling from the Gulf Stream.

In terms of the comparison of phytoplankton in post-dredge periods (i.e., spring and summer of 2014 and 2018), and similar seasons in other years, no reproducible differences were observed at any of the shoals. These observations suggest that any impacts of dredging on phytoplankton composition and biomass were comparatively short lived (i.e., not extending beyond one season).

17.3.2 Zooplankton and Meroplankton (Chapters 9 and 10)

For zooplankton (Chapter 9), the two most important groups in terms of biomass (mg carbon L⁻¹) throughout the study period and shoals were arthropods and protozoans. Similar to phytoplankton, the

high biomass levels of small-sized ciliates highlight the important role the microbial loop plays in food webs of the Cape Canaveral shelf. In terms of the influence of dredging activity on the zooplankton community, no major differences were observed in post-dredge seasons (i.e., spring and summer of 2014 and 2018) compared to similar seasons in other years, or in trends in biomass or composition among the four shoals, suggesting that any impacts of dredging were relatively short lived.

Meroplankton (Chapter 10) over the sand shoals off the east coast of Florida was dominated by bivalve and polychaete larvae. Season had the strongest impact, affecting all functional groups of meroplankton except molluscan larvae, with the greatest abundance in summer and the lowest in fall. In contrast, abundance was not different for most meroplankton groups between ridge and swale habitat, with the exception of molluscan larvae (mostly *Bivalvia*) that was significantly greater over ridges than swales. Overall meroplankton abundance, however, did not differ among any of the shoals, dredged or not. Echinoderm and polychaete larvae showed limited shoal effects, but were not different between the dredged shoal (CSII-BA) and the non-dredged shoal (CSII). Lack of any dredging impacts on meroplankton were consistent with prior research on zooplankton overall, showing that effects of sand removal were transient (Sullivan and Hancock 1977).

17.3.3 Benthic and Epifaunal Invertebrates Assessed With Benthic Grabs (Chapter 11)

Shoals, habitat (ridge versus swale), and seasons all significantly affected abundance of benthic invertebrates, including at dredged (CSII-BA) and non-dredged (CSII) portions of Canaveral Shoal II, Chester Shoal, and Bull Shoal. However, most of these differences were unrelated to dredging events. No general biological factors, including invertebrate abundance, biomass, species richness, or Simpson's Index of Diversity changed following either of the two dredging events at CSII-BA or at any of the reference shoals. The only taxon for which abundance clearly changed at CSII-BA following dredging were amphipods (small crustaceans) in the Family Haustoriidae, which increased in abundance following the second dredging event. This specific change was not observed at CSII, Chester, or Bull Shoals, nor were any changes observed during the first dredging event at any of the shoals. The most abundant taxa, such as amphipods, sand dollars, lancelets, and even colonial bryozoans, were motile, and thus able to quickly recolonize any disturbed area. Benthic invertebrates were expected to be the most directly impacted biota by the dredging events but, with limited exceptions, they were either not impacted or recovered from dredging faster than seasonal sampling could detect an effect.

17.3.4 Demersal and Epifaunal Invertebrates Assessed with Bottom Trawling (Chapter 12)

For demersal invertebrates, there were no clear effects of dredging on the diversity or mean biomass metrics used to assess demersal invertebrate communities on CSII-BA, CSII, Chester, and Bull Shoals. Bull Shoal had higher community diversity and mean biomass than Chester and CSII-All (the entirety of Canaveral Shoal II) but this pattern was not pronounced with respect to the dredged portion (CSII-BA). Diversity and biomass of demersal invertebrates varied from year to year for all shoals. Seasonal patterns in the community diversity and mean biomass were evident but not consistent. The most consistent pattern was higher diversity on the ridges and higher biomass in the swales, but this was also seasonal. Summer generally had the highest diversity of all seasons. The abundance of commercially important invertebrate species collected from trawls was low and did not include any high-value species.

The assessment of dredging on demersal invertebrates was consistent with similar studies on the effects of dredging on soft-bottom communities in high energy, subtropical environments that favor diverse communities of small, mobile, opportunistic species (Whittaker et al. 2001; Kotta et al. 2009). These characteristics may ameliorate any effect of dredging through a continuous and diverse supply of recruits available to colonize the recently disturbed habitat.

17.3.5 Demersal Fishes Assessed with Bottom Trawling and Acoustic Telemetry (Chapters 13 and 14)

Sand shoals off the east coast of Florida have diverse fish assemblages that are dynamic in all aspects of space and time. Fish species richness varied among shoals: it was highest on Bull Shoal (150 species), followed by Chester Shoal (134 species), and lastly CSII-BA and CSII (118 and 117 species, respectively). Both the Shannon Diversity Index and Simpson's Index of Diversity were moderate for all seasons except most winters, which was associated with low diversity but high abundance of juvenile sciaenids (drums and croakers). Swale habitat on the shoals had significantly greater diversity, abundance, and biomass compared to ridge habitat. In addition, fish assemblages were significantly different between swale and ridge habitats of the shoals. Temporally, fish abundance, biomass, and assemblages varied significantly on an annual (year to year), seasonal, and diel basis, with seasonal and diel factors contributing more to the temporal differences than year. Seasonal changes in abundance and biomass were highly significant, with fall having the greatest abundance and biomass, followed by winter, summer, and spring. While the fish assemblages were dynamic from season to season, there was a core assemblage of many fish species that clearly associated with all sand shoals or shoal complexes year-round and were not transient. Based on abundance, smaller fish species, such as juvenile sciaenids, anchovy species, juvenile cusk-eels, and Leopard Searobin were common among shoals. Larger fishes were more common on a biomass basis, including Atlantic Croaker, Banded Drum, Silver Seatrout and Leopard Searobin. Diel patterns were also observed, with abundance and biomass of fishes 2.5X and 2.8X higher, respectively, at night compared to day, and fish assemblages were also significantly different between day and night. Spatially, all the shoals had fish assemblages that were relatively different in multidimensional space based on both abundance and biomass (standardized on a per area basis). This indicated that the shoals all had some individualistic features and natural variation that could make it difficult to generalize to other shoals in the area that were not sampled.

For two dredging events (winter 2013/14 to spring 2014 and then again in spring 2018), there was no significant impact on the fish assemblages of CSII-BA (the dredged shoal), relative to the natural variability observed in the fish assemblages of the reference shoals (CSII, Chester and Bull Shoals).

Movements and residency of demersal fishes were also assessed through acoustically tagged fishes and an extensive receiver array off the east coast of Florida (Chapter 14). Rays, in particular, had focused detections in the southern portion of the receiver array and specifically at the entrance to Port Canaveral. However, these demersal fishes, including flatfishes, rays, and a skate, showed low residency over the sand shoals; on average, fish were detected over the sand shoals < 3% of the time that they were at large. This finding was also consistent with a concomitant telemetry study by Iafrate et al. (2019, 2022), which also determined that many species of sharks use the shoal area on a seasonal basis during their north-south migrations. In addition, they found high mobility and low residency rates in other teleosts, such as Redfish and Atlantic Croaker, with most fish spending less than an hour at the same location. These fishes had low residency rates and were considered to be ephemeral on the shoals.

Detections for a Summer Flounder, a Smooth Butterfly Ray, a Bluntnose Stingray, and a Clearnose Skate Ray all indicated that these fish were within the area of the dredged shoal (CSII-BA) either before or during dredging, and then again after dredging activities, indicating that they were not directly affected by dredging (specifically the second dredging event in spring 2018). Over a similar timeframe, Iafrate et al. (2019, 2022) also found that telemetered sharks and teleosts used a dredged shoal (CSII-BA) similarly to a nearby control shoal (Chester Shoal).

17.4 Trophic Pathways and Ecosystems (Chapters 15 and 16)

Stable isotopes of $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ were used to trace potential trophic changes in basal carbon resources and trophic level, respectively, for focal invertebrate and fish species from the dredged shoal (CSII-BA) compared to the non-dredged reference/control shoals (CSII, Bull, and Chester Shoals) (Chapter 15). Overall, seasonal isotopic composition and isotope niche sizes of amphipods, Roughneck Shrimp, Aviu Shrimp, Leopard Searobin, Spotted Whiff, and Atlantic Croaker varied among the reference shoals as much or more than between the reference shoals and the dredged shoal.

For all focal species except Aviu Shrimp, there were no differences in their trophic position based on their $\delta^{13}\text{C}$ or $\delta^{15}\text{N}$ isotope values in fall 2013 (i.e., immediately prior to the first dredging event in winter 2013/14 and spring 2014) compared to post-dredge fall seasons (fall 2014 and fall 2015) for CSII-BA, CSII, or Bull Shoals. Trophic positions of Aviu Shrimp changed between pre- and post-dredge fall periods associated with the first dredging event, but they did so simultaneously on all three shoals and therefore could not be ascribed to the dredging event itself. Isotope values of $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ for all focal species for all shoals were not different between the pre- and post-dredge seasons for the second dredging event in spring 2018, with few exceptions that were related to the natural variability among the non-dredged shoals rather than the dredging event. Overall, the stability in the trophic positions of the focal invertebrate and fish species during pre- versus post-dredge seasons, and within the dredged shoal relative to the reference shoals, indicated that dredging did not fundamentally change the trophic state and linkages of the food web of the shoals, as observed through focal species that spanned different trophic levels and feeding lifestyles.

At the ecosystem level (Chapter 16), data across multiple trophic levels were integrated into a modeling framework for a series of ecosystem models to evaluate the impacts of dredging activities in the borrow area of Canaveral Shoals II (CSII-BA) and reference (control) shoals (CSII and Chester Shoals). Ecological indicators were compared across shoals and seasons to determine whether dredging has impacted ecosystem structure and function. According to these modeled ecological indicators, there were no clear effects of dredging on the CSII-BA ecosystem based on seasonal sampling over a study period of 6 years. Most ecological indicators showed greater similarities between CSII-BA (dredged shoal) and Chester (reference shoal) over time than between CSII-BA and CSII, which indicated no impact on CSII-BA due to dredging events. Regarding ecosystem flows, in spring 2014 when CSII-BA was dredged it had a production (P) flow rate in proportion to TST (P/TST) and a consumption (Q) flow rate in proportion to TST (Q/TST) that was similar to CSII, which indicated that the rates were similar between dredged and non-dredged shoals. Relative to CSII, both CSII-BA and Chester Shoals had increases in P/TST and decreases in Q/TST during spring 2018 when CSII-BA was dredged. This suggested a potential change in the community structure and composition, but changes occurred on both shoals and therefore were unrelated to any direct dredging impact. Similarly, total biomass (excluding detritus) among the shoals in spring 2014 also indicated a lack of dredging impact on CSII-BA, and CSII-BA and Chester Shoals remained similar but relatively low compared to CSII. Although CSII-BA total biomass was lower in spring 2018 (second dredging event on CSII-BA) compared to both Chester and CSII, it was not significantly lower and had fully recovered by summer 2018, indicating that any impact from dredging was very short lived. Overall, the ecosystem model analysis suggested that the ecological indicators of the dredged shoal (CSII-BA) were mostly similar to the reference shoal (Chester), and whenever any differences were noted they were recovered with one season following the dredging event. After two dredging events at CSII-BA, the scale and nature of any differences in the dredged ecosystem was comparable to the control shoal, suggesting that this difference or change may also be explained by the dynamic variability in the system.

Ecological indicators did not suggest that there were any notable impacts from dredging at an ecosystem level, and the estimated oscillation of ecological indicators may be due to the inner variability in the

system at a seasonal level (Brooks et al. 2006). This was similar to a seasonal effect that was observed in invertebrate and fish assemblages (Chapters 11–13). Since high variability of ecological indicators were also exhibited at the reference shoal (Chester), indicators may suggest that dredging impact is not outside the bounds of natural variation in these systems. In addition, estimated values of ecological indicators were in the range of values of other ecosystems (Heymans et al. 2014).

17.5 Overall Dredging Impacts and Recovery

Recovery of perturbed biological communities and environments is usually indicated by the return of species assemblages (or specific organisms) to their former state prior to the perturbation (Michel et al. 2013). This was the primary definition used throughout the current study, where impacts and recovery were assessed using changes to species abundance, biomass, and assemblages. However, with very few exceptions, changes in the composition and abundance of any of the biological groups (zooplankton, meroplankton, benthic invertebrates, and demersal invertebrates and fishes) due to the impacts of dredging CSII-BA, twice, were not significant relative to the natural variability of these assemblages observed on the non-dredged shoals, including CSII, Chester and Bull Shoals. When there was any significant measurable difference, for example the increase in haustoriid amphipods in CSII-BA following dredging, the change returned to levels observed previous to the dredging within a single season. Besides the assemblages, isotopic niches, and ecosystem models all provide consistent results indicating that there was no to little impact from the two dredging events. Telemetered fish, such as Summer Flounder and Smooth Butterfly Rays, were also detected in the area of the dredged shoal (CSII-BA) before, during, and after the dredging event, showing no significant measurable impact on their movement and residency due to dredging. Physical properties of the dredged shoal (CSII-BA) also recovered to its pre-dredge condition within a year of the dredging event (i.e., depth change and draghead furrows reduced). Although not directly assessed using the BACI design, the occurrence of major storms and hurricanes were also considered as a source of change in abundance and biomass of the biological groups. Qualitative comparisons among the shoals during major storm events did not show a noticeable change that could be ascribed to the storm event directly, given the degree of natural variability that occurred during non-storm periods of time.

In summary, sand shoals off the east coast of Cape Canaveral, Florida, are dynamic ridge-swale ecosystems dominated by seasonal variation. Their position in relatively shallow, exposed waters is a contributing factor to their natural variability in both their physical and biological attributes. This natural variability in the biological assemblages among the shoals was as great or greater than any detectable changes due to dredging.

17.6 Lessons Learned and Future Suggestions

17.6.1 Use of Beyond-BACI Sampling Design

Using a Beyond-BACI sampling design in the current study, where there were multiple reference/control shoals and multiple sampling seasons before and after the dredging events, was time intensive and relatively expensive but proved to be invaluable in teasing out the effects due to natural variability versus effects due to dredging. Based on the high degree of variability in the biological communities of the sand shoals as a factor of season, day versus night periods, ridge versus swale habitat, year, and shoal itself, a simple BACI design would most likely not have provided a robust enough sampling design to assess the impact of dredging on the biological communities. At the very least, if only one control and one impacted shoal was available for the assessment, it would be imperative to sample them at least a full year before and after the dredging activity to account for the seasonal factor that was important as the major driving force in understanding the natural variability versus changes due to direct impacts. In addition, each shoal,

while similar, has individualistic properties that add to their natural variability and this was only possible to detect using multiple reference or control shoals.

17.6.2 A Call for Novel Approaches to Habitat Classification for Larval and Juvenile Fishes

The larval and juvenile fishes and demersal invertebrates that live on or close to the bottom were important components of the biological communities of the sand shoals. Classifying the bottom substrate in a manner that was relevant to them was difficult. The benthic grabs used for the benthic invertebrates were appropriate because the presence and abundance of many of those invertebrates rely on both the type of substrate (e.g., mud versus sand) and the grain size of the substrate. However, for demersal invertebrates and fishes, the substrate type is still important but the grain size of that substrate may be irrelevant other than its association with invertebrates that provide a prey base for them. We initially attempted to classify the bottom habitat of the sand shoals on a large scale using both a towed sled fitted with low-light cameras and a large metal drop-quadrat fitted with cameras. Both of these methods that have been used in other studies failed on the sand shoals because of the high turbidity caused primarily by sand suspended in the water column by tides, currents, and large waves (see Chapter 2). We therefore opted instead to collect over 200 benthic grabs over the area and classified the habitat both using conventional Coastal and Marine Ecological Classification Standard classification (FGDC 2012) as well as by viewing the surface of the grabs directly. Using the proportion of the surface area of the grabs to classify the habitat was useful in providing more information on the type and extent of substrate (e.g., 90% shell hash versus 10% sand) relevant to small invertebrates and larval and juvenile fishes that do not burrow in the substrate but use it as complex habitat for foraging and protection from predation. Deploying, collecting, and processing large benthic grabs is extremely time consuming both on the vessel and in the laboratory. With new methods arising with new technological advances, it would be invaluable if a method could be developed to assess benthic habitats over the sand shoals using a more automated methodology, both in collection of the data and processing. One promising technology may be Dual-Frequency Identification Sonar (DIDSON), which can provide detailed imaging in turbid waters. This sonar has been used primarily for fish surveys in turbid areas, for example estuaries (e.g., Able et al. 2014), but also for fish surveys that associate fish with specific habitats (e.g., Grabowski et al. 2012).

17.6.3 Quantitative Diet Data for Ecosystem Modeling: A Big Ask

For the ecosystem modeling, it was recognized that there was a considerable amount of uncertainty in the diet data because quantitative dietary information was not available for all of the species in all of the trophic groups. Obtaining diet data for ecosystem-level studies is always a daunting requirement, especially so in the sand shoals off the east coast of Florida because of the diverse number of both invertebrates and fishes encountered. In addition, many of the species were of various life stages, e.g., juveniles versus adults, which in many cases undergo ontogenetic shifts in their diets. The seasonal component of the assemblages on the sand shoals also dictates that both the availability of the species and their size may change over the seasons, varying based on recruitment. All of these factors increase the sampling requirements necessary to obtain robust estimates of diet composition and food consumption estimates.

17.7 References

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Appendix G. List of All Teleost and Elasmobranch Fish Species Sampled from All Shoals During Day and Night Trawling

Teleosts

Family_Superorder	Common Name	Scientific Name
Acanthuridae	Surgeonfish (Acanthurus spp. Juvenile)	<i>Acanthurus</i> spp.
Achiridae	Naked Sole	<i>Gymnachirus melas</i>
Actinopterygii	Unidentified fish (larvae/juvenile)	UNID fish (larvae/juvenile)
Anguilliformes	Anguilliformes (leptocephalus)	Anguilliformes (leptocephalus)
Antennariidae	Frogfish (Antennarius spp. juvenile)	<i>Antennarius</i> spp.
Apogonidae	Cardinalfish (Apogon spp. juvenile)	<i>Apogon</i> spp.
Ariidae	Hardhead Catfish	<i>Ariopsis felis</i>
Atherinidae	Reef Silverside	<i>Hypoatherina harringtonensis</i>
Balistidae	Gray Triggerfish	<i>Balistes capriscus</i>
Batrachoididae	Atlantic Midshipman	<i>Porichthys plectrodon</i>
Blenniidae	Blenny (Blenniidae juvenile)	Blenniidae (juvenile)
"	Seaweed Blenny	<i>Parablennius marmoratus</i>
Bothidae	Eyed Flounder	<i>Bothus ocellatus</i>
"	Flatfish (Bothidae larvae)	Bothidae (larvae)
"	Flatfish (Bothus spp.)	<i>Bothus</i> spp.
"	Twospot Flounder	<i>Bothus robinsi</i>
Carangidae	Atlantic Bumper	<i>Chloroscombrus chrysurus</i>
"	Atlantic Moonfish	<i>Selene setapinnis</i>
"	Bar Jack	<i>Caranx ruber</i>
"	Blue Runner	<i>Caranx crysos</i>
"	Carangidae (juvenile)	Carangidae (juvenile)
"	Jack (Caranx spp. juvenile)	<i>Caranx</i> spp.
"	Lookdown	<i>Selene vomer</i>
"	Round Scad	<i>Decapterus punctatus</i>
Clupeidae	Atlantic Menhaden	<i>Brevoortia tyrannus</i>
"	Atlantic Thread Herring	<i>Opisthonema oglinum</i>
"	Clupeidae (juvenile)	Clupeidae (juvenile)
"	Menhaden (Brevoortia spp. juvenile)	<i>Brevoortia</i> spp.
"	Scaled Sardine	<i>Harengula jaguana</i>
"	Spanish Sardine	<i>Sardinella aurita</i>
"	Yellowfin Menhaden	<i>Brevoortia smithi</i>
Clupeiformes	Clupeiformes (juvenile)	Clupeiformes (juvenile)
Congridae	Bandtooth Conger	<i>Ariosoma balearicum</i>
Cynoglossidae	Blackcheek Tonguefish	<i>Symphurus plagiusa</i>
"	Offshore Tonguefish	<i>Symphurus civitatum</i>

Family_Superorder	Common Name	Scientific Name
"	Tonguefish (Symphurus spp.)	<i>Symphurus</i> spp.
Dactylopteridae	Flying Gurnard	<i>Dactylopterus volitans</i>
Dactyloscopidae	Bigeye Stargazer	<i>Dactyloscopus crossotus</i>
"	Reticulate Stargazer	<i>Dactyloscopus foraminosus</i>
"	Sand Stargazer (Dactyloscopus spp.)	<i>Dactyloscopus</i> spp.
"	Shortchin Stargazer	<i>Dactyloscopus poeyi</i>
"	Speckled Stargazer	<i>Dactyloscopus moorei</i>
Diodontidae	Striped Burrfish	<i>Chilomycterus schoepfii</i>
Elopidae	Ladyfish/Malacho (leptocephalus)	<i>Elops</i> spp.
Elopomorpha (leptocephalus)	Elopomorpha (leptocephalus)	Elopomorpha (leptocephalus)
Engraulidae	Anchovy (Anchoa spp.)	<i>Anchoa</i> spp.
"	Dusky Anchovy	<i>Anchoa lyolepis</i>
"	Striped Anchovy	<i>Anchoa hepsetus</i>
Ephippidae	Atlantic Spadefish	<i>Chaetodipterus faber</i>
Fistulariidae	Bluespotted Cornetfish (juvenile)	<i>Fistularia tabacaria</i>
Gerreidae	Mojarra (Eucinostomus spp.)	<i>Eucinostomus</i> spp.
"	Silver Jenny	<i>Eucinostomus gula</i>
"	Slender Mojarra	<i>Eucinostomus jonesii</i>
"	Spotfin Mojarra	<i>Eucinostomus argenteus</i>
Gobiidae	Goby (Gobiidae unidentified)	Gobiidae (unidentified)
"	Goby (Gobiosoma spp.)	<i>Gobiosoma</i> spp.
"	Rockcut Goby	<i>Gobiosoma grosvenori</i>
"	Seaboard Goby	<i>Gobiosoma ginsburgi</i>
"	Seminole Goby	<i>Microgobius carri</i>
Haemulidae	Grunt (Haemulon spp. juvenile)	<i>Haemulon</i> spp.
"	Pigfish	<i>Orthopristis chrysoptera</i>
"	Tomtate	<i>Haemulon aurolineatum</i>
Hemiramphidae	Ballyhoo	<i>Hemiramphus brasiliensis</i>
Labridae	Pearly Razorfish	<i>Xyrichtys novacula</i>
Lutjanidae	Gray Snapper	<i>Lutjanus griseus</i>
"	Lane Snapper	<i>Lutjanus synagris</i>
"	Red Snapper	<i>Lutjanus campechanus</i>
"	Snapper (Lutjanus spp. juvenile)	<i>Lutjanus</i> spp.
Monacanthidae	Dotterel Filefish	<i>Aluterus heudelotii</i>
"	Filefish (Monacanthidae juvenile)	Monacanthidae (juvenile)
"	Orange Filefish	<i>Aluterus schoepfii</i>
"	Planehead Filefish	<i>Stephanolepis hispidus</i>
"	Pygmy Filefish	<i>Stephanolepis setifer</i>
"	Unicorn Filefish	<i>Aluterus monoceros</i>
Mugilidae	Mullet (Mugilidae juvenile)	Mugilidae (juvenile)
"	White Mullet	<i>Mugil curema</i>
Mullidae	Dwarf Goatfish	<i>Upeneus parvus</i>

Family_Superorder	Common Name	Scientific Name
"	Goatfish (Mullidae juvenile)	Mullidae (juvenile)
Ogcocephalidae	Roughback Batfish	<i>Ogcocephalus parvus</i>
"	Pancake Batfish	<i>Halieutichthys aculeatus</i>
Ophichthidae	Palespotted Eel	<i>Ophichthus puncticeps</i>
"	Shrimp Eel	<i>Ophichthus gomesii</i>
"	Snake Eel/Worm Eel (leptocephalus)	Ophichthidae (leptocephalus)
"	Speckled Worm Eel	<i>Myrophis punctatus</i>
Ophidiidae	Bank Cusk-eel	<i>Ophidion holbrookii</i>
"	Bearded Brotula (juvenile)	<i>Brotula barbata</i>
"	Blotched Cusk-eel	<i>Ophidion grayi</i>
"	Cusk-eel (juvenile)	Ophidiidae (juvenile)
"	Striped Cusk-eel	<i>Ophidion marginatum</i>
Opistognathidae	Spotfin Jawfish	<i>Opistognathus robinsi</i>
Ostraciidae	Boxfish (Ostraciidae juvenile)	Ostraciidae (juvenile)
"	Scrawled Cowfish	<i>Acanthostracion quadricornis</i>
Paralichthyidae	Bay Whiff	<i>Citharichthys spilopterus</i>
"	Dusky Flounder	<i>Syacium papillosum</i>
"	Flatfish (Etropus spp.)	<i>Etropus</i> spp.
"	Flatfish (Paralichthyidae juvenile)	Paralichthyidae (juvenile)
"	Flatfish (Paralichthys spp.)	<i>Paralichthys</i> spp.
"	Flatfish (Syacium spp.)	<i>Syacium</i> spp.
"	Fringed Flounder	<i>Etropus crossotus</i>
"	Gulf Flounder	<i>Paralichthys albigutta</i>
"	Ocellated Flounder	<i>Ancylopsetta ommata</i>
"	Shelf Flounder	<i>Etropus cyclosquamus</i>
"	Shoal Flounder	<i>Syacium guntieri</i>
"	Smallmouth Flounder	<i>Etropus microstomus</i>
"	Southern Flounder	<i>Paralichthys lethostigma</i>
"	Spotfin Flounder	<i>Cyclopsetta fimbriata</i>
"	Spotted Whiff	<i>Citharichthys macrops</i>
"	Summer Flounder	<i>Paralichthys dentatus</i>
Phycidae	Southern Hake	<i>Urophycis floridana</i>
"	Spotted Hake	<i>Urophycis regia</i>
Pleuronectiformes	Flatfish (Pleuronectiformes larvae)	Pleuronectiformes (larvae)
Pomatomidae	Bluefish	<i>Pomatomus saltatrix</i>
Priacanthidae	Short Bigeye (juvenile)	<i>Pristigenys alta</i>
Sciaenidae	Atlantic Croaker	<i>Micropogonias undulatus</i>
"	Banded Drum	<i>Larimus fasciatus</i>
"	Cubbyu	<i>Pareques acuminatus</i>
"	Cubbyu/High-hat (juvenile)	<i>Pareques</i> spp.
"	Gulf Kingfish	<i>Menticirrhus littoralis</i>

Family_Superorder	Common Name	Scientific Name
"	Kingfish (Menticirrhus spp. juvenile/unidentified)	<i>Menticirrhus</i> spp.
"	Kingfish spp. (adult)	<i>Menticirrhus</i> spp.
"	Sciaenidae (juvenile)	Sciaenidae (juvenile)
"	Silver Perch	<i>Bairdiella chrysoura</i>
"	Silver Seatrout	<i>Cynoscion nothus</i>
"	Southern Kingfish	<i>Menticirrhus americanus</i>
"	Spot	<i>Leiostomus xanthurus</i>
"	Star Drum	<i>Stellifer lanceolatus</i>
"	Weakfish	<i>Cynoscion regalis</i>
Scombridae	Atlantic Spanish Mackerel	<i>Scomberomorus maculatus</i>
Scorpaenidae	Scorpionfish (Scorpaena spp. Juvenile)	<i>Scorpaena</i> spp.
"	Smooth-head Scorpionfish	<i>Scorpaena calcarata</i>
Serranidae	Black Sea Bass	<i>Centropristis striata</i>
"	Dwarf Sand Perch	<i>Diplectrum bivittatum</i>
"	Pygmy Sea Bass	<i>Serraniculus pumilio</i>
"	Rock Sea Bass	<i>Centropristis philadelphica</i>
"	Sand Perch	<i>Diplectrum formosum</i>
"	Sand Perch (Diplectrum spp.)	<i>Diplectrum</i> spp.
"	Sea Bass (Centropristis spp. juvenile)	<i>Centropristis</i> spp.
"	Sea Bass/Sand Perch (juvenile)	<i>Centropristis/Diplectrum</i> spp.
"	Soapfish (Rypticus spp. juvenile)	<i>Rypticus</i> spp.
Sparidae	Pinfish	<i>Lagodon rhomboides</i>
"	Porgy (Calamus spp. juvenile)	<i>Calamus</i> spp.
Sphyraenidae	Barracuda (Sphyraena spp. juvenile)	<i>Sphyraena</i> spp.
"	Guaguanche	<i>Sphyraena guachancho</i>
Stromateidae	Butterfish (Peprilus spp. juvenile)	<i>Peprilus</i> spp.
"	Harvestfish	<i>Peprilus paru</i>
Syngnathidae	Bull Pipefish	<i>Syngnathus springeri</i>
"	Chain Pipefish	<i>Syngnathus louisianae</i>
"	Dusky Pipefish	<i>Syngnathus floridae</i>
"	Gulf Pipefish	<i>Syngnathus scovelli</i>
"	Lined Seahorse	<i>Hippocampus erectus</i>
"	Northern Pipefish	<i>Syngnathus fuscus</i>
"	Pipefish (Syngnathus spp. juvenile)	<i>Syngnathus</i> spp.
"	Sargassum Pipefish	<i>Syngnathus pelagicus</i>
Synodontidae	Inshore Lizardfish	<i>Synodus foetens</i>
"	Largescale Lizardfish	<i>Saurida brasiliensis</i>
"	Lizardfish (Synodus spp. juvenile)	<i>Synodus</i> spp.
"	Snakefish	<i>Trachinocephalus myops</i>
Synodontidae	Synodontidae (juvenile)	Synodontidae (juvenile)
Tetraodontidae	Bandtail Puffer	<i>Sphoeroides spengleri</i>

Family_Superorder	Common Name	Scientific Name
"	Puffer (Sphoeroides spp. juvenile)	<i>Sphoeroides</i> spp.
"	Puffer (Tetraodontidae juvenile)	Tetraodontidae (juvenile)
Trichiuridae	Atlantic Cutlassfish	<i>Trichiurus lepturus</i>
Triglidae	Bandtail Searobin	<i>Prionotus ophryas</i>
"	Bighead Searobin	<i>Prionotus tribulus</i>
"	Blackwing Searobin	<i>Prionotus rubio</i>
"	Bluespotted Searobin	<i>Prionotus roseus</i>
"	Leopard Searobin	<i>Prionotus scitulus</i>
"	Northern Searobin	<i>Prionotus carolinus</i>
"	Searobin (<i>Prionotus</i> spp. juvenile)	<i>Prionotus</i> spp.
"	Striped Searobin	<i>Prionotus evolans</i>
Uranoscopidae	Stargazer (<i>Astroscopus</i> spp. Juvenile)	<i>Astroscopus</i> spp.
Dasyatidae	Bluntnose Stingray	<i>Dasyatis say</i>
"	Southern Stingray	<i>Dasyatis americana</i>
Gymnuridae	Smooth Butterfly Ray	<i>Gymnura micrura</i>
Myliobatidae	Bullnose Eagle Ray	<i>Myliobatis freminvillei</i>
"	Cownose Ray	<i>Rhinoptera bonasus</i>
"	Southern Eagle Ray	<i>Myliobatis goodei</i>
Narcinidae	Lesser Electric Ray	<i>Narcine bancroftii</i>
Rajidae	Clearnose Skate	<i>Raja eglanteria</i>
Rhinobatidae	Atlantic Guitarfish	<i>Rhinobatos lentiginosus</i>
Sphyrnidae	Bonnethead Shark	<i>Sphyrna tiburo</i>

Elasmobranchs

Family_Superorder	Common Name	Scientific Name
Dasyatidae	Bluntnose Stingray	<i>Dasyatis say</i>
"	Southern Stingray	<i>Dasyatis americana</i>
Gymnuridae	Smooth Butterfly Ray	<i>Gymnura micrura</i>
Myliobatidae	Bullnose Eagle Ray	<i>Myliobatis freminvillei</i>
"	Cownose Ray	<i>Rhinoptera bonasus</i>
"	Southern Eagle Ray	<i>Myliobatis goodei</i>
Narcinidae	Lesser Electric Ray	<i>Narcine bancroftii</i>
Rajidae	Clearnose Skate	<i>Raja eglanteria</i>
Rhinobatidae	Atlantic Guitarfish	<i>Rhinobatos lentiginosus</i>
Sphyrnidae	Bonnethead Shark	<i>Sphyrna tiburo</i>

Appendix H. List of All Publications, Theses, Dissertations, and Conferences Presentations as a Result of this Study, as of Publication Date of the Final Report Volumes

Category	Year	First Author	Citation
Conference Presentation Poster	2014	Paniagua-Arroyave, Juan F	Paniagua-Arroyave JF, Parra SM, Adams PN, Valle-Levinson A. 2014. Observations of surface gravity waves within a bumpy environment: the "Canaveral 2 Shoals" borrow site, Florida. Summer Institute on Earth-surface Dynamics.
Conference Presentation Poster	2015	Paniagua-Arroyave, Juan F	Paniagua-Arroyave JF, Adams PN, Valle-Levinson A, Parra SM. 2015. Tidal variability of infragravity waves over cape-associated shoals. American Geophysical Union (AGU).
Conference Presentation Poster	2016	Paniagua-Arroyave, Juan F	Paniagua-Arroyave JF, Adams PN, Valle-Levinson A, Parra SM. 2016. Effects of cape-related shoals on the variability of long gravity waves. American Geophysical Union (AGU).
Conference Presentation Poster	2016	Parra, Sabrina M	Parra SM, Valle-Levinson A, Adams PN, Paniagua-Arroyave JF. 2016. Subtidal dynamics of cape-associated shoals. American Geophysical Union (AGU).
Conference Presentation Poster	2017	Paniagua-Arroyave, Juan F	Paniagua-Arroyave JF, Adams PN, Valle-Levinson A, Parra SM. 2017. Quantifying morphological changes of cape-related shoals. American Geophysical Union (AGU).
Other	2017	Paniagua-Arroyave, Juan F	Paniagua-Arroyave JF, Adams PN, Valle-Levinson A, Parra SM. 2017. Infragravity energy fluxes at inner and outer swales of Shoal E, Cape Canaveral, Florida: Experimental set-up, Tech. rep., Sustainable Environment through Actionable Data, National Center for Supercomputing Applications, University of Illinois at Urbana-Champaign.
Thesis	2017	Tate, Mary	Spatial, temporal, and water depth variation in the composition of phytoplankton in a subtropical coastal shelf environment. University of Florida.
Dissertation	2018	Alkhaldi, Mohammad	Turbulence over an inner shelf influenced by waves, tides, and thermal stratification. University of Florida.
Dissertation	2018	Paniagua-Arroyave, Juan F	Hydrodynamic processes associated with the evolution of cape-related shoals. University of Florida.
Peer-reviewed Journal Article	2019	Paniagua-Arroyave, Juan F	Paniagua-Arroyave JF, Adams PN, Parra SM, Valle-Levinson A. 2019. Observations of surface-gravity-wave scattering and dissipation by an isolated shoal related to a cusped foreland. Continental Shelf Research 173: 43-55.

Category	Year	First Author	Citation
Peer-reviewed Journal Article	2019	Paniagua-Arroyave, Juan F	Paniagua-Arroyave JF, Valle-Levinson A, Adams PN, Parra SM. 2019. Coherence between infragravity waves and ambient water motions over cape-associated shoals. <i>Continental Shelf Research</i> 173: 27-42.
Peer-reviewed Journal Article	2019	Paniagua-Arroyave, Juan F	Paniagua-Arroyave JF, Valle-Levinson A, Parra SM, Adams PN. 2019. Tidal distortions related to extreme atmospheric forcing over the inner shelf. <i>Journal of Geophysical Research: Oceans</i> 124: 6688-6701.
Conference Presentation Oral	2019	Paniagua-Arroyave, Juan F	Paniagua-Arroyave JF, Parra SM, Valle-Levinson A, Adams PN. 2019. Observations of bed elevation changes at Cape Canaveral shoals. <i>Coastal Sediments 2019</i> .
Conference Presentation Oral	2019	Stelling, Benjamin	Stelling B, Philips EJ, Badylak S, Tate M, Landauer L, West-Valle A, Hamazaki M. 2019. Investigating phytoplankton community trends off Cape Canaveral, FL—Different diatoms for a different season. 25th Biennial Coastal and Estuarine Research Federation Conference.
Conference Presentation Oral	2020	Paniagua-Arroyave, Juan F	Paniagua-Arroyave JF, Valle-Levinson A, Parra SM, Adams PN. 2020. Overtides at the inner shelf offshore of a cape during extreme atmospheric forcing. <i>Ocean Sciences Meeting 2020</i>
Conference Presentation Oral	2020	Saldarriaga, Anderson A	Saldarriaga AA, Chavarriaga ER, Paniagua-Arroyave JF. 2020. Application of a Coupled-Mode Theory for Wind-Wave Propagation Over an Idealized Cape-Related Shoal. <i>American Geophysical Union (AGU)</i>
Peer-reviewed Journal Article	2020	Tate, Mary	Tate MC, Philips EJ, Stelling B, Badylak S, Landauer L, West-Valle A, Murie D. 2020. Phytoplankton composition in the subtropical coastal shelf environment of Cape Canaveral, Florida. <i>Bulletin of Marine Science</i> 96: 593-615.
Peer-reviewed Journal Article	2021	AlYousif, Ahmad	AlYousif A, Laurel-Castillo JA, So S, Parra S, Adams P, Valle-Levinson A. 2021. Subinertial hydrodynamics around a cape influenced by a western boundary current. <i>Estuarine, Coastal and Shelf Science</i> 251: 107199.
Peer-reviewed Journal Article	2021	AlYousif, Ahmad	AlYousif A, Valle-Levinson A, Adams P, Laurel-Castillo JA. 2021. Tidal and subtidal hydrodynamics over ridge-swale bathymetry. <i>Continental Shelf Research</i> 2019: 104392.
Dissertation	2021	Stelling, Benjamin	Phytoplankton composition & abundance along depth & seasonal gradients in the South Atlantic Bight off the coast of Cape Canaveral, FL. University of Florida.

Category	Year	First Author	Citation
Peer-reviewed Journal Article	2022	Liao, Zhiling	Zhiling L, Li S, Paniagua-Arroyave JF, Liu Y, Zou Q. 2022. Infragravity wave amplification by isolated topography: Field observations and semi-analytical modeling. Applied Ocean Research 122: 103119.
Conference Presentation Poster	2023	Murie, Debra	Fish assemblages on offshore sand shoals and potential impacts due to dredging events. Florida Chapter of the American Fisheries Society.



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Bureau of Ocean Energy Management (BOEM)

BOEM's mission is to manage development of U.S. Outer Continental Shelf energy and mineral resources in an environmentally and economically responsible way.

BOEM Environmental Studies Program

The mission of the Environmental Studies Program is to provide the information needed to predict, assess, and manage impacts from offshore energy and marine mineral exploration, development, and production activities on human, marine, and coastal environments. The proposal, selection, research, review, collaboration, production, and dissemination of each of BOEM's Environmental Studies follows the DOI Code of Scientific and Scholarly Conduct, in support of a culture of scientific and professional integrity, as set out in the DOI Departmental Manual (305 DM 3).