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Ecosystem Status Report for the U.S. South Atlantic Region

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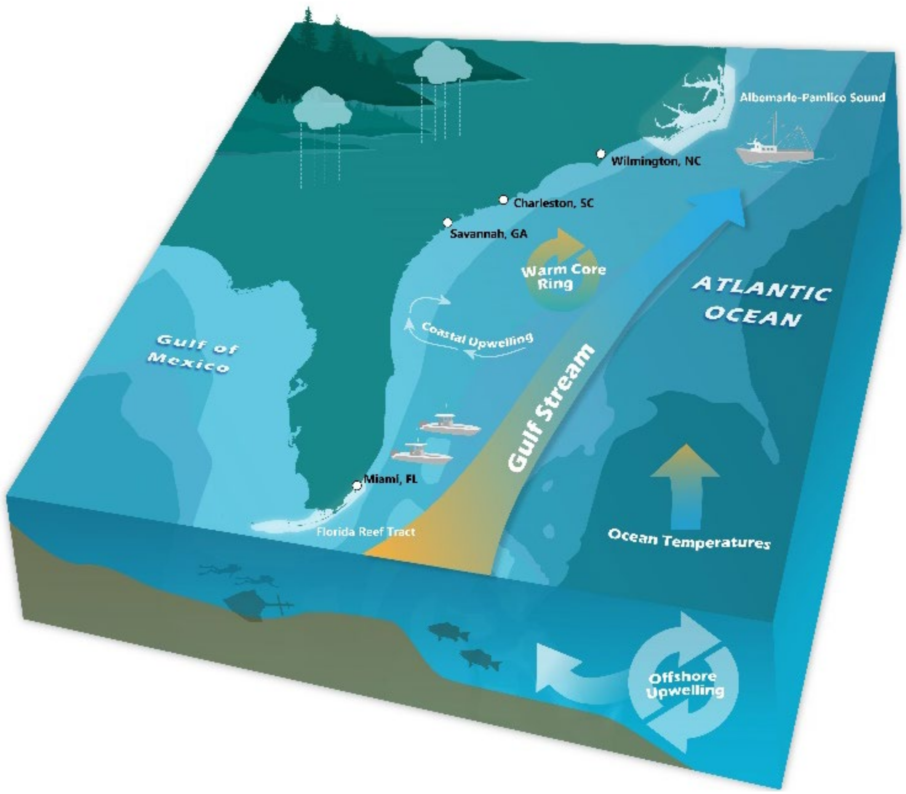

U.S. DEPARTMENT OF COMMERCE
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National Marine Fisheries Service
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1. HIGHLIGHTS

- **Climate Drivers** – The North Atlantic SST tripole, which was mostly in a positive phase from the mid-1990s to early 2010s, switched to a strongly negative phase in 2011 and has recently (2017) returned to near its long-term mean. The AMO switched to a positive phase in the mid-1990s. The NAO was in a strong positive phase in 2018 and negative phase in 2020.

- **South Atlantic Warming** – Multiple temperature indicators suggest warming in the South Atlantic at seasonal (winter-spring temperatures), annual (since the late 2010s), and decadal (since the 1980s) time scales. Thermal stress in coral reef ecosystems has also increased since the mid – 1980s.

- **Coastal Upwelling and Primary Productivity** – Coastal upwelling has declined since the early 2010s while primary productivity was low from 2010 – 2015 compared to earlier and later years. These observations, along with increases in winter-spring temperatures since 2014, suggest recent changes in ocean dynamics in the U.S. South Atlantic ecosystem.

- **Physical and Chemical Pressures Increase** – Several important physical and chemical pressures have increased over time, including sea level rise, nutrient loading to the coast from rivers and streams, and pCO_2 (an indicator of ocean acidification). The U.S. southeast region is becoming wetter with a corresponding decrease in the probability of extreme regional drought.

- **Status of Coastal Habitats Uncertain** – Limited data exists to assess the regional-scale status of important coastal habitats (wetlands, submerged aquatic vegetation (SAV), coral reefs, oyster reefs). While total wetland loss appears relatively small, forest cover and forested wetlands have decreased, consistent with increasing development in the region. Recent changes in SAV appear moderate, but data are generally not available prior to the 2000s when large losses likely occurred.

- **Status of Lower Trophic Levels Uncertain** – Limited data exist to assess the state of lower trophic levels in the U.S. South Atlantic. Measures of zooplankton biovolume and larval fish abundance and diversity are highly variable and likely reflect localized rather than regional trends.

- **Changes in Fish Community Structure** – Species richness and abundance of offshore hard bottom reef fishes have generally declined over time while richness and abundance of demersal fishes in soft sediment habitats on the nearshore shelf have increased. Potential explanations for these patterns include changes in harvest (directed and bycatch), trophic interactions, and environment effects on recruitment.

- **Mean Trophic Level Declines** – Mean trophic level (MTL) of the commercial and recreational landings has declined since the 1980s but has been stable in recent years. Declines in MTL are relatively small and driven in part by increases in landings of low trophic level species.

- **Patterns from Stock Assessments** – Biomass of most assessed species generally show declines from the 1970s through the 1990s with some species showing signs of recovery beginning in the early to mid-2000s. Recruitment of a number of snapper-grouper species has declined since the early 2010s whereas recruitment of Red Snapper and some pelagic species has increased in recent years.

- **Recreational Fishing Dominates Harvest** – Since the 1980s, recreational landings and effort have increased while commercial landings and revenue have declined. Commercial landings of estuarine Blue Crab and Penaeid Shrimp fisheries have generally declined since the mid – 1990s. Harvest in the South Atlantic is increasingly dominated by the recreational fishing sector, which accounts for > 80% of the landings of federally-managed species in recent years.

- **Overfishing Status Declines while Overfished Status Remains** – The proportion of stocks undergoing overfishing (high mortality) has declined since the early 2000s and has been relatively stable over the last five years. The number of overfished stocks (low biomass) was at a low point in 2005, but has changed little since then, suggesting some stocks have not recovered to sustainable biomass levels despite ending overfishing.

- **Status of Birds, Sea Turtles, and Cetaceans** – Marine bird abundances have been stable or increasing since the 1990s while Loggerhead Sea Turtle (*Caretta caretta*) nest counts have been increasing since the mid – 2000s. Marine mammal strandings have been sporadic with a large Bottlenose Dolphin disease-related mortality event in 2013 but no long-term trend.

- **Human Population Growing Rapidly** – The four South Atlantic states (NC, SC, GA, FL) are among the fastest in the nation in terms of human population growth. Urban land cover has increased at a rate of 18 – 23% and employment in the ocean economy and ocean-related GDP are at their highest levels in the most recent years.

- **Regional Variation in Socioeconomic Patterns –** Reliance on commercial and recreational fishing is highest in rural coastal regions of North Carolina, while engagement in commercial and recreational fishing is highest in Florida. Overall, social connectedness has declined in the U.S. South Atlantic, with the biggest losses occurring in North Carolina and South Carolina.

2. INTRODUCTION

The U.S. South Atlantic Ecosystem Status Report (ESR) is intended to support ecosystem-based management in the South Atlantic region by providing quantitative, scientific information about the ecosystem to scientists, managers, decision-makers, and stakeholders. For the purposes of this report, the South Atlantic ecosystem is defined as estuarine and marine waters from North Carolina through the Florida Keys. The term "South Atlantic Bight," which is variably defined as the coastal area between Cape Hatteras, North Carolina and Cape Canaveral, Florida (e.g., [1]) or more generally as offshore waters from North Carolina to Florida (e.g., [2]), is sometimes used in the report in reference to the literature. Several land-based indicators included in the report are based on coastal watershed counties. Coastal watershed counties are defined as counties where at least 15% of the total land area is located within a coastal watershed, and are considered areas where land use practices and other anthropogenic activities most directly affect coastal ecosystems [3]. The intent of the report is to provide a broad-level overview of the current state of the South Atlantic ecosystem through the development of multiple indicators depicting recent and historical trends in key components of the ecosystem. This is the first ESR for the South Atlantic region and was developed with contributions from multiple state and federal government agencies, academic partners, and non-governmental organizations. The report is considered a starting point for an ongoing dialog among the various stakeholders in the South Atlantic region, and is intended to be revised and updated as new indicators and additional data become available.

2.1 Indicator selection

Ecosystem indicators are specific, well-defined, and measurable quantities that reflect the status of some component of the ecosystem. For this report, indicator selection was broadly based on the DPSER (Drivers – Pressures – States – Ecosystem Services – Responses) conceptual modeling framework [4]. In addition, a number of demographic, social, and economic indicators that measure various aspects of human well-being were included.

Multiple criteria were used in the selection of indicators. In particular, annual or sub-annual (e.g., monthly) quantitative time series over at least a decade and at the scale of the entire South Atlantic region (North Carolina through the east coast of Florida) are emphasized. In some cases, time series were not available at broad spatial and temporal scales and only a change between two points in time (e.g., nutrient loadings) or for a key region or habitat (e.g., coral reefs) are reported. Available system-specific measurements are reported in tabular form for some indicators where limited data precluded the development of time series, such as various estuarine and marine habitats (e.g., seagrasses, oyster reefs). In addition to data availability and spatial and temporal scale, other criteria used to select indicators included: i) relevance to resource management, ii) familiarity to multiple stakeholder groups, iii) links to important human and societal dimensions, and iv) use in other regions. The selected indicators reported here span a wide range of ecosystem components, including climate drivers, physical and chemical pressures, habitat, lower and upper trophic levels, ecosystem services related to fisheries and species of conservation concern, as well as various measures of human well-being. In total, 154 indicators representing 47 ecosystem components across seven categories are reported for the South Atlantic ecosystem. The end year of each indicator reflects the most recent data available at the time the indicator was developed. Most indicators extend at least through 2017 with several extending to 2018, 2019, or 2020. We expect this interdisciplinary report will serve to identify linkages among different components of the South Atlantic ecosystem as well as identify knowledge gaps where additional research and synthesis is needed.

2.2 Notes on interpreting time series figures

Most indicators are plotted as annual or monthly time series in a standardized format for ease of interpretation (Fig. 2.1). The x-axis represents the temporal dimension and the y-axis represents the indicator value in units specified on the figure. Several time series are reported as standardized monthly anomalies, defined as the difference between the monthly mean and the time series mean divided by the time series standard deviation. The dashed horizontal line represents the mean indicator value across the entire time series, and the solid horizontal lines denote the mean plus or minus one standard deviation. Green and red shaded areas show years or months when the indicator value was above or below one standard deviation from the long-term mean, respectively. The blue vertical shaded box highlights the last five years of indicator values. Black circles to the right of each figure show whether the mean of the indicator over the last five years is greater than (plus sign), less than (minus sign), or within (black solid circle) one

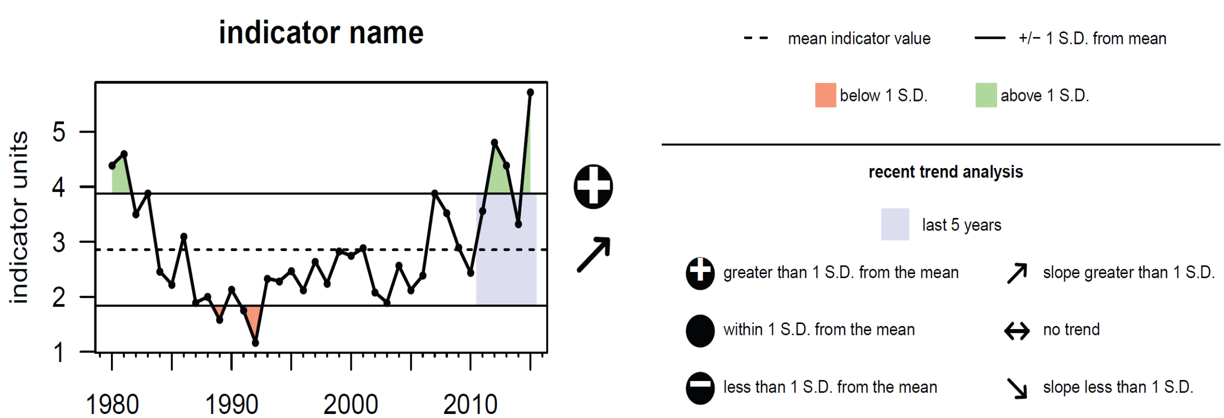


Figure 2.1. Example time series plot showing a hypothetical indicator plotted with its mean, standard deviation, and trend analysis for the five most recent years.

standard deviation from the mean of the overall time series. Arrows to the right of each figure indicate whether the slope of the least squares linear fit through the last five years of data is greater than one standard deviation from the mean in either the positive or the negative direction (upward- or downward-pointing arrows, respectively). A horizontal, right-pointing arrow indicates slopes less than one standard deviation from the mean. These trend indicators (circles and arrows) were not appropriate for all of the time series (e.g., those with limited data). Further, for many indicators, multi-panel plots show trends in the same indicator calculated for different species or over different spatial or temporal domains. In some cases, particularly where sufficient data to construct a time series were not available, other graphical or tabular representations of the available data were used.

3. CLIMATE DRIVERS

3.1 Atlantic Multidecadal Oscillation (AMO)

The Atlantic Multidecadal Oscillation (AMO) is a mode of multidecadal climate variability with alternating warm and cool phases over large portions of the North Atlantic [5]. The AMO manifests as oscillations in North Atlantic sea surface temperatures (SST) that are in part driven by fluctuations in the intensity of Atlantic thermohaline circulation, with a temperature range of ± 0.4° C over a period of 65 – 70 years [6]. Warm phases of the AMO are generally associated with decreased rainfall, decreased river flow, increased Atlantic hurricane activity, and increased drought severity in the southeastern U.S., while the opposite conditions occur during cool phases [5, 7, 8]. Like other modes of climate variability (e.g., El Niño Southern Oscillation), the effects of AMO extend over large geographic areas, and have been hypothesized to influence plankton biomass and fisheries production as well as the geographic range of mobile marine species and fisheries [9, 10, 11]. In the Gulf of Mexico, for example, changes in multiple physical and biological components of the ecosystem were associated with the 1995 – 1996 shift in the AMO from a cool phase to a warm phase [12].

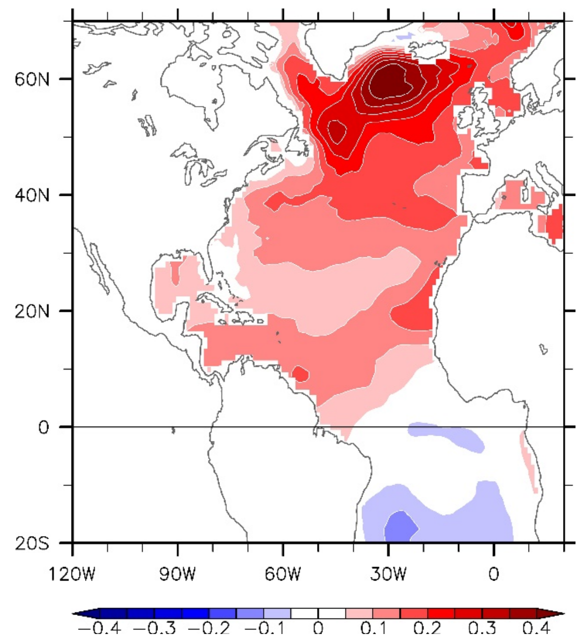


Figure 3.1a. Map of SST anomalies during a positive phase of the AMO.

Plots and standardized time series of the AMO were developed using the Centennial *in situ* Observation-Based Estimates (COBE) SST data [13, 14]. A map of the SST anomalies during the recent warm phase of the AMO is shown in Figure 3.1a and an 11-year running average of the AMO index is shown in Figure 3.1b. The AMO was in a warm phase from the 1950s to the 1960s, a cool phase from 1970s to the early 1990s, and then shifted back to a warm phase in the mid-1990s (Fig. 3.1b). The index has remained in a warm phase over the last five years.

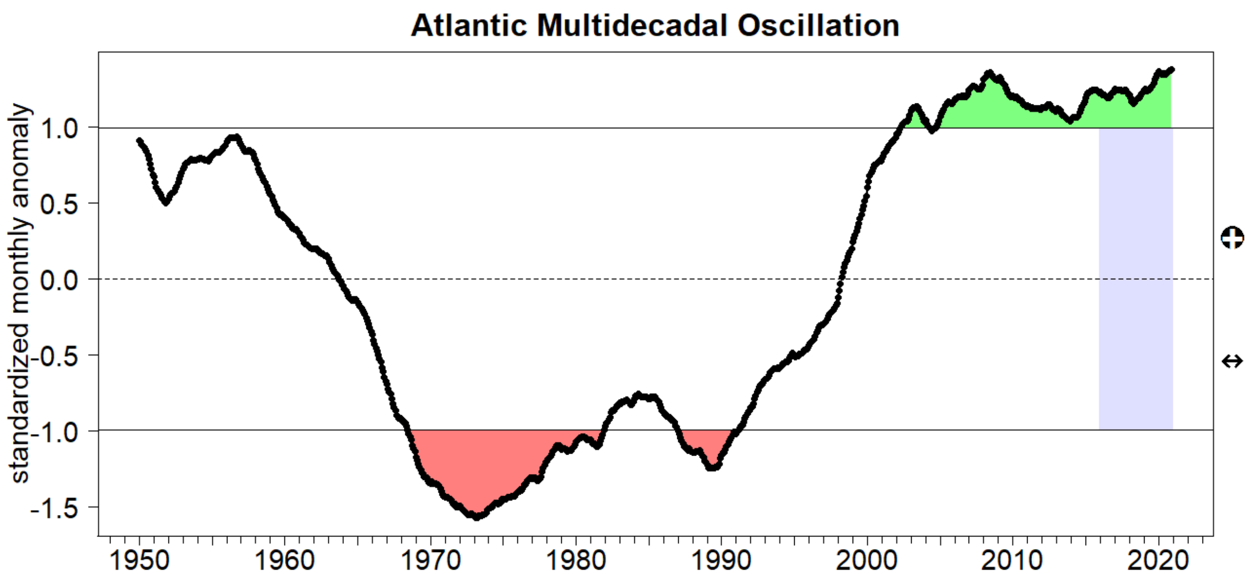


Figure 3.1b. Eleven-year running averaged standardized monthly anomalies of the Atlantic Multidecadal Oscillation (AMO).

3.2 North Atlantic Oscillation (NAO)

The North Atlantic Oscillation (NAO) is a dominant mode of atmospheric variability that influences wind patterns, air temperatures (particularly in winter), precipitation, and both the magnitude and frequency of storms over the mid latitudes of the North Atlantic Ocean [15, 16]. The NAO is defined as the difference in atmospheric pressure at sea level between the Icelandic low and the Azores high [17]. When the NAO is in a positive state, westerly winds intensify and shift northward. As a result, the eastern seaboard of the United States typically experiences warmer sea temperatures, a shallower winter mixed layer depth, and a modest decrease in winter storm activity. Changes in the NAO have been linked to changes in plankton community composition, larval fish growth and survival, and competition among temperate bird species [18].

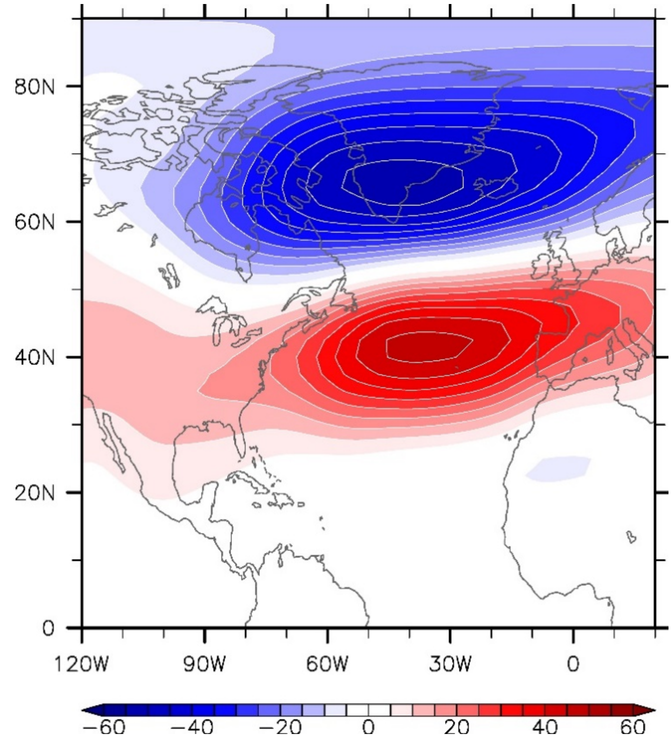


Figure 3.2a. Map of sea level pressure anomalies in December – February during a positive phase of the NAO.

Standardized monthly anomalies of the NAO were developed using methods described in [19]. A map of the winter (December – February) NAO is shown in Figure 3.2a and an 11-month running average of the NAO index is shown in Figure 3.2b. The NAO was in a mostly positive phase in the 1980s and early 1990s, a negative phase in the mid-2000s to 2010s and a mostly neutral phase (with the exception of 2018) in recent years (Figure 3.2b).

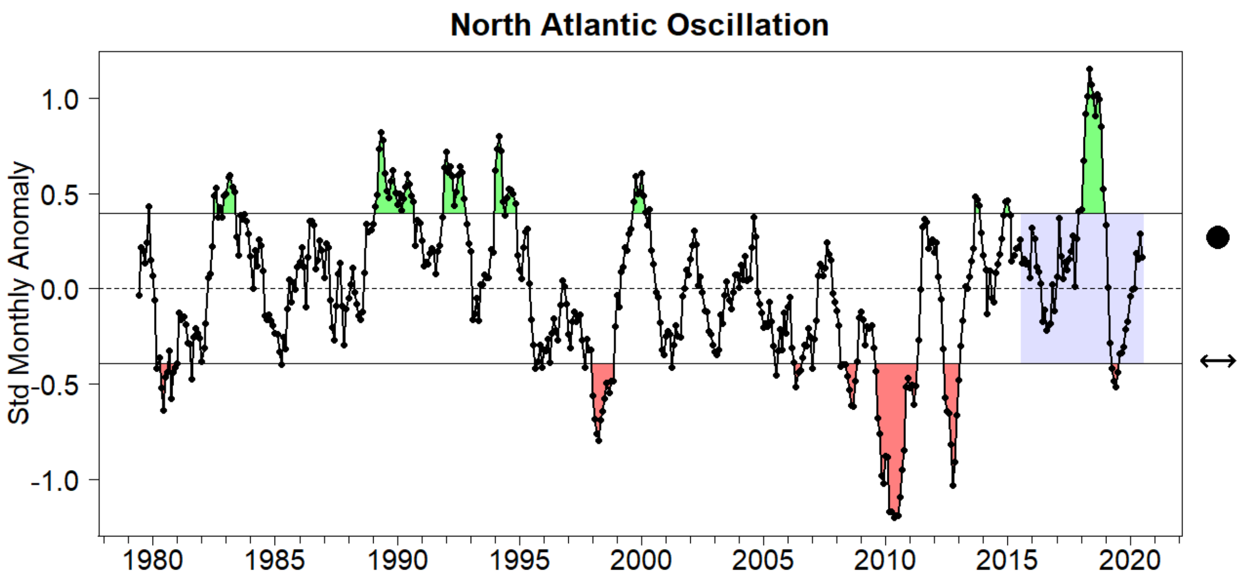


Figure 3.2b. Standardized monthly anomalies of the North Atlantic Oscillation index.

3.3 El Niño Southern Oscillation (ENSO)

The El Niño Southern Oscillation (ENSO) is an important mode of interannual climate variability centered in the tropical Pacific Ocean with strong effects on winter sea surface temperatures (SSTs) in the Pacific. However, ENSO has global effects, including in the North Atlantic Ocean and over the continental United States, which are mediated by atmospheric teleconnections [20]. El Niño conditions are characterized by warm SSTs in the tropical Pacific from the International Date Line to the west coast of South America, while La Niña conditions are characterized by relatively cool SSTs in this region. The Southern Oscillation is the atmospheric phenomenon linked to El Niño and involves the exchange of air between the eastern

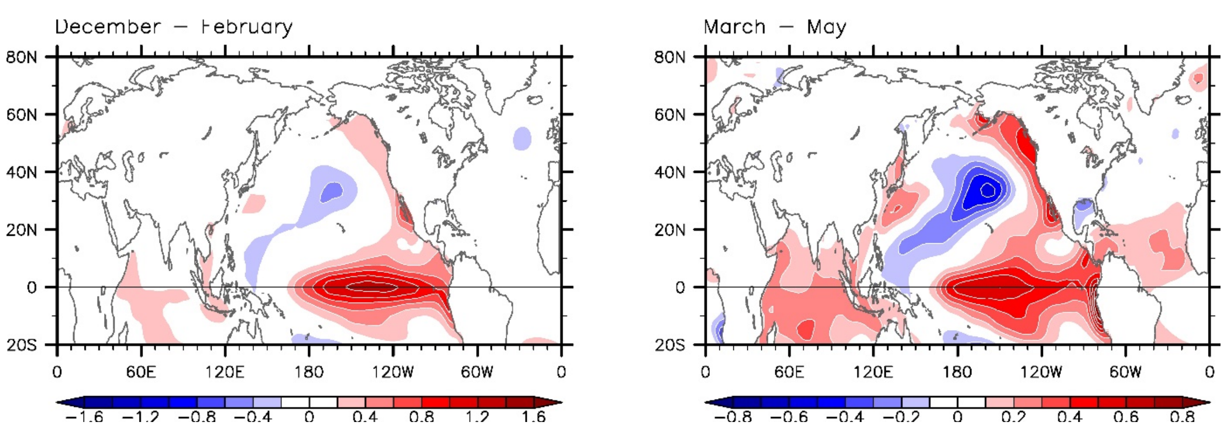


Figure 3.3a. Map of SST anomalies in winter (December – February; left panel) and spring (March – May; right panel) during a positive phase of ENSO (i.e., El Niño).

and western hemispheres mainly in tropical and subtropical latitudes. El Niño reduces the transport of warm, moist air into the middle and southern U.S. Atlantic states, leading to reduced precipitation and greater solar radiation in the U.S. Southeast region. Tropical cyclone activity in the North Atlantic tends to be reduced during El Niño years and increased during La Niña years [21]. ENSO also interacts with the AMO, the NAO, and the Atlantic Warm Pool (AWP) to influence climate on interannual to multidecadal time scales. These interactions influence North Atlantic sea surface temperature [22], regional patterns in precipitation and drought frequency [23], freshwater discharge [24], and hurricane tracks and intensity [25].

The Niño 3.4, one measure of the El Niño-Southern Oscillation, is a three-month running mean of SST anomalies in the Niño 3.4 region (5º N – 5º S, 120º – 170º W) based on centered 30-year base periods updated every 5 years. Standardized monthly anomalies of the Niño 3.4 were taken from [26]. Figure 3.3a shows the SST anomalies during winter and spring when ENSO is in a positive phase. The Niño 3.4 index has been near its long-term average conditions over the last five years with a strong positive phase in 2015-2016 and a slight positive or neutral phase since then (Fig. 3.3b).

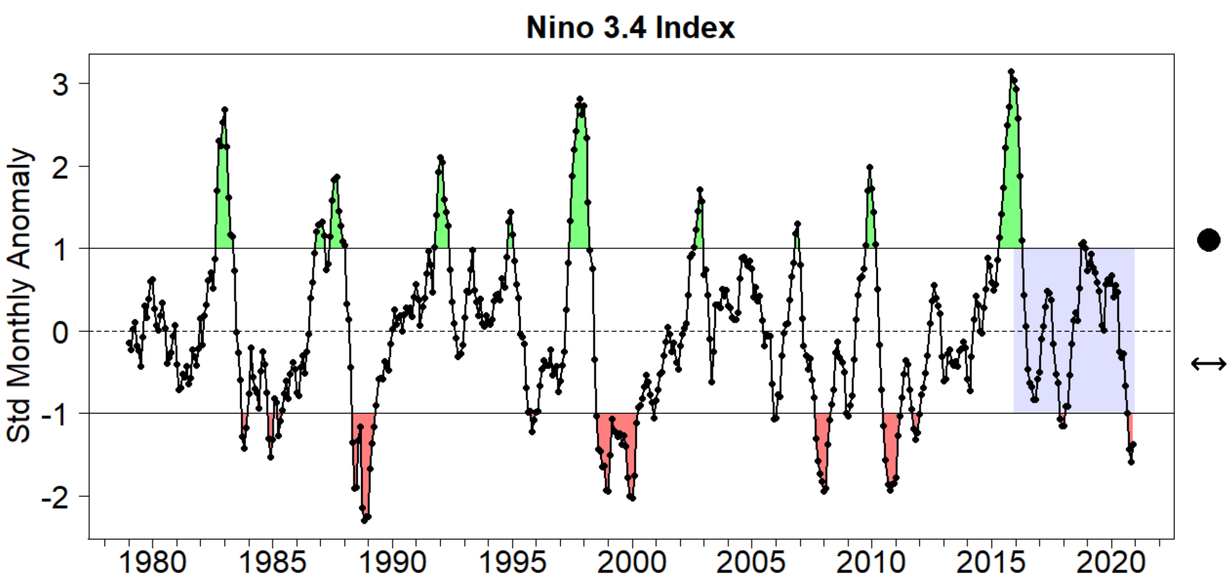


Figure 3.3b. Three-month running average of standardized monthly anomalies of the Niño 3.4.

3.4 North Atlantic Sea Surface Temperature Tripole

The North Atlantic sea surface temperature (SST) tripole is the dominant mode of interannual SST variability in the Atlantic Ocean during winter and spring. During a positive phase, it is characterized by warm, cold, and warm SST anomalies in the tropical, subtropical, and subpolar North Atlantic, respectively (Fig. 3.4a). The North Atlantic SST tripole is linked to multiple forcing mechanisms including the North Atlantic Oscillation and extratropical teleconnections [27, 28, 29, 30, 31, 32, 33]. Several studies have shown that the tropical component of the North Atlantic SST tripole feeds back into the overlying atmosphere and modulates atmospheric variability over the United States (34, 35, 36, 37, 38). During its negative phase (i.e., cold, warm, and cold in the tropical, subtropical, and subpolar North Atlantic, respectively), an anomalous anticyclone straddles the subtropical North Atlantic extending westward over the U.S. [35, 37].

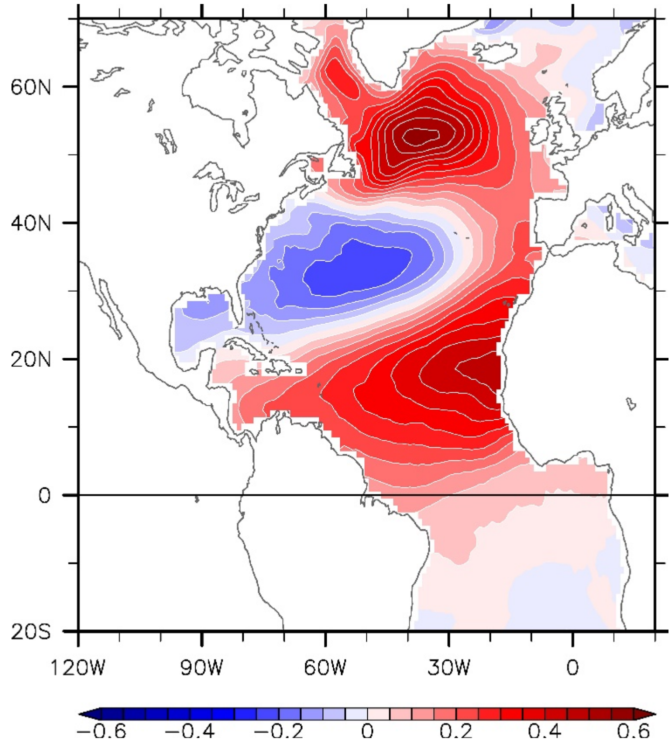


Figure 3.4a. Map of SST anomalies during the positive North Atlantic SST tripole, 1979 – 2017.

Convective available potential energy (CAPE) increases significantly over the western North Atlantic and Gulf of Mexico due to the warm SST anomalies in the regions. The anomalous anticyclone produces anomalous southeasterly winds across the U.S. east and Gulf coasts that, in turn, carry extra moisture toward the Southeast and Ohio Valley. These changes in the low-level vertical wind shear, moisture convergence, and CAPE increase rainfall and the probability of tornado outbreaks over the South, Ohio Valley and Southeastern U.S. [35]. These relationships are the opposite during the positive phase of the North Atlantic SST tripole.

An Empirical Orthogonal Function (EOF) analysis was applied to North Atlantic SST anomalies between 1979 and 2017 obtained from the Extended Reconstructed Sea Surface Temperature version 5 [39]. The North Atlantic SST tripole is the leading EOF mode and explains about 30% of the total variance. The principal component of the leading EOF mode (normalized by its variance) was considered an indicator the North Atlantic tripole (Figure 3.4b). The tripole was in an overall negative phase during most of the 1980s and early 1990s, but switched to an overall positive phase from the mid-1990s through the early 2010s. It shifted into a strong negative phase in early 2014, 2015 and 2018, but has mostly been in a neutral or slightly positive phase over the last five years.

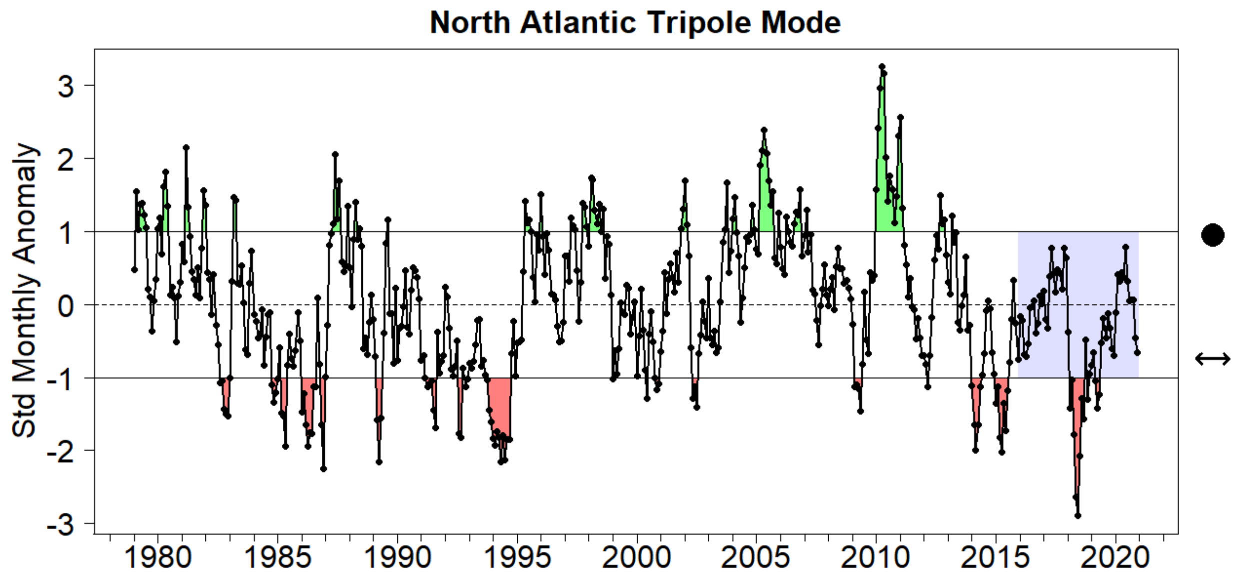


Figure 3.4b. Time series of the North Atlantic SST tripole mode.

3.5 Atlantic Warm Pool (AWP)

The Atlantic Warm Pool (AWP) is an area of warm water (≥ 28.5 °C) in the Gulf of Mexico, Caribbean, and western Tropical North Atlantic (TNA) [40]. The size of the AWP is highly variable and can fluctuate three-fold over interannual to decadal time scales [41]. The AWP affects moisture transport from the tropics to the north and, thus, patterns of mid-summer precipitation over North America, and plays a role in the severity of regional drought. The AWP also affects winds and moisture in the main hurricane development region off northern Africa, with a large AWP leading to increased Atlantic tropical cyclone activity [38]. While a large AWP is associated with an intensification of tropical storms and hurricanes, these storms tend to originate further east and track more northeastward so that the probability of landfall in the US is lower during years when the AWP is large [42]. The AWP is considered part of the mechanism by which the AMO influences climate through its effects on northward moisture transport and wind shear, and largely follows the AMO signal (larger AWP in the warm phase of the AMO). The AWP reaches its maximum extent in the summer (Aug – Sep), extending from the Gulf of Mexico and in some years well into the Central TNA (Fig. 3.5a). Since a large AWP manifests as an increase in SST in the U.S. South Atlantic region, which influences the depth and intensity of the surface mixed layer, it is important to the habitat use, spawning, and recruitment dynamics of many fish species [43].

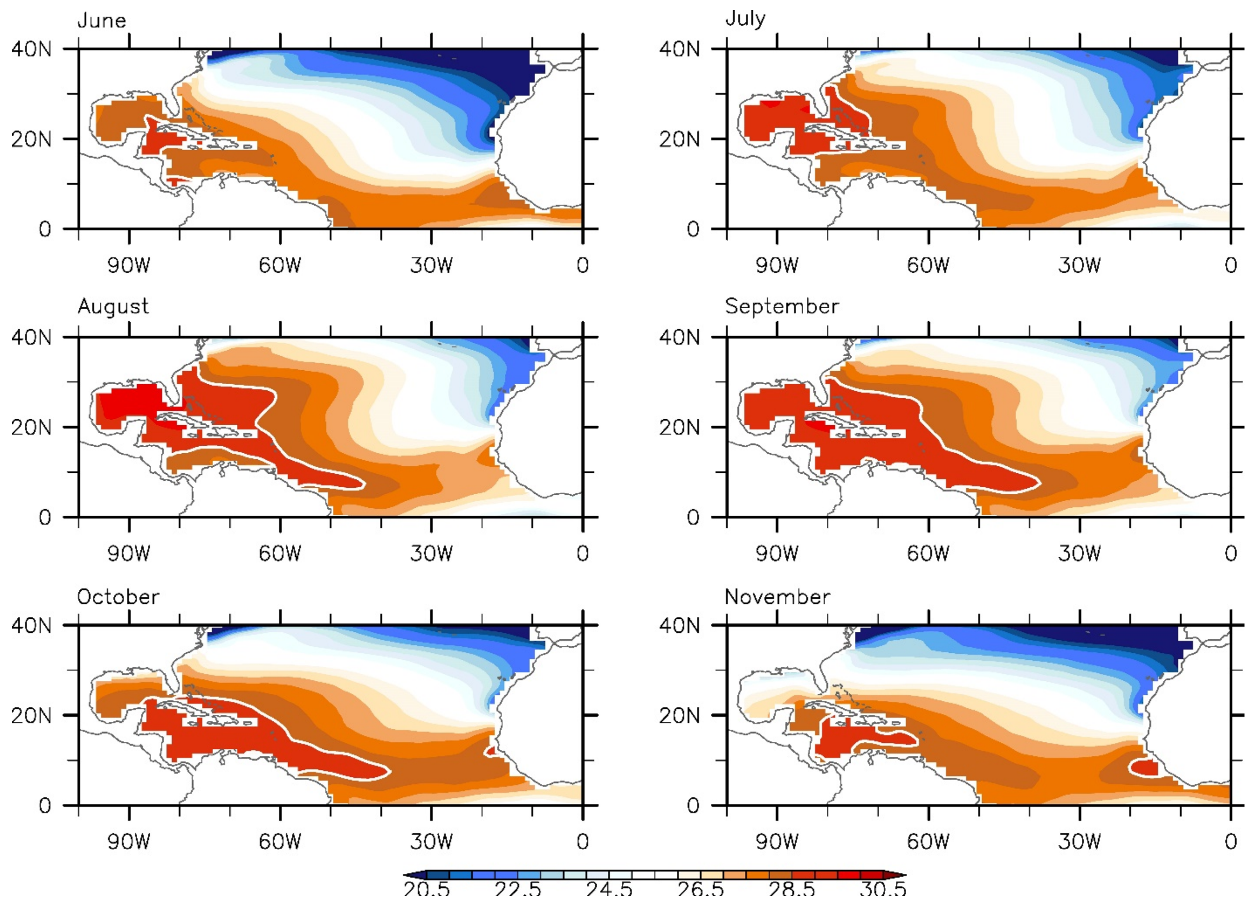


Figure 3.5a. Monthly SST evolutions of the Atlantic Warm Pool (AWP).

The area of the AWP was computed for Jun – Nov based on the Extended Reconstructed Sea Surface Temperature version 5 (ERSST5, [39]). The index represents the area with SST greater than 28.5°C expressed as an annual anomaly. For example, an AWP index value of 100% indicates the AWP area is twice as large as the climatological (i.e., mean) AWP. Over the last 30 years, the AWP has gradually increased, in part due to a change to the warm phase of the AMO and increasing SSTs (Fig. 3.5b). Currently, the AWP is somewhat larger than the historical average.

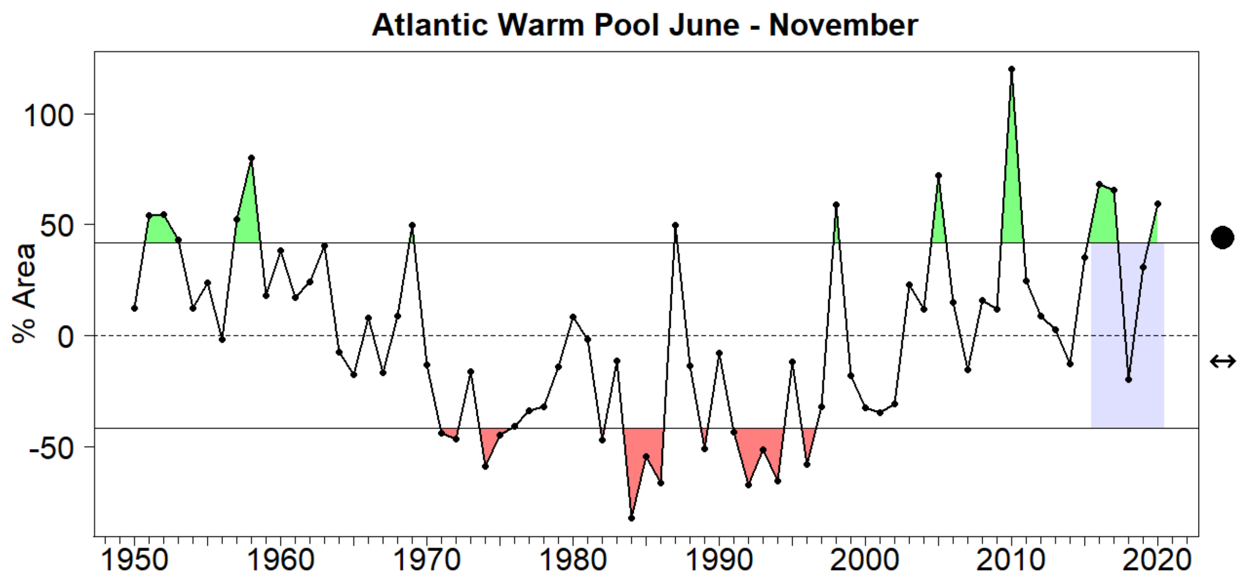


Figure 3.5b. Annual time series of the AWP from Jun – Nov. Values are annual anomalies (% area) from the climatological AWP.

4. PHYSICAL AND CHEMICAL PRESSURES

Several indicators of physical and chemical pressures influencing coastal watersheds, nearshore estuaries and coastal environments, and offshore oceanic environment were developed for this report. Temperature, in particular, is a fundamental property of marine ecosystems that has important effects on numerous physical and biological processes [44, 45], including shifts in the geographic distribution of mobile species [46, 47, 48], impaired growth or direct mortality for less mobile species [49, 50, 51] and interactive effects with other environmental stressors [52, 53]. Unusually cool temperatures can have negative consequences as well, such as cold stun mortality events [54, 55, 56]. An earl analysis of long-term (1875 – 2007) temperature data along the U.S. east coast indicated sea surface temperatures in the South Atlantic Bight have been relatively stable or weakly cooling (-0.1º per 100 years, [57]). Three temperature indicators reported below for the U.S. South Atlantic region suggest rising temperatures, particularly in the last decade.

4.1 Sea surface temperature

Two regionally-averaged monthly sea surface temperature (SST) time series were developed from satellite products with different temporal durations and spatial resolutions (Fig. 4.1). The spatial domain was 22.5° to 36.0 °N latitude and 82° to 74.5 °W longitude. The 1° resolution NOAA Reynolds OI SST data [58] were averaged monthly over this domain from December 1981 to December 2018. The 4 km resolution MODIS SST data [59] were averaged monthly from June 2002 to December 2018.

SST shows a strong seasonal signal with the highest temperatures in the summer months (Jun – Aug) and the coolest temperatures in the winter months (Dec – Feb). Standardized monthly anomalies show periods of relatively low winter temperature in the mid-2000s to the early-2010s, and increases in temperature since 2014, particularly during the winter and early spring (Fig. 4.1). Warm temperatures similar to those in recent years occurred occasionally throughout the time series, but typically over relatively brief, rather than sustained, periods of time. There has been an upward trend in SST over the last five years.

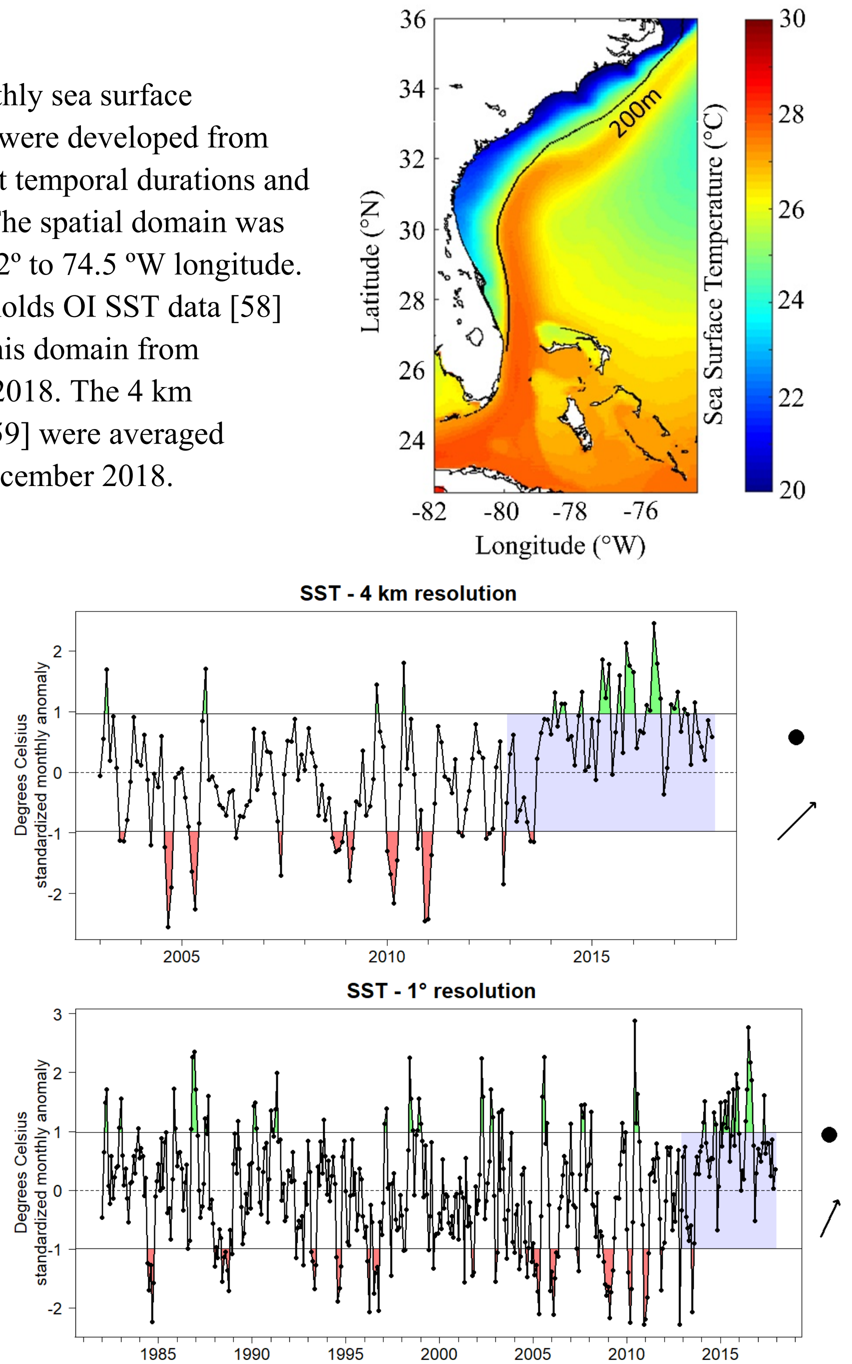


Figure 4.1. Standardized monthly anomalies of SST in the U.S. South Atlantic (map) based on short-term high resolution (middle panel) and long-term low resolution (bottom panel) satellite imagery.

4.2 Bottom temperature

Bottom water temperature data were collected from conductivity-temperature-depth (CTD) casts during the Southeast Reef Fish Survey (SERFS) [60] and the Southeast Area Monitoring and Assessment Program – South Atlantic (SEAMAP-SA) coastal trawl survey [61] between 1990 and 2017. SERFS conducted conductivity, temperature, depth (CTD) casts at stations ranging in depth from 14 to 100 m on the continental shelf between Cape Hatteras, North Carolina, and St. Lucie Inlet, Florida during April to October sampling cruises. SEAMAP-SA collected bottom water temperature during separate spring, summer, and fall research cruises, but much closer to shore (2–20 m depth). Bottom water temperatures were standardized by year, depth, day of the year, and spatial position using a generalized additive model [62].

Substantial variability in bottom temperature has occurred on the southeast U.S. Atlantic continental shelf since 1990 (Fig. 4.2). In 2003, a large, coastwide upwelling event occurred, causing bottom temperatures to drop to their lowest level in the time series [63]. The preceding year, 2002, was the warmest year in the time series, followed closely by 2016. The last four years (2016 – 2019) of the time series have been among the warmest observed and mostly above one standard deviation of the long-term mean, consistent with the observed pattern for SST (Fig. 4.1).

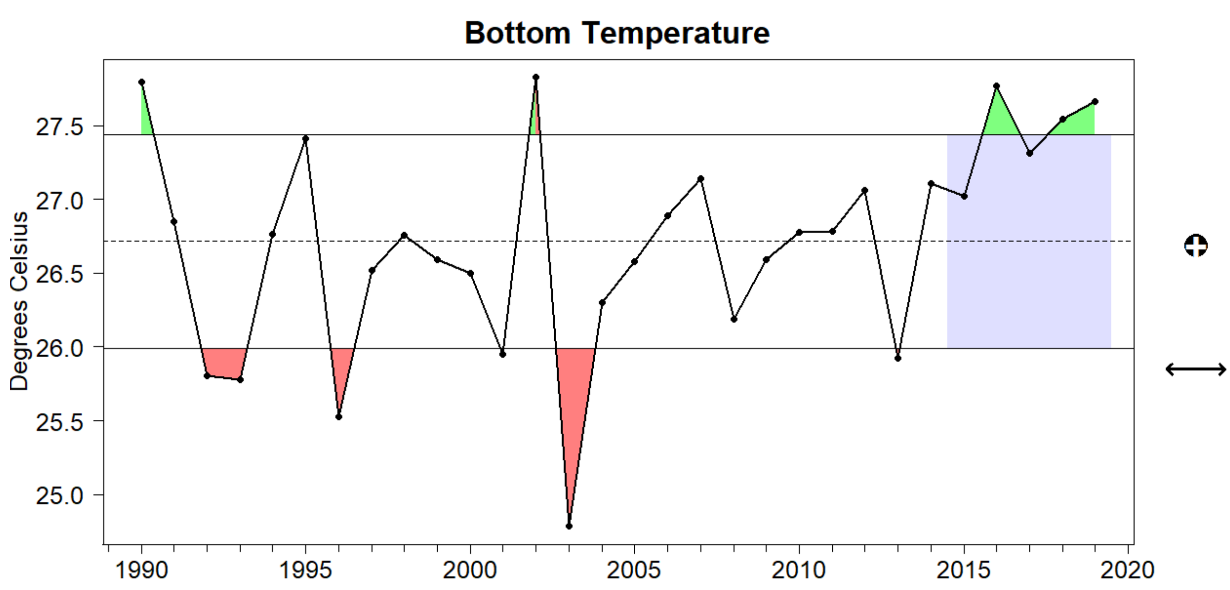


Figure 4.2. Mean annual (April to September) bottom temperature from shipboard surveys in the U.S. South Atlantic.

4.3 Decadal temperature

Mean decadal surface and bottom temperatures in the U.S. South Atlantic were constructed from the National Center for Environmental Information (NCEI) Southwest North Atlantic Regional Climatology (SWNA), with source data from 1955 – 2017 [64]. The SWNA climatology series comprises six decadal climatologies, the first centered on 1960 (i.e., data from 1955 – 1964), extending through 2010 (i.e., data from 2005 – 2017). The latest “decade” was extended by two years (to 2017) to take advantage of the most recent available data. The source data include all available *in situ* ocean temperature data from the NCEI World Ocean Database 2018, an all-source database containing quality controlled measurements from conductivity, temperature, depth probes (CTDs), expendable bathythermographs (XBTs), thermosalinographs, moored arrays, moored or profiling floats, and gliders [65]. For the period 1955 – 2017, a total of 37,448

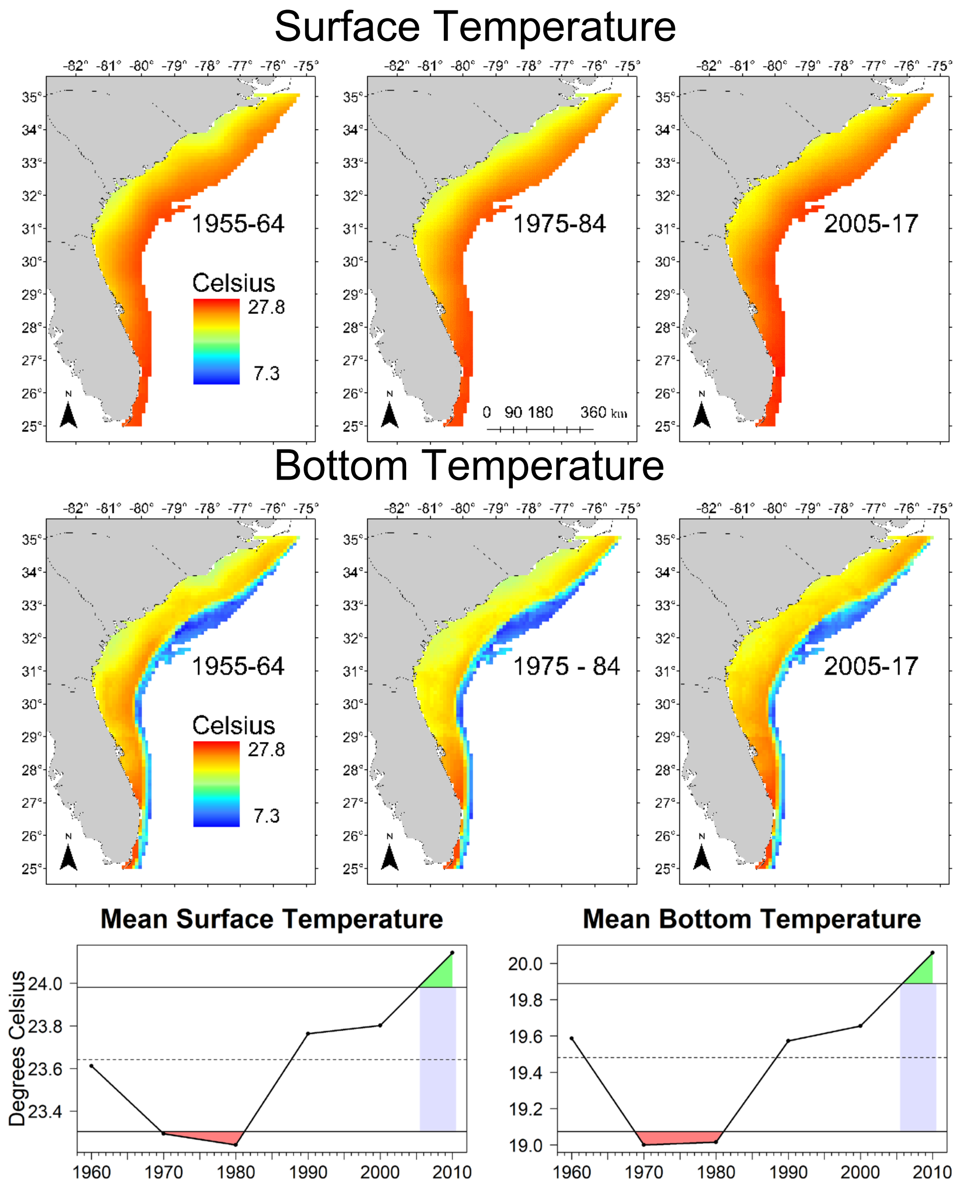


Figure 4.3. Decadal temperatures derived from the NCEI Ocean Database. Top two rows show distributions of surface and bottom temperatures in the South Atlantic for three decades. Bottom row shows mean temperature for each decade (1955 – 1965 through 2005 – 2017). Note the different scales on the y-axes.

surface measurements and 14,224 bottom measurements were available from the coastline to the 500 m isobath within the region bounded by 25 – 35 ºN (Fig. 4.3, maps).

The decadal mean fields that make up the SWNA were computed via the commonly used objective analysis procedure [66] following the approach used in NCEI's World Ocean Atlas 2018 [67]. The vertical grid resolution is 5 m down to 100 m depth, and 25 m down to 500 m depth. Bottom temperature fields were created by extracting the bottom (lowest) grid cell at each horizontal location. For more information on the construction of NCEI regional climatologies see [68].

Surface temperatures generally increase from inshore to offshore (Fig. 4.3, top row) while bottom temperatures (Fig. 4.3, middle row) generally decline from inshore to offshore. Mean decadal surface temperature in the South Atlantic decreased by ~ 0.3 ºC from the 1950s to the 1980s and then increased by ~ 1 ºC from the 1980s to the 2010s (Fig. 4.3, bottom). Mean decadal bottom temperatures showed a similar pattern, decreasing by ~ 0.6 ºC from the 1950s to the 1970s and increasing by ~ 1 ºC from the 1980s to the 2010s.

4.4 Florida Current transport

The Florida Current (FC) is a strong boundary current that represents the beginning of the Gulf Stream stretching from the Florida Straits north to Cape Hatteras, NC. The Gulf Stream provides the bulk of the northward heat and salt transport of the upper limb of the Atlantic Meridional Overturning Circulation (AMOC) and the western boundary component of the horizontal subtropical gyre, and plays an important role in climate variability in the North Atlantic Ocean. Numerical and analytical models suggest the annual cycle of the FC is driven by local winds over the Florida Straits [69, 70]. Since local winds are part of the large-scale atmospheric circulation over the North Atlantic, the annual cycle is likely modulated by the North Atlantic Oscillation (NAO) [71]. It has been hypothesized that NAO-modulated changes in wind stress curl over the North Atlantic and forces Rossby waves that propagate westward and affect the FC transport [72].

The FC transport has been measured since 1982 using submarine telephone cables extending from South Florida to the Bahamas (near 27º N latitude) [73]. The cables induce voltage differentials that are linearly proportional to the full-water-column transport of the current. The cable measurements provide daily estimates of the FC transport, which are routinely (14 – 16 times per year) calibrated with ship-borne measurements by dropsondes and lowered acoustic Doppler current profilers.

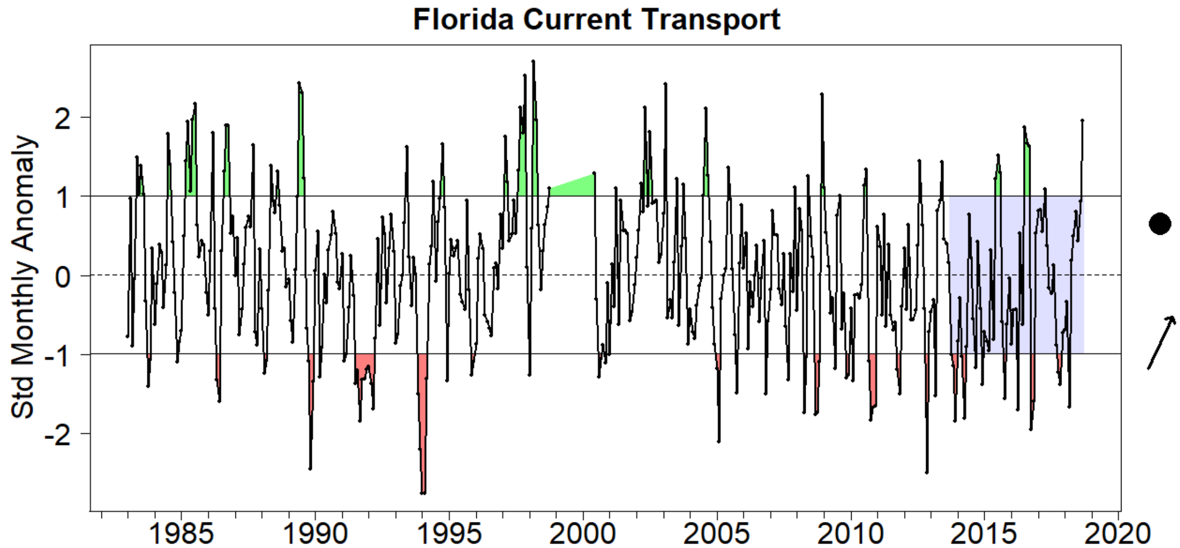


Figure 4.4. Standardized monthly anomalies of total current transport across the Florida Straits (27 °N latitude). A data gap exists from November 1998 through May 2000.

The mean (± standard deviation) transport of the FC over the last 32 years was 31.8 ± 3.4 Sv (1 Sv = 10^6 m^3 s^{-1}) (Fig. 4.4). The majority of the variability (~ 70%) is at periods less than annual, while the annual, interannual, and longer time scales each represent ~ 10% of the total variance [74]. Over the last five years, FC transport has been mostly within one standard deviation of the long-term mean with a slight increasing trend.

4.5 Gulf Stream position

The Gulf Stream (GS) is the western boundary current of the North Atlantic subtropical gyre, which plays an important role in regional and global climate via the poleward transport of heat and nutrients [75, 76, 77]. The high temperature water characteristic of the GS originates in the Florida Straits downstream of the Loop Current, and then separates from the shelf and upper slope and veers seaward near Cape Hatteras. The cross-shelf position of the GS influences coastal circulation, shelf break upwelling, and deep-ocean nutrient delivery and primary productivity, as well as the larval dispersal and spatial distributions of fish populations [78, 79, 80, 81] (Fig. 4.5a).

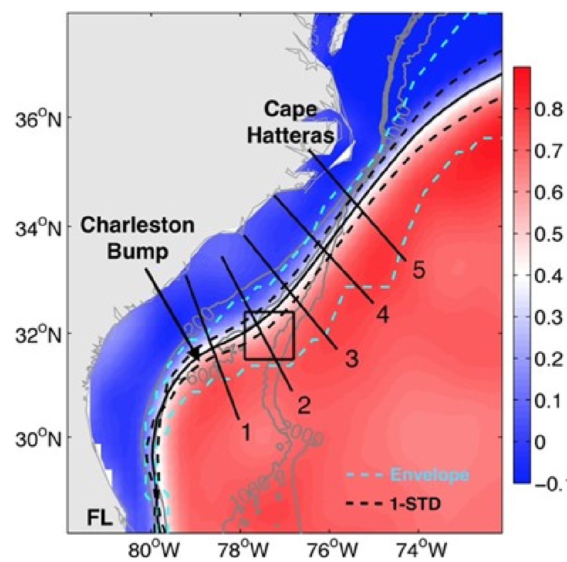


Figure 4.5a. Map showing the mean path of the Gulf Stream (solid black), one standard deviation (dashed black), and envelop (dashed cyan) from 1993-2013 taken from [81].

Monthly variations in GS position based on an extension of data reported in [81] are shown in Fig. 4.5b. The path of the GS in the South Atlantic Bight can be characterized as strongly deflected offshore or weakly deflected (either inshore or offshore). Offshore deflections occur relatively infrequently (approximately 26% of the time) and are more common in winter when water transport through the Florida Straits is relatively low. The largest offshore deflections over the last 25 years occurred in 2009 and 2010, and were associated with large GS offshore meanders. Over the last five years of the time series, the GS has shifted from being mostly offshore of its mean position (2014 – 2015) to either deflected inshore or near its long-term mean position (2016 – 2018).

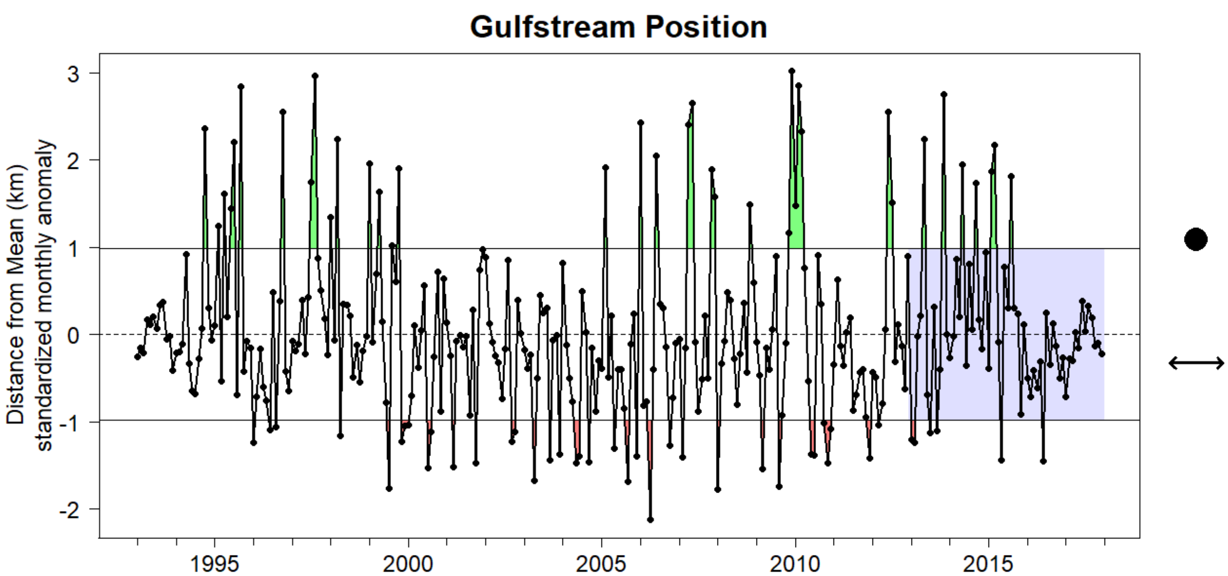


Figure 4.5b. Monthly Gulf Stream position relative to its mean path along transect 2 (See Fig. 4.5a). Positive and negative values indicate the offshore and onshore direction, respectively.

4.6 Upwelling

Wind-driven upwelling of cold, deep ocean water is an important mechanism that advects nutrients onto the U.S. South Atlantic continental shelf [82, 83]. Persistent southwesterly winds (blowing toward the north) primarily occur in the spring and summer months and promote upwelling favorable conditions on the continental shelf [84]. Event-scale intrusions of cold nutrient-rich waters onto the shelf are more likely during periods of upwelling-favorable winds and when the Gulf Stream is in a relatively onshore position ([63], Fig. 4.5b). In the U.S. South Atlantic, upwelling is particularly important in the southern (i.e., near Cape Canaveral) and northern (i.e., near Cape Hatteras) regions due to divergent and convergent isobaths in these respective areas [84]. Upwelled water from the shelf-slope break often reaches the mid shelf (20 – 40 m depth) and sometimes the inner shelf (0 – 20 m depth), enhancing primary productivity over large regions of the South Atlantic Bight. Upwelling also occurs inside cyclonic small-scale eddies that form on the western edge of the Gulf Stream due to interactions between the Gulf Stream and the continental shelf-slope break. These upwelling cells are short-lived, compared to wind-driven coastal upwelling [85].

An upwelling index (i.e., the offshore component of the Ekman transport) was computed from a wind time series collected at Grays Reef National Marine Sanctuary [86] following the method described in [87] (Fig. 4.6). The index can be considered a proxy for the intensity of upwelling favorable wind conditions. Further information on the data and the index methodology can be found in [63]. Periods of most intense upwelling over the 17-year time series occurred in winter of 2010 and 2011, with generally upwelling favorable conditions during at least some months through the mid-2010s. Since 2016, upwelling intensity in the region has been low and downwelling favorable winds have predominated.

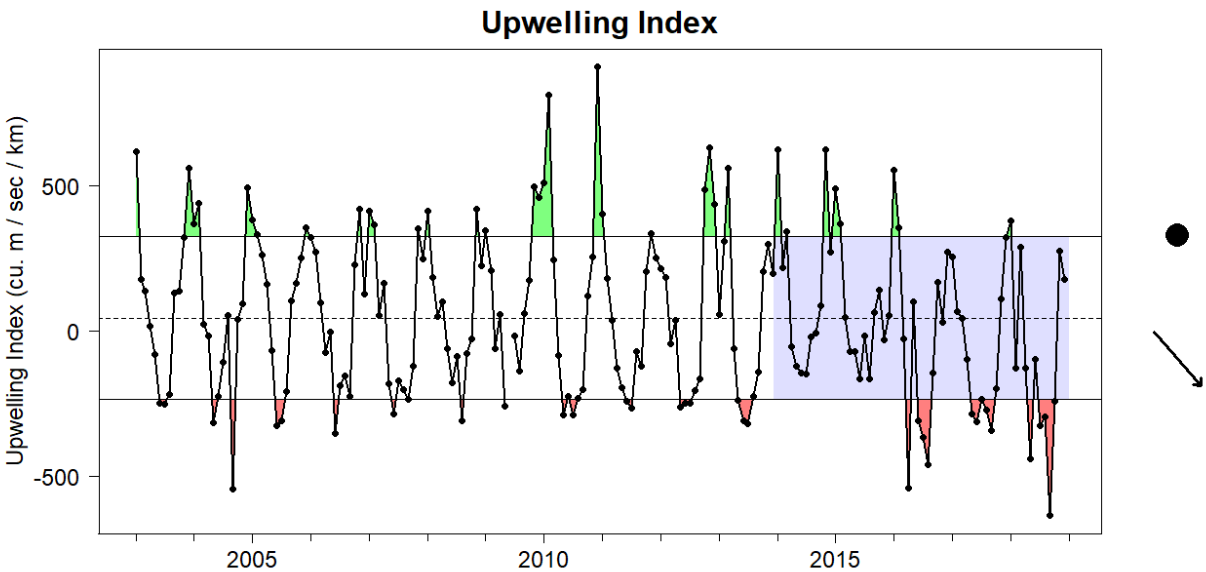


Figure 4.6. Monthly upwelling index calculated from buoy wind data (Gray Reef Station 41008) from 2003 to 2019.

4.7 Coastal salinity

Coastal salinity is an important environmental parameter that marks the transition from coastal (or estuarine) regions to the open ocean, and influences many biological processes, such as the distribution of estuarine and nearshore organisms [88, 89]. Salinity of coastal habitats is a function of riverine freshwater input, evaporation, precipitation, horizontal advection, and vertical mixing. The United States Geological Survey (USGS) reports a Coastal Salinity Index (CSI) from data collected at 17 monitoring sites in North Carolina, South Carolina, and Georgia (Fig. 4.7a) [90]. The data are a standardized probability index classified into groups ranging from exceptionally fresh conditions (low values) to exceptionally saline conditions (high values) for each of the 17 stations [91, 91]. The CSI is intended to characterize coastal drought conditions, monitor salinity, and improve understanding of salinity changes on freshwater and coastal ecosystems.

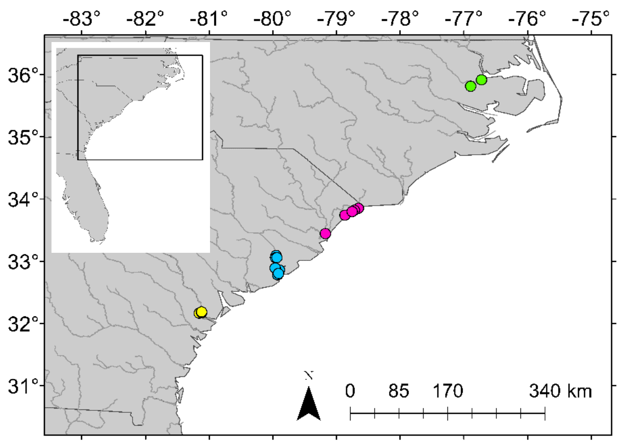


Figure 4.7a. Map of monitoring stations used to develop the coastal salinity index (CSI).

The proportion of measurements within various salinity categories for each year and geographic area was calculated as a regional indicator of coastal salinity (Fig. 4.7b). Annual time series were developed for the headwaters of the Roanoke River (NC), Myrtle Beach (SC), Charleston (SC), and Savannah (GA). The most saline conditions occurred in 2002, particularly in the northern areas, with high salinity years also occurring in the mid-2000s and early-2010s throughout the region. Increasingly fresh water conditions have occurred since 2013, consistent with increasing precipitation in the region (see Fig. 4.1).

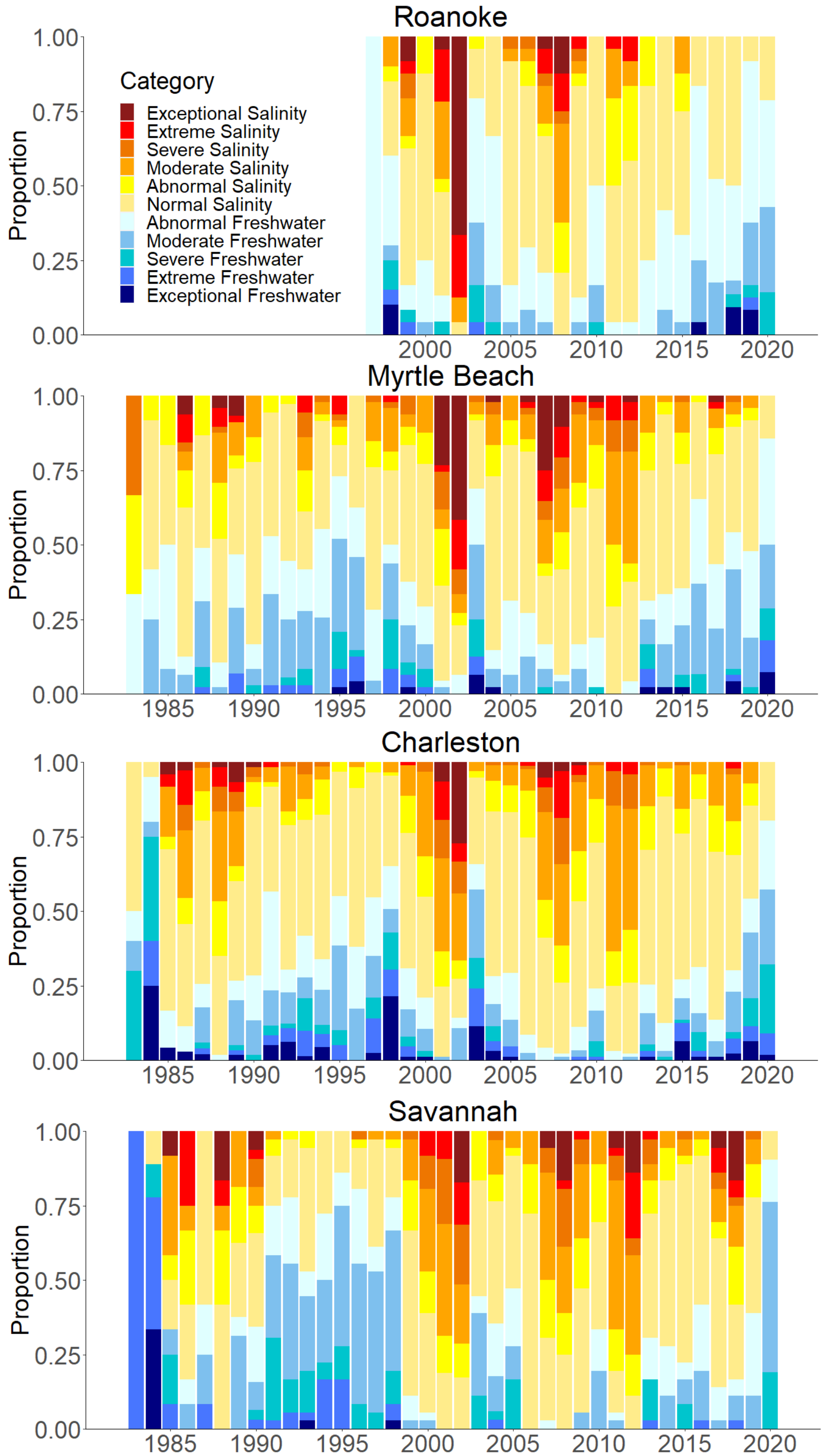


Figure 4.7b. Proportion of measurements within each coastal salinity classification group for North Carolina (Roanoke), South Carolina (Myrtle Beach, Charleston), and Georgia (Savannah).

4.8 Stream flow

There are more than 2.4 million kilometers of streams in the southeastern U.S. These streams harbor a high diversity of aquatic organisms, including more at-risk species (e.g., fish, mussels, crayfish) than any other region in the nation [93]. Stream flow in the southeast is driven primarily by precipitation, evapotranspiration, and spring discharge. Changing climate, land use, and increasing human populations have the potential to alter hydrologic conditions in the region. Low frequency variability in flow conditions of rivers and streams in the southeastern U.S. has been related to large-scale climate drivers (e.g., AMO, ENSO) and variable North Atlantic sea surface temperatures via effects on atmospheric moisture transport and precipitation [94]. Riverine inputs are an important driver of biogeochemical processes on the inner (0 – 20 m depth) and middle (20 – 40 m depth) continental shelf of the South Atlantic Bight, contributing to coastal stratification, the cross-shelf transport of nutrients, and coastal phytoplankton production [95].

Mean annual discharge data were obtained for the 360 streams entering coastal waters from North Carolina to northern Florida from the U.S. Geological Survey (USGS) National Water Information System (NWIS, [96]). Discharge data are reported in cubic feet per second for the water year (Oct 1 to Sep 30). Annual time series were converted to z-scores by subtracting each annual mean from the time series mean and dividing by the time series standard deviation. These standardized time series were then averaged across streams for each state (NC, SC, GA, FL east coast) in the Southeast region. It should be noted time series length varied considerably among streams, with some stream gauges in place since the 1930s. The number of gauged streams in the region increased from about 100 in 1950 to about 350 in 2010 and has varied from 310 to 357 since 2000. No adjustments were made for the number of streams that contributed to state-level average annual stream discharge.

Average annual discharge to South Atlantic coastal waters varied from 1950 – 2019 with no strong temporal trend (Fig. 4.8). Georgia and South Carolina showed very similar patterns in riverine discharge and were less similar to Florida and North Carolina. Stream discharge was generally high across all states from the mid-1950s to the mid-1970s, low in the 1980s and again in the mid-2000s, and has remained near the long-term mean in recent years.

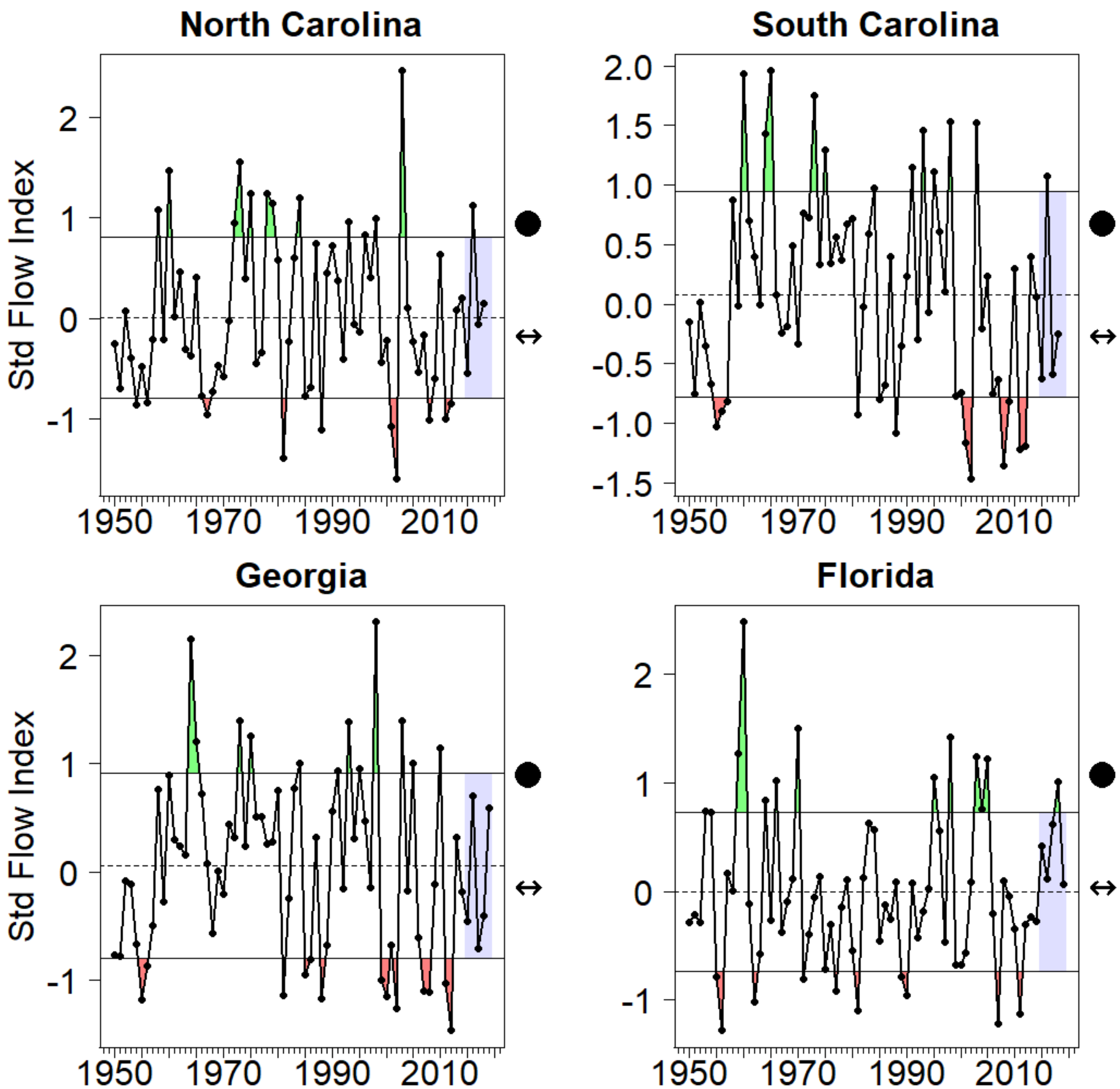


Figure 4.8. Mean annual relative stream flow from gauge monitoring data for 360 rivers and streams entering coastal waters of the U.S. South Atlantic region.

4.9 Nutrient loading

Water quality assessments indicate nitrogen and phosphorus concentrations in rivers and streams have either increased or shown little change since the 1990s for most regions of the U.S. [97]. Excessive nutrient inputs can lead to coastal eutrophication, defined as the increased rate of primary production and accumulation of organic matter, and results in a number of other ecosystem changes [98]. Increased nutrient loads are an important driver of harmful algal blooms, hypoxia, and coastal ocean acidification that have negative consequences for marine ecosystems and living resources [99, 100].

Coastal waters of the South Atlantic are the downstream terminus of multiple, rapidly developing watersheds that transport nutrients, primarily nitrogen and phosphorus, from agricultural and urban activities to the nearshore coastal zone. Annual time series of nutrient loadings from rivers and streams to coastal waters of the South Atlantic are not currently available. Therefore, the total nitrogen (TN) and total phosphorus (TP) load to basin outlets at the coast in 2002 and 2012 for the eight major watersheds in the South Atlantic region are reported (Fig. 4.9). These data were taken from two reports developed under the National Water Quality Assessment Program

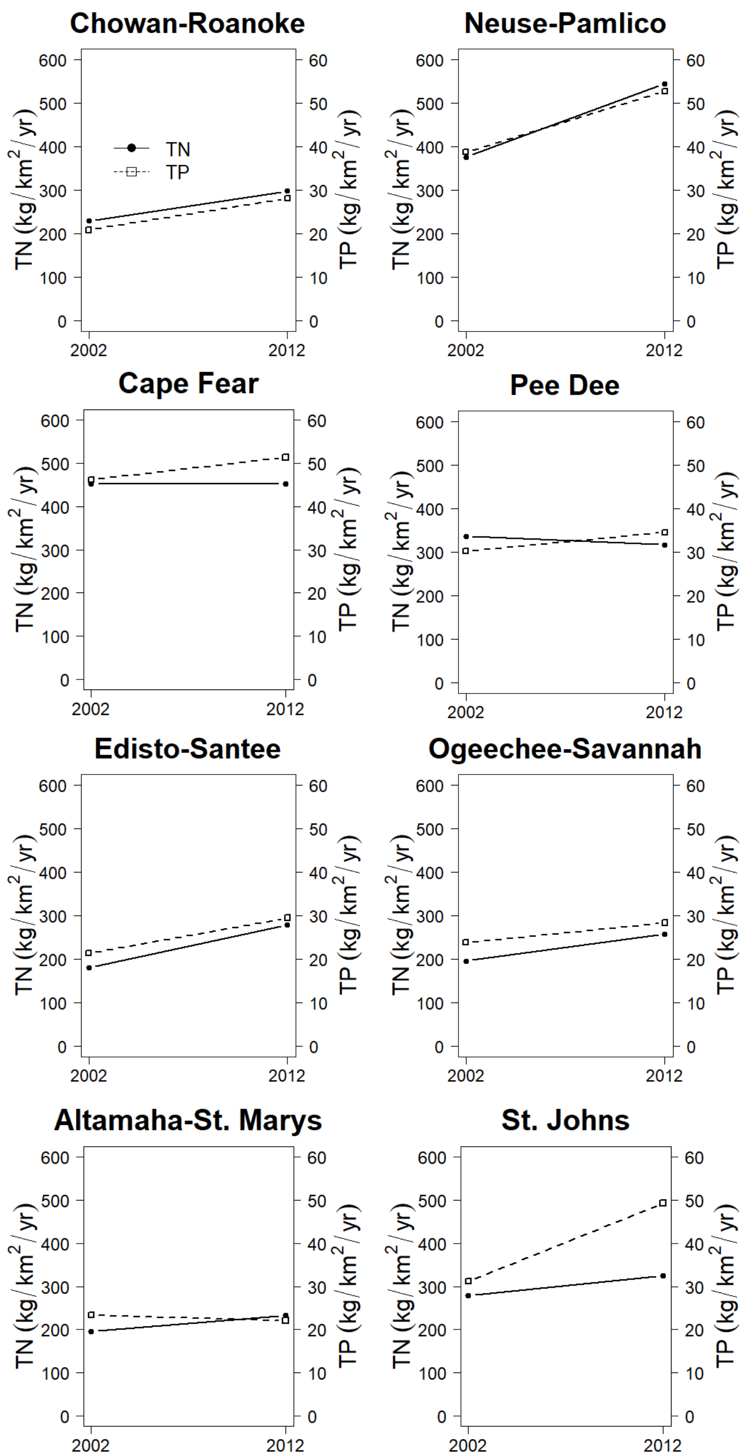


Figure 4.9. Total nitrogen (TN, left axis) and total phosphorus (TP, right axis) loading to coastal waters in the U.S. South Atlantic in 2002 and 2012. River systems are arranged from north to south along rows.

[101, 102], a program established by Congress in 1991 to address past and future changes in water quality in the U.S. Loading estimates were based on the Spatially Referenced Regression On Watershed (SPARROW) model, a hybrid statistical and process-based mass-balance model that estimates TN and TP loads based on constituent sources as well as watershed and stream characteristics [101].

On average, TN and TP loadings to coastal regions of the South Atlantic increased 22% and 20%, respectively, between 2002 and 2012 (Fig. 4.9). The magnitude of increase varied from -5.7 to 44.9% among basins for TN and from -5.5 to 58.3% for TP, with six of the eight major watersheds showing increases in TN and seven of the eight watersheds showing increases in TP.

4.10 Precipitation and drought

The hydroclimatology of the Southeastern U.S. is characterized by hot summers, mild winters, and high annual precipitation (mean ~ 50 inches annual rainfall) interspersed with periods of moderate to severe drought [103]. Variability in precipitation in the U.S. Southeast has increased since the 1960s with more frequent, longer-duration dry periods and infrequent but more extreme precipitation events [104]. Annual to decadal patterns in precipitation are related to variability in ENSO, NAO, and the Atlantic Warm Pool through the influence of these climate drivers on atmospheric moisture transport into the region. Seasonal patterns are influenced by the Bermuda High, convective activity in the Caribbean Sea, and interactions with the tropical Pacific-Atlantic sea surface temperature contrast [105].

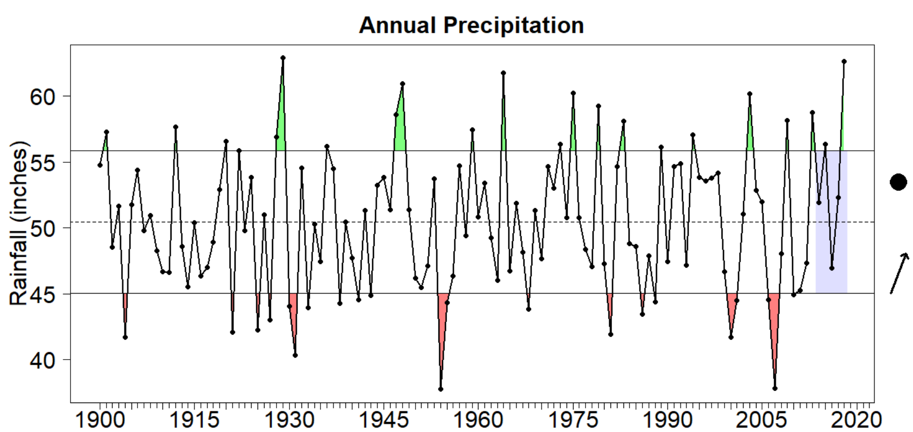


Figure 4.10a. Total annual precipitation in the U.S. Southeast region from 1900 – 2020.

Patterns in precipitation and drought are important drivers of variability in stream flow and nutrient loading to coastal ecosystems [106], and have implications for crop production [107], incidence of wildfires [108], and municipal and agricultural water supplies in the region [109, 110]. Future projections from General Circulation Models (GCMs) under anthropogenic forcing suggest an increase in precipitation and evaporation due to an increase in air temperatures in the Southeast region [111]. Peninsular Florida is projected to become drier under future climate while the rest of the Southeast is expected to become wetter [112].

For precipitation and drought indicators, the Southeastern U.S. was defined based on hydrological units, and encompasses Alabama, Florida, Georgia, South Carolina, North Carolina, and Virginia. Monthly precipitation data from rain gauges across the Southeastern U.S were downloaded from The Southeast Regional Climate Center [113] and summed across months to generate an annual estimate of land-based precipitation in the Southeast region. Drought data for the Southeastern U.S. were downloaded from The Drought Monitor [114], a site jointly managed by the National Drought Mitigation Center (NDMC) at the University of Nebraska-Lincoln, the National Oceanic and Atmospheric Administration (NOAA), and the U.S. Department of Agriculture (USDA). Areas of drought within the Southeast region are identified based on drought intensity, which is determined from a number of indicators (e.g., rainfall, stream flow, precipitation, soil moisture) categorized into five levels of severity. The average weekly drought severity was computed as a weighted average of the areas within each drought category. Weekly data from 2000 to 2018 were then summed and renormalized to develop a monthly indicator of drought severity for the Southeastern U.S.

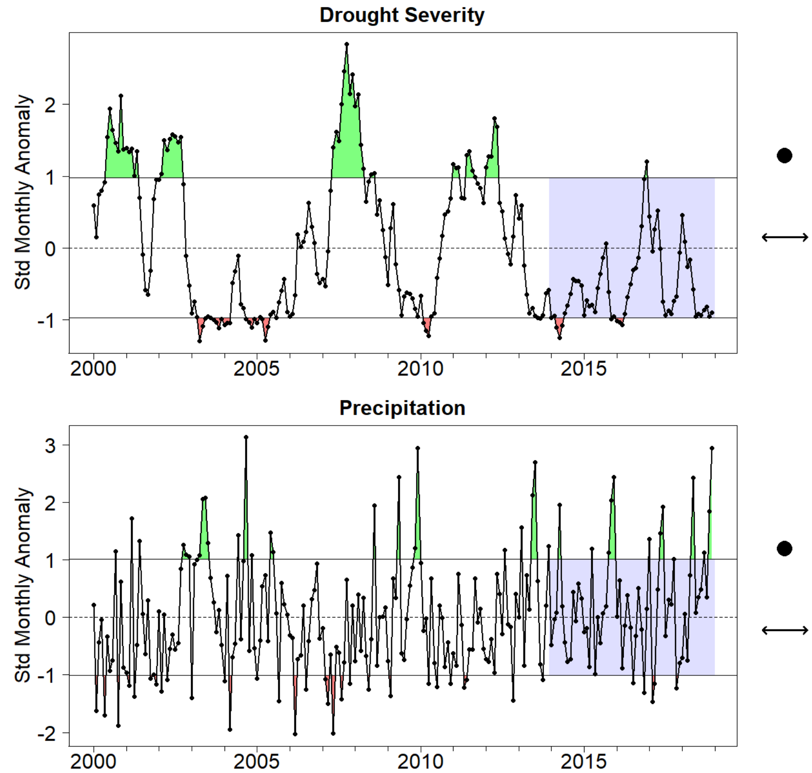


Figure 4.10b. Standardized monthly anomalies of drought severity (top panel) and precipitation (bottom panel) from 2000-2018.

Annual precipitation has shown high variability over the period of record (1900 – 2018) with little long-term trend (Fig. 4.10a). The year 2018 was the wettest on record since the 1920s for the continental U.S. and this is reflected in the high precipitation that occurred in the Southeast region in this year. Regional drought severity was high in the early and late-2000s and has been relatively low since the early 2010s (Fig. 4.10b, top panel). Monthly precipitation (Fig. 4.10b, bottom panel) generally showed an inverse relationship with the monthly drought severity index, with relatively high precipitation and low drought severity in the most recent years.

4.11 Sea level rise

Sea level rise is a global phenomenon primarily related to increased water volume in the oceans from melting land-based ice and expansion of seawater due to warming. Sea level rise along the U.S. South Atlantic coast has been comparable to or greater than the long-term (since 1900) global average rate of 1.7 mm per year [115]. In recent years (2011 – 2015), sea level rise has accelerated somewhat, with the highest annual rate of sea level rise projected for South Florida and the Cape Hatteras regions [116]. In addition to its impact on infrastructure and coastal communities, sea level rise is projected to have negative effects on a number of South Atlantic species, particularly those dependent on marsh and coastal beach habitat (marsh birds, some fishes and small mammals, [117]).

Data on long-term trends in local relative sea level (RSL) along the U.S. South Atlantic coast are available for 15 stations extending from Duck, NC to Key West, FL operated by the National Water Level Observation Network to collect detailed information on tides and local sea levels [118]. RSL is based on tide gauge measurements with respect to a local fixed reference on land, and is influenced by both sea level rise and vertical land movement.

Data on RSL were averaged across ten of the stations extending from Duck, NC to Key West, FL that had the longest and most consistent time series (at least 40 years; Fig. 4.11 map). Not all stations contributed equally to the average in each year due to variation in sampling at each station. Sea level trends ranged from a low of 2.01 to a high 4.55 mm/year across the U.S. South Atlantic stations. The station-averaged RSL is shown in Fig. 4.11. The average rates of sea level

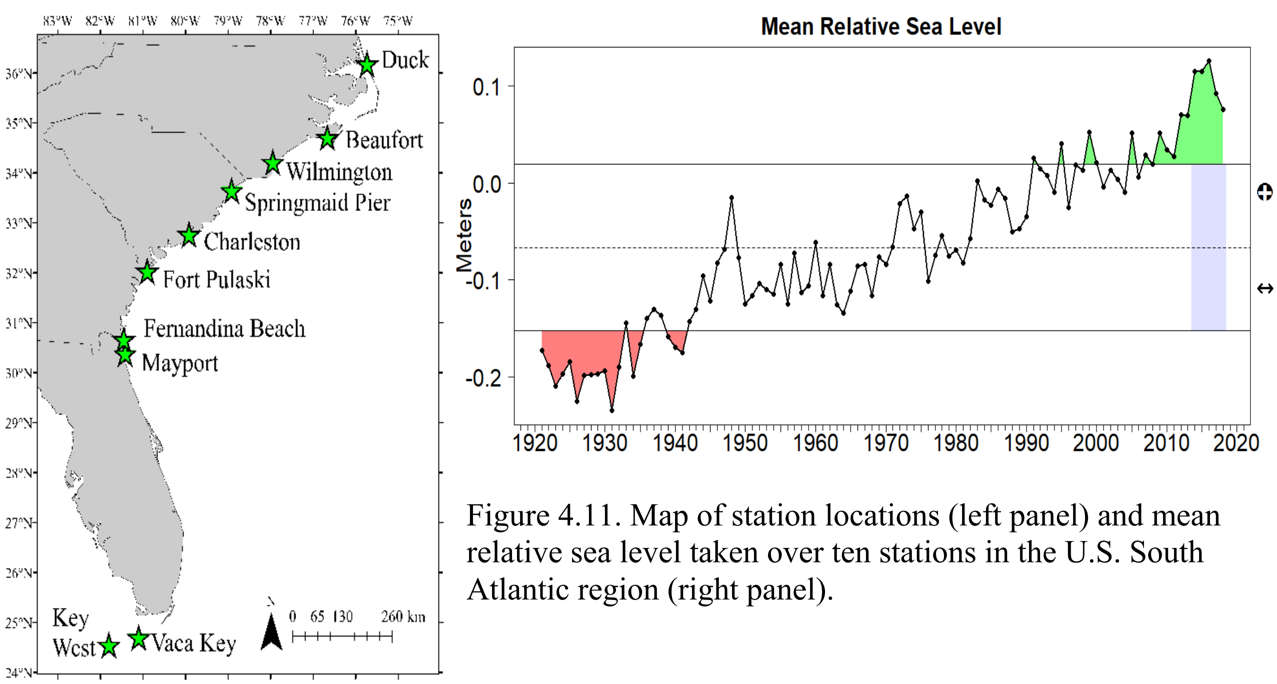


Figure 4.11. Map of station locations (left panel) and mean relative sea level taken over ten stations in the U.S. South Atlantic region (right panel).

rise for the South Atlantic region over the last ~ 100 years is about 2.6 mm/year. The rates of reported sea level rise range from 1.76 to 5.6 mm/year among sites along the entire Atlantic coast (Maine to Florida), 2.1 to 9.1 mm/year along the U.S. Gulf coast, and -1.7 to 4.6 mm/year along the U.S. Pacific coast [115].

4.12 Storms and hurricanes

Hurricanes and major storms are a significant hazard to coastal communities due to their high winds, rainfall, and coastal storm surge. Land-falling hurricanes in the continental U.S. have accounted for approximately 2 trillion USD in damages (normalized to 2018 dollars), since the early 1900s [119]. Long-term trends in Atlantic hurricane activity generally follow the phases of the Atlantic Multidecadal Oscillation (AMO), which influences storm patterns in the broader Northwest Atlantic basin [120]. The Accumulated Cyclone Energy (ACE) index is a measure of storm activity based on wind speeds that considers both the number and strength of named storms [121]. The ACE index was calculated for the U.S. South Atlantic Exclusive Economic Zone (EEZ, shore to 200 nautical miles offshore) using 6-hour storm tracks from the National Climate Data Center's International Best Track Archive for Climate Stewardship (IBTrACS) database [122, 123]. The index shows little long-term trend in hurricane activity since the 1960s, with particularly high activity in 1963, 1984, 1999, and 2005 (Fig. 4.12a). The year 1999 was a particularly active season in the U.S. South Atlantic with three land-falling hurricanes in one month in North Carolina [124]. The year 2005 was also an active season with numerous storms affecting Southeast Florida, including Hurricane Katrina.

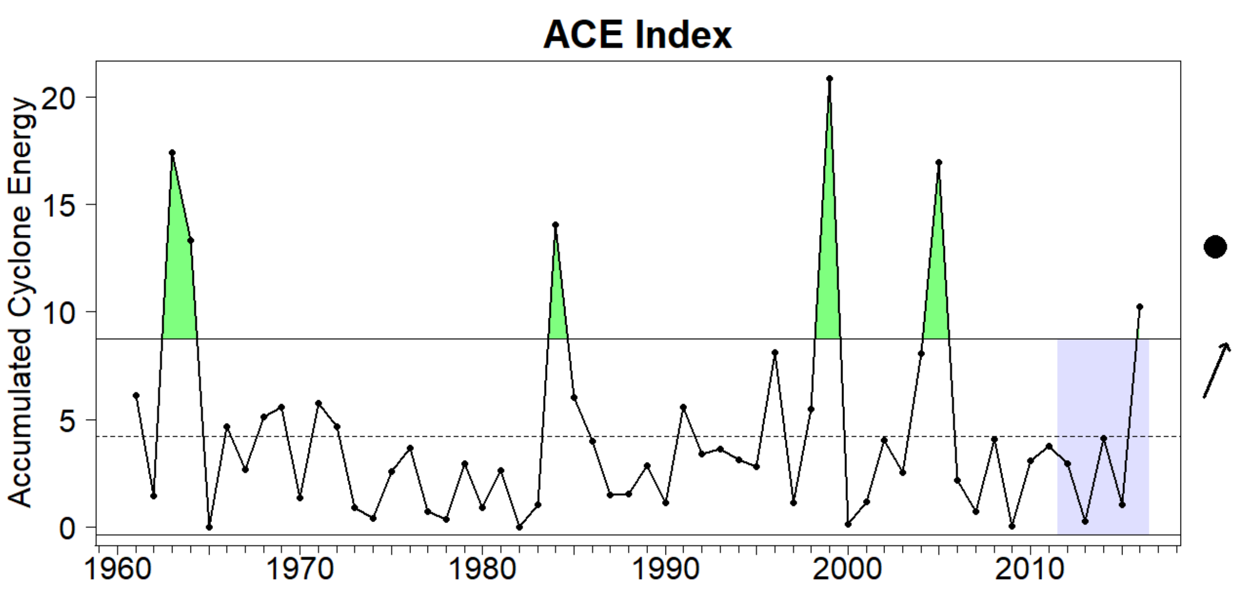


Figure 4.12a. The Accumulated Cyclone Energy (ACE) index calculated for the U.S. South Atlantic Exclusive Economic Zone.

The number of landfalling hurricanes in the South Atlantic states since 1900 (NC through east coast of FL) was tabulated by decade as another indicator of storm activity in the region [125]. The number of landfalling hurricanes was highest in the 1940s and 1950s and lowest in the 1960s and 1970s (Fig. 4.12b). The 1990s and 2000s were a period of increased hurricane activity, consistent with patterns in the ACE index. The 2020 Atlantic hurricane season was extremely active and destructive with 30 named storms and total damage in the U.S. of $42 billion [126].

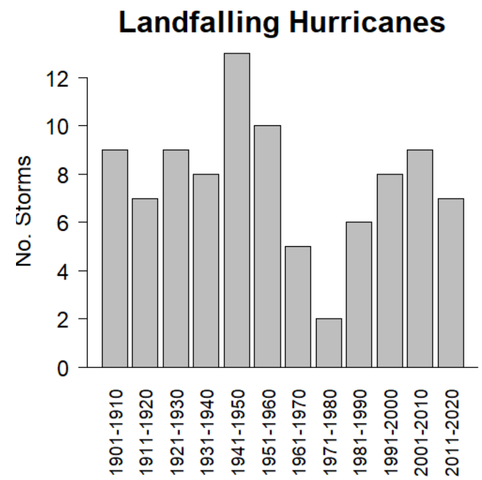


Figure 4.12b. Number of landfalling hurricanes by decade in the U.S. South Atlantic region.

4.13 Ocean acidification

The ocean removes approximately 26% of anthropogenic carbon emissions and is important for regulating the amount of CO_2 in the atmosphere [127]. Ocean acidification is mainly caused by the absorption of atmospheric CO_2 [128], whereas coastal acidification may be caused by a combination of *in situ* biological, thermal, and advective processes, some of which occur across the land-sea interface.

A recent study of acidification in U.S. South Atlantic coastal waters synthesized 26 years of cruise-based and moored autonomous observations [129]. The carbonate dynamics in the South Atlantic are driven primarily by coastal eutrophication and organic matter decay on the inner shelf (< 30 m depth) and in marsh regions, while temperature and atmospheric absorption are the primary influences on the middle (30 – 60 m depth) and outer (60 – 200 m depth) shelf. On multi-decadal time scales (1991 – 2017), the CO_2 partial pressure (pCO_2) is increasing at rates ranging from 3.3 ± 0.3 µatm y^{-1} on the outer shelf to 4.5 ± 0.6 µatm y^{-1} on the inner shelf [129]. Over the 26 years period, a sea surface temperature increase of 0.07 ± 0.3 °C y^{-1} contributed 1.1 µatm pCO_2 to the overall trend on the middle and outer shelf, with the remaining increase due to atmospheric absorption and other processes (2 to 3 µatm pCO_2). In the coastal region (< 15 m depth) and inner shelf, there was no temperature increase, and the atmospheric component contributed ~ 2 µatm y^{-1}, with the remaining 1 to 3 µatm increase likely due to biological processes. While the effect of rising global temperatures on seawater acidification has not yet been fully identified, increased temperatures generally result in higher pCO_2 concentrations (and lower pH), suggesting a compounding effect from coastal influences and rising temperatures in the U.S. South Atlantic and other marginal bodies of water.

Buoy data collected at Gray's Reef National Marine Sanctuary [86] show an increasing trend in air pCO_2 in the overlying atmosphere from 2006 – 2018 (Fig. 4.13, top). A similar trend is not evident in surface waters, though pCO_2 has been mostly above the long-term mean since 2015 (Fig. 4.13, bottom). Model estimates of pH derived from analyses reported in [129] indicate a declining trend in pH on the inner and outer continental shelf in the SAB of 0.003 to 0.004 pH units per year, respectively. Acidification can have negative effects on coastal resources, particular for larval stages and for sessile species (e.g., corals) but these effects have not been assessed for coastal waters in the U.S. South Atlantic region [130].

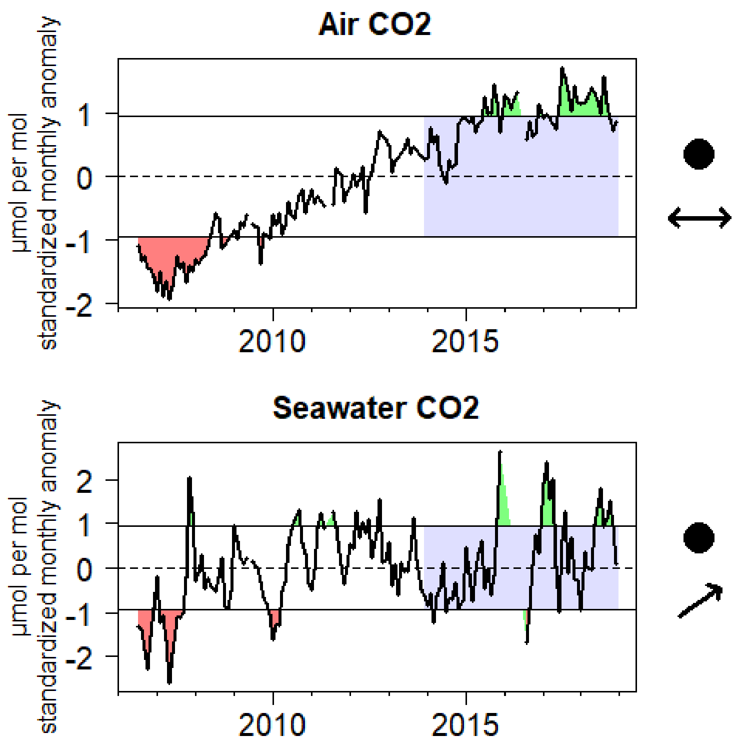


Figure 4.13. pCO_2 in the atmosphere (top) and underlying seawater (bottom) from buoy data at Gray's Reef National Marine Sanctuary from 2006-2018.

5. HABITAT STATES

5.1 Wetlands and forests

Wetlands and forested areas provide a number of ecosystem services, including enhanced water quality, fish and wildlife habitat, carbon storage, and flood protection [131]. Shallow estuarine marsh habitat has long been recognized as a primary nursery ground for a number of ecologically and economically important fish, invertebrates, birds, and mammals [132, 133, 134]. The Southeast is also considered the 'wood basket' of the U.S., producing half of the country's domestic timber supply, and is a biodiversity hotspot, with the highest native species richness of any temperate region in the continental U.S. [135, 136].

Data from the Coastal Change Analysis Program (C-CAP) were used to quantify changes in wetland and forest cover from 1996 to 2010 [137]. Wetlands represent a relatively large proportion of the total land area along the U.S. South Atlantic coast, making up approximately

35% of the total land cover within coastal counties in the region. This includes both palustrine wetlands (inland wetlands that lack flowing water and are non-tidal) and estuarine wetlands containing varying amounts of forested, shrub/scrub, and emergent vegetation.

Overall, total wetland cover (pooled over vegetative types) in the U.S. South Atlantic region declined by 323 square miles (mi^2), or about 1.5 %, from 1996 to 2010 (Fig. 5.1, right panel). However, this decline masks considerable variability among vegetation types, with palustrine forested wetlands declining by about 17% (~ 5000 mi^2) while palustrine scrub and emergent wetlands have increased by about 28% (~ 2000 mi^2). Consistent with the loss of palustrine forested wetlands, total forest cover (deciduous, evergreen, mixed forests) have also declined by ~ 17% from 1996 to 2010 (Fig 5.1, left panel). Forest and wetland losses are attributable in part to urban development and related infill needs (see Fig. 9.2). The degree of loss varies geographically, however, with the highest total wetland losses occurring near Myrtle Beach, SC and the Orlando, FL metropolitan area, and the highest forest loss occurring in Myrtle Beach and near Charleston SC, and the NC Albemarle-Pamlico region.

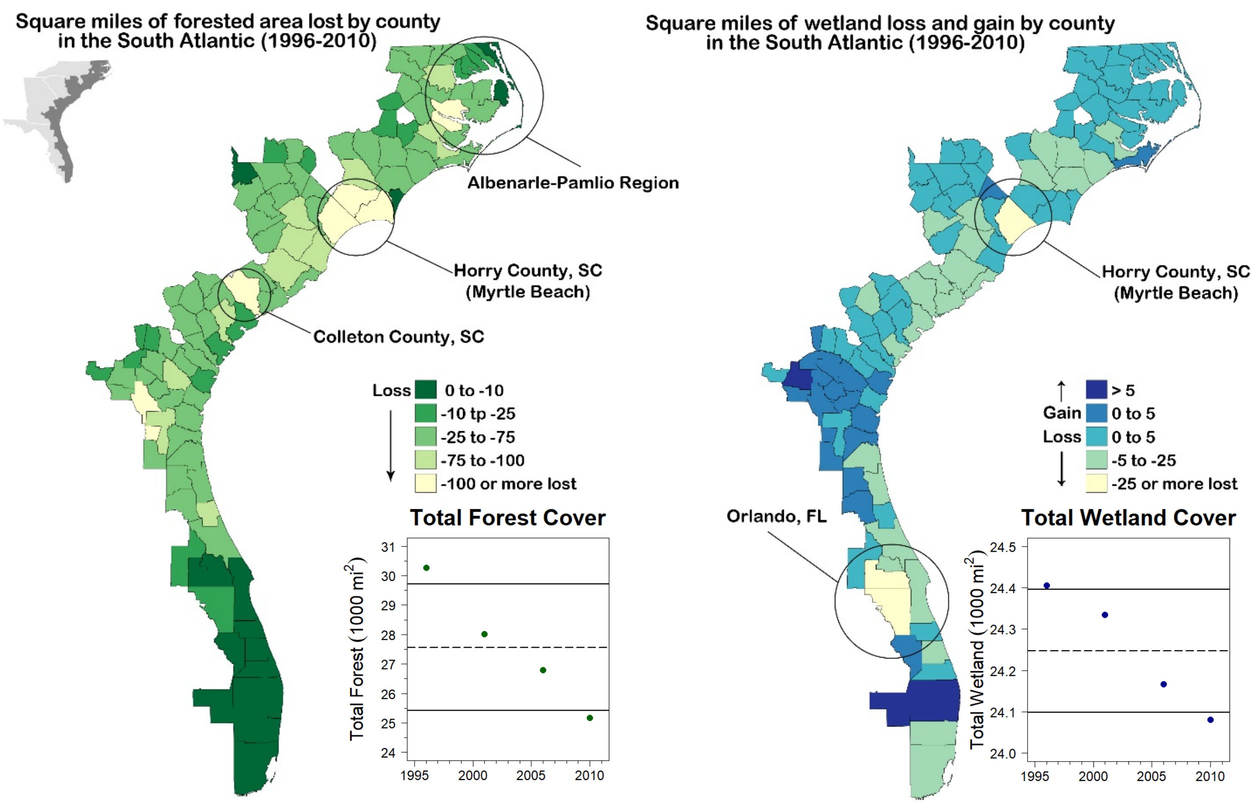


Figure 5.1. Maps of wetland (right) and forest (left) loss or gain across coastal counties in the U.S. South Atlantic in 2010. Graphs show the total loss by 5-yr increment from 1996 – 2010. Dashed line is the mean cover and solid lines represents the mean ± one standard deviation.

5.2 Submerged aquatic vegetation (SAV)

In coastal habitats of the U.S. South Atlantic, submerged aquatic vegetation (SAV) is comprised primarily of seagrasses, which provide a number of ecosystem services, including support for economically important fisheries, nutrient cycling, sediment stabilization and carbon sequestration and storage [138, 139, 140]. A global synthesis reported that the total amount of seagrass habitat is declining at an accelerating rate, and 29% of the known areal extent of seagrass has disappeared since measurements began [140]. Along the U.S. South Atlantic coast, seagrasses are prominent in North Carolina and Florida, but extremely limited along the South Carolina and Georgia coasts due to the generally high turbidity and limited light penetration [141]. Both eelgrass (*Zostera marina*), a temperate species at its southern limit, and shoalgrass (*Haloduili wrightii*), a subtropical species at its northern limit, are found in the Albermarle-Pamlico Sound region of North Carolina, while Florida coastal waters are dominated by turtle grass (*Thalassia testudinum*). Florida harbors the highest diversity of seagrasses with eight documented species.

No region-wide assessments of seagrass coverage have been conducted for the U.S. South Atlantic coast. Multiple site-specific assessments have been conducted in Florida and in North Carolina. Changes in the areal extent of seagrass in the Albemarle-Pamlico Estuarine System (APES) of North Carolina were taken from data collected via aerial surveys in 2006 – 2007 and 2013 [142]. Areal coverage of seagrasses along the east coast of Florida were taken from the Seagrass Integrated Mapping and Monitoring (SIMM) program [143]. Total seagrass coverage in NC declined by 5.6 % between 2006 – 2007 and 2013, with regional declines in the state ranging from 0.5 to 1.6% per year (Table 5.2). The largest declines occurred in the southern region (Core and Bogue Sounds). Multiple systems on the east coast of Florida have shown increases in seagrass coverage ranging from 0.3 to 6.3% per year (Table 5.2). However, these increases generally followed large losses in 2010 that resulted from storm runoff, poor water clarity, and widespread algal blooms, especially in the northern Indian River Lagoon system. Rates of change over time should be interpreted with caution as mapping methodologies may have changed.

Table 5.2. Seagrass areal coverage and percent change per year for North Carolina and Florida. For North Carolina locations, “Northern NC” ranges from the U.S. Highway 64 Bridge at Roanoke Island to Hatteras Inlet, “Central NC” from Hatteras Inlet to Ophelia Inlet, and “Southern NC” from Barden Inlet at Cape Lookout to Bogue Inlet [142].

Location	Earlier year(s)	Hectares	Recent year(s)	Hectares	% change/yr^{-1}
Northern NC	2006-2007	28,676	2013	26,889	-0.5
Central NC	2006-2007	9,766	2013	9,501	-0.4
Southern NC	2006-2007	2,367	2013	2,119	-1.6
Northern Indian River Lagoon, FL	2013	17,435	2015	19,631	6.3
Southern Indian River Lagoon, FL	2011	2,998	2013	3,267	4.5
Lake Worth Lagoon, FL	2001	667	2007	683	0.4
Biscayne Bay, FL	1992	62,252	2004-2005	64,492	0.3
Marquesas, FL Keys	1992	346,555	2006/2011	376,474	0.5
Dry Tortugas	NA	NA	2006/2010	3,724	NA

5.3 Oyster reefs

Eastern oysters (*Crassostrea virginica*, hereafter oysters) are distributed throughout the South Atlantic and Gulf of Mexico, where they build large reef structures in shallow estuarine waters [144]. These reef habitats provide a range of ecosystem services, including biogenic habitat for other species [145], structural stability and shoreline protection [146], and bio-filtration of surrounding waters [147]. Oysters have also been fished for hundreds of years and support important commercial, recreational, and subsistence fisheries in the region [148, 149]. Overexploitation, human modifications of coastal hydrology, increased prevalence of disease, and water quality degradation have led to significant declines in oyster populations [150, 151]. Based on a synthesis of data from estuaries throughout the United States, the spatial extent of oyster habitat has declined by about 64% and oyster biomass has declined by 88% over the last 100 years [153], though few measurements were available for the U.S. South Atlantic region.

Systematic, region-wide surveys of oyster reef habitat do not currently exist for the U.S. South Atlantic coast. Below we report available estimates of the spatial extent of oyster reef habitat from published papers and reports and from publicly available state mapping programs (Table 5.3). Collection and analysis of these data typically occurred over different time periods and used different methodologies with varying degrees of validation and spatial coverage. Further, most oyster reef habitats have not yet been fully mapped and mapping efforts are ongoing in most states. As mapping efforts continue to develop, more rigorous quantification of the spatial extent and temporal variability of oyster reef habitat will be possible.

Table 5.3. Historical and contemporary oyster reef coverage in the U.S. South Atlantic region.

State	Location	Contemporary	Hectares	Historical	Hectares
NC	North Carolina coast	1990-2020	8929[c]		
	Pamlico Sound	2007-2012	1,040[a]	1886-1887	3320[b]
	New River	2010	202	1886-1887	126[b]
	Bogue Sound	2009-2010	1,030	1886-1887	666[b]
SC	South Carolina coast	2003-2006	2,230[d]		
	Broad River	2006-2008	622[b]	1890	51[b]
	Charleston Harbor	2006-2008	57[b]	1890	27[b]
	St. Helena Sound	2006-2008	401[b]	1890	48[b]
	Stono/North Edisto Rivers	2006-2008	199[b]	1890	115[b]
GA	Georgia coast	2012-2013	276[e]		
	Altamaha River			1891	81[b]
	Savannah River			1890-1891	62[b]
	Ossabaw Sound			1891	71[b]
	St. Andrew/St. Simons Sounds			1891	185[b]
	St. Catherines/Sapelo Sounds			1891	271[b]
FL	Florida east coast	2003-2015	1,162[f]		
	Biscayne Bay			2005-2006	0[b]
	Indian River			2007-2010	90[b]

[a]Theurakaulf et al. [152]
[b]Zu Ermgassen et al. [153]
[c] Deaton [154]; excludes subtidal oyster bottom in Pamlico Sound
[d]South Carolina Department of Natural Resources [155]
[e]Georgia Department of Natural Resources [156]
[f]Florida Fish and Wildlife Commission [157]

5.4 Coral demographics

Coral reefs are highly valuable ecosystems that provide numerous goods and services, including food, storm protection, and recreational opportunities [158]. The global economic value of ecosystem services provided by coral reefs has been estimated at 9.8 trillion annually [159, 160]. Many reef ecosystems are highly impaired due to overfishing, pollution, acidification, and rising temperatures [161, 162, 163]. Shallow coral reef formations in the U.S. South Atlantic occur only in Florida from St. Lucie Inlet to the Florida Keys (i.e., Florida Reef Tract) where they support thousands of jobs and contribute several million dollars annually to local economies [164, 165].

Data reported here were compiled from 2005 – 2018 from the Florida Fish and Wildlife Conservation Commission (FWC) Disturbance Response Monitoring (DRM) Program [166] and the National Oceanographic and Atmospheric Administration (NOAA) National Coral Reef Monitoring Program (NCRMP) [167]. The Southeast Florida region extends from Martin County to Miami-Dade County, the Florida Keys region includes Biscayne National Park and the Florida Keys within Monroe County to Key West, and the Dry Tortugas region includes the area within the Dry Tortugas National Park and the Tortugas Bank. Data on coral density (mean number of coral colonies per m^2), disease prevalence (mean number of colonies with disease present), and mortality (mean % of colony surface covered with non-living skeletal structure) were collected from benthic demographic surveys conducted at randomly selected sample sites. All adult coral colonies (≥ 4 cm) were identified to genus or species, measured, and assessed for the presence or absence of coral disease. Site-level means were calculated from species level data and then aggregated and weighted by strata area to compute estimates for each region.

The density of coral colonies has increased slightly in the Florida Keys and Dry Tortugas but declined slightly in Southeast Florida, though with considerable annual variability (Fig. 5.4, left). Incidence of disease has been variable and without trend in Southeast FL, while disease

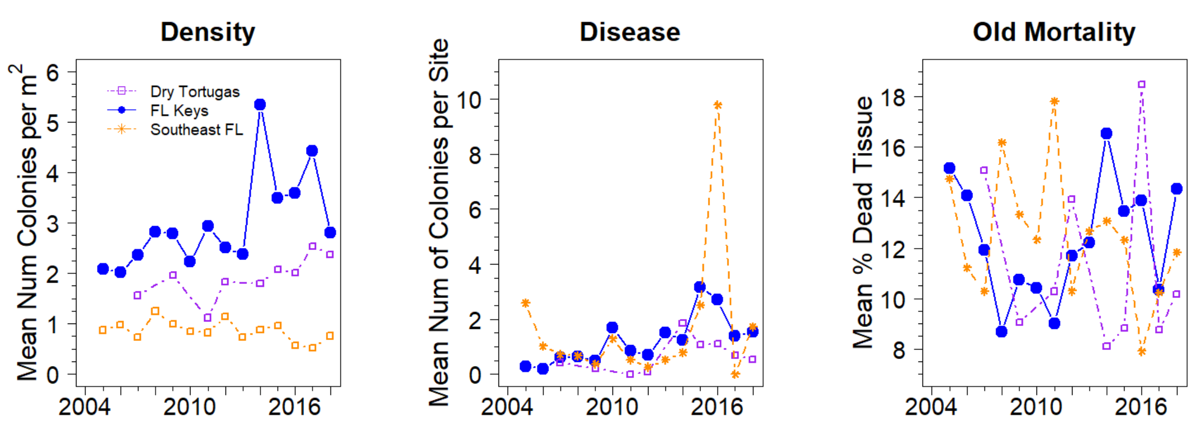


Figure 5.4. Coral density (left panel), disease prevalence (middle panel), and mortality (right panel) for three Florida reef ecosystems.

prevalence has increased gradually since 2004 in the Florida Keys and Dry Tortugas (Fig. 5.4, middle). Incidence of disease was particularly high in South Florida in 2016, two years after the first reports of Stony Coral Tissue Loss Disease (SCTLD) in the region [168]. By 2016, the disease had spread to the Florida Keys and has been confirmed in the Dry Tortugas National Park [169, 170]. Coral mortality has been highly variable, with recent high mortality in the Keys and the Dry Tortugas (2016), while mortality in southeast Florida was generally higher in the mid-2000s and has declined somewhat in recent years (Fig. 5.4, right).

5.5 Coral bleaching

Coral bleaching refers to the process by which corals expel the zooxanthellae (algal cells) symbionts that live within their tissue [171]. Coral bleaching can be caused by a number of environmental stressors, but is typically associated with unusually high sea temperatures. Corals that experience bleaching often recover if elevated temperatures are not too severe or prolonged,

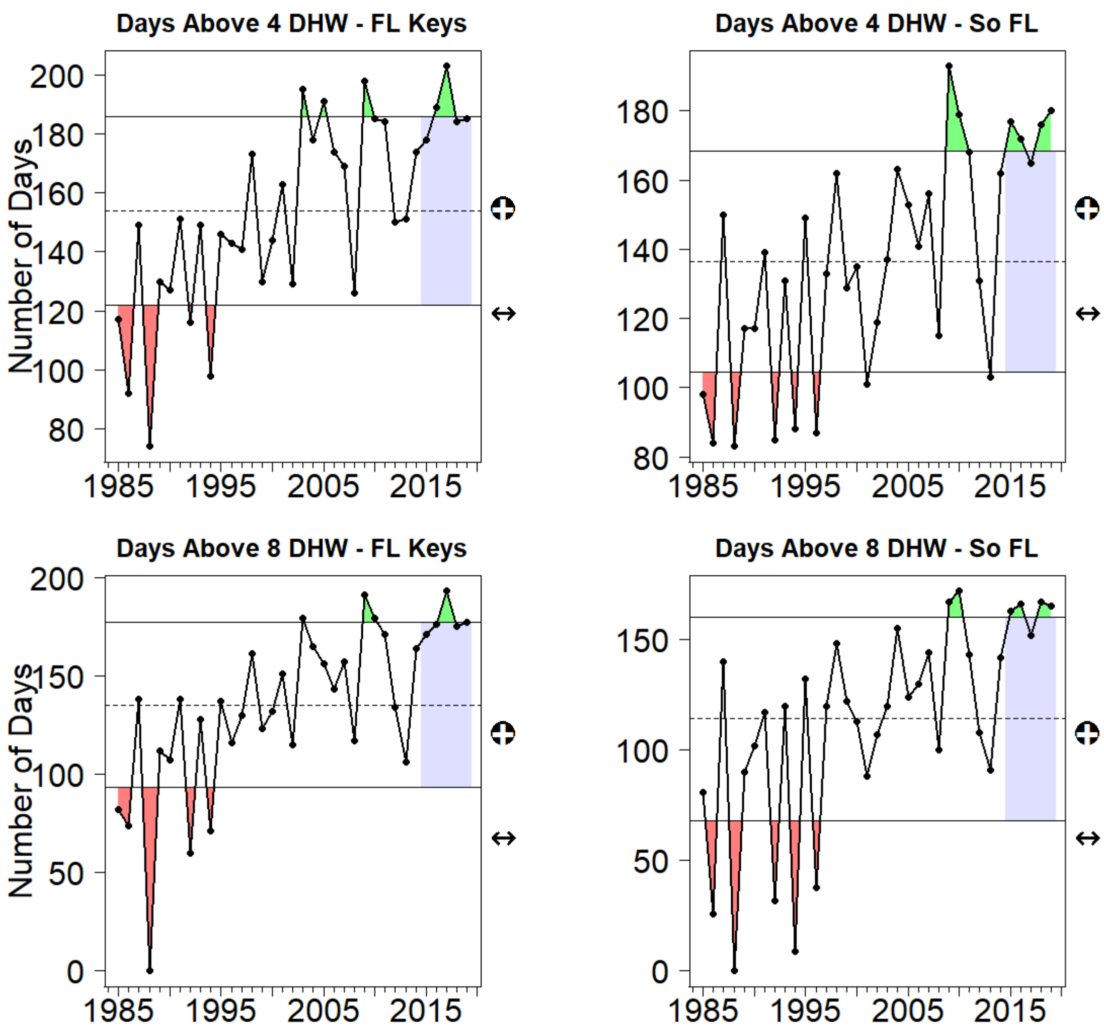


Figure 5.5a. Thermal stress indicators for corals in the Florida Keys (left panels) and South Florida (right panels). Note different y-axis scales.

though slower growth, reduced calcification, reproductive impairment, and increased incidence of disease may still occur [172]. High coral mortality is not unusual during severe bleaching episodes, such as those that occurred in 1982 – 1983, 1998, and in 2014 – 2017, the longest and most widespread bleaching event on record [167].

Data reported here were taken from the NOAA Coral Reef Watch (CRW) program daily global 5 km satellite coral bleaching heat stress monitoring product suite [173, 174]. Degree heating week (DHW) is a measure of the magnitude and duration that water temperatures have been above a maximum monthly mean (MMM) temperature considered stressful for corals. DHW was calculated by summing the difference between the observed maximum daily temperature and MMM, with MMM values of 29.6°C and 29.4°C for the Florida Keys and South Florida, respectively. Significant bleaching is expected when temperatures are above 4 DHW, while both severe, widespread bleaching and significant mortality are expected at temperatures above 8 DHW. The number of days above 4 DHW, the number of days above 8 DHW, the first day of the year that temperatures were above 4 DHW, and the last day of the year that temperatures were above 4 DHW were also computed annually from 1985 – 2019 as indicators of thermal stress for corals in the Florida Keys and in South Florida.

Days above 4 DHW and above 8 DHW have shown an increasing trend from the mid-1980s and 1990s to the late 2000s, suggesting increasing thermal stress on Florida coral reef ecosystems (Fig. 5.5a). Patterns between the two indicators (4 DHW and 8 DHW) and between the two regions (Keys and South Florida), were similar, though annual variability was greater for 8 DHW than for 4 DHW. The portion of the year during which stressful temperatures occur has increased nearly two-fold since the 1980s, with stressful temperatures now occurring earlier in the spring and persisting later into the fall (Figure 5.5b).

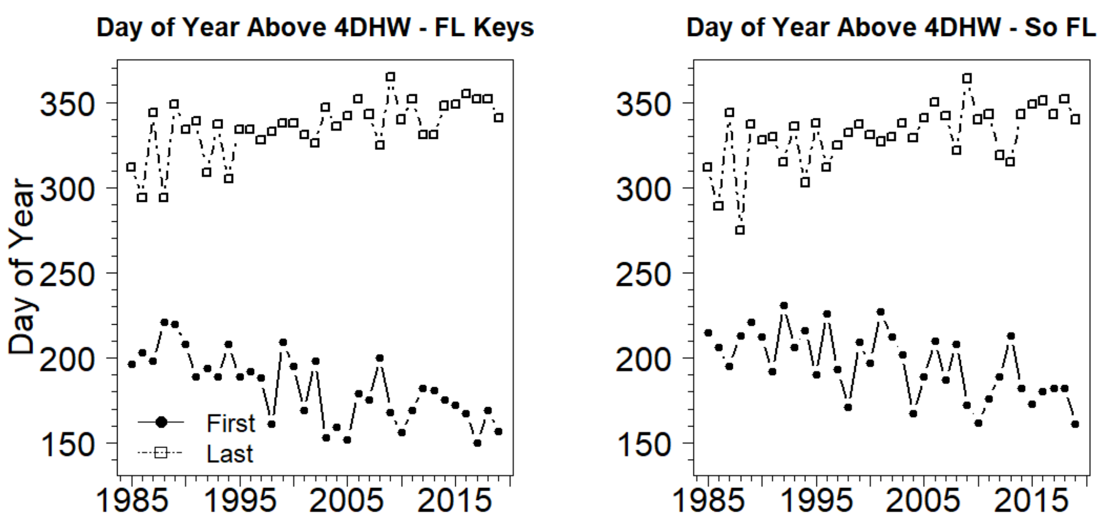


Figure 5.5b. First and last day of the year that temperatures were above 4DHW for corals in the Florida Keys (left panel) and South Florida (right panel).

6. LOWER TROPHIC LEVEL STATES

6.1 Primary productivity

Chlorophyll-a (Chl-a) concentration in ocean surface waters derived from satellite measurements is considered a proxy for phytoplankton biomass, the major primary producer in most continental shelf ecosystems [175]. The spatial and temporal dynamics of Chl-a reflect interactions among complex hydrographic and biological processes such as nutrient distributions, zooplankton grazing, upwelling dynamics, coastal circulation, temperature, and the light environment of the upper water column. The South Atlantic Bight is considered nutrient-poor, and primary productivity in the region is driven by nutrient inputs onto the shelf from shelf-break upwelling, Gulf Stream eddies, and riverine nutrient loading [63, 176, 177]. The importance of these processes to phytoplankton dynamics varies spatially and temporally, with the Gulf Stream intensity and location playing an important role in moderating nutrient dynamics [178, 179, 180]. Surface Chl-a is typically highest in coastal waters on the inner shelf (1.0 mg/m^2) and decreases by about three-fold toward nutrient-depleted Gulf Stream waters (0.3 mg/m^2). A persistent thin band of surface Chl-a is often present at the shelf break due to synoptic upwelling from Gulf Stream interactions with the upper continental slope.

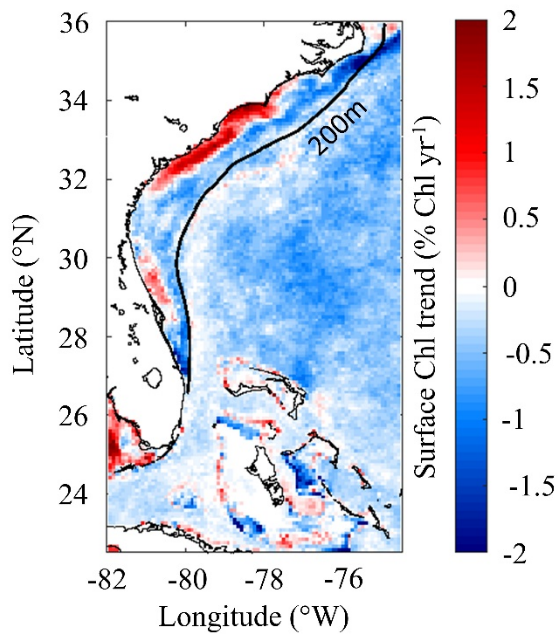


Figure 6.1a. Mean normalized Chlorophyll-a trend in the U.S. South Atlantic based on MODIS satellite imagery from 2003 to 2020. The thick black line denotes the 200 m isobath

A time series of monthly surface Chl-a anomalies in the U.S. South Atlantic was developed based on ocean color remote sensing images produced from MODIS (Moderate Resolution Imaging Spectroradiometer) data [181]. Specifically, the product used here was the mapped, level-3, daily, 4 km resolution images from MODIS-Aqua from January 1, 2003 to December 31, 2020, with a spatial domain of 82º to 74.5 ºW longitude and 22.5º to 36 ºE latitude. Every grid cell in each daily image was filtered by excluding observations outside of the 90% confidence interval for a given month and across years. Monthly means of surface Chl-a were then computed and expressed as an anomaly normalized by the standard deviation of observations.

Figure 6.1a shows the long-term trend in surface Chl-a and Fig. 6.1b shows the mean monthly standardized anomalies taken over the spatial domain for the 2003 – 2020 period. Throughout most of the region, Chl-a decreased slightly over the 18-year period with rates of approximately 0.5 – 1% yr^{-1} of the average Chl-a concentration (Fig. 6.1a). However, in the northern region of the shelf, surface Chl-a exhibited a positive trend approaching 2% yr^{-1}. The highest concentrations of surface Chl-a typically occur in winter and early spring (Jan – Mar) and the lowest occur in summer (Jun – Aug). From the early to late 2000s, mean surface Chl-a concentrations declined and remained low through 2015, followed by an increase and relatively stable concentrations since 2016 (Fig. 6.1b).

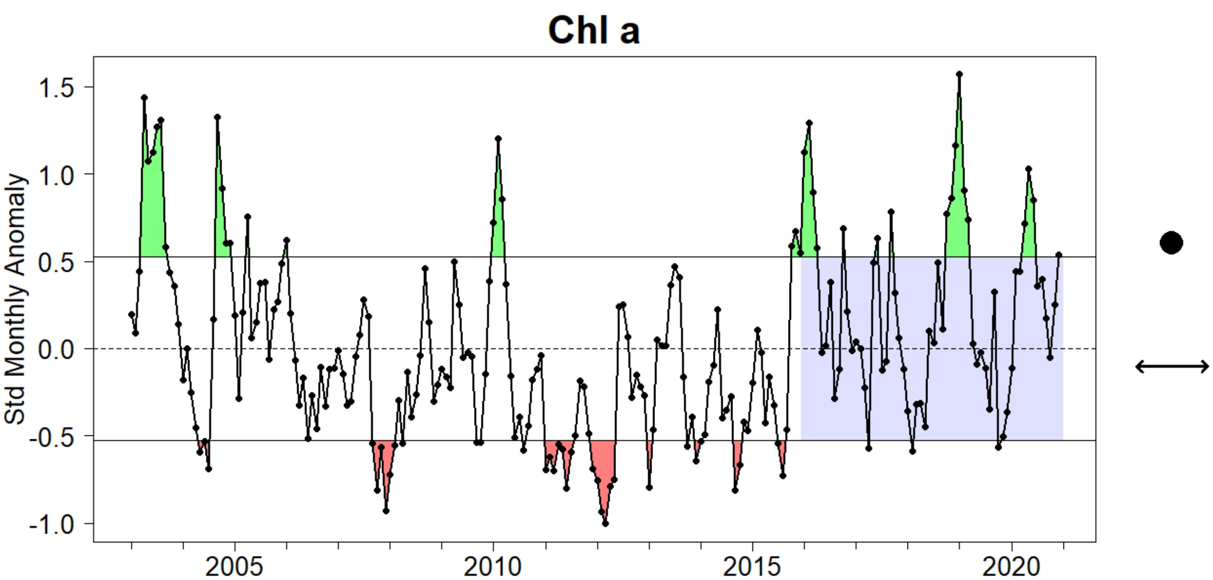


Figure 6.1b. Mean monthly MODIS surface chlorophyll-a anomalies normalized by the standard deviation of observations in the U.S. South Atlantic.

6.2 Zooplankton

Zooplankton play an important role in marine ecosystems because they link primary production to planktivorous fish [182, 183, 184] as well as mediate biogeochemical processes that determine the flow of carbon in pelagic ecosystems [185, 186, 187]. Most zooplankton sampling on the U.S. eastern seaboard has been conducted in the Mid-Atlantic Bight, on the Northeast continental shelf, and in the Gulf of Maine. The Marine Resources, Monitoring, Assessment, and Prediction (MARMAP) program sampled zooplankton as far south as Stuart, Florida in 1986 and to the South Carolina-Georgia border in 1985, but there are no large-scale survey time series of zooplankton on the U.S. South Atlantic continental shelf.

COPEPOD (Coastal and Oceanic Plankton Ecology, Production, and Observation Database) is a global plankton database that compiles quality-reviewed plankton data from multiple projects and surveys in marine ecosystems around the globe [188]. Data on zooplankton displacement volume from COPEPOD for the region south of 36.2 ºN latitude (Virginia-North Carolina border) were used to construct an index of zooplankton biomass [189]. Sampling rarely extended south of Cape Hatteras, NC, and this index encompasses only the northern area of the U.S. South Atlantic continental shelf (Fig. 6.2a). Sample sizes were relatively small, with a median annual sample size from 1977 to 2015 of 16 stations (range: 1 to 42 stations); no data were available from 1989-1992 or after 2015. Sampling occurred in all months, but was concentrated in early spring (e.g., March) and early fall (e.g., September). Preliminary analyses did not reveal consistent differences in time series for the January-June period and the July-December period; therefore, data were pooled across all months as an annual index.

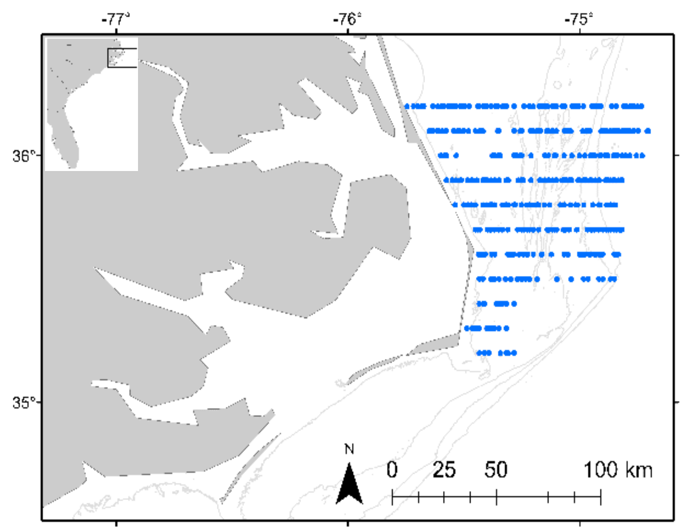


Figure 6.2a. Map of sampling locations in the U.S. South Atlantic (blue circles) from zooplankton surveys.

Zooplankton biovolume was highly variable and did not show strong trends. In general, biovolume was above the long-term mean in the early 1980s and the early to mid-2000s, and lower than the mean during most of the 1990s (Fig. 6.2b). A similar pattern was reported for the Gulf of Mexico [190] and the Mid-Atlantic Bight [191], suggesting the possibility of a weak, large-scale, quasi-decadal oscillation in zooplankton population dynamics. The peak year of 2004 reported here was also evident in time series of the Mid-Atlantic Bight [191], further supporting the similarity of the region between Cape Hatteras and the NC-VA border with the broader Mid-Atlantic. Over the last five years of the time series, zooplankton biovolume has been slightly above the long-term mean with little trend.

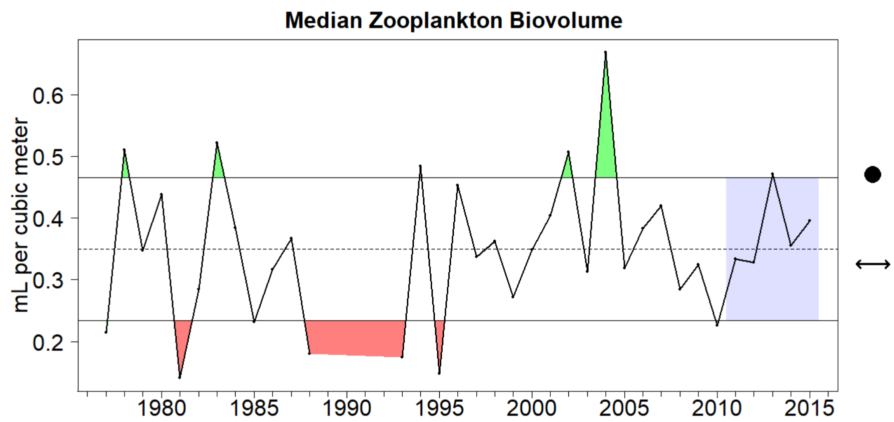


Figure 6.2b. Median zooplankton biovolume for the region between the North Carolina-Virginia border and Cape Hatteras, North Carolina from 1977 – 2015. Data were not available for 1989 – 1992.

6.3 Ichthyoplankton diversity and abundance

Ichthyoplankton abundance is affected by physical and biological forcing across a range of spatial scales and is a potential indicator of recruitment and year class strength of coastal fishes. There are no ongoing or recent large-scale ichthyoplankton surveys in U.S. South Atlantic waters. Since the 1980s, two separate, fixed-site ichthyoplankton surveys have been conducted near Beaufort Inlet, NC (Beaufort Bridgenet Ichthyoplankton Sampling Program, BBISP) and in the North Inlet estuary near Georgetown, SC (SCN). These long-term surveys have been used to investigate decadal changes in community composition in relation to warming patterns within estuaries [192, 193, 194, 195], as well as to investigate the recruitment dynamics of a number of South Atlantic species [196, 197, 198, 199, 200].

The NC survey conducts weekly nocturnal sampling (October – May, 1000 micron net) from a fixed station within a tidal estuarine channel ~ 1.5 km upstream from Beaufort Inlet. The SC survey conducts biweekly daytime sampling with an epibenthic sled (360 micron net), in which three consecutive tows of about 200 meters (5 minutes) are made in a large subtidal salt marsh channel (Town Creek, depth about 4 m) about 1.0 km from the North Inlet. Mean species richness, defined as the average total number of ichthyoplankton taxa collected per 100 cubic meters, and mean annual density, defined as the average total number of ichthyoplankton collected per 100 cubic meters, were calculated separately for each survey. For comparison between the surveys, the two metrics were calculated annually using only data from Oct – May, with the sampling year defined as the calendar year associated with the Jan – May portion of the sampling season. For the SC survey, these metrics were also computed on an annual basis (12 month, Jan – Dec) given the year-round sampling.

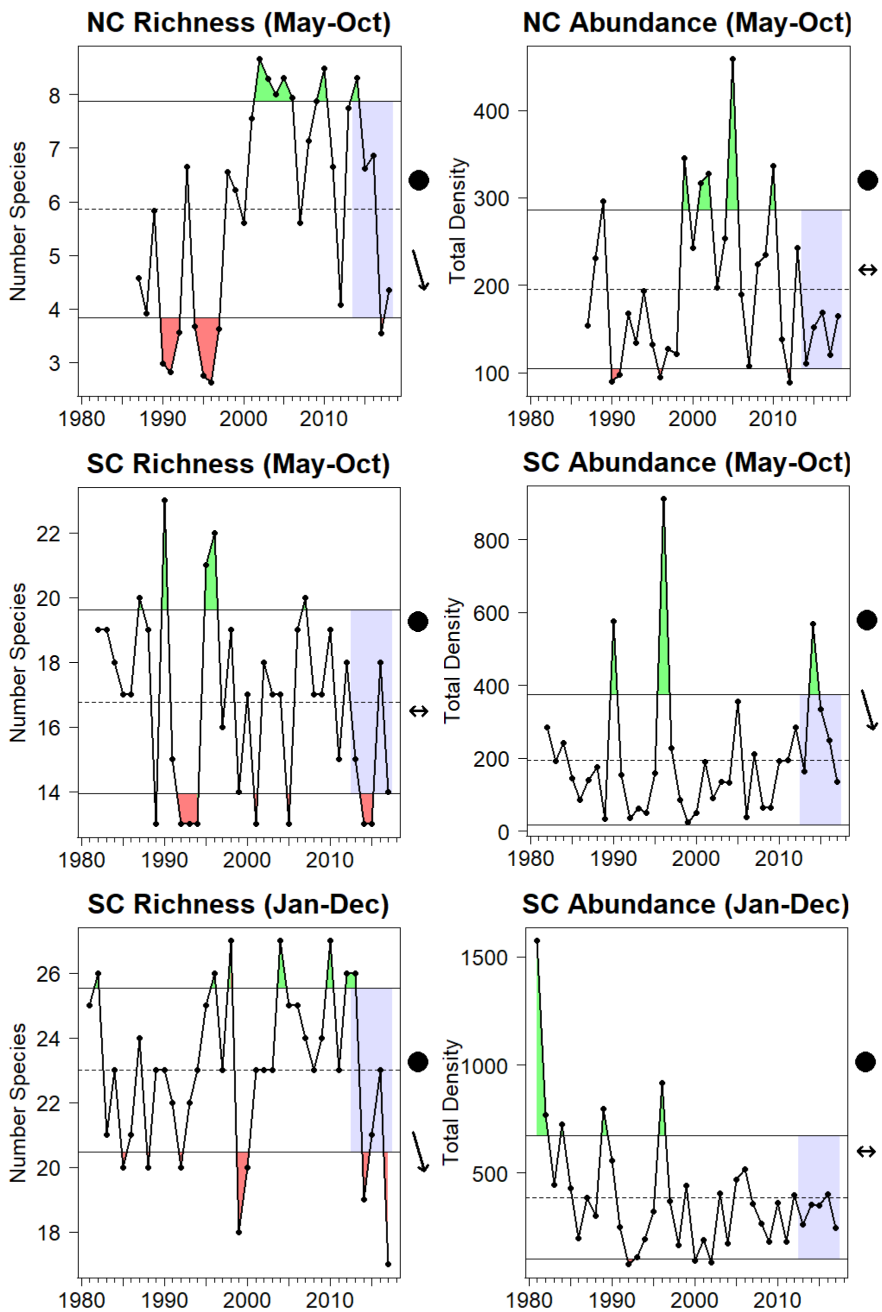


Figure 6.3. Time series of species richness (left panels) and total density (right panels) from bridgenet sampling in Beaufort Inlet, NC (top row) and North Inlet estuary, SC (middle and bottom rows). Numbers are per 100 cubic meters.

Mean ichthyoplankton species richness in NC was lower in the 1980s and 1990s compared to the 2000s and showed a highly variable but declining trend since the mid-2000s (Fig. 6.3, top left).

There was little long-term pattern in species richness in SC over either the 8-month or the 12-month sampling period, though both showed a slight declining trend since the mid to late-2000s (Fig. 6.3, bottom left). Mean annual ichthyoplankton density in NC varied almost five-fold, with relatively low values during 1990 – 1998 and 2014 – 2018, high values from 1999 – 2005, a highly variable but declining trend from 1999 – 2018, and relatively stable densities over the last five years (Fig. 6.3, top right). Ichthyoplankton densities in SC over a comparable sampling window (Oct – May) have been relatively stable over the long-term with the highest densities occurring in 1990, 1996, and 2014; however, there has been a declining trend over the most recent (2014 – 2018) five years (Fig. 6.3, middle right). A different pattern in ichthyoplankton densities in SC was evident based on the year-round sampling (Jan – Dec, Fig. 6.3, bottom right). The highest densities occurring in the early part of the time series (1981, 1984, 1989, 1996), while recent densities have been near or below the long-term mean but relatively stable over the last five years.

6.4 Forage fish abundance

Atlantic Menhaden (*Brevoortia tyrannus*) support one of the largest fisheries by volume in the United States and are an important forage species, contributing to the diets of a number of important piscivores (e.g., Striped Bass *Morone saxatilis*, Weakfish *Cynoscion regalis*, Bluefish *Pomatomus saltatrix*, and coastal sharks) in the northwest Atlantic shelf ecosystem [201, 202]. During the peak of the Atlantic Menhaden fishery in the 1950s, there were more than 20 processing plants along the Atlantic seaboard with over a third of those in the South Atlantic (North Carolina to Florida); currently a single plant operates in Reedville, Virginia. Coastwide landings of Atlantic Menhaden have declined to less than 20% of their historic highs due primarily to market forces. While landings in the reduction fishery, which accounts for ~ 74% of the total harvest, are at their lowest levels in the time series (1955 – 2017), landings in the smaller bait fishery, which supplies Atlantic Menhaden primarily to crab and lobster fisheries, have increased by 20% since the early 2000s. In the U.S. South Atlantic, North Carolina and Florida have significant bait fisheries for Atlantic Menhaden.

Based on the most recent stock assessment, coastwide population biomass of Atlantic Menhaden reached a low point in the 1970s, has increased steadily since the mid-1980s, and has not been considered overfished or undergoing overfishing for the last several decades (Fig. 6.4, top [201]). Estimated biomass over the last five years has fluctuated above the mean with a slight upward trend. A composite index of the relative abundance of adult Atlantic Menhaden (ages 1+) based on multiple surveys in the U.S. South Atlantic region (NC to FL) was developed as part of the most recent coastwide assessment. The index shows relatively high abundance of Atlantic Menhaden in 1990, 2006, and 2010, low abundance from the early-1990s to early-2000s, and stable abundance near the long-term mean since 2010 (Fig. 6.4, bottom). This pattern is similar to the coastwide biomass estimated by the stock assessment model, except that recent biomass appears to be increasing more rapidly than the southern index (i.e., since 2005), potentially due to an expanding age structure in other parts of the species' range.

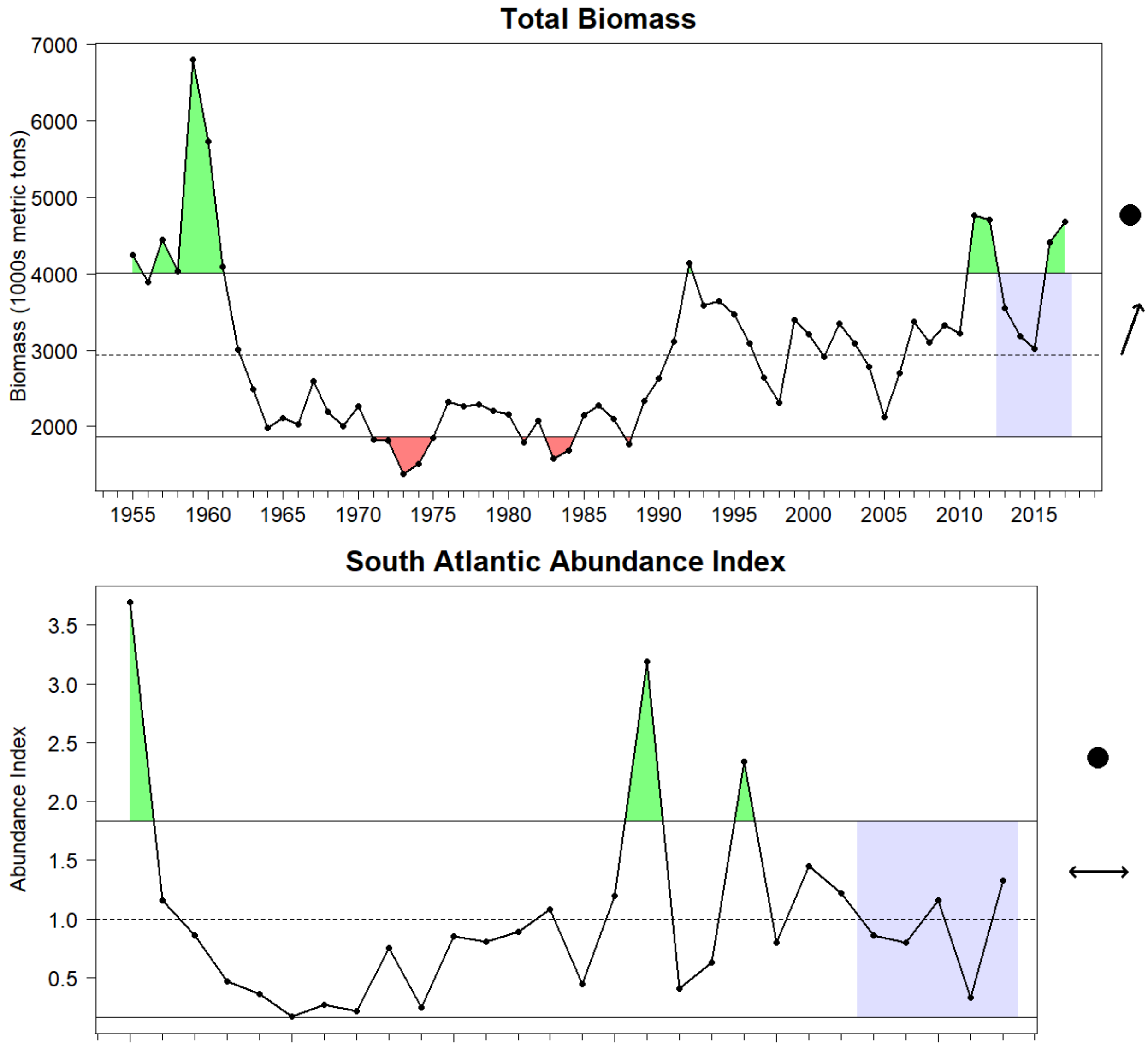


Figure 6.4. Total coastwide biomass estimated by the stock assessment (top panel) and the South Atlantic relative index of abundance (bottom panel).

7. UPPER TROPHIC LEVEL STATES

Multiple surveys that sample different portions of the upper trophic level fish community are conducted annually on the U.S. South Atlantic continental shelf. Similar indicators were developed across different survey platforms in order to assess changes over time for different components of the South Atlantic fish community. Two response variables were analyzed for each survey: The total number of species caught ("species richness") and the total catch-per-unit-effort (CPUE) of all individuals of all species ("total abundance").

7.1 Nearshore demersal fish diversity and abundance

The Southeast Area Monitoring and Assessment Program - South Atlantic (SEAMAP-SA) bottom trawl survey has sampled demersal nekton (e.g., sciaeneid fishes, Paralichthyid flounders, decapod crustaceans) using bottom trawls on the nearshore continental shelf between Cape Hatteras, NC and Cape Canaveral, FL since 1988 [61]. Data were pooled over three seasons (spring, summer, and fall) for the inner shelf (4 – 10 m depth) to assess annual trends from 1989 – 2017. Several taxonomic groups were pooled above the species levels due to potential identification issues, including anchovies (family Engraulidae), spider crabs (genus *Libinia*), mud crabs (family Xanthidae), true mud crabs (family Panopeidae), porgies (species of genus *Stenotomus*) and squids (species of genus *Loligo*). Miscellaneous taxa including algae, several invertebrates, cannonball jellyfish (*Stomolophus meleagris*), and horseshoe crabs (*Limulus polyphemus*) were also removed due to sampling or identification issues.

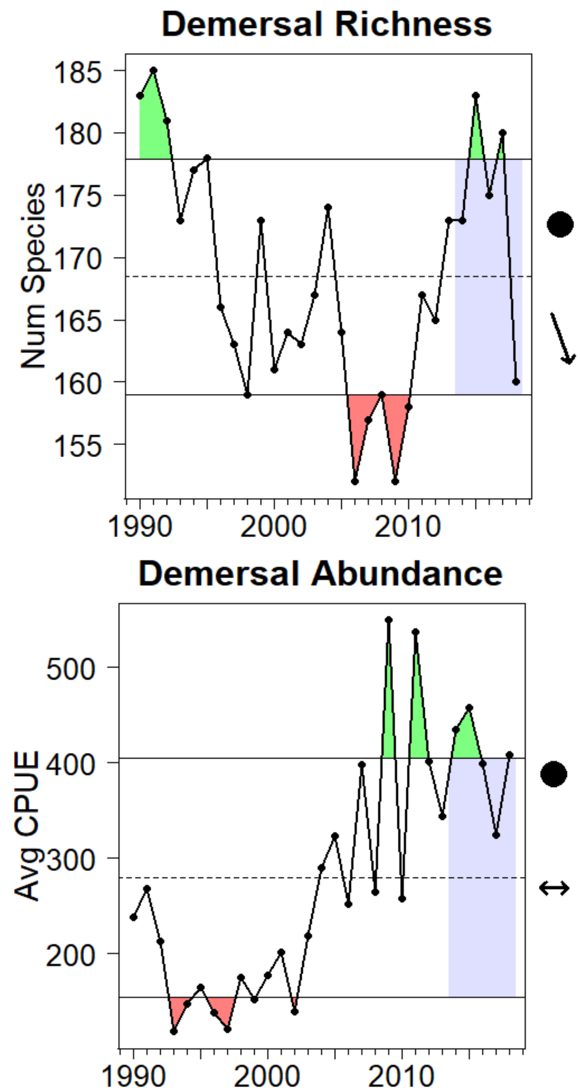


Figure 7.1 Species richness (top) and relative abundance (bottom) of nearshore demersal fishes from the SEAMAP-SA trawl survey on the nearshore South Atlantic continental shelf.

Demersal species richness declined from the 1990s to the mid-2000s and has increased since 2010 (Fig. 7.1). Demersal abundance was relatively stable through the 1990s, but has increased since the early 2000s and has been relatively stable in recent years. Potential explanations for these changes in abundance and diversity include shifts in spatial distribution or the timing of recruitment to the shelf, direct and indirect (bycatch) effects of fishing, particularly bycatch in the nearshore shrimp trawl fishery, and/or varying trophic interactions.

7.2 Offshore hard bottom fish diversity and abundance

The Southeast Reef Fish Survey (SERFS) is conducted jointly by the Marine Resources Monitoring, Assessment, and Prediction (MARMAP) program [203], the SEAMAP-SA survey [61], and the NMFS-SEFSC Southeast Fishery-Independent Survey (SEFIS) [60]. The programs use consistent trap-video methods to survey reef fishes on hard bottom habitat in the U.S. South Atlantic region. Thirty years (1990 – 2019) of SERFS trap data were used to quantify changes in reef fish abundance and species richness in the U.S. South Atlantic region. Trap data were standardized using generalized additive models to correct for potential biases due to uneven sampling or fluctuations in environmental conditions among years. Video data are not presented here because the video time series is relatively short (beginning in 2010). Species directly targeted by fisheries were analyzed separately from those not supporting directed fisheries [204].

Mean species richness of both targeted and non-targeted species has declined since the early 1990s (Fig. 7.2, left). Mean annual abundance has been variable over time, with periods of high abundance in the early 1990s, early 2000s, and early 2010s (Fig. 7.2, right). Over the last five years (2015 – 2019) total abundance as well as abundance of targeted species has declined, while abundance of nontargeted species has been relatively stable. Common species caught in the survey have a strong influence on overall trends. For example, Black Sea Bass (*Centropristis striata*), one of the most common "targeted" species caught in the SERFS trap survey, has declined precipitously over the last five years, which has likely contributed considerably to the overall decline in the abundance of targeted species over the same time frame.

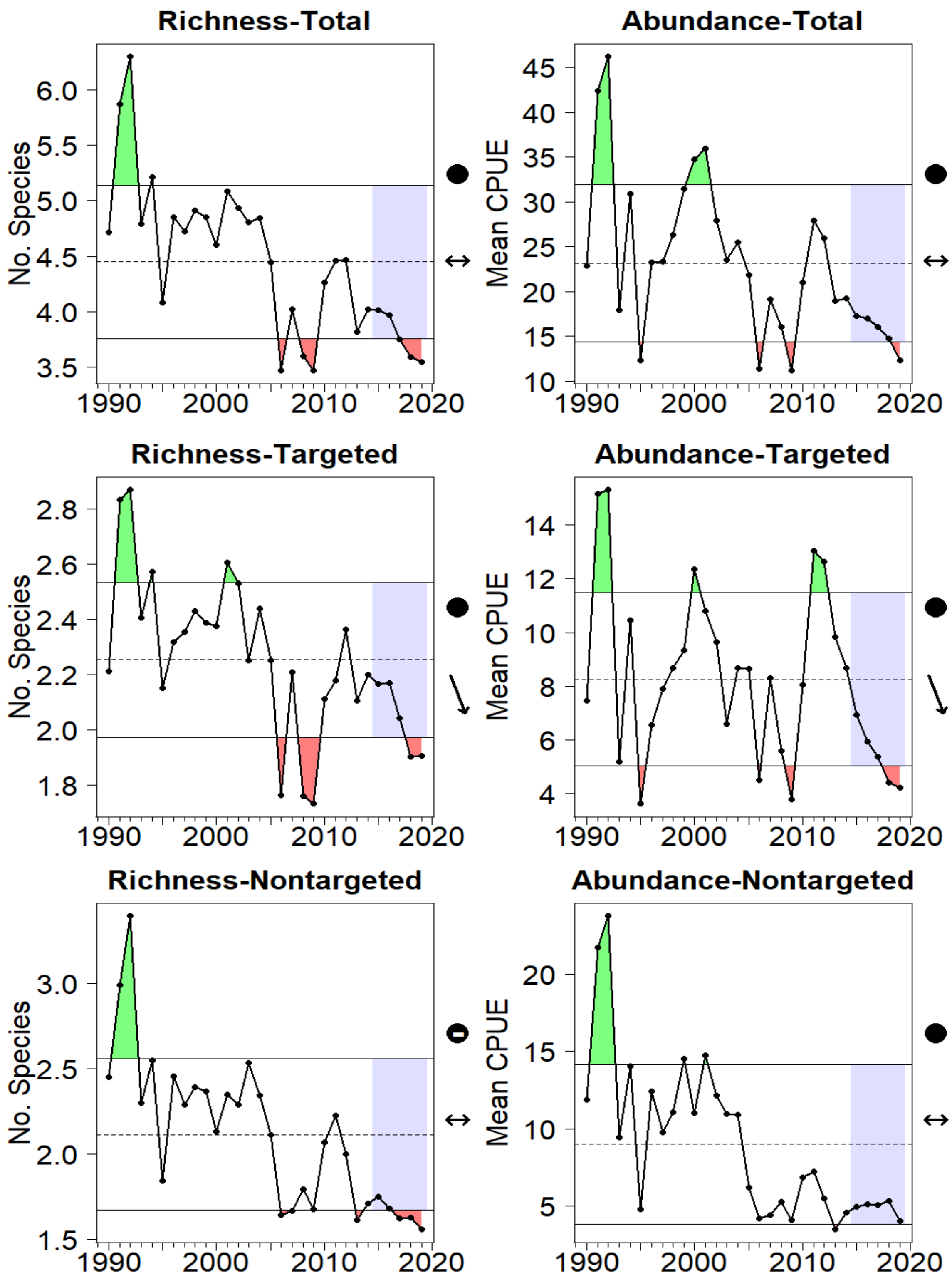


Figure 7.2. Species richness (left) and relative abundance (right) of hard-bottom reef fishes on the U.S. South Atlantic continental shelf. Metrics are reported for all species (top), species targeted by fisheries (middle), and species not typically targeted by fisheries (bottom).

7.3 Coastal shark diversity and abundance

The NMFS Fishery-Independent Bottom Longline Survey (NMFS BLL) samples coastal shelf waters from the east coast of Florida near Miami to the North Carolina Cape Hatteras region. The survey predominately samples shark species. Fourteen years of data (2002, 2004 – 2006, 2008 – 2017) were used to quantify annual patterns in shark abundance and species richness. Catch-per-unit-effort (CPUE) and species richness were standardized using a delta-lognormal model [205] to correct for potential biases due to uneven sampling or fluctuations in environmental conditions among years. A total of 10,735 sharks were captured at 690 stations sampled between 2002 and 2017. The total number of shark species per set ranged from 0 to 7 while the total number of sharks per set ranged from 0 to 72. Atlantic Sharpnose Sharks (*Rhizoprionodon terraenovae*) made up the majority of the catch (88%), followed by Tiger Sharks (*Galeocerdo cuvier*, 4.2%), Sandbar Sharks (*Carcharhinus plumbeus*, 3.1%), Blacknose Sharks (*Carcharhinus acronotus*, 1.4%), and Scalloped Hammerhead (*Sphyrna lewini*, 0.9%).

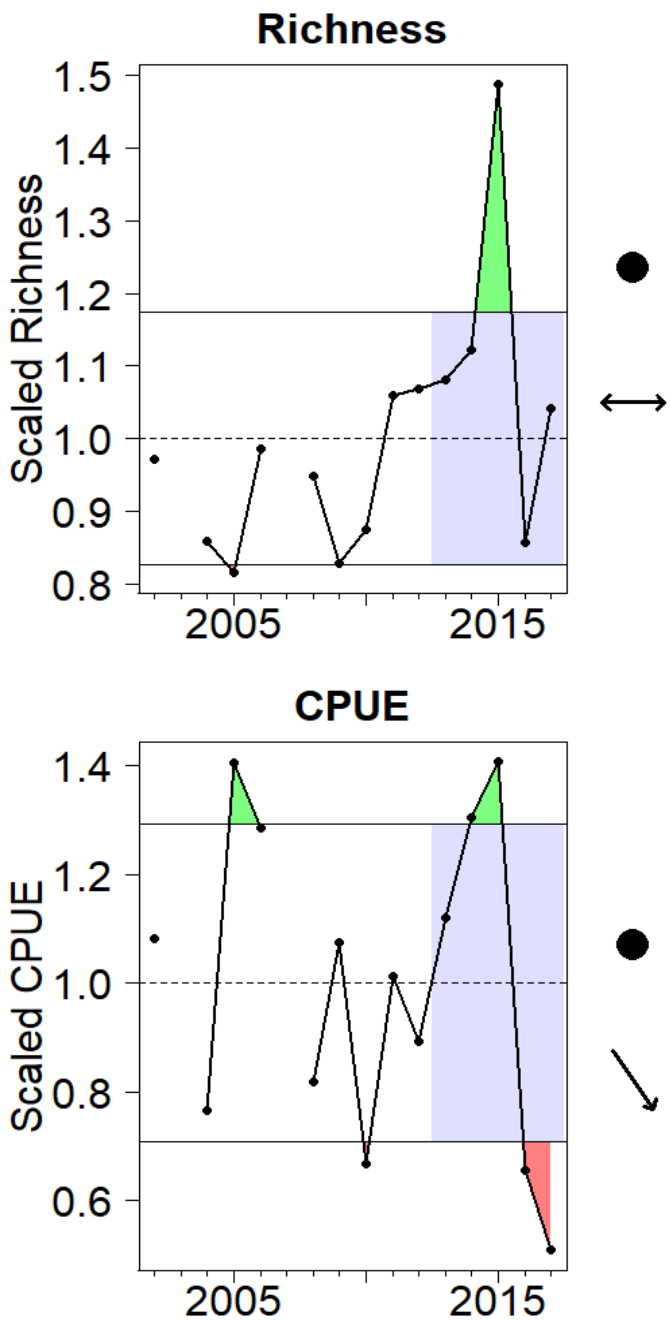


Figure 7.3. Species richness (top) and relative abundance (bottom) of coastal sharks from the NMFS BLL survey.

Species richness of sharks has been variable over the 14-year period with generally higher richness in the 2010s compared to the 2000s, and a peak in richness in 2015 followed by a sharp decline in 2016 (Fig. 7.3, top). Total relative abundance of sharks in the survey has varied with little trend through most of the 2000s, though abundance declined to its lowest level in the last two years (2001 – 2017) of the time series.

7.4 Coral reef fish diversity and abundance

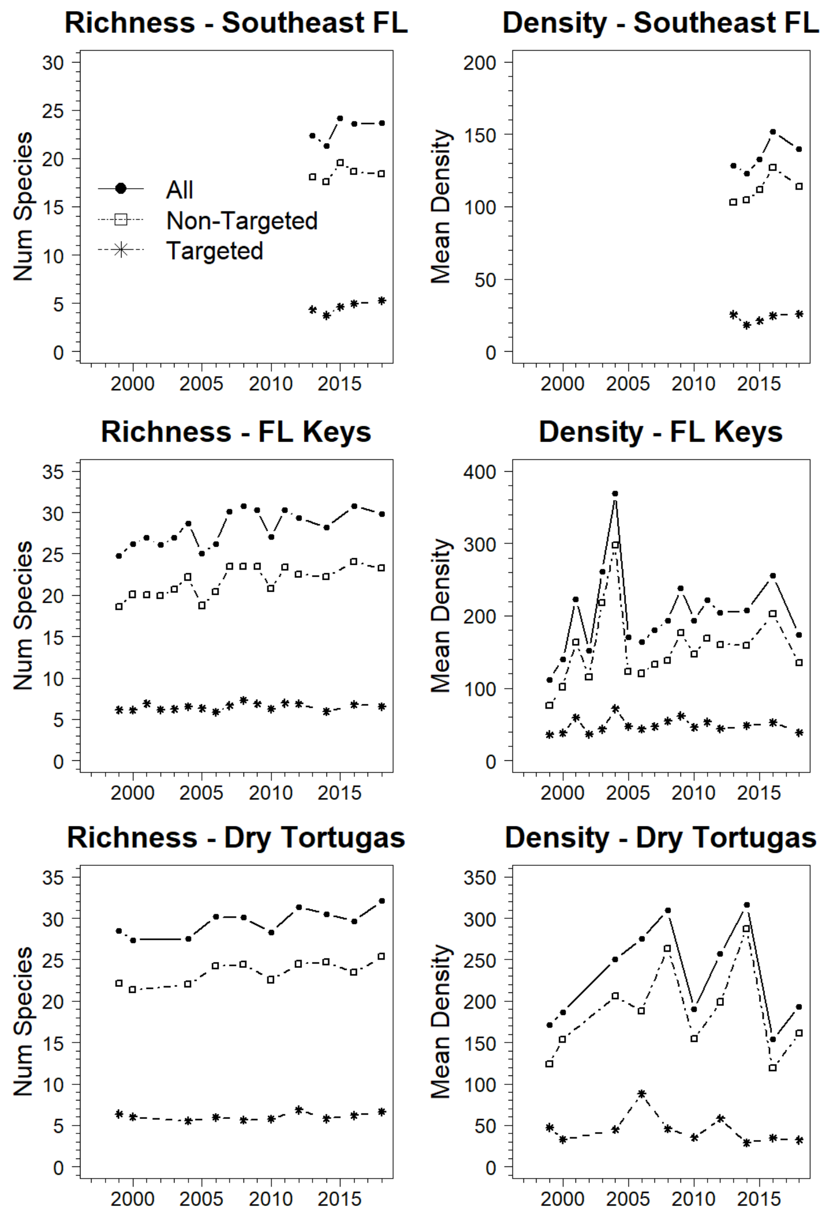


Figure 7.4. Species richness (left panels) and mean density (right panels) of targeted, non-targeted, and all reef fishes within three regions of Florida's coral reef tract.

Florida's shallow reef tract, the only shallow, warm-water coral reefs in the continental United States, contribute $8.5 billion to the local economy, tourism, and the fishing industry [206]. Though Florida's reefs continue to harbor numerous colorful ornamental fishes, such as Parrotfishes (family Scaridae) and Angelfishes (family Pomacanthidae), many years of heavy fishing pressure, advances in fishing technologies, and the life history characteristics of reef fish species (e.g., high site fidelity, late age-at-maturity, hermaphroditism) have resulted in substantial declines since the 1990s [207].

The Reef Visual Census (RVC) is a comprehensive fishery-independent SCUBA survey designed to assess the status and trends of reef fishes on Florida's coral reef tract [208, 209]. RVC surveys are a collaborative cross-agency effort led by the Southeast Fisheries Science Center, with the Florida's Department of Environmental Protection and Fish and Wildlife Conservation Commission, National Park Service, Nova Southeastern University, and University of Miami as key partners. Sampling occurs on hard bottom habitats less than 100 feet in depth within three distinct sampling domains: Southeast Florida, Florida Keys, and Dry Tortugas. Surveys in Southeast Florida were conducted annually from 2013 to 2017 and have occurred biannually since 2018. Surveys in the Florida Keys were conducted annually from 1999 to 2013 and biannually since 2014, while the Dry Tortugas have been surveyed biannually since 2004.

Patterns in species richness and density over time were driven primarily by non-targeted species, which comprise most of the catch in all three regions. For Southeast Florida reefs, time series are short (five years) and show little trend over time (Fig. 7.4, top row). In the Florida Keys, both species richness and density of non-targeted species have shown a slight upward trend since the early 2000s, while the same indicators have remained relatively constant over time for targeted species (Fig. 7.4, middle row). The Dry Tortugas showed similar, though less pronounced, patterns compared to the Florida Keys but with considerable annual variability (Fig. 7.4, bottom row).

7.5 Mean trophic level

Mean trophic level (MTL) is an indicator of ecosystem state driven by both top-down (e.g., fishing) and bottom-up (e.g., primary productivity) forces that affects the relative biomass of different species or functional groups [210, 211, 212]. The MTL is calculated annually as a weighted average of assigned trophic levels for species or species groups, weighted by the biomass or abundance of each group. Declines in MTL over time may indicate unsustainable patterns in fishing, changes in productivity at the base of the food web, or other changes in community structure.

MTL was computed from commercial (1950 – 2017) and recreational (1981 – 2018) landings data collected by the Atlantic Coastal Cooperative Statistics Program (ACCSP, [213]) and the Marine Recreational Information Program (MRIP, [214]), respectively. Trends prior to 1980 should be interpreted with caution because commercial landings were not reported with the same level of taxonomic resolution as later in the time series. In addition, the recreational landings were limited to fish species only, while the commercial data included both landed fish and invertebrates.

The MTL for all commercial landings in the U.S. South Atlantic increased from 1950 to 2017, with the largest increases occurring in the early 1970s and in the mid-2000s (Fig. 7.5a, top). Trends in MTL in the commercial landings were driving mainly by three low trophic-level fisheries, Atlantic Menhaden, Blue Crab (*Callinectes sapidus*), and Penaeid Shrimp (Pink Shrimp, *Farfantepenaeus duorarum*, Brown Shrimp, *F. aztecus*, and White Shrimp, *Litopenaeus setiferus*). These three groups combined account for 45 – 86% of the total annual commercial catch by weight. The pattern in MTL is driven mostly by Atlantic Menhaden landings, which peaked in the mid – 1950s, were relatively stable from the 1970s to the 1990s, and then declined through the 2000s. After 2005 there were no longer any menhaden reduction plants in the region, and Atlantic Menhaden accounted for only ~ 1% of commercial landings in the U.S. South Atlantic. The MTL of commercial landings excluding Atlantic Menhaden was relatively low from the 1950s to the early 1970s, relatively high from the mid-1970s through the 1990s, and has varied around the long-term mean since the early 2000s (Fig. 7.5a, middle). During the 1970s and 1980s, several low trophic level species (e.g., Blue Crab, and River Herring *Alosa pseudoharengus* and *A. aestivalis*) had low landings while several high trophic level species (e.g., Paralichthyid Flounders, Weakfish, *Cynoscion regalis*) had high landings, leading to the high MTL during this period. The general increase in MTL in the 1970s also coincides with the development of major U.S. domestic fisheries associated with the 1976 Magnuson-Stevens Fishery Conservation and Management Act, while the decline in the early to mid – 1990s corresponds to the implementation of many federal and state management regulations to rebuild stocks that had declined over the intervening 20 years since the Magnuson Act.

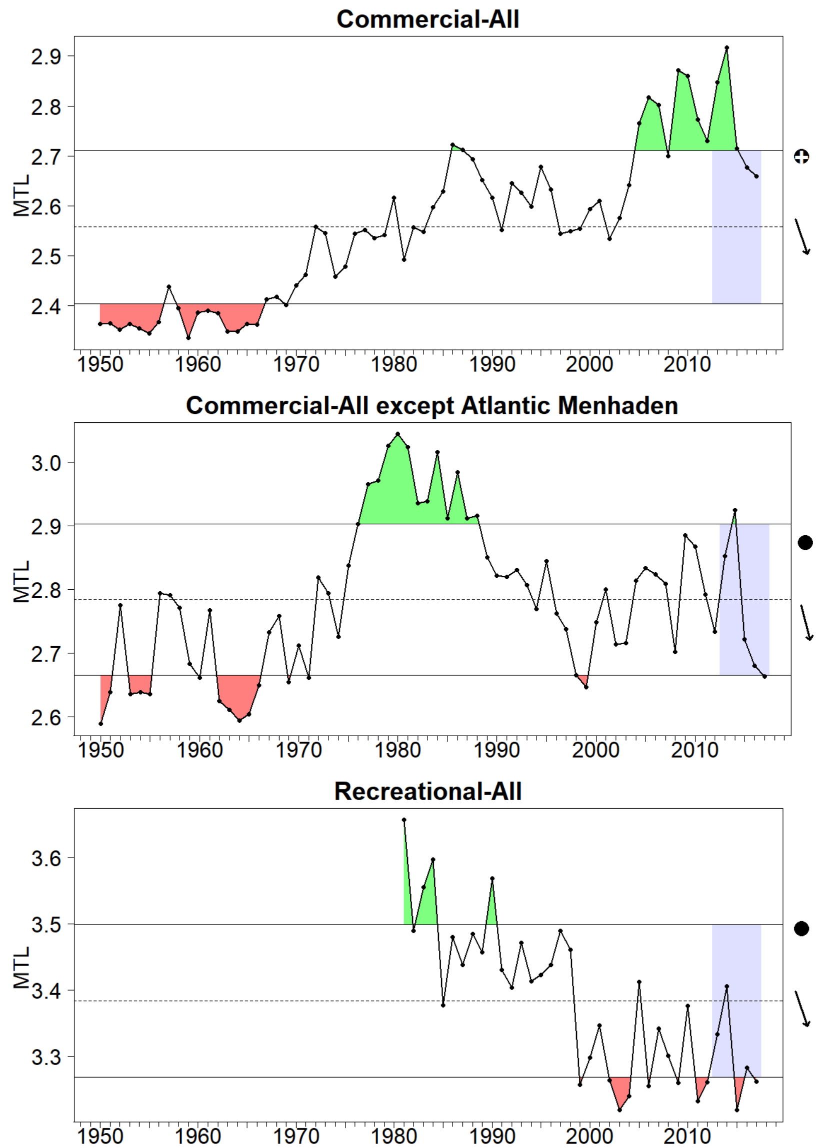


Figure 7.5a. Mean Trophic Level (MTL) of commercial and recreational landings in the U.S. South Atlantic. Note different y-axis scales.

The MTL of recreational landings declined from the early 1980s to the early 2000s and has been relatively low since then, though with considerable annual variability (Fig. 7.5a, bottom). This mirrors the pattern in recreational effort in the South Atlantic, which has steadily increased since the 1980s (see Fig. 8.4a). The overall magnitude of the decline in MTL of recreational landings is relatively small (~ one quarter of a trophic level), and reflects the harvest of more low trophic level species (e.g., Striped Mullet *Mugil cephalus*, Southern Kingfish *Menticirrhus americanus*, and Spot *Leiostomus xanthurus*), in addition to decreases in landings of upper trophic level species [215].

MTL was also computed from the two fishery-independent surveys described above, the Southeast Reef Fish Survey (SERFS) [60] and the Southeast Area Monitoring and Assessment Program South Atlantic (SEAMAP-SA) survey [61]. MTL was based on the trap data from SERFS and the bottom trawl data from SEAMAP-SA; only fish are included from both surveys.

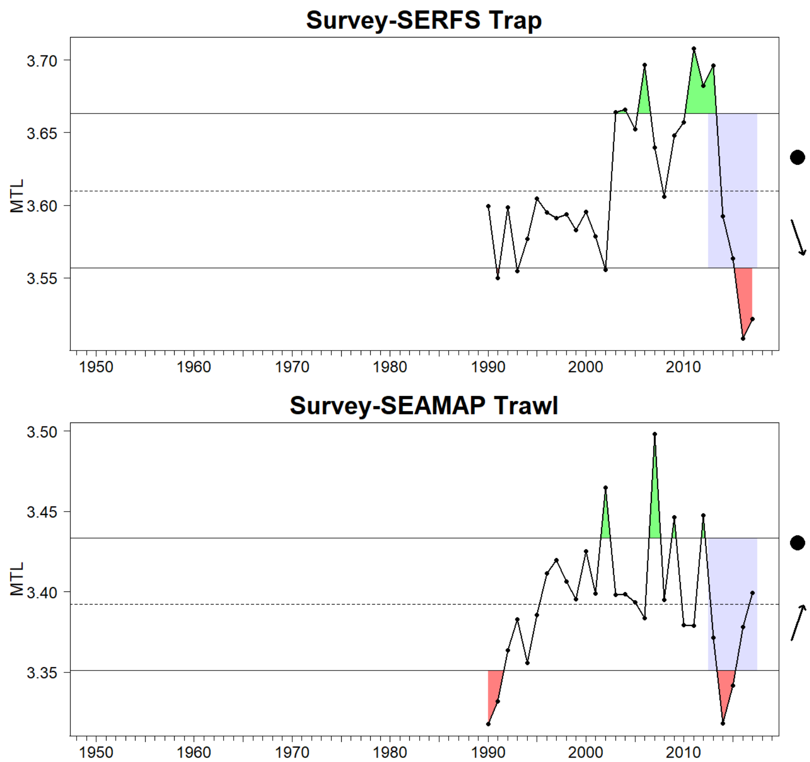


Figure 7.5b. Mean Trophic Level (MTL) of hard bottom reef fish species (top panel) and nearshore, soft-sediment demersal species (bottom panel) on the U.S. South Atlantic continental shelf.

The two fishery-independent surveys showed relatively small changes in MTL over time (Fig. 7.5b). Chevron traps used in the SERFS survey have captured 124 fish species since 1990, but are dominated primarily by four species or species groups, Tomtate (*Haemulon aurolineatum*), Porgies (primarily Red Porgy, *Pagrus pagrus*), Vermilion Snapper (*Rhomboplites aurorubens*), and Black Sea Bass (*Centropristis striata*), which account for at least 82% of the fish sampled annually. The decline in MTL in the most recent years is due to declining catch rates of Black Sea Bass, a relatively high trophic level species, and increasing catch rates of Tomtate, a low trophic level, non-targeted species (Fig. 7.5b, top). MTL within the nearshore demersal fish community has varied within a narrow range as well, but tends to be slightly lower in years with high catch rates of Atlantic Bumper (*Chloroscombrus chrysurus*), a pelagic planktivore, and slightly higher in years with high catch rates of Spot (*Leiostomus xanthurus*) and Atlantic Croaker (*Micropogonias undulatus*), two benthivores. The relatively narrow ranges of MTL are due to the similarity in trophic level among species captured within each survey.

7.6 Life history parameters

Life history characteristics of fish communities subject to intense harvest, such as maximum size, and age and size at maturity, can change due to genetic or phenotypic selection or changes in species composition within the community [216]. The U.S. South Atlantic reef fish fishery targets a multispecies complex of primarily demersal species that occupy hard bottom habitats on the South Atlantic continental shelf. The targeted and non-targeted species within this community have a wide range of life history strategies that can be differentially affected by fishing [217].

Annual catch-weighted mean life history traits were computed using the Southeast Reef Fish Survey (SERFS) [60] annual trap catch data (1990 to 2017) and published information on age at maturity, length at maturity, maximum size (Linf), and maximum age (longevity). Little information is available to investigate temporally varying life history traits of South Atlantic reef fishes; therefore, life history traits were assumed fixed over time. The reported trends reflect changes in the relative abundance of species within the South Atlantic reef fish community with different life history traits rather than genetic or phenotypic changes in the traits of individual species.

The species composition of the reef fish community (based on trap sampling) has changed in the U.S. South Atlantic over the last three decades [204]. All of the four life history indicators considered have been above their long-term mean in recent years, indicating the reef fish community as a whole has shifted toward greater age and length at maturity, larger maximum size, and older maximum age, though the absolute magnitude of these changes has been relatively small (Fig. 7.6). Mean age at maturity was constant or slightly declining from 1990 to 2013 but has shown an increasing trend over the last five years. Similarly, length at maturity was variable with no trend from 1990 to 2011, but has increased over the last five years as well. Maximum size (Linf) based on von Bertalanffy growth curves and maximum age have been increasing since the mid – 2000s but have shown slight declines in recent years. The increasing trend in life history traits over time is due to the combined effect of declines in catch rates of species with young age and small length at maturity, and young maximum ages (e.g., *Stenotomus* Porgies), as well as increases in catch rates of species with old maximum ages and large maximum sizes (e.g., Red Snapper) compared to earlier years. In addition, the period of highest landings for many reef fishes occurred in the 1970s and 1980s prior to when fishery-independent surveys were available, suggesting community changes in these life history traits may have already occurred prior to the 1990s.

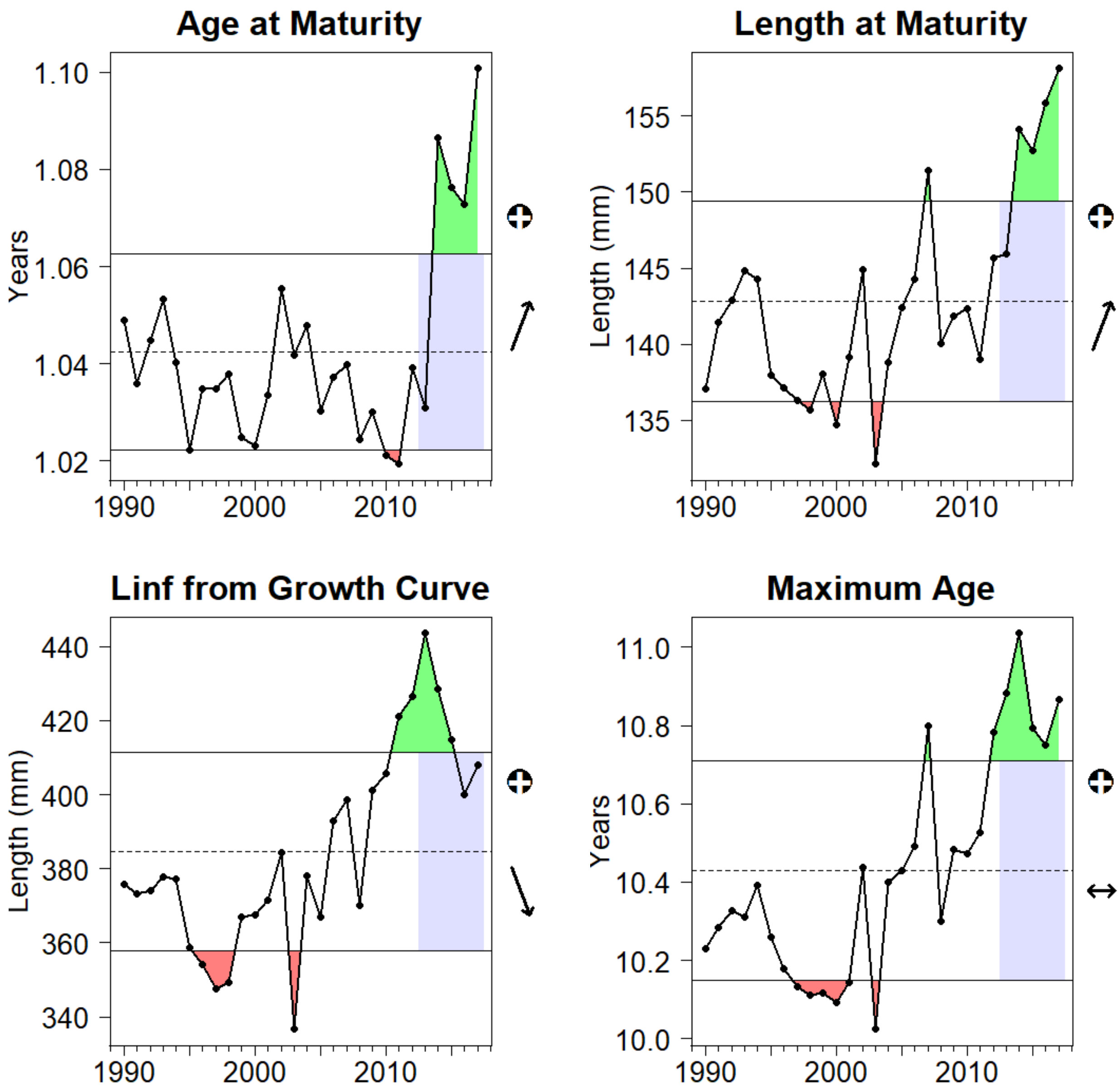


Figure 7.6. Mean life history characteristics of the U.S. South Atlantic reef fish community based on fishery-independent (SERFS) trap survey data.

8. ECOSYSTEM SERVICES

The U.S. South Atlantic supports populations of estuarine species, offshore reef fishes, coastal pelagics, and deepwater species, as well as protected species (e.g., marine mammals, sea turtles) and other species of concern (e.g., marine birds). The management of harvested species in federal waters is supported by stock assessments performed through the Southeast Data, Assessment, and Review (SEDAR) process [218]. Biomass and recruitment from the most recent SEDAR stock assessments are reported for 12 South Atlantic stocks: Vermilion Snapper (*Rhomboplites aurorubens*), Red Porgy (*Pagrus pagrus*), Black Sea Bass (*Centropristis striata*), Golden Tilefish (*Lopholatilus chamaeleonticeps*), Spanish Mackerel (*Scomberomorus maculatus*), Cobia (*Rachycentron canadum*), Red Snapper (*Lutjanus campechanus*), Red Grouper (*Epinephelus morio*), Greater Amberjack (*Seriola dumerili*), Gag Grouper (*Mycteroperca microlepis*), Snowy Grouper (*Hyporthodus niveatus*), and Blueline Tilefish (*Caulolatilus microps*). The terminal year of these assessments varies from stock to stock, depending on when each assessment was conducted. In general, model outputs are subject to a number of uncertainties related to data inputs, model specification, and parameter estimation [219]. In addition to fish stock assessment outputs, trends in stock status, fishery landings and effort, marine mammal strandings, turtle nest counts and relative abundance of multiple bird species are reported from various monitoring programs in the region.

8.1 Biomass of economically important species

Estimates of biomass from stock assessments provide indicators of stock trends over time. Assessment-derived estimates biomass may be preferable to indices from single surveys or fisheries because they integrate multiple data sources, typically cover the entire ontogeny and spatial distribution of the stock, and often have longer time series and a lower noise-to-signal ratio. Estimates of biomass (B) relative to unfished biomass (B0) are reported here for 12 U.S. South Atlantic stocks. These trends do not reflect the status of stocks relative to management benchmarks but, rather, are only used to show relative trends in biomass over time. Most fishery management plans aim for biomass benchmark somewhere between 20 to 50% of B/B0.

Though patterns vary considerably across stocks, the predominant pattern is a decline in biomass starting mostly in the 1970s and 1980s associated with the expanded development of commercial and recreational fisheries in the region (Fig. 8.1). Many of these stocks reached their lowest biomass in the 1990s, and since then have either remained relatively stable (e.g., Vermilion Snapper), continued to decline (e.g., Red Porgy, Gag Grouper) or increased (Greater Amberjack, Cobia). Recent increases for many stocks may reflect the effects of management efforts to limit overfishing and rebuild overfished stocks, which began in the mid-1990s for many offshore reef fishes and later for most coastal pelagics (e.g., Spanish Mackerel) and deepwater species (e.g., Blueline Tilefish).

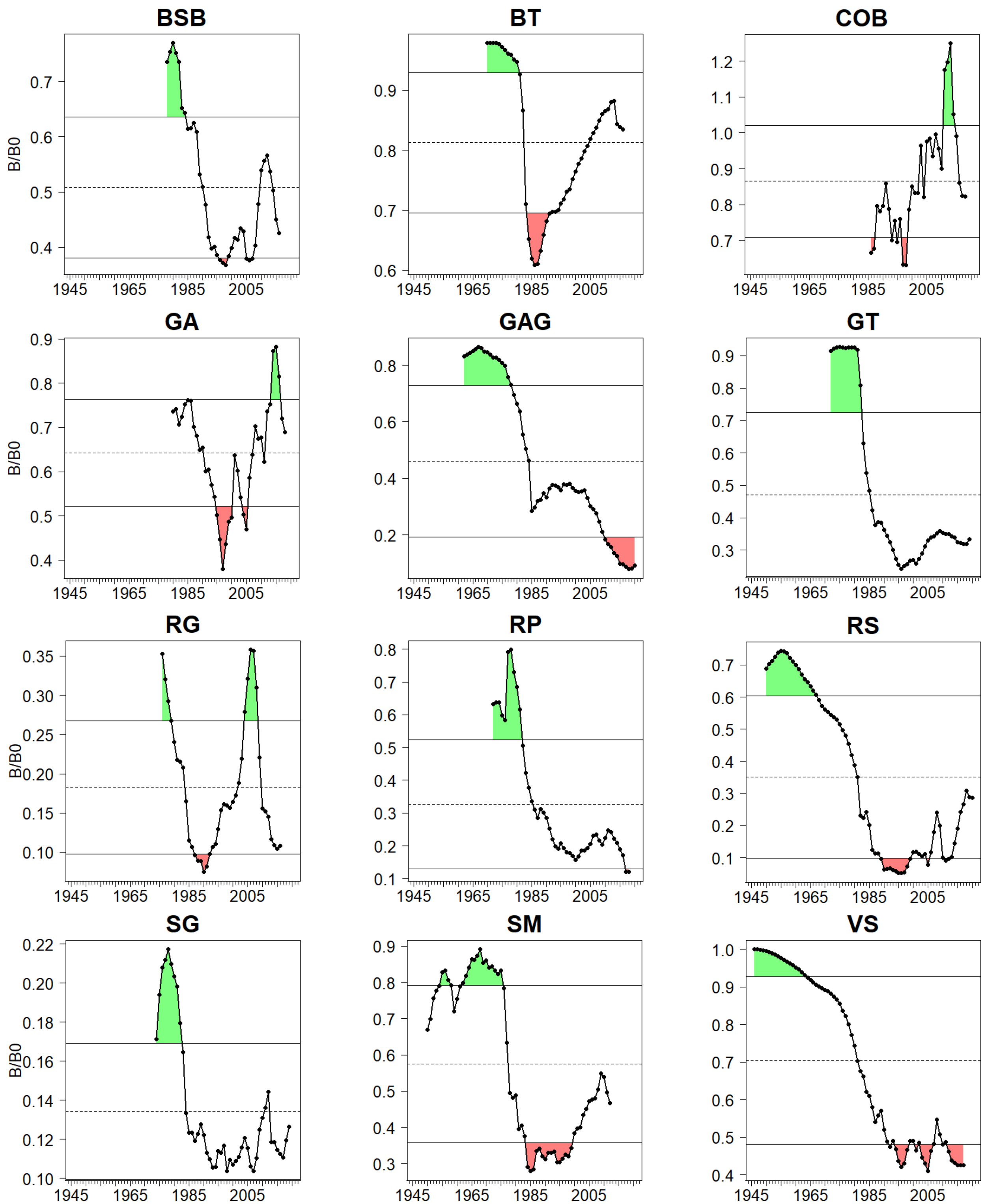


Figure 8.1. B/B0 for economically important, federally-managed South Atlantic stocks based on recent stock assessments. BSB = Black Sea Bass, BT = Blueline Tilefish, COB = Cobia, GA = Greater Amberjack, GAG = Gag Grouper, GT = Golden Tilefish, RG = Red Grouper, RP = Red Porgy, RS = Red Snapper, SG = Snowy Grouper, SM = Spanish Mackerel, and VS = Vermilion Snapper. Note different y-axis scales.

8.2 Recruitment of economically important species

Annual recruitment is a measure of the number of offspring produced each year by the adult stock. For South Atlantic species, annual recruitment is estimated by the stock assessment model as annual deviations around an estimated or assumed stock recruitment relationship for all or part of the assessment period. Variability in recruitment over time is summarized by time series of these recruitment deviations. For three species (Red Porgy, Black Sea Bass, and Spanish Mackerel), recruitment was estimated for age-0 fish in the assessments. For the remaining eight species, recruitment was estimated for age-1 fish, but annual values were shifted one year earlier to represent age-0 fish for consistency among all stocks. Recruitment deviations were not estimated by the Blueline Tilefish assessment model. Recent trends in recruitment were not statistically tested because estimates were rarely available for the most recent five years and recruitment is highly uncertain, and sometimes constrained by the model, for the most recent assessment years.

Recruitment is highly variable both within and among species (Fig. 8.2). Pelagic species, such as Spanish Mackerel, Cobia, and Greater Amberjack, have shown highly variable but generally stable or increasing recruitment in recent years. Within the snapper-grouper complex, Red Snapper has shown increases in recruitment since 2005 while Vermilion Snapper has shown variable but stable recruitment. In contrast, several species have shown declines in recruitment since the early to mid-2000s (e.g., Black Sea Bass, Gag Grouper, Red Grouper), or, in some cases, since the 1990s (e.g., Snowy Grouper, Red Porgy). Potential explanations for these declines include recruitment overfishing, increased predation (e.g., from invasive lionfish), or environmental factors (e.g., changing ocean conditions).

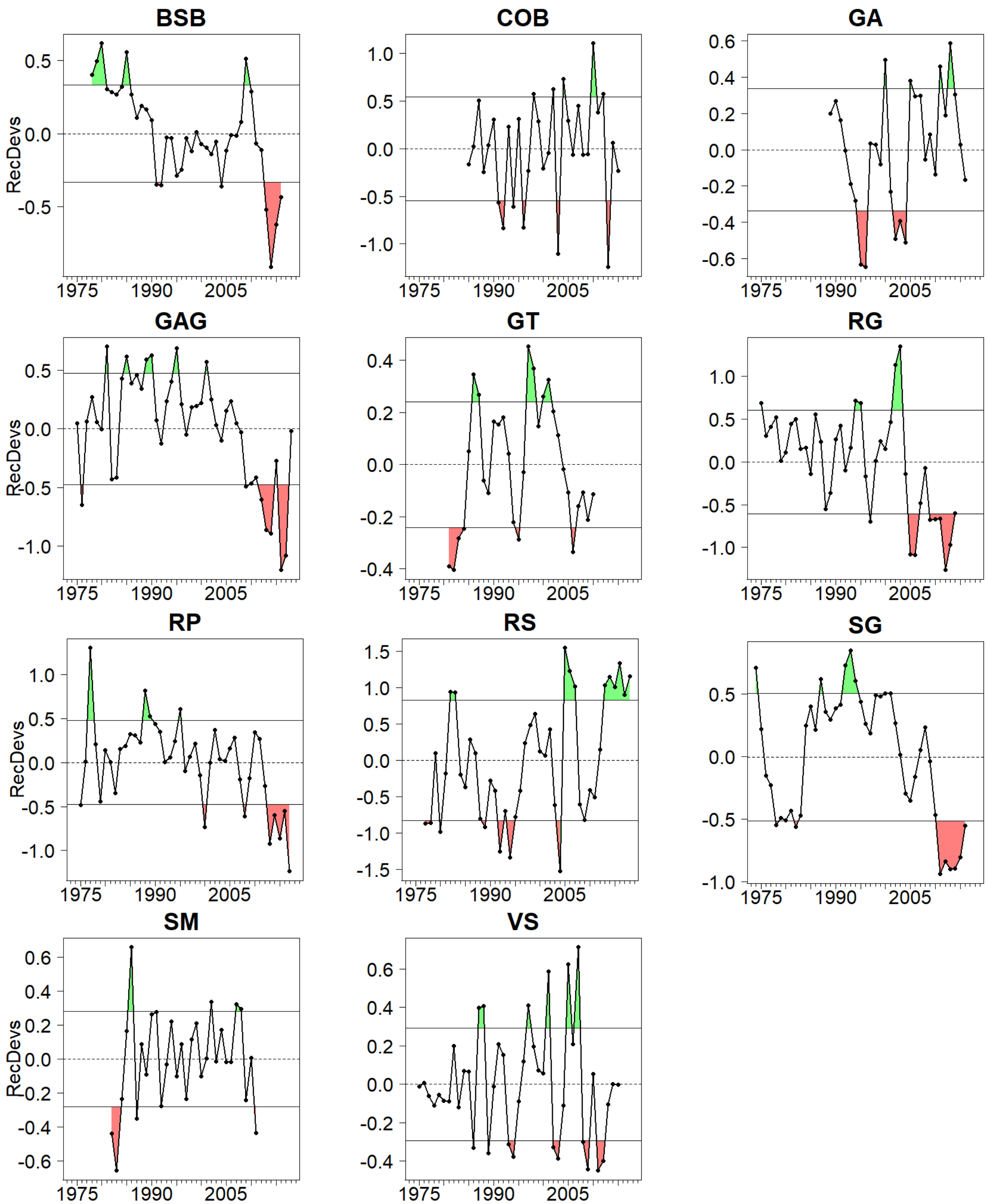


Figure 8.2. Patterns of recruitment (expressed as deviations from the mean recruitment in log space) for economically important, federally-managed South Atlantic stocks based on SEDAR stock assessments. BSB = Black Sea Bass, COB = Cobia, GA = Greater Amberjack, GAG = Gag Grouper, GT = Golden Tilefish, RG = Red Grouper, RP = Red Porgy, RS = Red Snapper, SG = Snowy Grouper, SM = Spanish Mackerel, and VS = Vermilion Snapper.

8.3 Commercial landings and revenue

Total landings from commercial fisheries in the U.S. South Atlantic (Fig. 8.3a) and inflation-adjusted revenues (Fig. 8.3b) are shown from 1950 – 2017. Commercial landings were highly variable prior to the 1980s with peaks in the 1950s and early 1980s, likely driven by domestic

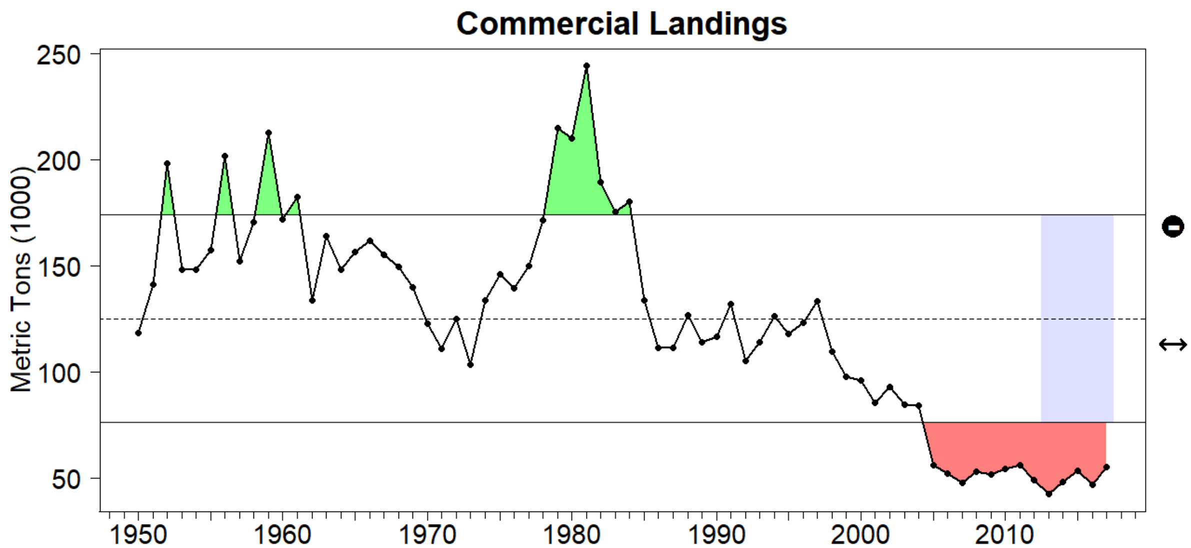


Figure 8.3a. Annual commercial landings for U.S. South Atlantic fisheries.

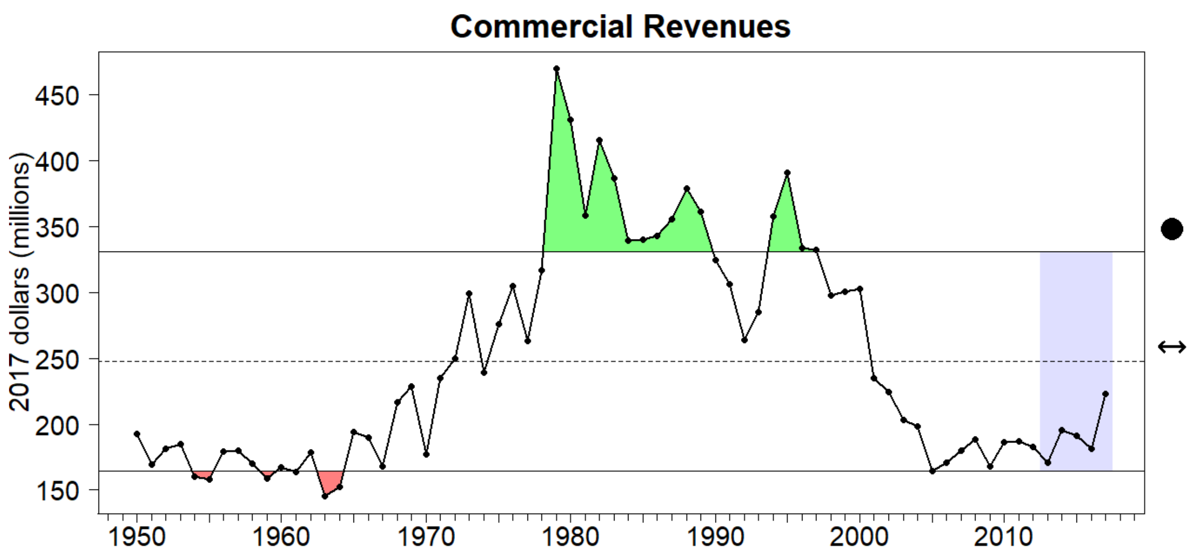


Figure 8.3b. Annual inflation-adjusted commercial revenues for U.S. South Atlantic fisheries.

fishery expansions after World War II and again after the 1976 Magnuson-Stevens Fishery Conservation and Management Act, which promoted the development of domestic fisheries. Commercial landings have declined since the 1980s and are currently less than half of the long-term mean. Commercial revenues rose steadily from the 1950s to 1980s, but have declined similar to landings since the early 1980s (Fig. 8.3b). Over the last five years, commercial revenues are about 25% below their long-term mean (adjusted for inflation).

Composition of the commercial landings is shown for species federally managed under Fishery Management Plans (FMPs) by the South Atlantic Fisheries Management Council (SAFMC) (Fig. 8.3c-e). Landings were categorized into three broad groups: (1) shelf species primarily from the snapper-grouper complex, (2) coastal pelagic species, and (3) deep-water species. Landings of all shelf snapper-grouper species declined since their peaks in the 1980s, with greater than a third of the recent catch comprised of Black Sea Bass and Vermilion Snapper (Fig. 8.3c). Coastal pelagic species showed a similar increase in landings from the 1950s to the 1980s as for the snapper-grouper complex but with more stable landings since then (Fig. 8.3d). Landings of coastal pelagic species were dominated by King Mackerel (*Scomberomorus cavalla*) and Spanish Mackerel *Scomberomorus maculatus)* which comprised > 80% of the landings. Deep-water species were not routinely identified in the landings data until the 1980s and have shown a general decline similar to the shelf species. Snowy Grouper dominated the landings in the early part of the time series but have been replaced by Blueline Tilefish, which have experienced increased harvest since the early to mid-2000s (Fig. 8.3e). Total landings of all three groups have been stable or slightly declining over the most recent five years.

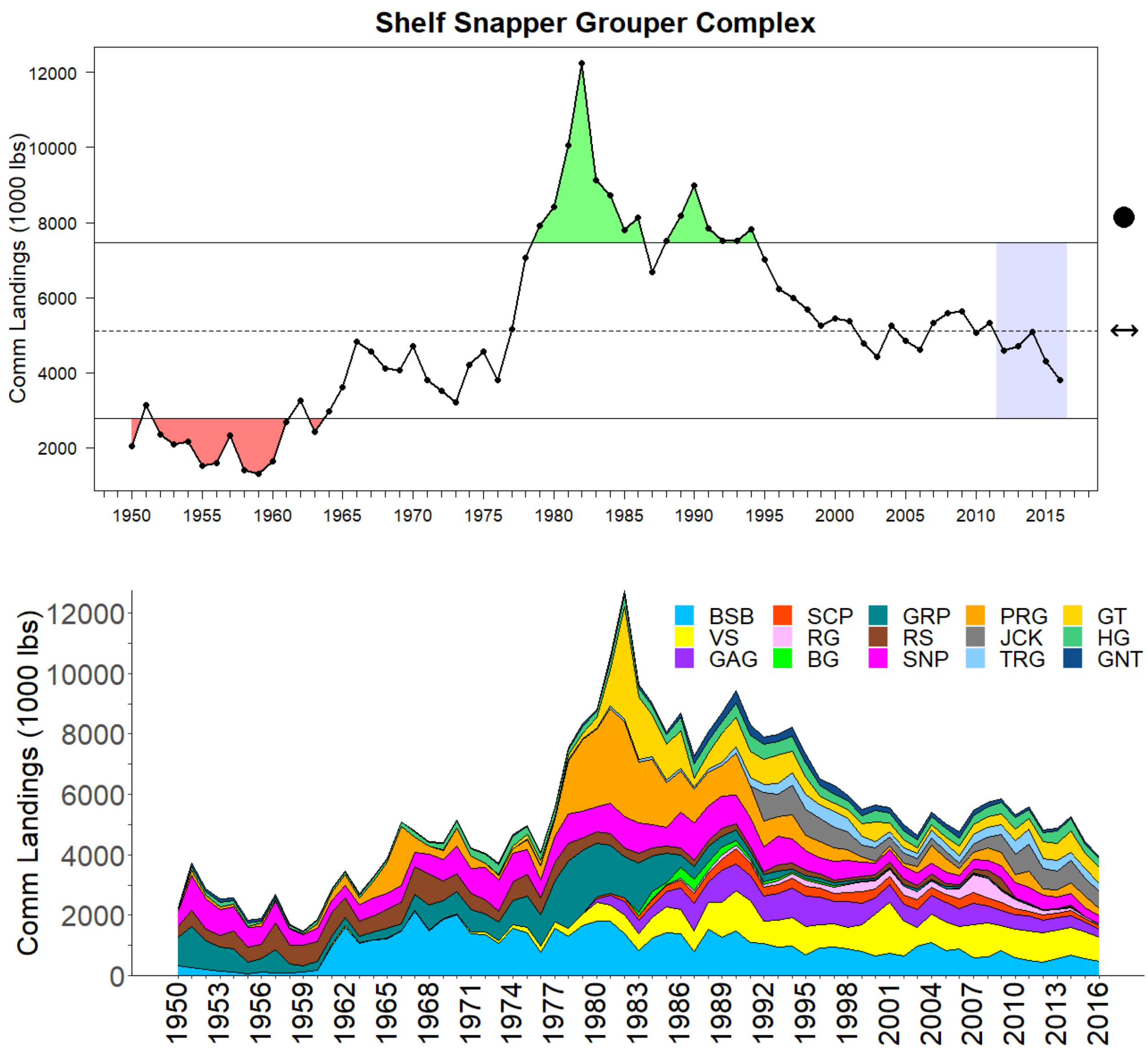


Figure 8.3c. Annual commercial landings (top panel) and species composition (bottom panel) for shelf snapper-grouper species in the U.S. South Atlantic. BSB = Black Sea Bass, VS = Vermilion Snapper, GAG = Gag, SCP = Scamp, RG = Red Grouper, BG = Black Grouper, GRP = unidentified and rare groupers, RS = Red Snapper, SNP = unidentified and rare snappers, PRG = Porgies, JCK = Jacks, TRG = Triggerfishes, GT = Golden Tilefish, HG = Hogfish and Spadefish, GNT = Grunts.

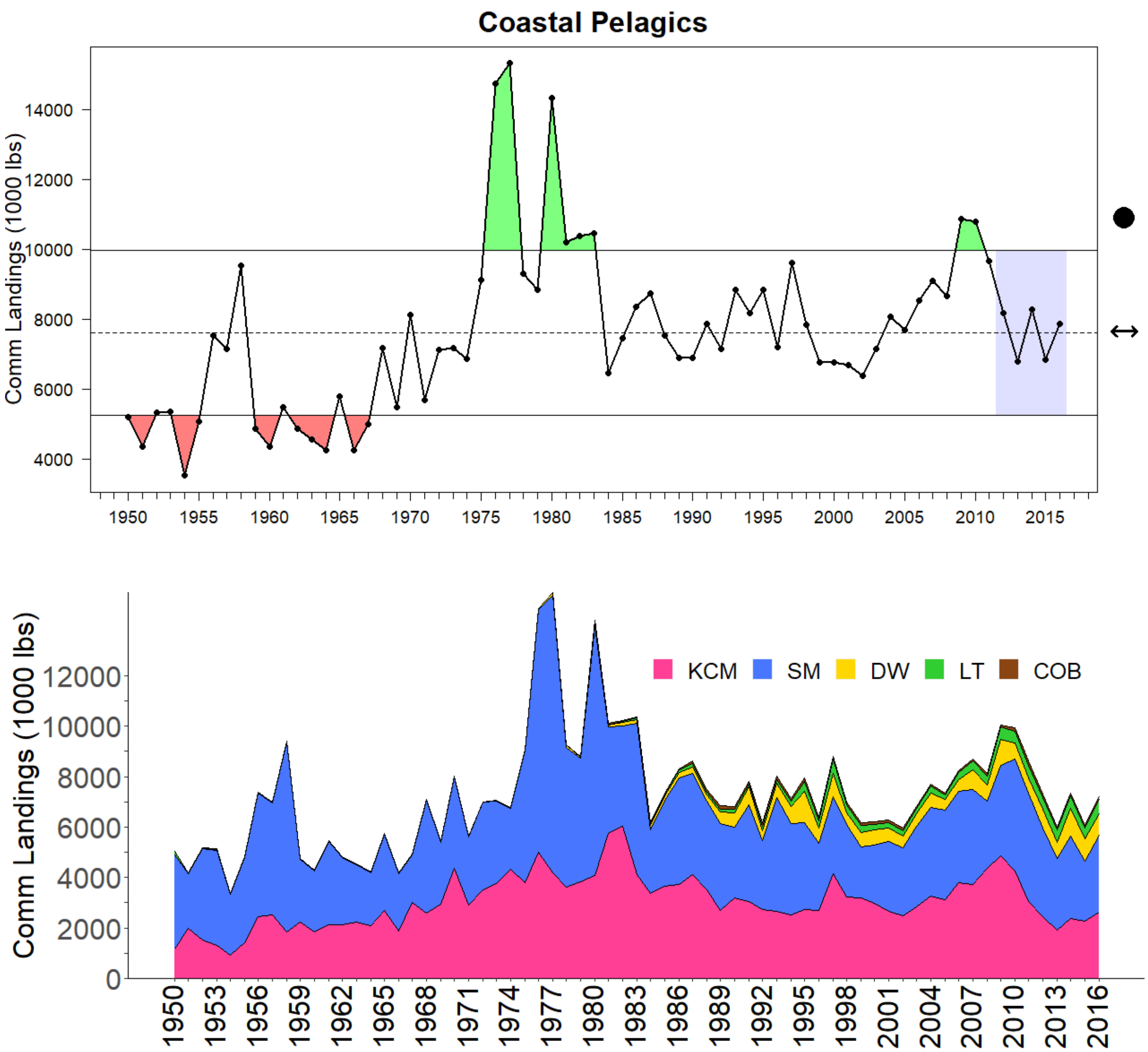


Figure 8.3d. Annual commercial landings (top panel) and species composition (bottom panel) for coastal pelagic species in the U.S. South Atlantic. KCM = King and Cero Mackerel, SM = Spanish Mackerel, DW = Dolphin and Wahoo, LT = Little Tunny, COB = Cobia.

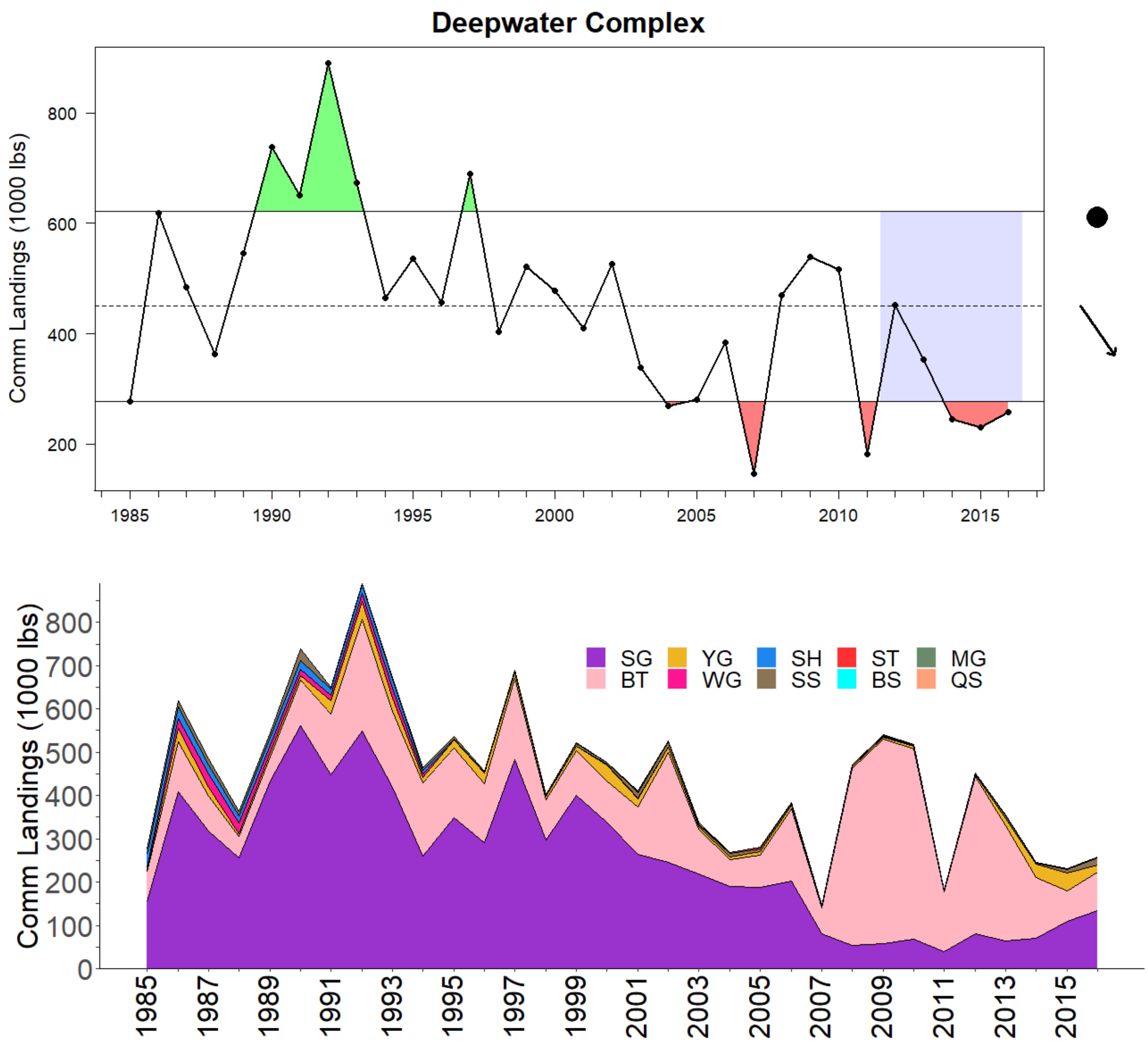


Figure 8.3e. Annual commercial landings (top panel) and species composition (bottom panel) for deep water species in the U.S. South Atlantic. SG = Snowy Grouper, BT = Blueline Tilefish, YG = Yellowedge Grouper, WG = Warsaw Grouper, SH = Speckled Hind, SS= Silk Snapper, ST = Sand Tilefish, BS = Blackfin Snapper, MG = Misty Grouper, and QS = Queen Snapper.

8.4 Recreational landings and effort

Total recreational landings and effort for the U.S. South Atlantic region are shown from 1981 to 2017 based on data from the Marine Recreation Information Program (MRIP [214]). Recreational landings generally increased from the 1980s and 1990s to the 2000s and 2010s, before declining slightly over the last five years (Fig. 8.4a top). Recreational effort, measured as the number of angler trips, increased from the early 1980s to the mid-2000s and has been relatively stable since then (Fig. 8.4a, middle). The proportion of total landings (commercial and recreational) for 22 federally managed species (primarily species in the snapper-grouper complex) from the recreational sector is shown from 1981 to 2016 (Fig. 8.4a, bottom). For all species combined, 71% of the landings from 1981 – 2016 were from the recreational sector [220]. Recreational landings were the dominant portion of the removals for 68% of the individual species. There is an increasing trend in the proportion of the total landings coming from the recreation sector, particularly after 2013, when recreational landings comprised > 80% of total landings.

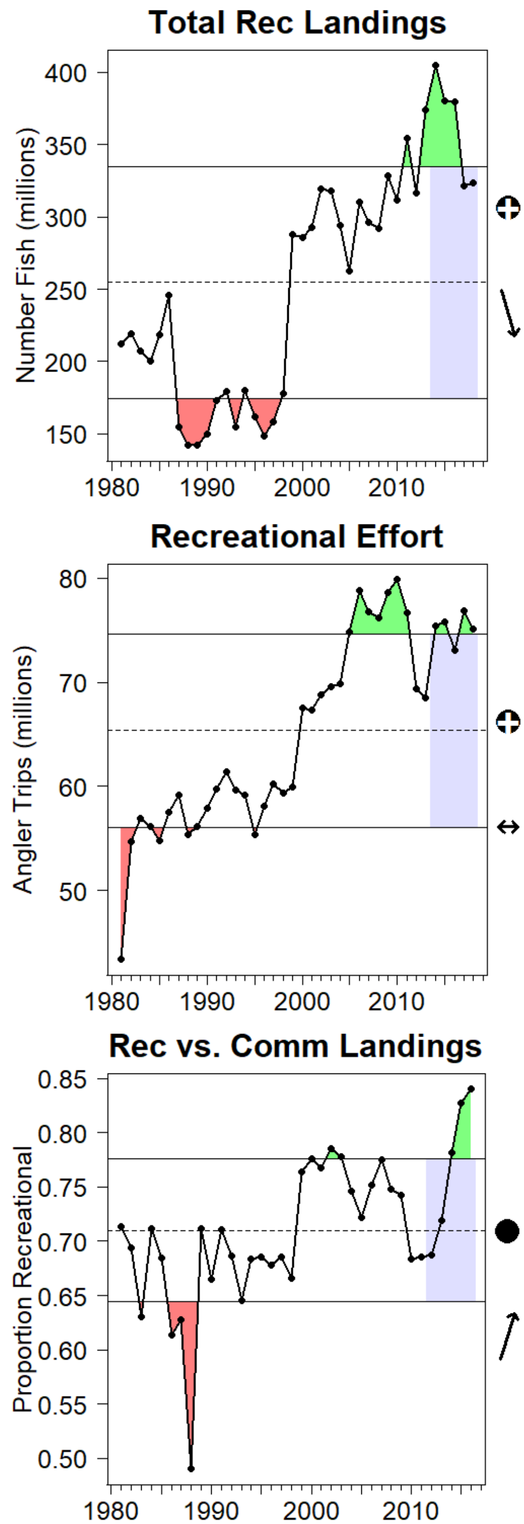


Figure 8.4a. Annual recreational landings (top panel), recreational effort (middle panel), and proportion of landings from the recreational sector (bottom panel) for the U.S. South Atlantic region.

Similar to commercial landings, species composition of the recreational landings was categorized into three broad groups: shelf snapper-grouper, coastal pelagic species, and deep-water species (Figs. 8.4b-d). Overall, recreational landings of shelf snapper-grouper species were highest in the early to mid-1980s, lower but stable through the 1990s and 2000s, and have increased in recent years (Fig. 8.4b). Recreational landings of snapper-grouper are dominated by Grunts, Black Sea Bass, and Gray Snapper. Recreational landings of coastal pelagic species have shown high annual variability but a declining trend over the last five years (Fig. 8.4c). Landings of coastal pelagics are dominated by Spanish Mackerel (*Scomberomorus maculatus*), Blue Runner (*Caranx crysos*), Mahi-Mahi (*Coryphaena hippurus*), and King Mackerel (*Scomberomorus cavalla*). Recreational landings of deep water species were relatively low from the 1980s to the mid 2000s, but have increased over the last 10 years, driven primarily by landings of Blueline Tilefish (*Caulolatilus microps*) (Fig. 8.4d).

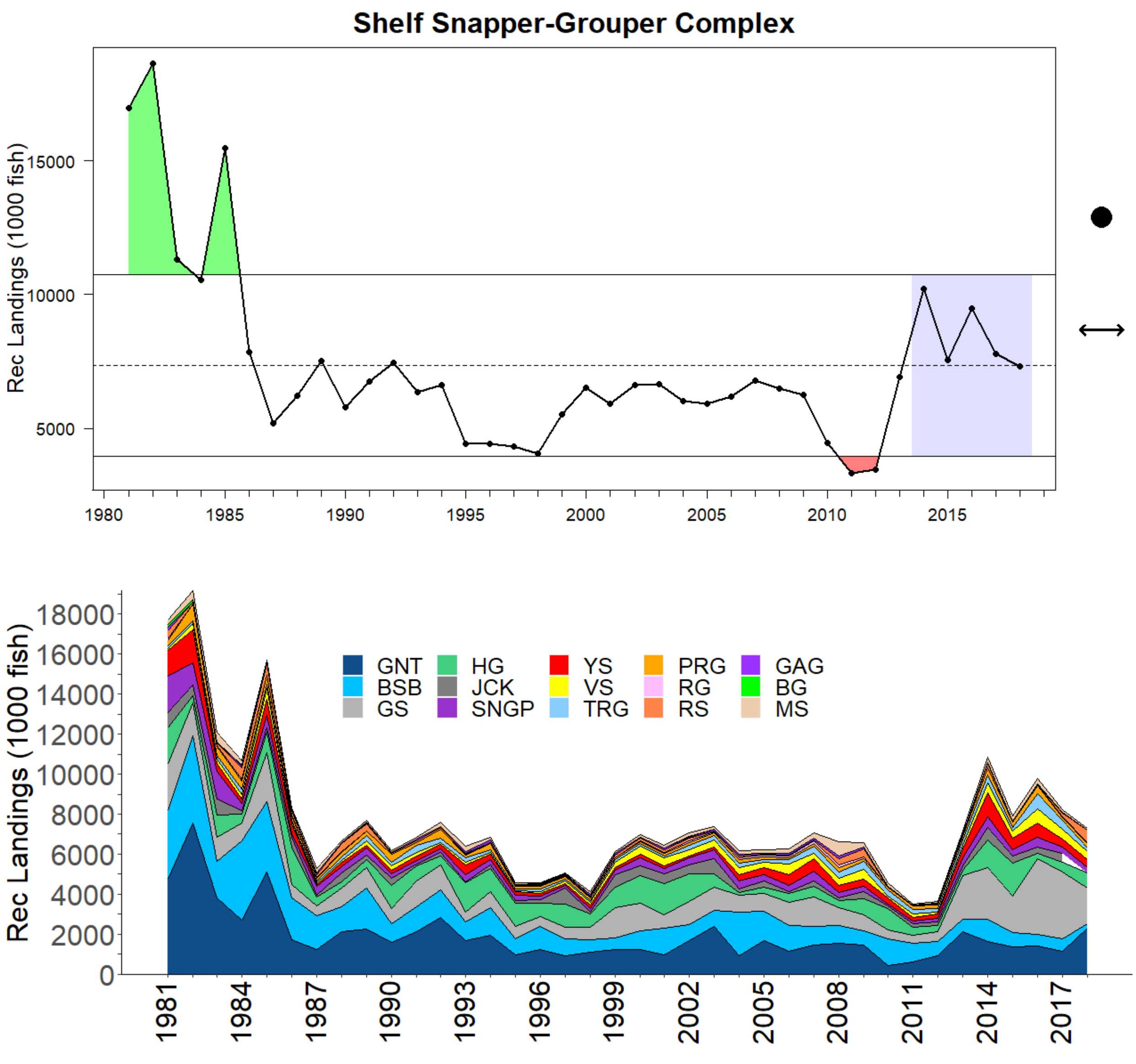


Figure 8.4b. Annual recreational landings (top panel) and species composition (bottom panel) for shelf snapper-grouper species in the U.S. South Atlantic. GNT = Grunts, BSB = Black Sea Bass, GS = Gray Snapper, HG = Hogfish and Spadefish, SNGP = Unidentified and rare snappers and groupers, YS = Yellowtail Snapper, VS = Vermilion Snapper, TRG = Triggerfish, PRG = Porgies, RG = Red Grouper, RS = Red Snapper, GAG = Gag, BG = Black Grouper, and MS = Mutton Snapper.

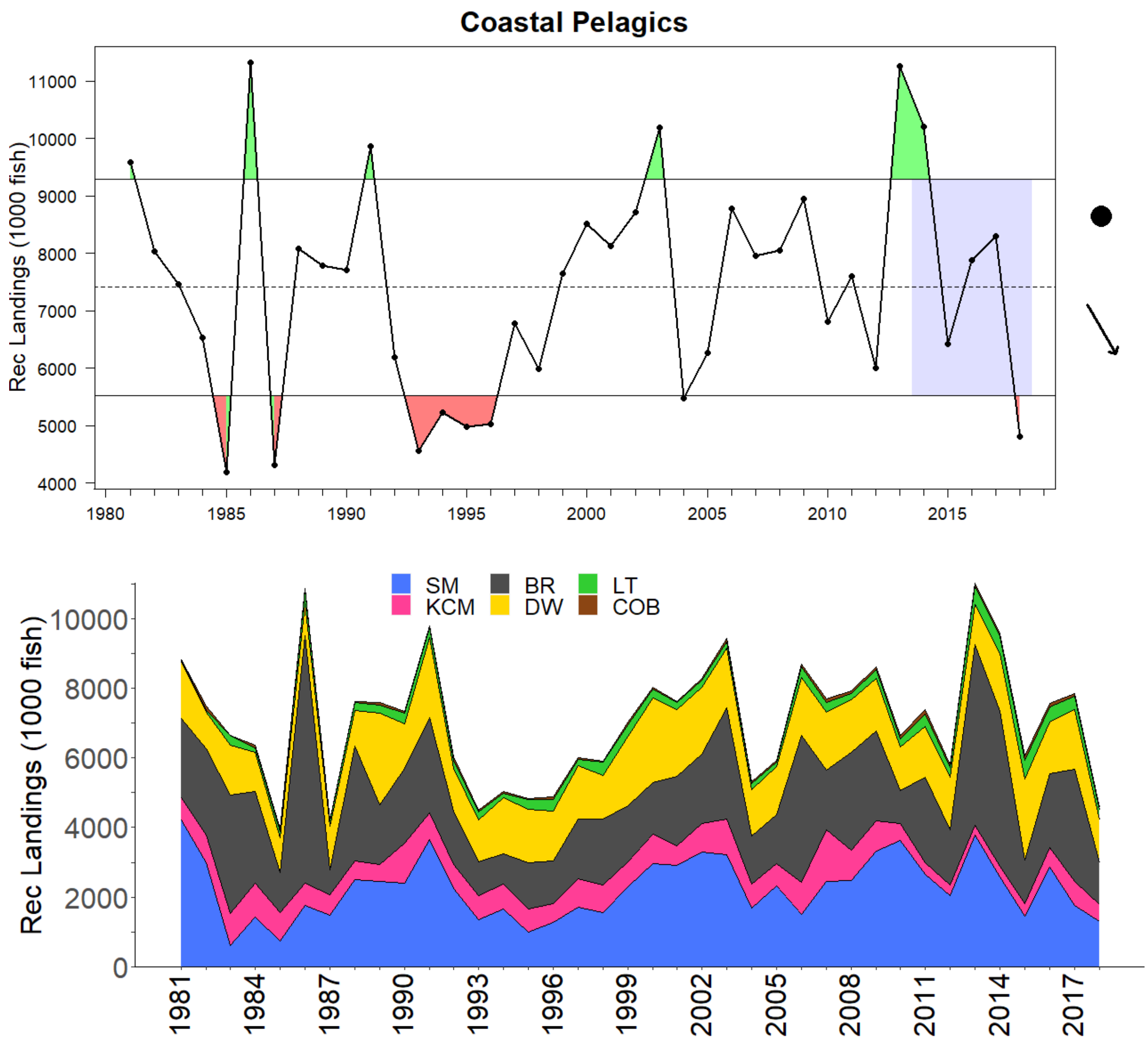


Figure 8.4c. Annual recreational landings (top panel) and species composition (bottom panel) for coastal pelagic species in the U.S. South Atlantic. SM = Spanish Mackerel, KCM = King and Cero Mackerel, BR = Blue Runner, DW = Dolphin and Wahoo, JCK = Jacks, LT = Little Tunny, and COB = Cobia.

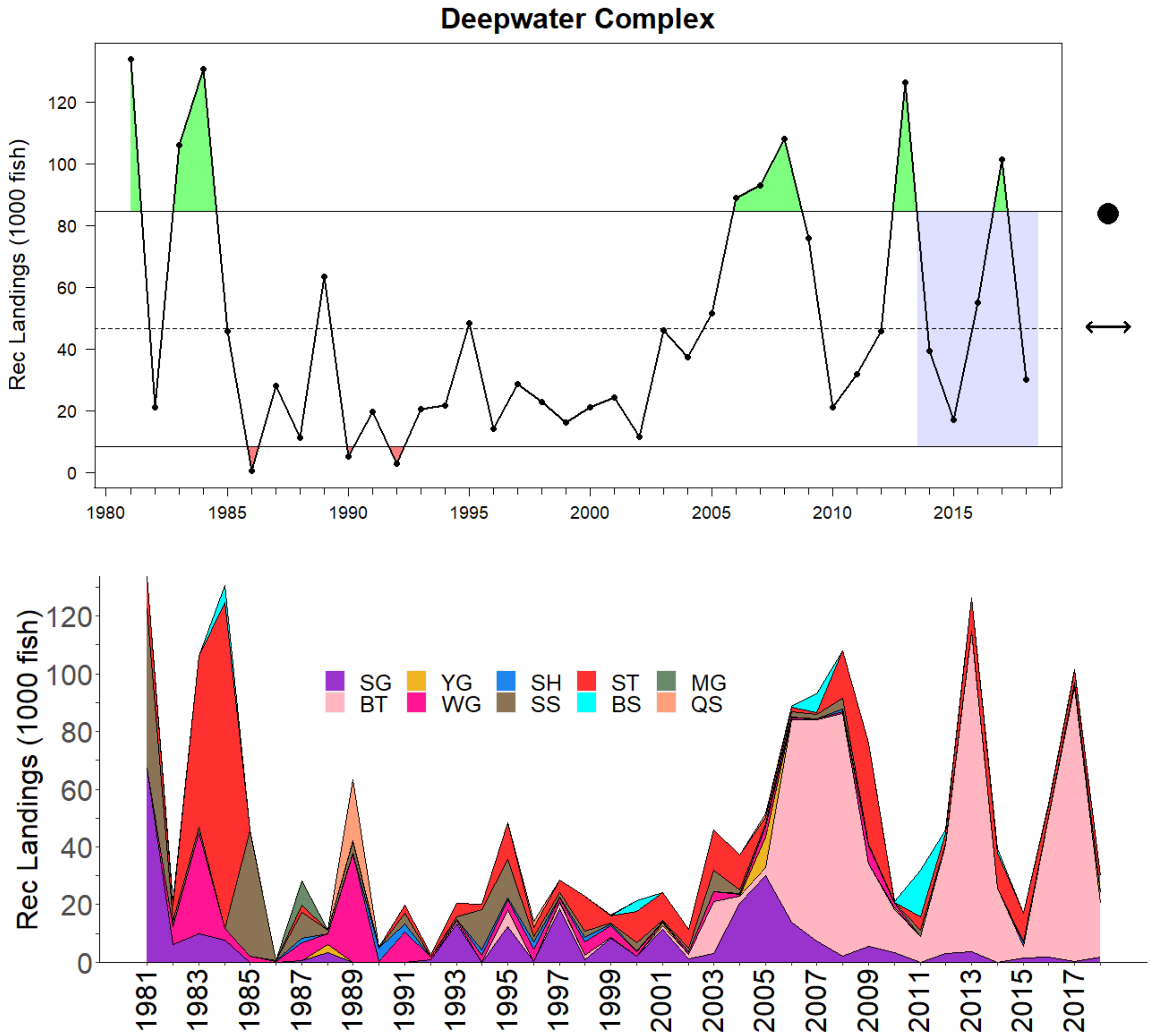


Figure 8.4d. Annual recreational landings (top panel) and species composition (bottom panel) for deep water species in the U.S. South Atlantic. SG = Snowy Grouper, BT = Blueline Tilefish, YG = Yellowedge Grouper, WG = Warsaw Grouper, SH = Speckled Hind, SS = Silk Snapper, ST = Sand Tilefish, BS = Blackfin Snapper, MG = Misty Grouper, and QS = Queen Snapper.

8.5 Estuarine shrimp, crab, and oyster landings

Invertebrate estuarine fisheries comprise a large portion of the economic value of commercial fisheries in the U.S. South Atlantic region. As an indicator of these valuable inshore fisheries, annual commercial landings are reported for Blue Crab and Penaeid Shrimp (Brown, White, and Pink combined) from 1950 through 2018 and for Eastern Oysters from 1991 through 2018. Landings data were compiled from each state (NC, SC, GA, and FL) and do not account for variation among states in reporting requirements or sampling programs, and are not necessarily indicative of trends in population abundance.

Blue Crab landings peaked in the mid to late-1990s and have shown a declining trend since then (Fig. 8.5, top). Most Blue Crab landings occur in North Carolina, which has accounted for approximately 73% of the total Blue Crab landings in the South Atlantic region since 1998. Many of the states from New York to Florida have reported declines in Blue Crab abundance and landings, presumably due to overfishing, environmental factors, or both [221]. Stock assessments indicate Blue Crab are overfished in North Carolina [222] but not on the east coast of Florida [223].

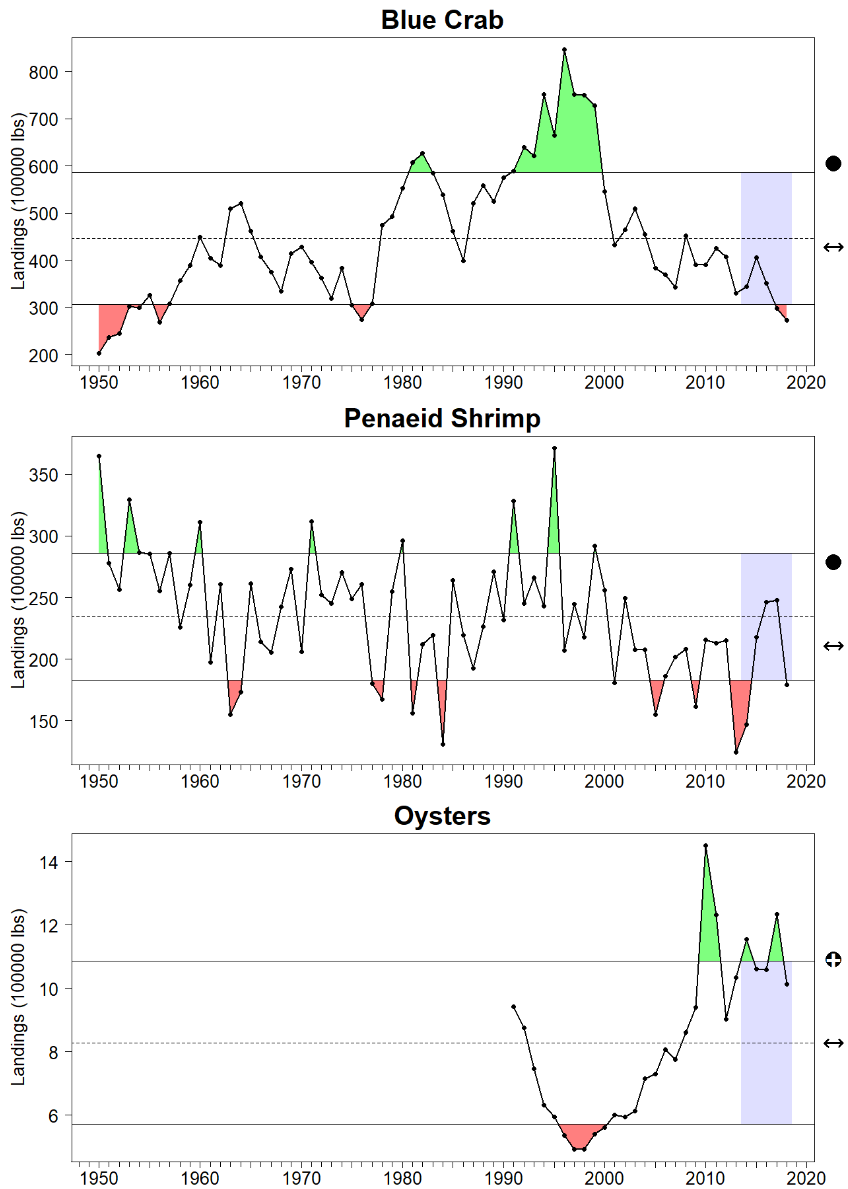


Figure 8.5. Landings of Blue Crab (top panel), Penaeid Shrimp (Brown, White, and Pink combined; middle panel), and Eastern Oysters (bottom panel) in the U.S. South Atlantic. Eastern Oyster landings are in meat weight.

Since its inception in the early 1900s, the Penaeid Shrimp fishery has been one of the most valuable fisheries in the United States, with over 20,000 vessels making greater than 300,000 fishing trips a year and generating over $500 million in landed value during the peak of the fishery [224]. The shrimp fishery primarily targets three species: Brown Shrimp, White Shrimp, and Pink Shrimp. In contrast to the Gulf of Mexico, there is no significant offshore fishery in the South Atlantic, with most of the landings occurring in estuaries or close to shore. Penaeid Shrimp are highly fecund and annual production is driven primarily by environmental factors. For example, White Shrimp landings are often low in years following severe winters, presumably due to cold-water kills during the winter spawning season. Asian Tiger Shrimp (*Penaeus monodon*) have also become established in the South Atlantic Bight in the last 10 years and may represent a threat to native Penaeid Shrimp species [225].

Annual landings of Penaeid Shrimp were stable but highly variable through most of the 1960s and 1970s, peaked in the early to mid-1990s and have declined by 25 – 30% since that time (Fig. 8.5, middle). Recent landings are near or below the long-term average. Downward pressure on ex vessel prices of domestic shrimp due to increasing imports as well as increasing fuel costs are likely contributors to the decline in shrimp landings [224].

Humans have utilized oysters for at least 4,000 years with oyster middens in the Southeast dating back to 2000 BC [226, 227]. The commercial oyster industry began in the late 19th century along the Atlantic seaboard and peaked in the 1930s to 1950s, fueled by a large canning and distributing enterprise. Changes in market demand for canned oysters, rising labor costs, competition from imports, along with overharvest, disease, and degraded water quality contributed to the decline of the commercial oyster fishery in the latter half of the 20th century [148, 149]. Globally, about 85% of oyster reefs have been lost with most population collapses along the eastern seaboard occurring prior to the 1950s [150]. Recreational oyster harvest and oyster culture have become increasingly important in the region. Increasing recognition of the many ecosystem services oysters provide (e.g., shoreline stabilization, improved water quality, and fish habitat) has led to oyster restoration efforts in many states [145, 228].

Developing regional time series of oyster harvest is hampered by differences among states in reporting requirements, variation in the units of landings (e.g., meat weight, shell weight, pounds, bushels) and difficulty in distinguishing wild and cultured harvest. Following a period of relatively low harvest in the 1990s, commercial oyster landings in the U.S. South Atlantic have increased and have been relatively stable at about 1.1 million pounds (meat weight) over the last five years (Fig. 8.5, bottom).

8.6 Status of federally managed stocks

Most fish stocks in federal waters of the U.S. South Atlantic are managed by the South Atlantic Fisheries Management Council (SAFMC) or jointly with the Gulf of Mexico Fisheries Management Council (GFMC). Some species harvested in federal waters are managed by state regulatory agencies in the region (e.g., Atlantic States Marine Fisheries Commission, ASMFC), and a number of highly migratory species are managed via international management authority (e.g., International Commission for the Conservation of Atlantic Tunas, ICCAT). The majority of federally managed species in the South Atlantic (55 species) are contained within the Snapper-Grouper Fishery Management Plan (FMP) with the remainder in the coastal migratory FMP (e.g., Mackerels and Cobia) or various species-specific FMPs (e.g., Dolphin-Wahoo *Acanthocybium solandri*, Golden Crab *Chaceon fenneri*). Currently, 38% of species managed by the SAFMC have stock assessments. While the status of many of the species within the FMPs is not known, those which comprise most of the landings and economic value have been assessed.

The status of federally managed stocks in the U.S. South Atlantic was compiled from annual reports to congress [229], SEDAR stock assessment documents [230], and web-based synthesis tools [231]. The assessment and identification of overfished stocks, implementation of rebuilding plans, and increased regulations over the past several decades have led to a general decrease in the number of managed stocks that are overfished (low biomass) or experiencing overfishing (high mortality) in the U.S. South Atlantic (Fig. 8.6, top panels). While this general trend is clear, species in the South Atlantic are not assessed on a regular schedule, overfishing/overfished definitions have changed over time, and the species composition of stock complexes has sometimes changed (e.g., coastal sharks); therefore, individual years should be interpreted with caution. Black Sea Bass, Vermilion Snapper, Yellowtail Snapper, and Blueline Tilefish are no longer considered overfished or experiencing overfishing in U.S. South Atlantic waters. Even so, a number of stocks remain at low biomass levels despite decreases in fishing mortality (e.g., Snowy Grouper, Red Porgy, Red Grouper), while some particularly long-lived species are considered commercially extinct after high exploitation in the 1970s (e.g., Warsaw Grouper

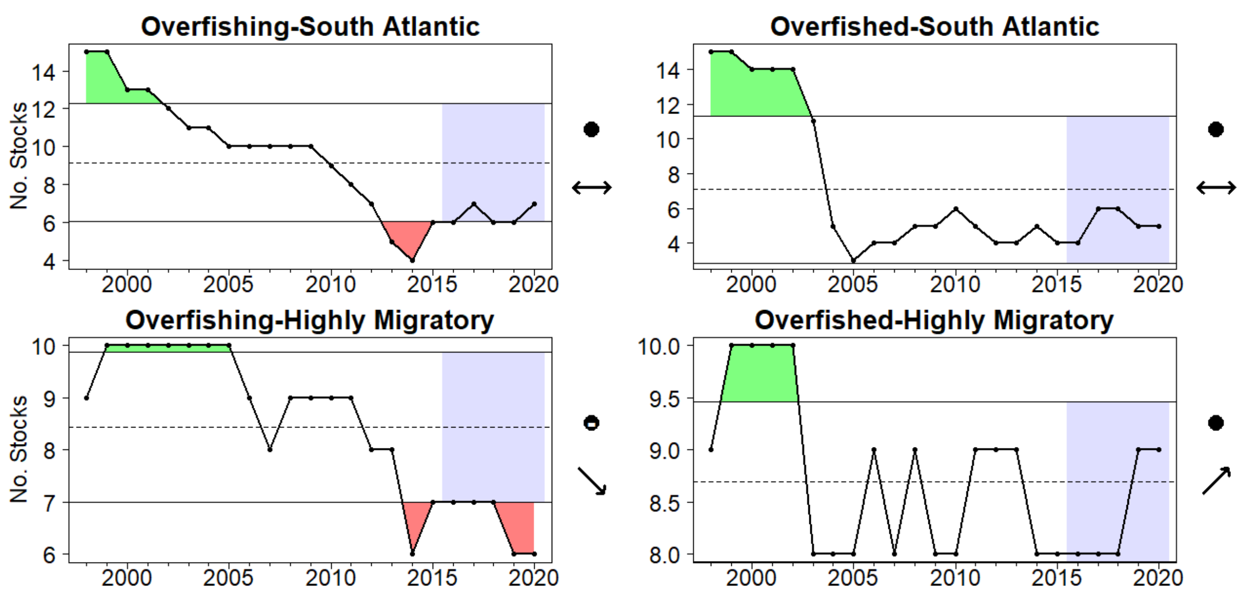


Figure 8.6. Number of assessed stocks that inhabit U.S. South Atlantic waters determined to be undergoing overfishing (left panels) or in an overfished state (right panels).

Hyporthodus nigritus, Speckled Hind *Epinephelus drummondhayi*). Reductions in estimated fishing mortality without corresponding increases in stock biomass may result from environmental effects on recruitment, food web dynamics, lags in re-building due to life history characteristics (old age at maturity, slow growth rate), or continued unintentional (bycatch) fishing mortality.

Highly migratory species (primarily Sharks, Billfishes, and Tunas) that inhabit South Atlantic waters during a portion of their life history have also shown declines in overfishing rates, but in many cases, biomass has not recovered to sustainable levels (Fig. 8.6, bottom). Overall, the number of stocks undergoing overfishing has decreased by two-fold for South Atlantic species and by 50% for highly migratory species, but the response of stock biomass to reductions in fishing pressure has been more moderate.

8.7 Marine bird abundance

Birds are important indicators of the state of coastal ecosystems because they are highly mobile, occupy relatively high trophic levels, use both coastal terrestrial and aquatic habitats, and are important functional components of marine food webs [232, 233, 234]. In addition, some birds have high tourism value [235] or are of conservation concern [236].

Trends in abundance are shown for eight species of marine birds that occur in coastal regions of the Southeastern U.S., are relatively easy to identify, have different life histories, and are typically associated with different coastal habitat types (Fig. 8.7). The Brown Pelican (*Pelecanus occidentalis*) is a relatively long-lived, broadly distributed, year-round resident that forages primarily on fish (particularly Atlantic Menhaden but also bycatch discarded from shrimp trawlers, [237]). Brown Pelicans were federally-listed as endangered in 1970, primarily due to reproductive impairment from exposure to organochlorine contaminants such as DDT, and were de-listed in 1985 [238]. The Double-Crested Cormorant (*Phalacrocorax auritus*) is a colonial nesting waterbird that has undergone a population expansion over the last 30 years, with an increase of overwintering birds in the Southeastern U.S. [239, 240]). The White Ibis (*Eudocimus albus*) and Wood Stork (*Mycteria americana*) are wading birds that are year-round residents and have been the subject of extensive monitoring efforts, particularly in South Florida where they are considered indicators for the restoration of the Florida Everglades System [241]. White Ibis use a widely distributed network of wetlands throughout the region, where they forage on benthic invertebrates in shallow water habitats [242]. Wood Stork (*Mycteria Americana*) were historically most common in the southern part of the region (southern South Carolina to south

Florida) but declined by approximately 75% from the 1930s to the 1980s due to habitat loss and hydrologic alteration of coastal wetland habitats, particularly in south Florida [243]. The species was federally-listed as endangered in 1984, but has since experienced some population recovery and expansion to the north [244]. The Piping Plover (*Charadrius melodus*) is a beach nester that primarily forages on broad, sparsely vegetated beaches, sandbars, and barrier islands [245]. The U.S. Atlantic coast breeding population is currently federally-listed as threatened [246]. American Oystercatcher (*Haematopus palliates*) are also primarily beach nesters but use a broad array of habitats including salt marshes and shell rakes for nesting, and shellfish reefs for foraging [247]. American Oystercatcher are a species of high conservation concern in the eastern U.S. [248], and are also considered a regional-scale indicator of other beach nesting birds [249]. The Clapper Rail (*Rallus crepitans*) is a cryptic species that is considered an indicator of the quality of estuarine marsh habitat due to its strong site fidelity and diet of mostly benthic marsh invertebrates [250]. The Northern Gannet (*Morus bassanus),* the only focal species that does not breed in the region, originates from colonies in

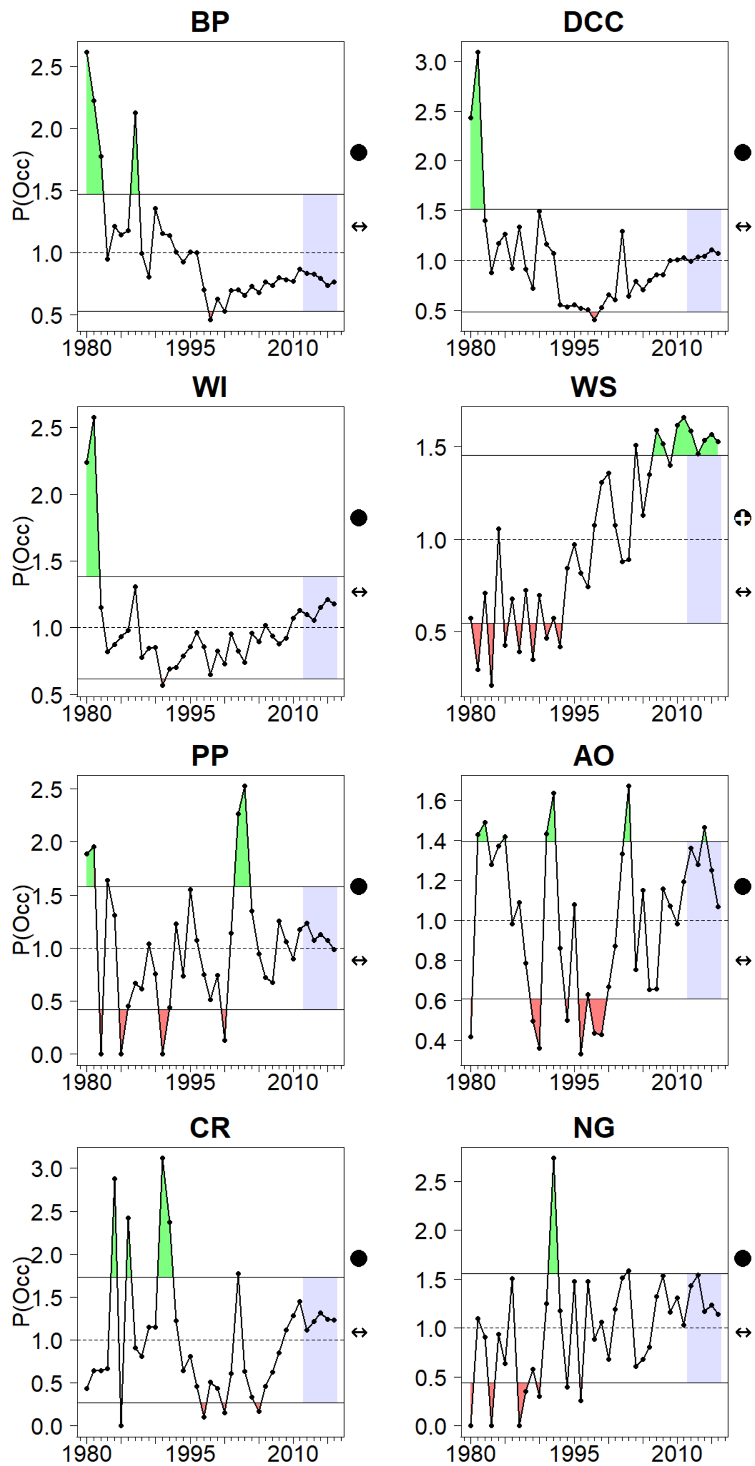


Figure 8.7. Indices of abundance for eight marine bird species in the U.S. South Atlantic based on probability of occurrence (P(Occ)). BP = Brown Pelican, DCC = Double-Crested Cormorant, WI = White Ibis, WS = Wood Stork, PP = Piping Plover, AO = American Oystercatcher, CR = Clapper Rail, and NG = Northern Gannet.

maritime Canada within the western north Atlantic and overwinters along the Atlantic coast of the U.S. and in the Gulf of Mexico [251]. Northern Gannets primarily occur in nearshore (< 20 m depth) coastal habitats where they feed on large schools of pelagic fishes, but also occur offshore and are capable of rapid, long-distance migrations, including trans-Atlantic crossings [252].

The Cornell Lab of Ornithology's eBird Reference Dataset, an extensive, standardized compilation of volunteer and professional bird sighting observations [253, 254], was used to construct indices of abundance for each of the eight species. Observations were limited to sightings reported from coastal watershed counties [3] from North Carolina through the east coast of Florida, and were standardized with respect to sampling effort, time of year, and geographic area using a generalized linear modeling framework.

Abundance patterns varied by species, but generally showed stable or declining trends from the 1980s through the 1990s and stable or increasing trends since the early 2000s (Fig. 8.7). Brown Pelican, Double-Crested Cormorant, and White Ibis showed similar declines beginning in the 1980s and gradual increases beginning in the early 2000s. Wood Stork, a species that has been the focus of considerable conservation effort in the region, increased beginning in the mid-1990s and has been relatively stable since the late 2000s. The remaining four species showed highly variable patterns in abundance prior to the mid-1990s and have been relatively stable since the mid-2000s. Abundances of all eight species in the U.S. South Atlantic coastal region have been relatively stable over the last five years of the times series.

8.8 Marine mammal strandings

Marine mammals are considered indicators of ecosystem change due to their dependence on the lower food web and their susceptibility to anthropogenic activities (e.g. fishing), bioaccumulation of toxins, and vulnerability to disease and other environmental factors [255, 256]. Stock assessments for all marine mammal stocks in U.S. waters are required under the amended Marine Mammal Protection Act of 1994. As of 2017, a total of 13 species that occur partly or wholly in U.S. South Atlantic waters have been assessed by the Southeast Fisheries Science Center, with a number of other species that extend into the South Atlantic from the north assessed by the Northeast Fisheries Science Center [257]. Further, Common Bottlenose Dolphins (*Tursiops truncatus*) are assessed as multiple distinct management units, with 17 recognized stocks in the U.S. South Atlantic.

Strandings data provide information on mortality needed for stock assessments, including identification of unusual mortality events [258], evaluation of anthropogenic activities [259, 260], and better understanding of disease transmission [261]. Strandings are reported by the U.S. marine mammal stranding network, which consists of a number of state, federal, and local government agencies, academic institutions, and the public. While strandings data can be indicative of changes in the ecosystem, they must be interpreted with caution due to variable reporting rates, uneven geographic coverage, and other factors [260, 262]. Collection of strandings data for many states in the South Atlantic region began in the 1970s and early 1980s but monitoring was not widespread and consistent until the 1990s; therefore, analyses were limited to 2000 – 2018 when most states had well-developed monitoring networks. Data reported here are from the NOAA Marine Mammal Health and Stranding Response Program's National Stranding Database [263].

From 2000 to 2018 an average of 359 cetacean stranding events per year involving 31 identified species of whales and dolphins were reported in the South Atlantic. The majority (53 – 88%) of these strandings were Common Bottlenose Dolphins (Fig. 8.8, left). Unusual Mortality Events (UMEs) are defined under the Marine Mammal Protection Act as a stranding that is unexpected, involves a significant die-off of a marine mammal population, and demands immediate response [264]. From 1991 to 2018 there have been 67 UMEs in the U.S. involving 17 species and multiple stocks, 20% of which have occurred partially or wholly within the South Atlantic region [264]. While the exact cause of many of these events is undetermined, infectious disease, biotoxins, and fishery interactions are the suspected or known cause in many cases. *Morbillivirus*, an infectious disease that affects many cetacean species, was the cause of a 2013 Common Bottlenose Dolphin UME [261], during which reported strandings increased three-fold compared to other years. The 2005 peak in strandings of marine mammals other than Common Bottlenose Dolphins (Fig. 8.8, right) was driven by strandings of Rough-Toothed Dolphins (*Steno bredanensis*), Harbor Porpoise (*Phocoena phocoena*), Short-Finned Pilot Whales (*Globicephala macrorhynchus*), and Pygmy Sperm Whales (*Kogia breviceps*). The latter three species also dominated the 2003 peak in strandings. The 2005 event of Harbor Porpoise strandings was a declared UME, with 43 strandings documented along the North Carolina coast [258]. Strandings of Common Bottlenose Dolphins as well as other cetaceans have been near their long-term mean in recent years.

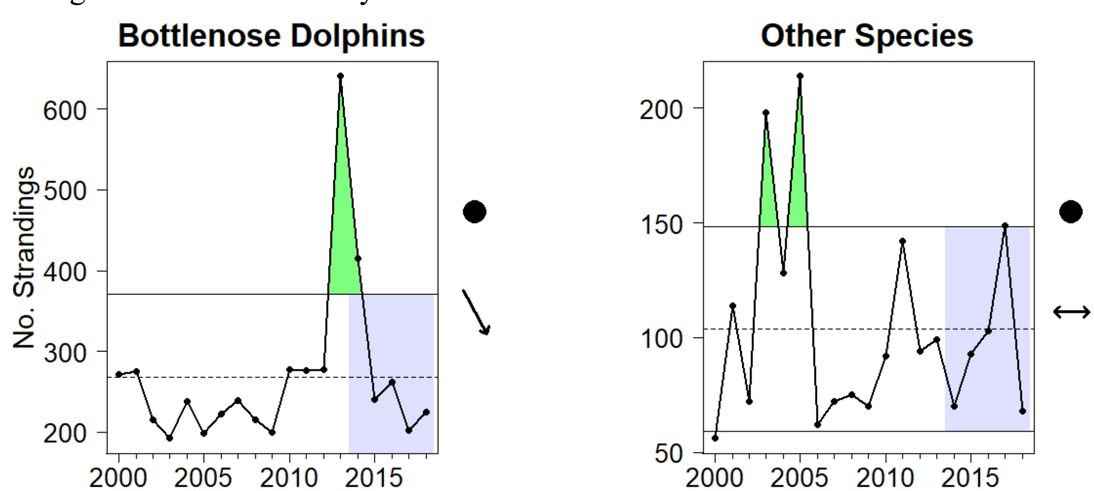


Figure 8.8. Annual reported Common Bottlenose Dolphin strandings (left panel) and other strandings (other cetaceans and pinnipeds, right panel) in the U.S. South Atlantic region.

8.9 Sea turtle nest counts

Five species of sea turtles nest on beaches along the coast of the U.S. South Atlantic. Loggerheads (*Caretta caretta*), Green Turtles (*Chelonia mydas*), and Leatherbacks (*Dermochelys coriacea*) regularly use coastal beaches and barrier islands in the region [265], with sporadic nesting by Kemp's Ridleys (*Lepidochelys kempii*; [266]) and Hawksbills (*Eretmochelys imbricata*; [267]). Of these species, loggerheads are by far the most abundant, with nesting in the southeast U.S. representing one of the two largest aggregations worldwide [268, 269]. Major threats to sea turtles include shoreline modification that alters nesting beaches, watercraft strikes, artificial lighting on beaches, fishery interactions with net and line gear, and occasional disease outbreaks and cold stuns. All five species of sea turtles are currently listed as either threatened or endangered under the U.S. Endangered Species Act.

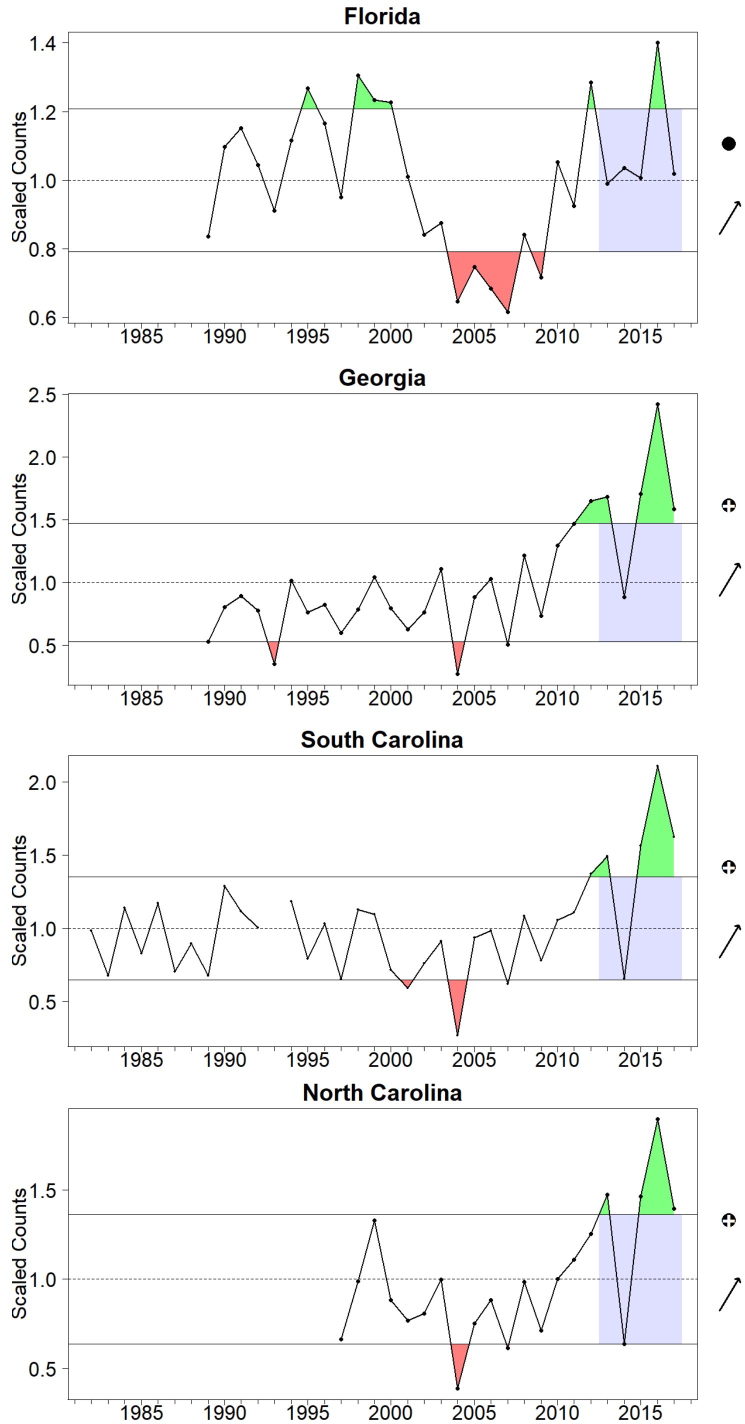


Figure 8.9. Nest counts of Loggerhead Sea Turtles from statewide monitoring programs. Annual counts were scaled to their long-term mean. Florida nest counts include the Florida Keys (Monroe County).

Statewide sampling programs to monitor trends in sea turtle nest counts on coastal beaches and barrier islands in the U.S. South Atlantic region began in the 1980s and 1990s. Counts of sea turtle nests provide an index of annual population productivity and an approximate index of adult female abundance, but are also affected by variation in remigration interval and number of nests per female [269]. Variation in annual sea turtle nest counts integrate demographic and ecosystem processes over temporal scales of at least 2 – 4 years. Loggerheads spend a considerable portion of their life history in South Atlantic waters, foraging primarily on the continental shelf and within estuaries during both the juvenile and adult life stages [270, 271, 272].

Annual nest count data for Loggerheads collected by standardized sampling protocols at index sites were provided by each state and then scaled to their respective long-term mean (Fig. 8.9). There has been a similar and increasing trend in Loggerhead nest counts since the mid-2000s across the four states. The decrease in Loggerhead nest counts during the early 2000s was also documented for several east coast Florida beaches and attributed to a decline in adult females in the population [268].

9. HUMAN DIMENSIONS

9.1 Human population

The rapidly growing human population in the U.S. Southeast is likely to increase pressure on coastal ecosystems in the region. Historical (since the 1800s) population estimates for the coastal region were developed from National Historical Geographic Information System data for states, cities, and metropolitan areas and interpolated to census tract definitions for visualization [273]. These population estimates used the current (as of 2010) census geographies integrated using areal interpolation from each decennial count in order to maintain geographic consistency where census tract boundaries may have shifted over time.

Maps of population density for the U.S. South Atlantic region in 1800, 1900, 1950, and 2010 illustrate the long-term population change that has occurred along the U.S. South Atlantic coast (Fig. 9.1a). The early enslaved peoples and plantation-based economy drove initial settlement in areas surrounding Charleston, South Carolina and Savannah, Georgia [274]. These port cities grew into important economic regions, first based around shipping industries and later around banking that led to increased economic diversification. By the 1950s cities in the South Atlantic such as Jacksonville, Florida and Miami, Florida began growing rapidly with the expansion of agricultural industries. A key turning point in re-development occurred in 1948 when Congress authorized the Central and South Florida Project, with the resulting road infrastructure and water-control systems bringing the potential for more urbanization and development. Migration and immigration drove population growth since the 1950s and created a large population boom in Florida. Additionally, Miami grew into a major metropolitan area with strong economic ties to Central and South America.

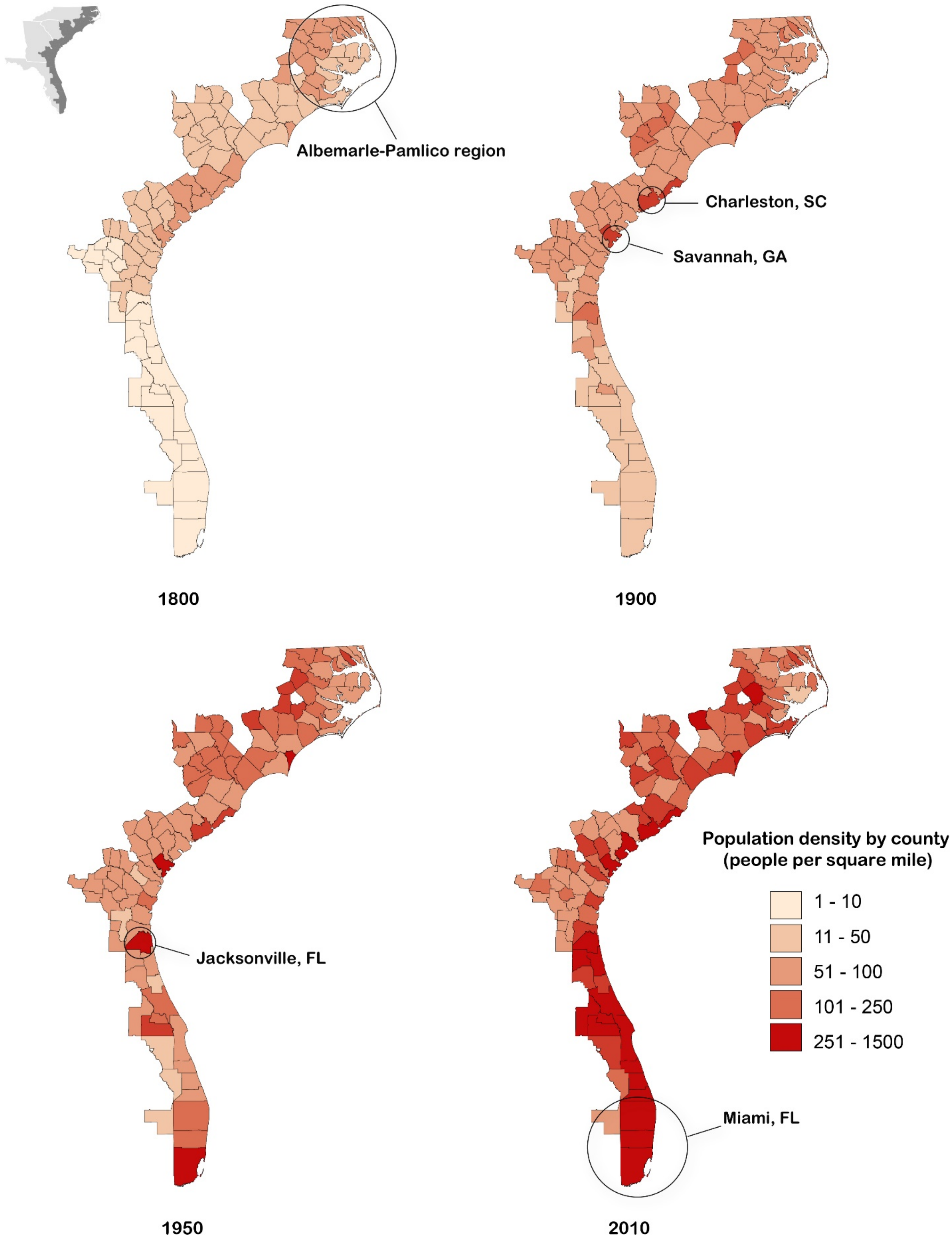


Figure 9.1a. Population density of coastal counties in the U.S. South Atlantic region from 1800 – 2010. Dark colors = high density, light colors = low density.

Based on 2019 American Community Survey (ACS) population estimates, there are approximately 48 million people living in the four U.S. South Atlantic states (FL, GA, SC, NC) (Fig. 9.1b). Florida is the most populous state in the region with multiple coastal counties each inhabited by more than one million people (Broward, largest city: Fort Lauderdale; Miami-Dade, largest city: Miami; Orange, largest city: Orlando; and Palm Beach, largest city: West Palm Beach).

Average population density of the 105 coastal and watershed counties in 2019 is shown in Fig. 9.1c. A majority of watershed counties in the Southeast gained population from 2000 to 2019. Of those counties experiencing a net loss in population, the total loss per county was on average around 2000 persons, whereas the average gain across counties that grew in population was around 50,000 persons. Florida has five urban counties, defined as over 1000 people per square mile, including the major metropolitan cities of Miami (Miami-Dade County), Orlando (Orange

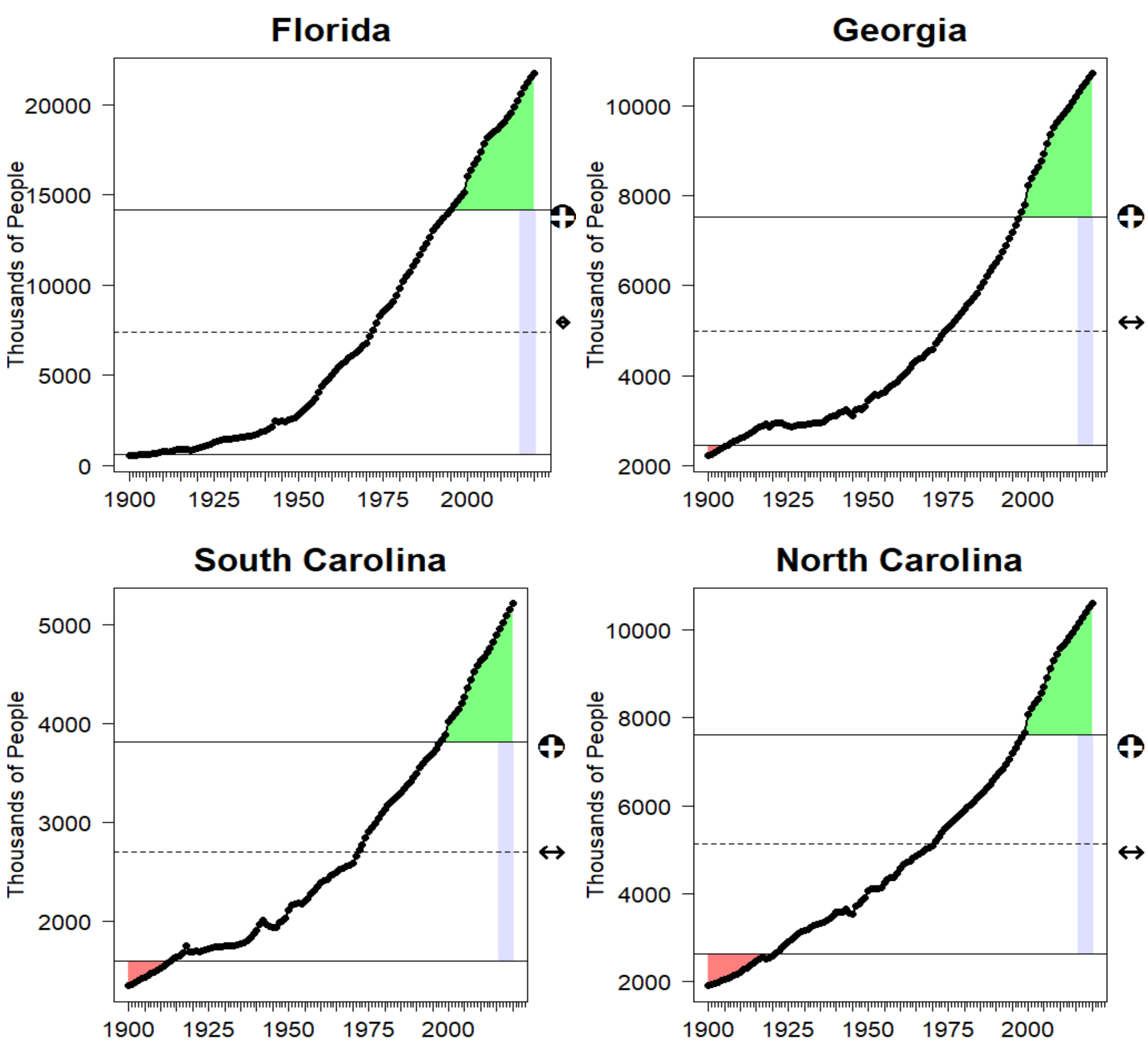


Figure 9.1b. Total population for states bordering the U.S. South Atlantic.

County) and Jacksonville (Duval County). Coastal population density in Florida has doubled since 1980 and has increased by more than 50% in the other three states.

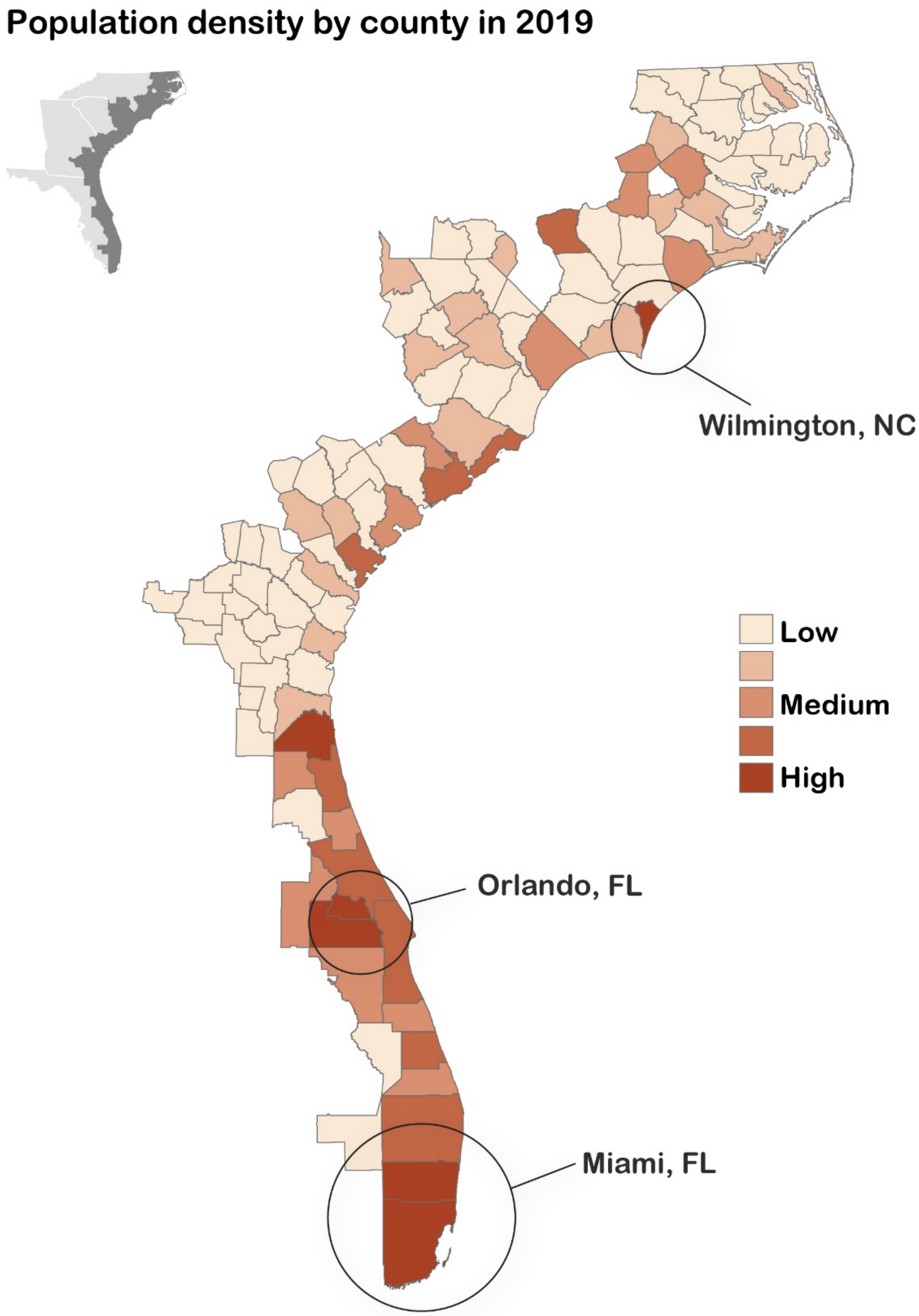


Figure 9.1c. Average population density (people per mi^2) in 2019 across coastal counties in the U.S. South Atlantic from the American Community Survey. Dark colors = high density, light colors = low density.

9.2 Coastal and urban land use

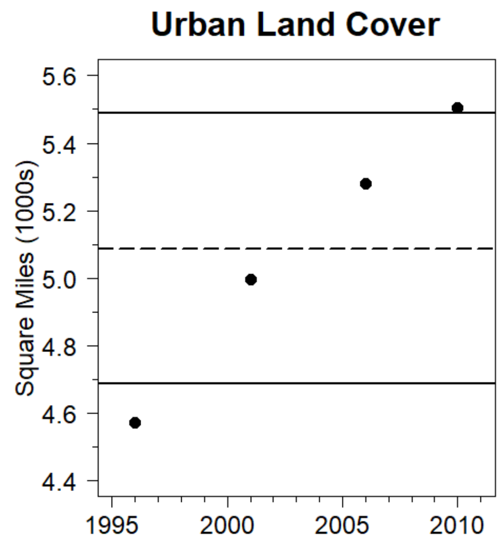


Figure 9.2a. Total area of urban land cover across coastal counties in the U.S. South Atlantic region. The dashed line represents the mean and the solid black lines represent the mean ± 1 standard deviation.

Land cover change derived from satellite imagery is an important indicator of coastal development in the U.S. South Atlantic region. Data on land cover change were taken from the Coastal Change Analysis Program (C-CAP) [137]. In addition to providing an indicator of coastal development, land cover change data is also a useful indicator of ecosystem services provided to humans [276]. Urban land cover classes for this analysis included low-intensity development (21 – 49% impervious surfaces), medium-intensity development (50 – 79% impervious surfaces), high-intensity development (80 – 100 % impervious surfaces), and developed open spaces. Urban land cover represents a relatively small amount of the total land cover in South Atlantic coastal counties, totaling approximately 8% of land area. However, this land use type exerts significant pressure on the ecosystem due to a number of effects, including a high degree of impervious surfaces, pollutant runoff, and high human population density.

Urban land cover increased in the South Atlantic between 1996 and 2010 at an average rate of 1.2% per year, from a total land area of about 4,500 square miles to more than 5,500 square miles (Fig. 9.2a). However, this increase in urbanization showed high spatial heterogeneity, with the highest levels of urbanization occurring along the Florida coast (Fig. 9.2b). The North Carolina and South Carolina coasts also exhibited localized areas with high rates of urbanization.

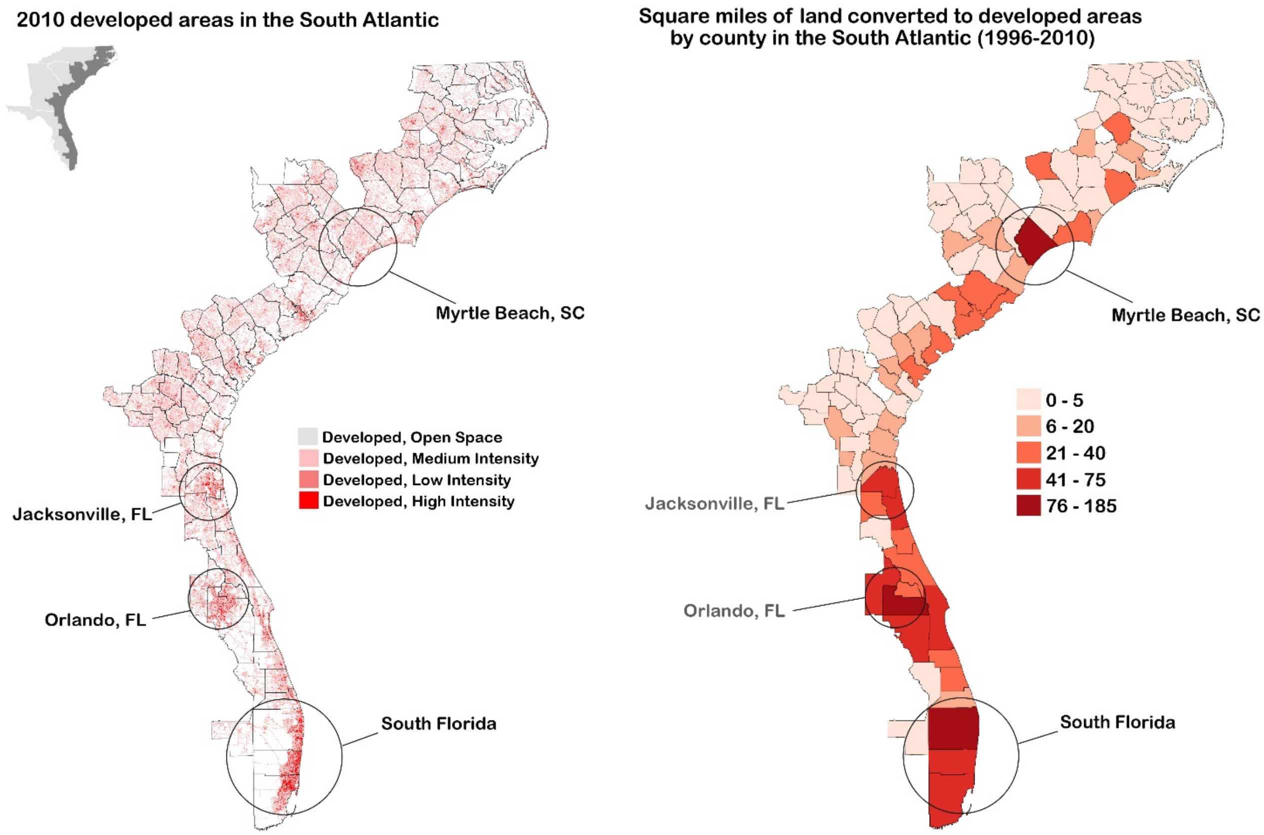


Figure 9.2b. Degree of development in 2010 (left panel) and total area of land converted to development between 1996 and 2010 (right panel) for U.S. South Atlantic coastal counties.

9.3 Total ocean economy

Data on the coastal economy (employment; gross domestic product, GDP) are collected by the Bureau of Labor Statistics and summarized for coastal counties as part of the Economics National Ocean Watch (ENOW) program [275]. The total ocean economy includes six categories: living resources, marine construction, marine transportation, offshore mineral resources, ship and boat building, and tourism and recreation. Coastal counties in the South Atlantic region collectively contributed about 600,000 jobs and over 35 billion dollars in gross domestic product (GDP) to the U.S. economy in 2014 (Fig. 9.3). Both employment and ocean-related GDP in coastal counties declined by 5 – 6% from 2007 to 2009 associated with the 2008 recession, but have since returned to or exceeded previous highs. Ocean-related employment is dominated by tourism and recreation, which accounted for 84% of total ocean-related employment in 2014. Because tourism and recreation are primarily driven by consumer spending, these patterns mirror overall economic growth (e.g., 2008 crisis and then recovery). Marine transportation (predominantly the transport of goods) is the second largest source of employment, accounting for 11% of employment in 2014. While GDP is an indicator of the overall size of the coastal economy, employment is a better indicator of how well economic growth is distributed across different population sectors. Both indicators have shown an upward trend over the last five years of the time series.

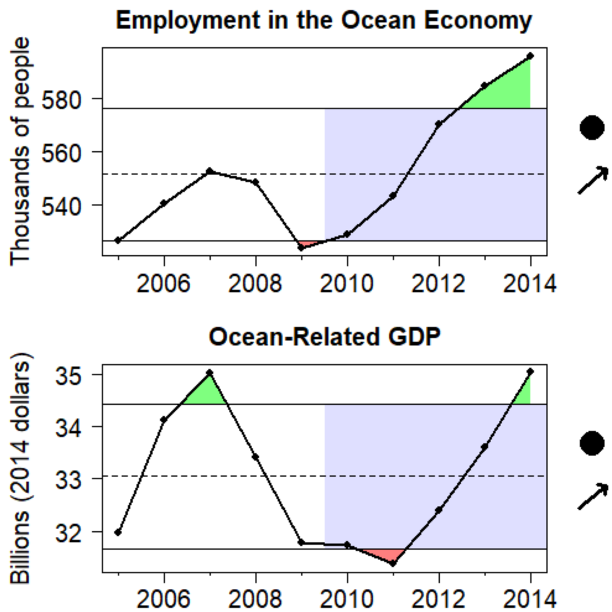


Figure 9.3. Ocean-related employment (top panel) and gross domestic product (adjusted to 2014 dollars; bottom panel) across the U.S. South Atlantic region.

9.4 Social connectedness

Social connectedness is the availability of social capital to the residents of a community and reflects the ability of people to rely on friends and neighbors for assistance, particular in times of need [276]. Social connectedness facilitates the efficient use of ecosystem services provided by the local environment, and is critical to recovery after hurricanes and other natural disasters, as it allows communities to share limited resources [277]. From a geographic perspective, social connectedness also reflects the importance of place attachment, which is associated with positive mental health outcomes and stewardship of natural and cultural resources in the community.

Social connectedness is measured as a composite indicator of six individual attributes of well-being in coastal communities: Participation in democracy, number of social gathering places, number of arts and cultural organizations, charitable giving, access to communication, and tenure in the community [278]. The social connectedness score for each county is the average value of each of the individual measures normalized to a scale of 0 – 1 (note this means the score is relative to other counties in the study region).

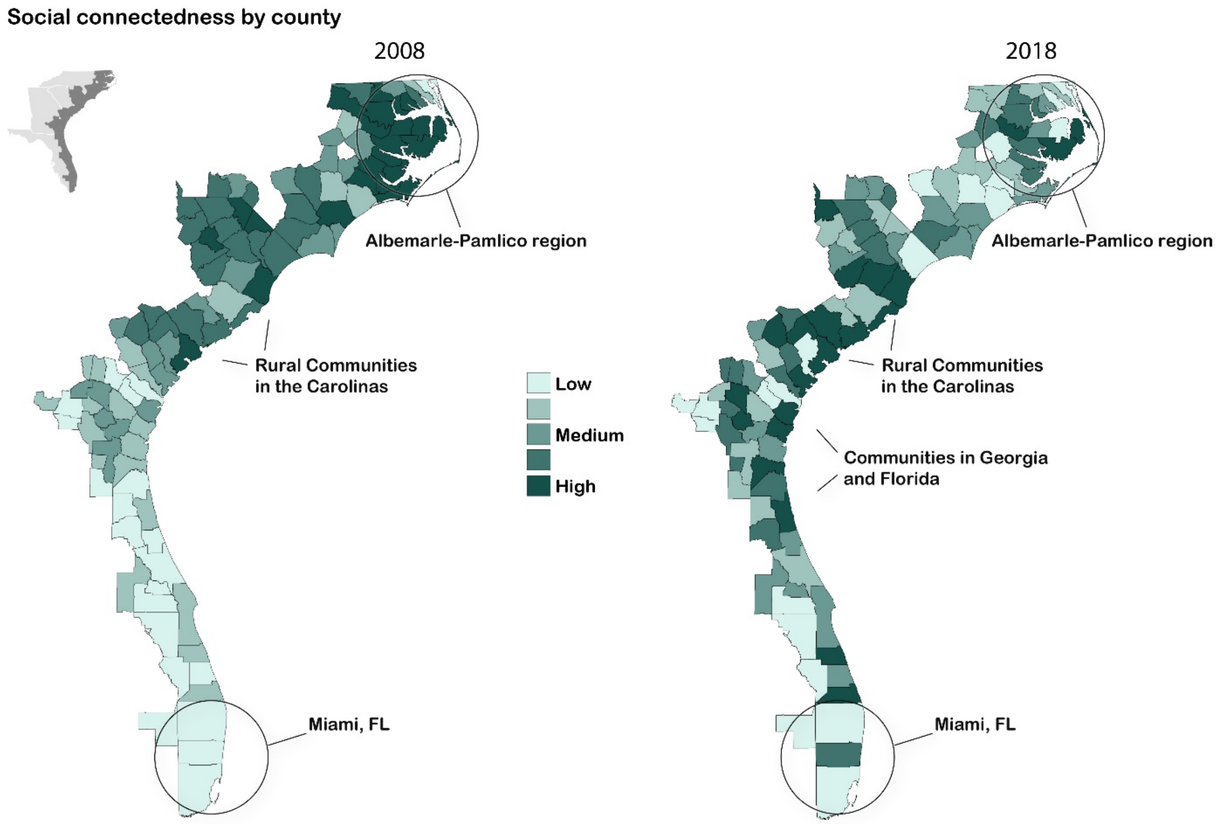


Figure 9.4. Social connectedness by county in 2008 (left panel) and in 2018 (right panel) for the U.S. South Atlantic region. Low, Medium, and High correspond to composite scores of < 0.33, ≥ 0.33 and < 0.66, and ≥ 0.66.

Social connectedness shows strong geographic variation in the U.S. South Atlantic region (Fig. 9.4). The top five socially connected counties in 2018, each scoring above 0.62, were in North Carolina (Martin county), South Carolina (Georgetown and Beaufort counties), and Georgia (Screven, Glynn, McIntosh counties). The least socially connected counties in 2018, each scoring below 0.4, were in North Carolina (Onslow County), Georgia (Atkinson and Long county), and Florida (Hendry county). The counties with the largest gains in social connectedness between 2008 and 2018 (all > 0.03) were in Georgia (Glynn county) and Florida (Putnam, Martin, Seminole, and Indian River counties). The counties with the biggest losses in connectedness from 2008 to 2018 were in North Carolina (Jones, Pasquotank and Onslow counties) and South Carolina (Marlboro county). Historically, the most socially connected counties were primarily rural communities with strong religious organizations. The maximum losses in social connectedness were an order of magnitude larger than the maximum gains, and only 31 of 105 counties showed gains, indicating that overall the U.S. South Atlantic region has experienced a decrease in social connectedness.

9.5 Commercial and recreational fishing engagement

Commercial and recreational fishing engagement and reliance are measures of fishing activity at the county level, measured by the absolute number of several activities available from federal fisheries datasets [279]. The commercial fishing engagement index is derived from the actual pounds of landings, number of commercial vessels by homeport address, number of commercial vessels by owner's address, and number of dealers with landings. The recreational engagement index is derived from the number of recreational vessels by homeport address, number of recreational vessels by owner's address, and number of recreational infrastructures (e.g., boat ramps). The commercial and recreational reliance indices are relative measures consisting of the same variables used to compute the engagement indices, but divided by the population of the community. These variables are then used in a principal component analysis with a single factor solution [279]. The factor score from the PCA is considered the engagement or reliance index score for the community. Data were available to compute these indices for coastal counties from North Carolina to Florida (east coast) from 2015 to 2017.

Counties with high levels of commercial fishing engagement in 2017 occurred along the South Atlantic coast from South Florida to North Carolina (Fig. 9.5, top). Particularly high levels occurred near Miami (Miami-Dade county), Charleston (Charleston county), and the Albemarle-Pamlico region (Carteret, Pamlico, High and Dare counties in NC). Reliance on commercial fishing was notably higher in the Albemarle-Pamlico region than in regions to the south, likely because southern regions with high engagement indicators are large metropolitan areas with a diverse economic base and alternative employment opportunities. In contrast, the Albemarle-Pamlico region of North Carolina relies primarily on a maritime and tourist economy. In 2017, commercial fishing engagement increased slightly while commercial reliance decreased compared to the previous two years.

Recreational fishing engagement and reliance showed a similar pattern to that for the commercial sector (Fig. 9.5, bottom), with the highest levels of engagement occurring in Brevard and Volusia counties near Cape Canaveral (Florida), Charleston County (South Carolina), and the Albemarle-Pamlico region of North Carolina, particularly Carteret and Dare counties. Recreational fishing reliance was highest in the northern areas, particularly Dare and Hyde counties (North Carolina) and Myrtle Beach, South Carolina (Horry County). Recreational fishing engagement and reliance decreased slightly in 2017 compared to the previous two years

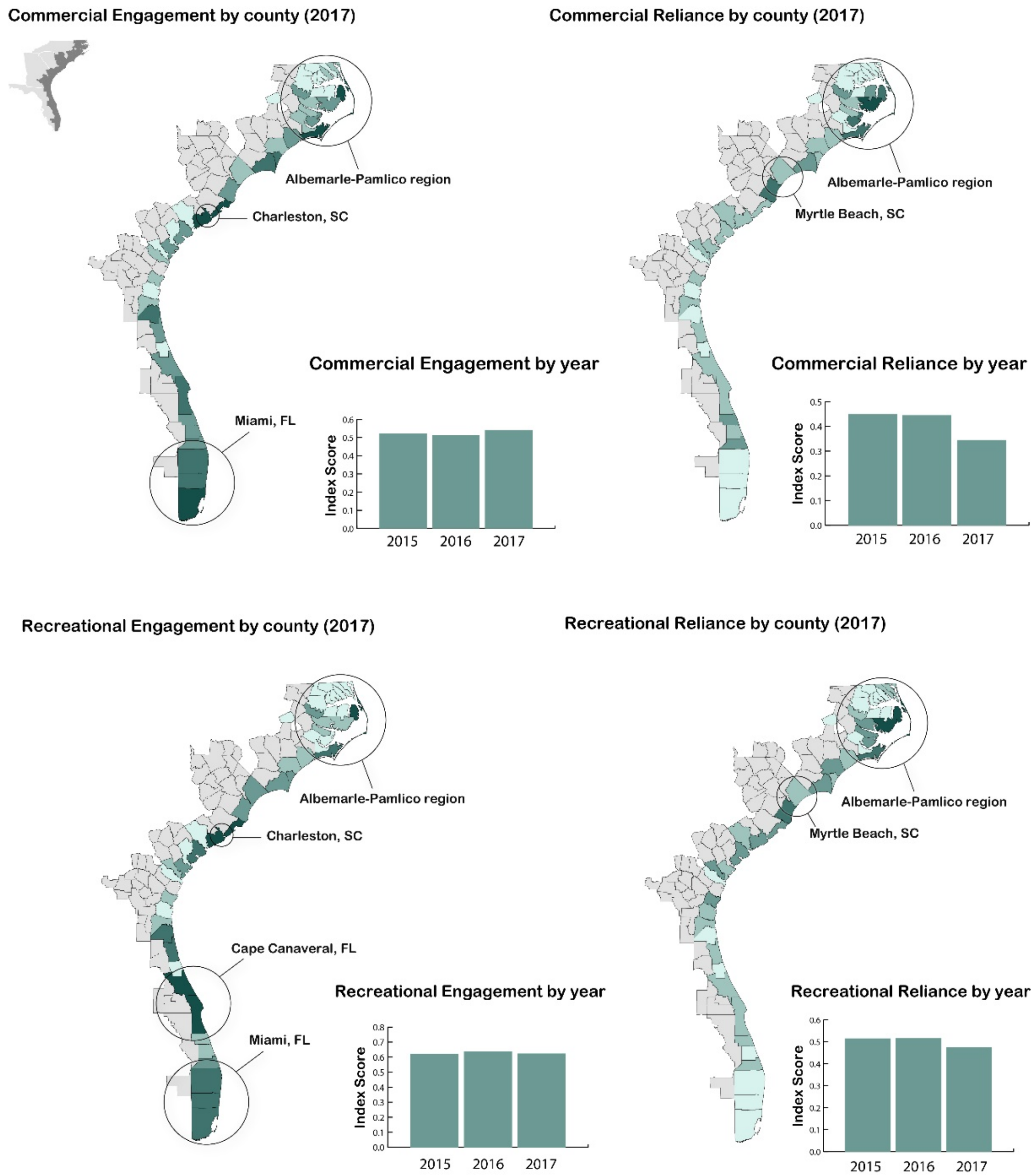


Figure 9.5. Level of commercial engagement (top left panel), commercial reliance (top right panel), recreational engagement (bottom left panel), and recreational reliance (bottom right panel) by county in 2017. Bar graphs represent the average engagement and reliance scores from 2015 – 2017.

10. INTEGRATED HUMAN DIMENSIONS PERSPECTIVE

Changes in the human dimension indicators for the U.S. South Atlantic region are rooted in rapid population growth, especially in cities, which is associated with a gradual diversification of coastal economies. This pattern is similar to many regions in the Sun Belt where immigration from northern regions and other countries has increased and more specialized industries, such as banking, have developed. Rapid population growth began fairly abruptly in the mid-1900s, enabled by policy changes allowing drainage of the South Florida wetlands and the invention of air conditioning after World War II [274]. These technological and policy changes made the coastal region of the U.S. Southeast an attractive place to live and visit, initiating the population increase that continues today.

The development and growth of cities in the U.S. South Atlantic was associated with a diversification of the economy and culture of the region. The natural resources that at first primarily supported fishing and agriculture, now also sustain additional important economic drivers, such as amenity migration and tourism. This social change coincides with areas that have also experienced declines in social connectedness, suggesting new residents may not yet have built the cultural institutions that still exist for the more rural parts of the Southeast. Within rural areas, such as the Albemarle-Pamlico region of North Carolina, social connectedness is strongest, as is reliance on traditional fishing industries. These areas are beginning to integrate tourism into their economies but remain dependent on fisheries.

There is considerable diversity and geographic variation in human dimension indicators across the region, from rural areas highly engaged in fishing, such as the Albemarle-Pamlico region, to the highly urbanized population centers of South Florida. While tourism is increasing as an important driver of economic activity in many regions of the U.S. South Atlantic, economic diversification is increasing as well, enhancing the resilience of local coastal economies to natural disasters and economic downturns. With the large impacts of COVID-19 on the global travel industry, areas highly dependent on tourism may take longer to recover both economically and socially. In a region that is projected to experience increases in sea level rise due to coastal inundation, storm surge, and high tide/nuisance flooding, as well as damages from intensifying hurricane and tropical storm activity, it remains to be seen how economic development and human population migration will proceed in the future.

11. INTEGRATED ECOSYSTEM PERSPECTIVE

In order to develop an ecosystem-wide perspective, the suite of indicators developed here for the U.S. South Atlantic region were synthesized using multivariate analyses. Traffic light plots, principal components analysis, and chronological clustering were applied to the full suite of indicators as well as to four indicator categories with sufficient time series for analysis (i.e., Climate Drivers, Physical and Chemical Pressure, Upper Trophic Level States, and Ecosystem Services).

Traffic light plots are useful for visualizing qualitative changes in different components of the ecosystem over time. Annual values of each indicator are represented by color-coded squares corresponding to quintiles along a color spectrum from blue to red. Blue and red colors correspond to years in which the indicator was above or below the long-term time series mean, respectively, while yellow colors correspond to years near the long-term mean. The indicators are ordered from top to bottom by their loadings from a principal component analysis (see below), so that indicators showing the most similar patterns over time are the most closely grouped in the traffic light plot.

Principal components analysis (PCA) is an ordination technique that condenses information from multiple time series into a smaller number of factors. Each factor is a linear combination of the original time series and is defined by the loadings (or correlation) of each time series with the factor. The matrix of indicator values by year was scaled to standardize for differences in absolute magnitude among indicators. The principal component scores from the first two PCA factors were plotted against each other to depict annual changes in the combined suite of indicators, with similar scores over time reflected as little annual variability (consecutive years close together in the PCA plot) and large changes over time reflected as large fluctuations, or potential shifts, in the suite of indicators (years far apart in the PCA plot).

Chronological clustering (CC) is a data reduction method that partitions time series into groups based on similarity or distance without *a priori* knowledge of relatedness among time series. Chronological clustering was performed on the Euclidean dissimilarity matrix constructed from the scaled indicator values. A constrained hierarchical clustering was performed on the year dissimilarity matrix with clusters constrained by sample order. Dendrograms show the grouping of years based on their degree of similarity with respect to the multivariate time series.

Both the PCA and CC analyses can be sensitive to the number of years with missing values as well as redundancy in the time series (e.g., the same indicator computed across multiple geographic regions or multiple species). To standardize the length of the time series among indicators, all analyses were run for the period 1990 – 2017. Only indicators with values for ≥ 90% of the years within this time period were retained in the analysis. The years with missing values (no more than three per time series) were replaced with the time series mean. To insure patterns were not affected by redundant indicators, analyses were also run with indicators aggregated by averaging across species or geographic areas.

11.1 Synthesis results

The traffic light plot for the full suite of indicators is shown in Fig. 11.1a and the names corresponding to the indicator abbreviations are shown in the appendix. The ecosystem showed evidence of consistent but gradual change from 1990 to 2010 and accelerated change thereafter (until 2017) (Fig. 11.1b, top panel). All seven of the indicator categories were represented among indicators with particularly high (≥ 0.1) or low (≤ 0.1) loadings on the first two principal components, suggesting there is considerable correlation among many of the indicators over time. The total amount of variation explained by the first two principal components was 29.8% for PC 1 and 13.9% for PC 2. The chronological clustering showed significant changes in the suite of indicators in 2005/2006, 2013/2014, and 1998/1999 (Fig. 11.1b, bottom panel).

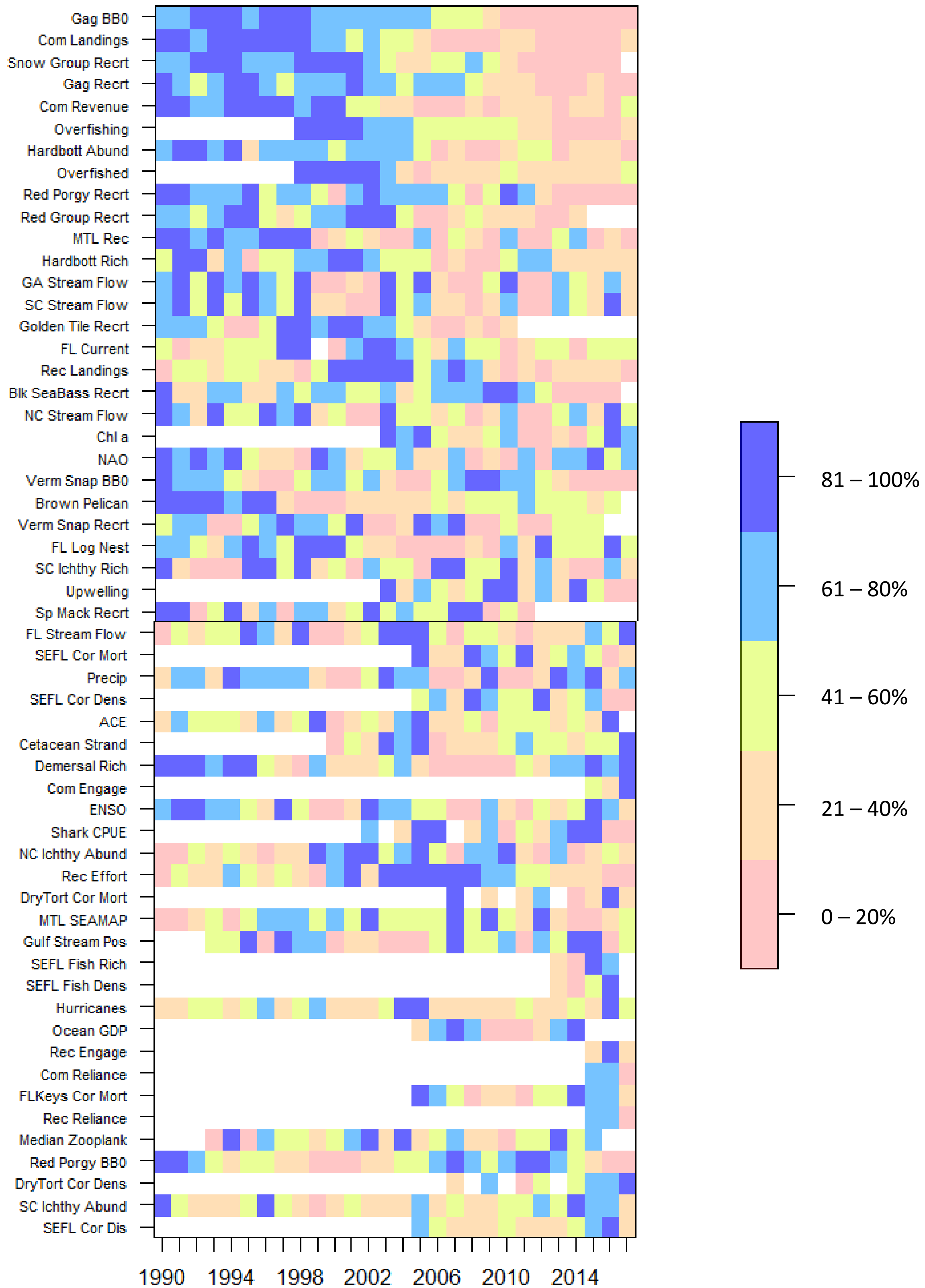


Figure 11.1a. Traffic light plot for all indicators in the order of their first principal component loadings from the PCA.

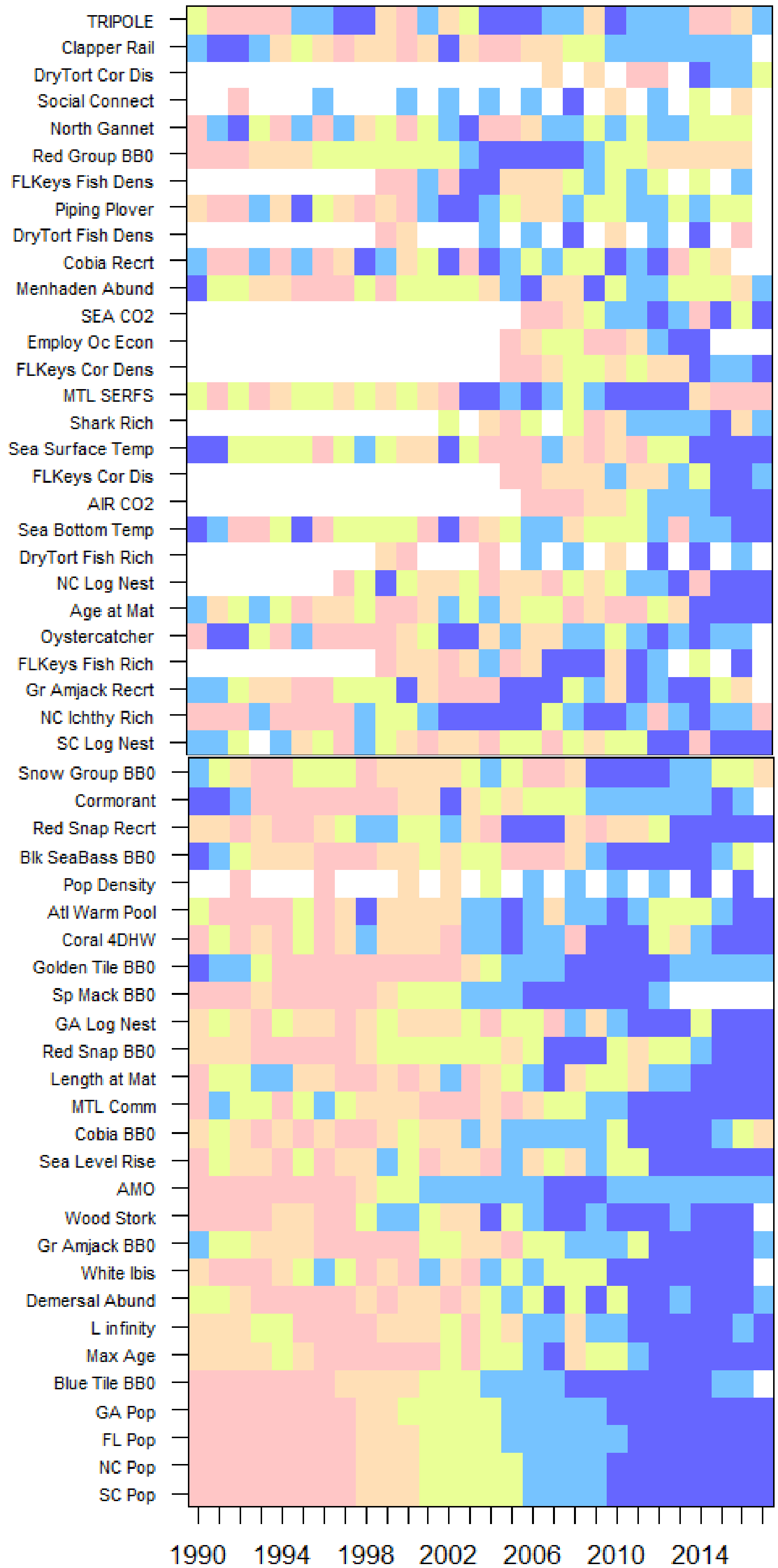


Figure 11.1 (continued) Traffic light plot for all indicators in the order of their first principal loadings from the PCA.

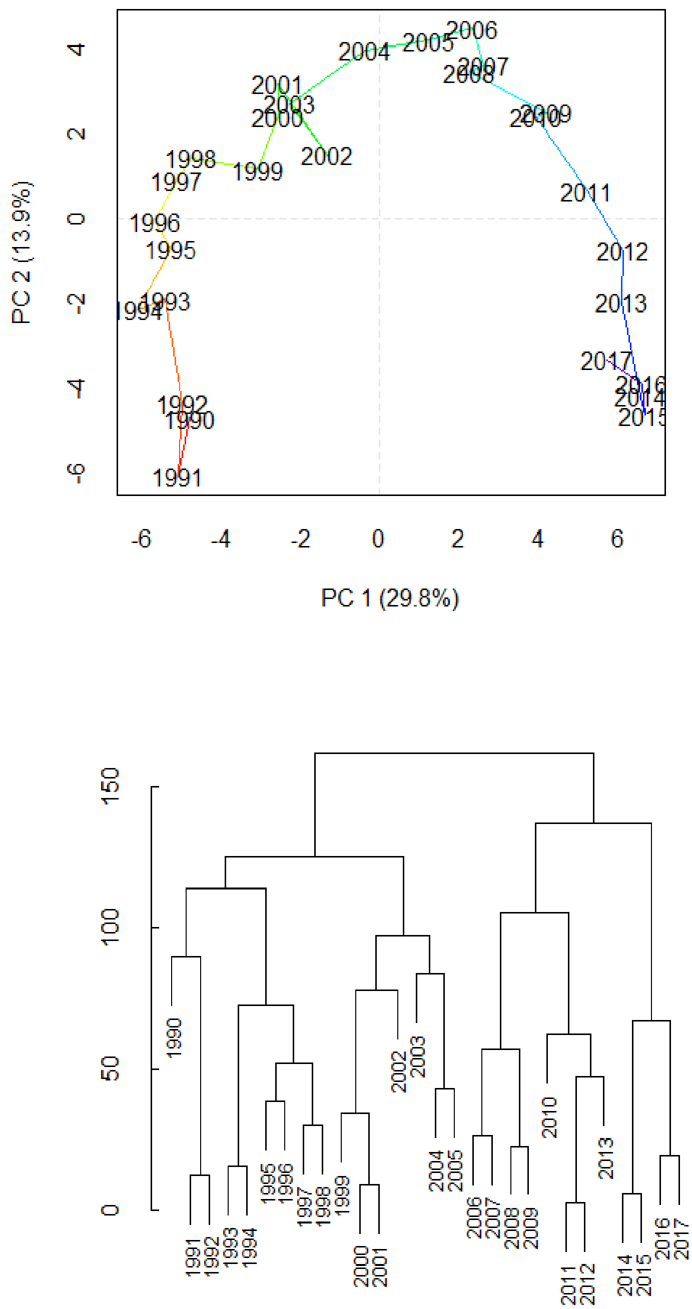
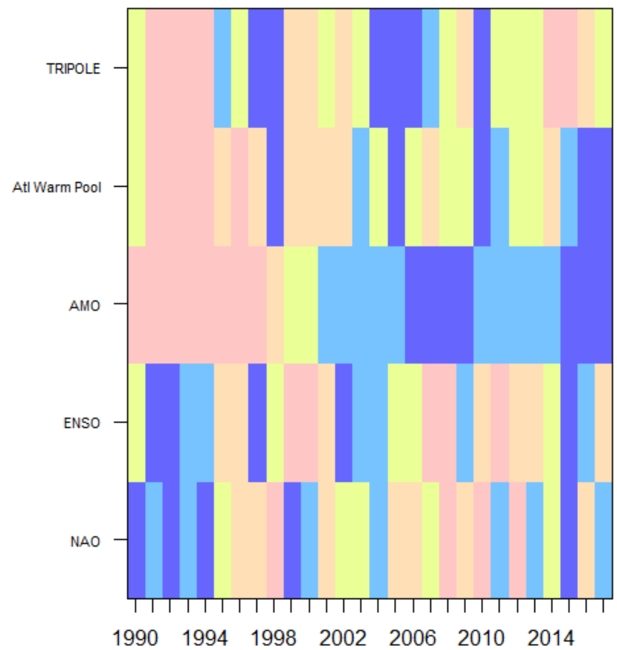


Figure 11.1b. (Left) Plot of the first two PCA axis scores. The percentage of variation explained by each principal component given in (). Trajectories indicate the relative direction and magnitude of annual change; (Right) Chronological clustering dendrogram based on an annual dissimiliarty matrix. Branches indicate breaks in the grouping of years based on euclidean distances among scaled indicator values. Data include indicators with three or fewer missing values from 1990 – 2017.

Multivariate analysis of the “Climate Drivers” indicators showed relative stability in the 1990s and larger and more rapid changes in the 2000s (Fig. 11.1c). This is evident in the PCA where the annual principal component scores in the 1990s are mostly contained in the upper left quadrant while the scores during the late 1990s and 2000s spanned a much broader range (Fig. 11.1c, bottom left). Chronological clustering indicated a break before and after 1994 – 1995 (Fig. 11.1c, bottom right), which corresponds to a documented shift in the AMO in the mid-1990s from a predominantly cool phase to a predominantly warm phase. This shift in the AMO has been associated with changes in a number of ecosystem components in the northern Gulf of Mexico [12]. The years 2010 and 2015 were highly anomalous years. 2010 was characterized by an extreme positive North Atlantic Tripole with an associated strong negative NAO and large Atlantic Warm Pool (see Fig. 3.2, 3.4, and 3.5), while 2015 was a strong El Niño year (see Fig. 3.3).



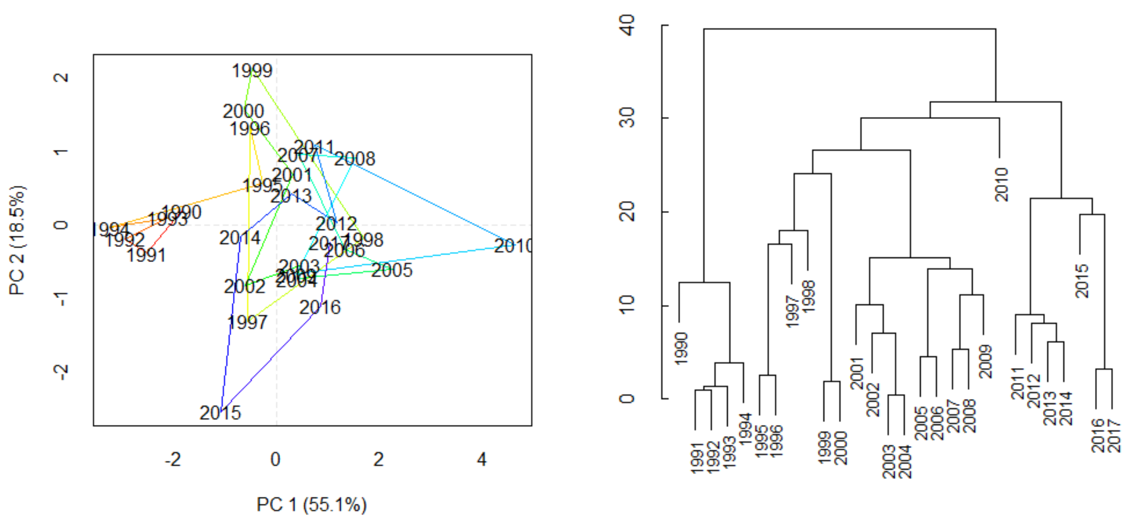


Figure 11.1c. Climate Drivers: (Top) Traffic light plot with indicators sorted by first PCA axis score (see appendix for abbreviation names); (Bottom Left) Plot of the first two PCA axis scores. The percentage of variation explained by each principal component given in (). Trajectories indicate the relative direction and magnitude of annual change; (Right) Chronological clustering dendrogram based on an annual dissimiliarty matrix. Branches indicate breaks in the grouping of years based on euclidean distances among scaled indicator values. Data include indicators with three or fewer missing values from 1990 – 2017.

Multivariate analysis of the “Physical and Chemical Pressures” indicators are shown in Figure 11.1d. PC1 was characterized by strong negative loadings for terrestrial and nearshore estuarine indicators (e.g., stream flow, precipitation) while PC2 was characterized by strong positive loadings for oceanic indicators, including multiple measures of temperature and Gulf Stream characteristics. The chronological clustering dendrogram showed a major break separating the most recent years in the time series (2013 – 2017) from the earlier year. These years also showed the most extreme values on PC2, suggesting recent, changes in ocean dynamics have occurred in the U.S. South Atlantic. In contrast, the combined ‘land-estuarine’ indicators did not show clear temporal trends in the PCA plot (PC1). The most recent years have been characterized by increased annual precipitation and decreased propensity for regional drought but little long-term pattern in stream flow and coastal salinity. Other indicators, such as hurricanes and storm intensity (as indicated by ACE) are more episodic in nature and did not show strong temporal trends.

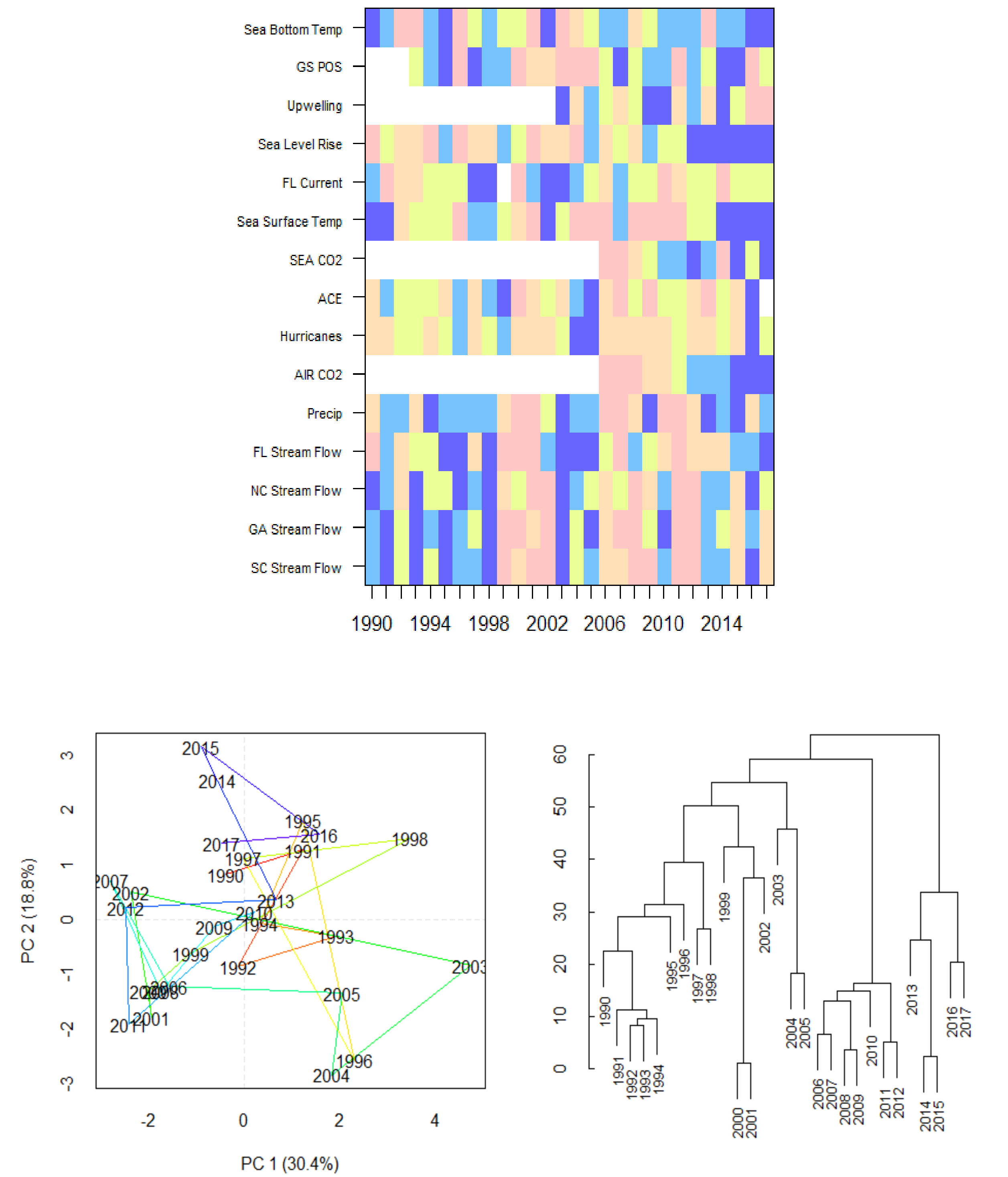


Figure 11.1d. Physical and Chemical Pressures: Traffic light plot with indicators sorted by first PCA axis score (see appendix for abbreviation names); (Bottom Left) Plot of the first two PCA axis scores. The percentage of variation explained by each principal component given in (). Trajectories indicate the relative direction and magnitude of annual change; (Right) Chronological clustering dendrogram based on an annual dissimiliarty matrix. Branches indicate breaks in the grouping of years based on euclidean distances among scaled indicator values. Data include indicators with three or fewer missing values from 1990 – 2017.

Multivariate analysis of the “Upper Trophic Level States” category showed relatively gradual changes over time punctuated by shifts in the early to mid-1990s, 2005/2006 and in 2013/2014 (Fig. 11.1e). This latter shift separating the most recent years (2014 – 2017) was also evident in the physical and chemical indicators discussed above, suggesting recent changes in the upper trophic level component of the South Atlantic ecosystem are anomalous compared to their historical (since 1990) pattern of variability. PC1 (40.9% of the variability) was characterized primarily by indicators related to offshore hard bottom reef fishes, with high positive loadings for indicators related to reef fish abundance and diversity and low negative loadings for indicators related to abundance-weighted life history parameters (e.g., maximum age and length, length at maturity). The shift in life history characteristics of the offshore hard bottom reef fish community reflects both declining catch rates for several small non-targeted species as well as increasing catch rates for some highly targeted species (Red Snapper) that may be responding to recent management measures. The break in the chronological clustering dendrogram before and after 2005 – 2006 reflects both the time when offshore hardbottom reef fishes declined to below their long-term mean and the abundance of demersal fishes in nearshore, soft-sediment habitats began to increase. This period of increasing annual variability is also reflected in the PCA ordination plot where years after 2005 have diverged from the historical (since 1990) pattern.

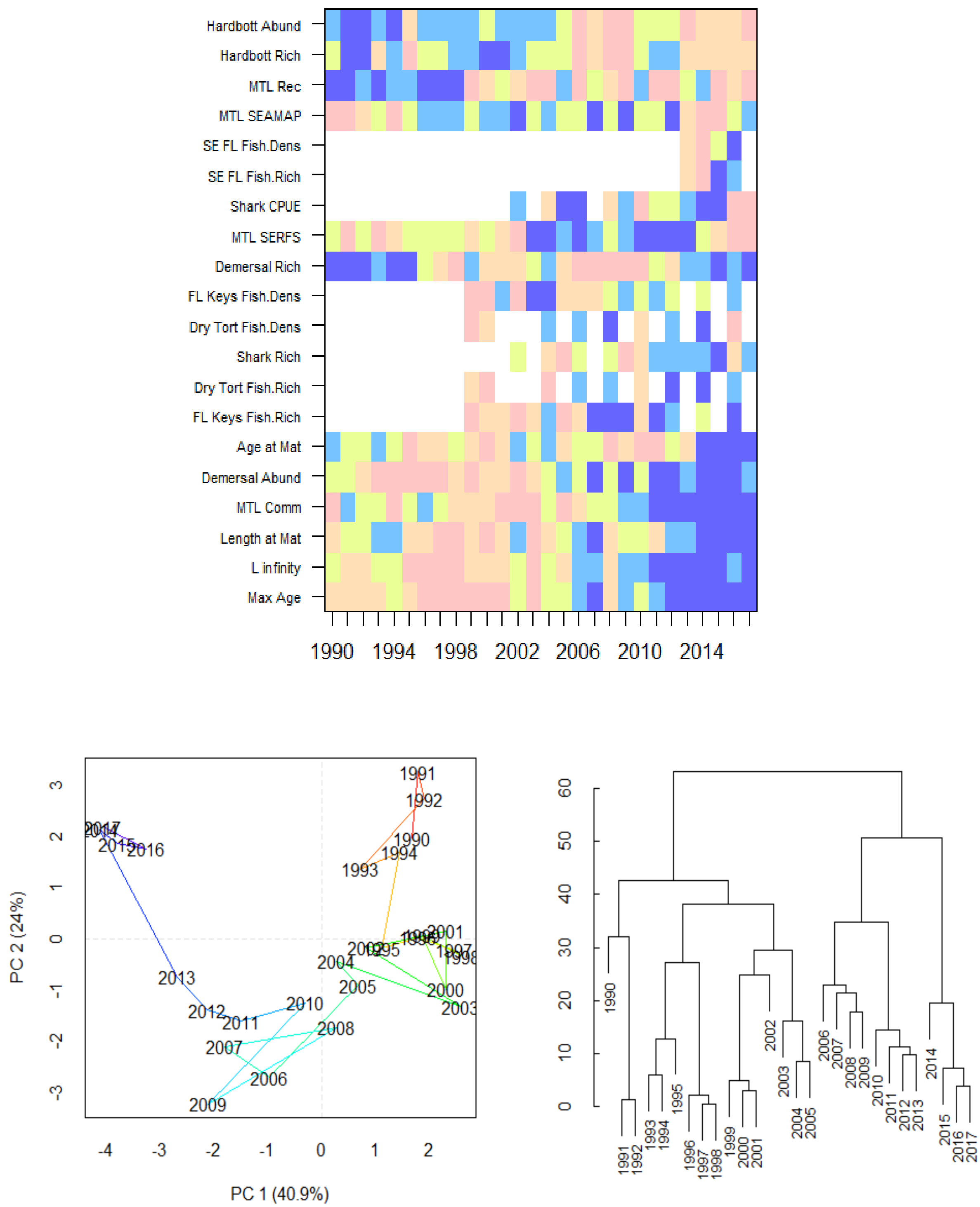


Figure 11.1e. Upper Trophic Level States: Traffic light plot with indicators sorted by first PCA axis score (see appendix for abbreviation names); (Bottom Left) Plot of the first two PCA axis scores. The percentage of variation explained by each principal component given in (). Trajectories indicate the relative direction and magnitude of annual change; (Right) Chronological clustering dendrogram based on an annual dissimiliarty matrix. Branches indicate breaks in the grouping of years based on euclidean distances among scaled indicator values. Data include indicators with three or fewer missing values from 1990 – 2017.

Multivariate analysis of indicators in the "Ecosystem Services" category is shown in Figure 11.1f. Similar to the upper trophic level indicators, the PCA plot of ecosystem services indicators showed mostly gradual change over time (Fig. 11.1f, bottom left), punctuated by shifts in the early 1990s, 2004/2005, and 2012/2013. PC1 explained 31.7% of the variability and was characterized by high positive loadings for indicators related to biomass of reef fishes and abundance of many species of marine birds, and negative loadings for stock status, commercial fishing indicators, and recruitment of several hard-bottom reef fishes. PC2 (15.2% of the variability) was defined primarily by indicators related to species of concern, such as loggerhead sea turtles and multiple marine birds (positive loadings) and recreational fishing indicators (negative loadings). Overall the ecosystem services indicators showed moderate changes over time, with the largest change occurring in the mid-2000s, and more stability in recent years (since 2013). These indicators are not expected to change rapidly because they integrate information across multiple age classes of long-lived species or represent aggregate measures (e.g., total landings or fishing effort), that are not expected to show large annual fluctuations. Similar to other indicator groups, the years at the end of the time series (2013 – 2017) showed the most extreme values, indicating some of the largest changes in ecosystem services have occurred recently. The similar patterns for both ecosystem services indicators and upper trophic level indicators is not surprising, as most of the ecosystem services included here are related to upper trophic levels either directly or indirectly influenced by fishing.

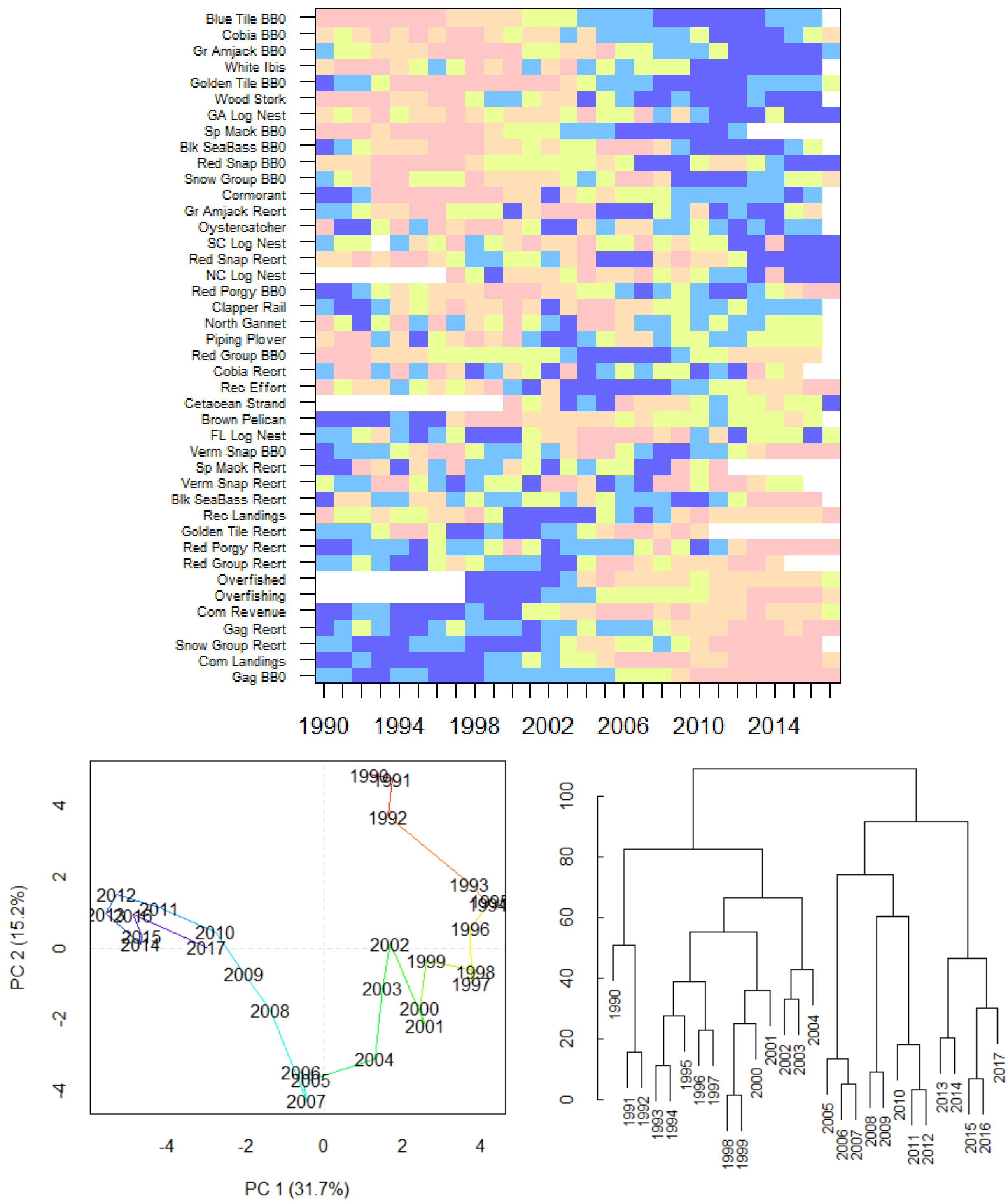


Figure 11.1f. Ecosystem Services: Traffic light plot with indicators sorted by first PCA axis score (see appendix for abbreviation names); (Bottom Left) Plot of the first two PCA axis scores. The percentage of variation explained by each principal component given in (). Trajectories indicate the relative direction and magnitude of annual change; (Right) Chronological clustering dendrogram based on an annual dissimiliarty matrix. Branches indicate breaks in the grouping of years based on euclidean distances among scaled indicator values. Data include indicators with three or fewer missing values from 1990 – 2017.

11.2 Synthesis summary

The intent of this report is to develop a broad perspective of the U.S. South Atlantic region through a set of quantitative indicators that reflect the current status and trends of the physical, biological, and socioeconomic components of the ecosystem. These indicators were grouped as "Climate Drivers," "Physical and Chemical Pressures," "Habitat States," "Lower Trophic Level States," "Upper Trophic Level States," "Ecosystem Services," and "Human Dimensions." The U.S. South Atlantic is a transitional ecosystem characterized by tropical conditions in the south (i.e., South Florida) and more temperate conditions in the north (i.e., North Carolina). As such, the region is influenced by multiple long-term modes of climate variability that interact to determine the physical conditions in the ecosystem. Because many of these drivers have contrasting effects on wind and moisture transport in the atmosphere, rainfall, sea surface temperatures, and storm activity, it is difficult to predict the consequences of annual to decadal shifts in these modes of climate variability on the ecosystem. In recent years the AMO and NAO have been in a mostly positive phase and the North Atlantic Tripole in a mostly negative phase. This suggests the U.S. South Atlantic is in a period of generally increased warming. Multiple indicators of temperature, including satellite-based SST (Fig. 4.1), the extent of the Atlantic Warm Pool (Fig. 3.5), bottom temperatures (Fig. 4.2), long-term climatology (Fig. 4.3), and indicators of coral thermal stress (Fig. 5.5) all indicate a period of recent warm, particularly since the early to mid-2010s. This suggests the U.S. South Atlantic is starting to see increases in temperature similar to other regions along the U.S. Atlantic seaboard where temperature increases have been more pronounced. In addition to increasing temperatures, other oceanographic aspects of the offshore ecosystem appear to be changing. The Gulf Stream has been in a more onshore position in recent years which has implications for coastal circulation, upwelling and nutrient delivery to the shelf, and coastal upwelling has declined since 2014 (Fig. 4.6), suggesting potential effects on delivery of nutrients to the photic zone. Chl-a was low from 2010 – 2015 compared to earlier and later in the time series and the overall long-term trend (since 2003) has been negative, though with considerable spatial variability. Linkages among these various physical components of the ecosystem and their effects on primary productivity are complex and in need of further investigation.

The South Atlantic ecosystem is experiencing a number of chronic stressors that have shown gradual increases over relatively long periods of time (i.e., decades). Florida marine ecosystems now experience temperatures that lead to coral bleaching and mortality for longer portions of the year than in the past (Fig. 5.5). Other examples of stressors that have increased over decadal time frames include nutrient loading (Fig. 4.9), sea level rise (Fig. 4.11), ocean acidification (Fig. 4.13), loss of marsh and coastal forest habitats (Fig. 5.1), and rising human populations (Fig. 9.1). While quantitative information on nearshore coastal habitats, such as seagrass beds (Table 5.2), oyster reefs (Table 5.3), and coral reefs (Fig. 5.4) is limited, the general consensus is that most of these habitats are a fraction of their historical distribution [140, 153, 161]. These degradations to various components of the ecosystem present a difficult problem for coastal managers. Because they occur gradually over time and are difficult to notice, society often perceives the new degraded state of the ecosystem as being acceptable, a phenomenon known as shifting baselines [280]. As a result, these stressors often do not garner the attention of managers, stakeholders, and the public.

The South Atlantic ecosystem has experienced a number of changes in the fish community, species of concern (e.g., marine mammals, sea turtles, birds), and in commercial and recreational fisheries. Demersal finfish in nearshore soft sediment communities have shown increases in diversity and abundance (Fig. 7.1) while offshore hard-bottom reef fishes, both targeted and not targeted by fisheries, have shown declines in abundance since the 1980s and 1990s (Fig. 7.1 and 8.1). In addition, many hard bottom reef fish species have shown recent (since 2010s) declines in recruitment (Fig. 8.2). While management has successfully ended overfishing of most stocks in federal waters, many stocks remain in an overfished state and have been slow to recover (Fig. 8.6). The underlying causes of many of these changes is unknown, though potential explanations include continued overfishing or changes in bycatch mortality, lags in recovery due to life history characteristics (e.g., long-lived, old age at maturity), or environmental factors that affect productivity. The U.S. South Atlantic region has always had a large recreational fishing sector, though the dominance of recreational fishing has continued to increase since the 1980s. This is evident in indicators of increasing recreational landings and effort (Fig. 8.4) and declining commercial landings and revenues (Fig. 8.3). The increase in recreational use of the ecosystem is driven in part by the growing coastal human population and the attraction of the region for tourism and retirement.

The U.S. South Atlantic is a complex ecosystem in terms of its physical, biological, and socioeconomic dynamics. The indicators developed here provide a broad overview of these various components and potential linkages, and provide a basis for future monitoring, research, and management. The indicators in this report represent the first attempt to develop a broad suite of quantitative metrics to characterize the status and trends of the U.S. South Atlantic ecosystem. Even so, they are based on relatively limited monitoring data (a few decades or less for most indicators) collected at a particular spatial and temporal scale, and often designed for other purposes. In particular, data were limited for developing regional-scale time series for habitat states, lower trophic levels, and most human dimensions indicators. As anthropogenic pressures continue to increase in the region, effective management will require continued and expanded monitoring efforts, more integrated consideration of the various components of the ecosystem, and a better understanding of the trade-offs inherent in the multiple uses of the ecosystem to support a diversity of human activities.

12. RESEARCH RECOMMENDATIONS

This report has identified a number of information gaps with respect to the U.S. South Atlantic ecosystem. Data to develop regional scale indicators were most limited for habitat, lower trophic levels, and human dimensions, and these are general areas in need of further data and synthesis. Sampling of various estuarine habitats (e.g., saltmarsh, seagrass beds, oyster reefs) is conducted by individual states using different sampling methodologies; additional work or more systematic surveys are needed to synthesize these data into region-wide estimates of important structured marine habitats. Similarly, while indicators of primary productivity are available from satellite imagery, there is little information on other lower trophic levels such as zooplankton, planktivorous fishes, or the mesopelagic zone in general for the U.S. South Atlantic. In addition, some physical indicators have limited (ocean acidification) or unknown (nearshore coastal upwelling) geographic scales. Data assimilative modeling, by providing time-space continuous ocean state conditions, can help to fill some of these data gaps. Time series of ichthyoplankton at particular point locations (Beaufort Inlet, Winyah Bay) showed disparate patterns, suggesting considerable localized variability in these indicators, and systematic ichthyoplankton and zooplankton surveys do not exist for the region. In addition, time series of commercial and recreational fishing engagement and reliance were relatively short (3 years). Longer time series of these indicators would be helpful in evaluating changes associated with the ongoing shift from a maritime economy dominated by commercial fishing to a tourist economy dominated by recreational fishing interests. Given that fisheries in the U.S. South Atlantic are increasingly dominated by the recreational sector, additional human dimensions indicators, such as gentrification, recreational fishing power, economic activity, and fisheries management complexity will be useful in tracking these ongoing changes. Finally, the indicators reported here were developed at a specific spatial and temporal scale, with an emphasis on monthly to annually averaged time series for the entire U.S. South Atlantic. Additional work is needed to determine the appropriate scale for these indicators. Given the geographic variability in the U.S. South Atlantic, with more tropical conditions at the southern limit and more temperate conditions at the northern limit, additional signals may be apparent at alternative spatial and temporal scales to those considered here. Specific research recommendations are listed below:

- The U.S. South Atlantic is a transitional ecosystem characterized by subtemperate/tropical conditions in the south and temperate/boreal conditions in the north. As such, the region is influenced by multiple climate drivers (AMO, NAO, North Atlantic tripole, ENSO). Additional research is needed to better understand the interactions among these drivers and their influence on the physical and biological dynamics of the U.S. South Atlantic region.

- Indicators developed here suggest changing ocean conditions in the U.S. South Atlantic beginning in the late 2010s. Additional research is needed to understand the relationship between SST, upwelling, and primary productivity, particularly in relation to the dynamics of the Gulf Stream, the dominated physical feature of the system.

- While bottom temperatures have been relatively high in recent years, the extent to which they have increased over time is not clear. Additional bottom temperature observations or sampling of outputs from regional hydrodynamic and biogeochemical models would be helpful in understanding three-dimension patterns in temperature variability on the continental shelf.

- Additional research is needed to better characterize the relationship between watershed precipitation, stream flow, coastal salinity, and nutrient inputs to coastal systems of the U.S. the South Atlantic. These indicators were computed as monthly and annual averages, or changes between two points in time (i.e., for nutrients), and often over different spatial scales. For example, variation in precipitation and stream flow may be better captured by the frequency of extreme events rather than an annual or monthly average.

- Sea level rise has shown a consistent long-term trend in the region, with some acceleration in the most recent years. A better understanding of how observed rates of sea level rise change the availability of nearshore coastal habitats (e.g., saltmarshes) or threaten infrastructure or species of concern is needed.

- Additional research is needed to develop indicators of seagrass and oyster reef availability, functionality, and change over time. The current indicators are based only on the areal coverage of these habitat types over relatively limited spatial and temporal scales and long-term time series are not available.

- Little information exists for lower trophic levels (e.g., zooplankton, ichthyoplankton, forage fish) in the U.S. South Atlantic and current indicators are highly localized in space. Better indicators of lower trophic levels are needed to determine the bottom-up effects of changing ocean conditions on upper trophic levels, including harvested species.

- Landings of deep-water species have increased, particularly in the recreational sector, due to increased interest, increases in fishing power, and declines in other species. A deep-water, fishery independent survey is in the early stages of development in the region. A better understanding of the life history and population dynamics of deep-water species is needed as harvest pressure on this component of the community increases.

- The Cape Hatteras region is considered a biogeographic break for many species along the Atlantic seaboard. Indicators of distribution shifts, potentially in relation to changing climate, are needed for ecologically and economically important species in the region.

- The identification of stock boundaries for assessed South Atlantic species (e.g., snapper-grouper) have historically been set at the North Carolina-Virginia border to the north and in the Florida Keys to the south. Given the potential for distribution shifts and evidence for connections between the Gulf of Mexico and South Atlantic via the transport of eggs and larvae, addition work is needed to better delineate stock boundaries for assessed species.

- Lionfish are an invasive species in the U.S. South Atlantic. A better understanding of the invasion history of lionfish as well as indicators that track their distribution and abundance are needed for the region.

- Multiple species of marine birds use coastal ecosystems in the U.S. South Atlantic. A better understanding of the reliance of coastal birds on marine habitats (e.g. beaches, barrier island, marshes, open ocean) as well as the contribution of these habitats to the larger-scale population dynamics of these species is needed. Pelagic birds, in particular, are not well-represented in the data for the region.

- Bottlenose Dolphins, other cetaceans, and pinnepeds play an important role in coastal ecosystems and are species of conservation concern. Long-term information on the relative abundance of these species is limited. Systematic surveys along with continued monitoring of strandings is needed to better understand the population dynamics of these protected species in the U.S. South Atlantic.

- The South Atlantic has historically had a large recreational fishing sector that has been increasing in recent years. Current indicators of recreational fishing track landings and fishing effort (e.g., number of trips), but do not capture the technological (vessel size, engine displacement, GPS and other technology) and social (communication through social media) innovations that continue to increase recreational fishing power.

- Engagement and reliance on commercial and recreational fishing and social connectedness are important socioeconomic indicators reflecting the dependence of human components on the biological ecosystem. Longer time series as well as additional human dimensions indicators (e.g., gentrification, fisheries management complexity) are needed to help track the changes in coastal communities.

13. ACKNOWLEDGEMENTS

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14. REFERENCES

1. Devictor, S., and S. Morton. 2010. Identification guide to the shallow water (0-200 m) octocorals of the South Atlantic Bight. Zootaxa 2599:1-62. doi:10.11646/zootaxa.2599.1.1.
2. Conley, M.F., M.G. Anderson, N. Steinberg, and A. Barnett, eds. 2017. The South Atlantic Bight Marine Assessment: Species, Habitats and Ecosystems. The Nature Conservancy, Eastern Conservation Science.
3. Ache, B.W., K.M. Crossett, P.A. Pacheco, J.E. Adkins and P.C. Wiley. 2015. "The Coast" is complicated: A model to consistently describe the nation's coastal population. Estuaries and Coasts 38 (Suppl. 1):151-155. doi:10.1007/s12237-013-9629-9.
4. Kelble, C.R., D.K. Loomis, S. Lovelace, W.K. Nuttle, P.B. Ortner, P. Fletcher, G.S. Cook, J.L. Lorenz, and J.N. Boyer. 2013. The EBM-DPSER conceptual model: Integrating ecosystem services into the DPSIR framework. PLoS ONE 8(8):e70766.
5. Knight, J.R., C.K. Folland, and A.A. Scaife. 2006. Climate impacts of the Atlantic Multidecadal Oscillation. Geophysical Research Letters 33:L17706. doi:10.1029/2006GL026242.
6. Enfield, D.B., A.M. Mestas-Nuñez, and P.J. Trimble. 2001. The Atlantic multidecadal oscillation and its relation to rainfall and river flows in the continental U.S. Geophysical Research Letters 28:2077-2080.
7. McCabe, G.J., M.A. Palecki, J.L. Betancourt, and I.Y. Fung. 2004. Pacific and Atlantic Ocean influences on multidecadal drought frequency in the United States. Proceedings of the National Academy of Sciences 101:4136-4141.
8. Chylek, P., and G. Lesins. 2008. Multidecadal variability of Atlantic hurricane activity: 1851-2007. Journal of Geophysical Research 113:D22106. doi:10.1029/2008JD010036.
9. Nye, J., M. Baker, R. Bell, A. Kenny, K. Kilbourne, K. Friedland, E. Martino, M.Stachura, K. Van Houtan and R. Wood. 2014. Ecosystem effects of the Atlantic Multidecadal Oscillation. Journal of Marine Systems 133:S103-S116.
10. Faillettaz, R., G. Beaugrand, E. Goberville, and R.R. Kirby. 2019. Atlantic Multidecadal Oscillations drive the basin-scale distribution of Atlantic Bluefin Tuna. Science Advances 5:eaar6993.
11. Gullestad, P., S. Sundby, and O.S. Kjesbu. 2020. Management of transboundary and straddling fish stocks in the Northeast Atlantic in view of climate-induced shifts in spatial distribution. Fish and Fisheries 2020;00:1-19. doi: 10.1111/faf.12485.
12. Karnauskas, M., M.J. Schirripa, J.K. Craig, G.S. Cook, C.R. Kelble, J.J. Agar, B.A. Black, D.B. Enfield, D. Lindo-Atichati, B.A. Muhling, K.M. Purcell, P.M. Richards, and C. Wang. 2015. Evidence of climate-driven ecosystem reorganization in the Gulf of Mexico. Global Change Biology 21:2554–2568. doi:10.1111/gcb.12894.

13. Physical Sciences Laboratory, "Climate Timeseries AMO (Atlantic Multidecadal Oscillation Index)" [Online]. Available: https://psl.noaa.gov/data/timeseries/AMO/. [Accessed 04 August 2021].
14. Ishi, M., Shouji, A., Sugimoto, S., and T. Matsumoto. 2005. Objective analyses of sea-surface temperature and marine meteorological variables for the 20th century using ICOADS and the Kobe collection. International Journal of Climatology 25:865-879.
15. Hurrell, J.W., and C. Deser. 2009. North Atlantic climate variability: The role of the North Atlantic Oscillation. Journal of Marine Systems 78:28-41.
16. Delworth, T.L., F. Zeng, G.A. Vecchi, X. Yang, L. Zhang, and R. Zhang. 2016. The North Atlantic Oscillation as a driver of rapid climate change in the northern hemisphere. Nature Geoscience doi:10.1038/NGEO2738.
17. Olsen, J., N.J. Anderson, and M.F. Knudsen. 2012. Variability of the North Atlantic Oscillation over the past 5,200 years. Nature Geoscience doi:10.1038/NGEO1589.
18. Stenseth, N.C., A. Mysterud, G. Ottersen, J.W. Hurrell, K. Chan, and M. Lima. 2002. Ecological effects of climate fluctuations. Science 297:1292-1296.
19. Barnston, A.G., and R.E. Livezey. 1987. Classification, seasonality and persistence of low-frequency atmospheric circulation patterns. Monthly Weather Review 115:1083-1126.
20. Garcia-Serrano, J., C. Cassou, H. Douville, A. Giannini, and G.J. Doblas-Reyes. 2017. Revisiting the ENSO teleconnection to the tropical North Atlantic. Journal of Climate 30:6945-6957.
21. Kim, H., P.J. Webster, and J.A. Curry. 2009. Impact of shifting patterns of Pacific Ocean warming on North Atlantic tropical cyclones. Science 325:77-79.
22. Chatterjee, S., M. Nuncio, and K. Satheesan. 2017. ENSO related SST anomalies and relation with surface heat fluxes over the South Pacific and Atlantic. Climate Dynamics 49:391-401.
23. Sehgal, V., and V. Sridhar. 2018. Effect of hydroclimatological teleconnections on the watershed-scale drought predictability in the southeastern United States. International Journal of Climatology 38(Suppl. 1):e1139-e1157.
24. Dai, A., T. Qian, K.E. Trenberth, and J.D. Milliman. 2009. Changes in continental freshwater discharge from 1948-2004. Journal of Climate 22:2773-2792.
25. Wang, C., X. Wang, R.H. Weisberg, and M.L. Black. 2017. Variability of tropical cyclone rapid intensification in the North Atlantic and its relationship with climate variations. Climate Dynamics 49:3627-3645.
26. National Weather Service Climate Prediction Center. “Monthly Atmospheric and SST Indices”. [Online]. Available: https://www.cpc.ncep.noaa.gov/data/indices. [Accessed 04 August 2021].
27. Deser, C., and M.S. Timlin. 1997. Atmosphere–ocean interaction on weekly timescales in the North Atlantic and Pacific. J. Climate 10:393-408.

28. Xie, S.-P., and Y. Tanimoto. 1998. A pan-Atlantic decadal climate oscillation Geophysical Research Letters 25:2185-2188.
29. Okumura Y., S.-P. Xie, A. Numaguti, and Y. Tanimoto. 2001. Tropical Atlantic air–sea interaction and its influence on the NAO. Geophysical Research Letters 28:1507-10.
30. Peng S., W.A. Robinson, and S. Li. 2002. North Atlantic SST forcing of the NAO and relationships with intrinsic hemispheric variability. Geophysical Research Letters 29:1171-1174.
31. Wu L., F. He, Z. Liu, and C. Li. 2007. Atmospheric teleconnections of tropical Atlantic variability: Interhemispheric, tropical–extratropical, and cross-basin interactions. Journal of Climatology 20:856-870.
32. Schneider, E.K., and M. Fan. 2012. Observed decadal North Atlantic tripole SST variability: II. Diagnosis of mechanisms. Journal of Atmospheric Science 69:51-64.
33. Yu, B., and H. Lin. 2016. Tropical atmospheric forcing of the wintertime North Atlantic Oscillation. Journal of Climatology 29:1755-72.
34. Sutton, R.T., and D.L.R Hodson. 2007. Climate response to basin-scale warming and cooling of the North Atlantic Ocean. Journal of Climatology 20:891-907.
35. Lee, S.-K., A.T. Wittenberg, D.B. Enfield, S.J. Weaver, C. Wang, and R. Atlas. 2016. U.S. regional tornado outbreaks and their links to ENSO flavors and North Atlantic SST variability. Environmental Research Letters 11:044008. doi.org/10.1088/1748-9326/11/4/044008.
36. Mo K.C., J.-K.E. Schemm, and S.-H. Yoo. 2009. Influence of ENSO and the Atlantic Multidecadal Oscillation on drought over the United States. Journal of Climatology 22:5962–5982.
37. Weaver S.J., S. Schubert, and H. Wang. 2009. Warm season variations in the low-level circulation and precipitation over the central United States in observations, AMIP simulations, and idealized SST experiments. Journal of Climatology 22:5401–5420.
38. Wang C., S.-K. Lee, and D.B. Enfield. 2008. Climate response to anomalously large and small Atlantic warm pools during the summer. Journal of Climatology 21:2437-2450.
39. Huang, B., P.W. Thorne, V.F. Banzon, T. Boyer, G. Chepurin, J.H. Larimore, M.J. Menne, T.M. Smith, R.S. Vose, and H.-M. Zhang. 2017. Extended reconstructed sea surface temperature version 5 (ERSSTv5): Upgrades, validations, and intercomparisons. Journal of Climate 30:8179-8205. doi: 10.1175/JCLI-D-16-0836.1.
40. Wang, C., D.B. Enfield, S. Lee, and C. Landsea. 2006. Influences of the Atlantic Warm Pool on western hemisphere summer rainfall and Atlantic hurricanes. Journal of Climate 19:3011-3028.
41. Wang, C., S. Lee, and D.B. Enfield. 2008. Atlantic Warm Pool acting as a link between Atlantic Multidecadal Oscillation and Atlantic tropical cyclone activity. Geochemistry, Geophysics, Geosystems: An Electronic Journal of the Earth Sciences 9:Q05V03. doi:10.11029/2007GC001 809.

42. Wang, C., H. Liu, S. Lee, and R. Atlas. 2011. Impact of the Atlantic Warm Pool on United States land falling hurricanes. Geophysical Research Letters 38:L19702. doi:10.1029/2011GL049265.
43. Muhling, B.A., S. Lee, J.T. Lamkin, and Y. Liu. 2011. Predicting the effects of climate change on bluefin tuna (*Thunnus thynnus*) spawning habitat in the Gulf of Mexico. ICES Journal of Marine Science 68:1051-1062.
44. Hughes, R.A., T.C. Hanley, A.F.P. Moore, C. Ransay-Newton, R.A. Zerebecki, and E.E. Sotka. 2019. Predicting the sensitivity of marine populations to rising temperatures. Frontiers in Ecology and the Environment 17:17-24.
45. Johnson, G.C., and J.M. Lyman. 2020. Warming trends increasingly dominate global ocean. Nature Climate Change 10:757-761.
46. Pinsky M.L., B. Worm, M.J. Fogarty, J.L. Sarmiento, and S.A. Levin. 2013. Marine taxa track local climate velocities. Science 341:1239-1242.
47. Morley, J.W., R.L. Selden, R.J. Latour, T.L. Frolicher, R.J. Seagraves, and M.L. Pinsky. 2018. Projecting shifts in thermal habitat for 686 species on the North American continental shelf. PloS ONE 13(5):e0196127.
48. Morley, J.W., R.D. Batt, and M.L. Pinsky. 2017. Marine assemblages respond rapidly to winter climate variability. Global Change Biology 23:2590-2601.
49. Carballo-Bolanos, R., D. Soto, and C.A. Chen. 2020. Thermal stress and resilience of corals in a climate-changing world. Journal of Marine Science and Engineering 8(1):15. doi.org/10.3390/jmse8010015.
50. Steeves, L.E., R. Filgueira, T. Guyondet, J. Chasse, and L. Comeau. 2018. Past, present, and future: Performance of two bivalve species under changing environmental conditions. Frontiers in Marine Science 5:184 doi:10.3389/fmars.2018.00184.
51. Collier, C.J., and M. Waycott. 2014. Temperature extremes reduce seagrass growth and induce mortality. Marine Pollution Bulletin 83(2): doi:10.1016/j.marpolbul.2014.03.050.
52. Tracy, A.M., M.L. Pielmeier, R.M. Hoshioka, S.F. Heron, and C.D. Harvell. 2019. Increases and decreases in marine disease reports in an era of global change. Proceedings of the Royal Society B 286:20191718.
53. Stevens, A.M., and C.J. Gobler. 2018. Interactive effects of acidification, hypoxia, and thermal stress on growth, respiration, and survival of four North Atlantic bivalves. Marine Ecology Progress Series 604:143-161.
54. Witherington, B.E., and L.M. Ehrhart. 1989. Hypothermic stunning and mortality of marine turtles in the Indian River Lagoon system. Copeia 1989:696-703.
55. Provancha, J.A., K.G. Mota, E.A. Holloway-Adkins, R.H. Reyier, D.M. Scheidt, and M. Epstein. 2005. Mosquito lagoon sea turtle cold stun event of January 2003, Kennedy Space Center/Merritt Island National Wildlife Refuge, Florida. Florida Scientist 68:114-121.

56. Ellis, T.A., J.A. Buckel, and J.E. Hightower. 2017. Winter severity influences spotted seatrout mortality in a southeast U.S. estuarine system. Marine Ecology Progress Series 564:145-161.
57. Shearman, R.K., and S.J. Lentz. 2010. Long-term sea surface temperature variability along the U.S. East Coast. Journal of Physical Oceanography 40:1004-1017.
58. Banzon, V., T.M. Smith, T.M. Chin, C. Liu, and W. Hankins. 2016. A long-term record of blended satellite and *in situ* sea-surface temperature for climate monitoring, modeling and environmental studies. Earth Systems Science Data 8:165-176. doi:10.5194/essd-8-165-2016.
59. OBPG. 2015. MODIS Aqua Level 3 SST Thermal IR Monthly 4km Daytime v2014.0. Ver 2014.0. PO.DAAC, CA, USA. [online] Available https://doi.org/10.5067/MODSA-MO4D4 [accessed 10 July 2018].
60. Bacheler, N.M., and J.C. Ballenger. 2018. Decadal-scale decline of scamp (*Mycteroperca phenax*) abundance along the southeast United States Atlantic coast. Fisheries Research 204:74-87.
61. South Carolina Department of Natural Resources Marine Resources Research Institute. Coastal Trawl Survey and Data Management. Available online at https://www.dnr.sc.gov/marine/mrri/CoastalResearch/SEAMAP/index.html.
62. Bacheler, N.M., N.R. Geraldi, M.L. Burton, R.C. Munoz, and G.T. Kellison. 2017. Comparing relative abundance, lengths, and habitat of temperate reef fishes using simultaneous underwater visual census, video, and trap sampling. Marine Ecology Progress Series 574:141-155.
63. Hyun, K.H., and R. He. 2010. Coastal upwelling in the South Atlantic Bight: A revisit of the 2003 cold event using long term observations and model hindcast solutions. Journal of Marine Systems 83:1-13.
64. Seidov, D., O.K. Baranova, T.P. Boyer, S.L. Cross, A.V. Mishonov, A.R. Parsons, J.R. Reagan, and K.A. Weathers. 2019. Southwest North Atlantic Regional Climatology (NCEI Accession 0201696). [Temperature]. NOAA National Centers for Environmental Information. Dataset. https://doi.org/10.25921/s3ag-2p18. Accessed [07-09-2020].
65. Garcia, H. E., T. P. Boyer, R. A. Locarnini, O. K. Baranova, and M. M. Zweng. 2018. World Ocean Database 2018: User's Manual. A.V. Mishonov, Technical Ed., NOAA, Silver Spring, MD (Available at https://www.NCEI.noaa.gov/OC5/WOD/pr_wod.html).
66. Thomson, R. E., and W. J. Emery. 2014: Data Analysis Methods in Physical Oceanography. 3rd ed. Elsevier. 716 pp.
67. Garcia H.E., T.P. Boyer, O.K. Baranova, R.A. Locarnini, A.V. Mishonov, A. Grodsky, C.R. Paver, K.W. Weathers, I.V. Smolyar, J.R. Reagan, D. Seidov, and M.M. Zweng. 2019. World Ocean Atlas 2018: Product Documentation. A.V. Mishonov (technical editor).
68. Seidov, D., O.K. Baranova, T. Boyer, S.L. Cross, A.V. Mishonov, and A.R. Parsons. 2016. Northwest Atlantic Regional Ocean Climatology. NOAA Atlas NESDIS 80, TA.V.

Mishonov (technical editor). Silver Spring, MD. 56pp.; doi:10.7289/V5/ATLAS-NESDIS-80, (dataset doi:10.7289/V5RF5S2Q).
69. Anderson, D.L.T., and R.A. Cory.1985. Seasonal transport variations in the Florida Straits: A model study. Journal of Physical Oceanography 15:773-786.
70. Lee, T.N., and E. Williams. 1988. Wind-forced transport fluctuations of the Florida Current. Journal of Physical Oceanography 18:937-946.
71. Peng, G., Z. Garraffo, G.R. Halliwell, O-M. Smedstad, C.S. Meinen, V. Kourafalou, and P. Hogan. 2009. Temporal variability of the Florida Current transport at 27N. In: Columbus, F. (editor), Ocean Circulation and El Nino: The New Research. NOVA Science Publishers, Hauppauge, NY.
72. DiNezio, P.N., L.J. Gramer, W.E. Johns, C.S. Meinen, and M.O. Baringer. 2009. Observed interannual variability of the Florida Current: Wind forcing and the North Atlantic Oscillation. Journal of Physical Oceanography 39:721-736. doi:10.1175/2008JPO4001.1.
73. Baringer, M.O., and J.C. Larsen. 2001. Sixteen years of Florida Current transport at 27N. Geophysical Research Letters 28:3179-3182.
74. Meinen, C.S., M.O. Baringer, and R.F. Garcia. 2010. Florida Current transport variability: An analysis of annual and longer-period signals. Deep-Sea Research I 57:835-846. doi:10.1016/j.dsr.2010.04.001.
75. Lee, T.N., J.A. Yoder, and L.P. Atkinson. 1991. Gulf Stream frontal eddy influence on productivity of the Southeast U.S. continental shelf. Journal of Geophysical Research: Oceans 96:22191-22205.
76. Schmitz, W.J., and M.S. McCartney.1993. On the North Atlantic Circulation. Reviews of Geophysics 31:29-49. doi:10.1029/92RG02583.
77. Frankignoul, C., G. de Coetlogon, T. M. Joyce, and S. Dong. 2001. Gulf Stream variability and ocean–atmosphere interactions. Journal of Physical Oceanography 31:3516-3529.
78. Yuan, Y., R.M. Castelao, and R. He. 2017. Variability in along-shelf and cross-shelf circulation in the South Atlantic Bight. Continental Shelf Research 134:52-62.
79. Stegmann, P.M., and J.A. Yoder 1996. Variability of sea-surface temperature in the South Atlantic Bight as observed from satellite: Implications for offshore-spawning fish. Continental Shelf Research 16:843-861.
80. Signorini, S.R., and C.R. McClain 2007. Large-scale forcing impact on biomass in the South Atlantic Bight. Geophysical Research Letters 34:L21605. doi:10.1029/2007GL031121.
81. Zeng, X., and R. He 2016. Gulf Stream variability and a triggering mechanism of its large meander in the South Atlantic Bight. Journal of Geophysical Research: Oceans 11:8021-8038. doi:10.1002/2016JC012077.

82. Blanton, B.O., L.P. Atkinson, L.J. Pietrafesa, and T.N. Lee. 1981. The intrusion of Gulf Stream water across the continental shelf due to topographically induced upwelling. Deep Sea Research Part A. 28:393-405. doi:10.1016/0198-0149(81)90006-6.
83. Castelao, R. 2011. Intrusions of Gulf Stream waters onto the South Atlantic Bight shelf. Journal of Geophysical Research 116:C10011. doi:10.1029/2011JC007178.
84. Miles, T.N., R. He, and M. Li. 2009. Characterizing the South Atlantic Bight seasonal variability and cold-water event in 2003 using a daily cloud-free SST and chlorophyll analysis. Geophysical Research Letters 36:L02604. doi:10.1029/2008GL036396.
85. Gula, J., M.J. Molemaker, and J.C. McWilliams. 2015. Topographic vorticity generation, submesoscale instability and vortex street formation in the Gulf Stream. Geophysical Research Letters 42(10):4054-4062.
86. US DOC/NOAA/NWS NDBC National Data Buoy Center. 1971. Meteorological and oceanographic data collected from the National Data Buoy Center Coastal-Marine Automated Network (C-MAN) and moored (weather buoys). [Station 41008 (LLNR 833) – Grays Reef – 40 NM Southeast of Savannah, GA]. NOAA National Centers for Environmental Information. Dataset. https://www.ncei.noaa.gov/archive/accession/NDBC-CMANWx. Accessed [09-17-2019].
87. Schwing, F.B., M. O. Farrell, J. Steger, and K. Baltz. 1996. Coastal upwelling indices, East coast of North America, 1946-1995. U.S. Department of Commerce, NOAA Technical Memorandum, NOAA-TM-NMFS-SWFC-231, 207 pp.
88. Smyth, K., and M. Elliott. 2016. Effects of changing salinity on the ecology of the marine environment. Chapter 9 *in* M. Solan and N. Whitely (editors), Stressors in the Marine Environment. Open University Press, Oxford Scholarship Online. doi: 10.1093/acprof:oso/9780198718826.001.0001.
89. Velasco, J., C. Gutierrez-Canovas, M. Botella-Cruz, D. Sanchez-Fernandex, P. Arribas, J. A. Carbonell, A. Millan, and S. Pallares. 2018. Effects of salinity changes on aquatic organisms in a multiple stressor context. Philosophical Transactions of the Royal Society B 374:20180011. https://doi.org/10.1098/rstb.2018.0011.
90. Conrads, P.A., and L. S. Darby 2017. Development of a coastal drought index using salinity data. Bulletin of the American Meterological Society 4:753-766. https://doi.org/10.1175/BAMS-D-15-00171.1.
91. Petkewich, M.D., B.J. McCloskey, L.F. Rouen, and P.A. Conrads. 2019. Coastal Salinity Index for Monitoring Drought: U.S. Geological Survey data release, https://doi.org/10.5066/P9MQLNL2.
92. Petkewich, M.D., K. Lackstrom, B.J. McCloskey, L.F. Rouen, and P.A. Conrads. 2019. Coastal Salinity Index along the southeastern Atlantic coast and the Gulf of Mexico, 1983 to 2018: U.S. Geological Survey Open-File Report 2019–1090, 26 p. https://doi.org/10.3133/ofr20191090.
93. Graham, J., and K. Hoenke. 2019. Building a network of partners working to reconnect fragmented streams in the Southeastern United States through actionable science. *In* D.C.

Dauwalter, T.W. Birdsong, and G.P. Garrett (editors), Multispecies and Watershed Approaches to Freshwater Fish Conservation. American Fisheries Society Symposium 91:40-413.

94. Sadeghi, S., G. Tootle, E. Elliott, V. Lakshmi, M. Therrell, J. Kam, and B. Bearden. 2019. Atlantic ocean sea surface temperatures and Southeast United States streamflow variability: Associations with the recent multi-decadal decline. Journal of Hydrology 576:422-429.
95. Kourafalou, V.H., T.N. Lee, L.Y. Oey, and J.D. Wang. 1996. The fate of river discharge on the continental shelf 2. Transport of coastal low-salinity waters under realistic wind and tidal forcing. Journal of Geophysical Research 101:3435-3455.
96. U.S. Geological Survey. 2016. National Water Information System data available on the World Wide Web (USGS Water Data for the Nation), accessed [July 6, 2020], at URL [http://waterdata.usgs.gov/nwis/]. http://dx.doi.org/10.5066/F7P55KJN.
97. Dubrovsky, N.M., K.R. Burow, G.M. Clark, J.M. Gronberg, P.A. Hamilton, K.J. Hitt, D.K. Mueller, M.D. Munn, B.T. Nolan, L.J. Puckett, M.G. Rupert, T.M. Short, N.E. Spahr, L.A. Sprague, and W.G. Wilber. 2010. The Quality of our Nation's Waters—Nutrients in the Nation's Streams and Groundwater, 1992–2004. US Geological Survey Circular 1350. Reston, VA: US Geological Survey. Available at: https://pubs.usgs.gov/circ/1350/.
98. Nixon S.W. 1995. Coastal marine eutrophication: A definition, social causes, and future concerns. Ophelia 41:199-219.
99. Rabalais, N.N., R.E. Turner, R.J. Díaz, and D. Justić. 2009. Global change and eutrophication of coastal waters. ICES Journal of Marine Science 66:1528–1537. doi.org/10.1093/icesjms/fsp047.
100. Breitburg D., S. Seitzinger, and J. Sanders. 1999. The effects of multiple stressors on freshwater and marine ecosystems. Limnology and Oceanography 44:739-972.
101. Moorman, M.C., A.B. Hoos, S.B. Bricker, R.B. Moore, A.M. García, and S.W. Ator. 2014. Nutrient load summaries for major lakes and estuaries of the Eastern United States, 2002: U.S. Geological Survey Data Series 820, 94 p. Available at: http://dx.doi.org/10.3133/ds820.
102. Hoos, A.B., and V.L. Roland II. 2019. Spatially referenced models of streamflow and nitrogen, phosphorus, and suspended-sediment loads in the Southeastern United States: U.S. Geological Survey Scientific Investigations Report 2019–5135, 87 p. Available at: https://doi.org/10.3133/sir20195135.
103. Labosier, C.F., and S.M. Quiring. 2013. Hydroclimatology of the Southeastern USA. Climate Research 57:157-171.
104. Wang, H., R. Fu, A. Kumar, and W. Li. 2010. Intensification of summer rainfall variability in the Southeastern United States during recent decades. Journal of Hydrometerology 11:1007-1018. doi.org/10.1175/2010JHM1229.1.

105. Kim, D., S.-K. Lee, H. Lopez, G.R. Foltz, V. Misra, and A. Kumar. 2020. On the role of Pacific-Atlantic SST contrast and associated Caribbean Sea Convection in August-October U.S. regional rainfall variability. Geophysical Research Letters. doi: https://doi.org/10.1029/2020GL087736.
106. Anandhi, A., C. Crandall, and C. Bentley. 2018. Hydrologic characteristics of streamflow in the Southeast Atlantic and Gulf coast hydrologic region during 1939-2016 and conceptual map of potential impacts. Hydrology 5:42. doi:10.3390/hydrology5030042.
107. Dourte, D.R., C.W. Fraisse, and W.L. Bartels. 2015. Exploring changes in rainfall intensity and seasonal variability in the Southeastern U.S.: Stakeholder engagement, observations, and adaptation. Climate Risk Management 7:11-19.
108. Mitchell, R.J., Y. Liu, J.J. O'Brien, K.J. Elliott, G. Starr, C.F. Miniat, and J.K. Hiers. 2014. Future climate and fire interactions in the southeastern region of the United States. Forest Ecology and Management 327:315-326.
109. Hester, C.M., and K.L. Larson. 2016. Time-series analysis of water demands in three North Carolina cities. Journal of Water Resources Planning and Management 142 (8):05016005.
110. Singh, V.P., A.K. Mishra, H. Chowdhary, and C.P. Khedun. 2014. Climate change and its impact on water resources. Pages 525-569 *in* L.K. Wang and C.T. Yang (editors) Handbook of Environmental Engineering, Volume 15: Modern Water Resources Engineering. Springer Science, NY, NY. doi:10.1007/978-1-62703-595-8_11.
111. Rocca, M.E., C.F. Miniat, and R.J. Mitchell. 2014. Introduction to the regional assessments: Climate change, wildfire, and forest ecosystem services in the USA. Forest Ecology and Management 327:265-268.
112. Selman, C., V. Misra, L. Stefanova, S. Dinapoli, and T.J. Smith. 2013. On the twenty-first-century wet season projections over the Southeastern United States. Regional Environmental Change 13:153-164.
113. The Southeast Climate Center, "Monthly and Seasonal Climate Information," [Online]. Available: https://sercc.com/climateinfo/monthly_seasonal. [Accessed 19 February 2019].
114. National Integrated Drought Information System, "U.S. Drought Monitor-Southeastern RCC," Available: https://www.drought.gov/drought/rcc/southeast. [Accessed 15 February 2019].
115. Church, J.A., and N.J. White. 2011. Sea-level rise from the late 19^{th} to the early 21^{st} century. Surveys in Geophysics 32:585-502.
116. Valle-Levinson, A., A. Dutton, and J.B. Martin. 2017. Spatial and temporal variability of sea level rise hot spots over the eastern United States. Geophysical Research Letters 44:7876-7882. doi:10.1002/2017GL073926.
117. Von Holle, B., J.L. Irish, A. Spivy, J.F. Weishampel, A. Meylan, M.H. Godfrey, M. Dodd, S.H. Schweitzer, T. Keyes, F. Sanders, M.K. Chaplin, and N.R. Taylor. 2019. Effects of future sea level rise on coastal habitat. The Journal of Wildlife Management 83:694-704. doi:10.1002/jwmg.21633.

118. NOAA Center for Operational Oceanographic Products and Services, “NOAA Tides and Currents,” [Online]. Available: https://tidesandcurrents.noaa.gov/sltrends/sltrends.html. [Accessed 12 December 2018].
119. Weinkle, J., C. Landsea, D. Collins, R. Musulin, R.P. Crompton, P.H. Klotzbach, and R. Pielke. 2018. Normalized hurricane damage in the continental United States 1900-2017. Nature Sustainability 1:808-813. doi.org/10.1038/s41893-018-0165-2.
120. Edwards, M., G. Beaugrand, P. Helaouet, J. Alheit and S. Coombs. 2013. Marine ecosystem response to the Atlantic Multidecadal Oscillation. PLoS ONE, 8(2):e57212. doi.org/10.1371/journal.pone.0057212.
121. Waple, A.M., J.H. Lawrimore, M.S. Halpert, G.D. Bell, W. Higgins, B. Lyon, M.J. Menne, K.L. Gleason, R.C. Schnell, J.R. Christy, W. Thiaw, W.J. Wright, M.J. Salinger, L. Alexander, R.S. Stone, and S.J. Camargo. 2002. Climate assessment for 2001. Bulletin of the American Meterological Society 83:S1-S62.
122. Knapp, K.R., M.C. Kruk, D.H. Levinson, H.J. Diamond, and C.J. Neumann. 2010. The International Best Track Archive for Climate Stewardship (IBTraCS): Unifying tropical cyclone best track data. Bulletin of the American Meteorological Society 91:363-376.
123. Knapp, K.R., H.J. Diamond, J.P. Kossin, M.C. Kruk, and C.J. Schreck. 2018. International Best Track Archive for Climate Stewardship (IBTraCS) Project, Version 4. [All.list]. NOAA National Centers for Environmental Information. https://doi.org/10.25921/82ty-9e16 [accessed 17 December 2018].
124. Paerl, H.W., J.D., Bales, L.W. Ausley, C.P. Buzzelli, L.B. Crowder, L.A. Eby, J.M. Fear, M. Go, B.L. Peierls, T.L. Richardson, and J.S. Ramus. 2001. Ecosystem impacts of three sequential hurricanes (Dennis, Floyd, and Irene) on the U.S.’s largest lagoonal estuary, Pamlico Sound, NC. Proceedings of the National Academy of Science USA 98:5655-5660.
125. NOAA Atlantic Oceanographic and Meterological Laboratory, Hurricane Research Division, “Hurricane Data,” [Online]. Available: http://www.aoml.noaa.gov/hrd/data_sub/hurr.html. [Accessed 04 December 2018].
126. Landsea, C., and E. Blake. An Incredibly Busy Hurricane Season. [Online]. Available: https://noaanhc.wordpress.com/2021/06/30/was-2020-a-record-breaking-hurricane-season-yes-but/. [Accessed 01 July 2021].
127. Le Quere, C., R.M. Andrew, J.G. Canadell, S.Sitch, J.I. Korsbakken, and 62 others. Global Carbon Budget 2016. Earth Systems Science Data 8:605-649. doi.org/10.5194/essd-8-605-2016.
128. McKinley, G. A., A.R. Fay, T. Takahashi, and N. Metzl. 2011. Convergence of atmospheric and North Atlantic carbon dioxide trends on multidecadal timescales. Nature Geoscience, 4(9):606–610. doi.org/10.1038/ngeo1193.
129. Reimer, J.J., H. Wang, R. Vargas, and W.J. Cai. 2017. Multidecadal fCO_2 increase along the United States Southeast Coastal Margin. Journal of Geophysical Research: Oceans 122:10061-10072. doi.org/10.1002/2017JC013170.

130. Hall. E.R., L. Wickes, L.E. Burnett, G.I. Scott, D. Hernandez, K.K. Yates, L. Barbero, J.J. Reimer, M. Baalousha, J. Mintz, W.-J. Cai, J.K. Craig, M.R. DeVoe, W.S. Fisher, T.K. Hathaway, E.B. Jewett, Z. Johnson, P. Keener, R.S. Mordecai, S. Noakes, C. Phillips, P.A. Sandifer, A. Schnetzer, and J. Styron. 2020. Acidification in the U.S. Southeast: causes, potential consequences and the role of the Southeast Ocean and Coastal Acidification Network. Frontiers in Marine Science 7:548. doi.org/10.3389/fmars.2020.00548.
131. Xu, X., M. Chen, G. Yang, B. Jiang, and J. Zhang. 2020. Wetland ecosystem services research: A critical review. Global Ecology and Conservation 22:e01027. doi.org/10.1016/j.gecco.2020.e01027.
132. Weinstein, M.P. 1978. Shallow marsh habitat as primary nurseries for fishes and shellfish, Cape Fear River, North Carolina. Fishery Bulletin U.S. 77:339-357.
133. Boesch, D.F., and R.E. Turner. 1984. Dependence of fishery species on salt marshes: The role of food and refuge. Estuaries 7:460-468.
134. Hughes, R.H. 2004. Climate change and loss of saltmarshes: Consequences for birds. Ibis 146:S21-S26.
135. Cartwright, J.M., and Wolfe, W.J. 2016. Insular ecosystems of the southeastern United States—A regional synthesis to support biodiversity conservation in a changing climate. U.S. Geological Survey Professional Paper 1828. 162 pp. http://dx.doi.org/10.3133/pp1828.
136. Lynch, A.J., B.J.E. Myers, C. Chu, L.A. Eby, J. Falke, R.P. Kovach, T.J. Krabbenhoft, T.J. Kwak, J.F. Lyons, C.P. Paukert, and J.E. Whitney. 2016. Climate change effects on North American inland fish populations and assemblages. Fisheries 41:346-361. doi:10.1080/03632415.2016.1186016.
137. NOAA Office for Coastal Management, “C-CAP Regional Land Cover and Change,” [Online]. Available: www.coast.noaa.gov/htdata/raster1/landcover/bulkdownload/30m_lc/. [Accessed 08 December 2017].
138. Orth, R.J., T.J.B. Carruthers, W.C. Dennison, C.M. Duarte, J.W. Fourqurean, K.L. Heck, Jr., A. R. Hughes, G.A. Kendrick, W. J. Kenworthy, S. Olyarnik, F.T. Short, M. Waycott, and S.L. Williams. 2006. A global crisis for seagrass ecosystems. Bioscience 56:987-996.
139. Hughes, A.R. 2009. Associations of concern: Declining seagrasses and threatened dependent species. Frontiers in Ecology and the Environment 7:242-246. doi:10.1890/080041.
140. Waycott, M., C.M. Duarte, T.J.B. Carruthers, R.J. Orth, W.C. Dennison, S. Olyarnik, A. Calladine, J.W. Fourqurean, K.L. Heck, A.R. Hughes, G.A. Kendrick, W.J. Kenworthy, F.T. Short, and S.L. Williams. 2009. Accelerating loss of seagrasses across the globe threatens coastal ecosystems. Proceedings of the National Academy of Science USA 106 (2009):12377-12381. doi.org/10.1073/pnas.0905620106.

141. Conley, M.F., M.G. Anderson, N. Steinberg, and A. Barnett (editors). 2017. The South Atlantic Bight Marine Assessment: Species, Habitats, and Ecosystems. The Nature Conservancy, Eastern Conservation Science.
142. Field, D., J. Kenworthy, and D. Carpenter. 2020. Metric Report: Extent of Submerged Aquatic Vegetation, High-Salinity Estuarine Waters. Albermarle-Pamlico National Estuary Partnership, Raleigh, NC, 19p. Retrieved at:
143. https://apnep.nc.gov/documents/files/metric-report-extent-submerged-aquaticvegetation-high-salinity-estuarine-waters.Yarbro, L. A., and P. R. Carlson, Jr. (editors). 2016. Seagrass Integrated Mapping and Monitoring Program: Mapping and Monitoring Report No. 2. Fish and Wildlife Research Institute Technical Report TR-17 version 2.vi + 281 p.
144. Bahr, L.M., and W.P. Lanier. 1981. The ecology of intertidal oyster reefs of the South Atlantic coast: A community profile. U.S. Fish and Wildlife Service, Office of Biological Services, Washington, D.C. FWS/OBS-81/15.105 pp.
145. Grabowski, J.H., R.D. Brumbaugh, R.F. Conrad, A.G. Keeler, J.J. Opaluch, C.H. Peterson, M.F. Piehler, S.P. Powers, and A.R. Smyth. 2012. Economic valuation of ecosystem services provided by oyster reefs. BioScience 62:900-909.
146. Piazza, B.P., P.D. Banks, and M.K. La Peyre. 2005. The potential for created oyster shell reefs as a sustainable shoreline protection strategy in Louisiana. Restoration Ecology 13:499-506.
147. Lai, Q.T., E.R. Irwin, and Y. Zhang. 2020. Estimating nitrogen removal services of eastern oyster (*Crassostrea virginica*) in Mobile Bay, Alabama. Ecological Indicators 117:106541. doi.org/10.1016/j.ecolind.2020.106541
148. MacKenzie, C.L. 2007. Causes underlying the historical decline in Eastern Oyster (*Crassostrea Virginica* Gmelin, 1791) landings. Journal of Shellfish Research 26:927-938.
149. Kirby, M.X. 2004. Fishing down the coast: Historical expansion and collapse of oyster fisheries along continental margins. Proceedings of the National Academy of Sciences 101:13096-13099.
150. Beck, M.W., R.D. Brumbaugh, L. Airoldi, A. Carranza, L.D. Coen, C. Crawford, O. Defeo, G.J. Edgar, B. Handcock, M.C. Kay, H.S. Lenihan, M. Luckenbach, C.L. Toropova, G. Zhang, and X. Guo. 2011. Oyster reefs at risk and recommendations for conservation, restoration, and management. BioScience 161:107-116.
151. Wilberg, M.J., M.E. Livings, J.S. Barkman, B.T. Morris, and J.M. Robinson. 2011. Overfishing, disease, habitat loss, and potential extirpation of oysters in upper Chesapeake Bay. Marine Ecology Progress Series 436:131-144.
152. Theuerkauf, S.J., B.J. Puckett, and D.B. Eggleston. 2021. Metapopulation dynamics of oysters: Sources, sinks and implications for conservation and restoration. Ecosphere 12(7):e03573. doi:10.1002/ecs2.3573.
153. Zu Ermgassen, P.S.E., M.D. Spalding, B. Blake, L.D. Coen, B. Dumbauld, S. Teiger, J.H. Grabowski, R. Grizzle, M. Luckenbach, K. McGraw, W. Rodney, J.L. Ruesink, S.P. Powers, and R. Brumbaugh. 2012. Historical ecology with real numbers: Past and present

extent and biomass of an imperiled estuarine habitat. Proceedings of the Royal Society B 279:3393-3400.

154. Deaton, A. North Carolina Division of Marine Fisheries. Personal Communication.
155. SCDNR (South Carolina Department of Natural Resources). http://www.dnr.sc.gov/GIS/descoysterbed.html.
156. GDNR (Georgia Department of Natural Resources), "Wetlands/OysterReefs2015," [Online]. Available: https://geospat.gatech.edu/arcgis/rest/services/Wetlands/OysterReefs2015/MapServer. [Accessed 09 July 2020].
157. Florida Fish and Wildlife Conservation Commission, "Oyster Beds in Florida," [Online]. Available: http://geodata.myfwc.com/datasets/oyster-beds-in-florida. [Accessed 09 July 2020].
158. Cinner, J.E., J. Zamborain-Mason, G.G. Gurney, N.A.J. Graham, M.A. MacNeil, and 32 others. 2020. Meeting fisheries, ecosystem function, and biodiversity goals in a human-dominated world. Science 368(6488):307-311. doi: 10.1126/science.aax9412.
159. Costanza, R., R. de Groot, P. Sutton, S. van der Ploeg, S.J. Anderson, I. Kubiszewski, S. Farber, and R.K. Turner. 2014. Changes in the global value of ecosystem services. Global Environmental Change 26:152-158.
160. de Groot, R., L. Brander, S. van der Ploeg, R. Costanza, F. Bernard, L. Braat, M. Christie, N. Crossman, A. Ghermandi, L. Hein, S. Hussain, P. Kumar, A. McVittie, R. Portela, L.C. Rodriguez, P. ten Brin, and P. van Beukering. 2012. Global estimates of the value of ecosystems and their services in monetary units. Ecosystem Services 1:50-61.
161. Carpenter, K.E., A. Muhammad, G. Aeby, R.B. Aronson, S. Banks, and 36 others. 2008. One-third of reef-building corals face elevated extinction risk from climate change and local impacts. Science 321:560-563.
162. Pandolfi, J.M., S.R. Connolly, D.J. Marshall, and A.L. Cohen. 2011. Projecting coral reef futures under global warming and ocean acidification. Science 333:418-422.
163. Spalding, M.D., and B.E. Brown. 2015. Warm-water coral reefs and climate change. Science 350:6262:769-771. doi: 10.1126/science.aad0349.
164. Johns, G.M., V.R. Leeworthy, F.W. Bell, and M.A. Bonn. 2001. Socioeconomic Study of Reefs in Southeast Florida: South Florida. Florida Fish and Wildlife Conservation Commission, National Oceanic and Atmospheric Administration (NOAA). Available: http://map.marineecosystemservices.org/node/7889 [Accessed 31 August 2020].
165. Brander, L., and P. van Beukering. 2013. The Total Economic Value of U.S. Coral Reefs: A Review of the Literature. Silver Spring, MD: NOAA Coral Reef Conservation Program. Available: http://lukebrander.com/wp-content/uploads/2013/07/Brander-and-van-Beukering-2013-The-total-economic-value-of-US-coral-reefs.pdf [Accessed 31 August 2020].

166. Florida Fish and Wildlife Conservation Commission, “Disturbance Response Monitoring ‘DRM’,” [Online]. Available: http://ocean.floridamarine.org/FRRP/ [Accessed 31 August 2020].
167. National Center for Coastal Ocean Science, “National Coral Reef Monitoring Program Implementation: Biological and Socioeconomic Monitoring,” [Online]. Available: https://coastalscience.noaa.gov/project/national-coral-reef-monitoring-program-biological-socioeconomic/. [Accessed 31 August 2020].
168. Precht, W.F., B.E. Gintert, M.L. Robbart, R. Fura, and R. van Woesik. 2016. Unprecedented disease-related coral mortality in southeastern Florida. Scientific Reports 6, 31374. doi: 10.1038/srep31374.
169. Muller, E.M., C. Sartor, N.I. Alcaraz, and R. van Woesik. 2020. Spatial epidemiology of the stony-coral-tissue-loss disease in Florida. Frontiers in Marine Science. doi:10.3389/fmars.2020.00163.
170. National Park Service. Stony coral tissue loss disease found at Dry Tortugas National Park. [online]. Available at https://www.nps.gov/drto/learn/news/stony-coral-tissue-loss-disease-found-at-dry-tortugas-national-park.htm. [Accessed 2 June 2021].
171. Glynn, P.W. 1993. Coral reef bleaching: Ecological perspectives. Coral Reefs 12:1-17.
172. Hoegh-Guldberg, O., P.J. Mumby, A.J. Hooten, R.S. Stenech, P. Greenfield, E. Gomex, C.D. Harvell, P.F. Sale, A.J. Edwards, K. Caldeira, N. Knowlton, C.M. Eakin, R. Iglesias-Prieto, N. Muthiga, R.H. Bradbury, A. Dubi, and M.E. Hatziolos. 2007. Coral reefs under rapid climate change and ocean acidification. Science 318:1737-1742.
173. Liu, G., S.F. Heron, C.M. Eakin, F.E. Muller-0Karger, M. Vega-Rodriguez, and 12 others. 2014. Reef-scale thermal stress monitoring of coral ecosystems: New 5-km global products from NOAA Coral Reef Watch. Remote Sensing 6:11579-11606. doi:10.3390/rs61111579.
174. NOAA Satellite and Information Service, National Environmental Satellite, Data, and Information Service (NESDIS), “Coral Reef Watch Satellite Monitoring and Modeled Outlooks,” [Online]. Available: https://coralreefwatch.noaa.gov/satellite/index.php. [Accessed 06 June 2020].
175. O’Reilly, J. E., S. Maritorena, B.G. Mitchell, D.A. Siegel, K.L. Carder, S.A. Garver, M. Kahru, and C.R. McClain. 1998. Ocean color chlorophyll algorithm for SeaWiFS. Journal of Geophysical Research 103:24937-24953. doi.org/10.1029/98JC02160.
176. Atkinson, L.P., J.A. Yoder, and T.N. Lee. 1984. Review of upwelling off the southeastern United States and its effect on continental shelf nutrient concentrations and primary productivity. Rapports et Process-verbaus des Reunions 183:70-78.
177. Verity, P.G., J.O. Blanton, J. Amft, C. Barnas, D. Knott, B. Stender, and E. Wenner. 1998. Influences of physical oceanographic processes on chlorophyll distributions in coastal and estuarine waters of the South Atlantic Bight. Journal of Marine Research 56:681-711.

178. Atkinson, L.P. 1985. Hydrography and nutrients of the Southeastern U.S. continental shelf. In: Atkinson, L.P., et al. (editors), Oceanography of the Southeastern U.S. Continental Shelf. Coastal and Estuarine Sciences, 2. AGU, Washington D.C. pp.77-92.
179. Lee, T.N., J.A. Yoder, and L.P. Atkinson. 1991. Gulf Stream frontal eddy influence on productivity of the southeast U.S. continental shelf. Journal of Geophysical Research 96:22191-22205.
180. Miles, T.N., and R. He. 2010. Temporal and spatial variability of Chl-a and SST on the South Atlantic Bight: Revisiting with cloud-free reconstructions of MODIS satellite imagery. Continental Shelf Research 30:1951-1962.
181. NASA EarthData, "MODIS-Aqua," [Online]. Available: https://oceancolor.gsfc.nasa.gov/data/aqua/. [Accessed 07 July 2018].
182. Micheli, F. 1999. Eutrophication, fisheries, and consumer-resource dynamics in marine pelagic ecosystems. Science 285:1396-1398.
183. Govoni, J.J., and J.A. Hare. 2001. The Charleston Gyre as a spawning and larval nursery habitat for fishes. American Fisheries Society Symposium 25:123-136.
184. Friedland, K.D., R.T. Leaf, J. Kane, D. Tommasi, R.G. Asch, N. Rebuck, R. Ji, S.I. Large, C. Stock, and V.S. Saba. 2015. Spring bloom dynamics and zooplankton biomass response on the U.S. Northeast Continental Shelf. Continental Shelf Research 102:47-61.
185. Turner, J.T. 2002. Zooplankton fecal pellets, marine snow, and sinking phytoplankton blooms. Aquatic Microbial Ecology 27:57-102.
186. Verity, P.G., D.G. Redalje, S.R. Lohrenz, C. Flagg, and R. Hristov. 2002. Coupling between primary production and pelagic consumption in temperate ocean margin pelagic ecosystems. Deep-Sea Research II 49:4553-4569.
187. Keister, J. E., D. Bonnet, S. Chiba, C. L. Johnson, D. L. Mackas, and R. Escribano. 2012. Zooplankton population connections, community dynamics, and climate variability. ICES Journal of Marine Science 69:347–350.
188. NOAA National Marine Fisheries Service, "COPEPOD: The Global Plankton Database Project," [Online]. Available at: https://www.st.nmfs.noaa.gov/copepod/. [Accessed 30 July 2020].
189. Postel, L., H.O. Fock, and W. Hagen. 2000. Biomass and abundance. In: R. Harris, H.R. Skjoldal, J. Lenz, P. Wiebe, and M. Huntley (editors), International Council for the Exploration of the Sea (ICES) Zooplankton Methodology Manual. Academic Press, New York.
190. Karnauskas, M., C.R. Kelble, S. Regan, C. Quenee, R. Allee, M. Jepson, A. Freitag, J.K. Craig, C. Carollo, L. Barbero, N. Trifonova, D. Hanisko, and G. Zapfe. 2017. Ecosystem status report update for the Gulf of Mexico. NOAA Technical Memorandum NMFS-SEFSC-706, 51 p.
191. Bi, H., J. Rubao, H. Liu, Y.H. Jo, and H.A. Hare. 2014. Decadal changes in zooplankton of the Northeast U.S. continental shelf. PlosOne 9(1):e87720.

192. Kimball, M.E., D.M. Allen, P.D. Kenny, and M.V. Ogburn. 2020. Decadal-scale changes in subtidal nekton assemblages in a warm-temperate estuary. Estuaries and Coasts 43:927-939.
193. Pfirrmann, B.W., M.E. Kimball, M.M. Mace, and B.D. Turley. 2021. Summer ichthyoplankton assemblage diversity within a southeastern United States estuary. Estuaries and Coasts 441(1):253-268.
194. Allen, D.M., V. Ogburn-Matthews, T. Buck, and E.M. Smith. 2008. Mesozooplankton response to climate change and variability in a southeastern U.S. estuary (1981-2003). Journal of Coastal Research SI(55):95-110.
195. Thaxton, W.C., J.C. Taylor, and R.G. Asch. 2020. Climate-associated trends and variability in icthyoplankton phenology from the longest continuous larval fish time series on the east coast of the United States. Marine Ecology Progress Series 650:269-287.
196. Taylor, J.C., W.A. Mitchell, J.A. Buckel, H.J. Walsh, K.W. Shertzer, G. Bath-Martin, and J.A. Hare. 2009. Relationships between larval and juvenile abundance of winter-spawned fishes in North Carolina, USA. Marine and Coastal Fisheries 1:12-21.
197. Taylor, J.C., J.M. Miller, L.J. Pietrafesa, D.A. Dickey, and S.W. Ross. 2010. Winter winds and river discharge determine juvenile southern flounder (*Paralichthys lethostigma*) recruitment and distribution in North Carolina estuaries. Journal of Sea Research 64:15-25.
198. Able, K.W., M.C. Sullivan, J.A. Hare, G. Bath-Martin, J.C. Taylor, and R. Hagan. 2011. Larval abundance of summer flounder (*Parealichthys dentatus*) as a measure of recruitment and stock status. Fishery Bulletin, U.S. 109:68-78.
199. Adamski, K.M., J.A. Buckel, K.W. Shertzer, G. Bath-Martin, J.C. Taylor. 2011. Developing fishery-independent indices of larval and juvenile Gag abundance in the Southeastern United States. Transactions of the American Fisheries Society 140:973-983.
200. Allen, D.M., and D.L. Barker. 1990. Interannual variations in larval fish recruitment to estuarine epibenthic habitats. Marine Ecology Progress Series 63:113-125.
201. SEDAR. 2020a. SEDAR 69 – Atlantic Menhaden Benchmark Stock Assessment Report. SEDAR, North Charleston SC. 691 pp. available online at: http://sedarweb.org/sedar-69.
202. SEDAR. 2020b. SEDAR 69 – Atlantic Menhaden Ecological Reference Points
203. South Carolina Department of Natural Resources Marine Resources Research Institute. Reef Fish Survey (RFS). https://www.dnr.sc.gov/marine/mrri/CoastalResearch/ReefFishSurvey/index.html. Accessed [6 November 2018].
204. Bacheler N.M., and T.I. Smart. 2016. Multi-decadal decline in reef fish abundance and species richness in the southeast USA assessed by standardized trap catches. Marine Biology 163:26. doi:10.1007/s00227-015-2774-x.
205. Pennington, M. 1983. Efficient estimators of abundance for fish and plankton surveys. Biometrics 39:281-286.

206. Edwards, P. E. T. (editor). 2013. Summary Report: The Economic Value of U.S. Coral Reefs. NOAA Coral Reef Conservation Program, Silver Spring, MD. 28 p. Available online at: https://www.coris.noaa.gov/activities/economic_summary/.
207. Ault, J.S., S.G. Smith, J.A. Bohnsack, J.G. Luo, N. Zurcher, D.B. McClellan, T.A. Ziegler, D.E. Hallac, M. Patterson, M.W. Feeley, B.I. Ruttenberg, J. Hunt, D. Kimball, and B. Causey. 2013. Assessing coral reef fish population and community changes in response to marine reserves in the Dry Tortugas, Florida, USA. Fisheries Research 144:S28-S37.
208. Brandt, M.E., N. Zurcher, A. Acosta, J.S. Ault, J.A. Bohnsack, M.W. Feeley, D.E. Harper, J.H. Hunt, T. Kellison, D.B. McClellan, M.E. Patterson, and S.G. Smith. 2009. A cooperative multi-agency reef fish monitoring protocol for the Florida Keys coral reef ecosystem. Natural Resource Report NPS/SFCN/NRR-2009/150. National Park Service, Fort Collins, Colorado.
209. Smith, S.G., J. Ault, J. Bohnsack, J.D. Harper, J. Luo, and D. McClellan. 2011. Multispecies survey design for assessing reef-fish stocks, spatially explicit management performance, and ecosystem condition. Fisheries Research 109:25-41. doi:10.1016/j.fishres.2011.01.012.
210. Pauly, D., V. Christensen, J. Dalsgaard, R. Froese, and F. Torres, Jr. 1998. Fishing down marine food webs. Science 279:860-863. doi:10.1126/science.279.5352.860.
211. Pauly, D., and R. Watson. 2005. Background and interpretation of the 'Marine Trophic Index' as a measure of biodiversity. Philosophical Transactions of the Royal Society B 360:415-423. doi:10.1098/rstb.2004.1597.
212. Branch, T. A., R. Watson, E. A. Fulton, S. Jennings, C. R. McGilliard, G. T. Pablico, D. Ricard, and S. R. Tracey. 2010. The trophic fingerprint of marine fisheries. Nature 468:431-435. doi:10.1038/nature09528.
213. Atlantic Coastal Cooperative Statistics Program (ACCSP), "Data Warehouse," [Online]. Available: https://www.accsp.org/ [Accessed 18 April 2019].
214. NOAA Fisheries, "Recreational Fishing Data," [Online]. Available: https://www.fisheries.noaa.gov/topic/recreational-fishing-data. [Accessed 16 October 2018].
215. Essington, T.E., A.H. Beaudreau, and J. Wiedenmann. 2006. Fishing through marine food webs. Proceedings of the National Academy of Sciences 103:3171-3175.
216. McClanahan, T.R., and C.C. Hicks. 2010. Changes in life history and ecological characteristics of coral reef fish catch composition with increasing fishery management. Fisheries Management and Ecology 18:50-60. doi:10.1111/j.1365-2400.2010.00768.x.
217. de Juan, S., H. Hinz, P. Sartor, and S. Vitale. 2020. Vulnerability of demersal fish assemblages to trawling activities: A traits-based index. Frontiers in Marine Science 7(44). doi:10.3389/fmars.2020.00044.
218. SEDAR (Southeast Data, Assessment, and Review). http://sedarweb.org/

219. Brooks, E.N., and J.J. Deroba. 2015. When "data" are not data: The pitfalls of post hoc analyses that use stock assessment model output. Canadian Journal of Fisheries and Aquatic Sciences 72:634–641.
220. Shertzer, K.W., E.H. Williams, J.K. Craig., E.E. Fitzpatrick, N. Klibansky, and K.I. Siegfried. 2019. Recreational sector is the dominant source of fishing mortality for oceanic fishes in the Southeastern United States Atlantic Ocean. Fisheries Management and Ecology 26:621-629. https://doi.org/10.1111/fme.12371.
221. ASMFS. 2004. Status of the Blue Crab (*Callinectes sapidus*) on the Atlantic coast. Special Report No. 80. 42 pp.
222. NCDMF (North Carolina Division of Marine Fisheries). 2018. Stock assessment of the North Carolina blue crab (*Callinectes sapidus*), 1995–2016. North Carolina Division of Marine Fisheries, NCDMF SAP-SAR-2018-02, Morehead City, North Carolina. 144 p.
223. FWC. 2013. A stock assessment for blue crab, *Callinectes sapidus*, in Florida waters through 2011. Florida Fish and Wildlife Commission, In House Report. 132 pp.
224. Keithly, W.R., Jr., and P. Poudel. 2008. The Southeast USA shrimp industry: Issues related to trade and antidumping duties. Marine Resource Economics 2:459-483.
225. Fuller, P.L., D.M. Knott, P.R. Kingsley-Smith, J.A. Morris, C.A. Buckel, M.E. Hunter, and L.D. Hartman. 2014. Invasion of Asian tiger shrimp, *Penaeus monodon* Fabricius, 1798, in the western north Atlantic and Gulf of Mexico. Aquatic Invasions 9:59-70.
226. MacKenzie, C.L., V.G Burrell, Jr., A. Rosenfield, and W.LO. Hobart. 1997. The history, present condition and future of the molluscan fisheries of North and Central America and Europe, vol. 1: Atlantic and Gulf Coasts. U.S. Department of Commerce, NOAA Technical Report 127, 234 pp. Available: https://spo.nmfs.noaa.gov/sites/default/files/tr127opt.pdf
227. Burrell,V.G., Jr. 2003. South Carolina Oyster Industry: A History. [Online]. Available: http://mrl.cofc.edu/pdf/OysterIndusSC.pdf. [Accessed 01 September 2020].
228. Hernandez, A.B., R.D. Brumbaugh, P. Frederick, R. Grizzle, M.W. Luckenbach, C.H. Peterson, and C. Angelini. 2018. Restoring the eastern oyster: How much progress has been made in 53 years? Frontiers in Ecology and the Environment doi:10.1002/fee.1935.
229. NOAA Fisheries Office of Sustainable Fisheries, "Status of Stocks 2019: Annual Report to Congress on the Status of U.S. Fisheries,"[Online]. Available: https://www.fisheries.noaa.gov/national/population-assessments/fishery-stock-status-updates. [Accessed 20 Aug 2020].
230. South Atlantic Fisheries Management Council, "SEDAR: Southeast Data, Assessment, and Review," [Online]. Available: http://sedarweb.org/. [Accessed 20 Aug 2020].
231. NOAA Fisheries Office of Science and Technology, "Stock Smart: Status, Management, Assessments, and Resource Trends," [Online]. Available: https://www.st.nmfs.noaa.gov/stocksmart. [Accessed 20 Aug 2020].
232. Crozier, G.E., and D.E. Gawlik. 2003. Wading bird nesting effort as an index to wetland ecosystem integrity. Waterbirds 26:303-324.

233. Ogden, J.C., J.D. Baldwin, O.L. Bass, J.A. Browder, M.I. Cook, P.C. Frederick, P.E. Frezza, R.A. Galvez, A.B. Hodgson, K.D. Meyer, L.D. Oberhofer, A.F. Paul, P.J. Fletcher, S.M. Davis, and J.J. Lorenz. 2014. Waterbirds as indicators of ecosystem health in the coastal marine habitats of Southern Florida: 2. Conceptual ecological models. Ecological Indicators 44:128-147.
234. Velarde, E., D.W. Anderson, and E. Ezcurra. 2019. Seabird clues to ecosystem health. Science 365:116-117. doi: 10.1126/science.aaw9999.
235. Furness, R. W., and J. Greenwood (editors). 1995. Birds as Monitors of Environmental Change, London:Chapman & Hall, 356 pp.
236. Rosenberg, K.V., A.M. Dokter, P.J. Blancer, J.R. Sauer, A.C. Smith, P.A. Smith, J.C. Stanton, A. Panjabi, L. Helft, M.Parr, and P.P. Marra. 2019. Decline of the North American avifauna. Science 366(6461):120-124.
237. Jodice, P.G.R., L.C. Wickliffe, and E.B. Sachs. 2011. Seabird use of discards from a nearshore shrimp fishery in the South Atlantic Bight, USA. Marine Biology 158:2289-2298.
238. Wilkinson, P.M., S.A. Nesbitt, and J.F. Parnell. 1994. Recent history and status of the Eastern Brown Pelican. Wildlife Society Bulletin 22:420-430.
239. Seamans, M.E., J.P. Ludwig, K. Stromborg, F.E. Ludwig, and F.E. Ludwig. 2012. Annual survival of Double-Crested Cormorants from the Great Lakes. Waterbirds 35(sp1):23-30. https://doi.org/10.1675/063.035.sp104.
240. Dorr, B.S., and D.G. Fielder. 2017. Double-Crested Cormorants: Too much of a good thing? Fisheries 42:468-477. doi:10.1080/03632415.2017.1356121.
241. Frederick, P.C, D.E. Gawlik, J.C. Ogden, M.I. Cook, and M. Lusk. 2009. The White Ibis and Wood Stork as indicators for restoration of the everglades ecosystem. Ecological Indicators 9S:S83-S95.
242. Frederick, P.C., and J.C. Ogden. 1997. Philopatry and nomadism: Contrasting long-term movement behavior and population dynamics of White Ibises and Wood Storks. Colonial Waterbirds 20:316-323. doi:10.2307/1521699.
243. Murphy, T.M., and. J.W. Coker. 2008. A twenty-six year history of Wood Stork nesting in South Carolina. Waterbirds 31(Special Publication 1):3-7 doi:10.1675/1524-4695(2008)31[3:ATYHOW]2.0.CO;2.
244. USFWS. 2007. Wood Stork (*Mycteria Americana*) 5-Year Review: Summary and Evaluation. Unpublished report, U.S. Department of the Interior, Fish and Wildlife Service, Southeast Region Jacksonville Ecological Services Field Office, Jacksonville, Florida.
245. Elliott-Smith, E., and S. M. Haig. 2004. Piping Plover (*Charadrius melodus*), version 2.0. In The Birds of North America (A. F. Poole, Editor). Cornell Lab of Ornithology, Ithaca, NY, USA. doi.org/10.2173/bna.2.
246. Hecht, A., and S.M. Melvin. 2009. Population trends of Atlantic coast Piping Plovers, 1986-2006. Waterbirds 32:479-479.

247. Jodice, P.G.R., J.M. Thibault, S.A. Collins, M. Spinks, F.J. Sanders. 2014. Reproductive ecology of American Oystercatchers nesting on shell rakes. Condor 116:588-598.
248. Simon, T.R. 2017. The American Oystercatcher (*Haematopus palliates*) working group: 15 years of collaborative focal species research and management. Waterbirds 40(suppl.):1-9.
249. Maslo, B., K. Leu, C. Faillace, M.A. Weston, T. Pover, and T.A. Schlacher. 2016. Selecting umbrella species for conservation: A test of habitat models and niche overlap for beach-nesting birds. Biological Conservation 203:233-242. doi:10.1016/j.biocon.2016.09.012.
250. Correll, M.D., W.A. Wiest, B.J. Olsen, W.G. Shriver, C.S. Elphick, and T.P. Hodgman. 2016. Habitat specialization explains avian persistence in tidal marshes. Ecosphere 7:e01506 doi: 10.1002/ecs2.1506.
251. Franci, C.D., M. Guillemette, E. Pelletier, O. Chastel, S. Bonnefoi, and J. Verreault. 2014. Endocrine status of a migratory bird potentially exposed to the Deepwater Horizon oil spill: A case study of northern gannets breeding on Bonaventure Island, Eastern Canada. Science of the Total Environment 473:110-116. doi:10.1016/j.scitotenv.2013.12.006.
252. Garthe, S., G.T. Hallgrimsson, W.A. Montevecchi, D. Fifield, and R.W. Furness. 2016. East or west? Migration routes and wintering sites of Northern Gannets *Morus bassanus* from south-eastern Iceland. Marine Biology 163:151 doi:10.1007/s00227-016-2918-7.
253. Munson, M.A., K. Webb, D. Sheldon, D. Fink, W.M. Hochachka, M. Iliff, M. Riedewald, D. Sorokina, B. Sullivan, C. Wood and S. Kelling. 2014. The eBird Reference Dataset, Version 4.0. Cornell Lab of Ornithology and National Audubon Society, Ithaca, NY.
254. Sullivan B.L., J.L. Aycrigg, J.H. Barry, R.E. Bonney, N. Bruns, and 27 others. 2014. The eBird enterprise: An integrated approach to development and application of citizen science. Biological Conservation 169:31-40.
255. Wells, R.S., H.L. Rhinehart, L.J. Hansen, J.C. Sweeney, F.I. Townsend, R. Stone, D.R. Casper, M.D. Scott, A.A. Hohn, and T.K. Rowles. 2004. Bottlenose dolphins as marine ecosystem sentinels: Developing a health monitoring system. EcoHealth 1:246-254.
256. Moore, S.E. 2008. Marine mammals as ecosystem sentinels. Journal of Mammalogy 89:534-540.
257. NOAA Fisheries, "Marine Mammal Stock Assessments," [Online]. Available: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-stock-assessments [Accessed 02 September 2020].
258. Hohn, A.A., D.S. Rotstein, and B.L. Byrd. 2013. Unusual mortality events of harbor porpoise strandings in North Carolina, 1997-2009. Journal of Marine Biology 2013:289892. 13 pp. doi.org/10.1155/2013/289892.
259. Byrd, B.L., A.A. Hohn, F.H. Munden, G.N. Lovewell, and R.E. Lo Piccolo. 2008. Effects of commercial fishing regulations on stranding rates of bottlenose dolphin (*Tursiops truncates*). Fishery Bulletin, U.S. 106:72-81.

260. Byrd, B.L., A.A. Hohn, G.N. Lovewell, K.M. Altman, S.G. Barco, A. Friedlaender, C.A. Harms, W.A. McLellan, K.T. Moore, P.E. Rosel, and V.G. Thayer. 2014. Strandings as indicators of marine mammal biodiversity and human interactions off the coast of North Carolina. Fishery Bulletin, U.S. 112:1-23. doi: 10.7755/FB.112.1.1.
261. Morris, S.E., J.L. Zelner, D.A. Bauquier, T.K. Rowles, P.E. Rose, F. Gulland, and B.T. Grenfell. 2015. Partially observed epidemics in wildlife hosts: Modelling an outbreak of dolphin morbillivirus in the northwestern Atlantic, June 2013-2014. Journal of the Royal Society Interface 12: 20150676. doi.org/10.1098/rsif.2015.0676.
262. Peltier, H., W. Dabin, P. Daniel, O. Van Canneyt, G. Dorémus, M. Huon, and V. Ridoux. 2012. The significance of stranding data as indicators of cetacean populations at sea: Modelling the drift of cetacean carcasses. Ecological Indicators 18:278–290.
263. NOAA Fisheries, "National Stranding Database Public Access," [Online]. Available: https://www.fisheries.noaa.gov/national/marine-life-distress/national-stranding-database-public-access. [Accessed 29 February 2019].
264. NOAA Fisheries, "National Stranding Database Public Access," [Online]. Available: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-unusual-mortality-events. [Accessed 29 February 2019].
265. Wallace, B.P., A.D. DiMatteo, B.J. Hurley, E.M. Finkbeiner, A.B. Bolten, and 28 others. 2010. Regional management units for marine turtles: A novel framework for prioritizing conservation and research across multiple scales. PloSOne 5(12):e15465. doi.org/10.1371/journal.pone.0015465
266. Pike, D.A. 2013. Forecasting range expansion into ecological traps: Climate-mediated shifts in sea turtle nesting beaches and human development. Global Change Biology 19:3082-3092. doi: 10.1111/gcb.12282.
267. Finn, S.A., W.P. Thompson, B.M. Shamblin, C.J. Nairn, and M.H. Godfrey. 2016. Northernmost records of Hawksbill Sea Turtle nests and possible trans-Atlantic colonization event. Marine Turtle Newsletter 151:27-29.
268. Witherington, B., P. Kubilis, B. Brost, and A. Meylan. 2009. Decreasing annual nest counts in a globally important loggerhead sea turtle population. Ecological Applications 19:30-54.
269. Richards, P.M., S.P. Epperly, S.S. Heppell, R.T. King, C.R. Sasso, F. Moncada, G. Nodarse, D.J. Shaver, Y. Medina, and J. Zurita. 2011. Sea turtle population estimates incorporating uncertainty: A new approach applied to western North Atlantic loggerheads *Caretta caretta*. Endangered Species Research 15:151-158.
270. Arendt, M.D., A.L. Segars, J.I. Byrd, J. Boynton, J.D. Whitaker, L. Parker, D.W. Owens, G. Blanvillain, J.M. Quattro, and M.A. Roberts. 2012. Seasonal distribution patterns of juvenile loggerhead sea turtles (*Caretta caretta*) following capture from a shipping channel in the Northwest Atlantic Ocean. Marine Biology 159:127-139.

271. Ceriani, S.A., J.D. Roth, D.R. Evans, J.F. Weishampel, and L.M. Ehrhart. 2012. Inferring foraging areas of nesting Loggerhead turtles using satellite telemetry and stable isotopes. PlosOne 7(9):e45335. doi.org/10.1371/journal.pone.0045335.
272. Pajuelo, M., K.A. Bjorndal, K.J. Reich, H.B. Vander Zanden, L.A. Hawkes, and A.B. Bolten. 2012. Assignment of nesting loggerhead turtles to their foraging areas in the Northwest Atlantic using stable isotopes. Ecosphere 3:89. doi.org/10.1890/ES12-00220.1.
273. U.S. Census Bureau. https://data.census.gov/cedsci/. [Accessed 04 August 2020].
274. Mormino, G.R. 2002. Sunbelt dreams and altered states: A cultural history of Florida, 1950-2000. The Florida Historical Quarterly 81(1):3-21. http://www.jstor.org/stable/30147612
275. NOAA Office for Coastal Management. DigitalCoast. https://coast.noaa.gov/digitalcoast/data/. [Accessed 7 March 2019].
276. Gough, I. 2004. Human well-being and social structures: Relating the universal and the local. Global Social Policy 4:289-311.
277. Burkhard, B., F. Kroll, F. Müller, and W. Windhorst. 2009. Landscapes' capacities to provide ecosystem services – a concept for land-cover based assessments. Landscape Online 15:1-22. doi:10.3097/LO.200915.
278. Dillard, M.K., T.L. Goedeke, S. Lovelace, and A.Orthmeyer. 2013. Monitoring well-being and changing environmental conditions in coastal communities: Development of an assessment method. NOAA Technical Memorandum NOS NCCOS 174. Silver Spring, MD. 176 pp.
279. Jepson, M., and L. Colburn. 2013. Development of social indicators of fishing community vulnerability and resilience in the U.S. Southeast and Northeast regions. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-F/SPO-129.
280. Pauly, D. 1995. Anecdotes and the shifting baseline syndrome of fisheries. Trends in Ecology and Evolution 10(10):430. doi: 10.1016/s0169-5347(00)89171-5.

15. APPENDIX

Table A1. Abbreviation, name, and figure number for indicators in the synthesis analysis.

Abbreviation	Name	Report Section	Figure Number
ACE	Accumulated Cyclone Energy	Physical and Chemical Pressures	Fig. 4.12a
Age at Mat	Age at Maturity	Upper Trophic Level States	Fig. 7.6, top left
AIR CO2	Air pCO_2	Physical and Chemical Pressures	Fig. 4.13 top
AMO	Atlantic Multidecadal Oscillation	Climate Drivers	Fig. 3.1b
Atl Warm Pool	Atlantic Warm Pool area June-Nov	Climate Drivers	Fig. 3.5b
Blk SeaBass BB0	Black Sea Bass B/B0	Ecosystem Services	Fig. 8.1, top left
Blk SeaBass Recrt	Black Sea Bass Recruitment	Ecosystem Services	Fig. 8.2, top left
Blue Tile BB0	Blueline Tilefish B/B0	Ecosystem Services	Fig. 8.1, top center
Brown Pelican	Brown Pelican Abundance	Ecosystem Services	Fig. 8.7, top left
Cetacean Strand	Cetacean and Pinniped Strandings	Ecosystem Services	Fig. 8.8
Chl a	Surface Chloraphyll-a	Lower Trophic Level States	Fig. 6.1b
Clapper Rail	Clapper Rail Abundance	Ecosystem Services	Fig. 8.7, bottom left
Cobia BB0	Cobia B/B0	Ecosystem Services	Fig. 8.1, top right
Cobia Recrt	Cobia Recruitment	Ecosystem Services	Fig. 8.2, top center
Com Engage	Commercial Fishing Engagement	Human Dimensions	Fig. 9.5, top left

Abbreviation	Name	Report Section	Figure Number
Com Landings	Commercial Fishing Landings	Ecosystem Services	Fig. 8.3a
Com Reliance	Commercial Fishing Reliance	Human Dimensions	Fig. 9.5, top right
Com Revenue	Commercial Fishing Revenues	Ecosystem Services	Fig. 8.3b
Coral 4DHW	Days Above 4 Degree Heat Weeks	Habitat State	Fig. 5.5, top
Cormorant	Double-Crested Cormorant Abundance	Ecosystem Services	Fig. 8.7, top right
Demersal Abund	Nearshore Demersal Fish Abundance	Upper Trophic Level States	Fig. 7.1, bottom
Demersal Rich	Nearshore Demersal Fish Richness	Upper Trophic Level States	Fig. 7.1, top
DryTort Cor Dens	Dry Tortugas Coral Density	Habitat State	Fig. 5.4, left
DryTort Cor Dis	Dry Tortugas Coral Disease Prevalence	Habitat State	Fig. 5.4, center
DryTort Cor Mort	Dry Tortugas Coral Mortality	Habitat State	Fig. 5.4 right
DryTort Fish Dens	Dry Tortugas Reef Fish Density	Upper Trophic Level States	Fig. 7.4, bottom right
DryTort Fish Rich	Dry Tortugas Reef Fish Richness	Upper Trophic Level States	Fig. 7.4, bottom left
Employ Oc Econ	Ocean-Related Employment	Human Dimensions	Fig. 9.3, top
ENSO	El Niño Southern Oscillation	Climate Drivers	Fig. 3.3b

Abbreviation	Name	Report Section	Figure Number
FL Current	Florida Current	Physical and Chemical Pressures	Fig. 4.4
FL Log Nest	Florida Loggerhead Sea Turtle Nest Count	Ecosystem Services	Fig. 8.9, top
FL Pop	Total Florida Human Population	Human Dimensions	Fig. 9.1, top left
FL Stream Flow	Florida Relative Stream Flow	Physical and Chemical Pressures	Fig. 4.8, bottom right
FLKeys Cor Dens	Florida Keys Coral Density	Habitat State	Fig. 5.4, left
FLKeys Cor Dis	Florida Keys Coral Disease Prevalence	Habitat State	Fig. 5.4, center
FLKeys Cor Mort	Florida Keys Coral Mortality	Habitat State	Fig. 5.4 right
FLKeys Fish Dens	Florida Keys Reef Fish Density	Upper Trophic Level States	Fig. 7.4, middle right
FLKeys Fish Rich	Florida Keys Reef Fish Richness	Upper Trophic Level States	Fig. 7.4, middle left
GA Log Nest	Georgia Loggerhead Sea Turtle Nest Count	Ecosystem Services	Fig. 8.9, mid upper
GA Pop	Total Georgia Human Population	Human Dimensions	Fig. 9.1, top right
GA Stream Flow	Georgia Relative Stream Flow	Physical and Chemical Pressures	Fig. 4.8, bottom left
Gag BB0	Gag B/B0	Ecosystem Services	Fig. 8.1, mid upper, center
Gag Recrt	Gag Recruitment	Ecosystem Services	Fig. 8.2, mid upper, left

Abbreviation	Name	Report Section	Figure Number
Golden Tile BB0	Golden Tilefish B/B0	Ecosystem Services	Fig. 8.1, mid upper, right
Golden Tile Recrt	Golden Tilefish Recruitment	Ecosystem Services	Fig. 8.2, mid upper, center
Gr Amjack BB0	Greater Amberjack B/B0	Ecosystem Services	Fig. 8.1, mid upper, left
Gr Amjack Recrt	Greater Amberjack Recruitment	Ecosystem Services	Fig. 8.2, top right
Gulf Stream Pos	Gulf Stream Position	Physical and Chemical Pressures	Fig. 4.5b
Hardbott Abund	Hard-Bottom Reef Fish Abundance	Upper Trophic Level States	Fig. 7.2, top right
Hardbott Rich	Hard-Bottom Reef Fish Richness	Upper Trophic Level States	Fig. 7.2, top left
Hurricanes	Number of Landfalling Hurricanes	Physical and Chemical Pressures	Fig. 4.12b
L infinity	Maximum Size ($L_{infinity}$)from Growth Curve	Upper Trophic Level States	Fig. 7.6, bottom left
Length at Mat	Length at Maturity	Upper Trophic Level States	Fig. 7.6, top right
Max Age	Maximum Age	Upper Trophic Level States	Fig. 7.6, bottom right
Median Zooplank	Median Zooplankton Biovolume	Lower Trophic Level States	Fig. 6.2b
Menhaden Abund	Atlantic Menhaden Abundance	Lower Trophic Level States	Fig. 6.4
MTL Comm	Commercial Fishing Mean Trophic Level	Upper Trophic Level States	Fig. 7.5a, top

Abbreviation	Name	Report Section	Figure Number
MTL Rec	Recreational Fishing Mean Trophic Level	Upper Trophic Level States	Fig. 7.5a, bottom
MTL SEAMAP	SEAMAP Demersal Species Mean Trophic Level	Upper Trophic Level States	Fig. 7.5b, bottom
MTL SERFS	SERFS Demersal Species Mean Trophic Level	Upper Trophic Level States	Fig. 7.5b, top
NAO	North Atlantic Oscillation	Climate Drivers	Fig. 3.2b
NC Ichthy Abund	North Carolina Ichthyoplankton Abundance	Lower Trophic Level States	Fig. 6.3, top right
NC Ichthy Rich	North Carolina Ichthyoplankton Richness	Lower Trophic Level States	Fig. 6.3, top left
NC Log Nest	North Carolina Loggerhead Sea Turtle Nest Count	Ecosystem Services	Fig. 8.9, bottom
NC Pop	Total North Carolina Human Population	Human Dimensions	Fig. 9.1, bottom right
NC Stream Flow	North Carolina Relative Stream Flow	Physical and Chemical Pressures	Fig. 4.8, top left
North Gannet	Northern Gannet Abundance	Ecosystem Services	Fig. 8.7, bottom right
Ocean GDP	Ocean Related Gross Domestic Product	Human Dimensions	Fig. 9.3, bottom
Overfished	Number of Overfished Stocks	Ecosystem Services	Fig. 8.6, top right

Abbreviation	Name	Report Section	Figure Number
Overfishing	Number of Stocks Undergoing Overfishing	Ecosystem Services	Fig. 8.6, top left
Oystercatcher	American Oystercatcher Abundance	Ecosystem Services	Fig. 8.7, mid lower, right
Piping Plover	Piping Plover Abundance	Ecosystem Services	Fig. 8.7, mid lower, left
Precip	Total Annual Precipitation	Physical and Chemical Pressures	Fig. 4.10b
Rec Effort	Recreational Fishing Effort	Ecosystem Services	Fig. 8.4a, middle
Rec Engage	Recreational Fishing Engagement	Human Dimensions	Fig. 9.5, bottom left
Rec Landings	Recreational Fishing Landings	Ecosystems Services	Fig. 8.4a, top
Rec Reliance	Recreational Fishing Reliance	Human Dimensions	Fig. 9.5, bottom right
Red Group BB0	Red Grouper B/B0	Ecosystem Services	Fig. 8.1, mid lower, left
Red Group Recrt	Red Grouper Recruitment	Ecosystem Services	Fig. 8.2, mid upper, right
Red Porgy BB0	Red Porgy B/B0	Ecosystem Services	Fig. 8.1, mid lower, center
Red Porgy Recrt	Red Porgy Recruitment	Ecosystem Services	Fig. 8.2, lower mid, right
Red Snap BB0	Red Snapper B/B0	Ecosystem Services	Fig. 8.1, lower mid, right
Red Snap Recrt	Red Snapper Recruitment	Ecosystem Services	Fig. 8.2, lower mid, center

Abbreviation	Name	Report Section	Figure Number
SC Ichthy Abund	South Carolina Ichthyoplankton Abundance	Lower Trophic Level States	Fig. 6.3, middle right
SC Ichthy Rich	South Carolina Ichthyoplankton Richness	Lower Trophic Level States	Fig. 6.3, middle left
SC Log Nest	South Carolina Loggerhead Sea Turtle Nest Count	Ecosystem Services	Fig. 8.9, mid lower
SC Pop	Total South Carolina Human Population	Human Dimensions	Fig. 9.1, bottom left
SC Stream Flow	South Carolina Relative Stream Flow	Physical and Chemical Pressures	Fig. 4.8, top right
Sea Bottom Temp	Sea Bottom Temperature	Physical and Chemical Pressures	Fig. 4.2
SEA CO2	Seawater pCO_2	Physical and Chemical Pressures	Fig. 4.13 top
Sea Level Rise	Sea Level Rise	Physical and Chemical Pressures	Fig. 4.11, right
Sea Surface Temp	Sea Surface Temperature	Physical and Chemical Pressures	Fig. 4.1, bottom
SEFL Cor Dens	Southeast Florida Coral Density	Habitat State	Fig. 5.4, left
SEFL Cor Dis	Southeast Florida Coral Disease Prevalence	Habitat State	Fig. 5.4, center
SEFL Cor Mort	Southeast Florida Coral Mortality	Habitat State	Fig. 5.4 right
SEFL Fish Dens	Southeast Florida Reef Fish Density	Upper Trophic Level States	Fig. 7.4, top right

Abbreviation	Name	Report Section	Figure Number
SEFL Fish Rich	Southeast Florida Reef Fish Richness	Upper Trophic Level States	Fig. 7.4, top left
Shark CPUE	Coastal Sharks Relative Abundance	Upper Trophic Level States	Fig. 7.3, bottom
Shark Rich	Coastal Sharks Species Richness	Upper Trophic Level States	Fig. 7.3, top
Snow Group BB0	Snowy Grouper B/B0	Ecosystem Services	Fig. 8.1, bottom left
Snow Group Recrt	Snowy Grouper Recruitment	Ecosystem Services	Fig. 8.2, lower mid, right
Social Connect	Social Connectedness	Human Dimensions	Fig. 9.4
Sp Mack BB0	Spanish Mackerel B/B0	Ecosystem Services	Fig. 8.1, bottom center
Sp Mack Recrt	Spanish Mackerel Recruitment	Ecosystem Services	Fig. 8.2, bottom left
TRIPOLE	North Atlantic Sea Surface Temperature Tripole	Climate Drivers	Fig. 3.4b
Upwelling	Upwelling	Physical and Chemical Drivers	Fig. 4.6
Verm Snap BB0	Vermilion Snapper B/B0	Ecosystem Services	Fig. 8.1, bottom right
Verm Snap Recrt	Vermilion Snapper Recruitment	Ecosystem Services	Fig. 8.2, bottom center
White Ibis	White Ibis Abundance	Ecosystem Services	Fig. 8.7, mid upper, left
Wood Stork	Wood Stork Abundance	Ecosystem Services	Fig. 8.7, mid upper, right