Chapter 4
Big Bend and Springs Coast

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Description of the region

The northern Gulf coast of peninsular Florida is commonly divided into two segments: the northern segment, from Wakulla to Levy County, is referred to as the Big Bend, while Citrus to Pasco County is called the Springs Coast due to the abundance of natural freshwater springs (Fig. 4.1). Also referred to as the Nature Coast, the region has limited urban development, low coastal pollution, and a lower population density than other coasts in Florida, in part due to the lack of extensive beaches (Livingston 1990). Large portions of the coast and nearshore waters are protected in a network of national wildlife refuges, state parks, and aquatic preserves including the Big Bend Seagrasses Aquatic Preserve. The region is also divided among three water management districts: Northwest Florida, Suwannee River, and Southwest Florida (NWFWMD, SRWMD, and SWFWMD, respectively; Fig. 4.1).

The coastal waters of the Big Bend and Springs Coast are characterized by gentle topography, little wave energy, low sediment supply, and limestone bedrock that lies at or near the land surface (Wolfe 1990, FDEP 2015). These conditions provide ideal habitat for extensive salt marshes and seagrass beds, which are interspersed with tidal flats and oyster reefs. Although the nearshore areas of the Big Bend and Springs Coast are not enclosed by barrier islands, coastal waters up to several kilometers offshore typically have lower salinity than ocean water throughout the year due to freshwater inputs from spring-fed rivers and submarine groundwater discharge (Orlando et al. 1993, FDEP 2015). Oysters in this region are more frequently found in intertidal areas with greater freshwater input, such as at the mouths of rivers and tidal creeks, as the lower salinity reduces the abundance of marine predators (Hine et al. 1988, Seavey et al. 2011). Oyster reefs in these low-salinity nearshore areas often have higher percent cover and population density than high-salinity offshore reefs (Bergquist et al. 2006). The eastern oyster (*Crassostrea virginica*) is the most common oyster species; the crested oyster (*Ostrea stentina*) is also found, but is much less common (Wolfe 1990).

Offshore reefs often form in linear patterns roughly parallel to shore, following paleoshorelines (Wright et al. 2005). Linear reef chains can act as semipermeable dams that slow the flow of fresh water from rivers and decrease salinity inshore of the reef, facilitating the development of inshore oyster reefs (Wright et al. 2005, Frederick et al. 2015, Kaplan et al. 2016). Oyster reefs can also lead to the development of a marsh island with smooth cordgrass (*Spartina alterniflora*; Wolfe 1990). Marsh islands are generally located closer to shore and have higher elevations than sand-oyster reefs, which are made up of coarse sand with shell fragments interspersed with oyster clumps (Seavey et al. 2011).

Apalachee Bay

Apalachee Bay (Fig. 4.2), located in the northern Big Bend, receives freshwater inflow from the Ochlockonee, Wakulla, St. Marks, Aucilla, Econfina, and Fenholloway rivers along with numerous other small creeks. Apalachee Bay is a broad, shallow embayment with an average depth

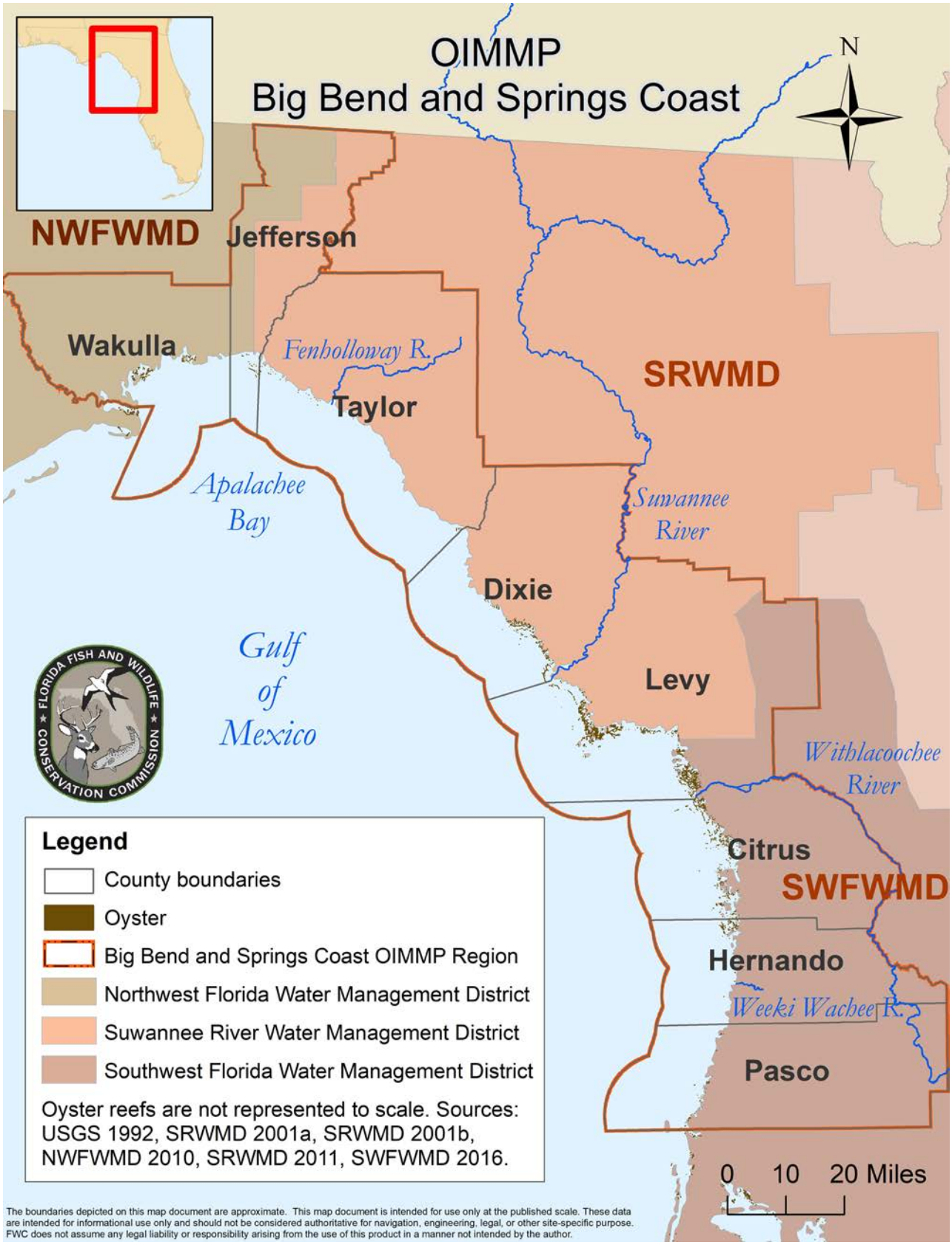

Figure 4.1. Oyster extent in the Big Bend and Springs Coast region.

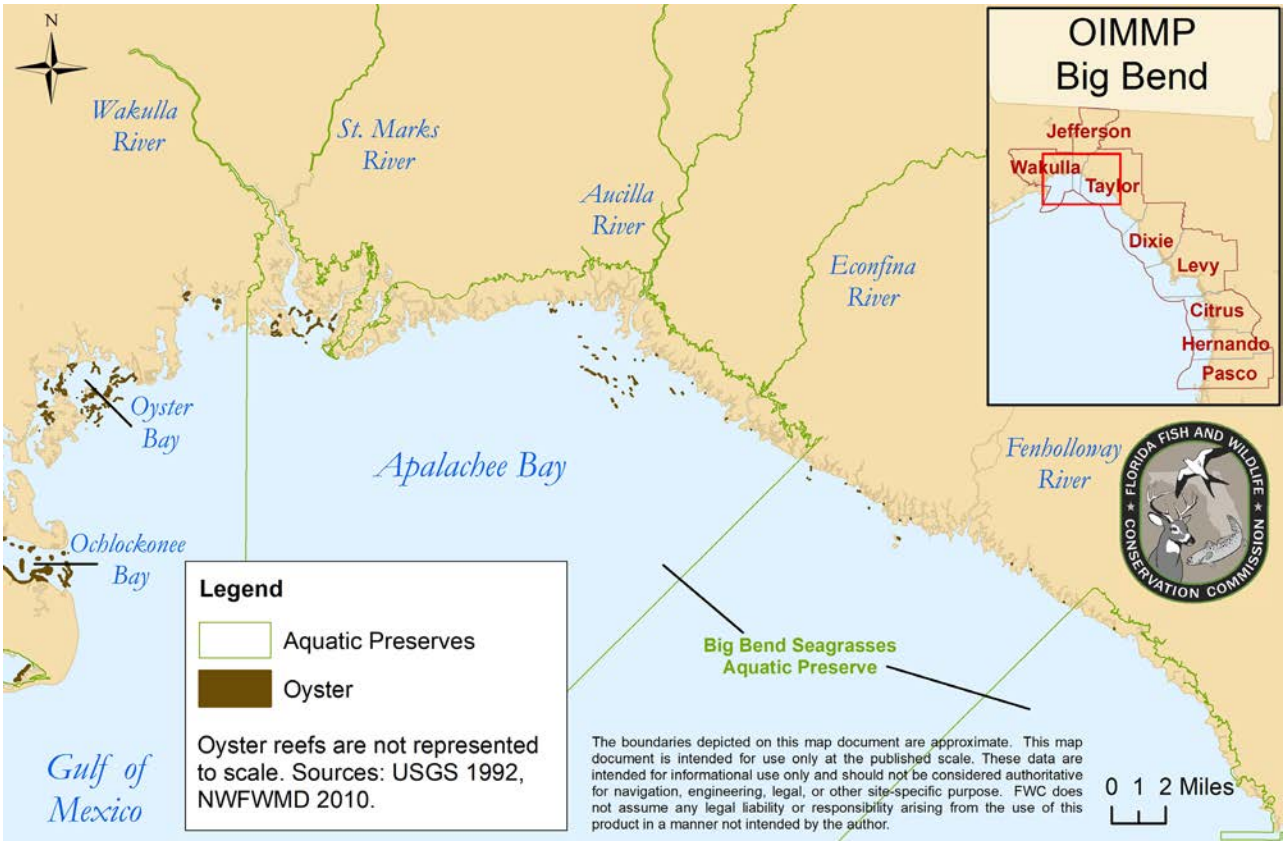

Figure 4.2. Oyster extent in the Big Bend of Florida. Oyster mapping sources: USGS 1992 (made from 1992–1993 aerial photographs) and NWFWMD 2010 (made from 2009–2010 photographs).

ranging from 1.5 to 3 m (5–10 ft) and an average salinity of 30 (FDEP 2015). A zone with moderate salinity (15–25) extends about 8 km (5 mi) offshore during months of high freshwater flow (Nelson 2015). Tidal range in the bay is 0.75 m (2.5 ft). Oyster reefs are found in the moderate- and low-salinity (5–15) areas near the mouths of creeks and rivers (Fig. 4.2; Nelson 2017). A large portion of Apalachee Bay is included within the Big Bend Seagrasses Aquatic Preserve, the boundaries of which extend up the coastline and navigable tributaries until the tidal mean high-water line (Fig. 4.2; FDEP 2015). The coastline is largely undeveloped, and the river systems, salt marshes, seagrass beds, and oyster reefs have relatively little human impact compared to other regions of Florida (FDEP 2015, SRWMD 2017). An exception is the Fenholloway River, located outside the Big Bend Seagrasses Aquatic Preserve, which was once classified by the Florida Department of Environmental Protection (FDEP) as a Class V water body (designated for industrial use) as it received point-source pollutants from the Buckeye Foley pulp mill, mining companies, and the city of Perry's wastewater treatment plant (FWC 2004). Water quality in the river has improved in recent decades, and the river was upgraded to a Class III water body (designated for fish consumption, recreation, and maintenance of fish and wildlife) in 1998. Nevertheless, water quality concerns and the need for environmental monitoring remain (FDEP 2012, 2014). Oysters are sparsely found at the mouth of the Fenholloway River; current mapping may underestimate extent (Fig. 4.2).

Deadman Bay

Farther south, the Steinhatchee River and several of its tidal tributaries discharge into Deadman Bay, a broad embayment with an average depth of 2.2 m (7.3 ft) (Fig. 4.3; FDEP 2015). Surface and subsurface freshwater discharge flows into Deadman Bay, creating an average salinity of 26 within the bay. Limited oyster reefs are found near the mouth of the Steinhatchee River (Fig. 4.3).

Suwannee Sound

The Suwannee River begins in Georgia and provides the largest source of fresh water to the Big Bend (FDEP 2015). Suwannee Sound, which extends roughly from Horseshoe Point to Cedar Key, has an average salinity of 16 (and a range of 1 to 26) due to freshwater input from the Suwannee River (Figs. 4.3 and 4.4; FDEP 2015,

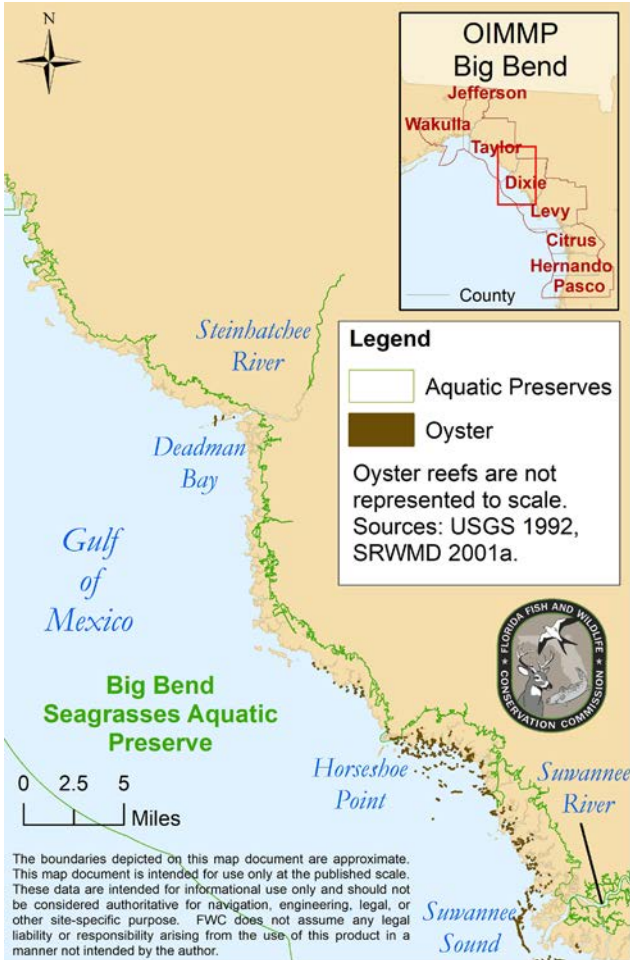

Figure 4.3. Oyster extent in Taylor and Dixie counties. Oyster mapping sources: USGS 1992 (made from 1992–1993 aerial photographs) and SRWMD 2001a (from 2001 photographs).

Frederick et al. 2015). Moderate-salinity (15–25) waters extend about 8 km (5 mi) offshore (Nelson 2015). Suwannee Sound contains oyster reefs scattered among salt marsh islands and historically hosted extensive linear offshore oyster reefs (Fig. 4.5). Evidence from the past 150 years suggests that many offshore, inshore, and nearshore oyster reefs migrated landward, with the exception of those offshore reefs that were largely lost (Fig. 4.5; Seavey et al. 2011). An aerial-photograph assessment of oyster habitat trends in the Suwannee Sound region revealed a 66% net loss of oyster reef area from 1982 to 2011 (Seavey et al. 2011). The loss was greatest in high-salinity offshore reefs (88% areal loss) and decreased in nearshore (61% loss) and inshore reefs (50% loss). Restoration efforts have shown that the physical reef structure can cause waters on the landward side of the reefs to have salinity that is an average of 3 (or maximum of 10) lower than waters on the seaward side of the reef (Frederick et al. 2015). Because offshore barrier reefs help to protect inshore reefs from wave action and create lower salinity conditions, the collapse of offshore oyster reefs leads to further oyster collapse inshore (Frederick et al. 2015, Kaplan et al. 2016).

Oyster collapse resulted in the erosion (~7 cm/2.8 in elevation loss per year) of many high-density, high-relief oyster reefs and conversion to low-relief tidal flats dominated by sand and scattered shells (Seavey et al. 2011). In some cases, oyster collapse led to an apparent increase in reef area as eroded shell spread across a wider expanse of substrate. This apparent expansion was temporary, as oyster spat were not able to successfully settle and establish on these collapsed oyster reefs due to limited substrate availability and stability. Following declines in oyster acreage in the 1980s and 1990s, oyster reef collapse was reported after 2000. The cause for the decline has been attributed to decreased freshwater discharge from the Suwannee River, which is linked with human usage and rainfall. Total discharge from the Suwannee River has declined relative to the amount of rain falling in the watershed, and months with low discharge have increased in frequency (Seavey et al. 2011). Seavey et al. (2011) suggest that mechanisms for the loss of oyster reefs include some combination of increased occurrence of disease, predation, and parasitism driven by increased salinity, as has also been reported in the Chesapeake Bay (White and Wilson 1996). Restoration actions that include the addition of substrates resistant to erosion may mitigate reef collapse (Frederick et al. 2015).

Springs Coast

The Springs Coast (Figs. 4.4 and 4.6) receives freshwater outflow from groundwater seeps and spring-fed rivers including the Crystal, Homosassa, Chassahowitzka, Weeki Wachee, and Pithlachascotee rivers (Hine et al. 1988). Oyster reefs are found mostly inshore among marsh islands, but occasionally offshore as linear reefs that fringe the shoreline. Mid-19th-century surveys of the Springs Coast indicate that offshore reefs were more abundant at the mouths of these spring-fed rivers than today (Fig. 4.7; Bache 1861, Raabe et al. 2004). The Crystal, Chassahowitzka, and Weeki Wachee rivers also had oyster reefs in the 1800s, but offshore reefs remain only at the Crystal River. At other rivers, remnants of these reefs are evident in the numerous shoals dominated by sand and scattered shell that extend across river mouths, many of which are no longer exposed at low tide (Hesterberg, pers. obs.). Spatial analysis of the historical distribution of oyster reef locations near the Crystal River compared with the best available mod-

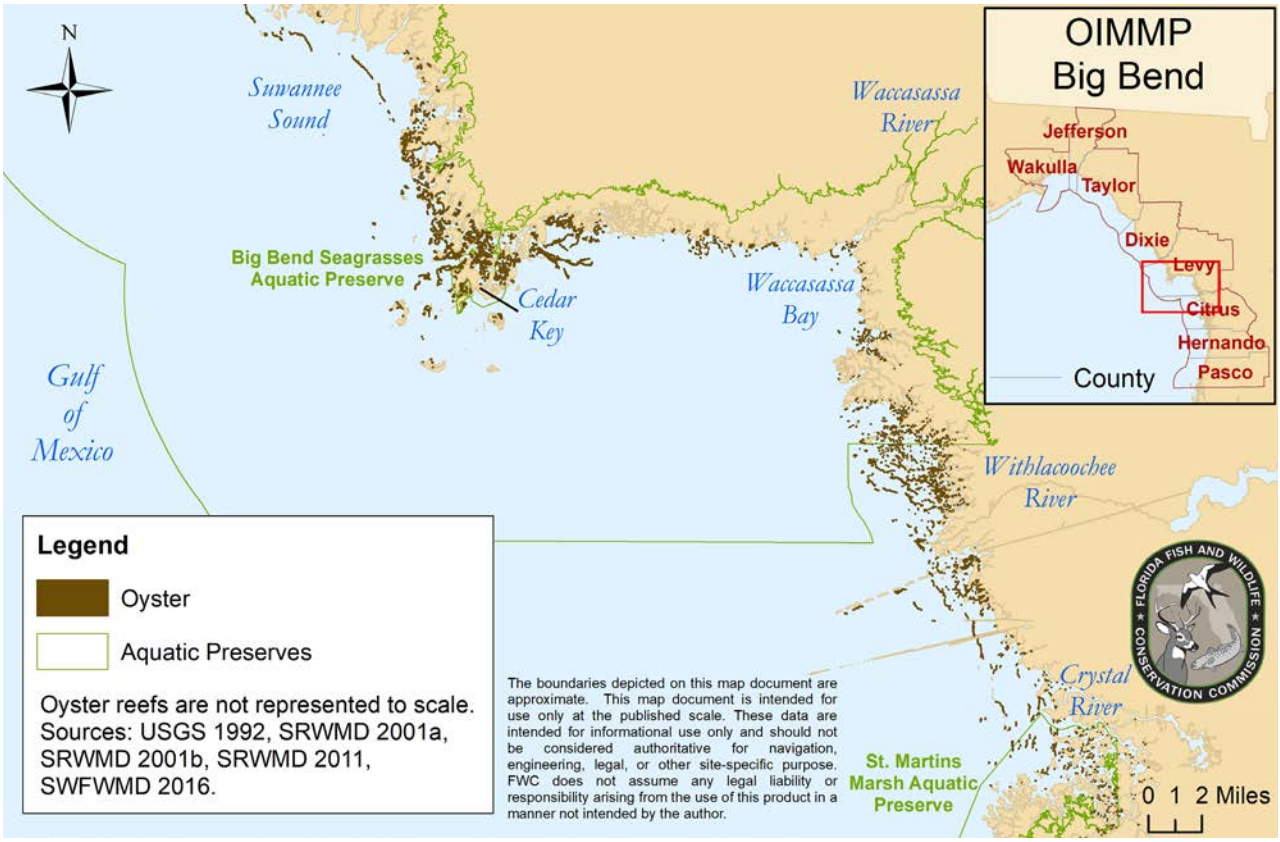

Figure 4.4. Oyster extent in Levy and Citrus counties. Oyster mapping sources: USGS 1992 (made from 1992–1993 aerial photographs), SRWMD 2001a and 2001b (from 2001 photographs), SRWMD 2011 (from 2010–2011 photographs), and SWFWMD 2016 (from 2016 photographs).

ern maps indicates a net landward movement of oyster habitat over the past 150 years (Fig. 4.7). These observations corroborate the findings of Seavey et al. (2011), who attributed shifts in reef distribution in the Suwannee area to decreasing freshwater input. The spatial loss of offshore oyster reefs near the Crystal River mirrors patterns of decline observed elsewhere in the Big Bend and corresponds with decreasing freshwater discharge from the Crystal River and increasing periods of higher salinity (SWFWMD 2015).

Excavations of prehistoric (1,500–1,000 ybp) shell middens from well-dated archaeological sites near the Crystal River suggests a dramatic decline in shell height of individual oysters over the last two millennia (Hesterberg et al., in review). Mean oyster size in the modern population is one third that of the archaeological baseline, and maximum body size has been sharply truncated at approximately 120 mm (4.7 in). These results are in stark contrast to frequent observations in middens of shells greater than 140 mm (5.5 in). Large oysters disproportionately contribute to increased water filtration, reproductive output, and shell budgets compared to smaller individuals (Riisgård 1988, Powell and Klinck 2007, Mroch et al. 2011, Mann et al. 2014). Thus, shifts in oyster size structure suggest a reduction in ecosystem services for reefs that have existed for thousands of years (Grinnell 1971, Wright et al. 2005).

Despite losses in both habitat extent and ecosystem services, the Crystal River still possesses the highest density of oyster reefs along the Springs Coast (Figs. 4.4 and 4.7). The clustering of oyster reefs in this area partially results from the large spring-fed discharge of the Crystal River. Like other river systems in the Big Bend, the substantial quantity of freshwater that empties into Crystal Bay creates lower-salinity conditions that extend for several kilometers offshore. The lower nearshore salinity, paired with a strong tidal exchange and often exposed limestone bedrock, produce favorable conditions for intertidal oyster reef formation (Hine et al. 1988). Although the Crystal River faces ongoing water quality challenges (SWFWMD 2015), this system might offer the best opportunity to protect and restore oyster reefs along the Springs Coast, given the amount of remaining habitat and its favorable hydrology.

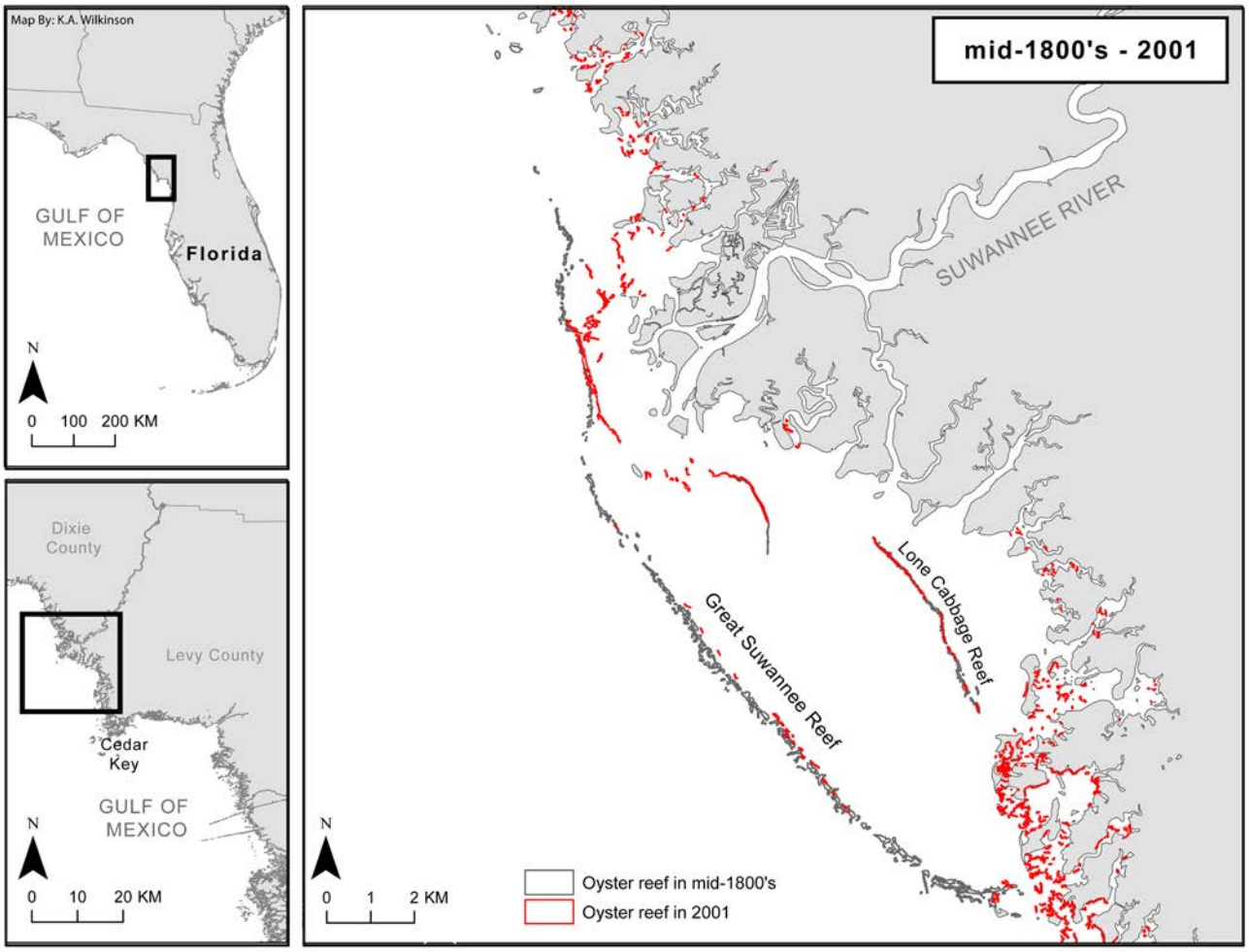

Figure 4.5. Oyster reef extent in Suwannee Sound in the mid-1800s (gray) and 2001 (red). Map by Krystan A. Wilkinson. Data sources: Raabe et al. 2004 (made from 19th century topographic sheets), SRWMD 2001a (made from 2001 photographs). Note that the increase in oyster extent along the coast in 2001 may reflect an increase in mapping effort in these areas.

Oyster harvesting

Numerous shell middens along the Big Bend and Springs Coast indicate that oysters have been harvested in this region for at least 1,400 years (Dean et al. 2004, Sassaman et al. 2013). Commercial harvesting began in the Big Bend in the late 1800s, but an intensive industry failed to materialize in the 20th century despite the presence of productive beds (Dawson 1955, Arnold and Berrigan 2002). The modern commercial oyster fishery in Florida has been dominated by landings from the Apalachicola Bay region in Franklin County, but the Big Bend area also has a history of commercial harvest. Annual yields in the Big Bend and Springs Coast varied throughout the 1900s and peaked in the 1980s before declining significantly in the 1990s (Fig. 4.8, Seavey et al. 2011). This decline can be attributed largely to the Florida Department of Agriculture and Consumer Services (FDACS) beginning to close shellfish harvesting areas in 1987 due to the presence of fecal coliform bacteria as well as a shift in emphasis in 1991 from wild oyster harvesting to hard-clam aquaculture (Colson and Sturmer 2000, Seavey et al. 2011). The Big Bend and Springs Coast region contains several shellfish harvesting areas, located in Wakulla, Dixie, Levy, and Citrus counties, that are monitored by FDACS (Fig. 4.9; FDACS 2017). Several of the areas with productive oyster reefs in the Big Bend are often closed to commercial harvest due to water quality concerns (FDEP 2015). Other factors have been proposed to explain the industry's limited development in the region, including periodic mortality events, habitat loss, pollution, and market pressures (Dawson 1955, Colson and Sturmer 2000, Arnold and Berrigan 2002). However, the relative contribution of each factor remains unclear and likely varies locally. Since the collapse of the Apalachicola oyster fishery in 2012, there has been increased focus on the Big Bend fishery. Although current oyster harvests in the Big Bend are far

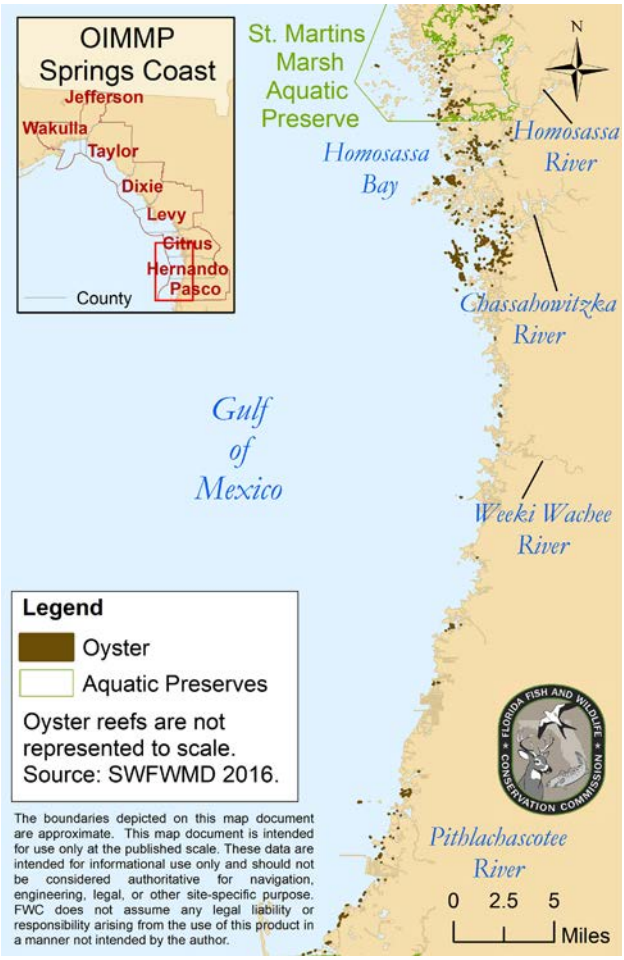

Figure 4.6. Oyster extent in Citrus, Hernando, and Pasco counties. Oyster mapping source: SWFWMD 2016 (made from 2016 aerial photographs).

smaller than the peak yields of the 1980s (Fig. 4.8), harvests are increasing again. In 2016, commercial oyster landings for both the Big Bend and the previously dominant Franklin County were both 170 metric tons (375,000 pounds; FWC 2018). In 2017, commercial oyster landings for the Big Bend increased to 219 metric tons (483,000 pounds), surpassing the Franklin County yield of 122 metric tons (268,000 pounds; FWC 2018). The role of harvest in structuring the distribution and viability of oyster reefs in the Big Bend is unclear, primarily because of uncertainty regarding the effects of fishing to the predominantly intertidal reefs along the Big Bend compared to the subtidal oyster reefs of Apalachicola.

Threats to oysters in the Big Bend and Springs Coast

- **Altered hydrology and salinity:** The primary threat to oyster reefs in the Big Bend and Springs Coast region is decreased freshwater input, which results in higher salinity (Geselbracht 2007). Many aspects of oyster biology are strongly influenced by local salinity patterns, including abundance, growth, mortality, recruitment, and reproduction (Bergquist et al. 2006, La Peyre et al. 2013, Miller et al. 2017). Evidence of the effects of increased salinity on oyster reefs can be seen in the preferential loss of offshore reefs and the net landward migration of extant habitat (Seavey et al. 2011; Figs. 4.5 and 4.7). It is unclear to what extent diminished freshwater input results from direct human use versus long-term climatic drivers. For example, the Atlantic Multidecadal Oscillation has been linked with declining rainfall on the Big Bend and Springs Coast since 1970, but human population and demand for freshwater have also increased during this time (Kelly and Gore 2008, SWFWMD 2015).
- **Thermal stress:** Extended periods of elevated water temperature can cause oyster mortality through oxygen limitation (Forster et al. 2012, Waples and Audzijonyte 2016) and disease intensification (Petes et al. 2012). Temperatures above 28 °C (82 °F) can cause stress and mortality in eastern oysters, especially when combined with other stressors (La Peyre et al. 2016, Rybovich et al. 2016, Southworth et al. 2017). High temperatures disproportionately affect large individuals as oxygen diffusivity and disease intensity decrease and increase with body size, respectively (Forster et al. 2012, Waples and Audzijonyte 2016). Thus, populations can be shifted towards a smaller body size over time as thermal stress causes higher mortality in larger oysters (Daufresne et al. 2009, Sheridan and Bickford 2011, Waples and Audzijonyte 2016). Such shifts have been observed on oyster reefs affected by thermal effluent near Crystal River; thermally affected reefs had size structures similar to those of unaffected reefs but lacked the largest oyster size class (Lehman 1974).
- **Sea-level rise:** Sea level rose at a rate of 2.02 mm/yr (0.80 in/decade) from 1914 to 2016 in the Big Bend and Springs Coast (Cedar Key tide gauge; NOAA 2017), increasing risk of exposure to parasites and predators as salinity and tidal immersion increased (Seavey et al. 2011). Oyster reefs accrete vertical elevation over time and thus can decrease the length of time spent immersed by tides (Rodriguez et al. 2014), but this requires a healthy reef, including large, long-lived individuals that can contribute ample shell to reef accretion at rates equal to or exceeding sea-level rise (Waldbusser et al. 2013). Thus, if oyster reefs are expected to keep pace with accelerating sea-level rise, managers must attempt to maximize oyster body size on reefs and keep other stressors, such as temperature and salinity, below thresholds that impact individual performance or mortality.

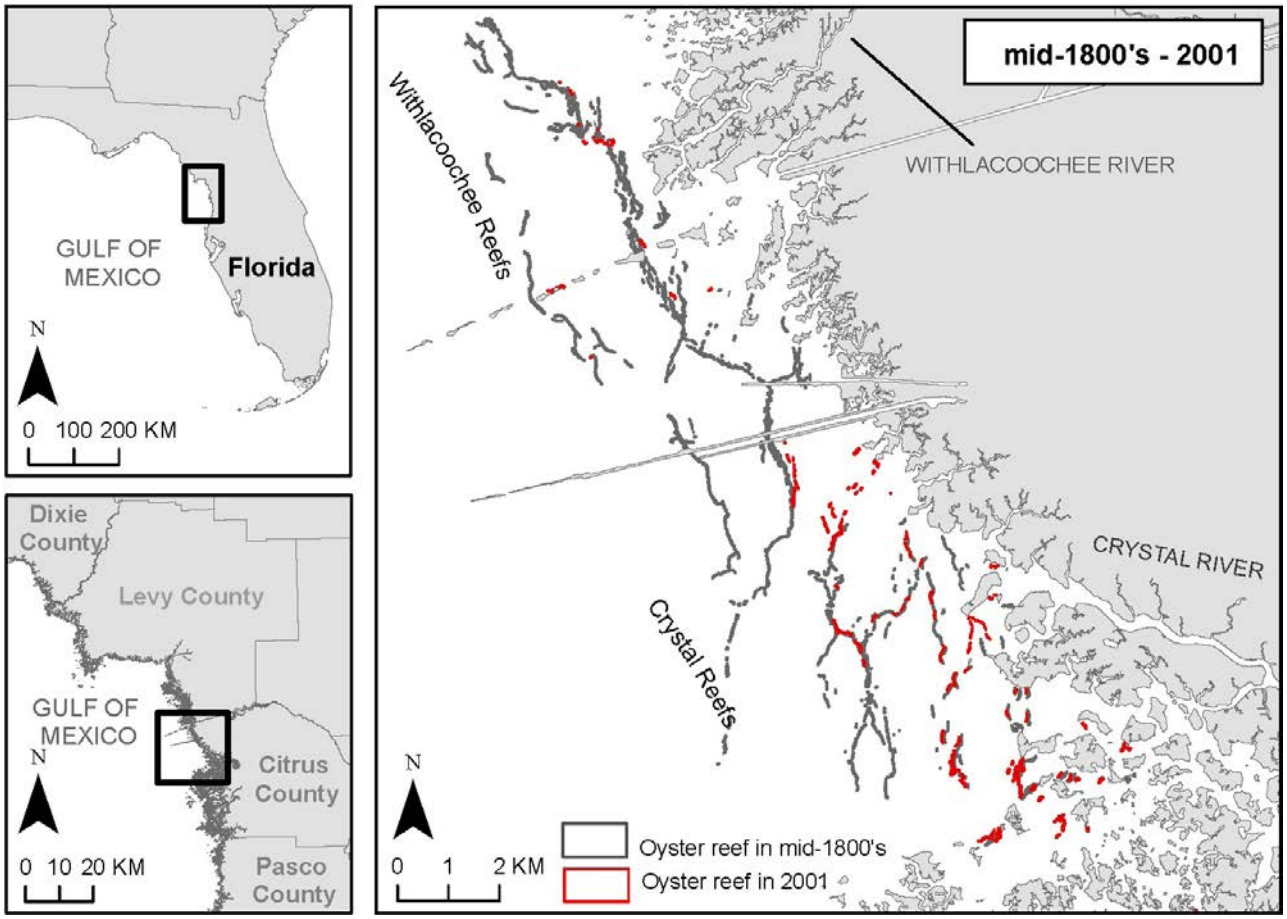

Figure 4.7. Oyster reef extent offshore of the Withlacoochee and Crystal rivers in the mid-1800s (gray) and 2001 (red). Map by Stephen Hesterberg. Data sources: Raabe et al. 2004 (made from 19th century topographic sheets), SRWMD 2001a (made from 2001 photographs).

- **Lack of adaptive, robust management strategies:** Current management strategies for oyster resources focus on a combination of size limits, bag limits, and seasonal closures to protect human health. It is not known how effective these regulations are in protecting oyster recruitment and growth from increasing fishing effort. Given that oyster populations declined before the recent increase in landings and trips in this region (Seavey et al. 2011), the concern that increased landings will result in high effective mortality is justified. Unlike Apalachicola Bay, where most of the oyster resources are found on subtidal oyster reefs, the Big Bend is characterized by large expanses of intertidal reefs. The population dynamics of intertidal reefs along the Big Bend and Springs Coast are poorly understood compared with areas where substantial fisheries exist. This creates additional challenges for managing exploited oyster reefs in the Big Bend and leads to the need for strategic, location-specific management strategies (further details provided below in the Recommendations for Management, Mapping, and Monitoring section).

Oyster reef mapping and monitoring efforts

The compilation of data used to create the oyster maps in this report is available for download at http://geodata.myfwc.com/datasets/oyster-beds-in-florida.

Northwest Florida Water Management District oyster mapping

The most recent NWFWMD land use/land cover (LULC) map that included a separate oyster reef layer was from 2009–2010 (NWFWMD 2010). Oysters were mapped following the Florida Land Use and Cover Classification System (FLUCCS), which includs a category for oyster bars (FLUCCS 6540; FDOT 1999). While more recent NWFWMD LULC maps from 2012–2013 are available, oyster bars were not mapped in those years. NWFWMD geographic information system (GIS) files are available for download at https://www.fgdl.org/metadataexplorer/explorer.jsp.

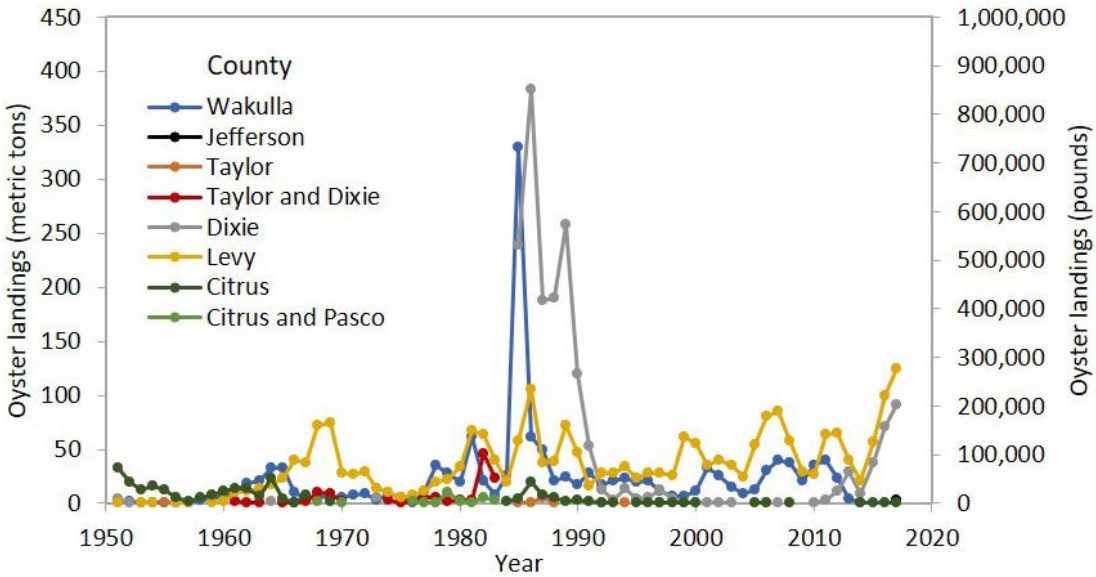

Figure 4.8. Commercial harvest of oyster meats from counties in the Big Bend and Springs Coast Region. Oyster landings before 1986 were collected under a voluntary reporting system. Data source: summary of Florida commercial marine fish landings (see Appendix A) and FWC 2018.

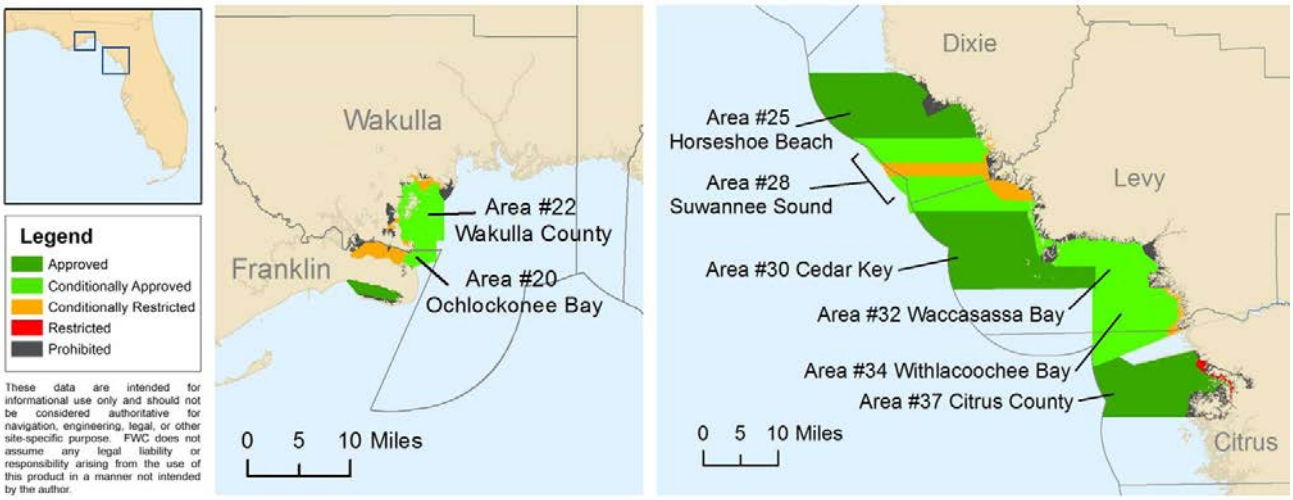

Figure 4.9. Shellfish harvesting areas in the Big Bend and Springs Coast. Data source: FDACS 2017.

Suwannee River Water Management District oyster mapping

In 2001, the SRWMD conducted a thorough seagrass and oyster mapping project in the Suwannee Estuary from Horseshoe Cove to Cedar Key using 1:24,000-scale true-color aerial photography (SRWMD 2001a, Patterson et al. 2002). The boundaries of oyster reefs were digitized with guidance from a photointerpretation key created for this effort and that differentiated between oyster reefs, patchy oyster reefs, and remnant oyster reefs. Twenty random locations were visited in the field to verify classification accuracy; overall mapping accuracy was 100% (Patterson et al. 2002).

Also in 2001, the SRWMD completed a seagrass and oyster mapping effort in Waccasassa Bay using 1:24,000-scale true-color aerial photography (SRWMD 2001b). Classifications were completed using FLUCCS categories, which included a category for oyster bars (FLUCCS 6540; FDOT 1999). Minimum mapping units were 0.25 ac (0.1 ha).

The most recent SRWMD LULC map that includes a separate oyster reef layer was from 2010–2011 (SRWMD 2011). Oysters were mapped using FLUCCS categories (FDOT 1999). While more recent SRWMD LULC maps from 2013–2014 are available, oyster reefs were not mapped in those years. SRWMD GIS files and all corre-

sponding reports for the 2001 seagrass and oyster mapping efforts are available for download at http://www.srwmd.state.fl.us/index.aspx?NID=319.

Southwest Florida Water Management District oyster mapping and surveys

The SWFWMD conducts periodic seagrass and oyster mapping using a modified version of FLUCCS (FDOT 1999). Subtidal habitats are mapped using natural-color aerial photography collected in winter at a scale of 1:24,000. Mapped habitats include tidal flats, oyster reefs, spoil areas, and seagrass. Dead and living oysters were mapped together and both included in the oyster reef class. The most recent seagrass and oyster mapping effort for the Springs Coast was completed in 2016 (SWFWMD 2016). SWFWMD shapefiles are available for download on the district website at https://data-swfwmd.opendata.arcgis.com/.

In addition to mapping programs, SWFWMD has conducted oyster surveys in the Homosassa and Chassahowitzka rivers as part of the process to determine acceptable minimum flows and levels for the rivers. Oyster surveys were conducted as part of mollusk community assessments in the Chassahowitzka River in 2007 (Estevez 2007) and the Homosassa River in 2008 (WAR 2010). More detailed oyster surveys were conducted in late 2017/early 2018 for the Homosassa, Chassahowitzka, and Withlacoochee rivers in order to evaluate minimum flows and levels and identify opportunities for oyster restoration projects.

1992 Panhandle Big Bend seagrass and oyster mapping project

In 1992–1993, the U.S. Geological Survey (USGS) National Wetlands Research Center completed an extensive mapping effort on seagrass extent from Anclote Key, located north of Tampa Bay, to Perdido Bay on the Florida–Alabama state line (USGS 1992). Aerial photos were collected at a scale of 1:24,000 in December 1992 and early 1993. Oysters were mapped if reefs were located near seagrass beds. This project was not specifically designed to provide a complete oyster reef data set, but the data set fills gaps in areas of Apalachee Bay and Waccasassa Bay that have not been covered by other mapping efforts.

Oyster restoration and monitoring in the Suwannee Sound

Recent research in the Suwannee Sound has focused on oyster population status (Seavey et al. 2011), the influence of oyster reefs on salinity (Kaplan et al. 2016), use of oyster reefs as bird habitat (Frederick et al. 2016), and effectiveness of restoration (Frederick et al. 2016). In 2017, the National Fish and Wildlife Foundation funded a multiyear project to restore the degraded Lone Cabbage Reef in Suwannee Sound. At present, a georeferenced database of these efforts is being developed, and this reef will be the subject of intensive elevation surveys as well as mapping and monitoring of oyster coverage to evaluate restoration projects. Information on the project is available at http://www.wec.ufl.edu/oysterproject/.

Historical habitat maps

Raabe et al. (2004) digitized 19th-century topographic sheets in a grid-based format to create georeferenced historical habitat maps of oyster reefs and coastal vegetation along the Big Bend and Springs Coast. The resulting map had an accuracy of ±8 m (26 ft) and showed marked shoreline erosion and landward migration of habitats. The open-file report and shapefiles of the historical habitats are available for download at https://pubs.usgs.gov/of/2002/of02-211/, enabling comparison with the modern-day locations of these habitats (Figs. 4.5 and 4.7). The use of historical topographic surveys to create habitat maps is not without complication, as these surveys sometimes had incomplete coverage due to the complexity of the shoreline and time-consuming nature of the effort (Raabe et al. 2004). Additionally, oysters in open water that presented navigation hazards were more likely to be mapped than were intertidal oysters adjacent to the shoreline.

NOAA Mussel Watch

The National Oceanic and Atmospheric Administration (NOAA) National Status and Trends Program has monitored pollutants in bivalves through the Mussel Watch program across the coastal United States from 1986 to present. Monitoring locations in this region include Black Point at Cedar Key, West Pass on the Suwannee River, and Spring Creek on Apalachee Bay (Kimbrough et al. 2008). High levels of arsenic and mercury were found in oysters on the Suwannee River (Kimbrough et al. 2008).

Recommendations for Management, Mapping, and Monitoring

- Monitoring of oyster reefs in this region should consider both the amount of habitat and its quality (i.e., ability to provide desired ecosystem functions and remain resilient in the face of threats identified above). Traditional mapping and ground-truthing of oyster

reefs is needed to help evaluate the continued impacts of reduced freshwater flow on oyster conditions and changes in reef location (FDEP 2015). Efforts to reconstruct the historical distribution of oyster habitat from aerial photographs and nautical charts could also prove useful for establishing reference conditions and elucidating the causes of habitat loss.

Although mapping the spatial distribution of oyster habitat is valuable, oyster monitoring must extend beyond reef aerial extent, which can increase due to reef collapse and therefore mask declines in habitat quality (Seavey et al. 2011, zu Ermgassen et al. 2013). The Conservation Action Plan for Marine and Estuarine Resources of the Big Bend specifically recommends monitoring oysters for population structure (e.g., size, abundance), disease, recruitment, and vertical relief of reefs (Geselbracht 2007). The NOAA Oyster Restoration Workgroup recommends monitoring similar metrics (Baggett et al. 2015). Yet widespread monitoring has not been implemented, and a severe lack of basic information on past and present reef conditions exists for the region. There are no standardized *Perkinsus marinus* (dermo) monitoring efforts for oysters within the Big Bend, nor are there standardized pre- or postharvest surveys of oyster population structure that might inform management decisions (as are conducted in Apalachicola Bay). This lack of information reduces the likelihood of detecting early warning signs of oyster population collapse and understanding what factors lead to the declines.

- Quantifying an oyster population's size structure can rapidly provide a snapshot of reef health, as large oysters are disproportionately important for reproductive output and shell budgets and provide information about the capacity of the habitat to withstand future stressors. The presence of large individuals can also be indicators of a reef's ability to cope with the threats outlined above, including salinity and thermal stress, overfishing, and sea-level rise. Change in body size has also been suggested as an early indicator prior to population collapse (Clements and Ozgul 2015, Clements et al. 2017). Given the limited time and resources available for monitoring, size structure of oyster populations should be emphasized in future assessments as it is both important and easy to quantify (Woodward et al. 2005).
- Large-scale oyster reef restoration should continue in the Suwannee Sound and expand to other sections of the coast, complementing habitat protection towards achieving regional conservation goals. Substrate-focused restoration efforts on Suwannee Sound oyster reefs have proved successful in improving oyster densities and reducing salinity inshore of the reefs, allowing for mitigation of some of the impacts of reduced freshwater flows (Frederick et al. 2015, Kaplan et al. 2016). But the long-term effectiveness of adding shell material to promote oyster reef growth is uncertain given the oyster population stock status and limited availability of cultch. If this restoration method proves successful, other viable locations, such as the Withlacoochee, Crystal, Homosassa, and Chassahowitzka rivers, should be considered for future restoration.

  An alternative strategy is to develop a better understanding of how positive shell budgets can be maintained naturally on oyster reefs without the addition of shell material. For harvested reefs, this may necessitate examining fishery practices including how and where culling takes place and determining survival rates of culls. It may also require implementing rotational harvest policies to allow for the accumulation of shell material through natural mortality processes on extant reefs. It is not certain whether restoration is more effective when creating large areas of low-relief habitat or creating smaller areas of high-relief habitat. Addressing these uncertainties will increase the effectiveness of management and restoration.
- The 2012 collapse of the Apalachicola oyster fishery and subsequent increase in oyster harvest in the Big Bend region have created a need to re-evaluate oyster fishery policies in the region. While size regulations, bag limits, and seasonal closures may once have promoted sustainable fisheries, new fishery management strategies should be considered, along with changing climate, fluctuating oyster fishing effort, and widespread anthropogenic changes, in efforts to reduce the risk of resource collapse (Camp et al. 2015). These strategies could include watershed-scale management and reef resilience, consideration of positive shell budgets and diverse size–age structure on remaining reefs (Quiros et al. 2017). Better enforcement is also needed to address harvesting below minimum size limits and returning culled oysters to their reef of origin. Experimental management policies that might be considered include rotational harvest of wild oyster areas (with long fallow periods) and the development of Territorial User Rights Fisheries (TURFs), which transfer (through leases) rights to bottom areas to individual fishers. The latter approach could promote innovative modes of reef conservation and restoration by providing TURF holders the incentive to better manage their individual resource. Finally, the growing oyster aquaculture industry should be viewed as a new alternative for providing direct benefits to oyster harvesters and local communities and reducing harvest pressure on wild oyster resources.

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