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Oyster Integrated Mapping and Monitoring Program Report for the State of Florida

KARA R. RADABAUGH, STEPHEN P. GEIGER, RYAN P. MOYER, EDITORS

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Technical Report 22 • 2019

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Cover photograph: An eastern oyster (*Crassostrea virginica*) reef in St. Petersburg, Florida. Photograph by Kara Radabaugh.

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An assortment of oysters on red mangrove (*Rhizophora mangle*) prop roots in the Ten Thousand Islands, Florida. Photo credit: Kara Radabaugh.

Summary of the Oyster Integrated Mapping and Monitoring Program

The Oyster Integrated Mapping and Monitoring Program (OIMMP) began as a joint effort between the Coastal Wetlands and Molluscan Fisheries research programs at the Florida Fish and Wildlife Conservation Commission's Fish and Wildlife Research Institute in St. Petersburg, Florida. OIMMP is based on the framework established by the Seagrass Integrated Mapping and Monitoring (SIMM) program (http://myfwc.com/research/habitat/seagrasses/projects/active/simm/) and the Coastal Habitat Integrated Mapping and Monitoring Program (CHIMMP) (http://myfwc.com/research/habitat/coastal-wetlands/projects/chimmp/), which rely upon a network of ecosystem experts to assemble regional summaries of mapping and monitoring data. The main objective of OIMMP was to build and maintain a collaborative network of stakeholders with interest in mapping and monitoring Florida's oyster habitats in order to identify the status of and management priorities for oysters and their habitats.

OIMMP workshops were held at the Guana Tolomato Matanzas National Estuarine Research Reserve in 2017 and 2018 and the Fish and Wildlife Research Institute in 2019 to bring together oyster researchers and managers from across the state. During these workshops, attendees gave presentations on oyster mapping and monitoring activities and made recommendations for future mapping, monitoring, and management of oyster resources. See http://ocean.floridamarine.org/OIMMP for detailed proceedings and outcomes of the OIMMP workshops.

Attendees of the 2017 workshop developed the regional boundaries for the chapters in this report, and many attendees also volunteered to contribute their expertise as coauthors. Additional regional coauthors were added based on need and personal recommendations (see below for a list of all regional contributors). Due to the collaborative nature of this report, the style, content, and level of detail varies among chapters based upon regional data availability, range of participating organizations, and expertise of the contributing authors.

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See Appendix for affiliation abbreviations.

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Restored oyster reef in the Mosquito Lagoon. Photo credit: Linda Walters.

Executive Summary

Oysters provide a variety of critical ecosystem services to coastal communities in Florida. They improve water quality and clarity as they filter feed, lessen shoreline erosion, and provide a habitat or food source for a wide variety of birds, fish, and invertebrates. Oysters are commercially valuable as a harvested food source, and historically their shell has been mined extensively for construction material. The eastern oyster (*Crassostrea virginica*) is the only reef-building oyster in Florida and forms both subtidal and intertidal reefs. Numerous other species of non-reef-building oysters are less frequent. This report focuses primarily on the eastern oyster, because it is the most abundant oyster in Florida and because it is important as both a keystone species and an ecosystem engineer.

The survival of an oyster reef depends on its shell budget, which is its rate of shell deposition from new oyster growth relative to the rate of shell loss. The rate of growth is limited by basic biological functions of living oysters (rates of growth and reproduction), while the rate of loss is a function of both biotic (e.g., predation, competition) and abiotic (e.g., salinity, temperature, pH) factors that can affect both living oysters and the shells of deceased oysters. Rates of bioerosion, chemical degradation, dissolution, and burial affect the length of time the dead shell remains on a reef as viable settlement substrate. The optimum salinity range for eastern oysters is 14 to 28, although they can temporarily tolerate salinity extremes from 5 to 40. Oysters have decreased growth and reproduction at low salinity and can quickly suffer high rates of mortality under freshwater conditions. While oysters can physiologically tolerate high salinity for extended periods of time, in such conditions they are more vulnerable to marine predators, disease, and parasitism. Tolerance of high or low salinity is significantly diminished at high temperatures, which oysters frequently encounter in Florida. Climate change and sea-level rise further alter the frequency and severity of temperature and salinity stress.

Many Florida estuaries have lost 80–90% of the oyster reefs that were present before human development. Altered surface-water flow is one of the major threats to oyster reefs today, as channelization or other mechanisms that concentrate stormwater runoff reduce salinity to levels less than those optimal for oyster survival, growth, and reproduction. Hydrologic alterations, coupled with freshwater withdrawal, also starve downstream areas of freshwater flow, resulting in increased salinity that makes oysters more vulnerable to predation and disease. Significant losses to oyster reefs have occurred due to anthropogenic activities including dredge-and-fill construction, harvesting without substrate replacement, and, in the past, shell mining. Oyster populations must also cope with water pollution, competition with invasive species, sedimentation, and accelerated erosion due to boat wakes. Many of these stressors are increasing as Florida's human population grows. The decline in the Florida oyster population has led to small and often isolated populations spread across Florida's estuaries. Although the long-term and large-scale ramifications of this decline are not well studied, the isolation of these small populations can limit genetic diversity and connectivity between estuaries.

Oyster harvesting is permitted in Florida within designated shellfish harvesting areas. The Florida Department of Agriculture and Consumer Services (FDACS) regulates the opening and closure of these harvesting areas based on health risks to consumers. Both FDACS and the Florida Fish and Wildlife Conservation Commission (FWC) monitor the waters in these shellfish harvesting areas for bacteria, red tide, and chemical pollutants. FWC reports commercial harvesting yields, which have had mandatory reporting since 1986. Historically 90% of the state's harvests originated from Apalachicola Bay in Franklin County; however, harvests from Apalachicola Bay (and consequently statewide harvests) have declined significantly since the 2012–2013 collapse of the bay's fishery.

Large-scale oyster reef mapping relies primarily on georeferenced multispectral or hyperspectral imagery with in situ ground truthing to verify mapping accuracy. Reefs are identified by patterns of color, texture, and shape, but reef identification can be confounded if oysters are intermixed with algae, mud, seagrass, or rubble. Oysters that grow on mangrove roots or seawalls are generally not included in mapping efforts because they are hard to see in aerial imagery. These oysters nevertheless contribute significantly to the oyster population in an estuary. Subtidal reefs can be mapped with side-scan or multibeam sonar or videography with simultaneous acquisition of global positioning system (GPS) data, but ground truthing is necessary to verify the presence of live oysters. Subtidal oyster mapping is complicated by murky water, variable water depth or shallow water, limited benthic relief, and oyster reefs co-occurring with multiple benthic habitats such as seagrass beds and hardbottom. Oyster maps in Florida generally focus on a specific region or estuary. FWC and the Oyster Integrated Mapping and Monitoring Program have compiled available oyster maps across the state, creating the most comprehensive map to date of oyster distribution in Florida (http://geodata.myfwc.com/datasets/oyster-beds-in-florida). This map identifies more than 7,920 ha (19,580 ac) of live oysters, yet gaps remain, and further mapping efforts are needed in several areas.

Oyster monitoring in Florida is conducted by a number of agencies and organizations with a variety of objectives, such as determining the efficacy of hydrologic restoration, the health of oyster fisheries, or the success of restoration efforts, as well as general ecological assessments. While methods used in monitoring programs may vary widely, commonly measured parameters include water quality (salinity, dissolved oxygen, and temperature), reef area, reef height, oyster density, degree of tidal emersion, and oyster size–frequency distribution.

The chapters in this report summarize mapping and monitoring programs for oyster reefs in each region of Florida. Content of each chapter includes a general introduction to the region, location-specific threats to oyster reefs, a summary of selected mapping and monitoring programs, and recommendations for oyster management, mapping, and monitoring. Regional figures include the FWC compilation of oyster maps, FDACS shellfish harvesting areas, and oyster harvesting data from 1951 to 2017.

Through the process of compiling this report and from feedback provided at the OIMMP workshops, several needs and recommendations were identified for Florida oysters:

Management priorities and recommendations

- Manage freshwater flow to mimic natural flow, avoiding rapid salinity changes and prolonged exposure to salinity extremes.
- Add shell and other materials to combat substrate limitation due to extensive harvesting, dredging, or past shell mining. Place substrate on firm sediments to prevent its sinking, and determine ideal locations based on current hydrologic conditions rather than historic reef extent.
- Create and implement a comprehensive oyster fishery management plan that takes into account climate change, variable oyster fishing effort, shell budgets, annually variable freshwater input, and widespread anthropogenic changes in order to prevent overfishing or loss of substrate.
- Replace or supplement hardened shorelines with living shorelines to create habitat and facilitate habitat migration upslope as sea level rises.
- Maintain genetic connectivity of oyster populations between estuaries across the state by rebuilding or maintaining stable oyster populations in all estuaries where they naturally occur.

Mapping priorities and recommendations

- Fill remaining mapping gaps in the Panhandle (Pensacola, Choctawhatchee, and St. Andrew bays), Big Bend and Springs Coast (Apalachee Bay and subtidal oysters), Everglades, and Indian River Lagoon (outside of its major tributaries).
- Complete regular mapping efforts every 5–7 years. Oyster extent is dynamic due to urban development, variable freshwater flow, and changing freshwater management, so maps should be updated regularly.
- Map all oysters, including subtidal oysters and oysters on mangrove roots and seawalls.
- Determine historical extent of oyster reefs to facilitate decision making regarding targets for future reef extent.
- Differentiate between live and dead sections on oyster reefs to track mortality or dead margins on live reefs over time.

Monitoring and research priorities and recommendations

- Conduct standardized and long-term monitoring across multiple estuaries to facilitate comparisons among oyster populations.
- Determine genetic diversity, life history, and habitat characteristics of high-salinity oyster reefs to determine why certain oyster populations survive in high salinity while others are decimated by predators and disease.
- Quantify oyster size structure of oyster populations. Shell height in an oyster population can provide a snapshot of reef resilience because large oysters are disproportionately important to reproductive output and shell budgets.
- Make high-frequency autonomous measurements of temperature and salinity near established oyster reefs in order to capture extreme events such as freshwater pulses, temperature extremes, and hypoxic conditions.

Chapter 1
Introduction

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Introduction to oysters in Florida

The dominant and only reef-forming bivalve in Florida is the eastern oyster, also known as the American oyster, *Crassostrea virginica* (Fig. 1.1). The eastern oyster is found intertidally and in shallow subtidal depths along Florida's nearshore and inshore estuarine waters (Fig. 1.2). Eastern oysters are ecosystem engineers as well as keystone species, and the reefs they build provide a variety of critical ecosystem services to coastal communities (Grabowski et al. 2012). Oyster reefs are commercially valuable as a harvested resource and indirectly valuable for protecting shorelines against erosion (Grabowski and Peterson 2007, Scyphers et al. 2011). As filter feeders, oysters improve water quality and clarity by removing nutrients and other pollutants from the water column (Kellogg et al. 2014). The Florida State Wildlife Action Plan (FWC 2012) identifies numerous species of greatest conservation need in Florida as being linked to habitat or food sources provided by oyster reefs, including the eastern oyster itself.

Figure 1.1. Eastern oyster (*Crassostrea virginica*) shells are easily recognizable by the purple or red-brown scar left by the adductor muscle. The morphology of the shell varies widely (Galstoff 1964). Photo credit: Kara Radabaugh.

Eastern oysters belong to the family Ostreidae, the true oysters. Florida contains several non-reef-forming members of this family including the mangrove oyster (*C. rhizophorae*), crested oyster (*Ostrea stentina*), threaded oyster (*Teskeyostrea weberi*), frond oyster (*Dendostrea frons,* typically found on soft corals), and possibly the commercial sponge oyster (*Ostrea permollis*), a Caribbean species. The crested oyster is difficult to distinguish from the eastern oyster without examining the inside of the shell. Florida also has oysters in the family Pteriidae (the pearl or winged oysters), including the scaly pearly oyster (*Pinctada longisquamosa*), Atlantic pearly oyster (*P. imbricata*), black-lipped pearly oyster (*P. margaritifera*), Atlantic wing oyster (*Pteria colymbus*), and glassy wing oyster (*P. hirundo*). Pteriidae tree oysters, including the flat tree oyster (*Isognomon alatus*), bicolor purse-oyster (*I. bicolor*), and radial purse-oyster (*I. radiatus*), co-occur in some locations associated with mangroves and are occasionally found in small numbers on reefs of the eastern oyster. One species of hammer oyster (family Malleidae), the Caribbean hammer oyster (*Malleus candeanus*), is found only in coral reef habitats. Finally, foam oysters (family Gryphaeidae) are also found on coral and other

Figure 1.2. Oyster reef extent in the state of Florida (FWC compilation of oyster maps, FWC 2018).

marine hardbottom. Florida has two native species of foam oysters, the Atlantic (*Hyotissa mcgintyi*) and deep-water (*Neopycnodonte cochlear*), as well as one introduced species, the giant foam oyster (*H. hyotis*) (Forbes 1966, Thayer et al. 2005, Mikkelsen and Bieler 2008).

Eastern oyster life history and ecology

Eastern oysters reproduce via broadcast spawning with external fertilization, while other oysters (e.g. *Ostrea* spp.) reproduce via internal fertilization and brooding before the release of larvae. Because multiple generations of oysters settle on top of one another, numerous generations of eastern oysters can contribute to a diverse mixture of genotypes during spawning (NOAA 2007). Areas with low oyster density have lower spawning biomass and consequent decreased rates of fertilization (Mann and Evans 1998). Spawning generally requires water temperatures above 20 °C (68 °F); while May–October is the peak period in Florida, reproduction may continue year-round in all but the coldest times of the year (Berrigan et al. 1991, Volety et al. 2009).

After hatching, eastern oysters spend 2–3 weeks as planktonic larvae before settling on a hard substrate (Stallworthy 1979, Kennedy 1996). During settlement, oyster larvae attach to a substrate and metamorphose into their sessile benthic form. Successful recruitment refers to both settlement and some period of postsettlement survival (Wildlish and Kristmanson 1997, Baggett et al. 2014). Rates of settlement are generally dictated by larval density, water residence time, water quality, substrate availability, and larval mortality. In contrast, recruitment and postsettlement survival are additionally influenced by rates of predation, environmental stress, and competition with other bivalve species or conspecifics (Mann and Evans 1998, Baggett et al. 2014). Recruitment is therefore highly variable spatially and temporally (on both seasonal and annual timescales). Estuaries with high rates of flushing tend to have low but more consistent recruitment, while estuaries with low flushing have higher but more variable recruitment (Kennedy 1996). After settling, oysters can reach reproductive maturity in as little as four weeks and grow to a length of 7–8 cm (3 in) in the first 18–24 months in Florida's warm waters (NOAA 2007, FWC 2010, VanderKooy 2012). Most eastern oysters are protandrous hermaphrodites, meaning that they begin life as males and later change to primarily female reproductive organs. Some females may revert to male later in life (Thompson et al. 1996).

Spawning and larval development may be reduced in response to high temperatures or abrupt changes in temperature (Kennedy et al. 1996). While oysters are tolerant of extreme high temperatures up to 36–40 °C (97–104 °F), their tolerance decreases above 28 °C (82 °F) if the high temperatures co-occur with disease, low oxygen, or salinity extremes (Shumway 1996, Coen and Bishop 2015, Rybovich et al. 2016, Southworth et al. 2017). Tolerance of salinity extremes are similarly limited if combined with higher temperatures (Shumway 1996). Temperature and salinity influence nearly every aspect of oyster physiology, including feeding, respiration, reproduction, larval life span, growth, and survival (Shumway 1996). While eastern oysters can briefly tolerate extreme salinity ranging from 5 to 40, prolonged exposure to salinity outside their ideal range of 14 to 28 can harm both subtidal and intertidal populations (Shumway 1996, Baggett et al. 2014, Coen and Bishop 2015). Growth and reproduction decrease at low salinity, and oysters can suffer high rates of mortality over a short period under freshwater conditions (Shumway 1996, Thayer et al. 2005, Turner 2006). While oysters can physiologically tolerate high salinity for extended periods, they are more vulnerable to disease and predation as marine predators and parasites of oysters can survive and reproduce in those saline conditions (Coen and Bishop 2015, Garland and Kimbro 2015). Several estuaries in Florida are home to significant populations of oysters that survive in an average salinity range of 30–35 (Parker et al. 2013). These populations are predominantly intertidal and so have daily protection from marine predators during exposure at low tide. Oysters in these reefs must have some combination of a reproductive potential that exceeds predation and parasitism or have a genetic aptitude towards survival at high salinity (Koehn et al. 1980a).

Eastern oysters in Florida are of two relatively distinct genotypes, which can be differentiated by genetic analyses. The Atlantic coast of the United States from Maine to Cape Canaveral, on the east coast of Florida, predominantly supports the Atlantic coast genetic stock, while the southeastern Atlantic and Gulf of Mexico coasts from Florida to Texas are home to the Gulf coast genetic stock (Buroker 1983, Hare and Avise 1996, FWC 2010). Marine invertebrate species including bivalves often exist as a metapopulation, a group of spatially separated local populations with some degree of interbreeding and genetic exchange (Kritzer and Sale 2006, Bert et al. 2014). For instance, Florida bay scallops (*Argopecten irradians concentricus*) exhibit genetic exchange between local populations within the larger metapopulation (Bert et al. 2014). The eastern oyster is also such a species; its local populations can be relatively isolated, but interbreeding and larval export sometimes occur (Reeb and Avise 1990, Murray and Hare 2006, Varney et al. 2009, Anderson et al. 2014, Arnold et al. 2017).

Larval export and settlement in neighboring estuaries are key to the maintenance of a population's genetic diversity (Kritzer and Sale 2006). Most of Florida's estuaries once had abundant oyster reefs, but the areal extent of these reefs has declined significantly (see discussion on threats below). When the oyster population in an estuary declines, it reduces the chances that larvae will be exported from that estuary and subsequently imported by a neighboring estuary. As a result, oyster populations in each estuary function largely as an isolated local population with only occasional larval exchange (Arnold et al. 2017). Detailed genetic analyses are needed to determine the degree of larval export and genetic isolation among Florida's remaining oyster reefs, as genetic diversity is key to oyster populations surviving the variety of environmental stressors they currently face (Koehn et al. 1980b, Hilbish and Koehn 1987, Arnold et al. 2017).

Oysters grow best in regions where water currents move settled particulates. These currents provide water exchange for feeding and keep oysters from being smothered in sediment, their own feces, or pseudofeces (incorporated particulates expelled by the oyster with mucus that did not pass through the digestive tract) (Lenihan 1999, Levinton et al. 2001). Oysters filter feed on phytoplankton, small zooplankton, bacteria, suspended particulate organic matter, and dissolved organic matter in the water column (Langdon and Newell 1996, Dame et al. 2001). Oysters remove excess organic matter and fine sediments from the water; this filter feeding increases water clarity and improves light conditions for seagrass and other benthic photosynthesizers (Booth and Heck 2009, Peterson and Heck 2001a, 2001b). Additional benefits may occur when the complex structure of oyster reefs entrains sediments by physical processes; these benefits include improved water quality and stabilized reef structure resulting from the filling of pore spaces between shells (Walles et al. 2015). Additionally, algae and associated organisms often form a complex film that encrusts oyster shells and serves as an important food source for invertebrates that live within the microhabitats of oyster reefs.

Figure 1.3. Oysters growing intertidally on red mangrove (*Rhizophora mangle*) prop roots. Photo credit: Kara Radabaugh.

Oyster reefs and shell budgets

Over many decades, multiple generations of oysters settle upon one another, constructing complex calcium carbonate reefs. Reef accretion depends on the rate of shell deposition relative to rate of shell loss; the balance between the two can be quantified as a shell budget (Powell et al. 2006). Natural shell deposition occurs through the growth and calcification of oysters. Large, long-lived oysters are particularly important contributors of shell material (Waldbusser et al. 2013). Oyster growth encourages further reef building as shell presence leads to increased settlement of calcifying organisms. For this reason, occasional die-offs can even increase the rate of carbonate addition to the reef (Kidwell and Jablonski 1983). Shell mass can be lost as a result of bioerosion (from boring sponges, worms, and mollusks), chemical degradation, dissolution, subsidence, burial, erosion, and habitat loss due to harvesting or dredging (Powell et al. 2006, Waldbusser et al. 2011, Rodriguez et al. 2014). Oyster reefs with a balanced shell budget can maintain their intertidal position or depth in the water column in response to changes in water depth as sea level rises. Yet many oyster reefs in Florida and the eastern United States have negative shell budgets; this shell loss results from a complex set of factors (see discussion on threats, below).

Figure 1.4. Oyster reefs are home to numerous other invertebrates, including predators of oysters such as the crown conch (*Melongena corona*) and reef associates such as the West Indian false cerith (*Batillaria minima*). Photo credit: Kara Radabaugh.

Oyster reefs are found in the majority of bays and lagoons in Florida (Fig. 1.2). Local oyster distribution is limited to locations with hard substrates for attachment, such as hardbottom, mangroves (Fig. 1.3), seawalls, pilings, or shell accumulations (FWC 2010, Drexler et al. 2014). Fringing intertidal oyster reefs occur on the edges of shallow embankments in and around estuaries and lagoons, where they stabilize shorelines and prevent erosion (Scyphers et al. 2011, Hanke et al. 2017). Subtidal reefs are generally found in water less than 4–5 m (13–16 ft) deep (MacKenzie 1996, NOAA 2007). The physical structure of the reef and its associated fauna provide a complex refuge and feeding habitat for many invertebrates including mollusks, echinoderms, fish, crustaceans, flatworms, boring sponges, polychaetes, mammals, and birds (Fig. 1.4; ASMFC 2007, Coen et al. 2007, zu Ermgassen et al. 2016). Over 30 species of greatest conservation need in Florida are linked to habitat or food sources provided by oyster

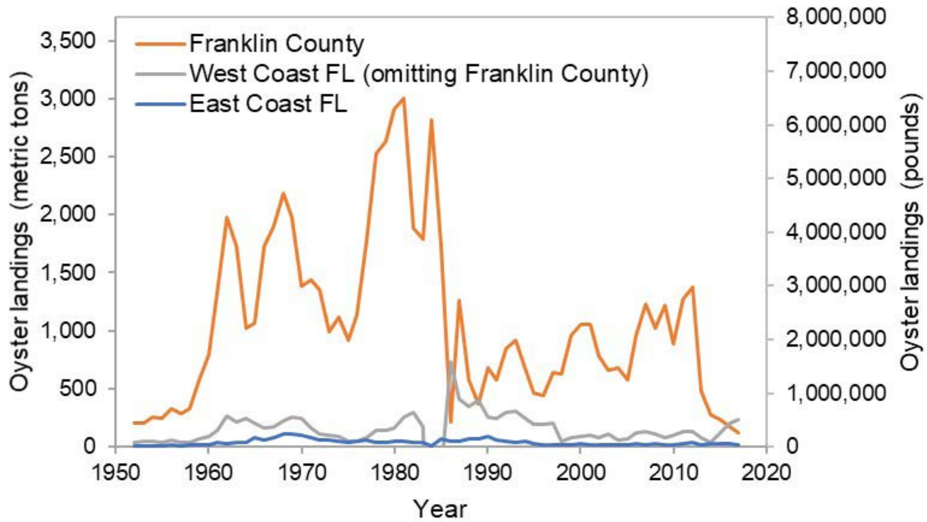

Figure 1.5. Oyster harvest in Franklin County (Apalachicola Bay), the remainder of the west coast of Florida, and the east coast of Florida from 1951 to 2017. Oyster landings data before 1986 were collected under a voluntary reporting system. Data sources: 1951–1983, Florida Commercial Marine Fish Landings (see Appendix A); 1984–1985, Berrigan (1990); 1986–2017, FWC (2018).

reefs, including the American Oystercatcher (*Haematopus palliates*), the Short-billed Dowitcher (*Limnodromus griseus*), the Marbled Godwit (*Limosa fedoa*), the diamond-backed terrapin (*Malaclemys terrapin*), the peppermint shrimp (*Lysmata wurdemanni*), and the banded tulip snail (*Fasciolaria lilium*) (FWC 2012).

Oyster harvesting in Florida

Oysters have high intrinsic economic value and have been both harvested as a food source and mined for shell (Coen et al. 2007, Grabowski et al. 2012). About 98% of Florida's oyster harvests come from the Gulf coast, and 90% of the state's historical harvests originated from Apalachicola Bay in Franklin County (Fig. 1.5; FWC 2010, VanderKooy 2012). Oysters are abundant along the Atlantic coast of Florida, though they are nearly all in intertidal reefs and harvesting is less common. Statewide harvests have varied significantly since 1950 (Fig. 1.5), largely as a result of varying salinity and the impacts of hurricanes on Apalachicola Bay (Berrigan et al. 1991, FWC 2010). In 2012–2013, a dramatic decline occurred in the oyster fishery in Apalachicola Bay due to a combination of low river flow, increased predation and disease, and removal of substrate by fishing; all these factors led to high mortality and low recruitment (Camp et al. 2015, Pine et al. 2015, Fisch and Pine 2016). Historical catch data and the fishery collapse in Apalachicola Bay are discussed in detail in Chapter 3 of this report.

Oyster decline has been even more dramatic on the mid-Atlantic coast of the United States, where commercial harvests have decreased to only 1–2% of previously recorded landings as a result of overharvesting and mortality from parasites (Beck et al. 2011). Oyster populations on the mid-Atlantic coast are now considered to be in poor condition or functionally extinct (Beck et al. 2011). Before the 2012 collapse in Apalachicola Bay, landings and oyster populations in the Gulf of Mexico were more stable than in the mid-Atlantic (NOAA 2007, Beck et al. 2011).

The Florida Shellfish Commission was established in 1913 to enact shellfish harvesting laws and implement a leasing program. Since that time, responsibility for shellfish harvesting regulations has passed through several state agencies (Arnold and Berrigan 2002). Data on historical commercial fishery yields from 1950 through 1983 were generated by a variety of agencies (see Appendix A). Commercial harvest records from 1984 to present are available from the Florida Fish and Wildlife Conservation Commission (FWC). Commercial landings from all Florida fisheries, including oysters, are available in a report generator that can sort landings by coast, region, or county at https://myfwc.com/research/saltwater/fishstats/commercial-fisheries/landings-in-florida/ (FWC 2018). Prior to 1986, data on trips and oyster landings were collected voluntarily; the FWC has since recorded this information via a mandatory reporting system (Camp et al. 2015). The FWC establishes limits and seasons for both commercial and recreational harvest. Current regulations

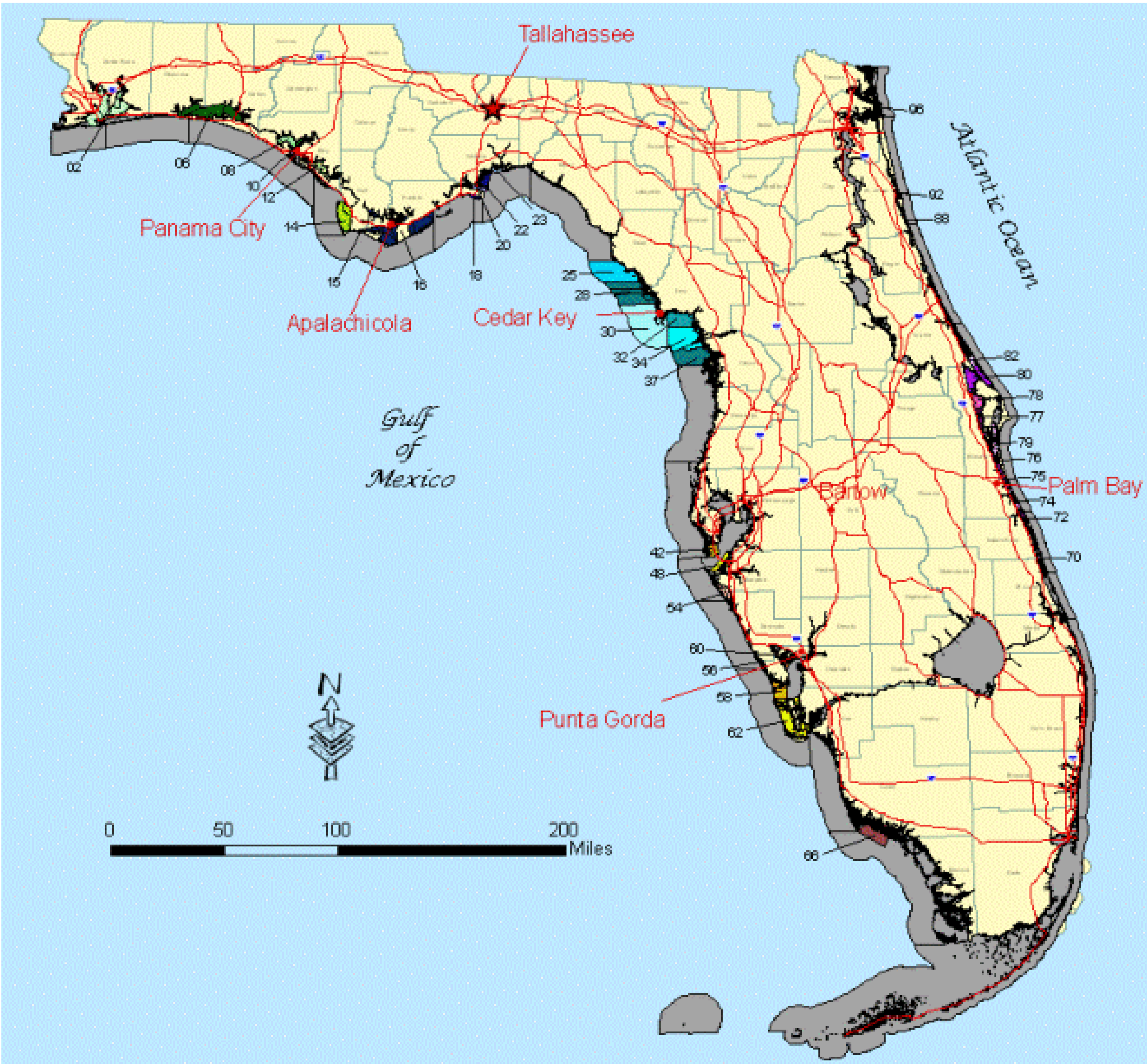

Figure 1.6. Shellfish harvesting areas in Florida. Figure credit: FDACS 2017.

for the commercial or recreational harvest of oysters may be found at http://myfwc.com/fishing/saltwater. States surrounding the Gulf of Mexico require a market size of at least 7.6 cm (3.0 in) shell height for oysters harvested from public reefs, but in Florida this size limit does not apply to private oyster leaseholders (VanderKooy 2012). Commercial oyster landings in Florida are reported as pounds of oyster meats. Conversion factors for commercial landings of oysters and other marine species may be found at https://myfwc.com/media/9085/sumfact.pdf.

The Florida Department of Agriculture and Consumer Services (FDACS) divides Florida into shellfish management areas (Fig. 1.6) and issues leases for the cultivation of oysters. Shellfish management areas are classified into sections that are deemed approved, conditionally approved, conditionally restricted, restricted, or prohibited for shellfish harvest. As filter feeders, oysters may accumulate harmful substances including heavy metals, toxins from harmful algal blooms (HABs), pathogenic bacteria such as *Vibrio parahaemolyticus,* and viruses like Norovirus. Therefore, bays in Florida in which oysters are commercially harvested for human consumption are monitored for bacteria, red tide, and other pollutants by FDACS and FWC. FDACS can issue closures of shellfish harvest areas when oysters are not safe for consumption. The regional status of shellfish harvesting areas in Florida is available at http://shellfish.floridaaquaculture.com/seas/seas_statusmap.htm.

Oyster shells have been dredged or mined in Florida for use in road construction, in decorative projects such

Figure 1.7. Bagged cultch is used to provide a substrate for oyster settlement. Photo credit: GTMNERR.

as driveways and walkways, or as material for oyster settlement (known as cultch) for oyster restoration efforts (Whitfield 1975). Extensive shell deposits across Florida can be found in middens from centuries of harvesting by indigenous populations (Dame 2009, Saunders and Russo 2011). Oyster shells were mined extensively in Florida from shell middens as well as from submerged and intertidal oyster reefs. Before 1947, companies could dredge shell from live oyster reefs that were deemed unproductive, so long as an artificial reef was constructed as replacement. But productive reefs were sometimes targeted for dredging, and restored extent often fell short of the original reefs (Whitfield 1975). Extensive shell dredging operations led to a decline in suitable oyster habitat in several estuaries due to a lack of suitable hard substrate and excess sedimentation (Whitfield 1975).

Because oysters provide essential ecosystem services and are so important economically, a great deal of effort is being exerted to enhance or restore oyster reefs. Restoration objectives may focus on increasing harvest potential or improving the value of the ecosystem services provided by oyster reefs. Oyster settlement is encouraged in many areas of Florida through the provision of cultch such as bagged or unconsolidated natural or fossil shell (Fig. 1.7; Brumbaugh et al. 2006, Walters et al. 2017). Management practices for fisheries often include replacing harvested substrate with cultch. As shell and fossil shell become more expensive or difficult to find, some oyster restoration projects, especially large commercial reefs, have switched to various types of rock, especially limestone. A variety of other alternative substrates can also be used in oyster reef restoration (concrete, porcelain, sandstone, granite, clam shell, engineered options, etc.; Goelz 2017). The Oyster Restoration Workgroup website, available at http://www.oyster-restoration.org, presents an array of resources regarding site selection, materials, gear, implementation, monitoring, and reports of oyster restoration projects.

Threats to oyster reefs

Globally, oyster reef habitat has declined by more than 85%, and remaining habitats are often in poor condition (Beck et al. 2011). In the United States, oyster spatial extent has decreased by as much as 64%, and oyster biomass has

Figure 1.8. A lightning whelk (*Sinistrofulgur sinistrum*) consuming oysters on a reef in Tampa Bay, Florida. Photo credit: Christine Russo.

declined by 88% (zu Ermgassen et al. 2012). Worldwide, declines are due primarily to unsustainable harvesting practices in combination with disease, pollution, sedimentation, and competition with nonnative species (Beck et al. 2009, Beck et al. 2011, Gillies et al. 2015). The loss of this keystone species can alter the trophic structure of an estuary. For example, in the Chesapeake Bay, dramatic declines in oyster populations have been linked to increased occurrences of hypoxia and shifting food chain dominance from benthic to pelagic organisms (Ulanowicz and Tuttle 1992, Thayer et al. 2005, Breitburg and Fulford 2006).

Although Florida oyster populations were considered to be more stable than those in many other regions (Beck et al. 2011, zu Ermgassen et al. 2012), the State still classifies reefs as being in relatively poor and declining condition with a very high level of habitat threat (FWC 2012). It is estimated that 80–90% of oyster reefs have been lost in several Florida estuaries (Meeder et al. 2001, Schmid et al. 2006, Estevez 2010, Boswell et al. 2012, Kaufman 2017). Although Apalachicola Bay historically dominated state harvests, it suffered serious declines in 2012–2013 and has not recovered (Fig. 1.5; Pine et al. 2015). The most critical stressors identified in Florida's State Wildlife Action Plan include altered hydrologic regimes, altered water quality, and habitat disturbance (FWC 2012). These and other threats are described in further detail below.

Altered hydrology and salinity: Altered hydrology as a result of stormwater management, canalization, freshwater withdrawal, and coastal development is the most significant threat to bivalve habitats in Florida (FWC 2012, Camp et al. 2015). Altered hydrology and low flushing can lead to extreme salinity events, increased sedimentation, low oxygen levels, and increased temperature. Channelization reduces sheetwater flow through coastal wetlands and concentrates freshwater runoff, reducing salinity around outflows beyond levels optimal for oyster growth and reproduction (Thayer et al. 2005, Turner 2006). Conversely, freshwater withdrawal, diverted stormwater runoff, and drought conditions can increase estuarine salinity, making oysters more vulnerable to predation and disease (Coen and Bishop 2015). Rapidly changing or seasonally variable salinity can also have detrimental effects on fish and invertebrate communities associated with oyster reefs (Tolley et al. 2006) but may provide relief from predation and disease. A large number of oyster reefs in Florida are stressed by salinity extremes brought about by altered hydrology, particularly in south Florida, Apalachicola Bay, and the Big Bend (Tolley et al. 2005, Parker et al. 2013, Camp et al. 2015, Frederick et al. 2016). The locations of oyster reefs have even shifted inshore or upriver in the Big Bend and the Everglades, following the lower salinity regimes (Volety et al. 2009, Seavey et al. 2011).

Figure 1.9. Boring-sponge infestation on an oyster shell. Photo credit: Linda Walters.

Predation: Rates of predation and density of predators (native and nonnative) on oyster reefs are influenced by a variety of physical factors including salinity, temperature, and dissolved oxygen (Eggleston 1990, White and Wilson 1996, Tolley et al. 2005, Garland and Kimbro 2015). High salinity allows for the survival of marine predators, making oysters vulnerable to high rates of predation. Predators of eastern oysters include crown conchs (*Melongena corona*), oyster drills (*Stramonita haemastoma* and *Urosalpinx cinerea*), mud crabs (*Panopeus herbstii*), black drum (*Pogonias cromis*), and occasionally lightning whelks (*Sinistrofulgur sinistrum*) (Fig. 1.8; Tolley et al. 2005). Predation occurs statewide, but high mortality due to predation has especially been noted in Apalachicola Bay and the Big Bend following periods of increased salinity (Camp et al. 2015, Frederick et al. 2016).

Development: Oysters are subject to direct habitat loss as a result of coastal development, shoreline hardening, and dredging, but they are also indirectly vulnerable to diminished water quality and increased pollutants and sedimentation associated with coastal development (Frazel 2009). Hardened shorelines interrupt the transition area from upland to benthic habitat. While seawalls often provide a substrate for oyster settlement, the surface area is often smaller than that of the intertidal habitat they replace. Reflected wave energy from seawalls can also undermine potential adjacent oyster habitat. Oyster habitat has been lost to development across Florida, most notably in areas of high population density such as Tampa Bay, Sarasota Bay, Charlotte Harbor, Naples Bay, and much of southeast Florida.

Substrate loss: The physical removal of oyster reefs and associated shell through harvesting, mining, construction, or dredging reduces the overall reef footprint and available substrate for settlement of new oysters. While reefs naturally lose substrate through degradation and dissolution, ocean acidification also presents challenges for all calcifying marine and estuarine organisms and is expected to lower rates of calcification and survival while increasing shell degradation (Hofmann 2010, Waldbusser et al. 2011). Substrate loss due to mining historically was common across Florida (particularly in Tampa Bay and Charlotte Harbor). Continued loss to live harvesting remains a concern and has had a particularly detrimental impact on reef extent in Apalachicola Bay and Suwannee Sound (Camp et al. 2015, Pine et al. 2015, Kaufman 2017).

Hypoxia: While oysters can tolerate occasional exposure to low dissolved oxygen, hypoxia and anoxia decrease settlement, growth rate, and survival (Baker and Mann 1992, Johnson et al. 2009). Dissolved oxygen under 2 mg/L can cause mortality in subtidal oysters and associated reef fauna (Lenihan and Peterson 1998). Benthic hypoxia may arise when water bodies are stratified as a result of freshwater flow or limited vertical mixing (Woithe and Brandt-Williams 2006). Water bodies with limited flushing, such as the Indian River Lagoon, are also susceptible to hypoxia, particularly in warm summer temperatures when oxygen solubility is low (FDEP 2014). Areas with high sedimentation of organic matter are also prone to decomposition-induced benthic hypoxia (Volety et al. 2008). Subtidal oyster reefs with sufficient vertical relief that elevates them off the bottom are less often exposed to hypoxic conditions (Coen and Humphries 2017). Intertidal oyster reefs encounter hypoxia less often as they are periodically submerged in surface water that has higher concentrations of dissolved oxygen (Coen and Humphries 2017).

Disease and parasitism: Two protozoan diseases can cause high mortality in oyster populations. *Perkinsus marinus* causes the disease dermo and *Haplosporidium nelsoni* causes the disease MSX (Ford and Tripp 1996, Fisher et al. 1999). Dermo is present in waters throughout Florida, although typically at a low intensity (Volety et al. 2009). The disease is usually recognized as a weakening factor for oysters rather than a primary cause of mortality in Florida. Dermo may have been a contributing factor to an oyster die-off in Pensacola Bay in 1971, although there was also poor water quality at the same time (USEPA 2004). MSX, first noted in the United States in the late 1950s, is present from Maine to northeast Florida and can cause local die-offs of as much as 90% mortality (Burreson et al. 2000). MSX has never been detected in the Gulf of Mexico (Ford et al. 2011), and its presence in Atlantic Florida waters is not pathogenic (Burreson and Ford 2004, Walters et al. 2007). Infestations of boring sponges (Fig. 1.9), polychaetes, and boring mollusks can also harm oysters and make shells more vulnerable to predators or breakage; this forces the oyster to dedicate more energy toward shell repair and away from growth and reproduction (Buschbaum et al. 2007, VanderKooy 2012). Oysters are more susceptible to these diseases and parasites at higher salinity (Camp et al. 2015, Coen and Bishop 2015).

Boating impacts: Boat wakes can cause significant local damage and erosion on intertidal oyster reefs, harming both established adults and new recruits (Grizzle et al. 2002, Wall et al. 2005). Erosion of intertidal reefs and salt marshes from boat wakes is a significant problem in many parts of northeast Florida, particularly along the Intracoastal Waterway. Mosquito Lagoon oyster reefs have extensive dead margins and in some cases have been reduced to intertidal sand flats as a result of erosion from boat wakes (Grizzle et al. 2002).

Sedimentation: Excessive sedimentation due to dredging or the lack of water currents can bury oysters and impede filter feeding and respiration (Thayer et al. 2005, Coen and Humphries 2017). Many of the reefs in Naples Bay were buried due to dredging, and seismic profiling has revealed remnant reefs buried below sediment (Savarese et al. 2006). Shell removal can also increase sedimentation on reefs, which can smother remaining oysters on low-relief reefs (Berrigan et al. 1991, Breitburg 1999, Lenihan 1999). Bottom currents help protect subtidal oysters from burial in sediments or their own feces and pseudofeces.

Overharvesting: Until recently, the oyster fisheries in the Gulf of Mexico were described as one of the last remaining areas in the United States (and perhaps globally) for which oyster conservation and sustainable wild fisheries were feasible (Beck et al. 2011). Areas such as Apalachicola Bay that have historically been central to the oyster fishery along the Gulf now face significantly reduced harvests as a result of numerous stressors including salinity variability, tropical storms and hurricanes, and substrate loss (Camp et al. 2015, NASEM 2017). While overharvesting is seldom considered a primary threat to oyster populations in Florida (in the sense that harvest does not limit recruitment; NOAA 2007, FWC 2012, FWC 2013), the substrate depletion associated with harvest may constitute a form of overfishing that results in loss of essential habitat (Pine et al. 2015, NASEM 2017). The effects of fishing pressure and substrate removal are of growing concern, particularly when paired with altered hydrology and sea-level rise.

Chemical contamination: Pesticides, fungicides, and herbicides enter the estuarine environment through runoff. Herbicides or antifouling chemicals such as tributyltin (TBT) can inhibit oyster growth, cause shell thickening, increase disease abundance, or decrease disease resistance (Alzieu 1998, Fisher et al. 1999, Bushek et al. 2007). According to Mussel Watch, a U.S. program that monitors bivalve contaminants, oysters at many sites in Florida have elevated levels of arsenic, copper, mercury, or lead (Kimbrough et al. 2008). Contamination by crude oil can also have detrimental impacts on oyster health (Barszcz et al. 1978). Following the Deepwater Horizon oil spill, the densities of spat, juvenile to young adult oysters, and market-size oysters decreased in several Gulf states (Grabowski et al. 2017, NASEM 2017). However, there is no evidence that the oil spill contaminated seafood from Apalachicola Bay (Havens et al. 2013).

Competition: Invasive species such as the striped barnacle (*Balanus amphitrite*), Asian green mussel (*Perna viridis*), charru mussel (*Mytella charruana*), and pink titan acorn barnacle (*Megabalanus coccopoma*) compete against native oysters for space and resources (Boudreaux 2003, Baker et al. 2007, Yuan et al. 2016). The presence of nonnative species can decrease larval settlement and survival of juveniles (Yuan et al. 2016). When present in high densities, native oysters and nonnative mussels may also

Figure 1.10. Asian green mussels (*Perna viridis*) on an oyster reef in Tampa Bay, Florida. Photo credit: Scott Adams.

compete with each other as they grow and mature (Galimany et al. 2017). A native of the Indo-Pacific, the Asian green mussel (Fig. 1.10) has established populations on both the Gulf and Atlantic coasts of Florida (Baker et al. 2007). While cold weather can cause die-offs and restrict expansion of the mussel, the range of this invasive species is expected to grow in the southeastern United States as a result of climate change (Firth et al. 2011, Urian et al. 2011).

Climate change: Rising sea level, altered precipitation patterns, increasing temperatures, and ocean acidification all pose significant threats for oysters (Hoegh-Guldberg and Bruno 2010, Rodriguez et al. 2014).

- **Sea-level rise:** Intertidal exposure offers oysters refuge from predation, pests, and disease (Bahr and Lanier 1981). Increased submergence times and salinity lead to increased susceptibility to predation and pathogens (Shumway 1996). As the rate of sea-level rise continues to accelerate, intertidal oysters will need to migrate landward or accrete sufficient substrate if they are to keep pace with water depth (Rodriguez et al. 2014). Similarly, subtidal reefs will need to colonize substrates with higher elevations or grow vertically to maintain viable depths. Oyster reefs with balanced shell budgets and manageable stressors are the most likely to keep pace with sea-level rise, as new shell material will enable the reef to grow vertically into the space provided by the rising water (Rodriguez et al. 2014). Reduced recruitment due to sea-level rise may move oyster reefs toward a shell budget deficit, as shell loss reduces carbonate supply and hampers reef building (Waldbusser et al. 2013, Solomon et al. 2014). Sea-level rise also causes estuaries to be increasingly saline and pushes seawater further up the estuary. Depending on the shape of the estuary, this shift can decrease the geographic area for which the salinity is suitable for oyster growth. As estuaries become more saline, oysters will become more vulnerable to predation, disease, and harmful algal blooms (Petes et al. 2012, Gobler et al. 2013, Garland and Kimbro 2015).
- **Altered precipitation patterns:** Global warming alters precipitation patterns by increasing evaporation and increasing the water-holding capacity of the atmosphere (Trenberth 2011). This effect can exaggerate weather patterns, including droughts and floods. Regions in Florida, such as parts of the southeast and southwest coast, Apalachicola Bay, and Suwannee Sound, that have historically been susceptible to high salinity as a result of both low precipitation and freshwater withdrawal, will continue to be vulnerable to more extreme variations in precipitation (Kelly and Gore 2008, Petes et al. 2012, SWFWMD 2015).

- **Increasing temperatures:** Oysters in Florida are already coping with temperatures near the upper limit of their physiological tolerance. Not only are oysters more susceptible to disease in high temperatures, but they can suffocate because the solubility of oxygen decreases as temperatures rise. Increased temperatures also can change the timing and frequency of oyster spawning (Hofmann et al. 1992, Wilson et al. 2005) and reduce larval survival and settlement (Shumway 1996). Not only will increasing temperatures expose oysters to temperature extremes more often, but also a lack of critical cool temperatures during winter months may force oysters to allocate energy toward survival and reduce energy input toward growth and reproduction (Kraeuter et al. 1982, Thompson et al. 1996). High temperatures have also been shown to disproportionately affect large oysters as oxygen diffusivity decreases and disease intensity increases with body size (Forster et al. 2012, Waples and Audzijonyte 2016). This phenomenon can result in the loss of large oysters (Lehman 1974), which are disproportionately important to reproduction and shell budgets (Waldbusser et al. 2013).
- **Ocean acidification:** Ocean acidification results from an increase in atmospheric carbon dioxide which dissolves in water, reducing carbonate ion concentrations in the water column and lowering pH. These changes make it difficult for calcifying organisms such as oysters to produce shell and can enhance dissolution of existing shell material (Hofmann 2010, Waldbusser et al. 2011, Waldbusser et al. 2013). Eastern oyster larvae reared under acidic conditions have shown stunted shell growth and reduced calcium content (Miller et al. 2009).

Mapping oyster reefs

The choice of techniques used to map oyster reefs depends on the size of the area to be mapped, which can vary from an individual restored reef (typically < 1 ha) to the regional or even statewide scale (thousands of ha). Relatively small intertidal reefs can be mapped directly by walking around the perimeter of oyster beds while they are exposed at low tide with a real-time kinematic global positioning system (RTK GPS) or differential GPS (dGPS) (Gambordella et al. 2007, Baggett et al. 2014). A surveyor's measuring wheel or transect tape may also be used to measure the perimeter, length, or width of the reef. Geographic information system (GIS) software may then be used to document the location and calculate reef areal coverage or reef footprint. For larger scales, oyster reefs are mapped from georeferenced multispectral or hyperspectral imagery (Grizzle et al. 2002, Le Bris et al. 2016). These remote images may be collected at low tide using satellites, airplanes, balloons, or drones. Oyster reefs are identifiable in aerial photographs by patterns of light and dark, texture, and shape (Grizzle et al. 2002). In traditional photo-interpretation, a person visually identifies oyster reefs in aerial images. This process can be automated or semiautomated using various object-recognition software packages (O'Keife et al. 2006, SCDNR 2008). All methods of remote sensing require some ground truthing for assessing the accuracy of the mapping products, which, for intertidal reefs, may be conducted by visual validation at low tide (SCDNR 2008, Meaux 2011).

Reef identification from aerial photography can be confounded if oysters are covered in mud or intermixed with algae, seagrass, rubble, or darkly colored sediment (O'Keife et al. 2006, Vincent 2006, SCDNR 2008, Le Bris et al. 2016). The spectral signature of reflectance will also vary depending on vertical or horizontal orientation of individual oysters, the sun's angle, and seasonally variable algal growth (Vincent 2006, SCDNR 2008, Le Bris et al. 2016). In Florida, the mangrove canopy can hide fringing oyster reefs and oysters growing on mangrove roots (Fig. 1.11). Mapping based on remote imagery is only possible on reefs with a horizontal footprint. Oysters growing on mangrove roots or seawalls are seldom mapped because these peripheral habitats are difficult to see in aerial photography, but those oysters still contribute significantly to an estuary's population (Drexler et al. 2014).

Subtidal reefs can be mapped indirectly with side-scan or multibeam sonar and simultaneous acquisition of

Figure 1.11. Oysters that grow on prop roots and as fringes under mangrove canopy are difficult to map using aerial imagery. Photo credit: Kara Radabaugh.

Table 1.1. Selected habitat classification schemes including oyster reefs. See text for affiliation acronyms. (Continues next page.)

Name	Affiliation	Region	Classification scheme	Reference
Florida Land Use and Cover Classification System (FLUCCS)	FDOT	Florida	Wetlands ○ Non-vegetated • Oyster bars	FDOT 1999
System for Classification of Habitats in Estuarine and Marine Environments (SCHEME)	FWC	Florida	Reef/hardbottom ○ Mollusk reef • Bivalve reef	Madley et al. 2002
Guide to the Natural Communities of Florida	FNAI	Florida	Marine and estuarine ○ Mollusk reef	FNAI 2010
Florida Land Cover Classification System	FWC	Florida	Estuarine ○ Intertidal • Oyster bar	Kawula 2009, 2014, Kawula and Redner 2018
Sarasota County Water Quality Planning Methods Manual for Field Mapping of Oysters	Sarasota County	Sarasota and Tampa bays	Oyster habitat characterization codes ○ Shell ○ Scattered shell ○ Oyster clumps ○ Scattered oyster clumps ○ Oyster reef ○ Oyster clumps/reef ○ Mangrove apron ○ Mangrove root oysters ○ Seawall ○ Riprap ○ Pilings ○ Floating docks	Meaux 2011
South Carolina Intertidal Oyster Survey and Related Reef Restoration/ Enhancement Program	SCDNR and NOAA Coastal Services Center	South Carolina	Background (no oysters) Vertical and horizontal oysters ○ Dense oyster clusters ○ Oysters tightly clustered on rocks ○ Vertical clusters on shell ○ Vertical clusters on horizontal oysters ○ Vertical oysters on mud ○ Separate vertical clusters on mud ○ Vertical clusters on mud Horizontal oysters ○ Very few clusters on shell ○ Few live oysters on shell Washed shell	SCDNR 2008

RTK GPS or dGPS data (Allen et al. 2005, Baggett et al. 2014). Acoustic backscatter of side-scan sonar data allow for differentiation between strong and weak acoustic returns, which provide, respectively, some indication of hard and soft substrate (Preston and Collins 2003, Twichell et al. 2007). A high-quality depth finder that uses side-scan technology may also be used to detect changes in bottom type. Multibeam sonar can extract additional directional information from the acoustic return of hundreds to thousands of points simultaneously, providing a high-resolution image with wide swath coverage. Single-beam sonar methods are less expensive than multibeam, but they provide data from a narrower footprint (Twitchell et al. 2007, Grizzle et al. 2008). Tidal depth is a critical concern in mapping using acoustic methods because shallow water limits boat access and the width of the sonar swath (Preston and Collins 2003). Divers, poles, or tongs can be used to validate the presence of live subtidal oysters (Baggett et al. 2014).

In areas with high water clarity, mapping can be completed using underwater video imagery with simultaneous collection of RTK GPS or dGPS data. Underwater

Table 1.1. (Continued.)

Name	Affiliation	Region	Classification scheme	Reference
Classification of Wetlands and Deepwater Habitats of the United States	USFWS	National	Estuarine ○ Subtidal • Reef ▪ Mollusk ○ Intertidal • Reef ▪ Mollusk ● Regularly flooded ● Irregularly flooded	Cowardin et al. 1979, FGDC 2013
Coastal Change Analysis Program (C-CAP) Classification System	NOAA	National	Marine/estuarine reef ○ Mollusk reef	Klemas et al. 1993, Dobson et al. 1995
Coastal and Marine Ecological Classification Standard (CMECS)	FGDC	National	Geoform origin: biogenic ○ Geoform: mollusk reef • Fringing mollusk reef • Linear mollusk reef • Patch mollusk reef • Washed shell mound Substrate: biogenic ○ Shell substrate • Shell reef substrate ▪ Oyster reef substrate • Shell rubble ▪ Oyster rubble • Shell hash ▪ Oyster hash Biotic setting: Benthic/attached biota ○ Reef biota • Mollusk reef biota ▪ Oyster reef ○ Faunal bed • Attached fauna ▪ Attached oysters • Soft sediment fauna ▪ Oyster bed	FGDC 2012

videography can map oyster reefs with high accuracy, but it covers a small swath and is greatly restricted by low-visibility conditions (Grizzle et al. 2005, 2008). Videography may also be used in combination with ground truthing to obtain information such as ratio of live to dead oysters, mean oyster size, or density (Grizzle et al. 2005, 2008). Subtidal oyster mapping is complicated by murky water, variable water depth, limited vertical relief, and oyster reefs interspersed with multiple benthic habitats such as seagrass beds and hardbottom.

Classification of oyster reef habitats

Benthic habitat maps use a variety of classification schemes, which are often hierarchical in structure. Most maps simply group all oyster structures into one category, but oyster habitats may be further subdivided based upon characteristics such as shell density, mean size, live/dead, reef complexity, dominant species, tidal exposure, or reef height (Table 1.1, Baggett et al. 2014). This report does not distinguish between reefs of differing vertical heights and refers to all oyster structures as reefs. However, some publications may refer to an oyster structure with a relief of less than 0.5 m (1.6 ft) as an oyster bed (Beck et al. 2009, Baggett et al. 2014, Gillies et al. 2015). Oysters growing on structures, such as mangrove roots, seawalls, or pilings, have sometimes been termed aggregations (ASMFC 2007, Beck et al. 2009). Relevant statewide and national classification schemes that include classification of oyster reefs are summarized in Table 1.1 and explained in further detail below.

The Florida Land Use and Cover Classification System (FLUCCS) was created by the Surveying and Mapping Office of the Florida Department of Transportation (FDOT). The original classifications were published in 1985 (FDOT 1985) and revised in 1999 (FDOT 1999).

Florida water management districts use FLUCCS for land classifications within their districts but may modify them for their region. Relevant FLUCCS classifications include:

- **6000 Wetlands:** areas where the water table is at or near the surface of the land for a significant portion of most years
    - **6500 Non-vegetated:** Hydric surfaces lacking vegetation
        - **6540 Oyster bars**

The System for Classification of Habitats in Estuarine and Marine Environments (SCHEME) was developed for the U.S. Environmental Protection Agency (USEPA) by FWC in an effort to make a standardized, hierarchical classification system for Florida (Madley et al. 2002). Relevant SCHEME classifications include:

- **3. Reef/hardbottom:** region dominated by calcium carbonate substrate formed by reef building organisms
    - **32. Mollusk reefs:** concentration of sessile mollusks attached to a hard substrate
        - **321. Bivalve reef:** oyster reef, partially exposed at low tide

The Guide to the Natural Communities of Florida was first published in 1990 by the Florida Natural Areas Inventory (FNAI 1990) and updated in 2010. Relevant FNAI (2010) classifications include:

- **Marine and estuarine:** includes subtidal, intertidal, and supratidal zones
    - **Mollusk reef:** subtidal or intertidal area with concentration of sessile mollusks

The Florida Land Cover Classification System (Kawula 2009, updated in Kawula 2014 and Kawula and Redner 2018) was developed to create a single land cover classification scheme for Florida by integrating established classification systems. The Florida Land Cover Classification System's hierarchy is based upon other mapping schemes including the FNAI's Guide to the Natural Communities of Florida (FNAI 1990) and FLUCCS classifications (FDOT 1999). Relevant classifications include:

- **5000 Estuarine**
    - **5200 Intertidal**
        - **5230 Oyster bar**

Sarasota County developed a Methods Manual for Field Mapping of Oysters for detailed mapping in Sarasota Bay and adjacent tidal creeks (Meaux 2011). Methods are based upon FWC protocols that were used to map oyster habitat in Tampa Bay. The training manual fully describes each category with accompanying photographs and provides protocols and data sheets for mapping (Meaux 2011). Specific classifications include the following:

- **Shell (S):** single shells, usually dead, scattered densely enough along a shoreline that a person would step on shells when walking through the area
- **Scattered shell (SS):** same as above, but shells are less dense, such that a person could walk through the area without stepping on shells
- **Oyster clumps (C):** clusters of two or more oysters that are cemented together; oysters may be live or dead. Clumps are dense enough that a person would step on shells when walking through the area
- **Scattered oyster clumps (SC):** same as above, but clumps are less dense, such that a person could walk through the area without stepping on clumps
- **Oyster reef (R):** includes patch reefs and string reefs, which may or may not be attached to the mainland and may or may not include mangroves growing out of the shell substrate in the center of the reef
- **Oyster clumps/reef (CR):** central solid oyster reef surrounded by clumps or scattered clumps
- **Mangrove apron (MA):** solid oyster reef growing in a narrow band around mangroves that are growing in sediment (not on a reef substrate). May be attached to the mainland or surrounding a mangrove island. Also known as a fringe oyster reef
- **Mangrove root oysters (MRO):** oysters grow on the prop roots and drop roots of *Rhizophora mangle* (red mangrove). May be single shells or clumps
- **Seawall (SW), riprap (RR), pilings (P), or floating docks (D):** oysters grow on solid structures such as seawalls, bulkheads, and riprap rather than on bottom substrate. Thickness and vertical height of oyster aggregations are subdivided, and oysters are classified as solid or scattered along the substrate:
    - **Light:** 1 or 2 layers of oysters in a band less than 15 cm (6 in) wide
    - **Medium:** more than 1 layer of oysters in a band 15–30 cm (6–12 in) wide
    - **Heavy:** more than 1 layer of oysters in a band 30–46 cm (12–18 in) wide
    - **Very heavy:** more than 1 layer of oysters in a band >46 cm (18 in) wide
        - **Solid: s**olid stretch of oysters along the seawall or riprap
        - **Scattered:** sporadic stretch of oysters along the seawall or riprap

The South Carolina Department of Natural Resources (SCDNR) and the NOAA Coastal Services Center mapped the coastline of South Carolina using multispectral imagery (SCDNR 2008). The effort included the development of classification system for oyster reefs, largely using different spectral signatures as a result of vertical or horizontal orientation of the oysters. The project report (SCDNR 2008) describes each category with accompanying photographs and provides protocols for classifying aerial imagery. Classes and oyster strata include the following:

- **Class 1:** background (no oysters)
- **Class 2:** vertical and horizontal oysters mixed, little or no mud or washed shell
    - **Stratum A:** dense oyster clusters with little exposed dead shell or mud
    - **Stratum E:** oysters tightly clustered on rocks, may have mud or *Spartina alterniflora* (smooth cordgrass) between clusters
    - **Stratum F:** vertical clusters with spatial separation. Substrate between clusters consists of shells with few horizontal live oysters and little mud
    - **Stratum F1:** small, vertical clusters on a substrate of single, horizontal oysters. Very little exposed mud
- **Class 3:** vertical oysters on a substrate of mud with few to no horizontal oysters
    - **Stratum C:** vertical clusters with up to 1 m spatial separation. Substrate between clusters is usually mud with little surrounding shell
    - **Stratum G:** close vertical clustered oysters separated by mud with little to no shells or oysters
- **Class 4:** horizontal oysters mixed with washed shells
    - **Stratum B:** little to no vertical oysters and few clusters, oysters frequently single. Found on heavily shelled substrate in the lower intertidal zone
    - **Stratum D:** mostly horizontal dead oyster shell with little live crop, generally found in lower intertidal zone
- **Class 5:** washed shell

The Classification of Wetlands and Deepwater Habitats of the United States, developed by Cowardin et al. (1979) for the U.S. Fish and Wildlife Service, was updated by the Federal Geographic Data Committee in 2013 (FGDC 2013). Classification codes directly relevant to oyster reefs include E1RF2L, E2RF2M, and E2RF2N. Oysters that are mixed with unconsolidated substrate may be classified as E2RF2/US2N, E2RF2/USM. The hierarchical structure of these relevant classifications is as follows:

- **System: estuarine (E):** impacted by seawater and by freshwater runoff
    - **Subsystem: subtidal (1):** exposed substrate flooded by tides
        - **Class: reef (RF):** ridge or moundlike structure formed by sessile invertebrates
            - **Subclass: mollusk (2):** dominance types include *Ostrea* and *Crassostrea*
                - **Water regime: subtidal (L):** substrate continuously inundated
    - **Subsystem: intertidal (2):** exposed substrate that is flooded by tides
        - **Class: reef (RF):** ridge or moundlike structure formed by sessile invertebrates
            - **Subclass: mollusk (2):** dominance types include *Ostrea* and *Crassostrea*
                - **Water regime: regularly flooded (N)**
                - **Water regime: irregularly exposed (M)**
        - **Class: unconsolidated shore (US):** >70% cover of stones, boulders, or bedrock and <30% vegetation cover
            - **Subclass: sand (2):** unconsolidated particles predominantly sand, although particles of other sizes may be mixed in
                - **Water regime: regularly flooded (N)**
                - **Water regime: irregularly exposed (M)**

The National Oceanic and Atmospheric Administration's (NOAA) Coastal Change Analysis Program (C-CAP) uses its own classification system. The original classification system was described in Klemas et al. (1993), and an updated summary is available in Dobson et al. (1995), which also explains how the land cover categories compare to Cowardin et al.'s (1979) classes. Relevant classifications include:

- **Class: marine/estuarine reef:** ridge or moundlike structure made from sedentary invertebrates
    - **Subclass: mollusk reef**

The Coastal and Marine Ecological Classification Standard (CMECS) was created by the Federal Geographic Data Committee and the Marine and Coastal Spatial Data Subcommittee (FGDC 2012). CMECS is a hierarchical classification scheme designed to use common terminology to classify marine and estuarine habitats. CMECS includes classifications based on two settings (biogeographic and aquatic) and four components (water column, geoform, substrate, and biotic). The relevant hierarchical classifications for oyster reefs in the geoform, substrate, and biotic components are listed below.

- **Geoform origin: biogenic:** physical features created by organisms, most commonly reefs made by corals, mollusks, or worm tubes
    - **Geoform: mollusk reef:** shell reefs intermixed with channels and unvegetated flats
        - **Geoform type: fringing mollusk reef:** narrow, linear reefs; generally intertidal and lower than the marsh along tidal creeks
        - **Geoform type: linear mollusk reef:** narrow, ridgelike reefs; generally intertidal and in areas with small tidal range
        - **Geoform type: patch mollusk reef:** mounded reefs with vertical relief above surrounding substrate; usual intertidal, occasionally subtidal
        - **Geoform type: washed shell mound:** accumulations of loose, dead shell in the high intertidal zone
- **Substrate origin: biogenic substrate:** majority of substrate is of nonliving biogenic origin rather than geologic or anthropogenic origin
    - **Substrate class: shell substrate:** substrate made of shells or shell fragments; may or may not include live reef-building fauna
        - **Substrate subclass: shell reef substrate:** cemented, conglomerated, or self-adhered shell reefs with median particle size >4 m
            - **Substrate group: oyster reef substrate**
        - **Substrate subclass: shell rubble:** shells with a median particle size of 0.064–4 m; may be loose, cemented, or conglomerated
            - **Substrate group: oyster rubble**
        - **Substrate subclass: shell hash:** loose shell, broken or whole, with median particle size of 2–64 mm
            - **Substrate group: oyster hash**
- **Biotic setting: benthic/attached biota:** biota live on or in the substrate
    - **Biotic class: reef biota:** reef-building fauna construct biogenic substrates
        - **Biotic subclass: mollusk reef biota:** living and dead mollusks or gastropods aggregate and attach in sufficient numbers to make a substrate
            - **Biotic group: oyster reef:** mounds or ridges formed by live oysters cementing to the substrate of live and dead conspecifics
                - **Biotic communities: *Crassostrea* reef, *Ostrea* reef**
    - **Biotic class: faunal bed:** seabed dominated by benthic fauna that have not created a reef
        - **Biotic subclass: attached fauna:** dominated by fauna that maintain contact with a hard substrate
            - **Biotic group: attached oysters:** oysters attach to a hard substrate other than conspecifics
                - **Biotic communities: attached *Crassostrea*, attached *Ostrea***
        - **Biotic subclass: soft-sediment fauna:** sand or mud with dominant presence of infauna, epifauna, or mobile fauna that create burrows
            - **Biotic group: oyster bed:** sand or mud with low densities of oysters that are not attached to a hard substrate.
                - **Biotic communities: *Crassostrea* bed, *Ostrea* bed**

Recent oyster mapping data in Florida

Oyster mapping data sets in Florida are often limited to a specific estuary or region. The FWC and OIMMP have combined many of these maps to create a statewide oyster map. This GIS shapefile is updated periodically and was used to create the maps in this report. The shapefile is available for download at http://geodata.myfwc.com/datasets/oyster-beds-in-florida. A listing of selected large-scale data providers, including the FWC compilation, is compiled in Table 1.2 and summarized in further detail below. These and other smaller-scale mapping efforts are described in the regional chapters of this report. While land classification schemes vary across agencies (Table 1.1) and may subdivide different types of oyster habitat, many maps simply plot oyster extent using one category. Land cover maps vary widely among agencies due to variable classification schemes, image resolution, and minimum mapping units. Oyster reef maps are subject to similar variability but also suffer from gaps as mapping efforts are generally regional and often focus on either subtidal or intertidal reefs.

For more than 30 years, the National Wetlands Inventory (NWI) has generated and updated highly detailed wetland maps following Cowardin et al.'s (1979) classification scheme using a variety of methods and data sources, including aerial images (Dahl et al. 2015). Most recently, NWI maps are available online at http://www.fws.gov/wetlands/index.html. The effort focused primarily on wetlands, but mollusk reefs are included in available maps. Not all bays in the state with oyster reefs included labeled mollusk reefs under the NWI scheme.

The Florida Water Management Districts (WMDs) periodically complete their own assessments of land use and land cover (LULC) in their jurisdictions. Land-cover analysis is based on remote imagery using FLUCCS categories (FDOT 1999) and does not always include oyster reefs. The Northwest Florida Water Management District (NWFWMD) sometimes includes oyster reefs (FLUCCS code 6540) in their LULC maps. The most recent Suwannee River Water Management District (SRWMD) LULC map that includes oyster reefs is from 2010. While LULC data are available from 2013–2014, this map does not include an oyster category. SRWMD also conducted an extensive oyster mapping effort in 2001 (Patterson 2002). LULC data are available on the water management district websites (Table 1.2).

The Southwest Florida Water Management District (SWFWMD) conducts periodic seagrass and oyster mapping within its district boundaries 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 bars, beaches, patchy seagrass, and continuous seagrass. Dead and live oysters were mapped together to form the oyster bar classification. Map files may be downloaded from the district website https://data-swfwmd.opendata.arcgis.com/.

The St. Johns River Water Management District (SJRWMD) mapped live and dead oyster reefs within the Northern Coastal Basin of Florida with the use of aerial photographs and a custom photo-interpretation key of oyster reef types. There was no minimum mapping unit. Field verification determined that the maps were 96% accurate. The data set is available for download at http://data-floridaswater.opendata.arcgis.com/.

FWC created the Oyster Beds in Florida GIS data set by compiling mapping data from a variety of sources. This map is a compilation of multiple studies and methodologies and was greatly expanded upon by OIMMP. Sources of data include the WMDs, FWC, U.S. Geological Survey (USGS), and university and city mapping efforts for seagrass, oysters, or general benthic habitat mapping. The data set is regularly updated and is available for download at http://geodata.myfwc.com/datasets/oyster-beds-in-florida. Although this is the most comprehensive oyster map available for Florida, gaps remain. There is need for updated detailed mapping in the panhandle (Perdido, Pensacola, Choctawhatchee, and St. Andrew bays), Big Bend and Springs Coast (for Apalachee Bay and subtidal oysters), much of the Everglades, and the Indian River Lagoon (outside of its tributaries).

The Cooperative Land Cover Map (CLC) is a collaboration between FNAI and FWC to support the goals of the Florida Comprehensive Wildlife Conservation Strategy (FNAI and FWC 2010). The CLC project compiles data from various sources and integrates them using aerial photography and local data collections. Data were obtained from Florida WMD LULC data, local mapping efforts, aerial photographs, and interviews with local experts (FNAI and FWC 2010). Each data set is assigned a confidence category to determine which data set takes precedence over other data sets with conflicting maps. Due to the diverse array of data sources, multiple land classification systems are used (FNAI 1990, FDOT 1999, Kawula 2014, and others). All classifications are related to the Florida Land Cover Classification System (Kawula 2009). Mapping layers are updated approximately every six months and can be downloaded at http://myfwc.com/research/gis/applications/articles/Cooperative-Land-Cover.

The Gulf of Mexico Data Atlas (http://gulfatlas.noaa.gov/), created by NOAA, compiles data from other sources. The oyster mapping layer for Florida is compiled from sources such as the water management districts, USGS, FWC, The Nature Conservancy, and the National Estuarine Research Reserves in the state. The data atlas includes an online mapping program that enables the viewing of maps for distribution of oysters and other invertebrates.

Oyster reef monitoring

Oyster monitoring in Florida is performed by a variety of state and local governments, water management districts, preserves, reserves, universities, and nongovernmental organizations. The goals of these efforts include monitoring the efficacy of the Comprehensive Everglades Restoration Program (CERP) (Volety et al. 2009), the health of oyster fisheries (FDACS 2012), the success of restoration efforts (Brumbaugh et al. 2006), identifying long-term changes (Seavey et al. 2011), and providing general ecological assessments (Garland and Kimbro 2015). Examples of protocols for oyster monitoring are cited in Table 1.3. Many focus specifically on monitoring restored reefs.

Monitoring parameters

Recommended universal monitoring metrics for oyster reef restoration efforts include reef areal dimension, reef height, oyster density, tidal emersion, and oyster size-frequency distribution (Baggett et al. 2014, Walles et

Table 1.2. Selected large-scale providers of oyster reef data in Florida. See text for affiliation acronyms.

Program	Affiliation	Region of map extent, live reef area mapped in Florida	Data origin, most recent data	Classification scheme	Website
National Wetlands Inventory (NWI)	USFWS	national, 197 ha/488 ac	Composite of multiple data and aerial image sources, image years vary from 1970s to 2010s	Cowardin et al. 1979	http://www.fws.gov/wetlands
Florida water management districts land use land cover (LULC) maps	NWFWMD	NWFWMD, 124 ha/306 ac	Color infrared or true color aerial photography, 2009–2010	FDOT 1999	https://www.fgdl.org/metadataexplorer/explorer.jsp
	SRWMD	SRWMD, 75 ha/185 ac	Color infrared or true color aerial photography, 2010–2011	FDOT 1999	http://www.srwmd.state.fl.us/319/Data-Directory
SRWMD oyster mapping	SRWMD	SRWMD, 590 ha/1,457 ac	Composite of multiple data and aerial image sources, 2001	customized FDOT 1999	http://www.srwmd.state.fl.us/319/Data-Directory
SWFWMD seagrass mapping	SWFWMD	SWFWMD, 1,330 ha/3,286 ac	Color aerial photography, oysters included in seagrass mapping efforts, 2016	customized FDOT 1999	http://data-swfwmd.opendata.arcgis.com/
Northern Coastal Basin Intercoastal Oysters	SJRWMD, UCF	Northeast Florida, 589 ha/1,456 ac	Color aerial photography, 2009–2016	custom classification	http://data-floridaswater.opendata.arcgis.com/
Oyster beds in Florida	FWC	Florida, 7,923 ha/19,579 ac	Compilation of many sources, see metadata. Source years vary; updated regularly	FDOT 1999 and others	http://geodata.myfwc.com/datasets/oyster-beds-in-florida
Cooperative land cover (CLC) map	FNAI, FWC	Florida, 235 ha/585 ac	Compilation of many sources, see metadata. Version 3.3 published 2018	FNAI 1990, FDOT 1999, Kawula 2014, and others	FNAI and FWC 2010 http://myfwc.com/research/gis/applications/articles/Cooperative-Land-Cover
Gulf of Mexico Data Atlas	NOAA	Gulf coast and east coast of Florida 6,906 ac	Compilation of many sources, see metadata. Source years vary; published 2011	FNAI 1990, FDOT 1999, and others	http://gulfatlas.noaa.gov/catalog/living-marine/

al. 2016). A variety of other parameters is also used, depending on objectives and goals of the monitoring. These parameters are briefly described below; see cited references for further detail.

Reef areal dimensions and footprint include the area of the reef (the summed area of patches of living and nonliving oyster shell or other substrate material) and reef footprint (entire area of the reef complex, including gaps between small patch reefs) (Baggett et al. 2014). Alternatively, data may be collected on the percent cover of oysters within the footprint (Coen et al. 2004). Data on reef area is collected with the same methodologies used to map oyster reefs (see previous section), which includes walking the perimeter of the reef with an RTK GPS or dGPS, use of aerial or underwater imagery, or use of side-scan or multibeam sonar.

Reef height and reef depth provide information on reef accretion and stability and offer an indicator of the reef's utility as habitat for associated species (Baggett et al. 2014). Subtidal reefs that are sufficiently elevated above the bottom substrate tend to be less vulnerable to hypoxia and sedimentation (Lenihan and Peterson 1998, Coen et al. 2004). A high-precision GPS unit or traditional surveying equipment may be used to determine intertidal reef elevation and topography (Baggett et al. 2014). Subtidal reefs can be assessed using side-scan sonar across a reef to determine the reef's relief and water depth. A sounding pole may also be used at intervals across a subtidal reef to determine variation in elevation of a given reef footprint.

Tidal emersion: The length of time portions of a reef are exposed to air at low tide can result in clear zonation in oyster development, performance, and ecosystem services (Walles et al. 2016, Hanke et al. 2017). Tidal emersion can be determined from temperature loggers or water-level gauges. It can also be assessed with the use of bathymetric or topographic maps, or the elevation of the top of the reef can be measured using an RTK GPS or dGPS. The elevation can then be converted to emersion time based on local sea level and tidal cycles (Rodriguez et al. 2014, Walles et al. 2016).

Oyster density is determined by counting live individuals of a particular size within quadrats on an oyster reef (0.25-m^2 quadrats recommended in high oyster densities, 1-m^2 in low densities) (Baggett et al. 2014). If necessary, the oysters in a given quadrat should be excavated to a depth of 10–15 cm to allow the counting of live oysters and articulated shells. Articulated shells, also called boxes, often indicate recent mortality (Christmas et al. 1997). Percent cover by live oysters can also be determined using a point–intercept method in a grid within a quadrat (Fig. 1.12). Grizzle et al. (2005) paired underwater videography with divers excavating oysters in grids in order to evaluate the accuracy of estimating live oyster counts from video stills. This method was found to work best in regions of low oyster density that did not have large numbers of spat or juveniles or a lot of dead shell.

Oyster size-frequency distribution is determined by using a ruler or caliper to measure the shell height of a subset of the oysters in a quadrat (Galtsoff 1964). A digital caliper system that wirelessly inputs data directly into a computer can also be used to efficiently measure a large number of oysters (Coen et al. 2004). The same set of oysters can generally be used for both density and size-frequency measurements (Coen et al. 2004, NASEM 2017). Baggett et al. (2014) recommend measuring at least 50 oysters per sample (or 250 oysters per reef). Size-frequency metrics can be used to gauge recruitment, to track a cohort over time, or to compare the age (size) structures of restored and natural reefs. Note that this can be difficult to implement on reefs with high recruitment or a high density of oysters.

Figure 1.12. Researchers use quadrats to assess cover of live and dead oysters. Photo credit: Kara Radabaugh.

Settlement can be monitored using arrays of replicate ceramic tiles, shells, or other materials appropriate for colonization to determine recruitment of spat (Figs. 1.13 and 1.14). Regular collection of these materials and counting the spat that have settled on the surfaces enables determination of the seasonal timing and rate of oyster settlement (Brumbaugh et al. 2006). On subtidal reefs with significant relief, separate measurements of oyster spat densities on different areas (e.g., the reef crest, slope, and base) can provide information on recruitment variability with depth (Lenihan 1999, Brumbaugh et al. 2006, Hanke et al. 2017).

Salinity, dissolved oxygen, and temperature are the three environmental metrics universally recommended for inclusion in an oyster monitoring plan (Baggett et al. 2014, NASEM 2017). These water quality metrics should be monitored continuously with automated sondes, and data should be collected as close to the reef as possible. Automated sampling is recommended because water quality measurements during infrequent site visits (weekly or monthly) provide only a snapshot of local conditions. These data are not very helpful for assessing impact of these parameters on growth, survival, or diseases. Estuarine water-quality parameters vary widely with tides, seasons, winds, and rainfall (see http://recon.sccf.org/ for real-time water-quality data associated with a number of oyster restoration efforts). Additional parameters such as total suspended solids, chlorophyll *a*, and water clarity also aid in ecosystem-wide water quality assessments (Brumbaugh et al. 2006). For intertidal reefs, air temperature at low tide should also be measured.

Condition index: The oyster condition index provides a method of comparing oyster condition across multiple locations (Lawrence and Scott 1982, Crosby and Gale 1990,

Table 1.3. Selected monitoring protocols for natural oyster reefs or shellfish restoration projects. See text for affiliation acronyms.

Name	Affiliation	Focus	Reference
Design and monitoring of shellfish restoration projects	The Nature Conservancy	Instructional guide for bivalve restoration projects and monitoring	Brumbaugh et al. 2006
Oyster habitat restoration monitoring and assessment handbook	NOAA, TNC, University of South Alabama, Florida Atlantic University	Instructional guide for monitoring and characterization of oyster restoration sites	Baggett et al. 2014
Science-based restoration monitoring of coastal habitats	NOAA	Volume 1: A framework for monitoring plans under the estuaries and clean waters act of 2000; Volume 2: Tools for monitoring coastal habitats	Thayer et al. 2003, Thayer et al. 2005
Best management practices for shellfish restoration	Interstate Shellfish Sanitation Conference, TNC, NOAA	Methods for shellfish restoration including community outreach and harvesting concerns	Leonard and Macfarlane 2011
Restoration goals, quantitative metrics and assessment protocols for evaluating success on restored oyster reef sanctuaries	Chesapeake Bay Program	Monitoring protocols and success metrics for restored oyster reefs	Oyster Metrics Workgroup 2011
Effective monitoring to evaluate ecological restoration in the Gulf of Mexico	National Academies of Sciences, Engineering, and Medicine	General and specific guidelines for monitoring numerous restored habitats, including oyster reefs	NASEM 2017
Oyster condition assessment protocol	UCF, SJRWMD, GTMNERR, NE Florida Aquatic Preserves	Instructional guide for standardized oyster reef monitoring	Walters et al. 2016
A South Carolina Sea Grant report of a 2004 workshop to examine and evaluate oyster restoration metrics to assess ecological function, sustainability, and success	South Carolina Sea Grant	Site selection parameters and metrics to assess reef restoration efforts	Coen et al. 2004
Sampling and analytical methods of the national status and trends program national benthic surveillance and Mussel Watch projects	NOAA Mussel Watch, national status and trends program	Chemical contamination monitoring for organic and inorganic contaminants in bivalves and sediment	Lauenstein and Cantillo 1993

Baggett et al. 2014). Condition index (CI) is calculated as follows (Crosby and Gale 1990, Baggett et al. 2014):

$$CI = (\text{tissue dry weight} \times 100) / (\text{whole wet weight} - \text{shell wet weight})$$

This dry-to-wet-weight ratio can provide a metric of the proportion of water in the tissue of a given oyster. A high amount of water within the tissue is a sign of depleted energy reserves (as occurs after spawning) or food limitation (Lucas and Beninger 1985, Rheault and Rice 1996). At least 25 oysters should be used to determine oyster condition at a location; the same oysters used for the size and density measurements as described above can be used for this purpose as well.

Oyster growth and survival can be determined by placing premeasured oysters in trays, mesh bags, or cages, and placing them back out on the reef. These oysters are then tracked for growth and survival over time (e.g., Kingsley-Smith et al. 2009). Comparison of survival in closed

Figure 1.13. Example spat on the interior side of an oyster shell. Photo credit: Christine Russo.

Figure 1.14. Suspended clean oyster shells are used to collect spat and assess oyster settlement. Photo credit: Kara Radabaugh.

vs. open cages allows for determination of predation on oysters of various sizes.

Oyster disease: Monitoring for the presence, frequency, and severity of diseases such as dermo and MSX can be achieved by collecting and examining 20–25 oysters per location (Coen et al. 2004). Dermo infections can be diagnosed by using Ray's fluid thioglycolate method (Ray 1952, Bushek et al. 1994, Dungan and Bushek 2015). In this method, oyster tissue is incubated in Ray's fluid thioglycolate medium, stained with iodine, and then examined for parasites under a microscope. The intensity of a dermo infection is scored on a scale of 0 to 5, where 0 indicates no infection and 5 indicates that protist density almost entirely obscures the oyster tissue (Fig. 1.15; Mackin 1962). Frequency of disease monitoring should be tailored to seasonal and annual variability of a given location. In some cases, seasonal variability of disease prevalence necessitates a sampling frequency of 4–5 times per year (Coen et al. 2004).

MSX is not found on the Gulf coast of Florida (Ford et al. 2011) and has not shown pathogenicity on the east coast of Florida (Burreson and Ford 2004, Walters et al. 2007). Disease monitoring may therefore be necessary only if disease prevalence is a problem in an area or there is unexplained high mortality (Baggett et al. 2014). MSX is more difficult to detect than is dermo and can be determined using the paraffin histology method (Burreson et al. 1988, Burreson and Ford 2004) or by polymerase chain reaction (PCR) amplification (Stokes et al. 1995), but suspected infections should be verified with histology (Burreson 2008).

Chemical contamination: Oysters are useful indicators of water quality and pollution because, as sessile filter feeders, their tissues provide a record of water quality and they can be used to quantify spatial variation in contaminant levels. Compounds of interest include polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), pesticides, and heavy metals. The methods used by NOAA's Mussel Watch program to monitor organic contaminants and trace elements in bivalve tissue and sediments are summarized in Lauenstein and Cantillo (1993), and two decades of results are summarized in Kimbrough et al. (2008) and Kim et al. (2008).

Monitoring of associated species: The presence and diversity of transient and resident species on oyster reefs provide an indicator of ecosystem status and function (Tolley et al. 2006, Coen et al. 2007). The biomass, abundance, and diversity of the fish and invertebrates that live near the oyster reef can be assessed with various types of nets (lift nets, drop nets, seines, gill nets, etc.), traps, embedded sampling trays, and visual surveys (Brumbaugh et al. 2006, ASMFC 2007, zu Ermgassen et al. 2016, Hanke et al. 2017, NASEM 2017). Animals may also be collected when shells are excavated during an oyster density survey; resident organisms such as crabs, mollusks, and other invertebrate species can be sampled in this way. While monitoring associated fauna may be time consuming and require significant knowledge of taxonomy, the resulting data are valuable for understanding the ecology of the reef (Coen et al. 2004, Tolley et al. 2005, Coen and Humphries 2017). In the case of restored reefs, faunal monitoring may focus on similarity of species composition with adjacent natural reefs (Walters and Coen 2006). Seagrass surveys may also be of interest after oyster restoration efforts, because enhanced water clarity as a result of oyster reefs has been noted to increase seagrass productivity (Peterson and Heck 2001a, 2001b, Newell and Koch 2004).

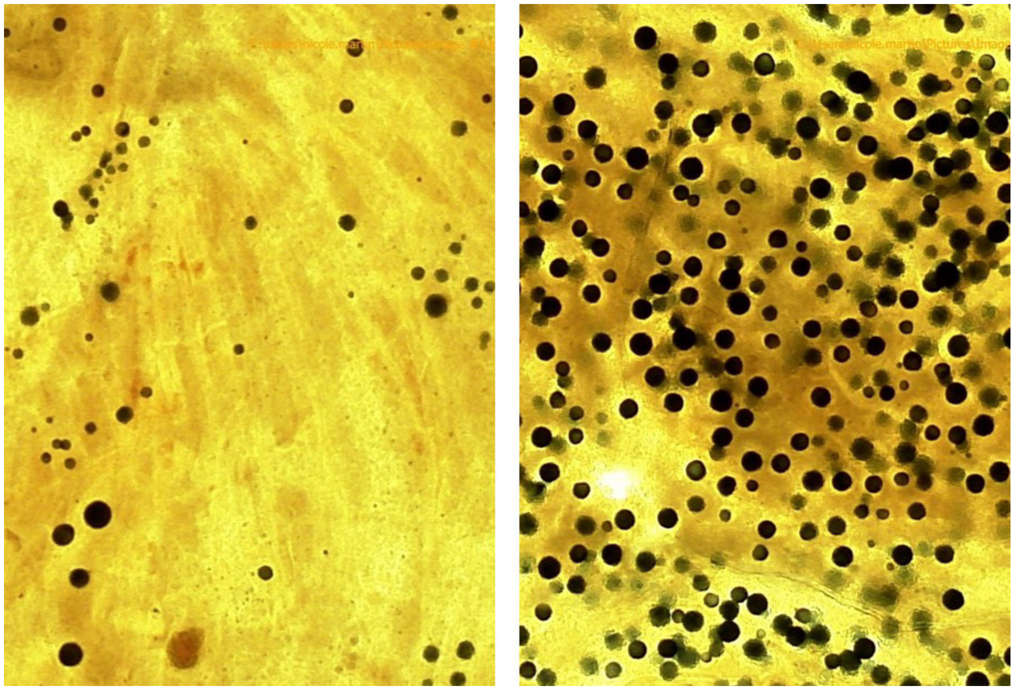

Figure 1.15. Example of dermo infections in oyster tissue. The intensities of these infections were classified as a three (left) and five (right) on the Mackin (1962) scale. Photo credit: Nicole Martin.

Restoration monitoring: Monitoring restoration sites calls for special consideration of sampling design (NASEM 2017). Monitoring data (physical and biological) collected before restoration activities begin allows for evaluation of the suitability of the habitat and its hydrology (Thayer et al. 2005, Coen and Humphries 2017). Frequent postrestoration monitoring of survival and erosion rates allows for early assessment of restoration success or any needed improvements (Baggett et al. 2014, NASEM 2017). Use of a before-after-control-impact (BACI) sampling design, which includes monitoring both the oyster restoration site and a control site before and after the restoration effort, enables identification of change as a result of restoration efforts rather than environmental factors (Thayer et al. 2005, Baggett et al. 2014, NASEM 2017). An interesting direction in restoration monitoring includes the estimation of ecosystem services derived from natural and restored oyster reefs. These can include production of fish and invertebrates, shoreline protection, or reduction of nutrients (Peterson et al. 2003, Grabowski et al. 2012, zu Ermgassen et al. 2016). Using an easily accessible tool (see http://oceanwealth.org/tools/oyster-calculator/), one can even calculate filtering capabilities of potential oyster habitat by area, estuary volume, residence time, and other variables.

Region-specific chapters

The remainder of this report documents region-specific ecosystems, monitoring, and mapping programs for oyster reefs across Florida. The eight OIMMP regions are separated as shown in Fig. 1.16. Each chapter includes a general introduction to the region, mapped oyster reefs, oyster harvesting records, location-specific threats to oyster reefs, a summary of selected mapping and monitoring programs, and recommendations for management, monitoring, and mapping efforts.

General references and additional information

OIMMP resources and workshop presentations: http://ocean.floridamarine.org/OIMMP/

FWC compilation of oyster maps in Florida: http://geodata.myfwc.com/datasets/oyster-beds-in-florida

FWC eastern oyster information: http://myfwc.com/research/saltwater/mollusc/eastern-oysters/

Commercial fisheries landings in Florida: https://myfwc.com/research/saltwater/fishstats/commercial-fisheries/landings-in-florida/

Florida saltwater fishing regulations: https://myfwc.com/fishing/saltwater/

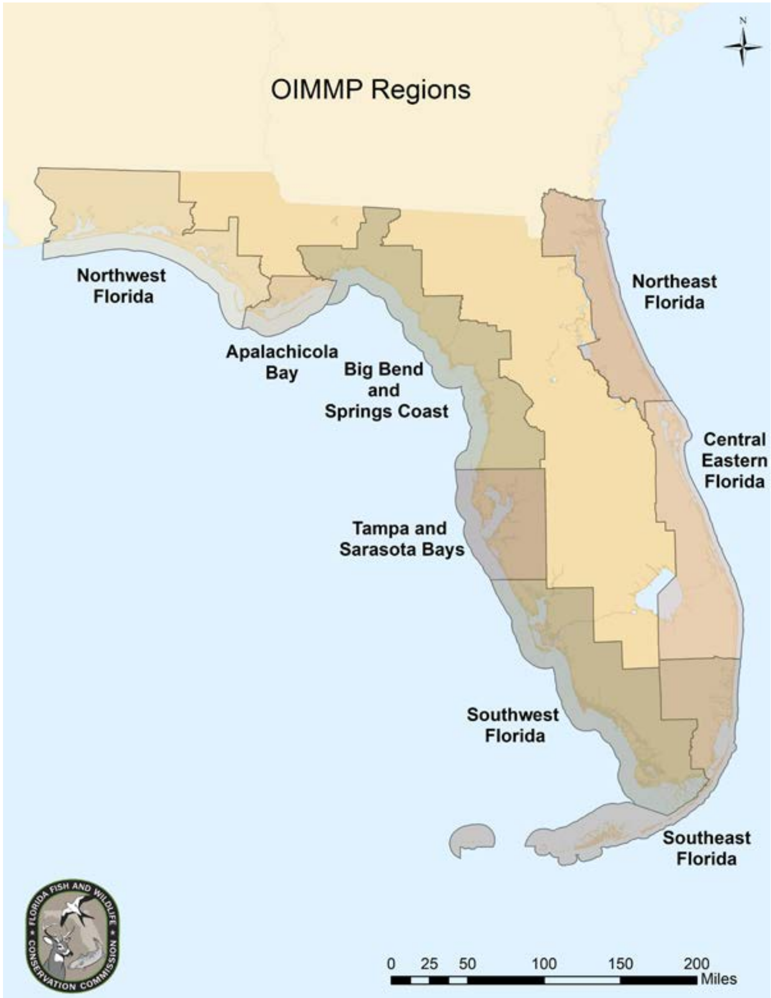

Figure 1.16. Regions of focus for the OIMMP report chapters.

Oyster restoration workgroup:
http://www.oyster-restoration.org/

NOAA's National Status and Trends program (includes Mussel Watch): https://products.coastalscience.noaa.gov/collections/ltmonitoring/nsandt/default.aspx

NOAA Chesapeake Bay Office: technical aspects of oyster restoration: https://chesapeakebay.noaa.gov/oysters/technical-aspects-of-oyster-restoration

NOAA Chesapeake Bay Office: oyster substrate literature review: https://chesapeakebay.noaa.gov/habitats-hot-topics/oyster-reef-alternative-substrate-literature-review

The Nature Conservancy's oyster calculator for water filtration and fish production provided by oyster reefs: http://oceanwealth.org/tools/oyster-calculator/

Shellfish Reef Restoration Network: https://www.shellfishrestoration.org.au/

Chesapeake Bay Foundation eastern oyster information: http://www.cbf.org/about-the-bay/more-than-just-the-bay/chesapeake-wildlife/eastern-oysters/

University of Maryland oyster hatchery information: http://hatchery.hpl.umces.edu/

Oyster Recovery Partnership:
https://oysterrecovery.org/

North Carolina Coastal Federation oyster information: https://www.nccoast.org/protect-the-coast/restore/oyster-habitat/

North Carolina Oyster Blueprint: oyster restoration, education, and research information:
https://ncoysters.org/

The Nature Conservancy: restoring North Carolina's oysters: https://www.nature.org/ourinitiatives/regions/northamerica/unitedstates/northcarolina/explore/oyster-reef-restoration-in-north-carolina-1.xml

Sink Your Shucks oyster recycling program:
http://oysterrecycling.org/

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Chapter 2
Northwest Florida

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Description of the region

Northwest Florida contains numerous barrier islands and peninsulas as well as five large bays (Fig. 2.1). The coast along the Gulf of Mexico is composed of sandy dunes and beaches, while salt marshes and tidal flats are commonly found in the estuaries protected by barrier islands. Hardened shorelines associated with urbanized areas are much less common in northwest Florida than in other regions of the state. Bays with moderate salinity provide habitat for eastern oysters (*Crassostrea virginica*), which are found in both subtidal and intertidal reefs. Eastern oysters thrive in a salinity range of 14 to 28; while they can briefly tolerate salinity outside this range, prolonged exposure can harm both subtidal and intertidal populations (Shumway 1996, Baggett et al. 2014, Coen and Bishop 2015). In high salinity, eastern oysters are vulnerable to predation and disease while at low salinity they have low rates of survival and reproduction. Crested oysters (*Ostrea stentina*) are present in higher salinity and do not generally create reef habitat.

Shellfish harvesting is prohibited in Perdido Bay. Pensacola, Choctawhatchee, St. Andrew, and St. Joseph bays all have areas of approved or conditionally approved harvest (Fig. 2.2). Historical harvests across the region are

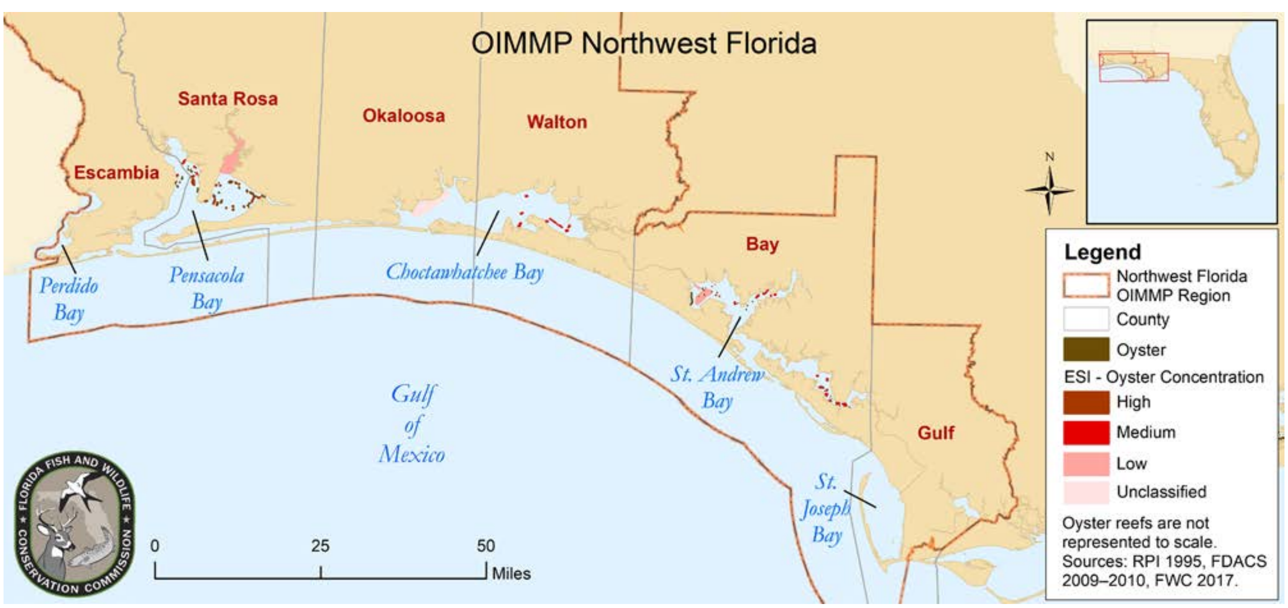

Figure 2.1. Mapped oyster extent in the northwest region of Florida.

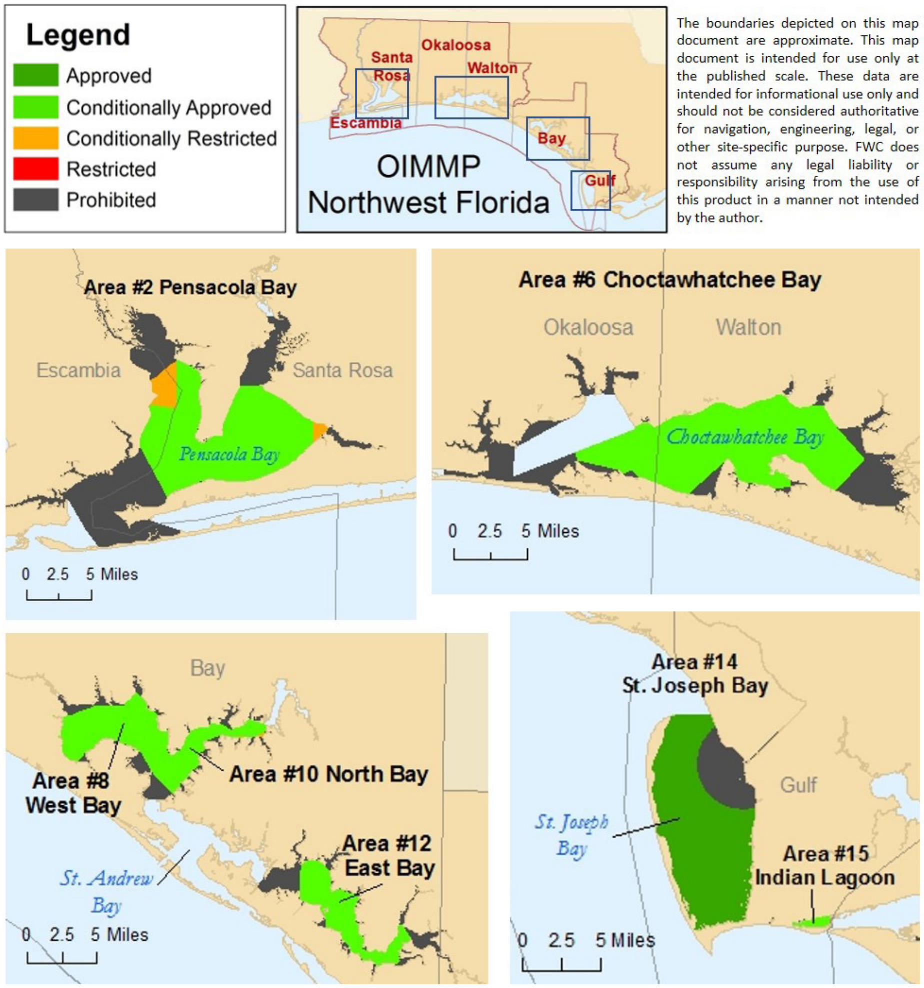

Figure 2.2. Shellfish Management Areas in the northwest region of Florida. Data source: FDACS 2017.

comparatively much lower than in neighboring Apalachicola Bay. East Pensacola Bay in Santa Rosa County and St. Andrew Bay in Bay County have provided the majority of commercially harvested oysters within the region (Fig. 2.3; FWC 2018).

Perdido Bay

Located on the border between Florida and Alabama, Perdido Bay receives freshwater flow from the Perdido River as well as other smaller rivers and creeks (Fig. 2.4). Sediment in the bay ranges from firm sand to soft mud (NWFWMD 2017a). Water quality issues include heavy metal pollution, high amounts of fecal coliform bacteria, and low dissolved oxygen (NWFWMD 2017a). The National Shellfish Sanitation Program categorizes Perdido Bay as an unclassified water, thus shellfish harvesting is prohibited, and the bay is not surveyed or mapped for oyster reefs (DWH NRDA Trustees 2017). There are no known continuous oyster reefs, but oysters do grow on piers, pilings, and rip rap (Beck and Odaya 2001, DWH NRDA Trustees 2017).

Pensacola Bay

The Pensacola Bay System includes Big Lagoon, Santa Rosa Sound, Pensacola Bay, Blackwater Bay, East Bay, and Escambia Bay (Fig. 2.5). The bay system is mostly enclosed by barrier islands. The average tidal range is 0.5 m (1.6 ft), and the main source for tidal exchange is through Pensacola Pass to the Gulf of Mexico, leading to low flushing and a long water residence time (USEPA 2004). Additional tidal connections include western Big Lagoon (which connects to Perdido Bay via the Intracoastal Waterway, ICW) and eastern Santa Rosa Sound (which connects to Choctawhatchee Bay). Upland forests are the dominant land cover within the watershed, with smaller areal extent occupied by agriculture and urban development including the city of Pensacola (FDEP 2012). The bottom of the bay is predominantly sandy in the lower bay, transitioning to silty clays in the upper region of the estuary (USEPA 2004).

Pensacola Bay provides appropriate salinity and temperature ranges for oyster habitat. Salinity in the upper

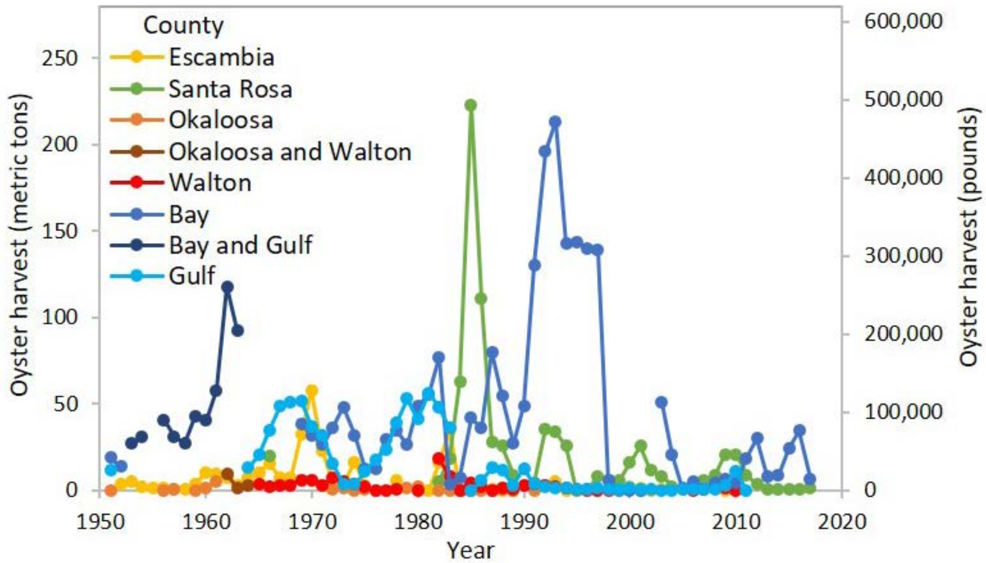

Figure 2.3. Commercial oyster landings in counties in the northwest region of Florida. Data sources: summary of Florida commercial marine fish landings (see Appendix A) and FWC 2018. Oyster landings prior to 1986 were collected under a voluntary reporting system.

part of the Pensacola Bay System ranges from 5–18, while salinity in the lower bay ranges from 18–30 (USEPA 2004). There are an estimated 95–99 ha (235–245 ac) of oyster reef within the Pensacola Bay system (Lewis et al. 2016); the majority of these reefs are located in East Bay. Water is

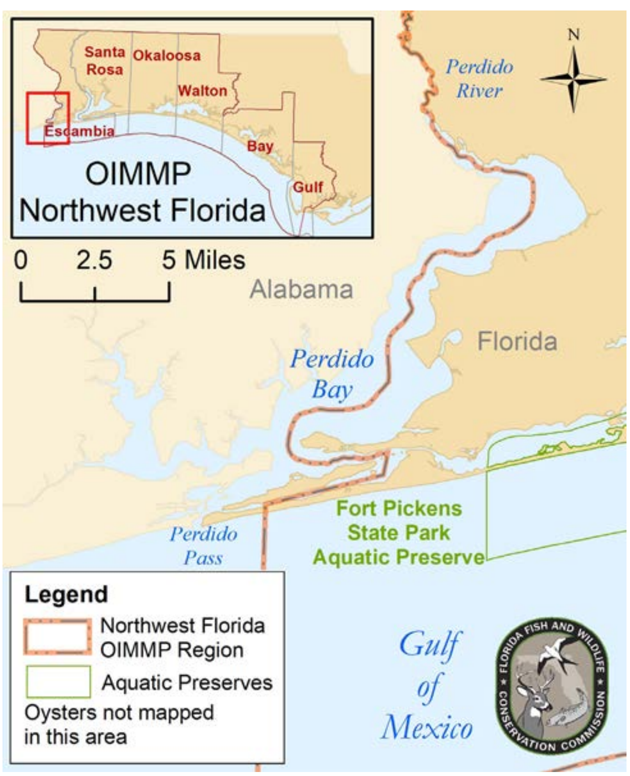

Figure 2.4. Perdido Bay and surrounding water bodies. There are no mapped oysters in Perdido Bay.

shallow in areas of Escambia Bay and East Bay where reefs are located (average depth 3 m/10 ft) and the water column is often stratified with a halocline present (FDEP 2012).

From the 1950s through the 1970s, Pensacola Bay faced water quality challenges including fish kills and algal blooms due to high-nutrient wastewater discharge. Oyster populations declined during the 1960s–1980s due to poor water quality, low salinity resulting from heavy rainfall, a lack of suitable hard substrate due to dredging, sediment contamination, and dermo (*Perkinsus marinus*) infections (USEPA 2004, Lewis et al. 2016, NWFWMD 2017b). Dermo infections contributed to the loss of more than 90% of oysters in 1971 (USEPA 2004). Compared to 1960 acreage, oyster reef area in Pensacola Bay has declined by 72% (a loss of 190–255 ha/470–630 ac) (Lewis et al. 2016). Water quality in the bay improved significantly since the passage of the Clean Water Act in the 1970s and the implementation of best land-use practices in the watershed. However, concerns remain high for sedimentation, excess nutrients, and water clarity near Pensacola and other urban areas (USEPA 2004, FDEP 2012). Oyster habitat restoration has been successful in several areas in the Pensacola Bay System, but the oyster population has been slow to recover following improvements to water quality due to lack of suitable substrate, disease, and natural variation in salinity and predation (USEPA 2004, Lewis et al. 2016). Escambia County used to have high oyster annual yields that peaked at 63 metric tons (140,000 pounds) in 1970, but reefs have been slow to recover following the die-offs of the 1970s (Fig. 2.3; Col-

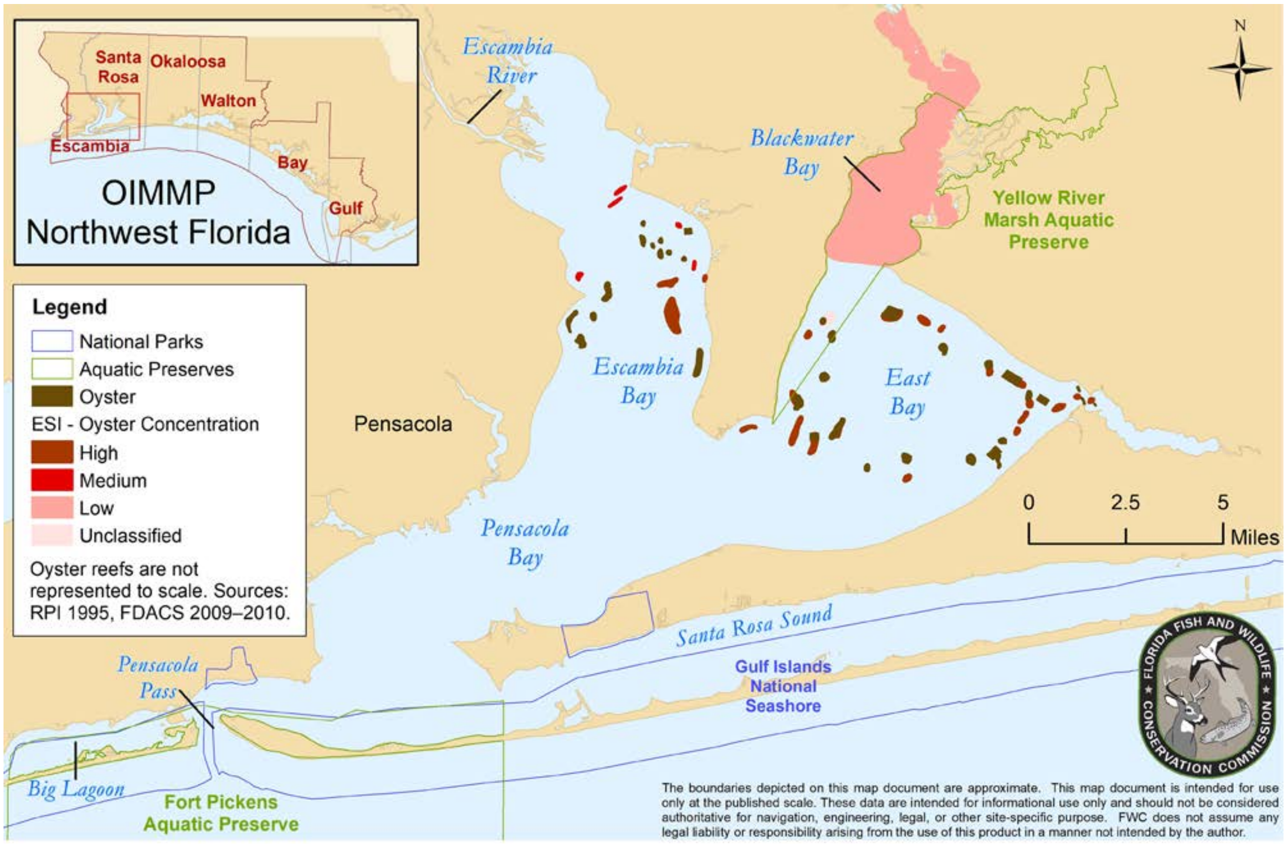

Figure 2.5. Mapped oyster extent in the Pensacola Bay system. Oyster mapping sources: RPI 1995 (from 1995 Environmental Sensitivity Index) and FDACS 2009–2010 (from navigation charts and local knowledge). Further description of mapping efforts in mapping section below.

lard 1991, USEPA 2004). Oyster landings in Santa Rosa County briefly peaked in the 1980s (Fig. 2.3).

Choctawhatchee Bay

Choctawhatchee Bay (Fig. 2.6) receives freshwater flow from the Choctawhatchee River, several smaller creeks, and groundwater from the Floridan aquifer system (NWFWMD 2017c). There is also a limited exchange of water with Santa Rosa Sound to the west and with St. Andrew Bay to the east through the ICW. As a result of limited hydrological connection with the Gulf of Mexico, the bay has a small tidal prism and limited flushing. Salinity in the bay varies widely depending on river input. Salinity is lowest in the eastern half of the bay near the Choctawhatchee River, and the bay is frequently stratified with a halocline present (Ruth and Handley 2007). Benthic substrate in the bay primarily includes sand, mud, seagrass beds, and scattered oyster reefs (NWFWMD 2017c).

Choctawhatchee Bay hosted variable oyster populations in the past; oyster extent was largely dependent upon increased tidal connectivity with the Gulf (CBA 2017). The 1500s were the most recent documented time when the bay hosted extensive oyster reefs (Thomas and Campbell 1993). The bay connects to the Gulf of Mexico at East Pass, which was an ephemeral tidal inlet until it was dredged and permanently opened in 1929 (Ruth and Handley 2007). The reefs that exist today were established shortly following the opening of the East Pass (CBA 2017). Choctawhatchee Bay has low oyster abundance, possibly due to limited hard substrate and changing water conditions from the previously ephemeral inlet.

Although there is limited information on early harvest yields in Choctawhatchee Bay, it is thought that the oyster harvest has declined since the early 1900s (Bahr and Lanier 1981, CBA 2017). Choctawhatchee Bay has undergone several substrate replenishment efforts coordinated by the Florida Department of Agriculture and Consumer Services (FDACS) using clam and oyster shells (including fossil shell) in efforts to improve the fishery (Berrigan 1988, CBA 2017). Replenishment and mapping efforts have focused on the eastern side of the bay in Walton County. While oyster extent in the western side of the bay is small, the extent of reefs is underrepresented by current maps (Fig. 2.6), particularly as there are known oyster restoration efforts located near Fort Walton Beach and Rocky

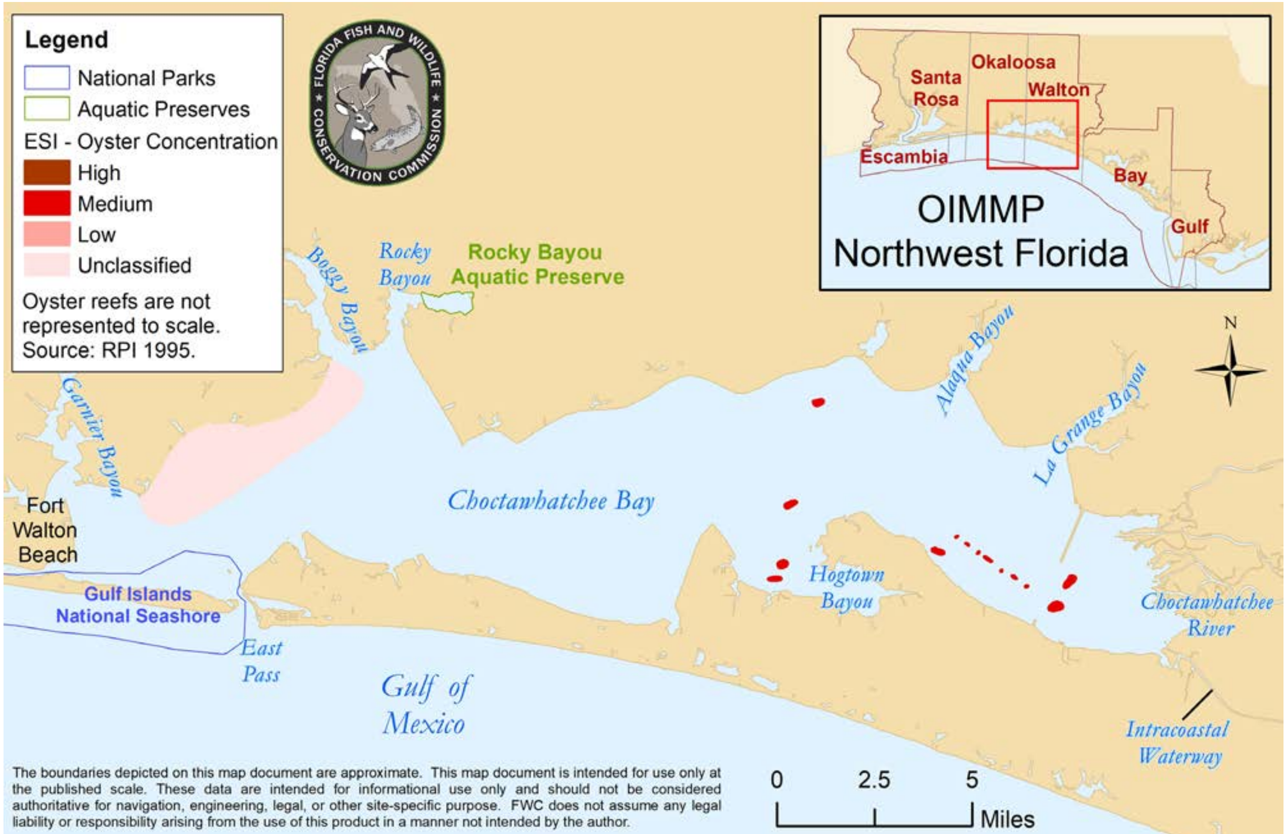

Figure 2.6. Mapped oyster extent in Choctawhatchee Bay. Oyster mapping source: RPI 1995 (from 1995 Environmental Sensitivity Index).

Bayou (CBA 2017). While parts of Okaloosa and Walton counties are conditionally approved for shellfish harvesting (Fig. 2.2), landings are reported infrequently, and harvest yields are low (Fig. 2.3, FWC 2018).

St. Andrew Bay

The West, North, and East bays that comprise St. Andrew Bay receive freshwater flow from 10 small creeks (Fig. 2.7). The largest flow originates from Econfina Creek, which drains into the northern portion of North Bay (FDEP 2016, Brim and Handley 2007). There is also a small hydrological exchange through the ICW in the west to Choctawhatchee Bay and in the east to St. Joseph Bay and the Apalachicola watershed. Approximately 2,000 ha (5,000 ac) of North Bay were impounded in 1961, disconnecting water flowing from Econfina, Bear, and Cedar Creeks and Bayou George into St. Andrew Bay proper. This impoundment is known as Deer Point Lake and provides water to Panama City and surrounding areas.

The water in St. Andrew Bay is relatively clear as little suspended sediment is brought in by the low freshwater flow (Brim and Handley 2007). The bay is protected from the Gulf by narrow peninsulas and barrier islands that have become welded to the mainland, which limit tidal flushing. Tidal range between neap and spring tides varies from 0.06–0.67 m (0.2–2.2 ft) (Brim and Handley 2007). Historically, St. Andrew Bay was connected to the Gulf of Mexico at East Pass at the end of Shell Island. A shipping channel was constructed through the center of the barrier peninsula in 1934 and sediment accumulation eventually closed East Pass in 1998 (FDEP 2016). Water in the bay has a long residence time and is susceptible to the accumulation of pollutants. The bay is a challenging habitat for oysters due to higher than optimal salinity as a result of low freshwater input (NWFWMD 2008). Little is known about rates of disease and predation on oyster reefs in St. Andrew Bay, although these rates are likely to be high because of high salinity (NWFWMD 2008). During certain weather conditions, such as stalled frontal systems, the salinity can decline rapidly throughout West and North Bays. The duration of these freshwater pulses is poorly understood but may persist for long enough to have deleterious effects on oysters found here. The extent to which such events impact East Bay is unknown. Additionally, the substrate in many parts of

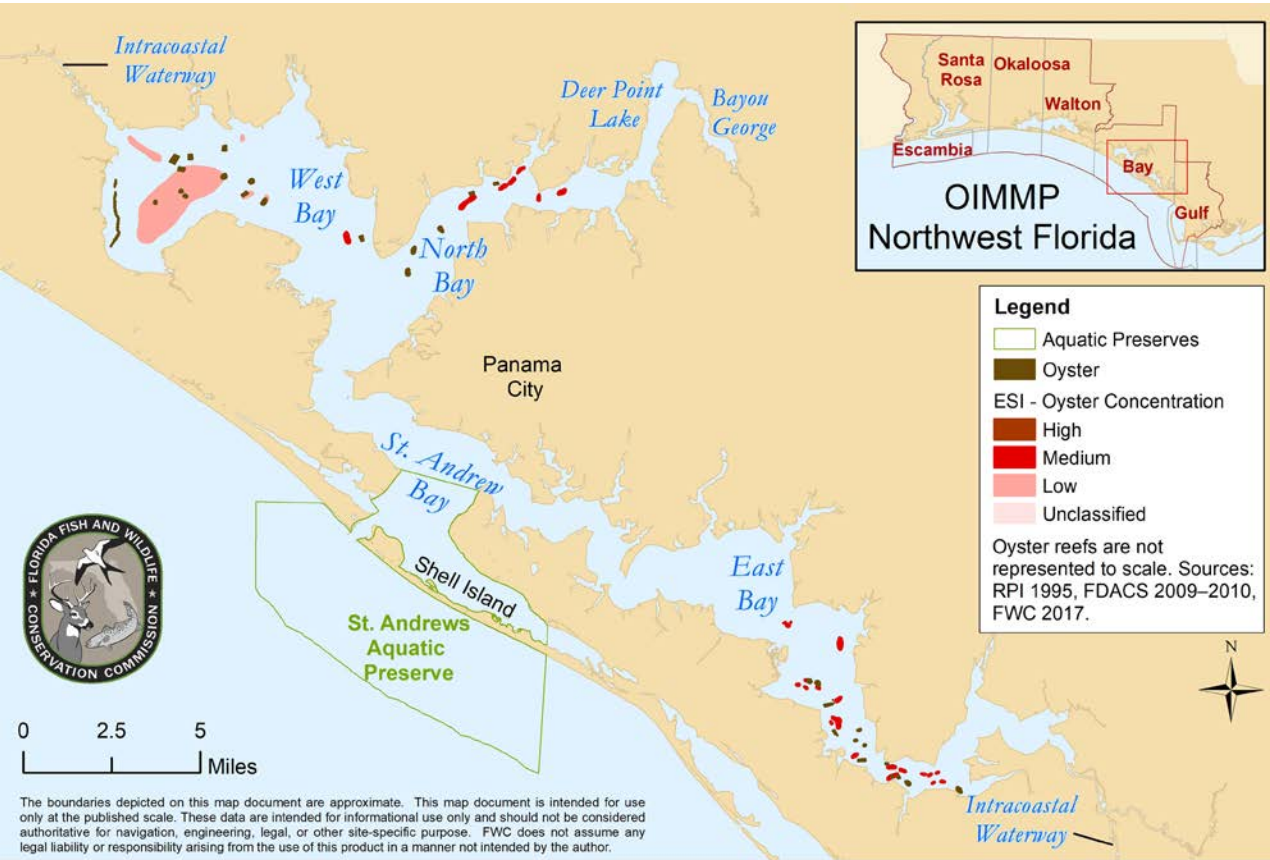

Figure 2.7. Mapped oyster extent in St. Andrew Bay. Oyster mapping sources: RPI 1995 (from 1995 Environmental Sensitivity Index), FDACS 2009–2010 (from navigation charts and local knowledge), and FWC 2017 (from 2017 side-scan sonar).

the upper bay is clay or silt and is therefore too soft for oyster reef establishment (Brim and Handley 2007). The impact of Hurricane Michael, a category 4 hurricane which made landfall at St. Andrew Bay in October 2018, on the bay's oyster reefs is unknown at the time of the writing of this report.

Both natural and planted reefs are found within the bay (NWFWMD 2008). However, limited data are available on oyster extent within the bay (NWFWMD 2017d) and existing maps (Fig. 2.7) may underestimate true extent (NWFWMD 2008). Parts of the West, North, and East Bays are conditionally approved for shellfish harvesting (Fig. 2.2). In 1975, the total oyster harvest area in Bay County was less than 60 ha (150 ac) (USEPA 1975). Annual harvest yields for Bay County peaked in 1993 at 213 metric tons (470,000 pounds) (Fig. 2.3).

St. Joseph Bay

St. Joseph Bay is partially enclosed by a spit of land extending north from Cape San Blas (Fig. 2.8). Salinity within the bay is similar to the Gulf of Mexico as a result of minimal freshwater input and a large tidal prism. Freshwater sources include groundwater input, precipitation, and the Gulf County Canal. The Gulf County Canal and ICW enable water exchange with East Bay of the St. Andrew Bay system and the Apalachicola River via Lake Wimico. Sediment load and turbidity is higher in the Gulf County Canal than the bay itself, which has consequently decreased seagrass coverage in the bay near the canal as a result of light limitation (Hand et al. 1996, Berndt and Franklin 1999). The salinity in St. Joseph Bay is too high for optimal oyster habitat as oysters are more vulnerable to predators and disease (Shumway 1996, Baggett et al. 2014, Coen and Bishop 2015). The bay is clear with predominantly sandy bottom and abundant seagrass, but lacks extensive oyster reefs (Beck and Odaya 2001, DWH NRDA Trustees 2017). Commercial oyster harvest yields that are reported for Gulf County (Fig. 2.3) are primarily derived from Indian Lagoon rather than from St. Joseph Bay (Fig. 2.2). Indian Lagoon is discussed in Chapter 3 of this report.

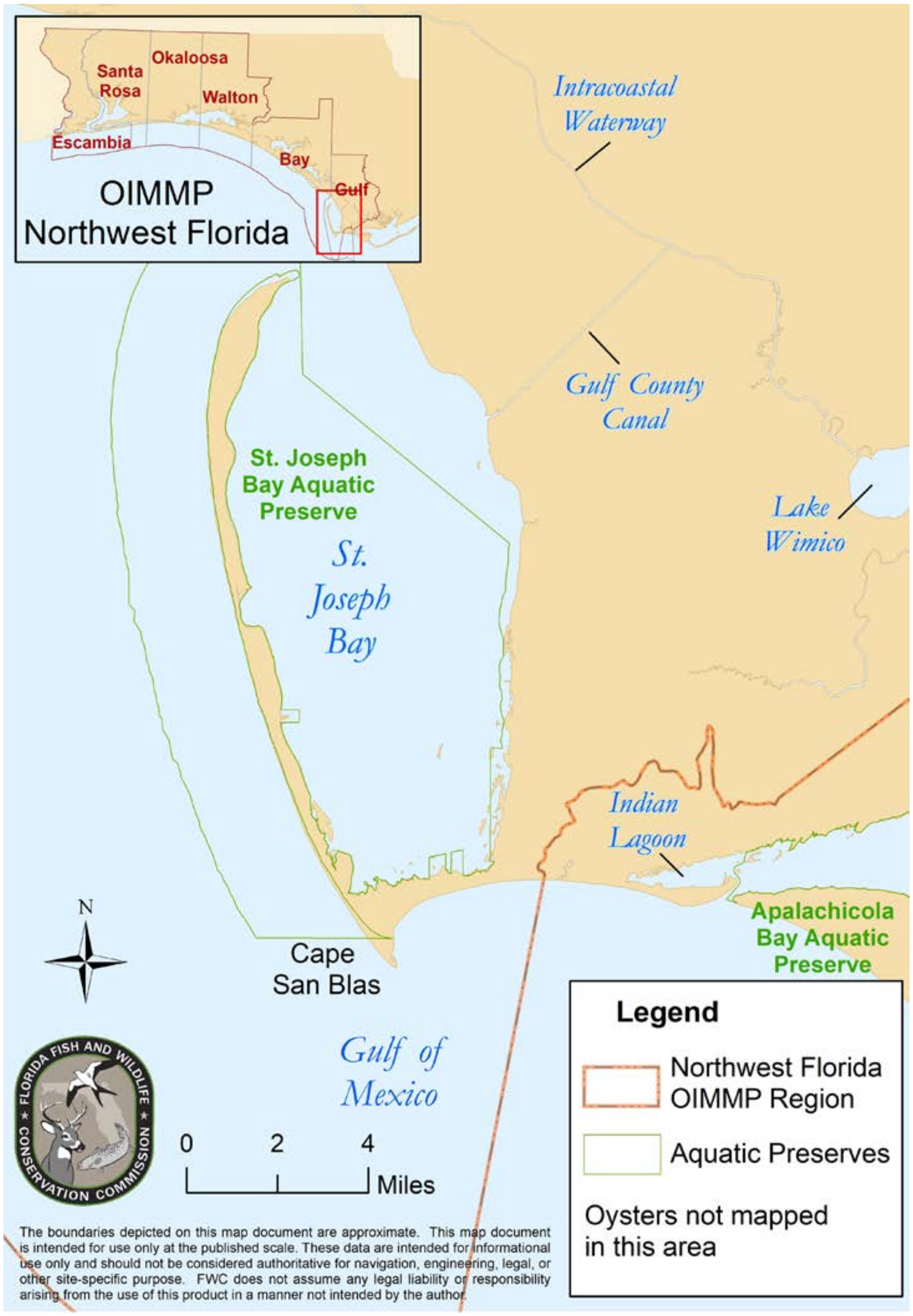

Figure 2.8. St. Joseph Bay and surrounding water bodies. There are no mapped oysters in St. Joseph Bay. Oysters in Indian Lagoon and Apalachicola Bay are described in Chapter 3.

Threats to oysters in northwest Florida

- **Suboptimal salinity:** Oyster distribution in the bays of northwest Florida is limited in many places by suboptimal salinity. Pensacola Bay faces widely variable salinity, which can make much of the system too fresh for oysters for months at a time. Choctawhatchee Bay is often stratified with a halocline. Much of St. Andrew Bay and all of St. Joseph Bay have high salinity due to low freshwater input. While these salinity regimes are not all the result of anthropogenic alterations, suboptimal salinity and its associated impact on disease and predation have slowed efforts to restore and repopulate oyster reefs in the panhandle of Florida (USEPA 2004, Lewis et al. 2016).
- **Sedimentation:** Oysters can be smothered by fine sediments and excess sedimentation can also limit oyster recruitment. Sedimentation is exacerbated by runoff in areas that lack vegetation, such as construction sites, dirt roads, and tree harvesting sites. Reducing erosion and sedimentation is one of the primary goals in water improvement plans across the region (NWFWMD 2017a, 2017b, 2017c, 2017d). Unconsolidated fine-grained sediments do not provide a sufficiently sturdy substrate for reef establishment. A lack of suitable substrate is a limiting factor for reef extent in several of the bays. Additional oyster shell or lime rock aggregate may be needed for the creation, restoration, or enhancement of reef habitat as long as these added materials can be supported without sinking into existing sediment (VanderKooy 2012).
- **Oil spill impacts:** The Deepwater Horizon oil spill of 2010 exposed the westernmost bays in the panhandle to crude oil and weathered residue. Oil exposure in Perdido Bay was light and primarily occurred on the Alabama side of the bay (Byron et al. 2016). Portions of Pensacola

Bay near inlets to the Gulf of Mexico were also exposed to oil, including areas near Pensacola Pass and Santa Rosa Sound (Harvey et al. 2016). Specific data on the impact of these oil exposures on oysters within these bays are not available; however, general studies have shown that direct oyster mortality was considerably higher in other Gulf states than in Florida (DWH NRDA Trustees 2017). Several oyster restoration activities have been financed by funds resulting from compensation for the oil spill, including cultch placement (in multiple bays in Florida panhandle) and construction of living shorelines (in Pensacola Bay) (DWH NRDA Trustees 2017).

- **Climate change and sea-level rise:** Increased temperatures have the potential to change timing and frequency of oyster spawning (Wilson et al. 2005, Hofmann et al. 1992) and reduce larval survival and settlement (Shumway 1996, DWH NRDA Trustees 2017). Sea-level rise will further increase salinity in bays along northwest Florida, making oysters even more vulnerable to predation and disease.
- **Harvesting:** Most of the panhandle bays have areas open to oyster harvest. During harvest, oyster shell is removed from the oyster bed. If the rate of shell removal exceeds the rate of growth, supplemental deposition of shell is required for the reefs to maintain suitable vertical relief and exposed surfaces for settlement to occur (VanderKooy 2012).
- **Isolated populations:** As each bay along the Gulf coast goes through periods of reduced abundance, genetic connectivity between populations in the bays is reduced. When the oyster population in a single bay declines, the chances of larvae being exported from one bay and subsequently imported by a neighboring bay decline.

Oyster reef mapping and monitoring efforts

The compilation of oyster maps used in figures in this report are available for download at http://geodata.myfwc.com/datasets/oyster-beds-in-florida.

Environmental Sensitivity Index maps

The National Oceanic and Atmospheric Administration (NOAA) office of Response and Restoration created Environmental Sensitivity Index (ESI) maps of coastal zone natural resources across the state of Florida. These maps were designed for use in damage evaluation, prevention, and clean-up efforts in the case of oil spills. Areas were mapped on a scale of sensitivity based on potential exposure, biological productivity, and ease of clean-up. ESI maps of oysters and several other shellfish species are divided into areas with low, medium, and high concentrations. These concentration categories were subjective and based upon the opinion of local experts. Oyster mapping data for northwest Florida was published in 1995 (RPI 1995). More information and ESI mapping data can be found at https://response.restoration.noaa.gov/resources/environmental-sensitivity-index-esi-maps.

FDACS oyster mapping

A set of hand-drawn oyster maps were created by FDACS personnel for the panhandle using NOAA navigation charts and verified in survey and monitoring efforts. These maps were then digitized by the Florida Fish and Wildlife Conservation Commission (FWC) to create the FDACS 2009–2010 dataset. While a report is not available regarding the methodology for the creation of these maps, these oyster maps were published in Section 17 of the Gulf States Marine Fisheries Commission Regional Management Plan (VanderKooy 2012). A combination of these FDACS maps and the ESI maps (RPI 1995) were used to create the figures in this chapter.

Northwest Florida Water Management District oyster mapping

NWFWMD land use/land cover (LULC) maps from 2006–2007 identified a few intertidal areas in St. Andrews Bay as oyster reefs, however these areas were later reclassified as sand in the 2009–2010 and 2012–2013 LULC maps and are thus not included in the maps in these chapters. LULC maps are created following the Florida Land Use and Cover Classification System (FLUCCS) classification system, which includes a category for oyster bars (FLUCCS 6540; FDOT 1999). NWFWMD geographic information system (GIS) shapefiles are available for download at https://www.fgdl.org/metadataexplorer/explorer.jsp.

NOAA Mussel Watch

The 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 the northwest include St. Andrew Bay, Choctawhatchee Bay, and Pensacola Bay. Oysters were monitored for concentrations of heavy metals and organics in each location. Oysters contained medium to high levels of arsenic, copper, mercury, and lead. Mercury was particularly high in Choctawhatchee Bay and Pensacola Bay oysters (Kimbrough et al. 2008).

Oyster Reef Restoration Database

Furlong (2012) compiled a database of 422 restored oyster reefs in the Gulf of Mexico by contacting a variety of universities, state and federal agencies, and non-profit organizations to obtain information on the location, management, and material construction of oyster reef restoration efforts. Twenty of these reefs were sampled, and it was found that only 65% of restored reefs successfully provided hard substrate with living oysters. Artificial reefs created out of rock were found to have a higher adult oyster density than reefs made from shell (Furlong 2012).

Pensacola Bay larval recruitment monitoring and modeling

Oyster recruitment and larval supply were monitored in 2007–2008 in Pensacola Bay (Arnold et al. 2017). These data were compared with data on wind, freshwater discharge, salinity, and water depth to model water circulation and larval dispersion throughout the bay. The model indicated that a very low proportion of oyster larvae were exported out of the bay. Thus, the oyster populations in the panhandle likely function as isolated local populations with occasional larval export events that allow for genetic exchange between the metapopulation among the bays (Arnold et al. 2017).

Pensacola Bay mapping and condition analysis

The Nature Conservancy (TNC) is leading a mapping and condition analysis effort on oyster reefs in Pensacola Bay using RESTORE Act Direct Component funding granted to Santa Rosa County by the Deepwater Horizon compensation funds. The project is anticipated to start in 2019 and will be implemented over 3 years. Phase 1 oyster habitat mapping includes an analysis of the data gaps of oyster resources in the East and Blackwater Bays to establish a baseline of the existing extent and condition of the oyster resources. TNC has initiated an oyster mapping and condition assessment protocol in Apalachicola Bay and will use similar methodology for this Pensacola Bay project. Phase 1 of the oyster habitat mapping consists of compiling and preparing information on aerial imagery, existing maps, and associated GIS shapefiles of current intertidal and subtidal oyster reef habitat in the project region. The data sources will be used to create preliminary maps in ArcGIS format of oyster bottom habitat, identify gaps within existing mapped areas, and identify gaps in areas not yet mapped throughout the bays. Phase 2 will be to ground-truth the maps developed in Phase 1, map oyster habitat in the identified gap areas, quantitatively characterize the general condition of the natural oyster habitat in the bay and make recommendations for restoration and management.

Pensacola and St. Andrew Bay restoration mapping and monitoring

A Florida Oyster Cultch Placement Project was included in the Deepwater Horizon Natural Resource Damage Assessment (NRDA) Final Programmatic and Phase III Early Restoration Plan (NOAA 2014). The project involved the placement of suitable cultch and lime rock aggregate on existing oyster reefs for new oyster colonization in the Pensacola Bay and St. Andrew Bay system. The geographic coordinates and description of these restoration efforts can be found in the project reports (FDACS 2016a, 2016b). Approximately 15,000 m^3 (20,000 yd^3) of a lime rock aggregate were placed over an estimated 36 ha (88 ac) of debilitated oyster reefs in the Pensacola Bay System in Escambia and Santa Rosa Counties, while approximately 13,000 m^3 (17,000 yd^3) of crushed granite was placed over an estimated 34 ha (84 ac) of debilitated oyster reefs in the St. Andrew Bay System in Bay County. The Florida Department of Environmental Protection's (FDEP) Central Panhandle Aquatic Preserves office is currently monitoring the success of this restoration effort, which also includes a mapping component for cultched reefs in the Pensacola and St. Andrew Bay systems to depict the extent of enhanced oyster reefs.

Choctawhatchee Bay mapping and monitoring

The first known oyster reef maps in Choctawhatchee Bay were developed in the late 1950s by FDACS. The mapping effort also included FDACS shell placement areas. Mapping efforts have focused on harvestable areas in Walton County. There have been no significant oyster mapping efforts in Okaloosa County or other non-harvestable areas of Choctawhatchee Bay (CBA 2017).

Over the last 10–20 years, the Choctawhatchee Basin Alliance has constructed, mapped, and monitored several intertidal oyster reefs in Choctawhatchee Bay as a part of a living shorelines program (CBA 2017). Monitoring parameters include size and density of oysters, sediment accumulation, water quality, and associated flora and fauna.

St. Andrew Bay restoration, mapping, and monitoring

In 2014, FWC received funding from the National Fish and Wildlife Foundation - Gulf Environmental Benefit Fund to implement a large-scale restoration project en-titled, Oyster Reef Habitat Restoration in St. Andrew Bay, FL (http://www.nfwf.org/gulf/Documents/fl-st-andrew-oyster-14.pdf). Through the placement of suitable oyster cultch, the project has created approximately 1.6 ha (4 ac) of subtidal oyster reef habitat as of April 2018 (shown on far left of Fig. 2.7) and plans to enhance over 80 ha (200 ac) of historical seagrass habitat in the West Bay segment of St. Andrew Bay. FWC conducts annual monitoring on the success of this restoration effort following the protocols of Baggett et al. (2014). Measured parameters include: oyster reef areal dimensions, oyster reef height, oyster density, oyster size-frequency abundance, and water quality. Restoration goal-based metrics include habitat enhancement for resident and transient fish, invertebrate species, and submerged aquatic vegetation (i.e., seagrasses). As a part of this monitoring effort, FWC is compiling fine-scale maps of oyster reef areal and height dimensions using side-scan sonar imaging technology and is assessing the feasibility of completing similar mapping surveys in other estuaries of Northwest Florida.

Recommendations for management, mapping, and monitoring

- Create updated maps of intertidal and subtidal oyster reefs for all bays in this region. While limited oyster maps are available for Pensacola, Choctawhatchee, and St. Andrew bays, these maps are based on data from 1995 or largely derived from hand-drawn maps and nautical charts. No maps are available for St. Joseph Bay or Perdido Bay, although oysters do grow peripherally along the shoreline in these areas (Beck and Odaya 2001, DWH NRDA Trustees 2017). Intertidal oysters growing on hardened shorelines or nested among salt marsh vegetation are generally not mapped by traditional mapping efforts which rely on aerial photography, therefore on-site ground truthing is necessary. Subtidal oyster reefs are mapped infrequently or not at all as labor-intensive efforts are required to map the benthos with sonar.
- Once subtidal and intertidal oyster reef habitat maps are established for Northwest Florida, a standardized and regularly repeated monitoring program is recommended to obtain current information on the status, conditions, and trends for those habitats. Monitoring programs should include methods tailored for commercially harvested as well as non-harvested reefs. Such monitoring and assessment programs have been highlighted as a watershed priority in each of the region's Surface Water Improvement and Management Plans (NWFWMD 2017a, 2017b, 2017c, 2017d).
- There are no plans for the management of oyster reef resources in Northwest Florida. Effective management planning should be stakeholder driven, involve the input of state resource management and policy agencies, and consider the full suite of economic and environmental services provided by oyster populations and the habitat they create. Oyster habitat management plans for each basin should consider managing the resource for sustainable human consumption (whether via wild harvest or aquaculture), shoreline protection, water quality improvement, the provision of fisheries habitat, and carbon sequestration. Bay-specific fishery management plans should be developed to include an estimate of sustainable harvest based on maintenance of the reef structure, including assessment of how much shell must be returned to the reef to offset loss due to harvest.
- Ensure that each bay has established oyster reefs in both upstream and downstream locations to increase genetic exchange among local populations within the metapopulation. By having a variety of reefs in each system, the resilience within each system is increased and the probability of exchanging larvae with neighboring bays increases. Create an oyster habitat suitability monitoring and modeling program to direct financial resources toward the areas that may be the most effective at enhancing the oyster population, enhancing ecosystem benefits, and sustaining economic use. Current understanding of areas suitable for maintaining existing oyster habitat and for creating, restoring, or enhancing degraded habitat is severely limited.
- Small-scale oyster shell recycling programs exist in Pensacola (http://keeppensacolabeautiful.org/) and Choctawhatchee Bays (http://www.basinalliance.org/). Additional programs are needed to support both the sustained reshelling of commercial reef habitat and the large number of oyster habitat or living shoreline projects anticipated for the region over the next 5 to 25 years. Oyster shell recycling hubs established in any of Florida's Northwest counties can build upon previously developed models (e.g., OYSTER or Shuck & Share) and engage the local community through school educational programs and volunteer events.

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General references and additional regional information

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http://www.basinalliance.org/

Florida Aquatic Preserve Program
https://floridadep.gov/fco/aquatic-preserve

Florida Living Shorelines
http://floridalivingshorelines.com/

Keep Pensacola Beautiful, Inc.–OYSTER: Offer Your Shell to Enhance Restoration
https://keeppensacolabeautiful.org/what-we-do/recycling/oyster_1/shell-recycling.html

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Description of the region

Apalachicola Bay is the largest of several estuarine systems in the panhandle region of northwestern Florida. It is confined hydrologically by a network of four barrier islands and is divided into four sections: St. Vincent Sound, Apalachicola Bay proper, East Bay, and St. George Sound (Fig. 3.1). The system is connected to the Gulf of Mexico through three natural tidal inlets (Indian Pass, West Pass, and East Pass) and one man-made inlet (Government Cut, also known as Sike's Cut). The bay is in a transition zone between diurnal tides to the west and semidiurnal tides to the southeast, resulting in a mixed tidal regime with one to five tides daily (Huang 2010, Oczkowski et al. 2011, Huang et al. 2015). Tides can be strongly affected by wind and are normally less than 1 m (3.3 ft) in range. Water currents are tidally driven but can also be strongly impacted by river discharge and winds. Currents generally do not exceed 1 m s^{-1} (3.3 ft s^{-1}) except in passes and tidal cuts. The system is wide and shallow, with an average depth of 2–3 m (6.5–10 ft), resulting in well-mixed and well-oxygenated waters with little stratification. Bottom types consist largely of sand and other soft sediments, with hardbottom in the form of extensive oyster reefs (Edmiston 2008). Water temperature typically ranges annually from 5–32 °C (41–90 °F). Salinity varies widely spatially and temporally and can range from less than 1 to 33. Overall water quality conditions in Apalachicola Bay are excellent, in part because the panhandle region is one of the least populated coastal areas in Florida (Livingston 1984, 2015, Edmiston 2008).

The bay receives most of its freshwater inflow from the Apalachicola River, the largest river in Florida in terms of flow. Average seasonal discharges range from 570 m^3 s^{-1} (20,000 ft^3 s^{-1}) in late summer and fall to 1,800 m^3 s^{-1} (65,000 ft^3 s^{-1}) in early spring (Edmiston 2008, Huang 2010). More than 80% of the water in the Apalachicola River comes from the Chattahoochee and Flint rivers (Fig. 3.2), which converge at Lake Seminole and the Jim Woodruff Dam at the Florida/Georgia border to form the Apalachicola. The Chipola River also provides smaller volumes of water as a tributary to the Apalachicola. The watershed of the Apalachicola–Chattahoochee–Flint (ACF) river system encompasses roughly 50,500 km^2 (20,000 mi^2) in Florida, Georgia, and Alabama. More than 7 million people, including many residents of Atlanta, live in the ACF watershed and rely on it as a major source of fresh water for drinking, recreation, and agriculture (Camp et al. 2015). The ACF river system includes 16 dams built to control alluvial flow and prevent flooding (la Cecilia et al. 2016).

Because of its productivity, biodiversity, and water quality, Apalachicola Bay has been designated as an Outstanding Florida Water, State Aquatic Preserve, International Biosphere Reserve, and National Estuarine Research Reserve (NERR; Livingston 1984, Edmiston 2008). The region is within the Northwest Florida Water Management District (NWFWMD). The Apalachicola National Estuarine Research Reserve (ANERR), which

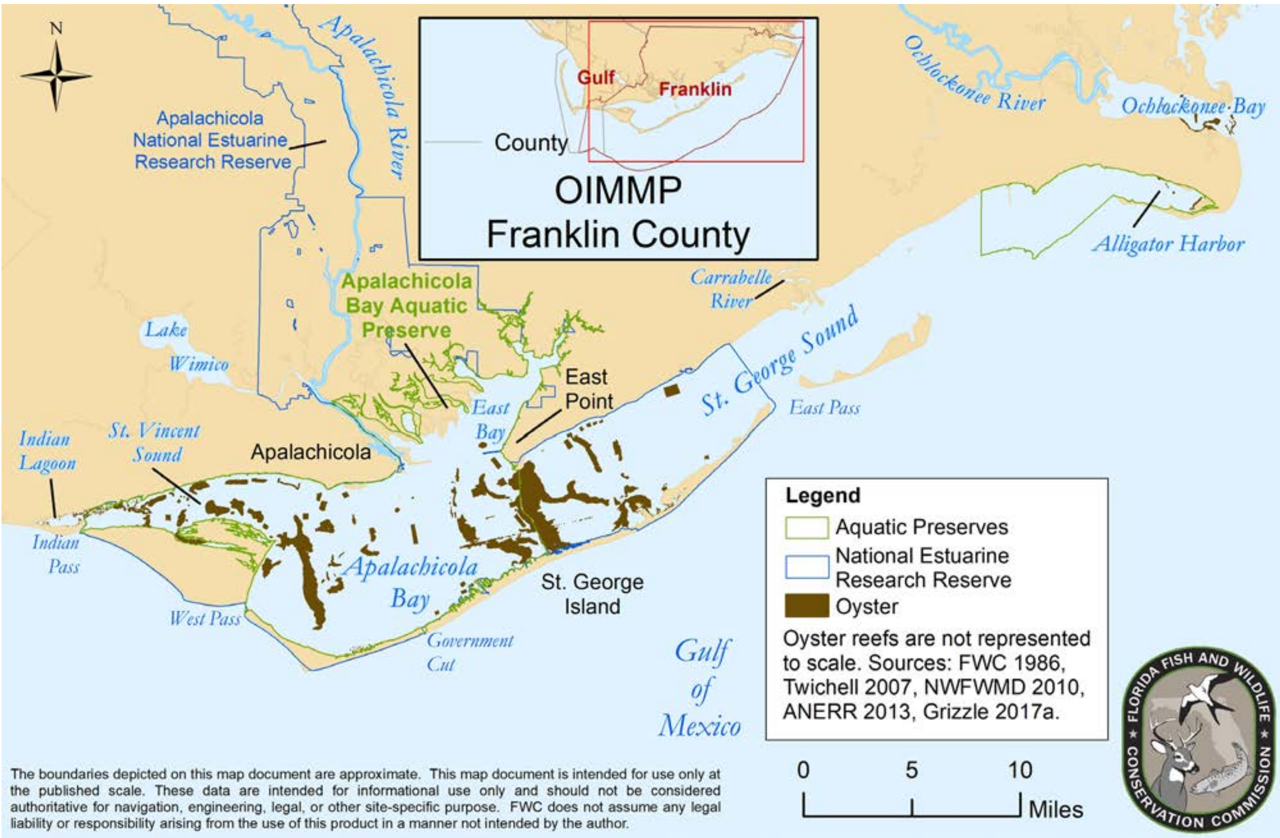

Figure 3.1. Major features of Franklin County, including the Apalachicola Bay estuarine system and oyster reef areas to the west (Indian Lagoon) and east (Alligator Harbor and Ochlockonee Bay). Oyster mapping sources: FWC 1986 (made from historical data and aerial photographs, years unknown), Twichell 2007 (from 2005–2006 side-scan sonar), NWFWMD 2010 (from 2009–2010 aerial photographs), ANERR 2013 (from 2007 and 2010 aerial photographs), and Grizzle 2017a (from 2012 satellite imagery).

encompasses roughly 1,000 km^2 (390 mi^2), spans the estuary and the lands surrounding the lower Apalachicola River (Fig. 3.1; FDEP 2014). Lands within ANERR are owned and managed by many partners including the Florida Fish and Wildlife Conservation Commission (FWC; Apalachicola River Wildlife Enhancement Area), NWFWMD (Apalachicola River Water Management Area), U.S. Fish and Wildlife Service (St. Vincent National Wildlife Refuge), the Florida Department of Environmental Protection's (FDEP's) Division of Recreation and Parks (St. George Island State Park), as well as FDEP's Florida Coastal Office (Apalachicola Bay Aquatic Preserve, Little St. George Island). Nearly the entire estuarine system provides potential habitat for the eastern oyster, *Crassostrea virginica*. The bay's history of providing most of the state's oyster harvest (until recently) is one indicator of how important oysters are in the bay's ecology and the region's economy. The extensive interstate watershed of the Apalachicola River, however, exacerbates the complexity of managing Apalachicola oyster resources.

Ecology of oysters in Apalachicola Bay

The autecology of oysters in the bay has been reasonably well studied (see summaries in Livingston 1984, Edmiston 2008). Spawning occurs mainly from April through October, typically with spring and fall peaks. Growth is continuous and rapid throughout the year, and market size (76 mm [3 in] shell height) is reached in approximately 18 months (Ingle and Dawson 1952). Oyster reefs cover perhaps 10% of the bay bottom and include both subtidal and intertidal reefs (Kennedy and Sanford 1989, Edmiston 2008). Subtidal reefs cover much more area than those in the intertidal zone, with 1,600–4,000 ha (Fig. 3.3; 4,000–10,000 ac) of subtidal oyster bottom mapped or estimated in recent decades (Livingston 1984, Twichell et al. 2007, ANERR 2013) compared to approximately 80 ha (200 ac) of intertidal reefs mapped in 2016 (Grizzle et al. 2017a). The intertidal reefs consist mainly of natural reefs, while the subtidal reefs consist of natural and planted reefs resulting from additions of clam shell,

Figure 3.2. Apalachicola–Chattahoochee–Flint River system (watershed boundary source: NERRS 2007).

fossil shell, and other hard materials to provide cultch (suitable substrate) for oyster larval settlement in support of the fishery (Berrigan 1990, Edmiston 2008, FDACS 2015a).

Based on extensive sonar mapping and field sampling, Twichell et al. (2010) concluded that the present-day subtidal reefs in the bay began to develop on the crests of broad, flat sand bars around approximately 400 BCE, most of which were oriented perpendicular to the long axis of the bay. The early reefs grew vertically and migrated westward, suggesting a net westward transport of sediments in the bay. This model contrasts somewhat with reef development in the Big Bend region to the south, where it is thought that oyster reefs initially developed on nearshore limestone outcrops (Hine et al. 1988). Core and seismic profile data indicate that oyster reefs were more extensive historically and have decreased at their edges due to fine sediment inputs from the Apalachicola River (Twichell et al. 2010). The current reef size and other characteristics reflect changes in the original spatial patterns resulting from more than two millennia of responses to changes in climate, sea level, water quality, sediment inputs from both freshwater and marine sources, and more recently by harvest and management practices.

Recent work has shown wide spatial variability in live oyster densities on both intertidal and subtidal reefs in the bay. The Florida Department of Agriculture and Consumer Services (FDACS) has annually monitored selected reefs in the bay for oyster density and size from 1990 to 2015, when monitoring responsibilities shifted to FWC (data summarized in Camp et al. 2015; also see Grabowski et al. 2017). From 1990 through 2011 (prior to the fisheries collapse discussed below), total oyster densities fluctuated between roughly 200 and 400 oysters/m^2 (19–37 oysters/ft^2). On intertidal reefs, Grizzle et al. (2017b) found an overall mean of ~400 oysters/m^2 (37 oysters/ft^2) in 2016 throughout the bay. However, the western and eastern portions of the bay differed greatly. Many intertidal reefs in the western bay were dead, and the overall mean density of live oysters was <50 oysters/m^2 (4.6 oysters/ft^2), compared with ~1,000 oysters/m^2 (93 oysters/ft^2) in the eastern bay. The same overall pattern was reported for subtidal reefs by Kimbro (2013).

Spatial patterns in mortality also vary widely across subtidal reefs in Apalachicola Bay (Berrigan 1988, Livingston et al. 2000, Edmiston 2008). For example, Livingston et al. (2000) produced maps of oyster mortality illustrating how river flow and salinity variations were related to mortality patterns across the bay in 1985 and 1986. Under moderate river flows, oyster mortality was reduced throughout the central portions of the bay. Under low-flow conditions, the area of high mortality in the outer bay increased. This effect is presumably because predators move from the Gulf of Mexico further into the bay when waters are more saline. The reverse—high river flows, such as during a hurricane—can result in essentially the opposite result with respect to spatial mortality patterns if salinity falls below the oyster's tolerance levels (Shumway 1996, Edmiston et al. 2008). The impacts of storms are more complicated, however, because storm-related factors other than salinity can increase oyster mortality. For example, Hurricane Elena in 1985 produced extreme tides, strong winds, heavy rainfall, and high river discharges that resulted in burial by sediments and other physical damage to reefs in western St. George Sound and eastern Apalachicola Bay (Berrigan 1988, 1990). Oyster production in most areas of the bay dropped by 90% following Hurricane Elena, resulting in closures to harvest, but rebounds in growth and recruitment quickly followed, particularly in areas with substrate restoration (Berrigan 1990). Edmiston et al. (2008) reviewed the literature on the impacts of subsequent storms on oysters in the bay, emphasizing that the effects of sporadic events such as

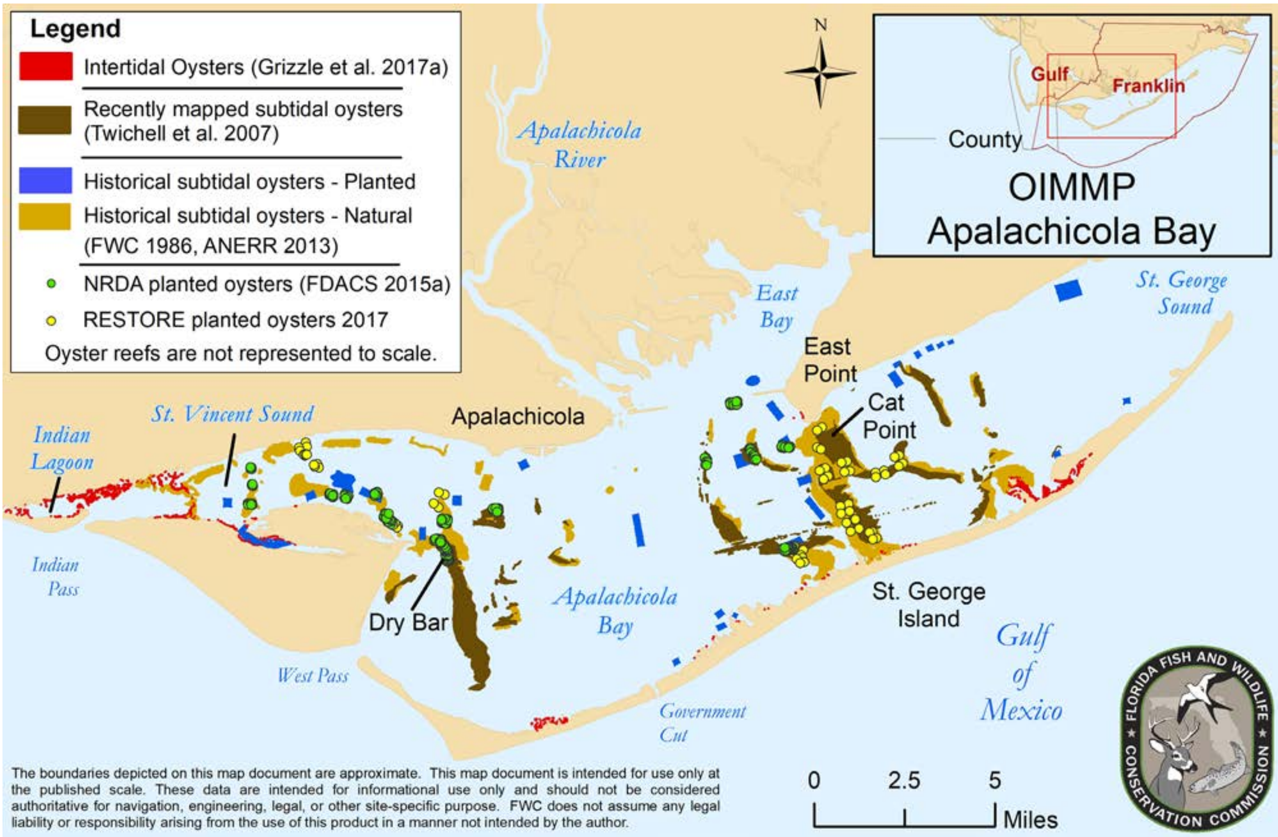

Figure 3.3. Extent of natural and planted subtidal reefs and intertidal reefs in Apalachicola Bay. Oyster mapping sources: FWC 1986 (made from historical data and aerial photographs, years unknown), Twichell 2007 (from 2005–2006 side-scan sonar), ANERR 2013 (from 2007 and 2010 aerial photographs), Grizzle 2017a (from 2012 satellite imagery), and planting sites from 2015 (FDACS 2015a) and 2017.

hurricanes can vary widely and involve multiple mortality factors. Thus, their effects are not easy to predict.

The spatial distribution of the bay's reefs today (Figs. 3.1 and 3.3) is a result of both natural processes and intensive management, which began in the late 1800s (Dugas et al. 1997; see review in Pine et al. 2015). Among the most important of the management actions was implementation of extensive shelling (shell planting) programs. It was soon recognized that loss of shell due to harvest threatened sustainability of oyster fisheries throughout Florida because it removed hard substrate needed for larval settlement. Shell additions to the bay were first recommended around 1885. The Florida Division of Agriculture planted the first known shell, 15,000 barrels' worth, in 1913. Shell distribution increased substantially around 1925 (P. Zajicek, FDACS, personal compilation from Biennial Reports of the Fish Commission, Biennial Reports Shellfish Division, Florida Department of Agriculture and Biennial Reports of the State Board of Conservation). Shell distribution continued more regularly after 1949 as the result of a State-mandated program requiring that harvested oyster shell be returned to public oyster beds, sometimes supplemented with limestone rock. Whitfield and Beaumariage (1977) wrote that as of 1977, more than 4 million bushels of shell and rock had been used to cover nearly 400 ha (1,000 ac) of bottom in Apalachicola Bay. Shell buy-back programs have been implemented to pay dealers for collected shell, but because these programs rely on grants, they do not have a permanent source of funding. Recent shelling programs have used primarily fossil shell (FDACS 2015a, 2015b).

Oyster harvesting in Apalachicola Bay

Much of the Apalachicola Bay system is classified by FDEP as Class II waters (those designated for shellfish propagation or harvesting) that are conditionally approved or restricted for harvest by FDACS dependent on prevailing water quality and seasonal closures (Fig. 3.4). Current oyster harvest regulation in Apalachicola Bay includes bag limits, size limits, and spatial closures. The oyster fishery is integral to the lives of many people living in the Apalachicola Bay region. Before the collapse, the fishery provided more than 2,500 jobs to nearby coastal communities, of-

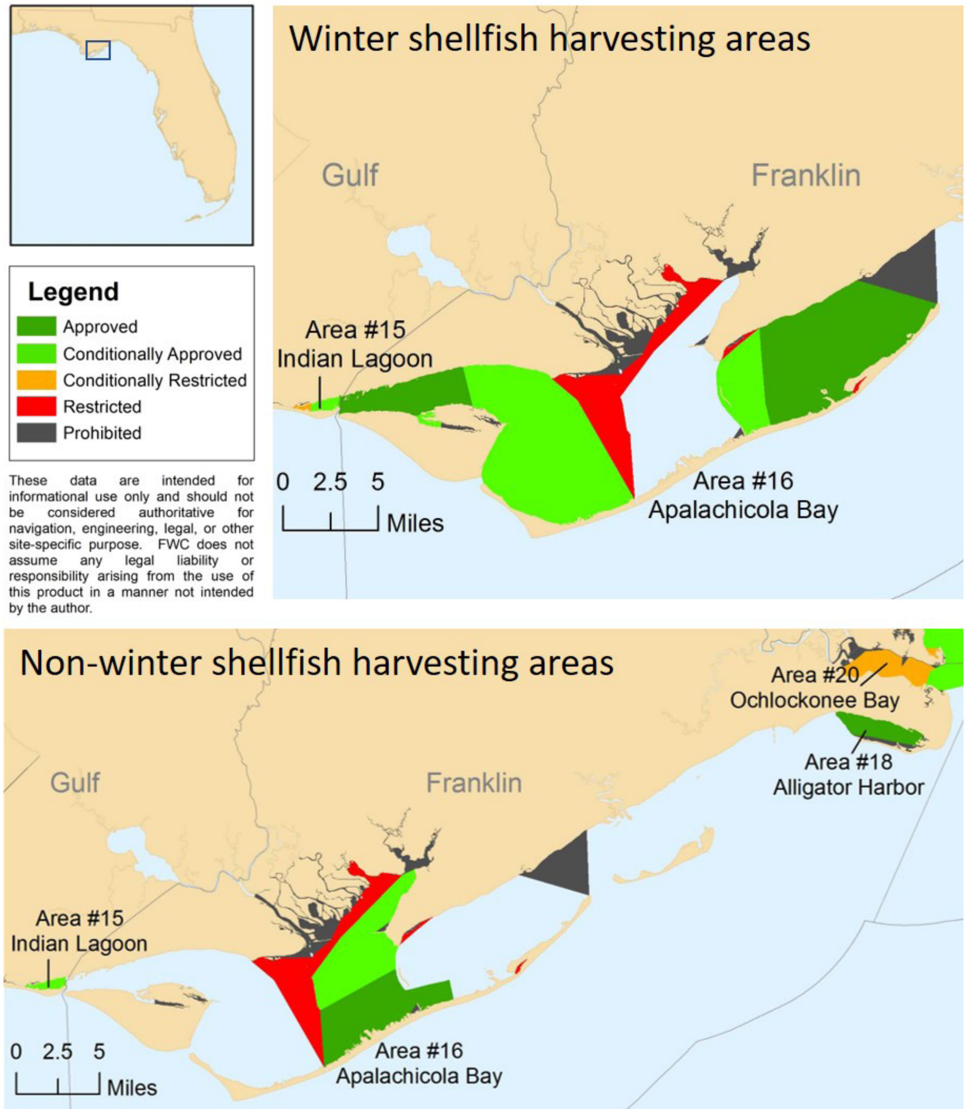

Figure 3.4. Shellfish harvesting areas within eastern Gulf County and Franklin County. Source: FDACS (2017a).

ten making up to half of their revenue (Havens et al. 2013, Camp et al. 2015). Harvest from portions of the bay in Franklin County has historically dominated oyster harvest in Florida, yielding more than 90% of the state's commercial landings (Fig. 3.5) and 10% of the oysters sold in the continental United States (Livingston 1984, Havens et al. 2013). Commercial landings data from 1895 to 1984 were reported by the U.S. Fish and Wildlife Service, Florida State Board of Conservation, Florida Department of Natural Resources, or National Marine Fisheries Service, but the FWC's Fish and Wildlife Research Institute took over these responsibilities in 1985. From 1986 onwards, FWC recorded the number of trip tickets and landings via a mandatory reporting system (Camp et al. 2015). Earlier, such data had been reported voluntarily. Despite the mandatory reporting system, Havens et al. (2013) found evidence of unreported harvest and harvest from closed areas that are difficult to quantify and reconcile with reported landings data.

Oyster landings from Franklin County (dominated by Apalachicola Bay) fluctuated but overall increased from 1950 through the early 1980s, peaking at 3,000 metric tons (6.6 million pounds) in 1981 (Fig. 3.5). In September 1985, Hurricane Elena caused extensive damage to the bay's reefs, particularly on the east end (Livingston et al. 1999; also see discussion above). Many of the reefs that had historically been the most productive suffered high mortality of live oysters, loss of cultch, and extensive sedimentation (Berrigan 1990). The bay was closed to harvest for several months for research and distribution of clam shells as substrate (Berrigan 1990). Commercial oyster harvest resumed in May 1986, but with harvest restrictions. Landings were nearly an order of magnitude lower than

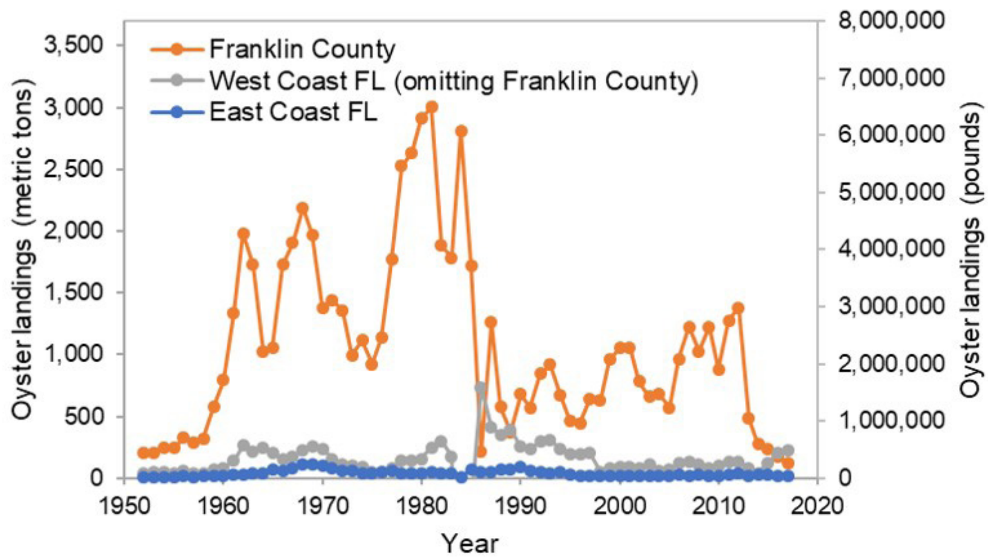

Figure 3.5. Commercial oyster landings for Franklin County and the east and west coasts of Florida. Oyster landings data before 1986 were collected under a voluntary reporting system. Data sources: 1951–1983, Florida Commercial Marine Fish Landings (see Appendix A); 1984–1985, Berrigan (1990); 1986–2017, FWC (2018).

the pre-hurricane harvest in 1985. Oyster populations recovered relatively quickly as a result of successful recruitment, shelling, and restricted harvests (Berrigan 1990, Livingston et al. 1999, Pine et al. 2015), but commercial harvests never returned to the levels recorded before Hurricane Elena (Fig. 3.5).

Landings as well as catch per unit effort (CPUE) estimates fluctuated, but generally increased, through the late 1980s and early 1990s (Fig. 3.6). Several hurricanes affected Apalachicola Bay after 1985; impacts to the oyster reefs and the fishery varied depending on storm-related physical disturbances and salinity extremes. In 1994, hurricanes caused record flooding in the region, resulting in near-freshwater conditions in the bay for nearly two weeks. While reefs were apparently not physically damaged by the hurricanes, mortality on the reefs varied from 10 to 100% as a result of low salinity (Edmiston 2008, Edmiston et al. 2008). Oysters at Dry Bar and St. Vincent reefs (Fig. 3.3) suffered particularly high mortality. In 2005, Hurricane Dennis caused a 3-m (10-ft) storm surge, but this had little impact on subtidal oyster reefs, as the extra water depth protected them from wave energy (Edmiston et al. 2008). Hurricane Katrina in 2005 did not have a measurable impact on the oysters, but hurricane winds pushed a red tide bloom into the bay, resulting in the closure of oyster harvesting for more than three months (Edmiston et al. 2008). Landings increased substantially after Katrina, but CPUE began to show a steady decline until the most recent collapse, in 2012. The impact of 2018's Hurricane Michael on Apalachicola Bay oysters was unknown when this report was written.

It should be noted that Apalachicola Bay was not directly affected by the Deepwater Horizon oil spill in 2010 (Grabowski et al. 2017). The fishery remained open—unlike those in large areas of Texas, Louisiana, and Alabama—and the shortage of oysters in other Gulf areas initially led to an increase in oyster harvesting and prices for oysters from Apalachicola Bay (Camp et al. 2015, Pine et al. 2015, Grabowski et al. 2017). Out of concern for possible future closures, the oyster harvesting season in Apalachicola Bay was opened early. While oyster harvesting is usually prohibited Friday through Sunday, harvesting was also allowed on weekends during that time (though no changes were made with regards to size limits or daily bag limits) (FWC 2013). Despite the extended season, oyster landings in 2010 were slightly lower than those before and after (Fig. 3.6), perhaps in part due to declining prices. Concern about the safety of post–oil spill Gulf oysters led to a decline in demand and oyster prices (Sumaila et al. 2012, Camp et al. 2015), though there has been no evidence that the oil spill contaminated seafood from Apalachicola Bay (Havens et al. 2013).

2012–2013 collapse of the oyster fishery

Oyster landings from Apalachicola Bay began a marked decline in 2012, dropping from 1,378 metric tons (3.0 million pounds) in 2012 to only 483 metric tons (1.1 million pounds) in 2013, followed by four years of historically low landings (Figs. 3.5 and 3.6). Fishery-independent sampling by FDACS has also shown a sharp decline

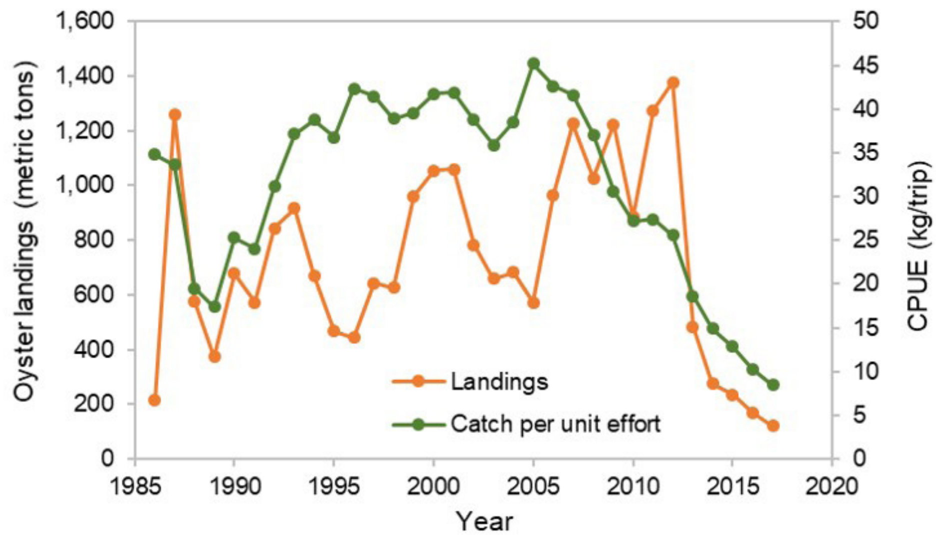

Figure 3.6. Commercial oyster landings and catch per unit effort (CPUE) data for Apalachicola Bay since 1985. Data source: FWC 2018.

in oyster density on subtidal reefs (results summarized in Camp et al. 2015). As previously mentioned, from 1990–2011 total oyster densities fluctuated between roughly 200 and 400 oysters/m^2 (19–37 oysters/ft^2). The density of oysters on subtidal reefs then fell below 100 oysters/m^2 (9 oysters/ft^2) during 2012 and 2013. Although many of the mapped subtidal reefs have not been monitored for density in recent years, a spatially extensive sampling program in 2016 by FWC found live oysters at only 66 of the 161 stations sampled on mapped reefs, and the overall average live oyster density at those stations was only 17 oysters/m^2 (1.5 oysters/ft^2) (Parker 2016).

The cause of the 2012–2013 fishery collapse has been linked to a combination of events. The conclusions of Camp et al. (2015), paraphrased in the following summary, provide a plausible scenario linking five likely contributing factors: 1) low river flow led to increased salinity in Apalachicola Bay for a multiyear period, which caused 2) an increase in oyster parasites, predators, or unknown pathogens, leading to 3) increased oyster mortality, particularly among juveniles, resulting in 4) recruitment failures (over several years) possibly worsened by shell removal by fishing or environmental events, finally leading to 5) collapse of adult oyster populations.

Numerous studies have assessed the role of river discharge in the long-term dynamics of oyster harvest from the bay, confirming the importance of freshwater discharges to the ecology, production, and harvest of oysters but also underscoring the complex nature of the relationship (Wilber 1992, Wang et al. 2008, Oczkowski et al. 2011, Fisch and Pine 2016). Unfortunately, sufficient data to fully support factors 2 and 3 are not available because studies of predators were not under way before the collapse. But, very high densities of boring sponges and predators have been observed in the bay since the collapse (Fig. 3.7, Camp et al. 2015). Camp et al. (2015; also see Fisch and Pine 2016) also discuss research that arrived at similar explanations for previous fishery collapses in Apalachicola Bay and other parts of the state. Fisch and Pine (2016) did not find a significant correlation between oyster CPUE and river discharge between 1987 and 2013; they posit that this lack of a relationship may be a result of the changes in fishery landings reporting requirements, a lack of a proportional relationship between CPUE and oyster populations, hurricane impacts, and changes to ecosystem dynamics in the bay. Overfishing is not thought to have directly contributed to the 2012 collapse, in the sense that recruitment was not limited by harvest (FWC 2013, Pine et al. 2015). Rather, the fishery may have indirectly exacerbated the collapse through the removal of shell substrate (Camp et al. 2015, Pine et al. 2015).

The fishery collapse resulted in a request by the State of Florida for a Federal Fisheries Disaster declaration. The request was granted in 2013 by the U.S. Secretary of Commerce, enabling the use of federal funds to support the community in the aftermath of the collapse (Havens et al. 2013). These funds, as well as funding from the Florida Department of Economic Opportunity, led to the Apalachicola Bay Fishery Disaster Recovery Project Plan, which included restoration of oyster habitat, monitoring of oyster resources and restoration efforts, vocational and educational training for affected oyster fishers and their communities, and processor facilities

Figure 3.7. An abundance of oyster drills (*Stramonita haemastoma*) and their egg cases on a concrete block left for one month to monitor oyster spat settlement in Apalachicola Bay in 2018 (photo credit: Nicole Martin).

upgrades. Several studies were also published focusing on various aspects of the ecological and social dimensions of the collapse. Fisch and Pine (2016) focused on the complexities of the relationship between freshwater discharge and oyster landings. Camp et al. (2015) and Pine et al. (2015) explored the relationship between ecological and social issues, focusing on management strategies that should be considered to enhance resiliency in the fishery. Kimbro et al. (2017) and Pusack et al. (2018) demonstrated the potential importance of predation as related to freshwater discharges to the bay in oyster population dynamics. Overall, recent research has provided new perspectives on the temporal and spatial dynamics of oyster populations in Apalachicola Bay, as well as the complexity and importance of the fishery to the regional economy and local communities.

Legal battles over water rights have been ongoing between the states of Florida, Alabama, and Georgia since the 1980s. But after the 2012–2013 oyster fishery collapse, the State of Florida sought to have the Court apportion water rights in the ACF watershed. The State of Florida argued that Georgia's water policies negatively affected Apalachicola's oyster fishery, resulting in the collapse of the oyster population and the loss of many of the ecosystem services that oysters provide. Florida stated its concern that upstream water use will continue to increase as urban and agricultural demands for water grow in Georgia, inhibiting the recovery of the fishery. In 2014, the U.S. Supreme Court agreed to hear *State of Florida v. State of Georgia* over the appropriation of water from the ACF basin (Fisch and Pine 2016). In 2017, the court-appointed special master recommended that the court side with Georgia because Florida had failed to prove that a water-consumption cap would have averted the fishery collapse (Lancaster 2017). In June 2018, however, the Supreme Court declared that the special master had applied too strict a standard in requiring Florida to prove its case and ordered reconsideration of the case (Florida v. Georgia 2018, Pittman 2018). Review of the case under a new special master is ongoing at the time of the writing of this report.

Little research has dealt with the substantial ecosystem services such as habitat provision, water filtration, and fish production that Apalachicola Bay's oyster reefs

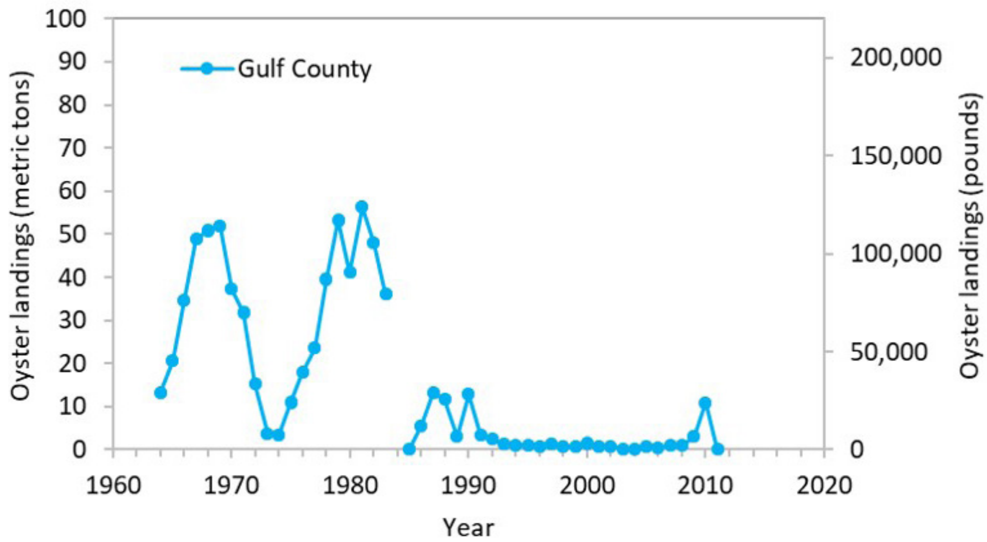

Figure 3.8. Commercial oyster harvest from Gulf County (western Apalachicola Bay). Data sources: FWC (2017) and Florida Commercial Marine Fish Landings (see Appendix A).

provide (Coen et al. 2007, Grabowski and Peterson 2007). In addition to oyster landings and economic impacts, the 2012 fishery collapse in the bay also resulted in a loss of some portion of the ecosystem services the oyster reefs provided. The collapse thus had ecological as well as economic and social effects. In their assessment of long-term changes in water filtration by oyster reefs in 13 estuaries in North America, zu Ermgassen et al. (2013) found only Apalachicola Bay showed an increase in filtration capacity. However, their assessment was based on 1990–2010 data, prior to the 2012 collapse (see Table 1 in zu Ermgassen et al. 2013). It is reasonable to assume that other ecosystem services provided by the bay's oyster reefs have also greatly diminished since 2012.

Finally, a recent result of the Apalachicola Bay oyster fishery collapse is that much of the oyster fishery (harvest and management) shifted to the Big Bend region. In 2016, yields from the Big Bend equaled those from Apalachicola Bay (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). There has also been a renewal of interest in oyster aquaculture in which oysters are grown in cages suspended in the water column, where they are safer from predators and less vulnerable to sedimentation or hypoxic conditions (Reiley 2018). This shift is similar to changes occurring in other estuaries, where traditional oyster fisheries that have failed or are greatly diminished are being supplemented with aquaculture practices.

Indian Lagoon

Located at the westernmost edge of Apalachicola Bay, Indian Lagoon (Fig. 3.1) is within the borders of Gulf County and is not part of ANERR. The lagoon is bounded by Indian Pass peninsula and opens to St. Vincent Sound to the east and the Gulf of Mexico to the southeast at Indian Pass. The lagoon is shallow with a bottom of fine organic sediments (FDEP 2014), and most oyster reefs in the lagoon are intertidal (Fig. 3.3; Grizzle et al. 2017a). Oysters from Indian Lagoon make up most of the landings from Gulf County, which were at substantial levels during the 1960s and 1980s but have been at record low levels since 1990 (Fig. 3.8).

Eastern Franklin County

Oyster reefs also exist in Alligator Harbor and Ochlockonee Bay in eastern Franklin County (Fig. 3.1). Alligator Harbor is a barrier-spit lagoon partly enclosed by Alligator Point peninsula. It has a mean low water depth of approximately 1.2 m (4 ft) (FDEP 2018). Salinity is similar to that in the Gulf of Mexico due to negligible freshwater input. There are some small areas of dense intertidal and subtidal oyster reef in the eastern end of the Harbor, as well as some scattered larger reefs and oyster growth associated with salt marshes (Fig. 3.1; FDNR 1986, FDEP 2018). Little data, however, are available on the condition of these reefs (FDEP 2018). Clam aquaculture was established in 2002 and off-bottom oyster aquaculture was approved in 2015 on leases in Alligator Harbor Aquatic Preserve. The University of Florida Institute of Food and Agricultural Sciences intermittently monitored water quality near these shellfish harvesting areas from 2002–2012; monitoring was discontinued in 2012 due to lack of funding (FDEP 2018).

Ochlockonee Bay receives freshwater flow from the Ochlockonee River. The watershed of this river covers 6,412 km^2 (2,476 mi^2), including parts of southern Georgia and the city of Tallahassee (NWFWMD 2017). Human population in the watershed is steadily increasing, and with population growth comes concerns for proper wastewater and stormwater management. Ochlockonee Bay includes extensive seagrass beds and coastal salt marshes. Salinity in the bay varies with river flow, and the bay is often stratified (NWFWMD 2017). Salinity has remained sufficiently low in the upper half of the bay to protect oysters there from key predators (Kimbro et al. 2017).

Threats to oysters in Apalachicola Bay

Several recent papers provide a comprehensive analysis of the relationship between the Apalachicola Bay's oyster fishery (and, indirectly, its oyster populations) and various environmental factors, thus providing an overview of threats to oysters (Camp et al. 2015, Pine et al. 2015, Fisch and Pine 2016, Kimbro et al. 2017, Pusack et al. 2018). From those papers, four of the most important factors are described below (and in some cases in sections above).

- **Altered hydrology:** Water withdrawals and other changes in the hydrology of the ACF river system represent a threat to oysters that has been at the center of debate and litigation for decades. A network of dams in the ACF river system alters freshwater flow rates, sediment delivery, and erosion patterns for the Apalachicola River and Bay. When this altered hydrology is coupled with low precipitation and urban and agricultural demand for fresh water, the resulting low freshwater flow and high salinity make oysters more vulnerable to dermo (*Perkinsus marinus*) and predators such as the stone crab (*Menippe mercenaria*) and southern oyster drill (*Stramonita haemastoma*) (Livingston et al. 2000, Kimbro et al. 2017, Pusack et al. 2018). Parasites, such as the boring sponge, boring clam, and polychaetes also cause damage to the oysters' shells, possibly resulting in death. The shells become weakened, leaving the oyster more vulnerable to predators (Havens et al. 2013).
- **Sea-level rise:** The combined impact of decreasing freshwater inflow and rising sea level will likely lead to more frequent instances of high salinity in the region. Even with modest increases in sea level, more saline water will enter the bay through East Pass, which will push river discharge toward the west with tidal currents (Huang et al. 2015). Cat Point is expected to experience greater increases in salinity than Dry Bar, as freshwater flow from Apalachicola River is pushed toward Dry Bar (Huang et al. 2015). While most oysters in Apalachicola Bay are subtidal, intertidal oysters will have to cope with increased submergence times. Solomon et al. (2014) found that shell length and recruitment are greatest at high rates of submergence for intertidal oysters in Apalachicola Bay. However, these submerged reef elevations also had the highest rates of sedimentation, which can smother reefs.
- **Hurricanes and tropical storms:** Hurricanes can negatively impact oysters and may cause erosion of reef substrate, sedimentation and burial of reefs, and extreme salinity changes (Edmiston et al. 2008). Hurricanes also redistribute shell off the reef, where it can be buried and lost in the mud (Twichell et al. 2010). Storms often bring heightened pollutant and nutrient loads with terrestrial runoff, which can feed algal blooms (including red tide) and lead to hypoxia (Edmiston et al. 2008).
- **Harvesting:** The effect of harvest on a fishery is generally considered a threat only if harvest exceeds the ability of the population to replenish itself. Pine et al. (2015) found that Apalachicola Bay was not experiencing recruitment overfishing, whereby the population of adults can no longer replace itself. There is no assessment of growth overfishing, whereby oysters may be harvested at a size too small to support a maximum sustainable yield, but one still might argue that if the number of legal-size oysters were extremely limiting, growth overfishing might be occurring. However, removal of shell substrate can cause impacts similar to overfishing as it results in the loss of substrate. Substrate loss is a significant factor for poor recruitment; therefore, fishing without shell replacement (as well as illegal fishing not complying with regulations) greatly reduces the chance that populations may recover (Havens et al. 2013). The effect of harvest on the ecosystem services oyster reefs provide, however, has not been well assessed and remains controversial (Beck et al. 2011).

Apalachicola Bay oyster mapping and monitoring efforts

Historical oyster mapping

Oyster maps for Apalachicola Bay date to the work of Franklin Swift who conducted a comprehensive survey in 1895–1896 for the U.S. Commission of Fish and Fisheries and published a detailed map based on 75,000 manual sounding points (Swift 1897). This map represents the

Figure 3.9. Composite map of the surficial geology of Apalachicola Bay based on 2005–2006 sonar-based mapping. (Figure from Twichell et al. 2007.)

modern starting point for the knowledge of the distribution of natural reefs in the bay before the extensive shell planting programs discussed above were started.

U.S. Geological Survey geophysical mapping of subtidal oysters

Following Swift (1897), another comprehensive survey of Apalachicola's subtidal reefs did not occur until 2005–2006, when the U.S. Geological Survey used interferometric multibeam bathymetry, side-scan sonar, and seismic-reflection techniques to create detailed maps of oyster reefs (Fig. 3.9; Twichell et al. 2007). Data were collected using an outboard-propelled boat, which was used to survey depths greater than 2 m (6.5 ft); an autonomous surface vehicle was used to survey depths between 0.75 and 2 m (2.5–6.5 ft). Approximately one-third of the total bottom area of the bay was not surveyed due to very shallow or very deep water, and they did not survey St. Vincent Sound. This effort characterized the relationship between current oyster reefs, bay floor morphology, and how the reefs likely developed in the long term (Twichell et al. 2010; see discussion in Ecology section above). Shapefiles from these surveys are available for download at https://catalog.data.gov/dataset/benthic-habitats-and-surficial-geology-of-apalachicola-bay-florida-2006-geodatabase.

FDACS compilation

The FDACS Division of Aquaculture compiled mapping data from Twichell et al. (2007) with information on shelling locations (Fig. 3.10). The reefs shown on the FDACS map are mainly subtidal, though some nearshore reefs are likely intertidal. This map is likely the most comprehensive map that differentiates between natural and constructed (restored) oyster reefs in the bay. It should be noted that the FDACS compilation and Twichell et al. (2007) focus on subtidal oyster reefs (Figs. 3.9–3.10) and provide only spatial data; i.e. no information on oyster reef condition is indicated or implied.

Intertidal reef mapping by the University of New Hampshire and The Nature Conservancy

Oyster reefs in the intertidal areas of Apalachicola Bay have largely been neglected in most mapping efforts because most of the oyster harvest has come from subtidal reefs, and the area covered by intertidal reefs is much less than by subtidal reefs. The University of New Hampshire (UNH) and The Nature Conservancy (TNC) developed new maps for oyster reefs in Apalachicola Bay and assessed the potential of high-resolution satellite imagery for mapping and monitoring (Grizzle et al. 2017a). The

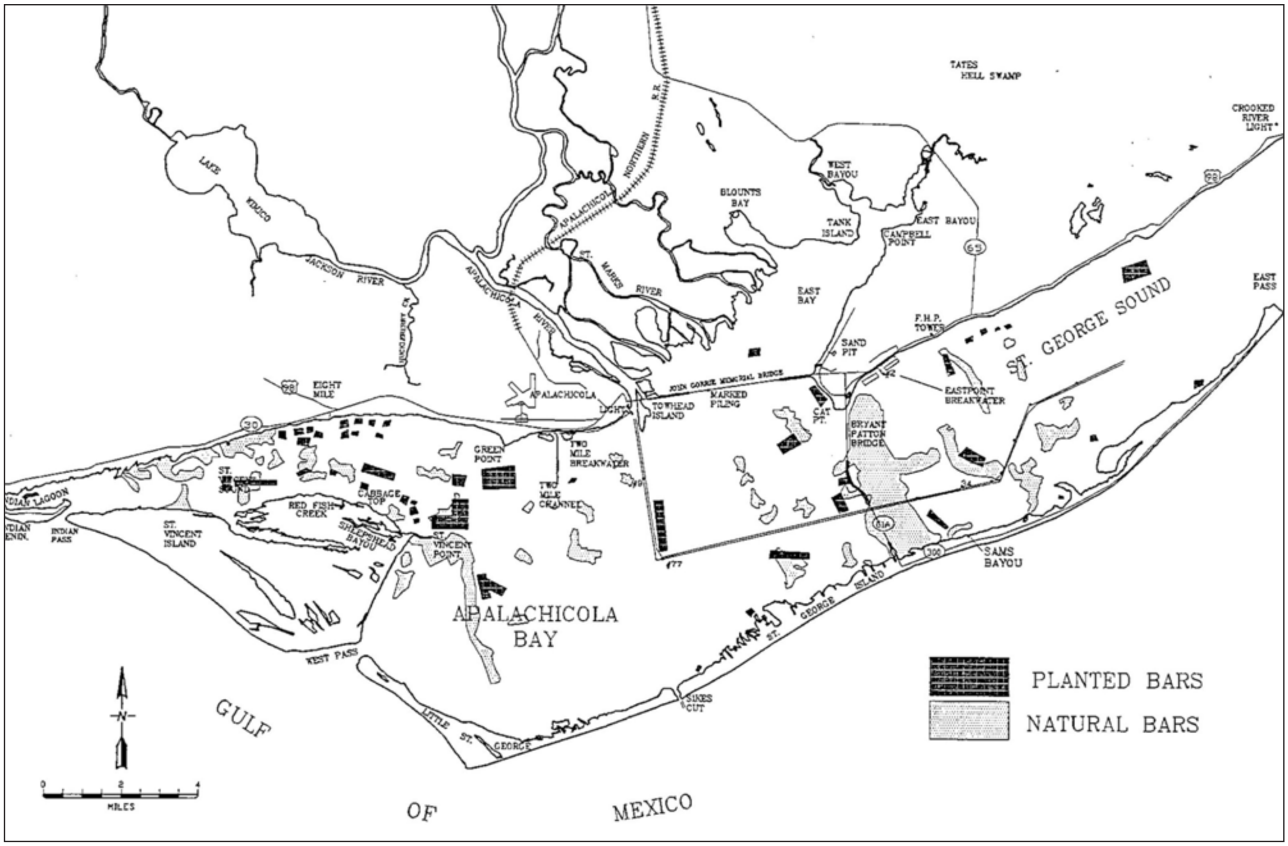

Figure 3.10. Composite map from FDACS' Division of Aquaculture showing oyster bottom areas (planted and natural bars) in Apalachicola Bay (provided by Joe Shields).

project used both high-resolution GeoEye satellite imagery from Grizzle et al. (2015) and ground truthing to assess the position and size of oyster reefs at the time. One hundred reefs were sampled, and oyster density was analyzed. This study concluded that most of the oyster reefs on the western side of the bay consisted of dead shells, indicative of a recent mass mortality, with little recent recruitment. Comparison of ground-truthing data to satellite imagery indicated a classification accuracy of 77–97%. A total of 777 reefs were mapped, covering 78.5 ha (194 ac) of bay bottom.

Figure 3.11 is a composite map that combines most of the data from previously published subtidal maps mentioned above with the intertidal oyster reefs mapped by Grizzle et al. 2017a. This compilation and the FWC compilation (Fig. 3.1 and discussed below) represent the most comprehensive maps available showing the shape, size, and location of the major oyster reefs in Apalachicola Bay. No spatially detailed data are available on the condition of these reefs apart from the intertidal reefs mapped by the University of Central Florida (UCF) and UNH (see data in Grizzle et al. 2016, 2017a, 2017b).

Intertidal reef mapping by the University of Central Florida

Researchers from UCF (Melinda Donnelly, Linda Walters, Stephanie Garvis, and Joshua Solomon) used Landsat imagery from 2012 (USGS) of Apalachicola Bay to map locations of intertidal oyster reefs. After initial mapping, ground truthing was used to evaluate the accuracy of the imagery interpretation. Field observations were conducted at a total of 100 random locations (50 oyster, 50 nonoyster) in summer 2013 (96% accuracy). A total of 603 intertidal reefs were identified, covering approximately 80 ha (198 ac); the majority of intertidal reefs were found near natural shorelines on lands managed by St. George Island State Park and St. Vincent National Wildlife Refuge. Mapping was supported by a grant from NOAA. Shapefiles are available by contacting Melinda Donnelly (Melinda.Donnelly@ucf.edu).

Northwest Florida Water Management District oyster mapping

The most recent NWFWMD land-use/land-cover (LULC) map that included a separate oyster reef layer is

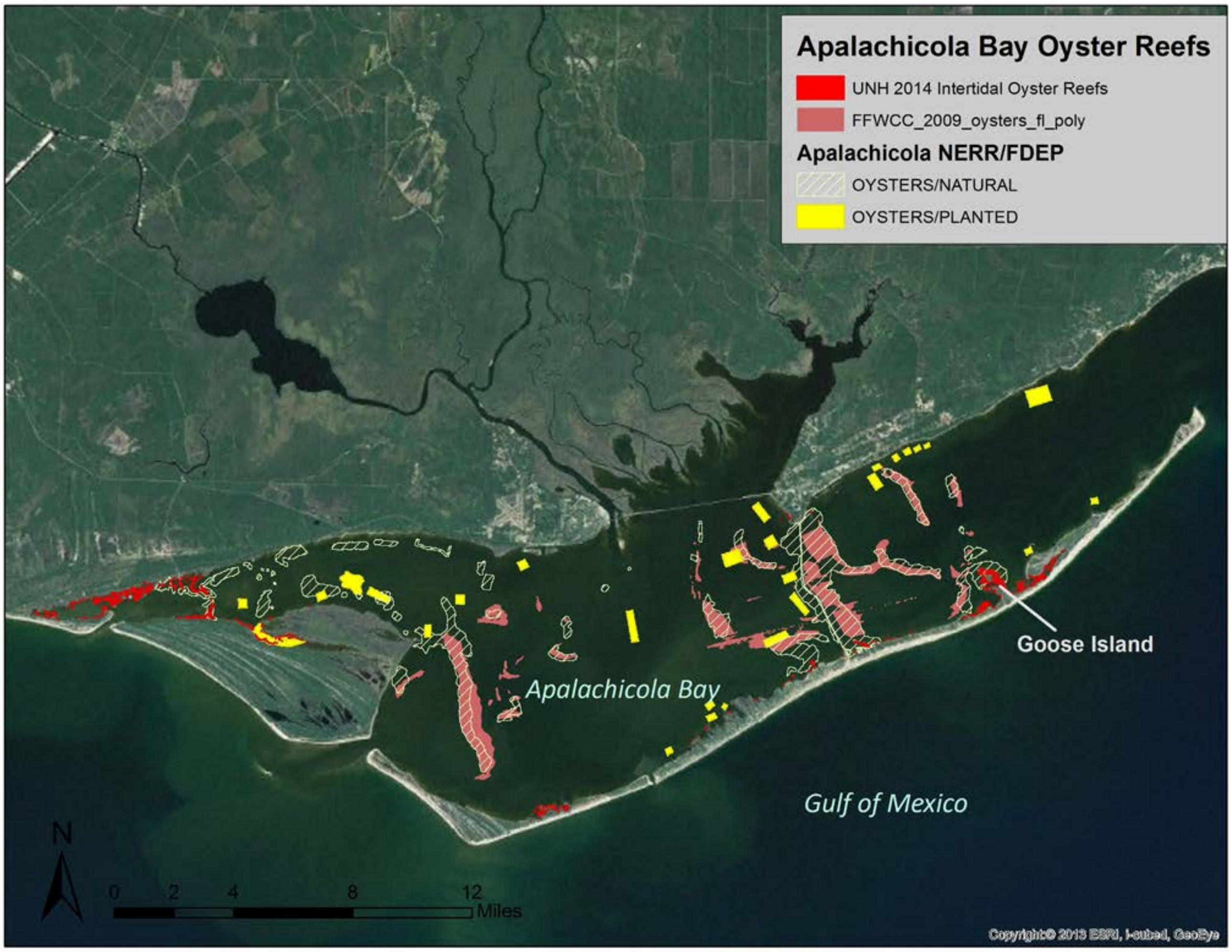

Figure 3.11. Composite map of intertidal and subtidal oyster reefs. UNH 2014 intertidal oyster reefs were mapped using 2012 satellite imagery (Grizzle et al. 2015, Grizzle et al. 2017a); FFWCC (FWC) 2009 layer compiled data from Twichell (2007) and historical data; Apalachicola NERR/FDEP data set from 2007 and 2010 aerial photographs and historical data sets (ANERR 2013). Figure from Grizzle 2017a.

from 2009–2010 (NWFWMD 2010). Oysters were mapped following the Florida Land Use and Cover Classification System (FLUCCS), which included a category for oyster bars (FLUCCS 6540; FDOT 1999). Mapped oyster reefs in Gulf and Franklin counties included intertidal oysters in Indian Lagoon, Alligator Harbor, and Ochlockonee Bay (Fig 3.1). While NWFWMD LULC maps from 2012–2013 are available, oyster bars were not mapped in those years. NWFWMD shapefiles are available for download at http://www.fgdl.org/metadataexplorer/explorer.jsp.

Apalachicola National Estuarine Research Reserve mapping and monitoring

ANERR mapped land cover and benthic cover in the reserve using high-resolution imagery from 2007 and 2010 (ANERR 2013). The minimum mapping unit was 0.02 ha (0.05 ac). Subtidal oyster reef extent was compiled by Twichell et al. (2007), and a lower-resolution data set of benthic communities was compiled using infrared photographs by the GIS group at FWC's Florida Marine Research Institute (since renamed Fish and Wildlife Research Institute) (FWC 1986). The ANERR shapefile is available for download at http://cdmo.baruch.sc.edu/get/gis.cfm.

Monitoring within ANERR includes its System-Wide Monitoring Program, which began in 1992 and monitors water quality at Cat Point, East Bay, and Dry Bar to study the effects of changing river flow on the environmental variables at those sites. More water quality stations were added at Pilot's Cove in 2015 and at Little St. Marks in 2016. Since 2002, monthly sampling for nutrient and chlorophyll-*a* began including sites throughout the bay, Apalachicola River, and offshore. ANERR also has a weather station that has collected meteorological data in East Bay since 1999. All data are available at http://cdmo.baruch.sc.edu/.

ANERR has also collaborated with multiple researchers for large-scale studies of oyster populations in relation to physical parameters within Apalachicola Bay. Petes et al. (2012) looked at oyster mortality in relation to salinity, temperature, and presence of dermo. They found that oysters suffered more disease-related mortality in high-salinity conditions, especially during warmer months, and that vulnerability was size specific; larger oysters were more susceptible. Kimbro et al. (2017) studied the effects of salinity on predation rates by the oyster drill on oysters in Apalachicola Bay; Pusack et al. (2018) further studied the impacts of predator density on predation rates.

FWC oyster map compilation

FWC has compiled many of the maps described above to create a comprehensive oyster map for Apalachicola Bay (compilation map shown in Fig. 3.1). Data sets include those from FWC (1986), Twichell (2007), NWFWMD (2010), ANERR (2013), and Grizzle (2017a). The compilation is available for download at http://geodata.myfwc.com/datasets/oyster-beds-in-florida.

Apalachicola Bay restoration mapping and monitoring

An oyster cultch placement project in Franklin County was funded by a Gulf Coast Ecosystem Restoration Council grant (GCERC 2016). This project is a continuation of a Deepwater Horizon Natural Resource Damage Assessment (NRDA) Phase III Early Restoration project and National Fish and Wildlife Foundation project. The project involved the placement of suitable cultch on depleted oyster reefs to promote new oyster colonization. The coordinates and description of these restoration efforts can be found in the project report (FDACS 2017b). Approximately 72,000 m^3 (95,000 yd^3) of lime rock aggregate were deposited onto an estimated 128 ha (317 ac) of depleted reefs in the fall of 2017. Site selection and cultch placement were coordinated through FDACS. FDEP's Central Panhandle Aquatic Preserves office is monitoring the success of this restoration effort. Cultched reefs will be mapped in the Apalachicola Bay system to depict the extent of enhanced oyster reefs.

A second cultch restoration project focused in Apalachicola Bay was initiated in 2014 and is also funded by oil spill reparation funding through the National Fish and Wildlife Foundation. This ongoing five-year project is a collaboration between FWC, the University of Florida (UF), FDACS, and UNH. The initial component of the project was overseen by FDACS and involved the placement of fossil shell at three experimental sites in Apalachicola Bay. At each of those experimental sites, five 2-ac parcels were delineated and cultched at different shell densities (0, 100, 200, 300, and 400 yd^3/ac) in order to identify optimal shell density for future restoration efforts. Following construction, UNH and Substructure Inc. conducted acoustic mapping and ground truthing of the experimental sites. FWC and UF are monitoring oyster density, size distribution, and oyster health and condition assessments. The coordinates and description of cultching efforts can be found in FDACS's final report summarizing its component of the project (FDACS 2015b). Details and results from the acoustic mapping component can be found in UNH's final report (Grizzle et al. 2017b).

Fishery disaster recovery project

Ongoing efforts for the recovery of Apalachicola Bay following the collapse of the oyster fishery in 2012–2013 are collaborative between the Florida Department of Economic Opportunity, FDACS, FDEP, and FWC. The ongoing monitoring component is conducted by FWC and includes pre– and post–commercial season metrics of oyster density at 15 oyster reefs located throughout the bay. In addition, monthly measures of larval settlement rates are recorded at those same reefs. The monitoring component also included a fishery-independent survey of oysters throughout Apalachicola Bay (mentioned in the Ecology section). Survey locations were randomly selected from areas deemed likely oyster habitat based on shapefiles and data sets. A total of 161 stations were sampled, and results indicate that many areas considered potential oyster habitat have experienced substantial loss of settlement substrate (Parker 2016).

FWC oyster population monitoring

In July 2015, the State of Florida provided funding for the establishment an annual monitoring program for Apalachicola's commercially fished oyster reefs for fishery management purposes. This program continues the annual oyster density and size monitoring that had been conducted by FDACS since 1990. In addition, the FWC program conducts monthly measures of oyster condition, dermo prevalence and intensity, reproductive development, and incidence and severity of shell pest infestations. The FWC program will continue to monitor monthly larval settlement rates after the Fishery Disaster Recovery Project concludes in 2019.

Modeling efforts in Apalachicola Bay

A series of papers has been published concerning the modeling of multiple abiotic parameters in Apalachicola Bay. Several directly relate their findings to the oyster population. Models include oyster population as a function of freshwater input (Livingston et al. 2000), wind effects on salinity (Huang et al. 2002), impacts of sea-level rise on salinity (Huang et al. 2014), oyster growth rate as a function of hydrodynamic models (Wang et al. 2008), and impacts of sea-level rise on salinity and oyster growth (Huang et al. 2015). Singh et al. (2015) modeled the impact of climate variability on baseline flow within the ACF river basin.

Environmental Sensitivity Index maps

Environmental Sensitivity Index (ESI) maps depict coastal zone natural resources. These maps are designed for use in damage evaluation, prevention, and clean-up for oil spills. Areas are mapped on a scale of sensitivity based on potential exposure, biological productivity, and ease of cleanup. ESI maps depict the locations of oysters and several other shellfish species in low, medium, and high concentrations. These concentration categories are subjective and based on the opinion of local experts. Oyster mapping data for northwest Florida was published in 1995 (RPI 1995). More information and ESI mapping data can be found at http://ocean.floridamarine.org/esimaps/.

Disease monitoring

The prevalence and intensity of dermo in the eastern oyster are monitored in several locations in Apalachicola Bay by the Oyster Sentinel, established by Thomas Soniat at the University of New Orleans. Monitoring locations and data are available at http://www.oystersentinel.org. Monitoring includes water temperature and salinity.

Contaminant monitoring

Oyster samples from Cat Point Bar and Dry Bar in Apalachicola Bay are included in the National Oceanic and Atmospheric Administration's (NOAA's) Mussel Watch program, which monitors sites around the United States for organic and inorganic pollutants. Oysters from Apalachicola Bay had moderate levels of arsenic and mercury (Kimbrough et al. 2008).

Recommendations for mapping, monitoring and management

- Design and implement periodic and extensive mapping of subtidal and intertidal reefs that is both practical and sustainable. While portions of intertidal reefs have recently been mapped and characterized, there has been no comprehensive mapping of the bay's entire subtidal oyster reef system since Swift (1897). Such a program should yield data that can be coupled with fishery-independent monitoring of the condition of the reefs in all areas of the bay to more fully assess the bay's oyster resources.
- Monitor the condition of the bay's oyster reefs, including harvested and nonharvested reefs. Fishery-dependent data are limited to reefs open to fishing, so fishery-independent monitoring should be expanded to include adequate sampling of both harvested and non-harvested reefs (Havens et al. 2013). The resulting data should be coupled with mapping programs to improve understanding of spatial and ecological relationships between the bay's oyster reefs and environmental variability.
- Better manage the fate of oyster shell removed during harvest. The importance of adequate hard substrate for larval settlement, and thus long-term sustainability of oyster reefs, has long been recognized (Swift 1898). Shelling programs have been conducted in Florida by FDACS, but no permanent funding source exists. These programs have focused on Apalachicola Bay and, more recently but to a lesser extent, St. Andrew and Pensacola bays, but they have relied on grants. The importance of shelling programs to oyster management is well established (Pine et al. 2015). Many questions remain, however, with respect to details in program design (Havens et al. 2013). Ongoing research is aimed at assessing optimal densities for deployment of fossil shell, but research also is needed on where shell plantings should be located, the types of substrate (e.g., fossil shell, recycled seasoned shell) that are most effective, and how to spread the shell.
- Investigate management needs and social issues related to salinity and river discharge requirements for Apalachicola Bay. Increased salinity as a result of reduced river discharges to the bay and sea-level rise is a major threat to oysters in this region, but also one of the most difficult to address. Litigation continues for water rights between states in the ACF watershed (Florida v. Georgia 2018, Pittman 2018). Although there has been a substantial amount of research on how oyster popu-

lations respond to increasing salinity, including research in Apalachicola Bay, research is also needed on how to address such complicated social issues from a management perspective.

- Continue research on the ecological roles of oysters in Apalachicola Bay. The bay's oyster resource has historically been managed almost entirely as a resource for human harvest, but oyster reefs also provide habitat, improve water quality through filter feeding, and are components of the estuary food chain. More information is needed particularly on how oyster harvest practices impact these ecological roles, and on how to optimize ecosystem functionality in a heavily harvested estuary.
- Further explore the role of oyster aquaculture in the bay. Oyster farming and oyster fishing are not mutually exclusive, but the tradition in Apalachicola Bay has not included aquaculture. In contrast, oyster farming is becoming increasingly common in other Florida estuaries, such as nearby Apalachee Bay and Alligator Harbor (FDEP 2018; Reiley 2018). The use of Territorial User Rights Fisheries (TURFs; Prince et al. 1998), in which oysters are harvested from areas leased to individual oyster farmers rather than from common-use public reefs (Havens et al. 2013, Camp et al. 2015), should also be explored. Individual leases help prevent unsustainable fishing practices and shell removal and encourage stewardship of reefs for long-term use, such as shelling to replace lost substrate.
- Involve all relevant state agencies, experts from academic institutions, and community organizations such as the Seafood Management Assistance Resource and Recovery Team (SMARRT) to develop an oyster management plan for the long-term well-being of oyster populations and the oyster industry in Apalachicola Bay. Although the present oyster shell height limit of 76 mm (3 in) is appropriate, it needs to be better enforced. Harvesting sublegal-size oysters is detrimental to future Apalachicola Bay oyster populations (Havens et al. 2013). Spatial restrictions and temporary closures need to be enforced and respected, including continuation or implementation of on-land and on-water checks. A bag tax used to fund research and monitoring programs that ended in the 1990s could also be reinstated (Pine et al. 2015).
- Enhance community outreach with partnerships in research, policy development, and education. The most effective policies will be those that result from a broad support base and are responsive to changes and new knowledge.

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Apalachicola River Wildlife and Environmental Area: https://myfwc.com/recreation/lead/apalachicola-river/

St. Vincent National Wildlife Refuge: https://www.fws.gov/refuge/st_vincent/

St. George Island State Park: https://www.floridastateparks.org/parks-and-trails/dr-julian-g-bruce-st-george-island-state-park

Apalachicola Bay Aquatic Preserve: https://floridadep.gov/rcp/aquatic-preserve/locations/apalachicola-bay-aquatic-preserve

Commercial oyster fishing regulations: https://myfwc.com/fishing/saltwater/commercial/oysters/

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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

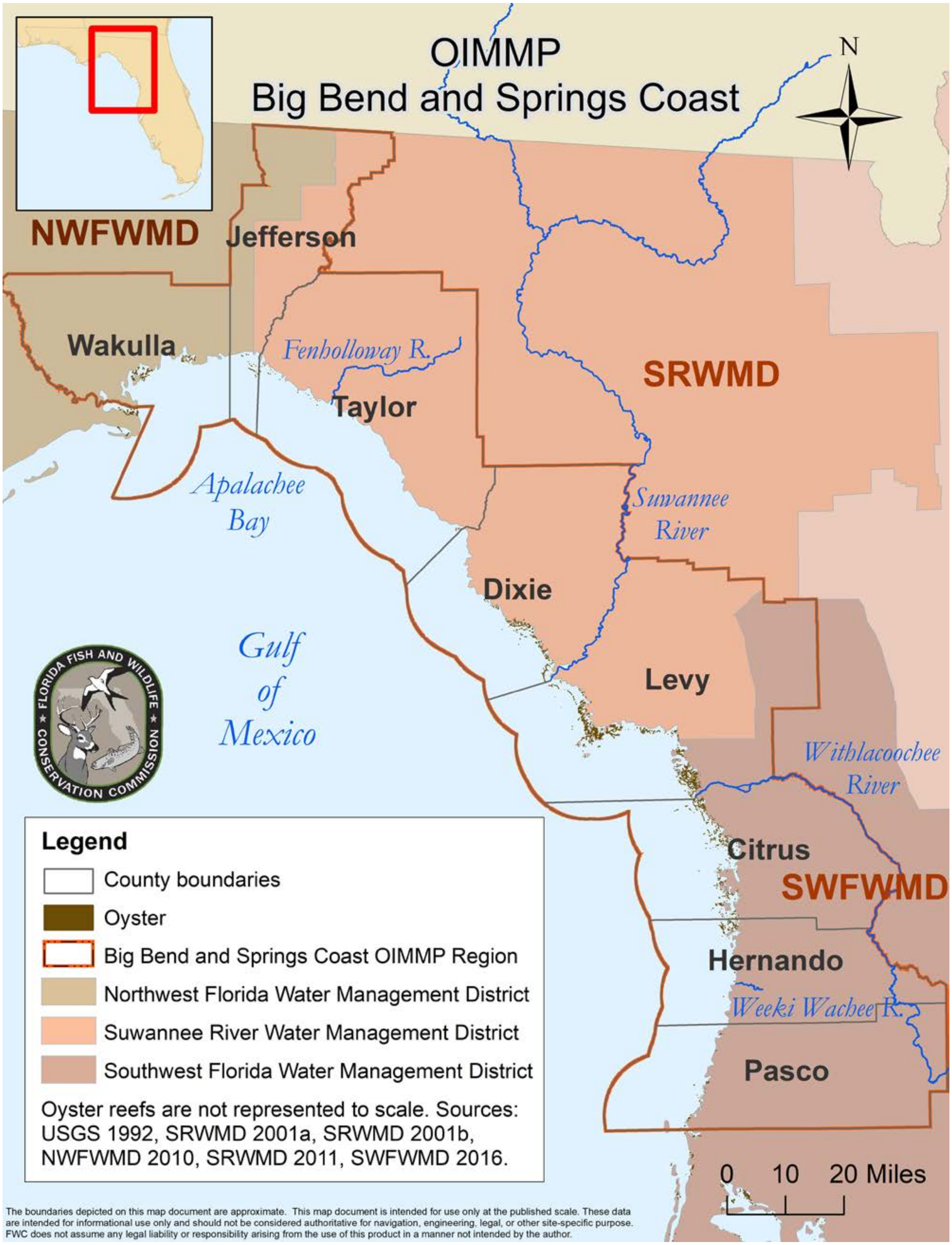

Figure 4.1. Oyster extent in the Big Bend and Springs Coast region.

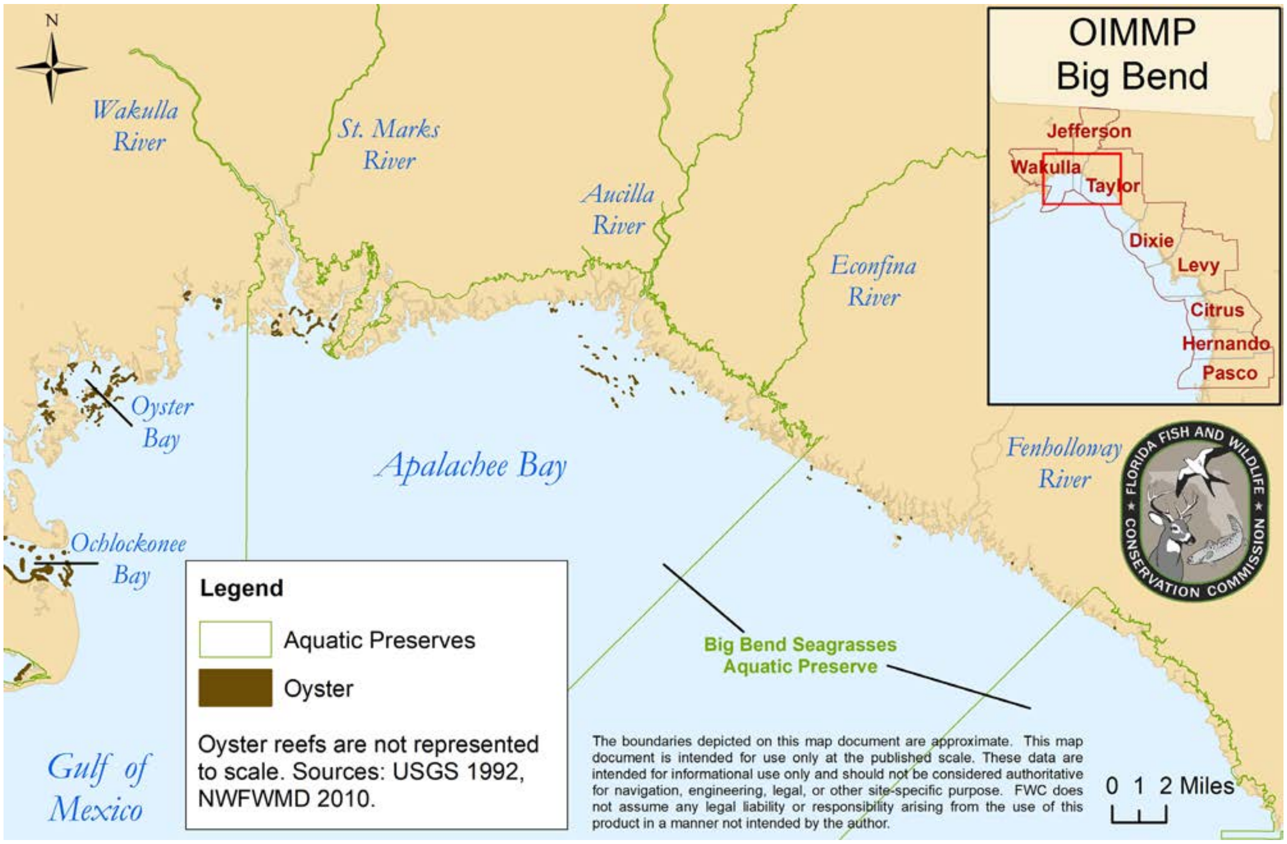

Figure 4.2. Oyster extent in the Big Bend of Florida. Oyster mapping sources: USGS 1992 (made from 1992–1993 aerial photographs) and NWFWMD 2010 (made from 2009–2010 photographs).

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,

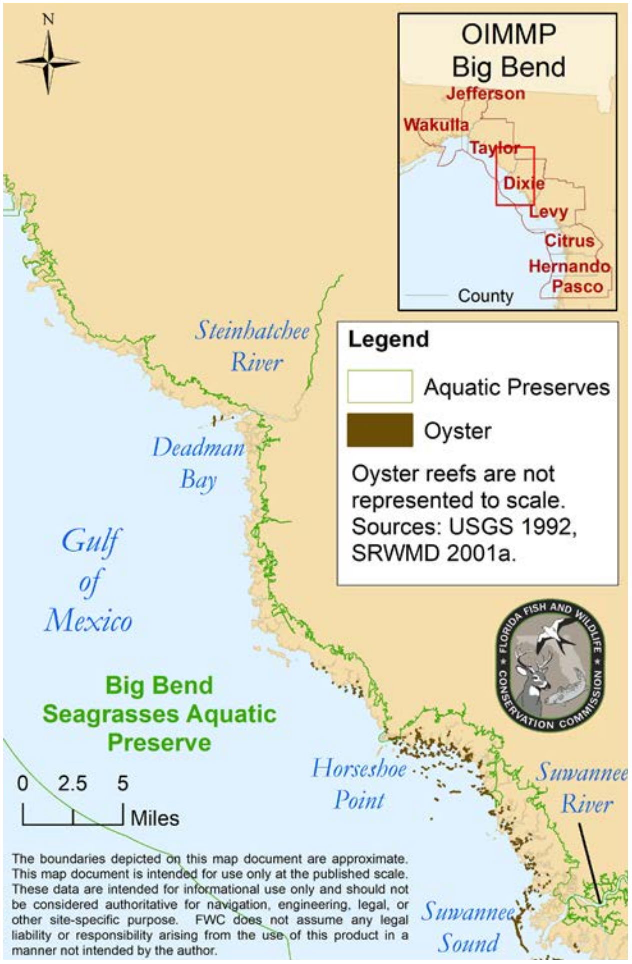

Figure 4.3. Oyster extent in Taylor and Dixie counties. Oyster mapping sources: USGS 1992 (made from 1992–1993 aerial photographs) and SRWMD 2001a (from 2001 photographs).

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-

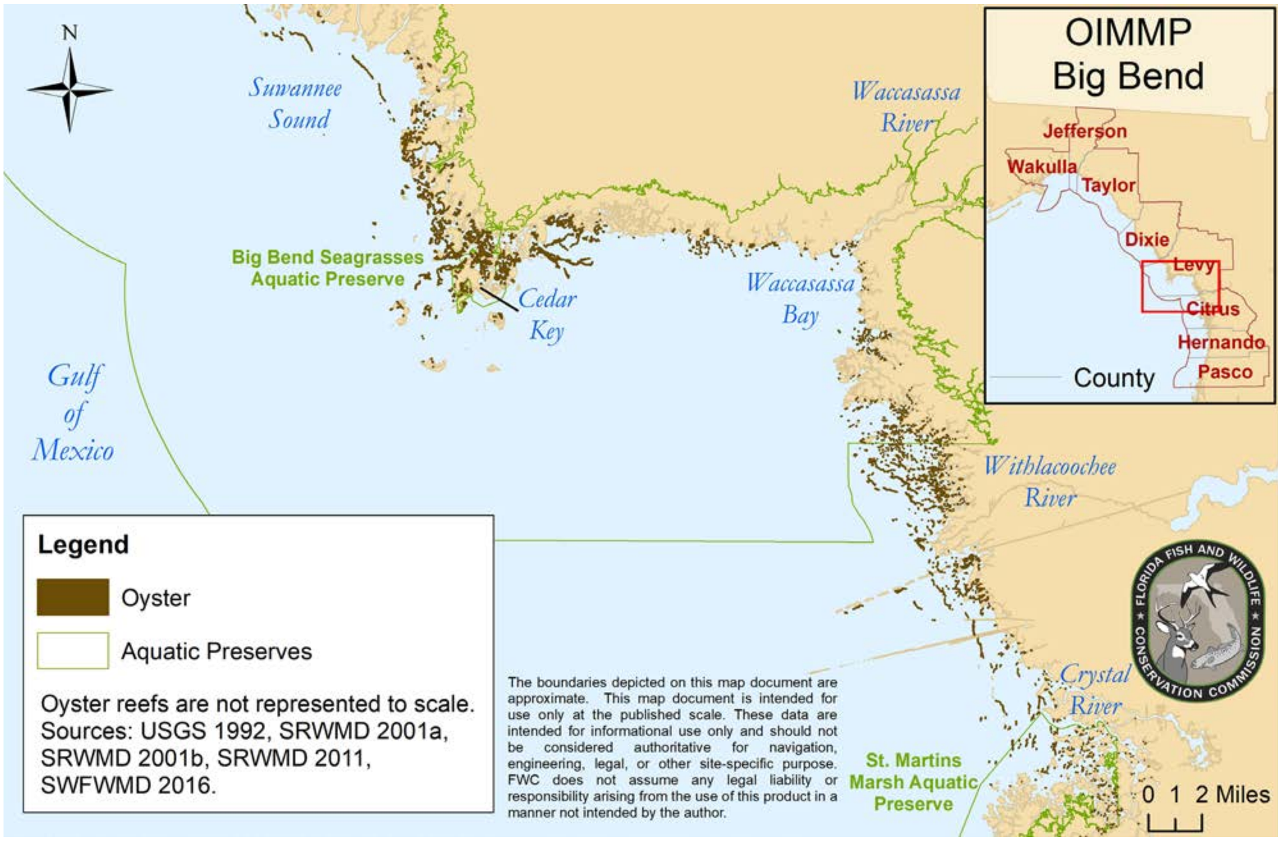

Figure 4.4. Oyster extent in Levy and Citrus counties. Oyster mapping sources: USGS 1992 (made from 1992–1993 aerial photographs), SRWMD 2001a and 2001b (from 2001 photographs), SRWMD 2011 (from 2010–2011 photographs), and SWFWMD 2016 (from 2016 photographs).

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.

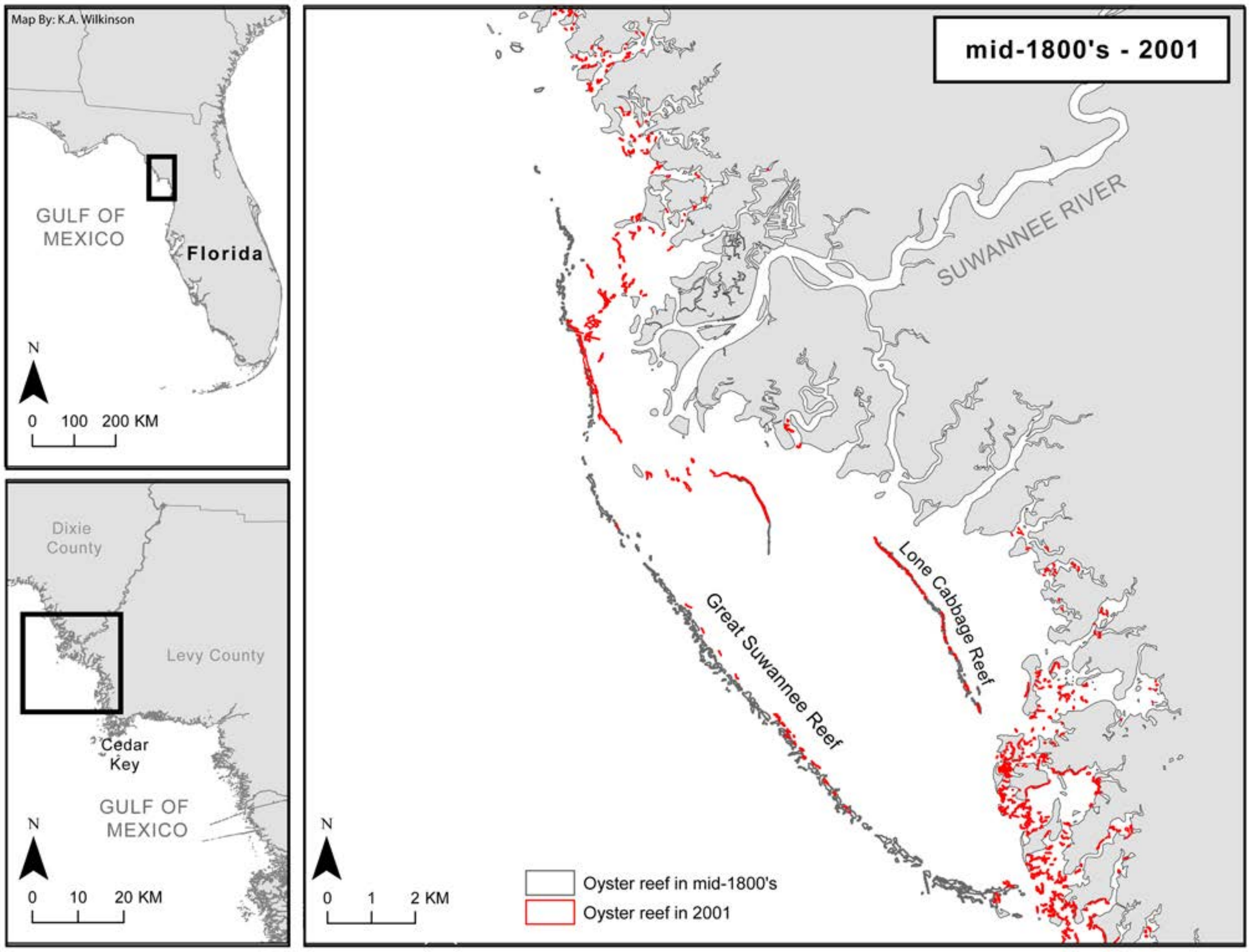

Figure 4.5. Oyster reef extent in Suwannee Sound in the mid-1800s (gray) and 2001 (red). Map by Krystan A. Wilkinson. Data sources: Raabe et al. 2004 (made from 19th century topographic sheets), SRWMD 2001a (made from 2001 photographs). Note that the increase in oyster extent along the coast in 2001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

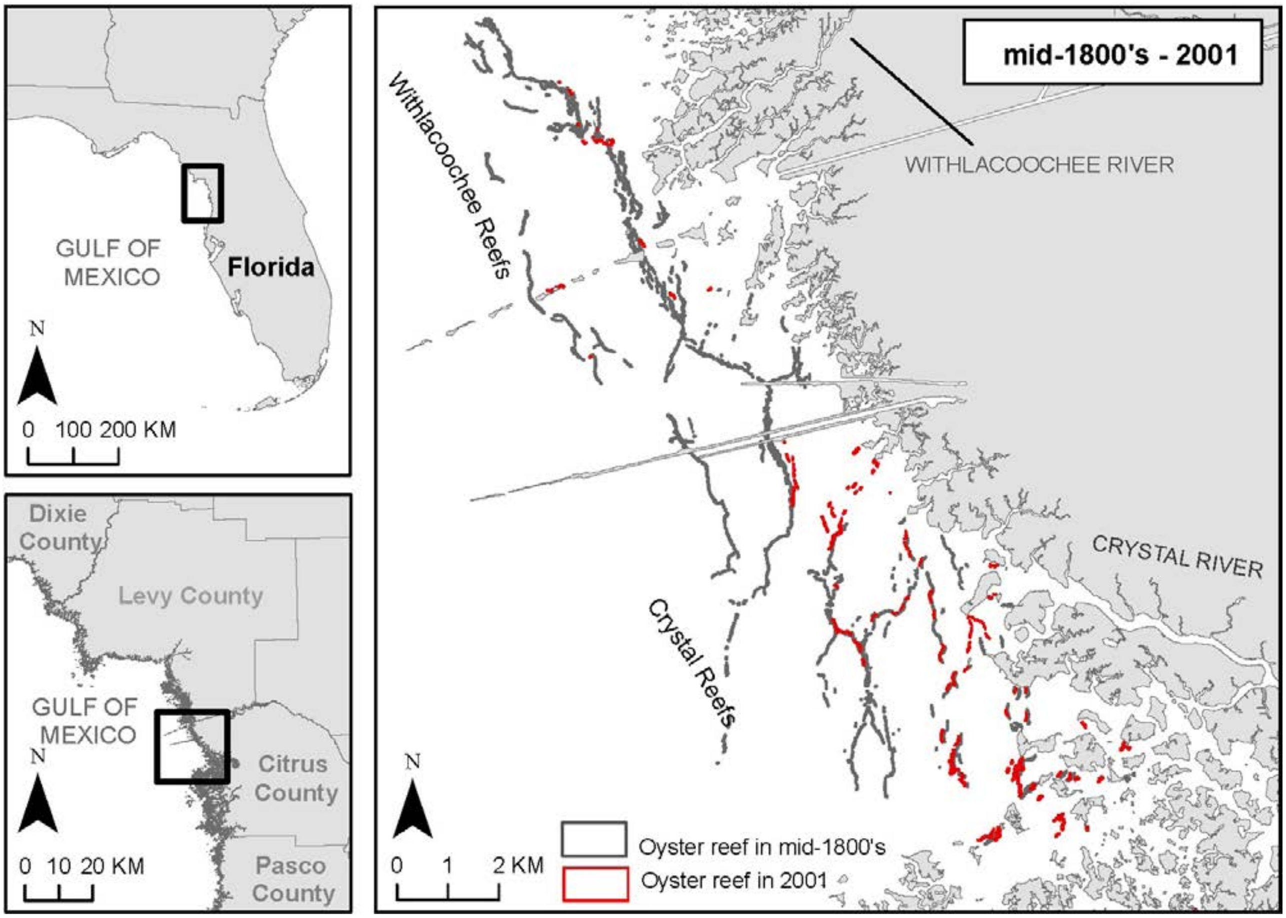

Figure 4.7. Oyster reef extent offshore of the Withlacoochee and Crystal rivers in the mid-1800s (gray) and 2001 (red). Map by Stephen Hesterberg. Data sources: Raabe et al. 2004 (made from 19th century topographic sheets), SRWMD 2001a (made from 2001 photographs).

- **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.

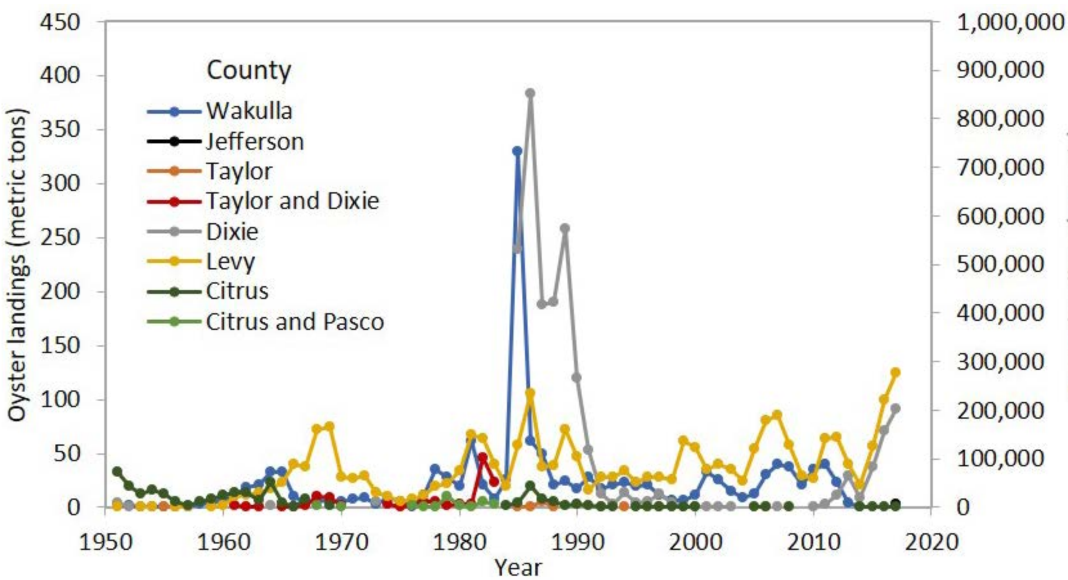

Figure 4.8. Commercial harvest of oyster meats from counties in the Big Bend and Springs Coast Region. Oyster landings before 1986 were collected under a voluntary reporting system. Data source: summary of Florida commercial marine fish landings (see Appendix A) and FWC 2018.

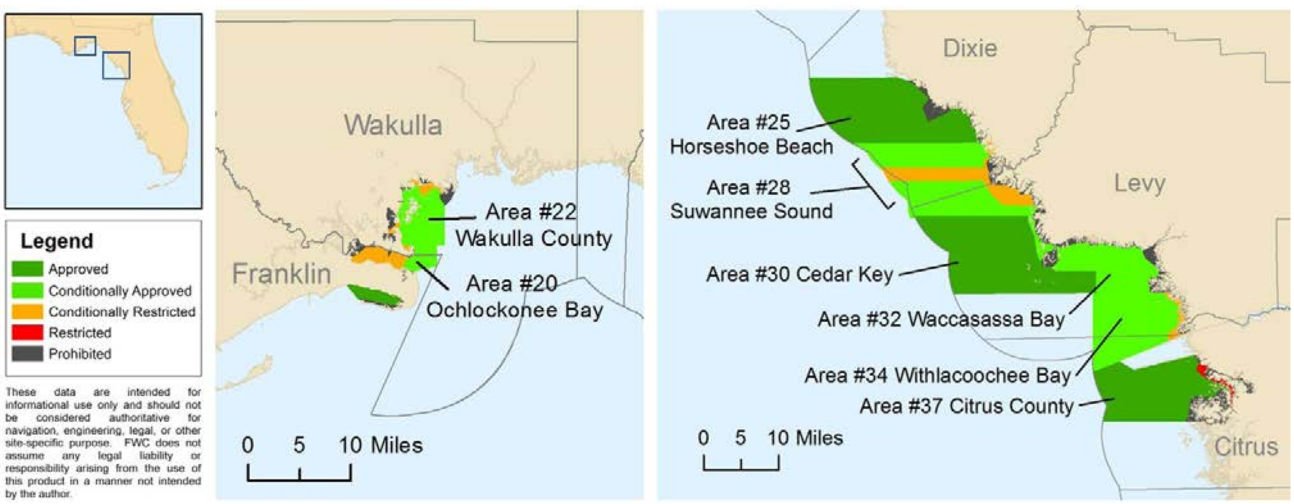

Figure 4.9. Shellfish harvesting areas in the Big Bend and Springs Coast. Data source: FDACS 2017.

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 corresponding 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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Big Bend Seagrasses Aquatic Preserve: https://floridadep.gov/fco/aquatic-preserve/locations/big-bend-seagrasses-aquatic-preserve

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Chapter 5
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Description of the region

Tampa Bay

Located on the central Gulf coast of Florida, Tampa Bay is Florida's largest open-water estuary with a surface area of 1,036 km^2 (400 mi^2) (Fig. 5.1; TBEP 2017). The bay receives freshwater flow from four major rivers (Hillsborough, Alafia, Manatee, and Little Manatee) and more than 100 small tributaries. Tampa Bay has a large, permanent connection to the Gulf of Mexico and has strong tidal currents at the mouth of the bay. There are five passes that connect to the Gulf of Mexico (Pass-a-Grille Channel, Bunces Pass, Egmont Channel, Southwest Channel, and Passage Key Inlet) as well as connections to Clearwater Harbor in the north through Boca Ciega Bay and the Intracoastal Waterway. Tampa Bay connects to the south to Sarasota Bay through Anna Maria Sound. The watershed area is about 5,700 km^2 (2,200 mi^2) and includes large portions of Hillsborough, Pinellas, and Manatee counties, as well as smaller portions of Pasco, Polk, and Sarasota counties (TBEP 2017). Outside of the Tampa Bay watershed, the greater region also includes Clearwater Harbor and St. Joseph Sound in northwestern Pinellas County (Fig. 5.1). The Tampa Bay area includes many city, county, and state parks and preserves, aquatic preserves, and national wildlife refuges. The entire region lies within the Southwest Florida Water Management District (SWFWMD) and the bay has one of the 28 National Estuary Programs in the United States.

The Tampa Bay area is highly urbanized with a population of over 3 million people. Local hydrology has been altered by freshwater withdrawal from tributaries and the construction of water reservoirs in order to meet urban water demands (Yates and Greening 2011). Urban construction has led to the channelization of stormwater, which increases the rate of surface water delivery and also transports nutrients and other pollutants to the bay. By the 1970s, Tampa Bay faced severe water quality problems and habitat degradation as a result of excessive nutrient input. These nutrient inputs were reduced through upgraded wastewater and stormwater treatment, lower industrial emissions, and other improvements in Tampa and St. Petersburg, reversing trends in eutrophication and improving habitat quality throughout the bay (Greening and Janicki 2006, Holland et al. 2006, Sherwood et al. 2017).

Tampa Bay has an average depth of 4 m (12 ft); this average depth has increased by more than 5% since 1900 as a result of channel dredging and rising sea level (Goodwin 1984, SWFWMD 1999). The bay has been dredged extensively for the creation of shipping channels and approximately 51% of Tampa Bay's shoreline has been altered (Coastal Environmental, Inc. 1994). In developed areas, the shoreline is generally hardened as rip-rap or a seawall; remaining natural shorelines are dominated by mangroves. The average tidal range in Tampa Bay is 0.4 m (1.2 ft) (SWFWMD 1999).

Eastern oyster (*Crassostrea virginica*) reefs are found in intertidal areas throughout Tampa Bay and are often clustered near freshwater inputs (Kaufman 2017). Mean

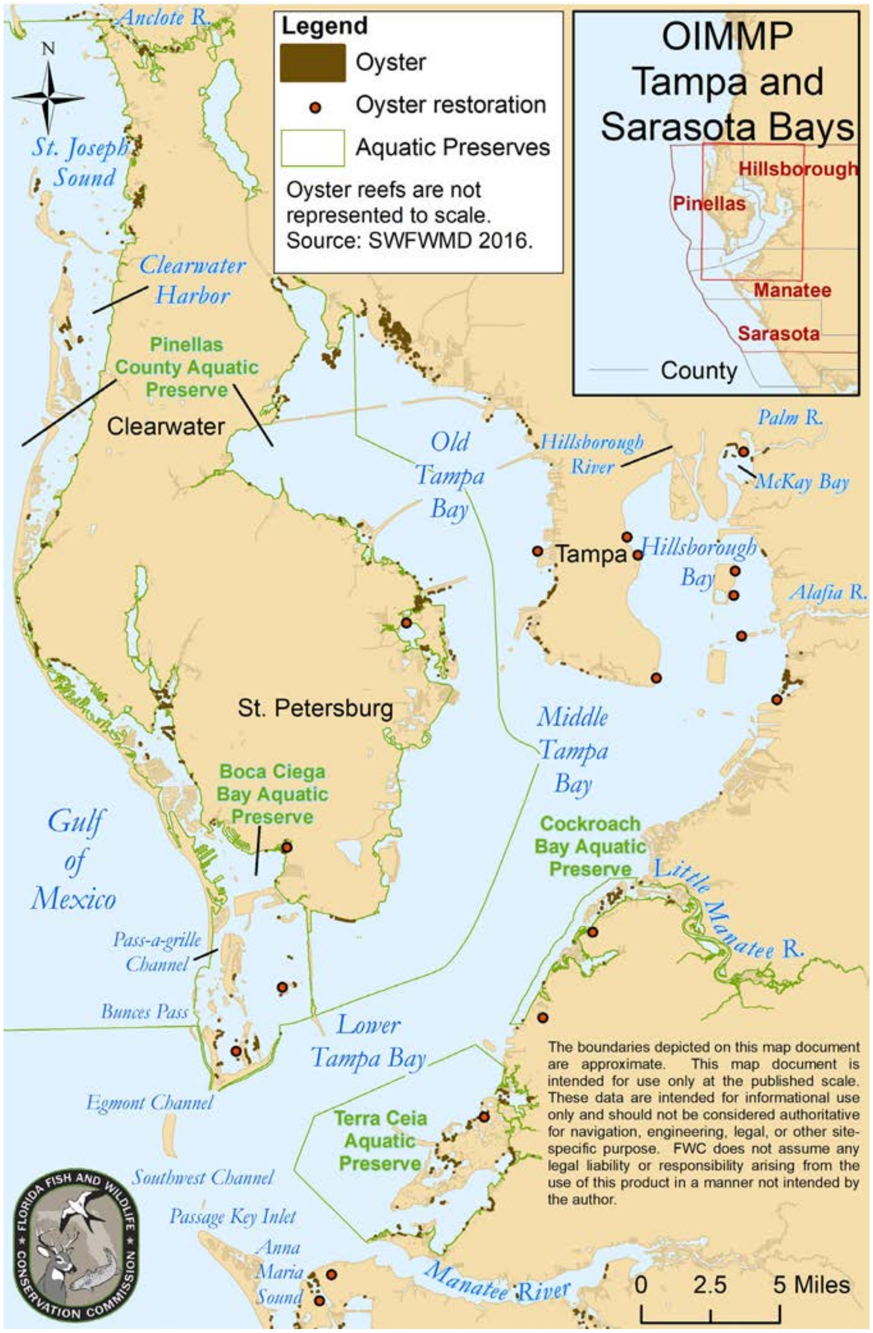

Figure 5.1. Mapped oyster extent in Tampa Bay. Oyster mapping source: SWFWMD 2016 (from 2016 aerial photographs).

monthly salinity on oyster reefs in Lower Tampa Bay ranges from 25 to 36 (Parker et al. 2013), but salinity can reach the low 20s during wet seasons (Drexler et al. 2014). Although Tampa Bay is highly urbanized, the bay does not face the extreme hydrologic impacts and salinity fluctuations seen in south Florida, making it a comparatively stable habitat for oysters (Arnold et al. 2008, Parker et al. 2013). However, tidal flow and flushing are restricted by multiple causeways in the upper reaches of Old Tampa Bay (located in northwestern Tampa Bay; Fig. 5.1). In a study of oyster density and recruitment across estuaries in central and southern Florida, Tampa Bay was found to have the most stable oyster population and a reef population density of 104–442 oysters/m^2 (10–41 oysters/ft^2) (Parker et al. 2013). As the lower bay provides relatively stable salinity, a lack of suitable hard substrate is the primary factor limiting oyster distribution (Morrison et al. 2011). The sediments of Tampa Bay are largely made up of sand and carbonate shell fragments, with clays and organic sediment becoming more prominent in the upper reaches of the bay (Brooks and Doyle 1998). Intermittent hard bottom, including subtidal oyster reefs and low-relief limestone outcroppings, are also found in the bay. Like oyster reefs, hard bottom is considered essential fish habitat and supports sponges, soft corals, algae, and live rock. Hard bottom has been mapped in selected parts of Tampa Bay (Savercool and Lewis 1994, Kaufman 2017), but the full extent of hard bottom and subtidal oyster reefs is unknown (TBEP 2017).

Oyster reefs in Tampa Bay were most recently mapped in 2016 by the SWFWMD seagrass and oyster mapping effort (Fig. 5.1; SWFWMD 2016). Oyster extent includes natural reefs and restoration sites created by multiple organizations since the early 2000s. Old Tampa Bay contained approximately 44% of Tampa Bay's oyster habitat in 2016 (Table 5.1; Kaufman 2017). Oysters that grow in close association with mangroves or seawalls are frequently not mapped or included in acreage estimates due to the difficulty of mapping these peripheral habitats using aerial imagery (O'Keife et al. 2006, Kaufman 2017). However, oyster populations on seawalls or mangrove roots often have a higher density and biomass than oysters in reefs in Tampa Bay (Drexler et al. 2014). Although oysters on mangrove

Table 5.1. 2016 mapped oyster reef extent in segments of Tampa Bay. Data source: SWFWMD, as presented in TBEP 2017.

Bay segment	Oyster reefs (ha)	Oyster reefs (ac)
Old Tampa Bay	29.8	73.6
Hillsborough Bay	5.0	12.2
Middle Tampa Bay	5.2	12.9
Lower Tampa Bay	6.3	15.6
Boca Ciega Bay	15.5	38.3
Manatee River	2.4	5.8
Terra Ceia Bay	3.2	8.0
Total	67.3	166.4

roots and seawalls are significantly smaller, they have similar condition index, fecundity, and recruitment as reef oysters and therefore are an important component of the bay-wide population (Drexler et al. 2014).

The oyster disease dermo (*Perkinsus marinus*) is relatively common year-round in Tampa Bay and has been found in 50% of oysters on natural reefs, 45% of oysters on seawalls, 42% of oysters on mangrove roots, and 38% of oysters in restoration habitats (Drexler 2011, Drexler et al. 2014). The increased prevalence of dermo on natural reefs may be linked to longer submergence times or greater age of oysters there than of oysters on seawalls or mangrove roots (Drexler et al. 2014). Although dermo infections are common, intensity is quite low and averages less than one on Mackin's (1962) scale of zero to five, where a zero indicates no infection and five is a severe infection (Arnold et al. 2008, Drexler et al. 2014). There is no evidence of MSX (*Haplosporidium nelsoni*) in Tampa Bay oysters (Arnold et al. 2008).

Several organizations including Tampa Bay Watch, Manatee County, and numerous supporting partners have conducted oyster restoration efforts since the early 2000s (TBEP 2017). In addition, intertidal restoration projects or living shorelines often directly (through use of oyster bags or reef modules) or indirectly (through oysters growing on red mangrove prop roots) result in increased oyster habitat (NOAA 2015, Brandt Henningsen pers. comm.). The Tampa Bay Estuary Program (TBEP) tracks restoration of coastal habitats, including oyster reefs, in the Tampa Bay area and provides summaries in a geo-spatial format on the Tampa Bay Water Atlas at http://www.tampabay.wateratlas.usf.edu/restoration/.

Oyster harvesting

Tampa Bay had a robust commercial fishery in the late 1800s that yielded up to 227 metric tons (500,000 pounds) annually (Finucane and Campbell 1968). Most of this harvest originated in northern parts of Old Tampa Bay. Yields dramatically decreased in the first half of the 1900s, and by the 1950s commercial oyster harvest declined to an annual yield around 2.3 metric tons (5,000 pounds) (Fig. 5.2; Finucane and Campbell 1968, Arnold and Berrigan 2002, Morrison et al. 2011). Harvest temporarily rebounded in the mid-1960s, largely as a result of cultch distribution on leased oyster grounds in Old Tampa Bay (Finucane and Campbell 1968). Cultch was also distributed in sections of Middle Tampa Bay for commercial fisheries (Whitfield 1975).

Based on records of shell production from leases, Estevez (2010) estimated that Tampa Bay once held 800 ha (1,980 ac) of oyster reefs and submerged shell. Oyster shell was extensively harvested for use in the construction industry and over 25.9 million metric tons (28.5 million US tons) of oyster shell were mined between 1931 and 1974 (Whitfield 1975). Kaufman (2017) estimated oyster reef extent in the 1970s was 75–140 ha (186–345 ac) while recent mapping efforts place current reef extent in Tampa Bay at 53–67 ha (131–166 ac) (SWFWMD 2016, Kaufman 2017).

Portions of Lower Tampa Bay are open to harvest (Fig. 5.3) contingent on levels of fecal coliform bacteria and other contaminants, which often correlate with intensity of rainfall events (FDACS 2017). The commercial harvest in Tampa Bay has often been closed since 1995 as a result of fecal coliform and other contaminant levels (Fig. 5.2; SWFWMD 1999).

Sarasota Bay

The Sarasota Bay region includes Sarasota Bay proper, Palma Sola Bay in Manatee County, and a series of smaller, contiguous bays to the south (Fig. 5.4). The bay connects with Tampa Bay to the north through Anna Maria Sound and to the Gulf of Mexico through four tidal inlets: Longboat, New, and Big Sarasota passes and Venice Inlet. Sarasota Bay is not an estuary under the influence of a major river, but rather a restricted coastal lagoon bounded by barrier islands. Phillippi Creek, which drains into Roberts Bay, is the largest of 16 tidal tributaries that flow into the system. Sarasota Bay has 135 km^2 (52 mi^2) of open water and a watershed comprising 390 km^2 (150 mi^2) (SBEP 2007). The region lies within the SWFWMD. The area from Palma Sola Bay south to Blackburn Bay comprises the Sarasota Bay Estuary Program (SBEP) while Dona and Roberts Bay in the extreme southern part of the system are within the boundaries of the Charlotte Harbor National Estuary Program (CHNEP).

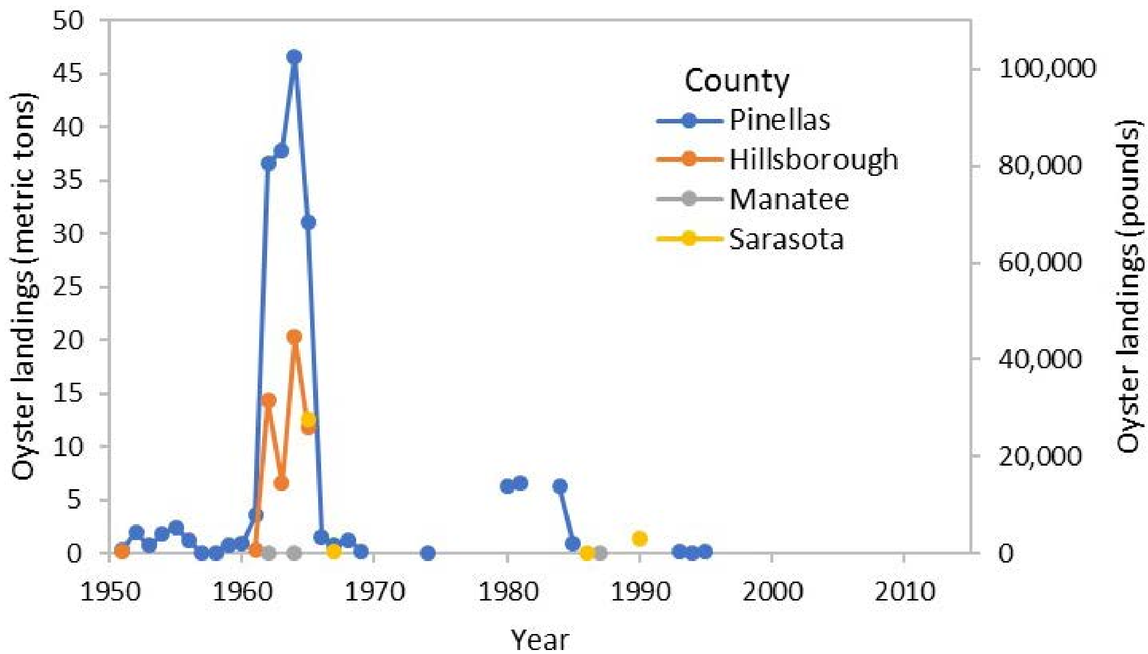

Figure 5.2. Oyster harvest in Tampa Bay (Pinellas, Hillsborough, and Manatee counties) and Sarasota Bay (Manatee and Sarasota counties). Data source: FWC 2018 and summary of Florida commercial marine fish landings (see Appendix A). Oyster landings prior to 1986 were collected under a voluntary reporting system.

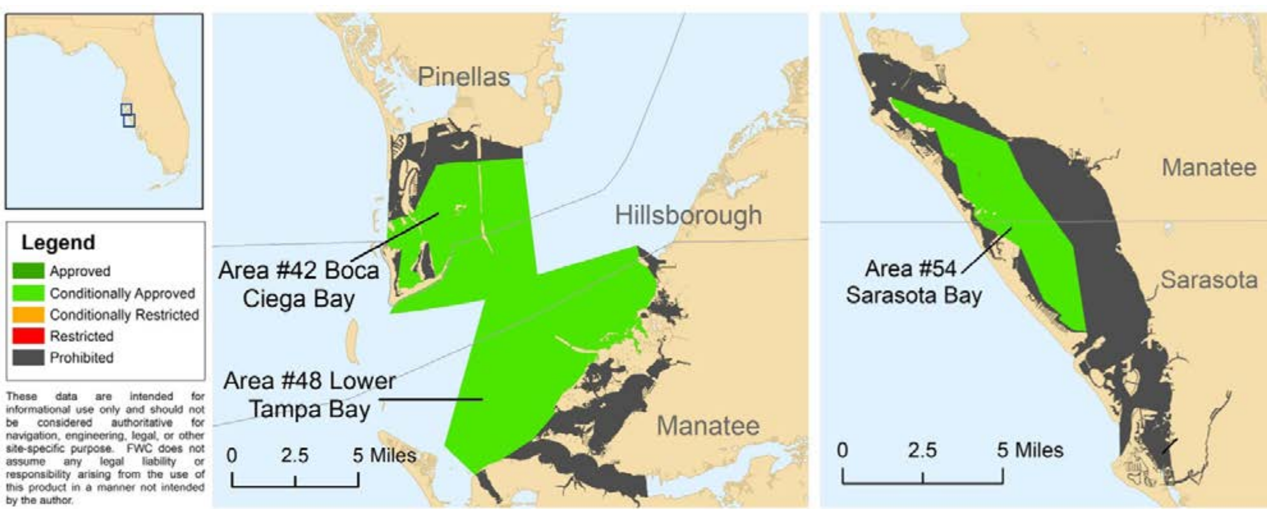

Figure 5.3. Current shellfish harvest areas in Tampa Bay and Sarasota Bay (FDACS 2017).

The cities of Sarasota and Bradenton are located along Sarasota Bay and the barrier islands are also highly developed. Many of the area's coastal wetlands have been lost to urban development and more than 160 km (100 mi) of seawalls are present today around Sarasota Bay, making up more than 80% of the bay's shoreline (SBEP 2010). As development expanded in the 1950s and 1960s, increased coastal runoff and poor wastewater management resulted in poor water quality due to excess sediments, nutrients, and other pollutants. Similarly, hydrologic alteration and canal systems around Dona and Roberts bays have tripled the historical freshwater flow into the small bays, reducing salinity and increasing nutrient load (CHNEP 2013). By the late 1980s, Sarasota Bay had reduced bivalve and fish harvests and diminished extent of seagrass beds (SBEP 2010). Nearly 1,820 ha (4,500 ac) of benthic habitat in Sarasota Bay, including oyster reefs and seagrass beds, were covered by dredge-and-fill operations (SBEP 2006). The SBEP was formed in 1989, and the community began to improve water quality and stormwater management in order to aid shoreline habitats, local fisheries, and seagrass extent (SBEP 1992). As a result of these efforts,

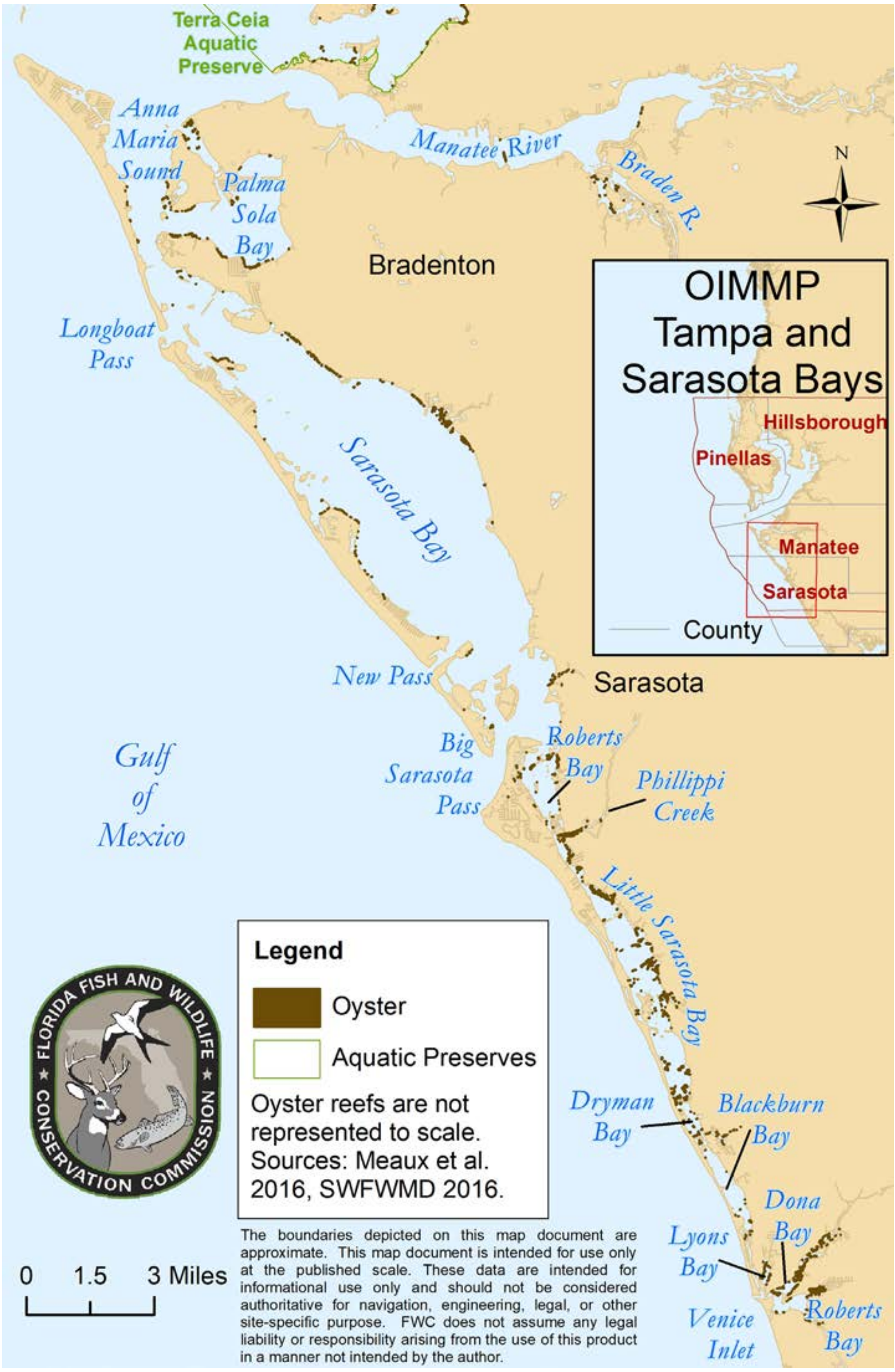

Figure 5.4. Mapped oyster extent in Sarasota Bay. Oyster mapping sources: Meaux et al. 2016 (from 2008–2012 field surveys) and SWFWMD 2016 (from 2016 aerial photographs).

nitrogen pollution decreased 64% from 1988 to 2010 and nitrogen loading from wastewater decreased 95% (SBEP 2010).

Oysters are found along the fringes of greater Sarasota Bay, particularly near areas of freshwater input where salinity is more suitable for oyster survival. Salinity within greater Sarasota Bay is generally 30–35, although in the rainy season salinity in Little Sarasota Bay varies between 15 and 30 (Sarasota County 2017). As a result of its more favorable salinity, Little Sarasota Bay contains the largest portion of the area's oysters (Table 5.2). The majority of oyster extent in the greater Sarasota Bay area is primarily made up of natural reefs, with smaller contributions from oysters growing on seawalls, rip rap, mangrove roots, and pilings (Meaux et al. 2016). Sarasota Bay has lost much of its fringing mangroves to hardened shorelines through ongoing urban development, resulting in a loss of mangrove root habitat (SBEP 2006). Like Tampa Bay, oysters in Sarasota are also substrate-limited (SBEP 2006, SBEP 2010, SBEP 2014). To provide additional substrate, restoration efforts include the addition of fossilized shells in mesh bags and prefabricated reef ball modules (SBEP 2006).

Mounds of oyster shells at Historic Spanish Point and elsewhere provide evidence that Sarasota Bay once hosted an active oyster fishery for Native Americans (SBEP 2006). While Sarasota Bay does include a conditionally approved shellfish harvesting area (Fig. 5.3), it is opened infrequently, and commercial oyster harvest was only recorded four times between 1951 and 2017 (Fig. 5.2).

Threats to oysters in Tampa and Sarasota bays

- **Habitat loss:** Large areas of oyster habitat in Tampa and Sarasota bays have been lost to dredge and fill construction, shell mining, and poorly-regulated commercial harvesting prior to the 1970s. Dredging efforts

Table 5.2. Oyster extent in regions of the greater Sarasota Bay area, as gauged by two different mapping efforts. Differences in area are due to different mapping methods and target areas (see mapping section below for details). Data sources: Meaux et al. 2016 and SWFWMD 2016.

Bay segment	Meaux et al. (2016)		SWFWMD (2016)	
	Oyster reef (ha)	Oyster reef (ac)	Oyster reef (ha)	Oyster reef (ac)
Palma Sola Bay			2.80	6.91
Sarasota Bay	2.95	7.29	9.64	23.83
Roberts Bay, Sarasota	3.74	9.23	8.63	21.33
Little Sarasota Bay	7.03	17.38	9.41	23.26
Dryman Bay	1.03	2.55		
Blackburn Bay	0.85	2.11	0.57	1.41
Lyons Bay	1.15	2.85		
Dona Bay	0.55	1.35		
Roberts Bay, Venice	0.60	1.49		
Lemon Bay	3.22	7.95		
Total	21.13	52.21	31.05	76.74

to maintain shipping channels move and suspend sediment, which can smother neighboring reefs. Hard bottom habitats and subtidal oyster reefs have also been removed for the construction of underwater pipelines and communications cables. Although mitigation regulations require the creation of new habitats, the resulting communities on the mitigated surfaces may be different than the natural habitats (TBEP 2017).

- **Altered hydrology:** Both natural and restored oyster reefs are indirectly impacted by proximity to urban development through nutrient and pollutant runoff, but water quality has improved substantially since the 1970s–1980s (Greening and Janicki 2006, SBEP 2010). In addition to water quality, changes to natural hydrology and salinity regimes by water control structures and consumptive use have also caused adverse impacts to oyster populations (Boswell 2012, CHNEP 2013). The construction of several causeways (Sunshine Skyway, Howard Franklin, Courtney Campbell, and McKay Bay) has certainly reduced tidal flushing within Tampa Bay but the net result for oysters is poorly understood.
- **Boating impacts:** Recreational boaters harm oyster reefs through erosion from boat wakes, boat groundings, collective damage from anchors (TBEP 2017).
- **Invasive species:** The Asian green mussel (*Perna viridis*) was first documented in Tampa Bay in 1999 and is presumed to have arrived as larvae in ballast water (Baker et al. 2007, Spinuzzi et al. 2013). The mussel does not have native predators and can outcompete native bivalves for space and food (Yuan et al. 2016). Asian green mussels have been observed outcompeting eastern oysters for space in at least one location in the intertidal zone in Tampa Bay (Baker et al. 2007).
- **Climate change and sea-level rise:** Changes in hydrology from sea-level rise or climate change may impact suitable areas for oyster reefs by altering the salinity regime. Additionally, most oyster reefs in Tampa Bay are intertidal. Sea-level rise may force vertical accretion on these existing reefs, result in the loss of this habitat type, or cause upslope migration of oyster reefs in areas where their movement is not inhibited by steep slopes, seawalls, or other factors. Changes in the frequency and severity of storms (both tropical and frontal systems) and weather patterns (El Niño Southern Oscillation) affect rainfall and subsequently the location and size of optimal oyster habitat. Decreases in pH levels worldwide (ocean acidification) can be detrimental to oysters and other calcifying marine and estuarine species (Hofmann et al. 2010). However, there are some indications that increased seagrass extent (as seen in Tampa Bay over the past three decades) in areas with carbonate sediment can locally combat pH trends toward acidification (Yates et al. 2016).

Oyster reef mapping and monitoring efforts

The compilation of oyster maps used in figures in this report are available for download at http://geodata.myfwc.com/datasets/oyster-beds-in-florida.

Southwest Florida Water Management District mapping

The SWFWMD has conducted seagrass mapping every two years since 1988 using a modified version of the Florida Land Use and Cover Classification System (FLUCCS; FDOT 1999) for coastal areas in the district. Subtidal habitats are mapped using natural-color 0.3-m (1-ft) resolution aerial imagery that is collected in winter. Mapped habitats include tidal flats, beaches, patchy seagrass, and continuous seagrass. The identification of monospecific oyster reefs was added in 2014 to form the oyster bar (FLUCCS 6540) classification. The most recent mapping effort was completed in 2016 (SWFWMD 2016), when SWFWMD mapped 57 ha (166 ac) of oysters in Tampa Bay (TBEP 2017) and 31.05 ha (76.74 ac) of oysters in Sarasota Bay (Table 5.2). While maps do not provide information on oyster condition (e.g., shell size, density, disease prevalence), the mapping data are important baseline information for habitat extent in the region. Oyster mapping is expected to continue at biennial intervals as a component of seagrass mapping surveys (TBEP 2017).

Tampa Bay oyster monitoring

Oysters are monitored biannually in south Florida for the Comprehensive Everglades Restoration Plan (CERP) by the Florida Fish and Wildlife Conservation Commission (FWC). Tampa Bay, Mosquito Lagoon, Sebastian River and Biscayne Bay were monitored from 2005–2007 to provide a comparison to oyster populations outside of CERP efforts (Arnold et al. 2008, Parker et al. 2013). Monitoring parameters include spatial and size distribution of adult oysters, recruitment, and growth and survival of juvenile oysters. Monitoring data also includes water quality parameters and identification of distribution and frequency of the diseases dermo and MSX (Arnold et al. 2008, Parker et al. 2013).

Tampa Bay Watch recently initiated a pilot project to monitor the outcomes of oyster restoration projects it has created over the past 20 years. Building on this work and that of other occasional projects (e.g., Drexler et al. 2014 and others), Tampa Bay Watch, in collaboration with FWC and TBEP, will create a consistent monitoring plan for natural and restored oyster reefs throughout Tampa Bay.

Investigation of automated oyster mapping

In 2004, FWC completed a TBEP-contracted effort to map oysters across Tampa Bay and develop and assess the utility of automated classification techniques for interpreting remote imagery (O'Keife et al. 2006). Hyperspectral imagery from Galileo Group flights and 2004 digital orthophoto quadrangles were used and interpreted using two automated approaches as well as traditional human interpretation. The accuracy of these three mapping efforts was assessed with ground truthing. Traditional photointerpretation was the most accurate, with 85% accuracy on free-standing reefs and 78% accuracy overall. Automated interpretation of hyperspectral imagery was not a useful tool at the time of the study but may prove more suitable with future advances in technology. Historical data from USGS 1927 topographic maps (T-sheets) and nautical charts from 1928–1988 were also geo-referenced and merged to create mosaics to facilitate tracking habitat changes over time (O'Keife et al. 2006).

Hard-bottom mapping in Tampa Bay

Kaufman (2017) created a comprehensive benthic map and report that characterized hard bottom, oyster, and tidal flat habitats for a portion of Tampa Bay. The primary objectives of the project were: 1) to create a benthic habitat map of hard and live bottom for portions of the Middle and Lower Tampa Bay and, 2) to provide data support for habitat restoration target setting of these three submerged habitat types. Approximately 105.6 ha (261.0 ac) of the assessed 8,950 ha (22,100 ac) in Lower Tampa Bay were determined to contain natural hard bottom habitat, with another 10.0 ha (24.8 ac) of artificial reef habitat (Kaufman 2017). The TBEP has an ongoing project to map hard bottom in portions of Pinellas County waters (project completion anticipated late 2019). It is recommended that this type of mapping be replicated for the entire bay.

Oyster mapping for SWFWMD minimum flows and levels

SWFWMD contracted Mote Marine Lab in 2001 to conduct field and aerial surveys of McKay Bay and the Tampa Bypass Canal to determine abundance of oysters (SWFWMD 2005). Oysters were present along the entire length of the canal and in northern McKay Bay. SWFWMD also contracted Florida Gulf Coast University in 2004 to evaluate flow levels needed to maintain appropriate salinity for oyster survival in the area. It was determined that the Tampa Bypass Canal does not need to sustain a minimum flow in order to maintain a salinity range appropriate for oysters within McKay Bay or the canal itself; however, there were times of high freshwater flow that decreased salinity beyond optimal levels for oysters within the canal (SWFWMD 2005).

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 Tampa Bay include Cockroach Bay, Hillsborough Bay, the Peter O. Knight airport, Old Tampa Bay, Papys Bayou, Mullet Key Bayou, and Navarez Park (Kimbrough et al. 2008). There are no Mussel Watch monitoring locations in Sarasota

Bay. Oysters were monitored for concentrations of heavy metals and organics in each location. Oysters contained high levels of mercury and lead as well as moderate to high levels of arsenic, copper, tin, and zinc (Kimbrough et al. 2008).

Oyster mapping in Sarasota County

Sarasota County developed methods for mapping oyster habitats with a significant focus on mapping oysters along seawalls and mangrove roots. The county used a rapid assessment technique which enabled rapid classification and large spatial coverage. The methods were published as an instructional manual (Meaux 2011) and include photographs demonstrating how to categorize types of oyster aggregations on seawalls and mangrove roots. Seventeen coastal creeks along Sarasota Bay, Venice Inlet, and Lemon Bay were mapped (example in Fig. 5.5). Results are summarized in Meaux et al. (2016) and mapping data are available at http://maps.wateratlas.usf.edu/oysters/.

Oyster monitoring in Sarasota County

The Sarasota County Oyster Monitoring Program monitors eastern oysters in greater Sarasota Bay to gauge the effect of freshwater flow on oyster condition. Monitoring locations and methods can be found in Jones (2006). Metrics include density of live and dead oysters as well as environmental parameters including rainfall, salinity, and discharge. A summary of monitoring results from 2003 to present is available at http://www.sarasota.wateratlas.usf.edu/oysters/?section=Oyster%20Monitoring%20Program.

Molluscan bioindicators of the tidal Myakka River and inshore waters of Venice

Mote Marine Laboratory conducted a survey of major mollusk species in the lower Myakka River and Dona and Roberts bays and their tributaries (Estevez 2005). Data were collected on density, shell size, and weather index values for live and dead eastern oysters. Oysters were noted

Figure 5.5. Example of the detail of oyster mapping completed in Sarasota Bay. Image credit: Meaux et al. 2016.

to be in much greater abundance in Dona and Roberts bays compared to the Myakka River (Estevez 2005).

Identification of suitable oyster habitat in Dona and Roberts bays

The CHNEP oyster habitat restoration plan was produced in 2012 in partnership with The Nature Conservancy (TNC). This plan provides an outline for identifying oyster habitat restoration goals, methods, and partnerships for the estuaries within the CHNEP study area. As part of this plan, a Restoration Suitability Model (RSM) was developed to guide future restoration decisions. The RSM uses GIS data to map the locations of suitable restoration areas on a scale of 0–100% suitability. The data layers include seagrass persistence, aquaculture lease areas, boat channels, bathymetry, and tidal river isohalines. The output for the RSM indicates that there are over 16,200 ha (40,000 ac) of highly suitable areas for oyster restoration within the CHNEP study area, including 60 ha (148 ac) within Dona and Roberts Bays (Fig. 5.6). Due to the limits of the locally-explicit data used to create the RSM model, the oyster habitat restoration plan recommends that prior to any oyster restoration, site-specific field evaluations should be conducted to further evaluate if a site is suitable for oyster restoration, and what type of methods are the most promising (Boswell 2012).

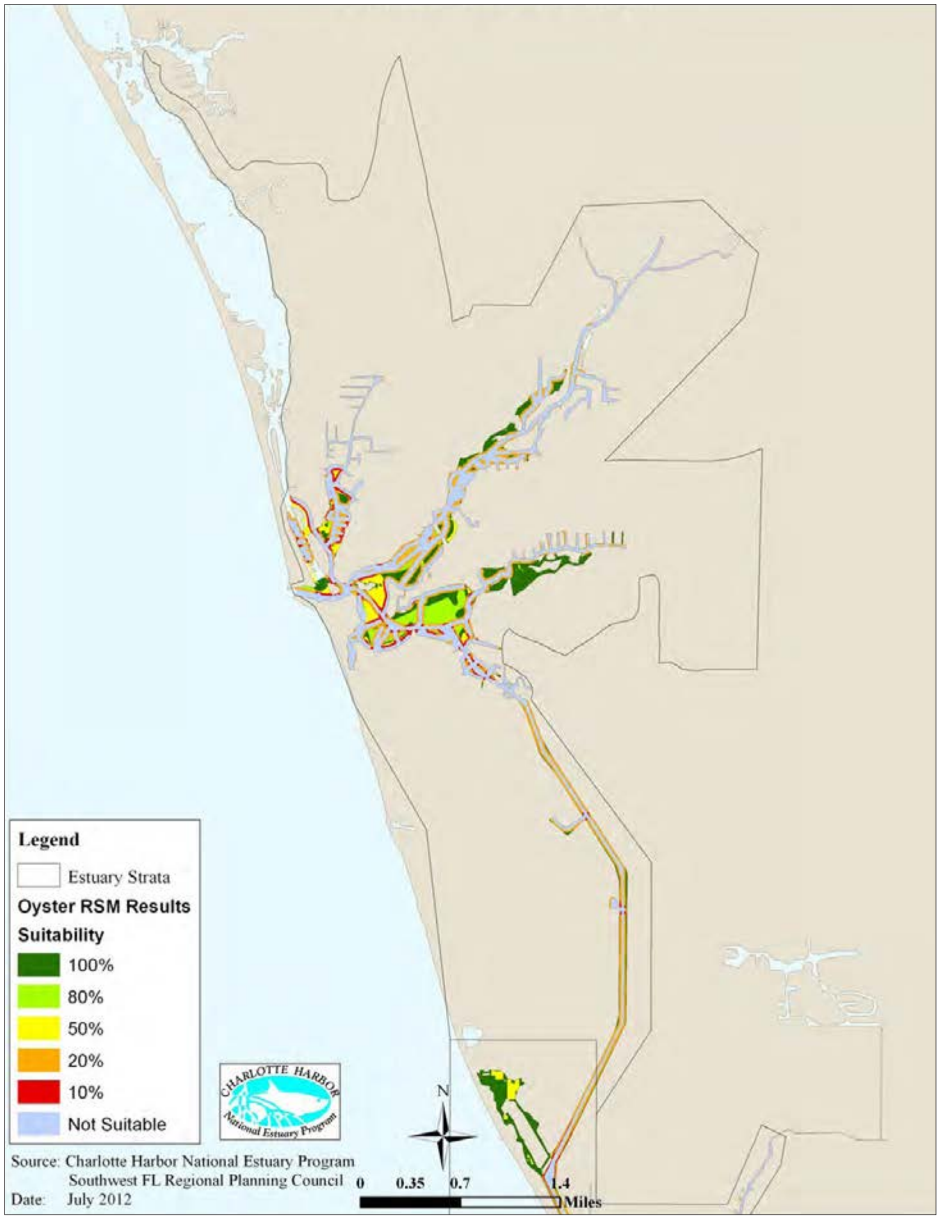

Figure 5.6. CHNEP oyster restoration suitability model results for Dona and Roberts bays. Image credit: Boswell 2012.

Recommendations for management, mapping, and monitoring

The 2017 revision of the Tampa Bay Comprehensive Conservation and Management Plan ("Management Plan"; TBEP 2017) includes 39 Actions within several different goals, such as improving water and sediment quality and improving bay habitats. Several Bay Habitat actions (BH-1, BH-2, BH-4, BH-6, and BH-8) directly address oyster and hard bottom habitat management, restoration, and monitoring. The SBEP Management Plan similarly has several actions specifically targeted at improving oyster resources (SBEP 2014). Additionally, the CHNEP Management Plan (CHNEP 2013) has been revised to include a Habitat Restoration Action Plan to address oyster habitat restoration, mapping, and monitoring in Dona and Roberts Bays Basin (CHNEP 2019). Relevant recommendations within the TBEP and SBEP

Management Plans and CHNEP oyster habitat restoration plan (Boswell 2012) include:

- Restore, mitigate, and protect priority habitats including oysters and hard bottom. Maintain proper freshwater inflow to maintain healthy oyster populations within the mouths of tidal tributaries (SBEP 2014, TBEP 2017). Continue to build partnerships to restore local oyster populations, including commercial and recreational fishing communities, with consideration for aquaculture needs (Boswell 2012).
- Continue to create oyster restoration projects in areas of the bays with suitable water quality that are substrate-limited (SBEP 2006, SBEP 2010). Increase the amount of living shorelines instead of seawalls along waterfront property (SBEP 2014, TBEP 2017).
- Create a complete rapid-assessment evaluation of the Tampa Bay shoreline for the presence of oysters on mangrove roots and hardened shorelines.
- Create and maintain a database of mitigation projects and monitoring reports of critical coastal habitats, including oyster reefs (TBEP 2017). Complete long-term monitoring on created oyster reefs to determine success of restoration efforts (Boswell 2012).
- Quantify the abundance of non-reef oysters (e.g., those growing on mangroves and seawalls). Quantify ecosystem services provided by oyster habitat in southwest Florida (Boswell 2012).
- Identify methods to improve contour plots of water quality parameters in local estuaries (Boswell 2012).
- Monitor distribution and abundance of oyster larvae in southwest Florida. Monitor biodiversity of resident and transient species associated with oyster reef communities (Boswell 2012).
- Increase targeted outreach to raise awareness regarding the ecosystem and economic value of oyster reefs in southwest Florida (e.g., press releases highlighting the benefits of oyster populations in relation to large-scale issues such as water quality, habitat conservation, water quality and water management) (Boswell 2012).

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Chapter 6
Southwest Florida

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Description of the region

The southwest Florida region includes Charlotte Harbor, Estero Bay, Rookery Bay, Ten Thousand Islands, and the Everglades. The surface water hydrology of southwest Florida has been severely affected by human modifications (e.g., impoundment and channelization), which have changed the quality, quantity, timing, and distribution of freshwater delivered to coastal wetlands and estuaries. Before development, the hydrology of the Everglades was tightly linked to a large watershed that encompassed much of central and southern Florida (Fig. 6.1; Chimney and Goforth 2001, Huber et al. 2006). Water meandered down the Kissimmee River to Lake Okeechobee, where it spilled over the southern edge of the lake into an expansive sawgrass marsh. The sheet of surface water then slowly made its way south, supporting a variety of freshwater marshes in the interior and mangroves, salt marshes, and oyster reefs along the coasts. Significant hydrological changes began in 1881, when the Caloosahatchee River was connected to Lake Okeechobee by Hamilton Disston for steamboat transportation (CHNEP 2013). In the 1950s and 1960s, the U.S. Army Corps of Engineers' Central and Southern Florida Project made major changes to the watershed in an effort to avert floods and drain wetlands for agriculture (Chimney and Goforth 2001). The Herbert Hoover Dike prevented water from seeping over the southern edge of Lake Okeechobee, stopping the sheet flow of surface water through the Everglades and instead diverting large amounts of freshwater flow into constructed channels with outlets in the Caloosahatchee and St. Lucie rivers (Fig. 6.2). These hydrological alterations resulted in markedly lower salinity and increased nutrient and sediment supply in the Caloosahatchee and St. Lucie estuaries. Meanwhile, draining and channelization of wetlands increased the severity

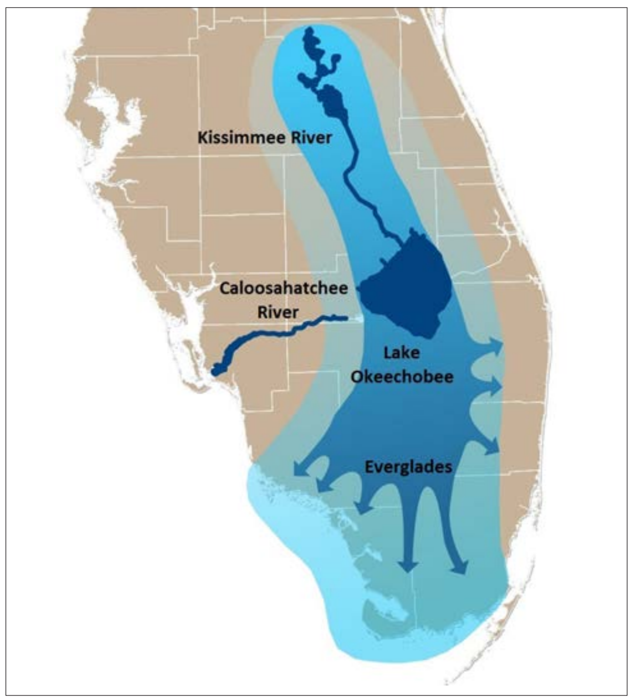

Figure 6.1. Historical surface water flow in south Florida. Figure credit: Chris Anderson, based on Chimney and Goforth 2001.

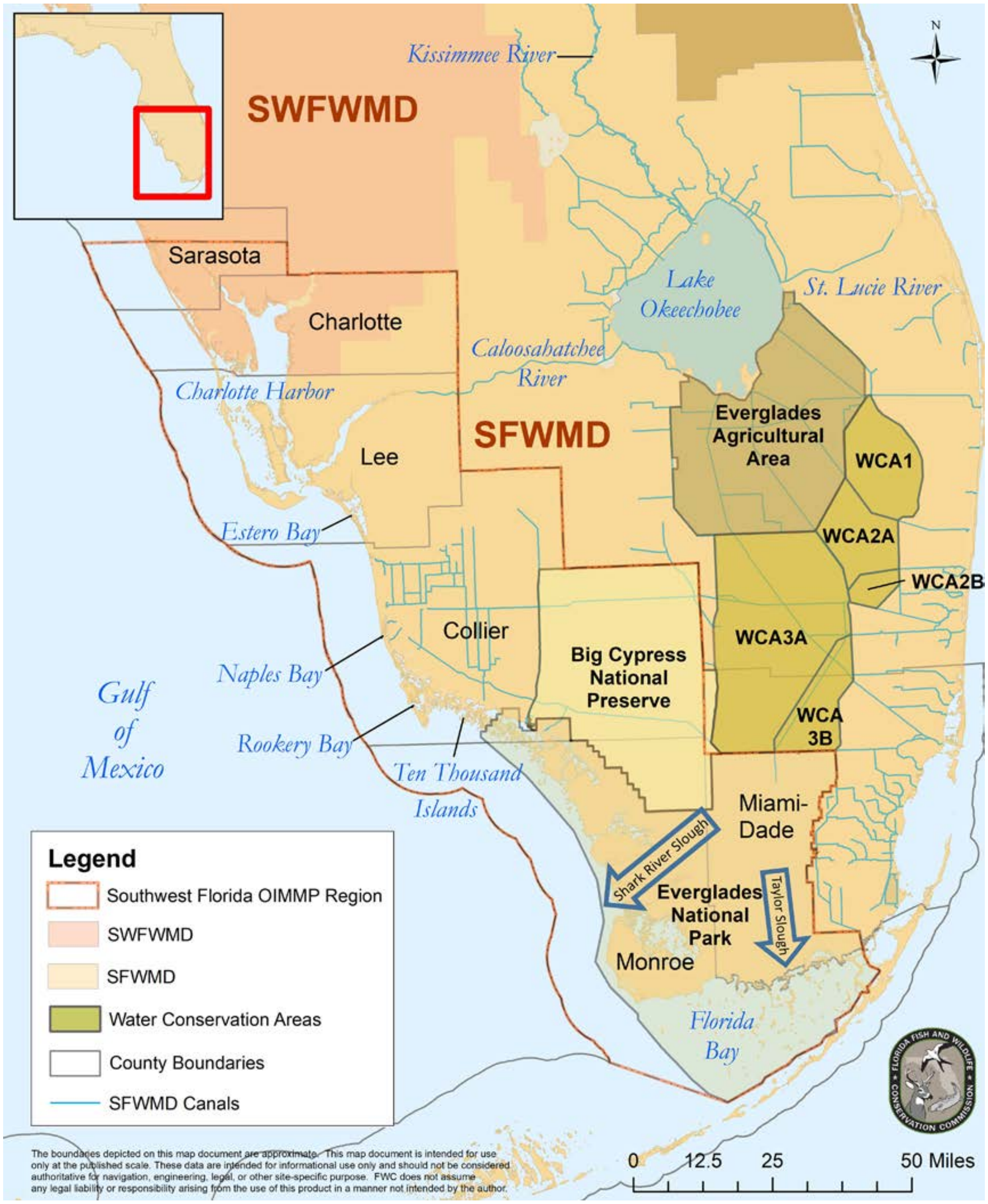

Figure 6.2. Major features impacting the hydrology of the southwest Florida region.

and frequency of droughts in other estuaries of the Everglades and Ten Thousand Islands, causing high salinity and disrupting the center of abundance of oysters in the major rivers and creeks (Chimney and Goforth 2001, Huber et al. 2006).

Awareness of the extensive environmental damage caused by this alteration of surface water flow prompted the Comprehensive Everglades Restoration Plan (CERP), which the U.S. Congress authorized in 2000. Major components of CERP include ecological management of Lake Okeechobee, decreasing channelized freshwater releases from the lake, increasing freshwater sheet flow to rehydrate the Everglades, and retaining water in reservoirs for slow release during the dry season (Huber et al. 2006, Volety et al. 2008). CERP also includes projects that focus on improving habitat for oysters, such as removing muck accumulations and providing substrate for oyster establishment (Volety et al. 2008).

Eastern oysters (*Crassostrea virginica*) are monitored as an indicator species in the Caloosahatchee estuary to gauge the impacts of CERP (Volety and Haynes 2012). Oysters provide a useful indicator of high flows because they are sensitive to prolonged periods of low salinity (below 10). Because they are sessile after settlement, they also integrate the impact of varying water quality at a location from weeks to months or even years, depending on the metric used (CERP 2004, Volety et al. 2014, CHNEP 2019). Oysters additionally provide an indicator of the overall biodiversity of an ecosystem, because biomass, density, and species richness of fish and invertebrates are all greater on oyster reefs than on unvegetated sand or mud bottoms (Tolley and Volety 2005).

Charlotte Harbor

Charlotte Harbor is a large, complex estuary bordered by barrier islands and includes the cities of Fort Myers, Cape Coral, Punta Gorda, and North Port. The estuary falls on the boundary between the jurisdictions of the Southwest and South Florida water management districts (SWFWMD and SFWMD, respectively) (Fig. 6.2). It receives freshwater input from the Myakka, Peace, and Caloosahatchee rivers along with many other minor tributaries. Management of the region is facilitated by the Charlotte Harbor National Estuary Program, CHNEP, which in 2019 was renamed the Coastal and Heartland National Estuary Partnership, and large areas are protected from development under the Florida Department of Environmental Protection (FDEP) Aquatic Preserve program (Fig. 6.3). The J.N. "Ding" Darling Wildlife Refuge complex, owned and managed by the U.S. Fish and Wildlife Service (USFWS), includes Island Bay, Pine Island Sound, Matlacha Pass, and Caloosahatchee National Wildlife Refuges (NWRs) and encompasses a significant area of mangrove islands.

Most oyster reefs in Charlotte Harbor are intertidal, although subtidal reefs are present at depths of at least 1.8 m (6 ft) in the Caloosahatchee River (Boswell et al. 2012). Reefs in open water are common in San Carlos Bay at the mouth of the Caloosahatchee, Matlacha Pass, and Pine Island Sound (Fig. 6.3). Historically, reefs in San Carlos Bay were typically elongated, usually perpendicular to the predominant direction of tidal or river flow, but today they more commonly persist as fringing reefs on low-relief spoil islands. Typical healthy oyster densities are 300–1,500 oysters/m^2 (28–139 oysters/ft^2) with shell heights of 21–35 mm (0.8–1.4 in). Many of the remaining reefs, however, have large areas with loose, scattered shell or shell fragments and appear to have lost substrate since the widespread development and land-use conversion that began in the late 1960s (Milbrandt, pers. obs.). Many of the reefs are surrounded by shallow sand bottom with submerged aquatic vegetation (SAV) growing up to the reef edge; the most common species of SAV are turtlegrass (*Thalassia testudinum)*, manateegrass (*Syringodium filiforme*), and shoalgrass (*Halodule wrightii*). Larval supply to these oyster reefs may depend on proximity to a healthy reef, but the relationship between hydrodynamics and larval transport remains poorly understood.

In addition to open-water reefs, oysters attach to mangrove prop roots and create fringing reefs along mangrove shorelines. FDEP's Aquatic Preserve and Buffer Preserve programs have protected much of the mangrove habitat in Charlotte Harbor from development. The shade provided by the canopy and the relatively stable root system creates an excellent place for oysters to thrive. However, settlement of oysters on prop roots may be limited because flow restrictions and shallow water may either prevent larvae from reaching the mangrove shorelines or limited water flow may restrict food supply.

While data are limited on the extent of oyster reefs before development, it has been estimated that Charlotte Harbor has lost 90% of its oyster reefs since the 1950s (Boswell et al. 2012, CHNEP 2019). These reefs were lost as a result of mining oyster shell for road construction, unsustainable harvesting rates, changes in surface-water hydrology, excess sedimentation, and direct habitat destruction due to coastal development. Commercial oyster harvests in Charlotte Harbor peaked in the 1960s (Fig. 6.4), and by the early 1970s much of Charlotte Harbor was closed to commercial oyster harvest due to pollution and declining oyster populations (Taylor 1974). Following a resurgence in the 1980s, oysters were not harvested commercially in Charlotte Harbor from 1992 through 2010. Landings resumed in 2011 and increased to 2.1 metric tons (4,666 pounds) in 2016. The impact of this resurgent commercial fishery on the resilience of remaining oyster reefs is a concern.

Southwest Florida contains five shellfish harvesting areas. Four are found in Charlotte Harbor and one is in the Ten Thousand Islands (Fig. 6.5; FDACS 2017). Shellfish are also harvested recreationally in the greater Charlotte Harbor region (Boswell et al. 2012). Removal of oysters for use as chum is largely unregulated in practice and does not adhere to shellfish harvesting areas.

Eastern oysters prefer a salinity range of 14–28, although they can survive brief exposure to salinity ranging from 0 to 42 (Shumway 1996, Volety et al. 2008, VanderKooy 2012). Oysters in the Caloosahatchee River Estuary must cope with the stress of freshwater releases from Lake Okeechobee that result in rapid and extended declines in salinity. Juvenile oysters are particularly susceptible to fresh water, and salinity <5 can result in 95% juvenile

Figure 6.3. Greater Charlotte Harbor mapped oyster extent, aquatic preserves, and national wildlife refuges. Oyster mapping sources: FWC and SFWMD 1999 (made from 1999 aerial photographs), Volety and Savarese 2004, and SWFWMD 2016 (from 2016 photographs).

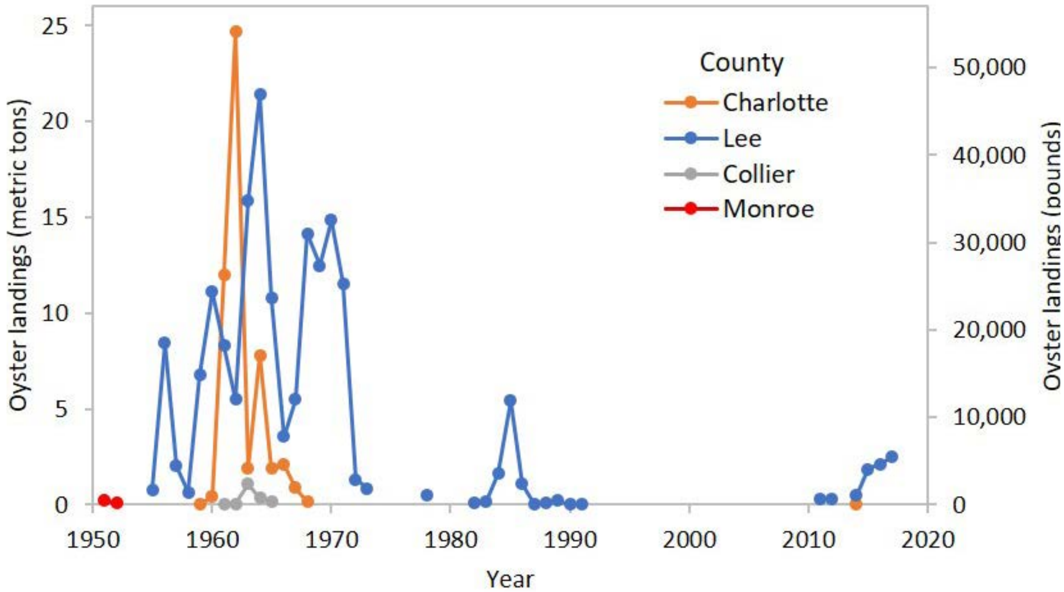

Figure 6.4. Commercial oyster landings in southwest Florida. Oyster landing data before 1986 were collected under a voluntary reporting system. Data source: FWC 2018 and summary of Florida commercial marine fish landings (see Appendix A).

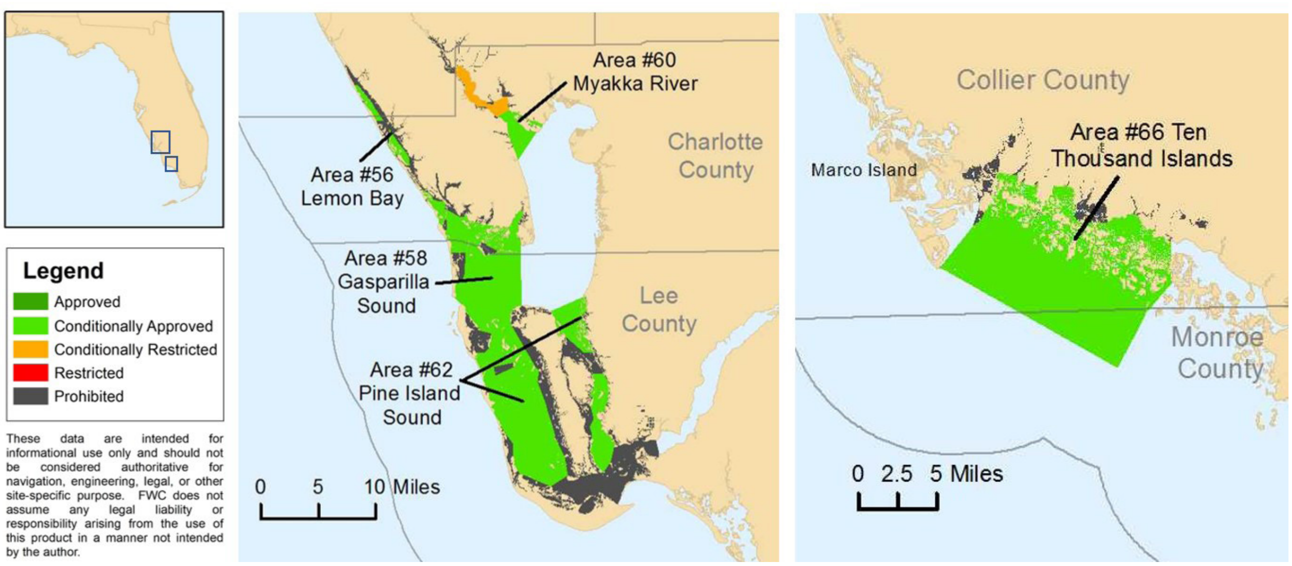

Figure 6.5. Shellfish harvesting areas in southwest Florida. Source: FDACS 2017.

mortality (Volety et al. 2003). The combination of low salinity and high flow in summer hinders larval recruitment in the Caloosahatchee (Volety et al. 2003, Volety et al. 2008). Oysters in San Carlos Bay, Pine Island Sound, and associated waters also must endure long periods of high salinity during periods of low river flow.

The CHNEP Oyster Habitat Restoration Plan (Boswell et al. 2012) was funded by The Nature Conservancy (TNC) and developed by the CHNEP Technical Advisory Committee as well as the oyster experts and stakeholders that made up the Southwest Florida Oyster Working Group. The purpose of the plan was to establish oyster restoration goals, methods, and partnerships in the region. While a 1999 mapping effort documented only 100 ha (247 ac) of oysters in the estuary (Avineon 2004), the restoration plan determined that the CHNEP study area should ideally support 400–2,400 ha (1,000–6,000 ac) of oyster reefs (Boswell et al. 2012). The plan also included an oyster restoration suitability model, which identified locations within the estuary that ranged from least to most optimal for oysters based on five key restoration factors: local seagrass persistence, aquaculture lease areas, boat channels, bathymetry, and salinity (Boswell et al. 2012). To advance oyster restoration in the Charlotte Harbor region and throughout the state, TNC facilitated a statewide team of stakeholders in the development of a new FDEP general permit specifically for low-profile oyster restoration efforts. The permit was approved by the state in 2013 and is the only state permit designed specifically for oyster restoration.

Estero Bay

South of Charlotte Harbor, Estero Bay is also lined by barrier islands and receives freshwater flow from many small rivers and creeks (Figs. 6.3 and 6.6). Much of the shoreline was protected from development and has been preserved as Estero Bay Preserve State Park, although there is extensive development on the barrier islands and uplands surrounding the bay. A large network of oyster reefs is present in northern Estero Bay, and smaller reefs are found in the southern part of this system (FDEP 2014; Fig. 6.6). The reefs in Estero Bay are subject to modified timing and delivery of freshwater flow, but to a lesser extent than the Caloosahatchee River.

The irregular shape of the coastline and mangrove islands in Estero Bay are a result of oyster reef formation and accretion of sediments in the early to middle Holocene (Savarese et al. 2004a). The sedimentation, along with shell growth and recruitment of young oysters, results in high rates of accretion and provides substrate for settlement of red mangroves (*Rhizophora mangle*). These mangroves eventually mature into mangrove islands, but evidence of past oyster reefs remains in the underlying stratigraphy (Savarese et al. 2004a).

Naples Bay

Historically, Naples Bay received the majority of its freshwater flow from rainfall-derived sheet flow and from small natural tributaries: Gordon River, Rock Creek, and Haldeman Creek. The watershed's drainage area was approximately 25 km^2 (10 mi^2) before it was developed and comprised mostly mesic and hydric flatwoods (Woithe and Brandt-Williams 2006). Mangroves dominated the shoreline of this shallow estuary, and the bay once thrived with oyster reefs, seagrass beds, and numerous fish species (Simpson et al. 1979).

Extensive development, watershed alterations, and dredge-and-fill activities since the 1950s transformed Naples Bay (Fig. 6.7) into a highly urbanized estuary (Laakkonen 2014). Residential development led to an increase in hardened shorelines in the 1950s and 1960s (Schmid et al. 2006). Navigation canals lined with seawalls and bulkheads replaced more than 70% of the mangrove shoreline. The bay has lost 90% of its seagrass beds and 80% of its oyster reefs (Fig. 6.8; Schmid et al. 2006). Today subtidal oyster reefs are present on sandy substrates in the northern part of the bay and fringing reefs are found in the southern part of the bay. Seismic profiling in Naples Bay revealed few buried oyster reefs, indicating that most of the historical reef extent was lost to dredging rather than sedimentation (Savarese et al. 2006). Today, Naples Bay remains a challenging environment for oysters due to water quality issues, muddy substrates, and suspended sediments, which can smother reefs (Savarese et al. 2006).

The natural tributaries of Naples Bay have been severely altered by urbanization and the addition of numerous canals, significantly changing the hydrology and disrupting the timing and magnitude of freshwater flow to the estuary. The construction of canal systems in residential areas has increased the perimeter of the bay by 53% and the water surface area by 23% (Schmid et al. 2006). Land use in the watershed is a mosaic of residential developments, industrial areas, and agricultural lands, which have increased the pollutant load to the bay (Woithe and Brandt-Williams 2006). Surface water that once traveled as a sheet flow through wetlands is now rapidly conveyed to Naples Bay via stormwater pipes and surface canals, resulting in degraded water quality including increased nutrients, sediments, and metals (CSF 2011).

The construction of the Golden Gate Canal (Fig. 6.7) in the 1960s to drain the Northern Golden Gate Estates increased the size of the Naples Bay watershed from approximately 25 km^2 (10 mi^2) to more than 300 km^2 (120 mi^2), drastically increasing rainy season freshwater flow into Naples Bay (Fig. 6.9; Woithe and Brandt-Williams 2006, CSF 2011). Average annual discharge from this weir-controlled canal is 7 m^3/sec (250 cfs), with flows exhibiting high seasonal variability ranging from 0 m^3/sec (0 cfs) in the dry season to 40 m^3/sec (1,400 cfs) in the wet season (CSF 2011). In the dry season (November–May), salinity is relatively uniform from Gordon Pass through the Gordon River. In the wet season (June–October), freshwater flow from the Golden Gate Canal causes severe stratification of the water column, which lowers benthic dissolved oxygen, inhibits vertical mixing, and decreases water clarity (Woithe and Brandt-Williams 2006). In addition, these large wet-season freshwater influxes greatly reduce salinity in the bay. When combined with the other stressors, this results in harmful impacts to the aquatic biota, including declines in oyster populations (Simpson et al. 1979).

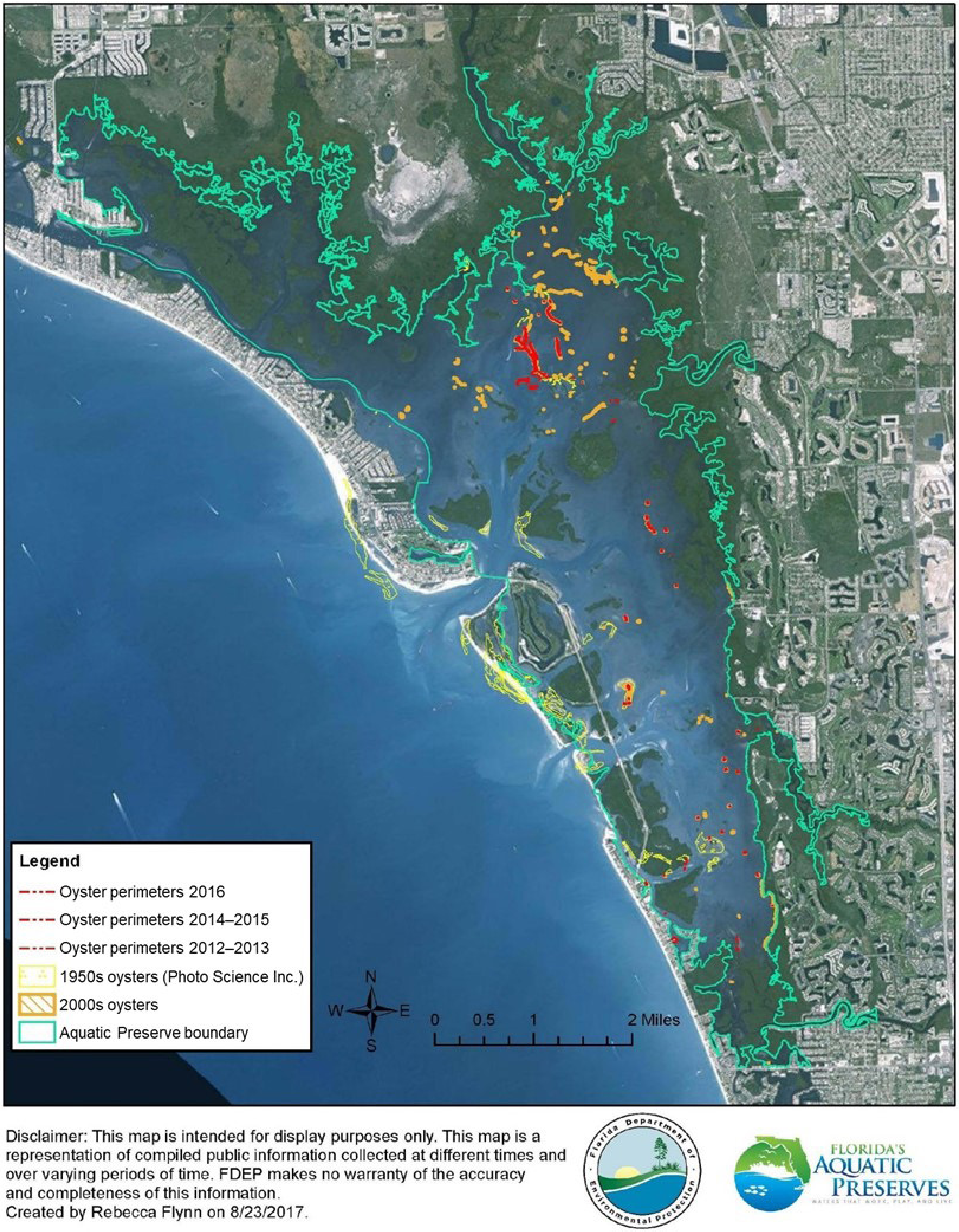

Figure 6.6. Map of historical known areas of oyster habitat and current (2012–2016) mapped oyster habitat in Estero Bay. Figure source: Rebecca Flynn, FDEP Estero Bay Aquatic Preserve.

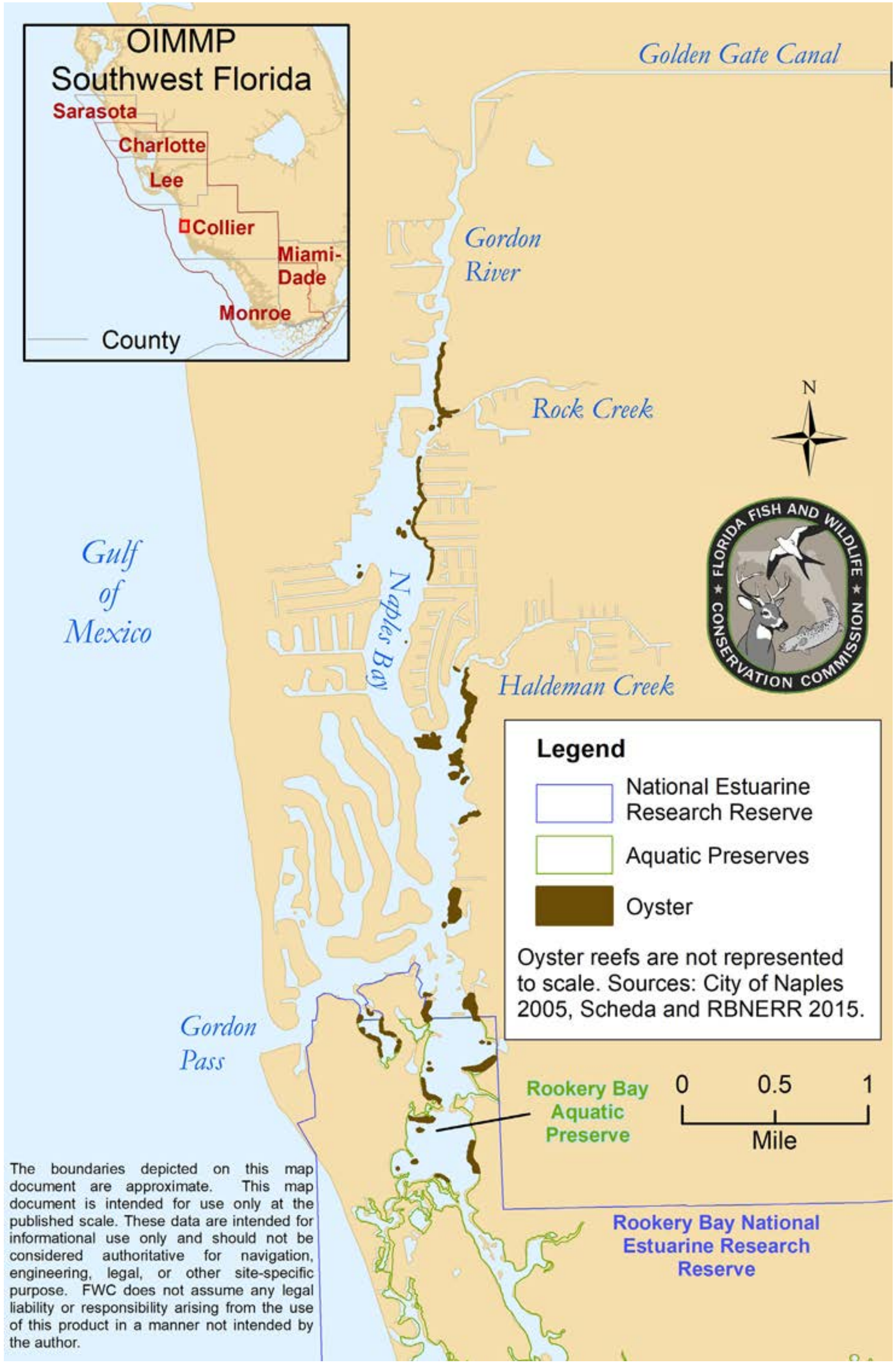

Figure 6.7. Mapped oyster extent within Naples Bay. Oyster mapping sources: City of Naples 2005 (made from 2005 sonar data) and Scheda and RBNERR 2015 (from 2014 aerial photography).

Rookery Bay and the Ten Thousand Islands

Rookery Bay and the Ten Thousand Islands (Fig. 6.10) are considerably less developed than the Naples and Charlotte Harbor watersheds. Apart from Marco Island, much of the watershed and coastal waters are protected from development by Rookery Bay National Estuarine Research Reserve (RBNERR), Rookery Bay Aquatic Preserve, Cape Romano–Ten Thousand Islands Aquatic Preserve, Ten Thousand Islands National Wildlife Refuge, Collier Seminole State Park, Picayune Strand State Forest, and

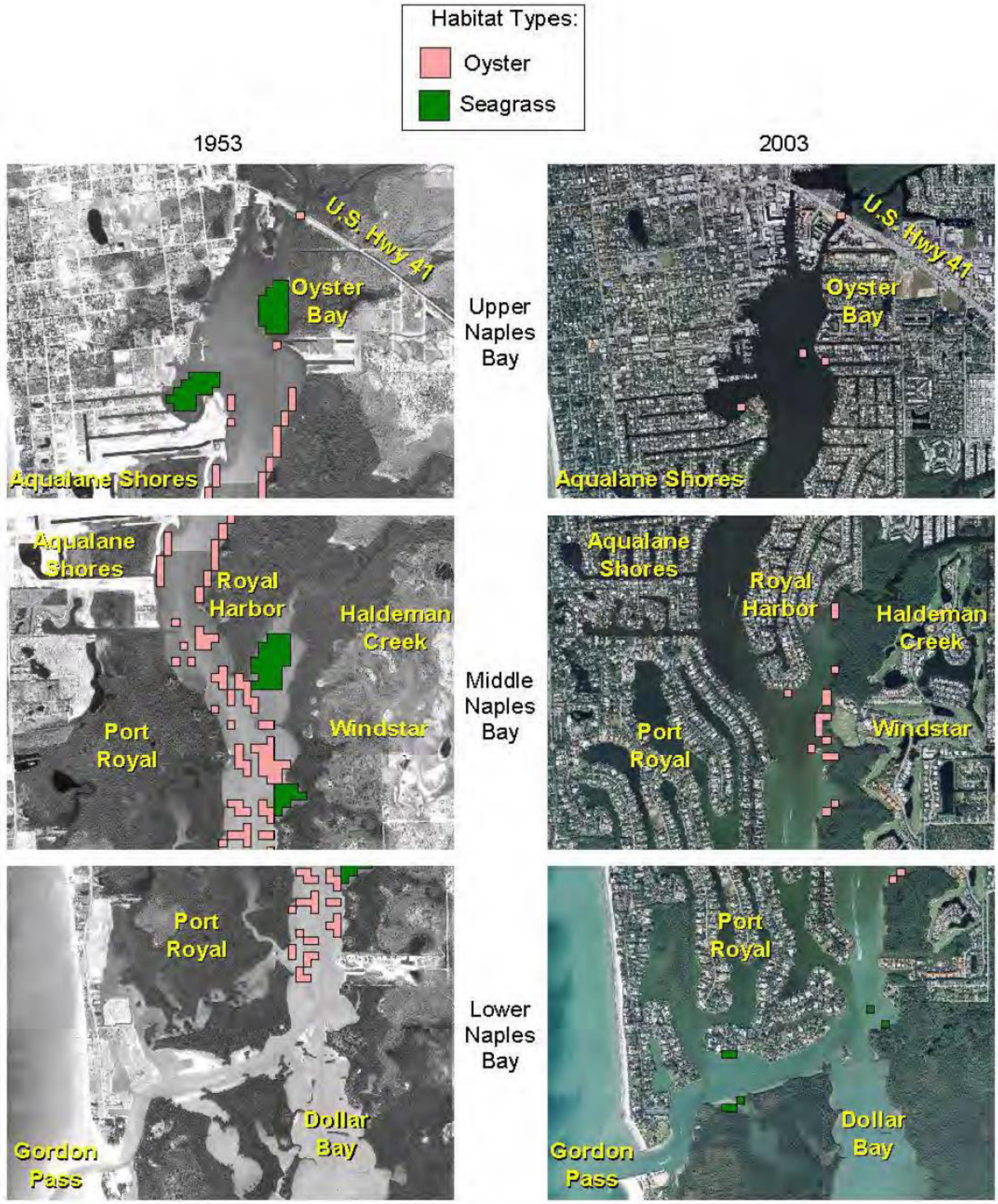

Figure 6.8. Coastline comparison between 1953 and 2003, with the approximate historical and present-day oyster and seagrass coverage. Figure source: Schmid et al. 2006.

Fakahatchee Strand State Preserve. Although much of the natural mangrove coastline remains intact, the watershed has been historically affected by widespread dredge-and-fill operations that drained large tracts of land for development and an increasing demand for fresh water by the growing population of southwest Florida. As a result, the timing and volume of fresh water entering the estuary has been altered (Scheda and RBNERR 2015). North of the Ten Thousand Islands, canals were dug to drain the failed planned community of the Southern Golden Gate Estates. These canals connect to the Faka Union Canal (Fig. 6.10), delivering large amounts of fresh water and increasing tur-

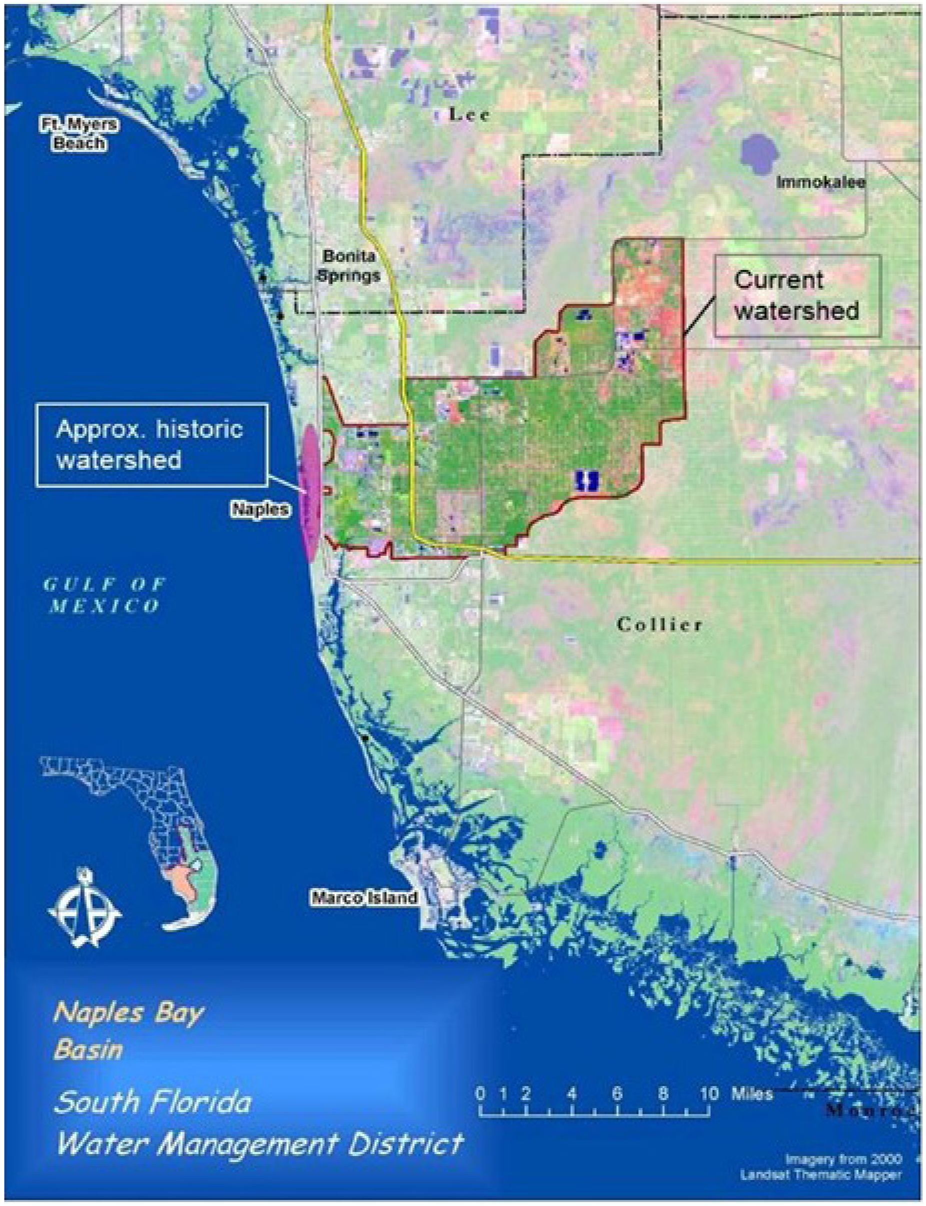

Figure 6.9. Historical (25 km^2/10 mi^2) watershed and expanded Naples watershed (300 km^2/120 mi^2) resulting from the dredging of the Golden Gate Canal. Figure credit: SFWMD.

bidity in the Ten Thousand Islands (USFWS 2000). The State of Florida purchased the private lots in the undeveloped community, and the land is now the Picayune Strand State Forest. Restoration efforts to improve hydrology are under way and include refilling canals, pumping water, and removing roads (SFWMD and USDA 2003). Salinity in the Ten Thousand Islands generally stays above 34 in the shallow coastal waters as a result of limited freshwater flow in the dry season; in the wet season it fluctuates between 20 and 32 (Soderqvist and Patino 2010). The region is susceptible to impaired water quality, including low dissolved oxygen and pollution from nutrients and heavy metals (CSF 2011). Shellfish harvesting is conditionally approved for a large part of the Ten Thousand Islands in shellfish harvesting area #66 (Fig. 6.5), although FWC records do not show any commercial harvest in Monroe or Collier counties since 1965 (FWC 2018).

Like Estero Bay, the geomorphology and formation of the Ten Thousand Islands can also be attributed to topography created by oyster and vermetid gastropod reefs formed during the early to middle Holocene (Volety et al. 2009a). Mud and shell accretion on these reefs provided a substrate for the settlement of mangrove propagules (Fig. 6.11), which matured into the mangrove islands that give

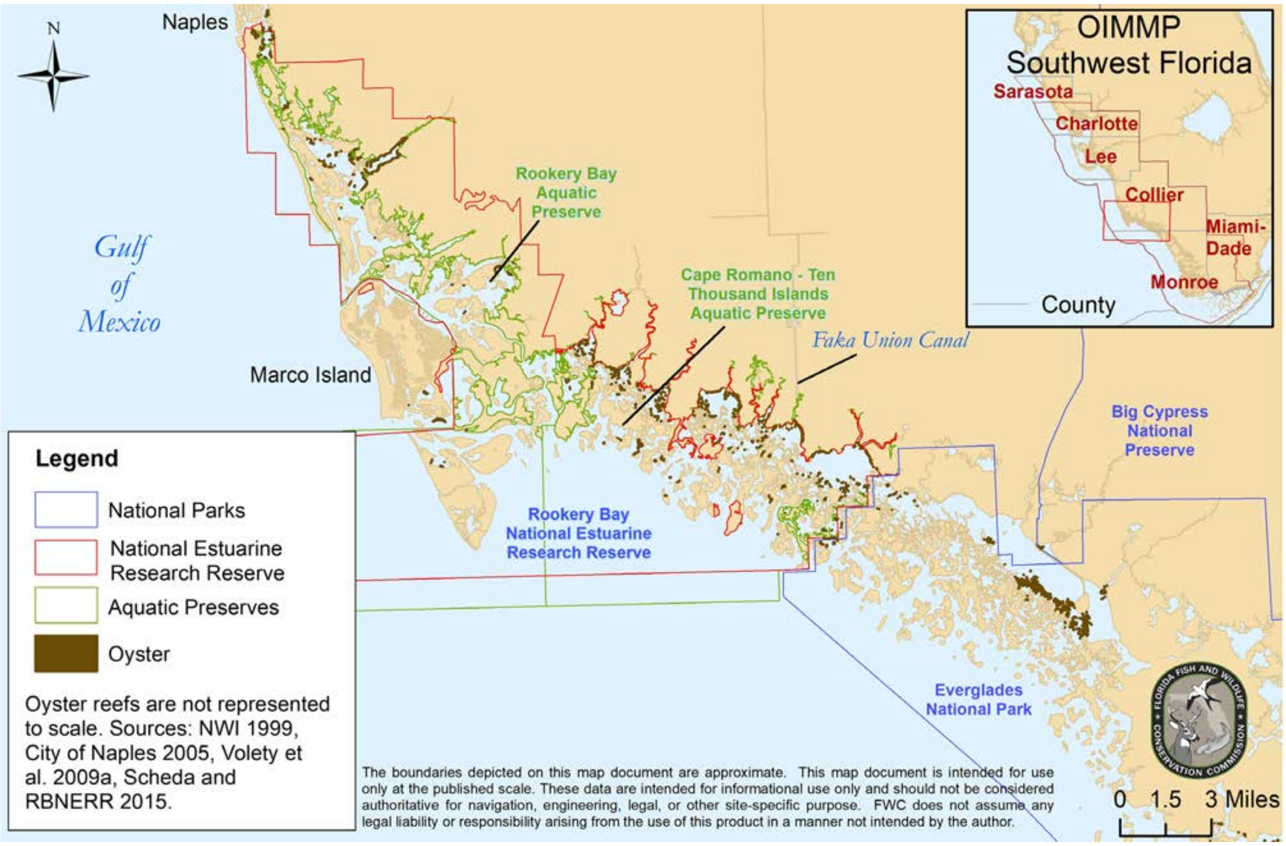

Figure 6.10. Oyster extent in Rookery Bay and Ten Thousand Islands. Oyster mapping sources: NWI 1999 (made from 1999 aerial photography), City of Naples 2005 (from 2005 sonar data), Volety et al. 2009a (from 2007 photography), and Scheda and RBNERR 2015 (from 2014 photography).

the Ten Thousand Islands its name (Volety et al. 2009a). A layer of oyster shell hash, a remnant of past oyster reefs, is common in subtidal stratigraphy in the area (Volety et al. 2009a, FDEP 2012).

Everglades and Florida Bay

Historically, sheet flow and groundwater flow made salinity favorable for oyster growth along the coast of the Everglades (Volety et al. 2009a). With reduced sheet flow, however, oysters are now restricted to brackish river mouths, inland bays, and channels (Fig. 6.12; Volety et al. 2009a). Mapping data are limited for much of Everglades National Park and may underestimate oyster extent. Limited oyster beds have been observed in the Taylor Creek and Shark River Slough systems, but since salinity is so dependent upon the timing and volume of upstream freshwater flows, their presence is highly variable (Penny Hall, pers. comm.).

Florida Bay is a broad, shallow bay south of the Everglades (Fig. 6.12). Some oysters are found in brackish waters on prop roots of mangroves bordering Florida Bay (Goodman et al. 1999), but these mangrove-root oysters are difficult to detect via remote imagery and so have not been mapped. Florida Bay itself does not support extensive oyster reefs and experiences widely variable water quality, including warm temperatures and hypersaline conditions as a result of reduced freshwater flow and evaporation (Fourqurean and Robblee 1999).

Threats to oysters in southwest Florida

- **Altered hydrology:** Oysters located in concentrated freshwater outflows such as the Caloosahatchee River Estuary or the Faka Union Canal must cope with very low salinity (<10) during the rainy season, which result in increased mortality and decreased reproduction (La Peyre et al. 2003, Volety et al. 2008). High nutrients in urban runoff and freshwater releases from Lake Okeechobee support algal blooms that smother oyster beds, cause hypoxia, and may release biotoxins (Volety et al. 2014). Most oysters in southwest Florida, including those near freshwater outflows, must also cope with low freshwater flow and high salinity, which increases rates of disease and predation (Volety et al. 2003, Volety

Figure 6.11. Red mangrove (*Rhizophora mangle*) propagules take root after settling on an intertidal oyster reef in the Ten Thousand Islands. Photo credit: Ryan P. Moyer.

et al. 2008). The consequences can be seen in the Everglades, where the range of oysters has shifted inland to follow lower salinity regimes (Volety et al. 2009a).

- **Sedimentation:** Several regions in Charlotte Harbor and Naples Bay must cope with excessive sedimentation as a result of runoff and altered hydrology. Fine sediment can interfere with filter feeding, cause decomposition-induced hypoxia, and accumulate as muck substrates that are unsuitable for oyster settlement (Volety et al. 2008). Frequent resuspension of sediments can occur due to boat wakes and waves intersecting with seawalls along the Caloosahatchee.
- **Habitat loss:** Large areas of oyster reef in Charlotte Harbor, Naples, and Marco Island were destroyed as a result of shell mining, commercial fishing, and dredge-and-fill construction (Boswell et al. 2012, Volety et al. 2014). Mangroves, and thus mangrove-root-oyster habitat, were also lost to coastal development.
- **Disease:** *Perkinsus marinus* (dermo) infections have been found in 65–95% of oysters in southwest Florida (Volety et al. 2009a, Volety et al. 2014). Although infection is common, infection intensity (as gauged on the scale developed by Mackin 1962) is often relatively low as warm summer temperatures are counteracted by the low salinity of the rainy season (La Peyre et al. 2003, Volety et al. 2008). Areas that do not receive high freshwater flow in the summer rainy season are more vulnerable to infection (Volety et al. 2009a).

Figure 6.12. Oyster extent in the Everglades and Florida Bay. Oyster mapping sources: NWI 1999 (made from 1999 aerial photography), Volety et al. 2009a (from 2007 photography), and Scheda and RBNERR 2015 (from 2014 photography).

- **Invasive species:** The Asian green mussel, *Perna viridis,* was first found in Charlotte Harbor in 2000 and in Estero Bay in 2002 (FDEP 2014). These mussels have no natural predators in Florida, reach sexual maturity quickly at an age of 2–3 months, and are tolerant of a wide range of salinity, so they can outcompete native bivalves. The Charlotte Harbor Aquatic Preserves have been monitoring green mussels since 2009 (FDEP 2017). Efforts to combat these invasive mussels include targeted removal efforts and educating the public on invasive species (FDEP 2014).
- **Climate change and sea-level rise:** Altered precipitation patterns, increasing temperatures, rising sea level, and ocean acidification all pose threats to oysters (Miller et al. 2009, Hoegh-Gulberg and Bruno 2010, Rodriguez et al. 2014), but their effect on oyster populations

in southwest Florida is uncertain, particularly when the many other anthropogenic impacts in the region are considered. This is a complex topic that requires further investigation and substantive review.

- **Harvesting:** Commercial harvesting in southwest Florida is not as extensive as it is in other coastal areas of the state, but Lee and Charlotte counties have experienced extensive shell mining and widely variable harvest intensity. The impact of the recent resurgence in commercial harvesting on severely reduced oyster populations needs to be assessed and managed.

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.

CHNEP and TNC volunteer oyster habitat monitoring

The CHNEP volunteer oyster habitat monitoring program is intended to enable the collection of meaningful long-term monitoring data from oyster habitat restoration sites by citizen scientists (CHNEP 2017). The CHNEP Oyster Habitat Restoration Plan (Boswell et al. 2012) provides monitoring protocols that can be used by all partner organizations in the Charlotte Harbor region and elsewhere. The handbook is available at http://www.chnep.wateratlas.usf.edu/upload/documents/Final_Volunteer-Coordinator-Manual-1517-with-Appendices.pdf.

The program was developed through hands-on experience gained working with volunteers to assist in monitoring TNC's Trabue Harborwalk Oyster Habitat Restoration Pilot Project in Punta Gorda. The protocols were vetted through both a scientific steering committee and a volunteer steering committee to ensure that the protocols were appropriate for volunteers and are also in line with those in the *Oyster Habitat Restoration Monitoring and Assessment Handbook* (Baggett et al. 2014). Results are therefore comparable to oyster habitat restoration projects in other regions.

By recruiting and training volunteers to conduct the majority of the monitoring using scientifically vetted protocols this program will:

- Enable long-term collection of monitoring data at a lower cost.
- Reduce the staff time needed to complete monitoring.
- Provide a source of leverage for grant funding through volunteer hours.
- Increase community support for habitat restoration projects through community engagement and stewardship of natural resources.
- Enhance understanding in the community about oyster habitat degradation and restoration and the ecosystem benefits of restoration.
- Provide meaningful volunteer opportunities for community members.

The Trabue Harborwalk Oyster Habitat Restoration Pilot Project includes nine pilot reefs—three each of bagged fossilized shell, loose fossilized shell surrounded by oyster bags, and oyster mats made with recycled shells. TNC is collaborating with Charlotte Harbor Aquatic Preserves, the CHNEP Volunteer Oyster Habitat Monitoring Program, Friends of the FDEP, and the Florida Fish and Wildlife Conservation Commission (FWC) to monitor these pilot reefs (Geselbracht 2016). Monitoring includes a focus on oyster recruitment, macroinvertebrate populations, bird presence, and use of the reefs by smalltooth sawfish (*Pristis pectinata*). The first of three years of planned postconstruction monitoring (2016) demonstrated that the reefs created a thriving community of oysters and other invertebrates. As of April 2017, 1,300 volunteers and citizen scientists had provided a total of 2,950 hours toward this restoration project, including material assembly, deployment, and monitoring (Geselbracht et al. 2017). The second annual postconstruction monitoring of the project was completed in November 2017. Monitoring reports and data for this project may be downloaded from http://chnep.wateratlas.usf.edu/oyster-habitat-restoration/.

Charlotte Harbor National Estuary Program Volunteer Tidal Shoreline Survey/Mapping

The CHNEP completed volunteer tidal shoreline surveys in 2007, 2010, and 2013 (CHNEP 2013). These surveys documented the condition of the estuary by assessing various characteristics of mangroves as well as gathering data on oyster presence or absence, invasive vegetation, and shoreline hardening. In 2013, 318 km (198 mi) of shoreline accessible by vessel were surveyed (80% of the shoreline length that had been identified for the survey); oysters were present along 18% of the surveyed shoreline. Oysters in the accessible areas tended to be located on the shorelines having the highest estuarine salinity and not along the banks of the rivers with lower salinity. The CHNEP plans to complete another survey in 2019. Survey results are available at http://www.chnep.wateratlas.usf.edu/shoreline-survey/.

Charlotte Harbor National Estuary Program mapping efforts

Several mapping studies have been conducted in this region (Harris et al. 1983, Avineon 2004, Photo Science 2007), but they are limited in scope, have used different methods, and the accuracy of results is not consistent. Comprehensive mapping of subtidal and intertidal oyster habitat is needed, as is condition analysis of reefs to inform restoration and management decisions. The Charlotte Harbor Oyster Restoration Plan provides a good summary of the past mapping efforts in the Charlotte Harbor estuary (Boswell et al. 2012).

Sanibel–Captiva Conservation Foundation (SCCF)

The SCCF Marine Laboratory has constructed or restored numerous oyster reefs from the mouth of the Caloosahatchee River to bayous west of Sanibel Island. All of its projects have included long-term monitoring of the restoration sites and adjacent reference sites. Monitoring metrics include density, size frequency, reef area, and reef height, plus ancillary measures including monthly settlement in Tarpon Bay, San Carlos Bay, and Matlacha Pass.

Comprehensive Everglades Restoration Program oyster monitoring

CERP conducts monitoring of oyster health and water quality on Florida's southeast and southwest coasts. Monitored locations on the Gulf coast include the Caloosahatchee River Estuary, San Carlos Bay, and the Ten Thousand Islands (Volety et al. 2008, 2009b, Volety and Haynes 2012, Volety et al. 2014). Florida Gulf Coast University (FGCU) led the first monitoring efforts on the Gulf Coast, while the FWC did so on the Atlantic coast, using a similar methodology (Boswell et al. 2012). In 2017, FWC assumed monitoring responsibilities in the Caloosahatchee River. Monitored parameters include water quality, spat recruitment, reproductive histology, dermo prevalence and intensity, recruitment, density, and juvenile oyster growth (Volety and Haynes 2012, Volety et al. 2014).

Oil spill impact study

Loren Coen and Ed Proffitt of Florida Atlantic University led a study on the impacts of the Deepwater Horizon oil spill on oyster reefs in Florida Gulf estuaries. Goals for this study include assessing oyster conditions, with components focusing on survival, growth, genetic diversity, and the concentration of polycyclic aromatic hydrocarbons in oyster tissues. More information may be found at http://research.gulfresearchinitiative.org/research-awards/projects/?pid=51.

Southwest Florida Water Management District mapping

The SWFWMD has conducted seagrass mapping every two years since 1988 using a modified version of the Florida Land Use and Cover Classification System (FLUCCS; FDOT 1999) for coastal areas in the district, including northern Charlotte Harbor. Subtidal habitats are mapped using natural-color aerial photography collected in winter at a scale of 1:24,000. Mapped habitats include tidal flats, beaches, patchy seagrass, and continuous seagrass. In 2014, the oyster bar (FLUCCS 6540) classification was added. The most recent effort was completed in 2016 (SWFWMD 2016).

Charlotte Harbor benthic habitat mapping and biodiversity efforts

Mote Marine Laboratory and CHNEP completed a mapping and biodiversity survey of benthic habitats in 10 basins of greater Charlotte Harbor (CHNEP and MML 2007). Surveyed habitats included mangroves, sandbars, mud flats, salt marsh, oyster reefs, and seagrass beds. More than 370 invertebrate taxa were identified from more than 44,000 organisms collected (CHNEP and MML 2007). The study also examined the impact of habitat and salinity on biodiversity.

Molluscan bioindicators of the tidal Myakka River and inshore waters of Venice

Mote Marine Laboratory conducted a survey of important mollusk species in the lower Myakka River and Dona and Roberts bays and their tributaries (Estevez 2005). Data were collected on density, shell size, and degree of shell weathering for living and dead oysters. Oysters were much more abundant in Dona and Roberts bays than in the Myakka River. Oyster shells were found in the Myakka River, but all shells were dead and highly weathered, indicating that the shells were old (Estevez 2005).

U.S. Army Corps of Engineers benthic habitat mapping

Dial Cordy and Associates was contracted by the U.S. Army Corps of Engineers to map benthic substrate in the Caloosahatchee River Estuary, St. Lucie Estuary, Loxahatchee Estuary, and Lake Worth Lagoon (Dial Cordy and Associates Inc. 2011). Resulting maps included bottom type (seagrass, oyster bed, shell, muck, etc.). Maps were created through the interpretation of high-resolution aerial photography as well as side-scan sonar. Substrates were verified with ground truthing, and the densities of living oysters were quantified on mapped oyster reefs.

Estero Bay Aquatic Preserve oyster mapping and monitoring

The Estero Bay Aquatic Preserve (EBAP) oyster mapping and monitoring program began in the winter of 2012. The original goal of the project was to map every oyster reef in Estero Bay Aquatic Preserve and compare reef extent to aerial photographs from the 1950s and 2000s. The focus has shifted toward assessing the health of a subset of oyster reefs in the bay (Stephanie Erickson, pers. comm.). Mapping and monitoring have typically been completed from October through March to take advantage of the low winter tides, but on a few occasions have been done in summer. In winter 2012–2013, 67 reefs were mapped and data were collected on reef status (natural live reef, natural live shell, natural nonliving, restored live reef, restored live shell, restored nonliving), presence of oysters, the perimeter of the intertidal oyster reef, presence of surrounding seagrass species, presence of green mussels, substrate, and water depth.

Since 2013, more parameters have been added to the protocol, including presence of mangroves and tide stage. In addition, each reef is surveyed using a transect and randomly placed quadrats. Data include the transect heading, type of oyster reef (patch, fringing, or string), reef length, shell height, reef height, percent cover (live oyster, oyster shell, sediment, or other), and presence of any other organisms. Several 0.25-m^2 quadrats are used to measure oyster density (including spat), size frequency distribution, and recruitment. Finally, water quality readings including temperature, salinity, dissolved oxygen, pH, and turbidity are collected near each reef. EBAP staff are developing a new long-term monitoring strategy for oyster health in Estero Bay that will comprise aerial mapping and annual monitoring of a handful of representative oyster reefs throughout the bay.

Naples Bay mapping

Savarese et al. (2004b and 2006) conducted substrate and subsurface mapping in Naples Bay and the Ten Thousand Islands. Substrate maps delineated oyster reefs using side-scan sonar (City of Naples 2005). Subsurface acoustic profiles were produced using shallow seismic chirp profiling, which could identify the presence of buried oyster reefs (Saverese et al. 2006). The effort also identified suitable locations for oyster reef restoration. GIS data are available at http://g.naplesgov.com/cityofnaplesgis/data.html.

Schmid et al. (2006) detailed the historical development and loss of estuarine habitat in Naples Bay. Historical and current maps of seagrass, oyster, and mangrove habitats were created in this effort. Benthic habitat maps were created using 1999 digital orthoquads, and sediment and biotic characteristics were verified with field sampling (Schmid et al. 2006).

Additionally, a master's thesis titled *Effects of salinity and other stressors on eastern oyster (*Crassostrea virginica*) health and a determination of restoration potential in Naples Bay, Florida* includes maps of historical oyster bed coverage, 2014 coverage, and restoration strategies for Naples Bay (Laakkonen 2014).

Rookery Bay National Estuarine Research Reserve (RBNERR) mapping

In 2014, RBNERR assessed estuarine conditions in Henderson Creek and the surrounding drainage basin. Scientists from Scheda Ecological Associates were hired to perform a literature review and analysis of available historical aerial imagery to assess the feasibility of documenting anthropogenic changes in estuarine habitats over time. This process included review of habitats (SAV, oyster beds, and hard bottom) based on visual signatures in aerial photographs. Aerial photography had never been taken specifically to identify submerged aquatic resources (seagrass or oysters) in the Rookery Bay Estuary. The project team acquired high-definition, geo-referenced aerial photography to provide baseline data that could be used in evaluating changes in the natural communities of Rookery Bay and the Ten Thousand Islands area in response to freshwater inflow alterations. Maps were created for SAV, seagrass, hard bottom, and oyster habitat (Scheda and RBNERR 2015).

National Wetlands Inventory (NWI) mapping

For more than 30 years, NWI generated and updated highly detailed wetland and intertidal habitat maps following the Cowardin et al. (1979) classification scheme. Estuarine intertidal mollusk reefs (coded as E2RF2) were mapped in Chokoloskee Bay using imagery from 1999 (NWI 1999). NWI maps are available at https://www.fws.gov/wetlands/index.html.

Ten Thousand Islands and the Everglades

An extensive geomorphologic study of the Ten Thousand Islands and the coastal Everglades was conducted to determine past and present oyster distribution (Volety et al. 2009a). Data products include maps of oyster extent, including those on reefs and mangrove roots, and information on oyster presence in the geologic record through stratigraphy. Mapping was based on digital orthophoto quarter quads (DOQQs); geographic information system (GIS) data layers of the mapping product include a confidence category that compares features visible on DOQQs to mapped and ground-truthed habitats. Monitoring parameters included oyster condition index, dermo prevalence and intensity, spat recruitment, size, survival and density (Volety et al. 2009a).

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 Bird Island in Charlotte Harbor, Naples Bay, Henderson Creek in Rookery Bay, Faka Union Bay in the Everglades, and Flamingo and Joe Bay in Florida Bay (Kimbrough et al. 2008). High levels of mercury were found in oysters at these locations, and high levels of copper were found in Naples Bay. Many oysters in Florida have high copper concentrations as a result of the use of copper in fungicides, algaecides, and antifouling paints (Kimbrough et al. 2008).

Recommendations for management, mapping, and monitoring

- CERP efforts to moderate freshwater flow are key to maintaining brackish salinity and are crucial for oyster survival. More frequent water sampling, such as sampling with autonomous instruments, is needed to capture data on freshwater pulses and their impact on oysters (CHNEP 2019).
- Compared with more urbanized areas of Florida, oyster mapping and monitoring are notably lacking for many regions of Everglades National Park. Oyster distribution in the region should be documented before completion of major improvements to U.S. 41 that will improve surface water flow. The Shark River Slough and Taylor Creek areas should be monitored after construction to assess whether the changes have improved conditions for oyster development (Penny Hall, pers. comm.).
- To gauge progress toward the goal of 400–2,400 ha (1,000–6,000 ac) of oyster habitat in Charlotte Harbor, the CHNEP Oyster Habitat Restoration Plan recommends mapping oyster habitats by type and implementing and monitoring oyster restoration efforts throughout the estuary by 2020 (Boswell et al. 2012).
- Continue efforts to determine patterns of oyster distribution before European settlement using historical records and sedimentary coring techniques (Boswell et al. 2012).
- Improve mechanistic understanding of reef-forming and reef-eroding processes (i.e., improve shell budget calculations).
- Improve understanding of how larval supply and transport among oyster reefs is impacted by river flow.
- Improve ecosystem service estimates (e.g., habitat provision, wave attenuation, water filtration, food-web contributions) for reef-forming and mangrove-root oysters.
- Improve estimates of predation and disease, especially as they relate to droughts.
- Assess whether recent increases in regional oyster harvests are sustainable.

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Chapter 7
Southeast Florida

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Description of the region

Southeast Florida features some of the most highly altered landscapes in the state, in Broward and Miami-Dade counties, as well as unique habitats in the Florida Keys archipelago. Broward and Miami-Dade counties are the most populous in Florida; the estimated 2017 population between the two counties was nearly 4.7 million (U.S. Census 2018). Oyster reefs are extremely limited, primarily due to suboptimal salinity, so there are no mapped oyster reefs or shellfish harvesting areas in this region of Florida.

Broward County

Before it was developed, much of the coast of Broward County (Fig. 7.1) was dominated by sawgrass (*Cladium jamaicense*) and other freshwater marsh plants (USFWS 1999a, FDEP 2006). The Intracoastal Waterway was constructed in Broward County in 1912, and the creation of inlets through barrier islands led to a brackish nearshore environment, killing freshwater species (FDEP 2006). The Intracoastal Waterway connects to the Atlantic Ocean in the north in Palm Beach County at Boca Raton Inlet, at the Hillsboro Inlet, and at Port Everglades. Dense urban development is now found along the coast of Broward County, including thousands of residences lining finger canals in dredge-and-fill developments. Most of this county has a hardened shoreline consisting of seawalls and riprap, and the main source of freshwater is urban runoff.

The few natural areas remaining in Broward County include Deerfield Island Park, Von D. Mizell and Eula Johnson State Park, Hugh Taylor Birch State Park (oysters uncommon), and the areas around West Lake (some oysters attached to mangrove roots). Small populations of oysters live on many of the extensive seawalls in this area. Oyster species include the flat tree oyster (*Isognomon alatus*) and the eastern oyster (*Crassostrea virginica*). Oysters have been noted during some seagrass surveys, but there are no known studies on the extent of oysters in Broward County (Linda Sunderland, pers. comm.).

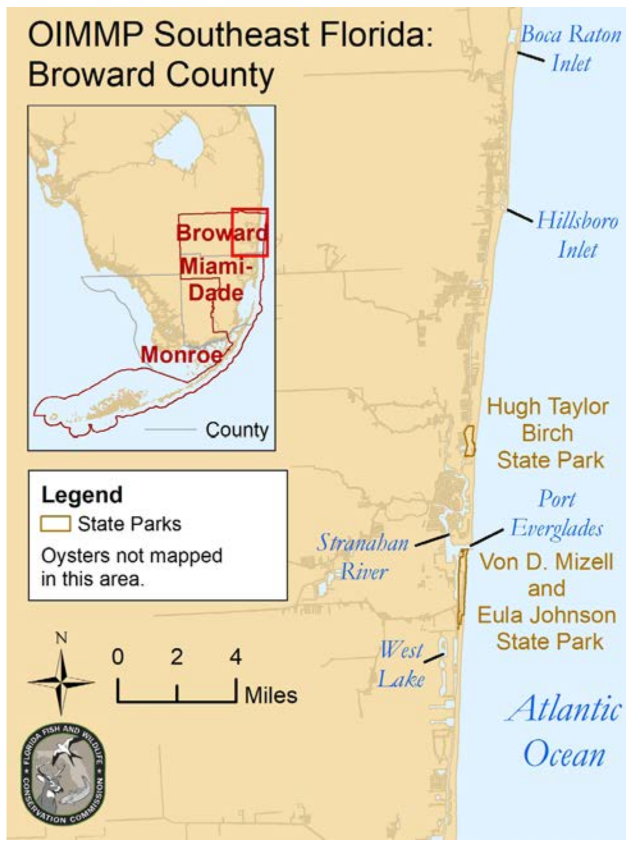

Figure 7.1. Major features of the Broward County coast. There are no mapped oyster reefs in this area.

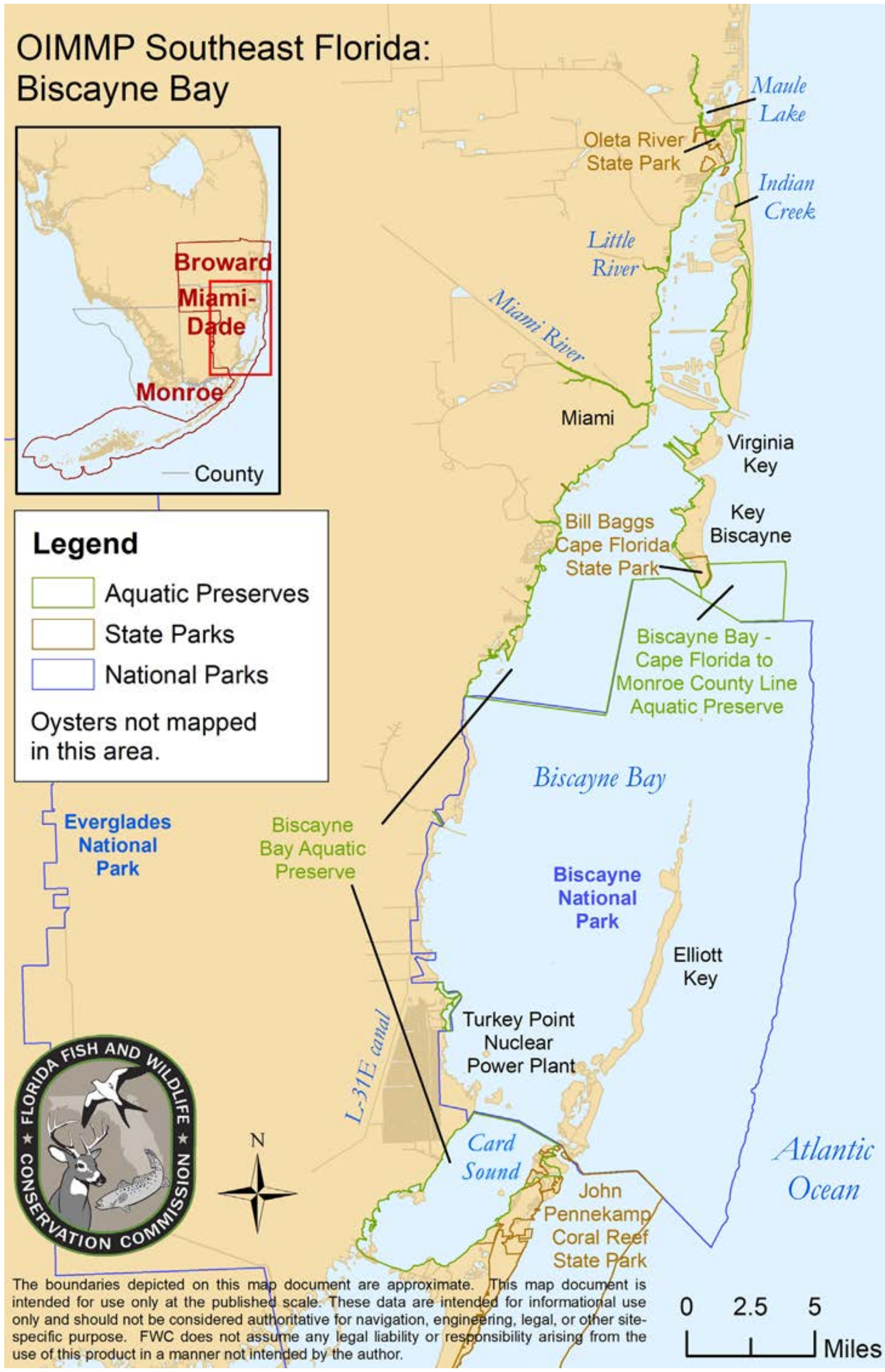

Figure 7.2. Major features of Biscayne Bay. Oyster extent is not mapped in Biscayne Bay.

Miami-Dade County: Biscayne Bay

Biscayne Bay is a semienclosed basin in Miami-Dade County approximately 75 km (47 mi) long and 16 km (10 mi) across at its widest point. The bay includes two aquatic preserves and Biscayne Bay National Park (Fig. 7.2). The average depth of the bay is around 2 m (6.5 ft), excluding dredged channels (Comp and Seaman 1985). Biscayne Bay was historically connected to the Everglades watershed via rivers and creeks that ran through and around the Atlantic Coastal Ridge, a ridge of limestone that runs along part of Biscayne Bay's western shore. Wetlands surrounded Biscayne Bay, and surface water entered the bay as a diffuse sheet of freshwater and through numerous small creeks and groundwater springs (Browder et al. 2005). Northern Biscayne Bay, which is partly enclosed and separated from the Atlantic Ocean by a series of barrier islands, used to be a brackish estuary as a result of freshwater input from the Everglades. Eastern oysters, whose survival and reproductive rates are greatest between 14 and 28 salinity (Shumway 1996), were relatively abundant in northern Biscayne Bay in those moderate-salinity waters, as evidenced by remnant oyster shells in sediments and historical accounts (Smith 1896, Meeder et al. 1999, 2001, CERP 2012). Shell middens along Biscayne Bay also show evidence of oyster harvest by Native Americans (Gambordella 2007, FDEP 2013). The northern part of the bay supported numerous oyster reefs and even a small oyster fishery in the late 1890s (Smith 1896, Meeder et al. 2001). Smith (1896) described oysters as being abundant on mangrove roots, pilings, boats, and submerged logs. Dense reefs also existed in the bay and near rivers, particularly near Little River and Indian Creek.

Central Biscayne Bay is broadly connected to the Atlantic Ocean. It is separated by a submerged ridge of Pleistocene-age coral reef, which is emergent at Key Biscayne, Elliott Key, and several smaller keys. Southern Biscayne Bay is partly sheltered from the Atlantic by Elliott Key and other islands in the northernmost extent of the Florida Keys. The central and southern parts of the bay historically had higher salinity than did northern Biscayne Bay, as determined by the marine mollusk species whose shells were found in sediment cores from those areas (Stone et al. 2000, Gambordella 2007). Relict oyster reefs have been found at the mouths of mangrove creeks in central Biscayne Bay (Meeder et al. 2001), but central and southern Biscayne Bay likely did not host extensive oyster reefs even before humans altered the watershed (Gambordella 2007).

The construction of canals, ditches, and levees in the 1800s and 1900s drastically altered the hydrology of Biscayne Bay and concentrated freshwater flow into constructed canals. This led to the loss of many of the mangrove creeks and resulted in large seasonal fluctuations in salinity due to variable freshwater input (Browder et al. 2005). The high-velocity release of fresh water does not readily mix with salt water in the bay, resulting in rapid changes in salinity that may kill estuarine organisms (Chin-Fatt and Wang 1987, CERP 2012). The Biscayne Aquifer, which contributes fresh groundwater to Biscayne Bay, is found in highly permeable limestone and sand from Miami-Dade County through Broward to southern Palm Beach County and supplies water for much of the population of south Florida (USFWS 1999b, CERP 2010). Canal construction in the early 1900s lowered the water table, and groundwater levels continue to decline due to urban and agricultural freshwater withdrawal (FDEP 2013).

The western shore of the southern half of the Biscayne Bay is largely lined with mangrove forests that are often cut through by tidal creeks. Historically, these creeks drained a small watershed, but as the area was developed, many of the creeks were disconnected by a series of water management canals and mosquito control ditches. The main waterflow is bisected by the L-31E canal, which disrupts the coastal sheet flow that once fed the creeks. The canal system still diverts most of the local watershed flow into a few canals, such that sheet flow from land into the bay has been largely eliminated by 1962 (Buchnan and Klein 1976). Near the southernmost point of the bay, the Turkey Point nuclear power plant dispenses water into an impoundment containing a large series of dredged canals, the intent of which is to cool thermal effluent.

Under the present hydrological regime, salinity exceeds 30 across Biscayne Bay in the spring and ranges from 25 to 30 in the fall, although areas along the coast with many canals can have an average salinity of 12–20 (CERP 2012). Today, the extent of oyster distribution is smaller than that before the hydrology was altered. In northern Biscayne Bay, where oysters had been abundant enough to support a fishery, they are now found, only occasionally, on mangrove roots, docks, or pilings. Oyster reefs are now limited to upstream areas of the Oleta River (FDEP 2013). Oleta River State Park has a large area of urban park with mangrove-lined creeks, but few oysters. Land between Oleta River State Park and Virginia Key is mostly urbanized with hardened shorelines. Oysters are occasionally found on these seawalls and riprap, especially in those areas that receive urban runoff from storm drains (Voss 1976). Most are eastern oysters. At some creeks, flat tree oysters are also found, but they do not form reefs or accumulate substrate. In central Biscayne Bay, relict oyster reefs are present near some tidal creeks (Meeder et al. 2001) and a small number of live oysters may occasionally be found associated with mangrove roots.

The Comprehensive Everglades Restoration Plan (CERP) includes efforts to redirect freshwater flow from canals into coastal wetlands to restore hydrology to a more natural state (CERP 2012). Projects relevant to Biscayne Bay include the Biscayne Bay coastal wetlands plan, the C-111 spreader project, and L31-N seepage management (Browder et al. 2005, FDEP 2013). These projects focus on four important stressors: altered freshwater inflow, input of toxicants and pathogens, altered input of solids and nutrients, and physical changes in structures.

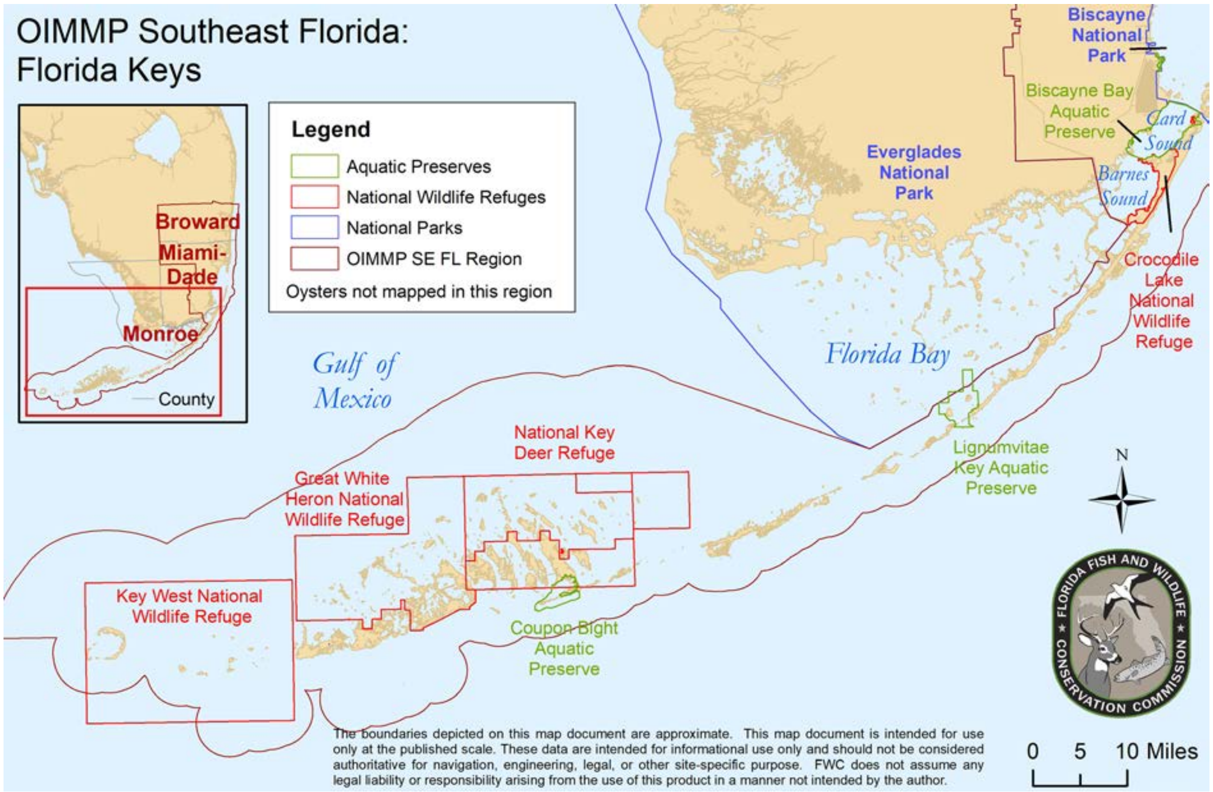

Figure 7.3. Major features of the Florida Keys. There are no mapped oyster reefs in the Florida Keys. Oysters in the Everglades and Florida Bay are mapped and discussed in Chapter 6.

Monroe County: Florida Keys

The Florida Keys are a 210-km-long (130 mi) archipelago on the southern edge of Florida (Fig. 7.3). The region is encompassed by the Florida Keys National Marine Sanctuary. Additional protected regions in the Sanctuary include Key West National Wildlife Refuge (NWR), Great White Heron NWR, National Key Deer Refuge, and Crocodile Lake NWR, as well as several state parks and aquatic preserves. Beginning near the border of Miami-Dade and Monroe counties, a series of sounds are found between the mainland and the primary tract of the Florida Keys. These sounds are all semienclosed and have low tidal amplitude and minimal exchange with other water bodies. The shoreline in these areas is mostly lined with mangroves. The lack of freshwater input and poor water circulation makes these sounds largely inhospitable to oysters. Oysters in the Everglades and Florida Bay are described in Chapter 6.

Continuing south, the middle and lower Florida Keys are a mix of hardened shorelines, mangroves, and occasional sandy beaches. Sources of fresh water include small amounts of terrestrial runoff, including runoff from watering lawns. Salinity is too high to support extensive populations of the eastern oyster, but they are occasionally found on hardened shorelines or mangrove prop roots (Mikkelsen and Bieler 2000). Several species of non-reef-building oysters are present in the Florida Keys, including two other species within the family Ostreidae: the root oyster (*C. rhizophorae*) and the crested oyster (*Ostrea stentina*) (Mikkelsen and Bieler 2000). The morphology of these oysters is plastic, as they often develop around their settlement substrate. The root oyster has a straighter and more elongate hinge than does the eastern oyster, as well as an elongate muscle scar. The crested oyster has a series of fine marginal teeth, lacking in both *Crassotrea* species, and a lighter muscle scar (Mikkelsen and Bieler 2007). The other most common oyster in the Keys is the scaly pearl oyster (*Pinctada longisquamosa*), which can at times be abundant in seagrass meadows. Other oyster species present include the sponge oyster (*O. permollis*), Atlantic pearl oyster (*P. imbricata*), black-lipped pearl oyster (*P. margaritifera*), Atlantic wing oyster (*Pteria colymbus*), glassy wing oyster (*P. hirundo*), tree oysters (*Isognomon alatus*, *I. bicolor*, and *I. radiatus*), hammer oyster (*Malleus candeanus*), frond oyster (*Dendostrea frons*), threaded oyster (*Teskeyostrea weberi*), and foam oysters (*Hyotissa mcgintyi*, *H. hyotis*, and *Neopycnodonte cochlear*) (Mikkelsen and Bieler 2000, 2007).

Threats to Oysters

Reef-building eastern oysters are rare in this region, largely as a result of suboptimal salinity regimes. The threats to oysters listed here are therefore restricted to those areas (namely Biscayne Bay) that are known to have hosted significant oyster reefs in the past.

- **Altered hydrology:** The hydrology of Biscayne Bay has been significantly altered by intense urbanization and the construction of drainage canals, which has decreased sheet flow through coastal wetlands and natural creeks. Water management canals preclude natural coastal flow and deliver large pulses of sediment-rich water through few large canals. Such canals allow rapid delivery of water that is often hypoxic and contaminated with urban runoff (SFWMD 1995). Additionally, the canals are generally steep-walled and linear, so their shorelines offer little habitat value. Restoration of minimal flows to the many mangrove-lined tidal creeks around the bay would enhance their chances of supporting small oyster communities (Meeder et al. 2001).
- **Isolated populations:** The survival and resilience of any remaining oyster populations are of concern due to their isolation from other oyster populations and vulnerability to continued urban development (Arnold et al. 2008). Biscayne Bay oysters are isolated from other oyster reefs on the southeast coast of Florida; the closest significant population of oysters is in the Lake Worth Lagoon, in Palm Beach County.
- **Habitat loss:** The coast of Broward County and northern Biscayne Bay is heavily developed, and much of the natural shoreline has been replaced by seawalls and other structures. Oyster reefs have directly faced habitat loss as a result of dredging and hardened shorelines.
- **Sea-level rise:** Sea-level rise is another contributor to the high salinity of Biscayne Bay. Sea-level rise will continue to suppress freshwater influence in the bay, such that brackish conditions may be completely eliminated in the bay except for waters adjacent to managed drainage canals. Sea-level rise may also result in increased submergence times, which would limit the intertidal exposure time for oysters that offers them refuge from predation, pests, and disease (Bahr and Lanier 1981). Increased submergence times as a result of sea-level rise, along with increasing salinity, lead to greater susceptibility to predation, pests (e.g., the boring sponge *Cliona celata*; Carroll et al. 2015), and pathogens (Shumway 1996).
- **Harmful algal blooms:** Harmful algal blooms have been tied to pollution from fertilizers, animal wastes, and increased soil erosion, all conditions that exist in south Florida watersheds. Toxins from cyanobacteria blooms have been detected in both Biscayne Bay and Florida Bay (Brand 2010).

Mapping and Monitoring Efforts

The compilation of oyster maps used in figures in this report are available for download at http://geodata.myfwc.com/datasets/oyster-beds-in-florida.

Historical oyster distribution study

Meeder et al. (2001) completed a study to identify tidal creek sites along western Biscayne Bay that were appropriate for restoration of freshwater discharge. Site selection was based on evidence of past oyster populations (determined by presence of oyster shell in sediments), and salinity targets were based upon physiological requirements of oysters.

FWC baseline mapping and monitoring

In the winter of 2005–2006, FWC and Golder Associates surveyed oyster distribution in the Sebastian River, Saint Lucie Estuary, Lake Worth Lagoon, and Biscayne Bay using a real-time kinematic global positioning system (RTK GPS) (Gambordella et al. 2007). Oyster reefs were identified using earlier oyster maps, helicopter aerial surveys, and sounding lines. Few live oysters and no oyster reefs were found in Biscayne Bay, so no oyster maps were produced for the bay.

Comprehensive Everglades Restoration Program oyster monitoring

Oysters were monitored as part of the CERP Restoration Coordination and Verification (RECOVER) program by FWC from 2005–2007 (RECOVER 2007, Arnold et al. 2008, Parker et al. 2013). Monitored basins included the Saint Lucie Estuary, Loxahatchee River, Mosquito Lagoon, Sebastian River, Lake Worth Lagoon, Biscayne Bay, and Tampa Bay (Arnold et al. 2008). Metrics included spatial and size distribution of oyster populations, physiological condition, disease frequency, and rates of reproduction, recruitment, growth, and survival in south Florida estuaries (RECOVER 2007). Too few oysters were present in Biscayne Bay for systematic density surveys. Most oysters existed as rare inhabitants of mangrove roots, and others were occasionally found along seawalls, where fresh water was supplied by lawn watering. Relict reefs were found at the mouths of several creeks (Parker et al. 2013). A few oysters did settle on settlement arrays monitored from July to November, but settlement rates were much lower than those in other Florida estuaries. Because oysters were found only inconsistently and in small numbers, there was not a clear pattern of reproductive development or condition indices (Arnold et al. 2008).

Growth and mortality were monitored in the estuaries mentioned above by monitoring planted juvenile oysters. Oysters in Biscayne Bay had a mean shell height in a middle to low range compared with oysters in the other estuaries (RECOVER 2007). Condition indices of oysters in Biscayne Bay were in the middle of the range compared with oysters in other south Florida estuaries. A few oysters were collected in 2005 and 2007 for investigation of disease intensity of dermo (*Perkinsus marinus*) and MSX (*Haplosporidium nelsoni*) (Arnold et al. 2008). Infection intensities and prevalence of dermo were low, and mean infection intensity was below 1 on the scale of 0 to 5 developed by Mackin (1962). There were no signs of MSX (Arnold et al. 2008).

NOAA Mussel Watch

The NOAA National Status and Trends Program has monitored pollutants in bivalves through the Mussel Watch program across the coastal United States since 1986. Monitoring locations in this region include Maule Lake in North Miami and Gould's Canal in Biscayne Bay (Kimbrough et al. 2008). High levels of copper, lead, and butyltins were found in oysters at these locations. Many oysters in Florida have high copper concentrations as a result of the use of copper in fungicides, algicides, and antifouling paints. Butyltins have also been used in antifouling paints, although their use was heavily regulated in 1988 (SFWMD 1995, Kimbrough et al. 2008).

Mollusk survey of the Florida Keys

Mikkelsen and Bieler published an extensive paper on marine bivalves (Mikkelsen and Bieler 2000) and a book on marine mollusks found in the Florida Keys (Mikkelsen and Bieler 2007). The publications include species lists with depth and location distribution throughout the Keys as well as records of dates of collection for museum specimens.

Recommendations for mapping, monitoring, and management

- Conduct a rapid assessment to determine if oysters are present in natural areas along Broward and Miami-Dade counties. Natural areas that may be able to support oyster communities include Deerfield Island Park, Hugh Taylor Birch State Park, and numerous tidal creeks along the western shoreline of Biscayne Bay. A presence–absence assessment would enable determination of need for future mapping and monitoring. These areas should also be assessed periodically for oyster larvae.
- Restore freshwater supply from the L-31E canal to some of the natural tidal creeks in Biscayne Bay. If oyster larvae are present, the placement of suitable substrate might allow settlement and rebuilding of oyster populations. If no larvae are present, transplanting of live oysters from the nearest suitable population, likely Lake Worth Lagoon, may be required. Small-scale efforts should be implemented before broader, large-scale efforts are planned.

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Chapter 8
Central Eastern Florida

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Description of the region

The Central Eastern region of Florida includes the Indian River Lagoon (IRL) system, St. Lucie Estuary, the Loxahatchee River, and Lake Worth Lagoon (Fig. 8.1). Water management in the region is divided between the jurisdictions of the St. Johns River Water Management District (SJRWMD) and the South Florida Water Management District (SFWMD). The coast includes sandy beaches on the Atlantic side of the barrier islands and a mosaic of mangroves, salt marshes, oyster reefs, and tidal flats in the protected inshore lagoons. Historically, these barrier islands were in a state of natural flux and inlets periodically opened or closed, altering salinity in the enclosed bays and lagoons. Human development has restricted that natural process, and many of the inlets are now permanently open to the Atlantic Ocean.

Hydrology in the region has been highly altered by urban development, freshwater management practices, mosquito ditching, dredge-and-fill operations, and construction of the Intracoastal Waterway (ICW). Regional watersheds are now considerably larger because of the construction of canals to drain inland wetlands for agriculture and development. Existing watershed-to-basin ratios for this region are 5:1 for the IRL (Engle et al. 2007); 98:1 for the St. Lucie Estuary (the estuary also receives freshwater releases from Lake Okeechobee; C. Buzzelli pers. comm., SFWMD et al. 2009); 175:1 for the Loxahatchee River and Estuary (Howard et al. 2011); and 7:1 for the Lake Worth Lagoon (Taylor Engineering Inc. 2009). Freshwater delivery to estuaries varies as a result of concentrated surface water flow, freshwater management, storms, and droughts, which in turn make salinity widely variable and result in challenging conditions for oyster growth.

The eastern oyster (*Crassostrea virginica*) has the highest rates of survival and reproduction at a salinity between 14 and 28 (Shumway 1996). Oyster reefs are therefore largely confined to bays, lagoons, and rivers with predominantly brackish salinity regimes along central and southeast Florida (Arnold et al. 2008, Parker and Geiger 2010). While the eastern oyster forms large reefs, the flat oyster (*Isognomon alatus*) can also be found on seawalls, pilings, and mangrove prop roots (Bachman et al. 2004) and can be a major component of some reefs (e.g., in the southwest fork of the Loxahatchee River). The crested oyster (*Ostrea stentina*) is also occasionally found in downstream, high-salinity areas of this region on mangrove prop roots (Abbott and Morris 1995, Puglisi 2008). Oyster reefs on the southeast coast of Florida often have lower densities than reefs on the southwest coast, although growth, condition index, and reproductive activity are comparable (LWLI 2013, Parker et al. 2013).

Indian River Lagoon

The IRL system consists of a series of coastal lagoons (Mosquito Lagoon, Banana River Lagoon, and Indian River Lagoon) enclosed by barrier islands (Fig. 8.2). The IRL covers 40% of Florida's east coast and includes seven

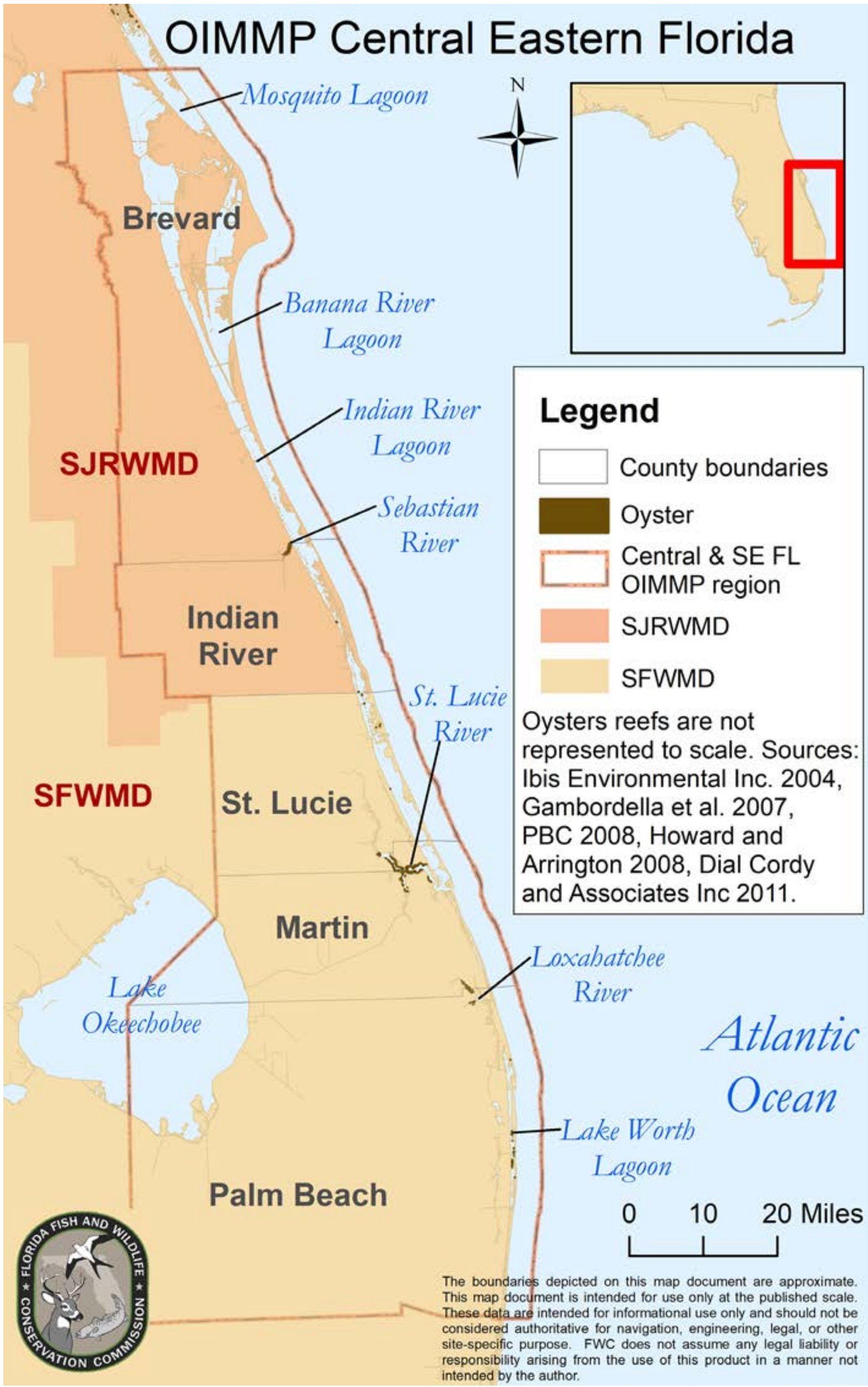

Figure 8.1. Oyster extent within the central eastern Florida region.

Florida Department of Environmental Protection (FDEP) aquatic preserves: Mosquito Lagoon, Banana River, North Fork St. Lucie River, Loxahatchee River-Lake Worth Creek, and three Indian River preserves (Malabar to Vero Beach, Vero Beach to Ft. Pierce, and Jensen Beach to Jupiter Inlet). The southern portion of Mosquito Lagoon is managed by Canaveral National Seashore and Merritt Island National Wildlife Refuge.

Historically, the IRL drainage basin was much smaller than today, and its boundary followed the Atlantic Coastal Ridge. Numerous canals were constructed from 1916 to the 1950s to drain wetlands for agriculture, which greatly increased both the size of the IRL watershed and the rate at which water was delivered to the lagoon. The canals concentrate stormwater runoff, delivering large quantities of freshwater flow and associated land pollutants into the lagoon during the rainy season (FDEP 2014). Upstream agricultural and urban consumption has decreased freshwater flow in the dry season. Other constructed hydrologic alterations include multiple causeways and impoundments on many of the remaining coastal wetlands. A prominent feature of the IRL system is the dredged ICW and associated spoil islands. The ICW was conceived in the 1800s and was dredged in the early to mid-1900s. For much of the IRL, the spoil was piled into a series of islands. Some of these islands were built as ring-shaped levees, which were intended to be flooded to reduce mosquito breeding but have now created enclosed basins of poor water quality. In some cases, spoil islands may create beneficial intertidal habitat. Some of the ring-shaped levees have been opened to improve water quality within the impoundments, creating small areas of suitable habitat for oysters.

The barrier islands that border the east side of the IRL system restrict tidal exchange and currents, resulting in a long water residence time and making the region susceptible to impaired water quality (FDEP 2009a, 2009b,

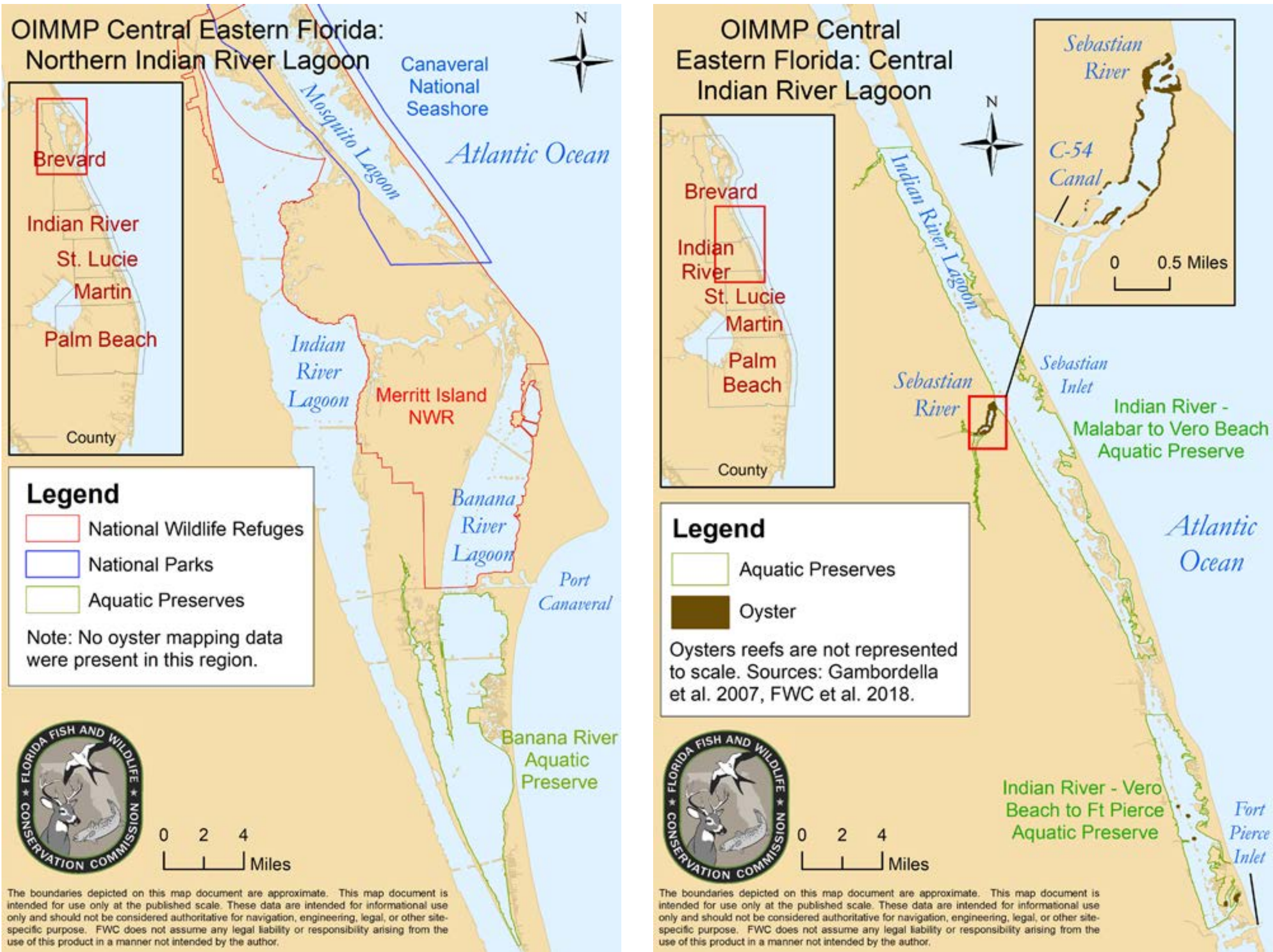

Figure 8.2. Mapped oyster extent within the northern (left) and central (right) Indian River Lagoon system. Oyster maps likely underestimate the extent of oysters in the IRL. Oyster mapping sources: Gambordella et al. 2007 (made from 2005–2006 field surveys) and FWC et al. 2018 (from 2018 field surveys and aerial photography).

2014). Along the 250-km (156-mile) extent of the IRL system there are five permanent inlets that connect with the Atlantic Ocean (Ponce, Sebastian, Fort Pierce, St. Lucie, and Jupiter inlets). The Banana River Lagoon requires two years for complete turnover (FDEP 2013). In extreme summer droughts, a combination of evaporation, low freshwater input, and limited flushing has caused salinity to reach 45 in the Banana River Lagoon (FDEP 2014). Turnover in the southern IRL is 10–15 times faster than in the northern lagoons (FDEP 2009b, FDEP 2014). The average depth of the IRL is 1.2 m (4 ft). The shallow water warms quickly in summer, decreasing oxygen solubility and facilitating development of hypoxia or anoxia (FDEP 2014). The region is also susceptible to cyanobacterial blooms, harmful algal blooms, and associated mortality of seagrass, marine mammals, and birds (FDEP 2014). Harmful algal species include those that cause brown tide (*Aureoumbra lagunensis*), red tide (*Karenia brevis*), and paralytic shellfish poisoning (*Pyrodinium bahamense*). Research into the ecosystem-wide relationships that lead to these algal blooms and mortality events is a key focus of scientific and management efforts in the IRL (SJRWMD et al. 2012).

Extensive oyster reefs are generally not found today within the main portion of the IRL, although the presence of shell middens along the coast of the IRL (FDEP 2014) and buried oyster shells (William Arnold, pers. comm.) indicate more extensive reefs existed in the past. The decline in oyster populations is likely a result of habitat degradation, altered salinity regimes, and decreased water quality (high salinity, low dissolved oxygen, algal blooms, etc.) (FDEP 2014). While oysters are abundant in the northern portion of Mosquito Lagoon (see Chapter 9), they are not common in the southern portion due to limited suitable substrate and salt marsh extent, minimal tidal flushing, and seasonally variable water levels. Seasonal precipitation patterns and winds can also result in changes in water levels that exceed tidal range, leading to seasonally variable water depths (FDEP 2009a, Smith 1993). Patches of oysters are occasionally present around spoil islands, and oysters can be found at the periphery of the IRL, often growing beneath or on the prop roots of red mangrove (*Rhizophora mangle*) trees (Aquino-Thomas and Proffitt 2014). The red mangroves facilitate oyster establishment by providing a substrate for growth and protection from erosion and sedimentation (Aquino-Thomas and Proffitt 2014). Oysters on mangrove roots are difficult to map and frequently do not appear on maps of oyster reefs, so IRL oyster maps likely underrepresent their true extent (Fig. 8.2). There are some oyster reefs in the central IRL located near the Northern Relief Canal near Vero Beach, and Jack Island and Wildcat Cove near Fort Pierce (lower right of Fig. 8.2). These three reefs are all found in the Indian River - Vero Beach to Fort Pierce Aquatic Preserve.

The most significant oyster populations along the IRL are found within tributaries, including the Sebastian River (Fig. 8.2). Oyster populations in the Sebastian River fluctuated after construction of the Sebastian Inlet, which increased salinity, and completion of the C-54 flood-control canal, which decreased salinity (Gambordella et al. 2007). Studies conducted by the Florida Fish and Wildlife Conservation Commission (FWC) during 2005–2007 showed that salinity in the Sebastian River was generally below 20 and that values below 5 were not uncommon (Arnold et al. 2008). These FWC monitoring efforts followed an extremely active 2004 storm season, the most active Atlantic hurricane season on record. From 2005–2007, considerable freshwater inputs from rainfall and runoff substantially reduced salinity to levels below the oyster tolerance range (Paperno et al. 2006, Steward et al. 2006). When the FWC study was initiated, in early 2005, oysters in the Sebastian River were scarce and likely recovering from a large-scale die-off in late 2004. Their recovery was further impeded by the active 2005 storm season. As a result, densities of live oysters in the Sebastian River were relatively low during that period (Parker et al. 2013). The oyster population in the Sebastian River recovered in 2007 after salinity increased to more suitable levels for oyster growth and survival (Arnold et al. 2008, Parker et al. 2013).

A commercial oyster fishery operated intermittently in the IRL during the 1800s, flourishing after the construction of the railroad and becoming well known by the 1900s (Gambordella et al. 2007). Periodic opening and closing of inlets to the IRL resulted in variable salinity, contributing to a decline in oyster extent and commercial harvest by the 1960s (Gambordella et al. 2007). Today the IRL contains nine shellfish harvesting areas with varying degrees of allowable harvest (Fig. 8.3, FDACS 2017). Oyster harvests for the region remain relatively low, particularly since the 1980s (Fig. 8.4, FWC 2018).

St. Lucie Estuary

The St. Lucie Estuary (SLE; Fig. 8.5) was a freshwater river until the St. Lucie Inlet was created in 1892, resulting in saltwater intrusion that extends as far as 26 km (16 mi) upstream (FDEP 2009c). Construction of the St. Lucie Canal (C-44) was completed in the 1920s, draining large areas west of the river and connecting Lake Okeechobee to the south fork of the St. Lucie River (FDEP 2014). The north fork was connected to the C-23/C-24 canal system in the 1950s as part of the Central and Southern Florida Project, further increasing freshwater diversions to the SLE. Water releases from Lake Okeechobee increased freshwater inflow as well as sediment and nutrient load to the SLE and southern IRL, decreasing water clarity and resulting in the loss of seagrass and oyster habitats (Bartell et al. 2004, FDEP 2014). Key components of the Comprehensive Everglades Restoration Plan (CERP) aim to improve the quality, quantity, timing and distribution of fresh water delivered to the SLE. The eastern oyster was chosen as an indicator species in the CERP REstoration, COordination, and VERification (RECOVER) Monitoring and Assessment Program due to its wide distribution and importance to estuarine ecosystems (CERP 2004, Parker et al. 2013). The density, population structure, and condition of oysters are therefore routinely monitored in the SLE. The SLE was estimated to contain 570 ha (1,400 ac) of oyster reefs in 1940–1960 (Fig. 8.6; URS Greiner Woodward Clyde Inc. 1998). Oyster extent had decreased to ~80 ha (200 ac) by the late 1990s (SFWMD 2002a). The goal of SLE restoration according to CERP and SFWMD's minimum flows and levels process is to "promote and sustain a healthy oyster population" and increase the acreage of oyster reefs (Dixon et al. 1994, SFWMD 2002a, 2007). SFWMD (2014) outlines the oyster monitoring results for live density, recruitment rate, and dermo (*Perkinsus marinus*) infection prevalence and intensity for the SLE by subbasin.

Salinity effects on different size classes are a key factor impacting oyster populations in the SLE, but other factors that are related to salinity (such as predation and parasitism) or sometimes correlated with it (e.g., turbidity, chlorophyll a, temperature) likely help determine population dynamics (Salewski and Proffitt 2016). Perhaps most important is the interaction of temperature and salinity. The most stressful time of year for an oyster in Florida is during the summer months when temperatures are high, often near their upper physiological tolerance limits. Water temperatures have exceeded 32°C for four months or more, and oysters can experience even higher temperatures if they are exposed during afternoon low tides. When oysters are subjected to environ-

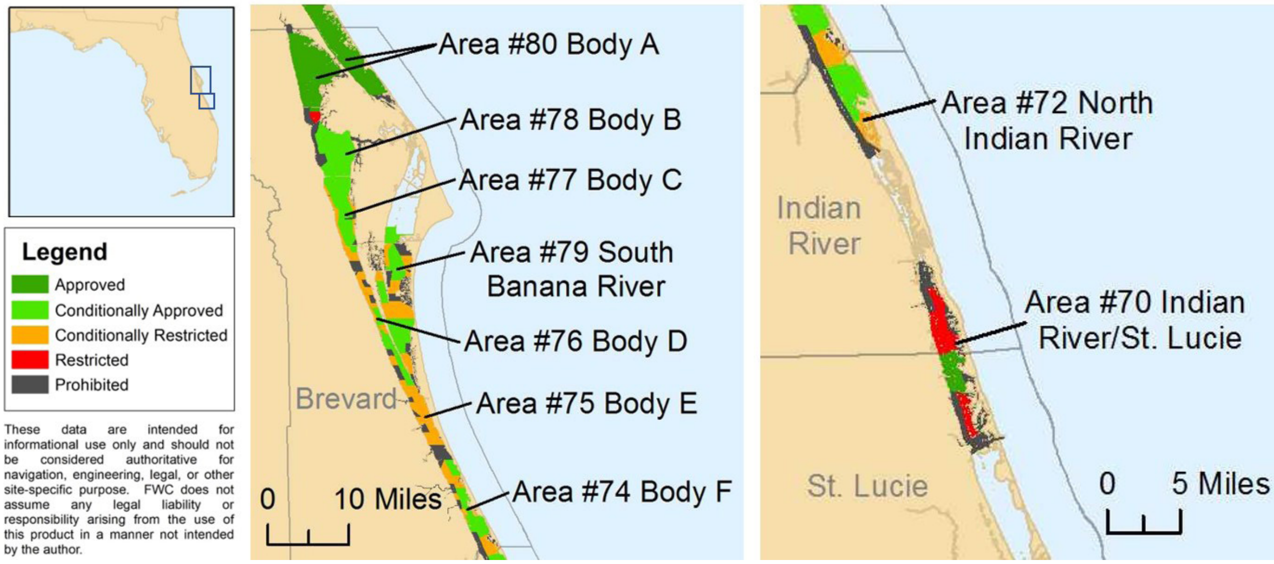

Figure 8.3. Shellfish harvesting areas in the central eastern Florida region. Data source: FDACS 2017.

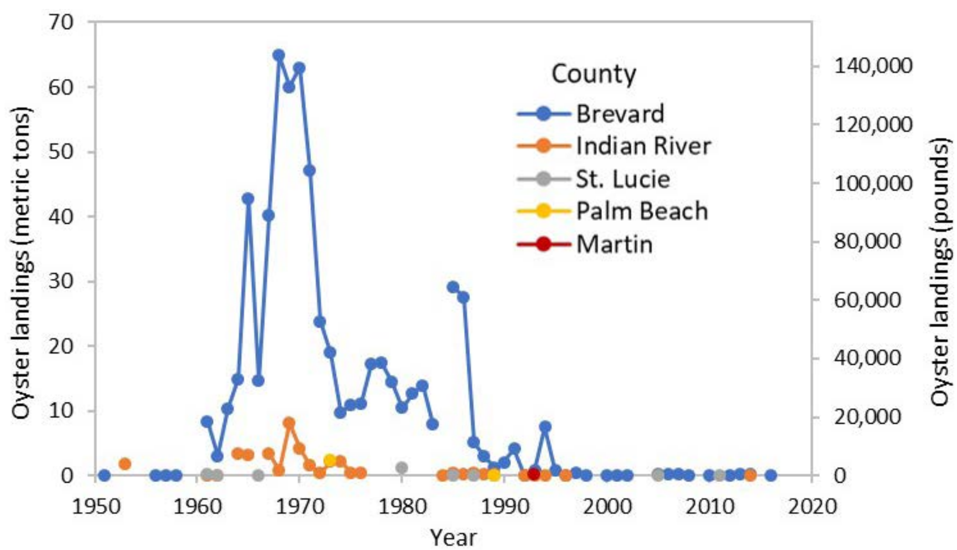

Figure 8.4. Commercial oyster landings for counties in the central eastern Florida region. Data sources: summary of Florida commercial marine fish landings (see Appendix A) and FWC 2018. Oyster landings data before 1986 were collected under a voluntary reporting system.

mental conditions that meet or exceed their tolerance limits, their energetic capacity to deal with additional stresses, such as salinity extremes, are diminished or lost (Parker 2015). Winter is an essential time for oysters in southeast Florida to allocate energetic reserves for growth and reproduction. Conditions are the least stressful when temperatures are lower, salinity is moderate, dissolved oxygen concentrations are greater, and predation and disease are less prevalent (Thompson et al. 1996, Arnold et al. 2008). Stressful events in the winter season may require oysters to allocate energy toward survival and reduce energy input toward gametogenesis (Kraeuter et al. 1982, Thompson et al. 1996), but these effects are less adverse than those incurred during the warm summer months.

The population and health of oysters in the SLE fluctuate dramatically due to variable freshwater inflows related to storm events and managed freshwater releases associated with Lake Okeechobee and the surrounding canals. The timing, magnitude, and duration of a freshwater release govern the severity of the impact on the local oyster population as well as the recovery time following the release. In 2004 and 2005, excess rainfall from hurricanes and large freshwater releases from Lake Okeechobee resulted in low salinity and widespread oyster mortality within the SLE (Arnold et al. 2008, Parker

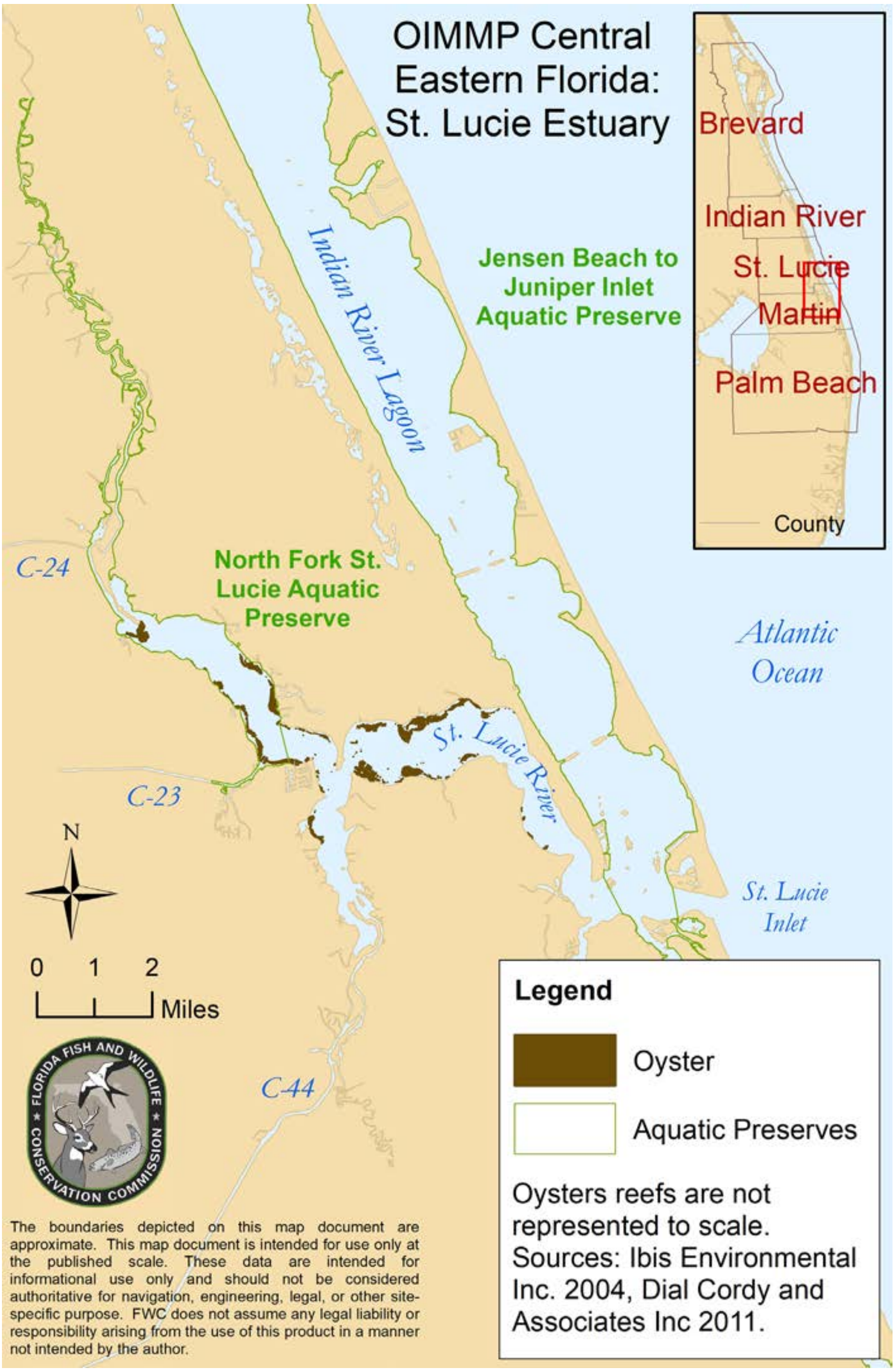

Figure 8.5. Oyster extent within the St. Lucie Estuary. Oyster mapping sources: Ibis Environmental Inc. 2004 (made from 2003 field surveys) and Dial Cordy and Associates Inc. 2011 (from 2011 side-scan sonar).

and Geiger 2013, Parker et al. 2013). Estuarine salinity remained below oyster tolerance levels through December 2005, so larvae from downstream locations could not settle and repopulate the SLE reefs, and recovery was delayed until late 2006. SLE oyster populations were again damaged when Tropical Storm Fay hit in August 2008. Though the event was of relatively short duration, the rainfall and freshwater inflows associated with it caused another oyster die-off. Salinity rapidly increased to tolerable levels immediately following the storm, allowing for settlement of larval recruits in late fall before the end of the 2008 spawning season. Those recruits successfully overwintered, allowing for a more rapid recovery in 2009 (Parker 2015).

The reprieve for oysters in the SLE was short-lived as salinity decreased again in 2010 due to a prolonged freshwater release that lasted from March through October. Although oyster mortality was not estuary-wide, upstream oysters were disappearing or exhibiting poor health during the warm summer months. In addition, the freshwater release coincided with the months of peak oyster reproductive development and spawning, and though oysters developed and spawned as expected, recruitment rates were significantly lower, suggesting that larvae were either killed by the low salinity or flushed out of the estuary by high water flow (Parker 2015).

Near-drought conditions in early 2012 kept SLE salinity levels relatively high and near optimal levels for oysters (Parker and Geiger 2013). Later in 2012, rainfall from Hurricane Isaac caused less damage to the SLE oyster reefs than storm-associated rains in 2005 or 2008, possibly due to management decisions to delay freshwater releases from Lake Okeechobee for a few weeks after the storm (Parker and Geiger 2013). SLE salinity recovered quickly in late 2012. For the first several months of 2013, drought conditions kept salinity so high it often exceeded the oyster tolerance range. High rainfall in the early wet season of 2013 led to high water

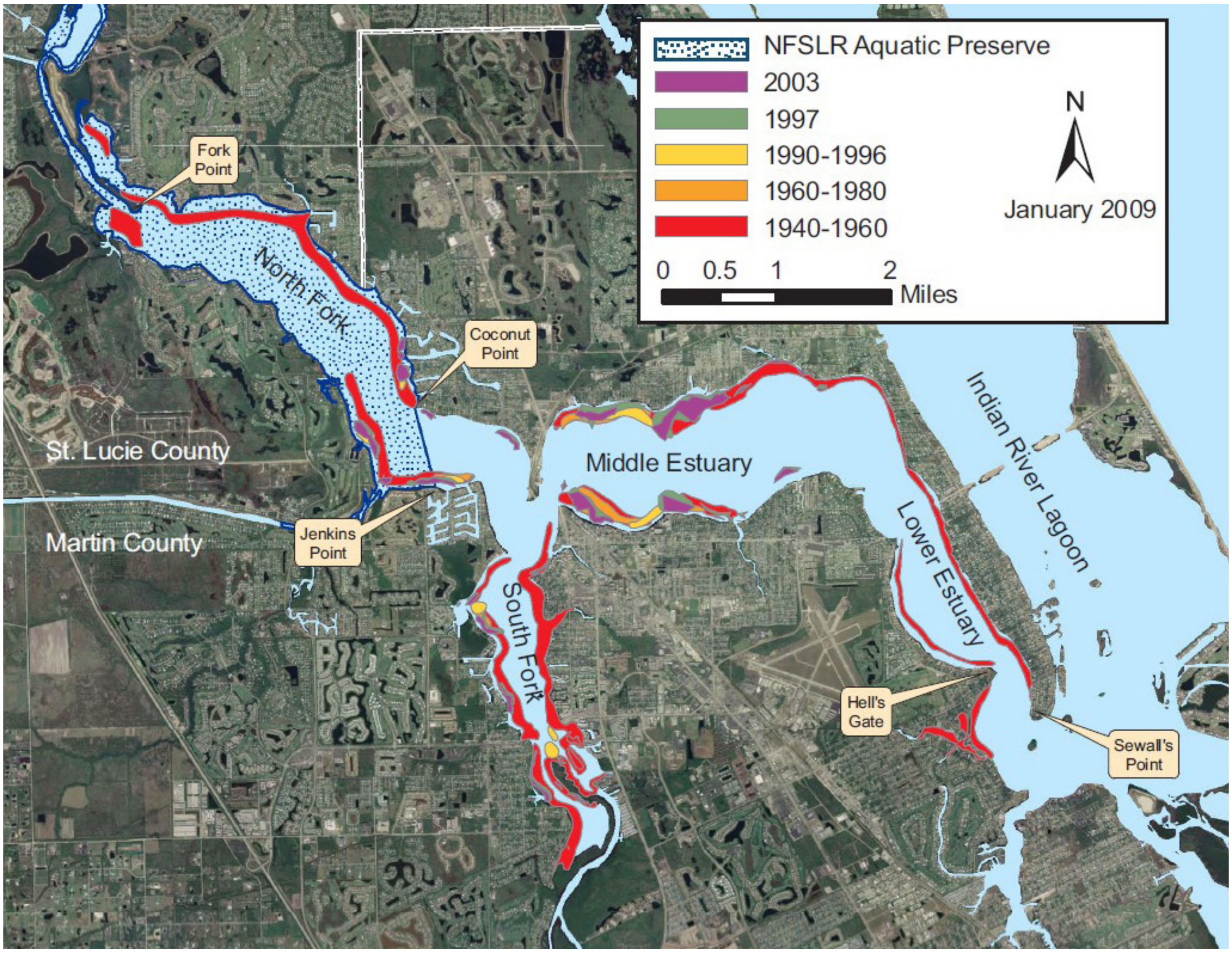

Figure 8.6. Historical extent of oyster reefs in the St. Lucie Estuary. Figure credit: FDEP 2009c.

levels in Lake Okeechobee, and subsequent freshwater discharges caused salinity to decrease rapidly to suboptimal levels. This event caused another widespread oyster die-off in the SLE and led to high levels of fecal coliform bacteria and a cyanobacterial bloom, both of which prompted the Martin County Department of Health to issue a health advisory regarding poor water quality. Density surveys conducted in December 2013 found only a few live oysters, all of which were spat that had settled in the past few weeks, and no live adults.

Excess freshwater releases from Lake Okeechobee during the 2015–2016 El Niño resulted in low salinity in the SLE from February through October 2016, causing a loss of oysters upstream (Parker and Radigan 2017). The oyster population response was similar to the response to the 2010 event, in that recruitment rates were poor even though spawning did occur. Freshwater discharges following Hurricane Irma in September 2017 had an even greater impact on oyster populations in the SLE, as they continued into late December, hindering the fall spawning and recruitment season.

Loxahatchee River

The Loxahatchee River crosses through both Martin and Palm Beach counties, draining a watershed of 700 km^2 (270 mi^2) before it reaches the Atlantic Ocean at the Jupiter Inlet (Fig. 8.7; SFWMD 2006, Howard and Arrington 2008). The tidal floodplains and estuary with its seagrasses, mangroves, and oyster beds are valuable ecological resources within the watershed, which also includes extensive urban development. Historically, the Jupiter Inlet opened or closed as a result of storm events and river flow. After the construction of the ICW and the Lake Worth Inlet, the Jupiter Inlet frequently remained closed until the 1940s, after which the Inlet was kept open through dredging operations and the construction of jetties and sand traps (SFWMD 2006). The Loxahatchee River watershed has been permanently altered by the stabilization of Jupiter Inlet, which heightens tidal amplitude and saltwater intrusion. Freshwater flow has also been altered by the construction and operation of drainage canal systems.

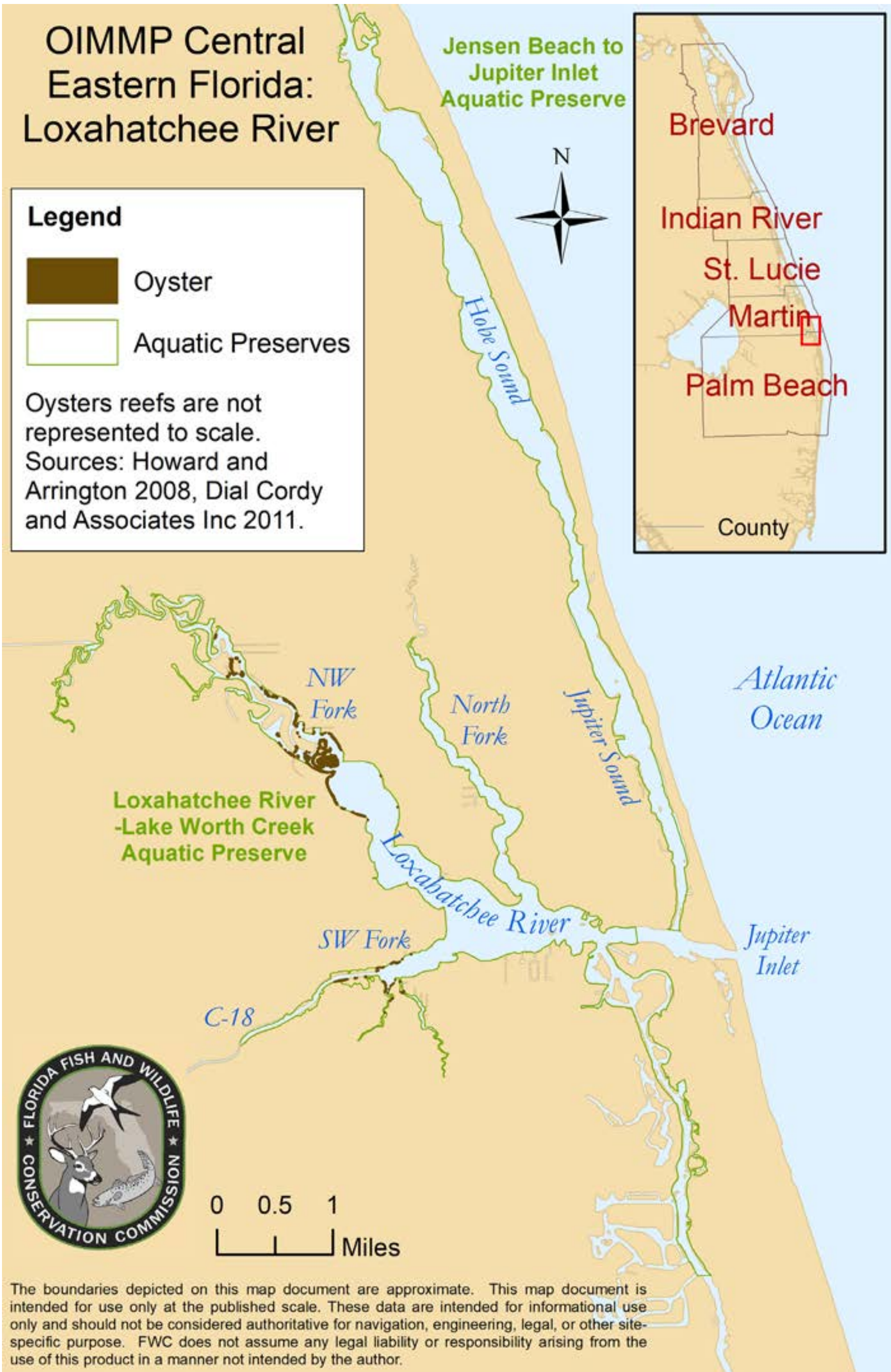

Figure 8.7. Oyster extent in the Loxahatchee River. Oyster mapping sources: Dial Cordy and Associates Inc. 2011 (made from 2011 side-scan sonar) and Howard and Arrington 2008 (from 2008 field surveys).

Oysters were intermittently present along Jupiter Inlet and Hobe Sound, their abundance influenced by the opening and closing of the inlet (Gambordella et al. 2007). Oyster populations declined by the 1990s; today a moderate population is found in the central embayment and northwest fork of the Loxahatchee River and a smaller assemblage in the southwest fork (Fig. 8.7; Gambordella et al. 2007, Howard and Arrington 2008). While some oysters have been documented on the north fork of the river (Law Environmental 1991), this area has not been mapped. In an average year, salinity ranges between 15 and 20 within the northwestern fork of the Loxahatchee River, within the optimal range for oyster survival. Salinity in the southwestern Loxahatchee River is consistently higher, closer to 30 (Parker and Geiger 2010). Both branches of the river support moderate to high densities of oysters, with the northwestern fork generally containing more suitable substrate for oysters and higher densities on reefs and associated with mangroves (Fig. 8.8), particularly 6.5–11 km (4–7 mi) upstream (Parker et al. 2013, SFWMD 2002b).

The Loxahatchee River has been affected by salinity fluctuations associated with storms and water management practices, but a large-scale oyster die-off has not been documented (Parker 2015). Densities of live oysters were lower in 2005 than in subsequent years, reflecting the impact of the active 2004/2005 storm seasons, but oyster populations rapidly recovered under the higher salinity in 2006. In 2016, salinity decreased in the Loxahatchee River as a result of increased freshwater flow associated with El Niño precipitation. Oysters upstream in the northwestern Loxahatchee River experienced a die-off, but higher salinity in the rest of the river supported a thriving population (Parker and Radigan 2017).

Although the effects of low-salinity events are often more acute, long-term exposure to high salinity can also be harmful. In the Loxahatchee River, salinity often exceeds the optimal range, and during such periods disease and predation rates increase substantially. Oysters that

Figure 8.8. Fringing eastern oyster (*Crassostrea virginica*) reefs adjacent to red mangrove (*Rhizophora mangle*) shorelines in the Loxahatchee River. Photo credits: Chris Kirby and Richard Radigan.

are physiologically acclimated to high salinity may be less able to cope with additional stresses, such as a rapid decrease in salinity. It follows that an oyster adapted to high salinity that has been weakened by a parasitic infection, e.g., dermo, would be even less likely to be able to withstand extreme changes and would more rapidly die when salinity decreases abruptly. As a consequence, oyster densities in the Loxahatchee River are likely moderated by the higher disease and predation rates.

Lake Worth Lagoon

The Lake Worth Lagoon (LWL) is a long, narrow lagoon that extends 35 km (22 mi) along coast of Palm Beach County (Fig. 8.9). The LWL began as Lake Worth, a freshwater lake that received sheet flows of surface water from the Everglades. The lake became a saline lagoon following construction of artificial inlets to the Atlantic Ocean in the early 1900s (CERP 2005). More than 81% of the lagoon's present shoreline is surrounded by urban development (PBC DERM 2008); mangroves occupy some islands and patchy regions of the shoreline. The LWL receives freshwater input from the Earman River, West Palm Beach Canal, and Boynton Canal (C-17, C-51, and C-16 canals, respectively) (SFWMD 2013).

Oysters were not present in Lake Worth while it was a freshwater lake, but they occurred intermittently after the artificial inlets created the brackish lagoon (Gambordella et al. 2007). The LWL supported an oyster fishery in the early 1900s, but it had declined by the 1950s (Linehan 1980, Gambordella et al. 2007). Oysters are not harvested in the LWL at present, nor does the region include any other shellfish-harvesting areas (FDACS 2017). Oysters can still be found on mangroves, jetties, edges of spoil islands, limestone rock revetments, and other hard substrates surrounding mangrove and salt marsh restoration projects, especially in the central LWL (Fig. 8.10; Gambordella et al. 2007, LWLI 2013).

Salinity in the LWL is generally in the range optimal for oysters only during the summer rainy season and is greater than 30 during the rest of the year (Parker and Geiger 2010). The LWL seldom experiences low-salinity events. The LWL and Loxahatchee River have higher salinity than the SLE and IRL, resulting in increased densities of predators (Arnold et al. 2008, Parker and Geiger 2013). Survivorship studies of tagged oysters left in cages in the LWL and Loxahatchee River showed poor survivorship, particularly in open cages in areas in which the oysters were exposed to predators (Parker and Geiger 2013). Dermo has been found to be relatively common in both the Loxahatchee River and the LWL, although intensity levels are low (below 2 on the scale of 0 to 5 developed by Mackin 1962; Arnold et al. 2008).

Threats to oysters in central and southeast Florida

- **Altered hydrology:** The hydrology of southeast Florida has been significantly altered by construction of drainage canals, both increasing the size of the watershed and altering the rate of surface water delivery. This has resulted in even more freshwater flow during the wet season and less during the dry season. While dredging and stabilization of inlets have increased the amount of salt water entering local estuaries, salinity is highly variable as a result of large freshwater pulses. Stormwater releases from Lake Okeechobee can reduce

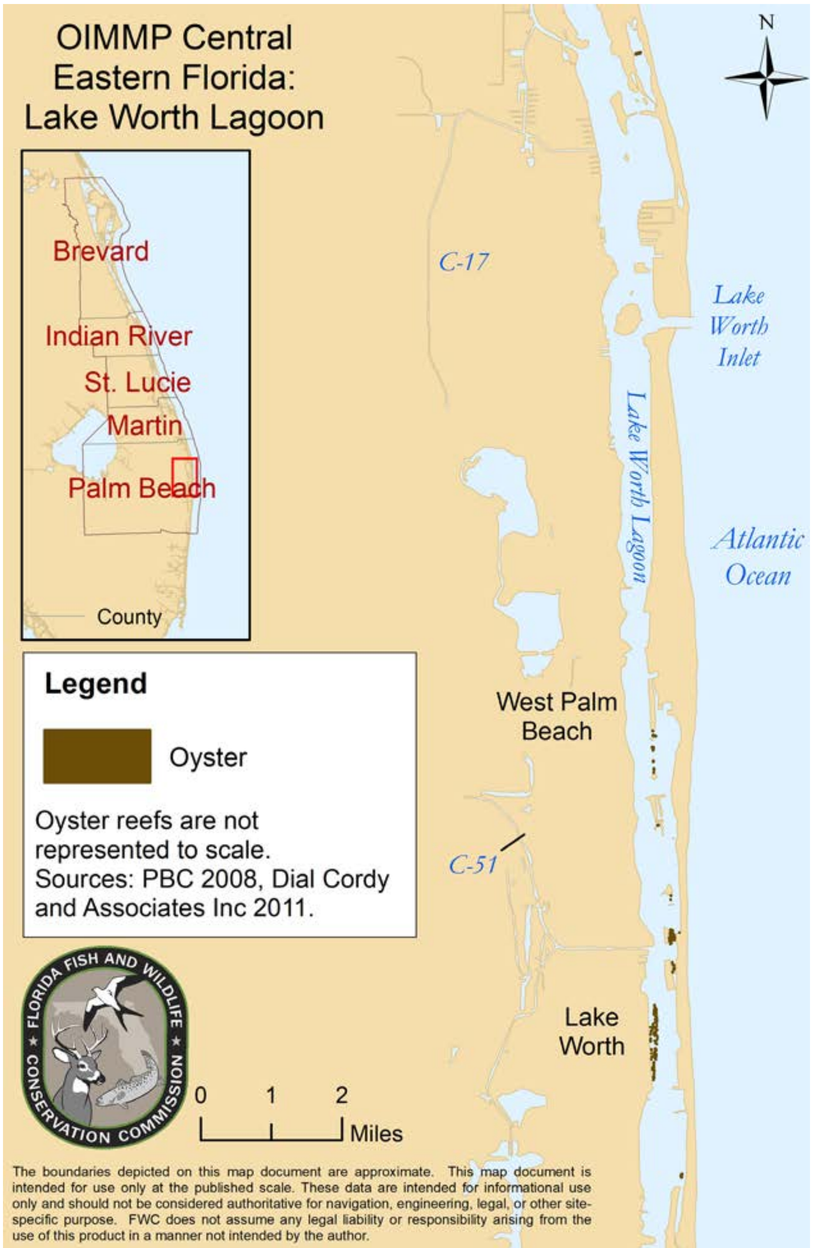

Figure 8.9. Mapped oyster extent in the Lake Worth Lagoon. Oyster mapping sources: PBC 2008 (made from 2007 aerial photographs) and Dial Cordy and Associates Inc. 2011 (from 2011 side-scan sonar).

salinity below tolerable levels for oysters in the SLE, resulting in periodic die-offs (Arnold et al. 2008, Parker et al. 2013, Parker and Radigan 2017). Sedimentation and muck accumulation brought by the extensive freshwater flow also pose a threat to affected reefs (Bartell et al. 2004). Dredging of the ICW and the creation and expansion of spoil islands further complicates natural water circulation in the basins.

- **Isolated populations:** The recovery of reefs after freshwater-induced mortality depends on the hydrologic transport of larval recruits from other reefs. The survival and resilience of local populations is of concern due to their isolation within rivers along the coast and vulnerability to continued urban development (Arnold et al. 2008).
- **Habitat loss:** The coastline of southeast Florida is heavily developed, and much of the natural shoreline has been replaced by seawalls and other hardened shorelines. Oyster reefs have directly faced habitat loss as a result of dredging and hardened shorelines, and remaining reefs must cope with the indirect impacts of a large human population including altered hydrology, nutrient loading, and associated phytoplankton blooms.
- **Sea-level rise:** Intertidal exposure offers oysters a temporary refuge from predation, pests, and disease (Bahr and Lanier 1981). Increased submergence times and higher salinity (due to both sea-level rise and reduced freshwater flow) lead to increased susceptibility to marine predators, pests (e.g., the boring sponge *Cliona celata*; Carroll et al. 2015), and pathogens (Shumway 1996). Sea-level rise will result in longer submergence times for intertidal habitats, decreasing the intertidal exposure time that is often the only respite from predation (optimal exposure frequency for eastern oysters is around 20–40% according to Ridge et al. 2015). This will encourage oyster settlement that results in vertical accretion on reefs or upslope migration onto shorelines with appropriate elevation. Natural tidal-phenomena, combined with sea-level rise, also impact oyster reef exposure time. The highest spring tides ("king tides") typically occur during the fall in south Florida and cause extended periods of high and low water.

  An additional effect of sea-level rise is that increased salt water intrusion into an estuary will push the zone of optimal salinity further upstream. In estuaries that narrow in geographic area upstream, this may decrease the size of the area of optimal salinity for oyster growth.

- **Boating impacts:** Boating channels and the ICW extend through the lagoons and rivers of southeast Flor-

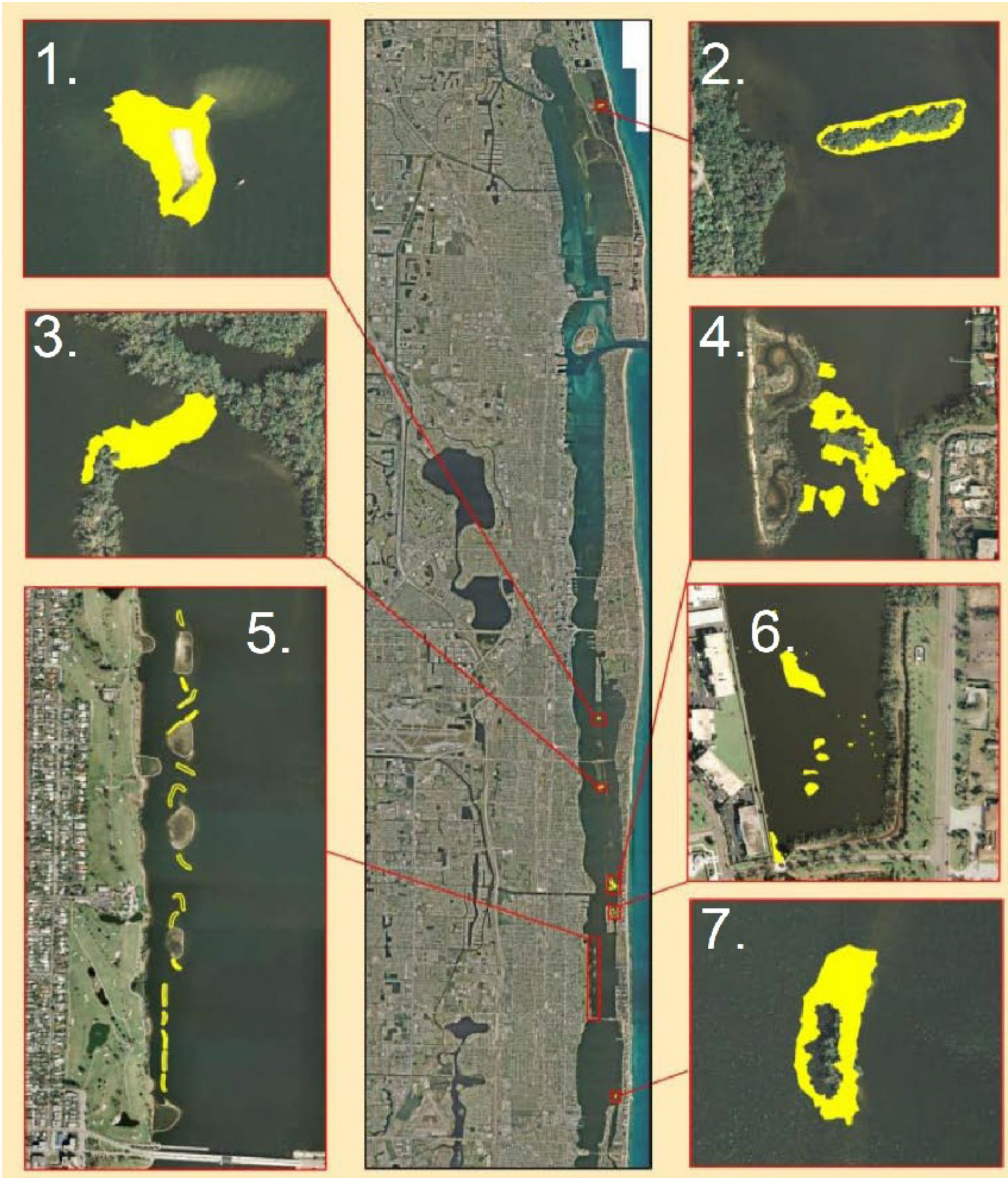

Figure 8.10. Detail of oyster reefs within the Lake Worth Lagoon. Image source: PBC 2011.

ida, and prop wash can erode the margins of oyster reefs. Boat wakes in the Mosquito Lagoon ICW have eroded salt marshes and oyster reefs to the extent that some have been reduced to intertidal sand flats (Grizzle et al. 2002, Wall et al. 2005). Campbell (2015) documented that boaters can lessen their impact on oyster reefs if they reduce wake by being on plane or boating at a slow speed.

- **Invasive species:** Invasive estuarine invertebrates along the east coast of Florida include the Asian green mussel (*Perna viridis*), charru mussel (*Mytella charruana*), and pink barnacle (*Megabalanus coccopoma*). The Asian green mussel has been found as far south as Cocoa on the IRL, and the pink barnacle has been found intermittently along the east coast of Florida (Spinuzzi et al. 2013). Charru mussels are native to Central and South America and were first noticed on the east coast of Florida in 1986, in Jacksonville (Galimany et al. 2017). These nonnative species are invasive, competing with native oysters for habitat and food (Gilg et al. 2012, 2014; Yuan 2016, Galimany et al. 2017). Population-genetic research suggests the Asian green mussel and pink barnacle were transported in ballast water or on ships' hulls (Spinuzzi et al. 2013, Cohen et al. 2014). The parasitic rhizarian *Bonamia* sp. was first reported in Florida in 2007 (Dungan et al. 2012) and has been documented in the LWL and southern IRL (Laramore et al. 2017). During 2016–2017, it was found in tissue sampled from oysters from the north, central, and southern IRL (Laramore 2017).

Oyster reef mapping and monitoring efforts

The compilation of oyster maps used in figures in this report are available for download at http://geodata.myfwc.com/datasets/oyster-beds-in-florida.

Comprehensive Everglades Restoration Program oyster monitoring

Oysters have been monitored as part of the CERP RECOVER program by FWC since 2005. Ongoing sampling efforts occur on oyster reefs in the SLE and Loxahatchee River; oyster populations were also previously monitored in the Mosquito Lagoon, Sebastian River, LWL, Biscayne Bay, and Tampa Bay (Arnold et al. 2008). Monitoring includes measuring oyster density and size distribution twice a year and monthly assessment of reproductive development, larval recruitment, dermo prevalence and intensity, growth, and survivorship. Methods for and results of these monitoring efforts can be found in a variety of reports (Arnold et al. 2008, Parker et al. 2013, Parker 2015, Parker and Radigan 2017).

U.S. Army Corps of Engineers benthic habitat mapping

Dial Cordy and Associates Inc. was contracted by the U.S. Army Corps of Engineers to map substrate in the Caloosahatchee River Estuary, SLE, Loxahatchee Estuary, and LWL (Dial Cordy and Associates Inc. 2011). Side-scan sonar was used to map areas from the 2.7-m (9-ft) bathymetric contour to shore (~1 m depth). Resulting maps of bottom type (seagrass, oyster bed, shell, muck, etc.) were created through the interpretation of high-resolution aerial photography as well as side-scan sonar. Substrates were verified with ground truthing, and the density of live oysters was also quantified on mapped oyster reefs.

FWC baseline mapping and monitoring

In the winter of 2005–2006, FWC and Golder Associates mapped oyster reefs in the Sebastian River, SLE, LWL, and Biscayne Bay using a real-time kinematic global positioning system (RTK GPS) (Gambordella et al. 2007). Oyster reefs were identified using oyster maps (URS Greiner Woodward Clyde Inc. 1999, Williams 1999, Bachman et al. 2004), helicopter aerial surveys, and sounding lines. Elevation, latitude, and longitude of reefs were mapped in 1-m intervals. Oysters were removed from 0.25-m^2 quadrats placed haphazardly on the reef to determine shell height and density of live oysters and of boxes (i.e., paired oyster shells, evidence of recent mortality). The effort mapped 152 reefs covering 12 ha (30.5 ac) (Gambordella et al. 2007).

Palm Beach County habitat mapping

In 2007, the Palm Beach County Habitat Mapping Project used aerial photography to map oyster reefs, seagrass, and coastal wetland habitat in LWL and the ICW; 1.7 ha (4.2 ac) of oyster reef was mapped (PBC 2008). Aerotriangulation, digital orthophotography, field work, photointerpretation, and trend analyses were used to map these coastal resources. A classification guide for identifying habitat from aerial photography was created and published in the final report (PBC 2008).

St. Lucie River oyster mapping

GIS data layers of oyster extent in the SLE have been assembled for 1940–1960, 1960–1980, and 1990–1996 (Fig. 8.6; URS Greiner Woodward Clyde Inc. 1998, FDEP 2009c). More recent mapping surveys were conducted in 1997 (URS Greiner Woodward Clyde Inc. 1999), 2003 (Ibis Environmental Inc. 2004), and 2006 (Gambordella et al. 2007). These surveys are compared and their vertical mapping techniques assessed by Gambordella et al. (2007). Although the 2006 survey did not map all reefs in the river, the reefs that were mapped were much smaller in area and had migrated toward shore than those in 1997 and 2003. In 2018, SFWMD mapped the SLE reefs to support CERP RECOVER monitoring efforts and forthcoming updates to RECOVER interim goals and targets. Following Hurricane Irma in September 2017, persistent freshwater releases from Lake Okeechobee and the SLE watershed resulted in an oyster die-off. An active rainy season in early 2018 delayed oyster recovery in the SLE until the fall, so work was postponed until 2019 to allow the newly settled oysters to grow prior to mapping.

St. Lucie Estuary water quality monitoring

Flow and salinity data for the SLE and the Caloosahatchee River are monitored by FDEP and graphed as seven-day averages at http://publicfiles.dep.state.fl.us/owp/SalinityReports/SalinityUpdate.html. The website also includes water height data for Lake Okeechobee. Real-time water quality monitoring stations have been established by the U.S. Geological Survey, in cooperation with the SFWMD, at several locations along the SLE (https://nwis.waterdata.usgs.gov). The SFWMD takes monthly grab samples at several stations throughout the main estuary and the north and south forks. Additional continuous-water-quality sondes have been or are deployed at several locations (Willoughby Creek, midestuary, Palm City Bridge, and Boy Scout Island in the south IRL) to supplement analyses conducted in-house by SFWMD scientists. Florida Atlantic University's Harbor Branch Oceanographic Institute operates five land/ocean biogeochemical observatory stations in the SLE that transmit data to its website (http://fau.loboviz.com/). The Ocean Research and Conservation Association also has five real-time water-quality observatories in the SLE (http://api.kilroydata.org/public/). Volunteers also take weekly water quality measurements in the Florida Oceanographic Society's citizen science program (www.floridaocean.org).

Lake Worth Lagoon monitoring

From 2008 to 2010, Palm Beach County Environmental Resources Management partnered with John Scarpa and Susan Laramore from the Harbor Branch Oceanographic Institute to monitor the health and productivity of reefs in LWL (Scarpa and Laramore 2010, LWLI 2013). This study was designed to build upon FWC's oyster monitoring in LWL. Monitoring parameters included size, physiological condition, reproductive potential, and density of adult oysters; presence and intensity of dermo and MSX (*Haplosporidium nelsoni*); larval recruitment and growth; and water quality. In 2015, FWC partnered with Palm Beach County to continue their monitoring efforts at the reef locations previously monitored as part of the CERP RECOVER program; three recently restored oyster reefs were also added to the established sampling regime.

Loxahatchee River mapping and monitoring

In 2003, the Loxahatchee River District's WildPine Ecological Laboratory and SFWMD examined the health and extent of oysters in the Loxahatchee River (Bachman et al. 2004). The perimeter of live oyster beds in the northwest and southwest forks of the Loxahatchee River were mapped using a hand-held GPS unit and the resulting outlines overlaid on color aerial photographs. A 1-m^2 quadrat was placed in 14 locations on the mapped reefs; the shell height of live and dead oysters within the quadrat were measured (Bachman et al. 2004). A follow-up mapping effort in 2008 was conducted by the WildPine Ecological Laboratory (Howard and Arrington 2008). The perimeters of live oyster reefs were mapped using a differential GPS, and oysters were removed from 0.25-m^2 quadrats to determine density and shell height. Spat recruitment was also monitored. A total of 91 reefs were mapped totaling 6.1 ha (15 ac). Mapping and monitoring reports for the Loxahatchee River as well as shapefiles of the 2008 mapping data are available at https://loxahatcheeriver.org/river/oyster/.

Central IRL mapping

During winter and spring of 2018, FWC, FDEP IRL aquatic preserves, and the Smithsonian Marine Station mapped previously unmapped reefs in the central IRL (e.g., Vero Beach, Fort Pierce) on foot using a handheld Trimble RTK GPS unit and aerial photography captured by drones (FWC et al. 2018). The map data and drone imagery will be processed in GIS by the FWC Spatial Analysis Program.

Oyster reef restoration projects

In 2009, NOAA awarded $4 million to Martin County to fund the Oyster Reef Restoration Project in the St. Lucie (8.4 ha/20.7 ac of reef) and Loxahatchee (1.4 ha/3.4 ac of reef) estuaries (LRD and FIU 2009, Parker and Geiger 2012). Construction of reefs was finished in 2010 and mapped and monitored in June 2012 (Parker and Geiger 2012). Managed by CSA International Inc., the effort involved the placement of more than 18,000 metric tons (40 million pounds) of concrete, limestone, and cultch at multiple sites in the two estuaries. A variety of parameters of the oysters and their associated communities was monitored by FWC, Florida International University, Florida Atlantic University, Florida Oceanographic Society (FOS), and Estuarine Coastal and Ocean Science Inc. (Jud and Layman 2011, Parker and Geiger 2012). The studies found that the substrates were rapidly colonized by oysters and that three of the five restored reefs achieved densities, reproduction, and mortality levels similar to those of natural reefs (Parker and Geiger 2012). The abundance and diversity of other reef fauna also reached levels approaching those natural reefs, although after 14 months there were still significant differences in species composition between the constructed reefs and natural reefs (Jud and Layman 2011). Some sites experienced mortality as a result of sedimentation and burial (Parker and Geiger 2012). Craig Layman's lab at Florida International University has been conducting oyster reef studies in the Loxahatchee River focusing on community ecology of natural and restored reefs and their filtration capacity (Yeager and Layman 2011, Layman 2012).

Additional restoration efforts in the SLE and IRL have been continued by Martin County since the initial projects in 2010. Oyster reef and living shorelines were installed at four sites in the SLE as community-based projects organized by Martin County and FOS. Two of these sites have been monitored for recruitment and oyster density and growth by FOS. In response to 2016 freshwater discharges, oyster survival was monitored at one restoration site in the central IRL. Additional living shoreline were established in the IRL at 10 sites by Martin County or FOS. Seven sites, established in 2015–2016, are monitored by FOS for recruitment, oyster density and growth. Associated species have been monitored at two of these sites. The St. Lucie County artificial reef program and its partners have used bagged oyster shell to establish reefs in Wildcat Cove and along City of Fort Pierce properties and some spoil islands in the IRL.

In Brevard County, numerous oyster reefs have been established and additional ones are planned, largely as part of Brevard's Save Our Lagoon program (http://www.brevardfl.gov/SaveOurLagoon/Home). The county has partnered with Brevard Zoo, the University of Central Florida, and others to create reefs, many of which are associated with living-shoreline projects whose design incorporates native-plant species. The reefs will be monitored so that their success can be evaluated and adaptive management strategies can be developed.

Most of Florida's east coast contains a comprehensive oyster shell recycling program. Numerous entities (mostly nongovernmental organizations as well as waste-management firms and the National Estuary Programs) in the IRL region have created a partnership called Shuck and Share to facilitate shell recycling from restaurants and shucking houses, shell curing, and provision of shell to those involved in regional restoration projects.

IRL oyster health snapshot

During 2016–2017, State agencies (FWC, FDEP) partnered with the Harbor Branch Oceanographic Institute to assess organismal health of oysters on natural and restored (or created) reefs throughout the IRL using standard histology techniques (Laramore 2017). The study provides baseline data for ongoing restoration efforts in the IRL. Preliminary findings show largely latitudinal and seasonal trends among health indicators, and restored reefs are similar to nearby natural reefs ones for most parameters studied. Few adults were observed on restored reefs, possibly because the reefs were so new. Prevalence and intensity of pests and disease were moderate to high across the IRL but varied latitudinally (e.g., dermo prevalence increased with latitude).

NOAA Mussel Watch

The NOAA National Status and Trends Program has monitored pollutants in bivalves through the Mussel Watch program across the coastal United States since 1986. The Sebastian River is the only Mussel Watch location in central and southeast Florida. Oysters are monitored for concentrations of heavy metals and organics. In 2004–2005, concentrations of copper, mercury, and lead in Sebastian River oysters were reported as medium to high (Kimbrough et al. 2008). Copper concentrations are often high in Florida oysters because copper is used in fungicides, algicides, and antifouling paints (Kimbrough et al. 2008).

MarineGEO and the Tennenbaum Marine Observatories Network

The Marine Global Earth Observatory (MarineGEO, https://marinegeo.si.edu/), directed by the Smithsonian Institution's Tennenbaum Marine Observatories Network (TMON), is a long-term, worldwide research program that focuses on understanding coastal marine biodiversity and its role in maintaining resilient ecosystems around the world. Under the program, the IRL Aquatic Preserves and the Smithsonian Marine Station, both in Fort Pierce, have teamed up to monitor and map six oyster reefs in the region (two in Indian River County and four in St. Lucie County). Monitored reefs include spoil island fringe reefs, small patch reefs and fringing reefs. Summer monitoring activities include recording oyster percent cover, density, and size-frequency distributions. A series of bio-boxes (50 × 50 × 10 cm) were also deployed for several months and retrieved to quantify abundance and diversity of species that inhabited the boxes. At three of the six sites, additional studies were conducted to examine predation, recruitment, and community development.

MarineGEO and the University of Florida formed the Florida Census of Marine Life, a working group focused on collecting and identifying marine species around Florida with a strong focus on marine invertebrates within the IRL. The goal of the working group is to create a baseline of biodiversity and a species inventory for future research and to facilitate monitoring of nonnative species. Marine invertebrates are collected, identified, photographed, DNA barcoded, and physical specimens are catalogued in the university's Florida Natural History Museum.

Recommendations for management, mapping, and monitoring

- Develop habitat-suitability guidelines for creating new oyster restoration projects to ensure proper site selection and increase rates of oyster settlement. The SFWMD is currently developing a habitat suitability model for the SLE based on temperature, salinity, and preferred substrates.
- Develop consistent restoration monitoring protocols and success criteria for evaluating restoration efforts in this region. Monitor indicators of oyster reef health (e.g., reef area, reef height, oyster density, and oyster size-frequency distribution) as well as environmental variables (water temperature, salinity, and dissolved oxygen) that will allow comparison of abiotic parameters to oyster reef metrics (Baggett et al. 2014, 2015). Duration of intertidal exposure should also be included among those environmental indicators (Walles et al. 2016), particularly in the context of sea-level rise. Oyster health and histology assessment should be included in routine monitoring (e.g., diseases, sex ratios, physiological status).
- While oysters have been mapped within the major tributaries of the IRL, oyster reef mapping data are lacking in the main portion of the lagoon and its minor tributaries. In central and southeast Florida, much oyster growth occurs on mangrove roots or along seawalls. Oysters at the periphery are seldom mapped by traditional efforts, which rely on aerial photography; on-site mapping should be completed to identify oysters on seawalls and mangrove roots.
- Oyster extent in southeast Florida fluctuates as a result of urban development, variable freshwater flow, and changing freshwater management. Maps of oyster extent in these estuaries should be updated every several years.
- Install living shorelines, replacing bulkheads and other artificial shorelines with oyster reefs, mangroves, and other native vegetation (FDEP 2014). Living shorelines mimic natural shorelines, which filter surface water runoff, provide habitat for other animals, and better allow for upslope migration of coastal habitats in the face of sea-level rise.

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Chapter 9
Northeast Florida

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Description of the region

The coast of northeast Florida contains an extensive network of salt marshes, estuarine lagoons, oyster reefs, and tidal creeks (Fig. 9.1). Spanning the transition between subtropical and temperate climates, mixed mangrove and salt marsh vegetation in coastal Volusia County gives way to salt marsh dominance in the north (Nassau and Duval Counties). Estuarine salt marshes and oyster reefs are protected from ocean energy by barrier islands and sand ridges. Freshwater input is provided by several rivers, the largest of which is the St. Johns River, as well as numerous natural creeks, man-made canals, and stormwater outfalls. The entire northeast region of Florida is within the jurisdiction of the St. Johns River Water Management District (SJRWMD).

Oyster reefs in northeast Florida are generally intertidal, rather than subtidal, and are commonly found as patches independent from the shoreline or as fringing reefs along the margins of salt marshes. Most oyster reefs occur at elevations between mean low water and mean sea level (Ridge et al. 2017). The northeast coast of Florida experiences a larger tidal range than do the southeast and Gulf coasts, and the range decreases from north to south. Regions with greater tidal energy generally have greater oyster biomass and more reef structure (Byers et al. 2015), and the northern coastal counties contain extensive eastern oyster (*Crassostrea virginica*) reefs in their tidal tributaries (Table 9.1).

Figure 9.1. Oyster extent in northeast Florida.

Table 9.1. Number and extent of oyster reefs in northeast Florida. Data updated from Walters et al. 2015.

County	Number of Reefs	Reef area (ha)	Reef area (ac)
Nassau	6,813	158.1	390.7
Duval	3,561	103.7	256.2
St. Johns	4,848	331.8	819.9
Flagler	1,099	18.1	44.8
North Volusia	1,632	40.1	99.2
Total	17,953	651.8	1,610.7

Nassau and Duval counties

The northernmost coastal counties, Nassau and Duval, have extensive oyster reefs in tidal tributaries and the St. Marys, Amelia, Nassau, Fort George, and St. Johns rivers (Fig. 9.2). These oyster reefs line shallow mud flats, tidal creeks, and fringe salt marshes dominated by smooth cordgrass (*Spartina alterniflora*). Large portions of the sovereign submerged lands are designated as outstanding Florida waters, aquatic preserves, or are located within the Timucuan Ecological and Historic Preserve. Spring tidal range is 1.9 m (6.4 ft) in the St. Marys River on the Georgia border and 1.4 m (4.5 ft) at the St. Johns River mouth (NOAA 2017). Waters in the lower 40 km (25 mi) of the St. Johns River are considered mesohaline, with a mean salinity of 14.5 (LSJRBR 2016). Mean salinity in the St. Johns River increases from around 3 near Jacksonville to 26 at the river's mouth; salinity in the Nassau River increases from around 7 near I-95 to 27 near Nassauville (Fig. 9.2; USNPS 1996). Average salinity in the St. Johns River increased from 1996 to 2007 as a result of sea-level rise and reduced freshwater input (LSJRBR 2016).

Alligator Creek, located south of the Amelia River, and parts of the Nassau River and Sound are classified as Class II waters (i.e., they are designated for fish consumption, recreation, and maintenance of fish and wildlife), but harvest has been prohibited since the mid-1980s. Shellfish harvesting area #96 includes the Fort George River and Nassau Sound (Fig. 9.2), but harvesting is prohibited there as well. The Florida Department of Environmental Protection (FDEP) has designated the Fort George River and the beaches at the south end of Amelia Island in Nassau Sound as 303(d) impaired water bodies because of bacteria in shellfish. South of the St. Johns River, Hopkins Creek, which flows from the barrier island into the intracoastal waterway (ICW), has designated total maximum daily loads (TMDLs) for dissolved oxygen, fecal coliform bacteria, and nutrients (Chl a) (Murray and Rhew 2010, SJRWMD and Frazel 2016b).

St. Johns and Flagler counties

The St. Johns River is connected to the Tolomato River via the ICW. Oysters exist in tidal creeks along this connection, but seawalls limit intertidal oyster extent around Palm Valley (Fig. 9.3). South of Palm Valley, the Tolomato River continues into the Guana Tolomato Matanzas National Estuarine Research Reserve (GTMNERR) and Guana River Marsh Aquatic Preserve. The GTMNERR includes approximately 30,350 ha (75,000 ac) of relatively undeveloped coastal and estuarine habitat. The reserve is separated into northern and southern components, with the city of St. Augustine at the center, and is named after the Guana, Tolomato, and Matanzas rivers (Fig. 9.3). Pellicer Creek Aquatic Preserve, located in the southern component of GTMNERR, is the largest tributary of the Matanzas River. The GTM estuary includes vast salt marshes dominated by smooth cordgrass, black needlerush (*Juncus roemerianus*), and other high-marsh species. Mangroves are expanding rapidly in the southern reaches of the Matanzas River. Salinity varies from near-freshwater conditions in tributaries to near-oceanic salinity (28–35) at the inlets to the Atlantic Ocean (Frazel 2009). Tidal range averages 1.5 m in the ICW (GTMNERR data). Intertidal oyster reefs are extensive in the low-energy lagoons and tidal creeks (Fig. 9.4).

Two conditionally approved areas for shellfish harvesting (St. Johns North, Area #92, and St. Johns South, Area #88; Fig. 9.5) and four shellfish leases held in perpetuity are located in the GTM estuary (FDACS 2017). Two of those leases are in conditionally restricted waters (Guana River), one is in conditionally approved waters (Matanzas River), and one is in the Summer Haven River, which filled with sand after a barrier island breach in 2008. Oyster harvesting is restricted in several Class II water bodies (those designated for shellfish propagation and harvesting) due to concerns about pollution. Modern oyster landings in northeast Florida peaked in 1990, at 87 metric tons (190,000 lb) (Fig. 9.6). Commercial oyster landings declined after 1990 but had begun to increase again by the mid-1990s. Despite declines in some years, the general increase in landings has apparently been driven by increased harvest effort, as overall catch per unit effort (CPUE; pounds harvested per trip) has declined slightly since 1986 (Fig. 9.6b). Landings before 1986 were reported voluntarily, so CPUE is not shown in Figure 9.6b before 1986. Recreational oyster harvesting continues in the region but remains unquantified and difficult to estimate.

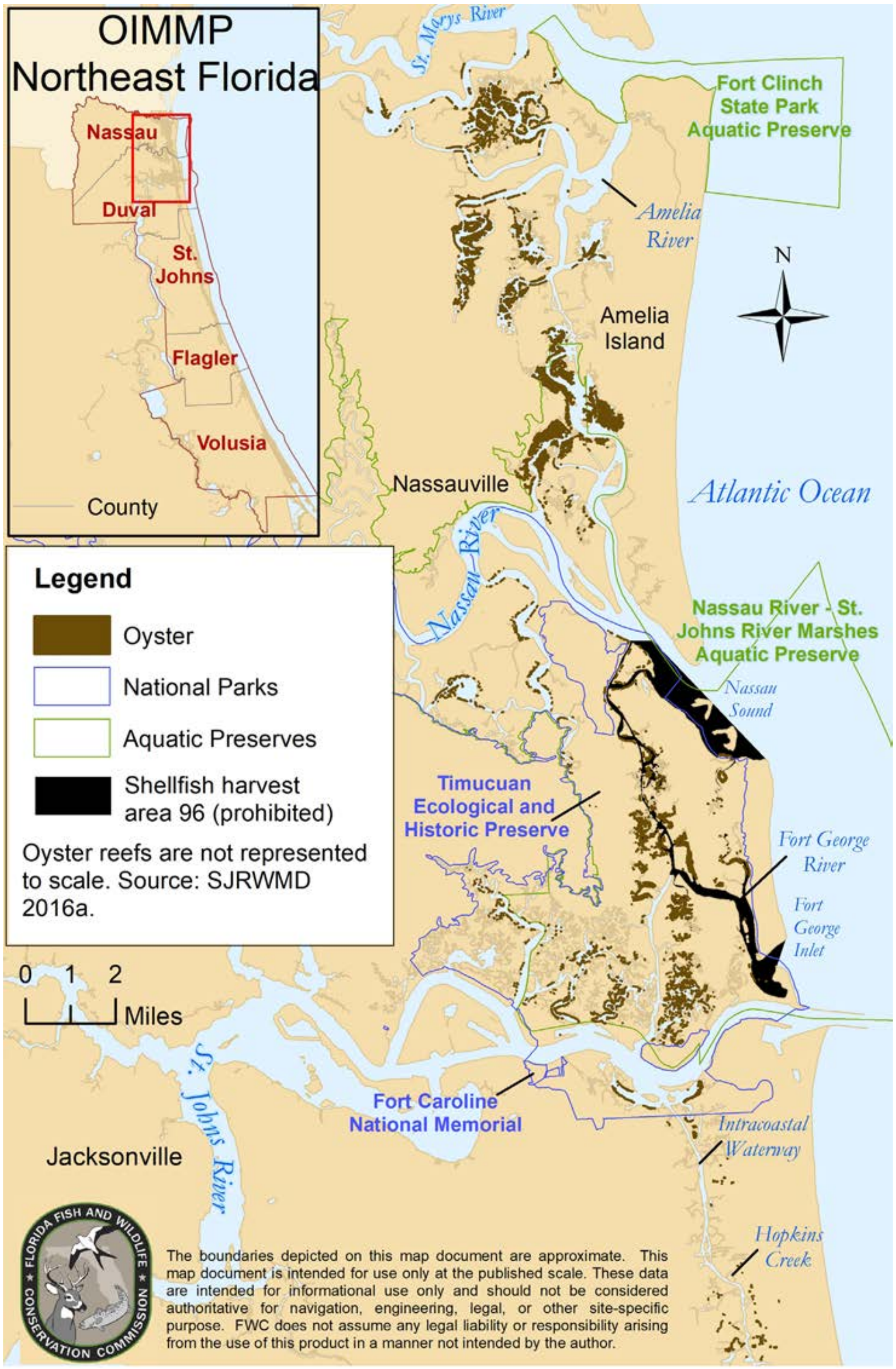

Figure 9.2. Oyster reef coverage in Nassau and Duval counties. Oyster mapping source: SJRWMD 2016a (made from 2002–2014 aerial photography).

A number of water bodies in the GTM estuary are listed as impaired due to bacteria and the Matanzas River south of Matanzas Inlet is impaired due to high nutrients (Chl a) (Frazel 2009, SJRWMD and Frazel 2016b). The only TMDL established for the GTM estuary is for fecal coliform bacteria in Pellicer Creek (Bridger 2012). South of Pellicer Creek, the ICW borders the Palm Coast and in Flagler County becomes the Halifax River (Figs. 9.3 and 9.7). Much of the ICW from Palm Coast to the Tomoka River has a developed shoreline. Several segments along the ICW are impaired due to low dissolved oxygen, high nutrients, and fecal coliform bacteria (SJRWMD and Fra-

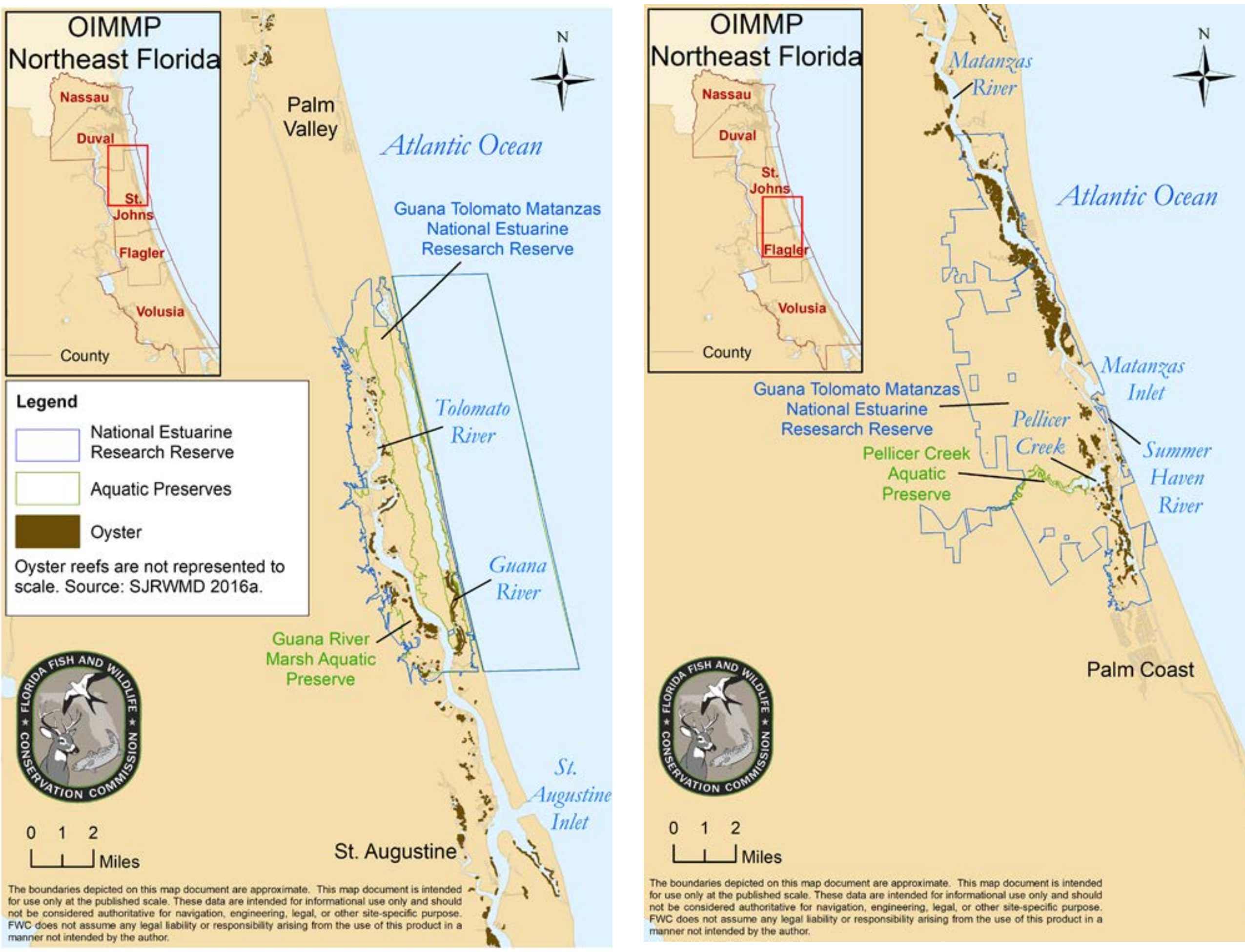

Figure 9.3. Oyster reef coverage in St. Johns and Flagler counties. Oyster mapping source: SJRWMD 2016a (made from 2002–2014 aerial photography).

zel 2016b). TDMLs for nutrients have been established for Palm Coast and the Tomoka River (Magley 2013a, Magley 2013b).

Use of living shorelines to restore patch and fringing oyster reefs has become more common in northeast Florida. Along the reach of Tolomato River within the GTMNERR, issues with shoreline erosion prompted the creation of ~260 linear meters (850 ft) of oyster reefs at Wright's Landing in 2012–2014. A new hybrid design consisting of breakwaters and oyster gabions was deployed at six new sites along the Tolomato River in 2017.

Volusia County

The Halifax River and Tomoka Basin form a narrow lagoon that connects to the Atlantic Ocean at Ponce de Leon Inlet (Fig. 9.7). The Halifax River has an average depth of 1.5 m (5 ft), and spring tidal range at Ponce de Leon Inlet is 1.0 m (3.2 ft) (NOAA 2017). Water levels are strongly influenced by wind speed and direction (FDEP 2017). Intertidal reefs are not found in great densities in the Tomoka Marsh Aquatic Preserve and Bulow Creek, but some sparse clusters of oysters are present along the shoreline. The waters of Bulow Creek have an average salinity of 11, but throughout the creek and Tomoka Marsh Aquatic Preserve salinity is dependent on rainfall, wind-driven tides, and storm surge (FDEP 2017). The small tidal amplitude around the Tomoka Basin (0.2 m; 0.75 ft) is not known to support intertidal reefs, although historical records indicate they have been present (SJRWMD and Frazel 2016b). Smith Creek and the waterway adjacent to High Bridge Road do support intertidal reefs. A large percentage of the shoreline on the Halifax River north of Port Orange is hardened and does not support shallow gradations of intertidal habitat (unpublished SJRWMD 2016 shoreline survey). Intertidal reefs become more abundant in the lower Halifax River near Port Orange (Fig. 9.7). Tomoka River and Spruce Creek are the only substantial freshwater inflows in the Halifax River.

Figure 9.4. Oyster reefs at low tide in the Matanzas River. Photo credit: Nikki Dix.

Mosquito Lagoon is a shallow lagoon with an average depth of 1.2 m (4 ft) that receives freshwater input predominantly through runoff, with small contributions from groundwater seepage, precipitation, small tributaries, and canals (FDEP 2009). Because it is enclosed by barrier islands and connects to the Atlantic Ocean only by the Ponce de Leon Inlet and to the Indian River Lagoon by the Haulover Canal, the lagoon is characterized by weak currents, a small tidal range (0.15 m/0.5 ft), and minimal flushing (Steward et al. 2010). Wind movement of water on time scales of weeks or months can exceed the normal diurnal tidal range, leading to irregular surface water levels (Smith 1993). Due to its restricted nature and small tidal prism, Mosquito Lagoon has a long water residence time and so is susceptible to poor water quality due to accumulation of pollutants (FDEP 2009). Salinity is generally between 25 and 36, although it can become hypersaline during times of high evaporation and low freshwater flow (Grizzle 1990, Parker et al. 2013).

The northern portion of Mosquito Lagoon is managed under the Mosquito Lagoon Aquatic Preserve, while Canaveral National Seashore and Merritt Island National Wildlife Refuge manage the remainder of the lagoon. Analysis of aerial imagery from 2010 revealed 2,542 oyster reefs in Mosquito Lagoon, 624 of which were within the boundaries of Canaveral National Seashore (Garvis et al. 2015). Of these totals, 8.9% of Mosquito Lagoon reefs were classified as dead, with a 24% loss of acreage when compared to 1943 aerial images. For Canaveral National Seashore, 17.5% of the reefs were classified as dead, with a 40% loss of acreage (Garvis et al. 2015). Many of the dead reefs in Mosquito Lagoon were associated with frequent boat wakes, which had dislodged oyster clusters from the sediment. South of Oak Hill/Eldora Hill, oyster abundance is minimal. This is potentially due to the lack of tidal range to support intertidal oysters as well as water levels that vary seasonally as a result of precipitation and wind patterns.

Canaveral National Seashore contains large shell mounds from pre-Columbian indigenous populations, including Turtle Mound, the largest shell midden on the east coast of Florida (FDEP 2009). These mounds and other Timucuan artifacts indicate a long history of oyster harvest by indigenous people in the area; archeological evidence indicates that the area was inhabited for more than 10,000 years before European settlers arrived. Many of these shell mounds were removed when they were mined for shell for use in roads and railroads or as fill (FDEP 2009).

A large portion of the Mosquito Lagoon is classified as a Class II water body (see Area #82 in Fig. 9.5), and oysters and clams are commonly harvested commercially and recreationally. Also, some active oyster leases in the southern portion of the lagoon were established be-

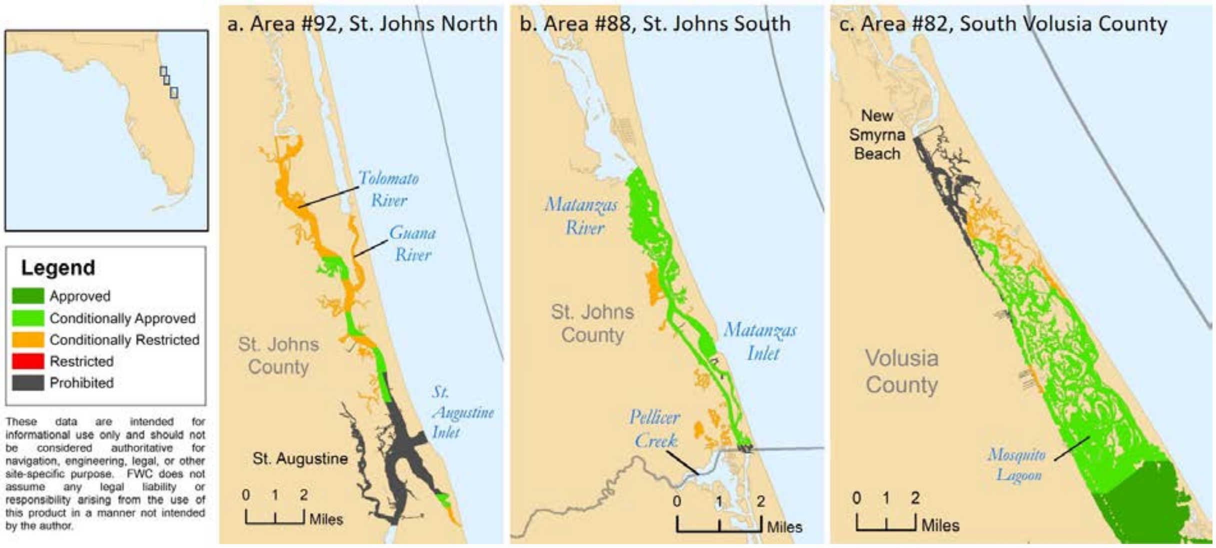

Figure 9.5. Shellfish harvesting areas on the northeast coast of Florida (Data source: FDACS 2017).

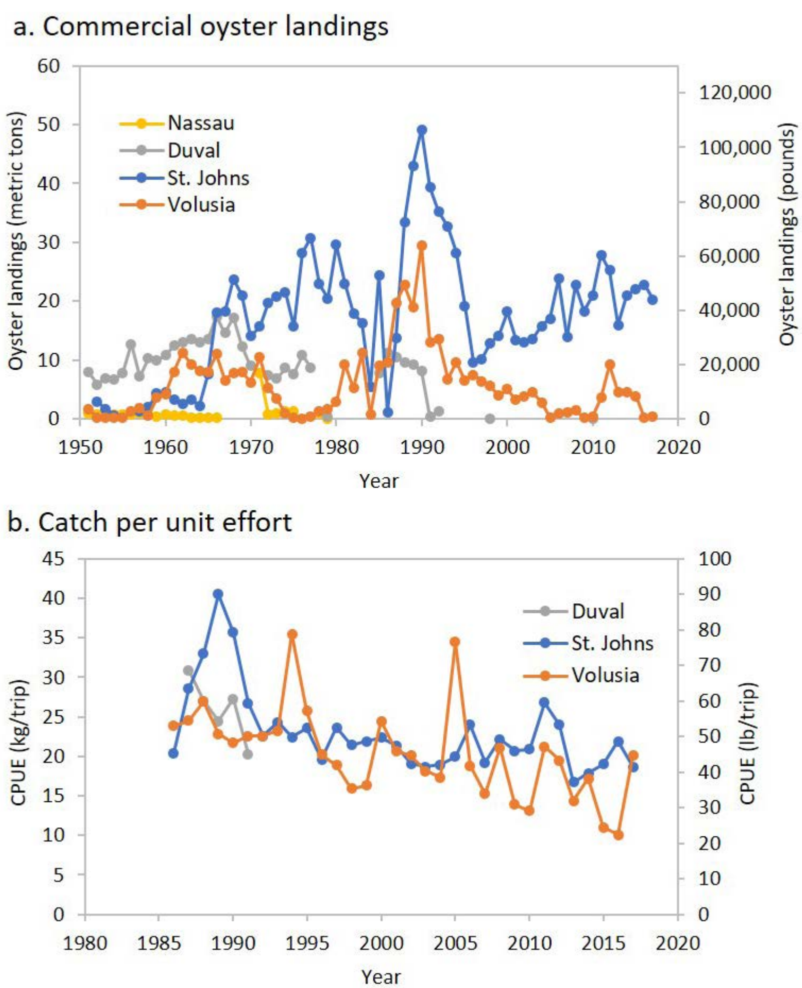

Figure 9.6. Commercial oyster landings (a) and catch per unit effort (CPUE; b) of commercial oyster landings in northeast Florida counties (Data source: FWC 2018 and Florida Commercial Marine Fish Landings, see Appendix A). Oyster landings before 1986 were collected under a voluntary reporting system.

fore the creation of national park (FDEP 2009). FDACS monitors water quality for concentrations of fecal coliform bacteria and toxic algae, including species that cause brown tide (*Aureoumbra lagunensis*), red tide (*Karenia brevis*), and paralytic shellfish poisoning (*Pyrodinium bahamense*). The lagoon has sometimes been closed to shellfish harvests due to high concentrations of toxins or algal cells (FDEP 2009).

The CPUE of oysters declined from 2000 to 2011 in the Mosquito Lagoon; this decline coincides with an increase in occurrence of the predatory crown conch (*Melongena corona*) associated with increased salinity and reduced freshwater input (Garland and Kimbro 2015). Crown conch larvae have higher survivorship in high salinity (Hathaway and Woodburn 1961, Garland and Kimbro 2015). Craig et al. (2016), however, found that crown conchs have a low density in this location, with an average of only 1 conch per 100 m^2 (1,074 ft^2). Thus, the decline in CPUE may also be related to other factors such as poor water quality, disease, and erosion due to boat wakes.

Oyster reef restoration has been carried out continuously in Mosquito Lagoon since 2007, and shell substrate has been prepared and deployed for the growth of an estimated 11 million oysters (Walters, unpublished data). Dead reefs have been transformed into living reefs by reducing the height of large piles of disarticulated shell above the high tide line and placing stabilized shell on the leveled area closer to or below the high tide line. The University of Central Florida (UCF) and its partners restored 83 reefs in Mosquito Lagoon with a footprint of approximately 1.2 ha (3 ac) and stabilized 1,865 m (6,120 ft) of shoreline between 2011 and 2017 in Canaveral National Seashore. Restored reefs have endured five hurricanes and three brown tides (e.g. Walters et al. 2007, Gobler et al. 2013). Restored reefs with more than 1,000 living oysters per m^2 (83 per ft^2) were present after nine years (Walters 2016). Significant recruitment of oysters along stabilized shorelines has also been documented (Walters et al. 2017). The wave energy hitting these shorelines is reduced by 69% when 1-year-old restored oysters and smooth cordgrass are present (Manis et al. 2015). Within one year of establishment, these restored oyster reefs achieve biogeochemical cycling similar to that of natural reefs (Chambers et al. 2018). Larval recruitment to restored reefs accumulates as much genetic diversity (expected heterozygosity and allelic richness in microsatellite loci) as to natural reefs as little as one month after shells are deployed. Likewise, harvesting did not impact genetic diversity of oyster reefs in Mosquito Lagoon (Arnaldi et al. 2018).

Threats to oysters in northeast Florida

Climate change and sea-level rise: Sea-level rise and variable precipitation patterns may increase (or already be increasing) predation, disease, and harmful algal blooms in this region (Petes et al. 2012, Gobler et al. 2013, Garland and Kimbro 2015). An emerging threat to the increasingly submerged reefs in Mosquito Lagoon is boring sponge (*Cliona* spp.) infestations that result in weakening and loss of shell (Walters and Fang, unpublished data). Areas of oyster reefs with moderate exposure and submergence times grow fastest, while areas that are entirely submerged have lower rates of growth and accretion because of exposure to disease, predation, and sedimentation (Ridge et al. 2015). Thus, increased submergence as a result of sea-level rise may put these intertidal reefs at risk.

Boating impacts: Oyster reefs are susceptible to wave exposure and boat wakes, which can result in erosion or accumulation of dead shell above the high tide line in piles known locally as shell rakes. Boat wakes in the ICW of northeast Florida have eroded salt marshes and oyster reefs to the point that they are reduced to intertidal sand flats (Grizzle et al. 2002, Price 2005, Wall et al. 2005, Frazel 2009, Walters et al. 2017). An analysis of aerial photos from 65 km (40 mi) of channel in the southern GTMNERR revealed that 70 ha (170 ac) of shoreline habitat eroded from 1970 to 2002 (Price 2005). Reefs in Mosquito Lagoon also underwent erosion as a result of boat wakes (Garvis et al. 2015). The width of dead margins increased from 1940 until 2000, when they made up 9% of the aerial extent of oyster reefs in Mosquito Lagoon (Grizzle et al. 2002). Some reefs migrated as much as 50 m (165 ft) away from the ICW as a result of the erosion (Grizzle et al. 2002). While wave exposure and shifting inlets can also alter locations of erosion, exposure to boat wakes and rising sea level are of greatest concern (Frazel 2009). Campbell (2015) documented that boaters do the most damage on oyster reefs when traveling at intermediate speeds with maximal wakes.

Habitat loss: Oysters are directly susceptible to habitat loss as a result of coastal development, shoreline hardening, and dredge-and-fill operations. In an SJRWMD study, more than half of the 2,000 locations surveyed in developed areas of northeast Florida were on a hardened shoreline edge, particularly in residential areas (Ron Brockmeyer, pers. comm.). Thirty percent of these areas lacked a natural intertidal zone. Hardened shorelines not only interrupt the transition area from upland to benthic habitat, but reflected wave energy can also undermine potential adjacent oyster habitat. Coastal

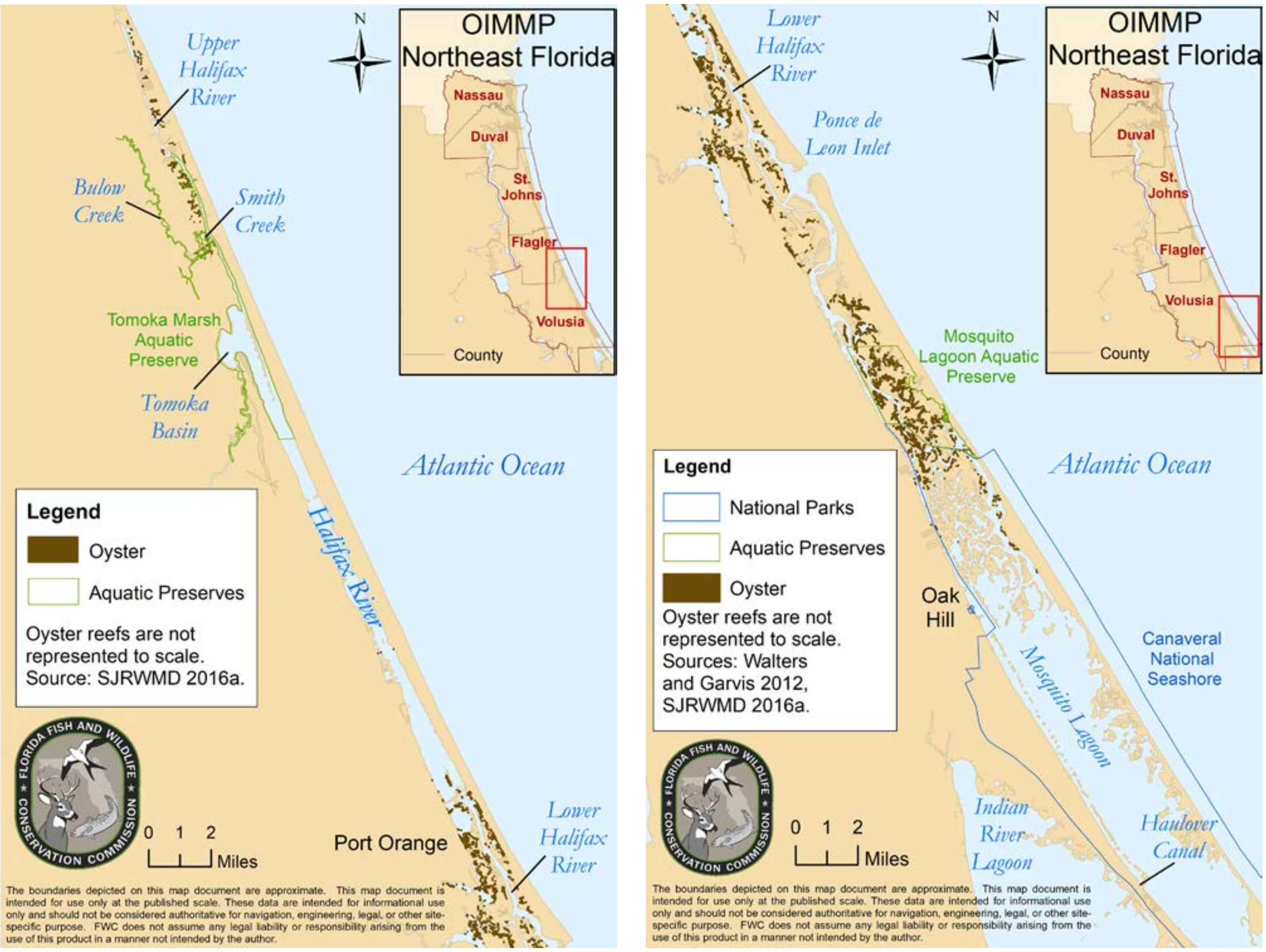

Figure 9.7. Oyster reef coverage in Flagler and Volusia counties. Oyster mapping sources: Walters and Garvis 2012 (made from 2006–2012 aerial photographs) and SJRWMD 2016a (from 2002–2014 photographs).

development also indirectly impacts oysters through reduced water quality, increased pollutants, and increased sedimentation (Frazel 2009).

Harvesting: Harvesting removes individuals from the population, removes available substrate for larval settlement, changes population size structure, and can widen the reef and reduce its height (Abbe 1988, Woods et al. 2005, Powell and Klinck 2007). Greater reef height reduces the effects of hypoxia and enhances current flow over oysters, both of which increase growth and survivorship rates (Rothschild et al. 1994, Lenihan and Peterson 1998). Loss of substrate due to overharvesting may further hinder the ability of reefs to keep up with sea-level rise (Rodriguez et al. 2014).

Diseases: In one study, the protozoan disease dermo (*Perkinsus marinus*) was common throughout the GTMNERR, but intensities were relatively low (0.75–1.16 on the scale of 0 to 5 developed by Mackin 1962) (Brandimarte et al. 2017). There is no evidence of elevated intensities or population-scale impacts due to either dermo or MSX (*Haplosporidium nelsoni*) in northeast Florida (Walters et al. 2007, Apeti et al. 2014). But because multiple stressors often interact and make oysters more susceptible to disease (Lenihan et al. 1999), monitoring of diseases and oyster condition should continue. In particular, warmer winters may allow disease to flourish year-round. The possible impacts of those conditions are poorly understood.

Invasive species: The Asian green mussel (*Perna viridis*), charru mussel (*Mytella charruana)*, and pink barnacle (*Megabalanus cocopoma*) are invasive species found in northeast Florida, where they compete with native oysters for habitat and food (Gilg et al. 2010, 2012, 2014; Yuan et al. 2016, Galimany et al. 2017). Charru mussels were first noticed on the east coast of Florida in Jacksonville in 1986; localized, long-term populations have been observed in several locations on the east coast (Boudreaux et al. 2006, FDEP 2009, Spinuzzi et al. 2013). Population genetics research suggests that all three species were transported via ballast water or ships' hulls (Gillis et al. 2009,

Figure 9.8. Field staff estimating percent cover along an oyster reef transect. Photo credit: GTMNERR.

Spinuzzi et al. 2013, Cohen et al. 2014, Calazans et al. 2017). The Asian green mussel has been found in greater numbers, particularly near Jacksonville (FDEP 2009).

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.

GTMNERR oyster monitoring

Pilot monitoring of intertidal oyster reefs within the GTMNERR and surrounding waters was initiated in 2014 (Fig. 9.8). The main objectives were to evaluate the status of oyster populations in the area; provide abundance and size estimates that would inform the quantification of ecosystem services provided by oysters; obtain baseline estimates of reef, population, and community structure metrics for future assessments; and evaluate methods for long-term monitoring. Oyster reefs were sampled in the winter (January–March) and summer (July–September) during 2014–2016 (summer only in 2014). A stratified-random design was used to sample 210 reefs in seven regions (Fig. 9.9).

In addition to reef sampling, larval settlement patterns were monitored following Fish and Wildlife Research Institute (FWRI) protocols (Parker 2015). Samples were collected using a spat tree, i.e., a T-shaped PVC device with cleaned oyster shell suspended on a wire from each end of the crossbar; rates of larval settlement were determined by counting the number of oyster spat on the bottom/inside of each interior shell (Fig. 9.10).

An average of 1,621 oysters per m^2 (150 oysters per ft^2) were observed. The proportion of spat-sized oysters (< 25 mm/1 in shell height) was higher in summer than winter, but there was no evidence of a seasonal difference for fishery-size oysters (≥ 76 mm/3 in shell height). Size-frequency distributions, an indicator of the age structure of a population (Baggett et al. 2014), were mostly skewed right with more abundant smaller individuals or size classes (Fig. 9.11). The decreasing abundance of larger individuals indicates higher mortality when they reach a subadult size; this pattern is common for intertidal oysters in the southeastern United States (Bahr and Lanier 1981, Coen and Luckenbach 2000, Volety and Savarese 2001). Understanding implications for the oyster fishery and long-term population sustainability will require estimation of growth and mortality rates and population modeling (Dame 2011). Primary mass settlement of new recruits (spat) occurred in the late spring–early summer in all regions during 2015–2016 (Fig. 9.12). Smaller settlement events occurred in each region throughout the summer and fall.

Reef height and slope measurements did not differ by season. Reefs were tallest and steepest in the northern regions and flattest in the most southern regions, similar to findings by Shirley et al. (2016). Reefs were also relatively flat in the Salt Run region, an easily accessible but relatively small oyster harvest area, resulting in relatively high harvest pressure. Local harvesting practice is to cull the reefs by hand, knocking off the small oysters and taking only fishery-size oysters. Salt Run reefs also had few clusters and lower oyster density but had one of the highest proportions of fishery-size oysters. The harvest activities that keep the reef profiles, clusters, and numbers low may also contribute to faster growth rates of oysters in this region.

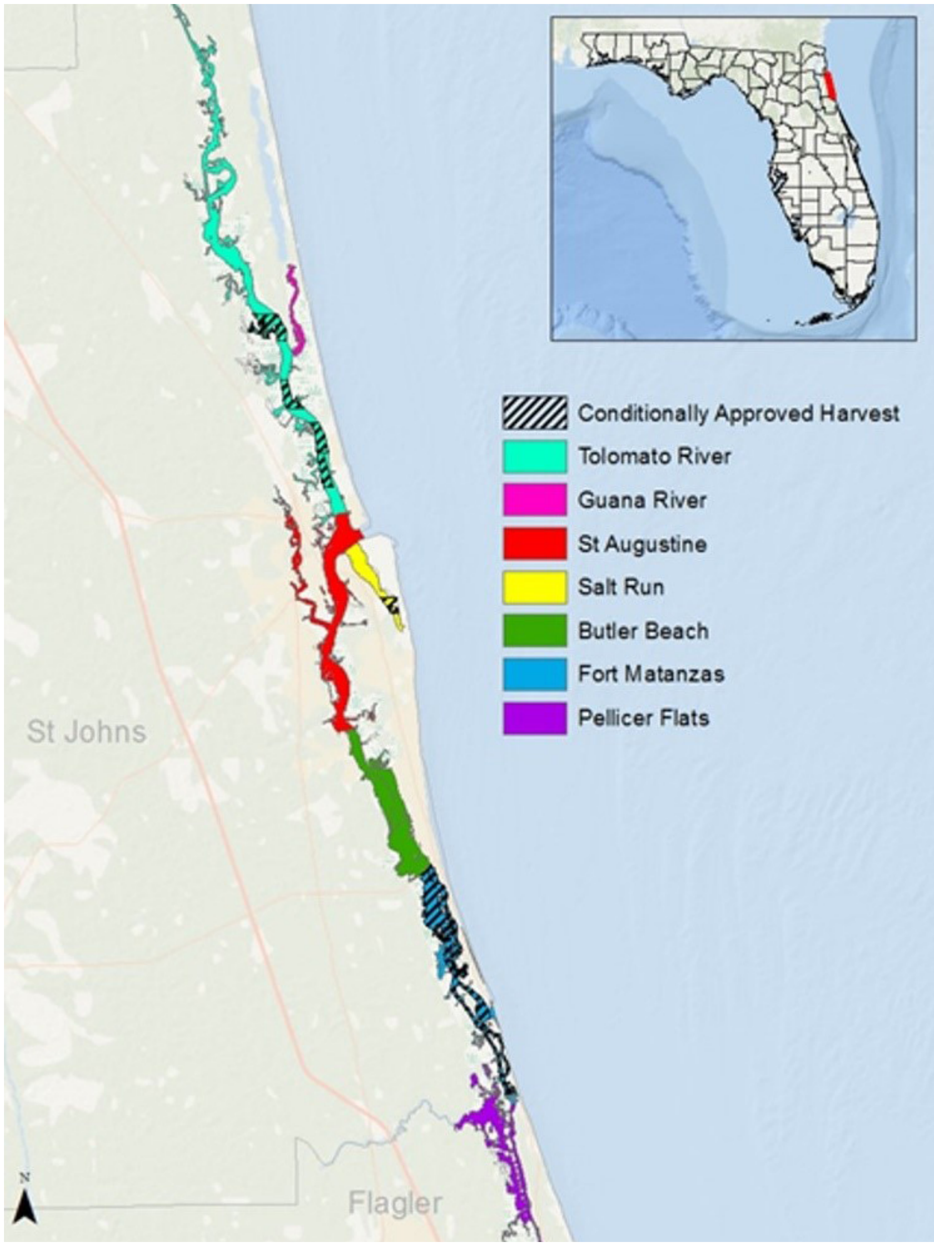

Figure 9.9. Map of oyster monitoring regions and conditionally approved shellfish harvest areas in the GTMNERR and surrounding waters.

Regional means in percent living cover were relatively similar (24–29%). Living oyster and shell cover were higher in winter; sediment cover was higher in the summer. Living oyster cover was positively correlated with densities of oysters, indicating that cover could be used to estimate oyster density (and relative quantities of the ecosystem services they provide). The ease and relative quickness of measuring percent cover would facilitate increased sample sizes and spatial coverage in a nondestructive manner.

Oyster density was correlated with densities of all associated fauna. The strongest relationship was with mussels ($R^2 = 0.69$). Associated fauna observed on oyster reefs throughout this study include annelids (*Polydora* spp.), quahog/hard clams (*Mercenaria campechiensis*), oyster drills (*Urosalpinx cinera*), white/striped barnacles (*Balanus amphitrite*), ribbed mussels (*Geukensia demissa*), mahogany date mussels (*Lithophaga bisculata*), crown conch, boring sponges, slipper snails (*Crepidula* spp.), porcelain crabs (*Petrolisthes armatus*), stone crabs (*Menippe mercenaria*), swimming crabs (*Callinectes* spp.), other xanthid crabs (Family Panopeidae), and hermit crabs.

Predatory crown conchs were found on reefs only in the Pellicer region, consistent with a study by Garland and Kimbro (2015) in the region. Mean crown conch density

Figure 9.10. Spat tree deployed on reef (left) and oyster spat settled on cleaned shell (right). Photo credits: GTMNERR.

was higher in this study (3.8 per m^2 vs. the 1.5 per m^2 found in Garland and Kimbro), but it is difficult to assess whether the difference is significant. The Pellicer region surrounds the mouth of a freshwater tributary (Pellicer Creek), and oyster growth rates tend to be lower in low salinity (Volety and Savarese 2001, Wang et al. 2008). Thus, the lack of large oysters in this region may also be a long-term consequence of freshwater discharge and associated factors.

NOAA Mussel Watch

The National Oceanic and Atmospheric Administration (NOAA) National Status and Trends Program has been monitoring pollutants in bivalves through the Mussel Watch program across the coastal United States since 1986. Monitoring locations on the northeast Florida coast included Chicopit Bay on the St. Johns River from 1989 to 2011 and Crescent Beach on the Matanzas River from 1989 to 2012. Oysters were monitored for concentrations of heavy metals and organics in each location. Polycyclic aromatic hydrocarbons (PAHs) in St. Johns River oysters were attributed to petroleum contamination associated with shipping and high boat traffic in the river (LSJRBR 2016). Medium to high concentrations of arsenic, mercury, nickel, and lead in St. Johns River oysters were reported based on data from 2004–2005 (Kimbrough et al. 2008). In the same report, medium to high concentrations of arsenic, mercury, nickel, and zinc were reported for Matanzas River oysters (Kimbrough et al. 2008). Oysters in the Matanzas River site had low copper concentrations, less than half the average concentration found in Florida oysters overall (Frazel 2009). Many oysters in Florida have high copper concentrations because copper is used in fungicides, algaecides, and antifouling paints (Kimbrough et al. 2008).

Lower St. Johns River Basin Report

The University of North Florida, Jacksonville University, and Valdosta [Georgia] State University complete an annual analysis of the health of the Lower St. Johns River Basin. Salinity has gradually increased in the St. Johns River Basin since the mid-1990s as a result of sea-level rise and decreased freshwater flow (LSJRBR 2016). Nutrient levels and chlorophyll-a levels remain high, but total nitrogen levels declined 31% from 1997 to 2015. Dilution of estuarine waters by low-nutrient ocean water and reduced freshwater flow may be contributing to the decrease in nutrients (LSJRBR 2016).

Northeast Florida oyster reef condition assessment

The SJRWMD, GTMNERR, UCF, and the Northeast Aquatic Preserves collaborated to develop an intertidal Oyster Condition Assessment (OCA) protocol built on their earlier research efforts but standardized for regional

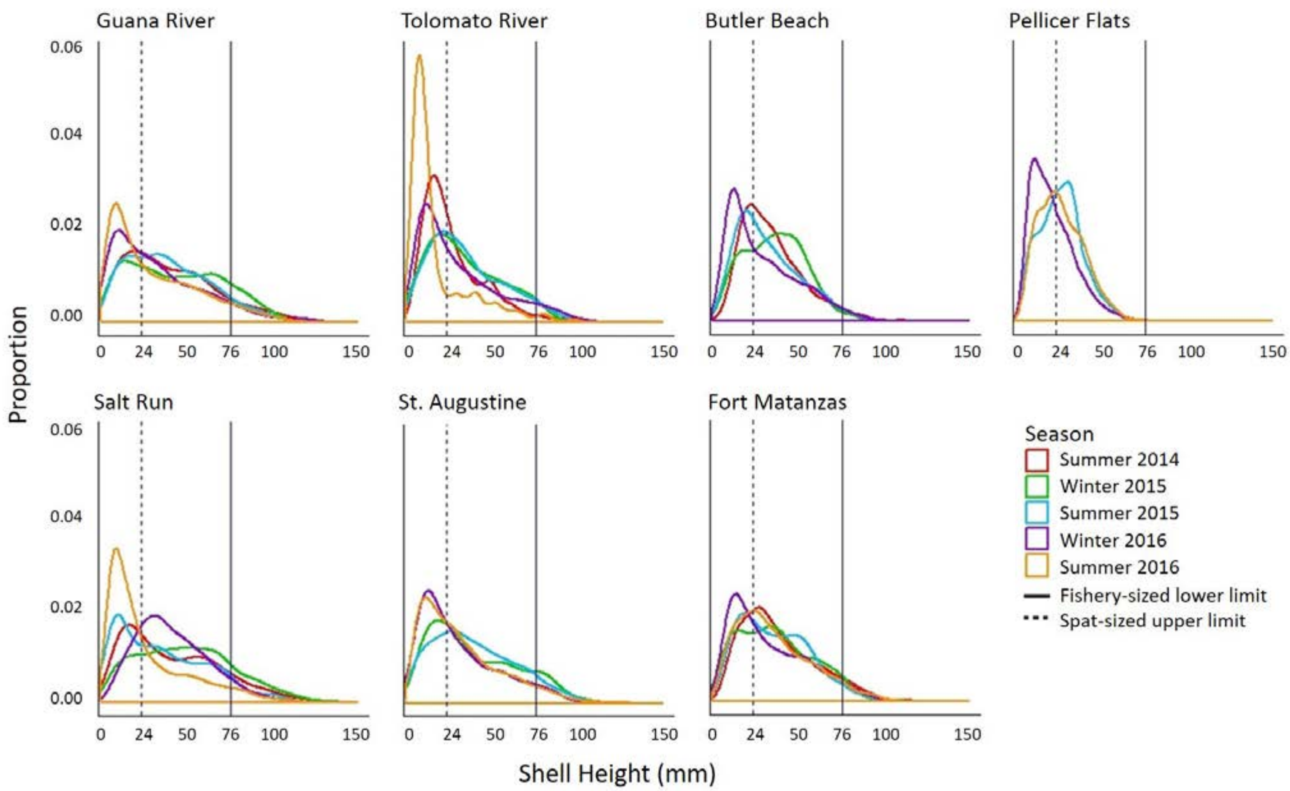

Figure 9.11. Oyster size class frequencies by region and season (summer or winter of 2014–2016).

application across geographic areas (Walters et al. 2016). With funding from SJRWMD and the Florida Coastal Management Program, partners applied the method in northeast Florida to test the repeatability and consistency of the method. The OCA sampling protocol captures universal metrics for monitoring and assessment of oyster habitat described by Baggett et al. (2014) but also complements research questions specific to the GTMNERR and UCF monitoring programs. Data will be used to assess the condition of the resource and provide baseline information describing northeast Florida estuarine ecosystems.

Metrics on oyster reef condition have been collected on more than 200 reefs in Nassau, Duval, St. Johns, Flagler, and Volusia counties during the summers of 2015–2017 and winter 2015–2016. The intertidal reefs are categorized as fringe, patch, or string and must be at least 5 m (16 ft) long per the monitoring protocol. Nested quadrat data are collected along a transect on the portion of the reef with highest oyster density (Fig. 9.8). Metrics include reef height, slope, and thickness; percent cover (living oysters, dead shell, or sediment); number of oyster clusters; oyster density; burial depth; and shell height. Total number and size of individuals of invasive, predatory, and commensal species are also recorded (Walters et al. 2016). The OCA protocol is available online at http://ocean.floridamarine.org/OIMMP/Resources/Walters%20et%20al%202016.pdf.

Mosquito Lagoon Aquatic Preserve oyster monitoring

The University of Central Florida has conducted annual monitoring of restored and natural oyster reefs in Mosquito Lagoon waters since 2008. Data collection includes density (on natural and restored reefs), shell length, type of cluster formation, presence of invasive species, amount of seagrass recruitment adjacent to reefs, and boat-strike frequency. Since 2014, monthly recruitment data have been collected on 10 reefs. Monitoring has overlapped with two brown tide events. In 2017–2019, additional data was collected on the impact of restoration on ecohydraulics, biogeochemistry, fisheries, invertebrates (including infauna), wading birds, and perceptions of volunteers involved with the project.

In 2016–2017, FWRI, along with Florida Atlantic University's Harbor Branch Oceanographic Institute and the FDEP Aquatic Preserves, sampled restored and natural reefs throughout the Indian River Lagoon, including six reefs within the Mosquito Lagoon Aquatic Preserve, to assess organismal health. Oysters sampled were evaluated for health indices such as gut condition, gonad development, and prevalence of disease.

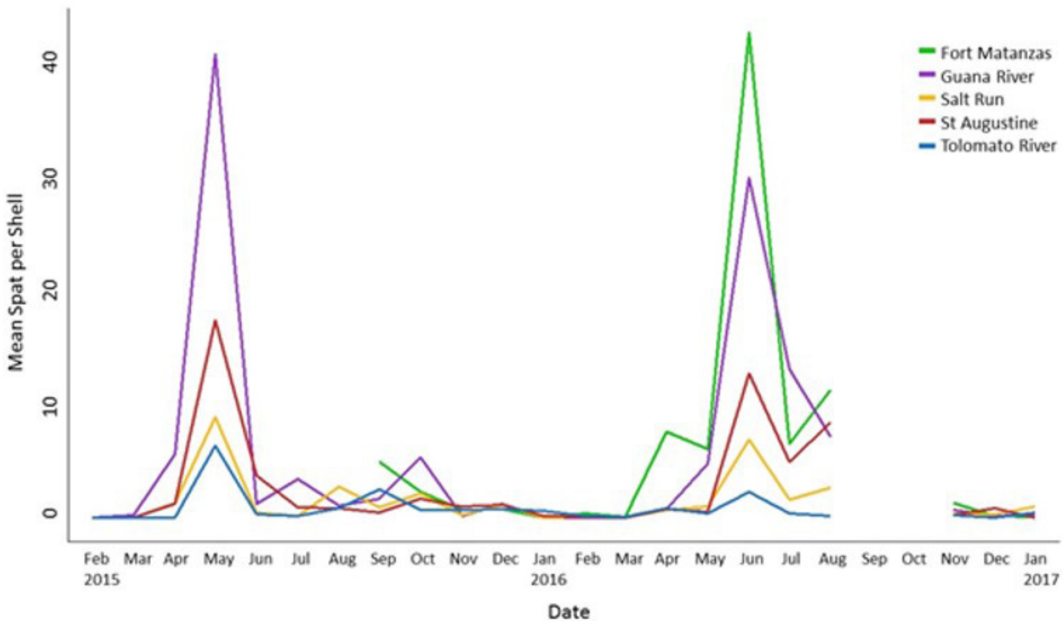

Figure 9.12. Spat settlement (mean number of spat per shell) by region.

Intertidal oyster mapping

Aerial photography was used to identify oyster reef signatures and map the distribution of intertidal oyster reefs throughout the northeast Florida region. ArcGIS software was used to delineate each reef perimeter. The goal was to create a continuous intertidal oyster reef habitat map (Walters and Garvis 2012, Walters et al. 2015, SJRWMD 2016a) which could serve as a baseline map of oyster distribution for future management and assessment. The mapping effort represents the first successful attempt at fine-scale oyster reef mapping across the entire northeast Florida region and resulted in the mapping of 17,953 reefs covering 650 ha (1,610 ac). Of these reefs, 6.1% were classified as dead, all of which were along important boating channels. Ground truthing found 98% accuracy for Mosquito Lagoon and 96% accuracy for the Northern Coastal Basins area.

Drone aerial oyster mapping in GTMNERR

A remote-imaging company, Prioria, was contracted to fly over a portion of Guana River in 2016 using a drone for a case study of mapping techniques. Oyster reefs were digitized in photos and compared with reefs mapped by the SJRWMD in 2008 and 2015. The 2008 oyster map was created from imagery collected by planes contracted specifically for the mapping effort. This 2008 map was the most accurate but it was also incomplete, likely because the survey had not been conducted at exact low tide or because the plane was flying at an altitude that did not allow reefs along the marsh edge to be resolved in the photographs. The 2015 map was created from 10 years of aerial imagery from a number of sources. That map estimated a similar area of oyster reef habitat to the drone-produced map, but the polygons were at a coarser scale than the drone-based polygons and the locations were not as accurate. These comparisons illustrate the challenges of mapping intertidal reefs, which are submerged for a significant portion of every day. While aerial imagery from planes can cover large geographic areas, tidal and atmospheric conditions can make the images difficult to interpret. On the other hand, drones can be flown in specific tide windows and give the most detailed coverage of reef area, but they can only cover small areas.

Recommendations for management, mapping, and monitoring

- Complete mapping and monitoring efforts that make note of unconsolidated substrate and dead margins (Fig. 9.13) to quantify migration or change in condition of oyster reefs (Grizzle et al. 2002, Price 2005, Frazel 2009, Garvis et al. 2015).
- Verify the presence or absence of subtidal reefs through dedicated nontraditional mapping efforts.
- Continue studies of species interactions (including predation, competition with invasive species, damage by boring sponges and algal blooms) and how they might be altered by a changing climate. Investigate effects of factors such as food limitation, nutrition, toxicity, and unpalatability for oysters in northeast Florida.

Figure 9.13. Researcher measuring the elevation profile on a reef with dead and living oysters in the Mosquito Lagoon. Photo credit: Linda Walters.

- Evaluate and monitor how harvesting impacts factors that influence reef resiliency (size structure, population, reef height, and accretion rate in the face of sea-level rise).
- Develop oyster population models to assist in predictions about long-term resource sustainability.

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http://www.gtmnerr.org/
https://floridadep.gov/fco/nerr-gtm
https://coast.noaa.gov/nerrs/reserves/gtm.html

Canaveral National Seashore general management plan: https://parkplanning.nps.gov/documentsList.cfm?parkID=360&projectID=13534

St. Johns River Water Management District: https://www.sjrwmd.com/

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Chapter 10
Conclusions and Recommendations

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Most of Florida's estuaries contain (or historically contained) significant populations of oysters. While the statewide extent of oyster reefs before European settlement is unknown, a marked decline in oyster extent has been documented in many areas that do have historical estimates. Tampa Bay and Charlotte Harbor have each lost 90% of their oyster reefs (Boswell et al. 2012, Kaufman 2017), Naples Bay has lost 80% (Schmid et al. 2006), Pensacola Bay has lost 72% (Lewis et al. 2016), Suwannee Sound has lost 66% of its reefs (Seavey et al. 2011), and Biscayne Bay has lost all its reefs (Meeder et al. 2001). A large portion of historical losses was due to activities that are now restricted by environmental regulations, such as dredge-and-fill construction and shell mining on live reefs. Altered hydrology and its associated stressors (especially salinity extremes) still threaten many reefs. Climate change and sea-level rise will increase the frequency and severity of high temperatures and salinity stress. Other, regional threats, such as erosion due to boat wakes and substrate loss from harvesting, also contribute strongly to losses. This combination of stressors makes Florida a challenging place for remaining oyster reefs and causes continuing degradation and habitat loss.

Multiple priorities and recommendations for the management, mapping, and monitoring of Florida's oyster reefs emerged during the writing of this report and as outcomes of the OIMMP workshops. Region-specific priorities and needs are identified in each chapter. Several priorities were frequently identified at the regional scale. These and additional statewide priorities are outlined below as key recommendations for the management, mapping, and monitoring of oyster habitats across the state.

Priorities and recommendations for ecosystem management of Florida's oyster reef habitats

- **Manage freshwater flow to mimic natural flow:** Many oyster reefs in Florida are stressed by either a lack or excess of freshwater flow. Variable freshwater inputs are largely due to surface water management efforts (e.g. South Florida) or limited river flow due to low precipitation and/or freshwater withdrawals (Apalachicola Bay and Suwannee Sound). Salinity is a primary factor for oyster survival and reproduction. Ensuring freshwater flow mimics natural flow helps to prevent rapid salinity changes and extreme salinity conditions; this is crucial to the survival of remaining oyster reefs.
- **Combat substrate limitation:** Many estuaries in Florida are substrate-limited due to extensive harvesting, shell mining, or dredging. Oyster restoration efforts that create new reef substrate or add shell to harvested reefs are key to maintaining oyster reef extent. Substrate placement should be based on present and predicted conditions, not principally on historic locations. New reef substrates should be placed on firm sediments to prevent their sinking. Although many materials can be used to create oyster substrates (Goelz 2017), shell recycling programs help replace the original substrate type removed during harvest and also engage the community through interaction with local businesses, school educational programs, and volunteer events. Limiting factors for substrate replenishment often include funding and materials. Areas like Apalachicola Bay need regular shell replenishment, yet the cost for purchase and distribution of this shell is often funded by grants, which are inher-

ently temporary. Long-term funding for shell replenishment is needed in areas that are heavily harvested and should perhaps be a requirement of harvesting. Efforts should also be made to reduce the use of plastic to hold together loose shell when creating artificial reefs due to the eventual degradation and release of microplastics into estuaries.

- **Create and implement a comprehensive fishery management plan:** Dynamic fishery-management strategies are needed to prevent overfishing or loss of substrate. These plans should incorporate changing climate, variable oyster fishing effort, annually variable rainfall, and widespread anthropogenic changes. Fishery management should consider maintaining positive shell budgets, oyster size structure (including large size classes), and the fluctuating hydrology of the watershed. Areas with high fishing pressure may also benefit from rotational harvest with fallow periods. These fallow periods allow for natural mortality on the reef and thereby production of natural shell, the preferred settlement substrate on a reef. The development of Territorial User Rights Fisheries (TURFs), which lease the rights of bottom areas to individual fishers, would incentivize care of the fishing resource. The growing oyster aquaculture industry is another means of reducing harvest pressure on wild oysters.
- **Replace or supplement hardened shorelines with living shorelines:** Living shorelines create habitat for oyster reefs and coastal wetlands and provide a gradual elevation change that facilitates the migration of these habitats upslope as sea level rises. Before new living shorelines or reefs are created, sites should be assessed for habitat suitability to ensure that they have appropriate environmental conditions for restoration success.
- **Maintain genetic connectivity of oyster populations:** Connectivity between oyster populations in multiple estuaries is important to maintaining genetic diversity, which is key to the survival of populations facing a variety of environmental stressors (Koehn et al. 1980a, Hilbish and Koehn 1987). Each estuary should ideally have established oyster reefs in both upstream and downstream locations to increase genetic exchange among local populations and maximize resiliency to local perturbations, stabilizing the regional metapopulation. Further study is also needed to elucidate the degree and temporal variability of existing genetic exchange across oyster populations.

Mapping priorities and recommendations

- **Fill remaining mapping gaps:** The FWC compilation used to create the maps in this report is the most comprehensive map of oyster reefs for Florida, but several gaps remain. Updated oyster mapping is needed for the Panhandle (Pensacola, Choctawhatchee, and St. Andrew bays), Big Bend and Springs Coast (Apalachee Bay and subtidal oysters), much of the Everglades, and the Indian River Lagoon (outside of its major tributaries).
- **Complete regular mapping:** Oyster extent is dynamic as a result of urban development, variability in salinity and temperature, and ongoing changes in freshwater management. Maps of oyster extent should be updated every 5–7 years. Some oyster maps in Florida are significantly out of date; for instance, parts of Apalachee Bay have not been mapped since 1992.
- **Map all types of oysters:** Intertidal oysters on hardened shorelines or on mangrove roots generally have not been mapped, as they are not easily identifiable from aerial imagery. Sarasota County is one of the few locations in Florida to have a focused oyster mapping effort for these peripheral habitats (Meaux et al. 2016). Oysters on mangrove roots and seawalls contribute a significant number of individuals to the breeding population in an estuary and provide many of the same ecosystem services as oyster reefs (Drexler et al. 2014). In more heavily developed estuaries (e.g., Biscayne Bay, Broward County), seawall and mangrove-root oysters may be the dominant form of oyster. Subtidal oyster reefs are also mapped infrequently or not at all, because it is so labor-intensive to map the benthos with sonar. Additional subtidal oyster mapping is needed across the panhandle, Big Bend, Tampa Bay, and possibly other locations where the extent of subtidal oysters is unknown.
- **Determine historical extent of oyster reefs:** Continue efforts to determine oyster distribution before European settlement using historical records and sedimentary coring techniques. In many regions of Florida, the historical (and sometimes current) extent of oyster reefs is unknown, which hinders decision making regarding targets for future reef extent.
- **Differentiate between live and dead extent on oyster reefs:** Mapping efforts vary as to whether they distinguish between live or dead oysters on a reef. Mapping should make note of dead reefs, unconsolidated substrate, and dead margins of shell on live reefs in order to track changes over time.

Monitoring and research priorities and recommendations

- **Conduct standardized and long-term monitoring:** Long-term monitoring conducted over a number of estuaries, such that conducted by FWC (Arnold et al. 2008, Parker et al. 2013), provides an invaluable resource for comparing the status and physiological tolerances of oyster populations across Florida. This type of standardized and regularly repeated monitoring program is recommended for all estuaries in Florida. While constant monitoring of all reefs in all estuaries may not be logistically feasible, a sample design that allows both regional and local monitoring at appropriate time and spatial scales would provide a better understanding of statewide oyster resources. Long-term monitoring is also needed to gauge the success and sustainability of oyster restoration efforts, which are frequently only monitored for a few years following installation.
- **Assess genetic diversity, life history, and habitat characteristics of high-salinity oyster reefs:** Several estuaries in Florida are home to significant intertidal populations of oysters that survive in environments with an average salinity range of 30–35 (Parker et al. 2013). These locations include lower Tampa Bay, Sarasota Bay, parts of the Ten Thousand Islands, the Mosquito Lagoon, and the southern Loxahatchee River. Oysters in these regions must have some combination of genetic aptitude towards survival at high salinity (Koehn et al. 1980b), adaptive life history traits, or only moderate amounts of parasitism and predation. The intertidal nature of these reefs does provide temporary relief from predation during exposure at low tide, but further study is needed on life history, genetics, and habitat characteristics to determine why certain oyster populations survive in high salinity while others are decimated by predators and disease.
- **Quantify size structure of oyster populations:** Measuring shell height in an oyster population can provide an easily measured indicator of reef resilience, as large oysters are disproportionately important to reproductive output and shell budgets (Waldbusser et al. 2013). Large oysters make a reef better able to cope with stressors such as salinity and thermal stress, overfishing, and sea-level rise.
- **Continuously sample abiotic parameters with autonomous instrumentation:** Frequent water sampling is needed to capture data on brief events including freshwater pulses or heat extremes. Oysters are vulnerable to rapid changes in salinity and temperature and are less resistant to environmental extremes when they occur simultaneously (Shumway 1996). Occasional snapshot water quality monitoring often does not capture these extreme events. Autonomous sampling also provides information regarding long-term trends and water quality variability within estuaries.

Conclusion

The Oyster Integrated Mapping and Monitoring Program will continue efforts to coordinate, facilitate collaboration toward, and address gaps in oyster mapping and monitoring in Florida. The information compiled in this report is meant not only to facilitate decision making for mapping and monitoring oyster reefs, but also to recommend priorities for the adaptive management of these unique coastal habitats and the numerous species that depend on them. Knowledge of the extent of, trends in, and threats to oyster reefs is crucial for the long-term management of these valuable habitats.

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Appendix A
Historical Yields of the Florida Oyster Fishery

The oyster fishery yield data reported in Table A-1 were compiled from published annual reports of Florida Commercial Marine Fish Landings. This data set is also available for download under the heading "1950 to 1983 oyster fisheries data" at http://ocean.floridamarine.org/OIMMP/. Florida Commercial Marine Fish Landings data from 1950 to 1963 were compiled by the U.S. Fish and Wildlife Service (USFWS), Florida State Board of Conservation, and University of Miami Institute of Marine Science. Data from 1964 to 1968 were compiled by the USFWS and the Florida State Board of Conservation. Data from 1969 to 1971 were compiled by the Florida Department of Natural Resources (FDNR) Division of Marine Resources and the USFWS. Data from 1972 to 1983 were compiled by the FDNR and the National Marine Fisheries Service. In 1984, the State assumed a mandatory trip ticket program for reporting commercial harvest yields. Data have been compiled by the Florida Fish and Wildlife Conservation Commission (FWC) and are available at http://myfwc.com/research/saltwater/fishstats/commercial-fisheries/landings-in-florida/.

All commercial oyster harvests are reported in Table A-1 as pounds of oyster meats. Oyster yields that were reported as gallons were converted to pounds of meats by a multiplier of 8.75. Oyster yields reported as bushels were converted to pounds of meats by a multiplier of 4.375 before 1984 and by a multiplier of 6.5625 from 1984 onward. Multipliers were updated in 1984 when the State assumed full responsibility for reporting commercial fishery yields. (Steve Brown, pers. comm.). Increased bushel size since the 1950s may have also contributed to the degree of change in this multiplier. The current conversion factors for commercial landings of oysters and other marine species may be found at https://myfwc.com/media/9085/sumfact.pdf. Note that counties in Table A-1 may alternate between reporting on their own and reporting jointly with neighboring counties.

Table A-1. Reported commercial yields of pounds of oyster meats harvested annually 1950–1983.

Year	Published east coast total	Published west coast total	Published statewide total	Bay, Gulf, and Washington	Bay and Gulf	Bay	Bay and Washington
1950†	22,715	872,553	895,248				
1951			735,304			42,368	
1952	20,907	542,080	562,987			30,304	
1953*	21,576	563,780	585,356		59,956		
1954	17,907	667,589	685,496		67,992		
1955	19,340	630,241	649,581	48,141			
1956	32,304	856,431	888,735		89,830		
1957	24,754	710,124	734,878		68,961		
1958	29,759	794,970	824,729		60,206		
1959	40,045	1,414,953	1,454,998		93,934		
1960*	44,644	1,930,756	1,975,400		89,458		
1961	72,542	3,254,059	3,326,601		127,367		
1962	67,091	4,952,680	5,019,771		259,664		
1963*‡	80,745	4,282,103	4,362,848		204,400		
1964	92,339	2,792,784	2,885,123				70,119
1965	166,013	2,788,732	2,954,745				79,998
1966	135,114	4,156,811	4,291,925				55,046
1967	182,772	4,578,358	4,761,130				61,475
1968	252,084	5,316,689	5,568,773				88,859
1969	241,320	4,911,422	5,152,742			85,337	
1970	212,725	3,573,794	3,786,519			70,856	
1971	181,988	3,528,554	3,710,542			58,078	
1972	126,404	3,230,967	3,357,371			80,249	
1973	122,389	2,408,936	2,531,325			106,743	
1974	97,724	2,653,661	2,751,385			70,436	
1975	79,413	2,133,652	2,213,065			28,473	
1976	111,781	2,602,669	2,714,450			28,160	
1977	125,896	4,071,688	4,197,584			64,634	
1978	93,651	5,880,212	5,973,863			76,220	
1979	87,349	6,124,910	6,206,259			58,245	
1980	97,379	6,755,931	6,853,310			107,115	
1981	99,055	7,170,329	7,269,384			122,901	
1982	81,752	4,816,936	4,898,688			169,946	
1983	77,959	4,326,494	4,404,453			7,964	

* Note that the county total does not add up to the published statewide total.

† No county data.

‡ County yields rounded to nearest 100 lbs.

Year	Brevard	Charlotte	Citrus	Citrus and Pasco	Collier	Broward and Dade	Dade
1950†							
1951	18		71,689				350
1952			44,411				
1953*			27,781				
1954			34,995				
1955			28,337				
1956	88		12,939				
1957	209		3,116				
1958	35		11,200				
1959		57	16,000				
1960*		891	26,119				
1961	18,477	26,431	31,393		102		
1962	6,759	54,412	29,184		123		
1963*‡	22,900	4,200	13,500		2,400		
1964	32,913	17,153	52,069		794		
1965	94,404	4,173	8,217		325		
1966	32,307	4,589	2,138				
1967	88,607	1,934	17,859				
1968	143,171	282		3,318			
1969	132,378		3,900				
1970	139,000			790			
1971	103,825						
1972	52,498						
1973	41,967						
1974	21,539						
1975	23,924						
1976	24,658			3,439			
1977	38,196			1,651			
1978	38,415			1,641			
1979	31,788			21,387		46	
1980	23,167			4,359			
1981	27,835			1,007			
1982	30,805			11,565			
1983	17,488			6,420			

Year	Dixie and Taylor	Dixie	Duval	Escambia and Santa Rosa	Escambia	Franklin	Gulf
1950†							
1951		8,278	17,386	3,535		546,560	26,504
1952		438	12,826		7,375	451,145	
1953*			15,236		11,514	459,225	
1954			14,717		4,288	553,946	
1955			17,282		2,756	542,874	
1956		2,602	28,009		2,967	722,046	
1957		267	15,769		2,202	624,222	
1958			22,524		2,147	713,230	
1959			21,781		8,371	1,268,757	
1960*			23,838		22,562	1,744,760	
1961	3,386		27,385		20,354	2,947,137	
1962	1,801		28,521		16,352	4,366,700	
1963*‡	1,400		29,700		6,600	3,810,500	
1964		4,450	28,612		16,531	2,252,377	28,936
1965	72		29,669		22,554	2,337,530	45,492
1966	1,838		38,352		34,467	3,809,941	76,483
1967	3,576		32,051		16,224	4,195,905	107,792
1968	21,340		37,843		16,038	4,825,668	112,034
1969	19,842		27,089		71,372	4,350,370	114,218
1970	4,000		19,746		126,520	3,044,401	82,014
1971					50,380	3,180,085	70,078
1972			16,184		9,253	2,980,543	33,443
1973		12,155	14,979		7,287	2,193,492	8,406
1974	7,347		19,027		36,536	2,453,995	7,664
1975	114		16,925		5,739	2,032,612	24,245
1976	2,668		23,814			2,503,441	39,612
1977	10,530		19,222			3,894,089	51,818
1978	12,038				12,198	5,566,464	87,175
1979	4,746		757			5,809,684	117,263
1980	6,444					6,409,909	90,516
1981	6,403				581	6,616,648	124,211
1982	100,378				35,093	4,153,182	105,873
1983	52,064				78,202	3,936,070	79,389

Year	Hills-borough	Indian River	Lee	Levy	Manatee	Monroe	Nassau	Okaloosa
1950†								
1951	481			665		438	2,130	35
1952						140	1,425	
1953*		3,894		153			2,087	
1954				394			1,583	
1955			1,689				1,677	
1956			18,603	401			1,487	297
1957			4,451	437			2,154	1,937
1958			1,458				1,764	
1959			14,869	1,637			905	26
1960*			24,511	3,691			1,793	2,460
1961	679	204	18,277	23,509			1,098	11,009
1962	31,639	40	12,131	25,350	65		1,232	
1963*‡	14,500		35,000	30,200			400	
1964	44,662	7,374	47,284	37,727	17		418	
1965	26,126	7,191	23,827	50,587			552	
1966			7,813	87,674			386	
1967		7,603	12,128	82,139				
1968		1,635	31,168	159,921				
1969		17,919	27,521	165,163				
1970		9,446	32,733	61,514				
1971		3,466	25,433	59,993			17,299	
1972		940	2,926	63,896			1,575	1,361
1973		4,815	1,890	30,390			2,185	3,121
1974		4,938		22,899			2,875	578
1975		955		12,179			2,684	1,761
1976		996		16,189				
1977				23,867				
1978			1,044	44,269			2,190	
1979				47,611			175	2,990
1980				75,908				4,652
1981				149,454				
1982			246	141,202				110
1983	1,224		382	87,132				44

Year	Okaloosa and Walton	Palm Beach	Pinellas	Santa Rosa	Sarasota	St. Johns and Flagler	St. Johns	St. Lucie
1950†								
1951			805			5,023		
1952			4,341				6,208	
1953*			1,531				3,789	
1954			3,949				1,344	
1955			5,306					
1956			2,722					
1957							2,801	
1958			26				4,286	
1959			1,795				9,438	
1960*			1,979				9,782	
1961			7,890				7,256	480
1962	21,441		80,620				5,660	132
1963*‡	3,000		83,400				7,200	
1964	6,168		102,825				4,916	
1965			68,653	12,207	27,639		16,717	
1966			3,220				39,813	125
1967			1,652	62,172	405		40,144	
1968			2,756	34,166			52,248	
1969			377	49,335			46,338	
1970				126,035			30,880	
1971				62,370			34,467	
1972				23,688			43,570	
1973		5,177		25,659			45,791	
1974			50	27,862			47,429	
1975				13,699			34,635	
1976				7,543			62,163	
1977				900			67,708	
1978				656			50,363	
1979							45,063	
1980			13,731				65,179	2,730
1981			14,351				50,756	
1982			132				39,520	
1983			1,715				35,967	

Year	Santa Rosa	Sarasota	Taylor	Volusia	Wakulla and Leon	Wakulla	Walton
1950†							
1951		1,426		3,588	4,025		
1952		1,258		448		2,668	
1953*		2,622		263		998	
1954		1,587		263		438	
1955		440	260	381		438	
1956		1,270	226	2,720		2,528	
1957		2,627		3,821		1,904	
1958		210		1,150		6,493	
1959		1,018		7,921		8,489	
1960*		131		9,231		14,114	
1961		12,473		17,642		24,052	
1962		13,115		24,747		40,083	
1963*‡		10,800		20,500		46,600	
1964		40,743		18,106		70,929	
1965				17,480		72,555	8,777
1966	44,049	3,763		24,131		21,127	4,663
1967				14,367		8,463	6,634
1968				17,187		13,886	7,253
1969				17,596		10,500	13,487
1970				13,653		12,723	12,208
1971				22,931		15,752	6,385
1972				11,637		19,242	16,366
1973				7,475		7,656	12,137
1974				1,916		18,514	7,780
1975				290		10,711	4,119
1976				150		1,345	272
1977				770		23,875	324
1978				2,683		76,736	1,771
1979				3,520		62,984	
1980				6,303		43,218	79
1981				20,464		134,773	
1982	11,922			11,427		47,173	40,114
1983	40,777			24,504		17,162	17,949

Appendix B

Abbreviations

Abbreviation	Meaning
ACF	Apalachicola–Chattahoochee–Flint
ANERR	Apalachicola National Estuarine Research Reserve
ASMFC	Atlantic States Marine Fisheries Commission
CBA	Choctawhatchee Bay Alliance
C-CAP	Coastal Change Analysis Program
CCMP	Comprehensive Conservation and Management Plan
CERP	Comprehensive Everglades Restoration Plan
cfs	cubic feet per second
CHIMMP	Coastal Habitat Integrated Mapping and Monitoring Program
CHNEP	Charlotte Harbor National Estuary Program
CI	condition index
CMECS	Coastal and Marine Ecological Classification Standard
CPAP	Central Panhandle Aquatic Preserves
CPUE	catch per unit effort
CSF	Conservancy of Southwest Florida
DERM	Department of Environmental Resources Management
dGPS	differential global positioning system
DOQQ	digital orthophoto quarter quadrangle
DWH	Deepwater Horizon
EBAP	Estero Bay Aquatic Preserve
ESA	Environmental Science Associates
ESI	Environmental Sensitivity Index
FAU	Florida Atlantic University
FDACS	Florida Department of Agriculture and Consumer Services
FDEP	Florida Department of Environmental Protection
FDOT	Florida Department of Transportation
FGCU	Florida Gulf Coast University
FGDC	Federal Geographic Data Committee

Abbreviation	Meaning
FIU	Florida International University
FLUCCS	Florida Land Use and Cover Classification System
FNAI	Florida Natural Areas Inventory
FOS	Florida Oceanographic Society
FWC	Florida Fish and Wildlife Conservation Commission
FWRI	Fish and Wildlife Research Institute
GIS	geographic information system
GPS	global positioning system
GTMNERR	Guana Tolomato Matanzas National Estuarine Research Reserve
HAB	harmful algal bloom
ICW	Intracoastal Waterway
IFAS	Institute of Food and Agricultural Sciences (University of Florida)
IRL	Indian River Lagoon
IRLAP	Indian River Lagoon Aquatic Preserve
LRD	Loxahatchee River District
LSJRBR	Lower St. Johns River Basin Report
LULC	Land Use/Land Cover
LWL	Lake Worth Lagoon
LWLI	Lake Worth Lagoon Initiative
MarineGEO	Marine Global Earth Observatory
MML	Mote Marine Laboratory
NASEM	The National Academies of Sciences, Engineering, and Medicine
NERR	National Estuarine Research Reserve
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NRDA	Natural Resource Damage Assessment
NWFWMD	Northwest Florida Water Management District
NWI	National Wetlands Inventory
NWR	National Wildlife Refuge
OCA	Oyster Condition Assessment
OIMMP	Oyster Integrated Mapping and Monitoring Program
OYSTER	Offer Your Shell to Enhance Restoration
PAH	polycyclic aromatic hydrocarbons
PBC	Palm Beach County
PCB	polychlorinated biphenyls
PCR	polymerase chain reaction

Abbreviation	Meaning
RBNERR	Rookery Bay National Estuarine Research Reserve
RECOVER	REstoration, COordination, and VERification
RPI	Research Planning Inc.
RSM	restoration suitability model
RTK	real-time kinematic
SAV	submerged aquatic vegetation
SBEP	Sarasota Bay Estuary Program
SCCF	Sanibel Captiva Conservation Foundation
SCDNR	South Carolina Department of Natural Resources
SCHEME	System for Classification of Habitats in Estuarine and Marine Environments
SFWMD	South Florida Water Management District
SGCN	species of greatest conservation need
SIMM	Seagrass Integrated Mapping and Monitoring Program
SJRWMD	St. Johns River Water Management District
SLE	St. Lucie Estuary
SMARRT	Seafood Management Assistance Resource and Recovery Team
SRWMD	Suwannee River Water Management District
SWFWMD	Southwest Florida Water Management District
TBEP	Tampa Bay Estuary Program
TBT	tributyltin
TBW	Tampa Bay Watch
TMDL	total maximum daily load
TMON	Tennenbaum Marine Observatories Network
TNC	The Nature Conservancy
TURF	Territorial User Rights Fishery
UCF	University of Central Florida
UF	University of Florida
UNH	University of New Hampshire
USACE	U.S. Army Corps of Engineers
USDA	U.S. Department of Agriculture
USEPA	U.S. Environmental Protection Agency
USF	University of South Florida
USFWS	U.S. Fish and Wildlife Service
USGS	U.S. Geological Survey
USNPS	U.S. National Park Service
WMD	water management district

Appendix C

Species List

Scientific name	Common name
Aureoumbra lagunensis	microscopic alga; causes brown tide
Balanus amphitrite	striped barnacle
Batillaria minima	West Indian false cerith
Bonamia spp.	parasitic rhizarians
Callinectes spp.	swimming crabs
Cladium jamaicense	sawgrass
Cliona celata	boring sponge
Crassostrea rhizophorae	mangrove oyster
Crassostrea virginica	eastern oyster
Crepidula spp.	slipper snails
Dendostrea frons	frond oyster
Fasciolaria lilium	banded tulip snail
Geukensia demissa	ribbed mussels
Haematopus palliates	American Oystercatcher
Halodule wrightii	shoalgrass
Haplosporidium nelsoni	protist parasite; causes the oyster disease MSX
Hyotissa hyotis	giant foam oyster
Hyotissa mcgintyi	Atlantic foam oyster
Isognomon alatus	flat tree oyster
Isognomon bicolor	bicolor purse-oyster
Isognomon radiatus	radial purse-oyster
Karenia brevis	microscopic alga; causes red tide
Limnodromus griseus	Short-billed Dowitcher
Limosa fedoa	Marbled Godwit
Lithophaga bisculata	mahogany date mussel
Lysmata wurdemanni	peppermint shrimp
Malaclemys terrapin	diamond-backed terrapin
Malleus candeanus	Caribbean hammer oyster
Megabalanus coccopoma	pink barnacle
Melongena corona	crown conch
Menippe mercenaria	stone crab
Mercenaria campechiensis	quahog/hard clam

Scientific name	Common name
Mytella charruana	charru mussel
Neopycnodonte cochlear	deepwater foam oyster
Ostrea permollis	sponge oyster
Ostrea stentina	crested oyster
Panopeus herbstii	Atlantic mud crab
Perkinsus marinus	protist parasite; causes the oyster disease dermo
Perna viridis	Asian green mussel
Petrolisthes armatus	green porcelain crab
Pinctada imbricata	Atlantic pearly oyster
Pinctada longisquamosa	scaly pearly oyster
Pinctada margaritifera	black-lipped pearly oyster
Pogonias cromis	black drum
Polydora spp.	genus of polychaetes
Pristis pectinata	smalltooth sawfish
Pteria colymbus	Atlantic wing oyster
Pteria hirundo	glassy wing oyster
Pyrodinium bahamense	microscopic alga; causes paralytic shellfish poisoning
Rhizophora mangle	red mangrove
Sinistrofulgur sinistrum	lightning whelk
Spartina alterniflora	smooth cordgrass
Stramonita haemastoma	southern oyster drill
Syringodium filiforme	manateegrass
Teskeyostrea weberi	threaded oyster
Thalassia testudinum	turtlegrass
Urosalpinx cinerea	Atlantic oyster drill
Vibrio parahaemolyticus	pathogenic bacterium occasionally found in oysters