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.

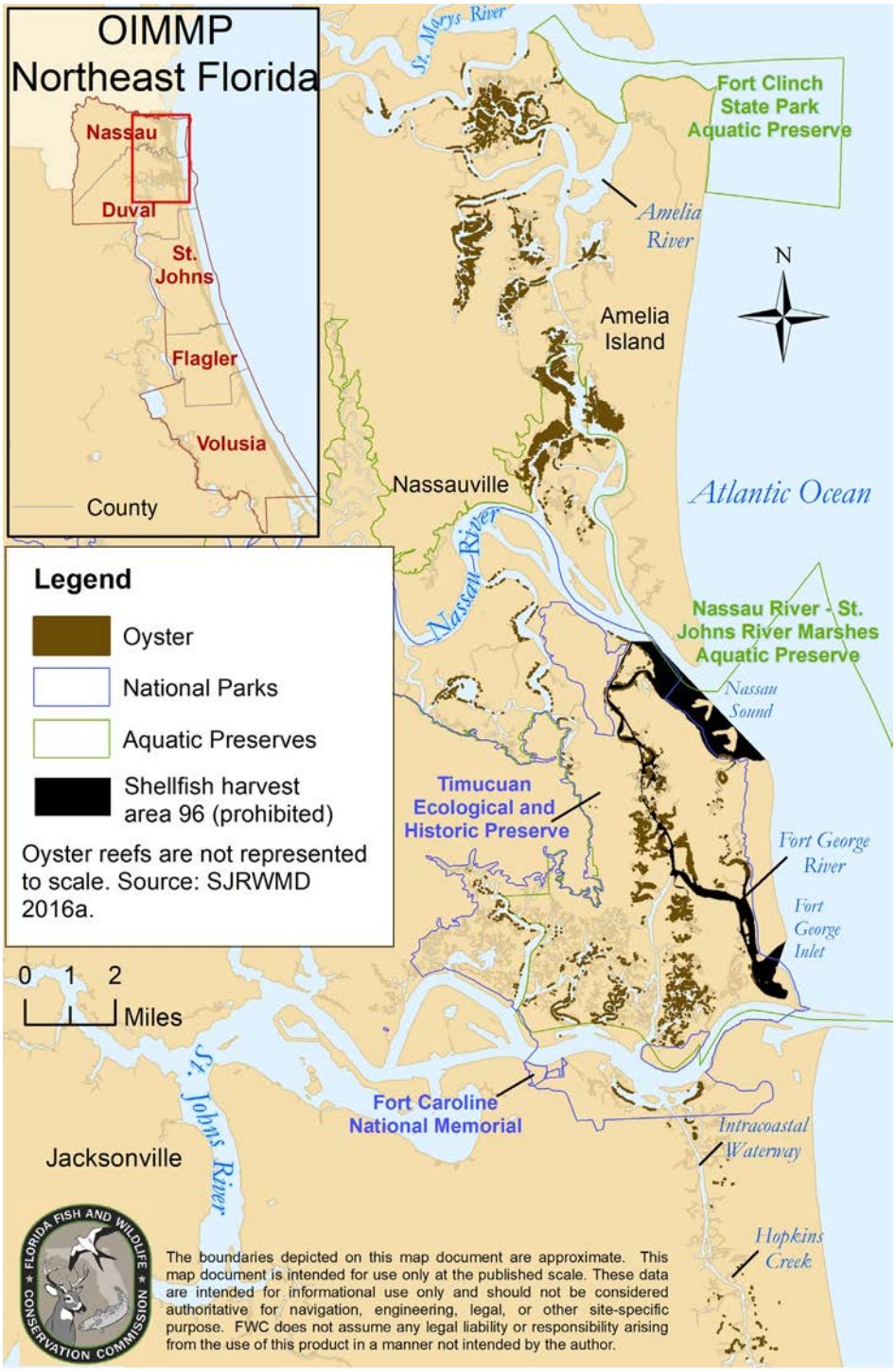

Figure 9.2. Oyster reef coverage in Nassau and Duval counties. Oyster mapping source: SJRWMD 2016a (made from 2002–2014 aerial photography).

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-

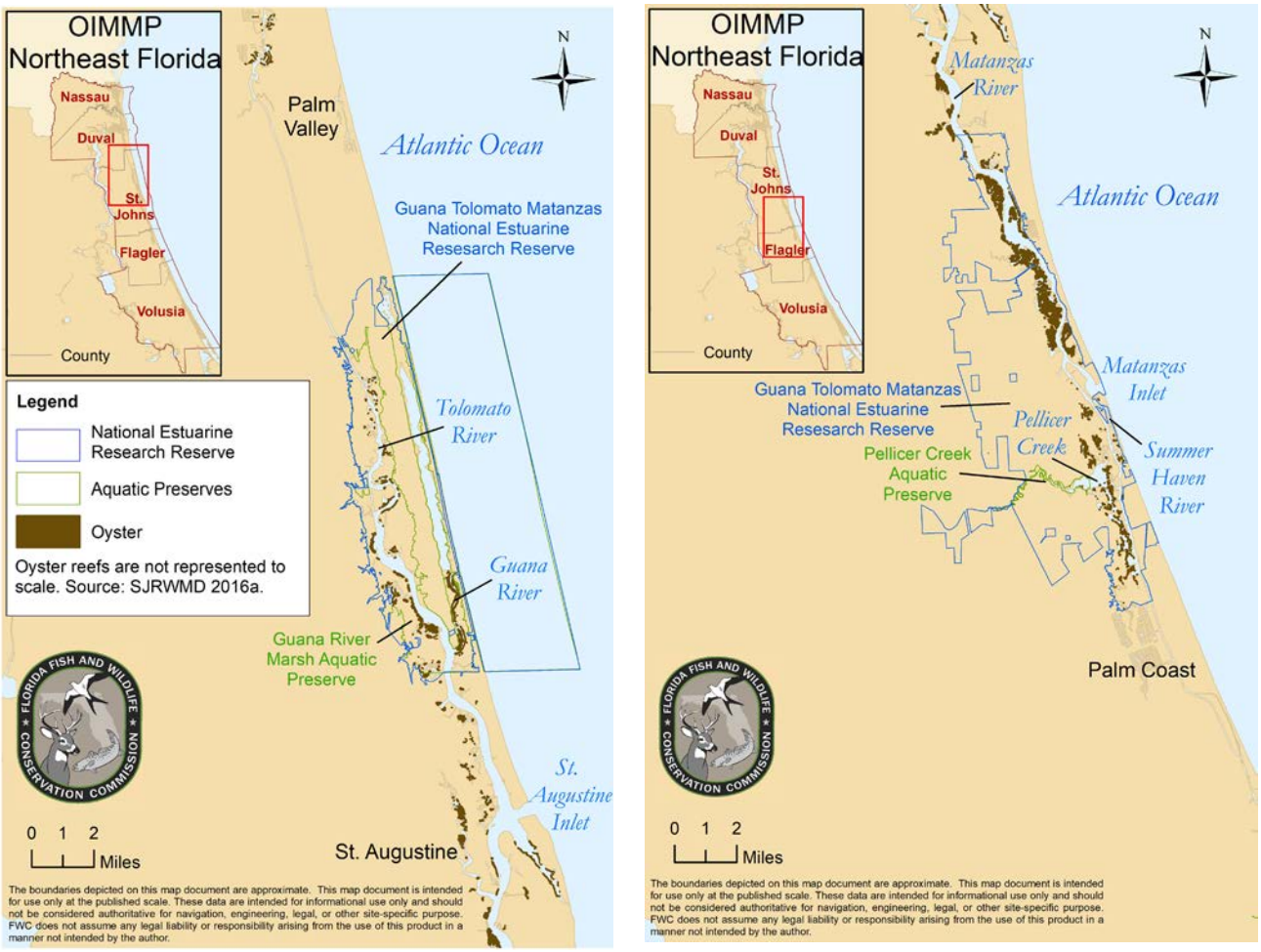

Figure 9.3. Oyster reef coverage in St. Johns and Flagler counties. Oyster mapping source: SJRWMD 2016a (made from 2002–2014 aerial photography).

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-

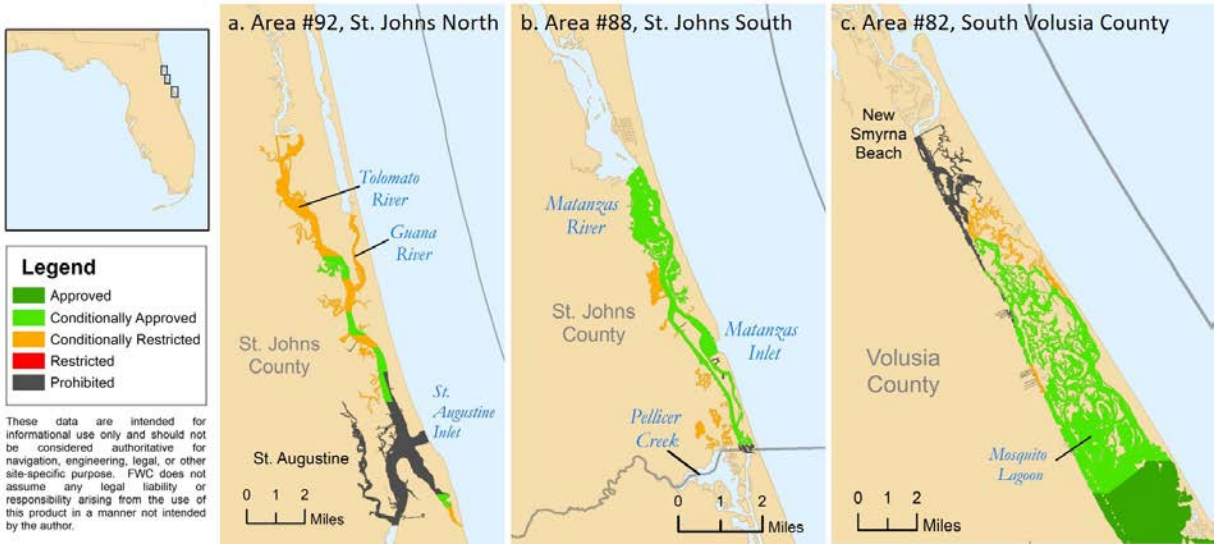

Figure 9.5. Shellfish harvesting areas on the northeast coast of Florida (Data source: FDACS 2017).

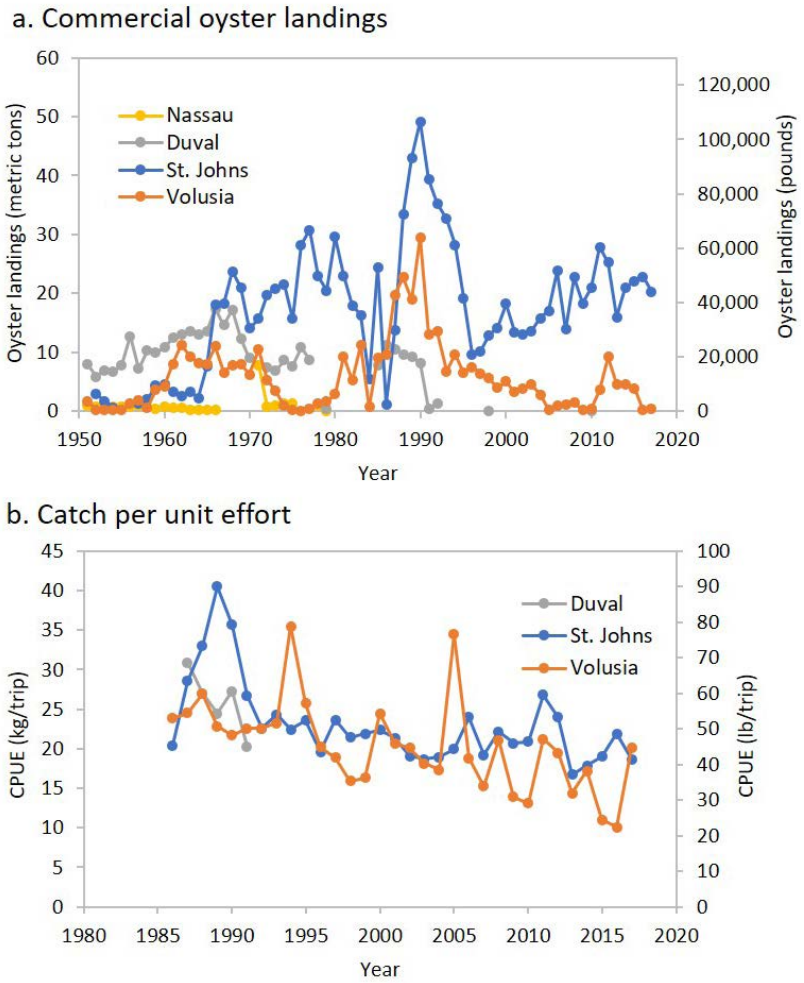

Figure 9.6. Commercial oyster landings (a) and catch per unit effort (CPUE; b) of commercial oyster landings in northeast Florida counties (Data source: FWC 2018 and Florida Commercial Marine Fish Landings, see Appendix A). Oyster landings before 1986 were collected under a voluntary reporting system.

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

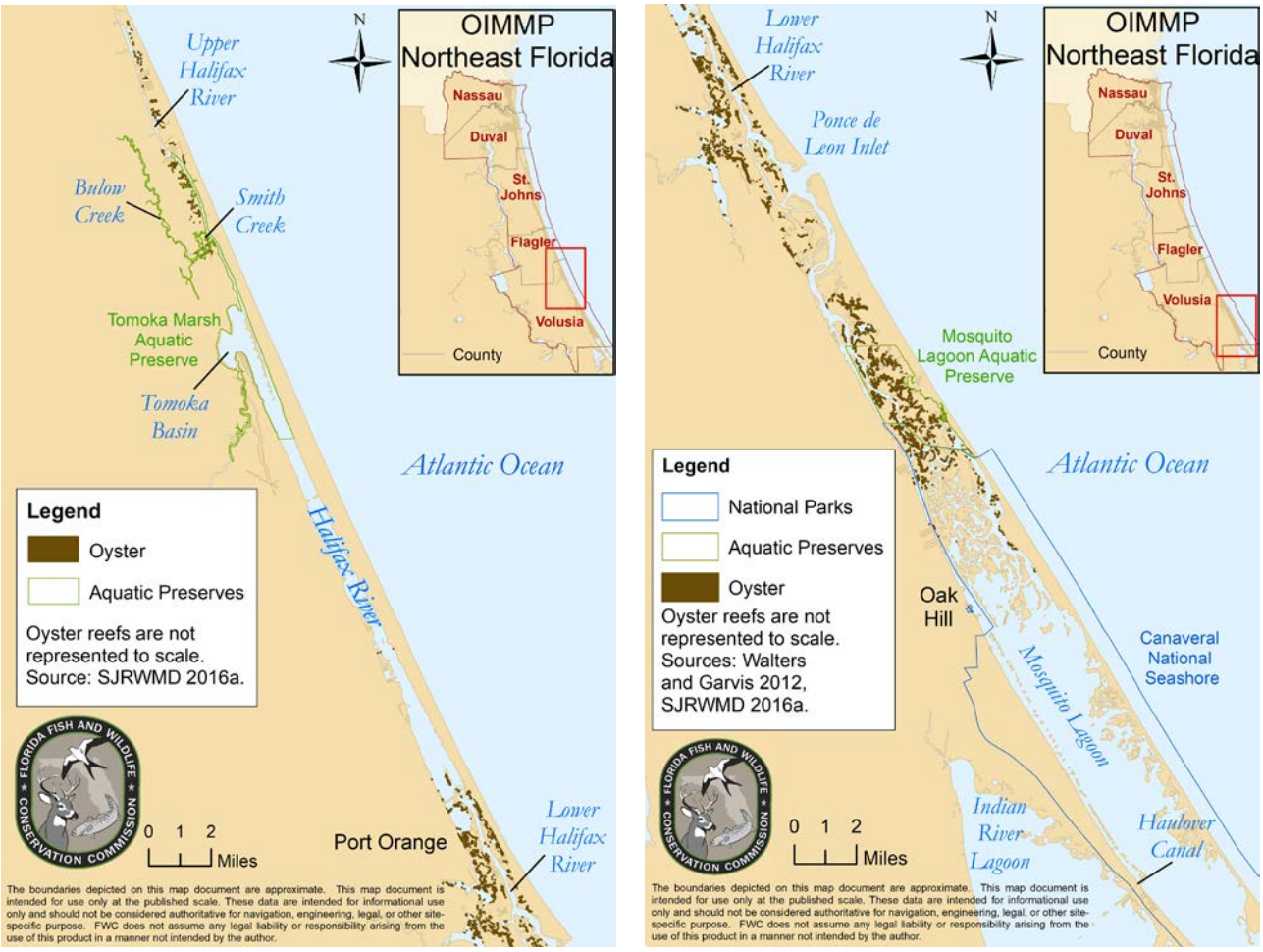

Figure 9.7. Oyster reef coverage in Flagler and Volusia counties. Oyster mapping sources: Walters and Garvis 2012 (made from 2006–2012 aerial photographs) and SJRWMD 2016a (from 2002–2014 photographs).

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.

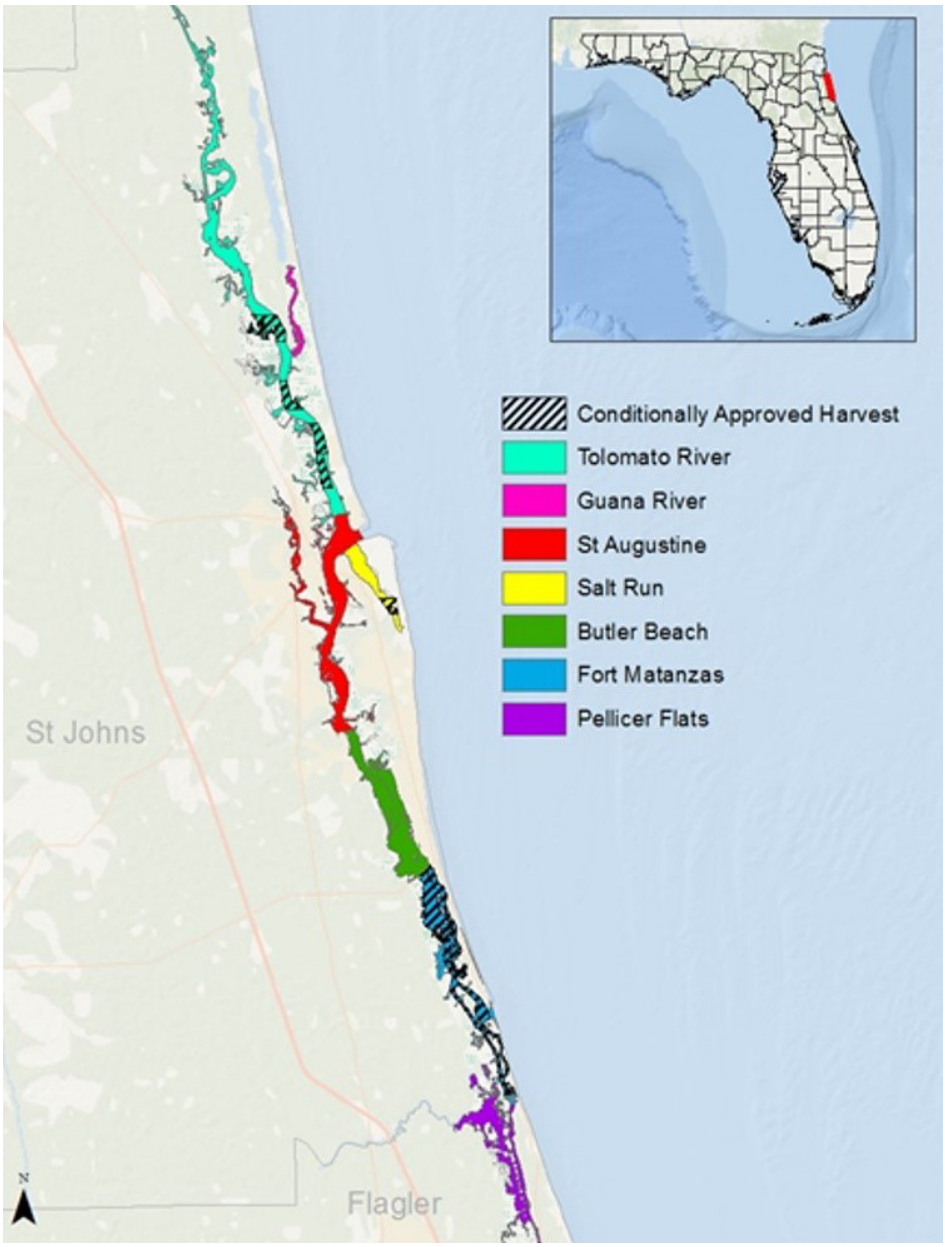

Figure 9.9. Map of oyster monitoring regions and conditionally approved shellfish harvest areas in the GTMNERR and surrounding waters.

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

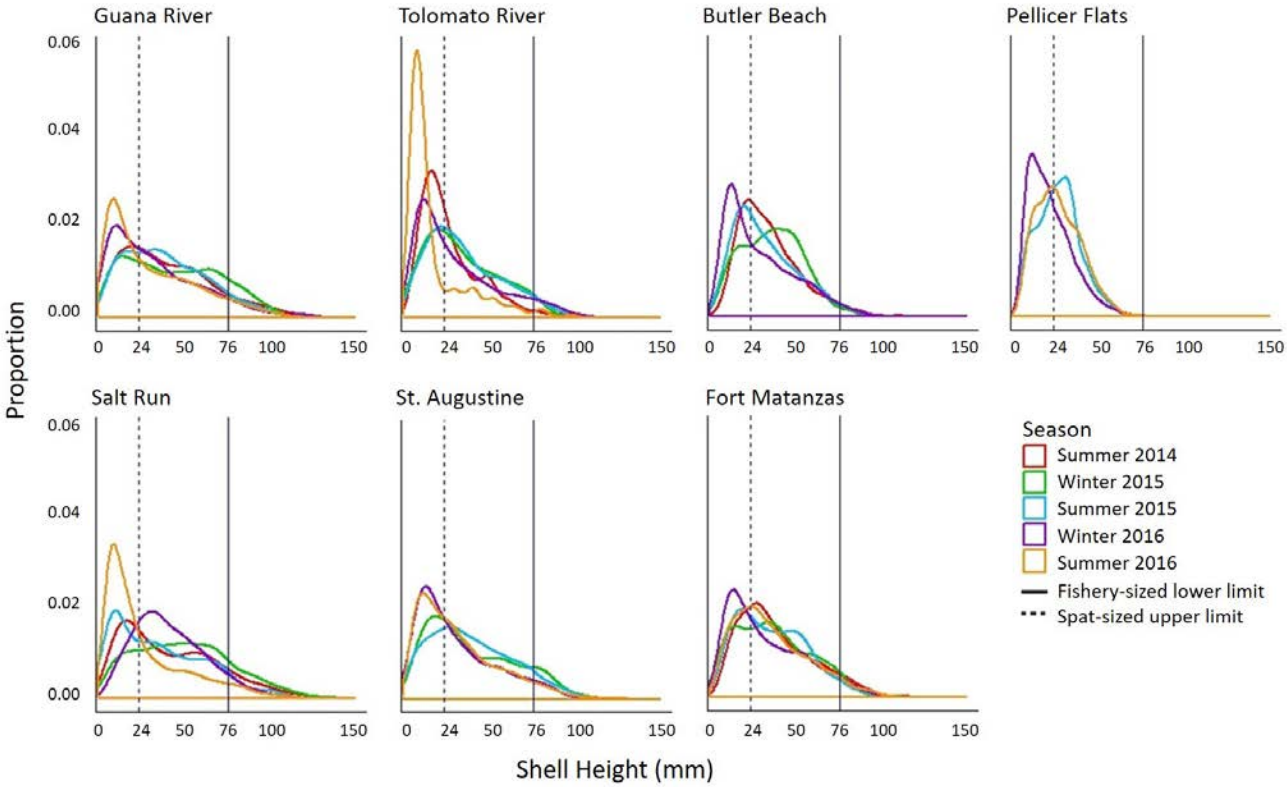

Figure 9.11. Oyster size class frequencies by region and season (summer or winter of 2014–2016).

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.

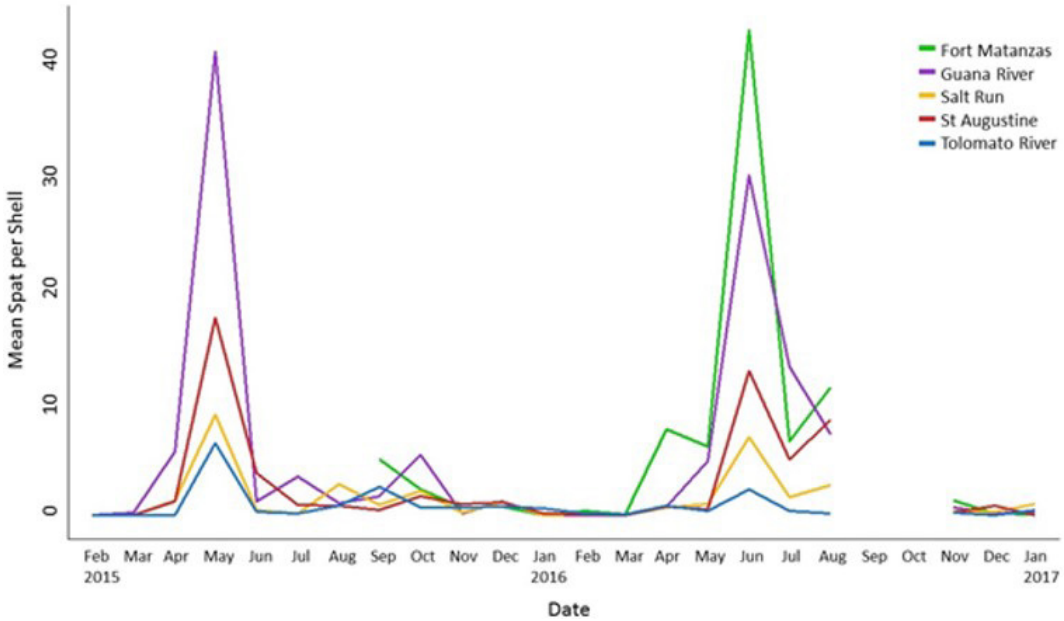

Figure 9.12. Spat settlement (mean number of spat per shell) by region.

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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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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