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FORMATION, MOVEMENT, AND RESTORATION OF DEAD INTERTIDAL OYSTER REEFS IN CANAVERAL NATIONAL SEASHORE AND MOSQUITO LAGOON, FLORIDA

STEPHANIE K. GARVIS,* PAUL E. SACKS AND LINDA J. WALTERS

Department of Biology, University of Central Florida, 4000 Central Florida Boulevard, Orlando, FL 32816-2386

ABSTRACT Globally, 85% of shellfish reefs have been lost during the past century. The severe loss of the eastern oyster C*rassostrea virginica* has encouraged different types of restoration efforts in the United States. In Mosquito Lagoon (ML), a shallow-water estuary on the east coast of central Florida, restoration focuses on providing additional substrate for larval recruitment via deployment of stabilized oyster shell. To assess the current number and area of natural, dead, and restored oyster reefs within ML, aerial photographs from 2009 were digitized using ArcGIS software. All reefs were screen digitized using a reef "signature" to estimate the surface area of each reef type. The maps from 2009 were then used as a guide to digitizing the historical aerial photographs (1943, 1951, 1967, 1971, 1984, 1995, and 2006). Oyster habitat within ML has decreased by almost 15 hectares between 1943 and 2009, which constitutes 24% of the 1943 lagoon-wide coverage. The impacts were greater in Canaveral National Seashore, which covers the southern ML; 40% of the oyster coverage within the park has been lost since 1943. Dead reefs were found adjacent to important boating channels. Tracked dead reefs exhibited a continuous migration into the mangrove islands located landward of the original live reefs, with some dead reefs completely washing up into the marsh. Restoration of dead reefs with stabilized oyster shells has added nearly 1 hectare of live oyster habitat to ML as of January 2009. This research demonstrates that dead reefs are increasing in number and coverage within ML, but this trend can be reversed with restoration.

KEY WORDS: *Crassostrea virginica*, eastern oyster, estuary, aerial photography, remote sensing, geospatial analysis, National Park Service

INTRODUCTION

The eastern oyster *Crassostrea virginica* (Gmelin, 1791) has been classified as an ecosystem engineer as well as a keystone species because of the important benefits that oysters and oyster reefs provide to estuarine systems (Dame 1996). Because of their three-dimensional structure, oyster reefs are able to maintain high levels of biodiversity (Coen & Luckenbach 2000). Many economically and ecologically important species use oyster reefs as habitat, including shrimp, stone crabs, blue crabs, and spotted sea trout (Coen & Grizzle 2007, Barber et al. 2010, Stunz et al. 2010). Research has demonstrated that oysters can have ecosystem-wide, positive effects on water quality and clarity (Newell 1988, Newell & Koch 2004). Oysters also act as soft armor and wave attenuators for shoreline stabilization (Scyphers et al. 2011, Manis et al. 2015).

It has been estimated that 85% of shellfish reefs have been lost worldwide (Beck et al. 2011). Global oyster coverage has decreased for several reasons: habitat destruction, disease, overharvesting, and reduced water quality (Wenner et al. 1996, Coen et al. 1999, Kirby 2004, Johnson et al. 2009). The severe loss of *Crassostrea virginica* has encouraged restoration efforts throughout its native range, which includes the eastern and Gulf coasts of North America. Restoration focuses on providing substrate for larval settlement and, if warranted, increasing larval abundances by introducing additional larvae or spat, either wild or cultivated. The simplest form of restoration is dumping the new substrate material directly on the seafloor (Kennedy et al. 2011). Another technique is the use of mesh bags filled with oyster shells, which can result in reefs more similar to the three-dimensional structure of natural reefs (Ertel & McCall 2005, Kennedy et al. 2011).

In Mosquito Lagoon (ML), boat wakes are a primary cause for oyster reef degradation (Grizzle et al. 2002, Wall et al. 2005). Water motion from boat wakes has caused live clusters and oyster shells to wash up on top of one another, pushing the oyster clusters above the mean high water (Grizzle et al. 2002, Wall et al. 2005). These clusters die due to lack of inundation, resulting in the formation of a dead reef (Grizzle et al. 2002, Wall et al. 2005, Stiner &Walters 2008).

A technique using stabilized oyster shell was developed to restore oyster reefs in the presence of large, repetitive boat wakes. Oyster restoration mats consist of aquaculture grade mesh squares that measure 0.25 m^2. Each mat has 36 disarticulated oyster shells, which each has a hole drilled in near the umbo, and are then firmly attached using zip ties so that the shells are perpendicular to the mesh. The mats are then deployed in a quilt-like fashion on top of a dead reef, which has been leveled to low intertidal height. Oyster restoration mats are held in place with cement weights. Restored oyster reefs have been shown to have equal densities of live oysters when compared with natural oyster reefs after 3.5 y (Birch & Walters 2012).

In 2002, Grizzle et al. conducted a study of the historical changes in intertidal oyster reefs in selected areas of Canaveral National Seashore (CANA) using year 2000 imagery. This represents southern ML. This study aims to expand on the Grizzle et al's. study to include the entirety of ML and incorporate recent restoration efforts that have occurred since 2007. The goals of this project are to (1) assess the change in oyster habitat coverage over time (1943 to 2009), (2) document the formation of dead reefs, and (3) quantify the amount of new oyster habitat created by restoration.

MATERIALS AND METHODS

Study Site

The study site was ML, which is the northern-most estuary in the Indian River Lagoon (IRL) system. The IRL system is a group of estuaries that comprises ~250 km of Florida Atlantic

*Corresponding author. E-mail: sgarvis@knights.ucf.edu
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coast [St. Johns River Water Management District (SJRWMD) 2006] (Fig. 1). The IRL is one of the most biologically diverse estuaries in the world, in part because it lies in a transitional area between temperate and subtropical climate zones (Dybas 2002). The majority of ML is a complex system of shallow, open water areas with almost 100 mangrove dominated islands (Walters et al. 2001). Mosquito Lagoon is microtidal, with a mean water depth of 1.7 m, and a tidal range of less than 1 m at Ponce de Leon inlet down to less than 1 cm at Haulover Canal (Smith 1993, Hall et al. 2001, Steward et al. 2006). The most influential water movement in the majority of this microtidal system is a direct result of wind-driven currents; however, some locations are more influenced by tidal currents (Dubbleday 1975, Hansell 2012, Smith 1987). Geographic information system analysis encompassed the entirety of ML, from Ponce de Leon Inlet in New Smyrna Beach in the north, to the Haulover Canal and Kennedy Space Center complex in the south.

Preliminary Field Checking

Pre-photointerpretation fieldwork was conducted in the summer of 2010 to ensure accurate photointerpretation. This preliminary field checking was focused on advanced signature identification prior to beginning digitization and was completed to identify and classify reef signatures appearing on the 2009 aerial photographs. Prior to field work, check sites were randomly selected to represent the assumed oyster signatures present within the project area.

Photointerpretation Process

To assess the aerial coverage of natural, restored, and dead oyster reefs within ML, imagery from 2009 was screen digitized using ArcGIS (v9.3) software. The three reef types were digitized separately to estimate the aerial coverage of each reef type. Each reef type had a distinct "signature" that was used to identify individual oyster reefs on the aerial photograph. After digitization was completed, historical aerial photography (1943 to 2006) was screen digitized using the 2009 map as a reference. Aerial photographs were predominantly free of glare and had high enough contrast to detect oyster reefs. Also, the aerial photography was captured during ideal conditions for oyster habitat study (i.e., winter at low tides). Issues with aerial

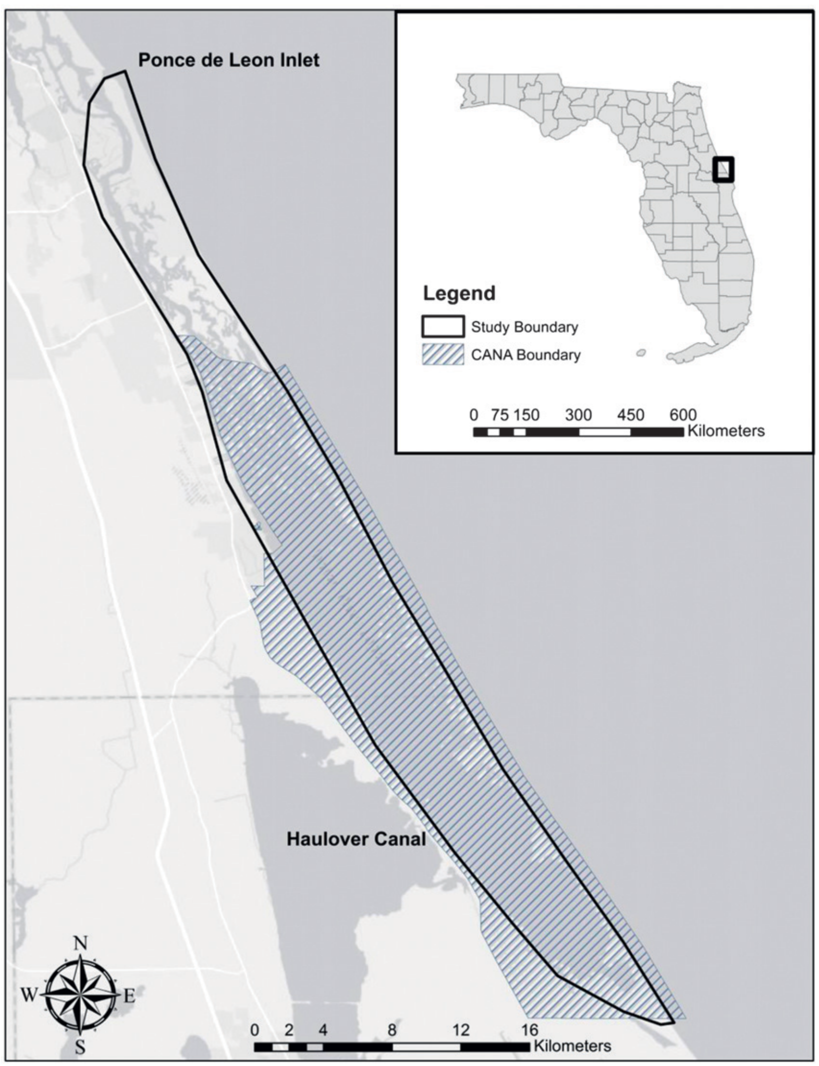

Figure 1. Map of the extent of the study area and CANA park boundary within ML, Florida. The inset shows the location of ML with respect to Florida.

photography such as glare or dark areas were resolved by interpolating reef positions between the previous and following years, a method also used by Grizzle et al. (2002). Narrow, fringing oyster reefs that were obscured by mangroves were not included in this study. Scattered live clusters were also not included. Only dense clusters of live oysters were included in the natural reef class. Aerial photographs used in this study were obtained from SJRWMD (2006) and Volusia County geographic information system (GIS) Department (Table 1). All oyster habitats were mapped to the level of detection (no minimum mapping unit).

Ground Truthing

A set of quality control points was independently created by a GIS Analyst at SJRWMD to avoid bias in the selection. The objective was to generate sample points for each class (oysters, nonoysters) using a spatially balanced method. A 1 × 1 km grid was generated for the study area (mapped oyster coverage) resulting in 60 grid cells across the lagoon. Random points were generated within the grid cells, across the final oyster layer (2009). A second set of random points was generated across a nonoyster layer consisting of areas that were likely to be oyster habitat. A minimum distance of 200 m was enforced between points. Ground truthing was conducted by both University of Central Florida and SJRWMD staff. A simple kappa coefficient was calculated to assess the agreement of the field data with the mapping classification.

Tracking of Individual Oyster Reefs

Dead margins and reefs in ML have narrow, distinct peaks created by the bleached, disarticulated oyster shells (Walters et al. 2007, Stiner & Walters 2008). Vertical reef profiles were used to track the movement of dead margins/reefs over time by tracking the location of the peaks relative to fixed reference points. To determine the highest point on each reef, vertical reef heights were measured at increments of 20 cm along a tape measure, using a stadia rod, and Johnson 9100/40–0909 laser level secured to a tripod. By measuring compass directions from permanent stakes, the same transects were able to be measured in multiple years. Reef height data collection began in 2006 and was tracked through 2013.

RESULTS

During preliminary field checking in the summer of 2010, unique reef signatures were developed for the three reef types: natural, dead, and restored (Fig. 2). Natural oyster reefs were identified based on the following criteria: globular or irregular in shape, with dark margins around the edge of the reef, and a lighter color in the center of the reef. Dead reefs were identified based on the following criteria: bright white reflection due to bleached, disarticulated shells that were continuously exposed, even at high tide. Dead reefs were found either adjacent to existing natural oyster reefs or standing alone, typically located on main boating channels (Fig. 3). Locations of all restored reefs were known before digitizing. Restored reefs were identified based on the following criteria: a more rectangular shape compared with the globular shape of the natural reefs.

After reef signatures were developed, aerial photography of ML in 2009 was digitized to identify different reef types (Fig. 3). A total of 2,542 natural reefs were identified (Table 2). Natural reefs ranged in size from <10 to over 5,000 m^2 and composed a total of 46.34 hectares of bottom habitat cover (Table 2). Five hundred and twenty four of the 2,542 reefs (20.6%) recorded in 2009 were located within CANA boundaries (Table 3). Natural reefs exhibited all three reef morphologies: patch reefs that are small and compact, fringing reefs that are found parallel to the shoreline (excluding those obscured by mangroves), and string reefs that are perpendicular to the shoreline (Kennedy & Sanford 1999). In 2009, 247 dead reefs or dead margins were identified in ML. These dead reefs ranged in size from <5 to over 1,300 m^2. Natural reefs with dead margins on the seaward edges were classified as two reefs with different reef types. Dead margins comprised anywhere from 10% to 99% of the original natural reef areal coverage, with 100% oyster loss being classified as a dead reef. As of the winter of 2009, 19 dead margins were restored to live oyster habitat. These 19 reefs totaled almost half an acre of bottom habitat in the lagoon. The restored reefs ranged in size from 10 to 317 m^2. All of these reefs were located within CANA boundaries. After the 2009 map was complete, extensive ground truthing occurred to ensure accuracy of reef signatures. The ground truthing was completed in the spring of 2011. Within the oyster class quality control points, there was 98% accuracy. Within the nonoyster class points, there was 100% accuracy. This resulted in a simple kappa coefficient of 0.9684, which indicates almost perfect agreement (Viera & Garrett 2005).

Over the 66-y study period, there was a steady decrease in natural reef coverage in ML, especially within CANA boundaries (Tables 2 and 3; Fig. 4). Losses in natural reef cover between 1943 and 1951 consisted mainly of reefs shrinking, potentially due to harvesting pressure or the impacts of dredging in the area. In 1943, four of the five dead reefs were located on the Atlantic Intracoastal Waterway, and one dead reef was located on the main channel in which the CANA boat ramp is located. By 1951, some dead margins had appeared north of Government Cut. By 1967, the number of dead reefs had nearly doubled (10–18) from 1951. This trend continued into 1971 and 1984, with dead reefs appearing on Shipyard Canal and additional boating channels. The largest increase in dead reefs was between 1984 and 1995 (Table 2). In 2006, dead reefs covered 3.61 hectares in bottom habitat (Table 2). In 2009, the areal coverage of dead reefs increased whereas the total number decreased, which is in part due to the converging of

TABLE 1.

Source information for aerial imagery used in study.

Year	Type	Source
1943	Black and white prints	St. John's River Water Management District
1951	Black and white prints	Volusia County
1967	Black and white prints	Volusia County
1971	Black and white prints	Volusia County
1984	National High Altitude Photography	St. John's River Water Management District
1995	Color IR prints	St. John's River Water Management District
2006	Digital	St. John's River Water Management District
2009	Digital	St. John's River Water Management District

Figure 2. Three reef signatures used to identify reef types during photointerpretation. Natural reefs are outlined in green (left), dead reefs are outlined in red (center), and restored reefs are outlined in blue (right).

several smaller dead reefs into one large dead reef. Dead reefs also displayed a shoreward migration compared with historical reef footprints (Fig. 5).

Because of restoration focused within CANA boundaries, there was a decrease in number and coverage of dead reefs within park boundaries from 2006 to 2009 (Table 3). Also, there was a small increase in natural reef acreage (<1 hectare) behind and near restored reefs within CANA (Table 3). Despite this gain within the park, the oyster habitat within all of ML decreased by almost 15 hectares, which constitutes 24% of the original coverage, with loss of 40% found in CANA boundaries.

The movement of 33 dead reefs was tracked for a minimum of 1 y and a maximum of 7 y (2006 to 2013) by comparing the

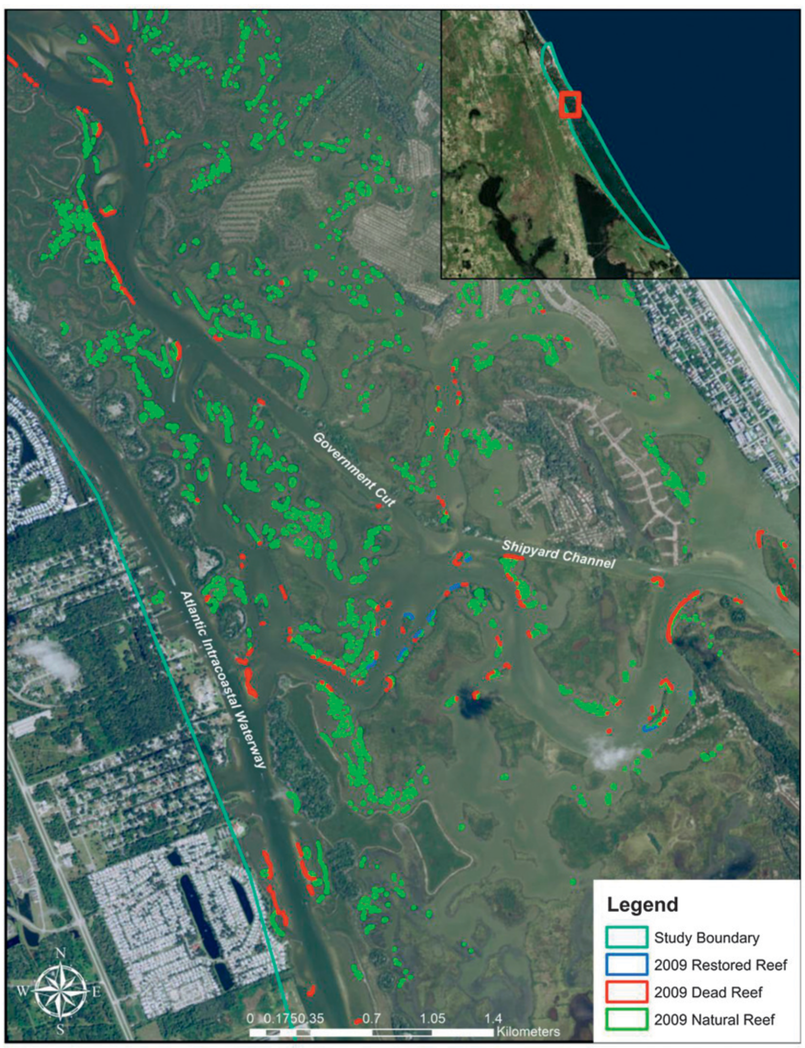

Figure 3. A selected portion of ML oyster reef coverage in 2009 (enlarged for visibility). Green represents natural reefs, red represents dead reefs, and blue represents restored reefs as of 2009. The inset shows the location of the selected portion of the study area (red) and the full extent of the study area (teal).

TABLE 2.

Oyster coverage within ML as determined by GIS. Percent coverage is relative to 1943 coverage.

Year	No. of Natural reefs	Hectares of natural reefs	No. of dead margins/reefs	Hectares of dead reefs	% Loss of natural coverage
1943	2,722	61.07	5	0.32	0
1951	2,716	57.46	10	0.75	6
1967	2,715	54.91	18	0.99	10
1971	2,705	54.17	32	1.06	11
1984	2,685	51.82	93	2.33	15
1995	2,609	48.28	230	3.12	21
2006	2,530	46.04	269	3.61	24
2009	2,542	46.34	247	4.30	24

highest points on the dead reefs. The primary difficulty was relocating the metal stakes that were covered by oyster shell washing into the marsh at some sites. Mean (±SE) dead reef movement toward shore was 0.86 ± 0.14 m/y. The range was 0.06–3.79 m/y. The largest movement occurred on a reef just north of the CANA north district boat ramp.

DISCUSSION

Many studies have shown large losses in oyster reef coverage compared with historical levels. Beck et al. (2011) created four categories of reef condition based on historical oyster abundance indicators or changes in reef coverage as measured by GIS: good = less than 50% lost; fair = 50%–89% lost; poor = 90%–99% lost, and functionally extinct = more than 99% lost. Of all the regions they evaluated, they found the majority of oyster reefs to be in poor condition (Beck et al. 2011). Few of the surveyed bays and estuaries were classified as good condition (i.e., >50% remaining) (Beck et al. 2011). Many ecoregions have been deemed functionally extinct (i.e., <1% remaining), especially in Europe, Australia, and North America (Beck et al. 2011). In Florida, the overall condition of oyster reefs is fair. For example, in the Big Bend region of Florida, researchers found a decrease of 123 hectares of oyster habitat between 1982 and 2010, which represented a 66% loss of the original oyster habitat, or fair reef condition (Seavey et al. 2011). Seavey et al. (2011) found these losses to be ecologically significant because the oyster reefs were 2,800–4,000 y old (Grinnell 1972, Wright et al. 2005). Thus, an important change has occurred over the last 30–40 y to cause such large losses in reef habitat. The researchers attributed losses in the Big Bend region to changes in freshwater inputs, which affected oyster spat recruitment and survival (Seavey et al. 2011). This study found that reef condition in ML is good, with losses of 24% within the entire lagoon and losses of 40% within CANA compared with 1943 levels.

Most of the oyster reef losses have occurred along major boating channels, suggesting that human impacts played a key role in these losses (Grizzle et al. 2002). Grizzle et al. also found little to no change in size or location over their 57-y study period on oyster reefs that were protected from boating channels via mangrove islands. Data from this study shows that dead reef formation does not perfectly mirror natural reef loss, which suggests that core reef area is also being lost. Figure 5 illustrates that in addition to changes in dead reef position, there is also compaction of dead reef area and loss of natural reef area behind the dead reef. This suggests that restoring these areas as soon as possible can possibly protect the live areas behind the impacted areas.

Mosquito Lagoon is a very popular tourist and fishing destination for boaters (Scheidt & Garreau 2007). A 2005 survey found over 100,000 registered boats within counties that are adjacent to ML (IRLNEP 2007). Approximately 46,000 boaters frequented ML between 2006 and 2007, and 76% of these were fishing boats (Scheidt & Garreau 2007). Advancement in boating technology has allowed boaters to travel into shallower waters, which can increase the negative impacts of boating (Scheidt & Garreau 2007). Boating traffic has been linked with dead reef formation as well as shoreline erosion (Gabet 1998, Grizzle et al. 2002, Wall et al. 2005). High levels of wave motion caused by boat wakes have been shown to dislodge oyster shells and potentially damage spat in the process (Walters et al. 2007). In contrast, in ML, less than 1% of deployed shells moved when sustained winds were 79 km/h (49 m/h), and shell movement due to wind-generated waves was less than 5 cm (Walters et al. 2007). Researchers have also found that boat wakes as small as 2 cm are capable of moving both individual

TABLE 3.

Oyster coverage within CANA as determined by GIS. Percent coverage is relative to 1943 coverage.

Year	No. of natural reefs	Hectares of natural reefs	No. of dead margins/reefs	Hectares of dead reefs	% Loss of natural coverage
1943	576	24.07	4	0.26	0
1951	573	20.72	6	0.52	14
1967	577	19.27	7	0.55	20
1971	573	18.62	14	0.56	23
1984	570	16.91	43	1.38	30
1995	541	15.26	84	1.54	37
2006	515	14.04	109	1.65	42
2009	524	14.46	103	1.45	40

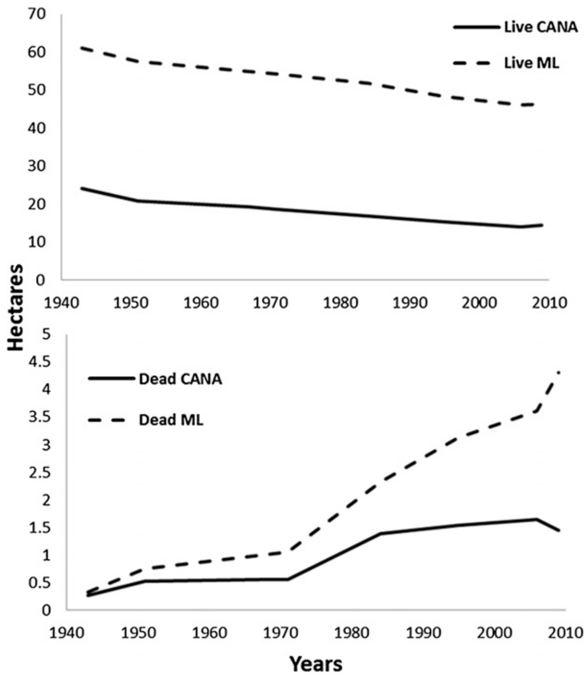

Figure 4. Change in natural reef coverage (top) and dead reef coverage (bottom) through the study period (1943 to 2009). The solid black lines represent coverage within CANA boundaries and the dashed black lines represent coverage within ML.

oysters as well as oyster clusters (Campbell 2014). Walters et al. (2007) additionally found that the highest points on natural and dead reefs did not significantly change when three hurricanes passed over the lagoon in the fall of 2004. Sites included in the multiyear reef monitoring have been repeatedly subjected to wakes from over 50 boats per hour (Walters, unpublished data).

It is possible that formation of dead areas could be related to other negative disturbance beyond recreational boat wakes. Sea level rise, overharvesting, and disease prevalence could potentially impact oyster reefs in this area. Recent studies, however, have found that intertidal oyster reefs were capable of accreting reef height in response to sea level rise (Rodriguez et al. 2014, Walles et al. 2015). Rodriguez et al. (2014) found that *Crassostrea virginica* could potentially outpace future-accelerated level of sea level rise. Walles et al. (2015) also found vertical reef growth that is in excess of local sea level rise rates for *Crassostrea gigas*. If sea level rise was affecting the formation of dead areas, one would expect a uniform distribution of dead areas within the lagoon. Within this study, however, there is a very strong correlation between dead areas and boat channels. Oyster reefs that are protected from boat wakes (via mangrove islands) are almost completely unchanged throughout the study period. In addition, catch logs from ML suggest limited harvesting pressure within CANA park boundaries (Kneifl personal communication) and low levels of disease (Walters et al. 2007).

When assessing reef conditions, it is important to understand the underlying mechanisms responsible for causing the rapid loss of reef habitat to guide future conservation planning. This study, as well as Grizzle et al. (2002), provides evidence that boat wakes are playing a major role in oyster reef declines in this shallow estuary. Understanding trends in oyster reef loss could help minimize the impact of boat wakes by implementing no wake zones in key areas. In addition, using the historical coverage data, natural resource managers can establish a baseline for oyster reef coverage in the area and demonstrate how oyster restoration is improving the current oyster coverage.

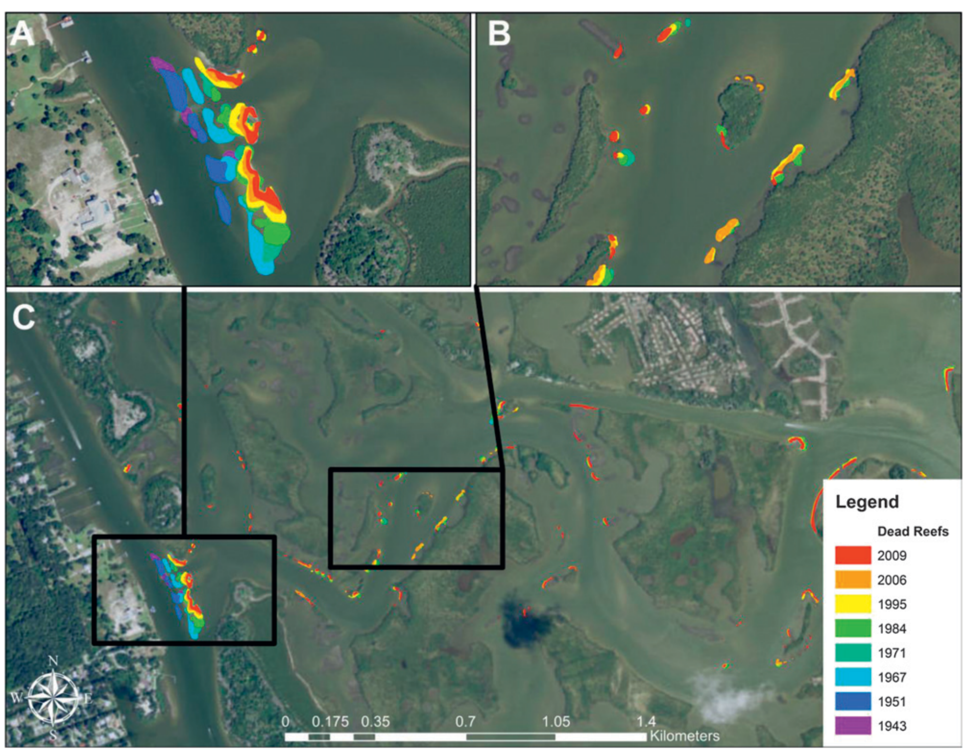

Figure 5. Selected dead reef footprints in CANA. Different colors correspond to the reef footprints within each mapped year in the study period (1943 to 2009). Panels A and B represent an enlarged view of selected areas within Panel C.

Resource managers can use all this information to set conservation and restoration goals. The loss of other marine habitats such as mangroves (30%–50%), coral reef (~20%), and seagrasses (~30%) has greatly influenced policy and conservation actions (Wilkinson 2002, Hughes et al. 2009, Spalding et al. 2010). With a global loss of 85%, oyster reef habitat protection needs to be the focus of conservation policy for estuaries, especially in protected waters (Beck et al. 2011).

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LITERATURE CITED

Barber, A., L. J. Walters & A. P. Birch. 2010. Potential for restoring biodiversity of macroflora and macrofauna on oyster reefs in Mosquito Lagoon, Florida. *Fla. Sci.* 73:47–62.

Beck, M. W., R. D. Brumbaugh, L. Airoldi, A. Carranza, L. D. Coen, C. Crawford, O. Defeo, G. J. Edgar, B. Hancock, M. C. Kay, H. S. Lenihan, M. W. Luckenbach, C. L. Toropova, G. F. Zhang & X. M. Guo. 2011. Oyster reefs at risk and recommendations for conservation, restoration, and management. *Bioscience* 61:107–116.

Birch, A. P. & L. J. Walters. 2012. Restoring intertidal oyster reefs in Mosquito Lagoon: the evolution of a successful model. Final Report, TNC-NOAA Community-Based Restoration Partnership #NA10NMF4630081. Orlando, FL: University of Central Florida. 66 pp.

Campbell, D. E. 2014. Quantifying the effects of boat wakes on intertidal oyster reefs in a shallow estuary. MS thesis, University of Central Florida.

Coen, L. D. & R. E. Grizzle. 2007. The importance of habitat created by molluscan shellfish to managed species along the Atlantic Coast of the United States. Atlantic States Marine Fisheries Commission Habitat Management Series No. 8. Washington, DC: Atlantic States Marine Fisheries Commission. pp. 1–108.

Coen, L. D. & M. W. Luckenbach. 2000. Developing success criteria and goals for evaluating oyster reef restoration: ecological function or resource exploitation? *Ecol. Eng.* 15:323–343.

Coen, L. D., M. W. Luckenbach & D. L. Breitburg. 1999. The role of oyster reefs as essential fish habitat: a review of current knowledge and some new perspectives. *Am. Fish. Soc. Symp.* 22:438–454.

Dame, R. F. 1996. Ecology of marine bivalves: an ecosystem approach. Boca Raton, FL: CRC Marine Science Series Press.

Dybas, C. L. 2002. Florida's Indian River Lagoon: an estuary in transition. *Bioscience* 52:554–559.

Dubbleday, P. S. 1975. Lagoonal circulation: ecological study of the lagoons around the John F. Kennedy Space Center. MS thesis, Florida Institute of Technology Melbourne, Florida.

Ertel, P. & A. McCall. 2005. The Nature Conservancy's Pamlico Sound reef restoration project. *J. Shellfish Res.* 24:320–321.

Gabet, E. J. 1998. Lateral migration and bank erosion in a salt marsh tidal channel in San Francisco Bay, California. *Estuaries* 21:745–753.

Grinnell, R. S. 1972. Structure and development of oyster reefs on the Suwannee River delta, Florida. Dissertation, State University of New York, Binghamton, New York, USA.

Grizzle, R. E., J. R. Adams & L. J. Walters. 2002. Historical changes in intertidal oyster (*Crassostrea virginica*) reefs in a Florida lagoon potentially related to boating activities. *J. Shellfish Res.* 21:749–756.

Hall, C. R., J. A. Provancha, D. M. Oddy, R. L. Lowers & J. D. Drese. 2001. Canaveral National Seashore water quality and aquatic resource inventory. NASA Technical Memorandum 2001-210261, Kennedy Space Center, Florida.

Hansell, H. 2012. Non-linear tidal distortion and the influence of the non-tidal forcing effects in a coastal lagoon. MS thesis, Florida Institute of Technology, Melbourne, Australia.

Hughes, A. R., S. L. Williams, C. M. Duarte, K. L. Heck & M. Waycott. 2009. Associations of concern: declining seagrasses and threatened dependent species. *Front. Ecol. Environ* 7:42–246.

Indian River Lagoon National Estuary Program. 2007. Indian River Lagoon: an introduction to a natural treasure. 40 pp.

Johnson, M. W., S. P. Powers, J. Senne & K. Park. 2009. Assessing *in situ* tolerances of eastern oysters (*Crassostrea virginica*) under moderate hypoxic regimes: implications for restoration. *J. Shellfish Res.* 28:185–192.

Kennedy, V. S. & L. P. Sanford. 1999. Characteristics of relatively unexploited beds of the eastern oyster, *Crassostrea virginica*, and early restoration programs. In: M. W. Luckenbach, R. Mann & J. A. Wesson, editors. Oyster reef habitat restoration: a synopsis and synthesis of approaches. Gloucester Point, VA: Virginia Institute of Marine Science Press. pp. 25–46.

Kennedy, V. S., D. L. Breitburg, M. C. Christman, M. W. Luckenbach, K. Paynter, J. Kramer, K. G. Sellner, J. Dew-Baxter, C. Keller & R. Mann. 2011. Lessons learned from efforts to restore oyster populations in Maryland and Virginia, 1990 to 2007. *J. Shellfish Res.* 30:719–731.

Kirby, M. X. 2004. Fishing down the coast: historical expansion and collapse of oyster fisheries along continental margins. *Proc. Natl. Acad. Sci. USA* 101:13096–13099.

Manis, J., S. Garvis, S. Jachec & L. Walters. 2015. Wave attenuation experiments over living shorelines over time: a wave tank study to assess recreational boating pressures. *J. Coast. Conserv.* 19:1–11.

Newell, R. I. E. 1988. Ecological changes in Chesapeake Bay: are they the result of overharvesting the American oyster (*Crassostrea virginica*)? In: M. Lynch & E. C. Krome, editors. Understanding the estuary. Chesapeake Research Consortium Publication 129. (CBP/TRS 24/88). Cambridge, MD: Horn Point Environmental Laboratories. pp. 536–546.

Newell, R. I. E. & E. W. Koch. 2004. Modeling seagrass density and distribution in response to changes in turbidity stemming from bivalve filtration and seagrass sediment stabilization. *Estuaries* 27:793–806.

Rodriguez, A. B., F. J. Fodrie, J. T. Ridge, N. L. Lindquist, E. J. Theuerkauf, S. E. Coleman, J. H. Grabowski, M. C. Brodeur, R. K. Gittman, D. A. Keller & M. D. Kenworthy. 2014. Oyster reefs can outpace sea-level rise. *Nat. Clim. Change* 4:493–497.

Seavey, J. R., W. E. Pine, III, P. Frederick, L. Sturmer & M. Berrigan. 2011. Decadal changes in oyster reefs in the Big Bend of Florida's Gulf Coast. *Ecosphere* 2:114.

Scheidt, D. M. & C. M. Garreau. 2007. Identification of watercraft use patterns in Canaveral National Seashore. Final report: Canaveral National Seashore PIMS Project no. 1021131.

Scyphers, S. B., S. P. Powers, K. L. Heck & D. Byron. 2011. Oyster reefs as natural breakwaters mitigate shoreline loss and facilitate fisheries. *PLoS One* 6:e22396.

Smith, N. P. 1987. An introduction to the tides of Florida's Indian River Lagoon. I. Water levels. *Fla. Sci.* 50:49–61.

Smith, N. P. 1993. Tidal and nontidal flushing of the Florida's Indian River Lagoon. *Estuaries* 16:739–746.

Spalding, M. D., M. Kainuma & L. Collins. 2010. World Atlas of mangroves. London, UK: Earthscan. 336 pp.

St. Johns River Water Management District. 2006. Water supply assessment: 2003. Technical Publication SJ2006-1. Palatka, FL: St. Johns River Water Management District.

Steward, J. S., R. V. Virnstein, M. A. Lasi, L. J. Morris, J. D. Miller, L. M. Hall & W. A. Tweedale. 2006. The impacts of the 2004 hurricanes on hydrology, water quality, and seagrass in the central Indian River Lagoon, Florida. *Estuaries Coasts* 29:954–956.

Stiner, J. L. & L. J. Walters. 2008. Effects of recreational boating on oyster reef architecture and species interactions. *Fla. Sci.* 71:31–44.

Stunz, G. W., T. J. Minello & L. P. Rozas. 2010. Relative value of oyster reef as habitat for estuarine nekten in Galveston Bay, Texas. *Mar. Ecol. Prog. Ser.* 406:147–159.

Viera, A. J. & J. M. Garrett. 2005. Understanding interobserver agreement: the kappa statistic. *Fam. Med.* 37:360–363.

Wall, L. M., L. J. Walters, R. E. Grizzle & P. E. Sacks. 2005. Recreational boating activity and its impact on the recruitment and survival of the oyster *Crassostrea virginica* on intertidal reefs in Mosquito Lagoon, Florida. *J. Shellfish Res.* 24:965–973.

Walles, B., R. Mann, T. Ysebaert, K. Troost, P. M. J. Herman & A. C. Smaal. 2015. Demography of the ecosystem engineer *Crassostrea gigas*, related to vertical reef accretion and reef persistence. *Estuar. Coast. Shelf Sci.* 154:224–233.

Walters, L. J., A. Roman, J. Stiner & D. Weeks. 2001. Water Resources Management Plan: Canaveral National Seashore, Florida. Titusville, FL: National Park Service, Canaveral National Seashore. 224 pp.

Walters, L. J., P. E. Sacks, Y. Bobo, D. Richardson & L. D. Coen. 2007. Impact of hurricanes on intertidal oyster reefs in Florida: reef profiles and disease prevalence. *Fla. Sci.* 70:506–521.

Wenner, E., H. R. Beatty & L. D. Coen. 1996. A method for quantitatively sampling nekton on intertidal oyster reefs. *J. Shellfish Res.* 15:769–775.

Wilkinson, C. 2002. Status of coral reefs of the world: 2002. Townsville, Australia: Australian Institute of Marine Science.

Wright, E. E., A. C. Hine, S. L. Goodbred & S. D. Locker. 2005. The effect of sea-level and climate change on the development of a mixed siliciclastic carbonate, deltaic coastline: Suwannee River, Florida, USA. *J. Sediment. Res.* 75:621–635.