Supporting Conservation of Mesquite Bay Reefs

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Final Report

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Summary

Oyster reefs in the Mesquite Bay complex were investigated for the current status and long-term changes in oyster populations, oyster reef structure/areal extent, and usage by avian populations. The current populations of oysters at depths less than 0.5 m at Ayres, Carlos, Cedar, and Third Chain of Islands Reefs (>100 oysters m^{-2}) are greater than many other harvested reefs along the Texas coast and are comparable to successfully restored and unharvested restored reefs. The oyster populations at Second Chain of Islands Reef and on the portions of other Mesquite Bay complex reefs greater than 0.5 m have densities of oysters comparable to other low-density, harvested reefs. Oyster populations in subtidal areas of Ayres, Ranch House, Second Chain of Islands, and Second Chain of Islands West have increased in live oyster density from 1976 to 2021. However, a decrease in sampling depth at the same locations means that either the reefs have been sampled in shallower areas over time, or the depths of the reefs have decreased. Mean oyster shell heights have not changed. Spat density has increased in at least four reef areas during the same 36-year period. Oyster and spat abundance was positively correlated with salinity but negatively correlated with depth spatiotemporally in the Mesquite Bay complex.

Areal extent of intertidal reef structure decreased at Carlos, Cedar, Third Chain of Islands, and Ayres Reefs from 1949 to 2016, despite short-term losses and gains in between. Intertidal reef extent at Second Chain of Islands Reef remained the same. Subtidal reefs were only able to be mapped on a short-term scale with three years (2004, 2009, and 2016) and results indicated that the areal extent of subtidal reef structure increased at Carlos and Cedar Reefs but decreased at Third Chain of Islands, Ayres Reef, and Second Chain of Islands from 2004 to 2016. There is less certainty about the apparent changes in subtidal reef structure because differences in aerial image quality may have affected the ability to delineate subtidal reefs.

The availability of suitable nesting area on natural islands in the Mesquite Bay complex can set an upper bound on nesting bird populations. Comparisons of bird populations in 1939 with those occurring from 1973 to 2022 indicate that the Mesquite Bay complex supports far fewer nesting birds than it did historically and that the trend is continuing downward. In the shorter-term (2004–2022), the loss of vegetated nesting area has been especially pronounced, and coincides with lower numbers of nesting wading birds.

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List of Acronyms

Acronym	Full Name
APSF	Aerial Photo Single Frame
CBBEP	Coastal Bend Bays and Estuaries Program
GIS	Geographic Information System
GIWW	Gulf Intracoastal Waterway
GPS	Global Positioning System
HRI	Harte Research Institute for Gulf of Mexico Studies
NAIP	National Agriculture Imagery Program
NOAA	National Oceanic and Atmospheric Administration
RMSE	Root Mean Square Error
SARA	San Antonio River Authority
TAMUCC	Texas A&M University-Corpus Christi
TCWS	Texas Colonial Waterbird Survey
TGLO	Texas General Land Office
TNRIS	Texas National Resources Information System
TOP	Texas Orthoimagery Program
TPWD	Texas Parks and Wildlife Department
T-Sheet	Topographic Sheet depicting a shoreline survey by NOAA
USDA	United States Department of Agriculture
USGS	United States Geological Survey
WGS	World Geodetic System

Introduction

Coastal habitats are recognized for providing ecological benefits and supporting coastal resiliency. However, the Texas coast is vulnerable to pressures from natural disasters and human activities. *Crassostrea virginica* oyster reefs rank highest among degraded marine systems, with an estimated 50-85% loss throughout the Gulf of Mexico. Loss of reef habitat translates into loss of biodiversity and associated ecosystem services, including viable fisheries, habitat provision, and water filtration. Conservation and restoration of oyster reefs can help communities become more resilient by providing natural buffers against storms, improving water quality, supplying critical habitat, and supporting coastal recreation and tourism.

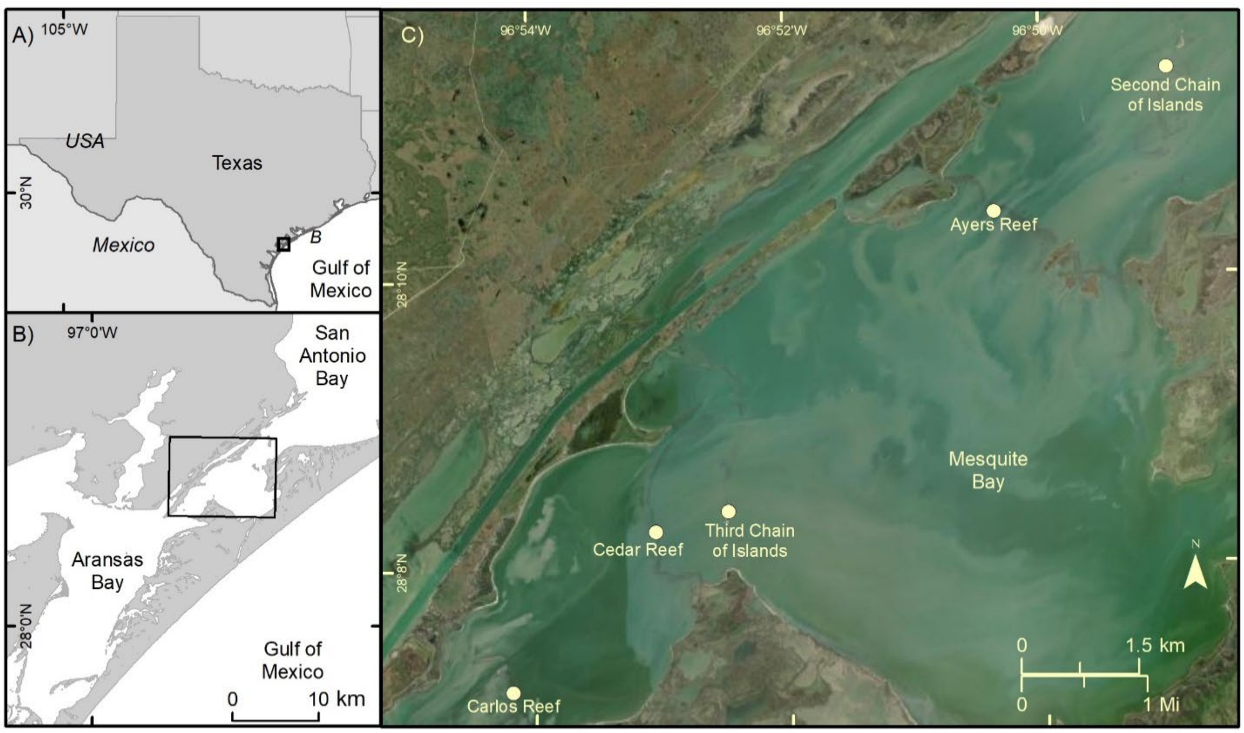

Figure 1. The Mesquite Bay complex, including prominent reefs.

In the Texas Coastal Bend, Second Chain of Islands, Ayres Reef, Third Chain of Islands, Cedar Reef, and Carlos Reef (hereafter referred to collectively as 'Mesquite Bay Reefs') are historically productive oyster reefs that support colonial waterbird populations and attenuate waves from vessels passing between the Mission-Aransas and Guadalupe Estuaries (Figure 1). However, in recent years, reduced oyster densities and overall degradation of reef habitat have been observed, likely due to the effects of Hurricane Harvey and ongoing commercial harvest activities (i.e., dredging, Figure 2). Second Chain of Islands, Third Chain of Islands, and Carlos Reef appear to have experienced the greatest declines, thus there is strong interest in protecting, conserving, and restoring the remaining Mesquite Bay Reef habitat.

Figure 2. Turbidity plumes caused from commercial oyster dredging around Carlos Reef. Imagery date: January 24, 2019.

The purpose of this study is to assess historical data and collect new data to support future conservation and restoration of Mesquite Bay Reefs, which may include:

(1) obtaining TGLO surface leases to conduct oyster reef restoration activities,

(2) working with TPWD to target oyster cultch placement efforts to these reefs, and

(3) bringing a request to the TPWD Commission for closure of these reefs to limit commercial harvest and allow oyster population recovery.

Objectives

The objectives of this study are to:

(1) Determine current densities and size distribution of oysters on Mesquite Bay Reefs by sampling in two seasons in 2022,
(2) Evaluate long-term trends in oyster relative abundance and size using TPWD Fisheries Independent Monitoring Program data (Martinez-Andrade et al. 2018),
(3) Investigate potential changes in physical reef structure using historical aerial imagery, and
(4) Assess colonial waterbird use of the reefs using CBBEP Coastal Bird Program data

Current Oyster Populations

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Methods

Sampling of oysters occurred at five sites along transects at two locations at five reefs in the Mesquite Bay complex: Second Chain of Islands, Ayres, Third Chain of Islands, Cedar and Carlos Reefs (n = 5 reefs x 2 locations x 5 stations = 50 samples; Figure 3) by HRI employees. Sampling occurred from 25–27 January 2022 and repeated on 26 May 2022. Locations on each reef were spaced approximately 100 m apart along the crest of the reef, as determined from aerial imagery and field scouting. PVC poles and a Global Positioning System (GPS) unit (latitude/longitude, WGS 84 datum) were used to mark and relocate sampling sites.

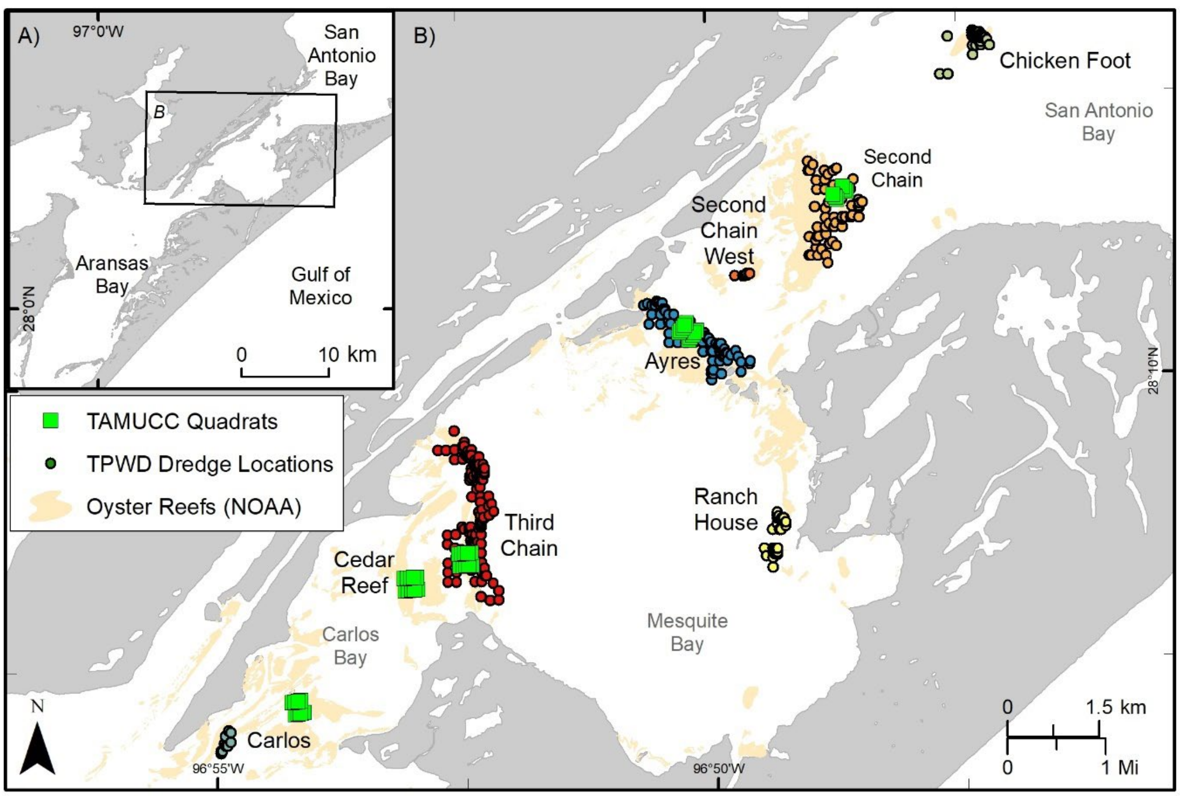

Figure 3. Map of reefs sampling locations sampled by TAMUCC and TPWD.

Oyster Sampling

Oyster sampling was conducted using quadrats at each of the Mesquite Bay Reefs sampling sites. A 0.25 or 0.5 m^2 quadrat was placed on a randomly selected area at each sampling site, and the approximate substrate composition of the bottom was noted (e.g., dominated by mud, sand, shell hash, whole shells). These qualitative notes about the substrate were determined visually and/or tactilely by estimating the areal coverage by whole shells (live and dead), shell hash (broken shell), sand, and/or mud occurring on the sediment/reef surface within the quadrat. The larger quadrat size was used where oyster densities appeared to be low. After substrate characterization, all live oysters/shell material down to approximately 5 cm from sediment/reef surface were manually collected and placed in a mesh bag. Live and dead oyster shells were quantified on the boat as follows:

1. Measure shell heights for all live oysters present up to 40 individuals (if more than 40 are present, count the rest and record the total number present).
2. Place live oysters in a bucket with volume demarcations.
3. Transfer a known volume of water into the bucket with live oysters and record the volume displaced (shell volume of live oysters).
4. Count all unoccupied ("dead") shells present and place them into a bucket, again recording the volume displaced (total shell volume).
5. Record whether spat and other encrusting fauna are present on oysters/shells in each sample.
6. After processing, all live oysters and dead shell will be returned to where they were collected.

Water Quality Sampling

A multiparameter instrument (YSI ProDSS data sonde or equivalent) was used to synoptically measure water temperature, dissolved oxygen, pH, salinity and turbidity at each reef during oyster sampling. Water depth, wind speed, wind direction, cloud cover, and wave height were also recorded with all sampling activities. At each collection location, the multiparameter instrument was deployed at the surface (0.1 m deep), and approximately 0.1 m above the bottom if deep enough. Field sampling and instrument calibrations procedures are documented within the project QAPP (Pollack 2021), and approved state and federal documents (USEPA 2010, TCEQ 2012).

Results

Oyster Populations

The transects sampled were 200-m long at Ayres, Cedar and Third Chain of Islands Reefs, but only 150-m long at Carlos Reef and 100-m long at Second Chain of Islands Reefs. The length of each reef sampled was shorter at Carlos and Second Chain of Islands Reefs because the reefs were thinner (horizontally) than the other reefs. The total depth of the sites sampled along the reefs were usually less than 1.0 m, although depths as deep as 2.0 m were sampled (Figure 4).

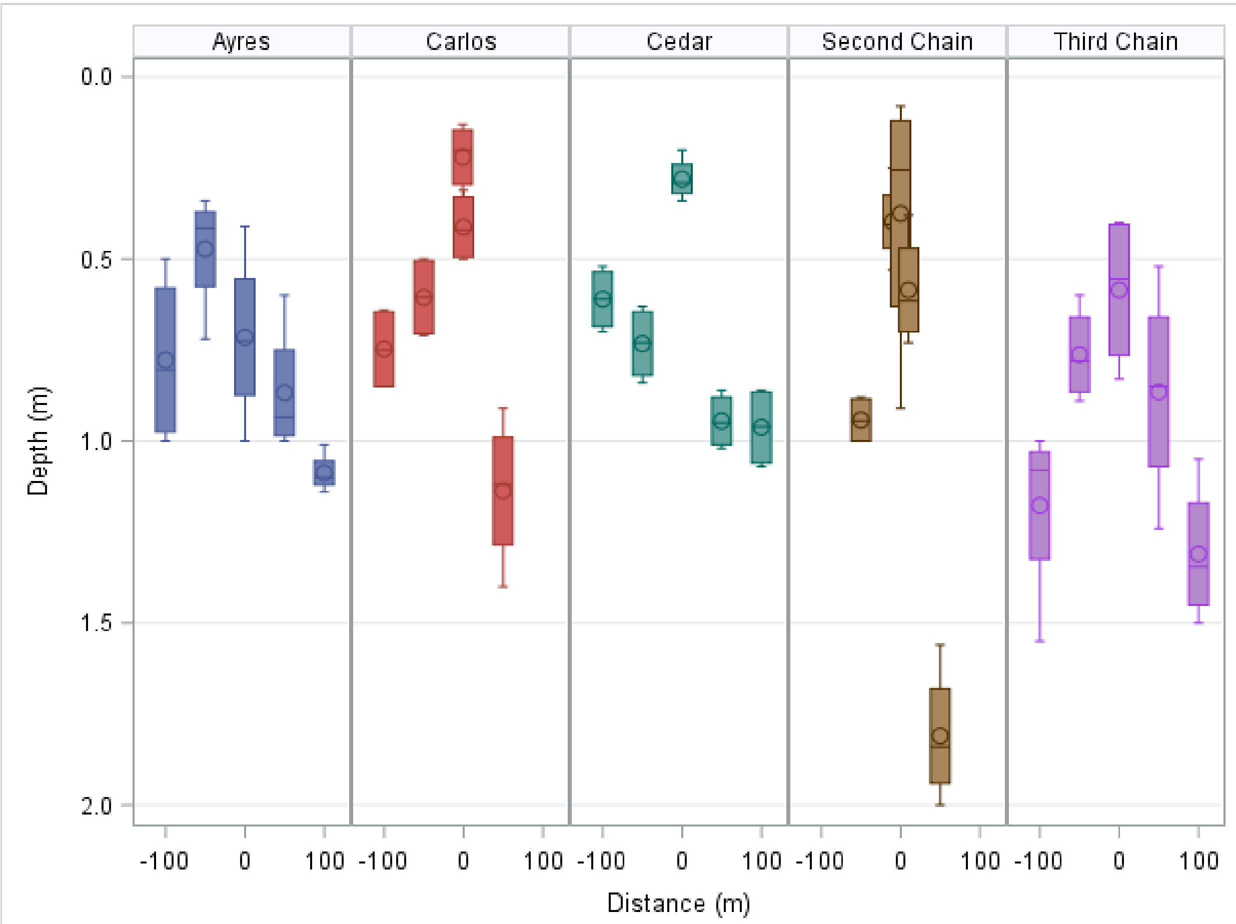

Figure 4. Water depth at oyster sampling locations at reefs in the Mesquite Bay complex in 2022 plotted by distance from the estimated reef crest.

Mean oyster density and the proportion of live shell (relative to dead shell) (± standard deviation) was greatest at Carlos (58 ± 94 m^{-2}, 17 ± 20% by shell abundance) and Cedar Reefs (55 ± 76 m^{-2}, 16 ± 14% by shell abundance; Table 1, Figure 6). Mean density and the proportion of live shell was much lower at Second Chain of Islands Reef (7 m ± 4 m^{-2}, 8 ± 8% by shell abundance) than at all other reefs. Mean densities were similar in January and May 2022, although the interquartile range was larger in January at all reefs (Figure 6). Oyster densities were greatest in shallower depths at all reefs except on Second Chain of Islands Reef, where there was no discernable difference among depths (Figure 7). The proportion of live oyster shells was greater in May than January 2022 at Carlos, Cedar, and Third Chain of Islands Reefs, but similar in both months in Ayres and Second Chain of Islands Reefs (Figure 8).

Mean oyster heights were higher at Ayres (67 mm) and Carlos Reefs (68 mm) than the other three reefs sampled (61 to 62 mm; Table 1), however the range of heights were similar among reefs (Figure 5). Mean and median oyster heights were greater in May than January at all five reefs sampled (Figure 9).

Table 1. Mean density, height, live oyster volume, and proportion of live shell (by abundance and by volume) at reefs sampled by TAMUCC in January and May 2022.
SD = Standard deviation.

	Density (m^{-2})		Height (mm)		Live oyster Volume (L)		% Live Shell by Abundance		% Live Shell by Volume	
Reef	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
Ayres	41.6	71.0	66.9	11.3	0.9	1.5	12.2	11.4	28.7	19.9
Carlos	58.4	93.9	67.8	5.7	1.4	2.1	16.6	19.7	38.4	24.3
Cedar	55.1	76.1	61.8	9.7	1.3	1.5	16.2	14.4	34.4	25.7
Second Chain	7.3	4.1	61.0	11.6	0.2	0.1	8.2	7.5	17.4	13.6
Third Chain	35.5	75.7	61.4	10.5	0.6	1.1	15.2	17.3	26.4	21.6

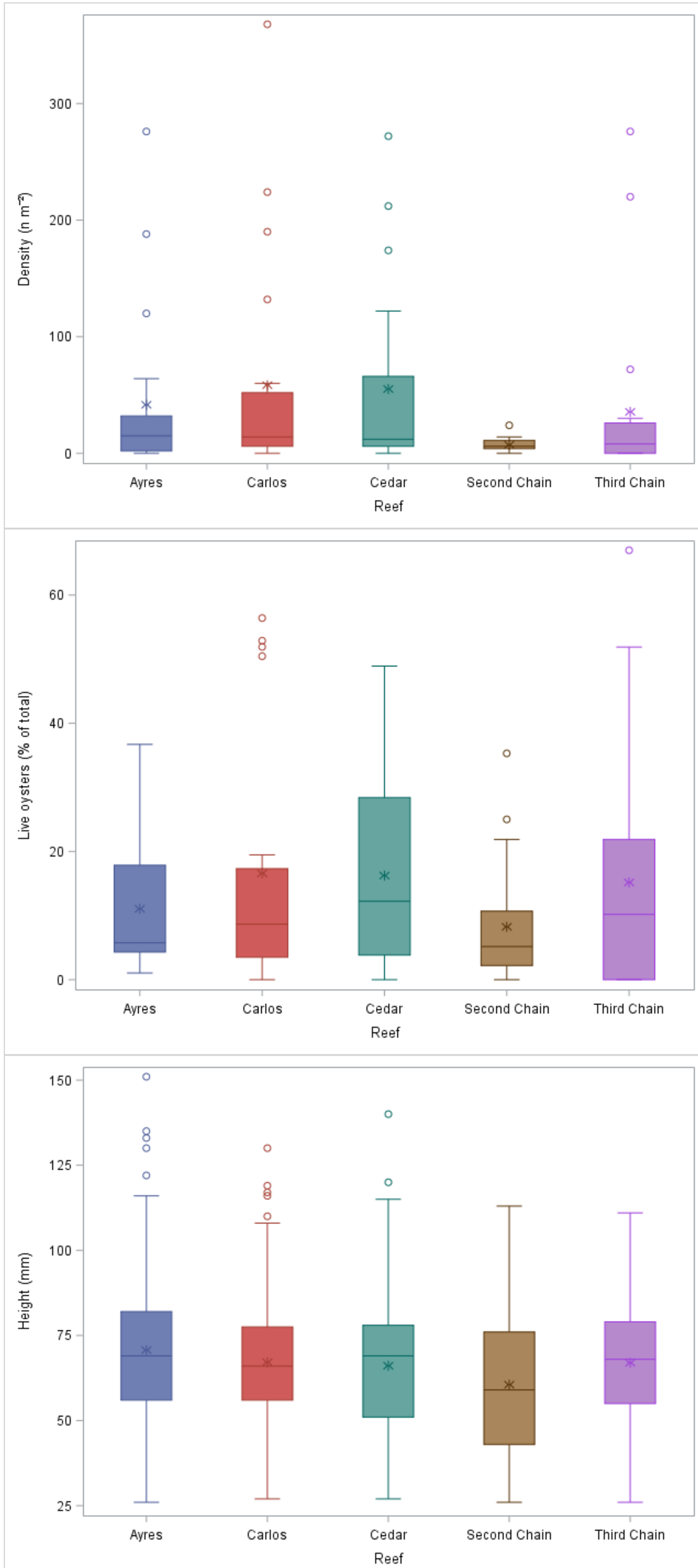

Figure 5. Live oyster density (top), proportion of live oysters (middle) and heights of live oysters (bottom) at the five reefs sampled by HRI in 2022.

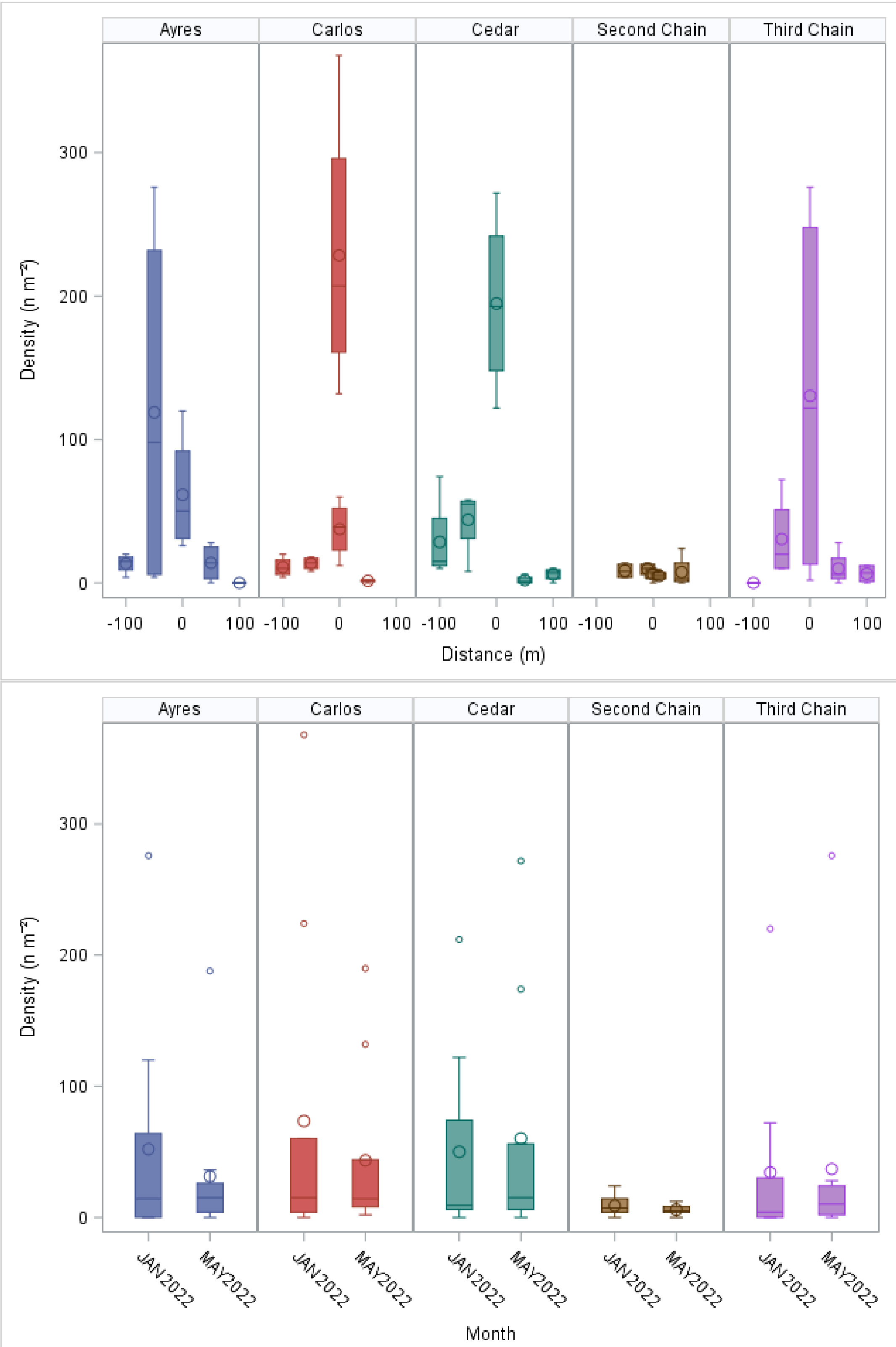

Figure 6. Oyster densities at reefs in the Mesquite Bay complex in 2022 separated by distance from the reef crest (top) and date sampled (bottom).

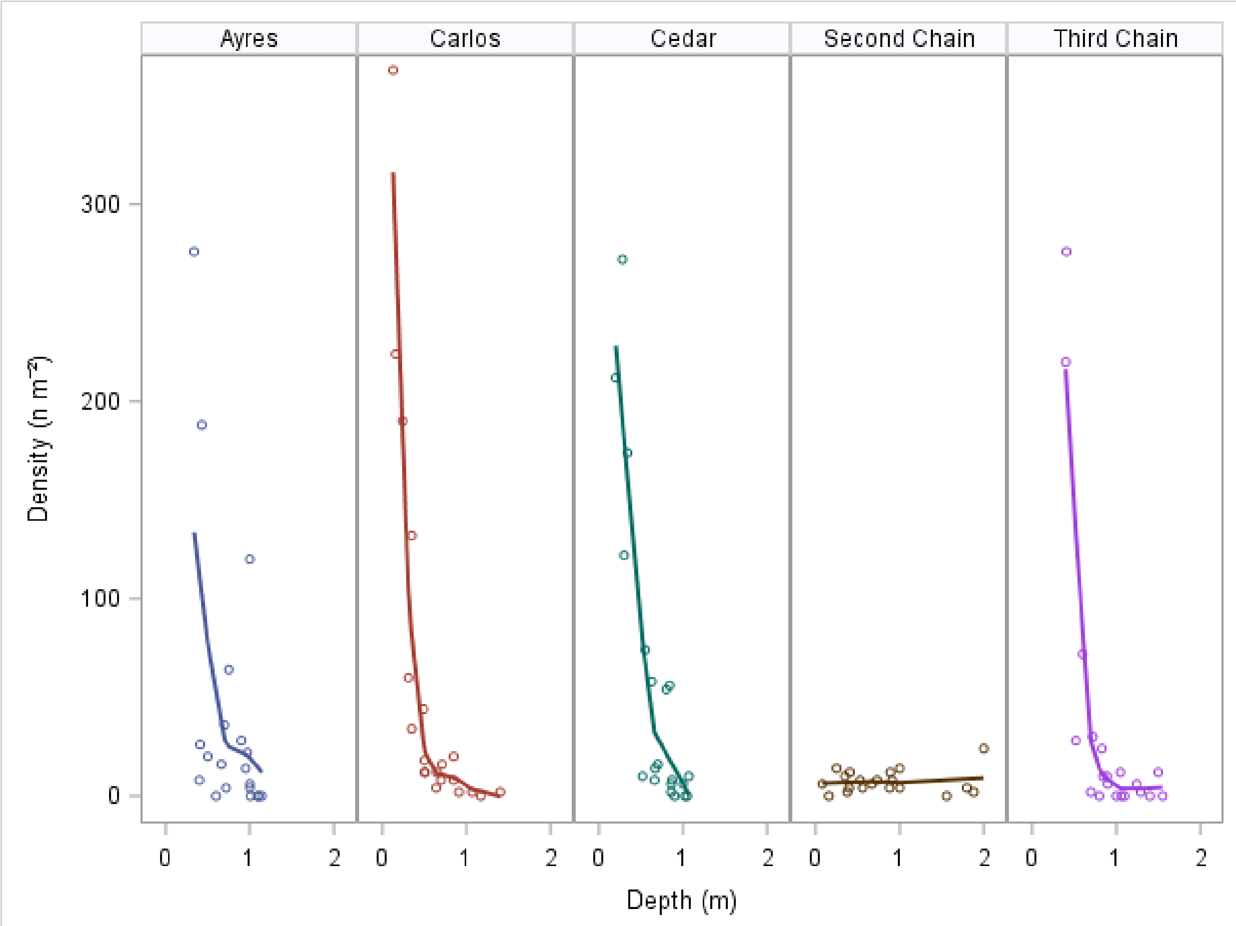

Figure 7. Loess regression of live oyster density and depth at HRI-sampled oyster reefs.

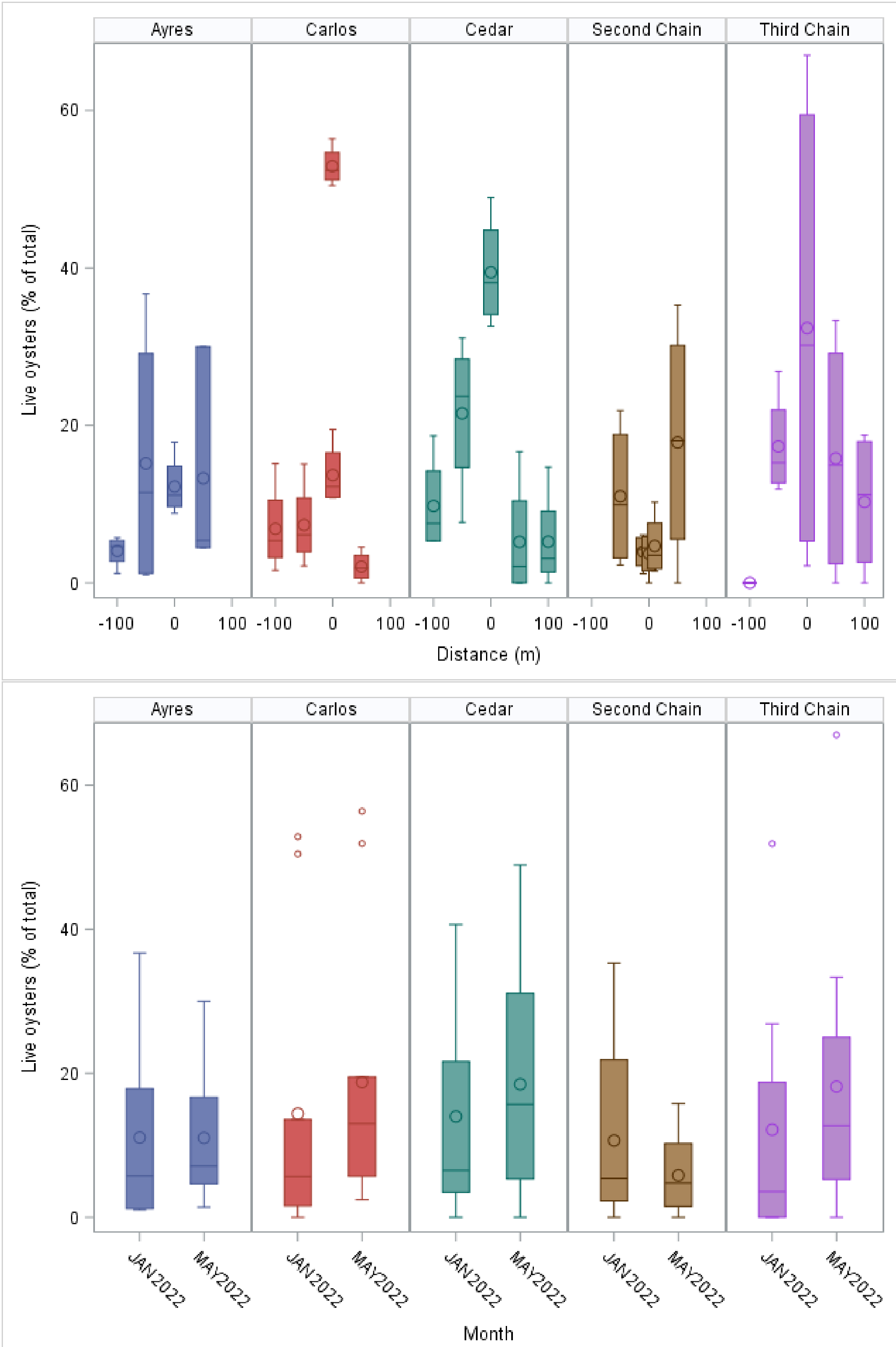

Figure 8. Proportion of live oysters at reefs in the Mesquite Bay complex in 2022 separated by distance from the reef crest (top) and date sampled (bottom).

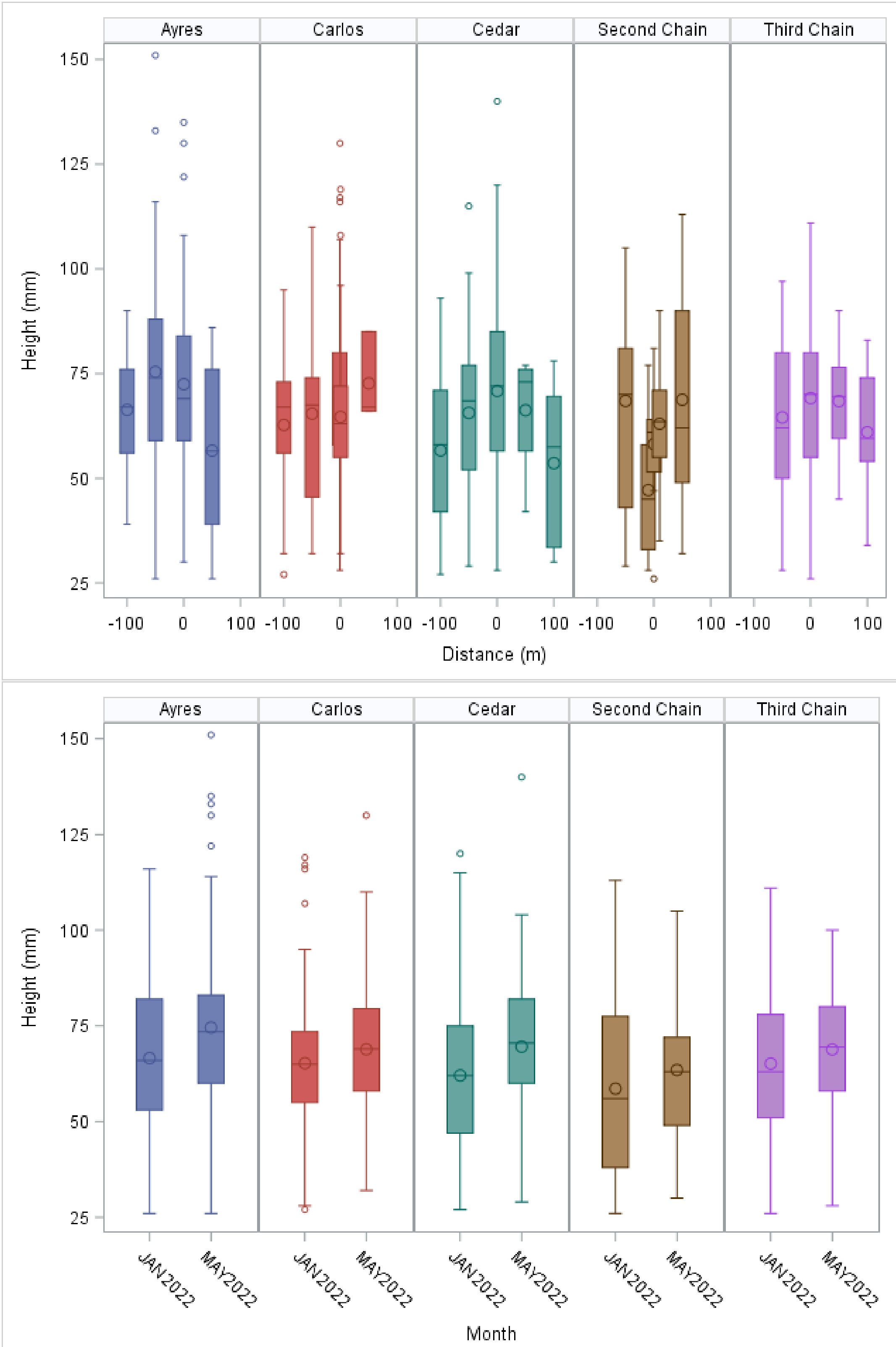

Figure 9. Oyster heights at reefs in the Mesquite Bay complex in 2022 plotted by distance along the reef transect (top) and date sampled (bottom).

Water Quality

Salinity increased from an overall mean of 20.7 in January to 27.8 in May 2022 (Table 2, Figure 10). Similarly, water temperature increased from an overall mean of 11.2 °C in January to 26.6 °C in May 2022. This combination of increased salinities and temperatures corresponded in decreases in dissolved oxygen concentration from 9.9 to 6.9 mg L^{-1}, and pH from 8.3 to 8.1 in the same four-month period. Water transparency, as measured by a secchi disk, was greater in January than May at three reefs, but lesser at two reefs. Water transparency was at least 0.3 m greater at Third Chain of Islands Reef than all four other reefs (Table 2, Figure 10).

Table 2. Water quality at the reefs sampled by HRI in January and May 2022.

	Salinity		Temperature (°C)		Dissolved Oxygen (mg L^{-1})		pH		Secchi depth (m)	
Reef	Jan	May	Jan	May	Jan	May	Jan	May	Jan	May
Ayres	19.9	27.9	10.7	25.6	10.2	5.8	8.3	8.0	0.3	0.5
Carlos	20.8	28.6	12.1	28.7	9.3	7.1	8.2	8.1	0.6	0.3
Cedar	20.9	27.8	12.3	27.0	9.6	7.3	8.2	8.1	0.4	0.5
Second Chain	20.2	26.9	9.8	25.5	10.1	7.2	8.3	8.1	0.7	0.3
Third Chain	21.5	27.8	11.2	26.2	10.3	7.0	8.3	8.2	1.1	0.8
Mean	20.7	27.8	11.2	26.6	9.9	6.9	8.3	8.1	0.6	0.5

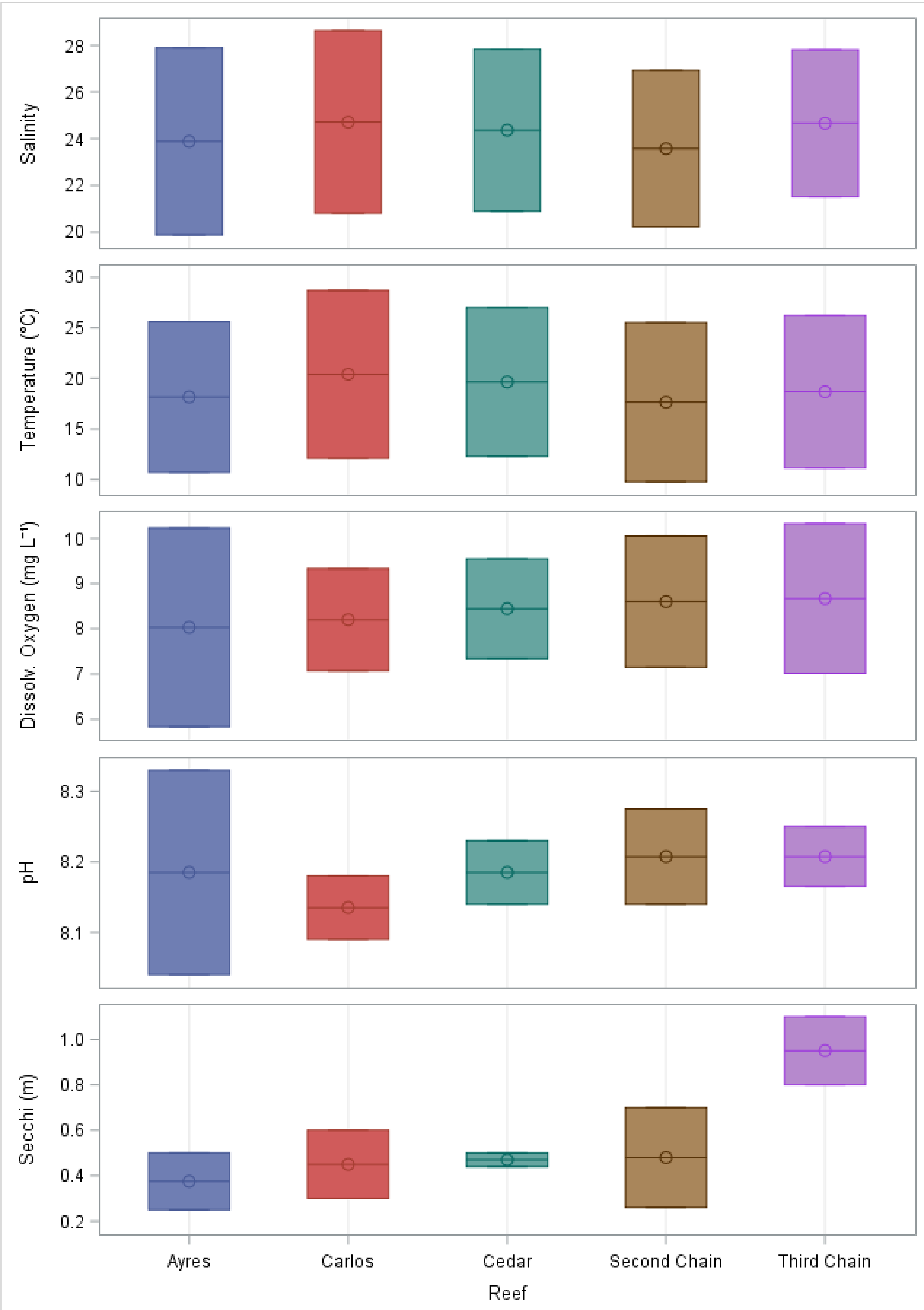

Figure 10. Water quality at reefs within the Mesquite Bay complex in January and May 2022.

Discussion

Peak oyster densities at all five reefs except Second Chain of Islands Reef were all >200 oysters m^{-2}. It is important to note that all occurrences of oyster densities ≥120 oysters m^{-2} occurred at depths <0.5 m except for one occurrence at Ayres Reef (120 oysters m^{-2} at 1.0 m). These peak densities of oysters are therefore out of the range of oyster dredging boats, except perhaps during extreme high tides. The peak densities observed in this 2022 survey are much greater than densities at 19 subtidal natural reefs along the Texas coast that were surveyed in 2018 (<95 oysters m^{-2}; Pollack and Palmer 2020). Aside from at Second Chain of Islands Reef, the mean densities at Mesquite Bay reefs are greater than densities at almost all of the 19 subtidal natural reefs along the Texas coast.

Peak densities at Mesquite Bay complex reefs are similar to densities at successfully restored artificial reefs in Texas — oyster densities at Grass Islands in Aransas Bay were 172 to 299 oysters m^{-2} (mean height 52 mm, unpublished data) and at Half Moon Reef in Matagorda Bay were 134 to 248 oysters m^{-2} (annual means from 2016 to 2019; Beseres Pollack et al. 2021). As with the shallow, dense parts of the oyster reefs in Mesquite Bay, these restored reefs are considered unharvestable by oyster dredge.

Long-Term Trends in Oyster Populations

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Methods

Historic oyster sizes and abundances were determined using Texas Parks and Wildlife Department (TPWD)-collected data. The TPWD, Coastal Fisheries Program, has been running a fisheries-independent monitoring program designed to manage fisheries by estuary throughout Texas (Martinez-Andrade et al. 2018, TPWD 2022). Oyster surveys were completed in a stratified random sampling design on defined oyster reefs in each major oyster-producing bay in Texas (including in Mesquite Bay) using oyster dredges (1984-present) pulled linearly for 30 seconds at approximately 1.3 m s^{-1} (~18 m^2). Oyster dredges used by the TPWD are Louisiana style 9-tooth: 46 cm wide, 25 cm tall with a 36 cm deep bag. The heights of up to twenty live oysters from each dredge sample were measured, and all live oysters were counted. The number of spat on up to five dead and five live oyster shells in a dredge sample were also counted. The number of dead oyster shells from each dredge sample were also recorded by TPWD but not analyzed in this report. Measurements of water temperature, salinity, pH and turbidity were taken during each oyster sampling event.

Oyster abundance and size data from 1986 to 2021 were analyzed by TAMUCC for this report. TPWD sampling stations were first entered into a Geographic Information System (GIS, ArcGIS Pro 2.9.0). Sampling stations were then labelled with the reef name that they were most-closely located (Figure 3). Any stations that did appear to be close to any reef were not used in our analyses. Oyster characteristics (oyster height, spat abundance, oyster abundance) and water quality (e.g., salinity, temperature) were averaged (mean) by year for each reef to minimize any biases attributed to spatiotemporal variations in sampling. Spat, abundance and height data were sampled by TPWD an average of one to nine times per year at each reef (Table 3).

Spearman rank correlations of oyster and water quality variables with sampling year (1986 to 2011) were made to determine any long-term increases or decreases over time. Linear regressions were also calculated to determine the rate of change if any temporal change occurred. Spearman rank correlations were also calculated between oyster variables and water quality variables to identify any potential causes of change to oyster communities.

Table 3. Sampling intensity at each reef (TPWD data).

Reef	Total number of samples			Mean number of samples per year		
	Spat	Abundance	Height	Spat	Abundance	Height
Ayres	130	130	107	3.9	3.9	3.2
Carlos Reef	144	177	104	4.1	5.1	3.0
Chicken Foot	247	255	204	6.9	7.1	5.7
Ranch House	82	176	44	2.3	4.9	1.2
Second Chain	144	151	122	4.0	4.2	3.4
Second Chain West	156	158	140	4.3	4.4	3.9
Third Chain	277	346	225	7.1	8.9	5.8

Results

Water Quality

There was no increase or decrease in salinity at any reef over the 36-year sampling period at any single reef area ($p \geq 0.17$; Table 4). However, temperature increased ($R \geq 0.37$, $p \leq 0.03$) and dissolved oxygen decreased ($R \leq -0.40$, $p \leq 0.02$) at Second Chain of Islands and Chicken Foot Reefs, and temperature only increased at Third Chain of Islands Reef ($R = 0.37$, $p = 0.03$). Turbidity decreased at Carlos, Ranch and Third Chain of Islands Reefs ($R \leq -0.32$, $p \leq 0.05$). The variable with the strongest correlation with time is total depth, which decreased at Ayres, Ranch House, Second Chain of Islands and Second Chain of Islands West Reefs ($R \leq -0.37$, $p \leq 0.03$; Figure 12). Decreases in sample depth varied from 5 to 9 mm/year (0.005 to 0.009 m/year; Table 6). Correlations of water quality variables over time were similar to those calculated using a partial correlation with depth (Table 9).

Table 4. Spearman correlations between water quality variables and year (TPWD data). Bolded numbers indicate correlations with $p < 0.05$. Grayed, bolded numbers indicate correlations with $p < 0.10$.

Reef area	n	Temperature (°C)		Dissolved Oxygen (mg L^{-1})		Salinity		Turbidity		Depth (m)	
		r	p	r	p	r	p	r	p	r	p
Ayres	33	0.10	0.5838	-0.06	0.7611	0.18	0.3296	-0.11	0.5510	**-0.42**	**0.0142**
Carlos	35	-0.11	0.5407	0.15	0.4010	0.04	0.8069	**-0.40**	**0.0166**	0.09	0.6253
Chicken Foot	36	**0.37**	**0.0252**	**-0.48**	**0.0028**	0.09	0.5930	-0.27	0.1137	0.10	0.5556
Ranch House	36	0.05	0.7919	-0.18	0.2923	0.12	0.4732	**-0.32**	**0.0546**	**-0.50**	**0.0017**
Second Chain	35	**0.42**	**0.0116**	**-0.40**	**0.0166**	0.15	0.3803	-0.28	0.1006	**-0.48**	**0.0036**
Second Chain West	36	0.12	0.4930	-0.22	0.1923	0.23	0.1696	-0.25	0.1396	**-0.37**	**0.0282**
Third Chain	36	**0.37**	**0.0265**	0.11	0.5278	-0.04	0.8080	**-0.34**	**0.0403**	-0.06	0.7366

Oyster Populations

Abundances of oysters were greatest at Second Chain of Islands (49.8 oysters/tow) and Second Chain of Islands West (44.4 oysters/tow), and least at Ranch House Reef (9.6 oysters/tow) in the 10-year period from 2012 to 2021 (Table 5). Mean oyster heights ranged from 55 and 64 mm at each reef, spat abundance ranged from 0.5 and 0.9 spat per shells during the same 10 period. Mean salinities ranged from 20.4 to 23.5 and dredge depths ranged from 0.8 to 1.4 m during oyster collections from 2012 to 2021. Slightly fewer, but on average larger oysters, were sampled in 2021 (31.1 oysters/tow, height 67 mm) than from 2012 to 2021 (31.5 oysters/tow, height 60 mm). Spat abundances and salinities were lesser in 2021 (0.2 spat/shell, salinity 16.0) than the 2012–2021 average (0.7 spat/shell, salinity 22.5), but dredge depths were similar (1.1 m for both periods). Caution must be taken when making comparisons between 2021 and other time periods for a single reef because each reef was sampled only two to five times in 2021.

Table 5. Mean oyster abundance, height, spat abundance and associated mean salinity and sample depth at each reef area from 2012 to 2021, and in 2021.

	Abundance		Height		Spat abundance		Salinity		Depth	
Reef area	2012-2021	2021	2012-2021	2021	2012-2021	2021	2012-2021	2021	2012-2021	2021
Ayres	35.8	31.4	60.1	66.2	0.9	0.1	23.5	17.3	0.8	0.6
Carlos	26.9	61.0	63.4	78.7	0.6	0.5	22.8	17.8	1.4	1.3
Chicken Foot	27.6	12.3	59.6	75.1	0.9	0.2	20.4	10.8	1.3	1.3
Ranch House	9.6	9.3	54.6	56.4	0.6	0.1	22.8	18.1	1.1	1.2
Second Chain	49.8	37.8	58.7	57.7	0.5	0.4	22.8	13.1	0.9	1.3
Second Chain West	44.4	40.4	56.2	68.2	0.7	0.1	24.2	22.3	0.8	0.8
Third Chain	26.3	25.4	64.4	67.7	0.7	0.1	20.8	12.3	1.2	1.1
Mean	31.5	31.1	59.6	67.2	0.7	0.2	22.5	16.0	1.1	1.1

Live oyster abundances increased over time at Ayres, Ranch House, Second Chain of Islands, and Second Chain of Islands West Reefs ($R \geq 0.48$, $p \leq 0.003$) and possibly Carlos Reef ($R \geq 0.32$, $p = 0.06$; Table 7, Figure 12). Increases in oyster abundance varied from 0.4 to 1.3 oysters/dredge/year (Table 6). Spat abundance increased at Carlos, Ranch House, Second Chain of Islands and Third Chain of Islands Reefs ($R \geq 0.43$, $p \leq 0.01$) and possibly Chicken Foot Reef ($R = 0.32$, $p = 0.06$; Figure 13). Increases in spat abundance varied from 0.01 to 0.02 spat/shell/year (Table 6). However, oyster heights did not increase or decrease over the 36-year sampling period at any reef ($p \geq 0.09$). When correlating with sampling year using a partial correlation with depth, live oyster abundance only increased at three reefs, whereas spat abundance increased in at least five reefs (Table 9).

Table 6. Regression analysis of oyster abundance, spat abundance and depth with year (TPWD data). RMSE = Root mean square error. Bolded numbers indicate Spearman rank correlations and linear regression with $p < 0.05$). Italicized numbers indicate Spearman correlations < 0.05 but linear regression > 0.05.

Reef area	Abundance			Spat			Depth		
	RMSE	Intercept	Slope	RMSE	Intercept	Slope	RMSE	Intercept	Slope
Ayres	**14.12**	**-1837**	**0.927**	0.66	-40.38	0.020	**0.15**	**14.41**	**-0.007**
Carlos	15.35	-784	0.401	**0.42**	**-30.88**	**0.016**	0.22	-3.42	0.002
Chicken Foot	21.43	-870	0.444	0.95	-31.12	0.016	0.19	-3.28	0.002
Ranch House	**8.40**	**-715**	**0.359**	**0.40**	**-30.81**	**0.016**	**0.18**	**17.59**	**-0.008**
Second Chain	**15.85**	**-2600**	**1.313**	*0.47*	*-22.18*	*0.011*	**0.18**	**19.43**	**-0.009**
Second Chain West	**20.63**	**-1921**	**0.972**	0.57	-24.28	0.012	**0.13**	**10.33**	**-0.005**
Third Chain	18.17	-501	0.262	**0.37**	**-43.09**	**0.022**	0.14	4.55	-0.002

Table 7. Spearman correlations between oyster variables and year (TPWD data). Bolded numbers indicate correlations with $p < 0.05$. Grayed, bolded numbers indicate correlations with $p < 0.10$.

Reef area	Abundance (/tow)			Height (mm)			Spat abundance (/oyster)		
	n	r	p	n	r	p	n	r	p
Ayres	33	**0.54**	**0.0011**	32	0.13	0.4728	33	0.30	0.0853
Carlos	35	**0.32**	**0.0600**	32	-0.09	0.6165	35	**0.43**	**0.0108**
Chicken Foot	36	0.03	0.8815	36	0.07	0.6751	36	**0.32**	**0.0558**
Ranch House	36	**0.63**	**<0.0001**	22	-0.26	0.2408	30	**0.58**	**0.0008**
Second Chain	35	**0.67**	**<0.0001**	35	-0.18	0.3105	35	**0.45**	**0.0063**
Second Chain West	36	**0.48**	**0.0031**	36	-0.28	0.0946	36	0.26	0.1296
Third Chain	36	0.20	0.2434	36	0.06	0.7226	36	**0.61**	**<0.0001**

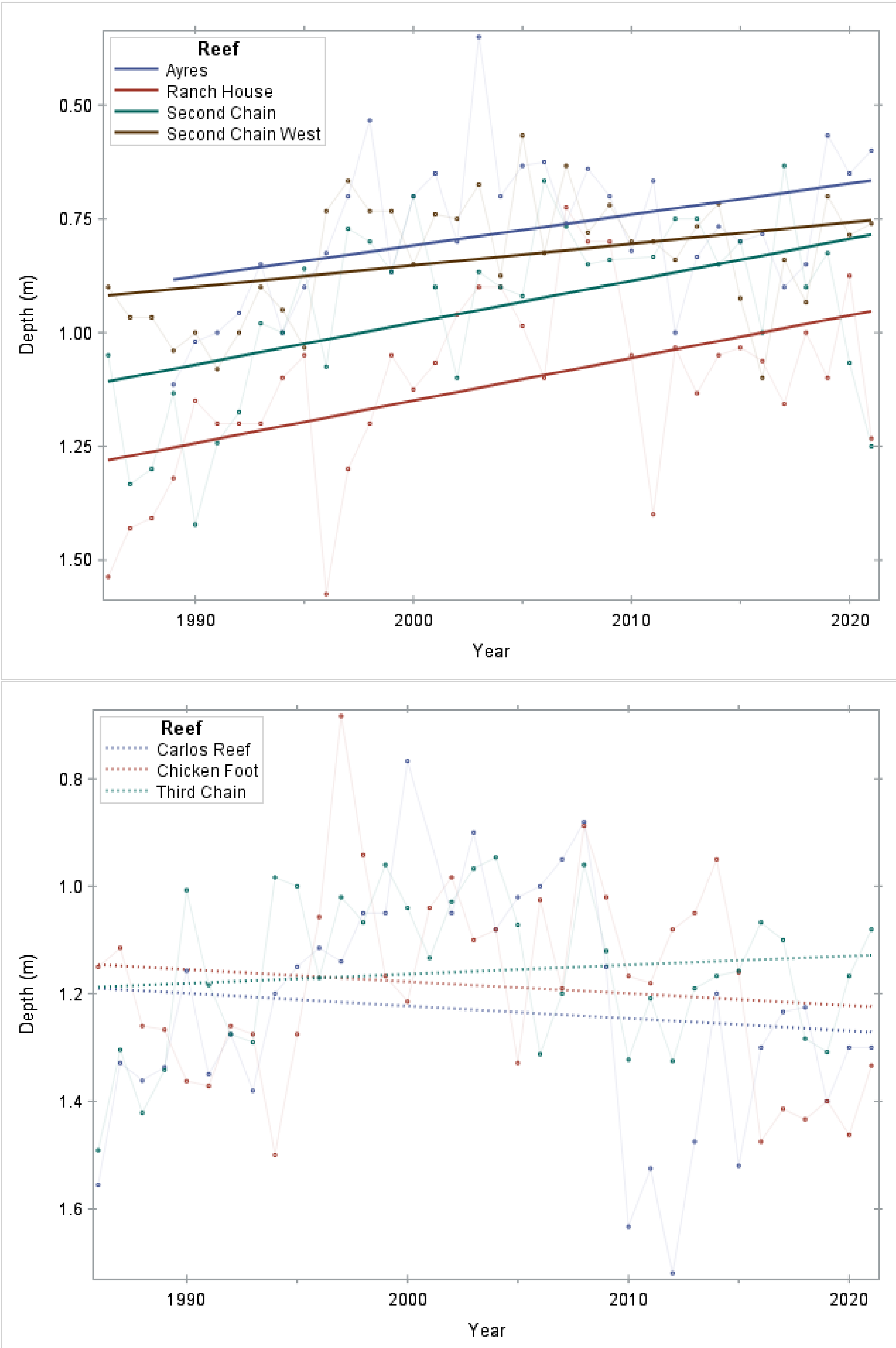

Figure 11. Linear regression of sample depth at each reef over time.
Solid regression lines indicate $p_{Spearman} < 0.05$.

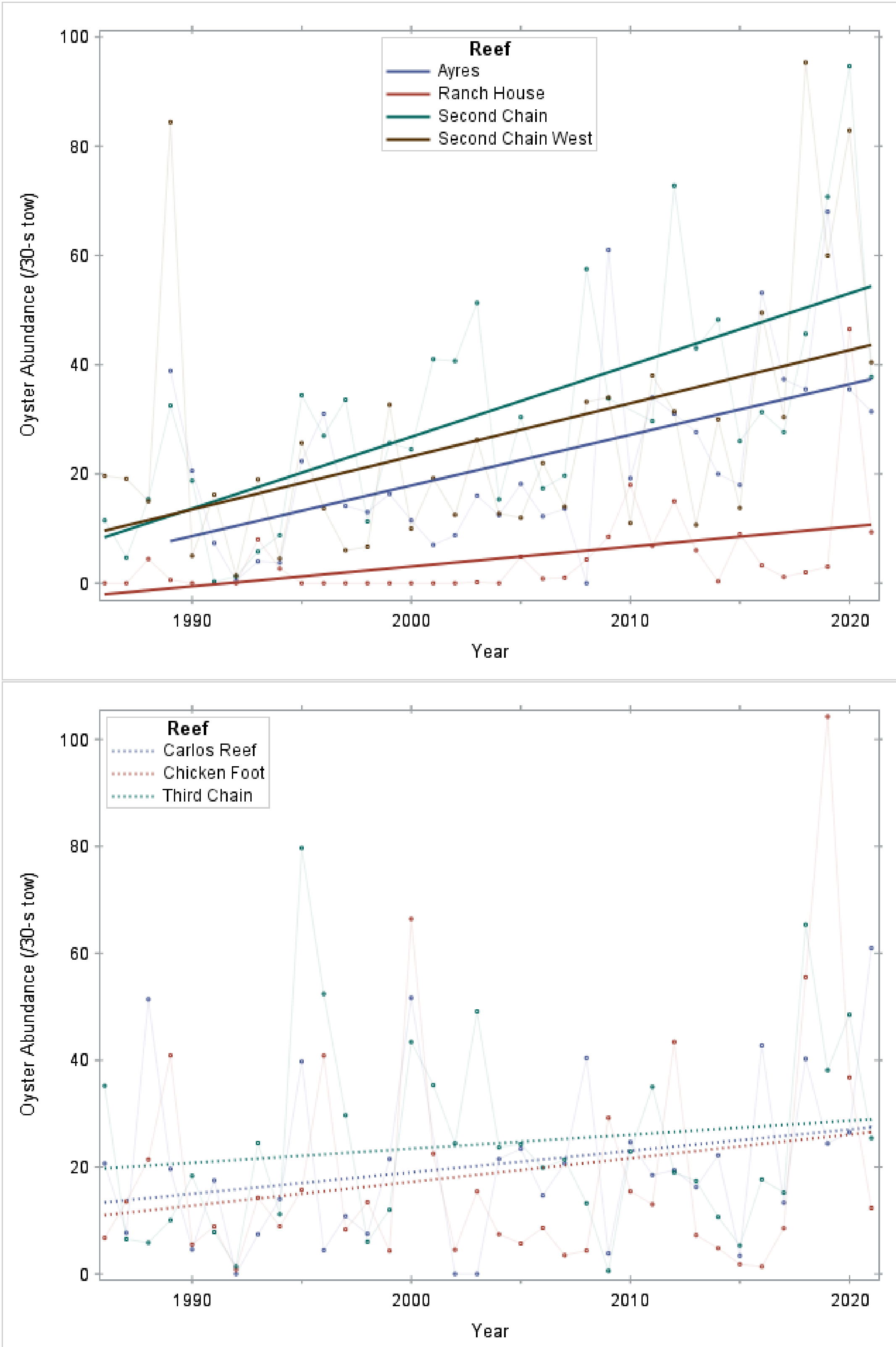

Figure 12. Linear regression of oyster abundance at each reef over time.
Solid regression lines indicate $p_{Spearman} < 0.05$.

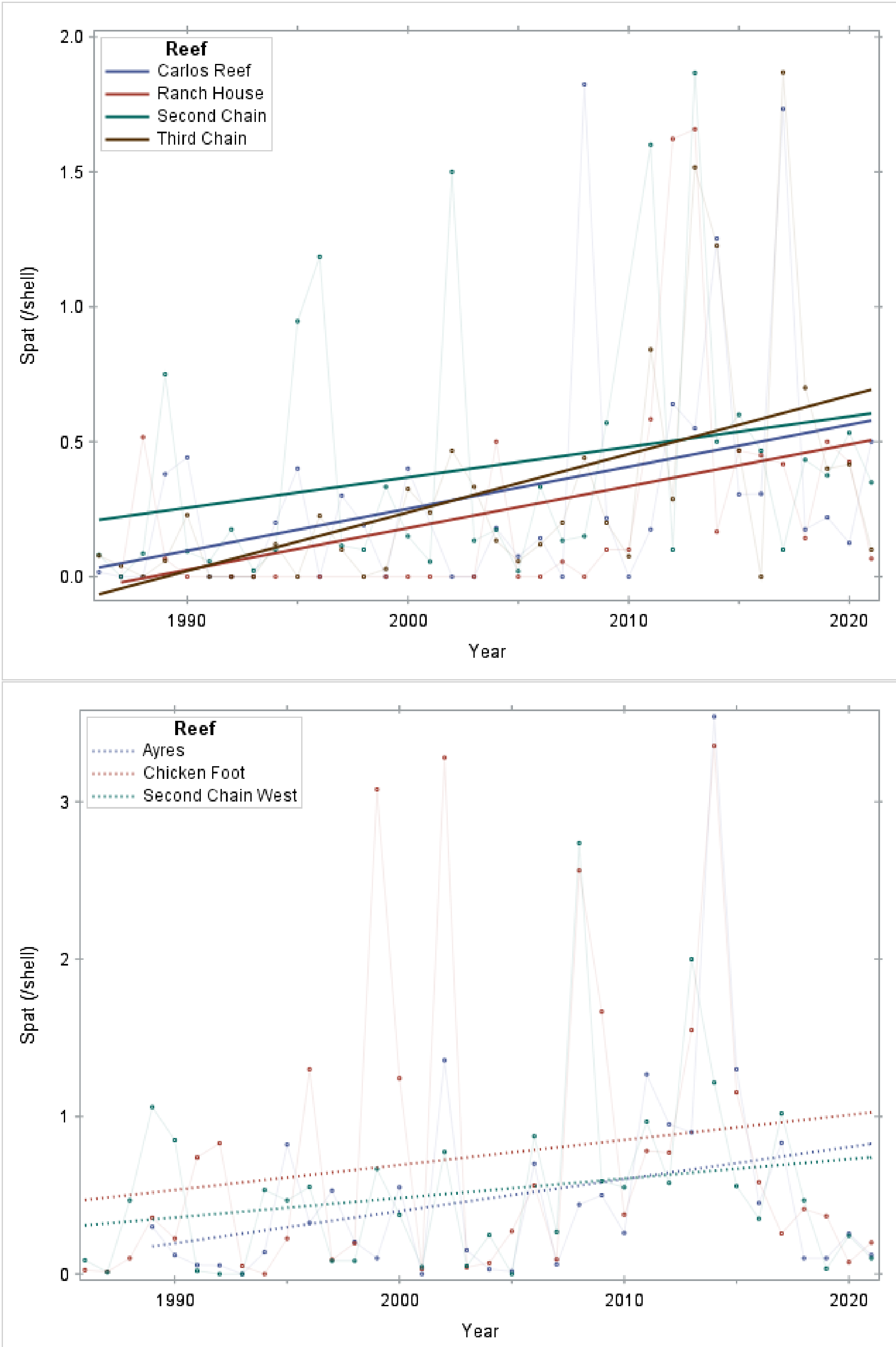

Figure 13. Linear regression of spat abundance at each reef over time. Solid regression lines indicate $p_{Spearman} < 0.05$.

Spat abundance was positively correlated with water temperature and salinity when comparing all reef-year combinations ($R \geq 0.23$, $p \leq 0.0003$) but negatively correlated with turbidity, total depth, and possibly dissolved oxygen concentration ($R \leq -0.12$, $p \leq 0.06$; Table 8). Live oyster abundance is positively correlated with salinity ($R = 0.18$, $p \leq 0.006$) but negatively correlated with total depth, and possibly turbidity ($R \leq -0.12$, $p \leq 0.05$). Oyster height is only correlated with salinity ($R = -0.19$, $p = 0.005$).

Table 8. Spearman correlations among oyster and water quality variables (TPWD data). Bolded numbers indicate correlations with $p < 0.05$. Grayed, bolded numbers indicate correlations with $p < 0.10$.

	Abundance (/tow)			Height (mm)			Spat abundance (/oyster)		
Water quality variable	n	r	p	n	r	p	n	r	p
Temperature (°C)	247	0.02	0.7303	229	-0.06	0.3643	**241**	**0.23**	**0.0003**
Dissolved Oxygen (mg/l)	247	-0.08	0.2286	229	0.03	0.6917	**241**	**-0.12**	**0.0607**
Salinity	**247**	**0.18**	**0.0056**	**229**	**-0.19**	**0.0048**	**241**	**0.40**	**<0.0001**
Turbidity	**247**	**-0.12**	**0.0531**	229	0.10	0.1317	**241**	**-0.25**	**<0.0001**
Sample Depth (m)	**247**	**-0.19**	**0.0023**	229	0.06	0.3841	**241**	**-0.14**	**0.0277**

Table 9. Spearman correlations between oyster and water quality variables and year, with a partial correlation with depth (TPWD data).

Reef	Abundance (/tow)		Height (mm)		Spat abundance (/oyster)		Temperature (°C)		Dissolved Oxygen (mg/l)		Salinity		Turbidity	
	r	p	r	p	r	p	r	p	r	p	r	p	r	p
Ayres	**0.62**	**0.0002**	0.01	0.9731	0.35	0.0506	0.11	0.5611	-0.01	0.9765	0.23	0.2090	-0.04	0.8375
Carlos Reef	**0.31**	**0.0893**	-0.07	0.7048	**0.42**	**0.0195**	-0.06	0.7608	0.07	0.7023	0.01	0.9404	**-0.39**	**0.0311**
Chicken Foot	0.01	0.9442	0.07	0.7010	**0.36**	**0.0313**	**0.41**	**0.0151**	**-0.52**	**0.0015**	0.10	0.5813	-0.27	0.1116
Ranch House	0.30	0.1896	-0.29	0.2105	**0.54**	**0.0110**	0.02	0.9287	0.05	0.8462	0.01	0.9619	-0.27	0.2394
Second Chain	**0.63**	**<.0001**	-0.19	0.2859	**0.42**	**0.0129**	**0.37**	**0.0303**	**-0.47**	**0.0049**	0.13	0.4752	-0.11	0.5452
Second Chain West	**0.52**	**0.0015**	-0.33	0.0510	0.29	0.0960	0.14	0.4300	-0.25	0.1427	0.28	0.0993	-0.20	0.2425
Third Chain	0.20	0.2605	0.06	0.7337	**0.60**	**0.0001**	**0.38**	**0.0261**	0.12	0.4872	-0.04	0.8371	**-0.35**	**0.0404**

Discussion

The TPWD oyster dredge dataset is useful for determining long-term trends of oyster communities in areas that can be dredged. The information on oyster communities from this chapter is most useful for managing oyster harvest because the majority of, if not all, oyster harvesting occurs by dredge in the Mesquite Bay complex.

There was an obvious long-term increase in oyster and spat abundances at several harvestable reefs in the Mesquite Bay complex from 1986 to 2021. Interestingly, there was also a decrease in sampling depth during the same period at many reefs. This decrease in sampling depth means that either the reef areas are getting shallower, or the sampling has moved to shallower areas of the reefs. It is expected that a decrease in sampling depth over time results in sampling more oysters because depth is negatively correlated with oyster and spat abundances.

The mean oyster abundance in the Mesquite Bay complex from 2012 to 2021 was 32 oysters/tow (reef means of 10 to 50 oysters/tow), which is greater than 83% of monthly means in Copano and Aransas Bays in 2007 and 2008, although the mean oyster height in most months in Copano and Aransas Bays was greater (mostly >70 mm; Beseres Pollack et al. 2011) than the 10-year mean in Mesquite Bay (60 mm). This indicates that the Mesquite Bay complex reefs have held healthy stocks of oysters in the recent past.

Abundances were lesser in 2021 than the 2012 to 2021 average at all seven reefs sampled, except for Carlos Reef, where abundances increased. This observation should be interpreted with caution because the sampling design was not created to make short-term comparisons. However, it does give an indication of a recent decrease in oyster abundances after a long period of increasing abundances.

Changes in Physical Reef Structure

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Methods

Oyster reefs in historic photos were delineated by tracing reef outlines as polygons from the various photographic layers, allowing for quantitative analyses of changes in reef size and shape following the general methods of Garvis et al. (2020). The study area was divided into five reef areas: Carlos Reef, Cedar Reef, Third Chain of Islands, Ayres Reef, and Second Chain of Islands (Figure 14). Over 20 data sources from 1860–2020 with varying spatial and spectral resolutions were investigated by ranking their useability in mapping intertidal and subtidal environments as good, intermediate, and poor (Table 10). Due to waves, turbidity, glare/glint, the presence of boats, and varying tide levels, we were only able to map the subtidal environment for three years (2004, 2009, and 2016). Figure 15 illustrates why subtidal mapping was not feasible in all years while Figure 16 – Figure 22 illustrate the differences between subtidal reefs for 2004, 2009, and 2016. However, additional years were useful in mapping intertidal reefs (Table 10). Reef structure was analyzed on three time-scales: 1) short-term (2004–2009–2016), 2) intermediate-term (1949–1972–1979–1996–2004–2009–2016), and 3) long-term (1949–2004).

Short-term reef changes illustrate changes in intertidal and subtidal areal extent among 2004, 2009, and 2016. Intermediate-term and long-term reef changes examine changes in intertidal reef extent only (no subtidal). All data for short-term and intermediate-term reef changes came from digitizing aerial photography (SARA 2022, TNRIS 2022, USDA 2022, USGS 2022), whereas data for long-term reef analyses came from pre-existing data; 1949 NOAA T-Sheets (NOAA 2022) and NOAA's benthic habitat maps from 2004 (NOAA 2007) already had intertidal reefs digitized. We also conducted a comparison between NOAA's 2004 intertidal/subtidal reef extent and our 2004 intertidal/subtidal reef extent.

Table 10. All aerial imagery dates investigated and their usability for mapping intertidal reefs.
G = Good and used in imagery analysis; G = Good; I = Intermediate; P = Poor. RGB = Red-Green-Blue, RGBN = Reg-Green-Blue-Near Infrared, CIR = Color-Infrared, BW = Black-White.*

Year	Data Type	Composite Bands	Cell Size (m)	Carlos	Cedar	3rd Chain	Ayres	2nd Chain	Data Source
1860s	NOAA T-Sheet	BW	Not rectified	P	P	P	P	P	NOAA
1934/1935	NOAA T-Sheet	BW	Not rectified	P	P	P	P	P	NOAA
1949	NOAA T-Sheet	BW	3.5	G*	G*	G*	G*	G*	NOAA
1969	APSF	BW; CIR	Not rectified	P	P	P	P	P	USGS
1971	APSF	CIR	Not rectified	P	P	P	P	P	USGS
1972	APSF	CIR	0.66; 2	I	I	I	G*	G*	USGS
1979	APSF	BW	2	G*	G*	I	G*	I	USGS
1982	APSF	CIR	Not rectified	P	P	P	P	P	USGS
1996	TOP	CIR	1	G*	G*	G*	G*	I	TNRIS
2004	NAIP	RGBN	1	G*	G*	G*	G*	G*	USDA
2005	NAIP	RGB	2	I	I	P	I	I	USDA
2008	NAIP	CIR	0.5	P	I	I	P	P	USDA
2009	NAIP	RGBN	0.5	G*	G*	G*	G*	G*	USDA
2010	NAIP	RGBN	1	P	I	P	P	P	USDA
2012	NAIP	RGBN	1	G	I	P	P	P	USDA
2014	NAIP	RGB	1	P	G	G	G	G	USDA
2015	TOP	RGBN	0.5	I	G	I	P	I	TNRIS
2016	NAIP	RGB	1	P	P	P	P	P	USDA
2016	SARA	RGBN	0.3	G*	G*	G*	G*	G*	SARA
2018	NAIP	RGBN	0.6	I	I	P	I	P	USDA
2020	NAIP	RGB	0.6	P	P	P	P	P	USDA

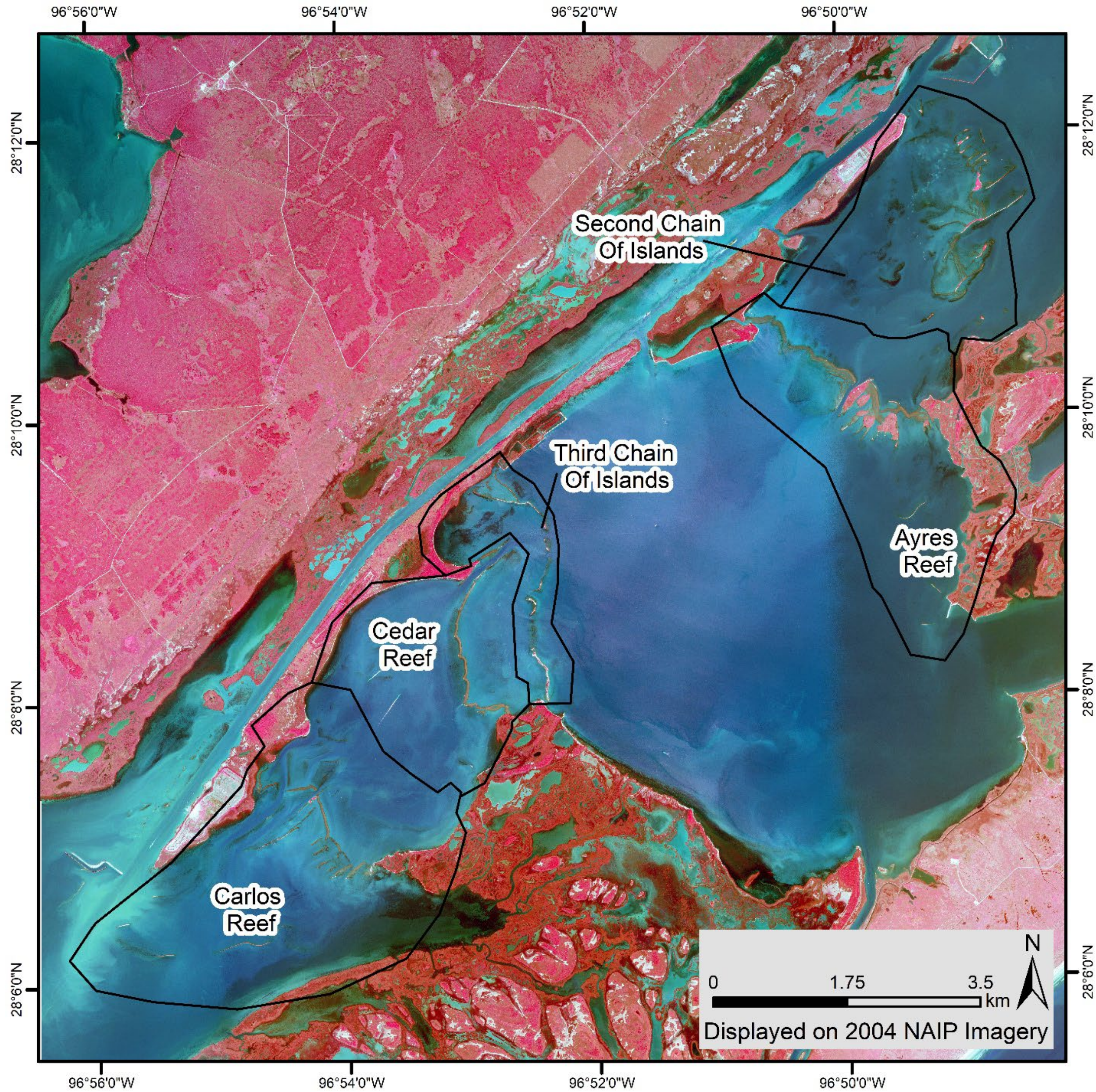

Figure 14. The study area as divided into five reef areas for change in areal extent analyses.

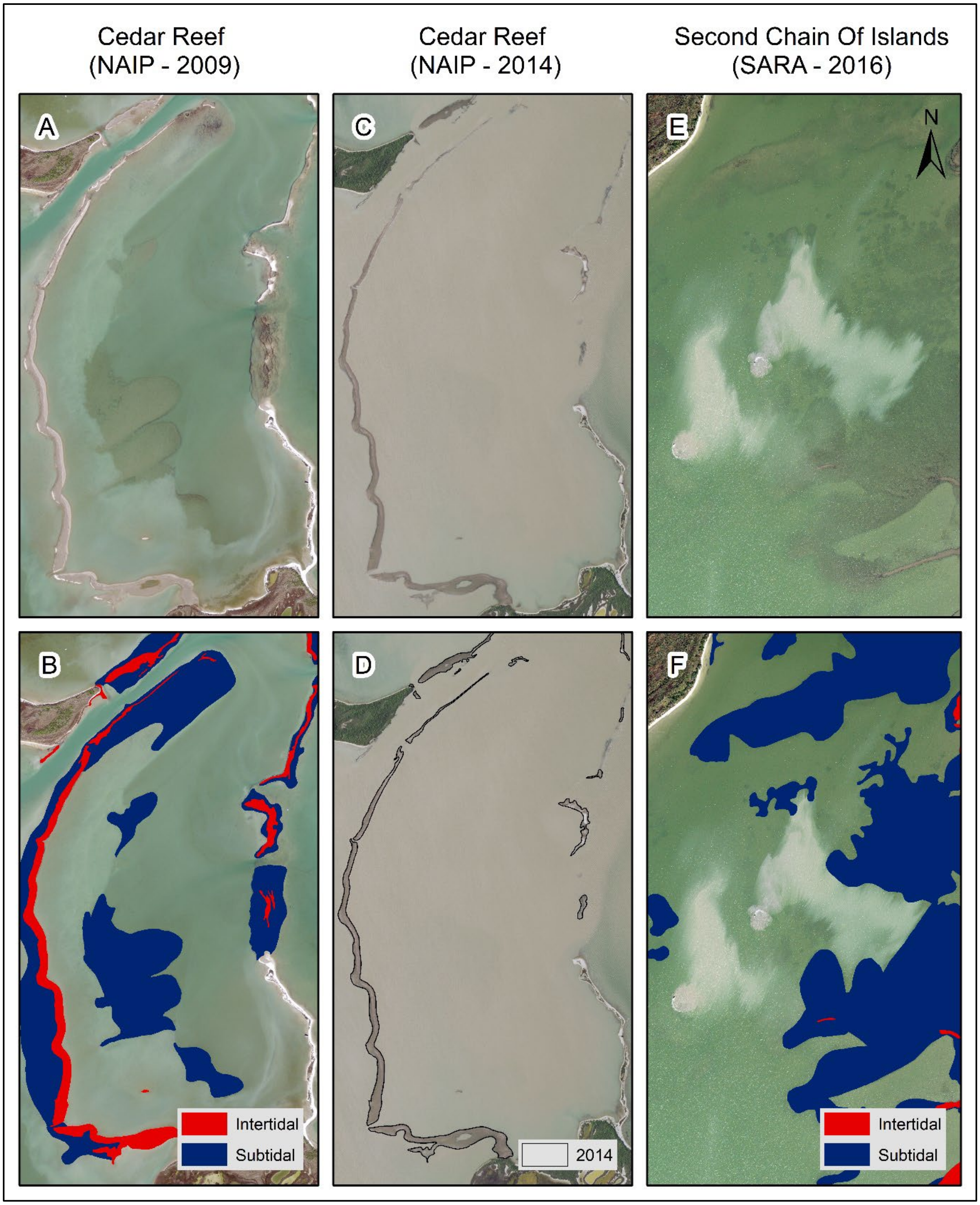

Figure 15. Imagery examples where subtidal reefs are and are not present.
A) 2009 NAIP imagery focused on Cedar Reef & Third Chain of Islands. This is a "good" imagery source where intertidal and subtidal reefs were distinguished, B) Digitized reef extent based on 2009 NAIP imagery, C) 2014 NAIP imagery focused on Cedar Reef & Third Chain of Islands. This imagery was "good" for mapping intertidal but "poor" for mapping subtidal. D) Digitized reef extent based on 2014 NAIP imagery, E) 2016 SARA imagery focused on Second Chain of Islands. This is a "good" imagery source where boats interfered with subtidal reef identification. F) Digitized reef extent based on 2016 SARA imagery.

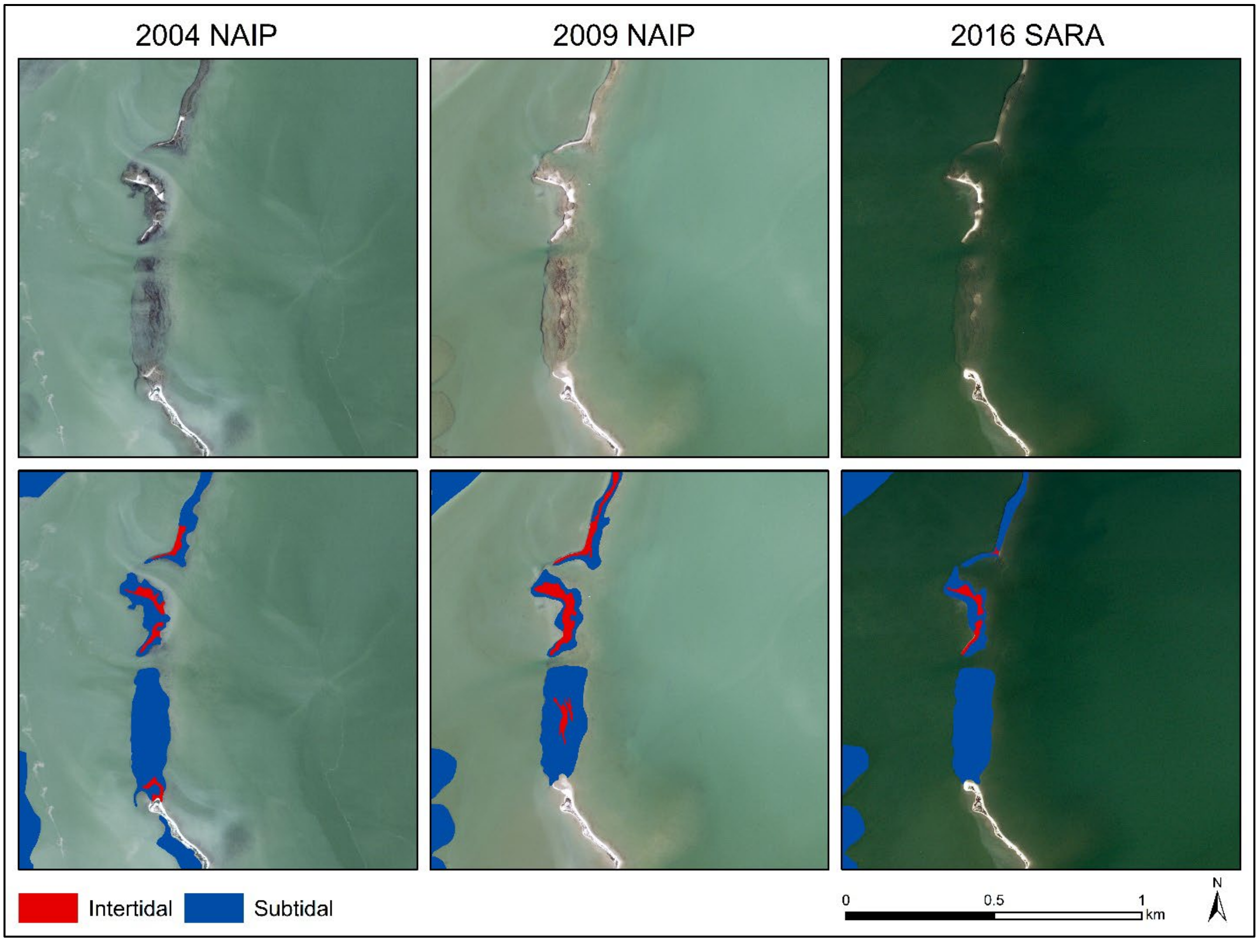

Figure 16. Differences between imagery used for Short-Term Reef Changes for Third Chain of Islands. The top row displays the imagery for a specific year (2004, 2009, and 2016) while the bottom row displays the digitized and classified landcovers for each year.

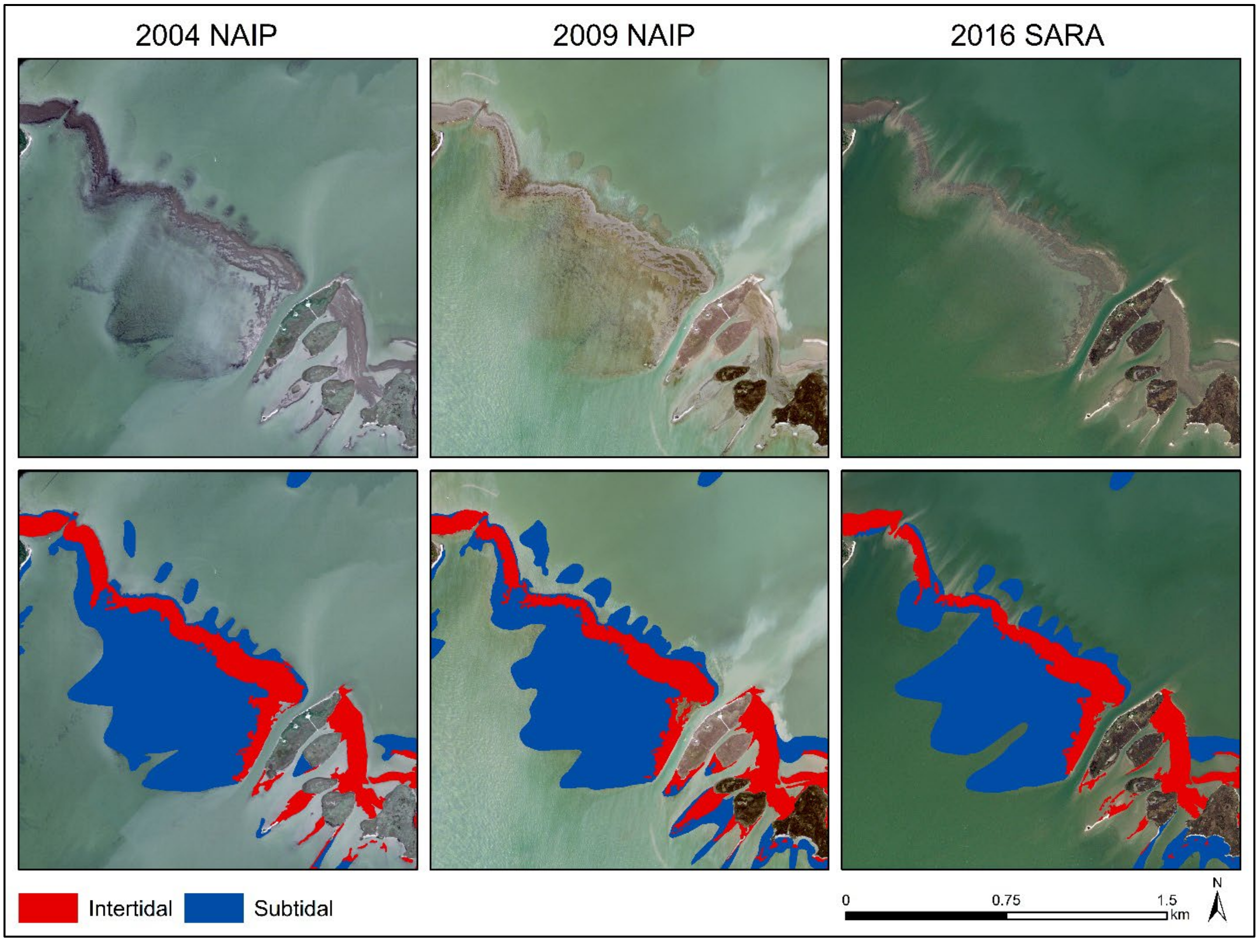

Figure 17. Differences between imagery used for Short-Term Reef Changes for all of Ayres Reef. The top row displays the imagery for a specific year (2004, 2009, and 2016) while the bottom row displays the digitized and classified landcovers for each year.

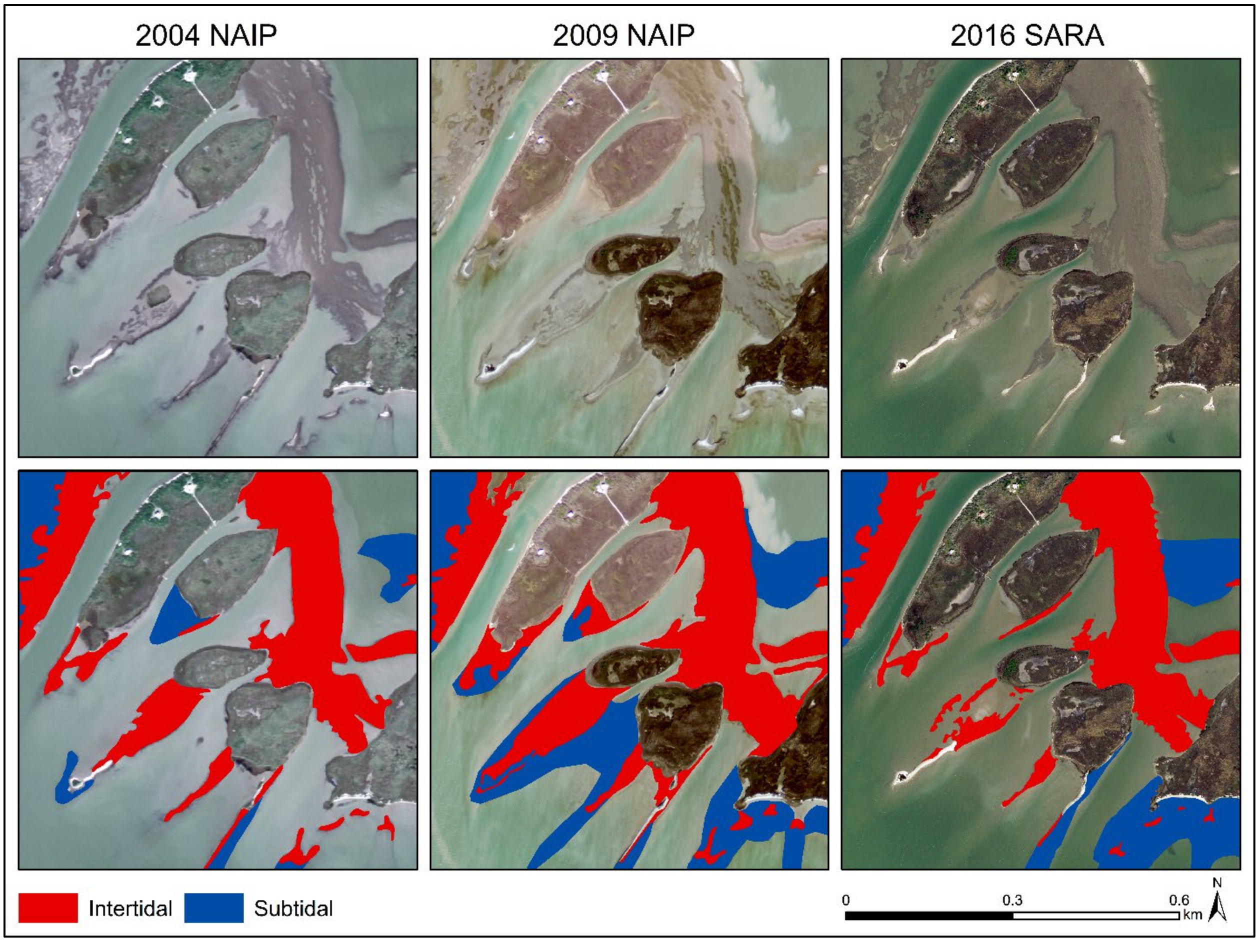

Figure 18. Differences between imagery used for Short-Term Reef Changes for a southeastern area of Ayres Reef. The top row displays the imagery for a specific year (2004, 2009, and 2016) while the bottom row displays the digitized and classified landcovers for each year.

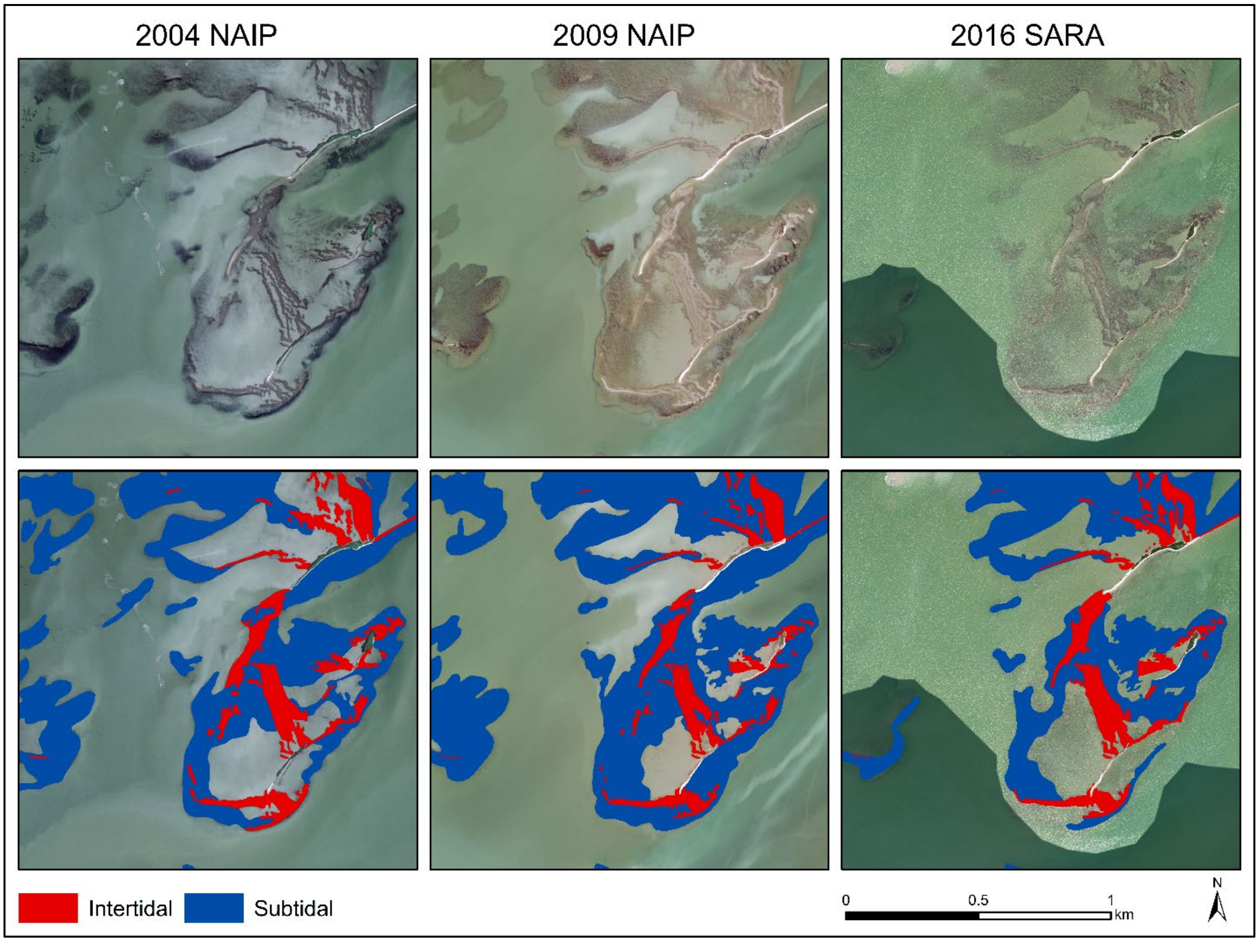

Figure 19. Differences between imagery used for Short-Term Reef Changes for Second Chain of Islands. The top row displays the imagery for a specific year (2004, 2009, and 2016) while the bottom row displays the digitized and classified landcovers for each year.

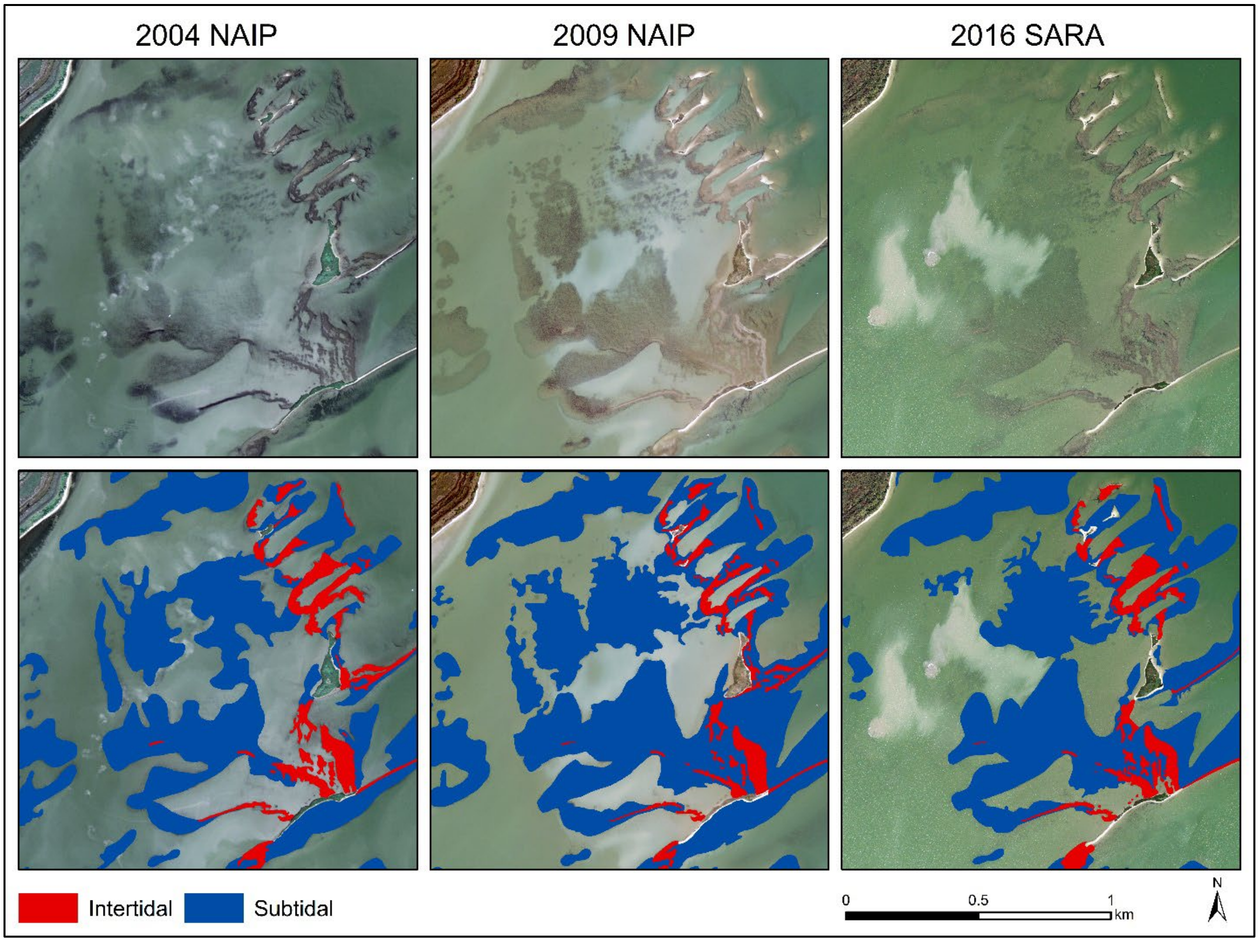

Figure 20. Differences between imagery used for Short-Term Reef Changes for Second Chain of Islands.
The top row displays the imagery for a specific year (2004, 2009, and 2016) while the bottom row displays the digitized and classified landcovers for each year.

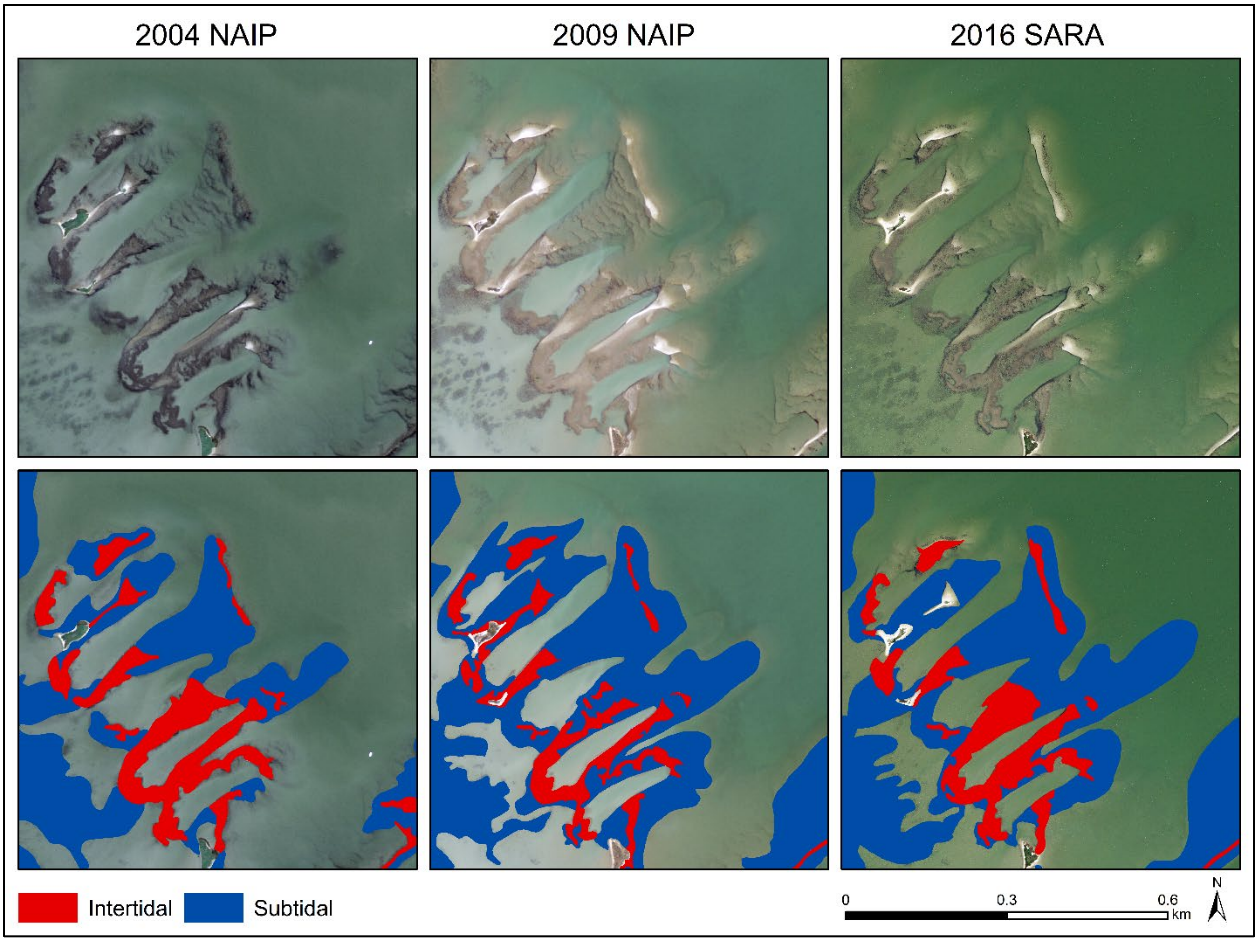

Figure 21. Differences between imagery used for Short-Term Reef Changes for a northern area within Second Chain of Islands. The top row displays the imagery for a specific year (2004, 2009, and 2016) while the bottom row displays the digitized and classified landcovers for each year.

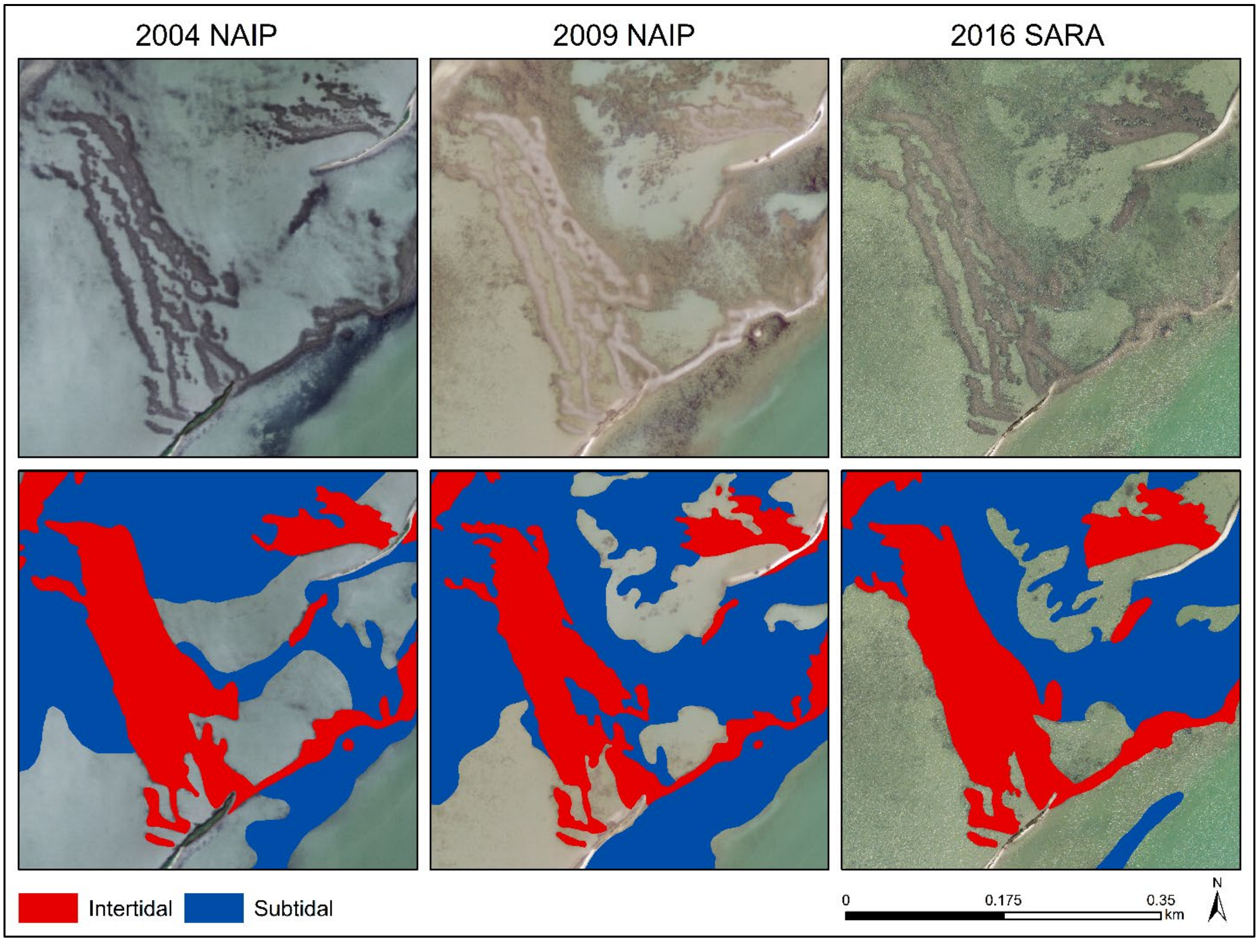

Figure 22. Differences between imagery used for Short-Term Reef Changes for a southern area within Second Chain of Islands. The top row displays the imagery for a specific year (2004, 2009, and 2016) while the bottom row displays the digitized and classified landcovers for each year.

Short-Term Reef Changes (2004–2009–2016)

The intertidal reefs were manually digitized for each aerial image (Table 10) within ArcMap 10.8 (Environmental Systems Research Institute, Inc., Redlands, CA) at a scale of 1:800. Since the subtidal reefs were not as clear, we had to zoom out, mapping the reefs at various scales ranging from 1:800 – 1:5,000. After digitization was complete, we 1) smoothed the data using the "Smooth" tool with 2m Peak to get rid of jagged edges, and 2) checked for topological errors to ensure there were no overlapping features. After digitization, we used the “Union” tool to combine sequential years together, allowing us to investigate specific transitions including: 1) Intertidal to Intertidal, 2) Intertidal to Subtidal, 3) Subtidal to Subtidal, 4) Subtidal to Intertidal, 5) Intertidal to No Data, 6) Subtidal to No Data, 7) No Data to Intertidal, and 8) No Data to Subtidal.

Intermediate-Term Reef Changes (1949–1972–1979–1996–2004–2009–2016)

Various data sources from both short-term and long-term reef changes as well as new sources were used in this analysis. We extracted the intertidal reefs from all sources used to conduct short-term reef changes (2004, 2009, and 2016). We also incorporated the 1949 intertidal reefs from long-term reef changes. Additional sources unique to intermediate-term reef changes consisted of 1972 Aerial Photo Single Frames (APSF), 1979 Aerial Photo Single Frames (APSF), and 1996 Texas Orthoimagery Program (TOP). However, not all years were useful in classifying each reef (Table 10). For example, Third Chain of Islands did not have good imagery in 1972 and 1979 while Second Chain of Islands did not have good imagery in 1979 and 1996. 1972 and 1979 imagery tiles were rectified (RMSE under 3m) using 1st order, 2nd order, Spline, or Adjust Transformations and mosaiced together to create one uniform image. We then manually digitized the intertidal reefs using the methodology presented in short-term reef changes.

Long-Term Reef Changes (1949–2004)

Two data sources were used to conduct a long-term study of reef changes. The first source consisted of 1949 NOAA T-Sheets (NOAA 2022). 1949 aerial photos were taken on 21 November 1946 and 9 December 1948, and field surveys were conducted in April 1949. Reefs were already outlined on the 1949 T-sheets (Figure 23), however there was no GIS file available for download. To get this data in a usable format, we rectified the two NOAA T-Sheets and traced the reefs at a scale of 1:5,000. The second source consisted of 2004 NOAA benthic habitat data (Figure 24) (NOAA 2007). According to metadata, NOAA classified benthic habitats below mean high water for various bay systems using 2004 NAIP imagery and the classifications were refined after conducting fieldwork in 2006 and 2007. This source differentiated reef from shells/shell hash, however for the purpose of comparison with 1949 data, we did not include shell/shell hash.

Comparing 2004 NOAA and HRI Digitizations

Lastly, we conducted a comparison between NOAA's 2004 intertidal/subtidal reef extent (NOAA 2007) and HRI's 2004 intertidal/subtidal reef extent. We used to "Union" tool to combine and compare both datasets to look for the following similarities and differences in the classifications: 1) Intertidal versus Intertidal, 2) Intertidal versus Subtidal, 3) Subtidal versus Subtidal, 4) Subtidal versus Intertidal, 5) Intertidal versus No Data, 6) Subtidal versus No Data, 7) No Data versus Intertidal, and 8) No Data versus Subtidal.

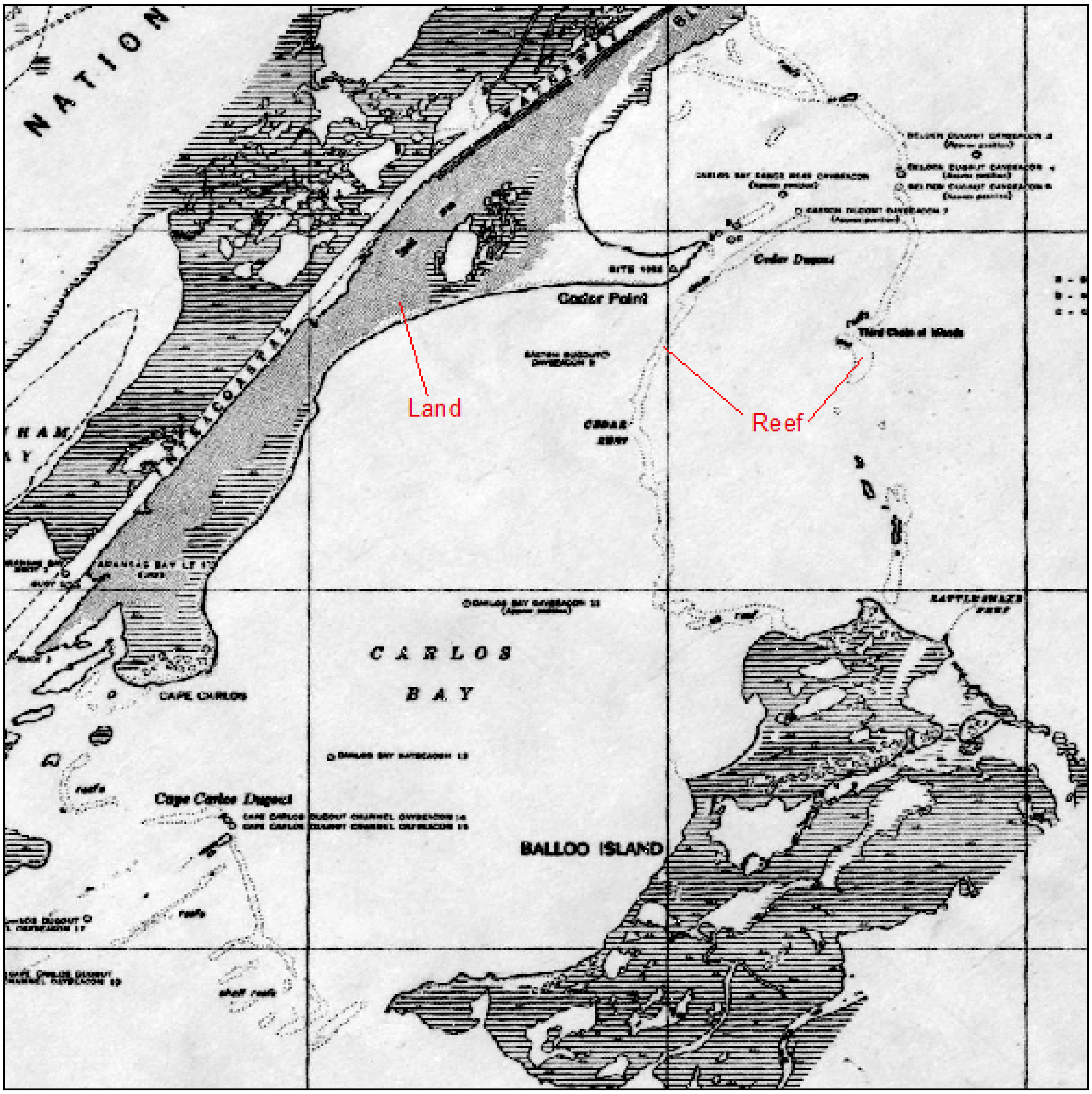

Figure 23. 1949 NOAA T-Sheet focused on Carlos Reef, Cedar Reef, and Third Chain of Islands depicting the difference between intertidal reef and land.

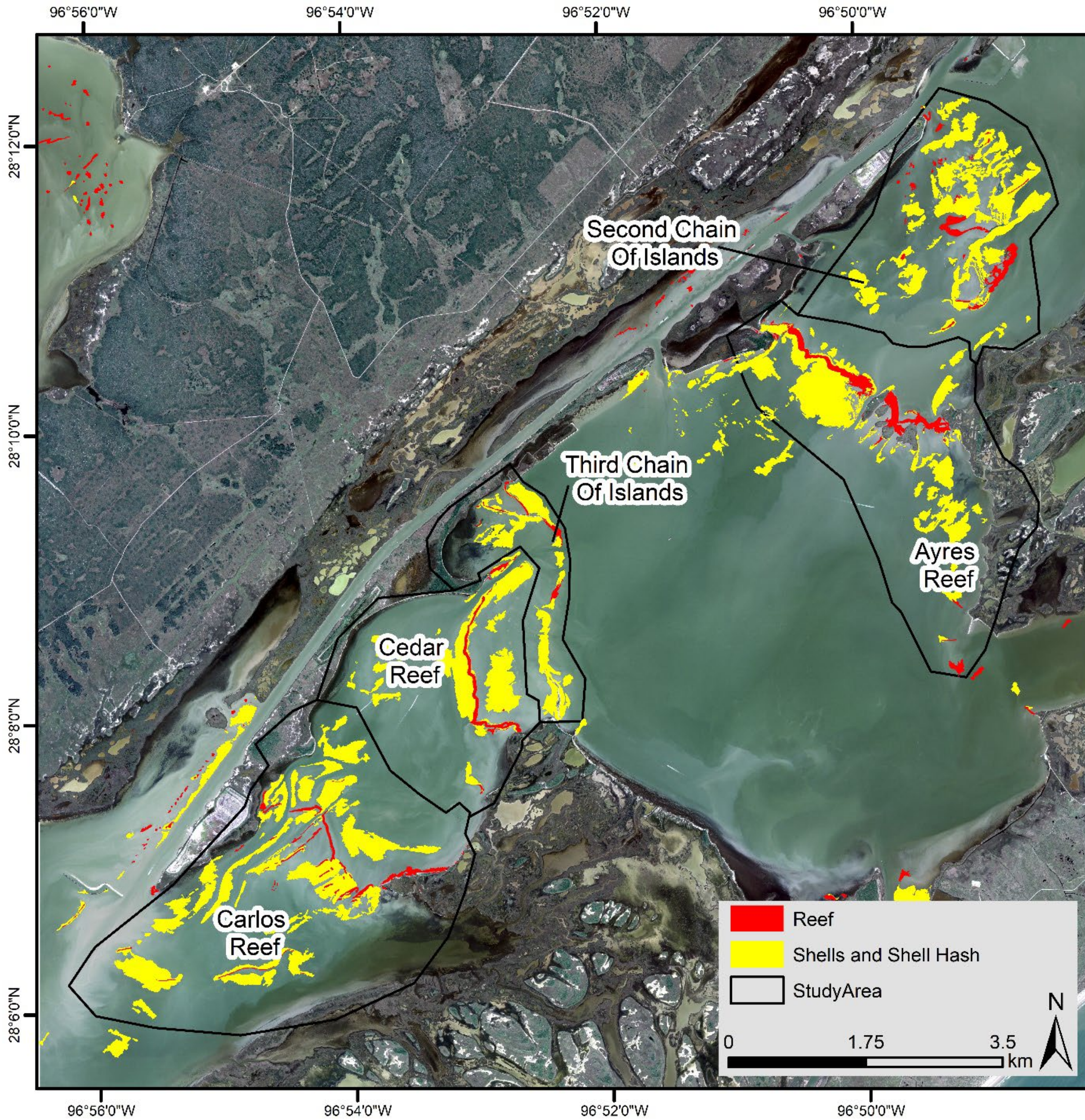

Figure 24. 2004 NOAA Benthic Habitat Data illustrating the difference between reef and shell/shell hash extent.

Results

Short-Term Reef Changes (2004–2009–2016)

When looking at all five reefs combined, from 2004–2016, there was a net decline in intertidal and subtidal reefs (Table 11). When looking at each reef individually, all reefs experienced a net decline in intertidal reefs from 2004–2016, however, between individual timesteps (2004–2009 and 2009–2016), there were instances of increased intertidal reef coverage (Table 11). Additionally, while there was a net decrease in total subtidal reefs, Carlos Reef and Cedar Reef experienced a net increase while Third Chain of Islands, Ayres Reef, and Second Chain of Islands experienced a net decrease (Table 11). We also looked at specific land cover transitions through time (1) Intertidal to Intertidal, 2) Intertidal to Subtidal, 3) Subtidal to Subtidal, 4) Subtidal to Intertidal, 5) Intertidal to No Data, 6) Subtidal to No Data, 7) No Data to Intertidal, and 8) No Data to Subtidal) and Table 12 represents their area of change (ac). Since the reefs vary in size, knowing the area of change is not as useful as knowing the percentage of change between transitions that occurred. Table 13 represents the percentage of change between land covers. Intertidal to No Data and No Data to Intertidal was negligible, only accounting for between 0.1–2.3% and 0.1–3.8% of the total transitions occurring between 2004–2009 and 2009–2016, respectively (Table 13). Conversely, Subtidal to No Data and No Data to Subtidal accounted for 4.6–42.3% and 4.6–40.5% of the total transitions occurring between 2004–2009 and 2009–2016. The variability in subtidal environments could be attributed to waves, turbidity, glare/glint, the presence of boats, and varying tide levels interfering with seeing the true subtidal reef extent. The large amount of Subtidal to No Data and No Data to Subtidal is evident in Figure 25.

Table 11. Short-Term: Area (ac) of intertidal and subtidal reefs as well as the net change in oyster coverage between timeframes.

Intertidal					
	2004	**2009**	**2016**	**Net Change 2004–2009**	**Net Change 2009–2016**
Carlos Reef	29.1	35.0	24.2	5.9	-10.8
Cedar Reef	29.5	29.4	23.5	-0.2	-5.9
Third Chain of Islands	8.7	11.0	4.2	2.3	-6.8
Ayres Reef	87.5	86.3	76.9	-1.2	-9.4
Second Chain of Islands	54.1	44.5	45.2	-9.6	0.7
Total	**208.9**	**206.1**	**173.9**	**-2.8**	**-32.2**
Subtidal					
	2004	**2009**	**2016**	**Net Change 2004–2009**	**Net Change 2009–2016**
Carlos Reef	375.3	545.3	394.4	170.0	-150.9
Cedar Reef	168.6	148.2	289.0	-20.4	140.8
Third Chain of Islands	62.4	33.1	30.3	-29.3	-2.8
Ayres Reef	353.6	279.7	222.3	-73.9	-57.4
Second Chain of Islands	322.0	432.8	282.3	110.8	-150.5
Total	**1281.9**	**1439.1**	**1218.3**	**157.2**	**-220.7**

Table 12. Short-Term: Area (ac) of change between sequential timeframes.
Ex: Amount of area classified as Intertidal in 2004 that was classified as No Data, Intertidal, or Subtidal in 2009

			2009		
			No Data	Intertidal	Subtidal
2004	Carlos Reef	No Data	-	3.0	298.4
		Intertidal	1.0	26.1	2.0
		Subtidal	124.6	5.9	244.8
	Cedar Reef	No Data	-	0.9	42.5
		Intertidal	0.6	27.7	1.3
		Subtidal	63.4	0.8	104.4
	Third Chain of Islands	No Data	-	0.6	11.7
		Intertidal	1.9	6.3	0.5
		Subtidal	37.4	4.2	20.8
	Ayres Reef	No Data	-	7.0	63.2
		Intertidal	3.4	76.6	7.5
		Subtidal	141.9	2.6	209.1
	Second Chain of Islands	No Data	-	1.6	123.4
		Intertidal	0.9	39.5	13.7
		Subtidal	22.9	3.4	295.7

			2016		
			No Data	Intertidal	Subtidal
2009	Carlos Reef	No Data	-	0.9	135.7
		Intertidal	7.7	21.4	5.9
		Subtidal	290.6	1.8	252.9
	Cedar Reef	No Data	-	0.1	156.7
		Intertidal	0.5	22.7	6.2
		Subtidal	21.4	0.6	126.1
	Third Chain of Islands	No Data	-	0.6	7.9
		Intertidal	2.0	3.6	5.5
		Subtidal	16.1	0.1	16.9
	Ayres Reef	No Data	-	3.6	43.3
		Intertidal	10.2	70.4	5.7
		Subtidal	103.5	2.9	173.3
	Second Chain of Islands	No Data	-	1.7	23.4
		Intertidal	2.6	34.3	7.6
		Subtidal	172.2	9.2	251.3

Table 13. Short- Term: Percentage of change for each transition between sequential timeframes.
Ex: Amount of area classified as Intertidal in 2004 that was classified as No Data, Intertidal, or Subtidal in 2009

			2009		
			No Data	Intertidal	Subtidal
2004	Carlos Reef	No Data	-	0.4	42.3
		Intertidal	0.1	3.7	0.3
		Subtidal	17.7	0.8	34.7
	Cedar Reef	No Data	-	0.4	17.6
		Intertidal	0.2	11.5	0.5
		Subtidal	26.2	0.3	43.2
	Third Chain of Islands	No Data	-	0.7	14.1
		Intertidal	2.3	7.5	0.6
		Subtidal	44.8	5.0	25.0
	Ayres Reef	No Data	-	1.4	12.4
		Intertidal	0.7	15.0	1.5
		Subtidal	27.8	0.5	40.9
	Second Chain of Islands	No Data	-	0.3	24.6
		Intertidal	0.2	7.9	2.7
		Subtidal	4.6	0.7	59.0

			2016		
			No Data	Intertidal	Subtidal
2009	Carlos Reef	No Data	-	0.1	18.9
		Intertidal	1.1	3.0	0.8
		Subtidal	40.5	0.3	35.3
	Cedar Reef	No Data	-	0.0	46.9
		Intertidal	0.2	6.8	1.8
		Subtidal	6.4	0.2	37.7
	Third Chain of Islands	No Data	-	1.1	15.0
		Intertidal	3.8	6.8	10.5
		Subtidal	30.7	0.1	32.2
	Ayres Reef	No Data	-	0.9	10.5
		Intertidal	2.5	17.0	1.4
		Subtidal	25.1	0.7	42.0
	Second Chain of Islands	No Data	-	0.3	4.6
		Intertidal	0.5	6.8	1.5
		Subtidal	34.3	1.8	50.0

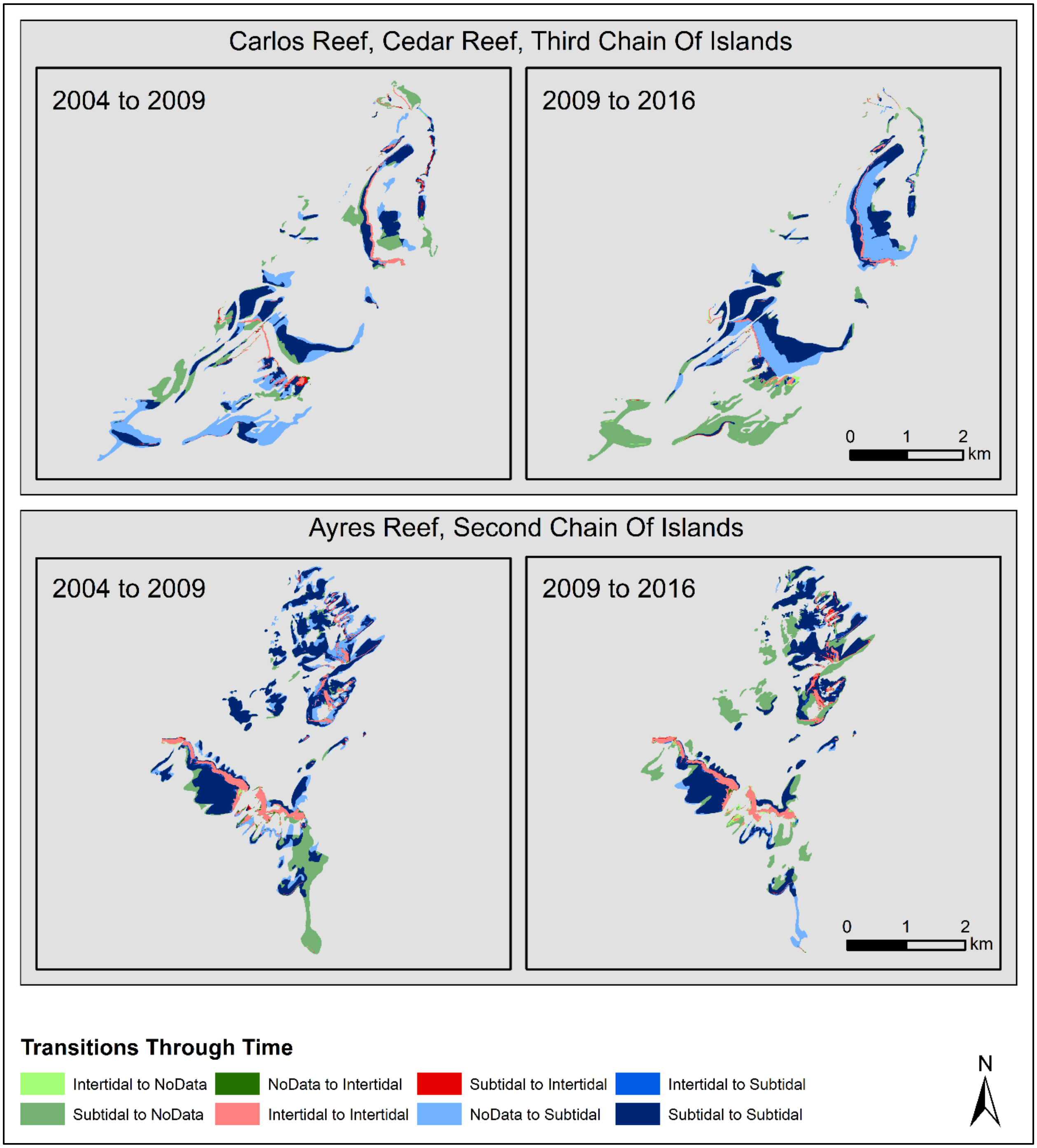

Figure 25. Short-Term Changes Between Reef Habitats emphasizing the large amount of Subtidal to No Data, and No Data to Subtidal.

Intermediate-Term Reef Changes (1949–1972–1979–1996–2004–2009–2016)

Carlos Reef, Cedar Reef, Third Chain of Islands, and Ayres Reef experienced a net loss of intertidal reefs while Second Chain of Islands remained the same from 1949 to 2016 (Figure 26 - Figure 31; Table 14). All reefs except Second Chain of Islands decreased from 1949–1996. Additionally, all reefs grew from 1996–2004 and Carlos Reef and Third Chain of Islands continued to grow from 2004–2009. Lastly, from 2009–2016, Second Chain of Islands remained the same while all other reefs decreased. Next, we intersected sequential timeframes to determine where reefs remained stable between timeframes and where reefs were lost or gained. The greatest amount of reef loss for sequential timeframes was observed over different timeframes for each reef: 1949–1972 (Ayres Reef and Second Chain of Islands), 1949–1979 (Carlos Reef and Cedar Reef), and 1949–1996 (Third Chain of Islands). Additionally, the greatest amount of reef loss was not always consistent with when the greatest net loss occurred. For example, in Carlos Reef, the greatest reef loss occurred from 1949–1979 (21.7 ac), however the net change was only a 1.6 ac loss because the reef also experienced a 20.1 ac growth of reefs in other areas. Similarly, the greatest net loss occurred from 1979–1996 (18.7 ac) because Carlos Reef lost 20.5 ac of reef and only gained 1.8 ac of reef. A Similar trend for Second Chain of Islands where the greatest net loss occurred from 2004–2009.

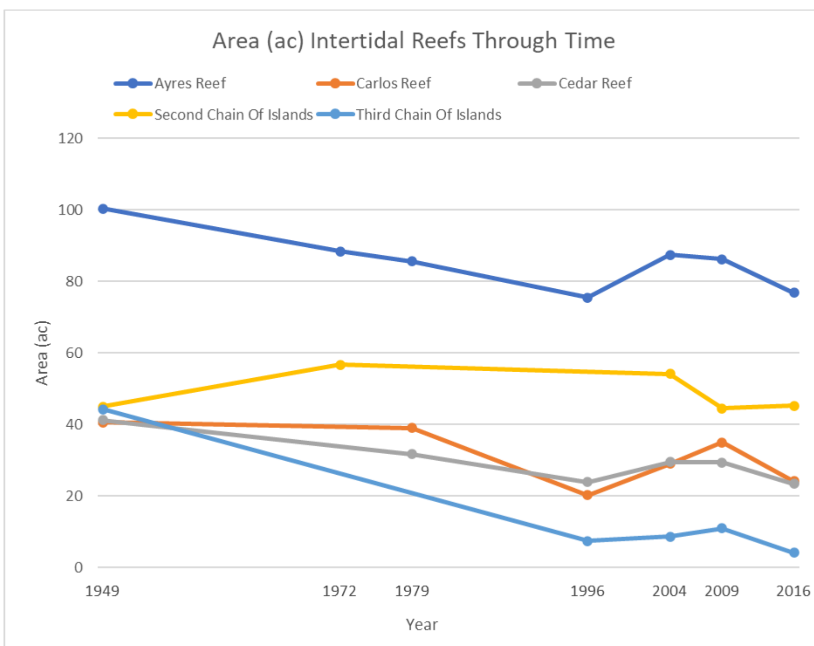

Figure 26. Intermediate-term intertidal reef area (ac) through time.

Table 14. Intermediate-Term: Area extent (ac) of intertidal oyster habitat as well as the amount of intertidal oyster habitat that was lost, gained, or remined the same between timeframes.
Cells highlighted in green represent the timeframe where the greatest net change in terms of loss occurred. Cells highlighted in grey represent the timeframe where the greatest loss occurred.

Carlos Reef						
Reef	Year 1	Year 2	Net Change	Area Reefs remained	Area Reefs lost since Year 1	Area Reefs gained since Year 2
1949 to 1979	40.6	39.0	-1.6	18.8	21.7	20.1
1979 to 1996	39.0	20.3	-18.7	18.5	20.5	1.8
1996 to 2004	20.3	29.1	8.8	18.3	2.0	10.8
2004 to 2009	29.1	35.0	5.9	26.1	3.0	8.9
2009 to 2016	35.0	24.2	-10.8	21.4	13.6	2.8
Cedar Reef						
Reef	Year 1	Year 2	Net Change	Area Reefs remained	Area Reefs lost since Year 1	Area Reefs gained since Year 2
1949 to 1979	41.3	31.7	-9.6	24.0	17.3	7.7
1979 to 1996	31.7	23.9	-7.8	20.2	11.5	3.7
1996 to 2004	23.9	29.5	5.6	22.5	1.4	7.0
2004 to 2009	29.5	29.4	-0.2	27.7	1.8	1.7
2009 to 2016	29.4	23.5	-5.9	22.7	6.7	0.8
Third Chain of Islands						
Reef	Year 1	Year 2	Net Change	Area Reefs remained	Area Reefs lost since Year 1	Area Reefs gained since Year 2
1949 to 1996	44.2	7.5	-36.8	4.6	39.6	2.8
1996 to 2004	7.5	8.7	1.2	5.0	2.5	3.8
2004 to 2009	8.7	11.0	2.3	6.3	2.4	4.8
2009 to 2016	11.0	4.2	-6.8	3.6	7.5	0.6
Ayres Reef						
Reef	Year 1	Year 2	Net Change	Area Reefs remained	Area Reefs lost since Year 1	Area Reefs gained since Year 2
1949 to 1972	100.3	88.4	-11.9	75.8	24.5	12.7
1972 to 1979	88.4	85.6	-2.8	75.6	12.8	10.0
1979 to 1996	85.6	75.5	-10.2	65.5	20.1	9.9
1996 to 2004	75.5	87.5	12.0	69.2	6.2	18.2
2004 to 2009	87.5	86.3	-1.2	76.6	10.8	9.6
2009 to 2016	86.3	76.9	-9.4	70.4	15.9	6.5
Second Chain of Islands						
Reef	Year 1	Year 2	Net Change	Area Reefs remained	Area Reefs lost since Year 1	Area Reefs gained since Year 2
1949 to 1972	45.0	56.7	11.7	19.7	25.4	37.0
1972 to 2004	56.7	54.1	-2.6	39.3	17.5	14.9
2004 to 2009	54.1	44.5	-9.6	39.5	14.6	4.9
2009 to 2016	44.5	45.2	0.7	34.3	10.2	10.9

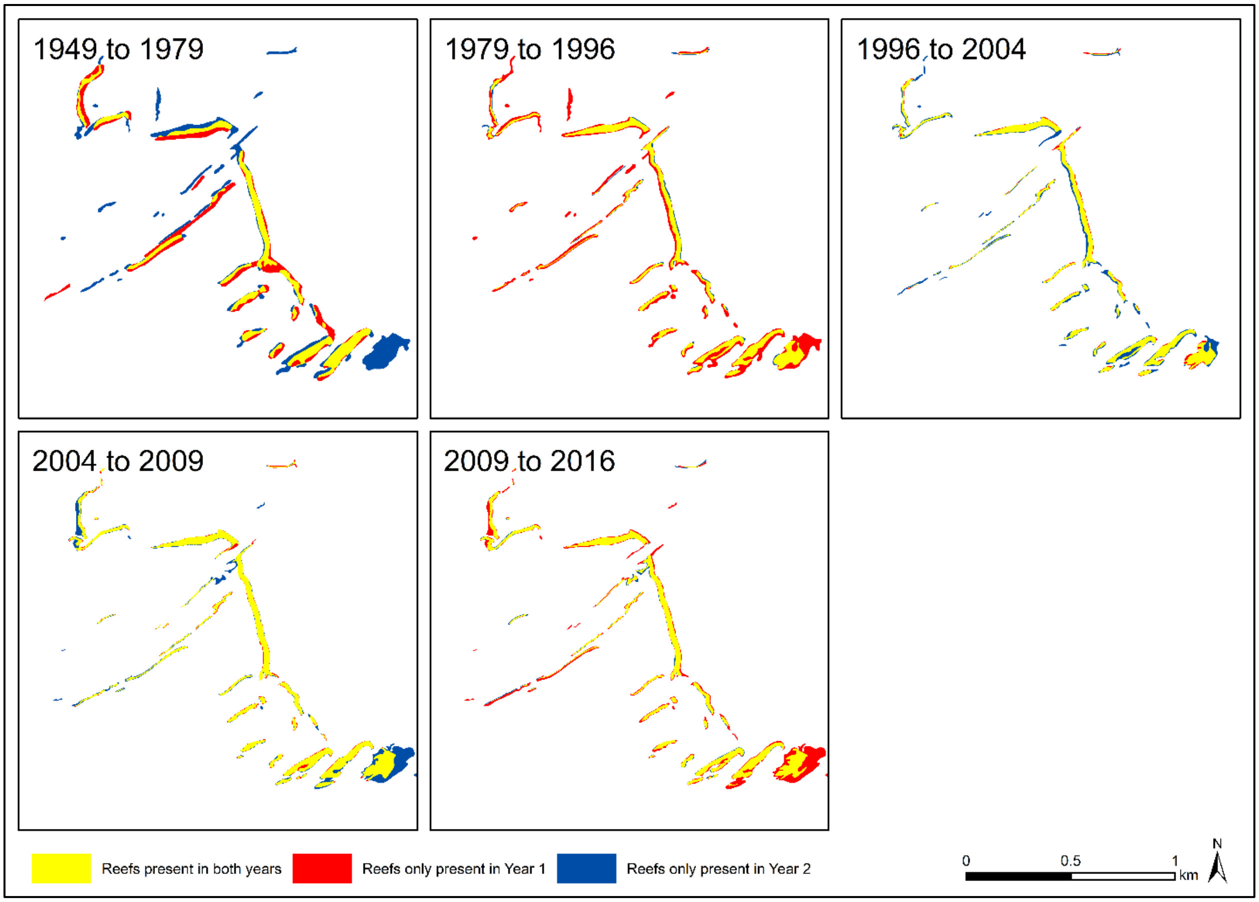

Figure 27. Intermediate-term reef extent for Carlos Reef.
This map illustrates where Intertidal Reefs were only mapped in year 1, where only mapped in year 2, and where mapped in both years for sequential timeframes.

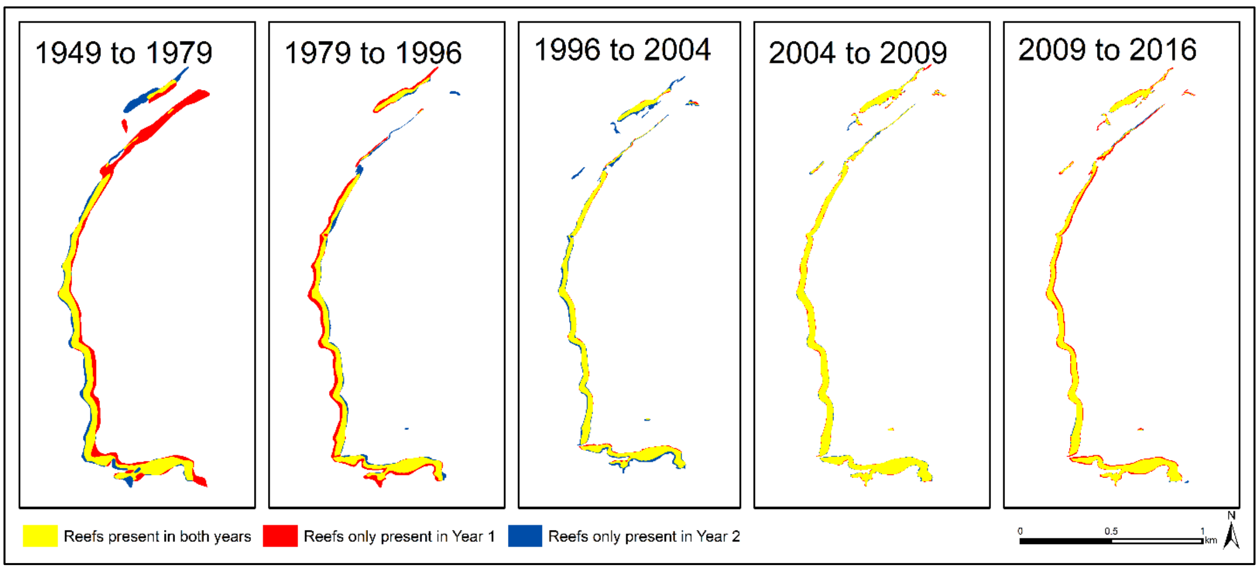

Figure 28. Intermediate-term reef extent for Cedar Reef.
This map illustrates where Intertidal Reefs were only mapped in year 1, where only mapped in year 2, and where mapped in both years for sequential timeframes.

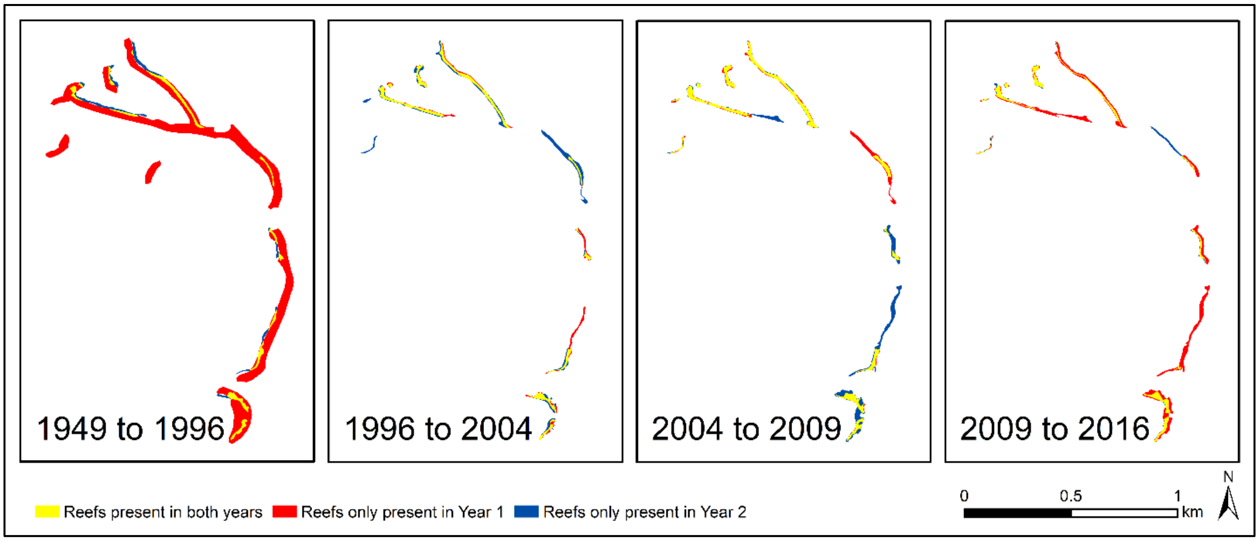

Figure 29. Intermediate-term reef extent for Third Chain of Islands.
This map illustrates where Intertidal Reefs were only mapped in year 1, where only mapped in year 2, and where mapped in both years for sequential timeframes.

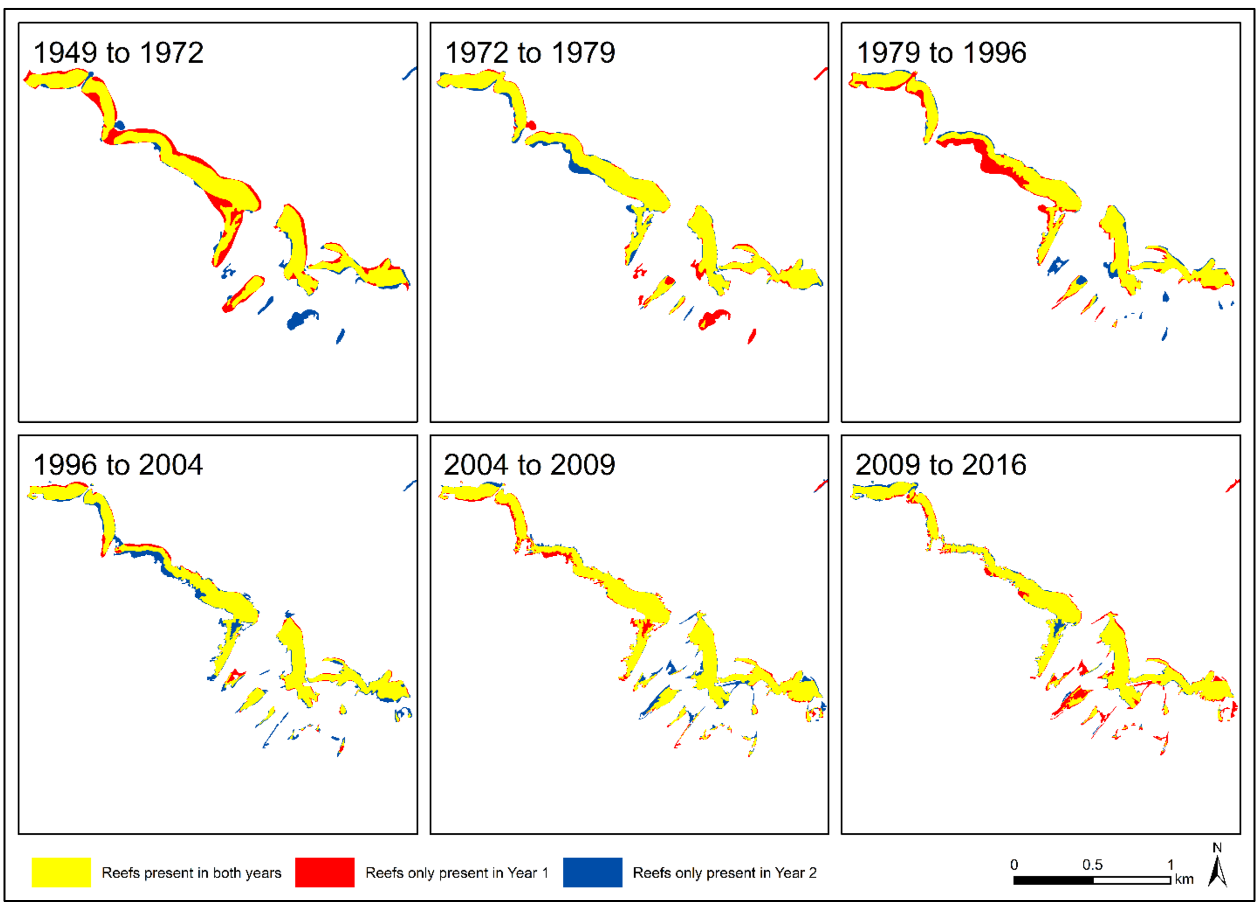

Figure 30. Intermediate-term reef extent for Ayres Reef.
This map illustrates where Intertidal Reefs were only mapped in year 1, where only mapped in year 2, and where mapped in both years for sequential timeframes.

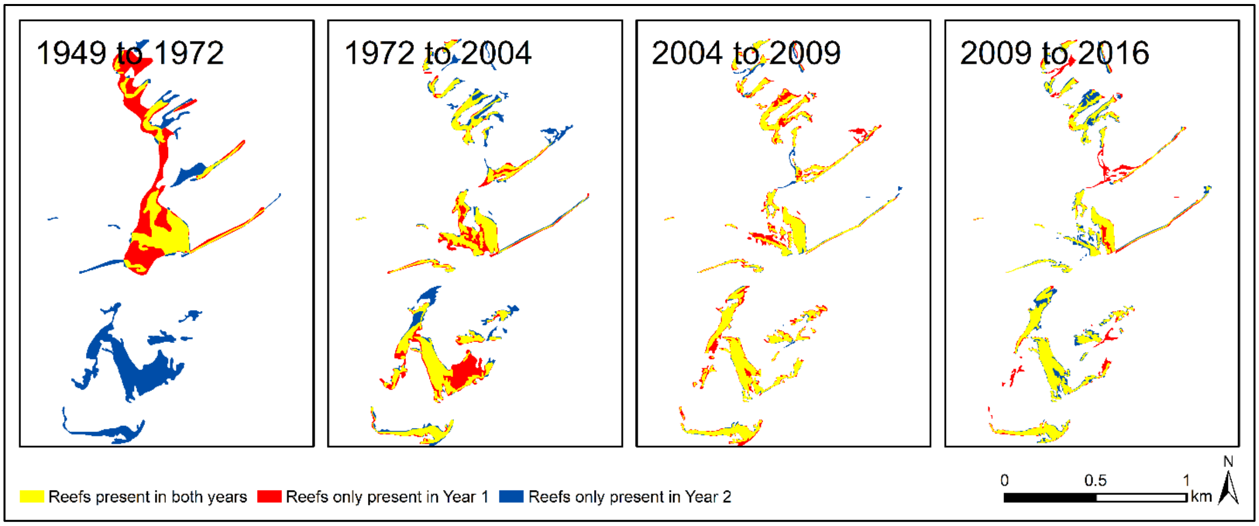

Figure 31. Intermediate-term reef extent for Second Chain of Islands.
This map illustrates where Intertidal Reefs were only mapped in year 1, where only mapped in year 2, and where mapped in both years for sequential timeframes.

Long-Term Reef Changes (1949–2004)

From 1949 to 2004 oyster coverage increased for Ayres Reef (33.8 ac), Third Chain of Islands (33.3 ac), and Cedar Reef (12.3 ac), did not change for Second Chain of Islands (0.3 ac) and decreased for Carlos Reef (1.6 ac) (Table 15). However, the net change between time frames does not give a complete picture of reef dynamics. We intersected the two data sources to determine where reefs remained stable between time frames and where reefs were lost or gained. Ayres Reef remained the most stable, with 54.6 ac remaining from 1949 to 2004 (Figure 32). Second Chain of Islands was the most dynamic with only 1.9 ac remaining from 1949 to 2004 while 42.8 ac was lost and 44.3 ac was gained between timeframes.

Table 15. Long-Term: Area extent (ac) of intertidal oyster habitat in 1949 and 2004 as well as the amount of intertidal oyster habitat that was lost, gained, or remined the same between timeframes.

Reef	1949	2004	Net Change	Area Reefs remained	Area Reefs lost since 1949	Area Reefs gained since 1949
Ayres Reef	66.5	100.3	33.8	54.6	12.0	45.7
Carlos Reef	42.5	40.6	-1.6	16.7	25.5	23.8
Cedar Reef	28.9	41.3	12.3	24.2	4.8	17.1
Second Chain of Islands	44.8	45.0	0.3	1.9	42.9	43.2
Third Chain of Islands	11.0	44.2	33.3	7.1	3.9	37.1

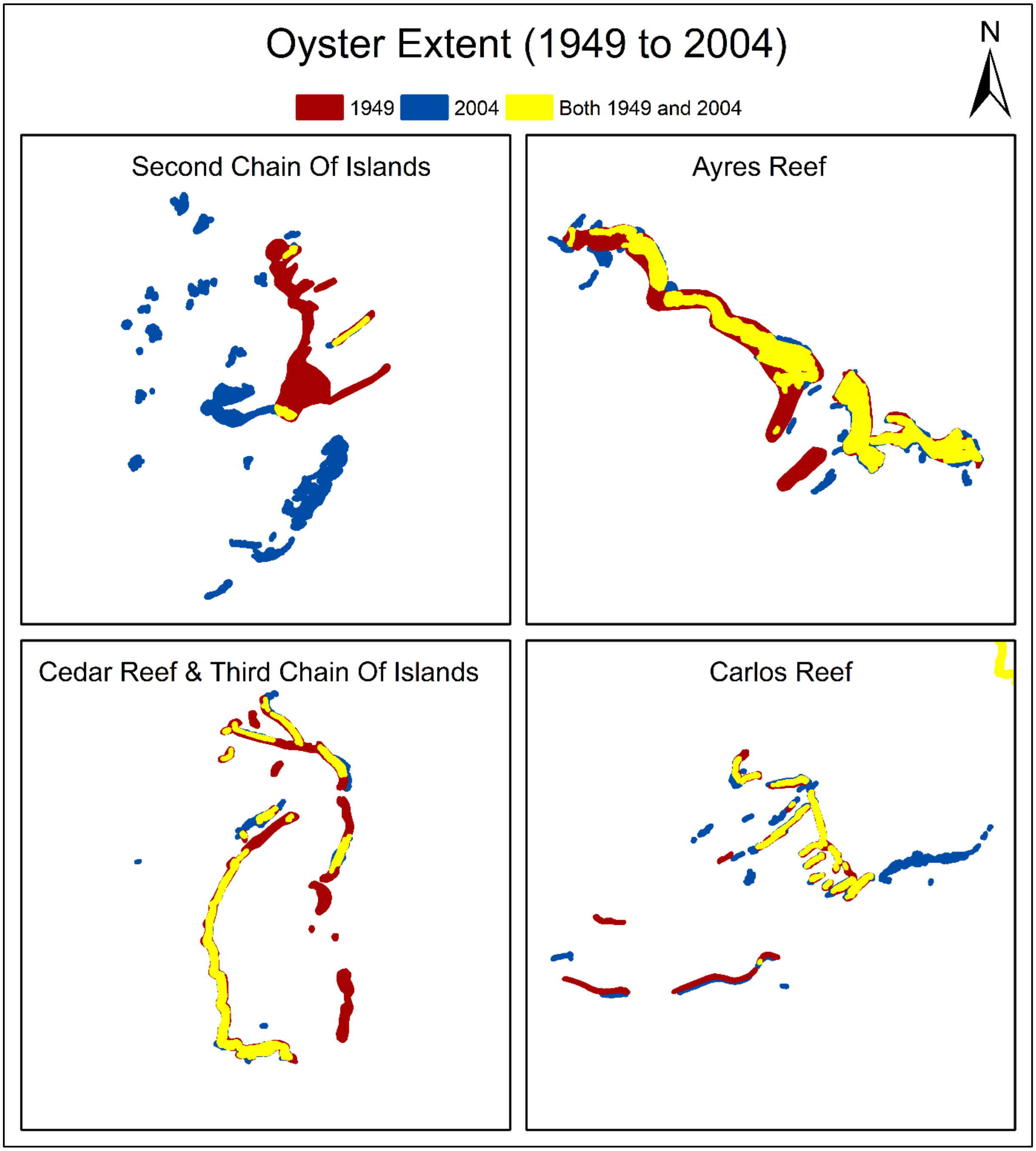

Figure 32. Long-term changes in reef extent.
Oyster extent in 1949 (digitization based on T-Sheet) and 2004 (NOAA Source). Locations of each relative to the entire study area can be seen in Figure 1.

Comparing 2004 NOAA and HRI Digitizations

The subtidal and intertidal reef habitats were mapped for this project over the entire study area and then compared to NOAA's 2004 benthic habitat classifications. Figure 33 – Figure 35 show how the classification digitized by HRI compares to NOAA's classifications. For the entire study area, NOAA classified 1756.28 ac of intertidal and subtidal habitats while HRI classified 1490.81 ac (Table 16). While the overall acreage of oysters classified is similar, the oysters classified as intertidal and subtidal varied drastically between both datasets. For example, NOAA classified 80.47 ac of subtidal that HRI classified as intertidal, and HRI classified 38.79 ac as subtidal and NOAA classified as intertidal.

Looking at each reef individually, both datasets mapped similar amounts of intertidal reefs in all reefs except Second Chain of Islands (Table 17 and Table 18; Figure 33 and Figure 34). HRI classified 71.7 ac (9% of all changes) as intertidal that NOAA classified as subtidal, and NOAA classified 26.6 ac (5.7% of all changes) as intertidal that HRI classified as subtidal. Additionally, for all reefs, NOAA classified significantly more subtidal reefs that HRI classified as No Data (Figure 35).

Table 16. Comparing 2004 NOAA and HRI digitizations: Change metrics depicting the area (ac) classified as intertidal and subtidal environments for NOAA 2004 and HRI 2004 classifications.

		NOAA (ac)		
		No Data	**Intertidal**	**Subtidal**
HRI (ac)	**No Data**	0.00	42.87	481.10
	Intertidal	11.49	116.96	80.47
	Subtidal	247.02	38.79	996.09

Table 17. Comparing 2004 NOAA and HRI digitizations: Change metrics depicting the area (ac) classified as intertidal and subtidal environments per reef for NOAA 2004 and HRI 2004 classifications.

			NOAA (ac)		
			No Data	Intertidal	Subtidal
HRI (ac)	Carlos Reef	No Data	0.0	16.5	197.4
		Intertidal	1.0	22.4	5.6
		Subtidal	84.0	3.6	287.8
	Cedar Reef	No Data	0.0	2.4	67.0
		Intertidal	0.6	25.1	3.9
		Subtidal	11.2	1.4	155.9
	Third Chain of Islands	No Data	0.0	3.4	72.8
		Intertidal	0.4	4.4	3.9
		Subtidal	4.0	3.2	55.1
	Ayres Reef	No Data	0.0	11.1	72.2
		Intertidal	5.3	56.3	25.8
		Subtidal	103.3	4.0	246.3
	Second Chain of Islands	No Data	0.0	9.4	71.7
		Intertidal	4.2	8.8	41.2
		Subtidal	44.5	26.6	250.9

Table 18. Comparing 2004 NOAA and HRI digitizations: Change statistics indicating the percentage of change each classification difference accounted for.

			NOAA (%)		
			No Data	Intertidal	Subtidal
HRI (%)	Carlos Reef	No Data	0.0	2.7	31.9
		Intertidal	0.2	3.6	0.9
		Subtidal	13.6	0.6	46.5
	Cedar Reef	No Data	0.0	0.9	25.0
		Intertidal	0.2	9.4	1.5
		Subtidal	4.2	0.5	58.3
	Third Chain of Islands	No Data	0.0	2.3	49.4
		Intertidal	0.3	3.0	2.7
		Subtidal	2.7	2.2	37.4
	Ayres Reef	No Data	0.0	2.1	13.8
		Intertidal	1.0	10.7	4.9
		Subtidal	19.7	0.8	47.0
	Second Chain of Islands	No Data	0.0	2.1	15.7
		Intertidal	0.9	1.9	9.0
		Subtidal	9.7	5.8	54.9

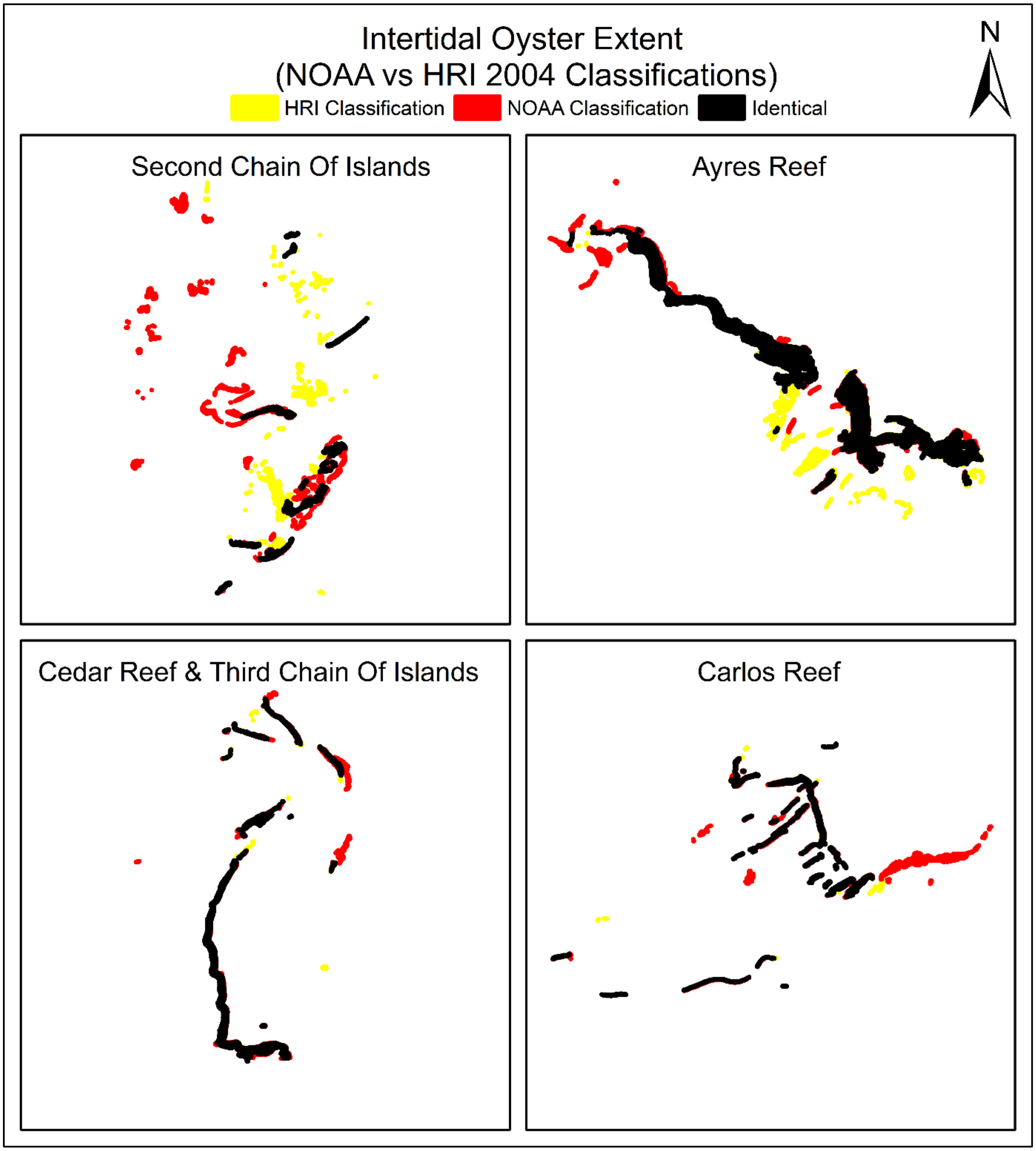

Figure 33. Comparison of HRI's Intertidal extent and NOAA's Intertidal extent from 2004 data.

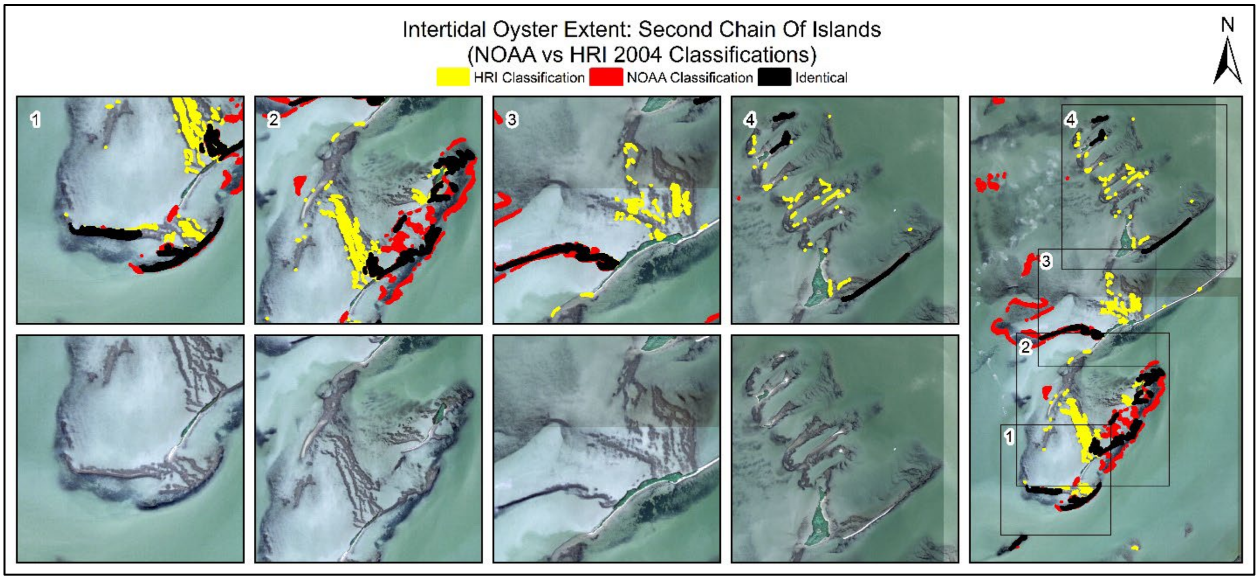

Figure 34. Comparison of HRI's Intertidal extent and NOAA's Intertidal extent from 2004 data for Second Chain of Islands.

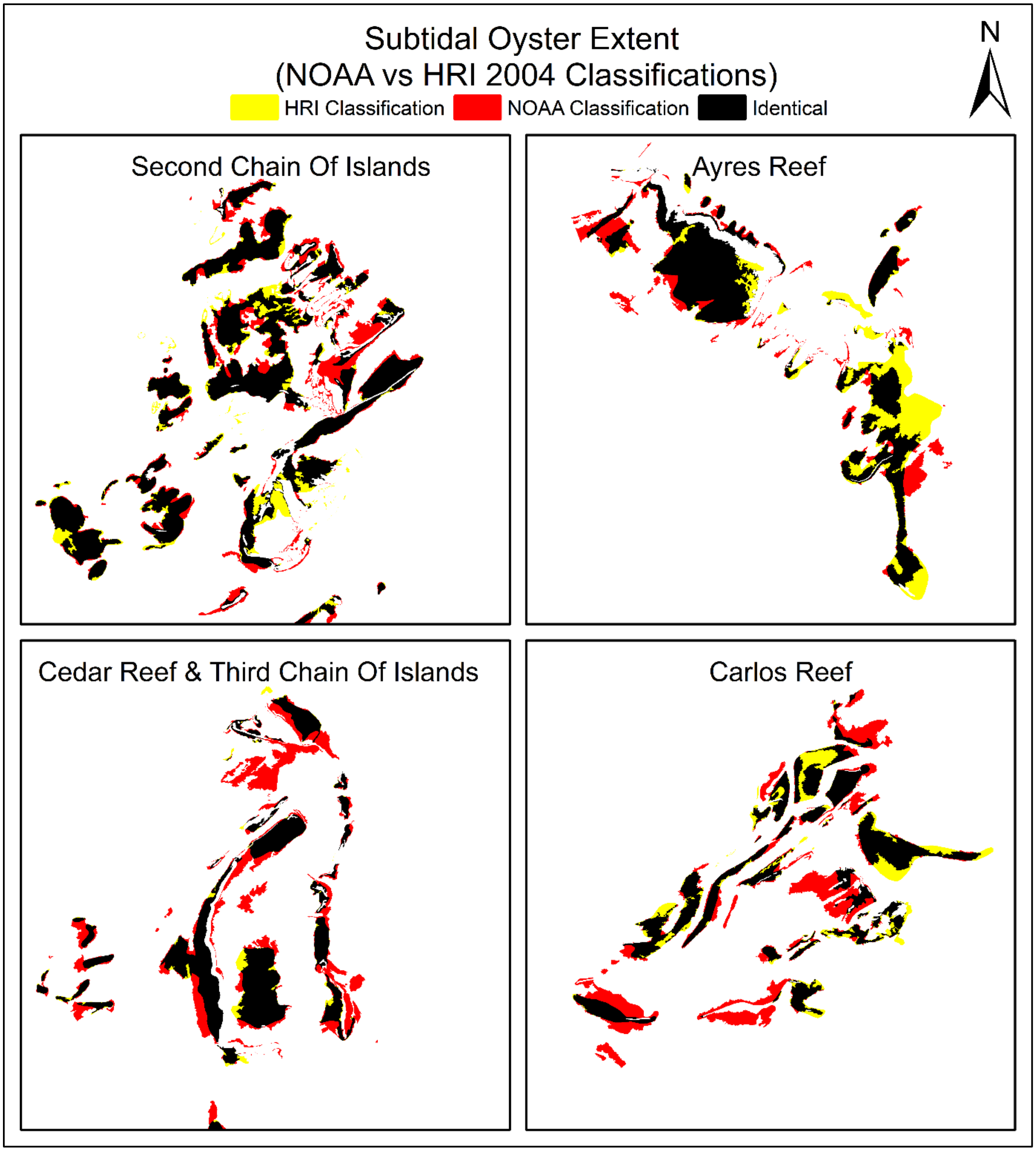

Figure 35. Comparison of HRI's Subtidal extent and NOAA's Subtidal extent from 2004 data.

Summary

Reef structures were analyzed on three time scales: 1) short-term (2004–2009–2016), 2) intermediate-term (1949–1972–1979–1996–2004–2009–2016), and 3) long-term (1949–2004). Additionally, a comparison between NOAA's 2004 intertidal/subtidal reef extent and HRI's digitized 2004 intertidal/subtidal reef extent was conducted. Through comparing both 2004 datasets, the overall acreage of oysters classified was similar, however, the area of oysters classified as intertidal and subtidal varied drastically between datasets. Without rigorous fieldwork documenting the subtidal and intertidal oyster extent, it would be difficult to determine which results are more accurate. Short-term and intermediate-term reef changes were focused on because those delineations were all consistently created (by the same person). While there were slight variations between the intertidal reef coverage of all five reefs, looking at the area of all five reefs combined illustrated that intertidal reef coverage decreased from 1949–1966, increased from 1966–2004, and then decreased from 2004–2016. Overall, from 1949 to 2016, Carlos Reef, Cedar Reef, Third Chain of Islands, and Ayres Reef experienced a net loss of intertidal reefs while Second Chain of Islands remained the same. Subtidal reefs were only able to be mapped on a short-term scale with three years (2004, 2009, and 2016) and results indicated that Carlos Reef and Cedar Reef experienced an increase in subtidal reefs while Third Chain of Islands, Ayres Reef, and Second Chain of Islands experienced a decrease in subtidal reefs. Results also indicated that between individual timesteps there was high variability between the area of subtidal environments while there was not much change between intertidal environments. The change in subtidal environments could have been attributed to either changes in reef structure, image quality, or a combination of the two.

Changes in Nesting Waterbird Populations and Habitat Availability

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Coastal Bend Bays & Estuaries Program

Colonial waterbird population and habitat analysis

The status of colonial nesting waterbirds has changed drastically over the past several hundred years along the Texas coast. To nest successfully, this group of birds congregate in dense concentrations on sites that are high enough to not risk tidal flooding, free of predators, and close to plentiful prey resources (mainly, fish). Historically, there were relatively few natural islands scattered between the state's extensive barrier islands and the mainland coast on which this group of birds encountered this condition. In the central Texas coast area between San Antonio Bay and Aransas Bay, these sites were particularly abundant due to the extensive oyster reef complexes that allowed for sediments to accrete in some formations that could remain above even the highest tides of the year. Within the Carlos/Mesquite/Ayres Bay system, Second Chain and Third Chain of Islands (Figure 36) were historically recognized as some of the most important nesting sites for colonial waterbirds along the Gulf coast, along with several other smaller islands.

Prior to the dredging of the Gulf Intracoastal Waterway (GIWW) in 1949, boat traffic navigated past this stretch of coast through a series of natural channels through the reefs. Upon cutting the GIWW, large dredge deposits were made on the south/east side of the channel and these have been used for subsequent maintenance dredge placement resulting in the creation of a nearly contiguous stretch of higher-elevation uplands from Dunham Island to Rattlesnake Island, much of which has become heavily vegetated with brush. Due to the proximity to an abundant source of mammalian mesopredators (e.g., raccoons, coyotes, feral hogs) from the mainland, these islands are not known to have supported any significant waterbird nesting since the GIWW was completed. This rendered the reef-based island chains even more critically important.

In the past two decades, several dredge material beneficial use sites have been created in and near the bay system with the objective of providing saltmarsh habitat for the Aransas-Wood Buffalo population of Whooping Cranes (*Grus americana*) that winters primarily on the Aransas National Wildlife Refuge and Matagorda Island. In some years following their initial creation, they have provided a nesting opportunity for bare-ground nesters before becoming vegetated or discovered by predators.

Our objectives were to quantify and summarize historical data on colonial waterbird populations nesting on islands in the Carlos/Mesquite/Ayres Bay system, and to quantify the changes in areal coverage of available vegetated nesting substrate.

Methods

There are three main islands or "chains" of islands in the Carlos/Mesquite/Ayres Bay system that have regularly supported colonial nesting waterbirds over the past century. The analysis of nesting birds and habitat focused on these three primary sites – Carlos Dugout, Third Chain of Islands (henceforth, Third Chain) and Second Chain of Islands (henceforth, Second Chain; Figure 36).

Colonial nesting waterbirds

Colonial nesting waterbird species vary in the types of preferred substrate on/in which to build nests. Some prefer to nest in vegetation (ranging from low halophytic forbs and grasses up to mature woody brush and trees several meters in height), and others utilize bare ground (especially sand or shell hash). While some species have some plasticity in nesting habitat preferences, there is general consistency within taxonomically-affiliated guilds.

Species were grouped into the following taxonomic groups: Wading Birds (Families Ardeidae and Threskiornithidae) which typically nest in vegetation ranging from perennial forbs to shrubs and trees; Seabirds (Family Laridae *excluding* gulls) which nest on emergent bare-ground substrates; Laughing Gulls (*Leucophaeus atricilla*) which nest on either sparse low vegetation or bare-ground; Brown Pelicans (*Pelecanus occidentalis*) and Neotropic Cormorants (*Phalacrocorax brasilianus*)). Pelicans can nest either in low vegetation or shrubs/trees, while cormorants are exclusively a shrub/tree nester. Since Laughing Gulls are often affiliated with anthropogenic subsidies (food waste, shrimp bycatch), they are enumerated and treated separately from the other ground-nesting seabirds. American Oystercatcher (*Haematopus palliatus*) is a ground-nesting shorebird common in shell-reef systems and utilizes many of the same islands as colonial nesters. However, it is solitary and territorial (of other oystercatchers) and often nests much earlier than most colonial-nesters and is thus not included in further analysis. A full taxonomic list of species in each guild is provided in Table 19.

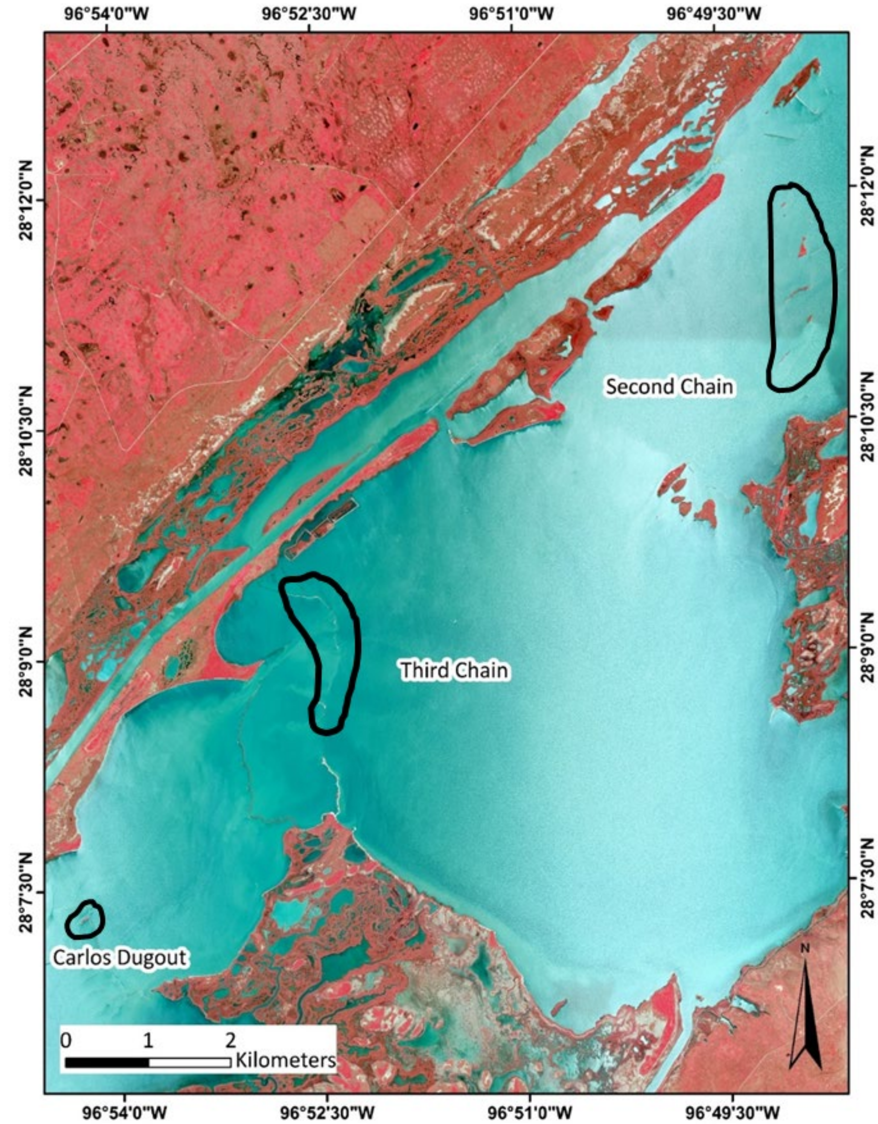


Figure 36. Carlos/Mesquite/Ayres Bay study area showing general areas of primary waterbird nesting habitats.

Table 19. Complete taxonomic list of species nesting on islands in the Carlos/Mesquite/Ayres Bay system based on historical and recent data analyzed in this report.

Order	Family	Latin Name	Common Name
Charadriiformes	Haematopodidae	*Haematopus palliatus*	American Oystercatcher
Charadriiformes	Laridae	*Leucophaeus atricilla*	Laughing Gull
Charadriiformes	Laridae	*Onychoprion fuscatus*	Sooty Tern
Charadriiformes	Laridae	*Sternula antillarum*	Least Tern
Charadriiformes	Laridae	*Gelochelidon nilotica*	Gull-billed Tern
Charadriiformes	Laridae	*Hydroprogne caspia*	Caspian Tern
Charadriiformes	Laridae	*Sterna forsteri*	Forster's Tern
Charadriiformes	Laridae	*Thalasseus maximus*	Royal Tern
Charadriiformes	Laridae	*Thalasseus sandvicensis*	Sandwich Tern
Charadriiformes	Laridae	*Rynchops niger*	Black Skimmer
Suliformes	Phalacrocoracidae	*Nannopterum brasilianum*	Neotropic Cormorant
Pelecaniformes	Pelecanidae	*Pelecanus occidentalis*	Brown Pelican
Pelecaniformes	Ardeidae	*Ardea herodias*	Great Blue Heron
Pelecaniformes	Ardeidae	*Ardea alba*	Great Egret
Pelecaniformes	Ardeidae	*Egretta thula*	Snowy Egret
Pelecaniformes	Ardeidae	*Egretta caerulea*	Little Blue Heron
Pelecaniformes	Ardeidae	*Egretta tricolor*	Tricolored Heron
Pelecaniformes	Ardeidae	*Egretta rufescens*	Reddish Egret
Pelecaniformes	Ardeidae	*Bubulcus ibis*	Cattle Egret
Pelecaniformes	Ardeidae	*Nycticorax nycticorax*	Black-crowned Night-Heron
Pelecaniformes	Ardeidae	*Nyctanassa violacea*	Yellow-crowned Night-Heron
Pelecaniformes	Threskiornithidae	*Eudocimus albus*	White Ibis
Pelecaniformes	Threskiornithidae	*Plegadis chihi*	White-faced Ibis
Pelecaniformes	Threskiornithidae	*Platalea ajaja*	Roseate Spoonbill

For historical reference, we utilized unpublished reports from National Audubon Society wardens from 1939, 1941, 1959, and 1965. We used these reports to describe the status of waterbird nesting in the system prior to large-scale hydrological alterations and at a time of lower human pressure (though many of these birds were still recovering from the plume trade of the late 19th century which decimated many waterbird populations). While the reports differed in methodology, reported metrics and geographic extent, they are nonetheless illustrative in describing the general status of waterbird nesting over that time.

We compiled colonial-nesting waterbird data gathered through the Texas Colonial Waterbird Survey (TCWS) for the 50-year time period from 1973–2022, excluding 1991 and 1992 when there were insufficient data for most islands in the system. The data consist of breeding pair estimates for each species detected during an approximate two-week window at the end of May and beginning of June when most nesting waterbirds are present.

The mean and standard deviation of each group was calculated for each of the three main island complexes (Second Chain, Third Chain, and Carlos Dugout) for the five decades beginning 1973 (the eight-year average was calculated for the period 1983–1990). Other islands that only occasionally supported nesting birds were grouped as "Other."

Nesting habitat change

As a historical reference, the National Oceanic and Atmospheric Administration (NOAA) National Geodetic Survey (T-sheet) data was obtained for 1939. The T-sheet is an authoritative coastline survey, delineating the U.S. high-water line. With respect to the islands, these polygons would include both the vegetated upland areas as well as any supratidal bare ground habitat that is typically found around the perimeter.

Preliminary analysis indicated that an unsupervised classification approach using National Agriculture Imagery Program (NAIP) imagery would not be suitable to delineate the habitats utilized by colonial nesting waterbirds. Though classification methods have been used in the area (Hackney et al., 2016), the spatiotemporal resolution adopted in this study was fine enough that manual digitization was favored because it incorporates user-based local knowledge when delineating potentially suitable habitat. To assess the change in upland vegetated habitats over time, NAIP orthoimagery were obtained and digitized for the years 2004, 2008, 2012, 2016, and 2020. The NAIP images were collected at a spatial resolution of one meter and the temporal resolution represented the prominent agricultural growing season for each year. Images were projected to the North American Datum of 1983 UTM Zone 14N coordinate system for all area calculations to be conducted in meters. Using the 1939 T-sheet as a reference for which islands to include in the study area, all NAIP imagery were then manually digitized. The T-sheet was only used as a general guideline because the delineated upland polygons include both emergent shoreline and vegetated area, while the NAIP imagery was referenced for delineating just emergent vegetated habitat. Images were displayed in the 4,1,2 RGB band combination, a widely used infrared visualization that displays healthy vegetation in red, in ArcMap 10.8.1. Starting in

2004, polygons were drawn, and their respective areas calculated (m^2) for each island of interest. The vertices for each polygon were placed in the center of red pixels, indicating the possible presence of healthy vegetation, that were surrounded by at least 4 other like pixels. The polygons from 2004 were then duplicated and overlayed over the 2008 NAIP imagery and adjustments were made only to vertices that appeared to have changed, and this process was repeated for subsequent years of imagery. By using duplicates of the polygons developed in previous time periods, user error derived from the random effects associated with new polygon creation was minimized and only the effects of erosion and land loss were included. However, this method still introduces some human formed errors. Moreover, though the majority of vegetation types analyzed within this study area do not experience significant annual changes in photosynthetic activity, it is important to note that other sensors and indices, such as MODIS EVI, may display greater seasonal variation of vegetation availability.

Relationship between vegetated island area and wading bird pairs

Since we could only effectively estimate the area of emergent vegetated island habitat, we limited our analysis of the relationship between area and birds to the wading bird group – the guild that predominantly utilizes vegetative structure to build nests.

To investigate the relationship between island area and waterbird pairs, we used the TCWS dataset for 2004–2020, and the area of vegetated upland calculated using the NAIP imagery used to describe the trend in habitat change. Since the area was calculated at four-year intervals, we assumed a constant rate of change between sequential years to interpolate the area available for each of the three years of the interval. We plotted the number of nesting wading birds against the area available for each year and determined the function of best fit by testing a linear, log, and second-order polynomial model to maximize the coefficient of determination (R^2).

Results

Colonial nesting waterbirds

Two historical reports are available prior to the dredging of the GIWW, referencing years 1939 and 1941.

1939

A total of 3,352 pairs of nine tree/shrub nesting species were reported in the system (Allen 1939; Table 20). These were all wading birds with the exception of 18 Neotropic Cormorants. Notes associated with the counts are unclear on two counts specifically: 1) for 200 Roseate Spoonbills on Ayres Island there is a question mark following the figure, but since over 1,000 other wading birds were nesting on the island at the time it is likely that the question mark indicates uncertainty of the estimate rather than uncertainty of their nesting status, so this figure is included in the total pair figure; and, 2) question marks follow most of the counts on Dunham Island, and for some species notes indicate the island was "deserted" of them. Notes do not clarify what is meant by the question marks so we do not consider this island in the summary of nesting birds for this year. Of all wading birds, 1,738 pairs were on Second Chain, and 1,588 pairs were on Ayres Island, with only 18 total wading birds distributed between Third Chain and "Shell Island." A total of 756 seabird and 174 Laughing Gulls pairs were distributed between Second Chain, Third Chain, and Shell Island. Additionally, 340 pairs of Brown Pelicans were documented on Third Chain.

The location of "Shell Island" referenced in this report is unknown. The island we now refer to as Carlos Dugout was not referenced in this or any of the other historical reports, though it would have been present/emergent at the time as it appears in the 1939 T-sheet. It is possible that Shell Island was a name previously attributed to Carlos Dugout.

1941

A report titled "1941 Production of young birds in National Audubon Sanctuaries" provided no information on authorship or methodology, though it is likely a compilation of data collected by Audubon wardens throughout the Gulf and Atlantic states. The report indicated young birds fledging from five islands in the system: Second Chain, Third Chain, Shell Island, Ayres Island, and Dunham Island. All but 36 of the 6,536 reported wading birds were ascribed to Second Chain, as were 1,800 Neotropic Cormorants. A total of 1,613 ground-nesters were reported to have been produced, in addition to 200 Brown Pelicans (on Second Chain). Since we have no data on nesting pairs, we cannot compare this to other such data from other years. For most of these species, a fledge rate exceeding 3 per pair would be considered very high, so conservatively we estimate that the young produced represented no less than 2,179 pairs of wading birds, 600 pairs of cormorants, and 538 pairs of ground-nesters (of which approximately ¼ were Laughing Gulls). Additionally, 200 Brown Pelicans were reported to have fledged. In a study spanning islands across the Gulf in 2014–2015 (Lamb et al. 2017), mean productivity never exceeded 1.64 young per nest. Conservatively assuming an even higher rate of 2.0 per nest, the 200 young would have represented a minimum 100 nesting pairs of Brown Pelicans.

1959 and 1965

A report produced in each of these two years (Unknown 1959, Unknown 1965) provides information on nesting pairs only for Second Chain, and is restricted to the wading bird species. A total of 965 and 1,062 wading bird pairs were reported for 1959 and 1965, respectively. These reports do not indicate whether other islands in the system were active or simply not surveyed, so they are of limited inferential value and not presented in tabulations for the full time period.

1973 through 2022

Texas Colonial Waterbird Survey data were available from 1973–2022 with the exception of 1991 and 1992. During that timeframe, Second Chain has supported over 96% of the 40,996 wading bird pairs recorded nesting in the system (Table 20). Total wading bird pairs declined from the beginning of the survey until the decadal period 1993–2002, increased slightly from 2003–2012, then declined drastically in the past decade. For ground-nesting seabirds, the decadal average increased between the first and second decades to a high of 668 (246), and declined steadily thereafter to 69 (52) in the most recent decade. Over the full dataset, Second Chain supported 61.6% and Third Chain supported 36.8% of the total of 16,069 pairs of seabirds, with other islands contributing less than 2%. Third Chain eroded more rapidly in the recent two decades and there have been no nesting attempts there by waders since 2015 (1 pair) or by seabirds since 2017 (15 pairs) due to the complete loss of emergent habitat. Laughing Gull trends showed a similar decadal pattern to that of the seabirds for the whole system, but were concentrated almost entirely (96.3%) on Second Chain. Brown Pelicans nested sporadically on Second Chain in numbers ranging from 4-22 pairs between 1973 and 1989, but have since ceased to nest there with the exception of one pair in 2011.

Nesting habitat change

NAIP orthoimagery were obtained for the years 2004, 2008, 2012, 2016, and 2020 and the 1939 NOAA T-sheet was used to characterize historic island locations. The digitization of the NAIP orthoimagery of the three main island chains revealed a 59% loss (Δ 11,514.7 m^2) of nesting habitat between 2004 and 2020. The area (m^2) of emergent vegetation suitable for nesting decreased in each period for all island chains (n=3; Table 21, Figure 37). Area loss was negatively correlated with time and the island chains expressed strong polynomial declines of area (all $R^2 > 0.92$), indicating that the rate of erosion also changed over time. Between 2016 and 2020, the Third Chain islands lost all remaining nesting habitat area. By 2020, the Carlos Dugout (Figure 38) and Second Chain of Islands (Figure 39 and Figure 40) decreased to 39% and 43%, respectively, of their areas in 2004. Both the Second (Δ 3,558.1 m^2) and Third Chains (Δ 551.9 m^2, Figure 41) experienced the greatest total area loss between 2008 and 2012, though the highest four-year percentage loss was between 2016 and 2020. The nesting habitat on Carlos Dugout experienced the largest decline (in total area and percentage) between 2016 and 2020 (Δ 596.0 m^2).

Table 20. Summary of colonial nesting waterbird data in the Mesquite Bay complex.
Numbers represent mean nesting pairs (standard deviation in parentheses). If no nesting within a site in a decade, standard deviation is (-). [a]Eight-year average as data is insufficient for 1991-1992

	Audubon (unpublished)	Texas Colonial Waterbird Survey - decadal averages				
Year(s)	**1939**	**1973-1982**	**1983-1990[a]**	**1993-2002**	**2003-2012**	**2013-2022**
Wading birds	**3334**	**1339 (355)**	**1279 (425)**	**656 (362)**	**817 (288)**	**64 (105)**
Second Chain	1738	1321 (358)	1264 (436)	640 (347)	752 (290)	245 (105)
Third Chain	8	4 (5)	5 (7)	3 (5)	3 (5)	0 (0)
Carlos Dugout	-	0 (-)	5 (7)	0 (1)	37 (59)	19 (13)
others	1588	15 (32)	5 (14)	13 (28)	25 (22)	0 (-)
Seabirds	**756**	**409 (141)**	**668 (246)**	**447 (228)**	**147 (83)**	**69 (52)**
Second Chain	352	229 (147)	385 (159)	285 (157)	102 (67)	66 (51)
Third Chain	248	165 (104)	283 (115)	162 (119)	37 (50)	3 (6)
Carlos Dugout	-	0 (-)	0 (-)	0 (-)	0 (-)	0 (-)
others	156	15 (35)	0 (-)	1 (3)	8 (26)	0 (-)
Laughing Gulls	**174**	**186 (196)**	**696 (506)**	**595 (467)**	**453 (201)**	**75 (96)**
Second Chain	96	147 (140)	684 (485)	580 (463)	449 (200)	75 (96)
Third Chain	52	3 (5)	12 (23)	15 (19)	3 (5)	0 (0)
Carlos Dugout	-	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
others	26	36 (114)	0 (0)	0 (0)	1 (3)	0 (0)
Pelicans/cormorants	**358**	**7 (8)**	**3 (6)**	**0 (-)**	**0 (0)**	**0 (-)**
Second Chain	18	7 (8)	3 (6)	0 (-)	0 (0)	0 (-)
Third Chain	340	0 (-)	0 (-)	0 (-)	0 (-)	0 (-)
Carlos Dugout	-	0 (-)	0 (-)	0 (-)	0 (-)	0 (-)
others	0	0 (-)	0 (-)	0 (-)	0 (-)	0 (-)
TOTAL	**4622**	**1940 (371)**	**2645 (1014)**	**1702 (804)**	**1422 (471)**	**411 (197)**
Second Chain	2204	1703 (451)	2336 (967)	1507 (770)	1305 (455)	389 (197)
Third Chain	648	171 (104)	299 (129)	181 (120)	45 (59)	4 (6)
Carlos Dugout	-	0 (-)	5 (7)	0 (1)	37 (59)	19 (13)
others	1770	66 (145)	5 (14)	14 (27)	34 (34)	0 (-)

Table 21. Change in nesting habitat area (m²) of three main island chains in Carlos, Mesquite, and Ayres Bays, Texas. Nesting habitat area was calculated as the total amount of emergent vegetation available in digitized NAIP orthoimagery for the years 2004, 2008, 2012, 2016, and 2020. The areas based on the 1939 NOAA T-sheet are assumed to include both vegetated and unvegetated supratidal habitats, so are not directly comparable.

Island Chain	1939	2004	2008	2012	2016	2020
Carlos Dugout	6,815.6	1,879.4	1,766.8	1,437.6	1,331.6	735.59
Second Chain	33,787.7	16,667.2	15,644.4	12,086.4	10,305.3	7,181.5
Third Chain	8,637.7	885.3	791.4	239.6	150.8	0
Total	49,240.9	19,431.9	18,202.6	13,763.5	11,787.7	7,917.1

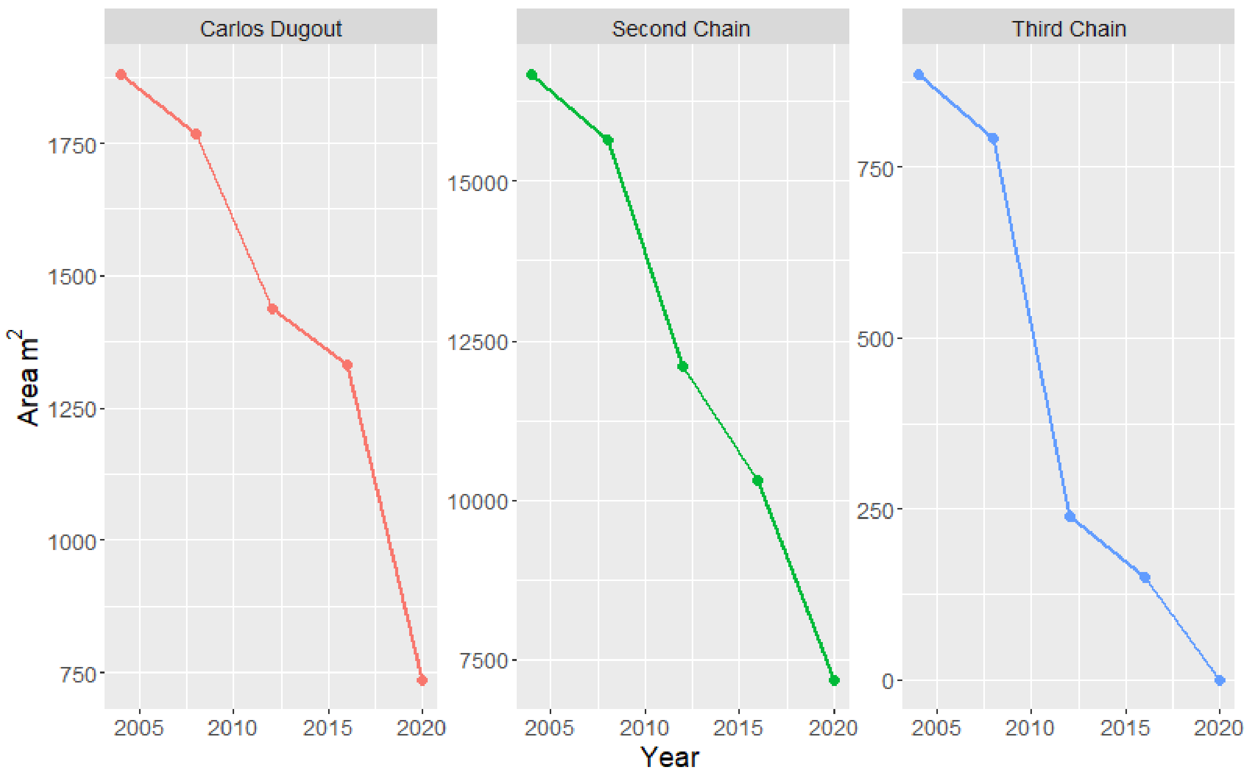


Figure 37. Change in available vegetated upland nesting habitat on three main island groups (Carlos Dugout, Second Chain, Third Chain) in Carlos, Mesquite and Ayres Bay, Texas between 2004-2020, based on NAIP imagery. Note different y-axis (area) for each.

Figure 38. Area (m^2) of Carlos Dugout Island, Texas. Area in 1939 was calculated based on the NOAA T-sheet shoreline mapping dataset (high-tide line). Areas for years 2004, 2008, 2012, 2016, and 2020 reflect emergent vegetation based on digitized NAIP imagery.

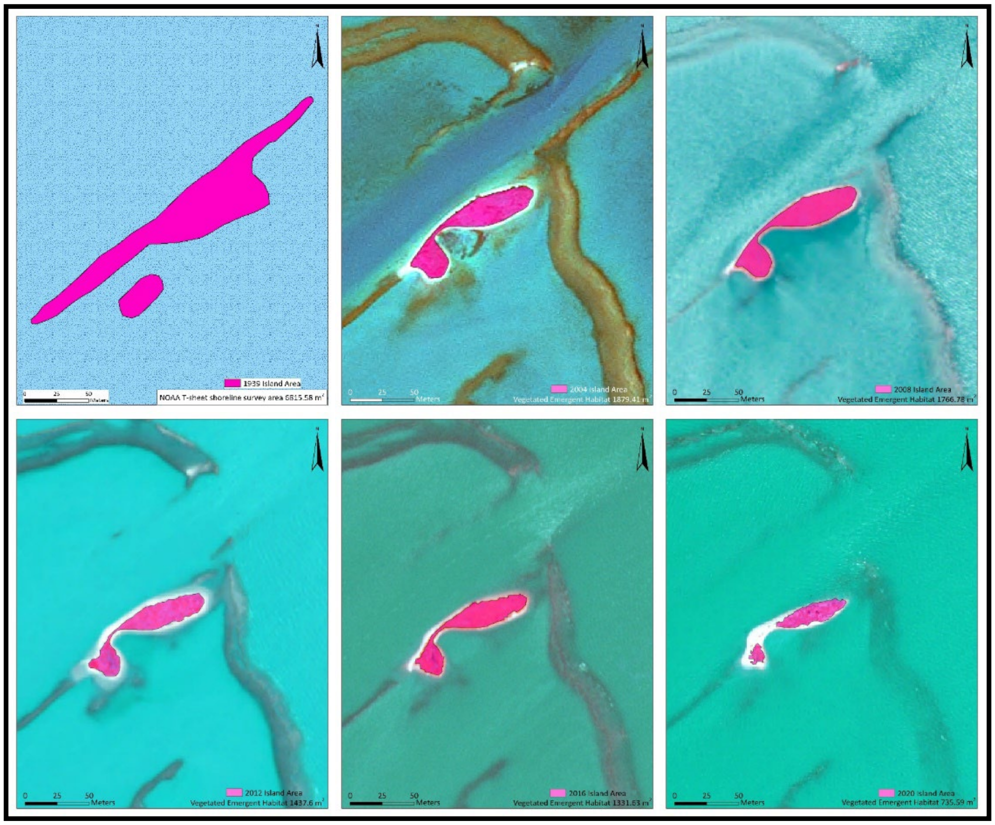


Figure 39. Area (m^2) of Second Chain of Islands, Texas. Area in 1939 was calculated based on the NOAA T-sheet shoreline mapping dataset (high-tide line). Areas for years 2004, 2008, 2012, 2016, and 2020 reflect emergent vegetation based on digitized NAIP imagery.

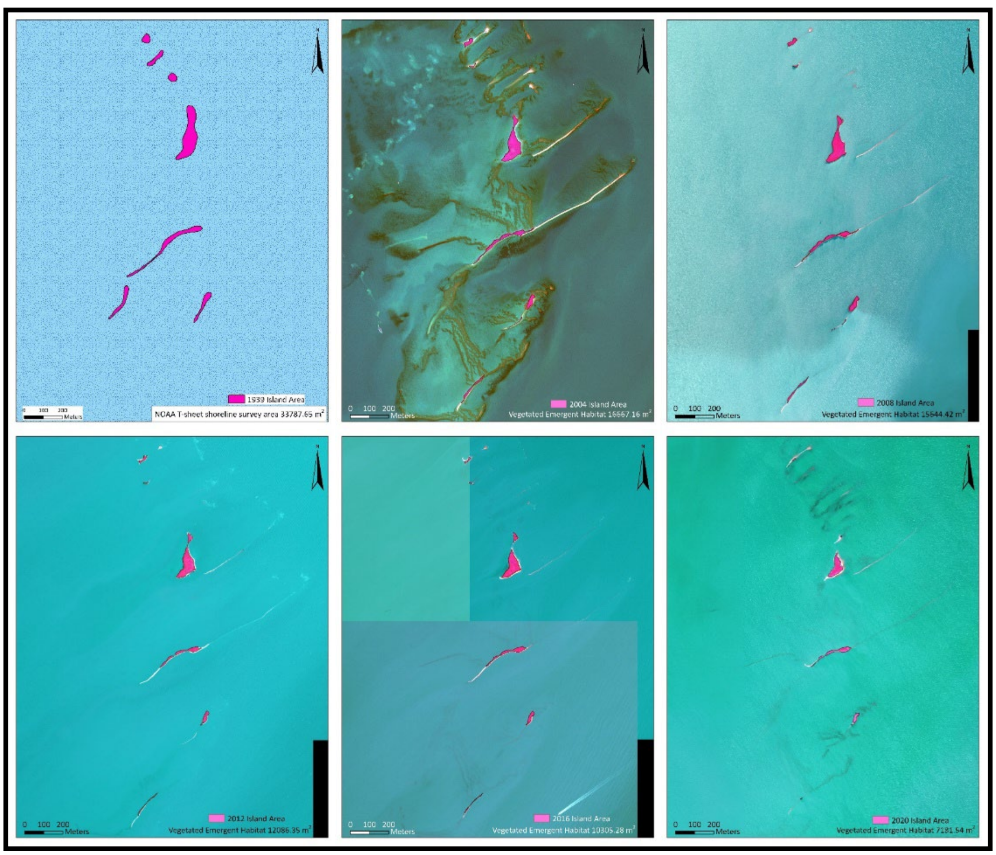


Figure 40 Area (m²) of the largest island in Second Chain of Islands, Texas. Area in 1939 was calculated based on the NOAA T-sheet shoreline mapping dataset (high-tide line). Areas for years 2004, 2008, 2012, 2016, and 2020 reflect emergent vegetation based on digitized NAIP imagery.

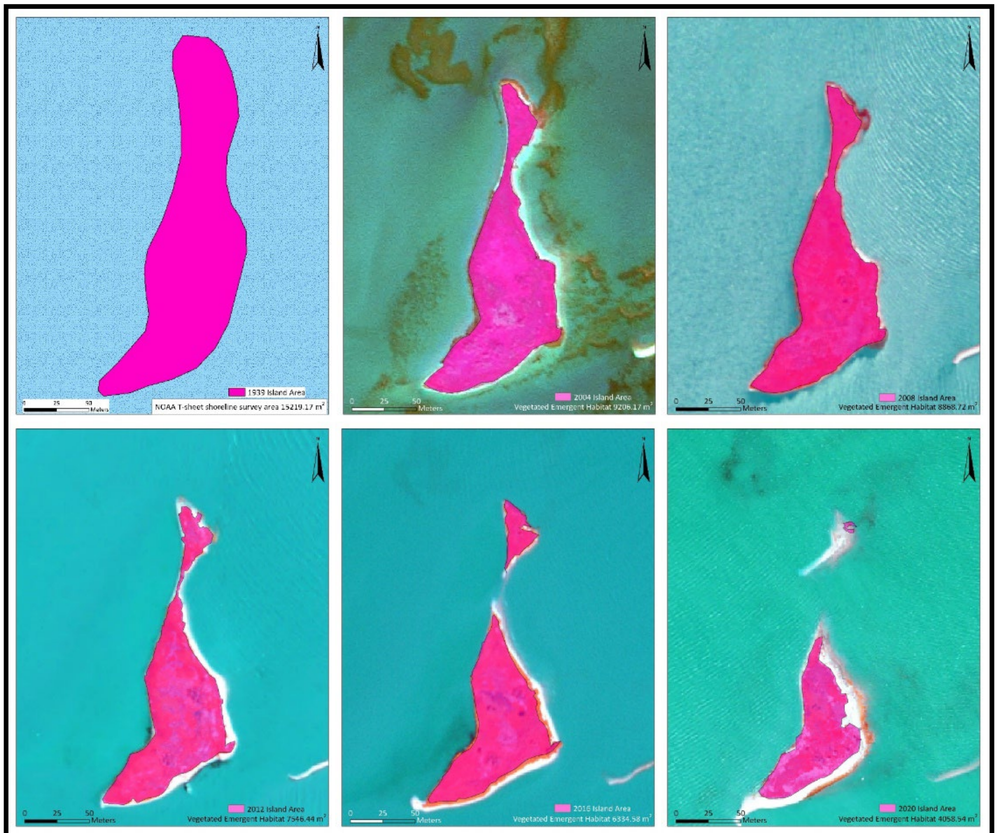
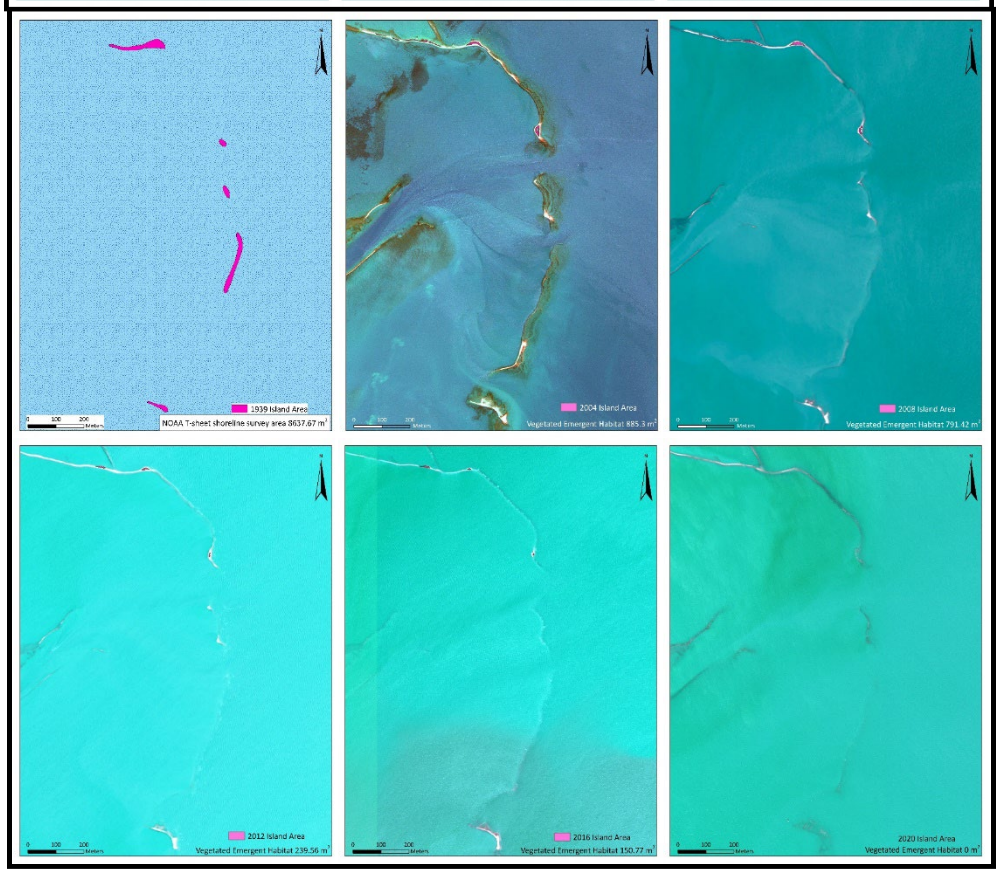


Figure 41. Area (m²) of Third Chain of Islands, Texas. Area in 1939 was calculated based on the NOAA T-sheet shoreline mapping dataset (high-tide line). Areas for years 2004, 2008, 2012, 2016, and 2020 reflect emergent vegetation based on digitized NAIP imagery.



Relationship between vegetated island area and wading bird pairs by island chain

On Second Chain, available nesting area accounted for 88% of the variability in the number of wading bird pairs, with the function of best fit being the 2nd order polynomial ($y = 9E\text{-}06x^2 - 0.1381x + 695.52$; $R^2 = 0.8779$; Figure 42).

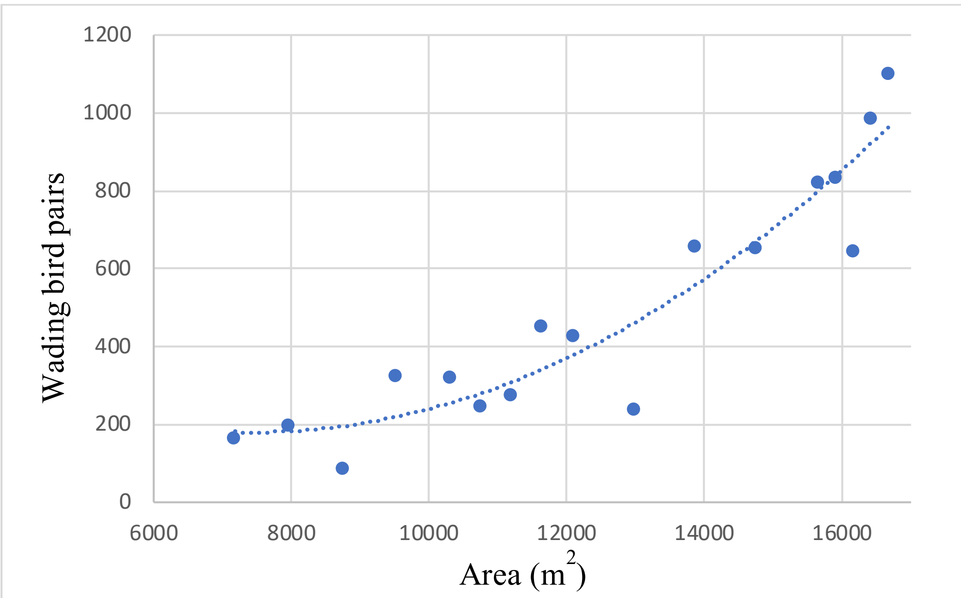


Figure 42. Plot of wading bird pairs reported to TCWS and available upland vegetated potential nesting habitat on Second Chain in the Mesquite Bay complex, Texas from 1973-2022.

On Carlos Dugout, small numbers (mostly less than 50, but three years with 60 or greater) of wading birds have occasionally utilized the island for nesting since 2006. Though the island has decreased in size over time, there was no evidence that island area accounted for a significant amount of variability in nesting pairs ($R^2 = 0.0446$).

On Third Chain, wading birds were recorded nesting in only five of the 17 years between 2004-2020, and only twice did they exceed 3 pairs (14 pairs in 2008, 13 pairs in 2010), so there were too little data to explore a relationship between vegetation and wading birds.

Discussion

Populations of nesting waterbirds can be highly variable from year to year, and these fluctuations are likely the result of a wide range of causative factors many of which are interrelated. This should be expected in a dynamic estuarine environment, and especially with birds which are relatively long-lived and highly motile. While prey availability may be considered to set an upper bound on a bay system's capacity to support birds energetically, the availability of suitable nesting area can also set an upper bound on nesting bird populations. The contrast between the historic report of the area in 1939 and the trends in the last 50 years (1973–2022) indicates that the Mesquite/Carlos/Ayres Bay system supports far fewer nesting birds than it did historically and that the trend is continuing downward. This trend reflects a similar trajectory to the availability of nesting area. In the shorter-term (the 2004–2022 period), the loss of vegetated nesting area has been especially pronounced, and the relationship to the numbers of nesting wading birds is clear.

While Second Chain continues to support the highest numbers of birds in the system, they are concentrated on fewer islands as they were previously, as several of the small islets in the chain have become completely submerged over the past thirty years.

Erosive forces work to erode the shorelines – the perimeters – of these islands and often the forces act on multiple sides. Meanwhile, no significant accretion is occurring to offset the losses due to erosion. Consequently, the same linear rate of erosion ($m \cdot y^{-1}$) has an exponentially increasing effect on an island's area as it gets smaller. The nesting habitat change analysis suggests that the rate of area loss has in fact accelerated in the latter part of the 2004–2020 time period. Indeed, only small parts of Third Chain have been emergent enough to allow for ground-nesting birds to attempt to nest sporadically since 2011, and none have been documented since 2017.

At least part of the apparently accelerated rate of nesting habitat loss associated with the 2016–2020 interval may be attributable to the effects of Hurricane Harvey which struck the area directly on August 25, 2017. Reconnaissance trips to the islands in the following weeks revealed major loss of stabilizing vegetation in the interior of islands in addition to severe erosion along the perimeter of islands. Previously, the shoreline of the main island of Second Chain was composed primarily of coarse oyster shell hash. Following the storm, much of this shell material had been dispersed or heaved up onto the higher vegetated part of the island, exposing the underlying clay layer along the shoreline. In addition to the protective function of surrounding oyster reefs which serve as breakwaters that reduce height and fetch of the wind-generated waves of the surrounding bays, a shoreline composed of shell serves as a secondary stabilizing buffer against the remaining energy by absorbing tiny waves while staying in place. The loss of shell hash shoreline surrounding the islands may be an exacerbating factor in the acceleration of island habitat loss.

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