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USING DRONE IMAGERY TO MAP INTERTIDAL OYSTER REEFS ALONG FLORIDA'S GULF OF MEXICO COAST

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Abstract: Eastern oyster (*Crassostrea virginica*) reefs offer vital ecosystem services and support economically and culturally important fisheries. However, environmental and anthropogenic stressors have led to significant decline in oyster reef coverage globally and locally in places like the Suwannee Sound in Florida, USA. Current monitoring methods are insufficient for timely and accurate assessment of oyster resources in the region. Here we demonstrate how drone imagery can be used to delineate intertidal oyster reef coverage rapidly and reliably. The high spatial resolution offered by drone imagery enables accurate delineations. We use a segmentation algorithm to delineate reefs, which produces consistently detailed outlines that are more representative of reef morphology than manual delineations. In total, 1,394 reefs were delineated, which corresponds with 497,670 m^2 of reef area. Of the delineated reefs, 236 (17%) were newly mapped, aligning with 19,848 m^2 of newly mapped intertidal oyster reef habitat. The overlapping drone imagery also enabled the production of digital surface models, which were used to calculate volume to area ratio as an indicator of reef condition. These delineations and features serve as accurate baseline data that can be compared to future surveys to monitor how reefs are changing over time in the Suwannee Sound. These methods can also be expanded to other geographical areas and can aid in identifying early signs of decline in oyster reefs.

Key words: coastal mapping, UAS, UAV, habitat mapping, image segmentation

Introduction

In Florida, USA, intertidal eastern oysters (*Crassostrea virginica*) offer ecosystem services such as shoreline erosion control, water filtration, and nutrient cycling (Coen et al. 2007, Grabowski and Peterson 2007, Scyphers et al. 2011). They also represent an economically and culturally important fishery, and support finfish and crustacean fisheries by building habitat (Coen et al. 1999, Peterson et al. 2003, Humphries and La Peyre 2015). However, stressors such as disease, overharvest, and an increase in drought conditions have resulted in a decline in oyster reef coverage globally and locally in Florida, threatening the services and fisheries they support (Beck et al. 2011, Seavey et al. 2011, Frederick et al. 2016). The collapse of oyster populations has resulted in the closure of fisheries in Florida, such as the historic Apalachicola fishery in 2020 (Brown et al. 2021). Given the documented decline, there is a critical need for efficient and consistent monitoring of oyster resources in the region, which requires robust baseline data.

Monitoring changes in oyster resources is often challenging, given the limitations associated with accessing intertidal areas. Conventional methods for delineating reef extent such as real—time kinematic (RTK) GPS surveys (Radabaugh et al. 2019) are also time—intensive and do not provide continuous coverage. Given the difficulties with field—based mapping, remote sensing alternatives such as satellite and occupied aircraft imagery have been relied on to delineate intertidal oyster reefs (Grizzle et al. 2018, Garvis et al. 2020, Grizzle et al. 2023). However, satellite and aircraft imagery carry their own limitations, as they are often not aligned with a favorable tide for maximum reef exposure, and low—cost options offer a relatively coarse resolution. Additionally, satellite imagery can be challenging to work with, given the limitations of cloud cover. Relying on resolutions consistent with satellite imagery (i.e., meters) or occupied aircraft imagery (i.e., decimeters) limits what can be definitively classified as oysters considering the spectral similarity of surrounding cover types in the intertidal environment, such as mudflats (Espriella et al. 2020).

Drones offer the flexibility to survey at low tide and can produce imagery with very fine resolutions (i.e., centimeters or millimeters). The fine resolutions achievable by drones enable the delineation of reefs with limited concern for including adjacent habitat classes, therefore producing high—accuracy habitat maps. An additional benefit that drone imagery has over satellite or occupied aerial imagery is the ability to produce high—resolution digital surface models (DSM) using Structure from Motion photogrammetry. The DSMs can serve as an additional monitoring tool when characterizing reefs and how they change by quantifying their 2D and 3D structure (Espriella et al. 2023).

Between 1982 and 2011, there was an estimated 66% net loss in oyster reef coverage along Florida's Big Bend coastline (along the west coast where Florida's peninsula transitions to Florida's panhandle; Seavey et al. 2011). Establishing accurate baseline data is essential for the long—term monitoring of oyster resources and restoration efforts in the region. Drones offer a platform to rapidly survey coastal areas at high spatial resolutions. Accurately documenting oyster reef coverage and changes in coverage will better inform management by providing time—series datasets on the success of management efforts such as seeding settlement substrate (Coen et al. 2007). Identifying areas of persistence of oyster reefs may also aid in understanding the physical settings that encourage larval settlement and reef persistence (Fodrie et al. 2014, Hogan and Reidenbach 2019). This information can then be used to inform and enhance restoration efforts. The objective of this paper was to update oyster reef spatial data in the Suwannee Sound, FL and provide a new baseline for monitoring changes in reef coverage.

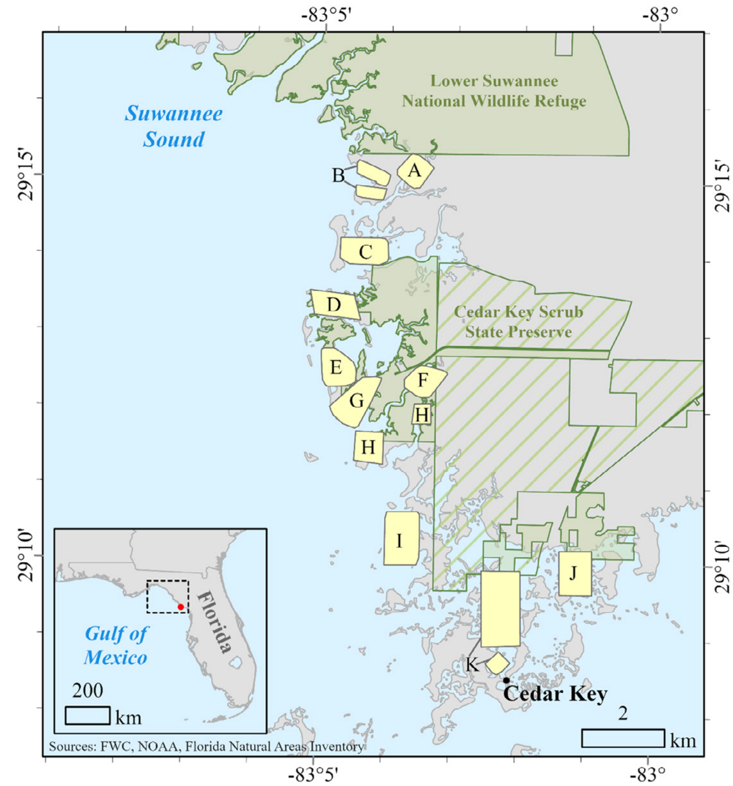

FIGURE 1. *Locations of the survey sites within the Suwanee Sound, FL. The Big Bend is outlined as the black rectangle within the inset map with the red circle representing the study area.*

Materials and Methods

Study Site

Intertidal oyster reefs were surveyed around Cedar Key, FL, USA, and within the Suwannee Sound (Figure 1). The Suwannee River is the primary source of freshwater to the Suwannee Sound and provides 60% of freshwater input to Florida's Big Bend coastline (Montague and Odum 1997, Mattson 2002). Intertidal oyster reefs, submerged aquatic vegetation, and salt marshes are present along the low—energy coastline (Mattson 2002). Oyster reefs are present in varying morphologies and settings including fringing reefs on the edges of embankments and salt marshes and patch reefs that form away from the shoreline (Radabaugh et al. 2019, Alonso et al. 2022). Oysters in this region are often intertidal and located near freshwater inputs due in part to the reduction in marine predators (Hine et al. 1988, Seavey et al. 2011, Radabaugh et al. 2019). The Suwannee Sound is the focus of the study in part because of the strong human interest and consequently strong management interest in the area and its resources. Oyster reef restoration is also ongoing in the region, such as the addition of limestone boulders as settlement substrate to restore the Lone Cabbage Reef (Frederick et al. 2015). Roughly 13 km separate the northernmost and southernmost sites surveyed in the study. Survey sites included a variety of settings such as tidal creeks, the landward side of barrier islands, and areas exposed to open waters (Figure 1).

Surveys and Image Processing

Surveys were conducted between July 2021 and March 2022 using a vertical take—off and landing (VTOL) Quantum Trinity F90+ drone equipped with a Micasense RedEdge—MX Camera. The camera system records 5 spectral bands: blue (475 nm), green (560 nm), red (668 nm), red—edge (717 nm), and near—infrared (842 nm). Surveys were performed as close to peak low tide as possible. Flying heights ranged from 100–110 m (Table 1). Surveys were pre—programmed using the QBase 3D v 2.3 software (QBase 3D 2021) and conducted in a single grid pattern. Across—track and along—track overlap were each at least 70% for all surveys. An iBase global navigation satel-

TABLE 1. *Survey parameters and tide information for all drone surveying flights. Tides are reported relative to Cedar Key, FL tide station (NOAA Tides and Currents 2023, station ID: 8727520) and are reported in mean sea level. Surveying site K required 2 flights. Flights are listed from North to South. Times are reported as local time with adjustment to Daylight Saving Time when appropriate. Sidelap refers to the overlap between adjacent survey lines, and frontlap refers to the overlap between successive images on the same survey line. GSD–ground sampling distance.*

Site	Date M/D/Y	Start Time	Tide at start (m)	Low tide time	Peak low Tide (m)	End Time	Tide at end (m)	Flying height (m)	Sidelap/ frontlap	GSD (cm)	Area (ha)
A	7/19/21	18:00	–0.35	16:54	–0.44	18:27	–0.30	110	75%/75%	7.64	30.4
B	3/17/22	08:56	–0.60	08:42	–0.60	09:23	–0.53	100	75%/75%	6.94	24.2
C	7/22/21	17:59	–0.43	20:00	–0.75	18:49	–0.60	105	75%/75%	7.29	76.9
D	3/17/22	09:27	–0.52	08:42	–0.60	10:11	–0.38	100	75%/75%	6.94	50.3
E	3/1/22	07:08	–0.88	07:36	–0.91	07:42	–0.91	110	74%/73%	7.64	49.9
F	3/1/22	08:09	–0.86	07:36	–0.91	08:43	–0.78	100	75%/72%	6.94	36.5
G	2/15/22	07:36	–0.83	07:42	–0.83	08:27	–0.77	100	75%/75%	6.94	68.7
H	3/4/22	08:33	–0.64	09:06	–0.68	09:18	–0.67	100	72%/72%	6.84	54.3
I	7/21/21	17:47	–0.50	19:06	–0/67	18:32	–0.62	110	75%/75%	7.64	71.7
J	3/2/22	07:28	–0.79	08:12	–0.86	08:09	–0.86	100	73%/70%	6.94	60.6
K1	3/3/22	07:31	–0.61	08:48	–0.80	08:30	–0.79	100	72%/72%	6.94	70.3
K2	3/3/22	08:36	–0.79	08:48	–0.80	09:32	–0.71	100	73%/73%	6.94	73.9

lite system (GNSS) base station logged reference measurements during each drone survey with a reported accuracy of 2—5 cm.

Following the surveys, image geotagging data were differentially corrected in the QBase software by using a post—processed kinematics (PPK) procedure that incorporated measurements from the iBase station. Elevation was recorded in reference to the WGS84 ellipsoid. Geotagged images were then exported from QBase and imported to Pix4D Mapper v 4.6.4 (2021). Within Pix4D mapper, an orthomosaic was generated for each spectral band for each respective scene, along with a DSM for each scene. All orthomosaics and DSMs were exported at the native resolution for each scene (Table 1).

Reef delineation was conducted using the multi—resolution segmentation algorithm in eCognition Developer 9.5 (2019), an object—based image analysis software. A segmentation algorithm was used rather than manually drawing polygons to limit any subjectivity introduced with manual delineation and to maintain a consistent level of detail depicting reef boundaries. Segmentations were tested for each spectral band and the DSM to visually assess which produced the most consistent delineations. The near—infrared band outperformed all other bands, likely in part due to its ability to differentiate between water and non—water bodies (Mondejar and Tongco 2019). Therefore, the near—infrared band was used to segment each scene with a scale parameter of 300, shape parameter of 0.1, and compactness parameter of 0.5. The scale parameter of 300 produces relatively small objects which were then manually merged to a single object per oyster reef. Objects were visually inspected to identify any objects that included both oyster reef and non—oyster reef cover types. Any objects that included other cover types were manually edited to exclude that area from the delineation. Oyster reef objects were then manually classified and exported from eCognition Developer as a shapefile for further analysis. Geometric features such as rectangular fit, maximum length and width, and asymmetry along with contextual features such as proximity to nearby reefs and marsh were exported with the shapefile to characterize each reef (see **Supplementary Material**).

The following criteria for delineating reefs were adapted from Patterson (2002; numbers 1–4 below) and Garvis et al. (2020; number 3 below), with additional criteria added (number 5 below). These conventions were largely developed for studies using occupied aircraft aerial imagery. However, these patterns remain consistent with drone imagery.

1. Linear or oval shape with some having branching arms.
2. Often occur in association with *Spartina* emergent vegetation.
3. Dark margins with a slightly lighter middle area or bright white signature due to the "hash" (dead oyster shells) that have been bleached due to exposure to the sun. The hash area typically occurs directly adjacent to living oyster bars. During photointerpretation the hash areas are included as part of the oyster bar.
4. Parallel ridges formed by remnant oyster shells deposited by waves washing over the bar.
5. Oyster aggregation must be continuous and have a distinct geometry to be classified as a reef.

Reefs were included if they were a minimum of 2 m^2 and completely contained within the survey bounds. While some areas of reef were not classified as a result, this ensures that reefs are depicted in their entirety, and there are no artificially straight reef outlines because of the survey boundary. Strict interpretation was used when defining the reef boundary. For example, assumptions were not made about reef being continuous around a marsh patch (Figure 2). If oysters were continuously visible around a patch of marsh, the marsh area was included in the delineation. The rationale for including the marsh was threefold: in the field we often observe oysters within the marsh patches, the marsh patch is within the geometry of the reef footprint, and this is consistent with previous mapping efforts. However, if a patch of marsh had fringing oysters that are discontinuous, each section was separately delineated. This is an important distinction, as the inclusion or exclusion of these regions of marsh adjacent to oyster reef can have a significant impact on reef area calculations. Additionally, oysters that were submerged were not delineated, even if visible in the imagery.

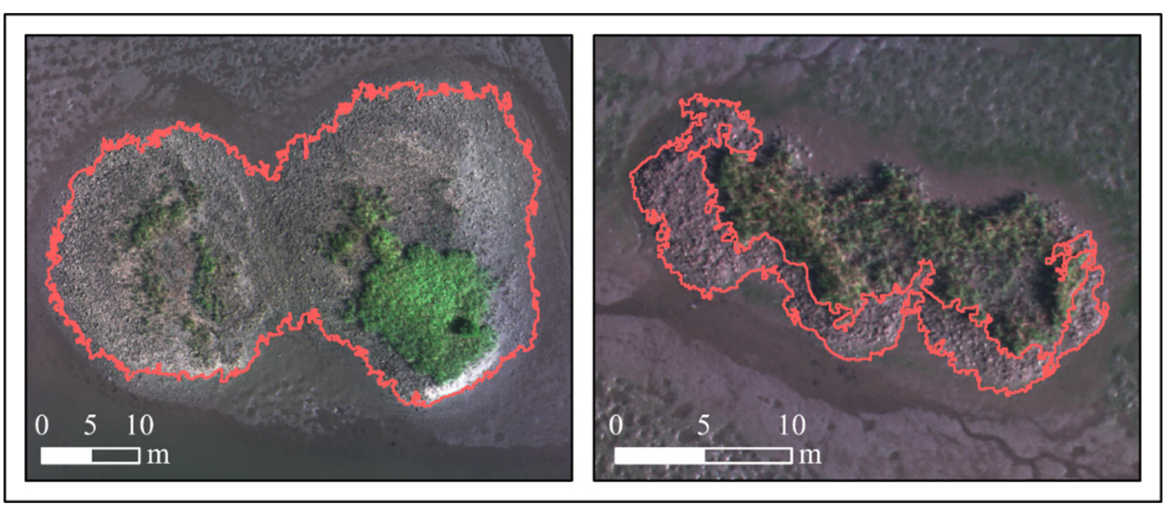

FIGURE 2. *Example of how oysters around marsh patches were delineated based on context. When oysters surrounded the marsh entirely (left), the marsh was included in the delineation. When oysters were discontinuous around the marsh (right), the marsh area was excluded.*

TABLE 2. *Summary statistics representing geometric data for delineated reefs.*

	Area (m²)	Perimeter (m)	Perimeter to area ratio	Max Length (m)	Max Width (m)	Volume to area ratio
Mean	357.01	232.55	1.61	26.29	14.61	0.30
Median	73.63	100.02	1.46	14.94	9.43	0.28
Min	2.22	8.27	0.16	2.07	1.53	<0.01
Max	23,792.10	12,785.17	5.69	560.66	346.40	1.88
Std. dev.	1,225.10	588.08	0.96	38.38	19.89	0.16
Skewness	10.84	11.79	0.83	5.82	7.40	2.07
Kurtosis	153.99	195.20	0.47	53.66	85.72	11.13

The exclusion of submerged oysters was for multiple reasons: the lack of spectral variability over water results in poor segmentations, it is more difficult to confidently discern what is reef and what is not when submerged, and water clarity will vary depending on when the survey is conducted.

Oyster reef shapefiles were merged to one layer and compared to the existing 'Oyster Beds in Florida' shapefile (accessed on 4/28/2023) made available by the Florida Fish and Wildlife Conservation Commission (FWC) Fish and Wildlife Research Institute (FWRI). The layer combines numerous mapping studies into one shapefile. Contributors to the data compilation include the Suwannee River Water Management District, FWRI, and the United States Geological Survey. However, these data are often outdated, with some reefs last delineated in 2001. Differences between the generated and existing layers were determined by identifying areas of overlap in ESRI ArcGIS Pro v 2.4 (2019) using the Select by Location tool. The existing layer was clipped to the drone survey extents to allow for direct comparison. Reefs that were partially within the survey bounds were only included when they corresponded with reef area in the updated layer for comparison purposes. In this instance, only the reef area outside of the survey bounds was excluded. This comparison allowed for calculations of existing reef areas that were updated and reefs that were mapped for the first time.

To characterize the reefs using DSMs, the shapefile was used as a mask to extract areas in the DSMs that corresponded with reefs. Areas of marsh within a reef footprint were masked out of the scene during this process, as these areas would inflate the volume derived from the DSM. Reef DSMs were extracted using the 'mask' function in the *raster* package in R v 4.2.3 (Hijmans 2022, R Core Team). These DSMs representing reefs were used to calculate a volume to area ratio for each reef using the 'Surface Volume' tool in ArcGIS Pro, as this metric was demonstrated to be a potential indicator for oyster reef condition (Espriella et al. 2023). For each reef, the minimum elevation within the extracted area was used as the reference plane for the 'Surface Volume' tool.

Results

About 8.9 km^2 of intertidal habitat was surveyed. A total of 1,394 reefs were delineated, representing 497,670 m^2 of reef area. Of the 1,394 delineated reefs, 236 (17%) had no overlap with previously mapped oyster reefs, corresponding with 19,848 m^2 of newly mapped intertidal oyster reef habitat. The high spatial resolution of the drone enabled the delineation of smaller reefs with 428 of the newly delineated reefs being <30 m^2. Conversely, 74 reefs in the existing dataset were <30 m^2 within the surveyed areas. Oyster reefs delineated in the present study ranged in size from 2 m^2 to 23,792 m^2 (Table 2). Some shadowed areas and reconstruction errors resulted in artifacts in the DSMs corresponding with reefs. As a result, 156 reefs were excluded from volume to area ratio calculations leaving a total of 1,238 included reefs.

While the number of reefs delineated in the new layer is higher than the existing data, it should be noted that this is partially due to larger reefs in the original layer being interpreted as many smaller reefs in the new layer (Figure 3). This difference happened most frequently with fringing reefs where small gaps existed between contiguous oyster patches.

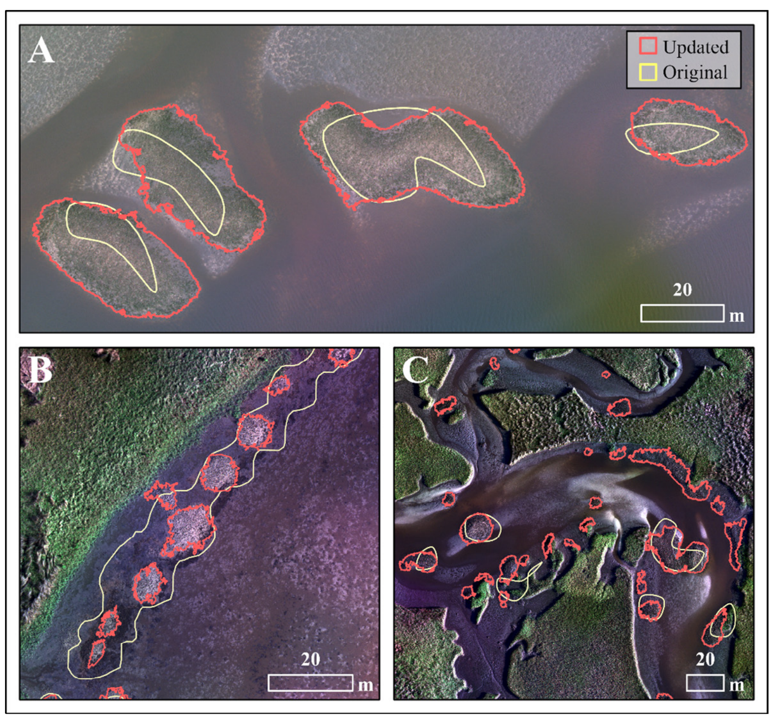

FIGURE 3. *Examples of original and updated oyster reef mapping. A. An area that underestimated reef coverage in the original layer. B. Fringing oysters that were classified as smaller patches in the updated layer, but one continuous reef in the original layer. C. An area with multiple newly classified reefs.*

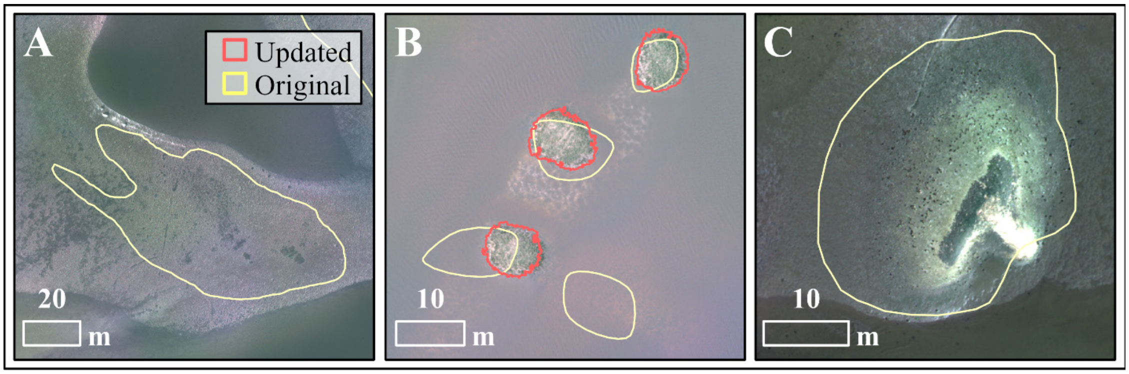

FIGURE 4. *Instances where the original layer indicates the presence of oyster reefs while the updated layer does not. A. Mudflats. B. Areas inundated with water at the time of the drone survey. C. An area of sparsely populated oysters.*

Within the drone surveyed areas, the original layer provided by FWC contains 910 reefs representing 910,020 m^2. A total of 178 reefs were identified in the original layer that were not delineated in the updated layer for a total area of 222,804 m^2. Those 178 classified areas are most commonly mudflats (85), which accounted for 147,340 m^2 (66%) of reef area delineated in the original layer but not the updated layer. Additional differences between the original layer and the updated layer include areas that were very sparsely populated with oysters and did not meet the criteria designated for a reef in the updated layer (26). Other differences include areas inundated with water at the time of the survey (14), areas at the edge of the surveyed area where reefs were excluded due to reconstruction artifacts (9), areas located in heavily shadowed areas (6), salt marsh (6), or some combination of these factors (32) (Figure 4).

Discussion

The reef delineations produced a total of 236 newly mapped reefs, representing 19,848 m^2 of area. The delineations also updated boundaries for 1,158 reefs in the Suwannee Sound. The high resolution of the drone imagery in combination with the segmentation algorithm provided detailed reef outlines that better represent reef morphology and complexity when compared to manual delineations from satellite or occupied aircraft imagery. The outlines generated by the segmentation also ensure consistent detail for all reefs. Of the newly delineated reefs, 66% were under 30 m^2, highlighting the utility of high—resolution drone imagery in classifying smaller reefs that are difficult to discern in other imagery sources. The high—resolution imagery facilitated the distinction between oysters and surrounding habitats, identifying areas of salt marsh or mudflats that were erroneously classified as oysters. Previous classification keys used *Spartina* presence as an indicator of oyster habitat, as at times it is the only cover visible in satellite or occupied aerial imagery (Patterson 2002). Additionally, the 147,340 m^2 of mudflats that were previously classified as oysters further illustrates the shortcomings of coarser imagery resolutions. Although there is the potential that mudflats misclassified as oyster reefs transitioned to mudflats between mapping efforts, there are misclassifications as recently as 2022. The high resolution achievable with drone imagery allows for confident delineation of oyster reefs from surrounding habitats.

The number of reefs added should be interpreted within context, as there are instances where one large reef from the original map is delineated as multiple smaller reefs in the current study. Although separating reefs into distinct patches provides a more accurate depiction, this can inflate the total number of reefs when comparing to existing maps. Additionally, the minimum reef size in the updated layer is 2 m^2, which is relatively small. There are no universal guidelines or consensus regarding the definition of an oyster reef. Here we included relatively small reefs to highlight the potential of drone imagery as a monitoring tool. All reefs 2 m^2 or larger were classified using the same criteria, regardless of size. Consistent monitoring using high—resolution imagery will allow researchers to quantify oyster reef habitat fragmentation, which has been documented in Florida (Benson et al. 2023). Given the coarse resolution of historical baseline imagery, it is challenging to differentiate between reef fragmentation and misclassifications or differences in methodology.

While observing changes in reef area is an important component in monitoring, it is also essential that these data are interpreted within context. For example, oyster collapse can result in an increase in reef area as eroded shell expands over a wider area of substrate (Radabaugh et al. 2019). Reef delineations and area should be evaluated in conjunction with metrics that can provide information on reef condition. Drone surveys can produce DSMs, which can provide numerous structural complexity metrics that offer additional insight into reef condition (Windle et al. 2022, Espriella et al. 2023). The inclusion of metrics such as volume to area ratio in monitoring efforts can aid in identifying areas prone to decline with more advance warning. Measuring structural complexity metrics consistently can provide valuable time—series datasets that can in turn be used to identify areas and contexts where reefs persist, informing management and restoration.

Limitations and considerations

Despite the advantages of drone imagery, some limitations persist. While drone surveys limit the need to access reefs directly, you must still be on site and have access to a suitable launch area and time. This may pose challenges in difficult to access intertidal areas, especially when attempting to survey at low tide when intertidal reefs are exposed. Selecting a suitable launch area involves assessing risk presented by obstacles and following all Federal Aviation Administration Part 107 guidelines for operating small unoccupied aircraft systems. For the Quantum Trinity F90+ used in this study, there is a recommended 20 m obstacle—free radius for safe take—off and landing. Atmospheric conditions can also pose challenges, as shadows from the sun angle can make it difficult to differentiate oyster reef from surrounding habitats and introduce artifacts in the DSM (James and Robson 2012). To avoid shadows, surveys can be conducted on overcast days or near solar noon whenever possible. However, this can be difficult as surveys of intertidal areas must also align with the tides. Many surveys conducted in this study were flown in early morning or late evening hours to coincide with low tides, resulting in some shadowed areas over oyster habitat caused by adjacent vegetation. Wind speed and direction is an additional consideration when conducting drone surveys, as pilots must be sure that conditions are within the limits of the drone's safe operation. For example, the Trinity F90+ has a wind tolerance of 9 m/s on the ground and 12 m/s while surveying. It should also be noted that as a fixed—wing drone the Trinity F90+ can cover larger areas than multi—rotor counterparts. Across the conducted surveys, the Trinity F90+ surveyed an average of 1.24 ha/min.

Delineating oyster reefs by manually drawing polygons or by using a segmentation algorithm to manually classify reefs can both be time consuming. Given the benefits of frequent and consistent oyster reef mapping, it may be more feasible to use semi—automated habitat classification techniques (Espriella et al. 2020, Ridge et al. 2020, Chand and Bollard 2021, Espriella and Lecours 2022). Using semi—automated classifications can yield overall accuracies of 80% or higher, making them a viable solution for consistent monitoring (Chand and Bollard 2021, Espriella and Lecours 2022). Semi—automated and repeatable workflows can encourage more consistent monitoring as the image interpretation time decreases significantly.

Conclusions

There is a critical need to increase monitoring of oyster resources to preserve the services and fisheries they provide. Conventional methods are time and labor intensive, limiting the ability to identify reef deterioration in time to address the decline. Further, photointerpretation that relies on satellite or occupied aircraft imagery often does not produce the resolution required to make confident conclusions about reef presence or morphology. This can result in misclassifications that produce inflated estimates of reef area. Alternatively, smaller reefs can be missed due to the coarse resolution. Drone surveys that produce high—resolution imagery expedite monitoring efforts and can help management pinpoint areas of concern, allowing more time to adjust regulations accordingly. Drone imagery also enables the production of DSMs which can provide more monitoring data including elevation, reef shell volume, and surface complexity metrics. Here, we updated baseline data that can be used for oyster monitoring using high—resolution drone surveys. Our results showed notable differences between the existing dataset and our data, which will help inform decision—making and future monitoring. Consistent drone surveys have the potential to serve as an additional tool to monitor these dynamic systems with little to no direct impact on the reefs themselves.

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