Original Article

Seagrass cover reduction in Zanzibar from 2006 to 2019

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Abstract

Seagrass meadows are abundant in Zanzibar, Tanzania and provide essential ecosystem services, yet they have been less researched or protected than other marine ecosystems. Evidence suggests that anthropogenic impacts, such as seaweed farming, small-scale fishing, and tourism, are their greatest threats. Using geographic information systems (GIS) to estimate seagrass cover, this study conducted Spearman's rank correlation analyses to estimate correlations between the scale of seaweed farming, fishing, and tourism or the implementation of marine protected area (MPA) management plans, and the change in percent of seagrass cover from 2006 to 2019. On average, seagrass cover decreased by 10.98 % over this period. The scale of tourism was negatively correlated with seagrass cover ($r_s(9) = -0.64$, $p = 0.044$). No other variables were correlated with declines, though seaweed farming had a weak protective effect on seagrass cover ($r_s(9) = 0.28$, $p = 0.426$). To improve the management of seagrass meadows, plans must identify seagrasses as critical ecosystems, expand seagrass restoration projects, and address harmful practices in the tourism industry.

Keywords: seagrass, change detection, anthropogenic impact, MPA, coastal management

Introduction

Seagrasses have been less commonly researched or protected than other flora and fauna in tropical coastal marine environments in Africa. However, there is increasing evidence that seagrasses are essential to humans and coastal environments and must be considered in coastal marine management (Githaiga *et al.*, 2016; Staehr *et al.*, 2018). Due to a lack of data on density and areal extent of seagrass meadows and their threats in Zanzibar, Tanzania (Aller *et al.*, 2019; Staehr *et al.*, 2018), coastal marine management plans cannot adequately target specific threats and address their underlying factors.

Seagrass meadows are located in the subtidal and intertidal zones around the islands of Zanzibar (Aller *et al.*, 2019; Khamis *et al.*, 2017); there have been 12 documented species in East Africa dominated by the climax species *Thalassodendron ciliatum* and *Thalassia hemprichii* (Lyimo *et al.*, 2008; Ochieng and Erftemeijer, 2003). They provide extensive ecosystem services in tropical and temperate regions around the world (Nordlund *et al.*, 2016; Staehr *et al.*, 2018). Seagrasses accumulate and stabilize sediment in the intertidal and subtidal zones, which facilitates coastal geomorphology and provides coastal protection by dampening waves (Belshe *et al.*, 2018; Nordlund *et al.*, 2016; Staehr *et al.*, 2018). They also fertilize sediment with nitrogen and other nutrients, creating ideal environments for feeding grounds, nurseries, and habitats for aquatic and nonaquatic species (Belshe *et al.*, 2018; Nordlund *et al.*, 2016; Staehr *et al.*, 2018). Globally, seagrasses also sequester an estimated 50-64 % of global organic carbon and a substantial proportion of blue carbon (Nchimbi and Lyimo, 2019; Nordlund *et al.*, 2016). One study concluded that the East African coast had the greatest total seagrass biomass in Africa, followed by the Southern Mediterranean coast (Githaiga *et al.*, 2016). Seagrasses are essential to coastal ecosystems and for mitigating climate change.

Previous research has focused primarily on the impacts of seaweed farming, small-scale fishing, and coastal tourism activities as drivers of seagrass degradation in Zanzibar. Commercial seaweed farming was introduced to Zanzibar in 1989 in Jambiani and Paje and expanded to Chwaka Bay in 1990 (Eklöf *et al*., 2012; Msuya, 2013). Commercial seaweed is the largest marine export from Zanzibar, with production reaching >16,500 metric tons in 2016 (Kamer, 2022), and seaweed farming remains a high-volume economic venture for several coastal villages (Moreira-Saporiti *et al*., 2021; United Republic of Tanzania, 2008). The historical practice of uprooting seagrass meadows to remove sea urchins that destroy seaweed has mostly ceased in favor of situating farms in nutrient-rich seagrass meadows that maintain low surface temperatures (Hedberg *et al*., 2018; Lyimo *et al*., 2006). The majority (92 %) of seaweed farms in Zanzibar are located at least partially on seagrass meadows, demonstrating their interlinkages (Hedberg *et al*., 2018). Physical disturbances like trampling or removing seagrasses and sediment disturbance can cause degradation, including a loss of seagrass biomass, shoot length, and cover (Lyimo *et al*., 2006; Moreira-Saporiti *et al*., 2021). Situating seaweed farms above seagrass meadows shades seagrasses and hinders their ability to photosynthesize (Lyimo *et al*., 2006). Among two rural villages where 50-60 % of villagers engaged in seaweed farming, Nchimbi and Lyimo (2019) found that seagrass meadows were visibly degraded and had lower shoot heights, biomass, and density.

The preferred fishing grounds for small-scale fisheries in Zanzibar are seagrass-dominated areas (de la Torre-Castro *et al*., 2014; Hedberg *et al*., 2018), yet due to the limited monitoring of seagrass, it is unclear how small-scale fishing practices have impacted seagrass meadows (Staehr *et al*., 2018). Seagrass meadows and proximate coral reefs support 70 % of small-scale fishing by providing habitats, nurseries, and feeding grounds for finfish, prawns, and bivalves (Khamis *et al*., 2017; Staehr *et al*., 2018; UNEP-Nairobi Convention and WIOMSA, 2021). The fishing industry, which provides livelihoods for about one-fifth of the Zanzibar population, primarily uses traditional low-technology gear, such as seine nets and wooden basket traps, and vessels, such as sail-powered dhows and canoes (de la Torre-Castro *et al*., 2014; Jiddawi and Öhman, 2002; Khamis *et al*., 2017; Staehr *et al*., 2018). Annual fish catches have increased from 4,100 tons in 1980 to 34,100 tons in 2015 (Staehr *et al*., 2018), yet the fish catch per fisherman has decreased, suggesting overfishing is occurring (Khamis *et al*., 2017). Destructive fishing techniques, such as the illegal use of seine nets, spear-guns, and dynamite, degrade and destroy seagrass meadows and coral reefs (Jiddawi and Öhman, 2002; Khamis *et al*., 2017). Overfishing and destructive fishing methods are increasingly used to satisfy the swelling demands of a rapidly growing population of inhabitants and tourists (Staehr *et al*., 2018). However, most Zanzibari fishermen make an income of less than 6 USD per day and cannot afford to alter their practices (de la Torre-Castro *et al*., 2014), and there is limited capacity to enforce laws that protect the coastal environment (Jiddawi and Öhman, 2002; Khamis *et al*., 2017).

Since 1985, tourism has grown more than sixteen-fold (Staehr *et al*., 2018), comprising 27 % of Zanzibar's gross domestic product in the mid-2010's (Khamis *et al*., 2017). Hoteliers build lodgings along the most attractive sections of the coast, clearing the coastline of seagrass meadows and dredging the seafloor of muddy silt, which is pushing the ecological capacity of coastal marine ecosystems to a state of degradation or destruction (Khamis *et al*., 2017). These harmful practices compromise the seafloor integrity, the health of seagrasses, coral reefs, and mangrove forests, and the welfare of vertebrates and invertebrates that depend on them (Khamis *et al*., 2017; Staehr *et al*., 2018). Other impacts of the booming tourism industry include the runoff of pollution and waste into the ocean and increased oil spills from maritime transport (Khamis *et al*., 2017; Staehr *et al*., 2018). Few hotels have onsite water treatment facilities, and many hotels discard waste directly into the ocean (Khamis *et al*., 2017; Staehr *et al*., 2018). Zanzibar does not have systematic waste management on the islands to manage the large increase of solid and sewage waste, which increasingly contain plastic products (Staehr *et al*., 2018). There is an inverse relationship between biomass of seagrass meadows and coastal development, which means that as the Zanzibar population, mariculture, and tourism increase, seagrass meadow densities decrease (Khamis *et al*., 2017; Staehr *et al*., 2018).

Although seagrasses are vulnerable to environmental changes, evidence suggests that they typically recover quickly to changing environmental conditions such as changing water temperature (Aller *et al*., 2019). For example, Aller *et al*. (2019) measured how changes in cloud cover, sunspot activity temperature, tidal amplitude and height, and storm occurrence affected seagrass cover and composition of six species in three transects of the Chumbe Island protected area over a 10-year period. Each transect gradually declined and then increased in composition and cover within the end of the 10-year period, demonstrating the resilience of the seagrass meadows to environmental variables. Although researchers speculate that increasing sea temperatures, sea levels, and storm occurrences due to climate change may be long-term environmental threats to seagrass viability, there is currently no evidence to suggest that environmental changes have affected long-term seagrass viability (Aller *et al*., 2019; Khamis *et al*., 2017; Lyimo *et al*., 2008; Staehr *et al*., 2018).

Zanzibar has made strides to protect coastal marine environments. Multiple public and private marine protected areas (MPAs) cover the majority of the Zanzibar's 370 km of coastline, protecting approximately 1,300 km^2 of Unguja, the main island commonly referred to as Zanzibar (see Table 1; IUCN, 2020). Private MPAs strictly regulate no-take zones in which mariculture is prohibited to enhance conservation efforts, whereas public MPA zones are regulated to ensure sustainable use of marine ecosystems, such as preventing overfishing (IUCN, 2020). Notably, seagrasses have not been identified as important or vulnerable ecosystems in coastal marine management plans, yet they may benefit from the protections implemented to preserve biodiversity and protect adjacent ecosystems, such as mangroves and coral reefs (Cullen-Unsworth *et al*., 2014; de la Torre-Castro *et al*., 2014; Unsworth and Cullen, 2010; Unsworth *et al*., 2018).

To date, previous studies on seagrasses in Zanzibar have focused on the overall health of seagrasses within particular regions of Zanzibar (e.g., Chwaka Bay) or specific anthropogenic impacts (e.g., seaweed farming) on seagrasses. Yet absent from the discussion is whether particular anthropogenic activities have caused greater harm to seagrasses over time and whether implementing MPA management plans has had a protective effect. Using geographic information systems (GIS) satellite imagery and field data, the current study assessed the percent of seagrass cover in five sites located within MPAs around Unguja Island, Zanzibar in 2006 and 2019 to evaluate whether 1) coastal activities (seaweed farming, small-scale fishing, and tourism) at varying scales, and 2) the implementation of MPA management plans are correlated with long-term changes in seagrass cover. It was hypothesized that high-intensity tourism will have the largest negative correlation with seagrass cover, and the implementation of MPA management plans will have the largest positive correlation.

Materials and methods

Study area

The study area included seagrasses on the coast of Unguja Island, which is located approximately 30 km off the coast of eastern Africa in the Western Indian Ocean. Zanzibar's climate is tropical and defined by two rainy seasons from March to May ("Masika" or the long rains) and October to December ("Vuli" or the short rains), and the Monsoon wind system influences the local currents with slightly stronger winds during the March-May rainy season (Khamis *et al*., 2017; Staehr *et al*., 2018). The coastlines alternate between rocky terrain, sandy beaches, and mangrove forests, and the coastal marine environments often include dense seagrass meadows, algae, and fringing coral reefs.

Table 1. Established marine protected areas (MPAs) on Unguja Island, Zanzibar, modified from IUCN (2020).

MPA	Declaration date	Status	Management plan implementation date	MPA area (km^2)
Chumbe Island Coral Park (CHICOP)	1994	Private MPA on western side of island	1994	0.55
Mnemba Island-Chwaka Marine Conservation Area (MIMCA)	2002	Public MPA, no-take zone around Mnemba Atoll	2010	337.3
Menai Bay Conservation Area (MBCA)	1997	Public MPA	2010	717.5
Tumbatu Marine Conservation Area (TUMCA)	2015	Public MPA	Not implemented	162.9
Changuu-Bawe Islands Marine Conservation Area (CHABAMCA)	2015	Public MPA	Not implemented	118.2

Most seagrass beds and meadows are located in shallow water depths of less than 5 m, and meadows are typically comprised of two or more seagrass species (Aller *et al*., 2017; Belshe *et al*., 2018; Ochieng and Erftemeijer, 2003). The dominant seagrass species in Zanzibar include *T. ciliatum*, *T. hemprichi*, *Cymodocea rotundata*, *Cymodocea serrulata*, and *Enhalus acoroides* (Lyimo *et al*., 2008).

There have been two major El Niño Southern Oscillation (ENSO) events in the Western Indian Ocean in the last 30 years, the first lasting from 1997-1998 and the second lasting from 2014-2016 (Lin *et al*., 2018; Nowicki *et al*., 2017). Heatwaves resulting from ENSO events have caused episodic declines in seagrasses, and there is currently little research on the recovery time for seagrasses following ENSO events, especially for seagrass meadows on the East African coast (Nowicki *et al*., 2017; UNdata, 2021).

Site selection

Seagrasses were mapped in five sites within MPAs on and off the coast of Unguja: Chumbe Island, Chwaka Bay, Fumba, Jambiani, and Nungwi (see Fig. 1). Sites were selected that represented a diversity of MPA protections and coastal activities. Chumbe Island, located 6 km off the southwest coast of Unguja, is a small private island surrounded by coral reefs and seagrasses with one small ecolodge. The western side of the island is a no-catch zone established and managed by the Chumbe Island Coral Park (CHICOP) MPA. Chwaka Bay is a large economically important bay on the eastern central coast that supports mariculture activities, including small-scale fishing and seaweed farming. Nungwi is the northernmost village on the island and economically relies on large-scale tourism and small-scale fishing. The Mnemba Island-Chwaka Marine Conservation Area (MIMCA) includes the coasts of Chwaka Bay and Nungwi. Jambiani is located on the southeastern coast of Unguja and supports small-scale fishing, seaweed farming, and large-scale tourism. Fumba is located on the southwest coast and relies primarily on small-scale fishing immediately off the coast. The Menai Bay Conservation Area (MBCA) includes the coasts of Jambiani and Fumba.

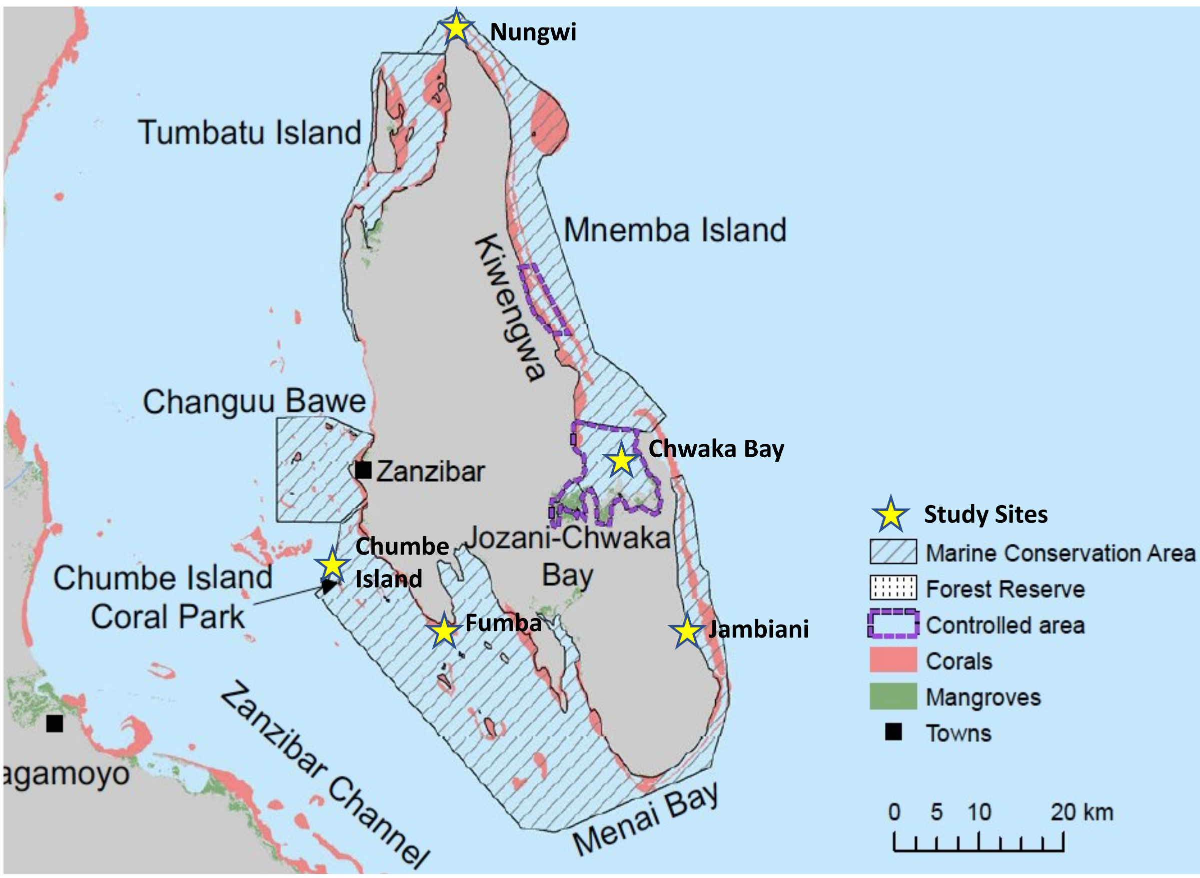

Figure 1. Study site map with MPA boundaries, modified from WIOMSA (UNEP-Nairobi Convention and WIOMSA, 2021).

Data sources

The study used Google Earth satellite images taken between 2005-2007 and 2019-2021, existing field data from a data repository, regional statistics, and evidence from peer-reviewed articles to map and analyze seagrass and habitat characteristics as well as MPA management plans and coastal activities. To compare the distribution and characteristics of seagrasses over time, site-specific satellite images and field data were included. The field data used for validation are part of SeagrassNet (Global Monitoring Network), were collected by CHICOP, Ltd., and were made available through the database at www.seagrassnet.org. Field data included the percentage of seagrass cover per 0.25 m^2 quadrat collected by CHICOP, Ltd. in October 2006 and September 2019, which aligned with the time periods of the Google Earth images to ensure field samples and satellite images reflected the same ecological conditions. Regional statistics, peer-reviewed articles, and Google Earth images provided information about the implementation and scale of MPA management plans and coastal activities, including seaweed farming, small-scale fishing, and tourism, occurring in coastal villages around Unguja. Due to the limited availability of data on the scale of coastal activities, all available relevant data was included.

Time period selection

The study compared the change in the percentage of seagrass coverage at each site from 2006 to 2019. Appropriate baselines were determined for seagrass cover based on timelines of the growth of Zanzibar's overall economy, mariculture, and tourism sectors and the availability of high-quality satellite images. Zanzibar's per capita GDP has steadily increased from 445 USD in 2006 to 1,111 USD in 2019 (UNdata, 2021), indicating a nearly threefold increase in capital in the measurement period. Tanzania's overall seaweed production increased from approximately 7,000 tons in 2004 to 10,000 tons in 2019 (Kamer, 2022; Msuya, 2020). Following a decline in fish catches in the 1980's, annual artisanal fish catches in Zanzibar have steadily increased from >20,000 tons in 2006 to >30,000 tons in 2016 (Rehren *et al.*, 2020). The number of tourists visiting Zanzibar has increased fivefold from >100,000 tourists in 2005 to >500,000 in 2018 (World Bank, 2019).

In addition to considering the increasing intensity of coastal activities, the limitations in available historical satellite imagery and field data had to be factored in. The earliest available high-quality satellite images on Google Earth Engine were between 2005 to 2007 for all sites, and CHICOP, Ltd. collected its first set of field data in 2006.

Study variables

Each site was assigned codes in 2006 and 2019 based on the relative scale of the following variables: MPA management plans, seaweed farming, small-scale fishing, and tourism. For MPA management plans, each site received a "1" or "2" if the site was contained within a partial or full MPA, respectively, that implemented a management plan for at least two years. A two-year minimum implementation period was included to account for the time it takes to implement new policies that lead to changes in practices in the use of coastal marine ecosystems. Sites contained outside of MPA boundaries or in MPA boundaries without management plans implemented for > two years were assigned a "0".

To assign the scale of seaweed farming at each site, the number of seaweed farmers were used as a proxy for assessing the impact of trampling, shading, and removal of seagrasses. The Joint Frame Survey scale of the number of seaweed farmers in each region of Unguja was used to assign codes (United Republic of Tanzania, 2008). Since the quantity of seaweed farmers is aggregated and reported at the regional level, sites that produced commercial seaweed within regions with >1,000 seaweed farmers were coded with "2"; commercial seaweed producing sites within regions with <1,000 seaweed farmers were coded with "1". Sites that did not harvest commercial seaweed were coded with "0".

Due to a lack of comprehensive fisheries data available for each site, a proxy of the number of fishermen was used to determine the scale of small-scale fishing at each site. As the number of fishermen increases, the number of destructive fishing practices and harmful equipment (e.g., use of seine nets and fishing boat engines) that can damage seagrasses are likely to increase as well. The Joint Frame Survey scale for the number of fishermen in each region was used to assign codes; sites within regions with >4,000 fishermen were coded with "3", sites within regions with 2,501-4,000 fishermen were coded with "2", and sites within regions with ≤2,500 fishermen were coded with "1" (United Republic of Tanzania, 2008). Sites within no-take zones were coded with "0". It should be noted that the mapping boundaries for Chumbe Island were limited to the no-take zone on the western side of the island.

The number of hotels situated on the coastline within each site mapping boundary were used as a proxy for the tourism variable, the same method employed by Khamis *et al*. (2017). This indicator is a suitable measure of the impact of tourism because the construction and operation of hotels on the coast can directly contribute to removal of seagrass meadows and other vegetation, coastal erosion, sewage and solid waste runoff, and increased degradation of coastal ecosystems due to higher volumes of tourists swimming in the intertidal and subtidal zones. Google Earth was used to identify and calculate the percent of hotels on the coastline in 2019. Sites with <10 %, 10-50 %, or >50 % of hotels located on the coastline were assigned a "1", "2", or "3", respectively. Due to a lack of available data on the number of hotels on the coast in 2006, it was estimated that the scale of tourism in 2006 was one-third of the volume in 2019 since the number of tourists had increased more than threefold from 2006 to 2019 (World Bank, 2019). Each site was, therefore, assigned a "1" for low-scale tourism in 2006.

A variable for the combined score of coastal activities was also created to assess whether coastal activities had a collective impact on seagrass health. For example, Chwaka Bay's combined score for coastal activities in 2006 was calculated as follows: 3 (high-scale seaweed farming) + 3 (high-scale fishing) + 1 (low-scale tourism) = 7. The MPA management plan status and scale of coastal activities for each site in 2006 and 2019 are displayed in Table 2.

Developing coastal maps

Remote sensing methods have been utilized globally to map the distribution of seagrass meadows and measure habitat characteristics, such as seagrass biomass, water depth, and water quality (Amran, 2017; Hossain *et al*., 2016; Knudby and Nordlund, 2011). The distribution of seagrass beds and meadows at both time points using Google Earth Engine images was mapped first at a resolution of approximately 15 m. Only seagrass beds and meadows located in relatively shallow areas (<5 m) were included to ensure data quality. Seagrass beds and meadows were mapped in heterogeneous polygons to measure the area of each patch.

For sites that did not have clear satellite images in 2006 or 2019 or during the same corresponding seasons, the primary image between the range of 2005-2007 and 2020-2021 was selected and the mappings compared to 2006 and 2019 images for accuracy. Images of final site mappings are displayed in Figures 2a-j, and the seasons during which each image was taken are listed in the figure caption. One challenge to mapping was distinguishing algae-covered coral reefs from seagrasses in sites with high coral mortality, such as at Nungwi, so potential seagrasses located in coral reefs were not included unless seagrass meadows within coral reefs were visibly dense (for example, see Fig. 3).

Once the mappings were completed, the percentage of seagrass cover for each site was calculated by

Table 2. Site characteristics of MPA management plan status and scale of coastal activities in 2006 and 2019.

Site	MPA zone	Year	Management plan status	Scale of Coastal Activities		
				Seaweed farming	Small-scale fishing	Tourism
Chumbe Island	CHICOP	2006	Implemented – Full MPA	None	None	Low
		2019	Implemented – Full MPA	None	None	Low
Chwaka Bay	MIMCA	2006	None	High	High	Low
		2019	Implemented – Partial MPA	High	High	Medium
Fumba	MBCA	2006	None	None	Medium	Low
		2019	Implemented – Partial MPA	None	Medium	Low
Jambiani	MIMCA	2006	None	High	High	Low
		2019	Implemented – Partial MPA	High	Low	High
Nungwi	MBCA	2006	None	None	High	Low
		2019	Implemented – Partial MPA	None	High	High

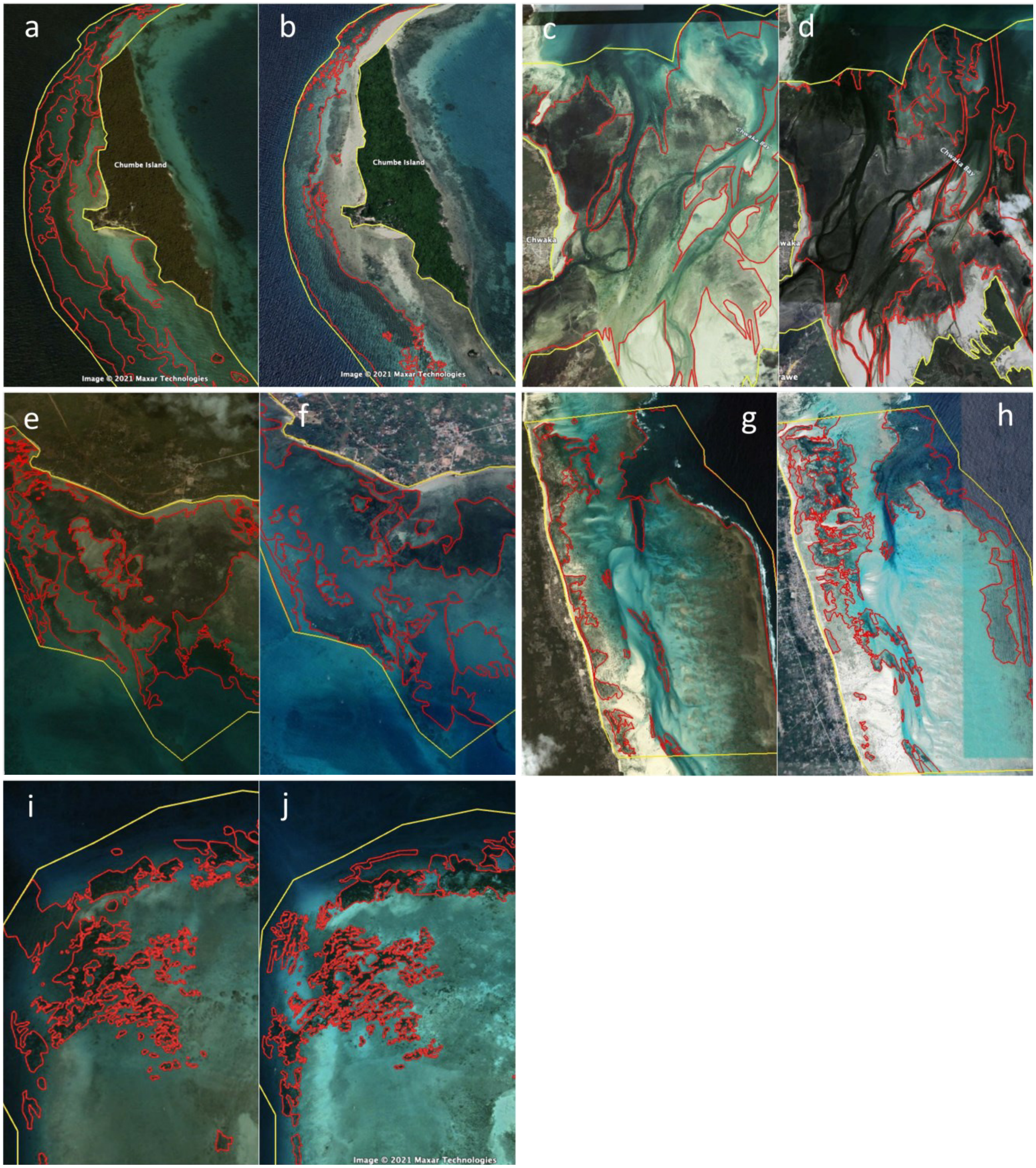

Figure 2a-j. Google Earth images of sites with site boundaries outlined in yellow and seagrass mappings outlined in red: (**a**) Chumbe Island, October 2005; (**b**) Chumbe Island, February 2021; (**c**) Chwaka Bay, February 2006; (**d**) Chwaka Bay, January 2019; (**e**) Fumba, January 2006; (**f**) Fumba, July 2019; (**g**) Jambiani, September 2005; (**h**) Jambiani, August 2020; (**i**) Nungwi, November 2005; and (**j**) Nungwi, February 2019.

summing the area of each polygon to calculate the total area (km^2) of seagrasses at each site and then dividing the total area of seagrass cover by the total mapping area for each site. The percent change in seagrass cover from 2006 to 2019 was then calculated per site.

Statistical analyses

All analyses were conducted with StatPlus:mac v5.0 statistical analysis software (AnalystSoft Inc., 2021). First, the mappings were validated by conducting a two-samples paired t-test to compare the percent of seagrass coverage in 2006 and 2019 in the validation field dataset

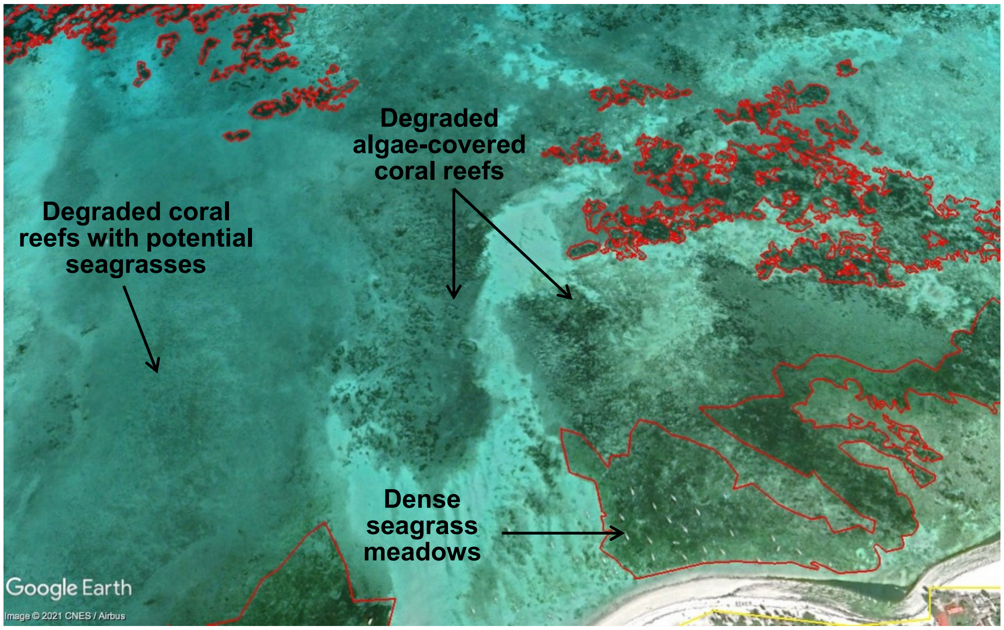

Figure 3. Examples of degraded coral reefs with potential seagrasses, degraded algae-covered coral reefs, and dense seagrass meadows located in Nungwi, February 2019.

and the study GIS dataset. To determine whether the implementation of MPA management plans, coastal activities (seaweed farming, small-scale fishing, and tourism), or the combination of coastal activities were correlated to changes in seagrass coverage from 2006 to 2019 at selected sites, Spearman's rank correlations tests were conducted and scatterplots generated.

Results

Total mapping areas, percent of seagrass cover in 2006 and 2019, and total percent change for each site are listed in Table 3. Chwaka Bay had the largest mapping area of 59.12 km², and Chumbe Island had the smallest area of 0.57 km², defined by the boundaries of the no-take zone. Chwaka Bay also had the highest percentage of seagrass cover in both measurement

Table 3. Total area and percentage of seagrass cover per site.

Site	Area included in mapping (km^2)	Total area of seagrass cover (km^2), 2006	Percentage of seagrass cover, 2006	Total area of seagrass cover (km^2), 2019	Percentage of seagrass cover, 2019	Percent change from 2006 to 2019
Chumbe Island	0.57	0.27	48.11%	0.22	39.08%	-18.76%
Chwaka Bay	59.12	37.58	63.57%	32.68	55.28%	-13.04%
Fumba	6.72	3.29	49.00%	3.09	45.91%	-6.30%
Jambiani	20.49	5.83	28.47%	6.01	29.34%	+3.07%
Nungwi	6.47	1.25	19.30%	1.00	15.47%	-19.86%

Table 4. Correlation coefficients and p-values for correlations between seagrass coverage and the implementation of marine protected area (MPA) management plan and the scale of coastal activities between 2006 and 2019.

MPA or coastal activity	Correlation coefficient (rs)	p-value
MPA management plan	-0.01	0.972
Seaweed farming	0.28	0.426
Small-scale fishing	-0.12	0.734
Tourism	-0.64	0.044
Combined coastal activities	-0.08	0.825

periods (63.57 % and 55.28 %), followed by Fumba (49 % and 45.91 %). Nungwi had the lowest percent of seagrass cover (19.3 % and 15.47 %) and had large areas of degraded or dead algae-covered coral reefs. On average, seagrass cover decreased by 10.98 % from 2006 to 2019. Nungwi had the highest decrease in seagrass cover (19.86 %), followed by Chumbe Island (18.76 %). Only Jambiani had an increase in seagrass cover from 2006 to 2019 (3.07 %).

The percent change of seagrass cover from 2006 to 2019 was -32.61 % for the validation field dataset compared to -18.76 % for the study GIS dataset. In the paired two-samples t-test, no significant difference was found in the percent change in seagrass cover between the validation and study datasets ($t(3) = 2.65$, $p = 0.230$). Therefore, it was concluded that the Chumbe Island GIS data adequately aligned with the field data and that the mappings of satellite images were valid.

Table 4 lists the correlation coefficients and p-values for MPA management plans and coastal activities. Spearman's rank correlation coefficients in the range of 0.60-0.69 are considered moderate to strong correlations (Akoglu, 2018). A strong negative correlation was found between changes in seagrass coverage from 2006 to 2019 and the scale of tourism ($r_s(9) = -0.64$, $p = 0.044$). In other words, seagrass coverage is more likely to be lower in sites with higher-scale tourism. Figure 4 displays a scatterplot of the relationship between tourism and the percent of seagrass cover. The linear slope of the line of best fit from the top left to the bottom right indicates a strong negative relationship between the variables.

There were no significant correlations between changes in seagrass coverage and MPA management plan implementation ($r_s(9) = -0.01$, $p = 0.972$), seaweed farming ($r_s(9) = 0.28$, $p = 0.426$), small-scale fishing ($r_s(9) = -0.12$, $p = 0.734$), or the combined score for coastal activities ($r_s(9) = -0.08$, $p = 0.825$).

Discussion

This study is the first to identify correlations between different coastal activities and decreases in seagrass cover over time. As expected, tourism had the greatest significant negative correlation with seagrass cover from 2006 to 2019 ($r_s(9) = -0.64$, $p = 0.044$), whereas the use of seaweed farming, small-scale fishing, or the combination of activities were not significantly correlated with changes in seagrass cover. Several studies evaluating nutrient concentrations in coastal waters near cities around Unguja have found higher nutrient concentrations in population and tourism centers and closer to the shore, which they attribute to untreated sewage and pollution (Limbu and Kyewalyanga, 2015; Moto and Kyewalyanga, 2017; Moynihan *et al*., 2012; Ngusaru, 2000). Staehr *et al*. (2018) suggested that eutrophication resulting from increased levels of dissolved organic matter and nutrients from untreated sewage in intertidal and subtidal zones were related to population and tourism increases in Zanzibar. Algal overgrowth due to eutrophication can harm seagrasses through light reduction and ammonium toxicity (Burkholder *et al*., 2007). Although the links between tourism and deteriorating marine environmental conditions are well established, the current study is the first to the authors' knowledge to demonstrate correlations between growing tourism and long-term seagrass declines in Zanzibar.

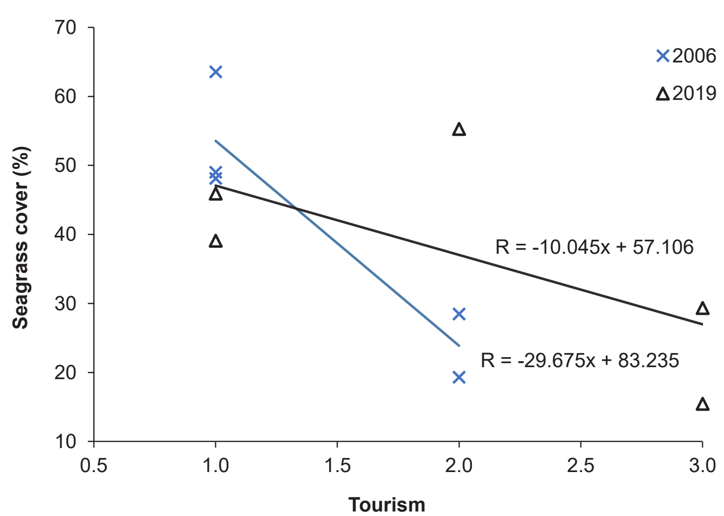

Figure 4. Scatter plot of the linear relationship between the scale of tourism and the percent of seagrass cover in 2006 compared to 2019.

Small-scale fishing had a weak negative correlation with the percent of seagrass cover ($r_s(9) = -0.12$, $p = 0.734$), suggesting that small-scale fishing still has a negative impact on seagrasses that has been documented in other research. For example, overfishing of finfish residing in seagrass meadows has caused multiple crown of thorn starfish (*Acanthaster planci*) outbreaks in Zanzibar, leading to significant coral reef damage (Staehr *et al.*, 2018). Additional research on the long-term impact of small-scale fishing practices on seagrass health is needed to establish whether small-scale fishing poses a substantial threat to seagrasses over time and should be identified as a threat in coastal marine management plans.

Contrary to hypothesis used in this study, seaweed farming had a weak positive correlation with seagrass cover ($r_s(9) = 0.28$, $p = 0.426$), suggesting that areas with higher scales of seaweed farming were more likely to have greater seagrass cover. Jambiani was the only site to show an increase in seagrass coverage from 2006 to 2019, which was likely reflective of the high baseline degradation of seagrass meadows and historical practices of uprooting seagrasses in seaweed farms (Gullström *et al.*, 2006; Lyimo *et al.*, 2006). For example, Figure 5 is a satellite image from 2006 of rows of seaweed farms in Jambiani that were built over bare sand that had been cleared of seagrasses. Seaweed farmers have since developed an understanding that the sediment nutrient enrichment of seagrasses supports better growth of seaweed, and the practice of uprooting seagrasses is less common (Lyimo *et al.*, 2008).

Contrary to the study hypothesis, the implementation of MPA management plans had no effect on seagrass cover ($r_s(9) = -0.01$, $p = 0.972$). However, this is not entirely surprising given the mixed evidence for the efficacy of MPAs to conserve seagrasses and other marine ecosystems (de la Torre-Castro *et al.*, 2014) and the complexity of developing MPA management plans that adequately account for the unique ecological, sociopolitical, and economic contexts both within and outside the boundaries of the MPA. Several studies have demonstrated the influence of coastal land-use on seagrass conservation within MPAs in Zanzibar and in other tropical environments. Aller (2018) compared seagrass cover and species composition in public MPAs, private MPAs, and unprotected sites in Zanzibar. Although the study's results suggested that MPAs increased the temporal stability of seagrass habitat-dependent fish, management of MPAs did not effectively protect seagrasses from negative land-use effects. Eklöf *et al.* (2009) compared the effectiveness of protecting seagrasses from sea urchin predation in two large Kenyan MPAs and ultimately found that the targeted approaches of MPAs were ineffective at preventing sea urchin overgrazing. They suggested that both ecological factors and impacts from coastal land use contributed to the unsuccessful management of sea urchin overgrazing. Additional research is needed to evaluate

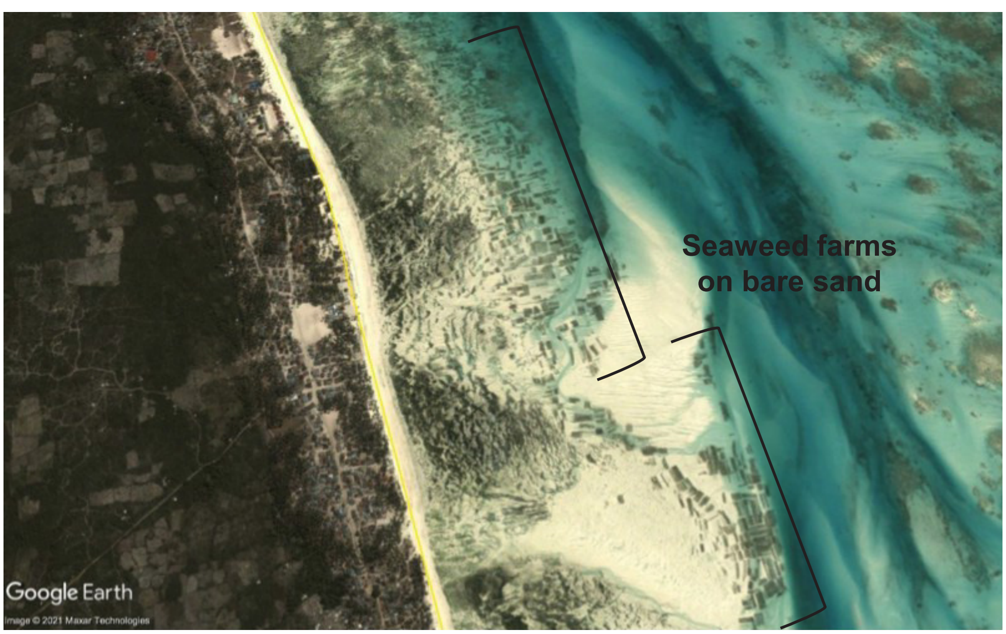

Figure 5. Rows of seaweed farms located on bare sand in the intertidal and subtidal zones of Jambiani, September 2005.

specific characteristics of Zanzibar MPAs, effects of coastal land use, water pollution, and associated seagrass conservation outcomes. In addition, a global framework for determining a site-specific hierarchy of variables (i.e., ecological factors within MPAs and outside of MPAs, sociopolitical factors, land-use conditions, among others) that influence the effectiveness of MPAs could be a useful tool for integrated coastal marine planning and management.

It remains a possibility that environmental factors are also driving seagrass declines, though the available literature does not support this conclusion. Aller *et al*. (2019) found that changes in cloud cover, sunspot activity temperature, tidal amplitude and height, and storm occurrence did not impede seagrass recovery in Chumbe Island within a 10-year period. The timing of seagrass declines and recovery varied by transect, rather than by the quarter of year, so they concluded that seasonality may not be the main driver of seagrass cover variation. In addition, two studies evaluating the seasonal variation of seagrass cover using satellite images and field data in Chwaka Bay and Chumbe Island found no seasonality in the percent of seagrass cover (Gullström *et al*., 2006; Knudby *et al*., 2010, respectively). More generally, Kamermans *et al*. (2002) found very little variability of porewater salinity in the intertidal zone across seasons, which may indicate that bi-annual monsoons do not substantially affect coastal water composition. More research is needed to evaluate the long-term impacts of environmental factors and climate change on seagrasses in Zanzibar.

There are several limitations to the current study. First, it was difficult to distinguish algae-covered coral reefs from seagrasses in several sites, so the percent of seagrass cover was likely underestimated in sites with high coral reef mortality, such as at Chumbe Island. More extensive and comprehensive field data could improve the accuracy of seagrass mappings. In addition, the percent of seagrass cover provides limited information on the overall health of seagrass meadows, therefore collecting in-situ data at multiple sites and utilizing more indicators in addition to seagrass cover can strengthen the assessment of overall seagrass health. Finally, results comparing the relative impact of each coastal activity should be interpreted with caution as detailed data on the number of seaweed farmers and small-scale fishers at each site and water quality indicators were not available.

Recommendations

First, it is recommended that Zanzibar's coastal marine management plans identify seagrass meadows as critical ecosystems; implement regional efforts to measure and track changes to seagrass cover, volume, and species composition; and develop evidence-based plans to restore and conserve seagrass meadows. Future research should evaluate the long-term return on investment for seagrass restoration projects to ensure they are cost-effective and beneficial to communities in Zanzibar.

Second, there is an opportunity for Zanzibar to promote sustainable practices and accountability in the tourism industry by creating an ecotourism network that uses a transparent rating system to evaluate participating hotels and lodges on their environmental practices. Although many tourists already participate in ecotourism in Zanzibar, there is currently no mechanism for tourists to determine the environmental and socioeconomic practices of hotels and lodges.

Finally, addressing harmful environmental practices of the tourism industry, such as polluting coastal waters and uprooting seagrass meadows on hotel coastlines, should become a high long-term priority for integrated coastal marine management. There are numerous models with demonstrable success from which Zanzibar could draw, such as levying fines on noncompliant hotels and lodges or instituting tourism taxes that fund waste management systems and conservation projects. Policymakers in Zanzibar should consult the appropriate stakeholders and conduct market-based analyses on each approach to determine the best approach to address harmful environmental practices in the tourism industry.

Conclusions

The current study found support for the hypothesis that tourism, compared to seaweed farming or small-scale fishing, was significantly correlated with a decline in seagrass cover in Zanzibar from 2006 to 2019. A weak insignificant correlation between small-scale fishing and seagrass cover decline was also found, as expected. In contrast, the results showed that seaweed farming had a weak, though insignificant, positive correlation with seagrass cover. Surprisingly, support was not found for the hypothesis that the implementation of MPA management plans would have a protective effect on seagrass cover. This may be due to community reliance on coastal resources and the ineffective enforcement of MPA regulations, which may limit the effectiveness of MPAs throughout Zanzibar (UNEP-Nairobi Convention and WIOMSA, 2021).

This research contributes to a growing body of evidence that tourism, urbanization, and other anthropogenic impacts are driving declines in seagrass ecosystems in Zanzibar, and implementing MPAs alone is not enough to protect seagrasses and the rest of the coastal marine environment. Yet this study is the first of its kind to compare the long-term impacts of coastal activities and the implementation of MPA management plans on seagrasses in Zanzibar.

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