August 2021

Current State of Seagrasses in Zanzibar: Impacts of Coastal Economic Activities and Marine Protected Areas on Seagrass Cover

Danielle Purvis
School for International Training Graduate Institute

Narriman Jiddawi
University of Dar es salaam

CURRENT STATE OF SEAGRASSES IN ZANZIBAR: IMPACTS OF COASTAL ECONOMIC ACTIVITIES AND MARINE PROTECTED AREAS ON SEAGRASS COVER

Danielle Purvis, MPH

A capstone paper submitted in partial fulfillment of the requirements for a Master of Arts in Climate Change and Global Sustainability at SIT Graduate Institute, USA

August 3, 2021

Academic advisor: Dr. Narriman Jiddawi

I hereby grant permission for World Learning to publish my capstone on its websites and in any of its digital/electronic collections, and to reproduce and transmit my capstone electronically. I understand that World Learning's websites and digital collections are publicly available via the Internet. I agree that World Learning is NOT responsible for any unauthorized use of my capstone by any third party who might access it on the Internet or otherwise.

Student Name: Danielle Purvis

Date: August 3, 2021

Table of Contents

Abstract

Seagrass meadows are located abundantly in Zanzibar, Tanzania and provide essential ecosystem services, such as sediment nutrient enrichment and blue carbon sequestration. However, seagrasses have been less researched or protected than other marine ecosystems. Although environmental variables affect seagrass health, evidence suggests that anthropogenic impacts are their greatest threats. The rapid expansion of seaweed farming and tourism and widespread use of harmful small-scale fishing practices in Zanzibar have contributed to the degradation and removal of seagrass meadows, disrupted coastal marine food chains, and reduced local biodiversity that seagrasses support. Public or private marine protected areas (MPAs) protect most of Zanzibar's coastal marine ecosystems, yet evidence is unclear whether MPAs effectively conserve marine ecosystems. Using geographic information systems (GIS) to estimate the change in percent of seagrass cover from 2006 to 2019, we conducted Spearman's rank correlation analyses to identify whether seagrass degradation was correlated with seaweed farming, fishing, or tourism and whether MPA management plans were protective. Tourism was negatively correlated with seagrass cover, $r(9) = -0.74$, $p = 0.044$, suggesting that tourism is an important driver of seagrass declines in Zanzibar. No other variables were significantly correlated with seagrass cover decline. 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 and other human impacts.

Acknowledgements

This project is a true testament to the power of community. I want to thank Dr. Narriman Jiddawi, whose deep knowledge of mariculture and marine ecology is a true asset to Zanzibar. Dr. Jiddawi's championship for seagrasses inspired my interest in the topic, and I hope this work contributes to this vital field. I am grateful for Dr. Xavier Silva's unending support and generous input, despite an ever-burgeoning workload. Thanks to my cohort for their patience and perseverance through a tumultuous year. I have no doubt we are stronger and more thoughtful scientists because of our experiences. This project would be half of what it is without Lillian Claire Chargualaf's meticulous assistance with data entry. Finally, to Tim Franzen, Jerry and Carolyn Purvis, Chase Conner, all of my mentors, and my entire support network, I have deep gratitude for your unconditional love and encouragement.

Introduction

Seagrasses have been less commonly researched or protected than other flora and fauna in tropical coastal marine environments, but there is increasing evidence that seagrasses are essential to the humans, flora, and fauna inhabiting the coast and must be considered in coastal marine management (Githaiga et al., 2016; Staehr et al., 2018). However, 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. To improve integrated coastal marine management and the welfare of seagrass meadows, adequate monitoring of density and distribution, addressing the human-caused threats on seagrass meadows, and consulting with coastal communities on sustainable methods for maintaining seagrasses will benefit the marine ecosystems, coastal communities, and all who enjoy the beautiful beaches in Zanzibar.

Figure 1. Images of a healthy seagrass meadow (left) and degraded seagrasses due to sea urchin predation (right). Source: UNEP, 2020.

Seagrass meadows are located abundantly in the archipelago of Zanzibar (Aller et al., 2019; Khamis et al., 2017), and there have been 10 documented species dominated by the climax species *Thalassia hemprichii* (see Figure 2; Lyimo et al., 2008; Ochieng & Erftemeijer, 1993). They primarily grow between the intertidal and subtidal zones, typically near coral reefs and mangrove forests, and 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 and fertilize the sediment with nitrogen and other nutrients (Belshe et al., 2018; Nordlund et al., 2016). The nutrient-rich meadows create ideal environments for feeding grounds, nurseries, and habitats for aquatic and nonaquatic species; the seagrass meadows are particularly important feeding grounds for sea turtles in Zanzibar (Staehr et al., 2018). Seagrass meadows play a role in coastal geomorphology and provide coastal protection by dampening waves (Nordlund et al., 2016). Globally, seagrasses also sequester an estimated 50-64% of global organic carbon and a substantial proportion of blue carbon (Nchimbi & Lyimo, 2019; Nordlund et al., 2016). Belshe et al. (2018) found that carbon content was significantly higher in environments with seagrass meadows compared to those without, regardless of the type of seagrass species present. One study compared the biomass density (of which approximately 50% is carbon) of seagrasses around Africa and concluded that the East African coast had the greatest total of biomass of 738.1 g DW/m^2, compared to the second highest total biomass of 370.8 g DW/m^2 in the Southern Mediterranean (Githaiga et al., 2016). While the average site in Africa had an average biomass density of 514.3 g DW/m^2, the mixed seagrass beds on the Jambiani coast in Zanzibar had an overall average biomass of 3,063.3 g DW/m^2. In summary, the ecosystem services of seagrasses in Zanzibar support a productive and biodiverse coastline and sequester a substantial amount of carbon.

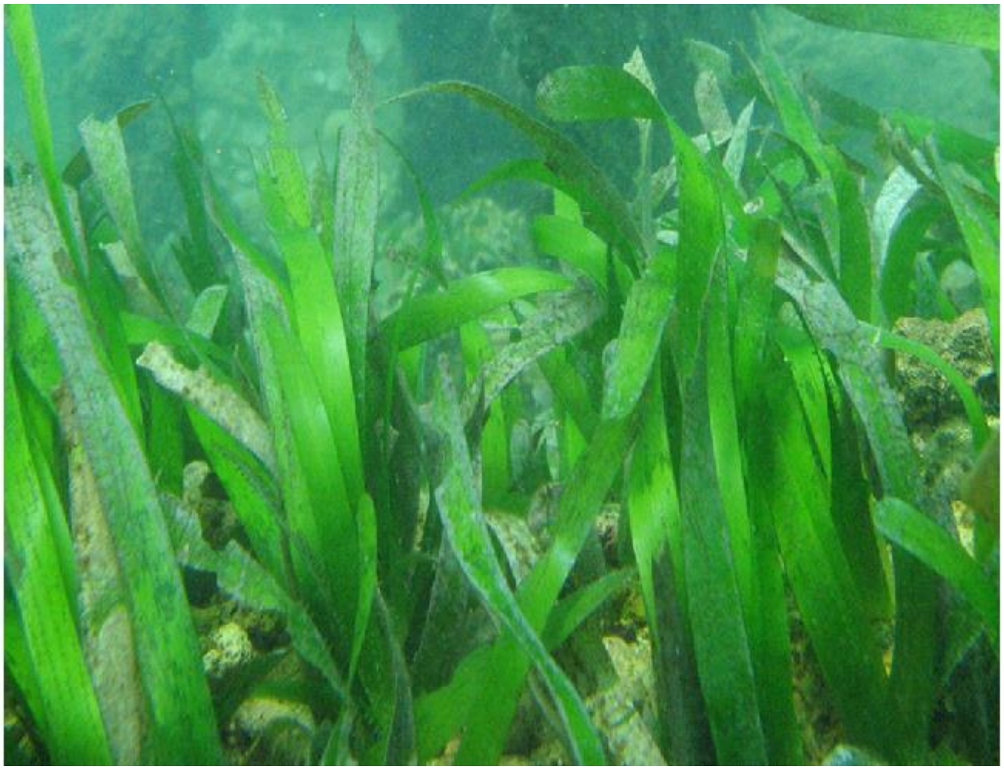


Figure 2. Image of *Thalassia hemprichii* seagrass species. Source: Jebasingh et al., 2016.

A timeline analysis of the health status of seagrasses in Zanzibar suggests that beds have been degraded and somewhat recovered since the 1990's, yet degradation due to anthropogenic influences continues to threaten the health of seagrass beds around Zanzibar (Nchimbi & Lyimo, 2019). Although seagrasses are vulnerable to environmental changes, they typically recover quickly to changing environmental conditions such as changing water temperature, suggesting that long-term decline is due to anthropogenic impacts (Aller et al., 2019). Previous research focuses primarily on the impacts of seaweed farming, small-scale fishing, and tourism as drivers of seagrass degradation. Evidence suggests numerous mechanisms in which each of these economic activities damage seagrasses, which are useful to understand when building policies that target the greatest threats to seagrasses.

Commercial seaweed farming was introduced to Zanzibar in 1989 in Jambiani and Paje and expanded to Chwaka Bay in 1990, starting with the non-native species *Eucheuma* *denticulatum* (Eklöf et al., 2012; Msuya, 2013). The primary use of commercial seaweed is to extract carrageenan, a thick gel used for food, cosmetics, pharmaceuticals, and textiles (Msuya, 2013). Zanzibar annual seaweed production reached >16,500 metric tons in 2016 (Zanzibar Department of Fisheries, 2021). Commercial seaweed is the largest marine export from Zanzibar, and seaweed farming remains a high-volume economic venture for several coastal villages on Unguja since it began in Jambiani, Paje, and Chwaka Bay around 1990 (United Republic of Tanzania, 2008; Moreira-Saporiti et al., 2021). Seaweed farms in Zanzibar are typically situated in the shallow intertidal range on sandy bottoms or in seagrass meadows because the water is shallow enough for primarily female farmers to create and maintain seaweed farms without having to swim (Lyimo et al., 2006; Moreira-Saporiti et al., 2021). Although some farmers have uprooted seagrass meadows to remove sea urchins that live in seagrasses and destroy the seaweed, most seaweed farmers situate their farms in seagrass meadows because of the high nutrient content and low surface temperatures in the meadows (Hedberg et al., 2018; Lyimo et al., 2006). However the majority (92%) of seaweed farms in Zanzibar are located at least partially on seagrass meadows, demonstrating their interlinkages (Hedberg et al., 2018), as demonstrated in Figure 3.

Figure 3. Seaweed farm situated above a seagrass meadow in Jambiani, Zanzibar. Source: Danielle Purvis, 2021.

Several studies demonstrate the negative impacts of seaweed farms on seagrass ecosystems that harbor farms, including lower density of seagrass shoots, biomass, canopy heights, and cover compared to areas without seaweed farms (Lyimo et al., 2006; Lyimo et al., 2008; Moreira-Saporiti et al., 2021; Nchimbi & Lyimo, 2019). Among two rural villages on the east and west coasts of Unguja Island 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, even though 90% of villagers rated seagrass status as very good. Physical disturbances like trampling or removing seagrasses and sediment disturbance rooting seagrasses 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). Although research is mixed on which mechanism of seaweed farming is most damaging to seagrass meadows, the results consistently link seaweed farming to the degradation or destruction of seagrasses.

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's unclear how overfishing or the resulting imbalance of food chains in the coastal marine ecosystems 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 (e.g., rabbitfish and parrotfish), prawns, and bivalves (Khamis et al., 2017; Staehr et al., 2018; UNEP-Nairobi Convention & WIOMSA, 2021). The fishing industry, which provides livelihoods for about one-fifth of the Zanzibar population, primarily uses traditional low-technology techniques, such as seine nets, wooden basket traps, handlines, and spears (de la Torre-Castro et al., 2014; 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 (as in, catch per unit effort) has decreased, suggesting overfishing is occurring (Khamis et al., 2018). Overfishing can disrupt the food chain of the entire coastal marine ecosystem. For example, overfishing of finfish residing in seagrass meadows has caused multiple Crown of thorn starfish (*Acanthaster planci*) outbreaks, leading to significant coral reef damage (Staehr et al., 2018). Destructive fishing techniques, such as the illegal use of seine nets, spear-guns, and dynamite (Jiddawi & Öhman, 2002; Khamis et al., 2017), degrade and destroy seagrass meadows and coral reefs. 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 & Öhman, 2002; Khamis et al., 2017).

Tourism has grown tremendously in Zanzibar over the last three and a half decades at the expense of the coastal marine environment, despite attracting visitors for ecotourism, white sandy beaches, and clean, clear waters (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 are building lodging along the most attractive sections of the coast of Unguja, clearing the coastline of seagrass meadows to make white sandy beaches, 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 are 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). However Zanzibar does not have systematic waste management on the islands to manage the large increase of solid and sewage waste, which increasingly contains plastic products (Staehr et al., 2018). There is an inverse relationship between biomass of seagrass meadows and coastal development; in other words, as the Zanzibari population and tourism increase, seagrass meadow density decreases (Khamis et al., 2017; Staehr et al., 2018).

Multiple public and private marine protected areas (MPAs) cover the majority of the Zanzibar's 370 km of coastline, protecting an area of approximately 1,300 km^2 (IUCN, 2020), the main island commonly referred to as Zanzibar, as displayed in Figure 4. The first MPA established in Zanzibar was the private Chumbe Island Coral Park Sanctuary (CHICOP) in 1994, which immediately implemented a management plan and established a no-take zone on the western side of the island (IUCN, 2020). Mnemba Island-Chwaka Marine Conservation Area (MIMCA) was declared in 2002, and the management plan was implemented in 2005 (IUCN, 2020). The MIMCA maintains a private no-take zone around the Mnemba Atoll, whereas the rest of the conservation area, including the northern tip of Unguja Island, is considered a managed resource protected area (IUCN, 2020; UNEP-Nairobi Convention & WIOMSA, 2021). The Jozani-Chwaka Bay National Park, established in 2004, is a smaller protected area and adds additional coastal and terrestrial forest protections, which likely have conservation implications for seagrasses in the Bay by proxy (UNEP-Nairobi Convention & WIOMSA, 2021).

In addition to the establishment of private MPAs, the Zanzibar Environmental Management Act of 2015 legally authorizes the Ministry of Livestock and Fisheries, or any minister responsible for marine natural resources, to establish public MPAs on Unguja Island, Zanzibar, of which there are currently four in effect (UNEP-Nairobi Convention & WIOMSA, 2021). The MIMCA is primarily a public MPA and includes the Mnemba Atoll no-take zone (UNEP-Nairobi Convention & WIOMSA, 2021). The Menai Bay Conservation Area (MBCA) was declared in 1997, and the management plan was implemented in 2010 (IUCN, 2020). As the largest MPA in Unguja, the MBCA boundaries cover the entire southern region of Unguja from the tip of the Urban West to the tip of the central region (IUCN, 2020). The Tumbatu Marine Conservation Area (TUMCA) and Changuu-Bawe Islands Marine Conservation Area (CHABAMCA) were declared in 2015 to protect coastlines on the western side of Unguja, and their first management plans were scheduled to be developed in 2018-2019 but have not yet been implemented (UNEP-Nairobi Convention & WIOMSA, 2021). MBCA and MIMCA were established by the Ministry of Livestock and Fisheries to maintain the sustainable use of fisheries resources and coastal marine ecosystems (IUCN, 2020; UNEP-Nairobi Convention & WIOMSA, 2021). 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 (de la Torre-Castro et al., 2014; Unsworth & Cullen, 2010; Cullen-Unsworth et al., 2014; Unsworth et al., 2018).

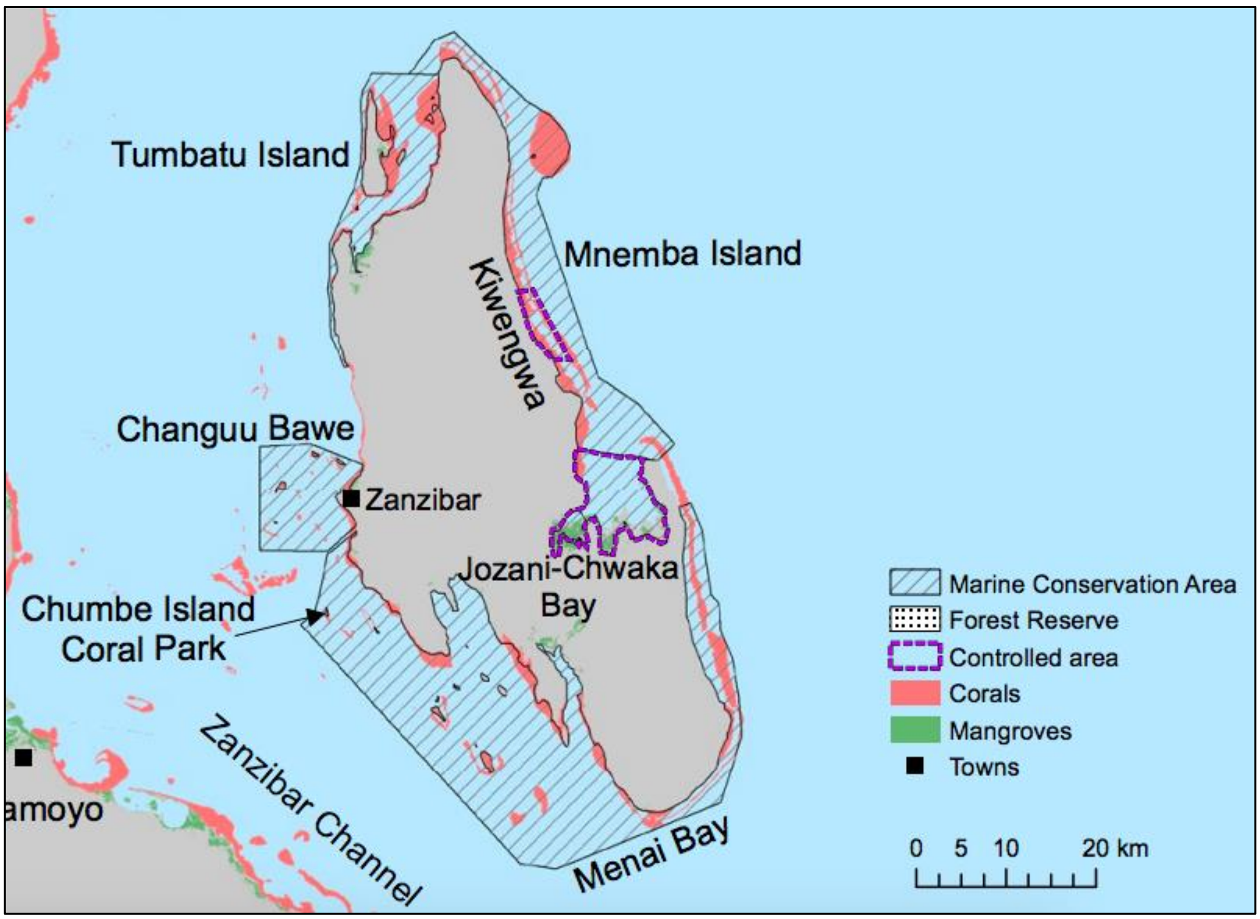


Figure 4. Marine protected area (MPA) map of Zanzibar. Source: UNEP-Nairobi Convention & WIOMSA, 2021.

There are several risks to the potential success of MPAs to protect and conserve coastal marine ecosystems in Zanzibar. The Western Indian Ocean Marine Science Association

(WIOMSA) has identified the high poverty levels and marine resource dependence of coastal communities, unsustainable fishing methods, and the downscaling of funding for MPA management and enforcement due to a lack of revenue generation as the greatest threats to the success of MPAs in Zanzibar (UNEP-Nairobi Convention & WIOMSA, 2021). It is currently unclear how influential these risks are on the effectiveness of MPAs.

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), on specific metrics of health (e.g., carbon content), or specific anthropogenic impacts (e.g., seaweed farming) on seagrasses. Yet absent from the discussion is which anthropogenic activities cause the most degradation or destruction of seagrasses over time and the types of measures Zanzibar has enacted that effectively protect these essential ecosystems. Using geographic information systems (GIS) satellite imagery and field data, the current study assessed the percent of seagrass cover in five sites around Unguja Island, Zanzibar in 2006 and 2019 and evaluated 1) whether coastal economic activities (seaweed farming, small-scale fishing, and tourism) at varying scales have negative impacts on seagrass cover, and 2) whether the implementation of MPA management plans had protective effects on seagrass cover. We hypothesized that high-intensity tourism will have the largest negative correlation with seagrass cover, and the implementation of MPA management plans will have the greatest protective effect on seagrasses.

Methods

Study Area

The study area included seagrasses on the coast of Unguja Island, which is located approximately 30 km off the coast of East Africa in the Western Indian Ocean. Zanzibar's climate is tropical and defined by two rainy seasons from March to May (the "long rains") and September to November (the "short rains"), and the Monsoon wind system influences the local currents with slightly stronger winds during the March-May rainy season (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. Most seagrass beds and meadows are located in shallow water depths of less than 5 m (Aller et al., 2017; Belshe et al., 2018).

There have been two major El Niño Southern Oscillation (ENSO) events in the Western Indian Ocean 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 (Lin et al., 2018; Nowicki et al., 2017), especially for seagrass meadows on the East African coast.

Site Selection

We mapped seagrasses in five sites on and off the coast of Unguja: Chumbe Island, Chwaka Bay, Fumba, Jambiani, and Nungwi (see Figure 5). We selected sites that represented a diversity of MPA protections and coastal economic activities. Chumbe Island is a small private island 6 km off the southwest coast of Unguja managed by CHICOP. Chumbe Island is surrounded by coral reefs and seagrasses, and the western side of the island is a no-catch zone established by the CHICOP MPA. There is one ecolodge on the island that economically supports research, conservation, and education programs. 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 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 MBCA includes the coasts of Jambiani and Fumba.

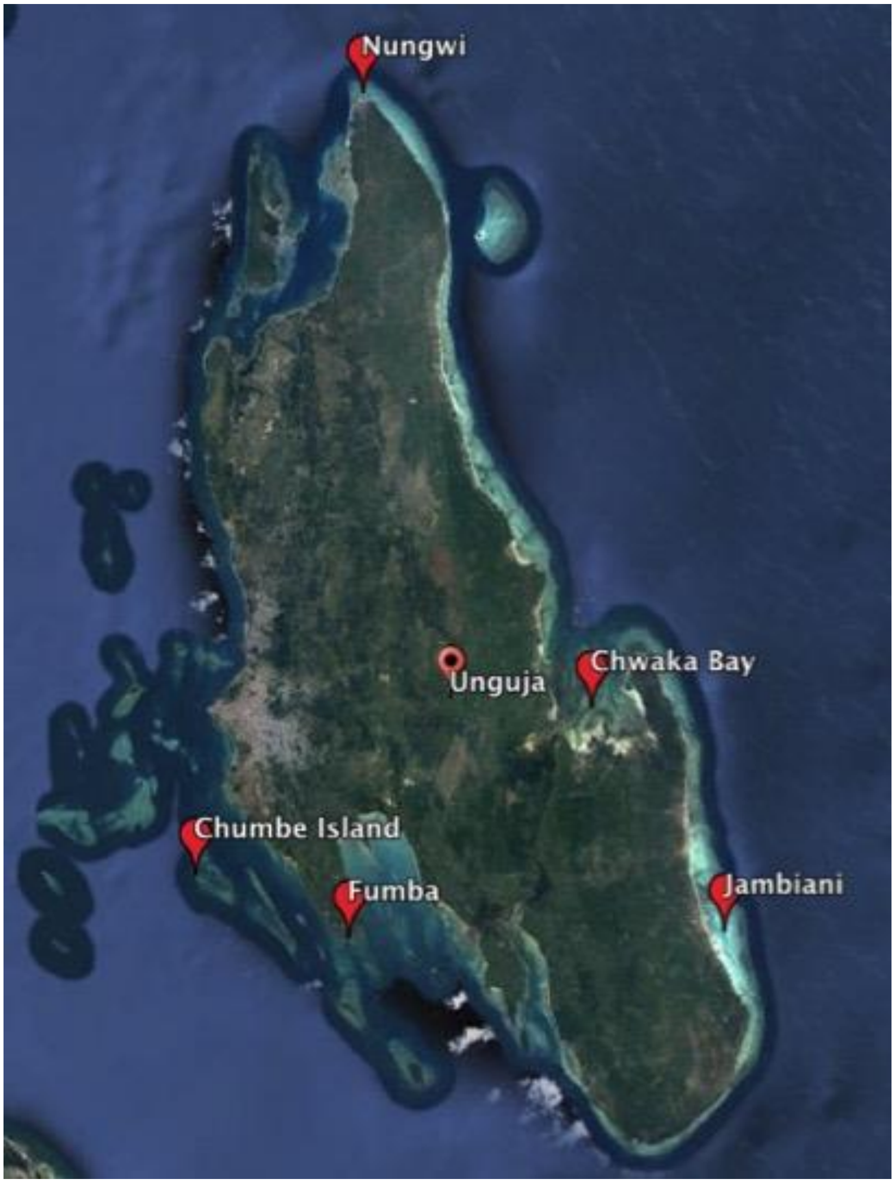


Figure 5. Study site map

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 economic activities. To compare the distribution and characteristics of seagrasses over time, site-specific satellite images and field data collected 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 data at www.seagrassnet.org. Field data included the percentage of seagrass cover per 0.25 m^2 quadrat collected by CHICOP 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 and peer-reviewed articles provided information about the implementation and scale of MPA management plans and coastal economic activities, including seaweed farming, small-scale fishing, and tourism, occurring in coastal villages around Unguja.

Time Period Selection

The study compared seagrass coverage at each site from 2006 to 2019. We determined appropriate baselines for seagrass cover based on the following three factors: timelines of the growth of Zanzibar's overall economy and mariculture and tourism sectors; the availability of high-quality satellite images; and the timing of ENSO events. 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 11,000 tons in 2018 (Msuya, 2020). Following a decline in fish catches in the 1980's, annual artisanal fish catches in Zanzibar have steadily increased since 1991 (Rehren et al., 2020). Estimated fish catches increased from >20,000 tons in 2006 to >30,000 tons in 2016 (Rehren et al., 2020); there are no available data on the quantity of fish catches in 2019. 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 economic activities, we had to factor in the limitations in available historical satellite imagery and field data. The earliest available high-quality satellite images on Google Earth Engine were between 2005 to 2007 for all sites, and CHICOP collected its first set of field data in 2006.

Finally, both time periods occurred >3 years following ENSO events to ensure seagrasses had adequate time to recover. Although Nowicki et al. (2017) found that seagrasses in Shark Bay, Australia did not fully recover 3 years following an El Niño event in 2011, Lin et al. (2018) found *Thalassia hemprichii*, the climax species in Zanzibar, to be particularly drought-resistant to ENSO events in Taiwan. Therefore we believe measuring seagrass cover at least 3 years following the ENSO events of 1997-1998 and 2014-2016 allowed sufficient time for seagrasses to recover.

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 2 years. We included a 2-year minimum implementation period 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 >2 years were assigned a "0".

To assign the scale of seaweed farming at each site, we used the number of seaweed farmers as a proxy for assessing the impact of trampling, shading, and removal of seagrasses. We used the Joint Frame Survey scale (United Republic of Tanzania, 2008) of the number of seaweed farmers in each region of Unguja to assign codes. 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 produce seaweed were coded with “0”.

Due to a lack of comprehensive fisheries data available for each site, we used a proxy of the number of fishermen 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. We used the Joint Frame Survey scale (United Republic of Tanzania, 2008; Rehren et al., 2020) for the number of fishermen in each region 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”. Sites within no-take zones were coded with “0”. Of note, 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. We used Google Earth 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, we estimated that the scale of tourism in 2006 was one-third of the volume in 2019 since the number of tourists have increased more than threefold from 2006 to 2019 (World Bank, 2019). Each site was, therefore, assigned a "1" for low-scale tourism in 2006.

We also created a variable for the combined score of coastal economic activities to assess whether coastal activities had a collective impact on seagrass health. For example, Chwaka Bay's combined score for coastal economic activities in 2006 was calculated as follows: 3 (high-scale seaweed farming) + 3 (high-scale fishing) + 1 (low-scale tourism) = 7.

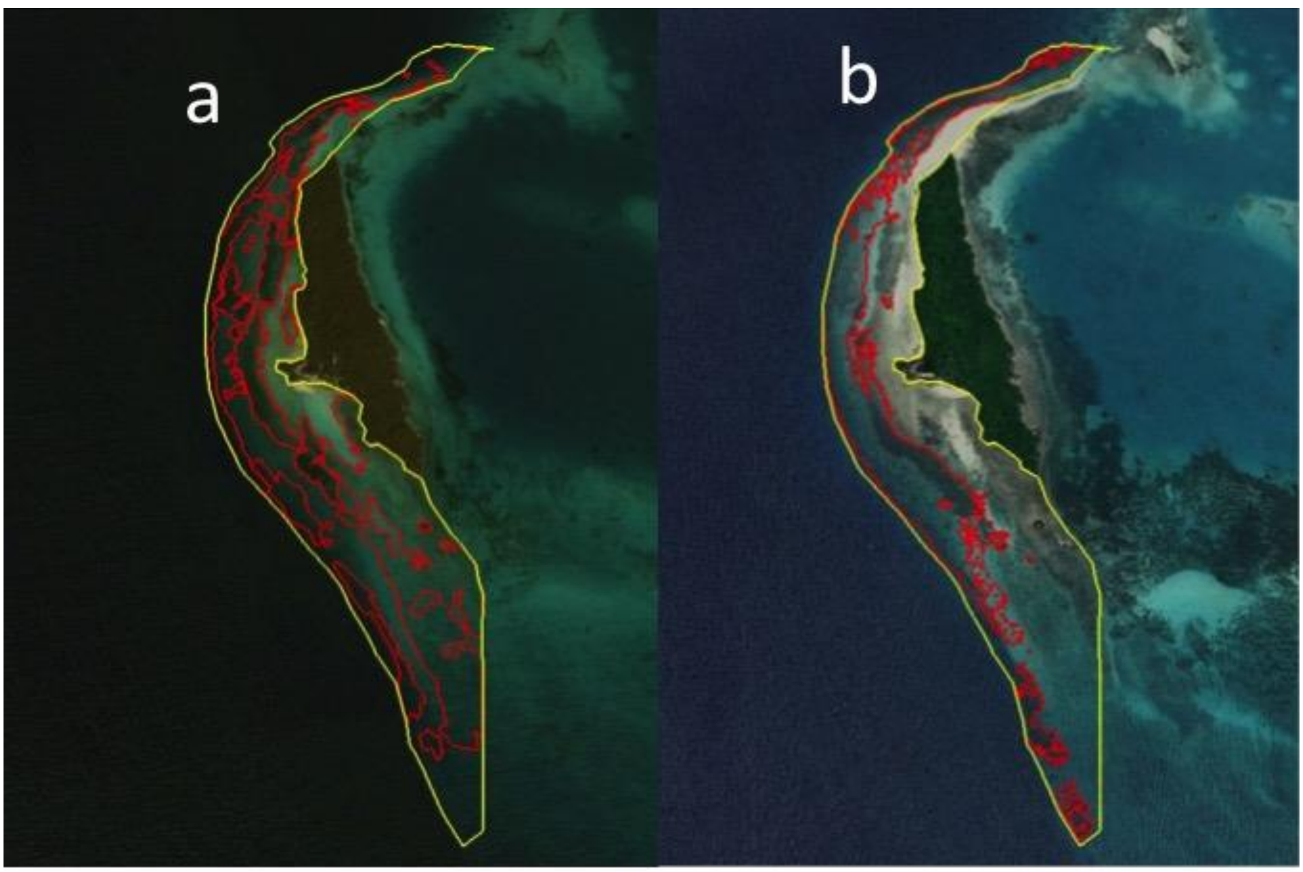
The MPA management plan status and scale of coastal economic activities for each site in 2006 and 2019 are displayed in Table 1.

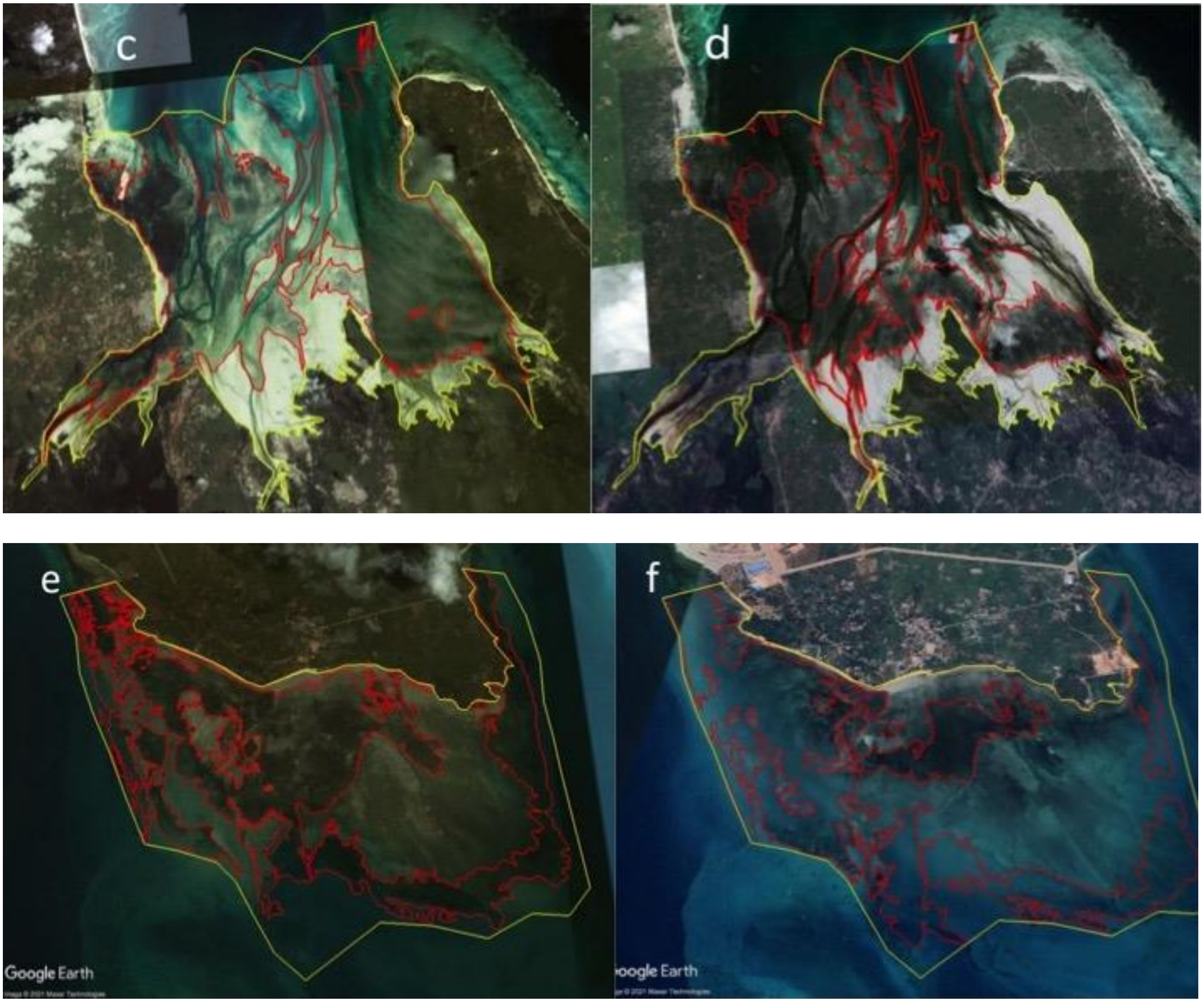
Site	**Year**	**MPA management plan status**	**Scale of coastal economic activities**		
			Seaweed farming	**Small-scale fishing**	**Tourism**
Chumbe Island	2006	Implemented – Full MPA	None	None	Low
	2019	Implemented – Full MPA	None	None	Low
Chwaka Bay	2006	None	High	High	Low
	2019	Implemented – Partial MPA	High	High	Medium
Fumba	2006	None	None	Medium	Low
	2019	Implemented – Partial MPA	None	Medium	Low
Jambiani	2006	None	High	High	Low
	2019	Implemented – Partial MPA	High	Low	High
Nungwi	2006	None	None	High	Low
	2019	Implemented – Partial MPA	None	High	High

Table 1. Site characteristics of MPA management plan status and scale of coastal economic activities in 2006 and 2019

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 (e.g., Amran, 2017; Hossain et al., 2016; Knudby & Nordlund, 2011). First, we mapped the distribution of seagrass beds and meadows at both time points using Google Earth Engine images at a resolution of approximately 15 m. We only included seagrass beds and meadows located in relatively shallow areas (<5 m) to ensure data quality. We mapped seagrass beds and meadows in heterogeneous polygons to measure the area of each patch. Studies suggest that seagrass density and cover are highly dynamic due to seasonal variances in rainfall and wind (Aller, 2018). Therefore we selected primary satellite images for each site to create polygons, and then we compared the polygons to images taken during other seasons within a 1- to 2-year period to ensure completeness. For sites that did not have clear satellite images in 2006 or 2019, we selected the primary image between the range of 2005-2007 and 2020-2021 and compared the mappings to 2006 and 2019 images for accuracy. Images of final site mappings are displayed in Figures 6a-j. One challenge to mapping was distinguishing algae-covered coral reefs from seagrasses in sites with high coral mortality, such as at Nungwi, so we did not include seagrasses located in coral reefs (for example, see Figure 7).




c
d
e
f
Google Earth
oogle Earth

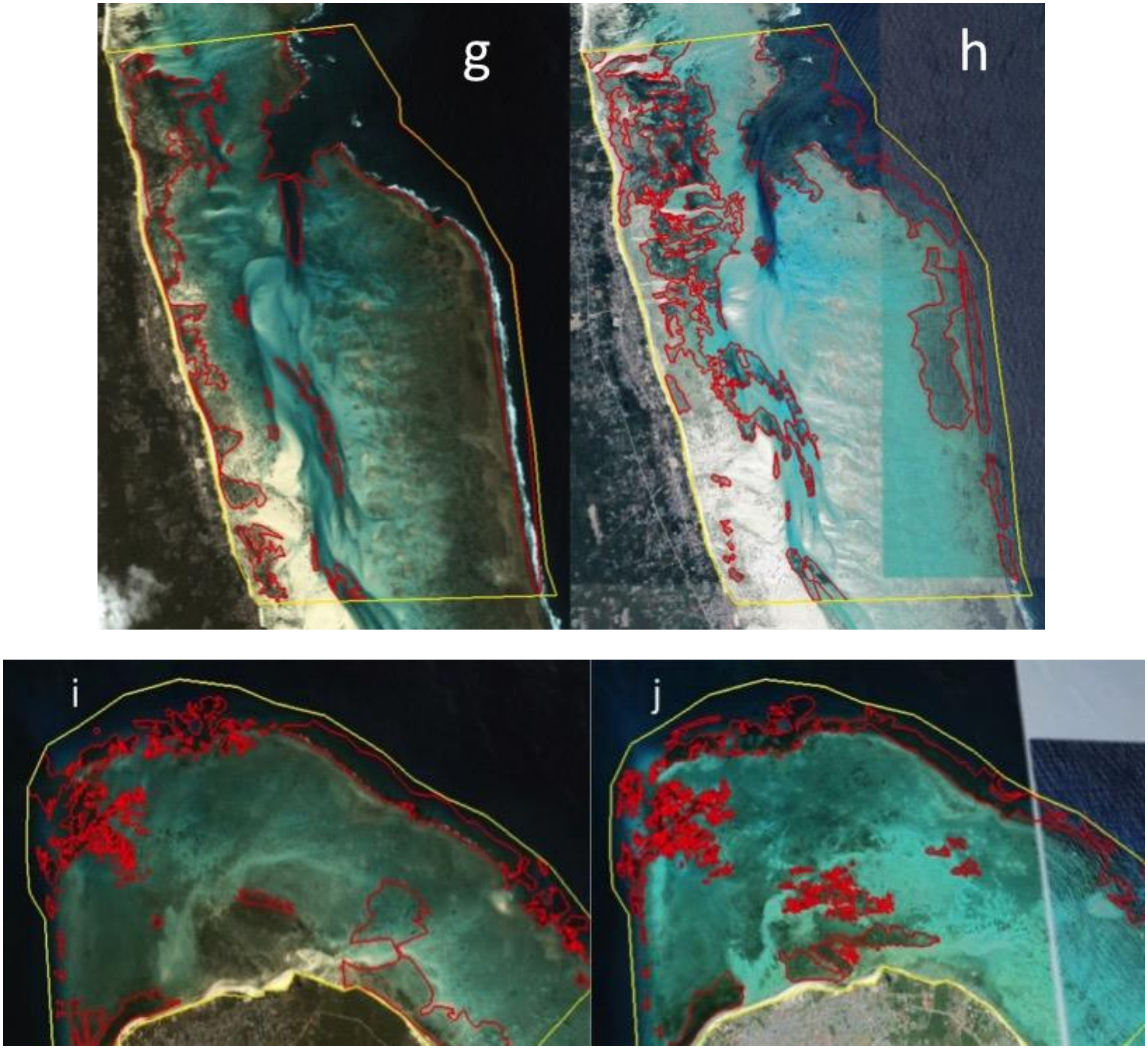


Figure 6a-j. Google Earth images of site mappings: a) Chumbe Island, October 2005; b) Chumbe Island, February 2021; c) Chwaka Bay, December 2006; d) Chwaka Bay, January 2019; e) Fumba, January 2006; f) Fumba, July 2019; g) Jambiani, September 2005; h) Jambiani, August 2020; i) Nungwi, November 2005; and j) Nungwi, February 2019

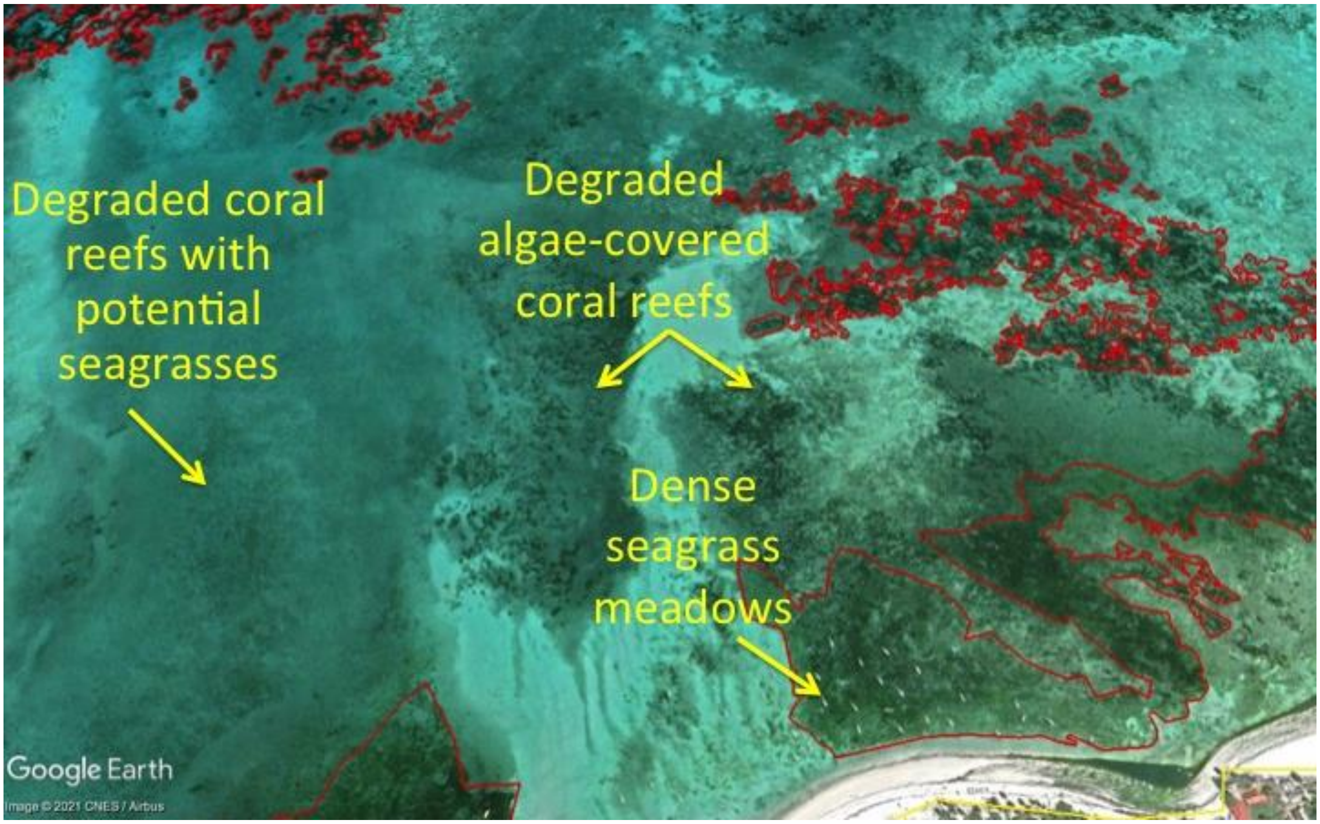


Figure 7. Examples of degraded coral reefs with potential seagrasses, degraded algae-covered coral reefs, and dense seagrass meadows located in Nungwi, February 2019

Once we completed the mappings, we calculated the percentage of seagrass cover for each site by 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. We then calculated the percent change in seagrass cover from 2006 to 2019 per site.

Statistical Analyses

All analyses were conducted with StatPlus:mac v5.0 statistical analysis software (AnalystSoft Inc., 2021). First, we ran two-samples paired t-tests to compare the percent of seagrass coverage in 2006 and 2019 in the validation field dataset and the study GIS dataset. To determine whether the implementation of MPA management plans, coastal economic activities (seaweed farming, small-scale fishing, and tourism), or the combination of coastal economic activities were correlated to changes in seagrass coverage from 2006 to 2019 at selected sites, we conducted Spearman's rank correlations tests and generated a scatterplot.

Results

Descriptive Statistics

Total mapping areas, percent of seagrass cover in 2006 and 2019, and total percent change for each site are listed in Table 2. Chwaka Bay had the largest mapping area of 59.12 km^2, and Chumbe Island had the smallest area of 0.57 km^2, defined by the boundaries of the no-take zone. Chwaka Bay also had the highest percentage of seagrass cover in both measurement periods (62.06% and 51.52%), 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 11.76% 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%). Seaweed farms were visible in both satellite images of Jambiani (see Figure 9).

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	36.69	62.06%	30.46	51.52%	-16.98%
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 2. Total Area and Percentage of Seagrass Cover per Site

Validation

Table 3 lists the percentage of seagrass cover in 2006 and 2019 and the percent change for the validation and study datasets. In the paired two-samples t-test, we found no significant difference between the validation and study datasets, $t(3) = 2.65$, $p = 0.230$. Therefore we concluded that the Chumbe Island GIS data adequately aligned with the field data and that the mappings of satellite images were valid.

Dataset	Percentage of seagrass cover in 2006	Percentage of seagrass cover in 2019	Percent change from 2006 to 2019
Validation	12.51%	8.43%	-32.61%
Study	48.11%	39.08%	-18.76%

Table 3. Percentage of seagrass cover and percent change from 2006 to 2019 for Validation and Study Datasets

Correlation Statistics

Table 4 lists the correlation coefficients and p-values for MPA management plans and coastal economic activities. Spearman's rank correlation coefficients in the range of 0.70-0.79 are considered strong correlations (Akoglu, 2018). We found a strong negative correlation between changes in seagrass coverage from 2006 to 2019 and the scale of tourism, $r_s(9) = -0.74$, $p = 0.044$ (see Figure 8). In other words, seagrass coverage is more likely to be lower in sites with higher-scale tourism. There were no significant correlations between changes in seagrass coverage and MPA management plan implementation ($r_s(9) = 0.06$, $p = 0.972$), seaweed farming ($r_s(9) = 0.23$, $p = 0.426$), small-scale fishing ($r_s(9) = -0.16$, $p = 0.734$), or the combined score for coastal economic activities ($r_s(9) = -0.08$, $p = 0.825$).

MPA or coastal economic activity	Correlation coefficient (r_s)	p-value
MPA management plan	0.06	0.972
Seaweed farming	0.23	0.426
Small-scale fishing	-0.16	0.734
Tourism	-0.74	0.044
Combined coastal economic activities	-0.08	0.825

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 economic activities between 2006 and 2019

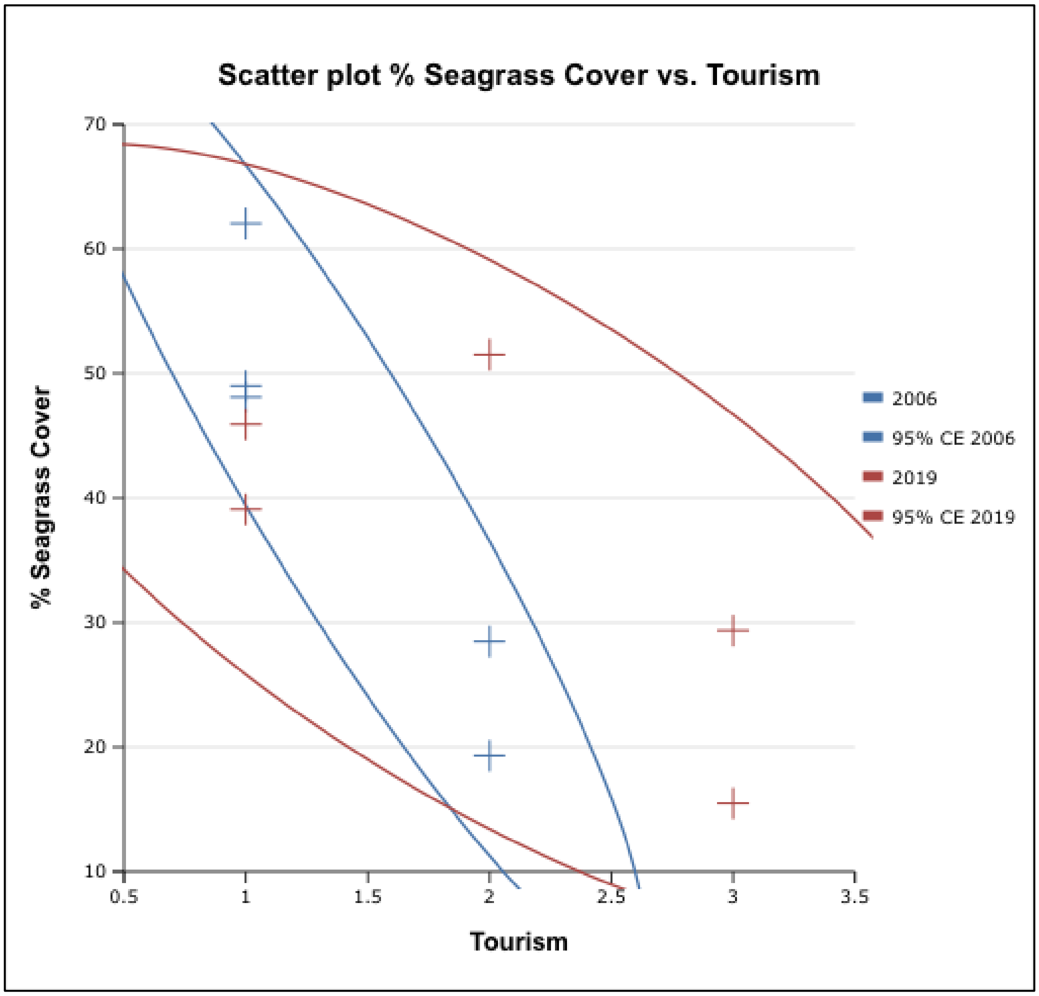


Figure 8. Scatter plot and 95% confidence intervals of the relationship between the scale of tourism and the percent of seagrass cover in 2006 compared to 2019

Discussion

This study is the first to measure and compare the relative impacts of different coastal economic activities on seagrass cover. Generally, coastal sites around Zanzibar have seen a decrease in seagrass cover over time, and our findings add to the literature demonstrating links between anthropogenic impacts and seagrass declines (Aller et al., 2019; Nchimbi & Lyimo, 2019). As expected, tourism demonstrated the greatest negative impact on seagrass cover from 2006 to 2019, whereas the use of seaweed farming, small-scale fishing, or the combination of activities were not significantly correlated with changes in seagrass cover. The results of the impact of tourism on seagrasses align with previous research conducted in Zanzibar and other coastal communities. 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). Another study in Indonesia found a similar trend of increased nutrient loading and eutrophication due to sewage discharge from beachside tourist cabins that resulted in significant decreases in the seagrass cover of multiple species (Short et al., 2014). Danovaro et al. (2020) evaluated the drivers of seagrass declines in the Adriatic Sea over a 40-year period and concluded that urban development and growth in blue tourism were the greatest predictors of seagrass declines from 1973-2013. Although the links between tourism and deteriorating marine environmental conditions are well established, the current study is the first to the authors' knowledge to directly draw correlations between growing tourism and long-term seagrass declines in Zanzibar.

It is somewhat surprising that neither seaweed farming nor small-scale fishing were significantly associated with seagrass degradation. Notably, small-scale fishing had a weak negative correlation with the percent of seagrass cover ($r_s(9) = -0.16$, $p = 0.734$), suggesting that small-scale fishing still has a negative impact on seagrasses that warrants further investigation. Prior evidence of the long-term impacts of both practices individually or in combination on seagrass health is inconclusive or lacking (Gullström et al., 2012). For example, in Chwaka Bay where high scales of fishing and seaweed farming have been practiced for over three decades, de la Torre-Castro et al. (2014) found that small-scale fishers fish in seagrass meadows more often than in coral reefs or mangrove ecosystems and most commonly use dragnets for their fishing equipment. Although they concluded that fishing pressures are highest among seagrasses, they did not directly assess whether these fishing practices led to long-term degradation to seagrasses. Moreira-Saporiti et al. (2021) assessed seaweed farming pressures on seagrasses over a 9-week period in Chwaka Bay. Seagrasses in seaweed farming plots had significant decreases in sunlight and shoot density of *Thalassia hemprichii* and *Halophila stipulacea*, but they only found a loss of shoots in *H. stipulacea*. Trampling did not significantly reduce seagrass shoots. Lyimo et al. (2008) compared seagrass biomass in intertidal zones with and without seaweed farms in Chwaka Bay and Jambiani six times over a 2-year period, and the results demonstrated that seagrasses in seaweed farm plots had significantly lower biomass in both sites. Notably, none of these studies assessed seagrass quantity and quality over a measurement period of >2 years. In contrast, Gullström et al. (2006) found seagrass cover in Chwaka Bay to be stable over a nearly 20-year period, suggesting that seaweed farming and fishing pressures are not exerting significant long-term declines in seagrass density. Contrary to the findings of Gullström et al. (2006), the current study found a 13-year trend of declining seagrass cover in Chwaka Bay and three other sites, but it is not yet clear what is driving these declines in areas where tourism is not exerting extreme pressures on the coastline. 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.

Interestingly, seaweed farming had a weak positive correlation with seagrass cover ($r_s(9) = 0.23$, $p = 0.426$), suggesting that areas with higher scales of seaweed farming were more likely to have greater seagrass cover. Jambiani was the only site to increase 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 (see Figure 9; Lyimo et al., 2006; Gullström et al., 2012). 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). The current study's findings of seagrass improvement in Jambiani and the weak positive correlation between seaweed farming and the percent of seagrass cover further suggest that trampling and shading seagrasses in seaweed farms may not have long-term negative impacts on seagrasses. These findings indicate that seaweed-farming regions may be more invested in the conservation of seagrasses due to the essential ecosystem services seagrasses provide to the benefit of seaweed mariculture. Additional research is needed to determine whether these findings can be replicated in Zanzibar and in seaweed farming communities in other countries, which may have important implications for the management and conservation of seagrass meadows globally.

Figure 9. Seaweed farms located on bare sand in the intertidal and subtidal zones of Jambiani, September 2005

Contrary to the study hypothesis, the implementation of MPA management plans did not have a significant protective effect on seagrass cover ($r_s(9) = 0.06$, $p = 0.972$). 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. Previous research evaluating MPA outcomes suggests that success depends on the size and location of the MPA, the integration of local and regional priorities in planning, the scale of implementation and enforcement, and environmental factors that originate outside the boundaries of MPAs (Aller, 2018; de la Torre-Castro et al., 2014; Smyth & Hanich, 2019). In particular, 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 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. Another study conducted in the Philippines determined that land use activities and maintenance of watersheds were stronger predictors of seagrass conservation successes than biological or environmental protections (Quiros et al., 2017). One explanation for the current study's findings that implementation of MPA management plans did not have a significant protective effect on seagrass cover is that land use conversion and increased water pollution related to increasing tourism and population levels have stronger impacts on seagrasses than conservation efforts within MPAs. Additional research is needed to evaluate specific characteristics of Zanzibar MPAs, effects of coastal land use, 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, etc.) that influence the effectiveness of MPAs could be a useful tool for integrated coastal marine planning and management.

The overall trend of the decline in seagrass cover across Unguja Island is consistent with the findings of other research (Gullström et al., 2012; Nchimbi & Lyimo, 2019). Seagrass declines have myriad implications for the coastal marine environment and Zanzibar's communities, including a loss of habitats, nurseries, and food sources for pelagic fish, sea turtles, dugongs, marine invertebrates, and other species; disruptions in sediment nutrient cycling; changes to geomorphology in the benthic environment that make coastlines more prone to floods and erosion (Nchimbi & Lyimo, 2019; Nordlund et al., 2016); and both decreases in carbon sequestration and increases in carbon emissions resulting from disturbances (Pendleton et al., 2012). For example, a study in Mauritius found the removal of seagrass beds during the development of a hotel property led to higher water turbidity due to seabed destabilization, loss of benthic infauna, and 65-72% reductions in seagrass biomass and carbon storage (Daby, 2003). Seagrass loss also has economic consequences for mariculturists, including seaweed farmers that rely on the sediment nutrient enrichment of seagrasses to produce seaweed and small-scale fishermen who fish in seagrass habitats (de la Torre-Castro et al., 2014; Nordlund et al., 2016). In addition, the loss of marine biodiversity supported by seagrasses may exacerbate food insecurity issues in Zanzibar (de la Torre-Castro et al., 2014) and will likely negatively impact the tourism industry. The continuing decline of seagrasses comes at great ecological, economic, and public health losses to 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 in 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 economic activity should be interpreted with caution as detailed data on the number of seaweed farmers and small-scale fishers at each site were not available. Other measures that are currently not publicly reported, such as annual volume of seaweed production and fishing catches per village, may be better indicators of the impacts of mariculture industries on seaweed ecosystems. In addition to collecting and reporting detailed data on the annual production of mariculture products such as seaweed and fish, future research should continue to evaluate the long-term impact of mariculture on seagrass quality and quantity in Zanzibar and determine whether some mariculture practices may have a protective on seagrasses, such as removing sea urchins from seaweed farms or restoring seagrass habitats for pelagic fish.

Recommendations

We propose several recommendations to improve the restoration, conservation, and management of seagrasses in Zanzibar. First, coastal marine management plans should identify seagrass meadows as critical ecosystems and develop evidence-based plans to restore and conserve seagrass meadows. Local seagrass restoration projects in Zanzibar have successfully demonstrated the feasibility of restoring and conserving seagrasses (J.K.U. Omar, personal communication, July 27, 2021), which could be scaled up with extension service support from the Ministry of Livestock and Agriculture. 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 create an ecotourism network similar to Ecotourism Kenya (https://ecotourismkenya.org/) that uses a transparent rating system to evaluate participating hotels and lodges on their environmental practices. Ecotourism is generally defined as nature-based tourism that aims to achieve one or more of the following: environmental conservation, economic and social benefits for local communities, preservation of local resources, and educational programing for visitors (Shasha et al., 2020). Global ecotourism has grown at an annual rate between 10-30% and is expected to continue growing (Shasha et al., 2020); a survey conducted in 2018 found that 87% of tourists desire to travel sustainably (Travel Agent Central Newsdesk, 2018). 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. Zanzibar's tourism sector has an opportunity to simultaneously attract the business of ecotourists and improve the restoration and conservation of seagrass ecosystems. Providing tourists with a transparent rating system that includes indicators relevant to seagrass conservation, such as implementing seagrass and coral reef restoration projects and instituting solid and sewage waste systems, could redirect a substantial proportion of tourism business to hotels and lodges taking actions to promote sustainable tourism.

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, requiring noncompliant businesses to compensate local communities for ecological damage, or instituting tourism taxes that fund waste management systems and conservation projects. Levy and taxation models implemented by municipalities and countries around the world have demonstrated a range of reductions in the use of single-use plastics between from 33-94%, resulting in large decreases in marine pollution (Schnurr et al., 2018). 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.

It is important to acknowledge that enforcing regulations, imposing taxes, or levying fines on hotels and lodges may have unintended consequences in Zanzibar. For example, hoteliers and investors may choose to relocate their businesses to areas without equivalent regulations, which can harm local communities by diminishing economic opportunities created directly or indirectly by tourism. This can further harm seagrasses and other coastal marine ecosystems by increasing the number of people who rely on environmental resources for their livelihoods, which can push the carrying capacity of coastal marine ecosystems to their limits. In addition, imposing taxes or levying fines without proper oversight and transparency can promote corruption. A potential approach to prevent or minimize these negative unintended consequences is to develop contingency plans in consultation with local community members. For example, in addition to identifying potential new ecotourism-focused investors, communities could apply for grants or low-interest loans that provide opportunities for local communities to start their own sustainable community-owned ecotourism businesses or other enterprises. We also recommend the formation of community oversight committees comprised of men and women who ensure that tourism taxes or fines are properly collected and reinvested in the community and in the environment. Although complex and multifaceted tourism sector reform would require a significant investment in time, financial resources, community input, and technical support, reform could benefit and empower local communities while simultaneously protecting and conserving seagrasses and other coastal marine ecosystems. It is with hope that decision-makers in Zanzibar act quickly because the importance of seagrasses and their surmounting threats cannot be ignored any longer.

Conclusion

The current study demonstrated a weak correlation between small-scale fishing and seagrass cover decline in Zanzibar from 2006 to 2019, as expected. We also found strong support for our hypothesis that tourism, compared to seaweed farming or small-scale fishing, significantly contributed to the decline in seagrass cover. Prior evidence suggests that harmful tourism practices, such as the disposal of hotel sewage into the ocean and the removal of seagrasses to manufacture attractive white beaches, are increasingly used in Zanzibar and degrade seagrasses and other coastal marine ecosystems (Khamis et al., 2017; Staehr et al., 2018). In contrast, the results showed that seaweed farming had a weak, though insignificant, protective effect on seagrass cover. This suggests that seagrasses and seaweed may have a symbiotic relationship; seagrasses create nutrient-rich environments that support seaweed growth, and seaweed farmers may protect seagrass meadows in turn (Hedberg et al., 2018; Lyimo et al., 2008). Surprisingly, we did not find support for our hypothesis that the implementation of MPA management plans would have a protective effect on seagrass cover, suggesting that poverty-driven community reliance on coastal resources and the ineffective enforcement of MPA regulations may limit the effectiveness of MPAs throughout Zanzibar (UNEP-Nairobi Convention & 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 it is the first of its kind to directly measure and compare the long-term impacts of coastal economic activities and the implementation of MPA management plans on seagrasses in Zanzibar. A key takeaway is that land use and other impacts that affect water quality, such as pollution runoff, can have a greater influence on seagrasses than short-term environmental impacts, such as increases in water temperature. As such, Zanzibar would benefit from developing an integrated coastal marine management plan that prioritizes increasingly vulnerable seagrass ecosystems and uses a multi-sectoral approach that expands its scope to include the impact of land uses and urban development. Additional research is needed to substantiate our findings on the anthropogenic threats toward seagrasses and to identify cost-effective methods for protecting seagrasses that can be incorporated in MPA management plans.

The decline of seagrasses in Zanzibar will have important implications for the essential ecosystem services seagrasses provide. The loss of seagrasses as food sources and habitats for aquatic and terrestrial species and the reduction of sediment nutrient enrichment threaten the unique biodiversity of the Western Indian Ocean (Nchimbi & Lyimo, 2019; Nordlund et al., 2016). The depletion of seagrasses as sediment stabilizers and natural storm buffers may compromise the resilience of coastlines in the face of increasing climate change-driven natural disasters (Nchimbi & Lyimo, 2019; Nordlund et al., 2016). Large-scale declines in seagrasses also decrease the volume of blue carbon they sequester, an essential ecosystem service for mitigating ocean acidification and climate change, and could increase carbon emissions during disturbances (Pendleton et al., 2012). Zanzibar's environment and economy also depend on finding the right balance of conservation and development; maintaining healthy, biodiverse coastal marine environments will ensure the sustainability of the natural resources on which the tourism and mariculture sectors depend. However, if the welfare of seagrasses and other marine ecosystems are ignored, the mariculture and tourism industries will suffer as well. These trends are not isolated to Zanzibar; global declines in seagrasses threaten marine environments and sustainable economic growth and contribute to climate change everywhere (Nchimbi & Lyimo, 2019).

Zanzibar has to make difficult decisions to determine the most appropriate and resource-efficient methods of integrated coastal marine management that protect vital seagrass ecosystems while balancing the growing demands of tourism and development. By implementing innovative solutions, such as instituting seagrass restoration and conservation projects, promoting more sustainable ecotourism practices, and increasing the accountability of the tourism sector, Zanzibar can reverse the trend of seagrass declines to the benefits of its people, the coastal marine environment, and the world.

References

Akoglu, H. (2018). User's guide to correlation coefficients. *Turkish journal of emergency medicine*, *18*(3), 91-93.

Aller, E. A. (2018). *Effects of marine protected areas on tropical seagrass ecosystems* (Doctoral dissertation, Department of Ecology, Environment and Plant Sciences, Stockholm University).

Aller, E. A., Eklöf, J. S., Gullström, M., Kloiber, U., Linderholm, H. W., & Nordlund, L. M. (2019). Temporal variability of a protected multispecific tropical seagrass meadow in response to environmental change. *Environmental monitoring and assessment*, *191*(12), 1-15.

Amran, M. A. (2017). Mapping Seagrass Condition Using Google Earth Imagery. *Journal of Engineering Science & Technology Review*, *10*(1).

Belshe, E. F., Hoeijmakers, D., Herran, N., Mtolera, M., & Teichberg, M. (2018). Seagrass community-level controls over organic carbon storage are constrained by geophysical attributes within meadows of Zanzibar, Tanzania. *Biogeosciences*, *15*(14), 4609-4626.

Burkholder, J. M., Tomasko, D. A., & Touchette, B. W. (2007). Seagrasses and eutrophication. *Journal of experimental marine biology and ecology*, *350*(1-2), 46-72.

Cullen-Unsworth, L. C., Nordlund, L. M., Paddock, J., Baker, S., McKenzie, L. J., & Unsworth, R. K. (2014). Seagrass meadows globally as a coupled social–ecological system: Implications for human wellbeing. Marine Pollution Bulletin, 83(2), 387-397.

Daby, D. (2003). Effects of seagrass bed removal for tourism purposes in a Mauritian bay. *Environmental Pollution*, *125*(3), 313-324.

Danovaro, R., Nepote, E., Martire, M. L., Carugati, L., Da Ros, Z., Torsani, F., ... & Corinaldesi, C. (2020). Multiple declines and recoveries of Adriatic seagrass meadows over forty years of investigation. *Marine Pollution Bulletin*, *161*, 111804.

de la Torre-Castro, M., Di Carlo, G., & Jiddawi, N. S. (2014). Seagrass importance for a small-scale fishery in the tropics: The need for seascape management. *Marine Pollution Bulletin*, *83*(2), 398-407.

Eklöf, J. S., Fröcklin, S., Lindvall, A., Stadlinger, N., Kimathi, A., Uku, J. N., & McClanahan, T. R. (2009). How effective are MPAs? Predation control and 'spill-in effects' in seagrass–coral reef lagoons under contrasting fishery management. *Marine Ecology Progress Series*, *384*, 83-96.

Eklöf, J. S., Msuya, F. E., Lyimo, T. J., & Buriyo, A. S. (2012). Seaweed farming in Chwaka Bay: a sustainable alternative in aquaculture. *People, nature and research in Chwaka Bay. WIOMSA, Zanzibar, Tanzania*, 213-233.

Githaiga, M. N., Gilpin, L., Kairo, J. G., & Huxham, M. (2016). Biomass and productivity of seagrasses in Africa. *Botanica marina*, *59*(2-3), 173-186.

Gullström, M., Lundén, B., Bodin, M., Kangwe, J., Öhman, M. C., Mtolera, M. S., & Björk, M. (2006). Assessment of changes in the seagrass-dominated submerged vegetation of tropical Chwaka Bay (Zanzibar) using satellite remote sensing. *Estuarine, Coastal and Shelf Science*, *67*(3), 399-408.

Gullström, M., Lyimo, T. J., Eklöf, J. S., Björk, M., Semesi, I. S., & de la Torre-Castro, M. (2012). Seagrass meadows in Chwaka Bay: Socio-ecological and management aspects.

Hedberg, N., von Schreeb, K., Charisiadou, S., Jiddawi, N. S., Tedengren, M., & Nordlund, L. M. (2018). Habitat preference for seaweed farming–A case study from Zanzibar, Tanzania. *Ocean & Coastal Management*, *154*, 186-195.

Hossain, M. S., Bujang, J. S., Zakaria, M. H., & Hashim, M. (2016). Marine and human habitat mapping for the Coral Triangle Initiative region of Sabah using Landsat and Google Earth imagery. *Marine Policy*, *72*, 176-191.

IUCN. (2020). Worldwide catalogue of case studies on Aquaculture and Marine Conservation, N°1: Zanzibar. IUCN - Gland, Switzerland.

Jebasingh, S. E. J., Lakshmikandan, M., Sivaraman, K., & Uthiralingam, M. (2016). Assessment of antibacterial, antifungal property and purification of bioactive compounds from seagrass, Thalassia hemprichii. *Proceedings of the National Academy of Sciences, India Section B: Biological Sciences*, *86*(4), 905-910.

Jiddawi, N. S., & Öhman, M. C. (2002). Marine fisheries in Tanzania. *Ambio: a journal of the Human Environment*, *31*(7), 518-527.

Khamis, Z. A., Kalliola, R., & Käyhkö, N. (2017). Geographical characterization of the Zanzibar coastal zone and its management perspectives. *Ocean & Coastal Management*, *149*, 116-134.

Knudby, A., & Nordlund, L. (2011). Remote sensing of seagrasses in a patchy multi-species environment. *International Journal of Remote Sensing*, *32*(8), 2227-2244.

Lin, H. J., Lee, C. L., Peng, S. E., Hung, M. C., Liu, P. J., & Mayfield, A. B. (2018). The effects of El Niño-Southern Oscillation events on intertidal seagrass beds over a long-term timescale. *Global change biology*, *24*(10), 4566-4580.

Lyimo, T. J., Mvungi, E. F., Lugomela, C., & Björk, M. (2006). Seagrass biomass and productivity in Seaweed and Non-Seaweed Farming areas in the East Coast of Zanzibar. *Western Indian Ocean Journal of Marine Science*, *5*(2), 141-152.

Lyimo, T. J., Mvungi, E. F., & Mgaya, Y. D. (2008). Abundance and diversity of seagrass and macrofauna in the intertidal areas with and without seaweed farming activities in the east coast of Zanzibar. *Tanzania Journal of Science*, *34*.

McLean, B., Hikmany, A. N., Mangora, M., & Shalli, M. (2012). Legal and institutional framework for effective management of marine managed areas in Tanzania, Zanzibar Report. *Zanzibar Report (January 1, 2012)*.

Moreira-Saporiti, A., Hoeijmakers, D., Msuya, F. E., Reuter, H., & Teichberg, M. (2021). Seaweed farming pressure affects seagrass and benthic macroalgae dynamics in Chwaka Bay (Zanzibar, Tanzania). *Regional Environmental Change*, *21*(1), 1-12.

Msuya, F.E. (2013). Social and economic dimensions of carrageenan seaweed farming in the United Republic of Tanzania. Social and economic dimensions of carrageenan seaweed farming, pp. 115–146. *Fisheries and Aquaculture Technical Paper No. 580*. Rome, FAO. 204 pp.

Msuya, F. E. (2020). Seaweed resources of Tanzania: status, potential species, challenges and development potentials. *Botanica Marina*, *63*(4), 371-380.

Nchimbi, A. A., & Lyimo, L. D. (2019). Socioeconomic Determinants of Mangrove Exploitation and Seagrass Degradation in Zanzibar: Implications for Sustainable Development. *Journal of Marine Biology, 2019*.

Nordlund, L.M., Koch, E. W., Barbier, E. B., & Creed, J. C. (2016). Seagrass ecosystem services and their variability across genera and geographical regions. *PLoS One*, *11*(10), e0163091.

Nowicki, R. J., Thomson, J. A., Burkholder, D. A., Fourqurean, J. W., & Heithaus, M. R. (2017). Predicting seagrass recovery times and their implications following an extreme climate event. *Marine Ecology Progress Series*, *567*, 79-93.

Ochieng, C. A., & Erftemeijer, P. L. A. (1993). Kenya and Tanzania. *World Atlas of Seagrasses*, *82*.

Pendleton, L., Donato, D. C., Murray, B. C., Crooks, S., Jenkins, W. A., Sifleet, S., ... & Baldera, A. (2012). Estimating global "blue carbon" emissions from conversion and degradation of vegetated coastal ecosystems.

Quiros, T. A. L., Croll, D., Tershy, B., Fortes, M. D., & Raimondi, P. (2017). Land use is a better predictor of tropical seagrass condition than marine protection. *Biological conservation*, *209*, 454-463.

Rehren, J., Samoilys, M., Reuter, H., Jiddawi, N., & Wolff, M. (2020). Integrating Resource Perception, Ecological Surveys, and Fisheries Statistics: A Review of the Fisheries in Zanzibar. *Reviews in Fisheries Science & Aquaculture*, 1-18.

Schnurr, R. E., Alboiu, V., Chaudhary, M., Corbett, R. A., Quanz, M. E., Sankar, K., ... & Walker, T. R. (2018). Reducing marine pollution from single-use plastics (SUPs): A review. *Marine pollution bulletin*, *137*, 157-171.

Shasha, Z. T., Geng, Y., Sun, H. P., Musakwa, W., & Sun, L. (2020). Past, current, and future perspectives on eco-tourism: a bibliometric review between 2001 and 2018. *Environmental Science and Pollution Research*, *27*(19), 23514-23528.

Short, F. T., Coles, R., Fortes, M. D., Victor, S., Salik, M., Isnain, I., ... & Seno, A. (2014). Monitoring in the Western Pacific region shows evidence of seagrass decline in line with global trends. *Marine Pollution Bulletin*, *83*(2), 408-416.

Smyth, C., & Hanich, Q. A. (2019). Large scale marine protected areas: Current status and consideration of socio-economic dimensions.

Staehr, P. A., Sheikh, M., Rashid, R., Ussi, A., Suleiman, M., Kloiber, U., ... & Muhando, C. (2018). Managing human pressures to restore ecosystem health of Zanzibar coastal waters. *J. Aquac. Mar. Biol*, *7*, 59-70.

Travel Agent Central Newsdesk. (2018). Stats: 87% of Travelers Want to Travel Sustainably. Available at: https://www.travelagentcentral.com/running-your-business/stats-87-travelers-want-to-travel-sustainably.

UNdata. (2021). Statistics: United Republic of Zanzibar per capita GDP at current prices – US dollars. Available at: http://data.un.org/Data.aspx?d=SNAAMA&f=grID:101;currID:USD;pcFlag:1;crID:836.

United Nations Environment Programme (UNEP), (2020). Opportunities and Challenges for Community-Based Seagrass Conservation. UNEP, Nairobi, Kenya.

UNEP-Nairobi Convention & WIOMSA. (2021). Western Indian Ocean Marine Protected Areas Outlook: Towards achievement of the Global Biodiversity Framework Targets. UNEP and WIOMSA, Nairobi, Kenya, 298 pp.

United Republic of Tanzania. (2008). Marine Fisheries Frame Survey Results for 2007. Ministry of Livestock and Fisheries Development (Mainland) and Ministry of Agriculture, Livestock and Environment (Zanzibar).

Unsworth, R. K., & Cullen, L. C. (2010). Recognising the necessity for Indo-Pacific seagrass conservation. *Conservation Letters*, *3*(2), 63-73.

Unsworth, R. K., McKenzie, L. J., Nordlund, L. M., & Cullen-Unsworth, L. C. (2018). A changing climate for seagrass conservation?. *Current Biology*, *28*(21), R1229-R1232.

World Bank. (2019). Zanzibar: A Pathway to Tourism for All. Integrated Strategic Action Plan. Available for download at: https://documents.worldbank.org/en/publication/documents-reports/documentdetail/993701565250577192/zanzibar-a-pathway-to-tourism-for-all-integrated-strategic-action-plan.

Zanzibar Department of Fisheries. (2021). Seaweed Production in Zanzibar 2010-2019. Zanzibar, Tanzania.