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Mangrove cover change detection in the Rufiji Delta in Tanzania

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Abstract
The Rufiji Delta is a critical ecosystem that comprises 50% of mangrove cover in mainland Tanzania, providing a plethora of ecosystem services that support diverse livelihood options. However, the rate of mangrove loss, especially due to rapid expansion of rice farming in the delta, is alarming. Landsat imagery from 1991, 2000, 2009 and 2015 was used to estimate total current mangrove area in the delta. There has been loss of mangroves from 51,941 ha estimated in 1991 to 45,519 ha in 2015, which is an annual loss rate of 0.52%. Clearance for rice farming expanded from 5,344 ha in 1991 to 12,642 ha in 2015. The average mangrove annual loss and gain from 1991 to 2015 were estimated to be 378 ha and 109 ha respectively. The consequences of this loss are not well appreciated. The present analysis serves as an updated baseline to inform the urgent need for coordination of multiple stakeholders to address the complex drivers of mangrove loss to secure the future of mangroves and ecosystem services in the delta.

Keywords: Rufiji Delta; mangrove forest change; Landsat; deforestation; rice farming

Introduction
Mangrove forests in mainland Tanzania are categorized as State Forest Reserves by the Forest Act of 2002 (URT, 2002). They occur along almost the entire coastline in continuous and fragmented stands (Mangora *et al.,* 2016). Recent estimates by the National Forest Resources Monitoring and Assessment (NAFORMA) indicate that mangroves cover approximately 158,100 ha, which is about 0.3% of the total forest area in the country (MNRT, 2015). Although this represents a small proportion at the country level, mangroves form a critical interface between terrestrial, estuarine, and near-shore marine ecosystems. Where they occur, mangroves are important economic and ecological resources for communities, providing useful products such as firewood, charcoal, poles and traditional medicines, and support for fisheries (Masalu, 2003; Wang *et al.,* 2003; Mangora *et al.,* 2016). For instance, about 80% of all wild shrimp catches in the country are associated with mangrove forests in the Rufiji Delta (Masalu, 2003).

Despite the critical value of mangroves, they are still exposed to degradation and deforestation due to weak law enforcement, poor management and land use prioritization, poverty and extreme dependence on natural capital (Mangora, 2011). Globally, mangroves have been reported to rapidly degrade at rates exceeding loss in many other tropical forests (Polidoro *et al.,* 2010; Giri *et al.,* 2011; Jones *et al.,* 2015). They are being lost at the rate of about 1% per year (FAO, 2007); in some areas, the rate may be as high as 8% per year (Miththapala, 2008). It is estimated that 20% to 35% of the world's mangrove area has been lost since 1980 (Giri *et al.,* 2011). The rates of loss are highest in developing countries where mangroves are cleared for coastal development, aquaculture, timber and fuel production (Polidoro *et al.,* 2010). Mangroves in Tanzania are not the exception; they are being rapidly degraded and deforested through over-exploitation for poles and timber, and the conversion of forests to other uses like agriculture, aquaculture and salt making, the impact of which is not well appreciated (Mangora *et al.,* 2016). As an attempt to address these management challenges, the government has invested in developing a National Mangrove Management Plan during 1989 – 1991 (Semesi, 1992). This Plan emphasized the need

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to have close coordination among the various users of the mangrove ecosystem. The Management Plan was however not effectively implemented due to weak institutional frameworks and inadequate financial and technical resources, and has remained without revision over the years, making it obsolete (Mangora, 2011).

Assessment of the status of mangrove forest cover is thus important for proper decision making on management of mangroves, including prioritization of management activities. Yet, studies on the forest cover and land use changes on mangroves in Tanzania are limited. Nindi *et al.* (2014) reported on mangrove cover change and land use only for the northern Rufiji Delta. Brown *et al.* (2016) used three ALOS PALSAR images to investigate the spatial-temporal patterns of backscatter mechanisms in mangrove forests using target decompositions, not actually dealing with land use and land cover changes in the delta. Mwansasu (2016) reported that although the potential of over-exploitation exists, there is no significant difference between the rate of mangrove loss and gain in the delta. This is contrary to observations by other similar studies carried out in the delta, which reported significant loss of mangroves (Wang *et al.* 2003; Peter, 2013). In this study remote sensing data combined with field surveys were used to assess mangrove forest cover and land use change in the entire Rufiji Delta from 1991 to 2015. The data forms a useful updated baseline for subsequent management planning, enforcement and monitoring.

Study area

The Rufiji Delta is located about 200 km south of Dar es Salaam between latitudes 8° 20′ 00″, 7°35′ 00″S and longitudes 39°10′ 00″, 39°20′ 00″E (Fig. 1). The delta is created by the Rufiji River, the largest river basin in Tanzania, which drains about 20% of the country, with a mean annual flow of some 800 m^3 s^{-1} (Duvail and Hamerlynck, 2007). The Rufiji has a strong seasonal flow pattern, with a main flood peak around April. The delta has the largest continuous mangrove forest, covering about 50% of the total area of mangroves in Tanzania (Wang et *al.,* 2003; URT, 2009). For local management purposes, the delta has been divided in three major blocks: northern; central; and southern blocks (Semesi, 1989). The northern block contains about 46% of the total mangrove coverage, and is characterized by more freshwater input, is more accessible, and therefore more frequently utilized for local and commercial interests than the other blocks (Brown *et al.,* 2016). Information about the central and southern blocks are limited compared to the northern block because of accessibility difficulties, and therefore attract limited research interests. Eight mangrove species have been reported to occur and are well represented in the delta, namely *Avicennia marina*, *Sonneratia alba*, *Ceriops tagal*, *Lumnitzera racemosa*, *Bruguiera gymnorrhiza*, *Rhizophora mucronata*, *Xylocarpus granatum* and *Heritiera littoralis* (Wagner and Sallema-Mtui, 2016). The two missing species, *Xylocarpus molluccensis* and *Pemphis acidula* are characteristically rare in the region, potentially due to their limited geomorphological niche.

The Rufiji mangrove forest was the first to be declared a forest reserve in Tanzania in 1898 during the German colonial period (Sunseri, 2007). One of the unique features of this forest reserve is that there are legally established village settlements within it (Mwansasu, 2016), who rely on mangroves and the associated marine environment for a range of resources and ecosystem services to enhance their livelihoods (Semesi 1991; Masalu*,* 2003; Wang *et al.,* 2003). Recent estimates indicate that over 49,000 people live in and around the delta, directly engaged in rice farming, mangrove cutting for poles and timber, and fishing activities for both food and income security (Peter, 2013). Mangroves are cleared for rice farming and timber to feed other parts of Tanzania including the islands of Zanzibar. Areas dominated by *H. littoralis* are more favored for rice farming while *C. tagal*, *R. mucronata* and *B. gymnorrhiza* are heavily cut for poles, *X. granatum*, and more recently *S. alba*, are logged for timber. Other species are not considered suitable for timber. Accordingly, mangrove cover in the delta has declined over time. An inventory carried out in 1989 combining aerial photographs and ground truthing, showed that the Rufiji Delta had about 53,255 ha of mangroves (Semesi, 1992). In the year 2000, Wang *et al.* (2003) used Landsat images and estimated the total area covered by mangroves in the Rufiji Delta to be 48,030 ha. Nindi *et al.* (2014) reported a loss in the northern block of 2,865 ha of mangrove forest from 25,312 ha reported in 1989 to 22,447 ha in 2010. The structural and floristic degradation may have taken place in the delta as well, but no detailed information is available, probably due to detection complexity by remote sensing. Unpublished inventory data indicate that there is a species change towards the dominance of *C. tagal* and *A. marina* in many areas of the delta. Mshale *et al.* (2017) reported a complex governance landscape in the delta, characterized by lack of mechanisms to coordinate a diversity of resource users and conflicting conservation actors' interests that further threaten the integrity of this unique coupled human-nature system.

Materials and methods

Landsat images

Free (https://www.usgs.gov/) Landsat TM 4, 7 and 8 (path 166, row 65 and 66) images for 1991, 2000, 2009 and 2015 with 30 m spatial resolution was acquired and used to analyze and quantify the mangrove forest change from 1991 to 2015.

Figure 1. Map of the Rufiji Delta in Tanzania showing the study site.

Image processing for cover change detection

The IMPACT toolbox version 2.1.12 was used for image pre-classification. Unzipped Landsat images were saved in the IMPACT raw data file while the study site shape file was saved in the vector data. The shape file was geo-referenced to WGS-84 UTM zone 37S, which is the geographic location of the Rufiji Delta.

Pixel based classification was performed for all the images to partition digital images into multiple segments based on spectral, geometrical or computed properties (texture), together with user-defined parameters describing the size, shape and similarity compared to adjacent segments. Due to clear spectral distinctness of the classes in the Rufiji Delta, supervised classification and segmentation of images were used to classify land cover classes. The layers were coded using the Impact Tool legend. The ground truthing data and the analyst's expert knowledge were used to obtain land cover classes for each year. The land cover map for the year 2015 was processed first and a copy of it was edited as per interpretation from the image from 2009 to derive a new land cover layer for that year. The same process was repeated to generate land cover maps for the years 2000 and 1991 retrospectively. Gain and loss statistics were computed with ArcGIS, and new layers showing the status of the mangroves for the years 1991, 2000, 2009 and 2015 were obtained. Changes of other land uses into mangroves were considered as a gain of mangroves, while conversion of mangroves into other land uses was considered as a loss of mangroves.

Field data collection for validation

A field mission was organized in November 2016 to collect ground truth data to validate classified images. A Garmin GPS (Garmin inReach Explorer+, made by Garmin, USA) was used to collect 20 coordinate points in the delta for each of the 5 classified land covers (mangroves, agriculture, bare lands, non-mangroves, and water). In this study, the bare land includes salt pans and mud flats, while non-mangroves are dominated by coconut trees and *Barringtonia racemosa.* In total, 100 coordinate points were collected randomly in the delta. The coordinate points were taken from the most representative land cover class. Photos were taken and consultations with Tanzania Forest Service (TFS) Agency field officers and local farmers were conducted to broaden our understanding of the land cover changes in the delta to help in the accuracy assessment.

Classification and accuracy assessment

As recommended by Smits *et al.* (2010), a confusion matrix was used to assess and compare accuracy of the classification results. Fifty coordinate points, 10 points for each classified land use class, collected directly from the study site were used to estimate user accuracy, producer accuracy, overall accuracy,

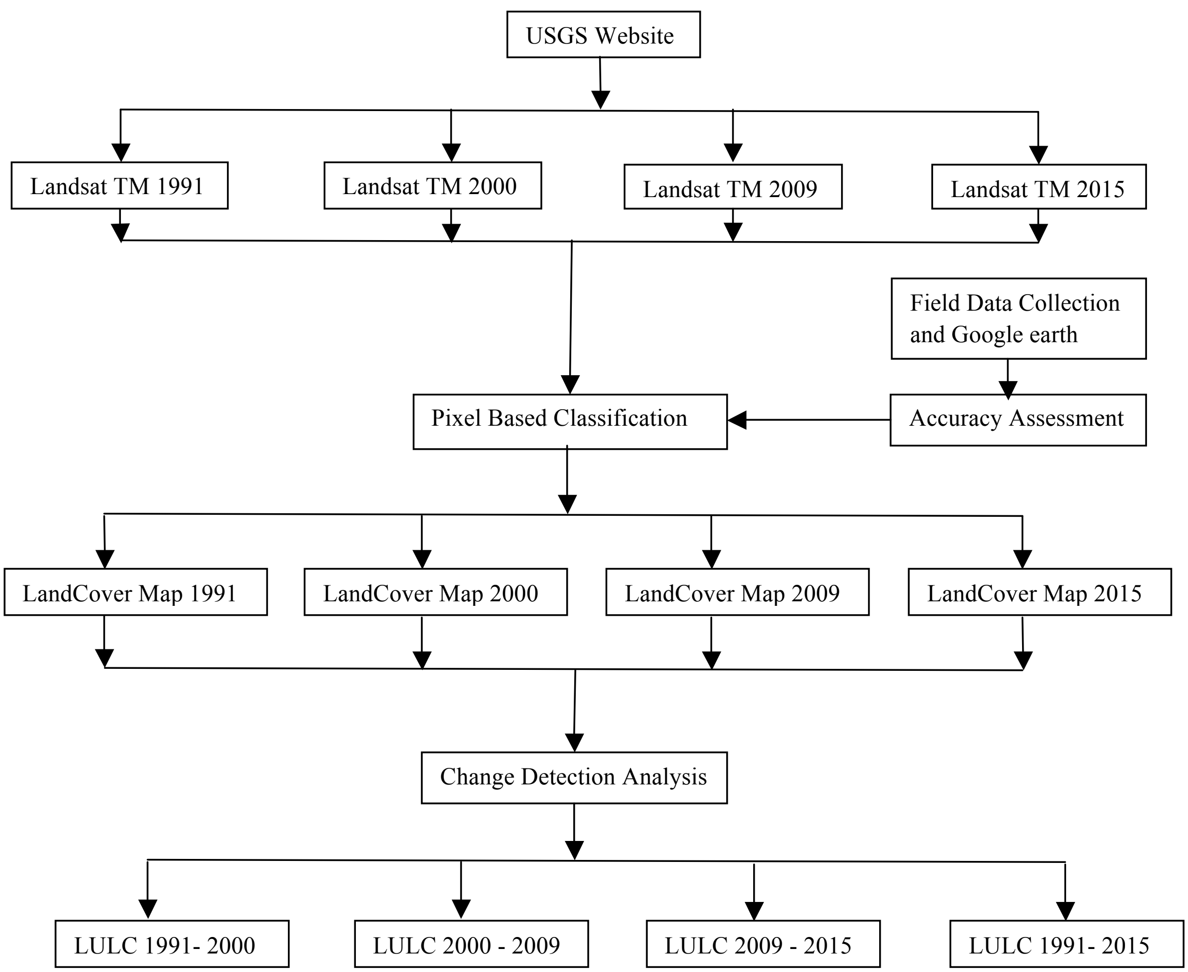

Figure 2. Flow chart showing procedures used for land use/ land cover classification.

and Kappa coefficient. A new shape file was created from fifty reference points. Reference points were aligned with pixels of classification to ensure classes in both references and classified images had similar codes. The reference points were then combined with classified images and displayed in a confusion matrix. The displayed data were then computed to obtain percentage user, producer and overall accuracies, and the Kappa coefficient.

Results

Landsat image classification showed an overall accuracy of 90%, user accuracy of 92%, and producer accuracy of about 88% for the mangrove cover in the year 2015, while the Kappa coefficient was calculated to be 86%. More cumulative loss than gain of mangrove cover was detected, from a total area of 51,941 ha estimated in 1991 to 45,519 ha in 2015 (Tables 1, 2 and Figs. 3a, b). It was estimated that 9,089 ha of mangroves were lost between 1991 and 2015 compared to 2,632 ha gained in the same period, translating to a net loss of 6,456 ha (12.4%) of mangroves in the period of 24 years (Table 2). Rice farming remains the main driver of mangrove loss in the delta, where conversion of mangrove areas for rice farming expanded from 5,344 ha in 1991 to 12,642 ha in 2015. Bare lands have constantly decreased from 19,993 ha in 1991 to 17,170 ha in 2015. Expansion and shrinking dynamics were evident for the river channels (water body) and non-mangrove areas over the analysis period.

Annual loss was estimated to be between 489 ha and 532 ha, whilst the gain was between 176 ha and 302 ha (Table 2). The average annual loss and gain were calculated at 378 ha and 109 ha respectively. Therefore, the average net annual mangrove change was calculated at -269 ha, translating into an annual rate of mangrove loss of 0.52%.

Discussion

The present analysis indicates that there have been greater mangrove losses in the Rufiji Delta in comparison with other mangrove areas in Eastern Africa between the 1980s and present. Nevertheless, the annual mangrove loss estimated in this study corresponds with the findings of Nindi *et al.* (2014), although that study concentrated on the northern block only. Similarity of these findings, regardless of differences in sizes of study sites, could be due to the fact that the major mangrove losses occur in the northern block where there is active rice farming and cutting for poles and timber (Fig. 4). Elsewhere, more annual mangrove gains than losses have been reported, for example in the Zambezi Delta (Shapiro *et al.,* 2015). Slight annual mangrove losses have also been reported in the Mahajamba Bays in Madagascar (Jones *et al.,* 2015), Mida Creek (Alemayehu *et al.,* 2014), and Tudor and Mwache creeks in Kenya (Bosire *et al.,* 2014). The greater mangrove losses observed in the Rufiji Delta might be due to the nature of the delta where, contrary to other parts of Eastern Africa, people have

Table 1. Land cover sizes (ha) for the classification years.

Land cover type\Year	1991	2000	2009	2015
Mangroves	51,941	49,687	46,862	45,519
Agriculture	5,344	8,395	11,172	12,642
Bare land	19,993	18,602	17,930	17,170
Other forest	3,921	2,522	3,548	4,268
Water	13,317	15,310	15,002	14,916

Table 2. Land cover changes for the 4 epochs of the classification years (annual loss/gain).

Change rate/epochs	1991-2000	2000-2009	2009-2015	1991-2015
Loss (ha)	4,468 (496)	4,409 (489)	3,192 (532)	9,089 (378)
Gain (ha)	2,324 (258)	1,584 (176)	1,814 (302)	2,632 (109)
No change (ha)	74,439	73,595	74,697	69,745
Water	13,284	14,926	14,812	13,049

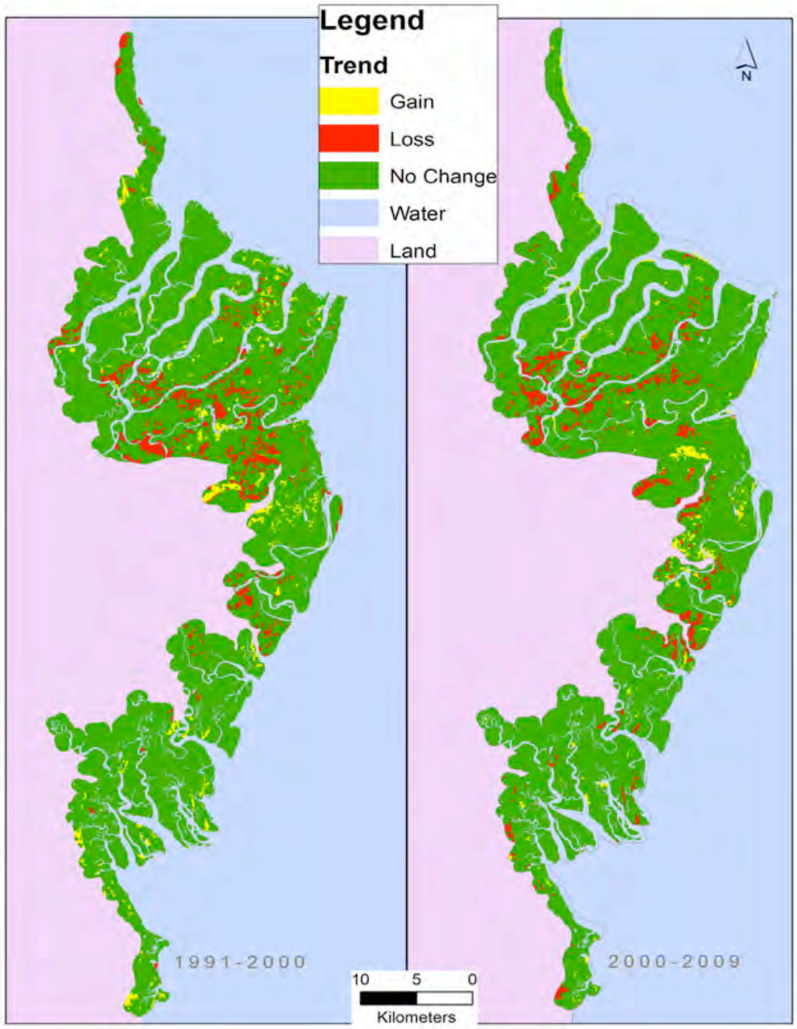

Figure 3a. Mangrove cover change detection (gain, loss and no change) in the Rufiji Delta for the classification periods between 1991-2000 and 2000-2009.

permanent settlements in the delta with sole dependence on mangrove resources for livelihoods, such as rice farming and cutting of mangroves for poles and timber (Mshale *et al.,* 2017). Although the Mangrove Management Plan developed in 1991 clearly categorized mangrove forests into four management zones: Zone I - forests for total protection; Zone II - forests for production; Zone III - degraded areas to be closed to allow recovery; Zone IV - areas to be set aside for different developments (Semesi 1992), there has been weak enforcement of the plan, allowing mangroves to become a tacit common pool resource (Mangora, 2011; Mangora *et al.,* 2016).

Areas surrounding the Rufiji river mouth in the northern block and the northern part of the central block are the areas severely under pressure from rice farming. These areas were initially dominated by *H. littoralis*, and through traditional knowledge, farmers understand these areas have low water salinity and high soil nutrients. Farming activities were also noted to expand to other areas dominated by *C. tagal* in the northern part of the northern block. The farmed areas are also easily accessible by local boats and therefore promote rice farming activities. Peter (2013) linked expansion of rice farming in the delta with rapid human population increase. Statistics show that human population in and around the delta has increased from 38,148 people in 2000 to 49,902 people in 2012. Mwansasu (2016) noted that the rapid expansion of rice farming in the northern block was contributed by the shift of the dominant fresh water-flow in the 1970s. The shift increased population in the northern block, with consequent increase in food demand and therefore increased rice farming activities. Due to poor agronomic knowledge, the farming approach in the delta is of a shifting nature where farms are cultivated in a rotation of 3-5 years before farmers move on to open new farm fields by clearing mangrove forests; and the vicious cycle continues (Fig. 5).

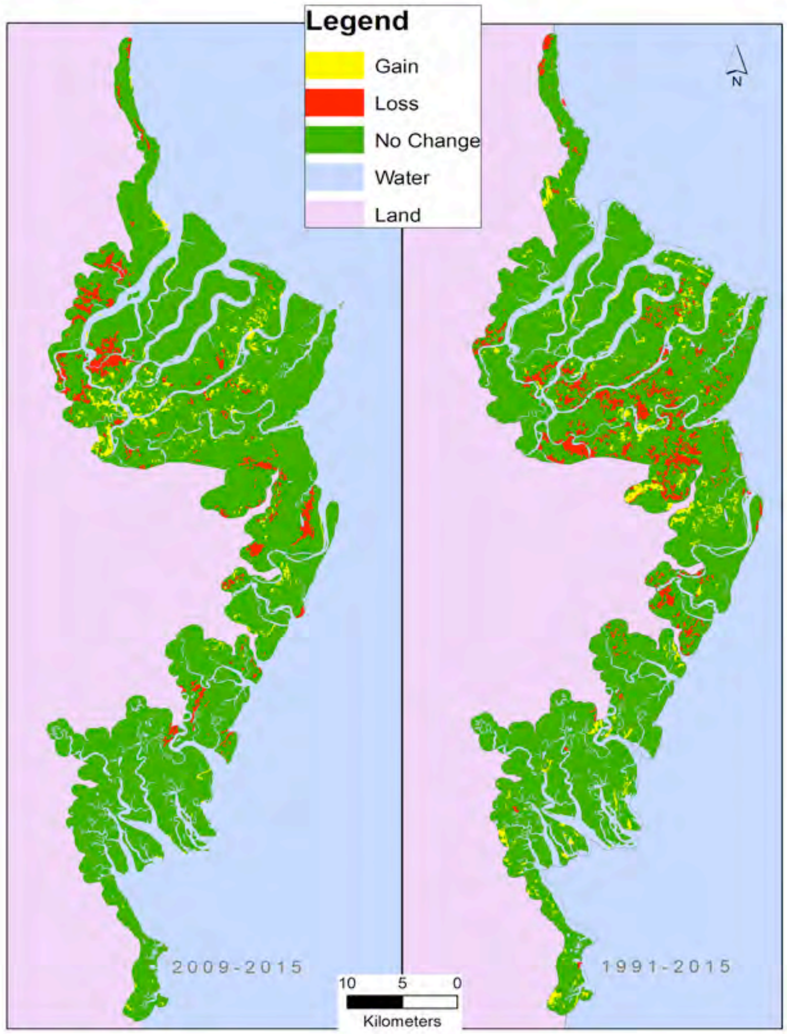

Figure 3b. Mangrove cover change detection (gain, loss and no change) in the Rufiji Delta for the classification periods between 2009-2015 and overall from 1991-2015.

There were greater mangroves gains between 2009 and 2015 than in other years. This is probably due to conservation projects initiated in the delta in late 1990s. Some of the projects were implemented by WWF-Tanzania, TFS Agency and the Rufiji District Authority and aimed at restoring deforested areas in the delta. Like any other large river delta, the Rufiji River Delta is also dynamic, with changing flow paths, shoreline position and development of new lands, which can result in changes to mangrove cover, notably directional changes or shifts along geomorphological patterns (Woodroffe, 1992; Beilfuss *et al.,* 2001). Newly formed islands and mud flats are colonized by *A. marina* and *S. alba*, representing some of the mangrove gain recorded in this study.

Non-mangroves, especially coconut trees, do not perform so well in the delta. The under-performance of coconut trees discourages farmers from planting more trees, therefore areas covered by non-mangroves remain relatively unchanged over the years. Decrease of bare land was mainly contributed by the collapse of salt making processes in the delta. Salt making collapsed in the delta because of the loss of a market for locally made salt from traditional salt pans that is not iodized. Some abandoned salt making areas have been observed to be colonized by mangroves, especially *A. marina* (Fig. 6). However the regenerated *A. marina* appears to be stunted, probably due to high soil salinity in these areas.

Conclusion

The present data analysis serves as an updated baseline on the mangrove cover and land use in the Rufiji Delta, where management planning should be a conservation priority. This retrospective analysis of mangrove cover change demonstrates that freely available Landsat images are an important source of data for land use/cover change studies, especially in developing countries where resources to purchase high resolution imagery data are limited. The loss of 12.4% of

Figure 4. Illegal mangrove pole cutting in the Rufiji Delta. Poles are cut for construction purposes by local communities and for business in the near towns.
Figure 5. Re-opened rice farm in the mangrove forest in the Delta. The area was left as fallow for about 3-5 years before being cleared again. The photo was taken in January 2016.
Figure 6. Mud flat in the Rufiji Delta. Some mud flat areas are colonized by stunted *Avicennia marina.* The photo was taken in 2018.

mangroves in the period of 24 years is alarming and the consequences are not well appreciated. This calls for more strategic collaboration between stakeholders to address the main drivers of mangrove loss in the delta. To address the challenge of mangrove loss, especially conversion of mangroves into rice farming, TFS should take the initiative of establishing multi-stakeholder platforms to regularly discuss opportunities and threats to the delta, and agree on the best way that they can work together to address the challenges for longer term benefits. There is also a need to establish a special management arrangement for the Rufiji Delta, which integrates various actors and communities. This management arrangement should facilitate agreement between TFS and communities where roles and responsibilities of communities in the delta are well clarified and managed. This will reduce the long-term ongoing friction between TFS and communities, which promotes illegal mangrove practices in the delta. Further research, especially on mangrove cover projection and analysis of species composition change for management planning, is relevant.

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Growth rates of *Eucheuma denticulatum* and *Kappaphycus alvarezii* (Rhodophyta; Gigartinales) cultured using modified off-bottom and floating raft techniques on the Kenyan coast

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Abstract

The study compared relative growth rates (RGR) of *Eucheuma denticulatum* and *Kappaphycus alvarezii* under modified off-bottom (MB) and floating raft (FR) culture techniques on the southern coast of Kenya. Seasonal variability in RGRs was evident over 10 months in both techniques and sites. RGRs were in the range of 0.9 – 10.2 % d^{-1} for *E. denticulatim* and 0.3 – 5.7 % d^{-1} for *K. alvarezii* at Mkwiro, and -0.2 – 7.3 % d^{-1} for *E. denticulatum* and -1.7 – 4 % d^{-1} for *K. alvarezii* at Kibuyuni. The RGR of 4.3 ± 0.4 % d^{-1} from the FR technique was significantly higher than the 3.2 ± 0.4 % d^{-1} from the MB technique ($P < 0.05$). Strong oceanic waves which were accompanied by the loss of seaweed thalli in the MB technique at Mkwiro between April and June led to significantly higher RGR in the FR than in the MB ($P < 0.05$). The higher percentage herbivory, epiphytes, and susceptibility to 'ice-ice' (white and soft thallus) associated with *K. alvarezii* than *E. denticulatum* in both techniques led to the former having significantly lower RGRs than the latter. Further research on the stability of MB in deep water and accessibility of FR techniques is recommended prior to commercial adoption.

Keywords: *Eucheuma denticulatum*, culture techniques, *Kappaphycus alvarezii*, relative growth rate, Mkwiro, Kibuyuni

Introduction

Seaweed biomass is a source of direct human food and phycocolloids which are extensively used in the processing of food additives and nutraceuticals, feeds, fertilizers, biofuels, cosmetics and medicines, among others (Bixler and Porse, 2011). They also act as anti-oxidants and antimicrobial agents (Gupta and Abu-Ghannam, 2011). In particular, the kappa carrageenan (phycocollide) which is predominantly extracted from *K. alvarezii* (Doty) Doty ex Silva, and iota-carrageenan from *E. denticulatum* (Collins and Hervey) are used in stabilizing food products, such as frozen desserts, chocolate milk, jellies, and pet foods (Anis *et al.*, 2017). Over the years, the industrial use of seaweed biomass has shifted from exploiting beach-cast seaweeds as fertilizers and a source of potash, to iodine production, and to hydrocolloid extraction (Synytsya *et al.*, 2015). The seaweed industry has demonstrated persistent relevance to current needs and has a bright future when compared to the past when the industry looked very different (Hafting *et al.*, 2015).

Sources of these seaweeds have been from wild harvesting or artificial cultivation (FAO, 2012). Due to increasing demand in seaweed-based products associated with carrageenan, commercial mariculture of *K. alvarezii* and *E. denticulatum* has been intensified over the last decade in many countries with tropical coastlines (Buschmann *et al.*, 2017). Conventional culture techniques such as floating rafts (FR), long lines, fixed off-bottom lines (FOB), fixed long lines (Hurtado *et al.,* 2008), swing hanging long lines, and single or multiple-raft long lines (Hurtado and Agbayani, 2002) have been used. The culture of eucheumoids using these techniques has spread from Asian countries such as the Philippines and Vietnam to those in the western

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Indian Ocean (WIO) such as Mozambique, Madagascar (Mollion and Braud, 1993), Tanzania (Msuya, 2007) and Kenya (Wakibia *et al*., 2006). Seaweed farming has been widely accepted as an alternative source of livelihood in these countries (Msuya, 2007).

While developing countries continue to use the popular fixed off-bottom techniques in shallow areas of the near shore environment (FAO 2015), developed and developing countries have ventured into deep water cultivation techniques to maximize production of seaweeds. This approach has been triggered by poor production of seaweeds cultivated in shallow water environments associated with perennial infestation by 'ice-ice' syndrome (white and soft thallus), epiphytes, and herbivory. These challenges have been mostly observed in seasons characterized by high water temperature (Msuya *et al*., 2014). As a result high disparity in production between regions due to diversification of culture techniques has been demonstrated (Buschmann *et al*., 2017). For instance, in 2014, Indonesia recorded more than ten million tons of carrageenophytes, while in the WIO region, Zanzibar had the highest production of a mere one hundred thousand tons (Buschmann *et al*., 2017). This scenario suggests that the farming techniques adopted has a direct impact on seaweed production. The deep water culture techniques require sophisticated materials such as motorised boats to access the farms in relatively deeper, stable and productive environments, as opposed to shallow water cultivation where farmers with low economic power access farms through walking at low tides. Although production statistics from other seaweed cultivating countries in Africa are scarce, limited production of carrageenophytes in the WIO region exists (Msuya *et al*., 2014). The latest report has revealed that even with the diversification of culture techniques in developed countries, the present sources of cultivated eucheumoids have not met the volume demands of the existing seaweed processing industry (Buschmann *et al*., 2017).

Recent studies have shown that relocation of seaweed farms from shallow water to deeper water environments reduced the risk of 'ice-ice' syndrome on these seaweeds thus improving their growth rate and biomass (Msuya *et al*., 2014). However, the studies have not focused on identifying the best culture technique for adoption in deep water environments. This aspect forms part of the background of this study. Recognizing the fact that seaweed biomass production could also be affected by other factors such as the wind patterns (Hurtado *et al*., 2001) and water quality (Msuya *et al*., 2014), the present study determined the growth rates of *E. denticulatum* and *K. alvarezii* cultured using two deep water culture techniques (FR and MB) on the southern coast of Kenya. The impact of these factors on farmed seaweeds was also investigated. The results from this study are critical in enhancing seaweed production strategies with a focus on formulating a sustainable national seaweed policy for Kenya to contribute to greater economic growth.

Figure 1. Map of the Kenyan coast showing the two study locations on the southern coast.

Materials and methods

Study sites

The study was conducted for ten months in the intertidal areas near Kibuyuni (4.38S, 39.20E) and Mkwiro villages (4.40S, 39.23E) on the southern coast of Kenya (Fig. 1). The two sites were sheltered from strong wave action and tidal currents by a fringing reef. The sandy substratum was colonized by seagrass species, *Thalosodendron ciliatum, Thalassia hemprichii*, *Syringodium isoetifolium*, and seaweeds, Glacilaria spp, *Turbinaria* spp and *Sargassum spp*. Echinoderms such as *Tripneustus gratila* and *Echinometra mathei* foraged within the seagrass beds and patchy coral heads. The two sites had been identified as suitable seaweed faming sites (Wakibia *et al.*, 2006).

Pilot scale commercial farming of *E. denticulatum* and *K. alvarezii* existed in these areas which were promoted by the World Bank–funded Kenya Coastal Development Project (KCDP). The climate on the southern coast of Kenya is influenced by northeast monsoon (NEM) winds which blow from the northeast between December and March (kaskazi) and southeast monsoon (SEM) winds blowing from the southeast between May and October (kusi). The SEM is characterized by strong winds that are accompanied by low air and water temperatures and solar radiation, with the lowest tides being measured at night. Conversely, the lowest tides of the NEM are measured during the day, the winds are relatively weaker, air and water temperatures are higher and rainfall is generally low. There is a one to two month transition period between the NEM and SEM characterized by variable and weaker winds. The seasons significantly affect the chemical and physical conditions of coastal waters (McClanahan, 1988).

Materials, cultivation techniques and growth experiments

Being a quantitative study, a completely randomized design (CRD) was used to assign stocked ropes for each of the two culture techniques at each site. Thirteen polypropylene ropes were used in each of the culture techniques; five ropes for each seaweed species, and three ropes as controls (without seaweed cutting). Fresh, young and clean seedlings of *E. denticulatum* and *K. alvarezii* were obtained from pre-existing seaweed farms at Kibuyuni and Mkwiro. To compare growth rates of the two eucheumoids, the seaweeds were cultured at these locations using FR and MB techniques, as illustrated in Figures 2 - 6. The MB technique (Figure 4) was a hybrid of fixed off-bottom (Figure 7) and FR techniques (Figure 8). At each site the FR and the MB techniques were deployed at 5m and 1m water depths respectively, at low tide. The FR site was accessed by use of a motorised boat while the MB site was accessed by foot. The distance between the FR and the MB at Kibuyuni was about 30 m, and 5 m at Mkwiro. The FR and MB techniques used in the present study were modified from those described by Lirasan and Twide (1993). The FR technique (Figure 2) consisted of a floating bamboo raft (4 x 5 m^2) anchored to the bottom of a deep lagoon (5 m at low tide) by a polypropylene rope (10 mm diameter) and a 100 kg weight. Similar ropes (6 mm diameter and 5 m long) were stocked with 25 seaweed cuttings of 50-80 g seed densities. Each seaweed cutting was attached to the rope using 'tie-tie' (soft, thin, tying material). Five ropes representing replicates of each species were used to test the daily growth performance of *E. denticulatum* and *K. alvarezii* at Kibuyuni and Mkwiro. Each of the stocked ropes was weighed and then stretched randomly along the length of the bamboo raft at 15-20 cm intervals. The same design was repeated in the MB technique at each site.

The MB technique (Figure 4) constituted mangrove poles of 6 cm diameter and 7 m length driven into the seabed and held upright at 6 m above ground. Similar polypropylene ropes as those in the FR technique also stocked with 25 seaweed cuttings were used in the MB technique. However, the seaweed cuttings in the MB technique were suspended by a floating mechanism that was designed to ensure that the seaweed cuttings remained immersed and close to the surface of the water. The design consisted of a two metal rings locked around the length of each pole at 20 cm above the seabed and on the uppermost part of the pole by a small wooden block nailed to the pole. These rings connected the polypropylene ropes from one pole to the other. Three 1 l empty plastic bottles were attached to each of the stocked ropes to enhance buoyancy. Field assistance provided by two young men and two ladies selected from the local communities enabled a complete set up of the experiments within the limited duration of the spring tide. However, due to ease of access at low tide, the MB technique was attended actively by both men and women thus enhancing the speed of the experimental set up. Only men with good swimming ability participated actively in the deep water deployed FR technique.

After a culture period of thirty days, the number of missing seaweed cuttings in each monoline displaying 'ice-ice' syndrome were recorded. The number of

Figure 2. The floating raft (FR) technique deployed in the sea.
Figure 3. One-month seaweeds being harvested from the floating raft at Mkwiro.
Figure 4. The field experimental set up modified off bottom technique.

Figure 5. The layout of modified off-bottom (MB) technique in the sea at high tide.
Figure 6. Monitoring growth of seaweeds cultured under the modified off-bottom technique at low tide.
Figure 7. Fixed off-bottom line seaweed culture technique. Figure 1. Map of the Kenyan coast showing the two study locations on the southern coast.

grazed and macro epiphyte-infested thalli was also visually counted before they were weighed using a spring balance (Satorius Model, Germany). The number of cuttings on the stocked rope which displayed 'ice-ice' syndrome or were grazed or infested with macro epiphytes was computed as a percentage of the original number of cuttings on the entire rope. Young healthy thalli from every harvest were then selected as the stocks for the next growing cycle. These procedures were repeated from September 2015 to June 2016 for the two culture techniques. The mean RGR expressed as percent increase in wet weight (wt) per day was calculated according to the formula by Wakibia *et al*. (2006):

$$RGR = [(w_t/w_o)^{1/t} - 1] \times 100\ \%$$

where, w_o = average wet weight of seaweed cutting at day 0; w_t = average seaweed cutting wet wt at time t and t = time intervals (days).

Plant tissue analysis

After the final growth of seaweeds was measured for each month of culture, approximately 300 g of wet seaweed was harvested from the thalli of the cultivated species. This biomass was then cleaned with seawater, packed in labelled plastic bags, and stored in a cooler box at 4°C before being transported to the Kenya Marine and Fisheries Institute (KMFRI) laboratory for further processing. From each of the 300 g wet seaweed samples a total of 200 g of the seaweed thalli was accurately measured, sun dried for one day and then oven dried at 40°C to a constant weight for the determination of total nitrogen (N) and phosphorus (P) content. Total N and P content was determined using the National Council for Air and Stream Improvement (NCASI) method TNTP-W10900 (NCASI, 2000). This method converted ammonia, organic and inorganic (excluding N_2) nitrogen to nitrate, and organic and inorganic forms of phosphorus to orthophosphate by means of an alkaline acid digestion, without affecting the native nitrate. The digestion was accomplished by heating acidified unfiltered samples in the presence of a persulfate (strong oxidizer) at 120°C and 15 psi for 1.5 hours using an autoclave. Following digestion, the samples were cooled and filtered using GFC (45 nm) micro filters. Using the same dilution factor, the samples were then diluted and their aliquots were calorimetrically analyzed using a QUAATRO autoanalyzer (UK).

Environmental parameters

Air and water temperature, salinity, water motion (diffusion factor), water nitrate and phosphates were determined at both cultivation sites. Water and air temperatures were recorded every two days at midday and the data was used to compute monthly temperature. A maximum/minimum thermometer (TFA model, Germany) was also deployed at the sites during each culture cycle. Seawater salinity was measured at each site fortnightly using a refractometer (Atago model, Japan). Water motion (diffusion factor) was also measured fortnightly by the rate of dissolution of clod cards made from Plaster of Paris (POP) using plastic balls as molds according to a modified method of Doty (1971). Three replicates of POP balls were left in the water column at sites of each culture technique and retrieved after 24 hours. Upon retrieval, the remaining POP balls were rinsed with fresh water, dried in the oven at 60°C, and weighed to a constant weight. To determine the diffusion factor, the average final weight (dry wt of POP balls in the field) was compared with the average final dry weight of POP balls suspended in a bucket of motionless seawater of equal salinity placed in the laboratory for 24 hours. Water samples for the determination of nitrate and phosphate levels were collected 20 cm below the surface of seawater at each site every fortnight using five 125 ml high density polyethylene bottles. The samples were fixed immediately with mercuric chloride, labeled, and stored in a cooler box at 4°C before being transported to the laboratory for analysis using the Technicon Auto Analyzer II system as described by Parsons *et al*., (1984).

The macro epiphytes were identified in the field using a field guide to seaweeds and seagrasses (Oliveira *et al*., 2005), while the herbivorous fish were monitored by swimming and were visually identified using the field identification guide to the living marine resources of Kenya (Anam and Mostarda, 2012).

Data analysis

All data were analyzed using Microsoft Excel and Minitab 17 Statistical Software (2010) for tabular and graphical presentations. Significant differences in relative growth rates between species, sites and culture techniques were determined using t-tests. The Pearson's product-moment correlation test was used to determine the relationships between relative growth rates of cultured *E. denticulatum*, *K. alvarezii* for the two culture techniques, and environmental factors ($P < 0.05$).

Results

Seaweed and environmental variables

Statistical analysis of the seaweed and environmental parameters measured between sites, species, and culture techniques on the southern coast of Kenya are presented in Tables 1, 2, and 3. The correlation coefficients of the variables with relative growth rates of *E. denticulatum* and *K. alvarezii* are presented in Table 4.

The minimum water temperature ranged from 27 to 30°C while maximum ranged from 29 to 33 °C. Air temperature had a range of 26 to 29.5°C and a mean of 27.3± 0.1°C. The minimum and maximum water temperatures varied greatly between months, with higher values being observed in January and February, and lowest values recorded in June. According to Table 1, there was no significant difference in minimum and maximum water temperatures between sites (P < 0.05). During the study period the diffusion factor (water motion) was in the range of 1.5 - 5.4 with a mean of 3.6 ± 0.9 on the southern coast of Kenya. The means were highly varied over the ten months with the highest (4.2 ± 0.5) being recorded in March, and lowest in January (3.2 ± 0.3) and February (3.2 ± 0.4). On average water motion of 3.63 ± 0.1 was oberved at Mkwiro and 3.51 ± 0.2 at Kibuyuni, 4.04 ± 0.14 and 4.03 ± 0.14 in the FR and MB technique, respectively. Statistical analysis of the mentioned variables in Tables 1 and 3 did not show any significantl difference between sites or culture techniques (P < 0.05). A salinity range of 35 - 35.4 ‰ was observed during the study with an average of 35.4 ± 0.1 ‰ at Kibuyuni and 35.2 ± 0.0 ‰ at Mkwiro.

Table 1. Seweed and environmental factors measured in seaweeds cultured at Kibuyuni and Mkwiro on the southern coast of Kenya (Mean ± SE, N = 141-285)

Variable	Kibuyuni	Mkwiro	P-value
Thallus N (%)	1.43 ± 0.01	1.44 ± 0.01	0.564
Thallus P (%)	0.04 ± 0.01	0.04 ± 0.01	0.752
‘Ice-ice’ syndrome (%)	19.2 ± 1.6	7.9 ± 1.5	0.000
Herbivory (%)	22.2 ± 1.8	12.1 ± 1.1	0.000
Epiphytic load (%)	24.5 ± 2.5	15.2 ± 1.1	0.001
Diffusion factor	3.51 ± 0.2	3.63 ± 0.1	0.549
Salinity (‰)	35.4 ± 0.1	35.2 ± 0.0	0.074
Minimum water temperature (°C)	27.8 ± 0.3	28.0 ± 0.2	0.611
Maximum water temperature (°C)	30.5 ± 0.4	30.2 ± 0.3	0.584
Air temperature (°C)	26.8 ± 0.3	27.8 ± 0.3	0.040
Nitrates (µmoles $^{-}$l)	1.31 ± 0.2	1.2 ± 0.1	0.722
Phosphates (µmoles $^{-}$l)	0.746 ± 0.1	0.624 ± 0.1	0.518
Plant loss (%)	15.3 ± 1.1	11.7 ± 1.4	0.040

Means are significant at P < 0.05

Table 2. Thallus N (%) and P (%) and biotic factors affecting the growth of *E. denticulatum* and *K. alvarezii* on the southern coast of Kenya (Mean ± SE, N = 141-285)

Variable	*E. denticulatum*	*K. alvarezii*	P-value
Thallus N (%)	1.44 ± 0.01	1.46 ± 0.01	0.340
Thallus P (%)	0.04 ± 0.01	0.04 ± 0.01	0.918
‘Ice-ice’ syndrome (%)	2.2 ± 0.1	9.2 ± 1.4	0.001
Herbivory (%)	6.2 ± 0.9	11.4 ± 0.9	0.001
Epiphytic load (%)	4.6 ± 0.1	10.9 ± 1.0	0.001

Means are significant at P < 0.05

The average thalli N was 1.44 ± 0.02 % and ranged between 1.35 - 1.55 %, with the highest value being obtained in October and the lowest in April. Thalli P had an average of 0.035 ± 0.002 % and ranged between 0.031 - 0.045 % with the highest and lowest being obtained in February and November, respectively. Concentrations of water nitrates and phosphates ranged from 0.034 to 5.35 µmoles⁻l and 0.004 to 1.896 µmoles ⁻l, respectively, with averages of 1.252 ± 0.1 µmoles⁻l and 0.685 ± 0.06 µmoles⁻l, respectively. There were no significant differences in thalli N (%) and P (%), nitrates (µmoles ⁻l) and phosphates (µmoles ⁻l) between sites (Table 1), and no significant difference in thalli N (%) and P (%), and nitrates (µmoles ⁻l) between species (Table 2) and between culture techniques (Table 3).

The percentage 'ice-ice' syndrome, herbivory and epiphytic load were higher at Kibuyuni than at Mkwiro (Table 1), higher in *K. alvarezii* than in *E. denticulatum* (Table 2), and higher in the FR than in the MB culture technique (Table 3). The highest levels of 'ice-ice' syndrome (%), herbivory (%) and epiphytes (%) in the present study coincided with the period of low RGR of eucheumoids. The epiphytes observed included *Ulva spp.*, *Enteromorpha ramulosa*, *Hypnea musciformis, Padina tetrastromatica, Gracillaria corticata, chaetomorpha indica* and the blue-green alga *Lyngbya majuscula.* These epiphytes mostly occurred in the warmer months as compared to the cooler months.

The main algal grazers observed included the herbivorous fish families Scaridae (Parrotfishes), Siganidae (Rabbitfishes), and the omnivorous Acanthuridae (Surgeonfishes). Grazing damage was mainly found on the thalli of affected seaweeds. Evidence of grazing by the sea-urchin *Tripneustus gratila* was also observed.

Relative growth rates of eucheumoids

The relative growth rates in both culture techniques varied greatly within the 10 months of investigation at the two sites (Figs. 2 and 3). At Mkwiro the two techniques showed a general pattern of high growth rates achieved between October (6.3 % d $^{-1}$) and December (6.0 % d $^{-1}$) before decreasing in February (3.2 % d $^{-1}$). A continuous decrease in relative growth rates appeared in the MB technique for the rest of the period while a sharp increase appeared from February to March in the FR technique. RGRs then decreased continuously to their minimum rates in the later months for both techniques. On the other hand, the growth rates achieved by the two culture techniques at Kibuyuni reveal a similar pattern as that at Mkwiro, but with relatively lower growth rates. The highest RGRs of 5.7 and 3.6 % d $^{-1}$ observed in FR and MB techniques at Mkwiro in October decreased to 1.7 and 1.4 % d $^{-1}$ in the FR and MB techniques in December, respectively. The highest relative growth rates of both species occurred in October and lowest in June at Mkwiro, while the highest at Kibuyuni occurred in September, and lowest in February.

The relative growth rates of both species were higher in the FR technique than in the MB technique at both sites (Fig. 10). The figure also shows that the growth rate of *E. denticulatum* was higher than *K. alvarezii* at both sites. When the growth rates achieved by each species cultured under each techniques were compared, it was established that RGR of *E. denticulatum* cultured using the FR technique was not significantly higher than that achieved when cultured under the MB technique at Mkwiro (P = 0.231), but differed significantly at Kibuyuni (P = 0.039). On the other hand, the RGR achieved by *K. alvarezii* cultured under the

Table 3. Thallus N (%) and P (%) of seaweeds, diffussion factor and biotic factors measured in the Modified off-bottom (MB) and the Floating raft (FR) culture techniques on the southern coast of Kenya (Mean ± SE, N = 141-285).

Variable	Modified off-bottom	Floating raft (FR)	P-value
Thallus N (%)	1.49 ± 0.04	1.42 ± 0.05	0.228
Thallus P (%)	0.04 ± 0.00	0.04 ± 0.00	0.992
Diffusion factor	4.04 ± 0.14	4.03 ± 0.14	0.960
'Ice-ice'syndrome (%)	8.13 ± 2.5	10.4 ± 5.3	0.696
Herbivory (%)	3.13 ± 1.51	3.57 ± 2.00	0.857
Epiphytic load (%)	17.3 ± 1.8	22.6 ± 2.1	0.060
Plant loss (%)	20.3 ± 1.1	11.7 ± 1.4	0.026

Means are significant at P < 0.05

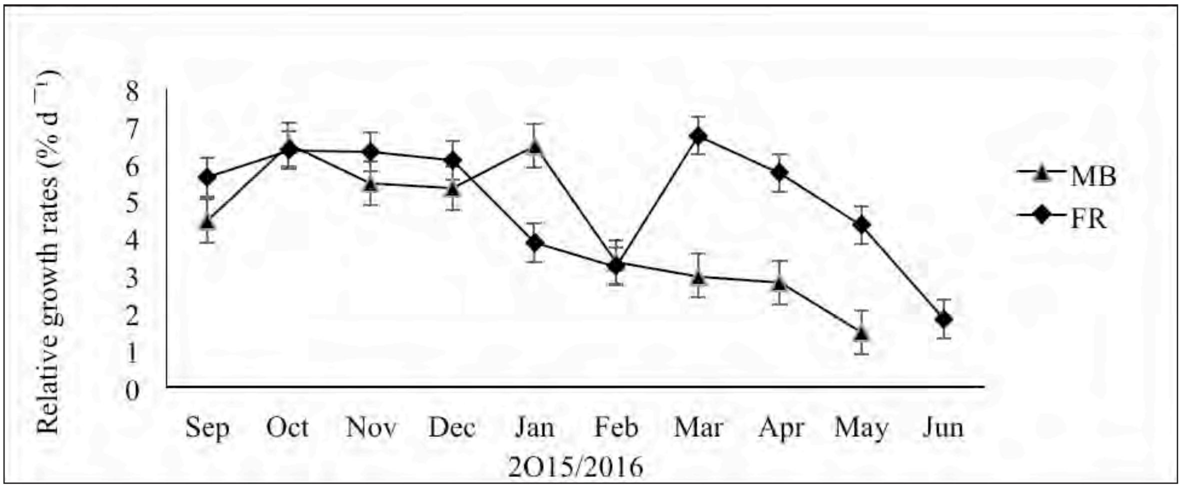

Figure 8. Monthly trends of growth rates achieved by the modified off-bottom (triangle) and floating raft (rhombus) techniques at Mkwiro.

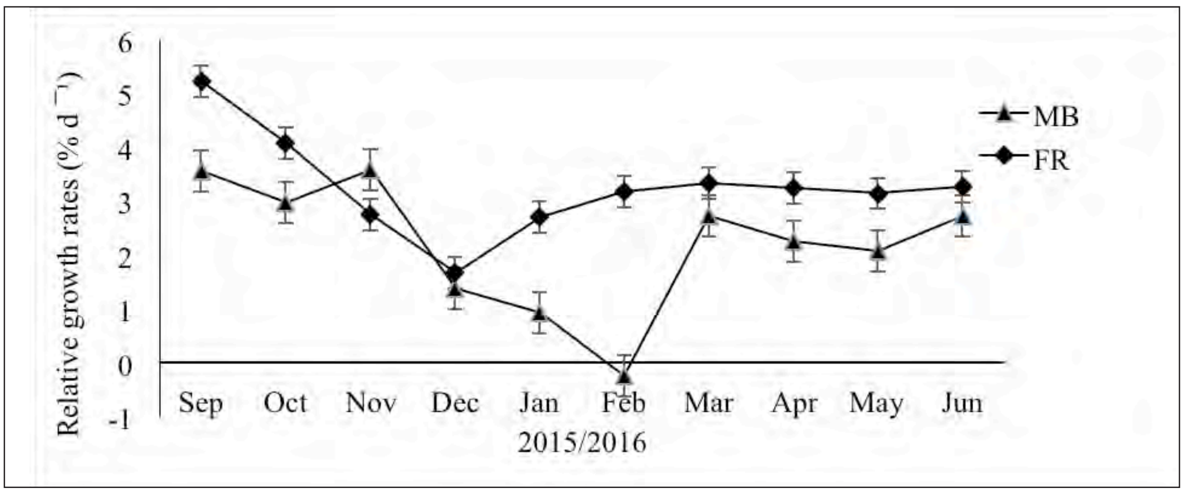

Figure 9. Monthly trends of growth rates achieved by the modified off-bottom (triangle) and floating raft (rhombus) techniques at Kibuyuni.

FR technique was not significantly different from that achieved when cultured under the MB technique at Mkwiro ($P = 0.504$) and Kibuyuni ($P = 0.130$).

Data subjected to the t-test analysis revealed that, regardless of the species cultured and the site selected, the RGR of 4.3 ± 0.4 % d^{-1} achieved by the FR technique was significantly higher than the 3.2 ± 0.4 % d^{-1} achieved by the MB technique ($P < 0.05$). It was also established that irrespective of the culture technique used and the species, the RGR of 4.7 ± 0.2 % d^{-1} at Mkwiro was significantly higher than the 2.7 ± 0.1 % d^{-1} at Kibuyuni ($P < 0.001$). With regard to species, the same test analysis showed that, irrespective of the culture technique and site, *E. denticulatum* performed significantly better than *K. alvarezii* on the south coast of Kenya with RGRs of 4.9 ± 0.2 % d^{-1} and 2.5 ± 0.1 % d^{-1}, respectively ($P < 0.001$).

Correlation coefficients of the RGR of the two seaweed species with environmental and seaweed parameters are presented in Table 4. Several of the correlation coefficients were statistically significant at $P < 0.05$ and at $P < 0.01$.

There was a positive correlation of RGRs of both eucheumoids with diffusion factor and water nitrates, and negative significant correlation with maximum water temperature, % plant loss, % 'ice-ice' syndrome, % epiphytic load and % plant loss. No correlation was found between RGRs of both seaweeds with thallus N (%). However, there was a negative significant correlation of RGR of *K. alvarezii* with % thallus P, and negative significant correlation of the RGR of *E. denticulatum* with salinity. Percentage herbivory showed significant negative correlation only with the RGR of *K. alvarezii*. Apart from the strong positive correlation

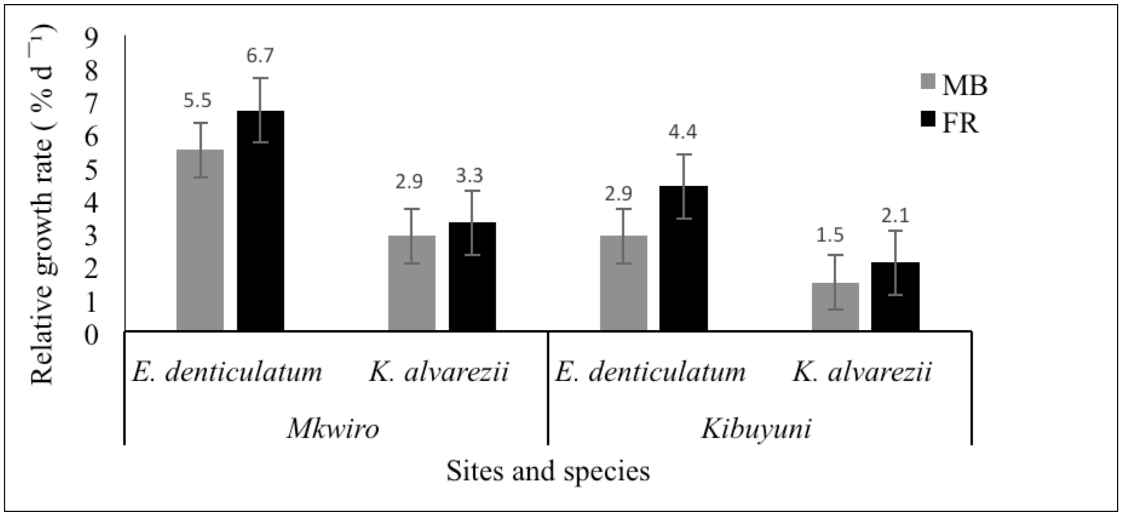

Figure 10. The relative growth rates of eucheumoids achieved under the two culture techniques at Mkwiro and Kibuyuni.

found between the RGR of *E. deticulatum* with the diffusion factor, all the other significant correlations were weak.

Discussion

The monthly trends in seaweed growth rates observed in this study, in which high growth rates occurred in September at Kibuyuni and October at Mkwiro, and the lowest between December and February, are typical of seasonal variations of eucheumoid growth rates observed in previous studies. For instance, in Igang Guimaras, Philipines, the growth rate of *E. denticulatum* was found to increase during the cool months of April and May (Ponce, 1992). Mollion and Braud (1993) found increased growth rate of *E. denticulatum* between April and February before decreasing in March due to grazing and 'ice-ice' attack in Madagascar. Recently in India, at Vizhinjam village, Kerala, high growth of *E. denticulatum* occurred in March and May (Bindu, 2011). Orbita (2013) reported highest growth rate of *K. alvarezii* during the cold period which occurred between June and September, and lowest during the

Table 4. Correlation coefficients of eucheumoids relative growth rates and environmental parameters on the southern coast of Kenya.

	RGR of *Eucheuma denticulatum*		RGR of *Kappaphycus alvarezii*	
Variable	**R**	**P-value**	**R**	**P-value**
Diffusion factor	0.597	0.001	0.267	0.012
Minimum temperature (^{0}C)	-0.127	0.231	-0.040	0.705
Maximum temperature (^{0}C)	-0.405	0.001	-0.208	0.047
Salinity (‰)	-0.368	0.001	-0.091	0.390
Herbivory (%)	-0.143	0.177	-0.317	0.002
'Ice-ice' syndrome (%)	-0.258	0.014	-0.645	0.001
Epiphytic load (%)	-0.239	0.023	-0.525	0.001
Plant loss (%)	-0.232	0.027	-0.393	0.001
Nitrate (µM)	0.424	0.001	0.315	0.002
Phosphate (µM)	-0.249	0.017	-0.101	0.338
Thalli N (%)	0.147	0.164	0.179	0.089
Thalli P (%)	0.028	0.793	-0.366	0.001

R is significant at $P < 0.05$

warmest period of the year between October and May (Ohno *et al*., 1996). A 63 % decrease in growth rate of *K. alvarezii* reported in Vietnam also coincided with the warmest month (August) of the year (Ohno *et al*., 1996; Saleh *et al*., 2016). High growth rates of *K. alvarezii* was also reported to occur between the southeast wind blowing period (April to December) in Fiji, while low growth rate was observed from January to March (Prakash, 1990). On the southern coast of Kenya, Wakibia *et al*. (2006) observed the highest growth rates of *K. alvarezii* during the cooler months (August and September), a period that usually occurs during SEM, and lowest in the hottest months (January and February), usually occurring during the NEM.

The present results coupled with the above references suggest a relationship between air temperature and seawater quality parameters for suitable seaweed farming. The absorption of high air temperature by the sea during hot periods appears to raise the average temperature in equal margin and consequently compromises the water quality parameters, as manifested by emergence of 'ice-ice' syndrome at this time. Seaweed farmers and investors should be weary of these dynamics to avoid sudden suffering from economic shock and also to enable them to make the necessary mitigation interventions in order to save the seaweed seeds from complete loss during the unconducive seasons.

RGRs at the two sites increased from February to March. RGR at Kibuyuni was relatively stable between March and June, but showed a decreasing trend at Mkwiro in the same period. The decrease at Mkwiro was attributed to massive plant loss caused by interference of the MB culture experiment by strong oceanic waves associated with the SEM. Such a scenario was not observed at Kibuyuni. Despite the big loss of plants from the MB culture technique, the RGRs of both species were significantly higher at Mkwiro than at Kibuyuni, because the loss was compensated by the high growth rates achieved by plants in the FR technique at that period. These results therefore suggested that if the structural set up of the MB technique could be improved to overcome the overwhelming sea waves between March and June, then Mkwiro could be a more suitable site for culture of both species as compared to Kibuyuni.

Previous studies have attributed site differences of eucheumoid growth rates to water quality parameters such as water motion (Wakibia *et al*., 2006). However, the results of the analysis shown in Table 1 indicate that there was no significant difference in diffusion factor between the two sites, but biotic factors including % herbivory, % epiphytes and % 'ice-ice' syndrome were significantly higher at Kibuyuni than at Mkwiro. These factors were therefore presumed to have fundamentally influenced variation in RGRs between Mkwiro and Kibuyuni. The negative effects of biotic factors on seaweed growth rate have been cited in previous studies (Msuya *et al*., 2014; Hurtado *et al*., 2014).

The maximum growth rate of 5.7 % d^{-1} for *K. alvarezii* obtained under the FR culture technique in the present study was comparatively higher than the 5.2 % d^{-1} observed in India (Kavale *et al*., 2016), 4.3 - 5.1 % day^{-1} in Brazil (Hayashi *et al*., 2010), 2 - 8 % d^{-1} in Mexico (Munoz *et al*., 2004), and the 5.0 % d^{-1} reported in Zanzibar (Msuya *et al.*, 2014). However it was lower than the 7 - 9 % d^{-1} reported by Ohno *et al*. (1996) in Vietnam, and the 10.7 % d^{-1} reported by Paula *et al*. (2002), and the 4.4-8.9 % d^{-1} in Zanzibar, Tanzania (Dawes *et al.*, 1994). Most importantly, the growth rate of *E. denticulatum* was above the recommended commercial rate of 3.5 % d^{-1} set by Doty (1987), and below this for *K. alvarezii*. The high variation in RGR between the species could have resulted from the difference in sensitivities to water temperature. The fact that *K. alvarezii* registered encouraging growth only during the cold period while the growth of *E. denticulatum* was relatively stable throughout the year under the two culture techniques could justify this explanation. However, several explanations have been made for these variations including difference in species' capabilities of tolerating wide range of ecological factors (Glenn and Doty, 1992), response to water motion (Wakibia *et al*., 2006), their morphological variability (Doty, 1987), and the underlying cell physiology and cell wall responses to the surrounding environment (Hurtado *et al*., 2014). The production of H_2O_2 by *E. denticulatum* as an oxidative burst was suspected to be part of its chemical defense mechanisms against epiphytic attack (Collen *et al.* (1994). Since this study never investigated the production of H_2O_2 by both eucheumoids, our explanation could only be limited to the analysis conducted.

The negative correlation between epiphytes (%) and RGR of eucheumoids revealed a scenario of exploitative competition for nutrient resources in which eucheumoids were outcompeted. Chaipart and Lewmanomont (2004) and Hanisak (1987) associated growth fluctuation of *Gracilaria fisheri with competitive growth of epiphytes. In the WIO region,* the negative effects of epiphytes on seaweed cultivation have been reported in previous studies in Tanzania (Msuya and Kyewalyanga, 2006) and Kenya (Wakibia *et a*l., 2006). However, according to Fujita (1985), some seaweed species overcome the effect of epiphyte infestation by storing enough nitrogen to allow them to grow at maximal rates for several days.

Although this study observed significantly lower levels of herbivory, epiphytes and 'ice–ice' syndrome in *E. denticulatum* than in *K. alvarezii*, it was not possible to accurately establish the cause of these variations since studies on the defense mechanisms of these species was not part of our investigation. Furthermore, the negative correlations between the RGRs of both species with maximum water temperature, water phosphate, 'ice-ice' syndrome (%), herbivory (%) and epiphytic load (%), were significant, but weak, suggesting that other confounding factors could be contributing to the species difference. Future research should therefore be focused on investigating possible distinguishing physiological and morphological characteristics in *E. denticulatum* and *K. alvarezii* that could be influencing resilience to extreme ecological conditions.

Based on the RGR of 4.5 % d^{-1} and range of 0.3 - 9.2 % d^{-1} observed in the FR, and RGR of 3.2 % d^{-1} and range of -1.7 - 10.7 % d^{-1} observed in the MB techniques, this study concludes that the two culture techniques are suitable for improving the growth rate of *K. alvarezii* and *E. denticulatum* in Kenya, thus improving their biomass. However due to seasonal environmental changes that affect the growth of seaweeds, variation in growth rate may occur between the species and techniques. New seaweed farmers and investors should be well sensitized to the seasonal dynamics of seaweed production in order to maximize production during the conducive seasons and to be ready to accept low production during unfavourable seasons. This could be achieved by developing a sense of optimism in the enterprise and exercising sobriety and patience during the various stages of production.

The advantage of higher impact of water velocity accorded to the seaweeds under the FR and the MB techniques coupled with lower grazing pressure and lower incidence of 'ice-ice syndrome explains the suitability of these culture techniques in improving the biomass of cultured seaweeds. However, mechanisms that ensure stability of the MB technique and accessibility of the FR technique by individuals with limited swimming capacities should be considered. This approach would address the plight of non-swimmers, particularly women, who constitute the majority of seaweed farmers in Kenya. To determine the economic viability, a comprehensive economic analysis of the techniques should be conducted prior to commercial up-scaling. Further studies should be conducted to: 1) investigate possible distinguishing physiological and morphological characteristics in *E. denticulatum* and *K. alvarezii* that could be influencing resilience to extreme ecological conditions; and 2) monitor the trends in herbivory attraction by the different techniques.

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The influence of physical-chemical variables on the spatial and seasonal variation of Chlorophyll-*a* in coastal waters of Unguja, Zanzibar, Tanzania

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Abstract

Chlorophyll-a (Chl-a) concentrations were measured at four sites around Unguja, Zanzibar during the northeast (NE) and southeast (SE) monsoon seasons. Data for Chl-*a*, nitrate, phosphate, ammonia, dissolved oxygen, sea surface temperature, pH and salinity were collected once a month from March 2008 to March 2009. The SE monsoon had insignificantly higher Chl-*a* compared to the NE monsoon season when Chl-*a* for Bawe, Chumbe, Pongwe and Mnemba were combined ($W = 234$, $p = 0.93$). The drivers of high Chl-*a* during the SE monsoon were ammonia and nitrate. Results from individual sites showed that Pongwe and Mnemba had higher median Chl-*a* during the SE- than the NE monsoon season. Temperature, dissolved oxygen and ammonia were the major factors that influenced high Chl-*a* at these sites. In contrast, Chumbe and Bawe had higher median Chl-*a* during the NE- than the SE monsoon season. The major factors influencing high Chl-*a* in the NE at Chumbe and Bawe were high levels of nutrients, mainly from sewage effluent and various human activities around the coast in Zanzibar town. The interaction of Chl-*a* between monsoon seasons (NE and SE) and sites (Bawe, Chumbe, Pongwe and Mnemba) was insignificant ($F_{(1,3)} = 1.3144$, $p = 0.2949$). The principal component analysis revealed that different physical and chemical environmental variables affect Chl-*a* concentration over time and location.

Keywords: Chlorophyll-*a*, monsoon seasons, physical-chemical variables, Zanzibar

Introduction

Phytoplankton are microscopic single-celled or colony-forming organisms dwelling in the water column. They are free-floating organisms that depend entirely on water movement such as surface currents for their movement (Bryceson, 1977). They thrive in the euphotic zone; the upper water column of the world's oceans, lakes and rivers (Letelier *et al.*, 2004). The euphotic zone receives energy from the sun that allows phytoplankton to photosynthesize; a chemical reaction that converts light energy into chemical energy (Letelier *et al.*, 2004).

Phytoplankton play an important role in ecological processes, which influence the structure and function of food webs, nutrient cycling and the flux of particles to deeper waters (Sá *et al.*, 2013). The distribution of phytoplankton in the euphotic zone varies both from the coastal to offshore areas (horizontal distribution), and from the surface to deeper waters (vertical distribution) (Barlow *et al.*, 2007; Leal *et al.*, 2009; Sá *et al.*, 2013). There are several factors that govern these distributions including temperature, nutrients, irradiance, water column stability, internal waves, grazing, salinity and ocean currents (Barlow *et al.*, 2007; Brunet and Lizon, 2003; Sá *et al.*, 2013). These environmental factors are not homogenous, and hence the way they influence phytoplankton distribution differs from one place to the other, and from tropical to temperate ecosystems (Sá *et al.*, 2013).

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Similar to terrestrial plants, phytoplankton contain Chl-*a*, an autotrophic component responsible for primary food production in aquatic systems (Limbu and Kyewalyanga, 2015). Boyce *et al.* (2010) affirmed that phytoplankton accounts for half of the production of organic matter on earth, hence serving as the primary source of food in oceans, rivers, seas and freshwater basins (Limbu and Kyewalyanga, 2015). The ecological role of Chl-*a* to convert light energy into food during photosynthesis attracted oceanographers to use it as an indicator of phytoplankton production and biomass (Baliarsingh *et al.,* 2015; Boyce *et al.,* 2010; Peter, 2013).

Some previous studies explored phytoplankton dynamics and distribution in the coastal waters of Tanzania. For example, Lugomela *et al.* (2002) studied the seasonal distribution of *Trichodesmium* species and found that *Trichodesmium* species follow a seasonal pattern with higher biomass during the NE monsoon compared to SE monsoon season. They also found that nitrate was higher in the NE than the SE monsoon seasons. Hamis and Mamboya (2014) explored the spatial and temporal variation of physico-chemical variables and phytoplankton at Ocean Road in Dar es Salaam, exposed to sewage discharge. They concluded that the sewer pipeline that drains the Dar es Salaam city center is the main cause of high nutrient levels in the area, which leads to higher phytoplankton biomass. Other studies like Barlow *et al.* (2011) investigated phytoplankton production and physiology in Unguja and Pemba Island, Zanzibar. They found that the phytoplankton communities are adaptable to changing environmental conditions. The availability of nitrates and phosphates were claimed as the primary factors that control the distribution of phytoplankton communities in Unguja and Pemba.

Most of these previous studies focused either on the distribution, diversity, and abundance of phytoplankton. Some looked at identification of phytoplankton species in the coastal waters of Tanzania (Lugomela, 1996; Bryceson, 1977). Other studies such as Hamis and Mamboya (2014) and Barlow *et al.* (2011) examined how sewage discharge and nutrient enrichment in coastal waters affect phytoplankton biomass. However, knowledge on how Chl-*a* concentration varies with monsoon season is poor. This study intended to fill this knowledge gap by exploring how Chl-*a* concentration varies with season (temporal) and space (spatial) along the coastal waters of Unguja Island, in relation to environmental factors.

Materials and methods

Study areas

This study was conducted in the coastal waters around Unguja Island, Zanzibar, Tanzania. The area is located between latitude 6.6 °S and 5.6 °S and longitude 39.15°E and 39.60°E (Fig. 1). Four study sites were selected for this study. The sites include Bawe, Chumbe, Mnemba and Pongwe. These study sites were chosen because of human activities that take place in these areas. While Bawe and Chumbe are found on the western side of Unguja Island, Pongwe and Mnemba are on the eastern side of the Island.

Like most western Indian Ocean countries, the coast of Tanzania is influenced by northerly and southerly monsoon winds (Nyandwi, 2013). From May to September the SE monsoon winds dominate and are usually strong and predominantly southerly (blowing from south to north) (Mahongo *et al.,* 2012), and this period is characterized by a mean sea surface temperature of 23°C (Semba *et al.,* 2016).

The NE monsoon winds are weaker and predominantly northerly (blowing from north to south), dominate from November to March, and this period is characterized by a mean sea surface temperature of 30°C (Semba *et al.,* 2016). April and October are the transition period when winds tend to subside (Mahongo and Shaghude, 2014). During this period of transition, there is a reversal in wind direction from the NE to the SE and vice versa (Mahongo *et al.,* 2012).

Furthermore, the island experiences two rainy seasons. The short rainy season is characterized by light rain and occurs around November and December. The heaviest and most prolonged rains occur between March and the end of May, with heavy rains throughout this period (Mahongo, 2015).

Data collection

Biological, physical and chemical variables at the selected study sites were recorded for the period of thirteen months between March 2008 and March 2009. At each study site, 5 l of surface water were collected in triplicate using plastic bottles for Chl-*a* determination. The samples were taken to the Institute of Marine Sciences for laboratory analysis. Water samples were filtered through 0.45μm pore size membrane filters followed by Chl-*a* extraction in 90 percent acetone overnight at 4 degree centigrade. The concentration of Chl-*a* was measured with a SHIMADZU spectrophotometer version UV-1201 following the method by Parsons *et al.* (1984). For nutrients, triplicate water samples were collected at each study site using a water sampler. Similar to Chl-*a*, phosphate, ammonia and nitrate were determined using a SHIMADZU spectrophotometer as documented in Parsons *et al.* (1984). A Hanna handheld instrument was used for in situ measurement of dissolved oxygen, pH, and temperature for each study site. An ATAGO refractometer was used to record *in situ* salinity values at each study site.

package to tidy the dataset structure and ensure each column formed a variable, and each row formed a measurement (Wickham and Henry, 2017). The tidy dataset was then transformed using the *dplyr* package (Wickham *et al.*, 2017). The *bind_ rows()* function from Wickham *et al.* (2017) was used to combine multiple variables (nitrate, phosphate, ammonia, dissolved oxygen, pH, salinity) by row, to form a single long format dataset.

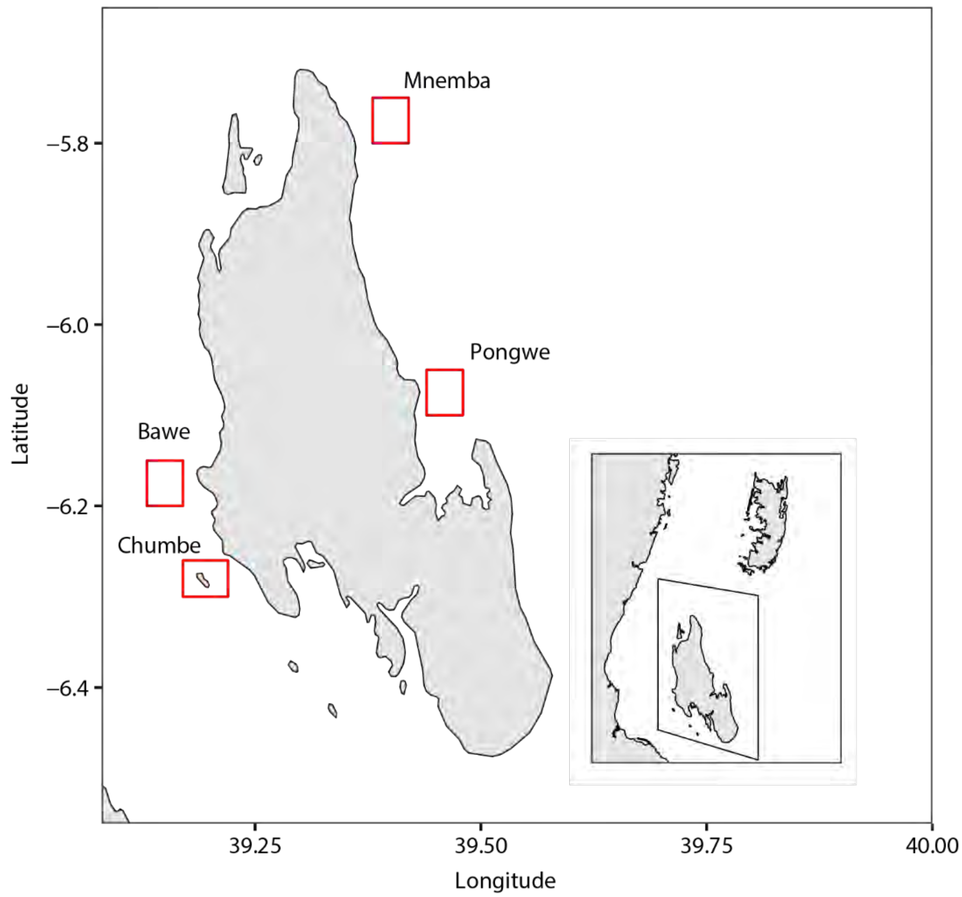

Figure 1. Map of Unguja Island showing the location of the study sites. The insert map indicates the location of Unguja Island in the Indian Ocean.

Data Processing

Data stored in a spreadsheet was imported and loaded into a data frame in R. Once the data was in R the format was ideal for tidying. Tidying is important because it puts the data in the correct format for different R functions. According to Wickham and Henry (2017), tidying data means storing it in a consistent form so that each column forms a variable, and each row forms an observation. Because the dataset was in a structure that limited the R functions, *spread()* and *gather()* functions were used in the *tidy*

Since environmental variables and Chl-*a* datasets were in separate data frames, The *full_join()* function was used to combine them, using key variables that were common to both datasets. The key variables included the station name and the date of sampling. The date of sampling was then separated to form year and month variables using *mutate()* in *lubridate* package (Grolemund and Wickham, 2011). The sampling months were used to determine the seasonality. Months between May and September were grouped into SE monsoon season, and months that fell between

November and March were grouped into NE monsoon season. April and October formed an intermediate season (inter-monsoon). The season variable was separated into *season* and *code* using the *separate()* function from the *tidyr* package (Wickham and Henry, 2017). The variable code was then dropped using the *mutate()* function in the *dplyr* package (Wickham *et al.,* 2017).

Data analysis

The aim of the analysis was to use the data to answer two key questions: the first was to assess if there was a difference in Chl-*a* between seasons (SE and NE) and among sampled sites; the second was to examine the drivers for the Chl-*a* variations. The Wilcoxon test was used to test the difference in median Chl-*a* between the NE and SE monsoon season.

The seasonal mean and standard deviation for each sampled site was computed to determine sites with high and low Chl-*a* concentrations. The computation was chained in *markdown* using the function in the *dplyr* package as described by Wickham *et al.* (2017). The statistics for the inter-monsoon season was dropped and the mean and standard deviation computed for Chl-*a* concentration, and the statistics by site and season were grouped.

The distribution of the data was tested using a histogram. Data that was not normally distributed was then transformed. The type of transformation method applied depended on the skewness of the data itself. According to Wickham *et al.* (2017), logarithmic transformation is used for variables with skewness values between 0.5 and 1, and square root transformation is appropriate for variables with skewness values of greater than 1, and no transformation is required for variables with skewness values below 0.5. Chl-*a* values were transformed for Bawe, Chumbe and Mnemba using square root transformation because their skewness values were greater than 1. The Shapiro test was used for groups as described by Millard (2013), to test for normality of transformed Chl-*a*. Because the samples in the current study were unbalanced (all months were sampled once, except for March, which was sampled twice in both 2008 and 2009), unbalanced two-way ANOVA was used to infer the difference between seasons and among sites. *Anova ()* in the *car* package was used for computation of two-way ANOVA for unbalanced designs (Fox and Weisberg, 2011). Before running the two-way ANOVA function, the Chl-*a* dataset was randomly sampled using the *sample_ n()* function (Wickham *et al.,* 2017).

Principal component analysis (*prcomp()* in R) was used to assess the drivers that influence the Chl-*a* variation. These include dissolved oxygen, pH, salinity, temperature, ammonia, nitrate and phosphate. Because *prcomp()* requires all rows to have values, the data was first cleaned by dropping all rows with NA using the *dplyr* package (Wickham *et al.,* 2017). The *prcomp()* function from R Core Team (2017) was then used to compute the principal component. For seasonality, the data were filtered into seasons in order to make a biplot of separate seasons. The *factoextra* package as described by Kassambara and Mundt (2017) was then used to visualize the drivers and their influence on Chl-*a*.

Results

In respect to seasonality, the southeast monsoon season had a median Chl-*a* concentration of 0.35 mg/m^3 as compared to 0.32 mg/m^3 for northeast monsoon season (Table 1). These results suggest that the SE monsoon has a relatively higher Chl-*a* concentration than the NE monsoon season. However, when the mean Chl-*a* concentration is considered, the results reveal that the NE season has a higher mean Chl-*a* (0.41 mg/m^3) than the SE monsoon season (0.36 mg/m^3). The high value of Chl-*a* during the NE season is contributed by the outliers of the samples collected during this season (Figure 2). However, the difference in the median of Chl-*a* between the two seasons was insignificant (W = 248, p = 0.86)

During the SE monsoon, Pongwe had a relatively higher Chl-*a* concentration (0.378 mg/m^3), followed by Mnemba, Bawe and Chumbe, which had the lowest of 0.251 mg/m^3 (Table 2). However, during the NE monsoon, the Chl-*a* concentration differed, with Bawe having relatively a higher Chl-*a* concentration (0.477

Table 1. Summary statistics of Chlorophyll-*a* with season.

Seasons	Min	Max	Median	Mean	STD
NE	0.094	0.886	0.324	0.413	0.211
SE	0.143	0.644	0.350	0.357	0.115

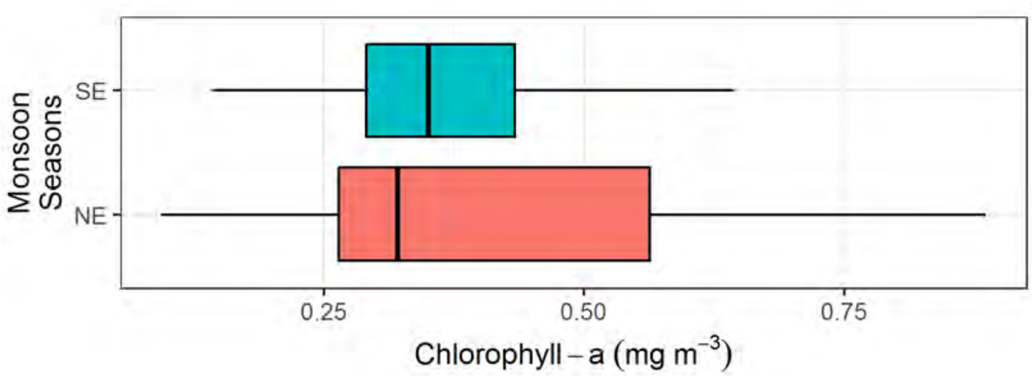

Figure 2. A boxplot showing variations in Chlorophyll-a concentration between monsoon seasons.

mg/m^3) followed by Chumbe, Pongwe and finally Mnemba, which had the lowest Chl-*a* concentration of 0.279 mg/m^3 (Table 3).

The boxplot shows that Mnemba and Pongwe had higher median Chl-*a* concentrations during the SE than the NE monsoon season (Fig. 3, Table 2). In contrast, Chumbe and Bawe had higher median Chl-*a* concentrations during the NE than the SE monsoon season (Fig. 3, Table 3). The results suggest that Chl-*a* concentration varied both in time (season) and space (study sites).

Since the statistic used to test for Chlorophyll-*a* and other chemical and physical variables were required to fit the normal distribution, the nature of the distribution of the data was explored prior to analysis. The results showed that combining the Chl-*a* data for all sites made the data unfit for normal distribution (GroupTest, p = 0.0028). However, observing the distribution at each site showed that Chl-*a* at Pongwe was normally distributed (p = 0.92), and the other study sites were not ($p < 0.05$). Normal distribution for Chl-*a* for Bawe, Chumbe and Mnemba sites was achieved after square root transformation (z = -1.25, p = 0.105).

The ANOVA table shows that the difference in mean Chl-*a* concentration between the Bawe, Chumbe, Mnemba and Pongwe sites was insignificant ($F_{(1,3)}$=1.55, p = 0.23). Similarly, the mean difference in concentration of Chl-*a* between the NE and SE monsoon seasons was not significant ($F_{(1,1)}$ = 1.14, p = 0.29). The interaction of mean Chl-*a* between study sites and seasons was also insignificant ($F_{(1,3)}$=1.31, p = 0.29). The results of the two-way ANOVA indicate that the spatial and temporal effects have little influence on Chl-*a* concentration.

Table 2. Summary statistics of Chlorophyll-a concentration during the southeast monsoon season for the four study sites.

Sites	Seasons	Min	Max	Median	Mean	STD
Pongwe	SE	0.355	0.644	0.378	0.447	0.123
Mnemba	SE	0.143	0.447	0.346	0.339	0.121
Bawe	SE	0.240	0.497	0.341	0.364	0.103
Chumbe	SE	0.222	0.368	0.251	0.277	0.061

Table 3. Summary statistics of Chlorophyll-a concentration during the northeast monsoon season for the four study sites.

Sites	Seasons	Min	Max	Median	Mean	STD
Bawe	NE	0.286	0.758	0.477	0.492	0.183
Chumbe	NE	0.218	0.791	0.349	0.426	0.214
Pongwe	NE	0.094	0.676	0.294	0.359	0.220
Mnemba	NE	0.226	0.886	0.279	0.376	0.252

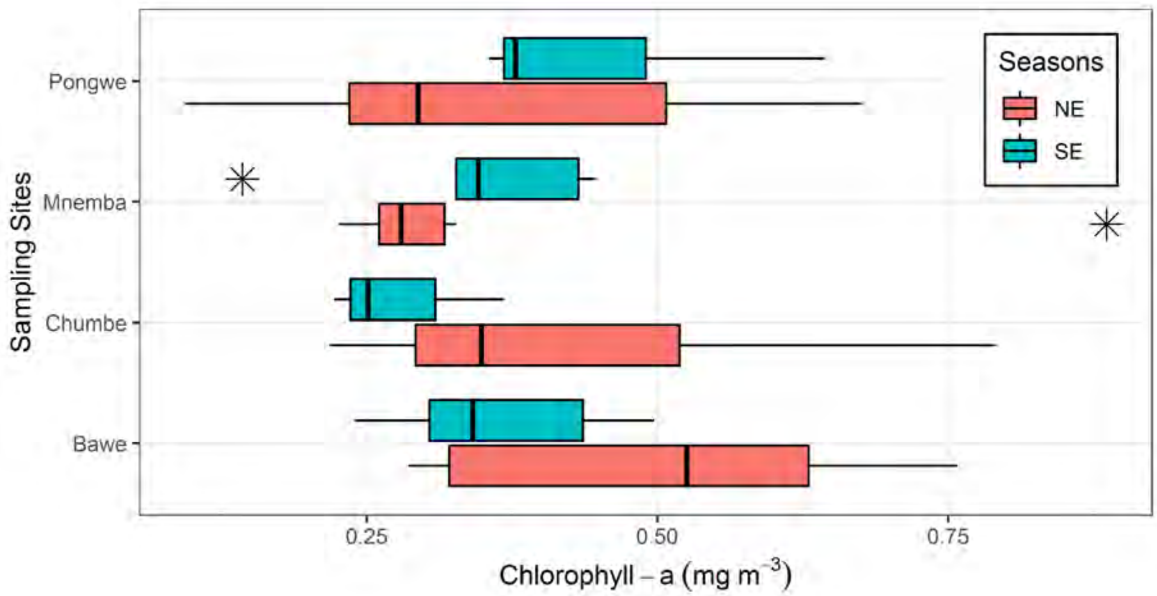

Figure 3. Boxplot showing Chlorophyll-a concentration variation with study sites and seasons.

Drivers of Chlorophyll-*a* variation

When the Principal Component Analysis (PCA) was run for all seasons together, it was found that in dimension 2 (Dim2), ammonia, nitrate, phosphate, Chl-*a*, pH, and temperature displayed negative coefficients, while dissolved oxygen and salinity displayed positive coefficients (Fig. 4). This shows that Chl-*a* had a positive linear relationship with ammonia and nitrate during the SE monsoon season, while phosphate, pH and temperature showed a positive relationship during the NE monsoon season. On the other hand, Chl-*a* had a negative linear relation with dissolved oxygen in the SE monsoon season, and salinity in the NE monsoon season.

However, when the NE monsoon season data was analysed on its own, it was found that in Dim2, ammonia, temperature, dissolved oxygen, pH, phosphate, nitrate and Chl-*a* displayed negative coefficients, whereas salinity displayed a positive coefficient (Fig. 5). The ellipse indicates that chlorophyll had a positive linear relationship with ammonia, nitrate, phosphate, dissolved oxygen, pH, and temperature at Chumbe, Bawe and Mnemba, and a negative linear

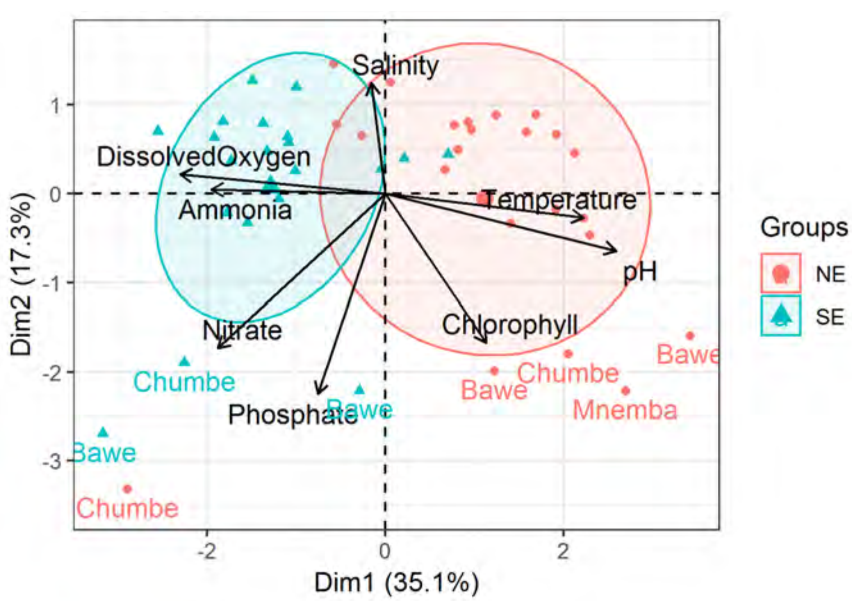

Figure 4. Biplot for combined northeast and southeast monsoon seasons at Bawe, Chumbe, Mnemba and Pongwe.

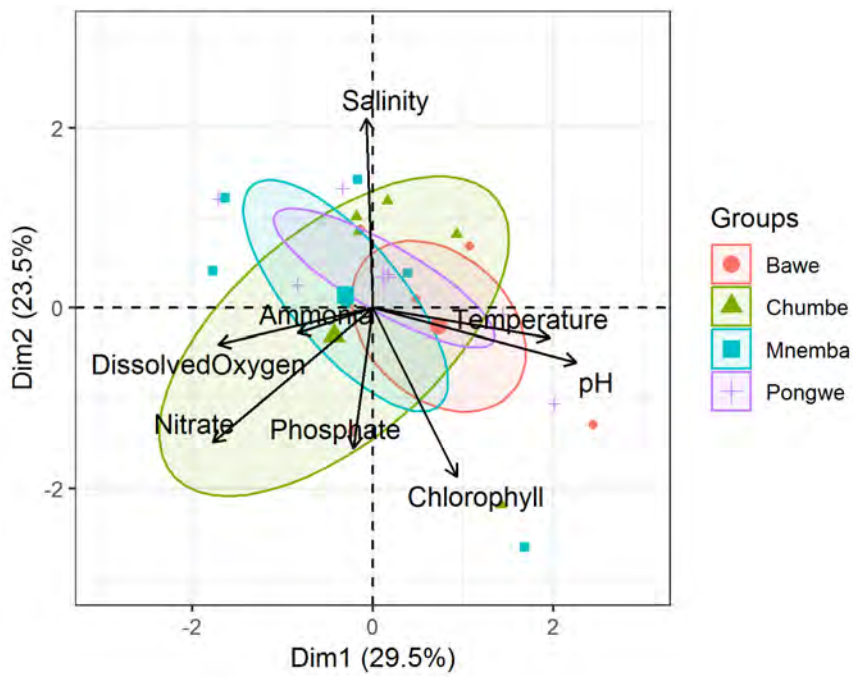

Figure 5. Biplot for northeast monsoon season at Bawe, Chumbe, Mnemba and Pongwe.

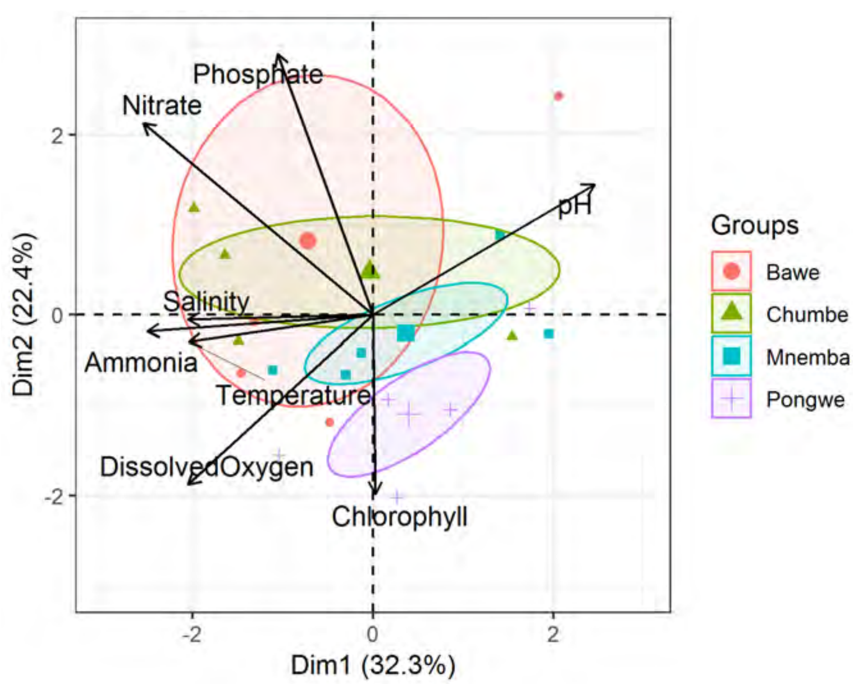

Figure 6. Biplot for southeast monsoon season at Bawe, Chumbe, Mnemba and Pongwe.

relation with salinity at Bawe, Chumbe, Pongwe and Mnemba during the NE monsoon.

Likewise, when data for the SE monsoon season were analyzed, it was found that in Dim2, ammonia, temperature, dissolved oxygen, and salinity, as well as Chl-*a* displayed negative coefficients, whereas nitrate, phosphate and pH displayed positive coefficients (Fig. 6). While ecllipses for Pongwe and Mnemba fall on negative coefficients, Chumbe and Bawe are on positive coefficients in Dim2. The figure indicates that chlorophyll had a positive linear relationship with ammonia, temperature, dissolved oxygen and salinity at Mnemba and Pongwe, and a negative linear relation with nitrate, phosphate, and pH at Bawe and Chumbe during the SE monsoon.

Discussion

Until now, there has been limited information on how Chl-*a* concentration varies in both space and time in the coastal waters of Zanzibar. This study has attempted to fill this gap by assessing Chl-*a* concentration at four selected sites during the SE and NE monsoon seasons. Previous studies have documented that the pattern of Chl-*a* concentration in the coastal water of Tanzania is associated with seasonal changes (Semba *et al.,* 2016; Peter, 2013). McClanahan (1988) reported that the coastal waters have relatively higher Chl-*a* concentrations during the SE monsoon as compared to the NE monsoon season. Similar seasonal patterns for Chl-*a* were found in this study as in Semba *et al.* (2016), Peter (2013), and McClanahan (1988) (Fig. 2). The SE monsoon had a median of 0.350 mg/m^3 Chl-*a* concentration compared to 0.324 mg/m^3 in the NE monsoon season. The SE season had a concentration of about 0.026 mg/m^3 higher than the NE season. However, the difference in median concentration of Chl-*a* between the NE and SE season was not significant ($W = 234$, $p = 0.93$).

The median concentration of Chl-*a* at individual sites showed a seasonal pattern. For example, there was a higher Chl-*a* concentration during the SE as compared to the NE monsoon season at Mnemba and Pongwe (Fig. 3, Table 2). In contrast, Chumbe and Bawe showed the opposite, having water of higher median Chl-*a* concentration during the NE as compared to the SE monsoon season (Fig. 3, Table 3). This difference in Chl-*a* concentration can be attributed to the location on the island. For example, Pongwe and Mnemba are distant from Zanzibar town, where there is little influence from human activities. The findings from Mnemba and Pongwe sites match the seasonal pattern of Chl-*a* found in previous studies (Semba *et al.,* 2016; McClanahan, 1988; Peter, 2013) that the SE season has higher Chl-*a* concentrations than the NE monsoon season; a characteristic widely experienced in the western Indian Ocean. The findings at Bawe and Chumbe Island is contrary to what is widely understood with regards to seasonal Chl-*a* concentration. The possible reason for this pattern might be due to the contribution of sewage inflow, the presence of the port, uncontrolled tourist activities, and fishing activities.

The waste water from Unguja Island is discharged into the ocean in the Zanzibar Channel. Because the current flows past Bawe and Chumbe Island (Nyandwi, 2013), nutrients in this waste water may affect these sites through stimulating localised phytoplankton growth. Nyandwi (2013) studied ocean circulation across the Zanzibar Channel and found that current velocity and direction is north easterly during the SE monsoon, but is reversed during the NE monsoon period. The reversal of the current during the NE monsoon season allows wastewater discharged from Zanzibar town to reach Chumbe and Bawe islands in Zanzibar channel. This suggests nutrient rich waters from Zanzibar town flow in a south westerly direction during the NE monsoon season, impacting the areas around Bawe and Chumbe islands, and enhancing primary production during the NE monsoon.

Although the difference in Chl-*a* concentration between the NE and SE monsoon seasons and among sites were insignificant ($F_{(1,3)} = 1.3144$, $p = 0.2949$), principal component analysis uncovered how environmental variables, which vary with season, influenced Chl-*a* concentration within the study sites (Fig. 4). Ammonia, phosphate and nitrate were found to be the dominant contributor to high Chl-*a* at Chumbe and Bawe during the NE monsoon season (Fig. 5). Ammonia, temperature, and dissolved oxygen were the dominant contributor to high Chl-*a* at Pongwe, Bawe and Mnemba during the SE monsoon season (Fig. 6). Surprisingly, nitrate and phosphate contribute negatively to Chl-*a* concentration during the SE monsoon season (Fig. 6).

Generally, environmental variables affect Chl-*a* concentration in time and space. The median Chl-*a* was insignificantly higher during the SE as compared to the NE monsoon season. While ammonia and nitrate were the main contributors to high Chl-*a* concentration in the SE monsoon season, pH, temperature and phosphate affected the amount of Chl-*a* in the NE monsoon season. Based on site-season interaction, ammonia, phosphate and nitrate had major influences on high Chl-*a* at Chumbe and Bawe during the NE monsoon season. In contrast, temperature, dissolved oxygen and ammonia affected Chl-*a* concentration at Pongwe, Mnemba and Bawe during the SE monsoon season. Since Chl-*a* is used as the proxy for phytoplankton biomass, understanding how these microscopic plants change over time and space is important, because any change in Chl-*a* concentration affects other marine organisms, and ecosystems in general.

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Integrated seaweed – sea cucumber farming in Tanzania

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Abstract

We review piloted co-culture experiments of the sea cucumber *Holothuria scabra* with different seaweed species in existing lagoon-based seaweed farms in Tanzania during 2011-2014. Key questions were whether stocking densities would influence growth rates of both species, and whether deposit feeders would modify organic components in the sediments. From a social perspective, we investigate if local people are readily willing to become involved in sea cucumber farming as an optional livelihood. Seaweed-specific growth rates between 0.32 and 4.1 $\%d^{-1}$ were reported, showing significantly higher values for those treatments combined with sea cucumbers than for the seaweed monoculture ($F_{3,1}$=3.20, $p<0.05$) at Zanzibar sites. Sea cucumber growth rates ranged from 0.14 to 1.6 gd^{-1}, and all of the studies showed that the treatments holding *H. scabra* at a low stocking density (average of 130 gm^{-2}) presented a higher growth performance than when it was stocked at more than 200 gm^{-2}. Total organic matter in sediments increased in all treatments over the sampling periods ($p<0.05$). Some 88 percent of the surveyed local people showed willingness to participate in this type of mariculture for livelihood. The survey identified theft and lack of credit as the main hindrances for this activity. *H. scabra* is viable for lagoon co-culture with seaweed when taking into account proper stocking density, implications on total organic matter and total organic carbon in the system, and local acceptance by local people.

Keywords: *Holothuria scabra, Eucheuma, Kappaphycus*, co-culture, IMTA, growth rates, organic matter, stocking density

Introduction

Seaweed is important for Tanzania, both from social, ecological and economic points of view. From an energetics point of view, seaweeds represent the result of the most significant and most economic transfer of sunlight into sugars, energy and a number of phycocolloids (Bresinsky *et al*., 2008). The direct use of seaweeds for human consumption worldwide is in the range of 300 000 tons/year with species such as Nori (*Porphyra* spp.), *Ulva*, *Fucus* and others being produced (CEVA, 2013). About 10–12 million tons/year of seaweed are harvested from the wild and from aquaculture, and some used in the seaweed processing industry (Nayar and Bott, 2014) with a value of US$ 6 billion. The largest share (US$ 5 billion) is for human food products, while US$ 1 billion is for industrial products such as hydrocolloids, commonly used in animal feeds, bioactives and fertilizers (1 million tons). In Tanzania about 11-15 thousand tons (dry weight) of *Eucheuma* and *Kappaphycus* species are produced yearly, with a total value of 1.6 – 2.0 billion Tsh (Msuya, 2012; Msuya *et al*., 2014).

Sea cucumbers play a key role in marine ecosystems through bioturbation, burrowing and feeding on organic matter in marine sediments (Purcell *et al*., 2016). Marketed as *beche-de-mer* or *trepang*, they are highly valued because they are rich in protein, popular

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as an aphrodisiac, used in traditional Chinese medicine, and it is known that they contain significant amounts of chondroitin-sulfates for cartilage support. Recently a number of bioactive compounds such as saponins, chondroitin sulfates, glycosaminoglycan, sulfated polysaccharides, sterols, phenolics, cerberosides, lectins, peptides, glycoprotein, glycosphingolipids and essential fatty acids have been reported (Bordbar *et al.*, 2011). Unfortunately, worldwide they are severely overexploited, including in Tanzania (Eriksson *et al.*, 2013). Some 66 out of more than 400 species of sea cucumbers are commercially used (Purcell, 2010), around 41,000 tons dried tons of global wild captures annually (FAO, 2018). The percentage of these coming from global aquaculture has increased from about 1% in 2002 to around 25% in 2011 (Purcell *et al*., 2011). In Tanzania there are about 20 species of sea cucumbers commercially used (Mbaga and Mgaya, 2004), with the sandfish, *Holothuria scabra*, having the highest market value.

While mariculture is gaining popularity in many developing countries, it's development comes with potential significant environmental impacts in coastal ecosystems, including threats to habitats such as mangroves, seagrass beds and coastal lagoons. Intensive mariculture is a potential source of pollution in terms of effluents or sediment eutrophication through bio-deposition (Black, 2001; Zhang *et al.,* 2012). Chopin *et al.* (2001) showed that Integrated MultiTrophic Aquaculture (IMTA) can be considered as a mitigation approach against excess nutrients and organic matter generated by intensive mariculture activities by using waste from one species as inputs for another. In an ideal case, fed species (such as fish) are combined with extractive species (such as algae, bivalves or sea cucumbers) to make use of surplus nutrients. This leads to balanced systems for environment remediation (biomitigation), provides economic stability (improved output, lower costs, product diversification and risk reduction) and leads to better social acceptability (e.g. through better management practices and higher yields). Initial steps for IMTA include different co-culturing approaches, by combining at least two species of different trophic level such as seaweed and sea cucumbers.

The Leibniz Centre for Tropical Marine Research (ZMT) has been collaborating with Tanzania in developing seaweed-sea cucumber co-culture research since 2012. There have been studies to co-culture seaweed and sea cucumbers since 2011, when the first of such experiments was done. This was followed by three more studies: Beltran-Gutierrez (2012); Fabiani (2013); and Namukose (2014). So far there have not been studies to integrate seaweed and sea cucumber in a set-up of more than two species in Tanzania. Furthermore, these scientific studies have not been appropriately disseminated to farmers although application of such results could improve livelihoods of coastal communities in Tanzania and the western Indian Ocean in general (Purcell and Eeckhaut, 2005; Eeckhaut *et al*., 2008; Robinson and Pascal, 2009).

While searching for optimizing growth and improving potential nutrient flow synergies between species, *Eucheuma denticulatum* was deliberately included as it is a much more widespread seaweed species in Zanzibar and apparently more resistant to adverse environmental conditions than *Kappaphycus* (Hayashi *et al.* 2010; Msuya and Porter, 2014).

A wide range of stocking densities in sea cucumber aquaculture have been applied in order to better understand trade-offs and limiting capacities, as well as relation to organic matter content of sediments (Zamora *et al*., 2016). This paper reviews research in integrated seaweed-sea cucumber culture in Tanzania and fills scientific gaps that could be used to produce more information on the cultivation of these high-valued organisms. In particular, the following open questions were addressed: How do stocking densities of sea cucumber (*Holothuria scabra*) and three seaweed species (*Kappaphycus striatus, K. alvarezii* and *Eucheuma denticulatum*) influence the growth and survival of both species when co-cultured in an integrated system?; and, Are local people with/without experience in seaweed farming readily willing to become involved in sea cucumber farming as an additional source of income?

Materials and Methods

Research was conducted in the years 2012 – 2014 at three different locations in Tanzania: Muungoni and Bweleo (Menai Bay) in Zanzibar; and Pangani (Pangani District) in mainland Tanzania (Fig. 1).

Surface seawater temperature, pH, salinity and oxygen concentration were measured at the sites during the experimental periods at regular intervals during the morning hours (8:00–11:00 am), using an HQ40d multimeter and refractometer.

Experiment set-up

Cages of 0.5 m height, with a base area of 2.25 m^2 were constructed at Zanzibar sites, and 1 m^2 at the mainland Pangani site, using wooden poles, 10mm coated wire mesh, cable ties and 4 mm nylon ropes. They were installed in the intertidal area, near local seaweed farms at each study site. The sites selected consisted of sand, sandy/muddy substrates and sea-grasses. The wire mesh of the cages was anchored into the sea sediment to a depth of 25 cm to avoid escaping of sea cucumbers according to the method described by Slater and Carton (2007).

All cages were thoroughly scrubbed with a hard nylon brush on a two-weekly basis to remove any fouling which would inhibit deposition of detritus from the seaweed to the sediment within the cage. Likewise, by scrubbing, the cages were maintained free of fouling and wrack/flotsam.

Holothuria scabra (Hamel *et al*., 2001) were collected from intertidal areas during spring low tides, weighed to the nearest 0.1 g, and allocated to cages according to specified stocking densities (Table 1). In the Zanzibar sites, sea cucumber juveniles were collected in intertidal pools at Bweleo and Unguja Ukuu (villages within Menai Bay). For the experimental trial in Muungoni, a total of 52 individuals with an average body weight of 97 ± 31g (mean ± SD) were allocated to experimental cages while 60 medium-sized individuals were allocated to experimental cages in Bweleo, with an average initial weight of 114 ± 37g. At Pangani District, younger juveniles found ranged from 29.1 to 66.6 g per animal with an average weight of 44 ± 15 g. In all trials, sea cucumbers had an acclimatization period to ensure they had not eviscerated prior to allocation into experimental cages. This period varied among the three studies consisting of 30 days at Muungoni, 14 days at Bweleo and 7 days at Pangani District. In addition, before starting the experiments, each *H. scabra* individual was photographed dorsally and ventrally, allowing characteristic markings to serve as means of recognition for individual growth and survival monitoring at every sampling event as suggested by Raj (1998), and used reliably for several sea cucumber species in previous studies (Slater and Carton, 2007; 2009).

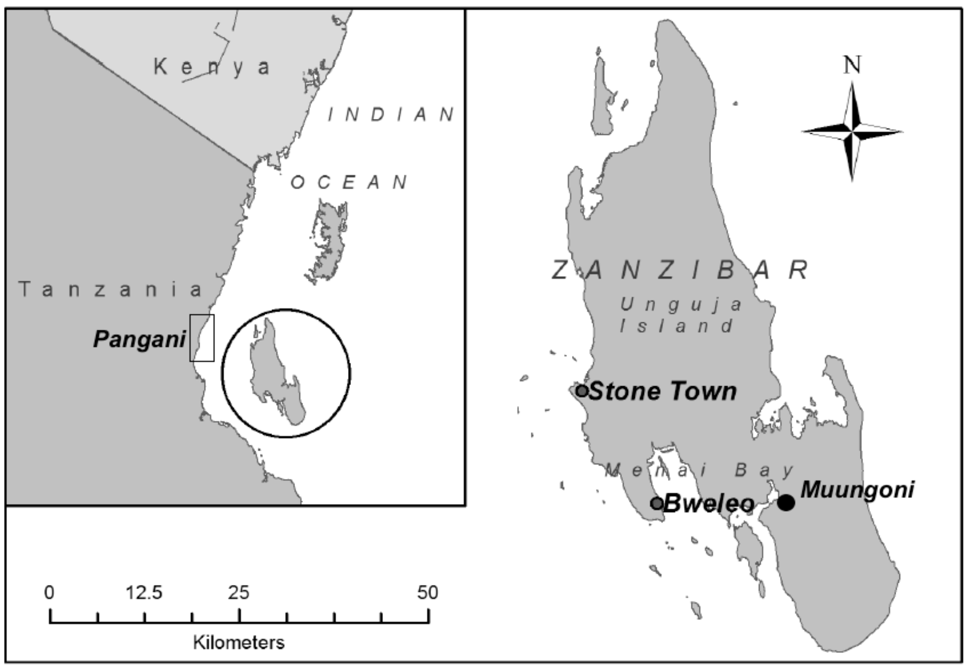

Figure 1. Location of the experimental sites in the coastal area of Muungoni and Bweleo on Unguja Island, Zanzibar (Beltran-Gutierrez, 2012; Namukose, 2014) and Pangani District on mainland Tanzania (Fabiani, 2013).

Meanwhile, seaweed fragments purchased from local farms at each site were planted above the sea cucumber cultures using the off-bottom method, commonly practiced for seaweed farming in Zanzibar (Fröcklin *et al*., 2012), as is shown in Fig. 2. Healthy bunches of seaweed were tied to a 4 mm Ø rope using a 1 mm Ø string (*tie-tie*), maintaining a distance of 20 cm between fragments/cuttings. The culture rope lines were tightened on top of the sea cucumbers (outside the cages at Muungoni site and inside the cages in Bweleo and Pangani studies), suspended around 50 cm above the sea floor, and stretched with wooden

Figure 2a, b. Exemplary experimental plot showing seaweed fragments planted using the off-bottom method, and sea cucumber culture cages.

stakes driven into the bottom. In the Muungoni study, the seaweed species used was *Kappaphycus striatus* var. *payaka brown,* locally known as *cottoni kikarafu*, whereas *Eucheuma denticulatum* was cultured in Bweleo and *Kappaphycus alvarezii* at Pangani. Two full seaweed production cycles of 6 weeks each (from planting to harvest) were completed during each study.

Experimental designs

Across the studies, treatments were established holding different densities of seaweed and sea cucumbers (Table 1) intending to reflect low (whereby optimal growth may be observed) and high density stocking (whereby growth limitation may occur) as per the literature (Battaglene *et al.*, 1999; Pitt and Duy, 2004; Purcell and Simutoga, 2008). However, at Bweleo, Zanzibar, a medium-density treatment was added to the experimental design at which higher growth performance was expected to occur. In every study, experimental plots or cages without seaweed and sea cucumbers were added as procedural controls and each of the treatments had four replicate plots.

In Muungoni and Bweleo, the initial seaweed density was constant across treatments (ca. 500 gm^{-2}), reflecting the actual cultivation method of the seaweed farms. On the other hand, in Pangani, low and high *K. alvarezii* stocking densities of 500 g and 1000 gm^{-2}, respectively, were used as per the study by Hurtado *et al.* (2008) for stocking of seaweed.

Therefore, at Muungoni and Bweleo, there was a control treatment that held seaweed in monoculture (no sea cucumbers), another control treatment with only sea cucumbers, and experimental treatments holding sea cucumbers at low, medium (only for Bweleo trial) and high stocking densities combined with seaweed.

In the study at Pangani, eight treatments were established. First and second cage treatments were stocked with seaweed only, while the third and fourth were stocked with sea cucumbers only. These served as the controls in this experiment. The remaining plot treatments (fifth, sixth, seventh, and eighth) were stocked with both seaweed and sea cucumbers at low and high stocking densities of each species (Table 1).

Monitoring of growth of seaweed and sea cucumbers

The production cycle of seaweed took six weeks in Muungoni and Bweleo, while it consisted of eight weeks in the Pangani study. Seaweeds are usually harvested after six weeks in most parts of Tanzania, and so far only two reported studies have kept the seaweed longer than six weeks (Msuya and Salum, 2012; Msuya *et al.*, 2012). Each rope with seaweed bunches was removed from the respective cages and shaken to reduce excess water before the wet weight was taken using a digital scale.

Furthermore, caged sea cucumbers were monitored every 14 days for a two-month period. Sea cucumbers were removed from cages and brought to the surface in a 10-litre watertight container. For growth assessment, every sea cucumber was measured on an individual basis (photo-identification) at every sampling. Before measurement, the animals were given a one minute period to allow water to be expelled from the respiratory trees (Slater and Carton, 2007). They were photo-identified by their colour patterns, weighed, and returned to their respective cages. Time outside the cages did not exceed 5 minutes.

Specific growth rates (SGR) for both seaweed and sea cucumbers were calculated by using the formula according to Dawes *et al.* (1993):

$$SGR = 100 \times (\ln W2 - \ln W1)/T,$$

where SGR denotes the specific growth rate (% d^{-1}); W2 represents mean wet weight on the sampling day (g); W1 refers to initial wet weight (g); and T stands for the period of cultivation in days.

Total Organic Matter and Total Organic Carbon in sediment

Before stocking of seaweed and sea cucumbers in experimental plots at all study sites, 40 g sediment samples were taken from each plot to estimate total organic matter (TOM), and in the Bweleo study, total organic carbon (TOC) was also considered. The same procedure was repeated at the end of the experiments. This allowed a determination of the amount of organic matter available for sea cucumbers feeding by comparing cages of different densities and without sea cucumbers, to get an idea of how much organic matter was consumed by the animals.

All collected samples (before and after) were dried on tin foil in the sun for six hours, then stored at room temperature for later analysis. Sediments TOM was determined by using an ash-free analysis method. Sediment samples were dried at 60°C for two days and weighed with a digital scale to the nearest milligram. After measuring, samples were placed in a furnace and burned at a temperature of 450°C for 6 hours. After that, the furnace was allowed to cool and the samples were weighed again. Percentage of organic matter was obtained by calculating the weight loss of the sediment samples after the burning in the furnace.

Estimation of TOC was done by weighing three homogenized samples of sediment (~25–35 g) per cage using a Mettler Teledo weighing scale (precision: 0.001 g) in pre-combusted (500 °C, 3 h) silver cups. Inorganic carbon removal from the samples prior to TOC analysis followed a protocol similar to that in Kennedy *et al.* (2005), and the elemental determination for carbon was

Table 1. Experimental treatments and stocking densities (gm^{-2}) of co-culture trials combining seaweed and sea cucumber, *H. scabra* in Tanzania during study periods between 2011 and 2014 (Beltran-Gutierrez, 2012; Fabiani, 2013; Namukose, 2014).

Study site	Time period	Plots	Treatments	Stocking density (g/m2)	
				Sea cucumber	Seaweed
					Kappaphycus striatus
Muungoni	From Dec 2011 to Mar 2012	16	Control seaweed	0.0	570 ± 22
			Control sea cucumber	218 ± 21	0.0
			Low sea cucumber	124 ± 22	582 ± 6
			High sea cucumber	218 ± 16	571 ± 21
					Eucheuma denticulatum
Bweleo	From Dec 2013 to Mar 2014	20	Control seaweed	0.0	500 ± 23
			Low sea cucumber	150 ± 5	500 ± 23
			Medium sea cucumber	236 ± 24	500 ± 23
			High sea cucumber	345 ± 48	500 ± 23
					Kappaphycus alvarezii
Pangani	From Jan 2013 to Mar 2013	32	Control low seaweed	0.0	506 ± 18
			Control high seaweed	0.0	1008 ± 4
			Control low sea cucumber	109 ± 16	
			Control high sea cucumber	192 ± 14	
			Low seaweed Low sea cucumber	105 ± 11	506 ± 4
			High seaweed Low sea cucumber	124 ± 20	1015 ± 5
			Low seaweed High sea cucumber	217 ± 18	506 ± 5
			High seaweed High sea cucumber	262 ± 52	1001 ± 2

carried out using a CN Analyzer (Eurovector EA3000). Laboratory analyses were done at the Institute of Marine Sciences, Zanzibar and the ChemLab at the Leibniz Centre for Tropical Marine Research, Bremen.

Economic viability

This component of the study was done at Muungoni and Pangani sites, where the economic viability of integrated culture of seaweeds and sea cucumbers was assessed. A calculation of productivity and economic returns was carried out by using data obtained from the experiments. Analysis for long-term profit was approximated by annual projections of the costs and revenues. Factors such as the initial investment costs, the farming period of seaweed and sea cucumbers, and the labour costs, usually in terms of opportunity costs, as well as the returns from production, were taken into consideration (Troell, 2009).

Perceptions on mariculture as a livelihood option

The willingness of local communities to adopt sea cucumber farming as a livelihood activity was assessed in Pangani District in the three coastal villages Kipumbwi, Mikocheni and Mkwaja. These villages were specifically chosen due to their dependence on marine related livelihood activities. Focus group meetings and semi-structured interviews were conducted with the participants involved in the farming before and after the trial experiment (Denscombe, 2007). The discussion was concentrated on topics such as preference as a job, strategy to improve the farming and harvesting, and problems associated to mariculture activities.

Statistical analyses

To compare the effect of stocking density and seaweed on sea cucumber growth performance, linear mixed models (GLMMs) were employed using the 'lme4' package (Bates *et al.*, 2012) of the statistical software R version 2.15.1 (R Development CoreTeam 2008 and 2013). The effect of stocking density on daily growth rate of sea cucumber was analyzed in a single model, with treatment and period as fixed terms, while the initial weight of individuals nested in cages was added to the model as a random term.

Also, the effect of stocking density of sea cucumbers on the survival and growth of seaweed was analyzed in two separate models for the two harvest cycles (where harvest cycle 2 was characterized by different weather conditions in the form of severe storms models), then later in a single model with treatment and harvest cycle as fixed terms, while blocks were added as random terms to the models. Negative values of seaweed data were interpreted as massive breakage loss that occurred, and hence not considered in the analysis. All interaction terms that were not significant were eliminated from the models. Where significant effects were detected, pairwise comparisons using Tukey's honest significant difference (HSD) post-hoc test was used.

For organic matter content, percentage of organic matter (% OM) was estimated by calculating the loss of weight (g) of the samples after the burning procedure. Then, a paired t-test on depended samples was applied to compare % OM in sediments between cages under the treatments sampled before and after the experiment. To analyze the effect of sea cucumber stocking density on sediment organic matter and total organic carbon, generalized linear mixed models (GLMMs) were applied.

Results

The abiotic parameters during the experimental periods did not vary much. Salinity averaged 35.1, temperature ranged from 29 to 34°C, pH was between 7.9 and 8.8, while oxygen values ranged from 6.01 to 8.15 mgL^{-1}.

Growth rates of seaweed and sea cucumbers

Seaweed growth rates

In Muungoni, Zanzibar, mean initial *K. striatus* density of $570gm^{-2}$ increased to $1.600gm^{-2}$ after one culture cycle of 41 days, reaching a specific growth rate (SGR) of $4.1\%d^{-1}$. This was the highest seaweed growth registered in the trial studies, which was recorded in co-culture with sea cucumbers at low stocking density. At Bweleo, Zanzibar, the highest *E. denticulatum* performance (SGR 2.3 $\%d^{-1}$) was observed in combination with a high-density of sea cucumbers (Table 2).

In both Zanzibar studies, lower values for seaweed growth were registered in the plots without sea cucumbers (Control seaweed treatment). SGRs of *E. denticulatum* in Bweleo differed significantly among treatments ($F_{3,1}$=3.20, P=0.03; Fig. 3) and post-hoc tests revealed that mean growth rates of seaweed in the medium and high culture density treatments differed significantly from those in monoculture ($P< 0.05$).

In Pangani, the average wet weight of *K.alvarezii* increased from 252 to 295.4 g per 1 m of culture line. The overall mean SGR across treatments was $0.54\%d^{-1}$

Table 2. Growth performance of co-cultured experimental trials between 2011 and 2014 (Beltran-Gutierrez, 2012; Fabiani, 2013; Namukose, 2014).

Study site	Experimental treatments	Growth performance		
		Sea cucumber	Seaweed SGR (% day-1)	
			Kappaphycus striatus	
Muungoni		**(g d^{-1})**	**First cycle**	**Second cycle**
	Control seaweed		0.73 ±0.6	3.7 ±0.7
	Control sea cucumber	0.9 ±0.1		
	Low sea cucumber	1.6 ±0.2	1.22 ±0.5	4.1 ±0.6
	High sea cucumber	0.9 ±0.1	1.40 ±0.8	3.6 ±0.6
			Eucheuma denticulatum	
Bweleo		**(g d^{-1})**	**First cycle**	**Second cycle**
	Control seaweed		0.79 ±0.4	0.75 ±0.4
	Low sea cucumber	0.8 ±0.3	1.59 ±0.5	0.45 ±0.4
	Medium sea cucumber	0.43 ±0.2	1.87 ±0.5	1.01 ±0.3
	High sea cucumber	0.14 ±0.4	2.33 ±0.7	1.01 ±0.3
			Kappaphycus alvarezii	
		SGR (% day^{-1})	**First cycle**	**Second cycle**
Pangani	Control low seaweed		1.19 ±0.6	0.72 ± 0.7
	Control high seaweed		1.63 ±0.4	0.32 ±0.2
	Control low sea cucumber	0.99 ±0.0		
	Control high sea cucumber	0.45 ±0.1		
	Low seaweed low sea cucumber	0.32 ±0.0	1.84 ±0.8	0.37 ±0.2
	High seaweed low sea cucumber	0.89 ±0.1	1.75 ±1.1	0.32 ±0.1
	Low seaweed high sea cucumber	0.38 ±0.1	1.73 ±1.0	0.52 ±0.1
	High seaweed high sea cucumber	0.68 ±0.2	1.67± 1.2	0.56 ±0.2

±0.1 (mean±SD), reaching the highest value (1.84 %d^{-1}) in the treatment of low seaweed density combined with low sea cucumber density, as was also shown in Muungoni (Table 3). Seaweed stocked at lower density (500 gm^{-2}) had a slightly higher average growth rate of 0.63% d^{-1} compared to the high stocking density (1000 gm^{-2}), which reached 0.45% d^{-1}. Overall, in the three studies species of seaweed showed lower growth rates in monoculture treatments (seaweed control) and better performance when co-cultured with sea cucumbers (Fig. 3).

Sea cucumber performance

Survival of *H. scabra* at the Muungoni, Bweleo and Pangani trials was 85, 77 and 56% respectively. In all cases, this was not dependent on the stocking density of the treatments, but rather on uncontrollable factors attributed to the escape of animals through the mesh, given the small initial sizes of individuals. Considering sea cucumber growth in Muungoni, the average growth rate at low stocking density (1.6 gd^{-1} ±0.2) (mean±SD) was significantly higher compared to those at high stocking density (0.9 gd^{-1} ±0.1) ($P<0.01$) (Table 3). The growth rate varied significantly between measurement periods ($P<0.001$), being highest in the last time period and lowest at intermediate time periods (Fig. 4).

In Bweleo, daily growth rates of *H. scabra* also differed significantly among treatments (low, medium and high) over the sampling periods ($F_{2,5}= 5.15$, P=0.0095). Sea cucumbers cultured at low stocking density with seaweed had the highest average growth rate of 0.8g d^{-1} ±0.3, whereas those cultured at high stocking density had the lowest average growth rate of 0.14 gd^{-1}

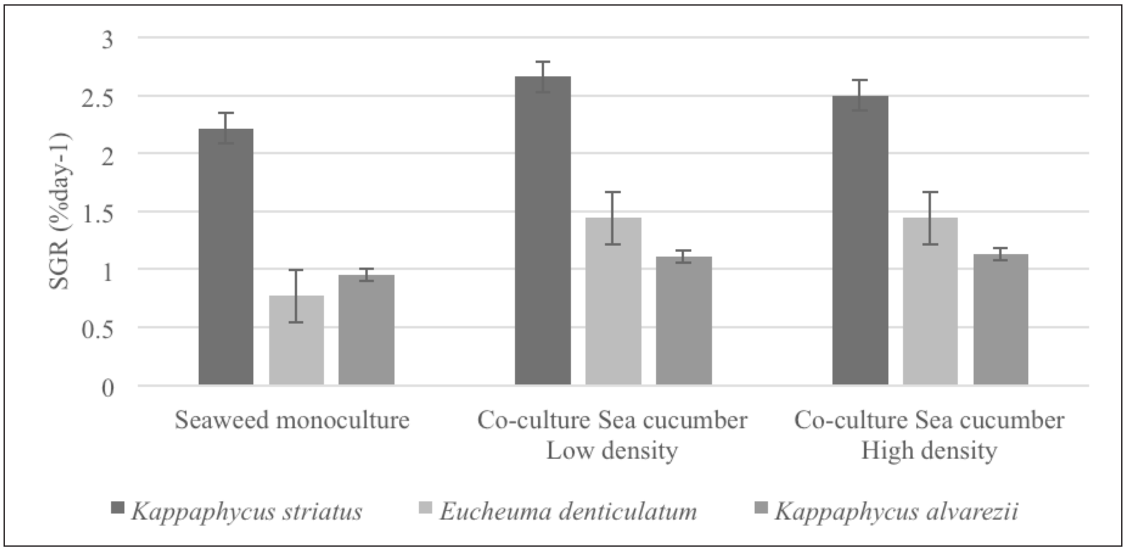

Figure 3. Average specific growth rates (SGR%day^{-1}) for the three species of seaweed *K. striatus*, *E. denticulatum* and *K. alvarezii*, each of them at 500 gm^{-2} stocking density in the following treatments: Control seaweed (monoculture), in combination with sea cucumber at low density (ca. 126g/m^{2}), and sea cucumber at high density (ca. 224 gm^{-2}). Error bars indicate standard error.

±0.2. Post-hoc tests revealed that mean growth rates in low and high density treatments differed significantly from each other ((Tukey's HSD test, $p<0.05$). At the end of the study, individuals cultured at low stocking density had the highest average weight (163 g ±34), while those cultured at high stocking density had the lowest value (118 g ±39).

In Pangani, the average specific growth rate (SGR) of sea cucumbers observed during the study was 0.62 %d^{-1} ±0.3 (mean±SD). In contrast to the Zanzibar studies, the lowest SGR of *H. scabra* in Pangani was recorded in the treatment with low stocking density (105 gm^{-2} of sea cucumber) combined with low seaweed density (506 gm^{-2} of *K. alvarezii*), while highest SGR was observed at low density monoculture of *H. scabra* (0.99 %day^{-1}). On average, the growth performance of sea cucumbers when co-cultured with high seaweed density was 55% higher than when combined with low seaweed density.

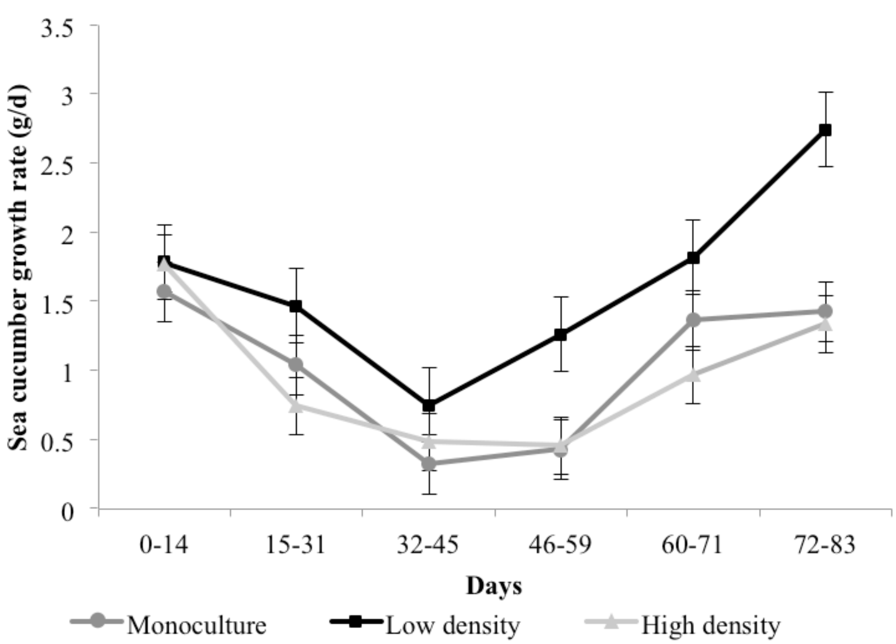

Figure 4. Mean growth rate (gd^{-1}) through sampling periods (days) of sea cucumber *H. scabra* in monoculture and co-cultured with seaweed *K. striatus* at two different sea cucumber stocking densities (low: 124 gm^{-2}, and high: 218 gm^{-2}) at Muungoni intertidal lagoons, Zanzibar. Error bars indicate standard error.

TOM and TOC in the sediment

In both the Muungoni and Bweleo trials, TOM in sediments increased generally in all treatments over the sampling periods. At Muungoni, a paired t-test showed that organic matter content percentages (OM%) of surface sediments in cages with treatment holding only sea cucumbers at a high density (without seaweed), were significantly higher after the trial time period (p=0.04). Despite the fact that OM% increased in cages with seaweed together with sea cucumbers at low and high stocking densities, differences in these plots were not significant (p=0.05355 and p=0.2284, respectively). Furthermore, TOM showed no variation in the treatment cages that held only seaweed in monoculture.

In contrast, at the Bweleo experimental site, TOM content differed significantly among treatments over the sampling periods (p=0.01). The interaction between treatment and culture period was marginally above the significance level (p=0.06, α=0.05). Generally, as in Muungoni, TOM in sediments increased in all treatments over the sampling periods, with the exception in the sea cucumber medium density treatment. TOM remained almost the same in the procedural control treatment (with neither seaweed nor sea cucumbers).

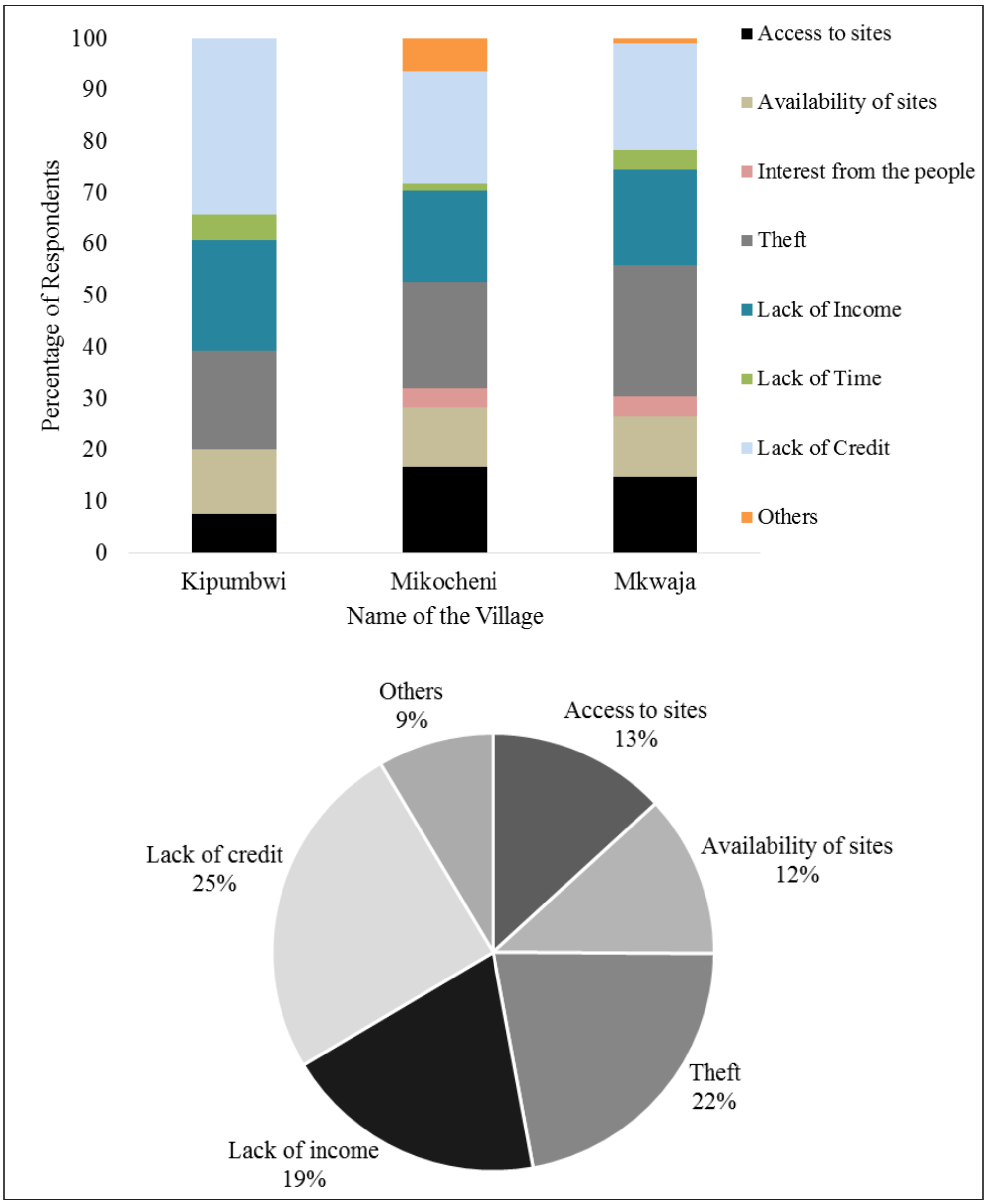

Figure 5a. Frequently mentioned problems likely to be encountered in sea cucumber farming in three villages of Pangani (n = 75).

Figure 5b. Percentage of responses to problems likely to be encountered in sea cucumber farming in Pangani, Tanzania (n = 75).

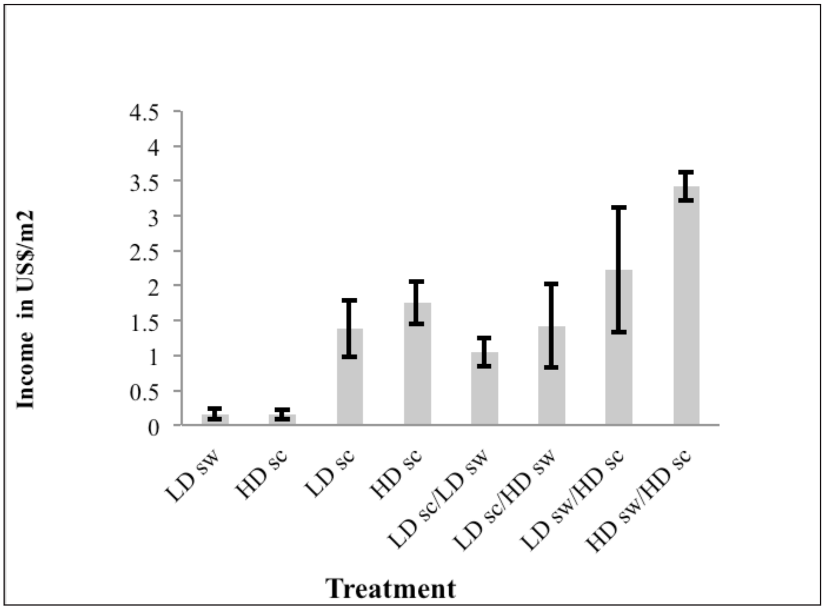

Figure 6. Income (US$m^{-2}) of *H. scabra* and *K. alvarezii*, at different low/high (LD/HD) stocking densities and combinations. Seaweed sw LD 500 gm^{-2}, sw HD 1000 gm^{-2}, sea cucumber sc LD 100 gm^{-2}, sc HD 200 gm^{-2}.

Additionally, TOC content of the sediment differed significantly among treatments over the sampling periods (p=0.03). The highest decrease in TOC was observed in the medium treatment, with the lowest overall mean value of 0.26 ±0.02% dry weight (mean ± SD) in the last sampling period. At the end of the cultivation period at Bweleo, TOM content in the guts of sea cucumbers was higher than that in the sediment.

Economic viability

Perceptions on mariculture as a livelihood activity

Respondents from the three villages were mainly involved in seaweed farming and fishing, and occasionally in other activities such as crop cultivation, food vending and rearing of animals.

The economic yield from different treatments was assessed. Overall performance in terms of returns differed substantially between both cultured species. Average yield for monoculture of sea cucumbers was US$ 1.57 ± 0.26 SD, while for seaweed it was US$ 0.16 ± 0.01 SD (Fig. 6). Across all treatments, a higher economic yield was noted in the integrated compared to the monoculture system.

The age of respondents showed a slight influence on people's willingness towards sea cucumber mariculture activities, since people between 21 and 41 years old showed high interest (93 %), compared to those between 41 and 50 years (86 %), as well as older than 51 years (80 %) (n=75). When asked whether they were willing to participate in sea cucumber mariculture to

Table 3. Cost and revenue for integrated aquaculture production at 500 g and 1000 g per m^2 experimental cages in Pangani with 1 m^2 each. Exchange rate is 1US$ = 1600Tsh.

	Integrated aquaculture production trial					Annualization		
	Investment cost per 1m^2 cage (US$)	Potential Production 1m^2 cage (Kg)	Unit Price	Income (US$)	Profit	Potential Production (kg)	Income (US$)	Profit (US$)
Seaweed	0.59	0.29	0.5	0.15	-0.44	2.03	1.02	0.43
SeSea cucumbers	18.26	0.14	12.5	1.75	-16.51	0.84	10.50	-7.76

earn more income, the majority of respondents in all three villages agreed (88 %), while only 10 respondents (11.8 %) did not show an interest.

When asked the kind of challenges they were likely to face during sea cucumber farming, most of the respondents frequently mentioned theft issues. Other problems that were given equally high attention included lack of credit and income. Difficulty in access to sites and space availability were moderately mentioned as well (Fig. 5).

Discussion

Integrating commercially valuable sea cucumbers into existing aquaculture can increase economic yields and reduce environmental impacts without increasing pressure on contested coastal spaces and resources. This review discusses piloted co-culture of the sea cucumber *H. scabra* with different seaweed species in existing lagoon-based seaweed farms in Tanzania. Survival and commercially viable growth of both algae and sea cucumbers clearly showed such a system's viability for up-scaling and commercial development.

Growth of seaweed and sea cucumbers

Stocking density showed an influence on seaweed growth. Seaweed at low stocking density grew better in these studies than those stocked at high density. These observations are in line with results reported by Hurtado *et al.* (2008) on *K. striatus*, where a lower stocking density (500 gm^{-2}) and shorter culture period (30 days) yielded a higher growth rate than a higher stocking density (1000 gm^{-2}) and longer culture period (i.e. more than 45 days). The observed specific growth rates (SGRs, expressed in % per day) of seaweed in the present studies at Bweleo are lower than those from previous studies on *E. denticulatum* in Zanzibar and other areas. Such higher growth rates reported are, notably, 3.50 in Hawaii (Glenn and Doty, 1990), 4.7 in Kenya (Wakibia *et al.*, 2006), and 3–11 in Tanzania (Msuya and Salum, 2012). Msuya *et al.* (2012) reported a growth rate of 5.1 for *E. denticulatum* in Uroa, Zanzibar. However, a low rate of 1.4 %day^{-1} of this species has been reported from the southwest coast of Madagascar (Mollion and Braud, 1993) as well as minimum values of less than 1%day^{-1} at Uroa, Zanzibar (Msuya *et al.*, 2012). As in the present study, lower seaweed growth rates have been attributed to the 'ice-ice' disease and severe storm events. These, in addition to epiphytism and fouling, are some of the major production challenges faced by seaweed farmers of Zanzibar and the western Indian Ocean region in general (Hayashi *et al.*, 2010; Msuya, 2012; 2013; Msuya and Porter, 2014). Nevertheless, the study here presented on *K. striatus* achieved growth rates of 4.1 %day^{-1} as compared to 1.1–4.0 %day^{-1} cultured in its native area of Philippines (Hurtado *et al.*, 2008).

Highest growth rates of *E. denticulatum* were achieved with stocking densities of *H. scabra* of 200 and 300 g per m^2. Similar results were obtained by Uthicke (2001) and Wolkenhauer *et al.* (2010), who observed that sea cucumbers enhance productivity of primary producers through their burrowing and nutrient recycling activities.

On the other hand, the growth rate of *H. scabra* can depend on the environmental situation and the time of the year (Agudo, 2006). Interestingly, the average growth rates of this species in the Zanzibar studies here reported, were different from 2012 (1.4 gd^{-1}) to 2014 (0.5 gd^{-1}), despite trials being performed during the same time of the year, but at different coastal lagoons (Fig. 1).

The mean growth rates from the present study are similar to peak growth rates (1.8 gd^{-1}) reported by Robinson and Pascal (2009) for *H. scabra* in commercial lagoon monoculture and exceed the 0.49–1.5 gd^{-1} reported for similar sized animals in pond culture (James 1999; Pitt and Duy, 2004; Purcell and Kirby, 2006). At Pangani District, specific growth rates (SGR) were estimated for *H. scabra* co-cultured with *K. alvarezii*, recording an average of 0.6 %day^{-1}.

All of the studies showed that the treatments holding sea cucumber at a low stocking density (average of 130 gm^{-2}) presented a higher growth performance than when it was stocked at more than 200 gm^{-2}. Sea cucumber growth in co-culture was stable at densities exceeding the 300 gm^{-2} reported for commercial lagoon culture and the maximum biomass densities for pond-based culture (Battaglene *et al.,* 1999; Pitt and Duy, 2004; Purcell and Simutoga, 2008). Other results indicate, however, that biomass densities of more than 370 gm^{-2} may be achieved in co-culture. Lavitra *et al.* (2010) even report potential maximum juvenile sea cucumber biomass of up to 700 gm^{-2} in optimum sea pen areas before growth is limited.

When comparing growth rates within the current studies, it seems that *H. scabra* grows better when integrated with *K. striatus* than with *E. denticulatum* and *K. alvarezii*. This aspect requires further studies using direct comparisons of both seaweed species as co-cultivars as there are some factors such as environmental conditions, and size and age of individuals, which could have an influence on the results (Battaglene *et al.,* 1999; Pitt *et al.,* 2004; Robinson *et al*., 2019).

TOM and TOC in sediment

Deposit-feeding sea cucumbers tend to modify their foraging behavior and digestive capabilities to optimize the intake of nutrients from the organic component in sediments (Roberts *et al*., 2000, Zamora *et al*., 2016). In the present study, an uptake of TOM through feeding by the sea cucumbers was evident in the density treatment holding c.a. 230 gm^{-2}, which showed a significant decrease in TOM and TOC over the experimental period. On the other hand, there seems to have been an accumulation of TOM in the sediments from plots without sea cucumbers. This could have been due to break-offs of seaweed thalli adding to the sediment inside the cages.

Studies on TOC utilization by *H. scabra* for comparison are limited in the literature. However, in other deposit-feeding holothurians, Michio *et al.* (2003) observed a decrease in TOC in surface sediments with sea cucumbers (*Apostichopus japonicas*) compared to sediments without sea cucumbers. Slater and Carton (2009) made a similar observation with *Australostichopus mollis* in coastal sediments impacted by mussel farm deposits. Sediment reworking and organic matter uptake by holothurians depends on individual numbers and sizes, food availability and local conditions (Dar and Ahmad, 2006). In the present study, it is evident that the stocking density was a key factor that controlled TOM and TOC utilization in the system, but it does not rule out other possible factors that were beyond the scope of this study.

Economic viability

Sea cucumber prices on the market are very attractive, which make them prone to theft. Theft issues have been reported in various places where sea cucumber farming has been operated, for instance in Madagascar (Robinson and Pascal, 2009). In the present study, when respondents were asked which problems they were likely to meet during sea cucumber farming, most of the respondents mentioned theft issues (Fig. 6). Other problems that were given equally high attention from respondents included access to sites and availability of the sites. Site selection is very important for sea cucumber farming. According to Rougier *et al.* (2013), important criteria to be considered during site selection include the presence of at least a 20 cm layer of sediments for strengthening cages, and a water depth of at least 10 cm during low spring tides. Robinson and Pascal (2009) also suggested that the site should be near a village for facilitating monitoring and necessary maintenance.

A fully functioning integrated system normally should result in better production according to Barrington *et al.* (2009). Revenues from the integrated systems in this study were calculated, and cages stocked with *H. scabra* at 200 gm^{-2} and *K. alvarezii* at 1000 gm^{-2} of seaweed (see Fabiani, 2013) had the highest potential economic yield of US$ 3.42 m^{-2} (Fig. 6) and other treatments yielded US$ 2.82 m^{-2} (Beltran-Gutierrez, 2012). The total cost of investment depends on materials used and availability of resources (e.g. in the Pangani study seaweed total costs were US$ 14.33, equivalent to US$ 0.59 per cage) (Table 3). In an estimation for the whole year, the potential yield for seaweed was around 12.39 kg for six cages, equivalent to 2.03 kg cage^{-1} year^{-1}. According to Msuya *et al*. (2014), production of dried seaweed per meter line per harvest normally ranges between 0.17 and 0.50 kg. In this study, average production per 1 m^2 cage was 0.11 kg dry weight. Therefore, considering dying off of seaweed, which was 33%, the total potential yield was 1.77 kg dry weight for all six cages. Potential economic yield was estimated at around US$ 0.89 per production cycle (45-60 days).

The price of sea cucumbers in Zanzibar is around Tsh 20,000 (US$ 12.5) per kg. Zanzibar is currently the only market as no company or agent for buying sea cucumbers exists in mainland Tanzania due to a moratorium imposed by the government (Eriksson, 2013). Sea cucumbers can be produced all year around. By annualizing the production, potential yield was approximated at around 0.84 kg per 1 m^2 cage per year, which can bring an income of US$ 10.50. The cost for initial start-up capital, which was US$ 18.26 per 1 m^2 cage is relatively high compared to seaweed farm establishments, mainly due to the cost of wire mesh used. Fishing nets are alternatively used instead of wire mesh. For example, Beltran-Gutierrez (2012) used 1 cm^2 mesh fishing net in Zanzibar for integrated seaweed and sea cucumbers and the result was successful. The cost for initial start-up capital can be significantly reduced when using fishing nets instead of wire mesh. Also the anticipated production of sea cucumber fingerlings in a hatchery in Zanzibar by initiatives currently being supported by the Korean International Cooperation Agency (KOICA) and FAO, would significantly reduce costs.

Practical potential of integrating sea cucumbers and seaweed

This study provides useful information for future mariculture development in Zanzibar and Tanzania as a whole, however information gaps concerning mariculture of sea cucumber resources in Zanzibar still remain. Aspects like the species' absorption efficiency need to be investigated for its potential use in IMTA systems. Mariculture integration of seaweed and sea cucumbers is a potential livelihood alternative, which if adopted, can diversify livelihood portfolios of Zanzibar's coastal communities, especially of women involved in seaweed farming. Therefore, integrating sea cucumbers in seaweed farms should be promoted in Zanzibar to boost seaweed production. Significant advantages in the current co-culture model include the established and accepted nature of the culture sites and synergies in terms of husbandry and presence requirements of producers/farmers which reduces poaching and predation (Robinson and Pascal, 2009; Robinson, 2011). The piloted co-culture system, effectively integrating detritivores into existing aquaculture, allows for a significant increase in biomass production over monoculture and, given the value of sea cucumber, will result in increased income per aquaculture unit with little or no increase in resource pressure. Zanzibar seaweed farms cover approximately 1000 ha of coastal lagoon area, constituting a large established area available for co-culture of viable co-culture species (Beltran-Gutierrez *et al.*, 2016). The results of the studies reviewed here clearly demonstrate the potential of integration of commercially valuable sea cucumbers with seaweed. This review contributes to a growing body of literature establishing suitability of sea cucumbers for co-culture with existing finfish (Mills *et al.*, 2012) and bivalve culture (Kang *et al.*, 2003; Slater and Carton, 2007; Paltzat *et al.* 2008; Zamora *et al.*, 2016).

Conclusions and Recommendations

In this review it was shown that high growth performance of seaweed in co-culture with sea cucumbers can be obtained in Tanzania, using high densities of seaweed (500 gm^{-2}) and low densities (200 gm^{-2}) of sea cucumbers.

Properly managed IMTA accelerates growth with no detrimental side effects on the organisms used or the environment. IMTA can increase profits and can reduce financial risks due to weather, disease and market fluctuations. Equally, co-culture offers a more efficient use of limited coastal space. The sea cucumber *H. scabra* is highly viable for lagoon co-culture with seaweed. However, care should be taken on stocking density because exceeding recommended stocking densities compromises the growth performance of sea cucumbers and may have implications on TOM and TOC budgets in the system. The appropriate adoption of an IMTA venture could increase income, acceptance, and finally create a new family business. The initial investment can be an obstacle but the use of local materials to construct culture facilities may help. In the future it is necessary also to look into secondary products (both from the seaweed and the sea cucumbers).

Integration of more organisms, with a focus on bivalves and finfish, to create real IMTA should be encouraged. For coastal environments it is necessary to consider that it is extremely difficult to convert fishermen into fish farmers. The focus should remain on smaller communities to increase the probability of success.

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Indicators of stock status for large-pelagic fish based on length composition from driftnet fisheries in Zanzibar

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Abstract

Small-scale fisheries (SSF) contribute to approximately half of the total landings of tuna and tuna-like species in the Indian Ocean and are an important form of employment and source of protein. Research into the properties and dynamics of SSF in East Africa are important for the assessment and sustainable management of fish stocks, however, detailed fisheries data are often inadequate or absent. Fisheries-dependent data on driftnet fisheries in Zanzibar, Tanzania, was collected during the northeast monsoon seasons in 2014 and 2015. The data describes the properties of the driftnet fisheries and allows for comparisons of the length composition of the landings of the SSF with large-scale industrial fisheries (IF) fishing in Tanzania's Exclusive Economic Zone (EEZ). This data also facilitates the calculation of stock indicators for the five most abundant tuna and tuna-like species landed in Zanzibar. Results show that the two fisheries (SSF and IF) exploit the same stocks, and landings are representative of a similar length composition, while operating in different parts of Tanzania's EEZ. High exploitation rates, above reference levels for all species were calculated, in agreement with official assessments by the IOTC, and suggest that calls for the expansion of the SSF should be reconsidered. The assessment and management of straddling stocks are discussed, as well as solutions to challenges faced by local observer programmes.

Keywords: Artisanal fisheries, billfish, IOTC, length-frequency data, tuna, Western Indian Ocean

Introduction

Although SSF contribute approximately half of global fish landings (FAO, 2015), few studies have directly compared SSF to IF (Chuenpagdee *et al.* 2006; Zeller *et al.* 2007). SSF are defined as "fishing households, using relatively small amounts of capital and energy, relatively small fishing vessels (if any), and making short fishing trips, close to shore" (Garcia, 2009). These fisheries can be commercial or subsistence, but provide fish mainly for local consumption (Garcia, 2009). In comparison to IF, SSF have access to far less technological equipment, storage capacity, and engine power, but involve a higher number of people and usually generate more yield per unit of fuel (Kolding *et al.*, 2014). Compared to other coastal fisheries, large-pelagic fish are harvested in almost equal quantities by IF and SSF in the Indian Ocean (Anonymous, 2009), and therefore approaches which consider the two in unison are required. Large-pelagic fish are highly migratory with populations spanning over large areas (Kaplan *et al.*, 2014; Ward *et al.*, 1997). Accordingly, the fisheries which target large-pelagics are under jurisdiction of regional fisheries management organisations (RFMOs), such as the Indian Ocean Tuna Commission (IOTC) for the Indian Ocean. While these organisations utilise fisheries data for the management of the IF, such as CPUE, and additional data such as from large tagging programs

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(Fonteneau and Hallier, 2015), catch information from SSF is often lacking and not incorporated in stock assessments. For this reason, management advice may not always be adequate for the stock under consideration (Costello *et al.*, 2012; Cullis-Suzuki and Pauly, 2010). The monitoring of SSF is complex due to the high number of small vessels involved, the usage of a variety of gears catching a large variety of species, and lastly, due to the decentralised landing sites along long coastal stretches (Anonymous, 2009; Salas *et al.*, 2007). Thus, fishery-dependent data for SSF such as catch and effort is either scarce or of low quality, for example, aggregated at a family level (Igulu and El Kharousy, 2013; Kolding *et al.*, 2014; Salas *et al.*, 2007). The use of length-frequency distributions (LFDs) in SSF are well established (Petersen, 1981), and provide a valuable and low-cost data source suitable for assessment. They allow for the estimation of population parameters, such as growth and mortality rates, and reference levels for the state of the fishery (Beverton and Holt, 1957; Mildenberger *et al.*, 2017a). Moreover, they allow for the inference of selectivity properties of the gear and fleet, such as selected length and age classes represented in the landings.

Tanzania is the largest country in East Africa with a coastline 850 km in length and an Exclusive Economic Zone (EEZ) of over 240,000 km^2 within the Western Indian Ocean (Fig. 1; Jiddawi and Ohman, 2002). Tanzania's fishing fleet is dominated by SSF, apart from three flagged long line vessels (Igulu and El Kharousy, 2015). In 2012, the SSF in Tanzania was comprised of 4259 dugout canoes, 6815 outrigger canoes, 3842 boats powered by outboard engines, and 313 boats powered by inboard engine (Igulu and El Kharousy, 2013). Tanzania's SSF is concentrated mainly on reef areas within 6 nm from shore, while some boats exploit large-pelagic fish, like tuna, billfish and sharks using surface driftnets and handlines (Igulu and El Kharousy, 2013). In 2014, landings for tuna and tuna-like species comprised around 3468 tonnes (Igulu and El Kharousy, 2015). With regards to the IF, Tanzania sells licenses to foreign vessels for access to fish resources within its EEZ. The number of foreign vessels has increased in recent years, with licences increasing from 12 in the year 2002 (Jiddawi and Ohman, 2002) to ~77 in the years 2013/2014 (Liganga, 2014). No information is available about sport fishermen and recreational fishing in Tanzania (Igulu and El Kharousy, 2015). Zanzibar, consisting of the two islands of Pemba and Unguja (Fig. 1), contributes around 30% of Tanzania's marine fish landings (FAO, 2016). While around two thirds of Zanzibar's landings consist of near-shore resources, one third consists of large-pelagic fish caught by driftnet fisheries (FAO, 2016). Driftnets are passive nets deployed at the water surface or in the water column and used for targeting pelagic species (European Commission, 1992; Caddell, 2010). While near-shore resources are thought to be overexploited (Jiddawi and Öhman, 2002; Mkenda and Folmer, 2001; Jacquet *et al.*, 2010; Colbert-Sangree, 2012), several authors have claimed that the offshore resources provide investment potential for fisheries expansion (Jiddawi and Öhman, 2002; Feidi, 2005; FAO, 2007), resulting in research programmes to assess the potential establishment of a national IF to release pressure on near-shore resources (Igulu and El Kharousy, 2015). Aside from official assessments for large-pelagic fish for the whole of the Indian Ocean, locally and regionally explicit studies investigating the exploitation status of stocks are rare or absent. However, such studies should be of particular importance when considering potential fleet expansions, and in national fisheries management.

This study describes the small scale driftnet fisheries in Zanzibar, as well as the length-composition of the five most abundant species, in terms of landings: Indo-Pacific sailfish (IPS - *Istiophorus platypterus*, Shaw and Nodder, 1792; Swahili: Mbasi), Yellowfin tuna (YFT -*Thunnus albacares*, Bonnaterre, 1788; Swahili: Jodari), Common dolphinfish (CPH - *Coryphaena hippurus*, Linnaeus, 1758; Swahili: Panje), Skipjack tuna (SKJ - *Katsuwonis pelamis*, Linnaeus, 1758; Swahili: Jodari), and Kawakawa (KAW - *Euthynnus affinis*, Cantor, 1849; Swahili: Jodari). The LFDs of these species are analysed in regards to differences between SSF and IF, and stock status indicators estimated. Results are discussed in the light of the application of this type of information and its implications for the management of driftnet fisheries in Zanzibar. All collected data are also provided as open source.

Methods

Sampling area and routine

Fisheries-dependent data for small-scale driftnet fisheries in Zanzibar were collected at the landing sites of Nungwi, Mkokotoni, and Fukuchani in the northern district of Unguja, Zanzibar (Fig. 1). These data were collected over seven to fourteen days each month, between October 2014 and March 2015, corresponding to the northeast monsoon season in the Western Indian Ocean. Unguja is the most important island for driftnet fisheries in Zanzibar during the northeast monsoon

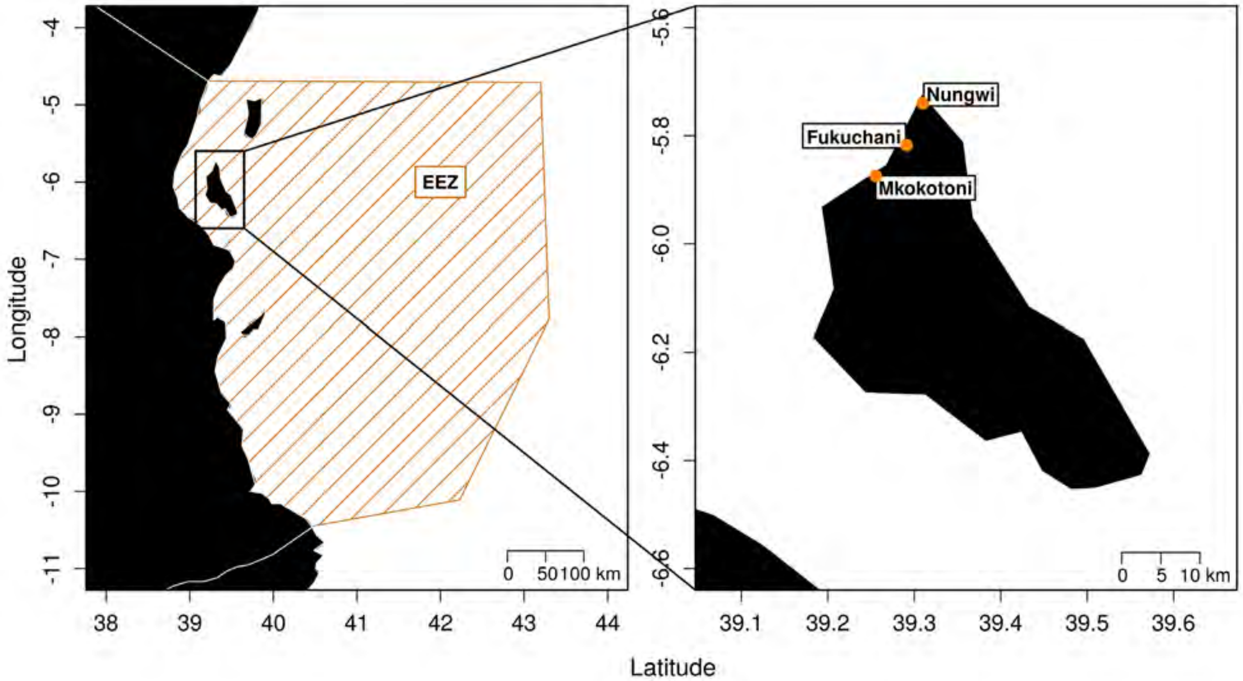

Figure 1. Map of the coastline and Exclusive Economic Zone (EEZ; hatched orange polygon) of Tanzania and close up of Unguja, the southern island of Zanzibar with the main sampling sites: Nungwi, Fukuchani, and Mkokotoni.

season, during which large-pelagic fish landings are highest (based on preliminary samplings and interviews with fishermen). Sampling effort was adjusted to the moon-dependent dynamics of fishing activities (no fishing effort for one week around full moon).

During sampling, fork length (FL: fish length from the front to the fork in the centre of the tail) of all fish landed by the driftnet fishermen was measured with a flexible tape, to the nearest 0.5 cm. In the case of sailfish, in addition to FL, the lower jaw fork length (LJFL) and the length from behind the eye to the fork (EFL) were measured (Cerdenares-Ladrón De Guevara *et al.*, 2011), allowing the inference of LJFL for individuals where the rostrum and the lower jaw was broken, damaged or missing (FL = -2.07 + 0.75 EFL and LJFL = -2.39 + 0.88 EFL). Between 14 and 940 fish lengths of the SSF were measured each month (Table 1). FL LJFL, and EFL are available on GitHub (see supporting online information). Maturity states and gender were not determined due to time limitations at auctions.

Length measurements from the IF were available for YFT and SKJ for the Tanzanian EEZ for the

Table 1. Number of fish measured for the small-scale fisheries (SSF) and industrial fisheries (IF) by month.

Species	Fisheries	Oct 2014	Nov 2014	Dec 2014	Jan 2015	Feb 2015	March 2015	Total
YFT	SSF	118	437	940	413	324	126	2358
	IF	132	204	2065	699	56	69205	72361
SKJ	SSF	151	354	523	110	135	82	1355
	IF						329686	329686
IPS	SSF	14	127	244	109	182	129	805
KAW	SSF	42	369	347	284	276	165	1483
CPH	SSF	20	151	467	121	365	433	1557

period from April 2014 to September 2015, and were retrieved from the corresponding IOTC working parties (Table 1; see supporting online information). IOTC compiles length-composition data of the catches collected by observers on board the fishing vessels, where length measurements are collected following two sampling strategies; stratified sampling or proportional sampling for multispecies fisheries, (IOTC, 2010). In both cases, 50 to 200 fish per haul are measured but the total number of measured fish is dependent on the size of fish, if catch consisted of one or multiple schools, and if a mode appears in the length frequency data. Observers record the FL for tuna and the LJFL for billfish to the nearest 1 cm using either large callipers (1.5 m), a measuring board, and/or flexible tape (IOTC, 2010). The IF length data covers the same period as the SSF data, but also provides data for the southeast monsoon season, and is therefore representative of a full meteorological year. The two main gears of the IF in Tanzania's EEZ are long lines (under Tawianese and Japanese flags) and purse seines (under French and Spanish flags) (Deep-sea Fishing Authority, pers. comm.). For IPS, KAW, and CPH, no length measurements from the IF were available, and our analysis was therefore focused on the SSF data only.

Data analysis

Length measurements were pooled over all gears into 2 to 4 cm length classes and compiled into LFDs for each species. The LFDs were compared between species and fisheries and used to construct length-converted catch curves, for the purpose of estimating the instantaneous total mortality rate (Z) from the slope of the regression of the descending part of the curve (Edeser, 1908; Pauly, 1983). The construction of the catch curves based on LFDs requires information about growth parameters to estimate the relative age of the individuals. Here, growth parameters (L_∞ and K) of the von Bertlanffy growth equation (VBGE) (von Bertalanffy, 1934; 1938) were gathered from other studies for the catch curves estimates (Table 3). The same growth parameters were used for data from the different fisheries. Growth parameters from nearby regions and the Indian Ocean were preferred, as well as parameters estimated with a single-stanza VBGE and age-based methods representative for both sexes. However, for SKJ and CPH, parameters derived by length-based methods were used (Pó *et al.*, 1992; CMFRI, 2016) and for IPS values representative of the sex-specific growth curves for the male fish were used (Hoolihan, 2006). Since all species are highly migratory (Kaplan *et al.*, 2014; Ward *et al.*, 1997), the parameters from chosen studies and regions were deemed to be representative.

Based on the growth parameters, the instantaneous natural mortality rate (M) can be approximated by means of the empirical formula of Then *et al.* (2015):

$$M = 4.118K^{0.73} L_\infty^{-0.33}$$

where K and L_∞ are the growth parameters of the VBGE. By subtraction of M from Z, an estimate of the instantaneous fishing mortality rate (F) can be approximated and an indicator of the exploitation rate be estimated (E = F / Z).

All analyses were done in R (R Core Team, 2018) using the TropFishR package (Mildenberger *et al.*, 2017b) and the following additional packages, gdata (Warnes *et al.*, 2017), SDMTools (VanDerWal *et al.*, 2014), reshape2 (Wickham, 2007), sp (Pebesma and Bivand, 2005; Bivand *et al.*, 2013), adehabitatHR (Calenge, 2006), maps (Brownrigg, 2018), and rgdal (Bivand *et al.*, 2018).

Results

The main fishing grounds of the driftnet fisheries in Zanzibar are located between the northern island of Pemba and the southern island of Unguja, as well as in the Zanzibar channel (between Unguja and Pemba and the mainland; Fig. 1). The maximum distance to the mainland shore and the islands was estimated by means of GPS trackers to be ~37 km. More than 28 different species were caught by the driftnet fishermen during the study period; however, the five selected species (YFT, SKJ, IPS, KAW, CPH) comprise 88.6% of the catch of the driftnet fisheries in Zanzibar (Table 2).

Seven to 14 pieces of black, blue, or grey gillnet (100 m x 15 m), either hand-made or industrially fabricated, are used in the driftnet fisheries. These are bound together, reaching a total length of around 1 km. Mesh sizes range from 3 to 6 inch (7.6 cm to 15.2 cm), and nets are often combined indiscriminately. The gillnets are deployed at the surface reaching depths of up to 15 m. Some fishermen also used handlines on the way to or from the fishing grounds, which influences the species- and length-composition of the landings. The fisheries are carried out exclusively at night, and results from interviews indicate that nets are deployed for around 7.7 ± 2 h per fishing trip. For about a week around full moon, no fishing takes

Table 2. Catch composition, total landings (kg) and percentage contribution to landings (%) for North A district over whole study period. The table excludes other species with a percentage contribution less than 0.04% (e.g. *Carangidae spp* and sharks). The shaded rows represent the five most abundant species in the landings of the small-scale driftnet fisheries, which were selected for further length-based analyses in the present study.

Species	Landings	Percentage
Indo-Pacific sailfish (*Istiophorus platypterus*)	57207	36.8
Yellowfin Tuna (*Thunnus albacares*)	34663	22.3
Common Dolphinfish (*Coryphaena hippurus*)	22427	14.4
Skipjack Tuna (*Katsuwonus pelamis*)	13124	8.4
Kawakawa (*Euthyunnus affinis*)	10411	6.7
Black marlin (*Istiompax indica*)	7937	5.1
Cobia (*Rachycentron canadum*)	1745	1.1
Longtail tuna (*Thunnus tonggol*)	1544	1
Wahoo (*Acanthocybium solandri*)	1340	0.9
Narrow-barred spanish mackerel (*Scomberomorus commerson*)	1326	0.9
Striped marlin (*Kajikia audax*)	1026	0.7
Blue marlin (*Makaira nigricans*)	791	0.5
Swordfish (*Xiphias gladius*)	673	0.4
Kingfish (*Scomberomorus plurilineatus*)	538	0.3
Frigate tuna (*Auxis thazard*)	535	0.3
Striped bonito (*Sarda orientalis*)	46	<0.05

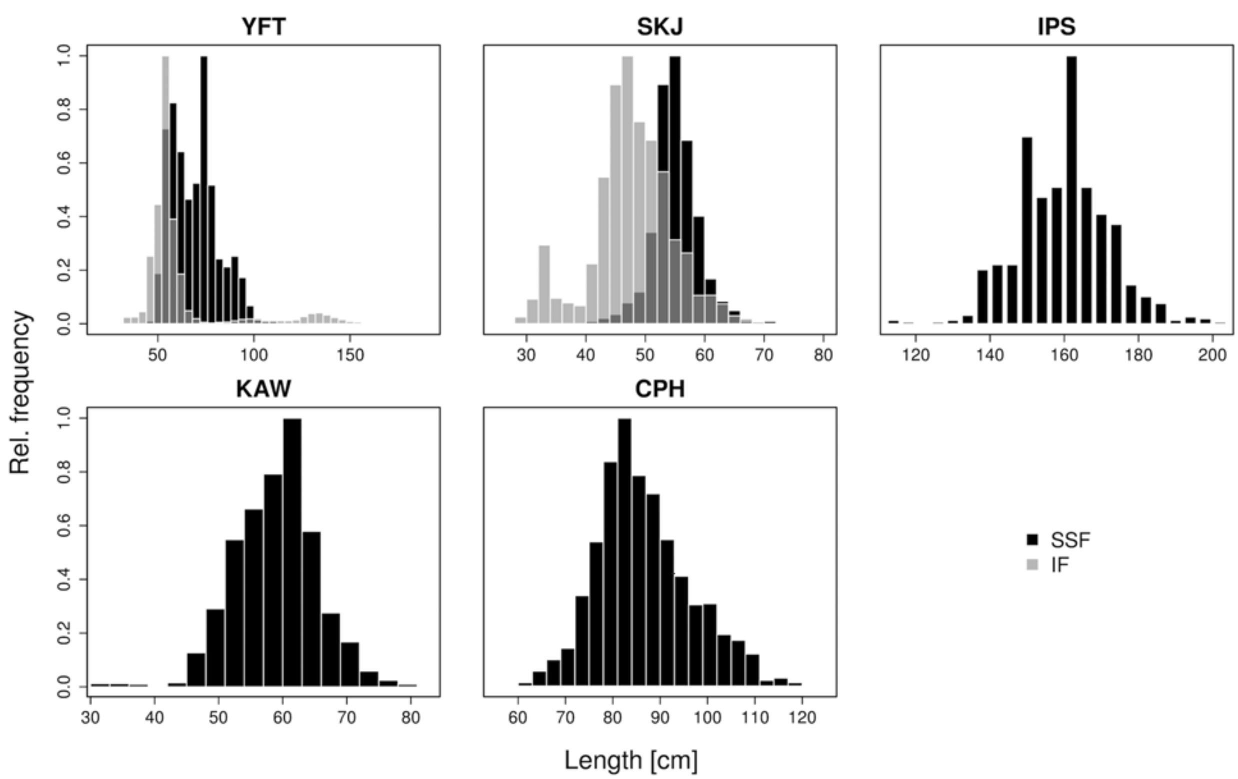

Figure 2. Length-frequency distributions (LFDs) of SSF landings for all species (black bars): Yellowfin tuna (YFT), Skipjack tuna (SKJ), Indo-Pacific sailfish (IPS), Kawakawa (KAW), and Common dolphinfish (CPH). Grey bars represent the LFDs of the IF landings for YFT and SKJ.

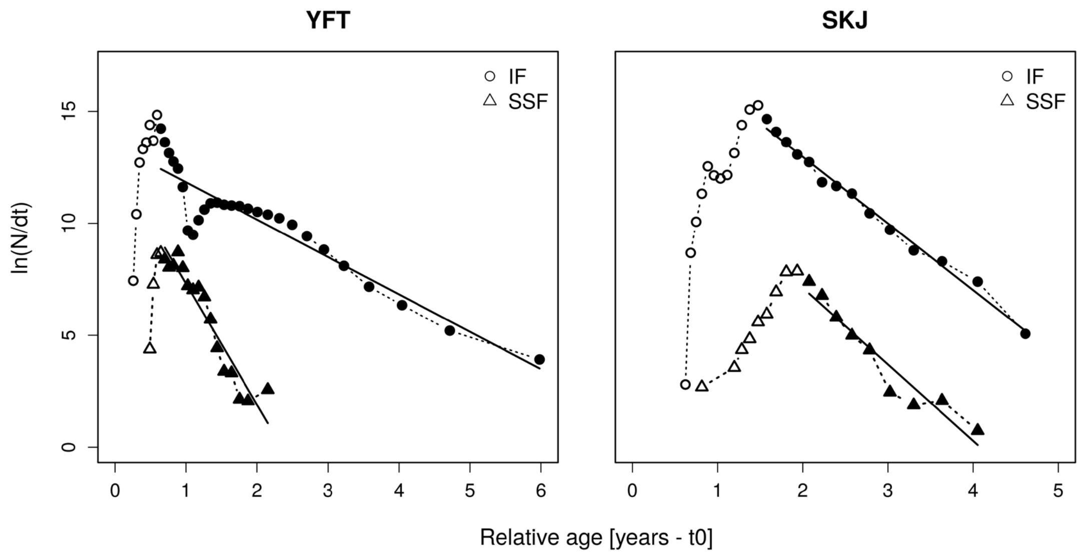

Figure 3. Length-converted catch curves for Yellowfin tuna (YFT) and Skipjack tuna (SKJ) based on data from the SSF (triangles) and from the IF (circles). Filled triangles and circles represent the data used for the regression line for both fisheries, respectively.

place, and shortly before and after full moon fishermen deploy the net for a shorter time (1.5 to 3 h).

While the length range of IF landings of YFT range from 25 to 199 cm, the SFF landed individuals between 45 and 143 cm during the study period. The length range differences between both fisheries for SKJ were less, with a range from 21 to 77 cm for IF and 32 to 79 cm for SFF. The length range for IPS, KAW, and CPH was from 118 to 200 cm, 34 to 80 cm, and 58 to 122 cm, respectively. The length composition of the landings for all species is displayed in Fig. 2.

While for SKJ, IPS, KAW, and CPH the LFDs of SFF shows a unimodal distribution with peaks around 56, 160, 60, and 84 cm, respectively, the distribution for YFT indicates three peaks at 55, 78, and 92 cm. The monthly LFDs indicate potential cohorts more clearly for YFT (Fig. 5a), while for the remaining species no multiple peaks and potential cohorts are visible (Fig. 5a and 5b). Comparing SSF and IF landings of YFT reveals that IF catch mainly smaller individuals of ~50 cm, and large individuals ~130 cm. IF landings for SKJ exhibit two peaks at 32 and 46 cm, below that of the SSF.

Overall, the catch curves show a clear pattern with a good representation of the descending part of the catch curves by the regression lines (Fig. 3 and 4). The high adjusted R^2 values (Table 3) reveal a good fit of the regression lines to the data. Estimates of the total

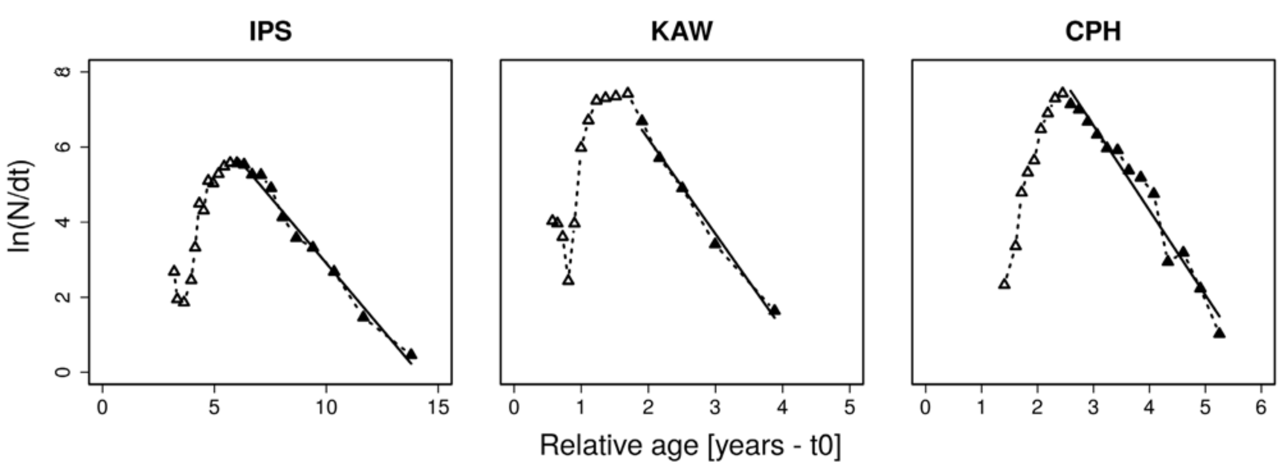

Figure 4. Length-converted catch curves for Indo-Pacific sailfish (IPS), Kawakawa (KAW), and Common dolphinfish (CPH) based on data from the SSF. Filled triangles represent the data used for the regression line for SSF.

mortality (slope of the regression line) have a wide range from 0.71 to 5.41, and differ between the five species sampled from the SSF. IPS demonstrated the lowest values, and SKJ the highest (Table 3).

Catch curves of KAW and CPH span a similar length range and show a similar slope of the regression lines of 2.53 and 2.26, respectively. A very different and much lower slope of 0.71 was found for IPS (Fig. 4 and Table 3). Overall, the descending part of the catch curves are well represented by the regression line with high adjusted R^2 values of 0.96 to 0.99 for those three species (Table 3).

Natural mortality estimates range from 0.3 and 0.36 for IPS and CPH, to 0.89 for KAW (Table 3). Fishing mortality is highest for YFT and lowest for IPS. For all species, the E values are above the reference level of 0.5, indicating overfishing (Gulland, 1983). While for IPS, E is only slightly larger than the reference level (E=0.58) for SKJ and CPH, E is with ~0.8 much higher than the reference level.

The Z estimates from the SSF were significantly higher than for the IF, with values of 5.41 and 3.42 vs 1.67 and 2.98 for YFT and SKJ, respectively. While the regression line of the catch curve for YFT starts at a similar relative age in both fisheries, the curve for IF is above the curve of SSF due to the higher number of samples. Furthermore, the slope is far steeper for the SSF than for the IF, but the exploited age range much shorter. As demonstrated in Fig. 3, a single regression line is a poor representation of the landings for the IF over the whole size range. Between relative ages of 0.6 to 1.3 the decline is very high, followed by an increase and second lower decline from ages 1.7 to 6. This is also reflected by a lower overall R^2 value of the regression line of 0.86. For SKJ, the catch curves of IF and SSF show a similar pattern (slope of decline and length range of declining part of the curve), however, there are more samples from the IF and the resulting curve is shifted vertically (Fig. 3). Similar to the Z estimates, the F estimates of the SSF are larger than for the IF. E values for SKJ are 0.8 and 0.78, almost identical between SSF and IF, respectively. However, for YFT, E is much higher for SSF than for IF (Table 3).

Discussion

This study describes SSF data from driftnet fisheries in Zanzibar, which target tuna and tuna-like species. Length-composition data were collected from SSF, and acquired for IF, for vessels operating in Tanzania's EEZ. These data were used to: (i) make a qualitative and quantitative comparison of the length composition of landings from both fisheries for the most abundant large-pelagic species; and (ii) for the estimation of stock indicators.

The length ranges of the landings highlight the selective nature of the gillnets used by the SSF, capturing only one cohort for all species. The exception to this was for YFT, where the monthly LFDs indicate three potential YFT cohorts, an observation which could not be determined from the yearly LFDs. SSF and IF operate in different parts of Tanzania's EEZ, due to the capacity of the vessels and jurisdictive/political boundaries (no distant water fleets are allowed in territorial waters). However, this study demonstrated that the two fisheries exploit the same resources of tuna and tuna-like species and the age ranges of these species overlap in the landings. Overall, IF catch a wider length range of fish, which can be attributed to the difference in gear type and fishery-dependent factors, as well as access of the IF to offshore fishing

Table 3. Literature parameters of the von Bertalanffy growth equation used for the estimation of the catch curve. Z represents the slope of the catch curve, however this might also include migration and is thus not referring to total mortality only (see discussion). The adjusted R^2 value is a goodness of fit statistic for the regression line of the catch curve analysis. The two values for Z, R^2, M, F, and E represent the estimates based on the data from the SSF and IF, respectively. *The set of growth parameters for the male individuals of common dolphinfish of the sex-specific growth analysis were used (Hoolihan, 2016).

Species	L_∞	K	t_0	Reference	Location	Z	R^2	M	F	E
YFT	165	0.878	-0.49	Nurdin *et al.* (2016)	Indian Ocean	5.41/1.67	0.9/0.86	0.57	4.84/1.1	0.89/0.66
SKJ	80	0.601	-	Pó *et al.* (1992)	Mozambique	3.42/2.98	0.92/0.99	0.67	2.75/2.31	0.8/0.78
IPS*	191	0.29	-4.31	Hoolihan (2006)	Arabic gulf	0.71	0.99	0.3	0.41	0.58
KAW	79	0.89	-0.08	CMFRI (2012)	India	2.53	0.98	0.89	1.64	0.65
CPH	146	0.34	-	CMFRI (2016)	India	2.26	0.96	0.36	1.9	0.84

grounds. Although the youngest YFT cohort might be the same in SSF and IF landings, the IF landings were observed to contain a much higher number of smaller individuals. For SKJ, IF landings demonstrated a second, younger cohort, which was not represented in SSF landings. Although the (length-converted) catch curve analysis is an often used assessment tool to derive estimates of the instantaneous total mortality rate and exploitation rates in data-limited environments, the findings in this study suggest that the slope of the regression line of the catch curve analysis does not just reflect the sum of natural and fishing mortality rates. This rate might also be influenced by migration processes (emigration and immigration to fishing grounds), as well as gear and fishery-dependent aspects, such as gear selectivity. This method can thus not be used to directly infer information about the instantaneous total, or fishing mortality rates, of the stocks directly, but it allows for the comparison of the slopes between different fisheries/fleets, and the estimation of exploitation rates. As the nominator F and denominator Z are affected by migration processes to the same extent, this bias cancels out when calculating their ratio (E). Furthermore, under the assumption that natural mortality and migration do not differ spatially within the EEZ of Tanzania, the only other factor affecting the slope of the regression line is fishing mortality, apart from gear selectivity effects. Based on this reasoning, it may be assumed that the fishing mortality for skipjack tuna is comparable between the two fisheries, while it differs greatly between fisheries for yellowfin tuna. From the deviations in the descending part of the catch curve for YFT and IF (filled circles in Fig. 3) it may be inferred that F is relatively high for individuals younger than a relative age of 1.3, and much lower for individuals older than a relative age of 1.7. This trend in the catch curve can either be related to the different fleets (and thus gears) in the IF data (purse seiners vs. long-liners), or due to different mortality and migration regimes for juveniles and sub-adults. Tuna and tuna-like species have been shown to form size-dependent (mixed) schools (Broadhead and Orange, 1960), which display specific migratory behaviour (Hu *et al.*, 2018). The catchability of SSF is much lower than of IF, as it uses a passive, highly selective gear, only accessing the upper 15 m of the water column (size dependent schooling in different depths), while the IF uses active gears (purse seines) and passive gears (long lines), bird radar, helicopters and Fish Aggregation Devices (FADs; Majkowski, 2007; Tidd *et al.*, 2017). This allows IF to sustain relatively high landings even when stock numbers are declining (Gulland, 1956; Tidd *et al.*, 2016). The SSF are limited in geographic range due to the size of their vessels, with many vessels operating without an engine and dependent on the wind.

The results show a high exploitation rate above reference levels, indicating overfishing in terms of fishing mortality for all species. No conclusion can be drawn regarding the stock status in terms of biomass. However, the estimated overfishing status corroborates official IOTC assessment results for YFT and IPS. IOTC classifies these species as overfished regarding fishing mortality (IPS), and fishing mortality and biomass (YFT; IOTC, 2017). The higher exploitation rate for YFT might reflect the fact that YFT is overfished in terms of F and biomass, while IPS only in terms of F, according to the IOTC. The smaller tuna species, SKJ and KAW are not classified as overfished according to the IOTC, but show high exploitation rates in this study. It is suggested that this may be due to the sub-population structures (Fonteneau, 2014) and local depletion of these stocks. While the official IOTC assessments consider data from the whole Indian Ocean for these species, only data from Tanzania's EEZ were considered in this study, and both data sources (SSF and IF) indicate high exploitation rates of 0.78 and 0.8, respectively. There are no official stock status estimates for CPH in the Indian Ocean or the region (Maguire *et al.*, 2006; IOTC, 2017). Although Benjamin and Kurup (2012) estimated a low exploitation rate for CPH between 2008-2009 in India, the findings from this study indicate that the exploitation rate of CPH in Tanzania might exceed biologically safe limits. The contrasting results between studies are likely explained by temporal or spatial differences in study period and location. Length-at-first-maturity (L_{50}) values of the studied species, SKJ: 41-43 cm, YFT: 100 cm, KAW: 38-50 cm, (IOTC, 2017), IPS: 157 cm EFL (Hernandez and Ramirez, 1998), and CPH: 47-49 cm (Rajesh *et al.*, 2016), show that 99% of the SSF length samples are smaller than L_{50} for YFT and 37% smaller for IPS, while the percentage for remaining species is close to 0%. This indicates that a large proportion of the tuna catches in Tanzania are juveniles (with 90% for IF YFT landings), which might reflect overfishing (Froese *et al.*, 2008; Myers and Mertz, 1998), the spatio-temporal occurrence of different length classes in the region, the gear selectivity, and a combination of these factors. Tanzania's EEZ forms an important section of the migration route of tuna (Hallier and Fonteneau, 2015), but is also part of the highly productive western Indian Ocean region (Bakun *et al.*, 1998;

Qasim, 1977) and therefore represents suitable habitat for the residence of juvenile tuna (Hu *et al.*, 2018).

Although the growth parameters used for the catch curve analysis were taken from regional studies, they allow for the comparison of the two fisheries (SSF and IF) as the same parameters were used. However, it should be noted that the growth of YFT could differ significantly between sexes (Shih *et al.*, 2014) and be better described by a gender-based two-stanza von Bertalanffy growth function (Dortel *et al.*, 2013). Furthermore, the results for SSF are based on data from one season only, and the variation in length-composition of the landings between seasons cannot determine. As with all fishery-dependent length-frequency data, results may be subject to biases due to recruitment variability, gear selectivity, and un-representative length measurements for the whole population (Punt *et al.*, 2001; Cope and Punt, 2009).

The management of straddling stocks has often been described as unsuccessful (Cullis-Suzuki and Pauly, 2010), although positive examples have also been acknowledged (Pons *et al.*, 2017). While spatial measures are prone to fail for highly-migratory species (Kaplan, 2013), input and output control measures might be more suitable. In any case, this study has demonstrated that IFs and SSFs targeting tuna and tuna-like species are interrelated, and must be managed in unison, a conclusion that was also shared by Leroy *et al.* (2016). Another challenge for management is the fact that two of the most abundant species in the SSF landings (KAW and CPH) are by-catch species in the IF (purse seiners (Ardill *et al.*, 2011), long liners (Huang and Liu, 2010), and any fishery associated with FADs (Dagorn *et al.*, 2013) and have an important market in Zanzibar, being sold to hotels and restaurants (Thyresson *et al.*, 2013). Accordingly, these species are important for the driftnet fisheries in Zanzibar, and are caught as by-catch by industrial. By-catch ratios greatly vary between areas and type of fisheries, but discard ratios for those fisheries can be as high as 60% (Dagorn *et al.*, 2013). The impact of sports and recreational fishermen should also be considered in the management of straddling stocks, particularly in light of growing tourism in Tanzania (MNRT, 2012).

Lastly, another crucial aspect of sustainable management concerns the monitoring of fishing activities and data collection, as this information is required for stock assessments and thus the definition of harvest control rules. The data collection on SSF must be improved and included in the official assessments by RFMOs. With regard to the monitoring strategies and procedures in Zanzibar, the following suggestions might improve the quality and value of fisheries-dependent data: (i) catch data should be collected at species level, as the stock status of different species can be contrasting (e.g. YFT and SKJ; IOTC, 2017). Experience and interviews with local observers show that the problem is not the lack of knowledge of species identification of the observers, but the fact that this type of information is not required by the official agencies, or is lost in the administrative process. The quality of catch data can also be improved by monitoring the work of beach recorders; (ii) Effort data should be collected by beach recorders more frequently than the national survey (e.g. at a monthly level); (iii) Data representing the catch and effort of recreational fishing are needed and should be implemented by an obligatory observer monitoring procedure for companies renting boats and organising trips for recreational fishing; (iv) The collection of a subsample of length composition data should be considered. This type of data allows the inference of information about gear selectivity and exploited size ranges of SSF, and might be used directly in length-based stock assessment models such as SS3 (Methot and Wetzel, 2013), and used for official assessments such as for YFT and SKJ (IOTC, 2017). In particular, for by-catch species of the IF (KAW and CPH), LFDs from SSF are valuable and even allow inferences to be made regarding stock status, where no data from IF are available. For example, newly developed methods (e.g. Schwamborn *et al.*, 2018) are being used to derive reference levels to quantify uncertainty in catch data (Herrón *et al.*, 2018), and could be applied to species such as CPH. As this study shows, length measurements are easy and cost-efficient to collect, even without the necessity of owning or damaging sampled fish; and (v) lastly, the use of a mobile application, such as ABALOBI (http://abalobi.info) poses great potential for simplifying and standardising data collection.

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Appendix

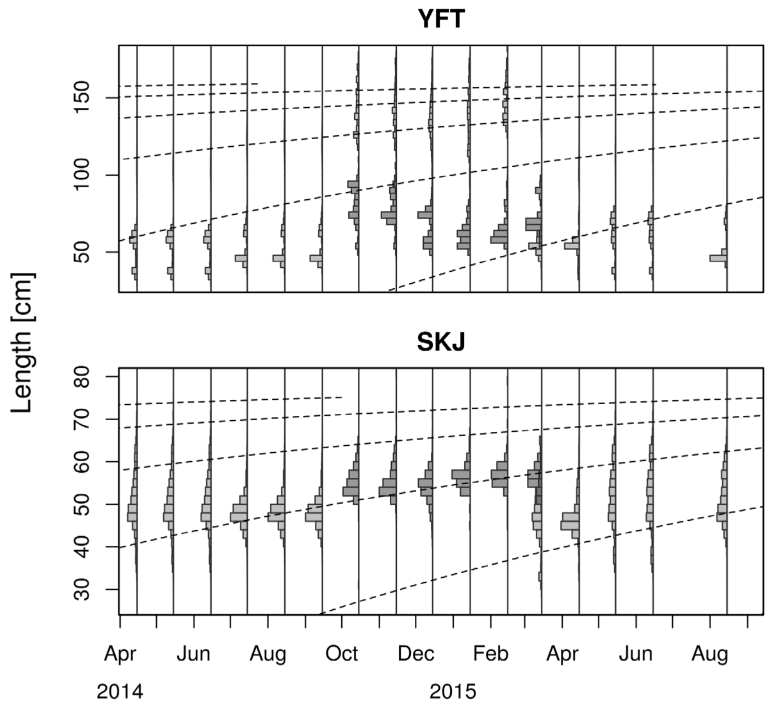

Figure 5a. Monthly length-frequency distributions (LFDs) of SSF and IF landings combined for Yellowfin tuna (YFT) and Skipjack tuna (SKJ) with von Bertalanffy growth curves. Dark grey bars represent the LFDs of the SSF while light grey bars represent the IF landings.

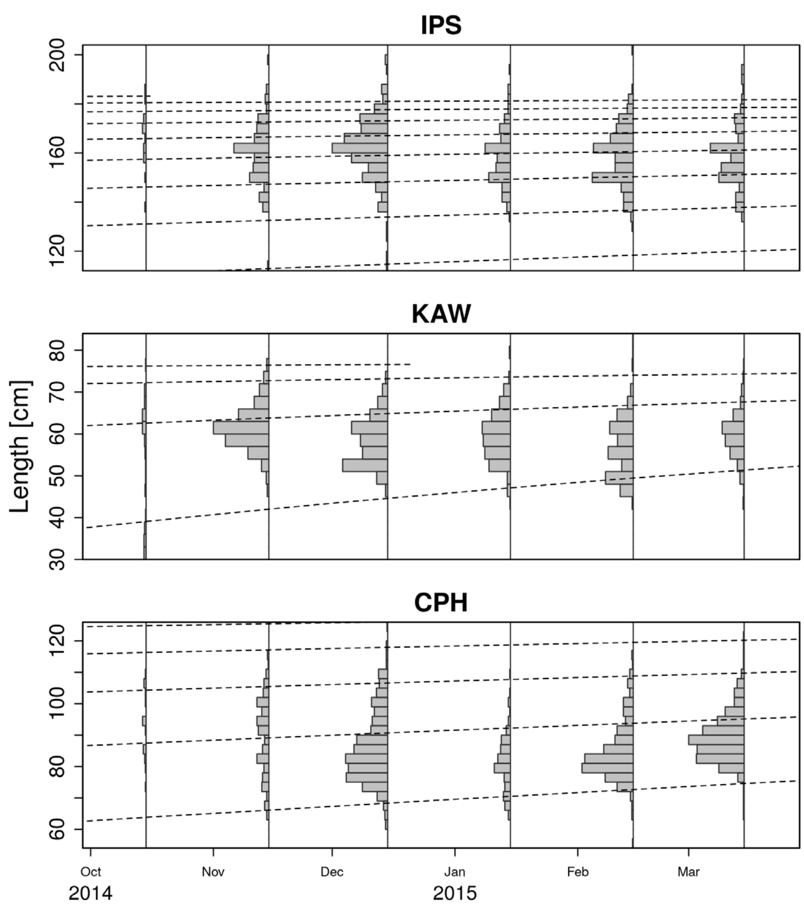

Figure 5b. Monthly length-frequency distributions (LFDs) of SSF landings for Indo-Pacific sailfish (IPS), Kawakawa (KAW), and Common dolphinfish (CPH) with von Bertalanffy growth curves.

Online supporting material

The length-composition data from IF was retrieved from the respective IOTC working groups (i) Neritic tunas (http://www.iotc.org/meetings/7th-working-party-neritic-tunas-wpnt07, accessed: 21/10/2017 1:30pm), (ii) Tropical tunas (http://www.iotc.org/meetings/19th-working-party-tropical-tunas-wptt19, accessed 21/10/2017 1:30pm), and (iii) Billfish (http://www.iotc.org/meetings/15th-working-party-billfish-wpb15, accessed: 21/10/2017 1:30pm). All collected length measurements for the five species: Yellowfin tuna, Skipjack tuna, Indo-Pacific sailfish, Kawakawa, and Common dolphinfish are available on GitHub at https://github.com/tokami/Jodari.

Previously unlisted scleractinian species recorded from the Great Reef of Toliara, southwest Madagascar

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Abstract

The scleractinian biodiversity of Madagascar is mainly known from one study performed in the Bay of Toliara (SW of Madagascar) in the 1970s. In the present study, this biodiversity was re-investigated 40 years later, at 2 sites previously considered as atypical, but now subject to high anthropogenic pressures. Results showed lower species diversity compared to the previous study, and to similar sites in the Indian Ocean region, but most of the well-represented genera were recorded. The occurrence of previously unrecorded species suggests that the scleractinian communities are changing, in addition to declining. The findings of the present study constitute a baseline of scleractinian structure studies, focused on diversity change. Further investigations on this reef must consider these changes, and management measures must be adapted to ensure greater efficiency.

Keywords: newly recorded species, scleractinians, biodiversity, Great Reef of Toliara, southwest Madagascar

Introduction

Coral reefs of Madagascar occupy an area of 2,400 km² along 1,400 km of the coastline (Cook *et al*., 2000). They are particularly well distributed along the southwest coast. The reef system is dominated by the Great Reef of Toliara (GRT), while most of the coral reef types (barrier reef, fringing reef, patch reef, coral bank) are recorded (Clausade *et al*., 1972). The GRT is among the most studied reefs of the Indian Ocean, especially in the 1960s and 70s (Pichon, 1978). It constitutes a refuge for diverse marine flora and fauna communities with more than 6,000 species recorded (ONE, 2002), including reef fishes (714 species; Harmelin-Vivien, 1979; Rasoarimalala, 2001), and benthic organisms such as sponges (125 species; Vacelet and Vasseur, 1971) and scleractinians (112 species; Pichon, 1978; Sheppard, 1998).

It is well known that the coral reefs of Madagascar, especially those in the southwest, are subject to intense social-ecological impacts that have caused unprecedented change. In the period of 50 years, the coral cover has decreased from >50% to around 5% (Harris *et al*., 2010; Bruggemann *et al*., 2012), reef fish species diversity has dropped to less than 30% (Ranaivomanana, 2006), while anthropogenic pressures, including destructive fishing, continues to increase due to the increasing number of fishing communities (Vasseur *et al*., 1988; Toany, 1995; Salimo, 1997; Vasseur, 1997). In the years 2006 – 2011, more than 50 locally managed marine reserves were created as a solution to coral reef degradation and the decline of fishery products (Voajanahary, 2011; Todinanahary, 2013). Initially, these locally managed marine areas (LMMA) were socially well perceived and accepted by fishing communities, who were the main users and the principal managers, and Mahafina (2011) reported some success and benefits generated by LMMAs. However, most of them were not properly managed, and their status as protected areas was often not respected since

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2014, except for few of them (e.g. Marine Reserve of Rose Garden, Ankaranjelita, Soariake and Velondriake) where there is strong support from non-governmental organizations (NGOs) (Rocliffe and Peabody, 2012; Belle *et al.*, 2009; Shane, 2012). The impacts of the establishment of these protected areas on the reef communities are poorly understood, especially for scleractinians of the GRT.

The study of Pichon (1978) is the only one that reported complete information about species richness and diversity of scleractinians in southwest Madagascar. Since then, new data about the reef building corals are very few (e.g. Sheppard, 1987; Sheppard, 1998; Harris *et al.*, 2010; Bruggemann, 2012). Sheppard (1998) updated the scleractinian biodiversity patterns in the Indian Ocean (including Madagascar, based on Pichon (1978)). This author analysed the effect of taxonomic error in data (including redundant synonyms and species marked as "spp"), and reported that the number of coral species on the GRT was only 112 belonging to 57 genera, rather than 135 (Sheppard, 1987; 1998). Harris *et al.* (2010) reported a loss in scleractinian diversity based on sampling in 2008, and noted that the once great barrier reef of Toliara was in serious decline. The present study presents the results of a survey on scleractinians of GRT in 2015, on a few of stations surveyed by Pichon (1978). The survey aimed at determining the present scleractinian biodiversity in the Bay of Toliara by comparing 2 sites of different geomorphological structure on the GRT, and compares the results with some of those documented previously.

Materials and methods

Area of study

Two distinct sites were chosen for the sampling during the biodiversity survey (Fig. 1). The first site was the "Grande Vasque" (GV)). The GV is a basin of about 1 km in diameter situated on the flat of the barrier reef. GV is well protected from the swell, around 15 m deep, and its slopes are colonized by scleractinians, mainly in the first 8 m. GV is located in front of the main harbour of the region, near Toliara city. Two stations were defined and sampled on the GV; one on its southern part (GV South), the other on its northern part (GV North). The second site of the study was Nosy Tafara. Nosy Tafara (NT) is a complex of patch reefs located on the southern tip of the Great Barrier Reef of Toliara. NT is exposed to the swell and the waves generated by the dominant SW wind. Two stations were also defined and sampled in NT: the outer slope of Arakaivo, exposed to the open sea, and Velomitahy, a station protected by Arakaivo. Both sites were chosen because of the existence of old and more recent data (e.g. Voajanahary, 2011; Mahafina, 2011; Bruggemann *et al.*, 2012; Andréfouët, 2013; Sheridan *et al.*, 2014a, 2014b; Todinanahary, 2013) and because they are among the most accessible sites in the Bay at any time of the year, and thus the most exposed to anthropogenic pressures.

Surveys and sampling

In the western Indian Ocean (WIO) region, the PRECOI ("Programme Régional Environnement de la Commission de l'Océan Indien") method is recommended for coral cover monitoring (Conand *et al.*, 1997). This method, based on a combination of transects and quadrats, has been widely used for coral reef studies in Madagascar, but was limited to category levels for coral identification (see details in Conand *et al.*, 1997). In the present study, the Point Intersept Transect (PIT) method was used (Hill and Wilkinson, 2004). Several coral reef research programmes have used the PIT method (e.g., Rogers *et al.*, 1994), recommended by English *et al.* (1997), and adapted by others to fit with regional aspects and research focus (Beenaerts and Berghe, 2005). It was chosen and adapted for its efficiency for coral species diversity monitoring (Beenaerts and Berghe, 2005).

At each station, 15 transects of 10 m were undertaken on the reef slope, at 8 to 15 m depth, by 3 to 4 divers between January and August 2015. The transect line was a flexible measuring tape, marked in millimetres. The line was kept close to the benthic communities using small weights. To allow the recording of small coral colonies (< 10 cm including juveniles which were abundant on the sites), the line was marked every 5 cm, and the sessile benthic organism or substrate directly beneath the mark was recorded. During the survey the common set of cover categories for the WIO (see details of categories in Conand *et al.*, 1997) were used. Live coral species were identified to genus level where possible, using the *in-situ* Coral finder identification guide (Kelley, 2011), followed by an in-lab skeletal morphology analysis based on the work of Veron (2000). All the observed colonies were photographed and two 2 cm to 5 cm branches were sampled for skeletal morphology analysis.

Calculation of ecological parameters

Coral species richness, species dominance and diversity were calculated for each station. Richness was

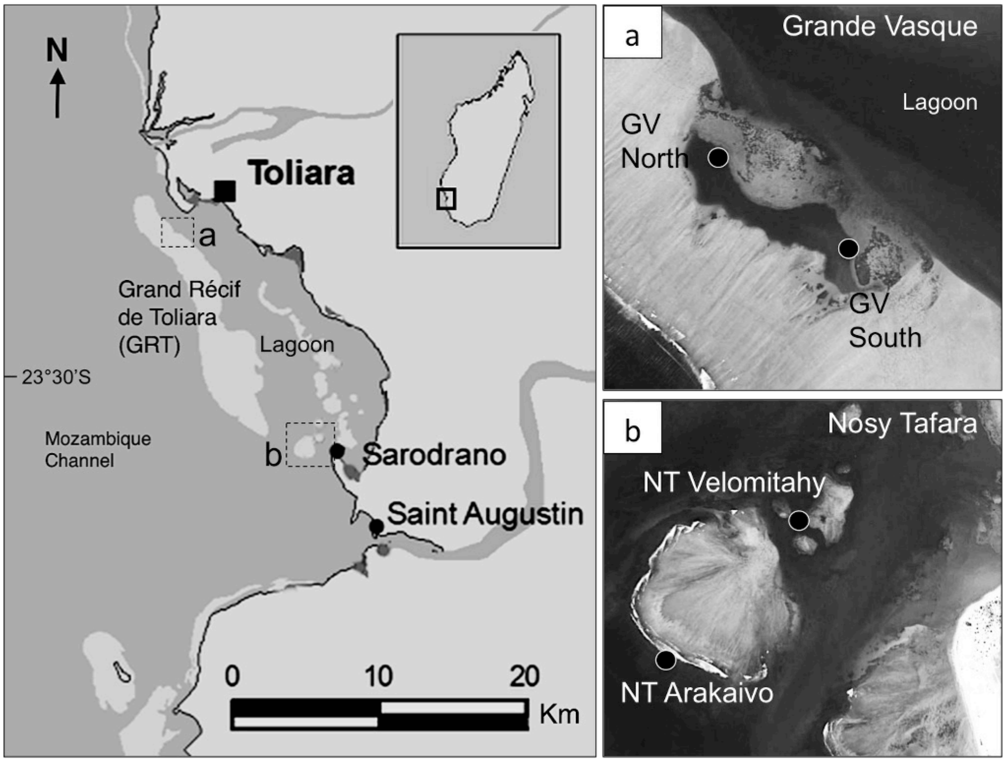

Figure 1. Locality of the sites and stations.

calculated as the total number of species under the transect line. Species dominance was calculated as the ratio of the abundance of each species and the total number of recorded colonies on the transect, reported as percentage. The Shannon diversity index (Shannon and Weaver, 1964) was calculated at the level of coral species.

To characterize the community at each station, the constancy and fidelity index of each species in the coral community (station) were also calculated. Constancy was calculated by dividing the number of records (transects) containing the species by the total number of records within the community. Fidelity was deduced by dividing the constancy of a species by the sum of the constancy of that species at all the stations as follows:

$$C_{A/1} = (R_A/R_1)*100$$
$$F_{A/i} = (C_{A/1} / \sum^{n}_{1} C_A)*100$$

where $C_{A/1}$: constancy of the species A at station 1
R_A: number of records of the species A
R_1: total number of records for the station 1
F_{Ai}: fidelity of the species 1 to the station i

The most characteristic species, and common or rare species, were identified for each station using the constancy and fidelity values on the basis of the following categories.

Constancy index:
75 % - 100 %: Constant species
50 % - 74.9 %: Common species
25 % - 49.9 %: Less common species
< 24.9 %: Rare species
Fidelity index:
75 % - 100 %: Selective species
50 % - 74.9 %: Preferential species
25 % - 49.9 %: Indifferent species
< 24.9 %: Occasional species

Statistical analysis

All statistical analyses were performed using the R software (R Core Team, 2015). Descriptive statistics were calculated first. Normality of the data was determined using a Shapiro-Wallis test, and homogeneity of the variance was calculated using Levene's test. For species richness analysis, data were transformation into log(x+1). Significance of difference in means were determined using one-way ANOVA, at a level of 5%. The Tukey multiple comparison test was used for pairwise comparison between stations. Principal component analysis (PCA) and hierarchical cluster dendrograms of species and stations were performed to characterize the distribution of the species and the similarity of the stations.

Results

Characterisation of the scleractinian communities

Richness and diversity of coral species

Species richness varied significantly from 4.2 ± 1.4 (mean ± SD) to 9.1 ± 2.2 (mean ± SD) at the studied stations ($p<0.001$). The lowest richness was observed at the GV site, while NT presented the highest values (significant difference between both sites, $p<0.001$). Arakaivo station had significantly higher species richness than the three other stations (Table 1 and 2), between which no significant difference was observed.

Similarly to richness patterns, Shannon diversity results show significantly higher diversity at NT as compared to GV ($p<0.001$). However, this difference was highly influenced by the station at Arakaivo, which had the highest and most significant diversity index, compared to than the three other stations, between which no significant difference was observed (Table 1 and 2). In addition, GV North and GV South showed no significant difference in richness and diversity (Table 2).

Recorded species: abundance, dominance and distribution

A non-exhaustive total of 36 species from 14 genera and 9 families were recorded at the 4 monitored stations (Table 3 and 4). Acroporidae was the most represented family with 14 species recorded, followed by Pocilloporidae with 9 species represented, and Poritidae with 4 species. Agaricidae, Oculidae and Fungidae were represented by 2 species each, and Favidae, Euphyllidae and Mussidae by 1 species each (Table 4). The overall dominance values placed *Porites rus* as the most dominant species (15.9%), followed by *Acropora robusta* (14.5), *Seriatopora hystrix* (7%), *Lithophyllon repanda* (6.3%) and *Acropora nasuta* (5.8%) (Fig. 2). These 5 species dominated 49.5% of the communities. However, the distribution of each species at the stations suggests that the dominance of *Porites rus* was due to its high dominance at GV South (53.7%), and that of *Acropora robusta* is due to its high dominance at GV North (43.5%), while the other species did not show obvious dominance at any station.

The principal component analysis (PCA) (Fig. 3) and the hierarchical cluster dendrogram of species and stations suggests that each station was mostly characterized by one to three species. Arakaivo had very different community species composition from the other stations. This station was characterized mostly by branching species such as *Acropora* and *Pocillopora*, which were, with *Echinopora gemmacea* and *Galaxea*

Table 1. Average specific richness and diversity at each station. SD: standard deviation

	Richness		Diversity	
	Mean	**SD**	**Mean**	**SD**
Arakaivo	9.067	2.017	0.891	0.097
Velomitahy	5.846	1.994	0.668	0.161
GV North	4.200	1.373	0.551	0.149
GV South	4.692	1.109	0.559	0.120
ANOVA (p-value)	< 0.001		< 0.001	

Table 2. Pairwise comparison between stations. Probability was calculated using the multiple comparison test of Tukey.

	Richness		Diversity	
Comparison	**t-value**	**p-value**	**t-value**	**p-value**
Arakaivo - Velomitahy	-4.246	< 0.001	-4.433	< 0.001
Arakaivo - GV North	7.270	< 0.001	6.993	< 0.001
Arakaivo - GV South	5.895	< 0.001	6.584	< 0.001
Velomitahy - GV North	2.760	0.0388	2.306	0.110
Velomitahy - GV South	1.594	0.3909	2.079	0.173
GV North - GV South	1.111	0.6846	0.154	0.999

Table 3. List of species recorded during the survey, total abundance, dominance, constancy and fidelity of each species. TA: total abundance (number of recorded individuals); D: dominance (in %); C: constancy (in %); F: fidelity (in %); OD: overall dominance (in %). Dark grey cells with bold font = C or F > 75% ; Grey cells = 50% < C or F < 74.99% ; Light grey cells = 25% < C or F < 49.99% ; White cells = C or F < 25%.

Species	TA	Arakaivo			Velomitahy			GV North			GV South		
		D	C	F	D	C	F	D	C	F	D	C	F
Acropora abrotanoides	5	2	20	**100**									
Acropora branchi	8	3	33.3	**100**									
Acropora clathrata	24	11.6	73.3	**100**									
Acropora cytherea	12	6.2	53.3	**100**									
Acropora digitifera	11	3.4	46.7	**77.8**				2.8	13.3	22.2			
Acropora divaricata	9	1.4	13.3	28.6	1.6	20	42.9	1.7	6.7	14.3	0.6	6.7	14.3
Acropora muricata	19	4	40	60	9.6	26.7	40						
Acropora latistella	6	2.4	33.3	**100**									
Acropora nasuta	45	6.4	60	36	4.7	33.3	20	9.1	40	24	2.9	33.3	20
Acropora robusta	83	5.4	20	13	8.1	26.7	17.4	43.5	**100**	65.2	1	6.7	4.3
Acropora retusa	12	4.9	26.7	57.1				1.4	20	42.9			
Acropora samoensis	13	5.4	53.3	**80**							1.5	13.3	20
Acropora tenuis	5	1.8	13.3	66.7				0.6	6.7	33.3			
Echinopora gemmacea	9	3.4	46.7	**77.8**				0.6	6.7	11.1	0.5	6.7	11.1
Lithophyllon repanda	35	0.6	13.3	9.1	5	40	27.3	15.7	73.3	50	3.7	20	13.6
Galaxea astreata	18				14.5	40	75	0.6	6.7	12.5	0.4	6.7	12.5
Galaxea fascicularis	7	2.8	33.3	**100**									
Goniastrea pectinata	2	0.6	6.7	50	0.4	6.7	50						
Herpolitha limax	8				1.1	20	42.9	4.4	26.7	57.1			
Lobophyllia corymbosa	7				2	13.3	50	1	6.7	25	1.7	6.7	25
Montipora undata	2	0.3	6.7	50	1.7	6.7	50						
Pavona cactus	40				14.9	53.3	72.7				1.9	20	27.3
Pavona clavus	11	0.3	6.7	25	2.2	6.7	25				4.1	13.3	50
Plerogyra sinuosa	29				13.9	60.0	64.3				5.9	33.3	35.7
Pocillopora damicormis	34	12	**93.3**	66.7	0.4	6.7	4.8	3	26.7	19	2.3	13.3	9.5
Pocillopora fungiformis	29	11.6	**80**	**100**									
Pocilopora grandis	8	1.9	20	75				1	6.7	25			
Pocilopora verrucosa	1	0.4	6.7	**100**									
Porites lutea	3	1.1	13.3	66.7							0.8	6.7	33.3
Porites profundus	16	0.4	6.7	9.1	1.7	6.7	9.1				7.8	60	**81.8**
Porites rus	94	1.4	20	11.1	5	40	22.2	3.7	20	11.1	53.7	**100**	55.6
Seriatopora caliendrum	3	0.6	6.7	33.3	1.2	13.3	66.7						
Seriatopora hystrix	49	4.2	53.3	25	8.1	60	28.1	5.8	33.3	15.6	9.9	66.7	31.3
Stylophora madagascarensis	3				1.3	13.3	66.7				1.3	6.7	33.3
Stylophora pistillata	6							3.7	20	**100**			
Stylophora subserata	8	0.6	6.7	16.7	2.6	20	50	1.7	13.3	33.3			

Table 4. Comparative list of genera and number of species of Scleractinia recorded on the coral reefs of the SW region of Madagascar and the present study (Genera in bold font). P: number of species recorded by Pichon (1978), T: number of species recorded by the authors at the selected stations. nr: not recorded.

Genera	P	T	Genera	P	T	Genera	P	T	Genera	P	T
Acropora	13	13	*Dendrophyllia*	2	nr	*Balanophyllia*	1	nr	*Merulina*	1	nr
Pavona	8	2	***Echinopora***	2	1	*Caryophillia*	1	nr	*Montastrea*	1	nr
Fungia	7	1	***Galaxea***	2	2	*Caulastrea*	1	nr	*Oxypora*	1	nr
Pocillopora	7	4	***Goniastrea***	2	1	*Culicia*	1	nr	*Paracyathus*	1	nr
Porites	7	3	*Goniopora*	2	nr	*Cycloseris*	1	nr	*Parascolymia?*	1	nr
Leptoseris	6	nr	*Hydnophora*	2	nr	*Cynarina*	1	nr	*Pectinia*	1	nr
Favites	5	nr	*Mycedium*	2	nr	*Diaseris*	1	nr	*Physogyra*	1	nr
Montipora	5	1	*Oulophyllia*	2	nr	*Diploastrea*	1	nr	*Platygyra*	1	nr
Turbinaria	5	nr	*Pachyseris*	2	nr	*Echinophyllia*	1	nr	*Plerogyra*	1	1
Favia	3	nr	*Platygyra*	2	nr	*Gyrosmilia*	1	nr	*Podabacia*	1	nr
Leptastrea	3	nr	*Plesiastrea*	2	nr	*Halomitra*	1	nr	*Polycyathus*	1	nr
Lobophyllia	3	1	*Turbastraea*	2	nr	***Herpolitha***	1	1	***Seriatopora***	1	2
Psammocora	3	nr	*Acanthastrea*	1	nr	*Heterocyathus*	1	nr	*Siderastrea*	1	nr
Stylophora	3	3	*Agariciclla*	1	nr	*Heteropsamia*	1	nr	*Sphenotrochus*	1	nr
Blastomussa	2	nr	*Alveopora*	1	nr	*Horastrea*	1	nr	*Stylocoeniella*	1	nr
Coscinarea	2	nr	*Anomastrea*	1	nr	*Leptoria*	1	nr	*Symphyllia*	1	nr
Cyphastrea	2	nr	*Astreopora*	1	nr	*Madracls*	1	nr	*Trachyphyllia*	1	nr

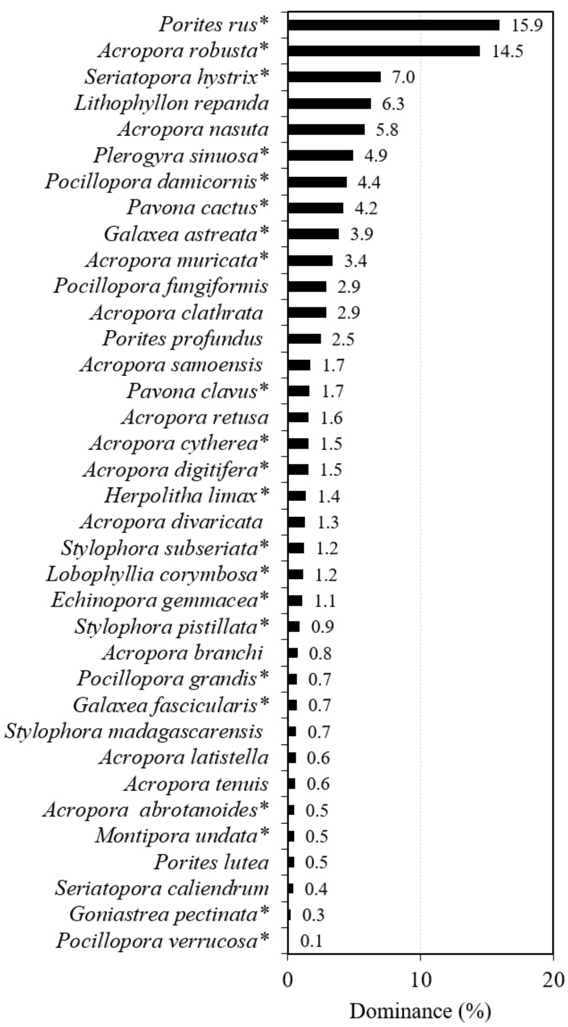

Figure 2. Total dominance of each species. Species followed by * were listed by Pichon (1978), including synonymized species name (WoRMS Editorial Board, 2016).

fascicularis, the less common but the most selective species (Fig. 4). Velomitahy was characterized by less common but indifferent species (Fig. 4a), particularly by *Pavona cactus*, *Plerogyra sinuosa* and *Galaxea astreata*. These species, with the others that are tolerant, are common to the region and recorded from at least 3 of the studied stations (Table 3). GV South was largely characterized by the species *Porites rus* whose dominance influences the whole community at this station. GV North presents a similar community as Velomitahy (Table 3, Fig. 4b). This station was particularly characterized by the free species *Lithophyllon repanda* and *Herpolita limax*, which are also indifferent, according to calculations.

Discussion

Thirty six species from 14 genera and 9 families were recorded at the 4 monitored stations. The scleractinian diversity was relatively low compared to similar studies in the WIO region (e.g. Sheppard, 1987; Beenaerts and Berghe, 2005; Obura, 2012), and especially the study of Pichon (1978) (corrected later by Sheppard, 1998) who observed 112 species belonging to 57 genera on the GRT. The results from the present study may be due to the smaller surface area sampled, compared to Pichon's (1978) study. Apart from the work of Pichon (1978), the only other study on the overall coral biodiversity of the GRT was carried out in the 2000s, and revealed the loss of from 8 to 18 coral

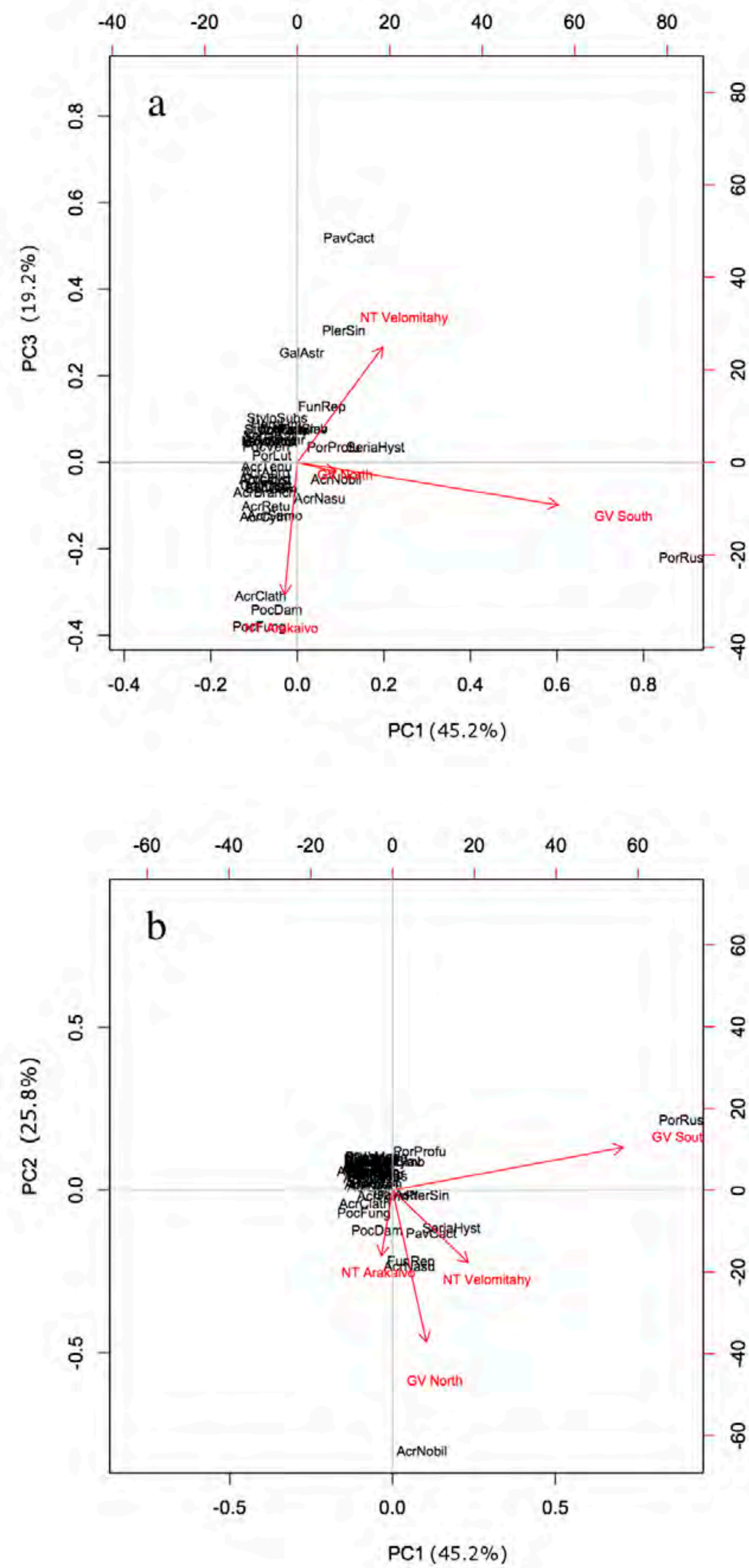

Figure 3. Principal component analysis of the stations and the species; a: according PCA1-PCA3 projection; b: according to the PCA1-PCA2 projection.

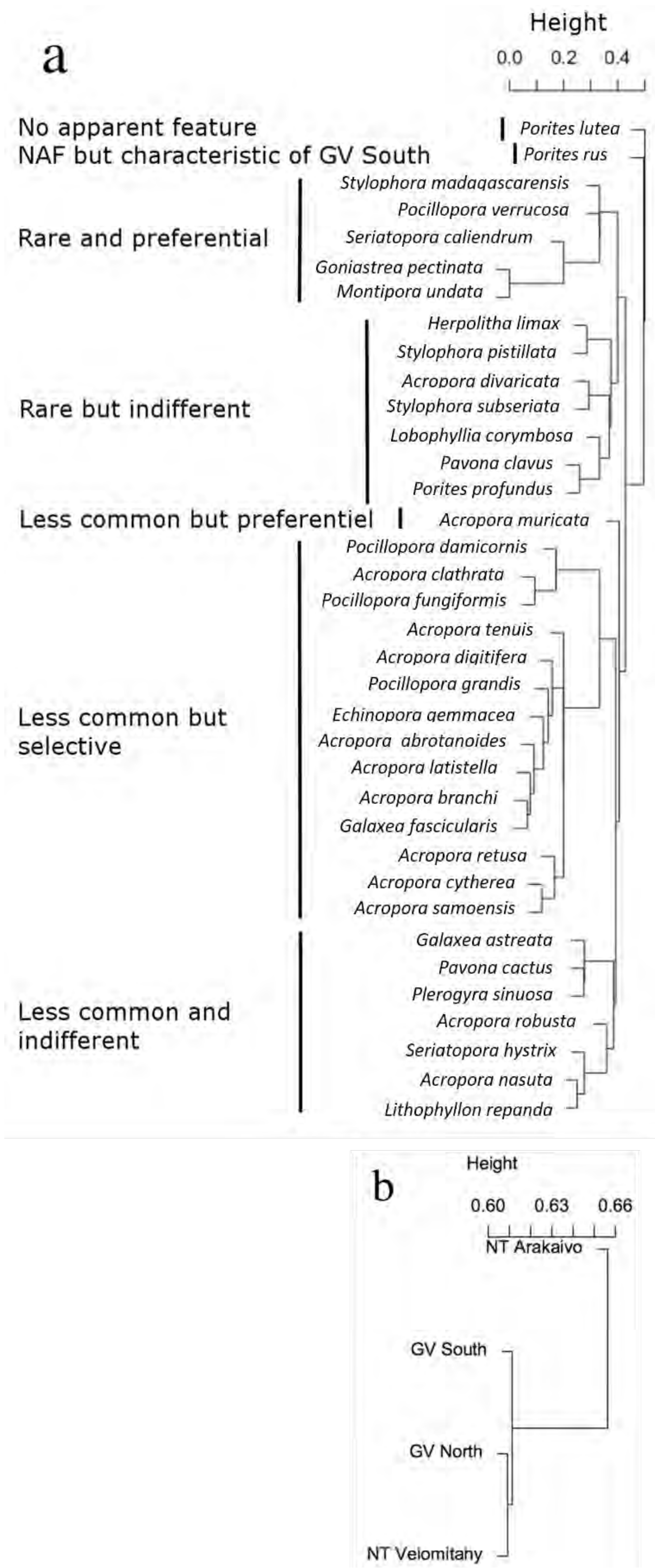

Figure 4. (a) Cluster dendrogram of species, based on their abundance at each station. The characteristics of each group of species are based on the results of constancy and fidelity (Table 3). NAF = No apparent feature; (b) Cluster dendrogram of the stations.

genera (Harris *et al*., 2010; Bruggemann *et al*., 2012). But, as in the present study, this work did not cover all the stations studied in 1978, or in 2008 (by Harris *et al*., 2010) and comparisons with the work conducted 40 years ago are difficult to make.

The present paper reports a total record of only 24.6% of the total number of genera, and 32.1% of the species recorded previously. Most of the genera observed by Pichon (1978) were recorded in the present study, including *Acropora*, *Pocillopora, Stylophora,* and *Porites*. Fourteen of the presently listed species were not listed by Pichon (1978), including eight species of *Acropora* (*A. branchi, A. clathratha, A. divaricata, A. latistella, A. nasuta, A. retusa, A. samoensis*, and *A. tenuis*), two *Porites* (*P. lutea* and *P. profundus*), one *Pocillopora* (*P. fungiformis*), one *Seriatopora* (*S. caliendrum*), one *Stylophora* (*S. madagascariensis*) and one *Lithophyllon* (*L. repanda*). *Lithophyllon* was the least represented with only 1 of 7 species recorded (present study; Pichon, 1978). *Acropora* was well represented in the present study with all 13 species recorded, while only 5 of the 13 were in the list of Pichon (1978).

The richness and diversity at Arakaivo appears very different from the three others staions. Arakaivo is the most exposed station to the water current and it hosts most of the branching species. The richness and diversity at Velomitahy, which is only separated by a few hundred metres from Arakaivo, are closer to those of the GV, a station well protected from the hydrodynamics of the open sea. In the GV, the South is dominated by the species *Porites rus* for an unknown reason, while the diversity of the North of the GV and Velomitahy are very similar. Pichon (1978) differentiated three coral communities according to the depth of the reef slope: many *Acropora* and species of Pocilloporidae inhabit the upper part of the slope; the massive species (e.g. *Pavona*, *Plerogyra* and *Galaxea*) are restricted to the lower part of the slope; and other species like *Porites*, *Montipora* and some *Acropora* occur from the top to the bottom of the slope. In the present paper a zone between 8 to 15 m depths was investigated. Branching colonies were observed in NT and in the GV, but principally at the hydrodynamically active station, Arakaivo. In addition to hydrodynamics, the difference in coral composition between the studied stations could also be explained by the change in habitat structure due to sedimentation and fishing pressure. Indeed, in 50 years, the sedimentation has increased on the GRT and certainly influences the structure of the habitats and consequently the structure of the benthic community, especially the scleractinians (Bruggemann *et al*., 2012; Andréfouët *et al*., 2013). Sheridan *et al.* (2015) showed that the GRT was more affected by diseases than coral reefs of the SW of Madagascar that were not subjected to sedimentation. Except for Arakaivo, the two other stations are highly accessible to fishermen and are the most frequented fishing zones. These stations are subjects to frequent trampling due to destructive fishing techniques (Salimo, 1997). The results from the present study suggest that the GRT scleractinian communities have undergone a significant change in terms of diversity and population structure. Further investigations on this reef, and the management recommendations for its exploitation (small-scale fishery and aquaculture) must consider these changes. In addition, management measures (implementation of protected areas, restoration programmes) must be adapted to ensure greater efficiency.

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Participatory assessment of priority fishery profiles in an overfished urban inshore seascape in Kenya

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Abstract

This study presents a participatory assessment of overfished small-scale fisheries from selected landing sites in coastal Kenya using mixed quantitative and qualitative research methods. A seven-criteria ranking using a modified Delphi questionnaire with a 5-point Likert scale was used. Through a process of scoring the fisheries on these criteria, marine aquarium fishing ranked highest with 86.7%, while beach seine ranked lowest at 55.3%. Averaging of scores and ranking across focus group discussions (FGDs) and key informant interviews (KIIs) at individual sites resulted in the following highest scorers: handlines (58.6%), octopus (55.9%), and basket traps (52.5%) for Bamburi; aquarium fishing (86.7%), handlines (85.5%), and reef-seines for Marina/Mtwapa; mixed pelagic (60.7%), octopus (60.5%), and rabbitfish (58.4%) for Nyali; and basket traps (70.4%), handline (57.8%), and monofilament nets (64.3%) for Reef. Destruction of critical habitats and prohibitive costs of fishing crafts were key management issues identified through scoring and ranking criteria, while 22 management issues were identified through FGDs. These findings suggest an existing spatial mixture of differences and commonalities among fisheries profiles and management issues. This study revealed trade-offs that should be incorporated in the co-management plans of the respective Beach Management Units (BMUs) in coastal Kenya.

Keywords: Mixed participatory approaches, Co-management value addition, Socio-ecological systems, Fishery profiling

Introduction

There is growing promotion for the uptake of, and capacity building for, co-management in the coastal fisheries of Kenya. This is based on the proposition that resource-user participation in management decision-making, commonly referred to as co-management, is likely to produce legitimacy and effective regulations (Wilson *et al.,* 2003; McClanahan *et al.,* 2008a). Successful management of small-scale fisheries, especially those characterized by mixed habitats, multiple gears, multiple species and multiple stakeholders, is vital to ensure continued provision of food, fishery-related jobs, economic profits and other ecosystem services for many resource-dependent communities locally, regionally, and globally.

The Kenyan inshore marine artisanal fishery resources, including the Nyali-Mtwapa urban seascape, continue to be heavily relied on by the coastal artisanal fishers as their main source of income, employment and food security (Okeyo, 2010; Hicks and McClanahan, 2012; FAO, 2014). Due to this high

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dependence, the fisheries in the studied locations continue to exhibit Malthusian over-fishing, exemplified by heavily exploited, multi-species, mixed-gear fisheries (Mangi *et al.,* 2007; McClanahan *et al.,* 2008b; Hicks and McClanahan, 2012).

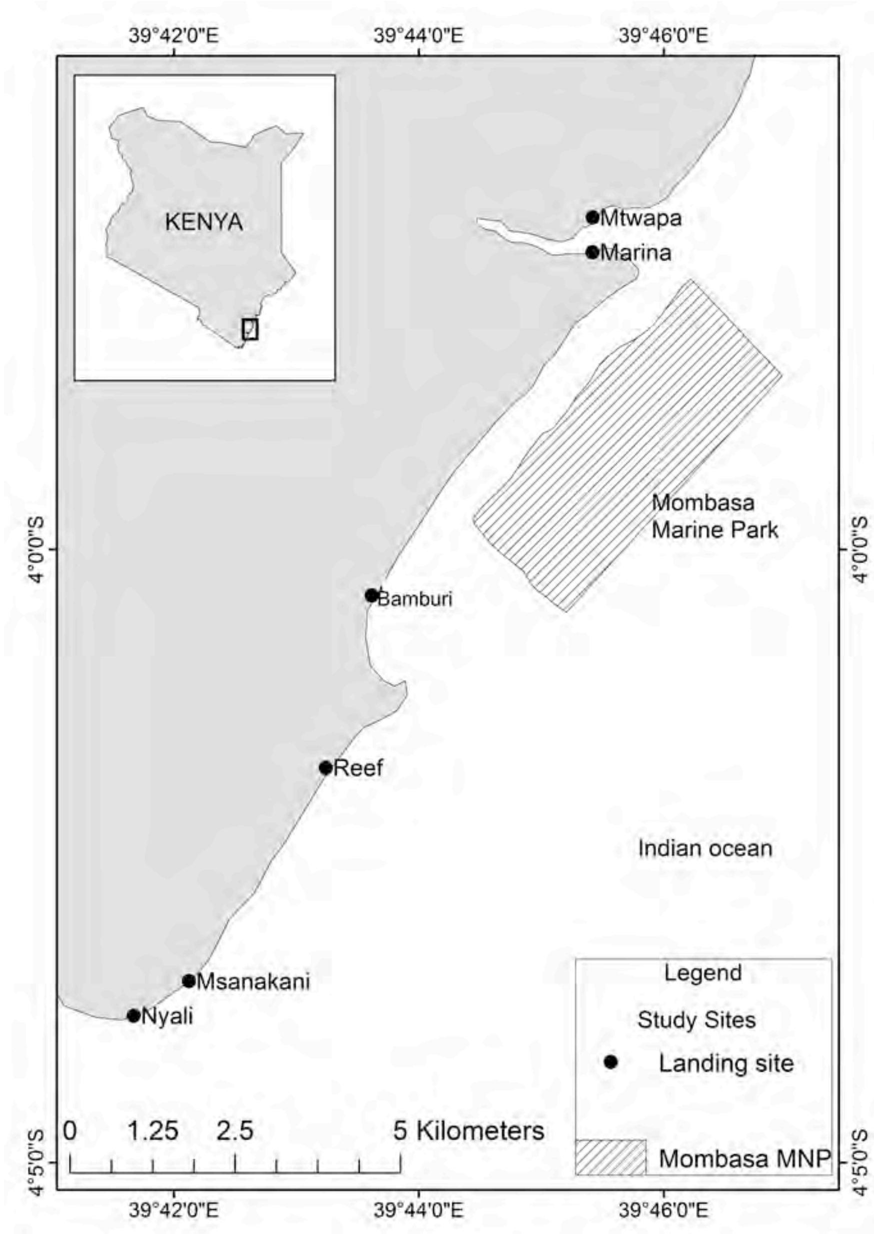

Figure 1. Map of the study sites showing Kenya (inset) and the inshore urban seascape of Nyali-Mtwapa along the Kenyan coastline.

Different management strategies such as traditional systems (e.g. *kayas* in coastal Kenya), gear regulations, size controls, fisheries closures, co-management (BMUs), and the ecosystem approach to fisheries (McClanahan *et. al,* 2005; McClanahan *et al.,* 2008a, b; McClanahan and Mangi, 2001; McClanahan, 2007; Aswani *et al.,* 2012) have been proposed to ensure sustainable fisheries. While co-management is plausible, compliance to management measures have always been hindered by, among other factors, high poverty levels, over-dependence on inshore fisheries resources, and the use of fishing gears that are simple and inexpensive with many fishers making their own gears such as spear guns and basket traps (Mangi *et al.,* 2007). Currently, marine and coastal resources in Kenya are managed at three main levels: the national level; the county level; and the community level (Aswani *et al.,* 2012). At the community level, the BMU movement is slowly increasing the capacity of local communities to manage resources within their fishing grounds (Aswani *et al.,* 2012).

One of the drawbacks with the BMUs in Kenya is the lack of focus and/or specific fisheries management objectives or measures, due to low or no technical capacity in community-generated fisheries management plans. Therefore, there is a need to support technical analysis to make the provisions in community fisheries management plans more specific and easy to identify and implement. To add value to co-management planning for artisanal marine fisheries in the Nyali-Mtwapa seascape, this study sought to support technical analysis of priority fisheries management needs. The study aimed at researching fisheries that are considered priority fisheries because of their high economic and social importance in this urban seascape. The study included investigating the management issues and priorities at local levels based on the views and perceptions of the resource-dependent communities. This was to help identify and provide an assessment of the status and local management needs of these priority fisheries, and support the incorporation of these needs in future fisheries co-management plans.

Materials and Methods

Study area

This study covered four BMUs representing five main landing sites of Nyali, Reef, Bamburi, Marina and Mtwapa within the Nyali-Mtwapa urban seascape (Fig. 1). These sites were selected based on their characteristic mix of different gear types, stakeholder interests, habitat types and multispecies fishery. Furthermore, this stretch of coast has been indicated in previous studies as experiencing Malthusian overfishing (McClanahan *et al.,* 2008a, b; McClanahan and Mangi, 2001). There is a marine no-take area (Mombasa Marine National Park) north of the Bamburi landing site. Around the park is the Mombasa Marine National Reserve (MMNR) that stretches about 1 km north of the park where the Marina and Mtwapa landing sites are located, and 12 km south of the Park where Bamburi, Reef and Nyali landing sites are located (McClanahan *et al.,* 2008b). About 500 fishermen and 50 fish traders derive their livelihoods and income from this seascape while fishing activity is controlled by the monsoon seasons affecting fishermen behaviour concerning target species and fishing methods (Obura, 2001).

Sampling

Purposive sampling that included eight fishers (at least 1 fisher per main gear type), two BMU officials, two female fish traders, two male fish traders, three non-governmental conservation organization representatives, and two government officers per workshop session were used for the participatory rural appraisal (PRA) approach. Thirty two fishers, eight BMU officials, seven female traders, eight male traders, four NGO representatives, and two government officers were involved in the FGDs and the Key Informant Interviews (KII). These stakeholders represented the resource users, researchers, and managers within the seascape.

Data collection procedure

A number of PRA tools were used for data collection based on the objectives of the study to enable different forms of cross-checking on responses, hence securing the validity and reliability of findings (Flick, 2004). The tools included Community Resource Maps (CRM), FGD, KII, and pair-wise scoring and ranking of fisheries management challenges (Fig. 2). This mixed approach used both quantitative and qualitative methods to answer the research question or questions by all means available (Tashakkori and Creswell, 2007; Wiggins, 2011) and for triangulation purposes.

Community Resource Maps

The stakeholders were taken through a community resource mapping exercise where the distribution of fishing grounds in relation to the shoreline, their names and relative positions, were sketched at the FGD workshop using a flipchart and felt pens of different colours. For all the fishing grounds, the main gears used were identified as well as the main species/fish groups targeted. The types of priority fisheries were thus identified based on the fish groups targeted as well as the gears used.

Focus Group Discussions

FGDs were guided by a modified Delphi questionnaire with a 5-point Likert scale (Appendix 1). The fishers were guided through a multi-criteria scoring system (scale of 1 (lowest) to 5 (highest)) that examined specific issues for each identified fishery based on seven criteria. These were: (i) level of community participation; (ii) income levels from the fishery; (iii) perceived catch production and trends; (iv) co-management initiatives; (v) ecosystem impacts of fishing; (vi) types of gears used; and (vii) types of vessels used. Each criteria included various questions (Appendix 1).

Figure 2. Triangulation between different PRA exercises for the participatory fishery profiling.

These scoring criteria were later used to rank the fisheries and the ranks were compared to see if the fisheries satisfied a specified level of compliance for the composite scoring criteria.

Key Informant Interviews

Five to six members representing the different stakeholders with experience in fishing, fish trade, conservation issues, and fisheries management were selected for priority fishery selection criteria scoring using the same questionnaire tool and procedure as above for validation of the FGD. These five to six respondents were considered as key informants. They were identified through a majority vote by the participants in the focus group discussions.

Pairwise scoring and ranking of management issues

The stakeholders identified major challenges faced in the fishery without any order of priority. Pair-wise ranking procedure was then used to score and rank the listed main challenges. Whenever there was a tie in the pair-wise ranking, the tied issues were compared and the one issue that was selected by the FGD as the most important among the two was given a higher ranking than the one considered as subordinate. Pairwise scoring and ranking methods provide a vital starting point, and sometimes the only practical means, for systems whose underlying complexities are not fully understood or agreed (Tucker *et al.*, 1996). The disadvantages of this technique are mainly associated with the arbitrary nature of assigning scores and these being described as overly simplistic (Thompson, 1990). However, if this is kept in mind, such scoring methods, if uniformly applied, can be designed to successfully compare a variety of different issues.

Data analysis

All scored criteria from FGDs and KIIs were converted to percentages of the maximum possible scores for each scoring item. The items were then aggregated to a percentage of the scoring criteria setting a lower cut off of ^{3}40% based on the classical expectations of performance of an open access fishery (being 50% at MSY). The approach of scoring used in the study was similar to other studies aimed at selecting indicators of performance in a fishery (Rademeyer *et al.*, 2007; Schnute *et al.*, 2007; Williams *et al.*, 2011) and specifically, a framework for selecting suitable indicators for fisheries management and development (Rice and Rochet, 2005). The environmental (ecosystem impact) percent scoring criteria was set at £25% based on the principles of the Natural Capital Index (NCI) Framework (PBL, 2012). NCI measures human impact on biodiversity and has been implemented in national, regional and global assessments (Czúcz *et al.*, 2012; Mayunga, 2007; Hambrey and Armstrong, 2010). The NCI is not so much one single-fixed indicator, but rather a flexible indicator framework which can be tailored to specific scale, available data, and demand. Stakeholder-specific profiling (scoring and ranking) across the FGDs and KIIs by site was done as well as overall scoring and ranking through averaging individual scores across them. The identified management issues were examined for patterns in their distribution among the landing sites using De-trended Correspondence Analysis (DCA), with sites as the independent variables and the ranks as the dependent variables.

Results

Criteria based assessment of priority fisheries in the Nyali-Mtwapa seascape

There were mixed categorizations of the coastal fishery into 14 main categories based on either gears used or the species/group of fish caught (Table 1). The marine aquarium fishery scored the highest at 86.7%, thus ranking first while the beach seine fishery scored the least at 55.3%, and was ranked last. Overall, there was highest scoring for criteria 6 on types of gears used, followed by criteria 3 on production and catch trends, and criteria 1 on community participation. The order of criteria however varied by individual BMUs and hence the differences observed in some of the priority fisheries.

Average ranking across FGD and KII indicated that the handline fishery scored the highest at 58.6%, while the gillnet fishery scored the lowest at 49.3% for Bamburi (Table 2). At Marina/Mtwapa the marine aquarium fishing scored highest at 86.7%, and beach seine fishing scored lowest at 55.3%. The Mixed pelagic fishery scored highest of 60.7%, and mixed demersal fishing the lowest at 56.5% for Nyali. The Reef landing site had the basket trap fishery scoring highest at 70.4%, while spear fishing scored least at 50.8% (Table 2).

Further analysis and pooling of the fishery by target species groups indicated four main fisheries in the seascape. These were marine aquarium fishing with a score of 86.7%, mixed pelagic fishery (mainly reef seine) at 79.7%, mixed demersal fishery (handlines, demersal longlines, speargun and beach seines) at 71.7%, and octopus fishery (spearguns and spears) at 67.1%. The results are discussed based on these four target species group categories.

Table 1. Overall priority fisheries types identified for the Nyali-Mtwapa seascape at all sites based on average scores across all the FGDs, KIIs and sites pooled. Scores ≥40% indicates higher ranking except for criteria 5, where ≤25% is used.

No.	Criteria	Criteria 1: Community Participation	Criteria 2: Income	Criteria 3: Production and Catch	Criteria 4: Co-Management	Criteria 5: Ecosystem Impacts	Criteria 6: Types of Gears Used	Criteria 7: Fishing Crafts Used	Overall	Ranking
	Criteria benchmark	≥40%	≥40%	≥40%	≥40%	≤25%	≥40%	≥40%	≥40%	
	Fishery									
1	Marine aquarium fishing	66.7	61.7	76.7	56.7	23.7	96.7	53.8	**86.7**	**1**
2	Reef seines	67.8	49.2	63.3	58.3	25.0	94.4	39.5	**79.7**	**2**
3	Longlines	78.3	57.5	86.7	67.5	41.0	60.0	57.1	**78.8**	**3**
4	Handline	73.3	48.8	66.8	52.8	34.4	88.3	36.9	**67.3**	**4**
5	Monofilament nets	67.5	49.2	83.3	51.3	41.3	96.7	36.7	**64.3**	**5**
6	Basket traps	69.8	50.7	75.3	49.7	34.2	71.9	33.7	**61.4**	**6**
7	Spear guns	77.5	51.7	100.0	48.3	38.0	100.0	7.6	**61.2**	**7**
8	Mixed Pelagic	63.0	55.0	44.0	55.5	40.0	61.3	41.1	**60.7**	**8**
9	Spear	61.4	39.3	64.5	47.6	27.5	100.0	18.1	**58.9**	**9**
10	Rabbitfish	70.0	49.0	24.0	59.0	39.6	62.7	32.6	**58.4**	**10**
11	Octopus	77.8	46.6	46.7	51.2	31.7	69.0	23.5	**58.2**	**11**
12	Mixed demersals	79.0	54.0	28.0	54.5	42.4	62.7	28.6	**56.5**	**12**
13	Gillnets	71.3	43.3	56.7	47.7	39.8	75.4	33.4	**55.4**	**13**
14	Beach seines	77.5	56.7	30.0	51.7	48.0	94.4	22.4	**55.3**	**14**

Table 2. Important fisheries types identified for Nyali-Mtwapa seascape at all sites based on average scores across all the FGDs and KIIs per landing site surveyed. Scores ≥40% indicate higher ranking, except for criteria 5 where ≤25% is used.

Criteria	Criteria 1: Community Participation	Criteria 2: Income	Criteria 3: Production and Catch	Criteria 4: Co-Management	Criteria 5: Ecosystem Impacts	Criteria 6: Types of Gears Used	Criteria 7: Fishing Crafts Used	Overall	Ranking
Criteria benchmark	≥40%	≥40%	≥40%	≥40%	≤25%	≥40%	≥40%	≥40%	
Nyali									
Mixed Pelagic	63.0	55.0	44.0	55.5	40.0	61.3	41.1	60.7	1
Octopus	78.0	44.0	40.0	56.5	30.8	78.7	17.1	60.5	2
Rabbitfish	70.0	49.0	24.0	59.0	39.6	62.7	32.6	58.4	3
Mixed demersals	79.0	54.0	28.0	54.5	42.4	62.7	28.6	56.5	4
Reef									
Basket traps	68.0	53.0	100.0	51.5	34.4	96.0	36.0	70.4	1
Handline	56.4	37.1	57.1	44.3	30.3	100.0	26.9	57.8	2
Monofilament nets	67.5	49.2	83.3	51.3	41.3	96.7	36.7	64.3	3
Gillnets	64.2	39.2	63.3	49.6	40.7	100.0	38.6	61.4	4
Speargun	77.5	51.7	100.0	48.3	38.0	100.0	7.6	61.2	5
Spear	53.6	37.9	65.7	41.4	27.4	100.0	5.3	50.8	6
Bamburi									
Handlines	81.7	52.5	63.3	46.7	33.7	64.5	29.1	58.6	1
Octopus	77.5	49.2	53.3	45.8	32.7	59.4	29.8	55.9	2
Basket traps	71.7	48.3	46.7	47.9	34.0	47.8	31.4	52.5	3
Gillnets	78.3	47.5	50.0	45.8	39.0	48.6	28.3	49.3	4
Marina/ Mtwapa									
Marine aquarium	66.7	61.7	76.7	56.7	23.7	96.7	53.8	86.7	1
Handlines	81.7	56.7	80.0	67.5	39.3	96.7	54.8	85.5	2
Reef seines	67.8	49.2	63.3	58.3	25.0	94.4	39.5	79.7	3
Longlines	78.3	57.5	86.7	67.5	41.0	60.0	57.1	78.8	4
Spear	69.2	40.8	63.3	53.8	27.7	88.9	31.0	67.1	5
Beach seines	77.5	56.7	30.0	51.7	48.0	94.4	22.4	55.3	6

All the fishery types prioritized by the participants scored highly on criteria 6 (types of gears used), and criteria 1 (community participation), indicating that these fisheries use gears that are locally available, affordable, legal, repairable and environmental-friendly, except the beach seines that were indicated to be destructive, and that the fishery is operated mainly by local community fishers (Table 2).

The participatory resource maps indicated that catches within the seascape included a mix of demersal finfishes from the families Lutjanidae (snappers), Lethrinidae (emperors), Siganidae (rabbitfish), Scaridae (parrotfish), Acanthuridae (surgeonfish and unicorn fish), Mullidae (goat fish), Haemulidae (sweetlips) and Serranidae (groupers); octopus; and mixed pelagics such as jacks and trevallies (family Carangidae), tuna and mackerels (family Scombridae), barracudas (family Sphyraenidae), halfbeaks (family Hemiramphidae), dolphin fishes (family Coryphaenidae) and sardines (family Clupeidae) (Appendix 2 and 3). Species identified included *Lethrinus lentjan*, *Siganus sutor*, *Leptoscarus vaigiensis*, *Lethrinus mahsena*, *Lutjanus fulviflamma*, *L. argentimaculatus*, *Sphyraena barracuda*,

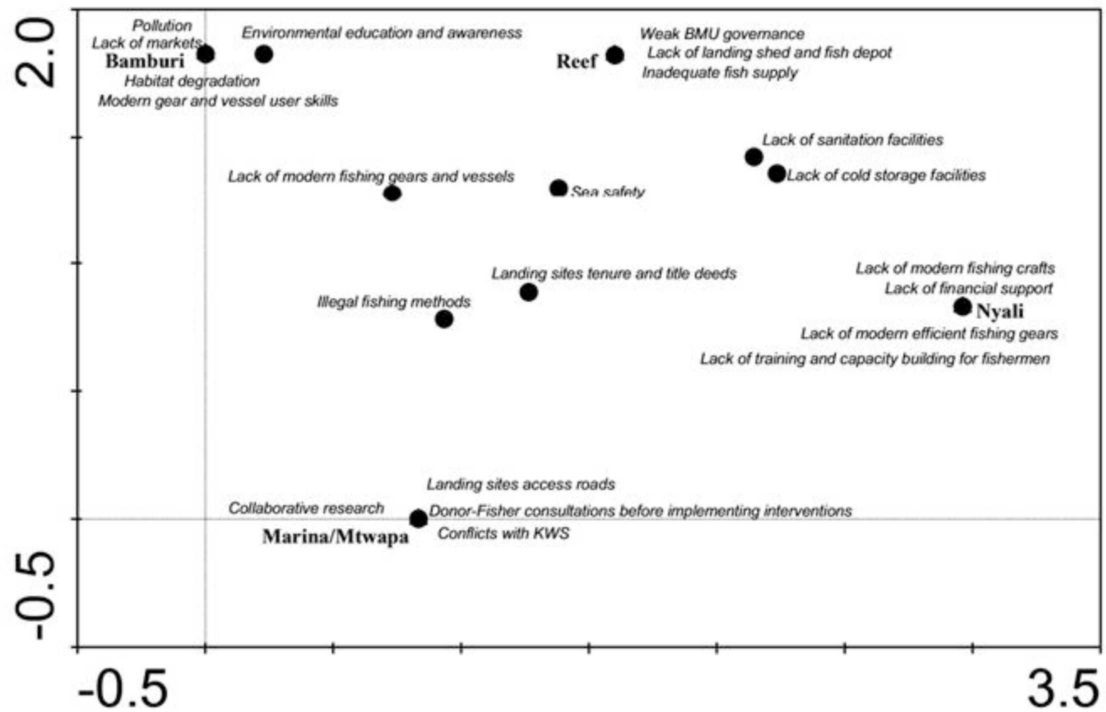

Figure 3. Spatial representation of the 22 management issues identified across the studied landing sites.

S. jello, *Hemiramphus far*, *Scomberoides tol*, *Plectorhinchus gaterinus* and *P. gibbus*.

Management issues

Pair-wise ranking of the various management issues showed a total of 22 management issues for all sites pooled (Fig. 3). Out of these, illegal fishing methods, lack of modern fishing gears and vessels, and lack of landing site title deeds were common across the three sites of Nyali, Bamburi and Marina/Mtwapa. The need for environmental education and awareness, lack of cold storage facilities, lack of sanitation facilities, and sea safety concerns were common issues across two sites (Fig. 3).

Discussion

This study utilized participatory rural appraisal approaches in the context of mixed methods research to examine fishery profiles and management issues in the Nyali-Mtwapa urban seascape in Kenya. The study broadly discusses marine aquarium, mixed demersal, mixed pelagic and octopus fisheries within this seascape that are targeted with a mix of gears relevant to each fishery type as prioritized by the participants in the FGD and KII.

Profiles for each selected priority fishery

Marine aquarium fishery

The Marine aquarium fishery was ranked highest based on criteria 3 that gears are locally available, easy to maintain and environmentally-friendly, and criteria 1 that local fishers are involved in it. The marine aquarium fishery in Kenya dates back to the 1960s, and since then it has grown in terms of volumes in numbers, species diversity, fishing effort, fishing grounds and trade (Draft Marine Aquarium Fishery Management Plan, 2016). This would be the reason why the fishery scored well on criteria 3 on participant perception that it had high production and catch trends. Marine aquarium products include live fish, invertebrates (soft corals, shrimp, and small clams), and coral rocks. Aquarium fishing is conducted by snorkelling in shallow depths of up to 3 metres, or diving using underwater breathing apparatus (SCUBA or hookah) in deeper depths. Barrier nets of varying mesh sizes and hand-held scoop nets are used to trap and collect the fish and invertebrates, while live rock is hand-picked from the bottom. The organisms are temporarily stored in plastic buckets while in the water, and transported in oxygenated plastic bags to holding facilities. Among the targeted finfishes are wrasses, damselfishes, anthiases, blennids, scorpionfishes, angelfishes, surgeonfishes, gobies, butterflyfishes, dartfishes and pufferfishes. The main export market for live aquarium products from Kenya is the United States of America. Others include the European Union (dominated by Germany and United Kingdom), South Africa, Israel and Japan. Currently there are eight established companies exporting to 26 countries.

Up to 90% of the aquarium fish catches are collected from five main fishing grounds off Shimoni (33%), Kanamai (20%), Mtwapa (18%), Kilifi (12%) and Ukunda (5%) (Draft Marine Aquarium Fishery Management Plan, 2016).

Despite the benefits indicated, there has been an increase in resource use conflict reported between aquarium fishers and other artisanal fishers. According to Okemwa *et al.* (2016), 57 marine aquarium fish species were recorded in artisanal catches, dominated by wrasses making up 39%, and comprising of 19 species of which handlines captured the highest diversity of marine aquarium fish species and also had the highest resource interaction constituting 46% the total marine aquarium catches in relative abundance. This was followed by spear guns (26%) and basket traps (17%). Over-exploitation of target species has also been indicated, thus negatively impacting on the sustainability of the fishery. To address these management challenges, the Marine Aquarium Fishery Management Plan (2016) is being developed following the Food and Agriculture Organization of the United Nations (FAO) Ecosystem Approach to Fisheries (EAF).

Mixed demersal fishery

This study established that the resource users within the seascape still prioritized mixed demersal fishes as a mainstay for their livelihood. The fishery scored highly on criteria 6 and 3 indicating prevailing perceptions among fishers that the gears they use are readily available, affordable, and repairable, and that production trends are stable. These mixed demersal fish species are targeted by a broad array of gears including beach seines, small meshed gill nets, monofilament nets, spear guns, longlines, handlines, and basket traps. From 70-80 percent of the marine fish catches in Kenya is demersal, mainly from the inshore shallow water coastal habitats (ICAM, 1996; Matiru *et al.,* 2002). The catch composition of the mixed demersal fishery generally consists of a very large number of species with greatly varying sizes. A large number of juveniles are also recorded in the demersal catches (Hicks and McClanahan, 2012). The possible reason why this fishery scored negatively on environmental impact was due to beach seine effects. Catches normally include finfish of the families Lethrinidae (emperors), Siganidae (rabbitfish), Scaridae (parrotfish), Lutjanidae (snappers), Acanthuridae (surgeonfish and unicorn fish), Mullidae (goat fish), Haemulidae (sweetlips) and Serranidae (groupers). The inshore reef fisheries are generally considered to be at maximum sustainable yields, or over-exploited (Matiru *et al.,* 2002; McClanahan *et al.,* 2008b; Mangi *et al.,* 2007; Tuda and Wolff, 2015). Although diverse, about 75 percent by weight of the 15 most abundant landed demersal fishes is composed of 3 species, namely *Lethrinus lentjan*, *Siganus sutor*, and *Leptoscarus vaigiensis*. Other species include *Lethrinus mahsena*, *Lutjanus fulviflamma, L. argentimaculatus, Plectorhinchus gaterinus* and *P. gibbus* (McClanahan *et al.,* 2008b, Hicks and McClanahan, 2012). These species were among those identified through the participatory resource mapping exercise. These fish groups are experiencing intense levels of fishing pressure in the seascape; however, the status of the stocks remains minimally described. There are general observations that most of the populations are showing evidence of growth and recruitment overfishing (McClanahan *et al.,* 2008b). Contributory factors include the use of non-selective gears such as beach seines, which are well documented to capture high numbers of juveniles; up to 80% in some areas (Hicks and McClanahan 2012), and targeting of spawning aggregations of especially groupers and rabbitfishes. Use of beach seines is banned, however, this gear is still popular in Nyali and Marina landing sites despite efforts to curb their use. These conflicts between fishermen and the Kenya Wildlife Service (KWS) were identified as one of the management issues at Marina/Mtwapa. There is evidence of the need for management interventions to enhance compliance in the regulations on mixed demersal fisheries.

Octopus fishery

The octopus fishery was prioritized highly based on criteria 6, as it is cheap and easy to make spears and spear guns that are the main fishing gears used in this fishery. The octopus fishery in Kenya is dominated by the common octopus (*Octopus vulgaris*) which contributes the bulk of all cephalopods landed on the Kenyan coast, and is one of the most desirable octopus species for food and commercial purposes for the majority of fishermen (Kivengea, 2014). The octopus catches have been reported to show an increase from values of 49 metric tonnes in 1992 to values of nearly 290 metric tonnes in the year 2008, with some high peaks in octopus landings occurring in the same time period when finfish landings were at their lowest (Kivengea, 2014). This may be an indication that, due to poor catches of finfishes, fishermen turn to the capture of invertebrate species such as octopus, but this needs further investigations (Kivengea, 2014). There is need for further research and assessment of the octopus fishery for detailed stock assessment status. There are no management interventions put in place specifically for this fishery, despite it being prioritized as important socio-economically in the studied seascape.

Mixed Pelagic Fishery

The mixed pelagic fishery was prioritized based on criteria 6 on perceived affordability, availability and maintenance of the gears involved, as well as criteria 1 on community involvement in the fishery. Mixed pelagic species are targeted by a variety of gears including reef seines, beach seines, small meshed gill nets, and longlines. The catch composition of the pelagic fishery generally consists of jacks and trevallies (Carangidae), tuna and mackerels (Scombridae), barracudas (Sphyraenidae), halfbeaks (Hemiramphidae), dolphin fishes (Coryphaenidae) and sardines (Clupeidae) (Munga *et al.,* 2016). In the Nyali-Mtwapa seascape this group was dominated by the small and medium pelagic fishes that include rainbow sardines, white sardinella, barred needle fishes, rainbow runners, trevallies, mackerel scads, chub mackerels, Indian mackerels, stripped bonitos, kingfish, queenfish, and great barracuda (pers. obs.). The full potential of small and medium pelagic fisheries along the Kenyan coast is not yet assessed. Consequently, these pelagic resources are assumed to be under-exploited due to lack of capacity of the artisanal fishers to venture far offshore.

The management of this fishery was initially captured in two draft management plans: the Draft Ringnet Fishery Management Plan (RFMP) (2013), whose broad objective was to enhance responsible exploitation of pelagic fish stocks through regulation of ringnet fishing practices that minimize resource use conflicts while providing long term biological and socioeconomic benefits; and the Draft Small and Medium Pelagic Fisheries Management Plan (SMP-FMP) aimed at management of the pelagic species to optimize social and economic benefits of the small and medium pelagic fisheries to the local community, national and regional economy, and to ensure long-term biological sustainability and ecological integrity of the pelagic fisheries, and develop and improve governance of the fishery locally, nationally and regionally. The Small Scale Purse Seine Fishery Management Plan (SSPSFMP) (2015) is currently under development. Its objectives include: regulating the small scale purse seine fishery in terms of catches, fishing effort, fishing grounds and trade; minimizing conflicts through capacity building of resource user organizations, benefit sharing strategies, licensing schemes, and environmental management; enhancing research and monitoring to support the development of optimum harvest strategies; improving the net income of small scale purse seine fisher communities and national revenues through value chain development and improvement; and developing mechanisms to enhance enforcement and compliance to prescribed measures for sustainable ecosystem management. These are emphasized in the current study due to the perceived importance of the fishery by the resource users within the seascape.

Priority fisheries and management issues

The findings of this study indicated both heterogeneities and homogeneities that exist in the priority fishery profiles over this short stretch of coastal urban seascape of the Nyali-Mtwapa system. This could be attributed to differences and similarities in resource use patterns by the users, as well as different or similar levels of local ecological knowledge (LEK). Similar findings were observed by Crona (2006), who indicated that differences in local ecological knowledge do occur even at small-scale level in coastal communities, although these have not been well studied to determine their interactions and associated outcomes. Likewise, LEK has been shown to be homogenous within Kenyan landing sites and may overlap groups of landing sites (Evans, 2010), as also indicated in this study. Points of consensus (homogeneity) and variance (heterogeneity) are important in adding value to the co-management initiatives in coastal Kenya since it entails integrated management where diverse stakeholder views, including trade-offs, are incorporated. This is because homogeneity amongst key individuals can often be a hurdle towards internalisation and recognition of changing ecological conditions (Bodin and Crona, 2008). Fishery systems that are open access and where multiple resource users exist would be characterised by scenarios where each user find ways to maximize utilization of resources for their individual benefit. Consequently, through participatory community processes, whose success is dependent on meeting practical needs, it was found that stakeholders within the seascape were mainly interested in ways of increasing their production and wealth, thus concurring with the study by Hirsch (1990), that changes in production relations may be linked with diversity among producers, emerging heterogeneity of interests, and the problems this produces for cooperative development interventions. There is therefore need to address observed heterogeneities among users as synergies are embraced in consensus towards improved co-management planning for fisheries management.

The priority fisheries identified through the participatory approaches are typical of those that have been studied in the seascape by various researchers over time (McClanahan *et al.,* 2008b, Mangi *et al.,* 2007, McClanahan and Mangi, 2001). These priority fisheries reflect the mixed species, mixed gear fishery of the Nyali-Mtwapa seascape as described and discussed in previous studies (McClanahan *et al.,* 2008a, b; McClanahan and Omukoto, 2011; Mangi *et al.,* 2007), thus resulting in the heterogeneity observed in the fishery. The participatory survey indicated a perceived mix of declines and stability of catch over the past 10 years for all the priority fisheries. This concurs with similar reports from previous studies that indicate these catches to be stable but with shifts towards the lower trophic groups such as the herbivores and detritivores (McClanahan *et al.,* 2008b).

The mix of artisanal fishing gears and techniques that included basket traps, handlines, beach seines, longlines, gillnets/set nets, monofilament nets, reef seines, spears, and spear guns were common across the study sites. The use of these gears and techniques is primarily driven by a range of geographical, contextual, financial and socio-cultural factors such as resident village, choice of landing site, financial capital, social networks, and age (Evans, 2010; Mangi *et al.,* 2007).

Management issues such as the landing of undersized fish and juveniles of other species were attributed to the use of deleterious fishing methods, especially beach seines, monofilament gillnets, undersized mesh gillnets and basket traps. Furthermore, the impacts of these gears on the habitats, amplified by environmental degradation due to pollution, were also among the key management issues requiring intervention. The issue of title deeds or entitlement to land ownership for BMUs remains critical in determining the stability and infrastructural developments for the fishers and BMUs.

Conclusions and Recommendations

The priority fisheries for the Nyali-Mtwapa system can be described at different levels depending on the method of resource extraction (different gear types and crafts), the target resource (fish species or groups) or a combination of the two. For management purposes and stock status assessments, addressing fisheries by species or species groups is advised. In this study, mixed demersals, mixed pelagics, octopus and marine aquarium fisheries have been broadly discussed within this seascape, that are targeted with a mix of gears relevant to each fishery type. The incorporation of these in co-management planning or for technical analysis purposes would be important. It was, however, noted that three main gears used in the seascape fishery (beach seines, monofilament gillnets, and spearguns), were illegal and therefore it is recommend that these should be the key focus of any future interventions, and a priority in co-management planning. While there are plans underway to finalize management plans for the marine aquarium fishery and the small and medium pelagics, the mixed demersal and octopus fisheries are still operational without management plans. There are a number of management gaps that require addressing in the selected priority fisheries. These include: fishery specific legislation and regulations; Monitoring, Control and Surveillance (MCS); development of the small-scale fishing fleets; addressing challenges in landing site ownership; and facilitating the training needs of fishers. It is recommended that the identified mix of both differences and commonalities in spatial variations in fishery profiles and management issues are incorporated in the co-management plans of the respective BMUs in Kenya, as added value to enhance legitimacy and acceptance of the co-management approaches and networks.

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Appendix 1. Modified Delphi questionnaire tool with a 5-point Likert scale.

The following criterion based on a scale of 1 – 5 is proposed for selecting priority fisheries profiles for the Nyali-Mtwapa Seascape based on stakeholders' consultation:

Criteria 1: Community Participation [Level are lowest (1) to highest (5)]

		Fishery types				
No	**Questions**					
i.	Are indigenous the main fishers involved in the fishery?					
ii.	Are the fish species eaten by the indigenous communities?					
iii.	Are women involved in the fishery?					
iv.	Are the youth involved in the fishery?					

Criteria 2: Income [Level are lowest (1) to highest (5)]

		Fishery types				
No	**Questions**					
i.	Does the fishery provide adequate income to the fishers?					
ii.	Does the fishery provide adequate income to fish traders?					
iii.	Is the fish traded beyond the local area?					
iv.	Does trading fish outside the fishing area give better income?					

Criteria 3: Production and Catch [Level are lowest (1) to highest (5)]

		Fishery types				
No	**Questions**					
i.	Do the fishers get enough catch on a daily basis?					
ii.	Have catches declined in the past 10 years?					
iii.	Are catches expected to be stable in the next 10 years?					

Criteria 4: Co-Management [Level are lowest (1) to highest (5)]

		Fishery types				
No	**Questions**					
i.	Is the BMU involved in managing the fishery?					
ii.	Does the BMU provide any services at the landing site?					
iii.	Is the BMU involved in marketing of the fish?					
iv.	Does the BMU have infrastructure for marketing?					
v.	Are there any illegalities in the fishery?					
vi.	Does the BMU participate in controlling these illegalities?					
vii.	Is the BMU effective in implementing regulations?					
viii.	Does the BMU collaborate with other partners?					
ix.	Are there clear management measures for the fishery?					
x.	Are legislations and regulations known to the BMU Assembly?					

Criteria 5: Ecosystem Impacts [Level are lowest (1) to highest (5)]

		Fishery types				
No	**Questions**					
i.	What is the intensity of fishing in the coral reefs?					
ii.	Does the fishing interfere with the corals?					
iii.	What is the intensity of fishing in the sea grass beds?					
iv.	Does the fishing interfere with the sea grasses?					
v.	Does fishing take place in the estuaries?					
vi.	Does the fishing interfere with the estuarine ecosystem?					
vii.	Does the fishery catch any juveniles of the targeted species?					
viii.	Does the fishery catch juveniles of non-targeted species?					
ix.	Does the fishery catch sea turtles?					
x.	Does the fishery catch dolphins and dugongs?					
xi.	Does the fishery catch sharks?					
xii.	Does the fishery catch rays and skates?					

Criteria 6: Types of Gears Used [Level are lowest (1) to highest (5)]

		Fishery types				
No	**Questions**					
i.	Are the gears used in the fishery available locally?					
ii.	Are the gears used in the fishery affordable? (probe for prices)					
iii.	Are the gears used in the fishery legally acceptable?					
iv.	Are the gears used in the fishery left in the sea or carried back to the shore after fishing?					
v.	Are the gears easy to repair?					
vi.	Do the gears last for a long time? (Durability)					
vii.	Are the gears frequently lost at sea?					

Criteria 7: Fishing Crafts Used [Level are lowest (1) to highest (5)]

		Fishery types				
No	**Questions**					
i.	Are the fishing crafts used in the fishery constructed locally?					
ii.	Are the fishing crafts used in the fishery affordable? (Probe cost)					
iii.	Are the fishing crafts used in the fishery considered seaworthy?					
iv.	Are the fishing crafts used in the fishery easy to maintain/repair?					
v.	Are the fishing crafts used in the fishery made of timber?					
vi.	Are the fishing crafts used in the fishery made of fibreglass?					
vii.	Are the fishing crafts used in the fishery made of metal?					
viii.	Are the fishing crafts propelled by paddles?					
ix.	Are the fishing crafts propelled by sail?					
x.	Are the fishing crafts propelled by outboard engines?					
xi.	Are the fishing crafts propelled by inboard engines?					

Satellite-derived bathymetry: A case study of Mombasa Port Channel and its approaches, Kenya

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Abstract

Bathymetry refers to the depth of the water column in relation to sea level. It is fundamental in marine spatial planning, resource exploration and sustainable management of marine resources. It is also vital for safety of navigation, and planning of coast-based infrastructural developments. However, acquisition of bathymetry data is very expensive due to the cost of equipment, expertise and technology needed to collect data and produce maps. Satellite-derived bathymetry (SDB) therefore offers an opportunity to generate shallow water bathymetry at extremely reduced costs, mainly due to freely-available multispectral satellite imagery and open-source processing software. This paper presents the application of an already developed and published shallow-water bathymetry derivation model and protocols. The results indicate that the technique could be effective for mapping shallow water bathymetry, with higher accuracy in low to non-turbid waters. The SDB of Mombasa Port was identical to the official charted depths upon comparison, with a R^2 value of ~90% and a RMSE of 1.61 m. SDB maps can be categorized as medium resolution due to their relatively low spatial resolution. SDB cannot be used as a stand-alone hydrographic tool but it rather offers a viable reconnaissance solution for mapping shallow coastal waters where hydrographic data gaps exist.

Keywords: bathymetry, remote sensing, satellite imagery, Landsat 8, models

Introduction

The costs of obtaining high resolution bathymetric data are extremely high. This is attributed to the high cost of equipment (e.g. multibeam and singlebeam echosounders, sidescan sonars, remotely and automatically operated vehicles), platforms (research vessels), specialized processing software, and expertise. Dedicated research vessels and the expertise needed are still inadequate in most developing countries (Pe'eri *et al.,* 2014). Traditionally, bathymetry data was mainly used for updating navigational charts, but it is now gaining prominence in mapping the seafloor for scientific purposes, including for fisheries management, tsunami propagation modelling, and for the oil and gas industry. The need for bathymetry data has also been raised by the heightened focus on the blue economy for the sustainable use of marine resources and development. Therefore, the development of freely-available resources, open source and cost-effective tools for obtaining bathymetry could improve the capability of developing countries to collect bathymetric data. These tools and resources should be able to assist in assessing geomorphological changes and provide reliable reconnaissance for further directed and detailed surveys.

One of the emerging tools for establishing bathymetry is the derivation of bathymetry from multispectral satellite imagery. Satellite-derived bathymetry (SDB) is a survey method founded on empirical and analytical modelling of light penetration through the water column in visible bands of multispectral satellites (Pacheco *et al.*, 2015; Laporte *et al.*, 2014; Pe'eri *et al.*, 2014; Lee *et al.*, 1999; Philpot, 1989; Lyzenga, 1985). Major advantages of these methods are the availability of freely-available satellite data (e.g. Landsat and Sentinel), new sensors with added capabilities, open source processing software (e.g. QGIS)

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and scientific literature and case studies (Barber *et al.*, 2016). SDB has been gaining acceptance in recent years (Pacheco *et al.* 2015; Pe'eri *et al.* 2014; Bramante *et al.*, 2013; Stumpf *et al.*, 2003; Lyzenga, 1985). Despite the developments over time, SDB still offers low resolution maps compared to some traditionally accepted acoustic surveying techniques such as multibeam surveys (MBES) which offer full seafloor coverage of up to 0.5 m resolution (IHO, 2008). According to Barber *et al.* (2016), this accuracy can be attributed to the large spatial resolution of available satellite platforms (2 m for sentinel-2, and 15 – 30 m for Landsat 8). However, SDB could offer better resolutions than point-based methods including leadlines and singlebeam (SBES) echosounders where data points could vary in the spatial scale. Although there has been gradual development of SDB methodology, published information has always concentrated on algorithm and model development, and it is only very recently that case studies have been conducted. This paper utilizes the water radiances of three bands of Landsat 8 (infrared 1560-1660 nm; green: 525-600 nm; blue: 450 – 515 nm) to retrieve and estimate shallow water bathymetry. Landsat 8 carries two sensors (the Operational Land Imager (OLI) and the Thermal Infrared Sensors (TIRS)) with eleven bands of various capabilities (Irons *et al.*, 2012). A typical multispectral satellite sensor has several channels individually capturing a broad (70 -150 nm) spectral range (Pe'eri *et al.*, 2014). They also collectively cover the entire electromagnetic spectrum (visible to infrared) ranging between 433 – 1390 nm. The fundamental concept that underpins SDB is the exponential wavelength-dependent light transmittance, penetration and attenuation in water with respect to depth (Laporte *et al.* 2015; Pacheco *et al.*, 2014; Pe'eri *et al.*, 2014). According to Pe'eri *et al.*, 2014 (originally in Jerlov, 1976), 350 nm (ultra-violet) – 700 nm (red) represents the range of wavelengths of light that are less attenuated in seawater for relatively considerable depths. However, wavelengths greater than 700 nm (near infrared) are attenuated easily in seawater and are therefore suitable for delineating the land and seawater boundary. It is therefore from the visible bands (red, green and blue) between 450 – 690 nm, that the principle of estimation of depth is anchored. In order to derive bathymetry estimates, existing bathymetric data (e.g. chart soundings, LiDAR) should be used to vertically reference the SDB and convert the SDB raster image to depth. The Lowest Astronomical Tide (L.A.T in metres) tidal datum used in this study is the official Kenyan chart datum for all nautical/navigation charts. For optimal use of the SDB method, several other environmental requirements that could affect the estimation of bathymetry and introduce errors need to be considered. These include water clarity, satellite images without and/or with minimal sun glint, and cloud and optically active materials in the water column such as suspended matter and aquatic vegetation.

Several models and algorithms have been developed over time for SDB (Pe'eri *et al.,* 2014; Stumpf *et al.*, 2003; Lyzenga, 1985; Lyzenga, 1978). The models, according to Pe'eri *et al.* (2014), can be categorized into three broad categories: i) analytical models that utilizes radiative transfer models for specific data (Lyzenga, 1978; Philpot, 1989); ii) comparative models that utilize large datasets generated from radiative transfer models (Bramante *et al.*, 2013; Louchard *et al.*, 2003); and finally iii) the optimization band ratio model used in this paper, based on the assumption of vertically-invariant water column and seabed condition (Pe'eri *et al.*, 2014; Stumpf *et al.*, 2003). The three models are mathematically hinged on the exponential decay of radiance (W m^{-2} sr^{-1} nm^{-1}) with water depth (Mobley, 2004). According to Philpot (1989), the observed radiance in shallow water can be represented as shown in the equation below:

$$L_{obs} = (L_b - L_w) \cdot e^{-2k(\lambda).z} + L_w$$

Where L_{obs} is the observed radiance, L_b is the radiance contribution from the sea bottom, L_w is the observed radiance over optically deep waters, z is the water depth (m), and k(λ) is the diffuse attenuation coefficient (Bramante *et al.*, 2013; Mobley, 2004). Therefore, the equations $L_{obs} = L_b$ (at Z=0) and $L_{obs}=L_b$ (Z=∞) holds.

The general assumption for the optimization band ratio model is that the sea bottom is homogenous and therefore the reflectance and attenuation is uniform. In reality though, shallow coastal areas are among the most dynamic in terms of wave and sediment dynamics. According to Pe'eri *et al.* (2014), the utilization of ratio transform in this approach yields robust bathymetry without necessarily sampling the dynamic environment.

For vertical referencing purposes, official Kenyan navigational charts were used. The official navigation chart for Mombasa and its approaches is based on diverse data from lead line (as old as 1880s), singlebeam data, and multibeam surveys. Kenya's official charting agency is the United Kingdom Hydrographic Office (UKHO), and they produce official charts for Kenya under the British Admiralty (BA). The area covering the port of Mombasa and its approaches is covered by the navigational chart BA666. A 30th May 2015 Edition 3 chart with a scale 1:12500 was used. The objective of this study was therefore to apply the optimization band ratio model to retrieve shallow water bathymetry for the Mombasa Port Channel and its approaches, and compare with officially charted depths.

Study Area

The coastal waters in and around Mombasa port and its environs (Fig. 1) were chosen as the site for the derivation of bathymetry from satellite imagery because of its importance to Kenya's maritime trade. The other reason was the availability of a high-resolution navigational chart for the area for referencing the SDB. The climate in Kenya is monsoon-influenced and there are two distinct seasons, namely the rainy and the dry seasons. The southeast monsoon (SEM) between April and July is characterized by long rains while the northeast monsoon (NEM) between October and November is characterized by short rains with a mean annual precipitation of 1144 mm (Verheyden *et al.*, 2005, originally in Lieth *et al.*, 1999). The Kilindini channel which hosts the Mombasa Port harbor is tide dominant but also receives seasonal inflow from the Mwache River during rainy seasons. Hydrodynamically, it is characterized by semi-diurnal tides with tide amplitudes of up to 3m with respect to the lowest astronomical tide (L.A.T).

Materials and Methods

In this paper, freely-available Landsat 8 satellite imagery were used because they were easy to obtain and readily available. Additionally, their 185 km swath coverage allows for processing images that cover larger areas; for example, only two images cover the entire Kenyan coast. The optimization model using band ratio calculation according to Pe'eri *et al.* (2014), Philpot *et al*. (2004), and Stumpf *et al.* (2003) was used to derive the shallow bathymetry as described in the introduction. Subsurface reflectance was transformed to spectral reflectance based on analytical equations for irradiance reflectance (W m^{-2} nm^{-1}), and remote-sensing reflectance according to Pacheco *et al.*

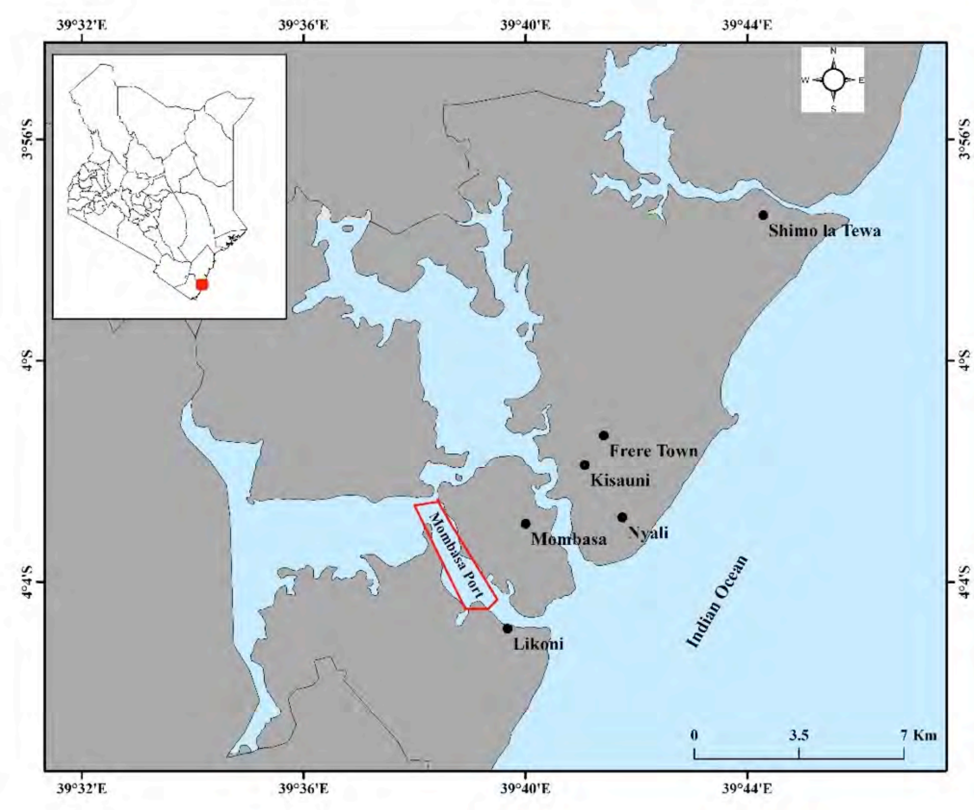

Figure 1. Mombasa Port channel and its approaches with the two distinct creeks, i.e. the southern Mwache Creek (Kilindini main channel of Mombasa Port), and northern Tudor Creek.

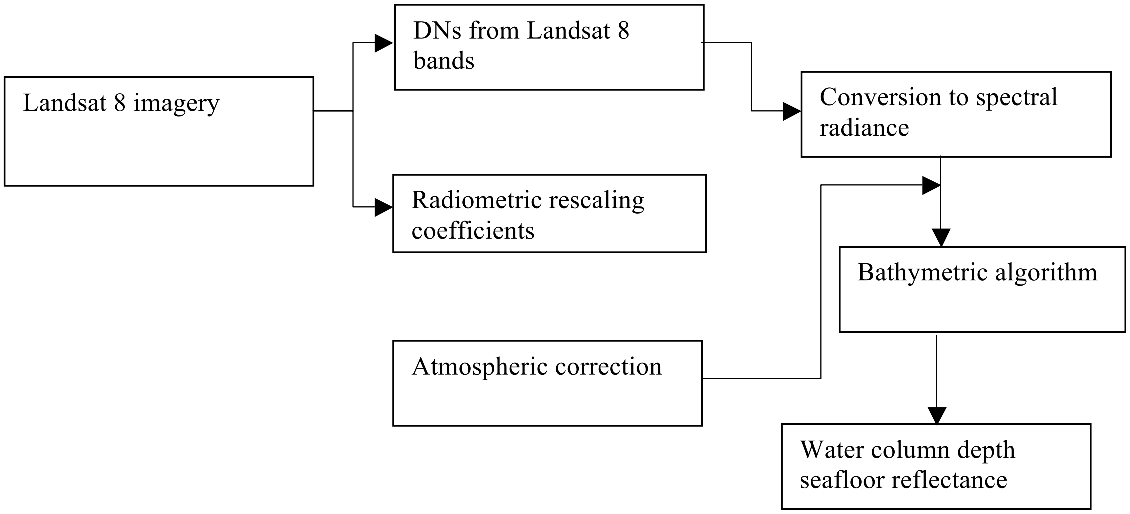

Figure 2. Workflow for deriving SDB maps from Landsat 8 images. This workflow is compatible with most accessible satellite data. (Modified from Pacheco et al., 2014).

(2014). The process of deriving bathymetry follows well-defined procedures from data (image) acquisition to referencing, as shown in the schematic workflow (Fig. 2).

Image Acquisition

Standard bulk Landsat 8 satellite images (band 1 to band 11) were downloaded from http://earthexplorer.usgs.gov/. Bands 1 to 7, and 9, have a spatial resolution of 30 m, while band 8 has a spatial resolution of 15 m, and bands 10 and 11 has a spatial resolution of 100 m (Pacheco *et al.*, 2014). Additionally, the standard bulk download of Landsat 8 imagery has an extra metadata file (MTL file) containing the radiometric Top of Atmosphere (TOA) rescaling coefficient, and this was also acquired. Satellite scenes from April 10, 2015 were downloaded considering earlier mentioned parameters (i.e. cloud free image and minimal sun glint). The satellite scenes downloaded covered the Kenyan south coast from Vanga to Kilifi, inclusive of the port

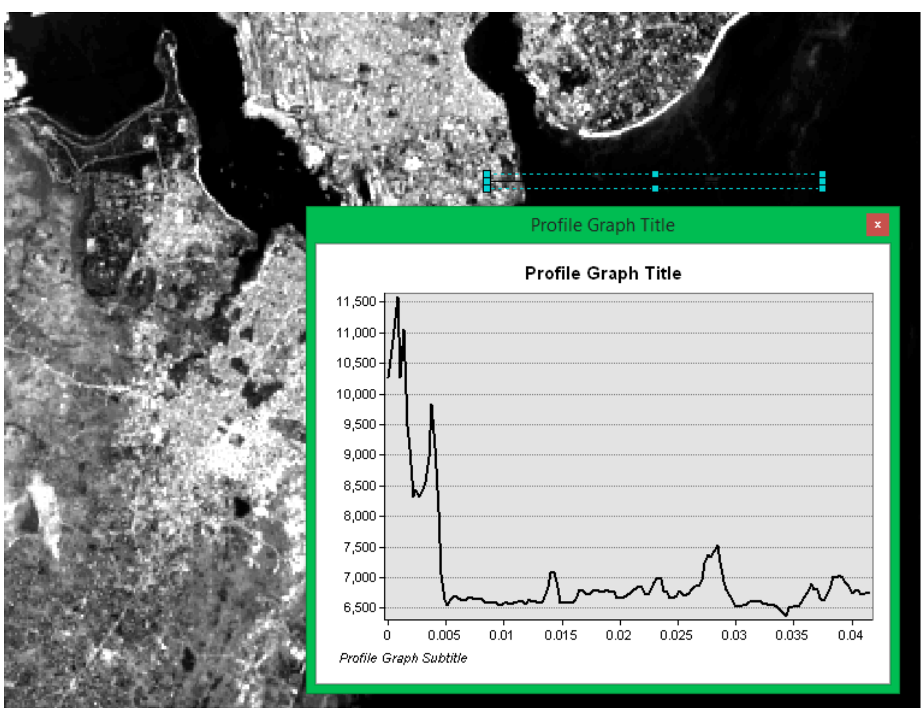

Figure 3. Land-Sea profile from a profile in NIR band with high (>6500-pixel) values corresponding to land and the low (<6500-pixel) values in the y-axis.

of Mombasa (Kilindini Harbour). The downloaded images were georeferenced to the WGS84 datum and projected to UTM zone 37S.

Preprocessing

A threshold depicting the transition between the land and water was defined by a combination of "identify" tool and the "profile" function in ArcMap toolbox. The profile method shows spiky sections representing land, and much smoother sections depicting water bodies (Fig. 3). The point of transition between the two regions signifies the threshold/cutoff value between land and water (Fig. 4). Additionally, the identify tool examines the pixel values both on land (high values) and water (low values), and since the water is close to opaque, it appears dark in the near-infrared (NIR), and bright on land due to reflection. From the selected values a threshold can be inferred from the transition between the low (water values) to high (land) pixel values. A low pass filter was applied to remove inherent radiometric noise in the imagery caused by "radiometric mal-adjustments" (Pe'eri *et al.*, 2013; Gao *et al.*, 2000) and brightness in the image (sun glint). Since the study site is situated in the tropics and is characterized by increased cloud coverage, cloud filtering was also performed as described in Pe'eri *et al.* (2014). Land, water and clouds are reflected well in near infrared (NIR), subsequently the histogram of the NIR band over a coastal area is bimodal, meaning the histogram will have two peaks; one corresponding to land and the other water. Furthermore, remaining cloud shadows and sun glint after masking and separation were removed using the Hedley *et al.* (2005) approach expressed in the equation below:

$$L'_{obs}(\lambda_i) = \left(\left(L_{obs}(\lambda_i)\right) - b_i \cdot \left(\left(L_{obs}(NIR)\right) - Min\left(L_{obs}(NIR)\right)\right)\right)$$

Where the pixel value in band i, $L_{obs}(\lambda_i)$ is a reduced by-product of the regression slope, b_i, and the difference between the pixel NIR value, $L_{obs}(NIR)$, and the ambient NIR level, $(Min(L_{obs}(NIR))$.

Deriving estimated bathymetry

The optimization band ratio model as described in Pe'eri *et al.* (2014) was then applied on the filtered images to estimate bathymetry by the ratios of blue and green bands. Specifically, the Stumpf *et al.* (2003) and Dierssen *et al.* (2003) log ratio model, which is a sub-category of the optimization band ratio model, was used. Based on the earlier described assumption of water column homogeneity, the difference of attenuation coefficient values (band ratio) will have a near-constant attenuation value. The use of two bands significantly reduces the number of parameters that would be required to compute the bathymetry (Stumpf *et al.*, 2003). Therefore, the variation in ratio between the bands with respect to depth can be deduced. In non-turbid shallow coastal waters, the light in the blue wavelengths attenuates faster than the light in the green wavelengths (Jerlov, 1976), and based on the fact that the two bands experience similar decay behavior, the model reduces errors associated with spectral reflectance in atmosphere,

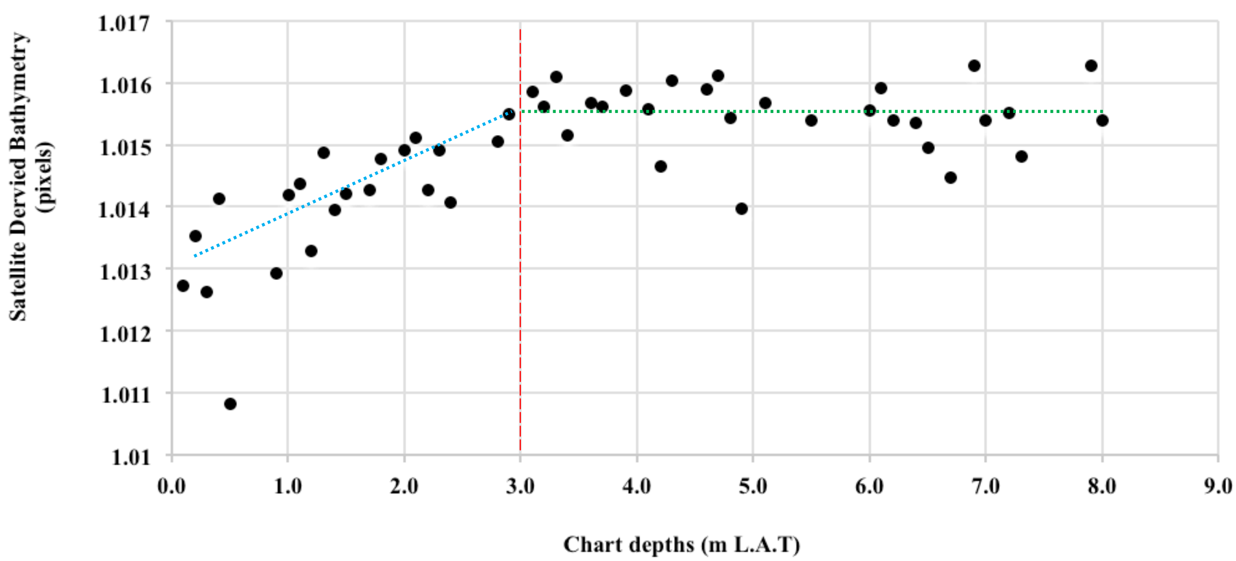

Figure 4. Scatter plot of chart depths and SDB (raster values) for the identification of the extinction depth (red dotted vertical line). In this case the ED is 3 m with visible inflection of curve from diagonal (blue dotted line) to horizontal (green dotted line).

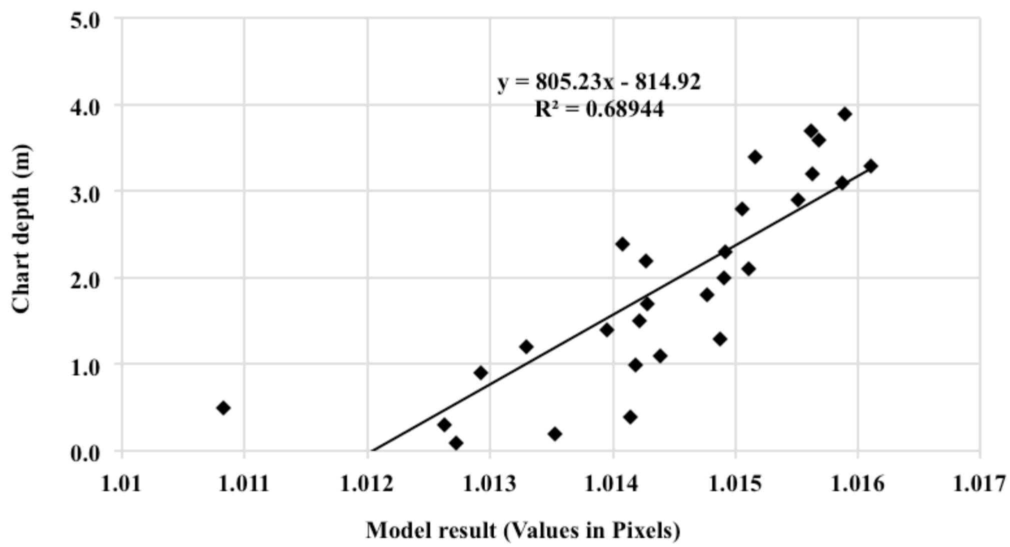

Figure 5. Regression analysis determining the gain and offsets for vertical referencing.

water column and sea bottom. The optimization band ratio model then estimates depth using reflectance of each spectral band, calculated with sensor calibration files and atmospheric artifacts corrected.

The optimization band ratio model is represented by the equation below;

$$Z = M_1 \left(\frac{\ln (nR_w(\lambda_1))}{\ln (nR_w(\lambda_2))} \right) - M_0$$

Where $R_W(\lambda_1)$ and $R_W(\lambda_2)$ are the pixel values corresponding to two reflectance bands λ_1 and λ_2 respectively. M_1 and M_0 are the gains and offset used to convert the algorithm result to depth relative to navigational chart datum. The value n is used to ensure that the logarithmic result is positive at all times, and the ratio subsequently produces a linear response.

Correlating the Landsat images with published chart soundings allowed for the determination of extinction depths, or the depth where bottom reflections affect the pixel values (typically in shallow waters). According to Pe'eri *et al.* (2014), only two points are adequate to determine the extinction depth, but several points are recommended for redundancy purposes (i.e. determination of "statistically optimal values of gain and offset"). Areas deeper than the extinction depth (equivalent to 2 secchi disk depths) are represented by value of water depth. The standard bathymetry algorithm has a theoretical derivation (Stumpf *et al.*, 2003; Dierssen *et al.*, 2003; Lyzenga, 1978), but also incorporates empirical tuning as an inherent part of the depth estimation process. According to available literature (Pacheco *et al.*, 2014; Lyzenga, 1985), it is preferable to minimize such tuning, particularly for remote regions where benthic and water quality parameters can be difficult to measure or estimate.

Referencing to a vertical datum

The algorithm results are referenced to a chart datum and it has been proven that shallower areas show linear correlation as opposed to the deep waters (Pe'eri *et al.*, 2014). This process also assists in checking the adequacy of navigational charts. The aforementioned linearity, or lack of it, can be determined by examining the regression coefficient (R^2) with values close to 1, signifying high correlation (Fig. 5).

Results

Vertical referencing and determination of extinction depth

This step required the determination of gain and offsets to be used in referencing the algorithm- derived bathymetry to the chart datum (see Pacheco *et al.*, 2014; Pe'eri *et al.*, 2014). The graph (Fig. 4) shows a gradual linear line at shallow depth that starts to curve and flatten out horizontally in deeper water. The exact point of inflection represents the extinction depth, which in this case was determined to be approximately 3 m (Fig. 4). However, this value could vary depending on the region and the variation can be attributed to variability in cloud cover and turbidity with time and area of interest. Variability could also arise from the fact that the extinction is based on the visual observation of the graph which varies from one person to another. From the scatter plot of chart depths obtained from digitizing the navigation chart (BA666) versus the raster values (Fig. 5), the slope (gain) and y-intercept (offset) values required

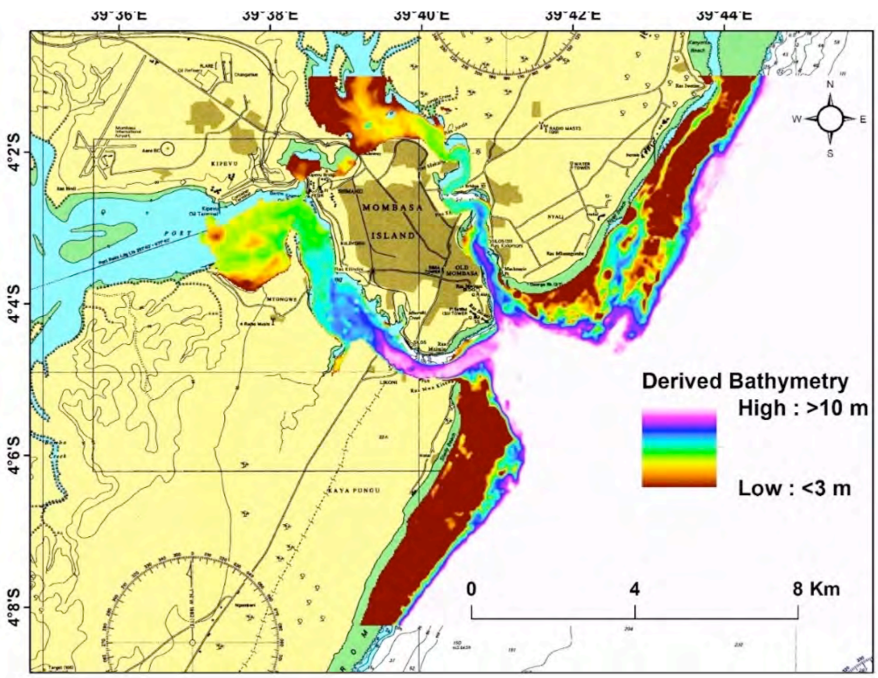

Figure 6. Derived shallow water bathymetry of Mombasa Port and its approaches.

for vertically referencing the SDB to chart datum were recorded as 805.23 and -814.92 respectively.

Derived bathymetry

The referenced SDB of Mombasa Port and its approaches is represented in Fig. 6. The depths generated showed a variation from as shallow as 5 m to depths greater than 10 m. This specific part of navigable areas of the sea present the highest risks to shipping, such as grounding. Selected charted depths and SDB showed great similarity (Fig. 7). For example, a charted depth of 2.2 m corresponded to a derived depth of 2.25 m. Furthermore, a linear regression analysis of the chart depth and the derived depths was

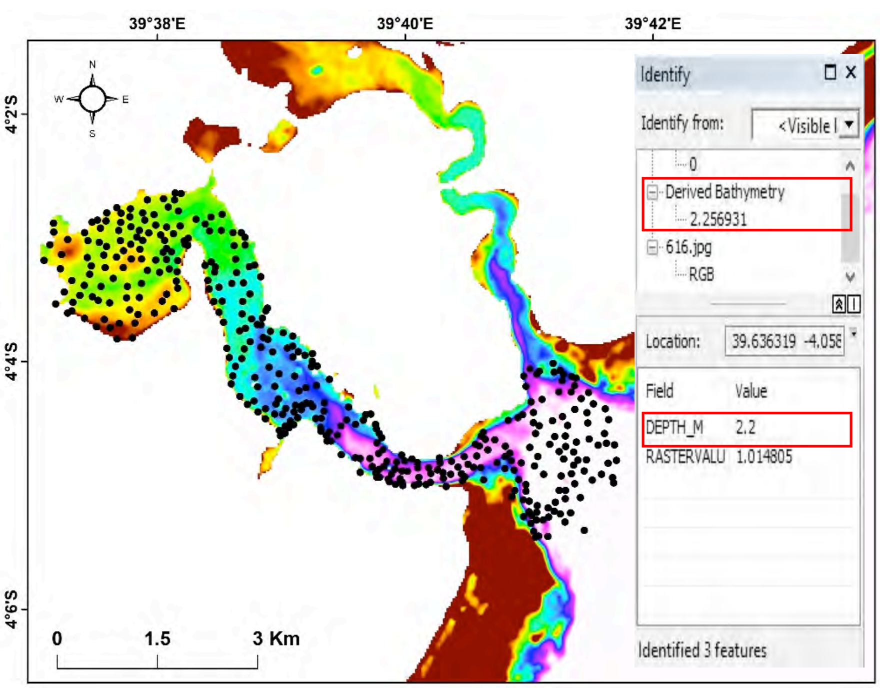

Figure 7. Comparison of derived bathymetry and the chart depths of Mombasa Port and its approaches. Inset is the identify tool in ArcMap showing the values of all visible layers; in this case the digitized water depth (black dots) and the derived depths.

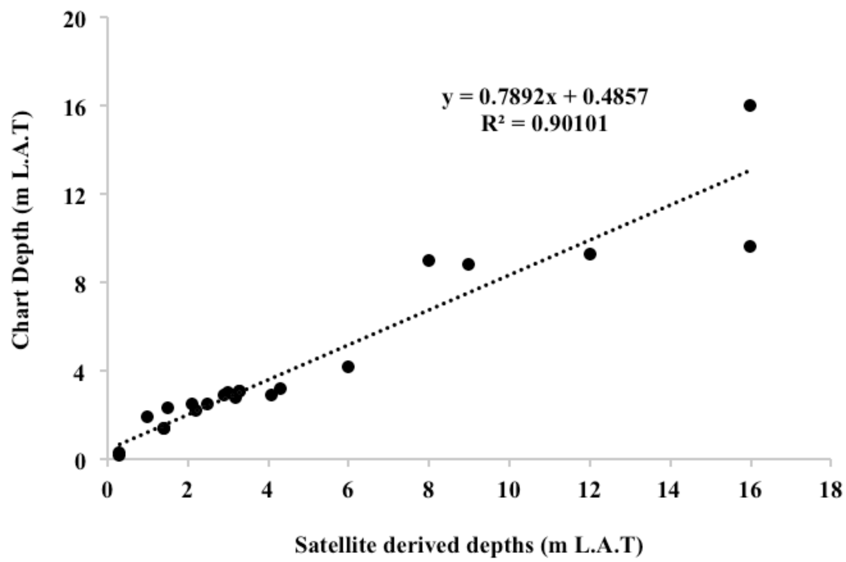

Figure 8. Regression graph of the chart depths (m L.A.T) vs derived depths (m L.A.T).

performed, and there was good linearity of the SDB versus the charted depths (Fig. 8), with a R^2 value of 0.90 and a root mean square error (RMSE) of 1.61 m.

Discussion

An excellent correlation between SDB and charted depths (R^2 =~90%) was obtained re-affirming the strength of analytical models in estimating bathymetry from satellite imagery. The results from this study are also similar to other previous studies (Jagalingam *et al.*, 2015; Pacheco *et al.*, 2015; Brando *et al.*, 2009). It is also worth noting that, except for Jagalingam *et al.* (2015), all the other studies did not use charted soundings to vertically reference their derived bathymetry. Pacheco *et al.* (2015) and Pe'eri *et al.* (2014) utilized airborne laser bathymetric (ALB) light detection and ranging (LiDAR). This shows that the band ratio model of retrieving bathymetry can be reproduced in a wide range of diverse environmental setting and at different times. Barber *et al.* (2016) used satellite imagery from different months and different locations and found that there was a good correlation (-/+ 1 m standard deviation). They also noted that even in remote areas, with favorable environmental conditions, bathymetry could still be retrieved. However, the main limitations of SDB is the reliance on good environmental conditions including water clarity (turbidity), depth, wave dynamics and meteorological conditions (e.g. presence/absence of clouds). To overcome some of these challenges, better estimation of bathymetry could be achieved by incorporating corrections for variables that could affect the estimation of bathymetry in the models. Some of the environmental variables that could cause errors in estimated bathymetry include the presence of optically active constituents in the water column (Pacheco *et al.*, 2015). These variables include chlorophyll-a, dissolved organic matter and particulate matter. SDB is suitable for calm waters with minimal turbulence because turbulence creates bubbles that are also optically active. These factors subsequently affect the derivation and accuracy of bathymetry (Laporte *et al.*, 2015; Pe'eri *et al.*, 2014; Louchard *et al.*, 2003; Lee *et al.*, 1999; Sandidge and Holver, 1989). For example, the low R^2 value observed could be attributed to a low-resolution hydrographic chart (Jagalingam *et al.*, 2015) and updates are recommended. Knowledge of these conditions, including their temporal and spatial occurrences, assists in a better understanding of bathymetry to be generated. SDB is modeled from light penetration and attenuation of variation in different spectral bands in water. Light penetration, attenuation and reflectance are inhibited and/or affected by the presence of suspended organic and inorganic materials. The depth of the seafloor can only be estimated to the extent of light penetration, and since turbidity will also lead to a "false shoaling" (Pe'eri *et al.*, 2014) incorrect bathymetry can be recorded. Suspended particles will therefore give a false reflection and jeopardize the accuracy of the generated bathymetry. Additionally, according to Lee *et al.* (1999), this technique avails itself to rapid data processing. However, it requires knowledge of a "few true depths for the regression parameters to be determined, and it cannot reveal in-water constituents". Therefore, the SDB described in this paper offers, among other benefits, a good coverage within depth and image limitations, and better depth resolution than traditional point-based methods such as lead line and singlebeam (SBES) acoustics.

Conclusion

From the statistics it can be concluded that the optimization band ratio model can retrieve depths in shallow waters. SDB offers a viable reconnaissance solution for areas of shallow coastal water where there is little or no existing hydrographic data, and no prospect of obtaining the resources required to proceed with extensive surveys using other higher accuracy methods in the foreseeable future. It is therefore a means of gaining information on bathymetry and reconnaissance where full-scale bathymetric mapping (by acoustic/sonar systems) cannot be carried out at a particular moment. It cannot therefore be used as a stand-alone hydrographic surveying tool, but rather as a support technique for hydrographic mapping of particular areas of interest.

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Sediment macro- and meiobenthic fauna distribution along the Kenyan continental shelf

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Abstract

Endobenthic fauna diversity, density and relative abundance were studied along the Kenyan continental shelf at four stations (Shimoni, Kwale, Mombasa and Kilifi) sampled during the *R/V Mtafiti's* maiden cruise between 12th and 21st December, 2015. A total of 16 macrobenthic taxa were recorded, with Amphipoda and Polychaeta dominating at all stations except Kilifi where no amphipods were recorded. Total densities of macrofauna ranged between 7 457 and 97 878 ind/m^2. The highest macrobenthic dominance (0.37), and the lowest H diversity and evenness (1.49 and 0.29, respectively) were recorded in Shimoni, while the highest H diversity values (2.18 and 2.11, respectively) were recorded at Kwale and Mombasa, respectively, and the highest evenness (0.74) at Kilifi. A total of 24 meiobenthic taxa were recorded, dominated by nematodes and copepods at all stations. Densities of meiofauna ranged between 96 and 1705 ind/10cm^2. The highest meiobenthic H diversities (1.67 and 1.63) were recorded at Shimoni and Mombasa, and the highest evenness (0.31) at Shimoni. The highest dominance (0.39), and lowest diversity (1.38) and evenness (0.21) were recorded at Kilifi. Sediment granulometry and total organic matter (TOM) displayed a north-south trend with the southern stations recording higher proportions of coarser sediments and low TOM, while the northern stations recorded higher proportions of finer sediments and higher TOM.

Keywords: Macrobenthic fauna, meiobenthic fauna, endobenthic, continental shelf

Introduction

The Western Indian Ocean is estimated to have a gross marine product of US$20.8 billion and an asset base valued at $333.8 billion. The main marine activities are based on fishing, maritime trade and marine resource use (Obura *et al.,* 2017). In Kenya, maritime commerce and tourism are the main economic activities directly related to the Indian Ocean in the urban centres (e.g. Mombasa) while artisanal fisheries and artisanal activities e.g. boat building are the main marine related activities in the Kenyan coastal rural areas (Kitheka *et al.*, 1998).

There are plans underway to increase economic activities along the Kenyan coast including the expansion of Mombasa port through dredging and the creation of new berths to increase the capacity of the port to handle more and larger ships, and the creation of a new port in Lamu to serve the hinterland in Ethiopia and South Sudan as part of a mega project (LaPSSET), and offshore oil and gas exploration activities. These developments necessitate baseline data and information on the status of the marine environment for continuous monitoring and evaluation of the impacts of these projects.

Most of these projects are likely to directly or indirectly impact on the seabed (UNEP, 2006) as most potential pollutants settle in the benthic zone. The benthic zone hosts benthic fauna which live an almost sedentary lifestyle with very close association with their environment, and are able to act as indicator species due to their sensitivities to specific changes in their environment (Ansari *et al.,* 2012; Tagliapietra and Sigovini, 2010; Josefson *et al.,* 2009; Theroux and Wigley, 1998). The species abundance patterns and community structure of these benthic invertebrates therefore provide important information on marine environmental quality (Josefson *et al.,* 2009).

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Benthic fauna can be categorized as either infauna (endo) or epifauna, with the former referring to those organisms that live in the sediments, while the latter refer to those organisms that live on the sediment surface. Generally, these benthic organisms are classified based on their size as either microbenthos (<0.063mm), meiobenthos (0.038-1.00mm), macrobenthos (>0.5mm), or megabenthos (>10mm) (Tagliapietra and Sigovini, 2010). Macro and meiobenthic invertebrates are among the main benthic fauna of interest in most continental shelf benthic studies.

Very limited information is available on the distribution, composition and ecology of the soft sediments and the fauna they support for the Western Indian Ocean's offshore regions (ASCLME/SWIOFP, 2012). Muthumbi *et al*. (2004) is among the recent studies conducted on the Kenyan shelf. This study reported the area as oligotrophic with higher organic contents recorded in the northernmost transect, which was attributed to the influence of the Somali upwelling system. Nematoda and Copepoda dominated the meiofauna component of the study.

The current study aims to provide more information on the status of the endobenthic communities of the Kenyan continental shelf.

Materials and methods

The study area was the Kenyan continental shelf of the Western Indian Ocean (WIO). Generally, the width of the Kenyan continental shelf varies, with the northern coast (north of Malindi) hosting a wider shelf compared to the southern coast (Schoolmeester and Baker, 2009; Kitheka *et al.,* 1998).

Four stations located along the shelf were identified and named based on the adjacent landward county, except for Shimoni. Replicate macrobenthic and meiobenthic samples were collected on board *R/V Mtafiti* during its maiden cruise between 12th and 21st December, 2015. Four stations, namely Shimoni Transitional Waters (Shimoni TW), Kwale (KTW), Mombasa (MTW) and Kilifi (KwTW) (Fig. 1) were sampled using a Van Veen grab (1000 cm^2) at water depths ranging between 15 and 80 m.

Sub-samples were taken using hand corers of 6.4 cm diameter for macrobenthos, and 3.4 cm diameter for meiobenthos and TOM. The sediment core depths of the samples were dependent on the depth of the grab samples hauled (Table 1). The samples were fixed in 4% formaldehyde solution and taken to the laboratory. No sediment samples for granulometry were collected during the maiden cruise and thus, samples for sediment grain size analysis were recovered from the samples for fauna analysis (detailed below).

Being the very first cruise, unforeseen challenges were encountered during benthic sampling. The main challenge that affected this research was the failure of the grab to haul complete samples (i.e. more than 10 cm deep sediment samples), even after several trials, and this affected sample collection for TOM analysis and sediment granulometry. No samples were specifically collected for sediment granulometry and for TOM analysis in Kwale.

Sediment granulometry was therefore undertaken using sediments that were recovered from the macrofauna samples during sieving, and after all the macrofauna individuals were collected during enumeration and identification. The sediments that passed through the 0.5 mm sieve and those trapped on the 2.00 mm sieve during macrobenthic sample preparation (sieving) were collected in a bucket. The sediments that were retained by the 0.5 mm sieve were recovered after the macrofauna individuals were identified and collected. These sediment portions were added to those collected from the 2.00 mm sieve and the filtrate from the 0.5 mm sieve, and they were then allowed to stand in the bucket until all the sediments had settled completely, decanted, air dried and then oven dried. Dry sieving, using a mechanical shaker, was then conducted using 2.00 mm, 1.00 mm, 0.5 mm, 0.25 mm, 0.125 mm, and 0.063 mm sieves. The sediments collected on these sieves were weighed and the relative grain distribution was calculated and classified based on the classification scale of Wentworth (1922). TOM samples were analysed by ashing, where the samples were oven dried at 70^{0}C until a constant weight was achieved. The samples where then ashed in a furnace for 6 hours at 500^{0}C, the ash free weight determined, and the difference between the two was used to calculate the TOM (Higgins and Thiel, 1988).

The samples for macrobenthic analysis were washed in the laboratory using tap water through a 2.00 mm sieve nested on a 0.5 mm sieve. The 2.00 mm sieve was used to remove larger material and organisms while the 0.5 mm sieve was used to retain and collect the macrobenthic fauna (Tagliapietra and Sigovini, 2010). The meiobenthic samples were washed through a 1.00 mm sieve nested on top of a 38 µm sieve with the 1.00 mm sieve used to remove larger material

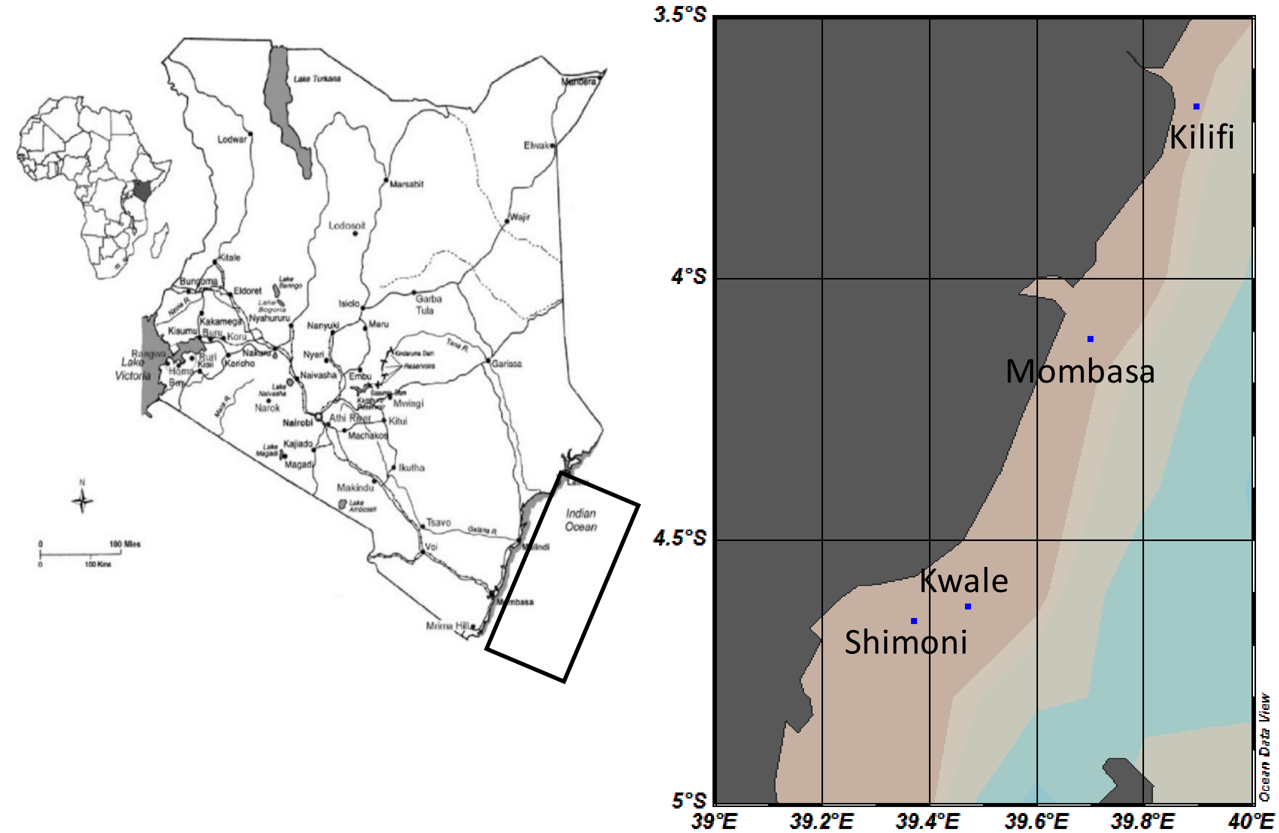

Figure 1. Map of study area showing the benthic sampling sites along the Kenyan shelf.

and larger organisms from the meiofauna samples. The meiofauna samples retained on the 38 µm sieve were then put in centrifugation tubes using $MgSO_4$ solution as the media (specific density of 1.28) and centrifuged to extract the meiobenthos from the sediment by density separation (Higgins and Thiel, 1988). Each sample was centrifuged three times and the supernatant was collected on the 38 µm sieve after each centrifugation and rinsed off the sieve using water. The samples were then preserved using 4% formaldehyde solution for further analysis. All samples (macrofauna and meiofauna) were stained in Rose Bengal solution to aid in the identification and sorting of fauna. Identification was carried out using Higgins and Thiel (1988).

Data analysis

The data was recorded and analysed for relative abundance and density in an Excel spreadsheet, with diversity indices calculated and the diversity *t* test carried out using the Paleontological Statistics Software package (PAST v2.17c) (Hammer *et al.*, 2001). Community analysis was carried out using Plymouth Routines in Multivariate Ecological Research (PRIMER v5.2.9) (Clarke and Gorley, 2002), where Bray-Curtis similarity, Analysis of Similarity (ANOSIM) and Similarity Percentages (SIMPER) were used to compare the similarities within stations, and among stations. Dendrogram and Multidimensional Scale (MDS) plots were also plotted using PRIMER based on Bray-Curtis

Table 1. Location of sampling stations and their depths along the Kenyan continental shelf.

Station	Code	Longitude	Latitude	Water Depth (m)	Sediment Depth (cm)	Number of replicates
Shimoni	ShTW	039'22.33 E	04'39.20 S	19	10	3
Kwale	KwTW	039'28.19 E	04'37.39 S	48	3	3
Mombasa	MTW	039'41.87 E	04'06.89 S	78	3	3
Kilifi	KTW	039'53.68 E	03'40.17 S	38	5	3

Simlarity. Statistical Package for the Social Sciences (SPSS) (IBM Corp. Released 2011. IBM SPSS Statistics for Windows, Version 20.0. Armonk, NY: IBM Corp.) was used to analyse for significant differences in the densities between stations using Analysis of Variance (ANOVA) and the subsequent post-hoc analysis was carried out using Tukey's Honestly Significant Difference (HSD). SPSS was also used to create the graphs.

Results

Abiotic parameters

The grain size analysis displayed a north-south trend, with the highest proportions of coarse sand (1-0.5 mm), very coarse sand (1-2 mm) and pebbles (>2 mm) recorded in the southernmost stations (Shimoni and Kwale), and their proportions reducing northwards where higher proportions of fine (0.25-0.125 mm) and very fine sand (0.125-0.063mm) were recorded (i.e. Mombasa and Kilifi). The highest median sand fraction was observed in Mombasa and Kwale (Fig. 2).

The total organic matter (%TOM) ranged between 2.86% and 3.59%, with Kilifi recording the highest TOM (3.59%), and Shimoni the lowest (2.86%), while Mombasa stations had 3.17% (Fig. 3). No samples for TOM analysis were collected for the Kwale station due to the challenges stated above.

Macrobenthic composition and abundance along the Kenyan Shelf

The densities varied significantly along the shelf stations (F (3,8)=15.003, p=0.001). Tukey's post hoc analysis indicated that Shimoni and Mombasa each differed significantly from Kwale (p=0.5 and p=0.021, respectively) and Kilifi (p=0.004 and p=0.002, respectively), while there was no significant difference between Shimoni and Mombasa, nor was there a significance difference between Kwale and Kilifi. The highest density was recorded at Mombasa (85 864±9 324 ind/m^2) despite the corer depth falling short of the recommended 10 cm depth. The second highest density was recorded at Shimoni (78 095±23 024ind/m^2), followed by Kwale (40 083±11 655ind/m^2) and finally Kilifi (16 261±10 137ind/m^2) (Fig. 4).

Sixteen macrobenthic taxa were recorded with Amphipoda and Polychaeta being the most abundant taxa over all, followed by Ostracoda, Tanaidacea, and Nematoda. The Amphipoda had high abundance at Shimoni (58%) and Mombasa (27%) and was the third most abundant at Kwale (12%), but were not recorded at Kilifi station. Polychaeta was the most abundant taxon in Kwale (30%) and Kilifi (26%), but less abundant at Mombasa (22%) and Shimoni (15%). Ostracoda showed similar abundance at all stations (15% at Kwale, 12% at Shimoni, 12% at Mombasa, 12% at Kilifi). Tanaidacea was the second most abundant taxon at Kilifi (16%) followed by Nematoda (13%). Other taxa with recognizable abundance during this study included Copepoda, Isopoda, Oligochaeta, Tubellaria, Gastropoda and Cumacea. Nemertina, Sipuncula, Gnathostomulida, Echinodermata and Halacaroidea were the least abundant taxa and were grouped as 'others' (Fig. 5).

Diversity of Macrobenthos along the Kenyan continental shelf

Shannon diversity t test (pairwise) indicated that Shimoni significantly differed from all the other

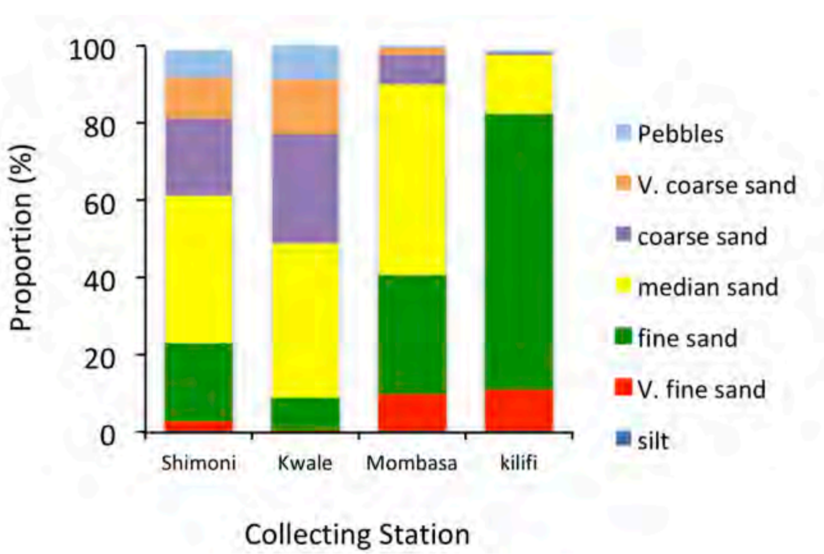

Figure 2. Granulometry distribution along the Kenyan continental shelf.

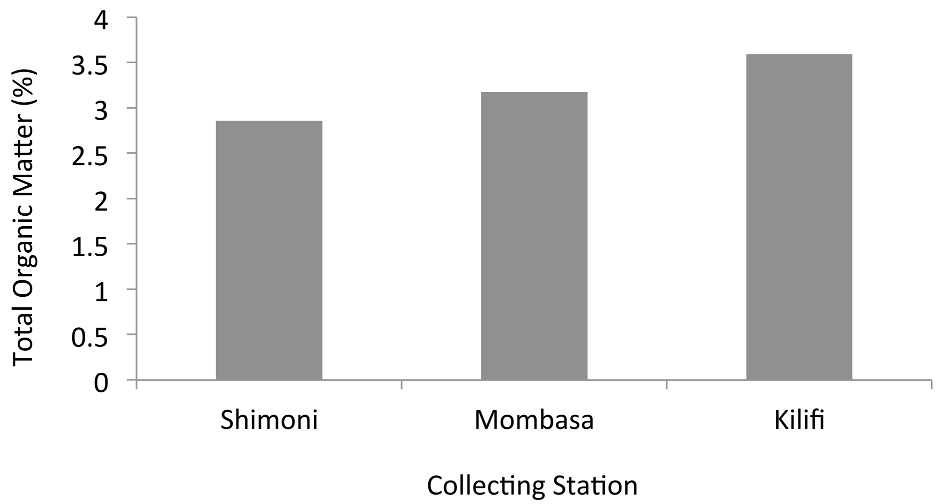

Figure 3. Percentage of total organic matter recorded from the Kenyan continental shelf.

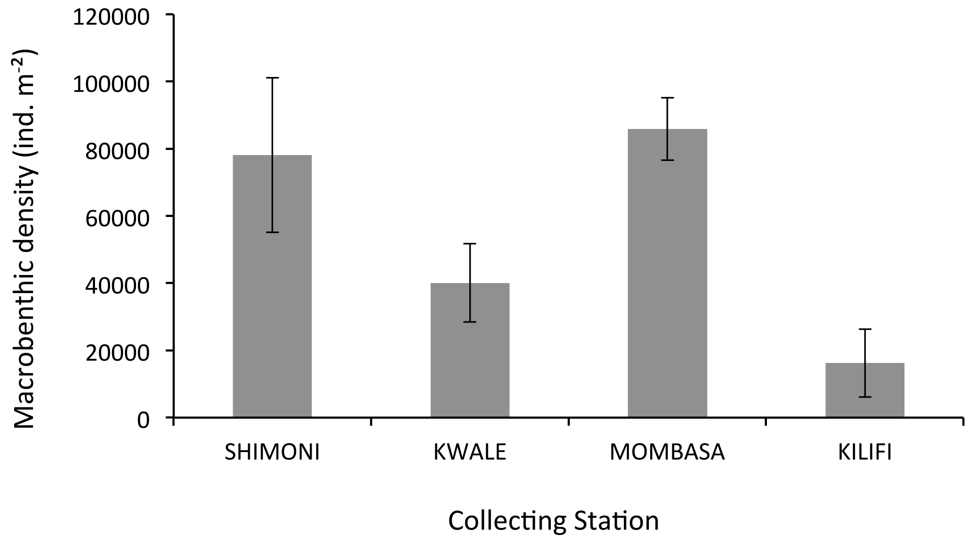

Figure 4. Macrobenthic densities from the stations along the Kenyan continental shelf.

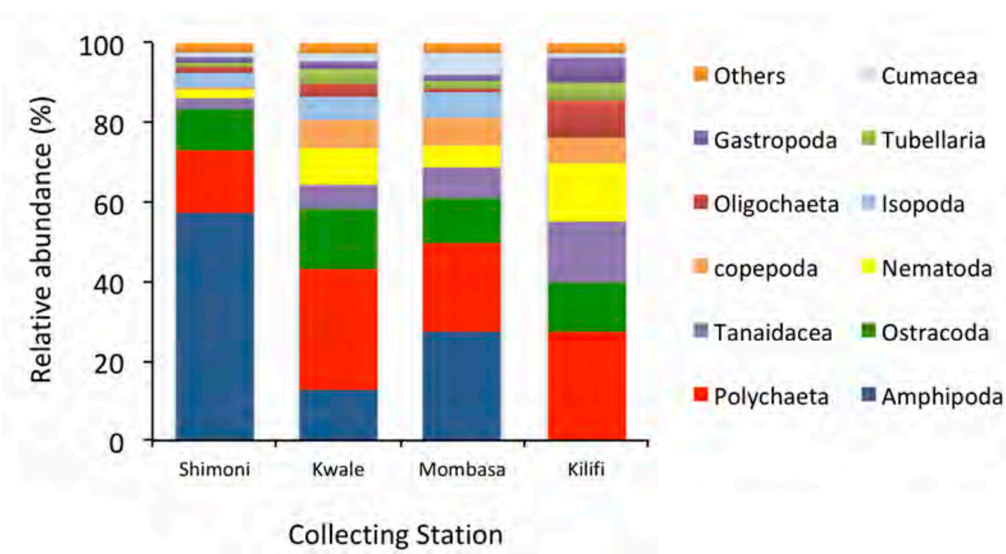

Figure 5. Macrobenthic relative abundance along the Kenyan continental shelf.

stations (Kwale ($t_{1052.9}$= 10.3; p<0.0001), Mombasa ($t_{1292.4}$=11.0;p<0.0001) and Kilifi ($t_{432.04}$=8.1;p<0.0001)).

Kwale had the highest H diversity (2.18) and the lowest dominance (0.16) while Shimoni had the highest dominance (0.37) and the lowest diversity (1.49) and evenness (0.3). Kilifi had the highest evenness (0.74), while Mombasa had the second highest H diversity (2.11) and evenness (0.59) (Table 2).

Community analysis of macrobenthic fauna along the Kenyan continental shelf

Analysis of Similarity (ANOSIM) indicated a high difference among the stations (R=0.756). Further analysis using SIMPER revealed low similarities between Shimoni and Kilifi (25.98%), Mombasa and Kilifi (28.59), while medium similarities were observed between Kwale and Kilifi (47.83), Shimoni and Kwale (47.75%), Kwale and Mombasa (59.55%), and Shimoni and Mombasa (62.32%). The within stations similarity was high in Mombasa (84.89%), Kwale (74.16%) and Shimoni (70.5%), and least in Kilifi (53.16%) (Fig. 6).

These similarity percentages resulted in Mombasa communities clustering together in the Multidimensional Scaling (MDS) due to the high similarity within the station (84.89%), and was also closer to Shimoni and Kwale, while Kilifi was furthest from the other stations, and its stations were also further apart due to the low similarity of the communities within the station (53.16%) and the other stations (Fig. 7).

Meiobenthic composition and abundance along the Kenyan continental shelf

The meiobenthic densities differed significantly among the stations according to ANOVA analysis

Table 2. Meiobenthic fauna tally; density, mean±SD, and diversity recorded from the stations along the Kenyan continental shelf.

	Shimoni	Kwale	Mombasa	Kilifi
	Mean±SD			
Amphipoda	145.3±50.8	15.7±6	75.3±18.6	0
Polychaeta	37±14	38.7±9.7	61.7±5.5	13.7±9.9
Isopoda	9±3	8.0±5.6	17.3±2.3	0
Tanaidacea	7.7±3.8	7.7±1.2	20±6.6	8.3±6.7
Oligochaeta	3.7±1.2	4±1.7	2.3±0.6	4±2
Ostracoda	28.7±36.7	19±3.5	32±3.6	6.3±5.5
Nematoda	4.3±2.5	12.3±5.9	15±4.6	7±3
Cumacea	2.7±0.6	2.3±1.5	15.3±0.6	1±1.7
Nemertina	3.7±2.1	1.3±1.5	4.3±2.1	1.3±1.2
Tubellaria	2.3±0.6	4.7±1.2	5.7±3.2	3.3±4.2
Sipuncula	0.3±0.6	0.7±1.2	0	0.3±0.6
Gastropoda	3.7±0.6	2.3±0.6	3.7±2.1	3.3±2.1
Gnathostomulida	0.7±1.2	0	0.7±0.6	0
copepoda	1.0±1.0	10.7±11	20.7±12.1	3.7±2.5
Echinodermata	1.3±1.2	1.3±1.5	2.3±2.3	0
Halacaroidea	0.0	0.3±0.6	0	0
Density	78095.4±23023.7	40083.5±11655.3	85863.5±9323.5	16261.2±10137.1
Taxa_S	15	15	14	11
Dominance_D	0.3725	0.1528	0.1593	0.146
Shannon_H	1.493	2.18	2.112	2.103
Evenness_e^H/S	0.2967	0.5898	0.5905	0.7444

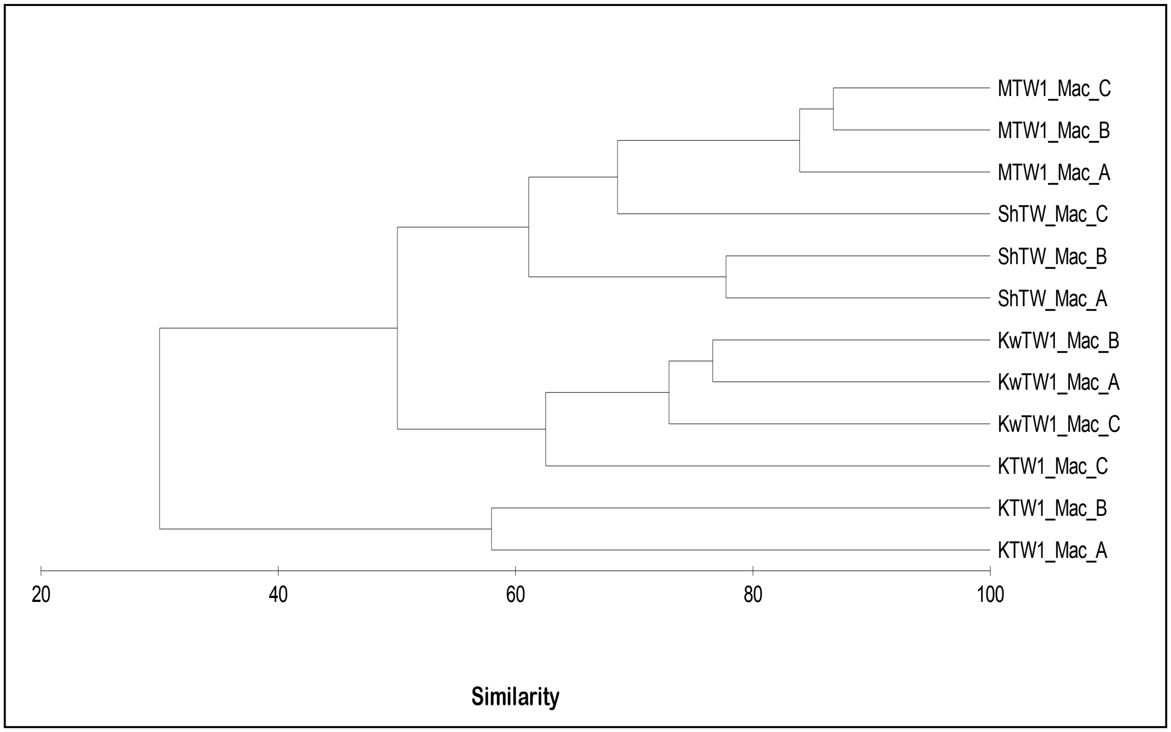

Figure 6. Dendrogram based on the Bray-Curtis Similarity analysis on the Macrobenthic fauna. MTW stands for Mombasa station, ShTW represents Shimoni Station, KwTW represents Kwale station, while KTW stands for Kilifi station. Mac represents Macrofauna samples, while (A-C) represents the replicate symbol.

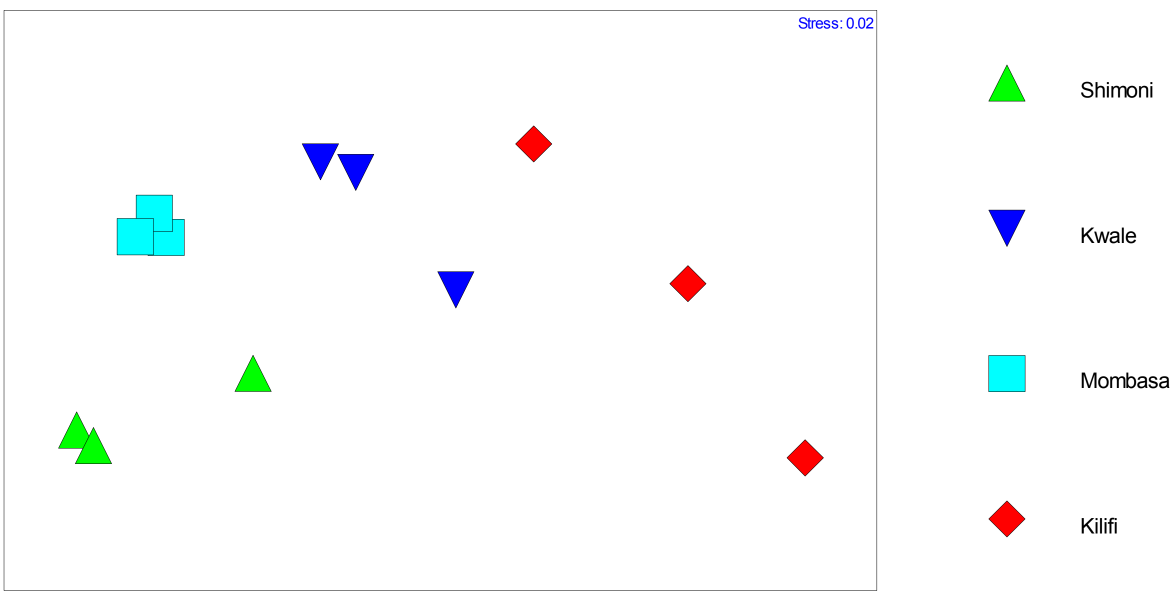

Figure 7. Multidemensional scale (MDS) plot on the macrofauna communities along the Kenyan shelf based on Bray-Curtis similarity.

($F(3,8)=19.397$, $p=<0.001$). Post-hoc analysis revealed that Mombasa's density was significantly different from all the other stations (Shimoni, $p=<0.001$; Kwale, $p=0.003$; Kilifi, $p=0.042$). Shimoni's density was significantly different to Kilifi ($p=0.029$) but not significantly different to Kwale ($p=0.229$), while Kwale and Kilifi had no significant differences between them ($p=0.483$). The highest meiobenthic density was recorded at Mombasa (1 286±365 ind/10 cm^2), despite the corer depth falling short of the recommended 10cm depth, and was almost double that of the second highest density in this study (i.e. Kilifi, 696±92ind/10 cm^2). The second lowest density was recorded at Kwale and the lowest in Shimoni (463±25 and 134±63ind/10 cm^2, respectively), despite achieving the recommended 10 cm core during sub-sampling (Fig. 8).

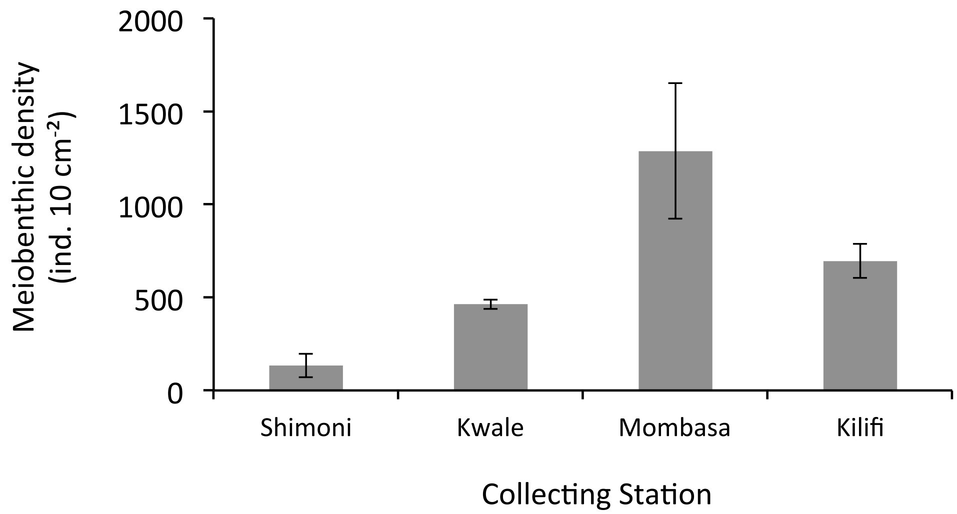

Figure 8. Meiobenthic density along the Kenyan continental shelf.

A total of 23 meiobenthic taxa were recorded and were dominated by Nematoda and Copepoda. The Nematoda had the highest relative abundance in Shimoni (43%), in Mombasa (46%) and Kilifi (58%), while Kwale was dominated by Copepoda (42%). Amphipoda was the second most abundant taxon in Shimoni (21%), while Polychaeta had the third highest abundance in Kwale (12%). Most of the other taxa (Tanaidacea, Isopoda, Halacaroidea, Priapaulida, Oligochaeta, Cumacea, Rotifera, Sipuncula, Gastropoda, Gnathostomulida. Pycnogonida, Kinorhyncha, Bivalvia, Echinodermata) recorded relative abundances of less than 3% in all stations and were grouped as 'others' (Fig. 9).

Meiobenthic diversity along the Kenyan continental shelf

The Shannon diversity t test indicated that there was no significant difference among the three stations with the highest H diversity; i.e. Shimoni, Mombasa and Kwale (Shimoni and Kwale ($t_{687.68}$= 1.36; p=0.175), Shimoni and Mombasa ($t_{511.5}$= 0.401; p=0.688), and Mombasa and Kwale ($t_{2560.6}$= 1.715; p=0.087)). However, all these stations differed significantly with Kilifi which recorded the lowest H diversity; Shimoni ($t_{619.8}$= 4.407; p<0.0001), Kwale ($t_{3157.8}$= 4.4873; p<0.0001), and Mombasa ($t_{4266.9}$= 7.47; p<0.0001)). The highest H diversity was recorded at Shimoni (1.67), albeit marginally, followed by Mombasa and Kwale

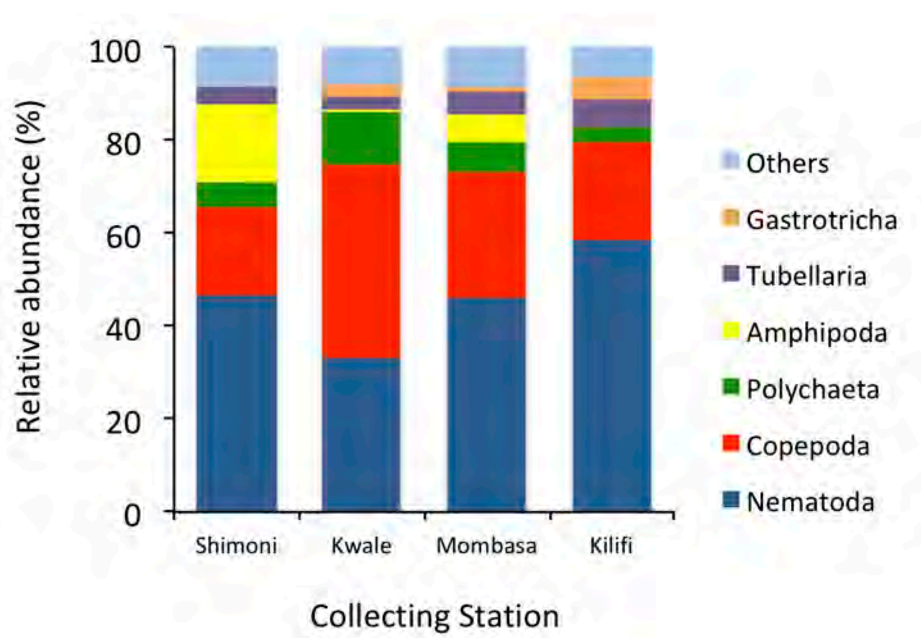

Figure 9. Meiobenthic relative abundance along the Kenyan continental shelf.

Table 3. Meiobenthic fauna tally; density, mean±SD, and diversity recorded from the stations along the Kenyan continental shelf.

	Shimoni	Kwale	Mombasa	Kilifi
	Mean±SD			
Nematoda	58.7±10.7	155.7±21.2	596±139.9	410.7±31.5
Copepoda	24.7±7	195.7±7	367.7±176.3	154±40.6
Polychaeta	8.3±7.5	54.7±2.5	83±23.6	22.7±1.5
Tubellaria	4.7±2.3	14.3±2.3	57.7±28.0	41±6.2
Amphipoda	28.3±33.5	1.7±1.2	80.3±25.1	0.3±0.6
Gastrotricha	0	11±2.6	20.7±9	32.7±5.7
Ostracoda	1.3±0.6	8.3±0.6	29.3±8.1	2±1
Nemertina	1±1	5±1	9±2.6	7±5
Isopoda	1±1.7	1.3±1.5	19±2	0
Tanaidacea	0.7±1.2	2.3±0.6	14.7±5	3.7±3.1
Tardigrada	3.7±2.5	8.7±5.7	7±2.6	1±1.7
Priapaulida	0	0	0	13±6.1
Cumacea	0.3±0.6	1±0	10.7±3.1	0.3±0.6
species 1	0	0	0	12±9.2
Halacaroidea	1±1	2.3±2.1	4.3±4.0	0
Rotifera	0.3±0.6	1.7±2.1	3±2	2.3±1.5
Oligochaeta	0.3±0.6	5.3±4	0.7±1.2	0.3±0.6
Gnathostomulida	0.7±1.2	1±1	0.3±0.6	2.3±1.2
Bivalvia	0	0	2.7±1.5	0.3±0.6
Gastropoda	0.3±0.6	0.3±0.6	1.7±1.5	0
Kinorhyncha	0	0.7±0.6	1.3±1.5	0
Sipuncula	0.7±1.2	0	0.3±0.6	2.7±2.5
Pycnogonida	0	0.3±0.6	0	0
Echinodermata (brittle star)	0	0	0.3±0.6	0
Density	133.6±62.9	462.9±24.7	1286.1±364.7	695.9±92.4
Taxa_S	17	19	21	19
Dominance_D	0.2684	0.2973	0.2969	0.3903
Shannon_H	1.674	1.573	1.633	1.382
Evenness_e^H/S	0.3138	0.2538	0.2438	0.2097

(1.63 and 1.57, respectively). The highest evenness (0.31) and lowest dominance (0.27) was recorded at Shimoni, while the highest dominance (0.39) and the lowest H diversity (1.38) and evenness (0.21) was recorded at Kilifi.

The meiobenthic community analysis based on ANOSIM portrayed a high separation among the stations (R= 0.914). Upon further analysis using SIMPER, it was shown that there was low similarities between Shimoni and all the other stations (Mombasa (18.87%), Kilifi (24.19%) and Kwale (34.87%)), while the other stations had modest similarities among their communities (Mombasa and Kilifi (66.68%), Kilifi and Kwale (62.65%), and Kwale and Mombasa (53.14%)) as summarized in the dendrogram (Fig. 10).

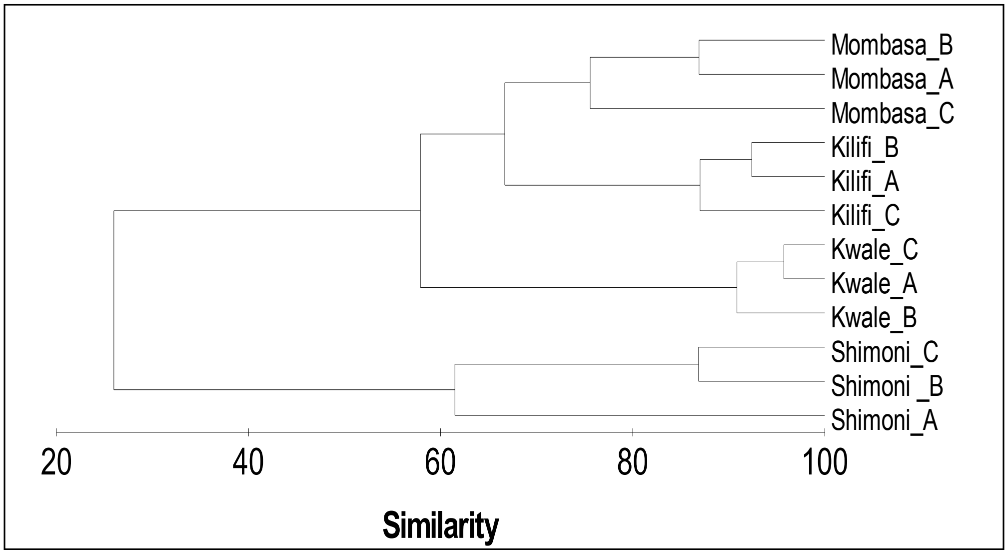

Figure 10. Dendrogram based on the Bray-Curtis similarity on the meiobenthic communities from the Kenyan continental shelf. (A-C) represents the replicate samples from the corresponding stations.

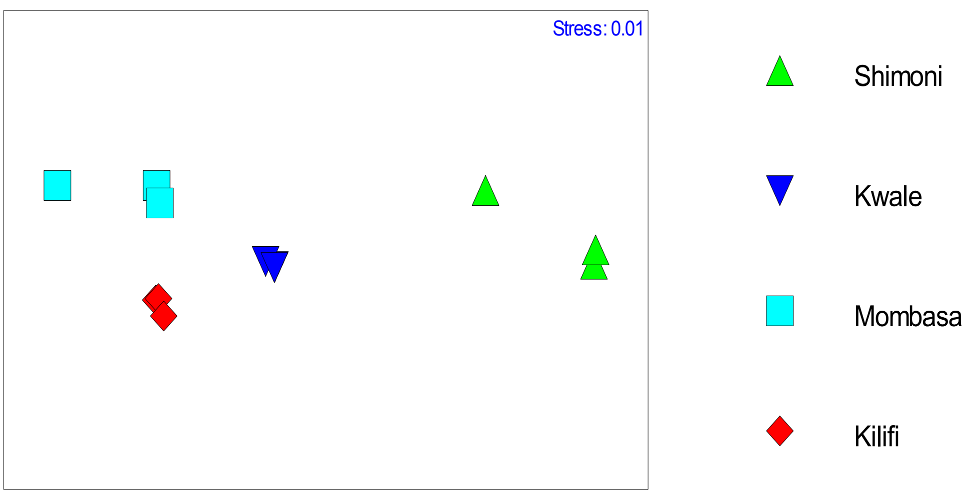

Figure 11. MDS plot of the meiobenthic communities along the Kenyan shelf.

The MDS plot showed clustering of points from the same stations, which, based on the results from SIMPER, indicated high levels of similarities within the stations, especially Kwale (92.49%), Kilifi (88.81%) and Mombasa (79%), while Shimoni's community had above average similarity among its communities (69.95%) (Fig. 11).

Discussion

The abiotic parameters portrayed a north-south trend as the southern stations had higher proportions of coarse particles which progressively reduced northwards, while the TOM values reduced southwards. The decantation method used to retrieve sediments from the macrofauna samples for sediment granulometry analysis appears to be an effective improvisation, as the results were similar to those reported by Muthumbi *et al.* (2004), where the north-south trend in grain size distribution was also reported.

This north-south trend can be explained based on the morphology of the Kenyan continental shelf which has been described as being wider in the northern parts and narrows southwards (Kitheka *et al.*, 1998). Narrower shelves have higher rates of erosion compared to wider shelves (Harris and Wiberg, 2002). Sediment deposition on the southern coast of the Kenyan continental shelf is also low due to the absence of permanent rivers draining into the ocean, resulting in lower proportions of finer sediments.

The northern coast, on the other hand, has two permanent rivers (Tana and Sabaki) resulting in higher rates of sediment deposition, especially silt and clay, and the wider shelf with gentle gradient reduces the rate of erosion (Fennessy and Green, 2016: Muthumbi *et al.,* 2004).

The TOM trend was similar to the granulometry trend, reducing southwards, and can be related to the grain size, as TOM in many cases tends to positively correlate with the proportion of fine grain size fraction compared to the coarser grain fraction (Secrieru and Oaie, 2009). The high TOM in the northern stations could also be a result of the influence of the Somali upwelling on the northern Kenyan shelf, in addition to the inputs of allochthonous organic matter through the permanent rivers (Muthumbi *et al.,* 2004).

The macrofaunal densities were high compared to most shelf studies (e.g. Wang *et al*., 2014; Gerdes and Montiel, 1999; Gupta and Desa, 2001). These high densities were reported despite most of the stations in this study failing to attain the recommended 10 cm deep cores during sub sampling. In fact, the station with the least core depth (Mombasa, 3 cm) recorded the highest density. This can be explained by the fact that most benthic infauna are located at the upper 2 cm of the sediment (Higgins and Thiel, 1988). Also, the high densities seemed to be a direct result of the high Amphipod abundance, with stations with high Amphipoda abundances recording the highest overall macrofauna densities (Mombasa and Shimoni), while Kilifi station, which had no Amphipods, recorded the lowest overall macrofauna density.

Carvalho *et al.* (2012), and Theroux and Wigley (1998) recorded high Amphipoda abundance in areas with medium and finer grain-size compared to coarser grain fractions, in their studies. Similarly, in this study, the stations with higher proportions of median and fine sand fractions recorded high Amphipoda abundance, implying that a combination of both fine and median sand influences occurrence of Amphipoda. The areas with high abundances of Amphipoda were also observed to have high abundance of Foraminifera (pers. obs.), which were not included in this study as they are not metazoan fauna. Foraminifera have a tendency to aggregate around settling phyto-detritus as their food source (Gooday and Turley, 1990; Higgins and Thiel, 1988), which is also the main food source for the Amphipods, and may explain their high abundances at these stations.

Polychaeta were the most dominant taxa in other macrofaunal studies (Wang *et al.,* 2014; Joydas and Damodaran, 2007; Gerdes and Montiel, 1999). In this study, they were in high abundance in Kilifi, Mombasa and Kwale. Kwale's high polychaeta abundance can be attributed to the nature of the station, having higher proportions of coarse grain size as compared to the other stations, implying strong wave action and currents which have an impact on the macrobenthic communities. Areas with high tidal currents and wave action have higher proportions of coarse clean sand fractions favouring filter feeders; most of which are polychaetes species as reported by Dutentre *et al.* (2013).

Musale and Desai (2011) related the abundance of macrobenthic polychaetes to grain size and organic matter along south Indian coast, and found that Polychaeta abundance was higher in areas with loose textured sediments (high sand and sandy silt), and that polychaetes preferred low organic matter habitats and avoided high organic carbon areas. This could explain the high relative abundance of polychaetes in Kwale and low counts in Kilifi. Kilifi had low Polychaeta counts compared to all the other stations and their relative abundance was only high due to the absence of most of the other taxa such as Amphipoda which dominated in Shimoni and Mombasa. The reverse can also be used to describe the lower relative abundance of Polychaeta in Shimoni and Mombasa. The average count of Polychaeta in Mombasa was higher than that of Kwale, while the average count in Shimoni was close to that of Kwale. This implies that the lower Polychaeta relative abundances in Shimoni and Mombasa was a result of high abundance of Amphipoda in these stations, and not low counts of the polychaetes *per se*.

Macrobenthic diversity has been reported to be affected by sediment grain size. Van Hoey *et al.* (2004) recorded higher species richness, abundance and diversity in fine to medium sandy sediments, similar to those reported in Mombasa and Kwale, where the highest H diversities were observed. On the other hand, Dubois *et al.* (2011) reported low diversity in coarse sediments, as observed in Shimoni, where the lowest H diversity was recorded.

Meiobenthic densities recorded in this study were lower than most continental shelf meiobenthic studies such as Sandulli *et al.* (2010), Grémare *et al.* (2002), Huys *et al.* (1992).This could be a result of food availability which is among the main factors regulating meiobenthic densities (Higgins and Thiel, 1988). The Kenyan shelf is described as an oligotrophic shelf (Muthumbi *et al.*, 2004), therefore it has low TOM, resulting in the low densities found in the present study. These densities fall within the range described by Soltwedel (2002) for tropical arid regions.

The meiobenthic fauna was dominated by Nematoda and Copepoda, which usually dominate in meiobenthic studies (Higgins and Thiel, 1988). Nematoda had the highest relative abundances at all stations except Kwale, where copepods had the highest abundance. Copepods have been observed to dominate in coarse grained sediments (Sajan and Dramodaran, 2007; Higgins and Thiel, 1988), as was the case in Kwale.

The relative abundance of Nematoda at the other stations fell within the range described by Soltwedel (2002) for tropical and arid regions of north eastern Africa, which were mainly influenced by sediment grain size and food availability in the form of TOM. This abundance therefore matches the study site which is categorized as being oligotrophic (Muthumbi *et al*., 2004), and can partly explain the nematode abundance patterns in the study area.

Other meiobenthic taxa usually occurin small numbers (Soltwedel, 2002) as observed in this study, however, it is important to note the high Amphipoda abundances recorded in Shimoni and Mombasa which corresponded with the high Amphipoda abundance in the macrobenthic fauna. Most juveniles of macrobenthic fauna fall within the meiofauna category, resulting in this high Amphipoda abundance in the meiobenthic fauna in Shimoni and Mombasa (Higgins and Thiel, 1988).

Meiofaunal H diversity indices were relatively higher in the coarse sand and medium sand dominated stations (Shimoni, Kwale and Mombasa) compared to Kilifi, which had very high proportions of very fine sand. Similar findings have been reported by Sandulli *et al.* (2010), where the stations with high proportions of coarse and medium sand fractions recorded the highest abundances compared to the stations with fine sand. This can be attributed to the increased habitat heterogeneity in the medium and coarse grained sediments, increasing various niches, and therefore increasing diversity (Higgins and Thiel, 1988). On the other hand, the slightly higher TOM in Mombasa may explain the slightly higher diversity and density compared to the other stations. The high dominance value in Kilifi could be due to the high abundance of Nematoda compared to the Copepoda. This may be due to the high proportion of finer sediments which favour nematodes over copepods (Sajan and Damodaran, 2007; Higgins and Thiel, 1988).

Conclusion

This study is a product of the maiden cruise of *R/V Mtafiti* which aim to provide information about the benthic community assemblages on the Kenyan shelf. Despite the various challenges faced, the results of the study agreed with earlier studies conducted along the Kenyan shelf, specifically with Muthumbi *et al*. (2004). The benthic fauna community assemblages were described based on the prevailing and measured abiotic parameters, mainly grain size and TOM, which are among the main factors shaping most benthic studies (Higgins and Thiel, 1988). However, it will be interesting to study the influence of the other abiotic parameters (such as current speed, sediment deposition, depth) on these communities.

This study has provided insights into how the benthic communities along the Kenyan shelf are assembled, and with further studies, it will be possible to better understand the status of the Kenyan shelf benthic communities, and thus provide much needed information that is a prerequisite for making informed decisions on any conservation or development initiatives along the Kenyan shelf.

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