Western Indian Ocean JOURNAL OF Marine Science

Volume 21 | Issue 1 | Jan – Jun 2022 | e-ISSN 2683-6416

Western Indian Ocean JOURNAL OF Marine Science

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e-ISSN 2683-6416

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JOURNAL OF
Marine Science

Volume 21 | Issue 1 | Jan – Jun 2022

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Abundance, spatial distribution and threats to Indo-Pacific bottlenose dolphins (*Tursiops aduncus*) in an Important Marine Mammal Area in Tanzania

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Abstract
Abundance estimates of cetaceans in the western Indian Ocean are rare, but important, as many cetacean populations are under threat, especially those in coastal habitats. This study aimed to generate first estimates of abundance for Indo-Pacific bottlenose dolphins (*Tursiops aduncus*), assessed as Near Threatened on the IUCN Red List, in an area identified by the Marine Mammal Protected Area Task Force as an 'Important Marine Mammal Area'. Two study sites were surveyed along the east and west coastlines of the Pemba Channel, Tanzania. Between 2014 and 2016, four boat-based visual surveys conducted a total of 2467 km of survey effort sighting a total of 16 groups of *T. aduncus* in west Pemba. Abundance was estimated using mark-recapture models of photo-identified individuals as 89 individuals (CV 7.7 %, 95 % CI 76-103) in the 1084 km² study area. In the Tanga study area in 2016 two boat-based visual surveys covered 1254 km of effort during which 15 groups of *T. aduncus* were sighted, resulting in a photo-ID based mark-recapture abundance estimate of 177 individuals (CV 8.6 %, 95 % CI 150-210) in the 1562 km² study site. Group encounter rate for this species in Tanga was at least double that recorded in the Pemba study site. A total of 23 % of identified dolphins bore the scars of interactions with fishing gear.

Keywords: Indo-Pacific bottlenose dolphin, *Tursiops aduncus*, abundance, photo-identification, important marine mammal area, western Indian ocean

Introduction
A lack of information on whales and dolphins due to limited research in many developing countries frequently prevents assessment of their status, and for informed management actions to aid in their conservation (Kuit *et al.*, 2021). Cetaceans typically occur at low densities, in hard to access habitats, and because they spend much of their lives underwater are also challenging to detect, with the result that gathering baseline data requires several years of study; requiring funds and time that are often unavailable. Obtaining an abundance estimate is an essential component of a baseline survey, and this is especially important for coastal species that are under threat from anthropogenic activities in their habitat impacting both survival and health. In these situations, baseline surveys and abundance estimates are important in order to guide and monitor species protection and habitat management (Kuit *et al.*, 2021; Avila *et al.*, 2018; de Vere *et al.*, 2018).

While there have been extensive long-term studies on coastal cetaceans around Unguja Island in Tanzania (see Amir *et al.*, 2002; Berggren *et al.*, 2007; Stensland *et al.*, 2006; Stensland and Berggren, 2007; Sharpe and Berggren, 2019), prior to 2016 information on cetaceans was almost completely lacking for the rest of the 800 km long Tanzanian mainland coastline. To fill the information gap, a rapid cetacean assessment was conducted which combined visual and acoustic surveys, compilation of citizen science records and documentation of skeletal material (Braulik *et al.*, 2017).

http://dx.doi.org/10.4314/wiojms.v21i1.1

A total of 19 species were documented from Tanzanian waters, and a cetacean fauna dominated primarily by tropical delphinids. The most frequently encountered species were spinner dolphins (*Stenella longilostris*) and Risso's dolphins (*Grampus griseus*) which occur primarily in deep waters, followed by Indo-Pacific bottlenose (*Tursiops aduncus*) and Indian Ocean humpback dolphins (*Sousa plumbea*) both in shallow coastal waters (Amir *et al.*, 2012; Braulik *et al.*, 2017). The rapid assessment concluded that the Pemba Channel was one of the most important areas for cetaceans in Tanzanian waters as relative species diversity and relative abundance were both higher than in all other areas (Braulik *et al.*, 2017). The Greater Pemba Channel was subsequently identified as an Important Marine Mammal Area (IMMA) by the IUCN/World Commission on Protected Areas Marine Mammal Protected Areas Task Force (IUCN-MMPATF, 2019), adding to its previous identification by the Convention on Biological Diversity as an Ecologically and Biologically Significant Area (EBSA). The Pemba Channel separates Pemba Island from the Tanzania mainland and is only 56 km wide but just over 900 m deep at its deepest point (Fig. 1). The biological importance of the Pemba Channel is likely to be because of the extremely steep bathymetric slopes on either side, and the rapid northward flowing East African Coastal Current (EACC), which can reach 2.5 ms^{-1} and brings cooler water and nutrients to the surface, leading to high productivity and resilience from increasing sea surface temperatures due to climate heating (McClanahan, 2020; Barlow *et al.*, 2011).

The current study was initiated to understand more about the status of cetaceans within the Pemba Channel focussing on Indo-Pacific bottlenose dolphins (*Tursiops aduncus*), coastal dolphins which are one of the species likely to be under greatest threat as their nearshore habitat is most extensively utilised and exploited by people. *T. aduncus* is listed as Near Threatened on the IUCN Red List and generally exists in small, semi-isolated populations in coastal areas, where they are impacted by habitat degradation, and are vulnerable to bycatch in fishing gear – primarily gillnets (Braulik *et al.*, 2019). Indo-Pacific bottlenose dolphins are one of the more common cetacean species recorded in coastal parts of Tanzania and in other regions in the western Indian Ocean (Berggren and Coles, 2009; Amir *et al.*, 2012, Braulik *et al.*, 2017). Although cetaceans are protected both in mainland Tanzania under the Fisheries Regulations, 2009, and in Zanzibar under the Zanzibar Forestry Act, 1996, cetacean protection measures are not routinely enforced. Fisheries bycatch has been identified as a large threat to this species both in Tanzania (Braulik *et al.*, 2017), in Unguja Island, Zanzibar (Amir, 2010; Amir *et al.*, 2002), as well as in other areas of the region including Kenya (Pérez-Jorge, 2016), the island of Mayotte (Kiszka *et al.*, 2008), Madagascar (Razafindrakoto *et al.*, 2004), and the Algoa Bay area of South Africa (Reisinger and Karczmarski, 2010).

The objective of this study was to generate a first estimate of abundance of Indo-Pacific bottlenose dolphins in coastal waters of the Pemba Channel. This three-year study is the first of its kind to be conducted for cetaceans in northern Tanzania. Baseline abundance estimates for this coastal cetacean are important for informed conservation and management planning in areas with high overlap of anthropogenic activities.

Material and methods

Study area

The Pemba Channel is located in the northern part of the coast of Tanzania in the western Indian Ocean. There were two study sites: 1) West Pemba Island, which occurs on the eastern side of the Pemba channel; and 2) Tanga coast, which occurs on the western side of the channel (Fig. 1). Both study areas included predominantly coastal waters extending to 10-15 km from shore depending on location.

Pemba Island is part of the Zanzibar archipelago, and lies about 40 km NNE of Unguja Island at approximately 4° south and 39° east. The western coast of Pemba Island has, since 2005, been designated as the Pemba Channel Conservational Area (PECCA), which is Tanzania's largest area legally set aside for marine conservation. (Fig. 1) (McLean *et al.*, 2012). The western coast of the island has a complex configuration and contains a variety of habitats; there are extensive shallows with mudflats, mangroves and sea grass beds, rich coral reefs and islets separated by deep tidal channels, as well as numerous rocky peninsulas that extend westwards (McLean *et al.*, 2012; McClanahan 2020). The study area covered was 1084 km^2 in size and encompassed all of the marine conservation area. The Tanga coast study area was located within the Tanga Coelacanth Marine Park (TACMP) located on the northern coastline of Tanga City, and covered 1562 km^2, extending from the Pangani River estuary along the coastal belt north to Kenya. These inshore waters are characterized by fringing and patch coral reefs, sea grass beds, mangrove forests, and several estuaries and bays. The marine park was established as a result of multiple records of coelacanth (*Latimeria chalumnae*) in the area, after which the park is named.

Field surveys

Between 2014 and 2016 six boat-based surveys were conducted each lasting 12-14 days. In Pemba four surveys were carried out, one each in October 2014, 2015 and 2016, and March 2016, while in Tanga two surveys were conducted, one in March and one in November 2016. These months were selected as they fall within the inter-monsoon period and are those with the lowest mean wind speeds annually (Mahongo *et al.*, 2011). A locally available wooden dhow 7 m in length, with a 40 hp outboard motor and viewing platform 2 m above the water surface, was used for the surveys.

Transects were systematically laid, running from east-west, perpendicular to the coast and depth contours, using the survey design function in the program DISTANCE (Thomas *et al.*, 2010). In the Pemba study area 38 transects, 2 km apart were laid, whereas in the Tanga study area there were 30 transects, 3km apart (Fig. 1). New transect positions were generated for each survey to avoid exact repetition.

Prior to all surveys, observers underwent a day of methods training and two days of field training. During all surveys, three trained observers scanned the ocean continuously for dolphins, each using a pair of 7x50 Fujinon marine binoculars, with internal compass. Dolphin position and survey effort were recorded via GPS using the WGS84 reference coordinate system. Survey effort was postponed when the sea state reached greater than Beaufort 4 because sighting rate declines with increasing wind speed (Nichol, 2009). When dolphin groups were sighted, survey effort stopped and the boat left the transect line and followed the group. A Canon EOS 60D camera with a 70-200 mm zoom lens was used to obtain

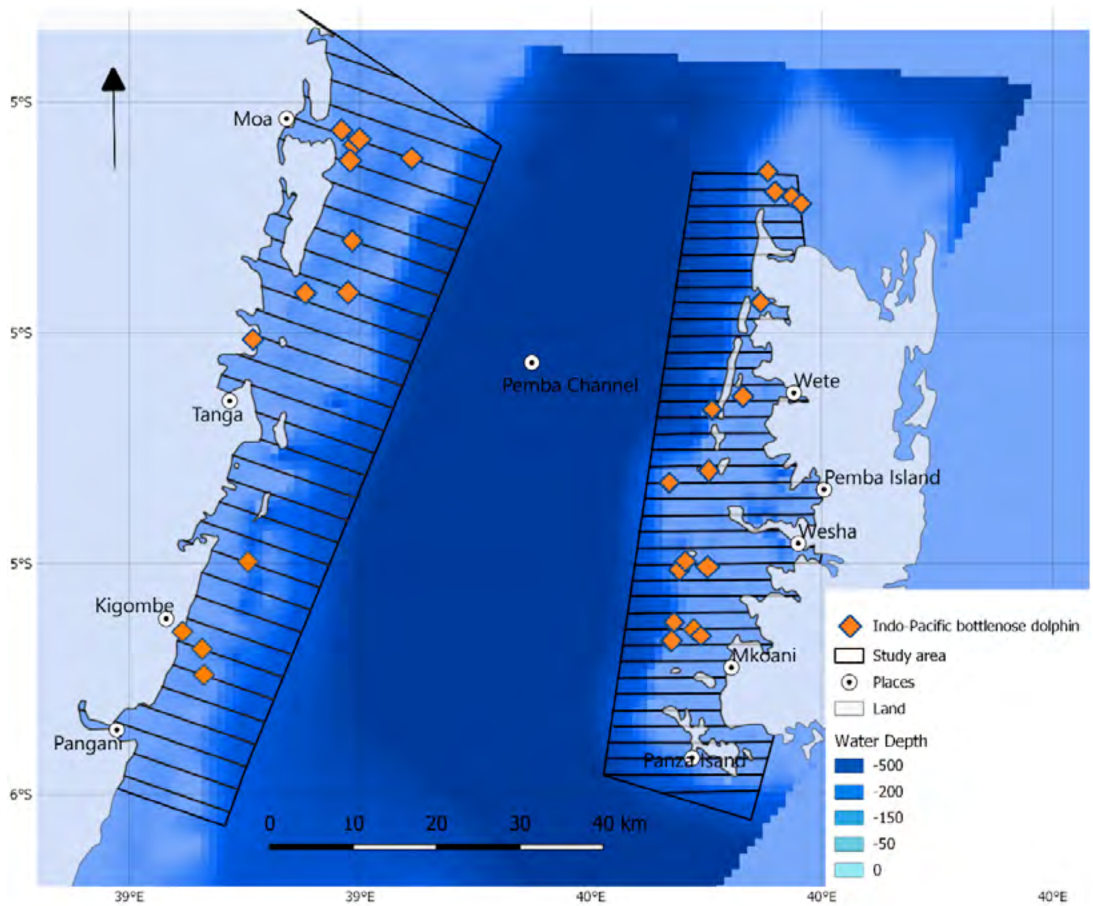

Figure 1. Location of the two study areas; Tanga in the west, and Pemba Island in the east, and Indo-Pacific bottlenose dolphin (*Tursiops aduncus*) sightings, transect lines and towns from the surveys conducted between 2014 and 2016.

high quality images of both sides of all animals in each group. Once all animals were photographed, or if the group was lost, the boat returned to the transect line and surveying efforts resumed. The following information was recorded during each sighting: species, best estimate of group size, high and low estimates of group size, number of calves, geographic location, sea state (wind), and whether photos were taken. Calves were defined as animals half the size of the mother or smaller, and groups were defined as animals sighted within 200 m of one another that appeared to be engaged in similar activities and moving in a similar direction. Although slightly counter-intuitive, as with most cetacean studies, the term 'group' is used to refer to all cetacean detections including those composed of only 1 or 2 animals, e.g. a group of one. Sightings recorded when transiting between transects or when following another dolphin group were recorded as 'off effort'. Photographs obtained from off-effort groups were used in analysis but not in calculations of encounter rate. After each day of field work all effort and sighting data were entered into Excel spreadsheets, and the GPS positions, survey track and photos were downloaded to a computer and stored in folders labelled by date.

Photo-processing

A separate photo-ID catalogue was created for each study area. Individual dolphins were identified on the basis of permanent marks and distinctive features on the dorsal fin and tail peduncle, including distinct fin shapes, notches, scars and cuts. Other secondary features such as colour pigmentation and tooth rakes were not used for identification, but only as supporting evidence, as these features are often not permanent. Images with no dolphins, or where the dorsal fin was not discernible, were discarded and those where dolphin fins were clearly visible were renamed, digitally enhanced and cropped, and given temporary identification numbers. Image quality was assessed and provided with codes based on four aspects: 1) focus, 2) angle of subject, 3) size of fin in the image, and 4) proportion of the entire fin visible in the photograph. Each of these were coded from 0 (very poor) to 3 (excellent). For each photograph, the four codes were summed to give a single quality (Q) score, with the maximum possible for the best images being 12. The quality rankings were made independently by two recorders (MK and GB), results were compared, and where there was a discrepancy a final revised quality code given. Photographs were sorted into three subfolders based on quality: Q0-8, Q9 and Q10-12. Only images of the highest quality Q10-12 were entered into the catalogue and used for photo-ID analyses. Dorsal fin distinctiveness was rated as highly distinct where fins were deformed or had major nicks or injuries; medium distinctiveness was assigned to fins with two or more small to medium sized nicks; and low distinctiveness were fins with one small nick or a unique fin shape. Photographs of left (LDFs) and right (RDFs) dorsal fins were treated as two separate datasets when creating the catalogues and during analysis. This was due to the inability in most cases, to definitively link the LDFs and RDFs of individual dolphins; this approach is commonly used in the studies of cetaceans (Minton *et al.*, 2013, Kuit *et al.*, 2021).

Matching was done by comparing each new fin image with all other existing fins in a catalogue; any individual that did not match was given a new identification number, entered into the catalogue as a new individual and was subsequently verified by an independent observer (GB). As is the case for most photo-identification studies, calves were excluded from the mark-recapture analysis as their probability of capture is not independent from that of their mothers; in addition, all calves were unmarked and could not be identified.

Abundance estimation

Capture–recapture models were applied in the program MARK (version 9, 2019) (Cooch and White, 2010) to the Indo-Pacific bottlenose dolphin catalogues giving separate abundance estimates for each study site. Closed models were selected as the species showed a good degree of residency to the areas (as shown by multiple recaptures), and due to the relatively short study period substantial demographic changes due to deaths, births or emigration were unlikely. Closed models have been used to estimate abundance of a number of other coastal dolphin populations that occur in similar habitat (Minton *et al.*, 2013; Stensland *et al.*, 2006; Sharpe and Berggren, 2019). Population closure was tested using the program Close Test to see if the assumption of closure was violated, and it was not for either study site (Stanley and Burnham, 1999). Sampling occasions corresponded to each 14-16 day survey; there were four sampling occasions in Pemba and two in Tanga. A capture history for each unique individual in each catalogue was created where 1 means that a dolphin was sighted on that sampling occasion, and 0 means that it was not seen. Individuals sighted multiple times in a survey were counted only once.

Five models were run (Mo, M_b, M_t, M_{tb} and the Pledger mixture model), as follows:

- The basic model, where capture and recapture probabilities were assumed to be the same and constant over time (model M_o) in program MARK
- Model capture and recapture probabilities assumed to be different, but not changing over time - equivalent to M_b
- Capture probabilities change over time - the model is equivalent to M_t in program MARK
- Models that assume change in capture probability over time and different recapture probability (M_{tb})
- Models that assume different capture probabilities for different classes of animals, referred to as heterogeneity in CAPTURE probabilities, formulated as a Pledger mixture model in program MARK.

Due to the small sample size AICc (Burnham and Anderson, 2003) was used to select the most appropriate model with the smallest number of parameters, according to the following guidelines: (1) differences of less than two in AICc values were taken to indicate that the models have approximately the same weight; (2) the differences of more than two but less than seven in AICc values indicate there is significant support for a real difference between the models; and (3) differences of more than seven between AICc values indicate that there is strong evidence of a difference between the two models (Burnham and Anderson, 2003). To account for uncertainty in model selection, if the best fitting models were separated by less than two AICc units, they were averaged based on their normalised AICc weights. Unlike other mark-recapture models, there is no good way to test goodness of fit for closed capture models, nevertheless, model averaged estimates of abundance, weighted according to AICc, are more robust than single model estimates. If this method is used, the necessity for testing goodness of fit is not maintained (Stanley and Burnham,1998).

The confidence intervals were constructed following Williams *et al.* (2002) by assuming that N is log-normally distributed according to the following: the lower and upper 95 % confidence limits were (N/c and N x c) where:

$$c=e^{1.96}\sqrt{\ln(1+CV^2N)}$$

The proportion of distinctly marked individuals in the population was calculated by examining all photographs of good quality (Q10-12) of all individuals in each sampling occasion and recording the number of marked and unmarked individuals in each photo. The proportion of marked animals in the population was calculated from the total number of individuals photographed divided by total number of photographed animals that were marked. The final abundance estimate was generated by correcting the estimated abundance of marked individuals generated by MARK.

The following assumptions associated with this mark-recapture model were adopted:

- The population probability of first capture is the same as recapture
- Unmarked animals have the same probability of being recaptured as marked
- Marks are not lost or missed
- Every marked animal has the same probability of survival

These assumptions are discussed below.

If dolphins changed their behaviour after being initially 'captured' photographically, they might then be less likely to be recaptured. For this study, the dolphins did not bow ride, or appear to actively move toward or away from the vessel. Given this behaviour, the probability of animals, whether marked or unmarked, being photographed on the first and subsequent occasions is likely to be the same and not to have changed over time. Behavioural bias affecting capture was included in the designed models (Mb). In dolphin photo-identification studies, the recapture probabilities are usually the same as capture probabilities (p=c), and capture probabilities are more likely to vary by sampling occasion (M_t) (Hammond, 2010).

A standard assumption of mark-recapture models is that there is equal capture probability for all individuals in all circumstances. It is possible that single animals are more likely to be missed than animals in larger more visible groups, leading to frequently sociable individuals being over-represented in the data. To account for this capture heterogeneity, the Pledger mixture model, which accounts for several groups of animals with different capture probabilities, was applied (Pledger, 2000). The assumption that marks are not missed or lost was addressed by using only high-quality photographs, thereby ensuring the maximum likelihood of not missing captures, and using only significant long-lasting marks on the dorsal fin, thereby reducing the likelihood of losing identifying features and missing recaptures.

Distribution and habitat use

Survey data was plotted in QGIS to illustrate the spatial distribution of sightings. The General Bathymetric Chart of the Oceans (GEBCO) 2014 Grid data at 30 arc-second intervals (equivalent in Tanzania to approximately 920 m square pixels) was used for bathymetry, however the resolution was too poor to accurately extract the depth at group locations. In the absence of any other form of suitably detailed digital depth data, images of digital Navionics navigation charts (webapp.navionics.com) were geo-rectified, which are the most accurate source of data available. Depth and distance to shore were then calculated for each sighting.

Assessment of dolphin-fishery interactions

An analysis was conducted to evaluate the impact of fisheries interactions on this population by determining the number of fishery-related scars present on individuals. For all individuals in the catalogue, injuries, wounds, lacerations and scars on the body or dorsal fin that may have been caused by fishing were identified following the detailed guide in Barco and Touhey (2006). Animals were classified as showing signs of fishing interactions if they had one of the following two injuries: 1) Linear marks - Linear cut, impression, scar or abrasion that was deep or shallow on the leading or trailing edge of the dorsal fin, or a series of parallel lines which is likely to be caused by fishing gear (lines/nets); and 2) Deformed or damaged fin - A partially or completely missing part of the fin likely to be caused by cutting from either fishing lines or nets. Based on the above evidence, fins were classified as having no evidence of fishing interactions, clear evidence of fishery interactions, and where there was some uncertainty, possible evidence of fishery interactions (Barco and Touhey, 2006; Kiszka *et al.*, 2008). Evaluation of the level of injuries to dorsal fins was evaluated following Kiszka *et al.* (2008), to determine a 'fishing gear exposure risk (R_i)' according to the following formulae:

$$R_i=(N^{dis}/\sum^{id})\ \%^{id}$$

Where N^{dis} is the number of individuals having clear evidence of fishery interaction, $\sum^{id}$ is the total number of identified individuals, and $\%^{id}$ is the proportion of identified individuals. Possible values of R_i range from a minimum of 0 %, where no animals have marks consistent with injuries sustained from interactions with fishing gear, to a maximum of 100 %, where all animals have fishing gear related injuries.

Results

Pemba: A total of 50 days of surveying was conducted in the Pemba study area, including 2467 km on effort searching and the detection of a total of 16 groups of *T. aduncus* at an encounter rate of 0.65 groups/ 100 km of survey effort, or 1 group per 154 km of survey. Group sizes ranging from 1 to 20 individuals (mean group size 17.65; SD=12.41). This species was the third most frequently encountered in the study area, after the spinner dolphin (*Stenella longirostris*; encounter rate of 2.03 groups/ 100 km), and Indian Ocean humpback dolphin (*Sousa plumbea*; encounter rate of 0.97 groups/ 100 km). Sightings occurred in water that was between 1 m and 46 m deep, with both the mean and median depth at sightings 15 m.

Tanga: A total of 26 days of surveying was conducted in the Tanga study area, including 1254 km of effort during which 15 groups of *T. aduncus* were sighted at

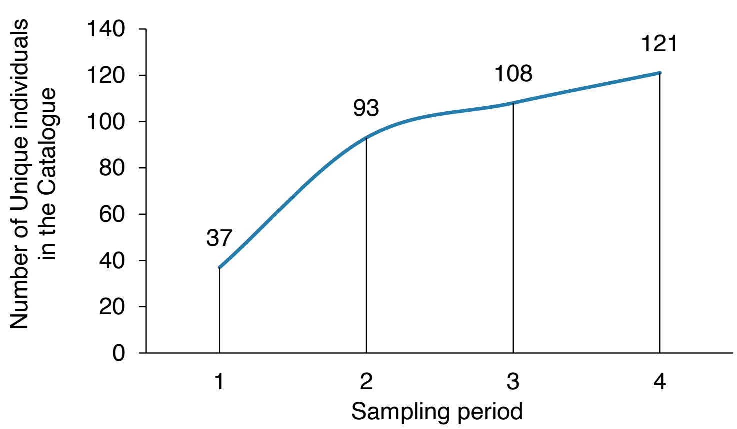

Figure 2. Discovery curve of new photo-identified Indo-Pacific bottlenose dolphins (*Tursiops aduncus*) individuals in the study site, west of Pemba Island, Tanzania.

an encounter rate of 1.20 groups/ 100 km of survey effort, or 1 group every 84 km. Only a single group was encountered during the November 2016 survey, while 14 groups were sighted in the March 2016 survey to give an encounter rate of 1.96 groups/ 100km of survey effort for that survey. Group sizes ranged from 1 to 37, with mean group size 11.75 (SD=9.58). *T. aduncus* was the most frequently encountered species along the Tanga coast, followed by spinner dolphins (1.12 / 100 km) and then Indian Ocean humpback dolphins (0.24 / 100 km). Sightings occurred in water that was between 10 m and 46 m deep, with the median depth 28 m and the mean 30 m. Indian Ocean humpback dolphins were in mixed species aggregations with Indo-Pacific bottlenose dolphins on two occasions. A Pearsons correlation of depth against group size for all bottlenose dolphin sightings data pooled gave a correlation coefficient of -0.22, which indicates no significant relationship.

Photo-ID based abundance estimates

Pemba: A total of 1631 photographs of Indo-Pacific bottlenose dolphin dorsal fins from Pemba (of which 303 were of quality 8 or less, 599 were of Q9, and 729 were of quality score 10-12) were examined for entry into the catalogue. The final catalogue contained 65 left dorsal fins (LDFs) and 56 right dorsal fins (RDFs). The rate of discovery of new individuals is shown in Figure 2 and was higher within the first two sampling occasions and started to decline in the subsequent two surveys. Although discovery of new individuals in the 2016 sampling periods slowed, it had not yet plateaued. A closure test was performed on both the LDF and RDF data separately, and no significant results were returned indicating that there was no evidence of significant losses or gains to the population between sampling intervals and the population could be considered to be closed for the purposes of analysis (Stanley and Burnham, 1999).

The abundance estimates derived from mark-recapture analysis were LDFs 76 (CV 2.3 %, 95 % CI 64-89) and RDFs 67 (CV 2.4%, 95 % CI 55-99). The best fitting model in both cases was one where the capture probability varied by time, with no heterogeneity in capture probabilities. The larger of the two estimates (LDF), which is based on the larger photo-ID catalogue is considered the best estimate of abundance for the study site. Determination of mark rate showed that 1246 dorsal fins were identified as marked and 199 as unmarked, giving a mark rate of 0.8623. The LDF final abundance estimate corrected for unmarked individuals was 89 individuals (CV 7.7 %, 95 % CI 76-103).

Repeated sightings of the same individuals across the four-year study, including 6 dorsal fins (5 % of all fins in the catalogue) that were seen in every survey, and 20 dorsal fins (16.5 % of all fins in the catalogue) that were seen in three or four surveys, suggest that a proportion of the animals are resident in the area over

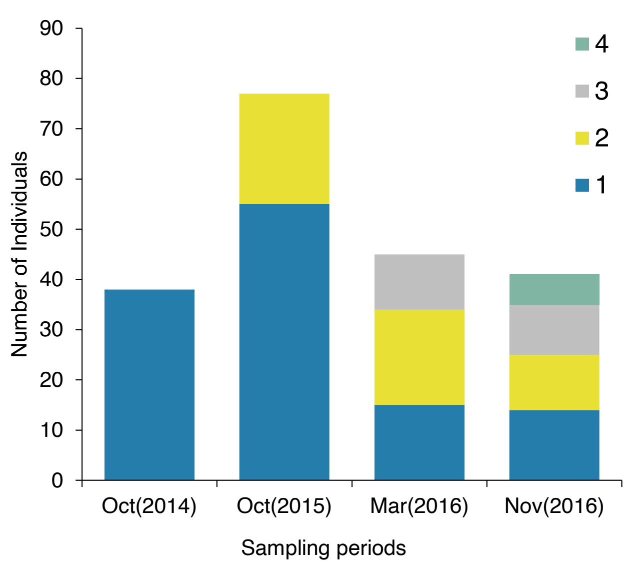

Figure 3. Number of sampling periods (equivalent to surveys) in which a photo-identified Indo-Pacific bottlenose dolphin (*Tursiops aduncus*) was sighted in the study area west of Pemba Island, Tanzania.

multiple years. However, 50 % of the dorsal fins in the catalogue were only encountered once (Fig. 3). Calves were sighted in all surveys.

Tanga: In Tanga a total of 1339 photographs were taken, of which 243 were of quality 8 or less, 471 were of quality 9 and 625 were of good or excellent quality (scored 10-12), and were examined for entry into the catalogue. The final catalogue of photos contained 57 unique LDF and 53 unique RDFs. Only one group was sighted during the second survey / sampling period, and there were very few (n=17, 15 % of dorsal fins) resights. The abundance estimate below is therefore fairly imprecise and should be considered preliminary pending more information. LDFs of 144 individuals (CV 6.0 %, 95 % CI 76-459) and RDFs 173.5 (CV 7.0 %, 95 % CI 72-821) were estimated. The best fitting models for both left and right fins was one where capture probabilities varied by time, with no individual heterogeneity and no difference in behaviour. The larger catalogue, and most precise abundance estimate was that of the LDFs and this was used for the final abundance estimate. The proportion of marked individuals in this population was 0.810, giving a total corrected abundance estimate of LDFs of 177 individuals (CV 8.6 %, 95 % CI 150-210) individuals.

Spatial distribution of sightings

Pemba: The distribution of Indo-Pacific bottlenose dolphin sightings was clumped geographically in the centre of the west coast of Pemba Island waters, adjacent to the towns of Mkoani and Wesha, and close to the islands of Misali, Uvinje and Njao, whereas, by contrast, few sightings occurred in the north of the study area, and none were seen in the southern part of the study area near Panza Island (Fig. 1). The location of the six photo-identified dolphins that were sighted on all four surveys are shown in Figure 4. The distance between the sighting locations of these six individuals ranged from a maximum of 59 km to a minimum of 21 km; four individuals were recorded in locations just over 50 km apart, while two were recorded on four occasions only 20 km apart.

Tanga: Indo-Pacific bottlenose dolphins occurred throughout the Tanga study area from north to south, but did not occur seaward of the fringing barrier islands and reefs that occur along the Tanga coast. One photo-identified individual was recorded just south of the village of Moa close to the Kenya border in April 2016, and was resighted 71 km south, near the Pangani river mouth in November 2016.

Photo-ID catalogues from the two study sites were compared and no matches were found.

Fisheries interactions

Many identified individuals in both study areas had marks that were likely to be injuries sustained from fishing gear. Among the 121 dorsal fins in the LDF and RDF catalogues for the Pemba study area, 28 (23.1 %) had marks clearly associated with fishery interactions (Fig. 5), 58 (47.9 %) had marks possibly associated with human interaction and 35 (28.9 %) individuals had no marks attributed to fishing gear. The calculated fishing gear exposure risk ratio, R_i: was 20.0 % of individuals that have definite marks, and 61.3 % of individuals if marks that are both definite and possibly caused by fishing gear are included. By comparison, in the Tanga study area, from a total of 117 unique individuals (left and right dorsal fins combined), 27 (23.1 %) had marks and injuries clearly associated with fishing gear interactions, 34 (29.1 %) had marks possibly associated with human interactions and 56 animals (47.9 %) had no marks from fishing interactions. The fishing gear exposure risk ratio was 18.7 % for animals with marks definitely related to fishing, and 42.2 % of animals if both definite and possible marks were included.

Discussion

Abundance

The findings of this study show that the population of Indo-Pacific bottlenose dolphins in the west Pemba study site is very small, numbering just under 100 individuals (89 individuals (CV 7.7 %, 95 % CI 76-103). The discovery curve of new individuals into the photo-ID catalogue was still slowly increasing at the end of the study (see Fig. 3) at a rate of approximately 10-15 individuals per 12-day survey suggesting that there are a number of dolphins that use the area that have not yet been identified. Based on the shape of the discovery curve it is likely that 20 to 30 animals would be added in subsequent surveys. The abundance estimates were similar to studies from nearby areas. For example, off the south coast of Unguja Island (Zanzibar) a small population of Indo-Pacific bottlenose dolphins was estimated to comprise 136 individuals (log-normal 95 % CI 124-172) from surveys conducted in 1999 and 2002 (Berggren *et al.*, 2007, Stensland *et al.*, 2006). In southern Kenya, a small population of this species was estimated as ranging from 19 individuals (95 % CI: 11–33) to a maximum of 104 dolphins (95 % CI: 78–139) (Pérez-Jorge *et al.*, 2016). For discrete, small populations, of slow to reproduce and long-lived species such as cetaceans, even low levels of anthropogenic mortality may be sufficient to extirpate them in a relatively short period of time (Thompson *et al.*, 2000). Species such as Indo-Pacific bottlenose dolphins in Pemba that show high site fidelity and coastal distribution are likely to be exposed to high levels of human disturbances and are the most likely to suffer declines (Pusineri *et al.*, 2014; Smith, 2012).

The abundance estimate from Tanga should be considered a preliminary estimate because there was only a single group encountered during the second survey, and a small number of recaptures which resulted in an imprecise abundance estimate. In addition, survey effort was lower in the Tanga study area than the Pemba study area because of funding constraints, so the abundance estimate is based on less data. However, the provisional estimate is larger (approximately double) the estimate for Pemba and indicative of a larger dolphin population. Similarly, the overall group encounter rate for Tanga (1.20 groups/ 100 km of survey effort) was approximately double the group encounter rate in Pemba (0.65 groups/ 100 km of survey effort), and the encounter rate for the March 2016 survey in Tanga (1.96 groups/ 100 km of survey effort) three times higher than the average for Pemba, again suggesting that many more animals use the Tanga

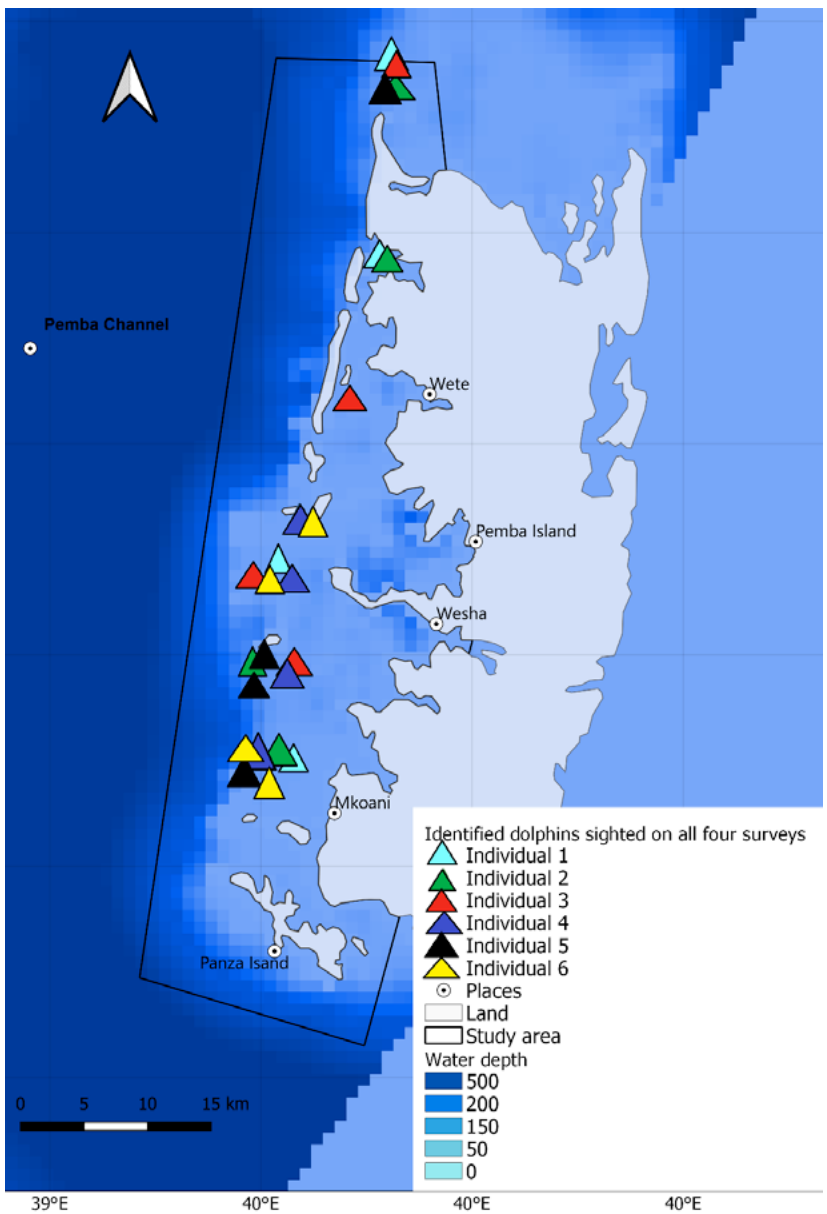

Figure 4. Sighting locations of the six most frequently sighted Indo-Pacific bottlenose dolphins (*Tursiops aduncus*) encountered during four surveys conducted between 2014 and 2016 in the west Pemba study area.

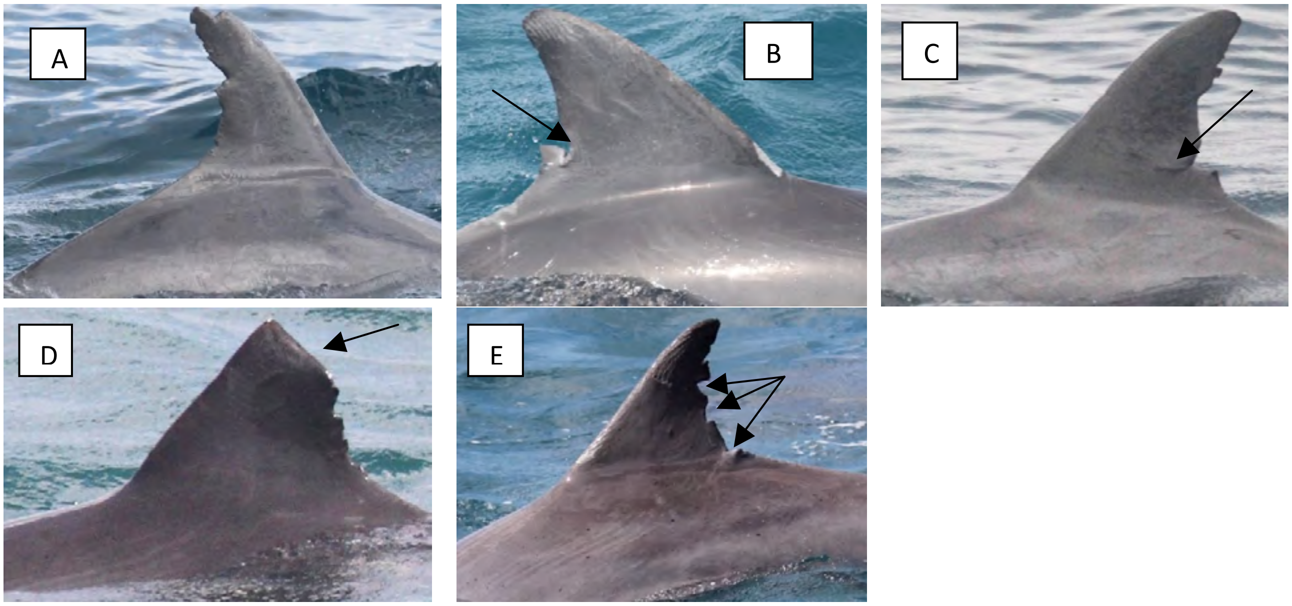

Figure 5. Examples of individual Indo-pacific bottlenose dolphin dorsal fin showing distinctive natural markings and deformed fins. A) shows dorsal fin mutilations likely caused by human impacts of unknown further classification; B) shows signs of major injuries (arrow) probably caused by net or unknown impact; C) has a clean-cut pointed notch and associating scar (arrow), which was possibly caused by fishing gear; D) has a linear section (arrow) probably due to fishing line interactions; and E) shows examples of major nicks (arrows) that can be easily recognized. (Photo credit: Magreth Kasuga and Gill Braulik).

coast than Pemba Island.

Connectivity and isolation

Indo-Pacific bottlenose dolphins are known to prefer predominantly coastal habitat (Braulik *et al.*, 2019) and throughout this study were found only in water less than 50 m deep. In Tanga one dolphin was recorded at locations 71 km apart, in the north and far south of the study area. Dolphins occurring within the Tanga study area are likely to be able to move along the coast, including into waters to the north in Kenya, and further south towards Bagamoyo, and are likely to form part of a connected coastal super-population on the coast of eastern Africa. By contrast Pemba is an island surrounded by water close to 1000 m deep on all sides, and it therefore seems plausible that the presence of deep-water habitat limits the regular movement of this coastal species from Pemba to the mainland (and vice versa). In addition, the marine habitat west of Pemba is quite different to the mainland coast, with less continental runoff, less exposure to onshore winds, and also small-scale upwelling leading to lower sea surface temperatures and localised high productivity, all of which might make Pemba more suitable for a resident population than the mainland environment (Kizenga *et al.*, 2021; Sekadende *et al.*, 2021). Further surveys in other seasons, studies of genetics, and comparison of photo-identification catalogues with other regions will provide a deeper understanding of the degree of isolation of this population.

Group size

Group size is highly variable in Indo-Pacific bottlenose dolphins, partly due to their fission-fusion society, and is thought to be related to activity, availability of food resources, predators and time of day (Shane *et al.*, 1986). Smaller groups are generally correlated with feeding and foraging and larger groups with socializing (Fury, 2009; Fury and Harrison, 2008). Mean group size in Tanga (11.75 individuals / group) was slightly smaller than in Pemba (17.65 individuals per group), however the difference was not statistically significant ($p=0.1712$, $t=1.4057$, $df=27$) and the range in group sizes was similar. Group sizes of this species in Menai Bay in southern Unguja had a median size of between 8 and 21 (Stensland and Berggren, 2007) and in Mayotte there was a mean of 6.5 animals per group (range 1-15) (Kiszka *et al.*, 2010). The highest mean group size reported for this species in the western Indian Ocean was in southern Kenya where 62 individuals/group (range 20-102) were reported (Pérez-Jorge *et al.*, 2016). The larger group size seen in southern Kenya may reflect differences in habitat and prey.

Fisheries interactions

Fishing gear entanglement and subsequent mortality is the largest threat to cetaceans worldwide and 75 % of odontocetes species are known to be at high risk from bycatch in gillnets (Reeves *et al.*, 2013). Because of its coastal distribution, *T. aduncus* is vulnerable to threats from anthropogenic activities throughout its range, including entanglement in fishing nets (Amir, 2010; Braulik *et al.*, 2019). Although both the Pemba and Tanga study sites are protected areas, fishing is permitted, and they are, in fact, some of the most intensively fished parts of the Tanzanian coastline (MLDF, 2010; ZMLF, 2010). In Tanzania bycaught animals are subsequently either discarded, eaten, the meat used as shark bait on longlines, or the meat is allowed to rot and the oil used as water proofing on boats (Amir, 2010; Braulik *et al.*, 2017; Robards and Reeves, 2011). Evidence of scars from injuries that are possibly linked to fishery interactions were observed on close to three-quarters of all identified dorsal fins in this study, and marks that were clearly due to fisheries were found on close to a quarter of all individuals, which means that both populations are clearly affected by fishery interactions. This mark rate is similar to other studies, such as in Bangladesh, where 28 % of identified individuals exhibited injuries related to entanglements with fishing gear (Smith *et al.*, 2015), and the island of Mayotte in the Mozambique channel where 19 % of the identified individuals showed significant marks and injuries that could be related to fishery interactions (Kiszka *et al.*, 2008). More than 30 % of Indo-Pacific humpback dolphins photo-identified off Taiwan, an intensely developed area, had scars or injuries most likely caused by interactions with fisheries (Dungan *et al.*, 2011; Reeves *et al.*, 2013). Investigation of fisheries bycatch conducted between 2003 and 2006 around Unguja Island showed that 48 % of all documented fishery-related dolphin mortalities were Indo-Pacific bottlenose dolphins (Amir, 2010). This large proportion is likely linked to their nearshore distribution and moderate local abundance of this species, and the overlap of core habitat with fishing effort (Amir, 2010). It is probable, similar to Unguja, that Indo-Pacific bottlenose dolphins are one of the most frequently captured cetacean species in the Pemba and Tanga areas, because of the overlap between their preferred habitat and intensive coastal fisheries. The mortality of only four individuals per year from a population of 100, or 7 from a population of 200 would result in a 50 % population decline over three generations (Moore, 2015). The existence of non-lethal injuries on many identified individuals (Dungan *et al.*, 2011; Kiszka *et al.*, 2008), combined with the small size of the population and the low reproductive rate of the species mean that the population of Indo-Pacific bottlenose dolphins west of Pemba may well be under threat and declining due to fishery interactions.

The issue of cetacean bycatch could be evaluated through fisheries observers or monitoring of fish landing sites, and a conservation strategy developed in conjunction with fishing communities and local governments. An approach such as that used by Verutes *et al.* (2020) which maps data on dolphin distribution and on fishing effort gained through community participation to identify high risk bycatch zones has potential to identify targeted areas where fishing effort may be restricted for maximum benefit to dolphin populations. The fisheries sector in Tanzania and Zanzibar is growing, and especially in Tanga and Pemba both of which are rural, fishing plays a key role in household food security, providing income and employment opportunities to communities where there are few other economic activities available. The enforcement capability of the government is weak and therefore communities need to be directly involved in management and conservation activities otherwise ecosystems are likely to continue to be overexploited (Jansen *et al.*, 2000).

Future studies

For conservation and management purposes these estimates are important and form a first baseline from which monitoring can be conducted. Future work may replicate these boat-based surveys to allow trends in abundance over time to be determined, as well as to provide more information on residency, movements and home range. The study areas were already quite large compared to other similar studies, many of which have study areas <500 km^2, however, especially in Pemba it would be useful to extend the current study site to include the remainder of the shallow water habitat to the northeast of the island, so that all probable habitat areas are included. To provide information that is helpful for managers, understanding the environmental drivers of distribution as well as identifying distribution hotspots is an important first step to prioritising locations that may be considered as areas for the removal of gillnet fishing, to reduce the risk of bycatch, which is likely to be the biggest threat to these coastal dolphin populations.

Acknowledgements

This work was conducted as partial fulfilment of a Master degree at the Open University in Tanzania. Thanks to the US Marine Mammal Commission (MMC) and the People's Trust for Endangered Species who funded this work via the Wildlife Conservation Society (WCS). The following administration and institutes are greatly acknowledged as they provided full support and collaboration for this project: Ministry of Agriculture, Natural Resource, Livestock and Fisheries Zanzibar, University of Dar es Salaam Institute of Marine Science (IMS), State University of Zanzibar (SUZA) Geospatial laboratory, park rangers on Misali Island, PECCA authorities, and Pemba Fisheries Department. We would like to thank Hamisi Sadallah, Katya Kalashnikova, Sarah Markes, Said Fakih and all who facilitated and supported the project. Sharif Mohammed, head of Fisheries Department Pemba, Ally Kassim manager of PECCA, Mohammed Said, Haji Mohammed, Haji Masoud, Said Shaib, Bakari Kassim, Mohammed captain and his crew and many others participated and contributed to field work during data collection.

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Use of physicochemical parameters and metal concentrations in assessing anthropogenic influences on coastal rivers in Dar es Salaam, Tanzania

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Abstract

Spatio-temporal variations in water quality of three rivers along the Indian Ocean coast in Dar es Salaam, Tanzania, were investigated based on physicochemical parameters and metal concentrations. A compliance analysis was performed based on the Tanzania Bureau of Standards (TBS) and World Health Organization (WHO) limits to examine the suitability of water for domestic use. The dataset was subjected to statistical analysis to determine differences and similarities amongst the rivers. Levels of pH (6.83-11.41), total dissolved solids (203–34,333 mg/L), electrical conductivity (9,408-68,014 µS/cm), turbidity (10.0-45.0 NTU), chloride (108-14,248 mg/L), sulphate (35-766 mg/L) and ammonium (40-5,468 µg/L) complied with neither TBS nor WHO limits. Dissolved oxygen (1.4-6.6 mg/L), chemical oxygen demand (91-1,863 mg/L), total suspended solids (11.9-50.7 mg/L), alkalinity (200-2,658 mg/L), total hardness (362-12,1312 mg/L), salinity (0.19-29.35 ppt) and phosphate (<method detection limit-3.01 µg/L) indicated polluted water in parts of the rivers. Pb (0.7-24.0 µg/L) exceeded both the TBS and WHO limits, whereas Cr, Cu, Fe, Zn and Cd were below limits. Water quality was poorer during the wet season. The results indicate that water from the rivers is unsafe for human consumption and the poor water quality probably also affects the ecology of the rivers. Strategic measures to protect the rivers from further contamination are suggested.

Keywords: water quality, physicochemical parameters, metal concentrations, coastal rivers, anthropogenic impacts, Tanzania

Introduction

The coastal area of Tanzania contains numerous resources associated with coral reefs, mangroves, seagrass beds, sand banks, wetlands, beaches and estuaries (Masalu, 2000). These resources and habitats provide life support to coastal communities through activities such as fishing, aquaculture and tourism that play an important role in social and economic development of the coastal region (Francis and Bryceson, 2000). The area is also endowed with freshwater resources, including rivers and streams that are used for different human activities. Moreover, in some locations, the river waters are used for irrigation in urban vegetable farming that is commonly conducted along the riverbanks (Sibomana *et al*., 2012; Mhache and Lyamuya, 2019). Growing coastal populations and emerging anthropogenic sources of pollution such as industrial effluents, urban runoffs, agricultural fields and solid waste dumps are exerting increasing pressure on the fresh and marine water quality of the area (Mihale, 2017). The effects have been investigated by several researchers, who revealed detectable levels of various contaminants, including pesticide residues (Mwevura *et al*., 2002), polycyclic aromatic hydrocarbons (Gaspare *et al*., 2009), metals (Mihale, 2021), nutrients and bacteria (Nyanda *et al*., 2016), and assortments of inorganic compounds such as residual acids, suspended solids, oils, greases and textile

http://dx.doi.org/10.4314/wiojms.v21i1.2

dyes, particularly from the discharge of unsuitably treated industrial wastewaters (Kihampa *et al.*, 2016). Furthermore, the coastal rivers of Dar es Salaam city have been reported to receive large quantities of solid wastes that are dumped directly into the rivers in commercial and residential areas, significantly contributing to water pollution (Bubegwa, 2012).

Physicochemical properties of water are key factors in evaluating its quality and suitability for various uses and its ability to sustain aquatic life. Due to the ease of their determination and interpretation, these parameters present a suitable means for the initial understanding of water characteristics, including any undesirable properties or possible health effects (WHO, 2011). Along with these, there are various chemical species of concern whose presence or enrichment in water quality monitoring are well known, e.g., nutrients and trace metals. For example, several metals have been identified as chemicals of significant public health concern by the WHO, although some of them, like Fe, Cu, Zn are beneficial to human health at permissible concentrations (WHO, 2011).

Owing to the ecological importance and the socio-economic significance of the rivers along the Indian Ocean coast in Dar es Salaam city, and considering their ever-changing dynamics, it is of paramount importance to periodically monitor the quality of their waters so as to identify changes in the underlying conditions. This study is one of such endeavors, in which concentrations of some physicochemical parameters, ionic species and selected metals are quantified in three the rivers of Kizinga, Msimbazi and Mbezi. To assess the extent to which the water quality has been affected, the levels are compared to the guidelines for such variables in potable waters set by the WHO (2011) and the Tanzania Bureau of Standards (TBS, 2008). Statistical analyses are used to highlight the observed concentrations in association with seasonality and land-use practices. The aims of the study were to examine trends in spatial and temporal variations, elucidate possible sources of contamination, provide insight into the current water quality status of the rivers and identify hotspots that may need closer monitoring.

Materials and Methods

The study area

Dar es Salaam city is located at 6°48’ South and 39°17’ East, on the coast of Tanzania. The total surface area of the city is about 1,800 km^2, comprising of about 1,400 km^2 of land mass (NBS, 2019). The city had a population of 4,364,541 according to the 2012 census and an intercensal growth rate of 5.6%. The population in 2021 is thus estimated at 6,015,000. Due to its proximity to the equator and the Indian Ocean, the city experiences tropical climatic conditions characterized by hot and humid weather through much of the year. The average annual precipitation is around 1100 mm (Mtoni *et al.*, 2012). December and January have an average precipitation of up to 195 mm, while in July and August it is much lower, up to 47 mm. The area, like many other parts of the country, has distinct rainy and dry seasons; the long rain season between March to May, followed by several months of dry season around June to September (Mahongo and Francis, 2012). River flows in the Dar-es-Salaam area is mainly controlled by the precipitation rate, and are normally high during the rainy season (up to 15 m^3/s) and low during the dry season (1 m^3/s or lower) (Van Camp *et al.*, 2013).

This study considered three rivers, namely the Kizinga, Mbezi and Msimbazi, that run through Dar es Salaam city towards the coast. The Kizinga River (about 30 km long) flows through the urbanised areas of Mbagala, Buza, Chang’ombe, Keko, Mtongani and Kurasini and through the Mtoni mangroves and over a mudflat before draining into the Indian Ocean. As a result, it is expected to collect mixed wastes from households, agriculture, industries, and car and truck washing activities along its catchment (Kruitwagen *et al.,* 2008). The Msimbazi River flows for about 36 km from the Kisarawe hills to the shores of the Indian Ocean, passing through highly populated and industrialized areas. Along its course it receives different industrial (dye, textiles, soap and detergent, breweries, building material and food), agricultural (fertilizers and pesticides) and household wastes. The Mbezi River is the shortest of the three, about 24 km long, flowing through the less populated areas of Kawe, Mbezi Beach and other nearby areas on the western side of the city, and discharges into the Indian Ocean. Along its course, the Mbezi River passes through fewer industrial areas but more human settlements that generate household wastes and sewage discharges (Mhina *et al.*, 2018).

Sample collection

Water samples were collected from fifteen stations, five on each river. Figure 1 is a map of Dar es Salaam, indicating the sampling locations in the three rivers.

Table 1 gives the code names and position descriptions of the sampling locations. Two sampling campaigns were conducted in March to April 2018 for the wet season and another two in August to September 2018 for the dry season. Samples were collected from the side of the riverbanks at the same spots during the two campaigns, as identified by a hand-held global positioning system (GPS).

were acidified after collection to pH < 2.0 using 5 mL concentrated HNO_3/L. All sample bottles were placed in ice-cases at <10 °C and transported to the Analytical Chemistry Laboratory at the Chemistry Department of the University of Dar Salaam for analyses.

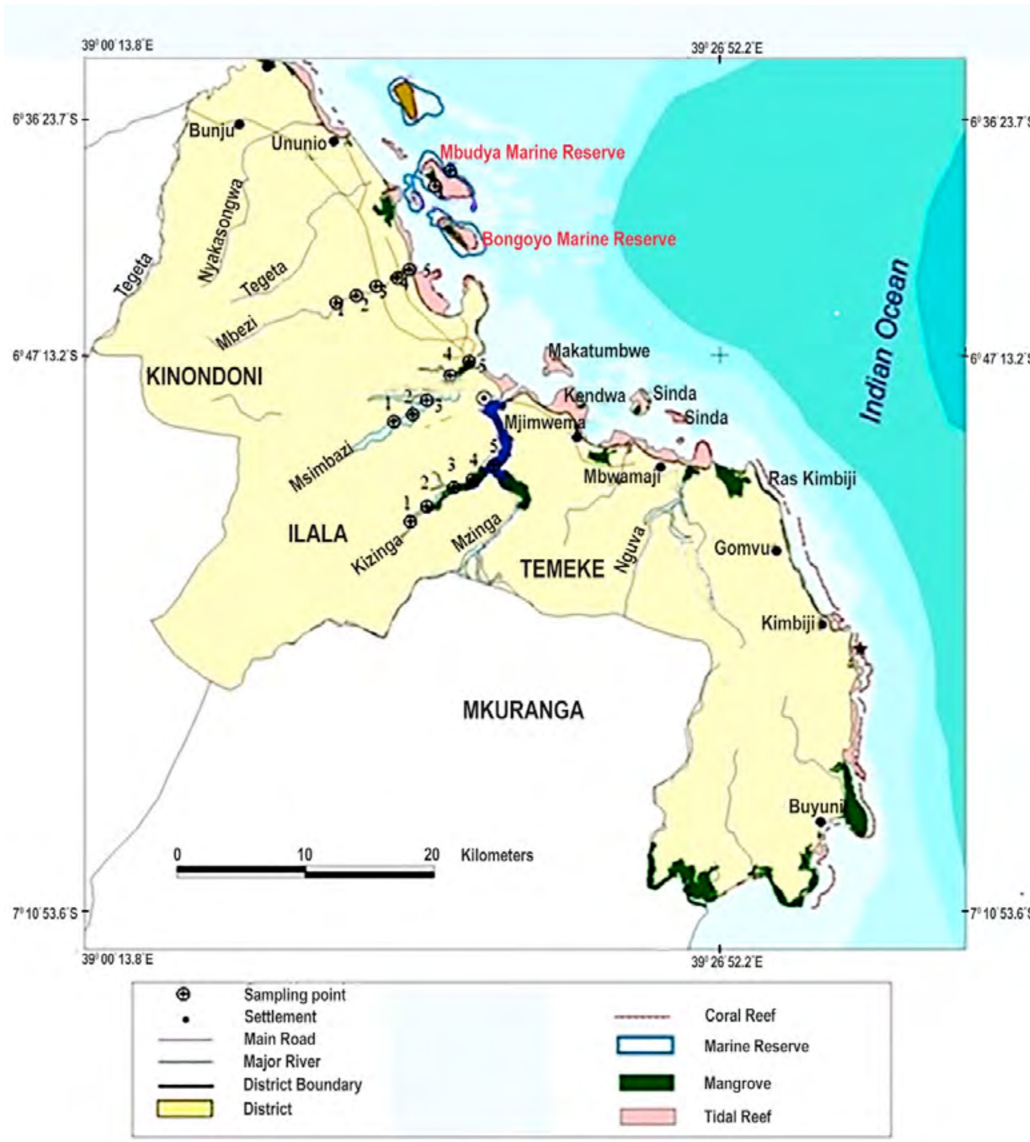

Figure 1. Map of Dar es Salaam showing the sampling locations along the three rivers.

To collect the samples, clean pre-labelled Teflon capped plastic bottles (1 litre) were vigorously rinsed twice and filled with river water at each station. Field parameters were measured and recorded on site. Two water samples were taken at each station, one for the determination of the conventional physicochemical parameters in the laboratory and the other for analysis of metals. Water samples for metal determination

Sample Analysis

Water samples were analysed for a total of twenty-one parameters, i.e., fourteen physicochemical parameters (pH, total dissolved solids (TDS), electrical conductivity (EC), total suspended solids (TSS), turbidity, dissolved oxygen (DO), chemical oxygen demand (COD), alkalinity, total hardness, salinity, chloride (Cl^-), sulphate (SO_4^{2-}), phosphate (PO_4^{3-}) and ammonium

Table 1. Description of the sampling sites.

River	Station Code	GPS Coordinates		Elevation (Meters)
Kizinga	Kz_1	S 06°53.060′	E 039°15.853′	8
	Kz_2	S 06°52.987′	E 039°16.176′	6
	Kz_3	S 06°52.903′	E 039°16.521′	8
	Kz_4	S 06°52.439′	E 039°17.054′	4
	Kz_5	S 06°52.107′	E 039°17.485′	4
Msimbazi	Ms_1	S 06°49.125′	E 039°15.318′	10
	Ms_2	S 06°48.943′	E 039°15.764′	12
	Ms_3	S 06°48.592′	E 039°15.921′	7
	Ms_4	S 06°48.236′	E 039°16.171′	11
	Ms_5	S 06°47.775′	E 039°16.849′	3
Mbezi	Mz_1	S 06°4.098′	E 039°1.020′	25
	Mz_2	S 06°42.728′	E 039°13.689′	8
	Mz_3	S 06°42.715′	E 039°13.901′	6
	Mz_4	S 06°42.696′	E 039°13.967′	6
	Mz_5	S 06°42.592′	E 039°13.984′	7

(NH_4^+), and seven metals (Al, Cr, Cu, Fe, Pb, Zn and Cd). Five parameters (pH, TDS, EC, turbidity and DO) were measured *in-situ* by hand-held portable water-quality instruments and their results recorded on site, whereas the rest of the parameters were analyzed in the laboratory upon arrival.

The measurements of pH, EC and TDS were done by a low range combo® pH/EC/TDS meter (Model HI 98129, HANNA Instruments Inc., USA) which measures the three parameters simultaneously. Calibration was done as per the manufacturer's instructions, using the manufacturer-supplied pH standard buffer solutions of pH 4.01, 7.01 and 10.01, and the conductivity standard 1413 μS/cm. Dissolved oxygen was measured by a digital DO meter (Model HI 98186, HANNA Instruments Inc., USA), which was calibrated as described in the manufacturer's manual. Turbidity was measured using an electronic turbidity meter (nephelometer) (Model HI 98713, HANNA Instruments Inc., USA) that utilizes the principle of light scattering to measure turbidity in terms of nephelometric turbidity units (NTU). Measurements involved calibration of the meter with the manufacturer's supplied turbidity standards.

The analysis of TSS was done by the filtration process, Standard Method 2540 D (APHA/AWWA/WEF, 2017). In this method, 250 ml water samples were filtered on pre-weighed Whatman® glass microfiber filters (2.0 μm particle retention) and the filters dried at 105 °C for 2-3 hours, then re-weighed to determine the weight change, which corresponded to the TSS content expressed in mass per volume of sample filtered (mg/L). The analysis of COD was done by the open reflux method, Standard Method 5220 B (APHA/AWWA/WEF, 2017), in which the organic matter in the sample (20 mL) was oxidized by refluxing in a mixture of concentrated H_2SO_4, 0.25N solution of $K_2Cr_2O_7$, and dried powders of Ag_2SO_4 and $HgSO_4$ for 2 hours on a heating mantle. The mixture was then titrated against standardized ferrous ammonia sulphate (FAS) solution with ferroin solution as an indicator. Finally, COD was calculated as mg of O_2 per mL of sample. Total hardness was determined by the Ethylene-diamine-tetra-acetic acid (EDTA) titrimetric method, i.e., Standard Method 2340 C (APHA/AWWA/WEF, 2017), in which 25 mL of water sample added with 1 mL of ammonia buffer reagent was titrated against EDTA solution using Eriochrome Black T indicator. Total hardness was then calculated as mg ($CaCO_3$)/mL sample. Alkalinity was determined by the titration method, Standard Method 2320 B (APHA/AWWA/WEF, 2017), in which 10 mL water sample was titrated against a standardized H_2SO_4 solution using phenolphthalein and methyl orange as indicators. The total volume of titrant was used to estimate the alkalinity of the sample. Salinity was determined by the conductivity method, Standard Method 2520 B (APHA/AWWA/WEF, 2017). Phosphate (PO_4^{3-}–P) was determined by the ascorbic acid method, Standard Method 4500-P E (APHA/AWWA/WEF, 2017), which involved treating water samples (50 mL) with ammonium molybdate and potassium antimony tartrate, which react with any orthophosphate present in the sample to form an antimony-phosphate-molybdate complex. This was then reduced by ascorbic acid, and the resulting colour intensity was measured spectrophotometrically using an Ultraviolet-Visible (UV) spectrophotometer at 880 nm. Sulphate was determined as SO_4^{2-} ions using the turbidimetric method, Standard Method 4500-SO_4^{2-} E (APHA/AWWA/WEF 2017), which is based on the principle of conversion of the SO_4^{2-} ion to barium sulfate ($BaSO_4$) under controlled conditions. In the procedure, about 50 g of $BaCl_2$ crystals were added to a 50 mL filtered water sample which contained a buffer solution prepared as per the description in the method. The resulting solution was then stirred for one minute and thereafter its absorbance was determined by a spectrophotometer at 420 nm. The SO_4^{2-} concentration was calculated by comparison with standard curves.

Chloride ion (Cl^-) was determined by the argentometric method, Standard Method 4500-Cl^- B (APHA/AWWA/WEF, 2017), in which a 50 mL water sample was directly titrated using K_2CrO_4 indicator against standardized $AgNO_3$ titrant until a yellow-colored solution was converted to a persistent brick red colour. The concentration of NH_4^+ was determined by the phenate method, Standard Method 4500-NH_3-F (APHA/AWWA/WEF, 2017). In the procedure, 20 mL of water sample in a 25 mL volumetric flask was added with 2 mL of phenol-nitroprusside solution, followed by 2 mL of alkaline hypochlorite solution and distilled water to make 25 mL of solution. The absorbance of the sample was recorded on a spectrophotometer at 635 nm.

Metals were analyzed instrumentally by Flame Atomic Absorption Spectrometry (FAAS) (iCE3000 SERIES, Thermo Scientific) using standard method 3111A as described in APHA/AWWA/WEF (2017). For each metal determined, the instrument was calibrated using working standards for different elements at different concentrations prepared in 5 % (v/v) HCl. The calibration curves that aid to estimate concentration of the intended analyte were made for each element. Determination of each element was achieved at specific wavelength, such as 309.2 nm for Al, 228.7 nm for Cd, 324.7 nm for Cu, 357.8 nm for Cr, 248.3 nm for Fe, 216.9 nm for Pb, and 213.8 nm for Zn. The detection levels were as described in the specific method (APHA/AWWA/WEF (2017).

Quality Assurance and Control

The quality of the analytical data was assured in all steps of the study, i.e., from sample collection, transport, laboratory analysis, and finally data checks. In the sampling procedure, all tools were thoroughly cleaned and rinsed with distilled water, and the instruments were calibrated using prescribed protocols by the instruments' manufacturers. Samples were carefully handled by ensuring correct labelling of the sample bottles, packaging and transporting of the samples to the laboratory. In the laboratory, the quality of analysis was assured by analyzing samples of distilled water as blank samples. After ensuring that the blank samples were free of the target analytes, they were spiked with standard solutions of the analytes at their method detection limits and processed by the same method as used for the ordinary samples, as described in the respective methods. The data checks for all field and laboratory measurements were done by ensuring that the obtained results were within the expected ranges of a particular parameter, that they were physically and scientifically possible, and that they were within the detection limits of the method used.

Data Analysis

The datasets were tested for normality using the Shapiro-Wilk test, in which a null hypothesis that a variable is normally distributed was rejected if $p < 0.05$. Descriptive statistics were used to summarize results for concentration ranges, means and standard deviations. The seasonal variation trends were studied using the Wilcoxon Signed Ranks test for paired samples, in which the difference in the measured values between the two seasons was tested for statistical significance at the 95 % confidence level. The Friedman test for multiple related samples was employed to compare levels of the measured parameters among the three rivers. The dataset was further subjected to Spearman Rank correlation analysis for non-normal data to evaluate the relationships among the measured parameters. Factor reduction using Principal Component Analysis (PCA) with Varimax rotation was used to reduce the large dataset to a small number of new variables that could account for at least 75 % of the total variance and explain the overall associations among the measured parameters. Those with eigenvalues ≥ 1.0 were considered significant. Hierarchical Cluster Analysis (HCA) was used in characterizing the different sampling locations to assess the similarities and differences among them and identify possible patterns of the measured parameters. All statistical analyses were carried out using the statistical package IBM SPSS Statistics 23.0.

Table 2. Concentration ranges, means and standard deviations (n = 30) of the measured water quality parameters in the rivers.

Parameter	Wet season (n = 15)		Dry season (n = 15)		TBS	WHO
	Range	Mean ± SD	Range	Mean ± SD		
pH	7.44 – 11.41	8.28 ± 1.08	6.83 – 9.13	7.92 ± 0.53	5.5 – 9.5	6.5 – 9.2
TDS (mg/L)	252 – 34333	9077 ± 12882	203 – 30236	7860 ± 11457	1500	1000
EC (μS/cm)	500 – 68014	17982 ± 25520	408 – 61205	17038 ± 23701	2500	2500
TSS (mg/L)	12.8 – 50.7	28.1 ± 11.9	11.9 – 34.7	22.1 ± 7.1	-	-
Turbidity (NTU)	10.8 – 45.0	23.7 ± 10.9	10.0 – 29.2	18.9 ± 6.61	5	< 5
DO (mg/L)	1.4 – 6.6	4.3 ± 1.7	2.6 – 6.4	4.9 ± 1.1	-	-
COD (mg/L)	105 – 1863	725 ± 590	91 – 1774	649 ± 523	-	
Alkalinity (mg/L)	200 – 2658	611 ± 625	207 – 1145	461 ± 269	-	500
Total Hardness (mg/L)	473 – 101333	21890 ± 30412	362 – 121312	23913 ± 37104	600	500
Salinity (ppt)	0.19 – 25.70	7.76 ± 9.97	0.21 – 29.35	7.47 ± 10.38	-	
Chloride (mg/L)	108 – 14248	4228 ± 5571	114 – 16245	4137 ± 5745	250	
Sulphate (mg/L)	42 – 766	273 ± 258	35 – 723	254 ± 239	400	500
Phosphate (μg/L)	ND – 2.74	1.31 ± 1.11	ND – 3.01	1.07 ± 1.00	-	
Ammonium (μg/L)	90 – 5468	1432 ± 1581	40 – 2131	585 ± 611	500	

Results and discussion

Physicochemical parameters

Descriptive data for measured water quality parameters at the fifteen stations during the two seasons are summarized in Table 2. The physicochemical quality of water was assessed based on a comparison to limits for potable water provided by the TBS (TBS, 2008) (TZS 789: 2008) and WHO (WHO, 2011). The normality test showed that all parameters were not normally distributed, due to the presence of extreme values.

Figures 2(a) and 2(b) depict the variation in trends for mean values for each parameter as measured at individual sampling stations.

The results of Spearman rank correlation analysis used to evaluate associations between pairs of measured parameters at the 15 stations are summarized in Table 3.

The data show that the river waters were near neutral to alkaline, with a pH ranging between 7.44 – 11.41 (8.28 ± 1.08) during the wet season and 6.83 – 9.13 (7.92 ± 0.53) during the dry season (Table 2). The pH of freshwater is an important parameter as it reflects on the state of pollution and productivity of the water. The Tanzanian pH limit for potable water is 5.5 – 9.5, whereas the maximum permissible limits for potable water according to the WHO is 6.5 – 9.2 (Table 2). The typical values for natural river and stream water are said to range from 4 to 11 (Chapman and Kimstach, 1996). Figure 2a shows that the lowest pH was recorded at the Mbezi River station Mz_2 and the highest pH of 11.41 was recorded at Msimbazi River at station Ms_1 during the wet season. This station, which is upstream and close to presumed pollution sources, has a pH that exceeds the maximum permissible limits for both the national and WHO limits. Generally, all the Msimbazi River stations were found to have a relatively higher pH compared to the other rivers. For the Mbezi and Kizinga Rivers, the pH was within the WHO and TBS limits. When data from the three rivers were subjected to the Friedman test, a statistically significant difference was confirmed based on their median ranks i.e., Msimbazi River (8.45), Kizinga River (7.88), Mbezi River (7.87), ($\chi^2(2) = 8.600$, $p = 0.014$). This could be an indication of inputs of compounds that make the water more alkaline in this river, given its surroundings and the human activities taking place around its banks. The pH was positively correlated with TSS, turbidity and alkalinity, as shown in Table 3. Studies have indicated that waters that have moderate to high levels of total alkalinity (> 50 mg/L)) usually have neutral to slightly basic pH (Nagwa, 2016). This is clearly depicted in this study where alkalinity was higher than 200 mg/L and pH higher than 6.8.

The TDS concentrations ranged between 252 – 34,333 mg/L during the wet season and 203 – 30,236 mg/L during the dry season (Table 2). The acceptable TBS limit of TDS in potable water is 1,500 mg/L while that of the WHO is 1,000 mg/L (Table 2). Both these limits were exceeded at about 50 % of the stations during both seasons, e.g., Kz_4, Kz_5, Ms_1, Ms_5, Mz_3, Mz_4 and Mz_5.

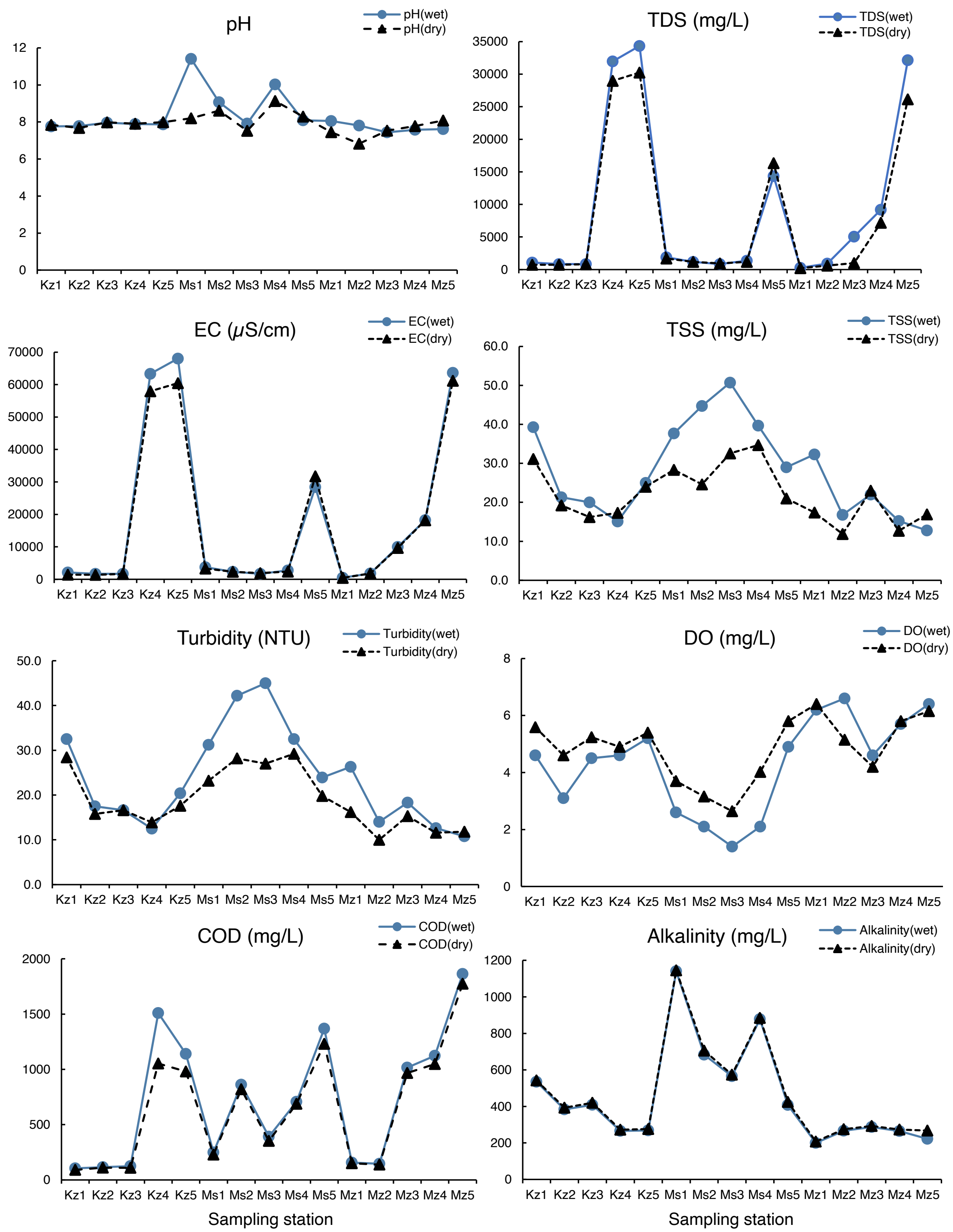

Figure 2(a). Variations of the mean values of pH, TDS, EC, TSS, Turbidity, DO, COD, and Alkalinity in the study area during the two seasons.

The highest concentrations during both seasons were measured at station Kz_5 followed by station Kz_4 (Fig. 2a). The high TDS concentration observed at these locations that are correlated to salinity may be due to saltwater intrusion (Anhwange *et al.*, 2012). The levels recorded in the three rivers in this study exceed by 30-fold the TBS permissible limit, which indicated significant deterioration of water quality. Based on the Friedman test results, the Kizinga River was found to have the highest TDS concentrations.

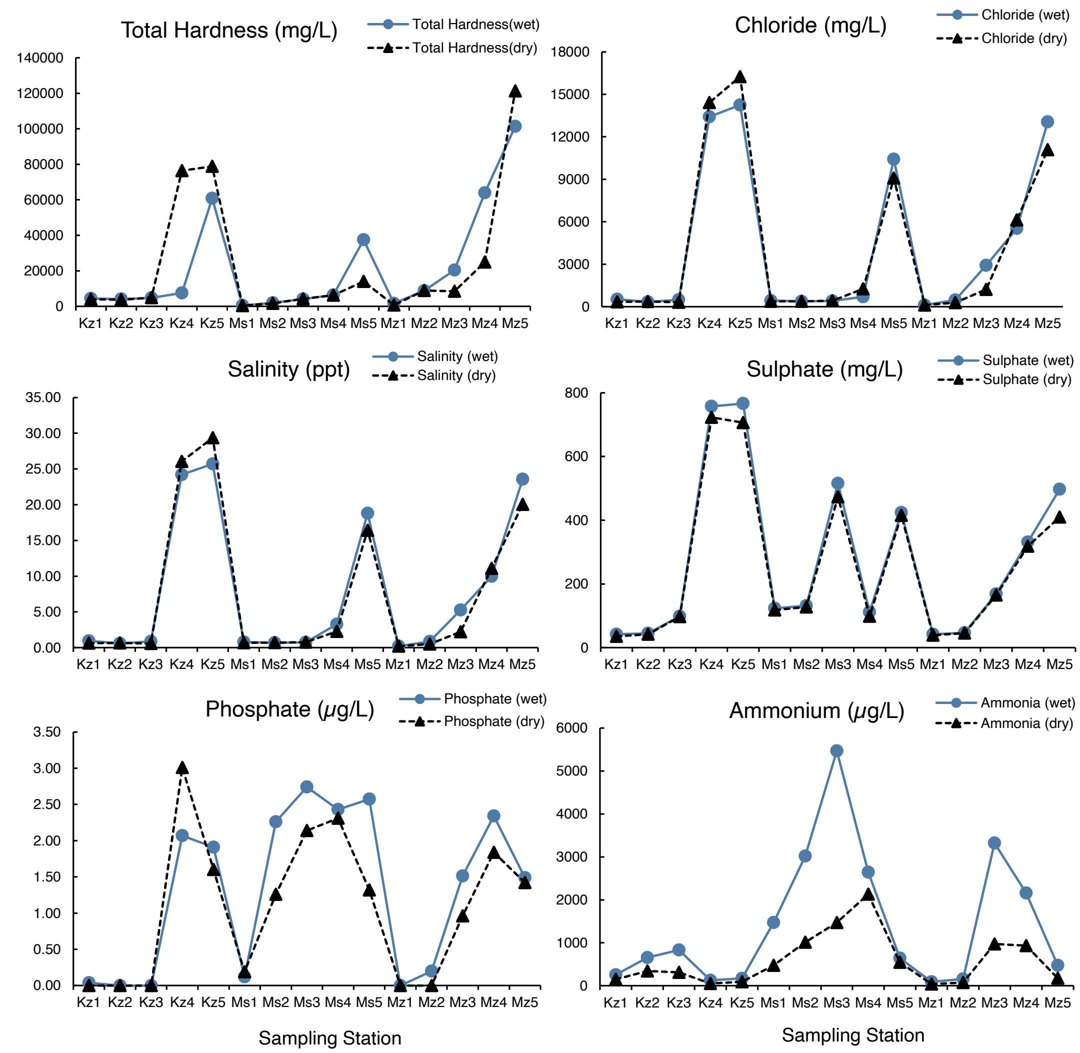

Figure 2(b). Variation of the mean values of Total Hardness, Chloride, Salinity, Sulphate, Phosphate, and Ammonium in the study area during the two seasons.

The ranges of EC recorded in this study were 500 – 68,014 µS/cm and 408 – 61,205 µS/cm during the wet and the dry seasons respectively (Table 2). These concentrations also significantly exceed both the national (TBS) and the WHO limits of 2,500 µS/cm in most of the locations as well as the normal range of EC in natural river waters, which is expected at 50 – 1500 µS/cm (Patil *et al.*, 2012). The highest concentrations were recorded at Kz_4 followed by Kz_5 and Mz_5 (Fig. 2a). The high EC concentrations probably reflect saltwater intrusion. Researchers have established that EC and TDS are highly positive correlated because the conduction of electric current primarily depends on the concentration of ionic species, and that EC values are almost usually twice those of TDS (Siosemarde *et al.*, 2010). The same pattern was observed in this study (Fig. 2a). TDS and EC were highly correlated (r = 0.977) as shown in Table 3. As expected, both TDS and EC were positively correlated with COD, total hardness, Cl^-, salinity, SO_4^{2-} and PO_4^{3-} (Table 3).

Total suspended solids (TSS) are particles of organic or inorganic nature that are larger than 2 microns found in the water column, anything smaller than that is considered a dissolved solid (Chapman and Kimstach, 1996). The concentrations of TSS recorded in this study were 12.8 – 50.7 mg/L during the wet season and 11.9 – 34.7 mg/L during the dry season (Table 2). The highest concentrations were measured at Ms_3 followed by Ms_2 and Ms_4 (Fig. 2a). The Friedman test showed the Msimbazi River to generally have the highest levels of TSS compared to the other rivers, with median ranks of Msimbazi River (33.65), Kizinga River (20.65), Mbezi River (16.85) ($\chi^2(2)$ = 9.800, p =

Table 3. Spearman Rank correlation matrix between pairs of water quality parameters (n = 30).

	pH	TDS	EC	TSS	Turb.	DO	COD	Alkal.	T. Hard.	Cl^-	Salin.	SO_4^{2-}	PO_4^{3-}	NH_4^+
pH	1.000													
TDS	0.243	1.000												
EC	0.149	0.977**	1.000											
TSS	0.485**	-0.132	-0.206	1.000										
Turb.	0.532**	-0.176	-0.277	0.960**	1.000									
DO	-0.318	0.106	0.143	-0.642**	-0.621**	1.000								
COD	0.137	0.861**	0.890**	-0.216	-0.287	0.187	1.000							
Alkal.	0.541**	-0.128	-0.190	0.679**	0.736**	-0.767**	-0.313	1.000						
T. Hard.	-0.208	0.666**	0.725**	-0.490**	-0.556**	0.472**	0.673**	0.502**	1.000					
Cl^-	0.096	0.920**	0.918**	-0.187	-0.263	0.181	0.804**	-0.232	0.800**	1.000				
Salin.	0.107	0.923**	0.917**	-0.180	-0.254	0.175	0.803**	-0.227	0.799**	0.999**	1.000			
SO_4^{2-}	0.101	0.827**	0.828**	-0.083	-0.147	-0.062	0.839**	-0.139	0.605**	0.798**	0.797**	1.000		
PO_4^{3-}	0.235	0.650**	0.616**	0.218	0.162	-0.255	0.697**	0.091	0.464**	0.651**	0.658**	0.760**	1.000	
NH_4^+	0.216	0.078	0.045	0.440*	0.454*	-0.651**	0.134	0.562**	-0.136	0.017	0.020	0.167	0.425*	1.000

**. Correlation is significant at the 0.01 level (2-tailed).
*. Correlation is significant at the 0.05 level (2-tailed).

0.007). One of the common sources of TSS in rivers is soil erosion along the riverbanks, particularly caused by human activities (Rossi *et al.*, 2006). The Msimbazi River banks are prone to erosion from vegetable farming taking place around them, especially during the rainy season. The TSS levels were found to positively correlate with alkalinity (Table 3).

Turbidity is a measure of the relative clarity of water that indicates the level of suspended and colloidal matter, such as clay, silt, finely divided organic and inorganic matter and microscopic organisms that may interfere with the passage of light through the water (Awoyemi *et al.*, 2014). The turbidity values recorded in this study were 10.8 – 45.0 NTU during the wet season and 10.0 – 29.2 NTU during the dry season. The limit set by both the WHO and TBS for turbidity in potable water is 5 NTU (Table 2). This was exceeded in all fifteen locations during both seasons. Turbidity and TSS measure approximately the same water quality property except that TSS provides an actual weight of the particulate material present in the sample (Araoye, 2009). The two parameters are often highly positively correlated as was also observed in this study ($r = 0.960$; Table 3). As was observed for the TSS concentrations, the Msimbazi River had the highest levels of turbidity, probably due to the same reasons of human activities such as vegetable farming that are taking place along the river banks.

Dissolved oxygen is an important water quality parameter since it is required for the maintenance of aerobic conditions in the water column. It is one of the major parameters of interest in water quality assessment in this study because it indicated changes in physical, chemical, and biological processes induced by both natural and anthropogenic activities within the water. Water is considered generally healthy when its DO range is 6.5 – 8.0 mg/L and 80 – 120 % saturation (Araoye, 2009). Data collected in this study indicated DO concentration ranges of 1.4 – 6.6 mg/L and 2.6 – 6.4 mg/L during the wet and dry season respectively. This showed depletion of the DO concentration at some locations, such as Ms_1, Ms_2, Ms_3 and Ms_4 (Fig. 2a). Among the reasons for low DO concentration in a body of fresh water is decomposition of organic matter (Araoye, 2009). The decrease in DO was also found to correlate with increase in TSS and turbidity (Table 3). The Friedman test showed that the Msimbazi River generally had the lowest concentrations of DO compared to the other rivers. This continues to assert the earlier suggestion of the possibility of relatively higher pollution load in the Msimbazi River compared to the other rivers.

The data gathered in this study indicated COD ranges of 105 – 1,863 mg/L and 91 – 1,774 mg/L during the wet and the dry season respectively. The COD measurements in natural river water is used to indicate the quantity of oxygen required to chemically oxidize organic contaminants in water into inorganic end products (Khan and Ali, 2018). It is useful for measuring human impact on water quality of rivers. The COD concentrations in surface water resources is expected to typically range from 20 mg/L or less in unpolluted waters to greater than 200 mg/L in waters receiving wastewater effluents (Chapman and Kimstach, 1996). The high COD concentrations exceeding 1,000 mg/L measured in some locations (Table 2) indicate that the rivers are generally impacted by organic wastes from human activities, likely to be from discharges of sewage and wastewaters. A study by Mihale et al. (2021) observed that sewage is an important anthropogenic source impacting the coastal rivers in this region. Furthermore, vegetable farming activities along the Msimbazi River could be producing organic wastes due to application of organic manure. The levels of COD were found to be positively correlated with TDS, EC, total hardness, Cl-, salinity, SO_4^{2-} and PO_4^{3-} (Table 3).

Alkalinity levels measured in this study ranged between 200 – 2,658 mg/L during the wet season and 207 – 1,145 mg/L during the dry season. Measuring the alkalinity of a freshwater body is important in determining its ability to neutralize acidic pollution from different sources such as acid rainfall, wastewaters, and agricultural practices. It is one of the best measures of the sensitivity of the river to acid inputs and its buffering capacity or resistance to pH changes upon the addition of acids or bases (Patil *et al.*, 2012). Although alkalinity has no known adverse health effects, its levels affects the palatability of water and its suitability to piping and use for domestic purposes (EWURA, 2020). The alkalinity of natural waters is primarily due to the presence of weak acid salts such as bicarbonate, although strong bases such as OH^- may also contribute in extreme environments. Normal river water is expected to have an alkalinity ranging between 100 and 250 mg/L (Mattson, 2014). The WHO set a permissible limit of 500 mg/L for drinking water (Table 2). This level was exceeded in a few locations, particularly in the Msimbazi River (Fig. 2a), probably due to inputs of alkali-inducing materials such as soap and detergent residues from industries along its catchment. The strong negative correlation with DO indicates respiration of organic matter as another source of increased alkalinity due to the production HCO_3^-, which has a potential of making the system highly heterotrophic. In this case high alkalinity is not a sign of good health for river water (Nyanda *et al.*, 2016). Alkalinity was also positively correlated to total hardness (Table 3), an association that has been established in other studies (e.g., Boyd *et al.*, 2016).

The total hardness measured in the three rivers in this study ranged from 473 – 101,333 mg/L during the wet season, and 362 – 121,312 mg/L during the dry season. Total hardness and alkalinity are often related because the main source of alkalinity is usually from carbonate rocks (limestone) which are mostly $CaCO_3$ (Boyd *et al.*, 2016). However, hardness in water is not caused by a single substance but by a variety of dissolved metallic ions, predominantly Ca^{2+} and Mg^{2+}, although other cations such as aluminium, barium, iron, manganese, strontium, and zinc, mostly from natural sources, may also contribute (WHO, 2010). The WHO potable water limit for total hardness is 500 mg/L, whereas the TBS limit is 600 mg/L (Table 2). Hard water is not a health hazard: in fact, some researchers have investigated the potential beneficial health effects of low levels of hardness in drinking water (Ong *et al.*, 2009). However, the levels measured in some locations in this study e.g., Kz_5, Ms_5, and Mz_5, exceed the beneficial level.

Chloride occurs naturally at low concentrations in freshwater bodies, such as the three rivers in this study. The main source is dissociation of salts, such as NaCl or $CaCl_2$, in water (Berger, 2019). The data obtained revealed chloride concentrations of 108 – 14,248 mg/L during the wet season and 114 – 16,245 mg/L during the dry season. Chloride levels in unpolluted river waters are expected to be below 40 mg/L and usually in the range 15 – 35 mg/L for rivers and other freshwater bodies (Alam *et al.*, 2007). The WHO sets a maximum limit of 250 mg/L chloride in drinking water. This value was exceeded at all stations except Mz_1. An increase in chloride concentration at levels above 250 mg/L will begin to make water taste salty (EWURA, 2020). The high chloride levels indicate inputs from the sea (sea water infiltration), particularly for the stations located close to the ocean, such as Kz_5, Mz_5, and Ms_5.

Salinity measured in the three rivers ranged from 0.19 – 25.70 ppt during the wet season, and 0.21 – 29.35 ppt during the dry season, with means of 7.76 ppt and 7.47 ppt respectively. Fresh water from rivers is expected to have 0.5 ppt or less of salinity, produced by natural processes such as weathering of rocks and rain deposits. These small amounts of dissolved salts are vital for the life of aquatic plants and animals. However, the salinity levels measured in these locations can be considered normal and natural for rivers connected to the oceans in estuarine systems (Berger *et al.*, 2019). The highest concentrations of both chloride and salinity were measured in samples from Kizinga River (Fig. 2b), particularly at stations Kz_4 and Kz_5, probably due to their proximity to the ocean. Salinity and Cl^- followed the same spatial trend as shown in Fig. 2(b).

The two parameters are highly positively correlated (r = 0.999) as shown in Table 3, because salinity is measure of the total salt concentration, comprised mostly of Na^+ and Cl^- ions, whereas Cl^- come from the dissociation of salts, such as NaCl in water (Berger *et al*., 2019).

Sulphates occur naturally in numerous minerals and are used commercially, principally in the chemical industry. They may be discharged into water in industrial wastes and through atmospheric deposition (WHO, 2003a). Sulphate levels measured in the three rivers in this study ranged from 41.7 to 766.2 mg/L during the wet season and 35.3 to 722.9 during the dry season. Both the WHO and TBS limits of sulphate in potable water (Table 2) were exceeded at some locations in the Kizinga and Msimbazi Rivers (Fig. 2b). The highest concentrations during both seasons were measured at Kz_4 and Kz_5. Seawater may have contributed to sulphate levels at some locations. However, the high sulphate concentration at the two stations are outliers and alluded to a point source, probably sewage treatment plants and industrial discharges from textile mills around the area. However, most of the other stations had rather low levels of sulphates. Noticeable changes in water taste are said to occur in concentrations exceeding 250 mg/L (WHO, 2003a), therefore water at locations Kz_4, Kz_5, Ms_5 and Mz_5 that have sulphate levels exceeding this concentration may be affected.

Concentrations of phosphate (PO_4^{3-}–P) were found to range from below the method detection limit to 3.01 µg/L. Phosphate is rarely found in high concentrations in fresh waters since it is actively taken up by plants, and its concentration in natural river waters is expected to range from 0.005 to 0.05 mg/L (Chapman and Kimstach, 1996). The phosphate concentrations in this study were well below this level at all sampling locations, but were the highest at Msimbazi River stations, especially during the wet season, probably due to run-off from agricultural activities. Wastewater is also an important source of phosphate, which is often a constituent of detergents.

Ammonium was detected at all sampling stations in concentrations ranging between 90 – 5,468 µg/L during the wet season, and 40 – 2,131 µg/L during the dry season. The TBS limit for ammonium in potable water is 500 µg/L, which was exceeded at most stations in the Msimbazi River and at some stations in the Mbezi River (Fig. 2b). The Msimbazi River was found to have the highest concentrations of NH_4^+ (Fig. 2b), especially at station Ms_3, whose water was detected to produce an unpleasant smell almost all along the sampling station. This was confirmed by the Friedman non-parametric test which revealed the median ranks of 211 (Kizinga River), 1,470 (Mbezi River) and 323 (Msimbazi River), ($\chi^2(2) = 15.200$, $p = 0.001$). The highest concentration of 5,468 µg/L was detected at Ms_3 (Fig. 2b), a station which also had lowest DO concentration. Ammonium occurs naturally in freshwaters, though in very small amounts, because of microbiological activity which causes the reduction of nitrogen-containing compounds. Usually the total ammonia concentration (NH_3–N + NH_4^+) in surface waters is <200 µg/L but may reach 2,000 – 3,000 µg/L. Its contamination sources in fresh water include wastewater and fertilizers (Chapman and Kimstach 1996). The high concentrations measured at some of the Msimbazi River stations are indications of contamination, most likely from sewage and farming activities.

Metals

The descriptive data for the measured metal concentrations from the fifteen stations are summarized in Table 4, with comparison to the Tanzanian national standards and the WHO limits.

Table 4. Ranges and mean ± SD (N = 30) concentrations (µg/L) of metals in the rivers.

Metal	Wet season		Dry season		Tanzania	WHO
	Range	Mean ± SD	Range	Mean ± SD		
Al	274.2 – 938.2	561.2 ± 223.2	241.5 – 746.0	437.4 ± 161.2	200	200
Cr	8.0 – 43.0	22.2 ± 10.9	1.2 – 31.6	18.2 ± 8.1	50	50
Cu	0.7 – 3.2	1.7 ± 0.7	0.3 – 3.4	1.7 ± 0.9	1000	2000
Fe	53.1 – 119.2	106.5 ± 35.5	42.6 – 112.5	82.6 ± 19.5	300	
Pb	0.7 – 24.0	9.0 ± 7.9	0.8 – 14.6	5.4 ± 4.2	10	10
Zn	114.6 – 175.6	153.3 ± 13.9	44.6 – 125.7	84.4 ± 23.3	500	300
Cd	BDL	-	BDL	-	3	3

BDL = Below Detection Limit

The data in Table 4 show that the Al concentration ranged between 274.2 – 938.2 μg/L during the wet season, and from 241.5 – 746.0 μg/L during the dry season. These concentrations exceed both the Tanzanian national standard and the WHO limit for Al in potable water of 200 μg/L. Aluminium occurs abundantly in nature in various concentrations depending on different physicochemical and mineralogical factors of a particular area, however it can also be easily enriched to high levels primarily due to its common use in construction, automotive, electricity, food packaging amongst other uses (WHO, 2003b). The highest concentrations during both seasons were measured at station Ms_1 followed by station Ms_2. The Msimbazi River generally had the highest concentrations of Al compared to the other rivers during both seasons (Fig. 3). This was also confirmed by the Friedman test, which revealed statistically significant difference in Al concentrations among the three rivers as Msimbazi River (n =10, M = 666), followed by Kizinga River (n = 10, M = 458) and Mbezi River (n = 10, M = 373), ($\chi^2(2) = 11.400$, $p = 0.003$). The concentrations measured in this study, especially in the Msimbazi River, are likely to be enriched from anthropogenic sources due to the range of human activities taking place in the area.

Chromium was found to range between 8.0 – 43.0 μg/L during the wet season and from 1.2 – 31.6 μg/L during the dry season. These concentrations were below the Tanzanian national standard and the WHO limit of 50 μg/L. The highest Cr concentration of 43.0 μg/L was measured at the Msimbazi River station Ms_1. The data show that generally the Msimbazi River had the highest levels of Cr during both seasons (Fig. 3). Chromium may contaminate river water through various diffuse sources arising from the use and disposal of materials and products from industrial and domestic activities.

The concentrations of Cu were 0.7 – 3.2 μg/L during the wet season and 0.3 – 3.4 μg/L during the dry season. These concentrations were far below the recommended Cu limit in potable water by both the TBS and the WHO limits (Table 4). The highest concentrations were again measured at the same two stations in the Msimbazi River, namely stations Ms_1 and Ms_2, which also had the highest concentrations of Al and Cr. However, the Cu concentrations detected in water in this study are too low to raise any alarm. The concentration of Fe of 53.1 – 119.2 μg/L during the wet season and 42.6 – 112.5 4 μg/L during the dry season were also far below the Tanzanian national standard of 300 μg/L. The highest concentrations during both seasons were again measured at the Msimbazi River station Ms_1. The statistical test revealed that Msimbazi River had generally the highest concentrations of Fe (n = 10, M = 105.1), followed by the Mbezi River (n = 10, M = 90.2) and Kizinga River (n = 10, M = 87.3), ($\chi^2(2) = 4.200$, $p = 0.022$).

The concentrations of Pb were 0.7 – 24.0 μg/L during the wet season and 0.8 – 14.6 μg/L during the dry season (Table 4). The maximum allowable limit of Pb in potable water by both TBS and WHO limits is 10.0 μg/L. This limit was exceeded at six of the 15 sampling locations, including three in the Msimbazi River and three in the Kizinga River. Station Ms_3 had the highest Pb concentration of 24 μg/L. Another station with a relatively high Pb concentration was Ms_4, which had 16.0 μg/L during the wet season and 14.6 μg/L during the dry season. The Friedman test performed on Pb data from the three rivers showed that Kizinga River had generally the highest levels, followed by Msimbazi River. Low levels of Pb are expected in river waters due to natural sources such as dissolution of rocks and soils. Anthropogenic sources include the production and use of lead-containing consumer products (WHO, 2003c). Tanzania is implementing legislation restricting the use of leaded fuels (Bultynck and Reliquet, 2003), therefore its importance as one of the largest sources of lead contamination is diminishing. A high concentration of Pb such as those found in some stations in this study is considered a health risk, since Pb is toxic and has no health benefit (WHO, 2003c).

The data in Table 4 also shows that Zn was detected in concentration ranges of 114.6 – 175.6 μg/L during the wet season and 44.6 – 125.7 μg/L during the dry season. These concentrations are well below the WHO limit of 300 μg/L as well as the TBS limit of 500 μg/L. The highest Zn concentration was detected at station Ms_4 during the wet season. The statistical test revealed that Msimbazi River had the highest concentrations of Zn. However, the Zn concentrations observed in this study are not alarming since they were all below the recommended limits, and were within the expected ranges for rivers that vary from <10 μg/L to >200 μg/L (Andarani *et al.*, 2021). Cadmium was not detected within the limits of the employed method in any of the sampling stations in this study.

The metal concentrations followed the order Al > Zn> Fe > Cr > Cu > Pb > Cd. When the whole dataset consisting of the physicochemical parameters and the metal concentrations was subjected to Spearman Rank correlation analysis, it was observed that Fe was positively correlated to pH (r = 0.563, at 0.05 level, 2-tailed). Studies have found that the pH of water is important for the solubility and biological availability of Fe since its concentration increases with increasing water pH (Nagwa, 2016). Concentrations of Al, Fe and Pb were all found to positively correlate with TSS (r = 0.488, 0.467 and 0.455 respectively). This indicated that TSS played an important role on the availability of the metals in water, although other factors might have been more significant. Studies have found that suspended particles in rivers can act as carriers of potentially bioavailable metal species (Nasrabadi *et al.*, 2018). Aluminium concentrations were also found to positively correlate with turbidity (r = 0.578), probably due to the same reason.

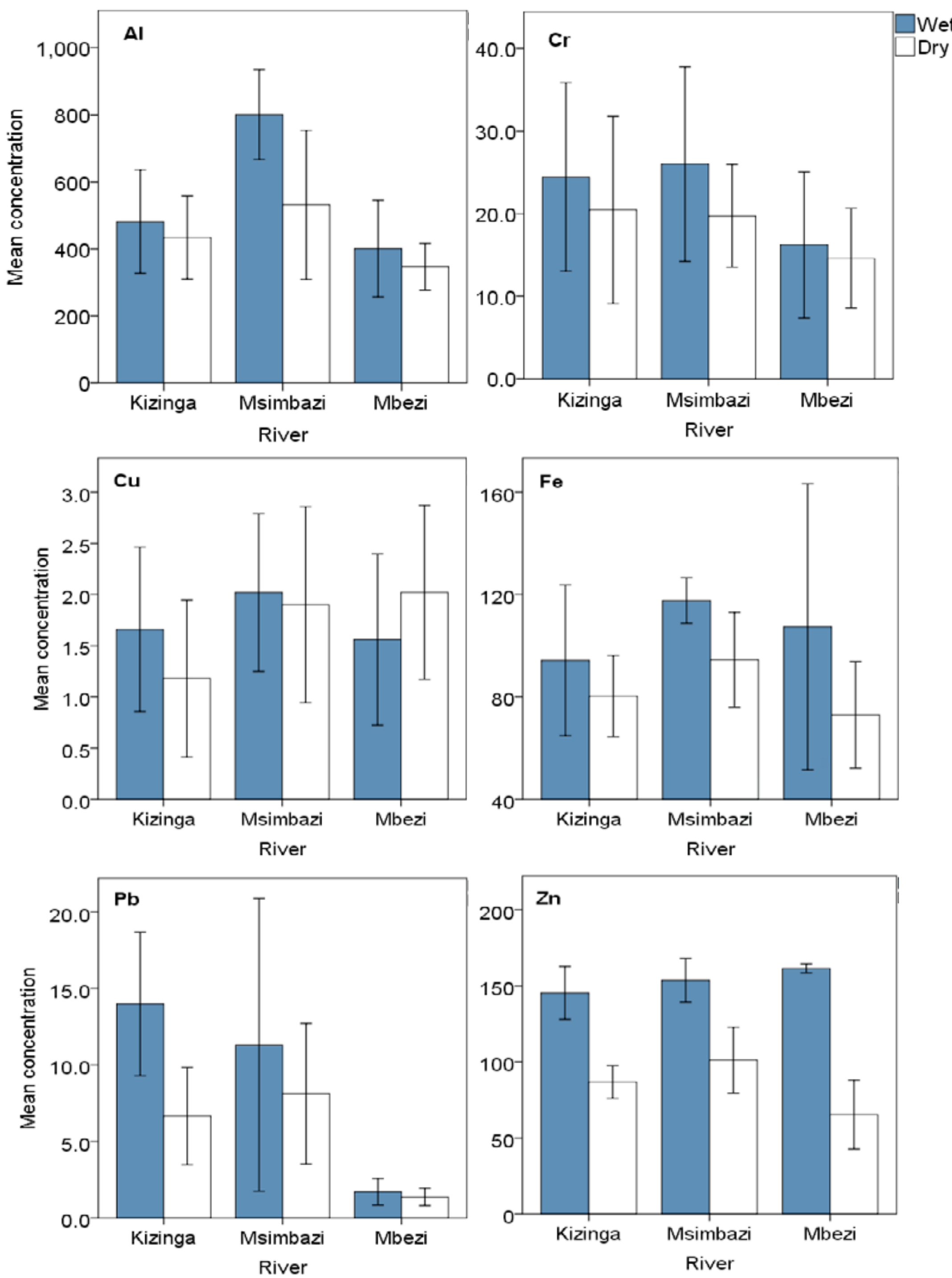

Figure 3. Mean concentrations (μg/L) of the six metals in the rivers during the wet and dry seasons.

Correlation among pairs of trace metal concentrations (Table 5) showed positive correlations between Al and Fe as well as between Pb and Cr. This indicated that the pairs were probably from the same sources.

Seasonal variations

On analyzing the seasonal variations of pH values by the Wilcoxon signed-ranks test, a statistically significant difference was observed, with higher values recorded during the wet season (n = 15, M = 8.28, SD = 1.08) than during the dry season (n = 15, M = 7.91, SD = 0.53), (z = -1.108, p = 0.021). This was accounted for by the fact that during the wet season rivers receive runoff that may increase water dilution and hence increase the concentrations of hydrogen ions that consequently reduce the pH. This has also been found by other researchers, e.g., Girardi *et al.* (2016) who studied water quality changes in rivers during rain events and found that pH showed some decreasing trends during rainfall. Statistically significant seasonal trends were also observed for both EC and TDS measurements. Higher EC values were recorded during the wet season (n = 15, M = 17,982) than the dry season (n = 15, M = 17,038), (z = -2.215, p = 0.027). Similarly, higher TDS values were observed during the wet season (n = 15, M = 9,077) than the dry season (n = 15, M = 7,860), (z = -2.542, p = 0.011). These variations might be attributed to increased inputs of dissolved materials from runoff during rainfall.

The statistical test also revealed that both TSS and turbidity were significantly higher during the wet season, with respective p values of 0.007 and 0.003. This may be attributed to increased rainfall runoff from the land that carries different sorts of materials into the rivers. Another contributing factor may be increased erosion of bottom material due to higher discharges. The same trend was also observed by other researchers such as Anhwange *et al.* (2012), who studied the seasonal variation of water quality parameters of the Benue River, Makurdi Metropolis, Nigeria. The levels of DO were statistically significantly lower during the wet season (n = 15, M = 4.31) than during the dry season (n = 15, M = 4.85), (z = -2.160, p = 0.031). There was also a marked seasonal trend in terms of COD, where higher concentrations were measured during the wet season at all stations (n = 15, M = 724.96) than during the dry season (n = 15, M= 649.53), (z = -3.408, p = 0.001). The increase in COD during the wet season indicated higher contamination levels, probably from reception of contaminated runoff during rainfall.

The seasonal variation of alkalinity followed the same trend as those for TSS and turbidity, with which it was positively correlated. The levels were higher during the wet season (n = 15, M = 611) than during the dry season (n = 15, M = 461), (z = -2.557, p = 0.011), probably due to the same reasons. The variation in total hardness was not statistically significant between the seasons. The seasonal trend for Cl^- and salinity had no statistically significant difference between the seasons for both, with p = 0.776 for Cl^- and p = 0.427 for salinity.

A statistically significant difference was observed for the levels of SO_4^{2-}, which were higher during the wet season (n = 15, M = 273.4) than during the dry season (n =15, M = 254.4), (z = -3.408, p = 0.001). Similarly, concentrations of PO_4^{3-} were significantly higher during the wet season (n = 15, M = 1.31) than during the dry season (n = 15, M = 1.07), (z = -2.040, p = 0.041). The same trend was again observed in concentrations of NH_4^+, where the wet season concentration was higher (n = 15, M = 1431) than the dry season concentration (n = 15, M = 585), (z= - 3.408, p = 0.001). The increase in concentration of the three parameters during the wet season was probably due to increased inputs of contaminated run-off from human activities during rainfall. This was also observed by researchers from other locations (e.g., Anhwange *et al.*, 2012; Tshibanda *et al.*, 2014).

Table 5. Spearman Rank correlation matrix between pairs of metals (n = 30) in the rivers.

	Al	Cr	Cu	Fe	Pb	Zn
Al	1.000					
Cr	0.179	1.000				
Cu	0.171	0.244	1.000			
Fe	0.512**	-0.091	-0.280	1.000		
Pb	0.240	0.530**	-0.127	0.163	1.000	
Zn	0.131	0.107	-0.182	0.364*	0.130	1.000

*. Correlation is significant at the 0.05 level (2-tailed).

**. Correlation is significant at the 0.01 level (2-tailed).

Table 6. Principal components matrix.

	Component				
	1	**2**	**3**	**4**	**5**
pH	-0.022	0.273	**0.859**	0.175	-0.002
TDS	**0.948**	-0.224	-0.044	0.096	-0.029
EC	**0.949**	-0.231	-0.055	0.063	-0.055
TSS	-0.243	**0.752**	0.330	0.292	0.134
Turbidity	-0.288	**0.774**	0.292	0.282	0.097
DO	0.189	**-0.793**	-0.366	0.183	-0.266
COD	**0.881**	0.017	-0.038	-0.102	-0.123
Alkalinity	-0.297	0.415	**0.730**	-0.079	0.056
Total Hardness	**0.807**	-0.245	-0.143	-0.062	-0.109
Chloride	**0.955**	-0.213	-0.076	0.086	-0.026
Salinity	**0.958**	-0.206	-0.067	0.091	-0.018
Sulphate	**0.904**	0.162	-0.146	0.074	0.012
Phosphate	**0.679**	**0.603**	-0.109	0.032	0.031
Ammonium	-0.072	**0.844**	0.004	-0.073	0.190
Al	-0.111	**0.674**	0.391	0.184	-0.112
Cr	-0.317	0.002	0.446	-0.204	**0.588**
Cu	-0.079	0.187	0.371	-0.697	-0.178
Fe	0.031	0.272	0.212	**0.818**	-0.205
Pb	-0.078	0.406	-0.064	0.160	**0.788**
Zn	0.126	0.194	0.324	**0.502**	0.322
Eigenvalue	8.08	4.37	1.61	1.30	1.06
% of Variance	40.39	21.85	8.06	6.60	5.32
Cumulative %	40.39	62.24	70.30	76.90	82.22

Extraction Method: Principal Component Analysis.
Rotation Method: Varimax with Kaiser Normalization.
Bold values = high loadings (> 0.50)

The data on concentrations of metals for the two seasons were further subjected to the Wilcoxon signed-rank test. The observed seasonal trends were such that, with the exception of Cd that was below the detection limit, and Cu which did not show any statistically significant difference between seasons, all other metal concentrations were significantly higher during the wet season than the dry season: i.e., Al ($p = 0.005$, $z = -2.840$); Cr ($p = 0.049$, $z = -1.676$); Fe ($p = 027$, $z = -2.215$, Pb ($p = 0.017$, $z = -2.385$) and Zn ($p = 0.001$, $z = -3.408$). These results showed that the metal concentrations were influenced by the weather conditions. Some possible factors that may have led to this variation include the reception of contaminated runoff from land-use practices such as agriculture and domestic and industrial activities that were conducted close to the rivers. This was also observed by researchers in locations with similar activities, e.g., Edokpayi *et al.*, (2016) observed increased concentrations of metals during the wet season compared to the dry season in the Mvudi River, South Africa.

Factor reduction and Cluster Analyses

The results of factor reduction by PCA extracted five principal components with eigenvalues ranging from 8.08 to 1.06 that accounted for 82.22 % of the total variance (Table 6).

The loadings of data in the principal components matrix showed that Component 1, contributing 40.39 % of the total variance, includes high loadings from TDS, EC, COD, total hardness, chloride, salinity, sulphate, and phosphate. It showed that these eight parameters played a major role in characterizing water quality in the rivers. The high levels of these parameters suggest that this component was associated with chemical contamination of the river waters, most likely from anthropogenic sources. The high contributions of TSS, turbidity, decreased DO and NH_4^+ in

Component 2 (21.85 %) may be associated with inputs from decaying particulate materials. Component 3 (8.06 %) had loadings from pH and alkalinity, which might be associated with water chemistry such as the types of dissolved ions present in the waters. Component 4 (6.60 %) was influenced by only Fe and Zn, which showed that these metals were probably from the natural geochemical composition of the catchment, since their levels did not constitute pollution, even though they were weakly correlated. Component 5, which contributed 5.32 % of the total variance, includes high loadings from the concentrations of Cr and Pb, which suggested that the two metals were probably from the same source. Since Pb was found to be anthropogenically enriched at some locations, this was probably associated with contamination from human activities.

The results of the classification of the sampling stations into homogeneous groups using HCA based on the measured parameters are summarized in a Dendrogram in Fig. 4.

The dendrogram showed an initial splitting of the tree to form two clusters; the top cluster (Cluster 1) that contained twelve sites, and the bottom cluster (Cluster 2) that contained only three sites (Kz_4, Kz_5 and Mz_5). As previously observed, the three stations in the bottom cluster were characterized by the highest levels of TDS, EC, chloride, and salinity (Figs. 3a and 3b). The initial clustering is therefore based on the four parameters. The bottom cluster is further split to group the two Kizinga River stations (Kz_4 and Kz_5) together, leaving Mz_5.

Cluster 1 is further split into two groups, the bottom one with two stations and the top one with 10 stations consisting of three Kizinga River stations (Kz_1, Kz_2, Kz_3), four Msimbazi River stations (Ms_4, Ms_3, Ms_1, Ms_2), and three Mbezi River stations (Mz_1, Mz_2 and Mz_2). The ten stations had somewhat similar trends in levels of the measured parameters. For example, the four Msimbazi River stations in this cluster had relatively high concentrations of NH_4^+, Al, Cr, Cu, Fe and Zn.

Conclusions

The characterization of water quality in the Kizinga, Msimbazi and Mbezi Rivers along the Indian Ocean coast in Dar es Salaam, Tanzania, revealed decreased water quality in the rivers. High levels of pH, TDS, EC, turbidity, chloride, sulphate and ammonium that do not comply with the TBS and the WHO limits for potable water were detected. Furthermore, other parameters that have no specific regulatory limits, such as TSS, DO, COD, alkalinity, total hardness, salinity, and phosphate, had levels that exceed those expected for uncontaminated river waters. Concentrations of five metals (Cr, Cu, Fe, Zn and Cd) out of the seven that

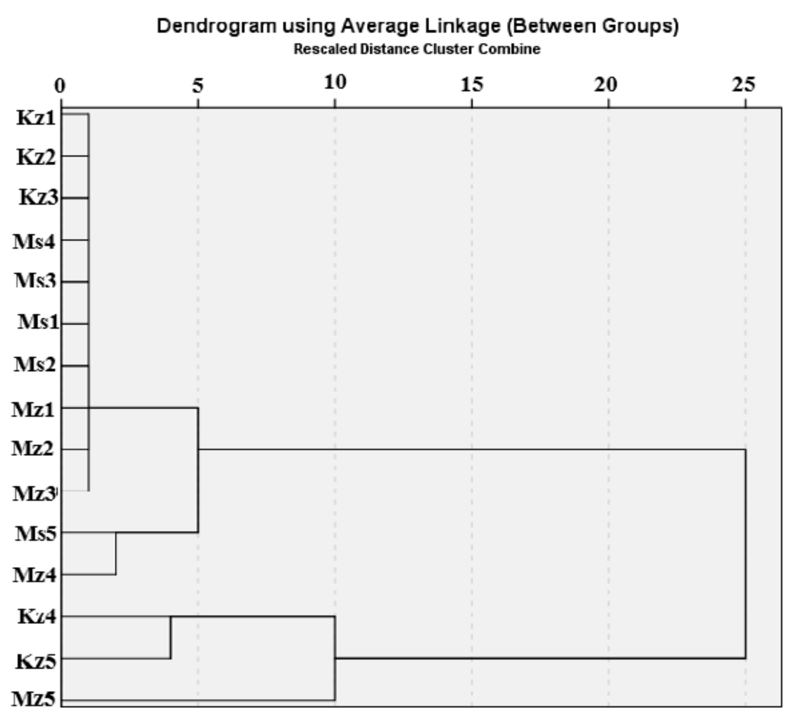

Figure 4. Dendrogram of the cluster analysis of the 15 stations.

were investigated, were found to be below the TBS and the WHO limits. Only Al and Pb were found to exceed the maximum recommended limits at some locations. The seasonal variation of the measured parameters showed that water quality was more impacted during the wet season than the dry season, with increased pH, EC, TDS, TSS, turbidity, COD, and decreased DO levels. Concentrations of SO_4^{2-}, PO_4^{3-}, NH_4^+ and the six metals (Al, Cr, Cu, Fe, Pb, Zn) were all higher during the wet season than the dry season. Natural influences were also observed in terms of increased Cl^- concentrations and salinity levels in locations closer to the ocean, due to seawater intrusion. Of the three rivers, the Msimbazi River was found to have the highest levels of Al, Cr, Cu, Pb and Zn. This is probably due to more exposure of its catchment to human activities compared to the other rivers.

Factor reduction by PCA revealed the main factors that control the observed variability to probably include chemical inputs from anthropogenic sources, decaying particulate materials, natural water chemistry in terms of types of dissolved ions present in the waters, and the geochemical composition of the catchment. These results showed that the quality of water in the three coastal rivers is not suitable for drinking and/or cooking. They also showed that the ecology of the rivers is probably impacted by human activities. These findings demonstrate that regular monitoring of the general environment and the associated land-use practices is important in this area. Strategic measures such as provision of environmental education at all levels and stipulation of clear policy statements on environmental management should therefore be adopted to ensure sustainability of these water resources.

Acknowledgements

This research was funded by the Flemish Inter University Council-University Development Cooperation (VLIR-UOS) Brussels, Belgium, under the South Initiative project No. ZEIN2014Z163 titled 'Tracing the status and impacts of terrestrial contamination in the coastal environments of Tanzania' (VLIR296). We are grateful for the support provided by the Chemistry Department, University of Dar es Salaam, Tanzania, for sample analyses.

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Perceived benefits and barriers to community participation in development projects – The case of *Hazina ya Maendeleo ya Pwani* on the Kenya coast

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Abstract
Benefits and barriers to participating in community development projects as perceived by participants were studied in coastal counties of Kenya through a World Bank-funded initiative known as *Hazina ya Maendeleo ya Pwani* (HMP). Primary data were collected from 326 randomly selected HMP beneficiaries using questionnaires. Data analysis using SPSS prioritized perceived benefits of participation as: acquisition of additional financial support (94 %); development of new skill (90.8 %); enhanced ability to meet own individual needs (90.8 %); development of valuable professional relationships (90.8 %); acquisition of useful knowledge (84.2 %); increased utilization of own expertise (77.9 %); heightened public profile (77.2 %); ability to contribute to community (71.9 %); ability to have greater impact (68.8 %); and enhanced ability to effect public policy (49.1 %). Perceived barriers were prioritized as: feeling unwelcome (89.4 %); lack of information or not knowing (87.9 %); feeling unable to make a difference (87.9 %); demanding work schedule at home or office (69.7 %); inadequate transportation (50.8 %); and concern for safety (43.2 %). The study concluded that while the perceived benefits still strengthen the argument for active involvement of communities, development practitioners need to incorporate "*what's in it for me?*" as an incentive for participation in future. They should also develop context-specific strategies to overcome participation barriers.

Keywords: Community participation, perceived benefits and barriers, community based organizations, *Hazina ya Maendeleo ya Pwani*, Kenyan coast

Introduction
Community participation in development initiatives is regarded as an important strategy that promotes project ownership and sustainable development. This participation has mostly been through common interest groups, such as Community Based Organizations (CBOs), which are voluntary membership groups consisting of individuals structured into self-defined communities. CBOs are seen as non-profit, non-governmental institutions where membership is placed on an equal level and whose main goal is to improve the socio-economic wellbeing of every member (Abegunde, 2004). CBOs serve as a link between citizens and the government and are often thought to be more responsive to community concerns than government agencies or private businesses (Mwaura and Ngugi, 2014). Due to their 'local' nature, CBOs have a better understanding of the needs, priorities and capabilities of the community in which they operate and by communicating these needs and capabilities to the policy makers, they give their communities a voice. CBOs are now increasingly asserting their importance as alternative economic vehicles that will spur development in Africa as they contribute 24 % to

http://dx.doi.org/10.4314/wiojms.v21i1.3

the gross national income of Africa's economy (Bhoke and Mwita, 2016). In Kenya, the numbers of registered CBOs have significantly increased and have now become the key target group for implementing development projects at the grass roots level (Mwaura and Ngugi, 2014). To a certain extent, local communities in Kenya rely on CBOs for the delivery of essential services such as education, health, water, among others.

Benefits of community participation have widely been described in the literature. Reed (2008) for example, highlighted benefits such as improved decision making, increased support and reduced costs as pragmatic benefits of community participation. In addition to these, increased representation, empowerment of marginalized groups, increased trust and promotion of social learning could be achieved. On the contrary, community participation is also associated with some disadvantages including its potential to cause conflict due to the increasing range of perspectives from different participants during decision making. It is expensive and time-consuming because it needs to involve all stakeholders thus leading to trade-offs between the individual interests and motivation for collective action of the group (Olson, 2013). It can also be susceptible to elite capture where wealthier or more powerful individuals gain a disproportionately large share of benefits, increasing inequalities and marginalizing weaker stakeholders (Persha and Anderson, 2014). While there are numerous and well-documented cases illustrating the problematic nature of participatory development, particularly in Africa (e.g. Kilewo and Frumence, 2015; Oketch, 2016; Lekaota, 2016; Osman, 2018; Setokoe and Ramukumba, 2020), there is still general optimism and support for community participation in development. As such, development in the full sense of the word is not possible without appropriate community participation (Botes and van Rensburg, 2000).

Barriers to community participation include poor means of communication, obscure information regarding the roles and responsibilities of the different participants, limited capacity due to lack of training, and insufficient financial resources to support the implementation of their activities (Kilewo and Frumence, 2015). A plethora of factors such as the paternalistic posture of authorities, the prescriptive role of the state, embellishment of successes, selective participation, inter-group conflicts, gate-keeping by leaders, excessive pressures for immediate results and disinterest within the primary beneficiary community were identified as barriers to community participation (Botes and van Rensburg, 2000). To encourage community participation in development, it is important to understand the principal variables that influence the perception of the local community regarding the barriers and benefits of community participation. As such, measuring subjective views and perceptions of the benefits and barriers of community participation is important to understand why people choose to or not to participate in development initiatives. In the absence of knowing what people perceive as limits to their community involvement, leaders of not-for-profit organizations such as CBOs are left guessing about why people do not participate (Torgerson and Edwards, 2012). This underlines the need to understand community perceptions relating to the benefits and barriers of community participation.

Much of the literature on community participation has focused on clarifying the concept (Chambers, 2007; Hodgkinson, 2004; Stukas and Dunlap, 2002), assessing the good that it does for those who are involved (Liu and Bessar, 2003; McBride *et al.*, 2006), factors affecting participation (Dorsner, 2004), and criticisms and challenges of participatory approaches altogether (Hayward *et al.*, 2004; Cornwall, 2009). While these issues are important, it is also interesting to understand the community's perception regarding the benefits and barriers of community participation. Perceptions are important in measuring human well-being (Woodhouse *et al.*, 2015), understanding and influencing human behavior (Ajzen, 1991) and enlisting stakeholders' support (Gurney *et al.*, 2015) that are critical to community participation in development initiatives. As such, recognizing perceived benefits and barriers of community participation in development is important for the successful implementation and management of government and donor-funded projects. Given the continued popularity of community participation, it is important to understand why people choose to participate in development initiatives, and the perceived benefits and barriers. A good understanding of community's perception of the benefits and barriers to community participation helps community leaders to identify potential challenges and address them, and design appropriate strategies to encourage the members to participate. No study has been conducted to identify the perceived benefits and barriers to community participation in the context of natural resources management and provision of services within coastal Kenya. The purpose of this study was, therefore, to fill this void in research through the use of a case study of a World Bank-funded community grant facility referred to in Kiswahili as *Hazina ya Maendeleo ya Pwani* (HMP). HMP takes the approach of Community Driven Development (CDD), which emphasizes the handing over of the entire development process, planning decisions and investment resources directly to community groups and the local government (Wong, 2012). Supported projects in resource management included promotion, conservation of and sustainable use of fisheries, forestry and other coastal resources. The supported community services included promoting social wellbeing by constructing early childhood classrooms, and enhancing provision of essential services such as water.

Various approaches have been used to determine individuals' perceptions towards particular programmes or issues. For example, the perceived benefits and barriers to community participation could be determined by administering Likert type questions to the survey respondents (Mpokigwa *et al*., 2011; Shan, 2012). The individual Likert questions are then analyzed and percentages or frequencies of each item described and presented (Shan, 2012). In the present study, benefits and barriers to community participation were assessed using multi-item measures adapted from the Synergy Model of Weiss *et al.* (2002) and indices from Khodyakov *et al*. (2011). Benefit of community participation was measured in terms of 10 closed-ended items organized into three main categories comprising capacity building, political impact, and professional development as presented in Table 1. Respondents used a five-point scale (1 = strongly disagree and 5 = strongly agree) to assess their experience of 10 benefits resulting from participating in a community-based project. The barriers to community participation scale comprised six closed-ended items (Table1), which were also scored on a five-point Likert scale ranging from strongly disagree to strongly agree.

Materials and methods

Study area

Figure 1 is a map of Kenya (inset) showing the location of the coastal region and the study sites. The study was conducted in all the six coastal counties of Kenya, A total of 150 projects were implemented to completion in Kwale (30), Taita Taveta (32), Mombasa (27), Kilifi (24), Tana River (20) and Lamu (17). Dots therein indicate sites of HMP projects in each county.

The coast region covers an area of 83,603 km^2 constituting about 11.5 % of the total area of the Republic of Kenya with a coastline of approximately 600 km long. The region is inhabited by a culturally heterogeneous

Table 1. Measuring benefits and barriers of community participation.

Benefits of Community Participation	
a. Capacity Building	
1	Development of new skills
2	Acquisition of useful knowledge about services, programs or people in my community
3	Ability to have a greater impact than I could have on my own
4	Enhanced ability to meet own needs
b. Political Impact	
5	Acquisition of additional financial support
6	Enhanced ability to affect public policy
7	Heightened public profile
c. Professional development	
8	Development of valuable professional relationship
9	Ability to contribute to my community
10	Increased utilization of my expertise or services
Barriers to Community participation	
1	Demanding work schedule at home or office
2	Inadequate transportation
3	Feeling unwelcome
4	Concerns for your safety
5	Lack of information or not knowing how to begin
6	Feeling that you can't make a difference

Source: Weiss *et al.*, 2002; Khodyakov *et al.*, 2011.

population with the Mijikenda being the largest ethnic group. Human density along the Kenya coast, with Mombasa leading with 5,495 per km², is higher than in many other parts of the country (Government of Kenya, 2009; KNBS, 2019). More than 57 % of coastal residents are classified as 'very poor', living on less than the international poverty line of 1.9 US$/day (Ferreira *et al.*, 2015). As documented by KNBS (2019), based on the national poverty line of Kenya Shillings 1,562 (rural) and 2,913 (urban), severity is high in the coast region especially in the counties of Tana River (44 %) and Kwale (49 %). Regardless, the region is endowed with a variety of resources that support livelihoods and economic development locally and nationally, and form the basis of maintaining the health and function of marine and coastal ecosystems (Ongoma and Onyango, 2014). These resources include mangroves and coastal forests, seagrass meadows, corals, fish and various fauna and flora in the near shore and the open marine environment. The region is also characterized by significant disparity in literacy between men and women; with literacy among the later being significantly low in the counties of Kilifi (68 %), Tana River (34 %) and Kwale (57 %) (Government of Kenya, 2008; Hoorweg *et al.*, 2000).

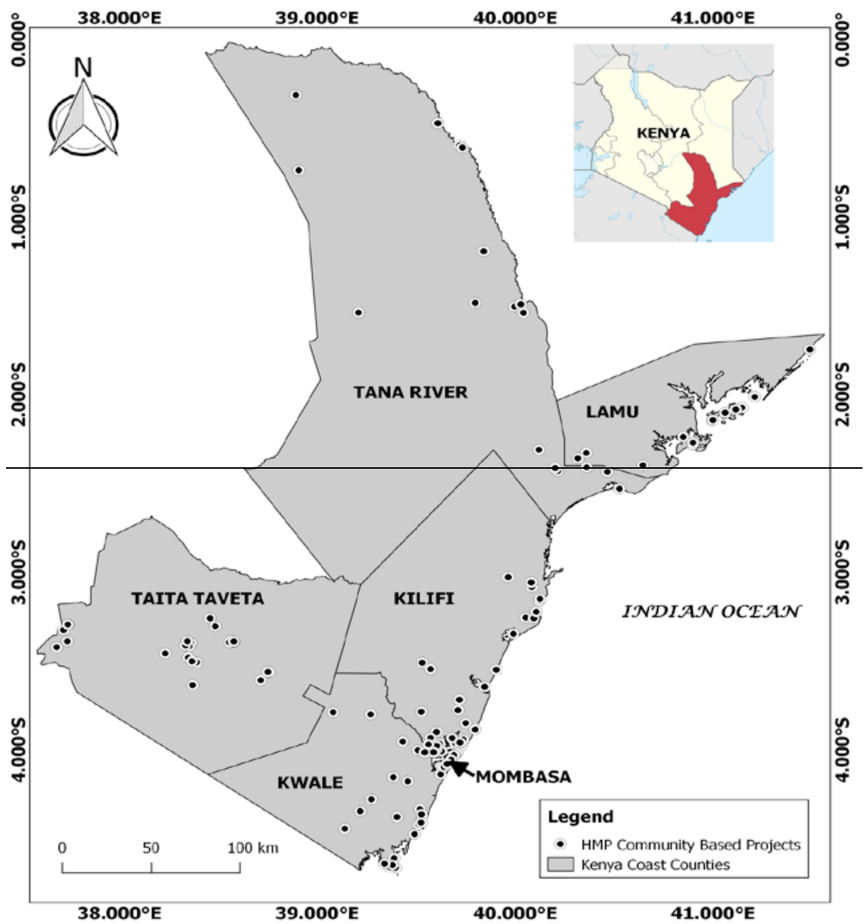

Figure 1. Map of Kenya showing coastal counties and location of HMP projects. (Source: Modified from Hassan et al., 2020)

Study population

The target study population comprised the communities living in the coastal region, at that time estimated to be 3.3 million people (KNBS, 2010), and currently 4.3 million (KNBS, 2019). The accessible population was the 2,160 community members drawn from the CBOs that participated in the implementation of HMP and were also the beneficiaries of the same.

Sample size and sampling procedure

A sample size of 326 persons was computed using Ross *et al.* (2002) as illustrated in the Equation below. The proportionate sampling technique was used to get a fair representation of the study sample from each of the six coastal counties. Simple random sampling techniques were used to obtain the study respondents using a sampling frame obtained from HMP records in the HMP Manual (Aura *et al.*, 2015).

Equation: Computation of study sample

$$n = \frac{NZ^2 \times 0.25}{(d^2 \times (N-1) + (Z^2 \times 0.25)}$$

Where n = sample size required
N = total population size (known or estimated)
d = precision level (usually 0.05 or 0.10)
Z = number of selected standard deviation units of the sampling distribution corresponding to the desired confidence level

Therefore, the following formula was used to compute the study sample

$$n = \frac{2{,}160 \times 1.96^2 \times 0.25}{(0.05^2 \times (2{,}160-1) + (1.96^2 \times 0.25)}$$

Resulting in n = 326

Data collection

Semi-structured questionnaires with two sections were used to collect primary data. Section I requested demographic information of the participants while Section II was used to identify the benefits and barriers of community participation. Desktop review of previously published and unpublished research, including internet material, was used to obtain secondary data about the study topics. Out of the 326 questionnaires distributed to randomly selected respondents in a face-to-face interview, 285 were correctly filled and returned resulting in a response rate of 87.4 %. Mugenda and Mugenda (1999) stipulated that a response rate of 50 % is adequate for analysis and reporting, a response rate of 60 % is good, and a response rate of 70 % is very good. Therefore, the 87.4 % response rate reported for this study formed an acceptable basis for analysis, reporting and drawing conclusions from the data obtained.

Data analysis

SPSS - Version 21 was used to analyze the data while descriptive statistics were used to report on the demographic characteristics of the sample.

Results and discussion

Socio-demographic and cognitive variables

Table 2 shows the distribution of some key socio-demographic variables of the respondents. Results show that more than half of the respondents were female (n = 179; 62.9 %) while 37.1 % were male. This infers that the range of respondents was dominated by females. This information is relevant because women play a vital role in community development projects. About half of the respondents were in the age category of 20 -30 years, while 35.6 % were 31-50 years. Regarding the level of education, just over half (n = 158; 55.3 %) of the respondents had primary education. These results are in agreement with Ochiewo (2004) who stated that the level of school dropouts at secondary school level in the coast region is very high. Gender, age, level of education, household size and occupation are among the factors that influence community perceptions of benefits and barriers.

The majority (n = 194; 68.2 %) of the families had a household size of 1-5 people while 30.3 % (n = 86) had 6-10 people. Concerning occupation, the majority of the respondents were either traders (n = 138; 48.5 %) or in both formal and informal employment (n = 114; 40.2 %). Very few respondents practiced farming or fishing.

Perceived benefits of community participation among coastal communities

Benefits of participation among members in community initiatives were evaluated using three parameters namely: a) capacity building; b) political impact; and c) professional development, and further broken down into various sub-categories as presented in Tables 3, 4 and 5.

Table 3 shows evaluation of various sub-categories statements measuring capacity building as a perceived benefit of community participation.

Capacity building as a benefit of community participation

This study evaluated the extent to which capacity building was perceived as a benefit of community participation in development initiatives. Table 3 shows the outcome of the evaluation. Capacity building was measured in terms of a set of parameters comprising 4 statements: development of new skills; acquisition of useful knowledge; ability to have a greater impact in the society; and enhanced ability to meet own needs. The first and fourth statements elicited positive reviews with 90.8 % (n = 259) agreeing that community participation encouraged the **development of new skill** and **enhanced ability to meet own needs** amongst community members. This finding corroborates that of Blackstock *et al.* (2007) that participation may also promote social learning. This is where

Table 2. Distribution of socio-demographic variables (N = 285).

Variables	Frequency (N)	Percentage (%)
Gender		
Male	106	37.1
Female	179	62.9
Age		
Below 20	39	13.6
20-30	136	47.7
31-50	101	35.6
Over 50	9	3
Level of Education		
Primary school	158	55.3
High school	110	38.6
College	13	4.5
University	4	1.5
Household size		
1-5	194	68.2
6-10	86	30.3
10-15	4	1.5
Occupation		
Farming	4	1.5
Fishing	2	0.8
Trading	138	48.5
Employment	114	40.2
Other	26	9.1

stakeholders and the wider society in which they live learn from one another. They learn through developing new relationships, building on existing relationships and transforming adversarial relationships as individuals learn about the other's trustworthiness and appreciate the legitimacy of the other's views (Leeuwis and Pyburn, 2002; Stringer *et al.*, 2006).

A large number of respondents, (n = 240; 84.2 %) perceived **acquisition of useful knowledge** as a benefit of community participation. Similar results were reported by Robinson *et al.* (2010) where training and capacity building for communities was an important end goal in itself which can support community participation in development initiatives in the longer term. On the contrary, lack of knowledge about co-management associations in conservation of protected areas, and how to join them have also been identified as factors that limit participation (Ward *et al.*, 2018).

A total of 196 (68.8 %) respondents perceived **ability to have greater impact** on their community as the reason for taking part in a community development initiative. They perceived "greater impact" to mean positive outcomes at community level such as better life, improved access to social services etc. The results of the present study concur with the findings of Uche-Nwachi *et al.* (2018) in which the citizenry highly rated improvement of rural economy and provision of employment as the main reasons for participating in community development projects. Similarly, De Vente *et al.* (2016) reported that meaningful community participation in protected areas is more likely to deliver positive outcomes for livelihoods and biodiversity, although local context is also an important predictor of success.

This therefore means that communities do consider the anticipated outcome of the project prior to choosing to participate or not. The findings correlate with those of Uche-Nwachi *et al.* (2018) who reported that the high percentage of farmers who participated in community development projects is an indicator that the project is based on their felt need. The finding is in congruence with that of Khadka and Nepal (2010) that local communities participate and support activities that they feel will bring them clear tangible and preferable benefits in terms of products or income. Alternatively, greater community participation may be experienced as burdensome, and have negative consequences for the individuals and communities involved if they are not adequately supported or if their expectations are not met (Greene, 2007).

Table 3. Capacity building as a perceived benefit of community participation (N = 285).

Statements measuring a particular perception	Strongly Agree (n)	Agree (n)	Neutral (n)	Disagree (n)	Strongly Disagree (n)
Development of new skill	22 (7.7%)	259 (90.8%)	2 (0.7%)	2 (0.7%)	0 (0%)
Acquisition of useful knowledge	45 (15.8%)	240 (84.2%)	0 (0%)	0 (0%)	0 (0%)
Ability to have greater impact	84 (29.5%)	196 (68.8%)	2 (0.7%)	2 (0.7%)	0 (0%)
Enhanced ability to meet own needs	24 (8.4%)	259 (90.8%)	0(0%)	0 (0%)	2(0.7%)

Political impact as a benefit of community participation

This study also assessed political impact as a benefit of community participation. It was measured in terms of acquisition of additional financial support, enhanced ability to affect public policy, and heightened political profile. Table 4 shows responses obtained about perceived benefit from political impact.

The majority of respondents 268 (94 %) agreed that they are likely to participate in a community project if they sensed a likelihood of **acquiring additional financial support** such as through political connections. Respondents described financial support to be in form of increasing financing for their community project or for individual benefit through paid casual labour. This finding is similar to those reported by Ward *et al.* (2018) that participation led to community members obtaining direct benefits from the NGOs such as paid work (e.g., "being a porter and building the new campsite"), training (e.g., "we get training on techniques for farming and growing crops"), and materials (e.g., provision of seeds and farming tools). Given the high poverty level amongst coastal residents and competing basic needs requiring the limited financial resources, the potential to access external financial support seems to be a motivation for individuals to participate in community development initiatives. On the flipside however, drained participants' energy levels as well as time and financial resources are some unintended negative consequences of community participation (Attree *et al.*, 2011).

The notion that community participation **enhanced the ability to affect public policy** drew a different response from the respondents. About half (n = 140, 49.1 %) of the respondents agreed that the view was true while 93 (32.6 %) respondents disagreed. Meaning that respondents do not strongly disagree but generally agree that political impact enhances ability to effect public policy. This finding is congruent with those of Beierle (2002) who reported improved quality of decisions made through addition of new information, ideas and analysis in the majority of the cases where communities participated in environmental decision making. Similar results were reported by Reed (2008) who opined that community participation can enhance the quality of environmental decisions by considering more comprehensive outputs. A significant proportion of the respondents (n = 220, 77.2 %) were of the view that community participation led to a **heightened public profile**. Active participants in community initiatives, inadvertently create future political profiles for themselves.

Professional development as a benefit of community participation

The responses obtained about perceived benefits from professional development is shown in Table 5. Professional development as a benefit of community participation was categorized into three variables comprising: i) development of valuable professional relationship; ii) ability to contribute to my community; and iii) increased utilization of own expertise.

From the study, the majority (90.8 %; n = 259) of the respondents considered **development of valuable professional relationship** as a perceived benefit and incentive for them to participate in community development initiatives. The study findings are paralleled

Table 4. Political Impact as a perceived benefit of community participation (N = 285).

Statements measuring a particular perception	Strongly Agree (n)	Agree (n)	Neutral (n)	Disagree (n)	Strongly Disagree (n)
Acquisition of additional financial support	15 (5.3%)	268 (94%)	2 (0.7%)	0 (0%)	0 (0%)
Enhanced ability to affect public policy	4 (1.4%)	140 (49.1%)	48 (16.8%)	93 (32.6%)	0 (0%)
Heightened public profile	28 (9.8%)	220 (77.2%)	13 (4.6%)	24 (8.4%)	0 (0%)

Table 5. Professional development as a benefit of community participation (N= 285).

Statements measuring a particular perception	Strongly Agree (n)	Agree (n)	Neutral (n)	Disagree (n)	Strongly Disagree (n)
Development of valuable professional relationship	22 (7.7%)	259 (90.8%)	2 (0.7%)	2 (0.7%)	0 (0%)
Ability to contribute to my community	80 (28.1%)	205 (71.9%)	0 (0%)	0 (0%)	0 (0%)
Increased utilization of own expertise	15 (5.3)	222 (77.9%)	15 (5.3%)	32 (11.2%)	0 (0%)

with those of Stringer *et al.* (2006) that participatory processes have the capacity to transform adversarial relationships, learn about others' trustworthiness and appreciate legitimacy of their views (Leeuwis and Pyburn, 2002; Stringer *et al*., 2006). This finding corroborates that of Blackstock *et al.* (2007) that participation may also promote social learning.

Experimental evidence also suggests that community engagement may benefit a community more widely, in terms of increasing mutual trust and understanding between different population groups (Callard and Friedli, 2005). On the flipside however, Ward *et al.* (2018) argued that participation can expand the range of perspectives in decision-making thus increasing the potential for conflict, and by extension poor relationships amongst community members. Community participation could also be seen as a potentially divisive factor within communities. A number of older Chinese people engaged in service planning, for example, reported that they had experienced disapproval, criticism and even bullying from other community members, who assumed that their primary motive for involvement was financial (Chau, 2007).

The study also revealed that a significant proportion of the respondents (n = 205; 71.9 %) identified the **ability to contribute to their own community** as a perceived benefit of community participation. This was expressed in terms of communities allocating both time and financial resources to the community project, which they would ordinarily not do if they were on their own. The study findings resonate with those of Attree *et al*. (2011) who reported that the majority of individuals who were actively involved in initiatives utilizing community participatory approaches experienced positive benefits, in terms of their self-confidence, self-esteem, social relationships and individual empowerment. Similar results were reported by Taylor *et al*. (2012) that community participation is instrumental as a means to achieve cost-effective, relevant and accessible health services; a priority issue for rural communities in Australia. The present finding is also in congruence with Okafor (2005) who contended that when communities participate in their own project, there is normally greater transparency and accountability, which enhances service delivery. This is where stakeholders and the wider society, in which they live, learn from each other through the development of new relationships, building on existing relationships and transforming adversarial relationships as individuals.

A significant proportion of the respondents (n = 222; 77.9 %) agreed with the opinion that community participation **increased utilization of one's own expertise**. Retired or even practicing professionals who take time to lend a hand in community projects in their own regions develop their professions and skills further and so does the community they serve.

Barriers to community participation among coastal communities

Gantt bars in Fig. 2 show the responses to a questionnaire relating to perceived barriers to community participation evaluated using six parameters namely: i) concern for safety; ii) demanding work schedule at home or office; iii) feeling that you cannot make a difference; iv) feeling unwelcome; v) inadequate transportation; and vi) lack of information or not knowing. Tabulated responses for each answer from the six questions are attached in the Appendix.

The majority of the respondents (43.2 %; n = 123) disagreed with the opinion that **concern for their safety** was a reason for not participating in community projects. This could be because most of the projects were located within a close geographical area where people are familiar to each other and concerns for their safety is not an issue.

A **demanding work schedule** was of interest to a significant proportion (69.7 %; n = 199) of the respondents who agreed that it was a major barrier among communities and in particular among females. This is especially so because the majority of the respondents were female who also assume other family

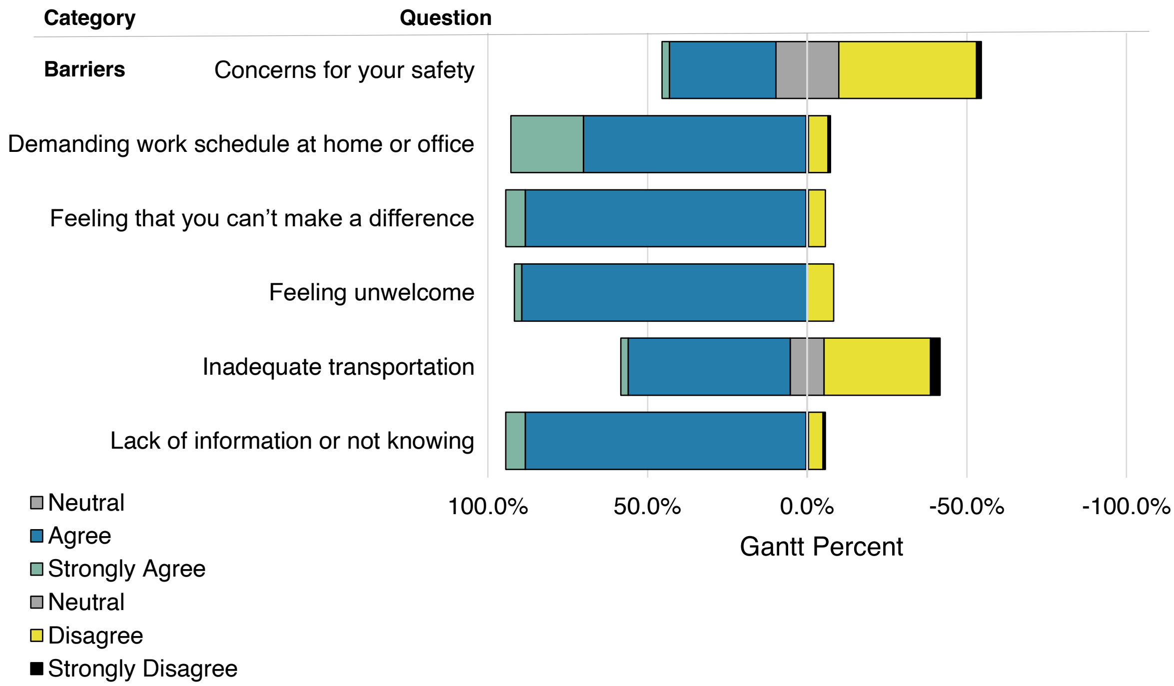

Figure 2. A Gantt bar chart visualizing the Likert scale survey data results of a questionnaire regarding barriers to community participation in development projects. Actual data showing categories of barriers, questions and answers as responded is shown in the Appendix.

responsibilities thus potentially reducing their overall time available for participation in community development initiatives. This is particularly true for single parents and families who work long hours or multiple jobs and therefore prioritize their limited free time to be with their children (McBride *et al.*, 2006). Torgerson and Edwards (2012) reported similar results that time-consuming family obligations disproportionately shouldered by women potentially constrain their participation. As such, the effect of demanding work schedules on a community's willingness to participate in development projects should not be underestimated, because it may work against such projects. Windle *et al.* (2009) stated that besides a demanding work schedule, the timing of community participation events and a lack of support to help particular groups to attend were barriers to community engagement, as different timings suit different groups of people. For instance, conducting community activities over the weekend would be suitable for the working adults and those with school going children while the holiday period would be more suitable for the youth. On the contrary, health managers sometimes rationalize non-participation because they perceive that people in small communities lack sufficient time, might become fatigued and eventually drop out (Brunger and Wall, 2016). The study findings are contrary to those of Mattingly and Bianchi (2003) who reported that in spite of the absence of work commitments, the unemployed individuals may find themselves struggling to maintain their livelihoods and having little free time to participate in community work. The results also contradict those of Wilson (2000) who opined that the employed are linked to greater social networks and may therefore be exposed to a wider range of possible volunteer activities to participate in.

The respondents gave their views on their perception of **"feeling that you cannot make a difference"** as a barrier to community participation. A significant majority (87.9 %; n = 250) of the respondents felt that this was true (agree). This could be caused by the technical nature of some of the community projects where most of the respondents felt that they cannot effectively contribute to the development and execution of community projects due to their low education (primary) level. This is supported by the socio-demographic findings (Table 2) that shows more than half (55 %; 158) of the respondents attained only primary school as their highest level of education. Similar results were reported by Stephen (2005) who argued that community members with less education may not feel that they can effectively contribute to processes that require extensive technical knowledge or civic skills. The study finding could also be because those in leadership positions do not take views of their less educated members leading to a seriously sense of cynicism towards the overall value of community participation. The feeling that one cannot make a difference could also be due to lack of capacity resulting from factors such as limited understanding and language skills, low confidence and self-esteem (White and Woodward, 2013).

Feeling unwelcome when participating in community projects was also a concern for a majority (89.4 %; n = 255) of the respondents. In their view, the respondents reported that the CBO leadership makes most of the decisions leaving the rest of the members as silent observers. This feeling mostly occurs in situations where there is poor relationship amongst community members or even between community members and the staff charged with the responsibility of overseeing project implementation. Some of the respondents cited cases of discrimination and exclusion by development practitioners and CBO leadership, which altogether contributed to their perception of not feeling welcomed, hence making their participation rather difficult. This finding correlates with those of Carlisle (2010) who reported that a history of poor relations between communities and engaging agencies could make it difficult to get community members to participate in development initiatives and to keep communities on-board. Similar results were reported by Robinson *et al.* (2010) that specific groups in the society described as stigmatized, isolated, marginalized or vulnerable comprising young people, older people, and ethnic minority groups, end up feeling unwelcome and as such finding it difficult to participate in development initiatives. This feeling of being unwelcome needs to be addressed because it could lead to local participants feeling disappointed and thus affecting their overall contribution to community work.

Inadequate transportation was identified as a perceived barrier to community participation by 50.8 % (n = 145) of the respondents. Access to transport services was considered difficult in areas where public transportation services are not reliable and with households dispersed and not easily accessible. However, 33.3 % (n= 95) of the respondents felt that transportation was not a barrier to community participation, probably because their households are not that dispersed and the community initiatives are located near to their homes. This finding contradicts Hartell (2008) who provided evidence from a social capital benchmark survey indicating that inadequate transportation is a barrier to community involvement.

The last aspect to be tested related to the barriers to community participation was the **lack of information or not knowing** what the project is all about. Most of the respondents (87.9 %; n = 250) agreed with the hypothesis that a major barrier to community participation in projects is the lack of information about it. Regardless of their enthusiastic participation in HMP projects, some community members require more time to grasp the details of the concepts, possibly due to limited command of English. Lack of information may also emerge from non-clarity of what the project is all about and the specific role of participating individuals. This finding resonates with those of Torgerson and Edwards (2012) that lack of information prevents community members from being as involved in development initiatives as they would wish. Similar findings were reported by Robinson *et al.* (2010), who stated that low levels of awareness and lack of understanding bar effective community participation. This is particularly true for those with limited education who may not feel that they can effectively contribute to processes that require extensive technical knowledge or civic skills (Stephan, 2005). For this reason, communicating the goals and expected outcomes of the development project clearly from the outset, and being transparent about the process aids effective community participation (Hatamian *et al.*, 2012). As such, providing a clear explanation of the purpose of community participation ensures 'buy-in' from participants, especially those who could not initially see why they were being involved.

Conclusions

This research gives insight on the human dimension of coastal communities in the localities where HMP was implemented, and contributes to capacity development and outreach. The results prioritized perceived benefits of community participation in HMP as: (i) acquisition of additional financial support; (ii) development of new skill; (iii) enhanced ability to meet own individual needs; (iv) enhanced ability to address important community issues; (v) development of valuable relationships; (vi) increased utilization of own expertise; (vii) heightened public profile; (viii) ability to contribute to community; (ix) ability to have greater impact; and (x) ability to influence public policy. The results suggest that whereas the welfare of the general community seems to be the main driver of community participation, personal benefits appear to be an important factor considered by individuals when deciding to participate. The implication of this finding is that development practitioners of projects similar to HMP may need to consider and incorporate "*What's in it for me*" as an incentive for community members to participate.

Perceived barriers to community participation in HMP projects were prioritized as: (i) feeling unwelcome; (ii) lack of information or not knowing; (iii) feeling unable to make a difference; (iv) demanding work schedule at home or office; (v) inadequate transportation; and vi) concern for one's safety. The implication of these findings is the importance of informing members fully about the project goals and benefits, preferably during pre-implementation training, their specific role, and agreeing on a convenient time for community members to participate in order to overcome potential barriers.

In conclusion, it is important to point out that while the study findings might not be transferable to other settings partly due to limited analyses; successful community participation in development work similar to HMP may be contingent upon enhancement of the perceived benefits to incentivize community participation. In the same spirit, development practitioners may need to take cognizance of the perceived barriers with a view to develop context specific strategies that overcome the perceived barriers to community participation. On this basis, the perceived benefit still strengthens the argument for active involvement of communities in development initiatives.

Acknowledgements

We acknowledge the County Liaison Officers of HMP, vehicle drivers, interns and community members from the six coastal counties of Kenya who participated in the study.

The HMP implementation strategy was developed at KMFRI and funded by the Kenya Government and the World Bank (Project ID P094692). We owe much gratitude to past and current Directors of the six KCDP partner agencies, and the World Bank team leaders, Anne Jeanette Glauber, Dinesh Aryal and Veruschka Schmidt. The outcome of this inspirational study is intended to contribute towards improvement of community performance of CBOs especially in coastal Kenya; notable among them is the newly established Mudzi Enterprise.

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APPENDIX

Response to the six questions regarding perceived barriers to community participation in projects.

Question	Answer	Respondents	Percent (%) Respondents
Concerns for your safety	Strongly Agree	6	2.3
	Agree	95	33.3
	Neutral	56	19.7
	Disagree	123	43.2
	Strongly Disagree	4	1.5
Demanding work schedule at home or office	Strongly Agree	65	22.7
	Agree	199	69.7
	Neutral	2	0.8
	Disagree	17	6.1
	Strongly Disagree	2	0.8
Feeling that you can't make a difference	Strongly Agree	17	6.1
	Agree	250	87.9
	Neutral	2	0.8
	Disagree	15	5.3
	Strongly Disagree	0	0
Feeling unwelcome	Strongly Agree	6	2.3
	Agree	255	89.4
	Neutral	0	0
	Disagree	24	8.3
	Strongly Disagree	0	0
Inadequate transportation	Strongly Agree	6	2.3
	Agree	145	50.8
	Neutral	30	10.6
	Disagree	95	33.3
	Strongly Disagree	9	3.0
Lack of information or not knowing	Strongly Agree	17	6.1
	Agree	250	87.9
	Neutral	2	0.8
	Disagree	13	4.5
	Strongly Disagree	2	0.8

Assessment of tropical cyclone-induced shoreline and riverbank changes at the Rufiji Delta using satellite remote sensing methods

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Abstract

The study aimed at quantifying changes in shoreline and the riverbanks caused by tropical cyclones (TCs) and associated rainfall in the Rufiji Delta, southern Tanzania. Long term changes of the shoreline and riverbanks were analysed using medium resolution (Landsat TM and ETM) satellite imagery (1991, 1997 and 2007), while short-term changes (2013 to 2014) were analysed using high resolution (Pleiades) satellite imagery. Delineation of the shoreline and riverbank changes were accomplished through the analysis of appropriate coloured image composites, Sobel filtering and maximum likelihood classification of land cover. Analysis of Landsat data showed a relatively higher magnitude of erosion between 1991 and 2007, followed by minor changes between 1997 and 2007. Simbauranga was the most severely eroding site, with an estimated magnitude of erosion of 83 to 100 m during the study period. The maximum magnitude of short-term changes of the riverbanks were estimated at about 31m^2. Apart from the erosion of the riverbanks, other changes were the conversion of water to vegetation covered areas (amounting to approximately 200 m^2). Short-term shoreline changes were up to 206 m with higher magnitude of accretion (142 m) than erosion (-4 m). The study conclusively calls for detailed research on shoreline and riverbank changes based on the impacts of TCs on land cover.

Keywords: shoreline and riverbanks, erosion and accretion, maximum likelihood classification, band combinations, Sobel filtering

Introduction

Tropical cyclones (TCs) are low pressure systems that form over warm tropical waters with gale force winds of at least 17.5 ms^{-1} and gusts exceeding 25 ms^{-1} near the centre (Holland, 1993). They are non-frontal low-pressure systems of synoptic scale having organized convection and a life span of at least six hours (Bengtsson, 2007). Actual wind speeds recorded from South Western Indian Ocean (SWIO) TCs include 70 ms^{-1} for Bondo (December, 2006) and 69 ms^{-1} for Hurry (March, 2012).

TCs have adverse impacts on society and their socio-economic well beings. Among the most common adverse impacts of TCs include the destruction of settlements and assets, public infrastructure (such as roads, telecommunication lines, schools, etc.) and crops (Charrua *et al.*, 2021). Such impacts are often caused by strong winds which are accompanied by heavy rains during the TCs and storm events. The extent of damage is often influenced by the coastal geomorphology, the nearshore ocean topography, the TC/storm track and the vertical changes in atmospheric pressure (Camargo *et al.* 2013; Yanxia *et al.*, 2013). Moreover, the associated coastal flooding (inundation) and the huge waves or swells which accompanied the TCs are considered to be the main cause of damage to coastal human settlements and public infrastructure (Chang-Seng and Jury, 2010).

http://dx.doi.org/10.4314/wiojms.v21i1.4

Apart from the socio-economic impacts, TCs are also associated with considerable environmental impacts which include coastal erosion and redistribution of nearshore and offshore sediments (Nyandwi, 2001; Lacombe and Cater, 2004; Cooper *et al.*, 2008), which may in turn adversely affect nearshore habitats such as coral reefs.

The problem of shoreline changes, particularly coastal erosion, is considered to be one of the major environmental management issues in Tanzania and most parts of the Western Indian Ocean Region (Shaghude *et al.*, 2007; Shaghude *et al.*, 2015). Although a wide range of studies have been undertaken to establish the magnitude of erosion at different locations, and identifying the main causative factors and impacts along the coast (Shaghude et *al.*, 2003; Muzuka, 2001; Muzuka and Shaghude, 2000; Shaghude, 2004; Shaghude *et al.*, 2015), no study has yet quantified the magnitude of shoreline or riverbank erosion associated with TCs along the Tanzanian coast. Thus, the main objective of this study was to describe and assess the impact of severe weather events such as TCs and the associated heavy rainfall on shoreline and riverbank changes in the Rufiji River using satellite remote sensing data (1991, 1997 and 2007). Specifically, the focus of the study was to examine and to quantify the long- and short-term shoreline and riverbank changes associated with TC events.

Materials and methods

Study area

The Rufiji Delta which is located along the southern coast of Tanzania has the largest mangrove forest cover in East Africa (Semesi, 1992). The mangrove forests fall within the protected forest reserves of Tanzania. One of the unique features of the mangrove forest reserve of the Rufiji Delta is that there are legally established village settlements (with an estimate of about 49,000 people (Mangora et al 2018) exists within it. These people depend on mangroves and the associated marine environment for a range of resources and ecosystem services to support their livelihoods. According to Monga et *al.* (2018) the local communities who live in and around the Delta are directly engaged in various socio-economic activities such as rice farming, cutting of mangrove poles and timber, and fisheries.

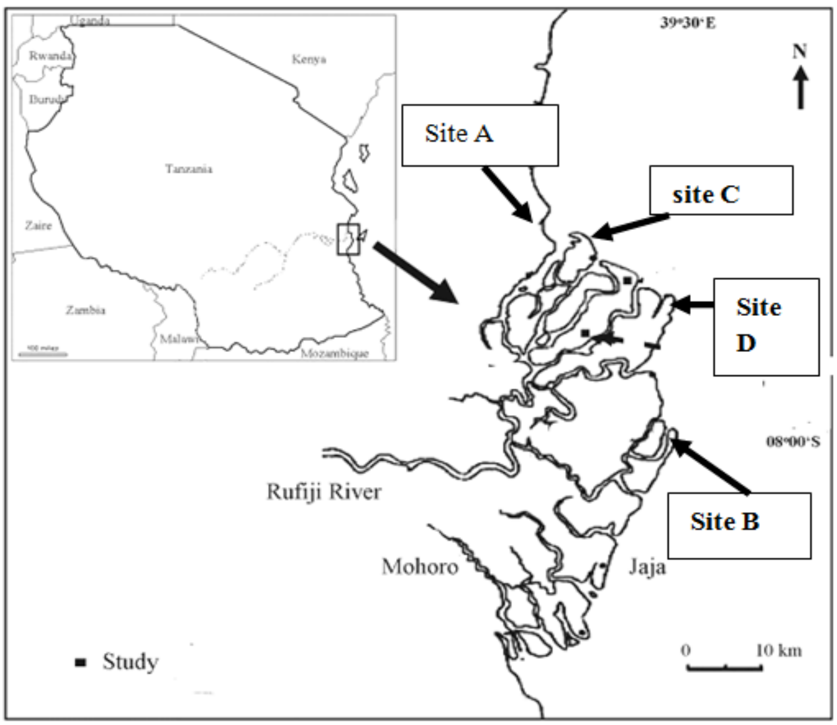

Figure 1. Study area location in the Rufiji Delta indicating of the four study sites (A to D). Modified after Punwong (2013).

The Rufiji Delta is approximately 65 km wide and extends for about 123 km inland (Semesi, 1992; Erftemeijert and Hamerlynck, 2005; Punwong, 2013). Within the Delta system, the Rufiji River branches into a series of distributary channels, including the Bomba, Kikale, Kyomboni, Mchingamfisini amongst others. The Delta is broadly divided into two morphological units, the southern and northern Deltas, which extend for about 23 km along a north-south direction and with an east-western extent of about 70 km (Doody and Hamerlynck, 2003). The Delta mangroves substrata forms the dominant component of the Delta substrates, while sand beaches and intertidal sand flats occur as subordinate substrates on the Delta. The mangrove substrata occur at approximately 2.5 m above mean sea level (Fisher *et al.*, 1994; Fisher and Overton, 1994).

The tides on the Delta are semi-diurnal with a tidal range varying from 2-2.5 m to about 3.3 - 4.3 m during high spring tides (Fisher *et al.*, 1994; Richmond *et al.*, 2002: Francis, 1992). The water and soil salinity along the Delta ranges from 10.60 ‰ to 32 ‰ (Francis, 1992; Fisher *et al.*, 1994), while observations show that the water in pools in non-mangrove covered areas can have a salinity as high as 85 ‰.

Inland, the Rufiji River is linked to three tributaries which drain the Upper Rufiji catchment. These three tributaries; the Luwegu, Kilombero and Great Ruaha Rivers, supply 18 %, 62 % and 15 % of the total inflow to the Rufiji River, respectively (Hufslund, 1980; RUBADA, 2001; Shaghude, 2016). Moving from the Delta towards the hinterland, the climate varies with the topography, with the Delta coastal areas, which are located at the lowest elevations, being characterized by high temperatures (24-28 °C) (Timiza, 2011; Ndesanjo *et al.*, 2012) and lowest annual rainfalls (700 mm) (FAO, 1960). By contrast, the areas over the Upper Rufiji catchment, located at 100-200 m above sea level are characterized by relatively lower air temperatures (between 23 °C and 28 °C) and higher precipitation (over 1000 mm per year). Both the Delta and the areas located over the Upper Rufiji catchment are geographically located in areas that are characterized by a unimodal rainfall pattern (Timiza, 2011) with the rainy season occurring from December to March (DJFM). The DJFM rainy season is also the peak time for TCs in the SWIO basin (Kai, 2018). The season is also characterized by frequent storms (Mavume *et al.*, 2006) and heavy rainfalls, which in turn generate huge volumes of water in the river drainage systems that is also associated with coastal inundations and river bank erosion (Kai, 2018; Kai *et al.*, 2021).

Data

Satellite data

The data used for this study consisted of: 1- Medium resolution (30 m by 30 m) Landsat satellite imagery from May, 1991, July, 1997 and June, 2007 (path 166, row 65 and 66) acquired from the Global Land Cover Facilities (GLCF; http:// landcover.org/); and 2- High resolution Pleiades satellite imagery covering an area of about 7 x 8 km^2 with a spatial resolution of 0.5 m x 0.5 m (panchromatic sensor) and 2 x 2 m (multispectral sensors). The satellite imagery was used to determine the historical shoreline positions at the Rufiji Delta using a cost-effective change detection technique (Lohani and Mason, 1999). In particular, the Landsat imagery from May, 1991, July, 1997 and June, 2007 were used to assess long term shoreline changes over the Delta, while the high-resolution Pleiades imagery was used to assess the short-term riverbank erosion/ accretion changes along the Rufiji Delta and its distributary channels. The acquisition dates of the three Landsat images were targeted to tie with the December to May rainfall season and peak TC period over the SWIO basin. An attempt was made to assess the importance of episodic shoreline changes occurring during the periods characterized by strong El Nino events (such as the 1997/1998 El Nino) and uniform rates of shoreline changes influenced by weak El Nino events. In this study, the images which was considered to qualify for the assessment of shoreline changes were those which were almost cloud free (at least 94 % cloud free) over the study area. As for short-term pre- and post-storm shoreline and riverbank changes, images from the 2013/2014 TC season were used to capture the entire DJFM period. The pre- and post-storm image acquisition dates for six storms during the 2013/2014 TC season are as shown in Table 1.

Tidal data/Tables

Tides influence the shoreline dynamics through the daily rise and fall of the sea water under the influence of the gravitational forces of the moon and to a lesser

Table 1. Pre- and post-storm images captured. Names and dates for six cyclones from December 2013 to March 2014.

Images captured		Storms (name and dates)					
Pre-storm	13/09/2013	Amara	Bruce	Collin	Besija	Fobane	Hellen
Post-storm	13/06/2014	14/12/2013 23/12/2013	19/12/2013 24/12 2013	09/01/2014 15/01/2014	28/12/2013 06/01/2014	6/02/2014 14/02/2014	27/03/2014 01/04 2014

extent also the sun (Williams and Thom, 2001; Williams *et al.*, 2003; Yanxia *et al.*, 2013). Thus, to eliminate the tidal influence on the assessment of shoreline changes due to TC events, it was important to use the images acquired during low tides of the daily tidal cycle (Pugh, 1996; Nyandwi, 2000; Nayak, 2002). As there was no tide information specifically for the Rufiji Delta, the study used the information for Dar es Salaam and Mtwara to estimate the time for the occurrence of low water at the Rufiji Delta which is geographically located approximately mid-way between Dar and Mtwara. The tide tables for Dar es Salaam and Mtwara were downloaded from http://tides.mobilegeographics.com/ locations/1490.html and http://tides.mobilegeographics.com/locations/3949.html, respectively. Furthermore, all images used in this study were preferably acquired at approximately the same low tide levels as recommended by other studies (Nyandwi, 2000; Nayak, 2002; Pugh; 2004; Shaghude, 2003, 2004; Boak and Turner, 2005). For short-term shoreline changes, the images were selected by taking into consideration that the key factors contributing to shoreline changes during the pre- and post-storm events are comparable. Moreover, during the field trips, Ground Control Points (GCPs) at specific monitoring sites (areas considered to be experiencing considerable changes as evidenced from digital image analyses of the satellite remotely sensed data) were taken and used for subsequent validation (geometrical rectification) of the remotely- sensed data.

Methods

The Landsat images were geo-referenced using the Arc map 10.2.2 (toolbar for projection) and WGS 1984 Zone 37°S Universal Transverse Mercator (UTM) coordinate system, which corresponds to the location of the eastern parts of Tanzania. During processing of the Landsat images, the atmospheric correction was not taken into consideration because the difference in shorelines between the two dates was determined by image differencing, which does not necessitate the atmospheric correction (Singh, 1989). The de-striping algorithm developed by USGS (Tsai and Chen, 2008) was used to remove the stripes for the Landsat 2007 image. Furthermore, atmospheric correction was not performed for Pleaders images because (i) the temporal variability of the selected images were within the same season, and (ii) the algorithm used for determining change detection used training data from the same location and same season and no multiple seasons or places were used (Singh, 1989; Song *et al.*, 2011; Chinsu *et al.*, 2015).

Analysis of shoreline changes (delineation) was carried out in two main steps; namely 1- Visual analysis of the images using colour composite images, and 2- digital image classification (Pardo-Pascual *et al.*, 2012; Dolan *et al.*, 1980). The visual analysis of the imagery was undertaken using Arc map 10.2.2 and GRASS software. Considerable contrast between water and land was clearly evident especially when the near infrared (NIR) and mid infrared (MIR) bands were used, as noted by Jensen (1996).

To get the best contrast between water and sand, the three bands 7 (Red), 5 (Green) and 4 (Blue) false colour composites were used. The three bands combination provided the best atmospheric penetration of the electromagnetic radiations, where the coastlines and shores became clearly delineated. The other alternative band combinations of 7 (Red), 5 (Green), and 3 (Blue) false colour composites also provided clearly delineated coastlines and water features within the image.

Classification of various types of land cover was accomplished using supervised digital imagery classification with maximum likelihood algorithms and two classes (i.e., water and land for the Landsat imageries) and five classes, (i.e., land, vegetation, grass, shallow water and deep water for the Pleiades imageries). During the maximum likelihood classification, Google Earth Imagery for the Rufiji Delta site was used to identify training signatures for water, sand (bare land), vegetation and grass land cover features.

Table 2. Description of the images and tidal records for both shoreline changes over short- and long-term settings. Note that the Pleiades images consisted of 4 bands (Bands 1 to 4), with spatial resolution of 2 x 2 m for the multispectral bands, and 0.5 x 0.5 m for the panchromatic band. The mean tidal height between Dar es Salaam and Mtwara was used to estimate the tidal height at the Rufiji Delta data.

No	Scene Sensor	acquisition date	Scene central time	Tidal height f for Dar (m)	Tidal height for Mtwara (m)
1	TM5	29/05/1991	07:20:55.5130750Z	0.2	0.8
2	TM5	16/07/1997	07:02:55.5130750Z	1.5	2.0
3	ETM7	18/06/2007	07:22:40.8780071Z	1.2	1.0
4	Pleiades	17/09/2013	12:56:11Z	2.8	3.0

Analysis of the high-resolution Pleiades imagery followed similar protocols, where the training classes were; shallow water, sand (bare land), deep water, vegetation, grasses (as defined from the Google Earth imagery through visual analysis). The Arc map (10.2.2) toolbar for measuring areas and lengths was used to measure the deviation of the extracted shoreline from the reference shoreline. Distances measured to the right of the reference shoreline (i.e., towards the water) were conventionally considered as positive (i.e., accretion of the shore) and those measured to the left of the reference shoreline (i.e., towards the land) were considered as negative (signifying erosion of the shore). The perpendicular distances between two corresponding points of the two historical shorelines were measured to determine the overall direction of change (mean accretion or erosion). The net magnitude of erosion and accretion was then determined. Lastly, the shoreline shape files were compared with the reference coastline to assess the level of precision of the digitizing method.

Results and discussion

The results showing riverbank changes during 1991 – 1997 and 1997 – 2007 are presented in Fig. 2. The results indicate that considerable riverbank changes occurred from 1991 - 1997 as compared to 1997-2007. The results further show considerable sediment accretion taking place along the Rufiji Delta distributary channels, during some of the flooding events (Fig. 2b). The highly pronounced bank accretion along the Rufiji Delta distributary channels could be attributed to the super El Nino event of 1997-1998. The distributary channel bank accretion observed in 1997/1998 are in marked contrast to the bank erosion that is clearly observable in 1991 (Fig. 2a). The results further revealed that the two periods (1991-1997 and 1997-2007) were characterized by a sequential decrease in vegetation cover over the Delta; a phenomenon which may explain the observed accelerated erosion/sedimentation pattern (Fig. 3 and 4).

The observed shoreline changes along the Rufiji Delta for the period 1991 - 1997 are presented in Fig. 3. The results show that considerable accretion of the shoreline occurred during the period 1991 - 1997, especially in the northern parts of the Delta (Fig. 3b). The observed changes are illustrated by the observed differences of the shoreline width between the 1991 and 1997 satellite imagery. Very few changes along the shoreline were observed during 1991 – 2007. However, considerable reduction of shoreline width during 1997 to 2007 was evident (Fig. 4).

Analysis of the magnitude of shoreline accretion at the four sites near the river mouth (Fig. 1) show different levels of shoreline accretion along the Rufiji River mouth (Table 3). Highest magnitude of shoreline changes during the cited periods (1991 to 1997, 1997 to 2007, 1991 to 2007) were are generally observed at sites A and B respectively, and relatively lower magnitudes of shoreline changes were observed at the remaining sites.

The results further revealed that shoreline digitization using both manual (band combinations) and automatic

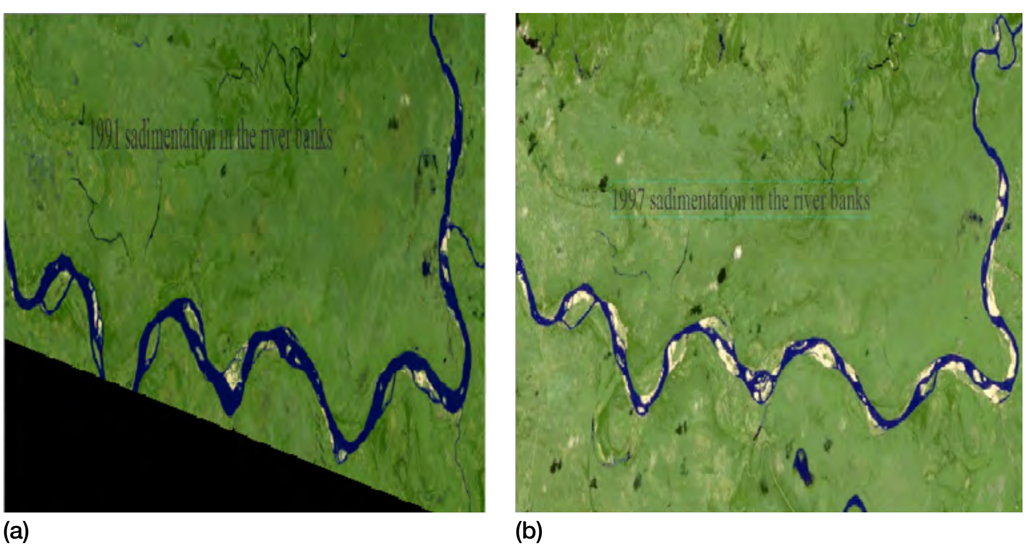

Figure 2. Variation of sediments in upstream meanders of selected parts of the Rufiji Delta distributary channel for the years 1991(a) and 1997(b). The white patches along the river bank shows the sand /sediment depositions. .

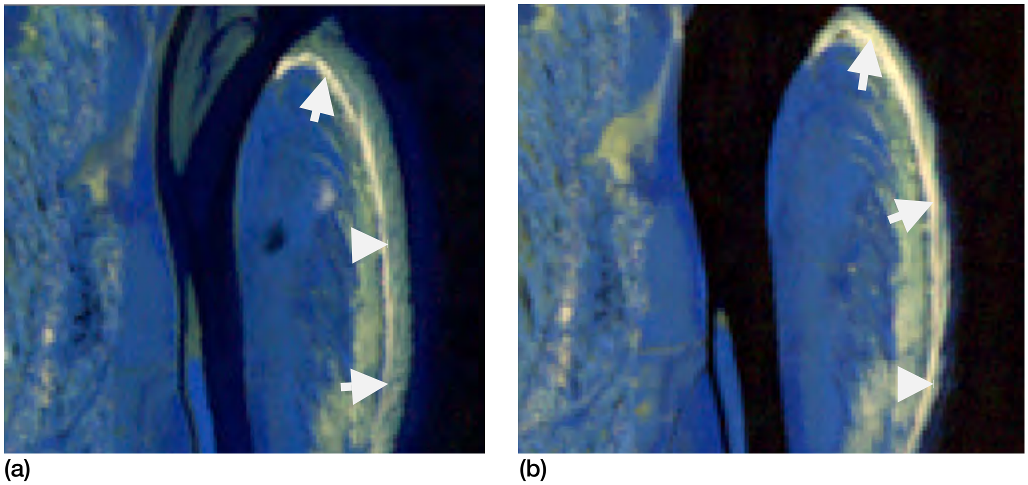

Figure 3. Observed shoreline changes along the Rufiji Delta during 1991(a) and 1997(b), as deduced by the false colour composite imagery represented by bands 7, 5 and 4 (Red, Green and Blue, respectively). The white arrows indicate the areas of signifcant shoreline changes from 1991-1997.

(supervised maximum likelihood classification) had comparable results. Moreover, the results revealed that 1997 was the period of highest shoreline erosion in the Rufiji Delta, while in 1991 there was considerable shoreline accretion due to deposition of sediments into the shoreline. The observed higher magnitudes of erosion could be explained by the strong El Nino event which is contrasted with the weak 2007 El Niño event, where considerable accretion of the riverbanks were observed. Furthermore, the results in Fig. 3 (a - b) revealed sequential erosion at Site A and Site C (Simbauranga) from 1991 to 2007, while Fig. 3 (c - d) shows that the erosion and accretion rates were altering from year to year at Site D and Site B.

The results presented in Table 3 were consistent with the observations deduced from the Google Earth satellite image of June, 2007 (Fig. not shown here) which also revealed that Site C (Simbauranga) had been consistently experiencing erosion throughout the three time periods; i.e., 1991 - 1997, 1997 - 2007 and 1991 - 2007.

The analysed images on riverbank erosion and accretion revealed that, while some parts displayed accretion at a magnitude of about 7 m^2, others had even higher magnitudes of accretion, reaching up to between 26 m^2 and 31m^2 (Fig. 5, 6). The results presented in Fig. 5 (a and b) show that during the 2013/2014 TC and heavy rainfall season, an area of about 0.2 km^2 which was previously under water was converted to a mangrove forest. This could be due to the sporadic sedimentation that took place under the influence of cyclonic or storm events. Generally, the results showed that the observed erosion and sedimentation on the riverbanks had direct linkages with the waves and storms that were accompanied by the heavy rainfall events (Fig. 6); where the magnitude of erosion and accretion of riverbanks ranged between 26 m^2 and 31 m^2.

Analyses of shoreline changes using various image classification methods based on high resolution data sets (Pleiaders images) for pre- and post-storm conditions at three selected sites in the Rufiji Delta are presented in Table 4. Maximum likelihood classification

Table 3. Magnitude of shoreline changes, including accretion (AC) or erosion (ER) at four selected sites along the Rufiji River Mouth of the Rufiji Delta during the three time periods (1991, 1997 and 2007). ER and AC indicate average magnitude of erosion and accretion, respectively, while NTC indicates the direction of the net change. At Simbauranga (Site C) (1997 - 2007) and Site D (1991-2007) the changes were both negative (erosion) and positive (accretion) and tended to cancel each other out.

Years	Site A			Site B			Site D			Site C		
	ER	AC	NTC	ER	AC	NTC	ER	AC	NTC	ER	AC	NTC
1991-1997	25.7 - 100	0 - 15	ER	14.3-64.3	6.4-21.7	ER	0 -18.3	0	ER	0-94.6	0-1	ER
1997-2007	0- 2	11.6 - 83	ER	5.6-18	2 - 29.9	ER	0	3.9 - 21.8	AC	0 -28	0-19.5	ER & AC
1991-2007	7.6 - 38	4.7 - 29.2	AC	3.8 - 35.6	6.73 - 18.7	ER	0 - 10.5	0 - 11	ER & AC	2.8- 48	0	ER

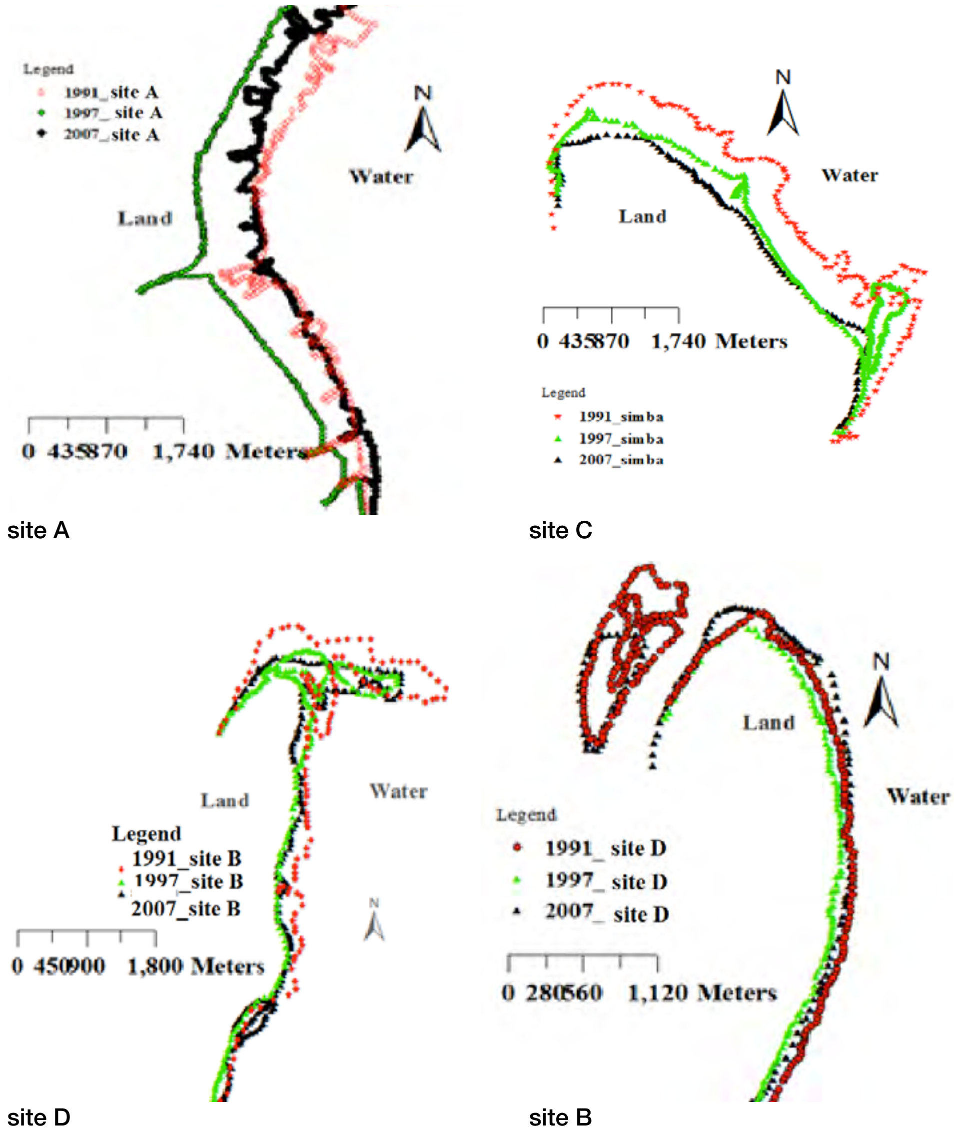

Figure 4. Manually digitized shorelines for different shoreline areas of the Rufiji Delta using the maximum likelihood classification with two classes of Land and Water. The red curve indicates the 1991 shoreline changes, the blue curve indicates the 1997 shoreline and the black curve indicates the 2007 shoreline changes at different sites near the mouth of the Rufiji River as shown in Fig. 1.

and false colour composite classification methods provided slightly different results, with the false colour composite method giving lower values for both the magnitude of accretion and overall change. Furthermore, Sobel filtering using band 4 (NIR) seemed to work better than the band combinations and the maximum likelihood classification methods for delineating the shoreline. This could be explained by the fact that the problem of shallow water radiances and water turbidity was solved in Sobel filtering after merging land and vegetation with high reflectance in band 4 into one class. Moreover, composite analysis of the imagery using bands 4, 3 and 2, or bands 3, 2 and 1 could not solve the shallow water and water turbidity problem.

Furthermore, the presented results show that the maximum likelihood classification was a better land cover classifier than the band combination classification methods. The results of the former method (maximum likelihood classification) consistently agreed with the visual analyses of the Google Earth satellite images, while the results of the later method were highly inconsistent from one another, and with the visual analyses of the Google Earth satellite images. This could be explained by the fact that the method lacks the ability to distinguish wet sands and turbid water (i.e., when the level of inorganic materials in turbid water increases the water looks like the wet sand). The problem of shallow water and water turbidity radiances could be solved by using change

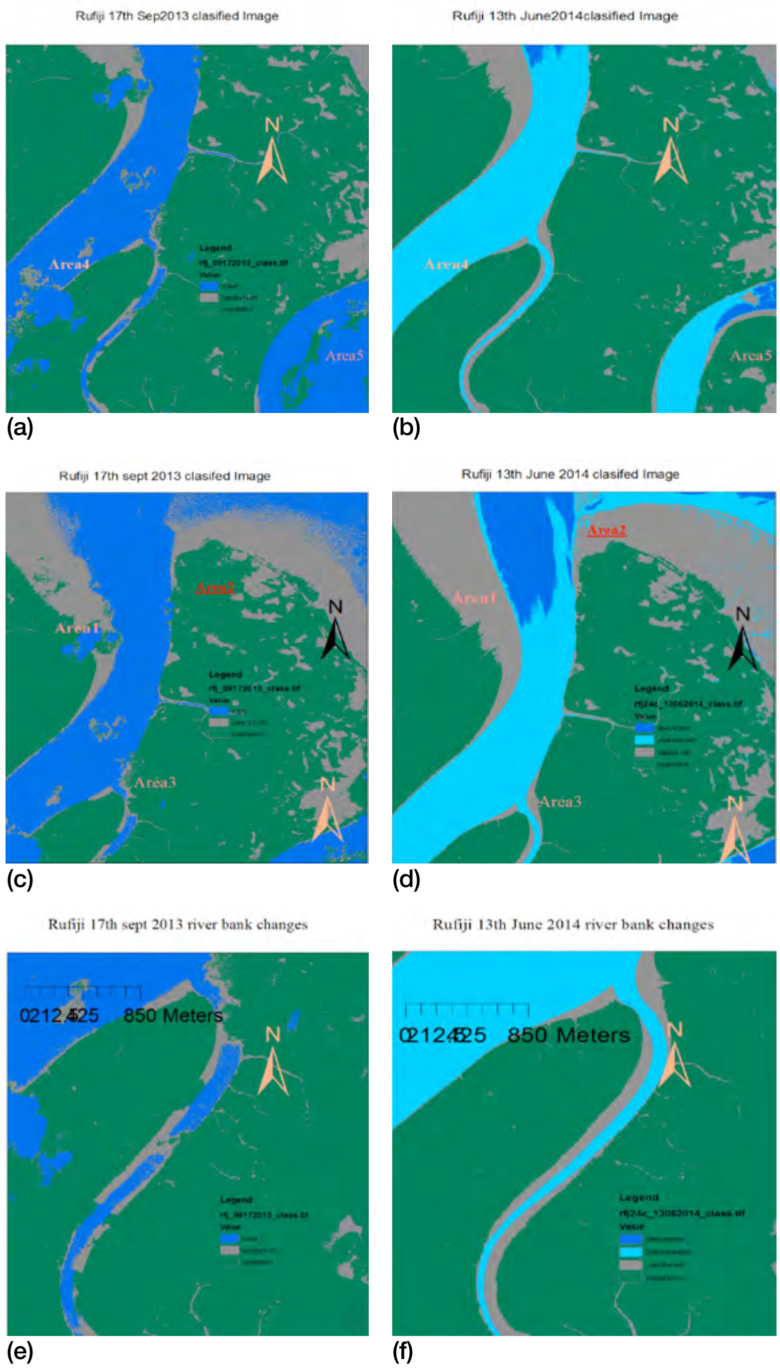

Figure 5. Sediment deposition and erosion along the Rufiji riverbanks as deduced from the classified images for the pre- (images on the left) and post- (images on the right) TC events, with areas covered by water (blue), sand (grey), and land (green). Images (a) and (b) show areas covered with water in pre-storm conditions, and covered with vegetation in post-storm conditions. Images (c) and (d) show three areas (Area 1, Area 2, and Area 3) with erosion in pre-storm conditions but sediment deposition in post storm conditions. Images (e) and (f) show extensive erosion in the river tributary in pre-storm conditions, but sediment deposition during post-storm conditions.

detection, employing subtraction between band 4 and band 3, or band 4 and band 2, or band rationing as discussed by Jensen (1996), Frouin *et al*. (1996) and Lohani and Mason, (1999). In this study these processes were not taken into consideration.

Additionally, the results presented on the magnitude of changes due to the influence of all cyclones over the given period, using three methods over the Rufiji Delta (Table 4) show that all the three methods showed relatively higher rates of accretion compared to erosion at

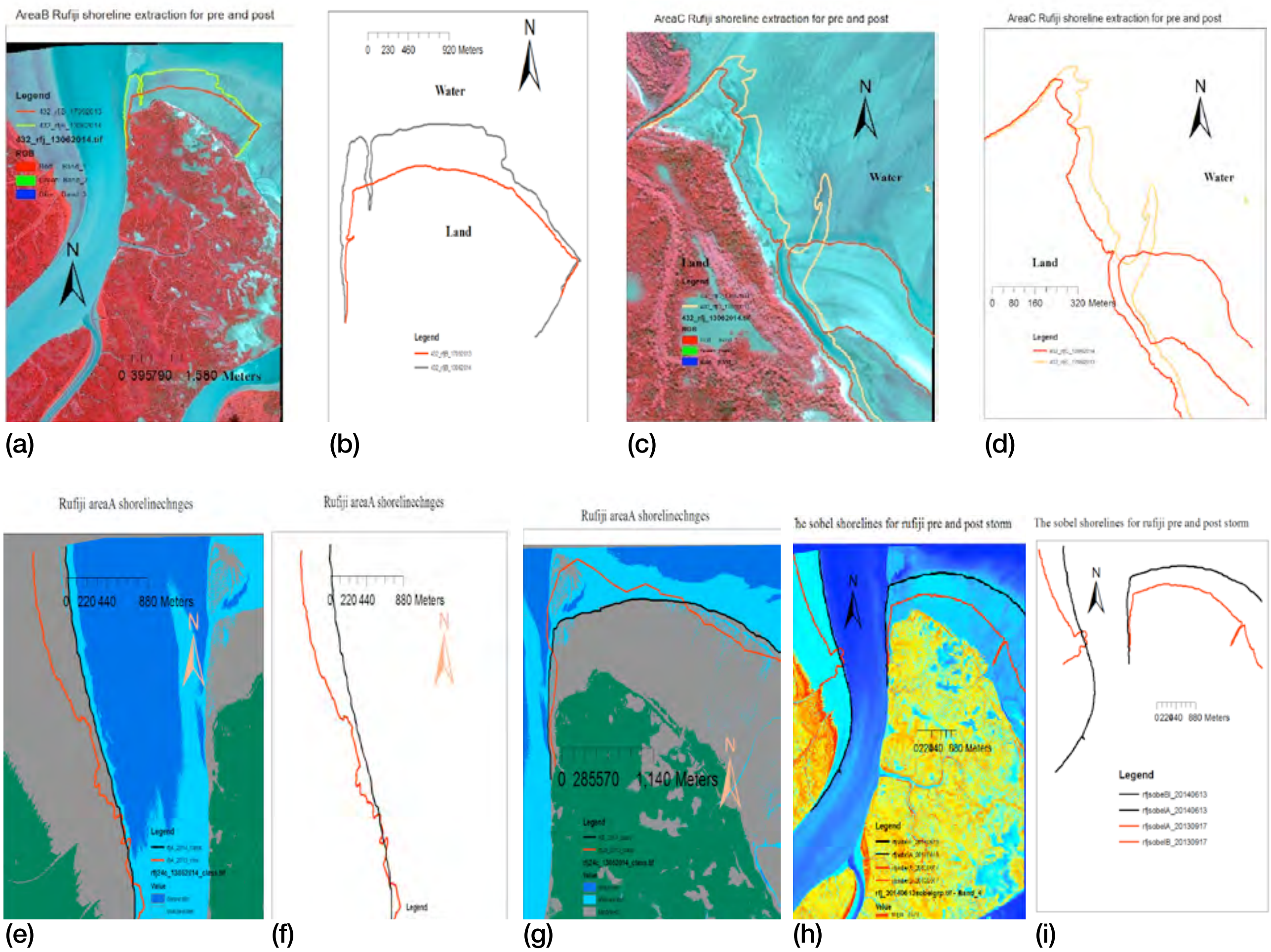

Figure 6. Digitized shorelines over the Rufiji delta using the true and false coluor composites (6a6d), maximum likelihood classification (6eand-6g) and Sobel filtering (6h and 6i) for pre- and post-cyclone conditions.

Site A. The Sobel filter method gave the highest magnitude of accretion (up to 393 m) followed by band combination (up to 244 m), and maximum likelihood classification gave the lowest magnitude of accretion (about 206 m). Moreover, all the three methods indicated that Site A had a relatively low magnitude of erosion, or no erosion at all. With the maximum likelihood classification (Fig. 6) Site C (Simbauranga) had the highest magnitude of erosion (of up to - 242 m) while the band combination and Sobel filter showed that this Site was not eroding and instead had been accreting. In general, the Sobel filter and band combinations classification methods were found to be good delineators for the accretion process while maximum likelihood classification seemed to be a good delineator for both accretion and erosion processes.

Further assessment of the three classification methods was made by digitizing the shoreline changes at Site C (Simbauranga) pre- and post-storm events as observed during the field trips to the Rufiji Delta (Fig. 7). The results showed that despite the apparent differences

Table 4. Results of the computation of the magnitude of erosion/accretion (in meters and not m^2) using maximum likelihood classification method, false colour composites, and Sobel filtering for the three selected sites in the Rufiji Delta.

Classification method	Maximum likelihood classification			False colour composite			Nonlinear egde enhancement (sobel)		
Type of change	Overall change	Accretion	Erosion	Overall change	Accretion	Erosion	Overall change	Accretion	Erosion
Site A	206	142	-42	244	244	-	393	393	-
Site B	-	-	-	37	52	-8.1	-	-	-
Site C	-8.7	100	-242	177	177	-	215	215	-

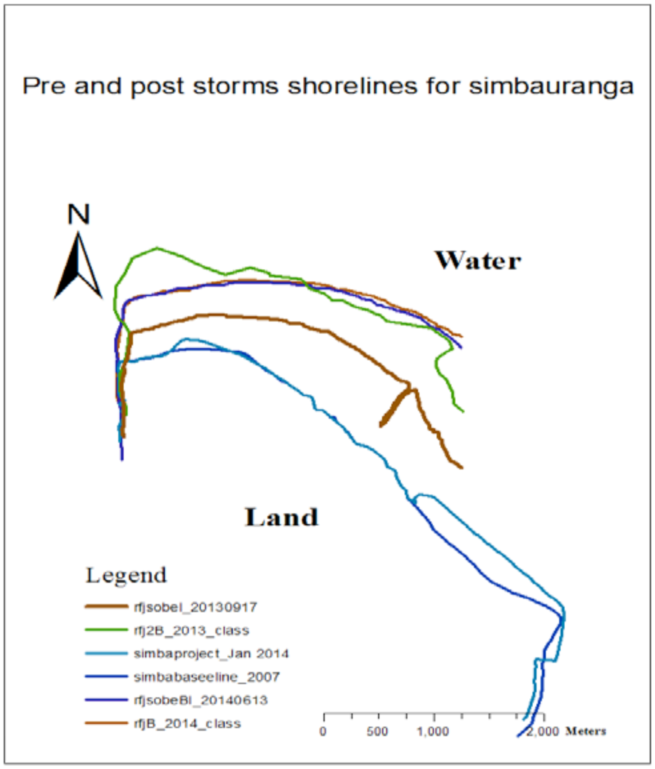

Figure 7. Digitized shorelines for Simbauranga pre- and post-storms of 2013/014, 2007 and the field survey of October, 2014.

between the three classification methods, all three methods revealed that Site A and Site C (Simbauranga) in the Rufiji Delta had been accreting over time.

Past studies had revealed that TCs along the coast of Tanzania mostly occur from December to May (Kai, 2018; Kai *et al*., 2021). Furthermore, the study by Kai (2018) identified that the December to May (DM) cyclone season is sub-divided in two sub-seasons; namely the December to February (DJF) wet sub-season, and the three wet months (March to May (MAM) sub-season. The two sub-seasons coincide with the DJFM TC and Tropical Storms (TS) peak season with a mean TC and TS frequency of 2.2 and 4.8 per season (Kai, 2018). The TCs not only influence the DJF and MAM (tending to enhance the rainfall during the two sub-seasons), but also have considerable influence on riverbank changes (erosion or accretion). The TCs influence on shorelines is mainly through the strong winds and waves which often accompany the TC events. When such waves collide with the shoreline they may head into the estuary and accelerate sediment erosion/deposition of the riverbanks. They may also give rise to longshore sediment transport, with localized areas of preferential shoreline accretion (Shaghude *et al*., 2015).

Other studies show that TCs may also be associated with sporadic incidences of sea level rise or coastal inundations (flooding) of low-lying coastal areas (Yanxia *et al.,* 2013; Jonathan et *al.,* 2013; Larcombe and Carter, 2004).

An anecdotal survey of various age groups of people residing in the Rufiji Delta consistently agreed with the results presented above. The interviewed residents of the Rufiji Delta reported that most environmental changes (sediment deposition and erosion in the Delta and the Delta distributary channels) occur during December to April. The residents of Simbauranga acknowledged that their village has been facing considerable environmental degradation over time under the influence of strong waves and winds and most of the degradation occurs during the DJFM period. They further revealed that during the last 10 to 15 years, their shoreline had transgressed (moved landward) for an estimated distance of about 150 - 200 m (from current position) seaward (based on GPS measurements). Regeneration of new mangrove trees due to sediment deposition along the riverbanks, shifting of the distributary channels and erosion of the riverbanks were among the issues that were reported during the anecdotal survey.

The main challenge encountered during this study was acquisition of at least 95 % cloud free Landsat satellite imageries that also matched with the low tidal phase and the DJFM tropical cyclone season. Future studies with a similar objective should therefore consider employing microwave remote sensing sensors which are not affected by clouds.

It has been demonstrated in this study that TCs affect the Rufiji River estuary, delta and shoreline, on both a short-term and long-term basis. This work has indicated that regular monitoring of these critical habitats in other parts of the Tanzanian coast through oceanographic studies using remotely-sensed satellite data is possible, and is necessary to inform coastal planning and management efforts.

Acknowledgements

The authors would like to acknowledge the CCIAM programme through Prof Lars Eik for supporting the idea of using high resolution data and financing the four pieces of data for the study areas of Chwaka Bay and Rufiji Delta. The authors would like to also thank the CCIAM programme for providing a 4-month scholarship that allowed the corresponding author to formulate the study and streamline its results. Last but not least, the authors would like to extend their sincere appreciation to the Tanzania Meteorology Authority (TMA) for technical and financial support to this study.

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Additions to the marine decapod (Crustacea: Decapoda) fauna of South Africa

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Abstract
This report adds five previously unreported species to the decapod crustacean fauna of South Africa, as well as removing one species previously listed in error. It also documents locality (and/or reference specimen) data for 12 other species, most of which had been depicted in regional field guides, but without reporting when and where they had been collected. Almost all the species added were already known from adjacent African countries and their ranges are here extended into South Africa. Although some of these records are based on photographs, rather than collected specimens, it is argued that such records should be accepted as adequate evidence for inclusion of at least visually-distinctive crustacean species into the regional fauna.

Keywords: fauna list, marine biodiversity, new records, photographic records, citizen science

Introduction
The last comprehensive monographic compilation of South African decapod Crustacea was that of Barnard (1950), although many of the more prominent regional species are depicted in Emmerson (2016), who also provides a comprehensive listing of the fauna of the wider southern African region. Reports of additional regional species are, however, continually coming to light, so that even Emmerson (2016) is now in need of revision. Maintaining accurate and up-to-date national species inventories and distribution records has a number of important functions. At the most basic level, species inventories are used to report on the taxonomic richness of the national fauna. For the South African marine fauna as a whole, for example, such compilations have been provided by Gibbons *et al.* (1999) and updated by Griffiths *et al.* (2010). These tabulations are also used to make international comparisons between species richness patterns between regions, such as that of Costello *et al*. (2010).

Recording the localities from which species have been reported is equally critical, as these data can then be used to plot geographical patterns of species richness, endemicity, etc. Such broader analyses of the marine invertebrate fauna in South Africa have been carried out by Awad *et al.* (2002), Griffiths *et al.* (2010) and Scott *et al.* (2012). The uses of such data include the design and placement of appropriate Marine Protected Area (MPA) networks (Awad *et al.,* 2002).

The methods by which researchers collate and formalise species records have undergone significant historical changes in recent years. For centuries, the documentation of marine species and their distribution patterns were based almost entirely on traditional collection and curation techniques and were undertaken, or at least reported, almost exclusively by professional marine scientists who obtained samples by a variety of destructive techniques, including grabs,

http://dx.doi.org/10.4314/wiojms.v21i1.5

trawls, dredges and cores, or in the case of coastal species, via manual collection. After preservation and identification, these specimens were then catalogued and lodged in museum collections and the results published in academic journals or monographs.

Recently, however, both methods of acquisition of species records and of publication, have changed radically. There has been a move towards non-destructive sampling, particularly photographic techniques, both remote (sledges, ROV's, BRUV's, etc.), and via proliferation of hand-held underwater cameras in the hands of both professional researchers and amateur SCUBA divers (Potts *et al.*, 2021). These methods often generate valuable species records, at least of those groups where external morphology and colour pattern are diagnostic, such as fishes, for which new photographic records have long been reported and accepted (e.g. Francis *et al.*, 1999). Amongst invertebrates, however, range expansions that are not supported by physical samples have seldom been reported on in formal taxonomic papers (but for a South African exception see Laird and Griffiths (2016) and Emmerson *et al*. (1990).

There has also been a trend towards publication of more popular photographic field guides, in place of technical taxonomic monographs, as well as a proliferation of 'virtual museums' to which the public can submit images. For example, many excellent images of South African decapod crustaceans (as well as many other groups) appear in recent photographic field guides by King and Fraser (2014), Branch *et al.* (2016) and Atkinson and Sink (2018). Many additional photographic records also appear on citizen science platforms, particularly iSpot (https://www.ispotnature.org), Echinomap (http://vmus.adu.org.za) and iNaturalist (https://www.inaturalist.org). Although all these sources provide valuable images, along with some descriptive text, they do not provide the full spectrum of information reported in formal taxonomic publications. Specifically, they generally do not explicitly state whether the organisms depicted are novel to the fauna, they often fail to provide precise collection data (date, location, depth, substratum, etc.), and such reports are hardly ever supported by reference specimens lodged in museum collections.

The objective of this paper is to add recent new records to the South African decapod fauna collected by the authors, as well as to formalise records that have already been reported in other media without providing adequate distributional and collection data.

Materials and methods

Some of the new records reported upon below derive from work done by the authors themselves, either though collection of physical specimens, using a variety of techniques, or through images taken with conventional underwater cameras. Other records were originally published in regional field guides, and original locality data has been obtained from authors or contributors linked to the images (indeed they often provided additional unpublished images and observations not mentioned in their books, or taken subsequent to their publication). In the case of records originating from virtual museums, locality data were collated for all images on the sites, thus establishing the known distributional ranges of each species. Where specimens were collected, these were deposited in the collections of the Iziko South African Museum and reference numbers were provided for these voucher specimens, thus making them accessible for future taxonomic research.

Results

Each of the species added to, or removed from, the South African decapod fauna is listed below and depicted in the plates, as indicated. The accompanying text gives relevant references, outlines the evidence upon which the species are included and provides locality records of their known distribution in South Africa.

Family Hymenoceridae

1. *Hymenocera picta* Dana, 1852

Fig 1a

Hymenocera picta King and Fraser, 2014: 280, Branch *et al.,* 2016:108; Emmerson, 2016, Vol 1: 212-215.

Remarkably, given the frequency with which regional divers have photographed this species, and the fact that both King and Fraser (2014) and Branch *et al.* (2016) indicate that it occurs in KwaZulu-Natal, there are no specific locality records for this species from within South Africa and its distribution is still listed by Emmerson (2016) as being from Maputo Bay, Mozambique, northwards. Here we document specific photographic locality records as follows:

Locality records: Sodwana Bay, Bikini Reef, 10 Jan 2007, S 27° 31.72', E 32° 41.31', 18-22 m, Kerry Sink; Sodwana Bay, Ribbon Reef, 2 Aug 2012, S 27° 29.40', E 32° 41.23', 15–20 m, also Aliwal Shoal off Scottburgh, photograph, Valda Fraser. iNaturalist records all Sodwana Bay: Stringer, 22 Apr 2016, S 27° 31.69', E 32°

Figure 1. a. *Hymenocera picta* Dana, 1852 Aliwal Shoal off Scottburgh; b. *Ancylocaris brevicarpalis* Schenkel, 1902, Sodwana Bay; c. *Cuapetes tenuipes* (Borradaile, 1898) and d. *Pontonides unciger* Calman, 1939), both Sodwana Bay, Bikini Reef; e. *Zenopontonia rex* (Kemp, 1922) and f. colour variant of *Z. rex*, both Sodwana Bay; photographs all Valda Fraser.

43.61', 14 m, UserID 'rosepalmer'; Rooneys, 10 Oct 2015, S 27° 32.12', E 32° 41.20', UserID 'jennyjj'; 7 Oct 2015, S 27° 31.44', E 32° 41.16', UserID 'Georgina Jones'; 6 Mile Reef, S 27° 31.91', E 32° 42.49', 4 Feb 2015, and 5 Mile Reef, S 27° 29.34', E 32° 41.37', 26 Apr 2012, UserID 'rowanwattpringle'.

Identification: Unmistakable and striking species, white with large red spots each fringed with blue. Legs banded with purple. Antennae leaf-like.

Distribution and habitat: Widespread Indo-Pacific, territorial and usually found in pairs; a specialist predator of echinoderms. In South Africa, regularly observed at Sodwana Bay, most southerly record Aliwal Shoal.

Family Palaemonidae

2. *Ancylocaris brevicarpalis* Schenkel, 1902

Fig 1b

Periclimenes brevicarpalis Gosliner *et al.,* 1996: 203; King and Fraser, 2014: 278; Emmerson, 2016, Vol 3: 432.

Emmerson (2016) lists this species in the checklist section of his book, but gives the distribution as Mozambique. King and Fraser (2014) previously reported the species as occurring in KwaZulu-Natal. The known locality records from South Africa are listed below.

Locality records: Sodwana Bay, 9 Mile Reef, S 27° 24.60', E 32° 43.80', 18 m, 13 Jan 2008, photograph, Valda Fraser; 5 Mile Reef, S 27° 36.91', E 32° 0.02', photograph, Kerry Sink. iNaturalist record: Sodwana Bay, S 27° 31.98', E 32° 40.98', 5 Feb 2013, User ID 'rowanwattpringle'.

Identification: Unmistakable species with large wart-like white patch on carapace, as series of white blotches on thorax and 5 large black spots with orange centres on uropods and telson.

Distribution and habitat: Widespread Indo-Pacific. Lives commensally on large sea anemones.

3. *Cuapetes tenuipes* (Borradaile, 1898)

Fig 1c

Cuapetes tenuipes King and Fraser, 2014: 278; Emmerson, 2016, Vol 3: 431.

Emmerson (2016) lists this species as a new record for KwaZulu-Natal in the checklist section of his book, but this record is based on the image and field-guide entry given by King and Fraser (2014). Neither reference provided detailed locality records, so this information is provided below.

Locality records: Sodwana Bay, Bikini Reef, S 27° 31.72', E 32°41.31', 18–22 m, 30 Jul 2012, photograph, Valda Fraser. iNaturalist records: Sodwana Bay, Elusive Reef (N of Sodwana) 10 May 2018, UserID 'robert-taylor'; two observations both 7 Mile Reef, S 27° 27.99', E 32° 42.60', 22 m, 5 Sep 2014, UserID 'rowanwattpringle'.

Identification: Very transparent species with a white line joining the eyes and red and white stripes along the abdomen, tips of chelae orange. Rostrum long with 8–10 dorsal and 6–9 ventral spines.

Distribution: Widespread Indo-Pacific, free-living, but sometimes on sea anemones. In South Africa distribution records not extending south of Sodwana Bay.

4. *Pontonides unciger* Calman, 1939

Fig 1d

Pontonides unciger Gosliner *et al.,* 1996: 207; King and Fraser, 2014: 280; Emmerson, 2016, Vol 3: 432.

Emmerson (2016) lists this species as a new record for KwaZulu-Natal in the checklist section of his book, but this record is based on the image and field-guide entry given by King and Fraser (2014). Neither reference provided specific locality records, so this information is provided below.

Locality records: Sodwana Bay, Bikini Reef, S 27° 31.72', E 32°41.31', 18–22 m, 8 Feb 2013, photograph, Valda Fraser. iNaturalist records: Sodwana Bay, 2 Mile Reef (Antons), S 27° 31.80', E 32° 40.80', 16 m, 17 Feb 2013, photograph, UserID 'rowanwattpringle'.

Identification: Yellowish with dark and light transverse bands across the body, eyes white. Mimics whip corals, on which it is found.

Distribution: Widespread Indo-Pacific, on whip corals. In South Africa, distribution not recorded as extending southwards beyond Sodwana Bay.

5. *Zenopontonia rex* (Kemp, 1922)

Fig 1e–f

Periclimenes rex Barnard, 1955: 47
Periclimenes imperator Emmerson, 2016, Vol 1: 250–254
Zenopontonia imperator King and Fraser, 2014: 278.

The occurrence of this well-known tropical Indo-Pacific species within the boundaries of South Africa has not previously been adequately documented. Both Barnard (1955) and Emmerson (2016) cite specific locality records from southern Mozambique and the latter gives the distribution as 'from Mozambique up though East Africa' (and beyond), despite the fact that King and Fraser (2014) had earlier stated that 'it occurs in southern KwaZulu-Natal'. Here the specific locality records are given based on observations by King and Fraser (2014) plus one additional independent observation.

Locality records: Sodwana Bay, Grants Beach, S 27° 32.20', E 32° 42.04', 54 m, Kerry Sink. Sodwana Bay, S 27° 31.98, E 32° 40.98', several occurrences on nudibranchs *Gymnodoris rubropapulosa, Chromodoris africana* and *Hexabranchus sanguineus*, off Pumula, KZN South Coast, S 30° 38.34', E 30° 32.94', occurrences on *Armina* sp. and *Miamina sinuata*, 2005–2014, photograph, Valda Fraser.

Identification: An attractive and familiar species; body red with numerous small white dots, broad sinuous white band extending from rostrum along back to cover uropods and telson; tips of chelae and legs purple.

Distribution and habitat: Widespread Indo-Pacific. Lives commensally on echinoderms and on various species of nudibranchs. In South Africa, fairly commonly seen at Sodwana Bay, extending southwards to Pumula on KZN South Coast.

6. *Zenopontonia soror* (Nobili, 1904) (removal from fauna list)

Periclimenes soror Gosliner *et al.,* 1996: 206
(Non) *Periclimenes soror* [misidentification] King and Fraser, 2014: 278; Emmerson, 2016, Vol 3: 432.

Emmerson (2016) lists this species as a new record for KwaZulu-Natal in the checklist section of his book, but that record is based on the image and field-guide entry given by King and Fraser (2014). However, it now appears that the images taken by King and Fraser (2014) represent a colour variant of *Zenopontonia rex* (above). Pending the collection of actual specimens that can be confirmed as *Z. soror* via examination of the rostral spines (which are fewer and larger than in *Z. rex*), it thus seems advisable to remove this species from the South African fauna list.

Family Alpheidae

7. *Alpheus bellulus* Miya and Miyake, 1969

Fig 2a

Alpheus bellulus Miya and Miyake, 1969: 307–314; Gosliner *et al.,* 1996; 208; King and Fraser, 2014: 282; Emmerson, 2016, Vol 3: 432.

This well-known and widespread Indo-Pacific species was reported as occurring in East Africa by Gosliner *et al.* (1996) and its range was extended into South Africa by King and Fraser (2014). It is thus listed from South Africa by Emmerson (2016), but specific locality records are not provided, so its range is still not documented. Observation details are provided for the first time below.

Locality records: 9 Mile Reef, Sodwana Bay, S 27° 31.98', E 32° 40.98', 18–23 m, 16 Nov 2010, photograph, Valda Fraser.

Identification: Body white with an attractive reticulated pattern of brown stripes and saddles. Legs banded in brown and white. Lives in association with goby species.

Distribution: Widespread Indo-Pacific, the above record appears to be the only one to date from South Africa, resulting in the recorded distribution not extending southwards beyond Sodwana Bay.

8. *Aretopsis amabilis* de Man, 1910, new record

Fig 2b

Aretopsis amabilis Marin, 2010: 1–4.

Exclusively found in association with large hermit crabs. Recorded while sampling for hermit crabs along the coasts of KwaZulu-Natal inside a *Conus* spp. shell inhabited by *Dardanus crassimanus.* Previously known from Somalia and Kenya (Vannini *et al.,* 1993), as well as Seychelles, Madagascar and Mozambique Channel (Poupin, 2016), but never as far south as South Africa.

Locality records: Off Pumula (KwaZulu-Natal), S 38° 20.40', E 30° 32.93', 20 m, 13 Oct 2015, SCUBA, SAMC–A066594, from sample SAMC–A066488, coll. Jannes Landschoff.

Identification: Left cheliped enlarged, colouration uniformly brown-orange to red with broad white or cream longitudinal dorsal stripe along the entire body. Photographs reported by Marin (2010) show

specimens with white dots and patches laterally.
Distribution: Red Sea, Somalia, Kenya, Madagascar, Seychelles and Maldives, Indonesia, Philippines, Vietnam, Japan, Australia and Marshall Islands. Reported from South Africa for the first time here.

Family Lysmatidae
9. *Lysmata debelius* Bruce, 1983, new record
Fig 2c
Lysmata debelius Bruce, 1983 [for 1982]: 115–120, Figs. 1–9; Burukovsky, 2000: 226, Fig. 4; Branch *et al.*, 2016:108; Emmerson, 2016: 434.

First described from the Philippines and popular in the aquarium trade. Appears to have a widespread Indo-Pacific distribution, but numerous locality records are from the ornamental pet industry (Baeza *et al.*, 2009). Here, a specimen photographed in South African waters in the wild is reported on that has the characteristic four large spots on the lateral carapace. It also has two additional small dots, one centrally between the four large ones, and another one posterior, but none on the pleon, like the most closely related species *Lysmata splendida* from the Maldives.

Locality records: Sodwana Bay, Bikini Reef, S 27° 31.72', E 32°41.31', 18–22 m, 28 Dec 2005, photograph Colin Odgen.

Identification: Identification by colour based on the uniformly bright red body with four large dots laterally on the carapace.

Distribution: Widespread Indo-Pacific, reported here for the first time from South African waters.

Family Enoplometopidae
10. *Enoplometopus crosnieri* Chan and Yu, 1998, new record
Fig 2d
Enoplometopus crosnieri Poupin, 2003: 645–646.

This ornamented reef lobster was originally described from Taiwan and has now been reported to have a more widespread distribution in the southwestern and central Pacific. This, however, is the first record for South Africa and to our knowledge also from the Indian Ocean. The image on which this new record is based was taken by Peter Timm, a nature enthusiast and one of the first SCUBA divers to see coelacanths in their natural environment in Sodwana Bay.

Locality records: Sodwana Bay, Triton 69, S 27° 30.91', E 32° 42.90', reef, 69 m, 16 Mar 2013, photograph, P. G. Timm.

Identification: Unmistakable by the white circle in distal half of lateral carapace.

Distribution: Central and southwestern Pacific. Range here extended to the Indian Ocean and into KwaZulu-Natal, South Africa.

Family Scyllaridae
11. *Scyllarides squammosus* (H. Milne Edwards, 1837), new record
Fig 2e
Scyllarides squammosus Emmerson, 2016, Vol 1: 296, Vol 3: 440.

A well-known Indo–Pacific species reported from Mozambique by Emmerson (2016), but range here extended into South Africa. Could be confused with the sympatric *E. elisabethae*, but that species has legs distinctively banded in red.

Locality records: South Coast KwaZulu-Natal, off Pumula, S 30° 38.52', E 30° 32.40', 35 m, 14 Apr 2014, photograph, Valda Fraser.

Identification: Mottled reddish-brown, body covered in small nodules. Terminal segment of antenna rounded, penultimate segment not strongly toothed. Carapace parallel-sided, anterior corners not acutely projecting.

Distribution: Widespread Indo-Pacific, here extended for the first time into South Africa.

Family Lithodidae
12. *Lithodes ferox* Filhol, 1885
Fig 2f
Lithodes ferox Emmerson, 2016, Vol 2: 85–89, Vol 3: 447; Griffiths *et al.*, 2018: 187.

Reported and illustrated from South Africa by Griffiths *et al.* (2018), but they fail to provide detailed station data or to point out that, as this species is listed by Emmerson (2016) as occurring only in Namibia and northwards, their records represent an addition to the South African fauna.

Locality records: West Coast Demersal Survey AFR291, sta A33086, S 31° 42.83', E 15° 59.81', 524 m, 15 Jan 2017, 1 female, SAMC–A066570, coll. Leila Nefdt.

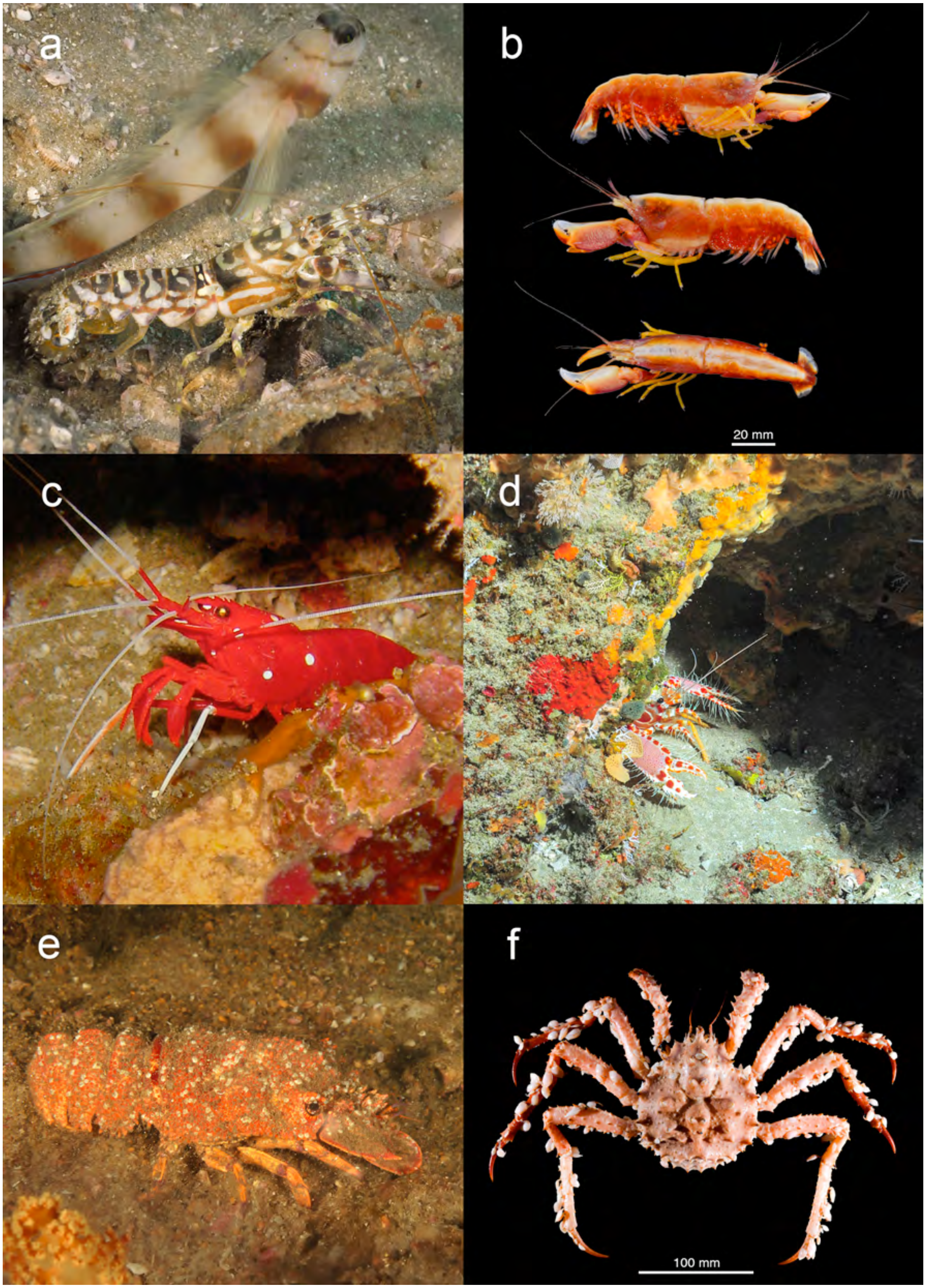

Figure 2. a. *Alpheus bellulus* Miya and Miyake, 1969, Sodwana Bay 9 Mile Reef, photograph Valda Fraser; b. *Aretopsis amabilis* de Man, 1910, off Pumula, left, right and dorsal aspect of same specimen, SAMC-A066594, photograph Jannes Landschoff; c. *Lysmata debelius* Bruce, 1983, Sodwana Bay, Bikini Reef, photograph Colin Odgen; d. *Enoplometopus crosnieri* Chan and Yu, 1998, Sodwana Bay, Triton 69, photograph Peter G. Timm; e. *Scyllarides squammosus* (H. Milne Edwards, 1837), off Pumula, photograph Valda Fraser; f. *Lithodes ferox* Filhol, 1885, West Coast offshore, SAMC-A066570, photograph Jannes Landschoff.

Identification: Distinguished from other regional stone crabs (*Neolithodes asperrimus* and *N. capensis*) by its smaller size, pattern and size of spines on the carapace and by shape of the rostrum, which is strongly produced and bifid with a branching pair of dorsal spines at corneal level and a downward-pointing ventral spine.

Distribution: Widely distributed across West Africa from Mauritania to Namibia at depths of 300–1000 m (Macpherson, 1988; Abello and Macpherson, 1991) and now extended into South Africa.

Family Lyreididae

13. *Lyreidus brevifrons* T. Sakai, 1937, new record

Fig 3a

Lyreidus brevifrons Emmerson, 2016, Vol 3: 454; Feldmann, 1992: 948; Griffin, 1970: 104.

Locality records: West Coast Demersal Survey, South of False Bay shelf edge, AFR300, sta A34550, S 35° 32.22', E 19° 01.20', 406 m, 2 March 2020, specimen damaged, SAMC–A066561, coll. Donia Wozniak.

Despite being damaged in the trawl the specimen collected here is well identifiable and also the only species of the genus known to extend into the western Indian Ocean. Genus new to the Southern African region, but it had been photographed before off the Mozambican coast by Tin-Yam Chan (Emmerson, 2016, page 455).

Identification: Rostrum acute, basal width equal to length. Anterolateral margin strongly granular, naked, concavo-convex from front backwards. One short, curved lateral carapace spine, with anterolaterally rounded sternum. Cheliped with one blunt merus dorsal spine a third from base and two long, sharp slender carpus dorsal spines. One small, blunt spine on third abdomen segment and one long, sharp spine on fourth abdomen segment.

Distribution: Known from Japan, South China Sea, west of the Philippines and Dar es Salaam. Here extended for the first time into South Africa.

Family Oziidae

14. *Lydia annulipes* (H. Milne Edwards, 1834)

Fig 3b

Euruppiella annulipes Barnard, 1950: 248; 1955: 4
Lydia annulipes Emmerson, 2016, Vol 2: 367–369.

There has been previous controversy as to whether this species occurs in South Africa (see detailed discussion in Emmerson, 2016). This is thus a confirmation, rather than a completely new record. Barnard (1950) documented earlier records of this species from 'the Cape' but expressed doubt about their authenticity, as he noted this as being a more tropical species. Although he then went on to confirm the species as occurring in Kosi Bay and St Lucia Bay, South Africa (Barnard, 1955), it was still not included in the regional fauna list of Kensley (1981). Moreover, Emmerson (2016) still listed the species as occurring from Inhaca Island, Mozambique northwards (thus not in South Africa), despite the fact that he reports on the records by Barnard (1955). The following records confirm its occurrence well into South Africa.

Locality records: Park Rynie, rock pool, S 30° 20.37, E 30° 44.11', intertidal, 2 Feb 2015, photograph, Alec Stansell.

Identification: Readily identified in the field by the red-orange body and cream legs with red bands at the joints. Carapace with marked anterior grooves medially and lumps laterally.

Distribution and habitat: Widespread Indo-Pacific. Intertidal, often under rocks. Distribution here confirmed to extend to South Coast of KwaZulu-Natal.

Family Leucosiidae

15. *Tanaoa pustulosus* (Wood-Mason, 1891)

Fig 3c

Tanaoa pustulosus Galil, 2003, 404–406, Figs. 1D, 3G–H; Emmerson, 2016, Vol 3: 460); Griffiths *et al.,* 2018: 226.

Although reported on and illustrated by Griffiths *et al.* (2018), they do not include detailed station data, which is given here. Emmerson (2016) lists this species from Mozambique so that the record reported here represents an important official range extension.

Locality records: South Coast Demersal Survey, South of Kenton-on-Sea, AND005 sta D00565, S 34°10.28', E 26° 46. 75, 425 m, 2 May 2015, SAMC–A091356, coll. Robin Leslie.

Identification: Carapace rounded, covered in fine granules, with distinctive long, pointed and upturned posterior spike and several smaller projections around posterior margin. Chelae elongate with narrow claws. Colour orange.

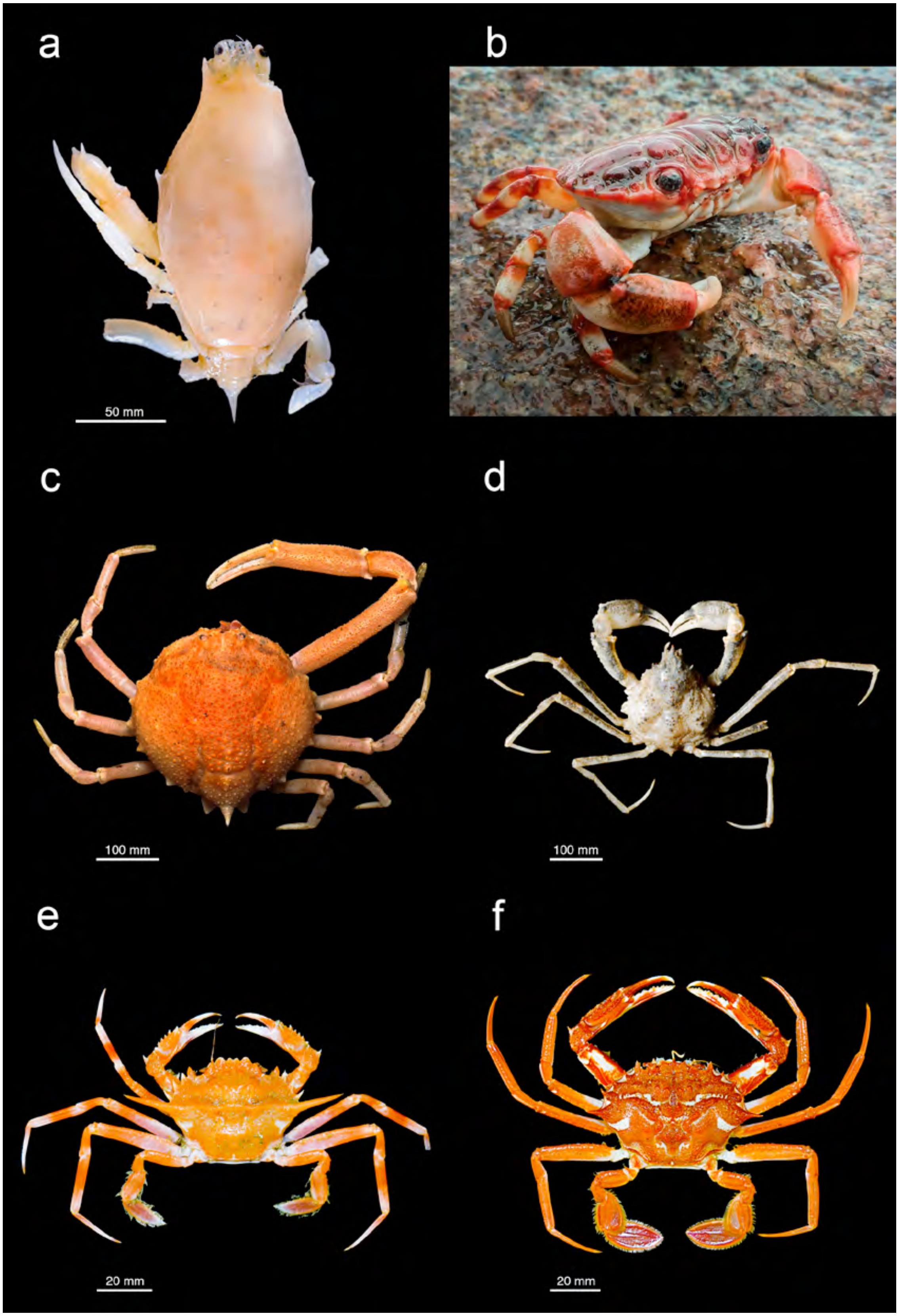

Figure 3. a. *Lyreidus brevifrons* T. Sakai, 1937, shelf edge south of False Bay, SAMC-A066561, photograph Jannes Landschoff; b. *Lydia annulipes* (H. Milne Edwards, 1834), Park Rynie, photograph Alec Stansell; c. *Tanaoa pustulosus* (Wood-Mason, 1891), South Coast off south Kenton-on-Sea, SAMC-A091356 and d. *Pyromaia tuberculata* (Lockington, 1877), South Coast south of Gouritz River Mouth, SAMC-A091355, both photograph Robin Leslie; e. *Bathynectes piperitus* Manning and Holthius, 1981, West Coast offshore, SAM-A091359 and f. *Macropipus australis* Guinot, 1961, West Coast offshore, SAMC-A091357, both photograph Donia Wozniak.

Distribution: Widespread Indo-Pacific, now extending into South Africa.

Family Inachoididae

16. *Pyromaia tuberculata* (Lockington, 1877)

Fig 3d

Pyromaia tuberculata Ahyong, 2005: 460–461; Griffiths *et al.,* 2018: 198.

Reported on and illustrated by Griffiths *et al.* (2018), but they fail to point out that, as this species is not listed by Emmerson (2016), it represents an addition to the regional fauna. This species was also recently reported from Eastern Australia by Ahyong (2005) and may represent an introduction to both locations.

Locality records: South Coast Demersal Survey, South of Gouritz River Mouth, AND005 sta D00596, S 34° 23.23', E 21° 54.10', 43 m, 10 May 2015, SAMC–A091355 coll. Robin Leslie.

Identification: Carapace oval, bearing one anterior and three larger posterior tubercles, each covered on smaller knobs. Chelae inflated in males, but slender in females. Legs long and slender, with elongate dactyls.

Distribution and habitat: Native to Pacific North America. A suspected introduction.

Family Polybiidae

17. *Bathynectes piperitus* Manning and Holthius, 1981

Fig 3e

Bathynectes piperitus Manning and Holthius, 1981: 77–83; Emmerson, 2016, Vol 2: 563–568; Griffiths *et al.,* 2018: 218.

Reported and illustrated by Griffiths *et al.* (2018), but they do not provide detailed station data and fail to point out that, as this species is listed by Emmerson (2016) as occurring only in Namibia, their records represent an addition to the South African fauna. A well-known and significant fisheries species (Abello and Macpherson, 1989).

Locality records: Caught by demersal trawl at several locations ranging from the border with Namibia (S 29° 47.30', E 14° 57.45') to just West of St Helena Bay (S 32° 45.162', E 17° 0.66'). Depth ranging from 353–415 m. *Reference specimens:* AFR296, sta A33877, S 29° 47.30' E 14°57.45', 381 m, 3 March 2019, SAMC–A091359, coll. Donia Wozniak.

Identification: Carapace oval with four teeth between eyes and four on each side before strong long lateral spine; scalloped ridge runs horizontally across centre of carapace and surrounds a square depression centrally. Colour orange with distinctive red and white banded legs, tips of chelae white.

Distribution: West Africa from Cape Verde Islands southwards to Namibia and now extended to along the west coast of South Africa.

18. *Macropipus australis* Guinot, 1961

Fig 3f

Macropipus australis Manning and Holthius, 1981: 85; Emmerson, 2016, Vol 3: 466; Griffiths *et al.,* 2018: 215.

Reported on and illustrated by Griffiths *et al.* (2018), but they do not list detailed station data as provided below that prove an extension record into SA waters.

Locality records: Caught by demersal trawl at several locations ranging from the border with Namibia (S 28° 52.15', E 16° 26.92') to just South of Hondeklip Bay (S 31° 7.914', E 17° 26.754'). Depth ranging from 80–210 m. *Reference specimens:* AFR296, sta A33882, S 29° 14.09', E 15° 41.44', 175 m, 2 males SAMC–A091357, 4 March 2019; AFR296, sta A33884, S 29° 46.83', E 16° 31.14', 154 m, ovigerous female SAMC–A091358, 5 March 2019, 154 m, coll. Donia Wozniak.

Identification: Carapace with three frontal teeth, four sharp curved anterolateral spines on each side and a large lateral spike. Colour pattern distinctive, brick-red, with bright iridescent patches.

Distribution: Widespread between 90–240 m depth along the coasts of Angola and Namibia, here extended into South Africa.

Discussion

The aim of this paper is to update and correct the list of decapod crustaceans known from South African waters, subsequent to the recent regional monograph and listing by Emmerson (2016).

The additional records reported fall into two categories. The first of these are species added to the fauna in the conventional way – that is the species are newly collected and reported as completely novel to South Africa. Many of these were in fact included in the decapod fauna lists for the wider southern African region published by Emmerson (2016), but based only on records from outside of the territory of South Africa. Additional locality records from within those borders are now reported, and these species can thus now be added to the South African national fauna list. The second group of species are those that have already been reported from South Africa in recent photographic field guides (King and Fraser, 2014; Griffiths *et al.,* 2018), and sometimes also by Emmerson (2016), but for which critical information, such as the date, location and/or depth (and museum catalogue number of specimen(s), where these exist) were not provided in the original source. By supplying these additional data, the usefulness of these reports is enhanced. A total of five species are reported from South Africa for the first time and formal locality records are provided to confirm the distributions of a further 12 species that were previously reported in field guides or as unconfirmed records. One other species is removed from the national species list as the previous field guide record appears to have been based on a misidentification.

The ease with which these numerous new records have been gathered illustrates how many decapods remain to be discovered in the region. Also, the majority of new records originate from the more tropical waters of the east coast, which have historically been more poorly sampled than the temperate South and West coasts, which support much larger commercial fisheries and have been subject to more intensive research and sampling to support their management (Griffiths *et al.*, 2010).

Acknowledgements

Our sincere thanks to Valda and Mike Fraser, whose photographic library of marine species from KwaZulu-Natal provided the basis for many of the above discoveries. We thank Valda and Mike, Peter Timm and Colin Ogden for allowing us to publish their photographs and for providing us with locality data for their images. Our thanks also to the citizen scientists and other photographers whose images on iNaturalist provided additional locality data. We are grateful to the curatorial staff of the Iziko South African Museum for access to their collections and for accessioning specimens collected during this study. We acknowledge the Master, crew and researchers who have supported data collection for marine invertebrates during demersal research surveys. SANBI acknowledges the SeaKeys Project (Grant 86950) funded by the Foundational Biodiversity Information Programme of the National Research Foundation for support for marine citizen science. Additionally, the Sea Change Trust is thanked for providing funding to the first author that allowed the completion of the manuscript.

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Assessment of vertical and horizontal distribution of meiobenthos along a salinity gradient in the Tana and Sabaki Estuaries, north coast Kenya

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Abstract

Meiobenthos respond to variability in salinity gradients in estuarine habitats and are thus suitable organisms for ecological studies. The vertical and horizontal distribution of the meiofauna community structure of two major estuaries (Sabaki and Tana) on the north coast of Kenya were compared. The aim was to draw a meiofaunal dataset inventory of the two estuaries and to examine how salinity gradient, seasonality and sediment characteristics influence their structure. Replicate samples were collected from three sampling stations along the salinity gradient of each estuary. A total of 3,556 individuals belonging to 26 taxa were recorded. Based on seasons and across stations, the upper surface (0-5 cm) layer recorded the highest meiobenthic density (90 ± 42 ind.10 cm^{-2}), followed by 46 ± 23 ind.10 cm^{-2} (5-10 cm) and 30 ± 8 ind.10 cm^{-2} in the deepest sediment layer (10-15 cm) studied. The southeast monsoon season recorded the highest mean density (160 ± 22 ind.10 cm^{-2}) compared to the northeast monsoon season (22 ± 12 ind.10 cm^{-2}) for both estuaries. Results of the non-Metric Multidimensional Scaling technique revealed distinct seasonal composition in meiobenthos but not between the estuaries. Results of the 2-way ANOSIM test confirmed no significant differences in meiobenthic composition between the estuaries ($p = 0.712$). However, seasonal difference was significant ($p = 0.001$) with higher densities for nematoda (166 ± 99 ind.10 cm^{-2} and 56 ± 29 ind.10 cm^{-2}) recorded in Tana and Sabaki, respectively during the southeast monsoon season. At least 7 taxa out of a total of 26 were present in both estuaries. Salinity gradient, season and sediment depth were found to influence the meiobenthic densities and taxa composition.

Keywords: meiobenthos, vertical and horizontal distribution, salinity gradient, river estuary, north coast of Kenya

Introduction

Meiobenthos (benthic fauna with a size range of between 32 and 1,000 µm) characterize sedimentary matter in estuarine habitats where they not only serve as diet to macrofauna but also play a key role in the ecological functioning of the ecosystem (Schratzberger *et al.,* 2017). The vertical and horizontal distributions of meiobenthos in the river estuaries are influenced by upstream anthropogenic activities coupled with a number of natural processes from the sea. Considering the extensive catchment areas of the Sabaki and Tana rivers on the north coast of Kenya, runoff from agricultural lands containing organic, inorganic and mineral matter influence water transparency, primary production and sediment loads whereas tidal movements regulate estuarine salinities. Consequently, this induces enormous variations in community composition and abundance in the estuaries (Kotwicki *et al.,* 2005).

http://dx.doi.org/10.4314/wiojms.v21i1.6

Naturally, meiobenthos occur in high densities in estuarine sediments (Coull, 1999; Dauer *et al.*, 2000) and their abundance, species composition and diversity depends largely on sediment grain size among other abiotic factors (Alongi, 1987a, b; Vanhove *et al.*, 1992; Mutua *et al.,* 2013). Since estuarine ecosystems are known to be globally stressed by anthropogenic activities (Dauer *et al.*, 2000), the integral role of meiobenthos in food webs and the ecological balance is threatened (Vincx and Heip, 1987; Coull, 1999; Dauer *et al.*, 2000; Costa *et al.,* 2016). These habitats are very productive despite the threats from upstream anthropogenic activities. Land use patterns within the watershed modify the receiving waters through inflow of nutrients, contaminants and tons of sediment (Dauer *et al.*, 2000; Burton and Thurman, 2001). The resultant increase in nutrients comes from extensive runoff from agricultural land and town wastes (Chapman and Wang, 2001), eventually influencing the biological and ecological structure of meiobenthos at the river mouths.

Previous studies on meiobenthos have mostly concentrated on temperate regions (Higgins and Thiel, 1988; Bongers and Ferris, 1999; Cryer *et al.*, 2002; Ingels *et al.*, 2009; Williams *et al.*, 2010; Dannheim *et al.*, 2014) focusing on various benthic environments and depths. In the Western Indian Ocean (WIO) region, literature on meiobenthos is very scanty or limited to bays (Annapurna *et al.,* 2015). No published work is available on the characterization of inter-estuary meiobenthos specifically on the rivers Sabaki and Tana on the north coast of Kenya except for a few studies in tropical habitats such as the tidal mangrove forests on the south coast of Kenya (Alongi, 1987a, b; Mutua *et al.*, 2013). Vanhove *et al.* (1992) described a total of 17 meiobenthic taxa from Gazi Bay on the south coast with nematodes accounting for 95 % of the total densities, and copepoda, turbellaria, oligochaeta, polychaeta, ostracoda and rotifera recording decreasing densities in that order.

The present study is therefore the first of its kind to describe the inter-estuary meiofaunal community structure, their vertical distribution and the influence of salinity gradient on taxa composition and diversity. The study also emphasizes the role of sediment characteristics on meiobenthic densities.

Materials and methods

The study area

The Sabaki (Kenya's second longest river) estuary is the point of entry of the river into the Indian Ocean. It is located on the north coast of Kenya at 03°09'S and 40°08'E, at distance of 5 km north of Malindi town (Figure 1). It is a relatively small estuary covering an area of about 6 km² and consists of sand and mud banks, dunes and seasonal and permanent freshwater pools, mangroves and scrubs (McClanahan, 1988; Marwick *et al.,* 2014). The river drains a basin area of approximately 70,000 km² where extensive human activities are undertaken including livestock keeping, growing of drought resistant crops, irrigated horticulture, fisheries and sand harvesting. The river rises at 1° 42' S as River Athi and empties into the Indian Ocean as River Sabaki. River Tana (2° 35′ 56.42″ S, 40° 20′ 19.04″ E), Kenya's longest river (with an estuarine area of 27 km²) drains into the Indian Ocean at Formosa Bay, Kipini, from its headwaters in the Aberdare Ranges and Mount Kenya region (Manyenze *et al.*, 2021). The river discharge varies with the season. During the southeast monsoon (SEM) season the river discharge is higher at 750 m^3s^{-1} and lower during the northeast monsoon (NEM) at 350 m^3s^{-1} (Kitheka *et al.,* 2005). Higher discharge occurs during the rainy SEM season in the months of May and November. This consequently results in daily variations in sediment load from 2,796 tons/day during the dry NEM season to 24,322 tons/day during the rainy SEM season (Kitheka *et al.,* 2005). Annually, the Tana estuary records a sediment load of 6.8×10^6 tons, though this is slightly lower than that recorded before the construction of the upper Tana Basin dams (Kitheka *et al.,* 2005). Numerous anthropogenic activities contribute to the structuring of meiobenthic biodiversity downstream.

Field sample collection and treatment

Sampling was carried out twice (14th and 15th) monthly for October and November 2016 (NEM season) and on 27th and 28th for May and June 2017 (SEM season). For each sampling site, three independent replicate sediment samples were collected across the salinity gradient (i.e., stations A, B & C) at each river estuary using a Plexiglas® corer tube (6.5 cm inner diameter) that was softly and slowly pushed into the sediment by hand up to a depth of 15 cm. Each sediment core obtained was divided into 2 halves longitudinally. One half was then sliced into three layers: 0 – 5 cm, 5 – 10 cm and 10 - 15 cm and taken for analysis of vertical distribution of meiobenthos. These samples were immediately treated with 70 % ethanol and taken for further laboratory analyses. The other longitudinal half samples were taken for the analysis of total organic carbon (TOC) and granulometry under refrigerated conditions in the laboratory. Processing of both meiobenthos, TOC and grain size sediment samples followed the procedures described in Heip *et al*. (1985), Higgins and Thiel (1988), EPA (2001), and Foti *et al*. (2014). Sediment temperature and salinity were measured *in situ*. Temperature was measured using the field thermometer (analogical thermometer, 0.1°C) whereas salinity was measured using a field hand-held refractometer (0 – 35 ‰).

The supernatant was carefully washed and rinsed to remove $MgSO_4$, after which rose Bengal was added to stain the organisms for 24 hours. Meiobenthos were extracted and stored in 70 % ethanol and were then identified, counted and classified at higher taxa using a binocular microscope (Leica S6E stereomicroscope, x50 magnification) following the Higgins and Thiel (1988) protocol. Meiobenthic taxa diversity and composition was analyzed by river estuary (Sabaki or Tana), salinity gradient and season.

Figure 1. Map of the study area showing the location of the estuaries of rivers Sabaki and Tana and sampling stations indicated in blue, red and green circles along a salinity gradient.

Meiobenthic analysis

Sediment samples were washed through a top 1,000 µm sieve and a bottom 38 µm sieve. The collected fraction was put in a centrifugation tube (Heip *et al*., 1985; Danovaro *et al.,* 2004) in which magnesium sulphate ($MgSO_4$) with specific density of 1.28 g/cm^3 was added and centrifuged three times at 6,000 rpm for 10 minutes. For every centrifugation cycle, the supernatant was retained and collected in a 38 µm mesh sieve.

Granulometric and total organic carbon (TOC) analysis

Refrigerated sediment samples were analyzed for both granulometry and TOC in the laboratory. Grain size (range 0.04 – 1600 mm) was determined following the

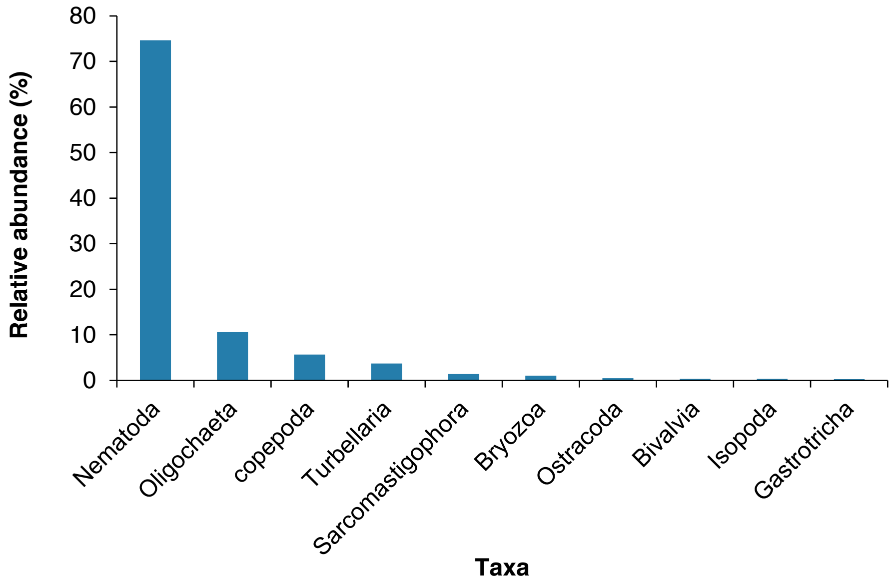

Figure 2. Relative abundance (%) of the ten most abundant meiobenthic taxa sampled during the study period for Tana estuary.

procedures described by Buchanan and Kain (1971), and the Wenthworth (1922) scale was applied to characterize sediment type. Thereafter, samples for TOC were put in an oven and dried at 90 °C for 24 hours to ensure moisture was removed. 5 g of the TOC sample was then taken from the oven and ashed at 600°C for six hours to attain ash free weight. TOC was determined as % of ashed content.

Data and statistical analyses

Descriptive statistics were conducted in Excel while multivariate analyses were performed using PRIMER v. 6.0 software and PERMANOVA+ for PRIMER (Clarke and Gorley, 2006; Anderson, 2005; Anderson *et al.*, 2008), and STATISTICA v.7.0 was used for all univariate analyses. Meiobenthic density was expressed as ind.10 cm^{-2}. For meiobenthos composition, data was initially standardized into relative abundance to minimize variations caused by the most abundant/dominant species and then similarities between pairs of their samples determined using the Bray-Curtis resemblance matrix (Clark and Warwick, 2001). A non-Metric Multidimensional Scaling (nMDS) technique was then used to visualize cluster

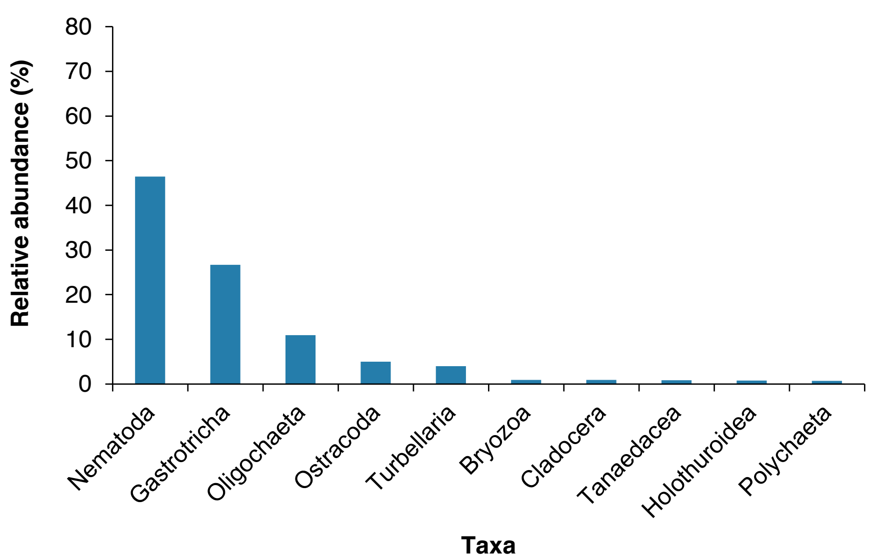

Figure 3. Relative abundance (%) of the ten most abundant meiobenthic taxa sampled during the study period for Sabaki estuary.

Table 1. Results of mean environmental variables for all stations sampled. Sediment grain size description (after Wentworth, 1922). SF = Sabaki fresh; SB = Sabaki brackish; SSA = Sabaki saline, TF = Tana fresh; TSA = Tana saline.

Season	Station	Clay%	Silt%	Very Fine Sand %	Fine Sand%	Medium Sand%	Coarse Sand%	Very Coarse Sand%	TOC%	Salinity (‰)	Temperature (°C)
SEM	SF-Station C	1.76	20.26	47.28	17.04	2.13	2.63	6.41	2.79	0.02	30.00
	SB-Station B	0.92	3.72	29.60	52.07	8.39	0.79	1.01	2.20	19.00	28.38
	SSA-Station A	0.02	2.60	39.08	45.54	6.43	0.88	1.52	0.48	30.33	27.11
	TF-Station C	2.16	52.96	31.06	2.80	1.81	2.79	2.78	4.22	0.01	26.11
	TB-Station B	1.18	11.11	43.53	13.16	4.27	8.26	10.98	6.39	9.86	26.06
	TSA-Station A	0.60	13.24	61.68	14.55	2.16	1.70	2.90	2.15	32.67	25.67
NEM	SF-Station C	9.95	47.00	20.03	6.14	7.08	4.48	1.70	1.15	0.03	31.15
	SB-Station B	7.64	36.59	15.36	12.27	13.56	8.06	2.30	2.27	17.50	29.00
	SSA-Station A	2.66	20.94	31.13	37.67	4.99	1.02	0.55	0.44	33.50	29.00
	TF-Station C	3.12	5.78	22.76	15.92	12.12	15.23	8.45	0.57	0.02	27.00
	TB-Station B	0.31	2.60	11.00	39.36	27.01	8.70	5.17	0.39	11.25	27.50
	TSA-Station A	0.01	1.16	16.82	57.75	20.72	1.93	0.66	0.76	34.30	28.15

(spatial differences in the composition of meiobenthic assemblages) groups by river estuary, salinity gradient, and season. Significant differences in the meiobenthic community composition between variables were tested by Analysis of Similarity (ANOSIM) for the nMDS assemblages. Complimentarily, a SIMPER analysis (cut-off of 50 %) was performed to unravel the percentage contribution of each taxon to the observed (dis)similarities between estuary, sampling station and season. Differences in taxa diversity (Shannon-Wiener diversity index) were tested using a 2-way ANOVA. Before the ANOVA test was performed, the normality and homoscedasticity of variances were checked using the Levene's test and accepted at $p > 0.05$. To meet the ANOVA assumptions, data were appropriately transformed and all significant differences were assigned at $p < 0.05$. Tukey HSD pair-wise comparison tests were performed for confirmation of differences between variables.

Results

Environmental factors

Sediment mean salinity varied across sampling stations between the two river estuaries. In the Sabaki, station C (upper estuary) recorded the lowest salinity of 0.02 ± 0.003 ‰, followed by Station B (19.00 ± 0.00 ‰) and the lower Station A (30.33 ± 0.83 ‰) had highest salinity level. In the Tana estuary, Station C recorded the lowest salinity of 0.01 ± 0.00 ‰, followed by Station B (9.86 ± 0.34 ‰) and the lower Station A (32.67 ± 0.44 ‰) had the highest salinity.

Sediment temperature varied between the estuaries and across sampling stations. For Sabaki, Station C recorded a highest mean of 30.0 ± 0.3 ^{0}C followed by Station B (28.4 ± 0.2 ^{0}C) and Station A (27.1 ± 0.2 ^{0}C). Sediment temperature across stations for the Tana estuary showed little variation where Station C recorded 26.1 ± 0.5 ^{0}C, Station B (26.1 ± 0.3 ^{0}C) and Station A (25.7 ± 0.2 ^{0}C). Silt and very fine sand proportions were higher in the sediment samples whereas TOC was generally higher during the SEM season (Table 1).

Meiofaunal composition

A total of 3,556 meiobenthic individuals belonging to 26 taxa were recorded. Overall, Nematoda was the most abundant taxon (62.1 %) followed by Gastrotricha (12.04 %) and Oligochaeta (10.74 %). The remaining 23 taxa recorded lower abundances of between 0.03 and 3.82 % (Table 2). By river estuary, Nematoda contributed 75 % in the Tana and 46 % in the Sabaki. Gastrotricha in Tana accounted for 0.4 % and 27 % in Sabaki (Figures 2 & 3). Oligochaeta on the other hand recorded 11 % in both the Tana and Sabaki. The rest of the 23 taxa were found in smaller abundances in both estuaries.

Densities of meiobenthos ranged from 1.0 ± 0.6 to 90 ± 42 and 1.2 ± 0.6 to 54 ± 42 ind.10 cm^{-2} for the NEM and SEM season, respectively (Figures 4 & 5). Meiobenthic total densities decreased with increase in sediment depth in both estuaries (Fig. 4 & 5) with highest densities recorded in the surface layers (0-5 cm), whereas the lowest density was recorded in the bottom-most layer (10-15 cm). The converse was true for taxa diversity in the aforementioned sediment depths. Generally, seasonal mean densities were higher in the

Table 2. Meiofauna densities (10 ind/cm²) of all the taxa sampled during the study period for Tana, Sabaki and Tana and Sabaki estuaries combined.

Meiobenthic Taxa	Sabaki	Tana	Sabaki&Tana	Meiobenthic Taxa	Sabaki	Tana	Sabaki&Tana
Polychaeta	0	0	0	Rotifera	1	0	1
Oligochaeta	2	18	10	Tardigrada	0	0	0
Nematoda	56	166	111	Bryozoa	3	4	4
Sarcomastigophora	0	0	0	Priapulida	0	0	0
Turbellaria	11	9	10	Aplacophora	0	1	1
Cumacea	0	0	0	Holothuroidea	2	0	1
Ostracoda	16	1	8	Cladocera	3	1	2
Copepoda	1	15	8	Insecta	1	0	1
Bivalve	0	0	0	Tunicata	0	0	0
Isopoda	0	0	0	Cnidaria	1	0	0
Tanaedacea	0	0	0	Laptoscala	0	0	0
Gastrotricha	0	1	1	Ciliophora	0	0	0
Amphipoda	0	1	0	Syncarida	0	0	0

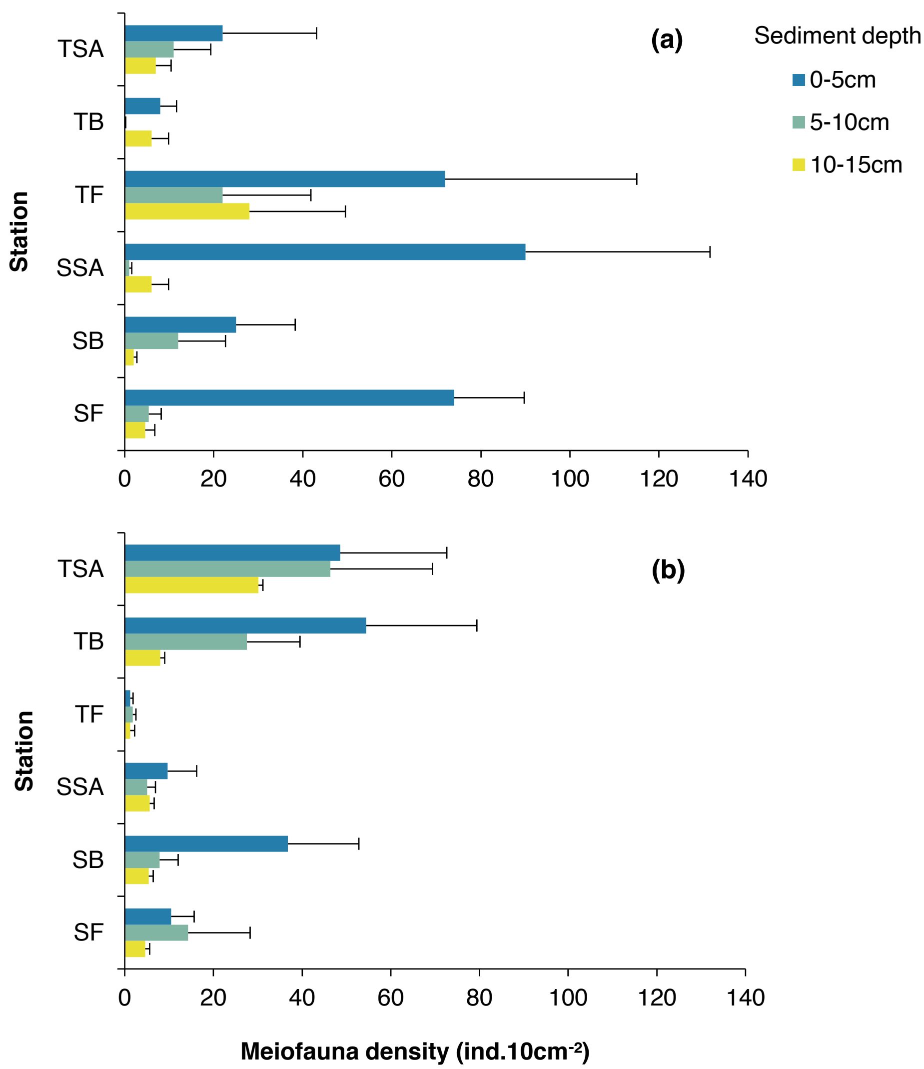

Figure 4. Mean ±SE meiobenthic density (ind/10cm²) distribution along the salinity gradient with sediment depths during the (a) north east monsoon and (b) south east monsoon season. TSA = Tana saline; TB = Tana brackish; TF = Tana fresh; SSA = Sabaki saline; SB = Sabaki brackish and SF = Sabaki fresh.

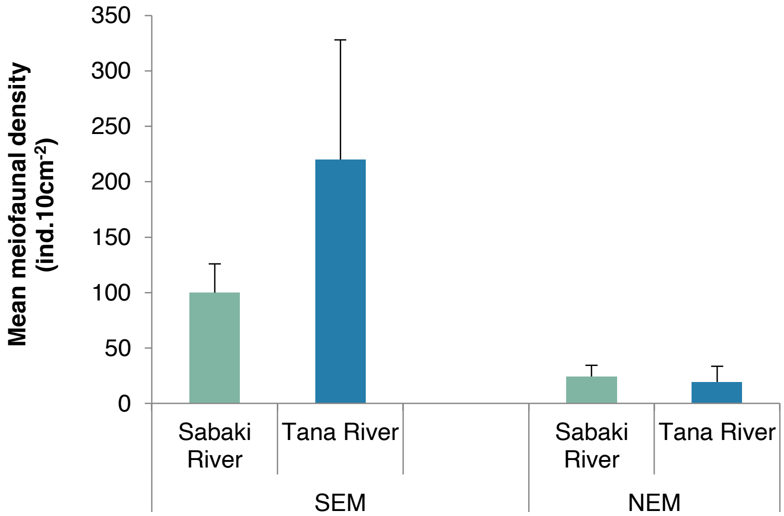

Figure 5. Comparison of mean meiobenthos density (ind/10cm^2) between estuaries with seasons. SEM = south east monsoon; NEM = north east monsoon.

Tana estuary (220 ± 108 ind. 10 cm^{-2}) compared to the Sabaki (100 ± 26 ind.10 cm^{-2}) during the SEM season. Results of non-Metric Multidimensional Scaling (nMDS) of the metazoan meiobenthic densities and structural composition showed distinct separation of meiobenthos composition between river estuaries with season combination (Figure 6). However, no distinct separation was observed for meiobenthos composition between river estuaries with habitat and with sediment depths. Results of the 1-way ANOSIM test confirmed a significant distinct meiobenthos composition between river estuaries with season combination ($R = 0.043$; $p = 0.004$). Results of the 1-way SIMPER analysis revealed that the dissimilarities observed in meiobenthos composition between river estuaries with season combination were attributed to more abundant Oligochaeta, Turbellaria, Ostracoda, Gastrotricha and Bivalvia (Table 2).

In terms of salinity gradient, meiobenthic densities were generally higher in Tana Station A, at 75 ind.10 cm^{-2} in the topmost (0-5 cm) sediment layer followed by Tana Station C and lastly Tana Station B. In the Sabaki, only surface layers recorded higher densities in all the stations with highest densities recorded in Station A followed by Station C and Station B in that order. In

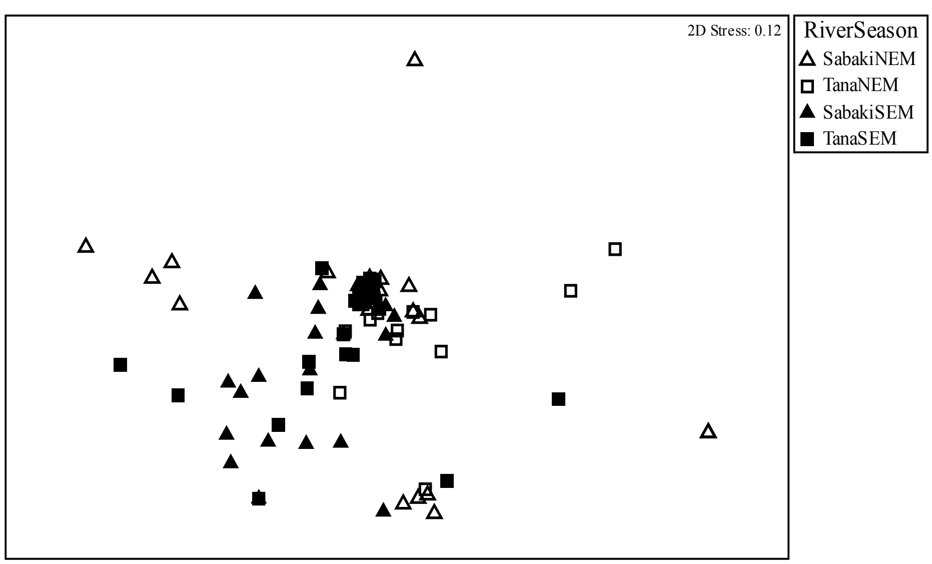

Figure 6. Non-metric multi-dimensional scaling plots on meiobenthos assemblages showing distinct clusters for both Sabaki and Tana rivers during southeast monsoon season.

Table 3. Results of 1-way SIMPER analysis: Sabaki river estuary showing meiobenthos taxa contributing to dissimilarity in terms of abundance (%) with river-season combination and an average dissimilarity of 73.20 %.

Taxa	Sabaki-NEM average abundance	Sabaki-SEM Average abundance	Av. Diss	% Contribution
Nematoda	50.65	43.81	22.6	30.86
Oligochaeta	14.37	6.65	9.21	12.57
Turbellaria	1.26	**16.03**	7.98	10.89
Ostracoda	0.00	**15.22**	7.61	10.39
Gastrotricha	14.92	0.00	7.46	10.19
Bivalvia	**8.74**	0.00	4.37	5.97
Polychaeta	4.36	0.79	2.5	3.42
Copepoda	3.10	0.62	1.79	2.45
Cladocera	0.00	3.18	1.59	2.17
Bryozoa	0.00	2.93	1.46	2.00

the same estuary the mid (5-10 cm) and lower (10-15 cm) sediment layers recorded low to moderate densities. Overall, both river estuaries showed higher densities in surface sediment layers designated as Station A for both estuaries (Fig. 4). Stations A and B of the Tana estuary recorded the highest mean meiobenthic densities in their surface sediment layers whereas station B in Sabaki recorded the highest density in the 0-5 cm sediment layer during the SEM season. Low to moderate densities were recorded in stations A and C along the Sabaki estuary (Figure 5). The nMDS plots revealed seasonal cluster separation for meiobenthic densities and taxa composition in both river estuaries with seasons (Figure 6).

Results of 1-way SIMPER analysis for the Sabaki meiobenthic taxa composition showed an average dissimilarity of 73.2 % between the seasons, with Nematoda (30.86 %) contributing most to the observed dissimilarities, followed by Oligochaeta (12.57 %), Turbellaria (10.89 %), Ostracoda (10.39 %), Gastrotricha (10.19 %) and Bivalvia (5.97 %) making up the meiobenthic genera responsible for the dissimilarities observed (Table 3). For the Tana estuary, 1-way SIMPER analysis for meiobenthos composition showed an average dissimilarity of 55.77 % between the seasons with Turbellaria (16.31 %) contributing the highest to the observed dissimilarities, followed by Copepoda (10.15 %), Isopoda (8.42 %) and Bryozoa (3.65 %) as the dominant meiobenthic taxa responsible for the observed dissimilarities (Table 4). By salinity gradient, Station B recorded the highest taxa diversity followed by Station C and A in that order (Figure 7). However, results of 2-way ANOVA indicated no significant difference in taxa diversity between river estuaries and across salinity gradient (df = 1; f = 0.018; p = 0.895 and df = 2; f = 1.837; p = 0.165, respectively). The same test indicated no significant effect due to estuary and station interaction (df = 2; f = 0.338; p = 0.714). By depth, lower sediment layers (10-15 cm) recorded the highest taxa diversity followed by the middle layers (5-10 cm) and surface layers (0-5 cm) (Figure 8). Results of the 2-way

Table 4. Results of 1-way SIMPER analysis: Tana river estuary showing meiobenthos taxa contributing to dissimilarity in terms of abundance (%) with river-season combination and an average dissimilarity of 55.77 %.

Taxa	Tana-NEM Average abundance	Tana-SEM Average abundance	Av.Diss	% Contribution
Nematoda	61.37	58.05	19.46	34.89
Turbellaria	8.44	14.41	9.1	16.31
Copepoda	4.55	8.39	5.66	10.15
Oligochaeta	7.73	4.38	5.27	9.46
Isopoda	**9.39**	0.00	4.70	8.42
Bryozoa	0.00	4.07	2.03	3.65
Bivalvia	3.32	0.00	1.66	2.97
Sarcomastigophora	3.24	0.00	1.62	2.91
Gastrotricha	0.00	3.11	1.56	2.79

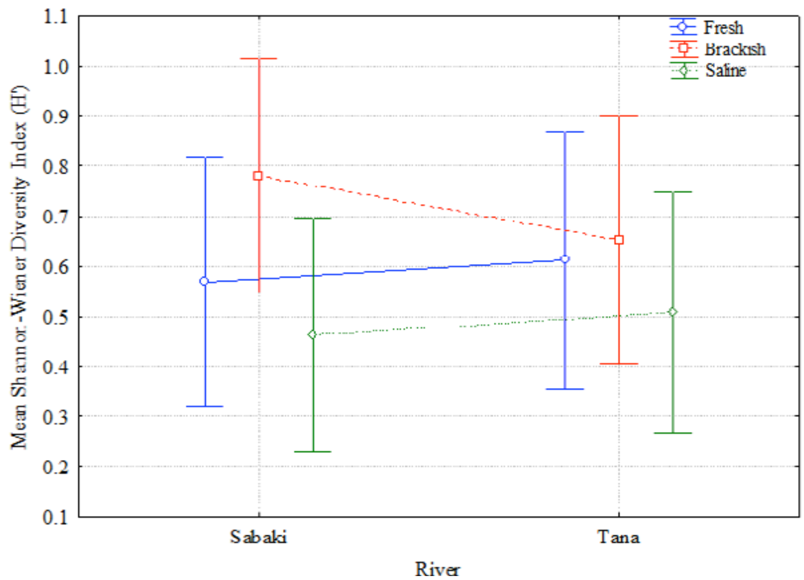

Figure 7. Mean ± SE Shannon-Wiener diversity index (H') of meiobenthos between rivers with habitat types sampled during the study period.

ANOVA test showed no significant difference in taxa diversity between river estuaries and across sediment depths (df = 1; f = 0.01; p = 0.922 and df = 2; f = 1.225; p = 0.299, respectively). The same test indicated no significant effect due to river estuary and sediment depth interaction (df = 2; f = 0.236; p = 0.791).

Discussion

Salinity gradient

Salinity gradient is a common phenomenon influencing the distribution and profiling of meiobenthic fauna. In the present study, salinity played a key role in community structuring for both fresh (Station C, brackish; Station B), and marine (Station A) habitats along the river estuaries as demonstrated by Montagna *et al.* (2002), Olafsson *et al.* (2000) and Merryl (2002). Salinity along the estuaries constantly keep changing with season and tidal influence. During the southeast monsoon (SEM) season, large volumes of fresh water with an influx of organic and inorganic materials enter the ocean resulting into reduced salinity levels in the estuarine ecosystem. Tidal movements also contribute to changes in salinity levels, where marine water pushes upstream during

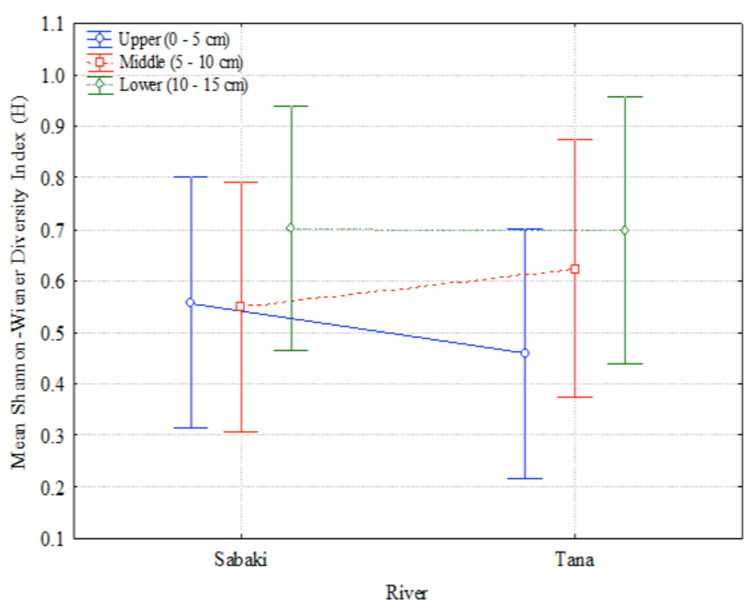

Figure 8. Mean ± SE Shannon-Wiener diversity index (H') of meiobenthos between rivers with sediment depths sampled during the study period.

high tide changing the salinity level (Olafsson *et al.* 2000). The Tana and Sabaki river estuaries exhibited this daily, and the seasonal dynamism in salinity levels influenced the diversity of taxa recorded.

Meiofaunal composition and structural assemblages

The Sabaki and Tana estuaries have continuously undergone upstream human pressure; particularly from agricultural activities, building and construction, mining and waste disposal, to mention a few. Determination of meiobenthic biodiversity trends from the present study provides a basis for describing their distribution along the salinity gradient. Meiofaunal structural composition and densities was aligned with the results from similar habitats across tropical zones (Guo *et al.,* 2008; Semprucci *et al.,* 2013; Costa *et al.,* 2016) and this study was the first of its kind in the WIO region. The results revealed relatively low densities and diversity in the two river estuaries. This observation can be accounted for by the fact that surface sediments (0-5cm) harbored higher abundance of meiobenthos with lower diversity whereas lower sediment depths (10-15 cm) harbored higher diversity with lower abundances (Vincx and Heip, 1987). Alongi and Pichon (1988) associated similar observations with inverse trend between meiobenthos abundance with depth. Vanhove *et al.* (1992) illustrated a declining pattern from the marine to freshwater habitat which is in accordance with the principle that abundance and diversity decreases from the marine zone towards the freshwater habitats. This scenario concurs with the study by Coull (1999) which further reveals that euryhaline estuarine species are rare, whereas euryhaline freshwater species do not exist. Alongi, (1987b) noted that physical characteristics, estuarine forest cover and productivity in addition to food availability determines meiobenthos community structure and densities.

In the findings from this study, it has been demonstrated that the WIO region does not have sufficient data on estuarine meiobenthos. It is therefore difficult to theorize on the elaborate mechanisms that shape their structure and composition. In fact, the current study established that salinity due to tidal action was the key factor in determining the community composition and structure (Fig. 4 & 5) which show the habitat prevalence of meiobenthos. More so, Annapurna *et al.* (2015) noted through a Canonical Correspondent Analysis (CCA) that community composition and structure was largely dependent on salinity and sediment texture. Other factors contributing to the observed patterns include seasonality, competition and predation, though the latter were not tested in this study. Tropical estuarine habitats incur severe physical stresses which can be reflected in the low numbers of species (5 to 13) living in these habitats. Similar trends have been recorded by others (Alongi, 1987b; Coull, 1999) citing low rates of organic matter deposition, speedy detritus utilization and enormous upstream to downstream disturbances as factors behind this scenario. In comparison, species richness and diversity across European and North American river mouths are much higher (Alongi, 1987a, b, c) than what the present study has revealed.

Nematodes were the most abundant meiobenthos in both river estuaries accounting for over 62.05 % of all the meiobenthic taxa identified. The other most dominant taxa were Gastrotricha, followed by Oligochaeta, Turbellaria, Copepoda, Ostracoda and Bryozoa in that order. These taxa are cosmopolitan with capabilities of being resilient to a wide range of environmental conditions (Alongi, 1987b; Ngo *et al.* 2013). This dominance pattern concurs with the structural assemblages for meiobenthic animals on the eastern African coast and other tropical estuaries (Vanhove *et al.*, 1992; Schrijvers *et al.*, 1997; Olaffson *et al.*, 2000; Mwonjoria, 2007). Vertical distribution of nematodes in sediment was biased with surface layers recording the highest densities where clay and silt dominated with a division of copepods occupying the medium and course sands. This finding agrees with that of Vanaverbeke *et al.* (2002), Mutua (2013) and Semprucci *et al.* (2013) on the ecology of nematodes and their preferred sediments to inhabit. De Troch *et al.* (2008) found that copepods preferred coarser and well oxygenated sediments. The current study yielded similar findings for both nematodes and copepods. Additionally, surface sediment layers possess higher total organic matter (TOM) which forms detritus and other food substances, thus supporting higher meiobenthos and especially high nematode densities (Mutua *et al.,* 2013).

Meiobenthic mean densities were higher in the Tana estuary during the SEM season as compared to the Sabaki, possibly due to enhanced riverine forest canopy in this estuary which implied that there was minimal environmental disturbance to meiobenthos (Mutua *et al.,* 2013). Additionally, this is associated with riverine productivity and hence food availability (Alongi, 1987b). The converse was true for the Sabaki estuary. Mutua *et al.,* (2013) further noted that sediment salinity and temperature increases with exposure, hence influencing the benthic microphytobenthos which form the primary food source for meiobenthos. This is indeed true and was confirmed by the present study where salinity and temperature values for the Sabaki estuary were higher compared to those of the Tana, hence moderate meiofauna densities and species diversity. This study, the first of its kind in estuarine meiobenthic community profiling on the east African coastline, has contributed to the body of scientific information on meiobenthic assemblages in these major river estuaries on north coast of Kenya. It has clearly shown that salinity gradient, coupled with temperature, sediment grain size and depth, TOC, and season control the community structure for meiofaunal assemblages which are known to be reliable biological indicators.

Conclusions

This study showed that meiobenthos densities and structural composition across the river estuaries were generally relatively low. The differences observed in densities and diversity could be attributed to the influence of salinity gradient, sediment depth and seasons. Unlike nematodes which are ubiquitous, all other meiofaunal groups identified clearly revealed that the variations in environmental factors and habitat heterogeneity in estuaries were responsible for the observed patterns. In the opinion of the authors, this implies that meiobenthos are good indicators of the environmental variations in river estuaries, though the relatively low densities and diversity signified high intensity stress levels both from river inflows and ocean tidal influences. It is recommended that similar studies are conducted across other estuarine systems within the WIO region to confirm the present findings. Including estuaries with minimal tidal actions may be necessary for comparison. Focused attention should be given to a taxon of interest such as the nematoda, which is not only useful for impact studies but also as a good indicator of habitat health.

Acknowledgements

Financial support was provided through the Marine Research Grant (MARG I) program of WIOMSA [MARG 1_2016_04]. We thank the Kenya Marine and Fisheries Research Institute for providing logistical field support and laboratory space during the entire project period. Much appreciation to Alfred Katana, Gladys Mwaka and Bernard Kimathi for their technical assistance both in the field and in the laboratory. Thanks to Mr Steven Odhiambo for designing the study area map. Finally, the authors are grateful to the anonymous reviewers for their valuable comments and suggestions which improved the quality of this publication.

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Patterns of fish community structure in protected and non-protected marine areas of mainland Tanzania

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Abstract

Information on the benefits of Marine Protected Areas (MPAs) for the condition of fish stocks is not well documented in Tanzania. Fish landing sites located in Tanga and Mtwara regions were surveyed to assess patterns of fish community structure; particularly fish abundance, species diversity, growth patterns, and maturity stages, based on catches landed at sites with different protection status. Fish abundance in the catch from protected areas was significantly lower than in non-protected areas *(p=0.002)*. Species diversity was relatively higher in catches from non-protected (H=2.742) compared to protected areas (H=2.232). A high percentage of species (63.24 %) exhibiting negative allometric growth was observed in catches from non-protected areas. Further, a large number of mature fish was observed in catches from protected areas compared to non-protected areas (p<0.01). These indices are useful indicators of the performance of MPAs. The observed negative allometric growth and reduced number of mature fishes in non-protected areas suggest that extractive pressure and disturbances from fishing gears have negative impacts on the fish stock. Continued high extraction may induce a decline in general fish size due to the constant selection for large-trait fish specimens, potentially causing evolutionally change in morphological traits. In contrast, the lower abundance and species diversity from the protected areas reflected low catch effort as a result of regulated fishing pressure in MPAs, rather than indicating the actual diversity in the fish stocks in these protected waters. Based on these findings it is recommended that more regulatory strategies are implemented in non-protected waters to allow more time for fish to attain appropriate harvest sizes and to ensure the effective protection of marine resources.

Keywords: fish growth, fish maturity, fish diversity, protected area, landing sites

Introduction

Coral reef fishes contribute significantly to tropical world fisheries (Jiddawi, 1997). While reef-associated fisheries officially make up about 10 % of global marine fishery landings, in some developing countries reef fish can contribute >35 % to national fisheries production (Jiddawi and Ohman, 2002). Fisheries resources are a vital source of food with high protein and omega 3 contents and make valuable economic contributions to the local communities involved in fishery activities, especially along the Tanzania coastline and its numerous islands (Jiddawi and Ohman, 2002; Tobey and Torell, 2006). About 95 % of fisheries in Tanzania are artisanal using traditional boats and gears such as dhows, outrigger canoes, nets, movable traps and fixed traps (Jiddawi and Ohman, 2002). The legal fishing gears which are encouraged to be use are indicated in the Fisheries Act no 22 of 2003 where the use of destructive methods is prohibited in both protected and non-protected areas. Fish caught in Tanzanian coastal waters are primarily traded in domestic markets, but the demand is increasing due to the increase in the human population (Kawarazuka *et al.*, 2017).

In Tanzania coral reef fishes form the basis of small-scale subsistence fisheries, often representing the

http://dx.doi.org/10.4314/wiojms.v21i1.7

major income for many coastal communities (Tobey and Torell, 2006). Reef fisheries are amongst the most important fisheries on the Tanzania mainland and provide a substantial part of the livelihood of coastal communities. However, reefs are highly subjected to human disturbances such as fishing (Muhando and Mohammed, 2002) undermining their sustainability and productivity potential. The impacts on coral reefs affect fish communities (Wagner, 2004) with considerable cascades on the economies of local coastal human populations. Reef fishes are highly targeted for consumption, and this pressure has secondary effects on the value of fish stocks (Rajasuriya *et al.,* 1998) by impacting on fish growth, maturation and fertilization (Wilson *et al*, 2006).

The contribution of the fisheries sector to economic development cannot be understated. The marine fishery industry contributes significant economic earnings to the gross domestic product (GDP) of several countries and a growing business worldwide (Sarpong, 2015). Lack of appropriate management practices have led to excessive fishing pressure and remarkable habitat degradation, among other environmental problems potentially undermining the economic viability of this sector. For example, the lucrative nature of the fishery business has, in most parts of the world, lead to over exploitation of the fishes (Mvula, 2009). Overfishing may cause the decline of fish stocks and increase risk of extinction of some fish species (Crowder *et al.*, 2008). These management and conservation challenges have led to many countries implementing control measures by delineating areas of marine waters where fishing activities are regulated. These areas, termed as Marine Protected Areas (MPAs), provide refuges and breeding havens for fish species and play an important role in replenishing otherwise depleted fisheries in areas affected by overfishing (Klein *et al.,* 2010). MPAs are viewed as important conservation areas similar to their terrestrial counterparts, yet they are under increasing pressure due to the increasing demand for fish protein from a rapidly growing human population worldwide (Kuboja, 2013).

MPAs in Tanzania were developed in the 1970s where several marine sites were established as marine reserves (Machumu and Yakupitiyage, 2013). The United Republic of Tanzania has improved the protection of marine resources by creating marine parks and marine reserve laws that allow the establishment of Marine Protected Areas (URT, 1994). Following these efforts, MPAs have contributed to an improvement in fish communities and increased resilience to anthropogenic disturbances (Alonso *et al.*, 2017). These areas are now recognized to be effective in providing refuges to fish populations (McClanahan and Arthur, 2001).

Ocean zoning provides a means for separating unsustainable human activities from marine resources as well as reducing conflicts among user groups (Crowder *et.al.,* 1994). Zoning aims to harmonize conflicting conservation and livelihood objectives by spatially separating extractive resource use areas from sensitive habitats (Lokina, 2005). Zoning has been useful in protecting critical species, species-rich habitats including sub-tidal areas, mangroves, forest, bird nesting, fish spawning as well as turtle breeding grounds. The existing forms of zoning in the MPAs of Tanzania include core zones, specified use zones and general-use zones (Kuboja, 2013). The levels of protection in these zones include core zones which provide the highest level of protection, also known as no take zones (Hamilton, 2012). Fishing and extractive activities without license are not allowed in MPAs. Other zones such as specified use zones act as buffers around marine parks, while multiple use zones allow fishing by resident fishers using traditional methods. Permitted fishing gears include hand-lines, basket-traps, pullnets (mesh size less than 2.5 mm), shark nets, set-nets (mesh size 2.5 mm-7 mm), while octopus collection and sport fishing is also allowed. These gears and fishing activities are only allowed in general use zone and specified use zone with special permits for residents and non-residents. The general use zones are intended to be used by residents in a sustainable manner, while extractive activities are allowed but only with permission from the park authorities (Kuboja, 2013). These areas also help to preserve coral reefs that provide breeding grounds to many fish species.

Despite these existing measures of protection there is still insufficient information from landing sites that provides a detailed picture on the status of marine resources in Tanzania's marine waters that differ in protection status. This study was conducted to fill this gap and to assess the fish community structure to determine the extent to which marine parks are effective in conserving fishery resources. Specifically, this research aimed at assessing the abundance and diversity of landed reef fishes, their growth patterns and maturity stages.

Materials and methods

Study area

Four landing sites in Mtwara and Tanga regions, Tanzania were selected. In each region, two sites were picked from unprotected waters and protected marine parks, respectively. The landing sites of Msimbati and Deep Sea are located within marine parks and the remaining two of Shangani and Moa are outside the marine parks. These areas were chosen based on protection status and ease of access (Sobo, 2004).

Msimbati landing site in Mtwara region is found within Mnazi Bay-Ruvuma Estuary Marine Park (MBREMP) located at 10° 00' 0.0" S, and 40° 00' 0" E. The park is located south of Mtwara town in southern Tanzania, stretching over 45 km of coastline from the headland of Ras Msanga Mkuu to the Ruvuma River that forms the border with Mozambique. The park covers an area of 650 km^2. The varied ecosystems of the park support a great diversity of marine life including mangrove forests (nursery grounds for many fish and crustacean species), sea grass beds (feeding ground for a number of marine species) and diverse coral reefs with approximately 250 species of hard coral, 400 species of fish, and 100 species of echinoderms (Obura, 2004). Shangani landing site is located at 10° 15' 56.880" S,

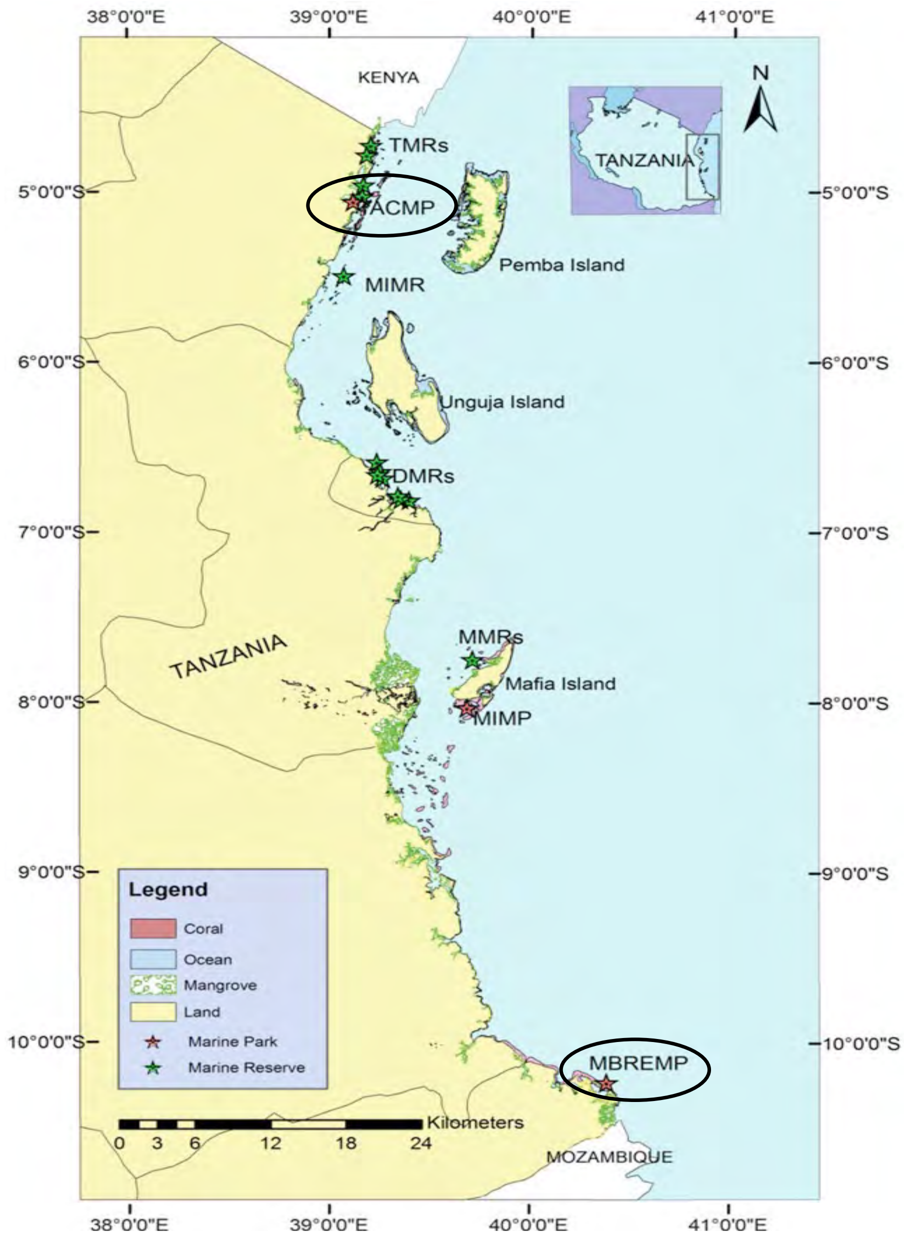

Figure 1. Map of Tanzanian coast showing the location of study areas. Source (Mangora *et al.*, 2014).

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and 40° 11' 4.560" E in the northern part of Mtwara region. It is one of the largest and busiest landing sites located on the Msanga Mkuu peninsula in Mtwara.

The third site was Deep Sea found within the Tanga Coelacanth Marine (TACMP) Park located at 8° 49′ 60 E and 5° 30′ 0″ S. The park extends for 100 km along the coastline from north of Pangani River estuary to Mafuriko village north of Tanga City. The park covers an area of about 552 km^2 of which 85 km^2 is terrestrial and 467 km^2 aquatic. The uniqueness of the park includes the occurrence and high rates of incidental catches of the CITES - listed and iconic Coelacanth, *Latimeria chalumnae*. TACMP is also home to other endangered species like dugongs, sea turtles, and migratory water- birds (Harrison, 2010).

The Moa unprotected landing site is located at 4°77′ 0″ S and 39° 15′ 0″ E in Mkinga district in the northern part of Tanga region. Mkinga coastal zone is rich in marine resources that include a variety of fishes, octopus, sea cucumber, spring lobster, prawns, sea crabs and seaweeds. A large part of the Mkinga coastline is covered with mangrove stands of considerable density (Harrison, 2010).

Data collection

Fish samples were collected for three days per month at landing sites during neap tides. The sampling was conducted during the Northern Monsoon period starting in October 2019 to March 2020, making a total of 18 fishing trips surveyed per site. A fishing vessel that utilized various fishing gears was used as a sampling unit. Soon after the fishing vessel had landed, the fishers were asked from which sites they had fished in order to record the fish samples in appropriate study sites based on protection status. The most encountered gears were traditional such as longlines, short handlines, box traps and pull nets (mesh size larger than 2.5 mm and smaller than 7 mm). Some of the fishing gears were regulated within MPAs; pull nets with mesh size of 2.5 mm and shark nets were only allowed within the general use zone. With the help of an expert in fish identification, the fish were selected and grouped based on their genera for further identification using a field guide (Lieske and Myers, 1994). The fish which were not identified in the field were photographed for further identification in the zoology laboratory at Sokoine University of Agriculture. Further, measurements of length and weight of each individual sampled fish were taken using a measuring board and weighing balance respectively. Fish maturity stages were assessed for a single selected species (*Lethrinus harak)*, which was selected because of its economic value in the local markets and availability across the coastline of Tanzania. The maturity stages were assessed by visual inspection of gonads after ventral dissection of the fish (Balci and Aktop, 2019). The modified five-point maturity scale (Burnett, 1989) based on the external appearance of the gonads was used to classify maturity stage. The features used to stage the gonads included, size, shape, colour, volume, and degree of vascularization and opacity in the ovary.

Length measurement was carried out by measuring and recording the standard length to the nearest centimeter. Weight was determined by measuring and recording the total body weight of the fish to the nearest gram. These data were used for assessing of reef fish size structure.

Data analysis

The relative abundance of species was calculated and the accumulation curves of abundance data from both protected and unprotected areas were visualized in Microsoft excel 2010. Further, the significant difference in fish abundance between protected and non-protected areas was tested using the Mann-Whitney test. The species diversity was calculated at the site level using the Shannon-Wiener diversity index. The data were then pooled from individual sites to the protected and non-protected site levels so that the assessment of the protection status on fishing grounds could be done.

The growth patterns of fish were calculated using the length-weight relationship through the regression equation $W = aL^b$, where W = weight (g), L = total length (cm), a = constant (intercept) and b = growth exponent (Thulasitha and Sivashanthini, 2012). The frequency of occurrence of growth patterns was plotted on a bar graph to assist with visualization. Furthermore, assessment of the significant difference of the growth patterns between the sites was tested using the student's t-test, after confirming normal distribution in the data using the Kolmogorov-Smirnov test. The Mann-Whitney test was used to compare fish maturity stages between protected and non-protected areas. A graph showing fish maturity stages against the sites was plotted to visualize the variation between the protection statuses. All the statistical analyses were performed in Paleontological Statistics (PAST 4.03).

Results

Fish composition and abundance

A total of 1548 fish samples from 50 species were examined. 6030 fish samples were collected from protected and 918 from non-protected areas. In Tanga region a total of 205 individual fish of 11 species, and 521 individual fish of 18 species were sampled from the protected and non-protected area landing sites, respectively. In Mtwara, the protected area and non-protected area landing sites provided a total of 415 fish of 10 species, and 503 fish of 13 species, repectively. Species composition of the catches was variable. The highest abundance within protected areas were *Lethrinus harak* (26.35 %, n=166), followed by *Siganus sutor* (21.43 %, n=135) and *Scarus ghoban* (13.02 %, n=82). Further, in non-protected areas the highest abundant species were *Lethrinus borbonicus* (21.35 %, n=196), followed by *Ctenochaetus striatus* (17.32 %, n=159) and *Scarus niger* (10.46 %, n=96) (Fig. 2). There was a significant difference in abundance of species between protected and non-protected areas (Mann-Whitney test, U=815.5, p=0.002). The accumulation curves for protected and unprotected sites showed increasing patterns indicating more sampling was needed to reach an asymptote (Fig. 3).

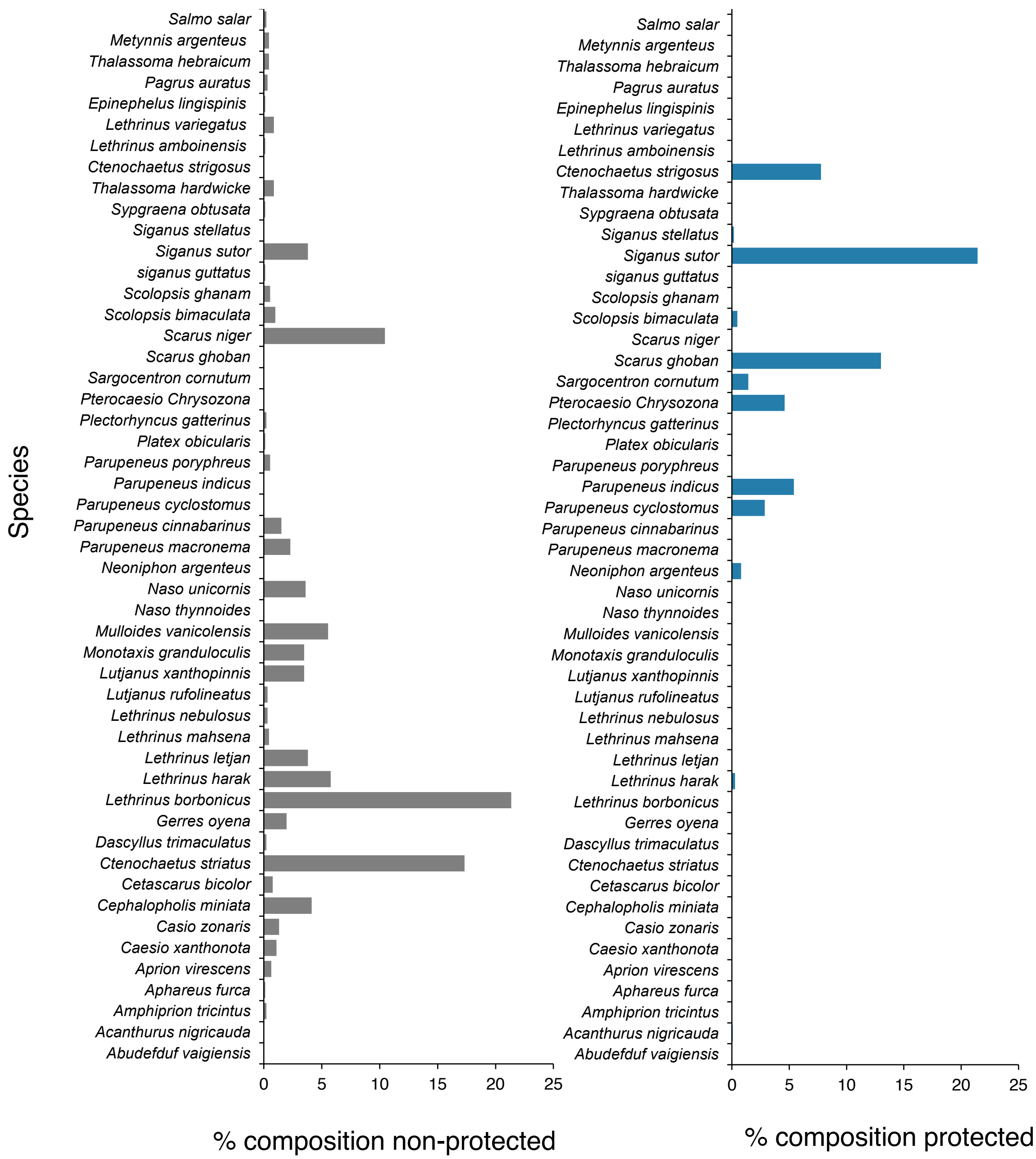

Figure 2. Composition and relative abundance of species in protected and non-protected sites.

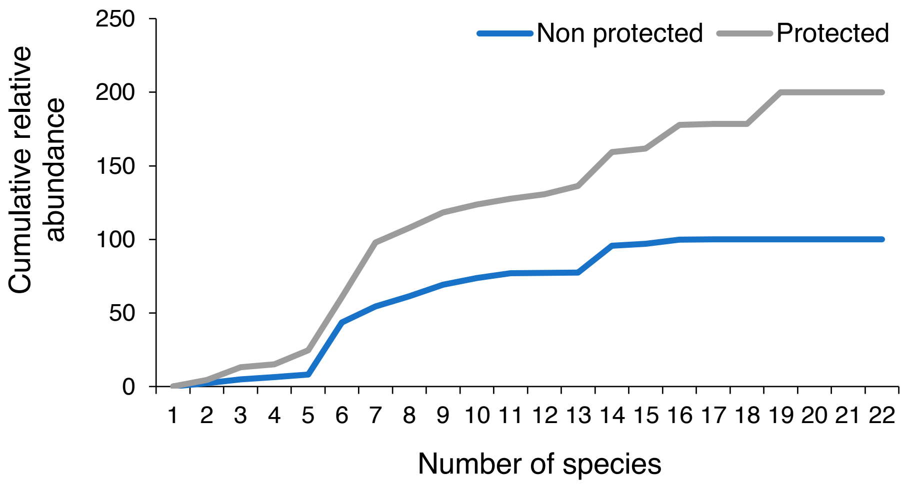

Figure 3. Cumulative relative abundance of landed fishes from protected and non-protected sites indicating more abundant catches in the former.

Fish species diversity

The diversity of landed species varied with protection status. In Mtwara the non-protected area had higher species diversity than the protected area (Fig. 4). A similar pattern was observed in Tanga region. Overall, the catch of fish landed from protected areas had lower species diversity than from non-protected sites.

Growth patterns and size distribution of landed fish

The growth patterns of collected samples in relation to protection status were analyzed allometrically. The majority (63.24 %) of species sampled exhibited a negative allometric growth form (Fig. 4). The percentage of growth types from fish landed from protected areas were variable: 70.83 % exhibited negative allometric growth; 12.5 % positive allometric growth; and 16.67 % isometric growth. Further, non-protected areas showed varying growth patterns: 56.82 % negative allometric growth; 25 % positive allometric growth; and 18.18 % isometric growth (Fig. 6).

Siganus sutor and *Lethrinus harak* were common in catches from both protected and non-protected sites. The mean size of species was higher in fishes landed from protected and lower from non-protected sites (Fig. 5). There was no significant difference in the counts of growth patterns within the protected areas of Tanga and Mtwara ($p > 0.05$).

Fish maturity status

The sampled fishes were grouped under five maturity stages (Table 1). The fishes had variable maturity stages across the protected and non-protected sites. Individuals in stages II, III, and IV (developing, maturing, and ripe) were dominant across both sites.

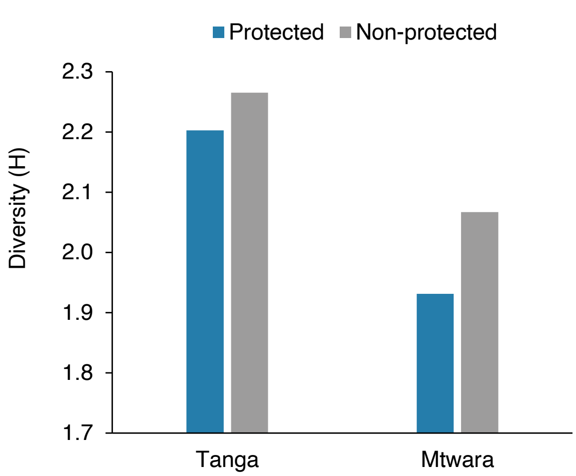

Figure 4. Shannon-diversity index of protected and non-protected areas in Mtwara and Tanga regions.

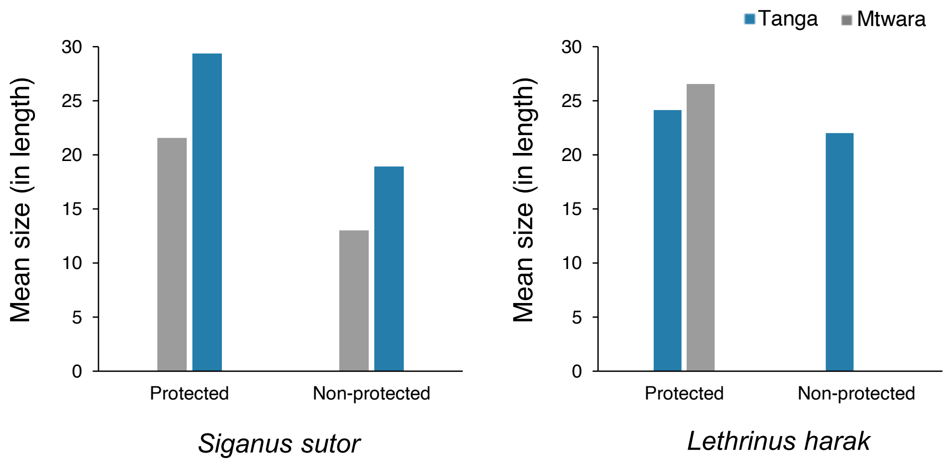

Figure 5. Mean size and size structure of *Siganus sutor* and *Lethrinus harak* from Tanga (blue) and Mtwara (grey) regions in protected and non-protected areas.

In Tanga region, 59 specimens of *Lethrinus harak* were examined from the landing site within the protected area (25 males, 35 females), while it was not possible to observe the gonads of specimens from the non-protected landing site as they were gutted at sea. A total of 107 individuals were collected from the Mtwara region landing site within the protected area (42 males. 65 females). In the non-protected landing site 49 individuals collected (28 males, 21 females). There was a significant difference in fish maturity stage (I-V) of females within protected and non-protected areas (Mann-Whitney, $U=0$, $p<0.01$). However, no significant difference was observed in counts of maturity stages of males between the sites (Mann-Whitney, $U = 4$, $p = 0.09$). Further, there was a higher number of females in catches at landing sites from protected areas (59.88 %, n=100) as compared to non-protected areas (40.12 %, n=21) ( ).

Discussion

This study aimed to assess the patterns of fish community structure in marine protected and non-protected areas of the Tanzania mainland. Significant differences were observed in species abundance, growth patterns, and female maturity stages. No significant differences were observed in the maturity stages of males between protected and non-protected areas. Also, maturity stages showed that there was a higher number of mature specimens from landing sites within protected areas as compared to those from non-protected areas (Fig. 7).

The high diversity and abundance at landing sites within non-protected areas could be influenced by differences in protection status and restrictions existing to regulate fishing. Fishers in marine parks are controlled by the park regulations and they are limited to using gears which are very selective, therefore they obtain small catches compared to the landing

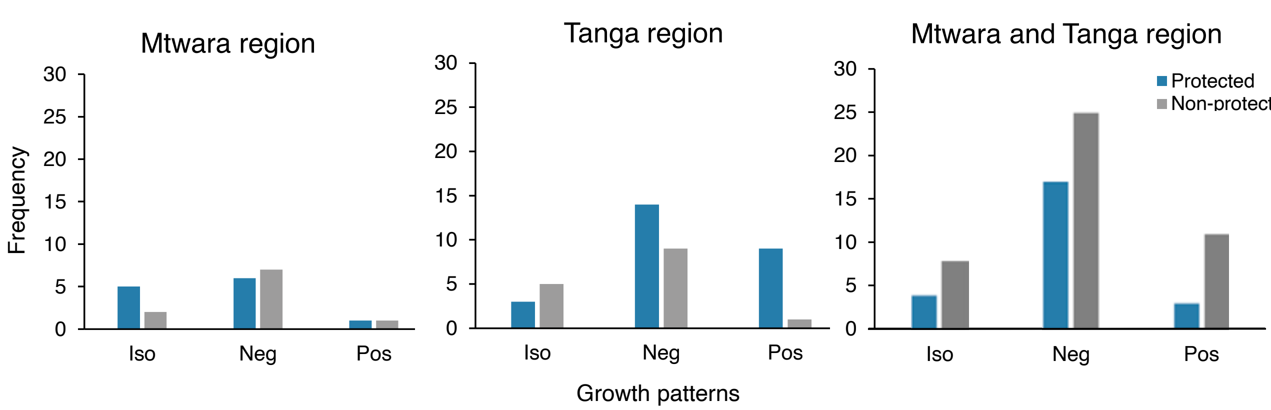

Figure 6. Frequency of growth patterns in different regions. Isometric (Iso), Negative allometric (Neg), Positive allometric (Pos).

Table 1. Fish maturity stages and their descriptions.

Maturity stage	Criteria		Description
	Male	Female	
I (Immature)			Males: testes very small and undeveloped; pinkish color Females: ovary small; light pink jelly
II (Developing)			Males: testes opaque with lobed or wavy appearance; color variable from red or pink to gray or white; milt may or may not be present in small amounts Females: ovaries opaque and enlarged with blood vessels becoming prominent
III (Maturing)			There is a further increase in the weight and volume of the ovaries, which have a deep yellow colour and occupy 2/3 to ¾ of the body cavity. Vascular supply increases and the blood capillaries become conspicuous
IV (Ripe)			Male: testes large, milt flows freely from testes Female: The ovaries are further enlarged occupying almost the entire body cavity. They are turgid, deep yellow, the blood supply increases considerably
V (Spent)			Males: testes emptying somewhat, still white Females: The ovaries are flaccid, shrunken and sac-like reduced in volume and have a dull colour. The vascular supply is reduced

sites outside marine parks where there are less restrictions (Tuda, 2018). Various studies have found higher fish diversity in MPAs as compared to non-protected areas (Sarkar *et al.,* 2013; Sweke *et al.,* 2013; Aller *et al.*, 2017). These studies contradict the findings reported here. The difference could be because of the sampling methods used; either collecting or counting fish directly in the fishing area, or from landed catches, as was the case in the present study. Further, the variation observed between the current study and other studies could be influenced by differences in the management of the resources (Sweke *et al.*, 2013). Restrictions on extractive activities within MPAs could limit the size of catches, which determined the sample size. Alternatively, the lower species abundance and diversity from catches within protected areas could be an

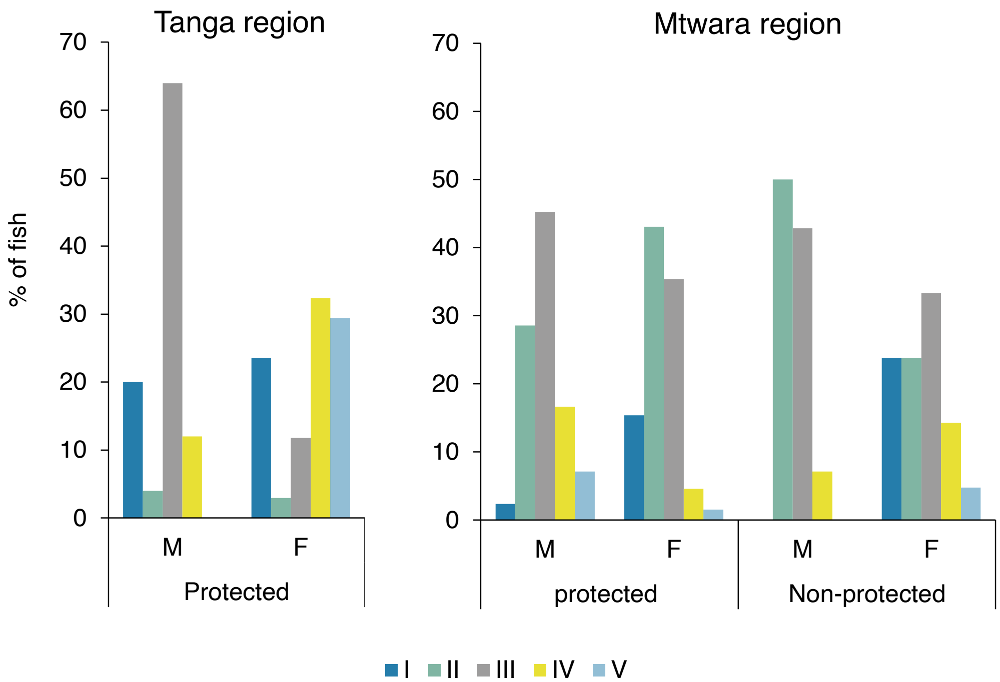

Figure 7. Maturity stages of *Lethrinus harak* in protected and non-protected sites from Tanga and Mtwara region. Fish maturity stages: immature (I); developing (II); maturing (III); ripe (IV); and spent (V). Fish sex: Male (M) and Female (F).

indicator of less exploitation and proper management practices which reduces fishing pressure (Silvano *et al.*, 2009; Samoilys *et al.,*2007).

The length and weight relationship helps to provide information on the wellbeing and growth of fish (Jisr *et al.*, 2018). This study indicated that catches at all landing sites, irrespective of their protection status, displayed a majority of reef fishes (63.24 %) exhibiting negative allometric growth. The high percentage of negative allometric growth from non-protected areas could be an indicator of overfishing and unregulated fishing activities (Jisr *et al.,* 2011). The rapid increase in fishing pressure over time on the Tanzanian coast could be the cause of non-proportional fish growth and could be considered as a clear sign of overfishing (Anderson *et al.*, 2008). Unrestricted fishing activities in non-protected marine areas may create disturbances which may interfere with normal proportional fish growth. In negative allometric growth, the fish becomes slender as it increases in length (Mazumder *et al.*, 2016). It is assumed that less disturbance favours healthy fish growth rates, thus the negative allometric growth observed in the majority of species could be influenced by disturbances caused by overfishing. Protected areas provided few individuals with negative allometric growth and their mean sizes were also higher than in non-protected areas.

Size structure is a critical component in maintaining the reproductive stability of fish populations as large individuals tend to produce many and high-quality eggs (Hsieh *et al.*, 2010). Size structure may be altered by several causes including fishing. The presence of large sized fish within protected areas may indicate the effectiveness of fishing regulations. The restrictions within protected areas help to create less disturbed environments for fish growth. Proportional growth is therefore expected to be higher within protected areas. (Magnussen, 2007). Experiences from MPAs show that limitations in fishing effort helps the fish stocks to recover, as individuals are able grow and mature in less disturbed environments (Hoof and Klaan, 2017).

Determination of sex and maturation provides an understanding of the reproductive biology of a species. The high percentage (78.57 %) of mature females from landing sites within protected areas may indicate limited fishing pressure which allows fish to mature in less disturbed areas (Wells *et al.*, 2012). Most species of fish mature when they reach 65-80 % of their maximum size (Beverton and Holt, 1959). The average length at maturity for *Lethrinus harak* is 21.5 cm in males and 22.5 cm in females (Badr *et al.,* 2019). The high number of mature females within protected areas is likely to be linked to their larger average sizes.

The restriction on type of gears within protected areas may greatly influence the sizes of female fishes in catches.

A higher female to male sex ratio increases fertilization success and productivity (Maskill *et al.*, 2017). The high percentage of mature females from catches within protected areas may indicate healthier productivity in the fish community, while the lower number of mature fish in non-protected areas suggests unregulated fishing pressure may lead to lower productivity.

Acknowledgements
This study was part of the MSc dissertation of Fausta G Sallema and was supported by grants from the World Bank/Southwest Indian Ocean Fisheries Governance and Shared Growth Program projects. We thank the Department of Wildlife Management at the Sokoine University of Agriculture for providing laboratory space and Madam Gladys for field assistance throughout this study.

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The influence of household human and social capitals on participation in agricultural development initiatives in the coastal region of Kenya

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Abstract
The present study sought to establish the human and social capital that determines rural households' participation in agricultural projects and programmes implemented by the Kenyan government and development partners. The research was carried out among rural households in the three counties of the coastal region of Kenya. Multi-stage sampling techniques (purposive, proportionate random and simple random sampling) were used to select the study area and the sample. Data were collected using a semi-structured questionnaire, Focus Group Discussion and observation schedules. The data analysis was carried out using descriptive statistics and regression analysis in Statistical Package for Social Sciences Version 22. Individuals with human capital; namely age (-0.15), primary education (-0.16), secondary education (-0.14), vocational training (0.35), and on the of job training (0.25), had a higher likelihood of participating in agricultural development initiatives. Households with the social capital of membership to groups (0.51), engaged in economic activities (0.53) and with linkages with development agencies (0.44) had a higher likelihood of participating in development initiatives. Key policy recommendations for county government and development partners include: encouraging community members to enrol in adult education; providing support for vocational and technical training; registering as members in existing groups or forming groups based on common interest, and engaging in economic activities. The county governments should enhance advisory services to ensure close contact with professionals who will facilitate training, meetings and interactions with groups leading to the empowerment of members.

Keywords: rural household, participation, human capital, social capital, agricultural development initiatives

Introduction
Rural household participation has been recognised by many international development agencies as a vital component for sustainable development (Cornwall, 2009). The concept of participation originated about 40 years ago from the community development movement of the late colonial era in parts of Africa and Asia. The concept has been recognized as an important element since the early 1990s as a means of improving local welfare, training people in local administration and extending government control through local self-help activities (Ayman, 2011; McCommon, 1993). Today, participation has developed as one of the major models of development gaining acceptance across the spectrum of development actors as a means of improving development practice related to grassroots community development initiatives and viewed as a basis for project success (Cornwall, 2009). In recognition of its vital role in community development, participation has been referred to as the heart that pumps the community life blood (Reid, 2000).

The concept of participation came to be popularised and institutionalised in the 1990s as a novel, common-sense way of addressing development discourses and practices of many mainstream development

http://dx.doi.org/10.4314/wiojms.v21i1.8

organizations. It has earned its status as an orthodoxy with promises of giving 'the poor' a voice and a choice in development, and an essential ingredient in getting development interventions and policies right (Cornwall, 2006). Participation is commonly understood as the collective involvement of local people in assessing their needs and organizing strategies to meet these needs in partnership with the national government, county government, local organizations and external development partners. (Zaku and Lysack, 1998 cited in Cuthill, 2010).

Extension practitioners seek to institute participatory practice in development initiatives implemented among rural households due to the benefits associated with this approach (Chambers, 1994). The benefits include: a) enhancement of the relevance of programmes to ensure that they are all suited for the needs and circumstances of the beneficiaries (Kironde and Kihirimbanyi, 2002 cited in Cuthill, 2010); b) ensuring that the views of many stakeholder groups are represented in the development process (Cullen *et al.*, 2011); c) expectations that the programmes decisions that feed on the insights of many stakeholders are not just relevant to the beneficiaries, and that they are generally smarter (Weaver and Cousin, 2004; Cullen *et al.*, 2011); d) greater programme outcomes such as greater access to social services (Bedelu *et al.*, 2007), consumption and demand for services (Kilpatrick *et al.*, 2009); e) ensuring programme sustainability due to a greater sense of ownership and responsibility by stakeholders. Their participation enables them to be willing to mobilize and commit local resources for continuity of some or all of the programmes proceeds after external support is withdrawn or reduced (Oakley, 1989).

In an attempt to understand effective community participation in development initiatives implemented by government and development partners, either on their own or in partnership, to attain the benefits associated with it, there is a crucial need for research to identify factors contributing to successful participatory practices in implementation of agricultural projects and programmes (Chambers, 1994). For instance, Bauma *et al.* (2000) argues that the level of participation in social and civic community life is significantly influenced by individual socio-economic status and other demographic characteristics. Supporting this line of thought, Plummer (2002) describes factors such as skills and knowledge, employment, cultural beliefs, gender, education and literacy, social and political marginalization to be key in affecting community participation. Recent research on community participation in development has broadened focus and included community capitals including human, social and institutional factors and the interaction among these components of the community (Cote, 2001, cited in Cuthill, 2010). A theoretical analysis of community participation by Nkwake, Trandafili and Hughey (2013) revealed that communities have seven types of capital which influence community or individual participation in development initiatives. Community capitals include cultural capital, social capital, human capital, built capital, natural capital and political capital. Assessing levels of community capital is an effective way of measuring community capacity to participate in development initiatives for change (Flora and Flora, 2008). It is important to examine the extent to which the community capital influences community participation in development initiatives among households.

Human capital includes characteristics of individuals that strengthen one's ability to earn a living and provide for one's community, family and self-improvement (Cadil et al., 2014). It consists of one's personal assets such as health of the individual, formal education, skills, intelligence, leadership and talents (Flora and Flora, 2008). While human capital consists of a variety of personal assets, Becker (2002) states that human capital which includes schooling, on-the-job training, health information and research, is the most important form of capital in successful economies which depend on how extensively and effectively people invest in themselves. Becker (2002) asserts that human capital stimulates technological innovations and the high-tech sector and identifies education and training as the most essential forms of human capital which are associated with individual occupation. In their theoretical analysis of the scientific literature Ciutiene and Railaite (2014) conclude that human capital includes a wide range of different components such as knowledge, experience, competency, and health among others which are necessary for achieving development.

While there are many definitions of social capital, Fine (2001) defines social capital as the development of networks in which community residents can identify problems, share information, and implement strategies designed to solve problems for the benefit of all. Putnam (1993) defines social capital as features of social organizations such as networks, norms and trust that improve performance of a society by facilitating coordination of actions for mutual benefits. Social capital is manifested in the relations among people (Coleman, 1988). According to Coleman (1988), social capital resides in people's minds and influences their relationships with each other or plans to interact and may produce potential benefits, advantages and preferential treatment from another person or group beyond that expected in an exchanged relationship. Narayan and Pritchett (1997, cited in Lindon *et al.* (2002) and Heller (1996) argue that increased social capital leads to increased community cooperative action and solves local community property problems and economic development, strengthens linkages among individuals that speed up the diffusion of innovations, quantity and quality of information, reduces transaction costs, pools risks and allows households to pursue more risky and higher return activities. Social capital falls within two contexts of economic development policy. The first is bottom-up development, and depends on intra-community ties which is referred to as integration, and extra-community networks referred to as linkages. The other is top-down development which involves state-society relations referred to as synergy, and institutional coherence, competence and capacity which are called organizational integrity (Woolcock, 1998). In other words, social capital is inherent in individuals and interactions with others.

In Kenya today, participation of the community is mostly ensured through group structures such Community Based Organizations (CBOs), Common Interest Groups (CIGs), and Faith Based Groups (FBGs), which according to Putnam (1993, cited in Cuthill, 2010) are social capital specifically formed for the purposes of achieving common good projects (Hassan et al., 2018; Ong'ayo *et al.*, 2017) and which are among the growing mechanisms for channelling development assistance (Khwaja, 2004). The groups have served as instruments for consultation with supposed beneficiaries about planning and implementation of community projects (Hassan *et al.*, 2018; Ong'ayo et al., 2017). The groups are formed on the basis of interest and for the purpose of sharing of technologies and information on new innovations, networking, forming linkages with other likeminded individuals, groups and professional. The viability of the groups is determined by both the acquired and inherent knowledge, skills and experience in the individual (Ong'ayo et al., 2017) Participation is strengthened by both inherent and acquired individual ability and anticipated gains which include literacy levels, gender, skills, knowledge, and training (Flora and Flora, 2008).

The Kenyan government, both at national and county level, and development agencies have implemented various development initiatives at the coastal region with the goal of alleviating poverty among rural households. Despite the implementation of many projects and programmes, coastal Kenya is the least developed region of the country with more than 62 % living below the poverty line with a poverty index being over 70 (World Bank, 2016). Many development initiatives have been implemented with a focus on ensuring community participation for empowerment. These projects include the Kenya Coastal Development Project (KCDP), Hazina Ya Maendeleo ya Pwani sub-component of KCDP, Health Service Project (HSP) funded by the Danish Development Agency (Danida), Agricultural Sector Projects (ASP) funded by the Kenyan Government in collaboration with development partners, Regional Water Development Projects, United Nation Development Programmes (UNDP) among others (Danida, 2004).

Objective of the study

The study was guided by the following specific objectives: To identify the human and social capital of the households, and to establish the extent the two forms of capitals determine rural household participation in agricultural development initiatives implemented among them by the government and development partners and organizations.

Methodology

The study was carried out in three counties in the coastal region of Kenya (Tana River, Kwale and Kilifi). The climate of the region varies with distance from the coast and it becomes drier towards the inland from the ocean and from south to north (Nicholson et al., 1999). Covering an area of approximately 83,000km^2, the coast region has a population of approximately 3.3 million people with a birth rate of 3 % (Government of Kenya, 2009). About 69.7 % percent of the coastal population lives below the poverty line, with some areas such as Ganze in Kilifi scoring an alarming 84 %, making it the second poorest region of Kenya's eight regions after the North Eastern region with 73.9 % (Government of Kenya, 2013).

The accessed population was the 2,160 community members drawn from households that participated in different development initiatives implemented in the region by the government either on its own or in partnership with development partners.

The study used a combination of simple random sampling, proportionate random sampling, and purposive sampling techniques. First, simple random sampling was used to select three counties since participatory approaches have been used for implementation of development initiatives in all the six counties. Purposive sampling was then used to select three sub-counties. Two hundred and eighty six households were proportionately sampled from the three counties using a sampling frame obtained from the respective County Population Coordinators as shown in Table 1. According to Kathuri and Pals (1993), a sample of 100 respondents or more is appropriate for a survey study. This is large enough for data collection. With a large sample, the researcher is confident that if another sample of the same size were to be drawn, findings from the two samples would be similar to a high degree (Bordens and Abbort, 2008). A sampling frame for households from the selected sub-counties was obtained and arrangements made on when to visit the field and administer the questionnaire to the selected household heads.

For successful data collection in the field, one set each of a semi-structured questionnaire, and a Focus Group Discussion schedule were used. The questionnaire was administered to households to collect the personal profile of the respondents which included, age, sex, education level, socio economic diversification and social engagements such as frequency of interaction with development professionals, and linkages with development agencies. Data collected were analysed using descriptive statistics including percentages and frequencies, and inferential statistics such as regression analysis with the help of the SPSS version 20.0 software. Regression analysis was used to determine the influence of human and social capital on household participation in development initiatives.

In this study human capital (HC) is captured in terms of gender, the education level, training, occupation and years of work experience. The data analysis was carried out using the following regression function predictor equation:

$$HP = ß_0 + ß_1Gd + ß_2Ag + ß_3Ed + ß_4Trn + ß_5Exp + \mathcal{E} \quad (1a)$$

HC is not observable. However, HP defined by the following formula was observable:

$$\left\{ \begin{array}{l} HP = 1 \text{ if } HC > 0 \\ \quad\quad\; 0 \text{ if } HC \leq 0 \end{array} \right\} \quad (1b)$$

Where
HP= Household participation
Gd = 1 if female, 0 if otherwise.
Ag = 1 if the household member 26 years, 0 if otherwise.

Educational level

Ed = A vector of dummy variables indicating household member's level of education

These are:
- Primary = If household member has primary level of education
- Secondary = If household member has secondary level of education
- Tertiary = If household member has tertiary level of education
  (Base category: no schooling)

Training

Trn = A vector of dummy variables indicating household member's type of training
These are:
- Vocational = If household member attended vocational training
- On-the-job training = If household member attended on-the-job training
  (Base category: no training)

Exp = 1 if the household member has 2 years of experience, 0 if otherwise.

Table 1. Proportionate distribution of the sample size.

County	Target population	Proportion	Sample size
Kwale	173176	32.1	92
Kilifi	298472	55.3	158
Tana River	68242	12.6	36
Total	539890	100	286

$ß_s$ are the coefficients to be estimated from equation (1b), while $\mathcal{E}$ is the error term with the assumption HP $(\mathcal{E}) = 0$.

Equation (1b) can be estimated using a Probit model because the dependent variable is binary.

The characteristics of the household such as education, age and gender of the individual may have either positive or negative relationships with HP. Households with basic or higher levels of education may influence the degree with which they participate in development positively because it enhances ones chances of participating in training such as workshops and seminars and other development initiatives. Individual marital status may also influence the participation in training and access to funds for economic activities due to lack of collaterals.

Social capital (SC) is captured in terms of membership of groups, interaction with other groups and linkages with development agencies. The data analysis was conducted using the following regression function:

$$HP = ß_0 + ß_1Ms + ß_2Mg + ß_3Ig + ß_4Lda + ß_5Se + ß_6Hs + \mathcal{E} \quad (2a)$$

Table 2. Biodata of the respondents.

Variables	Frequency (n)	Percentage (%)
Age:		
<25 Years	4	0
26 - 30 Years	20	7
31 - 50 Years	151	53
>50 Years	111	40
Gender:		
Male	124	44
Female	162	56
Marital status:		
Married	136	47.2
Single	70	24.6
Widow/widower	80	28.2
Membership to Group	160	55.9
Interaction with other groups	146	51.1
Linkages with devt. agencies	170	58.5
Level of education:		
None	89	31.2
Primary school	97	33.7
Secondary School	70	24.6
College	22	7.7
University	8	2.8
Training		
Vocational training	90	24.6
Informal training	129	45.0
None	67	23.4
Socio-economic activities		
Farming	184	64.3
Fishing	02	0.7
Trading	54	19.0
Formal employment	23	8.0
Others	23	8.0
Experience in agricultural activities		
>2 years	209	73.1
≤ 2 years	55	19.2
None	22	7.7

Field Survey data, 2018

HP is not observable. However, HP that is observable is defined by:

$$\left\{ \begin{array}{l} HP = 1 \text{ if } SC > 0 \ \ (2b) \\ \quad\quad 0 \text{ if } SC \le 0 \end{array} \right\} \quad (2b)$$

Where
Ms = 1 if married, 0 if otherwise.
Mg = 1 if the household is member of a group, 0 if otherwise.
Ig = 1 if interacts with other groups and development agencies, 0 if otherwise.
Lda = 1 if have linkages with development agencies, 0 if otherwise.
Se = 1 if engaged in economic activities, 0 if otherwise.
$ß_s$ are the coefficients to be estimated from equation (1b), while Ɛ is the error term with the assumption HP (Ɛ) = 0

Results

The biodata of the respondents as presented in Table 2 show that the majority (53 %; n =151) of the respondents fell within the age group of 31 - 50 years, whereas an additional 40 % (n = 111) were above 50 years of age, and only 3 respondents were below 20 years (Table 3). More than half (56 %, n = 161) of the respondents were females, while 46 % (n = 133) were males. More than half (56%, n = 161) of the respondents were single, widows and widowers. In terms of household sizes, slightly more (42.8 %, n = 122) of the respondents had small households of 1 - 5 persons while 41.4 % (n = 118) had a household size of 6 – 10 persons. The educational attainments of respondents were relatively low. Only 7.7 % (n = 22) and 2.8 % (n = 8) had college and university education respectively. More than 70 % (24.6 % and 45 %) had undergone training. Interaction with other groups occurred in over 50 % of the households while 51 % had linkages with development agencies. About 64 % (n = 183) of the respondents engaged in farming as their main source of livelihood. Very few respondents engaged in fishing (0.7 %, n=2).

Community participation by type of human capital

Using a Probit regression, the study assessed the influence of biodata comprising age, marital status, level of education, training, type of economic activity and experience attained by the household member on community participation in development initiatives (Table 3). In this model the reference category was "those who did not participate". Table 3 and 4 show the output from the Probit model and the z-statistics.

According to the results in Table 3, tertiary level of education does not predict the likelihood of a household heads participation in development initiatives. Household heads who are younger (25 years or below) are more likely to participate in development initiatives. The probability of participating is 16 % each. Although these results are weakly significant at the

Table 3. Human Capital influencing household participating in development initiatives.

Variables	Probit dF/dx.	z-stat
If female	0.23	1.12
If aged above 26 years	-0.16*	-1.67
Education level:		
Primary school	-0.16**	4.57
Secondary School	-0.14*	1.60
Tertiary	0.13	-1.44
Training (base no training):		
If attended Vocational training	0.35***	-0.18
If attended on-job training	0.23***	0.29
Experience (base no experience):		
>2 years	0.36***	0.19
≤ 2 years	0.21***	0.26
F-stat (wald chi^2)	97.40***	
R^2 (Pseudo-R^2	0.529	
Number of observation	286	

The coefficients on dummy variables indicate changes in probability for each outcome category when the value of the dummy variables changes from zero to one. The second column reports the z-statistics based on robust standard error.

*, **, and *** denotes significant at 10 %, 5 % and 1 % significant levels respectively.

Table 4. Social Capital influencing household participation in development initiatives.

Variables	Probit dF/dx.	z-stat
If married	0.13*	2.87
If member of a group	0.51**	2.21
Interaction with other groups	0.23*	0.29
Linkages with Devt agencies	0.44**	1.95
If engaged in Socio-economic activities:	0.53***	4.95
F-stat (wald chi^2)	92.40**	
R^2 (Pseudo-R^2	0.519	
Number of observation	286	

The coefficients on dummy variables indicate changes in probability for each outcome category when the value of the dummy variables changes from zero to one. The second column reports the z-statistics based on robust standard error.

*, **, and *** denotes significant at 10 %, 5 % and 1 % significant levels respectively.

10 % level, the results for age are consistent with those in Table 2. Household heads with primary education had attended vocational and on-the-job training with a probability of 35 %, 23 % and 53 % respectively and have a higher likelihood of participating in development initiatives. In general, the results show that household heads who have attained primary education and have undergone vocational or on-the-job training predict the chance of participating in development initiatives. This therefore means that basic education is a determinant of rural household participation in agricultural development projects and programmes. However, one of the participants in the FGD stated that:

"I did not go to school, but I am a member of a group engaged in agricultural production in which I am one of the executive members. The group has been in existence for eleven years and has networks. The group works and attracts support from Ministry of Agriculture and other organizations including FAO, KCep and KALRO among others".

The Probit results in Table 4 show that households who are members of groups and have linkages with government and development agencies that include private entities, NGOs and CBOs have a higher likelihood of 51% and 44 % of participating in development initiatives and 53 % being engaged in socio economic activities. The interaction with other groups has a lesser likelihood of the individual participating in development projects and programmes. Groups and engagement in socio economic activities as social capital provide opportunities for engaging in development activities implemented in the community. For instance, one participant in a FGD stated that:
"In Groups individuals gain synergy for demanding for services and engagement in socio economic activities. Membership to groups also serves as an avenue for participation in development initiatives".

Discussion

Human capital is inherent in an individual and is important in providing opportunities for households to engage in active participation in any development process or activities. Knowledge, skills and competency are among the important aspects of human capital acquired by individual households during the learning process, and are necessary for influencing decision-making and perceiving the benefits accrued from projects and programmes implemented in the community. Tanner et al. (2002) states that vocational training is an effective means of producing changes in practice, especially in relation to acquisition of individual human skills. Tanner *et al.* (2002) argues that lack of predictability of the likelihood of stakeholder participation in agricultural projects could be due to the lack of facilitation skills among professionals or extension workers. According to Nwake et al. (2013), the lack of the likelihood of participation could be attributed to the role that the professional development workers play even if the aim is to build the capacity of communities or empower them. Results obtained in a study done by Djomo and Sikod (2012) indicate that an additional year of experience and levels of education increases agricultural productivity. However, an additional year of experience acquired, led to a reduction in the level of inefficiency. Similarly, an additional unit of education reduces the level of inefficiency.

The negative significance findings on level of education are in agreement with those of Aworti (2012) who asserts that education as a human capital is in itself not entirely a determining variable in community participation. He suggests that many uneducated households scored even better than those with secondary school education in variables such as membership of community organizations, attendance of meetings and participation in planning, while those with good education levels speak more in meetings than those without education. High education level can also be a hindering factor in individual participation as explained by Dorsner (2004). Dorsner indicates that educated members of the communities at times are not available for their community even if they have interest, as they tend to have other business commitments. Although on-the-job training which allows acquisition of knowledge, skills and competency in performing various activities had a less likelihood than training, it is a more interactive and dialectic process of knowledge acquisition and provides these categories of participants with the necessary skills for them to participate in agricultural projects and programmes.

The results related to social capital imply that an increase per unit of each of the measures of social capital increases the probability of households participating in development initiatives. Membership of groups provide an avenue for collective action and helps individuals to negotiate the various challenges they face as they strive to pursue their individual goals and mutual interest. According to Seferiadis *et al*. (2015), membership of groups strengthens social fabric and interaction. Networks enable individuals to access resources and information and achievement of common goals. Brodie *et al.* (2009) found out that the socio-economic group a person belongs to has an impact on his/her level of participation, as people from lower socio-economic groups often have less access to resources and practical support making their participation in agricultural development initiatives rather difficult.

The formation of networks and linkages encourages residents to trust one another and therefore enables greater cooperation for mutual benefits. For instance, networks help in involving diverse players in the community in decision making processes, especially the vulnerable members of the community (Putman, 2000). Social networks provide useful information about the potential resources that can contribute to desired outcomes, such as socio-economic development, and in understanding factors that influence the capacity of isolated communities to make effective use of scarce natural or physical resources for achieving economic self-sufficiency (Tirmizi, 2005). Linkages open up avenues for creation of awareness to ideas and access to information and resources found outside the community which are necessary for development.

Conclusion

This study has shown that individual participation in development initiatives requires a set of human and social capital; elements that are closely intertwined. Human capital is associated with active and interactive engagement of the individual in development activities such as workshops, training and other practical activities. The interactive process inherent in group activities increases the ability of individual members to acquire knowledge and skills which are essential for decision making on the use of new ideas introduced to them, potentially leading to improved welfare. Although education as a human capital is necessary especially in the acquisition of technical knowledge, one does not require tertiary education to participate in development initiatives implemented in the community.

Policy recommendation

The Central Government, County Governments and development partners, should:
Emphasise the importance of community members enrolling in learning institutions to acquire basic education, and encourage group formation grounded on a strong foundation of trust. This will allow the individual members in the communities to engage in productive economic activities.

Acknowledgements

Acknowledgement goes to Pwani University for funding this research work.

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Phytoplankton distribution along a salinity gradient in two Kenyan saltworks (Tana and Kurawa)

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Abstract

The current study assessed the diversity and abundance of phytoplankton genera in two Kenyan saltworks (Tana and Kurawa) in March and September 2021. Water samples were obtained from ponds with salinities ranging from 30 to 200 ppt by filtering 40 l of water using a 20-μm phytoplankton net. Seventy-six genera of phytoplankton were identified. Genera richness, evenness and diversity decreased with increasing salinity while phytoplankton abundance increased with increasing salinity. Higher phytoplankton densities were observed in the Tana than in the Kurawa saltworks. Ponds of <100 ppt were dominated by Dinophyceae and Bacillariophyceae which accounted for >90 % of the phytoplankton community. Ponds of salinities >100 ppt were dominated by Cyanophyceae which accounted for >90 % of the phytoplankton community. From the results it was concluded that Kenyan saltworks host diverse phytoplankton genera whose richness decreases with increasing salinity and varies with seasons. The present data describes variation of phytoplankton assemblages in salt ponds between two selected seasons, but several samplings throughout the year would be more appropriate to describe variations of phytoplankton with season in these salt ponds.

Keywords: phytoplankton, hypersaline, saltworks, distribution, abundance, richness

Introduction

Saltworks are constituted of a series of interconnected ponds with increasing salinity from inlet to crystallizers. Natural seawater is drawn from a nearby creek or lagoon often by pumping into a large reservoir, and then allowed to flow by gravity through a series of shallow interconnected man-made ponds where evaporation fueled by wind and solar energy occurs (Korovessis and Lekkas, 1999). Since evaporation occurs in stages, ponds of narrow salinity ranges are created where different salts present in the original seawater precipitate based on their solubility. Low soluble salts such as calcium carbonate and sulphate precipitate in salinities below 200 ppt while sodium chloride, which is highly soluble, precipitates in the crystallizers (300–350 ppt) (Gongora *et al.*, 2005).

The natural seawater brought into saltworks also brings with it inorganic nutrients and several marine organisms, which establish stable populations at different salinity levels in saltworks ponds. Among the organisms described in saltworks include phytoplankton (Dolapsakis *et al.*, 2005; Elloumi *et al.*, 2009; Evagelopoulos *et al.*, 2009; Radhika *et al.*, 2011; Chatchawan *et al.*, 2011; Madkour and Gaballah, 2012; Devi *et al.*, 2019), ciliates (Elloumi *et al.*, 2006), halophilic bacteria and archaea (Casamayor *et al.*, 2002; Oren, 2008), zooplankton (Mustafa *et al.*, 1999), as well as macrobenthic invertebrates (Evagelopoulos *et al.*, 2009) and the brine shrimp (Elloumi *et al.*, 2009; Mohebbi, 2010). The consensus from most studies is that species richness, evenness and diversity declines with increasing salinity (Evagelopoulos *et al.*, 2009). Abundance on

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the other hand, has shown different trends for different organisms. In most studies, phytoplankton density and biomass have been reported to be highest in the high salinity ponds. Ciliates and most zooplankton except *Artemia* have shown a decreasing trend with increasing salinity in most studies (Elloumi *et al.*, 2006; Radhika *et al.*, 2011). The wide range of planktonic and benthic communities of marine organisms that develop along the salinity gradient constitute a biological system that plays an important role in the production of high-quality salt through preventing leakage of brine, improving solar energy absorption and water evaporation (Madkour and Gaballah, 2012).

Diversity and abundance of phytoplankton in saline lakes and saltworks has been an area of research focus by several scientists in the recent past. Phytoplankton sit at the base of marine food webs and act as primary producers where they convert solar energy into chemical energy. Zooplankton feed on the phytoplankton and pass the energy to higher trophic levels. In addition, phytoplankton produce oxygen and organic matter that support the flourishing of microbial communities in aquatic environments (Jakhar, 2013). The most common phytoplankton groups that have been described in saltworks include Dinophyceae, Bacillariophyceae, Chlorophyceae, Euglenophyceae, Raphidophyceae, Prymnesiophyceae, Cryptophyceae, and Silicoflagellates. The most dominant among the groups identified have been Bacillariophyceae and Dinophyceae in ponds of salinities below 100 ppt, and Cyanophyceae and Chlorophyceae (*Dunaliella*) in salinities above 100 ppt (Tiffany *et al.*, 2007; Evagelopoulos *et al.*, 2009; Madkour and Gaballah, 2012). Despite the numerous studies conducted on phytoplankton diversity in saltworks, there is limited information on the diversity and abundance of phytoplankton in tropical saltworks.

Kenya is an important salt producing country in the East African region with over 5 large commercial saltworks companies and several artisanal salt farms operating along the Kenyan coast (KNCHR, 2006). The salt farms have been identified for integrated production of salt and *Artemia*. *Artemia* is an important live feed in the larviculture of marine fish (FAO, 2011). Initial inoculation of *Artemia* in Kenyan saltworks was done in the mid-1980s at a time when similar initiatives were undertaken in some Asian and Latin American countries (FAO, 2011; Ogello *et al.*, 2014). Following successful post-inoculation colonization and establishment of stable *Artemia* populations in Kenya, efforts are underway to commercialize the integrated salt-*Artemia* farming model to generate enough *Artemia* cysts for the revitalised mariculture sector in the country, while at the same time improving overall revenues earned by salt producers. *Artemia* products are valued based on their nutritive value especially levels of essential highly unsaturated fatty acids, which are dependent on the diet of the *Artemia* (FAO, 2011). Primarily, *Artemia* feeds on phytoplankton present in the saltworks and this is a major determinant of *Artemia* quality. An understanding of the distribution and abundance of phytoplankton in Kenyan saltworks is important to inform efforts to improve the quality of *Artemia* cysts and overall biomass.

The present study seeks to determine the spatial distribution of phytoplankton along a salinity gradient in two Kenyan saltworks (Tana and Kurawa) using samples collected in March and September 2021. Sampling of phytoplankton was carried out at five salinity ranges (30-40, 50-60, 90-100, 150-160 and 190-200 ppt). It is expected that the information generated will benefit commercial and artisanal salt producers as well as policy makers to optimize *Artemia* production in saltworks.

Materials and methods

Study area

The study was conducted at the Tana and Kurawa salt farms, which are among the major salt producing companies in Kenya (Fig. 1). Kurawa saltworks is located in Kurawa area, Magarini Sub County, Kilifi County (2°44'3.00"S, 40°10'16.00"E) about 1 km off the Malindi-Garsen road at Kanagoni town. The saltworks covers an area of about 595.2 ha (KNHCR, 2006; Mumbah *et al.*, 2017). Tana saltworks is located in Tana River County (2°41'58.00"S, 40°11'5.00"E) about 14 km from Kanagoni town and occupies an area of approximately 380 ha (Yap and Landoy, 1986). Salinity of water circulated in the two saltworks ranges from 35 ppt at the inlet to 300 ppt in the crystallizers.

Sampling

Sampling was carried out selectively in salt ponds of varying salinity ranges (30-40, 50-60, 90-100, 150-160 and 190-200 ppt). In situ measurement of water temperature, salinity, total dissolved solids (TDS) and conductivity was carried out using a handheld multi-parameter sensor (YSI Professional Plus, USA) prior to collection of water samples. The water quality data was collected from four different points in one pond. The first sampling was done in the month of March 2021 at Kurawa and Tana between 0900 h to 1200

h. For each pond, 40 l of saline water were filtered using a 20 µm plankton net of 30 cm diameter and the remnant was transferred to 250 ml bottles. This procedure was repeated twice for each pond sampled. The samples were immediately fixed with 1ml of acidified Lugol's solution to prevent predation by zooplankton and to preserve the collected microalgae cells. Samples were transported in a cooler box to the laboratory, received and stored in the dark at room temperature awaiting laboratory analysis. The same procedure was repeated during the second sampling in the month of September 2021. March is the peak of the dry season in Kenya whereas September is the peak of the wet season. The choice of the two months was to optimize the ecological effects brought about by the two seasons (wet and dry).

Laboratory analysis

Microscopic identification and enumeration was conducted a week after sample collection. Firstly, a standard (concentrated) working volume of 50 ml across the collected batch of samples was obtained by incorporating sedimentation and further filtration using a small-modified sieve of 20 µm. Afterwards, 1 ml of the standardized sample was transferred to a Sedgewick rafter cell. One hundred small squares in the Sedgewick rafter cell (0.1 ml in duplicate) were taxonomically identified to genera level (Niklas *et al.*, 1978; Tomas, 1997; Cronberg and Annadotter, 2006), and counted under an inverted microscope (Euromex-Oxion inverso, 200x). Confirmatory identification for smaller species (<20 µm) was carried out using an ordinary microscope slide under an upright microscope (Zeiss-Primostar, 1000x).

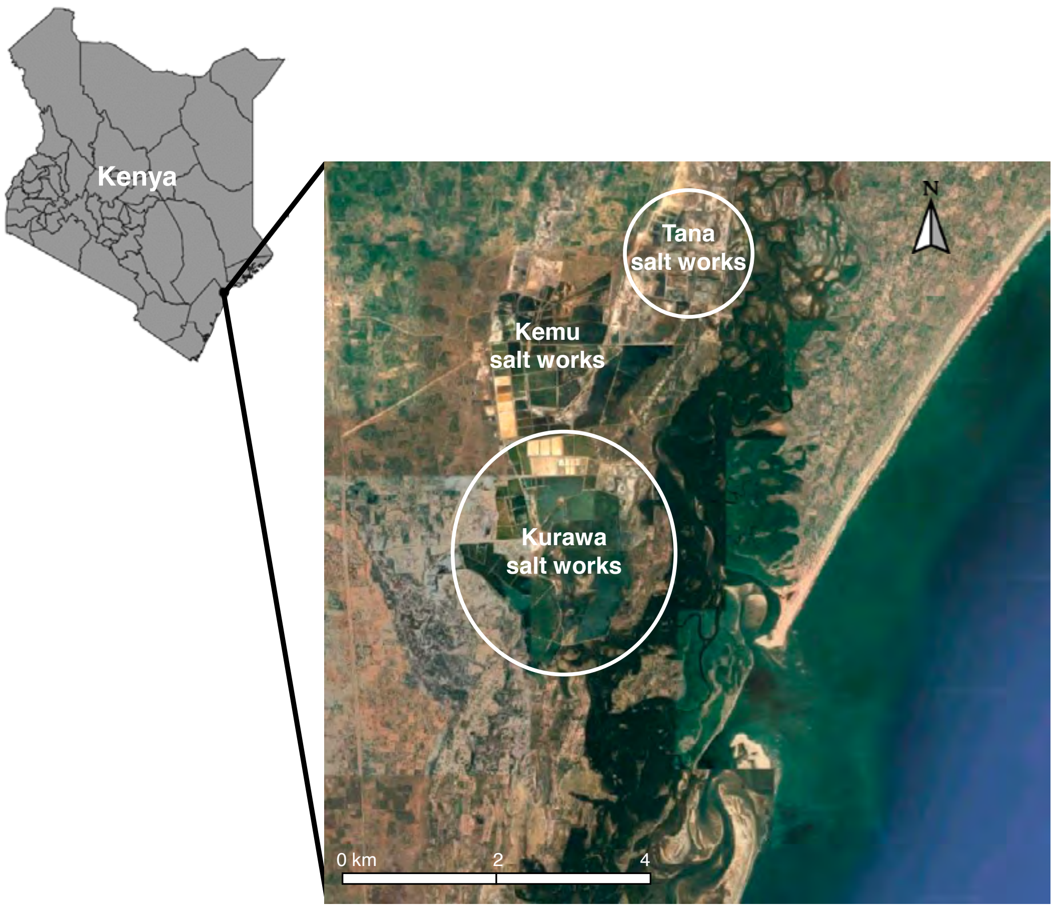

Figure 1. Map of Kenya showing Kurawa, Kemu and Tana saltworks. Sampling was conducted in Kurawa and Tana saltworks. Adapted from Google Earth. Accessed on 01/07/2022.

Statistical analysis

The data obtained from laboratory analysis was keyed in and processed on an excel spreadsheet. Normality of the data was tested using the Shapiro-Wilk test in R statistical software (version 4.0.2). Indices of diversity such as genera richness, abundance, diversity and relative abundance across the different

Table 1. Mean water quality parameters recorded from sampled ponds. Values in parenthesis represent ± standard deviation. Sal=salinity, Cond=conductivity and TDS=total dissolved solids

Site	Month	Pond	Sal.	Temp(°C)	pH	Cond(S/m) x1000	TDS (mg/l) x1000
Tana	**March**	1	37.95(0.07)	30.85(0.07)	7.20(0.03)	63.66(0.20)	37.28(0.05)
		2	55.70(0.14)	31.90(0.35)	7.85(0.08)	90.53(0.39)	52.03(0.14)
		3	92.76(0.34)	32.65(0.14)	7.74(0.22)	140.817(0.34)	80.20(0.83)
		4	142.90(0.99)	33.40(1.55)	7.98(0.05)	204.26(1.91)	12.39(0.74)
		5	193.45(1.48)	36.40(0.28)	7.20(0.04)	262.93(0.07)	40.35(0.66)
	September	1	39.15(0.21)	27.55(0.21)	8.27(0.09)	62.57(0.96)	38.21(0.02)
		2	51.75(0.01)	28.95(0.07)	11.87(0.04)	80.52(0.01)	48.72(0.04)
		3	93.55(0.21)	28.90(0.14)	11.16(0.01	132.58(0.12)	80.33(0.01)
		4	153.31(0.01)	33.15(0.21)	8.72(0.02)	209.62(0.23)	118.21(0.70)
		5	197.55(0.21)	36.15(0.31)	8.09(0.01)	266.03(0.06)	142.36(0.01)
Kurawa	**March**	1	39.85(0.07)	34.75(0.07)	8.25(0.02)	26.17(7.17)	39.03(0.05)
		2	50.72(0.311)	34.25(0.14)	8.00(0.00)	87.17(0.29)	48.32(0.04)
		3	93.94(0.51)	36.40(0.21)	8.30(0.14)	151.54(0.53)	80.89(0.41)
		4	155.55(1.77)	36.55(0.07)	7.40(0.00)	267.57(2.36)	42.23(0.88)
		5	196.90(1.55)	36.20(0.28)	7.80(0.14)	223.03(2.15)	19.76(1.06)
	September	1	40.31(0.11)	27.40(0.14)	6.91(0.02)	63.24(0.01)	39.19(0.01)
		2	54.97(0.05)	26.55(0.21)	7.24(0.03)	81.01(3.53)	51.23(0.01)
		3	94.62(0.78)	27.10(0.24)	7.06(0.05)	80.98(0.02)	129.25(0.01)
		4	152.25(0.21)	28.10(0.04)	6.43(0.02)	190.97(0.01)	117.32(0.11)
		5	191.61(0.02)	29.20(0.36)	7.20(0.01)	231.26(0.02)	138.97(0.23)

sampled ponds and seasons were calculated from this data using the formulas i, ii and iii listed below according to Kiteresi *et al.* (2011). A regression analysis was carried out between the indices of diversity calculated and salinity to establish any form of relationship. All regression analyses were considered significant at $p<0.05$.

i) Abundance (N) = A * 1000 * C / V * F * L

Where:

N: Number of plankton cells per liter of the original water

A: Total number of plankton counted

C: Volume of final concentrate of the sample in ml

V: Volume of a field in mm^3

F: Number of fields counted

ii) Shannon-Wiener Index (H) = $^{-}\Sigma\, n_i/N \log2\, n_i/N$

Where:

n_i: number of individuals of the i^{th} species

N: total number of individuals.

iii) Pielou evenness index (E) = H/ln S

Where:

H: Shannon Wiener's species diversity index

S: Species richness (number of species).

Results

Water quality parameters

Five water quality parameters were measured in each of the sampled ponds; namely salinity, pH, temperature, conductivity and total dissolved solids (Table 1). Salinity ranged from 37.95 to 197.55 ppt. Temperature ranged from 26.55 to 36.55 °C, whereas pH ranged from 6.43 to 11.87. Conductivity ranged from 26,172.00 to 267, 574.00 S/m while TDS ranged from 12, 385.50 to 142, 362.75 mg/l. Generally, higher salinity ponds were much warmer than the lower salinity ones.

Phytoplankton distribution

The present study identified 76 phytoplankton genera distributed among five classes, namely Dinophyceae (13), Bacillariophyceae (38), Cyanophyceae (13), Chlorophyceae (10) and Euglenophyceae (2) (Table 2). The five major genera of Cyanophyceae were *Synechococcus, Oscillatoria, Lyngbya, Anabaena* and *Spirulina*. For Dinophyceae, they were *Gymnodinium, Akashiwo, Prorocentrum, Scrippsiella* and *Dinophysis*. For Chlorophyceae, the major genera were *Dunaliella, Tetraselmis, Botryococcus, Pediastrum* and *Golenkinia*. The major genera of Bacillariophyceae were *Nitzschia, Pleurosigma, Thallassiosira, Navicula and Licmophora*. For Euglenophyceae, two genera were identified, namely *Eutreptiella* and *Phacus*. Notably, Cyanophyceae had the highest number of

Table 2. Distribution of major phytoplankton genera along a salinity gradient. + and – indicate present and absent, respectively. Values in parenthesis represent mean abundance of the genera in a given salinity.

		Salinity Ranges				
Class	**Genera**	**30-40**	**50-60**	**90-100**	**150-160**	**190-200**
Cyanophyceae	*Synechococcus*	+ (22.37)	-	+ (6.25)	+ (104, 906.30)	+ (118, 794.50)
	Oscillatoria	+ (69.70)	+ (463.28)	+ (2, 200)	+ (2, 307.03)	+ (529.68)
	Lyngbya	+ (6.25)	+ (421.88)	+ (12.50)	+ (6.25)	+ (6.25)
	Anabaena	+ (18.75)	+ (5.59)	+ (83.28)	-	-
	Spirulina	+ (60.27)	+ (12.50)	+ (6.25)	-	-
Dinophyceae	*Gymnodinium*	+ (15.24)	+ (15, 543.75)	+ (6.25)	-	+ (18.75)
	Akashiwo	-	+ (2, 395.31)	+ (14.58)	-	-
	Prorocentrum	+ (2,301.86)	+ (18.75)	+ (9.37)	+ (6.25)	-
	Scrippsiella	+ (1,539.31)	+ (295.83)	+ (12.50)	-	-
	Dinophysis	+ (21.49)	+ (414.06)	+ (25.00)	-	-
Chlorophyceae	*Dunaliella*	+ (45.73)	+(262.50)	+ (312.50)	+ (5, 523.43)	+ (1, 968.21)
	Tetraselmis	+ (30.48)	+ (128.12)	+ (12.50)	+ (22.09)	-
	Botryococcus	+ (62.5)	-	-	-	-
	Pediastrum	+ (59.37)	+ (31.25)	-	-	-
	Golenkinia	-	+ (18.75)	-	-	-
Bacillariophyceae	*Nitzschia*	+ (2, 500.48)	+ (53.47)	+ (229.16)	+ (6.25)	-
	Pleurosigma	+ (579.49)	+ (1, 356.25)	+ (99.21)	+ (14.06)	+ (8.33)
	Thalassiosira	+ (81.03)	+ (1, 112.50)	+ (6.25)	-	-
	Navicula	+ (140.66)	+ (214.06)	+ (117.18)	+ (168.69)	+ (8.83)
	Licmophora	+ (169.39)	+ (36.45)	-	+ (6.25)	-
Euglenophyceae	*Eutrptiella*	+(63.08)	-	+(6.25)	-	-
	Phacus	+(18.75)	-	-	-	-

genera with the widest salinity tolerance range (*Synechococcus, Oscillatoria* and *Lyngbya*) as compared to Bacillariophyceae (*Pleurosigma* and *Navicula*), Chlorophyceae (*Dunaliella*) and Dinophyceae (*Gymnodinium*).

Genera abundance

Genera abundance showed different patterns with increasing salinity in the two sampled saltworks. At Tana salt farm, genera abundance was lowest at salinities ranging from 30-100 ppt, and highest at salinities ranging from 100-200 ppt. Notably, samples collected in March reported higher genera abundance in most of the salinity ranges except at 150-160 ppt where samples collected in September reported significantly higher abundance than those collected in March. With respect to Tana, genera abundance was generally lower in all salinity ranges at Kurawa saltworks. Samples collected in March had higher genera abundance than those collected in September at the salinity range 190-200 ppt (Fig. 2).

Genera richness

Genera richness in the sampled saltworks was higher at the salinity range 30-40 ppt as compared to the other salinity ranges (Fig. 3). At Tana saltworks, there was no distinct trend in genera richness between seasons. At the Kurawa saltworks, genera richness was generally highest in September for most of the salinity ranges except 90-100 ppt where the highest genera richness was observed in March.

Genera diversity

Genera diversity in the sampled salt farms was higher in salinities ranging from 30-100 ppt than in salinities ranging from 100-200 ppt. At Tana salt farm, higher genera diversity was observed in most of the salinity ranges during September except at the salinity range 150-160 ppt where higher diversity was observed in March. In Kurawa, higher genera diversity was observed in September in all salinity ranges except 90-100 ppt where the highest diversity was observed in March (Fig. 4).

Genera evenness

Genera evenness was highest in salinity ranging from 30-100 ppt except in samples collected in September at Kurawa where genera evenness formed a peak at the salinity range of 150-160 ppt (Fig. 5). Generally, samples collected in September at Tana saltworks produced the highest genera evenness in most of the salinity ranges

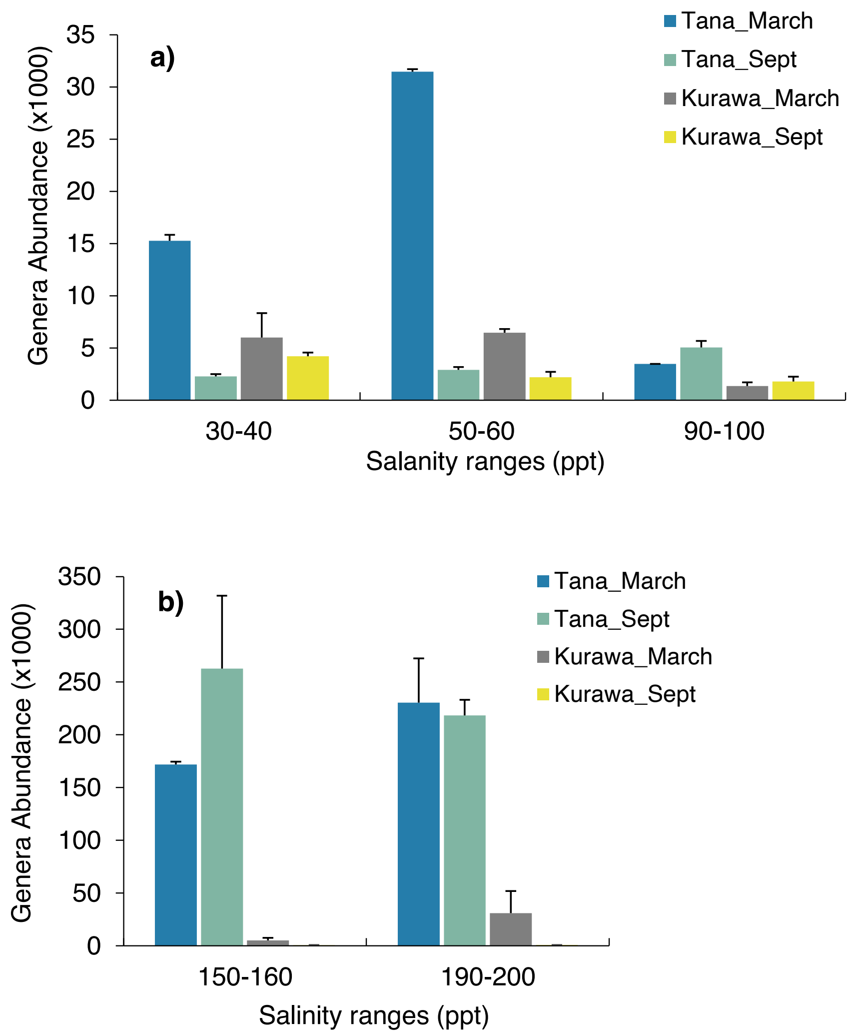

Figure 2. Phytoplankton genera abundance of sampled ponds at different saltworks and seasons. (a) shows abundance in salinities up to 100 whereas (b) shows abundance in salinities above 100. Error bars represent ±standard deviation.

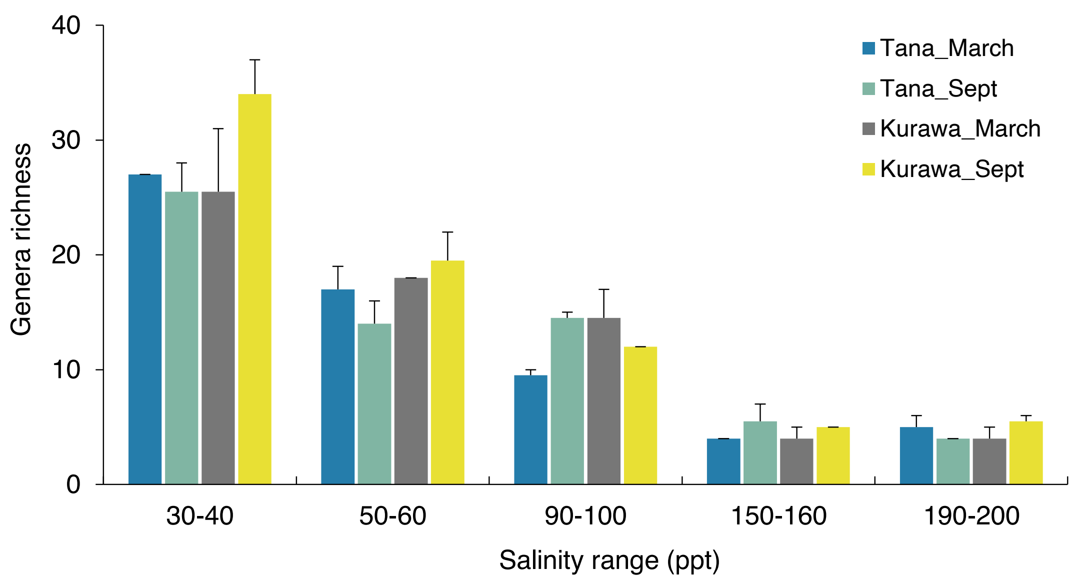

Figure 3. Genera richness of sampled ponds at different saltworks and seasons. Error bars represent ± standard deviation.

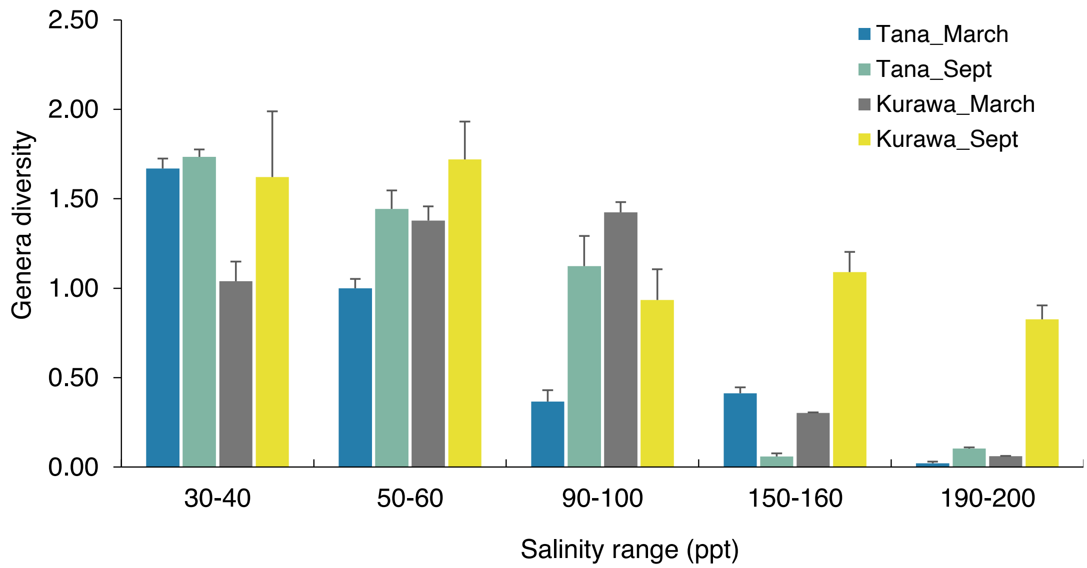

Figure 4. Genera diversity of sampled ponds at different saltworks and seasons. Error bars represent ±standard deviation.

except at 150-160 ppt whereas samples collected in September at Kurawa saltworks produced the highest genera evenness in most of the salinity ranges.

Relative abundance

Tana

At salinity ranging from 30-100 ppt, Dinophyceae and Bacillariophyceae (Fig. 6) dominated phytoplankton communities. Notably, Dinophyceae dominated phytoplankton communities in the salinity range 30-40 ppt in March while in September, Bacillariophyceae were the most dominant. At the salinity range 50-60 ppt, Bacillariophyceae were dominant in March and Dinophyceae were dominant in September. At salinity ranging from 100-200 ppt, Cyanophyceae dominated in both March and September samples.

Kurawa

Dinophyceae and Bacillariophyceae dominated phytoplankton communities at salinities ranging from 30-100 ppt (Fig. 7). Notably, Bacillariophyceae dominated in all samples collected in March and September at the salinity range of 30-40 ppt. At the salinity range 50-60 ppt, Bacillariophyceae dominated in September

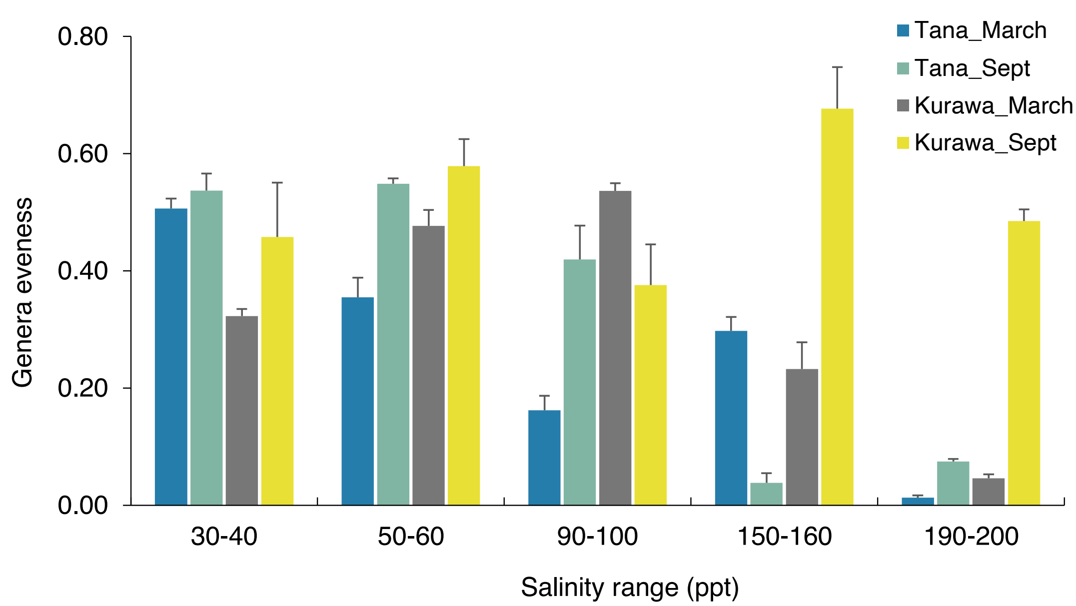

Figure 5. Genera evenness of sampled ponds at different saltworks and seasons. Error bars represent ±standard deviation.

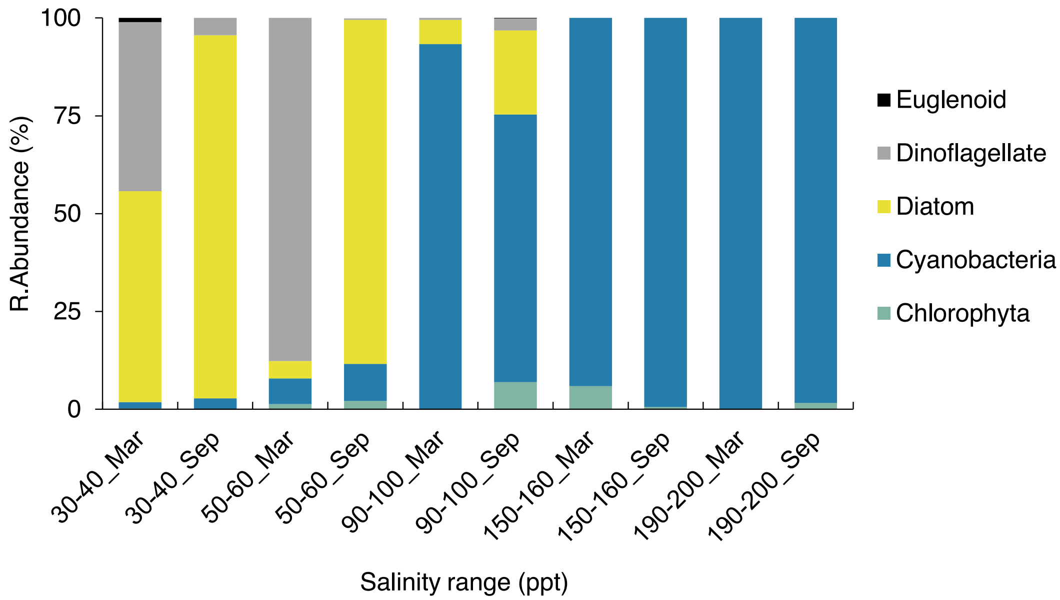

Figure 6. Relative abundance of phytoplankton groups in ponds sampled at different seasons in Tana saltworks.

whereas Dinophyceae dominated in March. At salinities between 100-200 ppt, Cyanophyceae dominated in both March and September samples.

Relationship between biodiversity indices and salinity

Genera diversity and richness showed negative relationships with increasing salinity at most of the sampling points and times (Table 3). Genera evenness showed a negative relationship with increasing salinity from 30 to 200 ppt in most of the sampling points except in September at Kurawa salt farm where no significant relationship was observed. Genera abundance on the other hand showed a positive relationship with salinity at Tana in both March and September. At Kurawa, a significant negative relationship between abundance and salinity was observed in September. However, no significant relationship between abundance and salinity was observed at Kurawa salt farm in March.

Discussion

The present study identified 76 phytoplankton genera distributed among five classes, namely Dinophyceae, Bacillariophyceae, Cyanophyceae, Chlorophyceae

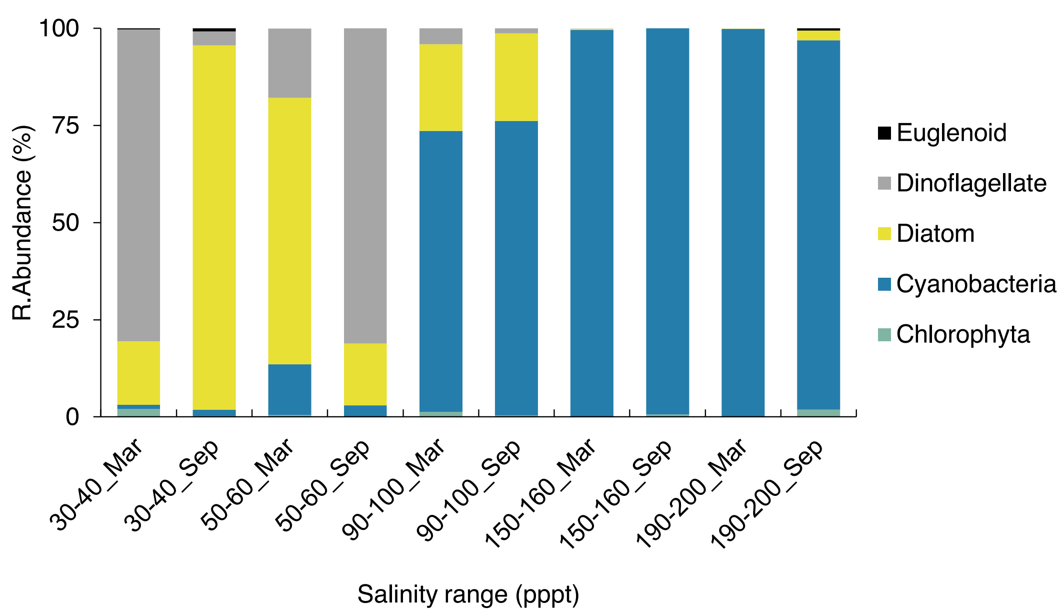

Figure 7. Relative abundance of phytoplankton groups in ponds sampled at different seasons in Kurawa saltworks.

Table 3. Regression analysis between salinity and phytoplankton community structure indices in Kurawa and Tana saltworks.

Index	Sampling site	Coefficient	R^2	*t* Stat	*P*-value
Genera diversity	Tana-March	-88.36	0.75	-4.91	<0.05
	Tana-Sept	-82.32	0.93	-10.45	<0.05
	Kurawa-March	-91.16	0.75	-4.89	<0.05
	Kurawa-Sept	-102.32	0.53	-3.00	<0.05
Genera abundance	Tana-March	0.00	0.83	6.29	<0.05
	Tana-Sept	0.00	0.76	5.05	<0.05
	Kurawa-March	0.00	0.27	1.71	>0.05
	Kurawa-Sept	-0.04	0.64	-3.79	<0.05
Genera richness	Tana-March	-6.02	0.77	-5.18	<0.05
	Tana-Sept	-6.66	0.78	-5.31	<0.05
	Kurawa-March	-6.14	0.82	-6.08	<0.05
	Kurawa-Sept	-4.69	0.75	-4.85	<0.05
Genera evenness	Tana-March	-276.11	0.62	-3.64	<0.05
	Tana-Sept	-251.61	0.91	-8.98	<0.05
	Kurawa-March	-251.89	0.57	-3.24	<0.05
	Kurawa-Sept	102.16	0.04	0.61	>0.05

and Euglenophyceae in the two Kenyan saltworks studied. The number of genera observed in the present study was much higher than in previous studies (Ayadi *et al.*, 2004; Madkour and Gaballah, 2012). Out of the five classes of phytoplankton, Cyanophyceae stood out as the most tolerant class with 3 genera that were present in most of the salinity ranges. High numbers of genera from the class Cyanophyceae (*Synechococcus, Oscillatoria* and *Lyngbya*) have also been reported by Nagasathya and Thajuddin (2008) at all sampling points ranging from 48-185 ppt when they studied the diversity of Cyanophyceae in hypersaline environments of saltpans on the southeast coast of India. Further, a review by Oren (2012) noted that some genera in the class Cyanophyceae (*Synechococcus, Oscillatoria, Lyngbya, Spirulina, Microcoleus* and *Synechocystis*) could grow at high salinities of up to 200 ppt.

The observed phytoplankton richness, evenness and diversity in both saltworks were highest in ponds with salinities ranging from 30-100 ppt followed by a sharp decline in ponds with salinity ranging from 100-200 ppt above. The declining trend of indices of diversity is supported by the results of the regression analysis obtained (Table 2). Nearly all the regression analyses between salinity and the three indices of diversity reflected a significant relationship ($R^2 > 0.5$, $P < 0.05$) with a negative coefficient. The higher richness and numbers of phytoplankton genera recorded in the ponds with salinity ranging from 30-100 ppt could be attributed to favorable conditions in terms of less harsh abiotic factors; particularly temperature and salinity levels, as compared to the ponds with salinity ranging from 100-200 ppt (Ahel *et al.*, 1996). A declining trend of species richness along a salinity gradient has also been reported in other studies (Ayadi *et al.*, 2004; Elloumi *et al.*, 2009; Marcarelli *et al.*, 2006; Madkour and Gaballah, 2012; Larson and Belovsky, 2013).

Unlike the other indices of phytoplankton diversity, numerical abundance increased with increasing salinity in all sampling sites and seasons. This was confirmed by the positive coefficient in all regression analyses conducted between numerical abundance and salinity, and this could be attributed to stable (less fluctuating) abiotic conditions associated with hypersaline environments that allow high productivity (Joint *et al.*, 2002). A similar increase in phytoplankton density with salinity was reported by Madkour and Gaballah (2012) when they investigated the phytoplankton assemblage of a solar saltern in Port Fouad, Egypt. Larson and Belovsky (2013) also observed an increase in abundance of *Dunaliela* spp with increasing salinity when they investigated the influence of salinity and nutrients on phytoplankton communities in microcosm experiments.

It is worth noting that samples collected in Tana saltworks reported higher numerical abundance of phytoplankton genera than samples collected from Kurawa saltworks at all seasons. At Kurawa, a high presence of *Artemia* in the ponds of higher salinity was observed. In fact, *Artemia* were present in some of the samples collected. In Tana saltworks, the density of *Artemia* was minimal and this was attributed to low inoculation by humans due to accessibility challenges, unlike in Kurawa where all ponds were easily accessible. The low numerical abundance of phytoplankton genera in Kurawa saltworks could be associated with the high grazing pressure of *Artemia* in the sampled ponds. The assertion that the presence of high grazing pressure could be responsible for the lower abundance of phytoplankton in Kurawa than in Tana is in agreement with Barnes and Wurtsbaugh (2015), who observed low phytoplankton chlorophyll levels in a microcosm experiment where *Artemia* and other grazers were present, but when the grazers were absent, the phytoplankton chlorophyll levels increased. Karacaoğlu *et al*. (2006) also noted that phytoplankton density changes could be because of predation and grazing pressure through the aquatic food web.

The results of the present have also indicated differences in the phytoplankton dominance among the ponds of different salinities. Ponds of salinities below 100 ppt were dominated by Dinophyceae, Bacillariophyceae and to a lesser extent Euglenophyceae and Chlorophyceae. In salinities above 100 ppt, Bacillariophyceae persisted due to the versatile genera *Pleurosigma* and *Navicula* that were reported even at extreme salinity ranges (Fig. 5 and 6). However, dominance of these two classes was extremely reduced in that their contribution to the phytoplankton communities at the higher salinities (>100 ppt) was negligible. The reduction of dominance of these two classes at salinities above 100 ppt was attributed to intolerance of higher salinities. The finding that Dinophyceae and Bacillariophyceae dominate low salinities (<100 ppt) and the reduction of their dominance at higher salinities (>100 ppt) has also been reported in other studies (Elloumi *et al*., 2009; Madkour and Gaballah, 2012).

Changing dominance between Bacillariophyceae and Dinophyceae in different seasons was also observed, which Ayedi *et al*. (2004) referred to as a negative correlation between Bacillariophyceae and Dinophyceae dominance. In the marine environment, the shift between Bacillariophyceae and Dinophyceae has been associated with seasonal variations. Hilaluddin *et al*. (2020) observed that Bacillariophyceae contributed the highest percentage (66.0 % to 98.9 %) of the phytoplankton community in most parts of the year under normal conditions, but their contribution declined (43 %) after the wet season with Dinophyceae contributing the highest percentage. While the results from the present study demonstrated the shift between Bacillariophyceae and Dinophyceae in different seasons, a clear association of season with the dominance of either group was not observed.

Cyanophyceae was the most dominant group in ponds of salinities above 70 ppt accounting for >90 %, and to a lesser extent Chlorophyceae, Dinophyceae and Bacillariophyceae. Three genera (*Synechococcus, Oscillatoria* and *Lyngbya*) were identified that were exclusively responsible for the dominance of Cyanophyceae. The observed dominance of Cyanophyceae in the present study is not surprising. Das Sarma and Arora (2002) noted that Cyanophyceae dominate the plankton biomass in many hypersaline lakes and form microbial mats. The ability of Cyanophyceae to dominate hypersaline environments has been associated with the accumulation of compatible solutes in their cells to counter high osmotic pressure. For instance, the unicellular species of Cyanophyceae (*Aphanothece halophytic*) has been observed to accumulate high glycerin in their cells and can tolerate salinity up to 124 ppt (Nagasathya and Thajuddin, 2008).

In terms of the effect of season on biodiversity indices, generally higher genera richness, evenness and diversity were reported in samples collected in September than in March. September coincides with the wet season (southeast monsoon) which is characterized by high organic and inorganic nutrient input into the oceans from the land, whereas March coincides with the dry season (northeast monsoon) which is characterized by limited nutrient input from land. According to Affan *et al.* (2005), phytoplankton abundance and taxonomic diversity depends on supply of nutrients in natural waters, and an increase in phytoplankton diversity with increase in nutrient concentrations in water has been observed. The seasonal differences in organic inputs could be responsible for the differences in phytoplankton diversities and abundances observed between the two seasons during which sampling was carried out.

Conclusions

The findings of the study have revealed the presence of highly diverse phytoplankton communities in the tropical saltworks of Kenya. Genera richness was highest in the low salinity ponds and decreased sharply with increasing salinity gradient. Conversely, the genera abundance increased with increasing salinity forming a peak at higher salinity. Generally, samples obtained in September from most of the sampled ponds reported higher genera richness, evenness and diversity. Ponds of salinity ranging from 30-100 ppt were dominated by Dinophyceae and Bacillariophyceae that showed a shifting dominance. However, it was not clear what the effect of season was on the shifting dominance observed, probably because sampling was restricted to only two months of the year. A follow-up study with year-round sampling is therefore recommended.

In salinities above 100 ppt, Cyanophyceae were the most dominant algae. From the result there was a clear indication that salinity influenced the occurrence and distribution of phytoplankton along the increasing salinity gradient. It is also important to note that genera abundance was much higher in the Tana as compared to the Kurawa saltworks and it is suggested that this was associated with low grazing pressure by *Artemia* that were present at different densities at the two sites. Lastly, the present study only focused on phytoplankton communities which sit at the base of the food chain powering the biological system in saltwork ponds. For a better understanding of the microorganisms that thrive in the Kenyan saltworks, similar studies investigating the assemblage of other organisms such as zooplankton, halophilic bacteria and macrobenthic fauna are encouraged.

Acknowledgements

The authors would wish to thank Kurawa Ltd for permission to collect samples from their ponds and Kenya Marine and Fisheries Research Institute for the staff and equipment availed during sampling and analysis of the samples collected. The Western Indian Ocean Marine Science Association (WIOMSA) is acknowledged for sponsoring the study through the 'Optimizing *Artemia* Production Technology for Sustainable Aquaculture Development (APTSAD)' project (MASMA/OP/2020/02).

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Review of fisheries and management of sea cucumbers in the Indian Ocean

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Abstract

Several sea cucumber species (Echinodermata: Holothuroidea) are fished, mostly for export of the dried product for Oriental consumers. Previous studies had analysed the historical trends at the world-scale until 2014. In the Western Indian Ocean (WIO) holothurian fisheries have a long history and several programmes have tried to ameliorate their management. Information has been recently gathered through a questionnaire and access to the most recent, yet unpublished available data (2015 to 2021) from different countries, through the evaluation of catches and/or processed product, present management systems, the imports of beche de mer and other products from Indian Ocean (IO) countries into the major market hub of Hong Kong SAR, and the Food and Agricuture Organisation (FAO) yearly statistics. The results are first presented for WIO countries, highlighting recent improvements in management. Imports from 16 WIO countries into the Hong Kong market (2017-2020 data) indicate the importance of the hub. The FAO world statistics are used to present the changes for the last few years, concentrating on the WIO countries. The recent trends show that demand for holothurians is still very high. Inconsistencies in the unit used in the reported statistics (fresh or dry weight) exist, and this needs to be addressed. The national data should be collected at the species level, to be able to follow the changes and the stock status. A regional approach is needed to encourage use of comparable management tools and follow future trends.

Keywords: holothurian, trepang, beche-de-mer, Indian Ocean, fishery, management

http://dx.doi.org/10.4314/wiojms.v21i1.10

Introduction

Several sea cucumber species (Echinodermata: Holothuroidea) are fished, mostly for export of the dried product (Beche-de-mer or trepang) for oriental consumers. Previous studies had analysed the historical trends at the world-scale (Conand, 1998, 2001, 2004, 2006a, 2006b, 2008). A paper by Conand (2017) showed that expansion during the past two decades outpaced management capacity. It was based on the last six years of the FAO capture data and Hong Kong statistics. Hong Kong remains the most important market for the imports and re-exports of the processed products, mainly the dry trepang, but also other more recently developed product forms, which raises difficulties for the analysis of catch trends. The catches are still increasing with additional countries developing export fisheries, changes in species targeted and new products traded. The exploitation is now qualified as 'serial' (Anderson *et al.*, 2011) and 'contagious' (Eriksson *et al*, 2015a). A recent paper by Conand (2018a) focused on the changes during the last decade of the tropical fisheries and trade patterns. These papers confirmed overexploitation in the traditional Indian Ocean and West Pacific export countries and territories. They also showed the active fisheries in several Latin-American countries. Despite the management and conservation issues which have recently received more attention at international, regional and national levels, more measures are still needed at all these levels, to ensure sustainable exploitations of these resources. This paper presents the recent stutus of IO sea cucumber fisheries which have a very long history. Several programmes have tried to improve the management of these fisheries but they appear to be now mostly overexploited (Conand, 2017, 2018a). This up-date presents the most recent data available for this region and discusses the trends and possible management tools.

Material and methods

Previous studies have allowed interactions between scientists studying holothurians within and outside the IO. They include a Western Indian Ocean Marine Science Association MASMA project (2000 – 2014) (see Conand and Muthiga, 2007, 2010; Muthiga and Conand, 2014), several FAO publications and meetings focussed on holothurians (Lovatelli *et al.*, 2004; Toral-Granda *et al.*, 2008; Purcell, 2010; Purcell *et al.*, 2012, 2013; FAO, 2013), and a number of publications have also shown the developing interest in these resources in the tropics (Conand, 2006a, 2006b, 2008; Uthicke *et al.*, 2010; Eriksson *et al.*, 2010, 2015b; Ochiewo *et al.*, 2010; Friedman *et al.*, 2008; Purcell *et al.*, 2013, 2014, 2016; Pakoa and Bertram, 2013; Conand *et al.*, 2016a; Conand, 2017, 2018; Léopold *et al.* 2019). Many publications, generally more focused on one country, are found in the yearly issues of the SPC Beche-de-mer Bulletin.

Information has been recently gathered through a questionnaire conducted to prepare an up-to-date review for the IO region, and through access to most recent, yet unpublished available data (2015 to 2020). The data used comes from: 1) The status of the holothurian fisheries in the different countries through the evaluation of the catches and/or the processed product, the present management and/or its recent changes; 2) The imports of beche-de-mer and other products coming from the IO countries, in the major market hub of Hong Kong SAR, where it appears under several categories; and 3) The FAO yearly statistics on this commodity, by country and globally.
The different data are presented, analysed and a synthesis has been prepared to show the main characteristics for the region and the problems encountered during the last few years. The relevant regulations in the different countries have also been compiled where this information is available.

Results

The situation of each of the fisheries will first be presented from the different countries, by geographic sub-regions in the IO, based on the responses from local experts to the questionnaire and the regional results presented earlier in Conand (2008), FAO (2013), Purcell *et al.* (2013), Eriksson *et al.* (2015b) and several local recent papers. The recent Hong Kong market data and FAO statistics, for the region are analysed. Finally, management tools utilised and the issue of illegal fisheries is discussed.

Situation in the different countries

Countries in WIO: Mainland

Information has been obtained on the present status of sea cucumber fisheries in Egypt, Oman, Tanzania and Mozambique. No recent information from Erythrea, Yemen, United Arab Emirates, Somalia, Kenya and Kenya was available since the syntheses by Conand (2008), FAO (2013), Purcell *et al.* (2013), Muthiga and Conand (2014), and Eriksson *et al.* (2015b).

In the Egyptian Red Sea, following the situation presented in Conand (2008) the resource is considered overexploited; the holothurian populations were monitored at intervals between 2000 and 2016 by Hasan (2019) who showed a decrease in number of species (from 13 to 7), abundance and densities (very low density in 2006 and 2016 after the high density recorded in 2000). The uncontrolled exploitation peaked around 2002; as depletion was observed an official ban was declared in 2001, but it was non effective and lifted in 2002, but re-decreed in 2003, with an increase of illegal fisheries, leading to depletion. Hasan and Johnson (2019) experimented with the restocking of populations of *Holothuria fuscogilva* by transplanting wild-captured juveniles in the Gulf of Aqaba and showed that it can be effective. Juveniles were translocated from a robust population into two sites where population density, growth rate and mortality were monitored for two years; the restocking was successful only at one site.

In Oman, the status of the fishery has often changed in the recent period: unregulated in 2004-2005 (Conand, 2008), overfished in 2010 (Al-Rashdi and Claereboudt, 2010), 3 tons of dried trepang were recorded in 2013 and 5 tons in 2015, banned in 2018 (decision 2018/69 AD prohibiting fishing), with seizure of illegal catches mostly of *H. scabra* and an extension of the ban for two years, in 2019, in order to prepare a management strategy (Al Jufaili *et al.*, 2021). Trials for aquaculture have now started (Al-Rashdi and Claereboudt, 2018; Al-Rashdi *et al.*, 2018).

For Tanzania, the information presented here concerns the mainland, as Zanzibar will be considered with the islands. During the MASMA project (Conand and Muthiga, 2007, Muthiga and Conand, 2014) several studies were conducted (Mgaya and Mmbaga, 2007). Despite a ban being implemented from 2003, some fishers continued to exploit the holothurians and the available data are from Mmbaga (2013, 2015). The main species harvested were *H. scabra* and *H. nobilis ;* the average CPUE (g/fisher/hour) values for *H. nobilis* are 0.5 in Kunduchi, 0.4 in Buyuni, 0.6 in Kitoni (near MPA) and 0.3 in Magemani, with the catch being made mainly by men (64:4, n= 68). A few women were seen collecting firewood and were involved in processing. The situation is complicated by the differences in regulations between the Mainland and Zanzibar, and illegal practices (Mmbaga, 2015). An up-date of the fishery status in this country will be useful. Co-culture trials for seaweed and sea cucumbers were started since 2011, and the synthesis from the experiments and the questions asked is detailed in Kunzmann *et al.* (2018).

In Mozambique, sea cucumbers are known as Magajojo and they are distributed throughout the coast. Sea cucumber fishing is not a new activity, it has been in existence since the 1950s. In 1983, the country earned exports worth ~600 000USD. The catch was 500 t in 1983, 700 t in 1993 (DNP, 1995), and had declined to 6 t in 1995 and 54 t in 1996 (DNP, 1997), and between 2000 and 2004, when the last reports are available, production has been >10 t per year (Conand and Muthiga, 2007). Unfortunately, it is difficult to know to what extent the variability in the catch is due to irregular reporting in the provinces or to over-exploitation, but given that there has been such a dramatic drop in the catch, overexploitation is probably a reasonable explanation.

The fishery intensified in the 1990s resulting in its virtual collapse mainly in the southern area of the country (Maputo and Inhambane) (Abdula, 1998). This fishery is most important and practiced in some Mozambican coastal provinces with rocky substrate like Cabo Delgado and Nampula. Despite not having had a directed fishery, the Chinese have always been the main buyers. Currently the fishery takes place in the northern part of the country (Cabo Delgado and Nampula province), where the fishermen have reported a decrease in catches forcing them to move to deeper zones (Fernando *et al.* 2012) and the main reasons for the decline of sea cucumber was an increase in the number of harvesters, and a change in approach to harvesting such as Tanzanian people in Palma (in northern Cabo Delgado province) using Scuba diving equipment and taking the most valued organisms.

The sea cucumber fishery is mainly smallscale, practiced by men using diving and hand catch on rocky substrates and coral reef shores at depths ranging from 10-20 m. In a few regions of the country children collect *H. scabra* over the seagrass beds (Fernando *et al.*, 2012). The market is driven by demand, with fishing occuring only if there is a buyer. The fishermen generally do not do any kind of processing but sell the wet product to Chinese traders who then procure an export license from the National Fisheries Administration, but in some regions, there are fishermen who boil the sea cucumber. The main commercial species harvested in Mozambique are *H. scabra, H. nobilis, H. fuscogilva, Actinopyga echinites, H. atra* and *Actinopyga mauritiana*. Other species also ocurr in Mozambique waters such as *Actinopyga lecanora, Stichopus chloronotus,* and *S. variegatus* with unknown catch levels, and a few more species remain unidentified (Fernando *et al.*, 2012).

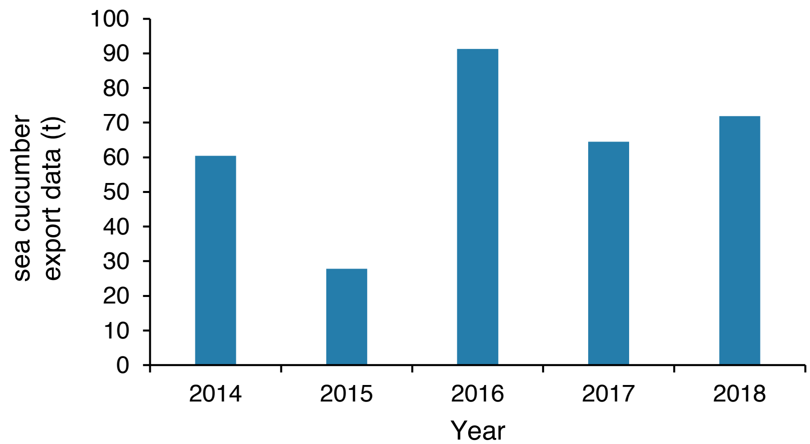

Figure 1. Annual exports of sea cucumbers from Mozambique.

According to Fernando *et al*. (2012) in Nampula and Cabo Delgado provinces the fisherman reported that the periods of high catch are related to the rainy season (October to April) in Nampula and dry season (July to August) at Cabo Delgado. CPUE ranges between 1 to 25 individual per fishermen per fishing day and daily profit ranges from 0.63 - 40. 63 USD. Stocks are severely depleted. The Mozambican legislation (REPMAR- General Regulation on Maritime Fisheries, Decree 89/2020) establishes, in its annex II, a minimum size and weight for the catch of sea cucumber of 20 cm and 250 g, respectively, for all species of live/fresh sea cucumber. It prohibits the harvesting of the three species of *H. scabra, H. lessoni and Thelentota ananas,* including harvest of larvae or juveniles of any species, unless it is for aquaculture. Since some species contract when handled, it is difficult to carry out this control using size, so it is recommended, in parallel, to use the minimum weight (especially for species of the genus *Actinopyga* and for *H. fuscogilva*). In the legislation it is reported that the use of diving equipment is prohibited.

In recent years there has been a marked increase in the request for sanitary licenses for the export of this resource. There are two companies in Mozambique that export sea cucumber and the National Institute of Fishery Inspection (INIP) is responsible for issuing sanitary licenses and has been recording exports for the last 7 years. The exports are shown in Figure 1 (2014-2018).

There is no management plan for the sea cucumber fishery in Mozambique and no cooperation between stakeholders.

Countries in WIO: Islands

Recent information has been obtained from most islands, except Comoros which had been presented in FAO (2013).

Maldives. The sea cucumber fishery (locally known as "Huifilan'daa") is recent as it started in the mid-1980, with the export of a trial shipment of about 30 kg of prickly redfish (*Thelenota ananas*) to Singapore (Naeen and Ahusan, pers comm). They explain that the number of foreign buyers as well as the exporters engaged locally, as well as the number of species harvested increased as the fishery progressed. The diversity of holothurians has been described by Muthiga (2008), and Ducarme (2015, 2016). Nine species are now commercially harvested, processed and exported. The processed sea cucumbers are either taken to Malé, or directly to the exporters or to the agents of some of the major exporters based in the islands. Maldives bêche-de-mer was exported mainly to Singapore and Hong Kong SAR. However, in recent years, Sri Lanka has also become an importer of Maldives bêche-de-mer. The overall exports over the period (1988-2018) show a first phase of very sharp increase, followed by an important drop in 1994, then less significant variations. Since 2014, the export quantities ranged between 50 and 100 mt. It is clear that the high-valued sea cucumber stocks are heavily over-fished and drastic measures are required if the resource is to recover. At present, the only data available is from the export statistics, which report sea cucumber shipments as "dried sea cucumbers", "Fresh or frozen sea cucumbers", without categorizing them by species, or any differentiation between the wild-caught and farmed animals. Therefore it is strongly recommended that the species composition

Figure 2. Women collecting sea cucumbers at low tide on the easy coast of Zanzibar (Photo credit: M de la Torre-Castro).

of the exports be determined and tracked over time to identify shifts in harvested species.

Zanzibar. The situation in Zanzibar is complex, as the Tanzanian legislation is different for the islands and the Mainland. The fishery has a rather long history, starting more than a century ago; it declined as elsewhere during the World Wars and then Chinese traders initiated more intense harvesting during the 1960s (Eriksson *et al.*, 2010, 2015c). The organisation of the fishery ranges from small-scale near-shore activities (Fig. 2) to more mobile industrialised avtivities using scuba-diving (Eriksson *et al.*, 2010, 2012a.). Comparisons between the small-scale fisheries in Zanzibar (open access) and Mayotte (precautionary closure) have been presented by Eriksson *et al.* (2015c). These authors observe that in terms of resource value, catch and exports differ markedly, with more than 30 species caught in Zanzibar, and 90 % of the catch composed of low-value species. More recently, Eggersten *et al.* (2020) made a more precise comparison between these islands with an analysis of the economic value under contrasting management regimes, using three species, *H nobilis* (high value), *B. atra* (medium value) and *H atra* (low value). They conclude that these fisheries have high potential for providing social-economic benefits if properly managed.

According to to data from the Department of Fisheries Development exports of holothurians were 35.26 t in 2015, 26.39 t in 2016, and 29.62 t in 2017. The recent data does not however provide information on species or value.

In 2015, the Government of Zanzibar, through the Department of Fisheries Development issued directives that prohibited sea cucumber fisheries and promoted farming of sea cucumbers in Zanzibar. Thus, export of holothurians from farms is allowed. The lack of tracebility initially meant that it was difficult to ensure that the farms are not just transit points for fished specimens. However, random spot checks on farms by fisheries officers have helped to control this gap.

As a result of implementation of the Zanzibar mariculture sector development program a government-run multi-species hatchery was inaugurated in 2018, which aims at producing 1 million sea cucumber juveniles annually (Menezes, 2018), alongside finfish and crab production. Between 2017 and 2018 there were thirteen spawning trials producing over 7 million eggs (Y. Yussuf pers. comm.). Hatching rate ranged between 39 to 87%. Average survival rate of juveniles was 2.2%. Small scale trials on ocean based nursery system in rearing post-settled juveniles of *H. scabra* using floating hapas have been successful (Yussuf and Yahya, 2021). Some biological studies have been carried out in recent years, specifically on *H. scabra* (Yussuf and Yahya 2020). Trials are currently ongoing in Pemba Island, at small scale, on integrating seaweed and sea cucumber farming.

Mayotte. The commercially important species in Mayotte have been described in numerous papers, some of them recently (Pouget, 2004, 2005; Eriksson *et al*. 2012b, 2015c; Ducarme, 2018; Mulochau, 2018a). Following previous publications, the most recent inventory of the Echinoderms includes 45 holothurians species (Ducarme, 2018). Several studies by Eriksson *et al*. (2012a, 2012b, 2015b, 2015c) and Mulochau (2018a) have delt with their ecology, fishery and conservation. A moratorium on the fishery has been in place since 2004 (Prefecture de Mayotte 2004), but some illegal fishing has probably occurred.

Seychelles. Reports of sea cucumber fisheries in Seychelles date back to the 1800s. However, the fishery remained small-scale until the late 1990s when a rapid increase in catch was observed (Aumeeruddy and Conand, 2008). The increase in demand and price for "bêche-de-mer" led to an evolution in the fishery from the collector type by wading in shallow areas, to the use of scuba gears as fishers moved to deeper waters (Aumeeruddy and Payet, 2004). Historically in Seychelles, the targeted species were the black teatfish *H. nobilis*, the white teatfish *H. fuscogilva*, the flower teatfish *H. Microthele* sp., the blackfish *Holothuria atra* and the prickly redfish *Thelonota ananas* (Aumeeruddy and Payet, 2004).

In the late 1990s signs of decline in sea cucumber catches were observed and in 1999 the Seychelles Fishing Authority (SFA) introduced management measures for the fishery (Fisheries (Amendment) Regulations, 1999). Measures included issuing fishing and processing licenses and a limit in the number of divers per license. However, the management measures were precautionary due to a lack of baseline information on the status of the stock. The established measures were regarded as insufficient and a lack of catch data from fishers raised concerns over potential overexploitation of the resource. This led to the temporary closure of the fishery in 2001. The fishery was reopened the same year with new management measures. The number of fishing licenses was capped at 25, logbook and receipt books were issued to fishers and processors and catch reports were to be submitted to the SFA on a monthly basis (Aumeeruddy and Payet, 2004).

To improve upon the established measures, the SFA carried out a fishery independent resource assessment of the holothurian population between 2004 and 2005 (Aumeeruddy *et al.*, 2005). The assessment indicated that some species like sandfish *H. scabra* and redsurf *Actinopyga mauritania* were overexploited while the white teatfish and flower teatfish were fully exploited, and some other species were underexploited. One of these underexploited species was the high-valued black teatfish. Recommendations for management measures were made which included controls of the fishing effort to not exceed the recommended Total Allowable Catch (TAC), control in fishing effort for high value species close to the main islands, proposal to establish minimum size limits and the implementation of continuous periodic surveys to regularly assess the performance of management strategies (Aumeeruddy *et al.*, 2005). Some of the recommendations such as the TAC limits were highly contested by stakeholders involved in the fishery and were not adopted in the reviewed management measures.

As part of a project examining the sea cucumber fishery ecology, a fisheries independent stock assessment survey was carried out during 2011 – 2013. Abundance of some species like lollyfish and black teatfish were found to have declined when compared to the 2005 SFA survey. However, the blackfish (*A. miliaris*) indicated an increased abundance (Koike, 2017).

Another stock assessment was carried out in 2012 by MRAG, based upon the catch and effort data reported by fishers to SFA via the logbooks. The assessment comprised of a spatially disaggregated analysis of catch and effort data, the creation of a standardized commercial Catch Per Unit Effort (CPUE) series which was used as an indicator for relative abundance, and using surplus production models to estimate biomass and Maximum Sustainable Yield (MSY). From this assessment recommendations for management were made. These included, introducing catch limits based on the recommended TAC suggested in 2005 and improvement in the collection of catch and effort data to enhance the quality of data for more comprehensive stock assessments in the future. Following the assessment, new management measures were introduced, including minimum landing weight for white teatfish, flower teatfish, prickly redfish and black teatfish (MRAG, 2012).

Table 1. Breakdown of quota distribution in Seychelles, per species and per vessel.

	Flower teatfish	Prickly redfish	White teatfish	Total
Pieces per species	281,250	37,500	56,250	375,000
Pieces per vessel	11,250	1,500	2,250	15,000

Following the recommendation from the 2012 stock assessment, an agreement was reached between SFA and the stakeholders to review the management plan after three years. This led to another fisheries dependent stock assessment by MRAG in 2017. Evidence of significant population decline was observed especially for black teatfish in some areas (MRAG, 2017). Based on this finding, further measures were introduced including, reduction in the fishing season from nine months to eight months, a complete ban on fishing of the black teatfish and a defined TAC for three species and quota allocation.

The TAC was set in reference to the 2016 season, at a total of 375,000 pieces of sea cucumber. Under this TAC, allocations were only made on three harvestable sea cucumber species. The TAC and non-transferable quota were first introduced during the 2017/2018 fishing season. The breakdown of the TAC is highlighted in Table 1.

As per the Seychelles Cabinet Memorandum 2017- 'New Management Measures for Sea Cucumber Fishery, the TAC is to be reviewed after three years or three consecutive opening seasons.' This exercice was carried out by the SFA in 2021, following the 2020/2021 fishing season. Table 2 and Figure 3 highlight the breakdown of the TAC consumed per species from the last four seasons (i.e., spanning from 2017 to 2021).

The R-squared (R^2) value is displayed for each species. The R^2 is a statistical measure that represents the proportion of the variance for a dependent variable (% Quota consumed) that is explained by an independent variable (Season).

Overall, the average quota for the White Teatfish and Prickly Redfish were well consumed. However, that of the Flower Teatfish (the species with the highest allocated quota) has been highly variable and averaging only 78 % over the period under review. The exact reasons for this remain unclear, although one possible factor could be the reduced natural abundance of this species. In order to determine the cause of this low quota intake for this species would require a more comprehensive analysis on a vessel-by-vessel basis.

A research project was conducted in 2017 and 2018 to strenghten the sea cucumber fishery co-management process through strong participation of fishers, skippers, and exporting companies in the research activities (Léopold and Govinden 2018). First, as part of a genetic study, 16, 19, and 25 microsatellite loci were isolated from White Teatfish, Flower Teatfish, and Black Teatfish DNA libraries, respectively (Oury et al. 2019a, 2019b). These loci were used for assessing genetic diversity and population structure of these taxa over the Mahé plateau and the Amirantes islands. The Flower Teatfish was structured in one single population (n=437 samples, 8 sites) while small, yet significant, genetic difference was observed among White Teatfish samples (n=348 samples, 7 sites). That spatial structure likely arose from the low abundance of White Teatfish over the fishing areas, which called for spatial management of White Teatfish resources that are protected by Appendix II of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). Second, the size at sexual maturity of the Flower Teatfish was estimated at 30.3 cm through a reproductive biology survey and macroscopic observations of the gonads of 93 specimens in the SFA lab (Cahuzac et al. 2019), which was informative to discuss a minimum catch size of that species. Third experimental dive fishing and commercial dive profiles showed that underwater visibility and depth impact catch rates, and, consequently, should be recorded in commercial logbooks for estimating CPUE index. Finally, an electronic logbook and an online database were experimented by SFA to facilitate fishery monitoring and fishery-dependent stock assessment. However, this monitoring system has not been deployed at the fishery scale yet due to technical constraints and acceptability issues among some skippers of the fishery.

Following lengthy discussions and presentations to the SFA Board of Directors, sea cucumber stakeholders and the Minister responsible for Fisheries, a decision was taken in 2021 to reduce the current allocated

Table 2. Breakdown of the consumed TAC in Seychelles, per species from the last four seasons. PR- Prickly Redfish, WT- White Teatfish, FT- Flower Teatfish

Season	Consumed PR	% Consumed	Consumed WT	% Consumed	ConsumedFT	% Consumed
2017-2018	32,130	86	47,862	85	193,103	69
2018-2019	37,554	100	50,249	89	177,929	63
2019-2020	37,061	99	66,491	118	268,047	95
2020-2021	38,494	103	53,241	95	243,169	86
Average	**36,310**	**97**	**54,461**	**97**	**220,562**	**78**

quota of the Flower Teatfish by 10 %. Therefore, the allocated quota set for the 2021-2022 sea cucumber fishing season is prescribed below in Table 3. It was also agreed that the quota and overall fishery would be reviewed following a scheduled independant stock assessment of sea cucumbers planned for late 2021 and early 2022. The Seychelles has adopted a co-management approach to the independent stock assessment exercise. Sea cucumber commercial divers participated in the data collection and data analysis segments of the survey, after undergoing extensive training by the SFA. Since fishing activities are mainly concentrated on the Mahé Plateau and the Amirantes groups, the same sites as for the 2005 assessment were surveyed (Fig. 4). This allowed a comparison over a 16-year time period. The specific survey points at each of the sites were selected using random stratified sampling where the Mahé Plateau and the Amirantes were stratified by habitat types, depths and the limits of the marine protected areas. Power analysis was then used to calculate the number of sites for each identified stratum based on fishing pressure and the sites were randomly selected within each stratum.

The survey was carried out in two phases; the first part was completed in November 2021, and the second in March 2022. The analysis and results of the stock assessment were recently finalized and presented to the Board of Directors of the SFA, as well as to the Ministry of Fisheries. The ultimate fate of the Seychelles' sea cucumber fishery will be disclosed to the sector through a National Sakeholder Workshop being planned for the first week of September 2022.

La Réunion. Several studies have been conducted on the biology and ecology of the main holothurians from La Reunion. An inventory of the 38 holothurian species has been published (Conand *et al.*, 2010;

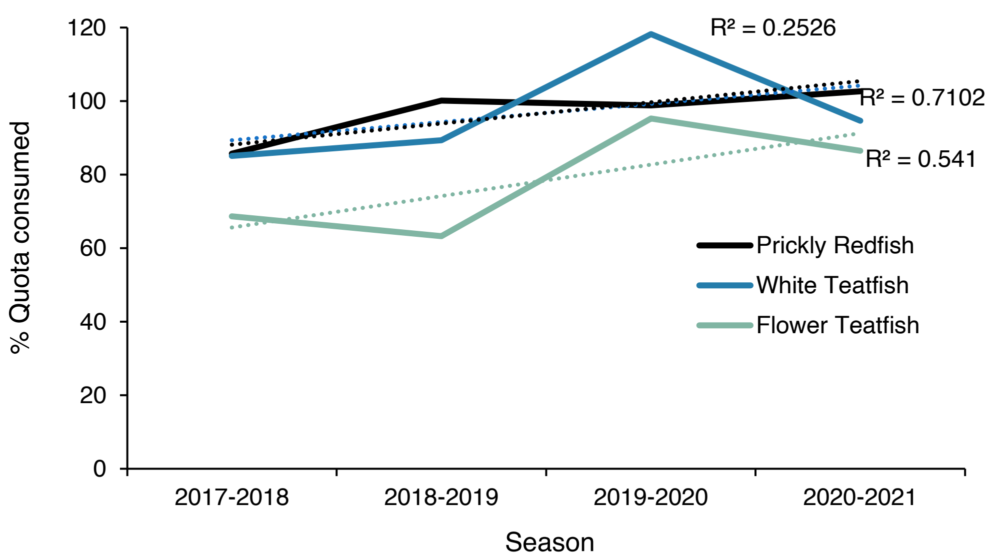

Figure 3. Line graph displaying the percentage quota in Seychelles consumed per species per season. Also displayed is the R^2 value for each species.

Table 3. Breakdown of quota distribution in Seychelles, per species and per vessel following a 10 % reduction of the Flower Teatfish in 2021.

	Flower Teatfish	Prickly Teatfish	White Teatfish	Total
Pieces per species	253,125	37,500	56,250	346,875
Pieces per vessel	10,125	1,500	2,250	13,875

Conand *et al.*, 2018); detailed information is also provided in a book on the echinoderms (Conand *et al.*, 2016a). Given the small size of the reefs, no fishery is authorised. Very recently, illegal fishing activities have been observed within the Marine Nature Reserve of Reunion Island. https://www.facebook.com/groups/1736481600015091/posts/2972490759747496

Eparses Islands (Scattered Islands, Canal du Mozambique). These isolated and inhabited islands are now a Natural Reserve and good sites for biodiversity inventories, but are also vulnerable to illegal fisheries (Conand *et al.*, 2016b). Several scientific programmes were conducted during the recent years in collaboration with the french TAAF. Inventories and/or abundances of the holothurians have been made for the different islands, such as for the Glorieuses Islands and Geyser Bank (Conand *et al.*, 2013a; Mulochau and Gigou, 2017; Mulochau, 2018b), Europa (Conand *et al.*, 2013b), and Juan de Nova (Mulochau *et al.*, 2015). A global inventory for the Echinoderms of the Eparses Islands includes the occurrence and habitats of 31 holothurians (Conand *et al.*, 2016b). Discussions between the Goverments of Madagascar and France for agreements and cooperation have taken place. Recent observations of illegal fishing (2021, pers. com.) have been made in the Glorieuses, Juan de Nova and Bassas da India. The TAAF regularly make fishing seizures around the Eparses Islands. Some catches were made on the Geyser Bank (Mulochau, 2018b), but the recent creation of the 'Réserve Naturelle Marine des Glorieuses' by TAAF will allow better monitoring.

Mauritius and outer islands. Sea cucumbers (*barbara* in Mauritian creole) have been exploited traditionally by the local Chinese communities for domestic consumption. The commercial exploitation started in late 2005 and after three years of commercial harvest

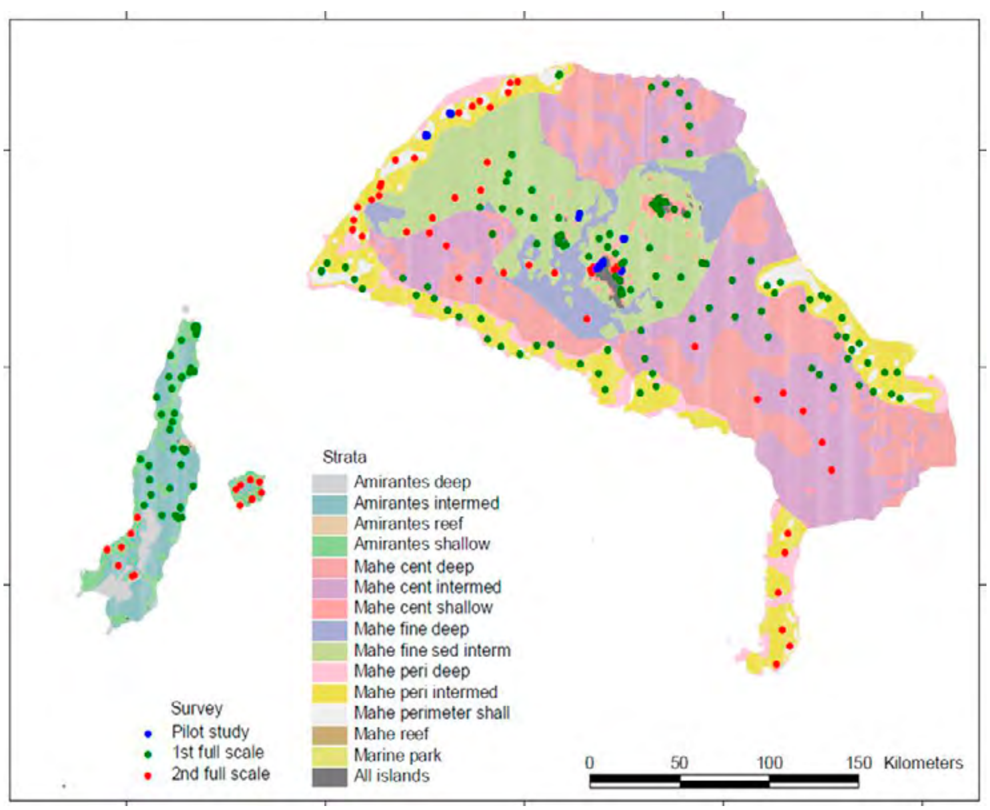

Figure 4. Sampling sites selected in Seychelles, based on the different stratifications for the 2005 resources assessment survey (Aumeeruddy *et al.*, 2005).

Table 4. Temporal relative abundances of nine species of sea cucumber around Mauritius and Outer Island lagoons. Note: data from 1998 are being used as measure of the virgin stock for Mauritius.

	Relative abundances in the lagoon of Mauritius				Relative abundances in the lagoon of Rodrigues		Relative abundances in the lagoon of Agalega	
	1998	**2011**	**2013**	**2016**	**2007**	**2016**	**2008**	**2016**
	n = 1129	n = 3411	n = 7488	n = 9963	n = 2734	n = 3397	n = 944	n = 2328
Average denssity (ind ha^{-1})	564	487	602	2640	854	1697	710	1290
Number of sites surveyed	2	16	23	12	16	10	8	8
Species composition (%):								
Holothuria scabra	0.44	0.00	0.01	0.00	0.04	0.00	0.00	0.00
Actinopyga echinites	3.45	1.08	0.75	0.10	0.15	0.09	0.00	0.00
Stichopus chloronotus	8.24	6.39	11.77	0.30	0.95	5.50	99.70	99.90
Thelenota ananas	0.35	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Stichopus variegatus	0.09	0.18	0.00	0.00	0.07	0.00	0.00	0.00
Bohadschia vitiensis	36.49	3.22	4.73	1.70	0.37	0.98	0.00	0.00
Holothuria atra	24.89	29.14	51.94	12.20	91.19	72.70	0.30	0.10
Holothuria leucospilota	19.31	11.93	3.21	85.60	7.21	20.70	0.00	0.00
Holothuria pervicax	6.73	1.99	1.54	0.00	0.04	0.03	0.00	0.00
Sources	Luchmun *et al.* (2001)	Lampe (2013)	Lampe *et al.* (2014)	AFRC (2016a)	AFRC (2007)	AFRC (2016b)	AFRC (2008)	AFRC (2016c)

Note: High commercial value (*Holothuria scabra*); Medium commercial value (*Actinopyga echinites*, *Stichopus chloronotus*, *Thelenota ananas*, *Stichopus variegatus*); Low commercial value (*Bohadschia vitiensis*, *Holothuria atra*, *Holothuria leucospilota*, *Holothuria pervicax*)

destined for export market, the resource was considered as depleted. The commercial exploitation of sea cucumber resources started with the issuance of export permits to two companies in 2006 for a six-month trial period. Following this trial period, at least 12 other companies benefited from an export permit for the same duration and under strict quota conditions set and monitored by the government. The first full stock assessment of sea cucumber resource in Mauritius was conducted in February 2007 by staff of Albion Fisheries Research Centre amid concerns of over-exploitation of the stock. Following the surveys, a Total Allowable Catch (TAC) of 550 tons (wet weight) was equally distributed among 5 companies for a six-month period and a regulation was promulgated under the Fisheries and Marine Resources Regulations of 2008, stating a minimum size limit of 15 cm for all species and a three-month closed season from 01st January to 31st March. The total production reported from all operators in 2007 was 408 tons (wet weight). Overall the production data (in dry weight) from 2006, 2007, 2008 and 2009 were 94 t, 50 t, 21 t and 5 t respectively (AFRC, unpublished data), amounted to approximately USD 1.8 million and the main export markets were Hong Kong, Singapore and Taiwan.

A new regulation came into force in October 2009 introducing a two-year moratorium period to ban the collection of sea cucumbers in Mauritian waters, including the outer islands (Rodrigues, Agalega and St Brandon). Mauritius was the first country in the Western Indian Ocean to introduce and enforce a moratorium on sea cucumber fishing. A second moratorium period was introduced from March 2012 to February 2016 and following stock assessment surveys carried out in August 2016 in Mauritian waters, the moratorium was extended for another 5 years.

Table 4 shows a summary of the relative abundances of sea cucumber from stock assessment surveys carried out during and after the exploitation period in Mauritius and the outer islands. The surveys conducted by AFRC employed the standardised belt transect method comprising a main transect of 100 m and five 20 m sub-transects. Sea cucumbers were enumerated within a 2.5 m belt on either side of the transect line, with a total surface area surveyed of 1000 m². Data from the lagoons around Mauritius were compared with data from Luchmun *et al.* (2001), used here as virgin stock since the surveys were done prior to commercial exploitation. The diversity had also been assessed in shallow sites by Lampe-Ramdoo *et al*. (2014). Table 4 shows a net loss in diversity of sea cucumber in Mauritian lagoons following a 4-year period of intensive fishery. A Simpson Diversity Index (SDI) was applied to the data in 1998 and 2016, showing a net drop in SDI from 0.75 (in 1998) to 0.25 (in 2016). Sea cucumber species of high to medium commercial value, such as *H. scabra*, *Thelenota ananas* and *Stichopus herrmani* have almost completely disappeared during the exploitation period and had not recovered by 2016 (the last surveys done). Species such as *H. atra*, *H. leucospilota* and *S. chloronotus* were among the most abundant in the years following the closure of commercial exploitation. Interestingly, these species are among the few sea cucumber species that can reproduce asexually through fission. The increase in average density (see Table 4) seems to indicate that these species have taken over the habitats left by the depleted ones.

Baseline data, prior to onset of commercial harvest in 2006, from the lagoons of Rodrigues Island (Rowe and Richmond, 2004) indicated 29 species of Holothurioidea, of which 18 were of commercial importance. Surveys done in July and August 2006 (Mrowicki, 2006), reported 8 species of sea cucumber (6 being commercially important) with *H. atra* (72 %), *H. leucospilota* (11%) and *S. chloronotus* (11%) being the most dominant and widespread species. Subsequent surveys done by AFRC staff in 2007 and 2016 (see Table 4) showed the same trends as in Mauritius, with a decrease in diversity from 18 commercially important species in 2004 to 6 in 2016. The most thriving species, after the exploitation from 2006 to 2009 and ban imposed in 2009, were those that could also reproduce by asexual reproduction. They were found to occupy all the different habitat types surveyed (sand, rock/coral rubbles, coral, seagrass and macroalgae).

Three surveys were done from 2006 to 2019 on the diversity and stock assessment of sea cucumber in the lagoons and outer reef flats of St Brandon. In 2006, the Ministry of Agro Industry and Fisheries commissioned a 6-day trip to St Brandon to survey the diversity of sea cucumber prior to allowing one company to exploit the resource. Ten species of sea cucumber were recorded from north to south of the atoll; these were *H. nobilis* (high value species); *Actinopyga mauritiana*, *A. miliaris*, *Thelenota ananas* and *Stichopus chloronotus* (medium value species); and *A. echinites*, *Bohadshia marmorata*, *H. atra*, *H. fuscopunctata* and *H. pervicax* (low value species). Three species were most abundant, namely *S. chloronotus* (24,000 ind/ha), *H. atra* (20,000 ind/ha) and, *A. miliaris* (6000 ind/ha) although they were patchily distributed. Eeckhaut (2010) conducted a survey on the atoll in view of determining its suitability for sea cucumber farming. Eleven species of sea cucumber were recorded, namely *H. nobilis* and *H. fuscogilva* (high value species); *T. ananas*, *S. chloronotus*, *S. hermanni*, *A.miliaris* and *A. mauritiana* (medium value species); and *A. echinites*, *H. atra*, *Holothuria sp*. and *Bohadshia vitiensis* (low value species). The most abundant species were *H. atra* (50,000 ind/ha), *S. chloronotus* (patches of 20,000 ind/ha), and *B. vitiensis* (2000 ind/ha). The latest survey was done in March 2019 and commissioned by the Mauritian Wildlife

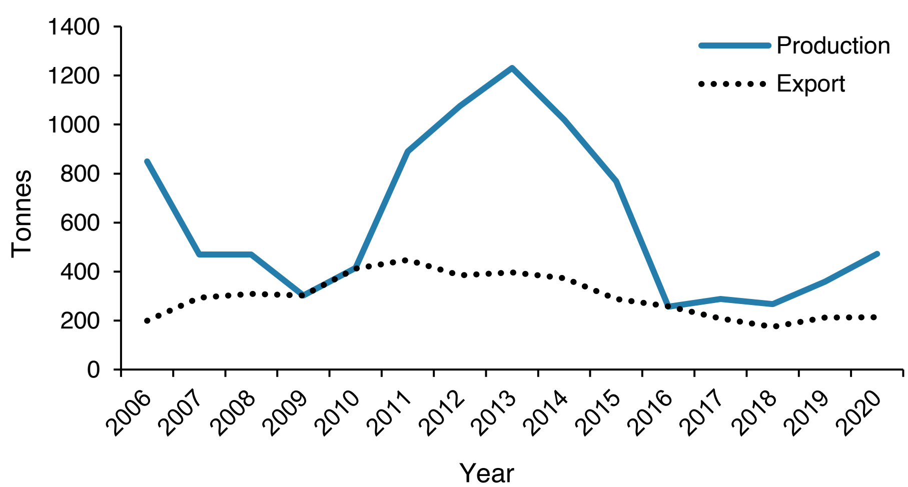

Figure 5. Recent yearly volumes of trepang production (data from MPEB) and exports from Madagascar (data from ASH).

Foundation (Ramah, 2019). Twenty-five sites on the atoll were surveyed and five species of sea cucumber were recorded, namely, *H. nobilis* (high value species); *S. chloronotus* (medium value species); and *H. atra*, *A. echinites* and *Bohadshia atra* (low value species). Two species were most abundant; *S. chloronotus* (25,000 ind/ha) and, *H. atra* (15,000 ind/ha). High value species such as *H. scabra* (sandfish) were not observed in all the surveys since they were conducted only during the day. Habitats suitable for sandfish were mostly observed on the northern part of the atoll (Eeckhaut, 2010).

Surveys done in 2008 and 2016, showed that the lagoons of Agalega have one predominant species, *S. chloronotus* (Table 4). Precaution should be taken while interpreting these data since no night surveys were done and it is possible that cryptic and nocturnal species were not recorded. The average density of *S. chloronotus* was 710 ind/ha in 2008 and increased to 1300 ind/ha in 2016, which is not as high as at St Brandon. *S. chloronotus* did not show any particular habitat preferences since it was observed on almost all substrates (coral rubbles, sand, and macroalgae). It is worth noting that local residents reported that about 30 t (wet weight) of sea cucumber were harvested in 2007 and since then, only the abovementioned species has recovered (AFRC 2008, 2016).

The sea cucumber stock in Mauritius and outer islands suffered from the 3 years of commercial exploitation. A drop in sea cucumber diversity was observed in all locations during the years following the open season. Some species have not recovered following a decade long moratorium and will probably never recover. An interesting observation is the increase in numbers of species such as *S. chloronotus* and *H. atra*, which, due to their ability to reproduce asexually are not affected by depensation. They can therefore proliferate quickly and occupy the habitats left by the other species that were locally depleted due to overfishing. Indeed, *S. chloronotus*, which is a species generally observed on coral rubbles of reef flats and upper slopes, was observed to occupy different habitat types such as coral rubbles, sandy patches, macroalga and seagrass patches.

Madagascar. The fishery is artisanal but the stocks of the high value species are mostly depleted and the fishers have adapted to find new sites and/or new species (Conand, 2008; Purcell *et al.*, 2013). The recent data on the production and the exports are shown on Figure 5.

A slight increase of the exports was observed from 2006 to 2011, where the maximum export level of trepang was reached (448 t). From 2012, a continuous decrease of trepang exports was recorded to 174 t in 2018 (source: Autorité Sanitaire Halieutique ASH). In apparent contract, a large increase of trepang production was also recorded from 2011 and a maximum production of 1,231 tonnes was reached in 2013. This increase in production is linked to the illegal fishing of sea cucumbers (the use of scuba diving) especially in the northeast region of Madagascar. A slow increase is noted from 2019 and 2020.

The differences between the production data (source: Service Statistique du MPEB) and the export data (ASH) could be explained by local sales and illegal exports.

Countries in East IO

Recent information and contributions have been submitted by the co-authors for Sri Lanka, India and Western Australia.

Sri Lanka. The sea cucumber fishery has a long history in Sri Lanka (Adithiya, 1969; Conand, 2008) being seasonal in relation to mooson winds, and several recent papers show its present importance (Dissanayake and Wijayarathne, 2007; Dissanayake and Stefansson, 2010, 2012; Nishanthan *et al.*, 2019, Dalpathadu, 2021).

An underwater visual census (UVC) carried out on the northwest and east coasts of Sri Lanka in 2008 reported the presence of 25 sea cucumber species. The results of this survey revealed that the overall average density (±SD) of sea cucumbers was higher in the northwest (350 ±648 ind ha^{-1}) than in the east (90 ± 130 ind ha^{-1}) and low-value species were predominant in both survey areas (79 ±125 ind ha^{-1} in the east, 244±488 in the northwest) when compared with the medium (10±34 ind ha^{-1}, 105±175 ind ha^{-1}, respectively) and highvalue category (<2 ind ha^{-1}, Dissanayake and Stefansson, 2010).

The detailed study by Nishanthan *et al.* (2019) based on data collected from November 2015 to January 2017, shows the fishing pattern, and the social and economic characteristics of sea cucumber fisher communities in the north, north-west and north-east regions of Sri Lanka. In total nine sea cucumber species are landed in these areas, using three methods: diving (SCUBA and breath-hold); gleaning; and surrounding nets. SCUBA diving is the dominant method and divers do both day and night fishing, reporting the highest catch rates (CPUE ± SD in numbers/person/day) compared to the other fishers. Gleaning is entirely carried out by fisherwomen in the northern region, while 28 surrounding nets are used targeting the low-value *Stichopus naso*. The CPUE varied with species, region and fishing method. *Bohadschia vitiensis* made up the highest percentage contribution (61.3 %) to the total sea cucumber landings during the study period. The high value teatfish species (*H. fuscogilva* and *H. nobilis*) are not reported any longer. Among all fishers, SCUBA divers reported the highest average net monthly income. Men play a dominant role in sea cucumber fishing (98 %), processing (99 %) and marketing (100%). Fishermen are in the age range of 25–67 yrs, having only primary education and mainly involved in the sea cucumber industry. The sea cucumber value chain consists of fishers (~ 2000), middlemen (12), processors (16), and exporters (7). A significant increase in overall price development from fishers to the exporters was reported for all nine species. Fishers received a proportionally higher share of the end-market price mainly for low-value *Bohadschia* spp. Although fishers receive disproportionately low returns compared to others, 47.8 % of fishers showed high satisfaction towards the prices they received from buyers. However, 83.6 % of fishers are not satisfied with existing management measures (Table 5). This information is important to strengthen the existing management measures in Sri Lanka.

Recently, Dalpathadu (2021) has evaluated the fisheries in the coastal waters of Trincomalee district, Eastern Sri Lanka, showing the population depletion of most species in the shallow waters. Logbook record-based surveys were conducted to cover the fishing season in 2019. The harvest was collected by skin diving from shallow nearshore waters and Scuba diving from coastal waters about 5 km away from the shore. *Thelenota anax* was the dominant species in the catch from both fishing grounds. The study revealed that the stock of *T. anax* along with other species in these coastal waters might be moving towards extinction if the fishery prevails without proper management.

A detailed study by Bondaroff (2021) analyses highly organised poaching and smuggling of sea cucumbers, which has been on the rise in Sri Lanka and the south of India. The 120 incidents analysed reveal 502 arrests.

Table 5. Holothurian fishery in Sri Lanka. A: informations from the north, north-west and north-east coast of Sri Lanka from November 2015 to January 2017. B: variation in CPUE for the different species (number/person/day), from the north, north-west and north-east coast of Sri Lanka from November 2015 to January 2017. Modified from Nishantan *et al.* (2019).

A_ Parameter	**Northwest**		**North**				**NorthEast**
	SCUBA	Snorkelling	SCUBA	Snorkelling	Gleaning	Netting	SCUBA
Fishing Effort	335	102	68	305	27	28	372
CPUE (nb/ person/day)	88	43	100	31	14	35	558
Distance to fishing (km)	29	15	10	8	2	16	31
Total income /person (US$) /day	68.9	23.2	21	12.1	5.1	9.8	80.7
nb of fishing days / month	14	15	14	15	4	20	15
Fishing months / Y	5	4.5	4.5	4.6	5	5	4.5
Net income /month (US$) /person	517	288	210	205	31	107	888
Total landings (Nbx 10^3)	2 926		2 017				12 303

B. Species	**North-West**		**North**				**North-East**
	SCUBA	Snorkelling	SCUBA	Snorkelling	Gleaning	Netting	SCUBA
H. scabra	109	43	3	10	14		
H. spinifera	25		17	4			72
T. anax	41						
S. chloronotus	3						
H. atra	1		32	6			
B. marmorata	3		2				
B. sp l	3		2				
B. vitiensis	1						486
S. naso	1		44	11		35	

He concludes that it should be treated as a form of transnational organised crime. Monitoring and enforcement efforts should be expanded and intergovernmental and interagency cooperation increased.

Finally, sandfish farming activities are rapidly developing in northern Sri Lanka. There are 2-3 functioning hatcheries at present, and two sea cucumber villages are proposed to be established. Sandfish fattening activities are currently carried out using lagoon pens (size range from 1 – 10 Ha) and there are 100 – 150 being operated at present. However, conflicts between sea cucumber farmers and fishers are very common as farmers collect small-sized sandfish from the natural environment. However, with the establishment of the proposed sea cucumber villages, more hatcheries will be established, and then the use of wild-collected individuals in farming activities will be banned.

India. Holothurians are mainly distributed in the Gulf of Mannar and Palk Bay, Andaman and Nicobar, Lakshadweep, Gulf of Kutch and along the mainland coast of India in variable numbers (Asha *et al*., 2019). The fishery was artisanal in nature and was introduced by the Chinese more than a thousand years ago in the Gulf of Mannar and Palk Bay (Hornell, 1917). The industry comprised fishermen, who are divers, the processors who act as middlemen, and the exporters. The fishery was mainly dependent on high valued *H. scabra* and the medium valued *H. spinifera*, and occasionally on medium valued *Actinopyga miliaris*, *A. echinites* and *Stichopus hermanni* based on their availability (Asha *et al*., 2017). Numerous studies have also been conducted by James (2004) after the industry came to a standstill when the Ministry of Environment, Forests and Climate Change of India imposed a total ban on the sea cucumber fishery in June 2001 and listed all holothurians as protected animals under the Schedule I of the Indian Wildlife, which severely affected the livelihood of the poor fishers of this region. A reliable long-term estimate was not available on the exploited as well as potential stocks of holothurians in Indian waters. A short-term survey conducted by the Zoological Survey of India (ZSI) along the Gulf of Mannar in 2007 reported the availability of seven species, with *H. atra* as the dominant one. Another survey conducted by the ZSI in 2011-12 also indicated the occurrence of nine and seven species in the Gulf of Mannar and Palk Bay, respectively (Venkataraman *et al.* 2012). A fundamental barrier to improved knowledge and management of sea cucumbers was the lack of data on population abundance/biomass and basic biological parameters of most of the species. The large dependent coastal population in the Gulf of Mannar and Palk Bay, high value of the species and the ease with which the sea cucumbers can be collected, allows illegal harvest, leading to potential biological and ecological vulnerability of the stocks.

Fourteen years of the sea cucumber fishing ban has caused considerable changes in the stock status. Scientifically supported information on the population characteristics of sea cucumber resources in the Gulf of Mannar and Palk Bay, two geographically distinct ecosystems, was very much needed for formulating sustainable management measures for these resources and the recent study by Asha *et al.* (2019) was aimed to fill the gap in this respect. It indicated variation in status and structure of sea cucumber resources in the Gulf of Mannar and Palk Bay. The difference is mainly linked to the unique environmental characteristics of both the ecosystems; hence, ecosystem-based management approaches should be given more emphasis while formulating conservation measures for these resources in the region.

The stock status of the sea cucumber population was assessed both by trawl and dive surveys following standard methods on a random basis. Comparatively, a higher average density of total sea cucumbers was estimated in the Gulf of Mannar (3853±152.3 nos. ha-1) than in Palk Bay (2428.5±504.6 nos. ha-1). The species diversity, average length and weight of individual species were also higher in the Gulf of Mannar. While comparing with previous post-ban surveys, this study also indicated wide fluctuation in the sea cucumber population status. Hence it is recommended to conduct fishery-independent and dependent surveys of the sea cucumber stock at a regular interval as per the standard survey and methodologies for the development of time series data on population metrics from this region. It is also suggested that long-term mark-recapture studies are implemented to assess the growth, dispersal, mortality rates and longevity of sea cucumber resources for better clarification on population structure in order to formulate management measures for the conservation and sustainable utilisation of the resources (Asha *et al*., 2019).

The poaching and smuggling that is increasing in Sri Lanka and the south of India has been analysed by Bondaroff (2021). It is recommended that cooperation between countries is necessary for effective management.

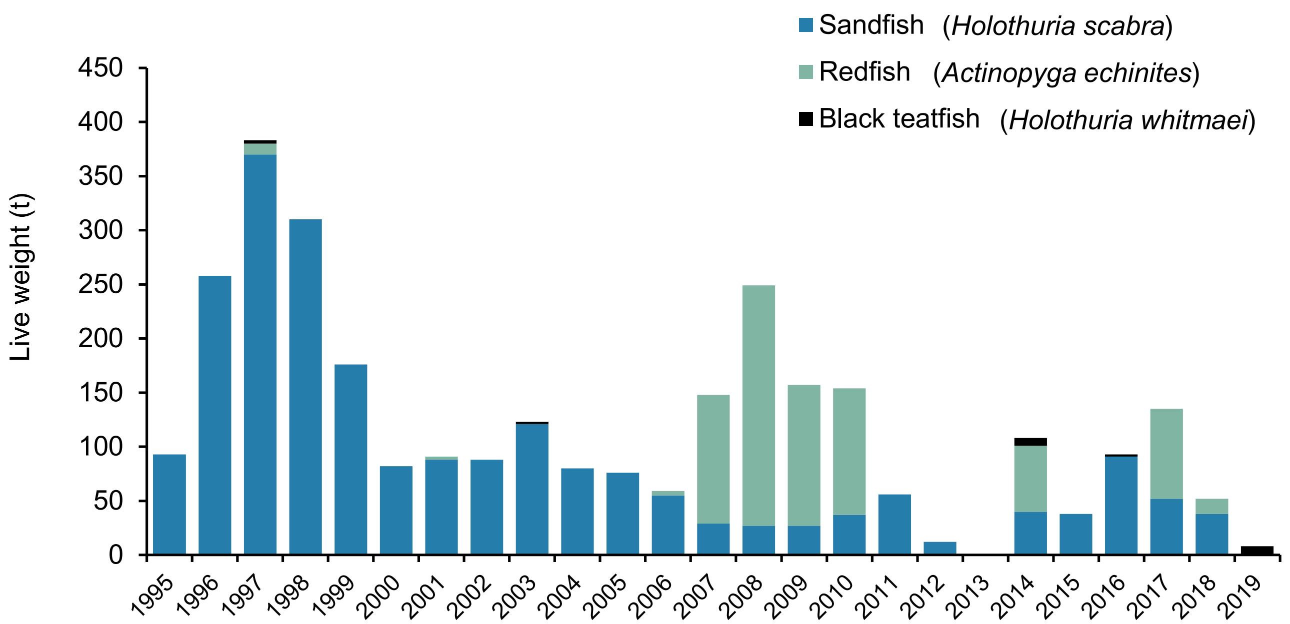

Figure 6. Annual total retained catches (tonnes) in the Western Australian sea cucumber fishery (WASCF) between 1995 and 2019. Adapted from Hart & Murphy (2021).

Western Australia. This reletively recent fishery for holothurians is a small commercial fishery as shown in Figure 6 where live weight captures since 1995 are presented (Gaughan *et al.*, 2019; Newman et al., 2021; Hart & Murphy, 2021). As elsewhere, the fishery followed the 'boom and bust' scenario. Essentially based on the sandfish *H. scabra* initially, it decreased after ten years and targeting the redfish *Actinopyga echinites* allowed a new peak in 2008. The teatfish *H. whitmaei* only represents occasional captures. The holothurians are mostly collected by diving in remote areas, accessed only by boat. The stock status is managed through input controls including limited entry, maximum number of divers, species-dependent minimum size limits, and gear restrictions. The Pilbara area of this fishery has achieved Marine Stewardship Council certification.

In 2019, two species were targeted, with a total catch of 6.9 t (Fig. 6). This catch comprised 2.1 t of sandfish (*H. scabra)* and 4.8 t of deepwater redfish (*A. echinites*). This was lower than the 2018 total of 61 t (sandfish 36 t; deepwater redfish 25 t). The industry has adopted a rotational fishing strategy for both sandfish and redfish with limited catch taken for either species in 2012 and 2013, and no fishing for sandfish in the Kimberley in 2018 and 2019.

Details are given on the different stocks (Hart *et al.*, 2022). For the Kimberley Sandfish (Sustainable – Adequate) based on the information and analyses available, the current risk level was estimated to be Medium, consistent with previous assessments of the fishery. For the Pilbara Sandfish (Sustainable – Adequate) based on the information and analyses available, the current risk level was estimated to be Low. For the Pilbara Redfish (Sustainable – Adequate) based on the information and analyses available, the current risk level was estimated to be Low. This is consistent with previous assessments of the fishery.

Given the hand only method of fishing, no bycatch is taken by the fishery and there are no known protected species interactions. This fishery harvests only a small amount of sandfish and redfish per annum. The effect from this harvesting on the rest of the ecosystem, given that the catch is spread over a wide region, would be insignificant.

Social and economic outcomes are also presented (Hart and Murphy, 2021): the social outcomes are at low risk level, as 4 to 6 crew are employed on a vessel, comprising a master, deckhand and divers. Additional individuals are employed for the processing of the product. These activities are mostly located in the Northern Territory and Victoria where the fishing fleet is based. The economic outcomes are also at low risk level, as the estimated annual value for 2019 was $29,325 based on a total live weight of 6.9 tonnes and $4.25 per kg. This is only a beach-price value and the processing sector adds significant value. Concerning

Table 6. Hong Kong imports 2017-2020. A- quantities in kg. B - values in HK$.

A-Quantities, Kg

Country	2017	RK	%	Country	2018	RK	%	Country	2019	RK	%	Country	2020	RK	%
Yemen	214 233	1	5.15%	Sri Lanka	196569	1	4.78 %	Sri Lanka	194495	1	6.58	Yemen	123472	1	7.38
Sri Lanka	176 170	2	4.24%	Yemen	179015	2	4.35%	Yemen	134507	2	4.55	Sri Lanka	118740	2	7.09
Madagascar	134 489	3	3.23%	Mozambique	88742	3	2.16%	Mozambique	57032	3	1.93	Madagascar	62690	3	3.75
Mozambique	119 675	4	2.88%	Madagascar	84509	4	2.05%	Seychelles	51969	4	1.76	Seychelles	44516	4	2.66
Maldives	76 114	5	1.83%	Maldives	69202	5	1.68%	Madagascar	47569	5	1.61	Tanzania	21873	5	1.31
Tanzania	54 865	6	1.32%	Seychelles	48991	6	1.19%	Maldives	42345	6	1.43	UAE	18621	6	1.11
Seychelles	51 820	7	1.25%	Kenya	39533	7	0.96%	Kenya	31533	7	1.07	Maldives	18175	7	1.09
Kenya	34 846	8	0.84%	Oman	25596	8	0.62%	Tanzania	27754	8	0.94	Kenya	13239	8	0.79
Oman	15 507	9	0.37%	Tanzania	25448	9	0.62%	Egypt	19786	9	0.67	Egypt	12140	9	0.73
Egypt	11 643	10	0.28%	Saudi Arabia	14110	10	0.34%	Saudi Arabia	10037	10	0.34	Saudi Arabia	4230	10	0.25
Saudi Arabia	8 879		0.21%	Mauritius	3880		0.09%	Oman	9756		0.33	Mozambique	2645		0.16
India	7 548		0.18%	Sudan	3739		0.09%	South Africa	6540		0.22	Mauritius	2551		0.13
UAE	7 245		0.17%	UAE	3281		0.08%	Somalia	6312		0.21	Oman	2215		0.13
Mauritius	4 786		0.12%	Somalia	2490		0.06%	Sudan	5935		0.20	South Africa	1348		0.08
Sudan	2 030		0.05%	South Africa	2615		0.06%	UAE	4128		0.14	Somalia	1250		0.07
Jordan	292		0.01%	Iran, islamic republic	945		0.02%	Mauritius	3675		0.12	Sudan	636		0.04
Somalia	405		0.01%	Egypt	853		0.02%	Ethiopia	1540		0.05	Qatar	171		0.01
South Africa	149		0.00%	Ethiopia	252		0.01%	Iran	663		0.02	Ethiopia	165		0.00
Ethiopia	110		0.00%	India	246		0.01%	India	616		0.02	Djibouti	0		0.00
Djibouti	70		0.00%	Pakistan	120		0.00%	Qatar	581		0.02	India	0		0.00
				Qatar	52		0.00%	Djibouti	42		0.00	Iran	0		0.00
total IO	**920 876**		**22.14%**	**total IO**	**790188**		**19%**	**total IO**	**656815**		**22%**	**total IO**	**448677**		**27%**
All countries	**4 158 818**			**All countries**	**4 116 795**			**All countries**	**2954775**			**All countries**	**1673908**		

B-Values, HK$

Country	2017	RK	Country	2018	RK	Country	2019	RK	Country	2020	RK
Yemen	65218	1	Sri Lanka	59572	1	Sri Lanka	68155	1	Yemen	60476	1
Sri Lanka	40442	2	Yemen	58104	2	Yemen	60491	2	Sri Lanka	53663	2
Madagascar	15562	5	Mozambique	3486	8	Mozambique	3851	8	Madagascar	9911	5
Mozambique	21582	3	Madagascar	31865	3	Seychelles	4665	7	Seychelles	15214	3
Maldives	14238	6	Maldives	16455	4	Madagascar	9727	4	Tanzania	12257	4
Tanzania	20855	4	Seychelles	3027		Maldives	16540	3	UAE	3113	8
Seychelles	4709	9	Kenya	3463	9	Kenya	2787		Maldives	6372	6
Kenya	2864		Oman	3323	10	Tanzania	3796	9	Kenya	2159	9
Oman	6273	7	Tanzania	7998	6	Egypt	7646	5	Egypt	3481	7
Egypt	3372		Saudi Arabia	12036	5	Saudi Arabia	6079	6	Saudi Arabia	1414	10
Saudi Arabia	5508	8	Mauritius	1289		Oman	997		Mozambique	442	
India	967		Sudan	671		South Africa	2061		Mauritius	219	
UAE	4466	10	UAE	4038	7	Somalia	2238		Oman	1131	
Mauritius	1131		Somalia	717		Sudan	3639	10	South Africa	349	
Sudan	92		South Africa	748		UAE	1735		Somalia	709	
Jordan	45		Iran, islamic republic	579		Mauritius	1878		Sudan	382	
Somalia	36		Egypt	215		Ethiopia	149		Qatar	86	
South Africa	391		Ethiopia	88		Iran	145		Ethiopia	53	
Ethiopia	75		India	70		India	156		Djibouti	0	
Djibouti	36		Pakistan	20		Qatar	436		India	0	
			Qatar	42		Djibouti	52		Iran	0	
total IO	207862		**total IO**	207806		**total IO**	197223		**total IO**	171431	

Table 7. Trade of sea cucumbers in tonnes and by product forms. From FAO data (FAO, 2021).

Commodity name	Flow	2015	2016	2017	%	Rank
1-SC live, fresh or chilled	**Import**	**5168**	**4782**	**6907**	30%	**2**
	Export	3858	3393	2519	9%	5
2-SC frozen	**Import**	**4934**	**4469**	**8402**	36%	**1**
	Export	4568	7031	9914	34%	2
3-SC dried, salted or in brine, smoked	**Import**	**5218**	**4968**	**6032**	26%	**3**
	Export	7732	8176	10214	35%	1
4-SC other than live, fresh or chilled	**Import**	**3031**	**3853**	**596**	3%	**5**
	Export	5769	4509	3378	12%	3
5-SC prepared or preserved	**Import**	**1346**	**1027**	**1361**	6%	**4**
	Export	3421	2706	2993	10%	4
TOTAL	**Import**	**19697**	**19099**	**23298**		
	Export	25348	25815	29018		

the Management Initiatives (Marine Stewardship Council MSC Assessment) the WA Sea Cucumber Fishery has been formally assessed against the sustainability standards. Sea cucumber stocks in the Pilbara Unit of Certification have passed the assessment, with the Kimberley region currently under review. A present conclusion is that the remoteness of the currently fished stock and the large tidal ranges where it occurs are natural barriers to uncontrolled expansion of fishing. Marine park planning has to date restricted this fishery from general use zones of some MPAs. Currently, lack of experienced fishers and suitable vessels is restricting catch to low levels.

Hong Kong statistics for the Indian Ocean region

Hong Kong is a very important hub for the trade and consumption of holothurian products (Conand, 2004, 2006b, 2008; To and Shea, 2012; Conand, 2017; To *et al.*, 2018).

The Census and Statistics from Hong Kong SAR for holothurians comprise several categories for the imports as well as the re-exports. In an early review based on 16 years of information, the main five supply countries of dried product were from the Western Pacific (To and Shea, 2012). A change in commodity codes occured in 2012 including an important frozen category and other categories such as live product, and prepared and preserved product (Conand *et al.*, 2014). A general figure showing the imports and re-exports of the dried and frozen product categories was published by To *et al.* (2018).

As the countries from the Indian ocean generally export dried product, this presentation (Table 6) is based on the last published Hong Kong import statistics, for the last 4 years from 2017 to 2020. Another change in categories had occured in 2016, therefore the data used here are from code 03081900 (sea cucumbers Holothuroidea, dried, salted or in brine, including those smoked; flours, meals and pellets of sea cucumbers fit for human consumption) for 2017 to 2020.

The Table 6-A shows the quantities (in kilograms) for each country of the Indian Ocean, imported into Hong Kong, the rank of the major exporters and the percent of the total world imports into Hong Kong. The total imports from the Indian Ocean have decreased during this period from 920 t in 2017 to 448 t in 2020. They represent approximately a fifth of the world total. The main exporters are Sri Lanka, Madagascar, Yemen and Mozambique. Table 6-B shows the values of these imports from the Indian Ocean countries in HK dollars, and the ranks of the major exporters with Yemen and Sri Lanka ranked as the main exporters.

A large part of these imports is then re-exported, as shown in Conand *et al.* (2014), Eriksson and Clarke (2015), and To *et al.* (2018). Locally, products in wholesale markets are typically displayed openly and in bulk, while retail markets offer a diverse array of dried and frozen products (To *et al.*, 2018).

FAO statistics for the Indian Ocean region

Statistics for holothurians are collected by FAO (2021) from the countries according to the five different forms of the product (Table 7).

Table 7 shows the general statistics at the world level for imports and the exports for the years 2015 to 2017.

Table 8. Holothurian captures by country in the Indian Ocean for the years 2015 to 2019. The sum for the Western (WIO) and Eastern (EIO) Indian Ocean areas are calculated and their percent of world total presented. From FAO data (FAO, 2021).

Country	Species	FAO major fishing area	FAO code	2015	2016	2017	2018	2019
Egypt		Indian Ocean, Western	51	0	0	0	0	0
Kenya		Indian Ocean, Western	51	171	6	13	15	15
Madagascar		Indian Ocean, Western	51	1598	535	922	853	1150
Maldives		Indian Ocean, Western	51	0	0	99	0	0
Mauritius		Indian Ocean, Western	51	0	0	0	0	0
Mozambique		Indian Ocean, Western	51	0	0	0	0	0
Seychelles	Black teatfish	Indian Ocean, Western	51	8	10	2	2	5
Seychelles	Prickly redfish	Indian Ocean, Western	51	108	79	72	72	75
Seychelles	Sandfish	Indian Ocean, Western	51	0	0	0	0	0
Seychelles		Indian Ocean, Western	51	467	270	194	194	195
Seychelles	White teatfish	Indian Ocean, Western	51	108	81	66	66	70
Tanzania, Unit. Rep.		Indian Ocean, Western	51	0	0	0	0	0
Yemen		Indian Ocean, Western	51	32	27	23	23	23
Sum WIO		Indian Ocean, Western	51	2492	1008	1391	1225	1533
% WIO				6.14%	2.20%	2.80%	2.42%	2.59%
Indonesia		Indian Ocean, Eastern	57	309	477	805	1013	1035
Malaysia		Indian Ocean, Eastern	57	0	0	0	0	0
Sri Lanka		Indian Ocean, Eastern	57	2560	1190	1090	1100	2350
Sum EIO		Indian Ocean, Eastern	57	2869	1667	1890	2113	3385
% EIO				7.07%	3.64%	3.50%	4.17%	5.71%
Sum IO				5361	3683	4677	3338	4918
% IO				13.2%	5.8%	6.4%	6.6%	8.3%
World Totals - Tonnes - live weight				40544	45773	49983	50661	59262

The category 'frozen' is now the largest in tersm of tonnage of the imports, followed by the 'live, fresh or chilled' category. This is an important change due to changes in transport facilities. The traditional category 'dried, salted or in brine, smoked' is now the third largest category with 26 % of the total. For the exports, the traditional category 'dried, salted or in brine, smoked' is still the largest, followed by 'frozen'. There are sometimes difficulties or inaccuracies in the way that these statistics are compiled, as the processed dry product is less than 10 % of the weight of the freshly fished product (Conand, 1989; Purcell *et al.*, 2009).

The FAO data are very important in showing the importance by country in terms of captures as presented in Table 8 for the years 2015 to 2019. The data for 2015 for Madagascar and Sri Lanka show very high levels compared to the following years, but again reached similar levels in 2019. The WIO countries, with more than1500 tonnes in 2019, represent 2.6 % of the world total, while the Eastern Indian Ocean countries represent 5.7%. On the whole for these recent years, the Indian Ocean countries have dropped from 13.2 to 8.3 % of the global total. Indonesia fishing zone 57 has shown an important increase during this period.

Discussion

Management of fisheries has developed during the 20th century with the Code of Conduct for Responsible Fisheries by FAO and partners developed in 1995 listing the main principles. The Ecosystem Approach (EAF) (FAO, 2003) is interdisciplinary and involves the overlap of the three objectives of stock productivity, biodiversity conservation and socio-economic issues. Sea cucumbers have been an issue for EAF management in international workshops organised by FAO (Lovatelli *et al.*, 2004) and CITES (Bruckner, 2006). More recently practical measures were presented at a global level by Friedmann *et al.* (2008), Purcell (2010), FAO (2010), and Purcell *et al.* (2013). Given the interest in the book on the commercial species by Purcell *et al.* (2012) produced by FAO, a second edition is now being prepared which will include more species. The conservation of the species had been examined by IUCN (Purcell *et al.*, 2014; Conand *et al.*, 2014) and projects are currently being developed through the IUCN

Sea Cucumber Specialist Group. CITES has recently put the teatfish species on Appendix II (Shedrawi *et al.*, 2019, Di Simone *et al.*, 2019, 2021) and Non Detrimental Findings (NDF) are now published (Setyastuti and Wirawati, 2019). Management measures were presented at the global regions level in Toral-Granda *et al.* (2008), including a chapter on the Indian Ocean (Conand, 2008). The regional approach for the Indian Ocean was later developed during a WIOMSA project (Conand and Muthiga, 2007; Muthiga and Conand, 2014) and a workshop by FAO (2013) and partners followed by publications (Eriksson *et al*, 2015b).

The **institutional regulatory measures** at a country level for holothurian fisheries are well presented in Purcell (2010); they use the answers to three questions which characterize the particular fishery to be managed: 1) the type of fishery (small-scale or industrial); 2) the status of the stock (underexploited, fully exploited, depleted); and 3) the management capacity of the country for management (strong, and will use recommended measures, or modest, and will use minimum measures and actions). The different types of fisheries for the Indian Ocean have been described in Purcell *et al.* (2013) and are updated here in the country reports.

The different measures, not detailed here, are the following: size-limits (to protect juvenile holothurians); gear limitation; fishing effort; catch quotas; market chain licensing and reporting; seasonal and short term closures; bans or moratoria; marine protected areas (MPA); rotational harvest closures; and territorial user rights. Each measure to be implemented needs to be supported by biological or socio-economical surveys, regulations and education.

The data reported in the literature is often at different levels depending on country and the different stakeholders consulted (Conand, 2001). The problem of reliability of the data has often been raised. A main issue concerns the measure of the catches of the different species; as fresh weight (or length), but also eviscerated, or salted, which differs largely from the processed weight (or length) and needs the knowledge of the specific values of conversion to be useful (Conand, 1990; Skewes *et al.*, 2004; Purcell *et al.*, 2009; Muthiga and Conand, 2014). The dried product weight was the main unit used in the trade until recently, but the frozen products are now increasing, as shown by Hong Kong imports, and this category will also need conversion values. Despite regulatory measures, illegal fisheries have developed in many countries; Conand (2018b) gives the information for the main global regions during the period 2014-2018. Bondaroff (2021) insists that illegal fisheries should be treated as a form of transnational organised crime, and intergovernmental and interagency cooperation should be increased to address this. The last issue of the review 'Traffic' by Ong and Chin (2022) analyses the online trade advertisments from Malaysia and Singapore. These are mostly (75 %) for the dried product, coming from 15 countries; most frequesntly from Indonesia and Australia. These authors emphasise that robust regulations and tracability systems are necessary.

As in other echinoderm species, holothurian populations show large 'boom - bust' variations coming from environmental and human influences with the latter being predominant for commercial species (Uthicke *et al.* 2009). Serial exploitations are described and analysed for many countries and will provide data for bio-economic models (Anderson *et al.*, 2011; Purcell *et al.*, 2013; Eriksson *et al.*, 2015b; Rawson and Hoagland, 2019; Wolfe and Byrne, 2022). Regional coordination for the management of holothurians is recommended for the future, as this seems to be the most appropriate scale based on the nature of the fisheries.

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9. Tables and Figures (maps, graphs and others): originals of all figures should follow the **Guidelines for illustrations** referred below. The lettering should be of a size readable after reduction for the final layout. Authors are requested to indicate the recommended position of figures and tables in the left-hand margin of the text. A separate sheet at the end of the document should be used for each table and figure, with the corresponding legend.

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a. Title Page: this should contain a concise title and the names of authors followed by affiliations and their complete postal addresses, phone numbers, and email addresses. The corresponding author and email address must be indicated.

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h. Citations: authors should be cited using their surnames, followed by the year of publication. Two authors should be separated by 'and'. If there are more than two authors, only the first author, followed by "*et al.*", should be given. This and other Latin or foreign terms should be italicized.

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- **Periodicals.** Here the following should be sequentially listed: author's name/s, initials, year of publication, full title of paper, periodical (in full), volume, first and last page numbers.

  Example: Richardson K, Beardall J, Raven J (1983) Adaptation of unicellular algae to irradiance: An analysis of strategies. The New Phytologist 93: 157-191

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- **Articles with a Digital Object Identifier (DOI).**

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l. Supplementary material: in case it is found relevant, authors may submit appendices with relevant information of major interest for the interpretation of the manuscript results. This is not applicable for the raw data of normal research. The editors will decide its eventual inclusion as appendices.

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Failure to follow any point of these guideline may delay or compromise the editorial process. Unless there are communication problems, all communications and exchange of documents shall be made strictly via the AJOL editorial interface. Authors must comply with timing for processing manuscripts, either requested revisions or proof check, or otherwise inform the editors of any delays.

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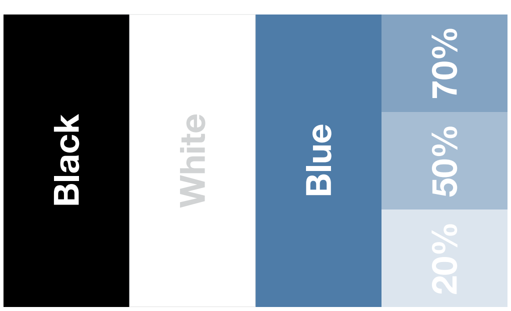
The original figures files should be sent separately from the text file to ensure the good quality of the figure reproduction.

Color palette

To maintain visual harmony in the illustrations, we recommend using mainly a monochromatic palette with the introduction of the Western Indian Ocean Journal's identity blue color.

When necessary to allow greater contrast between patterns and colors, the use of the secondary palette is considered.

main



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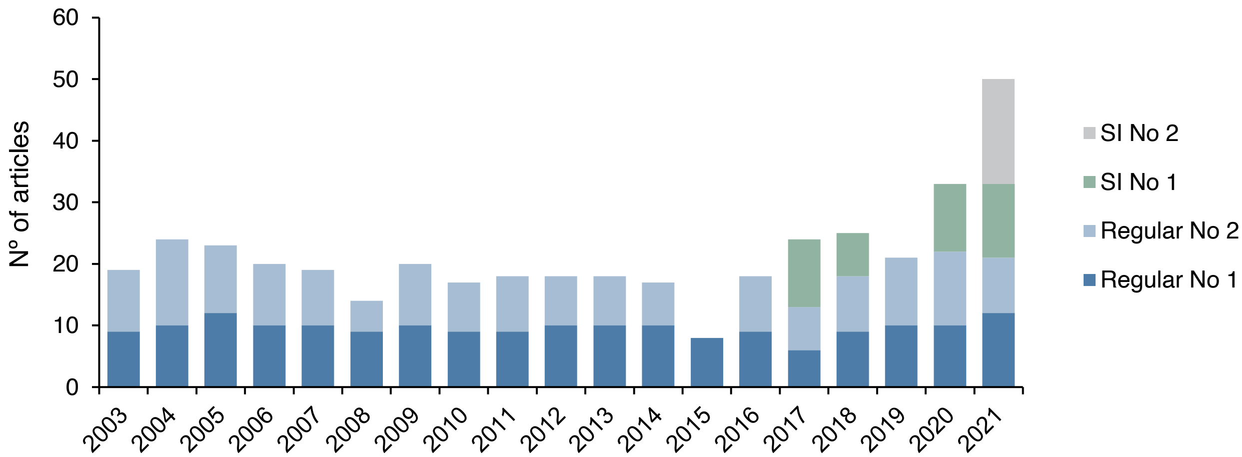

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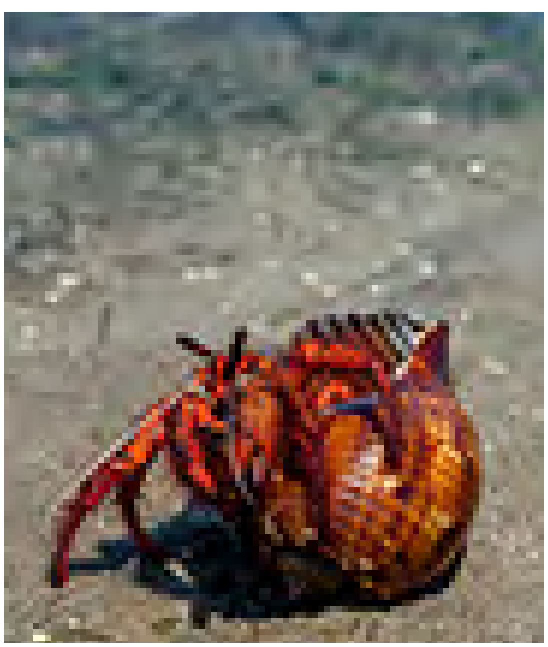

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Maps must have good resolution for efficient viewing and good reading

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Example of high resolution

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