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Ecological impact of salt farming in mangroves on the habitat and food sources of *Austruca occidentalis* and *Littoraria subvittata*

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ABSTRACT

The impact of salt farming on the habitats and food sources of *Austruca occidentalis* and *Littoraria subvittata* was studied in mangroves along the coast of Tanzania using stable isotopes (^{13}C and ^{15}N) and sediment particle size analysis. The ^{13}C and ^{15}N stable isotope composition in mangrove leaves, sediments and invertebrate tissues, were used to evaluate whether there are differences in feeding ecology of the crab *Austruca occidentalis* and the snail *Littoraria subvittata* collected from natural mangroves and mangroves around the salt ponds. Organic C, total N content and particle size distribution in sediments were used to assess if there are differences in habitat characteristics of mangroves around the salt ponds. Mangrove leaves and sediments were found to be ^{13}C enriched around salt ponds compared to those from natural mangroves. Likewise the macroinvertebrates collected from mangroves around salt ponds were found be enriched in ^{13}C relative to undisturbed mangroves. In addition, mangrove sediments around salt ponds were poorer in organic carbon and nitrogen and had more sand content compared to sediments from natural mangroves. These results indicate that salt pond activities have contributed to the modification of the habitats of macroinvertebrates by causing δ^{13}C stable isotopes enrichment and alteration of sediment characteristics in the ecosystem.

1. Introduction

High population pressures in tropical coastal zones resulting from increased human activities are prime cause of mangrove loss and degradation. Salt farming in mangroves is among the major factors that contribute to global loss of forested areas (Walters et al., 2008). For example, in Brazil it has been estimated that about 50,000 ha of forest have disappeared over the last 25 years and salt farming is one of the factors driving this loss (Ferreira and Lacerda, 2016). Moreover, in Kenya, about 10,000 ha of mangroves have been cleared between Ngomeni and Karawa due to salt farming (Abuodha and Kairo, 2001).

Salt production has been taking place for many years in Tanzania, in some places since 1920 (Mmochi et al., 2001; Liingilie et al., 2015). Several studies have highlighted threats on the ecosystem due to clearing of mangroves for salt ponds (Mwandya et al., 2009; Nehemia et al., 2017; Nehemia and Kochzius, 2017; Ocholla et al., 2013; Liingilie et al., 2015). Logging of trees from the mangrove forest can be detrimental to ecosystem functions because it can alter the microbial processes that are highly sensitive to the chemical quality and quantity of litter entering the soil. These processes include non-symbiotic N fixation, denitrification, net N mineralisation and nitrification (Pérez et al., 2009). Logging of mangrove trees during salt production can also cause changes in chemical, physical and biological parameters of the sediments (Ellegaard et al., 2014). A study on fish assemblages in Tanzanian mangrove creek systems revealed reduced abundances caused by the change in the hydrodynamics and sediment characteristics due to salt pond activities (Mwandya et al., 2009). The dykes (Fig. 1) constructed around the salt ponds become obstacles to the free movement of water, which are essential for dispersal of larvae of many invertebrates. Mangrove deforestation due to salt farming and other human activities has also been observed to cause unsteady tidal force, thereby increasing soil erosion and floods (Mazda et al., 2002; Ocholla et al., 2013). Physical disturbances in mangrove sediments have shown to have a significant effect on grain size distribution and sediment organic carbon content (Mazda et al., 2002; Balke et al., 2013).

Salt production in mangroves therefore has a major influence on the distribution and diet of macroinvertebrates. Faunal communities tend to utilise more mangrove carbon as their food source in systems with

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Fig. 1. Mangroves and salt ponds in Tanzania, Western Indian Ocean. a. Natural mangrove; b. a newly cleared mangrove to extend salt ponds; c. an old active salt pond with few scattered mangrove trees behind; d. a salt pond during the harvesting time; with dykes enclosing the ponds in b, c and d.

less input from adjacent waters – typically undisturbed systems (Bouillon et al., 2004). Understanding the impacts of salt farming activities on the habitats and feeding ecology of macroinvertebrates might thus be important for the effective management of mangroves ecosystem.

The crab, *Austruca occidentalis* is a dominant species in the mangroves along the coast of East Africa (Litulo, 2004). Gut content analysis, fatty acids composition and stable isotope analysis have revealed that this species feeds on various food sources, including microalgae, mangrove detritus, bacteria and meiofauna (France, 1998; Meziane et al., 2002; Bouillon et al., 2004). The distribution of *A. occidentalis* is known to be affected by many factors, including sediment grain size composition and organic and moisture content, as well as presence or absence of mangrove vegetation (Mokhtari et al., 2008, 2015).

The snail, *L. subvittata* is the dominant arboreal species in mangrove forest along the East African coast (Torres et al., 2008). Depending on the food availability, *Littoraria* snails species have been reported to feed on mixed diets that include plant material, phylloplane fungi and cyanobacteria (Lee et al., 2001). Gut content and stable isotope analysis also revealed that snails are opportunistic feeders, feeding on microalgae epiphytes from leaves and prop roots, suspended organic matter (OM) and mangrove detritus (Christensen et al., 2001; Alfaro, 2008). Their distribution pattern has been suggested to be controlled by tidal oscillations and can be affected by human-induced disturbances that influence environmental variables, such as temperature and oxygen concentration (Blanco and Cantera, 1999).

Studies carried out in tropical dry forest and highland forest in Madagascar highlighted the usefulness of naturally occurring stable isotope abundance in assessing impacts of deforestation to ecosystems (Vågen et al., 2006; Crowley et al., 2012). The study of stable isotopic composition can be used to identify mangrove nutrient status and has been used to trace feeding and movement patterns of different animal species (Rubenstein and Hobson, 2004; Wolters et al., 2016). However, we are not aware of any research conducted using natural stable isotope abundance to investigate the ecological consequences posed by salt pond activities on mangrove ecosystem. Using the stable isotopic composition of consumer tissues it is possible to trace the primary food sources (Miranda and Perissinotto, 2012; Bui and Lee, 2014). Stable isotope can also be used to investigate plant physiology and anthropogenic impacts. Stable isotopes for instance have been shown to be more useful for quantifying the edge effects resulting from forest fragmentation due to deforestation compared to conventionally measured variables, such as species richness, wind speed, light levels and air temperature (Swap et al., 2004; Crowley et al., 2012).

The general objective of this study is to determine the effects of mangrove deforestation by salt farming activities on the carbon and nitrogen food source availability and utilisation by macroinvertebrates. More specifically, we intend to determine whether the primary sources of carbon and nitrogen for *A. occidentalis* and *L. subvittata* is influenced by salt pond activities. We also want to assess if the OM content in sediments, an important potential food source for local invertebrates, is influenced by salt farming activities and correlates with changes in sediment particle size distribution. To achieve this, we determined the $\delta^{13}C$ and $\delta^{15}N$ in mangrove leaves, and tissues of *A. occidentalis* and *L. subvittata*. The sediment particle size, organic C and total N content as well as $\delta^{13}C$ and $\delta^{15}N$ composition were also determined. We then compared values obtained from mangroves around the salt ponds and from natural mangroves.

2. Material and methods

2.1. Description of study area

Tanzania is situated south of the equator at the Indian Ocean (Fig. 2), with a coastline of about 800 km (Masalu, 2000). The mangrove forest areas in this coast have declined by 3828 ha between 1990 and 2000 (Wang et al., 2005). Salt farming activities along this coast have been among the main contributing factors to this loss through

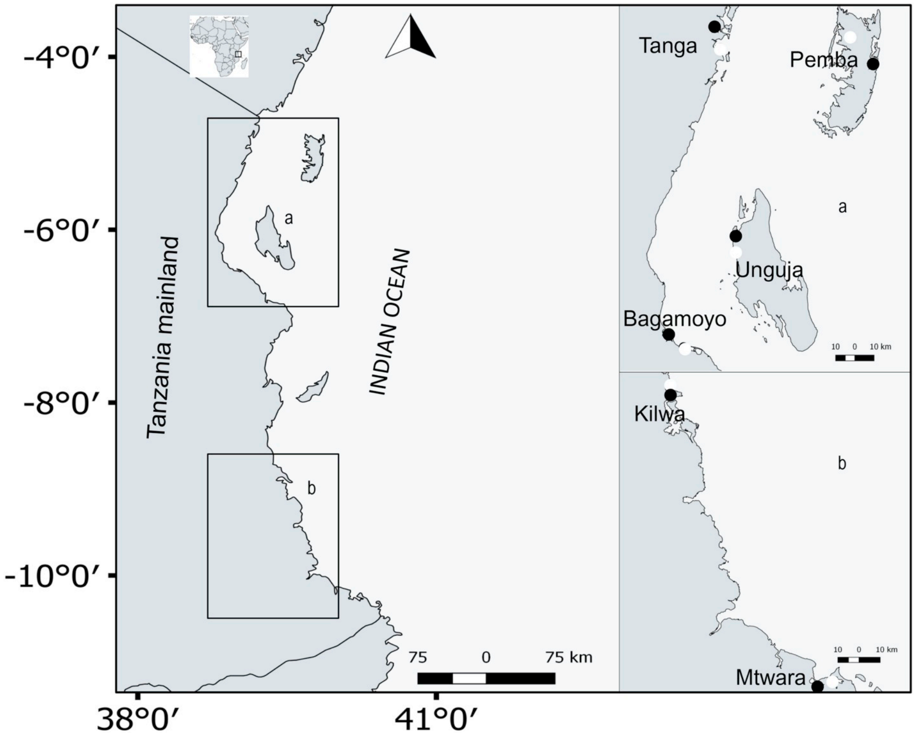

Fig. 2. Map showing the sample sites. White circles represent natural mangroves and black represent mangroves sites around the salt ponds.

clear cutting and selective logging of the trees for salt pond dyke and water channel construction. Mangrove deforestation in Bagamoyo for instance is reported to be so high that restoration projects are unlikely to have positive outcomes (Muzuka and Shunula, 2006). The climate of the coastal region is hot and humid throughout the year, with two seasonal cycles, namely the Southeast (SE) and Northeast (NE) monsoon. The wet NE monsoon occurs from October to March, while the dry SE monsoon occurs from April to October (Semesi, 1992). The total annual rainfall in this area ranges between 500 mm to 750 mm (Semesi, 1992). Salt production is mainly done in the dry season, with harvesting between August and October.

Samples were collected in six mangrove areas along the Tanzanian coast: Tanga, Bagamoyo, Pemba and Unguja, Kilwa and Mtwara. Four sites were located on the mainland and two on islands (Pemba and Unguja) and sampling took place at natural mangroves and salt pond sites (Fig. 2). The salt ponds are mainly located at the landside in mangroves dominated by stunted and isolated *Avicennia marina* trees. The natural mangrove sites are characterised by dense forest of *A. marina*. Other mangrove species, such as *Ceriops tagal,* were rarely seen at these sites. Various macroinvertebrates were observed, including the crab species *A. occidentalis* and *Cranuca inversa* and the snail species *L. subvittata* and *L. pallescens*.

2.2. Sampling and preparation of samples

Samples were collected during the dry season at low tide in July and August 2014 and 2015. For each study site of mangroves at salt ponds sampled, a natural mangrove site was also sampled for comparison. The mean distance between the salt pond site and its natural site was 10 ± 4.7 km. For each site in each area, three plots of 20 by 20 m were selected for the collection of samples. The collected samples included: (1) green mangrove leaves of *A. marina* and leaf scraps for assessing the impact of the salt pond activities on the food source and habitats of arboreal snails (*L. subvittata*), (2) brown mangrove leaves of *A. marina* for estimating the source of sedimentary organic C in natural mangroves and mangroves around the salt ponds, (3) surface sediments for evaluating the potential impact of salt pond activities on the food sources and habitats of benthic invertebrates (*A. occidentalis*), and (4) *A. occidentalis* and *L. subvittata* as bioindicators for environmental changes due to salt farming activities.

In each plot 10 random samples of surface sediments (0–5 cm depth) were collected by using a plastic spoon, cleared from shell remains and large debris, and mixed to form a homogeneous mixture. Two subsamples from the homogeneous mixture were taken from each plot, transported in a cool box and stored in a freezer at −20 °C until analysis. Ten young green mangrove leaves of *A. marina* were randomly collected by hand at the same height (2 m) in each plot. Additionally, 10 young mangrove leaves were also collected in each plot to obtain epiphytes on the leave surface. The scrapings were obtained by gently scraping on the leaf surface using a surgical blade without damaging the epidermis. In addition, 30 brown mangrove leaves of *A. marina* were randomly collected on the trees in the plots at each site. For each type of mangrove leaves and leaf scraps, samples were pooled together per plot/site, transported in a cool box and stored in a freezer at −20 °C.

Five individuals of *L. subvittata and A. occidentalis* with approximately the same size were randomly collected in the plots at each site. The shells and intestines of *L. subvittata* were removed and discarded. For *A. occidentalis* muscle tissues from the pincer were taken. The tissues from *L. subvittata* and *A. occidentalis* were rinsed with deionised water and frozen at −20 °C until further analysis.

All samples of sediments, macroinvertebrate tissues and mangrove leaves collected for organic C, total N content and stable isotope analysis were dried at 60 °C for 24 to 72 h and grounded to fine powder using a mortar and pestle. Mangrove leaves and tissues of macroinvertebrates were grounded by pre-treating with liquid nitrogen to make them brittle and obtaining homogeneous samples.

2.3. Sediments grain size

The frozen wet sediment samples were first lyophilised and then homogenised. Triplicates were individually analysed using a Malvern Mastersizer 2000. The coarse fraction (> 63 μm) was dry sieved using an American Standard Test Sieve Series (ASTM). Finally, particle size

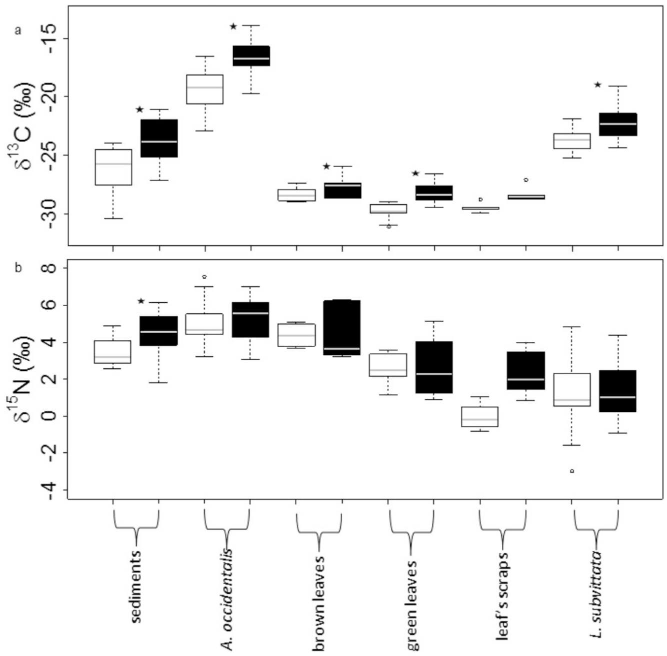

Fig. 3. Box plots with medians, 25th and 75th percentiles of (a) δ^{13}C and δ^{15}N ratio of sediments mangrove leaves and macroinvertebrates tissues from natural mangroves (white box plots) and salt pond sites (black box plots). Star at the box plots indicates sample from salt ponds that are significantly different from natural mangrove sites ($P < .05$).

distribution and composition were classified according to the Wentworth scale (Wentworth, 1922).

2.4. Elemental and stable isotope analysis

Samples were analysed for organic C and N content as well as δ^{13}C and δ^{15}N. For sediment organic C, samples were first acidified to remove carbonates by adding diluted (5%) HCl drop-by-drop, until no more release of CO_2 was observed. No acid treatment was performed on samples for nitrogen analysis, because it has been reported that acid affects δ^{15}N values (Vafeiadou et al., 2013). Consequently, tissue samples from crabs and snails were not acidified. Sediment and tissue samples alike were analysed using an Elemental Analyser (Thermo Flash1112) connected on-line via a Conflo III interface to an Isotope Ratio Mass Spectrometer (Thermo Delta + XL). Different standards were used for calibration, including Acetanilide and the certified reference materials IAEA-N1 Ammonium Sulphate (δ^{15}N: 0.43‰) and IAEA-CH6 Sucrose (δ^{13}C: −10.449‰). Reproducibility for both δ^{13}C and δ^{15}N measurements was ± 0.15‰. The isotope ratios were reported in the delta (δ) notation in per mil (‰) as follows:

$$\delta R(\%_0) = \{(X_{sample} - X_{s\tan dard})/X_{s\tan dard}\} \times 1000$$

where by $R = {}^{13}C$ or ${}^{15}N$, and $X = {}^{13}C/{}^{12}C$ or ${}^{15}N/{}^{14}N$

The carbon reference was Vienna Peedee Belemnite (VPDB) and the nitrogen reference was atmospheric nitrogen gas.

2.5. Mixing equations

The composition of the sedimentary OM reservoir was computed for both natural mangroves and mangroves around the salt ponds using a simple two end-member mixing model (Eq. (1)) considering the hypothesis that there were only two dominant sources of OM: mangrove litter and marine phytoplankton. For marine phytoplankton we used data from the literature for ocean phytoplankton at the latitude of Tanzania (Fontugne and Duplessy, 1981). This kind of literature data have also been used for assessing and quantifying the relative importance of different OM types in mangroves sediments elsewhere (Bouillon et al. 2003, 2004).

$$X_{\_mang} = \delta^{13}C_{\_sed} - \delta^{13}C_{\_phyt} \Big/ \delta^{13}C_{\_mang} - \delta^{13}C_{\_phyt} \quad (1)$$

Eq. (1) was derived from the following equations;

i. $\delta^{13}C_{\_sed} = (C_{\_phyt} \times \delta^{13}C_{\_phyt} + C_{\_mang} \times \delta^{13}C_{\_mang}) / C_{\_sed}$ (Middelburg et al., 1996)

ii. $X_{\_mang} = {}^{C_{\_mang}}\!/_{C_{\_sed}}$ (Bouillon et al., 2003)

$$X_{\_phyt} = 1 - X_{\_mang} = {}^{C_{\_phyt}}\!\Big/_{C_{\_sed}}$$

where δ^{13}C_sed, δ^{13}C_phyt and δ^{13}C_mang are the carbon stable isotope composition of OM in sediments, marine phytobenthos and the terrestrial mangrove component, respectively. C_sed, C_phyt and C_mang the organic C content of sediment, marine phytobenthos, and the mangrove component, respectively. X_mang and X_phyt are the fractions of organic C in sediments from mangrove litter and marine origin, respectively. For δ^{13}C_mang we used our measurements for brown leaves of *A. marina*, while for δ^{13}C_phyt we used the average value of −20.4 ± 0.2‰ as described for Indian Ocean phytoplankton (Fontugne and Duplessy, 1981).

2.6. Data analysis

Data analysis involved testing for homogeneity of variances, normality and differences among samples. Shapiro-Wilk test and Levene's test as implemented in the software R (version 3.1.2) were used to check for normality and homogeneity of variances, respectively. The δ^{13}C signatures were log (−x) transformed and percentage data were arcsine [square root (X/100)] transformed to improve variance homogeneity. To determine differences in δ^{13}C and δ^{15}N between samples collected from mangroves around the salt ponds and natural mangroves, a paired Wilcoxon signed-rank test statistic (V) was performed as implemented in R (version 3.1.2) since the assumptions for parametric statistics were not met. Bivariate least square regression (BLS) was performed to estimate the amount of inorganic nitrogen in natural mangroves and mangroves around the salt ponds. A Spearman rank correlation test was utilised to assess the relationship between median sediment particles (D50) and δ^{13}C, in both natural mangroves and mangroves around the salt ponds.

3. Results

3.1. Stable isotopes in mangrove leaves

Green mangrove leaves from salt ponds were enriched in ^{13}C by 1.1‰ compared to those from natural mangrove sites (V = 120, $P < .05$) (Fig. 3a). However, the δ^{13}C isotope composition of leaf's scraps from mangroves around the salt ponds (−28.3 ± 0.7‰) and natural mangrove sites (−29.5 ± 0.4‰) was not significantly different (Fig. 3a). Brown mangrove leaves at salt ponds had a higher δ^{13}C isotope composition compared to natural mangroves (V = 0, $P < .05$) (Fig. 3a). For nitrogen, no differences were detected between samples from natural mangroves and mangroves around the salt ponds (Fig. 3b).

3.2. Stable isotopes in macroinvertebrates and food sources

More depleted values and a wider range of δ^{13}C were recorded for *A. occidentalis* in natural mangroves (−22.9 to −16.5‰) compared to salt pond sites (−19.8 to −15.0‰) (Fig. 3a). The range and values of δ^{13}C for *L. subvittata* was higher in salt ponds (−23.9 to −19.1‰) compared to natural mangroves (−25.2 to −21.4‰) (Fig. 3a). *Austruca occidentalis* from natural mangroves and mangroves around the salt ponds was enriched in ^{13}C and characterised by higher δ^{15}N values

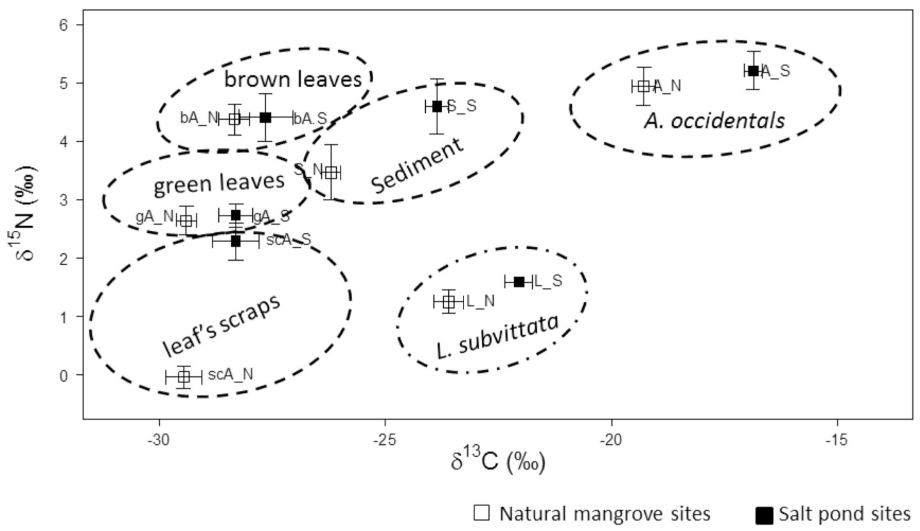

Fig. 4. δ^{13} C and δ^{15}N distribution in sediments, macroinvertebrates and mangrove leaves, with _N indicating natural settings and _S indicating salt pond settings. A_N and A_S; *Austruca occidentalis*, L_N and L_S; *Littoraria subvittata*, S_N and S_S; Sediments, gA_N and gA_S; green leaves of *Avicennia marina*, bA_N and bA_S; brown leaves of *Avicennia marina* and scA_N and scA_S; scraps from green leaves of *A. marina*.

compared to *L. subvittata* (Fig. 3a, b). The ^{13}C isotope composition of *A. occidentalis* and *L. subvittata* in natural mangroves and mangroves sites around the salt ponds were enriched compared to sediments, mangrove leaves and leaf's scraps (Fig. 4). *Austruca occidentalis* from natural mangroves and mangroves around the salt ponds was characterised by relative higher δ^{15}N values compared to sediments, mangrove leaves and leaf's scraps (Fig. 3b). In contrast, *L. subvittata* was characterised by lower δ^{15}N values compared to mangrove leave samples, with exception of scraps from green leaves from natural mangroves.

The difference in δ^{13}C between *A. occidentalis* and sediments at natural mangroves was 5.5‰ and around the salt pond sites 7.2‰. The difference in δ^{15}N value between *A. occidentalis* and sediments, in natural mangroves, was 1.5‰ and around the salt pond sites, 0.6‰ (Fig. 4). The difference in δ^{13}C values between *L. subvittata* and green mangrove leaves/leaf's scraps from leaves at natural mangroves was 5.8‰, while around the salt pond sites it was 6.3‰ (Fig. 4). A lower difference value of δ^{15}N was recorded between *L. subvittata* and green mangrove leaves/leaf's scraps (Fig. 4).

3.3. Stable isotopes, particle size distribution and organic matter content in sediments

The OM in sediments from mangrove sites around the salt ponds is enriched by 2.3‰ in ^{13}C compared to natural mangroves ($V = 0$, $P < .05$). The sediments are also enriched ($V = 13$, $P < .05$) in ^{15}N compared to natural mangroves (Fig. 3 b). The sediments sampled from natural mangroves showed a smaller particle size ($D_{50} = 265.4 \pm 47.9\,\mu m$) than those from salt ponds ($D_{50} = 343.5 \pm 47.1\,\mu m$) (Table 1). A significant positive correlation between median sediment particle size (D50) and δ^{13}C isotope ratio was observed at natural mangroves sites, but not for the salt pond sites (Fig. 5). Organic C and total N content in sediments were higher in natural mangroves than around the salt ponds ($V = 171$, $P < .05$) (Fig. 6a, b) and these variables were correlated with each other (Fig. 7a, b). The mean contribution of the marine component in OM in sediments around the salt ponds varied from 20 to 85% and was higher than in natural mangroves ($P < .05$) (Fig. 6C).

4. Discussion

4.1. Stable isotopes in mangrove leaves

The values of stable isotopic composition of organic C for green mangrove leaves of *A. marina* in natural mangroves ranged from −31.2 to −27.4‰ and in mangroves around the salt ponds −29.1 to −26.5‰, which is similar to δ^{13}C values reported in other studies (Rao et al., 1994; Al-Zaidan et al., 2006; Penha-Lopes et al., 2009; Briand et al., 2015). Ranges of δ^{13}C values for leaf's scraps (−30.0 to −27.1‰) were similar to δ^{13}C values reported from Fiji-Island (Alfaro, 2008). These ranges are typical for plants with a C3 photosynthetic pathway (Machiwa, 2000; Muzuka and Shunula, 2006). A study in mangrove forest (*Avicennia germinans*) from the Southern Caribbean shows how mangrove deforestation contributes to reduced canopy cover, changes in soil texture and increased temperature (Amortegui-Torres et al. 2013). The environmental stresses, such as high temperature caused by canopy openness and increased solar radiation, are known to influence the carboxylation process, which can lead to enrichment of ^{13}C in mangrove and terrestrial plant leaves (Crowley et al., 2012; McKee et al., 2002; Wei et al., 2008). The same reasons can be given for our higher δ^{13}C values observed in mangroves around the salt ponds, because they were characterised by a short and less abundant trees with poor canopy cover due to clear-cutting and selective logging. The other possible explanation for the differences observed in δ^{13}C values may be higher salinity in mangroves around the salt ponds. The existence of salt ponds may cause underground salt-water intrusion to the mangrove and cause hypersalinity in the soil that is beyond the tolerance level by these mangroves. High salinity in the mangroves has been reported to affect photosynthesis by decreasing the intercellular

Table 1
Mean ± SD for the characteristics of sediments from mangroves in Tanzania, Western Indian Ocean.

Variable	Mangroves around the salt pond sites	Mangroves at natural sites	*p*
particle size – D50 (μm)	343.5 ± 47.1	265.4 ± 47.9	< 0.05
Total N content (mmol/g)	0.02 ± 0.01	0.05 ± 0.02	< 0.05
organic C content (mmol/g)	0.51 ± 0.22	1.20 ± 0.52	< 0.05
C/N	19.55 ± 3.96	22.38 ± 2.69	< 0.05
δ^{13}C-organic C (‰)	−23.86 ± 2.00	−26.20 ± 2.00	< 0.05
δ^{15}N-total N (‰)	4.59 ± 1.06	3.47 ± 0.83	< 0.05

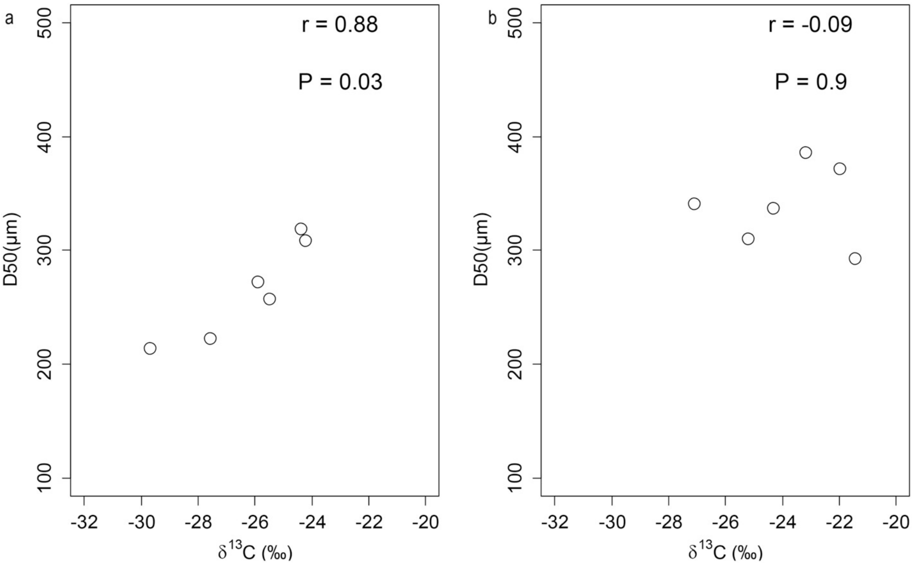

Fig. 5. Correlation (r) between sediment median particle size (D50) and carbon isotopic composition of organic matter in (a) natural mangroves and (b) mangroves at salt ponds.

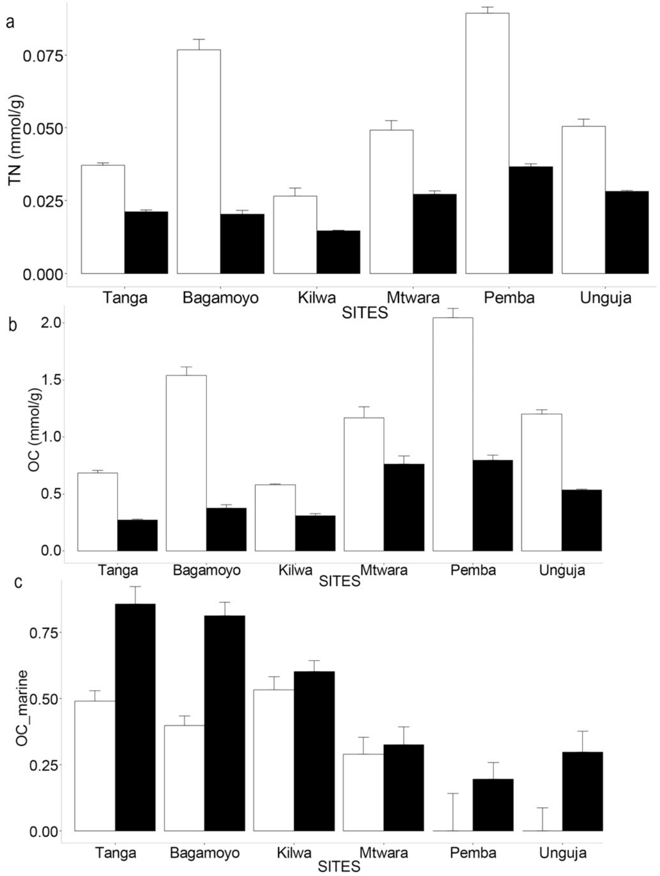

Fig. 6. (a) Mean ± SD of total nitrogen content in sediments; (b) organic carbon in sediments; (c) fraction of organic carbon in sediments that is of marine origin (OC_marine). White bars: natural mangroves; black bars: mangroves at salt ponds.

CO_2 concentration and leaf carbon isotope discrimination (Kao et al., 2001; Lin and Sternberg, 1992). The decrease in intercellular CO_2 may be linked to a lower number of stomata openings and results in an increase in the ^{13}C stable isotope composition of leaves (Kao et al., 2002). Also, salinity and $\delta^{13}C$ values of mangrove leaves were found to be positively correlated in an estuarine wetland of Southern China (Wei et al., 2008). In mangroves around the salt ponds the low evapotranspiration as a response to increased salinity inducing water stress may have stimulated lower number of opened stomata and results to enrichment in heavy carbon stable isotopes (Farquhar et al., 1989).

The mean $\delta^{15}N$ of green mangrove leaves of *A. marina* (2.7‰) was similar to the $\delta^{15}N$ values reported for the same coastal system in Tanzania (3.2‰; Muzuka and Shunula, 2006). Similar values of $\delta^{15}N$ have been also reported from Gazi Bay, Kenya (3‰; Bouillon et al., 2004). The mean $\delta^{15}N$ values of mangrove leaves from sites around the salt ponds and natural sites were not significantly different. However, leaf's scraps had higher $\delta^{15}N$ isotope values around the salt ponds (2.3 ± 1.2‰) than in natural mangroves (−0.04 ± 0.8‰). Lack of differences in most mangrove samples between natural and salt ponds sites might be explained by similar factors that influence ^{15}N composition in leaves indicating that the $\delta^{15}N$ isotope ratio in mangrove leaves is not a good indicator for impacts of mangrove deforestation. Lack of consistence in $\delta^{15}N$ compared to $\delta^{13}C$ signatures was also reported in samples collected in India, Sri Lanka and Kenya (Bouillon et al., 2004).

4.2. Stable isotopes in macroinvertebrates and food sources

The values of ^{13}C isotopic composition in tissues of *A. occidentalis* obtained in this study are similar to those reported in other studies (Bouillon et al., 2004; Kon et al., 2007). A consumer-diet discrimination factor ($\Delta = \delta^{13}C$ consumer − $\delta^{13}C$ diet) of about 1‰ is generally considered when estimating the diet of a consumer (Deniro and Epstein, 1981; Christensen et al., 2001). For $\delta^{15}N$ Δ is larger, in the range of 3–4‰ and the variations depend on food quality and environment factors (Caut et al., 2009).

In this study, the difference in $\delta^{13}C$ between *A. occidentalis* and OM in sediments (it's presumed main food source) was 5.5 and 7.2‰ respectively in natural and salt pond mangrove sites (Fig. 4). The differences which far exceed 1‰ may be indicating that OM in sediments is not the main food source of *A. occidentalis,* and that other enriched food sources should contribute to their diet as well. While the diet of *A. occidentalis* has indeed been observed to include mangrove detritus food materials from the sediment's OM (Meziane et al., 2002), it has also been reported that this species might be feeding selectively on food fractions enriched in ^{13}C, such as marine phytobenthos, benthic microalgae and bacteria from the sediment surface (Bouillon et al., 2004;

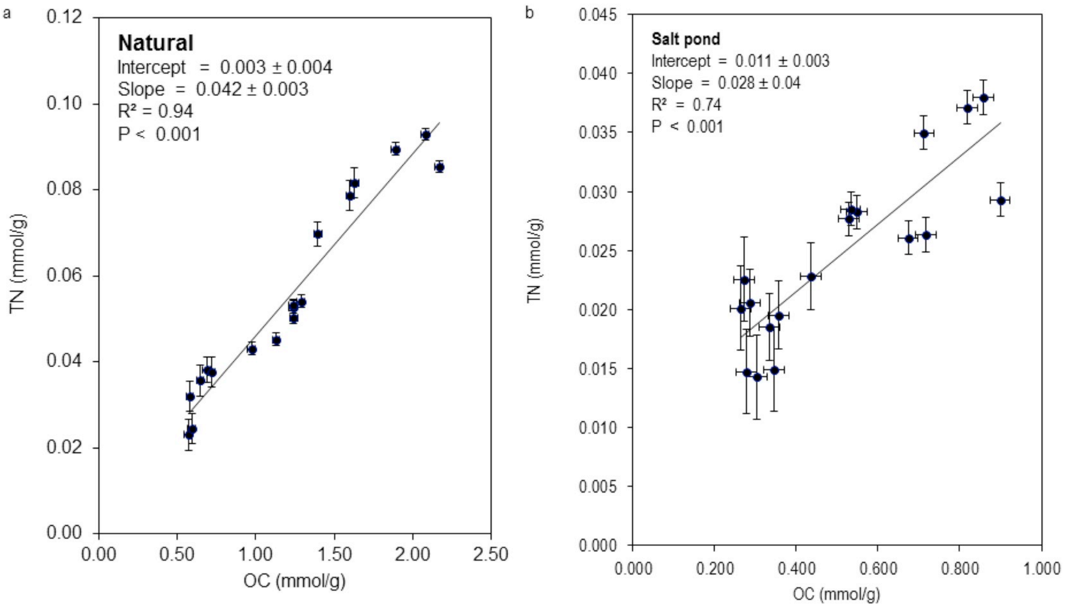

Fig. 7. Bivariate Least Square regression, indicating a relationship between organic carbon (OC) and total nitrogen (TN) in sediments. (a) natural mangroves and (b) mangroves at salt ponds in Tanzania, Western Indian Ocean.

Penha-Lopes et al., 2009; Kruitwagen et al., 2010). Therefore, the food source utilised by *A. occidentalis* is likely to be a mixture of sediment bulk organic matter and such other ^{13}C enriched fractions. The higher difference observed between salt pond sites and natural may then be explained by the fact that the ^{13}C of the mixed food source is likely more diluted by mangrove leaf litter in natural mangroves compared to mangroves close to salt ponds. In addition, the higher difference of enrichment in ^{13}C at salt pond sites compared to natural might also be induced by extreme conditions such as temperature and high light intensity which cause more enrichment in food sources in mangroves at salt ponds. Both experimental and field studies in fresh and marine environments have generally reported higher δ^{13}C values of food sources subjected to high light intensity and temperature (Durako and Hall, 1992; France, 1995; Hill et al., 2008).

Similar results were found for *L. subvittata* where the difference in δ^{13}C between the organisms and its presumed food source (mangrove leaves) was 5.8‰ and 6.3‰ at natural and salt pond mangroves, respectively, likely indicating that *L. subvittata* also feeds on a diet that is more ^{13}C enriched than mangrove leaves only. It has been suggested that *Littoraria* species feed on hairs of mangrove leaves and epiphytes found on the leaves, tree trunks and roots (Blanco and Cantera, 1999; Christensen et al., 2001; Bouillon et al., 2002). The values obtained in this study for δ^{13}C may be suggesting *L. subvittata* feed on microepiphytes, which have δ^{13}C values around −24‰ (Bouillon et al., 2004; Penha-Lopes et al., 2009). It is also possible that *L. subvittata* are exposed to marine food sources such as phytobenthos, microalgae and bacteria that stuck on the stems and roots of mangroves during high spring tides, when salt ponds are filled with water. Therefore, marine food sources might have also contributed to the observed higher δ^{13}C isotopic composition in *L. subvittata*. Although are opportunistic, snails prefer assimilation of microalgae and bacteria in their habitats (Alfaro, 2008).

Relative lower values of δ^{15}N than the ones obtained in this study in mangroves around the salt ponds (1.6 ± 1.6‰) and natural mangroves (1.2 ± 1.8‰) for *L. subvittata* were reported for other *Littoraria* species collected at the same coast (Lugendo et al., 2007). Low values of the δ^{15}N in snails have been suggested to be due to grazing on the leave epiphytes that are nitrogen fixing (Bouillon et al., 2002). Regarding difference in δ^{15}N (*L. subvittata* - mangrove leaves and *L. subvittata* –leaf's scraps), the obtained values are not concordant with the assumed discrimination factor of about 3‰ (Fig. 4). The small difference in δ^{15}N of lower than 3‰ recorded in this study might be supporting that mangrove material is not the potential food source of *L. subvittata*. This species might prefer food sources that have lower δ^{15}N values, such as microepiphytes available on leaves, tree trunks or roots (Bouillon et al., 2000; Christensen et al., 2001)

4.3. Stable isotopes, particle size distribution and organic matter content in sediments

The obtained ranges of δ^{13}C-values in sediments from natural mangroves (−23.9 to −30.3‰) and from mangroves around the salt ponds (−21.1 to −27.1‰) are close to those reported for other mangrove sediments (Bouillon et al., 2003; Muzuka and Shunula, 2006). On average, the δ^{15}N values (4.0 ± 0.9‰) obtained from sediments in this study are lower compared to δ^{15}N values (~ 10‰) reported for other mangroves sites along the Tanzanian coast (Muzuka and Shunula, 2006). Salt pond areas were dominated by larger sediment particle sizes, and lower organic C and total N contents as compared to natural areas. A significant positive correlation between median sediment particle size (D50) and δ^{13}C value was observed for natural mangroves sites, but not for sites around the salt ponds (Fig. 5). Similar results have been reported in other studies (Bosire et al., 2003; Aranibar et al., 2008; Alfaro, 2010; Tue et al., 2012; Wolters et al., 2016). Mangrove trees and their pneumatophore roots are known to promote particle sedimentation because of reduced tidal currents (Kathresan, 2003; Krauss et al., 2014). Due to increased tidal current movements in the deforested area, fine particles may be washed away by erosion preferentially, leaving coarser sandy material behind (Mazda et al., 2002; Balke et al., 2013; Ocholla et al., 2013). Lack of correlation between δ^{13}C values and median sediment particle size associated with a lower amount of OM in mangroves around the salt ponds may be an indicator for the influence of human disturbances. Lack of correlation among some sediment characteristics such as δ^{13}C and C/N ratios, has been suggested to be due to the influences of human activities (Gao et al., 2012).

The observed coarser sediments with lower organic C and total N content in mangroves close to salt ponds can be explained by this fine soil particle erosion. Sediments with high abundance of fine particles are indeed associated with higher OM content (Burone et al., 2003; Ramaswamy et al., 2008). Higher amounts of OM content in sediments in natural mangroves compared to deforested areas were also recorded at Kisakasaka in Zanzibar and Gazi Bay, Kenya (Bosire et al., 2003; Sjöling et al., 2005).

It has been observed that sediments with a higher organic C content are associated with lower values of $\delta^{13}C$ (Bouillon et al., 2003). We also measured lower $\delta^{13}C$ values and a higher amount of organic C in sediments from natural mangroves compared to mangroves around the salt ponds (Table. 1), suggesting a smaller contribution of mangrove material to the organic C in sediments in mangroves around the salt ponds. The mean contribution of the marine component in OM in sediments around the salt ponds varied from 20 to 85% and was higher than in natural mangroves ($P < .05$) (see Fig. 6C). This indicates that mangroves around the salt ponds might have relatively more marine input than natural mangroves.

Mangrove systems exhibit a low rate of nitrogen fixation and their high productivity is mainly driven by internal nutrient cycling that is coupled with organic matter mineralisation (Lee, 2006). Mineralisation and assimilation by trees contribute to high nitrogen dynamics in natural mangroves (Alongi, 2002). The importance of the inorganic N pool in soils can be estimated from the intercept of the regression line of TN vs organic C: a positive intercept reflects the presence of additional inorganic N in the sediments (Bergamaschi et al., 1997; Tue et al., 2012). In our study, the sediments from natural mangroves had the intercept at 0.003 ± 0.004 (mmol/g), which is not significantly different from zero, while the sediments from salt pond sites had the intercept at 0.011 ± 0.003 (mmol/g), which was significantly different from zero (Fig. 7 a, b), suggesting the presence of an important fraction of inorganic N in the sediments around the salt pond sites.

Influence of exogenous input from the ocean into sediments has been also documented to increase N availability in sediments (Bouillon, 2003; Molnar et al., 2014). However, it has been shown that higher mangrove tree biomass and number of roots per surface unit promotes OM availability and increases the uptake rates of nutrients by mangroves/microbes, thereby decreasing the concentration of inorganic nutrients, such as nitrates in the sediments and an isotope composition of enriched N (Middelburg et al., 1996; Fernandes et al., 2016). It is likely that the presence of a small number of trees that are isolated from each other might be contributing to the lower uptakes rates of nutrients in mangroves around the salt ponds, thereby increasing the concentration of inorganic nutrients such as nitrates and ammonium in the sediments.

5. Conclusion

The findings show that *A. occidentalis* and *L. subvittata* in mangrove around the salt ponds may be switching to the more available carbon source as their food, once mangroves are removed. This study revealed also that mangrove removal around the salt ponds may be contributing to large particle size distribution in sediments which ultimately lower the amount of organic C and N content in sediments. High enrichment in ^{13}C stable isotope composition in mangroves around the salt ponds compared to natural mangroves was also observed in OM of sediments and in leaves. The existence of salt ponds and selective logging of mangrove trees may be responsible for ^{13}C enrichment in sediments and leaves of mangroves where macroinvertebrates obtain their food. The amount of organic matter derived from mangroves might be having less influence on the feeding behaviour of the macroinvertebrates, because samples of *A. occidentalis* and *L. subvittata* collected in natural mangroves and around the salt ponds were found to have fed on food sources that are enriched in ^{13}C compared to mangrove-derived materials. The differences observed in the enrichment of heavy carbon stable isotope composition in sediments and mangrove leaves collected in mangroves around the salt ponds compared to natural mangroves might be linked to increased salinity, light intensity and temperature. Management strategies that involve restoration of mangroves at the salt ponds and halting of the on-going clearing of mangrove areas can help to improve the natural habitats of important macroinvertebrates. Based on the results of stable isotopes of the food source and macroinvertebrates analysed in this study, it would be interesting for further study to include microalgae, seagrasses, saltmarsh and phytoplankton in the analysis as potential food sources for the species studied.

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References

Abuodha, P.A.W., Kairo, J.G., 2001. Human-induced stresses on mangrove swamps along the Kenyan coast. Hydrobiologia 458, 255–256.
Alfaro, A.C., 2008. Diet of Littoraria scabra, while vertically migrating on mangrove trees: Gut content, fatty acid, and stable isotope analyses. Estuar. Coast. Shelf Sci. 79, 718–726.
Alfaro, A.C., 2010. Effects of mangrove removal on benthic communities and sediment characteristics at Mangawhai Harbour, northern New Zealand. ICES J. Mar. Sci. 67, 1087–1104.
Alongi, D.M., 2002. Present state and future of the world's mangrove forests. Environ. Conserv. 29, 331–349 2002.
Al-Zaidan, A.S.Y., Kennedy, H., Jones, D.A., Al-Mohanna, S.Y., 2006. Role of microbial mats in Sulaibikhat Bay (Kuwait) mudflat food webs: evidence from $\delta^{13}C$ analysis. Mar. Ecol. Prog. Ser. 308, 27–36.
Amortegui-Torres, V., Taborda-Marin, A., Juan, F., Blanco, J.F., 2013. Edge effect on a *Neritina virginea* (Neritimorpha, Neritinidae) population in a black mangrove stand (Magnoliopsida, Avicenniaceae: *Avicennia germinans*) in the Southern Caribbean. Panam. J. Aquat. Sci. 8, 68–78.
Aranibar, J.N., Anderson, I.C., Epstein, H.E., Feral, C.J.W., Swap, R.J., Ramontsho, J., et al., 2008. Nitrogen isotope composition of soils, C3 and C4 plants along land use gradients in southern Africa. J. Arid Environ. 72, 326–337.
Balke, T., Bouma, T.J., Herman, P.M.J., Horstman, E.M., Sudtongkong, C., Webb, E.L., 2013. Cross-shore gradients of physical disturbance in mangroves: Implications for seedling establishment. Biogeosciences 10, 5411–5419.
Bergamaschi, B.A., Tsamakis, E., Keil, R.G., Eglinton, T.I., Montluçon, D.B., Hedges, J.I., 1997. The effect of grain size and surface area on organic matter, lignin and carbohydrate concentration, and molecular compositions in Peru margin sediments. Geochim. Cosmochim. Acta 61, 1247–1260.
Blanco, J.F., Cantera, J.R., 1999. The vertical distribution of mangrove gastropods and environmental factors relative to tide level at Buenaventura Bay, Pacific coast of Colombia. Bull. Mar. Sci. 65, 617–630.
Bosire, J.O., Dahdouh-guebas, F., Kairo, J.G., Koedam, N., 2003. Colonization of non-planted mangrove species into restored mangrove stands. Aquat. Bot. 76, 267–279.
Bouillon, S., Chandra Mohan, P., Sreenivas, N., Dehairs, F., 2000. Sources of suspended organic matter and selective feeding by zooplankton in an estuarine mangrove ecosystem as traced by stable isotopes. Mar. Ecol. Prog. Ser. 208, 79–92.
Bouillon, S., Koedam, N., Raman, A.V., Dehairs, F., 2002. Primary producers sustaining macro-invertebrate communities in intertidal mangrove forests. Oecologia 130, 441–448.
Bouillon, S., Rao, A.V.V.S., Koedam, N., Dehairs, F., 2003. Sources of organic carbon in mangrove sediments: variability. Hydrobiologia 495, 33–39.
Bouillon, S., Moens, T., Overmeer, I., Koedam, N., Dehairs, F., 2004. Resource utilization patterns of epifauna from mangrove forests with contrasting inputs of local versus imported organic matter. Mar. Ecol. Prog. Ser. 278, 77–88.
Briand, M.J., Bonnet, X., Goiran, C., Guillou, G., Letourneur, Y., 2015. Major sources of organic matter in a complex coral reef lagoon: identification from isotopic signatures ($\delta^{13}C$ and $\delta^{15}N$). PLoS One 10, e0131555.
Bui, T.H.H., Lee, S.Y., 2014. Does 'you are what you eat' apply to Mangrove Grapsid Crabs? PLoS One 9, e89074.
Burone, L., Muniz, P., Pires-vanin, A.M.S., Rodrigues, M., 2003. Spatial distribution of organic matter in the surface sediments of Ubatuba Bay (Southeastern – Brazil). Ann. Brazilian Acad. Sci. 75, 77–90.
Caut, S., Angulo, E., Courchamp, F., 2009. Variation in discrimination factors ($\Delta^{13}C$ and $\Delta^{15}N$); the effect of diet isotopic values and applications for diet reconstruction. J. Appl. Ecol. 46, 443–453.
Christensen, J.T., Sauriau, P.G., Richard, P., Jensen, P.D., 2001. Diet in mangrove snails: preliminary data on gut contents and stable isotope analysis. J. Shellfish Res. 20, 423–426.
Crowley, B.E., McGoogan, K.C., Lehman, S.M., 2012. Edge effects on foliar stable isotope values in a Madagascan tropical dry forest. PLoS One 7, e44538.
DeNiro, M., Epstein, S., 1981. Influence of diet on the distribution of nitrogen isotopes in animals. Geochim. Cosmochim. Acta 45, 341–351 2007.

Durako, M.J., Hall, M.O., 1992. Effects of light on the stable carbon isotope composition of the seagrass Thalassia testudinum. Mar. Ecol. Prog. Ser. 86, 99–101.

Ellegaard, M., Nguyen, N.T.G., Andersen, T.J., Michelsen, A., Nguyen, N.L., et al., 2014. Temporal changes in physical, chemical and biological sediment parameters in a tropical estuary after mangrove deforestation. Estuar. Coast. Shelf Sci. 142, 32–40.

Farquhar, G.D., Ehleringer, J.R., Hubick, K.T., 1989. Carbon isotope discrimination and photosynthesis. Annu. Rev. Plant Physiol. Plant Mol. Biol. 40, 503–537.

Fernandes, S.O., Dutta, P., Gonsalves, M.J., Bonin, P.C., LokaBharathi, P.A., 2016. Denitrification activity in mangrove sediments varies with associated vegetation. Ecol. Eng. 95, 671–681.

Ferreira, A.C., Lacerda, L.D., 2016. Degradation and conservation of Brazilian mangroves, status and perspectives. Ocean Coast. Manag. 125, 38–46.

Fontugne, M.R., Duplessy, J.-C., 1981. Organic carbon isotopic fractionation by marine plankton in the temperature range −1 to 31 °C. Acta Oceanol. Sin. 4, 85–90.

France, R.L., 1995. Carbon-13 enrichment in benthic compared to planktonic algae: foodweb implications. Mar. Ecol. Prog. Ser. 124, 307–312.

France, R., 1998. Estimating the assimilation of mangrove detritus by fiddler crabs in Laguna Joyuda, Puerto Rico, using dual stable isotopes. J. Trop. Ecol. 14, 413–425.

Gao, X., Yang, Y., Wang, C., 2012. Geochemistry of organic carbon and nitrogen in surface sediments of coastal Bohai Bay inferred from their ratios and stable isotopic signatures. Mar. Pollut. Bull. 64, 1148–1155.

Hill, W.R., Fanta, S.E., Roberts, B.J., 2008. ^{13}C dynamics in benthic algae: effects of light, phosphorus, and biomass development. Limnol. Oceanogr. 53, 1217–1226.

Kao, W., Tsai, H., Tsai, T., 2001. Effect of NaCl and nitrogen availability on growth and photosynthesis of seedlings of a mangrove species, *Kandelia candel* (L.) Druce. J. Plant Physiol. 846, 841–846.

Kao, W.-Y., Tsai, H.-C., Chen-Ning, S., Tyng-Tyng, T., 2002. Nutrient contents, δ^{13}C and δ^{15}N during leaf senescence in the mangrove, Kandelia candel (L.) Druce. Bot. Bull. Acad. Sin. 43, 277–282.

Kathresan, K., 2003. How do mangrove forests induce sedimentation? Rev. Biol. Trop. 51, 355–360 2003.

Kon, K., Kurokura, H., Hayashizaki, K., 2007. Role of microhabitats in food webs of benthic communities in a mangrove forest. Mar. Ecol. Prog. Ser. 340, 55–62.

Krauss, K.W., McKee, K.L., Lovelock, C.E., Cahoon, D.R., Saintilan, N., Reef, R., et al., 2014. How mangrove forests adjust to rising sea level. New Phytol. 202, 19–34.

Kruitwagen, G., Nagelkerken, I., Lugendo, B.R., Mgaya, Y.D., SEW, Bonga, 2010. Importance of different carbon sources for macroinvertebrates and fishes of an interlinked mangrove-mudflat ecosystem (Tanzania). Estuar. Coast. Shelf Sci. 88, 464–472.

Lee, R.Y., 2006. Primary Production, Nitrogen Cycling and the Ecosystem Role of Mangrove Microbial Mats on Twin Cays, Belize. Dissertation University of Georgia, pp. 150.

Lee, O.H., Williams, G.A., Hyde, K., 2001. The diets of *Littoraria ardouiniana* and *L. melanostoma* in Hong Kong mangroves. J. Mar. Biol. Assoc. 81, 967–973.

Liingilie, A.S., Kilawe, C., Kimaro, A., Rubanza, C., Jona, E., 2015. Effects of salt making on growth and stocking of mangrove forests of south western Indian Ocean coast in Tanzania. Mediterranean J. Biosci. 1, 27–31.

Lin, G., Sternberg, L., 1992. Effect of growth form, salinity, nutrient and sulfide on photosynthesis, carbon isotope discrimination and growth of red mangrove (Rhizophora mangle L.). Aust. J. Plant Physiol. 19, 509–517.

Litulo, C., 2004. Reproductive aspects of a tropical population of the fiddler crab *Uca annulipes* (H. Milne Edwards, 1837) (Brachyura: Ocypodidae) at Costa do Sol Mangrove, Maputo Bay, southern Mozambique. Hydrobiologia 525, 167–173.

Lugendo, B.R., Nagelkerken, I., Kruitwagen, G., van Der Velde, G., Mgaya, Y.D., 2007. Relative importance of mangroves as feeding habitats for fishes: a comparison between mangrove habitats with different settings. Bull. Mar. Sci. 80, 497–512.

Machiwa, F., 2000. δ^{13}C signatures of flora, macrofauna and sediment of mangrove forest partly affected by sewage wastes. Tanzania J. Sci. 26, 15–27.

Masalu, D.C.P., 2000. Coastal and marine resource use conflicts and sustainable development in Tanzania. Ocean Coast. Manag. 43, 475–494.

Mazda, Y., Magi, M., Nanao, H., Kogo, M., Miyagi, T., Kanazawa, N., et al., 2002. Coastal erosion due to long-term human impact on mangrove forests. Wetl. Ecol. Manag. 10, 1–9.

McKee, K.L., Feller, I.C., Popp, M., Wanek, W., 2002. Mangrove isotopic (δ^{15}N and δ^{13}C) fractionation accross a nitrogen vs. phosphorous limitation gradient. Ecology 83, 1065–1075.

Meziane, T., Sanabe, M.C., Tsuchiya, M., 2002. Role of fiddler crabs of a subtropical intertidal flat on the fate of sedimentary fatty acids. J. Exp. Mar. Biol. Ecol. 270, 191–201.

Middelburg, J.J., Nieuwenhuize, J., Slim, F.J., Ohowa, B., 1996. Sediment biogeochemistry in an east African mangrove forest (Gazi Bay, Kenya). Biogeochemistry 34, 133–155.

Miranda, N.A.F., Perissinotto, R., 2012. Stable isotope evidence for dietary overlap between Alien and Native Gastropods in Coastal Lakes of Northern KwaZulu-Natal, South Africa. PLoS One 7, e31897.

Mmochi AJ, Mozes N, Dubi AM, Gordin H, Jiddawi N, Kissil G, Msuya F, Mwangamilo J. Design and preliminary results of an integrated mariculture pond system (IMPS) at Makoba, Zanzibar, Tanzania In: (M. D Richmond and J Francis edds), Proceedings of the. 20th Annivarsary on Advance Marine Science in Tanzania. WIOMSA book series. 2001; No 1. PP 431–450: Zanzibar, Tanzania.

Mokhtari, M., Savari, A., Rezai, H., Kochanian, P., Bitaab, A., 2008. Population ecology of fiddler crab, *Uca lactea annulipes* (Decapoda: Ocypodidae) in Sirik mangrove estuary. Iran. Estuar. Coast. Shelf Sci. 76, 273–281.

Mokhtari, M., Ghaffar, M.A., Usup, G., Cob, Z.C., 2015. Determination of key environmental factors responsible for distribution patterns of fiddler crabs in a tropical mangrove Ecosystem. PLoS One 10, e0117467.

Molnar, N., Marchand, C., Deborde, J., Della, P.L., Meziane, T., 2014. Seasonal pattern of the biogeochemical properties of mangrove sediments receiving shrimp farm effluents (New Caledonia). J. Aquac. Res. Develop. 5, 1–13.

Muzuka, A.N.N., Shunula, J.P., 2006. Stable isotope compositions of organic carbon and nitrogen of two mangrove stands along the Tanzanian coastal zone. Estuar. Coast. Shelf Sci. 66, 447–458.

Mwandya, A.W., Gullström, M., Öhman, M.C., Andersson, M.H., Mgaya, Y.D., 2009. Fish assemblages in Tanzanian mangrove creek systems influenced by solar salt farm constructions. Estuar. Coast. Shelf Sci. 82, 193–200.

Nehemia, A., Kochzius, M., 2017. Reduced genetic diversity and alteration of gene flow in a fiddler crab due to mangrove degradation. PLoS One 12, e0182987.

Nehemia, A., Huyghe, F., Kochzius, M., 2017. Genetic erosion in the snail *Littoraria subvittata* (Reid, 1986) due to mangrove deforestation. J. Molluscan Stud. 83, 1–10.

Ocholla, G.O., Bunyasi, M.M., Asoka, G.W., Pacha, O., Mbugua, H.K., Mbiti, S., et al., 2013. Environmental issues and socio-economic problems emanating from salt mining in Kenya; a case study of Magarini district. Int. J. Humanit. Soc. Sci. 3, 213–223.

Pe'rez, A.C., Carmona, R.M., Farin˜a, M.J., Armesto, J.J., 2009. Selective logging of lowland evergreen rainforests in Chiloe´ Island, Chile: effects of changing tree species composition on soil nitrogen transformations. For. Ecol. Manag. 258, 1660–1668.

Penha-Lopes, G., Bouillon, S., Mangion, P., Macia, A., Paula, J., 2009. Population structure, density and food sources of Terebralia palustris (Potamididae: Gastropoda) in a low intertidal Avicennia marina mangrove stand (Inhaca Island, Mozambique). Estuar. Coast. Shelf Sci. 84, 318–325.

Ramaswamy, V., Gaye, B., Rao, P., Chivas, A., Wheeler, D., Thwin, S., 2008. Distribution and sources of organic carbon, nitrogen and their isotopic signatures in sediments from the Ayeyarwady (Irrawaddy) continental shelf, northern Andaman Sea. Mar. Chem. 111, 137–150.

Rao, R.G., Woitchik, A.F., Goeyens, L., Vanriet, A., Kazungu, J., Dehairs, F., 1994. Carbon, nitrogen contents and stable carbon-isotope abundance in mangrove leaves from an East-African coastal lagoon (Kenya). Aquat. Bot. 47, 175–183.

Rubenstein, D.R., Hobson, K.A., 2004. From birds to butterflies: animal movement patterns and stable isotopes. Trends Ecol. Evol. 19, 256–263.

Semesi, A.K., 1992. Developing management plans for the mangrove reserves of mainland Tanzania. Hydrobiologia 247, 1–10.

Sjöling, S., Mohammed, S.M., Lyimo, T.J., Kyaruzi, J.J., 2005. Benthic bacterial diversity and nutrient processes in mangroves: Impact of deforestation. Estuar. Coast. Shelf Sci. 63, 397–406.

Swap, R.J., Aranibar, J.N., Dowty, P.R., Gilhooly, W.P., Macko, S.A., 2004. Natural abundance of ^{13}C and ^{15}N in C3 and C4 vegetation of southern Africa: patterns and implications. Glob. Chang. Biol. 10, 350–358.

Torres, P., Alfiado, A., Glassom, D., Jiddawi, N., Macia, A., Reid, D.G., Paula, J., 2008. Species composition, comparative size and abundance of the genus Littoraria (Gastropoda: Littorinidae) from different mangrove strata along the East African coast. Hydrobiologia 614, 339–351.

Tue, N.T., Ngoc, N.T., Quy, T.D., Hamaoka, H., Nhuan, M.T., Omori, K., 2012. A cross-system analysis of sedimentary organic carbon in the mangrove ecosystems of Xuan Thuy National Park, Vietnam. J. Sea Res. 67, 69–76.

Vafeiadou, A.-M., Adao, H., De Troch, M., Moens, T., 2013. Sample acidification effects on carbon and nitrogen stable isotope ratios of macrofauna from a Zostera noltii bed. Mar. Freshw. Res. 64, 741–745.

Vågen, T.G., Walsh, M.G., Shepherd, K.D., 2006. Stable isotopes for characterisation of trends in soil carbon following deforestation and land use change in the highlands of Madagascar. Geoderma 135, 133–139.

Walters, B.B., Rönnbäck, P., Kovacs, J.M., Crona, B., Hussain, S.A., Badola, R., et al., 2008. Ethnobiology, socio-economics and management of mangrove forests: a review. Aquat. Bot. 89, 220–236.

Wang, Y., Tobey, J., Bonynge, G., Nugranad, J., Makota, V., Ngusaru, A., et al., 2005. Involving geospatial information in the analysis of land-cover change along the Tanzania coast. Ocean Coast. Manag. 33, 87–99.

Wei, L.L., Yan, C.L., Ye, B.B., Guo, X.Y., 2008. Effects of salinity on leaf delta C-13 in three dominant mangrove species along salinity gradients in an estuarine wetland, Southeast China. J. Coast. Res. 24, 267–272.

Wentworth, C.K., 1922. A scale of grade and class terms for clastic sediments. J. Geol. 30, 377–392.

Wolters, J.-W., Gillis, L.G., Bouma, T.J., van Katwijk, M.M., Ziegler, A.D., 2016. Land use effects on mangrove nutrient status in Phang Nga Bay, Thailand. L. Degrad. Dev. 27, 68–76.