Available online at: www.mbai.org.in doi:10.6024/jmbai.2023.65.2.2393-02

Seasonal variability in the water quality, sediment characteristics and macrobenthic community structure in the vicinity of finfish cage culture sites in a tropical estuary along the south-west coast of India

V. Vineetha[1,2]*, V. Kripa[1,3], G. Vineetha[1], Shelton Padua[1] and D. Prema[1]
[1]ICAR-Central Marine Fisheries Research Institute, Kochi- 682 018, Kerala, India.
[2]Cochin University of Science and Technology, Kochi-682 022, Kerala, India.
[3]Coastal Aquaculture Authority, Chennai- 600 035, Tamil Nadu, India.

*Correspondence e-mail: vineetha.valsalan@gmail.com

Received: 20 Aug 2022 Revised: 16 Oct 2023
Accepted: 17 Oct 2023 Published: 14 Nov 2023

Original Article

Abstract

Cage culture in estuaries and backwater systems is augmented as an additional source of income among the coastal population. However, the growing popularity of cage culture has resulted in its expansion to a greater level. Hence the present study was carried out to identify the impacts associated with cage culture on the water quality, sediment characteristics, macrobenthic abundance, and the community structure of the fishing island Pizhala on the southwest coast of India on a seasonal scale. The sample was taken from six stations near the cage (<1m) and two reference stations (200-300 m away). The study revealed significant variation in the water quality between the seasons. The DO concentration during pre-monsoon (PRM) had considerable variation between the cage sites and reference sites (RF). The sediment organic carbon in cage sites was consistently higher than the RF irrespective of the season. The RF sites had higher species diversity than the cage sites contributed mainly by sensitive organisms like amphipods. The feeding guild analysis revealed a higher abundance of sub-surface deposit feeders (SSDF) and surface deposit feeders in the cage sites corresponding to an increase in the sediment organic carbon concentration.

Keywords: *Cochin estuary, habitat quality, cage aquaculture, macrobenthos, environmental impact*

Introduction

Commercial net cage culture operations form one of the most recently developed aquaculture practices with rapid expansion across the globe (Beveridge, 1996). A cornerstone of cage culture is that it can be installed in any water body with a free exchange of water (Beveridge, 1996). But in tropical estuaries, the exchange of water varies greatly with the monsoon precipitation, with very less discharges during pre-monsoon and with increased runoff during monsoon (John *et al*., 2020; Vineetha *et al*., 2020). The reduced water flow, especially during the pre-monsoon, affects the removal of waste particles from cage culture sites and in turn, alters the habitat quality of the region (Price *et al*., 2015; Tomasseti *et al*., 2016; Lima *et al*., 2019). Environmental impact assessments associated with cage culture have been conducted by several workers worldwide (Wu, 1995; Tomasseti *et al*., 2016). These studies investigated the impacts of cage culture on both the water column and the benthic ecosystem. The important consequences that cage culture impart on the ecosystem are on the sediment geochemistry and the distribution of the benthic fauna beneath the cages and in the vicinity of the cages (Wildish *et al*., 2004; Kaya and Pulatsu, 2017; Lima *et al*., 2019). Local changes in the water quality are also reported when the flushing rate is very low at the cage culture sites (Price *et al*., 2015).

Prior research indicated high organic contamination in cage sites as only 75% of feed given gets consumed by the fish stocked in the cage, whereas the rest gets deposited in the sediment beneath the cage (Holmer *et al*., 2005; Morata *et al*., 2015). Faecal particles together with the unconsumed feed exert pressure on the benthic environment by increasing the organic load (Wu, 1995; Tomasseti *et al*., 2016; Kaya and Pulatsu, 2017) thereby altering the benthic community structure concurrent to the tolerances of the diverse benthic organisms (Liao, 2019).

Since its commercialisation, cage culture has been widely adopted by the coastal fishermen community of the Ernakulam district, in Kerala, India in a fishing village, Pizhala. Pizhala is one of the prime locations for cage farming of finfish (Vineetha *et al.*, 2020). Hence, proper validation of the pros and cons of cage farming is essential to formulate policies, procedures and guidelines for ecosystem-friendly finfish cage farming in this area and for similar environments where cage culture activities are practised.

Material and methods

Study area

Pizhala is a small island located in the Cochin Estuary (CE), a micro-tidal estuary along the southwest coast of India. Small-scale fishing activities are using cast nets and Chinese dip nets and these form the chief source of livelihood of the population to date (Vineetha *et al.*, 2020). Since 2015, in addition to these fishing practices, Pizhala has turned out to be a prime location for cage fish culture practices in the CE. In this cage culture, fishes such as *Etroplus suratensis, Lates calcarifer, Mugil* spp., *Oreochromis* spp. are stocked in net mesh cages of size 4×4×2 m, fitted on to galvanised iron pipe frames attached with buoys for flotation (Joseph and Gopalakrishnan, 2017). The fish stocked in these cages were mainly fed with trash fish and artificial feeds. As cage farming has successfully emerged as a potential farming practice with large economic turnover within short periods, it is being widely adopted as a major fin fish farming practice by the residents of this island leading to its intensification in CE (Joseph and Gopalakrishnan, 2017). Hence the present study aimed to understand the impacts associated with intensified cage culture practices on the sediment chemistry, water quality, macrobenthic abundance and community structure along the selected cage culture sites of Pizhala Island (Fig. 1).

Sampling strategy

Sampling was carried out from six locations near the cage culture sites and two reference sites (RF) located 200-300 m away from the cages. The sampling was conducted twice during Pre-monsoon (PRM), Monsoon (MN) and Post-monsoon (PM) during the year 2018. Since Kerala experiences heavy monsoonal showers, it is seen that the top layer of the sediment in the CE is washed away and new sediment is replaced by runoff from rivers (John *et al.*, 2020). Kerala was confronted with a devastating flood in the year 2018, hence, the first monsoonal sampling was conducted before the flood and the second monsoonal sampling was carried out after the flood.

Hydrography

For the water quality analysis, samples were collected using a Niskin water sampler (1.7 l capacity, Norinco). The surface water temperature (ST) of the study region was recorded

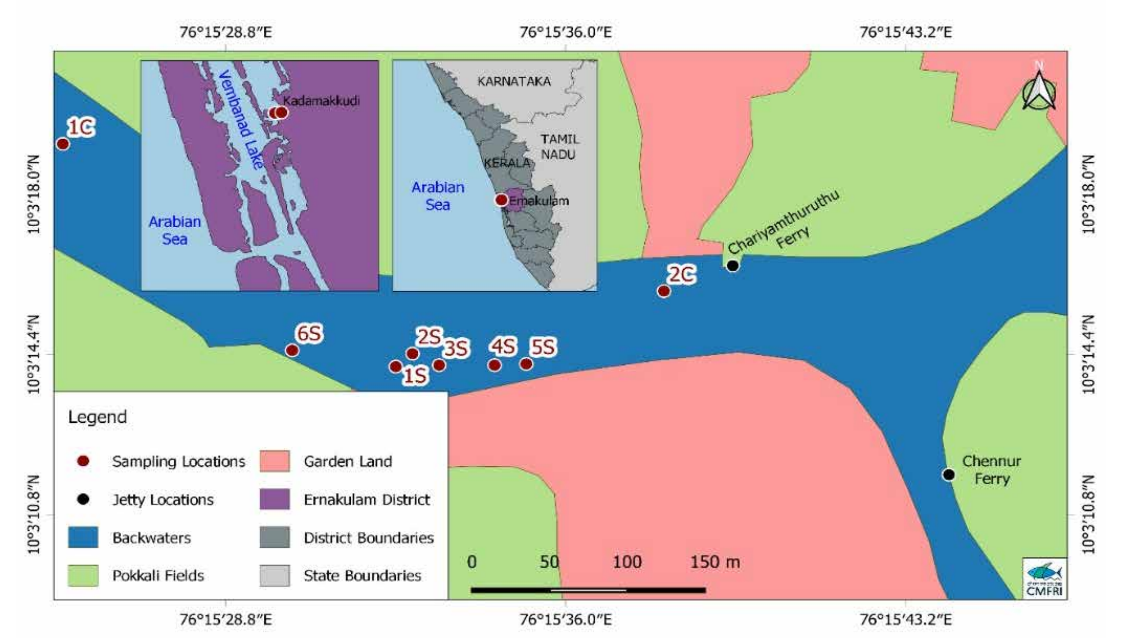

Fig. 1. Map of sampling locations

using a standard mercury thermometer. Surface salinity was determined using an optical refractometer (ERMA INC 0-100%). Dissolved oxygen (DO) and Biological Oxygen Demand (BOD) were determined by Winkler's titrimetric method (Grasshoff *et al.*, 1983). Chlorophyll *a* was estimated following the standard protocol of SCOR-UNESCO (1966). Total suspended solids (TSS) were determined by filtering 250 ml water through pre-weighed Millipore membrane filter paper (47mm dia.; pore size 0.45μm). The filter paper was then dried subsequently at 80˚C and re-weighed. The difference in the weight of filter paper before and after the filtering and drying process gives the amount of TSS (APHA, 2005). The dissolved inorganic nutrients (Ammonia, Nitrate, Nitrite, Phosphate and Silicate) were estimated using standard calorimetric methods (Grasshoff *et al.*, 1983)

Sediment characteristics

Sediment samples were collected using a Van Veen grab (0.05 m2). Sediment texture was analysed using the international pipette method (Krumbein and Pettijohn, 1938). The class to which the sediment belonged was examined using the textural triangle method available on the website of the United States Department of Agriculture (https://www.nrcs.usda.gov). Sediment organic carbon (OC) was analysed by the Walkley and Black titration method (Walkley and Black, 1934). Oxidation-reduction potential (E*h*) was analysed using a large screen ORP tester (ORPTestr10BNC -999 mV to +1000 mV).

Macrobenthic community

Macrobenthos were collected using a Van Veen grab (0.05 m^2). Sediment collected in the grab was passed through a 0.5 mm sieve (Birkett and McIntyre, 1971) and the organisms retained in the sieve were transferred to bottles and preserved in 5% Rose Bengal-formalin solution. Organisms collected were later identified up to the lowest possible taxa using standard identification manuals (Day, 1967; Gosner 1971; Fauchald, 1977). The numerical density was expressed as individuals per meter square (ind. m^{-2}) and the wet biomass (wet weight g m^{-2}) was determined using a high-precision electronic balance (Sartorius CP225D). Species identification of the macrobenthic taxa having a high numerical density in the benthic samples, such as Polychaeta and Amphipoda was carried out using standard identification manuals and keys (Day, 1967; Fauchald, 1977; WoRMS, 2020). For understanding the trophic ecology and feeding preferences of the macrobenthic community, relevant literature was used and was categorised accordingly as Carnivorous (CVR), Suspension feeders (SF), Surface Deposit Feeders (SDF), and Subsurface Deposit Feeders (SSDF) (Fauchald and Jumars, 1979; Jumars *et al.*, 2015; Jayachandran *et al.*, 2019; Rehitha *et al.*, 2019).

Statistical analysis

A paired test (with two-tailed P values and 95% confidence intervals) was performed between the abiotic and biotic variables of the cage sites and reference sites (RF) to understand the significance of variation existing between these two regions. To understand the seasonal variation in the distribution of the abiotic variables and macrobenthic biomass and abundance One-way ANOVA ($p<0.0001$) was performed using the statistical software Graphpad Prism version 5.01. Before the analysis, the datasets were checked for their normality in the distribution using the Kolmogorov-Smirnov test and based on the result, parametric t test and ANOVA were carried out. The variability in the polychaete species diversity between the cage and reference site and also along seasons were analysed using Univariate indices, Shannon Wiener index (H'; log2) for species diversity using PRIMER version 6.1.5 (Clark and Warwick, 2001).

Results and discussion

Stocking density and biomass

At the time of sampling, about 30 fin fish cages were installed at a length of about 200 m in the region. The average depth of the region only 2-3 m. In Pizhala, two cycles of finfish culture are practised every year with one cycle getting harvested by April and the other by January. On average, the farmers stock about 2000 fingerlings in the cages so that at least 500 fish reach adulthood for harvesting. During the pre-monsoon (PRM), the cages had a stocking density of average. 498±149 fishes/cage with an average biomass of 199.24±60 kg/cage. With the beginning of monsoon (MN), the cages had a stocking density of 645±144 fishes/cage with an average density of 96.75±29.33 kg/cage. In the devastating flood of 2018, the cages got inundated in the muddy water or were washed away as reported by Joseph *et al.* (2018). After the flood, the farmers re-established the cages and stocked them with new seeds. As a consequence, during the second sampling of the monsoon (September 2018), the cages were in the reviving stage, and hence the stocking density and biomass were much lesser (av. 298±61 fishes/ cage and biomass of av. 63±28 kg/cage). During the post-monsoon period (PM), stocking density was av. 276±59 fishes per cage and an average biomass of 151±31 kg per cage.

Hydrography

The surface water temperature was relatively higher during PRM (av. 31.83±0.98 ˚C) than monsoon (MN) (av. 27.71±1.30˚C) and PM (av. 28.31±2.73 ˚C) with statistically significant seasonal variation ($p<0.0001$). During both the pre-monsoon and post-monsoon, surface salinity showed more or less similar values (av. 20.5 and 21ppt during PRM and PM respectively) irrespective of cage sites and reference sites (RF). However, during MN, the salinity dropped throughout the sampling locations with an average salinity of 1 ppt in both the cage sites as well as RF (Fig. 2 b). Along a seasonal scale, though dissolved oxygen (DO) concentration was low during PRM, the RF sites showed relatively high DO compared to the cage sites (av. 3.73±0.52 and 5.27±1.54 mg l^{-1} in the cage and RF sites respectively). During MN and PM, the DO levels increased irrespective of cage and RF sites though a slight variation was evident between

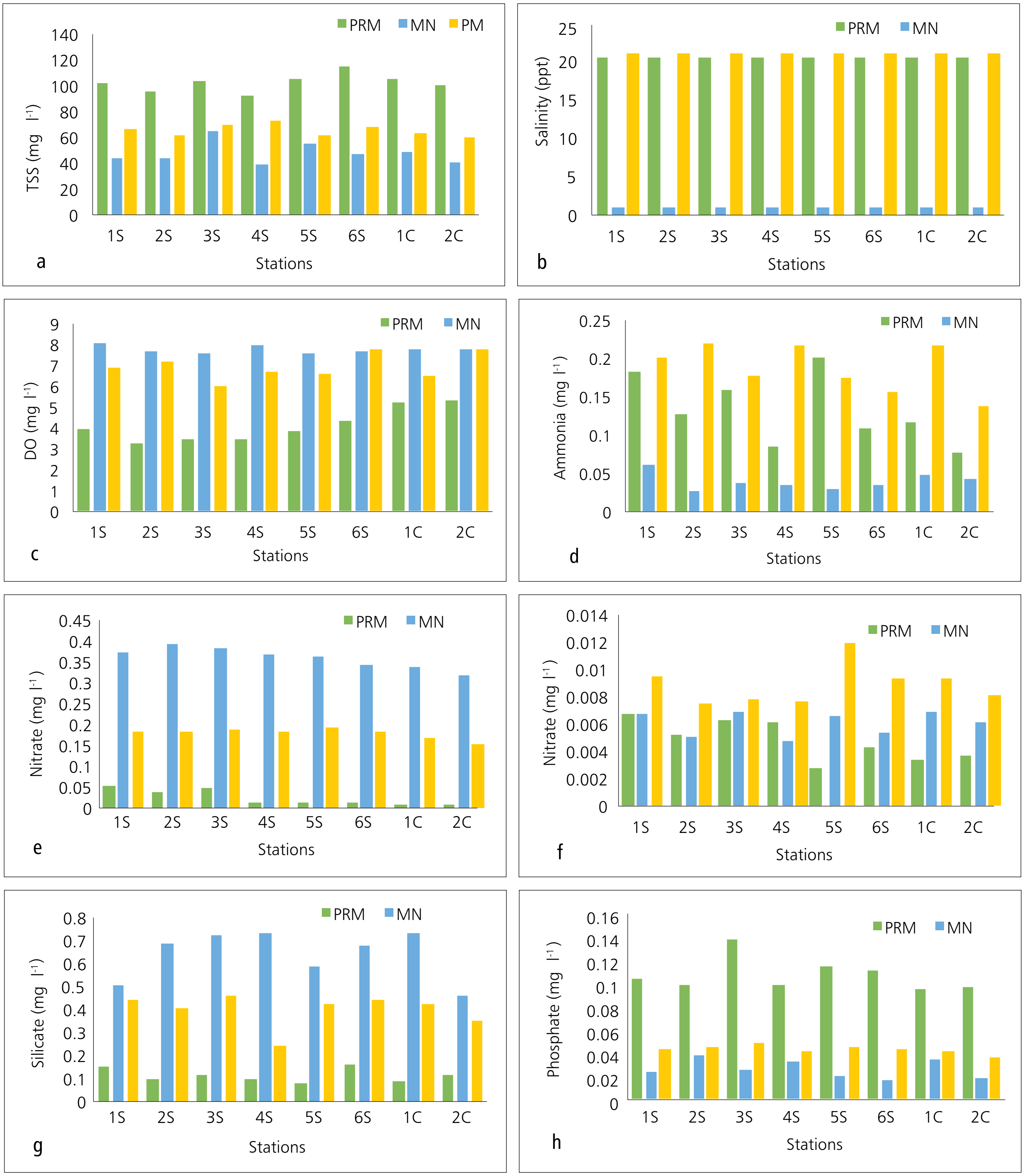

Fig. 2. Distribution of (a) TSS, (b) Salinity, (c) DO, (d) Ammonia, (e) Nitrate, (f) Nitrite, (g) Silicate, (h) Phosphate

them (av. 7.76±0.29 and 7.77±0.02 mg l^{-1} in cage and RF respectively during MN and av. 6.87±0.95 and 7.11±1.11 mg l^{-1} in cage sites and RF respectively during PM) (Fig. 2 c). The DO concentration showed statistically significant variation along seasons ($p<0.0001$). This was contrary to the widespread belief that cage culture and other aquaculture activities have an impact only on the bottom environment, some studies do mention the localized impact on the water quality (Wu, 1995; Pearson and Black, 2000). In the present study, DO was significantly lower during the PRM season in the cage sites (less than 4 mg l^{-1}) than RF which is considered as a threshold for causing stress to the fish stocked (Boyd, 2018). The localised depletion of DO levels may be due to increased respiration by fish stocked and also due to the microbial metabolism of their faecal matter (Price *et al.*, 2015). BOD was high during PRM in both cage sites and RF (av. 0.82±0.53 and 0.66±0.45 mg l^{-1} in cage sites and RF respectively). However, irrespective of the season, BOD was higher in the cage sites compared to RF (av. 0.61±0.31 and 0.67±0.48 in cage sites in MN and PM respectively and av. 0.42±0.06 mg l^{-1} and 0.48±0.25 mg l^{-1} in RF in MN and PM respectively. The high BOD in the cage sites can be linked to high microbial metabolism. However, proper siting of cages in areas with proper flushing rates is found to lower the depletion of oxygen (Braatan, 2007).

The TSS was high during PRM compared to the other two seasons in both the cage and RF sites (av. 102.22±10.95 and 102.89±13.24 mg l^{-1} in cage and RF sites respectively). When compared to MN and PM, TSS was relatively high during PM irrespective of the sampling location (Fig. 2 a) with a statistically significant variation along seasons ($p<0.0001$).

The dissolved inorganic nutrients (nitrate, nitrite, phosphate, ammonia and silicate) distribution determines the health of a water body. CE is considered a eutrophic estuary (Rajaneesh *et al.*, 2015) but during monsoon, CE receives heavy freshwater influxes that influence the seasonal trend of dissolved inorganic nutrient distribution (Menon *et al.*, 2000; Vineetha *et al.*, 2020). In the present study, no evident variation was observed between the cage sites and RF except during PRM where nutrient concentrations were observed to be relatively high in the cage sites than in RF. Ammonia concentration was comparatively high in cage sites with av. 0.14±0.07 mg l^{-1} while it was av. 0.10±0.05 mg L^{-1} in RF. Nitrate concentration was av. 0.03±0.03 mg l^{-1} in cage sites and av. 0.01±0.003 mg L^{-1} in RF and Nitrite was av. 0.005±0.002 mg l^{-1} in cage sites and av. 0.004±0.004 mg L^{-1} in RF. The highest phosphate concentration was observed during PRM with av. 0.11±0.03 mg L^{-1} in cage sites and av. 0.01±0.04 mg l^{-1} in RF. Silicate concentration was relatively lower during PRM (av. 0.12±0.07 mg l^{-1} and 0.10±0.07 mg l^{-1} in both cage sites and RF sites respectively). (Fig. 2). Though the spatial variation in dissolved nutrient distribution was not evident between the cage and RF sites, a strong seasonal variability was evident in nutrient concentrations and was statistically significant with the season ($p<0.0001$) (Fig. 2), especially for nitrite, ammonia, phosphate and silicate. Silicate and nitrate were relatively high during MN (av. 0.65±0.22 mg l^{-1} in cage sites and 0.60±0.32 mg l^{-1} in RF for silicate and av. 0.37±0.09 mg l^{-1} av. 0.33±0.11 mg l^{-1} for nitrate in cage and RF respectively). The highest concentrations of nitrite and ammonia were observed during PM on av. 0.009±0.003 mg l^{-1} in cage site and av. 0.009±0.003 mg l^{-1} in RF for nitrite and the av. 0.19±0.09 mg l^{-1} in cage site and av. 0.18±0.12 mg l^{-1} in RF for ammonia. No statistically significant variation in dissolved inorganic nutrients was observed between cage sites and RF except for Nitrate during MN and PM ($p<0.05$). Overall, an increase in nutrient concentration was observed in cage sites compared to RF during PRM. These local variations may be due to the increased residence time of the waste discharged from the cages contributed by the low flushing rate. In MN and PM, negligible variation was observed in the nutrient distribution between the cage and RF and can be attributed to the increased flushing replacing the water near cages (John *et al.*, 2020).

Sediment characteristics

Pronounced variations in the Organic Carbon (OC) concentration between the cage and RF sites were recorded irrespective of the seasons. During PRM, OC was av. 35.3±3.6 mg l^{-1} in cage sites and av. 18.9±10.6 mg l^{-1} in RF. During MN there was considerable variation within the cage sites which ranged from 13.5 mg l^{-1}

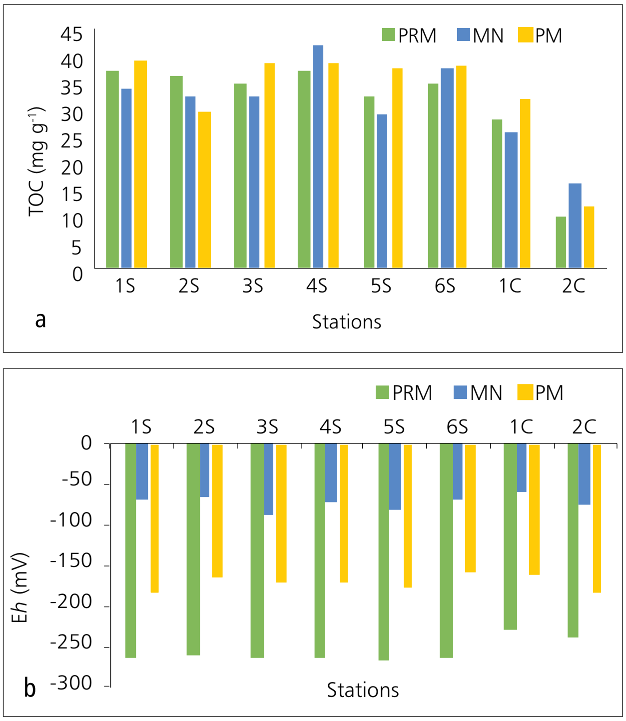

Fig. 3. (a) Total Organic carbon. (b) Redox potential (E*h*)

to 55.1 mg l^{-1} (av. 34.5±11.8 mg l^{-1}) (Fig. 3 a). While in RF, the average OC concentration was 20.8±9.0 mg l^{-1}. PM season had an OC concentration of av.36.8±5.0 mg l^{-1} and av. 21.6±13.5 mg l^{-1} in cage sites and RF, respectively. The t-test results show a statistically significant variation in OC concentration between the cage sites and RF sites ($p<0.05$) in all seasons. The significantly high OC in cage sites compared to RF irrespective of the season followed many earlier observations (Wu, 1995; Karakassis *et al.*, 2000; Sara *et al.*, 2004; Prema *et al.*, 2010; Tomasseti *et al.*, 2016). Even though similar sediment characteristics prevailed in 2C, the OC content was less when compared with the cage sites which substantiates the findings that cage culture increases the organic load and thereby increases the OC around the area.

The percentage contribution of sand, silt and clay showed marked spatial variations in both cage sites and RF. During PRM, all the sampling locations near the cage sites were clayey except 5S which was silty clay. In the RF sites, sampling location, 1C had clay loam whereas 2C was clayey. During MN, 5S showed a sandy clay texture while all other sites were clayey. 1C was clay loam and 2C was sandy clay. PM showed great variability in the sediment texture with 1S, 3S and 5S being Clayey, 4S, 2S and 6S being silty clay. Among RF, 1C had sandy clay loam substratum whereas 2C was clayey (Fig. 4).

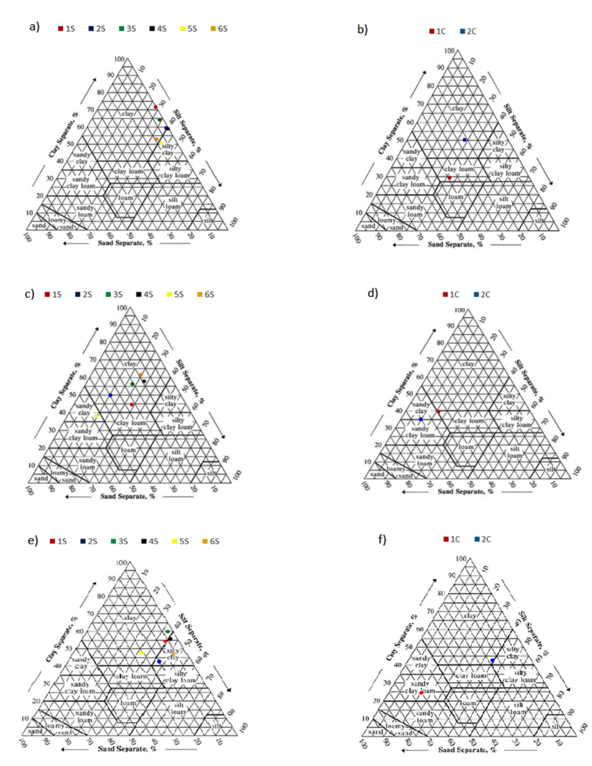

Fig. 4. Sediment texture triangle (a) PRM – cage sites, (b) PRM – RF, (c) MN – cage sites, (d) MN – RF, (e) PM – cage sites, (f) PM – RF

A change in the sediment texture at sampling locations during the second monsoonal sampling can be linked to the impact of the devastating flood that occurred before the sampling period. The higher freshwater discharges might have washed away the sediment and led to a shift in the sediment texture as observed by Vineetha *et al.* (2020). Soil texture plays a vital role in the distribution of the OC content as finer sediments characterised by more surface area have an increased ability to hold organic matter (Krull *et al.*, 2003; Nayar *et al.*, 2007). Baldock *et al.* (1992) graded the texture of sediment according to its capacity to hold organic particles of the order sand<silt<clay. As reported widely, cage culture imparts an evident impact on the sediment quality.

E*h* distribution had a similar trend in all the sampling locations irrespective of cage and RF sites (Fig. 3(b). However, the seasonal variation was prominent with PRM showing more negative (av. -248.65±41.1) redox potential compared to MN characterized by less negative redox potential (av. -71.6±78.1) with a statistically significant variation ($p<0.0001$). Increased organic carbon in the sediment affects the oxygen distribution as it consumes the oxygen available on the sediment surface resulting in hypoxia and finally culminating in anoxia (Holmer *et al.*, 2005). The intensity of anaerobic conditions in the sediment is generally assessed by measuring the redox potential (E*h*) (Wilding, 2012). Sediments having an E*h* value less than -100 mV are considered to be anoxic (Holmer *et al.*, 2005) and are found to have the highest organic carbon concentration since the surface sediment becomes completely reduced (Holmer *et al.*, 2005). In the present study, the observed higher negative E*h* during PRM and lower negative E*h* during MN indicate the increased oxygen replenishments of the sediment corresponding to increased freshwater discharges during monsoon. Compared to the MN sample collected before the flood, the sample collected after the flood was less negative. The higher E*h* values concurrent to the flood further substantiate the view (John *et al.*, 2020).

Biotic components

Phytoplankton biomass (Chlorophyll a mg/m^3)

Chlorophyll *a*, the representative measure for phytoplankton biomass exhibited higher concentration during PRM (av. 24.59±2.1 and 24.78±2.6 mg m^{-3} in cage and RF sites respectively) compared to MN (av. 7.86±3.9 and 5.05±3.69 mg m^{-3} in cage and RF respectively) and PM (Av. 10.51±1.97 and 9.49±2.75 mg m^{-3} in cage and RF respectively) with a statistically significant variation with seasons ($p<0.0001$).

Macrobenthic community

The physical and chemical characteristics of the sediment determine the size, distribution and composition of the macrobenthos (Rao and Sarma, 1982; Gray *et al.*, 2002; Hargrave *et al.*, 2008). Among the chemical attributes, OC and E*h* are often altered by the pollution from anthropogenic activities like aquaculture thereby altering the benthic abundance and community structure of the region. The macrobenthic community exhibited evident seasonal variations in the community structure with PM having the maximum benthic representatives and MN with the lowest representatives. Macrobenthic biomass was high during PM followed by PRM and MN irrespective of cage sites and RF (av. 2.9±2.6 g m^{-2}, 2.6±2.4 g m^{-2} and 5.1±3.8 g m^{-2} during PRM, MN and PM respectively in cage sites and 9.2±6.6 g m^{-2}, 5.6±4.5 g m^{-2} and 6.4±2 g m^{-2} during PRM, MN and PM respectively during RF) (Fig. 6b). Macrobenthic abundance (Fig. 6a) also showed a similar trend similar to the macrobenthic biomass (Table 1). The t-test results show a statistically significant variation ($p<0.05$) in the Macrobenthic abundance between cage sites and RF sites during both PRM and MN.

The seasonal abundance of macrobenthic communities from the cage site and reference site are shown in Fig. 5. The macrobenthic community of the Pizahala was represented by Polychaeta, Amphipoda, Gastropoda, Bivalvia, Nemertea, Tanaidacea and Oligochaeta during PRM with Nemertea and

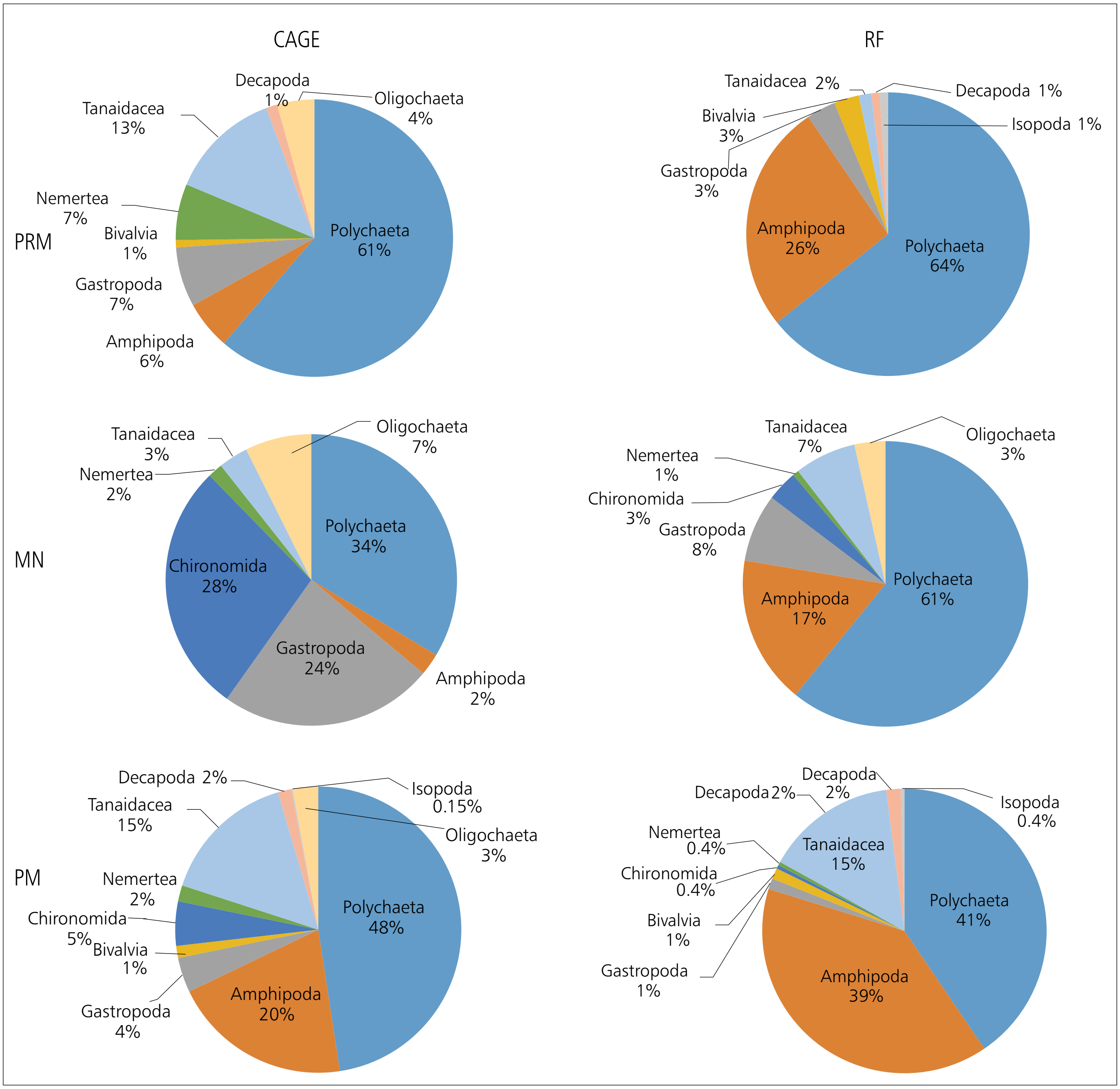

Fig. 5. Percentage composition of different groups of macrobenthos

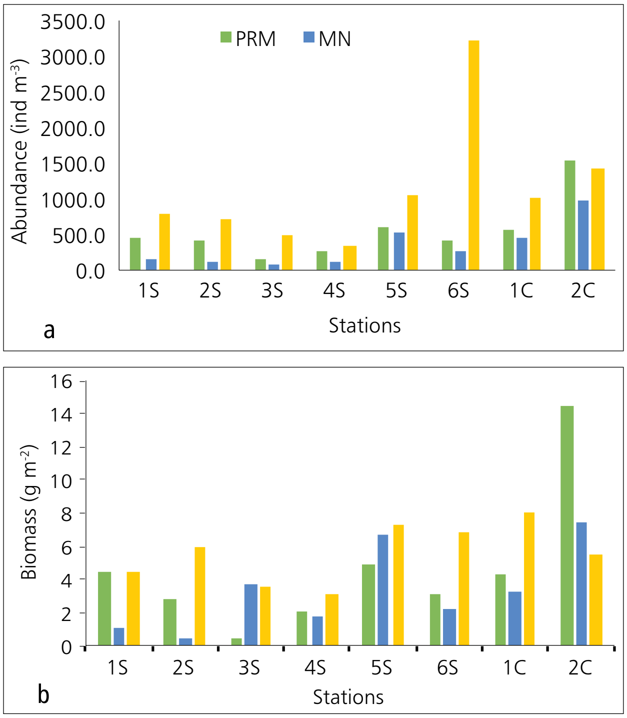

Fig. 6. Distribution of (a) macrobenthic abundance, (b) macrobenthic biomasswice the size of males, indicating sexual dimorphism in body size

Oligochaeta observed only in cage sites. Isopoda was observed only in RF sites during PRM. During MN, both cage sites and RF were represented by Polychaeta, Amphipoda, Gastropoda, Chironomidae, Nemertea, Tanaidacea and Oligochaeta. During PM, Isopoda was also observed in both cage sites and RF, but Oligochaeta was absent in RF. Polychaeta formed the most dominant taxa in the cage sites irrespective of seasons, though their percentage contribution to the benthic abundance varied among seasons (61.3%, 33.61% and 47.58% in PRM, MN and PM respectively in cage sites). However, in RF sites, Amphipoda had a similar share (39.26%) as that of Polychaeta (40.50%) to the benthic abundance during PM. In cage sites, the percentage contribution of Amphipoda was always lower compared to the RF sites irrespective of the season (5.65% and 26.07% during PRM, 2.46% and 16.78% during MN, and 20.30 and 39.26% during PM). During PRM in the cage sites, Tanaidaceae formed the second dominant taxa (13.04%) followed by Gastropoda (6.96%), Nemertea (6.52%), Amphipoda (5.65%), Oligochaeta (4.34%), Decapoda (1.30%) and Bivalvia (0.87%). During MN, Polychaeta was followed by Chironomidae (27.87%). Gastropods had a major contribution to the macrobenthic community of MN (23.77%) compared to PM (3.94%). Other macrobenthic taxa that were observed during MN were Oligochaeta (7.38%), Tanaidaceae (3.28%), Amphipoda (2.46%) and Nemertea (1.64%). Decapoda and Bivalvia were absent. Macrobenthic abundance and the number of macrobenthic taxa observed were relatively high during PM. Polychaeta formed the dominant taxa with a contribution of 47.59% followed by Amphipoda (20.30%), Tanaidaceae (15.45%) Chironomidae (5%), Gastropoda (3.94%), Oligochaeta (2.88%), Nemertea (1.82%), Decapoda (1.52%), Bivalvia (1.36%) and Isopoda (0.15%).

In RF sites, during PRM, Chironomidae, Bivalvia, and Nemertea were absent. Other groups present were Polychaeta (64%), Amphipoda (26.07%), Gastropoda (3.32%), Bivalvia (2.84%), Tanaidacea (1%), Decapoda (0.94%) and Oligochaeta (0.94%). During MN, the macrobenthic community was represented by Polychaeta (60.84%), followed by Amphipoda (16.78%), Gastropoda (7.69%), Tanaidacea (6.99%%), Chironomidae (3.50%), Oligochaeta (3.50%) and Nemertea (0.70%). In PM, Amphipoda and Polychaeta were followed by Tanaidaceae (14.88%), Decapoda (1.65%), Gastropoda (1.24%), Bivalvia (1.24%), Chironomidae (0.41%), Isopoda (0.41%) and Nemertea (0.41%).

Pearson and Rosenberg (1978) have done an elaborate study on the changes occurring in the benthic community in conjunction with the increase in organic pollution with species number and biomass increasing initially with higher organic carbon loads inhabited by small opportunistic species to a limit after which it gets reduced finally reaching an a-faunal state. Likewise, when the organic pollution reduces, a shift in the species distribution can also be found. An increase in the number of crustaceans in the RF sites compared to the cage sites in the present study substantiates Pearson and Rosenberg's findings. In CE, an earlier study conducted on the flood impacts on the benthopelagic community reported an increase in the crustacean abundance (Vineetha *et al*., 2020) which was observed to be similar to the present study. Though opportunistic benthic species have been reported from the CE frequently (Martin *et al*., 2011; Rehitha *et al*., 2019; Vineetha *et al*., 2020) their increased preponderances and dominance in the benthic community structure is a matter of concern.

The abundance of aquatic insect larvae particularly of the Chironomidae family in large numbers during monsoon might have happened by their aggregating behaviour upon favourable environmental conditions. The higher abundance of Chironomids during the rainy season has been reported by Santana *et al.* (2015) and pointed toward their tolerances to wide salinity (Thangasamy *et al*., 2016) and oxygen levels (Santana *et al*., 2015). Compared to cage sites, the lower abundance of Chironomidae in the RF sites indicates their adaptability to habitat changes. Other benthic groups like gastropods, crustaceans, bivalves, etc. are sensitive to varied pollutants (Dauvin and Ruellet, 2007) though the degree of sensitivity varies between different species. Gastropoda was represented exclusively by the species, *Nassodonta insignis*

Table 1. Average macrobenthic abundance (No/m^2) in cage site and reference site during pre-monsoon, monsoon and post-monsoon (av. ± 1 SD)

Name	Feeding guild	PRM		MN		PM	
		Cage	RF	Cage	RF	Cage	RF
Polychaeta							
Capitella capitata	SSDF†	82± 74	25 ± 17	12 ± 31	80 ± 102	19 ± 25	154 ± 234
Dendroneries aestuarina	CVR‡	0	10 ± 10	10 ± 20	164 ± 155	25 ± 69	88 ± 90
Marphysa macintoshi	CVR	2± 6	0	0	0	7 ± 14	0
Mediomastus capensis	SSDF	34± 36	145 ± 100	30 ± 48	172 ± 157	64 ± 102	66 ± 59
Namalycastis indicus	CVR	4± 8	0	15 ± 29	20 ± 24	89 ± 124	64 ± 56
Nephtys polybranchia	CVR	15± 26	30 ± 23	0	0	7 ± 14	8 ± 20
Prionospio cirrifera	SDF§	69± 53	220 ± 157	2 ± 6	14 ± 20	204 ± 250	72 ± 58
Prionospio cirrobranchaita	SDF	32± 60	245 ± 210	0	20 ± 24	112 ± 161	44 ± 69
Amphipoda							
Cheiriphotis geniculata	SF¶	12± 24	240 ± 337	5 ± 13	128 ± 180	50 ± 118	322 ± 427
Corrophium sp.	SF	0	0	0	0	5 ± 10	6 ± 10
Idunella sp.	SF	5± 10	20 ± 15	0	4 ± 10	44 ± 71	140 ± 128
Victoriopisa chilkensis	CVR	5± 13	15 ± 17	0	6 ± 10	125 ± 219	50 ± 42
Gastropoda							
Nassodonta insignis	SDF	27± 38	35 ± 22	49 ± 73	54 ± 56	44 ± 59	16 ± 20
Bivalvia							
Mytilus sp.	SF	4± 8	20 ± 15	0	0	4 ± 12	0
Villorita sp.	SF	0	10 ± 10	0	0	12 ± 41	18 ± 30
Chironomida	SDF	0	0	57 ± 45	20 ± 50	55 ± 101	6 ± 10
Nemertea	CVR	25± 34	5 ± 9	4 ± 8	6 ± 10	20 ± 37	4 ± 10
Tanaidacea	SDF	50± 137	15 ± 17	7 ± 18	50 ± 42	170 ± 353	172 ± 158
Isopoda	CVR	0	0	0	0	2 ± 6	4 ± 10
Decapoda	CVR	5± 10	10 ± 10	0	0	17 ± 38	16 ± 40
Oligochaeta	SDF	17± 31	10 ± 18	15 ± 29	28 ± 20	32 ± 92	0

†SSDF- Sub-surface Deposit feeders, ‡CVR- Carnivorous, §SDF- Surface Deposit feeders, ¶ SF- Suspended feeders.

which was observed throughout the year in the cage sites with higher abundance during MN. *N. insignis* is often observed in sediments enriched with organic carbon (Jayachandran *et al.*, 2019). Their lower abundance in the RF site might have resulted from the low levels of OC in those regions. Bivalvia was found to be less abundant throughout the study period. Nemerteans commonly found in relatively shallow environments can tolerate organic pollution (Albayrak *et al,* 2006). In the present study, the relatively higher abundance of nemerteans in cage sites compared to RF during PRM indicates their pollution tolerance ability. As the sediment resident time was more during PRM (John *et al.*, 2020) it might have provided an undisturbed predatory ground for nemerteans culminating in their higher abundance. Oligochaetes, also considered among opportunistic organisms able to tolerate high levels of organic carbon (Rehitha *et al.*, 2019) were also more in the cage sites in the present study thus corroborating their habitat preferences.

Polychaeta formed the most abundant and diverse taxa among the macrobenthic community throughout the sampling period. During PRM, seven Polychaete species were observed in the cage site while eight species were recorded during PM and five during MN. In RF sites, six species were observed during PRM and MN whereas in PM seven species were observed. During PRM and PM, genus *Prionospio* represented by *P.cirrifera* and *P.cirrobranchiata* dominated in both cage and RF sites contributing to 26.1% during PRM and 28.63% to total polychaete composition in cage sites and 44.1% and 10.7% during PRM and PM respectively in RF sites. During PRM, the family Capitellidae represented by *Capitella capitata* and *Mediomastis capensis* were abundant in cage sites whereas in RF they had a higher contribution during PM. The species diversity index varied from 0 to 2.2 in cage sites and 1.80 to 2.46 in RF. The monsoon period exhibited fewer species diversity (H–) in cage sites (av. 1.02±0.89) than PRM (av.

Table 2. Results of One-way ANOVA of abiotic variables and macrobenthic biomass and abundance along seasons

Parameter	p value
Water quality	
Temperature	< 0.0001
pH	< 0.0001
DO	< 0.0001
BOD	0.2557
Ammonia	< 0.0001
Nitrate	< 0.0001
Nitrite	0.0006
Phosphate	< 0.0001
Silicate	< 0.0001
Chlorophyll a	< 0.0001
TSS	< 0.0001
Sediment characteristics	
Organic carbon	0.5163
pH	< 0.0001
E*h*	< 0.0001
Macrobenthos	
Abundance	0.061
Biomass	0.0612

Table 3. Results of t-test analysis done between cage sites and RF sites in Pre-monsoon (PRM), Monsoon (MN) and Post-monsoon (PM)

Parameters	p value		
Water quality	PRM	MN	PM
SST	0.03683	0.27336	0.64376
DO	0.00224	0.9318	0.66209
BOD	0.35091	0.2777	0.26533
Ammonia	0.22656	0.57776	0.67899
Nitrate	0.23307	0.031044	0.00705
Nitrite	0.17667	0.60365	1
Phosphate	0.20769	0.80696	0.30182
Silicate	0.59483	0.54218	0.85793
Chlorophyll a	0.88108	0.12366	0.40352
TSS	0.91641	0.58701	0.2196
Sediment characteristics			
OC	0.01073	0.03052	0.03052
pH	0.13398	0.85462	0.85462
Eh	7.9E-05	0.9924	0.9924
Sand	0.00195	0.03743	0.037426
Silt	0.11318	0.1172	0.1172
Clay	0.03935	0.03485	
Macrobenthos			
Abundance	0.04142	0.02547	0.90328
Biomass	0.05285	0.21869	0.32419

1.55±0.19) and PM (1.40±0.77) but in RF sites Pre-monsoon and Monsoon had almost similar species diversity index (av. 2.06±0.38 and 2.07±0.18 during PRM and MN respectively) while PM had higher species diversity (2.24±0.31).

Species like *Capitella capitata, Mediomastus capensis* and *Prionospio* sp. are known to flourish in sediments characterised by higher levels of OC, sulphides and low levels of oxygen (Pearson and Rosenberg, 1978; Tomassetti and Porrello, 2005; Ansari *et al*., 2014). In the present study, the polychaetes, *C. capitata*, *M. capensis*, *P. cirrifera* and *P. cirrobranchiata* contributed a major share of the macrobenthic community of PRM and MN though their abundance was less compared to PM. These polychaete species are small sized offering a higher surface area to volume ratio. This strengthens their capacity to assimilate more oxygen (Levin, 2003). The polychaete species, *Marphysa gravelyi* was observed in one cage site during PRM and at three sampling locations during PM. *M. gravelyi* is generally found in brackish waters (Malathi *et al*., 2011). Mandario *et al.* (2019), observed *Marphysa* sp. having the ability to improve the sediment quality by reducing the OC and sulphur levels and hence have been recommended as bio remediators in organically enriched aquaculture farms. The absence of *M. gravelyi* in RF sites might have happened because of the reduction in OC.

The abundance of Amphipoda also varied greatly among seasons with PM having higher macrobenthic abundance and MN the least. Only one amphipod species, *Cheiriphotis geniculata* was observed in cage sites during MN while three species were observed in PRM and four species in PM. In the RF sites, three amphipod species were observed during PRM and MN while in PM four species were observed among which the species, *Victoriopisa chilkensis* (57.12%) dominated. The other species that were observed during PRM were *Corrophium* sp., *Idunella* sp. and *V. chilkensis*. Even though amphipods are considered to be sensitive to pollution, some species show exceptions (Afli *et al*., 2008). *Victoriopisa chilkensis* is reported in regions of high OC in CE (Nisha *et al*., 2007) which was also found to be true in the present study with reportedly higher abundance in the cage sites during PM. However, the occurrence of sensitive amphipod species like *Idunella* sp., *Cheriophotis geniculata* and *Corophium* sp. were restricted to very few sites in the cage sites whereas in RF they were the dominant amphipod species. In MN, a drop in amphipod abundance was noticed similar to all other macrobenthic taxa.

Macrobenthic feeding guild

Macrobenthos exhibits diverse feeding behaviours (Snelgrove, 1998) depending on the sediment texture, salinity, availability of detritus and other organic matters. Organisms are grouped into different feeding guilds based on their feeding patterns and characteristics (Fauchald and Jumars, 1979). Feeding guild analysis enables us to understand the type of food available in the habitat and also the carbon flow in the ecosystem (Fauchald, 1977). The collected specimens were assigned to four different feeding guilds such as surface deposit feeders (SDF), sub-surface deposit feeders (SSDF), suspension feeders (SF) and carnivorous (CVR). Cage sites showed a dominance of SDF in all seasons (50.4% in PRM, 63.1% in MN and 55.91% in PM) followed by SSDF (30%), CVR (14.3%) and SF (5.2%) during PRM. During MN, the trend in the feeding guild was similar to that of PRM in the cage sites with SSDF (20.49%) dominating followed by CVR (13.9%) and SF (2.5%). In PM, a change in the trend of distribution of feeding groups was observed. Though SDF dominated (55.91%), CVR (26.36%) formed the second dominant taxa followed by SF (10.3%) and SSDF (7.42%). In RF, the distribution pattern was slightly different from that of cage sites with the dominant feeding type varying from season to season. SDF dominated during PRM (49.8%) whereas SSDF (32.2%) and SF (36.4%) dominated during MN and PM respectively. During PRM, SDF formed the dominant feeding group followed by SF (27.5%) then SSDF (16.1%) and CVR (6.6%). During MN, the dominance of SDF and CVR were almost the same (26.6% and 25.2%) followed by SF (16.1%). Finally, during PM, SF was followed by SDF (27.3%) then by CVR (20.7%) and finally SSDF (15.7%).

The dominance of SDF in the cage sites indicates the increased availability of organic matter in the area such as faecal matter of fishes in the cage and the uneaten feed deposited. Apart from this, other sources of organic detritus include plankton, and suspended soft mud from rivers (Quasim and Sankaranarayanan, 1972). The dominance of SDF is considered a sign of a polluted environment (Hossain, 2018). Another indicator of a stressed and polluted environment is the predominance of SSDF, exhibiting varied abundance along seasons with the highest abundance during PRM and lowest during PM in the cage sites. During PM, CVR replaced SSDF as the second most abundant feeding guild in cage sites represented mainly by the polychaete species, *Namalycastis indicus*. The predominance of CVR in seasons other than monsoon is evidence of high carbon inflow (Rehitha *et al.*, 2019). The lower abundance of Suspension feeders (SF) in cage sites and their increased abundance in RF was an indication of a stressed environment in cage sites as these organisms are known to avoid environments with poor habitat quality (Jayachandran *et al.*, 2019).

The present results emphasise the necessity of monitoring the local biota, water quality parameters and sediment quality parameters of a region before, during, and after the installation of cages to ensure the sustainability of the environment and the farm. Since cage farming is effective in providing livelihood to the coastal population it is essential to ensure that there is sustainability without deteriorating the farm sites. Hence it is recommended that advisories on several cage farms that can be installed in open waters with instructions on the spacing of cages must be made available to farmers. Rules and regulations to prevent violations of such advisories should also be there so that the cage farmers give priority to estuarine health and its sustainability.

Acknowledgements

The authors thank the Director, CMFRI, Kochi for the constant encouragement. We are grateful to the staff and scholars of MBEM division of CMFRI for all the support rendered throughout the study period. The first author is thankful to UGC for providing financial assistance.

References

Afli, A., R. Ayari and S. Zaabi. 2008. Ecological quality of some Tunisian coast and lagoon locations, by using benthic community parameters and biotic indices. *Estua. Coast. Shelf. Sci*. 80 (2): 269-280.

Albayrak, S., H. Balkis, A. Zenetos, A. Kurun and C. Kubanc. 2006. Ecological quality status of coastal benthic ecosystems in the sea of Marmara. *Mar. Pollut. Bull*. 52: 790-799.

Ansari. Z. A., B. S. Ingole and S. A. H. Abidi. 2014. Organic enrichment and benthic fauna some ecological consideration, *Indian J. Geo-Mar. Sci*., 43 (4): 554-560.

APHA. 2005. Standard Methods for the Examination of Water and Wastewater. 21st Edition, American Public Health Association/American Water Works Association/ Water Environment Federation, Washington DC.

Baldock, J. A., J. M. Oades, A. G. Waters, X. Peng, A. M. Vassallo and M. A. Wilson. 1992. Aspects of the chemical structure of soil organic materials as revealed by solid-state ^{13}C-NMR spectroscopy. *Biogeochemistry,* 16: 1-42.

Beveridge, M. 1996. Cage Aquaculture. (2nd edition). Fishing News Book. 368 pp.

Birkett, L. and A. D. McIntyre. 1971. Treatment and sorting of samples methods for the study of marine benthos. IBP Handbook, p. 156-168.

Boyd, C. E., E. L. Torrans and C. S. Tucker. 2018. Dissolved oxygen and aeration in Ictalurid Catfish Aquaculture. *World Aqua. Soc*., 49 (1): 7-70.

Braatan, B. R. 2007. Cage aquaculture and environmental impacts. Chapter, Aquaculture engineering and environment. p. 49-92.

Clarke, K. R. and R. M. Warwick. 2001. Change in Marine communities: an approach to statistical analysis and interpreteation. 2nd edition PRIMER-E: Plymouth, p. 75-76.

Dauvin, J. C. and T. Ruellet. 2007. Polychaete/amphipod ratio revisited. *Mar. Pollut. Bull.,* 55: 215-224.

Day, J. H. 1967. A monograph on the Polychaeta of southern Africa, part I, Errantia and II, Sedentaria. Natural History London: Trustees of the British museum. 656 pp.

Fauchald, K. 1977. The polychaete worms: definitions and keys to the orders, families and genera. *Natural History Museum of Los Angeles County, Science Series*. 28: 1-188.

Fauchald, K. and P. A. Jumars. 1979. The diet of worms: a study of polychaete feeding guilds. *Annual Review–Oceanogr. Mar. Biol.,* 17:193-284.

Gosner, K. L. 1971. Guide to identification of marine and estuarine invertebrates. New York: Wiley-Interscience, John Wiley and Sons, Inc. *Mar. Biol*., 693 pp.

Grasshoff, K., M. Ehrhardt and K. Kremling. 1983. Methods of seawater analysis (2nd ed.). Deerfield Beach: Weinheim. 419 pp.

Gray, J. S., R. S. Wu and Y. Y. Or. 2002. Effects of hypoxia and organic enrichment on the coastal marine environment. *Mar. Ecol. Prog. Ser.,* 238: 249-279.

Hargrave, B. T., M. Holmer and C. P. Newcombe. 2008. Towards a classification of organic enrichment in marine sediments based on biogeochemical indicators. *Mar. Pollut. Bull*. 56: 810-824.

Holmer, M., D. Wildish and B. Hargrave. 2005. Organic enrichment from marine finfish aquaculture an`d effects on sediment processes. In B. T. Hargrave (Ed.), Water Pollution Environmental effects of marine finfish aquaculture. Springer. The Handbook of Environmental Chemistry. 5: 181-206.

Hossain, M. B. 2018. Trophic functioning of macrobenthic fauna in a tropical acidified Bornean estuary (Southeast Asia). *Int. J. Sediment Res*., 34: 48-57.

Jayachandran, P. R., S. B. Nandan, M. Jima, P. Joseph, N. D. Xavier, O. K. Sreedevi and K. J. Joseph. 2019. Macrobenthic functional feeding groups in a microtidal monsoonal estuary (Kodunganllur – Azheekode estuary, India). *Reg. Stud. Mar. Sci.,* 25: 100444.

Joseph, I. and A. Gopalakrishnan. 2017. Cage Farming Headed For Equal Opportunity In Aquaculture Development In Kerala, India. Gender in Aquaculture and Fisheries: Engendering Security in Fisheries and Aquaculture, Asian Fisheries Science Special Issue, 30S: 387-391.

John, S., K. Muraleedharan, C. Revichandran, S. A. Azeez, G. Seena and P. W. Cazenave. 2020. What Controls the Flushing Efficiency and Particle Transport Pathways in a Tropical Estuary? Cochin Estuary, Southwest Coast of India. *Water*, 12 (3): 908.

Jumars, P. A., K. M. Dorgan and S. M. Lindsay. 2015. Diet of worms emended: an update of polychaete feeding guilds. Annual Reviews in *Mar. Sci.*, 7: 497-520.

Joseph, I., S. Joseph, N. Aswathy and B. Bhaskaran. 2018. Impact of flashfloods on cage farms in Kerala. *Mar.Fish. Infor .Serv. T & E Ser.,* 237: 29-31.

Karakassis, M., E. Tsapakis, K. Hatziyanni, N. Papadopoulou and W. Plaiti. 2000. Impact of cage farming of fish on the seabed in three Mediterranean coastal areas, *ICES J. Mar. Sci*, 57: 1462–1471.

Kaya, D. and S. Pulatsu. 2017. Sediment-Focused Environmental Impact of Rainbow Trout (*Oncorhynchus* 2 mykiss Walbaum, 1792) Cage Farms: Almus Reservoir (Tokat), *Turk. J. Fish. Aquat. Sci.*, 17: 345-352.

Krull, E. S., J. A. Baldock and J. O. Skjemstad. 2003. Importance of Mechanisms and Processes of the Stabilisation of Soil Organic Matter for Modelling Carbon Turnover. *Funct. Plant Biol.*, 30: 207-222.

Krumbein, W. C. and F. J. Pettijohn. 1938. Manual of sedimentary petrography. Appleton– Century–Craft, Inc., New York, 549 pp.

Lima, L. S., Taciana K. Pinto, Bárbara de C. S. Brandão, Washington, Santos, Santiago, Hamilton., Ernesto C. Domingues, Ana, P. Klein., Carlos, A. Schettini, Luis, H. Poersch and Ronaldo, O. Cavalli. 2019. Impact of cage farming of cobia (Rachycentro canadum) on the benthic macrofauna in a tropical region, *Aquaculture*. 512: 734314.

Levin, L. A. 2003. Oxygen minimum zone benthos: adaptation and community response to hypoxia. Annual Review of *Ocenogr. Mar. Biol.*, 41:1-4.

Liao, Y., L. Shou, Z. Jiang, Y. Tang, P. Du, J. Zeng, Q. Chen, X. Yan and J. Chen. 2019. Effects of fish cage culture and suspended oyster culture on microbenthic communities in Xiangshan Bay, a semi-enclosed subtropical bay in eastern China. *Mar. Pollut. Bull.* 142: 475-483.

Mandario, M. A. E., V. R. Alava and N. C. Añasco. 2019. Evaluation of the bioremediation potential of mud polychaete *Marphysa* sp. in aquaculture pond sediments. *Environ. Sci. Pollut. Res.*, 26: 29810-29821.

Malathi, E., S. K. Sunder Raj and P. Mercy Bai. 2011. Larval development of *Marphysa gravelyi* (Polychaeta: Eunicidae) from Pulicat Lake, India. *Ital. J. Zool.*, 78 (1): 249-254.

Martin, G. D., P. A. Nisha, K. K. Balachandran, N. V. Madhu, M. Nair, P. Shaiju, T. Joseph, K. Srinivas and G. V. M. Gupta. 2011. Eutrophication induced changes in benthic community structure of a flow restricted tropical estuary (Cochin backwaters), India. *Environ. Monit. Assess.*, 176 (1-4): 427-438.

Menon, N. N., A. N. Balchand and N. R. Menon. 2000. Hydrobiology of Cochin backwater system–a review. *Hydrobiologia*, 430: 149-183.

Morata, T., S. Falco, I. Gadea, J. Sospedra and M. Rodilla. 2015. Environmental effects of a marine fish farm of gilthead seabream (Sparus aurata) in the NW Mediterranean Sea on water column and sediment. *Aquac. Res.*, 46: 59-74.

Nayar, S., D. J. Miller, A. Hunt, B. P. L. Goh and L. M. Chou. 2007. Environmental effects of dredging on sediment nutrients, carbon and granulometry in a tropical estuary. *Environ. Monit. Assess.*, 127: 1-13.

Nisha P. A., P. Sheeba, K. K. C. Nair and C. T. Achuthankutty. 2007. Life history and population dynamics of an estuarine amphipod, Eriopisa chilkensis Chilton (Gammaridae), *Estuar. Coast. Shelf Sci.*, 74: 87-95.

Pearson, T. H. and R. Rosenberg. 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Annual Review on *Oceanogr. Mar. Biol.,* 16: 229-311.

Pearson, T. H. and K. D. Black. 2000. The environmental impacts of marine fish cage culture. In: Black, K. D. (Ed.),Environmental impacts of aquaculture. Sheffield Academic Press, Sheffield, p. 1-31.

Prema, D., K. S. Sobhana, A. Lakshminarayana, Imelda Joseph, Shoji Joseph, Boby Ignatius, R. Jeyabaskaran, A. Nandakumar, I. R. Khambadkar, P. S. Anilkumar, G. Shylaja, and G. Syda Rao. 2010. Observations on selected characteristics of water and sediment at the open sea cage culture site of Asian seabass *Lates calcarifer* (Bloch) off Cochin, south-west coast of India. *Indian J. Fish.*, 57 (4): 53-59.

Price, C., K. D. Black, B. T. Hargrave, A. James and Jr. Morris. 2015. Marine cage culture and the environment: effects on water quality and primary production. *Aquacult. Environ. Interact.,* 6: 151-174.

Qasim, S. Z. and V. Sankaranarayanan. 1972. Organic detritus of a tropical estuary. *Mar. Biol.,* 15: 193-199.

Rajaneesh, K. M., S. Mitbavkar, A. C. Anil and S. S. Sawant. 2015. Synechoccus as an indicator of trophic status of Cochin backwaters, west coast of India. *Ecol. Indic.*, 55: 118-130.

Rao, D. S. and D. V. R. Sarma. 1983. Abundance and distribution of intertidal polychaetes in the Vasista Godavari estuary. *Mahasagar-Bull, natn, Inst. Oceanogr.*, 16 (3): 327-340.

Rehitha, T. V., N. V. Madhu, G. Vineetha, P. V. Vipindas, P. Resmi and C. Revichandran. 2019. Spatio-temporal variability in microbenthic communities and trophic structure of a tropical estuary and its adjacent coastal waters, *Environ. Monit. Assess.*, 191 (6): 341.

Santana, H. S., L. C. F. Silva, C. L. Pereira, J. Simiao-Ferreira and R. Angelini. 2015. The rainy season increases the abundance and richness of the aquatic insect community in a Neotropical reservoir. *Braz. J. Biol.*, 75 (1): 144-151.

Sara, G., D. Scilipoti., A. Mazzola and A. Modica. 2004. Effects of fish farming waste to sedimentary and particulate organic matter in a southern Mediterranean area (Gulf of Castellammare, Sicily): a multiple stable isotope study (d13C and d15N) *Aquaculture*, 234 (1-4): 199-213.

Snelgrove, P. V. R. 1998. The biodiversity of macrofaunal organisms in marine sediments. *Biodivers. Conserv.*, 7: 1123-1132.

Thangasamy, V., S. Veerasamy and S. Perumal. 2016. Population density of chironomus larvae (Diptera, Chironomidae) in selected Tuticorin salt pans, Southeast coast of India. *Int. J. Fish. Aqua. Stud.*, 4 (4): 434-439.

Tomassetti, P., P. Gennaro, L. Lattanzi, I. Mercatali, E. Persia, D. Vani and S. Porrello. 2016. Benthic community response to sediment organic enrichment by Mediterranean fish farms: case studies. *Aquaculture*, 450: 262-272.

Tomassetti, P. and S. Porrello. 2005. Polychaetes as indicators of marine fish farm organic enrichment. *Aquacult. Int.*, 13: 109-128.

SCOR-UNESCO. 1966. Determination of Photosynthetic Pigments in Sea Water. In: Monographs on Oceanographic methodology, 1. Paris, France, 368 pp.

Vineetha, G., V. Kripa, K. K. Karati, T. V. Rehitha, C. R. Vishal, V. Vineetha and M. Manu. 2020. Impact of a catastrophic flood on the heavy metal pollution status and the concurrent responses of the bentho-pelagic community in a tropical monsoonal estuary. *Mar. Poll. Bull.*, 155: 111191.

Vineetha, Valsalan, K. C., V. Kripa, Shelton Padua, R. Narayankumar and P. Vysakhan, 2020. Socio-economic analysis of finfish culture in cages in coastal waters of Kerala, south-west coast of India. *J. Mar. Biol. Ass. India*, 62 (1): 21-26.

Walkley, A. and I. A. Black. 1934. An examination of Degtjareff method for determining soil organic matter and a proposed modification of the chromic acid titration method. *Soil Science*, 37: 29-37.

Wilding, T. A. 2012. Changes in Sedimentary Redox Associated with Mussel (*Mytilus edulis* L.) Farms on the West-Coast of Scotland. *PLoS ONE*, 7 (9): e45159.

Wildish, D. J., J. E. Hughes-Clarke, G. W. Pohle, B. T. Hargrave and L. M. Mayer 2004. Acoustic detection of organic enrichment in sediments at a salmon farm is confirmed by independent ground truthing methods. *Mar. Ecol. Prog. Ser.* 267: 99-105.

WoRMS, 2020. World Register of Marine Species. Available from https://www.marinespecies.org Accessed 2020-01-20.

Wu, R. S. S. 1995. The environmental impact of marine fish culture: Towards a sustainable future *Mar. Pollut. Bull.,* 31 (4-12): 159-166.