



Impact of maintenance dredging on macrobenthic community structure of a tropical estuary



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ABSTRACT

This paper demonstrates the impact of maintenance dredging activities on the macrobenthic community structure of a tropical monsoonal estuary (Cochin estuary), located in the southwest coast of India for three consecutive years. The results of the study indicates apparent differences in benthic fauna and sediment characteristics between dredging and non-dredging sites, while most of the hydrographical parameters (temperature, pH, DO and BOD) exhibited inconspicuous variations. The dredging sites were characterized by significantly lower faunal density, biomass, and diversity and sustained distinct benthic faunal communities. The tubificid *Oligochaeta*, an opportunistic benthic taxon, was highly abundant in the dredging sites along with less density of Mollusca and Amphipoda. Prominent distinctions were evident in the feeding guilds of macrobenthic fauna between the dredging and non-dredging sites. The Benthic Opportunistic Annelida Amphipods Index (BO2A index), an index of benthic habitat quality showed relatively higher values (>0.24), which indicates the prevalence of poor environmental conditions in the dredging locations. The present study reveals the extent of impacts associated with maintenance dredging activities in a tropical estuary, which can be used to formulate effective management strategies for the protection of ecologically and economically significant benthic communities of estuarine ecosystems.

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1. Introduction

Estuaries, the transitional ecosystem between the marine and limnetic environments, are characterized by a highly dynamic and often unpredictable environmental scenario (Day, 1989). As they are endowed with rich bio resources, estuarine regions often form one of the most over exploited natural habitats on the Earth (Qasim, 2003). Being an ecologically significant region, a proper evaluation of the human associated changes in estuaries is of utmost importance for the healthy sustenance, and for the proper management of the bioresources they harbor. In order to abate the effects of siltation and to maintain navigable depth, regular dredging activities are carried out in the channels connecting estuaries to the sea. These recurrent dredging activities often have serious repercussions on the estuarine environment as they alter the bottom topography, sediment resuspension and composition, modifies the depth and current strength and also leads in the removal of a stable substratum (Jones et al., 2015; Newell et al., 1998, 2004).

Cochin estuary (CE), a tropical micro-tidal estuary, located in the southwest coast of India, running parallel to the Arabian Sea (AS), is remarkable for its rich biodiversity and productivity (Qasim, 2003). It is connected to the AS through two permanent inlets; one at Cochin (width 450 m) and the other at Azhikode (width 250 m). The estuary comes under the influence of the Indian Summer Monsoon (ISM), receiving heavy rainfall and associated runoff much larger than its volume during the wet monsoon season and hence is often categorized as a 'monsoonal estuary' (Vijith et al., 2009). Being shallow (2.5–15 m depth) the CE is often partially or completely mixed during the dry pre-monsoon season (Shivaprasad et al., 2013). The close proximity of the estuarine region to the bordering land and also the increased developmental activities along its coasts have adversely affected the estuarine ecology to a great extent (Gupta et al., 2009; Madhu et al., 2010a). Among the two inlets of the CE, the wider Cochin inlet forms the main navigational channel to the AS. Adjacent to the Cochin inlet, three channels (Fig. 1) are maintained for navigational purposes, i.e. the approach channel oriented along an east-west direction (~10 km length; 500 m width) and two inner channels (Balchand and Rasheed, 2000) located on either side of Willington Island,

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known as Ernakulum channel (~5 km length; 250–500 m width), and Mattancherry channel (~3 km length; 170–250 m width). The approach channel was constructed in 1928 by cutting a sand bar, situated 1.6 km west of the coast. As siltation often leads to a reduction in the depth of the channel, the materials silted up after the construction was removed by dredging (Gopinathan and Qasim, 1971). Since then, with the process of siltation, a synchronized siltation removal strategy through continuous dredging activity is being employed in these channels to ensure the depth for easy navigation.

Macrobenthos (>0.5 mm), ecologically significant faunal components of estuarine ecosystems, play a crucial role in the nutrient recycling, secondary production and pollutant metabolism, dispersion and burial (Snelgrove, 1998). Macrobenthic fauna inhabiting different substrata exhibits varied behavior and feeding modes to cope with their different functional needs (Forbes et al., 1994; Gutperlet et al., 2015; Kroncke, 2006), hence they are used as efficient indicators of physical disturbance such as dredging, which affects the sediment structure and composition (Taupp and Wetzel, 2013; Whomersly et al., 2008). As comprehensive knowledge on macrobenthic community structure gives a better insight on their responses to anthropogenic disturbances, it often becomes a prerequisite for evaluating the benthic community dynamics of a region (Berlow and Navarrete, 1997; Gutperlet et al., 2015). Recurring dredging activities often lead to substantial reduction in benthic standing crop and species diversity (Desprez, 2000; Guerra Garcia et al., 2003; Van Dalfsen et al., 2000). Although, studies on the impact of dredging activities on the benthic fauna is widely researched worldwide (Kaplan et al., 1975; Newell et al., 2004; Van Dolah et al., 1984), extensive studies providing detailed information on this aspect from tropical estuaries are scanty (Bemvenuti et al., 2005; Brown and Kumar, 1990; Ogbeibu et al., 2010). In CE, earlier studies on macrobenthic fauna mostly focused on their distribution and diversity (Devi et al., 1991; Martin et al., 2011;

Pillai, 1977; Kumar, 2002), but the impact of dredging on macrobenthic fauna have not been addressed comprehensively till date. In this context, the present study in the CE was designed to evaluate in detail whether the dredging activities carried out here (1) have any adverse impact on the water quality, sediment properties and community structure of macrobenthos (2) have any implications on the functional traits of the benthic community.

2. Material and methods

2.1. Study area

The CE is a semidiurnal micro-tidal estuary covering an area of ~25,600 ha along the south-west coast of India (Qasim, 2003). The estuary receives an annual freshwater influx of $22,000 \times 10^6 \text{ m}^3$ from two rivers via its northern limb and from five rivers via its southern limb (Revichandran et al., 2012; Srinivas et al., 2003). Annual precipitation in and around CE is about 320 cm and of which nearly 60–70% occurs during the south-west monsoon season (Qasim, 2003). Regular intrusion of sea water from the AS occurs through tidal intrusion (tidal range avg. 1 m), which gradually diminishes towards the head of the estuary (Martin et al., 2012). In the pretext of the construction of the Cochin port in 1936, an artificial island, (known as Willingdon Island), was created around a small pre-existing islet, using the dredged soil. After the construction of the port regular dredging activity is being carried out in and around the navigation channels to prevent shallowing of the estuary due to the increased siltation process. In the earlier years (during 1990s), intermittent dredging was carried out throughout the year (except monsoon) with a dredged volume ranging from 3.58 to 3.89 million cubic meters (Rasheed, 1997). At present, continuous dredging activities are being carried out in these channels throughout the year including the monsoon and an average of 13.38 million cubic meters of dredged materials are

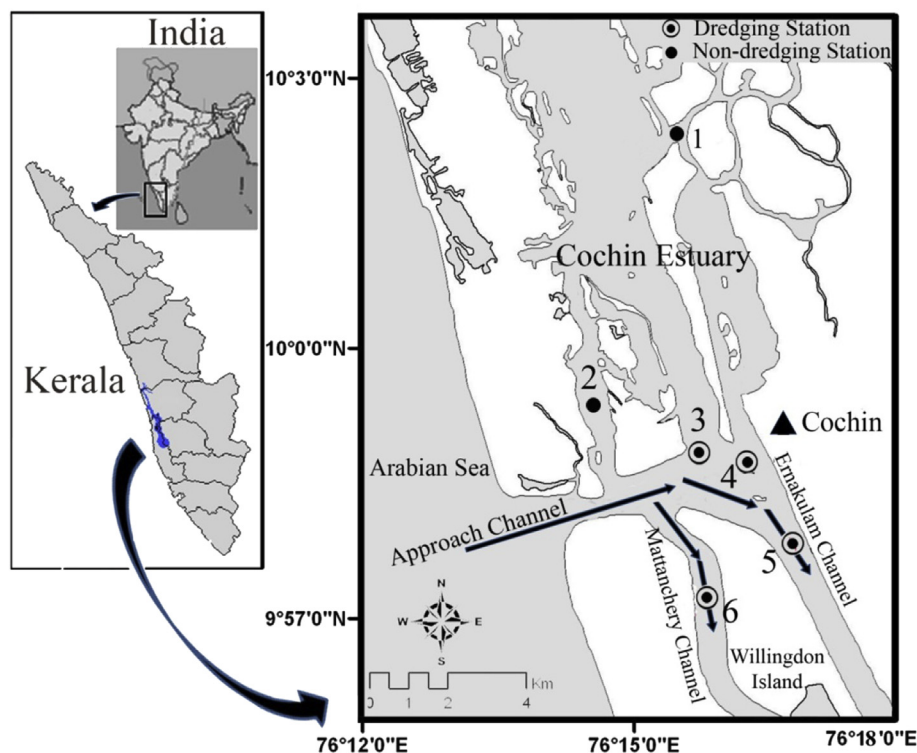


Fig. 1. Map of the Cochin estuary showing sampling locations.

being removed annually (Source, Cochin Port Trust, Kochi 2006–2012) thus maintaining a depth of 10–13 m for all the three channels (Menon et al., 2000).

2.2. Sampling strategy and laboratory procedures

In order to understand the impact of dredging on macrobenthic community of the CE, sampling was carried out at 6 locations for three consecutive years from 2009 to 2011. Two sampling locations positioned adjacent to the dredging sites, without any dredging activities were selected as the reference non-dredging sites (ND-stations 1&2). Sampling locations (D-stations 3 to 6), situated inside the channels, where continuous dredging activities were carried out were designated as dredging sites (Fig. 1). Since the CE is a tropical monsoonal estuary highly influenced by the Indian Summer Monsoon (ISM), sampling periods were categorized based on the rainfall and associated runoff. Pre-monsoon (PRM; February–May) period was characterized by little or no rainfall, monsoon (MN; June–September) was the period of heavy rainfall and post-monsoon (PM; October–January), was the retrieving phase of monsoon with relatively less rainfall (Qasim, 2003). A Global Positioning System (GPS) was used to locate the sampling stations. A conductivity-temperature-depth profiler (CTD 19 plus, Sea-Bird Electronics) was deployed at each station to obtain temperature profiles. A Niskin sampler (5 L capacity, Hydro-Bios, Kiel-Holtenau, Germany) was used to collect bottom water samples for the analysis of hydrographic variables, such as salinity, pH, dissolved oxygen (DO), biological oxygen demand (BOD) and suspended particulate matter (SPM). Salinity was determined using a Digi Auto Salinometer (Model TSK, accuracy ± 0.001) immediately after reaching the laboratory. pH was measured using a pH meter (ELICO LI610, accuracy ± 0.01). Samples for DO and BOD were collected carefully in glass bottles without trapping air bubbles. DO samples were fixed immediately onboard using 0.5 ml of Winkler A (3 M Manganous chloride) and 0.5 ml of Winkler B (8 M alkaline iodide). For BOD estimation, samples were fixed after 5 days of incubation and were later analyzed according to Winkler's method (Grasshoff et al., 1983). SPM was determined gravimetrically on Millipore membrane filters (nominal pore size, 0.45 μm) after drying at 70 °C for 6–8 h to reduce water content before weighing (APHA, 2005). For the determination of sediment characteristics, the collected sediment samples were dried in a hot air oven at 60 °C and subjected to textural analysis (Krumbein and Pettijohn, 1938). The organic carbon content of the sediment was estimated by wet oxidation method (El Wakeel and Riley, 1957).

Sediment samples were collected in duplicates using a Van-Veen Grab (area of 0.05 m²). For the macrobenthic community analysis, sediments collected were sieved onboard through a 0.5-mm test sieve (Birkett and McIntyre, 1971) and the organisms collected in the sieve were preserved in 5% (buffered) formalin-Rose Bengal mixture. All the organisms retained in the sieve were examined under a binocular stereozoom microscope (CATSCOPE CS-S 6080), and sorted for major macrobenthic taxa, for further analysis (e.g., polychaetes, molluscs, crustaceans etc.). The detailed identification of the macrobenthic fauna to the lowest possible taxonomic levels were carried out using standard identification manuals (Day, 1967; Fauchald, 1977; Fauvel, 1953; Gosner, 1971; Nayar, 1959) after the estimation of numerical density (ind m⁻²), and biomass (wet weight-g m⁻²). The feeding guilds of the dominant group, polychaetes, as well as the characterizing species of dredging and non-dredging sites (identified through SIMPER analysis-PRIMER 6.1.5) were identified using published literature (Aravind et al., 2007; Caine, 1977; Fauchald and Jumars, 1979; Gosner, 1971; Imrie et al., 1990; Leal and Matthews, 2013; Mondal et al., 2010) in order to get an overall idea about community

structure.

2.3. Statistical analysis

Significance of spatial variation in the biotic and abiotic parameters between the dredging and non-dredging sites was tested using an unpaired *t*-test (the Graph Pad Prism version 5.01). The significance of seasonal variation in the abiotic and biotic parameters was determined by one way analysis of variance (ANOVA). Before the analysis, the D'Agostino and Pearson omnibus normality test was carried out to check their normality in distribution, and based on the result, parametric or non-parametric ANOVA was performed for the variables. Univariate indices such as Shannon-Wiener index, (H' ; \log_2) for species diversity, and Margalef's richness (d) for species richness were carried out using the PRIMER (version 6.1.5); (Clarke and Warwick, 2001). To identify the different macrobenthic assemblages, multivariate analyses based on Bray–Curtis similarity index and group average linkage were carried out on fourth root transformed density data using which hierarchical cluster and non-metric multidimensional scaling (NMDS) were plotted. Based on the results of the cluster, the species having greatest contribution towards the grouping were determined using similarity percentage tool SIMPER. The significance of spatio-temporal variations in the macrobenthic density was tested using Analysis of Similarity (ANOSIM in PRIMER6) which define the differences between the sampling sites based on the macrobenthic density (Clarke and Gorley, 2006).

2.4. Benthic Opportunistic Annelida Amphipods index (BO2A)

Benthic Opportunistic Annelida Amphipods index (BO2A) was used to determine the environmental status of the study area.

$$BO2A = \log \left(\frac{foa}{fsa + 1} + 1 \right)$$

where *foa* is the opportunistic annelida (Clitellata and Polychaeta) frequency (i.e., the ratio of the total number of opportunistic annelid individuals to the total number of individuals in the samples containing ≥ 20 individuals), *fsa* is the amphipod frequency (i.e., the ratio of the total number of sensitive amphipod individuals, excluding the opportunistic Jassa amphipods, to the total number of individuals in the sample). The index values ranging between 0.15 and 0.24, refers to the moderate condition of pollution of the water body, and greater than 0.24 indicates the poor environmental condition (Dauvin and Ruellet, 2009).

3. Results

3.1. Hydrographical parameters

Hydrographical parameters such as bottom water temperature (D-25 to 33.5 °C, ND-26.2 to 33 °C), salinity (D & ND-0 to 35) and pH (D-6.5 to 8.7, ND-6.4 to 8.8) exhibited a prominent seasonal variation ($p < 0.05$) throughout the study period (Tables 1 and 2). However, except salinity and SPM most of the other hydrographic variables did not exhibit significant variation ($p > 0.05$) between the dredging and non-dredging sites (Table 3). Relatively high salinity was observed in dredging sites compared to non-dredging sites (Table 1). Though mean DO concentration was low in dredging sites (D-0.6–8.04 mg/l) relative to non-dredging sites (ND-0.2–7.4 mg/l), it was not statistically significant (Tables 1 and 3). The concentration of suspended particulate matter (SPM) was relatively higher in dredging sites (D-2 to 864 mg/l) as compared to the non-dredging sites (ND-2.8–97.6 mg/l) with statistically

Table 1
Distribution of the physico-chemical and biological variables (av.) in the Cochin estuary during 2009–2011.

Parameter	Temperature (°C)	Salinity	pH	DO (mg/l)	BOD (mg/l)	SPM (mg/l)	Sand (%)	Silt (%)	Clay (%)	Organic carbon (mg/g)	
2009 PRM	D	29.8 ± 1.6	27.2 ± 7.0	7.9 ± 0.1	3.1 ± 0.8	2.2 ± 0.9	163.8 ± 219	11.2 ± 12.2	36.1 ± 16.2	52.8 ± 13	25.5 ± 4.6
	ND	30.2 ± 1.5	19.3 ± 10.1	7.6 ± 0.3	3.1 ± 0.9	2.4 ± 0.9	43.9 ± 18.5	56.3 ± 22.1	15.9 ± 8.8	27.8 ± 20.7	14.1 ± 6.2
2010 PRM	D	32.3 ± 0.7	28.5 ± 6.1	7.7 ± 0.5	2.8 ± 0.3	1.6 ± 1.2	84.9 ± 64.7	23.6 ± 19.8	22.1 ± 26.6	54.5 ± 23.1	15.8 ± 7.8
	ND	32.6 ± 0.5	21.1 ± 11.4	7.9 ± 0.8	3.2 ± 0.4	1.6 ± 1	71.5 ± 19.8	32.1 ± 18.5	9.7 ± 15.7	58.1 ± 9.4	17.6 ± 4.1
2011 PRM	D	30.3 ± 0.9	16.0 ± 9.9	7.5 ± 0.5	5.0 ± 1.6	2.4 ± 0.9	40.6 ± 25.4	15.8 ± 24.1	32.5 ± 10.7	49.7 ± 18.3	19.5 ± 8.1
	ND	30.6 ± 0.7	11.2 ± 9.1	7.5 ± 0.5	5.7 ± 1.1	2.9 ± 1.3	26.7 ± 13.3	57.5 ± 27.7	17.9 ± 18.8	30.1 ± 22.4	12.2 ± 4.1
2009 MN	D	27.2 ± 1.9	17.9 ± 12.8	7.4 ± 0.5	2.5 ± 1.5	0.9 ± 0.6	72.8 ± 78.7	10.7 ± 9.9	27.4 ± 10.5	61.9 ± 9.1	24.5 ± 3.7
	ND	28.5 ± 1.3	3.8 ± 2.8	6.9 ± 0.4	3.7 ± 0.4	0.6 ± 0.3	17.1 ± 7.1	62.4 ± 28.3	17.0 ± 25.5	20.6 ± 7	15.3 ± 11.6
2010 MN	D	28.8 ± 1.1	9.5 ± 10.9	7.4 ± 0.3	3.9 ± 1.2	1.01 ± 0.6	35.4 ± 33.5	24.0 ± 17.0	19.0 ± 13.1	56.9 ± 18.6	26.8 ± 5.5
	ND	28.8 ± 1.2	1.7 ± 2.3	7.2 ± 0.4	4.2 ± 0.9	1.00 ± 0.7	21.2 ± 17.4	43.4 ± 13.4	18.1 ± 7.02	38.5 ± 8.5	15.7 ± 8.3
2011 MN	D	28.4 ± 1.5	7.1 ± 7.8	7.6 ± 0.3	4.7 ± 1.6	1.8 ± 1.1	41.5 ± 34.8	18.2 ± 16.6	33.7 ± 15.9	48.1 ± 20.1	14.5 ± 5.7
	ND	28.7 ± 1.9	2.9 ± 3.3	7.4 ± 0.4	5.0 ± 0.7	1.5 ± 0.5	22.5 ± 16.1	42.1 ± 26.6	17.7 ± 15.4	40.2 ± 22.5	11.1 ± 4.9
2009 PM	D	29.1 ± 0.8	21.9 ± 12.8	7.6 ± 0.5	2.9 ± 0.8	1.0 ± 0.5	70.8 ± 47.5	16.6 ± 26	37.5 ± 10.9	46.1 ± 19.6	26.7 ± 10.1
	ND	29.0 ± 1.1	11.3 ± 9.7	7.2 ± 0.7	3.3 ± 0.7	1.0 ± 0.8	24.8 ± 4.2	52.7 ± 28.3	25.1 ± 26.5	19.2 ± 9.4	15.9 ± 11.3
2010 PM	D	29.3 ± 1	16.4 ± 9.9	7.5 ± 0.5	4.6 ± 0.9	1.2 ± 0.5	38.1 ± 18.5	31.8 ± 21.4	17.5 ± 13.8	50.6 ± 19.9	26.3 ± 7.2
	ND	28.5 ± 1.3	9.1 ± 8.8	7.1 ± 0.5	4.4 ± 0.7	1.0 ± 0.4	19.9 ± 9.3	41.7 ± 19.3	22.5 ± 25.3	37.6 ± 17.5	16.3 ± 7.6
2011 PM	D	29.1 ± 1.1	18.1 ± 8.4	7.9 ± 0.3	4.5 ± 1.1	2.1 ± 0.4	51.5 ± 30.9	14.1 ± 12.3	36.6 ± 18.8	49.3 ± 0.9	20.9 ± 6.8
	ND	29.0 ± 0.9	13.2 ± 7.5	7.7 ± 0.4	4.1 ± 2.2	2.2 ± 0.3	30.3 ± 12.5	57.2 ± 25.2	13.1 ± 11.2	29.8 ± 16.7	13.6 ± 7.5

Table 2
Results of One way ANOVA of major environmental parameters in the dredging and non-dredging sites between seasons (* - $p < 0.05$, ** - $p < 0.01$).

Parameter	2009		2010		2011	
	D	ND	D	ND	D	ND
Temperature	0.001**	0.16	3.85	0.002**	0.0002**	0.02*
Salinity	0.08	0.04*	0.0003**	0.002**	0.003**	0.02*
pH	0.04*	0.04*	0.14	0.04*	0.02*	0.5
DO	0.39	0.56	0.001**	0.24	0.66	0.26
BOD	0.46	0.01*	0.17	0.30	0.21	0.01*
SPM	0.22	0.01*	0.01*	6.03	0.46	0.55
Sand	0.79	0.84	0.61	0.57	0.90	0.24
Silt	0.24	0.67	0.84	0.55	0.86	0.58
Clay	0.08	0.55	0.74	0.65	0.98	0.57
Organic carbon	0.76	0.93	0.002**	0.92	0.06	0.85
Density	0.25	0.28	0.30	0.54	0.84	0.90
Biomass	0.33	0.57	0.77	0.51	0.67	0.46

Table 3
P value of non parametric *t*-test between dredging and non-dredging sites (*- $p < 0.05$, **- $p < 0.01$).

Parameter	Year		
	2009	2010	2011
Temperature	0.31	0.02*	0.49
Salinity	0.003**	0.02*	0.05
pH	0.03*	0.19	0.18
DO	0.08	0.88	0.59
BOD	0.91	0.70	0.75
SPM	0.01*	0.12	0.01*
Sand	<0.0001**	0.02*	<0.0001**
Silt	0.002**	0.76	0.0001**
Clay	<0.0001**	0.05	0.002**
Organic carbon	<0.0001**	0.002**	0.001**
Density	0.001**	0.04*	0.21
Biomass	0.03*	0.17	0.001**

significant disparity (Tables 1 and 3).

3.2. Sediment characteristics

Bottom substratum at the dredging sites were mainly consisted of finer fractions of sediment i.e., clay (3.5–96.04%) and silt (0.62–85.5%), while the non-dredging sites were characterized with coarser particles especially sand (0.59–88.2%) (Table 1). Significant variation ($p < 0.05$) in sediment texture (sand, silt, and clay)

was observed between dredging and non-dredging sites (Table 3), whereas seasonal variation was insignificant ($p > 0.05$) (Table 2). Irrespective of seasons, the sediment organic carbon was significantly ($p < 0.05$) higher in dredging sites compared to the non-dredging sites (Tables 1 and 3).

3.3. Macrobenthic density and biomass

During the study, macrobenthic density varied from 6 to 12,600 ind. m^{-2} in dredging sites and from 101 to 28,140 ind. m^{-2} in the non-dredging sites (Fig. 2A). In dredging sites, the mean macrobenthic density was lower throughout the study period (PRM-av. 1325 ± 1566 , MN-av. 1844 ± 2432 , PM-av. 1678 ± 1678 ind. m^{-2}) compared to non-dredging sites (PRM-av. 4374 ± 5261 , MN-av.

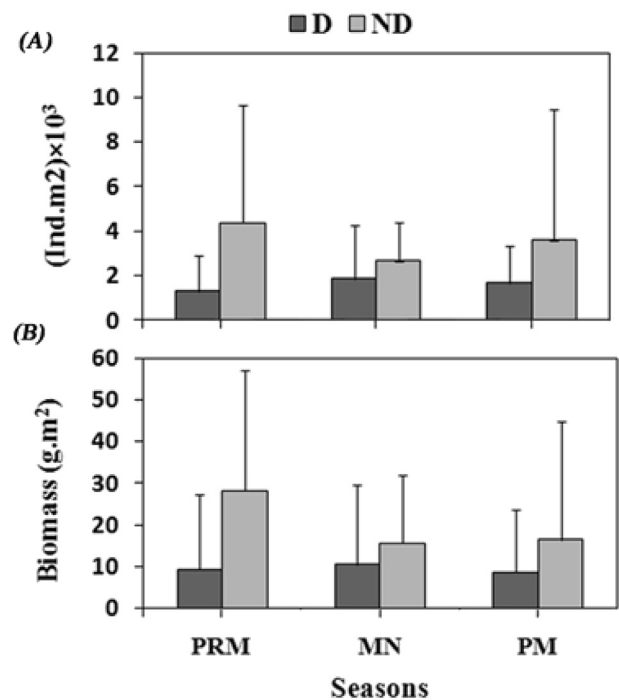


Fig. 2. Seasonal distribution of macrobenthic density (A) and biomass (B) in dredging (D) and non-dredging (ND) sites of the Cochin estuary (PRM-pre-monsoon; MN-monsoon; PM-post-monsoon).

2678 ± 1702, PM-av. 3619 ± 5849 ind m⁻²) (Fig. 2A). Macrobenthic density exhibited no significant seasonal patterns in the study area. Similar to density, macrobenthic biomass also exhibited similar trend with lower biomass in the dredging sites (PRM-av.9.36 ± 18.0, MN-av. 10.45 ± 19.0, PM-av. 8.78 ± 14.8 gm⁻²) compared to non-dredging sites (PRM-av. 28.32 ± 28.8, MN-av. 15.74 ± 16.2, PM-av. 16.52 ± 28.5 gm⁻²) (Fig. 2B). The variation in macrobenthic density and biomass between the dredging and non-dredging sites was statistically significant ($P < 0.05$) (Table 3).

3.4. Composition of macrobenthic groups

During the study, a total of 81 macrobenthic taxa belonging to 6 phyla were encountered. Among the different taxa observed, tubificid oligochaetes (av.7600 ind. m⁻², 48%) dominated along with polychaetes (av. 6053 ind. m⁻², 38%) in dredging sites, whereas in the non-dredging sites, polychaetes (av. 9573 ind. m⁻², 56%) and amphipods (av. 5407 ind. m⁻², 32%) formed the predominant taxa (Fig. 3). Tanaids (9%), amphipods (2.3%), and gastropods (0.42%) were the other groups observed in dredging sites while gastropods (2.3%), oligochaetes (1.6%), bivalves (1.4%), and tanaids (1.3%) constituted other groups in the non-dredging sites (Fig. 3).

3.5. Polychaete community composition

Among macrobenthic fauna, polychaetes belonging to the family Capitellidae, Spionidae, Nereidae, Eunicidae and Owenidae were dominant in the non-dredging sites, whereas Capitellidae, Nephtyidae, Pilargidae, Spionidae, and Cirratulidae were the major families observed in dredging sites. *Mediomastus capensis* (13–30%) dominated in non-dredging sites throughout the study period irrespective of seasons. While in dredging sites *Mediomastus capensis* (10%), *Sigambra parva* (7%), *Nephtys oligobranchia* (7%) and *Cossura coasta* (6%) were the predominant species observed during PRM. *Mediomastus capensis* (5%), *Nephtys oligobranchia* (4%), *Priospio cirrifera* (3%) and *Cirratulus filiformis* (3%) were observed during MN. Species such as *Sigambra parva* (8%), *Mediomastus capensis* (7.5%), and *Cirratulus filiformis* (7%) were predominated in dredging sites during PM period (Supplementary Table 1). The species diversity index (H') was relatively higher in non-dredging sites (PRM-av.3.2 ± 0.5, MN-av.3.1 ± 0.4, PM-av.2.8 ± 0.3) (Fig. 4A) compared to the dredging sites (PRM-2.5 ± 0.4, MN-av. 2.2 ± 0.6, PM-av.2.4 ± 0.4) irrespective of seasons. The trend was similar for the species richness index (d) also (Fig. 4B).

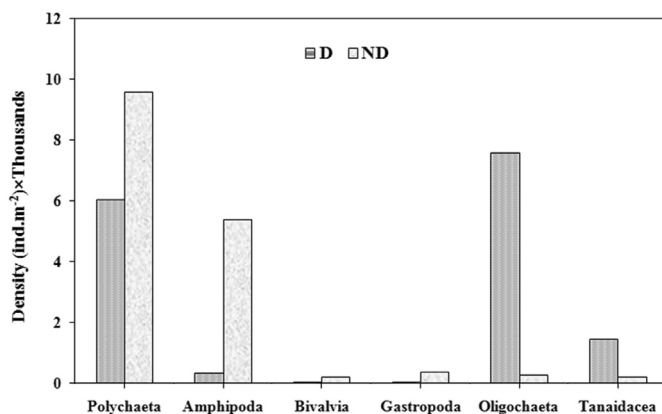


Fig. 3. Macrobenthic community composition in dredging (D) and non-dredging sites (ND) of the Cochin estuary.

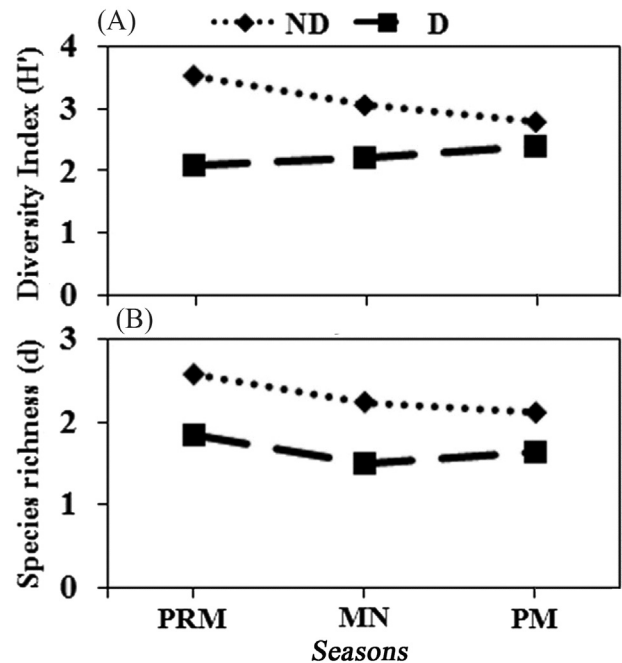


Fig. 4. Seasonal variation in (A) species diversity (H') and (B) species richness (d) in the dredging and non-dredging sites of the Cochin estuary (PRM-pre-monsoon, MN-monsoon, PM-post-monsoon).

The hierarchical clustering of sampling stations based on the fourth-root transformed macrobenthic density of the three seasons, revealed nine groups at 40% similarity level (Fig. 5A). Of these, two were formed by non-dredging sites whereas the other seven clusters were constituted by dredging sites. The results of the NMDS plot further affirmed the distinctness between the dredging and non-dredging sites (Fig. 5B).

Significant distinction in macrobenthic density between dredging and non-dredging sites were evident from the results of ANOSIM analysis (Global R-0.588, $p < 0.01$). The results of the SIMPER analysis, carried out to identify the characterizing and discriminating species of the dredging and non dredging sites, are depicted in Tables 4 and 5. Density of the major discriminating species overlaid on the nMDS plot clearly shows the distinctions between the dredging and non-dredging sites (Fig. 6).

3.6. Feeding guild composition

Assessment of feeding guilds of the dominant group, polychaetes revealed the dominance of carnivores (av.37.8%) in dredging sites, and sub-surface deposit feeders (SSDF-av.43.3%) in non-dredging sites (Fig. 7A–C). Seasonal variation in feeding guild composition was less in dredging sites, except during MN where carnivorous polychaetes (PRM-39.3%, MN-31.3%, PM-42.7%) were replaced by surface deposit feeders (SDF -36%). In the non-dredging sites, seasonal variation in feeding guild was obvious where SSDF (54.6%), dominated during PRM, replaced by SDF (50.7%) during MN and carnivores (38.7%) during PM period (Fig. 7A–C).

Feeding guild analysis of the characterizing species indicated the existence of four feeding modes in dredging sites and seven feeding modes in non-dredging sites (Fig. 7D–F). In dredging sites, SDF and carnivores (67–87% together) formed the predominant feeding modes whereas SDF and SSDF (62–91% together) constituted the major feeding types in non-dredging sites (Fig. 7D–F).

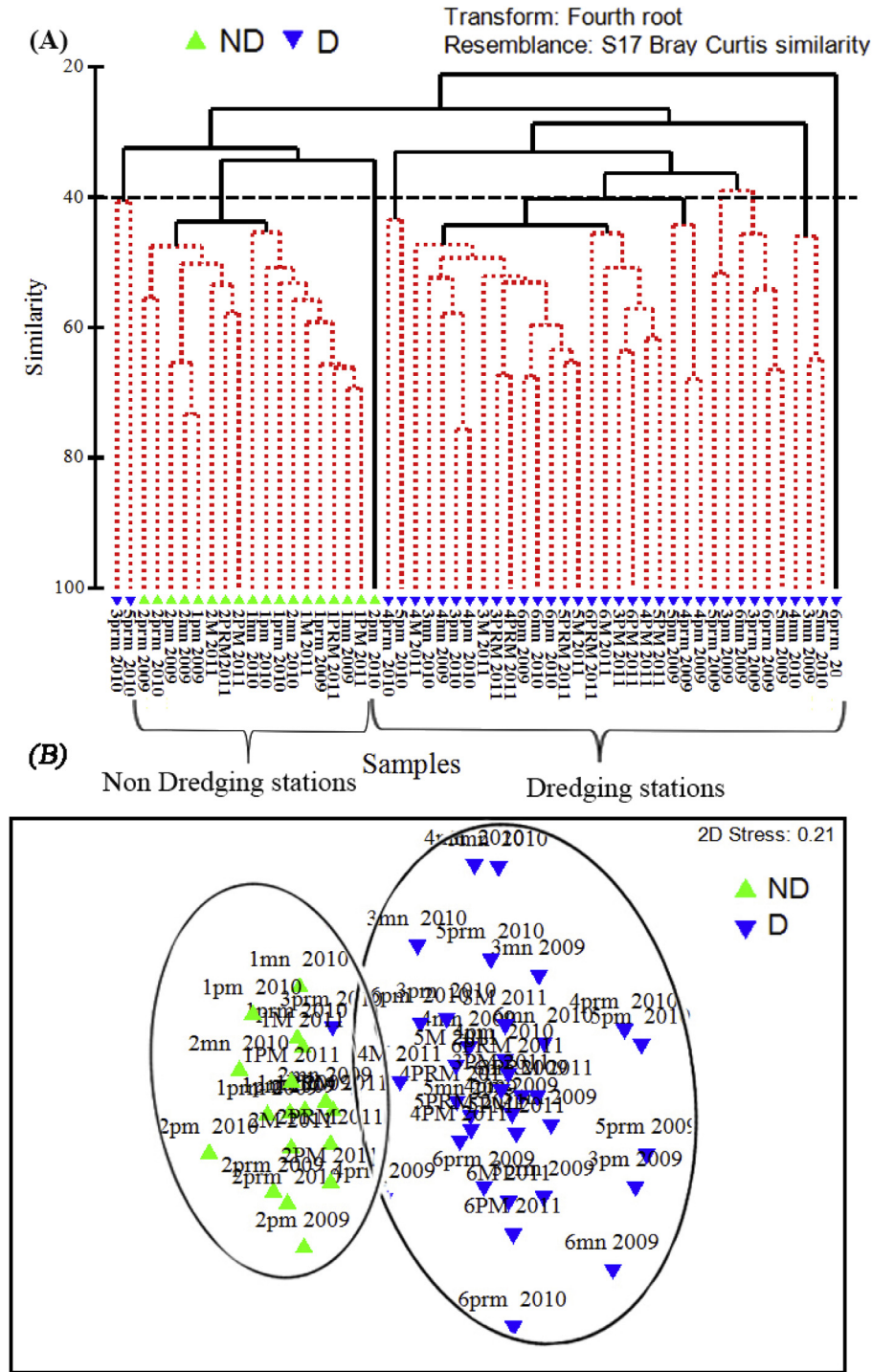


Fig. 5. Bray-curtis similarity based on hierarchical clustering of sampling stations depicted through (A) dendrogram and (B) NMDS (non-metric multidimensional scaling) showing macrobenthic assemblage pattern in dredging and non-dredging sites of the Cochin estuary.

3.7. BO2A index

The BO2A index was relatively higher in dredging sites (PRM-av.0.18 ± 0.08, MN-av. 0.22 ± 0.05, PM-0.20 ± 0.05), compared to non-dredging sites (PRM-av.0.14 ± 0.08, MN-0.17 ± 0.04, PM-0.10 ± 0.05). The index values of non-dredging sites indicated good to moderate condition while those in the dredging sites ranged from moderate or poor condition (Fig. 8). Among all the sampling locations, station 3 and station 5 (dredging sites) exhibited high

BO2A index and the values even reached >0.24 indicating the poor environmental conditions at these sites (Fig. 8).

4. Discussion

In estuaries, the increasing disturbances raised in the benthic habitats concurrent to the anthropogenic interventions have always been a matter of concern (Cooper, 2003). In the present study, a detailed monitoring of the distribution and community structure

Table 4

Characterizing species contributing to average similarity within sample groups identified through SIMPER.

Assemblages	Species	Av. Abundance	Av. Similarity	% Contribution
Sim: 46.54% Group I-ND	<i>Mediomastus capensis</i>	4.04	6.3	13.53
	<i>Prionospio cirrifera</i>	3.24	4.57	9.83
	<i>Paraheteromastus tenuis</i>	2.96	4.43	9.52
	<i>Caprella</i> sp	2.92	4.27	9.19
	<i>Dendronereis estuarina</i>	2.51	3.62	7.77
	<i>Littorina littorea</i>	1.91	2.44	5.25
	<i>Photis digitata</i>	1.91	2.26	4.85
	<i>Tubificidae</i> sp	1.72	1.77	3.81
	<i>Bivalvia</i> sp	1.67	1.71	3.68
	<i>Capitella capitata</i>	1.54	1.66	3.57
	<i>Eriopisa chilensis</i>	1.67	1.53	3.29
	<i>Apseudus chilensis</i>	1.55	1.5	3.21
	<i>Diopatra neapolitana</i>	2.1	1.37	2.95
	<i>Cheriphotis megacheles</i>	1.25	1	2.14
Sim: 37.64% Group II-D	<i>Coropium triaenonyx</i>	1.25	0.81	1.75
	<i>Owenia fusiformis</i>	1.11	0.69	1.49
	<i>Tubificidae</i> sp	3.47	6.57	17.46
	<i>Mediomastus capensis</i>	2.46	5.43	14.43
	<i>Sygambra parva</i>	2.39	5.09	13.52
	<i>Nephtys oligobranchia</i>	2.09	4.34	11.54
	<i>Prionospio cirrifera</i>	1.86	3.77	10
	<i>Apseudus chilensis</i>	1.93	3.04	8.09
	<i>Cirratulus filiformis</i>	1.56	2.29	6.08
	<i>Paraheteromastus tenuis</i>	1	0.87	2.3
	<i>Cossura coasta</i>	0.9	0.75	2
	<i>Cirratulus cirratus</i>	0.81	0.74	1.98
	<i>Decapod</i> sp	0.67	0.65	1.74
	<i>Bivalvia</i> sp	0.69	0.55	1.47

Table 5

Discriminating species contributing to average dissimilarity between dredging and non-dredging sites identified through SIMPER.

Average dissimilarity:74.01%	Group ND	Group D	Av.Diss	Diss/SD
Species	Av.Abund	Av.Abund		
<i>Caprella</i> sp	2.92	0	3.56	2.13
<i>Tubificidae</i> sp	1.72	3.47	3.27	1.25
<i>Paraheteromastus tenuis</i>	2.96	1	3.02	1.33
<i>Dendronereis estuarina</i>	2.51	0.39	2.77	1.67
<i>Mediomastus capensis</i>	4.04	2.46	2.66	1.26
<i>Sigambra parva</i>	0.55	2.39	2.6	1.43
<i>Prionospio cirrifera</i>	3.24	1.86	2.55	1.26
<i>Photis digitata</i>	1.91	0.2	2.54	1.12
<i>Nephtys oligobranchia</i>	0.12	2.09	2.51	1.44
<i>Diopatra neapolitana</i>	2.1	0.46	2.5	0.98
<i>Littorina littorea</i>	1.91	0.12	2.3	1.46
<i>Apseudus chilensis</i>	1.55	1.93	2.18	1.09
<i>Eriopisa chilensis</i>	1.67	0.42	2.01	1.07
<i>Cirratulus filiformis</i>	0.22	1.56	1.91	1
<i>Bivalvia</i> sp	1.67	0.69	1.85	1.19
<i>Capitella capitata</i>	1.54	0.58	1.84	1.1
<i>Cheriphotis megacheles</i>	1.25	0	1.57	0.9
<i>Cirratulus cirratus</i>	0	0.81	0.99	0.69

of the macrobenthic biota in a tropical monsoonal estuary was carried out for three consecutive years to understand the responses of the benthic biota to the incessant dredging activity performed to maintain the depth of the navigation channel.

The observed temporal variations in the major hydrographical variables were mostly driven by the heavy rainfall associated with the Indian summer monsoon and associated river discharges (Madhu et al., 2010b; Qasim, 2003). Higher salinity observed in the dredging sites was found to be associated with the intrusion of seawater through the dredged channels. A noticeable feature observed in the present study was the lack of significant variations in the hydrographical variables like temperature, pH, DO and BOD between the dredging and non-dredging sites, compared to the

variation observed in the SPM, sediment characteristics and the benthic biota. The churning up of bottom substratum as a consequence of the dredging activities, thus bringing up finer sediment particles into the water column might have contributed towards the higher SPM concentration in the dredging sites. Studies depicting the effect of dredging on the increased turbidity in the water column of tropical (Balchand and Rasheed, 2000; Johnston, 1981), sub-tropical (Hossain et al., 2004; Yeager et al., 2010) and temperate (de Jonge, 1983; de Jonge et al., 2014) water bodies further substantiates the observation.

In estuaries, the continuous removal of the substratum, brought about by intense dredging activities often lead to drastic changes in the bottom topography as well as the sediment composition (Junior et al., 2012). The present study evidenced a marked dominance of finer fractions of sediment in the dredging sites compared to the coarser particles in the non-dredging sites. Dredging of channels often leads to a modification of the bottom topography and increases the depth, resulting in lowering of the current velocities, thereby favoring the deposition of fine sediment particles (Desprez, 2000; Kaplan et al., 1975; Van der Veer et al., 1985). Newell et al. (1998) pointed out the dominance of finer sediments, dissolved particulate matter, strong current flows, and sediment bound contaminants associated with the dredging activities in estuaries. A more or less homogeneously sandy substratum was observed in the non-dredging sites. The substratum at the non-dredging sites consisted of high percentage of sand, throughout the year. This may be the consequence of combined sediment supply from the perennial river discharges and also from the sediment transport through tidal incursion in the region. Likewise, the report (Desprez, 2000) on marine dredging along the French coast of English Channel also corroborate the present findings as the non-dredged areas of the English channel was predominated by gravels and coarse sand, whereas the dredging sites were characterized by very fine sand. Further, the significant variation ($p < 0.05$) evident in the sand, silt and clay fractions between the dredging and the non-dredging sites further authenticates the observation in the

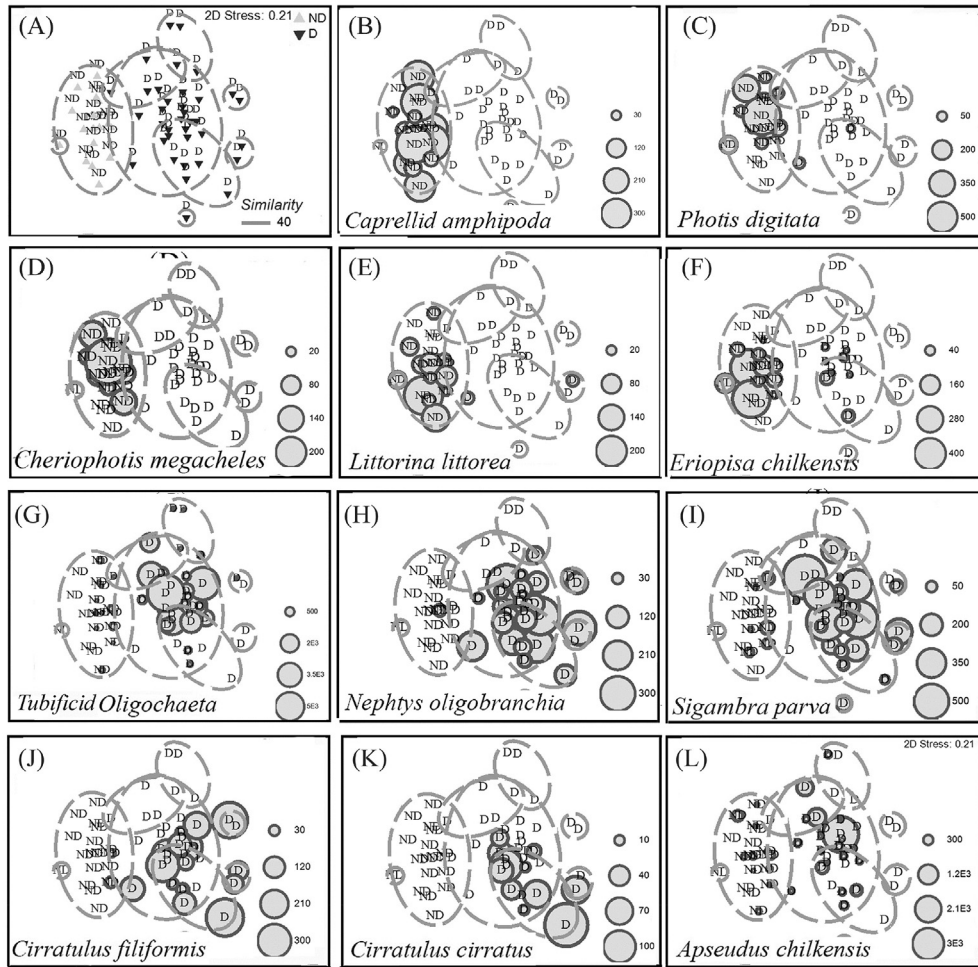


Fig. 6. Species responsible for the distinct difference between dredging and non-dredging areas identified using SIMPER analysis (A) nMDS plot of faunal distribution of macrobenthos, overlaid with bubbles indicating density of major discriminating species (B) Caprellid amphipod, (C) *Photis digitata*, (D) *Cheriophotis megacheles*, (E) *Littorina littorea*, (F) *Eriopisa chilkenis*, (G) *Tubificid Oligochaeta*, (H) *Nephtys oligobranchia*, (I) *Sigambra parva*, (J) *Cirratulus filiformis* (K) *Cirratulus cirratus* and (L) *Apseudus chilkenis*.

present study (Table 4).

The distribution of organic carbon is mainly associated with the type of sediment at the study site (Nayar et al., 2007; Nguyen et al., 1997) and finer fractions of sediment have greater surface area, and high retention capacity to entrap the organic matter (Flemming et al., 1996; Nayar et al., 2007; Venkatraman et al., 2013). Hence, finer sediments observed in the dredging sites retained higher organic carbon content compared to the coarser sediments in the non-dredging sites. In addition, organisms getting fragmented by dredging (Newell et al., 1999) and the inputs through sewage from land bordering the channels might also have contributed towards the enhancement of the organic carbon in the sediments of the dredging sites (Hossain et al., 2014; Robin et al., 2012).

Dredging activity involves the mechanical removal of sediments, which ultimately affects the bottom fauna by the alterations caused in their habitat. In comparison to the non-dredging sites, the remarkable reduction in the faunal density and biomass observed in the dredging sites further affirms the impact of the dredging activities on the benthic biota (Fig. 2). Newell et al. (1998) reported reduction in macrobenthic density and biomass associated with dredging activities from a variety of habitats such as mud, oyster shell deposits, sand and gravel deposits.

Over three consecutive years, marked variability was observed

in the benthic community composition between the dredging and non-dredging sites. In the dredging sites, a conspicuous dominance of opportunistic tubificid oligochaetes was evident, whereas in the non-dredging sites polychaetes predominated (Fig. 3). The lower density of molluscs and amphipods were also conspicuous in the dredging sites. As molluscs prefer a stable substratum, the unstable substratum in dredging sites contributed to the low density of molluscs in these sites. Studies revealing a negative impact on the distribution of bivalves associated dredging activities in Florida bay further supports our observation (Conner and Simon, 1979; Simon and Conner, 1977). As the churning of suspended particles associated with the dredging process, often clogs the feeding organs of these filter feeding organisms, they might prefer to avoid such turbid conditions (Bolam and Rees, 2003; Kennish, 1991). Moreover, dredging conducted throughout the year including monsoon in the CE also would negatively affect the recruitment of slow growing bivalves much more than the polychaetes. In addition, discernible change was also observed in the amphipod community composition with the predominance of species like *Melita zylanica* and *Ampelisca* sp in dredging sites and dominance of *Caprella* sp and *Eriopisa chilkenis* in the non-dredging sites.

The opportunistic tubificid oligochaetes observed in higher densities in the dredging sites might have been favored by their unique adaptations of rapid proliferation within short period of

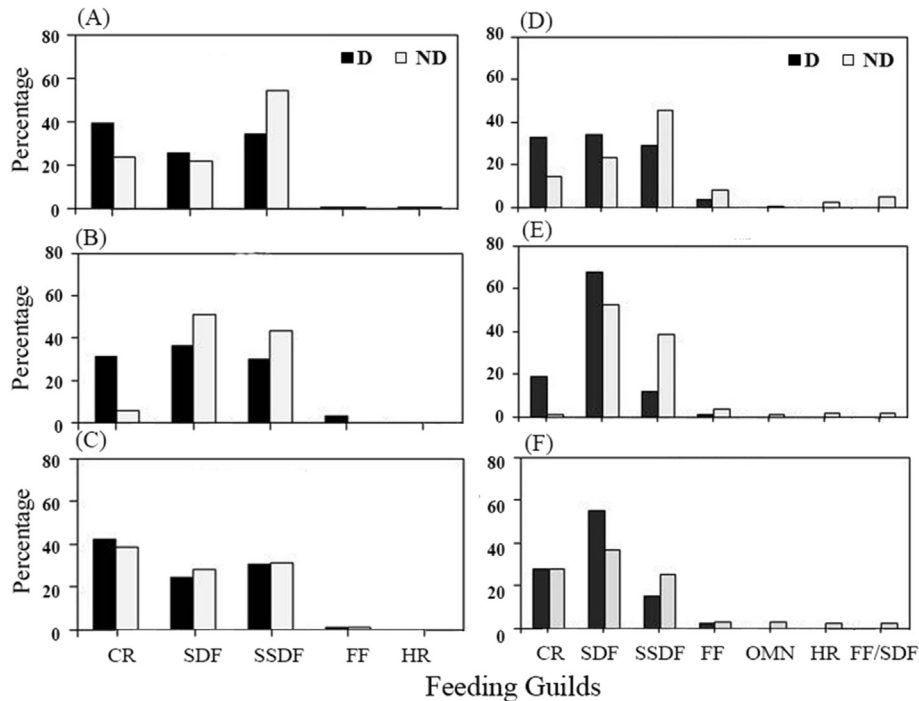


Fig. 7. Seasonal distribution of polychaete feeding guilds (A, B, C) and that of the characterizing species (D, E, F) in dredging and non-dredging areas of the CE during pre-monsoon (A&D), monsoon (B&E) and post-monsoon (C&F) period respectively (CR-Carnivores, SDF-surface deposit feeders, SSDF-sub-surface deposit feeders, FF-Filter feeders, HR-Herbivores, OMN-Omnivores, FF/SDF-either Filter feeder or SDF).

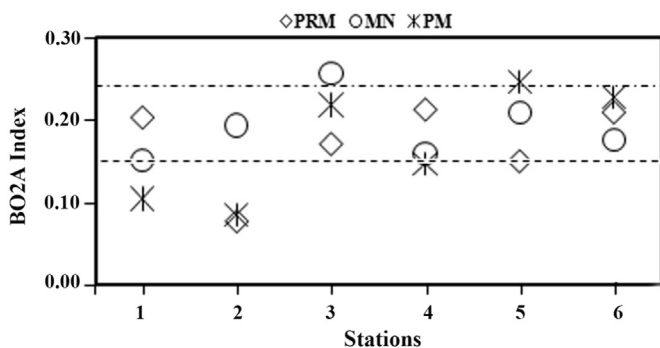


Fig. 8. Seasonal variation of BO2A index in Cochin estuary (index value > 0.15 to ≤ 0.24 indicates moderate, index > 0.24 indicates poor condition of water body) (PRM-pre-monsoon; MN-monsoon; PM-post-monsoon).

time (Giere and Pfannkuche, 1982), and because of their high tolerance to unsuitable environmental conditions like hypoxia and nutrient enrichments (Caspers, 1973). The presence of organic rich sediment and lack of competition from other macrobenthic fauna might also have favored their establishment in the disturbed environment of the dredging sites. However, the variations observed in benthic biomass in the dredging sites did not correspond with the variations in benthic density during the present study. The inconsistency evident between the biomass and density of the dredging locations can be attributed to the proliferation of the small sized opportunistic organisms. Similar observation of Besser et al. (1996) in Detroit River, USA substantiates the observations in the present study. Among polychaetes, the markedly higher density of species like *Mediomastus capensis*, *Prionospio cirrifera*, *Cirratulus cirratus*, *Cirratulus filiformis* and *Cossura coasta* in the dredging sites point towards the proliferation of r-selected opportunistic species. The occurrence of opportunists like

Prionospio cirrifera and *Cirratulus* sp as indicators of oxygen depletion and *Cossura coasta* as an indicator of sediment instability is reported among the macrobenthic communities of the AS continental margins (Abdul Jaleel et al., 2014; 2015) which demonstrates their adaptability to similar disturbed environmental conditions associated with the dredging process. As *Cirratulus cirratus* is an extremely asynchronous species with no seasonal breeding patterns and spawn at any time of the year (Giangrande, 1997), these attributes might have favored their dominance in the dredging locations when all the other species failed to establish themselves. Similar to the present study, Ceia et al. (2013) observed an increase in density of macrobenthic taxa such as *Mediomastus* sp, *Oligochaeta*, *C. capitata*, *Sigambra* sp, *Ampelisca* sp and a decrease of *Pectinaria* sp from the dredged sites of Mondego estuary, Portugal which further corroborates our observation.

Globally, extensive studies have been carried out on dredging activities and their impacts on benthic species composition, population density and biomass (Desprez, 2000; Sarda et al., 2000; Van Dalfsen et al., 2000). In the present study, the prominent decline observed in the benthic diversity and species richness in the dredging sites indicates the disturbance caused by the maintenance dredging on the benthic community in the CE. Similar results of decreased benthic diversity concurrent to the dredging processes in the Chesapeake Bay and Swedish estuary (Pfitzenmeyer, 1970; Rosenberg, 1977) further upholds this view.

Generally, a healthy aquatic ecosystem is characterized by a balanced benthic community constituted by functionally diverse forms and gives less opportunity for the predominance of one or a few taxa/species. The response of the benthic organisms to prolonged environmental stress often results in diminution of size, diversity and dominance by a single or group of opportunistic species (Gray, 1989). The low species diversity and dominance of opportunistic species in the dredging sites of the CE indicates the negative impact of maintenance dredging activities. Among the

discriminating species, the lower density of Caprellid amphipod *Caprella* sp, Photid amphipods such as *Photis digitata* and *Cherriophotis megacheles*, gammarid amphipod *Eriopisa chilensis*, gastropod species *Littorina littorea*, polychaete *Dendronereis estuarina* and *Bivalvia* sp in the dredging sites whereas the higher density of the oligochaete *Tubificidae* sp, polychaete species like *Sigambra parva*, *Nephtys oligobranchia*, *Cirratulus filiformis*, *Cirratulus cirratus* and tanaid species *Apeudus chilensis* further affirms their tolerance to the disturbed conditions (Fig. 6).

While assessing the impact of natural and anthropogenic disturbances on the benthic community, the feeding guild diversity is often the best method as a surrogate of the ecosystem functioning (Magalhaes and Barros, 2011; Pacheco et al., 2011). The analysis of the feeding guild composition of the polychaetes and that of the characterizing species revealed a clear dominance of carnivores in the dredging sites. As mobility helps organisms to efficiently adapt in a continuously disturbed environment, the relatively high motility of the carnivorous polychaetes might have favored higher density of this group in the dredging sites throughout the study period. Carnivores may also take advantage of the dead and injured organisms, damaged directly by dredging activities, as sources of food (Gutperlet et al., 2015). Interestingly, the non-dredging sites supported organisms with diverse feeding guilds such as surface deposit feeders, sub-surface deposit feeders, carnivores, herbivores, omnivores, and filter feeders and in turn indicates the availability of diverse food sources, resource partitioning and subsequently diverse trophic pathways (Ulanowicz, 1997). The BO2A index, also suggests the prevalence of contrasting environmental conditions in the dredging and non-dredging sites. Higher indices observed in the dredging sites (0.15–0.26), point towards the proliferation of opportunistic species and a decline in many sensitive species, and the index value reaching greater than 0.24 supports the poor environmental conditions prevailing in the dredging sites. The slightly higher BO2A indices (avg. 0.18) evident in the non-dredging sites during the monsoon may be due to the disturbances imparted by the monsoonal rainfall and associated run off.

Several studies suggest that macrobenthic re-colonization can only be possible if the ongoing dredging activities are stopped (Boyd et al., 2003; Guerra Garcia et al., 2003; Sarda et al., 2000). However, the re-establishment of the pre-dredging benthic communities can be attained only after the restoration of the sediment composition (Waye-Barker et al., 2015). The recovery time of the impacted areas depends on the magnitude and the frequency of disturbance activities (Lundquist et al., 2010) and the possibility of the recovery into a large stable community is less with the increase in the disturbance rate (Thrush and Dayton, 2002; Thrush et al., 2006). The communities inhabiting the fine mud is considered to recover more rapidly (1–18 months) than organisms dwelling in sand, gravel and corals reefs as they are often dominated by r-selected opportunistic species (Newell et al., 1998). Thus the recolonization process in highly variable shallow habitats like estuarine ecosystems is more rapid compared to more stable habitats. In the present study, the observed dominance of the opportunistic species in the dredging sites is an obvious indication of recolonization by the r-selected benthic community in the system impacted by the continuous dredging activities. The CE, a shallow dynamic estuary, where continuous dredging have been carried out in the channels which are intermittently influenced by the tidal activities, the recolonization of the opportunistic communities will be comparatively fast as the substratum is fine mud. Fast colonization can be possible after a physical disturbance in highly dynamic areas (Borja et al., 2010). The results of the study also suggest the possibility of recolonization by the opportunistic benthic community in the estuary. However the time required for the succession into a stable, complex community cannot be predicted,

as continuous dredging is being practiced in the CE. Since dredging of the navigational channels in the CE is a continuous activity, and considering the significant differences observed in the macrobenthos of dredging and non-dredging regions, the re-establishment of a stable benthic community in this area might not be possible within a short period of time.

As proper navigation in ports is critical for the trade and commerce, especially in a developing country like India, the deepening of the navigation channels by dredging activities cannot be avoided. However, various direct and indirect effects of intense dredging activities pose major environmental threats to the estuarine ecology. Hence for the proper management of the ecosystems, it is necessary to generate detailed information on the environmental conditions and ecology in such regions, and the influence of developmental and socio-economic activities on these ecosystems (de Jonge, 2000). In order to minimize the effects of dredging, these operations can be avoided especially during the sensitive breeding, spawning and larval dispersal periods of the estuarine organisms. Region specific evaluations should be undertaken on the local fauna and in order to ascertain their reproductive and growth cycles. Barletta et al. (2016) recommended avoiding the dredging activities during peak rainy season in order to conserve the recruitment of important fishery species. The recovery time for the macrobenthic fauna will be longer if the dredging frequency and time period is longer (Ceia et al., 2013). So frequency of dredging activities can be reduced in order to increase the possibility of recovery of the macrobenthic fauna. Further studies have to be carried out in the CE, giving consideration to all the accessible methods and tools such as statistical models, exploratory models and simulation models to mitigate the effects of the human intervention into these sensitive and vulnerable ecosystems as suggested by de Jonge et al. (2014). The present results emphasizing on the physico-chemical and biological attributes in relation with dredging activities in CE will be helpful in providing right direction to ecological management strategists.

5. Conclusions

Maintenance dredging activities have a significant impact on the sediment characteristics and macrobenthic community in the CE. The present study revealed a reduction in the macrobenthic density, biomass and species diversity in the dredging sites. Distinct communities were observed in the dredging and non-dredging sites, with a marked dominance of opportunistic taxa in dredging sites. Predominance of tubificid oligochaetes and opportunistic polychaete species, as well as the low density of amphipods and molluscs at the dredging sites revealed the varied impact and also the responses of the continuous dredging activities on the benthic community. Examination of feeding guilds revealed the proliferation of the carnivores and deposit feeders in the dredging sites, and organisms with diverse feeding modes in the non-dredging sites thus here reflecting the impact of dredging activities on ecosystem functioning. The BO2A index demonstrated a moderately-poor ecological quality in the dredging sites and further affirmed the poor ecological health of the estuarine ecosystem.

In order to minimize the effects of dredging, it is suggested to avoid dredging during the sensitive breeding, and recruitment periods of marine organisms, and the frequency of dredging operations may be reduced in order to attain fast recovery of the fauna. The information generated by the present study will be useful in the formulation of effective management measures needed for the protection and restoration of benthic community in this extremely vulnerable, yet highly valuable estuarine ecosystems.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at <http://dx.doi.org/10.1016/j.ocecoaman.2017.04.020>.

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