ECOLOGY OF MACROBENTHIC FAUNA OF THE COCHIN ESTUARY AND ADJACENT COASTAL WATERS

Thesis submitted to

COCHIN UNIVERSITY OF SCIENCE AND TECHNOLOGY

in partial fulfillment of the requirements
for the award of the degree of

DOCTOR OF PHILOSOPHY
in
MARINE BIOLOGY
under the
FACULTY OF MARINE SCIENCES

REHITHA T V
Reg. No. 4060

CSIR-NATIONAL INSTITUTE OF OCEANOGRAPHY
REGIONAL CENTRE,
KOCHI - 682018, INDIA

JUNE, 2017

Certificate

*This is to certify that the research work presented in this thesis entitled "**ECOLOGY OF MACROBENTHIC FAUNA OF THE COCHIN ESTUARY AND ADJACENT COASTAL WATERS**" is based on the original work done by **Mrs. Rehitha T V** (Reg. No. 4060), under my supervision at CSIR-National Institute of Oceanography, Regional Centre, Kochi, 682018, in partial fulfillment of the requirements for the degree of **Doctor of Philosophy** in Faculty of **Marine Sciences,** Cochin University of Science and Technology, Kochi, 682018 and that no part of this work has previously formed the basis for the award of any degree, diploma, associateship, fellowship or any other similar title or recognition in any universities or institutes.*

I also certify that all the relevant corrections and modifications suggested by the audience during the pre-synopsis seminar and recommended by the doctoral committee of the candidate has been incorporated in this thesis.

Dr. N V Madhu
(Supervising Guide)
Scientist
CSIR-National Institute of Oceanography
Regional Centre, Kochi- 682018

Kochi -18
June, 2017

Declaration

*The research work presented in this thesis entitled "**ECOLOGY OF MACROBENTHIC FAUNA OF THE COCHIN ESTUARY AND ADJACENT COASTAL WATERS**" submitted in partial fulfillment of the requirements for the degree of **Doctor of Philosophy**, is a bonafide record of the research work done by me under the supervision of **Dr. N V Madhu**, Scientist, CSIR-National Institute of Oceanography, Regional Centre, Kochi, 682018. No part of this work has previously formed the basis for the award of any degree, diploma, associateship, fellowship or any other similar title or recognition in any universities or institutes*

Kochi-18

June, 2017 ***Rehitha T V***

.. to my Family and Teachers

Acknowledgements

I express my sincere gratitude to my supervising guide Dr. N V Madhu, Scientist, Biological Oceanography Division, CSIR-National Institute of Oceanography, Regional Centre, Kochi. I will forever be grateful to my Research Guide for his infallible guidance, intellectual inputs, and the continuous encouragement during the course of my study. I take this opportunity to convey my respect and indebtedness to him.

I express my deep sense of gratitude to Dr. C Ravichandran, Senior Principal Scientist, CSIR-NIO, RC, Kochi for his support, encouragement and for the help rendered in field study and sample collection for my study.

I wish to pronounce my gratitude to Dr. T Pankajakshan, Scientist-in-Charge and Dr. P S Parameswaran, former SIC of this centre for providing necessary facilities to carry out my research. My deep sense of gratitude to Dr. S W Naqui, Director CSIR-NIO, Goa for providing me with the necessary infrastructure and excellent supporting facilities during the course of my Ph.D work.

I am grateful to Dr. Abdul Jaleel K V, Scientist, CSIR-NIO, RC, Kochi especially for his valuable suggestions and comments which have helped in improving the quality of my thesis.

I owe my gratitude to Dr. Kusum K K and Dr. Vineetha Gopinath for their inspiration, valuable suggestions and encouragement I received throughout the tenure of my doctoral programme. Interactions with them always provided encouragement for my work.

I am very much thankful to all the Doctoral and Research Committee members for their valuable suggestions. I also thank Mr. Thampy, Librarian, CSIR-NIO-RC Kochi for his cooperation. I also acknowledge the support extended by the Project, SSP 2179, and SSP 2686 for the financial support and fellowship during my Ph.D work.

I am thankful to my dear friends Mr. Ullas N, Mrs.Meenu Paul, Mrs. Lallu K R, Mr. Reny P D for the encouragement and support I received throughout the tenure of my doctoral programme. I am grateful to my dear collegues Mrs. Resmi P, Mr. Benny P Y, Mrs. Reshmi R, Mrs. Ashwini R, and Mrs. Pravitha for the assistance rendered during the study period. I also acknowledge Usha V parameswaran, for her timely help rendered during my study. I am also thankful to my colleugues Mr. Akhil, Mr. Rafeeq, Mr. Jineesh V K, Mr. Shinu Antony, Mr. Naveen k R, Mr. Dileep K., Mr. Monson, Mr. Deepak M P, Mr. Retheesh U, for their help rendered during the field sampling.

I sincerely thank all my colleagues and friends from CSIR-NIO, RC, Kochi for their timely help and support.

I sincerely thank my beloved parents, siblings and in laws for their immense support throughout life and hope that they would feel proud of this effort. My study would not be complete without the sincere support of my husband Dr. Vipindas P V who have supported and encouraged me throughout my doctoral programme. I fondly acknowledge my beloved son, Rithunath for all the blissful moments and happiness that he has given me.

Rehitha T V

INDEX

Contents

Chapter - 5 Impact of maintenance dredging on macrobenthic community in Cochin estuary 183-266

LIST OF ABBREVIATIONS

1. CE	-	Cochin Estuary
2. AS	-	Arabian Sea
3. ISM	-	Indian Summer Monsoon
4. DO	-	Dissolved Oxygen
5. BOD	-	Biological oxygen demand
6. SPM	-	Suspended Particulate Matter
7. CR	-	Carnivores
8. SDF	-	Surface deposit feeders
9. SSDF	-	Subsurface deposit feeders
10. FF	-	Filter feeders
11. OMN	-	Omnivores
12. HR	-	Herbivores
13. ANOVA	-	Analysis of Variance
14. ANOSIM	-	Analysis of Similarity
15. CCA -	-	Canonical correspondence analysis
16. BO2A index	-	Benthic Opportunistic Annelida Amphipods index

PREFACE

Cochin estuary (CE), the largest wetland ecosystem opening into the south east Arabian Sea, is identified as one of the most productive estuarine ecosystems along the west coast of India. The estuary is known for its fishery resources including fin fishes and shell fishes. Like most of the other tropical estuaries in the world, Cochin estuary also has been increasingly affected by several anthropogenic interventions such as intertidal land reclamation, sewage disposal, expansion of port and associated dredging activities and urbanization etc. Increasing impact of human activities along the coastal environments has been triggered attention towards the need for monitoring, assessing and managing its ecological integrity. Macrobenthic organisms play imperative role in ecosystem processes such as nutrient cycling, pollutant metabolism, dispersion and burial. Because of their limited mobility, they are widely used as biological indicators of ecosystem health and to reflect changes in the marine environment, such as deterioration of water and sediment conditions.

The present thesis on ecology, distribution and abundance of macrobenthic fauna in the Cochin estuary and adjacent coastal waters is presented in six chapters. Chapter 1 describes the characteristic features of benthos/macrobenthos, its classification and importance in the aquatic ecosystems, the influence of abiotic and biotic parameters on macrobenthos, and hydrographic features of the Cochin estuary. Chapter 2 deals with a detailed account of the study area, sampling methodology and analytical procedures. Chapter 3 includes the description of spatio-temporal distribution of macrobenthic community, feeding guilds of the polychaetes, and the influencing factors on their distribution and community structure in the CE and adjacent coastal waters. Chapter 4 consists of a detailed account of the taxonomy and population structure of the tube building amphipod, *Chelicorophium madrasensis* in the CE and factors influencing its population structure. Chapter 5 elucidates the impact of maintenance dredging activities on environmental parameters, sediment characteristics, and macrobenthic community and their functional ecology in the CE during three consecutive years (2009-2011). Chapter 6 includes summary of the thesis and conclusion.

Chapter 1

GENERAL INTRODUCTION

Contents

1.1 Estuary

Estuaries are highly complex and dynamic ecosystems in the world, which functions as a transition zone between marine and freshwater ecosystems. Most of the estuaries are shallow, characterised with high primary production and complex food chains, and serve as an habitat and breeding grounds for a variety of flora and fauna (McLusky, 1989). Estuaries provide the inhabitants with a highly variable physico-chemical environment and majority of the estuarine organisms are physiologically much adapted to these environmental fluctuations compared to the organisms in other aquatic systems. In estuaries, the tides and the river influxes have a prominent role in determining the hydrography of the estuarine ecosystems, mostly through its influence on the salinity patterns and water residence time. Of the many hydrographic variables, the sharpness of the salinity gradient from the sea to freshwater in estuaries forms the prime factor, regulating the biotic production of the estuaries through its profound influence on the growth and survival of the inhabiting organisms. Moreover, sediment inputs from rivers as well as the tidal pulsing resulting in high turbidity areas in estuaries, will also have a strong impact on the biotic potential through its effects on the estuarine primary productivity (Elliott and Whitfield, 2011). Estuaries are normally considered as the regions of high biological productivity, getting nourished by the regular inputs of anthropogenically generated allochthonous nutrients from the riverine and terrestrial regions (Ketchum, 1967). Inputs of surplus inorganic nutrients is reported to even cause eutrophication into the estuarine zones (Martin et al., 2011). Estuaries act as natural biological filters by filtering out sediments and pollutants carried by the rivers and streams before they flow into the oceans, thus providing a cleaner water for marine life (Day, 1989). In addition, the shallowness of the estuary often facilitates a stronger interaction between the water column and the bottom sediments and in turn promotes rapid regeneration and conservation of the nutrients (Day, 1989). Moreover, the behavioural and physiological adaptations exhibited by many estuarine organisms, to cope up with the stresses favours them in better exploitation of the available energy resources in the system. Their activities, such as building of burrows and tubes, and feeding in turn facilitate the movement of water and oxygen through the reduced sediments, making it well oxygenated.

Tropical estuaries are influenced by high rate of precipitation and evaporation, resulting in homogeneity in vertical salinity distribution during the dry season and weak to strong stratification in the wet season (Day, 1989). These estuaries are characterized by complex trophic dynamics, having diverse primary producers (such as phytoplankton, benthic algae, salt marsh plants, submerged seagrasses, and mangroves), grazers (zooplankton, benthos and fishes) and decomposers (bacteria, viruses and molds) and have an intense interaction between the water column and the bottom environment leading to high productivity. Among the varied biotic components of estuarine ecosystems, benthos forming an important intermediary link between the pelagic and benthic realm, have a prime role in the estuarine food web processes.

As many of the economically harvestable demersal fishes depend on the estuarine ecosystem for the completion of their life cycle, estuarine benthos contributes a major share in the commercial fishery landings. Estuaries form one of the most valuable aquatic ecosystems providing immense ecological services to the mankind and are one of the most over exploited natural habitats on earth. Massive changes in estuaries have occurred mostly in the twentieth century, when the human settlement grew dramatically in the coastal zone (McLusky, 1989). Increased anthropogenic intervention in estuaries as an area for draining and filling and as dredging channels for navigation have altered the estuarine bathymetry and morphology greatly (Pauly et al., 2005). Furthermore, the rapid industrialization and urbanization along the estuarine watersheds have introduced many toxic materials, like heavy metals, pesticides, and petroleum hydrocarbons into the estuarine water column resulting in poisoning the estuarine environment and raising the nutrients and organic matter levels leading to regular eutrophication events (Venugopal et al., 1982; Sujatha et al., 1993; Menon et al., 2000). In addition, the over fishing of commercially important shrimps and fishes, introduction of new species, either accidentally or purposefully, will also have a huge impact on the productivity and functioning of estuarine ecosystem as it leads to a replacement of ecologically and economically important indigenous species of the ecosystem.

1.2 Benthos and their classification

'Benthos' are collectively referred to all aquatic organisms which live in, on or near the bottom of a body of water. The term benthos is derived from the Greek word '*Bevoo'* meaning, "depths of the sea" (Haeckel, 1890). The benthic community encompasses a wide variety of plants, animals and microbes, having an integral role in the trophic dynamics of aquatic ecosystems. The autotrophic components inhabiting the benthic realm include various algae and rooted aquatic plants and are named the phytobenthos. The consumer community comprising of organisms ranging from protzoans to metazoans are called as the zoobenthos. Benthic microflora comprising the bacteria, fungi and many protozoans, constitute the decomposer community, are involved in the recycling of the essential nutrients.

Benthic fauna is categorized based on size into microfauna, meio fauna and macrofauna or macrobenthos (Mare, 1942). Organisms, which pass through sieves less than 63 μm mesh size are included in microfauna, and are mainly composed of prokaryotic microorganisms like bacteria and archaea and protozoans. Meiofauna include organisms that are retained by sieves of 63μm, and pass through sieves of 0.5 mm-1mm size and include organisms like gastrotrichs, kinorhyncs, nematodes, rotifers, and harpacticoid copepods; whereas organisms above 0.5μm (0.3 μ mesh is used in recent studies instead of 0.5 μ) size comes under the macrobenthic fauna. Macrobenthic faunal community usually encompasses polychaetes, crustaceans, mollusks, brittle stars (Echinoderm) and some icthyofaunal members like gobioid fishes.

Based on their habitat position, benthos are classified as epifauna, infauna and hyperbenthos (Pohle and Thomas, 2001). Epifaunal organisms live either attached or move on the surface of sediments and include amphipods, scallops, decapods etc. Infaunal organisms live within the sediments and move through the interstitial spaces or build tubes or burrows within sediments. Polychaetes, oligochaetes, and tube building amphipods often belong to this category. Hyper benthos are organisms that lives just above the surface of sediments.

1.3 Ecology of benthos

In estuarine ecosystems, benthic organisms have a crucial role in many ecological processes, and produce considerable physical and chemical changes in the water sediment interface (Gaudencio and Cabral, 2007). They form a food source for many epibenthic crustaceans, birds and fishes, and many species of shellfishes itself are consumed directly as food by human. Benthic organisms efficiently consume and convert the sedimentary organic matter into benthic biomass, dissolved organic matter and inorganic nutrients. Thus, they are involved in the transfer of nutrients from primary producers, through the detrital pool to higher trophic levels and also participates in the biogeochemical cycling of nutrients through mineralization process (Bryan and Langston, 1992). Hence, a comprehensive knowledge on the benthic fauna is utmost essential in elucidating the fishery potential, nutrient regeneration and biogeochemical cycling of aquatic ecosystems.

Most of the benthic organisms are sedentary or sessile and are considered to remain at same place even during conditions of environmental disturbances and hence, are used as potential indicators of environmental stress (Danulat et al., 2002). As benthic organisms live in close association with the bottom sediments, many pollutants ending up in the sediments will have an adverse impact on their density and survival and hence these organisms can be used as an efficient indicator of several anthropogenic activities (Bryan and Langston, 1992). Many of the benthic organisms are considered as sentinel organisms and biomarkers in the assessment of health of the ecosystem.

Abiotic parameters such as waves and tides, temperature, salinity, dissolved oxygen, nutrients, organic carbon, sediments and biotic parameters such as food availability, feeding activities, prey-predator relationship, recruitment and migration, are the prominent factors influencing the population dynamics and community structure of the benthic fauna. Though estuarine organisms have wide range of salinity tolerance, the rapid fluctuations in salinity associated with monsoonal precipitation and runoff in most of the tropical estuaries often cause a serious impact on benthic organisms. Furthermore, the nature and composition of the substratum play a crucial role on benthic fauna as they are the sediment inhabitants and fulfills their nutritional requirements from the sediments.

Many benthic organisms resort to different feeding habits depending upon the food availability. Several types of organic matter such as planktonic and benthic organisms, detritus, bacteria and dissolved organic matter often forms the potential food sources of the benthic fauna. According to the feeding preferences, benthic organisms are generally categorized mainly into five distinct trophic groups, as herbivores, filter feeders, surface deposit feeders, subsurface deposit feeders and carnivores. Herbivorous benthos feed mainly on algae and benthic diatoms; filter feeders filter organic matter and plankton in the water column, surface deposit feeders feed on organic matter, bacteria, detritus and benthic algae in the water-sediment interface, subsurface deposit feeders feed on organic matter by buried under sediments; and carnivores are predators or necrophagous species (Gaudêncio and Cabral, 2007).

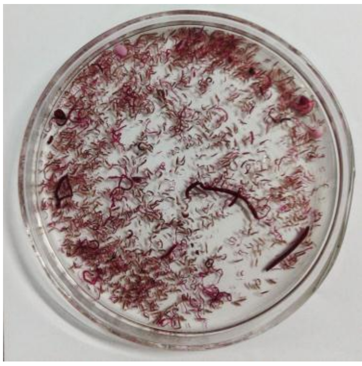


Figure 1.1 Macrobenthic assemblage

1.4 Polychaetes

Polychaetes, the dominant component of soft bottom macro-invertebrate benthic communities are diverse and ecologically significant functional constituent of estuarine and coastal ecosystems. These organisms are characterized by high stability and adaptability to different type of habitats (Simboura et al., 2000; Santos et al., 2005). They are the most abundant groups among the benthic fauna, both in terms of numerical abundance and species diversity, and contribute about 80% to the total macrobenthic abundance. About 16,000 polychaete species are reported so far and are placed fourth in the ranking of marine invertebrate species richness (Blake, 1995; Bouchet, 2000). Polychaetes exhibits varied reproductive modes and possess high reproductive rates(Wilson, 1991; Giangrande, 1997). Polychaetes are often the dominant players in the reworking, bioturbation of the marine sediments and in the recycling of the organic matter between the pelagic and benthic environments. Polychaete worms serve as a potential food source of several commercial fishes and shellfishes, and have a pivotal position in the benthic food chain. Their feeding activities endorse the decomposition of the organic matter and irrigate the sediments by allowing more oxygen to penetrate to the deeper sediments (Brown et al., 2000). Biotic factors, such as competition and predation, and abiotic factors, such as depth, current speed, temperature, salinity, sediment type, organic matter content and oxygen are regarded as the major factors affecting the polychaete community structure. In addition, anthropogenic activities are also found to influence the distribution and diversity of polychaetes.

Polychaetes have been used as indicators of environmental perturbations since earlier days and are also used as good indicators of species richness and diversity patterns in benthic invertebrate assemblages (Van Hoey et al., 2004). Polychaetous annelids display a wide variety of feeding modes such as surface and subsurface deposit feeding, suspension feeding, mud swallowing, carnivory and herbivory, and even parasitism (Fauchald and Jumars, 1979). This makes them the highly abundant group in many aquatic habitats compared to other macrobenthic fauna. Polychaetes have been used in ecological monitoring studies and also in experimental studies associated with the environmental pollution. *Capitella capitata,* the most widely used polychaete indicator species of pollution (opportunistic species), is found to dominate in organically rich sediments. Opportunistic species are often the pioneer forms dominating the initial stages of succession after disturbance and among polychaetes, species belonging to families Capitellidae, Cirratulidae, and Spionidae are taken as good indicators of organic pollution in sediments (Bellan et al., 1988). The presence or absence of a particular species in sediments can also be used as an indication of health of the benthic environments (Pocklington and Wells, 1992). Some of the important polychaetes used as positive indicators of stressed environments are the Capitellid species such as *C. capitata*, (Rivero et al., 2005) and *Heteromastus filiformis*(Ahn et al., 1995), the spionids *Malacocerus fulginosus*, *Paraprionospio pinnata*, and *Polydora ligni*(Mendez et al., 1998; Dix et al., 2004), the nereid *Neanthes (Hediste) diversicolor*, the dorvilleid *Ophryotrocha adherens* (Bailey-Brock, 2000) and the cirratulidae *Chaetozone setosa* (Rygg, 1985). Absence of the polychaetes such as the polynoid

Harmothoe imbricata and the maldanid *Maldane sarsi* indicates poor ecological conditions, while the absence of members belonging to the genera *Paramphinome* (Family Amphinomidae), *Ceratocephale* (Family Nereididae), Harmothoe (Family Polynoidae) and *Lumbrineris* (Family Lumbrineridae) indicates low species diversity raised by detrimental environmental conditions.

1.5 Amphipods

Amphipods, the dominant and diverse crustaceans among the benthic community form the most abundant members of the Super Order Peracarida. They live in a variety of habitats and forms major prey for fishes and larger invertebrates. Amphipods play diverse roles in the trophodynamics, as primary consumers, omnivores, carnivores and opportunistic feeders, and change their feeding modes according to the food availability. They lack pelagic larval stage and hence have specific habitat requirements. Amphipods brood their young in the marsupium and the recruitment occurs within the community itself. Since amphipods are characterized by higher numerical abundance and high sensitivity to a variety of toxicants and pollutants, are considered as sensitive environmental indicators (Ingole et al., 2009). Due to their ecological relevance, amenability and sediment tolerance, amphipods are used in many eco-toxicological studies (Onorati et al., 1999). Furthermore, their wide distribution, ease of handling and acclimatization to laboratory conditions often makes them a potential tools for bioassays (Schlekat et al., 1992).

The Order Amphipoda is composed of four suborders: Gammaroidea; Ingolfiellidea; Caprellidea; and Hyperiidea. Gammaridean amphipods represent the broad and diverse superorder under the Order Amphipoda, and displays high population densities, as well as high species diversity in a single area (Conlan, 1994). The species of the family Corophiidae are free living benthic gammaridians and most of them construct tubes on sessile objects in the sediments of harbors and estuaries. Many are able to tolerate slight variations of salinity in the surrounding water, and a few species are known to inhabit fresh habitats (Crawford, 1937). Corophideans live in tubes constructed by lipoprotein threads secreted from glands in their 3rd and 4th pereopods. *Corophium* Latreille, 1806 is a cosmopolitan genus in temperate and tropical waters (Barnard and Karaman, 1991). Most of them live in shallow marine regions, often in estuaries and harbors as burrowers or tube-dwellers. Getting nutritionally benefitted from the detritus and epipelic microalgae, they can establish themselves to high densities. Many species form important prey items for foraging shore-birds, demersal fishes and other crustaceans (Mattila and Bonsdorff, 1989; Eriksson et al., 2005). During their feeding, burrowing, tube construction and irrigation activities, they transport particles and fluids thereby affecting the physical and chemical properties of the sediment. The burrow ventilation of the amphipod causes the removal of NH_4^+ and hence their abundance enhances both nitrification and denitrification processes in the sediments (Pelegri et al., 1994).

1.6 Cochin estuary

Cochin estuary (CE), the second largest wetland in India (Qasim, 2003), is a geometrically complex estuarine system covering a total area of about 300 km^2. It is connected to the Arabian Sea through two permanent inlets, one at Cochin (450 m) and the other at Azhikodu (250 m). The CE is a micro-tidal estuary, regularly influenced by the tidal intrusion of seawater from the Arabian Sea and fresh water inflow from seven major rivers (Qasim, 2003). The northern limb of estuary receives runoff from two rivers (Periyar and Chalakudy) and the southern limb from five rivers (Muvattupuzha, Meenachil, Manimala, Pamba and Achancoil), thus contributing to an annual freshwater influx of 22,000×106 m^3(Srinivas et al., 2003). The environmental characteristics of this estuary is modulated by the Indian Summer Monsoon (ISM), with high runoff during wet monsoon season, and referred as 'monsoonal estuary'(Vijith et al., 2009). The CE receives about 320 cm rainfall annually, of which nearly 60-65% occurs during the southwest monsoon season (Qasim, 2003) and leading to a complete freshening of the estuary during the peak monsoon period (Revichandran et al., 2012). The depth of the estuary varies between 1.5 m and 6.0 m except in the dredging channels where depth is maintained to about 10-13 m.

As the southwest coast of India comes under the heavy influence of the Indian Summer Monsoon (ISM), seasons are classified according to the monsoonal rainfall and run off patterns. Accordingly, three seasonal conditions prevails over the region, i.e. Pre-monsoon (February-May), Monsoon (June-September) and Post-monsoon (October-January). Heavy rainfall and consequent freshwater influx causes salinity stratification in the CE during monsoon season. During post-monsoon season, the freshwater influx gradually decreases and tidal activity increases and the estuary become partially mixed with weak stratification. The estuary attains homogeneous well mixed condition only during the pre-monsoon period when maximum seawater incursion occurs into the estuary reaching even up to the upstream locations (Menon et al., 2000). The salinity gradient of the estuary supports diverse species of flora and fauna according to their tolerance to the varying salinity environments (Menon et al., 2000). Considering the nutrient status in CE, excess amounts of nutrients have been reported in the estuary throughout the year as a result of the inputs from the land drainages, agricultural runoff, industrial discharges, aquaculture, and from the terrestrial discharges (Madhu et al., 2007a; Martin et al., 2013). Along seasonal scale exceptionally high levels of nutrients occurs in the estuary during the summer monsoon compared to other two periods. Though nutrients are surplus in the estuary, light availability and turbidity determine the density of phytoplankton in CE, which exhibited high density during pre-monsoon season (Madhu et al., 2010a; Madhu et al., 2010b). This tropical estuary was characterized with high primary productivity (average gross primary production is 280 g C/ m^2/yr,) (Qasim et al., 1969). The population density of zooplankton in the estuary are not limited by primary production, as it often exceeds the consumption by zooplankton throughout the year (Menon et al., 2000). Copepods dominated sharing > 87 % of the mesozooplankton community in CE with a crucial role in the trophic dynamics of the CE (Madhu et al., 2007b; Vineetha et al., 2015). Concurrent to the high phytoplankton production, abundance of herbivorous or omnivorous copepods were notable in the estuary (Madhupratap, 1979).

According to the prior studies, the macrobenthic fauna of the CE was mainly composed of polychaetes, crustaceans and mollusks and few other organisms like coelenterates and gobioid fishes (Kurian et al., 1975; Pillai, 1977; Batcha, 1984). Similar to phytoplankton and zooplankton, the seasonal distribution and abundance of benthic fauna was influenced by salinity (Batcha, 1984). Crustacean benthic fauna in the CE was mainly represented by amphipods, isopods, tanaids, cumaceans, penaeid prawns and brachyurans (Batcha, 1984). Polychaetes, amphipods, isopods and tanaids form important macrobenthic links in the food web of the estuary (Anon, 1996). According to the previous studies, Polychaetes occupy a dominant position among the macrobenthic community in CE (Sunil Kumar, 1993; Sheeba, 2000). As benthic fauna reflects the conditions of the ecosystem, regions receiving high inputs of sewage have been reported to undergo drastic changes in the composition of benthic fauna (Nisha, 2008). The estimated macrobenthic and meiobenthic production in the CE was 2276.8 kg C/ km^2/yr and 215.9 kg C/ km^2/yr respectively (Feebarani, 2009).

Changes in the CE with the increasing anthropogenic interventions began with the construction of Cochin Port between 1930 and 1940 and by the creation of Willington Island. With the increasing developmental activities, like land reclamation, port expansion, dredging, construction of dams and bridges, blockages in the circulations patterns of the estuary leading to a reduction in the water flow was notable in the estuary. Moreover, industrial effluents discharged by various industries and fertilizer plants resulted an increase in the heavy metals (Nair et al., 1990), radionuclides (Paul and Pillai, 1981), pesticides (Menon et al., 2000), in the sediments and water column. Increasing concentration of PHC associated with the discharges from shipping, fishing vessel operation, transportation, urban run-off, accidental spillages during tanker operations (Menon & Menon, 1998), and also sewage inputs have drastic effects on the estuarine environment. (Devi and Venugopal, 1989) reported on the high abundance of pollution indicator species, *Capitella capitata* in the industrial discharge sites in the CE. The increased bacterial heterotrophic activity due to enhanced nutrient levels has resulted in CO2 supersaturation and subsequent oxygen undersaturation in the estuary (Gupta et al., 2009). Construction of the Vallarpadam International Container Transshipment Terminal (ICTT) is the recent developmental intervention in the CE. Reduction in the exchange volume of the estuary from 126 Mm^3/tidal cycles to 35 Mm^3/tidal cycles during the past 3 decades and a considerable reduction in the diversity of plankton and macrobenthic community with notable increase in the abundance of benthic deposit feeders and pollution indicator organisms were observed from the CE (Martin et al., 2011). In addition, the analysis of the benthic quality status by using ecological indices such as AMBI and M-AMBI revealed stress on macrobenthic community in the CE (Feebarani et al., 2016).

Maintenance dredging conducted to maintain the depth of the navigation channel, is a major human intervention in the estuary, which are considered to evoke potential damage to the water quality and biotic resources of the CE. Similar to other estuaries, continuous dredging has been conducted in CE to allow the easy transport of vehicles. Several studies have been conducted globally on the impact of dredging activities on the macrobenthic fauna and reported an initial reduction in species diversity, abundance, and biomass as direct result of dredging activities (Sutton et al., 2009). Even the nearby places of the dredging area are indirectly affected by the sediment resuspension, the release of nutrients and chemicals, and changes in food resources by shifts of plankton bloom seasons (Newell et al., 1998; Van Dalfsen et al., 2000; Simonini et al., 2007). Although, studies on the impact of dredging activities on the benthic fauna have become an important subject at the global level (Kaplan et al., 1975; Van Dolah et al., 1984; Clarke et al., 1993), the knowledge on this aspect in tropical monsoonal estuaries is very limited (Brown and Kumar, 1990; Bemvenuti et al., 2005; Ogbeibu et al., 2010). In CE, earlier studies on the impacts of dredging mostly focused on the changes in water quality parameters such as salinity and nutrients (Joseph et al., 1998), and impacts on physico-chemical parameters (Balchand and Rasheed, 2000) associated with dredging activities.

Considering the benthic community, the prior studies conducted in the CE mostly focused on the spatial variability of macrobenthic fauna. Very few studies have been focused on the temporal variability of macrobenthic fauna. Since Polychaetes form the major taxa in the macrobenthic fauna, many studies addressed on its community ecology while little information is available on its functional ecology (feeding ecology) from the estuary.

Coming to amphipods, the second dominant groups among macrobenthic fauna, population structure and production (Cunha et al., 2000a; Cunha et al., 2000b; Drolet and Barbeau, 2012), substratum preferences (Meadows, 1964), coexistence with other species (Commito, 1982; Flach and De Bruin, 1993, 1994) effect of grazing and bioturbation on sediment stability by tubecolous amphipods (Gerdol and Hughes, 1994; Mouritsen et al., 1998) have been seriously dealt with in temperate waters but very few information are available on these tubecolous amphipods from the Indian waters.

1.7 Scope and objectives of the study

Macrobenthos, an ecologically significant component of estuarine ecosystems, play an important role in the estuarine food web dynamics, by forming a major source of energy for higher trophic levels including the demersal fishes. Moreover, their sensitivity to the environmental variables makes them efficient indicators of the alterations caused by natural and anthropogenic activities. In most of the coastal ecosystems, community structure emerges as an outcome of the complex interaction between biotic and environmental variables. Ecological studies are important in procuring information on the structure and function of an ecosystem through the assessment of inhabiting communities along spatio-temporal scales. Tropical estuaries are recognized as an ecologically significant and complex habitats that have a critical role in the global ocean processing (Smith et al., 2003). Since the CE belongs to the tropical regime, seasons are primarily categorized based on the amount of monsoonal precipitation. The annual monsoonal rainfall exerts a significant influence on physico-chemical and biological characteristics of the CE and usually it brings about drastic changes to the environmental variables (Madhupratap, 1987; Menon et al., 2000). Temporal variability, the variability of communities through time has been used to evaluate the stability of aquatic ecosystems. Therefore, seasonal studies depicting the distribution of macrobenthic fauna in the CE and adjacent coastal waters are essential to understand the present status as well as the environmental/anthropogenic impact on benthic fauna. Since a continuous monitoring of macrobenthic fauna in terms of temporal scale are meagerly studied in the CE, the present study attempted to elucidate its community structure, with special emphasis on polychaetes in the CE and adjacent coastal waters with respect to the prevailing environmental conditions. A comprehensive knowledge on the impacts associated with the natural and anthropogenic activities on the estuarine biota will be helpful for the effective protection and management of the rich resources that estuarine ecosystem harbor.

Polychaeta, the predominant macrobenthic fauna, possess a variety of living strategies for adapting to various habitats such as, large variations in morphology, diverse feeding and reproductive modes. They have developed diverse feeding modes to utilize the nutritional needs effectively. In addition they are also involved in major functional roles such as recycling, reworking and bioturbation of marine sediments and in the burial of organic matter (Hutchings, 1998). Analysis of feeding guilds of polychaetes usually helps to understand the morphological mode of food acquisition and also to evaluate the environmental constraints on their trophic structure, and mobility. Since the information on feeding ecology of polychaetes are scanty in the CE, the present study focusing on the community structure and functional ecology of Polychaeta in the CE and nearby coastal waters. This information will helpful to interpret the polychaete assemblage patterns in the CE and to assess the environmental influence on them.

Even though studies have been came out regarding the benthic amphipods in the CE (Nair et al., 1983; Aravind et al., 2007), knowledge on the ecology of the tube building benthic amphipods is still lacking. For the first time, the study compiles knowledge on the population structure and ecology of corophid amphipod (*Chelicorophium madrasensis*) from a tropical estuary, which perform a key role in the estuarine benthic food web dynamics by acting as an intermediate link between the lower and upper trophic levels. The information on its ecology will be helpful in developing mass culture of these amphipods for their potential use as a suitable live feed organism for aquaculture purposes.

The present study also describes the impact of 'maintenance dredging' on the macrobenthic community in the navigation channels of the CE. Continuous dredging has been carried out to maintain the depth of the navigation channels in the CE. Therefore, an extensive study has been executed in the CE, taking into consideration the environmental changes brought about by dredging activities. The information generated from the study will be helpful in taking proper management strategies for the protection of the sensitive benthic ecosystem.

With this background the study propose the major objectives,

1) To study the spatio-temporal distribution of macrobenthic fauna in the CE and adjacent coastal waters
2) Understanding of feeding guilds of polychaetes in the CE and adjacent coastal waters
3) To study the ecology and population structure of a tube building amphipod *Chelicorophium madrasensis,* Nayar in the CE.
4) To study the impact of maintenance-dredging activities on macrobenthic community structure in the CE

*****$$*****

Chapter 2

MATERIALS AND METHODS

2.1 Description of the Study Area

2.1.1 Cochin estuary

The Cochin estuary (CE) is a semidiurnal micro-tidal estuary, covering an area of ~25600 ha (9˚ 30' - 10˚ 10' N to 76˚ 15' - 76˚ 25' E) and extending between Azhikode in the north and Alappuzha in the south (Qasim, 2003). The CE is characterized by its long axis lying parallel to the coastline, with several small islands and interconnected waterways. It has a surface area of 231 km^2 (calculated from satellite image) and a volume of 0.55 km^3(Gopalan et al., 1983). The estuary is generally wide (0.8-1.5 km) and deep (4-13 m) towards the south, but narrow (0.05-0.5 km) and shallow (0.5 2.5 m) towards the north (Gupta et al., 2009). It is running parallel to the Arabian Sea and connected by two permanent inlets, one at Cochin (Latitude 9°58' N) and other at Azhikode (Latitude 10°10' N). The Cochin inlet (~450m) is comparatively wider than Azhikode inlet (~250m).

Regular influence of sea water occurs in the estuary through the tidal intrusion (average tidal range is 1m), which diminishes considerably towards the head of the estuary (Martin et al., 2012). Tides in the CE are predominantly mixed semidiurnal, however, the tides at the southern most part is mixed semidiurnal to diurnal (Revichandran et al., 2012). The CE receives freshwater influx from seven rivers, two rivers from northern limb (Periyar and Chalakudy) and five from the southern limb (Muvattupuzha, Meenachil, Manimala, Pamba, and Achancoil), thus contributes an annual freshwater influx of 22,000×10^6 m^3 (Srinivas et al., 2003). Annual precipitation of the of the nearby regions of Cochin is ~320cm and of which ~60-70% come about during the south west monsoon period (Qasim, 2003). The total river discharge in the CE is several orders of magnitude higher as compared to its total volume and the complete freshening of the estuary occurs during peak monsoon period (Revichandran et al., 2012), hence it is considered as a monsoonal tropical estuary. This complex estuarine system (Joseph and Kurup, 1989) undergoes a characteristic transformation from a river-dominated system during the monsoon season (June–September) to a tide- dominated system during the pre-monsoon season (February–May). River Periyar draining into the northern estuary has a major influence on the salinity distribution of the CE (Madhupratap, 1987) and when the river discharge gradually diminishes, tidal influence (salinity) gains momentum to play a crucial role in the ecology of the system (Madhupratap, 1987; Menon et al., 2000).

The inorganic nutrients in the CE are highly influenced by the terrestrial anthropogenic inputs, fresh water discharge from the rivers, and seawater influx from the two inlets (Menon et al., 2000; Madhu et al., 2010). Earlier studies have revealed that the CE sustains excess inorganic nitrogen irrespective of the seasons (Sankaranarayanan and Qasim, 1969; Madhu et al., 2007). In accordance with the higher inorganic nutrient levels, the CE sustains higher level of chlorophyll a and phytoplankton abundance almost throughout the year (Jyothibabu et al., 2006; Madhu et al., 2010b). Considering the socio-economic importance, the CE has been included in the Ramsar site (no. 1214) of vulnerable wetlands to be protected in the year 2002 (Wetlands, 2002).

Anthropogenic activities in the CE have been started since the second half of the 19th century and it continues up to the present day. Without knowing the complex hydrodynamics, industries were permitted to establish at the upper reaches of the CE during the early stages of developments. Inadequate infrastructure and improper waste management strategies eventually led to the accumulation of pollutants (organic and inorganic) in the CE, especially in the northern region (Qasim, 2003; Babu et al., 2006). Approximately 1.04 x 10^5 m^3 d^{-1} of effluents and 260 m^3 d^{-1} of sewage introduced into the CE (Qasim, 2003; Balachandran et al., 2005). In addition, wastes from aquaculture fields (62 km^2) and agricultural fields (80 km^2) also have been dumped into the estuary (Thomson, 2002). The major sources of pollution in the CE were industries, municipal solid wastes, biomedical wastes, e-waste and domestic wastes etc. Presence of excessive nutrients have been reported in the system as a result of the impacts from industrial and agricultural runoff (Balachandran, 2001; Balachandran et al., 2002). Presence of high concentration of heavy metals; 3 fold in the case of Zn and 10 fold in the case of Cd was observed in the CE which positions the region among the impacted estuaries in the World. The flow-restrictions are found to be primarily responsible for the contamination of the sediment with heavy metals in the north estuary (Balachandran et al., 2005). Indiscriminate land reclamation resulted in the reduction of the estuarine volume by 40% nearly three decades ago (Gopalan et al., 1983). Thanneermukham bund has constructed in 1976 to prevent salt water intrusion, which is about 1,250 m long with 93 vent ways, each 12.2 m wide and 5.5 m high, and the sill is at an elevation of 3.38 m below mean sea level (Shivaprasad et al., 2013). It resulted in the reduction of upstream migration of marine fish and prawns, increased sedimentation and weed growth at upstream which affects the navigation and severely restricts the natural flushing of pollutants (Kannan, 1979; Revichandran et al., 2012). Hence the CE is experiencing high level of anthropogenic pressures during the last five decades (Menon et al., 2000). The greater Cochin Area bounded by CE occupies the 24th position amongst the critically polluted areas in our country (KSPCB, 2010) owing to the high population density and various types of manmade activities occurring in the city.

Siltation is a natural process occurring as a result of the river discharge and tidal flow, which is accelerated by the activities like deforestation, construction of dams, reservoirs and barriers and progressively led to swallowing of the estuary. In order to maintain the depth of the shipping channels large quantity of materials has been removed annually using dredgers to rectify the effects of siltation and for the easy water transport. These dredging activities led to many water quality changes such as, increase in the suspended particles, increase in inorganic nutrients, and exposure of sediment bound toxicants. Dredging and the disposal activities usually led to either the alternation or the removal of the habitats of sediment dwelling organisms, or smothering by the overlying sediments (Rasheed, 1997). In the estuarine and mangrove areas of the CE experienced enhanced (3-6 times higher) sediment accumulation rates than that in the adjacent inner shelf area (Manjunatha et al., 1998). The establishment of new Container Terminal at Vallarpadam, LNG terminal gas distribution system, and development of NTPC station at Puthuvaipu were the major industrial developments happened recently in the Cochin city.

2.1.2 Sampling locations

A Global Positioning System (GPS) was used to locate the sampling stations (Magellan NAV DLX 10, USA) and the details of the station positions were given in the following corresponding chapters.

2.2 Sampling strategy

2.2.1 Water quality parameters

Sampling was carried out from predetermined stations covering three different seasonal periods. A conductivity-temperature-depth (CTD 19 plus, Sea-Bird Electronics) profiler was deployed at each station to obtain the temperature and depth profiles. A Niskin sampler (5L capacity, Hydro-Bios, Kiel-Holtenau, Germany) was used to collect water samples from the bottom for the analysis of water quality parameters, such as salinity, pH, dissolved oxygen (DO), biological oxygen demand (BOD), inorganic nutrients and suspended particulate matter (SPM). The samples for DO (125 ml) and BOD (300ml) collected without air bubbles in acid washed (10% HCl) glass bottles, DO samples were fixed onboard and transported to laboratory. Water samples for the analyses of salinity, pH and inorganic nutrients were collected in cleaned polyethylene bottles and transported to the laboratory in ice boxes.

2.2.2 Benthos and sediment

The sediment and macrobenthic samples were collected in duplicates using a Van-Veen Grab with an area of 0.05 m^2. For the community analysis of macrobenthic fauna, the sediments collected were washed onboard through a 0.5-mm sieve (Birkett and McIntyre, 1971) and the organisms remaining in the sieve were transferred into a plastic container and preserved in neutral 5% formalin-Rose Bengal mixture. For the determination of sediment characteristics like organic carbon, texture and chlorophyll contents, separate sediments were collected, kept in iceboxes and transported to laboratory for further analysis.

2.3 Analytical methods

2.3.1 Environmental parameters

2.3.1.1 Temperature, salinity and pH

At every station CTD was used to get temperature and depth profiles. Salinity was estimated using a Digi Auto Salinometer (Model TSK, accuracy ±0.001) immediately after reaching the laboratory. The pH was measured using a pH meter (ELICO LI610, accuracy ±0.01).

2.3.1.2 Dissolved oxygen (DO)

Estimation of DO was carried out using Winkler's titrimetric method (Grasshoff et al., 1983). Water samples were fixed onboard by adding freshly prepared 0.5 ml of Winkler A (3 M Manganous chloride) and 0.5 ml of Winkler B (8 M alkaline iodide) and mixed properly (Grasshoff et al., 1983). The precipitate formed was dissolved using 1 ml of 10 N H_2SO_4 and titrated with 0.01 N sodium thiosulphate using starch as indicator. Concentration of oxygen is expressed as mg L^{-1}.

2.3.1.3 Biochemical Oxygen Demand (BOD)

The water samples for BOD estimation (Grasshoff et al., 1983) was incubated for 5 days at 20°C in the dark and after incubation, the samples were fixed by adding freshly prepared 0.5 ml of Winkler A and 0.5 ml of Winkler B and mixed properly. The reduction in the dissolved oxygen concentration from initial to final during the incubation period yields the biochemical oxygen demand. Concentration of oxygen is expressed as mg L^{-1}.

2.3.1.4 Suspended Particulate Matter (SPM)

For the estimation of SPM, 250 ml of water sample was filtered onto a pre-weighed Millipore membrane filter (47mm dia; nominal pore size- 0.45μm), and subsequently dried the residue at 80 °C to remove the water content. The weight of the filter was again measured and the differences in the weight indicate the amount of the suspended particulate matter on it and the value of the SPM was expressed in mg L^{-1}(APHA, 2005).

2.3.1.5 Inorganic nutrients

2.3.1.5.1 Ammonia-N (NH_4)

Ammonia-N was estimated according to the indophenol blue method of Koroleff (1983). In this method, both free dissolved ammonia gas and the ammonium ion was measured. This method estimates the sum of NH^{4+} and NH_3. Ammonia reacts with hypochlorite to form monochloramine, in a moderately alkaline medium, which in presence of phenol, catalytic amount of nitroprusside ions and excess of hypochlorite forms indophenol blue. A pH between 8 and 11.5 is required for the formation of monochloramine. Ammonia incompletely oxidized to nitrite at higher pH. Calcium and magnesium ions precipitate in seawater as hydroxide and carbonate respectively above pH 9.6. However, their precipitation can be prevented by complexing them with citrate buffer. The samples were 'fixed' by the addition of reagents immediately after collection and the absorbance, after the color development (after 6 hours) was measured at 630 nm using U V - Vis spectrophotometer (Shimadzu, 1650 PC Japan). The concentration was calculated based on the standard ammonium chloride (NH_4Cl) solution (precision: ±0.05). Concentration is expressed in µM of N-NH_4.

2.3.1.5.2 Nitrite-N (NO_2)

Nitrite was determined according to the method described by Bendschneider and Robinson (1952). In this method, nitrite was allowed to react with sulphanilamide in an acid solution. The resulting diazo compound reacted with N-(1-naphthyl)-ethylene diamine to form a highly coloured azodye. The absorbance was measured at 543 nm using a U V -

Vis spectrophotometer (Shimadzu, 1650 PC Japan). The concentration was calculated based on the standard Sodium nitrite ($NaNO_2$) solution (precision: ±0.05 μM). Concentration is expressed in μM N-NO_2.

2.3.1.5.3 Nitrate-N (NO_3)

Nitrate was estimated using the method described by Grasshoff (1970). In this method the nitrate present in the sample was reduced to nitrite using a reducter filled with copper coated-cadmium granules. The condition of reduction was adjusted so that nitrate is almost quantitatively converted to nitrite and not reduced further. Nitrite thus formed was estimated by the method of Bendschneider and Robinson (1952). Potassium nitrate (KNO_3) was used for standardization and concentration is expressed in μM of N-NO_3.

2.3.1.5.4 Phosphate (PO_4)

Inorganic phosphate was estimated by the method of Murphy and Riley(1962). Phosphate and ammonium molybdate were allowed to react in acid solution to give phosphomolybdic acid, which was reduced by ascorbic acid. Optical density was measured using a spectrophotometer (Shimadzu, Japan) after 10 min at 882 nm. Potassium dihydrogen phosphate (KH_2PO_4) was used as standard and the concentration is expressed in μM.

2.3.1.5.5 Silicate (SiO_4)

Silicate was estimated using protocol of Grasshoff(1964). Sample was allowed to react with ammonium molybdate resulting in the formation of silicomolybdate, phosphomolybdate and arsenomolybdate complexes and oxalic acid was added to reduce to silicomolybdous acid and the absorbance of blue color was measured at 810 nm. Sodium fluorosilicate (Na_2SiF_6) solution was used as standard and the he concentration is expressed in μM.

2.3.2 Sediment characteristics

2.3.2.1 Texture

The sediment sample from each station was dried overnight in a hot air oven 60°C. For the determination of sediment texture (sand, silt and clay), 5 g each of dried sample was accurately weighed and dispersed using Sodium Hexa Meta phosphate and kept overnight. samples were subjected to textural analysis (Krumbein and Pettijohn, 1938).

2.3.2.2 Organic carbon

Sediment samples from each station were subjected to chemical analysis to determine the organic carbon. The sediment samples were ground and powdered well after drying in hot air oven at 60^0C. The organic carbon content of the sediment sample was determined by wet oxidation method (El Wakeel and Riley, 1957).

Standardization

Add 75 ml of Conc. H_2SO_4 to a 500ml conical flask containing 10 ml of stock solution ($K_2Cr_2 O_7$ - 0.25N), and shake well. Then slowly add 200 ml of distilled water and shake well. Allowed to cool the solution in a trough of water, and add 1 or 2 drop of indicator. On titration the golden yellow color of the solution will slowly disappear and it became greenish. Then the light bluish color appears and finally to a brick red color that shows the end point of the reaction.

Weigh 0.2 g of the cleaned powdered sediment sample into a hard glass tube. Add 10 ml of chromic acid in to the glass tube. The test tubes were shaken well and heated in a water bath for 2 hrs until the sample was digested and then pour the content in to a conical flask containing 200 ml of distilled water. Two drops of indicator (Ferrous phenanthroline indicator) was added to it and titrated against 0.2 N Ferrous ammonium sulphate. The end point will be brick red color. A blank determination was also carried out in the same manner using 10 ml of prepared chromic acid. The organic carbon in the sediments was estimated by using the following formula.

$$\text{Organic carbon (mg/g)} = \frac{\text{B-S x1.15 x 0.6 X 1000}}{\text{Weight of the sample taken}}$$

B = Reading of Blank,

S = Reading of sample

1.15 = factor to be multiplied

2.3.2.3 Microphytobenthic biomass (Sediment Chlorophyll *a*)

Microphytobenthic biomass was estimated based on the concentration of chlorophyll *a* in the surficial sediment. A scoop of sediment collected during the field sampling was placed in a black plastic bottle and kept in freezer till analysis. The sediment sample was taken in a small petridish was allowed to lyophilize in a freeze drier. The lyophilized samples were grinded to powder. Add 1gm of lyophilized and grinded sediment into 10 ml of 90% acetone in a glass test tube, and kept at -4^{0}C in darkness for 24 hours. After incubation, centrifuge the samples for 10 minutes at 3000rpm and supernatant collected was measured spectrophotometrically before and after acidification, with a UV-visible spectrophotometer (UV-2550-Shimadzu) and quantified the pigment concentration using Lorenzen's equations (Lorenzen, 1967). The concentration was expressed in mg/g.

2.3.3 Benthos

For the macrobenthic community analysis, the sediments collected were washed onboard through a 0.5-mm sieve (Birkett and McIntyre, 1971) and the organisms remaining in the sieve, were preserved in neutral 5% formalin–Rose Bengal mixture. All the sieved organisms were examined, under a binocular stereozoom microscope (CATSCOPE CS-S 6080), and sorted out to the major macrobenthic taxa, for further analysis (*e.g.,* Polychaeta, Mollusca, Crustacea). The detailed identification of macrobenthic species, was carried out using the standard identification manuals, to the possible lowest taxonomic levels (Fauvel, 1953; Nayar, 1959; Day, 1967; Gosner, 1971; Fauchald, 1977) after the estimation of numerical abundance (ind.m^{-2}), and biomass (wet weight- g.m^{-2}).

2.3.3.1 Feeding Guilds

Feeding guilds in the study area was analyzed by identifying the feeding guild of the dominant macrobenthic fauna, Polychaeta and that of the characterizing species of the dredging and non-dredging stations (identified through SIMPER analysis- PRIMER 6.1.5) using published literatures (Gosner, 1971; Caine, 1977; Fauchald and Jumars, 1979; Imrie et al., 1990; Aravind et al., 2007; Mondal et al., 2010; Leal and Matthews, 2013). The various polychaetes encountered from the sampling locations of the CE represented in the present study were assigned to one of the following feeding guilds: carnivores (CR), surface deposit feeders (SDF), subsurface deposit feeders (SSDF), suspension feeders (FF), Omnivores (OMN), and Herbivores (HR).

2.4 Statistical Analysis

2.4.1 Karl Pearson's correlation

Karl Pearson's correlation was used for understanding the statistically significant relationship or associations between environmental variables, and biotic variables. In this correlation analysis, the basic quantity to determine the degree of correlation or correspondence between the two sets of variables is the average of the sum of all the products of deviations.

2.4.2 Analysis of variance (ANOVA)

Analysis of variance (ANOVA) was used to test for the differences between datasets and within areas for quantitative parameters. The data sets need not be equal in size. Data sets suitable for an ANOVA can be as small as three or four numbers, to infinitely large sets of numbers. One-way ANOVA uses one independent variable e.g site or month, season. In the present study one way ANOVA was used to test the significance of variance in the data sets between different seasons. 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.

2.4.3 t-test

t-test is based on t-distribution and is considered an appropriate test for judging the significance of a sample mean or for judging the significance of difference between the means of two samples in case of small sample(s) when population variance is not known. To know the variation in the biotic and abiotic parameters between the dredging and non-dredging stations in the CE, an unpaired non parametric t-test was performed, using the Graph Pad Prism (version 5.01).

2.4.4 Univariate Analysis

Various diversity indices were used for the comparison of communities. Univariate analyses of the macrobenthic density were carried out, using the PRIMER (version 6.1.5,); (Clarke and Warwick, 2001). Univariate measures included Margalef's richness (*d*) for species richness, Shannon-Wiener, (H') ($\log_2$) for species diversity, and Pielou's evenness (J') for species evenness. Species richness (Margalef): d = (S-1)/Log (N), is a measure of the number of species present, making some allowance for the number of individuals. Species diversity index (H') give the measure of the number of species in a sample and their relative abundance. The index is high in samples that have large numbers of unique species, or have greater species evenness. Pielou's evenness (J') is a measure of equitability, a measure of how evenly the individuals are distributed among the different species.

2.4.5 Multivariate analyses

Multivariate analysis was used to make out how multiple variables change together and allows us to condense the information for easy understanding. Multivariate analyses were carried out, using the PRIMER (version 6.1.5,); (Clarke and Warwick, 2001) and CANOCA software.

2.4.5.1 Cluster analysis & NMDS

Cluster analysis used to find natural groupings of samples such that samples within a group are more similar to each other, than samples in different groups. Agglomerative hierarchical cluster analysis (AHCA) on station versus macrobenthic species data through Bray–Curtis similarity and group-average linking were selected to categorize the assemblages/clusters (Clarke and Warwick, 1994). The Bray–Curtis similarity method, the most commonly-used similarity coefficient for biological community analysis, is used because it obeys many of the natural biological axioms in a way that most other coefficients do not (Clarke and Gorley, 2006).

To identify the different macrobenthic assemblages, multivariate analyses were carried out on fourth root transformed density data. Bray–Curtis similarity index and group average linkage, were used for the cluster analysis and non-metric multidimensional scaling (NMDS). NMDS represents relationships between multiple variables in two or three dimensions and used for visualization of similarities or dissimilarities in data and 2D and 3D results are produced together with their respective scatter plots. In this, stress measured from relationship between ranks of dissimilarities and ranks of distances.

2.4.5.2 SIMPER Analysis

Recognition of individual species contributing to the separation of two groups of samples, or the 'closeness' of samples within a group was carried out through the similarity percentages routine (SIMPER) implemented in PRIMER (Clarke and Gorley, 2006). For identifying 'characterizing species' in a particular assemblage, SIMPER calculates the average similarity (S) between all pairs of samples within a group. Because S is the algebraic sum of contribution from each species, within-group similarity can be expressed in terms of the average contribution from each variable. A good 'characterizing species' contributes heavily to intra-group similarity and has a small standard deviation. To identify 'discriminating species' between different groups of samples SIMPER calculates the average dissimilarity (d) for all pairs of inter-group samples. The analysis allowed us to determine the taxa responsible for patterns (resulting from cluster analysis and NMDS) and any differences between groups of sites. Different groups obtained from the results of cluster analysis, the species having the greatest contribution to this each group, were determined using similarity percentage tool SIMPER.

2.4.5.3 ANOSIM Analysis

One way analysis of similarity (ANOSIM, with 999 permutations), which tests for the differences between groups of (multivariate) samples, from different periods or locations were carried out, to define the differences between sites based on macrobenthic densities. The output statistic 'R' is said to be 0 if there is no separation of community structure and 1 if perfect separation occurs (Clarke and Gorley, 2006).

2.4.5.4 Canonical correspondence analysis

Canonical correspondence analysis (CCA) is a multivariate method to elucidate the relationships between biological assemblages of species and their environment. It is the combination between CA and multiple regressions. CCA maximizes the correlation between species scores and sample scores and the sample scores are constrained to be linear combinations of explanatory variables. CCA triplot simultaneously displays three pieces of information: samples as points, species as points, and environmental variables as arrows. Monte Carlo permutation test (with forward selection) was used to test the significance of environmental variables explained the variance of species distribution and abundance ($P < 0.05$ level).

In the present study CANOCO software (version 4.5) was employed to know the most influencing environmental variable with the species of dominant macrobenthic fauna, Polychaeta (in Chapter 3). It also tries to understand the relationship between the amphipod density and the measured environmental variables (in Chapter 4).

2.5 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 $\geq$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 <0.24 indicates the poor environmental condition (Dauvin and Ruellet, 2009). The index was proposed as an adaptation of the BOPA index (Dauvin and Ruellet, 2007), adding other annelids, namely Oligochaeta and Hirudinea (Clitellata), to the opportunistic polychaete species. As the Clitellata are very common in estuarine waters, index proposed by adding this group to the opportunistic Polychaeta.

*****$$*****

Chapter 3

SPATIO-TEMPORAL VARIATION OF MACROBENTHIC COMMUNITY IN THE COCHIN ESTUARY AND ADJACENT COASTAL WATERS

Contents

3.1 INTRODUCTION

The distribution of organisms and their interaction with the environmental variables in each habitat have an imperative role in ecology. Biotic communities in estuaries is often influenced by fluctuations in the physico-chemical factors at various spatial and temporal scales (McLusky and Elliott, 2004). These fluctuations are primarily brought about by the cyclic changes associated with seasons. Estuaries are naturally a highly variable ecosystem which exhibits wide fluctuations in the hydrographic parameters. Environmental variability in estuaries provides capability to the biota to attain stability by adapting to the alternation in the environment

(Elliott and Quintino, 2007). Temporal variability, the changes occurring in the communities through time is often used to evaluate the stability of the aquatic ecosystems. Ability of organisms to cope with the physical environment and in turn their interactions with the biotic and abiotic variables enable them to form community assemblages (Cadotte et al., 2013). Ecological studies are important in procuring information on the structure and function of an ecosystem through the assessment of the inhabiting communities in temporal and spatial scales.

Estuaries and coastal waters are endowed with complex and dynamic aquatic environments (Morris, 1995), having high biological productivity and rich resources. Estuaries, the transitional zone between the terrestrial and marine environments, have vital role in the global carbon balance (Smith and Hollibaugh, 1993), nutrient recycling (Fisher et al., 1982), provide habitat, shelter and nourishment for migratory and resident species and serve as fisheries resources, navigation routes, harbors, recreational purposes, etc. (Kennish, 1991; Paerl et al., 2006). CE belongs to the tropical regime and seasons are classified based on the availability of monsoonal rainfall, i.e., pre-monsoon (PRM-February to May), with little or no rainfall and with a domination of the tidal currents, monsoon (MN-June-September), characterized by heavy monsoonal rainfall and associated runoff, and post-monsoon (PM-October-January) with less precipitation and river discharges and with a gradual progression of seawater into the estuary. Dynamic nature of the CE is primarily attributed to the short-term changes caused by tides and the seasonal changes induced by the regional climate (Madhupratap and Rao, 1979; Iriarte and Purdie, 1994). In estuarine systems, the large scale fluctuations in the physico-chemical variables frequently affects the abundance, distribution and community structure of the estuarine biota and causes prominent spatial and temporal heterogeneity (Collins and Williams, 1981; Laprise and Dodson, 1994). In the CE also, the seasonal cycles induced by the monsoonal climate causes alterations in the hydrographical parameters, nutrient levels and sediment characteristics (Nair et al., 1993; Madhu et al., 2007; Martin et al., 2011), and also in the abundance and biomass of the primary and secondary producers, thereby affecting the food-web dynamics of the system (Qasim, 1972; Devassy and Bhattathiri, 1974; Wellershaus, 1974; Madhupratap, 1987). Hence, it is essential to understand the short-term and long term environmental changes occurring concurrent to the seasonality of the CE and also its subsequent influence on the pelagic and benthic biota.

Macrobenthic communities are principal components of estuarine and coastal environments, involved in the secondary production and nutrient exchange between the pelagic and benthic realm (Snelgrove, 1998). They act as indicators of environmental alterations brought about by the natural and anthropogenic activities (Danulet, 2002). Variations in several environmental factors, such as water mass movements, sediment deposition, salinity, turbidity, sediment grain size, total organic carbon, and anthropogenic activities (land resource management, urbanization and dredging) normally affect the macrobenthic fauna in the coastal water ecosystems (Kinne, 1966; McLusky and Elliott, 2004; Akoumianaki et al., 2013). The macrobenthic communities have been widely adopted as a tool for monitoring, evaluating the success of conservation efforts and also in the management of the health of coastal water ecosystems (Desroy and Retiere, 2004; Winberg et al., 2007; Ganesh et al., 2014).

Polychaetes, the dominant, diverse and ecologically essential functional benthic faunal components of the coastal ecosystems, have high stability and adaptability to a wide variety of habitats (Simboura et al., 2000; Santos et al., 2005). Polychaetes are involved in major functional roles such as recycling, bioturbation of marine sediments and in the burial of organic matter (Hutchings, 1998). The possession of high diversity of feeding modes (trophic levels) within this group and their extraordinary ability to adapt to a whole range of habitats and environmental variation, makes them excellent indicators of species richness and community patterns in benthic invertebrate assemblages (Fauchald and Jumars, 1979; Olsgard et al., 2003). They also indicate the long-term changes in the health of benthic communities (Papageorgiou et al., 2006). Feeding guilds of polychaetes are based on the relationships between food particle sizes, feeding habits and the motility patterns associated with the feeding (Fauchald and Jumars, 1979; Pagliosa, 2005). Studies on the ecology of feeding guilds is important to understand spatial and temporal changes in benthic communities (Heip, 1993; Wieking and Kroncke, 2003). As benthic fauna, especially polychaetes play a vital role in the trophic system by exploiting all forms of food available in the sediment and form an important link in aquatic food web structure (Crisp, 1971; Shou et al., 2009), studies on polychaete feeding ecology gets very much significant in the elucidation of trophodynamics of the estuarine ecosystems.

3.2 Background information on macrobenthic fauna

In earlier times (before AD 1900), studies on benthos were primarily based on qualitative aspects by exploring the world oceans through expeditions. After the turn of the century, studies focused on the ecological aspects, and thereafter, quantitative benthic studies were initiated (Peterson and Boysen-Jensen, 1911). Further, many comprehensive studies have been conducted on worldwide, mostly dealing with the factors influencing the benthic faunal distribution (Gerlach, 1971; Lamptey and Armah, 2008; Chao et al., 2012), spatio-temporal variability (Buchanan et al., 1978; Franz and Harris, 1988; Akoumianaki et al., 2013), community structure (Levin et al., 2000; Somerfield and Gage, 2000; Kennish et al., 2004), ecology (Maiorano et al., 2011; Chao et al., 2012; Akoumianaki et al., 2013), and feeding habits (Sanchez-Moyano and Garcia-Asencio, 2009; Antonio et al., 2012). Extensive studies have also been conducted on the macrobenthic responses to natural and anthropogenic perturbations (Gray et al., 1979; Elias et al., 2004; Neto et al., 2010; Dolbeth et al., 2011).

Pioneering information on the estuarine benthic fauna in India was generated from the Gangetic delta and Chilka Lake (Annandale, 1907; Annandale and Kemp, 1915). Later, the studies of Seshappa (1953), Ganapathi and Rao (1959) and Kurien (1967 and 1971) contributed immensely in bringing about valuable information on the benthic fauna from Indian waters. Several studies on benthic fauna have been carried out in the Indian estuaries and adjacent coastal waters pertaining to different aspects, such as, influence of environmental factors on the faunal distribution (Parulekar and Dwivedi, 1974; Jayaraj et al., 2007; Joydas and Damodaran, 2009; Sivadas et al., 2011), faunal composition and abundance

(Parulekar et al., 1975; Govindan et al., 1983), ecology and production of benthos (Parulekar et al., 1976; Nair et al., 1984; Musale and Desai, 2011), and anthropogenic impacts on the benthic community (Ansari et al., 1986; Nandan and Azis, 1995a, b; Ingole et al., 2009; Sukumaran and Saraladevi, 2009; Mandal and Harkantra, 2013; Ganesh et al., 2014).

As far as benthic studies in the CE is concerned, Desai and Kutty, (1967a, b) have carried out a detailed study in the estuary as well as its neighboring coastal waters. Most of the earlier studies on benthic fauna in the CE (Ansari et al., 1977; Pillai, 1977; Batcha, 1984; Devi et al., 1991; Sheeba, 2000; Nisha, 2008; Feebarani, 2009) provided information on the spatio-temporal distribution in relation to the environmental changes. Some studies provided information on the ecology of benthic fauna in different environments, such as mud banks (Damodaran, 1972), husk retting sites (Ambika Devi and Pillai, 1990), prawn culture fields (Aravindakshan et al., 1992), and mangrove swamps (Sunil Kumar, 1993). Apart from this, some of the studies have also addressed the influence of environmental pollution on the benthic faunal distribution and abundance in the CE (Unnithan et al., 1975; Remani et al., 1983; Devi and Venugopal, 1989; Martin et al., 2011).

After 1970s only systematic studies on the soft bottom macrobenthic community have been initiated in the coastal waters of India (Parulekar and Wagh, 1975; Ansari et al., 1977; Harkantra et al., 1980; Jayaraj et al., 2007). Studies on the soft bottom macrobenthic communities have usually been conducted to investigate the impact of anthropogenic perturbations (Pearson and Rosenberg, 1978). Studies on polychaetes have been extensively carried out in the coastal waters as a tool for environmental monitoring (Crema et al., 1991; Elias, 1992; Grall and Glemarec, 1997; Solis-Weiss et al., 2004) and for assessing the organic enrichment (Zajac and Whitlatch, 1982; Gray et al., 2002; Elias et al., 2005; Martin et al., 2011), heavy metal (Bryan and Langston, 1992; Gibbs et al., 2000; Rhee et al., 2007) and pesticide pollution (Briggs et al., 2002; Timmermann and Andersen, 2003).

The feeding-guild approach seems to be an effectual method for understanding the environmental constraints on the trophic structure, mobility and morphological mode of food acquisition of polychaetes (Magalhaes and Barros, 2011). Studies on the feeding guild composition of polychaetes in Indian coastal waters are very few (Ganesh and Raman, 2007; Jayaraj et al., 2008; Abdul Jaleel, 2012). Majority of the earlier studies conducted in the CE have mostly considered the spatial variability in comparison to the temporal variations in the community structure. Thus, the present study depicts the spatial and temporal variability of the macrobenthic fauna through continuous monthly sampling in two distinct environments, the CE and adjacent coastal waters. Though most of the prior studies from the CE, dealt with the community ecology of the dominant benthic taxon, Polycheata, very little information is available on their functional ecology (feeding ecology).

With this background, the major objectives of the study were (1) to understand the spatio-temporal distribution of the macrobenthic community (2) to acquire information on the feeding ecology of Polychaeta in the CE and the adjoining coastal waters.

3.3 Sampling strategy and methods

Monthly sampling (January 2011-December 2011) was conducted to study the macrobenthic community and also the feeding ecology of polychaetes of the CE (Fig. 3.1). A total of eleven stations were fixed for sampling and among them, 7 stations were in the CE and the remaining 4 stations were in the adjacent coastal waters.

Seasonal samplings were categorized based on the monsoonal precipitation and runoff in the regions in and around Cochin. Pre-monsoon was characterized by warm environmental conditions without much rainfall and runoff; Monsoon was the period of heavy rainfall and associated run off and the Post-monsoon was the transitional period with intermediate rainfall and run off.

Bottom water samples were collected to study the hydrographical parameters, such as temperature, suspended particulate matter (SPM), salinity, pH, dissolved oxygen (DO), biological oxygen demand (BOD) and inorganic nutrients and the measurements and analysis were carried out following standard protocols and using properly calibrated instruments. Sediment samples and benthos were collected using a Van-veen grab. Detailed taxonomic identification of the macrobenthic fauna was carried out microscopically based on the available literatures. Relevant statistical analyses (one way analysis of variance (ANOVA), t-test and Redundancy analysis) were also performed using available statistical softwares and packages (detailed in Chapter 2).

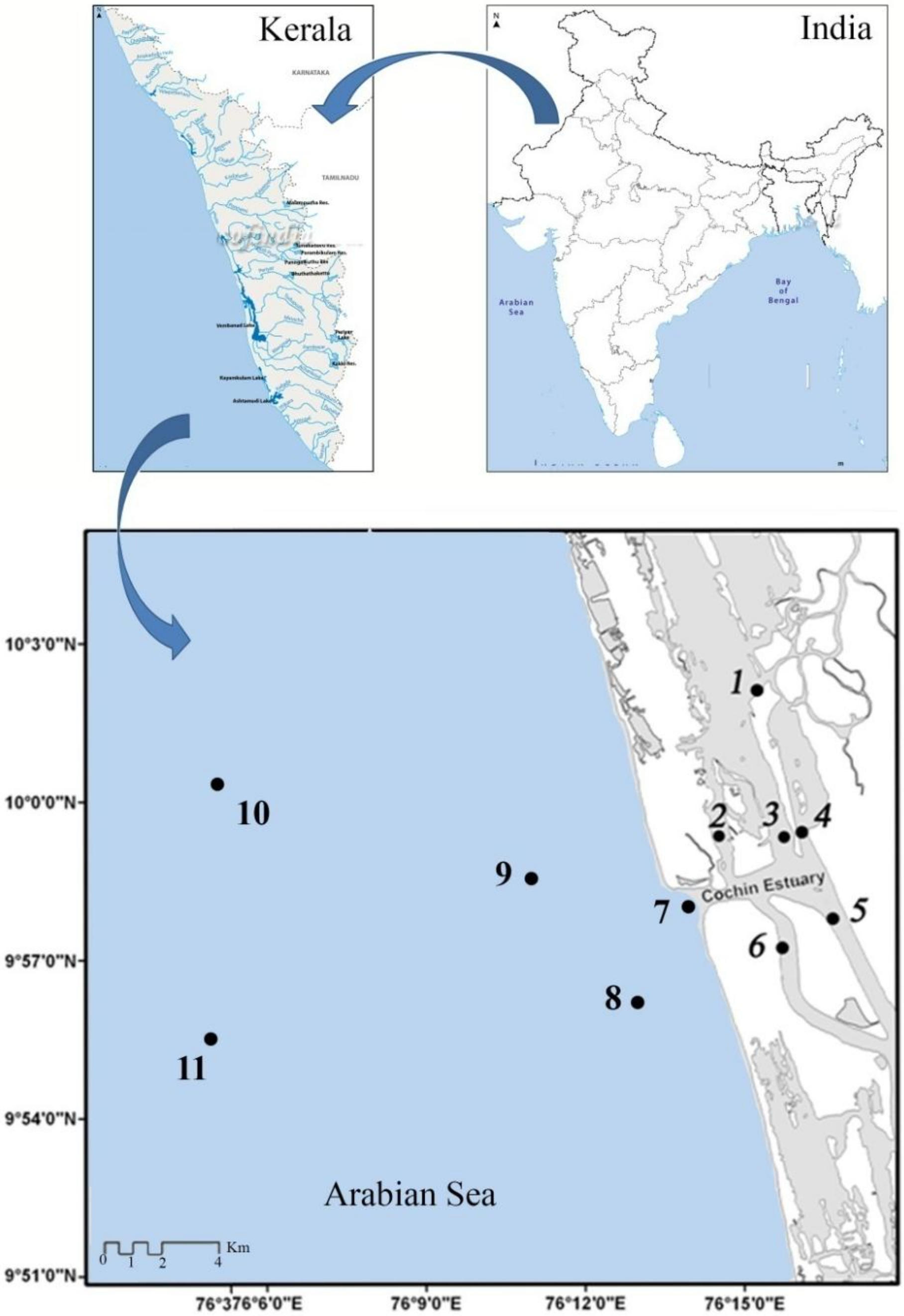


Figure 3.1 Study area and sampling locations

3.4 RESULTS

3.4.1 Environmental parameters

3.4.1.1 Rainfall

Rainfall in the Cochin area varied from 26.3 to 897.5 mm throughout the sampling period. Higher rainfall was recorded during MN (av. 659.05±163.1mm) whereas relatively low rainfall was recorded during PRM (av.130.9±86.1mm) and PM (av. 85.2±64.9mm) seasons, respectively (Fig 3.2)

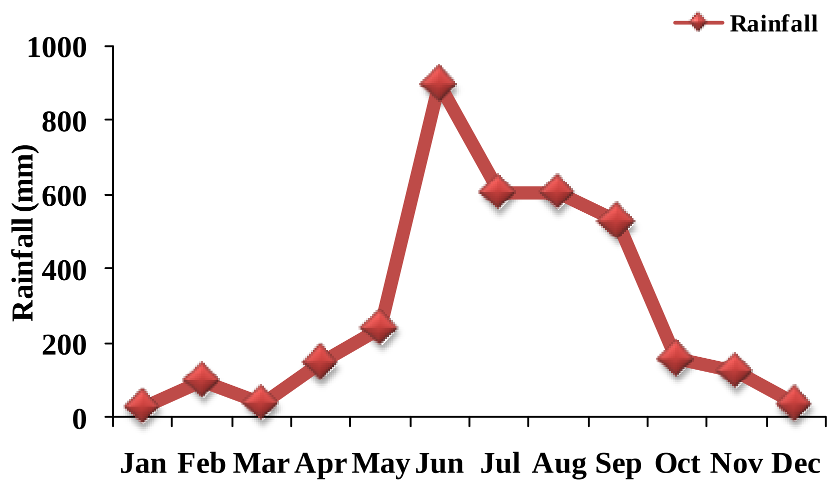


Figure 3.2 Seasonal distribution of rainfall in Cochin during 2011

3.4.1.2 Temperature

Temperature in the bottom water ranged between 26 and 33°C in the estuary throughout the year. Relatively low bottom water temperature was recorded during MN (av. 28.46±1.6 °C) and higher during PRM (av. 30.33±0.9 °C) (Fig. 3.3a). During PRM, bottom temperature varied between 28 and 32 °C with a minimum at station 7 and maximum at stations 2 and 5. During MN, temperature of the bottom waters showed high spatial fluctuation (26-33°C), with a minimum at station 4 and maximum at station 1. During PM, temperature varied between 26.5 and 30.5 °C (Fig. 3.4). Seasonal and spatial variation in the bottom temperature was statistically insignificant ($p>0.05$) in the estuary during the study period (Table 3.1 & 3.2).

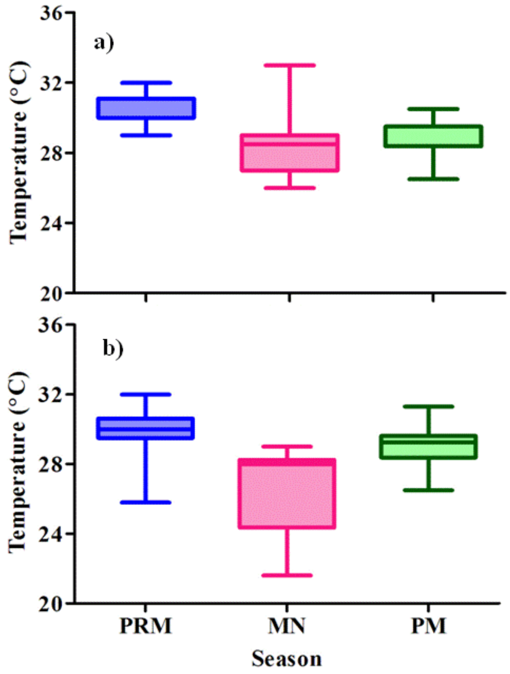


Figure 3.3 Seasonal distribution of bottom temperature in the (a) Cochin estuary and (b) adjacent coastal waters

Coastal waters exhibited relatively less bottom water temperature (21.6-31.5 °C) as compared to the estuary. Slightly lower temperature was observed during MN (av.24.86±2.9 °C) compared to PRM (av.29.20±1.6 °C) and PM (av.29.46±0.9 °C) periods (Fig. 3.3b). During PRM, temperature ranged between 25.8 and 31.5 °C with minimum at station 11 and maximum at station 9. Spatial variation was high (21.6-28°C) during

MN, with a minimum at station 12 and maximum at station 9 and station 10. During PM temperature varied between 28 and 31.3ºC with minimum at station 10 and maximum at station 9, and station 10 (Fig. 3.4). Seasonal and spatial variation in temperature was statistically insignificant ($p>0.05$) in the coastal waters during the study period (Table 3.1 & 3.2).

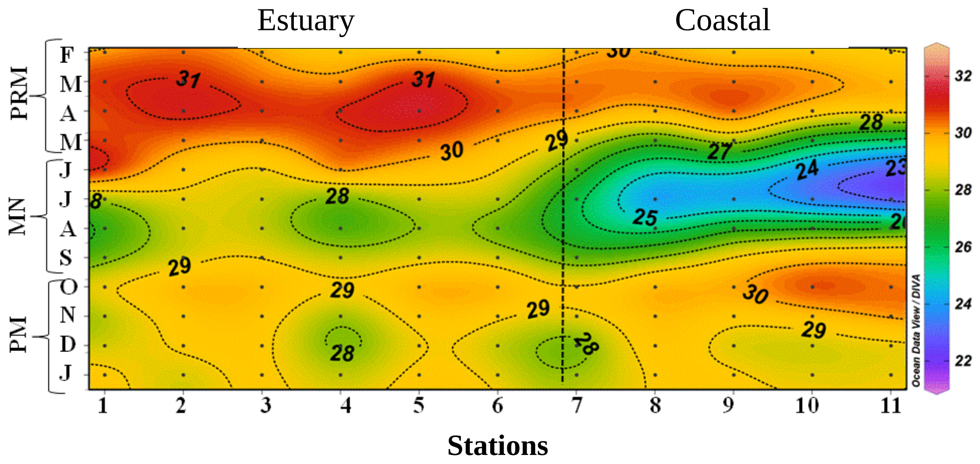


Figure 3.4 Spatial distribution of bottom temperature in the estuary (stations 1-7) and adjacent coastal waters (stations 8-11)

3.4.1.3 Salinity

Salinity of the bottom water ranged from 0 to 33.89 in the estuary with a statistically significant seasonal variation (<0.05) during the study period (Table 3.1). Relatively low salinity was recorded during MN (av. 7.47±8.7) and higher during PM (av. 18.49±9.2) (Fig. 3.5a). During PRM, salinity ranged between 1.7 and 33.89, with minimum salinity at station 1 and maximum at station 7. Spatial variation was high during MN (0 to 32) with lower salinity at station 1 and higher at station 7. During PM salinity ranged from 3.59 to 33.56, with lower salinity recorded at station 1 and higher at station 7 (Fig. 3.6). Salinity exhibited insignificant spatial variation during the study (Table 3.2).

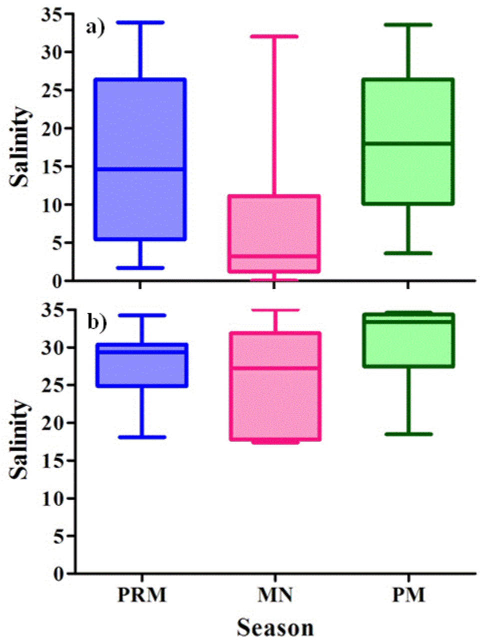


Figure 3.5 Seasonal distribution of bottom water salinity in the (a) estuary and (b) adjacent coastal waters

Compared to the CE, higher salinity was observed in the adjacent coastal waters (17.4 to 35) with less spatial variation. Lower salinity was observed during MN season (av.25.78±7.1) compared to PM (av. 30.84±4.8) and PRM (av. 28.02±4.4) (Fig. 3.5b). During PRM, salinity varied between 18.09 and 32.3 with a minimum at station 9 and maximum at the farthest station (station 11). During MN, fluctuation in salinity was high (17.41 to 35) with a minimum at station 8 and maximum at station 11. During PM, an increase in salinity was noticed (18.48 to 34.57) with minimum and maximum salinity recorded at station 11 (Fig. 3.6). Salinity exhibited statistically insignificant ($p>0.05$) spatial and seasonal variation in the adjacent coastal waters (Table 3.1 & 3.2).

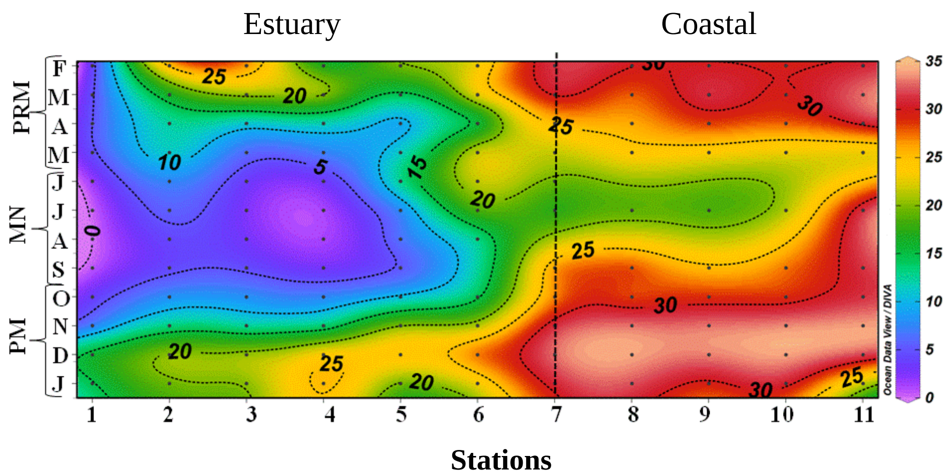


Figure 3.6 Spatial distribution of bottom water salinity in the estuary (stations 1-7) and adjacent coastal waters (stations 8-11)

3.4.1.4 pH

In the estuary, pH ranged from 6.66 to 8.46 with statistically significant ($p<0.05$) seasonal variation throughout the year. Relatively low pH was recorded during PRM (av. 7.55±0.5) and higher during PM (av. 7.87±0.3) (Fig. 3.7a). During PRM, pH exhibited distinct spatial variation (6.66 to 8.46) with a minimum at station 4 and maximum at station 2. During MN, pH varied between 6.9 and 8.1 with a minimum at station 1 and maximum at station 7. During PM pH fluctuated from 7.29 to 8.28 (Fig. 3.8). pH exhibited statistically significant spatial variation in the estuary during the study period (Table 3.1 & 3.2).

Compared to the estuary, pH was slightly alkaline in the adjacent coastal waters (7.53 to 8.45). Relatively low pH was observed during PRM (av. 8.0±0.29) and higher during PM (av. 8.3±0.1) (Fig. 3.7b). During PRM, pH ranged from 7.53 to 8.45 with minimum recorded at station 9 and maximum at station 10. pH exhibited less spatial variation during MN (7.92 to 8.29) and PM (8.09 to 8.43) (Fig. 3.8). In the coastal waters pH exhibited significant variation between seasons, while spatial variation was insignificant (Table 3.1 & 3.2).

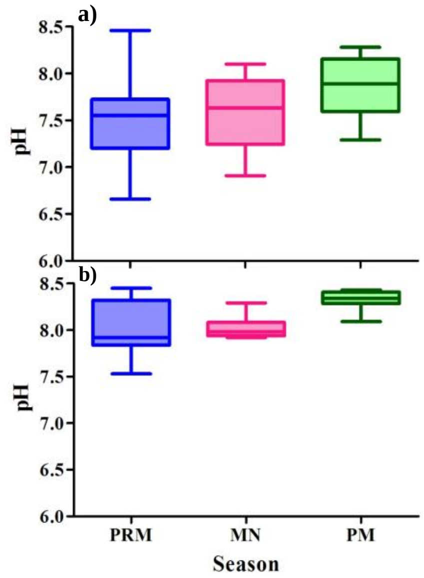


Figure 3.7 Seasonal distribution of bottom water pH in the (a) estuary and (b) adjacent coastal waters

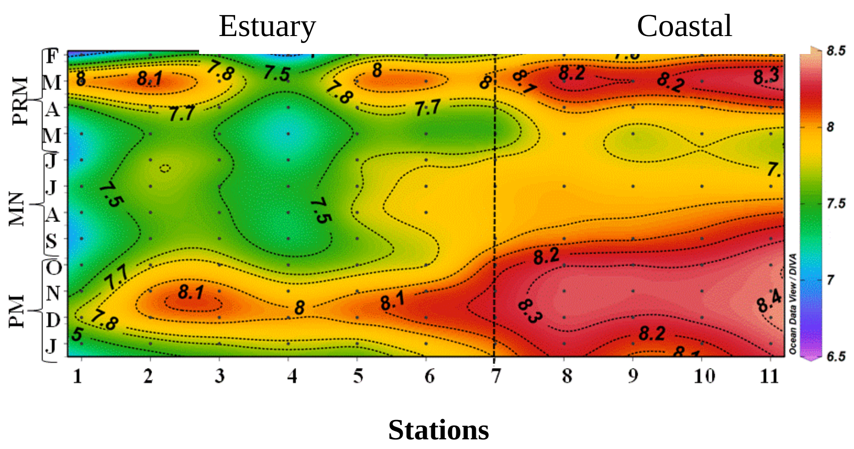


Figure 3.8 Spatial distribution of bottom water pH in the estuary (stations 1-7) and adjacent coastal waters (stations 8-11)

3.4.1.5 Dissolved Oxygen

In the estuary, DO concentration varied from 0.20 to 8.04 mg/L during the study period (Fig. 3.9a). Though DO was relatively high during PRM (av. 5.02±1.4 mg/L) compared to MN (av. 4.62±1.5 mg/L) and PM season (av.4.43±1.5 mg/L), the distinction was not statistically significant ($p>0.05$). DO concentration varied between 3.18 and 8.04 mg/L during PRM, with a minimum at station 5 and maximum at station 4. During MN, DO concentration varied from 1.14 to 6.52 mg/L with a minimum at station 6 and maximum at station 5. During PM, DO ranged between 0.20 and 7.29 mg/L with minimum concentration recorded at station 2 and maximum at station 1 (Fig. 3.10). Though DO varied spatially, the variation was not statistically significant (Table 3.1 & 3.2).

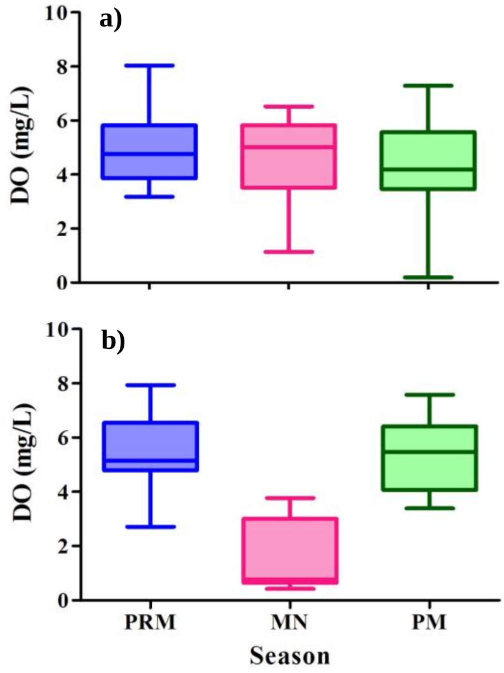


Figure 3.9 Seasonal distribution of dissolved oxygen in the (a) estuary and (b) adjacent coastal waters

Compared to the CE, DO concentration was slightly lower (0.42 to 7.94 mg/L) in the adjacent coastal waters. Relatively low DO concentration was recorded during MN (av.1.56±1.3mg/L) and higher during PRM (av. 5.46±1.3mg/L) (Fig. 3.9b). During PRM, DO concentration varied from 2.71 to 7.94 mg/L with a minimum at station 8 and maximum at station 9. A decrease in DO concentration was noticed during MN (0.42 to 3.77 mg/L). Spatially, DO concentration ranged between 3.39 and 7.58 mg/L during PM, with minimum observed at station 8 and maximum at station 9 (Fig. 3.10). Seasonal and spatial variation in DO was insignificant in the adjacent coastal waters (Table 3.1 & 3.2).

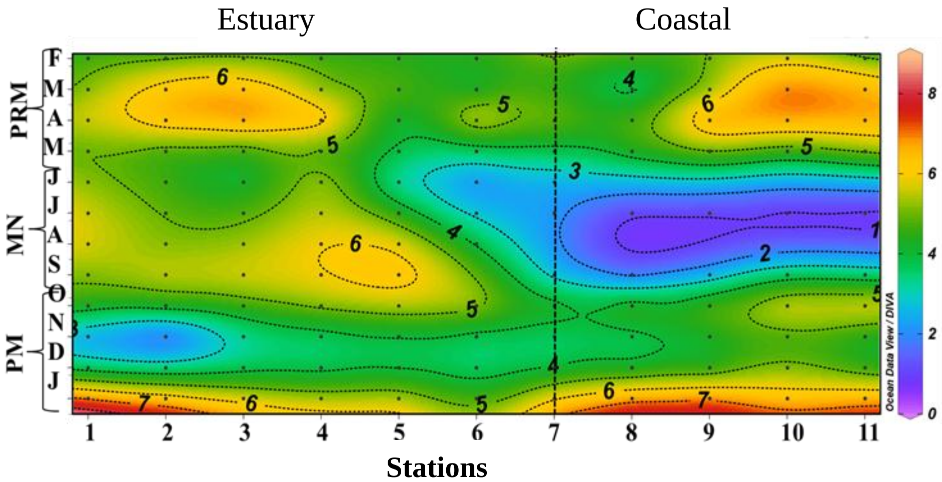


Figure 3.10 Spatial distribution of bottom water DO in the estuary (stations 1-7) and adjacent coastal waters (stations 8-11).

3.4.1.6 Biological Oxygen Demand

Biological oxygen demand varied from 0.59 to 4.60 mg/L in the estuary with significant seasonal variation ($p<0.05$) throughout the study (Table 3.1). Lower BOD concentration was observed during MN (av.1.71±0.82 mg/L) and higher during PRM (av.2.47±1.0 mg/L) (Fig. 3.11a). Spatial variation was high during PRM (0.59 to 4.60 mg/L) and MN (0.68 to 4.51 mg/L) season. During PM, BOD concentration varied from 1.56 to 2.87 mg/L with minimum concentration at station 7 and maximum concentration at station 3 (Fig 3.12). BOD concentration exhibited insignificant spatial variation in the estuary (Table 3.2).

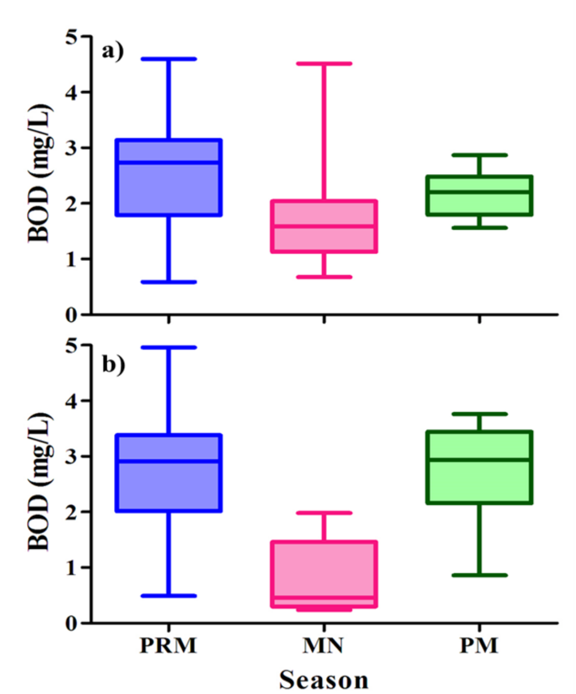


Figure 3.11 Seasonal distribution of bottom water biological oxygen demand in the (a) estuary and (b) adjacent coastal waters

BOD concentration fluctuated from 0.24 to 4.96 mg/L in the adjacent coastal waters. Higher BOD concentration was observed during PRM (av. 2.83±1.2mg/L) and lower observed during MN (av. 0. 81±0.7mg/L) (Fig. 3.11b). Spatial variation in BOD was high during PRM (0.49 to 4.96 mg/L) in the study area. A decrease in BOD was recorded during MN in the study area (0.24 to 1.98 mg/L) with lower concentration observed at station 10 and higher at station 9. During PM, BOD concentration fluctuated from 0.86 to 3.76 mg/L in the study area with minimum concentration observed at station 8 and maximum concentration at station 11 (Fig. 3.12). In the adjacent coastal waters BOD exhibited insignificant seasonal and spatial variation (Table 3.1 & 3.2).

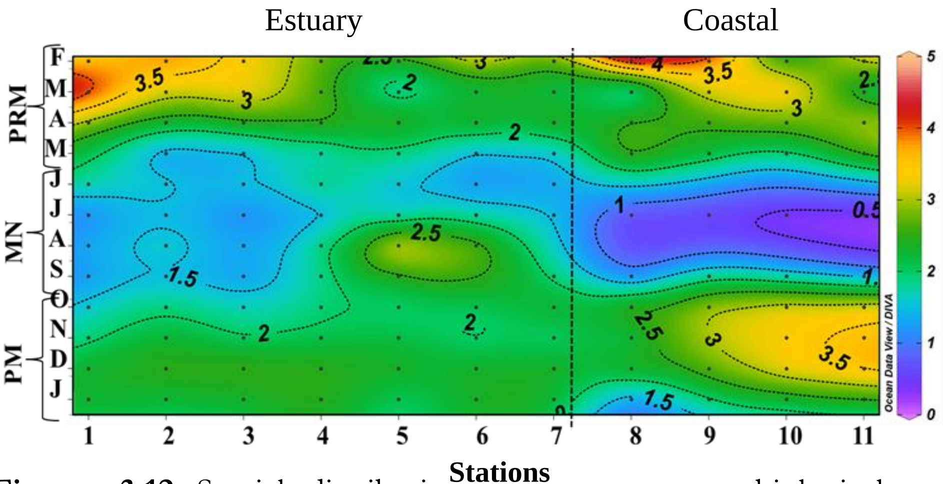


Fig.ure 3.12 Spatial distribution of bottom water biological oxygen demand in the estuary (stations 1-7) and adjacent coastal waters (stations 8-11)

3.4.1.7 Suspended Particulate Matter

In the estuary, SPM concentration varied from 4.4 to 155.60 mg/L during the study period (Fig. 3.13a). Lower concentration of SPM was observed during PRM (av.41.24±27.7mg/L) and higher during PM (av.52.00±35.1mg/L), but the variation was statistically insignificant ($p>0.05$) between seasons (Table 3.1). SPM ranged from 13.1to 120.8 mg/L during PRM, with lowest concentration at station 4 and highest at station 7. Spatial variation was prominent during MN (4.4 to 155.60 mg/L), with a minimum at station 1 and maximum at station 7. SPM concentration varied from 9.60 to 151.20 mg/L during PM, with lower concentration recorded at station 1 and higher at station 7 (Fig. 3.14). SPM exhibited insignificant spatial variation in the study area (Table 3.2).

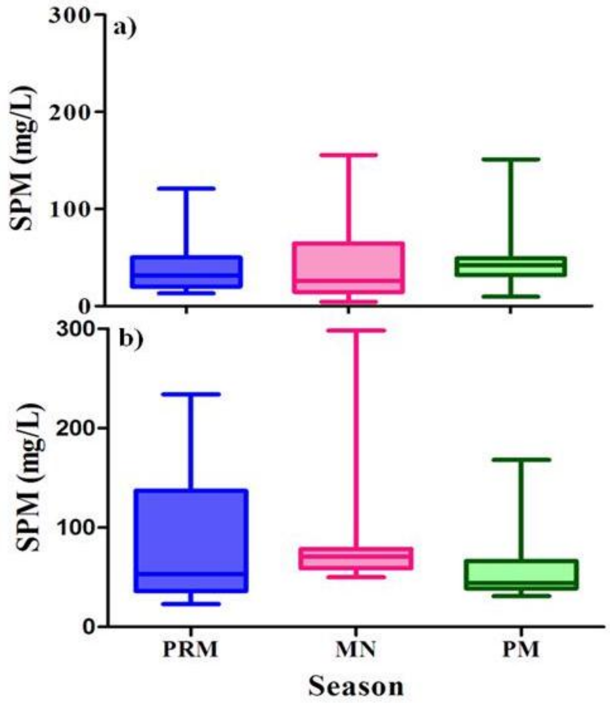


Figure 3.13 Seasonal distribution of bottom water suspended particulate matter in the (a) estuary and (b) adjacent coastal waters

In the coastal waters SPM ranged from 23.20 to 298.40 mg/L. Seasonal variation in SPM was insignificant ($p>0.05$) with lower concentration observed during PM (av. 58.23±34.5mg/L) compared to MN (av. 86.50±67.5mg/L) and PRM (av.86.38±67.1mg/L) (Fig. 3.13b, Table 3.1). Spatial variation was high during MN (50.0 to 298.40 mg/L), compared to PRM (23.20 to 234.00 mg/L) and PM (31.20 to 168.0 mg/L) (Fig. 3.14). SPM concentration exhibited insignificant spatial variation in the coastal waters (Table 3.2).

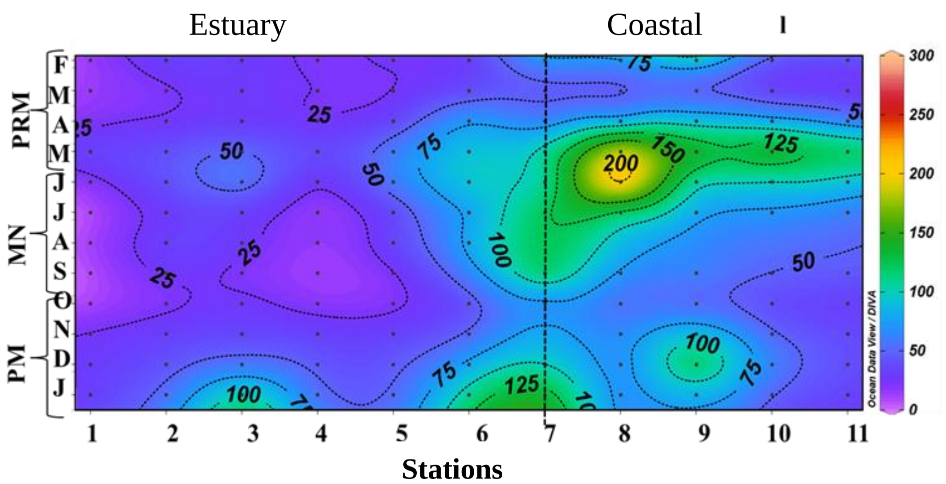


Figure 3.14 Spatial distribution of bottom water suspended particulate matter in the estuary (stations 1-7) and adjacent coastal waters (stations 8-11)

3.4.1.8 Inorganic nutrients

3.4.1.8.1 Nitrate

Nitrate ranged from 0.55 to 34.42 µM in the estuary with significant ($p<0.05$) seasonal variation (Table 3.1). Lower nitrate concentration observed during PM (av.4.81±3.4 µM) compared to MN (av. 9.69±7.4 µM) and PRM (av. 9.37±9.8 µM) (Fig. 3.15a). In the estuary, nitrate exhibited significant spatial variation during the study period (Table 3.2), with concentration varied from 0.80 to 34.42 µM during PRM, from 2.17 to 28.30 µM during MN, and from 0.55 to 15.19 µM during PM (Fig. 3.16).

Compared to the estuary less concentration of nitrate was recorded in the adjacent coastal waters (0.04 to 30.10 µM) (Fig. 3.16). Spatial variation in nitrate was high during PRM (0.04 to 30.1 µM) compared to MN (1.48 to 5.24 µM) and PM (0.43 to 5.70 µM) (Fig. 3.15b). Seasonal and spatial variation in nitrate was insignificant in the study (Table 3.1 & 3.2).

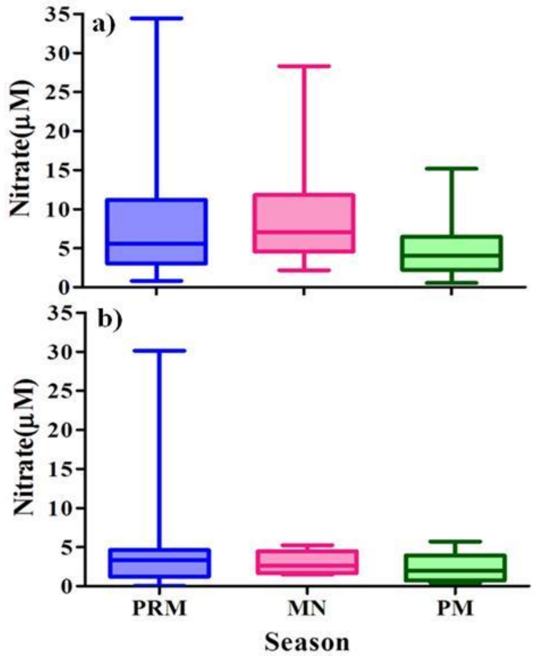


Figure 3.15 Seasonal distribution of bottom water nitrate in the (a) estuary and (b) adjacent coastal waters

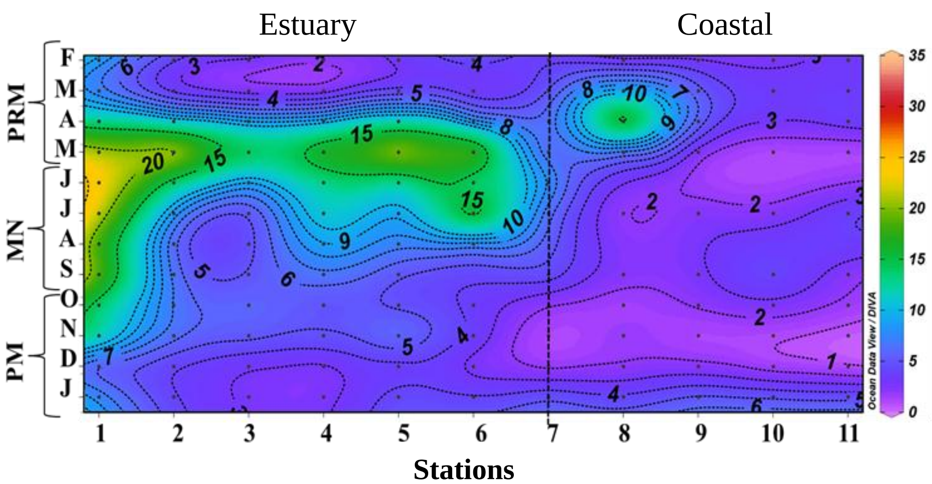


Figure 3.16 Spatial distribution of bottom water nitrate in the estuary (stations 1-7) and adjacent coastal waters (stations 8-11)

3.4.1.8.2 Nitrite

Nitrite concentration varied from 0.01 to 1.24 µM in the estuary. Nitrite concentration was relatively low during PRM (av.0.14±0.1 µM) compared to MN (av.0.63±0.3 µM) and PM (av. 40±0.2 µM) (Fig. 3.17a). During PRM, nitrite varied from 0.01 to 0.51 µM with a minimum recorded at station 3 and station 5 and maximum at station 4. Nitrite ranged between 0.26 and 1.24 µM during MN season with minimum concentration at station 4 and maximum at station 1. During PM, nitrite varied from 0.09 to 0.83 µM with a minimum at station 4 and maximum at station 1 (Fig. 3.18). Seasonal and spatial variation in nitrite was insignificant in the estuary during the study period (Table 3.1 & 3.2).

In the coastal waters, nitrite varied from 0.006 to 1.86 µM during the study period. Similar to estuary, nitrite was higher during MN (av.1.08±0.6 µM) compared to other periods (Fig. 3.17b). Nitrite concentration varied from 0.01 to 0.39 µM during PRM, with a minimum at station 9 and maximum at station 11. Spatial variation in nitrite was also high during MN (0.21 to 1.86 µM) with lower concentration recorded at station 8 and higher at station 10. During PM, nitrite fluctuated from 0.01 to 0.70 µM and stations nearer to the estuary have relatively high concentration (Fig. 3.18). Nitrite exhibited insignificant seasonal and spatial variation in the coastal waters during the study period (Table 3.1 & 3.2).

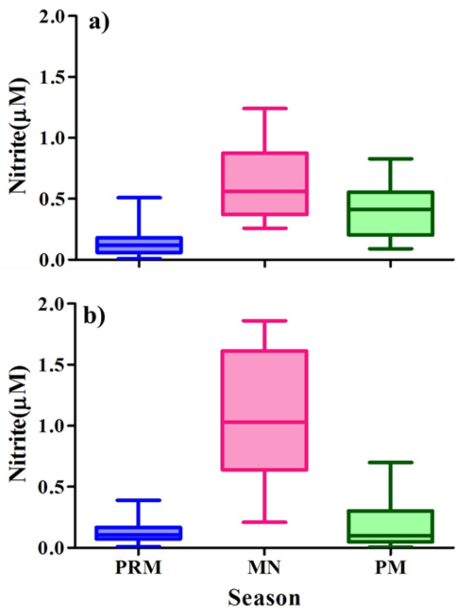


Figure 3.17 Seasonal distribution of bottom water nitrite in the (a) estuary and (b) adjacent coastal waters

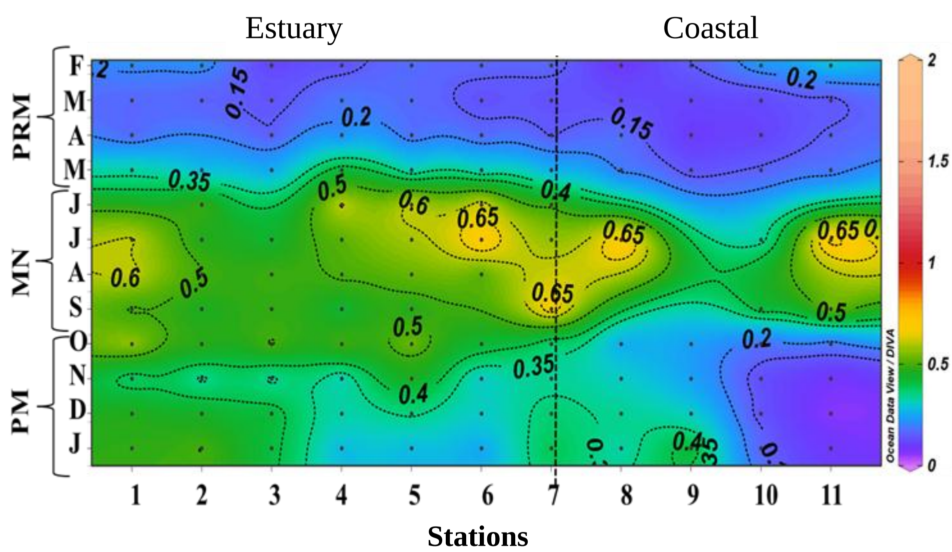


Figure 3.18 Spatial distribution of bottom water nitrite in the estuary (stations 1-7) and adjacent coastal waters (stations 8-11)

3.4.1.8.3 Ammonia

Ammonia concentration varied from 0.50 to 73.39 µM with significant seasonal variation in the estuary (Table 3.1). Lower ammonia was recorded during MN (av.6.97±3.1µM) compared to PM (av.18.54±9.6µM) and PRM (av. 15.02±17.4µM) periods (Fig. 3.19a). During PRM ammonia exhibited prominent spatial variation (0.50 to 73.39 µM), with minimum concentration was recorded at station 7 and maximum at station 4. Decrease in ammonia was observed during MN (1.81 to 14.14 µM) with lower concentration was noticed at station 2 and higher at station 5. During PM ammonia varied from 2.46 to 47.72 µM with lower concentration was recorded at station 7 and higher at station 1 (Fig. 3.20). Ammonia exhibited insignificant variation between stations (Table 3.2).

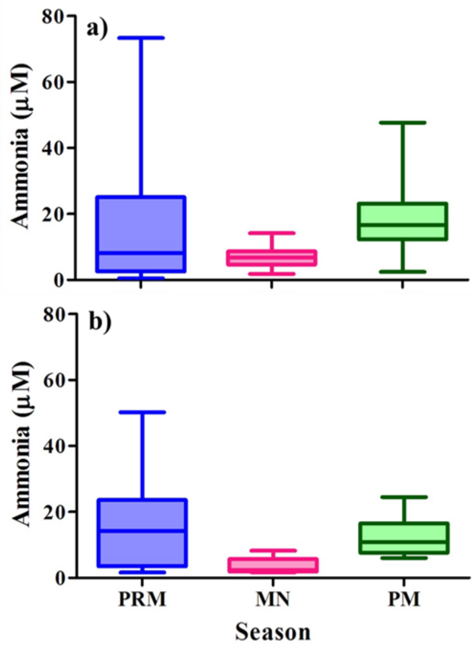


Figure 3.19 Seasonal distribution of bottom water ammonia in the (a) estuary and (b) adjacent coastal waters

Similar trend was observed in ammonia in the adjacent coastal waters (1.63 to 50.24 µM) like the estuary, but mean concentration was higher during PRM (av.16.22±13.7µM). Significant ($p<0.05$) seasonal variation (Table 3.1) was evident in the study. During PRM, ammonia varied from 1.63 to 50.24 µM with minimum concentration recorded at station 9 and maximum at station 8. During MN, ammonia ranged from 1.67 to 8.27 µM with lower concentration noticed at station 8 and higher at station 9. Ammonia varied from 6.0 to 24.50 µM during PM with minimum concentration observed at station 11 and maximum at station 8 (Fig. 3.20). Ammonia showed insignificant spatial variation in the adjacent coastal waters (Table 3.2).

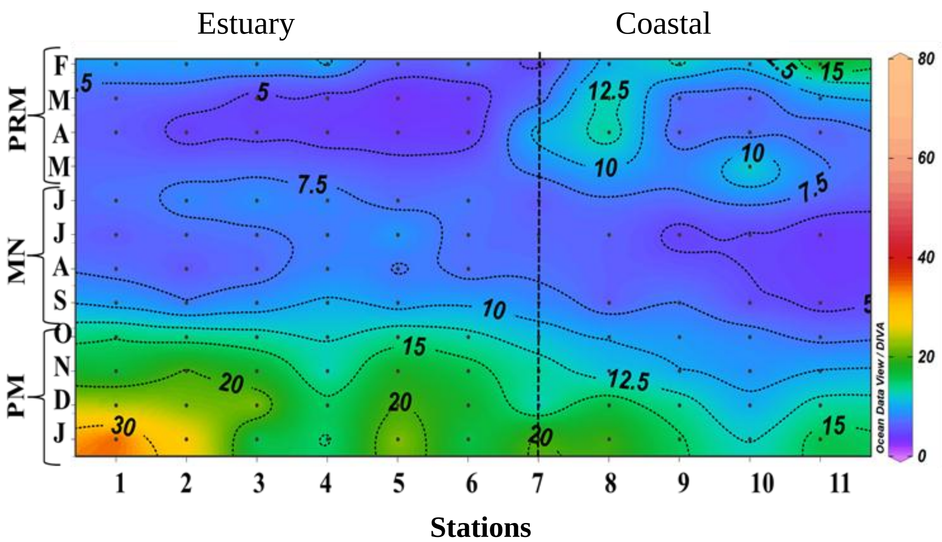


Figure 3.20 Spatial distribution of bottom water ammonia in the estuary (stations 1-7) and adjacent coastal waters (stations 8-11)

3.4.1.8.4 Phosphate

In the estuary phosphate varied from 0.10 to 2.84 µM during the study period. Among the seasons, high concentration of phosphate was observed during MN (av.1.81±0.6μM) and PM (av.1.21±0.6 μM) compared to PRM period (av. 0.53±0.3 μM) (Fig. 3.21a). During PRM phosphate concentration was < 1.2 μM, while during MN it varied from 0.43 to 2.84 μM and during PM, from 0.23 to 2.34 μM (Fig. 3.22). Phosphate exhibited insignificant spatial and temporal variation in the study area (Table 3.1 & 3.2).

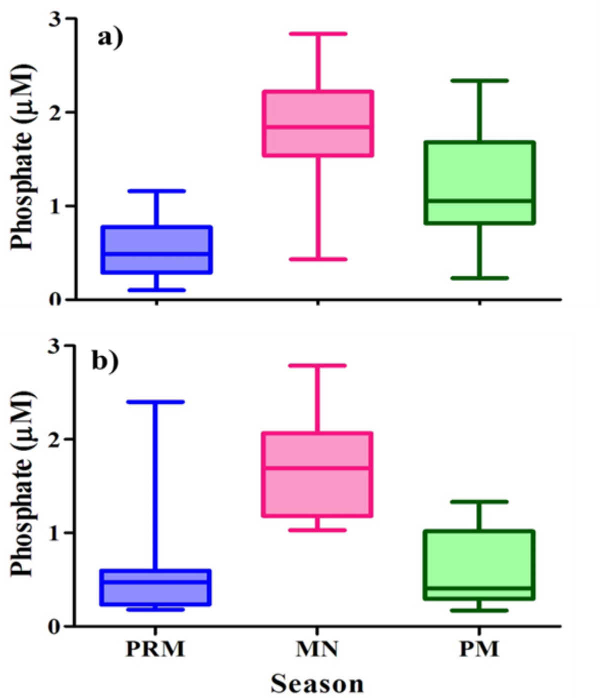


Figure 3.21 Seasonal distribution of bottom water phosphate in the (a) estuary and (b) adjacent coastal waters

Similar to estuary, insignificant temporal and spatial variation in phosphate was evident in the coastal waters (0.17 to 2.79 μM) (Table 3.1 & 3.2). Higher concentration of phosphate was observed during MN (av.1.72±0.6μM) compared to other seasons (PRM-av.0.53±0.5μM, PM-av.0.62±0.4 μM) (Fig. 3.21b). Spatial variation was high during PRM (0.18

to 2.40 µM) with minimum and maximum phosphate concentration recorded at station 10. During the study, all the stations exhibited high concentration of phosphate during MN (1.03 to 2.79 µM). During PM, Phosphate fluctuated from 0.17 to 1.33 µM with lower concentration was recorded at station 10 and higher concentration recorded at station 11 (Fig. 3.22).

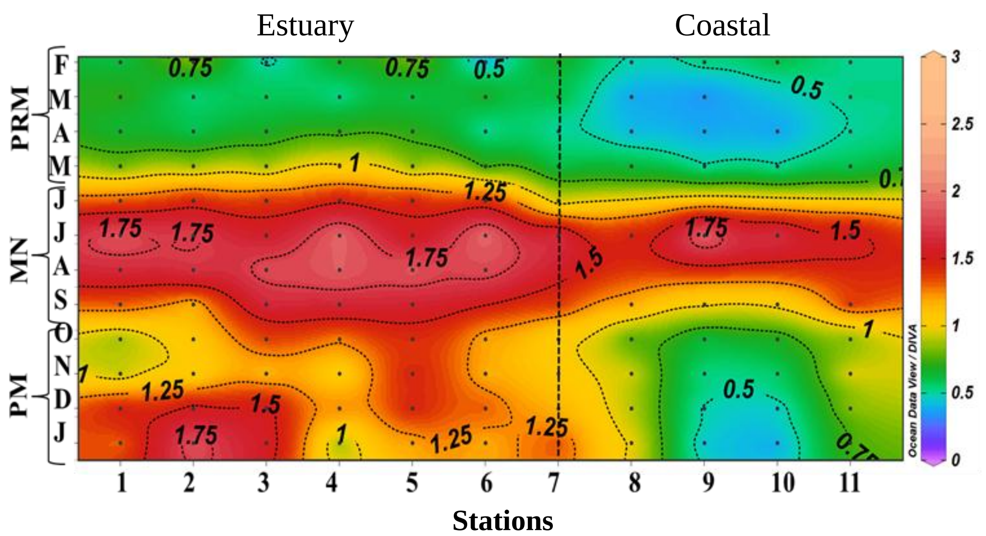


Figure 3.22 Spatial distribution of bottom water phosphate in the estuary (stations 1-7) and adjacent coastal waters (stations 8-11)

3.4.1.8.5 Silicate

In the estuary, silicate concentration (0.35 to 83.53 µM) was higher compared to the coastal waters (0.14 to 58.90 µM). Higher silicate was observed during MN season (av.36.38±21.7µM) and lower during PRM (av.14.17±7.5µM) (Fig. 3.23a). Spatial variation in silicate was high during MN (4.1 to 83.53 µM) compared to PRM (1.49 to 30.22 µM) and PM (0.35 to 36.27 µM) (Fig. 3.24). Throughout the study, maximum silicate concentration was recorded at the upstream station (station 1). Silicate exhibited no significant temporal and spatial variation during the study period (Table 3.1 & 3.2).

Similar to estuary, silicate exhibited insignificant spatial and temporal variation in the adjacent coastal waters (Table 3.1 & 3.2). Relatively high silicate was observed during PM (av.12.55±19.5µM) and lower during MN (av.7.41±5.2µM) (Fig. 3.23b). Throughout the study, silicate was observed higher at the station nearer to estuary (station 8) except during MN. (Fig. 3.24).

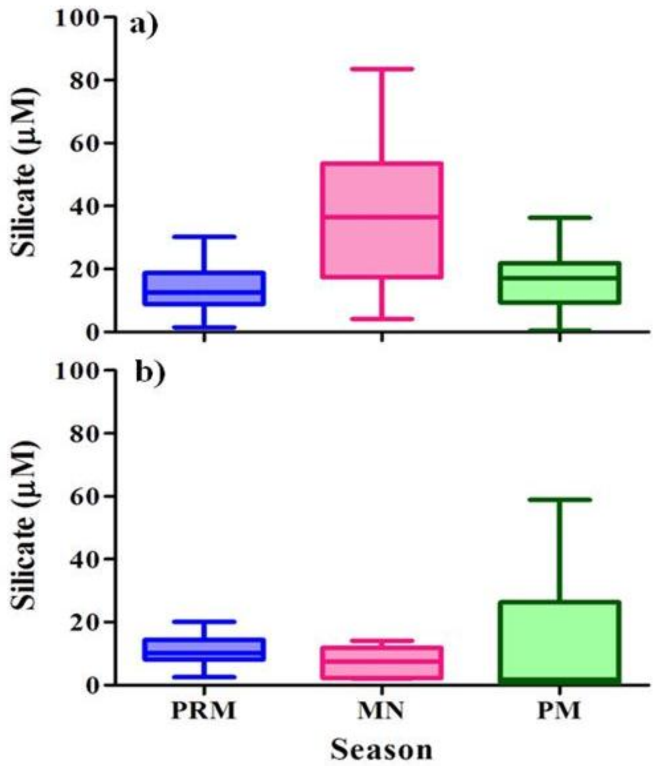


Figure 3.23 Seasonal distribution of bottom water silicate in the (a) estuary and (b) adjacent coastal waters

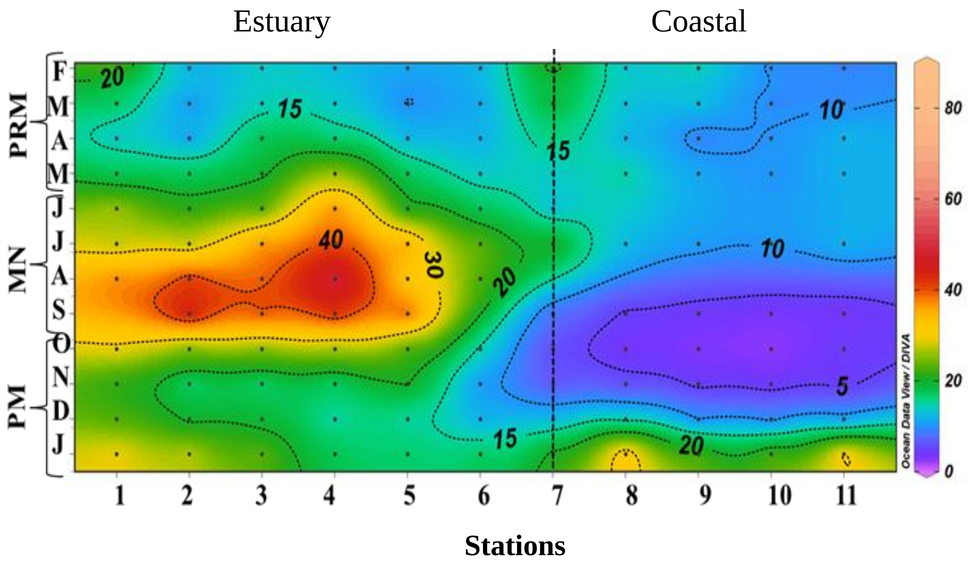


Figure 3.24 Seasonal distribution of bottom water silicate in the (a) estuary and (b) adjacent coastal waters

3.4.2 Sediment characteristics

3.4.2.1 Sediment texture

Sand varied from 0.05 to 82.30% in the estuary. Higher percentage of sand was observed during PRM (av.35.62±32.97%) and lower during MN (av. 27.03±22.31%). Sand varied from 0.05 to 82.3% during PRM, from 0.48 to 69.41% during MN and from 0.67 to 78.57% during PM respectively (Fig. 3.25a). During PRM, higher sand was noticed at station 7 (inlet) and lower at station 6. During MN, higher sand was noticed at the upstream station (station 1) and lower percentage was observed at station 6. During PM higher sand was recorded at station 2 (near to inlet) and lower at station 6 (Fig. 3.26). Spatial and temporal variation in sand was insignificant in the study period (Table 3.1 & 3.2).

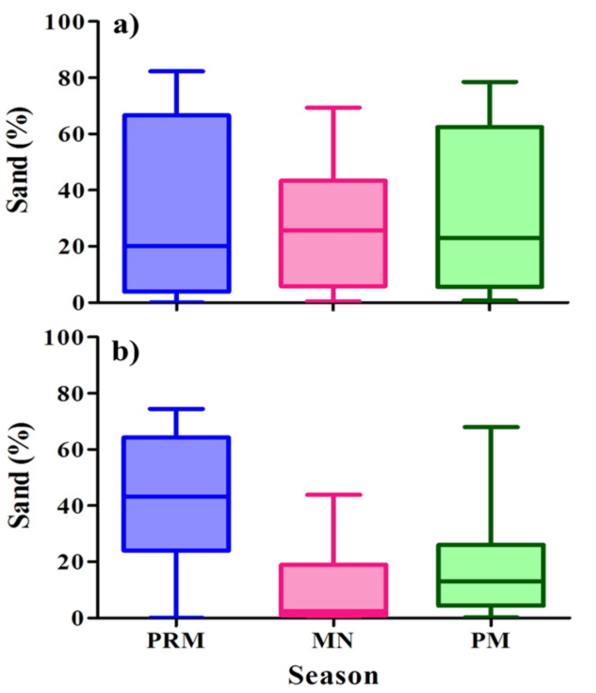


Figure 3.25 Seasonal distribution of sand in the (a) estuary and (b) adjacent coastal waters

Sand was relatively lower in the in the adjacent coastal waters (0.08 to 74.45%) than estuary during the study period. Sand fraction was higher during PRM (av.40.84±25.2%) and observed lower during MN (av. 11.11±15.4%). Sand varied from 0.08 to 74.45 % during PRM, from 0.59 to 43.87% during MN, and from 0.24 to 67.92% during PM respectively (Fig. 3.25b). Sand exhibited a significant seasonal variation, while the variation was insignificant between stations (Table 3.1 & 3.2).

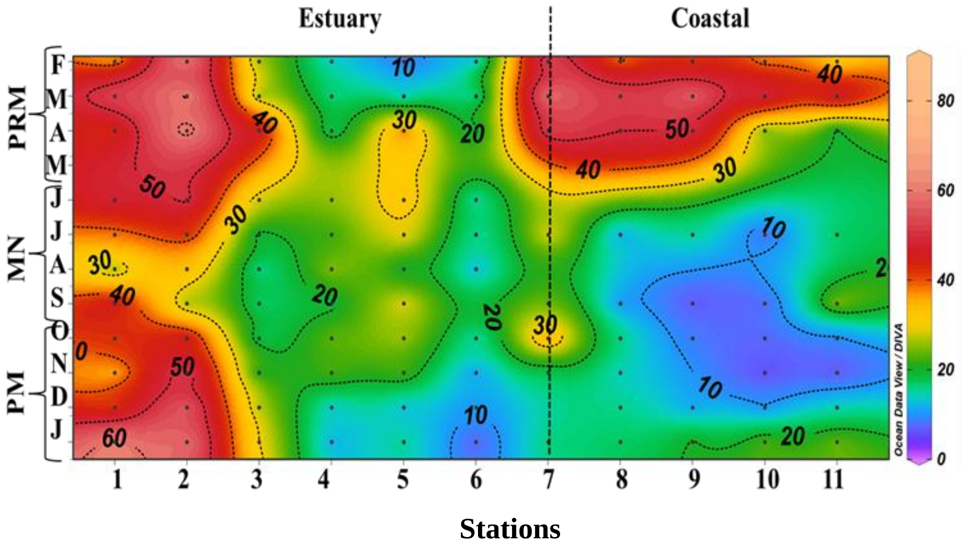


Figure 3.26 Spatial distribution of sand in the estuary (stations 1-7) and adjacent coastal waters (stations 8-11)

Silt fraction varied between 0.61 to 79.58% in the study area. Percentage of silt was higher during PM (av.30.41±23.2%) and lower during PRM (av.27.43±14.6%). Percentage of silt varied from 4.80 to 54.83% during PRM, from 1.86 to 66.99% during MN, and from 0.61 to 79.58% during PM respectively (Fig. 3.27a). Spatial variation in silt was significant in the estuary while temporal variation was insignificant (Table 3.1 & 3.2).

Silt fraction varied from 2.21 to 65.5% in the study area. Higher silt was observed during MN (av. 39.61±13.83%) and lower during PM (av. 30.93±15.03%). Silt fraction varied from 13.12 to 58.77% during PRM, from 21.59 to 65.50% during MN, and from 2.21 to 63.13% during PM (Fig. 3.27b). Seasonal and spatial variation in silt was insignificant in the study period (Table 3.1 & 3.2).

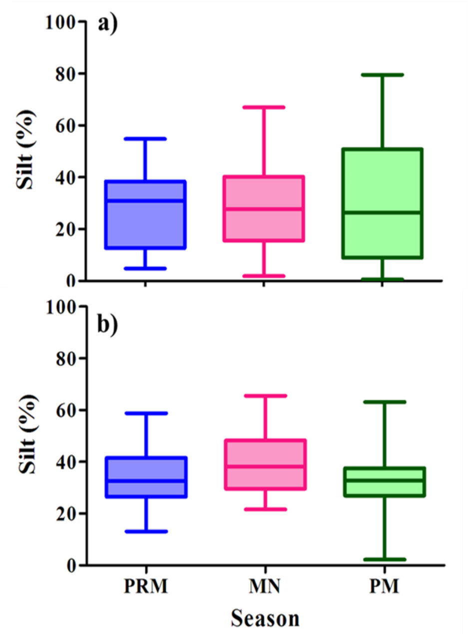


Figure 3.27 Seasonal distribution of silt in the (a) estuary and (b) adjacent coastal waters

Clay fraction ranged from 1.5 to 96.04% in the estuary. Percentage of clay was high during MN (av.43.74±20.7%) and was low during PRM (av.36.94±21.5%). The clay fraction ranged from 11.50 to 72.50% during PRM, from 16.68 to 96.04% during MN, and from 11.50 to 73.50% during PM respectively (Fig. 3.29a). Spatial and temporal variation in clay was insignificant in the study period (Table 3.1 & 3.2).

Clay fraction ranged from 0.09 to 69.88% in the adjacent coastal waters. Higher clay was observed during PM (52.52±14.2%) and lower during PRM (24.05±19.3%). Clay fraction ranged from 0.09 to 59.50% during PRM, from 28.96 to 69.88% during MN, and from 18.98 to 69.16% during PM (Fig. 3.29b). Seasonal variation in clay was significant in the coastal waters while spatial variation was insignificant (Table 3.1 & 3.2).

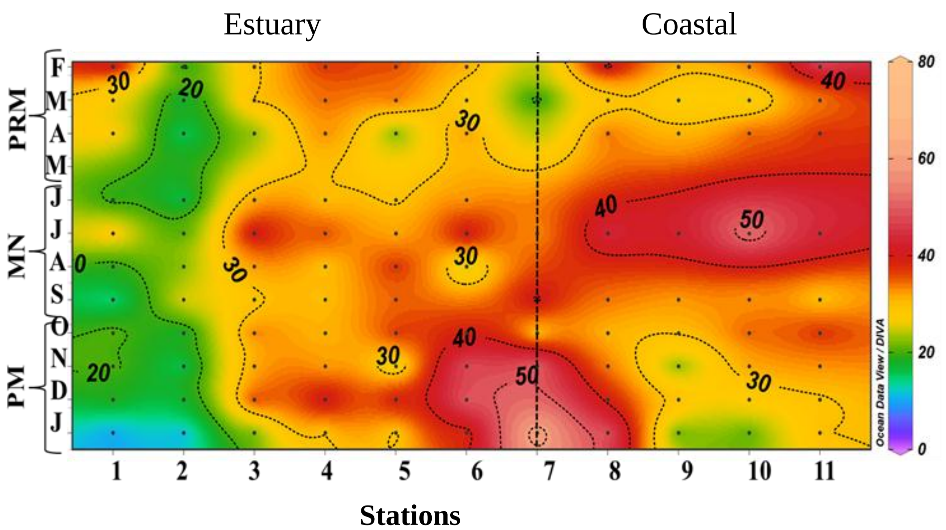


Figure 3.28 Spatial distribution of silt in the estuary (stations 1-7) and adjacent coastal waters (stations 8-11)

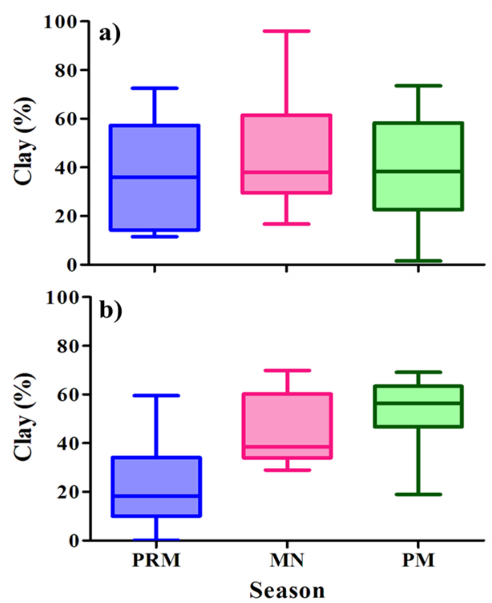


Figure 3.29 Seasonal distribution of clay in the (a) estuary and (b) adjacent coastal waters

Substratum of the station 1 and station 2 was dominated by clayey sand. Substratum of station 3, station 4, station 5, and station 6 was predominated by silt and clay particles, while station 7 was predominated by silty sand. Spatial variation in sediment texture was more pronounced in the estuary compared to the seasonal fluctuations (Fig. 3.26, Fig. 3.28, and Fig. 3.30).

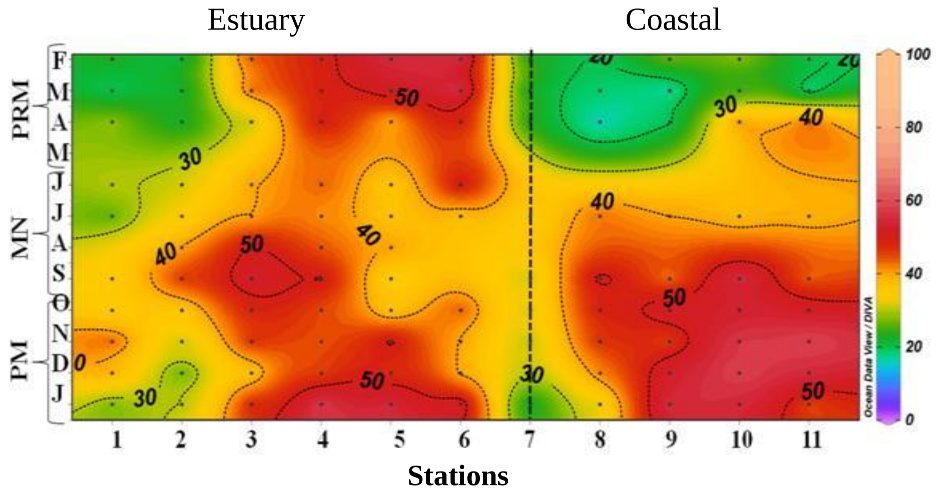


Figure 3.30 Spatial distribution of clay in the estuary (stations 1-7) and adjacent coastal waters (stations 8-11)

Substratum of station 8, station 10 and station 11 was predominated by clay and silt, while substratum of station 9 was predominated by clay and sand fractions. Seasonal variation in texture was more prominent than spatial variation in the adjacent coastal waters (Fig. 3.26, Fig. 3.28, and Fig. 3.30).

3.4.2.2 Sediment organic carbon

Sediment organic carbon ranged from 2.07 to 34.40 mg/g in the estuary. Lower organic carbon was observed during MN (av.13.06

±5.6mg/g) and higher during PRM (av.18.31±7.5mg/g) (Fig. 3.31a). Organic carbon varied from 5.44 to 31.40 mg/g during PRM, from 3.27 to 24.75 mg/g during MN, and from 2.07 to 34.40 mg/g during PM respectively. Organic carbon was high in stations having texture dominated by finer fractions of the sediment (Fig. 3.32). Low organic carbon was observed at stations having texture dominated by coarser sediment i. e, sand. Seasonal variation in organic carbon was significant in the estuary while spatial variation was insignificant (Table 3.1 & 3.2).

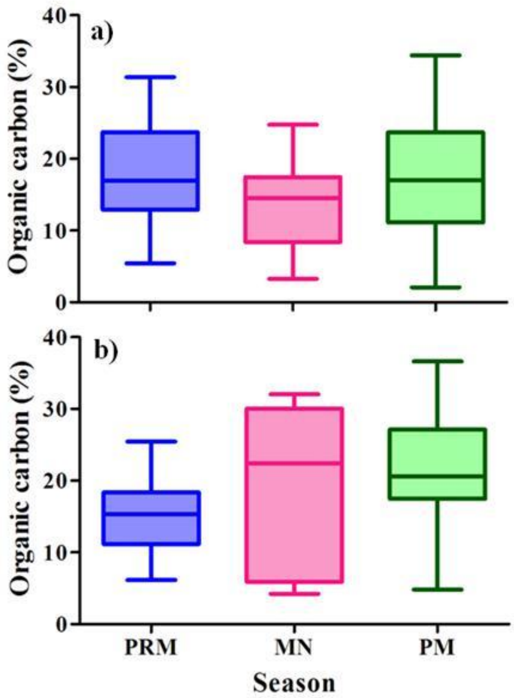


Figure 3.31 Seasonal distribution of sediment organic carbon in the (a) estuary and (b) adjacent coastal waters

Slightly higher organic carbon was observed in the adjacent coastal waters (4.21 to 36.62 mg/g) compared to the estuary. Organic carbon was relatively lower during PRM (av.15.22±5.1mg/g) and higher during PM (av.22.02±7.5mg/g) (Fig. 3.31b). Spatial variation was high during MN (4.21 to 32.06 mg/g) and PM (4.82 to 36.62 mg/g) compared to PRM (6.14 to 25.45mg/g). Temporal and spatial variation in organic carbon was insignificant in the study area (Table 3.1 & 3.2).

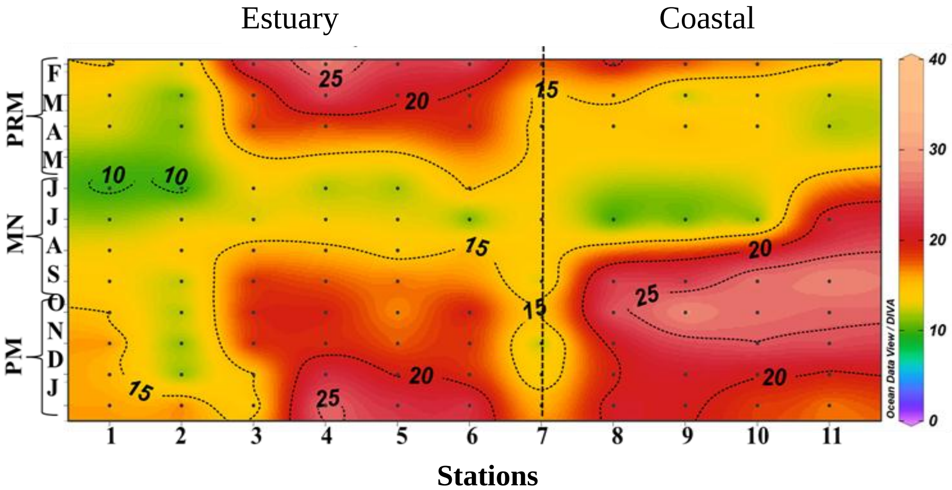


Figure 3.32 Spatial distribution of sediment organic carbon in the estuary (stations 1-7) and adjacent coastal waters (stations 8-11)

3.4.3 Macrobenthic community

3.4.3.1 Macrobenthic density

Macrobenthic density varied from 6 to 22680 ind.m^{-2} in the estuary with insignificant seasonal variation (Table 3.1). Lower density was observed during PRM (av. 1885±1769 ind.m^{-2}) and higher during PM (av. 2335±4378 ind.m^{-2}) (Fig. 3.33a). During PRM, density ranged from 40 to 7200 ind.m^{-2} with higher density at station 7 (barmouth) and lower at station 6. During MN (20 to 12600 ind.m^{-2}) and PM (6 to 22680 ind.m^{-2}) spatial variation in density was more prominent in the estuary (Fig. 3.33b). Spatial variation in macrobenthic density was significant in the estuary throughout the study period (Table 3.2).

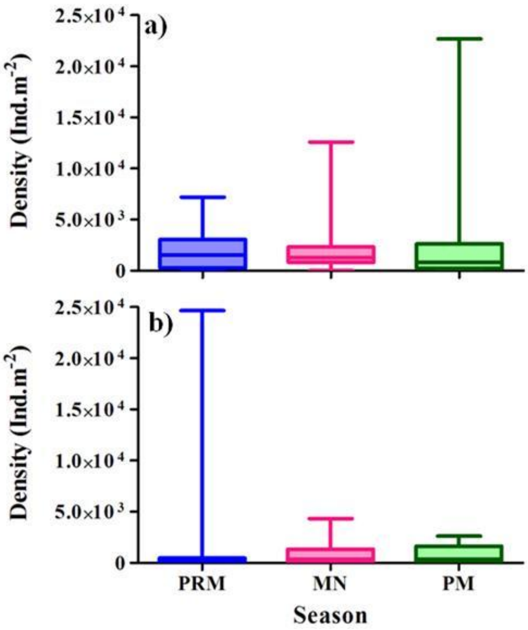


Figure 3.33 Seasonal distribution of macrobenthic density in the (a) estuary and (b) adjacent coastal waters

Relatively low density was observed in the adjacent coastal waters (14 to 2660 ind.m^{-2}) compared to the estuary. Density was higher during PRM season (av. 1814±1769 ind.m^{-2}) and lower during PM (av. 821±874 ind.m^{-2}) (Fig. 3.33b). Spatial variation was more pronounced during PRM (20 to 24660 ind.m^{-2}), with lower density at station 11 and higher at station 10. During MN, density ranged from 40 to 4320 ind.m^{-2} with minimum density recorded at the furthest station from estuary (station 11) and maximum at the station nearest to estuary (station 9). During PM, density varied from 14 to 2620 ind.m^{-2} with higher density was recorded at station 11 and lower at station 10 (Fig. 3.34). Macrobenthic density exhibited no significant spatial and temporal variation in the adjacent coastal waters (Table 3.1 & 3.2).

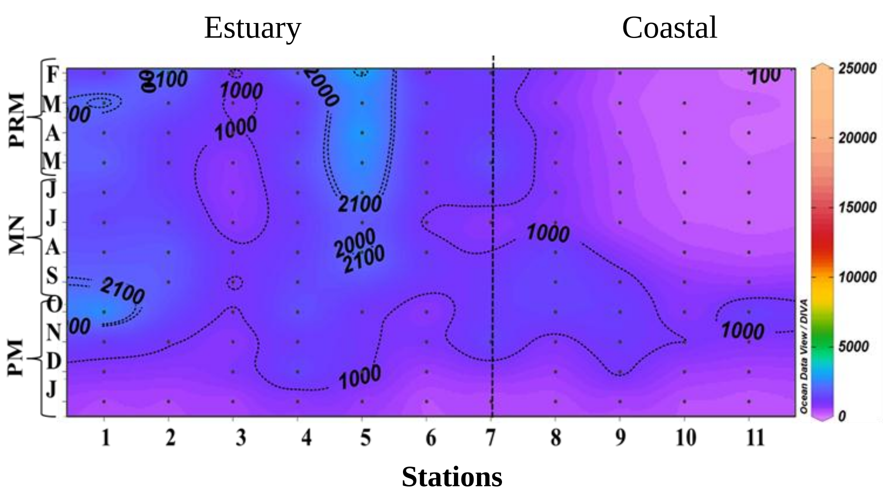


Figure 3.34 Spatial distribution of macrobenthic density in the estuary (stations 1-7) and adjacent coastal waters (stations 8-11)

3.4.3.2 Macrobenthic Biomass

Macrobenthic biomass was higher in the estuary (0.07 to 206.52 g.m^{-2}) compared to the adjacent coastal waters (0.04 to 164.98 g.m^{-2}). In the estuary, higher biomass was observed during PRM (av. 22.83±41.8 g.m^{-2}) and lower during MN (av. 8.20±11.8 g.m^{-2}) (Fig. 3.35a). Biomass varied from 0.26 to 206.52 g.m^{-2} during PRM, from 0.32 to 59.64 g.m^{-2} during MN, and from 0.07 to 200.16 g.m^{-2} during PM. Throughout the seasons higher biomass was recorded at station 7 (Fig. 3.36). Biomass exhibited insignificant temporal and spatial variation in the estuary (Table 3.1 & 3.2).

In the adjacent coastal waters, biomass was higher during PM (av. 15.29±40.39 g.m^{-2}) and lower during MN (av.3.18±4.51 g.m^{-2}) (Fig. 3.35b). Biomass varied from 0.16 to 18.40 g.m^{-2} during PRM, from 0.18 to 15.98 g.m^{-2} during MN, and from 0.04 to 164.98 g.m^{-2} during PM. Throughout the study, higher biomass was recorded at station 9 (Fig. 3.36). No significant seasonal and spatial variation was noticed in biomass during the study period (Table 3.1 & 3.2).

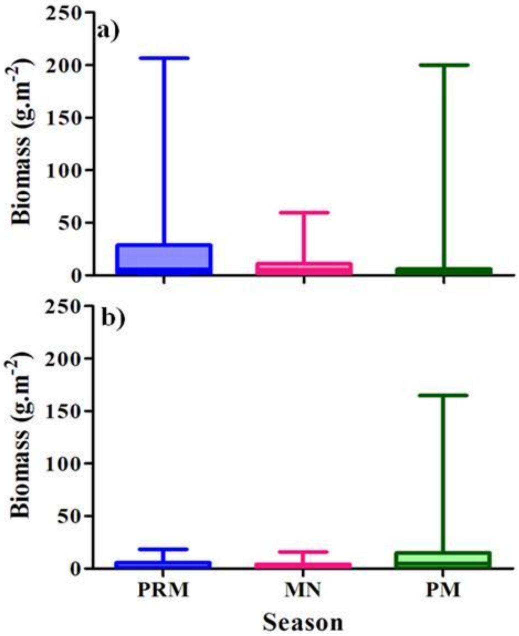


Figure 3.35 Seasonal distribution of macrobenthic biomass in the (a) estuary and (b) adjacent coastal waters

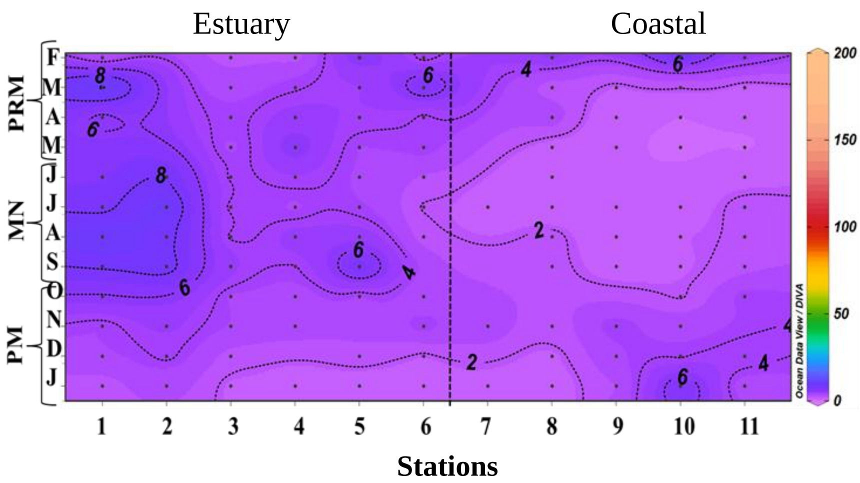


Figure 3.36 Spatial distribution of macrobenthic biomass in the estuary (stations 1-7) and adjacent coastal waters (stations 8-11)

Table 3.1 *Results of One way ANOVA of major biotic and abiotic parameters in the Cochin estuary (* - p<0.05, **- p<0.01)*

Parameter	**Seasonal**	**Spatial**
Temperature	1.20	0.45
Salinity	0.0001**	9.26
pH	0.005**	0.001**
DO	0.32	0.29
BOD	0.002**	0.99
SPM	0.48	4.41
Nitrate	0.03*	0.04*
Nitrite	8.77	0.84
Ammonia	0.003**	0.96
Phosphate	6.31	0.98
Silicate	1.36	0.12
Sand	0.58	2.71
Silt	0.84	0.01*
Clay	0.52	3.59
Organic carbon	7.07	0.04
Density	0.86	0.003*
Biomass	0.33	4.14

Table 3.2 *Results of One way ANOVA of major biotic and abiotic parameters in the adjacent coastal waters of Cochin estuary (* - p<0.05, **- p<0.01)*

Parameter	Seasonal	Spatial
Temperature	6.34	0.96
Salinity	0.08	0.86
pH	0.0001**	0.74
DO	1.45	0.60
BOD	1.77	0.84
SPM	0.30	0.32
Nitrate	0.38	0.55
Nitrite	2.53	0.34
Ammonia	0.01*	0.23
Phosphate	5.22	0.77
Silicate	0.54	0.80
Sand	0.003**	0.50
Silt	0.37	0.12
Clay	0.0002**	0.44
Organic carbon	0.10	0.82
Density	0.67	0.62
Biomass	0.26	0.38

3.4.3.3 Macrobenthic Community composition in the estuary

In the estuary, 88 macrobenthic taxa belonging to 5 phyla were encountered in the study. The estuarine macrobenthic fauna mainly comprised of polychaetes, oligochaetes, amphipods, tanaids, bivalves, gastropods, and isopods. Macrobenthic organisms that have minor contribution to the total density were considered as others which included decapods, cumaceans, mysids, chironomids, harpacticoids, brittle stars, sea anemones, and Nematodes. Polychaetes were the major macrobenthic taxa in the estuary (Fig. 3.37). During PRM, polychaetes (45.7%), oligochaetes (32.2%), amphipods (9.5%), and isopods (5.8%), were the major macrobenthic groups (Fig. 3.37a). Polychaetes (44.8%), oligochaetes (39.6%), amphipods (10.3%), dominated during MN (Fig. 3.37b). During PM, bivalves (36.0%), polychaetes (32.7%), oligochaetes (13.2%), amphipods (10.9%), and tanaids (3.7%) constituted the major macrobenthic fauna in the estuary (Fig. 3.37c). Considering the spatial distribution, at station 1, station 2 and station 7 polychaetes formed the predominant macrobenthic group, while at stations 3-6 oligochaetes constituted the major taxa.

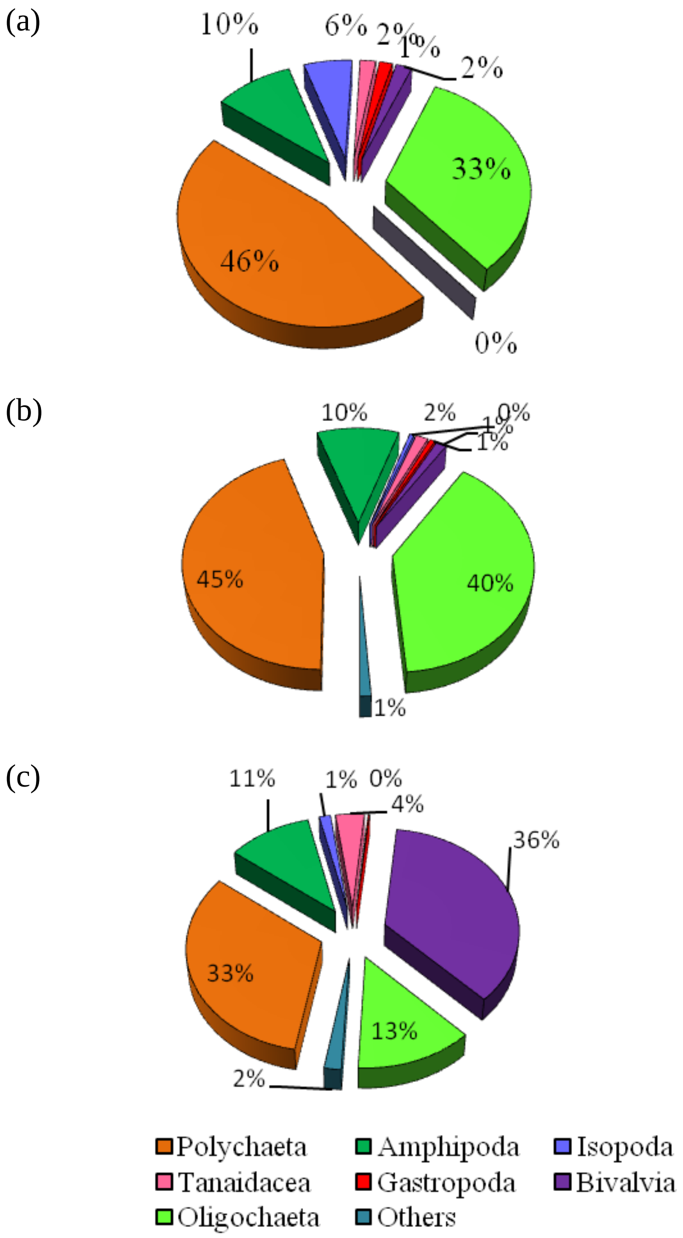


Figure 3.37 Macrobenthic community composition in CE during (a) pre-monsoon (b) monsoon and (c) post-monsoon seasons

3.4.3.4 Macrobenthic Community composition in the adjacent Coastal waters

In the adjacent coastal waters, 58 macrobenthic taxa belonging to 6 phyla were observed in the study. Five major taxa were encountered during the study period. Macrobenthos that contributed very minute abundance were considered as others which include brittle star, dentalium, and nematode etc. During PRM bivalves dominated (85%) in the coastal waters, followed by polychaetes (9%) and foraminiferans (5%) (Fig. 3.38a). Polychaetes (57%), foraminiferans (39%), and bivalves (3%) were the major taxa observed during MN (Fig. 3.38b).

During PM, polychaetes (86%) dominated in all stations followed by bivalves (8%), amphipods (3%) and cumaceans (2%) (Fig. 3.38c). At station 8 polychaetes and foraminiferans constituted the major macrobenthic groups. At station 9 and station 11 polychaetes formed the major macrobenthic groups while at station 10 polychaetes and bivalves comprised the major groups.

In the present study, macrobenthic fauna was dominated by polychaetes both in the estuary and the adjacent coastal waters. Hence detailed species composition of the polychaetes was also analyzed.

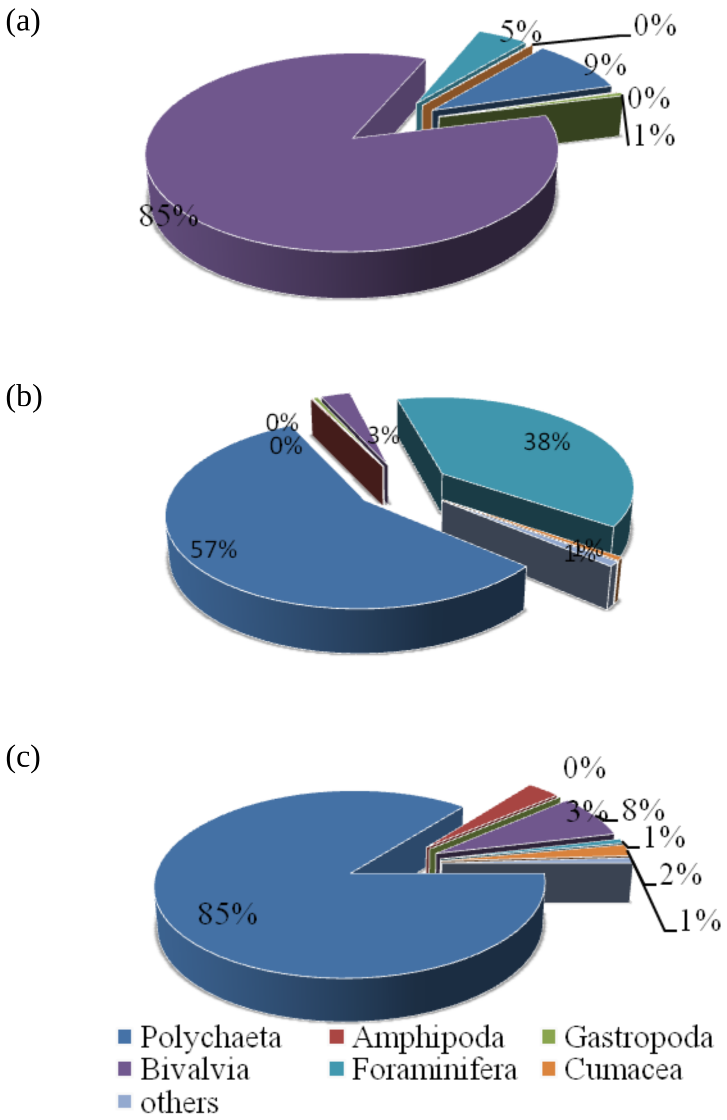


Figure 3.38 Macrobenthic community compositions in adjacent coastal waters during (a) pre-monsoon (b) monsoon and (c) post-monsoon seasons

3.4.3.5 Polychaete community composition in the estuary

During the study period a total of 25 polychaete families were observed in the CE. Among them 23 families were found during PRM, 14 during MN, and 17 families were observed during PM. Capitellidae (av. 670 ind.m^{-2}), Onuphidae (av. 580 ind.m^{-2}), Spionidae (av. 493 ind.m^{-2}), Cirratullidae (av. 248 ind.m^{-2}), Sigalionidae (av. 233 ind.m^{-2}), Pilargidae (av. 157 ind.m^{-2}), Nereidae (av. 123 ind.m^{-2}), Nephtydae (av. 65 ind.m^{-2}), Lumbrinereidae (av. 59 ind.m^{-2}), Pectinaridae (av. 32 ind.m^{-2}), Syllidae (av. 24 ind.m^{-2}), Sabellaridae (av. 24 ind.m^{-2}), and Cossuridae (av. 20 ind.m^{-2}) were the dominant families in the estuary. Twelve polychaete families such as Sigalionidae, Glyceridae, Pilargidae, Nephtydae, Nereidae, Cirratulidae, Spionidae, Capitellidae, Lumbrinereidae, Onuphidae, Cossuridae, and Pectinaridae were found throughout the study period in the estuary.

A total of 53 polychaete species were observed from the estuary. Polychaete species such as *Diopatra neapolitana* (20.8%), *Mediomastus capensis* (19.7%), *Prionospio cirrifera,* (11.7%), *Pisione sp* (8.3%), and *Sigambra parva* (5.5%), dominated in the estuary. In the estuary 46 polychaete species were observed during PRM. Polychaete species *Diopatra neapolitana* outnumbered all other species during PRM and PM while *Pisione* sp dominated during MN. During PRM *Diopatra neapolitana* (av. 292 ind.m^{-2}), *Mediomastus capensis* (av.229 ind.m^{-2}), *Sigambra parva* (av. 80 ind.m^{-2}), *Paraheteromastus tenuis* (av. 45 ind.m^{-2}), and *Prionospio cirrifera* (av. 44 ind.m^{-2}), were the major observed species. During MN and PM, 33 species of polychaetes were observed from the estuary. Species such as *Pisione* sp (av. 212 ind.m^{-2}), *Mediomastus capensis* (av. 184 ind.m^{-2}), *Prionospio cirrifera* (av. 176 ind.m^{-2}), *Caulleriella* *capensis* (av. 86 ind.m^{-2}), *Paraprionospio pinnata* (av. 81 ind.m^{-2}), *Dendronereis estuarina* (av. 50 ind.m^{-2}) and *Sigambra parva* (av. 47 ind.m^{-2}) were the major polychaetes during MN. During PM period, *Diopatra neapolitana* (av. 278 Ind.m^{-2}) *Mediomastus capensis* (av. 137 ind.m^{-2}), *Prionospio cirrifera* (av. 106 ind.m^{-2}), *Cirratulus filiformis* (av. 59 ind.m^{-2}) and *Dodecaceria* (av. 41 ind.m^{-2}) were the major species observed (Table 3.5 to 3.7).

Considering the spatial variation in the estuary, *Mediomastus capensis* formed the dominant species at station 1, station 3 and station 4. The species *Diopatra neapolitana* was observed to dominate station 2 (av. 979 ind.m^{-2}) and station 7 (av. 979 ind.m^{-2}). High density of spionid polychaetes such as *Caulleriella capensis, Paraprionospio pinnata*, and *Prionospio cirrifera* were observed at station 2 during MN. *Sigambra parva* dominated at station 5 during the entire study period. *Boccardia polybranchia* was the dominant species at station 6 and occurs in high densities during MN. In addition *Pisione sp* was observed in higher densities during MN and PM at station 7.

3.4.3.6 Polychaete community composition in the adjacent coastal waters

In the adjacent coastal waters, 20 polychaete families were observed during the study period. Among them 14 families were observed during PRM, 13 during MN, and 16 families during PM. Spionidae (av. 882 ind.m^{-2}), Cossuridae (av. 124 ind.m^{-2}), Cirratulidae (av. 55 ind.m^{-2}), Lumbrinereidae (av. 50 ind.m^{-2}), Aphroditidae (av. 45 ind.m^{-2}), Capitellidae (av. 28 ind.m^{-2}), Goniadidae (av. 27 ind.m^{-2}) and Nephtydae (av. 23 ind.m^{-2}) were the dominant polychaete families in the adjacent coastal waters. Polychaetes belonging to 7 families comprising Goniadidae, Cirratulidae, Spionidae, Capitellidae, Lumbrinereidae, Cossuridae and Magelonidae were present throughout the period.

A total of 39 polychaete species were observed from the coastal waters. Species such as *Paraprionospio pinnta* (av. 850 ind.m^{-2}), *Cossura coasta* (av. 124 ind.m^{-2}), *Caulleriella capensis* (av. 51 ind.m^{-2}), *Aphroditidae sp* (av. 45 Ind.m^{-2}), *Lumbriconereis latreilli* (av. 34 ind.m^{-2}), *Mediomastus capensis* (av. 27 ind.m^{-2}) and *Goniada emerita* (av. 27 ind.m^{-2}) dominated in the coastal waters. During PRM, 18 polychaete species were observed during the study period. Species such as *Cossura coasta* (av. 75 ind.m^{-2}), *Paraprionospio pinnata* (av. 28 ind.m^{-2}), *Sternaspis scutata* (av. 11 ind.m^{-2}), and *Mediomastus capensis* (av. 11 ind.m^{-2}), were the dominant polychaete species observed during PRM. During MN, 23 species of polychaetes have been observed, of which *Paraprionospio pinnata* (av. 364 ind.m^{-2}), *Cossura coasta* (av. 20 ind.m^{-2}), *Lumbriconereis latrelli* (av. 19 ind.m^{-2}), *Spionid sp* (av. 14 ind.m^{-2}), and *Caulleriella capensis* (av. 13 ind.m^{-2}), were the dominant species. During PM, 26 species of polychaetes have been recorded in the study area, among them species such as *Paraprionospio pinnata* (av. 459 ind.m^{-2}), *Aphroditidae sp* (av. 45 ind.m^{-2}), *Cossura coasta* (av. 29 ind.m^{-2}), and *Caulleriella capensis* (av. 39 ind.m^{-2}), were dominated. 6 species were observed throughout the study period.

3.4.3.7 Other macrobenthic taxa in the estuary

Among the identified species of amphipods, *Caprella sp*, *Photis digitata, Cheriophotis megacheles, Ampelisca sp, Eriopisa chilkensis, Corophium triaenonyx, Gammaropsis sp,* and *Melita zylanica* were dominant in the estuary.

Caprella sp dominated throughout the study period except during PM, where *Photis digitata* was the other dominant species. During PRM *Caprella sp* (av.74 ind.m^{-2}), *Gammaropsis sp* (av.23 ind.m^{-2}), *Eriopisa chilkensis* (av.15 ind.m^{-2}), and *Photis digitata* (av.14 ind.m^{-2}) were the major amphipod species. While during MN *Caprella sp* (av. 81.4 ind.m^{-2}), *Corophium triaenonyx* (av.30 ind.m^{-2}), *Photis digitata* (av.14 ind.m^{-2}), *Eriopisa chilkensis* (av.11 ind.m^{-2}), *Gammaropsis sp* (av.7 ind.m^{-2}), *Perioculoides longimanus* (av.6 ind.m^{-2}) and *Leucothoe sp* (av.6 ind.m^{-2}) were the major species observed. Species such as *Photis digitata* (av.72 ind.m^{-2}), *Cheriophotis megacheles* (av. 53 ind.m^{-2}), *Ampelisca sp* (av. 49 ind.m^{-2}), *Eriopisa chilkensis* (av.19 ind.m^{-2}), *Melita zylanica* (av.14 ind.m^{-2}), and *Caprella sp* (12 ind.m^{-2}) observed to dominate during PM (Table 3.5).

Isopods such as *Anthurid* sp, and *Cirolana fluviatilis;* Tanaid *Apseudus chilkensis;* Gastropod *Littorina littorea;* Bivalves such as *Villorita cyprinoides,* and *Perna* sp*;* Oligochaete *Tubificidae* sp were the other macrobenthic fauna found in the estuary. During PRM *Tubificidae* sp (av. 656 ind.m^{-2}), *Bivalvia* sp (av. 34 ind.m^{-2}), *Apseudus chilkensis* (av. 32 ind.m^{-2}), Isopod sp (av. 21 ind.m^{-2}), *Littorina littorea* (av.10 ind.m^{-2}), *Anthurid* sp (av. 6 ind.m^{-2}), and *Cirolana fluviatils* (av. 6 ind.m^{-2}) were the other macrobenthic fauna. During MN *Tubificidae sp* (av. 752 ind.m^{-2}), *Bivalvia* sp (av. 20 ind.m^{-2}), *Apseudus chilkensis* (av. 62 ind.m^{-2}), *Littorina littorea* (av.2 ind.m^{-2}), and *Cirolana fluviatils* (av. 6 Ind.m^{-2}) were constituted the benthic fauna (Table 3.6). During PM *Perna* sp (av. 611 ind.m^{-2}), *Tubificidae* sp (av. 349 ind.m^{-2}), *Apseudus chilkensis* (av. 62 ind.m^{-2}), *Villorita cyprinoides* (av. 53 ind.m^{-2}), Tanaid sp (av. 22 ind.m^{-2}), *Bivalvia* sp (av. 15 ind.m^{-2}), and *Anthurid* sp (av. 14 ind.m^{-2}), were the minor taxa observed in the estuary. Others include organisms such as

Herpacticoid copepodes (av.34 ind.m^{-2}), Cumcea (av. 12 ind.m^{-2}), Brittle stars (av.12 ind.m^{-2}), Decapods (av. 11 ind.m^{-2}), Sea anemone (av. 5 ind.m^{-2}), Mysid (av. 4 ind.m^{-2}), Nematodes (av. 3 ind.m^{-2}), and Chironomids (av.2 Ind.m^{-2}) were observed in the estuary (Table 3.7).

3.4.3.8 Other macrobenthic taxa in the coastal waters

Among the identified species of amphipods *Gammarid sp, Cheriophotis megacheles, Platyischnopus* sp, *Ampelisca* sp*, Photis digitata,* and *Melita zylanica* were the major species observed in the adjacent coastal waters. During PRM *Gammarid* sp (av. 1ind.m^{-2})*,* was the only representative species observed, while during MN *Ampelisca sp* was the dominant species. During PM *Gammarid* sp (av.8 ind.m^{-2})*, Cheriophotis megacheles* (av.6 ind.m^{-2})*, Platyischnopus* sp (av. 5 ind.m^{-2})*, Ampelisca* sp (av. 3 ind.m^{-2})*, Photis digitata* (av. 3 ind.m^{-2})*,* and *Melita zylanica* (av. 3 ind.m^{-2}) were the observed amphipod species. In addition bivalves such as *Perna* sp (av.1534 ind.m^{-2})*, Bivalvia* sp (av. 34 ind.m^{-2})*, Villorita cyprinoides* (av.33 ind.m^{-2})*, and Pholas orientalis* (av.29 ind.m^{-2})*;* Foraminifera (av. 411 ind.m^{-2}); Cumacea (av. 21 ind.m^{-2}); and Gastropod sp (av. 14 ind.m^{-2})*,* were also observed in the study region. Others include organisms such as *Dentalium* sp (av. 1 ind.m^{-2})*,* Brittle stars (av.5 ind.m^{-2})*,* and Nematodes (av.4 ind.m^{-2}) were also observed in the adjacent coastal waters (Table 3.5 to 3.7).

3.4.3.9 Diversity indices

3.4.3.9.1 Shannon-Weiner diversity index

Species diversity index varied from 0. 90 to 3.75 in the estuary during the study period. Higher diversity was observed during PM (av. 2.81±0.67) and lower during MN (av. 2.51±0.93) (Fig. 3.41a). During PRM diversity varied from 1.58 to 3.75, with higher diversity at station 2 and lower at station 4. During MN species diversity varied from 0.90 to 3.52 with higher diversity at station 6 and lower at station 5. During PM, diversity varied from 1.47 to 3.46 with higher diversity at station 2 and lower at station 5 (Fig. 3.39a).

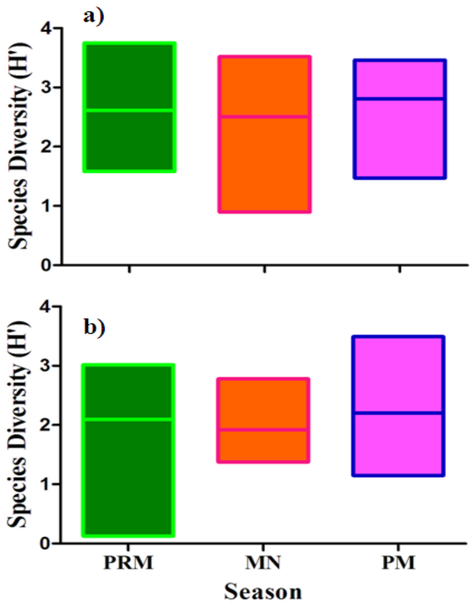


Figure 3.39 Seasonal variation of Shannon Weiner species diversity index in the (a) estuary and (b) adjacent coastal waters

In the coastal waters species diversity varied from 0.13 to 3.49 in the study period. Higher diversity was observed during PM (av. 2.20±0.97) and lower during MN (av. 1.92±0.65) (Fig. 3.41b). During PRM, diversity varied from 0.13 to 3.02 with higher diversity at station 11 and lower at station 10. During MN diversity ranged from 1.38 to 2.78 with higher diversity at station 11 and lower at station 8. During PM, species diversity varied from 1.15 to 3.49 with higher diversity at station 9 and lower at station 11 (Fig. 3.39b).

3.4.3.9.2 Margalef's Species Richness

Species richness varied from 1. 91 to 4.53 in the estuary during the study period. Higher richness was observed during PRM (av. 3.04±0.70) and lower during MN (av. 2.49±0.52) (Fig. 3.40a). During PRM richness ranged from 2.31 to 3.99, with higher richness observed at station 2 and lower at station 3. During MN species richness ranged from 1.91 to 3.26 with higher richness at station 2 and lower at station 3. During PM richness varied from 2.18 to 4.53 with higher richness observed at station 2 and lower at station 1.

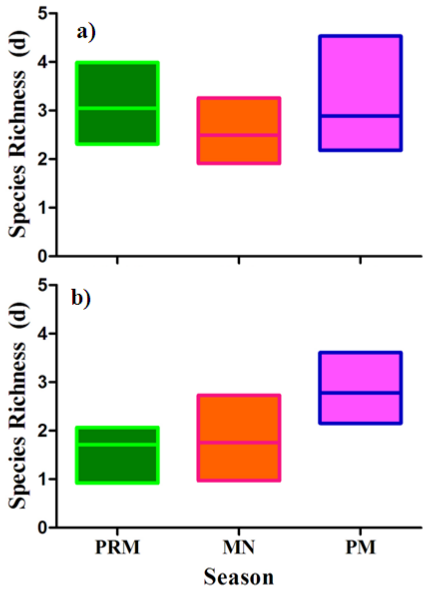


Figure 3.40 Seasonal variation of species richness in the CE and adjacent coastal waters

Species richness ranged from 0. 92 to 3.61 in the coastal waters during the study period. Higher richness was observed during PM (av. 2.77±0.72) and lower during PRM (av. 1.71±0.54) (Fig. 3.40b). During PRM richness ranged from 0.92 to 2.07, with higher richness at station 9 and lower at station 10. During MN species richness varied from 0.97 to 2.73 with higher richness at station 9 and lower at station 8. During PM, richness varied from 2.15 to 3.61 with higher richness at station 9 and lower at station 10.

3.4.3.9.3 Pielou's Evenness Index

Species evenness varied from 0.22 to 0.83 in the estuary during the study period. Higher evenness was observed during PM (av. 0.64±0.17) and lower observed during PRM (av. 0.58±0.15) (Fig. 3.41a). During PRM evenness ranged from 0.36 to 0.76, with higher evenness at station 2 and lower at station 4. During MN, species evenness varied from 0.22 to 0.83 with higher evenness at station 6 and lower at station 5. During PM, evenness varied from 0.32 to 0.82 with higher evenness at station 6 and lower at station 5.

Species evenness varied from 0.04 to 0.91 in the coastal waters during the study period. Higher evenness was observed during PRM (av. 0.60±0.39) and lower during PM (av. 0.51±0.18) (Fig. 3.41b). During PRM evenness varied from 0.04 to 0.91, with higher evenness observed at station 11 and lower at station 10. During MN, species evenness varied from 0.33 to 0.80 with higher evenness at station 11 and lower at station 9. During PM evenness varied from 0.29 to 0.73 with higher evenness observed at station 9 and lower at station 11.

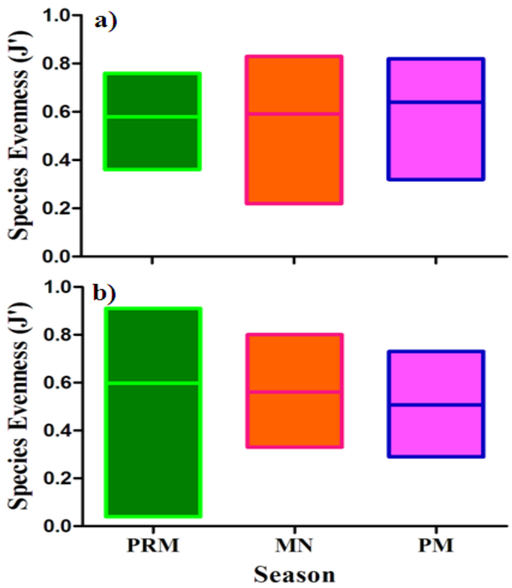


Figure 3.41 Seasonal variation of evenness in the estuary and adjacent coastal waters

3.4.3.10 Macrobenthic community structure

Bray-Curtis similarity (hierarchical clustering) based on species density categorized the estuarine stations into two groups at 40% similarity level (Fig. 3.42a). Station 1 and station 2 formed a cluster, stations.3, 4, 5, 6 formed another cluster and station 7 was positioned separately. The results of the NMDS also showed similar pattern of distinct groups of the stations (Fig. 3.42b). Throughout the study period, station 7 (inlet) was positioned as separate station apart from other estuarine stations. Seasonal variation in macrobenthic density was not prominent in the estuary during the study period while significant spatial variation was observed. Prominent disparity between the stations has been confirmed from the results of ANOSIM analysis (Global R value 0.658, p value 0.1%).

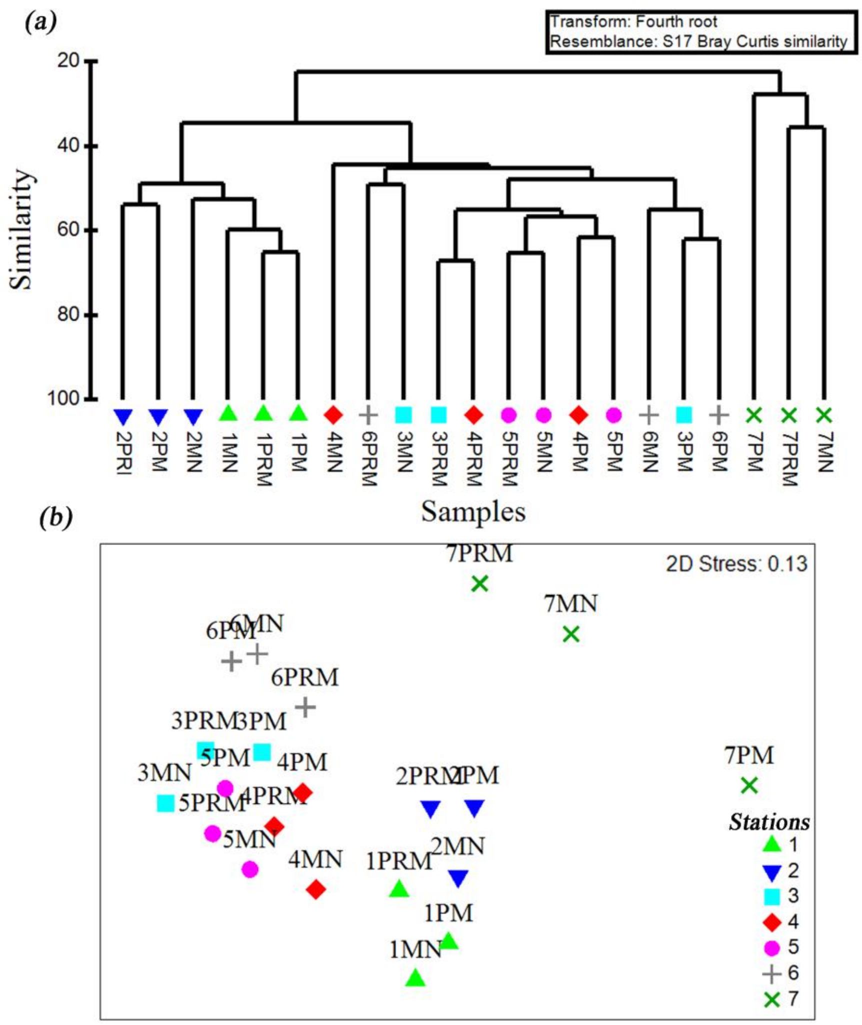


Figure 3.42 Bray Curtis similarity based on hierarchical clustering of stations manifested through a) dendrogram and b) NMDS (non-metric multidimensional scaling) showing macrobenthic assemblage pattern in the estuarine stations of the CE.

In the coastal waters Bray-Curtis similarity (hierarchical clustering) based on species density categorized the stations into three groups at 40% similarity (Fig. 3.43a-b). During PRM station 8 and station 9, situated nearest to the estuary formed a group. Other two stations were positioned as separately. During MN, station 9 and station 10 formed one group. Station 8 was more influenced by the monsoonal rains and subsequent land runoff, was situated apart from the other ones, and station 11, the farthest station kept separately. During PM all the stations formed a single group. Hence the clustering of the coastal stations was mainly based on seasons. Prominent seasonal variation was evident from the ANOSIM analysis (Global R.0.563, p value 0.3%) compared to the spatial variation.

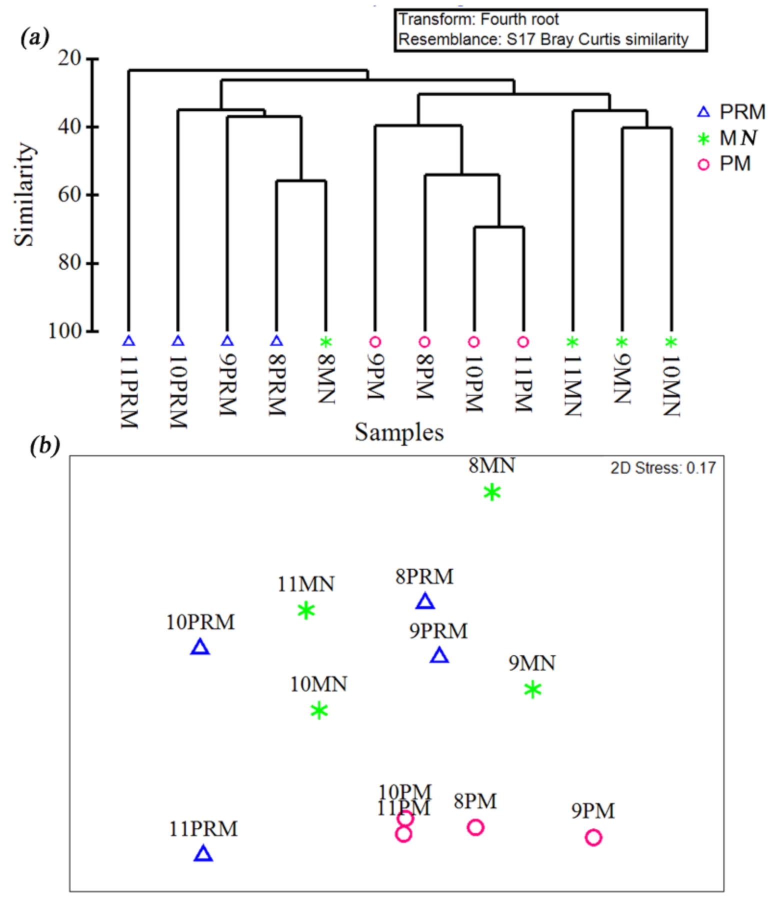


Figure 3.43 Bray Curtis similarity based on hierarchical clustering of stations manifested through a) dendrogram and b) NMDS (non-metric multidimensional scaling) showing macrobenthic assemblage pattern in the adjacent coastal regions of the CE.

Cluster analysis has been performed for all the stations together during the study period. The result showed a significant difference between the estuarine and coastal stations (Fig. 3.44). The results of the ANOSIM analysis (Global R. 0.844, p value 0.1%) also supported the clear disparity of the estuarine stations from the coastal stations.

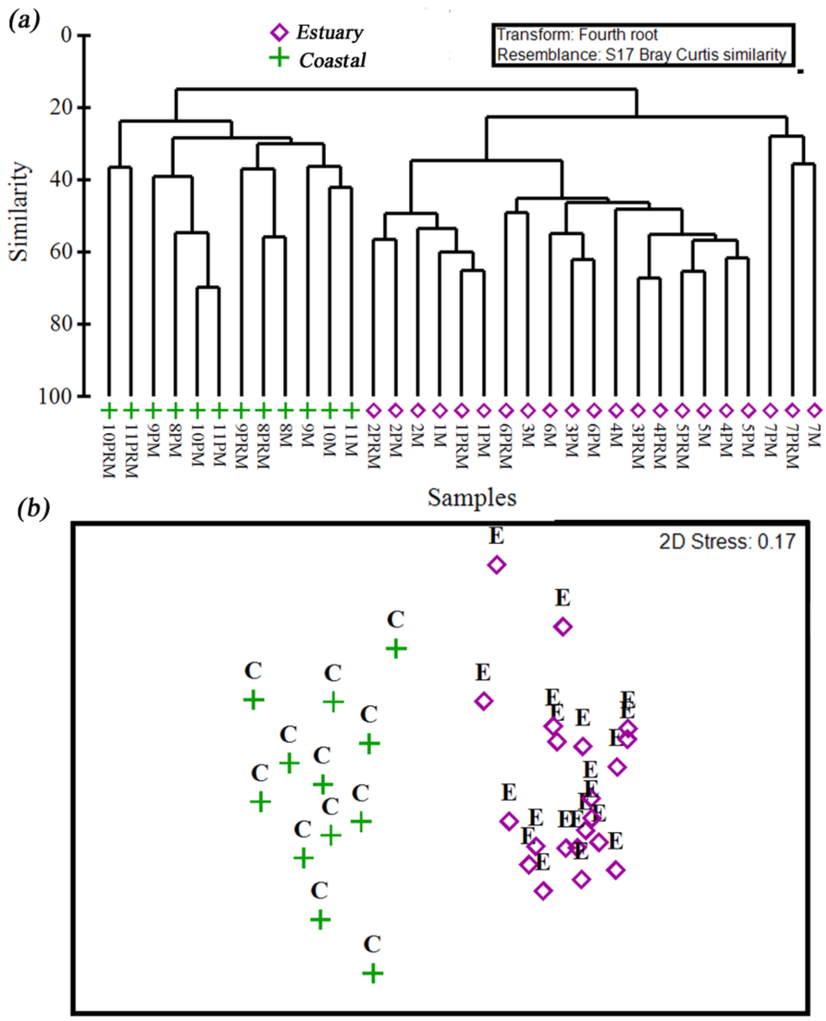


Figure 3.44 Bray Curtis similarity based on hierarchical clustering of stations manifested through a) dendrogram and b) NMDS (non-metric multidimensional scaling) showing macrobenthic assemblage pattern in the entire study area

As the results of the cluster and MDS analysis of the all stations together, depicts a clear separation of the estuarine stations from the coastal stations, SIMPER analysis has been performed to identify the major characterizing species that characterize each region and the discriminating species that differentiate the two regions. The results of SIMPER analysis has been shown in Table 3.3 and Table 3.4.

Table 3.3 *Major characterizing species identified through SIMPER that contribute to the average similarity within each assemblage*

Assemblages	Species	Av. Abundance	Av. Similarity	Sim/SD	% Contribution
Sim: 37.40%	*Tubificidae-Oligochaeta*	3.68	5.69	1.63	15.21
	Mediomastus capensis	2.85	3.93	1.25	10.5
Group I-Estuary	*Prionospio cirrifera*	2.35	3.15	1.47	8.42
	Sigambra parva	1.99	2.94	1.42	7.85
	Apseudus chilkensis	1.96	2.81	1.17	7.52
	Eriopisa chilkensis	1.44	1.84	0.95	4.91
	Nepthys oligobranchia	1.35	1.73	0.66	4.64
	Paraheteromastus tenuis	1.33	1.08	0.57	2.89
	Amphipoda	1.3	1.02	0.64	2.73
	Cirratulus filiformis	1.17	0.99	0.49	2.65
	Nereis sp	1	0.98	0.58	2.63
	Dendronereis estuarina	1.22	0.93	0.57	2.48
	Caulleriella capensis	1.08	0.77	0.49	2.06
	Melita zylanica	0.81	0.56	0.44	1.49
	Caprellid sp	1.18	0.56	0.37	1.49
	Diopatra neopolitana	1.36	0.47	0.35	1.26
	Cirratulus cirratus	0.67	0.41	0.33	1.1
	Bivalvia	1.5	1.69	0.95	4.53

Sim: 30.36%	*Paraprionospio pinnata*	3.31	8.9	1.77	29.31
Group II-Coastal	*Cossura coasta*	1.82	3.77	1.01	12.42
	Mediomastus capensis	1.25	2.9	0.94	9.55
	Lumbriconereis latreilli	1.24	2.64	0.82	8.7
	Bivalvia	1.18	2.5	0.65	8.22
	Goniada emerita	0.95	1.16	0.51	3.81
	Gastropda	0.72	1.11	0.41	3.66
	Magelona cinta	0.72	0.96	0.48	3.18
	Caulleriella capensis	0.85	0.75	0.29	2.48
	Sternaspis scutata	0.65	0.54	0.31	1.79
	Foraminifera	1.05	0.52	0.2	1.71

Table 3.4 *Discriminating species with mean abundances of species that contribute to the maximum dissimilarity between the assemblages*

Average dissimilarity:85.2% Species	Group I-E Av.Abund	Group II-C Av.Abund	Av. Diss	Diss/SD	Contrib%
Tubificidae-Oligochaeta	3.68	0	5.28	1.68	6.2
Paraprionospio pinnata	0.35	3.31	4.23	1.93	4.96
Prionospio cirrifera	2.35	0.15	3.02	1.72	3.54
Mediomastus capensis	2.85	1.25	2.82	1.6	3.31
Apseudus chilkensis	1.96	0	2.77	1.54	3.26
Sigambra parva	1.99	0.27	2.5	1.53	2.94
Cossura coasta	0.6	1.82	2.11	1.33	2.48
Eriopisa chilkensis	1.44	0	1.98	1.39	2.32
Nepthys oligobranchia	1.35	0.68	1.93	1.07	2.26
Lumbriconereis latreilli	0.6	1.24	1.76	1.32	2.06
Magelona cinta	0	0.72	0.98	0.86	1.15
Sternaspis scutata	0	0.65	0.87	0.67	1.03

The distribution of major discriminating species identified through SIMPER between the estuary and adjacent coastal region has been overlaid on the NMDS and represented as bubble plot (Fig. 3.45 & 3.46). Species such as *Tubificid Oligochaeta*, *Mediomastus capensis, Prionospio cirrifera, Sigambra parvae, Nephtys oligobranchiata, Apseudus chilkensis,* and *Eriopisa chilkensis* were mostly found in estuarine stations while species such as *Paraprionospio pinnata, Magelona cinta, Cossura coasta, Sternaspis scutata,* and *Lumbriconereis latreilli* were mostly observed in the coastal waters.

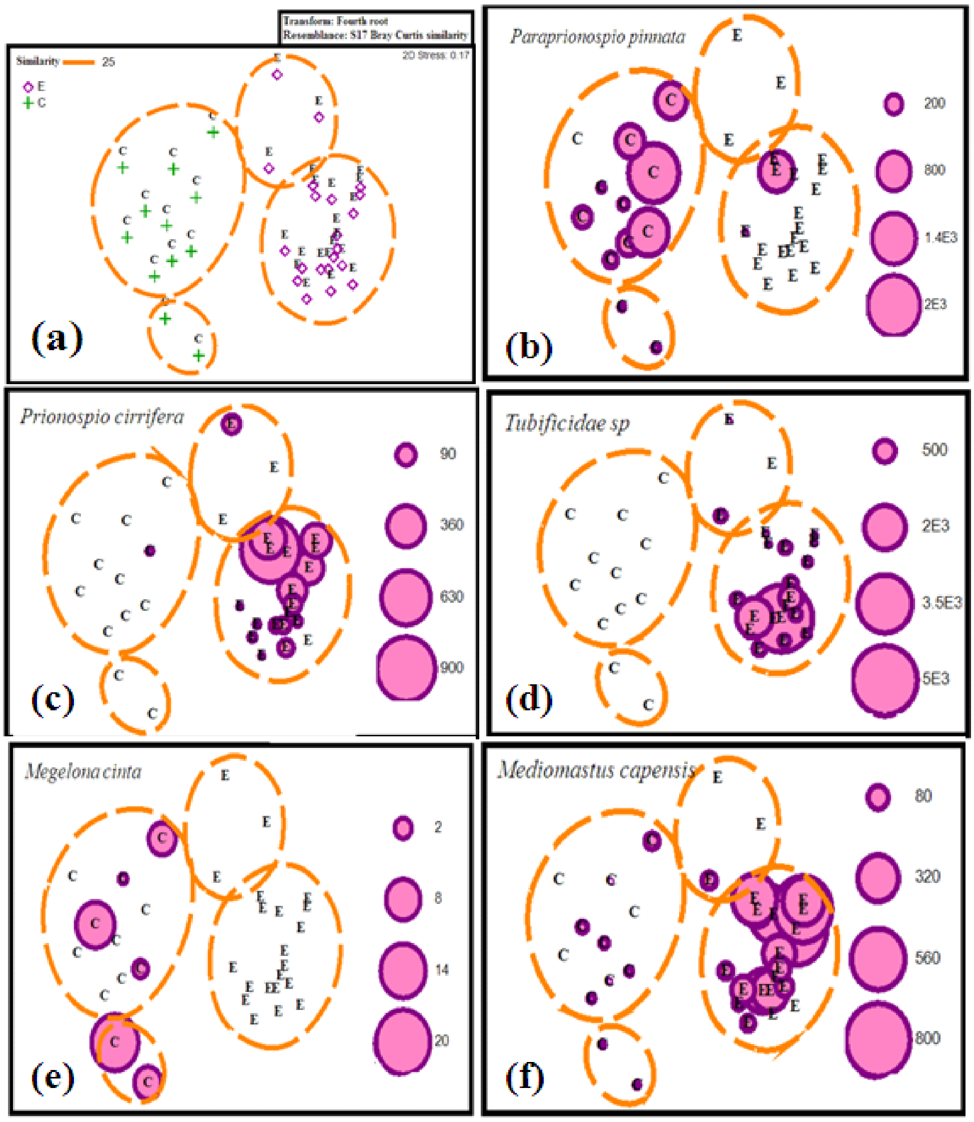


Figure 3.45 (a) NMDS- plot showing the distribution of macrobenthos, and (b to f) the bubble plot showing the distribution of major discriminating species overlaid on NMDS between estuary and adjacent coastal waters of CE.

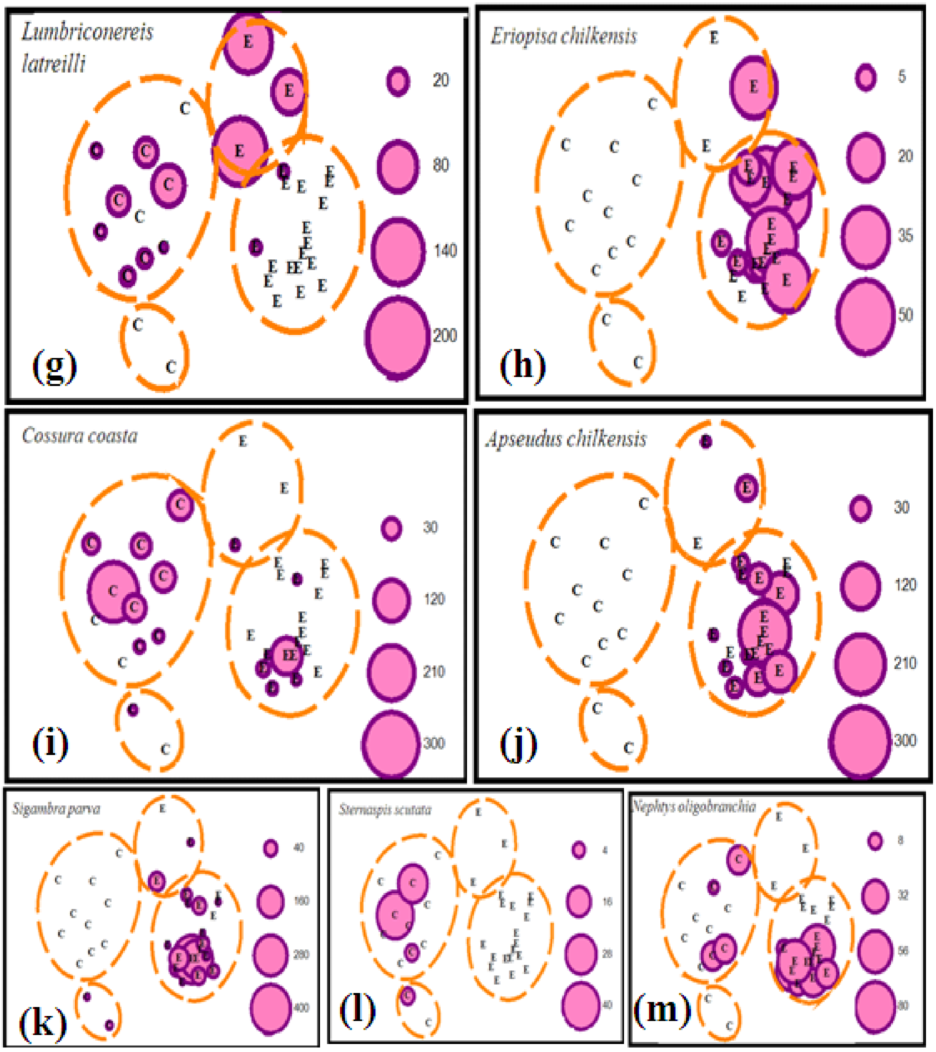


Figure 3.46 (g-m) the bubble plot showing the distribution of major discriminating species overlaid on NMDS between estuarine and adjacent coastal waters of CE.

3.4.3.11 Relation with environmental variables

Redundancy analysis was performed using the software CANOCA 4.5 to evaluate the relationship between the density and biomass of the macrobenthic fauna with the environmental parameters. The analysis helps to give a concurrent representation of the response variables (density and biomass) and the explanatory variables (environmental and sediment characteristics) in two or three dimensions and the variables in the triplot facilitate to visualize their interrelationships.

In the estuary, macrobenthic density exhibited a positive affinity with the salinity (during PRM-weak affinity), and sand during the entire study period. While the macrobenthic biomass showed a positive relation with the abiotic parameters such as salinity, SPM and sand (Fig. 3.47a-c).

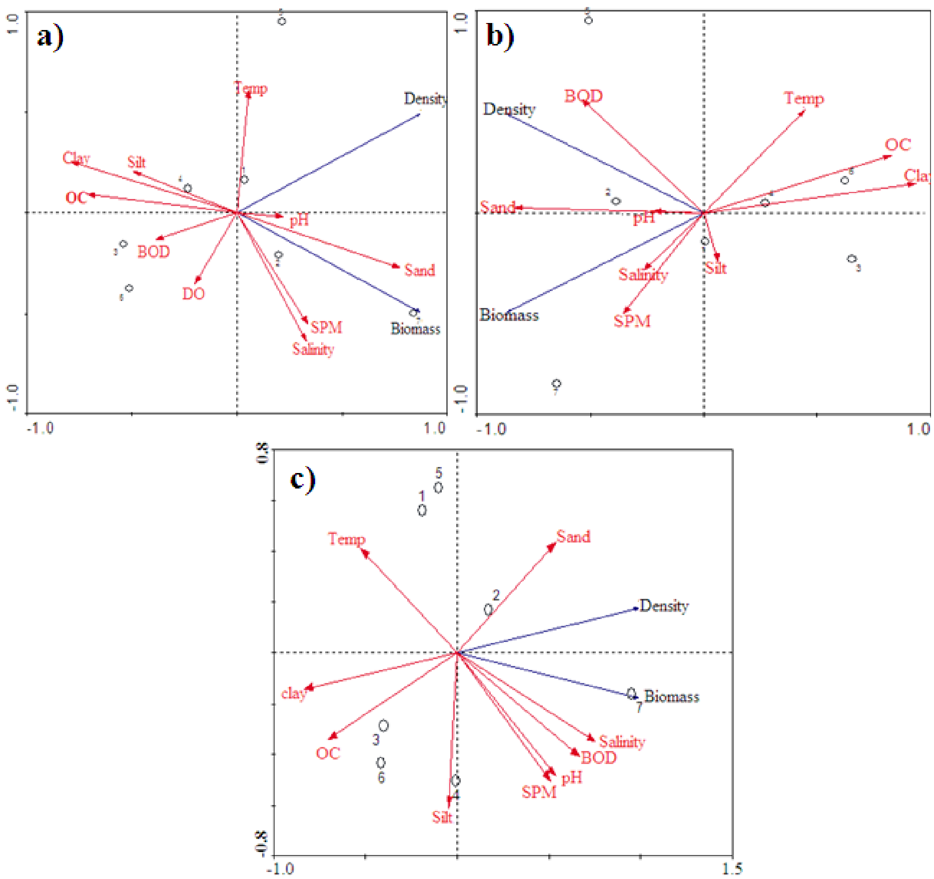


Figure 3.47 RDA plot showing the influencing factors on the macrobenthic density and biomass in the CE during a) pre-monsoon, b) monsoon and c) post monsoon season

In the coastal waters density showed a positive relation with clay during PRM and MN, while during PM a positive affinity observed with sand. In addition, density exhibited a positive relation with SPM during MN and PM. Biomass was positively related with SPM and sand during PRM and PM (Fig. 3.48a-c).

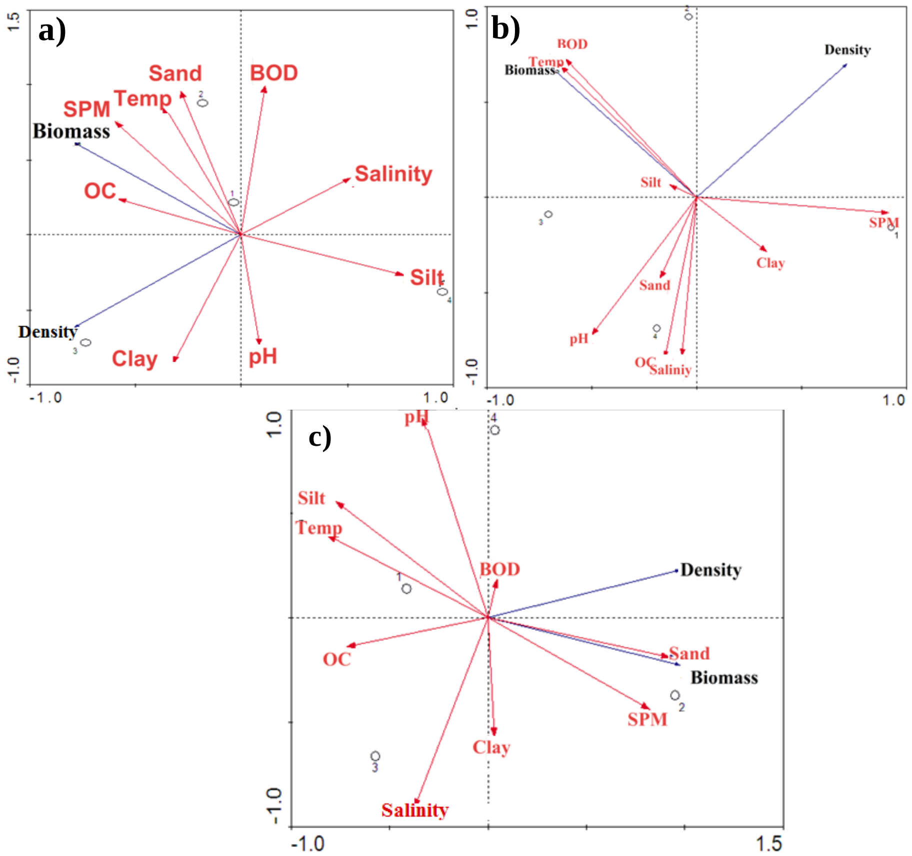


Figure 3.48 RDA plot showing the influencing factors on the macrobenthic density and biomass in the adjacent coastal waters during a) pre-monsoon, b) monsoon and c) post monsoon season

As polychaetes formed the dominant fauna in the estuary and coastal waters, redundancy analysis was performed between the major polychaete species and the abiotic variables to understand the interrelationship of polychaetes species with these factors. In the estuary, species such as *Sigambra parvae, Cossura coasta, Nephtys oligobranchiata,* and *Cirratulus filiformis*, exhibited positive relation with clay and organic carbon. Species like *Diopatra neapolitana, Paraprionospio pinnata,* and *Lumbriconereis latreilli,* showed positive affinity to sand and salinity. *Diopatra neapolitana, Sabellid* sp and *Syllid* sp were exhibited positive relation with SPM (Fig.3.49a-c).

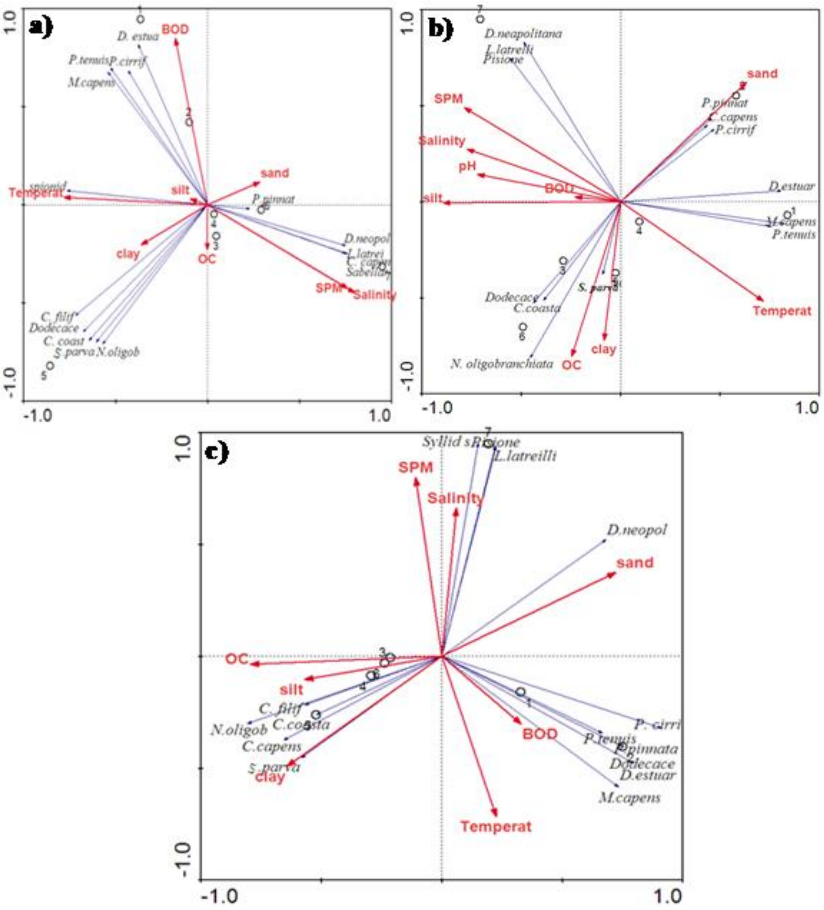


Figure 3.49 RDA plot showing the influencing factors on the macro-benthic species in the estuary during a) pre-monsoon, b) monsoon and c) post monsoon season

In the coastal waters, *Cossura coasta* exhibited positive affinity with SPM. Species *Magalona capensis* showed positive relation with sand and SPM during PRM and PM. Polychaetes such as *Goniada emerita*, and *Paraprionospio pinnta* showed positive affinity to sandy substratum (Fig.3.50a-c).

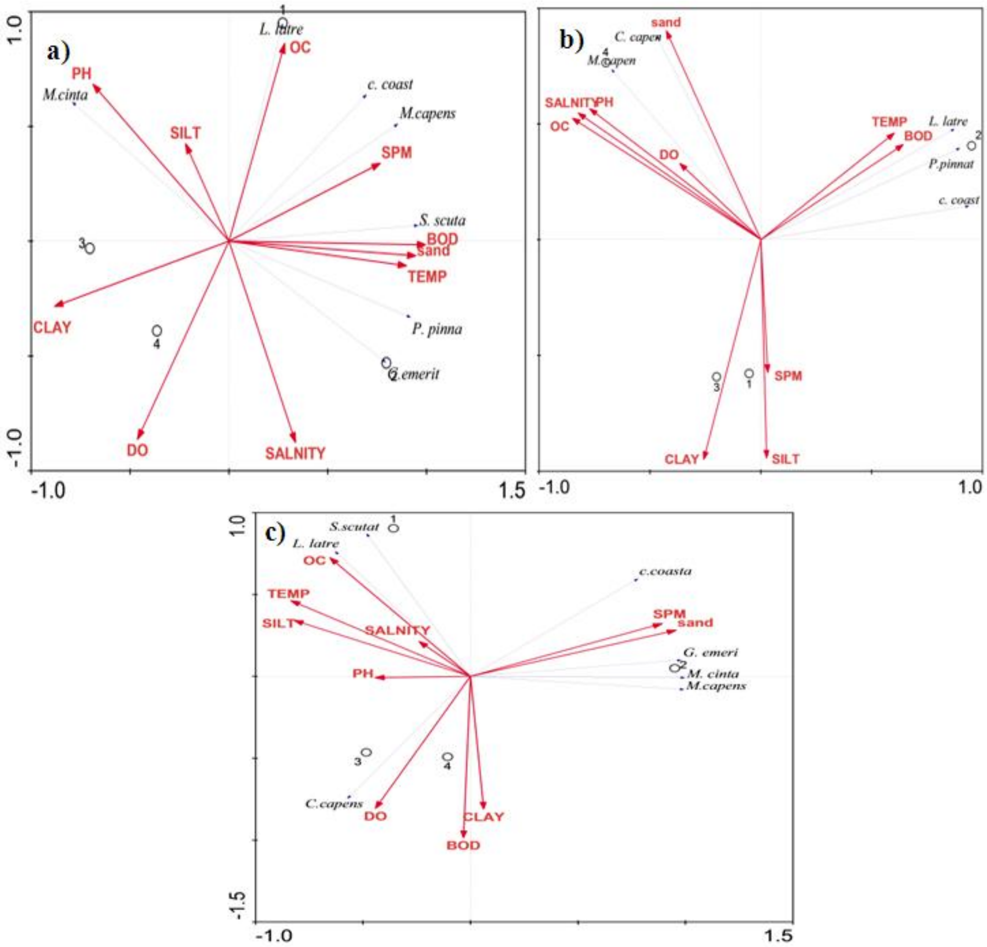


Figure 3.50 RDA plot showing the influencing factors on the macrobenthic species in the adjacent coastal waters during a) pre-monsoon, b) monsoon and c) post monsoon season

3.4.4 Polychaete feeding guilds

In the estuary, five polychaete feeding modes were identified comprising of carnivores, surface deposit feeders, sub-surface deposit feeders, filter feeders and herbivores. Carnivorous polychaetes (av. 43.0 %) dominated during the study period along with surface deposit feeders (av. 29.9%), sub-surface deposit feeders (av.24.7%), filter feeders (av. 2.1%) and herbovores (av. 0.1%). During PRM, carnivory (av. 49.7%) and sub-surface feeding (av. 32.5%) formed the major feeding mode in the estuary. Higher number of carnivores were observed at station 7 (2163 ind.m^{-2}) and surface deposit feeders had higher numerical abundance in station 1 (330 ind.m^{-2}) during the season (Fig. 3.51a). During MN, surface deposit feeding (av. 42.9%) formed the dominant feeding mode and higher number of surface deposit feeders were observed at station 2 (2170 ind.m^{-2}) (Fig. 3.51b). During PM carnivorus polychaetes dominated (av. 48.8%) with higher density at station 7 (1509 ind.m^{-2}) (Fig. 3.51c).

In the adjacent coastal waters five feeding modes were exhibited by polychaetes. Surface deposit feeding polychaetes (av. 59.8%) dominated in the coastal waters along with sub-surface deposit feeders (av. 27.4%), carnivores (av. 12.0%), filter feeders (av. 0.5%), and herbivores (av. 0.3%) (Fig.3.52). During PRM, sub-surface deposit feeding was the major feeding guild (av. 67.2%) and higher number of polychaetes exhibiting this mode was observed at station 8 (300 ind.m^{-2}) (Fig. 3.52a). During MN and PM surface deposit feeders (MN-av. 85.6%, PM-75.1%) were dominant with higher number of polychaetes exhibiting this feeding mode observed at station 9 (1305 ind.m^{-2}) and station 11 (826 ind.m^{-2}) (Fig. 3.52b-c).

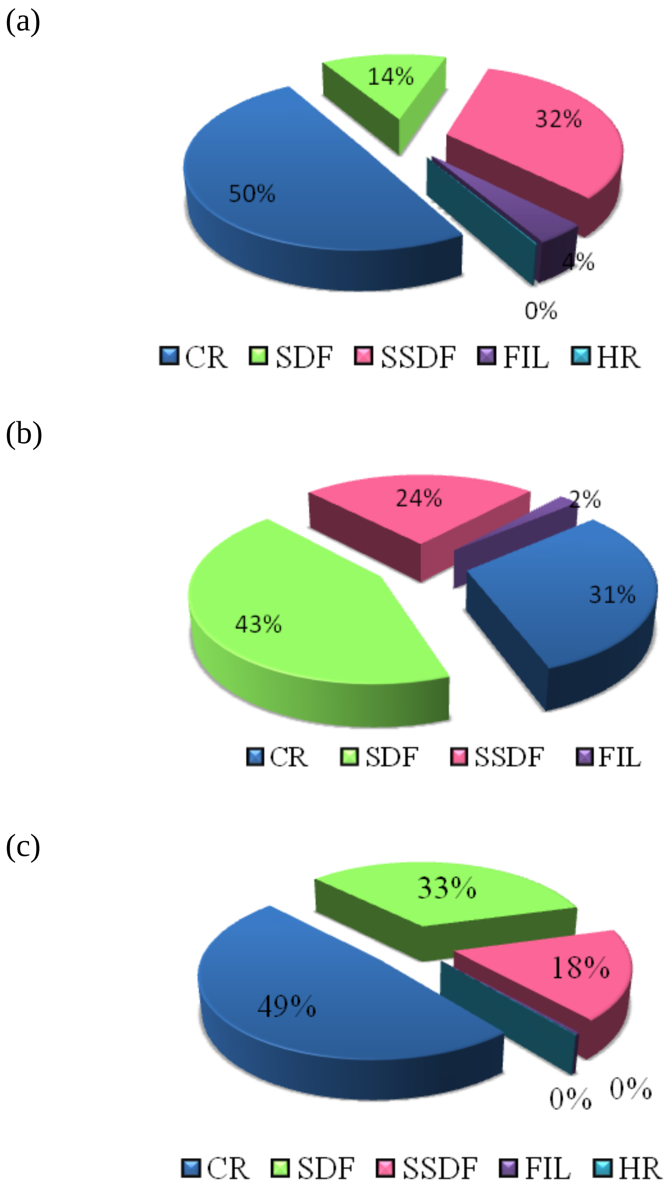


Figure 3.51 Polychaete feeding guilds in the estuary during (a) pre-monsoon (b) monsoon and (c) post-monsoon season

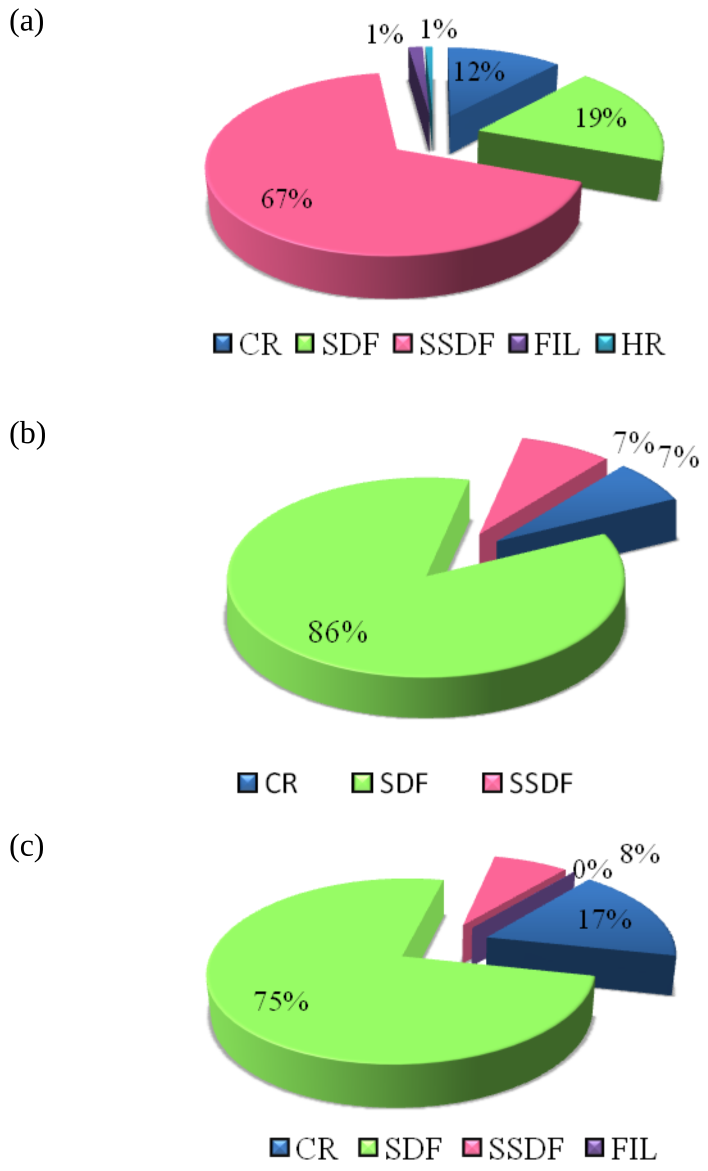


Figure 3.52 Polychaete feeding guilds in adjacent coastal waters during (a) pre-monsoon (b) monsoon and (c) post-monsoon seasons

3.5 DISCUSSION

The existence of biological communities in the aquatic ecosystems depends on the ability of these organisms to cope with the physical environment and their abiotic interactions (Cadotte et al., 2013). The persistent changes in the physico-chemical properties of an aquatic ecosystem directly or indirectly influence the survival of the biotic components of the system. The present study describes ecology and community dynamics of the macrobenthic fauna of the Cochin estuary and adjacent coastal waters, each characterized with distinct hydrographical properties.

Distribution of temperature in the CE chiefly depends on the magnitude of river discharges and seawater intrusion between seasons (Sankaranarayanan and Qasim, 1969). Higher water temperature observed in the estuary as compared to its adjacent coastal waters can be attributed to the shallow depth and the consequent higher penetration of solar insolation. Also, the influence of the warm fresh water from the rivers and tributaries causes an increase in the water temperature in the estuary compared to the coastal waters (Fig. 3.3 & 3.4). Present study observed higher bottom water temperature during PRM and lower during MN in the estuary and in the adjacent coastal waters. As the study area comes under the tropical regime, characterized by trivial spatio-temporal variations in temperature, temperature has a negligible role in limiting the distribution of benthic communities in tropical estuaries.

The CE, being a tropical monsoonal estuary is greatly influenced by the Indian Summer Monsoon (Vijith et al., 2009). Salinity of the estuary is primarily determined by the combined variations in the freshwater influx and the tidal activity. Along a seasonal scale, especially during MN, the higher precipitation and the subsequent runoff often leads to a remarkable decrease in salinity throughout the estuary (Qasim 2003; Madhu et al., 2010a), as the total runoff into the estuary during MN is several times higher than its volume, leading to a complete freshening of the estuary (Revichandran et al., 2012). In comparison to the estuarine regions heavily influenced by precipitation and runoff patterns during MN, the changes are markedly less in the adjacent coastal waters. Though the coastal stations located near to the estuarine zones exhibited less salinity, stations located far away from the estuarine zones showed relatively higher salinity. Hence, from this observation it can be inferred that monsoonal precipitation and associated influxes have an upper hand in regulating the salinity patterns of the estuarine and nearby coastal stations during MN. In case of the non-monsoon periods (both PRM and PM), the higher tidal incursion contributing towards the increased intrusion of high saline water from AS transforms the estuary into a high saline system which gets sustained until the beginning of the recurrent monsoon season (Ramamirtham and Jayaraman, 1963).

In estuaries, pH of the water is mainly influenced by the physico-chemical (rainfall and runoff), biological (water column productivity, bacterial activity) and anthropogenic factors (discharge of the sewage and industrial effluents) (Sarma et al., 2011; Hossain and Marshall, 2014). Along a spatial scale, the present study witnessed an increasing trend in salinity and pH from the estuary towards the sea. Due to the buffering effect of the sea water (Hossain and Marshall, 2014), high pH was evident in the coastal waters compared to the estuary (Fig. 3.8). Relatively high pH observed in the estuary during PM, can be attributed to the increased intrusion of the dense saline water through the bottom. Lower pH was observed during PRM in the estuary as well as in the adjacent coastal waters. Higher decomposition of the organic matter during PRM might be responsible for this noticeable reduction in the pH.

Physico-chemical factors such as temperature, salinity, pressure, tidal rhythm, freshwater flow, and biological factors like water column production, respiration and microbial decomposition have an imperative role in the dissolved oxygen concentrations in water bodies (Unnithan et al., 1975; Wetzel, 2001). Compared to the estuary, slightly lower DO was observed in the adjacent coastal waters. The lower dissolution of DO in high saline waters can be a prominent factor responsible for the lower DO of coastal waters. Relatively high DO was observed during PRM in both estuary and adjacent coastal waters. Similar results of higher DO in the CE during PRM by Sivadasan and Joseph (1997) further substantiate the results of the present study. During PRM the high phytoplankton production in the estuarine waters were observed to cause a substantial increase in DO (Madhu et al., 2007). The presence of upwelled waters might have resulted in the reduction of DO during MN in the coastal waters. Utilization of oxygen for the decomposition of organic matter by the microbial fauna increases the biological oxygen demand, which was relatively high during non-monsoonal months in the present study. In addition, increase in the temperature enhances the microbial decomposition of the organic matter.

The present study witnessed relatively higher SPM levels (>av.40mg/l) in the CE and adjacent coastal waters irrespective of seasons. Slightly higher SPM evident in the estuary during non-monsoon periods might have happened by the increased stirring up of sediments brought about by the bottom currents. In the coastal waters, relatively high SPM levels were observed during MN and PRM. In the coastal waters, the higher bottom hydrodynamics along with the inputs from monsoonal runoff and discharges contributed to the higher SPM levels during the periods. Spatial variation in SPM was more prominent in the study region relative to seasonal variations. Estuarine stations adjacent to the shipping channel and also the coastal stations nearer to the estuarine regions exhibited higher SPM levels. High tidal activities and frequent channel dredging in these regions might have contributed to the high spatial variability in the SPM levels (Qasim and Gopinathan, 1969; Balchand and Rasheed, 2000).

Even though the CE is considered to sustain high inorganic nutrients irrespective of seasons (Balachandran et al., 2002; Sankaranarayana and Qasim, 1969; Madhu et al., 2007 & 2010b), remarkably higher levels of inorganic nutrients (nitrate, nitrite, phosphate and silicate) observed during monsoon period is brought by the torrential rainfall and subsequent runoff. The present study also evidenced this characteristic nutrient feature in the CE (Fig. 3.16 to 3.24). Earlier studies reported on higher nutrient enrichment of the CE mainly through discharges of the domestic and industrial effluents in conjunction with the increased human settlement (Balachandran et al., 2002; Qasim, 2003; Madhu et al., 2007). Compared to the CE, nutrient levels in the coastal waters were less throughout the year. Higher concentrations of ammonia in the estuary and coastal waters during non-monsoon months have revealed that it might have originated from anthropogenic activities (Miranda et al., 2008).

The sediment texture exhibited pronounced spatial variation in the estuary than the seasonal variation. The bottom sediments in the CE comprise the sediment load brought from the rivers as well as sediment transport from sea to the estuary through tidal activities (Gopinathan and Qasim, 1971; Veerayya and Murty, 1974). Higher percentage of coarser particles (sand) was observed during PRM, whereas during MN the percentage of coarser particles were observed to be relatively less, indicating the transport of sand from sea to the estuary through tidal currents (Veerayya and Murty, 1974; Hossain et al., 2014). Higher percentage of sand particles recorded at station 7, inlet location (during PRM) and at station 2, near inlet location (during PM) validates the above mentioned statement. Similar to estuary, adjacent coastal waters also sustain high percentage of coarser particles during PRM. Spatially, higher percentage of sand particles apparent at station 1 in the estuary, during MN indicates its riverine origin. According to Liu et al. (2010) coarser particles are deposited to the upstream of the estuary as a result of the selective deposition. Sediment organic carbon was relatively less in the estuary during MN as estuarine waters normally undergo higher dilution during this period. The heavy flushing out of the estuary might have contributed to the higher organic carbon during MN and PM in the adjacent coastal waters. As per literature, fine sediment particles have greater surface area and also have greater ability to retain more organic carbon when compared to the coarser sediment particles (Nayar et al., 2007). This might be the reason for the prevalence of higher organic carbon content at stations having finer sediment particles. The organic load brought about by the discharge of domestic wastes from the nearby highly populated towns might also contribute towards its increased concentration in the estuarine channels and adjacent coastal area (Anon, 1996; Ingole et al., 2009).

Generally, macrobenthic density was higher in the estuary as compared to the adjacent coastal waters. High density of bivalves observed during PM in the estuary and during PRM in the coastal waters contributed correspondingly to higher macrobenthic density in the respective seasons. Increase in the density of macrobenthic fauna in the CE after MN period may be the outcome of intensive recruitment process as part of stabilization of environment (Harkantra and Rodrigues, 2003). Earlier studies also showed similar results of decreased macrobenthic density during MN and increased density during PM periods (Sivadas et al., 2011). Generally in tropical estuaries macrobenthic fauna exhibited a marked decline during MN, and an increased recruitment in PM (Parulekar et al., 1980; Sivadas et al., 2011). In the present study, though the macrobenthic density exhibited an increase after MN through PM, the decline in density evident during the stable PRM indicates towards the increased intensity of anthropogenic pressures on them. Higher macrobenthic biomass observed during PRM and PM in the CE was mainly contributed by the dominance of large sized polychaetes (*Diopatra neapolitana*) during the periods. In the coastal waters, the larger sized bivalves (*Villorita cyprinoides and Pholas orientalis)* contributed to the higher biomass during PM period. The observed macrobenthic biomass was not in accordance with the macrobenthic density, indicating the increase of small sized opportunistic organisms in the study area.

In addition to the standing crop, number of macrobenthic taxa observed was relatively higher in the CE (88 macrobenthic taxa belonging to 5 phyla) when compared to the coastal waters (58 macrobenthic taxa belonging to 6 phyla). Polychaeta, Oligochaeta, Amphipoda, Tanaidacea, Bivalvia, Gastropoda, and Isopoda formed the dominant macrobenthic groups in the estuary whereas in the coastal waters Polychaeta, Bivalvia, Foraminifera, Amphipoda, and Cumacea predominated. Higher number of macrobenthic groups evidenced in the estuary signifies the wide range of environmental variables and availability of varied food resources in the estuarine system than the adjacent coastal waters.

Species diversity was maximum during PM and minimum during MN period. Heavy rainfall and associated land runoff were observed to impose severe stress on benthic communities during MN period (Pillai, 1977; Alongi, 1989), resulting in lower diversity during MN. Coastal waters also exhibited a reduction in the benthic species diversity during MN.

Heavy precipitation associated with MN and the subsequent salinity variation impart various types of stress on benthic community such as mortality of adult and larval population or affecting their migration patterns (Alongi, 1990). Salinity drop can activate the gonadal release from the macrobenthic community (Kinne, 1977), and under extreme conditions, the larval forms tends to produce cysts and delay their larval settlement (Osman, 1977; Richer, 1977). Similarly, species diversity and species richness of macrobenthos was higher in the estuary than the adjacent coastal waters.

As per earlier reports macrobenthic community is normally influenced by a combination of factors such as temperature, salinity, DO, sediment texture and organic matter (Jayaraj et al., 2007). Present study showed that the CE sustained warm, oxygenated euryhaline nutrient enriched waters compared to the coastal waters. Previous studies have pointed out the influence of salinity on the distribution of macrobenthic fauna in estuaries (Sanders et al., 1965; Kennish et al., 2004; Sivadas et al., 2011). In the present study also, a positive affinity towards salinity was exhibited by the estuarine macrobenthic standing stock. High salinity in the coastal waters was conducive only to certain macrobenthic organisms and hence forms a limiting factor influencing the distribution of macrobenthic organisms. Hence salinity acting as an imperative factor in the distribution and density of macrobenthic fauna can be justified. The results of the multivariate analysis further give affirmation to the above statement (Fig. 3.47).

In addition to salinity, sediment texture and composition are considered to have a prominent influence on the community structure of the macrobenthic fauna (Gray, 1974; Mannino and Montagna, 1997; Kennish, 2001; Kennish et al., 2004). In the estuary, the macrobenthic standing stock (density and biomass) exhibited positive relation with sand throughout the study period (Fig. 3.47). However, in the coastal waters, macrobenthic density exhibited positive relation with clay except during PM (positive affinity with sand) (Fig. 3.48). Macrobenthic biomass showed positive affinity with sand except during MN (positive affinity with silt). Sediment texture has a profound influence on the macrobenthic density and often rich benthic fauna was observed in clayey sand and sandy substrate, while substrata with only clay showed poor abundance (Harkantra and Parulekar, 1985). Similar to present study Devi et al., (1999) also observed higher benthic biomass associated with sandy sediments.

In addition to sediment texture, SPM and organic carbon were also observed to influence the macrobenthic standing stock in the estuary and coastal waters (Fig. 3.47, 3.48). Hence in the present study, macrobenthic standing stock showed positive relation with the substratum and food availability, though it exhibited variation between seasons.

Polychaetes dominated the macrobenthic fauna of both sampling areas throughout the study period. As per the literature, polychaetes display high stability and adaptability to different habitats (Simboura et al., 1995; Simboura et al., 2000) with various feeding strategies (Fauchald, 1977), and often dominates in terms of species number and also in numerical abundance. Earlier studies in the tropical regions also showed the dominance of polychaetes in the macrobenthic community (Jayaraj et al., 2007; Ingole et al., 2009; Mandal and Harkantra, 2013). Among polychaetes, 53 species belonging to 25 families were observed in the estuary whereas 39 species belonging to 20 polychaete families were encountered in the coastal waters. Polychaete, *Diopatra neapolitana*, which are known to inhabit in muddy sand bottoms (Conti and Massa, 1998; Gambi et al., 1998; Dagli et al., 2005), dominated in the CE (at station 2 and station 7) during PRM and PM. In the RDA plot also a positive correlation was observed between the density of the species, salinity and sand (Fig. 3.49). The drop in salinity and change in the sediment texture during MN might have resulted in rapid reduction in the density of the species during the season. *Diopatra neapolitana* are tubicolous worms which are known to build tubes out of sand grains and detritus and harbor a variety of other invertebrates on their tubes. Density of Bivalves and amphipods noticed in these stations and were found in association with the tube of the species. But during MN, *D. neapolitana* was replaced by higher abundance of *Pisione* sp, at station 7. *Pisione* sp, is a small interstitial highly motile annelid usually found in high energy habitats with coarse sediments (Withers and Thorp, 1978; Vanosmael et al., 1982) or coastal waters stirred by waves (Peres, 1967; Quintino et al., 1989). Opportunistic polychaetes such as *Mediomastus capensis, Sigambra parva, Paraheteromastus tenuis,* and *Prionospio cirrifera* were also found abundant in the estuary. Rivero et al., (2005) reported the species *Mediomastus* sp as indicators of moderate environmental disturbances from Argentina coast. In addition, the species of the genus *Mediomastus* were observed to increase their abundance in areas of moderate organic carbon, while it disappears from the area of organic enrichment (Pearson and Rosenberg, 1978). Species of the family Spionidae (*Prionospio cirrifera, Paraprionospio pinnata*) and Capitellidae (*Mediomastus capensis, Paraheteromastus tenuis*) were widely distributed along the southwest coast of India (Gopalakrishna Pillai, 1978; Batcha, 1984). Earlier studies in CE observed two deposit-feeding polychaetes of genus *Prionospio* (*P. polybranchiata and P. pinnata*) from municipal discharge site (Remani et al., 1983). Further, reports on the occurrence of opportunistic benthic polychaetes as a consequence of organic enrichment (Feebarani, 2009; Martin et al., 2011) from the CE also corroborates the observation. Dominance of many opportunistic polychaetes recorded in the CE during the present study might have happened by the presence of organic rich sediments in combination with other anthropogenic disturbances such as dredging activities and discharge of sewage effluents.

In the coastal waters, the polychaete species *Paraprionospio pinnata* was observed to dominate throughout the sampling period except during PRM. Their life cycle normally enables them to inhabit disturbed areas quickly. Since they are surface deposit feeders, they can efficiently utilize the organic materials brought about by the monsoonal rainfall and runoff (Blake and Arnofsky, 1999; Rouse and Pleijel, 2001). The extended branchiae of *P. pinnata* aid them in efficiently capturing suspended food particles (Levin et al., 2003). *Cossura coasta,* another polychaete species considered as indicators of sediment instability (Ellis et al., 2000) were found to dominate in the coastal waters during PRM. This species has been reported from the coastal waters of AS, where the riverine inputs are more (Abdul Jaleel et al., 2014).

Among the non-polychaete taxa observed in the CE, tubificid Oligochaeta and Tanaid (*Apseudus chilkensis*) exhibited higher density from stations 3 to 6. Substrata of these stations are predominated by finer sediments with high organic carbon. In an anthropogenic influenced disturbed environment, species diversity declines and pollution-tolerant organisms such as small sized opportunistic oligochaetes were found to replace pollutant-sensitive species (Farara and Burt, 1993). Pollution indicator and tolerant species such *as Capitella capitata, Cossura coasta, Dendronereis estuarina, Lycastis indica, Mediomastus capensis, Paraheteromastus tenuis, Prionospio cirrobranchiata, Prionospio polybranchiata* and *Apseudus chilkensis* were also recorded in the CE during the study period. In coastal waters, foraminiferans were formed the major non-polychaete taxa, especially at station 8, situated nearer to the estuary. Their occurrence at coastal waters indicates their preference to marine habitat. Similarly, high density of foraminiferans in the estuarine mouths during MN period probably associated with the higher food availability brought in by the river run off.

Polychaetes usually exhibit wide range of feeding guilds (Snelgrove, 1998), which are known to vary depending on various physico-chemical (e.g. sediment characteristics, salinity) and biological factors (e.g. food availability from detritus, microbial and macrofauna) (Magalhaes and Barros, 2011). Feeding guild analysis of polychaetes revealed the existence of five types feeding modes in the CE throughout the year (Fig. 3.51). On the other hand, in the coastal waters polychaetes exhibited five types of feeding guilds during PRM, three during MN, and four during PM (Fig. 3.52). Carnivores dominated the estuary while surface deposit feeders dominated the adjacent coastal waters. Surface deposit feeding and sub-surface deposit feeding are the two modes of deposit feeding exhibited by polychaetes, in which former ingest the food particles from the sediment surface while latter ingest food particles by burying inside the sediments. Domination of deposit feeding mode observed both in the estuary and coastal waters (SDF and SSDF), indicates the significance of organic detritus as energy source for the macrobenthic fauna. Earlier studies have revealed the importance of detritus as food source for many estuarine and inshore organisms (Newell, 1965). The sources of organic detritus in the CE include plankton, benthic algae, rooted plants, animal matter, suspended soft mud and silt particles brought down by rivers (Qasim and Sankaranarayanan, 1972). The heavy monsoonal rains and runoff brings in high organic matter and detritus that might be well exploited by the surface deposit feeders leading to their higher proliferation in the estuary during the season. High density was maintained by surface deposit feeders during MN and PM season. Assemblages of deposit feeders vary according to the availability of food resources (Whitlatch, 1981; Rossi, 2003). In the estuary, carnivores dominated during PRM and PM periods of which, the polychaetes family Onuphidae was dominated. Decrease in density of the species (family Onuphidae) during MN might have resulted by the disturbance generated by heavy rainfall and runoff. In the coastal waters, sub-surface deposit feeders dominated during PRM. However, during MN and PM periods, sub-surface deposit feeders are replaced by surface deposit feeders, by the effective utilization of the food resources brought by the monsoonal run off. High production and terrestrial discharge resulted in higher organic matter in the surface sediments of the coastal waters of AS during MN (Abdul Jaleel et al., 2015). Previous studies in the near shore waters of south-west coast of India suggested the influence of estuarine inputs on sediments (Jayaraj et al., 2008; Joydas and Damodaran, 2009). Filter feeders, though less abundant were comparatively high during PRM in both estuary and coastal. As they need a stable substratum to attach and filter particles, the relatively stable PRM period might have supported the higher density of filter feeders. In spite of the availability of high suspended matter in the estuary, corresponding increase in number of filter feeders were not observed in the study area indicating the influence of anthropogenic disturbances in the estuary. The high macrobenthic standing stock, species diversity and diverse feeding guilds observed in the estuary compared to the coastal waters might be because of the wide range of environmental variables and wide variety food resources available in the estuary.

In the estuary spatial variation in macrobenthic community was more prominent than seasonal variations. Cluster analysis performed using the macrobenthic species density in the estuary revealed that resultant grouping was mainly based on spatial variability. Within the estuary the stations characterized with sandy substratum formed one group; the stations having finer sediments together formed another group whereas the inlet formed a separate group (having high salinity and coarser substratum). The results of ANOSIM analysis (Global R value 0.658, p value 0.1%)

based on macrobenthic density further revealed the pronounced disparity between the stations in the estuary. In the coastal waters, seasonal variation in macrobenthic density was more prominent than the spatial variation. The results of the cluster analysis exhibited by distinct groups mainly based on seasons supported the finding. The results of ANOSIM (Global R value 0.563, p value 0.3%) based on macrobenthic faunal density confirmed the variation between seasons. From these results it can be stated that macrobenthic faunal distribution both in the CE and adjacent coastal waters was primarily linked to the sediment texture. The spatial variation occurring in the sediment texture of the CE was reflected in macrobenthic density as well. In contrast, coastal waters exhibited seasonal variation in sediment texture which was evident in the macrobenthic density as well. When cluster analysis was performed using the entire sampling stations together (estuary and coastal waters) based on the macrobenthic species density, clusters of estuarine stations were kept apart from the clusters of coastal stations (Fig. 3.44). The results of the MDS and ANOSIM analysis (Global R. 0.844, p value 0.1%) also revealed the dissimilarity between the estuarine and coastal macrobenthic community. The discriminating species observed between the estuary and coastal waters (Average dissimilarity 85.2%) further affirms the above finding. The species such as Tubificid sp, *P. cirrifera, M. capensis, A. chilkensis, S. parva, E. chilkensis, N. oligobranchia,* were mostly occurred in the estuary while species like *P. pinnata, C.coasta, L. latrellei, M. cinta, S. scutata* were predominant in the coastal waters.

The present study showed salinity, sediment texture, and food availability are the major influencing factors in governing the distribution of macrobenthic community in the CE and adjacent coastal waters. The present study also evidenced the dominance of a variety of opportunistic macrobenthic species in the CE and adjacent coastal waters clearly pointing towards the influence of ongoing anthropogenic activities in and around the CE. Increased rate of sewage and industrial effluent discharges, oil and hydrocarbon pollution due to marine traffic and transportation, metal pollution from chemical industries, land reclamation, frequent dredging of shipping channels, and other developmental activities happening in and around the CE have apparently resulted severe ecological changes in the estuarine sediment characteristics and associated benthic fauna. Thus, the present study on macrobenthic community of the CE and adjacent coastal waters along spatial and temporal scales provides a glimpse on the recent status of sediment properties and concurrent faunal community structure which will ultimately help to better understand and evaluate the environmental changes associated with natural and anthropogenic activities in the CE.

Table 3.5 Macrobenthic density in the sampling sites during pre-monsoon (- <100 ind.m^{-2}, **100 to <500 ind.m^{-2}, ***500<1000 ind.m^{-2}, ****1000<3000 ind.m^{-2}, *\$-3000<7000 ind.m^{-2})*

	Estuarine stations							Coastal stations			
	1	2	3	4	5	6	7	8	9	10	11
Polychaeta											
Aphroditidae sp	-	-	-	-	-	*	*	-	-	-	-
Harmathoe sp	-	-	-	-	*	-	-	-	-	-	-
Sthenelais boa	-	-	-	-	-	-	*	-	-	-	-
Amphinomidae sp	-	*	-	*	-	-	-	-	-	-	-
Amphinomea rostrata	-	*	-	-	-	-	-	-	-	-	-
Ophiodromous sp	-	-	-	-	*	-	-	-	*	-	-
Glycera longipinnis	-	*	-	-	-	-	*	-	-	-	-
Goniada emerita	-	*	-	-	*	-	*	-	*	-	-
Sigambra parva	-	*	*	*	**	*	*	-	-	*	*
Sigambra sp	-	-	-	-	-	-	*	-	-	-	-
Nephtys oligobranchia	-	-	*	*	*	*	-	-	-	-	-
Nephtys dibranchia	-	-	*	-	-	-	-	*	-	-	-
Nephtys polybranchiata	*	-	-	-	-	-	-	-	-	-	-
Nereis sp	*	-	*	*	*	-	*	-	-	-	-
Dendronereis estuarina	*	*	-	-	*	-	-	-	-	-	-
Lycastis indica	*	-	-	*	-	-	-	-	-	-	-
Cirratulus cirratus	-	-	*	*	*	-	-	-	-	-	*
Cirratulus filiformis	*	-	*	-	*	*	-	-	-	-	-
Dodecaceria sp	-	-	-	-	*	-	-	-	-	-	-
Caulleriella capensis	-	-	-	-	-	-	*	-	-	-	-
Paraprionospio pinnata	-	-	-	-	-	*	-	*	*	*	*
Prionospio cirrifera	**	*	*	*	*	*	-	-	-	-	-
Prionospio cirrobranchiata	*	-	-	*	*	-	-	-	-	-	-
Laonice cirrata	*	-	*	*	-	-	-	-	-	-	-

Scolelepis squamata	*	-	-	*	-	-	-	-	-	-	-
Polydora ciliata	-	*	-	-	-	*	*	-	*	-	-
Polydora capensis	-	-	-	-	-	*	-	-	-	-	-
Prionospio sp	-	-	-	-	*	-	-	-	-	-	*
Boccardia polybranchia	-	*	-	-	-	-	*	-	-	-	-
Spionid sp	*	*	*	*	*	-	*	-	-	-	-
Mediomastus capensis	***	**	*	*	**	*	*	*	*	*	*
Capitella capitata	*	*	-	-	*	-	-	-	-	-	-
Paraheteromastus tenuis	**	**	-	*	*	-	-	-	-	-	-
Maldane sarsi	-	-	-	-	-	-	-	*	*	-	-
Lumbriconereis impatiens	-	*	-	-	-	-	-	*	-	-	-
Lumbriconereis latreilli	-	-	-	-	-	*	**	*	-	-	-
Lumbriconereis sp	-	-	-	-	-	-	*	-	-	-	*
Ninoe pulchra	-	-	-	-	-	-	*	-	-	-	-
Diopatra neapolitana	-	**	-	-	-	*	****	*	-	-	-
Cossura coasta	-	*	*	-	*	-	*	*	*	*	-
Aricidea longobranchiata	-	-	-	-	*	-	-	-	-	*	-
Owenia fusiformis	-	*	-	-	-	-	-	-	-	-	-
Magelona cinta	-	-	-	-	-	-	-	*	-	*	*
Phalacrostemma elegans	-	-	-	-	-	-	*	-	-	-	-
Sabellaria sp	-	-	-	-	-	-	**	-	-	-	-
Pectinaria sp	-	-	-	-	-	-	*	-	-	-	-
Streblosoma sp	-	-	-	-	-	-	*	-	-	-	-
Chaetopteridae sp	-	-	-	-	-	-	*	-	-	-	-
Serpulidae sp	*	-	-	-	-	-	-	-	-	-	-
Sternaspis scutata	-	-	-	-	-	-	-	*	-	*	-
Amphipoda											
Corophium triaenonyx	-	*	-	-	-	-	-	-	-	-	-

Photis digitata	*	*	-	-	-	-	-	-	-	-	-
Eriopisa chilkensis	*	*	-	*	*	*	-	-	-	-	-
Melita zylanica	*	-	-	*	-	-	-	-	-	-	-
Gammaropsis sp	*	*	-	-	-	-	**	-	-	-	-
Gammarid sp	-	*	-	-	*	-	-	-	*	-	-
Cheriophotis megacheles	*	-	-	-	-	-	-	-	-	-	-
Caprellid sp	**	**	-	-	-	-	*	-	-	-	-
Amphipod sp	*	*	-	*	-	*	**	-	*	-	-
Isopoda											
Anthuridae	-	*	-	-	-	-	-	-	-	-	-
Cirolana	-	*	-	-	-	-	*	-	-	-	-
Isopod sp	-	-	-	-	-	-	**	-	-	-	-
Tanaidacea	-	*	-	-	-	-	-	-	-	-	-
Apseudus chilkensis	**	*	*	*	*	*	-	-	-	-	-
Mollusca											
Littorina littorea	*	*	-	-	-	-	-	-	-	-	-
Gastropoda sp	*	-	-	-	-	-	*	*	*	*	-
Perna sp	-	-	-	-	-	-	-	-	-	*$	-
Bivalvia sp	**	*	*	*	-	*	**	*	*	*	-
Oligochaeta											
Tubificidae sp	*	**	**	***	****	**	**	-	-	-	-
Foraminifera	-	-	-	-	-	-	-	**	-	-	-
Decapoda	-	-	*	*	-	-	*	-	-	-	-
Mysids	-	-	-	-	-	-	*	-	-	-	-
Brittle star	-	-	-	-	-	-	*	-	-	-	-
Others	-	-	-	-	-	-	-	-	-	-	*

Table 3.6 Macrobenthic density in the sampling sites during monsoon (- <100 ind.m^{-2}, **100 to <500 ind.m^{-2}, ***500<1000 ind.m^{-2}, ****1000<3000 ind.m^{-2}, *\$-3000<7000 ind.m^{-2})*

	Estuarine stations							Coastal stations			
	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**
Polychaeta											
Pisione sp	-	-	-	-	-	-	****	-	-	-	-
Amphinomea rostrata	-	-	-	-	-	*	-	-	-	-	-
Ophiodromous sp	-	-	-	-	-	-	-	-	*	-	-
Glycera longipinnis	-	*	*	-	-	-	-	-	-	-	-
Goniada emerita	-	-	-	-	-	-	-	-	*	-	-
Sigambra parva	*	*	*	*	**	*	*	-	-	-	-
Sigambra sp	-	-	-	-	-	*	-	-	-	-	*
Nephtys oligobranchia	-	-	*	*	*	*	-	-	-	-	-
Nephtys dibranchia	-	-	-	-	-	-	-	-	-	-	-
Nephtys polybranchiata	-	-	-	-	-	*	-	-	-	-	-
Nephtys lyrochaeta	-	-	-	-	*	-	-	-	-	-	-
Nereis sp	-	-	-	-	-	*	*	-	*	-	-
Dendronereis estuarina	**	*	-	*	-	*	-	-	-	-	-
Lycastis indica	*	-	-	-	-	-	-	-	-	-	-
Cirratulus cirratus	-	-	-	-	*	-	-	-	*	-	-
Cirratulus filiformis	-	-	-	-	-	*	-	-	-	-	-
Dodecaceria sp	-	-	-	-	-	*	-	-	-	-	-
Caulleriella capensis	-	***	-	*	*	*	-	-	*	-	*
Paraprionospio pinnata	-	***	-	-	-	-	-	-	****	**	*
Prionospio cirrifera	*	***	*	**	*	*	-	-	*	-	-
Prionospio cirrobranchiata	-	-	-	*	-	-	-	-	-	-	-
Prionospio polybranchiata	-	*	-	-	-	-	-	-	-	-	-

Laonice cirrata	-	*	-	*	-	-	-	-	-	-	-
Scolelepis squamata	-	-	-	*	-	-	-	-	*	-	-
Prionospio sp	-	-	*	*	*	-	-	-	*	-	-
Boccardia polybranchia	-	-	-	-	-	**	*	-	-	-	-
Spionid sp	-	-	-	-	-	*	-	*	*	*	-
Mediomastus capensis	***	**	*	**	**	-	-	-	-	-	*
Capitella capitata	*	-	-	-	*	-	-	-	-	-	-
Paraheteromastus tenuis	**	*	*	*	*	-	-	-	-	-	-
Capitellid sp	*	-	*	*	*	-	-	-	*	-	-
Lumbriconereis impatiens	-	-	-	-	-	-	-	*	*	-	-
Lumbriconereis latreilli	-	-	-	-	-	-	*	*	*	*	*
Lumbriconereis sp	-	-	-	-	-	-	-	-	*	*	-
Ninoe pulchra	-	-	-	-	-	-	-	-	-	-	*
Diopatra neapolitana	-	*	-	-	-	-	*	-	-	-	-
Cossura coasta	-	-	*	-	-	*	-	*	*	-	-
Aricidea longobranchiata	-	-	-	-	*	-	-	-	-	-	*
Aricidea capensis	-	-	-	-	-	-	-	-	-	-	*
Owenia fusiformis	-	-	-	-	-	-	-	-	-	*	-
Megelona capensis	-	-	-	-	-	-	-	-	*	*	*
Pectinaria sp	-	-	-	-	-	-	*	-	*	-	-
Amphipoda											
Ampelisca sp	-	*	-	*	-	-	-	-	-	-	*
Corophium triaenonyx	-	*	-	**	-	-	*	-	-	-	-
Photis digitata	*	*	-	-	-	-	*	-	-	-	-
Eriopisa chilkensis	*	*	-	-	*	-	*	-	-	-	-
Melita zylanica	-	*	-	*	*	-	-	-	-	-	-
Gammaropsis	-	*	-	-	-	*	-	-	-	-	-
Gammarid sp	-	-	-	*	-	-	-	-	-	-	-
Cheriophotis megacheles	*	*	-	-	-	-	-	-	-	-	-

Perioculodes longimanus	-	-	-	*	-	-	-	-	-	-	-
Leucothoe sp	-	-	-	-	-	-	*	-	-	-	-
Caprellid sp	**	**	-	-	-	-	-	-	-	-	-
Amphipod sp	-	*	-	*	-	*	**	-	-	-	-
Isopoda											
Anthuridae	-	*	-	-	-	-	-	-	-	-	-
Cirolana sp	-	*	*	-	-	*	*	-	-	-	-
Tanaidacea											
Apseudus chilkensis	-	*	*	*	*	*	*	-	-	-	-
Mollusca											
Littorina littorea	-	*	-	-	-	-	-	-	-	-	-
Gastropoda sp	-	-	-	-	-	-	-	*	-	*	-
Umbonium sp	-	-	-	-	-	-	*	-	-	-	-
Bivalvia sp	*	*	-	*	*	-	*	*	*	*	*
Dentalium sp	-	-	-	-	-	-	-	-	*	-	-
Oligochaeta											
Tubificidae sp	*	*	**	**	*$	* *	-	-	-	-	-
Foraminifera	-	-	-	-	-	-	-	** **	-	-	-
Decapoda	-	-	-	-	*	-	*	-	-	-	-
Cumacea	*	*	-	-	-	-	*	-	*	-	-
Chironomid	*	-	-	-	-	-	-	-	-	-	-
Brittle star	-	-	-	-	-	-	-	-	*	-	-
Nematod	*	-	-	*	-	-	*	-	-	-	-
Others	-	-	-	-	-	-	-	-	-	*	-

Table 3.7 Macrobenthic density in the sampling sites during post-monsoon (- <100 ind.m^{-2}, **100 to <500 ind.m^{-2}, ***500<1000 ind.m^{-2}, ****1000<3000 ind.m^{-2}, *$-3000<7000 ind.$m^{-2}$)*

	Estuarine stations							Coastal stations			
	1	2	3	4	5	6	7	8	9	10	11
Polychaeta											
Aphroditidae sp	-	-	-	-	-	-	*	-	**	-	-
Etone sp	-	-	-	-	-	-	*	-	-	-	-
Pisione sp	-	*	-	-	-	-	**	-	-	-	-
Glycera longipinnis	-	*	*	-	*	-	-	-	-	-	-
Goniada emerita	-	*	-	-	-	-	-	*	*	*	*
Sigambra parva	-	*	*	*	*	*	-	-	-	-	-
Nephtys oligobranchia	-	-	*	*	*	*	-	*	*	*	*
Nephtys polybranchiata	-	-	*	-	-	-	-	-	*	-	-
Nereis sp	-	-	*	*	*	*	-	-	-	-	-
Dendronereis estuarina	*	*	-	*	-	*	-	-	-	-	-
Lycastis indica	*	*	-	*	-	-	*	-	*	-	-
Cirratulus cirratus	-	-	-	*	*	*	-	-	-	-	-
Cirratulus filiformis	-	*	*	**	*	*	-	-	-	-	-
Dodecaceria sp	-	**	-	-	-	-	-	*	-	-	*
Caulleriella capensis	-	*	*	*	*	*	-	-	-	**	*
Paraprionospio pinnata	-	*	-	-	-	-	-	**	**	**	***
Prionospio cirrifera	**	**	*	*	*	-	*	-	-	-	-
Prionospio cirrobranchiata	-	-	-	*	-	-	-	-	-	-	-
Polydora ciliata	-	-	-	-	-	-	-	-	*	-	*
Prionospio sp	-	-	-	-	*	-	-	-	-	-	-
Boccardia polybranchia	-	*	-	-	-	*	-	*	-	-	*
Spionid sp	-	*	-	-	-	-	-	*	-	-	-
Mediomastus capensis	**	**	*	*	**	-	-	*	*	*	*

Capitella capitata	*	-	-	*	*	-	-	-	-	-	-
Paraheteromastus tenuis	*	*	-	-	-	-	-	*	-	-	-
Capitellid sp	-	*	-	-	*	-	-	-	-	*	-
Lumbriconereis impatiens	-	*	-	-	-	-	*	-	-	-	-
Lumbriconereis latreilli	-	*	-	-	-	-	**	*	-	*	*
Lumbriconereis sp	-	*	-	-	-	-	*	-	-	-	-
Ninoe pulchra	-	-	-	-	-	-	-	*	-	*	*
Diopatra neapolitana	-	***	-	-	-	*	****	-	*	-	-
Cossura coasta	-	-	-	-	*	-	-	*	*	*	*
Owenia fusiformis	*	*	*	-	-	*	-	-	*	-	-
Megelona capensis	-	-	-	-	-	-	-	-	*	-	-
Magelona cinta	-	-	-	-	-	-	-	*	*	-	*
Pectinaria sp	-	-	-	-	-	-	**	-	*	-	-
Streblosoma sp	-	-	-	-	-	-	-	-	-	-	-
Syllid sp	-	-	-	*	*	*	**	-	*	-	-
Chaetopteridae sp	-	-	-	-	-	-	-	*	*	-	-
Sternaspis scutata	-	-	-	-	-	-	-	*	-	*	-
Amphipoda											
Ampelisca sp	-	-	-	*	-	-	**	-	*	-	-
Corophium triaenonyx	*	-	-	-	-	-	-	-	-	-	-
Photis digitata	**	-	-	-	-	-	*	*	-	-	-
Eriopisa chilkensis	*	*	*	*	*	*	-	-	-	-	-
Melita zylanica	*	*	*	-	*	-	-	-	*	-	*
Gammarid sp	-	*	-	-	-	-	-	-	*	-	-
Cheriophotis megacheles	**	*	-	-	-	-	**	*	*	-	-
Perioculodes longimanus	-	-	-	*	-	-	*	-	-	-	-
Leucothoe sp	-	*	-	-	-	-	-	-	-	-	-
Platyischnopus sp	-	-	-	-	-	-	-	-	*	-	-
Caprellid sp	*	*	*	-	-	-	-	-	-	-	-
Amphipod sp	*	**	-	*	-	-	-	-	-	-	-
Isopoda											

Anthuridae	-	*	-	-	-	-	*	-	-	-	-
Cirolana sp	-	-	-	-	-	*	-	-	-	-	-
Isopod sp	-	*	-	-	-	-	-	-	-	-	-
Tanaidacea	-	-	-	-	*	-	**	-	-	-	-
Apseudus chilkensis	-	*	*	**	-	*	*	-	-	-	-
Mollusca											
Littorina littorea	-	*	-	-	-	-	*	-	*	-	-
Gastropoda sp	-	-	-	-	*	-	-	-	-	-	-
Villorita cyprinoides	-	*	-	-	-	-	**	*	**	*	*
Pholas orientalis	-	-	-	-	-	-	-	-	**	-	-
Perna sp	-	-	-	-	-	-	*$	-	-	-	-
Bivalvia sp	*	*	-	*	*	-	-	-	-	-	-
Oligochaeta											
Tubificidae sp	*	*	**	***	****	*	*	-	-	-	-
Foraminifera	-	-	-	-	-	-	-	*	-	-	-
Decapoda	-	-	*	-	-	-	*	-	-	-	-
Cumacea	-	-	-	-	*	*	*	*	*	-	-
Mysids	*	-	-	*	*	-	-	-	-	-	-
Harpacticoid	-	-	-	-	-	-	**	-	-	-	-
Brittle star	-	-	-	-	-	-	*	*	*	-	-
Sea anemone	-	-	-	-	-	-	*	-	-	-	-
Nematods	-	-	-	-	-	-	*	-	*	-	*
Others	-	-	-	-	-	-	-	*	*	*	*
Unidentified	-	*	-	-	-	-	-	-	-	-	-

*****$$*****

Chapter 4

ECOLOGY AND POPULATION STRUCTURE OF A TUBE BUILDING AMPHIPOD *CHELICOROPHIUM MADRASENSIS,* NAYAR IN THE COCHIN ESTUARY

Contents

4.1 INTRODUCTION

Amphipods, the diverse and dominant group of pericaridan crustaceans, are widely distributed in a variety of habitats such as marine, freshwater and terrestrial environments. Amphipods are characterized by a laterally compressed body with seven distinct thoracic segments. They play multifaceted role in the benthic trophodynamics as herbivores, omnivores, carnivores and opportunistic feeders and are adapted to change their feeding modes according to the food availability in their habitat. They include a diverse community of tube dwellers, nestlers, algal inhabitants, commensals, and fossorial organisms. Amphipods are known to brood their eggs inside the thoracic pouch and upon hatching the hatchlings resemble miniature adults in morphological characteristics. Being an important prey of fishes, birds and larger invertebrates they play a pivotal role in the secondary production of the benthic food web. Besides being an ecologically significant and numerically abundant benthic community, these organisms are characterized by extreme sensitivity to a wide variety of toxicants and pollutants and hence are often considered as suitable environmental indicators (Hart and and Fuller, 1979; Ingole et al., 2009). They are widely employed in ecotoxicology studies related to polycyclic aromatic hydrocarbons (PHAs), polychlorinated biphenyls (PCBs), organochlorine pesticides (DDT), heavy metals, and ammonium or nitrite contaminants in the environment (Anderson et al., 2008; Ramos-Gomez et al., 2009; Riba et al., 2003), in sediment toxicity testing (Nendza, 2002), and also used for the assessment of organic enrichment (Esselink et al., 1989).

In general, the Order Amphipoda is comprised of four suborders among which the suborder Caprellidea occurs on solid surfaces; Hyperiidea, are pelagic parasites and commensals on marine macrozooplankton; Ingolfiellidea, includes wormlike interstitial amphipods and Gammaridea; forms the most abundant suborder having both pelagic and benthic inhabitants (Chapman, 2007). Being the most diverse and abundant super order, gammarid amphipods are remarkable for their high population density and species diversity. Approximately 7900 species of gammarid amphipods have been described till date (Foster et al., 2009). They exhibit sexual dimorphism and have morphologically distinguishable male and females.

Among the suborder Gammaridae, the subfamily Corophiinae are world widely distributed (Bousfield and Hoover, 1997) and are easily distinguishable by their gnathopods 1 and 2 modified to form a sieve with dense sieving setae on the posterior margins of its carpus and ischium (Myers and Lowry, 2009). Species belonging to Family Corophiidae are mostly free living and are highly remarkable for their tube building activities in muddy estuaries and on sessile objects and pilings in coastal harbors. Though most of the species living in estuarine zones are able to tolerate subtle variations in salinity of waters around their vicinity, only a few species live in fresh or nearly fresh water conditions (Crawford, 1937). Corophideans live in tubes constructed by lipoprotein threads secreted from specialized glands in their 3rd and 4thpereopods. These benthic amphipods have a potential role in estuarine food web (Conlan, 1994; Hawkins, 1985) as they form a major prey item of shorebirds (Hilton et al., 2002; Wilson, 1989, &1994; Wilson and Parker, 1996), demersal fishes (Mattila and Bonsdorff, 1989) and other crustaceans (Eriksson et al., 2005). Moreover, the bioturbation activities of the corophid amphipod increase the nitrification and denitrification rates in the sediments (Gilbert et al., 1998; Henriksen et al., 1983). They are also known to exert a considerable effect on the substratum stability of estuaries (Gerdol and Hughes, 1994a; Meadows and Tait, 1989). Some tube dwelling species of the genus Corophium exhibits tolerance towards metal contamination (Warwick, 2001), and sewage pollution (Lowe and Thompson, 1997). Burrowing amphipods are often known to actively avoid the polluted sediments by pumping of oxygenated water down into their burrows and tubes (De-la-Ossa-Carretero et al., 2012).

4.2 Background information on Amphipods

Earlier studies on amphipods have provided ample information on information on their systematics and taxonomy, which helped greatly in identifying the species inhabiting different geographic regions and habitats (Barnard and Karaman, 1991; Barnard, 1962 & 1969; Crawford, 1937; Giles, 1885; Nayar, 1959 & 1966; Sivaprakasam, 1968). In temperate waters, extensive studies have been carried out on corophid amphipods pertaining mainly to the ecology (Beukema and Flach, 1995; Holmstrom and Morgan, 2013), life history (Chapman, 2007; Cunha et al., 2000a; Cunha et al., 2000b; Moore, 1981; Prato and Biandolino, 2006; Wilson and Parker, 1996), population dynamics (Stevens et al., 2002), reproductive patterns (Cunha et al., 2000a; Rajagopal et al., 1999), feeding biology (Gerdol and Hughes, 1994b; Shillaker and Moore, 1987) and habitat preferences (Boyden and Little, 1973; Gee, 1961; Meadows, 1964; Watkin, 1941). In addition, laboratory culturing of many amphipod species improved understanding on their life cycle, tube building, growth rates, feeding modes, breeding and development (Dixon and Moore, 1997; Fenchel et al., 1975; Nair and Anger, 1979). Further, the activity of the corophid amphipods on sediment stability (Gerdol and Hughes, 1994a; Grant and Daborn, 1994; Meadows and Tait, 1989) and other sediment properties (Gerdol and Hughes, 1994a; Jones and Jago, 1993; Limia and Raffaelli, 1997) have also been a topic of intense researches in the temperate region.

Even though, many comprehensive studies on various aspects of corophidean amphipods have been given much focus in the temperate waters, there is very little information regarding the ecology of these important amphipod community from the tropical waters (Krishnan and John, 1974; Nair et al., 1983; Nayar, 1959). Since the last five decades, significant studies have been carried out on the ecology, tube building activity, and life history patterns of the corophidean, *Corophium triaenonyx* along Indian waters (Rao and Shyamasundari, 1963; Shyamasundari, 1972, 1973 & 1976). As far as the CE is concerned, very limited studies have been carried out on benthic amphipods so far (Nair et al., 1983; Aravind et al., 2007). As a pioneer attempt, Nair et al. (1983) have brought out baseline information on ecology and population dynamics of gammarid amphipods in the CE. Thereafter, Aravind et al. (2007) have conducted experimental studies on gammarid amphipod, *Eriopisa chilkensis* to understand its life history and ecological aspects, aiding greatly in improving the knowledge regarding this important but yet less investigated benthic taxa in the CE.

During the monthly sampling in the CE for understanding the ecology of macrobenthos, a significantly higher density of macrobenthic fauna has been encountered from a particular site throughout the year. Microscopic analysis using the relevant identification keys revealed that a tubecolous amphipod species, *Chelicorophium madrasensis* (Plate 1) was mainly responsible for this remarkable higher density. This particular species was first reported by Nayar (1950) from the Madras coast and thereafter from the Songkhla Lake, Thailand (Wongkamhaeng et al., 2015). As far as the southwest coast is concerned, no prior information is available on the ecology and population dynamics of *C. madrasensis* so far. Though there are a very few reports on *C. madrasensis,* as mentioned above, all of them were mainly dealt with its taxonomy. Since there was very scanty information on the ecology and population dynamics of *C. madrasensis* worldwide, an attempt has been made to study the ecological aspects, population structure of *C. madrasensis* using the data collected from the particular sampling site, located in the northern part of the CE.

Plate 1 Microphotograph of the amphipod *Chelicorophium madrasensis* (Male and female specimens)

4.3 Sampling Strategy and Methods

In order to understand the community structure of the macrobenthic fauna and to study the ecology and population structure of the amphipod species *Chelicorophium madrasensis*, monthly observations (January-December 2011) were carried out towards the northern part (North of panambukadu) of CE (Fig. 4.1). Being a tropical monsoonal estuary, the CE is heavily influenced by the Indian Summer Monsoon, and therefore the sampling periods were categorized based on the availability of monsoonal precipitation and runoff. Pre-monsoon (PRM- February-May) is characterized by least rainfall, monsoon (MN-June-September) with heavy rainfall, and post-monsoon (PM-October- January) as the transitional period with intermediate rainfall.

To understand the prevailing environmental conditions, bottom water samples were taken using Niskin sampler (5L capacity, Hydro-Bios). A portable CTD (Hydrobios) was used for measuring temperature in the water column. Water quality parameters such as salinity, pH, dissolved oxygen (DO), biological oxygen demand (BOD), nutrients and suspended particulate matter (SPM) were analyzed following the standard procedure. Sediment samples for macrobenthic fauna were collected using Van-Veen grab ($0.05m^{-2}$). Subsamples of sediments were further used for estimating benthic chlorophyll *a*, organic carbon content and texture. The detailed sampling and analytical methods followed for the analysis of above mentioned parameters were described in Chapter 2, Materials and methods. Taxonomic identification of amphipods and other macrobenthic fauna collected from the study area was carried out by using standard literatures and monographs. Detailed microscopic analysis was adopted for studying the morphological characters of *C. madrasensis.* Statistical analyses such as one-way analysis of variance (ANOVA), Karl Pearson's correlation, and Redundancy analysis were carried out using appropriate statistical softwares (detailed methodology in Chapter 2 Materials and methods) to substantiate the results of the study.

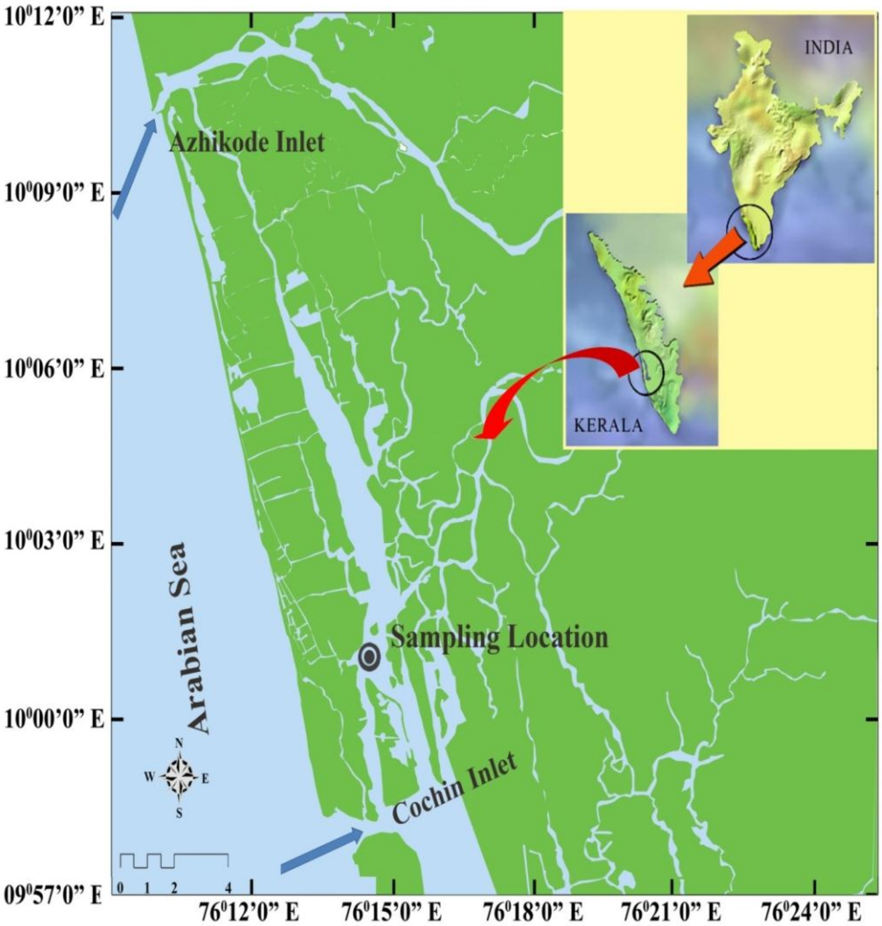


Figure 4.1 Map showing sampling location in the Cochin estuary

4.4 RESULTS

4.4.1 Environmental Parameters

4.4.1.1 Temperature

Bottom temperature varied from 28 to 32ºC in the sampling location throughout the year. Significant seasonal variation ($p<0.05$) in bottom water temperature was evident in the sampling period (Table 4.1), with relatively lower temperature recorded during MN (av. 28.6 ±0.48 ºC) and higher during PRM season (av.31.13±0.75ºC) (Fig. 4.2).

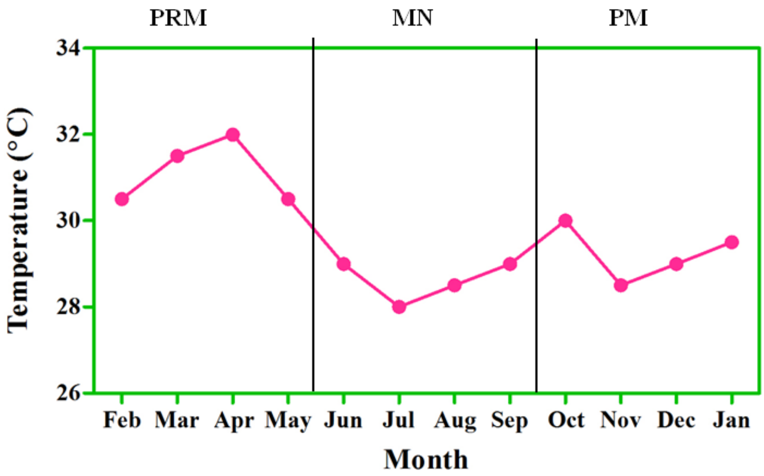


Figure 4.2 Distribution of bottom water temperature in the sampling location

4.4.1.2 Salinity

Salinity of bottom water ranged from 0.95 to 19.63 during the study period. Salinity exhibited a significant seasonal variation ($p<0.05$) in the study region (Table 4.1). Relatively higher salinity was recorded during PM season (av.11.6±6.4), with prominent monthly variation compared to MN (av.1.1±0.21) and PRM (av.11.3 ±2.4) respectively (Fig.4.3).

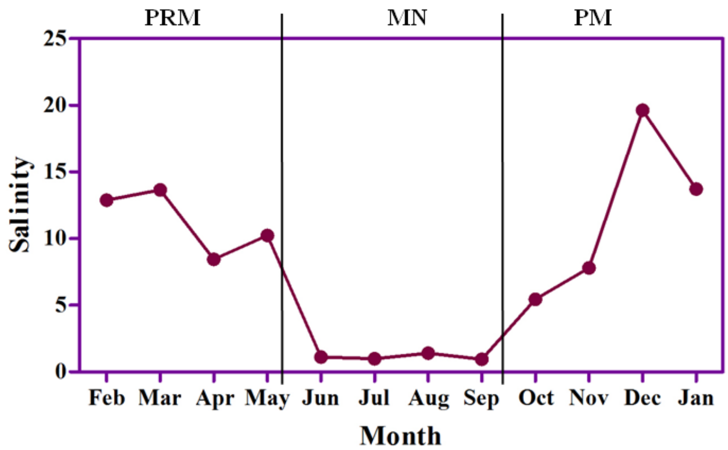


Figure 4.3 Distribution of bottom water salinity in the sampling location

4.4.1.3 pH

pH of bottom water ranged from 7.1 to 8.5 during the study period. Among the seasons pH was recorded high during PRM (av. 7.60 ±0.6) compared to MN (av.7.56±0.25) and PM (av.7.59 ±0.2) respectively (Fig.4.4). pH exhibited insignificant ($p>0.05$) seasonal variation during the study period.

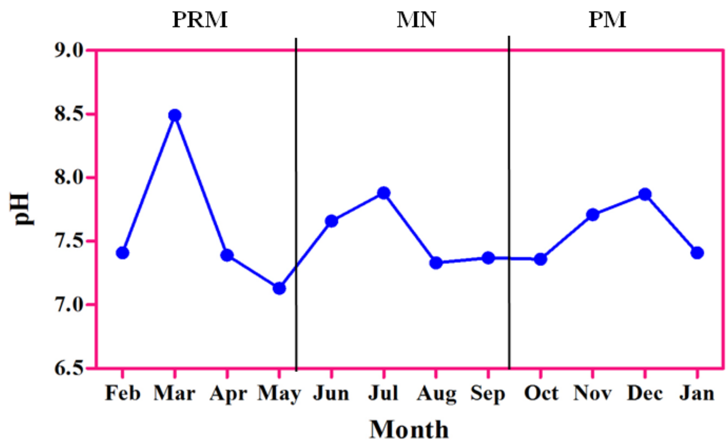


Figure 4.4 Distribution of bottom water pH in the sampling location

4.4.1.4 Dissolved Oxygen

Dissolved oxygen varied between 0.34 and 6.09mg/L during the study period. Mean DO concentration was higher during MN season (av. 5.45±0.39 mg/L) relative to PRM (av. 4.19±1.6 mg/L) and PM (av. 3.08±1.97 mg/L), with insignificant ($p>0.05$) seasonal variation (Table 4.1). Among seasons DO showed less monthly variation during MN relative to PM and PRM (Fig. 4.5).

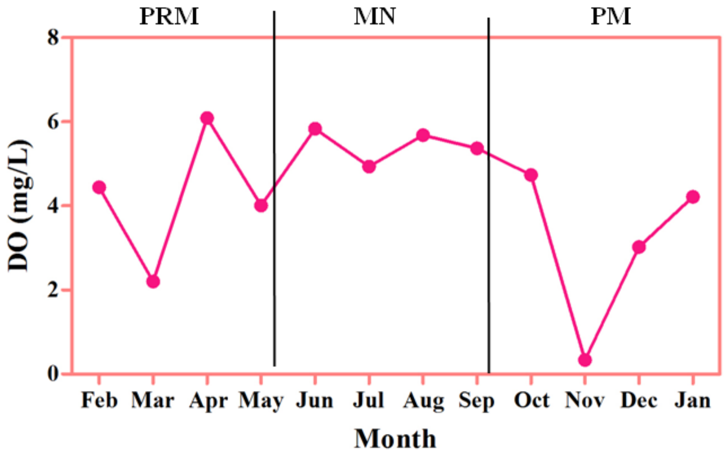


Figure 4.5 Distribution of bottom water dissolved oxygen in the sampling location

4.4.1.5 Biological Oxygen Demand

BOD ranged from 0.02 to 3.56 mg/L during the study period (Fig. 4.6). BOD exhibited insignificant ($p>0.05$) seasonal variation in the sampling location (Table 4.1), with lower concentration recorded during MN (av. 0.96±0.93mg/L) and higher during PRM season (av. 2.31±0.99 mg/L).

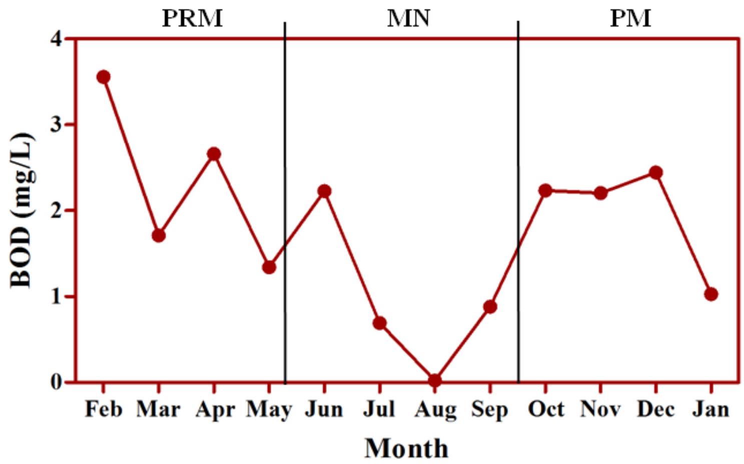


Figure 4.6 Distribution of biological oxygen demand in the sampling location

4.4.1.6 Suspended Particulate Matter

Suspended particulate matter varied from 6.4 to 50.4mg/L during the study period (Fig. 4.7). SPM exhibited statistically insignificant ($p>0.05$) variation between seasons (Table 4.1).

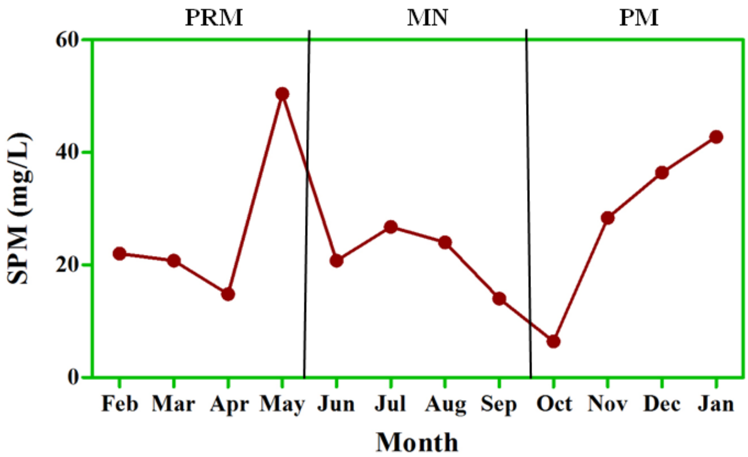


Figure 4.7 Distribution of suspended particulate matter in the sampling location

Lower SPM concentration was observed during MN season (av. 21.4±5.5 mg/L) while it was higher during PM (av. 28.5±15.9 mg/L). During PRM and PM, monthly variation in SPM concentration was high compared to MN

4.4.1.7 Nutrients

4.4.1.7.1 Nitrate

Nitrate concentration varied from 4.78 to 30.4 µM during the study period. Nitrate exhibited insignificant ($p>0.05$) seasonal variation in the sampling location (Table 4.1). Nitrate was lower during MN (av. 10.76±2.6 µM) while high concentration was recorded during PRM (av. 12.07±12.3 µM) (Fig. 4.8).

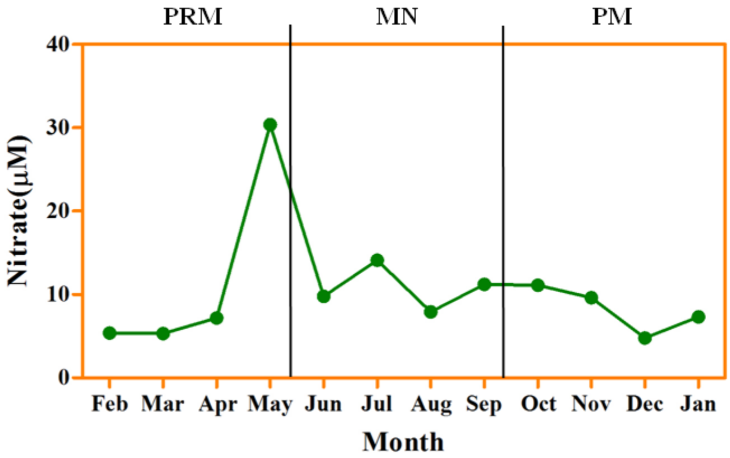


Figure 4.8 Distribution of bottom water nitrate in the sampling location

4.4.1.7.2 Nitrite

Nitrite varied between 0.04 and 0.71 µM during the study period, with significant ($p<0.05$) seasonal variation (Table 4.1). Lower nitrite concentration was noticed during PRM season (av.0.09±0.04µM) and higher recorded during MN (av.0.49±0.2µM). Monthly variation in nitrite was high during PM season, compared to MN and PRM (Fig. 4.9)

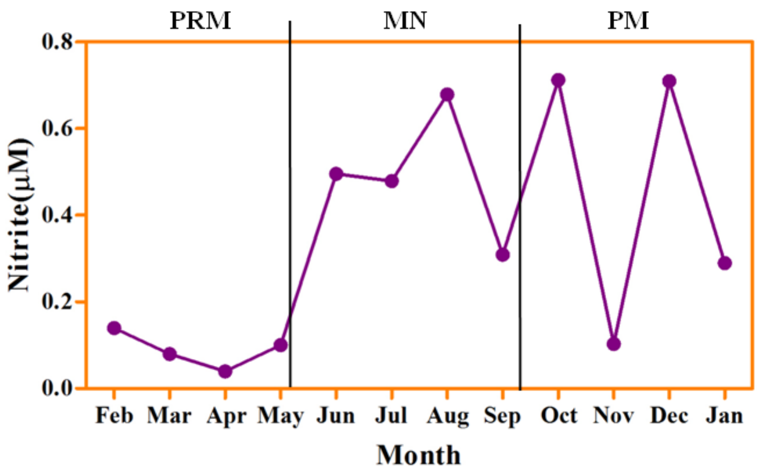


Figure 4.9 Distribution of bottom water nitrite in the sampling location

4.4.1.7.3 Ammonia

Ammonia concentration varied from 1.51 to 32.9 µM during the study with insignificant ($p>0.05$) seasonal variation (Table 4.1). Lower concentration of ammonia was recorded during MN season (av. 3.21±1.7 µM) and higher during PRM (av. 12.20±14.2 µM). Distinct monthly variation in ammonia concentration was evident during PRM comparative to other season (Fig. 4.10).

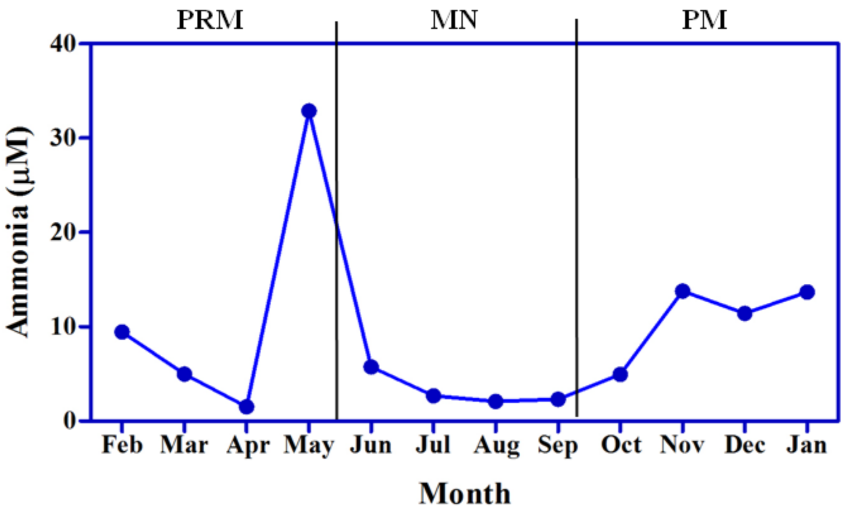


Figure 4.10 Distribution of bottom water ammonia in the sampling location

4.4.1.7.4 Phosphate

Phosphate concentration ranged from 0.23 to 3.01µM in the study region. Significant seasonal variation ($p<0.01$) in phosphate was observed during the study period (Table 4.1). Phosphate concentration was lower during PRM season (av. 0.47±0.2 µM) and was relatively high during MN (av. 2.53±0.3 µM) (Fig. 4.11).

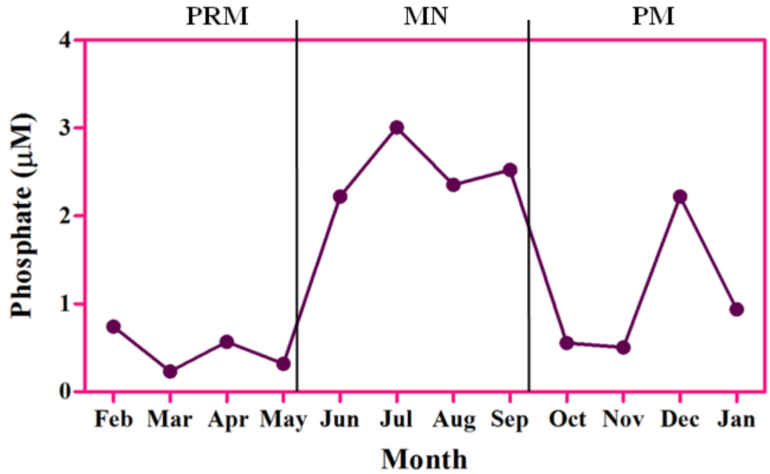


Figure 4.11 Distribution of bottom water phosphate in the sampling location

4.4.1.7.5 Silicate

Silicate concentration in the sampling location varied from 2.2 to 151.2 µM (Fig. 4.13). Relatively lower silicate concentration was recorded during PRM season (av. 12.4±7.2 µM) and higher during MN (av. 61.0±62.9 µM), with insignificant seasonal variation ($p>0.05$) (Table 4.1). Distinct monthly variation was evident in silicate during MN compared to other seasons (Fig. 4.12).

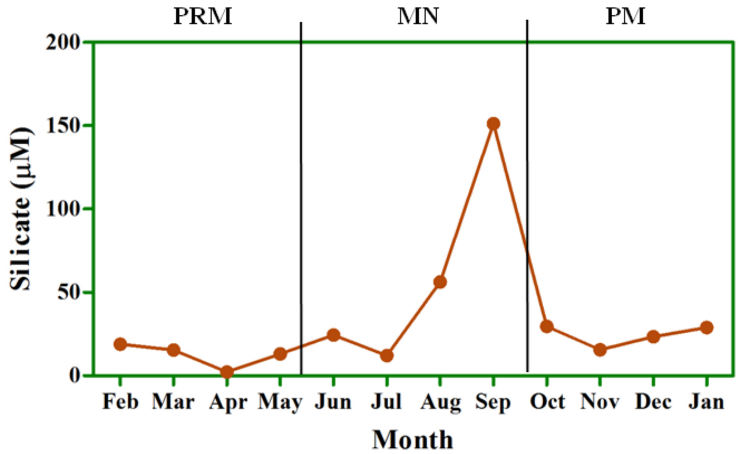


Figure 4.12 Distribution of bottom water silicate in the sampling location

4.4.2 Sediment characteristics

4.4.2.1 Sediment texture

In general, finer fractions of the sediment (clay and silt) dominated in the sampling location. While the percentage of silt increased considerably during December and texture became clayey silt. Sand fraction varied from 0.02 to 24.85% with lower percentage observed during PRM season (av.2.02±3.7%) and higher during PM (av.6.62±12.2%) (Fig. 4.13). Percentage of silt varied from 9.27 to 69.5% during the study. Low silt content observed during PRM (av. 24.87±9.3%) and high silt was recorded during PM season (av. 33.86±24.1%) (Fig. 4.13). Clay fraction varied from 7.25 to 87.21% in the sampling location having lower percentage observed during PM (av.59.52±24.8%) and higher during PRM season (av.73.13±7.3%). No significant seasonal variation was observed for the sediment texture during the study (Table 4.1).

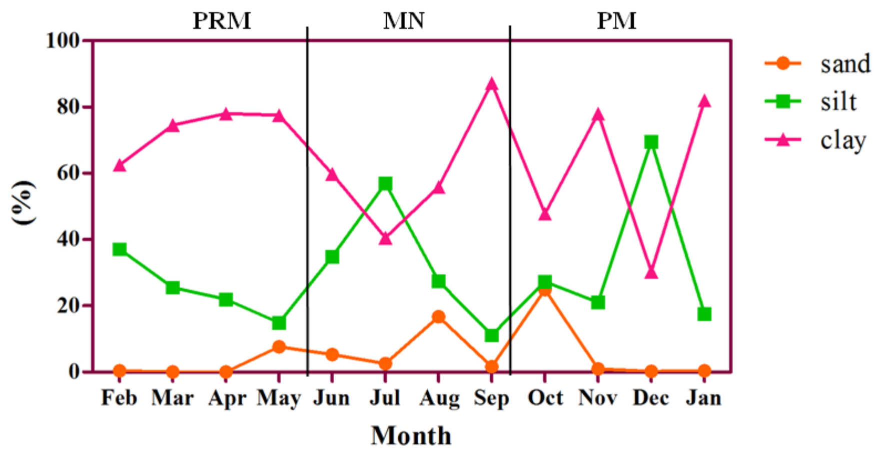


Figure 4.13 Distribution of sediment texture in the sampling location

4.4.2.2 Sediment Organic carbon

Organic carbon content of the sediment ranged from 2.0 to 26.91 mg/g during the study, with insignificant ($p>0.05$) seasonal variation (Table 4.1). Lower concentration was recorded during MN season, with an average value of 11.96±8.5mg/g and higher recorded during PM with an average value of 18.82±6.3 mg/g (Fig. 4.14). Monthly variation in organic carbon was evident during PRM and MN in the sampling location (Fig. 4.14).

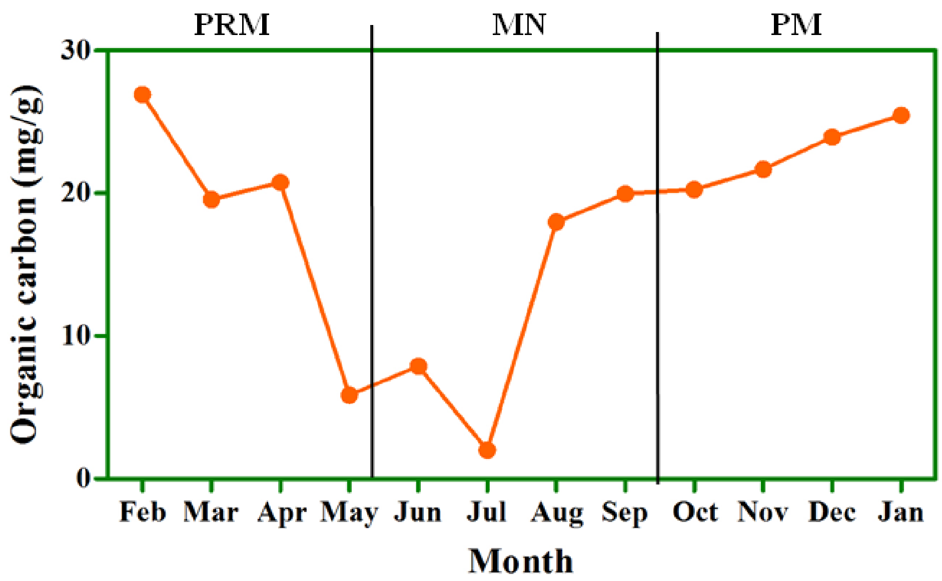


Figure 4.14 Distribution of sediment organic carbon in the sampling location

4.4.2.3 Sediment chlorophyll/ benthic chlorophyll *a*

Benthic chlorophyll *a* varied between 4.01 to 136.03 mg/g in the sampling location. Benthic chlorophyll *a* exhibited significant seasonal variation ($p<0.01$) during the study period (Table 4.1), having lower concentration recorded during MN season (av. 22.63±18.5 mg/g) and higher during PRM (av.96.60±43.2mg/g). Monthly fluctuation in benthic chlorophyll *a* was more visible during PRM compared to other seasons (Fig. 4.15).

Relatively higher concentration of benthic chlorophyll *a* (av. 69.4mg/g) was recorded in the sampling location when compared to other study locations in CE (Fig. 4.16) and it was more than twice that observed in other sites.

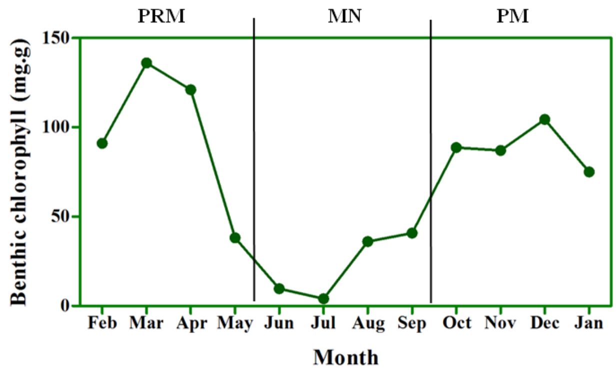


Figure 4.15 Distribution of benthic chlorophyll *a* in the sampling location

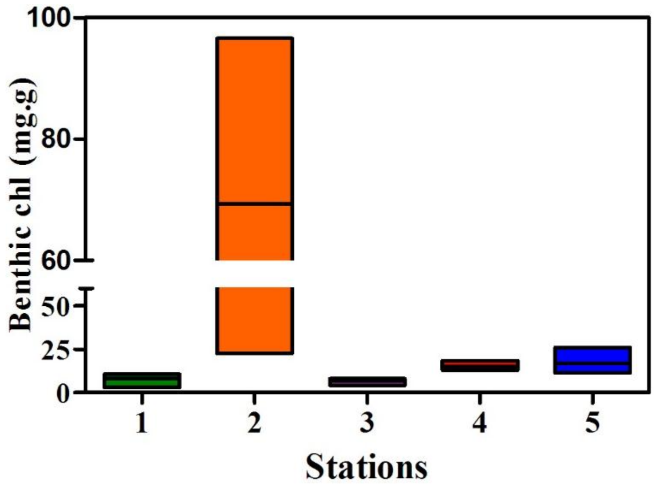


Figure 4.16 Comparison of benthic chlorophyll *a* with other regions of Cochin estuary (Stations 1- Chittoor, 2- north of Panambukadu (present study), 3- Vypeen, 4- Vallarpadam, 5- Bolgatty)

4.4.3 Macrobenthic community structure (at north of Panambukadu)

4.4.3.1 Macrobenthic density

Macrobenthic density at the sampling location (north of Panambukadu) varied between 49860 and 2149440 ind.m^{-2} in the sampling location. Higher density was observed during PRM (av. 1309500±592534 ind.m^{-2}) followed by PM (av. 470261±296028 ind.m^{-2}) and MN (av. 272016±173402 ind.m^{-2}) (Fig. 4.17). Significant ($p<0.05$) seasonal variation in density was apparent at this location during the study period (Table 4.1).

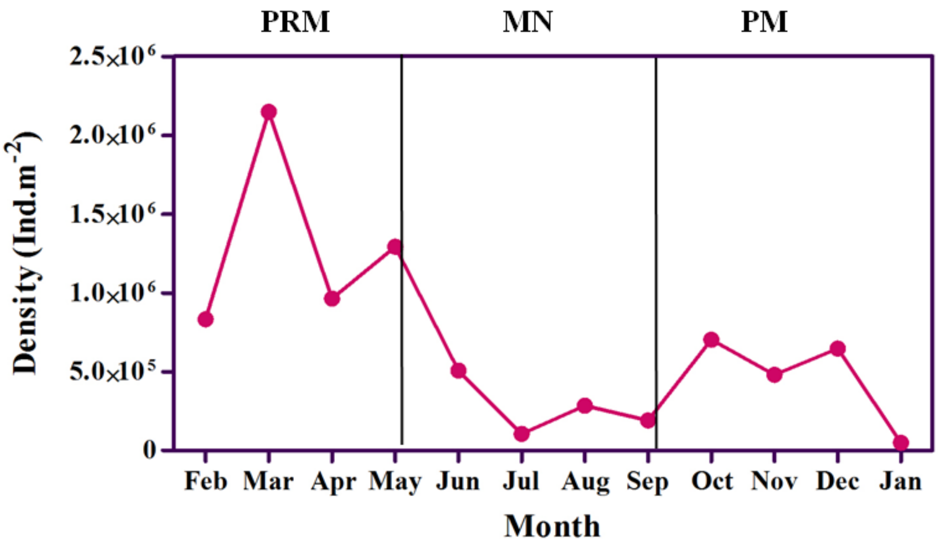


Figure 4.17 Distribution of macrobenthic density in the sampling location

Compared to other stations in CE, the macrobenthic density was very high at this location throughout the study (Fig 4.18). Average macrobenthic density at the sampling location was 683926 ind.m^{-2}, while it was 1817 ind.m^{-2} (varied from 6 to 18080 ind.m^{-2}) at other locations in the CE.

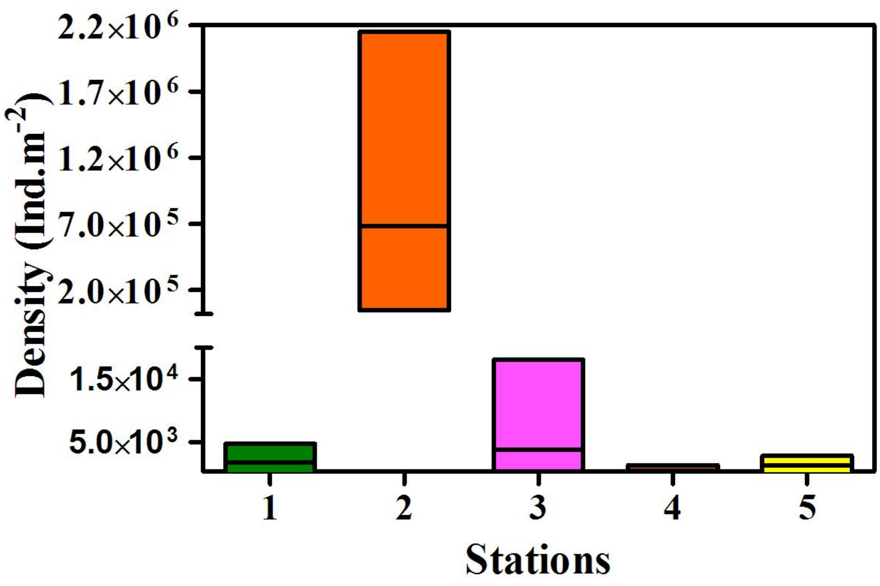


Figure 4.18 Comparison of macrobenthic density with other regions of Cochin estuary (Stations 1- Chittoor, 2- north of Panambukadu (present study), 3- Vypeen, 4- Vallarpadam, 5- Bolgatty)

4.4.3.2 Macrobenthic biomass

Macrobenthic biomass ranged between 33.98 and 1507.68 $g.m^{-2}$ in the study location (Fig. 4.19) with significant variation ($P<0.05$) between seasons. Comparatively lower biomass was recorded during PM season (av.338±302.2 $g.m^{-2}$) and higher observed during PRM (av. 1018.62±422.8 $g.m^{-2}$) (Table 4.1).

Similar to macrobenthic density, macrobenthic biomass was also very much high at this site compared to other locations in CE (Fig. 4.20). Average macrobenthic biomass at the sampling location was 567 $g.m^{-2}$ while it was below 10 $g.m^{-2}$ (varied from 0.28 to 57 $g.m^{-2}$) at other locations.

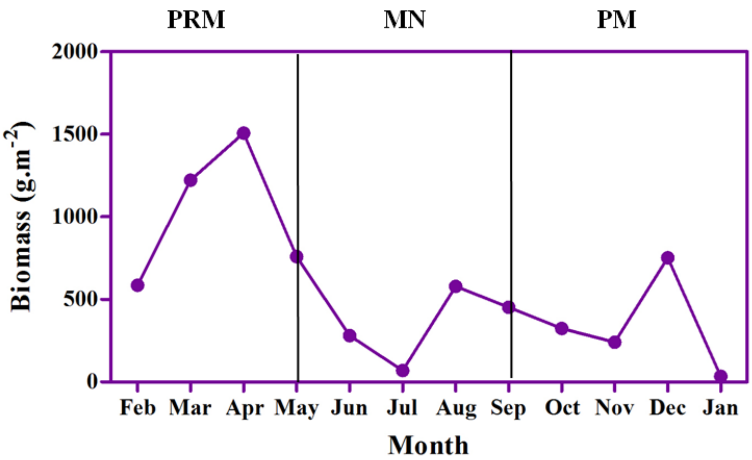


Figure 4.19 Distribution of macrobenthic biomass in the sampling location

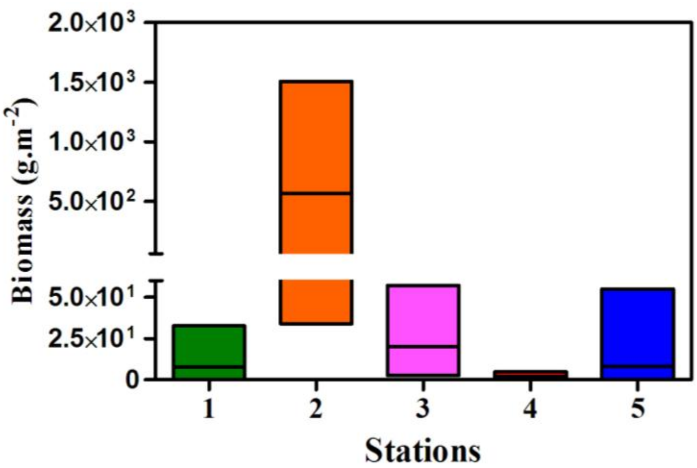


Figure 4.20 Comparison of macrobenthic biomass with other regions of Cochin estuary (Stations 1- Chittoor, 2- north of Panambukadu (present study), 3- Vypeen, 4- Vallarpadam, 5- Bolgatty)

4.4.3.3 Macrobenthic community composition (at north of Panambukadu)

Macrobenthic fauna at the sampling location was dominated by amphipods, followed by isopods, polychaetes, tanaids, oligochaetes, turbellarians, bivalves, and nematodes (Fig. 4.21). Of the observed groups, amphipods contributed 93% to the total density followed by isopods (4.84%), while oligochaetes and polychaetes contributed very less percentage. During PRM, amphipods (98%) and isopods (1.3%) formed the major groups, whereas oligochaetes, polychaetes, nematodes, tanaids, and bivalves constituted the minor macrobenthic fauna (Fig. 4.21A). During MN, amphipods and isopods together contributed 97.4% to total density (Fig. 4.21B), whereas during PM, they contributed 98.8% to the total macrobenthic density (Fig. 4.21C).

4.4.3.4 Macrobenthic species composition

Among Amphipoda, a single species *C. madrasensis* (93.4%) dominated at this site throughout the study. *C. madrasensis* contributed 93% during PRM, 92.7% during MN, and 94.5% during PM to the total amphipod density. During PRM, *Eriopisa chilkensis* (0.13%), *Photis digitata* (0.02%), and *Melita zylanica* (0.12%) were the other amphipod species observed at the site. During MN, *Melita zylanica* (0.15%), *Eriopisa chilkensis* (0.11%), *Ampelisca* sp (0.09%), *Caprellid* sp (0.04%), and *Platyischnopus* sp (0.02%) were the other species observed in the sampling location. *Melita zylanica* (0.51%), *Photis digitata* (0.05%), *Eriopisa chilkensis* (0.03%), and *Caprellid* sp (0.02%) were the minor species observed during PM (Table 4.3). Among Isopoda, *Cirolana fluviatilis* and *Anthurid* sp were the major species observed in the study (Fig. 4.22 & Fig. 4.23). These isopod species contributed 4.04% and *1.91%* during PRM, 2.29% and 2.14% during MN, and 1.82 %, and 1.92% during PM respectively (Table 4.3).

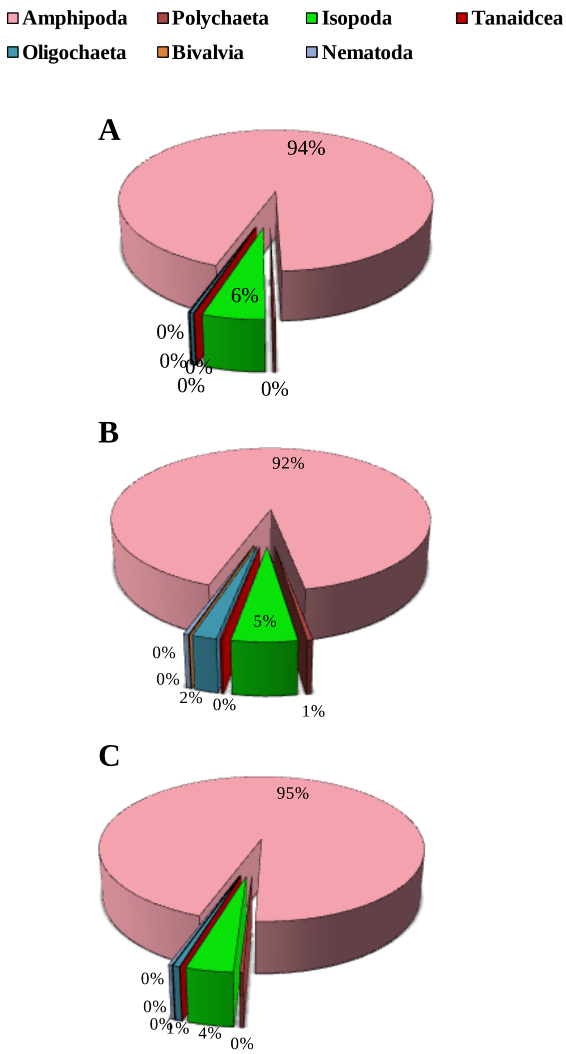


Figure 4.21 Macrobenthic compositions of groups during (A) premonsoon, (B) monsoon, and (C) post monsoon season in the sampling location

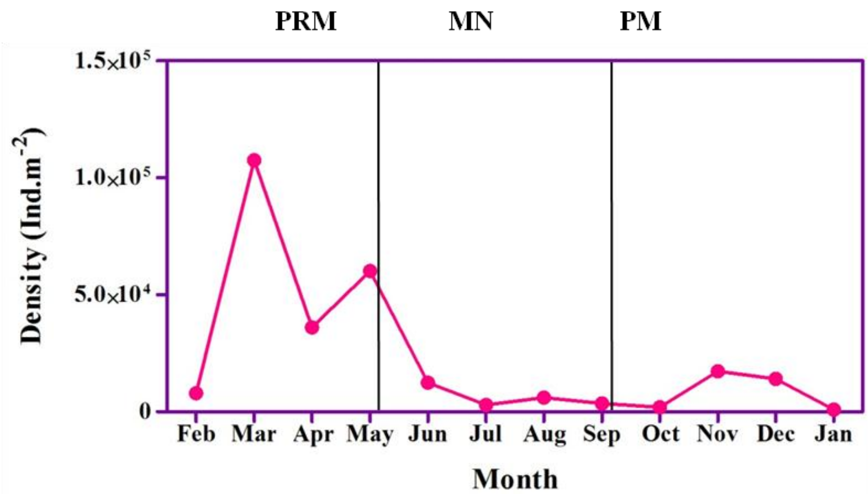


Figure 4.22 Variations in the density of *Cirolana fluviatilis* in the sampling location

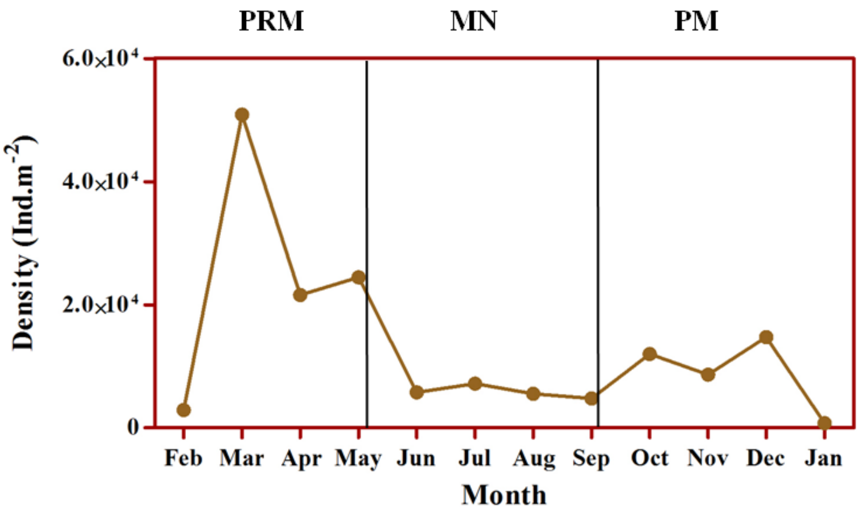


Figure 4.23 Variations in the density of *Anthurid sp* in the sampling location

Among polychaetes, species such as *Mediomastus capensis*, *Lumbriconereis*, *Lumbriconereis latreilli*, *Prionospio cirrifera*, *Nereis* sp, *Capitellid* sp, *Dendronereis estuarina. Lycastis indica, Ninoe pulchra,* and *Eunice indica* were observed in the sampling location (Table 4.3).

4.4.4 Taxonomy and population structure of *C. madrasensis*

4.4.4.1 Taxonomy and systematics of *C.madrasensis*

Phylum	**:**	**Arthropoda**
Order	**:**	**Amphipoda**
Super Order	**:**	**Gammaridae**
Family	**:**	**Corophiidae**
Subfamily	**:**	**Corophiinae**
Genus	**:**	**Corophium accepted as Chelicorophium (Bousfield and Hoover, 1997)**

The presence of characters such as short rostrum; strongly pediform and well developed antennae 2 in both sexes, strong bidentate postereo-distal process in peduncular segment; subchelate gnathopod 1; gnathopod 2 with slender propodus, short dactyl and tridentate; uropods 1 and 2 medium with distally broadening peduncles; uropod 3 with its ramus longer than peduncle with slightly broadened and setose at apex portion, medium broad sac like coxal gills on 3rd pereopods, and short sublinear brood lamellae position the specimen under the genus Chelicorophium. The distinguishing characters of the male and female are given below (Plate 1a-g). **Male**: The antennae 2, is larger and stronger than females; Black slightly oval eyes.

Antenna 1: It is reaching beyond the proximal end of the fifth joint of antennae 2; inner margin of the first joint have rather long setae and devoid of spines; the second joint of the antenna is little shorter than the first and slightly more than twice as long as the third; flagellum is shorter than the peduncle and provided with about 12 joints.

Antenna 2: Fourth joint of the antenna 2, longer than the third joint and provided with a distally produced strong forward-curving tooth below and a small tooth above and narrow teeth at the lower inside surface of the proximal end. Fifth joint is almost in same length as fourth. Flagellum composed of two unequal joints, groups of long setae present on the lower margins of the third, fourth, and fifth peduncular joints and the flagellum.

Gnathopod I: Third and fifth joints have dense setae, front margin of the fifth joint fringed with slender spines; smooth and curved seventh joint.

Gnathopod 2: The fourth joint of the appendage is fringed with two rows of extremely long setae and the seventh joint has three broad teeth.

Peraeopods: Second and fourth joints of peraeopods 1 and 2 moderately expanded; seventh and sixth joint have almost same length; Peraeopods 3 and 4 normal. Peraeopod 5 reaching beyond uropod 1.

Uropods:The peduncle of uropod 1 is provided with a row of about four spines on outer margin and three spines on the inner margin. The outer ramus has three spines on outer and devoid of spines on the inner margin. The peduncle of uropod 2 has one thin spine at the distal end on the outer side. The outer ramus has two lateral spines and the inner ramus bears only terminal spines. Uropod 3 is tiny. Triangular telson provided with obtusely pointed apex.

Female- **Antenna 1** reaching to the distal end of the fifth joint of antenna 2; inner margin of first joint of peduncle bears three proximal spines; flagellum consists of about eight to nine joints and is smaller than the peduncle.

Antenna 2- it is smaller and less stronger as compared to male; the lower distal end of the third joint bears two small spines; fourth joint bears four spines along the lower edge and two spines on the inner surface. The fifth joint has large number of setae (Plate 2e-g).

Gnathopods and peraeopods are similar to those of the male. A proximal tooth present on the inner surface of the fourth joint of the second gnathopod (Plate 2a-d).

Uropods: The outer margin of uropod 1 bears five or six spines and inner margin has four spines. Outer ramus has four spines on outer margin and without any spines on the inner margin. The inner ramus has four spines on the outer margin and no spines on the inner margin. The peduncle of uropod 2 has two small spines at its distal end. Uropod 3 is similar to that of the male.

4.4.4.2 Population density of *C. madrasensis*

Population density varied between 0.048 x10^6 ind.m^{-2} and 1.97 x 10^6 ind.m^{-2} during the study. *C. madrasensis* exhibited a clear seasonal disparity in population density (Table 4.1), with maximum density is recorded during PRM season (av. 1.219±0.53 x 10^6 ind.m^{-2}) and minimum density during MN (av. 0.25±0.17x10^6 ind.m^{-2}) (Fig 4.24).

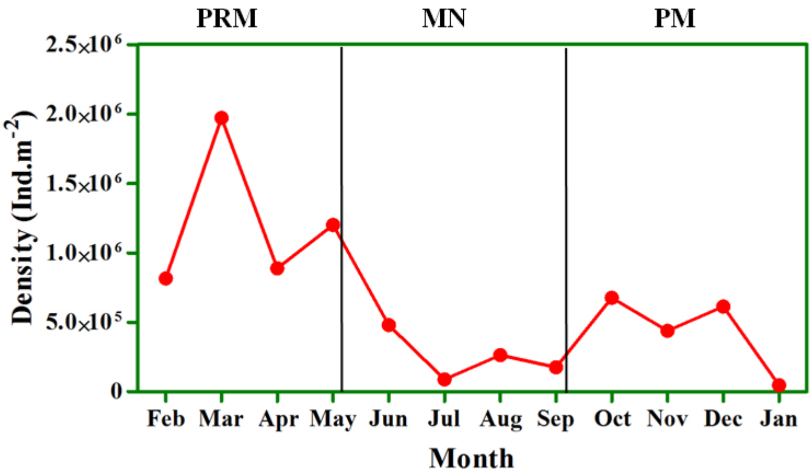


Figure 4.24 Variations in the population density of *Chelicorophium madrasensis* in the sampling location

Two peaks in density were observed during PRM, one in the month of March (1.97x 10^6 ind.m^{-2}) and the other in May (1.2x 10^6 ind.m^{-2}) and a low density observed in February (0.82 x 10^6 ind.m^{-2}). During MN season, high density was observed in June (0.48x 10^6 ind.m^{-2}) and low density was recorded during July (0.089x 10^6 ind.m^{-2}). During PM, higher density was observed in October (67.6x 10^6 ind.m^{-2}) and lower density recorded during January (0.048x 10^6 ind.m^{-2}). Significant seasonal variation in density ($p<0.05$) was observed from the sampling location (Table 4.1).

4.4.4.3 Biomass of *C. madrasensis*

Biomass of the species varied between 23.33 and 817.92 g.m^{-2} in the sampling location (Fig. 4.25) with insignificant seasonal variation (Table 4.1). Lowest biomass was observed during MN (av. 209.6±133.9 g.m^{-2}) season and highest during PRM (av.544±185.7 g.m^{-2}) during the study.

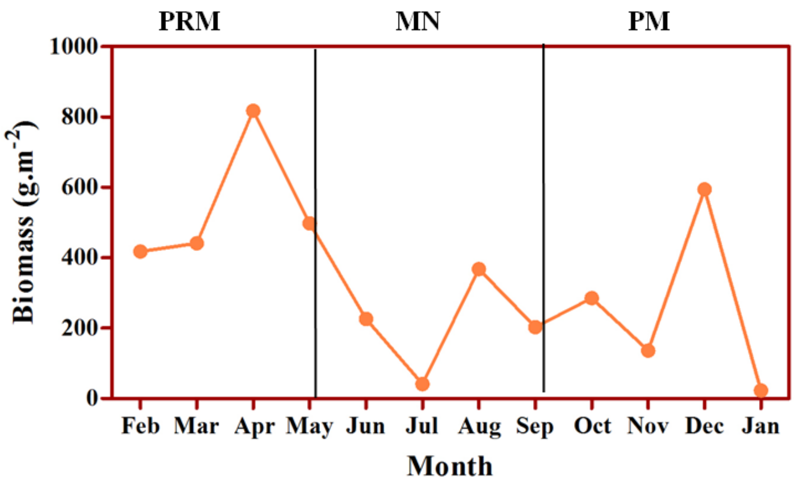


Figure 4.25 Variations in the biomass of *Chelicorophium madrasensis* in the sampling location

Biomass exhibited a peak in April (av. 817.92 g.m^{-2}), during PRM season, a peak in August (av.368.16 g.m^{-2}) during MN season, and a peak in December (av.594.24 g.m^{-2}) during PM season.

4.4.4.4 Mean individual body weight of *C. madrasensis*

Mean individual body weight of the species varied between 0.22x10^{-3} and 1.39x10^{-3}, with minimum ratio recorded during PRM (av.0.52±0.3x10^{-3}) and maximum during MN (av.0.87±0.4x10^{-3}) (Fig. 4.26). Among the PRM months lowest ratio was recorded during March which indicates higher number of smaller individuals.

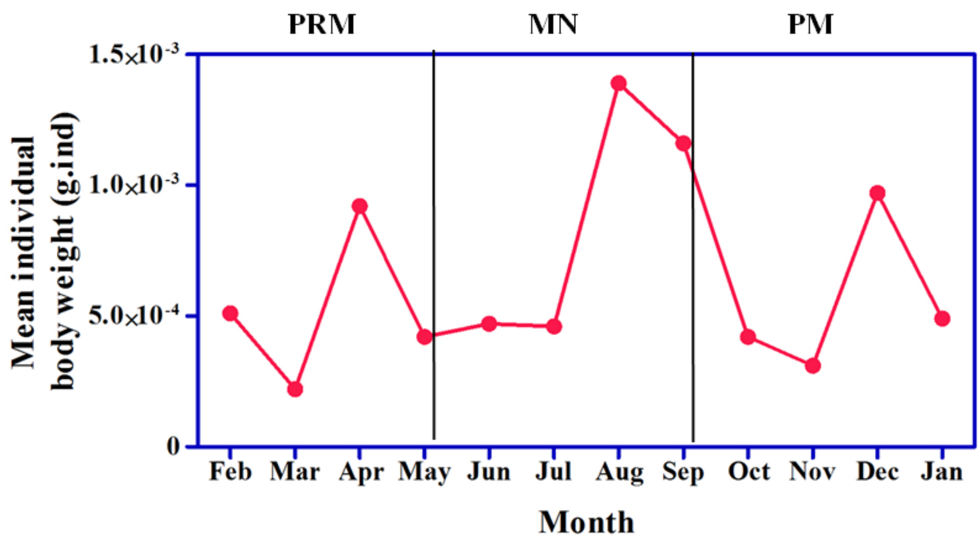


Figure 4.26 Variations in the mean individual body weight of *Chelicorophium madrasensis* in the sampling location

Table 4.1 *Results of One way ANOVA of major biotic and abiotic parameters in the Cochin estuary (* - $p<0.05$, **- $p<0.01$)*

Parameter	p value
Temperature	0.0009**
Salinity	0.006**
pH	0.987
DO	0.129
BOD	0.123
SPM	0.738
Nitrate	0.76
Nitrite	0.035*
Ammonia	0.321
Phosphate	0.0009**

Silicate	0.205
Sand	0.888
Silt	0.691
Clay	0.546
Organic carbon	0.167
Benthic chlorophyll *a*	0.008**
Macrobenthic Density	0.01*
Macrobenthic Biomass	0.024*
Density (*C.madasensis)*	0.01*
Biomass (*C.madasensis*)	0.07

4.4.5 Interrelation with biotic and abiotic variables

4.4.5.1 Karl Pearson's Correlation

Benthic chlorophyll *a* exhibited positive correlation with macrobenthic density, macrobenthic biomass and density of *C. madrasensis*, organic carbon, salinity and temperature (Table 4.2). It showed negative correlation with rainfall, and phosphate. Rainfall showed positive correlation with DO and phosphate, and a negative correlation with salinity and organic carbon. *C. madrasensis* density was positively correlated with salinity, biomass, macrobenthic density, density of *Anthurid* sp and with *C. fluviatilis*. Silt showed negative correlation to clay (Table 4.2).

Table 4.2 *Showing the results of the Karl Pearson's Correlation between the parameters (n=12, * denotes -5% significance and **- 1% significance)*

	Ben.Chl	Rainfall	C. madrasensis	Density	Biomass	Sand	Silt	Clay	OC	Silicate	SPM	PO4
Ben.Chl	1											
Rainfall	-0.853**	1										
C. madrasensis	0.592*	-0.426	1									
Density	0.809**	-0.449	0.660*	1								
Biomass	0.629*	-0.343	0.733**	0.781*	1							
Sand	-0.260	0.191	-0.005	-0.240	-0.156	1						
Silt	-0.022	0.056	-0.159	0.181	-0.112	-0.224	1					
Clay	0.143	-0.144	0.162	-0.069	0.184	-0.241	-0.892**	1				
OC	0.728**	-0.660*	0.035	0.472	0.191	-0.257	-0.072	0.191	1			
Silicate	-0.292	0.340	-0.377	-0.247	-0.196	0.029	-0.304	0.289	0.152	1		
SPM	-0.166	-0.220	-0.019	-0.506	-0.163	-0.146	0.000	0.068	-0.184	-0.289	1	
PO4	-0.675*	0.718**	-0.652*	-0.394	-0.405	-0.091	0.513	-0.468	-0.356	0.458	-0.084	1
NH4	-0.059	-0.321	0.227	-0.395	-0.070	0.144	-0.282	0.214	-0.175	-0.282	0.815**	-0.438
NO2	-0.348	0.366	-0.463	-0.220	-0.412	0.494	0.546*	-0.773	-0.099	0.184	-0.172	0.619*
NO3	-0.491	0.192	0.085	-0.561*	-0.111	0.439	-0.385	0.179	-0.680*	-0.031	0.469	-0.113
BOD	0.549*	-0.432	0.363	0.618*	0.309	-0.257	0.207	-0.087	0.401	-0.423	-0.220	-0.477
DO	-0.444	0.566*	-0.319	-0.082	0.047	0.266	-0.035	-0.089	-0.263	0.270	-0.316	0.431
Salinity	0.721**	-0.845**	0.450	0.420	0.349	-0.351	0.228	-0.064	0.544*	-0.417	0.462	-0.495
PH	0.338	-0.133	0.435	0.433	0.186	-0.466	0.447	-0.229	-0.022	-0.251	-0.104	0.037
Temp	0.683*	-0.524	0.771**	0.712**	0.805**	-0.074	-0.353	0.386	0.270	-0.315	-0.147	-0.736**
Anthuridae	0.546*	-0.368	0.916**	0.582*	0.724**	-0.059	-0.118	0.145	-0.079	-0.308	-0.002	-0.493
C. fluviatilis	0.461	-0.324	0.922**	0.471	0.684*	-0.131	-0.251	0.311	-0.112	-0.306	0.143	-0.541*
C.madra.biomass	0.544*	-0.327	0.580*	0.699**	0.910**	-0.008	0.055	-0.051	0.202	-0.259	-0.078	-0.326

	NH4	NO2	NO3	BOD	DO	Salinity	PH	Temp	Anthuridae	C. fluviatilis	C.madra.biomass
NH4	1										
NO2	-0.317	1									
NO3	0.710**	-0.173	1								
BOD	0.059	-0.278	-0.296	1							
DO	-0.381	0.301	0.064	-0.196	1						
Salinity	0.421	-0.229	-0.225	0.478	-0.508	1					
PH	-0.301	-0.056	-0.431	0.070	-0.497	0.265	1				
Temp	0.093	-0.584*	-0.025	0.480	0.062	0.430	0.039	1			
Anthuridae	0.111	-0.388	0.081	0.089	-0.339	0.364	0.587*	0.655*	1		
C. fluviatilis	0.254	-0.548	0.171	0.072	-0.364	0.359	0.520	0.655*	0.961**	1	
C.madra.biomass	0.052	-0.195	-0.047	0.441	0.143	0.407	-0.050	0.695**	0.496	0.439	1

4.4.5.2 Redundancy analysis (RDA)

Redundancy analysis helps to obtain a simultaneous representation of the observations of the response variables (species) and the explanatory variables (physicochemical variables and chlorophyll *a*) in two or three dimensions and the position of the variables in the triplot enable in visualizing their interrelationships. Hence a detailed understanding of the interrelation existing among the environmental, sediment characteristics and the amphipod population of the CE, RDA was carried out between these parameters using the statistical software, CANOCO 4.5.

The RDA plot helped to reveal the preferred environment for the amphipod species. Among the environmental parameters, the *Chelicorophium* species exhibited a positive relation with salinity, SPM, and pH while negative affinity with DO. Among the sediment characteristics, *Chelicorophium* showed close affinity with benthic chlorophyll *a*, clay, and organic carbon and it showed negative relation with sand (Fig. 4.27).

Among the biotic factors influencing the density of *Chelicorophium* species, benthic chlorophyll *a* and density of two isopod species (*Anthurid* sp and *Cirolana fluviatilis*) exhibited strong positive affinity. In addition other factors such as SPM and organic carbon also exhibited a positive but weak affinity (Fig. 4.28).

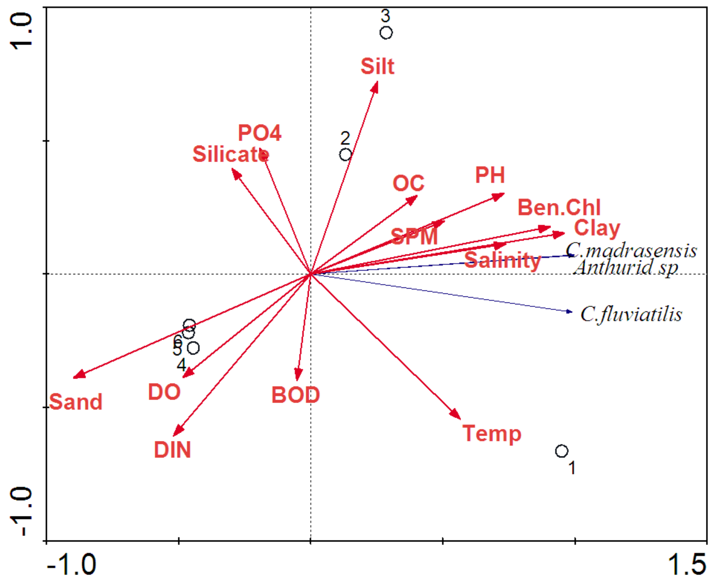


Figure 4.27 RDA plot showing the influencing factors on the population density of *C. madrasensis*

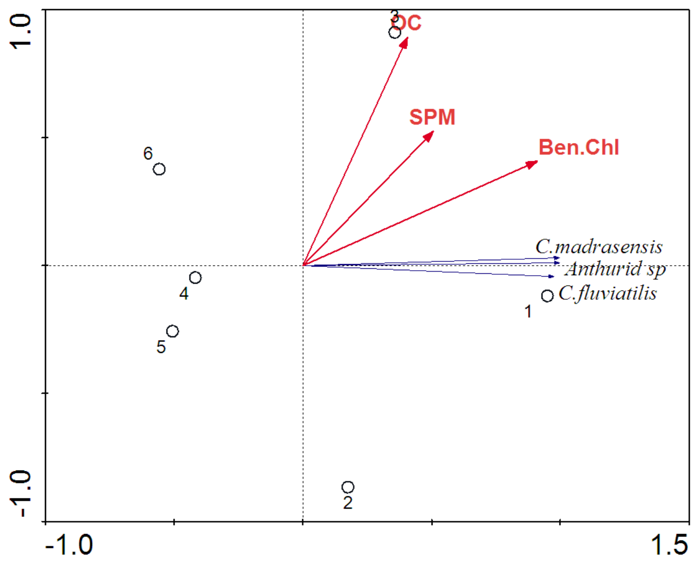


Figure 4.28 RDA plot showing the influencing factors on feeding of *C. madrasensis*

4.5 DISCUSSION

4.5.1 Taxonomic remarks of the species

Crawford (1937) classified the genus Corophium into three categories, Section A, Section B and Section C. The species having a segmented urosome belongs to section A, whereas in section B and section C the urosome segments are fused. In section B the uropods I and II are inserted in the notches of lateral margins of urosome. On the other hand uropods I and II are found to be ventrally attached in section C. Of the tube dwelling amphipod genus Corophium, three species have been reported from Indian waters so far -*Corophium triaenonyx* Stebbing, *C. crassicorne* Brazelius, and *C. madrasensis*, Nayar. As the current specimen, *C. madrasensis*, possessed with a separate urosome segment, it is categorized under section A. Although *C. madrasensis* resembles *C. triaenonyx* in the urosome and peraeopod features, some characteristic features which distinguishes it from the species, *C. triaenonyx* include 1) the first peraeon segment devoid of six plumose setae fringed on the apex of its side plates, 2) female possess only four spines in a row on the lower edge on the fourth segment of antenna 2, and 3) male possess a proximal tooth on the inner surface of the fourth joint of the second gnathopod (Plate 2). According to Nayar (1950) other morphological features such as, larger body size of the species, having only three teeth on the inner edge of the seventh joint of the gnathopod 2, 12 jointed flagellum of the first antenna in the male, found to separate the present species from other species of the genus and therefore named it as *Corophium madrasensis* by Nayar (1950). Later this species was accepted as *Chelicorophium madrasensis* (genus modified from Bousfield and Hoover, 1997).

In the present study, the remarkable density of macrobenthic fauna encountered in the sampling site was mainly constituted by a single species of tube building amphipod, *C. madrasensis* (av. 0.0477 to 1.972 x 10^6 ind.m^{-2}). The present chapter on ecology and population dynamics of *C. madrasensis* has its own significance as this particular amphipod species was reported first time from the CE. In Indian waters (Adayar estuary), there is only one report that has been published yet, which dealt with its taxonomy (Nayar, 1950) and the species reported to occur in high density. Similar observations of corophid amphipods in higher densities were also reported from many parts of the world like, *Corophium curvispinum* Sars in Lower Rhine at the Netherlands (Van den Brink et al., 1993), *C. multisetosum* from Ria de Aveiro, Portugal (Queiroga, 1990) and *Corophium volutator* from the estuarine and coastal mudflats of northwestern Europe (Hughes, 1988).

4.5.2 Ecology of *C. madrasensis*

In the present study, significant seasonality was observed in the environmental parameters such as temperature, salinity, inorganic nutrients (nitrite and phosphate) etc. Relatively low salinity (av.1.1±0.21) was recorded during MN period, while medium salinity prevailed in the sampling location, during the other two seasons (Fig 4.3). The monsoonal precipitation and associated runoff (Qasim, 2003), the characteristic feature of monsoon, significantly reduces salinity of the sampling location during MN. In contrast, the enhanced seawater incursion, the characteristic features of the CE during non-monsoon (PRM & PM) periods, led to the prevalence of high salinity water in the CE (Srinivas et al., 2003). Normally, seawater intrusion through bottom layers enable the entry of marine components into the estuary during non-monsoon periods (Sankaranarayanan and Qasim, 1969). Relatively higher levels of DO (av. 5.45±0.39 mg/L) observed in the sampling location during MN might have resulted by the increased river influx and runoff associated with the monsoonal rainfall, which solubilizes more oxygen compared to sea water (Qasim and Gopinathan, 1969). The insignificant seasonality of DO might have happened by the prevalence of higher phytoplankton standing stock, irrespective of the seasons (Madhu et al., 2007). Higher dilution and flushing of the estuary brought about by the monsoonal rains might be responsible for the lower BOD and pH in the sampling site. Higher concentration of nitrate and ammonia during PRM months may be derived from several non-point sources along the bank of the estuary (Menon et al., 2000). Earlier studies have revealed higher concentrations of inorganic nutrients in the upstream regions of the CE as a result of disposal of domestic and industrial wastes (Madhu et al., 2010a; Miranda et al., 2008). Similarly, higher silicate and phosphate levels observed in the sampling location during MN is apparently associated with the terrestrial runoff and concurrent turbulent activities (Sankaranarayanan and Qasim, 1969; Madhu et al., 2010b). In case of phosphate, other than the contribution from sediment resuspension (Martin et al., 2011; Sankaranarayanan and Panampunnayil, 1979), the external sources such as domestic sewage and industrial effluents particularly from phosphate fertilizer plants, contribute significantly to the phosphorus concentrations in the estuary.

Environmental factors such as temperature, salinity, DO, quality and quantity of food are found to influence the distribution and density of

Corophium species (Prato and Biandolino, 2006). Density of *C. madrasensis* was comparatively higher in the sampling site during PRM (av.1.219 ± 0.53 x10^6 ind.m^{-2}) and PM (av.0.44x10^6ind.m^{-2}) periods when mesohaline salinity conditions (mesohaline based on McLusky, 1993) prevailed. As per earlier reports *C. triaenonyx,* and *C. volutator* are reported to exhibit wide tolerance to salinity and temperature fluctuations (McLusky, 1968; Shyamasundari, 1973). The present study evidenced that, though *C. madrasensis* is found to occur in the sampling site between the salinity ranges 1 to 20, maximum density of this species was recorded during PRM and PM, when the sampling site attained mesohaline salinity (5-18). Therefore, it can be inferred that though this species are able to tolerate wide fluctuations in salinity they are known to flourish and obtain maximum population only when the optimum salinity conditions prevailed. The results of multivariate analysis also substantiate the influence of salinity on its density and distribution (Fig. 4.26). Water quality parameters such as DO, BOD, and nutrients did not show any correlation with the species density (Table 4.2). Bottom water was well oxygenated during the study period except few sampling occasions. Corophid amphipods are known to pump oxygenated water down into their burrows and tubes which facilitate them to avoid the low oxygen condition (De-la-Ossa-Carretero et al., 2012).

Nature of the substratum is another imperative factor controlling the distribution and density of amphipods (Fincham, 1969) and every species have a particular preference for a specific particle size. Prevalence of fine fractions of the sediment throughout the study period indicates that, this location is influenced by weak tidal currents (Satyanarayana Murty and Rao, 1959). Ramamirtham and Muthusamy (1986) reported that a differently oscillating null zone exists in the CE approximately 5-15 km north of the Cochin inlet. Being positioned near this region, the sampling location might have been well influenced by this zone. Further, Balachandran et al. (2005) remarked on the influence of the estuarine geomorphology (ox-bow shape) and meandering flow inducing the formation of these zones and the weak flushing rates often results in the entrapment of fine particles at the sampling location, further substantiates the predominance of clayey substratum of the region. Hence in the present study, the observed higher density of *C. madrasensis* in the sampling site apparently substantiates their preference to the clayey substratum. Corophium species were reported to occur abundantly in calm areas where physical stress on the sediment is minimal (Beukema & Flach, 1995). The multivariate analysis also corroborated the above finding as the species exhibited positive affinity with clay (strong affinity), and silt particles (weak affinity), while a negative relation with sand (Table 4.2). Earlier studies depicted the preference of the species of the Family Corophiidae to specific sediment texture. Similar to *C. madrasensis,* another species of the tube building amphipod *C. volutator* seems to prefer finer sediments while the species *C. arenarium* occurs in more sandy sediments (Boyden and Little, 1973; Gee, 1961; Watkin, 1941). As per literature Nguyen et al. (1997), finer sediments have maximum capacity for entrapping organic carbon and therefore the enhanced organic carbon content evidenced in the sampling site might have been associated with the predominance of finer clayey substratum.

Among the sediment characteristics, benthic chlorophyll *a* exhibited a positive relation with density and biomass of *C. madrasensis*. Benthic chlorophyll *a* concentration was relatively higher in the sampling site compared to other areas (Fig. 4.16) of the CE (Sanilkumar et al., 2009& 2011). During PRM, favorable conditions such as mesohaline estuarine water, adequate temperature, along with plenty of nutrients, supported the microphytobenthos to flourish in higher densities, which was reflected in the microphytobenthic biomass (Fig. 4.15 & Fig. 4.16). Non-monsoon periods usually experiences high phytoplankton growth in the euphotic water column in the CE due to the proliferation of diatoms (Madhu et al., 2010b). The sedimentation of this phytoplankton community may also have a contribution to the enhancement in the sediment chlorophyll *a* and organic carbon in the sampling site. Lower concentration of the benthic chlorophyll *a* observed during MN might be the result of the instability caused to the substratum by heavy river discharges and runoff associated with monsoonal precipitation. Decrease of microphytobenthic biomass as a result of the physical instability coupled with flow-induced velocity caused by runoff has been earlier reported by Sanilkumar et al. (2011). Though light forms the major limiting factor determining the production of microphytobenthos in turbid waters (Madhu et al., 2010b; Menon et al., 2000), microphytobenthos have several physiological adaptations for coping with low light fluxes and can survive even in dark without any damage to their photosynthetic potential (Wulff et al., 1997). They are also known to resume photosynthesis if gets resurfaced (Fielding et al., 1988), and even have the capacity to resort temporarily to heterotrophic nutrition (Admiraal et al., 1984) during adverse environmental situations. Higher density and biomass of *C. madrasensis* observed during PRM and PM months correspondingly with higher benthic chlorophyll *a* indicates their interrelation. Several studies have been shown that the species of the genus Corophium consume large number of mucus producing benthic microalgae (Gerdol and Hughes, 1994b; Grant and Daborn, 1994). Cunha et al. (2000a) reported on the significant correlation between benthic chlorophyll *a* and the density of *C. multisetosum* in the Aveiro estuary. Further, prior studies (Coles, 1979; Fenchel et al., 1975) suggesting on the intake of diatoms and bacteria associated with mineral particles as a major food item of similar species in the genus (*C. volutator*) and especially with 78% efficiency in the diatom ingestion (Lopez and Levinton, 1978) also adds valuable support on the dietary interaction existing between microphytobenthos and Corophium species in the CE. The significant correlation existing between their density and biomass with benthic chlorophyll *a* (Table 4.2) and the results of the multivariate analysis further affirms the above finding (Fig. 4.27).

Exceptionally higher macrobenthic standing crop recorded at the sampling site (north of Panambukadu Island) compared to previous studies in the CE (Sheeba, 2000; Martin et al., 2010), was contributed by the dominance of a single species, *C. madrasensis*. Like other species of Corophium, it has a characteristic behavior of tube building with fine sediments. Laboratory observation of the species also proved the tube building activity, when it was introduced in a glass bowl along with the mud obtained from the sampling sites. Each tube was appeared to be narrow, cylindrical in shape and opened just above the surface of the sediment. The aggregation of these organisms in large numbers at the particular site led to the formation of multiple tubes in sediments (Plate 3). Double-entrance cylindrical tubes are typical of Corophioids (Dixon and Moore, 1997) which are known to construct tubes by lipoprotein threads secreted from specialized glands in their pereopods. Report (Rao and Shyamasundari, 1963), on congenric species, *Corophium triaenonyx*, which builds tubes by using clay, silt, sand and detritus with the help of secretions from the glands in pereopods 3 and 4, further substantiates the tube building activity of the genus. Barnard et al. (1988) reported that Corophium sp. and Grandidierella sp, build tubes using amphipod silk in the presence or absence of foreign particles. *C. volutator* constructs its burrows in softer muddier substratum whereas *C. arenarium*, built its tubes in a firmer substratum (Gee, 1961).

The occurrence of high density of *C. madrasensis* in the sampling site, especially during PRM months, indicates the prevalence of favorable environmental conditions of the species, as recruitment of a large number of juveniles was present in the sampling site during this period. On the other hand, the observed decrease in density of the species during MN months are found to be associated with the instability of the substratum, due to the heavy river discharge and run off associated with the monsoonal rainfall. Higher density of preferred food favored the proliferation of Corophium species, and helped to attain the maximum density and biomass during non-monsoon periods. In addition, the presence of large number of small sized juvenile amphipods in the samples during the season indicates the most favorable recruitment period. In addition, analysis of the mean individual body weight of the species also confirms PRM period (during March) as its peak breeding season by its lowest resultant value (Fig. 4.26). However, the presence of juveniles throughout the study indicates its continuous breeding behavior. Tropical species such as *Melita zylanica* (Krishnan and John, 1974; Morino, 1978), *E. chilkensis* (Aravind et al., 2007), and *C. triaenonyx* (Shyamasundari, 1972) are known to breed continuously corresponding to the incessant availability of food and hence young ones are often observed in the population throughout the year (Steele and Steele, 1991).

4.5.3 Associate fauna of *C. madrasensis*

The noticeably higher density and biomass of *C. madrasensis* in the site supported the sustenance of two carnivorous isopods, *Cirolana fluviatilis* and *Anthurid* sp (Fig. 4.22 & Fig. 4.23). The significant correlation ($p<0.01$) observed between their densities with *C. madrasensis* density indicates the prevailing strong trophic relationship between these organisms. The RDA plot also validates the prevailing positive relationship of these two species with the abundance of *C. madrasensis* (Fig. 4.28). Of the isopods encountered in the sampling location, C*irolana fluviatilis* having a wide tolerance to salinity, temperature and DO conditions was the dominant one which commonly occurs in estuaries, (Newman et al., 2007). *C. fluviatilis* is a voracious carnivore and its higher abundance cause potential threat to the fishes and shellfishes and hence are commonly called as 'fish lice'(Mathew et al., 1994). It feeds on live as well as dead organisms such as wood borers, foulers, polychaetes, nematodes, etc and also attack on weak or dead prawns, fish baits, fishes and crustaceans trapped in nets (Mathew et al., 1994). Euryhaline nature of this organism is an adaptation to its scavenging mode of life, which has the ability to consume large amounts of food upon favorable environmental conditions (Smith and Baldwin, 1982). The higher density of the Corophium species in the sampling location might have contributed to the higher population densities of this isopod. Moreover, the muddy nature of the substratum also favored the isopod species (Newman et al., 2007). Mathew et al. (1994) reported the occurrence of large number of these isopods at Kumbalangi-Perumpadappu region of the CE, as a consequence of the improper flushing associated with the construction of an earthern bund. Another dominant isopod species was the *Anthurid* sp which not only showed a significant correlation ($p<0.01$) with the density of *C. madrasensis*, but also exhibited a positive affinity in the RDA plot (Fig. 4.28). Anthurideans, inhabiting littoral or shallow shelf environments are chiefly carnivores, feeding on small invertebrates (Brusca et al., 2001). Their biting mouthparts are indication of their mode of feeding. The two isopod species live on or within the tubes made by *C. madrasensis* and occur in high densities during PRM months (Fig. 4.22 & Fig. 4.23), when *C. madrasensis* attained its peak density. With the onset of MN density of both the amphipod and the isopods began to decline, due to the instability of the substratum. Predation have a significant role in structuring the benthic communities, and any factor that facilitate or inhibit predator-prey interactions is also have an imperative role in ecology (Eriksson et al., 2005). Natural events such as runoff and tidal currents not only lead to instability to the substratum, but also allow the prey organisms to be more exposed and easily available to predators.

In addition to the isopod species, 8 species of polychaetes belonging to 5 families were also recorded in the sampling site during the course of study. Of these species, 6 polychaetes species such as *Nereis* sp, *Lycastis indica*, *Lumbriconereis* sp, *Lumbriconereis latreilli*, and *Ninoe pulchra* are predominantly carnivorous in nature benefited from the higher prey availability in the study region. The occurrence of surface depositing feeding polychaetes such as, *Mediomastus capensis, Prionospio cirrifera, Dendronereis estuarina, Capitellid sp,* and tanaid (*Apseudus chilkensis*) indicates their opportunistic adaptability in the utilization of higher organic carbon available in the sampling location.

Previous report on various Corophid amphipods has pointed out its relationship with the sediment stability (Smith et al., 1996). Burrowing and feeding activities of Corophium known to cause instability to the substratum (Gerdol & Hughes 1994a; de Deckere et al., 2000). Though the corophid amphipod decreases the stability of the sediment by these activities, the tube building activity of the organisms has an opposite effect on the sediment stability. During tube building, Corophiidean amphipods binds sediment particles together using their mucous secretions, thus contributing to more stability to the substratum (Peterson, 1989; Meadows et al., 1990).

The present study on population dynamics of C. *madrasensis* in the CE inferred that the density and distribution of this particular amphipod was found to be influenced by combination of several ecological factors. Environmental parameters such as temperature, salinity, sediment properties like texture and composition, exerted a profound influence on the density and existence of the tube-building amphipod species in the CE. In addition, the unique feeding modes and the predation pressure experienced from the carnivorous isopods were also found to regulate their population structure in the CE. Thus an intricate and strong abiotic-biotic interaction between the flora, fauna and the physical environment helped in sustaining the high density of this species of amphipod in the sampling location.

Corophid amphipods, form a major food items for many fishes and larger invertebrates in many of the estuaries, and thus have a key role in the food web structure of an aquatic ecosystem. According to Elton (1927),"food is the burning question in animal society, and the whole structure and activities of the community are dependent upon the question of food-supply". Corophium act as a grazer on the micro flora and also form important prey items for the higher organisms. Thus they perform a key role in the estuarine benthic food web dynamics as an intermediate link between the lower and upper trophic levels. In addition, they are highly ecologically significant in the estuaries by the peculiar tube building activity, which not only strengthens the sediments but also harbors many invertebrates. The information on *C. madrasensis* achieved from the present study can be used as baseline information on the tube building amphipods of the CE. In the future, mass culture of these amphipods should be attempted for utilizing them as a suitable live feed organism in aquaculture purposes. In addition, more experimental studies have to be carried out to unravel the life history attributes of this ecologically significant tube building amphipods.

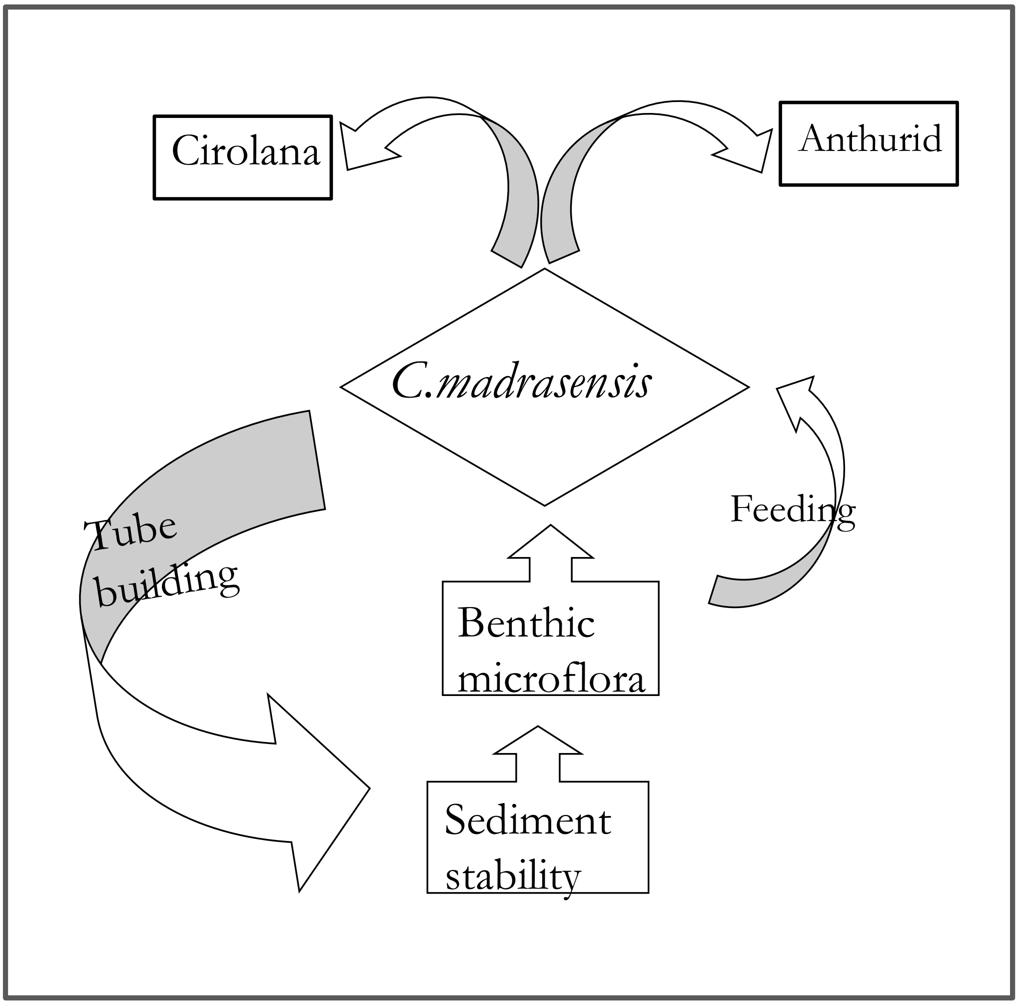


Fig. 4.28 Flow-Chart showing the ecological interaction of the amphipod species *C. madrasensis*

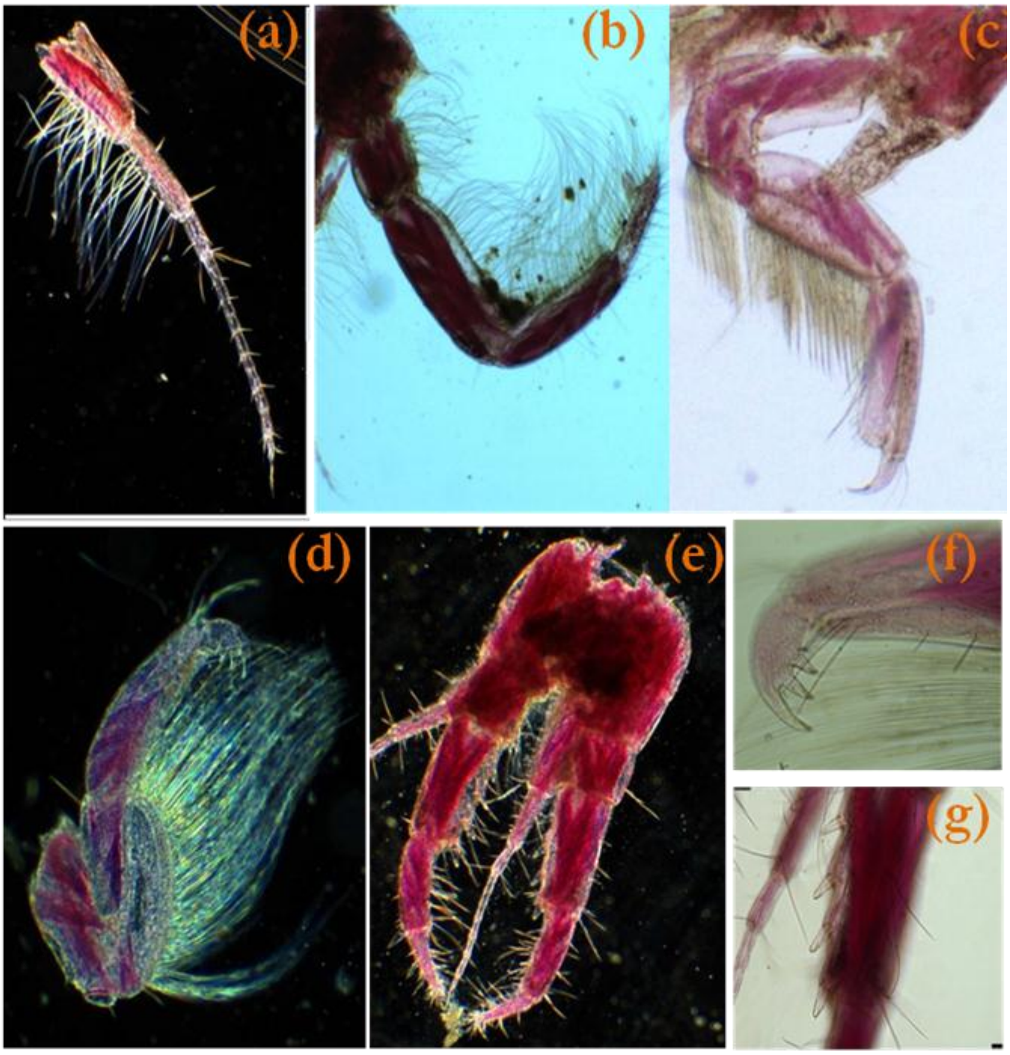


Plate 2 Microphotographs of Distinguishing features of *C. madrasensis* ;Male-(a) Antenna 1, (b) Antenna 2, (c) Gnathopod 1(d) Gnathopod 2; Female (e) Antenna 1&2, (f) Proximal tooth on 2nd Gnathopod (in 60X) (g) four spines the lower edge on the 4th segment of Antennae 2 (in 60X).

Plate 3 Photograph showing the mud formation by *Chelicorophium madrasensis* in the sampling location

Table 4.3 *showing the monthly variation in density of macrobenthic fauna (ind.m^{-2}) in the Cochin estuary (*-100 to 500, **-500 to 2000, ***-2000 to10000, ****-10000 to 50000, *#-50000 to 1 lakh, *$-1 lakh to 10 lakhs, *$# -above 10 lakhs)*

Taxa	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Polychaeta												
Mediomastus capensis	*	0	**	**	*	*	0	*	*	0	*	0
Capitellid sp	0	0	0	0	0	0	0	*	0	0	0	0
Nereis sp	0	0	0	0	*	0	*	0	0	0	0	0
Dendronereis estuarina	0	0	0	0	0	0	0	*	0	0	0	0
Lycastis indica	0	0	0	0	0	0	0	*	*	0	0	0
Lumbriconereis sp	*	**	0	**	*	0	0	**	0	0	0	**
Lumbriconereis latrei	0	0	**	0	0	**	0	0	0	**	**	0
Ninoe pulchra	0	0	0	0	0	0	0	*	0	0	0	0
Prionospio cirrifera	0	0	0	**	0	0	0	0	0	0	0	0
Eunice indica	0	0	0	0	0	*	0	0	0	0	0	0

Amphipoda	0	0	0	0	0	0	0	0	0	0	0	0
Corophium madrasen.	****	*$	*$#	*$	*$#	*$	*#	*$	*$	*$	*$	*$
Eriopisa chilkensis	*	**	0	***	0	0	*	*	*	0	*	0
Photis digitata	0	0	**	0	0	0	0	0	0	**	0	0
Melita zylanica	*	0	***	**	0	**	0	**	0	***	**	***
Ampelisca sp	0	0	0	0	0	0	0	0	**	0	0	0
Platyischnopus sp	0	0	0	0	0	0	0	0	*	0	0	0
Caprellidae sp	0	0	0	0	0	0	0	0	*	0	0	*
Isopoda												
Anthuridae	**	***	*#	****	****	***	***	***	***	****	***	****
Cirolana fluviatilis	**	***	*#	****	*#	****	***	***	***	**	****	****
Other Taxa												
Apseudus chilkensis	0	0	0	***	0	0	*	*	*	0	0	0
Bivalvia	0	0	0	0	***	0	0	0	*	0	0	0
Oligochaeta	*	***	***	***	0	***	***	***	***	*	***	**
Turbellarian	0	0	***	0	0	0	0	0	0	0	0	0
Nematoda	0	0	0	0	***	**	**	**	*	***	**	0
others	0	**	0	0	0	0	0	0	0	*	**	0

*****$$*****

Chapter 5

IMPACT OF MAINTENANCE DREDGING ON MACROBENTHIC COMMUNITY IN THE COCHIN ESTUARY

Contents

5.1 INTRODUCTION

Estuaries, the transitional water body 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, rendering immense economic and ecological services to the mankind, a proper evaluation of the impacts associated with the increasing human interventions in estuaries is of utmost importance for the healthy sustenance, and also for the proper management of the bio-resources they harbor. Dredging is one of the significant anthropogenic activities conducted in estuaries and nearby coastal bodies for several purposes. According to Bray et al. (1997), the dredging activities may be conveniently divided into two, based on the requirements; 1) capital dredging conducted for the construction of a new bed configuration in the marine environment and 2) maintenance dredging in the channels to abate the effects of siltation and to maintain a constant bed configuration.

Sedimentation, the process of increased accumulation of suspended particles occurs in rivers, estuaries and seas as a result of various natural processes, resulting in the reduction of the depth of the water bodies. Progressive accumulations of sediments lead to shallowing of estuaries and thereby causing troublesome for the navigation process. Estuaries provide ready access to the neighboring sea through deep water channels. In order to maintain the navigable depth, regular dredging activities are performed to provide access to the port and terminal facilities (Kennish, 1992). Maintenance dredging is often carried out with a trailing suction hopper dredger. Positive effects of dredging, includes the improvement of circulation in estuaries and shallow embayments, which causes an enhancement of primary production through the increase in nutrient levels in the system. Dredged materials can be utilized in salt marsh creation, island development, beach nourishment, substrate enhancement and also in the restoration of coastal habitats. Though these activities have positive benefits to some extent, the recurrent dredging activities often have serious repercussions on the estuarine environment directly or indirectly, as it alters the bottom topography, sediment composition, modifies the depth and current strength and also leads in the removal of a stable substratum (Hacking, 2003; Newell et al., 1998; Newell et al., 2004). Therefore, the recurrent dredging activities and associated dredge-spoil dumping tend to cause short-term as well as long term environmental changes in many of the coastal water ecosystems.

According to Keplan (1975) the disturbances caused by dredging activities are mainly divided into three groups; 1) physico-chemical changes such as siltation, changes in water chemistry and avoidance by plankton, 2) temporary changes such as the elimination of benthic organisms in the path of the dredge and 3) changes in sediment deposition patterns and current velocities. In addition, it leads to increase in dissolved oxygen consumption, alternations in the nutrients and primary productivity, generation of contaminated wastes that build up over the years (Johnston, 1981; Barletta et al., 2016), thus creating serious impacts to the ecosystem. The substantial increase in suspended particles usually results in temporary degradation of water quality and intensified bottom siltation (Berry et al., 2003; Mercaldo-Allen and Goldberg, 2011). Usually the suspended sediments evolved not only during the dredging operations but also during the disposal processes itself (by the overflow from barges or leakage of pipelines), cause significant effects in the environment (Jensen and Mogensen, 2000). Adverse effects to various aquatic fauna such as gill clogging, respiratory impairment, excretory dysfunction and feeding problems in filter feeding organisms (Sherk, 1971), mortality of larval forms (Rosenberg, 1977), decrease in the coral abundance, growth and diversity (Dodge and Vaisnys, 1977), and accumulation of heavy metals (Hedge et al., 2009) have been reported worldwide.

Sediments are important for the components of the ecosystem including organisms to settle and inhabit for their development and survival (Groot, 1980). Disturbances caused to the physico-chemical characteristics of the sediments, usually lead to alternations in benthic communities (Desprez, 2000; Piersma et al., 2001; van Dalfsen et al., 2001). Physical removal of the substratum along with its associated fauna and flora and subsequent burial by the dredged sediments during disposal are the major direct ecological effects of dredging activities. An initial diminution in the species density, biomass and, diversity of benthic fauna (Sutton and Boyd, 2009) has been recorded as direct effects of dredging activities. In addition, the adjacent areas around the dredging sites can be indirectly affected by sediment resuspension, the release of nutrients and chemicals, and variations in food resources by shifts of plankton bloom seasons (Boyd et al., 2005; Simonini et al., 2007). Alteration of the tidal range and salt water intrusion in estuaries have been also observed as a consequence of dredging activities (Colby et al., 2010; Yuan and Zhu, 2015).

In India very few studies have been conducted on dredging activities and associated environmental impacts in the coastal waters so far. These studies were mainly focused on the physico-chemical aspects such as siltation (Anto et al., 1977; Gopinathan and Qasim, 1971), movement of the dredge plumes (Chandramohan et al., 1996), short-term dredging impacts (Balchand and Rasheed, 2000), water quality changes (Joseph et al., 1998), effects on sediment nutrients, carbon and granulometry (Nayar et al., 2007). Recently Velamala et al. (2016) studied on the tidal propagation, flushing time and increase in the estuarine volume associated with the channel dredging activities in Amba estuary, west coast of India.

5.2 Background of dredging activities in Cochin estuary

Cochin estuary (CE), a tropical semi-diurnal micro-tidal estuary, located on the southwest coast of India, running parallel to the Arabian Sea (AS), is remarkable for its rich biological productivity and biodiversity (Qasim, 2003; Madhu et al., 2007). It is connected to the AS through two permanent inlets; one at Cochin (width 450m) and the other at Azhikode (width 250m). 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 have been maintaining for the entry of larger vessels and ships. The three channels in the CE (Fig. 1) were, one 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 the Willington Island, i.e. Ernakulam channel (~ 5 km length; 250-500m width) and Mattancherry channel (~3 km length; 170-250 m width) . The approach channel was constructed in 1928 by cutting a sand bar, situated at 1.6 km of west of the coast. In the pretext of the construction of the Cochin port in 1936, an artificial island was created from the dredged out soil around a pre-existing tiny island and henceforth named as the Willington Island. 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 the channels to ensure the depth for easy navigation. During earlier years (1990s), intermittent dredging was carried out in the channels 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 inclusive of monsoon period, thus a large quantity of dredged materials are being removed annually (Table 5.1), maintaining deeper depths (10-13 m) for all three channels (Menon et al., 2000).

Table 5.1 *Data of annual maintenance dredged volume between 2006-2012 (Source, Cochin Port Trust, Kochi).*

Year	Dredged Volume (million cubic meter)
2006-2007	12.13
2007-2008	15.74
2008-2009	13.68
2009-2010	12.00
2010-2011	15.00
2011-2012	11.72

Macrobenthos (>0.5mm), an ecologically significant faunal component of estuaries, 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) and therefore these organisms are used as efficient indicators of both natural as well as anthropogenic perturbations experienced in coastal waters (Taupp and Wetzel, 2013; Whomersly et al., 2008). Their sessile habit and in turn their close association with the bottom sediments makes them efficient indicating the environmental changes (Danulat et al., 2002). As comprehensive knowledge on the 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).

Although, studies on impact of dredging activities on the benthic fauna is widely researched worldwide (Clarke et al., 1993; Kaplan et al., 1975; Van Dolah et al., 1984), detailed studies of dredging impacts of on benthic fauna from tropical estuaries are very few (Bemvenuti et al., 2005; Sheeba et al., 2004). In the CE, most of the earlier studies on macrobenthic fauna were focused on their distribution and species diversity (Devi et al., 1991; Feebarani and Damodaran, 2014; Martin et al., 2011; Pillai, 1977), but the impact of dredging on macrobenthic fauna have not been addressed comprehensively till date. In this context, the present chapter was designed to evaluate in detail whether the dredging activities carried out in the CE (1) have any adverse impact on the water quality, sediment properties and on the community structure of macrobenthos (2) have any implications on the functional traits of the benthic community.

5.3 Sampling strategy and methods

In order to understand the impact of dredging on macrobenthic community structure of the CE, sampling was carried out at 6 locations for three consecutive years (2009-2011). Two sampling locations were selected in the CE without having any direct effects of dredging, are named as the reference non-dredging sites (ND- stations 1&2). Sampling locations (D-stations 3 to 6), located inside the channels, where continuous dredging activities were carried out were designated as treatment dredging sites (Fig. 5.1). Being a tropical monsoonal estuary influenced by the Indian Summer Monsoon, the sampling periods in the CE were categorized based on rainfall and runoff, i.e., pre-monsoon (PRM; February-May) characterized by very less or little rainfall, monsoon (MN; June-September) a period of heavy rainfall and river run off, and post-monsoon (PM ; October-January), a transitional period with intermediate rainfall.

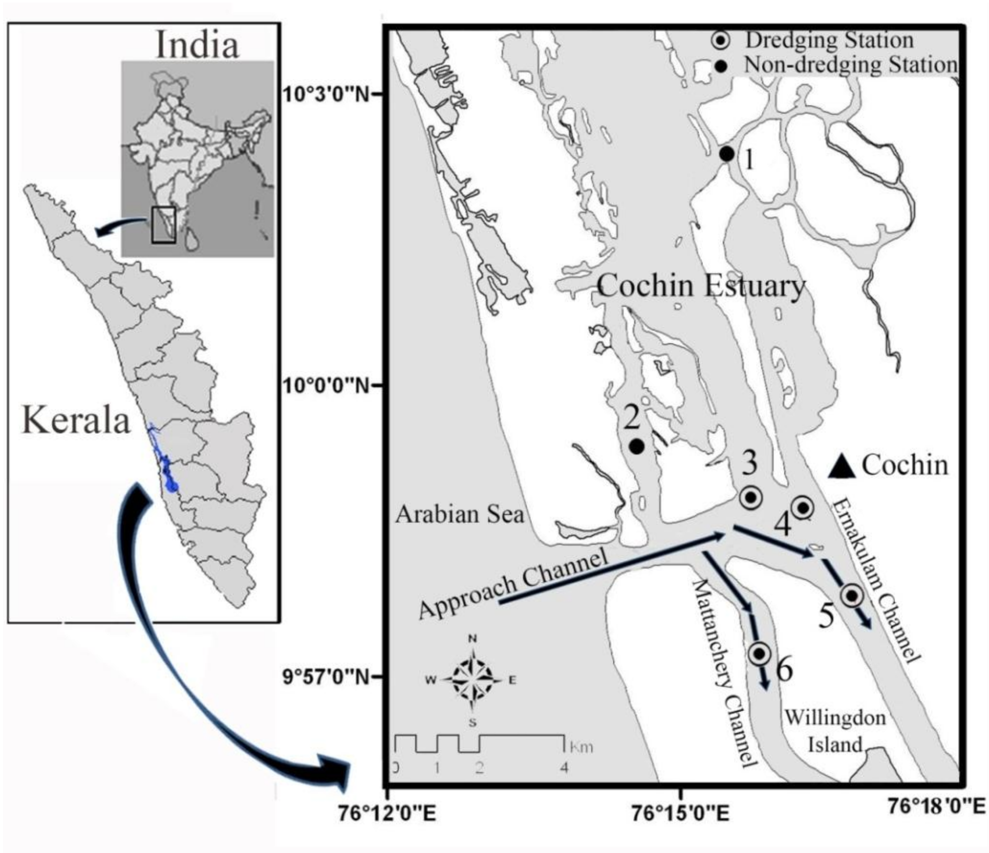


Figure 5.1 Map of the CE showing sampling locations.

Bottom water samples were collected for analyzing the environmental parameters such as temperature, salinity, pH, dissolved oxygen (DO), biological oxygen demand (BOD), nutrients and suspended particulate matter (SPM), and the analysis was carried out following standard procedures and protocols (mentioned in Chapter 2). Macrobenthos and sediment samples (for texture and organic carbon) were collected using Van-Veen grab and samples were analyzed using standard methodologies (mentioned in Chapter 2). Detailed taxonomic identification and feeding guild composition of the macrobenthic fauna was carried out using standard literatures (mentioned in Chapter 2). Statistical analysis such as unpaired t-test, one way analysis of variance (ANOVA), univariate analyses such as Shannon-Wiener index (*H'*), Margalef's richness (*d*), and multivariate analyses were also carried out using appropriate software packages (detailed in Chapter 2). Benthic Opportunistic Annelida Amphipods index (BO2A) was also used in the study to analyze the habitat quality status of the study area.

5.4 RESULTS

5.4.1 Environmental parameters

5.4.1.1 Rainfall

Annual rainfall in the nearby area of Cochin varied from 9.8 to 898 mm during the sampling period. Highest rainfall was observed during MN (av. 2009-540, 2010-588, 2011-659 mm) in the entire study period and lowest during PRM except 2011. Seasonal rainfall pattern during 2009-2011 was furnished in figure 5.2. During the study, higher rainfall was recorded in the year 2011 (26.3-897.5mm) compared to 2009 (12.6 -690.5 mm) and 2010 (9.8 -849.9 mm).

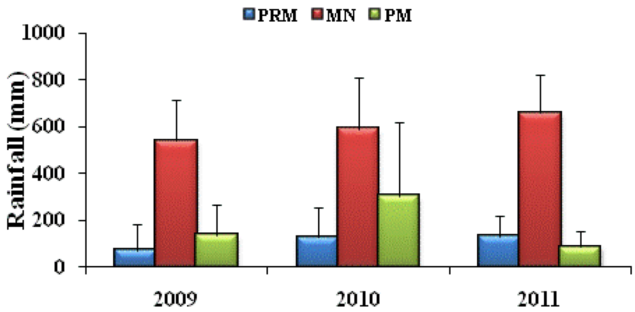


Figure 5.2 Seasonal distribution of rainfall during 2009-2011.

5.4.1.2 Temperature

During 2009, bottom water temperature varied from 25 to 32ºC in dredging sites and from 27 to 32.5 ºC in non-dredging sites. Slightly higher temperature was recorded during PRM and it varied from 27.8 to 32 ºC (av. 29.8 ±1.6 ºC) in dredging sites and from 28.5 to 32.5ºC (av.30.2 ± 1.5 ºC) in non-dredging sites. A decline in temperature was observed during MN and it varied from 25 to 29.5 ºC (av. 27.2 ± 1.9) in dredging sites and 27 to 30 ºC (av. 28.5 ± 1.3) in non-dredging sites. During PM, mean temperature was slightly high in dredging sites (av.29.13±0.18 ºC) compared to non-dredging sites (28.95±0.68 ºC). Seasonal variation in bottom water temperature, between dredging and non-dredging sites was given in figure 5.3a.

During 2010, bottom water temperature ranged from 26 to 33.5 ºC in dredging sites and from 26.2 to 33 ºC in non-dredging sites. Similar to previous year, comparatively higher water temperature was observed during PRM (D-av.32.3 ± 0.7 ºC, ND- av.32.6 ± 0.5 ºC), a decline during

MN (D-av.28.8 ± 1.1 °C, ND- av.28.8 ± 1.2 °C) and an increase during PM (D-av.29.0 ± 1.0 °C, ND- av.28.5 ± 1.3 °C) was observed. During PRM it varied from 31.5 to 33.5 °C in dredging sites and from 32 to 33 °C in non-dredging sites. During MN it ranged from 26 to 30 °C in dredging sites and 27 to 30 °C in non-dredging sites. During PM, temperature was ranged between 27 and 30.5 °C in dredging and between 26.2 and 30 °C in non-dredging sites (Fig. 5.3b).

During the study period 2011, bottom water temperature ranged from 26 to 32.5 °C in dredging sites and 27 to 33 °C in non-dredging sites (Fig. 5.3c). Seasonality of temperature was similar to previous years with higher temperature observed during PRM season in both dredging (av. PRM- 30.3±0.9, MN- 28.4±1.5, PM-29.1±1.0 °C) and non-dredging sites (av. PRM- 30.6±0.7, MN- 28.7±1.9, PM-29.0±0.90 °C).

The bottom temperature showed significant seasonal variation ($p<0.05$) during the entire study period while the variation was insignificant ($p>0.05$) between dredging and non-dredging sites (Table 5.2 and 5.3).

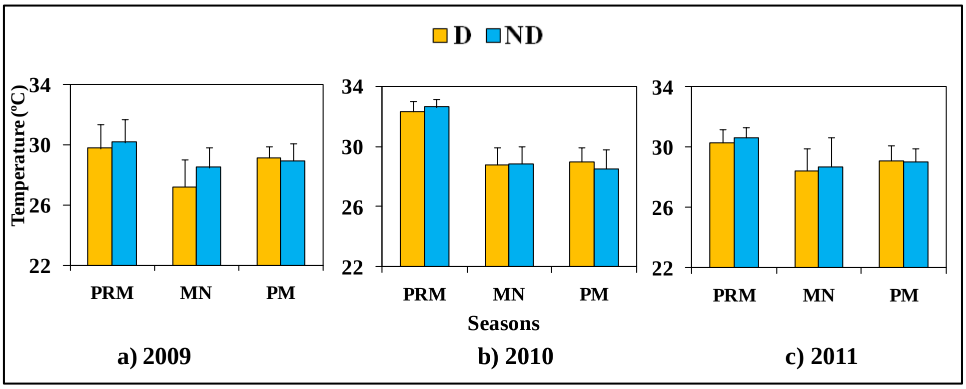


Figure 5.3 Seasonal distribution of the bottom water temperature, between dredging (D) and non-dredging sites (ND) of Cochin estuary.

5.4.1.3 Salinity

During 2009, bottom water salinity fluctuated from 0 to 35 in both dredging and non-dredging sites. Slightly higher salinity was observed in dredging sites relative to non-dredging sites (Fig. 5.4a). Seasonal variation was obvious with lower salinity recorded during MN (D- av.17.9±12.8, ND- av. 3.8±2.8) and higher during PRM (D-av.27.2 ± 7.0, ND- av.19.3 ± 10.1), in both dredging and non-dredging sites. An increase in salinity was observed during PM in dredging (av.21.9±9.9) and non-dredging sites (av.11.3±9.7). Seasonal variations in salinity, in dredging and non- dredging sites were given in figure 5.4a.

During 2010, relatively higher salinity was observed in dredging sites (0 to 34.3) compared to non-dredging sites (0 to 32.6). Similar to temperature, salinity also showed a decrease during MN in dredging (av. PRM-28.6±6.1, MN-9.5±10.9, PM-16.4±9.9) and non-dredging sites (av. PRM-21.1±11.4, MN-1.7±2. 3, PM-9.1±8.8) (Fig. 5.4b).

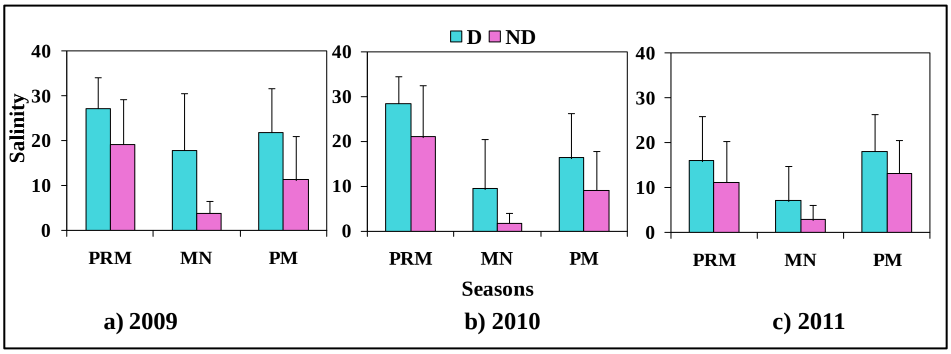


Figure 5.4 Seasonal distribution of the bottom salinity, between the dredging (D) and non-dredging sites (ND) of Cochin estuary.

Similar to previous year higher salinity was recorded in dredging sites (0. 8 to 32.7) during 2011, compared to non-dredging sites (0 to 28.9). Salinity exhibited a marked decrease during MN (D-av. 7.1 ± 7.8, ND-av.2.9 ± 3.3) in both regions. A slight increase in the salinity was observed during PM (D-av.18.1 ± 8.4, ND- av.13.2 ± 7.5) than PRM (D-av.16.0 ± 9.9, ND- av.11.2 ± 9.1) (Fig. 5.4c).

Significant seasonal variation ($p<0.05$) was observed in salinity (Table 5.2) during the study. Spatial variation in salinity was significant ($P<0.05$) during the study period except during 2011 (Table 5.3).

5.4.1.4 pH

In 2009, slightly higher pH was observed in dredging sites (6.6 to 8.3) relative to non-dredging sites (6.4 to 8.2). Low pH was observed during MN (D-av.7.4±0.5, ND-av.6.9±0.4) and high during PRM (D-av.7.9±0.1, ND-av.7.6±0.3) in both dredging and non-dredging sites. Distribution of pH in dredging and non-dredging sites of the CE was furnished in Figure 5.5a.

In 2010 also, pH was slightly higher in dredging sites (6.5 to 8.7) compared to non-dredging sites (6.6 to 8.8) except during PRM (Fig. 5.5b). Seasonal variation was evident with slightly higher pH recorded during PRM in both dredging (av. PRM-7.7±0.6, MN-7.4±0.3, PM-7.5±0.5) and non-dredging sites (av. PRM-7.9±0.8, MN-7.2±0.4, PM-7.1±0.5).

Similar to previous years, pH was higher in dredging sites (6.7 to 8.4) compared to non-dredging sites (6.8 to 8.5). Seasonal variation was obvious in pH with slightly higher concentration recorded during PM in both dredging (av.7.9±0.3) and non-dredging sites (av.7.7±0.4). Lower pH was observed during PRM (av.7.5±0.5) in dredging sites and during MN (av.7.4±0.4) in non-dredging sites (Fig. 5.5c).

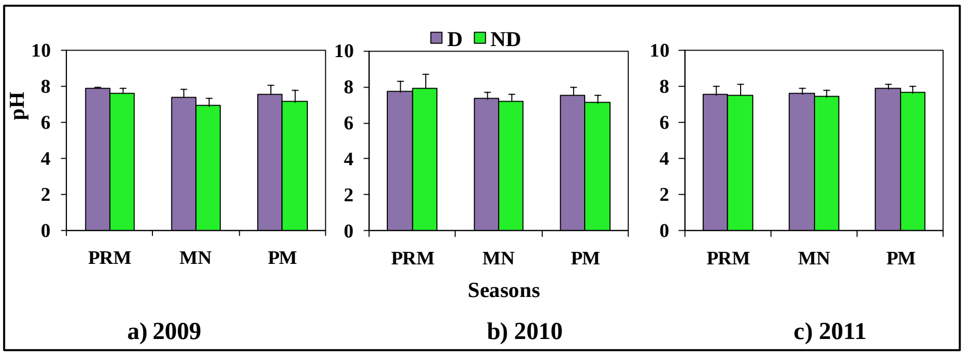


Figure 5.5 Distribution of pH in the dredging and non-dredging sites of Cochin estuary.

Significant temporal variation in pH ($p<0.05$) was observed in the study area (Table 5.2). Spatial variation was statistically insignificant ($p>0.05$) except during 2009 (Table 5.3).

5.4.1.5 Dissolved Oxygen

In 2009, dissolved oxygen was slightly lower in dredging sites (0.6 to 5.0 mg/L) compared to non-dredging sites (1.9 to 4.7 mg/L) (Fig. 5.6a). Lower DO observed during MN (av. 2.5 ± 1.5 mg/L) in dredging sites and during PRM (av. 3.1±0.9 mg/L) in non-dredging sites. Higher DO was recorded during PRM (av. 3.1±0.8 mg/L) in dredging sites and during MN (av. 3.7±0.4 mg/L) in non-dredging sites.

Similar to previous year, during 2010 also DO concentration was slightly lower (1.62 to 6.91 mg/L) in dredging sites compared to non-dredging sites (2.71 to 5.72 mg/L) except during PRM (Fig. 5.6b). Lower DO concentration was observed during PRM in both dredging (av. 2.77 ± 0.6 mg/L) and non-dredging sites (av. 3.23 ± 0.4 mg/L). Higher DO concentration was observed during PM in both dredging (av. 4.66 ± 0.9 mg/L) and non-dredging sites (av. 4.41 ± 0.7 mg/L).

In 2011, DO concentration ranged from 1.14 to 8.04 mg/L in dredging sites and from 0.22 to 7.44 mg/L in non-dredging sites. Lower concentration was recorded in dredging sites relative to non-dredging sites except during PM (Fig. 5.6c). Seasonally, in both dredging and non-dredging sites, lower DO concentration was observed during PM season (D- av. 4.51±1.1 mg/L; ND- av. 4.08±2.2 mg/L) while higher was observed during PRM (D- av. 4.96 ± 1.6 mg/L; ND- av. 5.67± 1.1 mg/L).

DO showed significant seasonal variation in dredging sites ($p<0.05$) only during 2010 while no significant spatial disparity ($p>0.05$) could be observed during the study period (Table 5.2, 5.3).

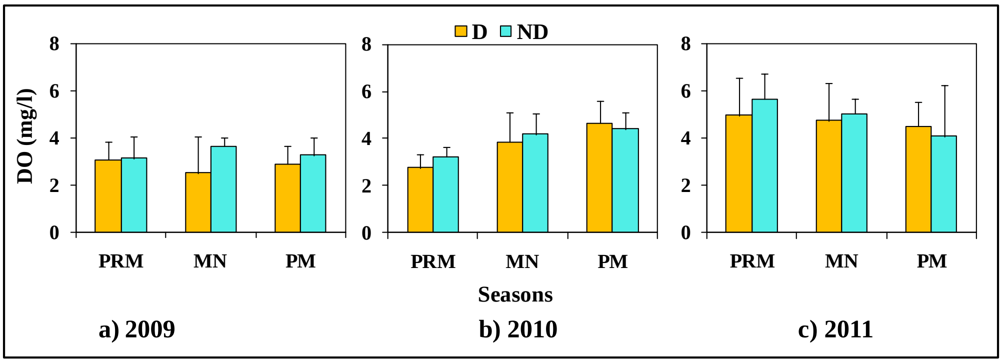


Figure 5.6 Distribution of dissolved oxygen (DO) in dredging and non-dredging sites of Cochin estuary.

5.4.1.6 Biological Oxygen Demand

In 2009, Lower BOD concentration was observed in dredging sites (0.37 and 3.35mg/L) compared to non-dredging sites (0.13 and 3.22 mg/L) except during MN. Lower BOD concentration was observed during MN (D- av. 0.87± 0.6 mg/L; ND-av. 0.62 ± 0.3 mg/L) and higher during PRM (D-av. 2.22 ± 0.9 mg/L; ND- av. 2.41 ± 0.9 mg/L) in dredging and non-dredging sites (Fig. 5.7a).

BOD concentration showed no remarkable variation between dredging (0.2 to 4.17mg/L) and non-dredging sites (0.2 to 2.41mg/L) in 2010 (Fig. 5.7b). In both the study sites, lower BOD concentration was observed during MN (D-av. 1.01±0.6 mg/L; ND-av. 1.0±0.7 mg/L) and slightly higher concentration was observed during PRM (D-av.1.58±1.2 mg/L) (ND-av. 1.61±1.0 mg/L).

In 2011, BOD ranged from 0.68 to 4.51 mg/L in dredging sites and 0.59 to 4.6 mg/L in non-dredging sites (Fig. 5.7c) Similar to 2010, BOD concentration showed no remarked variation between dredging and non-dredging sites. Lower BOD concentration was observed during MN (D-av. 1.82 ±1.1 mg/L; ND- av. 1.53±0.5 mg/L) while, higher BOD concentration was observed during PRM in both dredging (av. 2.34 ±0.9 mg/L) and non-dredging sites (av. 2.93 ±1.3 mg/L).

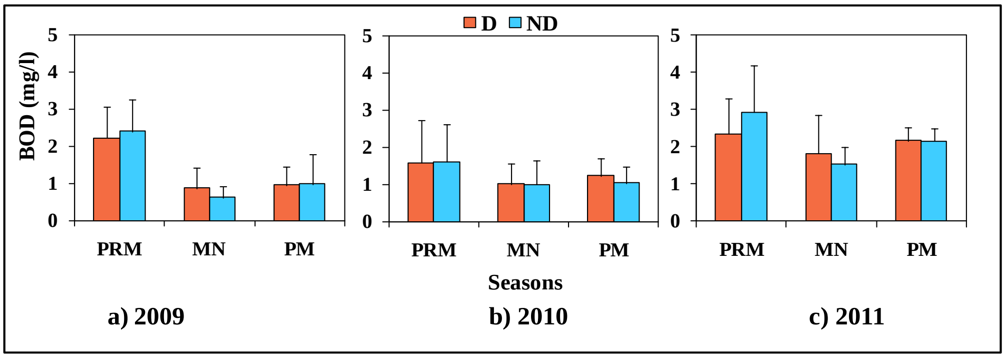


Figure 5.7 Distribution of biological oxygen demand (BOD) in dredging and non-dredging sites of Cochin estuary.

Significant seasonal variation in BOD ($p<0.05$) was observed in non-dredging sites during 2009 and 2011 (Table 5.2). However, the spatial variation in BOD was insignificant ($P>0.05$) (Table 5.3).

5.4.1.7 Suspended Particulate Matter

In 2009, SPM concentration ranged from 2.0 to 864 mg/L in dredging sites and from 9.6 to 73.6 mg/L in non-dredging sites (Fig. 5.8a). Slightly higher concentration of SPM was observed in dredging sites compared to non-dredging sites. Lower SPM was recorded during PM in dredging sites (av.70.80±47.5 mg/L) and during MN in non-dredging sites (av.17.10±7.1 mg/L). Higher SPM concentration was noticed during PRM in both the study sites (D-av.163.8±219.9 mg/L; ND- av. 43.9±18.6 mg/L).

Similar to 2009, SPM concentration was relatively higher in dredging sites (D-10.8 to 242 mg/L; ND- 2.8 to 97.6 mg/L) during 2010 (Fig. 5.8b). Maximum SPM concentration was recorded during PRM in dredging (av.84.90±64.7 mg/L) and non-dredging sites (av.71.50±19.8 mg/L). Minimum concentration was observed during MN in dredging sites (av. 35.35 ±33.5 mg/L) and during PM in non-dredging sites (av. 19.9±9.3 mg/L).

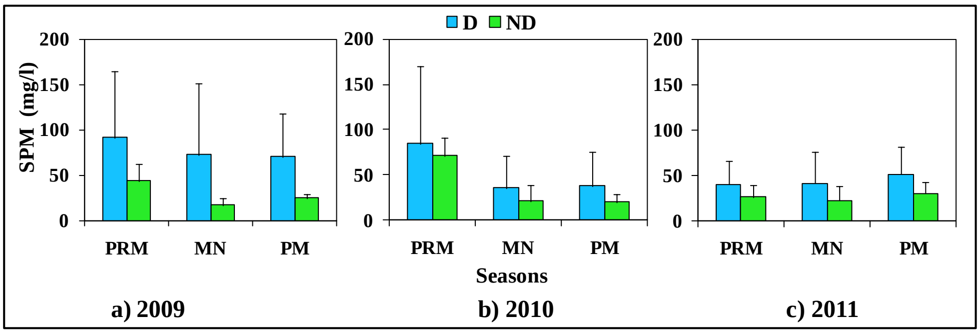


Figure 5.8 Distribution of Suspended particulate Matter (SPM) in dredging and non-dredging sites of Cochin estuary.

In 2011 also, dredging sites exhibited relatively higher SPM (7.2 to 131.2 mg/L) compared to non-dredging sites (4.4 to 46.4 mg/L). Relatively Lower SPM was recorded during PRM (av. 40.6±25.4 mg/L) in dredging sites and during MN (av. 22.5±16.1 mg/L) in non-dredging sites. Higher SPM was recorded during PM (D-av. 51.5±30.9 mg/L; ND- av. 30.3±12.5 mg/L) in dredging and non-dredging sites (Fig. 5.8c). Significant seasonal variation was visible during very few occasions and spatial variation between the two regions was significant ($p<0.05$) during the year 2009 and 2011 (during 2010 during PM) (Table 5.2, 5.3).

5.4.1.8 Inorganic nutrients

5.4.1.8.1 Nitrate

In 2009, nitrate-N ranged from 0.97 to 44.72 µM in dredging sites and from 0.82 to 45.17 µM in non-dredging sites (Fig. 5.9a). Lower concentration of nitrate-N was recorded in dredging sites except during

PRM. Lower nitrate was noticed during PM in dredging sites (av. 11. 4±7.8 μM) and during PRM in non-dredging sites (av. 9.58±10.8 μM). Higher nitrate was recorded during MN (D-av. 35.54±7.8 μM; ND-av. 39.46±6.9 μM) in dredging and non-dredging sites (Fig. 5.9a).

In 2010, nitrate-N varied between 2.02 and 40.32 μM in dredging sites and between 5.75 and 43.32 μM in non-dredging sites. Comparatively lower concentration of nitrate-N was observed in dredging sites relative to non-dredging sites. Higher nitrate-N was observed during MN (D-av. 21.69±10.2 μM; ND- av. 24.29±9.4 μM) in dredging and non-dredging sites (Fig. 5.9b). Lower nitrate was observed during PRM in dredging sites (av. 9.82±5.0μM) and during PM in non-dredging sites (av. 18.84±9.6 μM).

In 2011, lower concentration of nitrate-N (0.8 to 31.42 μM) was recorded in dredging sites compared to non-dredging sites (2.20 to 34.42 μM). Lower nitrate concentration was recorded during PM (D- av. 4.20±2.2 μM; ND- av. 7.36±4.4μM) in both dredging and non-dredging sites. In dredging sites higher nitrate was observed during PRM (av. 9.48±10.4 μM), while in non-dredging sites nitrate was higher during MN season (av. 14.19±8.6μM) (Fig. 5.9c).

Seasonal variation in nitrate-N was statistically significant only in non-dredging sites during 2009. During the study period, it exhibited insignificant ($p>0.05$) spatial variation between dredging and non-dredging sites (except during 2011) (Table 5.2 and 5.3).

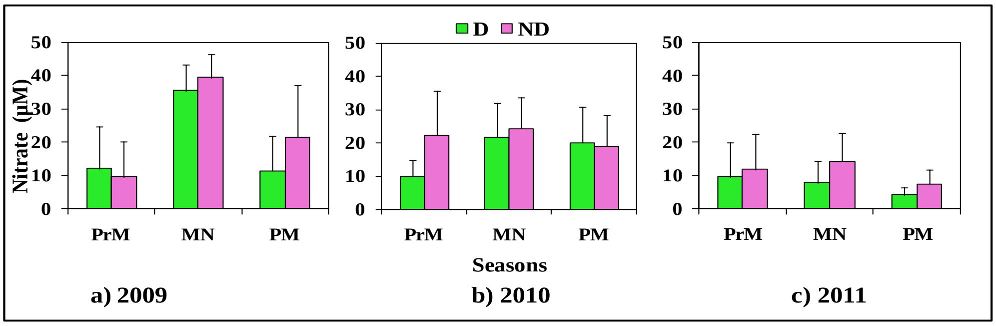


Figure 5.9 Distribution of nitrate in dredging and non-dredging sites of Cochin estuary.

5.4.1.8.2 Nitrite

During 2009, slightly lower nitrite was observed in dredging sites (0.09 to 3.44 µM) relative to non-dredging sites (0.22 to 5.37 µM) except during MN season (Fig. 5.10a). Lower nitrite concentration was recorded during PM season in dredging sites (av. 0.34±0.4 µM) and during MN in non-dredging sites (av. 0.47±0.1 µM). Higher nitrite concentration was observed during PRM in dredging (av. 1.46±1.1µM) and non-dredging sites (av. 1.73±1.8 µM).

Similar to 2009, slightly lower nitrite concentration was recorded in dredging sites (0.04 to 3.6 µM) relative to non-dredging sites (0.13 to 4.1 µM) during 2010. Lower nitrite concentration was observed during PM season (D-av. 0.36±0.2 µM; ND-av. 0.44±0.1 µM) and higher concentration was noticed during PRM (D-av. 1.33±1.3 µM; ND-av.1.19±0.6 µM) in both dredging and non-dredging sites (Fig. 5.10b).

During 2011, nitrite varied from 0.01 to 1.01 µM in dredging sites and from 0 to 1.24 µM in non-dredging sites (Fig. 5.10c). Nitrite-N concentration was relatively low in dredging sites compared to non-dredging sites. Nitrite exhibited lower concentration during PRM (D- av. 0.13±0.1 µM; ND- av.0.17±0.1 µM) and higher during MN (D- av.0.60±0.3µM; ND- av.0.64±0.3µM) in dredging and non-dredging sites.

Significant seasonal variation ($p<0.05$) in nitrite was observed in non-dredging sites during 2009 and 2011 (Table 5.2). Spatial variation in nitrite was insignificant ($p>0.05$) during the study period (Table 5.3).

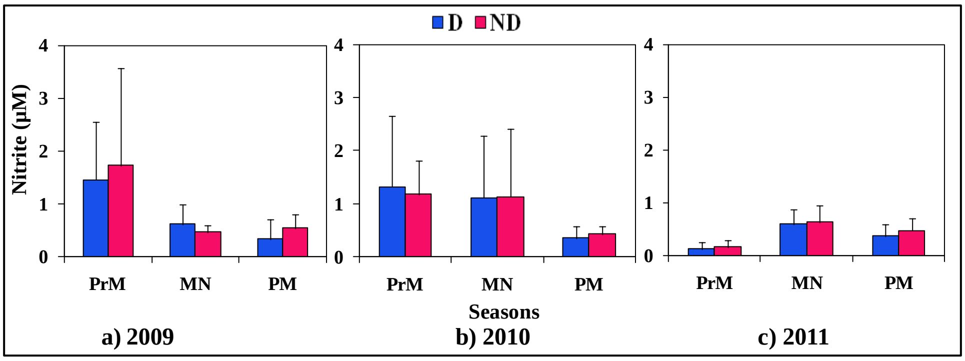


Figure 5.10 Distribution of nitrite in dredging and non-dredging sites of Cochin estuary.

5.4.1.8.3 Ammonia

Ammonia-N concentration during2009, ranged from 1.58 to 34.7 µM in dredging sites and from 3.89 to 152.5 µM in non-dredging sites (Fig. 5.11a). Lower ammonia-N was recorded in dredging sites compared to non-dredging sites except during MN. Lower concentration of ammonia was noticed during MN (D- av.9.37±3.2 µM; ND- av.8.04±2.5 µM) and higher during PRM (D- av.12.99±8.0 µM; ND- av.31.41±50.4 µM) in both dredging and non-dredging sites.

Ammonia concentration was slightly higher in dredging sites (1.97 to 45.82µM) relative to non-dredging sites (4.1 to 45.4 µM) except during PRM 2010 (Fig. 5.11b). In dredging sites higher concentration was observed during MN (av.20.8±10.3 µM) whereas in non-dredging sites concentration was high during PRM (av. 20.08±17.0µM).

Similar to 2009, ammonia concentration in 2011 was slightly lower in dredging sites (0.73 to 73.39 µM) compared to non-dredging sites (1.59 to 47.72 µM) except during MN season (Fig. 5.11c). Lower ammonia concentration was observed during MN season (D-av.7.66±3.1 µM; ND-av.6.19±3.2 µM), while higher concentration of ammonia was noticed during PM in both dredging (av.16.76±7.4 µM) and in non-dredging sites (av.23.89±11.7 µM).

Seasonal variation in ammonia-N was significant ($p<0.05$) in non-dredging sites only during 2011 and no variation ($p>0.05$) could be observed between dredging and non-dredging sites (Table 5.2, 5.3).

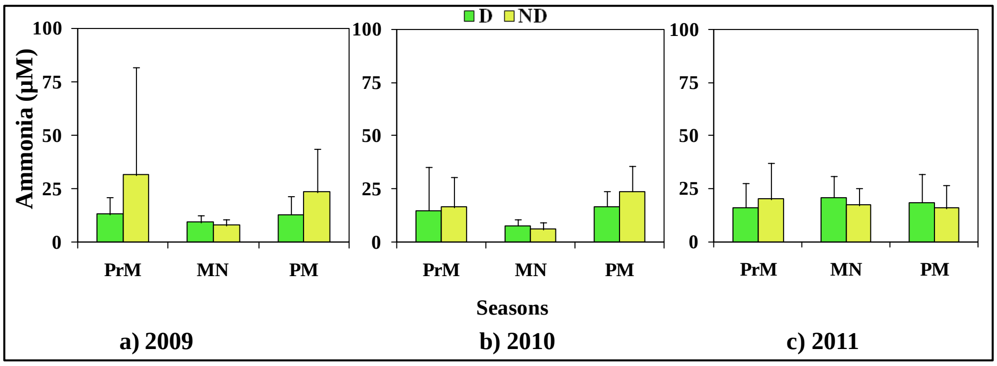


Figure 5.11 Distribution of ammonia in dredging and non-dredging sites of Cochin estuary

5.4.1.8.4 Phosphate

During 2009, phosphate concentration was slightly higher in dredging sites (0.31 to 4.48 μM) relative to non-dredging sites (0.47 to 4.23 μM) except during PRM (Fig. 5.12a). Lower phosphate concentration was noticed during PM season in both the study sites (D-av.1.78±1.1 μM; ND-av.1.74±0.9 μM). Higher phosphate was recorded during MN in dredging sites (av.2.44±0.9 μM), while during PRM (av.3.02±0.7 μM) in non-dredging sites.

During 2010, phosphate concentration ranged between 0.14 and 18.8 μM in dredging sites and 0.35 to 6.62 μM in non-dredging sites (Fig. 5.12b). Slightly higher concentration of phosphate was observed in dredging sites compared to non-dredging sites except during MN. Lower phosphate concentration was noticed during MN in dredging sites (av.1.24±1.2 μM) and during PM in non-dredging sites (av.0.87±0.5 μM). Phosphate concentration was higher during PRM (D-av.5.12±3.2 μM; ND-av. 4.54±2.5 μM) in both dredging and non-dredging sites.

Phosphate concentration during 2011 fluctuated from 0.10 to 2.84 μM in dredging sites and 0.29 to 2.52 μM in non-dredging sites (Fig. 5.12c). Slightly higher concentration of phosphate was observed in dredging sites compared to non-dredging sites except during PRM season. Phosphate concentration was lower during PRM season (D- av.0.51±0.3 μM; ND-av.0.57±0.3 μM), while higher during MN (D- av.1.85±0.6 μM; ND-av.1.70±0.6 μM) in dredging and non-dredging sites.

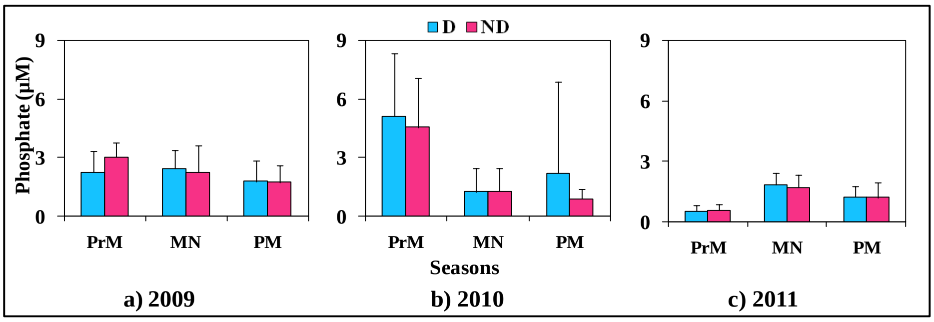


Figure 5.12 Distribution of phosphate in dredging and non-dredging sites of Cochin estuary.

Seasonal variation was significant ($p<0.05$) in study sites during 2010 and 2011 and spatial variation was insignificant ($p>0.05$) between the sampling sites throughout the study (Table 5.2, 5.3).

5.4.1.8.5 Silicate

During 2009, dredging sites exhibited lower silicate concentration (6.83 to 106.66 µM) compared to non-dredging sites (8.85 to 133.87 µM) (Fig. 5.13a). During study period, silicate concentration was lower during PRM (D- av.21.58±14.9 µM; ND- av.33.84±22.5 µM) and higher during MN (D-av.49.51±37.3 µM; ND- av. 74.42±52.1 µM) in both dredging and non-dredging sites.

Similar to 2009, silicate concentration exhibited lower concentration in dredging sites (0.64 to 102.5 µM) relative to non-dredging sites (2.91 to 107.53 µM) during 2010 (Fig. 5.13b). Lower silicate was recorded during PRM (D-av.21.08±16.3 µM; ND-av.28.89±30.3 µM), whereas higher silicate was recorded during MN (D-av.33.80±22.8 µM; ND-av.48.44±36.2 µM) in both the study sites.

In 2011 period, silicate ranged between 5.44 and 69.40 µM in dredging sites and between 1.9 and 83.53 µM in non-dredging sites (Fig. 5.13c). Concentration of silicate was relatively lower in dredging sites compared to non-dredging sites, which was recorded lower during PRM (D-av. 12.89±5.8 µM; ND- av. 13.20±9.3 µM) and higher during MN (D-av.37.91±19.3 µM; ND-av.42.22±24.2 µM).

Significant seasonal variation in silicate was observed only in dredging sites during 2009 and 2011(Table 5.2). Spatial variation was statistically insignificant ($p>0.05$) in the study period (Table 5.3).

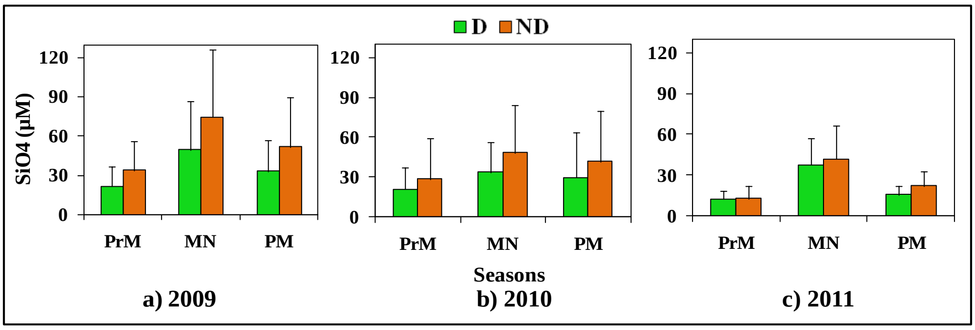


Figure 5.13 Distribution of silicate in dredging and non-dredging sites of Cochin estuary.

5.4.2 Sediment characteristics

5.4.2.1 Sediment texture

In 2009, finer fractions of the sediment (silt and clay) were dominated in dredging sites whereas coarser particles (sand fraction) predominated in non-dredging sites (Fig. 5.14a-c). Sand varied between 0.05 and 70.81% in dredging sites and between 8.63 and 88.15% in non-dredging sites (Fig. 5.14a). Lower sand was observed during MN in dredging sites (av.10.7±9.9%) and during PM in non-dredging sites (av.

53.3±28.3%). Higher percentage of sand was observed during PM in dredging sites (av. 16.57±25.9%) and during MN in non-dredging sites (av. 62.36±28.3%).

Relatively higher silt fraction was recorded in dredging sites (10.63 to 65.57%) compared to non-dredging sites (0.35 to 71.69%) (Fig.5.14b). Lower percentage of silt was recorded during MN in dredging sites (av.27.42±10.5%) and during PRM in non-dredging sites (av.15.8±8.8%) whereas higher silt was recorded during PM (D- av.37.46±10.9%; ND- av.25.12±26.5%).

Percentage of clay was higher in dredging sites compared to non-dredging sites and it varied from 3.5 to 73.5% and from 1.5 to 5.5% respectively (Fig. 5.14c). Lower clay was noticed during PM in dredging (av.45.98±19.6%) and non-dredging sites (av.19.20±9.4%). Higher percentage of clay was observed during MN in dredging sites (av.61.87±9.0%) and during PRM in non-dredging sites (av.27.8±20.7%).

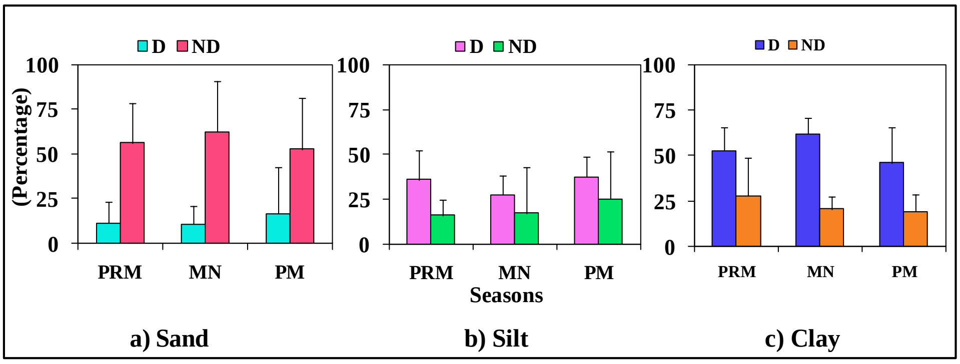


Figure 5.14 Distribution of sediment texture (**a**-Sand, **b**-Silt and **c**-Clay) in dredging and non-dredging sites of Cochin estuary during 2009.

Similar to previous year, sediment texture in dredging sites was dominated by finer fractions during 2010 (Fig. 5.15a-c). Sand varied between 1.60 to 71.3% in dredging sites and 7.88 to 67.20% in non-dredging sites. Percentage of sand was slightly higher during PM in dredging sites (av.31.84±21.4%) and during MN in non-dredging sites (av. 43.39±13.4%) (Fig. 5.15a).

Silt was comparatively higher in dredging sites (0.81 to 85.5%) than non-dredging sites (0.09 to 60.0%) except during PM season (Fig. 5.15b). Silt percentage was lower during PM in dredging sites (av. 17.54±13.8%) and during PRM in non-dredging sites (av. 9.74±15.7%). Higher silt was observed during PRM in dredging (av.22.11±26.6%) sites and during PM in non-dredging sites (av. 22.46±25.3%).

Clay percentage was higher in dredging sites compared to non-dredging sites except during PRM (Fig. 5.15c). Clay percentage varied from 8.08 to 80.23% in dredging sites and from 8.0 to 69.50 in non-dredging sites. Dredging sites contained greater than 50% of clay irrespective of seasons.

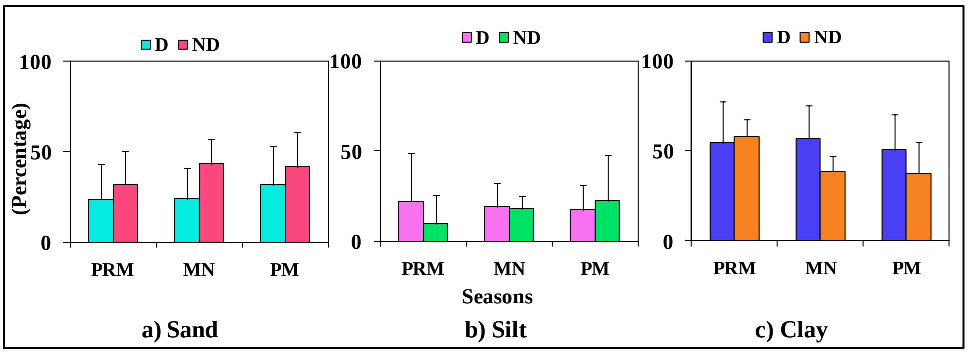


Figure 5.15 Distribution of Sediment texture (**a**-Sand, **b**-Silt and **c**-Clay) in dredging and non-dredging sites of Cochin estuary during 2010.

During 2011, sand fraction was lower, and finer fractions were dominated in dredging sites (Fig. 5.16). Sand fraction varied from 0.05 to 67.3% in dredging sites and from 0.59 to 82.2% in non-dredging sites (Fig. 5.16a). In dredging sites, percentage of sand was high during MN (av. 18.24±16.6%) whereas in non-dredging sites it was high during PRM (av. 57.47±27.7%). Lower sand percentage was noticed during PM in dredging sites (av. 14.08±12.3%) and during MN in non-dredging sites (av. 42.07±26.63%).

Irrespective of seasons silt was higher in dredging sites (0.62 to 66.99%) relative to non-dredging sites (0.09 to 54.83%) (Fig. 5.16b). Lower silt fraction was noticed during PRM in dredging sites (av. 32.45±10.7%) and during PM in non-dredging sites (av. 12.96±11.2%). Higher percentage of silt was observed during PM in dredging sites (av. 36.64±18.8%) and during PRM (av. 17.92±19.8%) in non-dredging sites.

Similar to silt fraction, clay was also higher in dredging sites (8.08 and 96.04%) compared to non-dredging sites (10.12 and 86.57%) (Fig. 5.16c). Minimum concentration of clay was noticed during MN in dredging sites (av. 48.06±20.1%) and during PM in non-dredging sites (av. 29.80±16.7%) while maximum clay fraction was observed during PRM in dredging sites (av. 49.69±18.3%) and during MN in non-dredging sites (av. 40.21±22.5%).

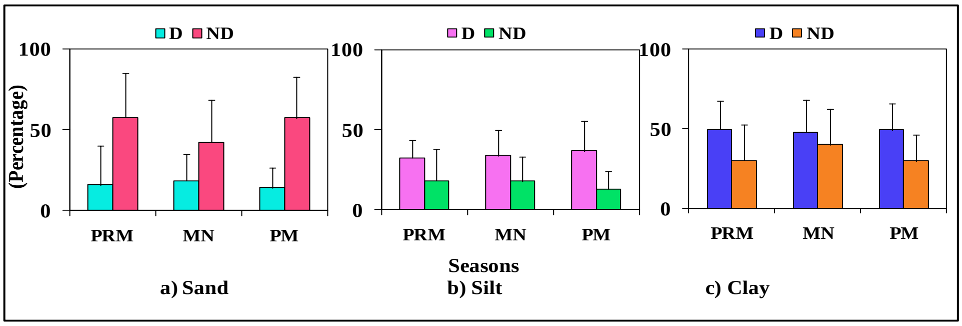


Figure 5.16 Distribution of sediment texture (**a**-Sand, **b**-Silt and **c**-Clay) in dredging and non-dredging sites of Cochin estuary during 2011.

Significant spatial variation ($p<0.05$) in sediment texture (sand, silt, and clay) was observed between dredging and non-dredging sites, whereas variation was insignificant between the seasons ($p>0.05$) (Table 5.2, 5.3).

5.4.2.2 Sediment organic carbon

Sediment organic carbon during 2009 fluctuated from 7.59 to 40.37 mg/g in dredging sites and from 2.07 to 32.09 mg/g in non-dredging sites. Organic carbon was comparatively higher in dredging sites than non-dredging sites irrespective of seasons (Fig. 5.17a). In dredging sites lower organic carbon was noticed during MN (av. 24.45±3.7 mg/g) and higher recorded during PM (av. 26.69±10.1 mg/g). In non-dredging sites lower organic carbon recorded during PRM (av. 14.10±6.2 mg/g) and higher during PM (av. 15.93±11.3 mg/g).

Similar to previous year during 2010 and 2011 also, organic carbon was apparently higher in dredging sites than non-dredging sites. During 2010 it varied from 5.86 to 41.39 mg/g in dredging sites and from 6.83 to 31.26 mg/g in non-dredging sites (Fig. 5.17b). In dredging sites lower organic carbon was recorded during PRM (av. 15.78±7.8mg/g), while higher during MN (av. 26.80±5.5 mg/g). In non-dredging sites lower organic carbon was recorded during MN (av. 15.71±8.3 mg/g) and higher during PRM (av. 17.59±4.05 mg/g).

Sediment organic carbon during 2011 fluctuated from 3.27 to 34.40 mg/g in dredging sites and from 3.40 to 22.30 mg/g in non-dredging sites (Fig. 5.17c). Lower organic carbon was recorded during MN in dredging (av. 14.51±5.7 mg/g) and non-dredging sites (av. 11.11±4.9 mg/g). The organic carbon was higher during PM in dredging (av. 20.93±6.8 mg/g) and non-dredging sites (av. 13.62±7.5 mg/g).

Significant spatial variation ($p<0.05$) in sediment organic carbon was noticed between dredging and non-dredging sites (Table 5.2) whereas statistically significant seasonal variation was noticeable only in dredging sites during 2010 (Table 5.3).

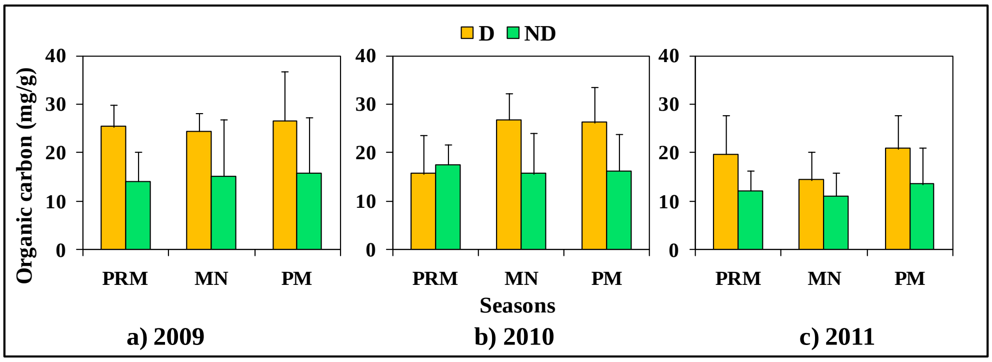


Figure 5.17 Distribution of Sediment organic carbon in dredging and non-dredging sites of Cochin estuary.

Table 5.2 *Results of One way ANOVA of major environmental parameters in dredging and non-dredging stations 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
Nitrate	3.38	0.004**	0.23	0.13	0.12	0.27
Nitrite	0.002**	0.16	0.40	0.23	1.28	0.002**
Ammonia	0.49	0.58	0.64	0.83	0.12	0.13
Phosphate	0.35	0.06	0.04*	0.001**	3.94	0.004**
Silicate	0.04*	0.20	0.45	0.68	0.003**	0.05
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
Abundance	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 5.3 *P value of results of non-parametric t-test between dredging and non-dredging stations (*-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*
Nitrate	0.50	0.51	0.04*
Nitrite	0.58	0.73	0.36
Ammonia	0.07	0.59	0.48
Phosphate	0.33	0.46	0.18
Silicate	0.06	0.68	0.97
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**

5.4.3 Macrobenthic community

5.4.3.1 Macrobenthic density

During 2009, macrobenthic density varied between 20 and 7440 ind.m^{-2} in dredging sites and between 347 and 24720ind.m^{-2} in non-dredging sites. Lower density was observed in dredging sites compared to non-dredging sites throughout the study period (Fig. 5.18a). Lower density was observed during MN in dredging sites (av.935±606 ind.m^{-2}) and during PM in non-dredging sites (av.2715±1868 ind.m^{-2}). In dredging sites higher density was recorded during PM (av.2178 ±2098 ind.m^{-2}) and in non-dredging sites, during PRM (av.7403±7399 ind.m^{-2}) season.

During 2010, density was slightly lower in dredging sites (40 to 6340 ind.m^{-2}) relative to non-dredging sites (101 to 28140 ind.m^{-2}), throughout the study period (Fig. 5.18b). Lower density was recorded during PRM (av.1020±724 ind.m^{-2}) in dredging sites and during MN season in non-dredging sites (av.2215±1287 ind.m^{-2}). Higher density was observed during MN in dredging sites (av.2194±2191 ind.m^{-2}) and during PM in non-dredging sites (av.5575±9281 ind.m^{-2}).

During 2011, macrobenthic density varied between 6.0 and 12600 ind.m^{-2} in dredging sites and between 154 and 7440 ind.m^{-2} in non-dredging sites. Macrobenthic density was higher in dredging sites relative to non-dredging sites (Fig. 5.18c). In dredging sites, lower density was recorded during PM (av.1126±1434 ind.m^{-2}) while higher recorded during MN (av. 1955±3179 ind.m^{-2}). In non-dredging sites, lower density was observed during PRM (av. 1905±1215 ind.m^{-2}) and higher during PM (av.2342±2659 ind.m^{-2}).

Macrobenthic density showed insignificant ($p>0.05$) variation between seasons (Table 5.2) while, the spatial variation was statistically significant ($p<0.05$) (Table 5.3).

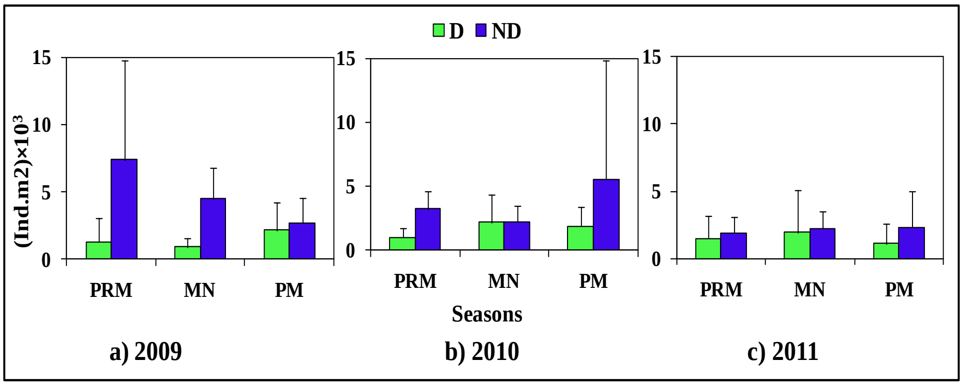


Figure 5.18 Distribution of macrobenthic density in dredging and non-dredging sites of the Cochin estuary.

5.4.3.2 Macrobenthic biomass

During 2009, macrobenthic biomass ranged from 0 to 99.42g.m^{-2} in the dredging sites and from1.21 to 66.14g.m^{-2} in non-dredging sites. Lower biomass was observed in dredging sites compared to non-dredging sites (Fig. 5.19a). Relatively higher biomass was recorded during MN in dredging sites (av.18.57±33.5 g.m^{-2}) and during PRM in non-dredging sites (av.25.8±21.6 g.m^{-2}). Lower biomass was recorded during PM in both dredging (av.6.02±5.69 g.m^{-2}) and non-dredging sites (av.15.73±11.7g.m^{-2}).

Similar to density, macrobenthic biomass in 2010 also was lower in dredging sites (0.18 to 106.46g.m^{-2}) compared to non-dredging sites (0.06 to 128.98g.m^{-2}). Higher macrobenthic biomass was observed during PRM in dredging (av.19.82±35.5 g.m^{-2}) and non-dredging sites (av.45.79±54.9 g.m^{-2}). Lower biomass was recorded during MN in dredging (av.12.81±17.1g.m^{-2}) and non-dredging sites (av.17.86±23.01 g.m^{-2}).

During 2011, the macrobenthic biomass varied from 0.07 to 55.1g.m^{-2} in dredging sites and from 0.26 to 57.28g.m^{-2} in non-dredging sites. Relatively lower biomass was noticed in dredging sites than non-dredging sites throughout the period (Fig. 5.19c). Macrobenthic biomass was higher during PM in dredging sites (av.5.92±13.6 g.m^{-2}) and during PRM (av.22.09±16.4 g.m^{-2}) in non-dredging sites. Relatively lower biomass was noticed during MN season in both dredging (av.3.61±3.9 g.m^{-2}) and non-dredging sites (av.9.88±3.5 g.m^{-2}).

No prominent seasonal variation in biomass ($p>0.05$) was evident throughout the study period, whereas statistically significant spatial variation ($p<0.05$) was noticeable during 2009 and 2011 (Table 5.2, 5.3).

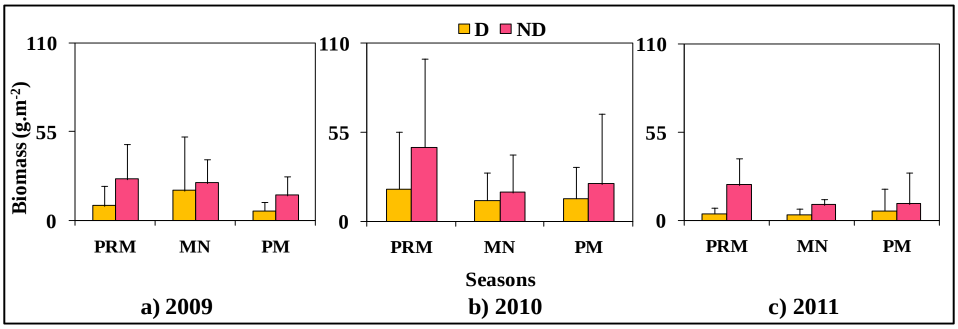


Figure 5.19 Distribution of macrobenthic biomass in dredging and non-dredging sites of Cochin estuary.

5.4.3.3 Macrobenthic community composition

During the study period from 2009 to 2011, 81 macrobenthic taxa belonging to 6 phyla were encountered in the study. Major macrobenthic groups identified in the study period include polychaetes, amphipods, oligochaetes, gastropods, bivalves, tanaids, isopods, decapods and cumaceans.

During 2009, among the different groups, polychaetes (70%) dominated in non-dredging sites along with amphipods (21%). Polychaetes

(59%) along with oligochaetes (24%) were dominated in dredging sites (Fig. 5.20). In non-dredging sites groups such as gastropods (3%), bivalves (2%), isopods (1.1%), oligochaetes (1%), tanaids (0.7%), and cumaceans (0.3%) contributed a minor portion to the total abundance. In dredging sites, amphipods (7%), tanaids (6%), gastropods (1.2%), bivalves (0.8%), and decapods (0.3%) were the minor groups observed. During PRM, polychaetes (79.7%), amphipods (13%), gastropods (3.5%), bivalves (2.4%), and isopods (0.7%) constituted the fauna in non-dredging sites, while polychaetes contributed 80.4% to the total density, amphipods to 7.6%, oligochaetes to 6.0%, gastropods to1.9%, and bivalves to 1.4% in dredging sites. During MN polychaetes contributed 60.3% to the total, amphipods to 32.6%, isopods to 1.6%, bivalves to 1.5%, tanaids to 1.3% etc in non-dredging sites, whereas 69.5% of polychaetes, 13.9% of amphipods and 8.8% of oligochaetes, and 2.4% of gastropods constituted the fauna in dredging sites. During PM, 59.6% contributed by polychaetes, 23.3% by amphipods, 4.6% by gastropods, 3.9% by oligochaetes, 1% by cumaceans and bivalves to the total fauna in non-dredging sites. In dredging sites oligochaetes were found to increase in number during MN and PM seasons. During PM 42.6% was contributed by oligochaetes, 40.1% by polychaetes, 11.3% by tanaids, 0.5% by bivalves and 4.7% by amphipods contributed to the total macrobenthic fauna in dredging sites.

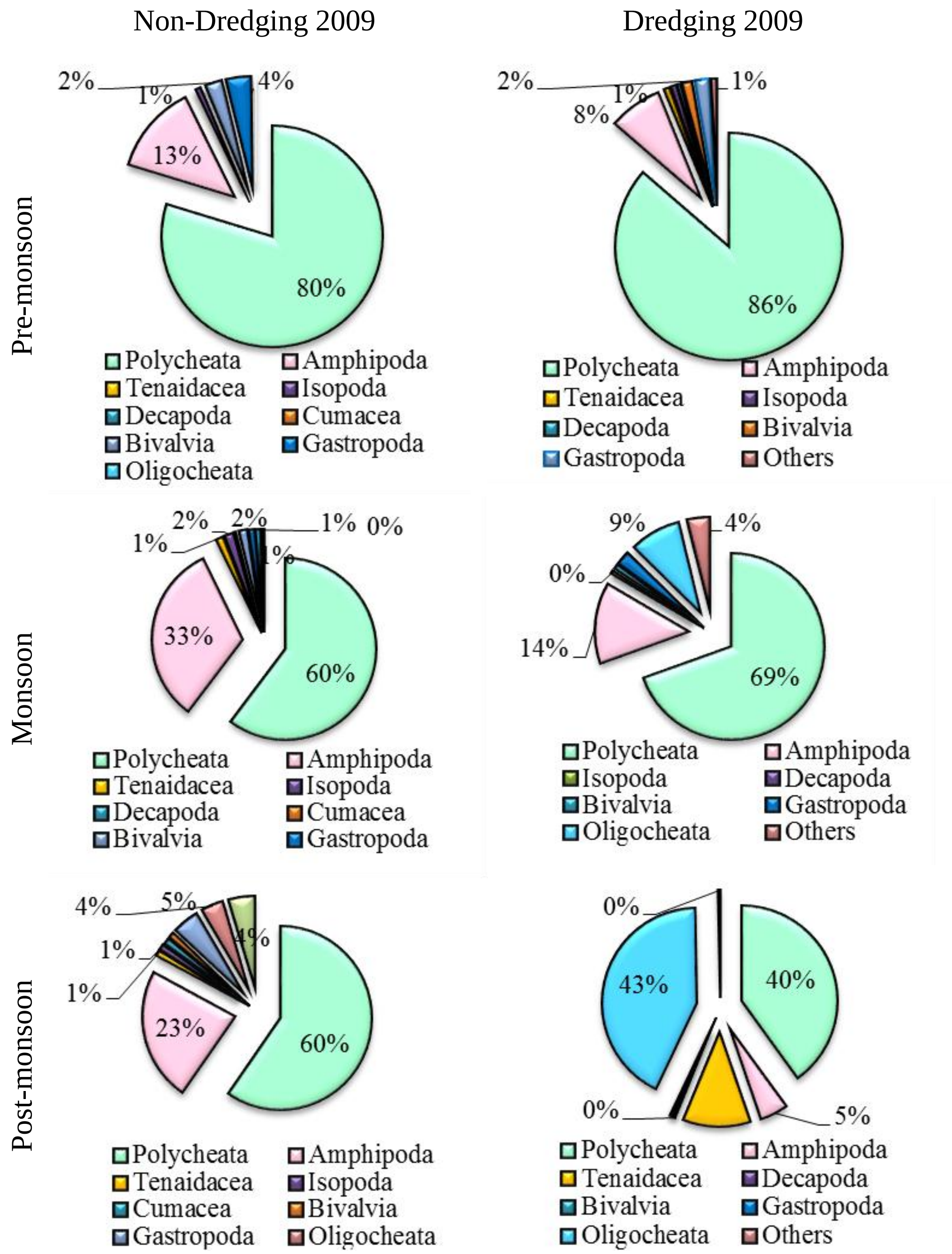


Figure 5.20 Macrobenthic community composition in dredging and non-dredging sites of Cochin estuary during 2009

Among the different macrobenthic groups observed during 2010, polychaetes (73.9%) and amphipods (15.1%) form major groups throughout the study period in non-dredging sites. In dredging sites, polychaetes (49.2%) along with oligochaetes (35.3%) formed the dominant fauna throughout the study period (Fig. 5.21).

During PRM, polychaetes (73.9%), amphipods (11.1%), bivalves (6.9%), gastropods (3.4%), and oligochaetes (3.2%) composed the fauna in the non-dredging sites, whereas polychaetes (71.3%), oligochaetes (13.7%), bivalves (5.6%), gastropods (3.4%), amphipods (2.5%), and tanaids (2.2%) constituted the fauna in dredging sites.

During MN, polychaetes (78%), amphipods (13.7%), gastropods (2.3%), bivalves (1.9%), and oligochaetes (1.9%) comprised the fauna in non-dredging sites, while oligochaetes (54.5%), polychaetes (37.5%), tanaids (4.6%) amphipods (2.5%), and decapods (0.4%) represented the fauna in dredging sites.

During PM, polychaetes (70%), amphipods (20.5%), isopods (3.4%), bivalves (2.1%), and tanaids (1.7%) composed the fauna in the non-dredging sites, while in dredging sites, polychaetes (38.7%), oligochaetes (37.8%), tanaidaceans (13.4%) amphipods (5%), isopods and amphipods (1.2%) comprised the fauna.

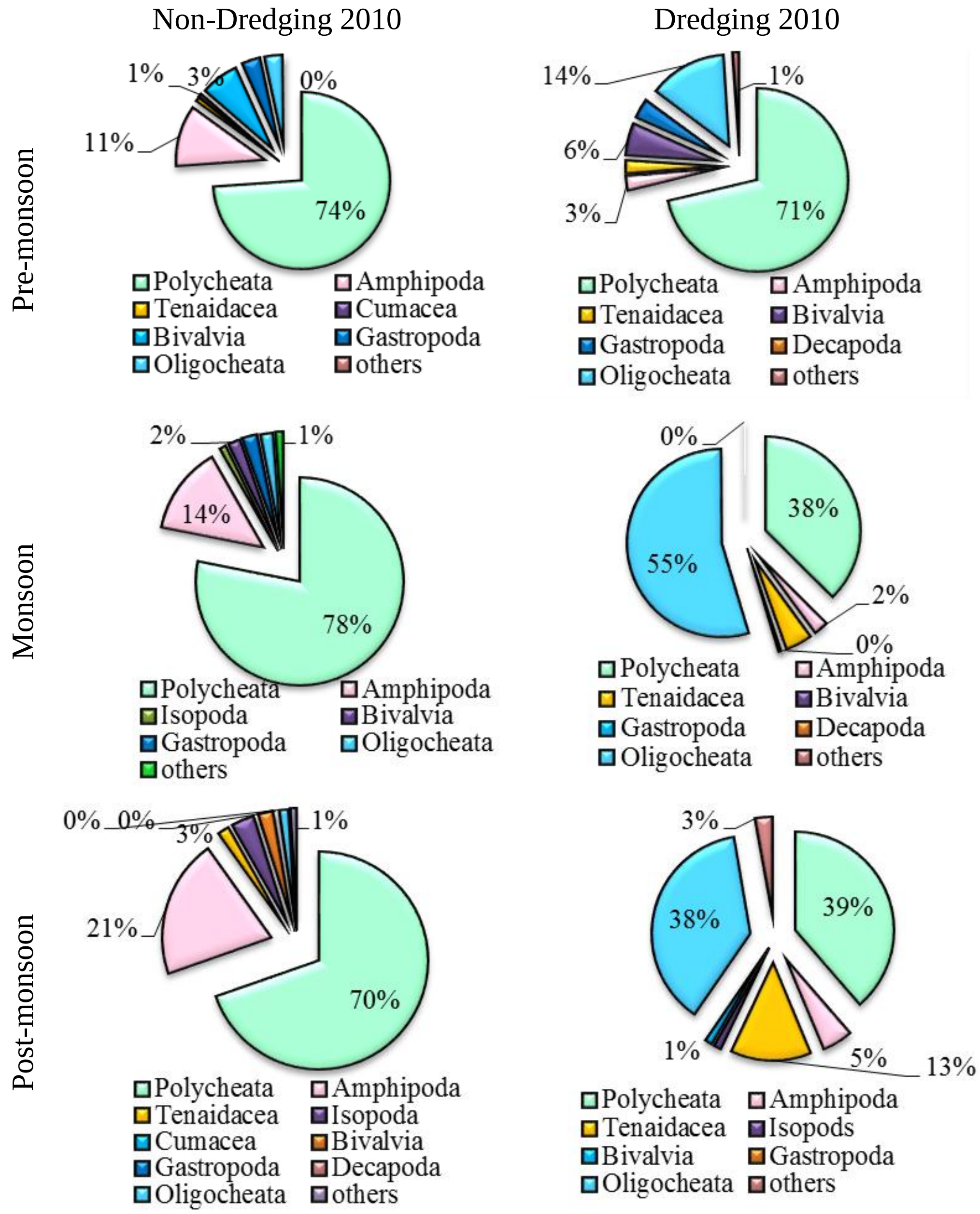


Figure 5.21 Macrobenthic community composition in dredging and non-dredging sites of Cochin estuary during 2010

Among the different groups identified in the present study during 2011, polychaetes (69.3%) and amphipods (23%) were dominated in non-dredging sites, while oligochaetes (64.8%) and polychaetes (27.6%) constituted the dominant fauna in dredging sites (Fig. 5.22).

During PRM, polychaetes (63.2%), amphipods (21.3%), oligochaetes (6.3%), tanaids (4.3%), bivalves (2.8%), and gastropods (2%) constituted the fauna in non-dredging sites. In dredging sites oligochaetes (69.6%), polychaetes (27.8%), and tanaids (1.2%) constituted the fauna.

During MN, polychaetes (78.6%), amphipods (17.5%), bivalves (1.5%), and oligochaetes (0.9%) comprise the fauna in non-dredging sites, while in dredging sites oligochaetes (71.3%), polychaetes (24%), amphipods (4.4%), isopods (0.2%), and bivalves (0.1%) constituted the fauna.

During the PM, polychaetes (66.2%), amphipods (30.3%), bivalves (1.8%), oligochaetes (0.7%), tanaids (0.5%) and isopods (0.3%) were the major groups present in non-dredging sites, while oligochaetes (53.5%), polychaetes (31.1%), tanaids (9.04%), amphipods (4.9%), bivalves (0.7%), and isopods (0.2%) and were the macrobenthic groups present in dredging sites.

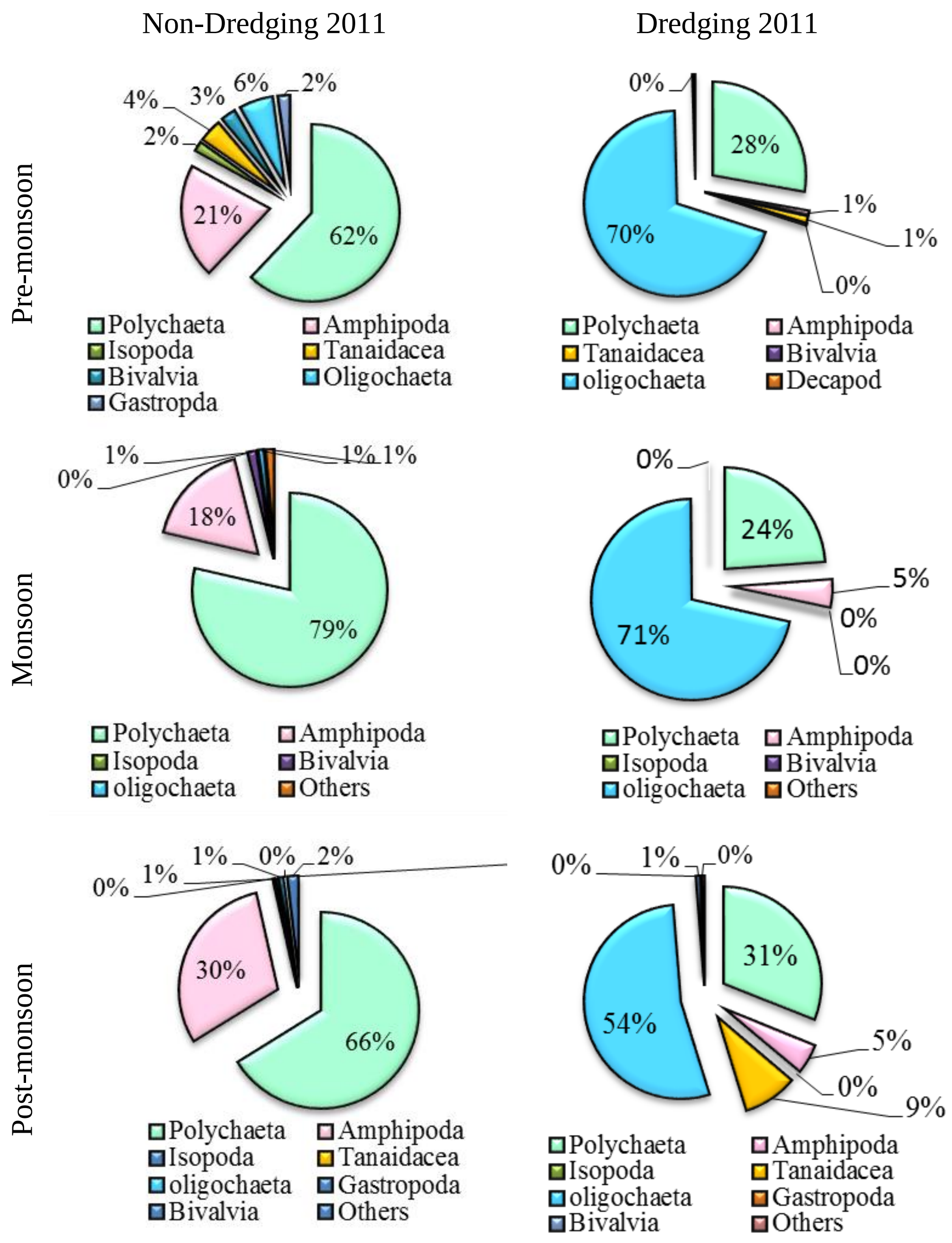


Figure 5.22 Macrobenthic community composition in the dredging and nondredging sites of Cochin estuary during 2011

5.4.3.4 Polychaete community composition

Among the macrobenthic fauna, polychaetes belongs to the family Capitellidae, Spionidae, Nereidae, Eunicidae and Owenidae were dominant in non-dredging sites, whereas Capitellidae, Nephtydae, Pilargidae, Spionidae, and Cirratulidae were the major families observed in dredging sites.

During 2009, *Nephtys oligobranchia* (11.2%), *Mediomastus capensis* (8.2%), *Sigambra parva* (7.9%), *Cirratulus filiformis* (7.3%), and *Prionospio cirrifera* (6.4%) were the major polychaete species observed in dredging sites while*, Mediomastus capensis* (14%), *Diopatra neapolitana* (11.3%), *Prionospio cirrifera* (7.7%), and *Paraheteromastus tenuis* (6.8%) constituted the major polychaete species in non-dredging sites. During PRM, *Nephtys oligobranchia* (14.6%), *Diopatra neapolitana* (12.3%), *Sigambra parva* (10.8%), and *Mediomastus capensis* (10.3%), were the major polychaete species observed in dredging sites wheras, *Mediomastus capensis* (20.6%), *Diopatra neapolitana* (15.6%), *Paraheteromastus tenuis* (10.1%), *Prionospio cirrifera* (5.1%), were the major polychaete species observed in non-dredging sites. During MN *Nephtys oligobranchia* (12.9%), *Cirratulus filiformis* (12.9%), *Mediomastus capensis* (9.7%), *Prionospio cirrifera* (9.7%), and *Cirratulus cirratus* (4.5%) constituted the polychaete species in dredging sites, whereas *Mediomastus capensis* (18.0%), *Prionospio cirrifera* (15.9%), *Paraheteromastus tenuis* (9%), and *Dendronereis estuarina* were the major species in non-dredging sites. During PM Sigambra *parva* (10.9%), *Nephtys oligobranchia* (6%), *Mediomastus capensis* (4.6%), and *Cirratulus filiformis* (4%) were the major species in dredging sites whereas *Diopatra neapolitana* (17%), *Mediomastus capensis* (4.6%), *and Prionospio cirrifera* (2.2%) formed the major species in non-dredging sites (Table 5.6).

During 2010, polychaete species such as *Mediomastus capensis* (10.29%), *Sigambra parva* (6.81%), *Cossura coasta* (4.86%), and *Paraheteromastus tenuis* (4.3%), constituted the major polychaete species in dredging sites whereas *Mediomastus capensis* (20.39%), *Paraheteromastus tenuis* (15.5%), *Prionospio cirrifera* (11.69%), and *Diopatra neapolitana* (4.7%), constituted the major polychaete species in non-dredging sites. During PRM, *Mediomastus capensis* (19. 9%), *Cossura coasta* (14.6%), *Sigambra parva* (8.6%), *Cirratulus filiformis* (8.6%), *Paraheteromastus tenuis* (8.6%), and *Pectinaria* sp (8.6%), were the major species in dredging sites while, *Mediomastus capensis* (37.9%), *Diopatra neapolitana*(12.6%), *Prionospio cirrifera* (8%), and *Prionospio cirrobranchiata* (7.7%) constituted the polychaete species in non-dredging sites. During MN *Paraheteromastus tenuis* (32.2%), *Prionospio cirrifera* (11.1%), and *Mediomastus capensis* (6.7%), were the major species in non-dredging sites, while *Sigambra parva* (4.2%), and *Paraheteromastus tenuis* (4.3%), were the major species in dredging sites. During PM *Mediomastus capensis* (16.6%), *Prionospio cirrifera* (16%), and *Paraheteromastus tenuis* (4.3%), constituted the major polychaete species in non-dredging sites whereas *Mediomastus capensis* (11.3%), *Sigambra parva* (7.6%), *and Cirratulus filiformis* (4.2%) were the major polychaete species in dredging sites (Table 5.7).

During 2011 *Cirratulus filiformis* (8.6%) *Mediomastus capensis* (4.9%), and *Nephtys oligobranchia* (3.7%), constituted the major polychaete species in dredging sites whereas *Mediomastus capensis* (6.7%), *Caulleriella capensis* (13.1%) *Prionospio cirrifera* (6.2%) and *Dendronereis estuarina* (4.7%), constituted the major polychaete species in non-dredging sites. During PRM *Nephtys oligobranchia* (4.1%), *Sigambra parva* (4.1%), *Mediomastus capensis*

(4.1%), and *Cossura coasta* (2.9%) were the major polychaete species in dredging sites whereas *Mediomastus capensis* (29.8%), *Paraheteromastus tenuis* (10.7%), *Prionospio cirrifera* (4%) and *Capitella capitata* (4%) were the major polychaete species in non-dredging sites. During MN *Cirratulus filiformis* (24.5%) *Nephtys oligobranchia* (4.6%), and *Mediomastus capensis* (3.7%), were the major polychaete species in dredging sites while *Mediomastus capensis* (19%), *Prionospio cirrifera* (12.5%) and *Dodecaceria* sp were the major polychaete species in non-dredging sites During MN *Cirratulus filiformis* (8.6%) *Mediomastus capensis* (4.9%), *Nephtys oligobranchia* (3.7%),and *Sigambra parva* (2.8%) were the major polychaete species in dredging sites, while *Mediomastus capensis* (22.1%), *Caulleriella capensis* (13.1%) (13.1%) *Prionospio cirrifera* (6%) and *Dendronereis estuarina* (4.7%) were the major polychaete species in non-dredging sites (Table 5.8).

5.4.3.5 Other macrobenthic taxa

Among the identified amphipods during 2009, species such as *Melita zylanica* (0.6%), *Photis digitata* (0.5%), *Gammaropsis sp* (0.3%), *Cheriophotis megacheles* (0.1%) and *Eriopisa chilkensis* (0.1%) were observed from dredging sites whereas *Eriopisa chilkensis* (4.7%), *Caprella* sp (4.1%), *Melita zylanica* (1.4%), *Photis digitata* (1.3%), *Corophium triaenonyx* (0.7%) *and Cheriophotis megacheles* (0.5%) were observed in non-dredging sites. During PRM, *Photis digitata* (1.1%) was observed in dredging sites whereas *Eriopisa chilkensis* (12.1%), *Caprella* sp (3.9%), *Melita zylanica* (2.3%), and *Cheriophotis megacheles* (0.8%) were observed in non-dredging sites. During MN season, *Melita zylanica* (1.3%) was noticed in dredging sites while *Caprella* sp (5.6%), *Photis digitata* (3.4%), *Eriopisa chilkensis* (2.1%), *Melita zylanica* (1.7%), *Corophium triaenonyx* (1.7%) and *Cheriophotis megacheles* (0.9%) were observed in non-dredging sites. During PM season *Gammaropsis sp* (0.3%), *Photis digitata* (0.3%), *Eriopisa chilkensis* (0.3%), and *Cheriophotis megacheles* (0.9%) were observed in dredging sites while *Caprella* sp (2.8%) dominated in non-dredging sites (Table 5.6).

Other macrobenthic taxa includes Tubificidae sp-22% (Oligochatea), Tanaidacea species *Apseudus chilkensis* (5.9%), and foraminifera (0.8%) were observed in dredging sites while foraminifera (6.7%), gastropod species *Littorina littorea* (1.6%), isopod species *Cirolana fluviatilis* (1.3%), *Apseudus chilkensis* (0.9%) were noticed from non-dredging sites.

During 2010 of the identified amphipods (1.5%), species such as *Corophium triaenonyx* (0.82%), *Photis digitata* (0.44%), and *Melita zylanica* (0.1%) observed in dredging sites while species such as *Corophium triaenonyx* (6.92%), *Caprella* sp (6.78%), *Photis digitata* (4.46%), *Eriopisa chilkensis* (1.67%), and *Cheriophotis megacheles* (1.06%) were observed in non-dredging sites. During PRM, *Corophium triaenonyx* (2.1%), and *Photis digitata* (1.3%), were the amphipod species observed in dredging sites whereas *Corophium triaenonyx* (9.2%), *Caprella* sp (2.2%), and *Photis digitata* (0.6%), were noticed from non-dredging sites. During MN season, no amphipods were observed in dredging sites while *Caprella* sp (13.3%), *Photis digitata* (4.4%), *Eriopisa chilkensis* (2.2%), *Corophium triaenonyx* (1.1%), and *Cheriophotis megacheles* (1.1%) were observed in non-dredging sites. During PM season *Corophium triaenonyx* (0.5%), and *Melita zylanica* (0.23%), were observed in dredging sites while *Corophium triaenonyx* (10.4%), *Photis digitata* (8.3%), *Caprella* sp

(4.9%), *Eriopisa chilkensis* (2.7%), and *Cheriophotis megacheles* (2.1%) were observed during non-dredging sites.

Other macrobenthic fauna such as Oligochatea, Tubificidae sp (39.98%), and Tanaidacea species *Apseudus chilkensis* (12.74%), were observed in dredging sites while Tubificidae sp (2.3%), Chironomid sp (2.1%), mollusks such as *Phalium* sp (1.54%), *Littorea littorina* (1.34%), *Modiolus* sp (1.23%), and *Dentalium* sp (1.1%), *Apseudus chilkensis* (0.94%), and *Anthurid sp* (0.84%),were noticed from non-dredging sites (Table 5.7).

During 2011 among the identified amphipods (1.5%), species such as *Ampelisca* sp (2.47%), *Gammaropsis* sp (0.19%), *Melita zylanica* (0.15%) and *Eriopisa chilkensis* (0.14%) were observed during dredging sites species such as *Caprella* sp (10.6%), *Photis digitata* (8.21%), *Cheriophotis megacheles* (3.50%), and *Gammaropsis* sp (1.07%) was observed in the non-dredging sites. During PRM, *Eriopisa chilkensis* (0.4%) was the amphipod species observed in dredging sites whereas *Caprella* sp (18.7%), *Photis digitata* (5.2%), *Eriopisa chilkensis* (1.6%), and *Cheriophotis megacheles* (0.4%) were observed in non-dredging sites. During MN season, *Gammaropsis* sp (0.6%) was observed in dredging sites while *Caprella* sp (11.9%), *Gammaropsis* sp (3.2%), *Photis digitata* (1.8%), *Cheriophotis megacheles* (1.4%) and *Melita zylanica* (0.5%), were observed in non-dredging sites. During PM season *Ampelisca* sp (7.4%), and *Melita zylanica* (0.5%), were observed in dredging sites while *Photis digitata* (17.7%), *Cheriophotis megacheles* (8.7%), *Gammarus* sp (2.2%), *Leucothoe* sp (1.7%), *Caprella* sp (1.3%), *Melita zylanica* (1.3%), and *Eriopisa chilkensis* (0.9%), were observed during non-dredging sites.

Other macrobenthic fauna such as Oligochatea, Tubificidae sp (63.3%), Tanaidacea species *Apseudus chilkensis* (3.78%), and *Cirolana fluviatilis* (0.09%) were observed in dredging sites while Tubificidae sp (2.7%), *Apseudus chilkensis* (2. 4%), *Littorea littorina* (0.74%), Chironomid sp (0.2%), and *Anthurid* sp (0.13%), were noticed from non-dredging sites (Table 5.8).

5.4.3.6 Macrobenthic species diversity indices

5.4.3.6.1 Number of species (S)

During 2009, species number (S) varied from 16-22 in non-dredging sites and 4-17 in dredging sites. The average number of species was 18 in non-dredging sites and 12 in dredging sites during PRM. Mean species number was higher (20) in non-dredging sites and lower in dredging sites (10) during MN season. During PM species number was 17 in non-dredging sites and 11 in dredging sites (Fig. 5.23).

In 2010, number (average) of species ranged from 9-18 in non-dredging sites and 4-19 in dredging sites. In non-dredging sites mean number of species was 17 during PRM, 12 during MN, and 10, during PM season. While in dredging sites, average number of species was 11 during PRM, 6 during MN, and 8 during PM season. Number of species was higher in non-dredging sites relative to dredging sites (Fig. 5.23).

During the period 2011, species number was higher in non-dredging sites and it ranged between 17-36 in non-dredging sites and 12-23 in dredging sites. In non-dredging sites it was 29 during PRM, 22 during

MN and 27 during PM season. In dredging sites it was 18 observed during PRM, and MN, and 19 during PM season (Fig. 5.23).

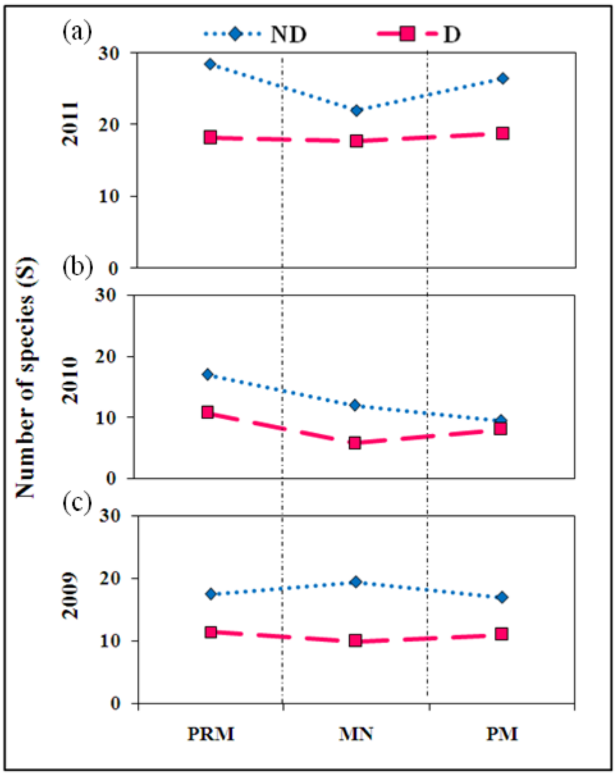


Figure 5.23 Variation in the number of species in dredging and non-dredging sites of Cochin estuary

5.4.3.6.2 Shannon diversity index (H' log2)

In the year 2009, species diversity was higher in non-dredging sites compared to dredging sites. It varied from 2.3 to 3.7 in non-dredging sites and from 1.4 to 3.3 in dredging sites (Fig. 5.30). During PRM, the diversity index in non-dredging sites was >3 (av.3.2±0.2) compared to dredging sites (av.2.9±0.4). During MN diversity was slightly higher than 2.5 in non-dredging (av.3.5±0.4) and dredging sites (av.2.6±0.5). Lower diversity was observed during PM season in dredging (av.2.4±0.9) and non-dredging sites (av.2.5±0.3) (Fig. 5.24a).

During 2010, lower diversity was observed in dredging sites (0.91 to 3.04) compared to non-dredging sites (2.2 to 3.3). Mean species diversity in non-dredging sites was higher during MN (av.2.9±) and lower during PRM period (av.2.6±0.6). In dredging sites high diversity was observed during PRM (av.2.5±0.6) season and low during MN (av.1.6±0.5) (Fig. 5.24b).

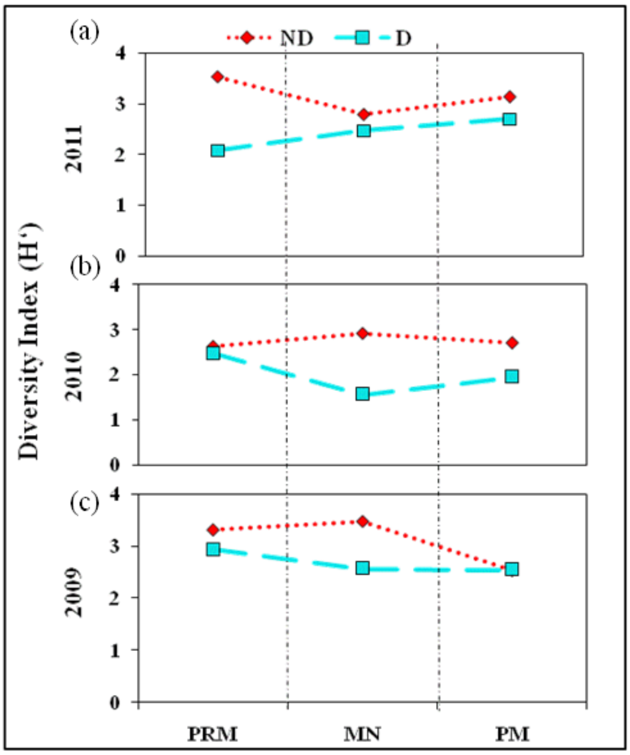


Figure 5.24 Variation in the diversity index in dredging and non-dredging sites of Cochin estuary

In 2011, species diversity varied from 0.9 to 3.2 in dredging sites and 2.5 to 3.8 in non-dredging sites. In non-dredging sites mean diversity was high during PRM (av.3.5±0.3) and low during MN (av.2.8±0.4), while in dredging sites species diversity was high during PM (av.2.7±0.8) and low during PRM (av.2.1±0.5) respectively (Fig. 5.24c).

5.4.3.6.3 Species Richness (d)

During 2009, similar to diversity, species richness (d) was also lower in dredging sites (av.1.5±0.2) than non-dredging sites (av.2.1±0.2). In non-dredging sites the mean species richness was higher during MN (av.2.4±0.4) and lower during PM (av.1.9±0.1). In dredging sites, the average species richness was high during PRM (av.1.54±0.6), and low during MN (av.1.35±0.6) and PM (av.1.39±0.6) respectively (Fig. 5.25a).

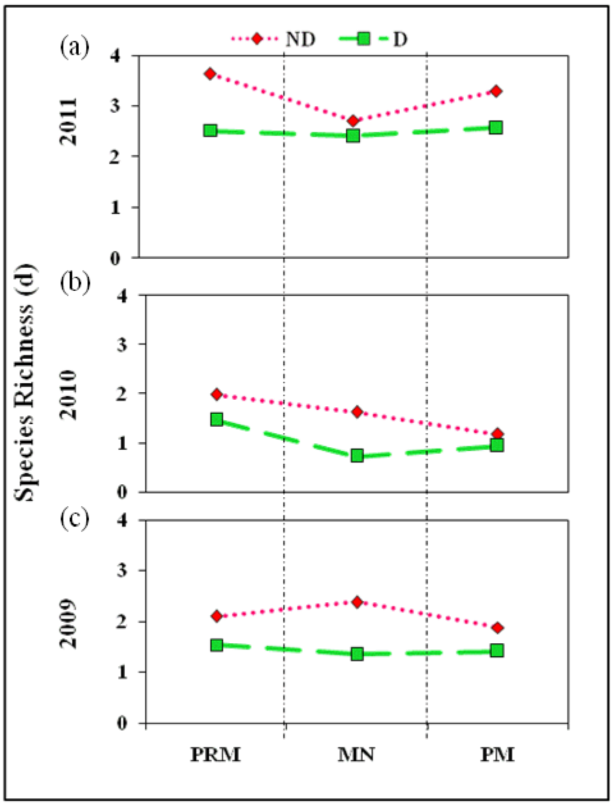


Figure 5.25 Variation in the species richness in dredging and non-dredging sites of Cochin estuary

In 2010, species richness was lower in dredging sites (av.1.0±0.5) compared to non-dredging sites (av.1.6±0.4). High values for species richness was observed during PRM (av.2.0±0.3) and lower during PM (av.1.2±0.03) in non-dredging sites. In dredging sites, average species richness was higher during PRM (av.1.5±0.7) and lower during MN (av.0.7±0.2) (Fig. 5.25b).

The species richness during 2011, ranged from 2.2 to 4.4 in non-dredging sites, and 1.9 to 3.0 in dredging sites with higher index in non-dredging sites. In non-dredging sites species richness was higher during PRM (av. 3.6±0.5) and lower during MN (av.2.7±0.8), while in dredging sites higher richness was observed during PM (av.2.6±0.3) and lower during MN (av.2.4±0.6) (Fig. 5.25c).

5.4.3.6.4 Pielou's Evenness index

During 2009, evenness index ranged from 0.4-0.9 in dredging sites and 0.6 to 0.8 in non-dredging sites. Slightly high values were observed in dredging sites compared to non-dredging sites. In dredging sites, high evenness was observed during PRM (av.0.86±0.05) and low during PM (av.0.76±0.28). In non-dredging sites variation in evenness was less between seasons (Fig. 5.26a).

During 2010, evenness index was higher in non-dredging sites (0.6 to 1.0) compared to dredging sites (0.3 to 1.0) (Fig. 5.26b). In non-dredging sites, higher index as observed during PM (av.0.83) and lower during PRM (av.0.64±0.12). In dredging sites index was higher during PRM (av.0.76±0.13) and lower during PM season (av.0.67±0.17).

During 2011 period, Higher index was observed in non-dredging sites (av.0.68±0.054) compared to dredging sites (av.0.58±0.21). In non-dredging sites high index was observed during PRM (av.0.73±0.04) and low index was observed during MN (av.0.63±0.03). In dredging sites, higher index was observed during PM (av.0.65±0.23) and lower during PRM (av.0.51±0.15) (Fig. 5.26c).

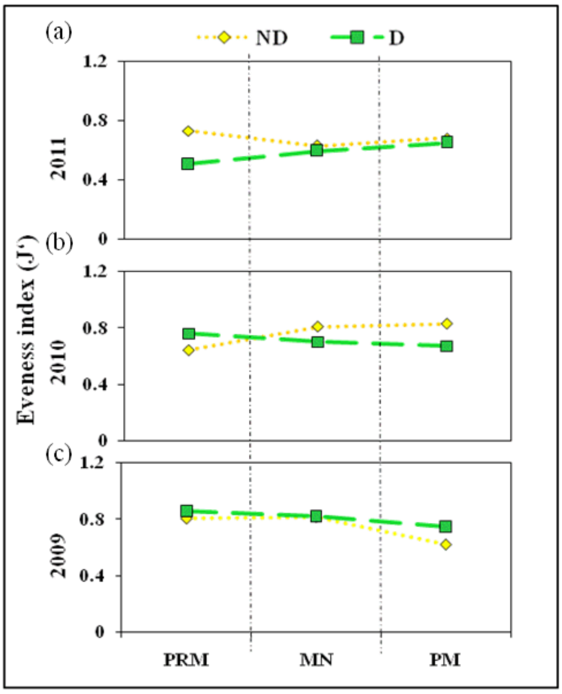


Figure 5.26 Variation in the evenness index in dredging and non-dredging sites of Cochin estuary

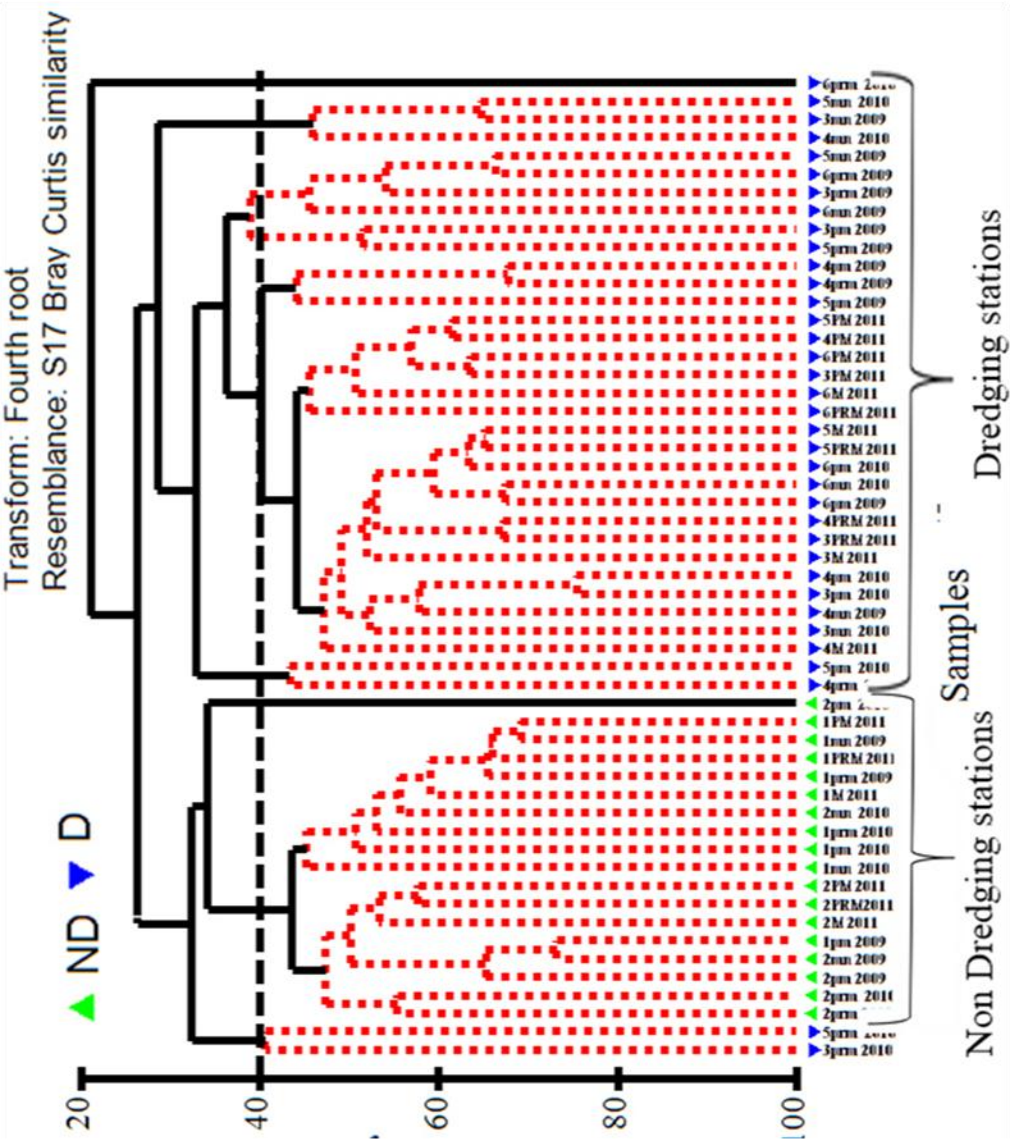
5.4.3.7 Macrobenthic community structure

In the present study, 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. The similarity matrix formed the basis for cluster analysis (hierarchical agglomerative method using group-average linking), similarity profile (SIMPROF) permutation test, analysis of similarity (ANOSIM) and non-metric multidimensional scaling (NMDS).

Multivariate analysis based on species density data, categorized the stations into nine groups at 40% similarity level (Fig. 5.27). Of these, two were formed by non-dredging sites whereas the other seven clusters were constituted by dredging sites. Thus the results of the cluster analysis distinguished by the cut off levels of the dendrogram, revealed distinct grouping of dredging and non-dredging sites. Hence two major groups were identified from the analysis, group I- dredging sites, group II – non-dredging sites.

The result of the NMDS also showed, similar pattern of distinct groupings of dredging and non-dredging sites (Fig. 5.28) with a stress value of 0.21. Significant disparity in density between the dredging and non-dredging sites were evident from the results of ANOSIM analysis (Global R- 0.588, $p<0.01$).

The SIMPER analysis was carried out considering the stations, as two major groups to identify the species, that characterized each region (characterizing species), and the major species that differentiate (discriminating species) between the dredging sites and non-dredging sites. Major characterizing species of the group-I dredging sites (av.similarity- 37.64%) were *Tubificidae* sp (17.46%), *Mediomastus capensis* (14.43%), *Sigambra parva* (13.52%), *Nephtys oligobranchia (11.54%),Prionospio cirrifera (10%), and Apseudus chilkensis (8.09%).* The major characterizing species identified in Group-II -non-dredging sites (av.similarity- 46.54%) were *Mediomastus capensis* (13.53%), *Prionospio cirrifera (9.83%), Paraheteromastus tenuis* (9.52%), *Caprella* sp (9.19%), *Dendronereis estuarina* (7.77%), and *Littorina littorea* (5.25%) (Table 5.4).



Similarity

Figure 5.27 Bray Curtis similarity based on hierarchical clustering of stations manifested through dendrogram showing macrobenthic assemblage pattern in the dredging and non-dredging areas of the Cochin estuary.

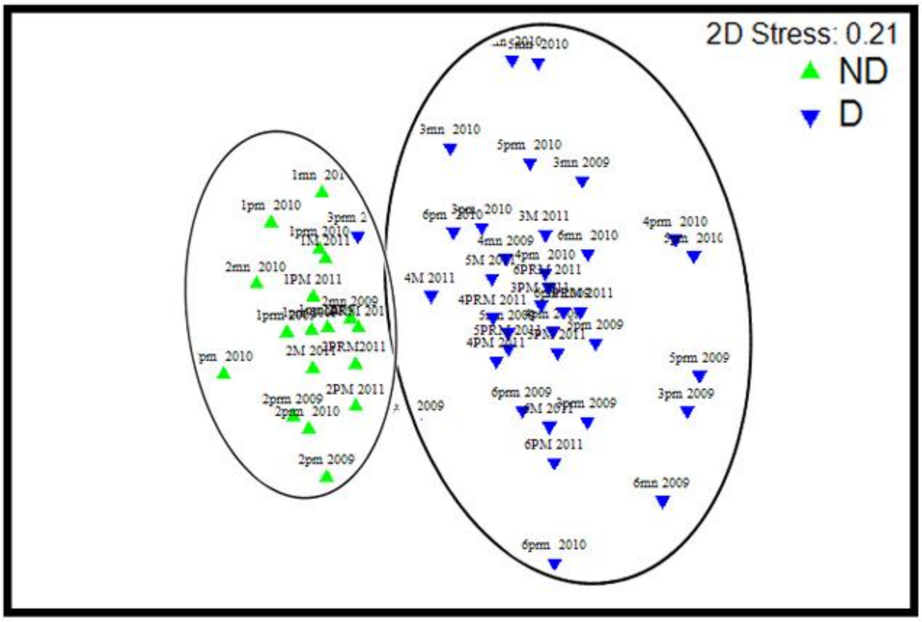


Figure 5.28 Bray-Curtis similarity based on hierarchical clustering of stations manifested through NMDS (non-metric multidimensional scaling) showing macrobenthic assemblage pattern in dredging and non-dredging areas of the Cochin estuary.

The major discriminating species identified through SIMPER tool between dredging and non-dredging sites have an average dissimilarity of 74.01%. The major discriminating species such as *Caprella* sp*, photis digitata, Littorina littorea, Eriopisa chilkensis,* and *Cheriophotis megacheles,* were mostly occurred in non-dredging sites while species such as *Tubificidae* sp*, Nephtys oligobranchia, Sigambra parva, Cirratulus filiformis, Apseudus chilkensis* and *Cirratulus cirratus* were observed abundantly from dredging sites. The variability of the spatial density of the discriminating species in dredging and non-dredging sites, overlaid on the NMDS plot also indicated their disparity between the two sites (Fig. 5.29).

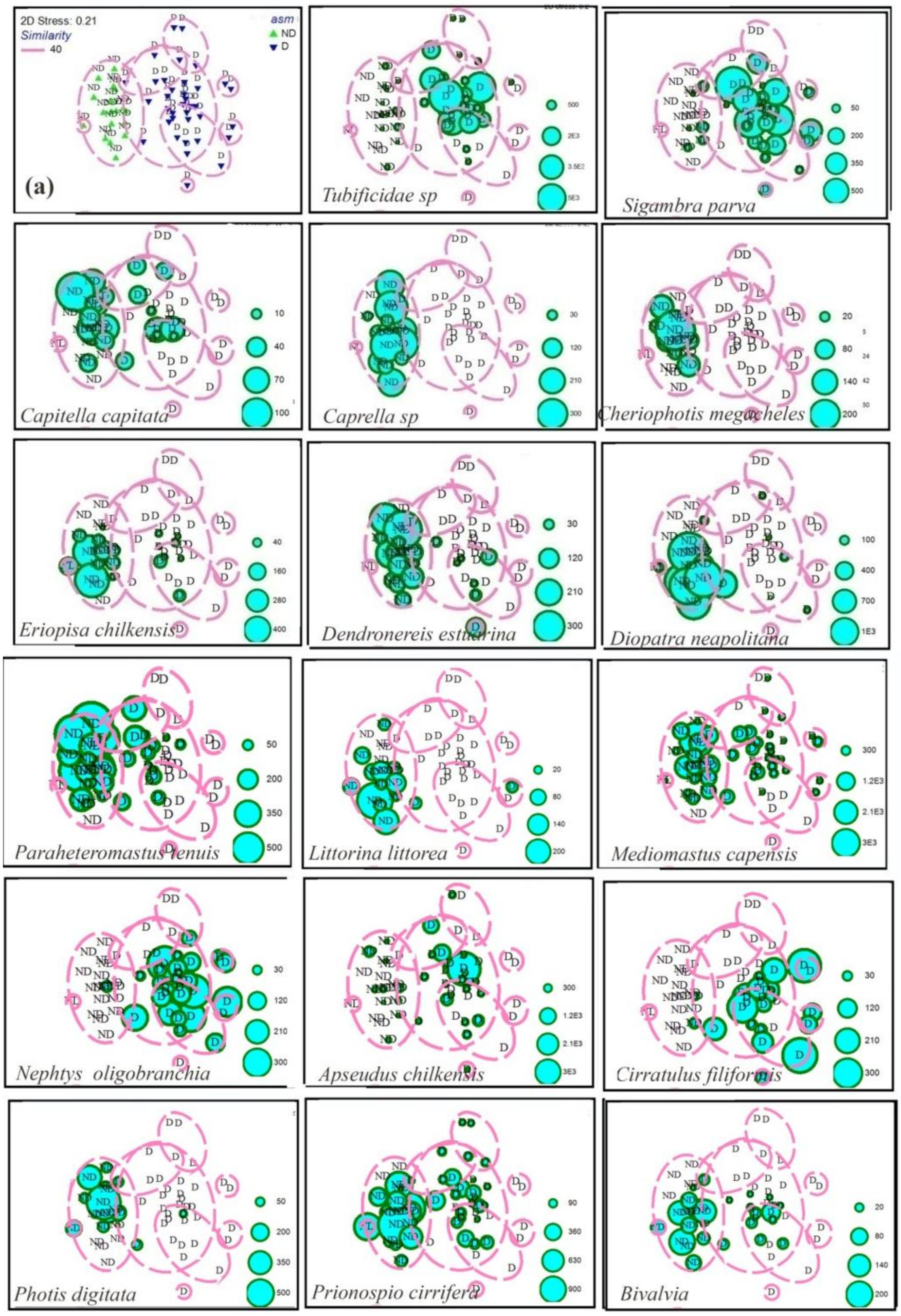


Figure 5.29 (a) NMDS- plot showing the distribution of macrobenthos, and the other bubble plot showing the distribution of major discriminating species overlaid on NMDS between dredging and non-dredging sites.

Table 5.4 *Major characterizing species identified through SIMPER that contribute to the average similarity within each assemblage*

Assemblages	Species	Av. Abundance	Av. Similarity	% Contribution
Sim: 46.54%	*Mediomastus capensis*	4.04	6.3	13.53
	Prionospio cirrifera	3.24	4.57	9.83
Group I-ND	*Paraheteromastus tenuis*	2.96	4.43	9.52
	Caprella sp	2.92	4.27	9.19
	Dendronereis estuarina	2.51	3.62	7.77
	Littorina littorea	1.91	2.44	5.25
	Photis digitata	1.91	2.26	4.85
	Tubificidae sp	1.72	1.77	3.81
	Bivalvia sp	1.67	1.71	3.68
	Capitella capitata	1.54	1.66	3.57
	Eriopisa chilkensis	1.67	1.53	3.29
	Apseudus chilkensis	1.55	1.5	3.21
	Diopatra neopolitana	2.1	1.37	2.95
	Cheriophotis megacheles	1.25	1	2.14
	Coropium triaenonyx	1.25	0.81	1.75
	Owenia fusiformis	1.11	0.69	1.49
Sim: 37.64%	*Tubificidae sp*	3.47	6.57	17.46
Group II- D	*Mediomastus capensis*	2.46	5.43	14.43
	Sygambra parva	2.39	5.09	13.52
	Nepthys oligobranchia	2.09	4.34	11.54
	Prionospio cirrifera	1.86	3.77	10
	Apseudus chilkensis	1.93	3.04	8.09
	Cirratulus filiformis	1.56	2.29	6.08

Paraheteromastus tenuis	1	0.87	2.3
Cossura coasta	0.9	0.75	2
Cirratulus cirratus	0.81	0.74	1.98
Decapod sp	0.67	0.65	1.74
Bivalvia sp	0.69	0.55	1.47

Table 5.5 *Discriminating species with mean abundances of species that contribute to the maximum dissimilarity between the assemblages*

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

5.4.4 Feeding guild Composition during 2009-2011

Assessment of the feeding guilds has been carried out by identifying the feeding modes of the dominant group (polychaetes) and also by identifying the feeding modes of the characterizing species of the respective region. As polychaetes formed the major taxa in the entire study area, by analyzing their feeding mode will give an idea on the dominant feeding modes they prefer to cope with the environment. In order to represent the entire taxa into the feeing guild composition analysis, feeding guilds composition of the identified characterizing species of each site (using SIMPER tool in PRIMER 6) was also analyzed.

The results of the feeding guild analysis of polychaetes revealed that, carnivores (av.37.8%) dominated in dredging sites, while sub-surface deposit feeders (SSDF-av.43.3%) dominated in the non-dredging sites. Seasonal variation in feeding guild composition was less in dredging sites where carnivorous polychaetes (PRM-39.3%, MN-31.3%, PM-42.7%) were replaced by SDF (36%) during MN season. While in non-dredging stations seasonal variation was obvious where SSDF (54.6%), dominated during PRM, replaced by SDF (50.7%) during MN, and carnivores (38.7%) dominated during PM (Fig. 5.30). During PRM, carnivores dominated (39.3%) in dredging sites, followed by sub-surface deposit feeders (34.5%), surface deposit feeders (25.5%), filter feeders (0.4%) and herbivores (0.3%). In non-dredging sites, dominance by the sub-surface deposit feeders (54.6%) were encountered followed by carnivores (23.3%), surface deposit feeders (21.5%), filter feeders (0.5%) and herbivores (0.1%) during the season (Fig. 5.30a). During MN replacement of the carnivores by surface deposit feeders (36.0%) was observed in the dredging sites, followed by carnivores (31.3%), sub-surface deposit feeders (29.7%), and (3.1%) filter feeders (Fig. 5.30b).

In non-dredging sites, higher percentage of surface deposit feeders (50.7%), and sub-surface deposit feeders (43.5%), were observed with low number of carnivores (5.8%). During PM, carnivores (42.7%) again dominated the region along with sub-surface deposit feeders (30.7%). Carnivores increased in non-dredging sites during PM season, followed by SSDF (31.8%), SDF (28.2%), and filter feeders (1.3%) (Fig. 5.30c). In both dredging and non-dredging sites, three prominent feeding guilds of polychaetes were observed throughout the study period.

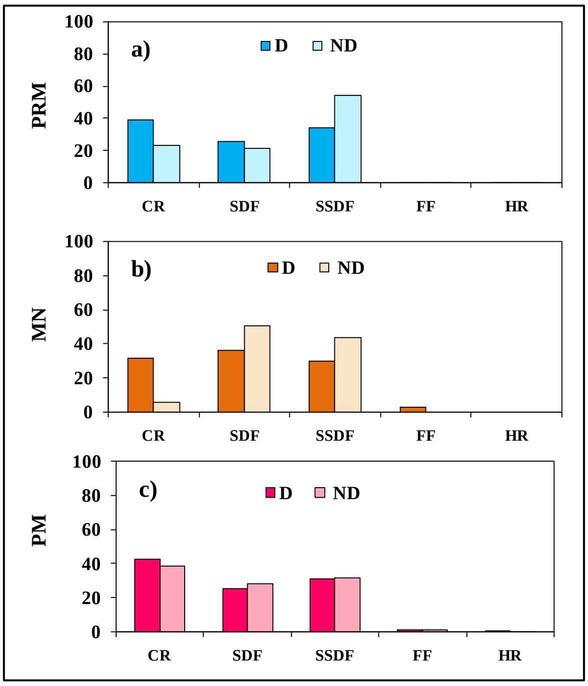


Figure 5.30 Variation in the macrobenthic polychaete feeding guild composition in dredging and non-dredging sites of Cochin estuary during pre-monsoon (a), monsoon (a), and post-monsoon (c).

The analysis of the feeding guild of the major characterizing species resulted in the identification of four types of feeding modes in dredging sites and seven types in non-dredging sites. SDF and carnivores (67-87% together) were the major feeding modes exhibited by macrobenthic fauna in dredging sites, whereas SDF and SSDF (62-91% together) constituted the major feeding guild in non-dredging sites (Fig. 5.31). In dredging sites surface deposit feeders (34.0%) and carnivores (33.0%) constituted major feeding guilds, followed by sub-surface deposit feeders (29.0%), and filter feeders (4%) during PRM (Fig. 5.31a). During MN, higher percentage of surface deposit feeders (68.0%) with minor abundance of carnivores (19.0%), sub-surface deposit feeders (12.0%), and filter feeders (1%) were observed at the site (Fig. 5.31b). During PM an increase in abundance of carnivores was observed and together with surface deposit feeders it contributed 83% to the total (Fig. 5.31c).

In non-dredging sites, sub-surface deposit feeding (45.0%) was the exhibited dominant feeding mode followed by surface deposit feeding (24.0%), carnivory (15.0%), filter feeding (8%), filter/surface deposit feeding (5.0%), herbivory (3.0%) and omnivory (1.0%). Surface deposit feeders (52.0%) replaced sub-surface deposit feeders (39.0%) and carnivores (1.0%) were immensely reduced in number during MN. Number of carnivores (28.0%) was increased during PM and surface deposit feeders (52.0%) and sub-surface deposit feeders contributed 64.0% together to total feeding guild.

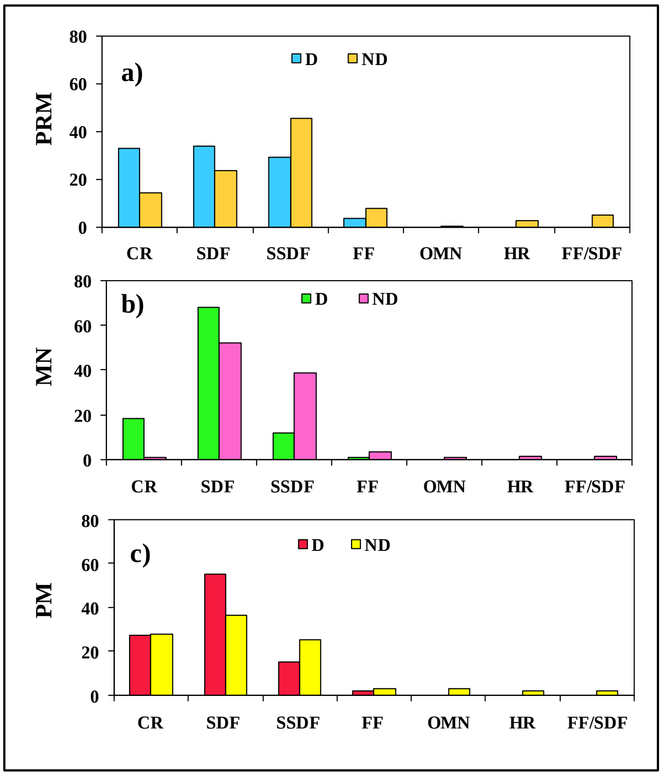


Figure 5.31 Variation in the macrobenthic feeding guild composition in dredging and non-dredging sites of Cochin estuary during (a) pre-monsoon, (b) monsoon, and (c) post-monsoon season.

5.4.5 BO2A Index 2009-2011

Benthic Opportunistic Annelid Amphipods Index (BO2A Index) varied from 0.08 to 0.29 in dredging sites and from 0.01 to 0.26 in non-dredging sites (Fig. 5.32). Mean BO2A index values were higher in dredging sites (av.0.20±0.04) compared to non-dredging sites (av.14±0.04). The index values ranged from 0.43 to 0.26 during PRM, from 0.11 to 0.22 during MN, and from 0.01 to 0.15 during PM in non-dredging sites. While in dredging sites index varied from 0.08 to 0.27 during PRM, from 0.11 to 0.27 during MN, and from 0.12 to 0.29 during PM. Mean index value was observed higher during the MN (av.0.17±0.03) in non-dredging sites and during PM in dredging sites (av.0.21±0.04). During MN and PM seasons the index values were higher than 0.2 in the dredging sites whereas in non-dredging sites the average index values were always below 0.18 during the study period. At the dredging sites (stations-3&5) index values even reached higher than 0.24 which indicating towards poor environmental condition of the index.

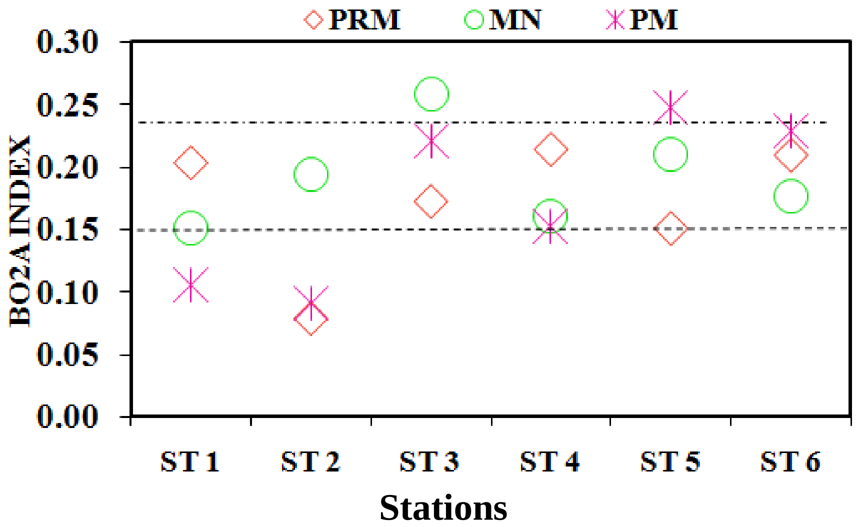


Figure 5.32 Variation in the BO2A index in the dredging and non-dredging sites of Cochin estuary (Stations 1 and 2-Non-dredging sites, Stations 3-6- Dredging sites).

5.5 DISCUSSION

Estuary, the dynamic ecosystem with highly variable physico-chemical environments supports high primary production and complex food chain. Though they are valuable aquatic ecosystem providing several ecological services to the human, they are one of the most over exploited natural habitats and used as major repositories of several anthropogenic activities. 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 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 hydrographical features of the CE are regulated by the incursion of the seawater and the inflow of the fresh water (Menon et al., 2000). In the present study, prominent seasonal variation was observed in the distribution of bottom temperature. Comparatively higher temperature recorded during pre-monsoon might have happened by the prevailing warmer weather and the maximum solar radiation (Qasim et al., 1968). The monsoonal rainfall and runoff (Sankaranarayanan and Qasim, 1968, Madhu et al., 2007) might have resulted in the decreased temperature during MN. No significant spatial variation was observed in temperature between the dredging and non-dredging sites, as dredging activity cause no remarkable variation in the temperature.

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., 2010; Qasim, 2003). During monsoon due to the heavy rainfall and associated runoff the entire estuary exhibited lower salinity values

(Vineetha et al., 2015). Large volume of freshwater (22×10^3 Mm^3/year) enters in to the estuary during the period and complete freshening of the estuary occurs during monsoon in the estuary (Revichandran et al., 2012). During non-monsoon periods, incursion of the seawater dominates over the markedly less freshwater influx resulted in an increase in the salinity. Higher salinity observed in the dredging sites was found to be associated with the intrusion of seawater through the dredged channels. As the observed variation in salinity in the CE is mainly regulated by the tidal and freshwater interaction, hence with respect to dredging activities no remarkable change could be observed in salinity between the study sites. Balchand and Rasheed (2000) reporting similar observations in the CE supported this finding. Channel dredging activities in the Amba estuary increased tidal inflow into the estuary leading to incursion of high saline waters even to the upstream (Velamala et al., 2016). Similar results were observed from Thampa bay by Zhu et al. (2015). In the present study a decrease in salinity was observed during 2009 to 2011, which can be attributed to the increased rainfall of 2010-2011 compared to the 2009-2010 period.

The pH of the underlying water is often influenced by a combined interaction of many physical and biological factors (Hossain and Marshall, 2014), such as bacterial activity, water turbulence, chemical constituents of water, and sewage overflows. Slightly high pH observed in dredging sites compared to non-dredging sites might have resulted by the influence of seawater. Observed decrease in pH during monsoon was due to the increased freshwater influx into the region. Previous studies from the CE and other Indian estuaries have also reported on low concentration of pH during monsoon (Nisha, 2008; Sarma et al., 2009, 2011). Spatial variation in pH was statistically insignificant between dredging and non-dredging sites during the study period.

Prior studies conducted on dredging impacts have reported on the slight reduction in dissolved oxygen concentration in dredged areas (Brown and Clark, 1968; Johnston, 1981). After dredging, low DO distribution can happen either due to the mixing of the reduced products like methane and hydrogen sulphide or due to the consumption of oxygen by the micororganisms attached to the resuspended particulate material.. Labile organic compounds released from the sediments also causes changes in the oxygen content (Riemann and Hoffmann, 1991). In the present study, though slight reduction in the oxygen levels was observed in dredging sites during most of the seasons, the observed change was statistically insignificant between dredging and non-dredging sites. Reduction in the DO levels can also be due to result of the increased turbidity raised by the suspended particles (up to 16-83%), which in turn increase the biological oxygen demand at the site (Johnston, 1981). In the present study, the average oxygen concentration was higher than 3mg/l throughout the sampling locations, indicating an oxygenated environment. High density of phytoplankton in the study area might contributed to the increased oxygen concentration irrespective of the season (Madhu et al., 2007). As the tidal exchange facilitate to replenish oxygen levels in estuaries, reduced oxygen levels caused by dredging activities will have cause only temporary effects on the environment.

Earlier studies also have documented on the increased biochemical oxygen demand associated with the higher turbidity and low oxygen levels (Brown and Clark, 1968; Johnston, 1981). Biological oxygen demand would be higher in regions where higher organic load has been discharged. In the present study no remarkable variation in BOD could be observed between dredging and non-dredging sites.

Sediment resuspension brought about by the dredging activities causes rapid release of nutrients into the overlying water column (Jones and Lee, 1981; Klump and Martens, 1981; Lohrer and Wetz, 2003). Fine grained sediments such as silt and mud would remain in suspension for longer time periods and can also retain greater concentrations of soluble regenerated nutrients (Lohrer and Watz, 2003). Changes occurring in the nutrient levels can led to a variety of impacts, such as overgrowth of seaweed and epiphytes, anoxia and hypoxia events, and nuisance and toxic algal blooms (Devlin et al., 2011; Pinon-Gimate et al., 2009; Teichberg et al., 2009). Preceding studies in the CE have reported on the increased nutrient levels arising from prolonged discharges of industrial and domestic sewage (Madhu et. al, 2007; Balachandran et al., 2005). The distribution of nutrients showed no statistically significant variation between the dredging and non-dredging sites. The changes brought about by the dredging activity on nutrient distribution are of short span as indicated by earlier studies conducted in CE (Joseph et al., 1998; Balchand and Rasheed, 2000).

In the present study, the churning up of bottom substratum as a consequence of the dredging activities, bringing up finer sediment particles into the water column might have contributed towards the higher SPM concentration in dredging sites. Studies depicting the effect of dredging on the increased turbidity in the water column of tropical (Balchand and Rasheed, 2000; Johnston Jr, 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. The dredging activities associated with resuspension of the bottom sediments as well as leaks and spills (Gutperlet et al., 2015) would increase the turbidity of the water column. Dredging activities cause an increase in turbidity levels higher than the natural turbidity levels in an aquatic system, which will negatively impact the fishes and other filter feeding organisms (Johnston, 1981). Several incidence of negative impacts of suspended particles associated with dredging such as gill clogging, impairment of the respiratory, excretory functioning and feeding activities of filter feeding organisms (Sherk, 1971), mortality of pelagic as well as settled larvae (Rosenberg, 1971) and decrease in the coral abundance, growth and species diversity (Dodge and Vaisnys, 1977) have been reported by many studies.

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 dredging sites compared to the coarser particles in 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. More or less, a homogenous sandy substratum was observed in non-dredging sites. The higher sand content observed in the substrata of non-dredging sites might 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 impact of marine dredging along the French coast of English Channel also corroborate the present findings as the non-dredged areas were 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 non-dredging sites further authenticates the observation in the present study (Table 5.2).

The substrata of dredging sites were characterized with relatively higher organic carbon levels thus leading to a significant variation ($p<0.05$) between dredging and non-dredging sites. 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). Fine grained fractions of sediment always have greater surface area, and thereby have high retention capacity to entrap the organic matter (Flemming et al., 1996; Nayar et al., 2007;Venkatramanan et al., 2013) compared to the other fractions. Hence, finer sediments observed in dredging sites retained higher organic carbon content compared to the coarser sediments in non-dredging sites. In addition, organisms getting fragmented by dredging (Newell et al., 1999) and the inputs through sewage from land bordering the channels, and fish landing centers might also have an immense contribution towards the enhancement of the organic carbon levels in the sediments of the dredging sites (Hossain et al., 2014; Robin et al., 2012). In the dredging sites, elevated levels of organic matter have been reported by earlier studies also (Lohrer and Wetz, 2003; Zimmerman et al., 2003). During dredging activities the organic rich underlying sediments getting exposed with the finer resuspended sediments contributed to the higher organic matter at the region. Disturbances in the physical chemical characteristics of the sediment (especially granulometric composition and SOM) in turn might cause changes in the benthic community composition (Lopez-Jamar and Mejuto, 1988; van Dalfsen et al., 2000).

Dredging activity involves the mechanical removal of sediments, which ultimately affects the bottom fauna by the alterations caused in their habitat. In comparison to non-dredging sites, the remarkable reduction in the faunal density and biomass observed in dredging sites further affirms the impact of dredging activities on the benthic biota. 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 the three consecutive years, marked variability was observed in the benthic community composition between dredging and non-dredging sites. In dredging sites, a conspicuous dominance of opportunistic tubificid oligochaetes was evident, whereas in non-dredging sites polychaetes predominated. The lower density of molluscs and amphipods were also conspicuous in 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 chilkensis* in non-dredging sites.

In the present study, noticeable higher abundance of tubificid oligochaetes in dredging sites throughout the study period indicating their adaptability in the disturbed environments. Being an opportunistic organism, they are known to rapidly proliferate within a short period of time upon favorable environmental conditions (Giere and Pfannkuche, 1982). The presence of organic rich sediment and also the less competition from other macrobenthic fauna might have favored the higher abundance of these opportunistic organisms in the dredging sites. Moreover, their higher tolerance to harsh environmental conditions such as hypoxia and nutrient enrichment further support their higher abundance when compared to the other benthic organisms (Caspers, 1973). Hence, their unique adaptability to the environmental conditions incident in the dredging site might have favored their higher preponderances. However, the observed biomass in dredging sites was not in accordance with the benthic abundance pattern. The inconsistency evident between the biomass and abundance value of the dredging locations can be attributed to the proliferation of the small sized opportunistic organisms. Similar observation in the Detroit River, USA (Besser et al., 1996) 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 shelf (Abdul Jaleel et al., 2014; Abdul Jaleel et al., 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 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 assemblages of benthic organisms 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 prevailing in dredging sites of the CE signifies the negative impact produced by the maintenance dredging activities. Among the discriminating species identified using SIMPER, Caprellid amphipod

Caprella sp, Photid amphipods such as *Photis digitata* and *Cheriophotis megacheles*, gammarid amphipod *Eriopisa chilkensis,* gastropod species *Littorina littorea*, polychaete *Dendronereis estuarina* and Bivalvia sp were in low in number compared to non-dredging sites whereas the higher density of the oligochaete *Tubificidae sp,* polychaetes like *Sigambra parva, Nephtys oligobranchia, Cirratulus filiformis, Cirratulus cirratus* andtanaid species *Apseudus chilkensis* in the dredging sites further affirms their tolerance to the disturbed conditions. The NMDS bubble plots overlaid with the density of the discriminating species also supports the differences in the taxa assemblages between dredging and non-dredging locations.

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 dominant polychaetes and that of the characterizing species (identified through SIMPER) revealed a clear domination of carnivores in dredging sites. As mobility helps organisms to efficiently adapt in a continuously disturbed environment, the relatively higher motility of the carnivorous polychaetes might have favored the higher density of this group in dredging sites throughout the study period. Carnivores may also take advantage 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 and sub-surface deposit feeders, carnivores, herbivores, omnivores, and filter feeders in turn indicates the availability of diverse food sources, resource partitioning and subsequently diverse trophic pathways (Ulanowicz, 1997). The BO2A index, depicting the habitat quality of a particular environment also suggests the prevalence of contrasting environmental conditions in dredging and non-dredging sites. Higher indices observed in 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 dredging sites. The slightly higher BO2A indices evident in non-dredging sites (avg. 0.18) during the monsoon may be due to the disturbances imparted by the monsoonal rainfall and associated run off.

Earlier studies on the benthic infaunal recovery suggested 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, 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). Recovery of disturbed community mostly depends upon the capability of an adjacent undisturbed community to provide migrating adults or larvae for the recruitment process (Zajac and Whitlatch, 1982; Diaz, 1994; Bolam and Rees, 2003). Recolonization of the fauna by larval settlement depends on the factors such as the sediment conditions (Crisp, 1965), time, and suitable season. So the larval recovery process will be slower compared to the recolonization by the adults (Van der Veer et al., 1985). The communities inhabiting the fine mud recover more rapidly (1-18 months) than sand, gravel and corals reefs because those communities are 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 the more stable habitats. In the present study, the observed dominance of opportunistic species from the dredging sites was an indication of recolonization by the r-selected benthic community in the system impacted by continuous dredging activities. The CE, a shallow dynamic estuary, where continuous dredging has been carried out in the channels which are intermittently influenced by tidal activities, the recolonization of the opportunistic communities will be comparatively fast as the substratum is fine mud. Fast recolonization 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.

Table 5.6 Macrobenthic density in the sampling sites during 2009 (- <100, **-100-500, ***- 500-1000, ****- 1000-3000)*

	2009					
	ND PRM	ND MN	ND PM	D PRM	D MN	D PM
Polychaeta						
Amphinomid sp	-	-	-	-	-	-
Amphinomea rostrata	-	-	*	-	-	-
Etone sp	-	-	*	-	*	-
Phylldoce sp	*	-	-	-	-	-
Harmothoe sp	-	-	-	*	-	-
Hesionidae sp	-	-	-	-	-	-
Sigambra parva	-	-	-	**	*	**
Glycera longipinnis	*	*	-	*	-	*
Goniada emerita	-	-	-	-	-	-
Nephtys dibranchia	-	-	-	-	-	-
Nephtys oligobranchia	-	*	-	**	**	**
Nephtys polybranchiata	-	-	-	-	*	-
Dendronereis estuarina	*	**	*	-	-	*
Nereis sp	*	*	*	-	-	*
Lycastis indica	-	*	*	-	-	-
Cirratulus cirratus	-	-	-	*	*	*
Cirratulus filiformis	-	-	-	*	**	*
Dodecaceria sp	-	-	-	-	-	-
Paraprionospio pinnata	*	-	-	-	-	*
Prionospio cirrifera	**	**	**	*	*	*
Prionospio cirrobranchiata	*	*	-	-	-	-
Polydora ciliata	-	-	-	-	-	-
Prionospio sp	-	-	-	-	-	-
Spionid sp	-	-	-	-	-	-
Caulleriella capensis	-	-	-	-	-	-

Scolelepis squamata	-	-	*	-	-	*
Boccardia polybranchia	-	-	-	-	-	-
Mediomastus capensis	***	**	**	*	*	*
Capitella capitata	-	*	*	*	*	*
Paraheteromastus tenuis	**	**	*	*	*	*
Capitellid sp	-	-	-	-	-	-
Lumbriconereis impatiens	-	-	-	-	-	*
Lumbriconereis latreilli	*	-	-	*	-	*
Diopatra neopolitana	**	*	***	**	*	*
Eunice sp	-	-	-	*	-	*
Cossura coasta	-	-	-	*	-	*
Aricidea longobranchiata	-	-	-	-	-	-
Owenia fusiformis	*	*	*	*	*	-
Serpulidae sp	-	-	-	-	-	-
Pectinaria crassa	-	-	-	*	-	*
Pisione sp	-	-	-	-	-	-
Amphipoda						
Coropium triaenonyx	*	*	-	-	-	-
Photis digitata	*	*	-	*	-	*
Eriopisa chilkensis	**	*	-	-	-	*
Melita zylanica	*	*	-	*	*	-
Cheriophotis megacheles	*	*	-	-	-	*
Ampelisca	-	-	-	-	-	-
Gammaropsis sp	-	-	-	-	-	*
Gammarid sp	-	-	-	-	-	-
Leucothoe sp	-	-	-	-	-	-
Caprella sp	**	**	**	-	-	*
Amphipoda sp	**	**	***	*	*	*
Isopoda						
Anthuridae sp	-	-	-	-	-	-
Cirolana fluviatilis	*	*	*	*	*	-

Tanaidacea						
Apseudus chilkensis	*	*	*	*	*	*
Mollusca						
Littorina littorea	**	-	-	-	-	-
Gastropoda sp	-	*	*	*	*	*
Bivalvia sp	**	*	*	*	*	*
Phalium sp	-	-	-	-	-	-
Villoria cyprinoides	-	-	-	-	-	-
Modiolus sp	-	-	-	-	-	-
Dentalium sp	-	-	-	-	-	-
Oligochaeta						
Tubificidae sp	-	*	*	-	**	***
Decapoda	*	*	-	*	*	*
Cumacea	*	*	*	-	-	-
Foraminifera	*	-	***	*	*	-
Chironomid	-	*	-	-	-	-
Others	-	-	-	-	-	*

Table 5.7 Macrobenthic density in the sampling sites during 2010 (- <100 **-100-500, ***- 500-1000, ****- 1000-3000)*

	2010					
	ND PRM	ND MN	ND PM	D PRM	D MN	D PM
Polychaeta						
Amphinomid sp	-	-	-	-	-	-
Amphinomea rostrata	-	-	-	-	-	-
Etone sp	-	-	-	-	-	-
Phylldoce sp	-	-	-	*	-	-
Harmothoe sp	-	-	-	-	-	-
Hesionidae sp	-	-	-	-	*	-
Sigambra parva	*	-	-	*	*	**
Glycera longipinnis	*	-	-	-	-	-
Goniada emerita	-	-	-	*	-	-
Nephtys dibranchia	*	-	-	*	*	-
Nephtys oligobranchia	-	-	-	*	*	*
Nephtys polybranchiata	-	-	-	-	-	-

Dendronereis estuarina	*	*	*	*	-	-
Nereis sp	*	*	-	-	-	-
Lycastis indica	-	-	-	-	-	*
Cirratulus cirratus	-	-	-	*	-	-
Cirratulus filiformis	-	-	-	*	*	*
Dodecaceria sp	-	-	-	-	-	-
Paraprionospio pinnata	-	-	-	*	-	-
Prionospio cirrifera	**	**	**	*	*	*
Prionospio cirrobranchiata	**	*	-	*	-	*
Polydora ciliata	-	-	-	*	-	-
Prionospio sp	-	-	-	*	*	-
Spionid sp	*	-	-	-	-	*
Caulleriella capensis	-	-	-	-	-	-
Scolelepis squamata	-	-	*	-	-	*
Boccardia polybranchia	-	-	-	-	-	-
Mediomastus capensis	****	*	**	**	*	**
Capitella capitata	*	*	*	*	*	*
Paraheteromastus tenuis	*	**	**	*	*	*
Capitellid sp	*	*	*	*	*	*
Lumbriconereis impatiens	-	-	-	*	-	-
Lumbriconereis latreilli	*	-	-	-	-	-
Diopatra neopolitana	**	-	*	*	-	-
Eunice sp	-	-	-	-	-	-
Cossura coasta	-	-	-	**	-	-
Aricidea longobranchiata	*	-	-	-	-	-
Owenia fusiformis	*	-	-	*	-	-
Serpulidae sp	-	-	-	-	-	-
Pectinaria crassa	-	-	-	*	-	-
Pisione sp	-	-	-	-	-	-
Amphipoda						
Coropium triaenonyx	**	*	**	*	-	*
Photis digitata	*	*	**	*	-	-
Eriopisa chilkensis	-	*	*	-	-	-
Melita zylanica	-	-	-	-	-	*
Cheriophotis megacheles	-	*	*	-	-	-
Ampelisca	-	-	-	-	-	-
Gammaropsis sp	-	-	-	-	-	-
Gammarid sp	-	-	-	-	-	-
Leucothoe sp	-	-	-	-	-	-

Caprella sp	*	**	*	-	-	-
Amphipoda sp	*	-	*	*	-	-
Isopoda						
Anthuridae sp	*	*	-	-	-	-
Cirolana fluviatilis	-	-	-	-	-	-
Tanaidacea						
Apseudus chilkensis	*	*	*	*	**	***
Mollusca						
Littorina littorea	-	*	*	-	-	-
Gastropoda sp	*	-	*	*	-	-
Bivalvia sp	-	-	*	-	-	-
Phalium sp	**	-	-	-	-	-
Villoria cyprinoides	-	-	-	*	-	-
Modiolus sp	*	-	-	-	-	-
Dentalium sp	-	*	-	-	-	-
Oligochaeta						
Tubificidae sp	*	*	-	*	****	***
Decapoda	-	-	-	*	*	-
Cumacea	*	-	-	-	-	-
Foraminifera	-	-	-	-	-	-
Chironomid	-	*	*	-	-	-
Others	-	-	*	*	-	*

Table 5.8 Macrobenthic density in the sampling sites during 2011(- <100 **-100-500, ***- 500-1000, ****- 1000-3000)*

			2011			
	ND PRM	ND MN	ND PM	D PRM	D MN	D PM
Polychaeta						
Amphinomid sp	*	-	-	*	-	-
Amphinomea rostrata	*	-	-	-	-	-
Etone sp	-	-	-	-	-	-
Phylldoce sp	-	-	-	-	-	-
Harmothoe sp	-	-	-	-	-	-
Hesionidae sp	-	-	-	-	-	-
Sigambra parva	*	-	*	*	*	*
Glycera longipinnis	-	*	-	-	-	*

Goniada emerita	-	-	*	-	-	-
Nephtys dibranchia	-	-	-	*	-	-
Nephtys oligobranchia	-	-	-	*	*	*
Nephtys polybranchiata	*	-	-	-	-	-
Dendronereis estuarina	-	**	**	*	-	-
Nereis sp	*	-	-	-	-	-
Lycastis indica	*	-	-	-	-	-
Cirratulus cirratus	-	-	-	*	-	-
Cirratulus filiformis	*	-	*	*	-	**
Dodecaceria sp	-	-	**	-	-	-
Paraprionospio pinnata	-	-	-	*	-	-
Prionospio cirrifera	**	*	***	*	*	*
Prionospio cirrobranchiata	*	-	-	*	-	-
Polydora ciliata	-	-	-	*	-	-
Prionospio sp	-	-	-	*	-	-
Spionid sp	-	-	-	*	-	-
Caulleriella capensis	-	***	-	-	*	*
Scolelepis squamata	-	-	-	-	*	-
Boccardia polybranchia	-	-	-	-	*	-
Mediomastus capensis	***	**	****	*	**	*
Capitella capitata	*	-	*	-	-	*
Paraheteromastus tenuis	**	**	*	-	*	-
Capitellid sp	-	*	-	-	-	-
Lumbriconereis impatiens	*	-	-	-	-	-
Lumbriconereis latreilli	-	-	-	*	-	-
Diopatra neopolitana	*	-	**	*	-	-
Eunice sp	-	-	-	-	-	-
Cossura coasta	-	-	-	*	-	-
Aricidea longobranchiata	-	-	-	*	-	-
Owenia fusiformis	-	-	*	-	-	*
Serpulidae sp	*	-	-	-	-	-
Pectinaria crassa	-	-	-	-	-	-

Pisione sp	-	-	*	-	-	-
Amphipoda						
Coropium triaenonyx	-	*	*	-	-	-
Photis digitata	**	*	***	-	-	-
Eriopisa chilkensis	*	*	*	*	-	-
Melita zylanica	-	*	*	-	-	*
Cheriophotis megacheles	*	*	**	-	-	-
Ampelisca	-	-	-	-	-	*
Gammaropsis sp	-	*	-	-	*	-
Gammarid sp	-	-	**	-	-	-
Leucothoe sp	-	-	*	-	-	-
Caprella sp	**	**	*	-	-	-
Amphipoda sp	-	-	***	*	*	-
Isopoda						
Anthuridae sp	*	-	-	-	-	-
Cirolana fluviatilis	-	-	-	-	*	-
Tanaidacea						
Apseudus chilkensis	**	-	*	*	*	*
Mollusca						
Littorina littorea	*	*	*	-	-	-
Gastropoda sp	*	-	-	-	-	-
Bivalvia sp	-	-	*	*	-	*
Phalium sp	-	-	-	-	-	-
Villoria cyprinoides	-	-	-	-	-	-
Modiolus sp	-	-	-	-	-	-
Dentalium sp	-	-	-	-	-	-
Oligochaeta						
Tubificidae sp	**	*	*	***	****	**
Decapoda	-	-	-	-	*	*
Cumacea	-	*	-	-	-	-
Foraminifera	-	-	-	-	-	-
Chironomid	-	*	-	-	-	-
Others	-	-	*	-	*	*

*****$$*****

Chapter 6

SUMMARY AND CONCLUSION

Tropical estuaries, complex and highly productive ecosystems have a crucial role in the global ocean processing (Smith et al., 2003). They are influenced by the high rate of precipitation and evaporation, with homogeneity in vertical salinity distribution during the dry season and weak to strong stratification in the wet season. The present study was undertaken in the Cochin estuary (CE), one of the largest estuarine systems (256 km^2) in the southwest coast of India, known for its high productivity and biodiversity (Qasim, 2003). The CE is a micro-tidal tropical estuary regularly influenced by the intrusion of seawater from the Arabian Sea (AS) and the inflow of freshwater from seven rivers and its tributaries. The estuary receives an annual freshwater influx of $22{,}000\times10^6$ m^3 and about 320 cm of annual rainfall, of which nearly 60-65% occurs during the southwest monsoon season (Qasim, 2003). The CE is referred as a 'monsoonal estuary' and hydrographical characteristics of this estuary are modulated by the Indian Summer Monsoon (ISM). The prevailing salinity gradient in the estuary supports high diversity of flora and fauna. This tropical estuary, possessed with high productivity (average gross primary production is 280 g C/ m^2/yr,), acts as a nursery ground for many species of marine and estuarine fin fishes and shell fishes (Qasim et al., 1969; Menon et al., 2000).

Macro-benthic community in the estuarine ecosystems plays a vital role in many of the ecosystem processes, such as cycling of nutrients by the transfer of materials from primary production through the detrital pool into higher trophic levels, pollutant metabolism, dispersion, burial, and secondary production. Macrobenthos are used to characterize trophic relationships in the aquatic ecosystems and act as indicators of energy transfer. As majority of the benthic fauna are sedentary and sessile and cannot avoid any environmental alternations, they have been used as indicators of environmental perturbations.

The present study is designed to procure information on the spatio-temporal distribution of macrobenthic fauna in the CE and the adjacent coastal waters. The study also delineates the feeding ecology of polychaetes, one of the major benthic faunal groups, used to assess the aquatic environmental conditions. The present study forms the first report pertaining to the occurrence, population structure, and ecology of a tube building amphipod, *Chelicorophium madrasensis,* Nayar from South-west coast of India. In addition, the study describes the impact of maintenance dredging activities on the macrobenthic fauna in the CE for three consecutive years (2009-2012), and forms one of the pioneering work dealing with the impacts of dredging on the macrobenthic faunal distribution, abundance, biomass, composition, diversity and also feeding guilds. The present thesis is comprised of six chapters and the salient features of each chapter are summarized below.

Chapter 1 includes an overview of benthos, classification and their importance in aquatic ecosystems, the significance of the study area, description of various groups of benthic organisms and their environmental responses. A brief description of the scope and objectives of the study also has been mentioned in this section. Chapter 2 comprised of a detailed description of the study area, sampling locations, sampling strategies, and also a comprehensive description of the methodology adopted for the collection and analysis of samples/parameters.

Chapter 3 describes the spatio-temporal distribution of macrobenthic community in the CE and adjacent coastal waters. This study was based on sampling from 11 stations, i.e., 7 stations, positioned in the CE at various salinity regimes and 4 stations in the adjacent coastal waters. Monthly sampling was conducted in these stations for one year duration from January to December 2011. In most of the ecosystems, community structure emerges as a result of the complex interaction between biotic and environmental variables. As a tropical estuary, the seasonal monsoonal heavy rainfall and associated run off brings about significant changes in the physico-chemical and biological characteristics of the CE and it usually creates a drastic change in the environmental and water quality parameters. Therefore, it is imperative to study the seasonal distribution and diversity of the macrobenthic fauna in the estuary and adjacent coastal waters.

Water quality parameters exhibited a distinct spatio-temporal changes in the estuary, whereas in the coastal waters it was comparatively less. Relatively higher water temperature observed in the estuary as compared to the adjacent coastal waters. A monsoonal drop in the water temperature and salinity was apparent in the estuary and coastal waters during the study period. Spatially, present study evidenced an increasing trend in salinity and pH from the estuary towards the sea. Lower pH observed during PRM in the estuary as well as in the adjacent coastal waters might be due to the higher decomposition of the organic matter during the period. Reduced ability of seawater to solubilize the dissolved oxygen might be responsible for the lower DO of coastal waters compared to the estuary. Spatial variation in SPM was more distinct in the study region than seasonal variation. Estuarine stations adjacent to the shipping channel and coastal stations located near to the estuary had higher level of SPM. Though the estuary was evidenced with higher level of inorganic nutrients irrespective of seasons, the exceptionally higher level of nutrients (nitrate, nitrite, phosphate and silicate) occurred during MN and it might have derived as a result of the torrential rainfall and the subsequent runoff. Higher concentration of ammonia observed during non-monsoon months in the estuary and coastal waters might have derived from the anthropogenic activities. The sediment texture exhibited prominent spatial variation in the estuary as compared to the seasonal fluctuations. The sediment organic carbon was relatively higher in the estuary during PRM and MN while in the coastal waters, it was higher during MN and PM period. As high dilution occurs in the estuarine waters during the MN season, the organic carbon was found to be less as compared to non-monsoon seasons. Higher macrobenthic density, biomass and diversity evidenced in the CE compared to the adjacent coastal waters. A total of 54 polychaete species were encountered from the CE and 39 species from the coastal waters. This variation could be due to the wide range of environmental variables and availability of varied food resources in the estuary. Both in the estuary and the adjacent coastal waters, lower diversity observed during MN may be the result of the heavy rainfall and associated land runoff which was observed to impose severe stress on benthic communities. Polychaetes formed the dominant macrobenthic fauna at both study area, which indicates their diverse feeding guilds. The RDA analysis revealed that salinity, sediment texture, and organic carbon formed the major influencing factors of macrobenthic distribution and density in the CE. Domination of deposit feeding mode observed both in the estuary and coastal waters (SDF and SSDF), shows the significance of organic detritus as an energy source for the macrobenthic fauna. The spatial variation in macrobenthic density was more prominent than seasonal variation in the estuary. However, the macrobenthic fauna in the coastal waters are concerned, seasonal variation was more pronounced than spatial variation. A distinct variation was quite apparent in the community structure of macrobenthic fauna between the CE and the adjacent coastal waters. From the overall analysis of the present study, it can be inferred that salinity and sediment texture had a major influence on the distribution and abundance of macrobenthic fauna in the Cochin estuary; however, high salinity in the coastal waters was conducive only to certain macrobenthic fauna and hence forms a limiting factor influencing the distribution of macrobenthic organisms. Apart from this, SPM and organic carbon showed a substantial influence on the distribution and abundance of macrobenthic fauna in the CE and the adjacent coastal waters. The present study has revealed the dominance of many opportunistic macrobenthic species in the CE and adjacent coastal waters, which apparently signifies the influence of ongoing anthropogenic activities in and around the study area.

Chapter 4 delineates the ecology and population structure of a tube building amphipod, *Chelicorophium madrasensis* in the CE. A remarkable density of macrobenthic fauna was observed at a specific site in the CE, north of Panambukadu Island, was the anxious motivation to study the ecology of this amphipod species. Monthly sampling was carried out in the estuary throughout the year in 2011. Higher macrobenthic density was sustained at the particular sampling location (Panambukad north) almost throughout the year and this remarkable macrobenthic density was due to the predominance of the tube building amphipod, *Chelicorophium madrasensis*. Since this tube-dwelling amphipod species was not reported elsewhere from the coastal waters of the southwest coast of India including the CE, the present study forms the first report on its occurrence and ecology from this area. High density of *C. madrasensis* (0.0477 to 1.972 x 10^6ind.m^{-2}) was encountered from the sampling site throughout the year. Major water quality parameters (temperature, salinity, nitrite and phosphate) exhibited significant seasonality in the study area. Fresh water domination was prevalent in the sampling location during MN period, whereas medium salinity condition sustained during the remaining periods. The density of *C. madrasensis* was quite higher during PRM and PM when mesohaline condition prevailed. The results of the multivariate analysis have revealed a significant correlation between the salinity and density of amphipod species and this relationship apparently show the influence of salinity on the density of this species. Environmental parameters such as DO, BOD and inorganic nutrients did not show much influence on this particular amphipod species. The sediment texture of the sampling location was mainly composed of silty clay. The occurrence of fine fractions of the sediment in the sampling location almost throughout the study period normally indicates the prevalence of low energy conditions. Earlier studies (Ramamirtham and Muthusamy, 1986; Balachandran et al., 2005) have referred this area as a null zone, due to the synchronous tidal activity experiencing from both inlets, i.e., the Cochin inlet in the south and Azhikode inlet in the north. The RDA plots also signify the above finding, as this amphipod species exhibited a positive affinity towards the clay (strong affinity) and silt particles (weak affinity), and a negative affinity towards the sand. Higher benthic chlorophyll *a* was noticed at this sampling site almost throughout the year as compared to other sampling locations in the CE. Earlier studies (Gerdol and Hughes, 1994b; Grant and Daborn, 1994) have revealed that amphipod species belong to the genus *Corophium* preferably feeds on benthic microalgal community. The positive correlation between the benthic chlorophyll *a* and the density of *C. madrasensis*, and also the results of the multivariate analysis further confirm the influence of benthic chlorophyll *a* on the abundance of *C. madrasensis*. Similar to other corophid amphipods, *C. madrasensis* also has a characteristic tube building behavior using fine sediments and mucous. Laboratory experiments conducted as part of the present study also confirmed the tube building activity of this amphipod species. The aggregation of these organisms in large numbers at the particular sampling site in the CE led to the formation of multiple tubes in sediments. The present study substantiates *C. madrasensis* as a continuous breeder, as juveniles of this species were observed in the sampling site almost throughout the year. Although this species is a considered as a continuous breeder, their most preferable breeding season was identified during premonsoon. Exceptionally higher density and biomass of *C. madrasensis* in the sampling site found to support the sustenance of two carnivorous isopods i.e., *Cirolana fluviatilis* and *Anthurid* sp in the same location. In conclusion it can be stated that the population dynamics of the *C. madrasensis* in the CE was influenced by the environmental parameters such as salinity and temperature, and also various sediment properties (texture and microflora) associated with the biota. The feeding modes and the predation pressure experienced from the carnivorous isopods also can be considered as one of the important biotic factors influencing their population structure. The prevailing ecological interaction between the physico-chemical and biological parameters helps in maintaining a higher density of *C. madrasensis* in the sampling location. The information generated from the present study can be used as a baseline data for further investigations about the ecological and morphological characteristics of the tube-building amphipods in the coastal waters of southwest coast of India.

Chapter 5 describes the impact of maintenance-dredging on the macrobenthic community structure in the CE. Inorder to maintain the navigable depth in estuaries, regular dredging is carried out in the shipping channels to prevent siltation. These recurrent dredging activities often have serious implications on the estuarine environment as it alters the bottom topography, sediment composition, alterations in depth, current strength and also the removal of the substratum. Comprehensive observations and data collections were carried out in the CE during three consecutive years (2009-2011) to study the dredging impact on macrobenthic fauna. Six stations were selected for this particular study, of which 4 stations located in the dredging area (D Stations) and 2 stations in the non-dredging area (ND stations). The results of the water quality parameters in the sampling locations did not show significant spatial variation between the dredging and non-dredging sites, except salinity and SPM. However, the sediment components and benthic fauna exhibited a significant spatial variation between the dredging and non-dredging sites. Being positioned near to the inlets, the dredging sites continuously received a large supply seawater through tidal incursion as compared to the non-dredging sites. The finer sediment particles derived by the dredging activities might have a crucial role in contributing towards a higher concentration of SPM in the dredging sites. Dominance of finer fractions of sediment was evident in the dredging sites as compared to the non-dredging sites. The substrata of the dredging sites were characterized by relatively higher level of organic carbon and this enhancement in organic carbon apparently led to the significant spatial variation ($p<0.05$) between the dredging and non-dredging sites. Remarkable reduction in the faunal biomass and density of the dredging sites along with a conspicuous variation in the community composition further substantiates the impact of dredging activities on benthic biota in the CE. The present study has documented prominent variation in the benthic diversity; species richness and species number between the dredging and non-dredging sites of the CE and apparently it indicate the disturbances induced by the maintenance dredging. The present study showed that the dredging sites were characterized by a conspicuous dominance of benthic opportunistic tubificid oligochaetes, whereas in the non-dredging sites polychaetes were dominated. The analysis of feeding guild composition of polychaetes and also the *characterizing species* revealed a clear domination of carnivores in the dredging sites. As motility of organisms often help them to efficiently adapt to a continuously disturbed environment, relatively higher motility of the carnivorous polychaetes might have favored their higher density in the dredging sites throughout the study period. Formation of distinct clusters (results of cluster analysis) of stations, based on the density of macrobenthic data clearly indicates the prevalence of different biotic environment in the dredging and non-dredging sites. The results of BO2A index, depicting the habitat quality of a particular environment, also suggest the pervasiveness of contrasting environmental scenario in the dredging and non-dredging sites. Higher BO2A index values recorded in the dredging sites indicate the proliferation of opportunistic species. In the present study, the observed dominance of opportunistic species in the dredging sites was a clear indication of recolonization of the r-selected benthic fauna impacted by continuous dredging activities. The CE is a shallow dynamic estuary, where continuous dredging has been carried out in the channels which are intermittently influenced by tidal activities. The recolonization of the opportunistic communities would be fast as the substratum is of fine mud. Though, the result of the study suggests the possibility of recolonization by the opportunistic benthic fauna in the CE, the succession of these r-selected benthic fauna into a stable complex benthic community could not be possible within a short period of time, as continuous dredging is going on in these channels.

As proper navigation through the aquatic environment 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, for the proper understanding and management of the coastal ecosystems, it is necessary to generate comprehensive data on the water quality and associated ecology of the biota. In order to minimize the effects of dredging, these operations should be avoided especially during the period of sensitive breeding, spawning and larval dispersal periods of the marine organisms. Region-specific evaluations should be undertaken to generate knowledge of the local fauna and their growth cycles. In addition frequency of dredging activities can be reduced in order to affirm the rapid recovery of the macrobenthic fauna in the dredging channels. Therefore, extensive studies should be executed in the CE with proper consideration to its physico-chemical and biological attributes in relation with dredging activities. It further helps to formulate proper management strategies to minimize the effects of dredging on the estuarine biota.

To conclude, the present study provided knowledge on the ecology of the macrobenthic community in and around CE. The current study also added valuable information regarding the utility of macrobenthic community in assessing the ecological changes associated with anthropogenic activities in the estuarine ecosystem.

$$

REFERENCES

Abdul Jaleel, K., 2013. Macrobenthosof the cotinental margin (200-1000 m) of South Eastern Arabian Sea with special reference to Polychaetes, Cochin university of science and Technology, Kochi.

Abdul Jaleel, K., Kumar, P.A., Khan, K.N., Correya, N.S., Jacob, J., Philip, R., Sanjeevan, V., Damodaran, R., 2014. Polychaete community structure in the South Eastern Arabian Sea continental margin (200–1000m). Deep Sea Research Part I: Oceanographic Research Papers 93, 60-71.

Abdul Jaleel, K., Parameswaran, U.V., Gopal, A., Khader, C., Ganesh, T., Sanjeevan, V., Shunmugaraj, T., Vijayan, A.K., Gupta, G., 2015. Evaluation of changes in macrobenthic standing stock and polychaete community structure along the south eastern Arabian Sea shelf during the monsoon trawl-ban. Cont. Shelf Res. 102, 9-18.

Admiraal, W., Peletier, H., Brouwer, T., 1984. The seasonal succession patterns of diatom species on an intertidal mudflat: an experimental analysis. Oikos, 30-40.

Ahn, I.-Y., Kang, Y.-C., Choi, J.-W., 1995. The influence of industrial effluents on intertidal benthic communities in Panweol, Kyeonggi Bay (Yellow Sea) on the west coast of Korea. Mar. Pollut. Bull. 30, 200-206.

Akoumianaki, I., Papaspyrou, S., Kormas, K.A., Nicolaidou, A., 2013. Environmental variation and macrofauna response in a coastal area influenced by land runoff. Estuar. Coast. Shelf Sci. 132, 34-44.

Alongi, D.D., 1989. The role of soft-bottom benthic communities in tropical mangrove and coral reef ecosystems. CRC Critical Review in Aquatic Sciences-pages: 1: 243-280.

Alongi, D.M., 1990. The ecology of tropical soft-bottom benthic ecosystems. Oceanogr. Mar. Biol. Ann. Rev 28, 381-496.

Ambika Devi, M., Pillai, N., 1990. Effect of pollution due to coconut husk retting on the species diversity of benthic communities of Cochin backwaters. Indian J. Fish. 37, 145-149.

Anderson, B.S., Lowe, S., Phillips, B.M., Hunt, J.W., Vorhees, J., Clark, S., Tjeerdema, R.S., 2008. Relative sensitivities of toxicity test protocols with the amphipods Eohaustorius estuarius and Ampelisca abdita. Ecotoxicol. Environ. Saf. 69, 24-31.

Annandale, N., 1907. The fauna of brackish ponds at Port Canning, Lower Bengal. Rec. Indian Mus 1, 35-43.

Annandale, N., Kemp, S., 1915. Fauna of the Chilka Lake. order of the Trustees of the Indian Museum.

Anon, 1996. Examination of environmental viability and sustainability of select projects. . NEERI Report to Supreme Court of India.

Ansari, Z., Harkantra, S., Nair, S., Parulekar, A., 1977. Benthos of the Bay of Bengal: A preliminary account. Mahasagar 10, 55-60.

Ansari, Z., Ingole, B., Parulekar, A., 1986. Effect of high organic enrichment of benthic polychaete population in an estuary. Mar. Pollut. Bull. 17, 361-365.

Ansari, Z., Parulekar, A., 1993. Distribution, abundance and ecology of the meiofauna in a tropical estuary along the west coast of India. Hydrobiologia 262, 115-126.

Anto, A., Varma, P.U., Iyer, H.K., 1977. Siltation in the outer channel of Cochin harbour. Ind J of Geo-mar sci, 6.

Antonio, E.S., Kasai, A., Ueno, M., Ishihi, Y., Yokoyama, H., Yamashita, Y., 2012. Spatial-temporal feeding dynamics of benthic communities in an estuary-marine gradient. Estuar. Coast. Shelf Sci. 112, 86-97.

APHA, 2005. Standard methods for the examination of water and wastewater, in: edn., s. (Ed.). American Public Health Association/American Water Works Association / Water Environment Federation,, Washington DC.

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

Aravindakshan, P., Balasubramanian, T., Devi, C.L., Nair, K.C., Gopalakrishnan, T., Jayalakshmy, K., Kutty, M.K., 1992. Benthos and substratum characteristics of prawn culture fields in and around the Cochin backwater.

Babu, M., Das, V.K., Vethamony, P., 2006. BOD–DO modeling and water quality analysis of a waste water outfall off Kochi, west coast of India. Environ. Int. 32, 165-173.

Bailey-Brock, J.H., 2000. A new record of the polychaete Boccardia proboscidea (Family Spionidae), imported to Hawai'i with oysters. Pac. Sci. 54, 27.

Balachandran, K., 2001. Chemical oceanographic studies of the coastal waters of Cochin. Ph. D. Thesis, Cochin University of Science and Technology.

Balachandran, K., Joseph, T., Nair, K., Nair, M., Joseph, P., 2002. The complex estuarine formation of six rivers (Cochin backwater system on west coast of India)—Sources and distribution of trace metals and nutrients, APN/SASCOM/LOICZ regional workshop on assessment of material fluxes to the coastal zone in South Asia and their Impacts. Negombo, pp. 8-11.

Balachandran, K.K., Raj, C.L., Nair, M., Joseph, T., Sheeba, P., Venugopal, P., 2005. Heavy metal accumulation in a flow restricted, tropical estuary. Estuar. Coast. Shelf Sci. 65, 361-370.

Balchand, A.N., Rasheed, K., 2000. Assessment of short term environmental impacts on dredging in a tropical estuary. Terra et Aqua, 16-25.

Barletta, M., Cysneiros, F., Lima, A., 2016. Effects of dredging operations on the demersal fish fauna of a South American tropical–subtropical transition estuary. J. Fish Biol. 89, 890-920.

Barnard, J., Karaman, G., 1991. The families and genera of ma rine gammaridean Amphioda (except marine gammaroids). Record of the Australian Museum, Supplement, 13: 1 417.

Barnard, J.L., 1962. South Atlantic abyssal amphipods collected by RV Vema. Columbia University Press.

Barnard, J.L., 1969. The families and genera of marine gammaridean Amphipoda. Smithsonian Institution Press Washington, DC.

Barnard, J.L., Thomas, J.D., Sandved, K.B., 1988. Behavior of gammaridean Amphipoda: Corophium, Grandidierella, Podocerus, and Gibberosus (American Megaluropus) in Florida. Crustaceana. Supplement, 234-244.

Batcha, A., 1984. Studies on the bottom fauna of north Vembanad Lake. Ph. D. Thesis, University of Cochin, India. relation to the deposition of degraded water fern Salvinia and other macrophytes. Proc. natn. Estuarine Management.

Bellan, G., Desrosiers, G., Willsie, A., 1988. Use of an annelid pollution index for monitoring a moderately polluted littoral zone. Mar. Pollut. Bull. 19, 662-665.

Bemvenuti, C.E., Angonesi, L.G., Gandra, M.S., 2005. Effects of dredging operations on soft bottom macrofauna in a harbor in the Patos Lagoon estuarine region of southern Brazil. Braz. J. Biol. 65, 573-581.

Bendschneider, K., Robinson, R.J., 1952. A new spectrophotometric method for the determination of nitrite in sea water.

Berlow, E.L., Navarrete, S.A., 1997. Spatial and temporal variation in rocky intertidal community organization: lessons from repeating field experiments. J. Exp. Mar. Biol. Ecol. 214, 195-229.

Berry W, Rubinstein N, Melzian B, Hill B (2003) The biological effects of suspended and bedded sediment (SABS) in aquatic systems: a review United States Environmental Protection Agency, Duluth

Besser, J.M., Giesy, J.P., Kubitz, J.A., Verbrugge, D.A., Coon, T.G., Braselton, W.E., 1996. Assessment of sediment quality in dredged and undredged areas of the Trenton Channel of the Detroit River, Michigan USA, using the sediment quality triad. J. Great Lakes Res. 22, 683-696.

Beukema, J., Flach, E., 1995. Factors controlling the upper and lower limits of the intertidal distribution of two Corophium species in the Wadden Sea. Mar. Ecol. Prog. Ser. 125, 117-126.

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

Blake, J., 1995. Introduction to the Polychaeta. Chapter 4. In: J. Blake, C. Erseus, B. Milbig (eds.), Taxonomic Atlas of the benthic fauna of the Santa Maria Basin and Western Santa Barbara Channel, pp. 39-51 Santa Barbara Museum of Natural History. California.

Blake, J.A., Arnofsky, P.L., 1999. Reproduction and larval development of the spioniform Polychaeta with application to systematics and phylogeny. Hydrobiologia 402, 57-106.

Bolam, S.G., Rees, H.L., 2003. Minimizing impacts of maintenance dredged material disposal in the coastal environment: a habitat approach. Environ. Manage. 32, 171-188.

Boltovskoy, E., Wright, R., 1976. The Systematic Position and Importance of the Foraminifera, Recent Foraminifera. Springer, pp. 1-3.

Borja, A, Dauer, D.M., Elliott, M., Simenstad, C.A., 2010. Medium-and long-term recovery of estuarine and coastal ecosystems: patterns, rates and restoration effectiveness. Estuar. Coast. 33, 1249-1260.

Bouchet, P., 2000. The elusive inventory of species. Research 333: 40-44.

Bousfield, E.L., Hoover, P., 1997. The amphipod superfamily Corophioidea on the Pacific coast of North America: 5. Family Corophiidae: Corophiinae, new subfamily: systematics and distributional ecology. Amphipacifica.

Boyd, S.E., Limpenny, D.S., Rees, H.L., Cooper, K.M., Campbell, S., 2003. Preliminary observations of the effects of dredging intensity on the re-colonisation of dredged sediments off the southeast coast of England (Area 222). Estuar. Coast. Shelf Sci. 57, 209-223.

Boyden, C., Little, C., 1973. Faunal distributions in soft sediments of the Severn Estuary. Estuarine and Coastal Marine Science 1, 203-223.

Bray, R.N., Bates, A., Land, J.M., 1997. Dredging: a handbook for engineers.

Briggs, J., Whitwell, T., Fernandez, R.T., Riley, M.B., 2002. Effect of integrated pest management strategies on chlorothalonil, metalaxyl, and thiophanate-methyl runoff at a container nursery. J. Am. Soc. Hort. Sci. 127, 1018-1024.

Brown, B.E., Kumar, A.J., 1990. Temporal and spatial variations in iron concentrations of tropical bivalves during a dredging event. Mar. Pollut. Bull. 21, 118-123.

Brown, S.S., Gaston, G.R., Rakocinski, C.F., Heard, R.W., 2000. Effects of sediment contaminants and environmental gradients on macrobenthic community trophic structure in Gulf of Mexico estuaries. Estuaries 23, 411-424.

Brusca, R.C., Coelho, V., Taiti, S., 2001. A guide to the coastal isopods of California. Internet address: http://tolweb. org/notes.

Bryan, G., Langston, W., 1992. Bioavailability, accumulation and effects of heavy metals in sediments with special reference to United Kingdom estuaries: a review. Environ. Pollut. 76, 89-131.

Buchanan, J., Sheader, M., Kingston, P., 1978. Sources of variability in the benthic macrofauna off the south Northumberland coast, 1971–1976. J. Mar. Biol. Assoc. U.K. 58, 191-209.

Cadotte, M., Albert, C.H., Walker, S.C., 2013. The ecology of differences: assessing community assembly with trait and evolutionary distances. Ecol. Lett. 16, 1234-1244.

Caine, E.A., 1977. Feeding mechanisms and possible resource partitioning of the Caprellidae (Crustacea: Amphipoda) from Puget Sound, USA. Mar. Biol. 42, 331-336.

Caspers, H., 1973. RO Brinkhurst and GBM Jamieson: Aquatic Oligochaeta of the World.—860 pp. Edinburgh: Oliver & Boyd 1971.£ 12. Internationale Revue der gesamten Hydrobiologie und Hydrographie 58, 147-147.

Chandramohan, P., Kumar, V.S., Kumar, S.J., 1996. Dredge spoil disposal off Kavaratti island, Lakshadweep, India

Chao, M., Shi, Y., Quan, W., Shen, X., An, C., Yuan, Q., Huang, H., 2012. Distribution of benthic macroinvertebrates in relation to environmental variables across the Yangtze River Estuary, China. J. Coast. Res. 28, 1008-1019.

Chapman, J.W., 2007. Amphipoda: Chapter 39 of The Light and Smith Manual: Intertidal Invertebrates from Central California to Oregon. Completely Revised and Expanded.

Ceia, F.R., Patrício, J., Franco, J., Pinto, R., Fernandez-Boo, S., Losi, V., Marques, J.C., Neto, J.M., 2013. Assessment of estuarine macrobenthic assemblages and ecological quality status at a dredging site in a southern Europe estuary. Ocean Coast. Manage. 72, 80-92.

Clarke, K.R., Gorley, R.N., 2006. User manual/tutorial. Primer-E Ltd., Plymouth, 93.

Clarke, K., Warwick, R., 1994. An approach to statistical analysis and interpretation. Change in Marine Communities 2.

Clarke, K.R., Warwick, R.M., 2001. An approach to statistical analysis and interpretation. Change in Marine Communities 2.

Clarke, K.R., Warwick, R.M., Brown, B.E., 1993. An index showing breakdown of seriation, related to disturbance, in a coral-reef assemblage. Mar. Ecol. Prog. Ser. 102, 153-160.

Coles, S., 1979. Benthic microalgal populations on intertidal sediments and their role as precursors to salt marsh development. Ecological processes in coastal environments, 25-42.

Colby, L.H., Maycock, S.D., Nelligan, F.A., Pocock, H.J., Walker, D.J., 2010. An investigation into the effect of dredging on tidal asymmetry at the river murray mouth J Coast Res:843-850.

Collins, N., Williams, R., 1981. Zooplankton of the Bristol Channel and Severn Estuary. The distribution of four copepods in relation to salinity. Mar. Biol. 64, 273-283.

Commito, J.A., 1982. Importance of predation by infaunal polychaetes in controlling the structure of a soft-bottom community in Maine, USA. Mar. Biol. 68, 77-81.

Conlan, K., 1994. Amphipod crustaceans and environmental disturbance: a review. J. Nat. Hist. 28, 519-554.

Conner, W.G., Simon, J.L., 1979. The effects of oyster shell dredging on an estuarine benthic community. Estuarine and coastal marine science 9, 749-758.

Conti, G., Massa, F., 1998. Experienze de allavamento del polichete Diopatra neapolitana Delle Chiaje, 1841 nella Laguna di S. Gilla (Sardegna Meridionale). Biologia Marina Meditteranea 5, 1473-1480.

Cooper, J.A.G., 2003. Anthropogenic impacts on estuaries. Coastal zones, estuaries. Encyclopedia of Life Support Systems (EOLSS). Oxford, UK, UNESCO, EOLSS Publishers, 246pp.

Crawford, G., 1937. A review of the amphipod genus Corophium, with notes on the British species. Journal of the Marine Biological Association of the United Kingdom (New Series) 21, 589-630.

Crema, R., Castelli, A., Prevedelli, D., 1991. Long term eutrophication effects on macrofaunal communities in northern Adriatic Sea. Mar. Pollut. Bull. 22, 503-508.

Crisp, D., 1965. Surface chemistry, a factor in the settlement of marine invertebrate larvae. Botanica Gothoburgensia III 3, 51-65.

Crisp, D., 1971. Energy flow measurements. In 'Methods for the Study of Marine Benthos'.(Eds NA Holme, AO McIntyre.) pp. 197-279. Blackwell Scientific Publications: Oxford, UK..

Cunha, M., Moreira, M., Sorbe, J., 2000a. The amphipod Corophium multisetosum (Corophiidae) in Ria de Aveiro (NW Portugal). II. Abundance, biomass and production. Mar. Biol. 137, 651-660.

Cunha, M., Sorbe, J., Moreira, M., 2000b. The amphipod Corophium multisetosum (Corophiidae) in Ria de Aveiro (NW Portugal). I. Life history and aspects of reproductive biology. Mar. Biol. 137, 637-650.

Dagli, E., Ergen, Z., Cinar, M.E., 2005. One-year observation on the population structure of Diopatra neapolitana Delle Chiaje (Polychaeta: Onuphidae) in Izmir Bay (Aegean Sea, eastern Mediterranean). Mar. Ecol. 26, 265-272.

Damodaran, R. 1972. Meiobenthos of the mudbanks of Kerala coast. Proc. Indian Acad. Sci (Sect. B) 38 B: 288-297.

Danulat, E., Muniz, P., García-Alonso, J., Yannicelli, B., 2002. First assessment of the highly contaminated harbour of Montevideo, Uruguay. Mar. Pollut. Bull. 44, 554-565.

Dauvin, J.C., Ruellet, T., 2009. The estuarine quality paradox: is it possible to define an ecological quality status for specific modified and naturally stressed estuarine ecosystems? Mar. Pollut. Bull. 59, 38-47.

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., 878 pp.

Day, J.W., 1989. Estuarine ecology. John Wiley & Sons.

De-la-Ossa-Carretero, J., Del-Pilar-Ruso, Y., Giménez-Casalduero, F., Sánchez-Lizaso, J., Dauvin, J.-C., 2012. Sensitivity of amphipods to sewage pollution. Estuar. Coast. Shelf Sci. 96, 129-138.

de Jonge, V.N., 1983. Relations between annual dredging activities, suspended matter concentrations, and the development of the tidal regime in the Ems estuary. Can. J. Fish. Aquat. Sci. 40, 289-300.

de Jonge, V.N., 2000. Importance of temporal and spatial scales in applying biological and physical process knowledge in coastal management, an example for the Ems estuary. Cont. Shelf Res. 20, 1655-1686.

de Jonge, V.N., Schuttelaars, H.M., van Beusekom, J.E., Talke, S.A., de Swart, H.E., 2014. The influence of channel deepening on estuarine turbidity levels and dynamics, as exemplified by the Ems estuary. Estuar. Coast. Shelf Sci. 139, 46-59.

Desprez, M., 2000. Physical and biological impact of marine aggregate extraction along the French coast of the Eastern English Channel: short-and long-term post-dredging restoration ICES J Mar Sci 57:1428-1438

Devi, K.S., Jayalakshmy, K., Venugopal, P., 1991. Communities and coexistence of benthos in northern limb of Cochin backwaters .Ind.J.Mar.Sci.20:249-254.

De Paiva, P., 1993. Trophic structure of a shelf polychaete taxocoenosis in southern Brazil. Cah. Biol. Mar. 35, 39-55.

Desai, B., Kutty, M.K., 1967a. A comparison of the marine and estuarine benthic fauna of the nearshore regions of the Arabian Sea. Bulletin of the National Institute of Science, India 38, 677-688.

Desai, B., Kutty, M.K., 1967b. Studies on the benthic fauna of Cochin backwater, Proceedings of the Indian Academy of Sciences-Section B. Springer, pp. 123-142.

Desprez, M., 2000. Physical and biological impact of marine aggregate extraction along the French coast of the Eastern English Channel: short-and long-term post-dredging restoration. ICES J. Mar. Sci. 57, 1428-1438.

Desroy, N., Retière, C., 2004. Using benthos as a tool for coastal management: The impact of the tidal power station on benthic communities of the Rance basin. Aquat. Ecosyst. Health Manage. 7, 59-72.

Devassy, V., Bhattathiri, P., 1974. Phytoplankton ecology of the Cochin backwater. Indian J. Mar. Sci 3, 46-50.

Devi, K.S., Jayalakshmy, K., Venugopal, P., 1991. Communities and coexistence of benthos in northern limb of Cochin backwaters.

Devi, K.S., Sheba, P., Balasubramanian, T., Venugopal, P., Sankaranarayanan, V., 1999. Benthic fauna of southwest and southeast coasts of India. The Fourth Indian Fisheries Forum, Proceedings, eds. Joseph, M.M.; Menon, N.R.; Nair, N.U. Kochi, Kerala., 9-12p.

Devi, K.S., Venugopal, P., 1989. Benthos of Cochin backwaters receiving industrial effluents. Indian Journal of Marine Sciences . 18, 165-169.

Dix, T.L., Markham, S.E., Goetting, B.K., Karlen, D.J., Holden, C.M., Grabe, S.A., 2004. Spionid Polychaetes as Environmental Indicators, Estuarine Indicators. CRC Press, pp. 277-295.

Dixon, I., Moore, P.G., 1997. A comparative study on the tubes and feeding behaviour of eight species of corophioid Amphipoda and their bearing on phylogenetic relationships within the Corophioidea. Philosophical Transactions of the Royal Society of London B: Biological Sciences 352, 93-112.

Dodge, R.E., Vaisnys, J.R., 1977. Coral populations and growth patterns: responses to sedimentation and turbidity associated with dredging. J Mar Res 35:715-730.

Dolbeth, M., Cardoso, P.G., Grilo, T.F., Bordalo, M.D., Raffaelli, D., Pardal, M.A., 2011. Long-term changes in the production by estuarine macrobenthos affected by multiple stressors. Estuar. Coast. Shelf Sci. 92, 10-18.

Drolet, D., Barbeau, M.A., 2012. Population structure of resident, immigrant, and swimming Corophium volutator (Amphipoda) on an intertidal mudflat in the Bay of Fundy, Canada. J. Sea Res. 70, 1-13.

El Wakeel, S., Riley, J., 1957. The determination of organic carbon in marine muds. Journal du conseil 22, 180-183.

Elias, R., 1992. Quantitative benthic community structure in Blanca Bay and its relationship with organic enrichment. Mar. Ecol. 13, 189-201.

Elias, R., Palacios, J., Rivero, M., Vallarino, E., 2005. Short-term responses to sewage discharge and storms of subtidal sand-bottom macrozoobenthic assemblages off Mar del Plata City, Argentina (SW Atlantic). J. Sea Res. 53, 231-242.

Elias, R., Vallarino, E.A., Scagliola, M., Isla, F.I., 2004. Macrobenthic distribution patterns at a sewage disposal site in the inner shelf off Mar del Plata (SW Atlantic). J. Coast. Res., 1176-1182.

Elliott, M., Quintino, V., 2007. The estuarine quality paradox, environmental homeostasis and the difficulty of detecting anthropogenic stress in naturally stressed areas. Mar. Pollut. Bull. 54, 640-645.

Elliott, M., Whitfield, A., 2011. Challenging paradigms in estuarine ecology and management. Estuar. Coast. Shelf Sci. 94, 306-314.

Ellis, J., Norkko, A., Thrush, S.F., 2000. Broad-scale disturbance of intertidal and shallow sublittoral soft-sediment habitats; effects on the benthic macrofauna. J. Aquat. Ecosyst. Stress Recovery 7, 57-74.

Elton, C., 1927. Animal Ecology, 1927. Sidgwick & Jackson, LTD, London, p.56.

Eriksson, S.P., Wennhage, H., Norkko, J., Norkko, A., 2005. Episodic disturbance events modify predator–prey interactions in soft sediments. Estuar. Coast. Shelf Sci. 64, 289-294.

Esselink, P., Van Belkum, J., Essink, K., 1989. The effect of organic pollution on local distribution of Nereis diversicolor and Corophium volutator. Netherlands Journal of Sea Research 23, 323-332.

Farara, D., Burt, A., 1993. Environmental assessment of western Lake Erie sediments and benthic communities: 1991. Report prepared for the Ontario Ministry of Environment and Energy, Water Resources Branch, Great Lakes Section by Beak Consultants, Brampton, Ontario, Canada.

Fauchald, K., 1977. The polychaete worms: definitions and keys to the orders, families and genera.

Fauchald, K., Jumars, P.A., 1979. The diet of worms: A study of polychaete feeding guilds. Aberdeen University Press.

Fauvel, P., 1953. The fauna of India (Annelida, Polychaeta), 507 pp. Allahabad: Indian Press Ltd.

Feebarani, J., 2009. Meiobenthos of Cochin backwaters in relation to macrobenthos and environmental parameters. Cochin University of Science and Technology.

Feebarani, J., Joydas, T., Damodaran, R., Borja, A., 2016. Benthic quality assessment in a naturally-and human-stressed tropical estuary. Ecol. Indicators 67, 380-390.

Fenchel, T., Kofoed, L., Lappalainen, A., 1975. Particle size-selection of two deposit feeders: the amphipod Corophium volutator and the prosobranch Hydrobia ulvae. Mar. Biol. 30, 119-128.

Fielding, P., Damstra, K.S.J., Branch, G., 1988. Benthic diatom biomass, production and sediment chlorophyll in Langebaan Lagoon, South Africa. Estuar. Coast. Shelf Sci. 27, 413-426.

Fincham, A., 1969. Amphipods of the shallow-water sand community in the northern Irish Sea. J. Mar. Biol. Assoc. U.K. 49, 1003-1024.

Fisher, T.R., Carlson, P.R., Barber, R., 1982. Sediment nutrient regeneration in three North Carolina estuaries. Estuar. Coast. Shelf Sci. 14, 101-116.

Flach, E., De Bruin, W., 1993. Effects of Arenicola marina and Cerastoderma edule on distribution, abundance and population structure of Corophium volutator in Gullmarsfjorden western Sweden. Sarsia 78, 105-118.

Flach, E., De Bruin, W., 1994. Does the activity of cockles, Cerastoderma edule (L.) and lugworms, Arenicola marina L., make Corophium volutator Pallas more vulnerable to epibenthic predators: a case of interaction modification? J. Exp. Mar. Biol. Ecol. 182, 265-285

Flemming, H.C., Schmitt, J., Marshall, K.C., 1996. Sorption properties of biofilms, Sediments and toxic substances. Springer, pp. 115-157.

Forbes T L , Forbes V E, H, D.M., 1994. Individual physiological responses to environmental hypoxia and organic enrichment:Implications for early soft-bottom community uccession. . J Mar Res 52, 1081–1100. .

Foster, J.M., LeCroy, S.E., Heard, R.W., Vargas, R., 2009. Gammaridean amphipods, Marine Biodiversity of Costa Rica, Central America. Springer, pp. 265-274.

Franz, D., Harris, W., 1988. Seasonal and spatial variability in macrobenthos communities in Jamaica Bay, New York—an urban estuary. Estuaries and Coasts 11, 15-28.

Gambi, M.C., Conti, G., Bremec, C.S., 1998. Polychate distribution, diversity and seasonality related to seagrass cover in shallow bottoms of the Tyrrenian Sea (Italy). Scientia Marina 62, 1-17.

Ganapathi, P., Rao, L., 1959. Preliminary observation on the bottom fauna of the continental shelf of the northeast coast of India, Proceedings of the First All India Congress of Zoology, pp. 8-13.

Ganesh, T., Rakhesh, M., Raman, A., Nanduri, S., Moore, S., Rajanna, B., 2014. Macrobenthos response to sewage pollution in a tropical inshore area. Environ. Monit. Assess. 186, 3553-3566.

Ganesh, T., Raman, A., 2007. Macrobenthic community structure of the northeast Indian shelf, Bay of Bengal. Mar. Ecol. Prog. Ser. 341, 59-73.

Gaudêncio, M.J., Cabral, H., 2007. Trophic structure of macrobenthos in the Tagus estuary and adjacent coastal shelf. Hydrobiologia 587, 241-251.

Gee, J., 1961. Ecological studies in South Benfleet Creek with special reference to the amphipod genus Corophium. Essex Nat 30, 291-309.

Gerdol, V., Hughes, R., 1994a. Effect of Corophium volutator on the abundance of benthic diatoms, bacteria and sediment stability in two estuaries in southeastern England. Marine ecology progress series. Oldendorf 114, 109-115.

Gerdol, V., Hughes, R., 1994b. Feeding behaviour and diet of Corophium volutator in an estuary in southeastern England. Marine Ecology-Progress Series 114, 103-103.

Gerlach, S., 1971. On the importance of marine meiofauna for benthos communities. Oecologia 6, 176-190.

Giangrande, A., 1997. Polychaete reproductive patterns, life cycles and life histories: an overview. Oceanography and marine Biology 35, 323-386.

Gibbs, P., Burt, G., Pascoe, P., Llewellyn, C., Ryan, K., 2000. Zinc, copper and chlorophyll-derivatives in the polychaete Owenia fusiformis. Journal of the Marine Biological Association of the UK 80, 235-248.

Giere, O., Pfannkuche, O., 1982. Biology and ecology of marine Oligochaeta, a review. Oceanography and Marine Biology 20, 173-309.

Gilbert, F., Stora, G., Bonin, P., 1998. Influence of bioturbation on denitrification activity in Mediterranean coastal sediments: an in situ experimental approach. Mar. Ecol. Prog. Ser. 163, 99-107.

Giles, G., 1885. Description of a new Species of the Amphipod Genus Melita from the Bay of Bengal. Baptist Mission P.

Gopalakrishna Pillai, N., 1978. Macrobenthos of a tropical estuary. Cochin University of Science and Technology.

Gopalan, U., Vengayil, D.T., Udayavarma, V., Krishnankutty, M., 1983. The shrinking backwaters of Kerala. Journal of the Marine Biological Association of India. Cochin 25, 131-141.

Gopinathan C, Qasim S (1971) Silting in navigational channels of the Cochin harbour area J Mar Biol Assoc India 13:14-26

Gosner, K.L.,197I Guide to identification of marine and estuarine invertebrates. Wiley-Interscience, John Wiley and Sons, Inc., New York, 693pp.

Govindan, K., Varshney, P., Desai, B., 1983. Benthic studies in South Gujarat estuaries. Mahasagar 16, 349-356.

Grall, J., Glemarec, M., 1997. Using biotic indices to estimate macrobenthic community perturbations in the Bay of Brest. Estuar. Coast. Shelf Sci. 44, 43-53.

Grant, J., Daborn, G., 1994. The effects of bioturbation on sediment transport on an intertidal mudflat. Netherlands Journal of Sea Research 32, 63-72.

Grasshoff, K., 1964. On the determination of silica in sea water, Deep Sea Research and Oceanographic Abstracts. Elsevier, pp. 597-604.

Grasshoff, K., 1970. A simultaneous multiple channel system for nutrient analysis in seawater with analog and digital data record. Technicon Q 3, 7-17.

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

Gray, J., 1974. Animal-sediment relationships. Oceanogr. Mar. Biol. Annu. Rev. 12, 223-261.

Gray, J., Waldichuk, M., Newton, A., Berry, R., Holden, A., Pearson, T., 1979. Pollution-induced changes in populations [and discussion]. Philosophical Transactions of the Royal Society of London B: Biological Sciences 286, 545-561.

Gray, J.S., 1989. Effects of environmental stress on species rich assemblages. Biol. J. Linn. Soc. 37, 19-32.

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

Groot, S.d., 1980. The consequences of marine gravel extraction on the spawning of herring, Clupea harengus Linné J Fish Biol 16:605-611

Guerra Garcia, J.M., Corzo, J., Carlos García Gomez, J., 2003. Short Term Benthic Recolonization after Dredging in the Harbour of Ceuta, North Africa. Mar. Ecol. 24, 217-229.

Gupta, G.V.M., Thottathil, S.D., Balachandran, K.K., Madhu, N.V., Madeswaran, P., Nair, S., 2009. CO2 supersaturation and net heterotrophy in a tropical estuary (Cochin, India): influence of anthropogenic effect. Ecosystems 12, 1145-1157.

Gutperlet, R., Capperucci, R.M., Bartholomä, A., Kröncke, I., 2015. Benthic biodiversity changes in response to dredging activities during the construction of a deep-water port. Marine Biodiversity, 1-21.

Hacking, N., 2003. A review of the ecology of offshore ocean sediments with particular reference to marine aggregate resources for beach nourishment in New South Wales Centre for Natural Resources Department of Infrastructure, Planning and Natural Resources

Hedge, L., Knott, N., Johnston, E., 2009. Dredging related metal bioaccumulation in oysters Mar Pollut Bull 58:832-840

Haeckel, E., 1890. Plankton-studien.

Harkantra, S., 1975. Benthos of the Kali estuary, Karwar. Mahasagar 8, 53-58.

Harkantra, S., Nair, A., Ansari, Z., Parulekar, A., 1980. Benthos of the shelf region along the west coast of India.Ind. J. Geo-Mar. Sci.9, 106-110.

Harkantra, S., Parulekar, A., 1985. Community structure of sand-dwelling macrofauna of an estuarine beach in Goa, India. Mar. Ecol. Prog. Ser. 30, 291-294.

Harkantra, S.N., Rodrigues, N.R., 2003. Pattern of species succession of soft-bottom macrofauna in the estuaries of Goa, west coast of India. Indian Academy of Sciences.

Hart, C.W., and Fuller, S.L.H., 1979. Pollution ecology of estuarine invertebrates. Academic Press, Newyork.

Hawkins, C., 1985. Population carbon budgets and the importance of the amphipod Corophium volutator in the carbon transfer on a Cumberland Basin mudflat, upper Bay of Fundy, Canada. Netherlands Journal of Sea Research 19, 165-176.

Heip, C., 1993. Benthic studies: summary and conclusions. NIOO, CEMO.

Henriksen, K., Rasmussen, M., Jensen, A., 1983. Effect of bioturbation on microbial nitrogen transformations in the sediment and fluxes of ammonium and nitrate to the overlaying water. Ecological Bulletins, 193-205.

Hilton, C., Walde, S.J., Leonard, M.L., 2002. Intense episodic predation by shorebirds may influence life history strategy of an intertidal amphipod. Oikos 99, 368-376.

Holland, A., Zingmark, R., Dean, J., 1974. Quantitative evidence concerning the stabilization of sediments by marine benthic diatoms. Mar. Biol. 27, 191-196.

Holmstrom, W.F., Morgan, E., 2013. Some Properties of the Tiddl Activity Rhythm in the Estuarine Amphipod Corophium volutofor, Cyclic Phenomena in Marine Plants and Animals: Proceedings of the 13th European Marine Biology Symposium, Isle of Man, 27 September-4 October 1978. Elsevier, p. 355.

Hossain, M.B., Marshall, D.J., 2014. Benthic infaunal community structuring in an acidified tropical estuarine system. Aquatic biosystems 10, 1-12.

Hossain, M.B., Marshall, D.J., Venkatramanan, S., 2014. Sediment granulometry and organic matter content in the intertidal zone of the Sungai Brunei estuarine system, northwest coast of Borneo. Carpath J Earth Env Sci 9, 231-239.

Hossain, S., Eyre, B.D., McKee, L.J., 2004. Impacts of dredging on dry season suspended sediment concentration in the Brisbane River estuary, Queensland, Australia. Estuar. Coast. Shelf Sci. 61, 539-545.

Hughes, R., 1988. Dispersal by benthic invertebrates: the in situ swimming behaviour of the amphipod Corophium volutator. J. Mar. Biol. Assoc. U.K. 68, 565-579.

Hutchings, P., 1998. Biodiversity and functioning of polychaetes in benthic sediments. Biodivers. Conserv. 7, 1133-1145.

Icely, J., Nott, J., 1985. Feeding and digestion in Corophium volutator (Crustacea: Amphipoda). Mar. Biol. 89, 183-195.

Imrie, D.W., McCrohan, C.R., Hawkins, S.J., 1990. Feeding behaviour in Littorina littorea: a study of the effects of ingestive conditioning and previous dietary history on food preference and rates of consumption, Progress in Littorinid and Muricid Biology. Springer, pp. 191-198.

Ingole, B., Sivadas, S., Nanajkar, M., Sautya, S., Nag, A., 2009. A comparative study of macrobenthic community from harbours along the central west coast of India. Environ. Monit. Assess. 154, 135-146.

Iriarte, A., Purdie, D.A., 1994. Size distribution of chlorophyll a biomass and primary production in a temperate estuary (Southampton Water): the contribution of photosynthetic picoplankton. Marine Ecology-Progress Series 115, 283-283.

Jayaraj, K., Jayalakshmi, K., Saraladevi, K., 2007. Influence of environmental properties on macrobenthos in the northwest Indian shelf. Environ. Monit. Assess. 127, 459-475.

Jayaraj, K., Josia, J., Kumar, P.D., 2008. Infaunal macrobenthic community of soft bottom sediment in a tropical shelf. J. Coast. Res., 708-718.

Jensen, A., Mogensen, B., 2000. Environmental Aspects of Dredging. Effects Ecology and Economy IADC/CEDA, Hague.

Jones, S., Jago, C., 1993. In situ assessment of modification of sediment properties by burrowing invertebrates. Mar. Biol. 115, 133-142.

Jones, R.A., Lee, G.F., 1981. The significance of dredging and dredged material disposal as a source of nitrogen and phosphorus for estuarine waters. In: Estuaries and Nutrients. 517-530

Joseph, J., Kurup, P., 1989. Volume transport and estuarine features at Cochin inlet. Mahasagar 22, 165-172.

Joseph, T., Balachandran, K., Nair, M., Venugopal, P., Sankaranarayanan, V., 1998. Changes in water quality at Cochin harbour dredging site, south west coast of India. Indian J. Mar. Sci. 27, 250-252p.

Joydas, T., Damodaran, R., 2009. Infaunal macrobenthos along the shelf waters of the west coast of India, Arabian Sea.

Johnston Jr, S.A., 1981. Estuarine dredge and fill activities: a review of impacts Environ Manage 5:427-440

Junior, S., Santos, S.R., Travassos, M., Vianna, M., 2012. Impact on a fish assemblage of the maintenance dredging of a navigation channel in a tropical coastal ecosystem Brazilian Journal of Oceanography 60:25-32.

Jyothibabu, R., Madhu, N., Jayalakshmi, K., Balachandran, K., Shiyas, C., Martin, G., Nair, K., 2006. Impact of freshwater influx on microzooplankton mediated food web in a tropical estuary (Cochin backwaters–India). Estuar. Coast. Shelf Sci. 69, 505-518.

Kannan, K., 1979. Ecological and socio-economic consequences of water-control projects in the Kuttanad region of Kerala. Proceedings of the Indian Academy of Sciences Section C: Engineering Sciences 2, 417-433.

Kaplan, E.H., Welker, J.R., Kraus, M.G., McCourt, S., 1975. Some factors affecting the colonization of a dredged channel. Mar. Biol. 32, 193-204.

Kennish, M.J., 1991. Ecology of estuaries: anthropogenic effects. CRC press.

Kennish, M., 1992. Ecology of Estuaries: Anthropogenic Effects CRC Press Boca Raton Ann Arbor, London

Kennish, M.J., 2001. Benthic Communities of the Barnegat Bay—Little Egg Harbor Estuary. J. Coast. Res., 167-177.

Kennish, M.J., Haag, S.M., Sakowicz, G.P., Durand, J.B., 2004. Benthic Macrofaunal Community Structure along a Well-Defined Salinity Gradient in the Mullica River–Great Bay Estuary. J. Coast. Res., 209-226.

Ketchum, B.H., 1967. Phytoplankton nutrients in estuaries.

Kinne, O., 1966. Physiological aspects of animal life in estuaries with special reference to salinity. Netherlands Journal of Sea Research 3, 222-244.

Kinne, O., 1977. Environmental factors-salinity: animal-invertebrates. Mar. Ecol., 821-996.

Klump, J.V., Martens, C.S., 1981. Biogeochemical cycling in an organic rich coastal marine basin—II. Nutrient sediment-water exchange processes Geochim Cosmochim Acta 45:101107-105121

Koroleff, F., 1983. Total and organic nitrogen. Methods of seawater analysis 2, 162-173.

Krishnan, L., John, P., 1974. Observations on the breeding biology of Melita zeylanica Stebbing, a brackish water amphipod. Hydrobiologia 44, 413-430.

Kroncke, l., 2006. Structure and function of macrofaunal communities influenced by hydrodynamically controlled food availabilty in theWadden Sea, the open North Sea and the Deep-sea. A synopsis. Senckenb Marit 36, 123–164.

Krumbein, W.C., Pettijohn, F.J., 1938. Manual of sedimentary petrography.

Kurian, C., 1971. Distribution of benthos on the southwest coast of India. Fertility of the sea 1, 225-239.

Kurien, C., 1967. Studies on the benthos of the South West Coast of India. Bulletin National Institute of Sciences of India 38, 649-656.

Kurian, C., Damodaran, R., Antony, A., 1975. Bottom fauna of the Vembanad lake. Bull. Dept. Mar. Sci. Univ. Cochin 7, 987-994.

Lamptey, E., Armah, A.K., 2008. Factors affecting macrobenthic fauna in a tropical hypersaline coastal lagoon in Ghana, West Africa. Estuaries and Coasts 31, 1006-1019.

Laprise, R., Dodson, J.J., 1994. Environmental variability as a factor controlling spatial patterns in distribution and species diversity of zooplankton in the St. Lawrence Estuary. Marine Ecology-Progress Series 107, 67-67.

Leal, J.H., Matthews, B., 2013. Bivalves. Shell Museum ,Florida,USA.

Levin, L.A., Gage, J.D., Martin, C., Lamont, P.A., 2000. Macrobenthic community structure within and beneath the oxygen minimum zone, NW Arabian Sea. Deep Sea Research Part II: Topical Studies in Oceanography 47, 189-226.

Levin, L.A., Ziebis, W., Mendoza, G.F., Growney, V.A., Tryon, M.D., Brown, K.M., Mahn, C., Gieskes, J.M., Rathburn, A.E., 2003. Spatial heterogeneity of macrofauna at northern California methane seeps: influence of sulfide concentration and fluid flow. Mar. Ecol. Prog. Ser. 265, 123-139.

Limia, J., Raffaelli, D., 1997. The effects of burrowing by the amphipod Corophium volutator on the ecology of intertidal sediments. J. Mar. Biol. Assoc. U.K. 77, 409-423.

Liu, H., He, Q., Wang, Z., Weltje, G.J., Zhang, J., 2010. Dynamics and spatial variability of near-bottom sediment exchange in the Yangtze Estuary, China. Estuar. Coast. Shelf Sci. 86, 322-330.

Lohrer, A.M,, Wetz, J.J., 2003. Dredging-induced nutrient release from sediments to the water column in a southeastern saltmarsh tidal creek Mar Pollut Bull 46:1156-1163

Lopez, G.R., Levinton, J.S., 1978. The availability of microorganisms attached to sediment particles as food for Hydrobia ventrosa Montagu (Gastropoda: Prosobranchia). Oecologia 32, 263-275.

Lorenzen, C.J., 1967. Determination of chlorophyll and pheo-pigments: spectrophotometric equations. Limnol. Oceanogr. 12, 343-346.

Lowe, S., Thompson, B., 1997. Identifying benthic indicators for San Francisco bay, regional monitoring program, annual report. San Francisco Estuary Institute, Oakland, CA.

Lundquist, C.J., Thrush, S.F., Coco, G., Hewitt, J.E., 2010. Interactions between disturbance and dispersal reduce persistence thresholds in a benthic community. Mar. Ecol. Prog. Ser. 413, 217-228.

Madhu, N.V., Balachandran, K.K., Martin, G.D., Jyothibabu, R., Thottathil, S.D., Nair, M., Joseph, T., Kusum, K.K., 2010a. Short-term variability of water quality and its implications on phytoplankton production in a tropical estuary (Cochin backwaters—India). Environ. Monit. Assess. 170, 287-300.

Madhu, N., Jyothibabu, R., Balachandran, K., Honey, U., Martin, G., Vijay, J., Shiyas, C., Gupta, G., Achuthankutty, C., 2007. Monsoonal impact on planktonic standing stock and abundance in a tropical estuary (Cochin backwaters–India). Estuar. Coast. Shelf Sci. 73, 54-64.

Madhu, N.V., Jyothibabu, R., Balachandran, K.K., 2010b. Monsoon-induced changes in the size-fractionated phytoplankton biomass and production rate in the estuarine and coastal waters of southwest coast of India. Environ. Monit. Assess. 166, 521-528.

Madhupratap, M., 1979. Distribution, community structure and species succession of copepods from Cochin backwaters.

Madhupratap, M., 1987. Status and strategy of zooplankton of tropical Indian estuaries: A review. Bulletin of Plankton Society of Japan.

Madhupratap, M., Rao, T., 1979. Tidal and diurnal influence on estuarine zooplankton, Indian journal of Geo-marine sciences, pp. 9-11.

Magalhaes, W.F., Barros, F., 2011. Structural and functional approaches to describe polychaete assemblages: ecological implications for estuarine ecosystems. Marine and Freshwater Research 62, 918-926.

Maiorano, P., Mastrototaro, F., Beqiraj, S., Costantino, G., Kashta, L., Gherardi, M., Sion, L., D'Ambrosio, P., Carlucci, R., D'Onghia, G., 2011. Bioecological study of the benthic communities on the soft bottom of the Vlora Gulf (Albania). J. Coast. Res., 95-105.

Mandal, S., Harkantra, S.N., 2013. Changes in the soft-bottom macrobenthic diversity and community structure from the ports of Mumbai, India. Environ. Monit. Assess. 185, 653-672.

Manjunatha, B., Yeats, P., Smith, J., Shankar, R., Narayana, A., Prakash, T., 1998. Accumulation of heavy metals in sediments of marine environments along the south west coast of India, Proc. Internatl. Symp. Mar. Pollution, Monaco, pp. 5-9.

Mannino, A., Montagna, P.A., 1997. Small-scale spatial variation of macrobenthic community structure. Estuaries and Coasts 20, 159-173.

Mare, M.F., 1942. A study of a marine benthic community with special reference to the micro-organisms. J. Mar. Biol. Assoc. U.K. 25, 517-554.

Martin, G., George, R., Shaiju, P., Muraleedharan, K., Nair, S., Chandramohanakumar, N., 2012. Toxic metals enrichment in the surficial sediments of a eutrophic tropical estuary (Cochin Backwaters, Southwest Coast of India). The Scientific World Journal 2012.

Martin, G., Jyothibabu, R., Madhu, N., Balachandran, K., Nair, M., Muraleedharan, K., Arun, P., Haridevi, C., Revichandran, C., 2013. Impact of eutrophication on the occurrence of Trichodesmium in the Cochin backwaters, the largest estuary along the west coast of India. Environ. Monit. Assess. 185, 1237-1253.

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

Mathew, K., Selvaraj, G., Naomi, T., Varghese, M., Sridhar, N., Sridhar, M., Antony, G., Leelabhai, K., Anilkumar, R., Solomon, K., 1994. Studies on the infestation of an isopod crustacean, Cirolana fluviatilis in some parts of the Cochin backwaters, Kerala. Marine Fisheries Information Service, Technical and Extension Series 131, 1-10.

Mattila, J., Bonsdorff, E., 1989. The impact of fish predation on shallow soft bottoms in brackish waters (SW Finland); an experimental study. Netherlands Journal of Sea Research 23, 69-81.

McLusky, D., 1989. The estuarine ecosystem, 215 pp. London: Blackie and Son.

McLusky, D.S., 1968. Some effects of salinity on the distribution and abundance of Corophium volutator in the Ythan estuary. J. Mar. Biol. Assoc. U.K. 48, 443-454.

McLusky, D.S., 1993. Marine and estuarine gradients-an overview. Aquat. Ecol. 27, 489-493.

McLusky, D.S., Elliott, M., 2004. The estuarine ecosystem: ecology, threats and management. Oxford University Press on Demand.

Meadows, P., 1964. Experiments on substrate selection by Corophium species: films and bacteria on sand particles. J. Exp. Biol. 41, 499-511.

Meadows, P., Tait, J., 1989. Modification of sediment permeability and shear strength by two burrowing invertebrates. Mar. Biol. 101, 75-82.

Meadows, P., Tait, J., Hussain, S., 1990. Effects of estuarine infauna on sediment stability and particle sedimentation. Hydrobiologia 190, 263-266.

Malone, T.C., Conley, D.J., Fisher, T.R., Glibert, P.M., Harding, L.W., Sellner, K.G., 1996. Scales of nutrient-limited phytoplankton productivity in Chesapeake Bay. Estuaries and Coasts 19, 371-385.

Mendez, N., Flos, J., Romero, J., 1998. Littoral soft-bottom polychaete communities in a pollution gradient in front of Barcelona (Western Mediterranean, Spain). Bull. Mar. Sci. 63, 167-178.

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

Menon, N. N. & N. R. Menon, 1998. Status of oil contamination in the Cochin estuary-Presented at the International Conference on Environmental Science (ICES 98) at RRL, Trivandrum from December 2-4, 1998.

Mercaldo-Allen, R., Goldberg, R., 2011. Review of the ecological effects of dredging in the cultivation and harvest of molluscan shellfish NOAA Technical Memorandum NMFS-NE 220

Miranda, J., Balachandran, K., Ramesh, R., Wafar, M., 2008. Nitrification in Kochi backwaters. Estuar. Coast. Shelf Sci. 78, 291-300.

Mondal, N., Rajkumar, M., Sun, J., Kundu, S., Lyla, P.S., Khan, S.A., Trilles, J.P., 2010. Biodiversity of brackish water amphipods (crustacean) in two estuaries, southeast coast of India. Environ. Monit. Assess. 171, 471-486.

Moore, P.G., 1981. The life histories of the amphipods Lembos websteri Bate and Corophium bonnellii Milne Edwards in kelp holdfasts. J. Exp. Mar. Biol. Ecol. 49, 1-50.

Morino, H., 1978. STUDIES ON THE TALITRIDAE (AMPHIPODA, CRUSTACEA) IN JAPAN-III. LIFE HISTORY AND BREEDING ACTIVITY OF ORCHESTIA PLATENSIS KRφYER.

Morris, J.T., 1995. The mass balance of salt and water in intertidal sediments: results from North Inlet, South Carolina. Estuaries and Coasts 18, 556-567.

Mouritsen, K.N., Mouritsen, L.T., Jensen, K.T., 1998. Change of topography and sediment characteristics on an intertidal mud-flat following mass-mortality of the amphipod Corophium volutator. J. Mar. Biol. Assoc. U.K. 78, 1167-1180.

Murphy, J., Riley, J.P., 1962. A modified single solution method for the determination of phosphate in natural waters. Anal. Chim. Acta 27, 31-36.

Musale, A.S., Desai, D.V., 2011. Distribution and abundance of macrobenthic polychaetes along the South Indian coast. Environ. Monit. Assess. 178, 423-436.

Muniz, P., Pires, A., 1999. Trophic structure of polychaetes in the Sao Sebastiao Channel (southeastern Brazil). Mar. Biol. 134, 517-528.

Myers, A., Lowry, J., 2009. The biogeography of Indo-West Pacific tropical amphipods with particular reference to Australia. Zootaxa 2260, 109-127.

Nair, K., Anger, K., 1979. Experimental studies on the life cycle of Jassa falcata (Crustacea, Amphipoda). Helgol Wiss Meeresunters 32, 444-452.

Nair, K., Gopalakrishnan, T., Venugopal, P., Peter, M.G., Jayalakshmi, K., Rao, T., 1983. Population dynamics of estuarine amphipods in Cochin backwaters. Marine ecology progress series. Oldendorf 10, 289-295.

Nair, N.B., Azis, P.A., Arunachalam, M., Dharmaraj, K., Krishnakumar, K., 1984. Ecology of Indian estuaries: Ecology and distribution of benthic macrofauna in the Ashtamudi Estuary, Kerala. Mahasagar 17, 89-101.

Nair, C.K., Balchand, A.N., Chacko, J., 1993. Sediment characteristics in relation to changing hydrography of Cochin estuary.

Nair, S., Balchand, A., Nambisan, P., 1990. Metal concentrations in recently deposited sediments of Cochin backwaters, India. Sci. Total Environ. 97, 507-524.

Nandan, S.B., Azis, P.A., 1995a. Benthic polychaetes in the anoxic sulphide biomes of the retting zones in the Kadinamkulam Kayal. Int. J. Environ. Stud. 47, 257-267.

Nandan, S.B., Azis, P.A., 1995b. Pollution indicators of coconut husk retting areas in the Kayals of Kerala. Int. J. Environ. Stud. 47, 19-25.

Nayar, K., 1950. Description of a new species of amphipod of the genus Corophium from Adyar, Madras, India. J. Wash. Acad. Sci. 40, 225-228.

Nayar, K.N., 1959. Amphipoda of the Madras Coast. Bulletin of the Madras Government Museum 6, 15-76.

Nayar, K.N., 1966. On the gammaridean Amphipoda of the Gulf of Mannar, with special reference to those of the pearl and chank beds.

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

Nendza, M., 2002. Inventory of marine biotest methods for the evaluation of dredged material and sediments. Chemosphere 48, 865-883.

Neto, J.M., Teixeira, H., Patrício, J., Baeta, A., Veríssimo, H., Pinto, R., Marques, J.C., 2010. The response of estuarine macrobenthic communities to natural-and human-induced changes: dynamics and ecological quality. Estuaries and Coasts 33, 1327-1339.

Newell, R., 1965. The role of detritus in the nutrition of two marine deposit feeders, the prosobranch Hydrobia ulvae and the bivalve Macoma balthica, Proceedings of the Zoological Society of London. Wiley Online Library, pp. 25-45.

Newell, R.C., Hitchcock, D.R., Seiderer, L.J., 1999. Organic enrichment associated with outwash from marine aggregates dredging: a probable explanation for surface sheens and enhanced benthic production in the vicinity of dredging operations. Mar. Pollut. Bull. 38, 809-818.

Newell, R.C., Seiderer, L.J., Hitchcock, D.R., 1998. The impact of dredging works in coastal waters: a review of the sensitivity to disturbance and subsequent recovery of biological resources on the sea bed. Oceanogr. Mar. Biol. Annu. Rev. 36, 127-178.

Newell, R.C., Seiderer, L.J., Simpson, N.M., Robinson, J.E., 2004. Impacts of marine aggregate dredging on benthic macrofauna off the south coast of the United Kingdom. J. Coast. Res., 115-125.

Newman, B.K., Wooldridge, T.H., Cockcroft, A.C., 2007. Aspects of the biology and ecology of the estuarine cirolanid isopod, Cirolana fluviatilis. Afr. Zool. 42, 12-22.

Nguyen, L.M., Cooke, J.G., McBride, G.B., 1997. Phosphorus retention and release characteristics of sewage-impacted wetland sediments. Water, Air, Soil Pollut. 100, 163-179.

Nisha, A.P., 2008. Studies on the macrobenthic community of Cochin backwaters with special reference to culture of Eriopisa chilkensis (Gammaridae-Amphipoda). Cochin University of Science and Technology.

Ogbeibu, A., Arazu, V., Nzei, N., Igho, R., 2010. Impact of dredging on the water quality and macrobenthic invertebrate fauna of the Ikpoba River in southern Nigeria. Trop. Freshwat. Biol. 19, 59.

Olsgard, F., Brattegard, T., Holthe, T., 2003. Polychaetes as surrogates for marine biodiversity: lower taxonomic resolution and indicator groups. Biodivers. Conserv. 12, 1033-1049.

Osman, R.W., 1977. The establishment and development of a marine epifaunal community. Ecol. Monogr. 47, 37-63.

Onorati, F., Bigongiari, N., Pellegrini, D., Giuliani, S., 1999. The suitability of Corophium orientale (Crustacea, Amphipoda) in harbour sediment toxicity bioassessment. Aquat. Ecosyst. Health Manage. 2, 465-473.

Pacheco, A.S., Gonzalez, M.T., Bremner, J., Oliva, M., Heilmayer, O., Laudien, J., Riascos, J.M., 2011. Functional diversity of marine macrobenthic communities from sublittoral soft-sediment habitats off northern Chile Helgol Mar Res 65:413-424

Paerl, H.W., Valdes, L.M., Joyner, A.R., Peierls, B.L., Piehler, M.F., Riggs, S.R., Christian, R.R., Eby, L.A., Crowder, L.B., Ramus, J.S., 2006. Ecological response to hurricane events in the Pamlico Sound system, North Carolina, and implications for assessment and management in a regime of increased frequency. Estuaries and Coasts 29, 1033-1045.

Pagliosa, P.R., 2005. Another diet of worms: the applicability of polychaete feeding guilds as a useful conceptual framework and biological variable. Mar. Ecol. 26, 246-254.

Papageorgiou, N., Arvanitidis, C., Eleftheriou, A., 2006. Multicausal environmental severity: A flexible framework for microtidal sandy beaches and the role of polychaetes as an indicator taxon. Estuar. Coast. Shelf Sci. 70, 643-653.

Parulekar, A., Dhargalkar, V., Singbal, S., 1980. Benthic Studies in Goa Estuaries. Part III-Annual Cycle of Macrofaunal Distribution, Production & Trophic Relations.

Parulekar, A., Dwivedi, S., 1974. Benthic studies in Goa estuaries. Part I–standing crop and faunal composition in relation to bottom salinity distribution and substratum characteristics in the estuary of Mandovi river. Indian Journal of Marine Science 3, 41-45.

Parulekar, A., Nair, S., Harkantra, S., Ansari, Z., 1976. Some quantitative studies on the benthos off Bombay. Mahasagar 9, 51-56.

Parulekar, A., Rajamanickam, G.V., Dwivedi, S., 1975. Benthic studies in Goa estuaries. Biomass and Faunal composition in the Zuari estuary. Indian J. Mar. Sci. 4, 202-205.

Parulekar, A., Wagh, A., 1975. Quantitative studies on benthic macrofauna of northeastern Arabian Sea shelf. Indian Journal of Marine Science 4, 174-176.

Paterson, D., Crawford, R., Little, C., 1990. Subaerial exposure and changes in the stability of intertidal estuarine sediments. Estuar. Coast. Shelf Sci. 30, 541-556.

Paul, N., Pillai, M.K., 1981. Dielectric studies of poly-p-toluidine films produced by the electrodeless glow discharge method. Thin Solid Films 76, 201-205.

Pauly, D., Watson, R., Alder, J., 2005. Global trends in world fisheries: impacts on marine ecosystems and food security. Philosophical Transactions of the Royal Society B: Biological Sciences 360, 5-12.

Pearson, T., Rosenberg, R., 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanogr. Mar. Biol. Ann. Rev 16, 229-311.

Pelegri, S.P., Nielsen, L.P., Blackburn, T.H., 1994. Corophium vo1utator. Mar. Ecol. Prog. Ser. 105, 285-290.

Peres, J.-M., 1967. The mediterranean benthos. Oceanogr. Mar. Biol. Annu. Rev.

Peterson, C.G.J., Boysen-Jensen, P., 1911. Valuation of the Sea: Animal Life of the Sea-bottom, Its Food and Quantity. Centraltrykkeriet.

Piersma, T., Koolhaas, A., Dekinga, A., Beukema, J.J., Dekker, R., Essink, K., 2001. Long-term indirect effects of mechanical cockle-dredging on intertidal bivalve stocks in the Wadden Sea J Appl Ecol 38:976-990

Pillai, N., 1977. Distribution and seasonal abundance of macrobenthos of the Cochin backwaters. Indian J. Mar. Sci. 6, 1-5.

Pinon-Gimate, A., Soto-Jimenez, M.F., Ochoa-Izaguirre, M.J., Garcia-Pages, E., Paez-Osuna, F., 2009. Macroalgae blooms and δ 15 N in subtropical coastal lagoons from the Southeastern Gulf of California: Discrimination among agricultural, shrimp farm and sewage effluents Mar Pollut Bull 58:1144-1151

Pfitzenmeyer, H.T., 1970. Benthos. Gross physical and biological effects of overboard spoil disposal in upper Chesapeake Bay. Univ. Md., Natural Resources Inst. Spec. Rep 3, 26-38.

Pocklington, P., Wells, P.G., 1992. Polychaetes key taxa for marine environmental quality monitoring. Mar. Pollut. Bull. 24, 593-598.

Pohle, G., Thomas, M., 2001. Monitoring protocol for marine benthos: Intertidal and subtidal macrofauna. Report for Atlantic Maritime Ecological Science Cooperative, St Andrews, New Brunswick.

Prato, E., Biandolino, F., 2006. Life history of the amphipod Corophium insidiosum (Crustacea: Amphipoda) from Mar Piccolo (Ionian Sea, Italy). Scientia marina 70, 355-362.

Qasim, S.Z., 2003. Indian estuaries. Allied publishers.

Qasim, S.Z., 1972. The dynamics of food and feeding habits of some marine fishes. Indian J. Fish. 19, 11-28.

Qasim, S.Z., Bhattathiri, P.M.A., Abidi, S.A.H., 1968. Solar radiation and its penetration in a tropical estuary J Exp Mar Biol Ecol 2:87-103.

Qasim, S., Gopinathan, C., 1969. Tidal cycle and the environmental features of Cochin Backwater (a tropical estuary), Proceedings of the Indian Academy of Sciences-Section B. Springer, pp. 336-348.

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

Qasim, S., Wellershaus, S., Bhattathiri, P., Abidi, S., 1969. Organic production in a tropical estuary, Proceedings of the Indian Academy of Sciences-Section B. Springer, pp. 51-94.

Queiroga, H., 1990. Corophium multisetosum (Amphipoda: Corophiidae) in Canal de Mira, Portugal: some factors that affect its distribution. Mar. Biol. 104, 397-402.

Quintino, V., Rodrigues, A., Gentil, F., 1989. Assessment of macrozoobenthic communities in the lagoon of Óbidos, western coast of Portugal. Sci. Mar. (Barc.).

Rajagopal, S., Van der Velde, G., Paffen, B., Van den Brink, F., Bij de Vaate, A., 1999. Life history and reproductive biology of the invasive amphipod Corophium curvispinum (Crustacea: Amphipoda) in the Lower Rhine. Archiv fur Hydrobiologie 144, 305-326.

Ramamirtham, C., Jayaraman, R., 1963. Some aspects of the hydrographical conditions of the backwaters around Willingdon Island (Cochin). J. Mar. Biol. Assoc. India 5, 170-177.

Ramamirtham, C., Muthusamy, S., 1986. Estuarine oceanography of the Vembanad lake Part II: The region between Cochin and Azhikode. Indian J. Fish. 33, 218-224.

Ramos-Gomez, J., Martin-Diaz, M., DelValls, T., 2009. Acute toxicity measured in the amphipod Ampelisca brevicornis after exposure to contaminated sediments from Spanish littoral. Ecotoxicology 18, 1068-1076.

Rao, K.K., Balasubramanian, T., 1996. Distribution of foraminifera in the Cochin estuary.

Rao, L.M., Shyamasundari, a.K., 1963. Tube-building habits of the fouling amphipod Corophium triaenonyx Stebbing at Visakhapatnam Harbour. J. zool. Soc. India 15, 133-140.

Rasheed, K., 1997. Studies on dredging impact assessment (DIA) at Cochin, a tropical estuarine harbour.

Remani, K., Devi, K.S., Venugopal, P., Unnithan, R., 1983. Indicator organisms of pollution in Cochin backwaters. Mahasagar 16, 199-207.

Revichandran, C., Srinivas, K., Muraleedharan, K., Rafeeq, M., Amaravayal, S., Vijayakumar, K., Jayalakshmy, K., 2012. Environmental set-up and tidal propagation in a tropical estuary with dual connection to the sea (SW Coast of India). Environmental Earth Sciences 66, 1031-1042.

Rhee, J.-S., Lee, Y.-M., Hwang, D.-S., Won, E.-J., Raisuddin, S., Shin, K.-H., Lee, J.-S., 2007. Molecular cloning, expression, biochemical characteristics, and biomarker potential of theta class glutathione S-transferase (GST-T) from the polychaete Neanthes succinea. Aquat. Toxicol. 83, 104-115.

Riba, I., DelValls, T.A., Forja, J., Gómez-Parra, A., 2003. Comparative toxicity of contaminated sediment from a mining spill using two amphipods species: Corophium volutator (Pallas, 1776) and Ampelisca brevicornis (A. Costa, 1853). Bull. Environ. Contam. Toxicol. 71, 1061-1068.

Richer, W., & Sarnthein, M. , 1977. Molluscan colonization of different submerged platforms in the western Baltic Sea. . In B. F. Keegen, P. O. Ceidigh & P. J. S. Boaden (Eds.), Biology of benthic organisms, New York: Pergamon., pp. 465–477.

Riemann, B., Hoffmann, E., 1991. Ecological consequences of dredging and bottom trawling in the Limfjord, Denmark Marine ecology progress series Oldendorf 69:171-178

Rivero, M.S., Vallarino, E.A., Elías, R., 2005. First survey of macroinfauna in the Mar del Plata Harbor (Argentina), and the use of polychaetes as pollution indicators. Rev. Biol. Mar. Oceanogr. 40, 101-108.

Robin, R.S., Muduli, P.R., Vardhan, K.V., Abhilash, K.R., Selvam, A.P., Kumar, B.C., Balasubramanian, T., 2012. Assessment of Hydrogeochemical

Characteristic in an Urbanized Estuary using Environmental Techniques Geosciences 2:81-92

Rosenberg , R., 1977. Effects of dredging operations on estuarine benthic macrofauna Mar Pollut Bull 8:102-104

Rossi, F., 2003. Short-term response of deposit-feeders to an increase of the nutritive value of the sediment through seasons in an intertidal mudflat (Western Mediterranean, Italy). J. Exp. Mar. Biol. Ecol. 290, 1-17.

Rouse, G., Pleijel, F., 2001. Polychaetes. Oxford university press.

Rygg, B., 1985. Distribution of species along pollution-induced diversity gradients in benthic communities in Norwegian fjords. Mar. Pollut. Bull. 16, 469-474.

Sanchez-Moyano, J.E., Garcia-Asencio, I., 2009. Distribution and trophic structure of annelid assemblages in a Caulerpa prolifera bed from southern Spain. Mar. Biol. Res. 5, 122-132.

Sanders, H., Mangelsdorf, P., Hampson, G., 1965. Salinity and faunal distribution in the Pocasset River, Massachusetts. Limnol. Oceanogr. 10.

Sanilkumar, M., Saramma, A., Joseph, K., 2009. Vertical distribution of microphytobenthos in Cochin estuary. J. Mar. Biol. Assoc. India 51, 61-68.

Sanilkumar, M., Saramma, A., Joseph, K., Cahoon, L., 2011. Monsoon Effects on Biomass and Composition of Microphytobenthic Diatoms in the Cochin Estuary, Southwest India. Journal of North Carolina Academy of Science 127, 1-12.

Sankaranarayanan, V., Panampunnayil, S., 1979. Studies on organic carbon, nitrogen and phosphorus in sediments of the Cochin Backwater. Indian. J. Mar. Sci 8, 27-30.

Sankaranarayanan, V., Qasim, S., 1968. The influence of some hydrographical factors on the fisheries of the Cochin area Bulletin of the National Institute of Sciences of India 38:846-853

Sankaranarayanan, V., Qasim, S., 1969. Nutrients of the Cochin backwater in relation to environmental characteristics. Mar. Biol. 2, 236-247.

Santos, C.D., Granadeiro, J.P., Palmeirim, J.M., 2005. Feeding ecology of dunlin Calidris alpina in a southern European estuary. Ardeola 52, 235-252.

Saraladevi, K., 1986. Effect of industrial pollution on the benthic communities of the estuary. Cochin Univ. Science and Technology, 386.

Sarda, R., Pinedo, S., Gremare, A., Taboada, S., 2000. Changes in the dynamics of shallow sandy-bottom assemblages due to sand extraction in the Catalan Western Mediterranean Sea. ICES J. Mar. Sci. 57, 1446-1453.

Sarma, V., Kumar, N., Prasad, V., Venkataramana, V., Appalanaidu, S., Sridevi, B., Kumar, B., Bharati, M., Subbaiah, C.V., Acharyya, T., 2011. High CO2 emissions from the tropical Godavari estuary (India) associated with monsoon river discharges. Geophys. Res. Lett. 38.

Sarma, V.V.S.S., Gupta, S.N.M., Babu, P.V.R., Acharya, T., Harikrishnachari, N., Vishnuvardhan, K., Rao, N.S., Reddy, N.P.C., Sarma, V.V., Sadhuram, Y. and Murty, T.V.R., 2009. Influence of river discharge on plankton metabolic rates in the tropical monsoon driven Godavari estuary, India. Estuarine, Coastal and Shelf Science, 85(4), pp.515-524. Satyanarayana Murthy, P., Rao K S Geological society of India, 13, 1959, 38.

Schlekat, C.E., Mcgee, B.L., Reinharz, E., 1992. Testing sediment toxicity in Chesapeake Bay with the amphipod Leptocheirus plumulosus: an evaluation. Environ. Toxicol. Chem. 11, 225-236.

Seshappa, G., 1953. Observations on the physical and biological features of the inshore sea bottom along the Malabar coast, Proc. Nat. Inst. Sci. India, pp. 257-279.

Sheeba, P., 2000. Distribution of Benthic In fauna in the Cochin Backwaters In Relation To Environmental Parameters. Cochin University of Science and Technology, Kochi.

Sheeba, P., Jayalakshmy, K., Devi, S., 2004. Bottom fauna of dredging and dredge spoil disposal sites of a tropical estuary. International Journal of Ecology and Environmental Sciences 30, 225-238p.

Sherk, J.A., 1971. Effects of suspended and deposited sediments on estuarine organisms. Chesapeake Biological Laboratory, University of Maryland Contribution No. 443.

Shillaker, R., Moore, P.G., 1987. The biology of brooding in the amphipods Lembos websteri Bate and Corophium bonnellii Milne Edwards. J. Exp. Mar. Biol. Ecol. 110, 113-132.

Shivaprasad, A., Vinita, J., Revichandran, C., Reny, P.D., Deepak, M.P., Muraleedharan, K. R., Naveen Kumar, K. R., 2013. Seasonal stratification and property distributions in a tropical estuary (Cochin estuary, west coast, India). Hydrol. Earth Syst. Sci. 17, 187-199.

Shou, L., Huang, Y., Zeng, J., Gao, A., Liao, Y., Chen, Q., 2009. Seasonal changes of macrobenthos distribution and diversity in Zhoushan sea area. Aquat. Ecosyst. Health Manage. 12, 110-115.

Shyamasundari, K., 1972. Studies on the tube building Amphipod corophium triaenonyx Stebbing from Visakhapatnam harbour. Annual life cycle. Rivista di biologia 65, 203.

Shyamasundari, K., 1973. Studies on the tube-building amphipod Corophium triaenonyx stebbing from Visakhapatnam harbor: effect of salinity and temperature. The Biological Bulletin 144, 503-510.

Shyamasundari, K., 1976. Effects of salinity and temperature on the development of eggs in the tube building amphipod Corophium triaenonyx Stebbing. The Biological Bulletin 150, 286-293.

Simboura, N., Nicolaidou, A., Thessalou-Legaki, M., 2000. Polychaete communities of Greece: an ecological overview. Mar. Ecol. 21, 129-144.

Simboura, N., Zenetos, A., Panayotidis, P., Makra, A., 1995. Changes in benthic community structure along an environmental pollution gradient. Mar. Pollut. Bull. 30, 470-474.

Simonini, R., Ansaloni, I., Bonini, P., Grandi, V., Graziosi, F., Iotti, M., Massamba-N'Siala, G., Mauri, M., Montanari, G., Preti, M., 2007. Recolonization and recovery dynamics of the macrozoobenthos after sand extraction in relict sand bottoms of the Northern Adriatic Sea. Mar. Environ. Res. 64, 574-589.

Simon, J.L., Conner, W.G., 1977. Response of a subtropical soft-bottom community to pertubation of oyster shell dredging, Am. Zool. Amer soc zoologists 1041 New Hampshire ST, Lawrence, ks 66044, pp. 907-907.

Sivadas, S., Ingole, B., Nanajkar, M., 2011. Temporal variability of macrofauna from a disturbed habitat in Zuari estuary, west coast of India. Environ. Monit. Assess. 173, 65-78.

Sivadasan, K., Joseph, K., 1995. Photosynthetic pigments of benthic microflora in Cochin estuary. Indian J. Mar. Sci. 24, 231-232.

Sivadasan, K., Joseph, K., 1997. Distribution and role of benthic micro algae in Cochin estuary. J. Mar. Biol. Assoc. India 39, 27-32.

Sivaprakasam, T., 1968. Amphipoda from the east coast of India. Part 1. Gammaridea. J. Mar. Biol. Assoc. India 8, 82-122.

Smith, D., Hughes, R., Cox, E.J., 1996. Predation of epipelic diatoms by the amphipod Corophium volutator and the polychaete Nereis diversicolor. Mar. Ecol. Prog. Ser. 145, 53-61.

Smith, G.C., Covich, A.P., Brasher, A.M., 2003. An ecological perspective on the biodiversity of tropical island streams. Bioscience 53, 1048-1051.

Smith, K., Baldwin, R., 1982. Scavenging deep-sea amphipods: effects of food odor on oxygen consumption and a proposed metabolic strategy. Mar. Biol. 68, 287-298.

Smith, S., Hollibaugh, J., 1993. Coastal metabolism and the oceanic organic carbon balance. Rev. Geophys. 31, 75-89.

Snelgrove, P.V.R., 1998. The biodiversity of macrofaunal organisms in marine sediments. Biodiversity & Conservation 7, 1123-1132.

Solis-Weiss, V., Aleffi, F., Bettoso, N., Rossin, P., Orel, G., Fonda-Umani, S., 2004. Effects of industrial and urban pollution on the benthic macrofauna in the Bay of Muggia (industrial port of Trieste, Italy). Sci. Total Environ. 328, 247-263.

Somerfield, P., Gage, J., 2000. Community structure of the benthos in Scottish Sea-lochs. IV. Multivariate spatial pattern. Mar. Biol. 136, 1133-1145.

Srinivas, K., Revichandran, C., Maheswaran, P., Asharaf, T.M., Murukesh, N., 2003. Propagation of tides in the Cochin estuarine system, southwest coast of India.

Steele, D.H., Steele, V.J., 1991. Effects of salinity on the survival, growth rate, and reproductive output of Gammarus lawrencianus (Crustacea, Amphipoda). Mar. Ecol. Prog. Ser. 78, 49-56.

Stevens, M.I., Hogg, I.D., Chapman, M.A., 2002. The corophiid amphipods of Tauranga Harbour, New Zealand: evidence of an Australian crustacean invader. Hydrobiologia 474, 147-154.

Sujatha, C., Nair, S., Kumar, N., Chacko, J., 1993. Distribution of organochlorine pesticides in a tropical waterway: HCH isomers. Toxicological & Environmental Chemistry 39, 103-111.

Sukumaran, S., Saraladevi, K., 2009. Polychaete diversity and its relevance in the rapid environmental assessment of Mumbai Port.

Sunil Kumar, R., 1993. Studies on the benthic fauna of the mangrove swamps of Cochin area, Cochin University of Science and Technology, kochi.

Sutton, G., Boyd, S., Augris, C., Bonne, W., Carlin, D., Cato, I., Cowling, M., van Dalfsen, J., Delpech, J.-P., Desprez, M., 2009. Effects of extraction of marine sediments on the marine environment 1998-2004. International Council for the Exploration of the Sea.

Sverdrup, H.U., Johnson, M.W., Fleming, R.H., 1942. The Oceans: Their physics, chemistry, and general biology. Prentice-Hall New York.

Taupp, T., Wetzel, M.A., 2013. Relocation of dredged material in estuaries under the aspect of theWater Framework Directive-A comparison of benthic quality indicators at dumping areas in the Elbe estuary. Ecol. Indicators 34, 323–331.

Teichberg, M., Fox, S.E., Olsen, Y.S., Valiela, I., Martinetto, P., Iribarne, O., Muto, E.Y., Petti, M.A., Corbisier, T.N., Soto-Jimenez, M.a.r.t.í.n. and Páez-osuna, F.e.d.e.r.i.c.o., 2010. Eutrophication and macroalgal blooms in temperate and tropical coastal waters: nutrient enrichment experiments with Ulva spp. Global Change Biology, 16(9), pp.2624-2637.

Thrush, S.F., Dayton, P.K., 2002. Disturbance to marine benthic habitats by trawling and dredging: implications for marine biodiversity. Annu. Rev. Ecol. Syst.33 (1), 449-473.

Thrush, S.F., Hewitt, J.E., Gibbs, M., Lundquist, C., Norkko, A., 2006. Functional role of large organisms in intertidal communities: community effects and ecosystem function. Ecosystems 9, 1029-1040.

Thomson, K., 2002. Economic and social issues of biodiversity loss in Cochin backwaters, Technical report. Cochin University of Science and Technology Cochin, India, pp. 51-82.

Timmermann, K., Andersen, O., 2003. Bioavailability of pyrene to the deposit-feeding polychaete Arenicola marina: importance of sediment versus water uptake routes. Mar. Ecol. Prog. Ser. 246, 163-172.

Ulanowicz, R.E., 1997. Ecology, the ascendent perspective: Robert E. Ulanowicz. Columbia University Press.

Underwood, G.J., Paterson, D.M., 1993. Seasonal changes in diatom biomass, sediment stability and biogenic stabilization in the Severn Estuary. J. Mar. Biol. Assoc. U.K. 73, 871-887.

Unnithan, R., Vijayan, M., Remani, K., 1975. Organic pollution in Cochin backwaters. Indian J. Mar. Sci. 4, 39-42.

Van Dalfsen, J.A., Essink, K., Madsen, H.T., Birklund, J., Romero, J., Manzanera, M., 2000. Differential response of macrozoobenthos to marine sand extraction in the North Sea and the Western Mediterranean. ICES Journal of Marine Science: Journal du Conseil 57, 1439-1445.

Van den Brink, F., Van der Velde, G., Bij de Vaate, A., 1993. Ecological aspects, explosive range extension and impact of a mass invader, Corophium curvispinum Sars, 1895 (Crustacea: Amphipoda), in the Lower Rhine (The Netherlands). Oecologia 93, 224-232.

Van der Veer, H.W., Bergman, M.J.N., Beukema, J.J., 1985. Dredging activities in the Dutch Wadden Sea: effects on macrobenthic infauna. Netherlands Journal of Sea Research 19, 183-190.

Van Dolah, R.F., Calder, D.R., Knott, D.M., 1984. Effects of dredging and open-water disposal on benthic macroinvertebrates in a South Carolina estuary. Estuaries 7, 28-37.

Van Hoey, G., Degraer, S., Vincx, M., 2004. Macrobenthic community structure of soft-bottom sediments at the Belgian Continental Shelf. Estuar. Coast. Shelf Sci. 59, 599-613.

Vanosmael, C., Willems, K., Claeys, D., Vincx, M., Heip, C., 1982. Macrobenthos of a sublittoral sandbank in the Southern Bight of the North Sea. J. Mar. Biol. Assoc. U.K. 62, 521-534.

Veerayya, M., Murty, P., 1974. Studies on the sediments of Vembanad Lake, Kerala State Part III Distribution and interpretation of bottom sediments. Indian Journal Marine Sciences 3, 16-27.

Velamala, S.N., Thomas, J., Bari, S., Kachave, S., 2016. The impact of dredging on residence time in the Amba estuary, west coast of India Environmental Earth Sciences 75:108

Venkatramanan, S., Ramkumar, T., Anithamary, I., 2013. Distribution of grain size, clay mineralogy and organic matter of surface sediments from Tirumalairajanar Estuary, Tamilnadu, east coast of India. Arabian Journal of Geosciences 6, 1371-1380.

Venugopal, P., Devi, K.S., Remani, K., Unnithan, R., 1982. Trace metal levels in the sediments of the Cochin backwaters. Mahasagar 15, 205-214.

Vijith, V., Sundar, D., Shetye, S., 2009. Time-dependence of salinity in monsoonal estuaries. Estuar. Coast. Shelf Sci. 85, 601-608.

Vineetha, G., Madhu, N., Kusum, K., Sooria, P., 2015. Seasonal dynamics of the copepod community in a tropical monsoonal estuary and the role of sex ratio in their abundance pattern. Zool. Stud. 54, 1-19.

Warwick, R., 2001. Evidence for the effects of metal contamination on the intertidal macrobenthic assemblages of the Fal Estuary. Mar. Pollut. Bull. 42, 145-148.

Watkin, E.E., 1941. The yearly life cycle of the amphipod, Corophium volutator. The Journal of Animal Ecology, 77-93.

Waye-Barker, G.A., McIlwaine, P., Lozach, S., Cooper, K.M., 2015. The effects of marine sand and gravel extraction on the sediment composition and macrofaunal community of a commercial dredging site (15 years post-dredging). Mar. Pollut. Bull. 99, 207-215.

Wellershaus, S., 1974. Seasonal changes in the zooplankton population in the Cochin Backwater (a South Indian estuary). Aquat. Ecol. 8, 213-223.

Wetlands, 2002. Ramsar sites for vulnerable wetlands http://www.wetlands.org

Wetzel, R.G., 2001. Limnology: lake and river ecosystems. Gulf Professional Publishing.

Whitlatch, R.B., 1981. Animal-sediment relationships in intertidal marine benthic habitats: some determinants of deposit-feeding species diversity. J. Exp. Mar. Biol. Ecol. 53, 31-45.

Whomersly P, Ware S, Rees HL, Mason C, Bolam T, Huxham M, H, B., , 2008. Biological indicators of disturbance at a dredged-material disposal site in Liverpool Bay, UK: an assessment using time-series data. ICES J Mar Sci 65, 1414-1420.

Wieking, G., Kroncke, I., 2003. Macrofauna communities of the Dogger Bank (central North Sea) in the late 1990s: spatial distribution, species composition and trophic structure. Helgol. Mar. Res. 57, 34.

Wilson, W.H., 1989. Predation and the mediation of intraspecific competition in an infaunal community in the Bay of Fundy. J. Exp. Mar. Biol. Ecol. 132, 221-245.

Wilson, W.H., 1991. Sexual reproductive modes in polychaetes: classification and diversity. Bull. Mar. Sci. 48, 500-516.

Wilson, W.H., 1994. The effects of episodic predation by migratory shorebirds in Grays Harbor, Washington. J. Exp. Mar. Biol. Ecol. 177, 15-25.

Wilson, W.H., Parker, K., 1996. The life history of the amphipod, Corophium volutator: the effects of temperature and shorebird predation. J. Exp. Mar. Biol. Ecol. 196, 239-250.

Winberg, P.C., Lynch, T.P., Murray, A., Jones, A.R., Davis, A.R., 2007. The importance of spatial scale for the conservation of tidal flat macrobenthos: an example from New South Wales, Australia. Biol. Conserv. 134, 310-320.

Withers, R., Thorp, C., 1978. The macrobenthos inhabiting sandbanks in Langstone Harbour, Hampshire. J. Nat. Hist. 12, 445-455.

Wongkamhaeng, K., Nabhitabhata, J., Towatana, P., 2015. Corophiine amphipods of the genera Chelicorophium and Paracorophium from the lower Gulf of Thailand (Crustacea, Amphipoda, Corophiidae, Corophiinae). ZooKeys, 35.

Wulff, A., Sundbäck, K., Nilsson, C., Carlson, L., Jönsson, B., 1997. Effect of sediment load on the microbenthic community of a shallow-water sandy sediment. Estuaries 20, 547-558.

Yeager, K.M., Brinkmeyer, R., Rakocinski, C.F., Schindler, K.J., Santschi, P.H., 2010. Impacts of dredging activities on the accumulation of dioxins in surface sediments of the Houston Ship Channel, Texas. J. Coast. Res.26, 743-752.

Yuan, R., Zhu, J., 2015. The effects of dredging on tidal range and saltwater intrusion in the Pearl River estuary J Coast Res 31:1357-1362.

Zajac, R.N., Whitlatch, R.B., 1982. Responses of estuarine infauna to disturbance. I. Spatial and temporal variation of initial recolonization. Mar. Ecol. Prog. Ser. 10,, 1-14.

Zhu, J., Weisberg, R.H., Zheng, L., Han, S., 2015. Influences of channel deepening and widening on the tidal and nontidal circulations of Tampa Bay Estuaries and coasts 38:132-150.

*****$$*****

- **T V Rehitha,** N Ullas, G Vineetha, P Y Benny, N V Madhu, C Revichandran, 2017. Impact of maintenance dredging on macrobenthic community structure of a tropical estuary. Ocean and Coastal Management, (Elsevier) 144, 71-82.

- N V Madhu, N Ullas, R Ashwini, Paul Meenu, **T V Rehitha**, K R Lallu, 2014. Characterization of phytoplankton pigments and functional community structure in the Gulf of Mannar and the Palk Bay using HPLC-CHEMTAX analysis. Continental Shelf Research, (Elsevier) 80 79-90.

- K R Lallu, K H Fausia, J Vinita, K K Balachandran, K R Naveen Kumar & **T V Rehitha,** 2014. Transport of dissolved nutrients and chlorophyll a in a tropical estuary, southwest coast of India. Environmental Monitoring Assessment (Springer): 186, 4829-4839.

- N V Madhu, Paul Meenu, N Ullas, R Ashwini, **T V Rehitha**, 2013. Occurence of cyanobacteria (*Richelia intracellularis*)-diatom (*Rhizosolenia hebetata*) consortium in the Palk Bay, Southeast coast of India. Indian Journal of Geo Marine Sciences 42(4), 453-457.

PAPER PRESENTATONS

- **T V Rehitha**, N V Madhu, R Reshmi, G Vijay John, C Revichandran. Seasonality in the distribution and abundance of macrobenthic fauna in the Cochin estuary and adjacent coastal shelf, **Osicon -2011**.

- **T V Rehitha**, P Reshmi, J G Vijay, M P Deepak and N V Madhu, 2010. Sediment Characteristics and Macro Benthic Fauna of the Cochin Estuary and Coastal Waters during Pre-Monsoon. **Proceedings of National Conference on coastal processes, resources & Management.** Thiruvananthapuram,

Ocean & Coastal Management 144 (2017) 71–82

Contents lists available at ScienceDirect

Ocean & Coastal Management

journal homepage: www.elsevier.com/locate/ocecoaman

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

T.V. Rehitha*, N. Ullas, G. Vineetha, P.Y. Benny, N.V. Madhu, C. Revichandran

CSIR-National Institute of Oceanography, Regional Centre, Cochin, 682 018, India

A R T I C L E I N F O

Article history:
Received 19 December 2016
Received in revised form
29 April 2017
Accepted 29 April 2017

Keywords:
Macrobenthos
Feeding guild
Dredging impact
Cochin estuary

A B S T R A C T

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.

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,

* Corresponding author.
E-mail address: rehitarajan@gmail.com (T.V. Rehitha).

http://dx.doi.org/10.1016/j.ocecoaman.2017.04.020

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 × 10^6 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

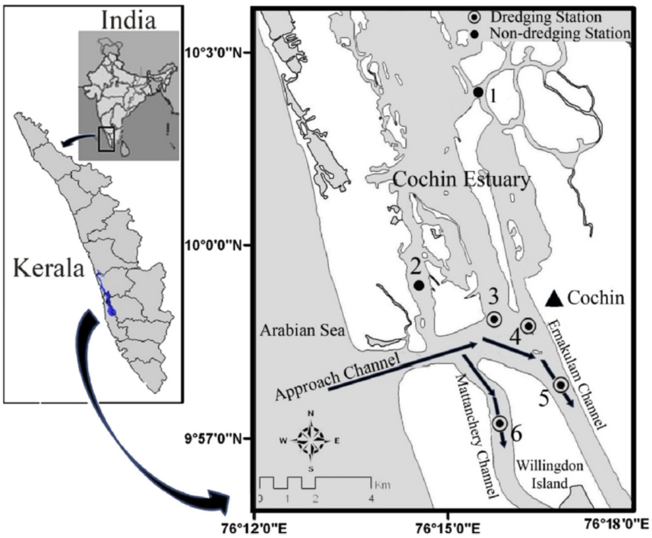


Fig. 1. Map of the Cochin estuary showing sampling locations.

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 m^2). 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	s.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.

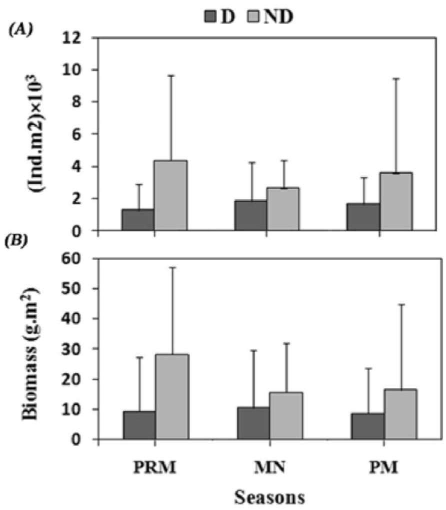


Fig. 2. Seasonal distribution of macrobenthic density (A) and biomass (B) in dredging (D) and non-dredging (ND) sites of the Cochin estuary (PRM-pre-monsoon; MN-monsoon; PM-post-monsoon).

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, Nephtydae, 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%), *Prionospio 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).

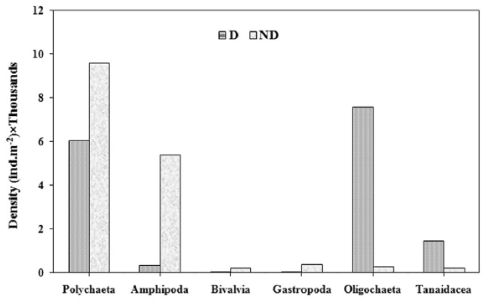


Fig. 3. Macrobenthic community composition in dredging (D) and non-dredging sites (ND) of the Cochin estuary.

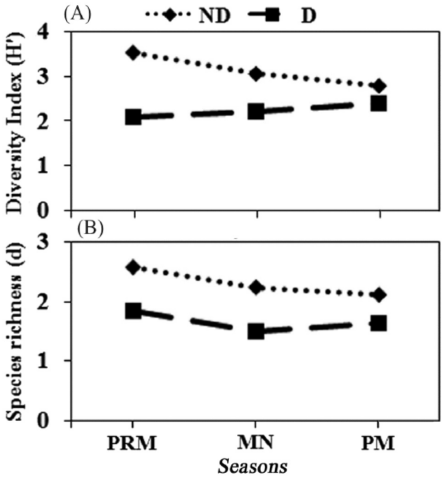


Fig. 4. Seasonal variation in (A) species diversity (H') and (B) species richness (d) in the dredging and non-dredging sites of the Cochin estuary (PRM-pre-monsoon, MN-monsoon, PM-post-monsoon).

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).

76 T.V. Rehitha et al. / Ocean & Coastal Management 144 (2017) 71–82

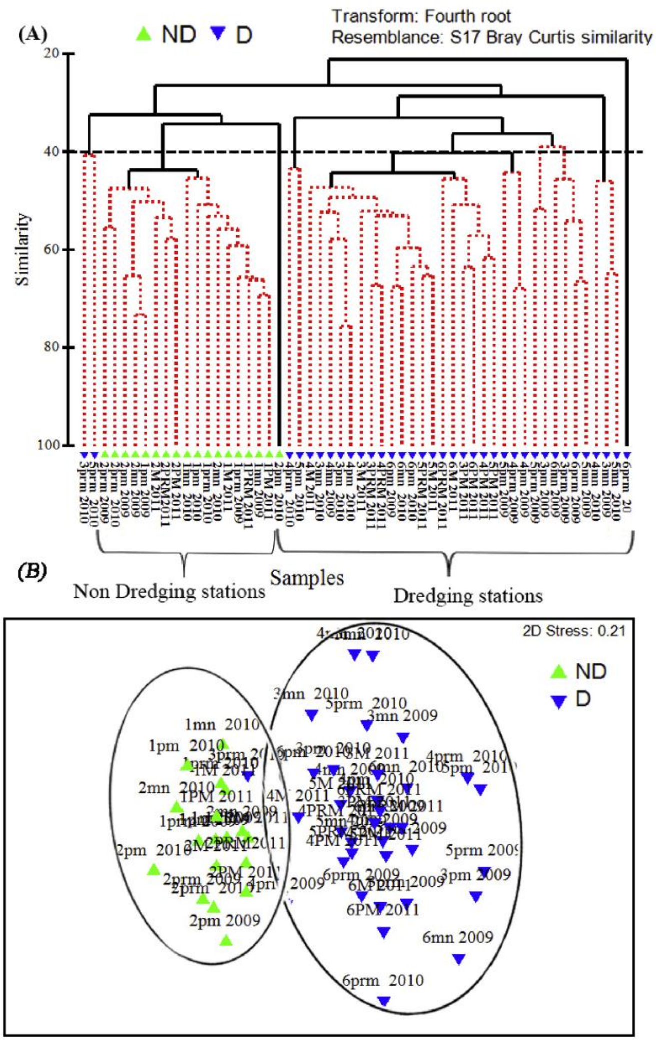


Fig. 5. Bray-curtis similarity based on hierarchical clustering of sampling stations depicted through (A) dendrogram and (B) NMDS (non-metric multidimensional scaling) showing macrobenthic assemblage pattern in dredging and non-dredging sites of the Cochin estuary.

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%	*Mediomastus capensis*	4.04	6.3	13.53
	Prionospio cirrifera	3.24	4.57	9.83
Group I-ND	*Paraheteromastus tenuis*	2.96	4.43	9.52
	Caprella sp	2.92	4.27	9.19
	Dendronereis estuarina	2.51	3.62	7.77
	Littorina littorea	1.91	2.44	5.25
	Photis digitata	1.91	2.26	4.85
	Tubificidae sp	1.72	1.77	3.81
	Bivalvia sp	1.67	1.71	3.68
	Capitella capitata	1.54	1.66	3.57
	Eriopisa chilkensis	1.67	1.53	3.29
	Apseudus chilkensis	1.55	1.5	3.21
	Diopatra neopolitana	2.1	1.37	2.95
	Cheriophotis megacheles	1.25	1	2.14
	Coropium triaenonyx	1.25	0.81	1.75
	Owenia fusiformis	1.11	0.69	1.49
Sim: 37.64%	*Tubificidae* sp	3.47	6.57	17.46
Group II-D	*Mediomastus capensis*	2.46	5.43	14.43
	Sygambra parva	2.39	5.09	13.52
	Nepthys oligobranchia	2.09	4.34	11.54
	Prionospio cirrifera	1.86	3.77	10
	Apseudus chilkensis	1.93	3.04	8.09
	Cirratulus filiformis	1.56	2.29	6.08
	Paraheteromastus tenuis	1	0.87	2.3
	Cossura coasta	0.9	0.75	2
	Cirratulus cirratus	0.81	0.74	1.98
	Decapod sp	0.67	0.65	1.74
	Bivalvia 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		
Caprella sp	2.92	0	3.56	2.13
Tubificidae sp	1.72	3.47	3.27	1.25
Paraheteromastus tenuis	2.96	1	3.02	1.33
Dendronereis estuarina	2.51	0.39	2.77	1.67
Mediomastus capensis	4.04	2.46	2.66	1.26
Sigambra parva	0.55	2.39	2.6	1.43
Prionospio cirrifera	3.24	1.86	2.55	1.26
Photis digitata	1.91	0.2	2.54	1.12
Nephtys oligobranchia	0.12	2.09	2.51	1.44
Diopatra neapolitana	2.1	0.46	2.5	0.98
Littorina littorea	1.91	0.12	2.3	1.46
Apseudus chilkensis	1.55	1.93	2.18	1.09
Eriopisa chilkensis	1.67	0.42	2.01	1.07
Cirratulus filiformis	0.22	1.56	1.91	1
Bivalvia sp	1.67	0.69	1.85	1.19
Capitella capitata	1.54	0.58	1.84	1.1
Cheriophotis megacheles	1.25	0	1.57	0.9
Cirratulus cirratus	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

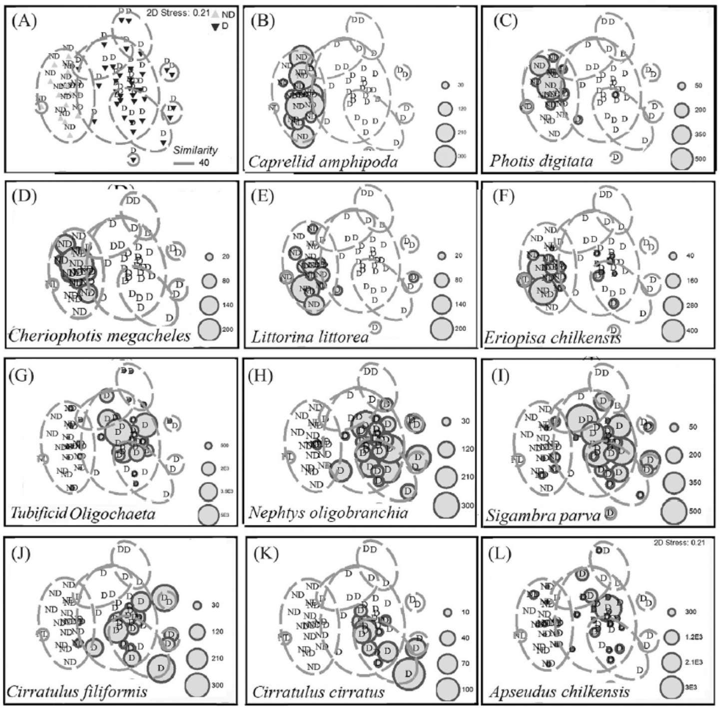


Fig. 6. Species responsible for the distinct difference between dredging and non-dredging areas identified using SIMPER analysis (A) nMDS plot of faunal distribution of macrobenthos, overlaid with bubbles indicating density of major discriminating species (B) Caprellid amphipod, (C) *Photis digitata*,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

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; Venkatramanan 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 chilkensis* 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

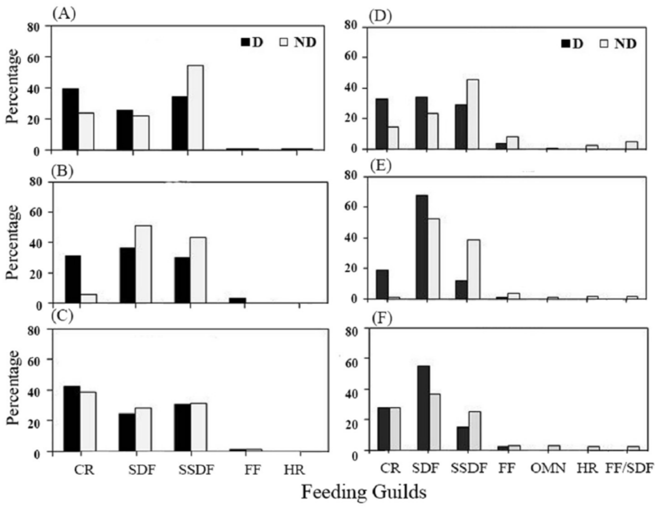


Fig. 7. Seasonal distribution of polychaete feeding guilds (A, B, C) and that of the characterizing species (D, E, F) in dredging and non-dredging areas of the CE during pre-monsoon (A&D), monsoon (B&E) and post-monsoon (C&F) period respectively (CR-Carnivores, SDF-surface deposit feeders, SSDF-sub-surface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

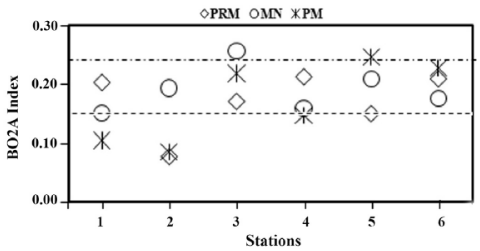


Fig. 8. Seasonal variation of BO2A index in Cochin estuary (index value > 0.15 to ≤0.24 indicates moderate, index>0.24 indicates poor condition of water body) (PRM-pre-monsoon; MN-monsoon; PM-post-monsoon).

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 *Cheriophotis megacheles*, gammarid amphipod *Eriopisa chilkensis*, 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 *Apseudus chilkensis* 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 oilgochaetes 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.

Acknowledgements

The authors are grateful to Dr. S W A Naqvi, Director, CSIR-National Institute of Oceanography (NIO), Goa and Dr. T Pankajakshan, Scientist-in-Charge, Regional Centre, CSIR-NIO, Kochi for the facilities provided. The study was supported by the project SSP-2179 (CSIR-NIO), funded by Cochin Port Trust (CPT), Cochin. This is NIO contribution number 6042.

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.

References

Abdul Jaleel, K., Kumar, P.A., Khan, K.N., Correya, N.S., Jacob, J., Philip, R., Sanjeevan, V., Damodaran, R., 2014. Polychaete community structure in the south eastern Arabian sea continental margin (200–1000m). Deep. Sea Res. Pt I 93, 60–71.

Abdul Jaleel, K., Parameswaran, U.V., Gopal, A., Khader, C., Ganesh, T., Sanjeevan, V., Shunmugaraj, T., Vijayan, A.K., Gupta, G., 2015. Evaluation of changes in macrobenthic standing stock and polychaete community structure along the south eastern Arabian Sea shelf during the monsoon trawl-ban. Cont. Shelf Res. 102, 9–18.

APHA, AWWA, WEF, 2005. Standard Methods for the Examination of Water and Wastewater, 21st ed. American Public Health Association, Washington, D.C.

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

Balchand, A.N., Rasheed, K., 2000. Assessment of short term environmental impacts on dredging in a tropical estuary. Terra Aqua 79, 16–26.

Barletta, M., Cysneiros, F., Lima, A., 2016. Effects of dredging operations on the demersal fish fauna of a South American tropical–subtropical transition estuary. J. Fish Biol. 89, 890–920.

Bemvenuti, C.E., Angonesi, L.G., Gandra, M.S., 2005. Effects of dredging operations on soft bottom macrofauna in a harbor in the Patos Lagoon estuarine region of southern Brazil. Braz. J. Biol. 65, 573–581.

Berlow, E.L., Navarrete, S.A., 1997. Spatial and temporal variation in rocky intertidal community organization: lessons from repeating field experiments. J. Exp. Mar. Biol. Ecol. 214, 195–229.

Besser, J.M., Giesy, J.P., Kubitz, J.A., Verbrugge, D.A., Coon, T.G., Braselton, W.E., 1996. Assessment of sediment quality in dredged and undredged areas of the Trenton Channel of the Detroit River, Michigan USA, using the sediment quality triad. J. Gt. Lakes Res. 22, 683–696.

Birkett, L., McIntyre, A.D., 1971. Treatment and Sorting of Samples. Methods for the Study of Marine Benthos. IBP Handbook, pp. 156–168.

Bolam, S.G., Rees, H.L., 2003. Minimizing impacts of maintenance dredged material disposal in the coastal environment: a habitat approach. Environ. Manage. 32, 171–188.

Borja, Á., Dauer, D.M., Elliott, M., Simenstad, C.A., 2010. Medium-and long-term recovery of estuarine and coastal ecosystems: patterns, rates and restoration effectiveness. Estuar. Coast 33, 1249–1260.

Boyd, S.E., Limpenny, D.S., Rees, H.L., Cooper, K.M., Campbell, S., 2003. Preliminary observations of the effects of dredging intensity on the re-colonisation of dredged sediments off the southeast coast of England (Area 222). Estuar. Coast. Shelf Sci. 57, 209–223.

Brown, B.E., Kumar, A.J., 1990. Temporal and spatial variations in iron concentrations of tropical bivalves during a dredging event. Mar. Pollut. Bull. 21, 118–123.

Caine, E.A., 1977. Feeding mechanisms and possible resource partitioning of the Caprellidae (Crustacea: Amphipoda) from Puget Sound, USA. Mar. Biol. 42, 331–336.

Caspers, H., 1973. RO Brinkhurst and GBM Jamieson: aquatic Oligochaeta of the World.—860 pp. Edinburgh: Oliver & Boyd. Int. Rev. Gesamten Hydrobiol. Hydrogr. 58, 147.

Ceia, F.R., Patrício, J., Franco, J., Pinto, R., Fernández-Boo, S., Losi, V., Marques, J.C., Neto, J.M., 2013. Assessment of estuarine macrobenthic assemblages and ecological quality status at a dredging site in a southern Europe estuary. Ocean. Coast. Manage. 72, 80–92.

Clarke, K.R., Gorley, R.N., 2006. User Manual/Tutorial. Primer-E Ltd., Plymouth, p. 93.

Clarke, K.R., Warwick, R.M., 2001. An approach to statistical analysis and interpretation. Change Mar. Communities 2, 75–76.

Conner, W.G., Simon, J.L., 1979. The effects of oyster shell dredging on an estuarine benthic community. Estuar. Coast Mar. Sci. 9, 749–758.

Cooper, J.A.G., 2003. Anthropogenic Impacts on Estuaries. Coastal Zones, Estuaries. Encyclopedia of Life Support Systems (EOLSS). UNESCO, EOLSS Publishers, Oxford, UK, 246 pp.

Dauvin, J.C., Ruellet, T., 2009. The estuarine quality paradox: is it possible to define an ecological quality status for specific modified and naturally stressed estuarine ecosystems? Mar. Pollut. Bull. 59, 38–47.

Day, J.H., 1967. A Monograph on the Polychaeta of Southern Africa,Part I, Errantia and II, Sedentaria. Natural History. Trustees of the British museum., London, 878 pp.

Day, J.W., 1989. Estuarine Ecology. John Wiley & Sons, pp. 5–7.

de Jonge, V.N., 1983. Relations between annual dredging activities, suspended matter concentrations, and the development of the tidal regime in the Ems estuary. Can. J. Fish. Aquat. Sci. 40, 289–300.

de Jonge, V.N., 2000. Importance of temporal and spatial scales in applying biological and physical process knowledge in coastal management, an example for the Ems estuary. Cont. Shelf Res. 20, 1655–1686.

de Jonge, V.N., Schuttelaars, H.M., van Beusekom, J.E., Talke, S.A., de Swart, H.E., 2014. The influence of channel deepening on estuarine turbidity levels and dynamics, as exemplified by the Ems estuary. Estuar. Coast. Shelf Sci. 139, 46–59.

Desprez, M., 2000. Physical and biological impact of marine aggregate extraction along the French coast of the Eastern English Channel: short-and long-term post-dredging restoration. ICES J. Mar. Sci. 57, 1428–1438.

Devi, K.S., Jayalakshmy, K., Venugopal, P., 1991. Communities and coexistence of benthos in northern limb of Cochin backwaters. Indian J. Geo-Mar. Sci. 20, 249–254.

El Wakeel, S.K., Riley, J.P., 1957. The determination of organic carbon in marine muds. J. Cons. Int. Explor. 22, 180–183.

Fauchald, K., 1977. The Polychaete Worms: Definitions and Keys to the Orders, Families and Genera. Science Series, 28 pp.

Fauchald, K., Jumars, P.A., 1979. The Diet of Worms: a Study of Polychaete Feeding Guilds. Aberdeen University Press.

Fauvel, P., 1953. The Fauna of India (Annelida, Polychaeta). Indian Press Ltd, Allahabad, 507 pp.

Flemming, H.C., Schmitt, J., Marshall, K.C., 1996. Sorption Properties of Biofilms, Sediments and Toxic Substances. Springer, pp. 115–157.

Forbes, T.L., Forbes, V.E., Depledge, M.H., 1994. Individual physiological responses to environmental hypoxia and organic enrichment: implications for early soft-bottom community succession. J. Mar. Res. 52 (6), 1081–1100. http://dx.doi.org/10.1357/0022240943076849.

Giangrande, A., 1997. Polychaete reproductive patterns, life cycles and life histories: an overview. Oceanogr. Mar. Biol. 35, 323–386.

Giere, O., Pfannkuche, O., 1982. Biology and ecology of marine Oligochaeta, a review. Oceanogr. Mar. Biol. 20, 173–309.

Gopinathan, C., Qasim, S., 1971. Silting in navigational channels of the Cochin harbour area. J. Mar. Biol. Assoc. India 13, 14–26.

Gosner, K.L., 1971. Guide to Identification of Marine and Estuarine Invertebrates. Wiley-Interscience, John Wiley and Sons, Inc., New York, 693 pp.

Grasshoff, K., Ehrhardt, M., Kremling, K., et al., 1983. Methods of Seawater Analysis, second ed. Deerfield Beach, Weinheim.

Gray, J.S., 1989. Effects of environmental stress on species rich assemblages. Biol. J. Linn. Soc. 37, 19–32.

Guerra Garcia, J.M., Corzo, J., Carlos García Gomez, J., 2003. Short term benthic recolonization after dredging in the harbour of Ceuta, North Africa. Mar. Ecol. 24, 217–229.

Gupta, G.V.M., Thottathil, S.D., Balachandran, K.K., Madhu, N.V., Madeswaran, P., Nair, S., 2009. CO2 supersaturation and net heterotrophy in a tropical estuary (Cochin, India): influence of anthropogenic effect. Ecosystems 12, 1145–1157.

Gutperlet, R., Capperucci, R.M., Bartholoma, A., Kroncke, I., 2015. Benthic biodiversity changes in response to dredging activities during the construction of a deep-water port. Mar. Biodivers. 45 (4), 819–839.

Hossain, M.B., Marshall, D.J., Venkatramanan, S., 2014. Sediment granulometry and organic matter content in the intertidal zone of the Sungai Brunei estuarine system, northwest coast of Borneo. Carpath. J. Earth Env. Sci. 9, 231–239.

Hossain, S., Eyre, B.D., McKee, L.J., 2004. Impacts of dredging on dry season suspended sediment concentration in the Brisbane River estuary, Queensland, Australia. Estuar. Coast. Shelf Sci. 61, 539–545.

Imrie, D.W., McCrohan, C.R., Hawkins, S.J., 1990. Feeding behaviour in Littorina littorea: a study of the effects of ingestive conditioning and previous dietary history on food preference and rates of consumption. Hydrobiologia 193, 191–198.

Johnston Jr., S.A., 1981. Estuarine dredge and fill activities: a review of impacts. Environ. Manage. 5, 427–440.

Jones, R., Fisher, R., Stark, C., Ridd, P., 2015. Temporal patterns in seawater quality from dredging in tropical environments. PLoS One 10 (10). http://dx.doi.org/10.1371/journal.pone.0137112.

Junior, S., Santos, S.R., Travassos, M., Vianna, M., 2012. Impact on a fish assemblage of the maintenance dredging of a navigation channel in a tropical coastal ecosystem. Braz. J. Oceanogr. 60, 25–32.

Kaplan, E.H., Welker, J.R., Kraus, M.G., McCourt, S., 1975. Some factors affecting the colonization of a dredged channel. Mar. Biol. 32, 193–204.

Kennish, M.J., 1991. Ecology of Estuaries: Anthropogenic Effects. CRC press, 1,363 pp.

Kroncke, I., 2006. Structure and function of macrofaunal communities influenced by hydrodynamically controlled food availability in the Wadden Sea, the open North Sea and the Deep-sea. A synopsis. Senckenberg. Marit. 36, 123–164.

Krumbein, W.C., Pettijohn, F.J., 1938. Manual of Sedimentary Petrography. Appleton-Century-Crafts, Inc., Newyork, p. 549.

Kumar, R.S., 2002. Blomass, horizontal zonation and vertical stratification of polychaete fauna in the littoral sediments of Cochin estuarine mangrove habitat, South west coast of India. Indian J. Geo.-Mar. Sci. 31 (2), 100–107.

Leal, J.H., Matthews, B., 2013. Bivalves. Shell Museum, Florida, USA, 52 pp.

Lundquist, C.J., Thrush, S.F., Coco, G., Hewitt, J.E., 2010. Interactions between disturbance and dispersal reduce persistence thresholds in a benthic community. Mar. Ecol. Prog. Ser. 413, 217–228.

Madhu, N.V., Balachandran, K.K., Martin, G.D., Jyothibabu, R., Thottathil, S.D., Nair, M., Joseph, T., Kusum, K.K., 2010a. Short-term variability of water quality and its implications on phytoplankton production in a tropical estuary (Cochin backwaters-India). Environ. Monit. Assess. 170, 287–300.

Madhu, N.V., Jyothibabu, R., Balachandran, K.K., 2010b. Monsoon-induced changes in the size-fractionated phytoplankton biomass and production rate in the estuarine and coastal waters of southwest coast of India. Environ. Monit. Assess. 166, 521–528.

Magalhaes, W.F., Barros, F., 2011. Structural and functional approaches to describe polychaete assemblages: ecological implications for estuarine ecosystems. Mar. Freshw. Res. 62, 918–926.

Martin, G., George, R., Shaiju, P., Muraleedharan, K., Nair, S., Chandramohanakumar, N., 2012. Toxic metals enrichment in the surficial sediments of a eutrophic tropical estuary (Cochin Backwaters, Southwest Coast of India). Sci. World J. 2012, 1–17.

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

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

Mondal, N., Rajkumar, M., Sun, J., Kundu, S., Lyla, P.S., Khan, S.A., Trilles, J.P., 2010. Biodiversity of brackish water amphipods (crustacean) in two estuaries, southeast coast of India. Environ. Monit. Assess. 171, 471–486.

Nayar, K.N., 1959. Amphipoda of the Madras coast. Bull. Madras Gov. Mus. 6, 15–76.

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

Newell, R.C., Hitchcock, D.R., Seiderer, L.J., 1999. Organic enrichment associated with outwash from marine aggregates dredging: a probable explanation for surface sheens and enhanced benthic production in the vicinity of dredging operations. Mar. Pollut. Bull. 38, 809–818.

Newell, R.C., Seiderer, L.J., Hitchcock, D.R., 1998. The impact of dredging works in coastal waters: a review of the sensitivity to disturbance and subsequent recovery of biological resources on the sea bed. Oceanogr. Mar. Biol. Annu. Rev. 36, 127–178.

Newell, R.C., Seiderer, L.J., Simpson, N.M., Robinson, J.E., 2004. Impacts of marine aggregate dredging on benthic macrofauna off the south coast of the United Kingdom. J. Coast. Res. 20, 115–125.

Nguyen, L.M., Cooke, J.G., McBride, G.B., 1997. Phosphorus retention and release characteristics of sewage-impacted wetland sediments. Water, Air, Soil Pollut. 100, 163–179.

Ogbeibu, A., Arazu, V., Nzei, N., Igho, R., 2010. Impact of dredging on the water quality and macrobenthic invertebrate fauna of the Ikpoba River in southern Nigeria. Trop. Freshw. Biol. 19 (1), 59–78.

Pacheco, A.S., Gonzalez, M.T., Bremner, J., Oliva, M., Heilmayer, O., Laudien, J., Riascos, J.M., 2011. Functional diversity of marine macrobenthic communities from sublittoral soft-sediment habitats off northern Chile. Helgol. Mar. Res. 65, 413–424.

Pfitzenmeyer, H.T., 1970. Benthos. Gross physical and biological effects of overboard spoil disposal in upper Chesapeake Bay. Univ. Md. Nat. Resour. Inst. Spec. Rep. 3, 26–38.

Pillai, N., 1977. Distribution and seasonal abundance of macrobenthos of the Cochin backwaters. Indian J. Mar. Sci. 6, 1–5.

Qasim, S.Z., 2003. Indian Estuaries. Allied Publishers, p. 259.

Rasheed, K., 1997. Studies on Dredging Impact Assessment (DIA) at Cochin, a Tropical Estuarine Harbour. Thesis. CUSAT, Kochi.

Revichandran, C., Srinivas, K., Muraleedharan, K., Rafeeq, M., Amaravayal, S., Vijayakumar, K., Jayalakshmy, K., 2012. Environmental set-up and tidal propagation in a tropical estuary with dual connection to the sea (SW Coast of India). Environ. Earth Sci. 66, 1031–1042.

Robin, R.S., Muduli, P.R., Vardhan, K.V., Abhilash, K.R., Selvam, A.P., Kumar, B.C., Balasubramanian, T., 2012. Assessment of hydrogeochemical characteristic in an urbanized estuary using environmental techniques. Geo Sci. J. 2, 81–92.

Rosenberg, R., 1977. Effects of dredging operations on estuarine benthic macrofauna. Mar. Pollut. Bull. 8, 102–104.

Sarda, R., Pinedo, S., Gremare, A., Taboada, S., 2000. Changes in the dynamics of shallow sandy-bottom assemblages due to sand extraction in the Catalan Western Mediterranean Sea. ICES J. Mar. Sci. 57, 1446–1453.

Shivaprasad, A., Vinita, J., Revichandran, C., Reny, P.D., Deepak, M.P., Muraleedharan, K.R., Naveen Kumar, K.R., 2013. Seasonal stratification and property distributions in a tropical estuary (Cochin estuary, west coast, India). Hydrol. Earth Syst. Sci. 17, 187–199.

Simon, J.L., Conner, W.G., 1977. Response of a Subtropical Soft-bottom Community to Pertubation of Oyster Shell Dredging. Am. Zool. Amer Soc Zoologists, 1041 New Hampshire ST, Lawrence, KS 66044, p. 907.

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

Srinivas, K., Revichandran, C., Maheswaran, P., Asharaf, T.M., Murukesh, N., 2003. Propagation of tides in the Cochin estuarine system, southwest coast of India. Indian J. Geo-Mar. Sci. 32 (1), 14–24.

Taupp, T., Wetzel, M.A., 2013. Relocation of dredged material in estuaries under the aspect of the water framework directive - a comparison of benthic quality indicators at dumping areas in the Elbe estuary. Ecol. Indic. 34, 323–331.

Thrush, S.F., Dayton, P.K., 2002. Disturbance to marine benthic habitats by trawling and dredging: implications for marine biodiversity. Annu. Rev. Ecol. Syst. 33 (1), 449–473.

Thrush, S.F., Hewitt, J.E., Gibbs, M., Lundquist, C., Norkko, A., 2006. Functional role of large organisms in intertidal communities: community effects and ecosystem function. Ecosystems 9, 1029–1040.

Ulanowicz, R.E., 1997. Ecology, the Ascendent Perspective. Columbia University Press, New York.

Van Dalfsen, J.A., Essink, K., Madsen, H.T., Birklund, J., Romero, J., Manzanera, M., 2000. Differential response of macrozoobenthos to marine sand extraction in the North Sea and the Western Mediterranean. ICES J. Mar. Sci. J. Cons. 57, 1439–1445.

Van der Veer, H.W., Bergman, M.J.N., Beukema, J.J., 1985. Dredging activities in the Dutch Wadden Sea: effects on macrobenthic infauna. Neth. J. Sea Res. 19, 183–190.

Van Dolah, R.F., Calder, D.R., Knott, D.M., 1984. Effects of dredging and open-water disposal on benthic macroinvertebrates in a South Carolina estuary. Estuaries 7, 28–37.

Venkatramanan, S., Ramkumar, T., Anithamary, I., 2013. Distribution of grain size, clay mineralogy and organic matter of surface sediments from Tirumalairajanar Estuary, Tamilnadu, east coast of India. Arab. J. Geosci. 6, 1371–1380.

Vijith, V., Sundar, D., Shetye, S., 2009. Time-dependence of salinity in monsoonal estuaries. Estuar. Coast. Shelf Sci. 85, 601–608.

Waye-Barker, G.A., McIlwaine, P., Lozach, S., Cooper, K.M., 2015. The effects of marine sand and gravel extraction on the sediment composition and macrofaunal community of a commercial dredging site (15 years post-dredging). Mar. Pollut. Bull. 99, 207–215.

Whomersly, P., Ware, S., Rees, H.L., Mason, C., Bolam, T., Huxham, M., Bates, H., 2008. Biological indicators of disturbance at a dredged-material disposal site in Liverpool Bay, UK: an assessment using time-series data. ICES J. Mar. Sci. 65, 1414–1420.

Yeager, K.M., Brinkmeyer, R., Rakocinski, C.F., Schindler, K.J., Santschi, P.H., 2010. Impacts of dredging activities on the accumulation of dioxins in surface sediments of the Houston Ship Channel, Texas. J. Coast. Res. 26, 743–752.

Continental Shelf Research 80 (2014) 79–90

Contents lists available at ScienceDirect

Continental Shelf Research

journal homepage: www.elsevier.com/locate/csr

Research papers

Characterization of phytoplankton pigments and functional community structure in the Gulf of Mannar and the Palk Bay using HPLC–CHEMTAX analysis

N.V. Madhu*, N. Ullas, R. Ashwini, Paul Meenu, T.V. Rehitha, K.R. Lallu

National Institute of Oceanography, Regional Centre, Kochi-18, India

A R T I C L E I N F O

Article history:
Received 3 October 2013
Received in revised form
4 March 2014
Accepted 9 March 2014
Available online 19 March 2014

Keywords:
Phytoplankton
HPLC
Biomarkers
Cyanobacteria
Palk Bay
Gulf of Mannar

A B S T R A C T

Phytoplankton marker pigments and their functional groups were identified for the first time in the Gulf of Mannar (GoM) and the Palk Bay (PB), located in the southeast coast of India using HPLC–CHEMTAX analytical techniques. The GoM generally remained more saline, productive (in terms of chlorophyll *a*) and less turbid than the PB during southwest and northeast monsoon periods. The diversity and concentration of marker pigments were high in the GoM, whereas the PB was characterized by high concentration of zeaxanthin, indicating the dominance of photosynthetic prokaryotes (cyanobacteria). The CHEMTAX analysis revealed that the phytoplankton biomass (chlorophyll *a*) in the PB was mainly derived from cyanobacterial community. However, abundance of fucoxanthin and peridinin in the GoM indicated microphytoplankton (20–200 μm) as the dominant group. The CHEMTAX results showed that more than 50% of chlorophyll *a* in the GoM was contributed by microphytoplankton, in particular diatoms and dinoflagellates. The substantial increase in the photoprotective carotenoids (PPCs) and photoprotection index (PI) in the PB was indicative of its low productivity, probably caused by the warm and turbid waters.

1. Introduction

Studies on phytoplankton are very useful for understanding the bio-geochemical processes and efficiency of the marine food web (Chisholm, 1992; Legendre and Rassoulzadegan, 1996). Since phytoplankton is a main source of organic production, information on their biomass, composition and community structure are so decisive for monitoring of an aquatic environment (Paerl et al., 2003). The growth and distribution of these microscopic communities are mainly controlled by a variety of abiotic and biotic factors (Chisholm, 1992). Besides chlorophyll *a*, (proxy for phytoplankton biomass), phytoplankton have evolved taxon-specific suites of various pigments (biomarkers) for performing photosynthetic activities (Wright and Jeffrey, 2006). These photopigments (e.g., fucoxanthin, peridinin, chlorophyll *b*, zeaxanthin, and alloxanthin) are diagnostic of specific phytoplankton functional groups (diatoms, dinoflagellates, chlorophytes, cyanobacteria and cryptophytes, respectively) and in a few cases can determine phytoplankton composition at the species level (Paerl et al., 2003; Barlow et al., 2008). Each of these phytoplankton functional groups (PFGs) are indicators of primary production that may change with varying environmental conditions (Barlow et al., 2007). Analysis of phytoplankton pigments using HPLC technique does not afford anything on species identification rather than the class level information (Suzuki et al., 1997; Barlow et al., 2008); however, this measurement is faster and reproducible for analysing a large set of samples compared to the time consuming and tedious microscopic identification (Jeffrey et al., 1997).

Since marker pigments can reveal a variety of taxonomic composition (Gibb et al., 2000; Barlow et al., 2007), physiological state and grazing effect (Trees et al., 2000; Veldhuis and Kraay, 2004), the pigment characterisation by HPLC is widely used to study the properties of phytoplankton. CHEMTAX is a statistical software used to estimate biomass of various PFGs by calculating the ratio of marker pigments to chlorophyll *a* (Mackey et al., 1996; Riegman and Kraay, 2001). Photosynthetic pigments and their ratios (indices) are normally used for determining the phytoplankton groups (Aiken et al., 2009) under varying environmental conditions (Vidussi et al., 2001; Griffith et al., 2010). For example, the carotenoid pigment ratio is used for understanding the productivity patterns of various water bodies (Kirk, 1994; Barlow et al., 2002; Moreno et al., 2012). High concentration of photosynthetic carotenoids (PSCs) normally indicates high productivity, whereas high photoprotective carotenoids (PPCs) signify low productivity (Gibb et al., 2000; Barlow et al., 2002). These variations are generally associated with the changes in the phytoplankton community of a system as well as their cellular responses (Barlow et al., 2008; Moreno et al., 2012).

* Corresponding author.
E-mail addresses: nmadhu@nio.org, madhunv12@gmail.com (N.V. Madhu).

http://dx.doi.org/10.1016/j.csr.2014.03.004
0278-4343/

Environ Monit Assess
DOI 10.1007/s10661-014-3741-6

Transport of dissolved nutrients and chlorophyll *a* in a tropical estuary, southwest coast of India

K. R. Lallu · K. H. Fausia · J. Vinita · K. K. Balachandran · K. R. Naveen Kumar · T. V. Rehitha

Received: 25 October 2013 / Accepted: 18 March 2014

Abstract Intra-tidal variability in the transport of materials through the Cochin estuary was studied over successive spring and neap tides to estimate the export fluxes of nutrients and chlorophyll *a* into the adjoining coastal zone. The results showed that there was a substantial increase in the freshwater flow into the estuary following heavy rains (~126 mm) prior to the spring tide observations. The estuary responded accordingly with a relatively larger export through the Cochin inlet during spring tide over neap tide. Despite an increased freshwater discharge during spring tide, the export fluxes of phosphate and ammonia were high during neap tide due to their input into the estuary through anthropogenic activities. The significance of this study is that the export fluxes from the Cochin estuary could be a major factor sustaining the spectacular monsoon fishery along the southwest coast of India.

Keywords Cochin estuary · Southwest coast of India · Coastal upwelling · Intra-tidal variations · Flux measurement · Nutrient fluxes · Spring-neap tides · Chlorophyll *a*

K. R. Lallu (✉) · K. H. Fausia · J. Vinita · K. K. Balachandran · K. R. Naveen Kumar · T. V. Rehitha
Regional Centre, CSIR National Institute of Oceanography, P.B. No. 1913, Cochin 682018, India
e-mail: lallukr@gmail.com

Present Address:
K. H. Fausia
Kerala Agricultural University, Aromatic and Medicinal Plants Research Station,
Asamannoor, Kochi, Kerala 683549, India

Introduction

Estuaries are the intermediate zones of land-sea interactions contributing significantly to the nutrient fluxes in to the ocean on varying time scales (Simpson et al. 2001). The nutrient delivery is influenced by catchment hydrology, freshwater flow and tides, which eventually determine the productivity of an estuary (Pennock et al. 1994; Justic et al. 1995; Doval et al. 1997; Mackas and Harrison 1997). Due to complex and varied nature of non-point sources, nutrient fluxes through estuaries are difficult to measure. Increased anthropogenic activities generally enhance nutrient levels in water bodies, leading to eutrophication and oxygen depletion (Beukema 1991; Parker and O'Reilly 1991). Estuaries are generally classified as matured systems when they sustain surplus nutrients (Dame et al. 1992; Dame and Allen 1996). Hence, transport measurements are important to study the nutrient economy and productivity patterns of coastal marine systems (Dehairs et al. 2000; McManus et al. 2001).

Cochin estuary is the second largest estuary along the west coast of India, that spreads over 250 km^2 and receives a freshwater input of 2×10^{10} m^3/y from six rivers (Srinivas et al. 2003). The estuary experiences mixed semidiurnal tides with clear variations from neap to spring phase. The tides at the inlet are ~1 m, which progressively attenuates towards upstream. Cochin is a major inlet (~450 m wide and 13.5 m deep), while there is another minor inlet at Azhikode, 25 km north of Cochin (Fig. 1). The region receives a mean annual precipitation of 3,200 mm, of which, >65 %

Published online: 26 April 2014

Indian Journal of Geo-Marine Sciences
Vol. 42(4), August 2013, pp. 453-457

Occurrence of cyanobacteria (*Richelia intracellularis*)-diatom (*Rhizosolenia hebetata)* consortium in the Palk Bay, southeast coast of India

[1]Madhu N V, Meenu Paul, Ullas N, Ashwini R & Rehitha T V
National Institute of Oceanography, Regional Centre, Kochi – 682 018, India
[E-Mail: nmadhu@nio.org]

[Received 11 November 2011; revised 30 January 2012]

Symbiotic association of heterocystous cyanobacterium, *Richelia intracellularis* Schmidt with oceanic centric diatom, *Rhizosolenia hebetata* is reported from the Palk Bay, southeast coast of India. One to six trichomes of *R. intracellularis* were occluded inside the periplasmic space of *R.hebetata*, with their prominent heterocyst pointing towards the valve of the host. Each of these trichomes had 14 to 23 vegetative cells caped by a terminal heterocyst enriched with the nitrogenase enzyme. Density of *Rhizosolenia* containing *R. intracellularis* ranged between 120 and 260 cells L^{-1}, and present uniformly in the water column. *R. intracellularis* is a diazotroph, can contribute substantially to the N_2 budgets thereby, promoting a different food web in the Palk Bay.

[Keywords: Cyanobacterium, Diatom, Diazotrophy, Palk Bay]

Introduction

Cyanobacteria, a photosynthetic prokaryote (blue-green algae) is a wide spread planktonic organism commonly found in the rivers, rocks and soil. Significance of this diverse community is its capability to fix nitrogen (N) in the warm, sunlit waters of tropical oceans[1]. It is estimated that the annual N fixation of 4.79×10^{12} g constitutes one quarter of the total input of nitrogen to the sea[2]. Morphologically, cyanobacteria exist as coccoid, filamentous, non-heterocystous or heterocystous forms and possess the unique cells of '*Heterocysts*', which can fix atmospheric nitrogen. *Trichodesmium* and *Richelia* are the most important nitrogen fixing (diazotrophic) marine cyanobacteria[3]. Of these, the *Trichodesmium* spp. usually found in the warm oligotrophic waters are devoid of heterocysts; whereas *Richelia intracellularis* is the only heterocystous cyanobacteria in the marine waters[2]. Many of the cyanobacterial communities are found symbiotically (inside or outside) associated with larger phytoplankton (diatoms and dinoflagellates) in oligotrophic waters[4,5].

Materials and Methods

Palk Bay (PB), a shallow water body located in the southeast coast of India between Sri Lanka and Indian sub-continent, seemed to be biologically productive than its counterpart, the Gulf of Mannar (GoM) as it receives Bay of Bengal water with lower salinity and higher nutrients[6]. Water quality monitoring in the PB and GoM was carried out in March 2010 as a part of the Environmental Impact Assessment (EIA) programme for alignment 4 A of Sethusamudram channel. Water samples were collected from the surface, mid- and bottom depths using a 5L Niskin bottles for analyzing inorganic nutrients[7] and phytoplankton characteristics (biomass and abundance). A portable CTD (Seabird SBE 26^{plus}) was used to measure water temperature (accuracy ± 0.001°C) and salinity (accuracy±0.0001). Estimation of chlorophyll *a* was done fluorometrically[8]. Lugol's (acid) iodine fixed phytoplankton counts[9] and identification[10] were made with a stereoscope inverted microscope (Olympus CK 30).

Results and Discussion

During the identification of phytoplankton samples collected from stations A1 and A2 (Fig. 1), observed a diatom-cyanobacterium symbiotic association. Inverted microscope images showing a filamentous structure inside the host cell (*Rhizosolenia hebetata,* a ubiquitous centric diatom) was identified as *Richelia intracellularis* Schmidt, a diazotrophic marine cyanobacterium. Furthermore, there were upto 6 trichomes of

[1]Corresponding author