ASSESSING THE STRUCTURAL AND FUNCTIONAL ECOLOGY OF BENTHIC FAUNA IN VEMBANAD ESTUARINE SYSTEM, INDIA

Thesis submitted to

Cochin University of Science and Technology

in Partial Fulfilment of the Requirements for

the Award of the Degree of

Doctor of Philosophy

in

Marine Biology

Under the Faculty of Marine Sciences

By

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COCHIN UNIVERSITY OF SCIENCE AND TECHNOLOGY

KOCHI –682 016, INDIA

August 2017

Assessing the structural and functional ecology of benthic fauna in Vembanad estuarine system, India

Ph.D. Thesis under the Faculty of Marine Sciences

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Certificate

This is to certify that the thesis entitled **"Assessing the structural and functional ecology of benthic fauna in Vembanad estuarine system, India"** is an authentic record of research work carried out by Mrs. Asha C. V. (Reg. No. 3882), under my supervision and guidance in the Department of Marine Biology, Microbiology and Biochemistry, Cochin University of Science and Technology, in partial fulfilment of the requirements for the Degree of Doctor of Philosophy in Marine Biology, Cochin University of Science and Technology under the faculty of Marine sciences and that no part of this has been presented before for the award of any other degree, diploma or associateship in any university.

It is also certified that all the relevant corrections and modifications suggested by the audience during the pre-synopsis seminar and recommended by the doctoral committee has been incorporated in the thesis.

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Declaration

I hereby declare that the thesis entitled **"Assessing the structural and functional ecology of benthic fauna in Vembanad estuarine system, India"** is an authentic record of research work carried out under the supervision and guidance of Prof. (Dr.) S. Bijoy Nandan, Department of Marine Biology, Microbiology and Biochemistry, School of Marine Sciences, Cochin University of Science and Technology, in partial fulfillment of the requirement for the Ph.D degree in Marine Biology and that no part thereof has been presented for the award of any other degree in any University.

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Dedicated to my parents…

Acknowledgement

*Firstly, I would like to express my heartfelt gratitude to my supervising guide **Prof. (Dr.) S. Bijoy Nandan**, Department of Marine Biology, Microbiology and Biochemistry, for accepting me into his group. I am grateful to his valuable guidance, continuous support and encouragement, inspiring discussions and patient criticisms throughout my research career. During my tenure, he inculcates my research experiences by providing opportunities in attending conferences and seminars and demanding a high quality of work in all my activities. His enlightening words towards life and achievements have been strong influence on my career and life. I acknowledge him with immense gratitude.*

I am grateful to Prof. (Dr.) A. N Balchand, Dean, Faculty of Marine Sciences and Prof. (Dr.) Sajan K, Director, School of Marine Sciences, for providing necessary facilities. I would like to express my sincere thanks to Prof. (Dr.) Rosamma Philip, Head of the Department, Dept. of Marine Biology, Microbiology and Biochemistry for providing necessary facilities. Her valuable suggestions and continuous encouragement is greatly acknowledged. I express my immense gratitude to Prof. (Dr.) A.V. Saramma, Prof. (Dr.) Aneykutty Joseph and Prof. (Dr.) Mohamed Hatha, my post-graduation teachers of the Dept. of Marine Biology, Microbiology and Biochemistry for their constant encouragement and consideration. I wish to thank Dr. Priyaja P, Dr. Swapna P. Antony, Dr. K. B. Padmakumar and Dr. Manjusha. K. P, faculties of the department for their support. I would like to thank Prof. (Dr.) Babu Philip, Prof. (Dr.) Radhakrishnan, Dr. N.G.K. Pillai and Dr. M. Harikrishnan for their inspiration. I am grateful to Prof. (Dr.) H. S. Ram Mohan, former Dean and Director, School of Marine Sciences and Prof. (Dr.) Mohan Kumar, former Director, School of Marine Sciences for their immense support to improve the course of Ph. D programme. I sincerely thank office staff and other non –teaching staffs of the Department of Marine Biology, Microbiology and Biochemistry, for their timely services throughout my research period. I am very much thankful to Dr. Prabhakaran, Dr. Santhosh, Dr. Mangala Unni, Dr. Abdul Jaleel, Dr. Mujeeb Rahman, Dr. Abish B, Dr. Smitha B.R, Mrs. Asha Devi for the valuable advices and support.

I am grateful to Dr. Ji-Hun Song and Dr. Gi-Sik Min (Inha University, South Korea); Dr. Alexey V. Chernyshev (Far Eastern Federal University, Russia); Dr. P. Graham Oliver (National Museum of Wales and School of Ocean Sciences, Bangor University North Wales, United Kingdom); Dr. N.V. Subba Rao for helping the macrofaunal species identifications. I extend my special gratitude towards Chairman, Pollution Control Board, for providing the lab facility for analyzing the sediment heavy metal.

I express my gratitude to Ministry of Environment, Forest and Climate Change (MOEFCC), Govt. of India, for the financial support as JRF in the Project entitled "Ecosystem Based Monitoring and Modelling of the Vembanad Kol Wetland in Kuttanad, Kerala, India" from 2010 to 2014 period. I am thankful to the Director, School of Industrial Fisheries and office staff of School of Industrial Fisheries for providing the vessel, King Fisher for field work. The innumerous, laborious and tedious field trips became an enjoyable experience and beautiful memory because of Mr. Anilajan, Mr. Suresh, Mr. Stephen and Mr. Raveendran, thanks for your continuous support. I thank Mr. P. J. Manuel, former Librarian, Mr. Balan and other staff members of School of Marine Sciences Library for their kind cooperation and support. I also thank all administrative and security staff in the School of Marine Sciences for their support. I also thank Cochin University of Science and Technology for helping the administrative work of my thesis.

I am grateful to my dear colleagues Retina I Cleetus and Suson P. S for their continuous support and co-operation from the initial stages of field sampling to the whole period of my Ph.D work. I acknowledge the help and encouragement provided by my dear fellow research scholars Philomina Joseph, Rani Varghese, Preethy C. M, Rakhi Goplan, Sreelekshmi S, Sreedevi O. K, Santu K. S, Jayachandran P. R, Akhilesh Vijay. The unconditional support and encouragement from them cherished throughout my career. I am really thankful to my dear friends Greeshmam M. B, Dr. Jayalakshmi, Ramya K. D, Jisha Kumaran, Savitha N, and Keerthi for their support and persistent confidence given throughout the entire course of my Ph.D programme. Special thanks to all the research scholars of the ecology lab especially, Sajana, Radhika, Ambily V, Anu P. R, Jima, Geetha P. N, Midhun, Don Xavier, Kapila, Sanu V. F, Krishna Priya, *Rejina, Neelima Vasu, Meenu, who had helped me in innumerable ways. I thank former project fellows in the ecology lab mainly Remya, Vineetha, Minu, Vishnu, Akhil Ghosh, Rithu, Aji Joseph and Rebin for their support during my research period.*

I am extremely thankful to my seniors Dr. Vineetha, Dr. Lathika Cicly Thomas, Dr. Deborah A, Dr. Anu Pavithran, Chaithanya E. R, Jimly C Jacob, Dr. Thasneem T. R, Dr. Naveen Sathyan, Anil Kumar, Jini Jacob, Dr. Shameeda, Dr. Smitha C. K, Mr. Shyam and Vijayalakshmi for their help and cooperation. I have pleasure in acknowledging my M.Sc. classmates Emilda Rosmine, Anjusha. A, Lakshmi Devi, Subin C.S, Jaliya Sara Jacob, Meera K. M., Ancy Thomas, Uma, Neetha for their words of encouragement. I also thank my dear friends Leena, Saritha, Lekshmi, Bini, Preethy John, Anju Mathew, Nithya, Ramya, Shinu, Ragi, Sruthi, Jabir, Solly, Minu, Sruthi for their support and friendship. I sincerely thank M Sc. and M Phil. students of this department who too have rendered all possible help. I also thank all my Athulya hostel mates, hostel matrons and mess workers for making my Ph. D period became an enjoyable experience with numerous lovely memories.

I am expressing my heartiest thanks to my father-Thampan P, mother - Padmaja C. V and grandmother -Thampai C. V for their support, patience and prayers during the entire course of this research work. I am grateful to my sister, Priya C. V, brother in law, Sudeep P. V and their little boy, Karthik for their inspiration and support. With deep respect and gratitude, I remember my grandfather late P. Sankaran Nair and father-in- law late N. V. Kunhiraman for their blessings. I am thankful to my mother in law, Kamalakshi M, and brother in law Naveen M for their kind support and help. I thank my husband Arun M for his continuous support and encouragement during the course of my work. I would also like to thank my baby daughter Pournami. I sincerely thank each and every person who helped me in various ways to complete this thesis in time.

Above all, I thank God Almighty for blessing me with the potential to complete this work successfully.

Asha C. V.

Contents

Chapter 3

PHYSICO-CHEMICAL FACTORS STRUCTURING THE TROPHIC ENVIRONMENT OF VEMBANAD ESTUARY............101 - 198

Chapter **4**

SEDIMENT CHARACTERISTICS OF THE VEMBANAD TROPHIC SYSTEM .... 199 - 264

Chapter **5**

ECOLOGY AND DISTRIBUTION OF SOFT BOTTOM BENTHIC COMMUNITIES OF VEMBANAD ESTUARY ....265 - 387

***Chapter* 6**

COMMUNITY STRUCTURE AND ECOLOGY OF POLYCHAETES IN THE VEMBANAD ESTUARY........................389 - 462

Chapter **9**

List of Tables

List of Figures

List of Plates

List of Annexures

List of Abbreviations and Symbols

%	-	Percentage
$\leq$	-	less than or equal to
$\geq$	-	Greater than or equal to
°C	-	degree Celsius
Anon	-	Anonymous
cm	-	Centimeters
et al.	-	And others
Fig.	-	Figure
g	-	gram
ha	-	Hectares
ind/ m^2	-	Individual per meter square
ie	-	that is
Km	-	Kilometer
Km^2	-	Square kilometre
L	-	Litres
m	-	Meters
m^2	-	Square meter
m^3	-	Cubic meters
mg	-	Milligram
Mha	-	Million hectare
ml	-	Milliliter
mm	-	millimeters
N	-	North
S	-	South
NW	-	North West
nm	-	Nanometer
No.	-	Number
ppm	-	Parts per million
SD	-	Standard Deviation
sp	-	Species
t	-	Tonnes
v6	-	Version 6
µ mol	-	Micro mol
µm	-	Micrometer

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Chapter 1

GENERAL INTRODUCTION

Contents

Wetlands are, one of the most productive ecosystems of the earth (Ghermandi et al., 2008). These are ecotones or transitional zones between dry lands and open water where the water level remains near or above the surface of the ground for most of the year. It has the unique characteristics of their own and also possesses the properties of both terrestrial and aquatic ecosystems. It supports a wide variety of flora and fauna which provide many ecological, climatic and societal functions. It covers about 6% of the earth's land surface. They are multidimensional, cross boundary resources which provide a range of inter related environmental functions and socio-economic benefits, which support a variety of livelihood strategies. They are critical for the maintenance of biodiversity and perform a great role in the biosphere and are often referred to as the "kidneys" of the earth. The wetlands cover diverse and heterogeneous assemblage of habitats ranging from lakes, estuaries, river flood plains, mangroves, coral reef and other related ecosystems. It provides many valuable services at population, ecosystem and global levels which form home to some of the richest, diverse and fragile natural resources. As they support a variety of plant and animal life, biologically they are one of the most productive systems in world. It is unique in having rich nutrient status and carrying capacity with immense production potential, hence considered as food and fodder resources for the entire community.

The value of the world's wetlands is increasingly receiving due attention as they contribute to a healthy environment in many ways, as they support different food chain, food webs, regulate hydrological cycle, recharge ground water, trapping of energy and shelter to large numbers of flora and fauna having great ecological and economical value. They provide suitable habitats for endangered and rare species of birds and animals, endemic plants, insects besides sustaining migratory birds. Based on the hydrological, ecological and geological characteristics of wetlands Cowardin et al. (1979) classified wetlands into marine (coastal wetlands), estuarine (including deltas, tidal marshes, and mangrove swamps), lacustarine (lakes), riverine (along rivers and streams), and palustarine ('marshy' – marshes, swamps and bogs). The Ramsar Convention (1971) defines wetlands as: *areas of marsh, fen, peat land or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salty including areas of marine water, the depth of which at low tide does not exceed 6 meters. It may also incorporate riparian and coastal zones adjacent to the wetlands and islands or bodies of marine water deeper than 6 meters at low tide lying within the wetlands* (www.ramsar.org). There are many kinds of coastal wetlands including salt marshes, estuaries, mudflats, swamps, bogs and lagoons.

Coastal ecosystems, situated along the margins of continents and oceanic islands, are areas of high productivity. They have key role in sustaining human wellbeing due to its enormous biological resources and the life-supporting services (WRI, 2001; UNISDR/UDNP, 2012). Within the coast, there is a variety of coastal ecosystems such as coral reefs to seagrass meadows, sand dunes, mangroves, salt marshes, tidal flats, lagoons and estuaries. Each of these ecosystems harbours a variety of species and maintains a range of ecosystem services essential for humans. In which coastal lagoons and estuaries deliver significant ecosystem goods and services to society and so most of the world's population live in coastal regions (Michael and Paerl, 2010). However, population expansion, variations in land and water use pattern, and habitat destruction in connecting coastal watersheds decrease the productivity and health of the coastal environment (Borja et al., 2008b). A comprehensive assessment of the coastal systems, ecological dynamics, and carrying capacity is necessary to evade additional ecological destruction that would damage the human economic and functional reliance on these important ecosystems (Borja and Dauer, 2008). Knowing the ecogeomorphology and biogeochemistry of coastal ecosystem needed the concern of humans as agents of change (Borja et al., 2008b; Bauer et al., 2013).

"An estuary is a semi-enclosed coastal body of water which has a free connection with the open sea and within which sea water is measurably diluted with fresh water derived from land drainage" (Pritchard, 1967). It is an area of transition from the tidal conditions seaward to the freshwater flows from landward. When the saline and freshwater bodies meet, mixing takes place, to a greater or lesser extent, and created a noticeable interface between the two water bodies and the formation of internal waves on the border between the two. Based on the size of the estuary, the salinity gradients produced density flows and it can be directed both along and across the estuary. Similarly the complexity of water movements was revealed in the sediment transport pathways. The sediments sourced from marine or freshwater input formed a complicated sediment reworking system within an estuary and erosional and depositional shores can exist in close proximity.

As transition zone between land and sea, estuarine environments were grouped to the richest and most productive ecosystems in the world. Coastal estuarine areas have developed to become some of the most densely populated and economically vibrant areas in the world. It was estimated that 'over 2.1 billion people are concentrated in coastal areas' (Vitousek and Mooney, 1997).

Morphology of estuaries, consist of the flat, fertile surrounding land, in association with their accessibility, has over the years led to the attraction of agriculture, industry, maritime commerce and urbanization. The coastal zones, preferred for habitations are formed the most crowded, developed and also overexploited regions in the world (Hinrichsen, 1995). The greatest population growth rates were observed in the coastal areas (Singh, 1999). Due to the population explosion in the coastal areas, extensive pressures were exerted on estuarine environments (Birch et al., 2016). Global population has increased from 2.5 billion people in 1950 to 6.5 billion people in 2005. It was expected that by 2050, this number could increase to more than 9 billion (http://www.prb.org/). Compared to the inland area, population density was more in coastal zones with an average 87 people/km^{-2} in coastal zones in comparison with 23 people/km^{-2} inland areas in 2000. The population pressure was more in Europe and Asia, where 31 % of the area had high population density. In the coastal zones of China, India, Japan, Bangladesh, Philippines, Vietnam and United Kingdom had 50% of the land with increased population densities in the coastal zone. The coastal zones of China, Bangladesh and India had high population density, with 93%, 89% and 86% respectively (Shi et al., 2001). Of the 32 largest cities in the world, 22 are located on estuaries (Ross, 1995). Out of ten most populous cities, eight are situated in the coastal areas. In terms of total global population coastal zones are “bright spots”, while at the same time being “hotspots" of biodiversity. Around 40.7 % of the world area of biodiversity hotspots was located in the coastal regions, in which only 10.45 % of coastal zones are designated as protected areas (Shi et al., 2001).

Estuaries are considered as biologically and economically invaluable natural resources. This high productive system were socio-economically valuable and sustainable because it provides several important ecosystem services such as flood control and also gives recreational areas, to retain a

Good Environmental Status (GEnS) (Elliott et al., 2015). Each estuary has unique physical features that control its ecology. It includes river discharge, depth and general topography, circulation patterns, climatic condition and tidal variations. These fragile ecosystems are vulnerable to any kind of disturbances. Natural disturbances include winds, tidal currents, waves, and ice. Anthropogenic disturbances include pollution, coastal development, and the invasion of non-native species. The major threat to estuaries includes large-scale alteration by draining, filling, damming or dredging. These actions are causing the sudden damage and modification of estuarine habitats. Degradation of remaining estuaries are continuing unabated. Pollution is considered as the main threat to water quality deterioration in estuaries. It affects most estuarine organisms, including commercially important fish and shellfish. The toxic materials such as chemicals and heavy metals, excess nutrients (eutrophication), pathogens and introduction of invasive species also negatively impacted the estuarine quality. Eutrophication is a common phenomenon in marine and coastal waters. The increase in the concentration of organic matter to an ecosystem (Eutrophication) (Nixon, 1995) was either from external sources or from biological production within the system. Estuaries were naturally eutrophicated due to the accumulation of land-derived nutrients and also the influence of marine water. As a result of eutrophication, a number of negative effects has happened, which include hypoxia, toxic algal bloom, loss of submerged vegetation, fish kill, reduction in water transparency etc. (Fig.1.1). Due to its complex nature and absence of reliable data, the consequences and the degree of eutrophication problem had never been addressed effectively (Bricker et al., 1999).

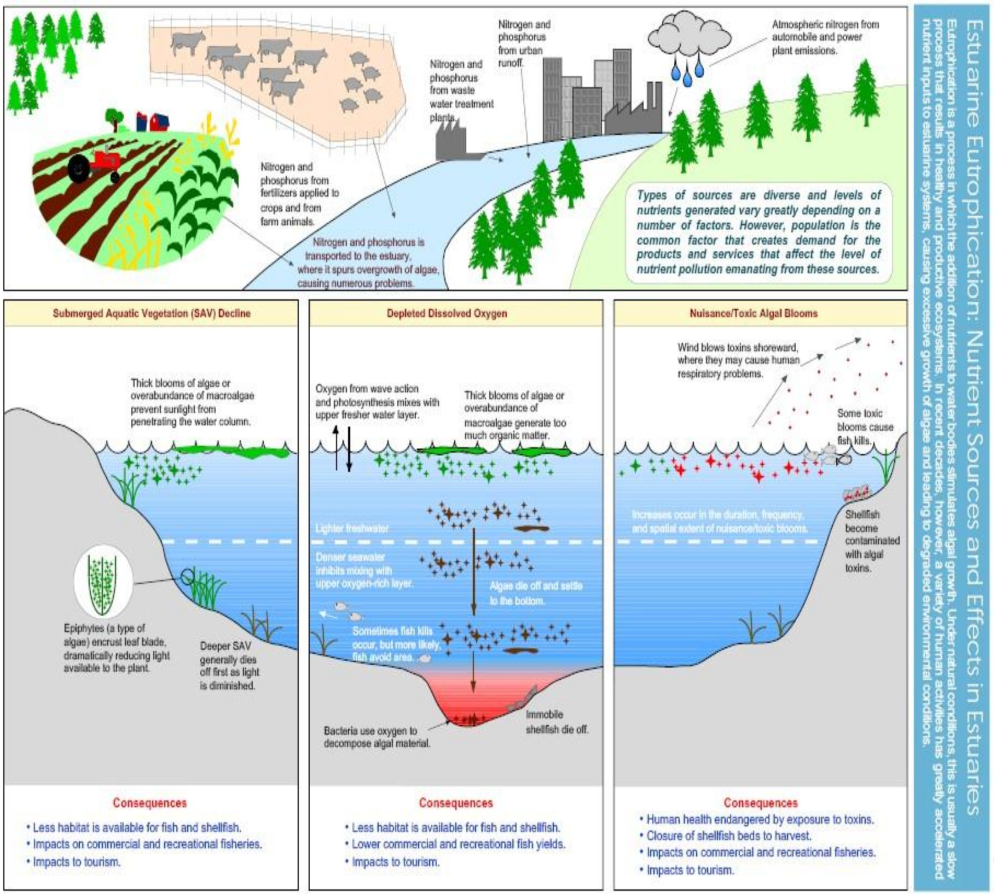


Fig. 1.1 Estuarine eutrophication: Nutrient sources and effects in estuaries (*source*: Bricker et al., 1999)

Estuaries undergoing variation with time and continuously face change in accordance with the natural phenomenon. In several ecological points estuarine ecosystem was more complex than Open Ocean. Estuaries provide direct way to sea via ports and deepwater channels, so various shipping activities (building and repairing) and several engineering works modified the estuarine shores. Periodic dredging of shipping channels greatly impacted the benthic environments and also altered the hydrology of the system through the variation of water velocity, stratification and turbidity (Cooper, 2009).

Climate affects the functioning, distribution and geomorphology of coastal and estuarine systems. Global climate change was likely to form an important factor of variations in wetland ecosystem (UNESCO, 2007). In the estuarine environments, variations in the mean sea levels were estimated to have a major role and the sea-level rise was impacted in several ways (Glamore et al., 2016) and the relative mean sea-level was important for coastal communities (Lewis et al., 2011). Another potential impact of climate change on estuaries was resulted from the changes in physical mixing process affected by the alteration in freshwater runoff (Scavia et al., 2002). The rate of sea level rise has been increased over the past century. By 2100, the global mean sea level was anticipated to rise about 44 to 74 cm (IPCC, 2013). The increasing sea level also directed to the loss of coastal wetlands, which filter nutrients, microbial agents, and chemical agents (McMichael et al., 2006).

Altered flow pattern was considered as the most serious and continuing threat to ecological sustainability of rivers and their associated floodplain wetlands (Ward et al., 1999). Due to impoundment and alteration of flow patterns of water, dams transformed the upstream habitats and bring out variations in the composition of aquatic biota (Baxter, 1977). Damming formed the one of the greatest stressors affecting the integrity of running waters (Garcia de Leaniz, 2008), because it can interfere or even stop the transport of sediment and nutrients along waterways and eventually disturb ecological connectivity, which underpins the transfer of materials and products of ecological functions and processes (Jenkins and Boulton, 2003). Majority of the life history characteristics of species have direct influence upon natural flow pattern, which affected the spawning and reproductive behaviour, larval recruitment and survival, and growth patterns (Bunn and Arthington, 2002). The interaction of flow pattern and physical habitat determine the distribution, abundance and diversity of organisms. Variation in the flow pattern resulted the invasion and succession of exotic species and also formed the taxonomic grouping of fauna and flora. Abundance, richness, and biotic integrity of fish and invertebrate assemblages were always lower in impoundments than in the free-flowing river (Santucci et al., 2005). Dominance of aquatic macrophyte communities in slow flowing and impounded water body, mainly by water hyacinth (*Eichhornia crassipes*) is common. Due to its free-floating habit and quick growth rate, covering large areas of open water, interfere with flow and water transport, disrupting recreation, obstruct the contact of water, and reduce the light penetration (Mitchell and Gopal, 1991). According to Sheaves et al. (2006), the important anthropogenic factor responsible for increasing sea level was the creation of more barriers. In the world, the most remarkable environmental effects were linked with the construction of very large impoundments, particularly in Africa: Lake Volta; Lake Nasser on the Nile, in Egypt; Lake Kariba on the Zambezi; also on the Zambezi, behind the Cobora Bassa Dam in Mozambique (Hall et al., 1976; Jackson and Davies, 1976). The majority of Queensland coastal streams, Australia were blocked by dams, weirs and other impoundments that obstruct fish migration (Sheaves et al., 2006). Dams have spectacular effect on rivers and aquatic biota changing the water quality and habitat, shifting the nutrient pathways, sediment movement and obstructing the fish and invertebrate migration (Santucci et al., 2005). The impact of dams or impoundments remain similar for water bodies of various size (ie. alteration of lotic to lentic environment) but the effect on biotic components were varying with size of the aquatic system and also the temperature pattern (Ward and Stanford, 1983) or with dam size and function (Poff and Hart, 2002).

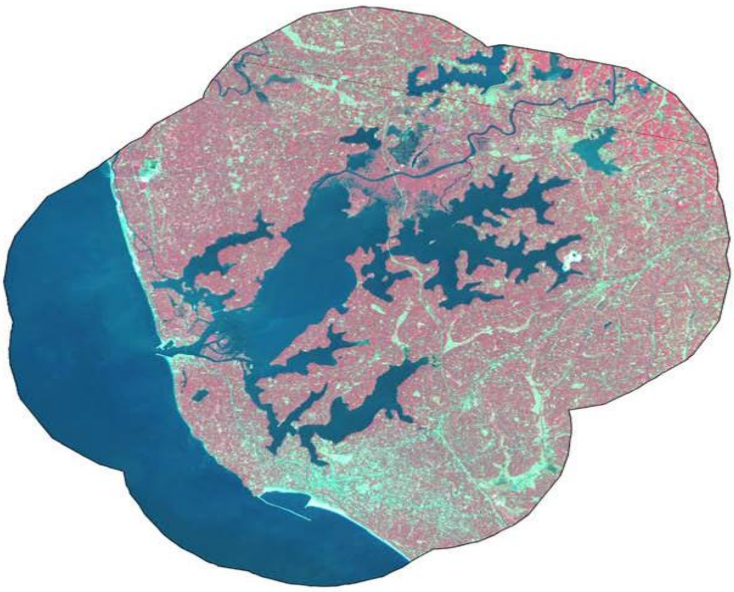
1.1 Coastal wetlands in India

India has long coast line and large area of wetlands coming under the coastal wetlands. The area estimates of various wetland categories for all the coastal states and Union Territories showed a total of 120019 wetlands (NWIA, 2011). In addition, 289459ha small wetlands (< 2.25 ha) have also been identified. Total wetland area estimated is 9.70 Mha which is around 6.94 per cent of the geographic area. Total inland wetlands are 5.58 Mha and coastal wetlands are 4.12 Mha. The most dominant type of coastal wetland is inter-tidal mud flats (2.39 Mha) occupying around 24.7 % of total wetland area. The other major coastal wetlands are mangrove (471407 ha), aquaculture pond (284589 ha), lagoon (246044 ha), creek (206698 ha) salt pan (148913 ha) and coral reef (142003 ha). There are 178 lagoons having area about 246044 ha which is 1.61 % of total wetland area of the country. Orissa has 89023 ha area under lagoons followed by Andhra Pradesh (47407 ha) and Kerala (38442 ha). Inter-tidal mud -flats are observed in all the coastal states except Lakshadweep and Kerala. Gujarat has large area under inter-tidal mud-flats (2260365 ha) followed by Tamil Nadu (33164 ha), and Andhra Pradesh (31767 ha).

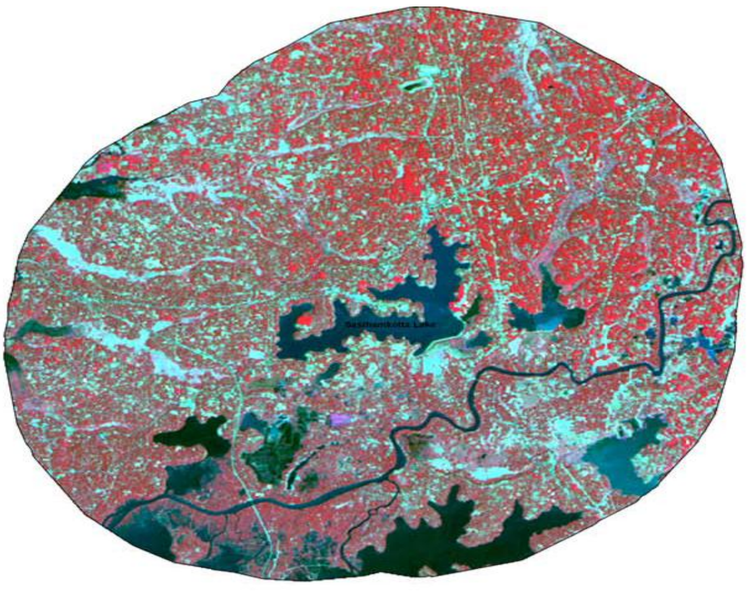
Coastal area of Kerala extending north-south direction with a coast line of 590 kms and width of 35 to 120 km as a barrier strip of land includes a chain of lagoons and backwaters with connection to the sea at various points. All the 44 rivers in the state drain the land, in which 41 are west flowing and join with the Arabian sea and remaining 3 flow east. Apart from these 44 rivers, their tributaries and distributaries and a countless number of streams and rivulets crisscross the land making it green and fertile and also serves as inland waterways. Beside these rivers, Kerala is bestowed with a number of lakes and backwater lagoon which add to the beauty of the land. Govt. of India marks nearly 1762 wetlands in Kerala (SAC, 2010). In addition, 2592 wetlands smaller than 2.25 ha have also been identified and the total wetland area estimated were 160590 ha. The major wetland types are river/stream (65162 ha), lagoons (38442 ha), reservoirs (26167 ha) and waterlogged areas (20305 ha). Among the fourteen districts of Kerala, four districts can be grouped to be rich in wetland habitat. Alappuzha district covering more wetland area with

26079 ha, mostly by the Vembnad kol wetland, the remaining three districts are Ernakulam (25065 ha), Kollam (13703 ha) and Thrissur (13285 ha) districts. The least wetland area was observed in Wayanad district (3866 ha). Vembanad, Ashtamudi and Sasthamkotta wetlands are important Ramsar sites of Kerala (Fig. 1.2). Vembanad-kol wetland is the brackish, tropical wetland ecosystem and typical of large estuarine system on the western coast.

a)



b)



c)

Fig. 1.2 IRS LISS-III FCC - 5 km buffer area of a) Vembanad, b) Ashtamudi, c) Sasthamkotta Wetland system *(source:* National Wetland Atlas: Kerala, 2010

1.2 Degradation of coastal wetlands

Humankind has been draining, in-filling and converting both coastal and inland wetlands for many centuries, as reported from Roman times in Europe (Davidson et al., 1991); at least the 17th century in North America (Dahl, 1990) and Southern Africa (Kotze et al., 1995); and for at least 2000 years in China (An et al., 2007). This conversion and degradation of wetlands continues with increasing economic and human population growth, and it abruptly causes alteration from extensive and intensive agriculture and aquaculture, water abstraction and major hydroengineering projects, increasing urbanization and infrastructure development, spread of invasive species, sea defences, port and industrial developments (Finlayson et al., 2005; Asselen et al., 2013). Globally the wetlands have declined from 64-71% in the 20th century and its degradation continues worldwide and is extreme in Asia (Davidson, 2014). According to Airoldi and Beck (2007) the greatest impacts to wetlands have consistently been reclamation and coastal development. However, coastal development and defence establishments, have had the greatest known impacts on soft-sediment. Most of the countries there have been estimated losses of coastal wetlands and seagrasses exceeding 50% of the original area with peaks above 80% for many regions. According to Dahl, (1990) the coastal wetlands of Southern California has lost its whole habitat and 91% of the world area (coastal and inland) has been transformed for different needs. Similarly many European coastal habitats have been lost or severely degraded, and it was found that only a small percentage of the European coastline (<15%) is in a healthy condition (EEA, 1999). In Mediterranean countries the surface area of natural coastal wetlands decreased by 10% from 1975 to 2005, losing 1,248 km^2. Wet meadows and marshes lost its surface area by 43% and 10% respectively in which construction of new dams and other climatic situation enhances the habitat loss (Mediterranean Wetlands Observatory, 2014). In the beginning of the 18th century, wetland conversion and loss in the long term was in excess of

50% later as much as 87% in 20th century, further the loss of wetland was four times faster than earlier, with losses of up to 70% of wetlands existing in 1900 AD, the transformation of coastal natural wetlands increased more than that of inland natural wetlands in the 20th century; and the alteration and loss was ongoing in all parts of the world, quickly in Asia and so the fate of the world's remaining wetlands was doubtful (Davidson, 2014). So Contracting Parties and their policymakers have been urged to take immediate action to meet the Ramsar Convention's objective to stop and reverse the loss and degradation of wetlands and services to people (Ramsar Briefing Note, 2015).

The poor, economic, social and environmental effect of declining water quality was a major concern for Indian wetlands. Many fresh water wetlands in India are in a threatened state and several are in a declining condition due to increased urbanization, population expansion, and economic activities (Central Pollution Control Board, 2008). In India, wetland tourism contributes a major share to Gross Domestic Product (GDP) and employment (Government of India, 2012). Annually an average of seven million tourist visit Kerala's backwaters, beaches and wildlife sanctuaries; three million visit Uttarakhand's lakes and other natural wetlands; one million visit Dal lake; and 20,000 visit lake Tsomoriri. Similarly wetlands in India play a significant role in fish production and its production increased from 0.2 million tonne in 1950–1951 to about 5.1 million tonne in 2010–2011 (Bassi et al., 2014). Based on the bio-geo-chemical processes and hydrology, wetlands act as a net sequesters or producers of greenhouse gases. The coastal wetlands in India have a great role in carbon sequestration and the Vembanad wetland releases up to 193.2 $mg/m^2/h$ of CH_4 (Verma et al., 2002). Poor governance and management was also a major problem in deteriorating these water bodies (Kumar et al., 2013). Most of the Indian wetlands were polluted due to agricultural runoff and untreated sewage discharges and other waste from urban areas. Due to the population explosion during 90 year period from 1901 to 1991 in India, the urban population has increased eightfold in India (Bassi and Kumar, 2012). This level of increase created great pressure on wetlands and flood plain areas, which formed large water and food demand for growing population in India. Water in most Asian rivers, lakes, streams and wetlands has been heavily degraded mainly due to the agriculture runoff of pesticides and fertilizers, untreated industrial and municipal wastewater discharges, all of which caused extensive eutrophication (Prasad et al., 2002). It is estimated that, due to the climate change related sea level rise of upto 1m will possibly decline around 84 % of coastal wetlands and 13 % of saline wetlands in India (Blankespoor et al., 2012). This will adversely affect the wetland and migratory species which utilize the wetland habitats for completing their life cycle. Even though India is included as a signatory to Ramsar Convention on Wetlands and also for enforced Wetland (Conversation and Management) Rules in 2010, but no significant improvement has occurred on the conservation and wise use of wetlands. It is complained that only few wetlands get the consideration from the national conservation policy process such as National Wetland Conservation Programme, National Lake Conservation Plan etc. (Bassi et al., 2014).

1.3 Structural and functional characteristics of benthic fauna

The functioning of an ecosystem is the interaction between the ecological processes which characterize the environment and the organisms which inhabit it. Benthos plays a crucial role in the functioning of estuaries. Coastal estuaries/wetlands are highly productive systems and their sediments are permanently or periodically inhabited by diverse assemblages of benthic organisms. The benthic invertebrates in aquatic ecosystems, plays an important role in the processing and transformation of sediment organic matter on the bottom to its base elements and subsequently that contribute to higher organisms (Fig. 1.3).

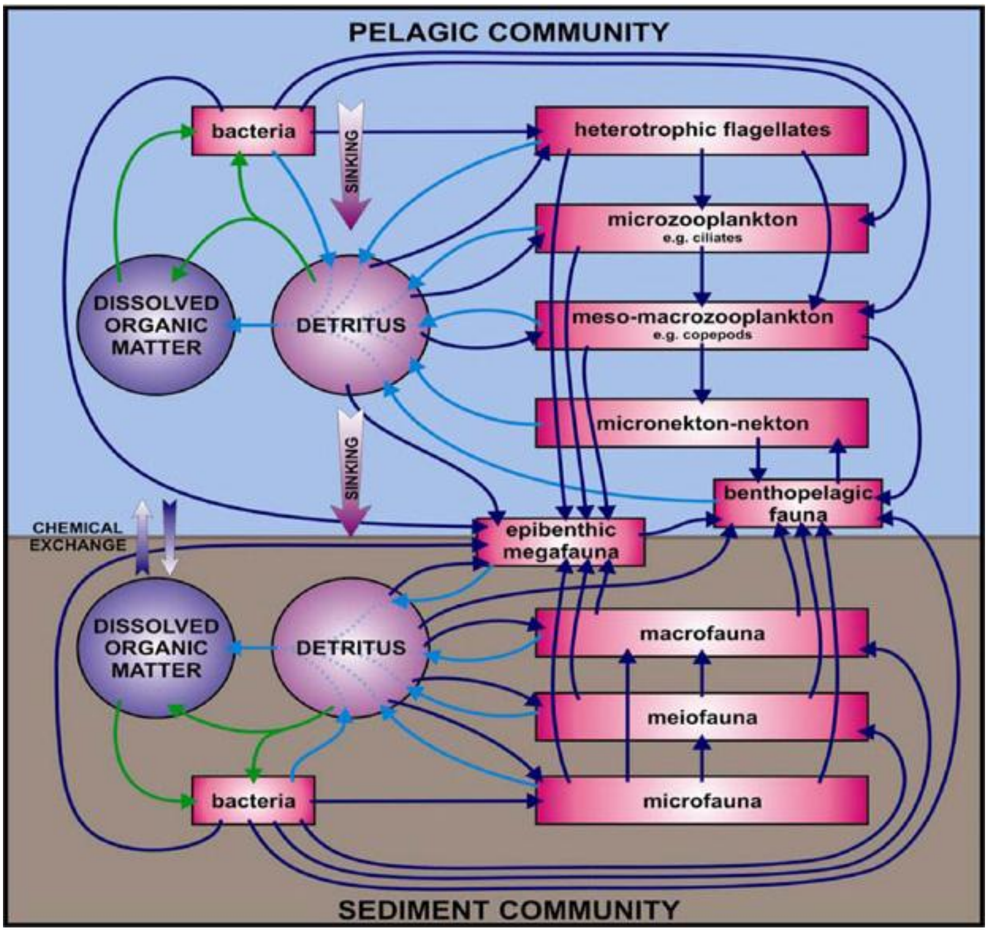


Fig. 1.3 Bottom boundary level ecosystem model (It does not include the photic zone. Purple arrows denote – feeding, blue - egestion and/or mortality, green arrows-microbial loop; respiration is not shown) (*source*: www.hub.globalccsinstitute.com)

These organisms range in size from the minute bacteria and protozoans to larger colonial animals as macrobenthos. The organisms based on size are divided into three groups- macrobenthos are those organisms which retained in a sieve of mesh size 0.5mm. These are the largest benthic animals including star fish, mussels, most clams etc. Meiobenthos are smaller than 0.5mm but larger the microbenthos, which are less than 0.1mm in size (Levinton, 2001). These are involved in burrowing activities that regulate pathways and rates of organic matter and other soil components (carbon, nitrogen, phosphorus etc.) by mineralisation process in bioturbated sediments. This diagenetic process is a source of energy to the benthic organisms and also influences the biogeochemical properties and their responses in the benthic environment. Similarly the functional biodiversity often influence essential ecosystem structure and functions related to the storing and cycling of organic material (Hooper et al., 2005). Organisms that are not consumed in the water column eventually die and sink to the seafloor to fuel the benthic community. Benthic-pelagic coupling, which refers to the linkages between benthic and pelagic environments in aquatic systems, play a major role in determining the production and biological structure of the aquatic systems (Valiela, 1995). The composition of benthic fauna has largely been considered as a good indicator of water quality because, unlike planktonic species, they form relatively stable communities in the sediments which do not change over a long time intervals and reflect characteristics of both sediments and upper water layer (El-Shabrawy and Khalil, 2003).

Globally the increased decline of aquatic health and rapid diminution of aquatic resources and biodiversity caused by human activities and climate change, directed to increased attention and anxiety. So several legislations has been taken on to determine the ecological integrity or status of various aquatic systems. The most important factor in determining ecological integrity is monitoring of biological integrity, which normally give emphasis to study of plankton, benthos, macroalgae and fishes, in which benthic macroinvertebrate communities are the most consistently emphasized biotic component of aquatic ecosystems (Borja and Daeur, 2008). So any type of alterations in the aquatic environment emphasizes the effective need for monitoring the ecology of benthos, the receptive indicators of such variation.

1.4 Significance of the Study

In the context of wetland conservation and management, the Vembanad wetland, a Ramsar site, on the south west coast of India is an important repository for biodiversity and ecosystem services. The Vembanad estuary with an area of providing the life support system for various aquatic organisms and the benefits to the livelihood sustenance. The region around Vembanad estuary (backwater) is one of the most thickly populated areas of Kerala and the estuary is the single ecosystem only next to the Arabian Sea in terms of supporting maximum livelihood activities. These backwaters have been considered as subservient to water needs for the developmental sectors and therefore heavily fragmented. The degradation of wetlands, therefore, creates a great threat for sustainability for economic growth for the entire region. The biological importance of Vembanad wetland ecosystem is to be viewed and assessed in this context.

Over the past few decades, significance in biological diversity and rising numerous human activities have led to the development of applied ecological research and impact studies on the macrobenthic communities residing in coastal and estuarine areas. The benthos, because of its sedentary nature has been affected directly by many of these perturbations and has also been the focus of numerous studies to assess their impacts. Thus it is imperative to have knowledge of the natural, spatial and temporal variability of benthic communities in order to derive the changes that may occur due to a man-induced modification of the environment and the subsequent recovery of the communities. The physical, chemical and geological process by both natural and manmade effects, disturb benthic communities. The abundance of benthic invertebrate species was closely correlated with nature of bottom sediments, organic matter, salinity and various anthropogenic activities including coastal developmental activities and intensive fishing practices etc. All these affects the standing crop of benthic fauna, so establishing a monitoring program to follow up the changes in the aquatic ecosystem will be of great help in the management of such important water body. The coastal wetland habitats of the country are facing serious anthropogenic impacts that have remarkably influenced the biodiversity and ecosystem character of the organisms inhabiting the benthic zone. Vembanad estuary is facing severe ecological problems due to massive reclamation associated with rapid urbanization and industrialization. Thaneermukkom barrage the salinity barrier, constructed across the Vembanad wetland system in 1975, has transformed the water body into two distinct ecosystems, a fresh water zone on the south and a brackish water zone on the north, resulting in gross changes in physical, chemical and biological characteristics of the ecosystem. Increased agriculture, urban development, industrialization, and tourism development caused the shrinkage of estuary both horizontally and vertically resulting its drastic environmental alterations (Gopalan et al., 1983; Sarala Devi, 1986; Menon et al., 2000; Balachandran et al., 2006; James, 2011; Dinesh Kumar et al., 2014). Apart from this, sewage pollution with increasing faecal coliforms, in conjunction with intense tourism related activities has seriously afflicted the estuary and major parts of the wetland by pollution problems from indiscriminate operation of house boats, waste dumping, oil and other contaminants in the ecosystem. So in this prevailing situation, understanding the trophic character and the organismal dynamics is all the more relevant. Benthic fauna form an important link in the transfer, mineralisation and re-mineralisation of particulate and dissolved organic matter in different levels of trophic food web maintaining the energy balance and carbon structuring of the aquatic system. But, the ecological imbalances, as noticed in Vembanad estuary can inflict long term damage on the ecology of the benthic fauna and associated communities possibly having its adverse repercussions on the trophic stability possibly even affecting the climate regime of the ecosystem.

Due to the lack of a comprehensive and worthy studies on the ecology and ecosystem production potential from the Vembanad estuarine system, based on long term data sets, the Ministry of Environment, Forest and Climate Change (MOEFCC), Govt. of India, funded a major research project on "**Ecosystem Based Monitoring and Modelling of the Vembanad Kol Wetland in Kuttanad, Kerala, India"** from 2010 to 2014 period in the Department of Marine Biology, Microbiology and Biochemistry, under the Principal Investigatorship of Prof. (Dr.) S. Bijoy Nandan. This Ph.D thesis and its significance has emoted as an important concept for the research project. Thus the objectives of the Ph.D thesis are as given below.

- Assess the physico- chemical characteristics of Vembanad estuarine system.
- Study the composition, distribution and abundance of meio and macrofauna, and its spatio - temporal variation.
- Assess the community structure and ecology of polychaetes and crustaceans.
- Establishing the ecological and benthic functionality in the estuary.

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Chapter 2

MATERIALS AND METHODS

Contents

2.1 Study Area

Wetlands are unique, productive ecosystems where terrestrial and aquatic habitats meet, sustaining many natural cycles and also supporting wide range of biodiversity entities. Wetlands perform numerous vital functions and, thus, need to be looked after and used wisely. They help in water storage and purification, flood control, ground water replenishment etc. It act as nurseries for freshwater and marine fish, provide shoreline stabilization and protection against nutrient and sediment retention and support biological diversity, mitigate effects of climate change, pollution and are also resources for recreation, tourism, transport and other services. Vemband-Kol wetland system, Ashtamudi and Sasthamkotta, are the three designated Ramsar sites of Kerala. Vembanad-Kol wetland is a large complex coastal lagoon in Kerala state. It represents the largest wetland system on the Arabian Sea coast (Malabar coastline) (09° 00' – 10° 40' N and 76° 00' -77° 30' E). It consists of Vembanad estuary bordered by Kuttanad on south and Kol lands on north, which are interlinked by river estuaries and mangrove marshes, all interconnected by a complex network of natural and manmade channels spreading over 1,780 km^2. Vembanad-Kol wetlands extend along the coastline of Alappuzha, Ernakulam and Thrissur districts over 145 km (WISA, 2013). It is a multi-functional ecosystem supporting rich biodiversity as well as livelihoods of dependent communities. It is aligned north-south, parallel to the coastline and widest at the southern side. Vembanad - Kol wetland has been designated as a Ramsar Site on 19th August 2002 based on the fulfillment of criteria 4, 5 and 6 of Ramsar convention. The Criterion 4 states that, a wetland should be considered internationally important if it supports plant and/or animal species at a critical stage in their life cycles, or provides refuge during adverse conditions. Criterion 5 encompass that, a wetland should be regularly supporting 20,000 or more water birds. Criterion 6 states that a wetland should be regularly supporting 1% of the individuals in a population of one species or subspecies of water bird.

Kuttanad is part of the Vembanad- Kol wetland system that comprises of marshy low lying area below mean sea level (1.5 to 2 m), coastal alluvial belt, uplands of higher elevation, river networks and backwaters, contributing to a unique ecology. The ecological character of the Kuttanad wetland is the capacity to provide a wide range of naturally occurring ecosystem services (MSSRF, 2007). Kuttanad, the rice bowl of Kerala, also commonly known as Kuttanad wetland system, included paddy fields; a marsh, lakes and rivers etc. located around the Vembanad Lake, and includes 304 km^2 of garden lands and 524 km^2 of low lying rice fields, the rest being aquatic systems (Thampatti and Padmakumar, 1999). The region extends from 9°17' to 9°40' N and 76°19' to 76°33' E and most of the region lying 0.6 - 2.2 m below mean sea level (Sreejith, 2013), water logged throughout the year, that is facing flood submergence during monsoon period and salt water intrusion during summer. Kuttanad region comprised of 79 revenue villages, 10 Taluks; Cherthala, Ambalapuzha, Chengannur, Kuttanad, Karthikappally and Mavelikara Taluks in Alappuzha districts, Thiruvalla taluk in Pathanamthitta District and Changanassery, Vaikom and Kottayam taluks in Kottayam districts covering an area of 870 Sq. km (Hazard Center and People's Science Institute, 2006). Above all, it is a deltaic formation of five river systems- Meenachil, Pamba, Manimala, Muvattupuzha and Achencovil, situated in the low-lying fertile areas of Vembanad lake (Velupillai, 1996). The Achenkovil-Pampa-Manimala River system feeds south and south-western parts of Kuttanad and the Meenachil feeds the eastern parts of Kuttanad. The Muvattupuzha River and the smaller Kariyar River feeds the Vaikkom area, which are in the north-eastern parts of Kuttanad wetland system (Fig. 2.1**)**. These rivers join into the Vembanad backwaters through a network of water ways and canals, and flow in a south to north direction, and then travel further north towards Cochin barmouth and empty into the Arabian Sea (MSSRF, 2007).

2.1.1 History of Kuttanad wetland system

In earlier days, the 'Kuttanad' was referred to a much larger area than what it represents now. Formerly the area in Kuttanad ranged from Karunagappally to Alwaye. Kuttanad was mentioned in Tamil literature like Venpai and Tholkappiyom as 12 Nadu-s (principalities), where the people spoke 'Kodumthamil' (non grammatical Tamil) and also it was cited in the great Thiruvaymozhi written in the 8^{th} century A.D by the renowned Vaishnavite Saint Nammalvar and in Periyapuranam (the great purana or epic) of the 11^{th} century A.D (Aravindakshan and Joseph, 1990). There are different myths and assumption on the origin of Kuttanad. Generally it is believed that the entire Kuttanad was reclaimed from waters at different periods; while "God created earth and waters, people created Kuttanad by raising it from waters" (Ramakrishna Pillai, 1974; Tharamangalam, 1981). Earlier period, Kuttanad region was a part of the shallow coastal region of the Arabian Sea. Due to geological uplift a shallow bay was formed, into which several rivers was discharged. The settled silt on the river mouth formed the current delta and the bay shaped into a lake-lagoon estuarine system connected to the Arabian sea through the Kochi barmouth (Fig. 2.2) Another assumption was, the entire area was a dense forest, legendary 'Khandava vana', of reference in the epic Mahabharatha was situated. Based on the myth which was destroyed completely in fire and later on, the area was submerged by the rising sea and afterward the sea water gradually retreat by exposing the land. This land known as 'Chutta nadu' (burned land) and later it named it as Kuttanad. Mining of bottom exposed that the whole area has characteristics of dense forest having high organic residues and remaining of burned wooden logs locally known as 'Kari'. Even now the durable burned logs are seen in the Karinilams of Vaikom, Thuravoor, Thakazhy, Purakkad etc. Based on the features of elevation, geological formation and soil quality, the formed Kuttanad deltaic area was grouped as karappadam, kayal (Malayalam term for a lake) and kari lands (black paddy fields) (Thampatti and Padmakumar, 1999). Based on the nature of bottom there are number of villages in Kuttanad having the name carrying the suffix kari- Ramankari, Puthukari, Mitrakari, Kainakari etc. According to Geologists, Kuttanad characterized a "Recent Sedimentary Formation". Due to the uplift of Warkallai Laterite formation, the tract of the recent formation became elevated forming a bay into which the rivers, draining from the mountains to the east, discharged their water. The silt carried by these rivers, deposited at their mouths gradually gave rise to the present sea coast, converting the shallow bay into an extensive lake-lagoon backwater track. These water bodies gradually silted up and formed the sedimentary structures which were transformed into rice fields and cultivatable lands by various reclamations (Aravindakshan and Joseph, 1990). The deltaic formation of rivers formed the Kuttanad and the deeper portions of the lagoon formed the present backwater. Another saying is that the name Kuttanad was connected with 'Karumadi kuttan', a stone engraved sculpture of Lord Buddha located at Karumadi. Years later, the local term 'place of Kuttan' was derived as

Kuttanad shortened/ abridged. Another legend is that early "Cheras" (an ancient Dravidian royal dynasty of Tamil origin) had their home in Kuttanad and they were called as "Kuttuvans" named after this place (Sylas, 2010).

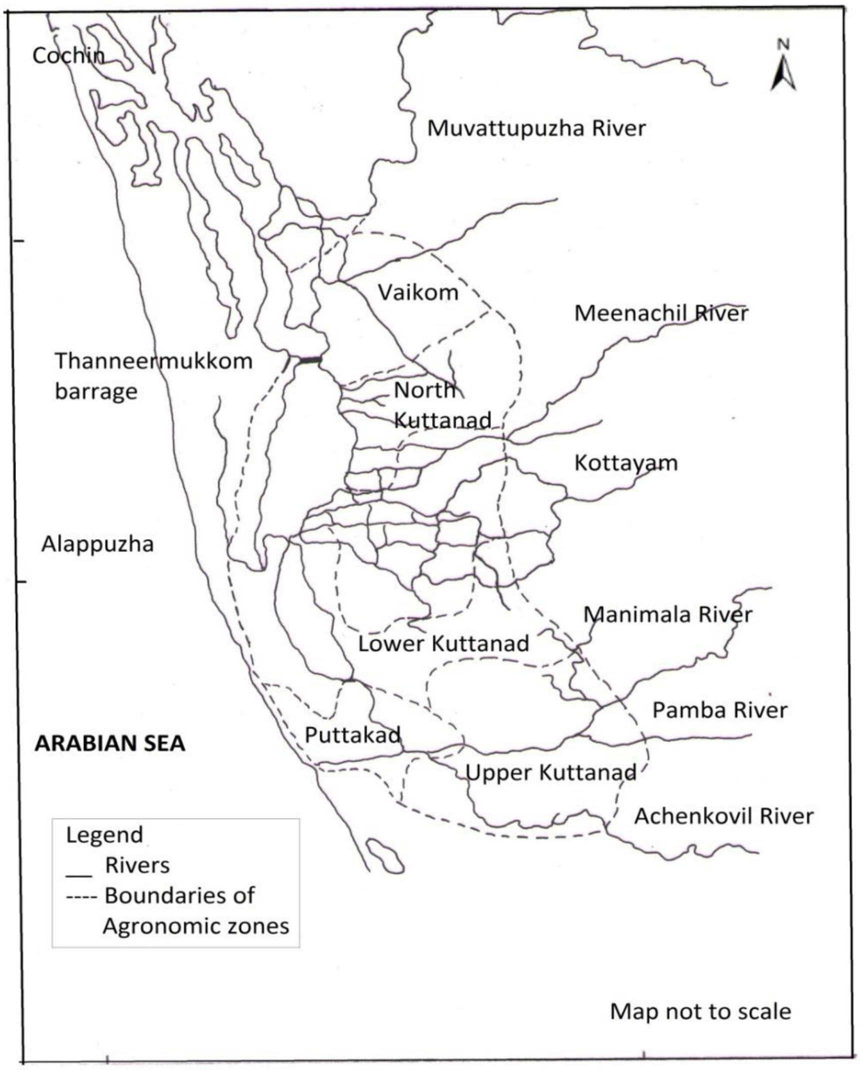


Fig. 2.1 River map of Vembanad backwater showing the tributaries and catchment area (*source*: modified from KWBS, 1989)

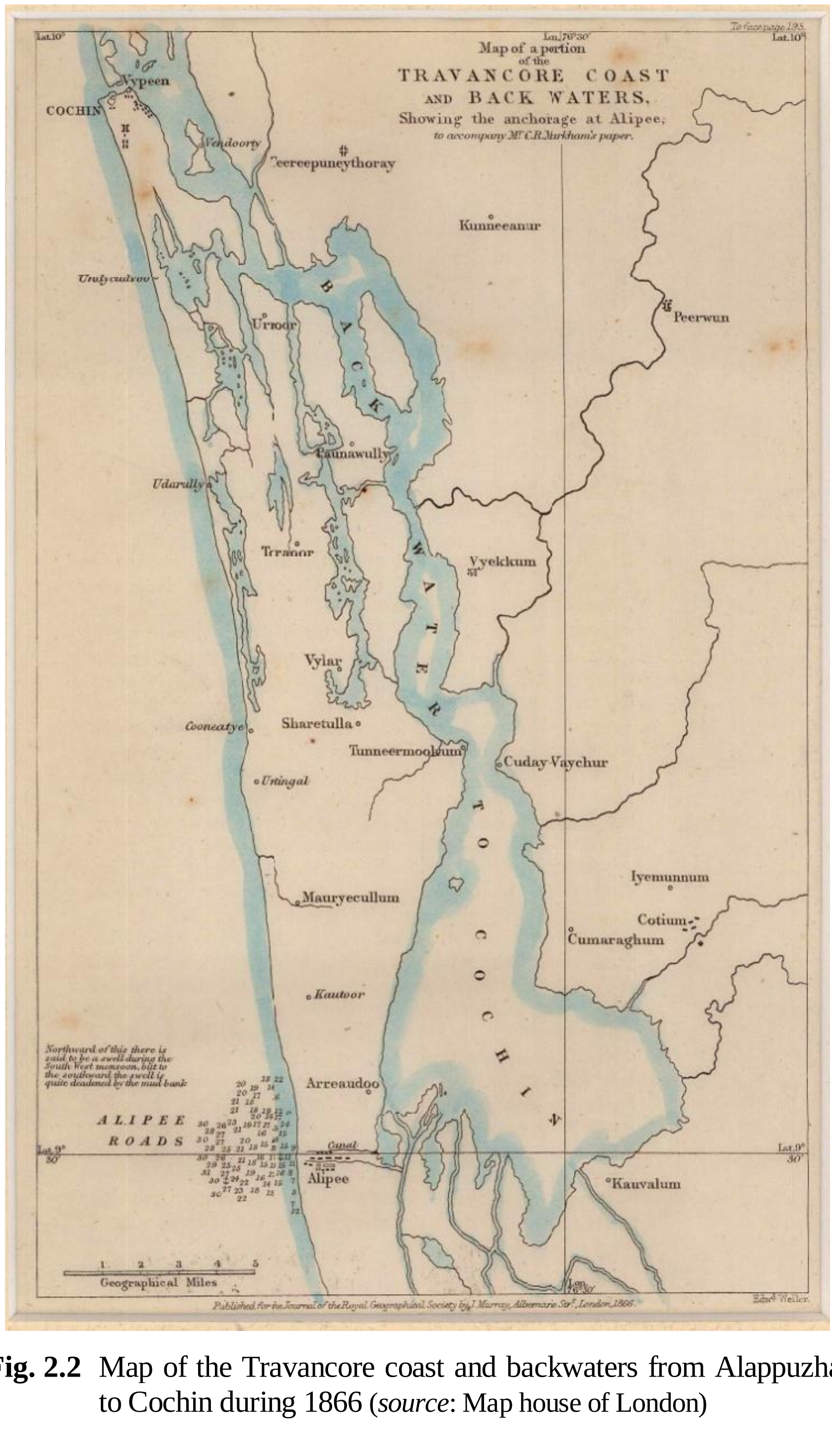


Fig. 2.2 Map of the Travancore coast and backwaters from Alappuzha to Cochin during 1866 (*source*: Map house of London)

2.1.2 Agriculture in Kuttanad and reclamation of lake

The golden era of Kuttanad Paddy cultivation started with the construction of a harbour at Alappuzha by Sri "Raja Kesava Das" the Divan of then Travancore "Empire Ramavarma". It was through these water ways of Kuttanad that spices, condiments and other exportable commodities from the Western Ghats reached Alappuzha harbour. Till that time "Nanchinaad" (now in Kanyakumari district of Tamil Nadu) was known as the rice bowl of Kerala. The rulers of Travancore realized the necessity of another rice bowl to meet the increasing demand for food grains which transformed Kuttanad into the rice bowl of Kerala. The history of paddy cultivation in Kuttanad can be traced back centuries and was performed in reclaimed portions of the shallow part of the Vembanad lake or from the periphery of the Pamba River. These reclamations constituted small areas of paddy fields called *padsekharams* (paddy lands). In 1912, the Madras Government approved a proposal from the Travancore Government for reclamations in three stages. Under this scheme kayal land was notified for reclamation in blocks each named with a letter of the English alphabet. These pioneering reclamation activities in kayal cultivation were made by the two brothers Mathai Luka Pallithanam and Ouseph Luka Pallithanam from Kainady village in Kuttanad. Out of the total area of 19,500 acres of kayal land, 12,000 acres were reclaimed between 1913 and 1920. After the removal of the ban in 1913, E-block Kayal measuring about 2,400 acres was reclaimed by Pallithanam Luca Matthai along with his partners and was known to be the biggest Kayal nilam (lake areas) in Kuttanad. The reclamations between 1914 and 1920, known as the new reclamations, were carried out in three phases (Table 2.1).

Pallithanam Luca Matthai, who had served as member of Maharajah Moolam Thirunal's Praja Sabha (Popular Assembly) was considered as the pioneer of cooperative agricultural movement in Kuttanad. His life marked the beginning of the epoch of first generation 'Kayal Raja's' of Kuttanad (Kayal Raja is the term generally used to refer to the prominent kayal cultivators in the Kuttanad region). In 1931, in order to strengthen the farming community in Kuttanad, he founded Kuttandu Karshaka Sangham (Kuttanad Agricultural Association). Due to the steep decline in the price of rice between 1920 and 1940 reclamation activities slowed down, but they gained momentum again in the early 1940s. During this period, in order to increase agricultural output, a government initiated "Grow More Food" campaign and the provision of incentives encouraged new reclamations. The advent of electric motors made reclamation relatively easier, cheaper and less risky as compared to in earlier periods. The last tract of the reclamations namely Q, S and T block were made during this period by Thomman Joseph Murickummoottil (Muricken Outhachan). R block is the largest cultivation area which covers to about 1400 acres **(**Fig. 2.3**)**. The construction of permanent outer bunds around the R Block kayal lands under the R Block - Holland Project that was started in 1961 is a landmark in the history of paddy cultivation in Kuttanad. The embankments built earlier as part of reclamation were made of mud and were not strong enough to withstand the incessant wave action. Most of them were submersible under floodwaters during the monsoon months. Under the R Block - Holland Scheme permanent and non-submersible bunds that stood six feet above the MSL with a top width of ten feet and a total length of 10.4 kilometers were erected around the R Block kayal lands. Later in 1970's the Kerala Land Development Corporation (KLDC) initiated a Bund Improvement Scheme for the repair and strengthening of ring bunds in kayal lands. However, the project had been dropped half way due to the paucity of sufficient funds (Thomas, 2002).

Table 2.1 Different phases of reclamation of Kayal lands in Kuttanad

1[st]Phase	**A-G Blocks, 6300 acres** E-Bloc - Erupathinalayiram Kayal (Biggest Kayal nilam - 2,400 acres), C-Block and D-Block - Attumukham Aarayiram (Attumuttu Kayal), Thekke Aarayiram and Vadakke Aarayiram Kayal, F-Block - Judge's Aarayiram Kayal, G-Block - Kochu Kayal
2[nd] Phase	**H - N Blocks, 3600 acres** Reclaimed by Pallithanam Luca Matthai, Cunnumpurathu Kurien, Vachaparampil Mathen, Pazhayaparmpil Chacko, Kunnathusseril Peious and Ettuparayil Xavier.
3[rd]Phase	**R Block** 1,400 acres Reclaimed by Sree Moolam Popular Assembly, Pallithanam Luca Matthai, Vachaparampil Mathen, Pazhayaparmpil Chacko, Ettuparayil Xavier, Meledom, Paruthickal, Pattasseril and Kandakudy.

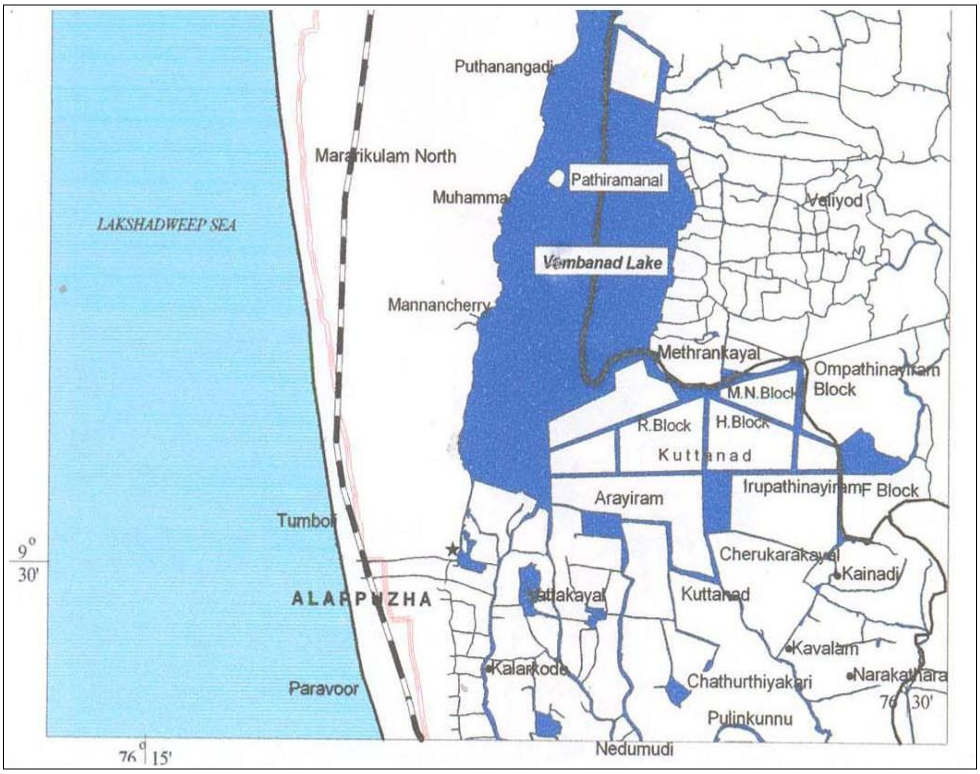


Fig. 2.3 Map showing the reclaimed estuarine lands

Kuttanad has an area of 1,10, 000 ha and is divided into four ecological zones. It include Garden or dry land with approximate area of 31, 000 ha, wetlands with area 11,000 ha, reclaimed land having 55,000 ha which is situated below sea level and water bodies including lake and canals etc. having area of 13,000 ha. Garden land is an important area for plantation crops situated above MSL of 0.5 to 2.5m and the dry lands were not facing floods or from saline water intrusion. The major portion of wetland areas (66,000 ha) are reclaimed from the backwater for paddy cultivation, which is situated either above MSL or below MSL. From this, 55,000 ha of reclaimed area situated below MSL are called as punja lands including cluster of fields are called padasekharams. Remaining 11,000 ha of wetland is part of Upper Kuttanad, where no influence of floods and salinity intrusion was reported. Agriculture is the main economic activity employing about 40% of the population. Kuttanad – the 'rice bowl of Kerala' has been declared as a Globally Important Agricultural Heritage Systems by United Nation's Food and Agriculture Organisation for its below sea-level farming system (Anon, 2013a). Punja lands of Kuttanad were categorized as Karappadam lands having 33,000 ha area, Kayal land with 13,000 ha area and Kari land with area of 9000 ha. Karappadam lands having alluvial soils, situated along the waterways, canals and included the lower reaches of eastern and southern boundary. During the flooding period, large amount of silt was deposited in the area. Kayal lands mainly included the padashekharams situated between 1.5 to 2.2 m below MSL. The area is situated in the revenue villages Chennenkari, Kainakari, Pulimkunnu of Kuttanad Taluk and Thiruvarppu and southern regions of Kumarakom of Kottayam Taluk. Kari lands having black peaty soils situated at or below MSL of north (Vaikom) west (Cherthala) and south-west (Purakkad) of Kuttanad. The soil in Kuttanad is a mixture of silt and clay in varying proportions. The soils of the area are highly acidic, saline and high in organic carbon content. Several parts of this delta have subsoil layers containing pyrites which on oxidation produces severe acidity (Manorama Thampatti and Jose, 2000).

2.1.3 Kuttanad Agro-ecological Zones

Based on the agro-ecological and climatic features, Kuttanad is classified into six agro-ecological zones. The distinctiveness includes mean sea level range, river influence, flooding, salt water intrusion, soil characteristics, fertility and cropping pattern (Fig. 2.4).

1) Upper Kuttanad: It is situated on the south eastern side of Kuttanad which includes comparatively high lands and covering 10,576 ha. The elevation varies from 0.5 to 6 m above MSL and bund levels vary from 0.3 m to 5.0 m above MSL. The zone experiences low risk of saline intrusion and flood. Rivers such as Achenkovil, Pampa and Manimala enter in this zone before flowing into Vembanad backwater.

2) Purakkad Kari: Situated on the western side of Upper Kuttanad having an area of 3,500 ha and spread across 43 *padasekharams* over four panchayats of Ambalappuzha and Karthikappally. This zone is close to the sea and located 1.5-2.0 m below MSL. This zone is undergoing the risk of floods and salt water intrusion through Thottapally spillway and Ambalappuzha- Thakazhy canals. Water and sediment of the area are highly acidic due to peaty soil (partially decomposed organic matter).

3) Lower Kuttanad: Is the main part of Kuttanad, having 16,280 ha area and situated on the southeastern side of Vembanad backwater. The elevations of padashekharams vary from 1.5 m below to 1.0 m above MSL and the bund levels at 0.3 to 1.3 m above MSL. This zone experiences a high risk of flood and salt

water intrusion and also noticeable with the presence of small islands with human habitation.

4) Kayal lands: Located on the north of Lower Kuttanad having 9,464 ha area at elevations 1.0 m to 2.0 m below MSL with bund levels ranging from 0.6 to 1.1 m above MSL. Most of the lands were reclaimed from the shallow portion of south eastern part of Vembanad lake. More than 600 *padashekharams* are included in this zone and is experienced with the risk of flood and saline water intrusion.

5) North Kuttanad: This zone is having an area of 6,556 ha, located on the north and eastern side of Vembanad *Kayal* falling in Kottayam district, including the Kumarakom town. The land is formed by deltaic formation of Meenachil River and its distributaries. The elevation of the western side was 0.5m below MSL and the eastern side was 1m above MSL. Salt water intrusion was the major risk faced by this zone.

6) Vaikom Kari: It is located on the northern most part of Kuttanad, ie northern side of Thanneermukkom barrage, spreading an area of 7,748 ha. The western portion having an elevation of 0.5m below MSL and in eastern portions it reaches upto 6m above. The zone is experienced with high risk of salt water intrusion during summer. Due to the presence of high organic carbon content, the soil of the area is black in colour. The soils of the area are highly acidic, heavy texture, hypoxic and low fertile in nature. Partially decayed (peat logs) wood is seen in this area.

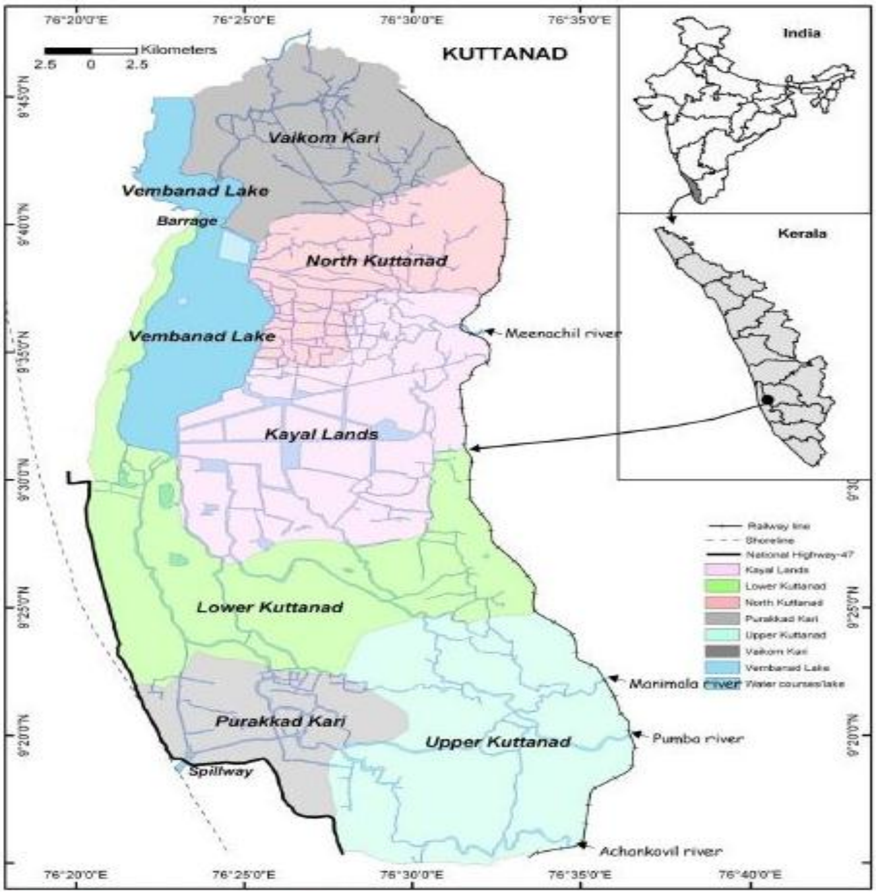


Fig. 2.4 Divisions of Kuttanad wetlands (*source*: Narayanan et al., 2011)

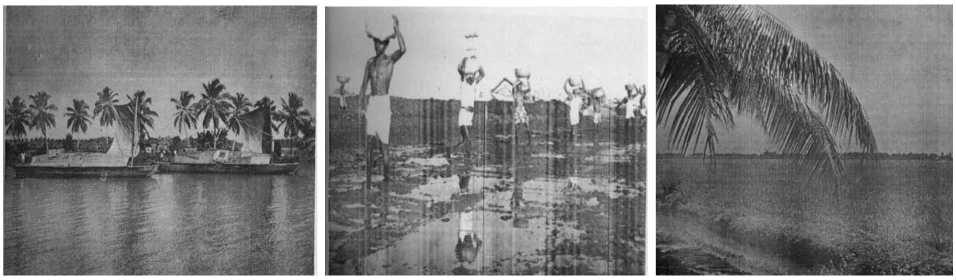
The rivers and the topography divide Kuttanad into three major ecological zones – highlands, lowlands and backwaters. Sandy beach ridges close to the sea have an elevation of 1.5 to 5 m above MSL. Upper Kuttanad has elevations of -0.5 m to +6.0 m MSL and elevations in lower Kuttanad areas range from -1.5 m to +1.0 m MSL. The backwaters or the Kayal lands are at elevations -1.0 m to -2.2 m MSL (MSSRF, 2007). According to MSSRF (2007) the rainfall and evapo-transpiration study in Kuttanad showed that the actual wet season in Kuttanad is only about 3-4 months, from mid-May to mid- August, where as a water deficit condition occurred in the remaining months. During this period, the tidal entry increases and reduces the water level in Kuttanad by 1.22 m within 6 hrs and this causes saline water intrusion into the lake. The intermixing of saline and fresh water in Kuttanad was varied annually and it mainly depend upon the rainfall and inflow of rivers. During January to April-May a brackish water condition was prevailed in Kuttanad.

Agriculture is the major economic activity in Kuttanad and other parts of Alappuzha. Rice and coconut are major crops contributing to about 80 % of agricultural income (Plate 2.1 and Plate 2.2). Area occupied by paddy and coconut is 38 and 45 % of the cropped area, respectively. Other crops include banana, tubers and vegetables. Dairying is not common in Kuttanad due to shortage of fodder, forages, and high price of concentrates. Kuttanad has highest number of ducks in the State and duck farming is also constrained with non-availability of paddy fields for foraging and affliction of serious diseases. Increased pesticide load in the water caused immune-suppression in ducks. The local duck breeds, 'Chara' and 'Champally' having better resistance (MSSRF, 2007). While capture fishing from water bodies is a major livelihood, culture fishing is not common, despite availability of water bodies. The culture fishing in paddy fields of wetland and *Pokkali* area under 'one paddy-one fish' system appears to have potential in limited areas for enhancing the farm income. There are two distinct paddy season in Kuttanad; Puncha (main summer crop – Rabi crop) and virippu (additional crop - Kharif crop). Puncha crop sown in November or December and harvested by the end of March whereas the additional crop grown from May to the end of June and harvested in September or October (http://www.kuttanadpackage.in). The cultivated punja lands of Kuttanad having an area of 40000 ha and that of additional crop cultivated approximately 10000 ha area consists of several padasekharams or polders separated by channels and bunds. The area is important in maintaining the food security of the state contributing 20% of the state rice production. Major part of the Kuttanad paddy lands were the reclaimed portion of the lake, as reclaimed on the basis of state policies for expanding the rice production due to Second World War and Bengal famine. Due to state policies and private involvement, a large portion of the shallow area of lake was converted for rice cultivation and reduces the lake area by 65%. In high ranges large portion of forest area were reclaimed for agriculture crops and cause silting up of the lake. River basins of Vembanad receive 32 million tones of sediments annually, which is mainly contributed by the human interference that occurred in the catchments of Western Ghats. Rice cultivation in Kuttanad faced several problems mainly because of the annual flooding and salt water intrusion during monsoon and summer respectively. Kuttanad experienced annual flooding and during the period majority of the areas get submerged under water. Vast areas of paddy fields get submerged for one or two weeks and the annual monsoon flood caused severe damage into the whole Kuttanad area. During monsoon floods, the bunds getting breached and lead to crop damage or result in complete lose of cultivation costs of farmers. The flood water enter the Vembanad backwater, from where the water is washed out to Arabian sea through the Cochin barmouth. During the flood period, communication and transportation facility of the area become worst and the whole area gets isolated as an island with no connection with the mainland. Flooding is worst where the rivers from the upper catchments enter the area in upper Kuttanad, and the least near the backwater. The flood storage capacity of the wetland in turn is closely related to the land use pattern, mainly the extent of rice cultivation in polders. In order to solve the flooding problems and salinity issues, Thottapally spillway was constructed during 1974, but a long term solution to the issues were not achieved. During 1976, the Thannermukkom barrage was specifically commissioned across the estuary to prevent the salinity intrusion in the dry season and retain a freshwater condition into the

Kuttanad region so as to make possible the punja cultivation. But the operation of barrage transforming the water body into two distinct ecosystems, a fresh water zone on the south and a brackish water zone on the north, resulting in gross changes in the physical, chemical and biological entities of the whole backwater system. Incomplete construction of the barrage, having a earthern zone in middle portion of the estuary intensified the situation of Kuttanad by reducing flood water receding during monsoon and tidal intrusion in all the seasons. Instead of earthen bunds, the reinforced granite bunds used for facilitating the additional crop of rice during monsoon created severe damage to the upper reaches of Kuttanad. Kuttanad area and the community associated with the region are facing extreme agrarian distress over the last five decades from multiple factors. Based on the request of Government of Kerala to address the perennial problems faced in Kuttanad, the Union Government entrusted Dr. M. S. Swaminathan Research Foundation (MSSFR), Chennai to conduct a detailed scientific study of the region and to suggest suitable measures to mitigate agrarian distress in Kutanad. The MSSRF recommended a variety of intervention to be executed as a **Kuttanad Package** with a total outlay of ₹ 1,840 crore which was accepted by Govt. of India. Thus Kuttanad package was implemented by the Kerala Govt. in 2008 on a time bound-manner, but was probably a big failure.

The objective of the package was to propose measures for strengthening ecological security of Kuttanad and for expanding sustainable livelihood opportunities for the local people through work and income security (http://www.kuttanadpackage.in/). The report contains a malady-remedy analysis of the problems and potential solutions and the suggestions include - creation of Special Agricultural Zone (SAZ), restoration of natural drainage systems, minimizing ecological damage caused by TMB, functional restoration of Thottapilly spillway, reducing pollution of Kuttanad waters, improving logistics of padashekarams, revitalization of fisheries, promoting environmentally sustainable water tourism, promoting group farming, enhancing income from coconut farming system, improving productivity and profitability of rice farming. The main tasks recommended in the package are: (1) protection and ecological restoration of the water spread area; (2) measures for salinity and flood management in Kuttanad; (3) measures for pollution control; (4) total elimination of aquatic weeds; (5) measures for augmenting biodiversity in the backwaters; (6) improving health and sanitation; (7) declaring Kuttanad a Special Agricultural Zone; (8) provide infrastructure support to paddy cultivation; (9) enforce crop calendar; (10) strengthening of research and extension; (11) strengthening economic viability of farming; (12) coconut-based enterprises and integrated farming; (13) actions to promote fishery wealth of Vembanad lake; (14) promoting fish infrastructure and (15) infrastructure support to facilitate responsible tourism (under water tourism and local ecology) (MSSRF, 2007).

The package aimed at ‘Mitigating Agrarian Distress in Alappuzha and Kuttanad Wetland Ecosystem,' may destroy the environment and ecology of Kuttanad and inflict severe damage to the midland and highlands of the state from where huge volumes of granite would be quarried for the construction of bunds. Criss-cross roads have come up in the State's granary by reclaiming paddy fields and canals. In the name of development, Kuttanad has been subjected to indiscriminate human interventions and the carrying capacity of Kuttanad has already reached its peak. Expressing his dissatisfaction over the implementation of the Kuttanad package M. S Swaminathan admitted the fault in the implementation of the Kuttanad package in a time bound-manner. He said that lack of time bound implementation and some kind of criteria/standards affect the productive nature of the project accomplishment. He added that the failure to set a time schedule and the lack of coordination among various government departments were the major factors that adversely affected the implementation of the package (Anon, 2013b). Poor research and development support for package lead to its failure of implementation and the lack of right resources with the right skills also increases the impact. Lack of establishment of professional implication trigger the complete failure of the package.



a) b) c)

d) e) f)

Plate 2.1 Agricultural and farming activities during the early years (*source*: KDP, 1972)

a) A navigational canal adjacent to the field in Kuttanad with canoes plying
b) Sowing in Kuttanad rice fields; c) Rice field in Kuttanad
d) Carrying harvested sheaths from the field to the threshing ground
e) Threshing of harvested rice sheaths in the threshing ground;
f) Winnowing


a) b)


c) d)


e) f)

Plate 2.2 Agrarian system of Kuttanad wetland system
a) A view of Kuttanad paddy fields before sowing; b) Growing stage of paddy cultivation in Kuttanad; c) A view of Kuttanad paddy fields showing irrigation channel; d) Paddy ready to harvest in the fields of Kuttanad; e) Duck rearing in paddy fields of Kuttanad; f) Flocks of birds in the paddy fields of Kuttanad

2.1.4 Vembanad estuarine system

The Vembanad estuarine system with its complex system of natural and artificial channels extends about 90km in North – South direction from Munambam in North to Alppuzha in South. The estuary has two permanent opening into the Arabian sea – one at Cochin and the other at Azhikode (Fig. 2.5). It is a brackish water system keep the unique biodiversity and resources and maintains the livelihood activities of a variety of communities who depends upon the resources. Diverse aquatic habits, endemic species and other variety of fauna and flora make Vembanad a major biodiversity hotspot. Many rare migratory birds including Siberian stork, egret, Ibis, darter, heron and teals etc. are a regular visitor. Geologically, the Vembanad estuary, a barrier-beach lagoon system, was primarily a marine environment (Rao and Balasubramanian, 1996). After the creation of Cochin port in 1936, the natural bar has been dredged out and maintain a constant depth of 14m for navigation (Strikwerda, 2004). It is believed to have had its origin during the postglacial sea level rise, 5000-6000 years ago. The Vembanad estuary has taken its name from the ancient kingdom of Vembolinad sometime before 1200 A.D. According to Narayan et al. (2002) the Vembanad estuary, the largest tropical estuarine systems in India, acts as a major burial ground for quaternary deposits. Although interrupted by the Arabian Sea at intervals, geomorphologically all the sectors of the existing backwater may not be of common origin. The present configuration of the Vembanad estuary has been attained in the 4th centuary A.D (Anon, 1973), bounded by an alluvial bar parallel to the coastline. Some parts of Alappuzha and Ernakulam districts and a number of islands arose due to the catastrophic deluge (a catastrophic cyclone occurred that made the river Periyar change its course; this resulted in the formation of a number of islands that took place in 1341 A.D separating the water body from the sea with interconnecting channels at Thottapally, Andhakaranazhi and Cochin (Menon, 1913).

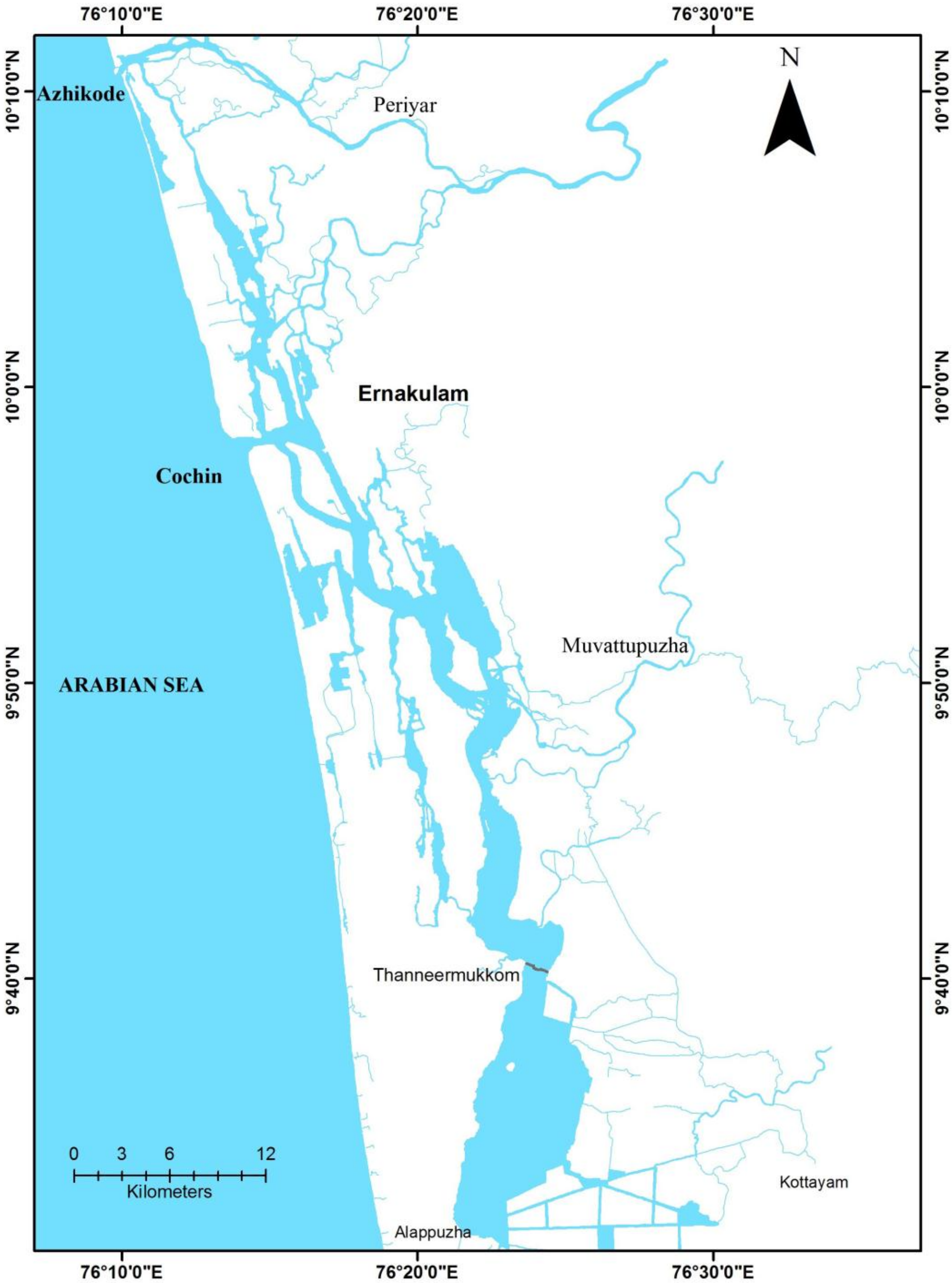


Fig. 2.5 Map of Vembanad estuarine system

The transformation of an originally marine environment into an estuarine system can be traced from the deposition of large quantities of typical marine shells in the Vembanad region (Preston, 1916; Gopalan et al., 1983). The emergence of several islands in the estuary from the time of its formation is a clear indication of siltation of fluvial supply from rivers. The rate and mode of accretion depend on the transportation, quality, and deposition of sediments, tidal influence, and prevailing hydrodynamic conditions. Some part it is known by different names (Velupillai, 1996). During southwest monsoon (June–September), the estuary is gets enhanced supply of alluvium, which is accumulated as bed sediment. The well-known clam fishery of this watershed, both live and dead, evidences the transformation of an originally marine environment into an estuary (Preston, 1916).

2.1.4.1 Physiography

The width of the estuary varies from 500 m to 4 km and the depth from <1m to 12m (Anon, 2008). The lowland regions of Vembanad estuary experience high rain fall and humidity. The mean annual rainfall varies from 250 to 400 cm (Asha et al., 2016). Similar to the other areas of Kerala, two rainy seasons are prevalent in Vembanad estuary. South west monsoon extends from June to September and provide > 60% of annual rainfall. Northeast monsoon extends from November to January, which contributes > 30% annual rainfall. The remaining rainfall occurs during February to May as occasional dry spell. Southwesterly from June to September and northeasterly from October to January are the major winds in Vembanad estuary and the mean wind velocity was about 10 to 12 km. (Balachandran 2001). The tides in the estuary are of a mixed, semidiurnal type with a maximum range of about 1 m (Qasim and Gopinathan, 1969). Vembanad estuary obtains freshwater discharge from six rivers most of which are short, steep, fast flowing and monsoon fed (Anon, 2008; www.nio.org). These rivers which originate from Western Ghats - the Biodiversity Hotspot drain to the estuary and eventually join the Arabian Sea. Pamba, Achankovil, Manimala and Meenachil are the major rivers that drain into the southern zone of Vembanad estuary (south of Thaneermukkom Barrage). Rivers such as Periyar and Muvattupuzha join with northern zone of Vembanad estuary (north of Thaneermukkom Barrage). Discharge of these rivers varies seasonally and maximum input was observed during monsoon season.

Pamba river is the third largest river in Kerala and is popularly called as Dakshina Ganga. It has a length of about 176 km and a catchment area of about 2235 km^2. The average annual stream flow was 3423.7 Mm^3 (http://www.kerenvis.nic.in). The river originates from Pulachimala in the Western Ghats at an altitude of about 1650m above msl and flows through highly varied geologic and geomorphic provinces of the state like Ranni, Ayroor, Pathanamthitta, Kuttanad etc. and then finally ends up in the Vembanad lake. The river drains through Pathanamthitta (Major portion) and Alappuzha (minor portion) district and enters into the Vembanad lake near Kainakkari. Pamba basin is surrounded by Western Ghats on the east, Manimala river basin in the north and Achankoil river in the south. The water quality of Pamba is influenced by the waste water from the pilgrim centre Sabarimala.

River **Achenkovil** originates from the hills of Achankovil in Patanamthitta district and flows through Mavelikkara, Thiruvalla and Karthikapally taluk and it join with river Pamba at Veeyapuram, in Alappuzha district, near the Vembanad lake (http://kerala-rivers.blogspot.in/). The total length of the river is 128km and its average annual stream flow was 2600 Mm^3. Achankovil river shares its northern boundary with Manimala river basin. The average rainfall in the river area was 2600mm.

Manimala river is one of the pernnial rivers of Kerala with a length of about 90 km and a catchment area of about 847 km^2. The river originates from the Thattamalai hills at an elevation of 1156m above msl and drains through the highland, midland and the lowland physiographic provinces of Kerala. The river spreads over Idukki, Kottayam and Pathanamthitta districts and empties into the Vembanad lake, after merging with the Pamba at Valanjavattom near Thiruvalla. The extensively developed sandy plains and point bars, used for holding annual religious congregations are vanishing at rapid rate due to indiscriminate sand mining. The average annual stream flow was 1560.74 Mm^3 (http://www.kerenvis.nic.in).

Meenachil river flows through the heart of Kottayam district of Kerala and have a length of 78km and water spread area of 1208.11km^2 and flows through Poonjar, Teekoy, Erattupetta, Palai, Ettumanoor and Kottayam before emptying itself into the Vembanad lake at Kumarakom, the famous tourist place of Kerala. The Meenachil River is formed by several streams originating from the Western Ghats. The river has 38 tributaries including major and minor ones. The river has 47 sub watersheds and 114 micro watersheds. The average annual stream flow was 1560.74 Mm^3.

Muvattupuzha river having a length of 121km and having average annual stream flow of 3560 Mm^3. The word “Muvattupuzha” is composed of three words 'Moonu' meaning three, 'Aaru' meaning small river and 'Puzha' which also means river. As the word indicates, Muvattupuzha River is also composed of three rivers namely Kothamangalam River, Kaliyar River and the Thodupuzha River, all of which together forms the Muvattupuzha river draining into Vembanad estuary near Vaikkom region. The industrial activities mainly by the Hindusthan Newsprint Limited situated near the banks of Muvattupuzha river and the effluent from the industry affect the water quality of the river. The average annual stream flow was 3560 Mm^3.

Periyar river is the longest river in Kerala state having a length of about 244km. It originates from Sivagiri hills of Western Ghats. The average annual stream flow of Periyar was 4867 Mm^3. The river flows along almost virgin forests in places such as Kokaripara, Neriamangalam, Edamalayar and Malayattoor. At Alwaye the river bifurcates into two, Marthandavarma and Mangalapuzha branches. The Mangalapuzha branch joins Chalakudy river and empties into the Arabian Sea at Munambam while the Marthandavarma branch flows southwards, through the Udyogmandal area and joins the Cochin backwater system at Varapuzha (Periyar Valley Irrigation Project (PVIP), 1972). Kochi city depend up on Periyar river for the drinking water source. Twenty five percent of Kerala's industries are situated along the banks of Periyar river. The major types of industries include fertilizers, pesticides, chemicals and allied industries, petroleum refining and heavy metal processing, radioactive mineral processing, rubber processing units, animal bone processing units, battery manufacturers, mercury products, acid manufacturers, pigment and latex producers etc. Through the river discharges large amount of industrial pollutants from these neighboring industries reached into the estuary. The average annual stream flow was 4867.9 Mm^3.

Chitrapuzha river, one of the tidal tributaries of Periyar river which drains to Vembanad estuarine system. The river receives effluents from chemical factories such as Fertilizers and Chemicals Travancore Limited (FACT) Cochin division, Hindustan Organic Chemicals Limited (HOCL), Cochin Refineries Ltd. (CR) and Carbon and Chemicals Ltd. (CC) and make southern part of Cochin estuary heavily polluted.

2.1.4.2 Socio-economic aspects of Vembanad estuary

The inland sector including the Vembanad estuary and other backwater systems of Kerala has a great potential for fisheries and ecotourism development. The ***backwater tourism***, tourist resorts and houseboat cruise is an economic evolution marketing the scenic and natural charm of Vembanad estuary. The Vembanad estuary with its vast expense of water, luxuriant greenery and very vibrant nature around the estuary attract tourist from India and abroad (Plate 2.3). There is a drastic growth in tourism sector in and around Vembanad estuary for the past two decades. According to Rajan et al. (2008), the tourism industry illegally encroached on many regions of Vembanad estuary. Several hundred house boats are operating in the estuary. The backwater tourism considered as backbone of the state major revenue. Rise of tourism sector in Alappuzha district resulted in employment of one or two members from each family of Kuttanad area to tourism industry as houseboat operators or other employment related to house boat and backwater tourism. Nearly 2% of the local adult population of Kumarakom village on the eastern banks of Vembanad backwater is employed in the tourism sector. The snake boat races in estuary have attained a wide popularity all over the world; especially the Nehru Trophy Boat race conducted annually in August. Even though the tourism sector in the region has greater economic benefits which are a source of employment and income foreign generation, there are also negative socio-environmental impacts (Vijayakumar, 2009). Thousands of houseboats, mainly concentrated on the Vembanad lake region (over 930 officially recorded in Alappuzha district), reject dumping wastes into the lake and accused the resorts on the banks of estuary. The high standard resort groups complained out the small ones for not having the proper waste treatment system (Anon, 2014a). Majority of the houseboats not utilizing the facility for the waste treatment plant operated by the District Tourism Promotion Council (DTPC) Govt. of Kerala in order to process and treat the septage from houseboats (Anon, 2016). In addition unsuccessful operation of sewage treatment plants has consistently impacted the water quality of the estuary (Asha et al., 2016). Municipal wastewater and septic tank runoff into estuaries and modification of natural water flow can also result in increased eutrophication. Similarly fuel leakage, emissions, plastic wastes and food wastes also affect water quality by adding metals, hydrocarbons and other pollutants to the water and enhances the degradation of the estuarine ecosystem by converting to cesspools of contamination in several zones.

a)

b)

c)

d)

Plate 2.3 Scenic beauty of the Vembanad estuarine system
a) House boat plying in the estuary; b) Men in small country boats going for collecting clams c) Flocks of Cormorants in the estuarine water; d) A distant view of uninhabited island in the estuary

Kumarakom bird sanctuary also known as Vembanad bird sanctuary is situated on the banks of Vembanad estuary, renowned for its avian population. Kumarakom, one of the famous bird sanctuaries in Kerala, that is unique in rare varieties of migratory birds like siberian stork, egret, darter, heron and teal. Sprawling over a wide expanse of 14 acres, the enchanting wooded beauty of the sanctuary lures a large number of foreign visitors (Plate 2.4). It is referred to as an ornithologist's paradise because of the presence of a wide variety of resident and migratory birds like egrets, darters, herons, teals, ducks including the Siberian storks. Kumarakom - a reclaimed portion of Vembanad lake is fertile and most suitable for cultivation of rice and coconut trees. The modern history of this area was started with the arrival of the foreigner Alfred George Baker, in 1847. He reclaimed 500 acres of Vembanad lake and garden land and paddy fields and built the Baker House. Besides his interest in agriculture, Baker also took care of the environment.

a)

b)

c)

d)

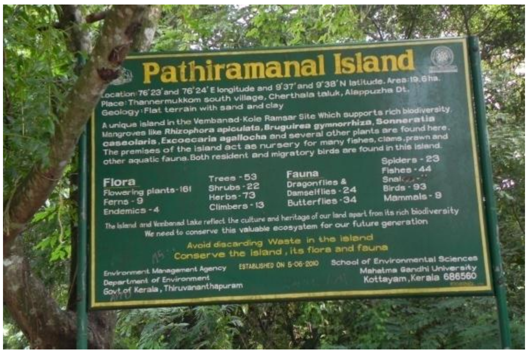
Plate 2.4 Various views of Kumarakom bird sanctuary
a) Inside view of walking trails in Kumarakom bird sanctuary
b) View from tower at Kumarakom bird sanctuary during off season
c) and d) Common species of mangrove - c) *Excoecaria indica* and d) *Bruguiera sexangula*

At Kumarakom, he planted mangroves for protecting the embankments. He also left a sizeable portion of the land undisturbed for the birds, which later developed into a bird sanctuary. This bird sanctuary is visited by travelers in large numbers these days. Baker's home at Kumarakom, locally known as Baker's Bungalow is now a five-star hotel - the Taj Garden Retreat, run by the Taj Group of hotels (www.keralatourism.org). Now, Kumarakom, a backwater tourism hub is gaining strategic place in on the tourist map due to its natural charm and aesthetic beauty. Tourism development in Kumarakom started with the lease of Kerala Tourism Development Corporation (KTDC) land in the bird sanctuary area to Taj Kerala Resorts Limited in the 1989. Indiscriminate land filling is taking place in Kumarakom for resort construction and other tourism amenities. The primary mangroves species in Kumarakom were *Acanthus ilicifolius, Acrostichum aureum, Bruguiera sexangula* and *Excoecaria indica* (Bijoy Nandan et al., 2015). The Kumarakom mangrove forests which was the biggest heronry in the State till early 1980s, is being systematically wiped out due to the growth of tourism. The population of Night Herons, Cormorants and Darters, which could be counted in thousands during the seventies and early eighties, had come down to a mere 258 in 1993. The 2001 survey spotted 198 birds in the tourist complex area. The number of Little Cormorants had declined to just six (http://www.birdskerala.com/). Ali (1984) reported that large flocks of wintering ducks roost in the calm waters of Vembanad lake. Narayanan and Vijayan (2007), observed ten species of waterbirds belonging to the families Anhingidae, Phalacrocoracidae, Ardeidae and Threskiornithidae were breeding in the Kumarakom Heronry. In their observation the Little Cormorant *Phalacrocorax niger* was the most numerous species, Little egret *Egretta garzetta* the least numerous and the Black-crowned Night Heron *Nycticorax nycticorax* was present in reduced numbers compared with earlier reports. According to Zoological survey of India (Editor-Director, 2009), 198 species of birds belonging to 50 families have been reported from Vembanad lake region. Of these, 54 species are winter migrants. Four species of birds, Spot-billed Pelican (*Pellecanus philippensis*), Oriental Darter (*Anhinga melanogaster*), Black-headed Ibis globally threatened category as per the Red Data Book (Birdlife International, 2001) are found here. The avifauna of Kuttanad was studied from January 1995 to June 2007 by Prasanth Narayanan et al. (2011). Two-hundred-and-twenty-five taxa of birds belonging to 15 orders and 59 families were recorded. Among the birds recorded, 38% were migrants. Fifty-five species were found to breed in the area. The bird Family Scolopaceidae showed maximum species diversity in the area. European Roller *Coracias garrulus* were recorded during the study and it was the first report of this species from Kerala. They noted that Kuttanad wetland shows greater species diversity, especially in the wetland birds, than the Kol wetlands of Kerala.

Pathiramanal island, a beautiful and tiny island with an area of 24h, located in the middle of Vembanad estuary, accessible only by boat from Kumarakom and Muhamma region (Plate 2.5). The island is another tourist attraction having considerable varieties of migratory birds from various parts of the world. The island is also an important breeding ground of Indian Otter (Protected Area Update, 2003). It provides habitat for resident and seasonal migratory water fowl, otters, fishes, clams, shrimps, crabs, aquatic insects and other aquatic organisms. True mangrove species in Pathiramanal island are *Rhizophora apiculata, R. mucronata, Ceriops tagal, Acanthus ilicifolius, Excoecaria agallocha. Dolichandrone spathacea, Cerbera odollam,* whereas *Derris scandens, D. trifoliate, Brringtonia racemosa, Hibiscus tiliaceus, Thespesis populnea, Pandanus odoratissimus, Canavalia obtusifolia* and *Clerodendrum inerme* were the mangrove associates (Balasubramanian and Azeez, 2012). The island is also renowned as a birdwatcher's paradise. It is home to around 91 local species of birds and 50 migratory birds. One can see pintail ducks, common teal, night heron, cormorant, darter, Indian shag, purple heron, gulls, terns, large egrets, intermediate egret, cattle egret, Indian pond heron, little egret, pheasant-tailed and bronze-winged jacanas, stork-billed kingfisher, watercock, whistling duck, cotton pygmy-goose, little cormorant and whiskered tern. Monarch flycatcher has been reported from this island. It provides a habitat for the vulnerable spot billed pelican *Pelicanus philippensis*, large populations of water fowls, besides a high species diversity of finfish and shellfish (WISA, 2014).

The island was in the possession of a private person and was taken over by the government in 1979, following the Land Reforms Acts. Later families residing on the island were rehabilitated and was handed over to the Kerala Tourism Department (http://wiienvis.nic.in/). When tourism gaining momentum in Kumarakom, the tourism department considered developing the adjacent Pathiramanal island for tourism activities. The former President Sankar Dayal Sharma laid the foundation stone for tourism development activities in the late nineties (https://archive.org). An entrance gate and walkway were created as part of the project. In 2000, there was a move by the government to long-term leasing of the Pathiramanal island to a private resort group. This was opposed by the Muhamma Panchayat and Kottayam Nature Society and challenged this move in the court by raising concern over the possible destruction of the ecologically fragile island. The Supreme Court in January, 2006 ordered that the status quo of the island would be maintained until a decision was taken by the government on the utilization of its land based on the Environment Impact Assessment (EIA) studies. Thus in 2007, Kerala State Biodiversity Board (KSBB) was entrusted by the Government of Kerala for necessary EIA studies. However, before completing the EIA study of KSBB, the Kerala government started to set up a bio-park in the island by violating the order of government and Supreme Court. In 2010, the KSBB submitted the EIA report to the state government where it was mentioned that no construction work should be allowed in the island. Tourism department tried to execute the bio-park project without doing any construction work. The plan was to set up a butterfly garden, Fish Park, floating restaurants and bird watching facility at Pathiramanal for attracting tourists. Development of freshwater ponds, major canals as well as rejuvenation of minor canals and small ponds were also part of the development plan. Kerala Industrial and Technical Consultancy Organisation Ltd. (KITCO) was appointed the executing agency of the project. But forest department and Kottayam Nature Society came up with objections that KITCO for having cut down trees in the island as part of the project. Following this KITCO quit the project (Anon, 2012b), but now the Pathiramanal island lost its naturality. Thus, the natural beauty of the island has been lost with lots of garbage and plastic waste that has accumulated in the island and the water along the island was with rotten smell and large scale encroachments.

a)



b)

c)

d)

Plate 2.5 Pathiramanal island
a) Distant view of Pathiramanal island; b) A board displaying the details of Pathiramanal island c) Inside view of the island; d) Rubbish things dumped inside the island

The ***fish*** and ***fisheries*** play a crucial role in the Kerala's economy, particularly among the communities in and around Vembanad estuary (Plate 2.6). It harbours a variety of marine and estuarine fishes, prawns, crabs and clams (Plate 2.7). Fishes like pearl spot, mullets, prawns and crabs are important priced fishes. One hundred and fifty species of fishes belonging to 100 genera under 56 families were identified from this estuary (Kurup and Samuel 1987). The edible clam, *Villorita cyprinoides* var. *cochinensis* from Vembanad sustains a regular fishery, providing a cheap source of protein food for a large section of people and a raw material for the manufacture of cement and lime. The major issue affecting clam fishery in the Vembanad estuary is the indiscriminate exploitation of undersized clams for industrial purpose and as an animal feed. The closure of TMB leads to changes in the ecology of Vembanad estuary affecting the distribution and survival of clams. In addition, pollution of the estuary through industrial sources is a serious concern for the sustainability of clam resources.

There are signs of decline of the Vembanad fishery resources, evident in the lesser number of species and decline of fishery production (Asha et al. 2014). The study observed that fishery resources comprised of eighty species of finfishes, five species of penaeid shrimps, three species of palaemonid prawns, two species of crabs. The estimates of annual fishery production indicated an annual landing of 4387.31 t, in which 480.98 t and 3906.33 t contributed by southern and northern zones respectively. The commercially important species such as *Mugil* sp., *Chanos chanos*, *Tachysurus* sp., *Etroplus suratensis*, *Macrobachium rosenbergii*, *Scylla serrata*, in which some of these species have now vanished from the area, while others have become a rarity. The native fish species such as *Horabagrus brachysoma* and *Hyporhamphus xanthopterus* of Vembanad estuary were vulnerable and other species suffered a decline in both number and distribution. The commissioning of the TMB and its prolonged closure has seriously affected the fish species diversity and population. The ecological changes in Vembanad estuary, particularly to the backwaters, has significantly affected the fisheries based livelihoods also.

Plate 2.6 Major fishing activities in Vembanad estuary
a) Gill net fishing; b) Stake net fishing; c) Clam fishing; d) Iron rake used for clam collection; e) Chinese dip net fishing; f) Fishing in coracle

The edible clam, *Villorita cyprinoides* var. *cochinensis* from Vembanad sustains a regular fishery, providing a cheap source of protein food for a large section of people and a raw material for the manufacture of cement and lime. The major issue affecting clam fishery in the Vembanad estuary is the indiscriminate exploitation of undersized clams for industrial purpose and as an animal feed. The closure of TMB leads to changes in the ecology of Vembanad estuary and affect the distribution and survival of clams. In addition, pollution of the estuary through industrial sources is a serious concern for the sustainability of clam resources.

a)*Macrobrachium rosenbergii*

b) *Villorita cyprinoides*

c) *Puntius spp.*

d) *Etroplus suratensis*

e) *Horabagrus brachysoma*

f) *Hyporhamphus xanthopterus*

Plate 2.7 Major fishery resources of Vembanad estuary from various landing centers

2.1.4.3 Thottappally Spillway

Due to the accumulation of floodwaters from the river systems in Kuttanad, water level used to rise beyond manageable limits soon after the onset of the southwest monsoon. The entire low lying areas of the region used to remain flooded till the end of north east monsoon making it impossible to raise a second crop during the autumn season. Detailed hydraulic surveys conducted from the early 1930's had shown that this problem could be mitigated by diverting the floodwaters directly to the Arabian sea at the extreme south of the flood limit itself. Accordingly, the construction of a spillway was started in 1951 at Thottappally located 20 kilometers south of Alappuzha town. A 368 metre long outlet to the sea was constructed. The regulator-cum-bridge has 40 spans of 25 ft. (KDP, 1972). It was designed to discharge 64,000 cusecs of water to the sea and to control floods in Kuttanad and the spillway was designed to discharge more than 90 percent of it directly to the sea (George, 1984). However, while designing the spillway the problem of piling up of water due to the raising sea level during the monsoon months and the consequent formation of sand bar on the seaward side of the spillway were not taken into account. Therefore after the spillway was completed in 1955, the realized capacity of it is found to be less than one - thirds of the estimated capacity and hence fails to serve its purpose to some extent (Thomas, 2002). So the remaining flood water in the Kuttanad area make least salinity in the upper part of the wetland system (Nayar, 1998).

2.1.4.4 Alappuzha-Changanassery (AC) Canal

Alappuzha-Changanassery (AC) Canal was mainly constructed for controlling the flood water and also envisaged to remove the block caused to the water flow by the Alappuzha Changanassery road (Alappuzha town is connected with Changanassery by AC road). The AC road was created in an east-west direction right across the direction of floodwater movement. The main functions of AC Canal include; it act as a shunt inter-connecting rivers being crossed by the road for re-distribution of flood water; give a water transport facility between Alappuzha and Changanachery and third, it provide the necessary mud for raising the AC road. Canal was not serving its three main functions because the construction work of canal was not complete in many places and also the construction of AC road dangerously took the natural width of rivers and rivulets. Now the paddy fields get water-logged soon after a heavy downpour in the upper Kuttanad area (MSSRF, 2007). The widening of the Alappuzha-Changanassery (AC) Canal, a project under the, Kuttanad Package, has not progressed due to the delay in the eviction of traders and families living on either side of the canal. The construction of AC canal from Manackachira to Onnamkara (Phase I - 11.72km) has been tendered and the construction work is to start soon.

2.1.4.5 Thanneermukkom Barrage (TMB)

The Thaneermukkom barrage was built (1975) as a part of Kuttanad Development Scheme to prevent salinity intrusion in the dry season and retain a freshwater condition into the Kuttanad region so as to make possible the punja cultivation (KWBS, 1989) (Plate 2.8). The barrage is located at Thannermukkom, 25 km north of Alappuzha considered the largest mud regulator in the country, which was built across narrowest region of backwater between Vechoor in east and Muhamma in west. It has a length of 1400 meters and includes a 470 m long reclaimed portion in the middle of the estuary. The plan was to build the TMB in three phases. The first phase at Muhamma end comprising 31 shutters and two locks for navigation was completed in 1968. The second phase at Vechoor-end with 31 shutters and one lock was completed in 1974. When the work on the third phase with 31 shutters was delayed, a cofferdam was erected in 1975 to stop the saltwater flow. However some of the reports states that, the barrage was then commissioned in 1976 (Anon, 2001; http://www.kuttanadpackage.in/). Another important issue associated with TMB is the deviation in the construction from its original design with the erection of the cofferdam at central section. This has changed the cross section of the Vembanad lake at its deepest central part. The anticipated benefit from TMB was safe *punja* paddy and intensification to a second crop (*virippu* crop) in about 18,500 ha of Kayal land, Lower Kuttanad**.** The barrage closure extended from three month period of December – March to even May. The barrage was operational and retaining a fresh water condition in the south of Thaneermukkom barrage and additional cropping was taking place during dry seasons in the Kuttanad area. The barrage has been relatively successful in ensuring freshwater conditions in Kuttanad and enabling cropping additional areas during dry seasons. However, there have been several ecological consequences triggered by changes in salinity regimes and impeded circulation and mixing patterns. There has been a decline in brackish water fisheries. Elimination of tidal flushing has impacted pollution levels in Kuttanad, further aggravated by increasing use of fertilizers and pesticides which also contributed to proliferation of invasive, water hyacinth affecting the light penetration in the southern zone of TMB. The fishery resources of the southern part of Vembanad estuary suffering a decline due to the construction of TMB, mainly its haphazard opening and closing and resultant ecological changes (MSSRF, 2007).The shutters have remained closed for a period upto six months (Fig. 2.6) invariably creating conflicts between farmers and fishers. Based on the expert committee constituted by Govt. of Kerala to settle the conflicts regarding the opening and closing of the barrage, to assess the environmental, ecological and socio-economic imbalances arising out of barrage operations, proposed to reduce the annual period of closure of barrage to mid- December to mid- March and also suggested for implementing entire environmental monitoring programme and a participatory structure for barrage operations.

Measures recommended include modernization of TMB with efficiently operable shutter, replacement of middle cofferdam with barrage, operation of TMB at scheduled time and appropriate enforcement new crop (*punja*) calendar and exploration of opportunity for reducing the scheduled time. It is also important to note that, based on the recommendations of M. S Swaminathan Commission's Kuttanad package, as a recent development in 2014, Govt. of Kerala initiative the coffer dams and other obstructions along the TMB are being suitably replaced by sluice gates (barrages) for overall rejuvenation of the ecology of the estuary, for the continuity of the water body on the south and northern sides and also the effective management for salinity regulation in the low lying paddy fields of the wetland. But such modifications could again lead to other ecological consequences on the ecosystem that is to evaluated possibly in a different perspective.

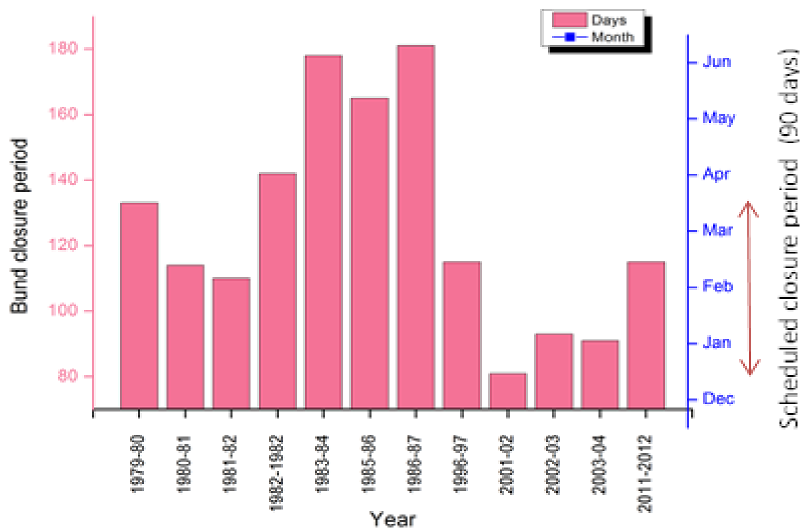


Fig. 2.6 The closure period of TMB from 1979 -2012 period

a)

b)

c)

d)

e)

f)

Plate 2.8 Different views of Thanneermukkom barrage

a) Thanneermukkom Barrage; b)National Water Highway crossing the barrage

c) TMB: open period; d) TMB: closed period

e) and f) Recent reclamation activities in the middle cofferdam

2.1.4.6 Alterations of Vembanad estuarine area

Major alterations that have taken place in the morphology and ecology of in the Vembanad estuary in the sequence of its history can be congregated as a combination of natural processes and human interferences of which latter has contributed much serious alterations in the system. Reclamations in the name of agriculture and aquaculture have been in its peak in the Kuttanad resulting in its shrinkage both horizontally and vertically (Gopalan et al., 1983). According to MSSRF (2007), the water spread area of Vembanad has decreased over the years both in square area and cubic area. In 1834, the Kayal having 36,329 ha area and this was reduced to about 23,750 ha in 1983. It further declined to 13,224 ha in 1992 and 12,504 ha in 2000 and its rate of decline has increased over the years (Fig. 2.7). The reclamations and economic activities in the Cochin estuary catalyzed with the development of the Cochin Port Trust in 1938 and the major reasons are given in the Table 2.2, presently ending with Vallarpadam International Container Transshipment Terminal, Kochi (2011) and Liquefied Natural Gas Terminal (LNG) (August, 2013). The annual rate of decline during the first phase (1834-1983) was 0.23 %, which during the second phase (1983-1992) increased to 4.93 % and to 0.68 % during the third phase (1992-2000). Along with the depth, the water carrying capacity of the lake system has reduced to 78%, ie from 2.4 km^3 to a mere 0.6 km^3 (MSSRF, 2007). The depth of the Kayal is decreasing and its rate increased after the construction of Thanneermukkom barrage particularly its cofferdam situated in the middle of the estuary. Reduced flow in the southern part cause the deposition of silts and ultimately reduces the depth of the system and it leads to decline in the carrying capacity of the water body (Rajan et al., 2008). Area wise shrinkage of Vembanad estuarine system from 1687 to 2005 is given in figure 2.8. According to Nair and Suresh Babu (2016) the total area wise loss of the Vembanad estuary was 12.28 sq km between 1972 and 2015 (Fig. 2.9).

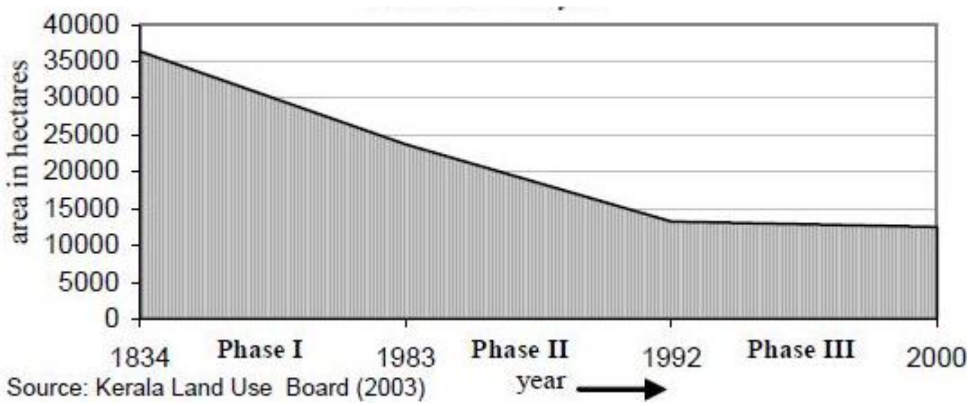


Fig. 2.7 Decline in area of Vembanad estuary over the years (*source*: MSSRF, 2007)

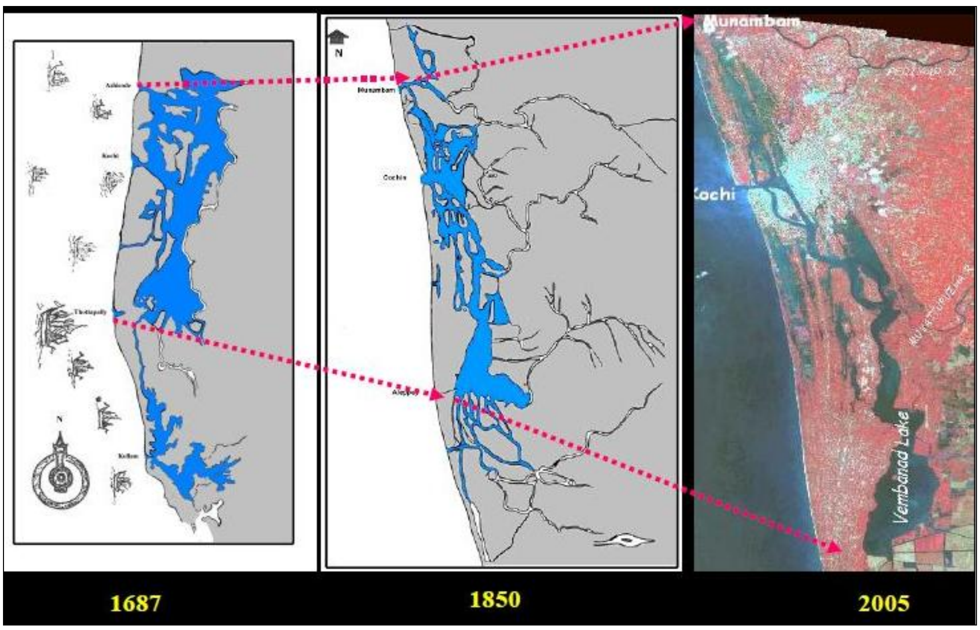


Fig. 2.8 Area wise shrinkage of Vembanad estuarine system from 1687 to 2005 (*source*: www.nio.org)

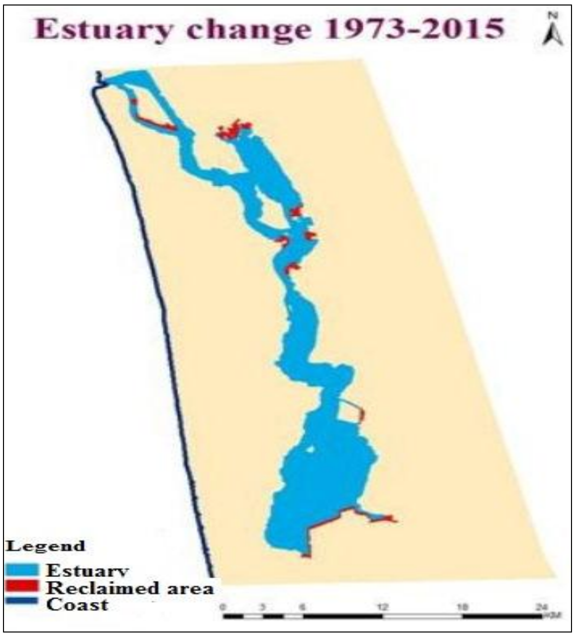


Fig. 2.9 Spatial changes of Vembanad estuary during 1973-2015 (*source*: Nair and Suresh Babu, 2016)

Table 2.2 Reclamation of Cochin estuary over the years (Menon et al., 2000)

Period	Area reclaimed (ha)	Purpose
1834-1903	2227	Agriculture
1912-1931	5253	Agriculture
1920-1936	364	Willington Island
1941-1950	1325	Agriculture
Till 1970	5100	Paddy-cum-shrimp culture
1970-1984	800	Paddy-cum-shrimp culture
1975	6900	Bunding at Thanneermukkom
1978	11	Fishing Harbour
1981-1985	142	Vallarpadam-Ramanthuruth, Candle island
1981-1985	142	Southern extension to Willington island
1981-1985	24	Urban development
1981-1985	12	Cochin shipyard and tanker berth
2011		Vallarpadam International Container Transshipment Terminal
2013		Liquefied Natural Gas Terminal (LNG)

2.2 Study stations

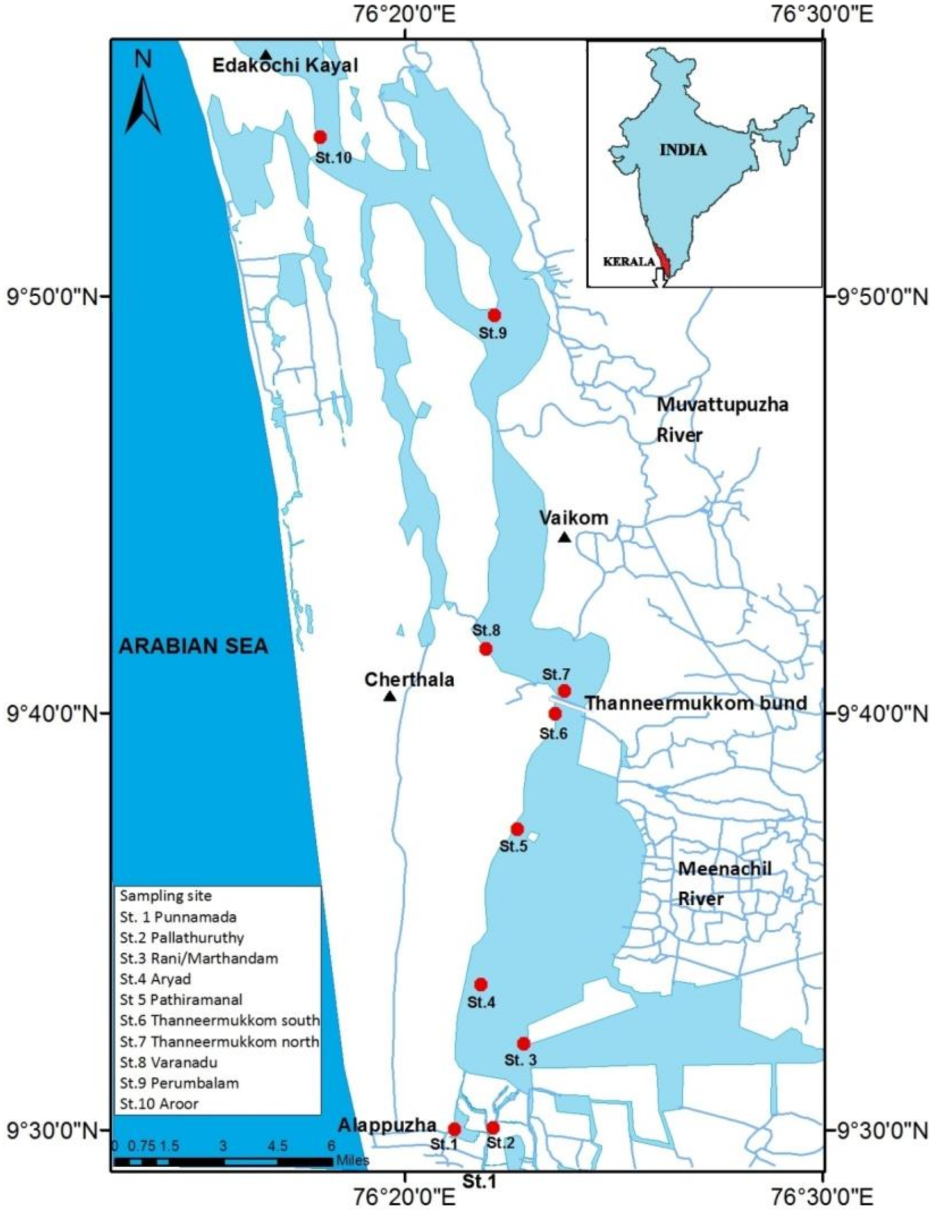


Fig. 2.10 Map of Vembanad estuarine system indicating the study stations (ArcMap prepared using ArcGIS v. 10.2.2 software)

Sampling was conducted in ten different stations of ecological importance, with six stations on southern side of the Thaneermukkom barrage and four on northern side of barrage (9° 30' 069" N - 9° 53' 519"N and 76° 21' 268" E - 76° 18' 139" E) extending from Punnamada in south to Aroor in north. The location of each study sites was statistically tested and the geographic positions were fixed using GPS (global positioning system). The study stations, station 1- Punnamada; station 2- Pallathuruthy; station 3- Marthandam; station 4- Aryad; station 5- Pathiramanal; station 6-Thaneermukkom South; station 7- Thaneermukkom North; station 8- Varanadu; station 9- Perumbalam and station 10- Aroor. The Vembanad estuary is enriched with the rivers Manimala, Meenachil, Pamba, Achenkovil on the south of Thanneermukkom barrage and Muvattupuzha river and Periyar river on the north. The map showing the study stations are given in figure 2.10

Station 1 – Punnamada (9° 30' 069" N and 76° 21' 268" E)

Punnamada: It is the portion of the Vembanad lake that is located in Kuttanad region of Alappuzha district. This zone is situated on the eastern side of the backwater and about 23 km south of the Thannermukkom barrage. It is in this portion of the lake where Nehru Trophy Boat race is conducted and is the finishing point of the national boating competition. This zone is the anchoring zone of the house boats and several of them along with inland transport boats ply across this part of the backwater connecting to the nearby towns/villages, which have lasting impact on the lake ecosystem. Oil spillage, solid waste disposal were observed in this study area. The floating weeds, *Salvinia* sp. and *Eichornia* sp. could be seen in the station probably obstructing dissolved oxygen diffusion into the water column. Construction of polders for agricultural purpose has adversely affected the free flow of water into the estuarine system. Severe encroachments and land filling are undergoing tremendously in and around the study station related to tourism development. The canals in and around Alappuzha town are choked by invasive water hyacinth and other aquatic weeds which further aggravates water logging which in turn prevent the proper washing out of municipal wastes leading to hyper-eutrophication. The colour of water is dark and turbid and several parts of this station. The average depth of the station was 2.5m and is highly influenced by the riverine inflow from Pamba, Achencovil and Manimala. The station is under severe influence of sewage containing drainages of Alappuzha town leading to drastic changes in the water quality of the area. The unregulated sewage discharges from resorts, houseboats and exhaust of motor boats forms a major source of aquatic pollution in Punnmada region. Unutilized organic and inorganic fertilizers from the Kuttanad paddy fields directly gets drained into this station (Plate 2.9 and Plate 2.10).

Station 2 – Pallathuruthy (9° 30' 145" N and 76° 22' 102" E)

Pallathuruthy: This station is situated on the south eastern side of the estuary with low lying paddy fields bordering it. It's about 26.5 km south of the Thannermukkom barrage. It is a water bound region surrounded by the Vembanad lake and the Pallathuruthy canal passed through the region. Pallathuruthy has become an inevitable tourist spot, vital vein for houseboats as well as artisanal fishing activities. Like the rest of Kuttanad, most of the land in Pallathuruthy is used for paddy cultivation. Riverine inflow from Pamba, Achankovil and Manimala influence the station. Human disturbance mainly by large scale tourism developments in Pallathuruthy is increasing and have a negative impact on its natural beauty. Agricultural runoff from the polders (padashekarams), sewage disposal from house boats, human settlements and floating weeds around seems to be affecting the station. Agrochemicals including fertilizers, insecticides, etc., domestic sewage and urban wastes from Alappuzha town is severely affecting the water quality of Pallathuruthy. Usage of these polluted waters for domestic purposes has led to several waterborne epidemics. The average depth of the station was 5.7m (Plate 2.11 and Plate 2.12).

Plate 2.9 Station 1 – Punnamada
Batteries of houseboats in the Punnamada region

Plate 2.10 Station 1 – Punnamada
Pavilion of famous Nehru trophy boat race located near to the study site.

Plate 2.11 Station 2 – Pallathuruthy
Open area with intense houseboat activities

Plate 2.12 Station 2 – Pallathuruthy
Local women row their small vanji (boat) near the study site

Station 3 – Marthandam (9° 30' 936" N and 76° 22' 665" E)

Marthandam: Situated on the eastern side of the estuary and is influenced by the discharges from Pamba and Manimala river. The station is about 18.5 km south of Thannermukkom barrage. The R Block and C Block are the nearest padasekharams situated on the eastern side of this station. Agricultural runoff including pesticide residues and fertilizers from these padashekarams is a major factor which affects the hydrology of this zone. It is also a major area of clam *(V. cyprinoides)* fishery. The station is a major site for night haulting of majority of the houseboats apart from Punnamada and Kainakari. The station gives way for the inland water way for transport boats. All these boating activities trigger fuel emissions from boat motors, suspension of bottom sediments, decreased water transparency, shoreline erosion, destruction of fish spawning areas, and loss of valuable fish. Uncontrolled houseboat tourism in the peak of tourism season subjects this station to indiscriminate exploitation beyond its carrying capacity. The solid waste from house boats and leachates from agricultural fields probably influences the station. The average depth of the station was 4.7m. Regular flow of flood water during monsoon season from Vatta Kayal, agriculture fields of C, H and R Blocks results in increased silting (Plate 2.13 and Plate 2.14).

Station 4 – Aryad (9° 30' 706" N and 76° 23' 583" E)

Aryad: It is the widest point and the main basin of the Vembanad estuarine system. It's around 16 km south of the barrage. Shorelines of the study station have also been reclaimed for development of infrastructure for tourism. Live clam fishing by traditional fishermen was a prominent activity in the station, providing livelihood and full time employment to over thousands of fishermen. Rich sources of live clam as well as sub fossil deposits of clam are the characteristic feature of this station. *Villorita cyprinoides* were the major clam species of the region. The rate of sand mining is higher than natural replenishment in the station and has resulted in the development of pits of various dimensions in the estuarine beds, affecting the water quality and depth of the station. House boat tourism and related activities were also influencing the station. Thanneermukkom Barrage plays a critical role in influencing salinity pattern of this station. Closure of Thanneermukkom Barrage during the summer months lead to a significant reduction in tidal flushing in this zone which affects the water quality of the region. The average depth of the station was 2.7m (Plate 2.15 and Plate 2.16).

Plate 2.13 Station 3 – Marthandam
Lakes bordered by the vast length of coconut trees

Plate 2.14 Station 3 – Marthandam
Halted houseboats near to the station

Plate 2.15 Station 4 – Aryad
Open area of backwater with clam dredging

Plate 2.16 Station 4 – Aryad
Houseboat navigation in Aryad region

Station 5 – Pathiramanal (9° 39' 708" N and 76° 23' 586" E)

Pathiramanal: The study station is situated near a small island Pathiramanal (28.505 ha.), in Muhamma panchayat of Alappuzha district, harboring significant biological diversity, especially of waterbirds. The island is fringed by mangroves and associates and is around 7.5 km south of the barrage. It emerged as an important roosting site for water birds due to availability of food and negligible human habitation in surrounding areas. Land drainage from the island could be observed particularly during the monsoon period. The island has been beautified by State Tourism Department by laying road ways and planting trees that has probably altered the topography of the area. The scenic beauty on both sides of the lake as well as that of the island is mind blowing. Fishing activity, clam fishery mining and tourism related activities are seen in the area. Operation of Thanneermukkom Barrage, have altered the natural ecological characteristics of the station. It is the fresh water zone of the backwater. The average depth of the station was 4.4m (Plate 2.17 and Plate 2.18).

Station 6: Thanneermukkom South (9° 39' 708" N and 76° 23' 586" E)

Thanneermukkom South: This station is located on the south of Thanneermukkom Barrage, about 400 m south of the barrage and on the narrowest part of Vembanad estuary. This station is influenced by tides and currents arising from the Cochin barmouth on the northern side of the water body. It is also part of the main basin of the estuary and is almost the central part of the water body extending from Alappuzha to Cochin. Intense fishing activity, sand mining and inland water transport activities are influencing the zone. Thanneermukkom fish market is situated near to the station. The unscientific operations of the Thanneermukkom barrage have blocked the connectivity of the water body to the sea for a good part of the year. TMB severely affects the ecological characteristics of this zone and restrained the seasonal intermixing of fresh and saline water and thereby interfering with natural cleansing mechanisms of wetland and threatening accelerated loss of habitats and biodiversity**.** The average depth of the station was 6.1m (Plate 2.19 and Plate 2.20).

Plate 2.17 Station 5 – Pathiramanal
Fishing activities near to the Pathiramanal island

Plate 2.18 Station 5 – Pathiramanal
Halted houseboats in the Pathiramanal island jetty

Plate 2.19 Station 6: Thanneermukkom South
A view of southern side of Thanneermukkom Barrage

Plate 2.20 Station 6: Thanneermukkom South
Thanneermukkom fish market situated close to the station

Station 7 - Thanneermukkom North (9° 40' 798" N and 76° 23' 664" E)

Thanneermukkom North: This zone is located on the northern side of Thanneermukkom Barrage about 150 m north of the barrage and 40 km from the Cochin barmouth. The station acts as a transition zone between north and south of the backwater. This station experiences the action of tides and currents arising from the barmouth at Cochin. Intense fishing activity, sand mining and inland water transport activities were seen actively in this zone. Large number of stake nets is operated in the study station. As the station is a major source clam, *V. cyprinoides* that provides livelihood for thousands of fisherfolk. The average depth of the station was 3.3m (Plate 2.21 and Plate 2.22).

Station 8 – Varanadu (9° 41' 586" N and 76° 22' 185" E)

Varanadu: The station is influenced by probable discharge from Mc Dowell and Company's distillery unit situated on the western side of the water body. Both raw and treated effluents from the distillery are being directly discharged into this station with high organic load causing severe fouling of the atmosphere. It is about 34 km south of Cochin barmouth and 5 km north of the barrage. Fishing activity by local fishermen using cast nets and traps could be seen around. Continuous dredging for clam shell beds for making white cement by the Travancore Cements Ltd. (TCL) was evident in the area, which adversely affected the water quality and benthic biodiversity. Conflict between TCL industries and fisheries is common phenomenon of this zone. The area is also under the influence of inland navigation boats criss-crossing the area. The average depth of the zone was 1.7m (Plate 2.23 and Plate 2.24).

Plate 2.21 Station 7 - Thanneermukkom North
A view of northern side of Thanneermukkom Barrage in the background

Plate 2.22 Station 7 - Thanneermukkom North
Large number of stake nets is operated in the study station

Plate 2.23 Station 8 – Varanadu
A view of distillery unit situated on the western side of the water body

Plate 2.24 Station 8 – Varanadu
Lime shell dredging by Travancore Cements Ltd (TCL) near to the station

Station 9 – Perumbalam (9° 49' 731" N and 76° 22' 189" E)

Perumbalam: Perumbalam is a beautiful island, located within the central part of the Vembanad lake, that is wedge shaped, pointing towards the bar mouth. It is bordered by Ernakulam districts on the north, and Kottayam on the South. Most beautiful attraction of the island is backwaters and traditional fishing. Perumbalam is also famous for its coir products and a land of lush paddy fields. The station is about 14 km south of Cochin barmouth and is highly influenced by anthropogenic interventions like encroachment, land filling and construction activities in the small islands of the water body. Land filling and reclaiming parts of the backwater could be seen here affecting the environment of the study area. Mining by local fishermen for clam fishery is widely witnessed in the area and also fishing by cast nets, stake nets could be seen here. Dredging for clam shell beds for making white cement by the Travancore Cements Ltd was evident in this area. The average depth of the area was 3.1m.There were number of illegal construction activities in the lake especially in small private island Nediyathuruthu situated near to the Perumbalam island. The luxury resort, promoted by MiniMuthoot and Kuwait-based Kapico group reclaimed the surrounding backwaters. Resorts and apartment complexes numbering over 20 and worth over ₹ 1, 500 crore have come up in violation of the Coastal Zone Regulation (CRZ) guidelines on the island, Nediyathuruthu, in the Vembanad lake. The Kerala High Court, July 25, 2013 ordered the removal of illegal construction of resort complex while noticing the violation of the various laws and norms like the Coastal Regulation Zone (CRZ), EIA Notification-2006 etc. (Plate 2.24 and Plate 2.25).

Station 10 – Aroor (9° 53' 519" N and 76° 18' 139" E)

Aroor: The station is about 9 km south of Cochin bar mouth and is highly influenced by the tidal action, currents and saline water intrusion from the nearby Lakshadweep Sea. The station is characterized by the presence of patches of mangrove swamps. Aroor is known as one of the three industrial estates in Alappuzha District. The abundance of marine wealth and logistical advantages has helped the seafood export to grow especially around Aroor. The station is also affected by the discharges from sea food industries situated in the banks of the estuary. Sewage wastes from the Kochi metro city is also a major threat to the station. The Cochin Shipyard and the Cochin Port are releasing sizable quantities of waste oil, paints, metal and paint scrapings into the study area and surrounding locations. Fishing activities with stake nets and cast nets by traditional fishermen could be observed here. The heavy movement of motor boats are seen across the station. The average depth of the station was 7.74m (Plate 2.26 and Plate 2.27).

Plate 2.25 Station 9 – Perumbalam
Illegal construction activities in the private island, Nediyathuruthu situated near to the study site

Plate 2.26 Station 9 – Perumbalam
Local fishermen harvest clam by hand while diving in waters

Plate 2.27 Station 10 – Aroor
Construction of residential apartments near to the study site

Plate 2.28 Station 10 – Aroor
Large number of stake nets operated in the study station

2.3 Field Sampling

Monthly field sampling was carried out from research vessel KINGFISHER CP NO. 62, a 25 feet fibre boat of the School of Industrial Fisheries, Cochin University of Science and Technology (CUSAT), Cochin. Sampling locations were located between Alappuzha and Ernakulam district so that ten stations were covered in two consecutive days. Bottom water samples were collected on monthly basis from ten selected stations, for two years between March 2011 and February 2013, and the field sampling was restricted to early morning hours. During the study, first year is chosen as 2011-2012 and second year 2012-2013 period respectively. Based on the prevailing meteorological conditions, the field sampling was also distinguished seasonally as, the premonsoon (PRM) (February – May), monsoon (MN) (June – September) and post monsoon (PM) (October – January).

2.4 Methodology

2.4.1 Physico-chemical parameters of estuarine water

Rainfall data was obtained from India Meteorological Department (IMD) (www.imd.gov.in). River discharge data were obtained from Central Water Commission (CWC), Govt. of India web site (http://www.india-wris.nrsc.gov.in/). The monthly discharge data of river flow from rivers such as Muvatupuzha, Meenachil, Achenkovil, Manimala and Pampa into the backwater were collected from the gauging stations situated at Ramamangalam, Kidangoor, Thumpamon, Kalloopara and Malakkara respectively. The tidal data during the study period were obtained from the automatic tide gauge retained by the Cochin Port Trust (Govt. of India). Data collections were organized on days of high tide.

Bottom water samples were collected using a standard Niskin sampler (General Oceanics, 5L capacity). The water temperature, water depth, transparency and carbon dioxide were measured at each site. Water transparency was usually measured with the Secchi disc (20 cm in diameter) and expressed in meters (Strickland and Parsons, 1972) and light extinction coefficient (ke, m^{-1}) was calculated following the relationship ke = 1.7/Secchi depth (Qasim, 2004).

Water temperature was measured onboard using a 0-50^0C precision alcohol based thermometer. The pH, Total dissolved solids (TDS) and conductivity was determined using Systronics Water Analyser (Model No. 371; accuracy ± 0.01) calibrated with standard sea water. Turbidity was measured using Nephelo– Turbidity meter – Systronics model no: 132 (APHA, 2005) and it was expressed in NTU (Nephelometric Turbidity Unit). Salinity was estimated by Mohr-Knudsen titrimetric method (Grasshoff et al., 1983) and expressed in parts per thousand (ppt).

The water samples for dissolved oxygen (DO) and biological oxygen demand were collected in DO glass bottles. The samples for dissolved oxygen were fixed on-board and estimated using the modified Winkler's method (APHA, 2005; Strickland and Parsons, 1972). The remaining water samples for the analysis of other parameters were collected in thoroughly washed 1 litre polythene bottles and stored in containers with ice. The preserved samples were brought to the laboratory at the earliest and analysis was completed soon. Free carbon dioxide and total alkalinity was measured by using titrimetric method (APHA, 2005). It was expressed in milligrams per litre (mg L^{-1}). Biological oxygen demand (BOD) was measured by APHA (2005); Kale and Mehotra (2009) and it was expressed in milligrams per litre (mg L^{-1}).

The inorganic nutrients were analysed immediately after filtering through Whatman No:1 filter papers, following standard procedures (Strickland and Parsons, 1972; Grasshoff et al., 1983) and using a spectrophotometer (Systronics UV-VIS spectrophotometer, Model No.117), after proper calibration. The values were expressed in the unit of micromole per litre (µmol L^{-1}). Dissolved inorganic phosphate-phosphorus was measured by ascorbic acid method (Strickland and Parsons, 1972; Grasshoff et al., 1983) and absorbance was measured at 882nm; that of silicate-silicon in the water was estimated by molybdosilicate method (Grasshoff et al., 1983) at 810 nm; ammonia-nitrogen by the phenate method (Grasshoff et al., 1983); nitrite-nitrogen diazotised method and absorbance was measured at 543 nm (Strickland and Parsons, 1968; Grasshoff et al., 1983) and nitrate-nitrogen by resorcinol method (Zhang and Fischer, 2006) at 505 nm.

The chlorophyll *a* were estimated by vacuum filtration, acetone - extraction method by the membrane filtration assembly (APHA, 2005).

Redfield ratio (Nitrogen-Phosphorus ratio) is the atomic ratio of carbon, nitrogen and phosphorus throughout the world oceans in phytoplankton biomass and in dissolved nutrient pools. This empirically developed stoichiometric ratio was C:N:P = 106:16:1 (Redfield, 1934) and the ratios of carbon to nitrogen to phosphorus remained the same from coastal to open ocean regions.

2.4.2 Trophic index (TRIX)

Trophic Index analysis (TRIX) was used for the estimation of trophic status of Vembanad estuarine system (Vollenweider et al., 1998).

The following formula was used to quantify the eutrophication.

TRIX = (log10 [chl.a *Ab %O *DIN *DIP]+K) / M

Where:-

Chl.a	:	Concentration of chlorophyll a in mg/m^3
Ab %O	:	Absolute value of the percentage of dissolved oxygen saturation [Abs (100-%O)]
DIN	:	Dissolved inorganic nitrogen = (nitrate-NO_3 + nitrite- NO_2 + ammonium –NH4), in mg/m^3
DIP	:	Dissolved inorganic phosphorus in mg/m^3
K=1.5, M=1.2	:	Are the scale values introduced to adjust TRIX scale values with the levels of eutrophication.

TRIX was scaled from 0 to 10, covering a range of four trophic states (0-4 high quality and low trophic level; 4-5 good quality and moderate trophic level; 5-6 moderate quality and high trophic level and 6-10 degraded and very high trophic level) (Giovanardi and Vollenweider, 2004; Penna et al., 2004).

2.4.3 Sediment analysis

The sediment samples were collected using a standard van Veen grab of size $0.04m^2$. Sediment temperature was determined using standard degree centigrade thermometer in the field. A separate core sample of sediment was taken for further analysis of sediment parameters. pH were measured using Systronics model 371 water analyser and Eh (oxidation reduction potential) measured using Systronics digital Eh meter (Model No; 318) (APHA, 2005) and expressed in mV. The moisture content was determined by dry weight method (Jackson, 1973) and expressed in percentage. For further analysis the sediment samples was dried in shade and one portion of dried sediment was kept as such for textural analysis. The remaining portion was powdered using mortar and pestle and then the sediment sieved through a 2 mm sieve for the analysis of organic carbon, available nitrogen and available phosphorus (Jackson, 1973). Sediment organic carbon measured by Walkley-Black method (1934) method modified by Trivedy and Goel (1986) and dichromate digestion methods (Nelson and Sommers, 1982). Total carbon was analysed by using Analytik Jena TOC analyzer multi N/C 2100 S. Sediment organic matter was estimated by multiplying the sediment organic carbon value with a factor 1.724 (Trask, 1939). Available nitrogen was estimated by Kjeldhal method using Nitrogen Distillation System (Model: KEL PLUS DISTYL EM (VA) (Jackson, 1973; Carter, 1993). Available phosphorus was determined by Olsen's method (Olsen et al., 1954). Particle size analysis was done to estimate the percentage composition of sand, silt and clay of sediment samples. Particle size distributions were obtained using a laser scattering particle size analyzer and the result was expressed in percentage of sand, silt and clay. Each sample were dispersed by stirring in a solution of sodium hexametaphosphate in distilled water overnight (Folk, 1968), after which sand was removed by sieving (180 µm mesh), dried and weighed and the remaining suspension of silts and clays was sized using a particle size analyzer- Helos Sensor control (Sucell- Model KFS MAGIC, SYMPATEC

H70010) (Germany, 2000). Sodium, potassium and calcium were determined using flame photometer (Model – Systronic 128 μc) (Jackson, 1973).

Heavy metals, Copper (Cu), Zinc (Zn), Lead (Pb) and Cadmium (Cd) were analyzed seasonally during 2011-2012 period. For heavy metal analysis, 0.5g sediment samples were treated with 5:1 mixture of concentrated HNO_3 and $HClO_4$ (AOAC, 1995) and digest were analyzed using atomic absorption Spectrometer (Perkin Elemer Model AA700) in the Pollution Control Board Laboratory (PCB), Ernakulam. Based on the sediment quality guidelines (SQGS) (Perin et al., 1997), the pollution status of heavy metals were given in Table 2.3.

Table 2.3 Concentration of heavy metals and its pollution status based on the sediment quality guidelines (SQGS)

Element (mg/Kg)	Elemental background concentration	SQG non-polluted	SQG moderate polluted	SQG heavily polluted
Cu	55	<25	25-50	>50
Zn	70	<90	90-200	>200
Cd	0.2	<1	1-2	>6
Pb	12.5	<40	40-60	>60

2.4.4 Benthic faunal analysis

For meiofaunal analysis sampling was done seasonally during both the years. For quantitative sampling, sub samples (triplicate) for meiofauna were taken from the undisturbed van Veen grab (0.04 m^2) samples by using a cylindrical glass corer of 2.5cm inner diameter and 30 cm long (Eleftheriou and Mclntyre, 2005). In order to reduce the bow wave effects (wave remove the surface material and meiofauna), the sampling vessel was maintained in stationary and descending of sampler was at minimum speed (ICES, 1994). Before processing samples, it is important to check that the laboratory fresh water supply does not contain any meiofauna. For that tap water was run through a 63 µm sieve to make sure that water does not contain any meiofauna (Somerfield and Warwick, 2013). In the laboratory, core samples were extracted for meiofauna by $MgCl_2$ decantation (Crezee, 1976) using a set of 500 µm (remove larger polychaetes and other debris) and 63 µm sieves. The sediment retained in the 63 µm sieve was decanted to extract meiofauna (Higgins and Thiel, 1988). In order to extract the meiofaunal organisms from silt/clay sediment, the classical method of decantation by hand, using a 63 µm sieve was used (Somerfield and Warwick, 1996). The meiofaunal organisms were stained with Rose Bengal prior to extraction and were sorted and enumerated under a stereomicroscope. The meiofauna were identified into group level and were expressed in individual/10cm^2 (ind./10cm^2).

Grabs are generally used for quantitative sampling of macrofauna. For macrofauna, monthly triplicate samples were taken at each site by using standard van Veen grab of size 0.04m^2 and the sediment samples were sieved onboard through a 0.5 mm mesh sieve for two year period. The sieved macrobenthos with residual sediment samples were then preserved in 4 - 7 % neutral buffered formaldehyde containing Rose Bengal, which facilitate sorting of the organisms from other components of the soil in the laboratory (Holme and Mclntyre, 1984; Eleftheriou and Mclntyre, 2005; APHA, 2005). The sieved samples were then labeled and stored for further examination. For qualitative enumeration, each sample was examined under a binocular microscope. The organisms were separated into different taxonomic groups and then enumerated and expressed as ind./m^2. Organisms were identified upto lowest possible level as using standard references. The fragmented specimen, difficult for species level identification was limited to its possible level. Taxonomic identification of oligochaetes, polychaetes, molluscs and crustaceans were done up to species level (Brinkhurst and Jamieson, 1971; Naidu, 2005; http://apps.biodiversityireland.ie/OligochaeteIdKey/key; Fauvel, 1953; Day, 1967; Fauchald 1977; Cunha et al., 1999; Moreira et al., 2008;

Wang et al., 2010; Guerra-Garcia, 2010; Esquete et al., 2011). Validity and taxonomic status of species checked from the World Register of Marine Species (WoRMS, www.marinespecies.org).

Macrofaunal biomass was estimated by the wet weight method. The shells of molluscs and the tubes of the tube dwelling polychaetes were removed prior to weighing. Wet weight of benthic fauna was taken by using high precision electronic balance and it was expressed in g/m^2. Single organisms with greater than 0.5g wet weight was not taken for extrapolating to $1m^2$.

2.4.5 Molecular analysis

For molecular analysis separate grab sample were taken and sieved onboard through a 0.5 mm mesh sieve and the samples were stored into 95% Ethanol and stored at 4°C. DNA barcoding of polychaete species were carried out by DNeasy Blood and Tissue Kit (Qiagen, Valenica, CA). The standard mitochondrial cytochrome oxidase I (COI) gene was selected as a DNA barcode region and was amplified using universal primers (Folmer et al., 1994). PCR amplifications were performed in 25 µL reaction volumes using PCR reagents as described by Blanco-Bercial et al. (2011) by using a gradient thermal cycler (Bio - Rad, Hercules, CA, Model Number 621BR07085). Purified products were given to SciGenom Labs (SciGenom Labs Pvt, Ltd., Kerala) for sequencing and sequences obtained were compared with public data base, Basic Local Alignment Tool (BLAST) - National Center for Biotechnology Information (NCBI) and also the sequences were submitted to NCBI’s Genbank.

2.4.6 Biotic indices

The ecological quality of benthic environment was assessed by using indices such as AZTI-Marine Biotic Index (AMBI), Multivariate AMBI (M-AMBI), BENTIX and Benthic Opportunistic Polychaetes Amphipods (BOPA).

2.4.6.1 AZTI-Marine Biotic Index (AMBI)

The AMBI is based on the distribution of the abundance of benthic communities into five ecological groups (Grall and Glemarec, 1997) and the AMBI was calculated using the AMBI software available on the AZTI site (http://www.azti.es) (Borja et al., 2000; Borja et al., 2007) (Table 2.4). Based on the sensitivity to the toxicant, benthic biota were classified into five ecological groups,

EG I : species sensitive to organic enrichment and found in unpolluted areas
EG II : species which are unaffected by disturbances and found in low densities
EG III : species tolerant to high organic matter enrichment
EG IV : second order opportunistic species
EG V : first order opportunistic species

The index value was calculated using the following equation

AMBI = [(0 x %EGI) + (1.5 x %EGII) + (3 x %EGIII) + (4.5 x %EGIV) = (6 x %EGV)]/100

Table 2.4 Summary of the AMBI values and their equivalences (modified from Borja et al., 2000)

Biotic coefficient	Dominating ecological group	Benthic community health	Site disturbance classification
$0.0 < AMBI \leq 0.2$ $0.2 < AMBI \leq 1.2$	I	Normal Impoverished	Undisturbed
$1.2 < AMBI \leq 3.3$	III	Unbalanced	Slightly disturbed
$3.3 < AMBI \leq 4.3$ $4.3 < AMBI \leq 5.0$	IV - V	Transitional to pollution Polluted	Moderately disturbed
$5.0 < AMBI \leq 5.5$ $5.5 < AMBI \leq 6.0$	V	Transitional to heavy pollution Heavy polluted	Heavily disturbed
$6.0 < AMBI \leq 7.0$	Azoic	Azoic	Extremely disturbed

The species/groups which were not assigned to various groups in the AMBI classification were not included in the analysis. Based on this, during the present investigation only 31 and 27 species/groups of macrofauna were selected during first and second year respectively.

2.4.6.2 M – AMBI (Multivariate AMBI)

It was proposed by Borja et al. (2004) and then detailed by Muxika et al. (2007). The results are the combination of Shannon's diversity, richness and AMBI, into a factor analysis (FA) multivariate approach which appears to be a suitable method to evaluate the ecological conditions using AMBI software. Based on the M-AMBI classification the 'high' status was considered as an optimum where the M-AMBI value was 1. At 'bad' condition, the M-AMBI was 0 (Muxika et al., 2007). The reference values were 'High' quality, > 0.8; 'Good', 0.57–0.80; 'Moderate', 0.38–0.57; 'Poor', 0.20–0.38; and 'Bad', < 0.20.

2.4.6.3 BENTIX

The biotic index, BENTIX, was calculated for the assessment of the ecological quality (ECoQ) of the study area (Table 2.5). It was calculated by using the formula,

$$\text{BENTIX} = \frac{6 \times GS + 2 \times GT}{100}$$

Where,

GS = EG1 + EG11 and GT = EGIII + EGIV + EGV

It assigns the numerical factor '6' for the sensitive taxa group and the factor '2' for the tolerant taxa groups.

Table 2.5 Benthic habitat classification based on the Bentix index (Simboura and Zenetos, 2002).

Pollution Classification	**Bentix**	**EQS**
Normal/pristine	4.5 ≤ Bentix < 6	High
Slightly polluted, transitional	3.5 ≤ Bentix < 4.5	Good
Moderately polluted	2.5 ≤ Bentix < 3.5	Moderate
Heavily polluted	2.5 ≤ Bentix < 2.5	Poor
Azoic	0	Bad

2.4.6.4 Benthic Opportunistic Polychaete Amphipod ratio (BOPA index)

It was based upon the comparative frequencies of opportunistic polychaetes and amphipods (except the genus Jassa (opportunistic) (Dauvin and Ruellet, 2007). BOPA index was calculated by using the formula,

$$BOPA = \log [(fp / fa + 1) + 1],$$

where fp is the opportunistic polychaete frequency (ratio of total opportunistic polychaetes to the total individuals in the sample); fa is the amphipod frequency (ratio of total amphipods to the total population in the sample). Its value can range between 0 and 0.3 (Pinto et al., 2009). BOPA values grouped the environment into five different classes from unpolluted to extremely polluted or azoic sites (de-la-Ossa-Carretero et al., 2009). The classification is

High (unpolluted sites)	0 ≤BOPA≤0.025
Good (slightly polluted)	0. 025≤BOPA≤0.13
Moderate (moderately polluted)	0.13≤BOPA ≤0.199
Poor (heavily polluted)	0.199≤BOPA≤0.255
Bad (extremely polluted)	0.255≤BOPA≤0.3

2.4.7 Macrobenthic functional group analysis

2.4.7.1 Feeding guild assignments

Based on the food type each taxon was grouped to a feeding guild and this information was get from peer-reviewed literature, species identification guides, online databases and from personal expert consultations. Based on the trophic guild classification, macrofauna were grouped as surface deposit feeder (SDF), subsurface deposit feeder (SSDF), suspension feeder/filter feeder (FF), grazer (GR), omnivore (O), and carnivore (C) (Fauchald and Jumars, 1979; Gaston, 1987; Mancinelli et al., 1998; Wieking and Kroncke, 2005; MarLin species database - http://www.marlin.ac.uk/sah/species information.php; Macdonald et al., 2010; Sivadas et al., 2016). Species which were not strongly classified and so it was grouped as unknown (U). Some cases missing data were add-on by using information referring to closely related species.

2.4.8 Data analysis/statistical analysis

For statistical analysis software programs like SPSS Vs.16.0 (Statistical Programme for Social Sciences version 16.0) were employed to find out the two way ANOVA (Analysis of Variance), standard deviation, correlation etc. among parameters. Pearson's correlation shows the linear relationship between two sets of data. ORIGIN 8 software was also used for the graphical representation of data. PRIMER Vs.6.1.8 (Plymouth Routines in Multivariate Ecological Research, version 6.1.8) was used for univariate and multivariate statistical analysis and plotting of data (Clarke and Gorley, 2006). Principal Component Analysis (PCA) and Canonical Correspondence Analysis (CCA) performed using PAST V. 2.17c software.

2.4.8.1 Univariate analysis

Diversity indices and taxonomic distinctness index

Species diversity-Shannon index (Shannon and Wiener, 1949). It explains both abundance and evenness of species present in the community. It is explained by the formula:

H' = -ΣS Pi log 2 Pi.....
i = 1

where

H' = the species diversity

S = the number of species

pi = the proportion of individuals of each species belonging to the species of the total number of individuals (number of individuals of the i^{th} species)

Species richness- Margalef's index is a measure of the number of species present for a given number of individuals (Margalef, 1958). It is given by

d = (S-1) / log N

where

d = species richness

S = total number of species

N = total number of individuals

Species evenness- Pielou's index (Pielou, 1966). The index measures equitability and allows comparison of Shannon Weaver index with the distribution of individuals in the observed species that would have the maximum diversity. It is calculated as

j' = H'/log2 S or H'/ln S

where

J' = evenness,
H' = species diversity
S = total number of species

Species dominance- Simpson's index (Simpson, 1949)

The **Simpson's** index is a measure of both the richness and proportion (percentage) of each species. It is calculated by the formula

$$D = 1/\lambda$$

Where

$\lambda = \Sigma Pi^2$

$Pi = ni/N$

ni = number of individuals of i, i^2 etc.

N = total number of individuals.

Taxonomic diversity (Δ) and the taxonomic distinctness (Δ*) are indices based on the taxonomic spread of species. The indices assess average taxonomic separation of all pairs of individuals in a sample. The taxonomic diversity (Δ) is empirically related to the Shannon–Wiener index but includes an additional component of taxonomic separation, whereas the taxonomic distinctness (Δ*) is a measure purely of taxonomic distinctness (Warwick and Clarke, 1995). Taxonomic distinctness is reduced in respect to increasing environmental stress and this response of the community lies at the base of this index's concept.

Taxonomic diversity index (Δ) - measures the average taxonomic distance between any two individuals chosen at random belonging to separate species was calculated using the formula,

$$\Delta = [\sum\sum_{i<j} \omega ij\ x_i\ x_j]/[N(N-1)/2],$$

where $N = \sum ixi$, the total number of individuals in the sample.

Average taxonomic distinctness index (Δ^+) - It is the average taxonomic distance apart of all its pairs of species.

$$\Delta^+ = [\sum\sum_{i<j} \omega ij]/[S(S-1)/2],$$

where S is the number of species present, the double summation is over the set $\{i = 1; j = 1, S;... S,$ such that $i < j\}$ and ωij is the 'distinctness weight' between species i and j.

Variation in taxonomic distinctness index (Λ^+) - is a measure of the evenness of the spread of species across higher taxa

$$\Lambda^+ = [\sum\sum_{i<j} (\omega ij - \Delta^+)^2]/[S(S-1)/2],$$

where, ωij is the variance of the taxonomic distance between each pair of species i and j about their mean value of Δ^+ and S is the number of species

Total phylogenetic diversity index (sPhi+) - It is simply a cumulative branch length of the full tree drawn using the Linnaean classification.

95% confidence limit funnel plot- Average taxonomic distinctness index (Δ^+) and variation in taxonomic distinctness (Λ^+) were used to construct funnel plots to test for any significant departure of species from the expectation (Warwick and Clarke, 2001).

Species accumulation plot

In order to estimate the species richness of polychaete fauna, various species estimators such as Chao 1 (Chao's estimator based on number of rare species) Chao 2 (Chao's estimator using just presence-absence data), Jacknife 1 (based on species that only occur in one sample) Jacknife 2 (Second order jacknife estimator), Bootstrap (based on proportion of quadrats containing each specie s), UGE (Calculated species accumulation curve (Ugland et al., 2003), SOBS (Curve of observed species counts) and MM (Michaelis-Menton -Curve fitted to observed S curve) were done in PRIMER.

2.4.8.2 Multivariate analysis

Bray-Curtis similarity index: Multivariate analysis consisted of estimating Bray-Curtis similarity after suitable transformation of sample abundance data. The similarity matrix was subjected to both clustering (hierarchical agglomerative method using group average linking) and ordination (nonmetric multidimensional scaling, NMDS). Cluster analysis was done to find out the similarities between groups. Hierarchical clustering methods are commonly used. The Bray- Curtis similarity measure was used to produce the similarity matrix and the similarity percentage was used to determine the degree of similarity. The dendrogram with the X-axis representing the full set of samples and the Y – axis defining the similarity level at which the samples or groups are fused.

Nonmetric multidimensional scaling (NMDS): was used to graphically display the two-dimensional ordination plots representing the similarity ranking of groups/species composition between different stations. A stress value of < 0.2 gives a useful representation of results (Clarke and Warwick, 2001). In nMDS "stress coefficient" showed the extent to which the relationship between the samples was adequately represented in the resulting 2-D ordination. The red lines represent 80% similarity contour, blue line 40% and green 20%.

Bubble plots: The abundance of individual species over the station was represented in bubble plots, where the smaller bubble is denoting less abundance and larger bubble denoting higher abundance.

K-dominance plot: which rank species in decreasing array of their abundance on the x-axis (logarithmic scale) with the percentage of the total abundance of all species in the sample on the y-axis (cumulative scale). These were broadly applied for finding out the pollution effects on macrobenthos (Clarke, 1990). The curve of 'J' shaped representing the dominance of opportunistic species (disturbed condition) whereas 'S' shaped curve indicate occurrence of conservative species (undisturbed condition).

Abundance Biomass Comparison curve (ABC plot): used to evaluate the disturbances based on the trend of ABC curve at the particular site without any reference site (Warwick, 1986). Uniformity in the distribution of abundance and biomass values represents the level of stress in the community. In undisturbed communities the biomass curve lies above the curve for abundance. Under moderate pollution (or disturbance), the biomass and abundance curves are closely coincident and may cross each other one or more times. In polluted condition, abundance curve lies above the biomass curve throughout its length. The W- value (Warwick value) were used to statistically define the relationship between curves and quantify the level of stress that a community experiences. When the biomass curve is above the abundance curve the W-value will be positive. The negative W-value occurred when the abundance curve is above the biomass curve, with intermediate cases tending toward zero.

BIO-ENV Analysis: BIO-ENV procedure was used to determine which environmental data, such as physico-chemical data, is most responsible for structuring the benthic community (Clarke and Warwick, 2001). The BIO-ENV procedure involves the evaluation of link between the two similarity matrices: the fixed biotic similarity matrix (using Bray-Curtis similarity on biotic data) and each of the possible abiotic matrices (PCA on combinations of the environmental data). The BIOENV routine, which is based on weighted Spearman rank correlations (ρ) between the similarity matrices underlying the biotic MDS and the Euclidean distance matrices for all combinations of environmental variables, thereby defining suites of environmental variables that best explain the measured biological structure.

Similarity Profile Analysis (SIMPROF): SIMPROF test was carried out for detecting statistically significant cluster (Clarke and Gorley, 2006). It conducts a series of permutation tests to determine if clusters in the dendrogram have statistically significant structure.

Analysis of Similarity (ANOSIM): ANOSIM significance test was performed in order to provide a test for significant differences between two or more groups of sampling units. Here, calculate the significance level by referring the observed value of R to its permutation distribution (Clarke and Warwick, 2001). R value varied between -1 to +1. When R value close to zero, denote the clear distinction between samples (Clarke et al., 2006).

Similarity Percentage Analysis (SIMPER): This analysis assesses the average percentage contribution of individual variables to the dissimilarity between objects in a Bray-Curtis dissimilarity matrix. This allows observing the variables that are important in contributing any similarity/difference between groups detected by methods such as ANOSIM.

Principal Component Analysis (PCA): It is a powerful tool that tries to elucidate the variance of a large dataset of intercorrelated variables with a smaller set of independent variables (Simeonov et al., 2003). This analysis was performed using PAST V. 2.17c software (Paleontological Statistic software package) (Hammer et al., 2001). The assessment of the index was done using a Pearson correlation between the scores in PCA axis and the environmental variables. The most significant variables in the components represented by high loadings (>0.6) are taken into consideration for evaluating the components.

Canonical Correspondence Analysis (CCA): This analysis is a multivariate method to elucidate the relationships between biological assemblages of species and their environment. It is an example of direct gradient analysis, where the gradient in environmental variables is known a prior and the species abundances (or presence/absences) are considered to be a response to this gradient. In the ordination diagram, where the taxa are represented by points in relation to major ordination axis, and environmental factors indicated as vectors from the centre of the ordination. A perpendicular line is traced between the species and environmental vector, represents the relation between the species and the factor (TerBraak and Verdonschot, 1995). This analysis was performed using PAST V. 2.17c software.

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Chapter 3

PHYSICO-CHEMICAL FACTORS STRUCTURING THE TROPHIC ENVIRONMENT OF VEMBANAD ESTUARY

Contents

3.1 Introduction

Estuaries are transition zones between river and ocean environments influencing productivity of both the systems. Estuarine environment is a complex, continuously changing habitats, that has wide range of physical and chemical variations that occur with several events. It represents one of the most sensitive and ecologically important habitats on earth. Physical condition and functioning of majority of the estuaries in the world are threatened due to various levels of interactions mainly from anthropogenic pressure, which stands top; industrial and agriculture pollution, over exploitation of natural resources and various developmental activities within the estuarine environment (Morant and Quinn, 1999). Most of the coastal estuaries are experiencing degradation mainly due to urban growth and development. Consequently input of nutrients especially nitrogen and phosphorus is increasing tremendously causing several environmental problems including water quality deterioration and probable biodiversity loss (Glibert et al., 2010). Of late tropical estuaries, the most active biogeochemical zones in the biosphere, are more easily getting affected by anthropogenic nutrient loading than temperate estuaries (Yule et al., 2010).

Throughout the world, estuaries and coastal waters have experienced degradation. The successful management of estuaries and coastal waters requires an ecohydrology based approach. Without effective thinking and management concepts, estuaries and coastal waters will continue to degrade, whatever integrated coastal management plans are implemented. To help in this process of change there is a need to develop a profound understanding on the effects of biota and biotic processes on mediating estuary response to changing hydrology, sediment and nutrient flux on the biota and hydrology. About 60% of the world's population lives along the estuaries and the coast (Lindeboom, 2002). The increase of human populations in the river basins, from natural growth and internal migration within the hinterland, has resulted in a doubling of the population along many coasts over the last 20 years. This is degrading estuarine and coastal waters through pollution, increasing turbidity from eutrophication, overfishing and habitat destruction. The pollutants include increasing nutrients, mud from eroded soil, heavy metals, radionuclides, hydrocarbons, and a number of chemicals including new synthetic products. Estuaries are often regarded as sites for future development and expansion, and have been increasingly canalized and dyked for flood protection, and their wetlands were modified for residential areas (Wolanski et al., 2004).

Water management is an essential component in the organization and conservation of natural resources. Generally the water quality characteristics of any ecosystem give considerable knowledge about the existing life supporting condition of that system. The interactions of important environmental factors vary with space and time and play an important role in deciding the existence of many biotic communities. The information about these parameters was vital for understanding the dynamics of communities and also essential for obtaining the knowledge about different trophic levels and food webs. So achieving a detailed and systematic understanding of the ecosystem status of an estuary is remarkable for the estuarine management and also essential for predicting the response of the coastal ecosystem to the continuously progressing anthropogenic stresses.

3.1.1 Significance of the study

Estuaries provide multiple functions, significant to social, economic and environmental values. Over the years the availability of domestic and industrial wastes, urban discharges and agricultural effluents increase the nutrient concentration by many times to the levels that occur naturally causing several environmental modifications in estuarine and coastal waters. Sustaining good water quality condition was important not only for ecosystem health, but also for providing service and health to people. The Vembanad estuary is passing through a phase of rapid ecological modifications. Several of the changes have been taken place on the hydrology and environmental quality in the estuary. There has been lot of alterations in the upland areas of Vembanad, on account of the various projects as well as the increased use of water for a wide array of activities. There are also *in-situ* changes in the Vembanad area on account of various interventions. Scientific information on the ecosystem based analysis of the ecological processes on a comprehensive basis is severely lacking from the Vembanad estuarine system after the commissioning of the barrage. It was in this context that, this study is undertaken on the appended objective to precisely investigate the ecohydrological characteristics of the Vembanad estuarine system for its wise use and also enhance ecosystem services for society.

3.1.2 Review of Literature

Estuaries are the meeting place of fresh and salt water and formed complex physical, chemical, and biological environments, where the salinity and flow of water are influenced by constant changes in river inflow and tides. This complexity has made the study of estuaries and their associated tidal rivers very difficult; as a result the processes that take place within them are poorly understood (Callender et al., 1984). In order to understand this complex environment several studies has been conducted by several authors over the years. Bally and McQuaid (1985) investigated the physical and chemical characteristics of the waters of the Bot river estuary, South Africa. The study noted that physical and chemical properties were mostly affected by artificial openings of and water-levels in the estuary. Wolanski et al. (2004) pointed out that ecohydrology as a new tool for sustainable management of estuaries and coastal waters. Ecohydrology is the science of the interplay between hydrology and biota (Zalewski, 2002). Long-term trends of water quality in Chesapeake Bay from 1986 to 2008 was evaluated by Williams et al. (2010). They evaluated the ecological health-related water quality parameters such as chlorophyll-a, dissolved oxygen, Secchi depth, biotic metrics such as phytoplankton and benthic indices. They found that over the 23-year period, water quality have generally shown little improvement and water clarity and chlorophyll-a have considerably worsened. Assessment of physical-chemical characteristics of water and sediments from a Brazilian tropical estuary was made by Pereira et al. (2012). Eyre and Balls (1999) made a comparative study of nutrient behavior along the salinity gradient of three tropical estuarine systems from North Queensland, Australia (Jardine, Annan, Daintree) and three temperate estuarine systems from Scotland, United Kingdom (Inverness, Cromarty, Dornoch Firths) for comparison. Xu et al. (2012) observed the climate forcing and salinity variability in Chesapeake Bay, USA. They noted that understanding of the salinity structure, salinity variability is necessary to advance our understanding of the physical, chemical, and biological dynamics of estuaries.

Trophic state measurements serve as standards for measuring the health of an estuarine system. Various techniques and methods have been recognized to assess the trophic status and eutrophication of aquatic systems

(Carlson 1977; Vollenweider et al., 1998). Human-induced nutrient enrichment is becoming a serious problem for coastal marine areas around the world (Garmendia et al., 2012). Eutrophication of coastal waters has resulted in a number of adverse effects on ecosystem, including algal growth, hypoxia, significant loss of marine life and habitat and the contamination or impairment of commercial shellfish stocks (De Jonge et al., 2002; Cook et al. 2010). Tropical estuaries, which formed the most active biogeochemical zones in the biosphere, are more easily affected by anthropogenic nutrient loading than higher latitude (Yule et al., 2010). Based on selected biotic and abiotic parameters Nawrock et al. (2011) determined the trophic state of the Vistula Lagoon, brackish water lagoon on the Baltic Sea. Baodong (2007) made the assessment of trophic status of Changjiang (Yangtze) River estuary, China (one of the largest estuaries in the world). Based on the assessment of estuarine trophic status (ASSETS), the overall status of estuary was graded as bad. Nagy et al. (2002) studied the distribution patterns of nutrients and symptoms of eutrophication in the Rio de la Plata river estuary system, Uruguay. The status and characteristics of eutrophication in the Yangtze River (Changjiang) estuary was investigated by Chai et al. (2006). They investigated the nitrate, ammonium, nitrite, soluble reactive phosphorus, dissolved reactive silica, dissolved oxygen, phytoplankton chlorophyll *a* and suspended particulate matter. Alves et al. (2013) evaluated the eutrophication and water quality in a tropical Massangana river estuary, Brazil. They used TRIX index and an ASSETS model for evaluating the level of eutrophication in the estuary and found that TRIX quantified the Massangana river estuary at a level medium of eutrophication and ASSETS qualified it as moderate. Based on the Carlson’s trophic state index Elmaci et al. (2009) indicated that lake Uluabat, Turkey is an eutrophic system. Monteiro et al. (2016) investigated effects of anthropogenic process on the trophic status of an Amazonian estuary (Caete estuary), Brazil under different climate conditions. Based on the trophic index (TRIX), the waters of the Caete estuary were characterized by moderate levels of eutrophication and good water quality during the wet season, shifting to high levels of eutrophication and bad water quality during the dry season. The study also found that eutrophication was very less during neap tides when compared to spring tides.

Several studies have been carried out on the ecology of estuaries and backwaters of India. Along the east coast Hoogly-Matlah estuarine complex (West Bengal), Chilka lake (Orissa), Pulicat lake and Vellar estuary (Tamil Nadu) are the major brackish water systems. Mandovi-Zuari estuarine complex (Goa), Gurpur Nethravati, Mulki-Pavanje and Kali estuary (Karnataka) and Vembanad estuarine system (Kerala) are the major estuarine systems along the west coast. Estuaries of east and west coasts of India differ from each other by the geographic condition, monsoonal regime, the river water influx and tidal range. Environmental characteristics especially the primary productivity of Ashtamudi estuary in Kerala was investigated by Balakrishnan Nair et al. (1984c). Balakrishnan Nair and Abdul Azis (1987) studied the hydrobiology of the Ashtamudi estuary in Kerala. Raman et al. (1990) examined the physico-chemical features of water quality parameters and phytoplankton characteristics of Chilka lake. The study revealed that the distribution of phytoplankton species in the lake was determined by salinity gradient. Bijoy Nandan and Abdul Azis (1990), discussed about the biological oxygen demand and its relation to the dissolved oxygen in the coconut husk retting zones in Kadinamkulam backwater. Raman (1995) conducted a 23 years of investigations and monitoring on the pollution effects in Visakhapatnam harbour from 1976. The study found that over the years industrial and urban wastes caused high degree of pollution in the harbour. Similarly the nutrient status revealed that concentration of nitrites (max. 5 mg L^{-1}) and phosphates (9.6 mg L^{-1}) has increased over the years. Environmental status of the estuarine ecosystem of Kerala was investigated by Bijoy Nandan (2004). Water quality status of the

Adimalathura estuary on the south west coast of India was observed by Anila Kumary et al. (2007). Various research aspects have been conducted in Chilka lake, Orissa. Annadle and Kemp, in 1915 started the hydrographic investigations on Chilka lake. Banerjee and Roychoudhury, (1966) observed the physico-chemical features of Chilka lake. Sarkar (1977) studied the seasonal variations of salinity whereas Subba Rao et al. (1981) observed the hydrological features of the lake. Panda et al. (1989) studied the distribution of nutrients in Chilka lake. Other major studies on the environmental characteristics of Chilka lagoon was carried out Nayak et al., 2004; Panigrahi et al., 2007; Bramha et al., 2008; Pradhan et al., 2012. Temporal and spatial distribution of physico-chemical properties of water of Dharma estuary, Orissa was investigated by Prasanna et al. (2010). Impact of dredging on the water quality of Dharma estuary was observed by Sangita et al. (2014). Water quality assessment of Gautami - Godavari mangrove estuarine system was conducted by Tripathy et al. (2005). Manna et al. (2012) investigated the interplay of physical, chemical and biological components in the Sundarban estuarine ecosystem, India. Influence of southwest monsoon on the water quality and assimilative capacity of Mandovi estuary, India was investigated by Vishnu Radhana et al. (2015). Vijith et al. (2009) investigated the time dependence of salinity in the Mandovi Zuari estuary, India. Sarma et al. (2011; 2012) investigated the CO_2 emissions from Godavari estuary and other major estuaries of India respectively. Physico-chemical features of Indian estuaries was studied by several authors, Kottakudi and Nari backwaters, south east of Tamil Nadu (Thirunavukkarasu et al., 2014), Veli lake, Thiruvananthapuram, Kerala (Leena Grace, 2007), Mulki estuary, Karnataka, south-west coast of India (Vijaya Kumar et al., 2010), the Arasalar estuary south-east coast of India (Muthukumaravel et al., 2012), Thengapattanam estuary, Tamil Nadu, south west coastal zone, (Anitha and Kumar, 2013), Agniar estuary, Tamil Nadu, south-east coast of India (Sukumaran et al., 2013), Tapi estuary, Gujarat, West coast of India (Ranjana et al., 2013), Thengaithittu estuary, Puducherry, south east-coast of India (Vijayakumar et al., 2014), Poonthura estuary, Kerala, southwest coast of India (Beslin, 2014), Manakudy estuary, Tamil Nadu, south west coast of India (Arumugam et al., 2014), and Sarada and Varaha estuarine complex, Andhra Pradesh, east coast of India (Vijaya Pratap and Ramesh Babu, 2015).

Monitoring of basic hydrological and ecological characteristics of the Vembanad estuary especially the Cochin estuarine area was carried out by several authors. Majority of the studies were reported during 1960 to 1980 period. Balakrishnan (1957) was one of the earliest to study the surface salinity of the Ernakulam Channel, attributed to the rapid salinity fluctuations due to the influence of tide. George and Kartha (1963) recorded the surface salinity of Cochin backwaters in relation to the tidal regime and did not find any tidal influence on the surface salinity of the Channel. Ramamirtham and Jayaraman (1963) observed a vertical gradient of temperature of 4-6°C during MN, in which the surface having flood water from rivers and in bottom having low temperature, well oxygenated upwelled water. Before in PRM the vertical gradient disappears, and complete mixing occurs. Qasim and Gopinathan, 1969; Sankaranarayan and Qasim, 1969; Devassy and Gopinathan, 1970; Haridas et al., 1973; Balakrishnan and Shynamma, 1975; Sarala Devi et al., 1979; Remani et al., 1980; Udayavarma et al., 1981; Ramamirtham and Muthuswamy, 1986 conducted extensive studies in Cochin estuary especially on the physico-chemical and biological characteristics. Josanto (1971) analysed the bottom salinity characteristics and the factors that influence the saltwater penetration in the Vembanad estuary extending from the Cochin bar mouth to Alappuzha. Pillai et al. (1973) analyzed the tidal currents around Cochin bar mouth and their influence on the hydrographic parameters. According to Ramamirtham et al. (1986), the monsoon season, in Cochin estuary from Pallipuram to Thevara behaves as a highly stratified salt wedge type of estuary, whereas during the premonsoon and post monsoon periods, it behaves fairly partially mixed type and fairly vertically homogeneous type. Vertically homogeneous type character was found in the shallowest regions from Perumbalam and Pallipuram. Anirudhan et al. (1987) conducted intensive studies on various ecological aspects of Cochin estuary.

Nutrient status and its dynamics in Cochin estuary were investigated by several researchers. Qasim and Sankaranarayanan, 1972; Joseph, 1974; Sreedharan Manikoth and Salih, 1974; Pillai et al., 1975; Balakrishnan and Shynamma, 1976; Sankaranarayanan et al., 1986; Lakshmanan et al., 1987; Anirudhan and Nambisan, 1990; Sarala Devi et al., 1991b etc. gave a detailed account on the hydrographic condition and distribution and variability of nutrients in Cochin estuary. Gopalan et al. (1983) pointed out the alarming rate of vertical and horizontal shrinkage of the Vembanad estuary by about 35%, and also on the siltation caused by river discharge. Pillai (1991) pointed out that, the physico-chemical characteristics of Vembanad estuarine system showed significant variation with time and space. It was mainly due to the tidal flow from Arabian sea as well as from the fresh water discharges from the rivers. The suspended sediment transport and residual salt flux in the lower reaches of Cochin estuary and west of Kumbalam Island was investigated by Revichandran et al. (1993). Kumar et al. (1994) made a detailed review on the historical trends and implications for the future of the Vembanad estuary. He mentioned about the importance of the significant role played by the estuarine system in the socio-economic and cultural aspects and emphasized upon the need to protect the system by new methodologies. Spatial and temporal variations of hydrography, floral and faunal composition and the impact of the anthropogenic activities of Cochin backwater was reviewed by Menon et al. (2000). Time scale changes in the Vembanad wetland ecosystem due to Thanneermukkom Barrage was investigated by Bijoy Nandan and Unnithan

(2004). Hydrodynamic and geomorphic controls on the morphology of an island ecosystem in the Vembanad lake was done by Laluraj et al. (2008).

Studies on tide dependent salt-silt wedge and identification of turbidity maxima zone in Cochin estuary was observed by Renosh et al. (2010). The study consisted of development of salt-silt wedge in the estuarine environment and the identification of turbidity maxima zone in the pre-monsoon season in southern arm of Cochin estuary. Similarly oscillatory behaviour of the turbidity maxima zone was also analysed with respect to the tidal rhythm. Primary productivity and water quality status of Valanthakad backwater, part of Cochin estuary was reported by Meera and Bijoy Nandan (2010). Jayachandran et al. (2012) focused on the water quality characteristics in relation to various anthropogenic activities in the Kodungallur – Azhikode estuary (northern part of Vembanad estuarine system) and observed a mixo - mesohaline condition with clear vertical stratification along with the influence of high river discharge on nutrient and silt loading. Assessment of nutrients using multivariate statistical techniques in estuarine systems and its management implications in Cochin estuary was done by Joseph and Ouseph (2010). They noted that nutrient enrichment from run-off formed a major influencing factor for the estuarine water quality variation. Madhu et al. (2007) made an environmental study in Cochin estuary and showed that seasonal variation in salinity was associated with increased monsoonal rainfall and run off that was an important element regulating the micro - and mesozooplankton distribution and abundance. Fresh water influence on the nutrient stoichiometry of Cochin estuary was investigated by Martin et al. (2008). The study showed that monsoon altered the estuarine hydrology by modifying the nutrient concentration and their stoichiometric ratio and also secondary production etc. Hydrographic characteristics exclusively in the bottom water of Cochin estuary were studied by Martin et al. (2010). They also reported the formation of anoxia and denitrification in the bottom waters of lower Cochin estuary and it was the first time report in the estuaries along the West coast of India. Jacob et al. (2013) studied the salt water intrusion in Cochin estuary using empirical model. They observed that stratification parameter of salinity varied with the distance along the estuary, tide and fresh water discharge indicating that Cochin estuary experiences a transition from partially or well mixed estuary during post and premonsoon to a strongly stratified estuary during monsoon season. Janardanan et al. (2015) studied the hydrographic conditions of Cochin estuary and showed that river discharge have a strong influence on the seasonal variability of salinity. During peak runoff periods Cochin estuary confirmed a transition from salt wedge type to a well-mixed condition during peak dry season. Balachandran et al. (2008) developed a model on the tidal hydrodynamics and pollutant dispersion and its implication in Cochin estuary. He divided the whole estuary into three zones includes, a central dynamic estuary and northern and southern estuary having flow restrictions that was sensitive to pollution. Robin et al. (2012) studied the hydrogeochemical characteristics of Cochin estuary by using environmental techniques. Water quality assessment and isotope studies of Vembanad wetland system indicated that the physico-chemical and microbiological status of the wetland system had been worsened by pollutants (Nasir, 2010). The study reported that, water quality has dangerously ruined by different organic and inorganic pollutants of different origin. Nutrient level in water was increasing to cause eutrophication and noted hypereutrophic stage in many parts of the water body.

Based on the salinity variation of Cochin estuary, ranging from completely riverine to completely saline conditions, Shivaprasad et al. (2013c) proposed a new terminology to Cochin estuary as “Monsoonal Estuarine Bay”. He states that new terminology provides basic information about the physiographic, hydrographic, salinity and ecological features of the water body. Radhika (2013) assessed the influence of general hydrobiology and nutrient dynamics on trophic interactions in Cochin backwater system. Karlaganis and Narayanan (2014) made a case study on the environmental regulations and challenges of houseboat tourism in Vembanad lake. They observed that tourism related environmental degradation in Vembanad lake was mainly due to the boat owner’s desire for short-term profit maximization and lack of infrastructure. By using the Geographical Information System (GIS) software, spatial distribution of non-point source pollution in Vembanad lake was observed by Paul et al. (2014). Safoora Beevi and Devadas (2014) made an assessment on the impact of tourism on Vembanad lake system in Alappuzha District. They observed different types of human intervention on the lake, which affects the function, structure and entire composition of the wetland system. Residual fluxes of water and nutrient transport through the inlet of Cochin estuary were investigated by Revichandran et al. (2015). In Cochin estuary, the nitrogen uptake and its dynamics was investigated by Bhavya et al. (2015). Bijoy Nandan et al. (2014) observed alarming changes in the ecology of Vembanad estuarine system influencing the regional dynamics of planktonic, benthic and fishery structure. Similarly Asha et al. (2016) gives an over view of the degrading status of Vembanad estuarine ecosystem and signifies the need for better sustainable restoration programmes. Cleetus et al. (2016) studied about the relationship between hydrographic parameters and mesozooplankton in the Vembanad estuarine system.

3.2 Results

The physical and chemical properties of bottom water in the study area and its spatial, temporal and seasonal variations are explained in this part. The parameters such as rainfall, depth, pH, temperature, transparency, dissolved oxygen, turbidity, conductivity, TDS, salinity was determined. Nutrients such as nitrite-nitrogen, nitrate-nitrogen, ammonia- nitrogen, phosphate-phosphorus and silicate-silicon were also determined from the bottom water samples. The temporal variations, results of ANOVA and correlation analysis of physico-chemical parameters are given in annexure 3.1 to 3.9. Comparison of physico-chemical parameters on the south and north of TMB and also during the open and closure period of TMB are given in Annexure 3.10 to 3.11.

3.2.1 Meteorological Parameters

3.2.1.1 Rainfall

During 2011 – 2012 period rainfall was maximum observed during June 2011(442.2 mm) and the minimum was in January 2012 (5.16mm) (Fig. 3.1). The annual rain fall during 2011- 2012 period was 2412.5mm. Seasonal analysis showed that maximum rainfall observed during MN (1478.49 mm) followed by PM (510.93 mm) and PRM (394.44mm). The South West monsoon contributed to the major share of annual rainfall (62 %). During 2012-2013 period maximum rainfall observed during October 2012 (555.45 mm) and minimum in January 2013 (6.06 mm). Seasonally, maximum rainfall observed during MN (837.54mm) and minimum in PRM (483.66mm).

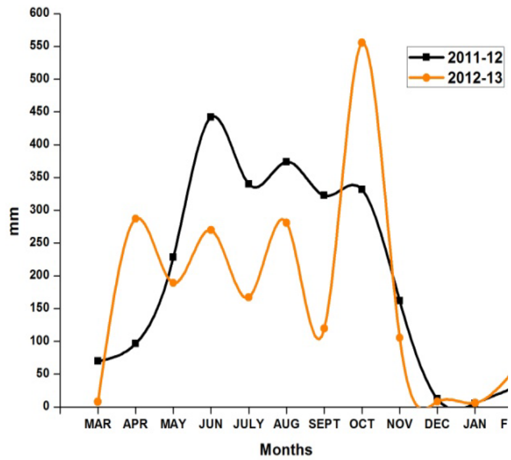

Fig. 3.1 Monthly variation in rainfall in the estuarine area during 2011-2013 period

3.2.1.2 River discharge

The maximum annual net discharge was recorded at Muvattupuzha river (68373 Mm^3), whereas the minimum discharge was recorded at Achencoil river (10825.3 Mm^3) (Annexure 3.12). Monthly river discharge data showed maximum in June 2011 (av. 6428 Mm^3) and minimum in February 2012 (av. 562.6 Mm^3) (Fig. 3.2). Maximum average river discharge was observed during MN (27225.64 ± 1925.1Mm^3) and minimum during PRM (5297.17 ± 1096.6 Mm^3) season.

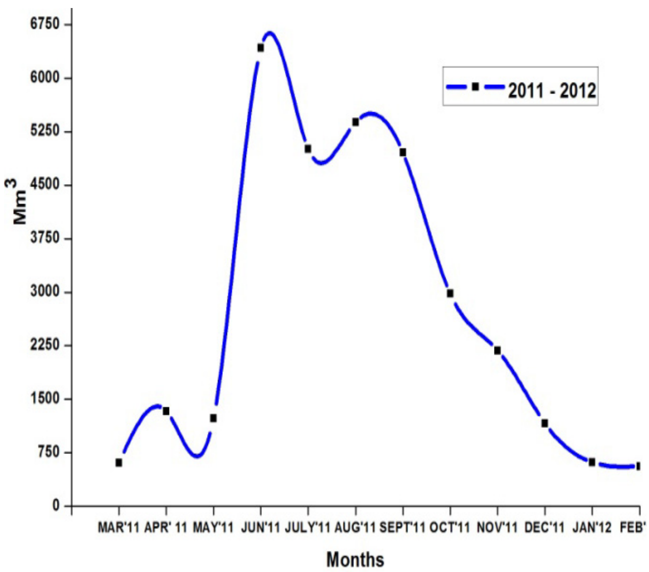

Fig. 3.2 Monthly variation in river discharge in Vembanad estuary during 2011-2012 period

3.2.2 Physical parameters

3.2.2.1 Tide

Field sampling was conducted during high tide time except (Cruise 1, 3, 4, 17, 18 and 24). The type of tides and tidal height were given in annexure 3.1 and 3.3. During 2011-2012 period, the maximum tidal height (96 cm) were observed during Cruise ≠7 (September 2011) whereas during

2012 -2012 period maximum tidal height (98 cm) observed during Cruise ≠ 21 (November 2012).

3.2.2.2 Depth

In 2011-2012 period, the average depth was found to be 4.15 ± 1.94 m and it ranged between 1.45 to 9m. The minimum depth occurs in station 8 during July 2011 and maximum depth in station 10 (Fig. 3.3). Out of the ten sampling stations, minimum spatial average in depth was observed in station 8 (1.95 ± 0.78 m) and maximum in station 10 (7.55 ± 0.6 m). When comparing the month wise variations, the minimum average depth was observed during May 2011 (3.87 ± 2.2 m) and maximum in June 2011 (4.59 ± 2.2 m). During MN, the average depth was maximum as compared to PRM and PM periods. ANOVA showed significant ($P \leq 0.01$) spatial variation of depth.

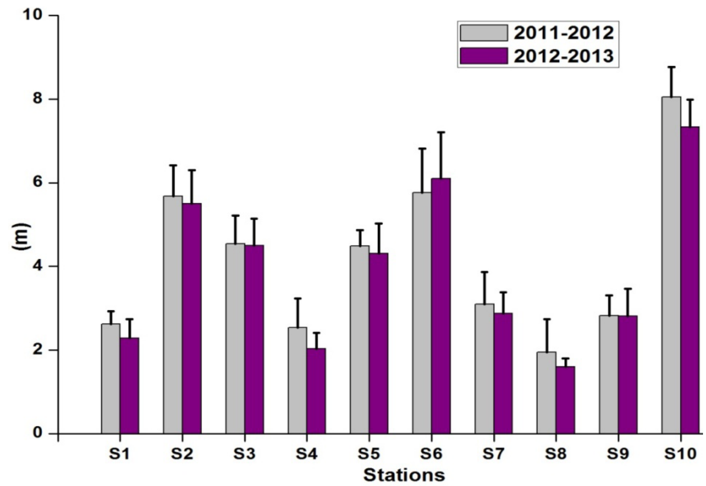

Fig. 3.3 Mean spatial variation of depth in Vembanad estuary during 2011-2013 period

During 2012-2013 period, the depth of the water column varied from 1.4 to 8.5m. The minimum depth of 1.4m was observed in station 8 during June 2012 and maximum of 8.5m was observed in station 10 during April 2012. Seasonally a slight difference of depth was observed. Seasonal variation of average depth was minimum. Here PRM season showing maximum depth followed by PM season. Similar to 2011-2012 period, ANOVA showed significant ($P \leq 0.01$) spatial variation of depth during 2012-2013 period.

3.2.2.3 Water Temperature

During 2011-2012 period, the temperature ranged from 25 – 32.5 °C with an average of 29.29 ± 1.85 °C. Station 2 showed lowest values of 25 °C in December 2011 and station 8 showed highest value of 32.5 °C in September 2011. The estimated minimum spatial average of temperature was observed in station 1 (28.21 ± 1.67 °C) and maximum in station 9 (30.33 ± 1.47 °C) (Fig. 3.4). Minimum temporal variation was observed in July 2011 (27 ± 0.91°C) and maximum in May 2011 (31.45±0.44 °C). When comparing the south and north zone, temperature was higher in north (30.05 ± 0.37 °C) than south (28.78 ± 0.45°C). Premonsoon season showed maximum average of temperature than MN and PM season. ANOVA detected significant spatial and temporal variation of water temperature ($P \leq 0.01$).

During 2012-2013 period, the temperature varied from 29 – 34 °C with an average of 31.35 °C. The minimum temperature was observed in all stations except station 5, 6 and 8. Similar to 2011-2012 period seasonal average was maximum during PRM. Station wise minimum mean temperature was observed at station 4 (30.58 ± 0.99 °C) and maximum at station 9 (32.17 ± 1.59 °C). Temporal analysis showed that maximum mean temperature was observed during May 2012 (32.4±0.66 °C) followed by February 2013 (32.2 ±1.03°C). In the present study, there was not much variations observed in the temperature in south (31.07 ± 0.37 °C) and north zones (31.77 ± 0.28 °C). ANOVA showed significant spatial ($P \leq 0.05$) and temporal ($P \leq 0.01$) variation of water temperature.

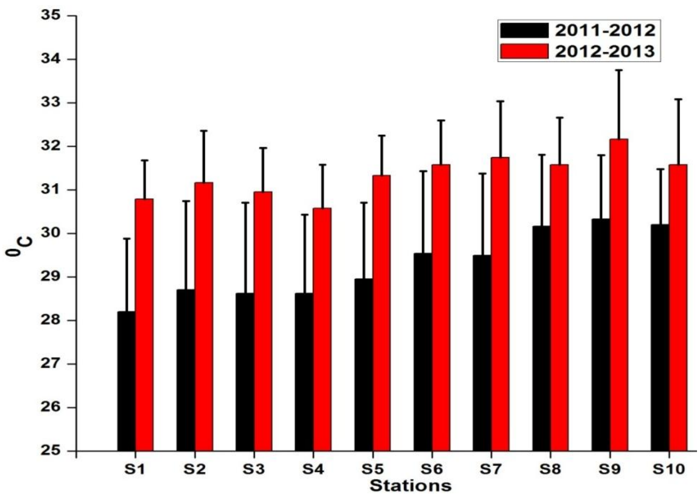

Fig. 3.4 Mean spatial variation of water temperature in Vembanad estuary during 2011-2013 period

3.2.2.4 pH

In the present study the pH value varied between 5.94 to 8.75 and the average value was found to be 7.13 ± 0.52. The lower pH value was observed in station 1 during September 2011 and an alkaline condition was observed in station 10 during February 2012. Station 2 and station 10 showed the minimum (6.98 ± 0.46) and maximum (7.55 ± 0.6) spatial average of pH respectively (Fig. 3.5). Minimum monthly average (6.7 ± 0.25) was observed during July 2011 and maximum during February 2012 (7.97 ± 0.47). In the present investigation, seasonal variation of average pH was displaying maximum value during PM and minimum during MN. When comparing the southern zone and northern zone, the bottom water pH was slightly higher in north (7.23 ± 0.22) than south (7.06 ± 0.06). During 2011 -2012 period, the ANOVA results showed that the seasonal variation of pH were significant ($P \leq 0.01$).

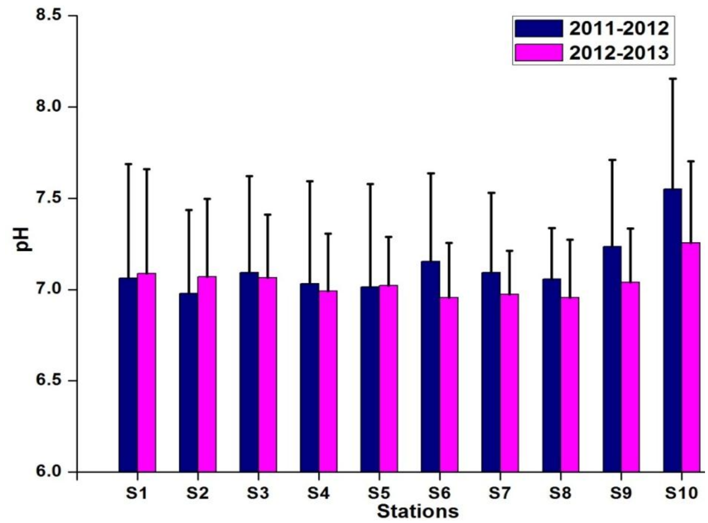

Fig. 3.5 Mean spatial variation of pH in Vembanad estuary during 2011-2013 period

During 2012 – 2013 period, pH varied from a lower value of 6.22 observed at station 1 during October 2012 and a higher value of 7.99 observed at station 10 during February 2013 with an annual average of 7.04 ± 0.36. Spatial distribution of pH showed that station 10 was having a slightly alkaline condition (av. 7.26 ± 0.45) followed by station 1 (av. 7.09 ± 0.57). February 2013 showing the maximum temporal average (7.58 ± 0.24), followed by January 2013 (7.49 ± 0.22). In contrast to 2011-2012 periods, the maximum seasonal average was observed during PRM followed by PM and minimum during MN. There was no variations were observed in pH between the average value of southern and northern zones. During 2012 – 2013 period, the ANOVA showed that the seasonal variation of pH were significant ($P \leq 0.01$).

3.2.2.5 Transparency- light extinction coefficient

During 2011-2012 period, the transparency value ranged between 0.2 – 2m with an average of 0.99 ± 0.31m. During March 2011, station 9 showed the lower transparency and during September 2011, station 4 showed the higher value. Temporal variation of transparency showed that the minimum value was observed during April 2011 (0.75 ± 0.22 m) and maximum during September 2011 (1.23 ± 0.33 m). Spatial variation of transparency showed that the minimum average value was observed in station 10 (0.69 ± 0.17 m) and maximum in station 5 (1.31 ± 0.22 m) (Fig. 3.6).

In the present investigation, the seasonal variation of transparency was minimum. During PRM, the average transparency was minimum and PM exhibited maximum values as compared to MN. Transparency of the water column was more in south (1.05 ± 0.21) than northern zones (0.91 ± 0.2m). Light extinction coefficient (k) value ranges from 0.85 to 8.5 with an average value of 1.9 in 2011-2012 period. The spatial average observed a minimum in station 5 (1.33) and maximum in station 9 (2.56) (Fig. 3.7). The temporal analysis showed that minimum average value of Light extinction coefficient was observed during September 2011 (1.48) and maximum in June 2011 (2.42).

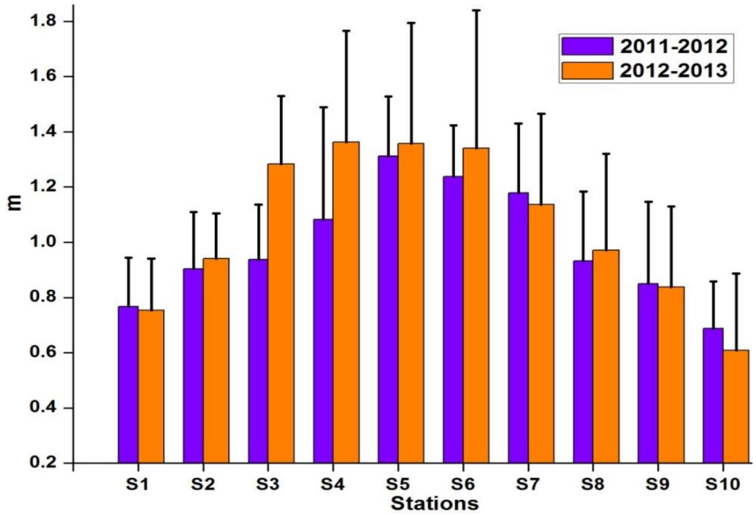

Fig. 3.6 Mean spatial variation of transparency in Vembanad estuary during 2011-2013 period

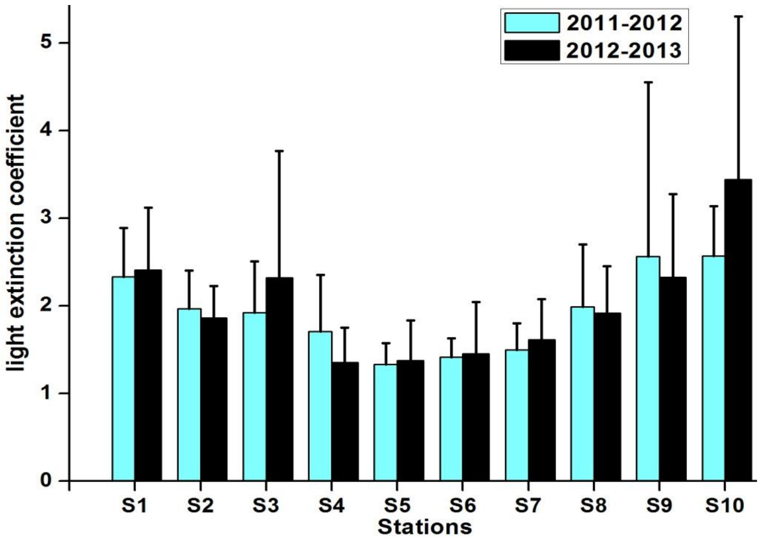

Fig. 3.7 Mean spatial variation of light extinction coefficient in Vembanad estuary during 2011-2013 period

During 2012-2013 period it was observed that, the transparency value varied between 0.2 to 2.25m. The lower value was observed at station 10 during February 2013 and higher value at station 6 during December 2012. Seasonal variation in transparency showed a similar pattern for both the years, in which maximum seasonal average was observed during PM followed by MN and PRM. In the present study, it was observed that southern zone had higher transparency (1.1 ± 0.28 m) than northern zone (0.89 ± 0.22 m). The annual spatial variation showed that station 4 and 5 (1.36 ± 0.4 m) depicted higher transparency and station 10 showing less transparency (0.6 ± 0.28 m). Monthly variation of transparency showed that, the minimum average transparency was observed (0.75 ± 0.23 m) during June 2012 and maximum during January 2013 (1.41 ± 0.46 m). During 2012-2013 period, light extinction coefficient (k) value varied between 0.76 to 8.5 with an annual average of 2. The minimum value was observed in station 6 during December 2012 and maximum in station 10 during February 2013. The spatial maximum average was observed in station 10 (3.4) and minimum in station 4 and 5 (1.4).

3.2.2.6 Conductivity

During 2011-2012 period, conductivity recorded an average value of 5.63 ± 8.96 mS cm^{-1} with a variation from 0.09 mS cm^{-1} to 45.5 mS cm^{-1}. The lower range of conductivity was observed in station 1, 3, and 4 during July 2011 and higher range in station 10 during December 2011. Station 3 and station 10 showed the minimum (2.15 ± 3.65) and maximum (17.05 ± 16.78) spatial average of conductivity respectively (Fig. 3.8). The minimum average conductivity was observed during June 2011 (0.22 ± 0.04 mS cm^{-1}) and maximum value during May 2011 (16.83 ± 4.86 mS cm^{-1}). In the present study, seasonal variation of conductivity was very distinct. The maximum average of conductivity was observed during PRM and minimum value in MN. In between the southern and northern zone conductivity value showed a clear variation, in which southern zone always indicated a lower value. In the southern and northern zone the average value viewed as 2.81 ± 0.65 mS cm^{-1} and 9.85 ± 5.32 mS cm^{-1} respectively. Similar to bottom water temperature, ANOVA of conductivity showed significant spatial and temporal variation ($P \leq 0.01$).

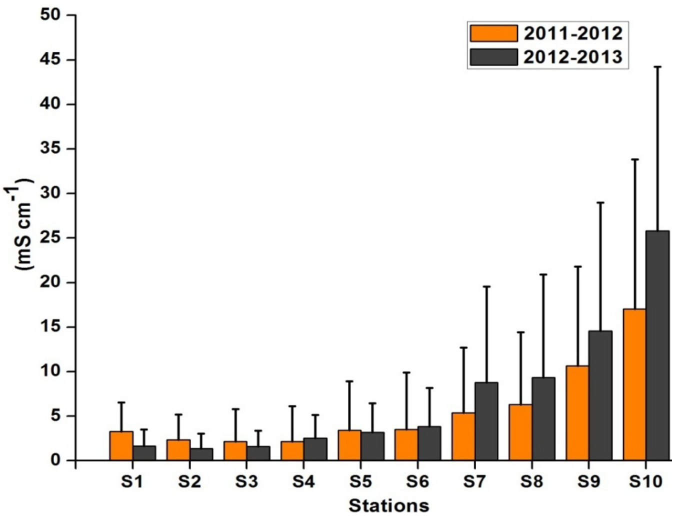

Fig. 3.8 Mean spatial variation of conductivity in Vembanad estuary during 2011-2013 period

During 2012-2013 period, the conductivity value ranged from 0.07 to 47.9 mS cm^{-1} with minimum at station 5 for the month of August 2012 and maximum at station 10 in September 2012. The annual average value observed was 7.27 ± 1.16 mS cm^{-1}. Station wise mean value of conductivity recorded highest in station 10 (25.81 ± 1.84 mS cm^{-1}) followed by station 9 (14.58 ± 1.44 mS cm^{-1}). Similarly the lowest mean spatial value observed was in station 2 (1.35 ± 1.69 mS cm^{-1}) followed by station 1 (1.35 ± 1.69 mS cm^{-1}). Temporally highest mean value was observed during February 2013 (17.11 ± 1.47 mS cm^{-1}), followed by January 2013 (16.34 ± 1.43 mS cm^{-1}). Similar to 2011-2012 period, conductivity followed a similar pattern of variation during 2012-2013 period, in which the maximum value was observed during PRM followed by PM and MN. When comparing the south and north sectors, the average conductivity value was maximum in north (14.63 ± 7.9 mS cm^{-1}) than south (2.36 ± 0.99 mS cm^{-1}). ANOVA showed that spatial and temporal variation of conductivity was significant ($P \leq 0.01$).

3.2.2.7 Total Dissolved Solids (TDS)

During 2011-2012 period, TDS value ranged from 0.1 to 22.4 ppm with an average value of 3.28 ± 4.47 ppm. The minimum value was observed during June and August 2011 in station 5 and June 2011 in station 8 (Fig. 3.9). At the same time, the maximum value was observed in station 10 during January 2011. Spatial mean value of TDS recorded the highest in station 10 (10.39 ± 8.39 ppm) and lowest in station 3 (0.64 ± 0.73 ppm). Highest monthly average of TDS was present in May 2011 (5.25 ± 2.10 ppm) and lowest in June 2011 (0.46 ± 0.63 ppm). Seasonal average of TDS shows that PM is having a maximum value and MN having a minimum value. ANOVA test indicated that spatial and temporal variation of TDS values were significant ($P \leq 0.01$).

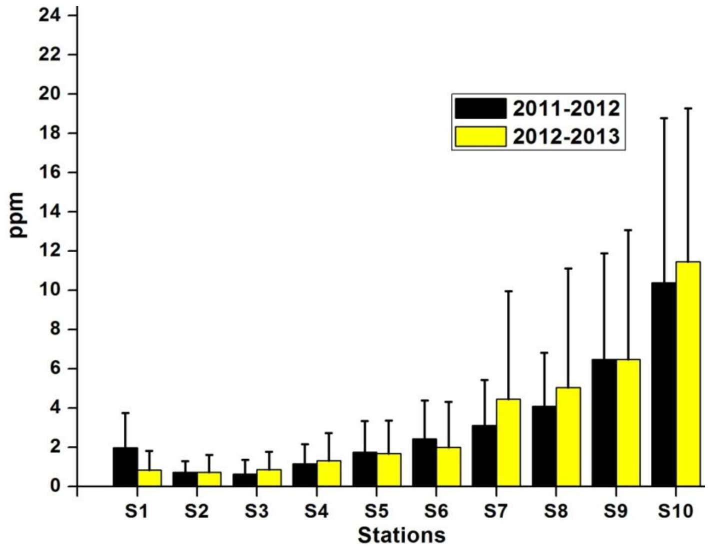

Fig. 3.9 Mean spatial variation of total dissolved solids in Vembanad estuary during 2011-2013 period

During 2012-2013 period, the annual average of TDS observed was 3.49 ± 5.28 ppm. It ranged between 0.09 ppm at station 2 in October 2012 to 23.1ppm at station 10 during March 2012. The spatial analysis displayed that as compared to other stations, station 2 was having low value (0.72 ± 0.88 ppm) and station 10, having higher value (11.46 ± 7.82 ppm) compared to the other stations. Seasonal analysis of TDS shows that highest value was observed during PRM and lowest during MN. On a temporal scale, it was noted that, the lowest TDS was observed in August 2012 (0.29 ± 0.48 ppm) and second lowest during July 2012 (0.44 ± 0.4 ppm). Similar to turbidity, during both the years average value of TDS was found to be maximum in northern zone than southern zone. ANOVA showed that spatial and temporal variation of TDS was significant ($P \leq 0.01$).

3.2.2.8 Turbidity

During 2011-2012 period, turbidity value recorded an average of 1.93 ± 1.01 NTU, with a minimum of 0.1 NTU at station 5 for the month of February 2012 and maximum of 5.1NTU at station 10 in February 2012 (Fig. 3.10). Spatial analysis of turbidity showed highest average value at station 10 (3.03 ± 1.47 NTU) and lowest in station 5 (1.38 ± 0.9 NTU). The monthly average value was minimum during November 2011 (0.92 ± 0.64 NTU) and maximum during April 2011 (2.64±1.02). Seasonal analysis of turbidity showed that highest value was observed during MN followed by PM and PRM. During the study period, northern zone was showing a more turbid condition (2.19 ± 0.65 NTU) than southern zones (1.74 ± 0.39 NTU). The ANOVA showed that turbidity value had a significant variation both spatially ($P \leq 0.01$) and temporally ($P \leq 0.01$).

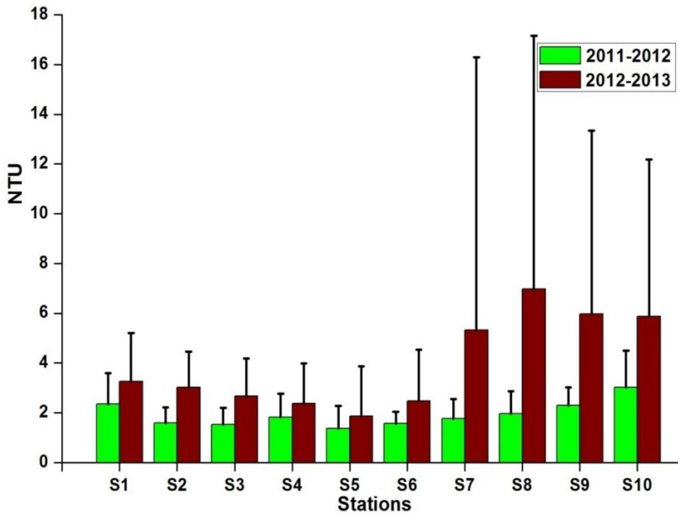

Fig. 3.10 Mean spatial variation of turbidity in Vembanad estuary during 2011-2013 period

During 2012-2013 period, turbidity value ranged between 0.1 to 40 NTU with an average of 3.77 ± 5.43 NTU. The lower value was observed at station 1 (July 2012) and station 3(January 2013) and the higher value observed at station 8 during March 2012. Spatial variation of turbidity showed that, the minimum average value was observed in station 5 (1.5 ± 7.75 NTU) and maximum in station 8 (7.99 ± 7.89 NTU). The monthly average value was minimum during January 2013 (0.49 ± 0.4) and maximum during March 2012 (12.17 ± 12.19 NTU). Seasonal variation of turbidity showed that maximum average value was observed during PRM followed by MN and PM. Similar to 2011-2012 period, the northern zone exhibited more turbidity than southern zone. The ANOVA showed that turbidity had a significant temporal variation ($P \leq 0.01$).

3.2.2.9 Salinity

Wide variations in salinity were observed in the estuary. During 2011-2012 period, the salinity ranged from 0 to 31ppt with minimum in most of the stations during monsoon months and maximum at station 10 in November 2011 (Fig. 3.11). The average salinity during 2011-2012 period was 3.23 ± 5.87 ppt. Station wise mean value of salinity recorded the highest in station 10 (11.32 ± 1.18 ppt) and lowest in station 2 (0.53 ± 0.5 ppt). Maximum average salinity was observed during December 2011 (9.2 ± 0.64 ppt) and nil values during August and September 2011. During the present study, seasonal variation of salinity was very distinct, where maximum average salinity was observed during PM followed by PRM and MN. With the onset of monsoon, a sudden decrease in salinity was observed. When comparing the southern zone and northern zone, the bottom water salinity was higher in northern zone. The average salinity exhibited a value of 1.3 ± 0.5 ppt and 6.11 ± 3.8 ppt in the southern and northern zone respectively. ANOVA results of salinity showed that the variations were significant at 1% level between stations ($P \leq 0.01$) and between seasons ($P \leq 0.01$).

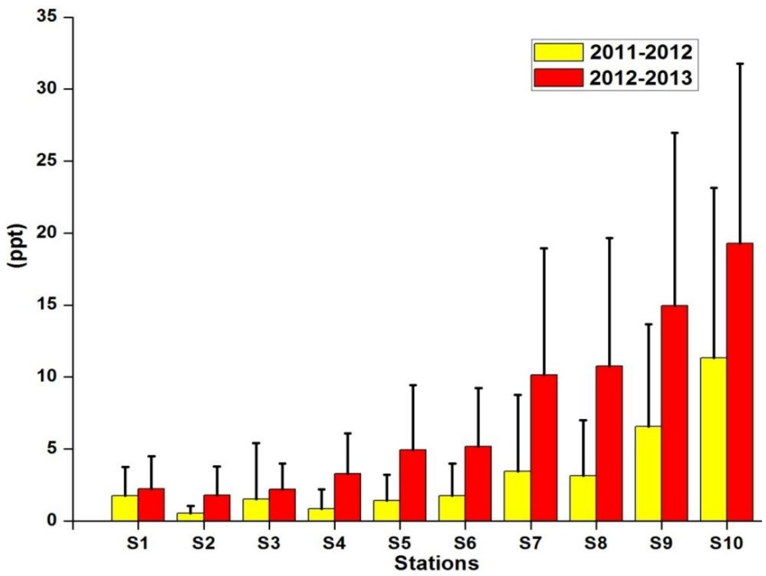

Fig. 3.11 Mean spatial variation of salinity in Vembanad estuary during 2011-2013 period

In 2012-2013 period, the annual average salinity was 7.48 ± 8.99 ppt and the value ranged from 0.16 to 33.1ppt. The minimum value was observed at station 5 during July 2012 and August 2012 and the maximum value at station 10 during February 2013. The minimum spatial average was observed in station 2 (1.8 ± 1.97 ppt) and maximum in station 10 (19.28 ± 1.25 ppt). In the case of temporal analysis, the month wise average was maximum during February 2013 (15.72 ± 9.46) and minimum during July 2012 (0.49 ± 0.59 ppt). In contrast with 2011-2012 period, seasonally salinity showed a bimodal pattern during 2012-2013 period. Maximum salinity was observed during PRM and decreased during MN and again increased during PM. When comparing the southern zone and northern zone, the bottom water salinity was higher in northern zone (13.78 ± 4.24 ppt) than southern (3.28 ± 1.47 ppt) zone. ANOVA of salinity showed significant variation between stations ($P \leq 0.01$) and between seasons ($P \leq 0.01$).

3.2.3 Chemical parameters

3.2.3.1 Free carbon dioxide (CO_2)

During 2011-2012 period, the annual average free CO_2 was 6.4 ± 4.43 mg L^{-1} that ranged from 1 to 19 mg L^{-1}. The minimum value was observed at station 1 during March, April, May and August 2011 and maximum value at station 10 during October 2011 (Fig. 3.12). Station wise variation of CO_2 showed that lowest value in station 6 (4.92 ± 3.32 mg L^{-1}) and highest in station 10 (8.17 ± 5.24 mg L^{-1}). Monthly variation of free CO_2 showed that April 2011, having minimum (1.4 ± 0.52 mg L-1) and October 2011 having maximum (13.6 ± 3.47 mg L^{-1}) value. In the present study, a slight variation of free CO_2 was observed between southern and northern zones. A slight increase of average free CO_2 was observed in south (6.5 ± 1.18 mg L^{-1}) than north (6.23 ±1.3 mg L^{-1}) zone. Seasonal analysis showed that PM season was having maximum average CO_2 than MN and PRM. ANOVA of free CO_2 was significant at 1% level between stations ($P \leq 0.01$) and between seasons ($P \leq 0.01$).

During 2012-2013 period, the free CO_2 value recorded a minimum value of 0.5 to maximum of 16 mg L^{-1} with an average of 2.4 ± .87 mg L^{-1}. The lowest free CO_2 value was observed in station 5 and 6 and the highest value in station 1 during February 2012. Spatial analysis showed that station 1 was having highest CO_2 value (3.96 ± 4.11 mg L^{-1}) and lowest in station 6 (1.29 ± 0.99 mg L^{-1}). Monthly variation of CO_2 showed that the minimum monthly average was observed during October 2012 (1.2 ± 0.42 mg L^{-1}) and maximum during August 2012 (3.9 ± 1.79 mg L^{-1}). In contrast with 2011-2012 period, the average bottom water CO_2 was higher in northern zone (2.35 ± 0.89 mg L^{-1}) than in southern zone (2.48 ± 0.66 mg L^{-1}). Seasonal analysis showed that PRM season was having maximum average free CO_2 than MN and PM. During the study period, the ANOVA showed that the temporal variation of free CO_2 were significant ($P \leq 0.01$).

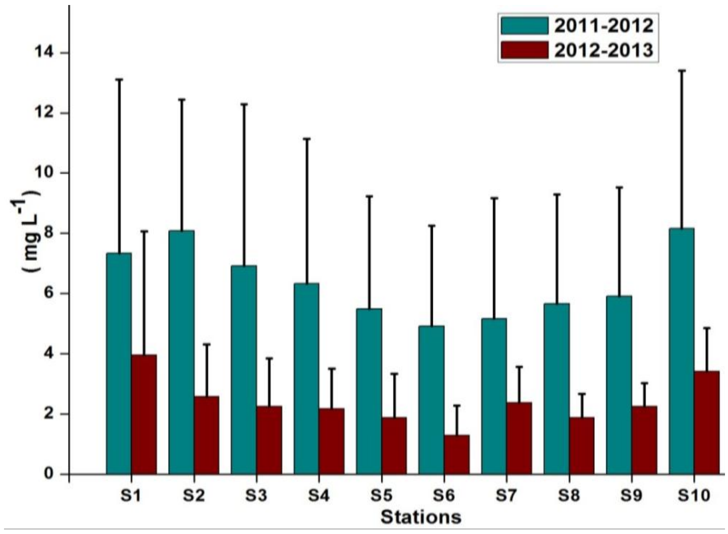

Fig. 3.12 Mean spatial variation of free carbon dioxide in Vembanad estuary during 2011-2013 period

3.2.3.2 Total alkalinity

During 2011-2012 period, alkalinity values ranged from 16 to 220 mg L^{-1} with an average value of 45.57 ± 3.65 mg L^{-1}. Station wise mean value of alkalinity recorded the highest in station 10 (76 ± 4.97 mg L^{-1}) and lowest in station 2 (33.67 ± 3.09 mg L^{-1}) (Fig. 3.13). Seasonal average of alkalinity showed that PRM was having maximum value and MN having minimum value. ANOVA of alkalinity was significant at 1% level between seasons ($P \leq 0.01$).

During 2012 – 2013 period, the annual average of alkalinity was observed to be 28.77 ± 1.48 mg L^{-1}. It ranged between 12 mg L^{-1} at station 2, station 3 and station 5 to 120 mg L^{-1} at station 10 during February 2013. The spatial analysis displayed that, as compared to other stations, station 5 was having low values (19. 67 ± 4.66 mg L^{-1}) and station 10 having higher value (54.67 ± 2.89 mg L^{-1}) than other stations. Seasonal analysis of alkalinity shows that highest value was observed during PRM and lowest during PM. During both the years, average value of alkalinity was found to be maximum in northern zone than southern zone. The ANOVA analysis of variations of alkalinity was significant at 1% level between stations ($P \leq 0.01$) and between seasons ($P \leq 0.01$).

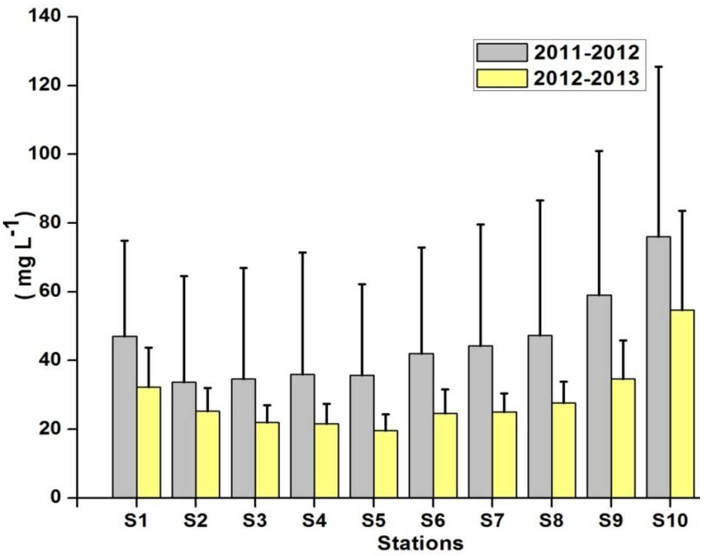

Fig. 3.13 Mean spatial variation of alkalinity in Vembanad estuary during 2011-2013 period

3.2.3.3 Dissolved Oxygen (DO)

During 2011-2012 period, the annual average of DO was 7.53 ± 0.93 mg L^{-1} and it ranged between 5.69 to 9.76 mg L^{-1}. The minimum value was observed in station 1 (July 2011), 2 (April 2011), 6 (January 2012), 7 (January 2012) and 9 (December 2011) and maximum value was observed at station 4 during May 2011 (Fig. 3.14). Station wise variation of DO showed that lowest value was in station 2 (7.03 ± 1.09 mg L^{-1}) and highest in station 5 (7.92 ± 0.5 mg L^{-1}). Monthly variation of dissolved oxygen showed that November 2011 was having minimum DO value (6.5 ± 0.53 mg L^{-1}) and May 2011 having maximum (8.46 ± 0.78 mg L^{-1}). In the present study, a slight variation of DO was observed between southern and northern zone. A slight increase of average dissolved oxygen was observed in north zone (7.66 ± 0.22 mg L^{-1}) than south (7.35 ± 0.38 mg L^{-1}). Seasonal analysis showed that PRM was having maximum average DO than MN and PM periods.

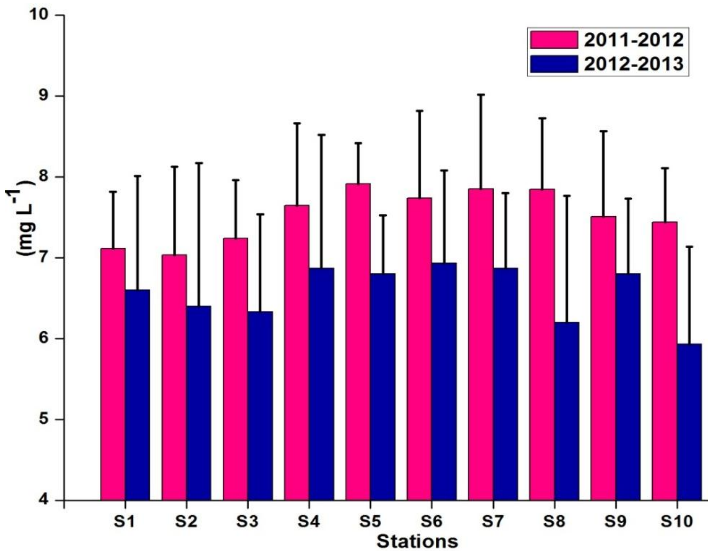

Fig. 3.14 Mean spatial variation of dissolved oxygen in Vembanad estuary during 2011-2013 period

During 2012-2013 period, the DO value recorded a minimum value of 3.2 to maximum of 11.2 mg L^{-1} with an average of 6.57 ± 1.29 mg L^{-1}. The lowest DO value was observed in station 2 during March 2012 and November 2012, whereas the highest value was observed at station 4 during June 2012. Spatial analysis showed that station 6 was having highest DO value (6.93 ± 1.15 mg L^{-1}) and lowest in Station 10 (5.93 ± 1.2 mg L^{-1}). Monthly variation of dissolved oxygen showed that the minimum monthly average was observed during November 2012 (5.36 ± 1.0 mg L^{-1}) and maximum during May 2012 (8.8 ± 0.99 mg L^{-1}). When comparing the southern and northern zones, the bottom water was showing higher value in southern zone (6.66 ± 0.25 mg L^{-1}) than in northern zone (6.45 ± 0.46 mg L^{-1}).

Seasonal analysis showed that MN was having maximum average DO than PRM and PM. During the study period, the ANOVA showed that the seasonal variation of DO were significant ($P \leq 0.01$).

3.2.3.4 Biological Oxygen Demand (BOD)

In the present investigation, the BOD value varied between 0.7 to 7.32 mg L^{-1} with an annual average of 2.94 ± 1.39 mg L^{-1}. The lowest value was recorded in March 2011 at station 7 and highest value in May 2011 at station 4. Month wise mean value was recorded lowest in August 2011 (1.47 ± 0.52 mg L^{-1}) and highest in May 2011 (4.72 ± 1.66 mg L^{-1}) (Fig. 3.15). Station 1 showed the highest average value (3.39 ± 1.14 mg L^{-1}) followed by station 8 (3.31 ± 1.6 mg L^{-1}) and lowest in station 6 (2.7 ± 1.49 mg L^{-1}). Seasonally it demonstrates that PRM was showing maximum value followed by PM and the minimum in MN. During 2011-2012 period there was no observed variation in the average BOD value in north and south of TMB.

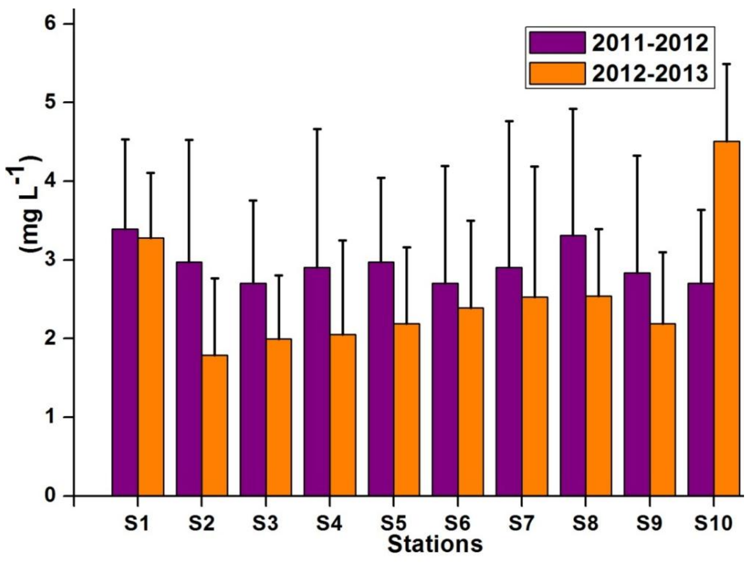

Fig. 3.15 Mean spatial variation of biological oxygen demand in Vembanad estuary during 2011-2013 period

During 2012 – 2013 period, BOD showed an annual average of 2.55 ± 1.27 mg L^{-1} and it ranged from a minimum of 0.68 mg L^{-1} during May 2012 in station 9 and the maximum value of 6.36 mg L^{-1} was observed at station 7 during September 2012. Spatial analysis showed that station 10 having highest BOD value followed by station 1. Seasonal study showed that maximum average concentration was observed during MN and minimum during PM. Similar to 2011-2012 period, during 2012-2013 period the BOD was slightly higher in north zone (2.94 ±1.06 mg L^{-1}) than south zone (2.29 ± 0.53 mg L^{-1}). ANOVA of BOD showed significant variation between stations ($P \leq 0.01$) and between seasons ($P \leq 0.05$).

3.2.4 Nutrients

3.2.4.1 Silicate-silicon

A wide range of variation in SiO_4-Si was observed during the study period. During 2011 – 2012 period, the highest value of 105.74 µmol L^{-1} was observed in September 2011 at station 1 and lowest value of 2.13 was recorded from station 6 during May 2011 with an annual average of 19.32 ± 1.52 µmol L^{-1} (Fig. 3.16). The highest average concentration of SiO_4-Si was observed in September 2011 (45.32 ± 2.88 µmol L^{-1}) followed by March 2011(29.48 ± 9.1 µmol L^{-1}), whereas lowest concentration was observed in May 2011 (3.19 ± 1.5 µmol L^{-1}). Spatial average of maximum was observed in station 1 (25.13 ± 2.83 µmol L^{-1}) followed by station 10 (21.13 ±1.37 µmol L^{-1}). Seasonal study illustrated that maximum average concentration was observed during MN followed by PRM and PM periods. On comparing the southern and northern zones, the SiO_4-Si varied with lower values in northern (18.59 ± 2.25 µmol L^{-1}) than southern zones (19.65 ± 3.24 µmol L^{-1}).

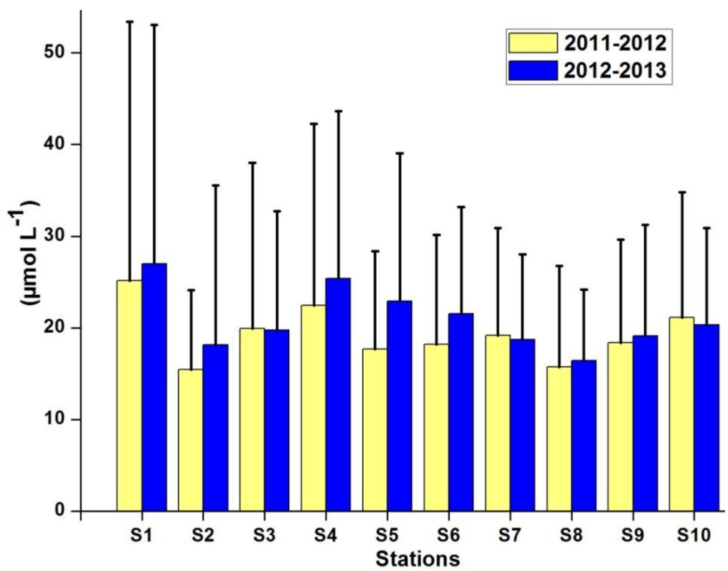

Fig. 3.16 Mean spatial variation of silicate-silicon in Vembanad estuary during 2011-2013 period

During 2012 – 2013 period, the SiO_4-Si concentration varied from 2.51 to 90.4 µmol L^{-1}µmol/l and recorded an average of 20.91± 1.49 µmol L^{-1}. The minimum value was recorded during May 2012 at station 9 and maximum during November 2012 at station 1. Station wise variation of the SiO_4-Si concentration showed that maximum average concentration was observed in station 1 (27 ± 2.6 µmol L^{-1}) and minimum in station 8 (16.42 ± 7.77 µmol L^{-1}). A clear monthly variation of SiO_4-Si was observed during the study period. Maximum monthly average was observed during April 2012 (39.69 ± 9.97 µmol L^{-1}) and followed by February 2013 (31.92 ± 1.74 µmol L^{-1}). Similarly May 2012, recorded the minimum average (4.11 ± 0.89 µmol L^{-1}), followed by June 2012 (8.55 ± 1.97 µmol L^{-1}). In the southern region, the SiO_4-Si concentration was always higher during both the years. Similar to 2011-2012 period, SiO_4-Si varied with lower values in northern (18.63 ± 1.62µmol µmol L^{-1}) than southern zones (22.44 ± 3.36 µmol L^{-1}) during 2012-2013 period.

3.2.4.2 Phosphate-phosphorus

In the present study PO_4-P concentration ranged from 0.13 to 4.63 µmol L^{-1} with an average of 1.17 ± 0.78 µmol L^{-1}. The minimum concentration was observed in station 2 (July 2011), station 5 (February 2012) and station 9 (July 2011) and maximum concentration was observed in station 10 during March 2011 (Fig. 3.17). Seasonal observation showed that higher concentration was observed during PM and lower concentration during MN. The spatial analysis revealed that station10 (2.28 ± 1.12 µmol L^{-1}) and station 4 (0.79 ± 0.36 µmol L^{-1}) was having maximum and minimum average concentration respectively. Similarly temporal analysis showed minimum average concentration during July 2011 (0.36 ± 0.23 µmol L^{-1}) and maximum during November 2011 (1.75 ± 0.85 µmol L^{-1}). During the study period the PO_4-P concentration was higher in northern zone. Northern sector was having an average concentration of 1.43 ± 0.57 and that in the southern sector, it was 1 ± 0.24 µmol L^{-1}. ANOVA analysis revealed that the spatial and temporal variation of PO_4-P was varied significantly ($P \leq 0.01$).

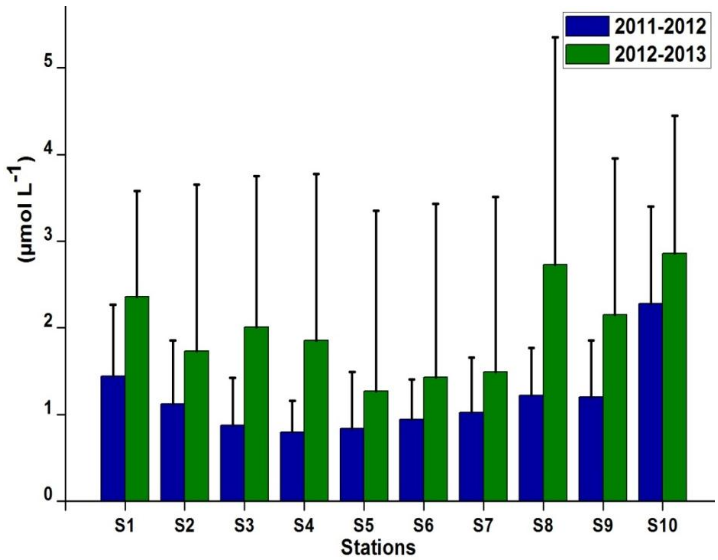

Fig. 3.17 Mean spatial variation of phosphate – phosphorous in Vembanad estuary during 2011-2013 period

In 2012 – 2013 period, the PO_4-P concentration varied from 0.07 to 8.58 µmol L^{-1}, having an average of 1.99 ± 1.92 µmol L^{-1}. The minimum concentration was recorded at station 10 during February 2013 and the maximum value at station 8 during March 2012. Spatial distribution pattern showed that, station 10 (2.86 ± 1.59 µmol L^{-1}) recorded increasing PO_4-P followed by station 8 (2.73 ± 2.63 µmol L^{-1}) and station 5 recording lower value (1.27 ± 2.08 µmol L^{-1}). Seasonal study showed that higher value of PO_4-P was detected during PM followed by MN and PRM. Both the years the phosphate concentration was higher in northern than the southern zone.

3.2.4.3 Nitrite - Nitrogen

During 2011-2012 period, the average NO_2-N concentration was 0.29 ± 0.57 µmol L^{-1}. It varied from a minimum of 0.03 µmol L^{-1} to a maximum of 5.53 µmol L^{-1}. The minimum value was observed in station 2 (November 2011), station 3 (November 2011) and station 4 (June 2011) and maximum concentration was observed in station 4 during September 2011 (Fig. 3.18). In the present investigation seasonal variation of NO_2-N concentration was very distinct. Monsoon season showed the maximum value followed by PRM and PM. Station wise mean value recorded the highest in Station 4 (0.59 ± 1.56 µmol L^{-1}) followed by Station 10 (0.56 ± 0.71 µmol L^{-1}) and lowest in Station 5 (0.13 ± 0.09 µmol L^{-1}). Month wise mean value recorded the highest in September 2011 (0.91 ± 1.63 µmol L^{-1}) and lowest in March 2011 (0.57 ± 0.9µmol L^{-1}). The annual average concentration of NO_2-N e was slightly lower in south (0.24 ± 0.18 µmol L^{-1}) than in north (0.35 ± 0.16µmol L^{-1}) of the barrage.

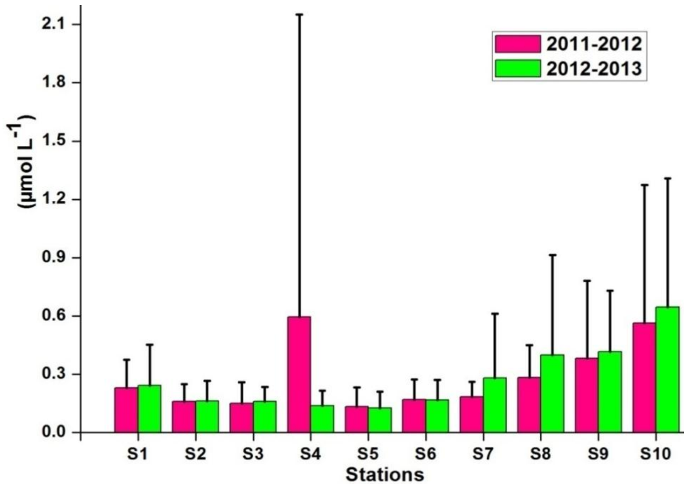

Fig. 3.18 Mean spatial variation of nitrite –nitrogen in Vembanad estuary during 2011-2013 period

During 2012-2013 period, the NO_2-N concentration varied from 0.04 µmol L^{-1} to 2.04 µmol L^{-1} with an average of 0.27 ± 0.34 µmol L^{-1}. The minimum value was recorded in station 2 (July 2012), station 4 (November 2012) and station 5 (December 2012) and the maximum value recorded during February in station 10. Seasonally it revealed that maximum concentration was observed during PRM and minimum during MN. Spatial distribution of NO_2-N concentration depicted that, station 10 was having maximum average concentration (0.59 ± 1.56 µmol L^{-1}) and station 5 the minimum value (0.13 ± 0.09 µmol L^{-1}). On comparing the southern and northern zones, the NO_2-N varied widely with lower values (0.17 ± 0.04 µmol L^{-1}) in southern than northern zones (0.44 ± 0.15 µmol L^{-1}). ANOVA analysis showed that the spatial and temporal variation of NO_2-N concentration were significant ($P \leq 0.01$).

3.2.4.4 Nitrate - Nitrogen

In the present investigation, the NO_3-N concentration varied between 0.11 to 4.21 µmol L^{-1} with an annual average of 1.51 ± 0.89 µmol L^{-1}. The lowest value of 0.11 µmol L^{-1} was recorded in January 2011 in station 8 and highest of 4.21 µmol L^{-1} in December 2011 in station 3 (Fig. 3.19). Monthly the mean value recorded the lowest in January 2012 (0.51 ± 0.34 µmol L^{-1}) and highest in June 2011 (2.52 ± 0.56 µmol L^{-1}) and December 2011 (2.29 ± 1.03 µmol L^{-1}). Station 10 was showing the highest average value (2.03 ± 0.95 µmol L^{-1}) followed by station 9 (1.86 ± 0.9 µmol L^{-1}) and lowest in station 5 (0.94 ± 0.66). Seasonal study demonstrates that MN exhibited maximum concentration followed by PM and the minimum concentration in PRM. During 2011-2012 period the NO_3-N concentration was slightly higher in north (1.61 ± 0.39 µmol L^{-1}) than south (1.45 ± 0.29 µmol L^{-1}) zone. The ANOVA result of NO_3-N was significant at 1% level between seasons ($P \leq 0.01$).

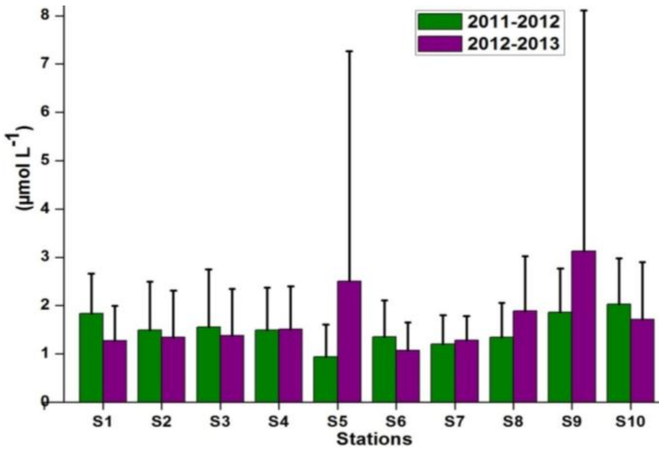

Fig. 3.19 Mean spatial variation of nitrate – nitrogen in Vembanad estuary during 2011-2013 period

During 2012 – 2013 period, NO_3-N showed an annual average of 1.71 ± 2.32 µmol L^{-1} and it ranged from a minimum of 0.06 µmol L^{-1} at station 1 during April 2012 and the maximum value at station 9 during June 2012 (18.59 µmol L^{-1}). Seasonal study showed that, maximum average concentration was observed during MN and minimum during PM. Spatial average value showed that station 9 was having the highest average value and lowest in station 6. Similar to 2011-2012 period, during 2012-2013 period the NO_3-N concentration was slightly higher in northern (2 ± 0.79 µmol L^{-1}) than the southern zone (1.5 ± 0.5 µmol L^{-1}). ANOVA analysis showed that temporal variation of NO_3-N was significant at 5 % level ($P \leq 0.05$).

3.2.4.5 Ammonia – Nitrogen

The concentration of NH_4-N ranged between 0.06 and 37.31 µmol L^{-1} and the estimated average concentration was found to be 7.29 ± 8.41 µmol L^{-1}. The spatial average of NH_4-N displayed a lowest value in station 5 (5.24 ± 6.55 µmol L^{-1}) and highest in station 1 (12.28 ± 10.74 µmol L^{-1}). Temporal variation exhibited maximum value in October 2011 (25.02 ± 6.19 µmol L^{-1}) and minimum during January 2012 (1.23 ± 1.77 µmol L^{-1}) (Fig. 3.20). During PRM, the average NH_4-N concentration was minimum as compared to MN. Post monsoon season was showing the highest average seasonal value. During 2011-2012 period, the NH_4-N concentration was slightly higher in southern (7.38 ± 3.23µmol L^{-1}) than northern zone (7.16 ± 1.19µmol L^{-1}). The ANOVA result of NH_4-N was significant at 1% level between seasons ($P \leq 0.01$).

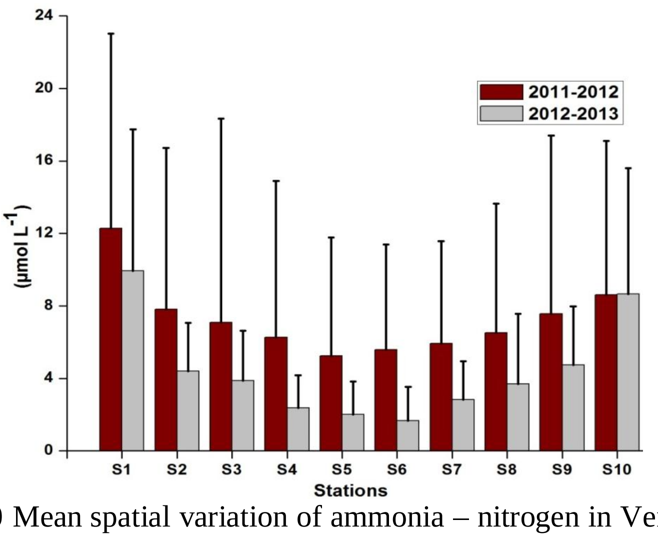

Fig. 3.20 Mean spatial variation of ammonia – nitrogen in Vembanad estuary during 2011-2013 period

During 2012 – 2013 period, the NH_4-N recorded an annual average of 4.42 ± 4.71 µmol L^{-1}. The minimum range of NH_4-N value (0.06 µmol L^{-1}) was observed at station 6 during November 2012 and the highest value (25.33 µmol L^{-1}) recorded at station 10 during November 2012. The spatial analysis showed that the station 1 recorded the maximum average value of NH_4-N (9.93 ± 7.82 µmol L^{-1}) followed by station 10 (8.66 ± 6.94 µmol L^{-1}). Temporal analysis showed that the maximum monthly average was observed during April 2012 (7.5 ± 4.83 µmol L^{-1}) followed by March 2012 (6.41 ± 4.76 µmol L^{-1}) and minimum monthly average was observed during July 2012 (1.73 ± 0.64 µmol L^{-1}). ANOVA result of NH_4-N showed that the variations were significant at 1% level between stations ($P \leq 0.01$) and seasons ($P \leq 0.01$).

3.2.4.6 Total dissolved inorganic nitrogen

Total dissolved inorganic nitrogen (DIN) recorded an average value of 9.08 ± 8.73 µmol L^{-1} with a variation from 0.54 to 40.06 µmol L^{-1}. The minimum value in station 6 in March 2011 and maximum value was observed at station 1 during December 2011(Fig. 3.21). Seasonal variation of total nitrogen was clearly distinct during the study period. The seasonal average value showed that maximum concentration was observed during PM followed by MN and PRM. Temporally total nitrogen showed higher average value during October 2011(26.57 ± 6.58 µmol L^{-1}) and lower during January 2012 (1.98 ± 1.92 µmol L^{-1}). In the present study northern sector showed a slight higher value (9.06 ± 2.83 µmol L^{-1}) of DIN than southern sector (9.1 ± 1.74 µmol L^{-1}). The ANOVA analysis showed that seasonal variation of DIN was significant at 1% ($P \leq 0.01$).

During 2012 – 2013 period, total DIN varied from 1.21 to 26.92 µmol L^{-1} having an average of 6.39 ± 5.32 µmol L^{-1}. The lowest range of DIN was observed during December 2012 in station 2 and the highest range in station 10 during November 2012. Spatial analysis showed that station 1 had the maximum average concentration (11.45 ± 7.82 µmol L^{-1}) followed by station 10 (11.02 ± 6.43 µmol L^{-1}). The lowest average minimum concentration was observed in station 6 (2.92 ± 1.87 µmol L^{-1}). Monthly trends indicated, that a maximum mean value was observed during June 2012 (10.05 ± 7.89 µmol L^{-1}), followed by May 2011 (9.54 ± 5.44 µmol L^{-1}). Similar to 2011-2012 period, northern sector (7.42 ± 2.88) always showed a higher value than southern sector (5.72 ± 2.99 µmol L^{-1}). The ANOVA result showed that the spatial and seasonal variation of DIN concentration were significant ($P \leq 0.01$).

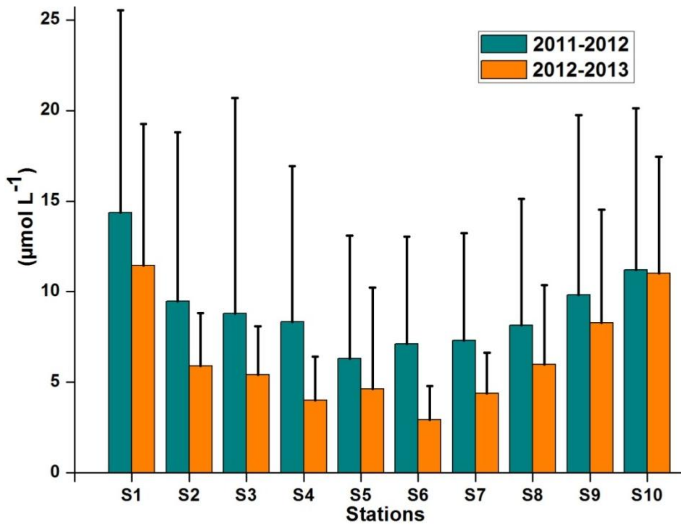

Fig. 3.21 Mean spatial variation of total inorganic nitrogen in Vembanad estuary during 2011-2013 period

3.2.5 Trophic condition

3.2.5.1 N: P Ratio

The mean N: P ratio was below the normal Redfield ratio (16:1) that varied from 5.49 ± 3.95 in station 10 to 15.38 ± 22.39 in station 2 with an average value of 10.96 ± 13.57 for 2011-2012 period. Postmonsoon season showed maximum N : P ratio (15.61 ± 19.33) followed by MN (11.42 ± 10.23) and PRM (5.59 ± 4.79).

For 2012-2013 period, N : P ratio varied from 3.71 ± 2.53 (station 4) to 15.52 ± 34.58 (station 10) with an average N:P ratio of 6.47 ± 4.3. Seasonal analysis showed that, N : P ratio was higher in PRM (10.62 ± 19.46) and lower in PM (3.99 ±3.65).

3.2.5.2 Chlorophyll *a*

During 2011-2012 period chlorophyll *a* value ranged between 0.01 at station 3 in January 2012 to 16.16 mg m^{-3} at station 6 in April 2011. The annual average of chlorophyll *a* during 2011 - 2012 period was 7.72 ± 1.27 mg m^{-3}. Spatial distribution showed a highest average value of 17.84 ± 3.78 mg m^{-3} in station 6 and lower value of 4.61 ± 7.12 mg m^{-3}in station 10. Temporal variation showed that, maximum average value was found in November 2011 (18.55 ± 4.16 mg m^{-3}) and minimum in January 2012 (1.1 ± 0.95 mg m^{-3}) (Fig. 3.22). During 2011-2012 period, the seasonal dynamics of chlorophyll *a* was distinct. The seasonal average was maximum during PRM followed by PM and MN. The chlorophyll- *a* content was higher in southern zone (7.35 ± 0.56 mg m^{-3}) than northern zone (6.63 ± 1.02 mg m^{-3}).

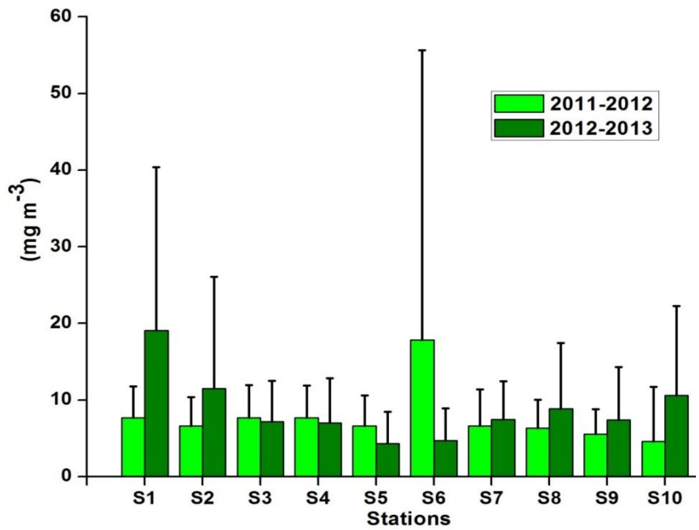

Fig. 3.22 Mean spatial variation of chlorophyll *a* in Vembanad estuary during 2011-2013 period

In 2012- 2013 period, chlorophyll *a* value showed wide range of variation between 0.03 to 76.84 mg m^{-3} with an average of 8.8 ± 1.06 mg m^{-3}. Minimum chlorophyll *a* value was observed in station 6 during July 2012 and maximum at station 10 during March 2012. Spatial analysis showed that higher value was found at station 1 (19.07 ± 2.13 mg m^{-3}) and lower value at station 5 (4.3 ± 4.14 mg m^{-3}). Temporal variation showed that highest value was observed during February 2013 (17.6 ± 2.59 mg m^{-3}) and lowest value in August 2012 (0.37 ± 0.35 mg m^{-3}). The seasonal average was maximum during PRM followed by PM and MN. When comparing the southern zone and northern zone, the bottom water chlorophyll *a* content was higher in northern (8.57 ± 1.51 mg m^{-3}) than southern zone (6.79 ± 1.94 mg m^{-3}). ANOVA also showed significant ($P \leq 0.01$) temporal variation of chlorophyll *a*.

3.2.5.3 TRIX Index

During 2011-2012 period, the average TRIX value were 6.49 ± 0.69. It varied from a minimum of 3.56 to a maximum of 8.04. The minimum value was observed in station 3 during January 2012 and maximum value was in station 10 during October 2011 (Fig. 3.23). Premonsoon season showed the maximum value followed by MN and PM. Spatial variation of TRIX value found that, station 10 was having maximum average concentration (7.12 ± 0.47), followed by station 1 (6.88 ± 0.46) and station 8 that recorded the minimum value (6.1 ± 0.57). Month wise mean value recorded the highest in October 2011 (7.09 ± 0.41) and lowest in January 2012 (5.45 ± 0.85). In the southern zone, the average TRIX index showed higher value (6.53 ± 0.25) than the northern sector (6.48 ± 0.39).

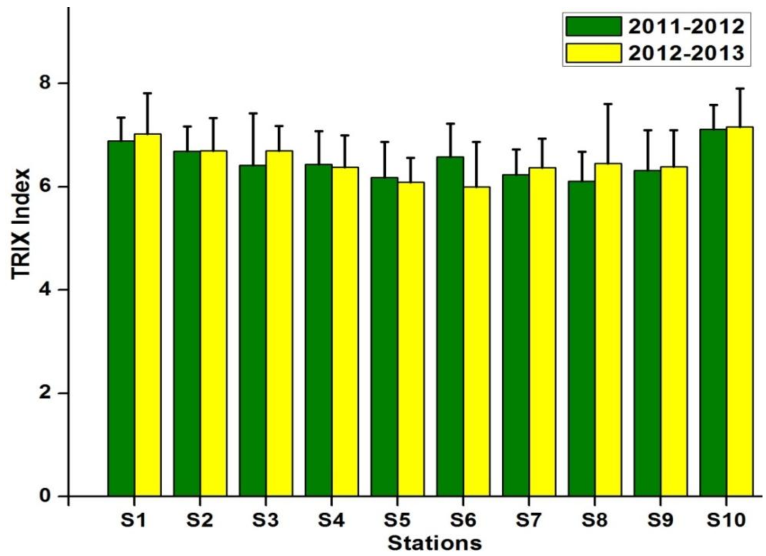

Fig. 3.23 Mean spatial variation of TRIX value in Vembanad estuary during 2011-2013 period

During 2012-2013 period, the TRIX value varied from 3.31 to 8.1 with an average of 6.53 ± 0.77. The minimum value was recorded in station 2 during August 2012 and the maximum value in September 2012 at station 10. Seasonally it revealed that maximum concentration was observed during PRM and minimum during MN. Station wise mean value recorded the highest TRIX value in station 10 (7.16 ± 0.74) followed by station 1 (7.03 ± 0.78) and lowest in station 6 (6 ± 0.87). On comparing the southern and northern zones, the TRIX value varied with lower values (6.44 ± 0.46) in southern than northern zones (6.59 ± 0.38). The ANOVA analysis of TRIX index showed that the spatial and temporal variation were significant ($P \leq 0.01$).

3.2.6 Data Analysis

3.2.6.1 Cluster analysis

Spatial average of physico-chemical parameters were subjected for cluster analysis. During 2011-2012 period the resulting dendrogram showed two main clusters (Fig. 3.24). Cluster one was further divided into two clusters, where station 1 stands separately. Stations on the southern sector (except station 1) formed one group and station 7 and station 8 formed another group. Cluster two corresponds to the northern most stations; its solid line represented the significant spatial variation from other stations. During 2012-2013 period, the cluster analysis revealed that the physico-chemical parameters of station 10 was distinct from other stations (Fig. 3.25). Station 10 stands apart from other stations and all other stations formed a single cluster. It further divided into two, in which stations on the southern zone of TMB formed one group and northern stations (except station 10), formed another group.

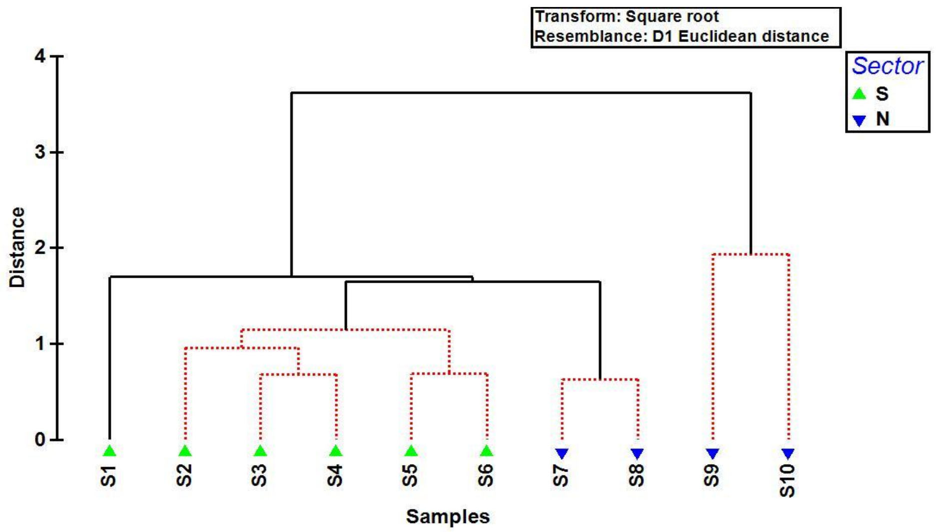

Fig. 3.24 Dendrogram showing the relationship of physico-chemical parameters among the stations in Vembanad estuary during 2011-2012 period

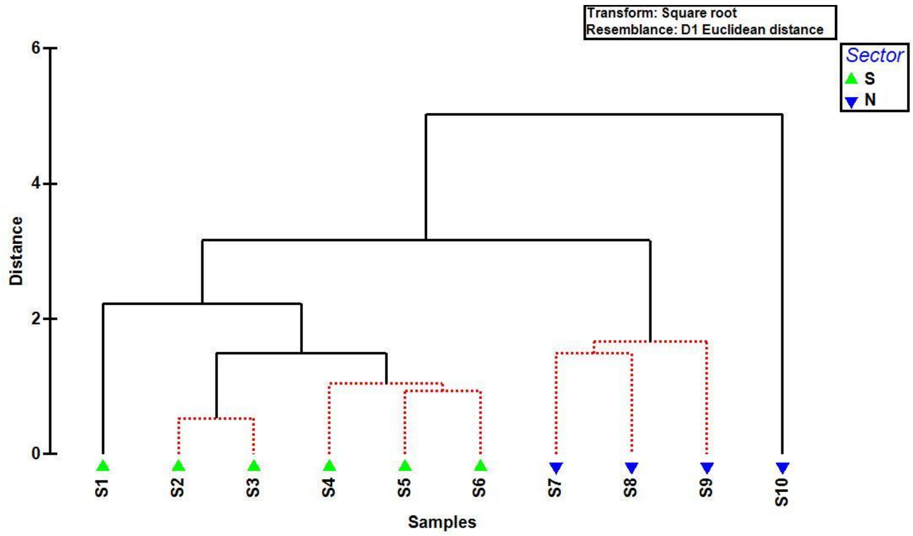

Fig. 3.25 Dendrogram showing the relationship of physico-chemical parameters among the stations in Vembanad estuary during 2012-2013 period

3.2.6.2 Principal Component Analysis

Principal component analysis (PCA) extracted two principal components (PCs) from the variance present in the data. The PCA was carried out with factors having eigenvectors higher than one. Factor loadings (correlations between the variables and the extracted factors) for the two retain Eigenvalues are given in Table 3.1. The first two PCA axes accounted for 87 % of the variation in water quality, whereas the remaining PCA axes each explained ≤ 10% of the variation. The variables with highest loadings on PC1 were alkalinity, salinity and TDS and the second PC had high loadings values for DO, NH_4-N and temperature (Fig. 3.26). Similar to 2011-2012 period, during 2012 – 2013 period the first two PCs accounted for nearly 81% of the total variance (Table 3.2). In the first component strong loading were obtained for NO_2-N, alkalinity, conductivity, TDS, salinity, turbidity, PO_4-P and pH. The second component had positive loading with NO_3-N and negative loading with NH_4-N (Fig. 3.27).

Table 3.1 Results of PCA analysis of physico- chemical parameters in Vembanad estuary during 2011-2012 period.

Parameter	**PC1**	**PC2**	**PC3**	**PC4**
pH	0.930	0.171	-0.029	-0.040
Temperature (TEM)	0.612	0.708	-0.042	-0.251
Salinity (SAL)	0.954	0.249	0.007	-0.018
Dissolved Oxygen (DO)	-0.037	0.920	0.304	0.171
Phosphate-phosphorus (PO_4)	0.919	-0.119	0.232	0.152
Ammonia –Nitrogen (NH_3)	0.420	-0.782	0.414	0.054
Nitrate – Nitrogen (NO_3)	0.789	-0.534	-0.035	-0.286
Nitrite – Nitrogen (NO_2)	0.656	-0.131	-0.701	0.202
Turbidity (TUR)	0.934	-0.307	0.070	0.128
Total Dissolved Solids (TDS)	0.950	0.304	0.008	0.011
Alkalinity (ALK)	0.982	0.149	0.086	0.016

*The code of parameter used for PCA analysis was given in parenthesis

Table 3.2 Results of PCA analysis of physico- chemical parameters parameters in Vembanad estuary during 2012-2013 period.

Parameter	**PC1**	**PC2**	**PC3**	**PC4**
Nitrite – Nitrogen (NO_2)	0.982	0.139	-0.042	-0.080
Alkalinity (ALK)	0.947	-0.092	0.198	-0.175
Conductivity (COND)	0.924	0.304	0.103	-0.199
Total Dissolved Solids (TDS)	0.919	0.330	0.039	-0.203
Salinity (SAL)	0.862	0.480	0.041	-0.138
Turbidity (TUR)	0.852	0.075	-0.465	0.195
Phosphate-phosphorus (PO_4)	0.839	-0.300	-0.318	0.250
pH	0.617	-0.538	0.515	0.023
Ammonia – Nitrogen (NH_3)	0.595	-0.648	0.248	0.151
Nitrate – Nitrogen (NO_3)	0.290	0.620	0.331	0.645
Dissolved Oxygen (DO)	-0.714	0.449	0.295	-0.093

*The code of parameter used for PCA analysis was given in parenthesis

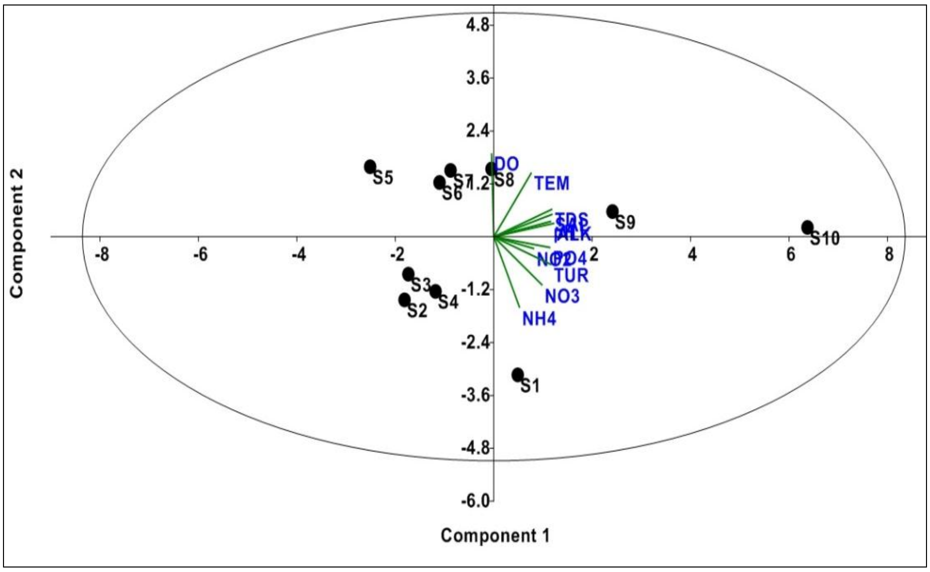

Fig. 3.26 PCA ordination of physico-chemical parameters of Vembanad estuary during 2011-2012 period

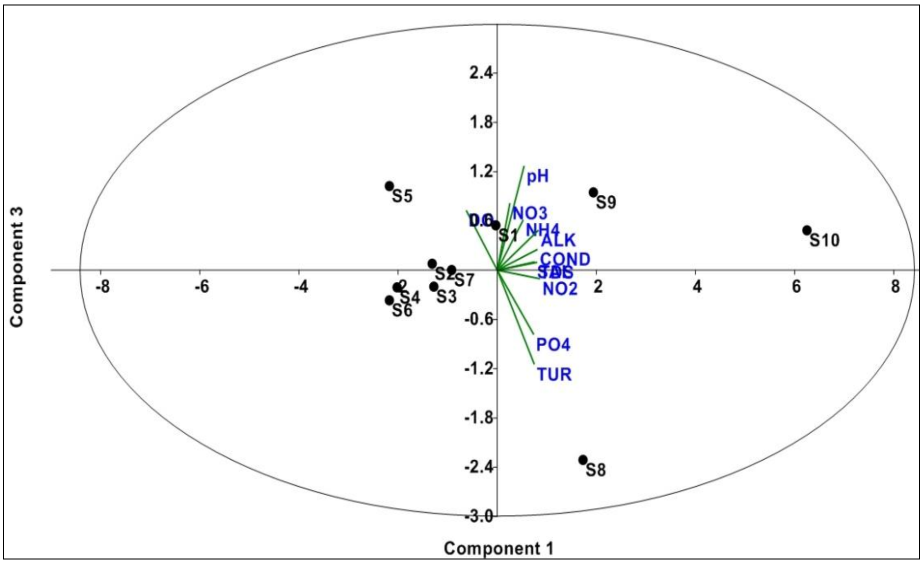

Fig. 3.27 PCA ordination of physico-chemical parameters of Vembanad estuary during 2012-2013 period

3.3 Discussion

Estuaries are highly dynamic environments that display strong temporal and spatial variation in their physical, chemical and biological characteristics. So, this dynamism leads to a large variability in estuarine characteristics. Generally no two estuaries are identical, where each have its own size, shape and bathymetry, length of tidal influence, tidal range, freshwater inflow, turbidity and flushing time etc. Thus, each estuary is unique and there is no general parameter to readily assess the 'health risk' of an estuary from natural and human influences. The ecological health of estuaries depends on the 'successful' interaction – and the limits to the possible interaction that are imposed by the temporal and physical variability between organisms and variations in salinity, currents, waves, suspended particulate matter (SPM), bed sediments, temperature, air exposure, hypoxia, wetland contaminants and biodiversity (Wolanski et al., 2004). During monsoon season, the amount of precipitation and land runoff naturally carry about the dynamic variation from a marine environment to a brackish water characteristics in estuaries (Qasim and Sen Gupta, 1981). The tidal mixing and estuarine circulation are vital factors controlling the hydrographic condition of an estuary which in turn influence the nutrient dynamics and overall productivity of the system. Tropical estuaries, which formed the most active biogeochemical zones in the biosphere, are more easily affected by anthropogenic nutrient loading than higher latitude (Yule et al., 2010; Smith et al., 2012). Estuaries are facing threats of degradation from multiple human impacts such as sewage discharge, pollution from pesticides, fertilizers and municipal wastes, conversion of land for agriculture, land reclamation for tourism activities and other construction activities. These overall intensified by the rapid expansion of the coastal population (Galloway et al., 2008). Human activities increased the nutrient path from land to sea in the past decades that resulted in environmental deterioration and biogeochemical processes modification. Human-induced nutrient enrichment is becoming a serious problem in coastal marine areas around the world. Eutrophication of coastal waters has resulted in a number of adverse effects on ecosystem, including algal growth, hypoxia, significant loss of marine life and habitat and the contamination or impairment of commercial shellfish stocks (Cook et al., 2010). Trophic state measurements serve as standards for measuring the health of an estuarine system. Over the decades, there were number of interventions and alterations faced by Vembanad estuarine system, mainly through estuarine reclamation for agriculture, urban growth and industrial development, tourism exploitation etc. This would have severe adverse impact on the environmental quality of the estuarine system.

The west coast of India obtains the heaviest rainfall in the Indian subcontinent due to the orography of the region (Suprit et al., 2012). Additionally, due to the geographical peculiarities, the major share of the heavy summer-monsoon rainfall reach the Arabian Sea through the rivers (Suprit et al., 2012). Rainfall intensity effects in the nutrient transport in surface runoff. During periods of heavy rainfall, increased runoff brings more nutrients into the estuary, as it carries terrestrial washouts along with the surface runoff. According to Alongi (1990), the life cycle pattern of tropical coastal fauna are matched with monsoonal rainfall. Any alteration in the rainfall pattern have a negative impact on the benthic community (Grilo et al., 2011; Sivadas et al., 2012). The tropical estuarine organisms spawn during the monsoon season followed by their recruitment and so variation in natural monsoon pattern affect the recruitment of macrobenthos and its overall distribution (Gaonkar et al., 2013). According to Krishnakumar et al. (2009) the weather systems in recent years over peninsula have collapsed and can cause a drastic decline in southwest monsoon rainfall over Kerala. It is supported by Rupa Kumar et al. (1992) and Guhathakurta and Rajeevan (2007), that June and July are the rainiest months in Kerala, but, nowadays it showed a drastic decline in rainfall. Similarly, the percentage contribution of rainfall during the southwest monsoon has declined whereas that during pre- and post-monsoon season in the State has increased. The above phenomenon was more significant in recent decades. Seasonal alteration of river discharge induced by the monsoon, play an important role in the spatial and temporal variations in nutrients and other biotic components in estuarine waters. The increased freshwater discharges into the estuary during the rainy season and decreased discharges during the dry season presumably lead to increase in delivery of terrestrial materials into the estuarine region during MN and low terrestrial input during dry season. In the present investigation 66% of river discharge was obtained during the MN season and minimum discharge was received during PRM period. Four rivers such as Pamba, Manimala, Achenkovil and Meenachil were drain into the southern zone and receive 57.9 % of the total river discharge, whereas Muvattupuzha River carries 42.1 % in the northern sector. In Cochin estuary, Revichandran et al. (2012) noted that, the annual total river discharge was 22 × 10^3 Mm^3/year from the seven rivers and from the total discharge value, around 60-70% of received during MN season and the annual total discharges from river Periyar (6795 Mm^3) and Achenkovil (1250 Mm^3) brought the maximum and minimum rate respectively. The MN, propels flow towards the sea during the entire tidal cycle, except near to the barmouth, then with the decrease in river discharge, flow reversal occurs resulting in reduced river discharge during PRM season in the estuary (MSSRF, 2007). The flow pattern in the estuary depends mainly on the tidal conditions. During MN season both tidal and river discharges play a great role in maintaining the flow pattern. According to Joseph (1989), the hydrographic characteristics of Vembanad vary annually, particularly during July and August the estuary behaves as a salt-wedge type. But, during PM months, it shows significant stratification and in PRM a well-mixed type estuarine condition evolves. According to the modified tidal prism method in Cochin estuary, the flushing time was found to be of the order of 16 – 21 days during summer (www.ramsarkerala). Towards Cochin barmouth (Arabian Sea), the freshwater river discharges become saltier. Based on the ratio of river discharge to volume of estuary which was noted as nearly 42, Revichandran et al. (2012) grouped Cochin estuary as monsoonal estuary where the river discharge shows large seasonal variation and complete freshening of estuary happened during MN season. Knowledge on tidal dynamics is important for analyzing many oceanographic processes in any location, especially in the coastal and estuarine sites, because tide is an important factor controlling the mixing, circulation and other processes in the sites (Srinivas et al., 2003). All the estuaries have specific tidal features that affect the mixing, circulation and hydrodynamic reaction of the system (Paeer Selvam, 2014). According to Srinivas et al. (2003) the variation of sea level was subjected to tides but inside the estuary its power decreases quickly. In estuaries, tidal force is affected by factors such as friction, river runoff, nature of bathymetry and also the shape of the channel (Vinita et al., 2015). Tides in Cochin estuary are mixed, predominantly semi-diurnal type with an average tidal range of 1m (Qasim and Gopinathan, 1969). Tides with two high and two low tide of unequal heights take place every day and the tidal influence that was strongest near the mouth and decreased in the upstream direction (Revichandran et al., 2012). During spring tide, the maximum tidal range of 65.22cm in the barmouth region decreased to a minimum of 13.44 cm at Thanneermukkom (40 km south of barmouth). During neap tide, the tidal range was 30.64 cm at barmouth region that decreased to 9.4cm at Thaneermmukkom. They also noted the impact of closure of TMB on the tidal amplitude and water level ranges in the area. During the prebarrage phase Josanto (1971), noted the influence of mixed nature of tide upto Pulikkezhu, 80 km south of barthouth, with maximum range of tidal amplitude upto 5cm, and at Punnamada, it was 22 cm. But due to the construction of TMB and its closure during summer, cutoff the tidal inflow into the southern sector completely (Anon, 2001). Similar condition was existed during the present study period also. This may adversely impacted the hydrography of the region by preventing the transport of nutrients and saline water exchange into the southern sector with the tide. An unusual amplification of Msf tidal constituent (14.7 day period) in the attenuating tidal regime of Cochin estuary was reported by Joseph et al. (2007). It was reported that the Msf tide has considerable impact on the water level regime and salinity distribution in the coastal lagoon. In Cochin estuary, comparatively large Msf tidal amplitude recorded in the upstream regions of estuary influenced by the bottom frictional effects at fortnightly periodicity. They also noted that amplification of Msf tides were high in the sandy bottom (bed-roughness more) environment of Punnamada than the fine clay substratum of barmouth region. Shivaprasad et al. (2013b) observed that gradual closing of TMB associated with the decreased river flow and increased tidal force from Cochin inlet generate the horizontal salinity distribution in the estuary.

According to MSSRF (2007), the rapid shrinkage of flood carrying capacity of Vembanad lake by 78% occurred due to the reduction in the lake area and depth. They observed that the rapid decrease in area was mainly due to unrestricted encroachments and reclamation of lake area and raising lakebed due to silting. Until, 1980s paddy culture was responsible for the reclamation but the intensive tourism activities was accountable for the reclamation of lake area now. The 36,329 ha of open water area of Vembanad estuary existed before agriculture practice in Kuttanad, reduced to 12,700 ha (34.8%) (Gopalan et al., 1983). It further declined to 13,224 ha in 1992 and 12,504 ha in 2000 and its rate of decline has increased over the years. The annual rate of decline during the first phase (1834-1983) was 0.23 %, in the second phase (1983-1992), which increased to 4.93 % and to 0.68 % during the third phase (1992-2000). Remote Sensing and GIS analysis of Vembanad estuary revealed that from 1944 to 2009, the estuarine area was declined to 12.96% (15.11 km^2) (Dipson, 2012). The depth of the estuary is also decreasing and its rate increased after the construction of Thanneermukkom barrage particularly the cofferdam situated in the middle of the estuary. Depth of Vembanad estuary reveals a declining trend from early 1900's onwards (Gopalan et al., 1983) and the average depth at various parts reduced from 6.7 m to 4.4 m from 1920 to 1980 (Table 3.3). During the current study period an appreciable decrease in the depth of the estuary from an average of 4.4 m during 1980 (Gopalan et al., 1983) to 4.1 m during the present study period. Compared to northern zone, southern zone was deeper. Indiscriminate mining of lime shells and river sands are more in the southern zone, which causes severe damage to the lake system (Maya, 2006). From the river catchment of Vembanad lake, an average of 11.73 million ty^{-1} of sand and gravel were removed from the channel and 0.414 million ty^{-1} of sand from the river floodplains (Padmalal et al., 2008). The depth between Varanadu to Perumbalam showed an average depth of 4.5 m in 1980s which declined to 2.9 m in the present study (Fig. 3.28). During the current study, the minimum depth range in station 8 was responsible for the mechanized dredging for lime shell collection in the area and the dredged material was washed for collecting shells. During 2005-2006 period, in the Varanadu region had an average depth of 2.68 m (Lekshmi and Jaya, 2007). Dredging along with waste disposal and accumulation is altering the depth of the region. Land reclamations and other illegal constructional activities have also caused serious alterations in the depth of the estuary. Along with the depth, the water carrying capacity of the lake system has reduced to 78%, *i.e* from 2.4 km^3 to a mere 0.6km^3 (MSSRF, 2007). The geospatial analysis of Cochin estuary showed that the estuarine system is being inflicted with major geomorphic changes at several segments and also its reduction in area. From 1967 to 2011 onwards, the shrinking rate of the system was estimated as 0.288 km^2/year. Quantitative comparison of inner island segments of the estuary shows significant changes in island widths and area through time (Dinesh Kumar et al., 2014).

Table 3.3 Reduction in depth range in different locations of Vembanad estuarine system over the years

Source: (Gopalan et al., 1983)	Depth range (m)		
Location	**Year 1920**	**Year 1980**	**Present study**
South of Thaneermukkom	8 - 9	3 - 3.5	1.6 - 4.5
Between TMB north to Varanadu	8 - 9	3 – 4	2.5 - 7
Between Varanadu to Perumbalam	7 - 9	4 – 5	1.5 - 4.3
Aroor region	7- 8	7- 8	6.0 - 8.5

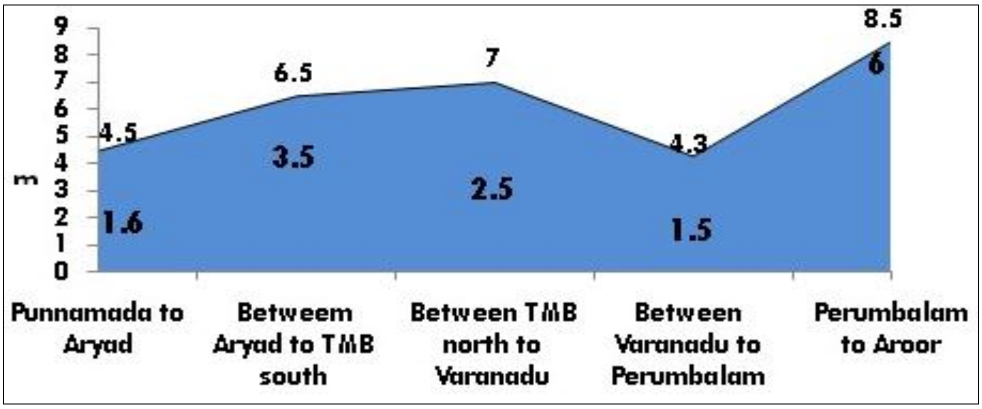

Fig. 3.28 Average sector wise depth variation in Vembanad estuary during 2011-2013 period

In aquatic habitat, water temperature is an important parameter affecting the chemical process such as dissolution-precipitation, adsorption–desorption, oxidation–reduction and physiology of biotic community (Aken, 2008). According to Brey and Clarke (1993), the differences in growth and productivity of benthic populations from different regions are mainly related to temperature. Changes in temperature may directly influence mortality, reproduction, onset of spawning, and the embryonic and gonad development of benthic species, and thus may change phenological processes (Birchenough et al., 2011). In the present investigation higher temperature observed during PRM and lower during MN. According to Pillai et al. (1975), the temperature values for the entire period of estuarine observation reflect climatic variations to a certain extent and also due to different timing of collection (Jayaraman et al., 2003). Sankaranarayanan and Qasim (1969) stated that the influx of fresh water and influx of cold water from the sea in to the estuarine system may be a significant factor which brings down the water temperature in the estuary. The increased trend of temperature in summer months was attributed by the high intensity of solar radiation (Govindasamy et al., 2000). During MN season the increased freshwater riverine inflow from Pamba, Manimala, Achenkovil and Muvatupuzha river may bring down the temperature. Qasim (2004) gives a detailed temperature regime of various estuaries in India. In Hoogly estuary, surface water temperature ranged from 18 to 33°C; that in Godavary estuary from 26 to 30 °C, that in Ashtamudi estuary from 26 to 35.5 °C; that in Cochin backwater from 28.4 to 31.8 °C, and in most of the Mumbai estuaries, it was 28.5 to 31 °C. Over the years, when comparing the previous studies, the temperature regime of Vembanad estuary was slightly higher. During 1967 the temperature in Vembanad estuarine system ranged from 26.7 to 31.1 °C (Devassy and Gopinathan, 1970); that in 1972 from 26.5 to 30 °C (Haridas et al., 1973), that in 1995 from 28.6 to 31.9 °C (James, 1997) and that in 1998 it ranged between 23.3 and 31 °C (CWRDM, 2006). These results reflect a rising temperature pattern over the years possibly associated with impacts of climate change (Asha et al., 2016). The reduction in temperature in the southern zone was mainly due to the increased river discharge in this zone.

From the physico-chemical parameters, pH is considered as an important ecological factor whose variation in any aquatic system affects the inhabitants as they are adapted to an optimum pH and cannot withstand abrupt changes. The processes of primary production, respiration and mineralization of organic matter may alter the pH of the system because they can cause significant changes in the oxygen and carbon dioxide concentrations of aquatic environments. During the study period, a slight acidic to alkaline condition was observed in the Vembanad estuary, indicating the sea water influence through Cochin bar mouth (Joseph and Ouseph, 2010; Jayachandran et al., 2013; Asha et al., 2015). The observed variations in pH in the southern most stations could be ascribed to the relatively higher freshwater discharge from rivers like Manimala, Meenachil, Pamba and Achencovil which drain into the Vembanad estuary on the Southern region. These rivers and streams transport large quantities of humic material in colloidal suspension which are frequently slightly acidic. Which upon meeting sea water, the colloidal particles get coagulated and the pH shifts towards the alkaline side (Ried, 1961). Hence, the values were higher toward downstream, in the northern zone. Compared to other seasons, pH in MN season was always less in both the years. Annual monsoonal flood water flushed into the estuary during MN season (Anon, 2007) and the acidic organic sediments beneath reflects the slight acidic nature of pH during MN. During both the years, station wise mean values of pH was recorded the highest in station10. Compared to other stations, in station 10, which is located close to the Cochin barmouth, seawater plays a significant role in controlling the pH. Southern stations such as station 1 (Punnamada) and station 2 (Pallathuruthy) have minimum range of pH in both the years. It was noted that the soil of Kuttanad area was highly acidic (Thampatti and Jose, 2000) and this will impart low pH in the southern stations. The ‘kari’ soils in Kuttanad region are acidic in nature (due to rich organic carbon) especially in the dry months. The runoff from these agricultural fields with the onset of the southwest monsoon may also attribute to the acidic pH in both station 1 and 2. Removal of CO_2 by photosynthesis through bicarbonate degradation, dilution of seawater by freshwater influx, reduction of salinity and temperature and decomposition of organic matter can be related with the fluctuations in pH values (Upadhyay, 1988; Rajasegar, 2003; Paramasivam and Kannan, 2005). The low variation of pH observed in the estuary, suggests that the water mass remained well buffered throughout the study period and indicates the presence of biodegradable organic matter in the water column (Sarkar et al., 2007). In both the years, southern zone of the Vembanad estuary showed lower pH as compared to north. As stated earlier, this may be due to the combined effect of the runoff from the acidic soils of the 'kari lands'and riverine inflow to these southern stations. Time scale study found that the average pH of the southern zone was increasing over the years. During 1980-1981, the average pH of the southern zone was 6.5 (Gopalan et al., 1983), whereas during 1994 – 1997 period it become 6.8 (Anon, 2001). During closure period of TMB, the average pH in the southern zone was little higher than (7.5) the open period (6.8). Due to the lack of tidal flushing all the pollutants including house hold wastes, house boat discharges etc. accumulate in the southern zone and make the area alkaline. Correlation analysis showed that, during second year pH showed significant positive correlation with temperature (r^2 = 0.3). As pH is a measure of the hydrogen ion concentration, a change in the temperature of a solution will be reflected by a subsequent change in pH. Similar type of correlation was observed in Valanthakad back water (Meera and Bijoy Nandan, 2010) and Cochin estuary, Kerala (Shivaprasad et al., 2013a). According to Riley and Chester (1971), pH value is positively related to the temperature.

In the present investigation, the transparency of the water column was maximum during PM season and minimum during PRM season. This is in contrast with the observations of Madhu et al. (2007) and Radhika (2013), in Cochin estuary. Variation in the seasonal trend of transparency was attributed to the heavy precipitation and associated riverine runoff prior to the field sampling in the study stations. Similarly the main basin of the estuary get clogged with the water plants, *Eichhornia*, *Salvinia* and other aquatic weeds during the PRM season that also reduced the visibility of the water body. Increased river discharge, intense rain fall and the cloud cover during MN reduces the solar radiation and large input of suspended materials makes the estuarine water turbid. According to Madhu et al. (2010) PRM season was associated with reduced fresh water flow and elevated solar radiation making the water column more transparent. Qasim et al. (1968) mentioned about the reduced transparency during MN and its implications on phytoplankton composition and physiology in Cochin estuary. During PRM season study Bijoy Nandan (2008), observed a lower transparency in the retting zones of backwater systems of Kerala, mainly due to the accumulation of coir pith and ret liquor containing organic acids like pectin, pentosan, phenol, tannin, etc. in the water body. Renjith (2006) observed that lower transparency during MN was not mainly from the suspended matter, it was due to the cloudiness associated with rain fall. Similarly after the end of MN the transparency also increased. Comparing to previous studies (av. 1.4 m) (Anon, 2001), the transparency of water column in the southern zone of Vembanad estuary was slightly higher (av. 1.1 m). During closure period of TMB, the average transparency of water column in the southern zone was little lower than (0.98 m) the open period (1.04 m). Relatively low transparency in the dam sites was mainly due to the absorption and scattering of sunlight by dissolved substances, inorganic particles, detritus and plankton (Francisco and Martinez, 1993). Downstream stations were clearer by trapping the sediments by the impoundments and allowed clearer water to the downstream (Edokpayi and Osimen, 2002). According to Madhu et al. (2007) due to less turbidity, low value of the light attenuation coefficient was observed during PRM (av. 1.2 ± 0.2 m^{-1}) as compared to MN (av. 1.7 ± 0.2 m^{-1}) and PM (av. 2.4 ± 0.4 m^{-1}). According to

Saha et al. (2001) the turbidity of water inhibited penetration of incident light and limited the euphotic depth within a narrow range of 0.075 to 0.228 m and this condition is not favorable for photosynthetic production. Particulates and dissolved organics introduced by the freshwater runoff may increase light attenuation, limiting the growth of primary producers. Qasim et al. (1968) obtained light extinction coefficient value of 1.5 for the very turbid estuarine waters in Cochin estuary. The minimum spatial average of transparency in station 10 was mainly attributed by the increased turbidity of the area. Similarly the increased clay fraction in the sediments of the area may be re-suspended by strong tidal currents that reduce the transparency and increase turbidity.

Electrical conductivity represents the total concentration of soluble salts in water (Purandara et al., 2003). Higher values of conductivity may be attributed to high ionic concentration (Young et al., 1972) and release of various dissolved substances into the aquatic medium through death and decomposition of organisms (Raja et al., 2008). When comparing the conductivity values for both the years, the average value was found to be maximum in northern than the southern zone. The highest conductivity value was mostly observed in station 10. This was mainly attributed to the sea water inflow from the Arabian Sea which increases the electrical conductivity value. In the southern zone, fresh water runoff from Meenachil, Manimala, Achenkovil and Pamba river reduces the conductivity. In the post barrage phase, reduction in soil conductivity value was noted by Thampatti and Jose (2000). They observed that the closure of barrage leads to the prevention of saline water intrusion from the sea, which was the major source of salts in to the southern zone of Vembanad and it ultimately decreased the electrical conductivity of the soil system. The electrical conductivity was low during MN and high during PRM that was associated with the variations of salts mainly through river runoff and tidal intrusion. The increased fresh water runoff during MN also imparts a low conductivity and similarly increased influx of seawater control the high conductivity during premosoon season (Narendra Babu et al., 2009). A similar condition was observed in the present study. A large variation in the conductivity pattern in the southern and northern zone of Vembanad was mainly attributed by the operation of TMB. During the closure period of TMB, the saline water intrusion from Arabian sea was prevented and ultimately the entry of salts into the southern zone stopped, which reduces the conductivity of the southern zone. Comparing to previous studies (av. 2.8 mS cm^{-1}) (Anon, 2001), the conductivity of water column in the southern zone of Vembanad estuary was slightly lower (av. 2.59 mS cm^{-1}). Nair and Pillai (1990), observed about 90% salinity reduction during the months of March to May as compared to that of pre-barrage period. In the southern zone of Vembanad estuary Vincy et al. (2012) observed the highest conductivity during the post-tourism periods (April-May) and also found that inorganic salts dissolved in the water influenced the characteristics of conductivity and also noted a positive correlation of conductivity with TDS, silicate and BOD and negative correlation with DO. In the present investigation, both the years conductivity showed a significant ($P \leq 0.01$) positive correlation with water temperature and pH. The conductivity value of mangrove swamps of Ariyankuppam estuarine complex, Tamil Nadu ranged from 26.41–41.33 mS cm^{-1} and it showed a positive correlation with pH (Satheeshkumar and Khan, 2012).

The total dissolved solids (TDS) in water mainly consist of inorganic salts and dissolved substances which typically comprises of anions and cations. Water with high TDS value decreases its quality and cause unfavorable condition for biotic components. In both years, TDS was minimum during MN. This was due to the increased precipitation and river runoff during MN and increased dilution. The increased TDS value in the northern zone could be due to the salt water influence from the adjoining Arabian sea. Similar condition of low TDS during MN and higher value in the estuarine stations of Mahanadi river–estuarine system was reported by Sundaray et al. (2006). In contrast, maximum TDS value was observed during MN season in Kodungallur -Azhikode estuary (Jayachandran et al., 2012). During closure period of TMB, the average TDS of water column in the southern zone was little higher than (1.7 ppm) the open period (1.5 ppm). The study report of Atree (2012), stressed that during December to May period, the TDS value in the southern zone of Vembanad increased and from June to November period it decreased. This type of situation was attributed by the effect of closure of TMB. The increased TDS rate in station 8 could be due to the effluent discharge from the distillery situated on the banks of the estuary at Varanadu. Lekshmi and Jaya (2007), has also reported on the inefficient treatment process of distillery resulting in higher TDS of the water body. The increasing unregulated dredging activities and illegal constructions on the Perumbalam islands and nearby region may contribute the elevated level of TDS in station 9. Discharge from seafood industries along with municipal waste discharge from Cochin metro city and salt water incursion from the Arabian Sea play a significant role in increasing TDS in the northern zone, especially in Aroor (station 10) which is the hub of sea food processing industries. This zone is also influenced by riverine discharge from the upstream by Muvattupuzha and the Periyar river in the downstream, which could directly influence the mixing of dissolved solids. Dredging activities as a part of deepening the boating channels in the northern zone is disturbing the benthic zone by releasing the fine sand particles, resulting in higher TDS value. Both the years, TDS showed significant ($P \leq 0.01$) positive correlation with water temperature, pH and conductivity. According to Gopinathan and Qasim (1971) the total dissolved solids give the indication of the conductivity values. In Cochin estuary, TDS established a strong significant positive correlation with conductivity ($P \leq 0.01$) (Thasneem, 2016).

Turbidity indicates water clarity and it is the measured by light scattering by suspended particles in the water column. Several factors are responsible for water turbidity; suspended soil particles including clay, silt, and sand; tiny floating organisms such as phytoplankton, zooplankton, and bacterioplankton and small fragments of dead plants (Voluntary Estuary Monitoring Manual, 2006). Turbidity forms the primary factor responsible for low light penetration as noticed in MN fed tidal estuaries (Chandran and Ramamoorthi, 1984; Balakrishnan Nair et al., 1984c). In the coastal wetlands of south India the turbidity value in the retting zone and non-retting zone was 2.62 NTU to 1.60 NTU respectively (Bijoy Nandan and Abdul Azis, 1997). In the retting zone of Paravur backwater, Kerala the turbidity value during the PRM and PM season had a value of 1.66 NTU (Suja, 2014). Generally the turbidity level of Vembanad Kol wetland is lower (NWIA, 2010). According to Gopinathan and Qasim (1971), comparatively low level of sediment concentrations was observed in Cochin estuary. According to Mustapha et al. (2012) a slight change in solids in the water body would have large effects on turbidity levels. In Sundarbans estuary turbidity ranges from 35-150 NTU, and highest index being observed during MN (Manna et al., 2012). In the two year study period the seasonal variation of turbidity follow different pattern. The maximum turbidity was observed in station 10 (Aroor) during both the years. High riverine inflow, dredging and municipal discharge from the Cochin metro city all together make the water in these northernmost stations turbulent, resulting in higher turbidity. Human activities and urban discharges contribute large amounts of suspended material, increases the turbidity of water column. Similarly seawater intrusion from Arabian sea, along with mixing of decaying organic matter enhanced the turbidity. Increase in turbidity level, especially in MN season may attributed to high influx of silt content from the perinial rivers, agricultural runoff and other allochthonous organic matters (Jayachandran et al., 2012). During second year, elevated spatial level of turbidity in the

Varanadu region (station 8) might be from the effluent discharge from the distillery operations. During the sampling period, the estuarine surroundings also had odour alcohol/ester as part of distillery activities. Lekshmi and Jaya (2007), reported about the inefficient treatment process from the effluent generated after the distillery operations also releasing into the estuary may lead to increased turbidity. In the southern arm of Cochin estuary, Renosh et al. (2010) observed higher values of turbidity during high tide phase and a moderate decrease in its values during ebb phase. He noted the relation between development of turbidity maxima and salinity factors and also the riverine inputs of suspended solids. They identified a higher turbidity values and turbidity maxima zones near Thevara, near Kumbalam Bridge and Perumbalam and also realized its oscillating nature with respect to the tide. On comparing the values of previous studies (Renosh et al., 2010) Aroor region is considered as estuarine turbidity maxima zone. When compared to the open period of barrage, the turbidity of southern zone was higher during the closure period of barrage. During the closure period of barrage, the southern part of estuary was virtually stagnant and large quantities of contaminants from various sources accumulate in the water column and adversely influenced the turbidity of the southern zone (MoEFCC, 2014). During 2011 – 2012 period, turbidity value showed a significant negative correlation with transparency ($P \leq 0.01$). Turbidity usually reduces the visibility of the water column (Palic et al., 2007). Due to the presence of large quantity of detritus and sediment resulted the turbid water column with low light penetration in Cochin estuary (Qasim and Sankaranarayanan, 1972). During 2012-2013 period, turbidity showed a significant ($P \leq 0.01$) positive correlation with conductivity ($r^2 = 0.48$) and TDS ($r^2 = 0.57$).

The $CaCO_3$ level, organic matter, temperature and partial pressure of CO_2 are the major factors contributing to alkalinity of the water body. According to the Bureau of Indian Standards (BIS), the desirable limit of alkalinity of drinking water specification is 200 mg L^{-1}.Beyond this limit, taste becomes unpleasant. Range of alkalinity in the present study area was within that limit. Compared to other stations, station 10 signified with higher alkaline condition in both years. station 10, which is located close to the Cochin barmouth region where, seawater plays a significant role in controlling the alkalinity of the station. At the same time, in stations 2 and 3, fresh water from rivers such as Pamba and Achenkovil uphold a reduced condition of alkalinity. Tropical river basins play a major role in chemical weathering and transfer of dissolved silicate and alkalinity to rivers and oceans due to their geological and climatic settings (Jennerjahn et al., 2006). The weathering rates are influenced by climatic factors such as temperature and precipitation (Bouwman et al., 2013). The usage of dissolved carbon dioxide for photosynthesis by phytoplankton leads to decrease in CO_2 in the aquatic medium which in turn creates an alkaline condition. Mogalekar et al. (2015) noted a relatively higher alkalinity (139 mg L^{-1}) in the Vembanad estuary at Panangad-Kumbalam mangrove patches of Kochi. During 2011-2012 period, the estimated average alkalinity value was found to be maximum during PRM and minimum in MN. Similar condition was observed in Cochin estuary by Radhika (2013). Qasim and Gopinathan (1969), observed a marked variation in alkalinity value during the MN months. The southern zone of Vembanad was having less alkalinity as compared to northern zone. The saline water containing high amount of carbonate, bicarbonate and hydroxide compounds from the Arabian Sea increases the alkalinity in north. Based on the alkalinity value (av. ~ 37 mg L^{-1}) Meera and Bijoy Nandan (2010), reported the poor condition for fish production in Valanthakad backwater. From the various industrial discharge points into the Cochin backwater, situated in the Eloor (Udyogamandal) region, Sarala Devi et al. (1979) noted an alkalinity range of 10.8 (Indian Rare Earths Ltd.) to 195.8 mg L^{-1} (Zinc Smelters) in the effluents. In the Eloor- Edayar region of Periyar river before the breakage of Pathalam bund in 2014, in the downstream region the surface water alkalinity was recorded as 43.32 ml/L and 51.72 ml/L in bottom water. But after the breakage of the bund average alkalinity was changed into 20.10 ml/L in surface water and 26.07 ml/L in bottom water (KSPCB, 2014). Ratheesh Kumar et al. (2010), noted a variation of alkalinity from 68 mg $CaCO_3$ /L during PM to 216 mg L^{-1} during PRM in industrial area of Cochin estuary. The indiscriminate discharge of the untreated effluents containing heavy metals and pesticides from the Eloor industrial belt is a common point source of pollution to the estuary. When comparing the alkalinity of retting and non-retting zones in the coastal wetlands of Kerala, retting zone having higher alkalinity value of 103 mg L^{-1} and non-retting area having 91 mg L^{-1} (Bijoy Nandan, 2008). During 2011-2012 period, correlation analysis showed that alkalinity have a significant positive correlation with water temperature (r^2= 0.3; $P \leq 0.01$) and salinity (r^2= 0.4; $P \leq 0.01$). During 2012-2013 period, alkalinity showed a significant positive correlation with pH (r^2= 0.33; $P \leq 0.01$), conductivity (r^2= 0.59; $P \leq 0.01$), TDS (r^2= 0.59; $P \leq 0.01$) and salinity (r^2= 0.63; $P \leq 0.01$). Meera and Bijoy Nandan (2010), observed a significant positive correlation with alkalinity and salinity in Valanthakad backwater. Similarly Radhika (2013), observed a significant positive correlation of alkalinity with pH and salinity in Cochin back water. Comparing to previous studies (Anon, 2001), both the years southern zone showing lower alkalinity (av. 21.9 mg L^{-1}). The decreased alkalinity level may be due to the increased lime shell and live clams collection from the region over the years. Similar condition of low alkalinity related to the low lime content of rock was noted in the Ibiekuma stream, southern Nigeria (Edokpayi and Osimen, 2002).

Mixing of sea water with fresh water in estuaries created brackish water to be more saline than fresh water but less saline than sea water. Salinity of estuaries usually increases away from a freshwater source such as a river, although evaporation sometimes causes the salinity at the head of an estuary to exceed seawater (Manna et al., 2012). According to Santra et al. (1989) vertical salinity structure and nature of salinity variation along an estuary is the unique feature of coastal water ways. Salinity regime of the Vembanad estuary is governed by inflow from Arabian sea through Cochin barmouth. Based on the Venice System for the Classification of Marine Waters, a salinity classification (Anon, 1958) was adopted. According to this, in the Vembanad estuary the observed zones were limnetic (< ± 0.5 ppt), Oligohaline (± 5 - ± 0.5 ppt), Mesohaline (± 18 - ± 5 ppt), Polyhaline (± 30 - ± 18 ppt), Mixoeuhaline (> ± 30 ppt but < adjacent euhaline sea). During the study period the salinity values ranged from 0 ppt to 33.1 ppt. During both the years the average salinity pattern in station 1 to 5 denoted as an oligohaline zone. During first year stations 6 to 8 included in the oligohaline zone, whereas stations 9 and 10 were grouped as a mesohaline zone. During second year, station 7 to 9 denoted as mesohaline and station 10 as polyhaline zone. In the southern stations, the high fresh water influx from Pamba, Manimala and Achenkovil rivers were the main factor for the low salinity level. Agricultural runoff from the nearby paddy fields and the southwest monsoon also bring enormous amounts of freshwater into these southern stations, creating a pure limnetic condition. As the salinity incursion from Arabian Sea is blocked during the bund closure period, stations in the southern part of TMB remains oligohaline throughout all the seasons. Menon et al. (2000) stated that the major hydrological variable in the Cochin backwaters is salinity, and noted a salinity of 30 ppt at the entrance of the estuary to 0.2 ppt at the point of entry of the rivers and found that salinity gradient in the Cochin backwaters supports diverse species of flora and fauna depending on their capacity to tolerate oligohaline, mesohaline or marine conditions. Qasim (2004) has given the details of salinity pattern of most of the estuaries in India and noted that in most of the estuaries annual variation in salinity was large from the mouth to upstream. In Hoogly estuary, seasonal changes of salinity was evident and it ranged to almost fresh water conditions (1.6 ppt) during MN to 30ppt during PRM. In Rushikullya estuary, it ranged from 1-2 ppt during MN to 30-34 ppt during PRM. In the estuarine mouth of Godavari, during MN season, the salinity was 12 ppt and during dry season it was 30 ppt. In Vellar estuary, Tamil Nadu the seasonal changes of salinity was 0.01 ppt (November) to 36.3 ppt (July). In Ashtamudi estuary during MN season the salinity was 0.12 ppt and PRM it was 35.05 ppt. In Kali estuary, Karnataka the salinity ranged from 1.33 ppt to 31.12 ppt. According to Martin et al. (2013) in Cochin estuary during PRM season, due to increased sea water intrusion, the lower reaches of the backwater behaved as an extension of Arabian sea with fairly high salinity ranges to 31-33 ppt. Compared to other stations in the present study, station 10, Aroor was closest to Arabian sea having maximum salinity of 33.1ppt. Joseph (1974) pointed out that in Cochin estuary salinity varied from time to time depending on the fresh water influence and penetration of sea water into the Cochin estuary. High value of salinity was observed during PRM and when fresh water discharges were minimum. A homogeneous condition was established until the outbreak of south west monsoon. He also observed that during MN period the estuary turned completely to fresh condition due to the enormous volume of discharges from the rivers.

In the present study, during southwest monsoon season the salinity declined nearly zero in all the stations (stations 1 to 10, during June, July, August, September months) was mainly due to the heavy rainfall and river discharge. The results obtained on the rainfall and river discharge value support to this condition. Earlier, Devassy and Gopinathan (1970), observed similar decreasing trend in salinity and fresh water condition during the monsoon months. With the onset of MN rain fall, the hydrographic condition of estuary undergoes considerable transformation with condition of fresh water dominating the area (Wyatt and Qasim, 1973). During MN season, the tidal incursion into the estuary was not apparent due to the increased fresh water runoff from rivers. So the sea water intrusion during flood tides only limited to the bottom layers. The salinity in the Kol wetland and a part of Vembanad north of TMB was observed to range between 10 – 15 ppt during 1990s. Inter basin transfer of water from Periyar to Muvattupuzha and discharge of water from tail races of hydropower projects have changed salinity gradient in the central part of estuary which presently tends towards freshwater conditions. The TMB plays a critical role influencing the salinity pattern in the estuary. Shivaprasad et al. (2013 b) noted that during the closure period of barrage, the freshwater runoff was reduced as flow from the four rivers was blocked by TMB and so salinity was increased towards the northern region of TMB. In the present study, the annual average salinity in southern region of the estuary was oligohaline whereas northern region was mesohaline in nature. During PRM period (1985-1986) the average salinity of surface and bottom water in the proximity of TMB, that was 10 ppt and 12 ppt respectively (Anirudhan et al., 1987). In TMB region the salinity decreased to 8.6ppt during the PRM season of 1980-1981 period (Gopalan et al., 1987). During the prebarrage period the salinity range of Punnamada area showed an average salinity of 8ppt during PRM, and nil values during both MN and PM periods (Haridas et al., 1973). In the present study, the average salinity in the Punnamada region during PRM season was 2.25 ppt, 0.05 ppt during MN and 0.25 ppt during PM season. During the PRM, the closure of TMB blocks the tidal flow of saline water into the southern sector and reduces the salinity into an oligohaline condition. During the closure period of barrage, the average surface water salinity was 2.16 ppt during 2011-2012 period (Asha et al. 2016), which was very less as compared to earlier studies by Haridas et al., 1973; Pillai et al., 1975; Jacob et al., 1987 from the same estuary. In the southern zone, the average salinity during the pre-barrgae phase was 18.47 ppt (Josanto, 1971), but reduced to an average of 2.8 ppt during the post-barrage phase (1994 to 1996) (Anon, 2001). At the same time, the average salinity in the southern zone reduced to 2.29 ppt during 2011-2013 period (Table 3.4). During prebarrage phase, Devassy and Gopinathan (1970) noted that in the PRM season, the intrusion of sea-water seems to extend throughout the backwater system and it progressively decreased from Cochin barmouth (31.72 ppt) to Punnamada (13.29 ppt) region. When comparing the salinity pattern during the closure period of TMB, the average salinity in the southern zone was 3.3 ppt and in the northern zone it was 17.98 ppt during 2011-2013 period (Fig. 3.29). According to Jacob et al. (1987) during the closure period of TMB the water level was low in Kuttanad region and this prevents the ingress of brackish water into the rivers and channels of the region. According to Carvalho et al. (2010), salinity fluctuation in Bacanga River Estuary, Brazil depended on the management and behavior of the dam gate that regulates the outflow of freshwater and the inflow of saltwater. Balakrishnan and Shynamma (1976) pointed out that pattern of salinity distribution in the Cochin backwater mainly in the Cochin harbour area did not follow the tidal rhythm, maximum salinity was observed much later than the attainment of high tide and low salinity much later than the low tide. Continuation of positive correlation of salinity towards temperature, pH, conductivity and TDS showed the interrelationship between these environmental parameters. In the coastal waters of southwest coast of India, similar correlation was observed between salinity and combination of the environmental parameters (Udaya Kumar et al., 2014). Mustapha et al. (2012) observed a linear relationship between conductivity and salinity in Jakara River Basin. During 2011-2012 period, salinity showed a significant positive correlation with pH (r^2= 0.43; $P \leq 0.01$), conductivity (r^2= 0.80; $P \leq 0.01$), TDS (r^2= 0.75; P < 0.01) and alkalinity (r^2= 0.4; $P \leq 0.01$). During 2012-2013 period, correlation analysis showed that salinity value has significant positive correlation with water temperature (r^2= 0.52; $P \leq 0.01$), pH (r^2= 0.41; $P \leq 0.01$), conductivity (r^2= 0.89; $P \leq 0.01$), TDS (r^2= 0.91; $P \leq 0.01$) and alkalinity (r^2= 0.6; $P \leq 0.01$).

The strong correlation between conductivity and salinity was due to the reduced fresh water discharge from the rivers which would act to increase the conductivity and salinity of a water body. Satheeshkumar and Khan (2012), observed a significant positive correlation between salinity and water temperature in the mangrove area along the Ariankuppam estuary, Tamil Nadu and point out the seasonal factor for its correlation. The significant positive correlation of pH, salinity and alkalinity was mainly due to the fact that increases in salinity related with the increase in pH and alkalinity.

Table 3.4 Time scale variation in salinity in the southern zone of Vembanad estuary

Reference cited	Salinity (ppt)
Josanto, 1971	23
KWBS, 1989	6
Anon, 2001	3.87
Padmakumar et al., 2002	4.31
Present study	2.29

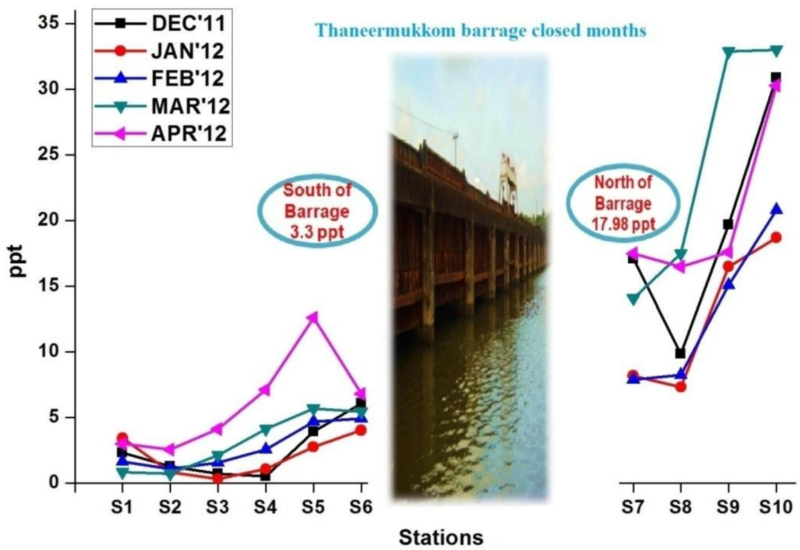

Fig. 3.29 Comparison of salinity pattern in the south and north of TMB during the closure period.

Rivers and estuaries play a key role in the transport and transformation of carbon from the continent to the adjacent coastal zone and naturally act as source of CO_2 to the atmosphere (Araujo et al., 2013). It is closely linked to the organic carbon pool through photosynthesis and respiration. This CO_2 is produced in high concentrations in estuarine regions which were known to be heterotrophic systems, where organic carbon transported by rivers is partly mineralized (Gattuso et al., 1998). Carbon dioxide is also a greenhouse gas and therefore its air-sea exchange is an important issue for climate change studies (Cai and Wang, 1998). According to Frankignoulle et al. (1998), estuaries are considered as sources of CO_2 to the atmosphere because of the presence of higher amount of detritus organic matter that cause intense respiration. Normally the uptake or release of carbon dioxide by organisms can change the proportion of carbonate and bicarbonate ions in water. Related with the affinity towards both carbon dioxide and bicarbonate level, the composition and species of submerged aquatic vegetation varies (Beer and Koch, 1996). It is certain that, CO_2 levels will keep on rising throughout the 21st century. Any variation in atmospheric CO_2 changes is likely to be followed by variations in surface water CO_2 causing lowering of pH and carbonate ion concentration, (CO_3^{2-}), which is estimated to affect $CaCO_3$ secreting organisms, including shellfish (Najjar et al., 2010). Calcareous bivalves, foraminiferan is sensitive to pH related developmental activities to build calcium carbonate shells and other structures. Therefore, CO_2 increases could dramatically alter calcification in these animals (Gazeau et al., 2007). The pH of water also depends upon relative contents of free CO_2, carbonates, bicarbonates and calcium. Similarly the alkalinity of water increases when it contains carbonates and it gets decreased along with increased bicarbonates, free CO_2 and calcium (George et al., 2012). During both the years, a distinct annual average value of dissolved carbon dioxide value was observed in Vembanad estuarine system in which higher value was observed in southern most and northern most stations. Socioeconomic development especially backwater tourism (house boat) along the southern zone have led to many illegal construction of tourism related amenities such as resorts, hotels, home stays and apartments on the banks of estuary and easily disposing off the sewages and waste- water either directly into the estuary creating more vulnerability in the southern zone. Similarly industrial as well as anthropogenic interventions into the northern estuarine station (Aroor) result in discharge of partially-treated and untreated wastewater into the fragile ecosystem. The rate of primary production, respiration and mineralization cause significant changes to the oxygen and carbon dioxide concentrations of the aquatic environments (Vasudevan Nayar, 1992). The southernmost stations (station 1 and 2) also exhibited high carbon dioxide values, where the pH seemed to be acidic in nature, indicating the direct correlation of bicarbonates and pH. Decomposition of organic matter, respiration and the acidic soil substratum influences the release of carbon dioxide in water. The range of dissolved carbon dioxide in the present investigation was similar to that of other tropical estuaries. But its seasonal variation was more conspicuous, where the range of CO_2 value was slightly higher (0.5 to 19 mg L^{-1}) in the estuary. In Cochin estuary, Radhika (2013) noted the dissolved carbon dioxide is in the range of 0.04 to 3.0 mg L^{-1}. In the Kodungaloor-Azhikode estuary the average dissolved carbon dioxide concentration ranges from 5.3 ± 1.8 to 7.1 ± 2.1 mg L^{-1} (Jayachandran et al., 2012). During the post barrage period (1994-1996 period), the average free CO_2 in the southern zone of Vembanad was 6.1 mg L^{-1} (Anon 2001), whereas during 2011-2012 period a slight increase was observed in the southern zone (6.5 ± 1.18 mg L^{-1}). During closure period of TMB, the average dissolved CO_2 in the southern zone was little higher than (6.2) the open period (5.8). The southernmost stations (1 and 2) also exhibited high carbon dioxide values, where the pH seemed to be acidic in nature, indicating the direct correlation of bicarbonates and pH. Decomposition of organic matter, respiration and the acidic soil subratum influences the release of carbon dioxide in water.

According to Bijoy Nandan (2008), retting of coconut husk in the coastal backwaters led to acidic pH conditions with anoxia resulting in the production of high amounts of dissolved carbon dioxide leading to drastic reduction in the incidence and abundance of plankton, benthic fauna and the fishery resources. In the retting zone, the mean value of dissolved carbon dioxide was 6.4 mg L^{-1}. Vembanad estuary is famous for its black clam (*Villorita cyprinoides*) fishery and large deposits of clam shells (white clam). Around 30,000 families in the region depended on clam shell fishery for their livelihood (Anon, 2014b). Any variation in the dissolved carbon dioxide level and pH in the estuary could adversely affect the growth and development of *V. cyprinoides.* During 2011-2012 period, CO_2 concentration showed a significant negative correlation with water temperature (r^2= 0.3; $P \leq 0.01$), turbidity (r^2= 0.3; $P \leq 0.01$), dissolved oxygen (r^2= 0.30; $P \leq 0.01$) and alkalinity (r^2= 0.3; $P \leq 0.01$). In contrast with first year, a significant positive correlation was existed between CO_2 and alkalinity (r^2= 0.31; $P \leq 0.01$). Both the years, relatively a weak correlation observed between CO_2 and pH (r^2= 0.2; $P \leq 0.05$).

The concentration of dissolved oxygen in an aquatic environment is an important indicator of the environment water quality. Physical and biological factors contributed to the seasonal reduction in the bottom water DO (Boynton and Kemp, 2000). The diversity of organism is directly dependent upon the DO concentrations. Depletion of DO in coastal and marine waters is a common phenomenon that appears to be growing globally and impacts the physiology and energetics of coastal ecosystems (Rabalais and Gilbert, 2009). The dynamics of dissolved oxygen level in Vembanad estuary is showing wide variations. No strict pattern of spatial or temporal variations of dissolved oxygen was observed during the present study. The DO concentration was increasing from morning to mid-afternoon where the rate of photosynthesis exceeds respiration. Similarly the photosynthesis rate decreases during late afternoon or evening and only respiration occurs during night. Most of the field sampling was completed before afternoon. In the current study, the average DO was slightly higher in Vembanad when comparing with other water bodies. The average DO in other estuaries of India observed were that in Vasishta Godavari estuary, it was 4.5 ml L^{-1} (Srinivasa Rao et al., 2009); that in tropical brackish water wetlands of West Bengal, 7 ml L^{-1} (Roy et al., 2013); that in Gautami–Godavari mangrove estuarine ecosystem of Andhra Pradesh, 5.26 mg L^{-1} (Tripathy et al., 2005); that in Dhamara estuary, Bay of Bengal, 4.78 mg L^{-1} (Mahapatro et al., 2011); that in Chilika lake, ~ 6 mg L^{-1} (Nayak et al., 2004); that in Rushikulya estuary, Orissa 4.64 mg L^{-1} (Paikaray et al., 2012) and 4.4 mg L^{-1} in Mandovi and Zuari estuaries of Goa (Harkantra and Rodrigues, 2004). This could be due to the fresh water runoff from the rivers of Pamba, Manimala, Achenkovil, Meenachil and Muvattupuzha as oxygen solubility is higher for fresh water. Higher DO during MN season was due to the increased precipitation and river discharge during the season. The decreased salinity and temperature were also linked to increased DO during MN. The lowest level of DO in station 2 during first year was mainly responsible for the proliferation of water plants in the surface layer for the period of sampling, which reduces the oxygen dissolution in water, light penetration and ultimately the photosynthesis. Similarly decomposition of sewage effluent and organic discharges from the house boats, along with nutrient rich runoff from agricultural lands situated close proximity to the station. In different time periods, the estuary gets clogged with water plants, *Eichhornia*, *Salvinia* and other aquatic weeds. This will reduce the rate of dissolution of oxygen in the water column; along with the decaying weeds also probably affect the DO level. Similarly the decreased DO level in station 10 was mainly augmented by the resuspended silt particles from the dredging activities in the Cochin navigational channel. According to Brown and Clark (1968), due to dredging the resuspension of oxidizable bottom sediments in a tidal waterway caused significant reduction in the dissolved oxygen concentration of water in Raritan Bay in

U.S, where during dredging, DO was reduced between 16-83% below normal. In the present investigation, the maximum average DO during PRM season during first year period was mainly attributed to the heavy precipitation occurred just prior to the field sampling in the PRM months. Whereas during second year higher DO value observed in MN, that may be due to cumulative effect of wind and tidal action from the Arabian Sea coupled with rainfall and resultant freshwater mixing (Das et al., 1997). Similar findings were observed in Kodugalloor-Azhikode region (Jayachandran et al., 2012). Seasonal southwest monsoon and increased river discharges enhance the flushing of bottom waters and irregular winds in summer season that helped to increase the dilution and interrupt the development of hypoxia in the bottom waters of the estuary. Similarly the monsoon rainfall also affected the trophic food web in Cochin estuarine system (Madhu et al., 2007). Considerable variation in DO has been reported in Indian estuaries from season to season and from one region to the other in the same estuary (Qasim, 2004). In Hoogly estuary, it varied from 4.79 - 4.92 ml L^{-1} (De et al., 1994); Rushikulya estuary DO fluctuated from 3 to 7.4 ml L^{-1}; in Godavary estuary it fluctuated from 2.6 to 8.1 ml L^{-1}; in Vellar estuary it was 3.9 to 7.62 ml L^{-1} (Qasim, 2004). Devassy and Gopinathan (1970) noted that the Vembanad estuary was having a DO of 4.50 to 5.0 ml $L^{-1}$and Martin et al. (2013) reported a range of 4.1 - 7 ml L^{-1} in Cochin estuary. The average DO in and around Vembanad estuary varied considerably. In the Vallanthakad estuary, it was 4.7 mg L^{-1} (Meera and Bijoy Nandan, 2010), 6.35 mg L^{-1} in Cochin estuary (DOECC, 2013); 6.35 mg L^{-1} in Kodungaliur-Azhikode estuary (KSCSTE, 2013), 5.06 mg L^{-1} Cochin estuary (Joseph and Ouseph 2010).

Comparing the open and closure period of TMB, dissolved oxygen in the closure period was always low in the southern zone. The thick layer of water hyacinth and the oil film from the house boat and other boats in the southern zone may prevent the dissolving of atmospheric oxygen into the water column. Similarly during the closure period, the organic wastes from animals and humans, and also crop debris, food wastes, rubber, plastic wastes are increasingly accumulated in the southern zone. Such wastes require more oxygen to decompose and which reducing the oxygen level in the southern zone. During the pre barrage phase Devassy and Gopinathan (1970) reported the range of DO value as 3.37 - 4.76 ml L^{-1} and 3.85–5 ml L^{-1} during premonsoon and monsoon respectively. During monsoon season Pillai et al. (1975) noted that the dissolved oxygen concentration ranged from above 1 ml L^{-1} and maximum of 6 ml L^{-1} in the pre barrage period. During the prebarrage period Devassy and Gopinathan (1970) noted that the in the Punnamada region, DO level was always high (7ml/l) and similar trend was not observed in the current study period. This may be due to the increased tourism activities and other urban development in the Punnamada region which could be affected the phytoplankton production. Similarly during 1994 to 1996 period, in the southern zone of barrage the DO ranged from 6.0 to 7.2 mg L^{-1} with an average of 6.25 mg L^{-1} at all the stations except Punnamada where municipal discharge waters brought down the dissolved oxygen to 3.16 mg L^{-1} during the pre monsoon (Anon, 2001). Houseboat tourism and related activities, sewage discharge and spread of invasive water plants play a crucial role in dissolved oxygen content of the southernmost stations of Vembanad estuary. Alterations of the natural hydrological regime have led to serious clogging of the channels of the Kuttanad area, which obviously alters the dissolved oxygen level of the area (Asha et al., 2016). Both the years, DO showed a significant negative correlation with salinity during first (r^2= 0.3; $P \leq 0.01$) and second year (r^2= 0.4; $P \leq 0.01$) period. Apart from salinity during second year, it showed a significant negative correlation with water temperature (r^2= 0.3; $P \leq 0.01$), conductivity (r^2= 0.3; $P \leq 0.01$) and TDS (r^2= 0.3; $P \leq 0.01$). Compared to first year, decreased level of spatial average of DO during second year in all stations possibly related with the increased level of water temperature, turbidity and salinity. Similar to present investigation, a strong negative correlation was observed between DO and salinity in Cochin estuary (Robin et al., 2012). In Bahuda Estuary, Bay of Bengal dissolved oxygen has negatively correlated with temperature and salinity (Behera and Padhy, 2012).

Biological Oxygen Demand is a measurement of the amount of dissolved oxygen (DO) that is utilized by aerobic microorganisms when decomposing organic matter in water. Permissible limit of BOD is 5 mg L^{-1} (WHO, 1984). Increased BOD level is attributed to the biodegradation of organic materials which exerts oxygen stress in the water (Gupta, 2009) and also gives an idea about the extent of pollution in the water body. Over the two years periods, water bodies assessed are considered to have an acceptable level of BOD. Both the years, the maximum level of BOD was present in station 10 (4.51 ± 0.98 mg L^{-1}) and station 1 (3.28 ± 0.83 mg L^{-1}) and station 8 (3.31 ± 1.61 mg L^{-1}). The organic rich effluents primarily from shellfish industries and sewage effluent from human settlements along the banks of estuary has enhanced the BOD level in station 10, Aroor. As a result of increased industrialization and urbanization in the Cochin area, a daily discharge of about 104 million litres of untreated effluents and 260 m^3 of raw sewage enter into the estuarine system (Qasim, 2003; Balachandran et al., 2005). Numerous open toilets on the banks of the Vembanad estuary especially on the northern zone have also increased the sewage load of whole Vembanad estuarine system and which might be increased the oxygen demand. In the Tapi estuary Borade et al. (2014), clearly noted a strong relationship between BOD and microbial populations and found that BOD level in the estuarine system was mainly controlled by the fecal indicator bacteria. Increased BOD level in station 1, Punnamada was mainly due to the enhanced organic load from house boat tourism activities which resulting the microbial oxidation of organic matter. The polluted and flow restricted Alappuzha canal carrying waste water from the town directly entering the Punnamada region of Vembanad estuary and these events enhances the BOD level in Punnamada station. Faecal contaminants from house boats, municipal garbage from Alappuzha town and organic pollutants from nearby agriculture fields leads to the proliferation of coliform in southern region (Asha et al., 2016) and the southern zone were more prone to waterborne diseases (Chandran et al., 2013). The desire for short term profit from house boat tourism caused environmental degradation of the system, even though the operators are understand that sustainability of the industry is connected to the environmental quality of the system (Karlaganis and Narayanan, 2014). The increased BOD in Varanadu region (station 8) was attributed to the effluent discharges from the distillery. Lekshmi and Jaya (2007) noted that after the treatment of effluent discharges from the distillery situated near Varanadu, there was 99% drop in BOD load and 98% drop in the nutrient level of effluent. But the water quality monitoring on the estuarine water near the discharge point in Varanadu revealed that the mean value of BOD from the estuarine water were comparable to that of effluent from the distillery. During 2011-2012 period, increased BOD level was observed during the PRM season. Similar situation was observed in Cochin estuary (Radhika, 2013). Increased temperature during the PRM season enhances the microbial activity and ultimately the BOD level (Thomas and Abdul 2000). When comparing the average BOD level in Vembanad estuarine system with different estuarine systems of India, it was noted that the average value was not much varied each other. The BOD level in Chilika lake, Orissa was ≤ 5 mg L^{-1} (Nayak et al., 2004), in Gautami–Godavari mangrove estuarine ecosystem of Andhra Pradesh, it was 3.99 mg L^{-1} (Tripathy et al., 2005), 3.1 mg L^{-1} in Vallanthakad backwater, Kerala (Meera and Bijoy Nandan, 2010), 2.55 mg L^{-1} in Kodungallur-Azhikode estuary (Jayachandran et al., 2013). During 2011-2012 period, BOD value showed a significant positive correlation with water temperature (r^2= 0.31; $P \leq 0.01$) whereas during 2012-2013 period, BOD showed positive correlation with DO (r^2= 0.33; $P \leq 0.01$). Similar observations were made from the Valanthakad backwater, Kerala (Meera and Bijoy Nandan, 2010).

Photosynthetic pigments are the index of primary production of an area and playing a significant role in the ecological characteristics of an ecosystem. Quantitative estimation of pigments in aquatic ecological studies can be used to estimate phytoplankton biomass. Among the phytoplankton pigments, chlorophyll *a* is the most active and universal pigment. Worldwide, chlorophyll *a* is used as a prime indicator given with information for assessing ecological integrity in estuarine and coastal systems (Borja et al., 2008b). A definite trend in the spatial and temporal distribution of chlorophyll *a* was not observed in the present study. Each water body has its own hydrographical features which determine the biological parameters and its production. Seasonal analysis showed that the average seasonal maximum was observed during PRM followed by PM and MN. Madhu et al. (2010) observed a mesotrophic condition in the Cochin estuary with a significant increase in chlorophyll *a* and primary production mainly by the growth of nanoplankton during PRM season. Chlorophyll *a* was higher during PM season in Mandovi-Zuari estuary (Devassy and Goes, 1989; Krishna Kumari et al., 2002). According to Senthil Kumaret al. (2008), chlorophyll *a* showed significant seasonal variations in a tropical mangrove estuary, southeast coast of India and its value varied from 6.5 µg L^{-1} to 12.92 µg L^{-1}. They also noted that, concentration of chlorophyll *a* was highest and lowest in wet and dry season respectively and found that high concentration of chlorophyll *a* during wet season was due to monsoonal runoff. Similarly the high concentration of chlorophyll *a* in the upper reaches of the mangrove estuary suggests the impact of sewage and effluent from aqua farms on phytoplankton biomass. In the present study, the lower chlorophyll *a* value observed during monsoon season is attributed to the increased river discharge and flushing rate as well as precipitation along with decreased salinity. According to Shivaprasad et al. (2013c), higher flushing during MN resulted in low chlorophyll concentration in surface water of Cochin estuary, where salinity was lower. During the dry season, the low discharge and the highest water residence conditions resulting in relatively higher biomass accumulation. Jyothibabu et al. (2006) noted that enhancement of chlorophyll *a*, in Cochin estuary was due to the enrichment of nutrients mainly from domestic and industrial actions. Earlier studies on chlorophyll *a* conducted in the Vembanad lake reported an annual range of 2 to 21 mg m^{-3} (Nair, 1975). According to Bijoy Nandan et al. (2014), the average chlorophyll *a* content in the Kodungalur-Azhikode estuary was higher (av. 5.3 ± 1.8 mg m^{-3}), as compared to inshore waters of Indian coast. In the present study, Vembanad estuary displayed higher chlorophyll *a* than Kodungalur-Azhikode estuary. During the eutrophication condition, the concentrations of chlorophyll *a* in the estuary of the Odra river were high and exceeded to 320 mg m^{-3} (Nedzarek and Torz, 2002). In the Mandovi estuary, Devassy and Goes (1989) observed a very high column value of 30.30 mg m^{-2} of chlorophyll *a*. Chlorophyll *a* value was varying in opened and closed period of TMB, where it was higher during the opened than in closed condition. Shivaprasad et al. (2013b) observed similar condition in the northern side of barrage. They noted that during the TMB open period, higher chlorophyll *a* was observed close to the north of barrage, which was mainly attributed by freshwater species due to increased river runoff. However, during the closure period the decreased fresh water discharge during summer season lowered the chlorophyll *a*. During first year, the chlorophyll *a* content was higher in southern zone than northern zone. Direct discharge of phosphorus containing fertilizers from agrarian zone in the southern sector and increased nutrient rich river runoff was favouring the phytoplankton community. The increased chlorophyll *a* value in the northern zone during second year may be supported by dominance of marine species. Similar type of observation was noted in Cochin estuary by Shivaprasad et al. (2013a). Comparing chlorophyll a production during the prebarrage period, it was always < 4 mg m^{-3} (Nair et al., 1975). Whereas during the current study the average chlorophyll a value was 7.72 ± 1.27 mg m^{-3} (2011-2012) and 8.8 ± 1.06 mg m^{-3} (2012-2013). Over the decades, increasing anthropogenic inputs of nutrients cause eutrophication, which can considerably increase estuarine phytoplankton production and biomass. During second year correlation analysis showed that chlorophyll-*a* value showed significant positive correlation with pH (r^2= 0.43; $P \leq 0.01$). Such type of correlation between pH and chlorophyll *a* concentrations was reported by El-Otify (2015), in Lake Nasser area, Egypt and noted that photosynthetic activities of phytoplankton play a major role in demonstrating such type of relationships.

Dissolved silica is a key nutrient for the aquatic ecosystem as it is required by diatoms to build their frustule made of particulate amorphous silica or biogenic silica (Martin-Jezequel et al., 2000). Silicate showed large fluctuations and its annual seasonal variations were irregular in each stations. During the study period, the increased spatial average of silicate concentration in station 1 could be due to the influence of discharges from Pamba and Achenkovil rivers and due to the terrestrial runoff from the padashekarams. The municipal discharges from the Alappuzha town may also be enhancing the silicate input. Tripathy et al. (2005) noted the high concentration of SiO_4-Si (142.0 µmol L^{-1}) concentration in the Gautami–Godavari estuarine system of Andhra Pradesh that indicates the impact of terrestrial runoff. There were several reports that high silicate value observed during MN months the major source of silicate is from the soil. Seasonal variation of SiO_4-Si showed that, during 2011-2012 period, maximum concentration was observed during MN whereas during 2012-2013 period, it was highest during PRM. The observation of increased concentration during MN agreed with the results of Balakrishnan Nair et al. (1984c) in Ashtamudi estuary. The increased river fall and land runoff in the MN season may become a major contributor of silicon (Sankaranarayanan and Qasim, 1969; Anirudhan and Nambisan, 1990; Bhattacharya et al., 2002; Renjith, 2006; Sundaray et al., 2006). The concentration of SiO_4-Si in the estuary is influenced by mixing of fresh water with seawater (Anirudhan et al., 1987) and the irregular summer showers during the month of April and May that affects the distribution pattern of silicate. The effect of local rains and its uneven intensity rain fall and associated runoffs altogether affects the spatio-temporal variation of nutrients in Vembanad estuary. There was a major shift in rainfall pattern during recent years as seasonal rainfall during the southwest monsoon was declining while increasing in PM season (Krishnakumar et al., 2009). Sankaranarayanan et al. (1984) reported maximum SiO_4-Si concentration in Cochin backwaters during south west monsoon. Madhu et al. (2010) also reported a two fold increase of SiO_4-Si during MN and PM season in Cochin estuary. According to Shivaprasad et al. (2013a), elevated levels of silicate (120.66 µmol L^{-1}) level in the Cochin estuary were found at the surface water indicating that freshwater runoff is the principal source of silicate inputs. This is substantiated by the fact that higher silicate concentrations in wet season were due to heavy runoff from the river input. During the present study, silicate showed a weak correlation with alkalinity during first year (r^2= 0.23; $P \leq 0.05$) and chlorophyll *a* during second year (r^2= 0.22; $P \leq 0.05$). Shivaprasad et al. (2013a) noted a negative relationship of salinity with silicate in Cochin estuary. Manna et al. (2010) observed a positive correlation of chlorophyll *a* with silicate in the Sundarbans estuary. These results indicated the influence of silicate in regulating the phytoplankton production in Vembanad estuary. Similar type of observation was seen in eutrophic estuary of Taiwan (Wu and Chou, 2003). The release of dissolved silicate from the regeneration of diatom frustules is also a temperature dependent process with the highest concentrations of dissolved silicate often observed in estuaries during the summer (Conley and Malone, 1992). Diatom production can be seasonally limited by dissolved silicate concentrations in a number of estuarine systems (Conley, 2000). In the Chesapeake Bay, diatom bloom during the spring was determined by dissolved silicate concentrations and its variation in the magnitude, collapsed the spring diatom bloom and shifted the phytoplankton communities with species not requiring dissolved silicate for growth. The high rates of regeneration and sediment release observed during summer for both dissolved inorganic phosphorus and dissolved silicate, brings high concentrations back into the overlying water, creating an internal source of nutrients. During the post barrage period (1994-1996 period), the average silicate in the southern zone of Vembanad was 5.1 mg L^{-1} (Anon 2001), whereas during the study period the silicate concentration was increased in the southern zone. The increased silicate concentration in the southern zone was mainly attributed to the large amount of silicate input from rivers such as Pamba, Manimala, Achenkovil and Meenachil, which may retained over the year into the Vembanad estuarine system. Increased amount of waste water inputs from agricultural runoff and industrial activities might be contributed a major share.

In the case of phosphorus, increasing anthropogenic events especially, agricultural activities including use of superphosphate fertilizers and urban expansion and the diffuse nature of its associated nonpoint source pollution from surface runoff and ground water contamination have become a major water quality problem. Phosphorus undergoes sedimentary cycle with no significant role in the gaseous phase so it tends to accumulate in wetland systems (Renjith, 2006). Spatial analysis of phosphorus showed that maximum concentration was observed in station 10 during both the years. Saline incursion, discharge from sea food industries, urban and municipal discharge from Cochin metro city makes this station rich in inorganic materials. Higher PO_4-P value in station 8 was might be due to the effluent discharged from the distillery unit situated on the banks of the estuary. Studies of Lekshmi and Jaya (2007), on the effluent from the distillery revealed that the conventional primary and secondary effluent treatment processes reduce the pollution potential of the effluent with certain parameters including nutrients, but it was not much effective for reducing the concentrations of sodium, potassium, oil and grease, total solids, total dissolved solids etc. According to Qasim and Gopinathan (1969), concentrations of all the nutrients changes with tidal actions. The phosphorus value increased during flood and decreased at ebb tide time, representing that the phosphorus accompanied with the incursion of sea water. Nair et al. (1975) reported higher values of PO_4-P in Vembanad lake during MN and PRM period representing about 15µg L^{-1}. Madhu et al. (2010) reported that in Cochin estuary the dissolved inorganic phosphorus (DIP) showed slight increase during the PM (av. 2.6 ±1.2 µmol L^{-1}) as compared to the PRM (av. 1.7 ± 0.8 µmol L^{-1}). In the present observation, PM season showed the maximum concentration followed by PRM. This is not agreement with the observations of Qasim (2004). According to Pomeroy et al. (1965), seasonal variations of phosphate concentration was connected to various processes like adsorption and desorption of phosphate and buffering action of sediments under different environmental situations. In estuaries, distribution of nutrients was mainly depending up on the marine influence and fresh water discharge. Qasim (2004), pointed out that during MN the increased rainfall and land runoff is associated with increased phosphorus runoff gets settled to the bottom, from where it is released to the surface. Over the years, the rainfall in June and July showed a decreasing trend and August, September, October and November months showed an increasing trend (Krishnakumar et al., 2009). So the seasonal dynamics of nutrient distribution was undergoing periodic shifts in the estuary. According to Shivaprasad et al. (2013a), the salt wedge circulation during high tides brought upwelled water to Cochin estuary and was evident by the bottom hypoxic, high chlorophyll a and nutrient- rich conditions. They also noted that higher concentrations of nutrients such as phosphate and nitrite at the bottom saline conditions. During first year, PO_4-P concentration showed a significant positive correlation with conductivity (r^2= 0.46; $P \leq 0.01$), salinity (r^2= 0.45; $P \leq 0.01$), and TDS (r^2= 0.54; $P \leq 0.01$). Tripathi et al.

(2014) observed high significant positive relationship with phosphate and TDS in River Ganga. Dissolved inorganic nutrients normally exhibited an inverse relationship with salinity (Senthilkumar et al., 2008; Pradhan et al., 2009; Meera and Bijoy Nandan, 2010). The correlation of phosphate with salinity indicates source of phosphorous is not fresh water origin. Similar type of observation has also been reported in Cochin estuary by Robin et al. (2012). According to Liu et al. (2009), in coastal and estuarine waters, sea water act as a major source of phosphate except those get freshwater containing domestic wastes including detergents as well as discharges of phosphate-phosphorous fertilizers and pesticides from agriculture field. In the present investigation both sea water and fresh water discharges act as a major phosphate contributor because the highest spatial average of phosphate was observed in the extreme south and northern stations.

Sujatha et al. (2009) reported higher levels of inorganic phosphate in Vembanad estuary as compared to Ashtamudi estuary. According to Jyothibabu et al. (2006), Cochin estuary receives large amount of nutrients from rivers, apart from this, increased supply of nutrients are mainly received from various domestic and industrial activities. The excessive input of inorganic phosphate fertilizers, detergents, fungicide etc in the Vembanad estuary has been reported from several studies (Remani et al., 1983; James, 2011). In Kuttanad agrarian zone the annual consumption of fertilizer was 8409 tonnes of nitrogen, 5044 tonnes of phosphorus and 6786 tonnes of potassium and the pesticides/fungicides/weedicides were about 500 tonnes/annum (James, 2011). The estuaries of Mandovi and Zuari reported a phosphorus concentration that ranged from 1-2.8 $\mu g\ L^{-1}$ and 1- 2.4 $\mu g\ L^{-1}$ respectively. Similarly, in estuaries of Mumbai, phosphorus concentration ranged between 2 and 9$\mu g\ L^{-1}$ and in Mahim estuary it was 6 to 46 $\mu g\ L^{-1}$ (Qasim, 2004). On comparing with the southern zone of the estuary, during 2011-2012 and 2012-2013 period, the northern zone showed maximum phosphate-phosphorus concentration. Comparing the prebarrage and post barrage phase the highest phosphate-phosphorus concentration was observed in the post barrage period. Temporal and seasonal variation of phosphate in the southern stations of Vembanad was dissimilar in both pre (Pillai et al., 1975) and post barrage period. During the prebarrage phase the highest value was observed during December 1971 in Punnamada region (12 µg L^{-1}) and in south of TMB highest of 18 µg L^{-1} was observed in July 1971. From the southern station, the maximum spatial average of phosphate (2.36 ± 1.22 µmol L^{-1}) was observed in Punnamada region. During the pre-barrage period, Nair et al. (1975) also reported very high concentration of phosphate in the southern region of the same estuary during MN and PM period and decreased concentration towards the lower reaches of the estuary and noted that land drainage and fresh water runoff lead to the wide variations in phosphate distribution. The re-suspension of phosphate from sediments also contributes a major portion of these nutrients to the estuarine water (Vidal, 1994).

In estuaries, the main sources of nutrients are river discharges containing load of urban wastewater, agriculture fertilizers and organic manure (Bricker et al., 2008). Nitrogen exists in a range of oxidation states in estuaries from ammonia up to nitrate which coexist with intermediate species such as nitrite and gaseous forms including nitrous oxide and nitrogen. Nitrite is the major component in the form of dissolved inorganic nitrogen followed by ammonia and nitrate in the process of nitrification. Many studies have been subjected to the behaviour of nitrogen within estuaries and coastal areas together with the nitrification and denitrification processes (Balls, 1992). In Cochin estuary runoff from agriculture, industrial discharge, aquaculture and domestic wastes are the major sources of nutrients (Sarala Devi et al., 1991b; Vijayan et al., 1976; Madhu et al., 2007). During first year, higher values for nitrite in station 4 may be supported by some nitrogenous compounds being added to the estuary from external source around the station. The water and effluent released from the clam processing units on the banks of the estuary at Aryad and Muhamma region lead to higher nitrogen condition in the estuary. The increased level of nitrite during second year in station 10 was mainly due to the increased discharges from Kochi metro city along with the effluent from shell fish industry situated along the estuarine bank at Aroor. There were a number of reclamation activities on the banks of Vembanad estuary mainly in the Aroor, Thevara region under the guise of industrial, tourism and urban developments has threatened the bathymetry of the region. The reclamation and economic activities in the Cochin estuary initiated with the development of the Cochin Port Trust in 1938, later with Vallarpadam International Container Transshipment Terminal, Kochi (2011), the Liquefied Natural Gas (LNG) regasification terminal in Puthuvype, Kochi (2013) and other establishments. Apart from this, the population of the area is expected to increase by 5×10^6 by 2025 (Balachandran et al., 2005) that also, signifies the need for environmental conservation plans to minimize the excessive pollution in the estuarine area. According to Thottathil et al. (2008), the higher nitrite concentration in the bottom water is related to increased oxygen demand and strong heterotrophic behavior. The rate of nitrification is controlled by temperature unless there is an insufficient supply of oxygen and ammonium. During first year, higher nitrite was observed during MN. According to Govindasamy et al. (2000), during MN season, increased phytoplankton excretion, oxidation of ammonia and reduction of nitrate and by recycling of nitrogen and bacterial decomposition of planktonic detritus in the system contributed the higher nitrite. During second year, the nitrite concentration was higher during PRM season. Similar condition was observed in Kodungaloor Azhikode estuary (Jayachandran et al., 2012). The lower freshwater input, higher salinity, pH and uptake by phytoplankton may also have contributed to the elevated level of nitrite concentration during PRM

(Radhika, 2013). Similar to phosphate-phosphorus, the nitrite-nitrogen were high in the post barrage period. During the pre-barrage phase a very low (< 0.6 μg L^{-1}) nitrite-nitrogen was observed in the southern zone of Vembanad estuary (Pillai et al., 1975). In Vemaband estuary, Nair et al. (1975) had reported an average nitrite level of 1μgL^{-1} and Pillai (1991), a value of 0.965 μgL^{-1}. In the effluent discharge point of Periyar river estuary (Northern most part of Vembanad estuary), Sarala Devi et al. (1991b) noted a nitrite concentration of 0.8 – 2.67 µmol L^{-1}. Over the years several investigators reported the eutrophic condition of Vembanad estuary (Balachandran, 2001; Robin et al., 2012; Martin et al., 2013). Nitrogen loading to the Vembanad estuary increased mainly due to extended aquaculture, agricultural practices and aqua-tourism (Martin et al., 2010). During closure period of TMB, the average nitrite concentration in the southern zone was little lower than (0.19 µmol L^{-1}) the open period (0.26 µmol L^{-1}). Similarly both the years, maximum average value of nitrite was observed in the northern zone. This indicates the source of nitrogen is from the coastal waters of Arabian Sea and this was confirmed from the correlation analysis. Decreased nitrite in southern zone was presumably, it got washed to the northern zone during increased river discharge. During first year correlation analysis of NO_2-N concentration showed positive correlation with silicate – silicon (r^2= 0.4; $P \leq 0.01$) and during second year, NO_2-N showed significant positive correlation between pH (r^2= 0.4; $P \leq 0.01$), conductivity (r^2= 0.67; $P \leq 0.01$), TDS (r^2= 0.7; $P \leq 0.01$), salinity (r^2= 0.62; $P \leq 0.01$), turbidity (r^2= 0.7; $P \leq 0.01$) and alkalinity (r^2= 0.48; $P \leq 0.01$). Correlation between NO_2 and SiO_4 can be assumed that these were mainly from same source such as land runoff and urban discharge, while there existed several nonpoint sources for NO_2 and silicate – silicon. Natesan et al. (2015) observed a positive correlation between nitrite and silicate during northeast monsoon and summer in the Kalpakkam coastal area, India. The positive correlation between pH and NO_2-N indicates that the increasing pH of water was connected with increasing nitrogenous materials. Temperature, salinity, substrate concentration, suspended particulate matter and pH influenced the nitrification process at one time or the other (Thasneem, 2016).

Nitrates are the most oxidized forms of nitrogen and the end product of the aerobic decomposition of organic nitrogenous matter. The cycling of N is more influenced by external input due to the open characteristic of the N cycle (Quiros, 2003). Natural waters in their unpolluted state contain only minute quantities of nitrates (Manikannan et al., 2011). The increased spatial average of nitrate in station 10 during 2011 -2012 period (2.03 ± 0.95 µmol L^{-1}) and station 9 during 2012 -2013 period (3.13 ± 4.98 µmol L^{-1}) indicating a nitrate input from either sea water or effluent and sewage discharges from the residential and urban areas situated close proximity to the stations. The increased concentration of nitrate during monsoon months (June 2012) in station 9 (18.59 µmol L^{-1}) was mainly contributed by the river inflow from Muvattupuzha river. Balchand (1983), noted that after receiving effluent from paper mill located on the Muvattupuzha river bank, the water quality of the river was considerably polluted in its downstream region on receiving the discharges, the nitrate concentration in river water increased to a range from 0.1 - 1 to 6 -10 µg at/l. The lower levels of nitrate observed in station 5 and 6 were due to the variation in freshwater input, salinity, pH and also their utilization by phytoplankton. The increased nitrate concentration in station 8 during January 2011, was corroborated with the effluent discharge from the distillery located on the estuarine bank. During the sampling period, the estuarine surroundings also had odour of alcohol as part of distillery activities. Increased concentration of nitrate in the northern most stations authenticated the influence of seawater intrusion from Arabian sea and also from urban discharges. In the present investigation nitrate concentration was maximum during MN season. Similar type of observation was noted by Santhanam and Perumal (2003), in Parangipettai coastal waters, Southeast coast of India. According to Vijayakumar et al. (2009), in an estuarine zone, increased nitrate concentration during MN season was due to freshwater influx. Similarly Parab et al. (2013) noted that increased land runoff during the monsoon brought nitrate into the Mandovi estuary and observed an addition and dilution of nitrate in the MN season which was responsible for its elevated levels. Earlier studies in Cochin backwater revealed that the nitrate values were very low than those of nitrite and noted relatively higher concentration of inorganic nutrients during MN and low concentration during the PRM, when the system remained predominantly on a marine phase (Sankaranarayanan and Qasim, 1969). Similar situation was also observed in the present study. During closure period of TMB, the average NO_3-N in the southern zone was little lower than (1.12 µmol L^{-1}) the open period (1.49 µmol L^{-1}). NO_3-N concentration showed a significant negative correlation with water temperature (r^2= - 0.4; $P \leq 0.01$) and a positive correlation with ammonia – nitrogen (r^2= 0.3; $P \leq 0.01$). Thasneem (2016), observed a significant negative correlation between nitrate and water temperature. Water column nitrification was controlled by temperature (Eyre and Balls, 1999) and ammonium concentration (Berounsky and Nixon,1993).

Ammonia is a product of excretion and decay, it is released during the breakdown of dead organic material, and it is also produced during the denitrification which converts oxidised nitrogen species eg. nitrate, nitrite ultimately to nitrogen. In estuaries, nitrogen compounds exert a significant oxygen demand through microbially mediated nitrogen conversion (Grabemann et al., 1990). Organic nitrogen is hydrolyzed to ammonia and the excess ammonia enters the estuaries via the tributary of rivers and also as wastewater discharges. Ammonia utilizes dissolved oxygen during nitrification to produce nitrate, via nitrite (Owens, 1986). The increased concentration of ammonia in station 1 has considerably been impacted by house boat tourism activities. This area is the hub of house boat tourism activities, in which more than thousands of house boats are plying in the region. The untreated wastes from the house boats are directly dumbed into the water body. Similarly the polluted Alappuzha canals with restricted flow, carrying waste water from the town directly enter the Punnamada region of Vembanad estuary which also enhances the ammonia-nitrogen in the estuary. Occurrence of ammonia indicates the presence of sewage in the water body. Apart from this, sewage load from Sabarimala through Pamba and Manimala also contibute a major share of ammonia in the water body. The usage of NPK fertilzers and Factamphos (20N:20P) from the padashekarams (Q, S, T, R and C blocks may get leached out during mosoonal rains to these southern stations can also contribute high ammonia concentration. There were several reports indicating the sewage pollution with increasing faecal coliforms, along with unobstructed tourism in conjunction with operation of thousands of house boats in the estuary and also by the waste disposal and oil pollution causing severe ecological implications in the southern zone of Vembanad estuary (Hatha et al., 2008; Anon, 2012a; Safoora Beevi and Devadas, 2014). The shoddy operation of sewage treatment plants by District Tourism Promotion Council, Govt. of Kerala for processing the sewage and other effluents from the house boats and other domestic sources has consistently affected the water quality of the southern zone (Asha et al., 2016).

In the southern zone, ammonia concentration was higher during the closure period of TMB than the open period. During the closure period it was 8.92 μmol L^{-1} and 6.38 μmol L^{-1} for open period. During the closure period of TMB, the increased levelof organic wastes and its decomposition, increased pH enhances the ammonia level in the water column and may contribute to non-point pollution and its excessive increase cause fish kill (Pillai, 1991). According to Joseph and Ouseph (2010), the ammonia content in Cochin estuary was contributed by the Periyar River and its associated tributaries including Chitrapuzha, which flow through the industrial zones. On a time scale the nitrate and phosphate in Vembanad estuary were lower in early 1970s but with it increasing industrial and agriculture activities the concentration was found to increase (Martin et al., 2013). During 1965, the surface phosphate and nitrate was 0.75 and 2.0 µM, which increased to 2.9 and 6 µM respectively by 2000. The overall trend shows a prominent increase of nitrate and phosphate after 1975; then from 1980 onwards, the concentration remained high (Balachandran, 2001). It is important to note that, this comparison is based on available data from the lower reaches of the backwaters as several researchers have sampled this region since 1965. The increased concentration of total inorganic nitrogen in Punnamada was due to the unregulated sewage discharge from resorts, houseboats and exhaust from motor boats which forms an important source of nitrogen. In Cochin estuary, Madhu et al. (2010) noted the seasonal variation of total DIN that showed high values during both PM (av. 45.2 ± 28.2 µM) and PRM (av. 53.5 ± 36.9 µM) and among these, ammonia contributed 68.3 % (av. 35.8 ± 34.5 µM) and 70.2 % (av. 30.8 ± 28.3µM) during the PM and the PRM respectively. Nitrogen fixation rates in DIN-enriched waters are generally expected to be lower because high ammonium ion (NH_4^+) concentrations (>1 µM) can suppress the nitrogenase activity (Mulholland and Capone, 2001). Bhavya et al. (2015) observed an elevated level of DIN uptake rates and confirmed the anthropogenic effects on nitrogen uptake dynamics in the Cochin estuary. The inverse relationship of NH_4^+ uptake and N_2 fixation rates with total nitrogen: total phosphorus (TN:TP) means that the nitrogen assimilation is associated more on the relative balance of nitrogen and phosphorus rather than distinct concentration of nitrogen or phosphorus. During 2011-2012 period, correlation analysis of ammonia – nitrogen showed a significant negative correlation with water temperature (r^2= - 0.4; $P \leq 0.01$) and positive correlation with CO_2 (r^2= 0.55; $P \leq 0.01$).

During 2012-2013 period, ammonia – nitrogen showed a significant positive correlation with alkalinity (r^2= 0.51; $P \leq 0.01$), CO_2 (r^2= 0.49; $P \leq 0.01$) and chlorophyll *a* (r^2= 0.47; $P \leq 0.01$). Positive correlation of ammonia and chlorophyll *a* indicates the contribution of phytoplankton proliferation (George et al., 2012) and increasing its biomass (Udaya Kumar et al., 2014).

Estimation of the trophic state of a system depends upon the balance of inorganic nutrients from various sources and their transport through assimilation, sedimentation and export (Herrera-Silveira et al., 2002). N:P ratio is one of the important components for the calculation of the trophic state index of estuaries. Several studies denoted that N : P ratio ≤ 10:1 that appears to support algal blooms, especially of blue-green algae (Sigua et al., 2000). According to Tilman et al. (1996) decrease in the ratio of N: P results in primary production shifting from primarily diatoms and other small edible algae, towards larger cyanobacteria that are stronger competitors for nitrogen. In the present study a nitrogen limiting condition prevailed in the estuary (10.96 ± 13.57 for 2011-2012 period and 6.47 ± 4.3 for 2012-2013 period), which was affected by some phytoplankton groups. Low N/P ratio denoted that DIN limits photosynthesis by phytoplankton and it continues throughout the year (Buranaprather, 2002) and also specifies the surplus loading of phosphorus into the estuary (Asha et al., 2016). Schindler (2008), indicated that a reduced N input into lakes increasingly favors the N_2-fixing activity of cyanobacteria as a response by the phytoplankton community to seasonal N limitations. Similar trend was observed in Vembanad estuarine system, where abundance of cyanobacteria increased in the southern part of estuary during 2011-2012 period. During the closure period of TMB support the condition for cyanobacterial proliferations with low nitrogen to phosphorus ratio, increased turbidity, increased retention time and low flow of water due to reduced tidal flushing in the southern zone (Asha et al., 2016). The barrage in the Fitzroy river estuary, Australia prevents any tidal movement of saline water into the upstream freshwater region and also let the flood water flow towards downstream (Currie and Small, 2002). High nutrient concentration in Vembanad was due to the continuous inflow of phosphate from the neighboring crop fields (agricultural fertilizers) and municipal wastes, which in turn enhances the growth of phytoplankton and other macrophytes in the lakes. Elmaci (2009), indicated that Uluabat lake, a Ramsar site in Turkey is a eutrophic system, based on the high load of phosphate. Closer to Redfield ratio (16:1) during PM of 2011-2012 period (15.61 ± 19.33), suggest that nutrients are not limiting factors for production during this season. According to Seitzinger (1988), chemical and biological processes for N and P uptake are different. After sedimentation, most of the N load is removed by denitrification, whereas a major fraction of P is reversibly bound to sediments and become available again after its concentration has decreased in water. According to Madhu et al. (2010), the variation in N : P ratio (6.8-262) in Cochin backwater was due to the irregular contribution of ammonia from the river Periyar. Similarly during PRM season, a mesotrophic condition established in the estuary mainly through a significant increase in chlorophyll *a* and primary production associated with the growth of nanoplankton (Madhu et al., 2010). Jyothibabu et al. (2006) reported that enhancement of Chlorophyll *a* in Cochin estuary was due to the enrichment of nutrients mainly from domestic and industrial actions. Direct discharges of phosphatic fertilizers from agrarian zone of Kuttanad and nutrient enriched wastes from domestic sewages and other pollutants, was favouring the phytoplankton community. The long water residence time during the closure period of Thaneermukkom barrage enhances the tendency towards eutrophication. According to Bakan (2010), intensification of agriculture in the last decades is considered as a most important non-point source of water pollution. It was mainly due to the nutrients such as nitrogen and phosphorus which are transported from fertilized agricultural lands to surface waters via runoff and erosion that accelerate the eutrophication process. The most important source of the nitrogen is biological oxidation of organic nitrogenous substances, which derived from sewage and industrial waste or produced indigenously in the water (Sharma et al., 2008).

TRIX index denote the eutrophication condition and give information regarding over a wide range of trophic situations. The TRIX value showed that the Vembanad estuarine system is under stress and also identified its upper zone and lower zone was the areas most affected by eutrophication. In Kodungallur-Azhikode estuary, south west coast of India is also experiencing a high degree of eutrophication with an annual mean value of 6.91 TRIX value. Premonsoon season showed the maximum TRIX value of 7.15 and minimum value of 6.51during PM period (Jayachandran and Bijoy Nandan, 2012). In the southern zone of TMB, the average TRIX value was little higher in the open period (6.7) than the closure period (6.4) and its variations were linked to the nutrient distribution. In the southern zone, both nitrite-nitrogen and nitrate-nitrogen were higher during the open period of TMB. Nutrient enrichment of coastal areas may have far reaching distresses, such as fish kill, exclusion of shellfish aquaculture, loss or degradation of sea grass beds and overwhelming of bivalves and other benthic organisms (Joint et al., 1997; McGlathery, 2001). Kochi city alone generates 2,550 million L/day of urban sewage that directly enters Vembanad estuary largely untreated (WISA, 2013). The total dissolved solid content of water was as high as 53,750 mg L^{-1} during summer that gets reduced to 160 mg L^{-1} during rainy season. In Kochi, the existing sewage facilities process water from only 1% of the population. A septic tank sewage system is used in most of the region, large number of toilets located on the banks of the estuary lead to direct faecal contamination. Wastes and sewage collection system of Alappuzha municipality directly releases the particulate organic matter containing wastes into the estuary (WISA, 2013).

Backwater tourism considered as a backbone of the state major revenue and house boat tourism has brought flourishing trend in Vembanad estuarine system and it negatively affects the estuarine system in many ways. According to Rajan (2008), the tourism industry illegally encroaches many regions of Vembanad estuary. Several thousand house boats are operating in the estuary and its unregulated operation, altering the natural estuarine characteristics (Safoora Bevi and Devadas, 2014). The sewage discharges from resorts, houseboats and exhaust from motor boats form an important source of chemical pollution. The waves generated by motorboat propellers increases phosphorus on average by 28%–55% (Yousef et al., 1980) and also sediment re-suspension, higher turbidity and larger total phosphate concentrations tended to be greater in shallower lakes than deeper lakes (Asplund and Cook, 1997). Fuel leakage and emissions affect water quality by adding metals, hydrocarbons and other pollutants to the water. All these accelerate the eutrophication process in Vembanad estuary. During 2011-2012 period, TRIX index showed a significant positive correlation with phosphate – phosphorus (r^2= 0.45; $P \leq 0.01$), ammonia – nitrogen (r^2= 0.53; $P \leq 0.01$) and nitrate r^2= 0.47; $P \leq 0.01$) and total nitrogen (r^2= 0.56; $P \leq 0.01$). During 2012-2013 period, TRIX index showed a significant positive correlation with phosphate– phosphorus (r^2= 0.51; $P \leq 0.01$), chlorophyll *a* (r^2= 0.57; $P \leq 0.01$), ammonia – nitrogen (r^2= 0.54; $P \leq 0.01$) and total nitrogen (r^2= 0.56; $P \leq 0.01$).

Euclidean distance usually gives the resemblance between two samples, and a distance can be denoted by the variation between analytical values from the samples (Otto, 1998). During 2011- 2012 period, cluster analysis of physico-chemical characteristics of Vembanad estuary represented that there was no significant spatial variation between stations 2 to 6. A significant spatial difference was observed in station 1 and it stands separated from other southern stations. The solid line represented the significant differences between clusters and dotted line reflects no statistical significance for the subgroup within the cluster. Station 9 and 10 spatially separated from other stations, which were mainly due to the wide salinity variations of the region. The close proximity towards the Arabian Sea and influence of tidal water makes it a distinct water quality condition in the estuary. Similarly, the salinity pattern of the region varied from an oligohaline during MN to a polyhaline condition during PRM season. During 2012-2013, the cluster analysis revealed that, there was a clear spatial variation of physico-chemical parameters between southern zone of TMB as well as the northern zone. In which the northern most station, station 10 (Aroor) exhibited a significant spatial variation. Salinity variation may be a prime factor in controlling the spatial variation of environmental condition in Vembanad estuary. During the closure period of TMB (December – May), there was a complete absence of flushing in the upper reaches causing serious ecological degradation in the southern zone of Vembanad estuary (Padmakumar et al., 2002). The unscientific operation and the duration of opening and closing of the barrage always created conflict between fisher folk and farmers (Asha et al., 2014). Bijoy Nandan and Unnithan (2004) informed the necessity for maintaining the continuity of the estuary on both sides of the TMB for reducing the stress on the environment by natural tidal flushing of the system.

Principal Component Analysis (PCA) was carried in order to find out the environmental parameters, which was influencing the study area. The first year result of PCA ordination using physico-chemical parameters gives two factors explained by 87 % of the total variance and second year it was 81%. The strong loading of alkalinity, salinity, TDS, turbidity and pH in the first component indicated the influx of marine water into the estuary from Arabian sea. The high loading of nutrients such as phosphate, nitrate and nitrite was mainly due to the release of nutrients from municipal waste from the surrounding Cochin metro city and also through river discharges from six major rivers. Similarly increased tourism activities in the southern zone and northern region (Cochin estuary) also imparted high loading of all these factors. Negative loading of DO observed in the PCA analysis might be due to the consumption of large amounts of oxygen by the organic matter. During the last 50 years, the effluent discharge from Cochin Industrial City has increased from a negligible level to 6.5 millionm^3/day (Gopalan, 2002). The indiscriminate use of fertilizers, insecticides, fungicides and municipal waste causes considerable damages to the water quality of riverine, estuarine and marine systems (Ouseph et al., 2004).). In Kuttanad agrarian zone the annual consumption of fertilizer was 8409 tonnes of nitrogen, 5044 tonnes of phosphorus and 6786 tonnes of potassium and the pesticides/ fungicides/ weedicides were about 500 tonnes/annum (James, 2011). In addition, increased organic and inorganic load from house boats into the estuary during the peak tourism period was also increasing the inorganic nutrients. Closure period of TMB maintain the condition for increased turbidity, longer water residence time, proliferation of invasive water plants, increased water column stability and low flow of water due to reduced tidal flushing in the southern zone that enhanced the rate of eutrophication to a certain extent (Asha et al., 2016).

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Chapter 4

SEDIMENT CHARACTERISTICS OF THE VEMBANAD TROPHIC SYSTEM

Contents

4.1 Introduction

Sediments are extensively studied all around the world, because it acts as sink and source of contaminants in aquatic systems and for their variable physical and chemical properties. Sediment particles reach rivers, lakes, streams, estuaries and ocean through wind, water and ice. In estuaries, sediments are brought in through river inflow and tidal currents and also from various physio-chemical and biological activities within the estuaries. According to Dethier (1990), the primary physical parameters influencing the distribution of organisms in estuaries are sediment composition, salinity and elevation. Any variation in sediment property affects the quality and cause habitat variation, ultimately leading to the alteration of biological communities. Understanding the textural characteristics of estuarine sediments is important for distinguishing different depositional microenvironments (Ganesh et al., 2013). Sediment transport and its depositional patterns are affected by the interaction of fresh and saline waters in estuaries. Fine sediments are flocculated due to saline water and vary the sediment transport pattern (Baker, 1978). Capacity of sediment loading within an estuary is related to the sedimentation rate and the energy available for transport. Weathering and erosion of continental areas act as a major source of sediment in coastal areas. Sediments are mainly transported by rivers in the form of suspended load and most of the aquatic system especially river system is facing flow restriction due to the construction of dams and reservoirs. It acts as an obstacle for the movement of sediment and forms permanent sediment traps.

Apart from providing a habitat for many organisms, sediments are significant because many toxic substances which are found in trace amounts in water are found in elevated levels in them. As such, sediments serve both as reservoirs and as potential sources of contaminants to the water column. In addition to their potential to degrade surface water quality, sediment-associated contaminants have the potential to affect benthic and other sediment-associated organisms directly (Chapman, 1989). Sediment contaminants are transferred to the water column through physical disturbance, diffusion, and biological activities. Therefore, sediment quality data provide essential information of ambient environmental conditions in aquatic ecosystems. According to Sarala Devi et al. (1992), the organic carbon content in the sediment of the estuarine and riverine system forms a potential source of food for the benthic fauna. The major portion of organic matter settles down the water column and eventually gets preserved in the sediments (Hu et al., 2006). Due to changes in the land use and land cover pattern, the excessive nutrient input and increased sedimentation adversely affects the aquatic ecosystem (Vitousek et al., 1997) by bringing changes in the carbon storage and nutrient retention ability (Morris and Bradley, 1999). Understanding the chemical characteristics of the estuarine sediment is important for assessing the water quality and management of the ecosystem. Similarly knowledge of the sediment – nutrient interactions of estuarine environment is valuable in assessing the productivity of the system (Balakrishnan Nair et al., 1984b). The quality of organic matter in sediments is critical to the partitioning and bioavailability of sediment associated contaminants. Influence of total organic matter present in sediments is important to assess the fate, transport, deposition and retention of various inorganic and organic constituents in estuaries (Akhil et al., 2013).

Normally oxygen in sediment is mainly restricted to the upper few centimeters. The rate of oxygen penetration and diffusion from the overlying water column is accelerated by the burrowing action of benthos. Similarly the bioturbating benthic infauna alters the physical and chemical structure of sediments and involve in the interactive process between water column and sediment. As sediment provides a habitation for benthic organisms it forms the primary source of contaminant exposure to benthos and also other organisms which feed on benthos. This exposure can produce adverse impacts on benthic communities and can also lead to indirect effects on wildlife and human health due to the accumulation of contaminants in the food chain. Sediment quality criteria is critical with respect to dredging activities related to navigation, water quality protection connected to aquatic and fisheries management, channelization of water ways and recreational aspects and other restoration programmes. Knowledge of sediment conditions and the spatial distribution of various parameters could hold the key to understand–water purification processes and the magnitude and direction of sediment transport (Huzarska, 2013). So sediment quality data give necessary information for assessing the environmental status of an aquatic ecosystem and is widely used for estimating the level of pollution also (Caccia et al., 2003).

4.1.1 Significance of the study

Sediment is a complex matrix of components and forms. Sediment quality has an important influence on the overall status of an aquatic body. It typically directs the nutrient stochiometry of an aquatic ecosystem and so the information on the characteristics of sediment nutrient is valuable in understanding the sediment-water interaction, which directly influences the productivity pattern. Due to increased urbanization, the deterioration of estuary can be attributed to huge quantity of domestic sewage and industrial effluent released directly into the estuary. Reclamation and construction activities also obstruct natural flow of river, tidal flow and affect flushing. Local variations in the granulometry or substratum modification of the bottom sediment caused by changes in various natural or human interactions make long-term effect on benthic communities. Similarly, the trace metal concentration in sediments vary greatly with sediment characteristics like grain-size, organic carbon, pH, dissolved oxygen, salinity, sediment water content and anthropogenic inputs etc. So knowledge on the sediment features and other environmental characters are essential for the better management of estuaries. The current study gives a renewed insight on the sediment quality of Vembanad estuary and its trophic characteristics for its long term management.

4.1.2 Review of Literature

The physical and chemical characteristics of bottom sediments in James River estuary, Virginia were investigated by Moncure and Nichols (1968). Henriksen and Michael Kemp (1988) explained the nitrification process in estuarine sediment. Spatial and temporal variations of suspended and sedimented nutrients in the eutrophicated Gulf of Riga (Baltic Sea) were investigated by Reigstad et al. (1999) and noted that the major supply of organic matter to the benthos is determined by the intensity of phyoplankton bloom. Temporal changes in grain size and organic-mineral aggregates in surficial sediments near the Massachusetts Bay outfall site was investigated by Rendigs and Bothner (2001). Kamaruzzaman (2002) analysed the grain size of surface sediment samples to understand the sedimentation processes in the Kemaman River estuarine system, Malaysia. The study noted that the characteristics of deposited sediments dependent upon the combination of physical forces such as freshwater runoffs, tidal currents and waves. Quantitative measures of sedimentation in an estuarine system and its relationship with intertidal soft-sediment infauna was studied by Anderson et al. (2004). Geochemical variations in estuarine sediments of Odiel estuary (southwest of Spain) were investigated by Lopez-Gonzalez et al. (2006). Marchand et al. (2006) investigated the heavy metals distribution in mangrove sediments along the mobile coastline of French Guiana. Lesen (2006) investigated the seasonal variation of benthic organic matter and chlorophyll in channel sediments and noted the relationship between seasonal patterns in benthic organic matter and water column chlorophyll in San Francisco Bay, California. The study revealed that sediment organic matter levels in the San Francisco Bay channel show seasonal cycles that follow patterns of water column production. Peaks in water column chlorophyll are generally followed by later peaks in sediment chlorophyll and organic matter. Most of the benthic studies were associated with the bottom fauna and its relation between physico-chemical parameters in sediments of lakes, rivers, estuaries and marine environment. The structure of the benthic macrofaunal assemblages of the estuarine portion of Paraguacu estuarine system, NE Brazil, and its relationship with surface sediment characteristics such as trace metals, Polycyclic Aromatic Hydrocarbons, nutrients and grain size and physical variables were investigated by Barros et al. (2008). Jaeger et al. (2009) studied the anthropogenic impacts on sedimentary sources and processes in a small urbanized subtropical estuary, Florida. Rodil et al. (2009) conducted a study to determine the relationship between macrobenthic assemblages, sediment type and food availability in Ria of Vigo, a partially stratified estuary on the NW coast of the Iberian Peninsula. The study showed that biopolymeric carbon fraction (BPC) and lipids form the important component of sediment nutritional value, demonstrating the importance of food quality as a structuring factor in macrobenthic communities. The study suggested the use of BPC and lipids as an effective tool in the study of soft bottom benthic ecosystem dynamics. Surface elevation changes and sediment characteristics of intertidal surfaces of Waikaraka estuary, New Zealand was investigated by Stokes et al. (2009). Specchiulli et al. (2010) analyzed the sediment characteristics and macrobenthic assemblages of two Italian coastal lagoons (Lesina and Varano) to evaluate the different relationship between sediment matrix and benthic organisms within and between the two lagoons. It demonstrated that variation in marine and fresh water input and sediment properties especially sand and clay components play a significant role. Season and tide effects on sediment characteristics of trans Okpoka Creek, Bonny estuary of Nigeria was studied by Davies and Tawari (2010). Venturini et al. (2012) gave information on biochemical characteristics and trophic state of the sediments in the Rio de la Plata estuary, South America. Oyedotun et al. (2013) focused on the sedimentological characteristics of surface and shallow intertidal sediments within the Camel estuary system, England. Particle size characterization of a closed estuarine lagoon of Central California was studied by Watson et al. (2013). Ali et al. (2013) investigated the sediment dynamics of a very shallow subtropical estuarine system on the Coombabah Lake, southern Moreton Bay, Australia. The study identified that sediment dynamics were subjected to advection processes due to the influence of tides and with wind and wave forces playing minor roles. Mitchell and Uncles (2013) studied the estuarine sediments and identified the key challenges in estuarine management in terms of the issues related to safe and practical solutions for redistribution or disposal of sediments in macrotidal estuaries and tidal waters. Mil Homens et al. (2014) investigated the major factors affecting the elemental composition of surface sediments of estuaries in Portugal. The study noted that the hydrological regime and estuarine morphologies along with lithological influences and anthropogenic effects influence the textural and chemical composition of sediments. Percuoco et al. (2015) examined the release of nutrients from estuarine sediments by diffusion and resuspension process in a temperate estuary (Great Bay) on the northeast coast of the United States. The study found that during the summer season, sediments form an important source for dissolved inorganic nitrogen to the primary producers along with the nutrient loading from rivers, which accelerates eutrophication.

Many studies have been carried out so far on sediment quality and nutrients in various estuaries and shelf waters of India. The grain size variation of the Vasishta Godavari river sediments were studied by Dora and Borreswara Rao, (1970). Sivakumar et al. (1983) studied the seasonal variations in sediment nutrients of Vellar estuary. Sesamal et al. (1986) analyzed the sediment texture and composition of Hoogly estuary and near shore regions. Mohan (1990), worked on the sediment texture, mineralogy and geochemistry of modern sediments of Vellar estuary. Kumar et al. (2003) investigated the heavy metals in the sediment and macrobenthos of the coastal waters off Mangalore. Manjappa et al. (2003) documented the characteristics of sediments in the mangrove areas of brackish water impoundments along the Nethravathi estuary, Mangalore. Organic carbon in the sediments of Mandovi estuary, Goa was studied by Alagarsamy (1991). Joseph et al. (1992) worked on the sediment characteristics of the estuaries around Madras in relation to environmental degradation. The study revealed that environmental deterioration was evident from various sediment characteristics in which higher organic matter was observed in polluted estuaries than non polluted ones. Sediment characteristics and benthic biomass in relation to growth of *Penaeus monodon* in low saline small ponds of Orissa was studied by Das et al. (2001). Moorthy et al. (2005) investigated the sediment characteristics of Madhurantakam lake, Tamil Nadu. Bragadeeswaran et al. (2007) studied the sediment texture and nutrients of Arasalar estuary, Karaikkal, south-east coast of India. Ranjan et al. (2008) assessed the metal enrichments in tsunami -genic sediments of Pichavaram mangroves, southeast coast of India. Prusty et al. (2010) analysed the macronutrients along the sediment of monsoonal wetland in India. The study found that the nutrient content in the sediment appeared to be a cumulative function of plant uptake, detritus decomposition, period of inundation, evapotranspiration and ambient temperature. Similarly temporal distribution of sediment organic matter and other nutrient element in the wetland system signifies that input water brought nutrients and sediments along with allochthonous materials. Shaikh and Tiwari (2012) conducted a study to assess the physico-chemical parameters of sediments of open mudflats of Sewri located along the Arabian Sea. Ramya et al. (2013) observed the biogeochemistry of the shelf sediments of south eastern Arabian Sea and its relation with the effect of benthic bacterial heterotrophs. Distribution and sources of sedimentary organic matter in Cochin estuary were studied by Gireeshkumar et al. (2013). Anitha and Kumar (2013), studied about the physicochemical characteristics of sediment in Thengapattanam estuary, southwest coastal zone of India. Ganesh et al. (2013) made a detailed textural and clay mineralogical study of Gosthani estuary, Bheemunipatnam, East coast of India. Sediment characteristics of Manakudy estuary and Tamiraparani estuary, Tamil Nadu was analyzed by Kumar and Sheela (2013) and Suresh Gandhi et al. (2009). Characteristics and distribution of elements in the sediments of Manakudy estuary on the South West coast of India has been studied by Jeena Pearl and Abbas Fenreji (2010). Seasonal changes of sedimentation rates and sediment characteristics in the Gulf of Mannar coral island, India was investigated by Yogesh Kumar et al. (2013). Chandrakiran and Sharma (2013) investigated the relationship of sediment community and macrobenthic community composition in lower Himalayan Lake, Mansar. Kessarkar et al. (2013) analyzed the geochemistry of the suspended particulate matter and its source in the Mandovi and Zuari River estuaries. The study found that the compositions of suspended particulate matter were controlled by the particulates from ore dust, geology of the drainage basins and various physico-chemical processes in the estuaries. Sankaran et al. (2014) observed the physicochemical characteristics of coastal sediments along the southeast coast of India. The study noted that coarser sediments contain moderate concentrations of soil organic carbon and total nitrogen than finer and well sorted sediments. Similarly the textural characteristics and mud contents was varied due to the distinction of sediment sources, topography and its reaction towards currents and waves. Purnachandra Rao et al. (2014) conducted a study of bottom sediment, clay minerals and isotopic analysis of suspended particulate matter to show the function of estuarine processes on their distribution in Mandovi and Zuari estuary. Similarly Shynu et al. (2015) investigated the organic carbon, total nitrogen and stable carbon and nitrogen isotopic compositions in suspended particulate matter and surface sediment of Mandovi and Zuari estuaries. The study showed that the rivers draining into the estuary contribute at least 20 % terrestrial organic carbon to the coastal system throughout the season.

The nature of sediment and distribution dynamics of nutrients in the Vembanad estuary particularly in the Cochin estuary has been a major area of research by several authors. Sankaranarayanan and Qasim (1969) concentrated their research on the nutrient status in the water and sediment phases of Cochin estuary in relation to other environmental features. Murty and Veerayya (1972) gave a detailed account of the phosphorus content in the sediments of the Vembanad estuary. The studies showed that the grain size varied from place to place and the sediments were relatively finer in the northern part of the estuary when compared with those in the southern part. In Cochin estuary Ansari and Rajagopal (1973) made a study on the distribution of phosphate in the mud and its exchange between the mud and the overlying water and also its relation to the texture of the sediment. The study noted that higher percentage of organic matter was associated with finer sediments and comparatively lower percentage of organic matter was associated with coarser sediments. Unnithan et al. (1975), Vijayan et al. (1976) and Sarala Devi et al. (1979) documented the effect of organic pollution due to industrial pollution on the quality of sediment in Cochin backwater in relation to pollution aspects. Sarala Devi (1989) and Sarala Devi et al. (1992) investigated on the temporal and spatial distribution of particulate organic carbon and particulate matter in Cochin backwaters, especially in the lower reaches of the Periyar river. Characteristics of sediments in the prawn field in and around Cochin estuary was conducted by Aravindakshan et al. (1992). They observed a direct relationship between the organic carbon content and texture of sediment. Biogeo-organics in the sedimentary environments of Cochin estuary was done by Nayar (1992). Sediment texture analysis, carbon and nitrogen and phosphorus contents, analyses of chlorophyll *a*, *b*, *c*, carotenoids, pheopigments, carbohydrates, lipids, protein, humic acid and hydroxylated aromatic compounds have been carried out in his study. Studies on carbon, nitrogen and phosphorus in the sediment indicated a polluted condition, mainly from high organics and phosphorus content and that these inputs were mostly of allochthonous in nature. Ramachandran (1992) investigated the texture and composition of inner-shelf sediments between Narakkel and Purakkad, Kerala and mainly focused on the formation of mud banks. Source, sedimentation and geochemistry of the modern newline sediments of the mud banks of Kerala coast were investigated by Purandara (1990). The geochemical analysis of sediment of Vembanad lake and mud bank sediments showed that the iron and manganese were widely distributed. Padmalal (1992) studied the mineralogy and geochemistry of the sediments of Muvattupuzha river and central Vembanad estuary. Sediment textures have also been noted to authenticate the mineralogical and geochemical study. Rajamani Amma (1994) investigated the distribution and partition of some of the trace metals in sediments and waters of the coastal environment of Kerala. Distribution of organic carbon in the sediment of Cochin mangroves was studied by Sunil Kumar (1996). Geochemistry of interstitial waters and sediments of Vembanad estuary was studied by Abdula Bava (1996). The role of sediments in nutrient dynamics and fertility of Kuttanad waters in Alappuzha were studied by Mathews (2000). The study observed that, the concentration and dynamics of the inorganic nutrients are controlled by the sedimentary exchange, and the quantitative and qualitative studies of nitrogen and phosphorus in sediments are important for understanding the basic processes governing the distribution and biogeochemical cycling of nutrients. Geochemistry of bottom sediments in a river-estuary-shelf mixing zone on the Cochin estuary was done by Paul (2001). The study indicates that the sediment texture is the major controlling factor in the distribution of elements in the area. Sediment characteristics such as texture, redox potential and organic carbon of Poonthura estuary, Kerala was evaluated by Anila Kumary (2001). Similarly sediment quality of Poonthura backwater was studied by Beslin (2014). Sediment quality of Ashtamudi estuary was studied by Babu et al. (2010) and Soumya et al. (2011). Martin et al. (2012) estimated the concentration and distribution of trace metals in surficial sediments of Cochin estuary and found that spatial variations were in accordance with textural characteristics and organic matter content. According to the study increased level of trace metal pollution in sediments of Cochin estuary was attributed to the anthropogenic contribution of industrial, domestic and agricultural effluents. The nutrient dynamics in the sediments of Kerala coast were studied by Nair and Sujatha (2013). Akhil et al. (2013) assessed the relationship between the biogeochemical constituents of sediment in the Cochin estuarine system and found that characterization of organic matter helped to trace out the source of allochthonous and autochthonous inputs into the estuary. Sedimentary organic carbon source and its characterisation in Cochin estuary was done by Gireesh Kumar (2013). Shiji et al. (2015) assessed the sediment quality of Kavvayi wetland in the South coast of India and found that sediments were polluted with heavy metals such as Fe, Mn, Cu, Pb, Cd, Ni, and Zn.

Numerous studies has been conducted on toxic metals found in the sediments of Vembanad estuary especially in Cochin estuary Nair et al. (1992); Shankaranarayanan et al. (1998); Balachandran et al. (2002); Kaladharan et al. (2005); Balachandran et al. (2006); Harikumar, (2009); Ratheesh Kumar et al. (2010); Sudhanandh et al. (2011); Anju et al. (2011); Mahesh Mohan, (2012); Paneer Selvam et al. (2012); Ramasamy et al. (2012); Manju et al. (2013). All these studies revealed the increasing levels of heavy metal pollution in the estuary.

4.2 Results

4.2.1 Physico-chemical characteristics

The temporal variations, results of ANOVA and correlation analysis of physico-chemical characteristics of sediment is given in annexure 4.1 to 4.7. The comparison of sediment parameters on south and north of TMB is given in Annexure 4.8.

4.2.1.1 Sediment colour

In station 1 the sediment was black in colour. In station 2 it was orange-brown in appearance. Sediment in station 3 was dark gray in colour. While in station 4 it was orange brown in colour. Station 5, 6, 7 and 9 the colour varied from light brown to blackish and light gray to black. In station 8 and station 10, it was black and olive gray colour respectively.

4.2.1.2 Sediment temperature

Sediment temperature for 2011-2012 period-ranged from 23 to 32 ^{o}C with an average of 28.6 ^{o}C. The highest temperature was recorded in the month of May 2011 in stations 2, 6 and 8 and August 2011 in station 9 and the lowest in the month of October 2011 from all stations except 3, 5 and 6. Spatially the minimum temperature was observed in station 3(27.86 ± 2.01 ^{o}C) and maximum in station 8 (29.07 ± 2.60 ^{o}C) (Fig. 4.1). Seasonal values for the estuary were 30.48 ± 0.83 °C, 28.5 ± 2.3 °C and 26.79 ± 2.3 °C respectively for PRM, MN and PM periods. ANOVA showed that temporal variation of sediment temperature was significant ($P \leq 0.01$). During 2012 – 2013 period, the sediment temperature ranged from 29 °C to 34 °C. Spatially the minimum average value of temperature was observed in station 1 (30.83 ± 1.27 °C) and maximum in station 7 (31.75 ± 1.14 °C). Seasonal analysis showed that the average value of sediment temperature during PRM, MN and PM was 32.03 ± 0.85 °C, 30.21 ± 1.15 °C and 31.56 ± 0.63 °C respectively. Similar to 2011-2012 period, ANOVA showed significant ($P \leq 0.01$) temporal variation of sediment temperature during 2012-2013 period.

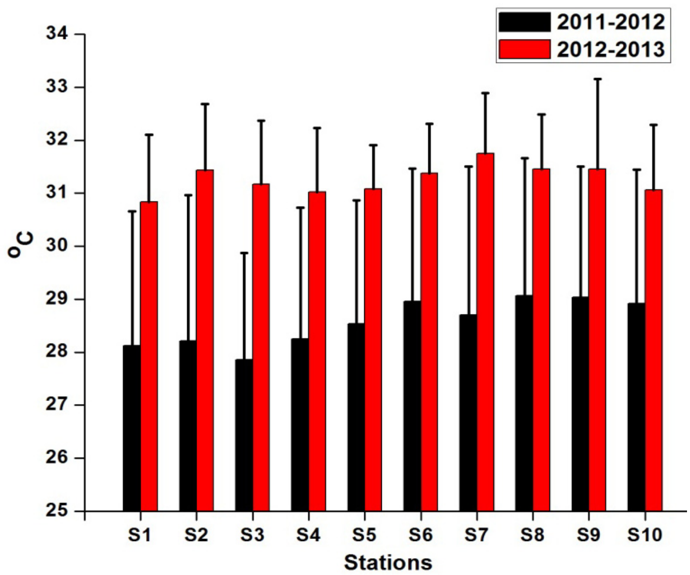

Fig. 4.1 Mean spatial variation of sediment temperature in Vembanad estuary during 2011-2013 period

4.2.1.3 Sediment pH

During the period 2011-2012, the sediment pH ranged from 3.73 to 7.48. The maximum pH was observed in station 10 (February, 2012) and minimum in station 2 during March 2011. Spatially the lowest average pH was observed in station 5 (6.29 ± 0.43) and highest in station 10 (6.83 ± 0.36) (Fig. 4.2).

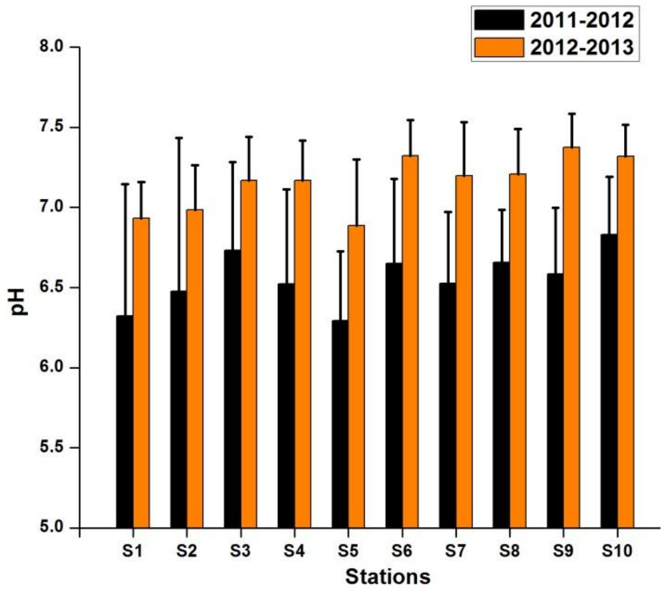

Fig. 4.2 Mean spatial variation of sediment pH in Vembanad estuary during 2011-2013 period

Seasonal observation demonstrated that, PM season was having maximum pH (6.88 ± 0.31) followed by MN (6.50 ± 0.36) and PRM (6.30 ± 0.78). On comparing the southern zone and northern zone, the sediment pH was slightly higher in north (6.65 ± 0.39) than the south (6.5 ± 0.67). ANOVA showed that temporal variation of sediment pH was significant ($P \leq 0.01$). During 2012 – 2013 period sediment pH varied between 6 (station5, May 2012) and 7.85 (station 8, May 2012) with an annual average of 7.16 ± 0.31. Mean spatial variation of sediment pH was lowest in station 5 (6.89 ± 0.41) and highest in station 9 (7.38 ± 0.21). Premonsoon season exhibited a moderately higher sediment pH (7.25 ± 0.38), followed by MN (7.18 ± 0.29) and PM (7.05 ± 0.22). ANOVA showed that, both spatial and temporal variations of sediment pH were significant ($P \leq 0.01$). On comparing the southern zone and northern zone, the sediment pH was slightly higher in north zone (7.27 ± 0.26) than south (7.08 ± 0.32).

4.2.1.4 Sediment Eh

Oxidation reduction value (Eh) varied from 86 to -463 mV during 2011-2012 period. Lowest reduced condition was observed in station 1 (March 2011) and highest reduced condition in station 2 (January 2012). Spatially highest reduced condition was observed in station 1(-288.83 ± 136.01 mV) and lowest in station 7 (-74.42 ± 56.56 mV) (Fig. 4.3). Seasonal analysis of Eh observed that PM was having a highly reduced condition (-226.15 ± 118.31mV) followed by MN (-163.83 ± 109.36 mV) and PRM (-98.78 ± 90.55 mV). ANOVA of Eh showed that variations between stations and seasons were significant ($P \leq 0.01$). During second year the Eh value ranged between -13 to -599 mV. The lowest reduced condition was observed in station 9 during August 2012 and highest in station 4 during August 2012. Seasonal variation of Eh was not much distinct. The highest mean seasonal value was observed during PM (-218.30 ± 110.82 mV) season. But its variation between PRM (-210.15 ± 111.39 mV) and MN (-210.45 ± 129.18 mV) were not much distinct. ANOVA of Eh was significant at 5% level between stations ($P \leq 0.05$).

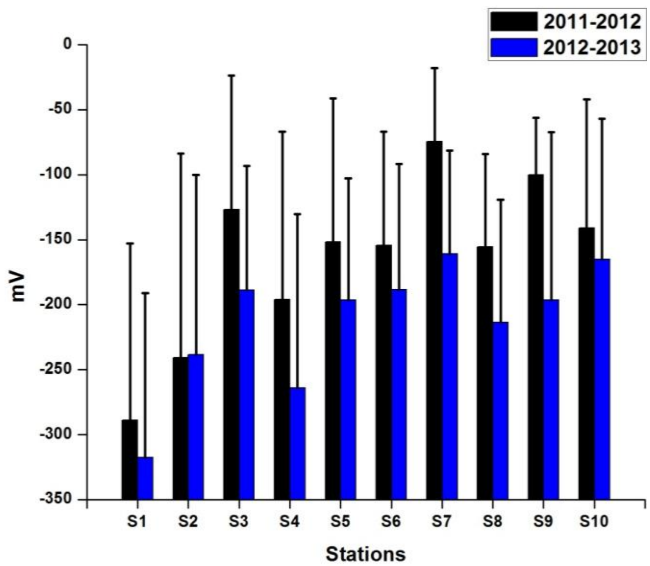

Fig. 4.3 Mean spatial variation of sediment Eh in Vembanad estuary during 2011-2013 period

4.2.1.5 Sediment water content (moisture content)

Sediment water content varied from a minimum of 2.97 % in station 9 (September 2011) to maximum of 15.96 % in station 10 (August 2011) with an annual average of 7.06 ± 2.62 %. Spatially mean value of moisture content recorded the lowest in station 7 (6.09 ± 03.1%) and highest in station10 (8.3 ± 3.3 %) (Fig. 4.4). Mean seasonal variation of water content was highest during MN (8.5 ± 3.06%) and the lowest was observed during PRM (6.03 ± 1.95%). Seasonal variation of sediment water content was significant at 1% level ($P \leq 0.01$). During 2012 – 2013 period, water content varied from 1.8 % to 11.88 % with an annual mean value of 4.99 ± 1.38%. Spatial variation of water content was lowest in station 9 (4.00 ± 1.04 %) and highest in station 2 (6.03 ± 2.08). Premonsoon season showed the maximum sediment water content (5.20 ± 1.34%), followed by MN (5.17 ± 1.62%) and PM (4.61 ± 1.10%). ANOVA of water content was significant at 1% level between stations ($P \leq 0.01$).

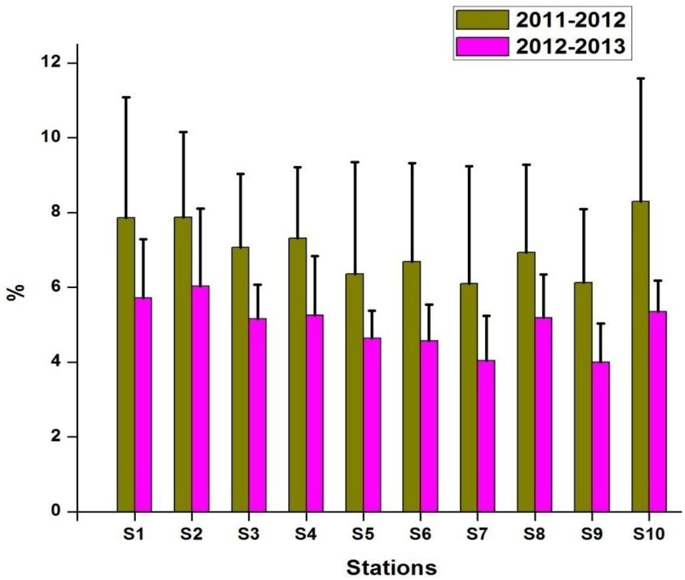

Fig. 4.4 Mean spatial variation of sediment water content in Vembanad estuary during 2011-2013 period

4.2.1.6 Organic carbon

During 2011-2012 period sediment organic carbon varied from 0.03 to 9.4 % with an average value of 2.69 ± 2.23 %. The lowest value of 0.03 % was observed in station 9 during December 2011 and highest value of 9.4% was observed in station 1 during September 2011. Station wise mean values of organic carbon recorded the lowest in station 5 (0.91 ± 0.39 %) and highest in station 1 (6.81 ± 1.45 %) and station 2 (4.79 ± 1.57%) (Fig. 4.5). Seasonal analysis showed that the highest value of organic carbon was noticed during MN (3.29 ± 2.59 %) and lowest during PM (2.04 ± 1.6 %). When comparing the southern zone and northern zone, the organic carbon was slightly higher in south (3.2 ± 2.34) than north (1.93 ± 1.81). ANOVA result of organic carbon showed that it was significant at 1% level between seasons ($P \leq 0.01$) between stations ($P \leq 0.01$) and between seasons and stations ($P \leq 0.05$).

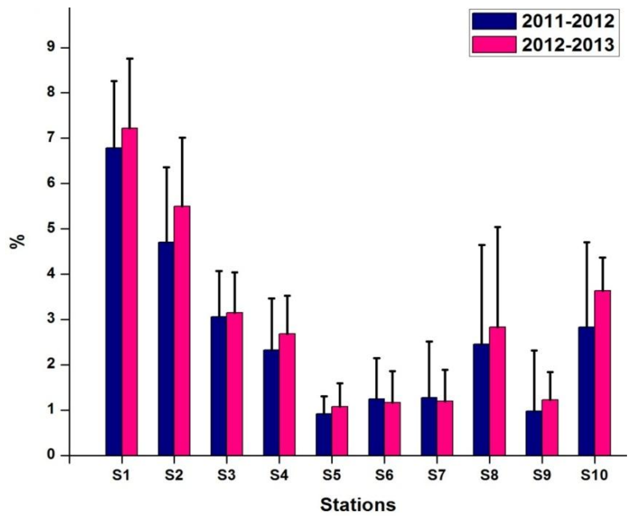

Fig. 4.5 Mean spatial variation of organic carbon in the sediment samples of Vembanad estuary during 2011-2013 period

During 2012 – 2013 period the organic carbon varied from 0.31 to 9.8 % with an annual average of 2.97 ± 2.25 %. During this period, the spatial variation of organic carbon was similar to first year, where the lowest value was observed in station 5 (1.08 ± 0.52 %) and highest in station 1 (7.22 ± 1.54 %) and station 2 (5.50 ± 1.51 %). Similar to first year, seasonal average of organic carbon was highest during MN (3.40 ± 2.51 %) and lowest in PM (2.52 ± 1.73 %). It was noted that the annual average of organic carbon in the southern zone was slightly higher (3.46 ± 2.48 %) than the northern zone (2.22 ± 1.61 %). Similar to first year ANOVA of organic carbon exhibited a significant spatial ($P \leq 0.01$) and seasonal ($P \leq 0.01$) variation. The spatial and seasonal interaction was also significant ($P \leq 0.01$).

4.2.1.7 Organic matter

The organic matter ranged from 0.05 % to 16.17 % with an annual average of 5.04 ± 3.8 %. The low value was observed in station 9 during December 2011 and high value of 16.17 % was observed in station 1 during September 2011. Spatial variation of organic matter was maximum in station 1 (11.70 ± 2.54 %) followed by station 2 (8.10 ± 2.86 %) (Fig. 4.6). The maximum mean value of organic matter was observed during MN (5.67 ± 4.46 %) and minimum was in PM (3.78 ± 2.88 %). ANOVA result of organic matter showed that it was significant at 1% level between stations ($P \leq 0.01$) and significant at 1% level between seasons ($P \leq 0.01$).

During 2012 – 2013 period, the highest organic matter was found at station 1 (16.9 %) during July 2012 and lowest in station 5 (0.54 %) during August 2012. Similar to first year, the highest seasonal value was observed during MN (5.86 ± 4.33 %), followed by PRM (4.92 ± 3.94 %) and PM (4.35 ± 2.98 %). ANOVA of organic matter exhibited a significant spatial ($P \leq 0.01$) and temporal variation ($P \leq 0.01$).

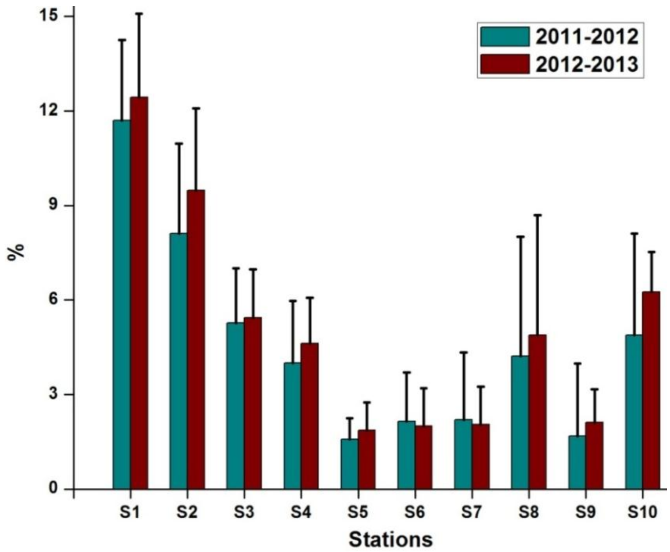

Fig. 4.6 Mean spatial variation of organic matter in the sediment samples of Vembanad estuary during 2011-2013 period

4.2.1.8 Total carbon

Total carbon in the sediment varied widely from 6.87 to 121.4 g/Kg with an annual average of 30.17 ± 25.05 g/Kg. The lowest value of 6.87 g/Kg was observed in station 5 during February 2012 and highest in station 1 during April 2011. Mean spatial variation of total carbon was lowest (12.09 ± 2.63 g/Kg) in station 9 and highest in station 1 (93.59 ± 18.18 g/Kg) and 2 (47.15 ± 13.97 g/Kg) (Fig. 4.7). Seasonal variation was not distinct in the case of total carbon. Spatial variation of total carbon in sediment was significant at 1% level ($P \leq 0.01$). Total carbon content of sediment showed a wide variation between south and north zone. The annual average was maximum in southern part (37.50 ± 29.07 g/Kg) than northern part (19.17 ± 10.34 %).

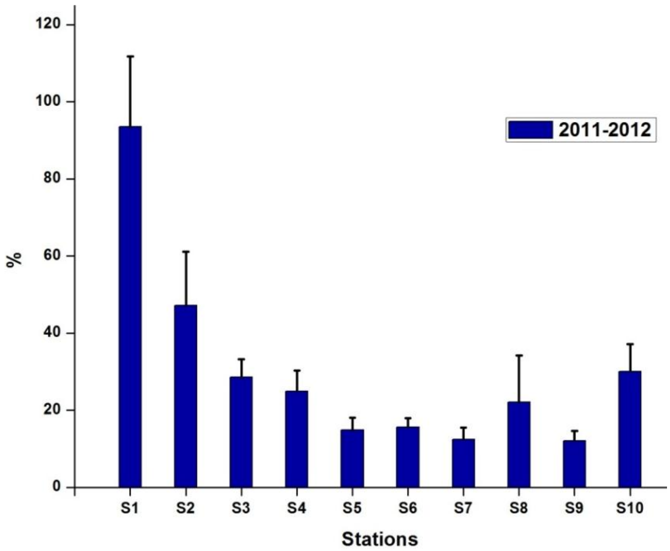

Fig. 4.7 Mean spatial variation of total carbon in the sediment samples of Vembanad estuary during 2011-2012 period

4.2.1.9 Available phosphorus

Available phosphorus concentration in sediment showed a wide variation from 1.2 to 301.6 mg/Kg with an average value of 38.61 ± 44.45 mg/Kg in 2011-2012 period. The lowest value of available phosphorus was observed in station 6 during February 2012 and highest in station 10 during October 2011. Station wise mean values indicated that available phosphorus recorded the lowest in station 7 (13.45 ± 17.13 mg/Kg) and highest in station 10 (87.13 ± 73.12 mg/Kg) followed by station 1 (80.97 ± 27.68 mg/Kg) (Fig. 4.8). Compared to other seasons, relatively higher concentration of available phosphorus was observed during PM (55.77 ± 62.83 mg/Kg) followed by MN (35.64 ± 31.09 mg/Kg). In contrast, a significant reduction (24.42 ± 24.43 mg/Kg) in available phosphorus was observed during PRM. ANOVA revealed that available phosphorus values exhibited a significant spatial ($P \leq 0.01$) and seasonal ($P \leq 0.01$) variation. Available phosphorus distribution showed that the maximum average value was observed in southern than northern (37.73 ± 49.24 mg/Kg) zone. During second year also available phosphorus concentration in sediment showed a wide variation form 0 to 470.2 mg/Kg with an annual average of 31.63 ± 52.03 mg/Kg. Seasonal average of available phosphorus was highest during PM (44.35 ± 77.65 mg/Kg), followed by PRM (31.92 ± 32.71 mg/Kg) and MN (18.63 ± 28.66 mg/Kg). The distribution of available phosphorus in south and northern zone of Vembanad showed a clear delineation, in which, the maximum average value was observed in the southern zone (34.70 ± 62.02 mg/Kg) than the northern zone (27.02 ± 31.75 mg/Kg). ANOVA showed that available phosphorus values exhibited a significant spatial variation ($P \leq 0.01$).

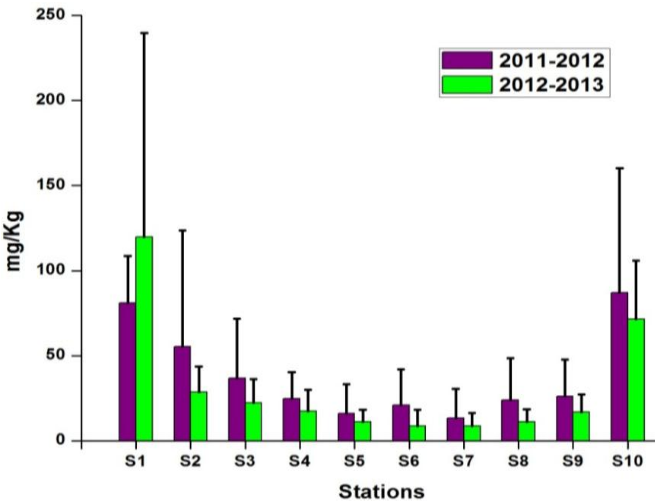

Fig. 4.8 Mean spatial variation of available phosphorous in the sediment samples of Vembanad estuary during 2011-2013 period

4.2.1.10 Available nitrogen

During 2011-2012 period, available nitrogen in the sediment ranged from 0.001 % and 0.17 % with an annual average of 0.03 ± 0.03 %. Spatial analysis showed that average minimum available nitrogen was observed in stations 5, 6, 7 and 9 (0.01 %) and maximum in station 1 (0.08 %) (Fig. 4.9). Considering the seasonal average, a slight increase in available nitrogen was observed during the PM (0.05 ± 0.04 %) and MN (0.03 ± 0.02 %), whereas the lowest seasonal average of available nitrogen was observed during PRM (0.02 ± 0.03 %).

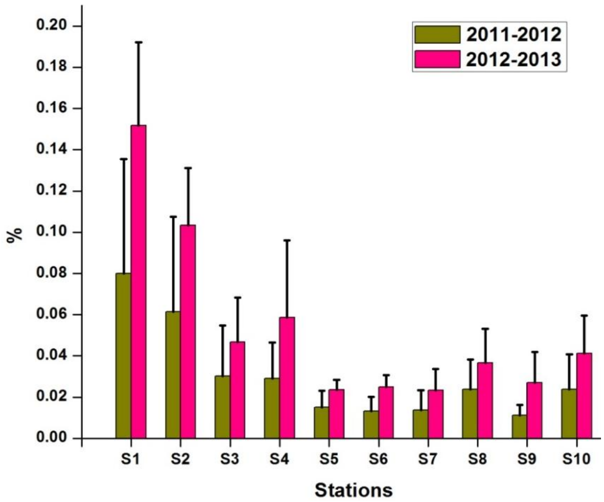

Fig. 4.9 Mean spatial variation of available nitrogen in the sediment samples of Vembanad estuary during 2011-2013 period

The average value of available nitrogen was maximum in the southern zone (0.04 ± 0.04 %) than north (0.02 ± 0.01 %). ANOVA result of available nitrogen showed that it was significant at 1% level between stations ($P \leq 0.01$) and between seasons ($P \leq 0.01$). During 2012 – 2013 period available nitrogen in the sediment varied from 0.01 to 0.21% with an annual average of 0.05 ± 0.05 %. The minimum value was observed in station 3 (February 2013) and station 8 (July 2012) and maximum value in station 1 (September 2012). Considering seasonal analysis, the available nitrogen did not show significant variation between MN (0.06 ± 0.05 %) and PM (0.06 ± 0.04 %) periods. The lowest seasonal value was observed during PRM (0.04 ± 0.04 %). Distribution of available nitrogen showed a similar trend as that of first year, in which maximum values were observed in the southern zone (0.07 ± 0.05 %) than north (0.03 ± 0.02 %). Similar to first year, the ANOVA of available nitrogen exhibited a significant spatial ($P \leq 0.01$) and seasonal variation ($P \leq 0.01$).

4.2.1.11 Sodium

During 2011-2012 period, sediment sodium concentration showed a wide variation from 1 to 55200 mg/Kg. The lowest value was observed in station 4 during December 2011and highest in station 10 during February 2012. Spatial mean variation of sodium was minimum in station 2 (356.67 ± 431.58 mg/Kg) and maximum in station 10 (16439.0 ± 13567.3 mg/Kg) (Fig. 4.10). Relatively high average concentration of sodium was observed during PRM (3461.0 ± 9343.88 mg/Kg). But a slight variation was observed between MN (1630.6 ± 4328.35 mg/Kg) and PM (1672.70 ± 3802.3 mg/Kg) season. In contrast with other sediment parameters, the maximum average level of sodium was observed in the northern zone (4728.70 ± 9502.81 mg/Kg) than south (605.47 ± 863.57 mg/Kg). ANOVA showed that sodium values exhibited a significant spatial variation ($P \leq 0.01$).

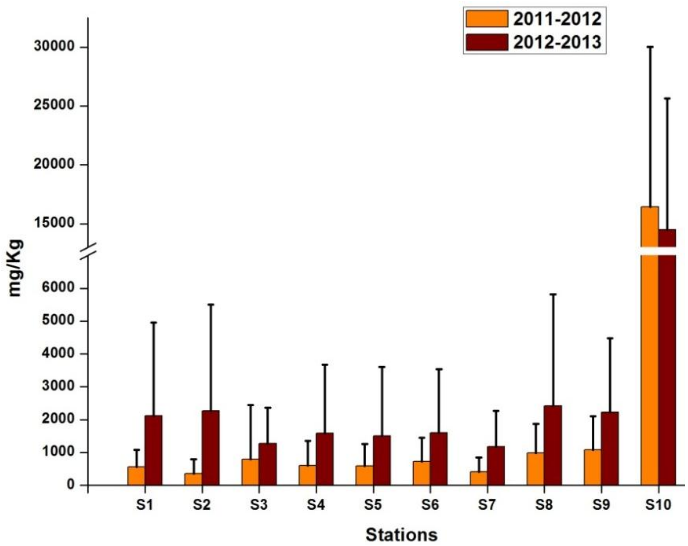

Fig. 4.10 Mean spatial variation of sodium in the sediment samples of Vembanad estuary during 2011-2013 period

During second year the highest value of 32500 mg/Kg of sodium was recorded in station 10 (February 2013) and lowest value of 3 mg/Kg in station 5 (June 2012). Spatial average of sodium was maximum in station 10 (14512.00 ± 11147.3 mg/Kg) and minimum in station 7 (1173.20 ± 1098.5 mg/Kg). In contrast with first year the highest average value was observed during PM (4264.0 ± 6469.66 mg/Kg) and lowest during MN (1073.3 ± 3109.81 mg/Kg). In the south and north zone, the distribution of sodium followed a similar pattern as that of first year. The maximum average value was observed in north (5083.00 ± 7984.20 mg/Kg) than south zone (1726.00 ± 2265.03 mg/Kg). Similar to 2011-2012 period, sodium exhibited a significant spatial ($P \leq 0.01$) and seasonal variation ($P \leq 0.01$).

4.2.1.12 Potassium

During 2011-2012 period sediment potassium varied from 165 to 3976 mg/Kg with an average value of 706.58 ± 820.42 mg/Kg. The lowest value of 165 mg/Kg was observed in station 8 during September 2011 and highest value of 3976 mg/Kg in station 10 during October 2011. Spatial mean values recorded the lowest in station 5 (356.50 ± 84.95 mg/Kg) and highest in station 10 (2856.42 ± 1000.04 mg/Kg) (Fig. 4.11). Considering seasonally, the mean variation of potassium was highest during MN (761.70 ± 856.2 mg/Kg) and the lowest during PRM (656.33 ± 633.74 mg/Kg). Similar to sodium, the highest average value of potassium was observed in north zone (1031.21 ± 1203.62 mg/Kg) than south (490.15 ± 224.47 mg/Kg). ANOVA revealed that spatial variation of potassium was significant at 1% level ($P \leq 0.01$). The spatial and seasonal interaction of potassium was significant at 5% level ($P \leq 0.05$).

During 2012-2013 period, the sediment potassium level varied from 2 to 2984 mg/Kg with an annual average of 400.69 ± 616.38 mg/Kg. The lowest value was found in station 3 during February 2013 and highest in station10 during October 2012. In contrast with first year the highest seasonal average was observed during PRM (413.68 ± 422.33 mg/Kg). During MN the value decreased to an average of 375.50 ± 710.97 mg/Kg and again increased to 412.90 ± 688.81 mg/Kg during PM. Similar to first year the highest average of potassium value was observed in north zone followed by the south (238.61 ± 167.74 mg). Potassium content of sediment exhibited a significant spatial variation ($P \leq 0.01$) and its spatial and seasonal interaction was significant at 1% level ($P \leq 0.01$).

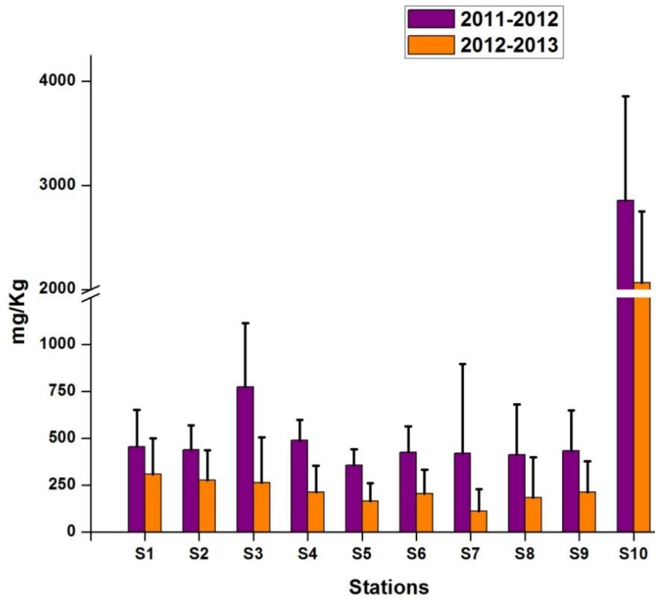

Fig. 4.11 Mean spatial variation of potassium in the sediment samples of Vembanad estuary during 2011-2013 period

4.2.1.13 Calcium

During 2011-2012 period the average sediment calcium was 868.17 ± 809.36 mg/Kg with lowest of 89 mg/Kg in station 7 during July 2011 and highest of 3832 mg/Kg in station 10 during June 2011. The lowest spatial average was observed in station 5 (361.83 ± 97.09 mg/Kg) and highest in station 10 (2545.7 ± 1152.28 mg/Kg) (Fig. 4.12). Seasonal analysis showed that maximum average calcium was observed during MN (983.08 ± 899.33 mg/Kg) followed by PM (852.45 ± 819.08 mg/Kg) and PRM (768.98 ± 703.69 mg/Kg). Similar to sodium and potassium the highest average value of calcium was observed in northern (1008.20 ± 1115.90 mg/Kg) than the southern sector (774.79 ± 501.49 mg/Kg). ANOVA of calcium showed a significant seasonal variation at 1% level ($P \leq 0.01$). During second year the calcium content in the sediment varied between 154 to 7404 mg/Kg, with an annual average of 1244 ± 1554.98 mg/Kg. Spatial minimum average of 361.92 ± 140.66 mg/Kg was observed in station 7 whereas a maximum of 5079.20 ± 2080.18 mg/Kg was observed in station 10. Similar to first year the highest average value was observed during MN (1519.60 ± 2011.47 mg/Kg), followed by PM (1171.10 ± 1524.14 mg/Kg) and PRM (1041.20 ± 939.86 mg/Kg). Calcium content of sediment exhibited a significant spatial ($P \leq 0.01$) and seasonal ($P \leq 0.01$) variation.

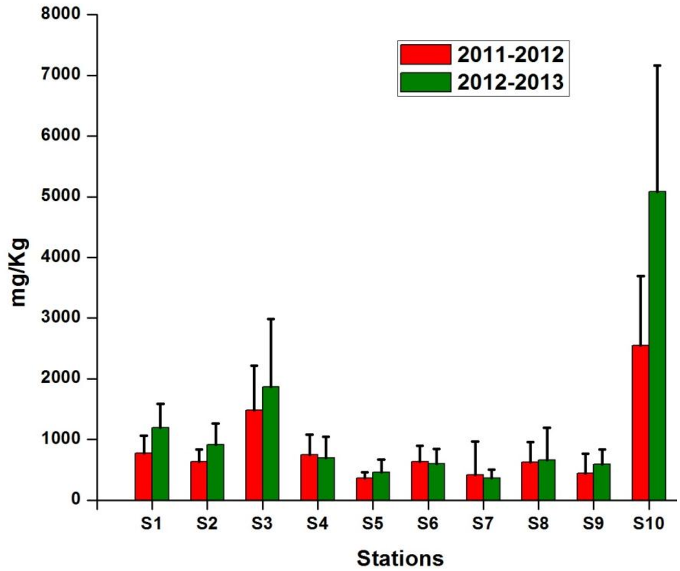

Fig. 4.12 Mean spatial variation of calcium in the sediment samples of Vembanad estuary during 2011-2013 period

4.2.1.14 Sediment Texture

During 2011- 2012 period, the annual average value for the sediment particle was 54.86 ± 26.96 %, 26.63 ± 21.68 %, and 18.5 ± 11.28 % for sand, silt and clay respectively (Fig. 4.13). Spatial average of sand fraction was highest in station 9 (86.32 ± 5.38 %), followed by station 7 (79.22 ± 12.58 %)

and lowest in station 10 (13.80 ± 9.65 %) (Fig. 4.14). The silt content varied between 0.04 to 71.26 %. The minimum percentage value of silt (0.04 %) was observed in station 9 during December 2011 and maximum value of 71.26% in station 10 during October 2011. Highest spatial average of silt content was observed in station 1 (53.28 ± 7.95 %), followed by station 10 (50.75 ± 16.94 %) and lowest silt content was in station 9 (4.30 ± 5.32 %) (Fig. 4.15). In the sediment texture, clay percentage ranged between 1.26 % in station 7 during December 2011 to 66.51% in station 3 during August 2011. The mean spatial variation of clay percentage was minimum in station 9 (9.38 ± 5.34 %) and maximum in station 10 (35.45 ± 14.83 %) (Fig. 4.16). Seasonally there was not much variation observed in the sediment texture. Sand fraction was highest in all the seasons followed by silt and clay. During PRM, sand fraction contributed 54.31 ± 26.42 %, followed by silt (26.18 ± 20.87 %) and clay (19.51 ± 10.63 %). During the MN sediment texture consisted of 55.83 ± 26.08 % of sand, 23.93 ± 20.27 % of silt and 20.24 ± 13.27 % of clay. During PM it was 54.45 ± 28.95 % of sand, 29.79 ± 23.88 % of silt and 15.76 ± 9.37 % of clay. Comparing the southern and northern zones of Vembanad, the percentage composition of sediment particle varied significantly. It was noted that, the silt and clay fraction was highest in southern zone than north, whereas the sand fraction was lowest. ANOVA results of sand fraction showed a significant seasonal variation at 1% level ($P \leq 0.01$), whereas spatial variation of silt ($P \leq 0.01$) and clay ($P \leq 0.01$) fraction was significant at 1% level.

During 2012 – 2013 period, the annual average of sand, silt and clay was 48.65 ± 34.46 %, 42.87 ± 28.75 % and 8.49 ± 12.09 % respectively (Fig. 4.17). The percentage of sand content in the sediment varied from 0.55 to 96.55 %. The maximum percentage content of sand was observed in station 7 during September 2012 and minimum in station 10 during August 2012. The minimum mean spatial average was observed in station 10 (2.20 ± 1.57 %)

and maximum in station 7 (93.64 ± 2.32 %) (Fig. 4.14). ANOVA of sand fraction showed that its spatial variation was significant at 1% level ($P \leq 0.01$). The silt fraction in the sediment varied from 0.44 (station 9, July 2012) to 89.89 % (station 10, September 2012). The maximum average percentage was observed in station 1 (76.38 ± 5.31%) and minimum in station 9 (4.57 ± 2.87 %) (Fig. 4.15). The clay fraction in the sediment varied between 0.04 to 60.19 %. The highest percentage of clay fraction was observed in station 10 during January 2013 and lowest in station 5 during April 2012. Spatially the minimum average of clay fraction was observed in station 7 (1.79 ± 1.09 %) and maximum in station 10 (36.06 ± 19.12 %) (Fig. 4.16). Seasonally the sand fraction in the sediment was not distinct, whereas the silt and clay fraction varied seasonally. In the case of silt the highest percentage was observed during MN (45.38 ± 31.71 %) followed by PM (42.18 ± 27.71 %) and lowest in PRM (41.06 ± 27.18 %). The clay fraction was highest during PRM (10.18 ± 13.68 %), followed by PM (9.23 ± 14.98 %) and MN (6.08 ± 4.97 %). Compared to southern and northern zone of Vembanad, the percentage composition of sediment particle varied each other. Similar to first year sand fraction was highest in the north zone than the south. In contrast to 2011-2012 period, the slit fraction was highest in south and clay fraction the highest in north. The ANOVA result of silt content in sediment showed that, its spatial ($P \leq 0.01$) and seasonal ($P \leq 0.05$) variation was significant. The ANOVA result of clay fraction showed that variations between stations and season were significant at 1% level ($P \leq 0.01$). The interaction between stations and seasons of silt ($P \leq 0.01$) and clay ($P \leq 0.01$) fraction was also significant.

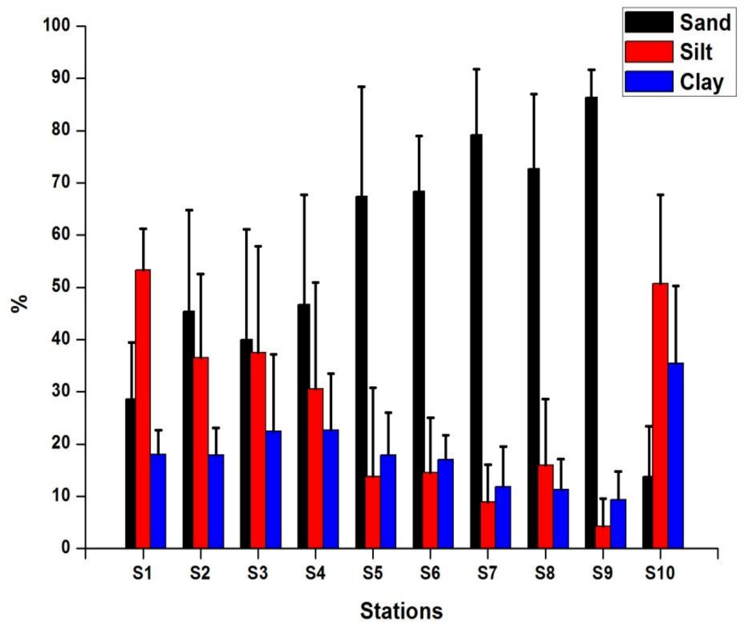

Fig. 4.13 Mean spatial variation of sediment texture in Vembanad estuary during 2011-2012 period

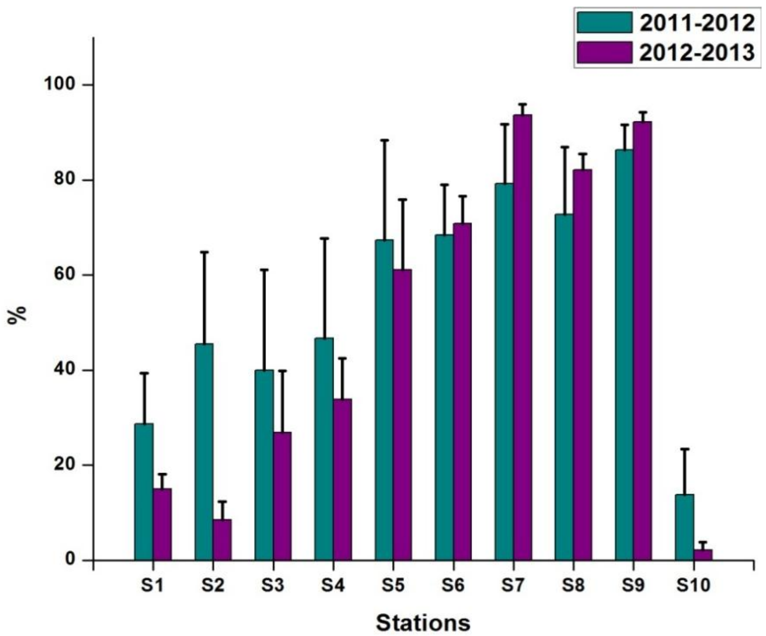

Fig. 4.14 Mean spatial variation of sand content in the sediment samples of Vembanad estuary during 2011-2013 period

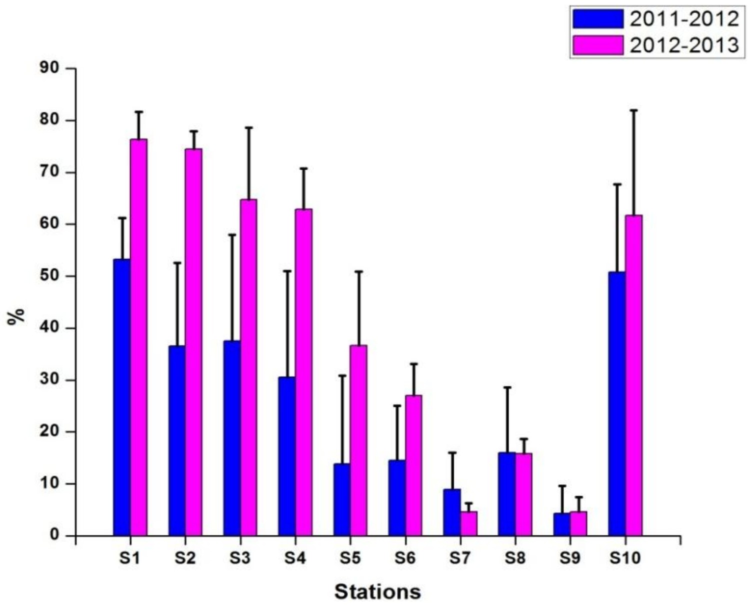

Fig. 4.15 Mean spatial variation of silt content in the sediment samples of Vembanad estuary during 2011-2013 period

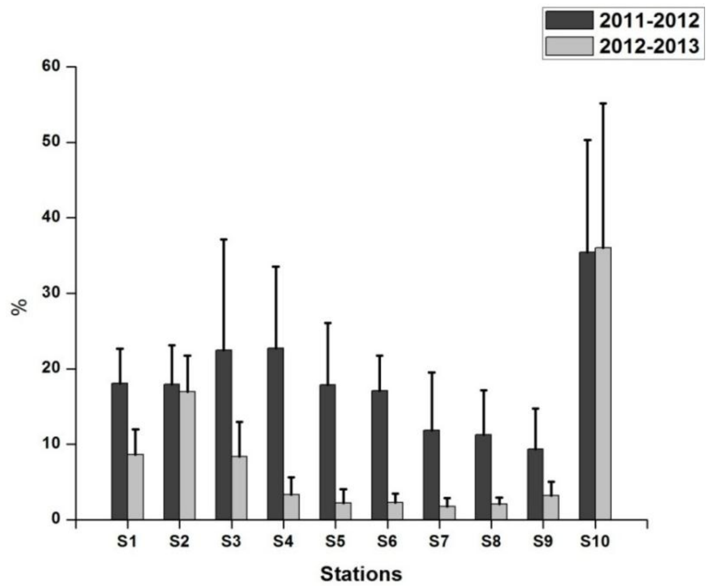

Fig. 4.16 Mean spatial variation of clay content in the sediment samples of Vembanad estuary during 2011-2013 period

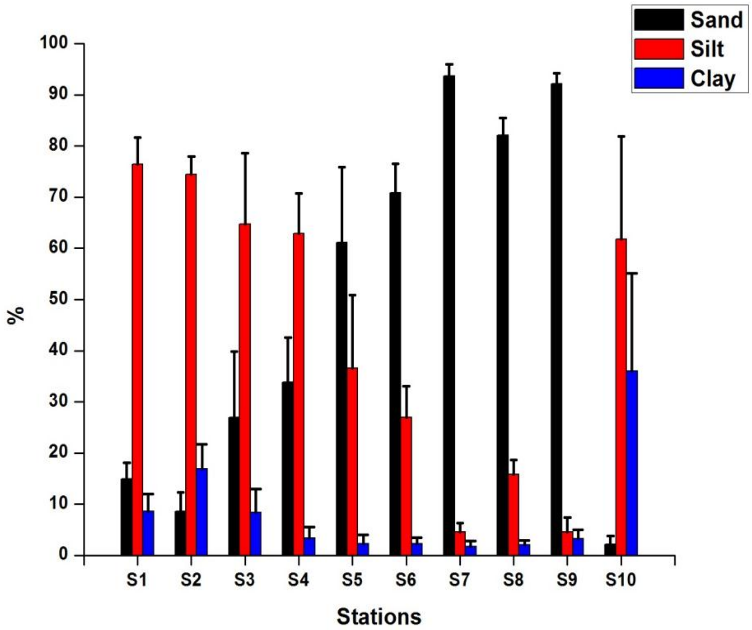

Fig. 4.17 Mean spatial variation of sediment texture in Vembanad estuary during 2012-2013 period

4.2.2 Distribution of Trace Metals

During 2011-2012 period concentration of copper showed maximum concentration (232.7 mg/Kg) during PRM in station 1 and minimum (3 mg/Kg) during PM season in station 8 (Fig. 4.18). Its annual average concentration was 33.27 ± 52.41 mg/Kg. Spatially highest average concentration of Cu was observed in station 1 (173.8 ± 72.45 mg/Kg) and station 2 (38.97 ± 6.17 mg/Kg). The lowest spatial average was observed in station 7 (7.27 ± 4.11). Zinc concentration varied between 9 to 8950 mg/Kg with an annual average of 589.01 ± 2057.81 mg/Kg. Similar to Cu, the spatial highest average of Zn was observed in station 1 (5467 ± 4660.55 mg/Kg) but the second highest concentration observed in station 10 (95.1 ± 13.59 mg/Kg). The lowest spatial average of Zn was observed in station 8 (21.37 ± 10.71 mg/Kg). Seasonally zinc showed maximum concentration in PRM with an average of 940.58 mg/Kg and minimum in PM (52.51 mg/Kg) (Fig. 4.19).

Throughout the seasons, cadmium concentrations in the sediment depict a heavily polluted condition. Its concentration varied between 0 to 3.1 mg/Kg with an annual average of 1.39 ± 0.77 mg/Kg. Seasonally the maximum average Cd concentration was observed during MN (1.61 ± 0.64 mg/Kg) and minimum during PM (1.13 ± 0.8 mg/Kg) (Fig. 4.20). Lead concentration in sediment varied between 12 mg/Kg in station 7 during PRM to 81.7 mg/Kg in station 1 during PM season. The seasonal average was highest during PM 59.74 ± 15.2 mg/Kg) followed by MN (44.57 ± 14.83 mg/Kg) and PRM (27.44 ± 19.13 mg/Kg) (Fig. 4.21). The highest spatial average of Cd and Pb were observed in station 1 and second highest in station 10. In station 1 the average concentration of Cd and Pb was 2.83 ± 0.38 mg/Kg and 79.53 ± 2.25 mg/Kg respectively. In station 10 the concentration of Cd and Pb was 2.03 ± 0.38 mg/Kg and 59.1 ± 22.91 mg/Kg respectively.

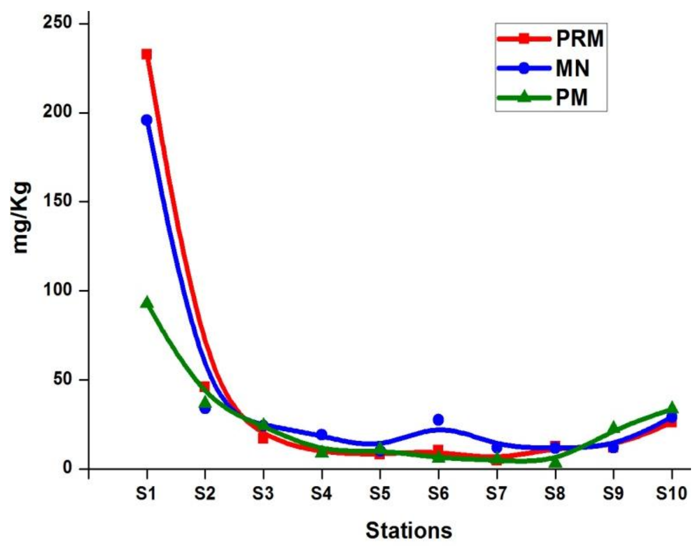

Fig. 4.18 Spatial variation of copper in the sediment samples of Vembanad estuary during 2011-2012 period

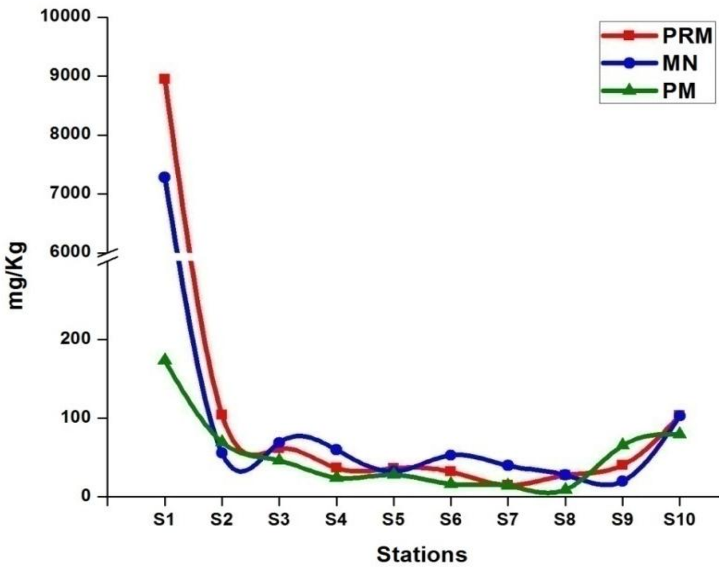

Fig. 4.19 Spatial variation of zinc in the sediment samples of Vembanad estuary during 2011-2012 period

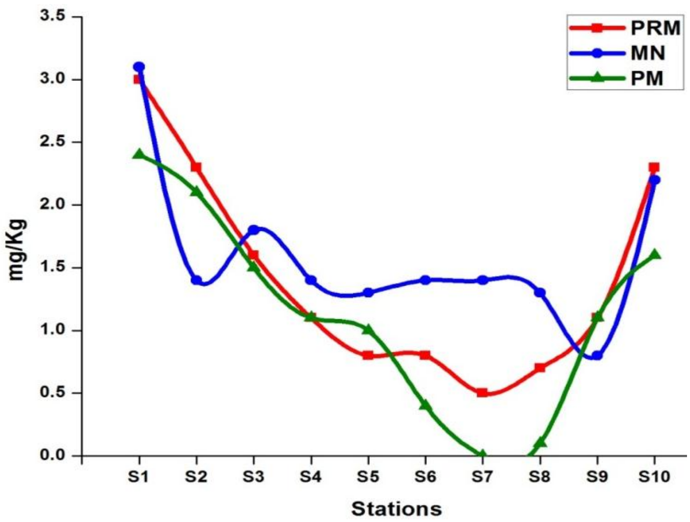

Fig. 4.20 Spatial variation of cadmium in the sediment samples of Vembanad estuary during 2011-2012 period

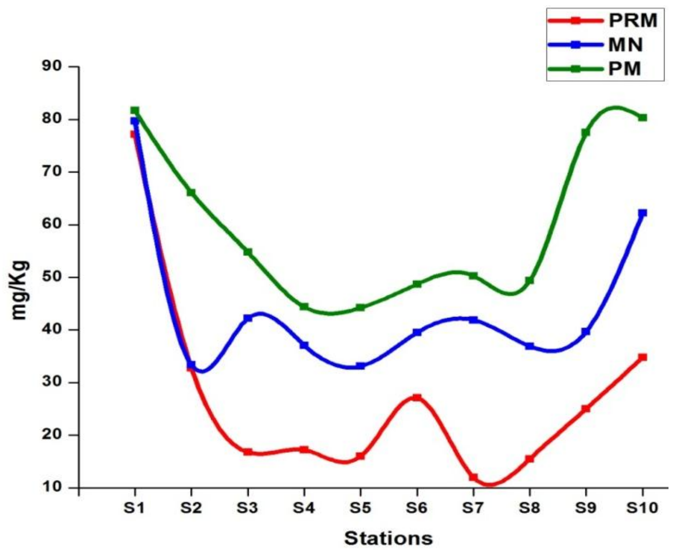

Fig. 4.21 Spatial variation of lead in the sediment samples of Vembanad estuary during 2011-2012 period

4.2.3 Data Analysis

4.2.3.1 Principal Component Analysis (PCA)

The data for the two years of study were processed by applying PCA in order to gain information on the distribution of sediment parameters by detecting relationship and also differences among the parameters. The PCA was carried out with factors having eigenvectors higher than one and correlation matrix was used. The amount of variance captured by each of the three PCs is given in Table 4.1 and Figure 4.22 and Figure 4.23. During first year first two PCs of the sediment parameters account for a total variance of 86.04 %. The first PC had higher loadings for the parameters cadmium, silt, organic carbon, organic matter, total carbon, available phosphorus, available nitrogen, water content, copper, lead and zinc. The second PC had higher loadings for potassium, sodium, calcium, pH, clay. During 2012-2013 period, the PCA analysis produced two principal components (PC) and accounted for 84.24% of the observed variation in sediment quality parameters. The first PC have a total variance of 55.24 % with high positive loading values for organic matter, organic carbon, silt, water content, available phosphorus, available nitrogen and clay particles. Similarly the first PC has a strong negative loading value for sand, temperature and Eh. The second PC (28.72%) had strong loading of sodium, potassium, calcium, clay, Eh and pH.

Table 4.1 Results of PCA analysis of sediment parameters in Vembanad estuary during 2011-2012 and 2012-2013 period.

2011-2012				**2012-2013**		
Eigen	11.1973	5.15095	1.31661	**Eigen**	7.77338	4.02035
%Variation	58.933	27.11	6.9295	**%Variation**	55.524	28.717
Cum.% Variation	58.933	86.043	92.9725	**Cum.% Variation**	55.524	84.241
Variable	**PC1**	**PC 2**	**PC3**	**Variable**	**PC1**	**PC 2**
Cd	0.98	-0.01	0	TEMP	-0.73	0.07
SILT	0.95	0.24	-0.19	pH	-0.45	0.61
OC	0.93	-0.28	-0.05	Eh	-0.62	0.69
OM	0.93	-0.28	-0.05	WC	0.88	-0.15
TC	0.92	-0.37	0.09	OC	0.92	-0.26
AP	0.92	0.31	0.2	OM	0.92	-0.26
AN	0.88	-0.42	-0.1	AN	0.83	-0.48
WC	0.88	0.33	-0.16	Na	0.42	0.88
Cu	0.84	-0.41	0.3	K	0.47	0.87
Pb	0.84	-0.04	0.51	Ca	0.54	0.83
Zn	0.75	-0.48	0.35	SAND	-0.93	-0.15
CLAY	0.53	0.74	-0.28	SILT	0.91	-0.09
Ca	0.48	0.86	-0.04	CLAY	0.65	0.72
K	0.37	0.91	0.15	AP	0.85	0.02
Na	0.32	0.89	0.25			
pH	-0.11	0.81	0.01			
TEMP	-0.53	0.32	0.7			
Eh	-0.79	0.43	0.14			
SAND	-0.87	-0.47	0.23			

(**PCA code**- Cd-Cadmium; OC-Organic Carbon; OM-Organic Matter; TC- Total Carbon; AP-Available Phosphorus; AN-Available Nitrogen; WC-Water Content; Cu- Copper; Pb-Lead; Zn-Zinc; Ca-Calcium; K- Potassium; Na-Sodium; TEMP-temperature)

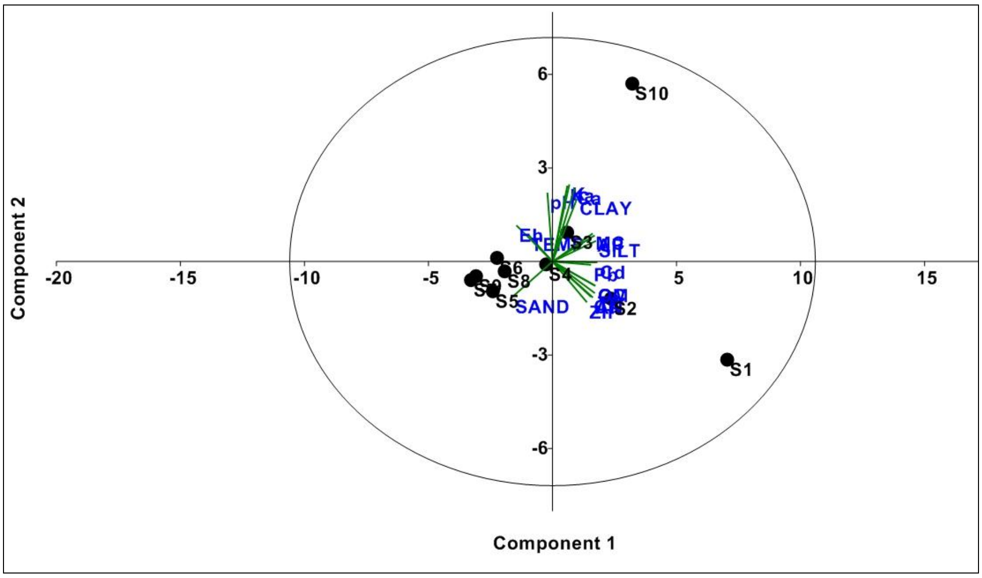

Fig. 4.22 PCA ordination of sediment parameters of Vembanad estuary during 2011-2012 period

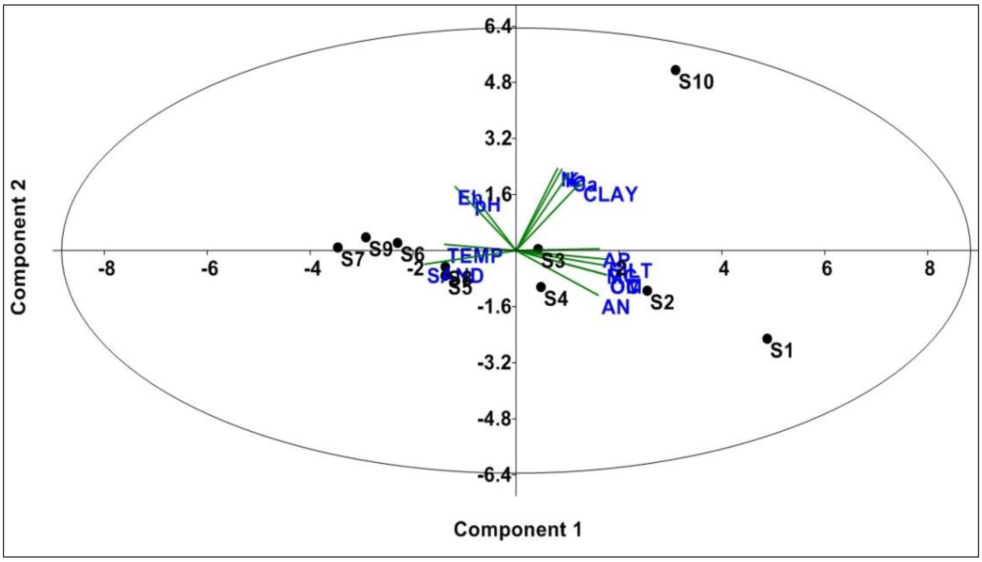

Fig. 4.23 PCA ordination of sediment parameters of Vembanad estuary during 2012-2013 period

4.3 Discussion

Sediment characteristics within the estuarine system have many insinuations for the health of the overlying water body. When the sediment is stirred up, trace metals, nutrients and organic contaminants are released into the water column and limit the light penetration essential for plant growth (Dyer, 1995). Contaminants such as heavy metals, nutrients, pesticides and herbicides have a strong tendency to get adsorbed on to fine-grained sediments, thus relating pollutant transport and storage strongly to sediment dynamics. An understanding of the sediment sources carried to estuaries, stored within and exported from estuaries/lagoons is important for explaining number of environmental issues. These include maintenance of navigational channels, primary productivity, turbidity, reduction of dissolved oxygen concentrations, transport and accumulation of particle bound nutrients and contaminants and their eventual transport to the continental shelf (Eyre et al., 1998). Coastal wetlands provide important ecological services to the coastal zone, one of which is sediment retention. The sediment deposited in estuaries was either allochthonous sediment originated from external sources such as terrestrial or oceanic or autochthonous, where sediment gets resuspended. In various ways, human activities have changed the composition of sediments in estuaries, such as increased erosion rate and changing the direction of water – flow and sediment movement. Normally the development of port or marina affects the sedimentation rate of fine sediments, so regular dredging is required for maintaining the optimum depth for navigation. Due to increased sedimentation, annual dredging is required in Cochin harbour and other regions. Similarly the coastal urbanisation increased the rate of erosion by clearfelling of mangrove forests, reclamation and land filling for construction projects especially for tourism amenities and residential requirement has perverted the Vembanad estuarine banks. As a result, the increased amount of fine particles and eroded materials may end up in estuaries. Due to tourism development large portion of mangrove swamps in the Kumarakom were cleared. The swamps were the only breeding ground for Night Herons in Kerala in the eighties. In the name of tourism development natural banks of Vembanad estuary, once covered for thick mangrove forest were removed and various types of reclamation has occurred that is difficult to be revoked. There have been numerous encroachments and illegal land filling occurring in and around the estuaries and small islands in the estuaries by violating various laws and norms, which is continuing unabated.

Sediment colour is an important property, which provides significant indication about the organic matter, drainage and redox conditions of the soil (Biswas and Mukherjee, 1992). Black colour of sediment in station 1 may be due to the increased organic matter in the station. Szafran (2004), recorded olive grey to black colour of sediments comprising of rather higher organic matter. According to Folger (1972), silt and clay are mostly olive grey and sandy layers are lighter in color. In the present investigation the silt and clay fractions were more in station 3 and 10. The oxidized sediments indicated a reddish brown colour, whereas light grey colour sediment represented the presence of terrigeneous matter, and in the effluent discharge point sediments were black in colour (Remani et al., 1980). Anoxic sediment also denoted black colour due to the presence of H_2S.

Sediment temperature is considered as one of the key environmental factors affecting the life of marine organisms. Any variations in temperature affects the geographic and bathymetric species distribution patterns (Portner, 2001) and it act as a determinant parameter that controls the reproductive cycle and duration of the planktonic larval phase and also regulate the benthic ecosystem dynamics in temperate areas (Kinne, 1970). The temporal changes in abundance of polychaetes, amphipods, and bivalves recruits corresponded to increases in sediment temperatures (Powers et al., 2002). According to Bergstrom et al. (2010) spatial and seasonal variation in sediment temperature influences the CO_2 production in lake sediments and out of the total production, 83% was released during the spring and summer season. Similarly the sediment organic carbon mineralization was controlled by temperature (Cardoso et al., 2014). The chemical processes in the sediments and the distribution of benthic organisms depended upon the sediment temperature (Ramachandra et al., 1984). In the current investigation, the most important spatial variations in temperature occurred mainly due to the differences in sampling times. The sediment temperature fluctuations are influenced by atmospheric and water temperature. Similarly it presumably increases in relation to bottom water temperatures. In the present investigation sediment temperature corresponded to ambient bottom water temperature with lowest during PM and MN season for the first and second year respectively. This seasonal variability related to the increased rainfall and river discharge during southwest and northeast MN. The sediment temperatures recorded were maximum during PRM in both the years and minimum during PM in first year. A similar condition was observed in Ashtamudi estuarine system (Soumya et al., 2011). During warmer period the sediments of the lake absorb heat from the water and during winter it transmits (Wetzel, 1975). Higher temperature accelerates the decomposition of organic materials which leads to oxygen depletion and release of total organic carbon.

The pH of sediment plays a key role in recycling of nutrients between water and sediment (Nasnolkar et al., 1996). So it can be used for predicting other chemical properties of sediments (Mohamed, 2005). In the present study the range of sediment pH agreed with previous studies in the region (Anon, 2001; Nasir, 2010) but its spatial variation was very distinct. During the current study period the lowest spatial average of sediment pH was mostly observed in southern stations. River discharge brought large amounts of allochthonous matter into the southern stations of estuary and its decomposition may reduce the soil pH. Berner (1971), pointed out that lowering of pH in the sediment could be due to release of biogenic carbon dioxide by bacterial decomposition of organic matter. When comparing the south and north zone of TMB, sediment pH was minimum in southern zone than the northern zone. This was possibly due to the decreased saline water flushing in the southern zone, where the tidal inflow completely cut-off during the closure. pyrites which on oxidation produces higher acidity. They noted that, after the construction of TMB, there was severe depletion of soil organic matter content with an increase in soil acidity in the agrarian zone of Kuttanad. According to Anon (2001), the acidic nature of sediment in southern sector of Vembanad lake was due to the exposure of relatively higher acidic subsurface basin soil, on subsequent to MN turbulence and bottom current it reduced the pH of the estuary. The highest alkaline pH in the northern most station indicated the presence of more hydroxyl ions in the sediments. The pH can become more alkaline if the soil solution contains high sodium ion, which was mainly contributed by the adjacent Arabian Sea, influenced by influx of sea water by tides. Both the years, the annual seasonal variation of sediment was varied from each other. During first year higher pH was observed during PM season, whereas in second year it was during PRM. According to Rajasegar et al. (2002), higher value of pH during PRM was due to the redox changes in the sediment and water column apart from the influence of fresh water. During 2011-2012 period, sediment pH showed a significant negative correlation ($P \leq 0.01$) with temperature ($r^2 = 0.31$).

The redox potential (Eh) is a quantitative measure of reducing power which provides a diagnostic index on the degree of anoxia. Anoxic sediments have redox potentials below -0.2 V, while typical oxygenated soils have potential of above +0.3V (Prema et al., 2012). The decomposition of organic materials reduces the Eh (vanLoon and Duffy, 2011). It depends on several factors, including diffusion from the surface of the sediment and the activities of infaunal organisms, so it can be used as surrogate measure of benthic conditions (Kroop, 2004). The negative value of Eh in majority of the stations represented a reduced condition. In Cochin estuary, Geetha et al. (2010) reported a reduced condition of sediments ranging between -11 to -645 mV. Bijoy Nandan and Abdul Azis (1997) reported Eh value of -34 to -400 mV from coconut husk retting areas of Kerala. During the present investigation the whole estuarine sediment had higher Eh value in PM season. This observation was in contrast with the seasonal variation of Eh in Cochin estuary (Geetha et al., 2010) and Manakudy estuary (Helen, 2015), where highest average was observed during MN season. Radhika (2013) observed a highest Eh value during PRM (av. -161.5 mV) and lowest during MN (av. -80.75 mV) in Cochin estuary. However a comparison with the northern stations, in both the years in the southernmost stations (station 1 and 2) showed a highly reduced condition. This was mainly due to the accumulation of large amount of decomposable organic material in this zone. After, due to increase in salinity the invasive water plants, *Eichhornia* start to sink and its decay occurred in the bottom. Earlier studies also reported the decay of macrophyte and associated high organic matter and depletion of oxygen in Vembanad estuary (Unnithan et al., 1975; Sheeba, 2000). The soils of Kuttanad showed acidic nature, as a result of excessive microbial oxidation of organic matter and anerobic decomposition (WISA, 2013). Highest Eh value in station 4 (- 599 mV) was probably due to the clam dredging occurred in the area before sampling and it disturbed the sediment and *affecting* the balance of ionic compounds. According to Whitfield (1969), Eh values around -200 mV, indicate high sulfide activity and presence of hydrogen sulfide. So in this zone, the sediment may act as a trap for electron acceptors in the overlying water and oxygen depletion may arise if water movement along the bottom is restricted. Thus the present investigation gives an alert on the reducing environment in Vembanad estuarine system. Persistence of such reduced conditions, will adversely affect the bottom fauna and flora. Correlation analysis showed that sediment Eh exhibited significant positive and negative correlation ($P \leq 0.01$) with sediment temperature ($r^2 = 0.37$) and pH ($r^2 = -0.41$) respectively.

Sediment water (moisture) is the water that is held in the spaces between sediment particles, which reflects its accumulation state and interaction strength among particles (Qiao et al., 2016). It is an important property of bottom sediments being directly related to the bulk density, porosity and organic matter content of sediments (Chandrakiran, 2012). Bottom with higher water content are more easily eroded than bottom having less water content. Water content is related to sediment porosity, an important factor controlling the diffusivity of chemical species. Benthic faunl burrowing action, deposit feeding and pelletizing activities affect the sediment water content. Mobile burrowing bottom fauna increases the sediment water content of top of fine mud and also sands (McCall and Teves, 1982). Dissanayake and Chandrasekara (2014) observed that in the mangrove environment soil water content controlled the distribution of certain benthic fauna. Similarly, the burrowing actions of benthos change the sediment water content (Qiao et al., 2016). In the present investigation higher spatial average of water content was recorded at station 10 and 2 and lowest in 7 and 9. The percentage of water content was greatly associated with the sediment texture. The higher percentage of fine sediment fraction can hold more water than coarse fraction. Lower mean percentage of water content at station 7 and 9 was associated with the sandy nature of sediments found in these stations. It was noted that compared to other stations, station 7 and 9 have sandy substratum and station 2 and 10 have increased percentage of clay and silt. According to Shaikh and Tiwari (2012), the water content was slightly higher in silty clay to clayey silt nature of sediment and presence of clay in the sediment increases the water holding capacity, whereas sandy substrate holds less water. Musale et al. (2005) also observed comparatively higher water content in silty sediments in Mavinhole Creek, Karwar. Similar type of observation was observed in the mangrove ecosystem of Kochi (John, 2003) and Sewri Mudflats, Mumbai (Shaikh and Tiwari, 2012). According to Bhat (1984), the sediment water and organic matter were linearly related. Seasonal analysis of water content showed that during first year, the highest water content was observed during MN and lowest in PRM season. Similar condition was observed in Manakudy estuary (Helen, 2015) and Shahpura lake (Anu et al., 2010). On the contrary, in Sewri Mudflats, the highest seasonal value was observed during MN and lowest during PM (Shaikh and Tiwari, 2012).

Soils and lake sediments are major identified sites for the storage of large amounts of organic carbon (Benoy et al., 2007). The settled organic carbon in the sediment reached there either originated from internal primary production (autochthonous) or from external sources (allochthonous). Most of the bottom sediments consist of terrigenous detritus, biogenic debris, and pollutants. This is a monitoring parameter analyzed in environmental investigation programs. It can influence the concentration of other compounds and affects biogeochemical processes, nutrient cycling, biological availability, and chemical transport and interactions. Organic carbon generally makes up less than 5 % of the bottom sediments except in swampy areas, fjords, or where pollutants are abundant (Folger, 1972). In the present investigation the lowest spatial mean value of organic carbon content was found in station 5 and highest in station 1 and 2. The decreased sediment organic carbon content in station 5 may be related to the increased oxygenation and remineralization of organic carbon in the sandy substratum. During 1996-1997 period, the average organic carbon content (2.7 %) in the southern zone of TMB was slightly higher (Anon, 2001). Relatively increased organic carbon in the southernmost stations may be due to the discharge of large amount of humus brought to this region by rivers, along with the sewage inputs from house boats. The southernmost stations such as Punnamada and Pallathuruthy are the major hub of house boat tourism activities in Vembanad estuary. More than a thousand houseboats have been operating in the Punnamada basin. Most of the houseboats are not using the sewage treatment facility of the District Tourism Promotion Council (DTPC), Govt. of Kerala but continue to dump the untreated waste directly in to the estuary, increasing the organic load. The decaying invasive water plant matters also generate increased organic load in the benthic part (sediment) of southern zone. The reported average organic carbon content of different estuaries of India are that in Vembanad estuary was 1.48 % (Murty and Veerayya, 1972); that in Cochin estuary was 2.06 % (Reddy and Sankaranarayanan, 1972); that in Vellar estuary was 2.32 % (Thangaraj, 1984); that in Kakinada Bay was 4.56 % (Vijayakumar et al., 1991); that in Mandovi estuary organic carbon varied from 0.1 to 3 % (Alagarsamy, 1991); that in Periyar river-estuarine area it varied from 0.1 to 2.9 % (Sarala Devi et al., 1992); that in Poonthura estuary it varied between 0.25 to 0.85 mg/g (Anila Kumary, 2001); in Cochin estuary ranged from 0.09 to 4.29 % (Geetha et al., 2010) and that in Mandovi – Zuari its ranged from 0.33 to 1.34 % during wet season and 0.24 to 0.54 % during dry season (Shynu et al., 2015). Over the period of three decades, a fourfold increase in sediment organic carbon in Cochin estuary reported by Martin et al. (2011), that could be due to autochthonous and allochthonous inputs of various sources as elaborated above.

In the present investigation higher organic carbon in the estuary was observed during the MN season. This was mainly contributed by the increased land runoff and river discharge during the MN season. Similar type of observation was reported in Mandovi estuary by Alagarsamy (1991). He mentioned that increased organic carbon during MN season was mainly due to the increased flooding and large amount of land derived organic load by estuarine tributaries. However in Poonthura estuary seasonal maximum of organic carbon content was observed during PRM and minimum during PM period (Anila Kumary, 2001). In their observation the heavy rainfall and increased river discharge and heavy flooding during the MN and PM season are not suitable for the settlement of organic matter. Whereas reduced rainfall, low river discharge and closure of mouth of estuary during PRM created a stagnant type estuary with increasing organic carbon settlement. Similar type of observation was earlier reported from the southern zone of Vembanad estuary, as during PRM season stagnant water condition support higher values of organic carbon (Anon, 2001). According to Reddy et al. (1986), the distributional status of total organic carbon was closely related to the distribution of sediment type, where sediment with low clay content was having lower in organic carbon, but with increasing clay content, the total organic carbon content also increased. Several studies revealed the clear relationship between organic carbon and sediment texture (Sarala Devi et al., 1992; Nair et al., 1993; Anila Kumary 2001; Balachandran et al., 2006). Finer particles having increased surface area can absorb more heavy metals and other contaminants containing organic matter. The spatial variation of organic carbon is in quite agreement with the widely accepted observation of several earlier reports stating that fine grained sediments generally have higher amounts of organic carbon than coarse grained ones. According to Hyland et al. (2005), the organic carbon content above 3.5% acts as a reducing factor for benthic abundance, biomass and species diversity. Harkantra (1980) noted that when organic carbon content was > 4 %, a decrease was observed in benthic fauna of the west coast of India, whereas Joydas (2009) observed that, there was no direct relationship between organic matter and benthic fauna in the west coast of India. According to Cocito et al. (1990), only a few species survive in sediments having high amount of organic input, so the species diversity decreases in the organic rich area (Albayrak et al., 2006) and maximum diversity was found in moderate organic carbon areas (Sivadas et al., 2011). According to Musale et al. (2011), polychaete abundance was lower in high organic carbon areas because of avoidance of organisms to organic matter, which indicates that high organic matter adversely affects the polychaete abundance and distribution. In the present investigation, a significant ($P \leq 0.01$) positive correlation was observed between organic carbon and water content ($r^2 = 0.35$). Similar type of observation was noted in Maranchery Kol wetland, Kerala by Vineetha (2015). Water holding capacity of sediments was improved by increased organic matter (Reddy and Delaune, 2008).

Nowadays, the organic matter enrichment in estuaries leading to cause serious ecological problems. In estuaries water column organic matter and primary production have direct connection on sediment organic matter content and benthic ecology (Zimmerman and Canuel, 2001). Organic matter content in the surface sediment forms an important source of food for benthic fauna and an elevated organic matter may result in an enhancement of benthic metabolism (Gray 1981; Meksumpun and Meksumpun, 1999). But its surplus amount and associated decomposition leads to oxygen depletion and accumulation of toxic materials which eventually affect the benthic species richness, abundance, and biomass (Pearson and Rosenberg, 1978). Organic matter differs substantially in biochemical composition and availability to consumers, depending on whether it is imported by rivers or produced locally by various primary producers (plankton, seagrasses, macroalgae, and mangroves). Exchange of organic matter across ecosystem boundaries thus has important consequences for the availability of organic matter and the relative importance of burial, mineralization, and consumption by fauna. It has often been proposed that organic matter exported from tidal wetlands enhances secondary production in the coastal zone, thus contributing to higher fishery production. High organic matter improves water holding capacity of sediments (Reddy and Delaune, 2008). Fine-grained sediment substrata generally have high organic content and chemical contaminants because it provides greater surface area for attachment of both organic and chemical pollutants (Thompson and Lowe, 2004). The sediment organic matter in estuaries is received from sources like land run off; water column production through organic productivity and influence of the seas in the estuarine part (Murty and Veerayya, 1972). According to Zhu et al. (2011), the hydrodynamic characteristics influence the re-suspension and remobilization of surface sediments and so it affects the quantity and type of buried organic matter in different depositional settings of estuarine and coastal ecosystems. Similarly organic matter distribution in lake sediments follow the same pattern of sediment distribution as that of finer sediments, silty clays and clayey silts, that have high organic matter and sand have lower organic content. In the present investigation organic matter distribution followed a similar pattern in estuarine sediment. The organic matter content of sediment ranged between 0.05 to 17 % in the present study where as during the pre-barrage phase (1969), the organic matter in the sediment varied between 0.1 to 6 % (Murty and Veerayya, 1972). Over the three decades a threefold increase in sediment organic carbon was observed. So the present study signified the probable impact of organic enrichment. As a consequence of organic enrichment, alterations in the benthic community structure and depletion of species diversity were reported by several authors (Gray, 1971; Ansari, 2000; McLeod et al., 2010; Quiroga et al., 2012; Ansari et al., 2014). Organic rich sediments are directly ingested by deposit feeders either selectively or non-selectively. The suspension feeding organisms decrease when organic load in the sediment increases and at the same time deposit feeders increases. During the pre-barrage period, in the southern part of TMB, the sand zone running along the western margin of lake, organic matter was not more than 1.5% and in the southernmost end it was 2 to 3 % (Murty and Veerayya, 1972). Based on the organic matter percentage, earlier Purandara and Dora (1987) divided Vembanad estuary into three zones: zone 1 (from Azhikode to Vypeen), where organic matter was moderate amounts; zone 2 (from Vypeen to Thaneermukkom) contained high organic matter; zone 3 (Thaneermukkom to Alappuzha) composed of low organic matter. They noted that, the fresh water conditions and coarser and sandy sediments were responsible for the lower amounts of the organic matter in the southern part of estuary. In the present investigation, south of TMB (station 4 to 6) the average value reached up to 2% and southernmost Punnamada region, organic matter was >11 %. Therefore it clearly indicates that southernmost part of Vembanad is facing severe organic enrichment over the years. Similarly, the black colour of the sediment in Punnamada region indicated high organic matter. Szafran (2004) recorded olive grey to black colour of bottom sediments that comprised of comparatively higher organic matter. Increased human impacts such as sewage discharge from house boats, pesticides and fertilizers run-off from Kuttanad agrarian zone and municipal wastes from Alappuzha town, land reclamation for tourism activities etc. were more pronounced in the region. The failing septic systems along the estuarine bank also led to direct sewage pollution in the water body. During both the years, correlation analysis showed that organic matter showed a significant positive correlation with available nitrogen and silt ($P \leq 0.01$). At the same time a significant negative correlation was observed with sand ($P \leq 0.01$). Significant negative correlation of organic matter with sand was observed in Fort Cochin beach (Anu Pavithran, 2015).

Total carbon concentration is a fundamental parameter for describing the abundance of organic matter and inorganic carbon in sediments (Meyers and Lallier-Verges, 1999). In carbon rich sediments organic carbon constituted about 60% of total carbon, whereas in low carbon sediments organic carbon constituted 80 – 90 % (Renjith, 2006). The annual average of total carbon in the estuarine sediment was comparatively higher in the present investigation. Similar to organic carbon distribution, total carbon was comparatively higher in the southernmost stations and also it did not show any marked seasonal fluctuations in the whole estuary. Similar findings were observed in Cochin estuary. Renjith (2006) and Ratheesh Kumar et al. (2010) did not observe any seasonal trend of total carbon in the sediments of Cochin estuary, where the value ranged between 0.06 to

4.05 % and 0.38 to 3.04 % respectively. Similar to Southern most stations, station Aroor (station 10) was also having higher total carbon, where increasing sewage discharges from Kochi metro city and waste water from sea food industries forms a major threat in this zone. Similarly the annual dredging of navigation channel by the Cochin Port Trust also seriously affected the sediment quality in the region. In Ponggol estuary (Singapore) Nayar et al. (2007) observed that the total carbon in the sediment increased during the post dredged phase (34.6 ± 12.0 mg/Kg) than prior to dredging (18.5 ± 3.7 mg/Kg) and during dredging (20.2 ± 3.5 mg/Kg) period. Due to dredging a shift in sedimentary carbon from inorganic carbon to organic carbon was also observed as dredging operations exposed the organically enriched historically contaminated layer. Total carbon possessed a significant ($P \leq 0.01$) positive correlation with organic carbon (r^2 = 0.76), organic matter (r^2 = 0.77), available nitrogen (r^2 = 0.6) and silt (r^2 = 0.6). Where as a significant ($P \leq 0.01$) negative correlation was observed with Eh (r^2 = - 0.43) and sand (r^2 = -0.53). In freshly deposited sediments of Hoogly estuary Mukherjee and Kumar (2012), reported 8.2 ± 2.8 mg/g of total carbon and observed good correlation with sediment grain size. In the present investigation, total carbon showed a significant positive correlation with silt and negative correlation with sand. This result was contrasting with the observation of Jacob et al. (2013) in the shelf sediments of Bay of Bengal, where total carbon was positively correlated with sand and negatively with silt. According to Li and Pang (2014), clay particles possessed higher organic carbon enrichment ability than silt and sand and organic carbon concentration was lowest in sand. Gireesh Kumar (2015) noted a significant positive correlation of total carbon with organic carbon, silt and total nitrogen in Cochin estuary.

Phosphorus (P) is an important nutrient which has a significant role in global biogeochemical cycles. In estuarine environments P is subjected to an extreme chemical gradient. The sediment may act as a phosphorus sink, thus liberating phosphate under specific conditions. It can be found in the sediment matrix in the form of calcium, iron, aluminum complex salts and organic species, or adsorbed onto the surface minerals (Pettersson et al., 1988), but the amount of mobile or bioavailable phosphorus in the surface sediments is an important parameter for predicting future internal loading and its release to the water column (Rydin, 2000). Changes in physical, chemical, and biological conditions can induce release and transformation of organic and inorganic phosphorus from sediments (Tallberg et al., 2008). According to Schenau and De Lange (2001), the information of phosphorus speciation is important to monitor the environmental variations, phosphorus cycle, bioavailability, carbon, nitrogen, sulfur and oxygen cycles. Due to various extraction procedures, evaluation of phosphorus fractionation in sediment was not easy. Due to many different extraction schemes and their modifications it is quite difficult to compare literature data in the field of phosphorus fractionation (Nasir, 2010).The deposited form of phosphorus in the sediment is not accessible to the organisms, but due to various biogeochemical reactions, some forms gets converted to bioavailable form as dissolved orthophosphate (Singh et al., 2015). This phenomenon occurred due to the Fe(III)-bound-P dissolution when conditions change from oxic to anoxic, to the desorption of adsorbed phosphate when fine particles are resuspended (Andrieux-Loyer and Aminot, 2001) and to the assimilation of organic phosphorus by the bacterial community (Aviles et al., 2006). Due to high tidal energy and tidal flows in many estuarine systems, the sediment material often gets resuspended and translocated (Boynton et al., 1995). The released P may significantly increase the biologically available pool of P in the water, counteracting decrease in the external loads. In some estuaries, intrinsic supplies of phosphate by desorption, reduction or mineralization, may exceed external supplies by a factor 2 to 4 due to high concentrations in sediments (Schlungbaum and Nausch, 1990). Therefore, the knowledge of sedimentary phosphorus speciation and phosphate fluxes is essential to understand the functioning of a coastal system under anthropological constraints. Increased P load in estuaries occurred as a result of catchment sources brought about by the river or direct discharge into the estuary (Billen et al., 1991). Similarly the transport of phosphorus in sediment-water interface was controlled by the soil characteristics (Wang et al., 2013).

During the study period, the highest spatial average of phosphorus was observed in Punnamada (station 1) and Aroor (station 10) region. In Punnamada, the increased river discharge from rivers Pamba, Manimala and Achenkovil may contribute a major share of phosphorus. According to Singh et al. (2015) rivers play a great role in the transfer and deposition of sediment load in to the deltaic lowland, in which detrital apatite might be a key source of phosphorus in the sediments. The elevated level of phosphorus in Aroor region was mainly supported by the increased clay fraction, which absorb large amount of phosphorus. Earlier Sankaranarayanan and Panampunayil (1979) reported the increasing concentration of phosphorus in the muddy substratum in Vembanad estuary. During the pre-barrage phase, Murty and Veerayya (1972) observed a decrease in phosphorus content from estuarine mouth to fresh water zone in Vembanad estuary. This was in contrasting with the present observation. The effect of anthropogenic disturbances in the southernmost zone of Vembanad estuary was more severe over the last few decades especially due to urbanization and tourism development. In the southern zone discharge of phosphatic fertilizer residue from Kuttanad region also plays a key role in maintaining the phosphorus concentration in sediment. Kuttanad agrarian zone consumed 5044 tonnes of phosphorus fertilizer and 500 tonnes of pesticides/fungicides/weedicides annually (James, 2011). Apart from this, oil pollution from tourism and navigational boat, unplanned construction and industrial input, domestic sewage, hanging toilets along the banks of the estuary and open defecation into the water body had great impact. Nasir (2010) also observed the relatively increasing magnitude of phosphorus in the Punnamada region and mentioned about the continuous discharge of urban waste and organic load from Alappuzha town. The lowest concentration of available phosphorus in station 6 was mainly associated with decreasing organic carbon and increasing sand fraction in the sediment of the region. Annual seasonal variation of available phosphorus was not showing any distinct trend. The lowest value was observed in PRM and MN season in both first and second year respectively. According to Nasnolkar et al. (1996), during MN season the topmost layer of sediment removed due to heavy flooding or deposition of sand or re-suspension of phosphate from the sediments to the overlying water column caused reduced value in Mandovi estuary. Similar observation was reported in Cochin estuary (Sankaranarayanan and Panampunayil, 1979); Vellar estuary (Ragasegar et al., 2002) and Manakudy estuary (Helen, 2015). During the study period the available phosphorus ranged from 0 to 470.2 mg/Kg. The average available phosphorus in the Pulicat, Ennore and Adayar estuary were 36 mg/Kg, 125 mg/Kg and 163 mg/Kg respectively (Joseph et al., 1992). The bioavailability of phosphorus content in the Pearl River estuary of China varied from 128.6 mg/kg to 442.9 mg/kg, indicating a potential risk of eutrophication (Wang et al., 2013). Nasir (2010), estimated >400 mg/Kg ortho-phosphate and >850 mg/Kg of total phosphorus in the Alappuzha region, and forewarn the continued threat of eutrophication in the estuary for a longer period even if the phosphorus source into the estuary gets limited. Balachandran et al. (2002) reported that Cochin estuary is receiving 42.4×10^3 mol d^{-1} inorganic phosphate and 37.6×10^3 mol d^{-1} organic nitrogen through Periyar river. The export from coastal waters is only 28.2×10^3 mol d^{-1} inorganic phosphate and 24×10^3 mol d^{-1} of inorganic nitrate, thus the estuary acts as a nutrient sink. The phosphorus released from the sediment maintains the eutrophication and cycles between overlying water and sediments through algal growth, organic deposition, decomposition and release (McManus et al., 1997). So, eutrophication process could persist for a longer period even after the phosphorus inputs are stopped due to remobilization and phosphorus is bound to sediments (Sondergaard et al., 1993). So there is a clear chance of incidence of eutrophication in Vembanad estuarine system. During 2011-2012 period, available phosphorus displayed a significant positive correlation ($P \leq 0.01$) with organic carbon ($r^2 = 0.34$), silt ($r^2 = 0.49$), available nitrogen ($r^2 = 0.43$), sodium ($r^2 = 0.34$), potassium ($r^2 = 0.42$) and total carbon ($r^2 = 0.42$). Available phosphorus showed significant ($P \leq 0.01$) negative correlation with Eh ($r^2 = - 0.3$) and sand ($r^2 = -0.47$). During 2012-2013 period, correlation analysis of available phosphorus revealed a significant positive correlation with available nitrogen ($r^2 = 0.48$), organic carbon ($r^2 = 0.4$), organic matter ($r^2 = 0.4$) and silt ($r^2 = 0.36$). A significant negative correlation ($P \leq 0.01$) was exhibited between available phosphorus and sand ($r^2 = -0.42$). In Kuttanad ecosystem available phosphorus has significant correlation with organic carbon and available nitrogen (Sylas, 2010). In the earlier studies in Cochin estuary a significant positive correlation of available phosphorus with organic carbon was not observed (Reddy and Sankaranarayanan, 1972). Distribution of phosphorus is also determined by the texture of the sediments. Purandara (1990), Padmalal (1992) and Ampili and Shiny Sreedhar (2016) observed a linear relationship of phosphorus towards fine grained sediments, viz., silty clays and clayey silts. Results of the present study agreed with the relationship of phosphorus distribution and sediment texture.

Nitrogen is essential for life, but most of the nitrogen in the biosphere is unreactive and unavailable to organisms. Unreactive nitrogen can be converted into reactive nitrogen by natural processes and human activities (Castro et al., 2003). In recent decades, due to increased human activities along with greater use of nitrogen fertilizer and fossil fuel combustion have greatly modified the balance of the nitrogen cycle in estuarine and coastal environments (Bricker et al., 2008) and has caused widespread eutrophication, hypoxia and anoxia, loss of biodiversity, and increased harmful algal blooms (Anderson, 2002). Most of the nitrogen leading to eutrophic estuaries comes from land-based sources and not from the ocean, so the coastal nutrient pollution is mainly from the modification of nitrogen sources made by human controlled alterations with the natural nitrogen fixation (Nixon et al., 1996). From this perspective, human-controlled processes may now exceed the natural processes or at least are approaching this if the natural rate of nitrogen fixation is at the higher end of the range suggested by Cleveland et al. (1999). Sediments are important sites for organic matter accumulation and nutrient cycling in estuarine and coastal environments (Dunn et al., 2012). The methods of sediment nitrogen mineralization are closely linked with sediment nitrogen input and less, which has great ecological role in maintain the ecosystem health (Bai et al., 2012). In the present investigation available nitrogen (0.001- 0.21 %) was comparable and slightly higher to that from Thengapattanam estuary (0.012 to 0.052 %) reported by Anitha and Kumar (2013), that from Maranchery wetland (0.0015 to 0.375 %) by Vineetha, (2015) and from Kuttanad wetland (0.004 to 0.088 %) by Sylas, (2010). The average available nitrogen content in estuaries of Pulicat, Ennore and Adayar were 67 mg/Kg, 69 mg/Kg and 93 mg/Kg respectively (Joseph et al., 1992). The indiscriminate and unscientific application of fertilizers, insecticides, oil pollution from tourism and navigational boat, unplanned construction and industrial input, domestic sewage, hanging toilets along the banks of the estuary and open defecation into the water body have all impacted to the nitrogen loading in the system. During both the years, the maximum spatial average of available nitrogen was observed in station 1. The increased river and agricultural discharges may form a major source of nitrogen in the estuary. Apart from this, sewage discharges from houseboats and municipal waste from Alappuzha town also contribute a major share. The sediment nitrogen was mainly derived by decomposition of plants and animals or plankton or anthropogenic sources such as chemical contaminants, fertilizers or organic rich waste (Avramidis et al., 2015). Seasonal trend of available nitrogen showed that the minimum seasonal average was observed during MN season. Similar type of result was earlier reported in Kuttanad wetland system by Sylas (2010). Comparatively higher concentration of available nitrogen in the southern zone may be attributed to the decomposition of settled water hyacinth (*Eichhornia crassipes*) in the region. Decomposition of Eichhornia was also contributing to increased N_2 load in the sediment through the absorption and recycling process (Sylas, 2010). The spatial variations in the available nitrogen were mainly attributed to the variation of bacteriological processes in the sediment. The distribution and cycling of nitrogen species in sediments has primarily been attributed to various bacterial metabolic processes (Fenchel and Blackburn, 1979). During 2011-2012 period, available nitrogen showed a significant ($P \leq 0.01$) positive correlation between organic carbon (r^2= 0.41) and silt (r^2 = 0.5). It also exhibited a significant negative correlation with sediment temperature (r^2 = -0.37), Eh (r^2 = -0.55) and sand (r^2 = -0.43). During 2012-2013 period, correlation analysis of available nitrogen were similar to that of 2011-2012 period. During both the years, available nitrogen showed a significant positive correlation with sediment organic carbon. These results were in agreement with several studies that, organic carbon plays a major role in nitrogen mineralization process, because organic carbon forms the major energy source for heterotrophic microbial metabolism (Keuskamp, 2013). In oxygen – depleted area with high organic matter, mineralization of organic matter occurs and leading to release of ammonium. So nitrogen mineralization becomes a significant internal source and accelerates the process of eutrophication in the system (Lin et al., 2016).

Sodium is an important constituent in sediment. The carbonates, sulphates, nitrates and chlorides of sodium are found abundantly in nature.

Sodium and potassium are the most important minerals occurring naturally. Na is more mobile than K and dominates the natural solutions (Milliot, 1970). Na and K are not precipitated by hydrolysis, it is possible that they are fixed up in clay minerals either by adsorption and/or cation exchange (Abdulla Bava, 1996). Weathering of rocks, in addition to sewage and industrial effluents forms its main source (Annalakshmi and Amsath, 2012). In the present investigation average sodium content was 2254.8 mg/Kg during first year and 3068.8 mg/Kg during second year. On comparing the sediment sodium content of other estuaries, the Vembanad estuary had comparatively lower values. Ramanathan et al. (1999) reported that the maximum average level of sodium was in Pichavaram mangrove, which was observed as av. 28205.8 mg/Kg, in Vellar estuary (av. 29620 mg/Kg) and in Coleroon estuary (av. 35061 mg/Kg). The large volume of riverine inflow and complete flushing out of estuarine water during MN season may reduce the sodium level in the estuarine sediment. In Vembanad estuary, the fluctuations in salinity are of extreme kind ranging from entirely riverine to entirely saline. The high runoff months are characterized by monsoonal spells causing intense flushing (Shivaprasad et al., 2013c). Purandara (1990) also noted that the concentration of sodium was comparatively higher in the northern part of the estuarine mouth and minimum percentage was observed in the southern most part of the estuary. The present findings agree well with Purandara (1990) as maximum sodium level was observed in the northern most station 10 (Aroor). This was mainly due to the saline interaction from tidal water from Arabian Sea. According to Sayles and Mangelsdorf (1977), increased level of sodium in the saline region was mainly due to the replacement of calcium by sodium. According to Abdulla Bava (1996), higher content of Na in sediments of estuarine and marine region of Vembanad estuary was texturally controlled one. As these sediments are highly muddy in nature with more clay content than river, Na gets fixed more to clayey minerals. He also noted that, maximum Na concentration in surface sediment was 7989 ppm in river, 9989 ppm in estuary and 11985 ppm in marine sediments of Cochin estuary. Seasonal analysis showed that the lowest seasonal value of sodium was during MN. A similar observation was noted in Manakudy estuary (Helen, 2015). Correlation analysis revealed that sodium exhibited a significant ($P \leq 0.01$) and positive correlation with clay ($r^2 = 0.37$) and potassium ($r^2 = 0.69$). Simultaneously it exhibited a significant negative correlation with sand ($r^2 = 0.39$). In the present investigation, significant positive correlation with sodium and potassium denoting that they are derived from the same source. Similar type of correlation was observed in Pichavaram sediments (Ramanathan et al., 1999). According to Alagarsamy and Zhang (2010), high degree of correlation of elements in the coastal environment indicates the identical behavior in its distribution.

In sediments, the leachate from the agricultural field and the canal waters supply potassium. The polluted water containing domestic waste normally increases the potasium level (Sobha et al. 2008) as well as the lechate from untreated sewage. When compared to sodium, the overall concentration of potassium was lower in Vembanad estuarine system. In the Pichavaram mangrove ecosystem, the sodium was dominant (80%) than potassium and higher values were observed during non MN periods (Ramanathan et al., 1999). Similar type of observation was noted in the Vembanad estuarine system, as sodium was dominant than potassium. In Ashtamudi estuary, the lowest mean seasonal value of potassium was observed during MN and highest during PRM (Ampili and Shiny Sreedhar, 2016). In the present investigation the seasonal variation of potassium varied annually. During first year the lowest average value of potassium was observed during PRM and highest during MN, whereas in the second year, lowest value was recorded during MN season and the highest in PRM season. Similar to sodium, higher value of potassium was found in the estuarine mouth region. Higher values in this region were mainly due to the influence of seawater, as this station 10 (Aroor) was situated close to the Cochin barmouth. Similar type of observation was noted by Purandara (1990). According to Evangelou (1998), higher salinity in pore waters caused higher adsorption of sodium and potassium, which leads to its higher concentrations in sediments. In the present investigation potassium showed a positive correlation with clay fraction of sediment. Similar type of observation was reported in Ashtamudi estuary (Ampili and Shiny Sreedhar, 2016). During 2011 – 2012 period, correlation analysis showed that potassium exhibited a significant ($P \leq 0.01$) positive correlation with silt ($r^2 = 0.37$) and clay ($r^2 = 0.49$) and negative correlation with sand ($r^2 = - 0.5$). During 2012 – 2013 period, correlation analysis revealed that potassium showed a significant positive correlation between clay ($r^2 = 0.67$) and significant negative correlation between sand ($r^2 = - 0.47$).

Calcium is the major source of biogenic carbonates, which is abundant and main constituent in shallow marine biotic components. It also plays an important role in the marine biogeochemical cycle. In the present investigation calcium level in sediment were comparatively higher than potassium. This was mainly due to the presence of large scale deposit of white lime shell in the depths of Vembanad estuary. In the Manakudy estuary Kumar and Sheela (2013), observed an elevated level of calcium in the sediment samples having molluscan and foraminiferan shells. The average calcium level in the sediments of Vellar estuary was 9658 mg/Kg (Ramanathan et al., 1999), 18123 mg/Kg in Coleroon estuary (Ramanathan et al., 1999), 2654 mg/Kg in Vembanad estuary (Nasir, 2010) and 2398 mg/Kg in Manakudy estuary (Kumar and Sheela, 2013). Similar to sodium and potassium concentrations, the highest spatial average value of calcium was observed in the northern estuarine station. Such type of observation was earlier reported by Purandara (1990). He also reported an elevated value of calcium in the Alappuzha region. But in the present investigation, such increasing level of calcium was not distinct. Due to the constant operation of Thaneermukkom barrage, the incursion of saline water through tidal inflow was restricted towards the southern zone. This altered the environmental conditions of the estuary and created an oligohaline condition in the southern zone which might affect the biogenic activity. During both the years, the highest seasonal values were observed during PRM and lowest values during MN. According to Pillai (1991), the heavy rain during MN period resulted in lowering of cations mainly due to the substantial leaching that occurred during MN period. It was also noted that among other elements calcium exhibited maximum variation. During 2011-2012 period, correlation analysis revealed that calcium showed a significant ($P \leq 0.01$) positive correlation with potassium (r^2 = 0.8), silt (r^2 = 0.4) and clay (r^2 = 0.47) and also it exhibited a significant negative correlation with sand (r^2 = - 0.52). Similar to 2011-2012 period, calcium exhibited a significant ($P \leq 0.01$) positive correlation with clay (r^2 = 0.58) and potassium (r^2 = 0.9). Simultaneously it exhibited a significant ($P \leq 0.01$) negative correlation with sand (r^2= - 0.53) during 2012-2013 period. Pillai (1991) noted that the concentration of cations showed inverse relationship with sand and direct relationship with silt fraction. Similarly it showed positive correlation with potassium but in the coastal sediments of India calcium is inversely correlated with potassium (Alagarsamy and Zhang, 2010).

A sediment substrate is an unconsolidated surface that helps the organisms to live in or on the surface (Patel and Desai, 2009). The spatial variation of sand, silt and clay particles and its ratio were used for understand and envisage the distribution of benthic organisms (Hossain, 2011). According to Sesamel et al. (1986) the division of sediment in estuaries and near shore region is linked to the source and texture of sediment loaded and also the topography of the particular region. The sediment grain size correlates with the quantitative distribution of benthos (Ansari et. al., 2012). Any change in chemical properties of the sediments can affect benthic communities, including their survival. Anthropogenic originated materials from dredged wastes, other desecrate materials from the urban surroundings changed the sediment composition and lead to the enrichment of organic matter (Robin et al., 2012). The burrowing activities of various benthic organisms such as crabs, polychaetes and gastropods in the muddy substratum increases the sediment water contents and affects the sediment textural composition (Davis, 1993). Whereas the tube-dwelling polychaete (*Diopatra cuprea*) increases the sediment accumulation rate and decreases the erosion rate and ultimately stabilize the bottom (Sarkar et al., 2005). The sediment particle size analysis had a number of ecological and environmental associations. The important aspect was that, the sediment particle size had a strong influence upon the macrobenthic distribution (Alongi, 1990). Increased abundance and diversities of macrobenthic organisms were observed in the silty and clayey (finer particles) substratum characterized with higher amount of organic matter and reduced hydrodynamic features (Sarkar et al., 2005; Gray and Elliott, 2009). The increased surface area of fine particles retained higher percentage of organic matter (Flemming et al., 1996; Venkatramanan et al., 2013). Based on the percentage composition of sediment type in Vembanad estuary, the sediment composed of five main types of sediment fractions such as sandy, silty sand, clayey sand, sandy silt and clayey silt. Of these, silty sand and sandy fractions were the major types.

The spatial annual variation of sediment fraction was very distinct during the study period. In both the years, the silt fraction was highest in station 1 and 2. But its percentage contribution varied distinctly. Compared to first year, during the second year the silt fraction of sediment in majority of the stations increased considerably. Sediment texture analysis in the present study reveals that, southernmost stations of the Vembanad estuary had high silt content, where as northern stations was sandy in nature. From the annual comparison of sediment it was found that during second year sediment composition of station 1 and 2 was silty in nature, whereas in station 6 to 9 had more than 70% of sand. In the southern side of TMB, monthly percentage contribution of sediment particles were varied, where during most of the period, except in March, April and May the substratum was silty sand in type; sandy silt during March and clayey silt in April and May (Arun, 2009). But, in the northern side of TMB, sand fraction was dominant. In the present investigation when comparing the average percentage contribution, the substratum was observed as clayey sand and silty sand in type during first and second year periods respectively. In the entire months sand fraction was contributing more than 60% in the sediment particles except in March 2011, 40 %. This type of substratum was more preferable for clam fishery. In Vembanad estuary black clams, *Villorita* occurred in a mixture of fine sand, clay and silt and distributed over wide areas (Suja and Mohamed, 2010). Textural characteristics of sediments in the northern most station (Aroor) appeared as clayey silt in nature. Pillai (1977), proposed four major sedimentological zones in Cochin estuary: (i) area covered by clayey silt with very little sand, ii) area with dominance of sand fraction, (iii) area covered by sand, silt and clay in more or less equal proportions and (iv) area covered by sandy mud. He found that nature of substratum was found to be another important factor influencing the distribution and abundance of bottom fauna. During 1994-1996, the sediment texture analysis on the southern zone showed that sand was the major fraction (44.7%) and also noted a gradual increase of clayey fraction in the sediment (Anon, 2001). Silty-sand condition was observed in the pre barrage phase of Vembanad estuary (Murty and Veerayya, 1972). Continuous annual dredging operation takes place especially in the Cochin harbour area of the estuary, which increased the silt fraction (suspended sediment particle) and siltation in various parts of the estuary leading to decreasing depth and declining fishing activities in various parts of the estuary. From the grain size analysis, Purandara and Dora (1987) divided Vembanad estuary into three zones. The northern zone (Azhikode in north to Perumbalam in south) contained silty clays. Central zone (Perumbalam to TMB), composed of mixture of sand, silt and clay in varying proportions and noted as sand content was much higher. Southern zone (TMB to Alappuzha) consisted of coarser grained sediment when compared to the northern zone but finer than central zone. However, in the present investigation, the region between Perumbalam to TMB was entirely shifted to sandy substratum. During 2011-2012 period, correlation analysis revealed that silt fraction showed significant ($P \leq 0.01$) positive correlation with organic carbon ($r^2 = 0.52$) and clay ($r^2 = 0.3$). Similarly the sand fraction showed a significant negative correlation with organic carbon ($r^2 = - 0.45$), clay ($r^2 = - 0.63$) and silt ($r^2 = - 0.92$). Similar to 2011-2012 period, correlation analysis of sand fraction showed significant ($P \leq 0.01$) negative correlation with organic carbon ($r^2 = - 0.65$), clay ($r^2 = - 0.61$) and silt ($r^2 = - 0.94$) during 2012-2013 period. During second year, correlation analysis of silt fraction showed significant ($P \leq 0.01$) positive correlation with water content ($r^2 = 0.39$), organic carbon ($r^2 = 0.64$) and clay ($r^2 = 0.31$) and clay fraction showed significant positive correlation with organic carbon ($r^2 = 0.3$). Overall, correlation analysis revealed that sand, silt, clay and organic matter were significantly related with each other. Similar observations were noted in Brunei estuarine system, northwest coast of Borneo (Hossain et al., 2014), where a strong negative correlation of sand fraction towards silt, clay and organic matter and also a positive correlation between organic matter and silt. Balachandran et al. (2006) reported a strong correlation between organic carbon and clay and silty – clay type fraction of sediment with organic carbon in Cochin estuary.

High organic matter (2.062 to 5.098 %) was reported in the nearshore sediments from Azhikode to Alappuzha, where the sediments are relatively finer in size composed of clays and silty clays (Purandara and Dora, 1987).

According to Sanders (1956), most of the clay minerals except kaolin bind organic matter and area with high clay percentage is capable of holding increased proportions of organic matter. But it was not supported by Harkantra et al. (1982), when he mentioned that even though fine particles of clay containing ample amount of food but it causes clogging of the feeding apparatus of the filter feeders. According to Rodrigues et al. (1982), the coarse sediment support rich fauna while the finer sediments do not support much fauna due to its relatively poor nutrients. Ingole et al. (2002) detected that more benthos were sustained by medium sand grain size.

The majority of the sediments in coastal estuarine zones are enriched with contaminants from municipal, industrial and surface run-off etc. Heavy metals are toxic and carcinogenic to living organisms (Clement et al., 1995). Because of its toxicity and non biodegradability, the presence of heavy metals in aquatic system cause great anxiety. The increased discharges from mining, electroplating works, painting and printing works, automobile battery, and petrochemical industries contain high levels of copper, zinc, cadmium and lead (Dali-youcef et al., 2006; Mohan et al., 2007). Due to increasing industrialization and urbanization, heavy metal pollution cause serious threats in the coastal waters of tropical and subtropical countries (Ratheesh Kumar et al., 2010). Concern of people over the heavy metals is increasing day by day mainly due to its elevated level in the aquatic ecosystems and also its higher chance of entering the food chains (Ivan et al., 2011) and accumulation in biosystems (Lokeshwari and Chandrappa, 2006). Due to various physico-chemical processes in the aquatic environment, river estuarine sediment forms an important sink for heavy metals. Sediment linked metals cause serious threat to detritus and deposit feeding benthic fishes and act as sources of contamination when the metals are released into water and impose serious impacts on living organisms and act as good indicators of environmental pollution (Asa and Rath, 2014). Detection of elevated levels of heavy metals in the aquatic system is increasing nowadays, which leads to bioaccumulation and bio-magnification of various metals. An alarmingly high concentration of copper and zinc was observed mainly in the Punnamada region, where Zn concentration ranged from 9 to an increasing concentration of upto 8950 mg/Kg. Earlier, in the estuarine sediment of Vembanad an average Zn concentration of 53.3 ppm (Purandara, 1990) was observed. He noted a gradual increase in the concentration of Zn (70.83 ppm) towards south of Vembanad, especially in the Alappuzha region. In the southern zone of Vembanad Zn concentration varied between 103 to 305 mg/Kg whereas in the northern zone (including the industrial area) it varied from 42.7 to 3157.6 mg/Kg (Nasir, 2010). Thus the results clearly denote that the sediments of Punnamada region were heavily polluted with Zn. Throughout the seasons, Cadmium concentrations in the sediment depicted an heavily polluted condition. Presence of Pb, the non-essential element in the biological system, is extremely harmful and is very difficult to remove once it enters the system (Raveendran and Sujatha, 2011). In the present study the higher range of Cu (3-232.7 mg/kg) and Zn (9-8950mg/kg) denoted a higher chance of anthropogenic impact on the distribution of these metals in the estuary over the years (Table 4.2). Increased concentration of Zn affects the aquatic animal health, mainly in fishes as it causes osmoregulatory dysfunction, ultimately leading to its death. Large number of rusted iron boats (both navigational and tourist boats) festered along the canals of Alappuzha and also the repairing activities mainly painting of house boats etc. contribute a major share of such metal pollution. Apart from this, municipal runoff and other developmental activities forms the next contestant in anthropogenic enrichment of heavy metals in the estuarine system. Ahamed et al. (2011) reported the highest concentration of heavy metal in the macrobenthic samples collected from the Sundarbans mangrove areas, which was impacted in the Shipyard area of Khulna and Khalishpur, Bangladesh where ship breaking and repairing activities are widely carried out. Harikumar et al. (2009) observed southern part of Vembanad lake is facing heavy metal pollution. The deposition rate of heavy metals in the lake increases with time and various heavy metals follow the order of Hg < Cd < Cr < Pd < Cu < Ni < Zn < Mn < Fe in sediment. Based on the sediment quality guidelines, the rate of pollution in the lake was moderate for copper (16.73- 56.13 mg/kg), moderate to high rate for zinc (103.39 to 305.29 mg/kg) and nickel (36.53 to 74.47 mg/kg) and non polluted to moderate for lead (0.61 to 80.03 mg/kg) (Harikumar et al., 2009). Studies of Ratheesh Kumar et al. (2010) observed a very strongly polluted condition for Cd (ND to 11mg/kg) and a moderately polluted condition for Zn (51.93 to 716.93 mg/kg) in the sediments of Cochin estuary. In Cochin estuary, the range of value reported for the metals Cd, Cu, Pb and Zn, are 0.05- 40.7 mg/kg, 0.28-124 mg/kg, 0.25-190 mg/kg and 3.1-2230 mg/kg, respectively (Anu et al., 2014).

During 2011-2012 period, high loading of silt, organic carbon, organic matter, available phosphorus, available nitrogen, water content, copper, lead and zinc, in the PCA results denoted the input of these components from same sources. Thus it was implicit that sediment physicochemical characteristics of Vembanad estuary are mainly controlled by the impact of organic loading from various point and nonpoint sources. The freshwater input from five rivers, the urban discharges from Alappuzha and Cochin town played a significant role in determining the water quality status of the estuary. During the pre-barrage phase Murty and Veerayya (1972) noted comparatively low organic carbon in the southern part of Vembanad estuary especially Punnamada. This signified the buildup of organic carbon over the years in this zone. In the present study, the observed significant positive correlation between silt, organic carbon, organic matter, total carbon, available phosphorus, available nitrogen, water content, copper, lead and zinc reflected their increased adsorption by silt particles. Similar results were earlier reported by Paneer Selvam et al. (2012) in Vembanad lake. So due to the increased discharge of silt fraction from rivers together with sewage discharge from urban areas, houseboat wastes, agricultural runoff, and industrial wastes makes the estuarine sediment a major sink for these materials. Thus, a clear spatial gradient was observed with high organically enriched sediments in the southern sector of Vembanad estuary. Based on the PCA analysis such type of spatial gradient was observed in Rio de la Plata estuary, South America by natural and human induced pressures (Venturini et al., 2012). The strong loading of second component can be attributed to their source of origin or strong influence by seawater intrusion from Arabian Sea. So the northern zone behaved as a distinct environment. However, the PCA results analysis were almost similar for both the years of the study.

Table 4.2 Comparison of heavy metals in surface sediments of Vembanad estuary especially in the Cochin estuary over the years

Location	**Zn (mg/Kg)**	**Cd (mg/Kg)**	**Cu (mg/Kg)**	**Pb (mg/Kg)**
Purandara, 1990	34.5 – 70.83	NM	51-271	NM
Rajamani Amma, 1994	29.29 – 290.5	ND –10	31.87 - 333.78	20-120
Martin et al., 2012	10.0–2233	0.2–40.7	3.6–123.5	6.8–99.6
Balachandran, et al., 2005	92–1266.0	0.59 –14.9	5.4–53.2	19.3–71.3
Dipu and Kumar, 2012	72.75 – 1306.5	1.375–24.75	6.36 - 655.125	12-79.625
Ratheesh Kumar et al., 2010	51.93–741.93	ND-11	0.28-41.8	ND-34.5
Nasir and Harikumar, 2011	70.07 –1963.67	0.27 - 26.35	38.87 - 1723.75	21.70 - 162.59
Harikumar et al., 2009	103.39 – 305.29	0.07 - 2	16.73 – 56.13	0.61 – 80.03
Present study	9-8950	ND-3.1	3-232.7	12-81.7

NM – Not measured; ND - Not detectable

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Chapter 5

ECOLOGY AND DISTRIBUTION OF SOFT BOTTOM BENTHIC COMMUNITIES OF VEMBANAD ESTUARY

Contents

5.1 Introduction

Taxonomically and morphologically meiofauna form a diverse group including a wide variety and range of invertebrate fauna. They constitute an important component of benthic habitat and they exhibit high abundance, diversity, and productivity in many sedimentary habitats and play an important role in benthic food webs. The most important meiofaunal groups in the shallow estuarine areas and coastal marine habitats are composed of nematodes, harpacticoid copepods, foraminifera etc. Meiofaunal organisms play a great role in benthic food webs as they act both as consumers and producers. They consume on detritus, diatoms and algae and also feed upon small metazoans (Mascart et al., 2013); simultaneousily they form a major food source for macrobenthos and fishes (Lebreton et al., 2011; Carpentier et al., 2014). Normally meiofauna feed on benthic microalgae, other microbes, and various detritus matter and in turn form an important food sources for a variety of higher trophic levels including shrimps and juvenile fish that use these areas as nursery habitat. The meiofaunal feeding and burrowing activities enhance the microbial activities, thus accelerating the recycling process of nutrients, biomineralization of organic matter and productivity (http://web.vims.edu/).

Meiofaunal organisms are generally abundant in estuarine environments (Coull, 1999). The combined impacts of natural and anthropogenic stressors on the estuarine system are responsible for a continuous change in the biochemical, physical and biological characteristics of the sediment including the distribution of aquatic animals along the estuarine gradient. Therefore the benthic meiofauna, which lives in close association with the sediment and which represents the numerically dominant faunal group, was studied in relation to relevant environmental factors in order to establish a base line against which past and future changes can be measured. The estuarine meiofaunal communities are showing numerous advantages over macrofaunal communities due to their diminutive size, abundance, short generation time and lack of planktonic life stages. The changes in their density, diversity, structure and functioning indicate alterations in the system and so these are widely used as indicators of ecological quality assessment (Alves et al., 2013).

Macrobenthic organisms are important functional constituents of aquatic ecosystems. They act as a connecting link between the biotopes of substratum and water column, also involve in the breakdown of particulate organic matter, transfer energy to higher trophic level and support various communities (Anbuchezhian et al., 2009). Macrobenthos show different body shapes, feeding styles and reproductive modes. It consumes all types of organic matter including bacteria, planktonic and benthic organisms, detritus etc. and which in turn forms the food for many demersal fishes, birds and other invertebrates. The survival, distribution and abundance of macrobenthos depend on the characteristics of their environment such as salinity, organic matter, soil texture, sediment particles and the ability to build permanent burrows in the substratum (Perkins, 1974). Among physical factors, sediment type is of paramount importance in dictating the type of community that could be established in a particular locality. The relative proportion of epifauna and infauna vary according to the nature of the bottom surface, whether it is composed of rock, mud, sand or some combinations of these three types. At the same time, the activities of organisms themselves alter the substrate. Deposit feeders are usually found in fine sediments, which are deposited in relatively quiet water. Biological determinants of benthic community structure include competition, predation and the type of development and recruitment leading to the adult population. The physical, chemical and geological process by both natural and manmade effects, disturb all the communities. Members of the macrobenthic community serve as suitable bio-indicators of environmental variation and key elements of many aquatic monitoring programs, due to their sedentary behaviour and reduced responses to environmental changes (Tweedley et al., 2012). The sensitivity of macroinvertebrates to changes in environmental quality renders them an integral part of any biomonitoring program. Macroinvertebrate communities therefore reside in an aquatic system long enough to reflect the chronic effects of pollutants and yet short enough to respond to relatively acute changes in water quality. They tend to be relatively immobile and as a result are continuously exposed to the constituents of the surface water they inhabit. Thus, because of the limited mobility and their relative inability to move away from adverse conditions, the location of chronic sources of pollution often can be pinpointed by comparing communities of these organisms.

Among benthic organisms, macrofauna forms the most widely studied entity which reside beneath the sediment surface in burrows and tubes. Burrowing and feeding activities of the infauna continually disturb and alter soft sediments (bioturbation) and sessile organisms that attach to hard surfaces alter the topography of their environment. The rates of bioturbation provide an overall physical activity of sediment assemblages. Biogenic sediment mixing has an impact on chemical reactions rate in sediments, especially on the recycling and burial of organic carbon and particle bound pollutants (Smith, 1992). In addition to providing food sources, another important functional value of benthic habitat is the shelter and refuge from predators, provided by its structure. Each macroinvertebrate's response to environmental perturbations produces measurable, and often predictable shifts in abundance and composition at the community level. The physico – chemical properties of sediment play an important regulatory role in determining the distribution and abundance of benthos in the brackish water environment (Gowda et al., 2008). In estuarine and brackish water habitat, macrobenthic organisms are mainly composed of polychaetes, molluscs and crustaceans. Benthos play a vital role in the storage, transformation, release and recycling of nutrients to the overlying water column and promote primary productivity. Physical disturbances associated with construction, recreational activities, dredging and tourism development also have negatively impacted the assemblages of benthic fauna in estuaries. The reclamation and destruction of wetlands such as mangroves and salt marshes have caused major impacts on fauna,that rely on them for their habitat, shelter and food source. According to Thrush et al. (2004) anthropogenic embankment and dredging destroy benthic habitat and thus affect sediment biogeochemistry, biodiversity, ecosystem functioning and overall ecological value of the impacted sediments.

5.1.1 Significance of the study

Temporal changes in ecosystems are normally unpredictable mainly due to the undocumented natural events occurring at various periodicities (Gray and Christie, 1983). Benthic organisms form the most sensitive faunal communities in estuaries and are important in observing the status of an aquatic body (Dauer 1993). Most of the benthic organisms are sedentary, display long life cycles and show different tolerance to stress (Day et at., 1989). These organisms play an important role in altering the physical and chemical environment at the sediment – water interface, supporting the decomposition of organic matter, recycling of nutrients and transfer of energy through the food chain and food web. In recent years the coastal environment has been the focus of numerous human activities such as waste disposal and energy exploitation which often resulted in disturbances to the environment. The benthos, because of its sedentary nature has been affected directly by many of these perturbations and has also been the focus of numerous studies to assess their impacts. Thus it is imperative to have knowledge of the natural spatial and temporal variability of benthic communities in order to deduce the changes that may occur due to a man-induced modification of the environment and the subsequent recovery of the communities. All these affect the standing crop of benthic fauna, so establishing a monitoring program to follow up the changes in the aquatic ecosystem will be of great help in the management of such important water bodies. The coastal wetland habitats of the country are facing serious anthropogenic impacts that have remarkably influenced the biodiversity and ecosystem character of the organisms inhabiting the benthic zone. Vembanad estuary is no exception, facing severe ecological problems due to massive reclamation associated with rapid urbanization and industrialization. Most of the benthic studies were concentrated on the downstream section of the estuary, mainly the Cochin estuarine region. Since the commissioning of TMB, a detailed study on its impact on the benthic faunal distribution and diversity along the whole Vembanad estuary is still lacking. So understanding the spatial distribution patterns of soft bottom benthic communities along the environmental parameters might help to know the relation between benthic community distribution and ecological interactions and ultimately the functioning of the aquatic ecosystem.

5.1.2 Review of Literature

Benthos are the organisms which are found on or in the bottom sediments of aquatic system. The term benthos, was coined by Haeckel in 1891, meaning "depth of the sea"; derived from the Greek word "Ben" meaning the collection of organisms living in or on the sea or lakes and "Thos" the bottom of sea or lakes (Kabir et al., 2012). Benthic animals are classified into two ecological categories related to the substrate where they live. Epibenthic organisms live attached to a hard substratum or rooted to a shallow depth below the surface, but most of the epibenthic organisms project into the water column. About 80% of the larger macrobenthic animals belong to this category. Infaunal organisms live below the sediment – water interface. It includes burrowers, such as clams and polychaetes or borers such as wood boring shipworms and isopods. Infaunal organisms usually dominate in soft substrates and are the most diverse and abundant in subtidal regions. Semi-infaunal organisms live partially below the sediment water interface but protrude from it. The interstitial organisms live and move in the pore waters among sedimentary grains. Nematodes, foraminifera and harpacticoid copepods are the major interstitial organisms. Benthic organisms are collected and separated on sieves, so a classification based on size iss also followed. Based on the size, benthos can be divided into three groups macrobenthos, meiobenthos and microbenthos. Macrobenthos are those organisms which retain in a sieve of mesh size 0.5mm. These are the largest benthic animals including star fish, mussels, most clams etc. Meiobenthos are smaller than 0.5mm but larger the microbenthos, which are less than 0.1mm in size (Levinton, 2001).

The term meiofauna was introduced and defined by Mare (1942). It was derived from the Greek word 'meio' meaning 'the smaller'. Mare (1942) suggested 2 mm for the topmost size limit for meiofauna organisms, today 1 mm is commonly accepted as the upper size limit for meiobenthic investigations. The lower size limit for meiofauna remains different for various studies. The first quantitative sampling of meiofauna in the shelf break area was done by Wigley and McIntyre (1964), using a 74 µm sieve. Thiel (1966) decreased the size limit of sieve to 65µm; but during 1970s Dinet fixed 50 µm as lower limit; whereas Thiel (1971) used 42 µm for including smaller and abundant specimens (Soltwedel, 2000). Among the benthic invertebrates, meiofauna (= meiobenthos) have small size and can be noticed with the help of magnification; which will pass through the sieve of 500 µm and retain on meshes of 40–60 µm sieve. Meiofauna are not a homogeneous ecological group; it is distributed from a range of habitats from Alpine lakes to deep sea trenches (both freshwater and marine habitats), however most of the ecological studies on meiofauna were concentrated on the marine environment. In the marine environment meiofauna represents the most diversified group; among 35 of the animal phyla, 24 have meiofaunal representatives, in which protists and foraminifera form an important group (Giere 2009). According to Hentschel and Jumars (1994) permanent meiofaunal organisms are able to complete its life cycle stages without reaching the size class greater than 1mm, whereas temporary meiofauna are only found in this size class only during a part of their life cycle. The juveniles of macrofaunal groups such as annelids, and bivalves also form a major component of temporary meiofauna, their newly settled larvae eventually grow to become macrofauna (Giere, 2009).

In muddy substratum, most of the fauna are found in the top 2 cms of the sediment while in more oxygenated sandy sediments meiofauna can be observed in deeper sediments (Vincx, 1996). According to Smith and Coull (1987), the muddy sediments have more meiofauna in the upper first cm than that of the first 10 cms of sandy sediments. It was documented that, the meiofauna found worldwide in every natural and uncontaminated estuarine sediment, on an average was 10^6 m^{-2} and its dry weight biomass was 0.75–2 gm^{-2} (Coull and Bell, 1979). In each habitat several factors such as season, latitude, water depth, tidal exposure, grain size, habitat type, and other abiotic and biotic factors play a crucial role in determining the variation of abundance and biomass of meiofauna. They are distributed in all types of sediments from clay to gravel. The meiofaunal abundance was normally highest in intertidal muddy estuarine habitats and lowest in deep seas (Balsamo et al., 2010). Similarly the abundance and biomass values were highest in mud having higher organic matter and lowest in clean sands. Among the total meiofuana, nematodes generally form the most abundant taxa contributing 60–90%. Worldwide around 4000-5000 species of free-living marine nematodes were identified and described (Eyualem-Abebe et al., 2008).

Meiofaunal characters were considered as good indicators of environmental status and any variation in their composition, diversity and assemblage structure denote modification in the system. There were a variety of physical factors known to control meiofaunal abundance and species composition. Among the physical factors, sediment particle size was identified as a primary important factor controlling meiofauna in estuaries (Coull, 1999). Meiofaunal assemblages were also found to be varied in different regions of estuaries; the assemblage in muddy substratum was varied from sandy substratum; similarly the distributional pattern of meiofauna in the low saline region varied from that of high saline region (Rutledge and Fleeger, 1993). Several researchers reported the role of physical factors in structuring the meiofauna. In meso-scale, abiotic factors such as sediment texture, salinity and hydrodynamics, were the most important factors controlling the meiofauna (Heip et al., 1985; Alongi, 1987; Sarma and Wilsanand, 1996; Li and Vincx, 1993; Smol et al., 1994; Soetaert et al., 1994., Rosa and Bemvenutti, 2005., Kapusta, 2005). In a micro-scale, biological parameters such as inter and intra-specific relationships, the occurrence of biogenic structures, accessibility of food sources and reproductive strategies etc. formed the key factors (Santos et al., 1996; Moens et al., 1999; Pinto and Bemvenuti, 2003; Moreno et al., 2011; Vanaverbeke et al., 2011; Harguinteguy et al., 2012; Semprucci and Balsamo, 2012; Ngo et al., 2013). El-Serehy et al. (2015) studied about the population fluctuation and vertical distribution of meiofauna in the Red Sea interstitial environment. The study revealed that nematodes, harpacticoid copepods, polycheates and ostracods formed the dominant meiofaunal groups and its density was controlled by both sediment parameters and the occurrence of biogenic structures. Zeppilli et al. (2015) made a review about the role of meiofauna in global change impact studies. The review noted that foraminifers, nematodes, copepods and ostracods formed the important meiofaunal groups for forecasting the global changes, in which foraminifers and copepods were the more sensitive to environmental disturbances, whereas nematodes were tolerant to various types of stress.

There were several studies undertaken in and around Indian waters. Krishnaswamy (1957) reported the occurrence of marine gastrotrichs in the Indian coast for the first time. Ganapati and Rao (1962), in a pioneering effort reported the occurrence of gastrotrichs from the interstitial meiofauna of Waltair coast. Desai and Krishnan Kutty (1967) observed high abundance of foraminifera and nematodes in meiofaunal groups in Cochin backwater. Kurian (1972) studied the ecology of benthos of the Cochin backwaters and observed that the meiofauna were more abundant in the finer sediments and their abundance was not affected by tidal changes. Damodaran (1973) investigated the benthos of mud banks of Kerala coast and correlated them with prawn fishery. This was the first quantitative study on meiobenthos along the Indian coast. Ansari (1978) studied the meiofauna of Karwar estuarine area and observed that nematodes and foraminifera formed the dominant groups among the meiofauna. Varshney et al. (1981) observed the abundance range of meiofauna in the sandy and muddy substratum of Narmada estuary as nil to 3164 no./10cm^2. Abdul Azis and Balakrishnan Nair (1983) investigated the occurrence and relative abundance of meiofauna of Edava – Nadayara and Paravur backwaters and they also observed the effect of coconut husk retting on the fauna. Ansari and Parulekar (1993) studied the distribution and abundance of subtidal meiofauna in Mandovi estuary of Goa. Nematodes formed the dominant group, in which deposit feeders and epigrowth feeders were abundant in the fine mud and sandy substratum. Ansari and Parulekar (1998) studied the community structure of meiobenthos from a tropical estuary. Ingole and Parulekar (1998) studied the role of salinity on meiofauna in the estuarine intertidal beach at Siridao, Goa (India). The study noted that the variation in community structure of meiofauna was associated with salinity fluctuations. The distribution and abundance of meiobenthos, hydrography and sediment characteristics of three different types of traditional prawn culture systems around Cochin were studied by Preetha and Pillai (2000). The effect of sewage pollution on meiofauna was studied in the Mandovi estuary where the meiofaunal population density showed a gradual increase from the discharge point (Ansari et al., 2003). A complete picture of the gastrotrichs recorded from India and the neighboring islands was reviewed and illustrated by Naidu and Rao (2004). In the mangrove areas of Parangipettai, five new nematode species were reported by Chinnadurai and Fernando (2006a).

Diversity of meiobenthic nematodes in the Poonthura estuary (southwest coast of India) was studied by Kumary (2008). Meiobenthos of Cochin backwaters in relation to macrobenthos and environmental parameters were studied by John (2009). Sajan and Damodaran (2007) investigated the faunal composition of meiobenthos from the shelf regions of the west coast of India. Sajan et al. (2010) studied the meiofaunal standing stock and community structure of nematodes, the predominant group, in western continental shelf of India. The study found that meiofauna had affinity towards finer sediments. Furthermore the distribution of meiofauna was influenced by depth, latitude, organic matter content and clay factors. Thilagavathi et al. (2011) distinguished the benthic meiofaunal composition and community structure in Sethukuda mangrove area and adjacent open sea in the East Coast of India. The study found that the open sea area was largely enriched with nutrients and benthic organisms when compared to Sethukuda mangrove creek. Spatial distribution of nematodes in the subtidal community of the Central West Coast of India with special reference to *Tershellingia longicaudata* (Nematoda: Linhomoeidae) was observed by Nanajkar et al. (2011). Priyalakshmi and Menon (2014) surveyed the interstitial fauna in the sandy beaches of Kerala. The study revealed the presence of large numbers of gastrotrich (unusual phyla of invertebrates) in the area, and eight species reported among them were the first records from Kerala coast. The study noted that gastrotrich showed a distinct discontinuity in its occurrence and abundance. The faunal composition of meiobenthos in the sediments of the southeast continental shelf of India was studied by Ansari et al. (2012). During the study, nematoda, copepoda and foraminifera were the dominant meiofaunal groups, comprising 192 species of freeliving marine nematodes, 39 species of harpacticoid copepods and 39 species of foraminifera. Mohan et al. (2012) made a study to find out the influence of sediment characteristics, organic carbon and carbonate on the meiofauna of Junglighat Bay and Car Nicobar, Andaman and Nicobar Islands. The study found that the abundance of nematodes was influenced less by organic carbon and more by carbonate content. Dhivya and Mohan (2013) made a review on meiofaunal studies in India. Ansari et al. (2014) investigated the quantitative distribution of meiofauna in the north-east Andaman Sea. The study found that the meiofauna was dominated by nematodes (80%), benthic copepods (5.9 %), and foraminifera (2.8 %). Distribution of meiobenthos off Kakinada Bay, Gaderu and Coringa estuarine complex were studied by

Annapurna et al. (2015). Among the meiofauna, nematodes (37%) were the dominant group followed by copepods (15.0%), foraminifera (13.1%) and polychaetes (9.9%). They also noted that salinity and sediment texture played an important role in determining meiofaunal abundance. Mohan and Dhivya (2015) studied about meiofaunal succession in the newly formed marine environment after the tsunami at Port Blair, Andaman. Ansari et al. (2016) studied the diversity pattern of marine nematodes in the continental shelf off Bay of Bengal. The study also found that among the abiotic parameters, sediment texture, organic matter, pressure and depth were the important factors controlling the community assemblages of nematodes. Singh and Ingole (2016) studied about the structure and function of nematode community from the Indian western continental margin to determine the influence of habitat heterogeneity and oxygen minimum zone. Based on the published literature Ghosh and Mandal (2016) prepared the taxonomic list of free-living nematode community from Indian coasts and Andaman and Nicobar Islands. Ansari et al. (2017) made a quantitative study of metazoan meiofauna across the oxygen minimum zone (OMZ) of continental margin in the depth range 500-1965 m in the north east Arabian Sea.

During 1750 Marsigli and Donati studied benthos using dredge (Murray and Hjort, 1965). During British - Antarctic expedition, HMS Erebun and HMS Terror (1839-43) dredged benthic fauna down from 730m. During H.M. S. Challenger (1872-1876) expedition, benthic samples were collected from Pacific, Atlantic and Antarctic Ocean. The systematic attention on the identity and composition of bottom communities was initiated by Petersen (1914, 1915). It was followed by research in various place of the world, mainly for estimating the fish production potential (Parker, 1975). Since the pioneering works of Peterson (1918) and Thorson (1957), a majority of marine benthic ecology investigations had been concentrated on short-term sampling surveys of specific region to demarcate certain macrobenthic species assemblages (Whitlatch, 1977). Thorson (1957) explained the concepts of marine benthic communities. Remane (1940) distinguished the thought of marine benthic communities which were classified according to the dominant species. The spatial heterogeneity of macrobenthos along the estuarine gradient, in relation to salinity and sediment composition was noted by several authors (Carriker, 1967; Wolff, 1973; Gray, 1974; Boesch, 1977; Michaelis, 1983; Holland et al., 1987).

The benthic macrofauna along the estuarine gradient of the Schelde estuary (France) was studied by Ysebaert et al. (1993); which showed that species richness, diversity and total biomass of the benthic macrofauna decreased along the salinity gradient wheras no variation was observed in total density. The oligohaline and freshwater tidal part of the Schelde estuary was characterized by a very impoverished benthic community, composed only of Oligochaeta. Meire et al. (1994) observed the effects of the storm-surge barrier on macrozoobenthos and analysed the spatial and temporal distribution of macrozoobenthos in Oosterschelde and its relationship with some environmental parameters. The effect of proximity to an offshore hard-bottom reef on infaunal abundances in the South East of Wrightsville Beach, North Carolina was investigated by Posey and Ambrose (1994). This Study suggested that there were potentially important indirect effects of predator - prey interactions among the rock ledge-associated predators and soft-bottom prey. Zonation of intertidal macrobenthos in the estuaries of Schelde (Netherlands and Belgium) and Ems (northeast of the Netherlands) was investigated by Ysebaert et al. (1998). The study emphasized that for Schelde estuary and to a lesser extent for Ems estuary, anthropogenic stress had a negative effect on the intertidal macrobenthic communities of the oligohaline/freshwater tidal zone. Oligochaeta were the only dominating group, whereas the very euryhaline and/or true limnetic species were absent. The mesohaline zone of the Schelde estuary was dominated by a large number of short living, opportunistic species, whereas in the Ems estuary, relatively more stable macrobenthic communities were present. A comparison with some other European estuaries also showed a similar trend as those observed for the Schelde and Ems estuaries. There were several studies of macrofaunal communities in estuaries and their associated environments in the world (Moverley et al., 1986; Pridmore et al., 1990; Chardy and Clavier, 1998; Mancinelli et al., 1998; Talley et al., 2000; Dittmann, 2002; Morrisey et al., 2003; Van Hoey et al., 2004; Puente et al., 2008; Mooraki et al., 2009; Tweedley et al., 2012; Abdelsalam and Tanida, 2013; Zalmon et al., 2013; Abdel Gawad et al., 2014; Mola andAbdel Gawad, 2014). Ritter and Montagna (1999) gave a detailed account on estaurine macrobenthic community succession. He mentioned about the influence of hypoxia, salinity fluctuations, sediment resuspension and disturbance frequency on macrofaunal community. Reinicke (2000) studied the small scale patchiness of benthos and sediment parameters in a fresh water tidal mudflat of the river Elbe estuary. He observed that fine grained sediments sheltered a specialized and opportunistic benthic community, largely dominated by tubificid Oligochaeta and the benthic Cladocera among macrofauna, together with Chironomidae. They also noticed that benthic colonisation was mainly in the uppermost 1cm sediment layer. Carvalho et al. (2001) studied the macrobenthic community characterization of Sado estuary from the western coast of Portugal. The study was a part of an environmental assessment undertaken prior to dredging operations in the estuary. Spatial and temporal variation in benthic macrofauna and its relationships with environmental variables in an estuarine, intertidal soft-sediment environment in the Schelde estuary, Netherlands was investigated by Ysebaert and Herman (2002). They observed that salinity, mud content, chlorophyll *a* concentration and bed-level height were responsible for variation of macrobenthic species. Edgar and Barrett (2002) gave information on the distribution of species richness, faunal density, biomass and estimated productivity of benthic invertebrates in Tasmanian estuaries. They quantified a variety of spatial and temporal scales to assess general hypothesis relating community metrics to environmental variables.

Ysebaert et al. (2003) related the macrobenthic species distributions and community structure to salinity, depth, current velocities and sediment characteristics in the Schelde estuary, NW Europe. The study showed that at a large estuarine scale, a considerable fraction of the variation in abundance and biomass of benthic macrofauna correlated very well with environmental factors. Multivariate analyses showed that, a strong relationship existed between the macrobenthic assemblages and the predominant environmental gradients in the Schelde estuary. The most important environmental factors were depth and salinity, which explained a significant part of the macrobenthic community structure. The spatial distribution of macrobenthic communities (biomass and abundance) in the Rio de la Plata estuary, Argentinae - Uruguay and the adjacent shelf waters were investigated by Giberto et al. (2004). They observed that bottom type, salinity and the presence of a turbidity front were considered as the main physical variables in structuring benthic communities of the Rio de la Plata estuary. The potential impact of sedimentation on benthic infauna was quantitatively investigated in the Okura estuary, New Zealand by Anderson et al. (2004). They observed that sedimentation become an important structuring force in the estuarine macrobenthic assemblages. They noted that bivalves had a negative relationship with sedimentation, wheras certain burrowing crabs and polychaetes were more abundant in high-deposition environments.

Long-term temporal and spatial differences in benthic community structure in an east Australian sub-tropical estuary (Port Curtis) were quantitatively examined by Currie and Small (2005). Their findings suggested that high levels of turbidity promote recruitment and growth of benthic organisms in Port Curtis. Strong correlations existed between regional rainfall, freshwater inflow, nutrient and chlorophyll a concentrations and benthic community in the estuary. Benthic variability in intertidal soft-sediments in the mesohaline part of the Schelde estuary was studied by Ysebaert et al. (2005). The study showed that in estuarine systems with high seasonal variability in river flow rate and salinity, the upper-middle estuarine fauna may switch each year between an oligohaline and a mesohaline fauna, which result in communities that progress beyond early benthic-community succession. The majority of the benthic infauna was found in the upper 3 cm in both seasons, with surface deposit feeders mainly occurring in the top 1 cm of the sediment, showing a strong correlation with both elevation and pigment contents and subsurface deposit feeders occurring deeper in the sediment, showing a positive correlation with the mud content of the sediment. Hastie and Smith (2006) evaluated the benthic faunal communities in intermittent estuaries during the drought period. They also compare the effect of drought on macrofaunal communities between closed, intermittent estuaries and permanently open estuaries within the region. They found that physico-chemical variables were highly variable among estuaries and were not strongly correlated with assemblage patterns of macrofauna. Lucero et al. (2006) observed the variability of macrobenthic assemblages under abnormal climatic conditions in Dagua River estuary of Colombian coast (Tropical Eastern Pacific). The typical macrobenthic assemblage structure was varied with the unusual hydrographical and sediment conditions generated by the cold phase of El Niño-Southern Oscillation (ENSO) in water temperature, higher deposition of organic matter and low salinity. Spatial subtidal macrobenthic distribution in relation to abiotic conditions in the Lima estuary, NW of Portugal was studied by Sousa et al. (2006). The multivariate analysis revealed a macrobenthic community with five distinct groups, particularly related to the sedimentological characteristics and salinity. So they observed significant differences in macrobenthic assemblage's composition along an estuarine gradient.

Biodiversity and distribution of soft bottom macrobenthos and its zonation pattern in an area of Dese estuary (lagoon of Venice, Italy) was carried out by Maggiore and Keppel (2007). The study showed that molluscs and polychaetes showed a zonation of marine species which colonize brackish environments in the outer and intermediate areas of the estuary and of brackish water species in the inner one. In the outer area different assemblages were attributed to different sediment types also. Spatial patterns of benthic macrofauna in relation to environmental variables in an intertidal habitat in the Humber estuary, UK was studied by Fujii (2007). The structure of the benthic macrofaunal assemblages of the estuarine portion of Paraguaçu River, NE, Brazil, and its relationship with surface sediment characteristics and physical variables were investigated by Barros et al. (2008). They found that benthic assemblages in the upper estuary were different from the lower estuary and a clear replacement of benthic taxa along the estuary was observed. They also prepared new models of faunal distribution and recommended detailed studies on the effects of salinity variation using functional approaches. The spatial and temporal variability of benthic oxygen demand and nutrient regeneration in an anthropogenically impacted New England estuary was studied by Fulweiler et al. (2010). Specchiulli et al. (2010) analysed sediment characteristics and macrobenthic assemblages in two very close Italian coastal lagoons (Lesina and Varano). They assessed the difference between the two basins and the relationship between sediment matrix and benthic organisms within and between the two lagoons. Benthic faunal distribution and abundance in the Mfolozi-Msunduzi estuarine system, South Africa was examined by Ngqulana et al. (2010). The study showed that polychaetes, crabs and tanaids formed the dominant macrofaunal groups in the system. Oxygen concentration, temperature, the open or closed state of the estuarine mouth, salinity, particle size and organic content of the substratum being important factors in determining the benthic community structure there.

The distibution of macrobenthic community in the Quequen estuary was analyzed by Godoy et al. (2011). The study found that the distribution of macrofauna was related to environmental variables of water and sediment and also benthic assemblages were highly sensitive to pollution. Composition and spatial distribution of the benthic macrofauna in the Cachoeira River estuary, Brazil was observed by Ourives et al. (2011). They observed that salinity, sediment characteristics, and dietary habits were important in the composition and distribution of organisms in this estuary. Wildsmith et al. (2011) used benthic macro invertebrates as indicators to study the severity of environmental degradation in microtidal estuaries such as Swan-Canning and Peel-Harvey estuaries of Australia. Composition and spatial distribution of benthic macrofauna in the Cachoeira River estuary, Bahia, Brazil was observed by Ourives et al. (2011). They observed that salinity, sediment characteristics, and dietary habits greatly influenced the spatial distribution and composition of macrobenthos in the estuary. Effects of extreme climate events on the macrobenthic communities structure and functioning of Mondego estuary (Portugal) was investigated by Grilo et al. (2011). The study revealed that floods and heat waves had a stronger negative effect on macrobenthic assemblages than the droughts, imposing a total abundance decline. Bivalves and oligochaetes decreased with the flooding events along with subsurface-deposit feeders and suspension feeders, while crustaceans were particularly sensitive to heat waves. The environmental effects on the macrobenthic community of Han River Estuary, Korea were studied by Yu et al. (2012) during summer and spring. Their study revealed that salinity was the most important factor affecting macrobenthic communities in the Han River Estuary, followed by other factors such as sediment grain size, bottom dissolved oxygen, and total organic carbon. Picanco et al. (2014) studied the spatial and seasonal variations of the macrobenthic community structure of the Minho estuary, saltmarsh (Portugal). The study involved examination of relationships between the macrobenthic community and the abiotic characteristics of the surrounding sediments: texture, organic matter content and metal concentrations. The study revealed that eventhough macrobenthic community showed no standard spatial distribution throughout the saltmarsh, species abundances showed high seasonal variations that were directly dependent on the natural environmental conditions of each season.

In the past numerous studies have been conducted on the ecology of macrobenthic populations in the tropical estuarine systems of India. Annandale (1907) and Annandale and Kemp (1915) initiated the study on the ecology of Gangetic delta and the fauna of Chilka Lake in India. The bottom fauna of the brackish water region of Madras city was studied by Panikkar and Aiyar (1937). Samuel (1944) worked on the benthic animal communities of the Madras coast. The bottom fauna along the Travancore coast were surveyed by Kurian (1953). Similarly the bottom fauna of Malabar Coast was investigated by Seshappa (1953). Near the shore region of Cochin Desai and Krishnan Kutty (1967) carried out a survey on meiofauna and macrofauna. Kurian (1967) made a survey on the bottom fauna of the Southwest coast of India. Kurian (1972) made study on the ecology of benthos in Cochin estuary. Similarly Harkantra (1975) studied the benthos of the Kali estuary, Karwar. Macrobenthic production in the Vembanad lake was investigated by Ansari (1974). He observed that macrobenthic fauna was mainly composed of polychaetes, bivalves, decapodes and amphipods. He also noted a progressive decrease in the animal population from lower part of backwater towards the upper reaches which had low salinity. Occurrence, abundance and seasonal distribution of the benthic fauna of the mud bank area at Alappuzha were investigated by Reghunathan et al. (1974). They observed that in the mud bank area, benthos had a low intensity mainly due to the unconsolidated nature of the sediments which does not give a stable substratum for the animals to settle.

Kurian et al. (1975) studied the distribution of benthic macrofauna of Vembanad Lake. In their investigation bottom fauna was mainly composed of bivalve molluscs, polychaetes and amphipods. They observed that the bivalve, *Villorita* sp. found in the sandy substratum and the dominance of polychaete and crustacean were noted in the fine sediments. Raman et al. (1975) investigated the hydrobiology and benthic ecology of the Pulicat lake. They noted that bottom fauna consisted mostly of polychaetes, tanaids, amphipods, and mollusks; their abundance was more during monsoon and post- monsoon seasons. A survey of benthic macrofauna in the Cochin backwater was carried out by Ansari (1977). Pillai (1977) studied the pattern of distribution and seasonal abundance of important benthic groups in Cochin backwaters and observed that polychaetes formed the major fauna throughout the year. The maximum benthic population occurred during December to April and minimum during south-west monsoon period (July to September). Ecology of soft bottom benthic fauna in Kadinamkulam Backwater, southwest coast of India was studied by Balakrishnan Nair et al. (1984a). The study revealed that monsoon season was the most productive and post monsoon least. Similarly they observed that coconut husk retting activity along the shores seems to adversely affect the benthic fauna. Batcha (1984) studied the distribution and composition, diversity and biomass of macrobenthos in Cochin estuary. The occurrence and relative abundance of benthic fauna in the Ashtamudi estuary have been studied by Balakrishnan Nair et al. (1984b). In the Visakhapatnam harbour area, there were several studies conducted to know the impact of pollution (Ganapati, 1969; Ganapati and Raman, 1979; Raman and Ganapati, 1983). Raman (1995) made an overview of 23 years of investigations and monitoring in the major changes occurred in the water quality and benthic organisms in the pollution effected Visakhapatnam harbour.

In Vellar estuary, the relationship between benthic fauna and physico-chemical and sediment prameters was studied by Chandran (1987). The study indicated that polychaetes, bivalves, gastropods and crustaceans were the dominant benthic fauna. A significant negative relationship was obtained between sand content and benthic fauna while clay content showed a positive relationship. Sarala Devi et al. (1991a) gave an account of the benthic communities and its co-existence in the northern limb of Cochin backwaters. The study noted high densities of *Capitella capitata* near the effluent discharge site which was considered as an indicator of pollution. Environmental impact assessment of benthic community stability in an estuarine complex was investigated by Ansari and Abidi (1993). The study revealed that composition of the community was influenced by increasing environmental disturbances caused by human activities such as mining and sand excavation. Studies on the benthic fauna of the mangrove swamps of Cochin area was studied by Sunil Kumar (1993). Trophic model of an estuarine ecosystem at the southeast coast of India was studied by Santhanam et al. (1995). The investigation detailed the trophic connections existing among the planktonic, pelagic and benthic components of Pullavali brackishwater. Benthic fauna of southwest and southeast coasts of India was studied by Sarala Devi et al. (1999). In the study it was observed that number of benthic groups varied with stations and also with depths. Sheeba (2000) studied about the distribution of benthic infauna in the Cochin backwaters in relation to environmental parameters.

Macrobenthic community of a shallow subtidal (5-20 m) muddy deposit off Dabhol, west coast of India was investigated by Ingole et al. (2002). They observed that macrofaunal community were numerically dominated by polychaetes and also noted a high benthic production from the area of high organic content evident from a good fish catch in the experimental bottom trawling. Increased macrobenthic density and diversity, an indication of recovery of Chilka lake from environmental degradation was evaluated by Ingole (2002). The fauna was comprised mainly of molluscs, crustaceans, polychaetes etc. In terms of number the polychaetes was dominated followed by molluscs and crustaceans. The result showed that there was two fold increase in macrofauna abundance during dredging and eight fold increase in post dredging phase, mainly due to exposure of deep burrowing organisms to the sediment surface and due to the improved food supply.

Raj et al. (2002) did the experiments in restoration of benthic biodiversity in Pulicat Lake. Within one year of deploying artificial habitats, they observed that 83 species of biodiversity colonised in the experimental artificial habitats, where the edible oysters served as the 'keystone' species. Response of benthic fauna to different pollutants and some case studies and the relevance of benthos to environmental impact assessment were studied by Ansari et al. (2003). Benthos were exposed to different stress conditions and suggested the distinctness of benthic data as a tool in environmental impact assessment studies. Economic and social management of estuarine biodiversity in the west coast of India was studied by Thomson (2003). In this study he tried to characterise the nature of fish and shellfish diversity and to describe the ecological services of major estuaries along the West Coast of India. Benthos in the nearshore waters off Visakhapatnam was studied by Vijayakumaran (2003). The faunal composition, population density, biomass, volume of benthos in relation to sediment, depth range and lat - long squares off Visakhapatnam were studied. The study showed that Polychaeta (63 %) was the dominant group of fauna followed by Amphipoda (17 %). Environmental influences on the species diversity, biomass and population density of soft bottom macrofauna in the estuarine system of Goa, West coast of India were studied by Harkantra and Rodrigues (2004). They recorded a total of 58 species belonging to polychaete, molluscs and crustaceans and other minor groups in order of species abundance; eighteen species were new to the local fauna. They observed that higher species diversity was observed at high salinity, fine sand and high sedimentary biochemical parameters of total organic carbon, total organic nitrogen and carbon of biopolymeric fraction (C-BPF) sites. They found that the multiple linear regression model of the system revealed that the environmental parameters were significantly influencing the species diversity, biomass and population density of macrofauna.

Numerical analyses of soft bottom macro invertebrates to diagnose the pollution in tropical coastal waters of off Mangalore were studied by Harkantra and Rodrigues (2003). Moorthy et al. (2005) studied the quantitative and qualitative investigations of the ostracod fauna of the benthos of Madhurantakam Lake in Tamilnadu, India. The study found high ostracod abundance in organic rich areas. Raut et al. (2005) studied the macrobenthos of Kakinada bay in the Godavari delta, East coast of India. They made a comparison in the decadal changes. They found that benthic densities were highest at sites close to mangrove outlets. Their observations revealed marked changes in benthic community structure compared to an earlier investigation held in 1958 – 1963 in this area. Ecology and biodiversity of macrofauna in a mangrove fringed lagoon situated on the south-west coast of India were examined by Marakala et al. (2005). Among the macrofauna polychaetes were the dominat group followed by molluscs and finfish. Seasonal study showed that macrofaunal density was maximum during premonsoon followed by postmonsoon and monsoon.

Coastal and marine biodiversity of India was reviewed by Venkataraman and Wafar (2005). The paper reviewed the literature on coastal and marine biodiversity from India and examined the constrains to its sustainability. Influence of environmental properties on macrobenthos in the northwest Indian shelf was investigated by Jayaraj et al. (2007). They observed the standing stock of macrobenthic infauna and associated environmental factors influencing the benthic community. The polychaetes were the dominant group and were more abundant in shallow and middle depths with moderate organic matter, clay and relatively high dissolved oxygen. The crustaceans and molluscs were more abundant in deeper areas having sandy sediment and low temperature. Acoustic characterization of seafloor habitats on the western continental shelf of India was investigated by Chakraborty et al. (2007). They included the interaction effects of the dual frequency (210 and 33 kHz) backscatter signal with seafloor sediment and benthic biota along a transect in water 27–83 m deep offshore of the Goaregion of India’s central west coast. Pawar and Kulkarni (2007) investigated the diversity indices of selected macrobenthos in Karanja creek, west coast of India. Polychaetes, gastropods, crustaceans, pelecypods and echinoderms were the major faunal groups encountered at Karanja creek. The higher values of evenness index denoted suitable for the growth and abundance of biota therein. The infaunal benthic community of soft bottom sediment of the tropical eastern Arabian Sea shelf was investigated by Jayaraj et al. (2008). The faunal composition, abundance, and diversity of species together with environmental parameters were observed. The study showed that mean benthic abundance was high in the shallow and intermediate depths and decreased toward deeper areas. Diversity increased with depths up to 100 m during both seasons and decreased beyond 100 m depth. The dominant benthic group was polychaetes during both seasons. Shallow depths were dominated by deposit-feeding polychaetes, and deeper depths were dominated by deposit and filter feeders such as crustaceans and molluscs.

Abundance and vertical distribution of macrobenthos in a mangrove - fringed brackish water pond in Mangalore, India was studied by Gowda et al. (2008). The study revealed that polychaetes were the most abundant group (50.86 %) followed by molluscs (30.88 %) and crustaceans (18.26 %). Species diversity was highest at the top layer and decreased with sediment depth. The study pointed out that abundance and vertical distribution of macrobenthos were useful indicators of the condition of the pond bottom and of the pond habitat as a whole. Distribution, density and community ecology of macrobenthic intertidal polychaetes in the coastal tract of Midnapore, West Bengal was studied by Chandra and Chakraborty (2008). They assesed the effects of ecological parameters such as temperature, salinity, dissolved oxygen, pH, sediment texture and organic matter on the distribution, density and community structure of polychaetes. Meiobenthos of Cochin backwaters in relation to macrobenthos and environmental parameters were studied by John (2009). Importance of sieve size in deep-sea macrobenthic studies was assessed by Pavithran et al. (2009a). In this study, they tested the effectiveness of sieves of two different mesh sizes - 0.3 and 0.5 mm in assessing macrofaunal diversity, density and biomass. Using the larger mesh resulted in a significant loss in biomass of 90% and 78% for polychaetes and nematodes respectively. The result showed a significant reduction in the number of species, organism density and biomass of macrofauna with use of a 0.5 mm mesh rather than a 0.3 mm mesh and that a sieve of lower mesh size was more suitable for evaluation of deep-sea macrofauna.

Pavithran et al. (2009b) studied the composition of macrobenthos from the Central Indian Ocean basin. It was the first attempt to document the Indian Ocean abyssal benthic diversity of macro invertebrates and to investigate its relation to the surface primary production (chl-*a*), sediment labile organic matter, organic carbon and texture. They observed that the values for population density were strongly correlated with surface water chl-*a* and sediment protein, indicating supply of fresh organic matter as a critical factor for maintaining deep sea benthic diversity and abundance. Macrobenthos and its relation to ecosystem dynamics in Cochin estuary were studied by Geetha et al. (2010). Distribution and seasonal abundance of Gangavali estuary was studied by Shirodkar and Nayak (2010) and found that polycheates were dominant among macrofauna. Its diversity was maximum towards lower reaches of the estuary. The benthic faunal components exhibit bimodal pattern of distribution with monsoon as a period of minimum density and post-monsoon and early pre-monsoon with maximum density. The diversity and distribution pattern of benthic macroinvertebrates in two backwaters viz., Veli and Kadinamkulam of Kerala were assessed using diversity indices by Latha and Thanga (2010). Seasonal abundance of macrobenthic composition and diversity along the south east coast of India was studied by Varadharajan et al. (2010). Its results indicated that polychaetes were dominated among macrofauna, followed by decapods, bivalves, gastropods, amphipods and isopods. The study conformed that pollution in the intertidal coastal area was affecting the species diversity directly and the fishery potential indirectly. Bivalve resource survey of Moorad estuary in the northern Kerala was carried out by Laxmilatha et al. (2011). The biodiversity and abundance of benthos along the southeast coast of India was monitored by Manoharan et al. (2011). They found that habitat played an important role in the life cycle of many benthic organisms. Deposit composition was vital to the marine benthic organisms which provide shelter and food stuff in the form of organic matter. They noted that the macro, meio and micro fauna and flora was recorded maximum in muddy shores and minimum in sand beaches. They also noted that benthic organisms were highly sensitive to environmental stress due to pollution and the observed variation in the physico-chemical and sediment quality was responsible for the disparity in the species abundance. Mahapatro et al. (2011) studied the macrobenthos of shelf zone off Dhamara estuary, Bay of Bengal. A total of 1870 ind./m^2 of macrobenthic organisms were encountered; the population density and species diversity was increased along with increasing depth. The study noted that environmental parameters like salinity, pH and dissolved oxygen exhibited strong correlation with population density of macrobenthic organisms. Martin et al. (2011) studied the eutrophication induced changes in benthic community structure of Cochin estuary. They observed that the dissolved inorganic nitrogen and phosphorus caused high abundance of chlorophyll a and accumulation of organic carbon in sediments. Principal component analysis distinguished 3 zones in the estuary; the central zone was characterized by organic enrichment, low species diversity and increased pollution tolerant species. In their study deterioration of the estuary was indicated by an increase in the nutrients and chlorophyll levels by 6-fold during the last few decades. The estuarine variation caused a reduction in the benthic diversity followed by an invasion of opportunistic polychaetes indicating a stress in the estuary.

Temporal variability of macrofauna from a disturbed habitat in Zuari estuary, west coast of India was investigated by Sivadas et al. (2011). They assessed the changes in macrobenthic community by using abundance, biomass and species diversity indices and noted that the macrofauna showed a drastic decline during the stable pre-monsoon season, a period when highest abundance of fauna was observed in the tropical estuary. So they conclude that, the macrobenthic community in the disturbed habitat of Zuari estuary did not follow the seasonal trend generally observed in a tropical estuary. In the continental shelf of west coast of India distribution, abundance and diversity of macrobenthos and the influence of environmental parameters on the macrofaunal community was studied by Joydas and Damodaran (2009); Joydas et al. (2009). Smitha (2011) investigated the marine benthos of the continental shelf of southwest and southeast coast of India. Abdul Jaleel (2012) studied about the macrobenthos of the continental margin (200-1000m) of south eastern Arabian Sea with special reference to polychaetes. A change in the soft bottom macrobenthic diversity and community structure from the ports of Mumbai, India was studied by Mandal and Harkantra (2013). The study showed that polychaetes were the most dominant macrobenthic group (72.09%) followed by decapoda, amphipoda and bivalves. The spatial distribution and diversity of macrobenthos and their relationships between physico-chemical parameters of the water and sediment in different mangrove habitats of Tamil Nadu, India were studied by Samidurai et al. (2012). The study showed that, there were obvious differences among the macrofaunal community structures in different mangrove habitats. The result implied that different mangrove ecosystem had different effects on the macrofauna communities revealing the macrofaunal adaptation capability to specific habitats. Benthic biodiversity in the Pichavaram mangroves, southeast coast of India was studied by Pravinkumar et al. (2013). During the study as many as 22 species of benthic macrofauna were recorded in the study area and its population density varied between 394 and 23,888 ind./m^2. The spatial distribution and macrofaunal composition of two mangrove ecosystems of Tamil Nadu were studied by Sekar et al. (2013). The study indicated that the differences in environmental parameters were responsible for the assemblages of benthic community between stations as well as temporal seasons. Macrobenthos in anthropogenically influenced zones of a coralline marine protected area in the Gulf of Kachchh, India was studied by Sukumaran et al. (2013). The study provided valuable baseline data of macrobenthos along with prevailing environmental conditions in the eco sensitive area. They used polychaete/amphipod ratio to verify the environmental status and viewed oil-sensitive amphipods on a major part area.

Diversity of benthic fauna in Coleroon estuary, south east coast of India was investigated by Muniasamy et al. (2013) and found that polychaetes formed the most abundant macrofaunal group followed by crustaceans, bivalves and gastropods. Distribution and abundance of macrobenthos in the mangroves ecosystem of Kali estuary, Karnataka was investigated by Vasanth Kumar et al. (2013). From the fourteen macrofaunal groups, polychaetes and bivalves contributed maximum abundance in the area. Effect of tropical rainfall in structuring the macrobenthic community of Mandovi estuary, west coast of India was investigated by Gaonkar et al., 2013. The study showed that the monsoon regimes increased the food input into the system which favored the recruitment of benthic species and also ensured the wide dispersal of the sedentary benthic species. The macrobenthic communities of the Vellar estuary was studied by Chertoprud et al. (2013). The study found that salinity, depth, and bottom type are the leading ecological factors affecting the distribution of communities. The study revealed that polyhalinic area of Vellar estuary was dominated by marine species, which are most abundant in biomass and species richness, whereas the mesohalinic area was inhabited by brackish water communities with low abundance. Ansari et al. (2014) analyzed the macroinfauna and total organic content of sediment for assessing the relationship between organic carbon and benthic fauna. The study observed that the nature of organic matter act as a source of food or as a pressure factor for macrofaunal community. Hemalatha et al. (2014) investigated the diversity of infaunal macrobenthic community in the intertidal zone of Vellar estuary, Southeast coast of India. Pawar (2015) evaluated the density, biomass and diversity indices of selected macrobenthos along the creek mangrove ecosystems of west coast of India. Compared to monsoon, during premonsoon and postmonsoon elevated values of density, biomass and diversity indices of macrobenthos were observed. Using mitochondrial COI marker Jayachandran et al. (2016) confirmed the occurrence of *Nassodonta insignis* H. Adams, 1867 (Gastropoda: Nassariidae) from the Kodungallur - Azhikode backwater, southwest coast of India.

5.2 Results

5.2.1 Community structure of meiofauna in the Vembanad estuary

5.2.1.1 Seasonal variations of meiofaunal communities

During 2011-2012 period, the meiofauna taxa appeared to be well represented with a total of 12 groups. It included foraminifera, kinorhynchs, nematodes, oligochaetes, polychaetes, chironomids, ostracods, harpacticoid copepods, amphipods, tanaids, isopods and bivalves. In all the seasons nematodes were the dominant group. Its percentage contribution was maximum during MN (75.2 %), followed by PRM (50.4) and PM (47.4 %) (Fig. 5.1). During PRM foraminifera formed the second dominant group (9.4 %) followed by bivalves (8.8 %) and harpacticoid copepods (8.6 %), whereas during MN and PM harpacticoid copepods formed the second dominant group, which contributed 11.4 % and 20.3 % respectively. Throughout the season, only oligochaetes (100 %) were present in station 1 showing the maximum annual density of 565/10 cm^2. In station 2, only oligochaetes and chironomids were present during PRM and PM season, whereas during MN only oligochaetes were present there. In station 3, amphipods were the dominant group during PRM (66.7 %) followed by nematodes (33.3 %). During MN from the meiofauna 98.4% were contributed by nematodes and chironomids, harpacticoid copepods and bivalves constituted 0.5 % each. During PM amphipods and isopods were accounted for 36.4 % each. Chironomids and tanaids formed the next important group represented 18.2 % and 9.1 % respectively. Maximum annual density of 17804/10 cm^2 of nematode was observed in station 3. In station 4, foraminifera accounted 43.8 % of the total meiofauna during PRM. The next most abundant taxa were the nematodes (18.8 %) and isopods (12.5 %). During MN nematodes (54.8 %), bivalves (22.6 %), polychaetes (9.7 %) and isopods (9.7 %) were the most abundant meiofaunal taxa. Similar to MN, nematodes were the dominant taxon during PM in station 4, where isopods formed the second dominant group and contributed 23.1 % to the total meiofaunal population. In station 4, annual meiofaunal density was maximum for nematodes (3768 /10 cm^2) and isopods (1036/10 cm^2). In station 5 and 6, nematodes were the dominant group during all the seasons. During PRM nematodes attributed 73.8 %, followed by harpacticoid copepods (13.1 %) and foraminifera (4.9 %). During MN polychaetes formed the second dominant group (22.2 %) followed by foraminifera (11.1 %). In contrast with PRM and MN, bivalves formed the second most abundant taxa (13.3 %) during PM. Isopods (15.7 %), foraminifera (25 %) and bivalves (18.2 %) become the second most abundant meiofaunal taxa in station 6 during PRM, MN and PM. Among the meiofauna, nematodes were having the maximum average numerical density in station 7 (7536/10 cm^2) followed by harpacticoid copepods (1036/10 cm^2). In station 7, foraminifera (34.4 %) and harpacticoid copepods (40.5 %) formed the second dominant meiofaunal taxa during PRM and MN respectively. Similar to MN, harpacticoid copepods formed the second dominant taxa (17.1 %) followed by isopods (14.6 %) during PM. During PRM, meiofauna comprised of bivalves (75 %), nematodes (12.5 %) and harpacticoid copepods (12.5 %) in station 8. During PM meiofauna included harpacticoid copepods (88.9 %), tanaids (6.7 %), nematodes (2.2 %) and polychaetes (2.2 %), whereas during MN, tanaids accounted 100 % of meiofaunal taxa in station 8. In station 9 nematodes were the most abundant taxa, accounting 30.2 % and 57.1% during PRM and MN respectively. During PM in station 9, bivalves and nematodes constituted 52.8 % and 41.7 % of the total meiofaunal organisms respectively. In station 10, annual meiofaunal density was maximum for nematodes (1884/10 cm^2), followed by harpacticoid copepods (471/10 cm^2). Seasonally, nematodes (60.7 %), harpacticoid copepods (60 %) and polychaetes (75 %) were the dominant organisms during PRM, MN and PM respectively. Among the total meiofaunal groups nematodes represented 60.1% followed by harpacticoid copepods (12.5 %) and bivalves (8.5 %). Throughout the season, the harpacticoid copepods were not regularly present in any stations. Only a single observation of amphipods was noted in station 3 during PRM. Foraminifera, kinorhynchs, ostracods and amphipods were not observed during MN. During MN, taxa such as oligochaetes, polychaetes, harpacticoid copepods, tanaids, isopods and bivalves were not regularly present in any station. Similar to MN foraminifera, kinorhynchs and ostracods were not observed during PM. Similar to MN, taxa such as oligochaetes, polychaetes, harpacticoid copepods, amphipods, tanaids and isopods were not regularly present in any station during PM.

During 2012-2013 period, the meiofauna taxa included foraminifera, turbellarians, kinorhynchs, nematodes, oligochaetes, polychaetes, ostracods, harpacticoid copepods, amphipods, tanaids, isopods, bivalves and invertebrate eggs. Similar to 2011-2012 period, nematodes were the dominant group in all seasons. Its percentage contribution was maximum during MN (66.6 %), followed by PRM (51.8 %) and PM (42.8 %) (Fig. 5.2). In contrast with first year, harpacticoid copepods formed the second dominant group (22.5 %) during PRM. During MN polychaetes (13.8%) formed the second dominant group followed by oligochaetes (10.9 %). During PM, foraminifera formed the second most abundant group with 17.4 % of total meiofauna, followed by harpacticoid copepods (9.1 %). There were no oligochaetes in station 1 during PRM and MN, whereas during PM bivalves formed the dominant group (85.7 %), followed by nematodes (14.3 %). In station 2, only two meiofaunal groups were present. The annual meiofaunal density was maximum for oligochaetes (283/10 cm^2) followed by amphipods (94/10 cm^2). Seasonally, similar to first year, oligochaetes represented the most abundant group during PRM in station 2. Whereas during MN and PM there was no meiofaunal taxa, that was observed in station 2. In station 3, oligochaetes (42.5 %) and amphipods (42.9 %) dominated during PRM and PM respectively. In station 4, polychaetes and harpacticoid copepods accounted 58 % and 23.5 % of the total meiofauna during PRM. Oligochaetes were most abundant during MN, constituting 91.7 % of the total meiofauna. During PM, foraminifera and nematodes contributed 74.7 % and 25.3 % respectively. In station 5 annual meiofaunal density was maximum for harpacticoid copepods (25246/10 cm^2), followed by oligochaetes (14224/10 cm^2) and nematodes (6123/10 cm^2). During PRM harpacticoid copepods attributed 54.7 % of the fauna, followed by oligochaetes (30.8 %) and nematodes (10 %) in station 5.

During MN nematodes formed the most dominant group (44.4 %) followed by polychaetes (29.6 %). Foraminifera and nematodes contributed 40 % each to the total meiofaunal population and polychaetes formed the next dominant group contributing 20 %. In station 6, annual meiofaunal density was maximum for nematodes (2732/10 cm^2), followed by harpacticoid copepods (2072/10 cm^2). Seasonally harpacticoid copepods and oligochaetes contributed 44.4 % and 31.3 % each during PRM in station 6. During MN polychaetes formed the most dominant group (52.2 %) followed by nematodes (43.5 %) whereas during PM nematodes formed the most dominant group (50 %). Throughout the season nematodes formed the most dominant group. Among the meiofauna, nematodes (12811/10 cm^2) and polychaetes (848/10 cm^2) were having maximum annual density in station 7 and seasonally nematodes formed the dominant group. In station 8, 9 and 10, the annual average density of nematodes was maximum and it was 27412/10 cm^2, 9514/10 cm^2 and 42013/10 cm^2 respectively. Seasonally, nematodes (83.2 %) and oligochaetes (45.2%) became the most abundant meiofaunal taxa in station 8 during PRM and PM. No meiofaunal groups were encountered in station 8 during MN. In both stations 9 and 10 nematodes were the most abundant group in all the seasons.

Among the total meiofaunal groups nematodes represented 52.5 % followed by harpacticoid copepods (16.7 %) and oligochaetes (14.9 %). During PRM taxa such as tanaids (station 5), isopods (station 5), kinorhynchs (station 8) and bivalves (station 4) were observed only once. During MN, there was no meiofaunal taxa observed in station 1, 2, 3 and 8. The taxa such as ostracods and harpacticoid copepods were not encountered during MN. Turbellarians (station 5), tanaids (station 10), isopods (station 10) and bivalves (station 7) were observed only once during MN. The taxa such as tanaids and isopods were not encountered during PM. At the same time only a single observation of ostracods were made in station 8. Similarly

Kinorhynchs (station 4 and 9) and turbellarians (station 8 and 9) were observed only twice during PM.

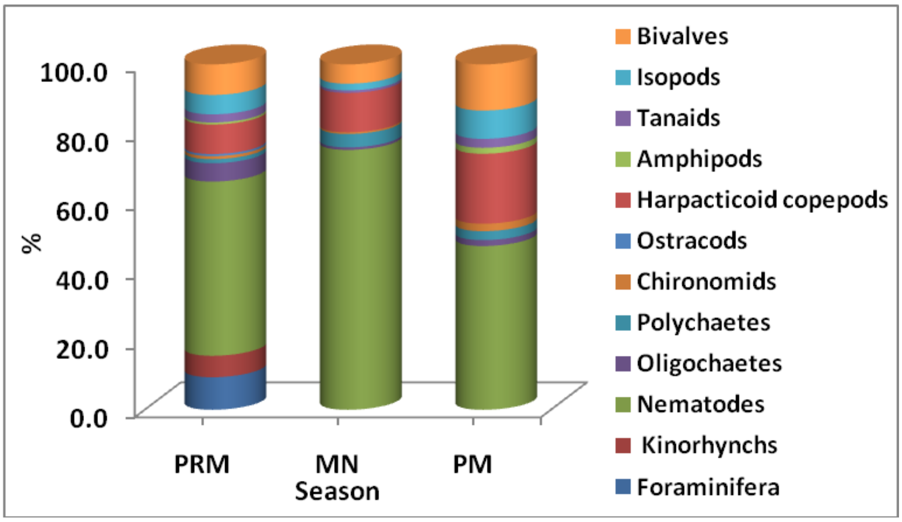


Fig. 5.1 Seasonal variation of percentage abundance of meiofauna in Vembanad estuary during 2011 – 2012 period

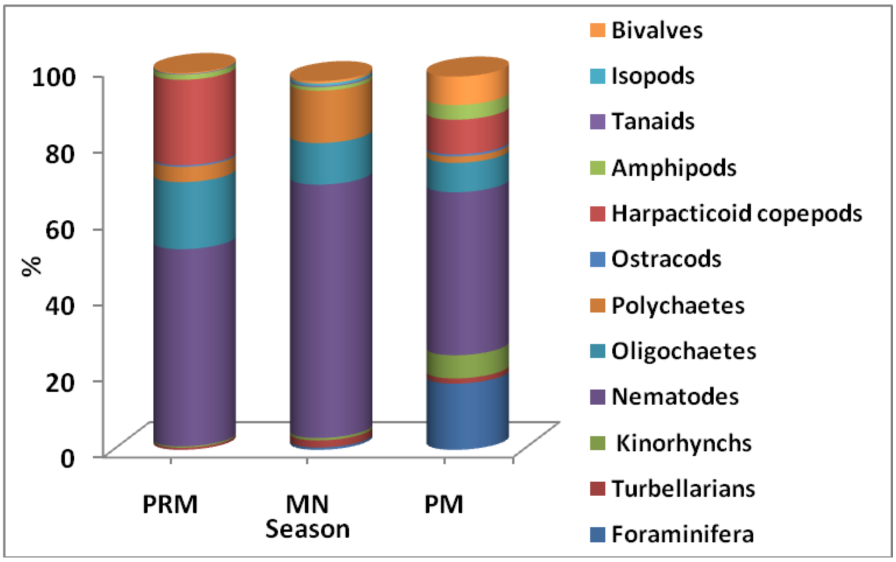


Fig. 5.2 Seasonal variation of percentage abundance of meiofauna in Vembanad estuary during 2012 – 2013 period

5.2.2 Data Analysis

5.2.2.1 Variations in the diversity pattern of meiofauna

Spatially the Shannon diversity index varied from 0 to 2.2 during 2011-2012 period. The minimum diversity value was observed in station 1 and maximum in station 6 (Fig. 5.3). The richness values fluctuated between 0 and 0.97 with minimum value in station 1 and maximum in station 9. The Pielou's evenness ranged from 0 to 0.88 with lowest value observed in station 1 and highest in station 2. The dominance index ranged from 0.3 (station 9) to 1 (station 1). During second year, the richness value ranged from 0.15 (station 1) to 1.02 (station 5) with an average of 0.67 (Fig. 5.4). The diversity values fluctuated between 0.59 (station 1) to 2.42 (station 3) with an average of 1.53. The evenness index varied from a minimum of 0.28 in station 10 to 0.81 in station 2 with an average of 0.59. The dominance index was minimum in station 4 (0.21) and maximum in station 1 (0.75) with an average of 0.49.

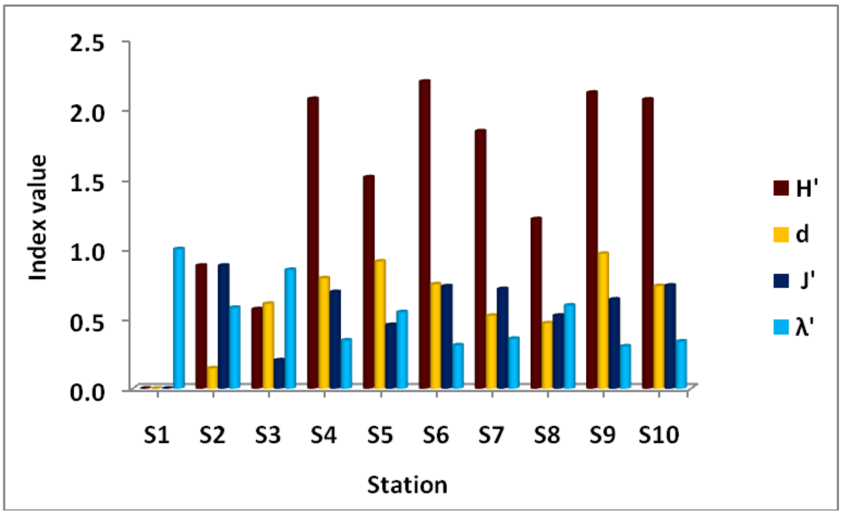


Fig. 5.3 Spatial variation of meiofaunal diversity indices in Vembanad estuary during 2011 – 2012 period

During first year the seasonal variation of diversity index and species richness index was maximum during PRM and minimum during MN (Fig. 5.5). The maximum and mnimum values of evenness index were observed during PM and MN respectively. During second year the values of diversity index and evenness index were maximum during PM and minimum during MN respectively (Fig. 5.5).

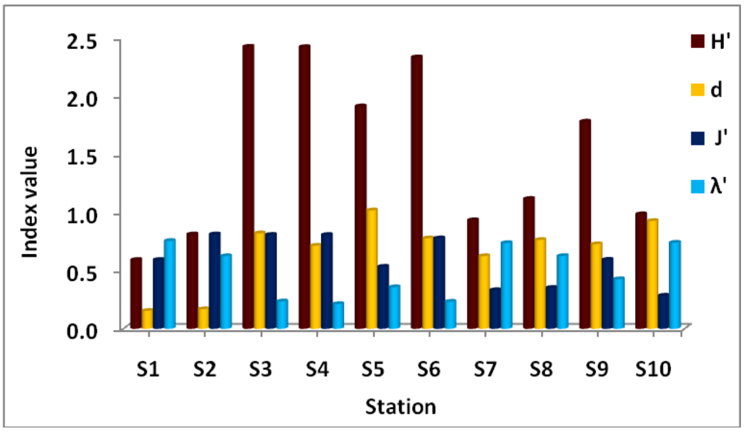


Fig. 5.4 Spatial variation of meiofaunal diversity indices in Vembanad estuary during 2012 – 2013 period

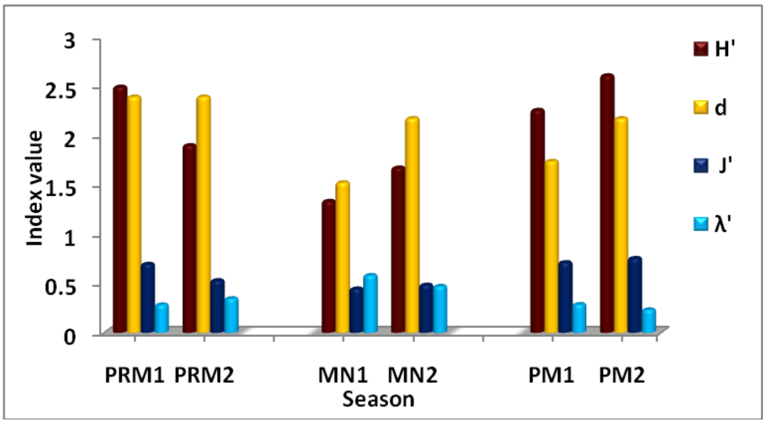


Fig. 5.5 Seasonal variation of meiofaunal diversity indices in Vembanad estuary during 2011-2013 period.

5.2.2.2 Comparison of meiofaunal communities in the south and north zone of TMB

During 2011-2012 period, differences in the composition of the meiofaunal assemblages between sites were determined using cluster analysis and it showed a significant spatial differentiation between the sites where the samples differentiated at a similarity of 10.26 %, into two distinct groups, i.e southern most stations showed grouping (stations 1 to 2) and the remaining stations (stations 3 to 10) formed another group (Fig. 5.6). During 2012-2013 period, cluster analysis depicted that station 1 was separated from all other stations. In the second group station 2 stood apart while the remaining stations formed another group (Fig. 5.7). In 2011-2012 period, the average meiofaunal density was higher in southern (8638/10 cm^2) than the northern zone (8478/10 cm^2), whereas during 2012-2013 period the average meiofaunal density was higher in northern (28684/10 cm^2) than the southern zone (10626/10 cm^2). Differences in the meiofaunal community structure on the south and north of TMB were tested using ANOSIM. No significant differences were detected between southern and northern sectors during the first year (R = - 0.012; $P > 0.05$) and second year (R = -0.83; $P > 0.05$). The results of SIMPER analyses showed that the average dissimilarity of meiofauna in the southern and northern sectors of Vembanad estuary was 54.9 % during first year and 48.45% during second year. During first year, nematodes and harpacticoid copepods contributed 15.37 % and 13.5 % respectively to dissimilarity (Table 5.1). Similarly, during second year nematodes showed the highest dissimilarity (20.16%) followed by harpacticoid copepods (10.68 %) (Table 5.2).

During first year diversity, richness and evenness indices was higher in the northern zone than southern zone of TMB. The Shannon diversity index in the north zone was 1.81 and that of south zone was 1.21. The highest Margalef's richness and Pielou's index in the northern zone was

0.67 and 0.65 and that of south zone was 0.53 and 0.49 respectively. Simpson index was maximum in southern (0.61) than the northern zone (0.4). During second year, by comparing the Shannon diversity value of south and north zone, the highest diversity value was observed in southern zone (1.75) than the northern zone (1.2). Similar to Shannon diversity index, highest Pielou's index was observed in south (0.72) than the north zone (0.39) of TMB. The highest Margalef's richness and dominance index was observed in north. The highest Margalef's richness and dominance index in the northern zone was 0.76 and 0.63 that of south zone was 0.61 and 0.4 respectively.

Table 5.1 SIMPER test results showing the dissimilarity of meiofaunal communities between southern and northern sector of TMB during 2011-2012 period

2011-2012	Average dissimilarity = 54.91 %				
	Group South	Group North			
Groups	Average Abundance	Average Abundance	Average Dissimilarity	Percentage Contribution (%)	Cumulative percentage (%)
Nematodes	6.51	6.94	8.44	15.37	15.37
Harpacticoid copepods	3.17	5.08	7.41	13.5	28.87
Bivalves	3.1	4.61	5.78	10.53	39.41
Polychaetes	1.74	2.91	4.76	8.68	48.08
Isopods	3.45	2.16	4.61	8.39	56.48
Oligochaetes	3.02	2.48	4.2	7.65	64.13
Foraminiferans	2.43	2.44	4.19	7.63	71.76
Tanaids	2.24	3.23	4.18	7.61	79.37

Table 5.2 SIMPER test results showing the dissimilarity of meiofaunal communities between southern and northern sector of TMB during 2012-2013 period

2011-2012	Average dissimilarity = 48.45 %				
	Group South	Group North			
Groups	Average Abundance	Average Abundance	Average Dissimilarity	Percentage Contribution (%)	Cumulative percentage (%)
Nematodes	5.15	11.92	9.77	20.16	20.16
Harpacticoid copepods	4.83	5.28	5.17	10.68	30.84
Polychaetes	3.9	4.85	4.21	8.69	39.53
Oligochaetes	5.95	5.62	4.21	8.69	48.21
Bivalves	1.5	3.56	3.97	8.19	56.4
Foraminifera	2.15	1.88	3.46	7.15	78.89
Amphipods	3.47	3.31	2.9	5.98	84.87

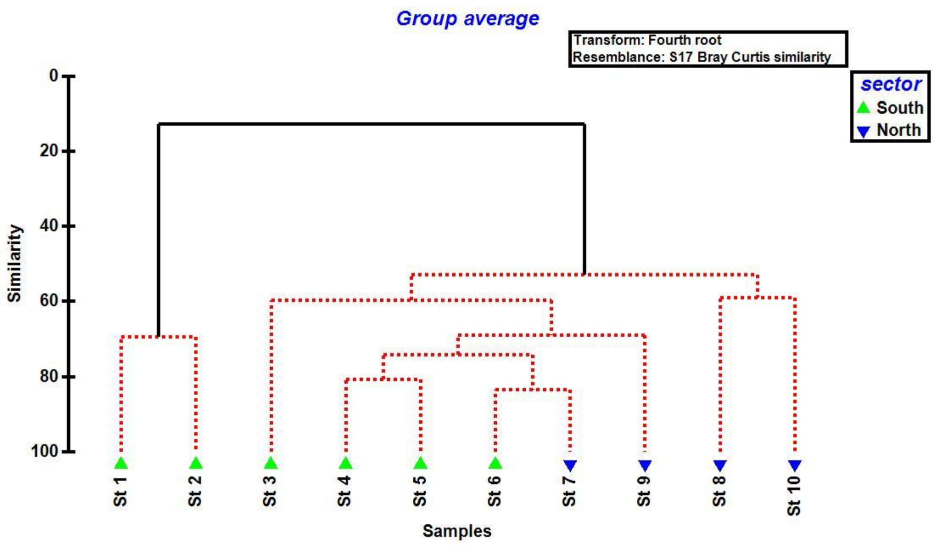


Fig. 5.6 Dendrogram showing the spatial similarities of meiofaunal abundance in the Vembanad estuary during 2011 – 2012 perio**d**

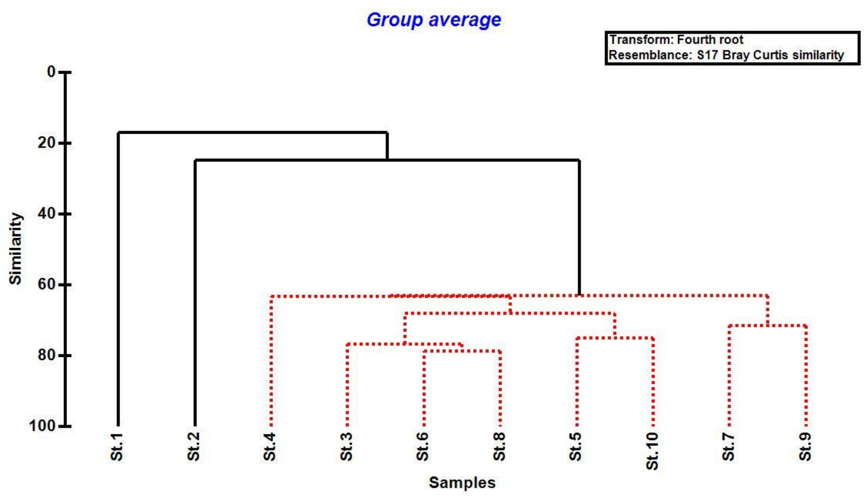


Fig. 5.7 Dendrogram showing the spatial similarities of meiofaunal abundance in the Vembanad estuary during 2012 – 2013period

5.2.2.3 BIOENV analysis of meiofaunal communities

During 2011-2012 period, the BIOENV analyses revealed a strong correlation between environmental parameters and meiofaunal abundance (Table 5.3 and Fig. 5.8). Of all the parameters, available nitrogen alone showed the highest correlation (ρ = 0.696) and also parameters such as organic carbon, depth, salinity, sand and clay fraction best described the meiofaunal community structure. Similar to first year, during 2012-2013 period available nitrogen alone showed a strong correlation (ρ = 0.687) (Table 5.4 and Fig. 5.9). Parameters such as organic carbon, salinity, pH and silt also formed an important factor that determined the community structure of meiofauna.

Table 5.3 Results of BIOENV analysis for meiofauna and environmental parameters in Vembanad estuary during 2011 - 2012 period

Sl. No.	Variables	Variables Selected	Correlation Values (Rho)
1	Depth	7	0.696
2	Salinity	6, 7	0.648
3	Dissolved oxygen	1, 6, 7	0.614
4	Chlorophyll a	2, 6,7	0.587
5	Sediment pH	6-8	0.577
6	Organic carbon	1, 6-8	0.568
7	Available nitrogen	1, 2, 6, 7	0.562
8	Sand	1, 7	0.553
9	Silt	6, 7,10	0.55
10	Clay	6	0.544

Table 5.4 Results of BIOENV analysis for meiofauna and environmental parameters in Vembanad estuary during 2012 - 2013 period

Sl. No.	Variables	Variables Selected	Correlation Values (Rho)
1	Depth	7	0.687
2	Salinity	5-7	0.561
3	Dissolved oxygen	6, 7	0.56
4	Chlorophyll a	5, 7	0.549
5	Sediment pH	7, 9	0.541
6	Organic carbon	2, 7	0.534
7	Available nitrogen	2, 5 - 7	0.518
8	Sand	5-7, 9	0.514
9	Silt	2, 6, 7	0.512
10	Clay	6, 7, 9	0.511

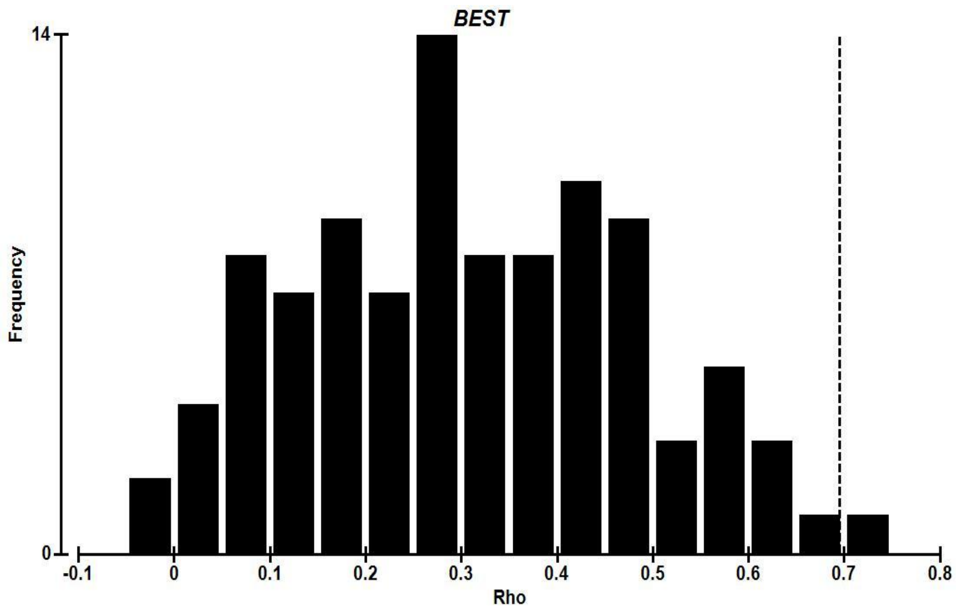


Fig. 5.8 Histogram showing the results of BEST analysis of meiofauna with environmental parameters in Vembanad estuary during 2011-2012 period

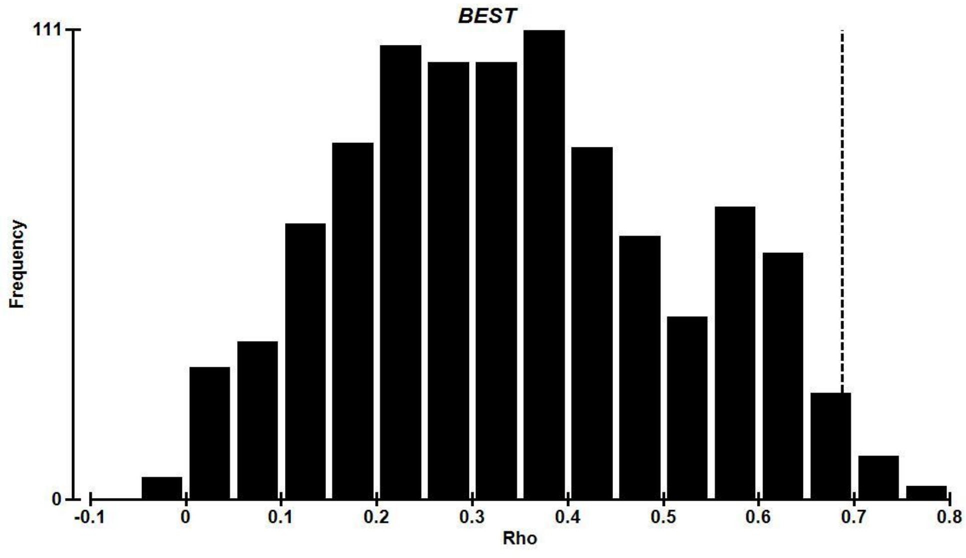


Fig. 5.9 Histogram showing the results of BEST analysis of meiofauna with environmental parameters in Vembanad estuary during 2012-2013 period.

5.2.3 Community structure of macrofauna in the Vembanad estuary

5.2.3.1 Spatio – temporal variation of macrofaunal communities in the Vembanad estuary

During first year, oligochaetes, polychaetes, bivalves and insects were the most abundant macrofaunal groups in station 1. Spatial variation of macrofaunal groups in the Vembanad estuary during both years are given in annexure 5.1 and annexure 5.2. Polychaetes formed the dominant group (783 ± 730 ind./m^2) followed by oligochaetes (388 ± 499 ind./m^2) during the study. While comparing other stations, the highest abundance of oligochaetes was observed in station 1(1634 ind./m^2) during March 2011. Polychaetes and oligochaetes contributed 50.43 % and 24.97 % to the total macrofaunal community (Fig. 5.10). Temporal variation of polychaetes showed the minimum density of 295 ind./m^2 during June, August and October 2011 and the maximum density of 2088 ind./m^2 observed September 2011. Hydrozoans and nemerteans were present only during January 2012 (23 ind./m^2). Distribution of insects showed a low density during most of the months whereas it was not observed during monsoon months. In the case of temporal distribution of bivalaves, tanaids, amphipods and isopods, their presnce was limited to a few months with very low density. During second year macrofaunal composition varied greatly and also its density varied temporally and annually. Bivalves, polychaetes, amphipods, isopods, tanaids, oligochaetes and decapods were the dominant groups. The bivalve density increased greatly and contributed 36% to the total macrofauna. The annual average of macrofaunal groups was maximum for polychaetes (1793 ± 1542 ind./m^2) followed by amphipods (931 ± 1106 ind./m^2) and oligochaetes (916 ± 1489 ind./m^2). The polychaetes and amphipods contributed 48.37 % and 25.1 % to the total macrofaunal density. The density of polychaetes was minimum during March and September 2012 (681 ind./m^2) and maximum during October 2012 (6265 ind./m^2). In the case of amphipods the maximum density was observed during August 2012 (3450 ind./m^2). The density of oligochaetes varied from nil (October and December 2012) to 5448 ind./m^2 (August 2012) and contributed 24.69 % to the total macrofauna.

During 2011-2012 period, polychaetes formed the most abundant macrofaunal group followed by oligochaetes, insects and amphipods in station 2, whereas during 2012-2013 period, amphipods, polychaetes, oligochaetes were the most abundant macrofaunal group. During second year macrofaunal groups such as oligochaetes, polychaetes, amphipods were present during most of the months, whereas during first year faunal density was neglible in most of the months except polychaetes. Polychaete density was negligible during March 2011 whereas the maximum density of 13416 ind./m^2 was recorded during September 2011, with an annual average of 3532 ± 3808 ind./m^2 in the estuary. It contributed 95.3 % to the total macrofaunal population. Oligochaetes forms the second dominant group with an annual average of 64 ± 113 ind./m^2. In the second year period amphipod density was maximum over the other groups and highest number was observed during June 2012 (8013 ind./m^2) with an annual average of 3292 ± 2992 ind./m^2 (47.72%) (Fig. 5.11). Polychaetes, the second dominant group (37.27%) showed maximum density during June 2012 (7741 ind./m^2) with an annual average of 2497 ± 2509 ind./m^2. Oligochaete density was highest in May 2012 (2361 ind./m^2) with an annual average of 880 ± 941 ind./m^2 (12.75%).

During 2011-2012 period, turbellarians, oligochaetes, polychaetes, insects, amphipods, cumaceans, isopods and bivalves were the major macrofaunal groups present in station 3. Similar to station 1, polychaetes formed the most common macrofaunal group and most of the other macrofaunal groups were not observed regularly. The density of polychaetes reached upto 3065 ind./m^2 during August 2011 with an annual average of 1438 ± 1153 ind./m^2, contributing 54.68 % to the total macrofauna (Fig. 5.12).

Amphipods formed the second dominant group with highest density during April 2011 (5198 ind./m^2) with an annual average of 902 ± 1740 ind./m^2. Apart from stations 1 and 2, isopod formed the third dominant group in station 3 showing the highest density of 1339 ind./m^2 during September 2011, with an annual average of 199 ± 383 ind./m^2. During the second year period, distribution and abundance of macrofaunal groups varied significantly. Amphipods forms the dominant group (4254 ± 6833 ind./m^2) followed by polychaetes (1035 ± 900 ind./m^2) and isopods (910 ±1484 ind./m^2). The other minor macrofaunal groups were oligochaetes (42 ± 137 ind./m^2), insects (32 ± 53 ind./m^2), bivalves (30 ± 49 ind./m^2), nematodes (8 ± 26 ind./m^2) etc. Miscellaneous forms in the station included unidentified fragments of organisms.

In 2011-2012 period, macrofaunal composition of station 4 was represented by turbellarians, nematodes, oligochaetes, polychaetes, amphipods, cumaceans, tanaids, isopods and bivalves etc. In which polychaetes formed the most dominant group followed by isopods, tanaids and amphipods. Polychaetes showed the maximum density during October 2011 (5312 ind./m^2) with an annual average of 1795 ± 1731 ind./m^2 (54.98%) (Fig. 5.13). The maximum density of isopods were observed during April 2011 (2906 ind./m^2) with an annual average of 866 ± 1027 ind./m^2. They constituted 26.54 % to the total macrofauna. Maximum abundance of tanaids was observed during November (1771 ind./m^2) with an annual average of 272 ± 504 ind./m^2 (8.34%). Similar to isopods, higher abundance of amphipods were observed during April 2011 (1021 ind./m^2) and noted a lower annual average of 138 ± 287 ind./m^2 (4.23%). During 2012- 2013 period, the macrofaunal composition and abundance value varied considerably. Isopods, amphipods, polychaetes, bivalves and oligochaetes were the most dominant macrofaunal components. Maximum density of isopods was observed during July 2012 (11554 ind./m^2) and minimum

(90.8 ind./m^2) during March 2012 (71.45%). Comparing with other stations, the maximum annual average density of isopods was observed in station 4 (3153 ± 3477 ind./m^2). The annual average denisity of amphipods was 532 ± 867 ind./m^2 and it contributed 12.04 % to the total macrofauna. Compared to the other macrofaunal groups, polychaetes were the permanent inhabitants of the area, though its abundance varies temporally and contributed 7.33% to the total macrofauna. The minimum density of 91 ind./m^2 was observed during April 2012 and maximum of 817 ind./m^2 during June 2012.

During first year, altogether nine macrofaunal groups were observed in station 5. Turbellarians, oligochaets, polychaetes, insects, harpacticoid copepods, isopods, amphipods, tanaids and bivalves were encountered in the macrofaunal groups. Polychaetes formed the most abundant and widely distributed groups in the area. Its abundance varied from a negligible level during March 2011, July 2011 and February 2012 to 885 ind./m^2during August 2011 and October 2011. Compared to stations 1 to 4, the annual average of polychaetes was very low in Station 5 (507 ± 487 ind./m^2) and constituted 75.49 % to the total macrofauna (Fig. 5.14). Oligochaetes formed the second dominant macrofaunal group with an annual average of 62 ± 128 ind./m^2 (9.29%). Isopods, polychaetes and amphipods were the dominant macrofauna observed during second year. Similar to station 4 isopods formed the most abundant group in the area (35.1%). Polychaetes formed the second dominant group and its maximum abundance was observed during May 2012 (1452) with an annual average of 236± 407 ind./m^2. Other benthic groups showed an irregular pattern of distribution with very low abundance.

In station 6, polychaetes were common in all the months and occupied first position with regard to the abundance of macrobenthos. The maximum value was recorded during March 2011 (1952 ind./m^2) with an annual average of 496 ± 630 ind./m^2 (68.23%) (Fig. 5.15). Bivalves showed its second highest density (108 ± 277 ind./m^2) (14.84%) followed by isopods (47 ± 72 ind./m^2) (6.51%). During the study period oligochaetes were not present in this zone. Similar to stations 4 and 5, isopods were most common and showed the highest density among macrobenthos (420 ± 502 ind./m^2) (41.49%) during second year. Amphipods formed the second dominant group (175.93 ± 320 ind./m^2) and constituted 17.38 % of total macrobenthos. The next dominant groups, such as tanaids and polychaetes contributed 16.26 % and 15.7 % to the total macrobenthos. Bivalves were found only in few numbers in limited months with an annual average of 30 ± 41 ind./m^2.

In station 7 macrofauna included 13 major groups such as turbellarians, nematodes, oligochaetes, polychaetes, insects, harpacticoid copepods, isopods, amphipods, cumaceans, decapods, mysids and bivalves. Polychaetes dominated by contributing 68.23% to the total macrobenthos with an annual average of 815 ± 539 ind./m^2 (Fig. 5.16) Among macrofauna polychaetes were observed in all months except November 2011, whereas other minor groups were observed only during a few months. Isopods formed the second dominant group (549 ± 954 ind./m^2) followed by amphipods (274 ± 522 ind./m^2). When comparing other stations, highest numerical abundance of insects was observed in this area during September 2011 (590 ind./m^2). During 2012-2013 period, the variation in dominance of macrofaunal groups were observed. Bivalves formed the dominant group with annual average value of 1999 ± 5900 ind./m^2 contributing 56.98 % to the total fauna. Isopods were the second dominant group (620.47 ± 783 ind./m^2), followed by polychaetes (384 ± 363 ind./m^2) and amphipods (295 ± 375 ind./m^2).

During 2011-2012 period, the macrofaunal composition in station 8 mainly consisted of polychaetes, bivalves and tanaids. Turbellarians, oligochaetes, insects, amphipods, cumaceans, decapods formed the minor contributors. When comparing with other stations highest abundance of turbellarians were observed during February 2012 (272 ind./m^2). Simultaneously the numerical abundance of macrofaunal groups were very less in this station. Polychaetes showed the maximum average density (568 ± 927 ind./m^2) followed by bivalves (157± 254 ind./m^2) and tanaids (110 ± 158 ind./m^2). During 2012-2013 period, bivalves formed the most dominant and abundant group (2189 ± 2704 ind./m^2) and contributed 63.71 % to the total macrofauna (Fig. 5.17). Isopods (495.6 ± 719.6 ind./m^2), tanaids (255.38 ± 359.7 ind./m^2) and amphipods (255.38 ± 205 ind./m^2) formed the other dominant groups contributing 14.43%, 7.4% and 7.4% respectively.

In station 9 polychaetes forms the most abundant and dominant macrofaunal group (2418 ± 2586 ind./m^2) contributing 83.25 % to the total macrofauna (Fig. 5.18). The other macrofaunal groups included isopods, amphipods and bivalves which contributed 9.9 %, 3.45% and 1.69 % to the total macrofaunal population respectively. During second year bivalves formed the most abundant and dominant macrofaunal group (12640 ± 16551 ind./m^2) (84.6%). Isopods formed the second dominant group (1089 ± 3521 ind./m^2), contributing 7.29 % to the total macrofauna. Amphipods and tanaids formed the other dominant macrofaunal groups contributing 4.9% and 1.48% to the total macrofauna.

During 2011-2012 period, macrofauna were composed of 14 groups in station 10. Polychaetes formed the most abundant group followed by bivalves, contributing 47.73 % and 37.91% to the total macrofauna respectively (Fig. 5.19). Among macrofaunal groups polychaetes showed the highest density. It was present in all the months and its abundance varied from a minimum of 250 ind./m^2 to a maximum of 20180 ind./m^2 with an annual average of 4230 ± 5630 ind./m^2. The aggregation of tubiculous polychaetes forming massive structures with the bivalve, *Modiolus* was observed in this region. Its abundance reached upto 34413 ind./m^2 during

February 2012. Hydrozoans were encountered in this station during October 2011 (4404 ind./m^2) and February 2012 (772 ind./m^2). Similarly ostracods were present in this zone. The highest numerical abundance of ostracods were observed during October 2011. Similarly when comparing with the other stations, highest abundance of decapods and nematodes were present during October 2011 (409 ind./m^2) and December 2011 (204 ind./m^2) respectively. During 2012-2013 period polychaetes formed the most abundant group (6951.8 ± 9490 ind./m^2) and contributed 65.4% to the total macrofauna population. Bivalves formed the second dominant group with an annual average of 2957± 3724 ind./m^2 (27.82%). The lowest numerical abundance of polychaetes was observed during June and July 2012.

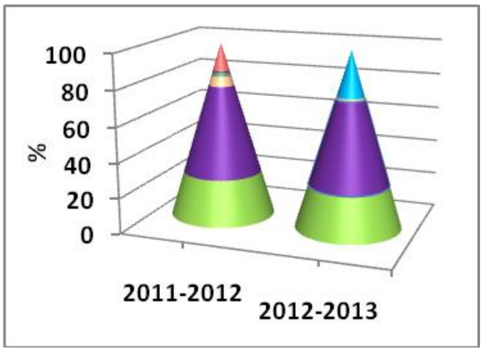


Fig. 5.10 station 1

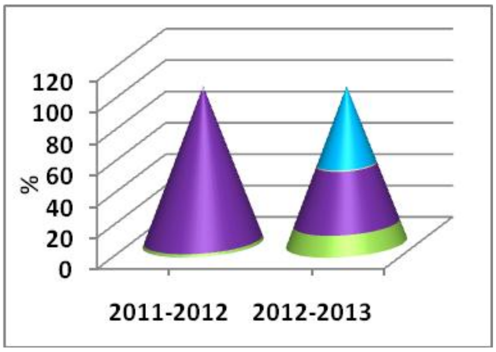


Fig. 5.11 station 2

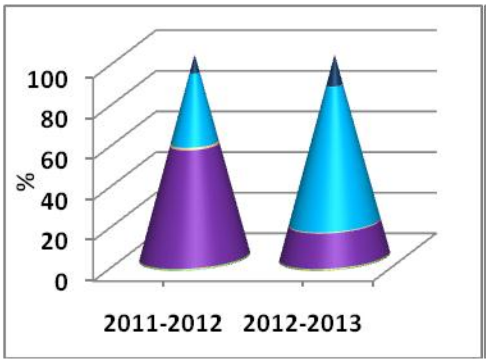


Fig. 5.12 station 3

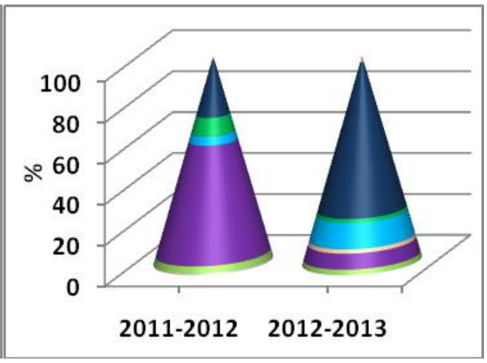


Fig. 5.13 station 4

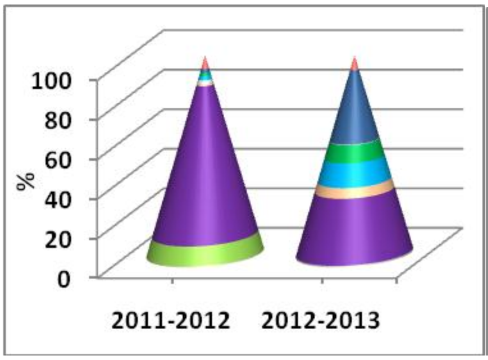


Fig. 5.14 station 5

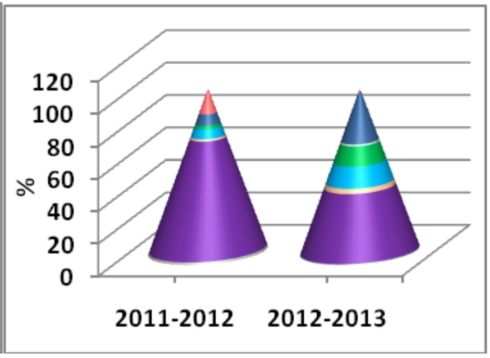


Fig. 5.15 station 6

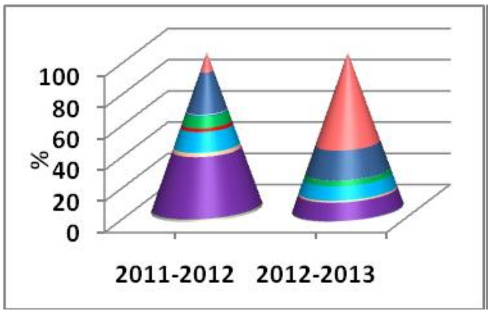


Fig. 5.16 Station 7

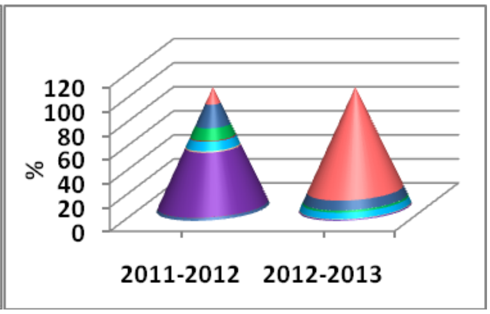


Fig. 5.17 station 8

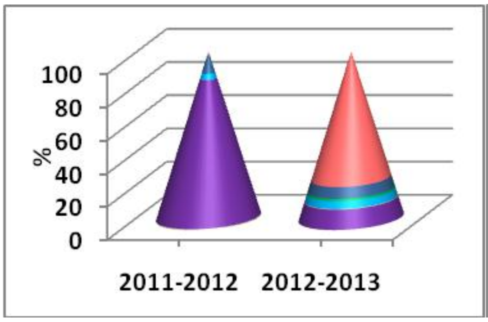


Fig. 5.18 station 9

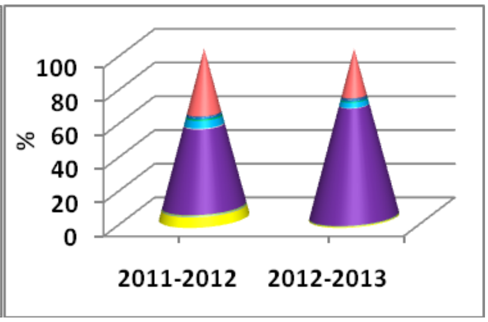


Fig. 5.19 station 10

Fishes
Gastropods
Bivalves
Isopods
Mysids
Decapods
Tanaids
Cumaceansns
Amphipods
Harpacticoids
Ostracods
Insects
Polychaetes
Oligochaetes
Nematodes
Turbellariansns
Nemerteans
Hydrozoans

Fig. 5.10 to 5.19 Spatial percentage composition of macrofaunal groups in Vembanad estuary during 2011-2012 and 2012-2013 period

Hydrozoans, nemerteans, turbellarians, nematodes, oligochaetes, polychaetes, insects, ostracods, harpacticoid copepods, amphipods, cumaceans, tanaids, isopods, mysids, decapods, gastrapods, bivalves and fishes were the different macrofaunal observed in the Vembanad estuarine system. During 2011-2012 period, the spatial abundance of macrofauna showed the maximum average abundance in station 10 (5317 ind./m^2), followed by station 2 (2223 ind./m^2) and station 4 (1959 ind./m^2). The lowest numerical abundance was observed in station 5 (403 ind.m^2) and second lowest in station 6 (436 ind./m^2). When comparing the sum of spatial average of macrofaunal groups, polychaetes showed the maximum average abundance (av. 16580 ind./m^2). Bivalves (4328 ind./m^2), isopods (2262 ind./m^2), amphipods (1977 ind./m^2) and oligochaetes (706 ind./m^2) were the other dominant groups. Considering the total percentage contribution of macrofaunal groups, 59.8% was contributed by polychaetes, 15.6% by bivalves, 8.2% by isopods, 7.1% by amphipods and 2.5% by oligochaetes (Fig. 5.20). ANOVA results of macrofaunal groups are given in annexure 5.3. During 2011-2012 period, ANOVA of oligochaetes showed that there was a significant spatial variation at 1% level ($P \leq 0.01$). Similar to oligochaetes, ANOVA of polychaetes were significant at 1% level ($P \leq 0.01$). The ANOVA of amphipods showed that there was a significant variation between stations ($P \leq 0.01$), seasons ($P \leq 0.05$) and interaction between season and station ($P \leq 0.01$).The ANOVA results showed that the spatial variation of tanaids were significant at 5% level ($P \leq 0.05$). Similar to tanaids, the spatial variation of isopods were significant at 5% level ($P \leq 0.05$). The ANOVA results of decapods showed that the spatial variation ($P \leq 0.01$) and seasonal variation ($P \leq 0.01$) were significant at 1% level. The interaction between season and station were also significant at 1% level ($P \leq 0.01$). In the case of harpacticoid copepods ($P \leq 0.01$) and ostracods ($P \leq 0.01$), there was a significant spatial variation at 1% level. The less abundant macrofunal groups such as mysids ($P \leq 0.05$), gastropods ($P \leq 0.05$) and fishes ($P \leq 0.01$) showed a significant spatial variation in its distribution. During first year, temporal distribution of macrofauna showed that maximum average abundance was observed during February 2012 (373 ind./m^2) followed by October 2011 (257 ind./m^2) and September 2011 (186 ind./m^2). The minimum average abundance was observed during July 2011 (64 ind./m^2).

During 2012-2013 period, the sum of spatial average of macrofaunal density showed that bivalves formed the most abundant group (20059 ind./m^2) followed by polychaetes (13684 ind./m^2), amphipods (10882 ind./m^2) and isopods (7133 ind./m^2). Bivalves contributed 36.06 %, polychaetes (24.69%), amphipods (19.56%) and isopods (12.82%) (Fig. 5.21) The spatial distribution of macrofauna showed that the maximum average density was observed in station 9 (8964 ind./m^2) followed by station 10 (6376 ind./m^2) and station 2 (4138 ind./m^2). The lowest density was observed during station 5 (449 ind./m^2) and second lowest in station 6 (607 ind./m^2). During second year ANOVA results (Annexure 5.3) showed a significant variation in oligochaetes distribution among station ($P \leq 0.01$). Similarly polychaetes showed a significant spatial variation in abundance at 1% level ($P \leq 0.01$). Additionally a significant seasonal and spatial interaction ($P \leq 0.01$) was observed in the abundance of polychaetes in Vembanad estuarine system. ANOVA analysis detected a significant difference in amphipod abundance among stations at 5% level ($P \leq 0.05$). Similar to amphipods, tanaids ($P \leq 0.05$) and isopods ($P \leq 0.05$) showed significant spatial differences at 5% level. In contrast, the first year two-way ANOVA of bivalves showed a significant spatial variation ($P \leq 0.05$). In the case of decapods two-way ANOVA detected a significant difference in interaction among season and station ($P \leq 0.05$). Turbellaria ($P \leq 0.01$) and harpacticoid copepods ($P \leq 0.01$) showed a significant spatial difference in its abundance at 1% level. The spatial variation in the abundance of gastropods ($P \leq 0.05$) and fishes ($P \leq 0.05$) were significant at 5% level. Monthly variation of isopods noted that maximum average abundance was observed during March 2012 (437 ind./m^2) followed by December 2012 (407 ind./m^2) and January 2013 (353 ind./m^2).

During both the years the maximum average abundance of polychaetes was observed in station 10 (Fig. 5.22). In the case of oligochaetes, during both the years the maximum average abundance was observed in station 1 (Fig. 5.23). During the first year, maximum abundance of bivalves was observed in station 10, whereas during second year it was in station 9 (Fig. 5.24). Both the years, maximum abundance of amphipods was observed in station 3 (Fig. 5.25). During first year the average abundance of isopods was maximum in station 4 (3153 ind./m^2) followed by station 9 (1089 ind./m^2). During the second year the average abundance of isopods was maximum in station 4 (866 ind./m^2), followed by station 7 (548 ind./m^2) (Fig. 5.26). During the first year, maximum average abundance of tanaids was observed in station 4 and in second year it was in station 8 (Fig. 5.27). Hydrozoans include hydroid (hydra) in station 1and obelia colony in station 10 and Gobidae was the only representative family of fishes.Among oligochaetes five species, *Pristina foreli, Tubifex tubifex, Monopylophorus* sp., *Aulodrilus pigueti* and *Limnodrilus* sp. and from gastropods, *Nassodonta insignis* were the only representative species identified during the study period.

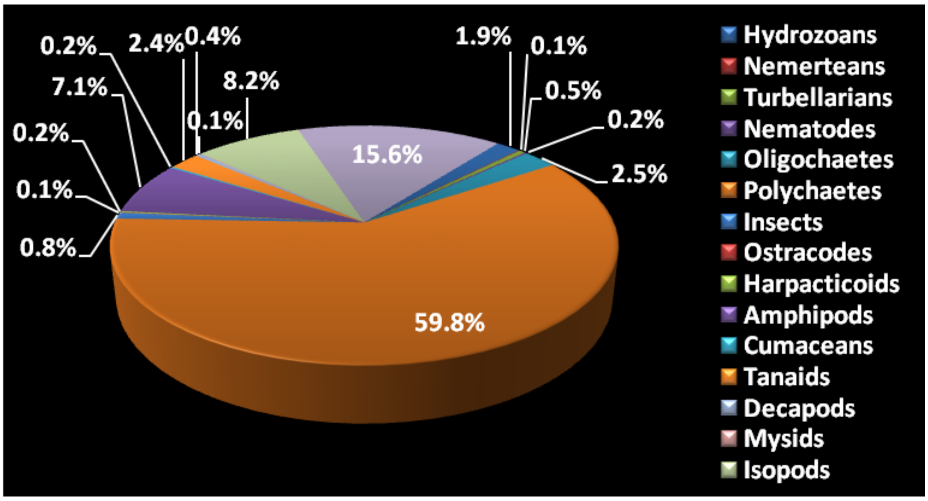


Fig. 5.20 Mean percentage abundance of macrofauna on a spatial scale in Vembanad estuary during 2011 – 2012 period

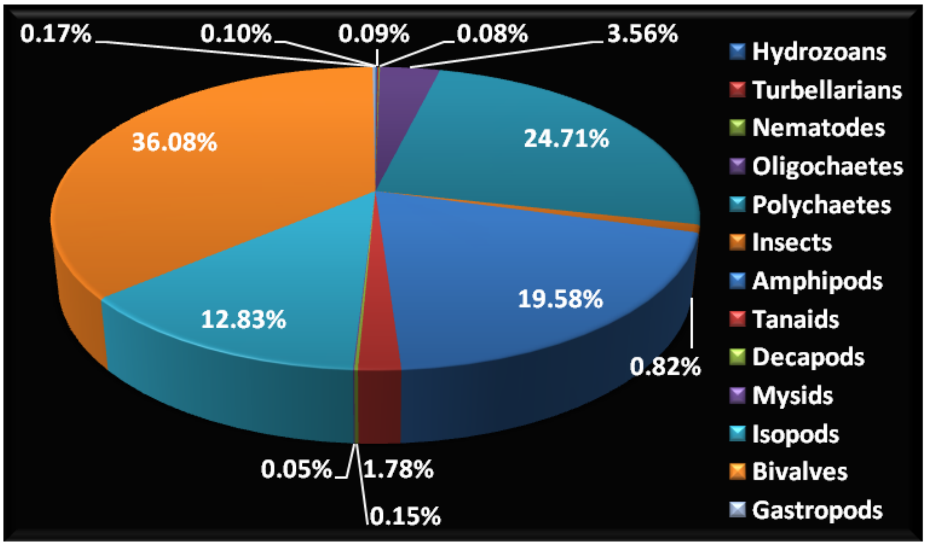


Fig. 5.21 Mean percentage abundance of macrofauna on a spatial scale in Vembanad estuary during 2012 – 2013 period

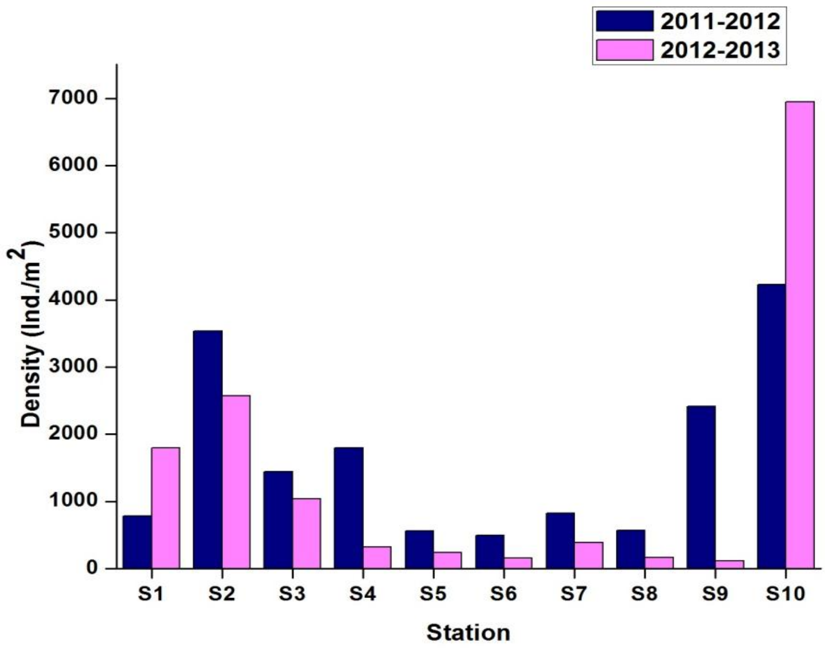


Fig. 5.22 Variation of mean density of polychaetes in Vembanad estuary both during 2011-2012 and 2012-2013 period

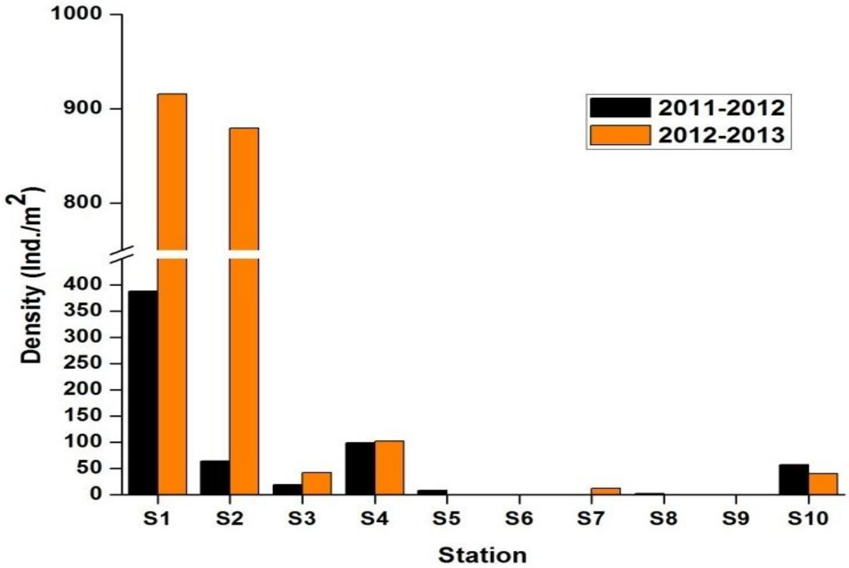


Fig. 5.23 Variation of mean density of oligochaetes in Vembanad estuary both during 2011-2012 and 2012-2013 period

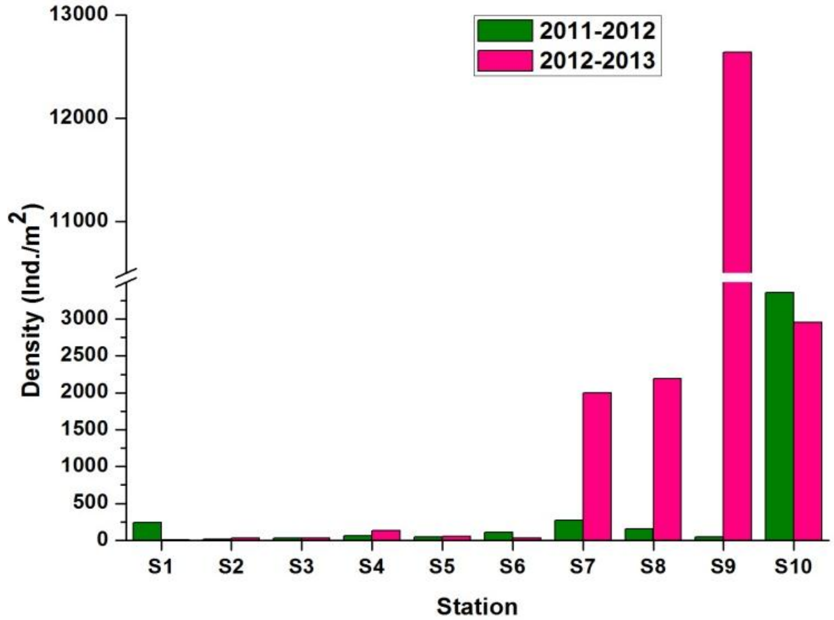


Fig. 5.24 Variation of mean density of bivalves in Vembanad estuary both during 2011-2012 and 2012-2013 period

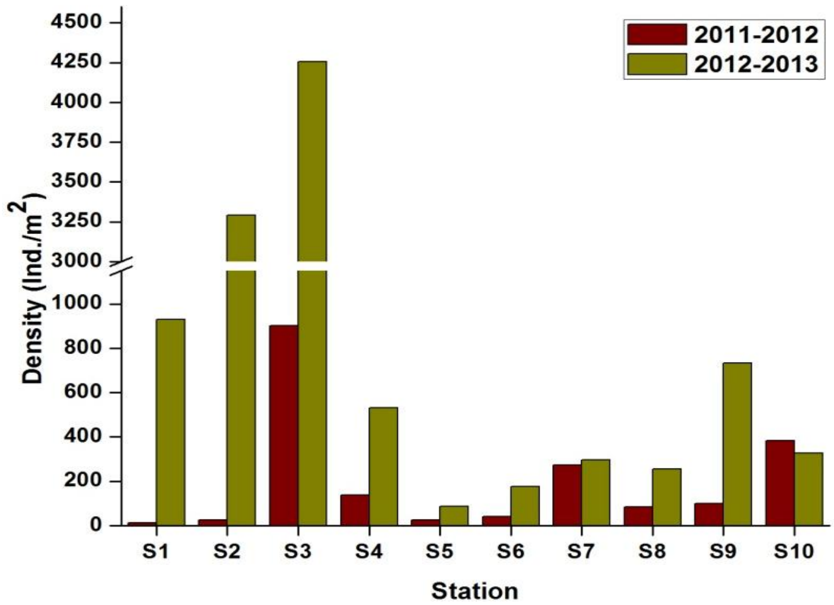


Fig. 5.25 Variation of mean density of amphipods in Vembanad estuary both during 2011-2012 and 2012-2013 period

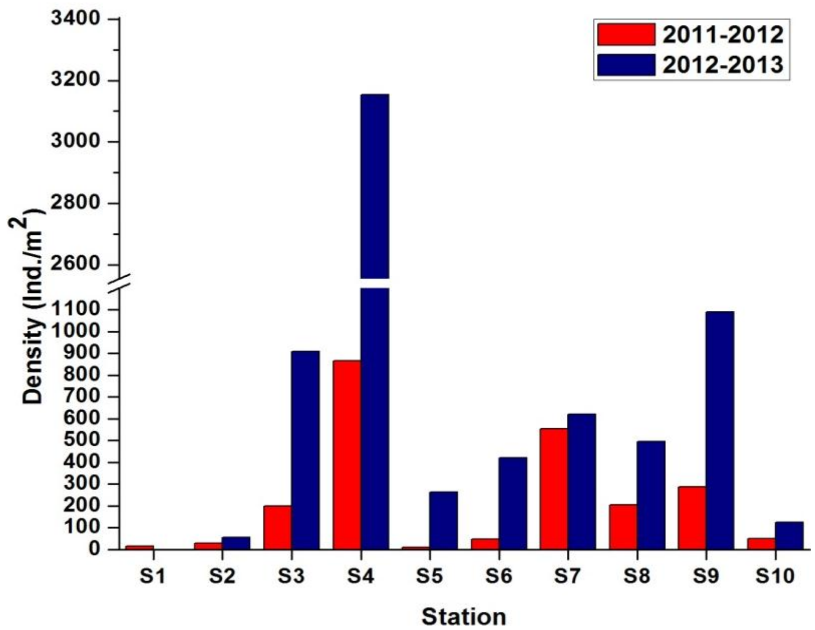


Fig. 5.26 Variation of mean density of isopods in Vembanad estuary both during 2011-2012 and 2012-2013 period

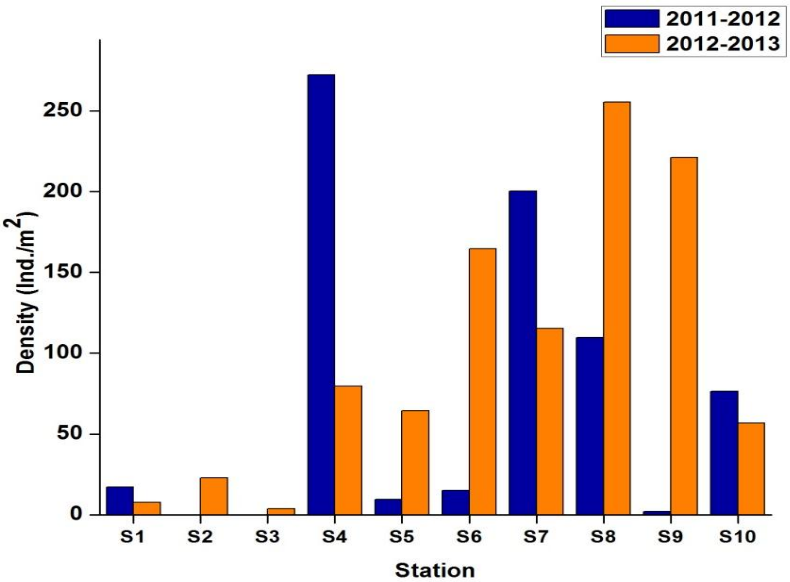


Fig. 5.27 Variation of mean density of tanaids in Vembanad estuary both during 2011-2012 and 2012-2013 period

5.2.3.2 Seasonal variation of macrofaunal communities in Vembanad estuary

Seasonal variation of macrofaunal communities showed that in all the seasons polychaetes formed the dominant group in station 1. During 2011-2012 period, polychaetes contributed maximum during MN (67%) followed by PM (52.1%) and PRM (43%) (Fig. 5.28); whereas during second year, the percentage contribution of polychaetes varied with different seasons and oligochaetes dominated over it during PRM and MN. During first year, bivalves formed the second dominant group (26.4%) during PRM; whereas in monsoon it was oligochaetes (21.8%). Similar to MN, oligochaetes formed the second dominant group during PM (34%). During second year the seasonal variation of macrofaunal groups were clearly distinct from first year. The oligochaetes during PRM formed the dominant group (35.4%) followed by polychaetes. Polychaetes percentage contribution was maximum during PM (67.8%) followed by PRM (31%) and MN (30%). In the case of monsoon oligochaetes forms the dominant group (38.7%)

followed by amphipods (30.3%). During PM polychaetes became the dominant group (67.8%) followed by amphipods (22.7%).

In station 2 polychaetes formed the dominant group in all season and contributed more than 80% during 2011-2012 period (Fig. 5.29). Isopods formed the second dominant group during PRM (7.8%) whereas during MN (98 %) and PM (93 %) polychaetes contributed a major share in the macrofaunal composition. During 2012-2013 period the seasonal variation of macrofauna varied greatly. Amphipods formed the major dominant group during PRM (41.9%) followed by polychaetes (29%) and oligochaetes (27.8%). During monsoon polychaete formed the dominant group (49.8%) followed by amphipods (38.5%). Similar to PRM amphipods showed the maximum contribution (61.9%) during PM followed by polychaetes (22.6%) and oligochaetes (13.9%).

In both the years, amphipods were the dominant group in station 3 during PRM. It contributed 67% and 80.49% during first year and second year period respectively (Fig. 5.30). Polychaete was the second dominant group which contributed 27.4 % and 10.35% during 2011-2012 and 2012-2013 periods respectively. Whereas during MN of first year period, polychaete was dominating (77.9%) followed by isopods that formed the second dominant group (15.9%). Isopods increased in dominance (47%) during MN of 2012-2013 period, followed by polychaetes (27.17%). Similar to MN of 2011-2012 period polychaetes formed the major contributing group (83.1%) during PM followed by isopods (8.4%), whereas during 2012-2013 period amphipods topped the list with 68.42%, followed by polychaetes (18.6%).

During 2011-2012 period isopods were found to be the dominant group by constituting 49.8% of the total macrofauna during PRM in station 4 (Fig. 5.31). Polychaetes formed the next dominant group with a contribution of 26.9%. During MN, polychaetes ranked first with a contribution of 78.6% and contribution of isopods decreased to 9.6%. Similarly during PM, polychaetes ranked first with a contribution of 58.5% and isopods (22.3%) and tanaids came next in the order with a maximum percentage contribution (12.4%). During second year period, isopod topped the list with greater than 60% contribution in all the seasons. Similar to isopods, amphipods formed the next dominant group in all the season.

In station 5 polychaetes formed the dominant group in all the seasons during 2011-2012 period. During PRM and MN it contributes 92% and 83.8% respectively (Fig. 5.32), whereas during PM, the percentage contribution of polychaetes decreased to 57.2%. Similar to first year, polychaetes were the dominant group during PRM (54%) of second year and its percentage contribution decreased during MN (21.43 %) and PM (21.43%). Isopods ranked the second dominant group during MN (57.22%) and PM (27.38%).

During 2011 – 2012 period polychaetes formed the dominant group in all the seasons of station 6 (Fig. 5.33). Bivalves ranked second with contribution of 25.8% in PRM. During MN amphipods came next in the order (14.9%) and during PM it was isopods (11.2%). During second year period, polychaetes became the dominant macrofaunal groups during PRM (32.25%) followed by amphipods (30.96 %) and tanaids (17.42 %). In contrast with first year, during MN, isopods were found to be the dominant group that constituted 52.88 %, followed by amphipods (15.55%) and tanaids (15.55%). Similar to MN, isopods formed the dominant group during PM (50.32 %) followed by tanaids (16.12%).

In station 7, polychaetes formed the dominant group in PRM during both the years (Fig. 5.34). It contributed 56.2% and 30.51 % during 2011-2012 and 2012-2013 period respectively. Similarly bivalves formed the next dominant group during PRM. During MN isopods topped the macrofaunal group with 40.4% and 50.99 % contribution during both the years. Similarly polychaetes formed the next dominant group contributing 22.62% and 15.23% in 2011-2012 and 2012-2013 periods respectively. There was no seasonal similarity of macrofaunal distribution during PM of both the years. During 2011-2012 period polychaetes formed the dominant group (38.8%) followed by bivalves (25.4%) and amphipods (19.6%), whereas during 2012-2013 period bivalves were the dominant group (79.19%), followed by isopods (11.26%) and polychaetes (3.27%).

In station 8 during premonsoon season polychaetes formed the dominant group (62.7%) followed by bivalves (16.4%) (Fig. 5.35). Similarly during MN polychaetes formed the dominant group (52.8%) but amphipods become the second dominant group contributing 33.3%, whereas during PM, bivalves were the most dominant group contributing 53.2% followed by polychaetes (17.2%) and tanids (9.7%). During 2012-2013 period isopods formed the dominant group (37.68%) followed by tanaids (16.77%) and bivalves (16.35%) in PRM season. But in MN, the bivalves (70.76%) formed the dominant group and isopods followed in the second position (10.21%).

Polychaetes formed the dominant group in all the seasons during 2011-2012 period in station 9 (Fig. 5.36). In all the seasons, polychaetes contributed greater than 65% of the benthic fauna. During 2012-2013 period bivalves contributed greater than 90% to the total macrofauna in PRM and MN season, whereas in PM, bivalve contribution decreased towards 49.05% and isopods contribution increased upto 35.57%.

The bivalves (2011-2012) formed the dominant group during PRM (63.9%) followed by polychaetes (29.2%) in station 10 (Fig. 5.37). During MN and PM polychaetes formed the dominant group contributing 89.4%

and 62.9% respectively. Similar to first year, bivalves were the dominant group during PRM but its percentage contribution decreases to 45.14% in second year and polychaetes were the second dominant group (44.92%). Similar to first year, polychaetes were the dominant group contributing 61.54% and 81.45% during MN and PM respectively. Bivalves formed the second dominant group during both MN (34%) and PM (14.12%) respectively.

Considering the overall macrofaunal density of the ten stations together, macrofaunal population density were maximum during PRM in 2011-2012 period. When comparing PRM (40.9%), a considerable decrease in the population density of macrofauna was observed during MN (24.5%) and PM (34.6%). During 2012-2013 period the seasonal variation of macrofaunal groups were not clearly distinct from each other and showed a minimal variation except during PM. In contrast with first year, maximum population density was observed during PM (37.09 %) followed by MN (31.94%) and PRM (30.97%) in second year.When comparing the seasonal variation of macrofaunal groups in the whole Vembanad estuarine system, during 2011-2012 period, polychaetes formed the dominant group in all the season and maximum contribution was observed during PM (65%) followed by MN (60.5%) and PRM (46.7%). During PRM bivalves formed the second dominant group (31.1%) followed by amphipods (10.8%). During MN oligochaetes and isopod ranked second and third with contribution of 17.3% and 8.7% respectively. Isopods and bivalves contributed 8.9% and 7.9% respectively to the total benthic organisms during PM. During 2012-2013 period bivalves formed the dominant group during PRM and MN with a contribution of 44% and 41% respectively. Polychaetes formed dominant group during PM contributing 34% to the total macrofauna.

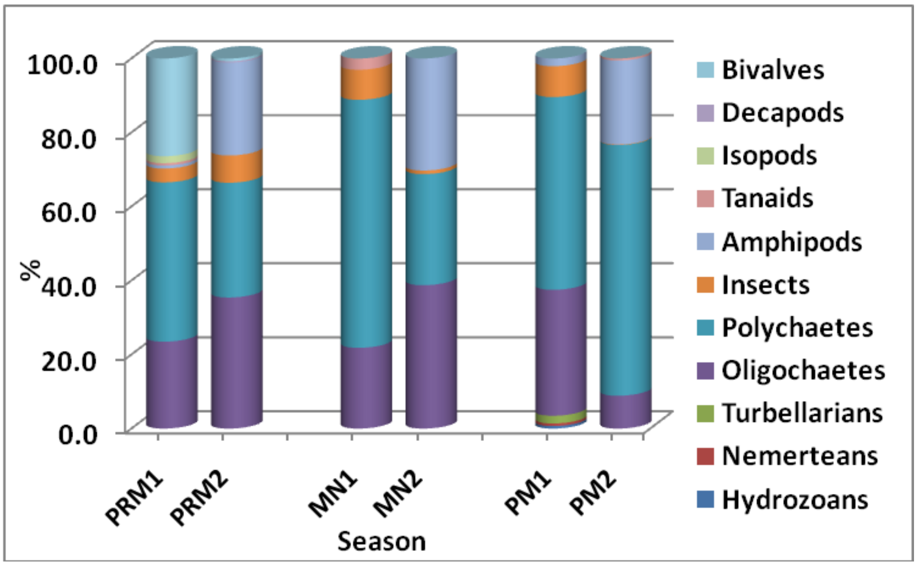


Fig. 5.28 Mean percentage abundance of macrofauna in station 1 in Vembanad estuary during 2011-2012 and 2012-2013 period

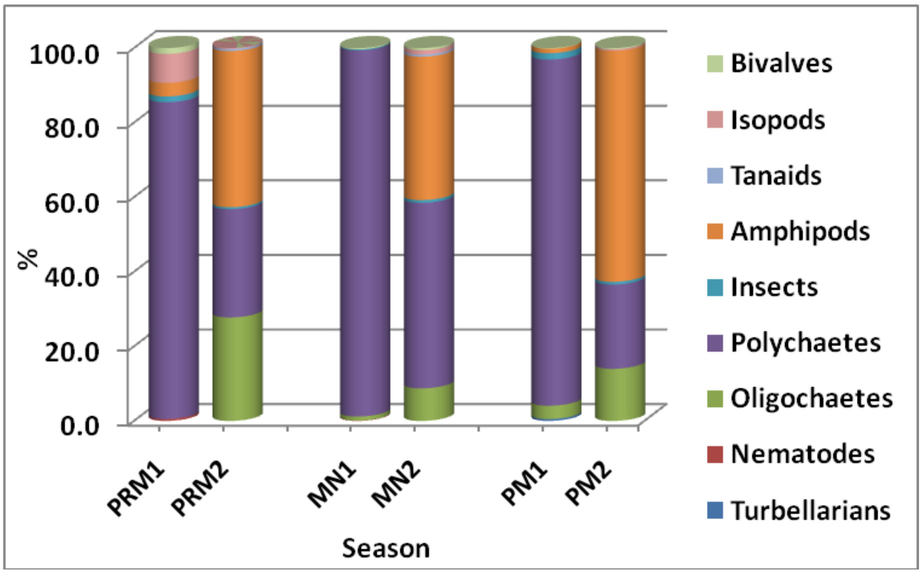


Fig. 5.29 Mean percentage abundance of macrofauna in station 2 in Vembanad estuary during 2011-2012 and 2012-2013 period

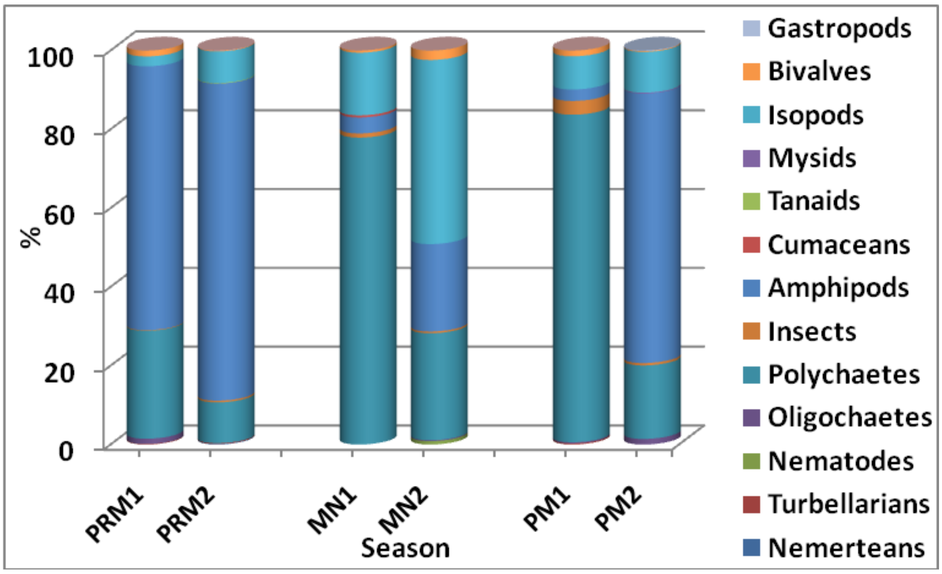


Fig. 5.30 Mean percentage abundance of macrofauna in station 3 in Vembanad estuary during 2011-2012 and 2012-2013 period

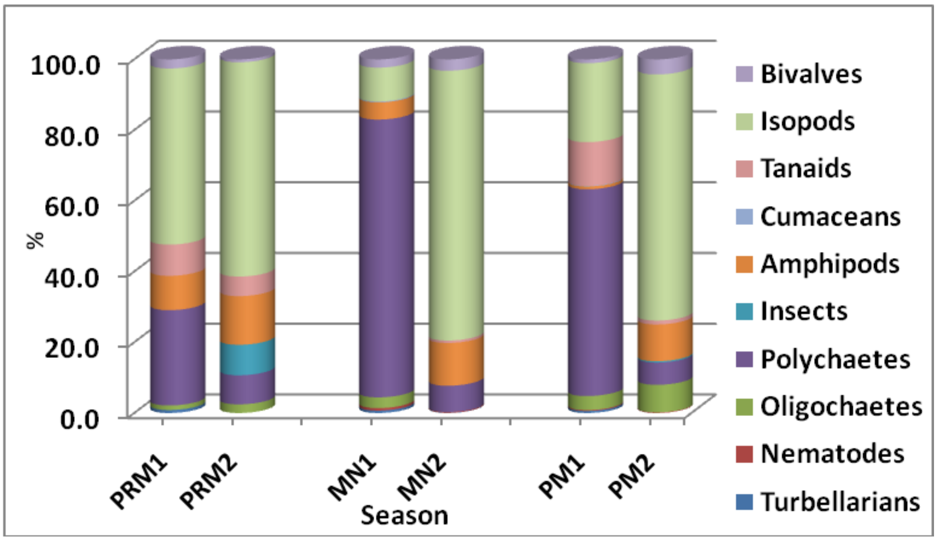


Fig. 5.31 Mean percentage abundance of macrofauna in station 4 in Vembanad estuary during 2011-2012 and 2012-2013 period

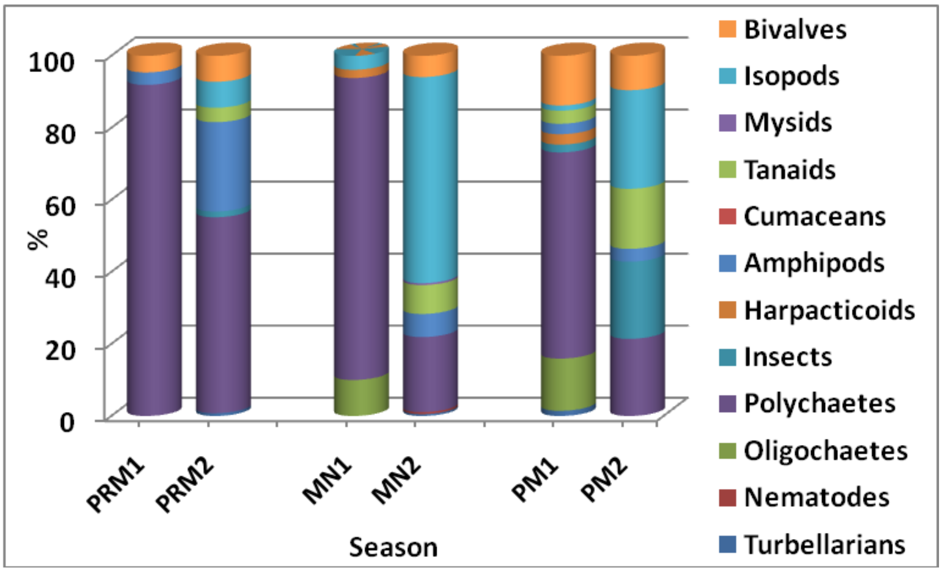


Fig. 5.32 Mean percentage abundance of macrofauna in station 5 in Vembanad estuary during 2011-2012 and 2012-2013 period

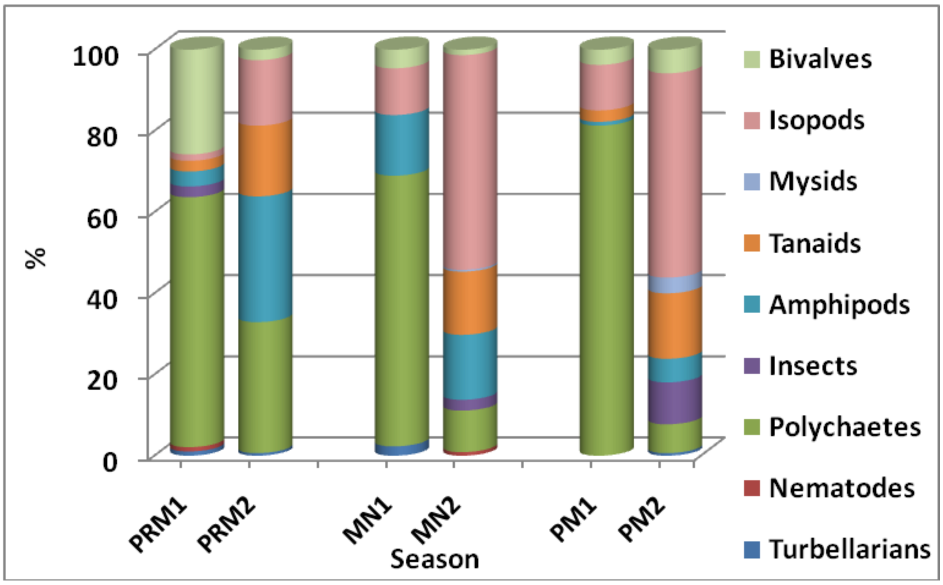


Fig. 5.33 Mean percentage abundance of macrofauna in station 6 in Vembanad estuary during 2011-2012 and 2012-2013 period

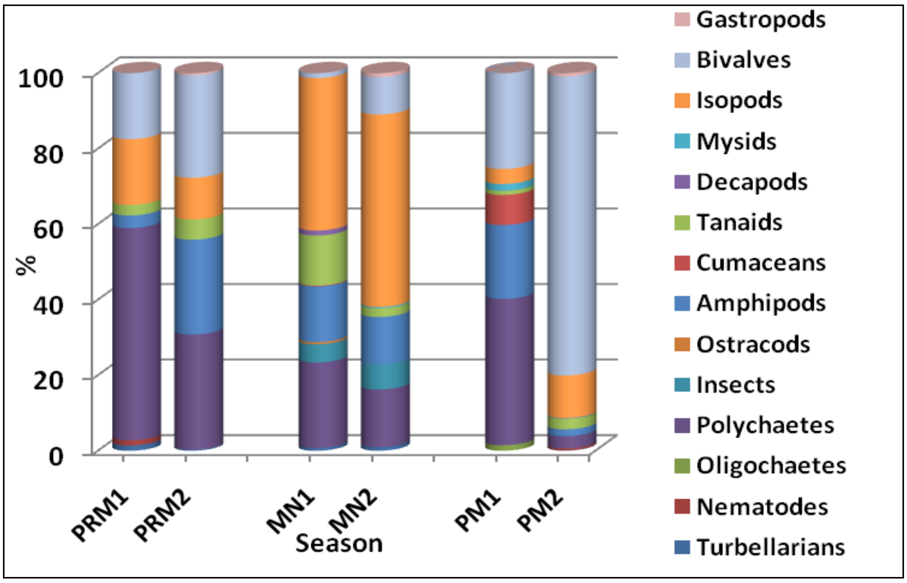


Fig. 5.34 Mean percentage abundance of macrofauna in station 7 in Vembanad estuary during 2011-2012 and 2012-2013 period

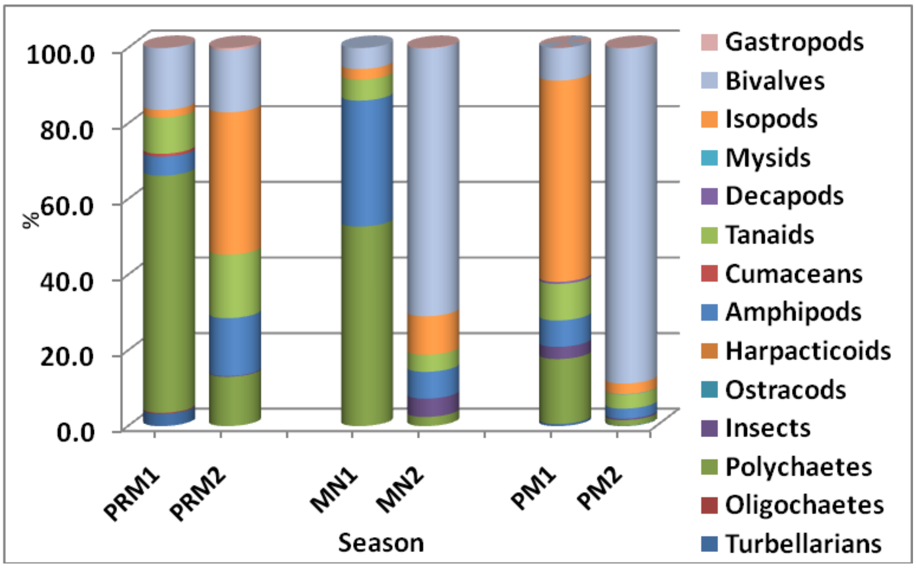


Fig. 5.35 Mean percentage abundance of macrofauna in station 8 in Vembanad estuary during 2011-2012 and 2012-2013 period

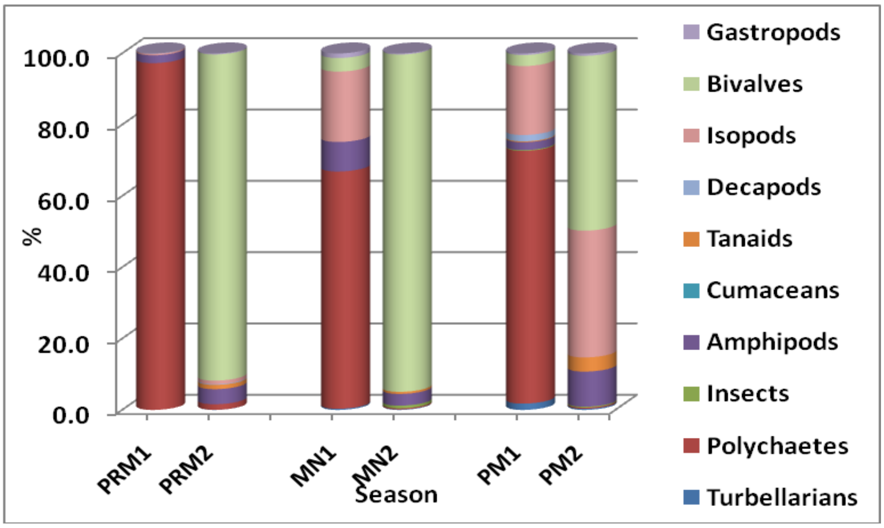


Fig. 5.36 Mean percentage abundance of macrofauna in station 9 in Vembanad estuary during 2011-2012 and 2012-2013 period

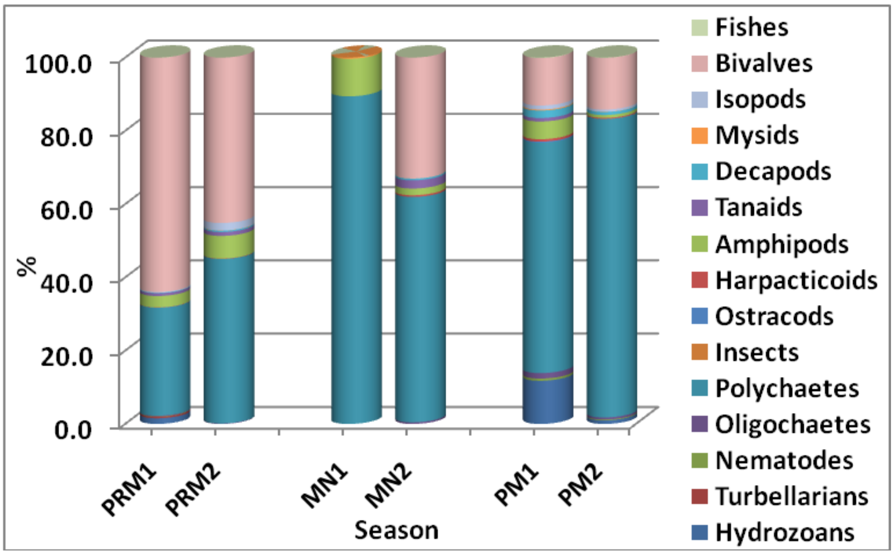


Fig. 5.37 Mean percentage abundance of macrofauna in station 10 in Vembanad estuary during 2011-2012 and 2012-2013 period

5.2.4 Biomass variations of major macrofaunal communities

5.2.4.1 Spatial variation of macrofaunal biomass

Mean spatial variation of macrofaunal biomass in the Vembanad estuary for both years is given in annexure 5.4 and annexure 5.5. During 2011-2012 period, oligochaetes showed the highest average biomass value (0.89 ± 1.78 g/m^2) when compared to polychaetes (0.7 ± 0.66 g/m^2) in station 1. The biomass estimates of oligochaetes ranged from 0 to 5.54 g/m^2 and that for polychaetes it varied from 0.08 to 2.07 g/m^2. Mean biomass of bivalves were 0.34 ± 1.19 g/m^2 and that of insects were 0.09 ± 0.14 g/m^2. Whereas during 2012-2013 period, polychaetes had high average biomass value (4.63 ± 3.98 g/m^2) followed by oligochaetes (0.58 0.64 g/m^2). Amphipods contributed an average biomass of 0.4 ±0.56 g/m^2 and the biomass estimates ranged from 0 to 1.64 g/m^2.

During 2011-2012 period the most important biomass component in station 2 was polychaetes. It showed the highest average biomass value (11.02 ±14.21 g/m^2) that ranged from 0 to 49.11 g/m^2. Relatively higher average biomass was observed for bivalves (0.99 ± 3.36 g/m^2) followed by oligochaetes (0.29 ± 0.71 g/m^2) and isopods (0.22 ± 0.67 g/m^2). Similar to first year, the major biomass component during second year was polychaetes, bivalves and oligochaetes in station 2. The biomass value of polychaetes varied from 0.18 to 135.01 g/m^2 with an annual average of 31.08 ± 39.7 g/m^2. The biomass estimates of bivalves ranged from 0 to 79.99 g/m^2 with an annual average of 17.78 ± 3.00 g/m^2 and that for oligochaetes it varied between 0 to 24.97 g/m^2 having an annual average of 2.39 ± 7.11 g/m^2.

In station 3 polychaetes contributed the major share of macrofaunal biomass (av. 2.18 1.7 g/m^2) during 2011-2012 period, followed by the bivalves (1.51 2.62 g/m^2). Compared to station 2, oligochaetes (0.74 ± 2.57) contributed much higher biomass followed by amphipods (0.26 ± 0.59 g/m^2). The biomass estimates of bivalves and amphipods varied from 0 to 8.4 g/m^2

and 0 to 2.04 g/m^2 respectively. During second year period the average biomass estimates of bivalves increased tremendously and formed the dominant contributor (55.85 ± 1.29 g/m^2) followed by amphipods (4.72 ± 9.33 g/m^2).

Polychaetes formed the major contributor of macrofaunal biomass in station 4 during 2011-2012 period. Its value ranged from 0 to 8.23 g/m^2 with an annual average of 2.33 2.59 g/m^2. Other macrofaunal groups such as amphipods, tanaids, oligochaetes and isopods showed a lower average biomass value. The amphipod biomass values varied from 0 to 3.18 g/m^2 with a mean value of 0.42 ± 0.95 g/m^2. In the case of tanaids the biomass value accounted a range of 0 to 1.23 g/m^2 with an average of 0.33 ± 0.45. g/m^2 Oligochaetes and isopods accounted a mean biomass value of 0.25 ± 0.65 g/m^2 and 0.25 ± 0.44 g/m^2 respectively. During 2012-2013 period, polychaetes formed the most important biomass component in macrobenthos (3.96± 7.28 g/m^2) followed by isopods (1.82 ± 2.49) and bivalves (1.66 ± 2.54).

During 2011-2012 period, polychaetes and bivalves had higher average biomass value in station 5. The biomass of bivalves accounted a range of 0 to 301.01 g/m^2 with an average of 27.94 ± 8.64 g/m^2. The biomass of polychaetes value ranged from 0 to 10.11 g/m^2 with an estimated mean value of 1.57 ± 2.83 g/m^2. Similar to first year, bivalves and polychaetes were the dominant macrofaunal groups contributing a major share of macrofaunal biomass. Bivalves and polychaetes were having an average biomass value of 42.92 ± 6.1 g/m^2 and 1.49 ± 1.72 g/m^2 respectively. The third important macrofaunal group was isopods, its biomass value ranged from 0 to 1.16 g/m^2 with mean value of 0.22 ± 0.39 g/m^2.

In station 6, bivalves, polychaetes and oligochaetes formed the important biomass component during 2011-2012 period. Bivalves had maximum average biomass value (3.49 ± 6.94 g/m^2), followed by polychaetes (0.72 ± 1.06 g/m^2) and oligochaetes (0.39 ± 1.37 g/m^2). The polychaete biomass value ranged from 0.01 to 3.84 g/m^2 and in the case of oligochaetes its biomass value ranged from 0 to18.16 g/m^2. Similar to first year, bivalves and polychaetes formed the major macrofaunal group contributing a major share of biomass during the second year. Bivalves and polychaetes accounted a mean biomass value of 41.96 ± 9.76 g/m^2 and 1.94 ± 3.49 g/m^2 respectively. Tanaids and isopods occupied the third and fourth place contributing an average biomass of 0.28 ± 0.39 g/m^2 and 0.26 ± 0.46 g/m^2 respectively.

During 2011-2012 period, bivalves formed the dominant biomass component in station 7. The biomass value of bivalves ranged from 0 to 376.71 g/m^2 with a mean value of 42.79 ± 1.08 g/m^2. Polychaetes and isopods were the other major contributors to the macrofaunal biomass. Bivalve biomass ranged from 0 to 376.71 g/m^2, whereas for polychaetes it was 0.05 to 6 g/m^2. Compared to first year, the average biomass of bivalves decreased to 8.96 ± 24.39 g/m^2 and polychaetes increased to 3.5 ± 5.16 g/m^2 during the second year.

During 2011-2012 period, among the macrofaunal groups the highest mean biomass was found for bivalves (46.37 ± 8.69 g/m^2) followed by polychaetes (0.64 ± 1.1 g/m^2) and tanaids (0.23± 0.63 g/m^2) in station 8. During the second year, polychaetes and bivalves formed the dominant biomass component. But its mean weight significantly decreased during 2012-2013 period. Bivalves and polychaetes were having an average biomass value of 1.12 ± 1.6 g/m^2 and 1.28 ± 1.33 g/m^2 respectively.

In station 9, the mean standing crop of bivalves was 3.27 ± 8.19 g/m^2 and that for polychaetes was 2.47 ± 2.66 g/m^2. Gastropods formed the major contributor of biomass in station 9. The biomass value of gastropods accounted a range of 0 to 13.54 g/m^2 with an average of 1.41 ± 3.89 g/m^2. Similarly decapods also contributed a major share of biomass here. The biomass estimated a range of 0 to 1.97 g/m^2 with an average of 0.32 ± 0.74 g/m^2.

During second year, polychaetes and gastropods formed the major contributor of macrofaunal biomass. The mean biomass value of polychaetes was 6.0 ± 8.07 g/m^2 and that of gastropods was 3.91 ± 5.88 g/m^2. Bivalves were the third important contributor to the macrofaunal biomass and the mean biomass estimates ranged from 0 to 352.08 g/m^2.

Polychaetes and bivalves contributed the major share of macrofaunal biomass in station 10. The biomass of polychaetes accounted a range of 0.95 to 86.67 g/m^2 with an average of 15.63 ± 24.91 g/m^2. Bivalve biomass value ranged from 0 to 133.27 g/m^2 with an estimated mean value of 14.89 ± 3.83 g/m^2. Decapods and amphipods also contributed the major share of macrofaunal biomass. The average biomass of decapods and amphipods were 0.65 ± 1.33 g/m^2 and 0.47 ± 0.76 g/m^2 respectively. Similar to the first year, during second year bivalves and polychaetes contributed the major share of macrofaunal biomass. Compared to other stations, its mean biomass was comparatively higher in station 10. The mean biomass of polychaete was 23.5 ± 32.1 g/m^2 and that for bivalves was 56.83 ± 8.05 g/m^2.

5.2.4.2 Monthly variation of macrofaunal biomass

During 2011 - 2012 period, monthly variation of polychaete biomass showed the maximum average value during October 2011 (10.96 ± 26.72 g/m^2) and minimum during March 2011 (0.58 ± 0.6 g/m^2). In the case of bivalves, minimum value was observed during May 2011 (0.001± 0.003 g/m^2) and maximum during February 2012 (75.64 ± 1.32 g/m^2) (Fig. 5.38). The oligochaete biomass contribution was negligible during June 2011 and maximum during March 2011 (1.92 ± 3.25 g/m^2). In the case of amphipods, higher value of biomass was observed during April 2011 (0.40 ± 0.71 g/m^2) and lower (0.019 g/m^2) during March 2011. There was no detectable level of biomass contribution by insects during May, June, July, November 2011and February 2012, whereas higher biomass of insects were observed during April 2011 (0.045 ± 0.143 g/m^2). During June and August 2011, the biomass estimates of tanaids were insignificant but during October 2011 it contributed a maximum of 0.26 ± 0.69 g/m^2. Maximum biomass contribution of isopods was observed during March 2011 (0.78 ± 1.64 g/m^2) and minimum during August 2011 (0.0003 ± 0.0007 g/m^2). Decapods and gastropods showed higher average biomass contribution during October 2011 (0.633 ± 1.45 g/m^2) and June 2011 (1.35 ± 4.28 g/m^2) respectively.

During 2012-2013 period, polychaetes showed the maximum average macrofaunal abundance during August 2012 (16.93 ± 41.57 g/m^2) followed by January 2013 (13.69 ± 28.86 g/m^2). However the lowest polychaete biomass was observed during February 2013 (0.97 ± 0.85 g/m^2). The amphipod biomass was highest during May 2012 (3.66 ± 9.34 g/m^2) and lowest during July 2012 (0.16 ± 0.27g/m^2). Among the macrofaunal groups the highest mean biomass was contributed by bivalves. The estimated mean biomass value of bivalves was highest during September 2012 (157.85 ± 243.14 g/m^2) and July 2012 (132.47 ± 225.68 g/m^2) (Fig. 5.39), whereas, the lowest mean value was observed during November 2012 (17.13 ± 37.69 g/m^2). In the case of oligochaetes biomass was not significant observing nil values in February 2012 and having maximum contribution observed during October 2012 (2.49 ± 7.89 g/m^2). The estimated biomass contribution of tanaids and decapods was highest during June 2012 (0.269 ± 0.44 g/m^2) and January 2013 (0.107 ± 0.34 g/m^2) respectively. Higher biomass values of isopods were observed during December 2012 (1.87 ± 3.9 g/m^2) and July 2012 (1.14 ± 2.6 g/m^2). The lowest mean value of isopod biomass was observed during April 2012 (0.009 ± 0.02 g/m^2). The biomass contribution of gastropods and fishes were negligible during many months. The highest biomass value of gastropods (10.35 ± 26.84 g/m^2) and fishes (0.134 ± 0.422 g/m^2) were observed during October 2012 and January 2013 respectively.

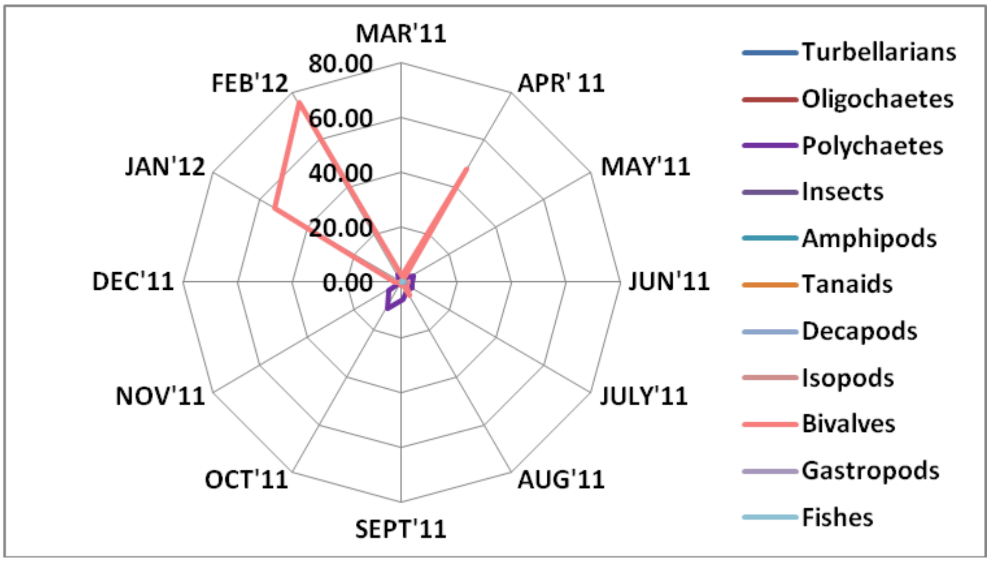


Fig. 5.38 Radar chart showing the monthly variation of macrofaunal biomass (g/m^2) in Vembanad estuary during 2011 – 2012 period

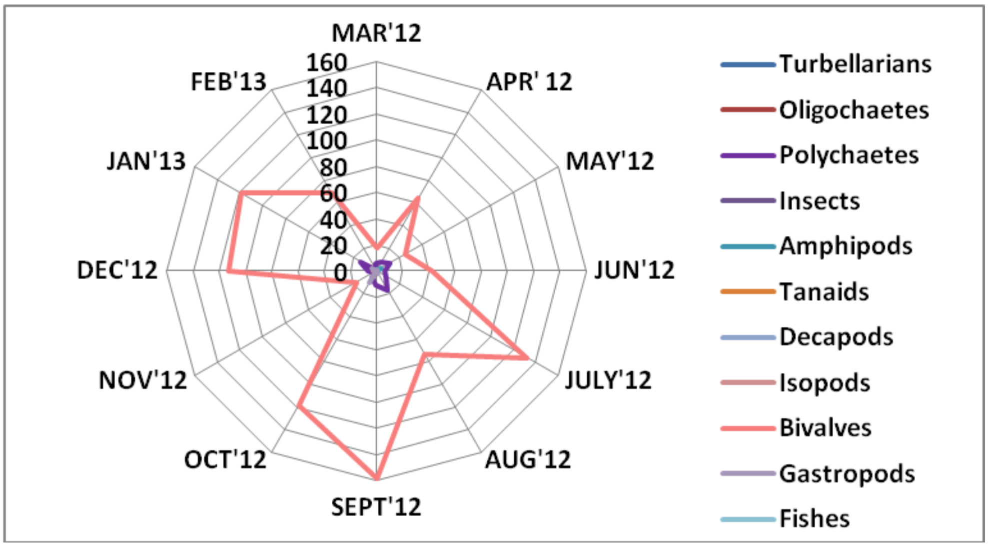


Fig. 5.39 Radar chart showing the monthly variation of macrofaunal biomass (g/m^2) in Vembanad estuary during 2012 – 2013 period

5.2.4.3 Seasonal variation of macrofaunal biomass

When comparing the seasonal variation of biomass of macrofaunal groups in the whole Vembanad estuarine system, the biomass value of fauna varied considerably. During 2011-2012 period, bivalves formed the dominant group in PRM and PM and contributed about 88.51% and 71.68% to the total biomass (Fig. 5.40). During PRM the average biomass of bivalve was 31.36 ± 7.94 g/m^2 followed by polychaetes (2.58 ± 5.79 g/m^2) contributing 7.28% to the total biomass. During MN, polychaetes become the major contributor of macrofaunal biomass, contributing 58.29% to the total macrofaunal biomass. Bivalves become the second major contributor of (33.13%) macrofaunal biomass. During second year period seasonal variation of biomass of major macrofaunal groups were showing a similar pattern as that for the first year. But its percentage contribution varied seasonally. The average biomass of bivalves during PRM season was 43.74 ± 8.85 g/m^2 and contributing 82.92% to the total fauna. The next important group such as polychaetes and amphipods had a mean biomass of 6.79 ± 14.73 g/m^2 and 1.26 ± 4.75 g/m^2 and contributing 12.87% and 2.4 % respectively. During MN polychaetes contributed 78.52% of total macrofaunal biomass. During PM bivalves showed the maximum biomass (87.66%) followed by polychaetes (6.5%).

Overall, during 2011-2012 period, the highest spatial annual average macrofaunal biomass was recorded at station 8 (28.39 g/m^2) followed by station 7 (27.05 g/m^2) and station 10 (19.25 g/m^2) and the lowest value was recorded at station 1 (1.23 g/m^2) and 6 (2.82 g/m^2). Among the macrofaunal groups the highest mean biomass was found in bivalves (16.06 ± 5.71 g/m^2) followed by polychaetes (3.85 ± 10.14 g/m^2) (Fig. 5.41). Bivalves dominated the macrofaunal structure by weight, accounting for 74.53% of the total macrofaunal biomass followed by polychaetes (17.88%). The bivalve biomass value varied from a negligible level to a maximum of 376.71 g/m^2. Similarly polychaete biomass ranged from an insignificant level to 86.67 g/m^2. ANOVA results of macrofaunal biomass are given in annexure 5.6. ANOVA analysis showed that there was a significant spatial variation of polychaetes biomass at 1% level ($P \leq 0.01$). Oligochaetes contributed an average biomass of 0.27 ± 1.12 g/m^2 and its value varied from 0 to 8.9 g/m^2. Amphipods showed an annual average biomass of 0.14 ± 0.45 g/m^2 and its value ranged from a minimum of 0 to a maximum of 3.18 g/m^2. Other macrofaunal groups such as gastropods and isopods was having an annual average biomass of 0.14 ± 1.26 g/m^2 and 0.12 ± 0.52 g/m^2 respectively. To the total macrofaunal biomass, turbellarians and oligochaetes accounted for 3.17% and 1.24% respectively.

Isopods, gastropods and amphipods contributed 0.56%, 0.66% and 0.67% to the total macrofaunal biomass respectively. ANOVA analysis showed that the spatial variation of tanaids biomass significantly varied between stations $P \leq 0.05$). ANOVA analysis detected a significant difference in decapods biomass among stations at 5% level ($P \leq 0.05$) and seasons ($P \leq 0.05$) and interaction between seasons and stations were also significant at 5% levlel ($P \leq 0.05$). Similar to polychaetes, spatial variation of insects ($P \leq 0.01$) and fish ($P \leq 0.05$) biomass were also significant. During 2011 - 2012 period, the overall monthly distribution of macrofaunal biomass showed the maximum average biomass during February 2012 (5.28 ± 9.11 g/m^2) followed by January 2012 (3.72 ± 7.27 g/m^2) and April 2011 (3.39 ± 5.27 g/m^2). The minimum biomass value was observed during December 2011 (0.24 ± 0.6 g/m^2). Considering the overall seasonal variation of macrofaunal biomass, it was maximum during PRM in 2011-2012 period. Compared to other seasons, a considerable decrease in the biomass value of macrofauna was observed during MN (11.56%) and maximum biomass was observed during PRM (56.42%) followed by PM (32.02%).

During 2012-2013 period, the highest spatial annual average macrofaunal biomass was recorded at station 4 (103.54 g/m^2) followed by station 9 (98.92 g/m^2) and station 7 (93.99 g/m^2) and the lowest value was recorded at station 1 (3.48 g/m^2). Among the macrofaunal groups, the biomass of bivalves dominated by weight (79.07 ± 1.51 g/m^2) (Fig. 5.41), accounting for 87.37% to the total macrofaunal biomass. Polychaetes (8.14 ± 18.88 g/m^2) and gastropods (1.48 ± 8.32 g/m^2) formed the next dominant biomass components and contributed 8.99% and 1.63% respectively. The ANOVA of polychaetes showed that there was a significant variation between stations ($P \leq 0.01$) and interaction between season and station were also significant ($P \leq 0.01$). To the total macrofaunal biomass, amphipods and isopods accounted for 0.9% and 0.55% respectively. The ANOVA of amphipods and bivalves biomass showed a significant spatial variation ($P \leq 0.05$). The ANOVA analysis of decapod biomass showed that the spatial variation ($P \leq 0.01$) and interaction between seasons and stations were significant ($P \leq 0.01$). ANOVA of fish biomass showed that the interaction between seasons and stations were significant at 1% level ($P \leq 0.01$). On an overall scale during 2012 - 2013 period, the maximum average biomass of macrofauna was observed during September 2012 (16.98 ± 26.83 g/m^2) followed by July 2012 (14.29 ± 24.31 g/m^2) and January 2013 (13.91 ± 19.51 g/m^2). The minimum macrofaunal biomass contribution was observed during November 2012 (2.06 ± 4.54 g/m^2). During 2012-2013 period, the maximum biomass was observed during PM (61.18 %), followed by MN (30.97%) and minimum in PRM (30.79%).

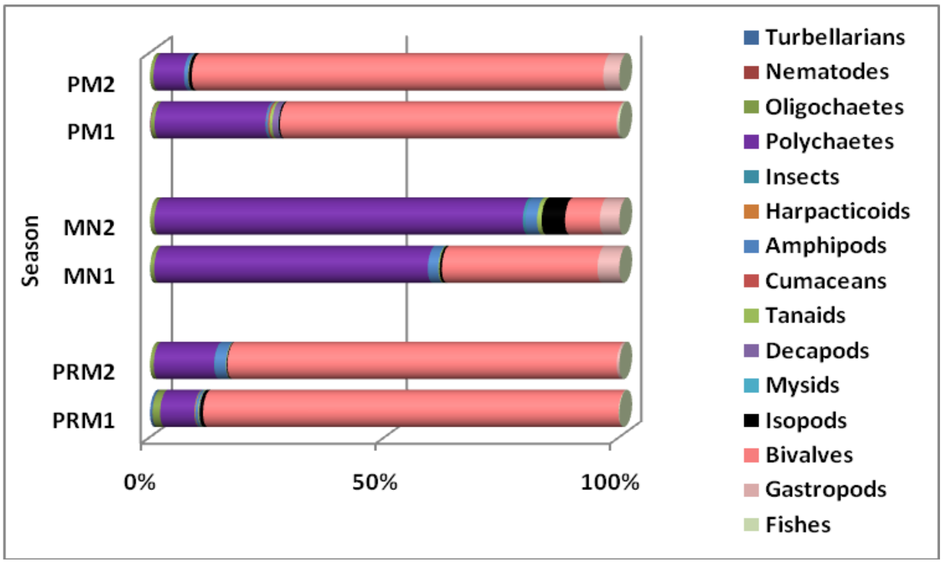


Fig. 5.40 Seasonal variation of percentage contribution of macrofaunal biomass in Vembanad estuary during 2011-2013 period.

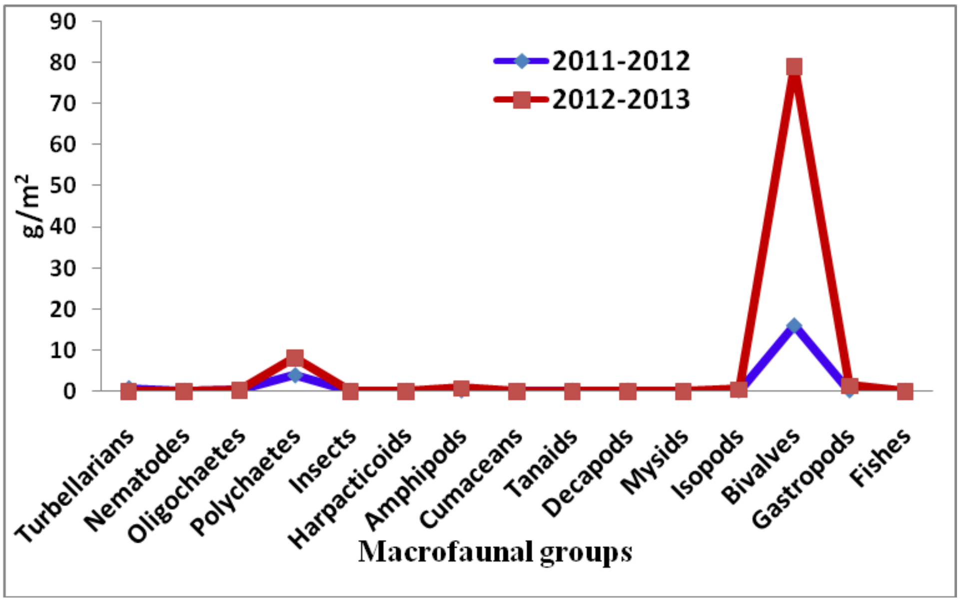


Fig. 5.41 Annual variation of macrofaunal biomass in the Vembanad estuary during 2011-2013 period.

5.2.5 Data Analysis

5.2.5.1 Variation in diversity patternof macrobenthos

Using the abundance data, the diversity of benthic macro-invertebrates in the study area was estimated both spatially and temporally by the Shannon index (H'), Margalef index (d), Pielou index (J') and Simpson index (λ'). During 2011-2012 period, the Shannon diversity values for the entire study area ranged from 0.39 (station 2) to 2.49 (station 7) with an annual average value of 1.61 (Fig. 5.42). Margalef's richness index showed that the maximum value was observed in station 7 (1.56) and minimum in station 2 (0.85). Pielou's evenness index, varied between 0.13 (station 2) to 0.67 (station 7) with an annual average value of 0.49. Simpson dominance indices were found to be highest in station 2 (0.91) and lowest in station 7 (0.23) with an average of 0.47. Temporal variation of diversity indices reveled that richness showed the highest values in October 2011 (1.87) and lowest in June 2011 (0.68) with an average of 1.27 (Table 5.5). The diversity and evenness index followed similar temporal patterns with maximum value was observed in December 2011 and minimum in June 2011. In contrast with diversity and evenness, the dominance index was highest during June 2011 (0.71) and lowest during December 2011 (0.22). Seasonally the highest Shannon diversity index was observed during PRM (2.02) followed by PM (1.95) and MN (1.9). Post-monsoon season was having higher Margalef index (3.69) followed by PRM (3.25) and MN (2.82). During PRM and MN Pielou's evenness index was 0.5 and during PM it was 0.47. Simpson dominance index was found to be highest during PM (0.44) followed by MN (0.4) and PRM (0.32).

During 2012-2013 period, the spatial variation of Shannon diversity was more or less similar from station 1 to 4. Station 5 and 6 showed highest diversity value (2.29) and it decreased from station 7 to station 10 and the lowest diversity value was observed in station 9 (0.92) with an annual average of 1.66 (Fig. 5.43). Maraglef's richness index varied between a minimum value of 0.68 to a maximum value of 1.4. Pielou's evenness index, fluctuated from 0.29 (station 9) to 0.72 (station 6) with an annual average of 0.51. Low dominance was observed in station 5 (0.25) and high dominance in station 5 (0.25). Temporal variation of macrofaunal diversity indices varied considerably during 2012-2013 period (Table 5.5).

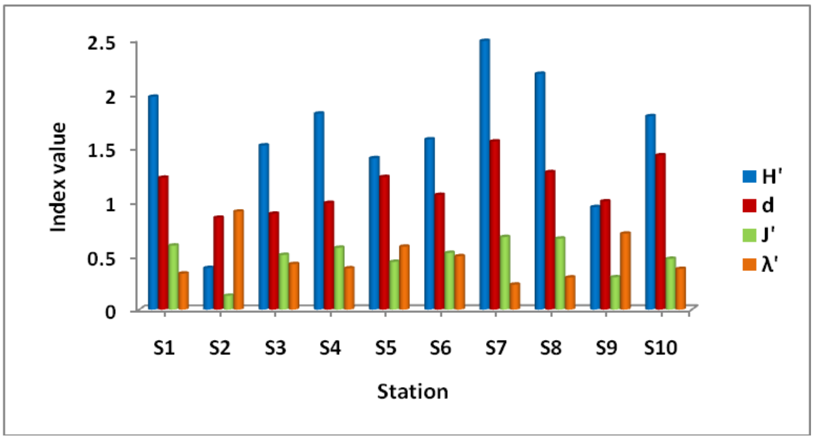


Fig. 5.42 Spatial variation of macrofaunal diversity indices in Vembanad estuary during 2011 – 2012 period

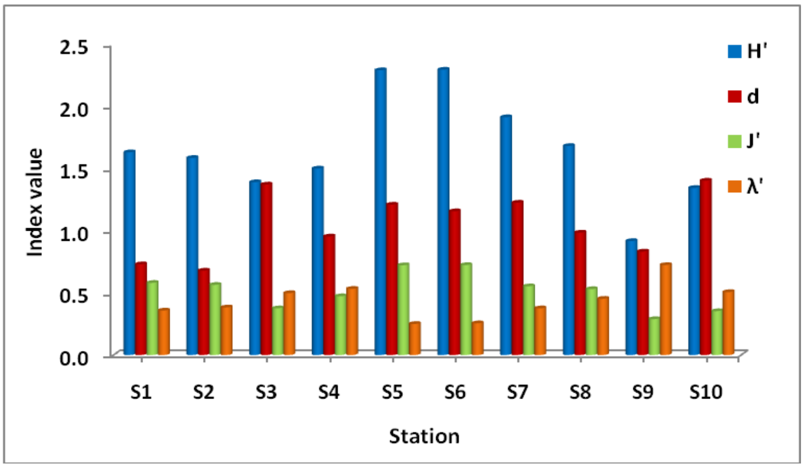


Fig. 5.43 Spatial variation of macrofaunal diversity indices in Vembanad estuary during 2012 – 2013 period. (H' - Shannon index; d- Margalef richness; J'- Pielou's evenness; λ'- Simpson index)

Table 5.5 Temporal variation of diversity indices of macrofauna in Vembanad estuary during 2011-2013 period

	H'		d		J'		λ'	
Month	2011-2012	2012 - 2013	2011-2012	2012 - 2013	2011-2012	2012 - 2013	2011-2012	2012 - 2013
MAR	2.29	1.12	1.30	0.88	0.66	0.35	0.27	0.65
APR	1.80	2.13	1.13	1.18	0.54	0.64	0.38	0.28
MAY	1.28	2.09	0.69	1.16	0.50	0.60	0.58	0.34
JUN	0.83	2.06	0.68	0.90	0.32	0.65	0.71	0.29
JUL	1.14	1.95	1.14	1.37	0.36	0.53	0.63	0.33
AUG	1.08	2.44	1.07	1.09	0.34	0.73	0.66	0.20
SEP	1.53	2.22	1.22	1.09	0.44	0.67	0.52	0.27
OCT	1.48	1.88	1.87	1.15	0.36	0.54	0.58	0.39
NOV	1.65	2.10	1.23	1.60	0.48	0.55	0.49	0.31
DEC	2.62	2.11	1.82	1.44	0.69	0.55	0.22	0.30
JAN	2.15	2.00	1.74	1.47	0.56	0.53	0.32	0.29
FEB	1.58	1.84	1.35	1.07	0.43	0.55	0.41	0.37
Average	1.62	1.99	1.27	1.20	0.47	0.58	0.48	0.34

The minimum diversity value was noted in March 2012 (1.12) which considerably increased in successive months but fluctuating in July 2012 (1.95), October 2012 (1.88) and February 2013 (1.84). The maximum diversity value was noted in August 2012 with a highest value of 2.44. Margalef's richness index showed maximum value in samples of November 2012 (1.59) and minimum in March 2012 (0.88). The range of variations of Simpson's dominance index ranged from 0.2 to 0.65, in which the higher values were observed in March 2012 and lower in August 2012. The minimum evenness value was observed in March 2012 and maximum in August 2012, the values being 0.35 and 0.73 respectively. In contrast with first year, the highest Shannon diversity index was observed in PM (2.28) followed by MN (2.25) and PRM (2.13) during second year seasonally. During second year, seasonal variation of Margalef richness index was following similar pattern as that of first year. The highest richness value was observed during PM (3.26) followed by PRM (2.82) and MN (2.61). Pielou's evenness index was highest in MN (0.61) followed by PM (0.57) and PRM (0.56). Simpson dominance index was found to be highest during PRM (0.28) followed by MN (0.25) and PM (0.24).

5.2.5.2 K dominance plot

Multiple k-dominance plots facilitate the discrimination of macrobenthos based on its relative contribution to abundance. During first year, the k-dominance plot of macrofaun showed the highest dominance of faun in station 2 followed by station 9. The maximum diversity was observed in station 7, so the curve for this station was lying lower than the others whereas curve of station 2 was lying in the top, representing lowest diversity (Fig. 5.44). During 2012-2013 period, k- dominance plot of macrofauna according to stations shows a 'S' shaped curve indicating the high diversity of macrofauna in station 6 without disturbance; station 5 and station 6 showed maximum diversity as the curves for these stations was lying lower to the other plots, whereas the plots for station 9 was lying in the top representing lower diversity (Fig. 5.45).

5.2.5.3 ABC plot

ABC curves were used to investigate macrofaunal community stress in Vembanad estuarine system (Fig. 5.46 and Fig. 5.47). During first year in station 1, ABC curve showed that initially abundance curve intersected biomass curve and lied below the abundance curve, representing a slightly disturbed site and also low W value of the area indicating the environmental disturbances. Stations 2 and 3 indicated a disturbed condition. ABC curve designated an unstressed condition in stations 4 to 8. The negative W – value of 0.024, 0.069 and 0.147 characterized a disturbed macrofaunal community in station 2, station 3 and station 9 respectively. The low positive W value in station 10 represented slightly disturbed condition.

During second year ABC plot of macrofaunal community showed that in most of the stations biomass curve lied above the abundance curve indicating normal condition of the system. In stations 2, 3, 9 and 10 biomass and abundance curves closely coincided and crossed each other indicating moderate disturbances in the respective area.

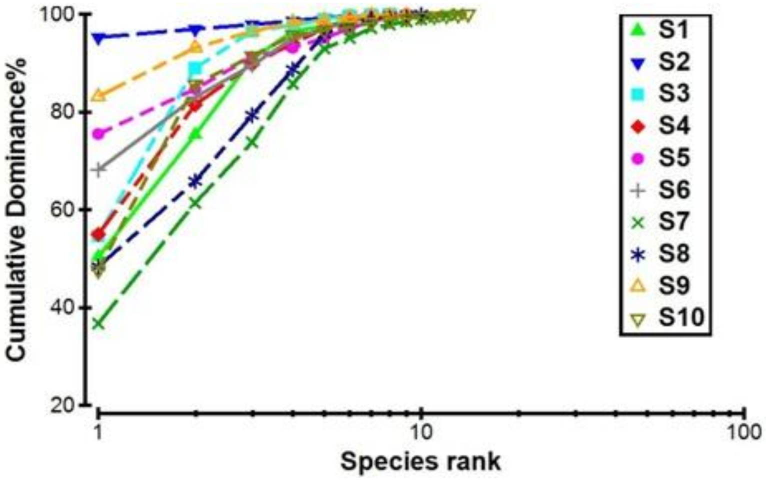


Fig. 5.44 K-dominance plot of macrofaunal groups in Vembanad estuary during 2011-2012 period

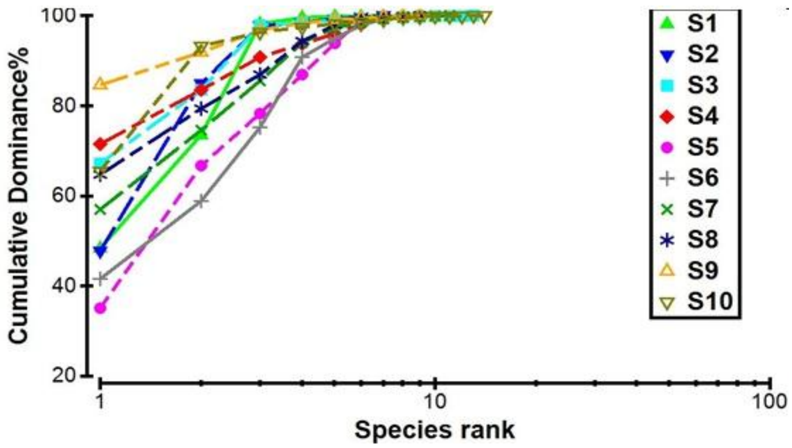


Fig. 5.45 K-dominance plot of macrofaunal groups in Vembanad estuary during 2012-2013 period

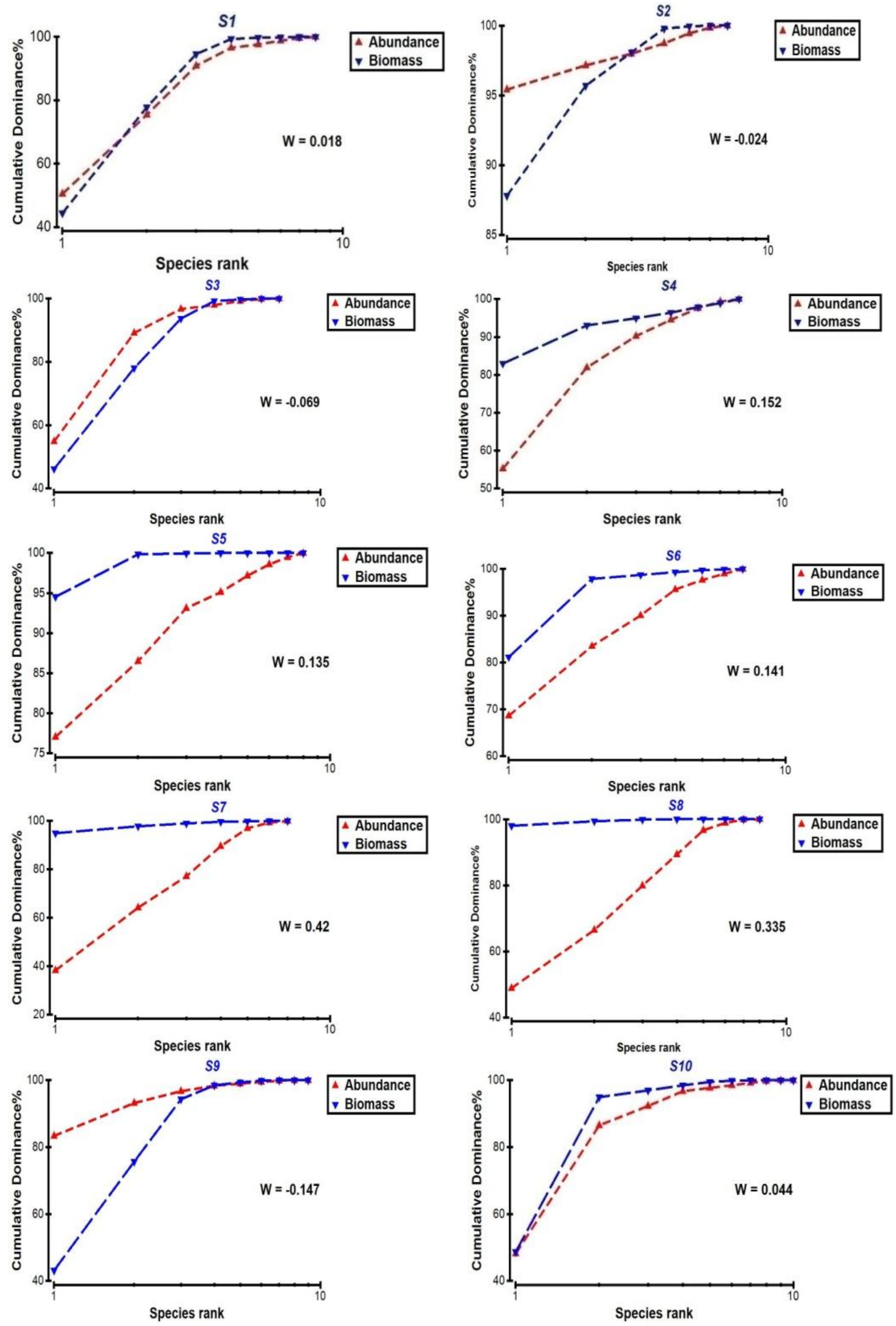


Fig. 5.46 ABC curve of macrofaunal groups in different stations of Vembanad estuary during 2011-2012 period

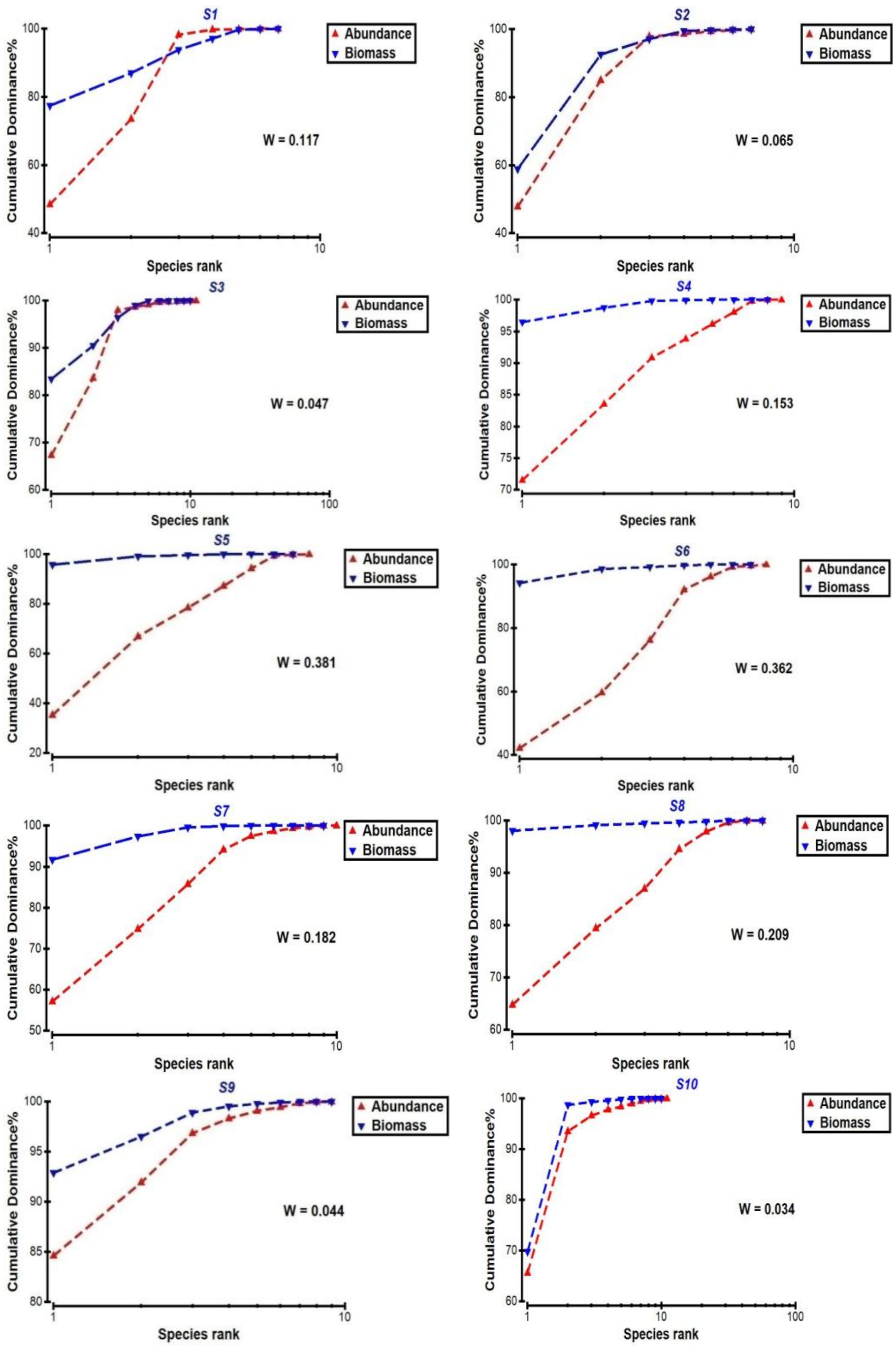


Fig. 5.47 ABC curve of macrofaunal groups in different stations of Vembanad estuary during 2012-2013 period

5.2.5.4 BIOENV analysis of macrofaunal community

BIO-ENV procedure was used for characterizing the relationship of more than one ecological factor on the abundance of macrofauna. A total of ten parameters including water and sediment were employed for the analysis (Table 5.6). During the first year a combination of bottom water pH, salinity, sediment potassium content (ρ=0.47) were found to be the major environmental factors influencing the distribution of macrofauna (Fig. 5.48). During second year the interaction of salinity, organic carbon, available nitrogen and sand (ρ= 0.89) were the major influencing factors (Table 5.7 and Fig. 5.49). Silt and clay fraction were the other important factors that determined the community structure of macrobenthos.

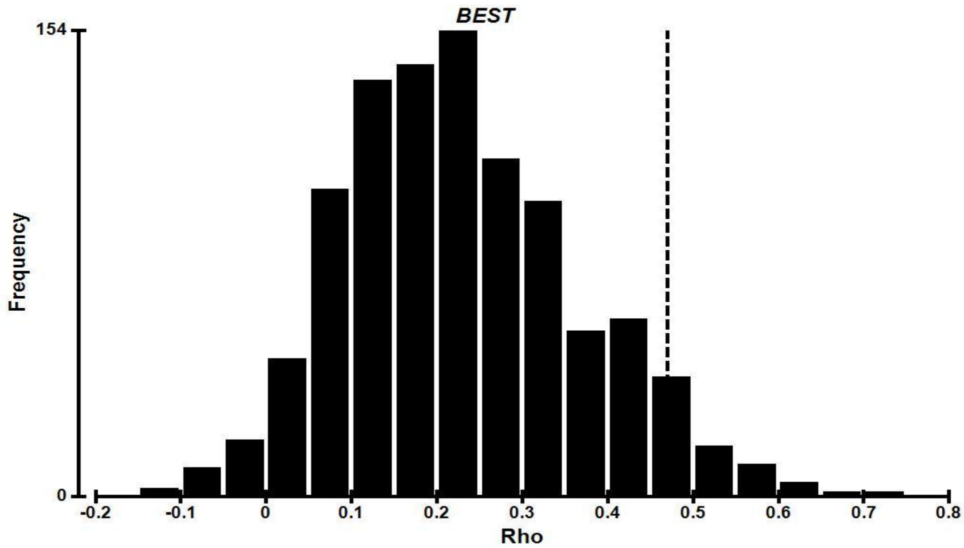


Fig. 5.48 Histogram showing the results of BEST analysis of macrofauna with environmental parameters in Vembanad estuary during 2011-2012 period.

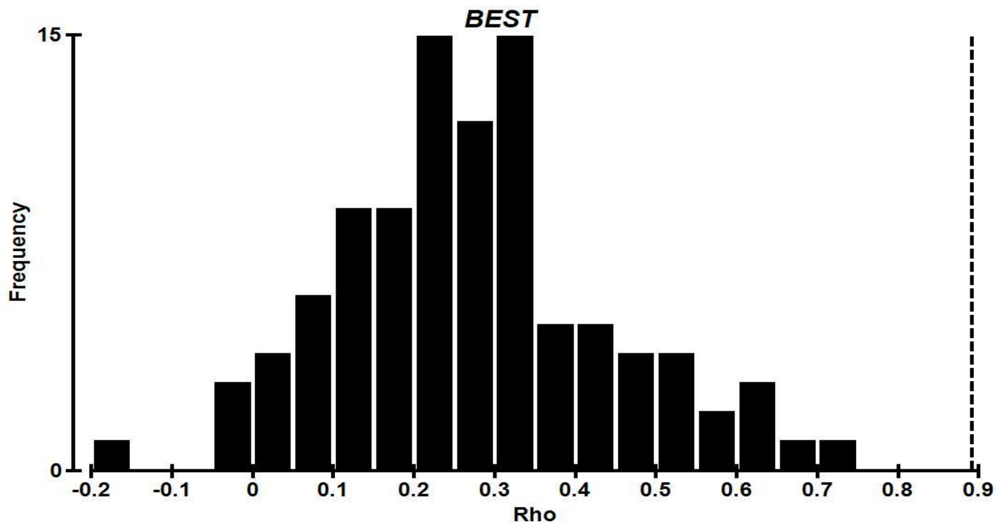


Fig. 5.49 Histogram showing the results of BEST analysis of macrofauna with environmental parameters in Vembanad estuary during 2012-2013 period

Table 5.6 Results of BIOENV analysis for macrofauna and environmental parameters in Vembanad estuary during 2011 – 2012period.

Sl. No.	Variables	Variables Selected	Correlation Values (Rho)
1	Water pH	2,3,7	0.47
2	Salinity	2,3,6,7	0.468
3	Sediment pH	2,3,6	0.463
4	Organic Carbon	2,3,5,7	0.457
5	Available Nitrogen	2,3,5-7	0.454
6	Sodium	1-3,7	0.452
7	Potassium	2,7,8	0.45
8	Sand	2,7	0.45
9	Silt	2,3,7,10	0.45
10	Clay	1-3,6,7	0.449

Table 5.7 Results of BIOENV analysis for macrofauna and environmental parameters in Vembanad estuary during 2012 - 2013period.

Sl. No.	Variables	Variables Selected	Correlation Values (Rho)
1	Water temperature	2-4,8	0.891
	Salinity	2,4,8	0.887
3	Organic Carbon	2-4,9,10	0.881
4	Available Nitrogen	2-4,10	0.881
5	Sediment pH	2-4,8,9	0.874
6	Sodium	1-4,8	0.872
7	Potassium	2-4	0.872
8	Sand	2-4,9	0.865
9	Silt	2-5,8	0.863
10	Clay	1-4,10	0.863

5.2.5.5 CCA analysisof macrofaunal community

To support the BIOENV analysis, Canonical Correlation Analysis (CCA) was intended to find the relationship between spatial changes in benthic community composition and environmental variables in the Vembanad estuary. The vector length of a given variable showed the significant correlation with stations. During first year a total of 10 dominant macrofaunal groups were analysed with 11 environmetal parameters (Fig. 5.50). Eigen – value of axis 1 (0.223), 2 (0.157) and 3 (0.118) explained 84.59% of the relation between benthic macrofauna and environmental parameters. Salinity (r = 0.56), bottom water pH (r = 0.68), clay (r = 0.52) and silt (r = 0.49) showed positive correlation with axis 1, indicating the influence of sea water intrusion.Whereas sand fraction showed a negative correlation with axis 1 (r = -0.54). Comparatively large vectors representing salinity and bottom water pH with axis 1, specified their role in shaping macrofaunal assemblage structure across the site. During second year a total of 11 dominant macrofaunal groups were analysed with 10 environmental parameters including both bottom water and sediment parameters (Fig. 5.51). Eigen value of axis 1 (0.579), 2

(0.359) and 3 (0.166) explained a total of 92.35 % variation. Silt fraction (r = 0.91), organic carbon (r = 0.73), available nitrogen (r = 0.69), and bottom water pH (r = 0.52) showed strong positive correlation with axis 1. Whereas sand (r = -0.85), salinity (r = -0.63) and sediment pH (r = -0.62), and sediment temperature (r = -0.59), showed negative correlation to axis 1.

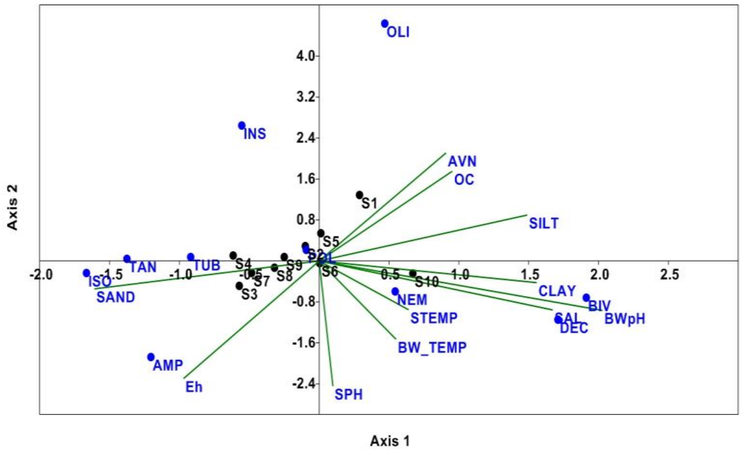


Fig. 5.50 CCA analysis of macrofauna with environmental parameters in Vembanad estuary during 2011 – 2012 period

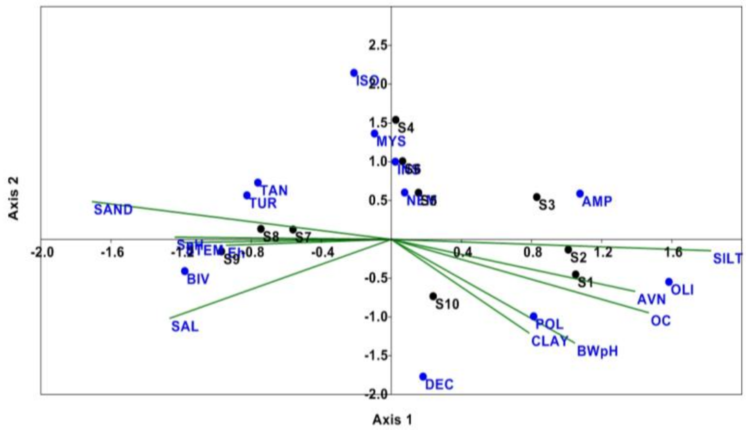


Fig. 5.51 CCA analysis of macrofauna with environmental parameters in Vembanad estuary during 2012 – 2013 period. (CCA Code- TUR- turbellarians, NEM- nematodes, OLI-oligochaetes, POL- polychaetes, INS- insects, AMP- amphipods, TAN- tanaids, DEC- decapods, MYS- mysids, ISO- isopods, BIV- bivalves)

5.2.5.6 Comparison of macrofaunal communities on the south and north of TMB

Percentage similarity (Bray–Curtis with SIMPROF test) of cluster analyses on the macrofaunal abundance and distribution showed variation between the sampling stations. From the results on dendrogram, two clusters were observed (Fig. 5.52). Apart from station 10, the other 9 stations were grouped into two clusters and station 10 stood separately. First cluster matched to the southern stations of TMB except station 4 and northern stations grouped in second cluster. During second year, the resulting dendrogram showed two clusters, from which station 3, 4 and 10 stood alone (Fig. 5.53). It indicated cluster one of stations 1 and 2 showing similarity of 85.11 % and cluster two incorporated in all the other stations showing similarity of 73.69 %. The macrofaunal abundance in stations located on the either side of TMB (station 5 and 6) showed a homogenous situation with 92.49% similarity. ANOSIM analysis was used to test the occurrence of statistical differences in the spatial variation of macrofaunal abundance. Differences in the benthic community structure on the south and north of TMB were tested using ANOSIM. Significant differences were detected between southern and northern sectors during the first year ($P \leq 0.05$) and second year ($P \leq 0.05$). SIMPER analysis was conducted to determine which macrofaunal groups were responsible for differences between northern and southern sectors. The results showed that the average dissimilarity of macrofauna in the southern and northern sectors of Vembanad estuary was 33.55 % during first year (Table 5.8) and 35.83 % during second year (Table 5.9). During first year, decapods and oligochaetes contributed 10.12 % and 9.02% dissimilarity respectively. At the same time, during second year bivalves showed the highest dissimilarity (23.68 %) followed by oligochaetes (11.15%).

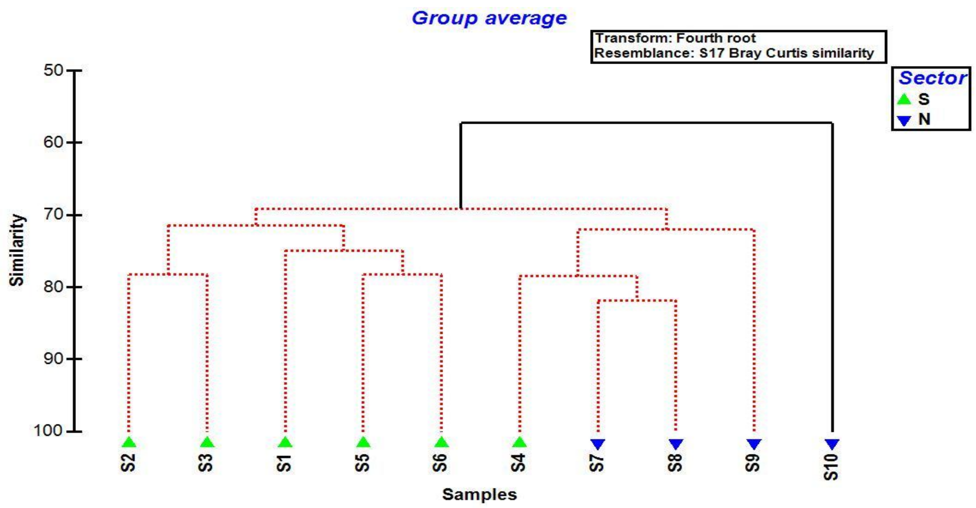


Fig. 5.52 Dendrogram showing spatial similarities of macrofaunal groups in the south and north zone of Vembanad estuary during 2011 – 2012 period

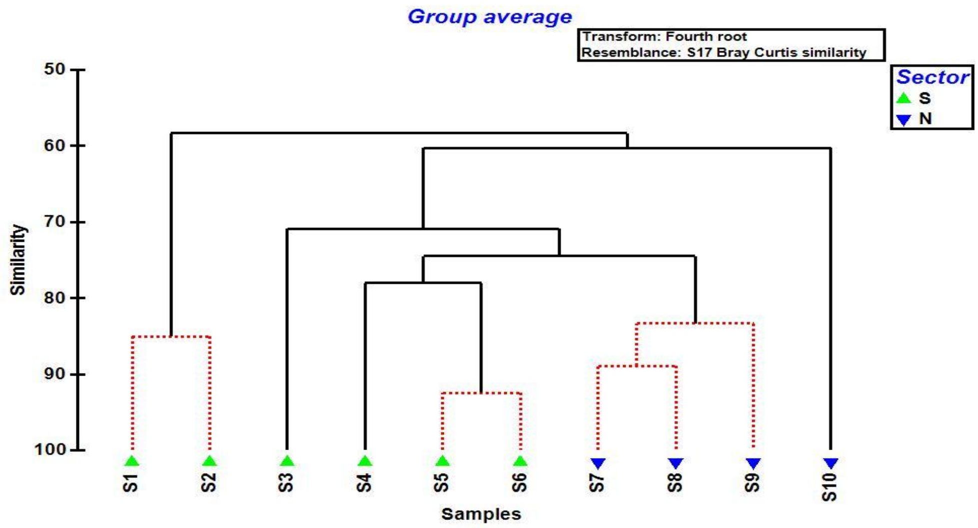


Fig. 5.53 Dendrogram showing spatial similarities of macrofaunal groups in the south and north zone of Vembanad estuary during 2012 – 2013 period

Table 5.8 SIMPER test results showing the dissimilarity of macrofaunal communities between southern and northern sector of TMB during 2011-2012 period

2011-2012	Average dissimilarity = 33.55 %				
	Group South	Group North			
Groups	Average Abundance	Average Abundance	Average Dissimilarity	Percentage Contribution (%)	Cumulative percentage (%)
Decapods	0	2.02	3.4	10.12	10.12
Oligochaetes	2.55	1.42	3.03	9.02	19.14
Tanaids	1.64	2.71	2.97	8.85	27.99
Bivalves	2.81	4.46	2.84	8.48	36.47
Isopods	3.01	3.85	2.59	7.72	44.19
Amphipods	2.92	3.67	2.4	7.16	51.35
Polychaetes	5.85	6.33	2.4	7.14	58.49
Insects	1.85	1.41	1.89	5.63	64.11
Cumaceans	0.46	1.03	1.78	5.32	69.43
Nematodes	0.76	1.01	1.68	5.01	74.44
Hydrozoans	0	1.19	1.65	4.91	79.35
Harpacticoid	0.32	0.95	1.55	4.62	83.97
Mysids	0	0.9	1.34	4	87.98
Gastropods	0	0.48	0.97	2.89	90.86

Table 5.9 SIMPER test results showing the dissimilarity of macrofaunal communities between southern and northern sector of TMB during 2012-2013 period

2012-2013	Average dissimilarity = 35.83 %				
	Group South	Group North			
Groups	Average Abundance	Average Abundance	Average Dissimilarity	Percentage Contribution (%)	Cumulative percentage (%)
Bivalves	2.41	7.88	8.49	23.68	23.68
Oligochaetes	2.78	0.63	4	11.15	34.84
Isopods	5.17	5.03	3.4	9.48	44.32
Amphipods	4.05	4.7	3.17	8.84	53.16
Gastropods	5.45	4.4	2.7	7.54	60.7
Decapods	0.23	1.69	2.53	7.06	67.75
Tanaids	0.2	1.45	1.99	5.54	73.3
Polychaetes	2.44	3.47	1.8	5.03	78.32
Nematods	0.98	0.94	1.46	4.07	82.39
Hydrozoans	0	0.69	0.95	2.65	92.3

5.2.5.7 Comparison of diversity pattern of macrofauna in the south and north zone of TMB

During the study period, a clear spatial difference was observed in the diversity pattern of macrofauna in the south and north of TMB (Table 5.10). During first year except dominance value; diversity, richness and evenness indices were maximum in the northern region, whereas during second year, in contrast with first year diversity and evenness values were highest in southern zone than north. During 2011-2012 period, the Shannon diversity index in the north zone was 1.85 and that of south zone was 1.44. In contrast with first year, during second year the highest diversity value was observed in the southern (1.78) than the northern zone (1.46). Both the years, the highest Margalef's richness index was noted in the northern zone. Similar to Shannon diversity index, during first and second year Pielou's index was maximum in northern (0.53) and southern zone (0.57) respectively. During first year, Simpson index was maximum during southern (0.52) than the northern zone (0.4), whereas during second year, the maximum value was observed in the northern (0.51) than the southern zone (0.38).

Table 5.10 Comparison of diversity indices of macrofauna in the south and north zone of Vembanad estuary during 2011-2013 period

	2011-2012		2012-2013	
	South	**North**	**South**	**North**
H'	1.44	1.85	1.78	1.46
d	1.02	1.32	1.02	1.11
J'	0.46	0.53	0.57	0.43
λ'	0.52	0.40	0.38	0.51

5.2.5.8 Relationship of meio and macrofaunal communities on the south and north of TMB

During first year, among the meiofauna oligochaetes (22.89 %), nematodes (20.94 %) and isopods (11.2 %) were more abundant, contributing a major share in the southern zone whereas in the northern zone nematodes (28.76 %), harpacticoid copepods (17.68 %) and bivalves (13.43 %) contributed the major share of meiofauna. Similar to first year, in the southern zone, oligochaetes (25.14 %) and nematodes (20.98 %) were the dominant group whereas in the northern zone nematodes (32.7 %), harpacticoid copepods (14.29 %) and bivalves (13.43 %) were the dominant group. Both the southern (29.5 %) and northern zone (23.33 %), polychaetes contributed the major share of macrofauna during first year. At the same time, bivalves (13.73 %) and isopods (14.67 %) formed the second dominant group in the southern and northern zone respectively. During second year, polychaetes (21.46 %) and amphipods (20.83 %) contributed a major share among macrofauna in the southern zone, whereas in the northern zone bivalves (21.46 %) and isopods (25.46 %) contributed a major share in the southern zone. During first year meio and macrofaunal diversity, richness and evenness indices were higher in northern zone than southern zone of TMB. During second year, both meio and macrofaunal diversity and evenness were highest in southern zone than north, whereas richness and dominance indices highest in northern than southern zone.

5.3 Discussion

A total of 12 groups of meiofauna belonging to 7 phylum, 7 class and 3 orders were identified during the study period. It includes foraminifera, kinorhynchs, nematodes, oligochaetes, polychaetes, chironomids, ostracods, harpacticoid copepods, amphipods, tanaids, isopods and bivalves, in which nematodes formed the dominant group. Many studies have reported the dominance of nematodes in meiofaunal population in the Indian estuaries

(Damodaran, 1973; Abdul Azis and Balakrishnan Nair, 1983; Rao and Sarama, 1994; Ansari and Parulekar, 1998; Chinnadurai and Fernando, 2006b; Annapurna et al., 2015). In Cochin estuary John (2009) reported a total of 14 taxa of meiobenthos, in which major groups were nematodes, polychaetes, copepods; foraminifea and nematodes being the most dominant group. Earlier in Cochin estuary Desai and Krishnan Kutty (1967) noted three groups of meiofauna: a) temporary settling stages of bivalves and gastropods b) permanent meiobenthos and occasionally observed forms.Ciliates, copepods, sipunculids and other unidentified worms were found occasionally and the permanent groups such as foraminifera and nematodes were the dominant groups. In the present investigation, juveniles of polychaetes, oligochaetes, bivalves, amphipods, tanaids, isopods and chironomids were considered as temporary settling stages; foraminifera, kinorhynchs andostracods were found occasionally; nematodes were the permanent meiofauna. In the present investigation, settling stages of bivalve, *Villorita cyprinoides* were common. Hakenkamp and Morin (2000) stated that increased meiofaunal contribution depended upon the permanent meiofauna (taxa that complete their entire life cycle in meiofaunal size classes) but Stead et al. (2005) observed that 51% of the total secondary production was contributed by temporary meiofauna (taxa that grow later into the macrofaunal size class). Along the central west coast of India, stretching from Ratnagiri to Mangalore the major meiofauna taxa included nematodes, ostracodes, turbellarians, polychaetes, harpacticoid copepods, hydroids, nauplii and gastropods, bivalves and oligochaetes (Nanajkar et al., 2011). In the mangrove shore on the west coast of peninsular Malaysia Sasekumar (1994) noted that among the meiofauna nematodes, harpacticoid copepods, oligochaeta, and kinorhynchs were dominant. Ansari et al. (2012) observed that the faunal composition of meiobenthos in the southeast continental shelf of India was dominated by nematodes (57.63%), harpacticoid copepods (14.77%), foraminifera (13.89%). The other groups included polychaetes, ostracods, isopods, cumaceans, tanaidacea and tardigarda and contributed 13.71% of the fauna. During first year, nematodes were not encountered in station 1 and 2. This may be due to the influence of high densities of polychaete species *Namalycastis indica*. Tita et al. (2000) confirmed that increased abundance of nereididae polychaete species influenced the meiofauna both by predation and disturbance of the complex nematode sedimental tube system and thereby decreasing its feeding opportunities.

In the soft-bottom habitats of Patos Lagoon estuary (south Brazil), meiofauna was represented by 8 taxa, where nematodes and ostracods contributed more than 90% of total meiofauna (Rosa and Bemvenuti, 2005). In the south east continental shelf of India, meiofauna were represented by 9 groups; nematodes, harpacticoid copepods, foraminifera, polychaetes, ostracods, isopods, cumaceans, tanaidacea and tardigarda (Ansari et al., 2012). From the total meiofaunal groups, nematodes contributed 57.63%, followed by harpacticoid copepods (14.77%), foraminifera (13.89%) and others (13.71%). In the retting zone of Edava-Nadayara backwater and in the non retting zone of Paravur backwater meiofauna was composed of 8 and 6 taxa respectively (Abdul Azis and Balakrishnan Nair 1983). They noted that due to the effect of sulphide pollution in the retting zone of Edava-Nadayara backwater the meiofauna represented only 12 species, whereas in Paravur backwater the meiofauna was represented by 28 species. Earlier Desai and Krishnan Kutty (1967) observed that, among the meiofauna foraminiferns and nematodes were more abundant and maximum density of foraminiferns reached upto 1872371/m^2 in Cochin estuary. In the present investigation foraminiferns were observed occasionally and its maximum density was 15826/10 cm^2.

In the Sethukuda mangrove area and adjacent open sea, East Coast of India, the meiofaunal density ranged from 12029 to 23493 individuals

10 cm/m^2. Where the meiofaunal groups were represented by foraminifera, nematodes, harpacticoid copepods, ostrocodes, rotifers, ciliophora, cnidaria, gnathostomulida, insecta, propulida, bryozoa, turbullarians, tanaidaceans and polychaete larvae (Thilagavathi et al., 2011). In which the range of percentage composition of foraminifera was more (51.67 % to 96.866 %), followed by nematodes (1.39 % to 1.625 %), ostracodes (1.12 to 1.19%), harpactcoid copepods (0.27% to 0.22). From the meiobenthos in the Gulf of Martaban, Myanmar coast, north-east Andaman Sea, Ansari et al. (2014) identified nematodes, nemertines, foraminifera, harpacticoid copepods, turbellarian, polychaetes, kinorhynchs, ostracods, tardigrada and crustacean nauplii, in which nematodes were the dominant group (80 %) followed by benthic copepods (5.9 %) and foraminifera (2.8 %). Meiofauna, the most diversified element of the marine biota represented as many as 24 members from the 35 animal phyla, either showing a permanent life or just temporarily (Balsamo et al., 2010).

Ansari and Parulekar (1998) observed the distinct seasonal variation of meiofaunal population density in the Zuari estuary of Goa, where maximum density was observed during PRM and minimum during MN season and after monsoon, the meiofaunal recovery was also noted. Earlier similar trend in the seasonal variation of meiobenthos were noted in the Indian coast by Damodaran (1973) and Kondala Rao and Ramanamurty (1988). This type of observation corroborated with the meiofaunal distribution during second year. At the same time contrasting observation was observed during first year. Throughout the season high abundance of nematodes were found in the Poonthura estuary, Thiruvananthapuram, Kerala (Anila Kumary, 2016). Nematodes showed a distinct seasonal variation with minimum abundance during the MN season than pre and PM periods. In the Vembanad estuarine system nematodes were the dominant meiofauna taxa during all the seasons. But its maximum abundance was observed during MN and minimum during PM. Similar type of maximum abundance of nematodes during monsoon season was noted in the Coringa estuarine complex, Visakhapatnam (Annapurna et al., 2015). Tietjen and Lee (1972) noted that temperature played a key role in the reproduction of nematodes and its maximum density was observed during summer season.

In the present investigation, it was noted that harpacticoid copepods were completely absent during MN season. Rao and Sarma (1994) pointed out that during low salinity period the density of harpacticoid copepods was reduced. Similarly high density of harpacticoid copepods was observed during PRM, where salinity was maximum in the current study. According to (Ingole and Parulekar, 1998), harpacticoid density was closely related to salinity fluctuations. During PRM season, the meiobenthic copepods were reproductively active and these were fit to the climatic condition by their tolerant and adaptive behavior (Annapurna et al., 2015). According to Ansari and Parulekar (1998), during MN season, increased salinity reduction and resuspension of sediments lead to the mortality of metazoan meiofauna. Annapurna et al. (2015) revealed that increased numerical abundance of meiobenthos was observed with increased and constant salinity whereas decreased abundance was observed with decreased and varying salinity, during south west monsoon. Meiobenthic abundance was controlled by the availability of food because it aggressively consumed on diatoms, bacteria, protozoans, detritus, and dissolved organic carbon. In the case of meiobenthos of the Gosthani estuary on the east coast of India, Rao and Sarma (1990) noted that the decline in abundance of meiofaunal organisms during post south-west monsoon and northeast monsoon was mainly due to various factors such as decrease in salinity, cleaning of bed and banks of estuary, absence of tidal inputs and the resultant ending of meiofaunal recruitment in the neritic end of estuary.

Comparatively lower meiofaunal diversity indices were observed in Vembanad estuarine system. The lowest diversity, richness and evenness in station 1 were attributed to the dominance of oligochaetes, which was the only meiofaunal group present there. Due to the high dominance of only one or a few species of benthic organisms generally formed lowest diversity index value (Levin et al., 2009). In the Sethukuda mangrove area, Tamil Nadu and adjacent open sea of East Coast of India diversity index varied from 3.5 to 3.7, richness value varied between 6.4 and 8.5, and evenness index ranged from 0.84 to 0.88 (Thilagavathi et al., 2011). Meiofaunal diversity in the beaches along the Kerala coast revealed that Shannon diversity index varied between 1.14 and 2.03 and richness index varied from 0.59 to 0.98 (Priyalakshmi and Menon, 2014).

Different grouping of environmental parameters were answerable for the community structure of meiofauna and nematodes (Coull, 1999). According to Raes et al. (2007), numerous environmental variables are essential for the community structure of meiobenthos. Which include grain size, clay-silt content, sorting, oxygen content, sediment redox potential discontinuity layer, organic content, bioturbation by macrobenthos, macrofaunal competition and predation, water depth. Most of these factors were associated with sediment and hydrodynamic conditions. Ansari and Parulekar (1998) observed that salinity and availability of food influenced the horizontal distribution and chlorophyll a and interstitial water affected the vertical distribution of meiofauna in Zuari estuary of Goa respectively. BIOENV analysis showed that the distribution of meiofaunal communities was explained by distinct environmental factors in Vembanad estuary. During both the years, available nitrogen and organic carbon acted as a main structuring factor for meiofauna distribution. Positive correlation between the sediment organic carbon and meiofaunal density and also nematode abundance was noted in central West Coast of India by Nanajkar et al.

(2011). The influence of organic matter in structuring the nematode community was reported by Pusceddu et al. (2009). Mohan et al. (2012) noted that higher amount of organic carbon, silt content and carbonate affected the turbellaria distribution. At the same time, increased abundance of copepods was supported by higher sand and carbonate level and lower value of organic carbon in Junglighat Bay and Car Nicobar, Andaman and Nicobar Islands. The nematode community feed up on the bacterial and organic matter settled on the sediments (Danovaro, 1996). Preetha and Pillai (2000) found that meiobenthos were more abundant in the sandy and lowest in clay substratum. Similarly higher organic carbon in the sediments did not support more meiofauna. BIOENV analysis showed that water depth, salinity, Chl-a, Phaeo-a and silt–clay content were directly connected to variation in meiofauna especially nematode in the Changjiang estuary, East China Sea (Hua et al., 2009). Ansari et al. (2014) observed that meiofaunal density was maximum in fine silty clay and minimum in sandy substratum. In Andaman Sea, Ansari and Parulekar (1982) observed the increased abundance of meiofauna in the silty clay substratum and less abundant in sand silt clay type sediment. Fine sediment particles adsorb more organics and so more microbial flora and meiofauna were attracted there for feeding (Ansari, 2004). Ngo et al. (2013) observed that grain size, coliform concentrations, ammonium and DO formed the best combination to explain the meiofauna community structure in the estuaries of the Mekong river system, Vietnam. According to Cook et al. (2000), low level of DO did not influence on the abundance of nematodes. But the abundance of foraminifera, polychaetes, turbellarians and ostracods were very few in the low oxygen area (Gooday et al., 2000). Rosa and Bemvenuti (2005) observed the correlation of meiofaunal density and sediment particles in Patos Lagoon estuary (south Brazil), where density of meiofauna was lower in fine sediment. Meiofauna structures in Tramandai-Armazem estuary (South of Brazil) were influenced by transparency, depth and pH (Kapusta et al., 2005). Annapurna et al. (2015) observed that salinity along with sediment texture had great role in the meiobenthic abundance in the Kakinada Bay, Gaderu and Coringa estuarine complex.

Natural temporal changes to ecosystems are basically unpredictable because of undocumented natural cycles of long or unknown periodicity (Gray and Christie,1983). These cyclical changes and random between-year variations make long-term human changes in benthic communities difficult to detect (Currie and Parry, 1999). Owing to the dynamic physical conditions of estuaries, the spatial variation of macrofaunal composition extend from metres to kilometers, and its temporal variation ranged from days to years (Morrisey et al.,1992). So the knowledge of the spatial distribution patterns of macrobenthos along estuarine gradients might help to identify the linkages between species distributions and ecological processes and therefore to gain insight into the functioning of estuarine ecosystems (Thrush et al., 1999) and which is essential for implementation of integrated estuarine management. According to Abdelsalam and Tanida (2013), the macrofaunal assemblages are controlled according tophysical and chemical variables that characterize otherbiological parameters that influence their reproductive success. The reactions of benthos to natural and anthropogenic perturbations are especially important because of their ability to incorporate over time with changes to the water column and sedimentary characteristics (Elliott and Taylor, 1989).

A total of 18 taxa were representing the macrofaunal community structure of Vembanad estuarine system. The benthic macrofaunal community of the estuary, mainly consisted of polychaetes, molluscs and crustaceans, which carry out important functions such as breakdown of organic matter, nutrient recycling and materials transfer (Hutchings 1998; McLenaghan et al., 2011). These macrofaunal groups were widespread in most of the estuaries such as Negombo estuary, Sri Lanka (Dahanayaka and

Wijey Aratne, 2006); Schelde estuary (Ysebaert et al., 1998); Meghna river estuarine bed (Hossain et al., 2009); Han River Estuary, Korea (Yu et al., 2012); Kali estuary, Karwar (Harkantra, 1975); Pulicat Lake (Raman et al., 1975); Chilka lake (Ingole, 2002); Mandovi estuary (Gaonkar et al., 2013). In the present study, polychaetes and bivalves formed the abundant macrofaunal group in the first and second years of the study respectively. In benthic community, polychaetes form a major component because of their high species richness, biomass and density and their tolerance to pollution and other natural disturbances (Omena and Creed, 2004). During 1974, the macrobenthic fauna in Vembanad estuary was mainly composed of polychaetes, bivalves, decapods and amphipods and its abundance variedfrom 347 to 793 ind./m^2 and also a progressive decrease in the benthic population from lower reaches (maximum density of 1100 ind./ m^2) of estuary to upper region was noted (Ansari, 1974). Similar observation was also noticed in the present investigation. According to Gopalan et al. (1987), the major macrobenthic group in Vembanad estuary were bivalves, polychaetes, amphipods, isopods and tanaidaceans. Others such as decapods, sea anemones, echinoderms and fish larvae were present in minor proportions. In the present study polychaetes, amphipods, bivalves, isopods, chironomids, oligochaetes, tanaeids, fishes, crabs, shrimps and gastropods were the abundant groups. Fauna such as sea anemones, echinoderms were completely absent in the present study. In both the years, the minimum abundance was observed in station 5 (403 ind./m^2). During 2011-2012 period and 2012-2013 period, the sum of spatial abundance of macrofauna showed that the maximum average abundance was observed in station 10 (5317 ind./m^2) and station 9 (8964 ind./m^2) respectively. Compared to other stations, station 9 and 10 located in the northern most region of Vembanad estuary, close to the barmouth. Station 10 was distinct with increased percentage of silt and clay, which influenced the higher abundance of polychaetes. Salinity and bottom water pH were the key factors controlling the distribution and abundance of macrofauna, that was clearly reflected in the CCA plot. According to Yu et al. (2012), salinity was the prime abiotic factor affecting the macrobenthic community distribution. He mentioned that other parameters such as sediment grain size, dissolved oxygen and sediment organic carbon were only secondary factors. Similar observation was reported from tropical hypersaline coastal lagoon in Ghana (Lamptey and Armah, 2008). The present mesohaline condition in station 10 would have supported macrofaunal communities. During the study period the salinity of bottom water in the region ranged from 0.08 to 33.1 ppt. Similarly the increased abundance of macrofaunal communities in station 10 might be due to the presence of aggregation of tubiculous worms like *Diopatra neapolitana* and *Modiolus* in the region. These structures provided habitat for other macrofaunal communities. Compared to previous studies in the same estuary during 2011-2013 period, the macrofaunal communities were varied and its density was comparatively higher in the present investigation (Desai and Krishnan Kutty, 1967; Kurian et al., 1972; Ansari, 1974; Pillai, 1977 and Gopalan et al., 1987; Anon, 2001) (Table. 5.11). During 1966 bivalves, settling stages of gastropods and polychaetes were the dominant forms of macrofauna in Cochin estuary and its density was lower in September (Desai and Krishnan Kutty, 1967). In the bar mouth region large abundance of tubiculous polychaetes and bivalves were also reported. It has been reported that polychaetes contributed 80% of the benthic macrofauna in Cochin estuary; the central part of estuary showed the average density of 2261 ind./m^2 and northern part 2267ind./m^2 and the region was dominated by sandy sediments (Martin et al., 2011). In majority of the estuarine and coastal environments, polychaetes formed the dominant macrofaunal groups. In the Mumbai port area polychaetes were the most dominant macrobenthic group (72.09%) followed by decapods, amphipods and bivalves (4.56%) (Mandal et al., 2013). In Dhamara estuary Mahapatro et al. (2011) reported a total of 1870 ind./m^2 of macrobenthic organisms in which polychaetes formed as a dominant macrobenthic group. In the shelf waters of Visakhapatnam, more than forty groups of benthic fauna were reported by Vijayakumaran (2003). Among the benthic fauna, polychaetes were the dominant taxa contributing 62.5% followed by amphipods (17%). In the mangrove ecosystems of Tamil Nadu, Thilagavathi et al. (2011) noted a total of six diverse groups of macrofauna, of which polychaetes, gastropods, bivalves, amphipods, isopods and cumacea were the most important groups. In the present study, during second year in station 9, bivalves dominated and having a maximum abundance in March 2012 (55524 ind/m^2). It was noted that majority of the bivalves were juveniles. Vembanad estuary is famous for its clam fishery and this zone is one of the major clam fishing ground. In Vembanad estuary, the bivalve, *Villorita* showed continuous spawning that peaked in November-December and February-April (Anon 2006). In association with *Villorita,* the gastropod *Nassodonta insignis* were observed in stations 3, 7, 8 and 9. Such type of observation was reported by Jayachandran et al. (2016) in Kodungallur-Azhikode backwaters. Pillai (1977) reported fifteen species of molluscs from Cochin estuary, where *Modiolus undulates* and *Nuculana mauritiana* were the most abundant and its abundance varied from 44340/m^2 and 13720/m^2 respectively. In the present investigation only two species of bivalves were obtained from the macrofaunal samples, in which *Villorita cyprinoides* and *Modiolus sp.* was most abundant. *Modiolus sp.*was present only in the northern most region (station 10) and its maximum abundance occurred during February 2012 (34413 ind./m^2). The decreased abundance of macrofauna in station 5 was mainly attributed to the intense clam dredging in the surroundings of Pathiramanal island, which was disturbing other macrofaunal communities.

During the first year isopods formed the third dominant group and it was followed by amphipods and oligochaetes. During second year a shift in the numerical abundance of macrofauna were observed and amphipods and isopods become the third and fourth major macrofaunal groups. In the coastal marine biodiversity amphipods and isopods are important biotic components (Venkataraman, 2005). Benthic macrofauna in Tasmanian estuaries, Australia were dominated by crustaceans, in which amphipods and isopods comprised the richest taxonomic group followed by polychaetes (Edgar and Barrett, 2002). In Cochin estuary, during 1965-1966 period, amphipods were not recorded from the macrobenthic samples (Desai and Krishnan Kutty, 1967). While Kurian (1972) reported the dominance of amphipods in the area with highest density of 35000/m^2. Among the crustaceans, Pillai (1977) obtained amphipods as the dominant group from Cochin estuary. According to Gopalan et al. (1987), in the Vembanad estuary the maximum density of amphipods (70290/m^2) was mainly promoted by the input of macrophytic biomass especially from *Salvinia.* Similarly, the dominance of amphipod population was supported by low salinity and high dissolved oxygen and plant matter. In the macrofaunal crustacean communities amphipods, isopods and tanaids formed the important faunal groups (Geetha and Bijoy Nandan, 2014). Krishna Pillai (1963) reported the presence of isopods and amphipods in the southern zone of Vembanad estuary especially the Alappuzha and Vattakayal regions. During second year in the present study, the average abundance of isopods was maximum in the Aryad region. Among benthic fauna peracarid crustaceans were the most diverse and numerically abundant organism (Dauby et al., 2001). Peracarid crustaceans such as amphipods and isopods have ancient marine origin and its subgroups settled in brackish and freshwater habitats. It was characterized with direct development from their young in the brood pouch and the adults selected the suitable habitats where the broods were directly released (Parr et al., 2007). Peracarid crustaceans played a significant role in the structuring of benthic assemblages and were affected by organic pollution in the coastal areas (Tomassetti et al., 2009).

The gut content of demersal fish from Vembanad estuary contained large amount of isopods and amphipods (Kurup, 1982). From the 80 species of finfishes reported from Vembanad estuary, 44% were demersal (Asha et al., 2014). So the abundance of benthic macrofaunal communities in Vembanad estuary was important in maintaining the fishery production in the area. In Great Lake Superior, the percentage of fine sand was positively related to the abundance of amphipods and isopods (Breneman et al., 2000). In the present investigation amphipods and isopods were predominantly abundant in stations with highest sand percentage. Tanaids were abundant in stations 4 and 8 during first and second year respectively. Tanaids are present in both clean and polluted water and its abundance was controlled by sediment particle size and the amount of organic matter (Tomassetti et al., 2009). Ates et al. (2014) reported the highest dominance of tanaid, *Apseudopsis latreillii* in highest ratio of sand, gravel and muddy substratum. In this context, the present result was similar to station 4, characterized with silty sand and sandy silt and station 8 with sand fraction. Geetha and Bijoy Nandan (2014) noted that the highest abundance of tanaids in the middle zone of Cochin estuary was favoured by the availability of food and salinity. During the study period the average abundance of isopods was maximum in station 4, 9 and 7. These regions were involved as major fishing areas in Vembanad estuary, where isopods and amphipods were the major food items for the resident fishes. Benthic macrofauna species were important as food for economically important fish and shellfish species in most aquatic environment where they are the major secondary producers (Ajao and Fagade, 2002).The important commercial fishes of Vembanad estuary, *Etroplus suratensis, Horabagrus brachysoma* were omnivorous feeders mainly fed on micro and macro vegetation, apart from this it feds on crustaceans, insect larvae, bivalves, mysids, worms etc. (Jhingran and Natarajan, 1973; Keshava et al., 1988; Padmakumar et al., 2009). In the present investigation a distinct spatial pattern of macrofaunal groups was observed. Among the macrofaunal communities, oligochaetes dominated in the southern most stations such as station 1 and station 2. This may be related to the increased organic matter there. CCA analysis revealed that the organic carbon had a promising influence on the spatial distribution of oligochaetes in stations 1 and 2. During 1980 – 1981, in the same estuary oligochaetes were not recorded from the macrofaunal samples collected in this region (Gopalan et al., 1987). According to Iskaros and Dardir (2010), the predominance of oligochaetes in lakes was possibly due to their ability to adapt to various habitats and also to their tolerance to low oxygen content in anoxic conditions of the bottom sediment. Oligochaete abundance increased as organic content increased (Day, 1981), mainly through the intense asexual reproduction that results its quick increase in population within a short period of time (Giere and Pfannukuche, 1982). In the present investigation, increasing abundance of oligochaetes was noted in the southern zone, where the organically enriched sediments probably were responsible for the oligochaetes abundance. This also indicated the fresh water dominance in the ecosystem. During the present study a three fold increase in sediment organic carbon was observed in the southern most region especially station 1, Punnamada, where the oligochaetes utilized this condition and flourished.

Seasonal variation of macrofaunal communities was distinct in both the years. During first year the maximum percentage abundance was observed during PRM and minimum in MN, whereas in second year, PM season was having higher abundance and lowest in PRM. Earlier Anon (2001) reported ten groups of benthic fauna in Vembanad lake and observed that postmonsoon period was most favorable period for the occurrence of benthic fauna. In Vellar estuary the highest macrofaunal density was recorded during PM and lowest during MN (Ansari et al., 1982). Similar pattern of observations were reported from Cochin estuary (Sarala Devi and Venugopal, 1989) and also from south west coast of India (Harkantra and

Parulekar, 1986; Bijoy Nandan and Abdul Azis, 1996; Sarala Devi et al., 1996). These authors supported that influence of monsoon rain was influencing the lower density of macrofauna. During first year, percentage abundance of polychaetes was maximum in all the seasons. Such type of seasonal variation was observed in Rameswaram backwaters (Magdoom et al., 2010). Earlier Pillai (1977) observed a distinct seasonal variation of macrofauna in Cochin backwater. The maximum benthic population was observed during December to April and minimum during south-west monsoon period (July to September) and also observed that polychaetes dominated throughout the year. In Pulicat lake, the incidence and abundance of macrofauna was more during MN and PM seasons (Raman et al., 1975). Geetha et al. (2010) reported that MN was most favorable for benthic faunal abundance followed by the PM season in Cochin estuary.

During second year period, in Vembanad estuary, bivalves were dominant during PRM and MN, whereas polychaetes were dominant during PM season. This was in contrast with the observation from Zuari estuary where polychaetes dominated during the MN (92 %) and PRM (83%), whereas bivalves predominated in PM (57.7%) (Sivadas et al., 2011). Due to the monsoonal fluctuation, the natural environmental parameters showed a significant seasonal variation. So the variation of environmental settings caused differences in the macrofaunal community structure. The annual rainfall controlled the macrobenthic assemblage structuring of Mandovi estuary, where the average minimum faunal abundance was in monsoon onset regime (June) and maximum during the end monsoon regime (September-October) (Gaonkar et al., 2013). Seasonal variations of benthic fauna in tropical estuaries were prominent due to the monsoonal rains and its life cycle pattern was synchronized with monsoonal rain (Alongi, 1990), in which heavy rains caused increased mortality of benthic communities and ultimately lead to its defaunation. According to Sivadas et al. (2011) the faunal diversity was negatively impacted with monsoon. Monsoon onset regime characterised by effects in defaunation, migration and spawning of the macrofauna and increased organic matter and primary productivity during the peak monsoon regime and post monsoon period supported the recruitment process (Gaonkar et al., 2013). The seasonal variation of salinity in the whole estuary was highly related to rainfall and river discharge. Salinity formed the key factor of spatial heterogeneity among macrobenthos (Day et al., 1989; McLusky and Elliot, 2004; Fujii, 2007). In Vembanad estuary, the low salinity during monsoon was attributed to the increased riverine runoff which fully flushed the estuary, turning the estuarine water fresh (Shivaprasad et al., 2013b). Over the years the percentage contribution of macrofaunal groups varied considerably. Earlier during premonsoon season in the Punnamada region bivalves were the dominant group (77%) followed by polychaetes (12%) and amphipods (11%) (Gopalan et al., 1987). Whereas in the present investigation, polychaetes become dominant during first year and in second year oligochaetes were the dominant groups. The decreased abundance of bivalves in the region mainly resulted due to the salinity variation or from the environmental distress caused by intense tourism. After the construction of TMB, the southernmost part of estuary remained fresh or oligohaline condition. According to Suja and Mohamed (2010), the black clams could not reproduce well in low salinity and it required 10–12 ppt salinity for spawning. In the southern region the average bottom water salinity decreased upto < 3ppt. Similarly in the southern zone of Vembanad, monsoon rain and flood water brought large amount of mud and silt by five rivers into the lower reaches and this causes mortality of live clams (Anon, 2006).

Table 5.11 Time scale variation of macrofaunal communities in the Vembanad estuary (Values given in parenthesis are maximum density of macrofauna during the respective study)

Desai and Krishnan Kutty, 1967	**Kurian, 1972**	**Ansari, 1974**	**Pillai, 1977**	**Gopalan et al., 1987**	**Anon, 2001**	**2011-2013**
Polychaetes, Decapods, Gastropods, Bivalves, Ophiuroids, Sea anemones, Sipunculids, Fish)	Polychaetes, Isopods, Amphipods, Decapods, Gastropods, Bivalves, Sea anemone, Goboid Fishes	Polychaetes, Amphipods, Decapods, Gastropods, Bivalves	Polychaetes, Cumaceans, Isopods, Tanids, Amphipods, Decapods, Molluscs	Polychaetes, Isopods, Tanids, Amphipods, Decapods, Bivalves, Echinoderms, Sea anemones, Fish larvae	Polychaetes, Oligochaetes, Nemerteans, Amphipods , Gastropods, Bivalves	Polychaetes, Oligochaetes, Nemerteans, Cumaceans, Isopods, Tanids, Amphipods, Insects, Decapods, Gastropods, Bivalves, Fish
(8137/m^2)	(35000/m^2)	(1100/m^2)	(44340/m^2)	(73500/m^2)	-	(46036/m^2)

In Vembanad estuary among the macrofauna, bivalves and polychaetes contributed the major share of macrofaunal biomass. Such type of observation was reported by Kurian (1972). Among bivalves the occurrence of relatively small number of large bivalves contributes maximum macrofaunal biomass in stations 6, 7, 8 and 9. Similar observation was reported by Desai and Krishnan Kutty (1967) who obtained greatest average biomass (dry weight) of bivalves accounting to 354.95 g.m^{-2}. Earlier Ansari (1977) reported that, the average benthic biomass varied from 50.5 to 311.8 g/m^2 in Vembanad lake and the increase in biomass was mainly contributed by large sized bivalves. In the present investigation polychaete biomass ranged from an insignificant level to 86.67 g/m^2. Compared to other stations, the average biomass of polychaetes was highest in station 10. This region was particularly marked by the presence of large aggregation of polychaetes and bivalves. According to Diaz Castaneda and Harris (2004), polychaetes constitute one of the largest faunal assemblages on earth and the biomass in the sediments. Sunil Kumar (2002) noted that the biomass values of polychaetes in the Cochin estuarine mangrove habitat varied from 2.57 to 128.28 g.m^{-2}. In the Carson submarine canyon, Atlantic Ocean the biomass estimates of macrofauna ranged from 0.2 to 43.0gm^{-2} (Houston and Haedrich, 1984). Harkantra and Rodrigues (2004) found that the total wet biomass of soft bottom macrofauna in the Mandovi and Zuari estuaries of Goa varied from 1.5 to 6.7 g/m^2. They found that environmental parameters especially salinity and total organic nitrogen had a significant influencing role on macrofaunal biomass that explained 32-72% of the total variance. Earlier Devassy and Gopinathan (1970) reported that increase in benthic biomass from marine to fresh water habitat during monsoon in the estuary was attributed by the disappearance of transition from marine to freshwater condition. In the Cochin estuarine region the macrofaunal biomass decreased over the years. It varied from 27.8 g/m^2 in 1981 (Sarala Devi, 1986), 21.1g/m^2 in 1996 (Sheeba, 2005) to 15.4 g/m^2 in 2005 (Martin et al., 2011). In the current investigation, seasonal variation of macrofaunal biomass followed a similar trend as that of macrofaunal abundance. During first and second year, the highest macrofaunal biomass was observed during PRM and PM respectively. In the Zuari estuary macrofaunal biomass distribution had a similar pattern of its abundance, where maximum biomass value was observed during PM (72.07 g m^{-2}) and lowest in MN (4.27 g m^{-2}) (Sivadas et al., 2011).

Interpreting the effects of disturbance is complex with the assessment of natural or anthro- pogenic impacts on macrobenthic integrity. Because of various human interventions and urban development, the natural estuarine characteristics of Vembanad estuary were negatively impacted. Therefore, it is very vital to evaluate the ecological quality status of Vembanad estuary based on the composition and distribution of the macrobenthic community. In the aquatic environment use of macrobenthos as bioindicators to detect environmental disturbances and also the application of abundance and biomass curves to determine the level of pollution was common. Based on the abundance/ biomass comparison (ABC) curve half of the stations were in undisturbed condition and stations such as 1, 2, 3, 9 and 10 were in a disturbed condition. The stress factors varied from one site to another. Stressed regions were characterized with large number of small sized individuals, where the abundance curve lying above the biomass curve and in undisturbed condition, the biomass was dominated by one or few individuals, leading to a curve of abundance which lied below biomass (Specchiulli et al., 2010). Murugesan et al. (2011) used the ABC curve for assessing the impact of thermal plant discharges on benthic community of Tuticorin Bay and Vaan Island, East Coast of India. The curve confirmed polluted condition of Tuticorin Bay by representing the abundance curve lying above the biomass curve. In contrast, in Vaan Island the cumulative abundance curve was lying below the biomass curve denoting the undisturbed condition. The higher concentration of Cu and Zn in the southern most stations especially Punnamada, denoted a higher chance of heavy metal pollution and associated impact on the distribution of macrofauna in the estuary. In the coastal waters, physical disturbance of sediment and heavy-metal contamination had great impact on macrobenthic composition (Zhang et al., 2012). Rygg (1985) reported a negative correlation of sediment copper concentration and macrofaunal community diversity. Similarly heavy metal such as Zn, Cu, Pb, and Cr were also toxic to macrobenthos (Dauvin, 2008). Stations 1and 2 was highly disturbed with the continuous traffic of house boats and navigational boats whereas, station 3 and 9 by dredging operations of clam fishers. According to Sousa et al. (2007) constant disturbance of sediment by dredging, navigation and heavy metal contamination led to disturbance of macrofaunal community in the Lima estuary, as such the ABC curve in the estuarine region showed stressed condition with negative W value. Similarly in the present investigation, stations such as stations 1 and 2 were characterized with organically rich sediment (>4 %). In the Indian waters Ganapati and Raman (1973) noted that, organic carbon > 6% is anoxic to the marine life. Harkantra (1980) noted that when organic carbon content was >4 %, a decrease was observed in benthic fauna in the west coast of India. Similarly increased organic carbon lead to the proliferation of opportunistic species (Pearson and Rosenberg, 1978). So in stations 1 and 2, the numerical dominance of polychaetes, oligochaetes increased and their biomass contribution was less due to its smaller size. In the disturbed environment, polychaetes were the dominant group capable to reproduce quickly but with a smaller biomass (Pawhestri et al., 2015). Majority of the benthic organisms were characterized with increased rate of fecundity and so it could quickly convalesce from various stresses and able to settle to new habitats, increasing its abundance (Manoharan et al., 2011). However, the biomass dominance observed in other stations was due to the occurrence of larger and a few bivalves, *V. cyprinoides*. In contrast, station 9 having higher abundance of juveniles of bivalves, *V. cyprinoides*, but its biomass contribution was lower. The gastropods formed the major contributor of biomass in station 9. In station 10, the slightly disturbed condition was mainly due to the unstable association of small bivalve *Modiolus* having high density but low biomass with the tubiculous polychaetes structures. The k-dominance plot clearly confirmed the trend as observed in the variation of diversity indices. During the first year in station 2, there were only eight macrofauanl groups; in which polychaetes represented highest dominance over other groups. Similar situation was observed in station 9. Here the average abundance of polychaete was ten times higher than the second dominant group, the isopods, whereas in station 7 the increasing diversity and decreasing dominance of macrofaunal groups also occurred. From thirteen macrofaunal groups that were observed, the over dominance of any particular macrofaunal group was absent. In undisturbed condition, k-selected groups characterized with slow growing, large and late maturing groups settled. At the same time, r-selected species (opportunistic) was characterized with fast-growing small organisms (Warwick, 1986). During second year in station 9, a complete dominance of bivalves over other macrofaunal groups was observed denoting a stressed condition, whereas in station 6, the macrofaunal groups were evenly distributed and the dominance of any particular fauna were absent. During 2011-2012 period, the average diversity (H') value was 1.61; richness (d) was 1.15; evenness index, (J') was 0.49 and dominance (λ') was 0.47. During 2012-2013 period, the average diversity (H') value was 1.66; richness (d) was 1.09; evenness index, (J') was 0.51 and dominance (λ') was 0.44. In a healthy environment the Shannon diversity and Margalef richness were always higher (2.5 - 3.5) (Ajmal Khan et al., 2004). Higher values of evenness in station 6 and 7 indicated that macrofaunal community structure of this region was less modified due to ecological variations, whereas low evenness values in station 2 and 9 denoted the irregularity of macrofaunal distributions in the region.The low evenness values in station 2 and 9 were mainly attributed to the dominance of polychaetes. According to Wohlgemuth et al. (2016) the most dominant species exert a disproportionate influence on functioning at low levels of evenness. During the study period, there were number of illegal construction activities in the estuary came up with the violation of various laws, the Coastal Zone Regulation (CRZ) guidelines, EIA Notification-2006 etc. on the small private island Nediyathuruthu situated near to station 9 (The Kerala High Court, July 25, 2013 ordered the destruction of illegal construction in the estuary while noticing the violation of various laws). The associated dredging and reclamation activities of the region might have disturbed the bottom topography of the estuary and ultimately affected the benthic faua. Similarly dredging for clam shell beds for making white cement by the Travancore Cements Ltd. created a disturbed bottom and also in the present investigation it was noted that the depth of the region was drastically decreased over the years. According

Picanco et al. (2014), higher evenness value in the region indicated that the macrofaunal community of the area gets highly adapted at the same time undergoes severe ecological changes, but low evenness indicated that the fauna of the region were able to adapt easily and survive the changing environmental conditions. In the Schelde estuary, France species richness, diversity and total biomass of the benthic macrofauna decreased along the salinity gradient (Ysebaert et al., 1993). In the mesohaline zone, large numbers of short living, opportunistic species was dominated (Ysebaert et al., 1998). The increased dominance and decreasing evenness affect benthic community contributions to ecosystem process such as bioturbation and nutrient cycling. Similarly decrease in evenness caused more variation and a divergence in ecosystem performance (Wohlgemuth et al., 2016). In the present investigation, during first year diversity, richness and evenness indices was maximum in the northern zone of TMB than the southern zone. Mcintosh et al. (2008) reported that variations in the flow pattern after the dam construction limited the colonization and growth of macroinvertebrates and also represented as reduced community density and biomass.

Based on the results of ANOSIM and SIMPER test in Vembanad estuarine system, two groups of stations were identified. SIMPER analysis of macrofauna revealed that the dissimilarity of southern and northern sector of TMB was mainly because of the high abundance of oligochaetes on the southern stations and bivalves and decapods on the northern stations. It was observed that during second year higher abundance of bivalve population in the station 9 was distinct with other stations that would have contributed to a major difference. In southern most stations, oligochaetes, an opportunistic group was more abundant in the organic rich sediments. Over all the average abundance of tanaids and amphipods were abundant in the northern sector. According to Begun et al. (2010) salinity, plays a decisive role in the development of macrobenthic communities. Due to the construction of

TMB and its closure during the premonsoon months prevented the ingress of marine and estuarine migratory fauna into the southern sector of the estuary. It also reduced the tidal flushing and altering the circulation and mixing process, especially in the southern part of estuary. The barrage in the Fitzroy river estuary, Australia prevents the tidal movement of saline water into the upstream freshwater region and also let the flood water flow towards downstream (Currie and Small, 2005). Alterations in the flow pattern lead to the accumulation of pollutants especially fertilizers and insecticides etc. from the agrarian zones of Kuttanad; industrial effluents, domestic and urban sewages from the Alappuzha town and the estuarine banks resulting in eutrophication and it was confirmed through the TRIX analysis. In such eutrophic estuaries, the macrofaunal density frequently increased, while its species number declined (Karlson et al., 2002). Degradation of estuaries was also associated with the change in macrofaunal composition where the dominance of polychaetes increases and the sensitive taxa such as molluscs and crustaceans declineed (Pearson and Rosenberg, 1978). The closure of TMB obstructed the downwardly migrating ovigerous females of *Macrobrachium rosenbergii* and also the upwardly migrating post larvae (Kurup et al., 1992). Similarly the marine fish and prawn migration become reduced towards the southern zone and the commercially important decapods *M. rosenbergii* and *Scylla serrata* become a rarity (Asha et al., 2014). According to Suja and Mohamed (2010) the long-term effects of the TMB and industrial pollution in the southern zone caused a decline in bivalve abundances. The salinity variation in estuaries affected the reduction in the number of species and limited its distribution based on its salinity tolerance (Edgar and Barrett, 2002). Harkantra and Rodgris (2004) found that the euryhaline species of echiuran worm, recorded in high salinity region and oligohaline bivalve, *Meretrix casta* observed in low salinity region. Earlier Desai and Krishnan Kutty (1967) reported that higher salinity, favourable substratum and rich nutrients in the Cochin barmouth region maintain more benthic fauna than the interior part of Cochin estuary. According to Kurian (1972), salinity acted as a major factor controlling the distribution of organisms in Cochin estuary. He noted that the intensity of macrofauna were more abundant in the lower parts of the estuary than the upper reaches of estuary. In addition, in the less saline regions the bivalve mollusc Meretrix was replaced by *Villorita* sp. In the present investigation, the bivalve mollusc Meretrix was not encountered in the Vembanad estuary, whereas *V. cyprinoides* was observed in most of the stations except station 10. TMB played a crucial role in influencing the salinity pattern of Vembanad estuarine system, where the salinity incursion from Arabian sea was restricted during the barrage closure period that made the southern part of TMB predominantly oligohaline in all the seasons (Asha et al., 2016). According to Arun (2009), moderate level of salinity was found to be a favourable factor for the growth of *V. cyprinoides* in Cochin estuary. He reported that periodic opening and closing of TMB had been influencing the ecology of clam beds and threatening the survival of clams in the estuary. Compared to previous studies, the macrobenthic communities and its density varied over the years in Vembanad estuary. During 1980-1981 period, among the macrofauna there was no oligochaetes reported in the Vembanad estuary (Gopalan et al., 1987). In the present investigation, increased abundance of oligochaetes was noted in the southern zone, where the organically enriched sediments was probably responsible for the oligochaetes abundance. This also indicated the fresh water dominance in the ecosystem also. Dam construction made the transport of organic material impossible and favour the colonization of oligochaetes and chironomids in River Vistula (Southern Poland) (Krzyzanek, 1986). Similarly the amphipod density in the southern zone decreased whereas, the density of insects, polychaetes, isopods and tanaids increased over the years (Table 5.12). In the present study, sea anemones and echinoderms were not recorded in the estuary. Earlier Kurian (1972) reported that in Cochin estuary, sea anemones and brittle stars occur during the PRM months and continue until the rains become intense and were absent during MN, which showed its intrusion from the marine environment.

Table 5.12 Comparison of macrofaunal density (ind. /m^2) in different regions of Vembanad estuary between 1980-1981 and 2011-2012 period

	Alappuzha		**Thaneermukkom**		**Vaikom**		**Aroor**	
	'80-'81	**'11-'12**	**'80-'81**	**'11-'12**	**'80-'81**	**'11-'12**	**'80-'81**	**'11-'12**
Bivalves	460	721	0	278	0	369	8	8603
Polychaetes	70	1174	160	664	113	1413	228	3961
Amphipods	65	23	60	40	20	114	40	420
Isopods and tanaids	0	68	40	45	7	261	264	131
Oligochaetes	0	641	0	0	0	6	0	23
Insects	0	102	0	28	0	0	0	0

The current tourism activities in the southern zone of Vembanad estuary had various negative impacts. More than thousands of house boats were operating in the Punnamada (southern) region without any carrying capacity studies. Houseboats were country boats that were used in the early days for the transport of goods especially rice harvested from Kuttanad paddy fields. But after the regional developmental initiatives, its use was diversified. During the 1990s, houseboats became major features of backwater tourism (Rajan et al., 2011; Nansen, 2011). The toilet wastes from houseboats and oil and grease leakages were a major problem in the region (Rajan et al., 2011). In the present investigation, an increasing accumulation of toxic heavy metals in the sediment was also noted. Generally, the polluted area become unreceptive for native benthic species and a few native and exotic stress tolerant opportunistic species become dominant (Galil, 2000). The fishing boats and small tourism boats fitted with outboard engines and other navigational boats releases large amount of hydrocarbons into the system (www.ramsar). In the oil polluted area, the oligochaetes showed a positive loading with oil concentration and negative with bivalve biomass in subarctic lakes (Skvortsov, 1997). This was confirmed by comparing the macrofaunal abundance from the previous studies. Compared to other macrofaunal groups, bivalves were the dominant group in the Alappuzha region during 1980s, where no oligochaetes were reported at that time (Gopalan et al., 1987).

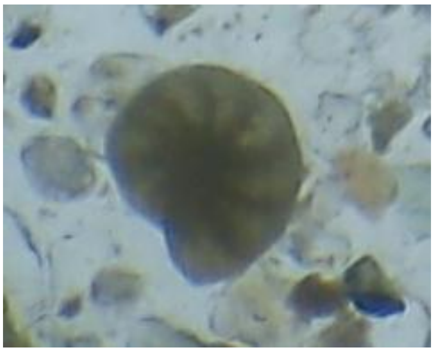
Macrofaunal community structure and its distributional pattern were influenced by a number of environmental variables (Nouri et al., 2008; Gaonkar et al., 2013). So the variations of macrofauna have been recommended for a suitable biological indication of environmental disturbances (Morrisey et al., 2003). Due to the pressure of human activities on the coastal estuarine areas, the value of studies at large spatial scales has recently been accepted in order to protect and conserve the habitats and resources healthier. Normally abiotic parameters influenced largely on the distributional patterns of benthic fauna, whereas the combined effects of both abiotic and biotic factors have limited impacts (Ellingsen, 2002). Sediment characteristics have been identified as important factors responsible for macrofaunal distribution (Hily et al., 2008). Similarly physicochemical properties of the water column directly or indirectly shaped the presence or absence of benthic species by controlling food supply, larval dispersion and metabolism (Pearson and Rosenberg, 1978). Several studies established the relation between environmental parameters and benthic communities and found significant correlation of parameters such as salinity, pH, sediment characteristics and dissolved oxygen with macrobenthic communities (Raut et al., 2005). In the coastal environment especially in estuarine areas, the tolerance of benthic fauna to the short term variation of environmental parameters has great impact on their spatial distribution (Laprise and Dodson, 1993). BIO-ENV analysis revealed that both water and sediment parameters played an important role in influencing the distribution and abundance of soft bottom macrobenthic assemblages of the Vembanad estuary. Among the hydrological parameters, salinity and among sediment parameters, sediment pH, potassium, organic carbon, available nitrogen and sand fraction were the most important drives in the ecology of macrobenthic communities. Silt and clay fraction of sediment also showed best relation with macrofaunal abundance. Spatial variation in the community structure of macrobenthos along the estuarine gradients have been linked to fluctuations in salinity, depth, sediment grain size, and organic carbon content (Day et al., 1989). No single factor was responsible for determining soft-sediment benthic community settlement and colonization (Snelgrove and Butman, 1994). A combination of factors such as temperature, salinity and DO together with sediment characteristics have major role in controlling the distribution and abundance of benthic fauna (Jayaraj et al., 2007). According to Mahapatro et al. (2011) salinity, water temperature, dissolved oxygenand pH showed strong correlation with macrobenthic density in Dhamara estuary, Bay of Bengal. BIO-ENV analyses in the Han River estuary, Korea showed that environmental variables such as benthic salinity, total organic carbon, dissolved oxygen in bottom water, and silt and sorting values of the sediment showed the maximum correlations with macrofaunal composition (Rho = 0.7, $P < 0.01$). Apart from this, low salinity due to the freshwater input from the Han River was the important environmental variable affecting macrobenthic community structure during the rainy season (Yu et al., 2012). According to Gilberto et al. (2004) type of the estuarine bottom, salinity and turbidity front were the main physical variables in structuring benthic communities of the Rio de la Plata estuary, Argentinae – Uruguay.

In the present study, CCA results indicated the spatial influence of environmental parameters on the benthic community structure. The first canonical axis accounted for 84.59 % and second axis accounted for 92.35%

of variance. During first year, the CCA plot indicated that oligochaetes preferred silty substratum having high organic carbon load and available nitrogen. The increased percentage of clay fraction as well as rising salinity in station 10 was suitable for the bivalve, *Modiolus*, decapods and nematodes. Isopods and tanaids were abundant in sandy substratum having an increased percentage in stations 5, 6, 7, 8 and 9. During second year, similar to first year oligochaetes had higher affinity towards organic carbon, available nitrogen and silt fraction. Polychaetes showed good association towards clay and bottom water pH. The average polychaete abundance was highest in 10, which was characterized with increased percentage of clay. In station 10, the sediment was mostly clayey silt in nature and contain more organic matter. Sukumaran et al. (2013) observed the preference of some polychaete families on silty-clayey substratum with high organic content in the Gulf of Kachchh. In the coastal waters of Dabhol, west coast of India, among the macrofaunal communities polychaetes dominated in the clayey silt substratum (Ingole et al., 2002). Normally benthos preferred with medium grain sized sediment containing low organic matter (Jayaraj et al., 2007). Several researchers reported the high density and biomass of polychaetes in the sand, clayey-sand, sandy silt or sandy clay substratum (Ansari et al., 1977; Harkantra et al., 1982; Musale and Desai, 2011). Tanaids and bivalves showed affinity towards sand and the long vector length of sand fraction showed its significant correlation with stations such as 7, 8 and 9. Isopods and mysids showed close association towards stations 4 and 6 respectively. Molluscs were positively correlated with salinity and sand in the northwest shelf waters of India (Jayaraj et al., 2007). The CCA analysis between the benthic assemblage and abiotic variables in the Mfolozi-Msunduzi estuarine system, KwaZulu-Natal, South Africa found that dissolved oxygen concentration was an important variable in structuring the benthic community, followed by salinity, temperature, percent organic content in the sediment and particle size (Ngqulana et al., 2010). Through the CCA analysis Gogina et al. (2010) observed that distribution of benthic macrofaunal communities in the western Baltic Sea was correlated with depth, organic carbon, salinity and grain size etc.

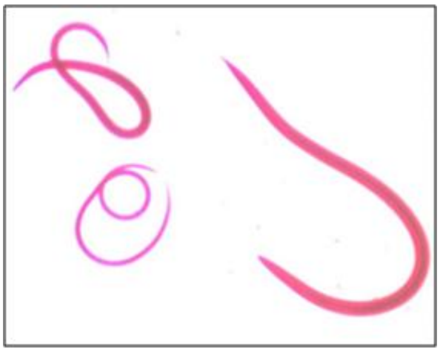
Meiofauna and macrofauna were closely linked to various types of interactions, but comparative studies are rare (Baldrighi and Manin, 2015). According to Netto and Gallucci (2003), the variations on the diversity and total abundance of the benthic faunal components such as meiofauna and macrofauna were mainly dependent on detritus biomass. In both years, among the macrofauna, oligochaetes were observed only in the southern zone of TMB, whereas in meiofauna, its abundance was highest in southern zone of TMB during both the years. During first year, bivalve was present both the southern and northern zone in meio and macrofaunal composition. But during second year bivalves was observed in the macrofaunal samples of both southern and northern zone and its abundance was maximum in the northern zone, whereas in the meiofaunal samples bivalves were observed in the northern zone only. In the present investigation, temporary meiobenthos such as juveniles of bivalves and oligochaetes were observed mainly in the northern and southern zone respectively. According to Desai and Krishnan Kutty (1967), occurrence of temporary meiobenthos such as settling stages of bivalves and gastropods indicated the spawning period of adults. According to Pinto and Bemvenuti (2003), macrofauna was modifying the physical characteristics of its habitat, which ultimately affected the distributional patterns of meiofauna in the region. Both the years, in the Vembanad estuary, similar pattern were found for meio and macrofaunal diversity variation. Similarly during first year meio and macrofaunal diversity, richness and evenness indices were higher in northern zone than southern zone of TMB. Ingels et al. (2014) revealed that macrofauna can act positively by improving the substrate heterogeneity through bioturbation and formed micro-habitats or negatively by predation pressure, physical disturbance by sediment browsing or no effect on the meiofaunal communities and diversity. According to Warwick (1984), meio and macrofauna have different mechanisms for diversity protection; meiofauna followed a particular feeding behavior and macrofauna generally unselective for its feeding behavior.

a)



b)

c)

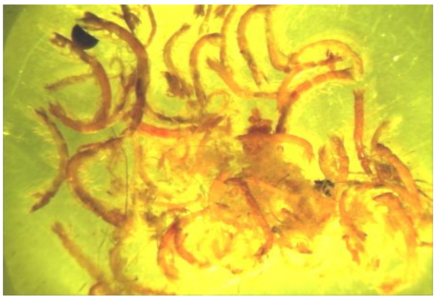


d)

e)

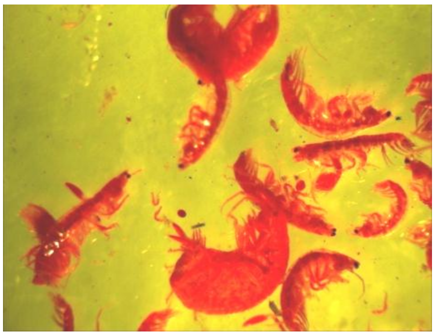
f)

g)



h)

i)



j)

k)

l)

m)

n)

o)

p)

q)

Plate 5.1 **(a-q)** Major benthic fauna observed in Vembanad estuary during 2011-2013 period (Magnification: a to n – X4)
a) Obelia; b) Foraminifera; c) Nemerteans; d) Nematodes; e) Oligochaetes; f) Polychaetes; g) Cumaceans; h) Isopods; i) Tanaids; j) Amphipods; k) Chironomids; l) Decapods; m) Harpacticoid copepods; n) *Nassodonta insignis*; o) *Villorita cyripnoides* ; p) *Modiolus* sp.; q) Gobidae.

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Chapter 6

COMMUNITY STRUCTURE AND ECOLOGY OF POLYCHAETES IN THE VEMBANAD ESTUARY

Contents

6.1 Introduction

Human pressure linked to population expansion and economic developments are disturbing the health and ecological stability of coastal and marine environments, creating them one of the most endangered ecosystems on earth. Apart from anthropogenic pressures, the oceanic biosphere is experiencing rapid global climate change, which comprised an unprecedented threat to human and natural systems (Grilo et al., 2011). Number of biotic and abiotic factors interacts with life-history features of benthic species, which could directly or indirectly, influenced by climate change. As a result, benthic ecosystems demonstrate complex interactions and species-specific responses in relation to climate change (Birchenough et al., 2010). Climate change could transform population dynamics over time and space, phenology, and the geographical distribution of communities (Dulvy et al., 2008). These alterations could cause habitat loss and species extinctions over time with impact on biogeochemical fluxes, ecosystem functioning, and biodiversity. It was evident that organisms are responding to climate change either through its presence or absence. As human activities increases tremendously in and around the water bodies, it caused problem such as habitat reduction and fragmentation. It affected the growth and development of many species of benthic polychaetes and crustaceans. Now the signature of primary climate change elements such as temperature, precipitation and hydrological systems fluctuations are more prominent. According to Richey et al. (1990) the major link of global carbon cycle was the transfer of organic matter through rivers from land to ocean waters. In coastal aquatic systems nutrient enrichment from agricultural fields might act synergistically with warming water temperatures due to climate change and it could lead to eutrophication issues.

In the soft bottom habitats, polychaetes are qualitative and quantitative key components of the benthic fauna (Fauchald, 1977). It forms a major component because of their high species richness, biomass and density and their tolerance to pollution and other natural disturbances (Mendez et al., 1998; Omena and Creed, 2004; Tomassetti and Porrello, 2005). So they play a great role in stability and functioning of the benthic communities and the ecology in general (Hutchings, 1998). According to Rouse and Pleijel (2001) around 12,000 polychaete species were depicted world widely. Investigation on the dynamics of polychaete distribution has been considered as an alternate for the total macrofaunal studies (Cacabelos et al., 2008; Lourido et al., 2008; Quiroz-Martinez et al., 2011). Polychaete are segmented worms belonging to phylum Annelida, class Polychaeta and formed the dominant and abundant group, actively involved in bioturbation, burial of organic matter, recycling and reworking of bottom sediments (Hutchings, 1998; Gholizadeh et al., 2012). So the activities of benthic fauna have a profound impact on the physical, chemical and biological conditions in aquatic sediments. Sediment alteration and water irrigation or aeration move both particles and porewater within the sediment (Aller, 1982). Among the macrofaunal communities, polychaetes contributed about 80% to the total community and its diet consists of microbial organisms including bacteria, microalgae, protists and fungi; meiobial and organic materials (Shou et al., 2009). It forms an important component in the marine food chain (Musale and Desai, 2011) especially for bottom fish and some mammals (Rao et al., 2009; Parulekar et al., 1982; Herman et al., 2000). Majority of the benthic macrofauna have pelagic larval stages and important for inter-habitat energy transfer by forming food for many nektonic organisms (Baustian et al., 2014). Among the macrofauna, polychaetes play a significant role in the benthic food web as well as pelagic, that bring them to a major link in benthic – pelagic coupling (Cattaneo-Vietti et al., 1999).

Benthic indicator species are used to describe the nature of habitat and also used for grouping ecological quality. The zoobenthic indicator species are classified into two, sensitive and tolerant. The sensitive groups differentiated a habitat by their abundance or dominance or its special presence in the specific habitats. The tolerant groups are the resistant species and the opportunistic ones. The tolerant species are usually opportunistic with low ecological necessities but at times showed special relation with specific conditions (Simboura and Zenetos, 2002). Among the macrofauna, polychaetes are the initial macrobenthic colonist in the region faced with severe physical or chemical distress, and several taxa that survived in the most disturbed site of pollution (Pearson and Rosenberg, 1978). These greatly diverse taxa followed different modes of reproductive approaches and feeding guilds (Hutchings, 1998). These contain both sensitive and tolerant species, in which several species have been identified as species that responds quickly to environmental distress (Pearson and Rosenberg, 1978; Zajac et al., 2003). It is mainly considered as organic pollution indicators to monitor the health of the marine environment (Remani et al., 1983; Warwick and Ruswahyuni, 1987; Ingole et al., 2001; Belan 2003; Surugiu, 2005; Jayaraj et al., 2007) mainly due to its diversity of feeding modes, differential tolerance to environmental imbalances and limited motility studies (Hughes et al., 2009). Polychaetes are considered as a suitable indicator of assemblage models of macrobenthic structure and species richness (Fauchald and Jumars, 1979). According to Cinar et al. (2006), in disturbed soft-bottom sediments 90-100% of the zoobenthic populations are contributed by polychaetes. According to Rhoads and Young (1970), polychaetes were the major contamination tolerant marine organism in which it lives at the water-sediment interface and considered that the region is active in biologically as well as chemically. Both soft and hard bottom marine environment polychaetes established significant abundance and diversity (Cinar et al., 2006; Musco, 2012). Due to its high adaptation towards various environmental situation, polychaetes took a major role in ecosystem functioning and so it was exploited as an alternative for the evaluation of benthic community dynamics (Giangrande et al., 2005; Papageorgiou et al., 2006). The impacts of human activities on the biological diversity, extending from gene to ecosystem, are most apparent in coastal areas. Besides eutrophication, various activities recognized to affect significantly the biodiversity of coastal ecosystems consist of shipping (oil spills, exotic species), industry (chemical effluents), dredging and dumping, fishing and mariculture, biological invasions, tourism, etc. (Simboura and Zenetos, 2002). Polychaete composition and abundance varied in the sewage discharge sites, mainly due to the colonization behavior of Capitellidae or Spionidae families (Cabral Oliveira and Pardal, 2016). These opportunistic species were tolerant to pollution and increased their abundance in the organic enriched areas where the competition for food and space was very limited (Elias et al., 2006).

Polychaete species distribution mainly associated with the particle size of sediment in which it reside and form the largest faunal grouping on earth (Diaz Castaneda and Harris, 2004). According to Lardicci et al. (1993) and Mendez (2002), the distribution of soft-bottom polychaetes was influenced by water movement, dissolved oxygen, granulometry of sediment and organic matter. Generally benthic marine assemblages are described by observing the spatial variation of species structures due to its interaction of various environmental processes occurring at various ranges and depths. The fluctuations of key environmental parameters including temperature, salinity, wave action etc. have a great influence on the spatial heterogeneity among benthic assemblages (Benedetti-Cecchi et al., 2000). Natural habitat disorder could result in variation of salinity, temperature, currents and wave generated resuspension of sediments (Bhattacharya et al., 2000).

6.1.1 Significance of study

Nowadays biological diversity studies along the estuaries are a subject of understanding the impact of human activities and climate change. Coastal estuarine systems are highly biodiverse aquatic system that has been receiving increasing attention in the last decade. These habitats of the country are facing serious anthropogenic impacts that have remarkably influenced the biodiversity and ecosystem character of the organisms inhabiting the benthic zone. The value of estuarine ecosystems for its diversity and ecology are important at the same time various human interventions and developing activities are altering the normal habitats and lead to the loss of their natural function (Nienhuis and Smaal, 1994). But it's monitoring and assessments are very few and lacking especially in the important Ramsar sites.

Estuaries are transitional zones, where salinity varied between fresh water and seawater. It served as a nursery ground for various commercially and ecologically important species of marine and migrating finfish and shellfish (Ragnarsson and Raffaelli, 1999). Most of the world cities are located near this productive and complex estuarine ecosystem and received all types of contaminants (http://www.fao.org). Similarly various impacts from harbour development and continuous dredging operations, discharge of nutrient enriched water from domestic, industrial and agricultural areas modified the bathymetry along with significant hydrodynamic variations increasing the eutrophication processes in the estuarine system (Sousa, 2001). Vembanad estuarine system is not an exclusion from these types of impacts mainly in the upper and lower reaches of the estuary. Monitoring of the macrobenthic estuarine species variation relative to their environment is essential to the improvement of estuarine ecology and also benthic fauna are considered as an important bioindicators of various environmental consequences (Ysebaert et al., 2003). Ecological information on polychaete species is scarce from the entire Vembanad estuary. It is essential to regularly monitoring the distribution, abundance and ecology of polychaete species with respect to the time and space. However, little is known about the ecological information on polychaete habitat types and species distribution from the Vembanad estuary especially on the southern zone. After the construction and operation of TMB, the ecological and environmental condition of the southern and northern zone of TMB that have radically been altered. So it is necessary to monitor the nature of polychaete and its species–environment relations in a changing estuary-related with various coastal development and anthropogenic impacts. The results of the present study would attempt to bridge our knowledge on the community structure of polychaetes in terms of its ecology and taxonomy in an ecosystem that is very vital for evolving the benthic diversity conservation for trophic health of the system.

6.1.2 Review of Literature

A number of studies have been made on the various aspects of macrofaunal polychaetes in the world. Polychaetes are usually the most abundant taxon in benthic communities and have been most often utilized as indicator species of environmental conditions (Elias et al., 2005; Papageorgiou et al., 2006), ecotoxicological studies (Mauri et al., 2003; Xie et al., 2005; King et al., 2004; Cornelissen et al., 2006 and Gopalakrishnan et al., 2007); as indicators of benthic species diversity (Pearson and Rosenberg, 1978; Mendez et al., 1998; Bailey-Brock et al., 2002; Olsgard et al., 2003; Elias et al., 2005; Dix et al., 2005); as indicators of organic enrichment and organic contaminants (Pearson and Rosenberg, 1978; Pridmore et al., 1992; Gray et al., 2002; Klaoudatos et al., 2006; Surugiu, 2009; Sanchez et al., 2011; Ansari et al., 2014); indicator species of heavy metal pollution (Athalye and Gokhale, 1991; Gibbs et al., 2000; Berthet et al., 2003; Trannum et al., 2004; Chiarelli and Roccheri, 2014).

Polychaete diversity in the estuarine habitats of Terminos Lagoon, southern Gulf of Mexico was studied by Hernandez-Alcantara et al. (2014). In the soft bottom habitat they identified 3,398 specimens belonging to 119 species and 33 families of polychaetes. Fujii, (2007) observed the linkages between environmental variables and patterns in the distribution, abundance and biomass of estuarine intertidal macrobenthos, in order to provide a basis for describing the effect of future sea level rise in the Humber estuary, UK. The effect of brine discharge over bottom polychaete was studied by Nabavi et al. (2013). The study found that the abundance, richness and diversity indices of polychaetes were decreased in the regions near the discharges area comparing to control sites far from the brine discharges zone. The Canonical Correspondence Analysis (CCA) revealed that salinity and temperature have influenced the community structure of polychaetes. Distribution patterns of shallow water polychaetes along the coast of Alexandria, Egypt indicated that its distribution was related to variation of grain size and the assemblage distribution affected by parameters such as BOD, dissolved oxygen, organic carbon, organic matter, salinity, temperature and pH (Dorgham et al., 2014). Polychaete assemblage studies of impacted estuary, Guanabara Bay, Brazil revealed that high polychaete diversity observed in the hydro dynamically exposed areas composed of sandy, oxidized or moderately reduced sediments with normal oxygen conditions in the water column and low diversity and high biomass of deposit and suspension feeding polychaetes observed in the area having equal proportions of silt, clay and fine sand (Santi and Tavares, 2009).

Banse (1959) studied the marine polychaetes of Mandapam, southeast coast of India. Thampi (1963) noted the swarming of heteronereids in the Gulf of Mannar coast (southeast coast of India). Pollution effects on ecobiology of benthic polychaetes in Visakhapatnam harbour was studied by Raman and Ganapati (1983). Ansari et al. (1986) studied the effect of high organic enrichment on the benthic polychaete population in Mandovi estuary of Goa. The study revealed that the population density, biomass and species diversity of polychaetes were greater at the sewage receiving site. Bijoy Nandan and Abdul Azis (1995) have made observations on the benthic polychaetes of the retting zone in the Kadinamkulam kayal, Kerala. The study revealed that retting of coconut husk led to the depletion in the diversity, richness and eveness of the polychaetes. Distribution of polychaetes in Hooghly estuary was observed by Datta et al. (1995). They recorded the polychaete species *Dendronereides gangetica* in the tip of estuary, which was contrast with the earlier reports of the distribution of this species in the mouth of the estuary. A checklist of soil-dwelling polychaetous annelids from some Indian mangrove habitats was made by Sunil Kumar (2001). Benthic fauna of Southwest and Southeast coasts of India was studied by Sarala Devi et al. (1999) and recorded 36 species of polychaetes along the southwest coast, and 19 from the southeast coast. Benthic ecology of prawn culture fields and ponds near Cochin was studied by Preetha (1993). Biomass, horizontal zonation and vertical stratification of polychaete fauna in the littoral sediment of Cochin estuarine mangrove habitat, South West coast of India was studied by Sunil Kumar (2002). The study observed that a substantial difference in percentage composition of fauna up to 15cm depth of mangrove soil. The study analysed that a high numerical abundance and coexistence of certain euryhaline species showed significant similarity index among polychaete fauna. It was reported that there was a decrease in macrobenthos distribution from 1981 to 2005 period in the Cochin estuarine area. During 1981 the total number of species were observed as 76 (Sarala Devi, 1986); during 1996 it was 56 species (Sheeba, 2000) and during 2005 it was decreased to 47 species (Martin et al., 2011). John (2009) observed that benthic community structure in Cochin estuary was dominated by polychaetes and a total of 32 polychaete species was recorded. Sivadas et al. (2010) studied the benthic polychaetes of Marmugao harbour and found that benthic polychaetes as good indicators of anthropogenic impacts. A total of 71 polychaete species were identified during the study. The harbour stations were dominated by *Prionospio pinnata*, *Magelona* sp., *Tharyx* sp. and *Cossura* sp. Dominance of opportunistic deposit feeding species and low polychaete diversity in the inner harbour indicate negative impact of harbour activities on the species composition. Ajmal Khan and Murugeshan (2005) discussed about the polychaete diversity of Indian estuaries. He pointed out that information on polychaetes is available only from 8 estuaries out of 33 on the east coast and only from 4 out of 34 on the west coast. The study also mentioned for a strong need to understand the variations of polychaete diversity in relation to environmental changes, especially pollution. The community structure of polychaete fauna from the Vasishta Godavari estuary was analyzed by Rao et al. (2009). This study provides key information on the biodiversity of polychaete communities of the Vasishta Godavari estuary. Changes in the occurrence of hard substratum fauna in the Mumbai harbour, India was investigated by Gaonkar et al. (2010). In the investigation they found that acron barnacle *Balanus amphirit*e and tube dwelling polychaete *Protula tubularia* were the dominant hard substratum species. The study envisages that hard substratum faunal composition in the Mumbai harbour environment was changing due to ever increasing human perturbation. Spatio temporal variation in benthic polychaetes and relationship with environmental variables in Sundarban Biosphere Reserve on the northeast coast of India was investigated by Sarkar et al. (2005). The study demonstrates that textural composition of the sediments, together with hydrodynamic and geotechnical properties, have the greatest control to quantify the differences in polychaete community. Biodiversity of benthic polychaetes from the coastal waters of Paradip, Bay of Bengal was studied on a seasonal basis by Ingole (2007). Composition, distribution and diversity of macrobenthic assemblage were investigated. The composition of macrobenthos differed considerably between the seasons and its diversity was higher at offshore deeper stations where as density was higher at near shore shallower stations. Sukumaran et al. (2011) identified thirty polychaete species including18 families in the Ratnagiri bay, India. *Parheteromastus tenuis*, *Sabellides* sp., *Paraprionospio pinnata* and *Cirriformia tentaculata* was the most dominant species.

Musale and Desai (2011) carried out a study to observe the abundance and diversity of soft bottom macrobenthic polychaetes along the South Indian coast, along with observations on sediment characteristics. They observed 63 different forms of polychaetes along the coast, which represented the major macrofauna. Occurrence of *Prionospio pinnata* and *Capitella capitata* the deposit feeders and indicators of organic pollution denoted the area is organically rich. Canonical correspondence analysis indicated that majority of polychaete species preferred low organic carbon, sandy silt, or sandy-clay substratum. The lower polychaete abundance at high organic carbon and high silt and clay areas can be attributed to avoidance of organisms to rich organic matter and suboxic levels, being a possible indication that these characteristics adversely affecting the polychaete abundance and distribution. Polychaete community structure in the south eastern Arabian Sea continental margin (200–1000m) was investigated by Abdul Jaleel et al. (2014). They observed that the gradients in dissolved oxygen concentration in the Oxygen Minimum Zone, textural variation of sediment with different bathymetric zones were affect the differences in polychaete size and community structure. Mandal and Harkantra (2013) studied upon the macrobenthic community structure in the ports of Mumbai, India. The study found that sediment texture, temperature and suspended particulate matter (SPM) were the most important environmental variables influencing polychaete species composition. Manokaran et al. (2013) studied the feeding guild composition of macrobenthic polychaetes from continental shelf of southeast coast of India. Mahapatro et al. (2015) described the polychaete species in the Chilika Lake after opening of a new lagoon inlet. Among the total 45 polychaete species *Nereis reducta*, *Capitella capitata*, *Heteromastus filliformis*, *Minuspio cirrifera* were dominant.

6.2 Results

6.2.1 Community structure of polychaete species in the Vembanad estuarine system

6.2.1.1 Spatio-temporal variation of polychaete species

During 2011-2012 periods, *Namalycastis indica* and *Dendronereis aestuarina* were two polychaete species present in station 1. In which the annual average density was maximum for *Namalycastis indica* (30 ± 78 ind./m^2) than *Dendronereis aestuarina* (380 ± 327 ind./m^2). The maximum density of *Dendronereis aestuarina* was observed during April 2012 (272 ind./m^2) and majority of the months this species was not obtained from the sample. The species *Namalycastis indica* found in all the months but its varied from a minimum of 45 ind./m^2 (October 2011) to a maximum of 1271 ind./m^2 (April 2011). Similar to first year, during second year *Namalycastis indica* and *Dendronereis aestuarina* were two polychaete species present in station 1. The maximum density of *Dendronereis aestuarina* observed during March 2012 (204 ind./m^2) and it was not observed during April, May, September, November and December 2012. Its average density in the station was 89 ± 98 ind./m^2, whereas *Namalycastis indica* was observed in all the months and the maximum density was observed during October (2860 ind./m^2) and minimum during March 2012 (477 ind./m^2) with an average density of 1443 ± 757 ind./m^2.

During 2011 – 2012 periods, similar to station 1, *Namalycastis indica* and *Dendronereis aestuarina* were the two polychaete species present in station 2. The species *Dendronereis aestuarina* was found only during the months of March 2011 (91 ind./m^2), April 2011 (68 ind./m^2) and November 2011(23 ind./m^2) with an annual average of 15 ± 31 ind./m^2. Here, the density of *Namalycastis indica* varied between 250 ind./m^2 (Aril 2011) to 13416 ind./m^2 (September 2011) with an annual average of 3131 ± 3938 ind./m^2. Similar to first year *Namalycastis indica* and *Dendronereis aestuarina* were the two polychaete species present in station 2 during second year. In the case of *Dendronereis aestuarina*, the maximum density was observed during June 2012 (1180 ind./m^2) and it was absent during the month of March, April, May and January 2012. Its average density was observed as 219 ± 358 ind./m^2. Similar to *Dendronereis aestuarina*, the maximum density of *Namalycastis indica* was observed during June 2012 (6560 ind./m^2) and it was absent during April 2012. Compared to

Dendronereis aestuarina, the average density of *Namalycastis indica* was higher in station 2 (2351 ± 2252 ind./m^2).

During both the years, the distribution of polychaete species in station 3 was not different from station 1 and 2. *Namalycastis indica* and *Dendronereis aestuarina* were the two polychaete species present in station 3. During first year the annual average density of *Namalycastis indica* and *Dendronereis aestuarina* were 732 ± 698 ind./m^2 and 195 515 ind./m^2 respectively. The density of *Namalycastis indica* varied from a minimum of 45 ind./m^2 (March 2011) to 2224 ind./m^2 (October 2011). During 2012 – 2013 period, the average density of *Namalycastis indica* was 847 ± 797 ind./m^2 and for *Dendronereis aestuarina* was 187 ± 283 ind./m^2. The monthly, maximum density of *Dendronereis aestuarina* was observed during May 2012 (998 ind./m^2) and was absent during July and November 2012. In the case of *Namalycastis indica* the maximum density was observed during October 2012 (2906 ind./m^2) and was absent during November 2012.

During 2011 – 2012 periods, a total of six polychaete species were found in station 4. Among the polychaete species, *Namalycastis indica* showed the maximum average density (433 ± 450 ind./m^2), followed by *Owenia* sp.(138 ± 199 ind./m^2) and *Dendronereis aestuarina* (85 ± 171 ind./m^2). Here, the density of *Namalycastis indica* varied between 45 ind./m^2 (September 2011 and February 2012) and 1407 ind./m^2 (November 2011). The species such as *Parheteromastus tenui, Mediomastus* sp., *Glycera alba* were not found regularly and its abundance was 68 ind./m^2 (May 2011), 341 ind./m^2 (October 2011) and 68 ind./m^2 (July 2011) respectively. Compared to first year, during second year only three species of polychaetes such as *Dendronereis aestuarina, Namalycastis indica* and *Owenia* sp. were found in station 4. In which the average density was maximum for *Namalycastis indica* (199 ± 209 ind./m^2) followed by *Dendronereis aestuarina* (98 ± 101 ind./m^2). *Owenia* sp. was present only in the month of March (23 ind./m^2), July (159 ind./m^2) and August (136 ind./m^2) 2012.

During first year 5 species of polychaetes were observed in station 5. *Dendronereis aestuarina* showed maximum average density (148 ± 186 ind./m^2), followed by *Namalycastis indica* (127 ± 127 ind./m^2) and *Prionospio cirrifera* (25 ± 38 ind./m^2). During second year in station 5, three species of polychaetes were present. Compared to other species, the average density of *Prionospio cirrifera* was higher in station 5 (424 ± 728 ind./m^2), followed by *Dendronereis aestuarina* (180 ± 310 ind./m^2) and *Namalycastis indica* (57 ± 105 ind./m^2).

During first year, five species of polychaetes were observed in station 6. Among the polychaete species, the average density of *Prionospio cirrifera* was maximum with 242 ± 526 ind./m^2. *Prionospio cirrifera* was not found in station 6 in most of the months and its maximum density was observed during January 2012 (1544 ind./m^2), followed by March 2011 (1158 ind./m^2). Compared to other species *Dendronereis aestuarina* was observed in most of the months in station 6. But its monthly variation was varied significantly. Its maximum density was observed during September 2011 (272 ind./m^2) and not observed in December 2011and February 2012. Similar to first year during second year also, *Prionospio cirrifera* was the dominant group, having an average density of 399 ± 829 ind./m^2 in station 6. *Dendronereis aestuarina* formed the second dominant group with an annual average of 153 ± 248 ind./m^2. Monthly variation of *Dendronereis aestuarina* found that the maximum density was observed during May 2012 (885 ind./m^2), followed by June 2012 (341 ind./m^2). Similar to *Dendronereis aestuarina*, the maximum density of *Prionospio cirrifera* was observed during May 2012 (2928 ind./m^2).

During 2011-2012 periods, four polychaete species were observed in station 7. Among these polychaete species, the maximum annual average density was observed for *Namalycastis indica* (178 ± 214 ind./m^2), followed by *Prionospio cirrifera* (153 ± 275 ind./m^2) and *Dendronereis aestuarina* (110 ± 96 ind./m^2). During second year, *Prionospio* cirrifera, *Dendronereis aestuarina*, *Namalycastis indica*, *Parheteromastus tenuis* and *Cossura coasta* were the polychaete species present in the station 7. *Prionospio cirrifera* formed the dominant group, with an annual average of 1069 ± 1412 ind./m^2 followed by *Dendronereis aestuarina* (371 ± 370 ind./m^2).

During first year, seven species of polychaete were observed in station 8, in which *Prionospio cirrifera* (176 ± 322 ind./m^2) and *Dendronereis aestuarina* (68 ± 72 ind./m^2) showed the maximum average density. Monthly distribution of *Prionospio cirrifera* noted that, it was observed only during March 2011 (704 ind./m^2), April 2011 (863 ind./m^2), May 2011(23 ind./m^2) and January 2012 (522 ind./m^2). During second year a total three polychaete species were observed in station 8, in which *Prionospio cirrifera* (274 ± 351 ind./m^2) formed the dominant group, followed by *Dendronereis aestuarina* (132 ± 134 ind./m^2) and *Parheteromastus tenuis* (85 ± 91 ind./m^2). Monthly, the species *Prionospio cirrifera* was maximum during May 2012 (1044 ind./m^2) and January 2013 (817 ind./m^2). Maximum density of *Dendronereis aestuarina* was observed during March 2012 (408 ind./m^2) and it was absent during July, September and January 2012. Maximum density of *Parheteromastus tenuis* was observed during June 2012 (295 ind./m^2), followed by April 2012 (159 ind./m^2) and February 2012 (159 ind./m^2).

During first year, fourteen species of polychaetes were observed in station 9, in which *Prionospio cirrifera* (1487 ± 1754 ind./m^2), *Parheteromastus tenuis* (643 ± 548 ind./m^2) and *Dendronereis aestuarina* (125 ± 140 ind./m^2) was having maximum average density. The maximum abundance of *Prionospio cirrifera* (6560 ind./m^2) and *Parheteromastus tenuis* (1998 ind./m^2) were observed during February 2012. Total eight species of polychaetes were observed in station 9 during 2012 – 2013 periods. *Prionospio cirrifera* formed the dominant group (1347 ± 1909 ind./m^2) followed by *Parheteromastus tenuis* (554 ± 661 ind./m^2). The maximum density of *Prionospio cirrifera* was observed during the month of March 2012 (7037 ind./m^2) and followed by May 2012 (2179 ind./m^2) and it was not observed during September 2012. *Scoletoma impatiens* (4 ± 13 ind./m^2), *Sabellaria pectinata* (4 ± 9 ind./m^2) and *Owenia* sp. (10 ± 18 ind./m^2) were the other major species.

During 2011-2012 period, nineteen species of polychaetes were observed in station 10, in which *Diopatra neapolitana* (4137 ± 5467 ind./m^2), *Prionospio cirrifera* (454 ± 307 ind./m^2), *Micronephthys oligobranchia* (363 ± 378 ind./m^2) and *Parheteromastus tenuis* (363 ± 272 ind./m^2) was having maximum annual average density. The abundance of *Diopatra neapolitana* varied from a minimum of 91 ind./m^2 in August 2011 to maximum of 18342 ind./m^2 in October 2011. During 2012-2013 period, total twelve species of polychaetes were observed in station 10, in which *Diopatra neapolitana* showed the maximum average density (6042 ± 9099 ind./m^2), followed by *Prionospio cirrifera* (845 ± 934 ind./m^2) and *Parheteromastus tenuis* (819 ± 665 ind./m^2).

Overall during 2011 – 2012 period, the total distribution of polychaete species in Vembanad estuary, showed a maximum percentage contribution by *Namalycastis indica* (34.22 %), followed by *Diopatra neapolitana* (27.98 %), *Prionospio cirrifera* (17.15 %), *Parheteromastus tenuis* (7.18 %) and *Dendronereis aestuarina* (6.03 %). Spatially the maximum average density of *Namalycastis indica* was observed in station 2 (3131 ind./m^2) followed by station 3 (732ind./m^2) and station 4 (433 ind./m^2) (Annexure 6.1 and Fig. 6.1). The minimum average density was observed in station

10 (2 ind./m^2). It was noticed that the ANOVA of spatial distribution of *Namalycastis indica* was significant at 1% level ($P \leq 0.01$). ANOVA of *Namalycastis indica* was significant at 1% level between stations ($P \leq 0.01$) and between seasons and stations ($P \leq 0.01$). The species *Prionospio cirrifera* was observed between station 5 and station 10. Its maximum average density was observed in station 9 (1487 ind./m^2) and station 10 (454 ind./m^2). In the case of *Parheteromastus tenuis* the maximum average density was observed in station 9 (643 ind./m^2) and station 10 (363 ind./m^2). Based on the ANOVA analysis, the species such as *Prionospio cirrifera* ($P \leq 0.01$) and *Parheteromastus tenuis* ($P \leq 0.01$) showed significant spatial variation. The species such as *Micronephthys oligobranchia*, *Sigambra parva*, *Cossura coasta*, *Aphelochaeta filiformis*, *Scoletoma impatiens*, *Sabellaria pectinata* and *Pectinaria* sp. was only present in station 9 and 10, in which maximum average density of these species was observed in station 10. The ANOVA showed that the spatial variation of species such as *Micronephthys oligobranchia* ($P \leq 0.01$), *Sigambra parva* ($P \leq 0.01$), *Cossura coasta* ($P \leq 0.01$), *Aphelochaeta filiformis* ($P \leq 0.01$), *Scoletoma impatiens* ($P \leq 0.01$) and *Sabellaria pectinata* ($P \leq 0.01$) were significant at 1% level. The species such as *Paraprionospio pinnata* (av. 62 ind./m^2), *Dipolydora flava* (av. 44 ind./m^2) and *Diopatra neapolitana* (4137 ind./m^2) was obtained only in station 10. The ANOVA showed that the spatial variation of species such as *Ceratonereis* sp. ($P \leq 0.01$), *Paraprionospio pinnata* ($P \leq 0.01$), *Dipolydora flava* ($P \leq 0.01$), *Glycera alba* ($P \leq 0.01$), *Owenia* sp. ($P \leq 0.01$), *Diopatra neapolitana* ($P \leq 0.01$) and *Nereis* sp. ($P \leq 0.01$) showed significant variation at 1% level.

During 2012 – 2013 period, among the total distribution of polychaete species in Vembanad estuary, the maximum percentage abundance was observed by *Diopatra neapolitana* (31.3 %), followed by *Namalycastis indica* (25.41 %), *Prionospio cirrifera* (22.58 %), *Dendronereis aestuarina* (8.02 %) and *Parheteromastus tenuis* (7.6 %). The ANOVA analysis revealed that the spatial variation of species such as *Diopatra neapolitana* ($P \leq 0.01$), *Namalycastis indica* ($P \leq 0.01$), *Prionospio cirrifera* ($P \leq 0.01$), *Parheteromastus tenuis* ($P \leq 0.01$) were significant at 1% level. Spatially the maximum average density of *Dendronereis aestuarina* was observed in station 7 (371 ind./m^2) followed by station 2 (219 ind./m^2) and station 3 (187 ind./m^2) (Annexure 6.2 and Fig. 6.2). The minimum average density was observed in station 9 (44 ind./m^2) and station 10 (74 ind./m^2). In the case of *Namalycastis indica,* the maximum average density was observed in station 2 (2351 ind./m^2) and station 1 (1443 ind./m^2). But it was not found in station 8 to 10. Similar to first year, the species *Micronephthys oligobranchia* was present only in station 9 and 10, in which maximum average density was observed in station 10 (450 ind./m^2). The species *Parheteromastus tenuis* was observed from station 6 to station 10. Its maximum density was observed in station 10 (819 ind./m^2) and station 9 (554 ind./m^2). The polychaete species *Sigambra parva* was present only in station 10, with an annual average of 176 ind./m^2. The polychaete species, *Prionospio cirrifera* was observed in most of the northern stations. It was absent in southern most stations such as Punnamada, Pallathuruthy, Rani/Marthadam and Aryad. Similar to *Sigambra parva*, the species such as *Dipolydora flava* and *Diopatra neapolitana* was present only in station 10 (Aroor). The species such as *Glycera alba*, *Scoletoma impatiens* and *Sabellaria pectinata* were present only in stations 9 and 10. The maximum density of *Cossura coasta* and *Scoletoma impatiens* were observed in station 10, with an annual average of 110 ind./m^2 and 53 ind./m^2 respectively. The maximum average density of *Sabellaria pectinata* was observed in station 9 (4 ind./m^2). The average density of *Owenia* sp. in station 4 and station 9 was 27 ind./m^2 and 10 ind./m^2 respectively. ANOVA result showed that spatial variation of *Micronephthys oligobranchia* ($P \leq 0.01$), *Sigambra parva* ($P \leq 0.01$), *Dipolydora flava* ($P \leq 0.01$), *Glycera alba* ($P \leq 0.01$), *Scoletoma impatiens* ($P \leq 0.01$) were significant. The ANOVA showed that the spatial variation of *Cossura coasta* ($P \leq 0.05$) and *Owenia* sp. ($P \leq 0.05$) were significant at 5 % level.

6.2.1.2 Seasonal distribution and abundance of polychaetes

During 2011 – 2012 period, in PRM and PM season *Namalycastis indica* contributed > 87 % in station 1. During MN season, its contribution reached to 100 %. Similar to station 1, during MN season *Namalycastis indica* contributed 100 % in station 2. In station 3, during PRM season *Dendronereis aestuarina* contributed 61.7 % followed by *Namalycastis indica* (38.3%), whereas during MN and PM season, *Namalycastis indica* contributed 99 % to the total polychaete population. In station 4, the major variations in percentage contribution of polychaete species occurred during PRM. In this season, *Dendronereis aestuarina* (48.28 %) and *Namalycastis indica* (32.18 %) formed the dominant species. At both MN and PM seasons *Namalycastis indica* showed higher percentage value with 73.49 % and 70.71% respectively. The percentage values of *Owenia* sp. varied seasonally and its maximum value was observed during MN (22.89 %), followed by PM (20.2 %) and PRM (16.09 %). In all the season, *Dendronereis aestuarina* and *Namalycastis indica* exhibited dominance in station 5. At the same time, its percentage contribution varied seasonally. The species such as *Prionospio cirrifera* and *Owenia* sp. have a percentage value of 7.69 % and 10.26 % respectively during PRM season and during PM season it was 13.46 % and 1.92 % respectively. At the same time, these two species were not observed during MN season. In station 6, during PRM and PM season, *Prionospio cirrifera* exhibited dominance over other polychaete species and contributed 64.04 % and 82.56 % during PRM and PM respectively. During MN season, it was completely absent and *Dendronereis aestuarina* formed the dominant group (70.3 %). Similar to station 6, *Prionospio cirrifera* become the dominant group during PRM (56.4 %) and PM (44.6 %) season in station 7. In contrast, *Namalycastis indica* exhibited dominance during

MN season (63.86 %). The second dominant species, *Dendronereis aestuarina* contributed 21.79 % and 31.33 % during PRM and MN respectively. During PM season, *Namalycastis indica* formed the second important contributor among polychaete species (35.14 %), followed by *Dendronereis aestuarina* (20.27 %). Similar to station 6 and station 7 *Prionospio cirrifera* become the dominant group during PRM (60.87 %) and PM (62.16 %) season in station 8. Similar to station 6, *Dendronereis aestuarina* exhibited dominance during MN season (66.67 %). *Parheteromastus tenuis* formed the second dominant species during PRM (20.87 %), followed by *Dendronereis aestuarina* (13.04 %). Apart from *Dendronereis aestuarina*, *Namalycastis indica* (22.2%) and *Parheteromastus tenuis* (5.56 %) formed the major contributor during MN season. In station 9, all the seasons, *Prionospio cirrifera*, *Parheteromastus tenuis* and *Dendronereis aestuarina* were the major polychaete species where its percentage contribution varied seasonally. In station 10, *Diopatra neapolitana* become the dominant species in PRM (63.1 %) and PM (87.3 %). Whereas during MN, *Micronephthys oligobranchia* become a dominant species (29.7 %) followed by *Diopatra neapolitana* (25.5 %).

During 2012-2013 period, in station 1 *Namalycastis indica* become the dominant species in all the season contributing > 90% in all the seasons followed by *Dendronereis aestuarina.* Similar to station 1, in all the seasons *Namalycastis indica* formed the dominant species in station 2. During PRM season, the percentage contribution of *Namalycastis indica* becomes 99 % and in MN and PM season it contributed > 90 %. *Dendronereis aestuarina* contributed 9.2 % and 9.3 % during MN and PM season respectively. In station 3 during PRM season, *Namalycastis indica* (56.16 %) and *Dendronereis aestuarina* (43.84 %) formed the major dominant species. During MN and PM season *Namalycastis indica* contributed 95.2 % and 89.49 % respectively. In station 4, in each season the percentage contribution of polychaete species varied considerably. During PRM *Namalycastis indica* formed the dominant species, contributing 57.5 % of total polychaetes. Second dominant species, *Dendronereis aestuarina* contributes 40 % to the total species composition followed by *Owenia* sp. 2.5 %. Similar to PRM, *Namalycastis indica* become the dominant species (68 %) in MN, followed by *Dendronereis aestuarina* (18.6 %) and *Owenia* sp. (13.4 %). Whereas during PM, the percentage contribution of *Dendronereis aestuarina* increased upto 52.9 % and the percentage contribution of *Namalycastis indica* decreased to 47 %. Compared to other seasons, the *Owenia* sp. was completely absent during PM. In staion 5, *Prionospio cirrifera* become the dominant species in all the season and its percentage contribution was not much varied in each season. Similar to *Prionospio cirrifera*, the seasonal variation of *Dendronereis aestuarina* was not much differed during the study period. Similar to station 5, *Prionospio cirrifera* and *Dendronereis aestuarina* become the dominant species in station 6. But its percentage contribution varied seasonally. *Prionospio cirrifera* contributed 73.4 %, 64.6 % and 69.3 % during PRM, MN and PM seasons respectively. At the same time, during PRM, MN and PM season *Dendronereis aestuarina* contributed 26.1 %, 35.4 % and 20.4 % respectively. During PM season, *Parheteromastus tenuis* contributed 8 % to the the total polychaete species. In station 7, *Prionospio cirrifera* were the dominant species during all the seasons. The highest percentage of *Prionospio cirrifera* was present during PM (78 %) followed by PRM (75.5 %) and MN (56.8 %). Similar to station 5 and 6, *Dendronereis aestuarina* was the second highest polychaete species observed during all the season in station 7. In station 8, the most dominant species during PRM season was *Prionospio cirrifera* (57.9 %), followed by (29.6 %) and *Parheteromastus tenuis* (12.5 %). During MN season *Parheteromastus tenuis* become the dominant species (41.3 %) followed by *Dendronereis aestuarina* (30.4 %) and *Prionospio cirrifera* (28.3 %). In station 8, during PM season

Prionospio cirrifera (57.89 %) and *Parheteromastus tenuis* (38.5 %) were the dominant polychaete species. In station 9, *Prionospio cirrifera* were the dominant polychaete species in all the seasons. Percentage of *Prionospio cirrifera* was recorded maximum during PRM (73.6 %), followed by MN (62.9 %) and PM (57.89 %). *Parheteromastus tenuis* were recorded the second dominant species in all the seasons. Its maximum percentage contribution was observed during PM (38.5 %), followed by MN (30.1 %) and PRM (23.8 %). During the study period, the seasonal variation of polychaetes was distinct in station 10. *Diopatra neapolitana* recorded the maximum percentage during PRM (66 %) and PM season (79 %), whereas its percentage contribution was very much decreased during MN (0.8 %) season. During PRM *Prionospio cirrifera* (12 %) and *Parheteromastus tenuis* (11 %) formed the second and third dominant polychaete species. During MN, *Parheteromastus tenuis* formed the dominant species (30 %), followed by *Micronephthys oligobranchia* (26.5 %) and *Prionospio cirrifera* (16.4 %). During PM season *Prionospio cirrifera* (7.9 %) and *Parheteromastus tenuis* (6.3 %) formed thze second and third dominant species.

By observing the overall seasonal variation of polychaete species, in each season the variation of polychaete species density was distinct. During first year the maximum polychaete species density was observed during PRM (74.05 %), followed by PM (20.7 %). In contrast with first year, during second year the maximum seasonal density of polychaetes was observed during PM (45.83 %). During both the years, the minimum density was observed during MN, where 5.25 % observed during first year and 22.31 % during second year. During first year, *Diopatra neapolitana* formed the dominant species (68.32 %), followed by *Prionospio cirrifera* (13.32 %) and *Namalycastis indica* (7.14 %) in PRM (Fig. 6.3.a). Whereas during MN, the overall density of *Namalycastis indica* increased up to 71.55 % and the density of *Diopatra neapolitana* decreased up to 4.97 % (Fig. 6.3.b). The second dominant density was shown by *Prionospio cirrifera* and it contributed 6.19 %. During PM the contribution of *Diopatra neapolitana* reached the maximum percentage of 81.5 % and the second dominant species *Namalycastis indica* contributed 8.75 % (Fig. 6.3.c). The ANOVA analysis showed that the seasonal variation of polychaete species such as *Dendronereis aestuarina* ($P \leq 0.05$), *Namalycastis indica* ($P \leq 0.05$), *Sigambra parva* ($P \leq 0.05$) and *Dipolydora flava* ($P \leq 0.05$) was significant at 5% level. During second year, PRM season *Prionospio cirrifera* become the dominant group (40.2 %), followed by *Diopatra neapolitana (*22.9%) and *Namalycastis indica* (12.3 %) and *Dendronereis aestuarina* (11.7%) (Fig. 6.4.a). Similar to first year, during MN season *Namalycastis indica* become the dominant group (55.9 %) followed by *Prionospio cirrifera* (15.6 %) and *Dendronereis aestuarina* (11.1 %) (Fig. 6.4.b). At the same time, the percentage abundance of *Diopatra neapolitana* was very negligible during MN season (0.1 %). Whereas during PM season, *Diopatra neapolitana* become the dominant group (52.3 %), followed by *Namalycastis indica* (19.7 %) and *Prionospio cirrifera* (13.7 %) (Fig. 6.4.c). The ANOVA analysis showed that the seasonal variation of *Prionospio cirrifera* were significant ($P \leq 0.01$).

6.2.1.3 Succession of polychaete species

In 2011-2012 periods, during the opening period of TMB, *Namalycastis indica* (av. 328.39 ind. /m^2) formed the copious species followed by *Dendronereis aestuarina (*av.124.09 ind. /m^2*)* and *Prionospio cirrifera* (av.106.7 ind. /m^2). During the closure period of barrage, the average numerical abundance of *Namalycastis indica* was (1061.23ind. /m^2) increased and *Dendronereis aestuarina* (91.18 ind. /m^2) decreased (Table 6.1). The species such as *Prionospio cirrifera* disappeared during the closure period of barrage and the capitellid polychaetes *Parheteromastus tenuis* and *Mediomastus* sp. appeared in this period. Similar to first year, during the open period of TMB *Namalycastis indica* (av. 861 ind. /m^2), *Dendronereis aestuarina (*av. 200 ind. /m^2*)* and *Prionospio cirrifera* (av. 238 ind. /m^2) were the abundant polychaete species during second year. Compared to the open period, during the closure period the average density of polychaete species decreased as *Namalycastis indica* (av. 617 ind. /m^2), *Dendronereis aestuarina (*av. 103 ind. /m^2*)* and *Prionospio cirrifera* (av. 46 ind. /m^2).

6.2.1.4 DNA barcoding of polychaete species

From the 19 species of polychaetes only 4 species were sequenced. The taxonomic status of the four polychaete species, *Prionospio cirrifera*, *Glycera alba*, *Micronephtyis oligobranchia*, and *Dendronereis aestuarina* were established and their DNA sequences from molecular analysis have been deposited in NCBI's GenBank under accession numbers KX098491- (*Prionospio cirrifera)*, KX098492 –(*Glycera alba)*, KX098493 (*Micronephthys oligobranchia)* and KU205267 (*Dendronereis aestuarina)*. *Dendronereis aestuarina* were the first submissions made to NCBI.

Table 6.1 Comparison of average polychaete species abundance (ind./m^2) during the open and close period of TMB in the southern zone of Vembanad estuary

	2011-2012		2012-2013	
	open	**close**	**open**	**close**
Dendronereis aestuarina	124	91	200	103
Namalycastis indica	328	1061	861	617
Parheteromastus tenuis	1	2	3	0
Mediomastus sp.	0	11	0	0
Prionospio cirrifera	107	0	238	46
Glycera alba	0	2	0	0
Owenia sp.	20	39	5	1

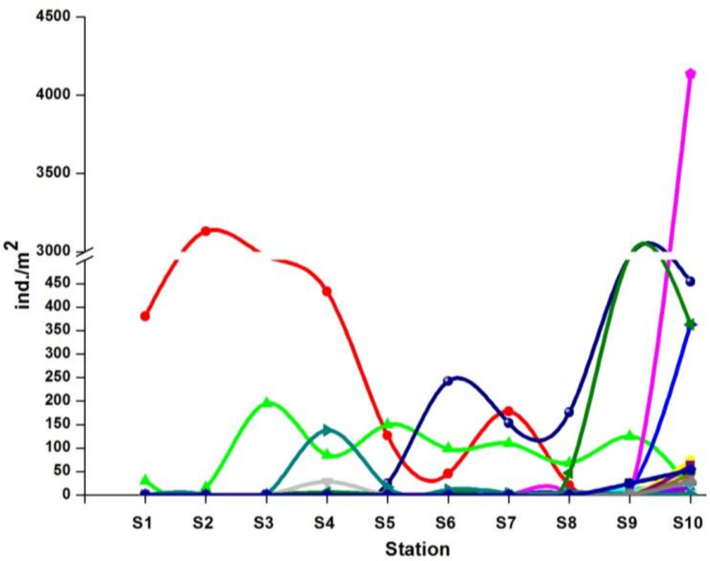


Fig. 6.1 Spatial distribution polychaete species in Vembanad estuary during 2011- 2012 period

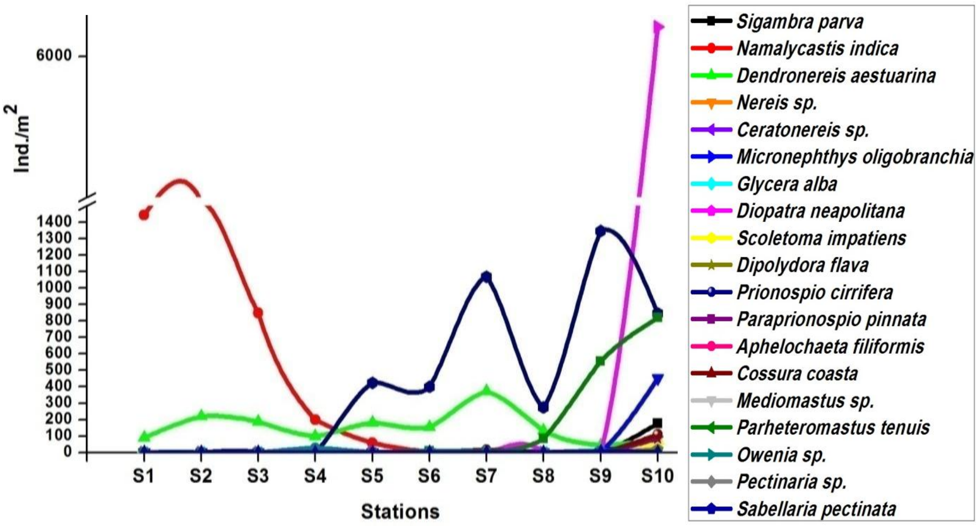
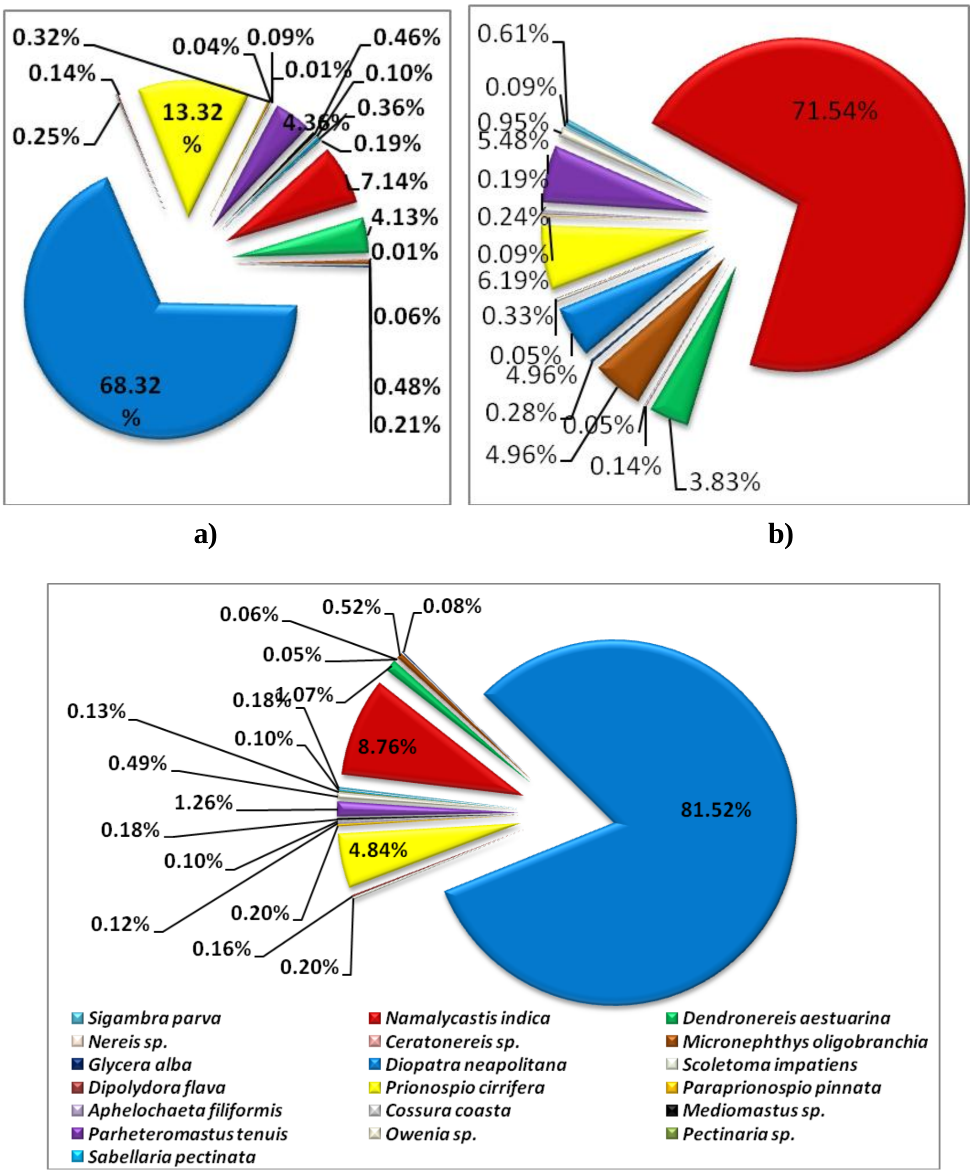
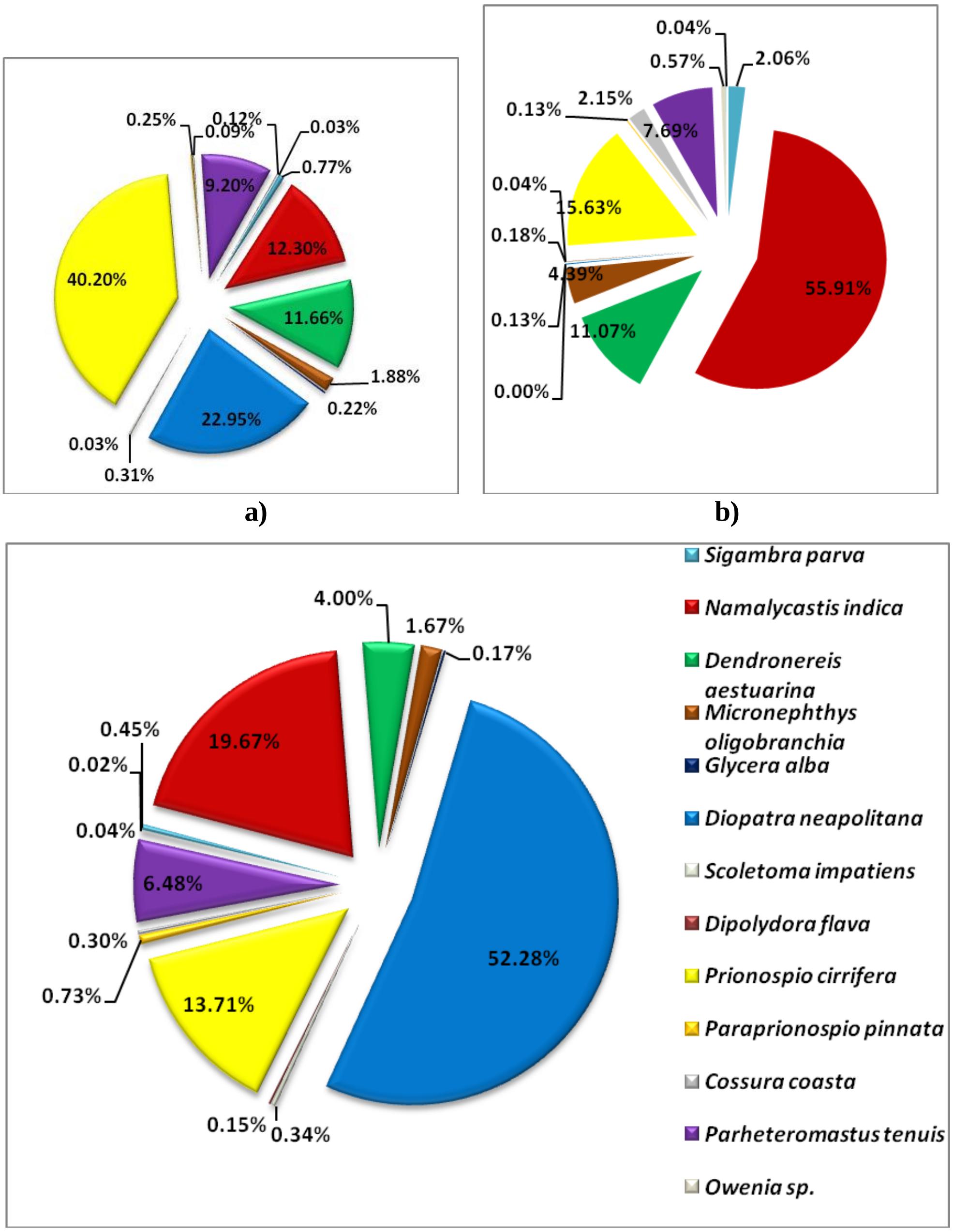


Fig. 6.2 Spatial distribution polychaete species in Vembanad estuary during 2012- 2013 period



c)

Fig. 6.3 Seasonal variation of percentage density of polychaete species in Vembanad during 2011 – 2012 period [a) Premonsoon b) Monsoon c) Postmonsoon]



c)

Fig. 6.4 Seasonal variation of percentage density of polychaete species in Vembanad estuary during 2012 – 2013 period [a) Premonsoon b) Monsoon c) Postmonsoon]

6.2.2 Data Analysis

6.2.2.1 Variation in diversity pattern and taxonomic distinctness of polychaetes in Vembanad estuary

The spatial abundance patterns of polychaetes were used for analyzing the diversity, richness, evenness index and are given in Fig. 6.5 and Fig. 6.6. During first year, the highest Shannon diversity was recorded in station 8 (1.8) and lowest in station 2 (0.04) with an annual average of 1.24. The polychaete species richness increased towards northern stations, where maximum values were observed in stations 10 (2.08) and minimum value in station 2 (0.12) with an annual average of 0.78. The evenness value was ranged from 0.04 to 0.81 with an average of 0.53, where the higher index value was recorded at station 7 and lower value at station 2. The Simpson dominance index was lower in station 7 (0.34) and highest in station 2 (0.99) with an average of 0.55. During second year the Shannon diversity index ranged between 0.32 (station 1) and 1.64 (station 10) with an average value of 1. Similar to first year, the richness index showed that maximum species richness was observed in station 10 (1.21) and minimum in station 2 (0.13) with an annual spatial average of 0.45. The evenness values varied between 0.32 (station 1) to 0.89 (station 8) with an average of 0.56. In the case of dominance index, the lowest value was observed in station 8 (0.41) and highest in station 1 (0.89) and its spatial average index was 0.60.

During first year taxonomic diversity index (Δ) of polychaetes varied from 0.2373 to 45.72 and total phylogenetic diversity index (sPhi+) varied from 125 to 975. Average Taxonomic Distinctness (Δ+) and Variation in Taxonomic Distinctness (Λ+) ranged between 25 to 70 and 0 to 510 respectively (Table 6.2). The 95% confidence funnel generated for the Average Taxonomic Distinctness (Δ+) and Variation in Taxonomic Distinctness (Λ+) values of all the stations is shown in Figure 6.7 and Figure 6.8. The Δ+ values of all the stations except stations 1 to 3 were found to fall within the main body of the simulated distribution, where stations 1 to 3 have lowest Δ+ value (25), denoting different diversity of polychaetes. Variation in Taxonomic Distinctness (Λ+) value was observed that, station 8 was not in the expected level, having highest Λ+ value (510). During second year taxonomic diversity index (Δ) of polychaetes varied from 2.738 to 44.07 and total phylogenetic diversity index (sPhi+) varied from 125 to 700. Average Taxonomic Distinctness (Δ+) and Variation in Taxonomic Distinctness (Λ+) ranged between 25 to 75 and 0 to 555.6 respectively (Table 6.2). The variation in taxonomic distinctness value and average taxonomic distinctness value of polychaetes was superimposed on the funnel to find out the deviation from the normal distribution (within the 95% confidence limit). The Δ+ value of polychaetes in stations 1 to 3 lying lower the 95% probability limit of the funnel and station 8 in the borderline also denoting significantly different diversity of polychaetes (Fig. 6.9). The funnel plot of Λ+ for all stations were not come together within the 95% confidence level represented that polychaete diversity was varied significantly among the stations (Fig. 6.10).

6.2.2.2 Species accumulation plot

Species accumulation plot shows the cumulative species count against sample number and a number of estimators. The curve denoted that whether data represented the larger population is suitably given the real patters in biodiversity. During the first year sampling, from the species estimators, the maximum number of species estimated by Sobs, Chao1, Chao2, Jacknife1, Bootstrap, MM and UGE were 19, 19, 32, 23, 21, 19 and 19 respectively (Fig. 6.11). During the second year sampling the maximum estimate was obtained from Chao2 (23), followed by Jacknife2 (20) and Jacknife1 (18) (Fig. 6.12).

6.2.2.3 K dominance plot

Multiple K- dominance plots was employed according to the polychaete species contribution in ten stations. In both the years, in the K- dominance plots the curve for stations 1 to 3 was lying above the other stations and showed the lowest diversity. In these stations *Namalycastis indica* and *Dendronereis aestuarina* were the only two polychaete species dominant during the study period. During first year the dominance curve of station 7 were found to lie below the other curves, denoting the absence of dominance of any polychaete species. The 'S' shaped curve of station 8 and station 10 indicating maximum diversity of polychaete species during both first (Fig. 6.13) and second (Fig. 6.14) year respectively.

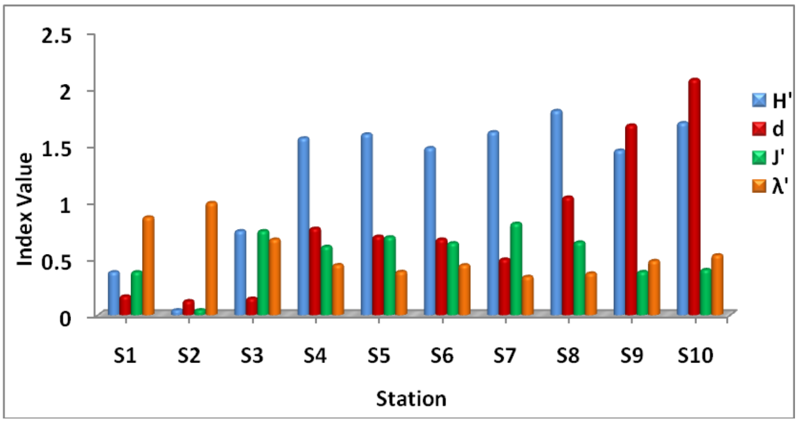


Fig. 6.5 Spatial variation of polychaete diversity indices in Vembanad estuary during 2011 – 2012 period

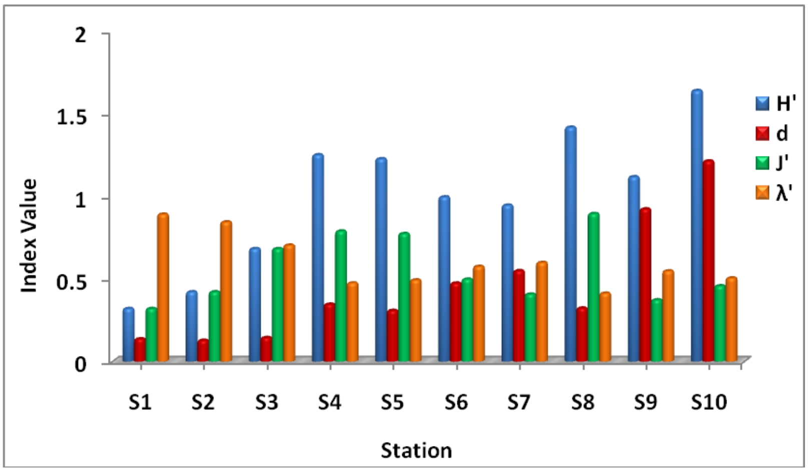


Fig. 6.6 Spatial variation of polychaete diversity indices in Vembanad estuary during 2012 – 2013 period

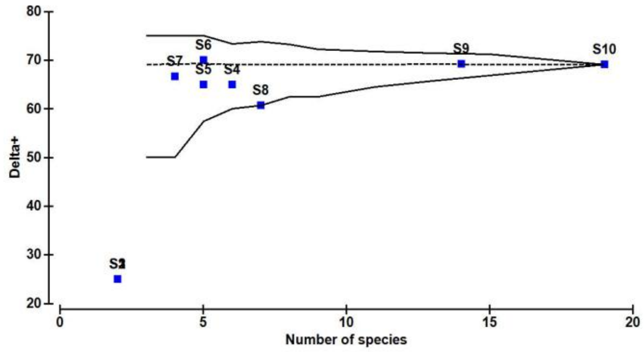


Fig. 6.7 The 95% confidence funnel for the average taxonomic distinctness (Delta+) values of polychaete diversity in Vembanad estuary during 2011 – 2012 period

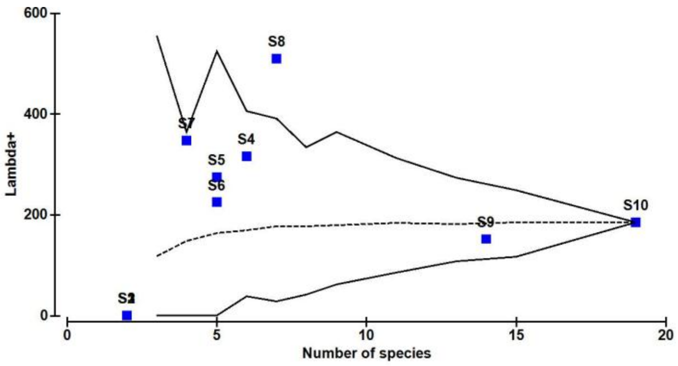


Fig. 6.8 The 95% confidence funnel for variation in taxonomic distinctness values (lambda+) of polychaete diversity in Vembanad estuary during 2011 – 2012 period

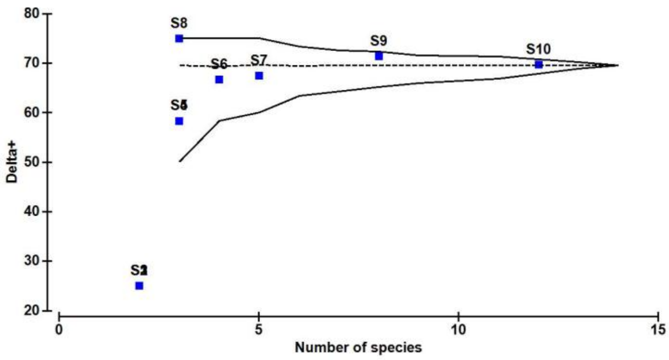


Fig. 6.9 The 95% confidence funnel for the average taxonomic distinctness (Delta+) values of polychaete diversity in Vembanad estuary during 2012 – 2013 period

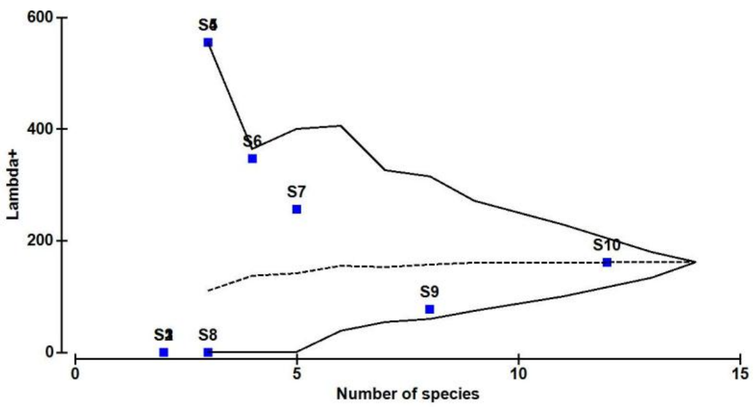


Fig. 6.10 The 95% confidence funnel for variation in taxonomic distinctness values (lambda+) of polychaete diversity in Vembanad estuary during 2012 – 2013 period

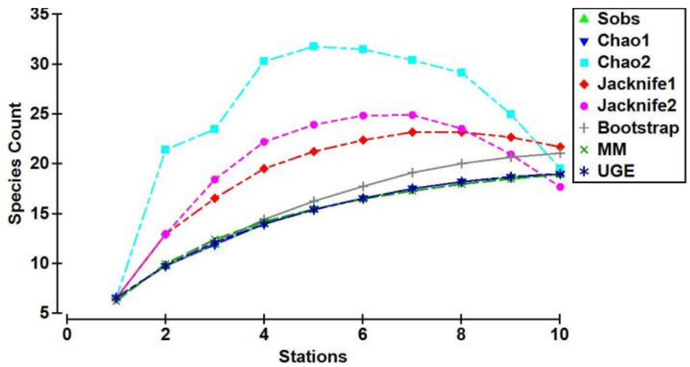


Fig. 6.11 Species estimators for polychaete species in the study stations of Vembanad estuary during 2011 – 2012 period

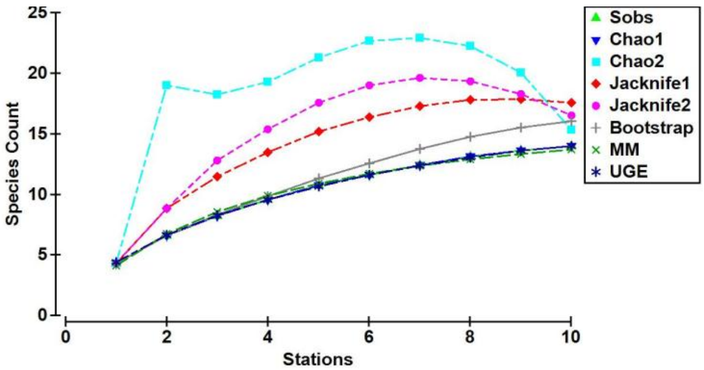


Fig. 6.12 Species estimators for polychaete species in the study stations of Vembanad estuary during 2012 – 2013 period

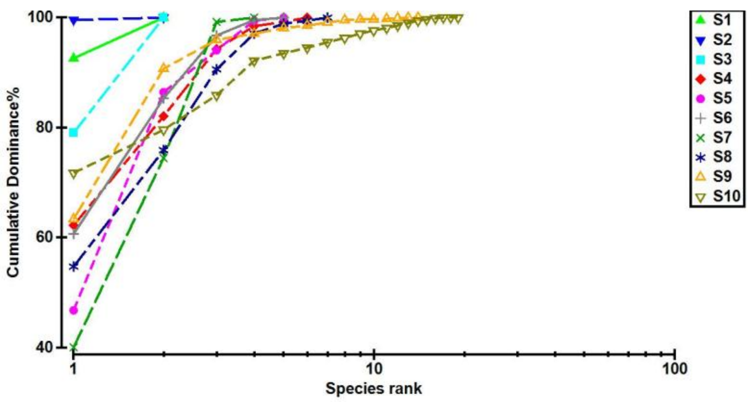


Fig. 6.13 K-dominance plot of polychaete species in Vembanad estuary during 2011-2012 period

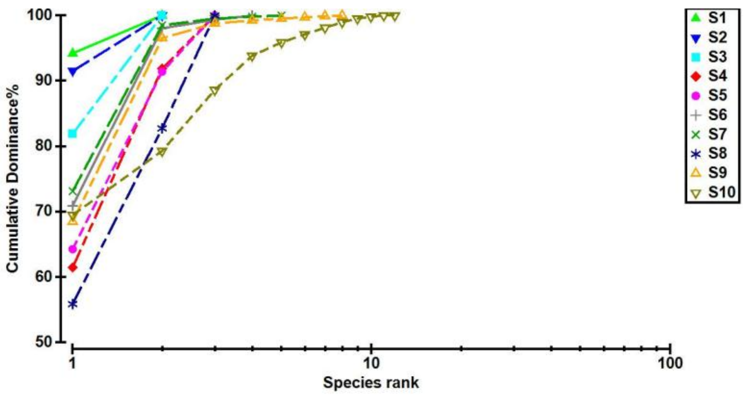


Fig. 6.14 K-dominance plot of polychaete species in Vembanad estuary during 2012-2013 period

Table 6.2 Diversity indices and taxonomic distinctness values of polychaete species in Vembanad estuary during 2011 – 2013 period

Station	Delta (Δ)		Δ* (Delta*)		Δ+ (Delta+)		Λ+ (Lambda+)		sPhi+	
	2011-2012	2012-2013	2011-2012	2012-2013	2011-2012	2012-2013	2011-2012	2012-2013	2011-2012	2012-2013
S1	3.399	2.738	25	25	25	25	0	0	125	125
S2	0.2373	3.899	25	25	25	25	0	0	125	125
S3	8.314	7.414	25	25	25	25	0	0	125	125
S4	33.87	20.51	60.72	39.18	65	58.33	316.7	555.6	350	200
S5	27.59	35.73	44.6	70.37	65	58.33	275	555.6	325	200
S6	39.02	31.79	69.99	74.55	70	66.67	225	347.2	350	275
S7	39.75	30.07	60.03	74.82	66.67	67.5	347.2	256.3	275	325
S8	45.72	44.07	72.27	75	60.71	75	510.2	0	400	250
S9	39.03	33.98	74.8	74.99	69.23	71.43	152.2	76.53	825	550
S10	34.62	36.69	73.49	74.03	69.15	69.7	185.1	161.3	975	700

6.2.2.4 Cluster and MDS analysis

Similarity of stations based on the abundance of polychaete species were established by Bray-Curtis and MDS analysis. Based on the polychaete density, during first year the Bray Curtis similarity index grouped the stations into two main clusters (Fig. 6.15). Cluster one further divided into two; in which the southern most stations, station 1 to 3 formed a cluster with 78.41% similarity. The dendrogram showed a clear pattern of grouping of stations on the south – north sector, in which northern most stations 9 and 10 showed a similarity of 63.87 %. Variations in the composition of the polychaete species assemblages between stations were also observed using non-metric multidimentional scaling (nMDS). The MDS stress value of 0.02 gives a valuable 2-dimensional plot representing an overall spatial similarity of 20 % (Fig. 6.16). From the MDS plot, it was observed that the distribution of polychaetes showed 80% similarity between stations 5, 6 and 7 and station 1 and 3 also showed 80% similarity. Except station 9 and 10, in all other stations the distribution of polychaete species showed an overall similarity of 40%. These separated stations have different polychaete composition when compared to other stations. During second year, from the resulting dendrogram it was able to group the stations into two main clusters (Fig. 6.17). Similar to first year, cluster one includes all stations except station 9 and 10 which formed cluster 2. Station wise ordination of MDS analysis exhibited a 2D stress value of 0.01 which showed that the results were credible (Fig. 6.18). Similar to first year, station 9 and 10 were dissimilar from other stations and showed 40% similarity with each other, in which the southern most stations 1 to 4 differentiated from stations 5 to 8.

6.2.2.5 Comparison of distribution of polychaetes in the southern and northern zone of Vembanad estuary

The polychaete species distribution on either side of southern and northern zone of TMB is given in Table 6.3.

Table 6.3 Polychaete species distribution in the south and north of TMB in Vembanad estuary during 2011-2013 period

Sl. No.	Polychaetes	South of TMB	North of TMB
1	*Sigambra parva*	-	+
2	*Namalycastis indica*	+	+
3	*Dendronereis aestuarina*	+	+
4	*Nereis* sp.	-	+
5	*Ceratonereis* sp.	-	+
6	*Micronephthys oligobranchia*	-	+
7	*Glycera alba*	+	+
8	*Diopatra neapolitana*	-	+
9	*Scoletoma impatiens*	-	+
10	*Dipolydora flava*	-	+
11	*Prionospio cirrifera*	+	+
12	*Paraprionospio pinnata*	-	+
13	*Aphelochaeta filiformis*	-	+
14	*Cossura coasta*	-	+
15	*Mediomastus* sp.	+	+
16	*Parheteromastus tenuis*	+	+
17	*Owenia* sp.	+	+
18	*Pectinaria* sp.	-	+
19	*Sabellaria pectinata*	-	+

(Note: '+' denote present; '-' denote absent)

Most of the polychaete species recorded during the study was euryhaline. During first year the ANOSIM revealed that there was a significant spatial variation of polychaete species between southern and northern zones (ANOSIM Global R=0.409, P <0.05). SIMPER analyses revealed that the polychaete population from southern and northern zones varied from each other mainly due to the dissimilarity in species abundance and composition. In the southern and northern zone of estuary, the mean similarity of polychaete species was 62.34 % and 50.13 % respectively and the dissimilarity percentage was 57.02 % (Table 6.4). The polychaete species generally answerable for the noted variation between both sectors were *Prionospio cirrifera*, *Namalycastis indica* and *Parheteromastus tenuis* and having mean dissimilarity contribution of 10.15 %, 7.03 % and 6.82 % respectively. During second year, the ANOSIM results indicated that a significant variation was observed in the polychaete distribution on the southern and northern sites (Global R=0.683, $P < 0.01$). SIMPER analysis elucidate that, in the southern and northern zones of estuary, the mean similarity of polychaete species was 66.79 % and 52.71 % respectively and the dissimilarity percentage of 63.31 % (Table 6.5). Similar to first year the polychaete species such as, *Namalycastis indica*, *Prionospio cirrifera* and *Parheteromastus tenuis* were mainly responsible for the dissimilarity between southern and northern zone, contributing dissimilarity in percentage of 14.06 %, 13.36 % and 10.63 % respectively.

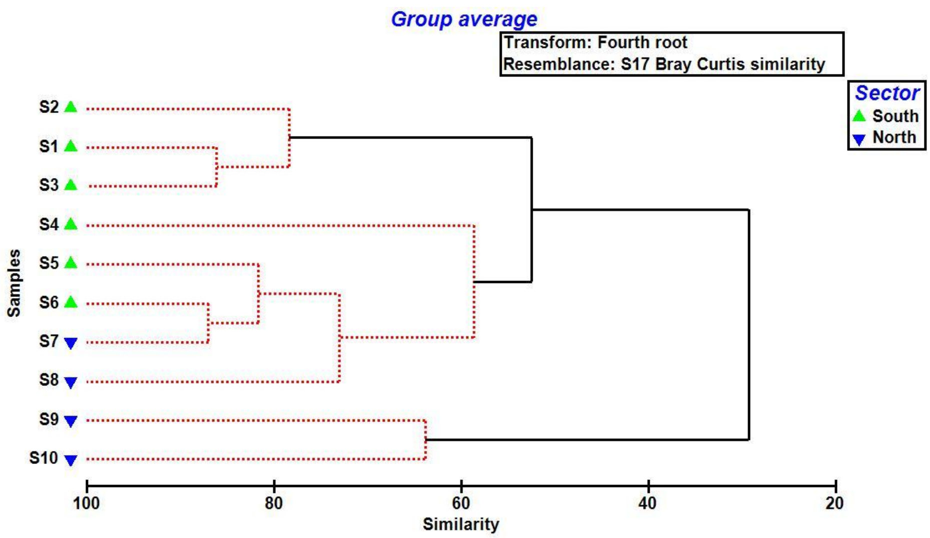


Fig. 6.15 Dendrogram showing the spatial similarities of polychaete species abundance in the Vembanad estuary during 2011 – 2012 period

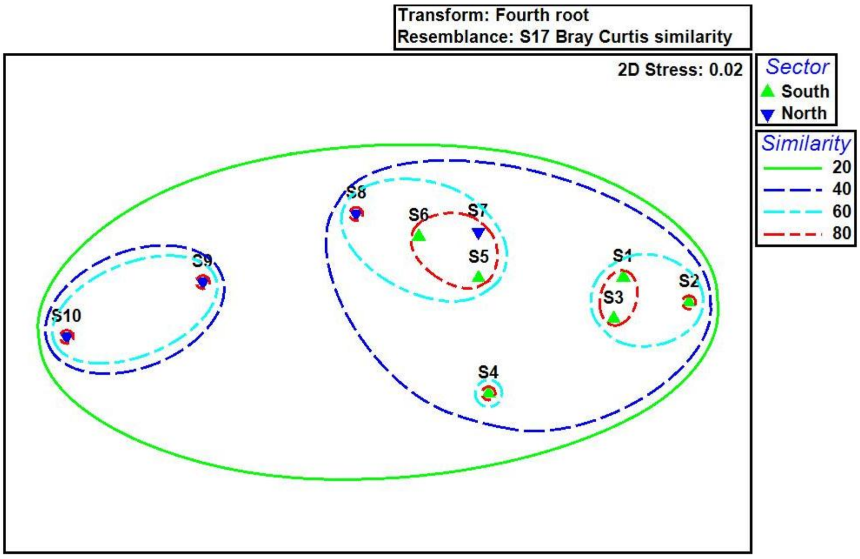


Fig. 6.16 MDS plot of polychaete species similarity in Vembanad estuary during 2011-2012 period

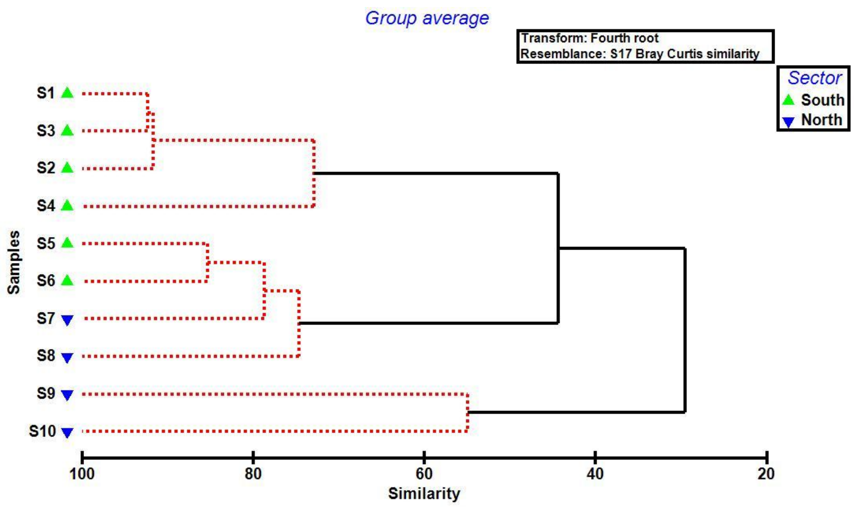


Fig. 6.17 Dendrogram showing the spatial similarities of polychaete species abundance in the Vembanad estuary during 2012 – 2013 period

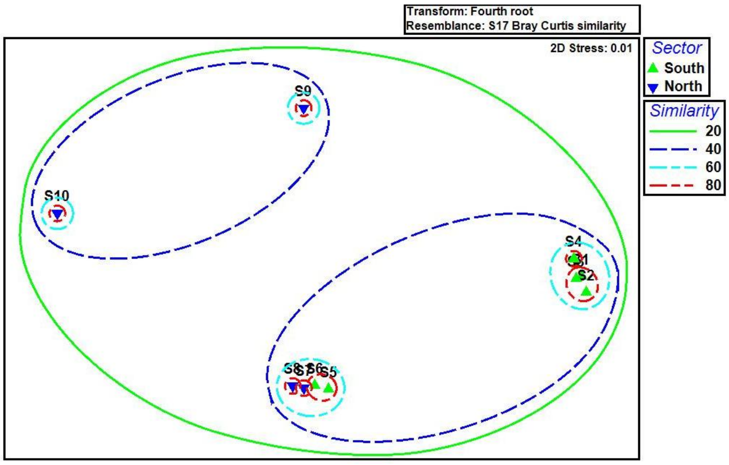


Fig. 6.18 MDS plot of polychaete species similarity in Vembanad estuary during 2012-2013 period

Table 6.4 SIMPER test results showing the dissimilarity of polychaete species between southern and northern sector of TMB during 2011-2012 period

2011-2012	Average dissimilarity = 57.02 %				
	Group South	**Group North**			
Species	**Average Abundance**	**Average Abundance**	**Average Dissimilarity**	**Percentage Contribution (%)**	**Cumulative percentage (%)**
Prionospio cirrifera	1.03	4.5	10.15	17.8	17.8
Namalycastis indica	4.6	2.2	7.03	12.32	30.12
Parheteromastus tenuis	0.45	3.01	6.82	11.96	42.08
Owenia sp.	1.22	1.32	3.71	6.51	48.59
Diopatra neapolitana	0	2	3.09	5.43	54.02
Micronephthys oligobranchia	0	1.55	2.77	4.85	58.87
Ceratonereis sp.	0	1.08	2.58	4.53	63.4
Sabellaria pectinata	0	1.23	2.35	4.13	67.52
Glycera alba	0.45	1.02	2.34	4.11	71.63
Sigambra parva	0	1.22	2.33	4.09	75.72
Nereis sp.	0	0.84	1.97	3.46	79.18
Dendronereis aestuarina	2.96	2.86	1.92	3.37	82.55
Scoletoma impatiens	0	1.03	1.82	3.19	85.74
Aphelochaeta filiformis	0	0.88	1.6	2.8	88.54
Cossura coasta	0	0.87	1.58	2.77	91.32

Table 6.5 SIMPER test results showing the dissimilarity of polychaete species between southern and northern sector of TMB during 2012-2013 period

2012-2013	Average dissimilarity = 63.31 %				
	Group South	Group North			
Species	Average Abundance	Average Abundance	Average Dissimilarity	Percentage Contribution (%)	Cumulative percentage (%)
Namalycastis indica	4.4	0.35	14.06	22.21	22.21
Prionospio cirrifera	1.5	5.31	13.36	21.1	43.31
Parheteromastus tenuis	0.28	3.66	10.63	16.79	60.1
Diopatra neapolitana	0	2.2	4	6.31	66.41
Cossura coasta	0	1.3	3.45	5.45	71.86
Micronephthys oligobranchia	0	1.44	3.04	4.8	76.67
Scoletoma impatiens	0	1.02	2.36	3.72	80.39
Owenia sp.	0.38	0.44	2.3	3.63	84.02
Dendronereis aestuarina	3.49	3.32	2.26	3.56	87.59
Glycera alba	0	0.93	2.24	3.54	91.13

6.2.2.6 Relation between polychaete species distribution and environmental factors

6.2.2.6.1 Correlation analysis

The Pearson's correlation co-efficient was used to study the relationship between polychaete species distribution and environmental variables. The results of correlation analysis for first year and second year are given in annexure 6.3 and annexure 6.4 respectively.

6.2.2.6.2 BIOENV analysis of polychaete community

BIO-ENV analyses were applied to analyze polychaetes community structure and their relationships with environmental variables. During first year based on the BIO-ENV analysis, of bottom water parameters and polychaete species, the best combination of variables to explain the biological data (ρ = 0.851) was obtained from the combination of five variables including water pH, water temperature, dissolved oxygen, salinity and alkalinity (Table 6.6 and Fig. 6.19). Similarly among the sediment parameters a combination of four variables was responsible for the polychaete species distribution (ρ = 0.659), from which parameters such as sediment temperature, sand, silt and clay fraction formed the major combinations (Table 6.7 and Fig. 6.20). During second year also the relationships of corresponding environmental and biotic parameters were analyzed (Table 6.8 and Fig. 6.21.; Table 6.9 and Fig. 6.22). No single water quality parameter except salinity (ρ = 0.883), denoted any strong correlation between polychaete community structure. Among the water quality parameters a combination of temperature, pH and salinity best explained the community structure of polychaete species (ρ = 0.937). In the case of sediment parameters, a combination of silt and clay was mainly responsible for the abundance and distribution of polychaete species (ρ = 0.714).

Table 6.6 Results of BIOENV analysis for polychaete species and water quality parameters in Vembanad estuary during 2011 – 2012 period

Sl. No.	Variables	Variables Selected	Correlation Values (Rho)
1	Water pH	1-4, 7	0.851
2	Water temperature	1, 2, 4, 7	0.838
3	Salinity	1-4	0.835
4	Dissolved oxygen	2-4, 6, 7	0.833
5	Silicate-silicon	1-4, 6	0.831
6	Phosphate-phosphorus	2-4, 7, 9	0.830
7	Alkalinity	2-4, 7	0.826
8	Chlorophyll *a*	1, 2, 4, 6, 7	0.826
9	Depth	1-3	0.825
10	Dissolved inorganic nitrogen	1-4, 8	0.825

Table 6.7 Results of BIOENV analysis for polychaete species and sediment quality parameters in Vembanad estuary during 2011 - 2012 period

Sl. No.	Variables	Variables Selected	Correlation Values (Rho)
1	Sediment temperature	1, 6-8	0.659
2	Sediment pH	1, 6, 8, 9	0.655
3	Moisture content	1, 3,6, 8	0.654
4	Organic carbon	1, 6-9	0.653
5	Available nitrogen	1, 3, 6, 8, 9	0.650
6	Sand	1, 6, 8	0.647
7	Silt	1, 5, 6, 8	0.647
8	Clay	1, 8, 9	0.642
9	Available phosphorus	1, 6, 7	0.639

Table 6.8 Results of BIOENV analysis for polychaete species and water quality parameters in Vembanad estuary during 2012 - 2013 period

Sl. No.	Variables	Variables Selected	Correlation Values (Rho)
1	Water pH	1, 2,9	0.937
2	Salinity	2, 5, 9	0.932
3	Dissolved oxygen	1, 2, 5, 9	0.914
4	Phosphate-phosphorus	1, 2, 4, 9	0.884
5	Alkalinity	2	0.883
6	Chlorophyll *a*	1-3, 5, 9	0.88
7	Depth	1-3, 9	0.871
8	Silicate-silicon	2, 3, 5, 9	0.869
9	Water temperature	2, 9	0.868
10	Dissolved inorganic nitrogen	1, 2	0.861

Table 6.9 Results of BIOENV analysis for polychaete species and sediment quality parameters in Vembanad estuary during 2012 - 2013 period

Sl. No.	Variables	Variables Selected	Correlation Values (Rho)
1	Sediment temperature	7, 8	0.714
2	Sediment pH	6-8	0.683
3	Moisture content	6,8	0.650
4	Organic carbon	2, 6-8	0.650
5	Available nitrogen	2, 7, 8	0.641
6	Sand	3, 7, 8	0.64
7	Silt	3, 6-8	0.639
8	Clay	1, 6-8	0.623
9	Available phosphorus	5-8	0.615

6.2.2.6.3 CCA analysis of polychaete community

CCA analysis showed that environmental parameters had a strong control over the distribution of polychaete species in the Vembanad estuarine system. By observing the relationship of polychaetes and bottom water quality parameters, the result of CCA analysis revealed that, the first axis explained 50.21 % of total variance followed by second (32.29 %) and third axis (10.57 %) with eigen values of 0.873, 0.562 and 0.184 respectively. Monte Carlo permutation test specified that water temperature (r = 0.96), TDS (r = 0.88), conductivity (r = 0.84) and salinity (0.82) was negatively correlated with Axis 1 and dissolved oxygen (0.63) was positively correlated with Axis 2 (Fig. 6.23). During first year, the first 3 axis of CCA triplot explained 99.66 % of total variability of sediment parameters with eigen values of the first three axis having 0.869, 0.559 and 0.178 respectively, in which the first axis that explained 52.32 % of the total variance and the second and third axis represented 33.67 % and 10.74 % of the total variance respectively (Fig. 6.24). When considering the CCA analysis of the influence of sediment parameters on the polychaete community, the parameters such as sediment temperature (r = - 0.91), available nitrogen (r = 0.63), pH (r = - 0.6), sodium (r = - 0.59), Eh (r = - 0.57), organic carbon (r = 0.52) were most important.

The CCA analysis during second year showed that first two CCA axis explained 57.79 % and 34.66 % of variability of sediment parameters with eigen values of the first two axis having 0.869 and 0.52 respectively. Whereas in the case of water quality parameters first two axis having eigen values of 0.855 and 0.519 and it explained 57.34 % and 34.79 % of the variability. Similar to first year, the first CCA axis negatively correlated with water temperature (r = - 0.848), salinity (r = - 0.82), TDS (r = - 0.74) and conductivity (r = - 0.72) and second axis positively correlated with DO (r = 0.64) (Fig. 6.25). In the case of sediment parameters, such as silt fraction (r =0.79), available nitrogen (r = 0.78), organic carbon (r = 0.71) and moisture content (r = 0.70) exhibited positive correlation with axis 1, whereas sand fraction (r = 0.63), Eh (r = 0.72) and pH (r = 0.6) having negative correlation with axis 1.The second axis having strong positive correlation with sand (r = 0.95) and temperature (r =0.60) and negative correlation with clay (r =0.81) and silt (r =0.87) (Fig. 6.26).

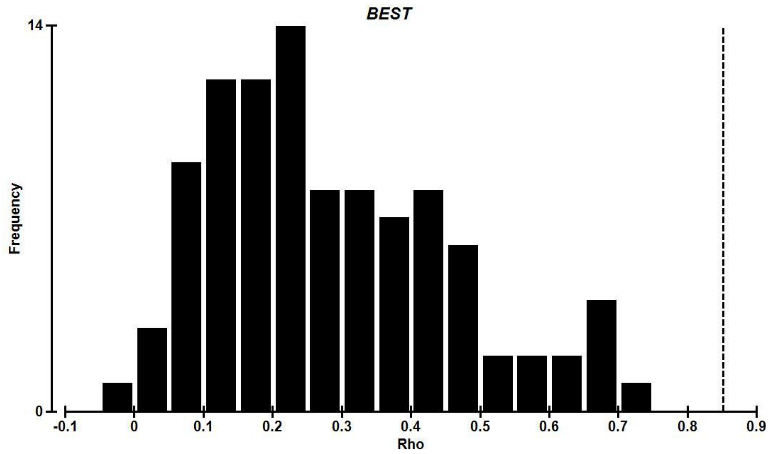


Fig. 6.19 Histogram showing the results of BEST analysis of polychaetes with water quality parameters in Vembanad estuary during 2011-2012 period.

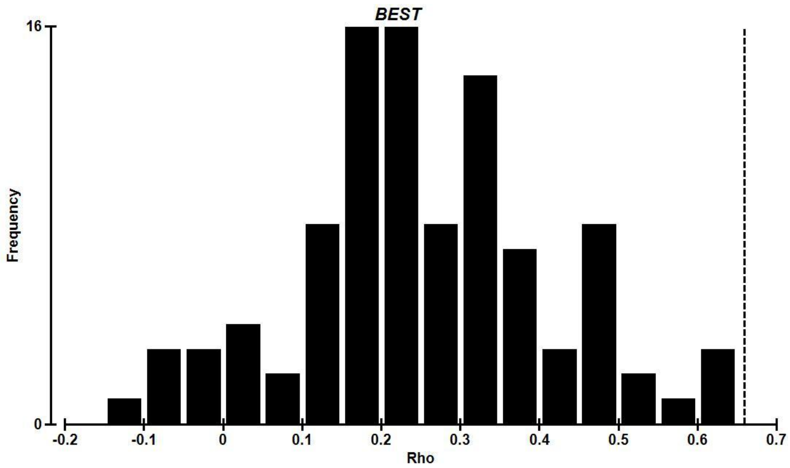


Fig. 6.20 Histogram showing the results of BEST analysis of polychaetes with sediment parameters in Vembanad estuary during 2011-2012 period.

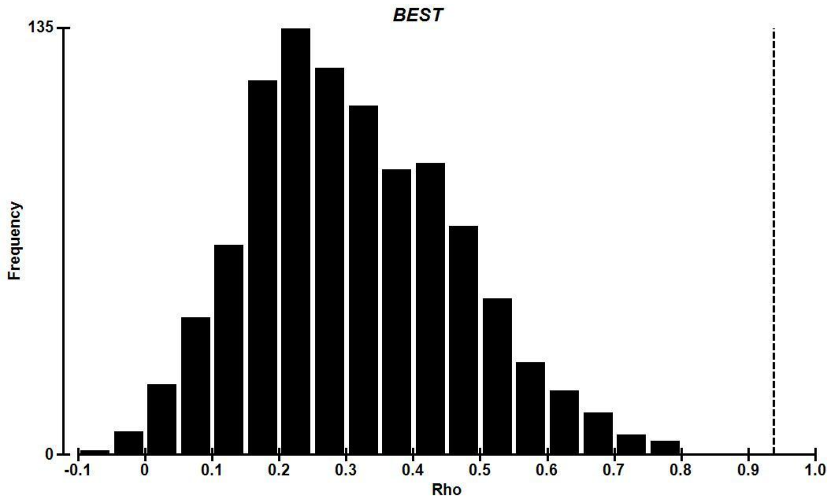


Fig. 6.21 Histogram showing the results of BEST analysis of polychaetes with water quality parameters in Vembanad estuary during 2012-2013 period.

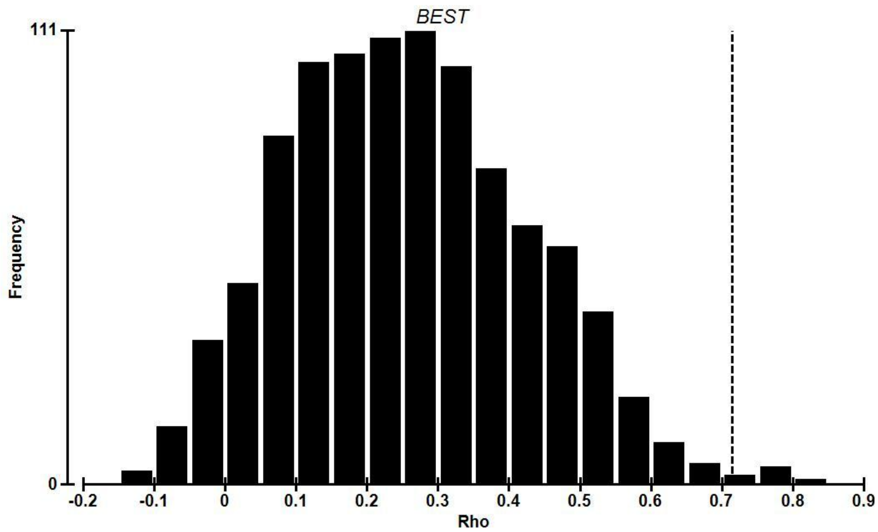


Fig. 6.22 Histogram showing the results of BEST analysis of polychaetes with sediment parameters in Vembanad estuary during 2012-2013 period.

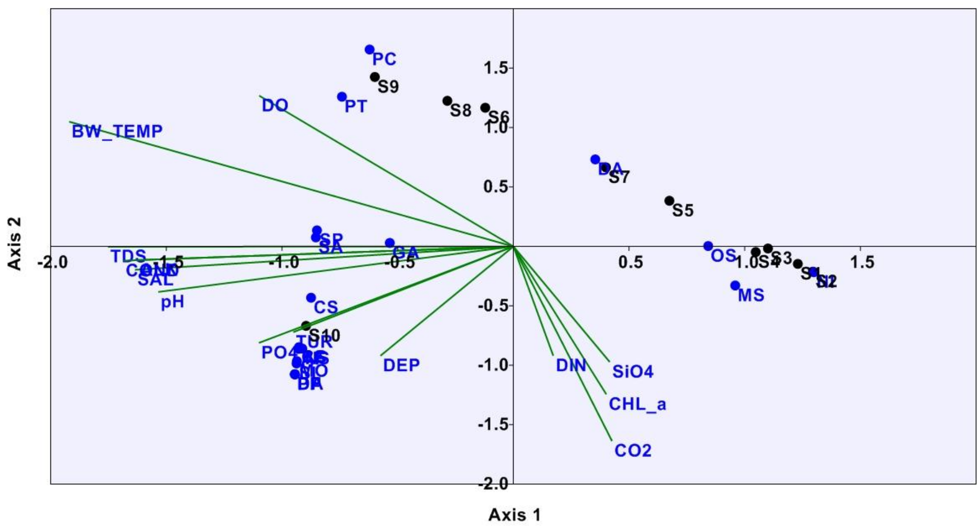


Fig. 6.23 CCA analysis of polychaete species and water quality parameters in Vembanad estuary during 2011 – 2012 period

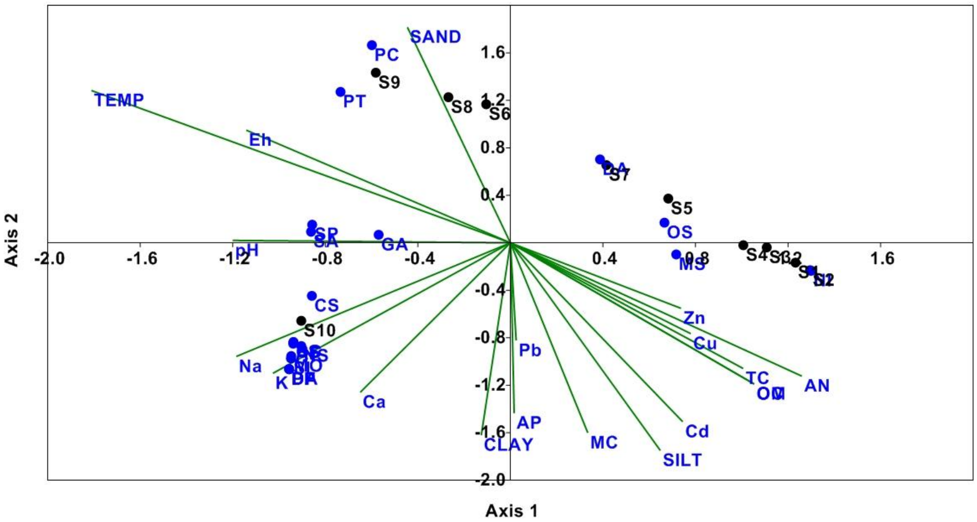


Fig. 6.24 CCA analysis of polychaete species and sediment parameters in Vembanad estuary during 2011 – 2012 period

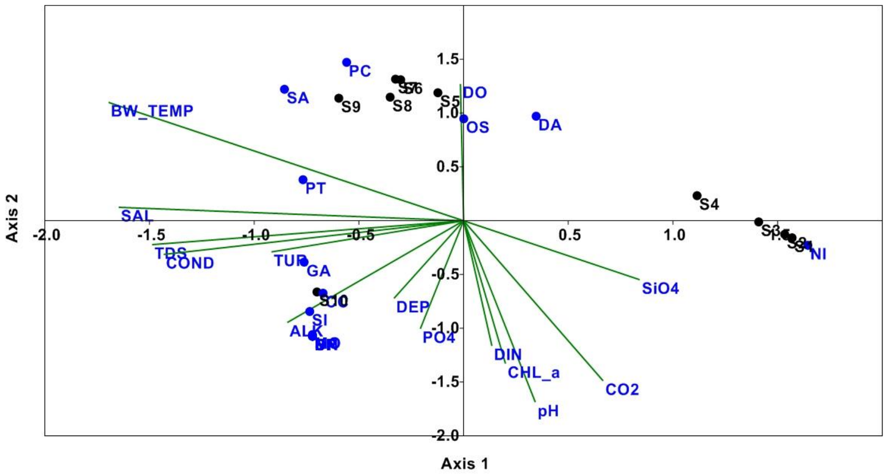


Fig. 6.25 CCA analysis of polychaete species and water quality parameters in Vembanad estuary during 2012 – 2013 period

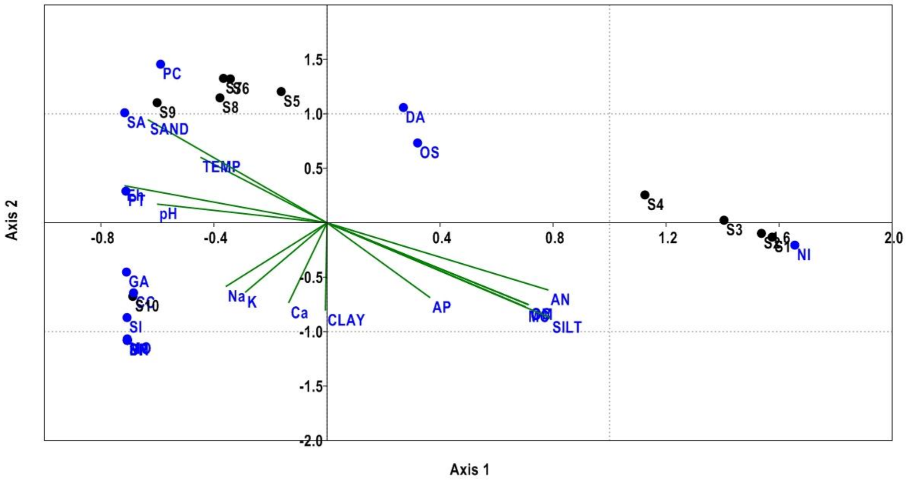


Fig. 6.26 CCA analysis of polychaete species and sediment parameters in Vembanad estuary during 2012 – 2013 period

6.3 Discussion

From the 33 estuaries on the east coast and 34 on the west coast of India, the information on polychaetes are available from 8 and 4 estuaries of east coast and west coasts respectively (Ajmal Khan and Murugesan, 2005). They noted a total of 153 species of polychaetes, contributing 37.46 % of the total species from the Indian coast. From the one hundred and forty two polychaete species were reported from east coast estuaries, whereas one hundred and nineteen species were reported from east coast, 11 species from west coast and 23 species reporting both the coasts. From the estuaries of east coast, the maximum number of polychaete species were identified from Vellar estuary as 98; in Hoogli-Matla estuary - 69; in Vasista—Godavari estuary - 44; in Coleroon estuary – 37; in Mahanadhi estuary - 33 species. In west coast, 19 species were recorded from Cochin estuary, 10 species from Mandovi and Zuari estuarine complex, 7 species from Ashtamudi estuary and 5 species from Mulki estuary (Ajmal Khan and Murugesan, 2005). A total of 19 species of polychaetes belonging to 13 families and 19 genera were observed in the present investigation (Plate 6.1), in which only 7 species were observed in the southern zone. In the northern part of Vembanad estuary the species composition of polychaetes were varied in each of the studies. During 1977, Pillai reported 32 species of polychaetes, in which *Ancistrosyllis constricta*, *Lumbriconereis simplex* and *Nephthys oligobranchiata* were common and *P. polybranchiata* was the dominant species. In Cochin estuary Remani et al. (1983), observed a total of 20 and 11 species of polychaetes in the municipal discharge site and coconut husk retting zone respectively. In the northern limb of Vembanad estuary Sarala Devi et al. (1991a) observed a total of 30 species of polychaetes, in which *Capitella capitata, D. aestuarina, N. indica and P. tenuis* more abundant. They also found high densities of *C. capitata* in the effluent discharge site. Sheeba (2000) reported 54 species of polychaetes from 19 families in Cochin estuary. Among the 54 species of polychaetes, *C. capitata* was common in all the stations followed by *D. aestuarina, P. polybranchiata, H. bifidus* and *P. tenuis.* John (2009) reported 32 species of polychaetes in Cochin estuary and noted an increasing colonization of *P. cirrifera, P. tenuis* and *Heteromastus bifidus* in the stressed stations of Cochin estuary. During 1996- 1997 period, in the southern zone of Vembanad estuary 10 species of polychaetes were observed (Anon, 2001), in which *Dendronereis* sp., D. *aestuarina*, D. *arborifera* were abundant. From Cochin estuary Martin et al. (2011) observed 19 species of polychaetes during 2005 period. They observed that in the organically enriched central part of Cochin estuary, the tolerant species such as *C. capitata, P. tenius* and *Heteromastus similis* were abundant and showed a wider distribution. In the present study, *N. indica* and *D. aestuarina* were the common polychaete species in the southern part of Vembanad estuary (Table 6.10). These two species seems to live in waters of lower salinity in the south than they do in the north. These animals occurring in water in a dense substrate composed of sand mixed with various percentage combinations of clay, and vegetable debris mainly the decaying water hyacinth *Eichhornia crassipes.* According to

Sheeba (2000), *D. aestuarina* and *N. indica* depend upon the dissolved oxygen concentration and particulate organic carbon. In the present investigation higher sediment organic carbon was observed in the Punnamada region, where *D. aestuarina* and *N. indica* were the only two polychaete species seen. The study found that there was a radical shift in the polychaete species distribution in the southern zone of TMB. Time scale study found that since 1981, there was a drastic variation in the occurrence of polychaete species in Cochin estuary. Several polychaete species, which were reported earlier was not observed in the present study (Table 6.11). During 1996-1997 period there were 10 polychaete species such as *Dendronereis* sp., D. *aestuarina*, D. *arborifera*, *Nereis chingrihanensis*, *Ceratonereis mirabilis*, *Ceratonereis* sp., *Prionospio cirrifera*, *Prionospio* sp. and *Capitellid* sp. were observed in the southern zone (Anon, 2001) where it reduced to seven species now. Apart from D. *aestuarina* and *Prionospio cirrifera*, the remaining species were not observed in the current study. During the present study, the southern zone was characterized by the dominance of *N. indica.* According to Glasby (1999), the genus Namalycastis was mainly species rich within *Namanereidinae* and found in coastal polluted regions. The species was mainly adapted to live in the semi-terrestrial habitat with low saline areas and also coupled with the decaying plant matter with high organic materials (Magesh et al., 2012). Similar condition was existed in the southern sector of Vembanad estuarine system, where the decaying and decomposing dead water hyacinth formed a thick mat on the floor of southern stations and tolerant of a wide range of salinities in Vembanad estuarine system. The increased organic carbon in the southern stations may not support more polychaete species. Similar type of observations was reported in the Vishakhapatnam harbour area (Musale et al., 2015). According to Harkantra et al. (1982) low abundance of polychaetes in areas with greater than 3% organic matter indicates its evasion for high organic matter.

Table 6.10 Spatial distribution of polychaete species in Vembanad estuary during 2011-2013 period

Sl. No.	Polychaete species	S1	S2	S3	S4	S5	S6	S7	S8	S9	S10
1	*Sigambra parva*	-	-	-	-	-	-	-	-	+	+
2	*Namalycastis indica*	+	+	+	+	+	+	+	+	+	+
3	*Dendronereis aestuarina*	+	+	+	+	+	+	+	+	+	+
4	*Nereis* sp.	-	-	-	-	-	-	-	+	-	+
5	*Ceratonereis* sp.	-	-	-	-	-	-	-	+	+	+
6	*Micronephthys oligobranchia*	-	-	-	-	-	-	-	-	+	+
7	*Glycera alba*	-	-	-	+	+	-	-	-	+	+
8	*Diopatra neapolitana*	-	-	-	-	-	-	-	-	-	+
9	*Scoletoma impatiens*	-	-	-	-	-	-	-	-	+	+
10	*Dipolydora flava*	-	-	-	-	-	-	-	-	-	+
11	*Prionospio cirrifera*	-	-	-	-	+	+	+	+	+	+
12	*Paraprionospio pinnata*	-	-	-	-	-	-	-	-	-	+
13	*Aphelochaeta filiformis*	-	-	-	-	-	-	-	-	+	+
14	*Cossura coasta*	-	-	-	-	-	-	+	-	+	+
15	*Mediomastus* sp.	-	-	-	+	-	-	-	-	-	+
16	*Parheteromastus tenuis*	-	-	-	+	-	+	+	+	+	+
17	*Owenia* sp.	-	-	-	+	+	+	+	+	+	+
18	*Pectinaria* sp.	-	-	-	-	-	-	-	-	+	+
19	*Sabellaria pectinata*	-	-	-	-	-	-	-	-	+	+

(Note: '+' denote present; '-' denote absent)

Large number of polychaete species *D. neapolitana* was observed in the muddy or clayey sand area of Cochin estuary (Kurian, 1972) which was comparable to the current study. The massive aggregations of tube dwelling polychaetes, *D. neapolitana* were observed in station 10 (Plate 6.2). These aggregation of tube structures of *D. neapolitana* were absent during MN season and it was possible that *Diopatra* species is sensitive to the sediment disturbance during MN mainly due to under water movement and associated churning process in the sediment. Range shift, species diversity variation and biogenic reef structure variation observed in ecosystem engineer polychaete species, *Diopatra* might be due to tidal currents, storm tides and other environmental factors (Berke, 2010; Vorberg, 2000). By changing the geochemical properties of sediment, occurrence of *Diopatra cuprea* play a great role in controlling the distribution of sub-surface feeding species (Gaston, 1987). The tube structures provide refuge from predation and so increasing the infaunal density (Woodin, 1978). Such observation was obtained in the present investigation. In the Sundarban Biosphere Reserve on the northeast coast of India, Sarkar et al. (2005) observed the formation of mudflats by continuous silting in the 'Diopatra' zone. Similarly Badve (1996) observed tube aggregates of reef building polychaete worm, *Sabellaria* along the west coast of India. Bailey-Brock, (1984) reported the occurrence of large number of tube- building polychaete *Diopatra leuckatri* in Hawaiian reef-Bat, giving stable environment and provide abundant resources for the diverse invertebrate community. Sanchez et al. (2011) reported that massive polychaete reefs as indicator of both increase sewage-contamination and chlorination process in Argentina. According to Bailey-Brock, (1984) the packing of sediments provides substratum strength for tube dwellers and burrowers, and the high organic content of the trapped material serves as a food source for selective and burrowing detritivores. Consequently a community has developed around the *Diopatra* tubes taking advantage of these resources. In the present investigation a total of 11 species of polychaetes were associated with these *Diopatra* structure and all these species are considered as classical indicators of disturbance reacting to organic enrichment.

Table 6.11 Time scale changes in polychaete species from 1980 to present in Vembanad estuary especially Cochin estuarine area.

		Study period					
		1981	1996-1997	1994-1996	2002-2004	2005	201-2013
Sl. No.	Polychaetes	(Sarala Devi et al., 1991a)	(Sheeba, 2000)	(Anon, 2001) South of TMB only	(John, 2009)	(Martin et al., 2011)	(Present study)
1	*Aphrodita alta*	+	+	-	-	+	-
2	*Lepidonotus* sp.	-	+	-	-	-	-
3	*Sthenelais boa*	+	+	-	-	-	-
4	*Sthenelais* sp.	-	-	-	+	-	-
5	*Amphinome rostrata*	-	+	-	-	-	-
6	*Ancistrosyllis constricta*	+	+	-	+	+	-
7	*Sigambra parva*	-	-	-	-	-	+
8	*Vanadis formosa*	-	+	-	-	-	-
9	*Syllis spongicola*	-	+	-	-	-	-
10	*Syllis spongicola*	-	-	-	+	-	-
11	*Scoloplos madagascariensis*	-	+	-	-	-	-
12	*Namalycastis indica*	+	+	-	+	+	+
13	*Dendronereis aestuarina*	+	+	+	+	+	+
14	*Dendronereis* sp.	-	-	+	-	-	-
15	D. *arborifer*a	-	-	+	-	-	-
16	*Perinereidae*	-	+	-	-	-	-
17	*Perinereis cavifrons*	+	+	-	-	+	-
18	*Platynereis* sp.	-	+	-	-	-	-
19	*Nereis chingrihanensis*	-	-	+	-	-	-
20	*Nereis* sp.	-	-	-	+	-	+
21	*Ceratonereis mirabilis*	-	-	+	-	-	-
22	*Ceratonereis* sp.	-	-	+	-	-	+
23	*Nephthys dibranchis*	-	+	-	-	+	-
24	*Nepthys polybranchia*	+	-	-	-	+	-
25	*Micronephthys oligobranchia*	+	-	-	+	-	+
26	*Marphysa mossambica*	-	+	-	-	-	-
27	*Diopatra neapolitana*	+	+	-	+	+	+

28	*Lumbrinereis simplex*	+	+	-	+	-	-
29	*L. polydesma*	-	+	-	-	-	-
30	*L. latreilli*	-	+	-	+	+	-
31	*Scoletoma impatiens*	-	+	-	-	-	+
32	*L. notocirrata*	+	+	-	-	-	-
33	*L. pseudobifiliaris*	+	-	-	-	-	-
34	*Gonida emerita*	+	+	-	-	+	-
35	*G. inceria*	-	+	-	-	-	-
36	*Goniadopsis maskallensis*	-	+	-	-	-	-
37	*Glycera longipinnis*	-	+	-	-	-	-
38	*G. alba*	+	+	-	-	-	+
39	*G. benguellana*	-	+	-	-	-	-
40	*Glycera convoluta*	+	-	-	-	+	-
41	*Glycera* sp.	-	+	-	-	-	-
42	*Glycera trydactyla*	-	-	-	+	-	-
43	*Scolelepis indica*	-	+	-	-	-	-
44	*Prionospio pinnata*	+	+	-	+	-	+
45	*P. cirrifera*	-	+	+	+	-	+
46	*P.cirrobranchiata*	-	+	-	-	+	-
47	*Prionospio* sp.	-	+	+		-	-
48	*Prionospio polybranchiata*	+	-	-	+	+	-
49	*Polydora kempi*	-	-	-	+	-	-
50	*Dipolydora flava*	-	-	-	-	-	+
51	*Disoma orissae*	-	+	-	-	-	-
52	*Magelona capensis*	-	+	-	-	-	-
53	*Cossura coasta*	-	+	-	+	+	+
54	*Aphelochaeta filiformis*	-	-	-	-	-	+
55	*Capitella capitata*	+	+	-	+	+	-
56	*Capitellid* sp.	-	-	+	-	-	-
57	*Notomastus aberans*	-	+	-	-	-	-
58	*N. latericeus*	-	+	-	+	-	-
59	*N. fauveli*	-	+	-	-	-	-
60	*Heteromastus similis*	+	+	-	-	+	-
61	*H. filiformis*	-	+	-	-		-
62	*Heteromastides bifidus*	+	+	-	+	+	-
63	*Mediomastus capensis*	-	+	-	-	+	-
64	*Mediomastus* sp.	-	-	-	-	-	+

65	*Leiochrides africanus*	-	+	-	-	-	
66	*Parheteromastus tenuis*	+	+	-	+	+	+
67	*Branchiocapitella singularis*	-	+	-	-	+	-
68	*Talehsapio annandalei*	+	-	-	-	-	-
69	*Scyphoproctus djiboutiensis*	-	+	-	+	-	-
70	*Pulliella armata*	-	+	-	-	-	-
71	*Branchiomaldane vincenti*	-	+	-	-	-	-
72	*Maldane sarsi*	-	+	-	-	-	-
73	*Owenia fusiformis*	-	+	-	-	-	-
74	*Owenia* sp.	+	+	-	-	-	+
75	*Pista indica*	+	+	-	-	-	-
76	*Polycirrus coccineus*	-	+	-	-	-	-
77	*Eunice tubifex*	+	-	-	-	-	-
78	*Nais* sp.	+	-	-	-	-	-
79	*Notopygos*	+	-	-	+	-	-
80	*Odontosyllis gravelyi*	+	-	-	-	-	-
81	*Serpula vermicularis*	+	-	-	-	-	-
82	*Sternapsis scutata*	-	-	-	+	-	-
83	*Glycinde bonhourei*	-	-	-	+	-	-
84	*Eteone* sp.	-	-	-	+	-	-
85	*Exogone* sp.	-	-	-	+	-	-
86	*Ophelia capensis*	-	-	-	+	-	-
87	*Maldane* sp.	-	-	-	+	-	-
88	*Sabellidae*	+	-	-	+	-	-
89	*Pectinaria* sp.	-	-	-	-	-	+
90	*Sabellaria pectinata*	-	-	-	-	-	+
	Total	**30**	**54**	**10**	**32**	**19**	**19**

Throughout the study period, in the southern most stations having a stable community structure of *N. indica* were established. The southern zone was characterized with higher organic carbon, which leads to a stressed environment in the region. Organic enrichment is considered as one of the important factors affecting the macrobenthic communities (Ansari, 2014). In Cochin estuary, high abundance of *N. indica* was favored by pollution

(Remani et al., 1983). It is noted that the annual monsoonal flooding, intense rain fall and increased river discharges in the southern zone do not cause any major impact on the organic carbon loading. Thousands of house boats are operating in the southern zone of Vembanad alone (Rajan et al., 2008) and in which most of these house boats are lacking proper sewage treatment system. Similarly the unregulated sewage discharges from resorts, exhaust from motor boats, fuel leakage, agricultural runoff etc. add the hydrocarbon and other organic pollutants to the southern part of Vembanad estuary. According to Musale et al. (2015) the species *Cirratulus* sp. and *Cossura coasta* were common and dominant in sandy-silt substratum with moderate OC (< 3%). At the same time, the pollution tolerant *Cirratulus* sp. was more in the clayey and sandy substratum with high organic carbon. In Cochin estuary Pillai (1977) noted that the seasonal increase in abundance of species such as *P. pinnata*, *P. polybranchiata*, *P. tenuis* was mainly due to its recruitment. Similarly the maximum number of polychaete species was observed during PRM and minimum during MN season. During the study period in Vembanad estuarine system, seasonal minimum polychaete density was observed during MN season. Whereas the maximum seasonal density was varied in both the years, in which the maximum density was observed during PRM and PM season during first and second year respectively. During MN season, the low density of polychaetes was observed in Nethravathi estuary (Gowda et al., 2009) and Coleroon estuary (Prabhadevi, 1994). The increased rainfall and freshwater runoff during MN season, modified the substratum and which in turn disturbed the settled bottom fauna for their choice of substratum selection and ultimately affected its distribution (Prabhadevi, 1994). Sivadas et al. (2010) observed that polychaete abundance, biomass and species diversity was maximum during PM and minimum during PRM. This was mainly attributed by the recruitment of young ones found more in MN season. Similarly most of the polychaetes observed during PRM were mature adults having eggs.

Seasonal variations of benthic fauna in tropical estuaries are prominent due to the monsoonal rains. During MN season higher riverine runoff flushes the estuary and the estuarine water becomes fresh (Shetye et al., 2007). The faunal diversity of benthic fauna was decreased during MN period (Sivadas et al., 2011). In contrat with onset of MN an increased defaunation, migration and spawning of the macrofauna observed and also increased organic matter and primary productivity during the peak MN regime and PM period support recruitment of more macrofauna (Gaonkar et al., 2013). According to Bemvenuti (1997), in tropical estuaries increased salinity during PRM accelerates the reproduction of majority of the benthic fauna and subsequently its recolonization occurs during PM season.

In a healthy environment, the Shannon diversity and Margalef richness having high value, which varied from 2.5–3.5 (Ajmal Khan, et al., 2004). In the present investigation Shannon diversity and Margalef richness were always less than the corresponding range of value and its maximum value in the study area was 1.8 and 2.1 respectively. The highest richness value in station 10 was attributed by the maximum number of polychaete species and lowest in station 2. Comparing with other stations, the maximum number of polychaete species (19 species) was recorded in station 10, whereas it's lowest value in station 2 that was attributed by the increased dominance of two polychaete species, *D. aestuarina* and *N. indica* that were also found to be pollution tolerant (Remani et al., 1983). Species richness in the southern zone of Vembanad was always very lower as observed in stations such as Punnamada, Pallathuruthy and Marthandam. In the funnel plot also southern stations falling outside the confidence limit clearly indicates that these station were influenced by external factors. The increased silt and organic carbon content in the sediment also explained the lowest richness value in the southern most stations. Such type of observations were made by Shimabukuro et al. (2016) in the southwestern Atlantic continental shelf where the lowest species richness and abundance of polychaetes associated with the occurrence of fine sediment and higher concentration of organic matter. These stations representing a stressed and unhealthy condition in all the period mainly because of the intense pressure from the house boat tourism and agricultural runoff from paddy lands of Kuttanad and low salinity. Overall in the southern part organic enrichment from municipal wastes, hydrocarbon contamination from house boats, fertilizers and pesticides from adjoining paddy fields and closure of TMB jointly pollute the aquatic system (Asha et al., 2016). The commercial and recreational utilization of patellid limpets were lost in the intertidal region of Atlantic islands impacted with sewage discharges mainly due to the rampant tourism growth (Martins et al., 2008). The unhealthy benthic habitats were represented by reduced species diversity, abundance and biomass and also increased dominance of small bodied pollution – tolerant species (Dauer, 1997). That is where the conservative species were replaced by opportunistic species having small body size and a short generation time, leading to its abundance and following lower species diversity (Warwick, 1986). Positive pollution indicators are normally opportunistic species increased its dominance with pollution (Rygg, 1985). The frequent houseboat and transport boat navigation causes the shallow estuarine bottom more unstable and may cause the lowest diversity and evenness value in station 2. When comparing the species diversity, richness and evenness of polychaetes in the retting and nonretting zones, the lowest value was always observed in the retting zones characterized with anoxic conditions along with the increase in hydrogen sulfide and BOD (Bijoy Nandan and Abdul Azis, 1995). Increasing organic input into the sediment leads to anoxic condition and this will cause the exclusion of all species distribution and bear only species with the ability to adapt the stress (Ansari et al., 1986). It was possible that the high organic carbon especially due to the decomposition of plant matter in station 2 may sometimes create an oxygen limiting condition. The polychaete species richness and diversity are rather lower in polluted areas with the absence of oxygen in the water column (Dhainaut Courtois et al., 2000). According to Diaz and Rosenberg (1995), among the macrofauna, polychaetes are the taxon most tolerant to low oxygen than bivalves and crustaceans. Increasing organic input into the sediment leads to anoxic condition and this will cause the exclusion of all species distribution and bear only species with the ability to adapt the stressed condition (Ansari et al., 1986).

According to Sukumaran and Sarala Devi (2009) the regular and continuous vessel navigation in the Mumbai Port area, cause pollution and also affected the benthic habitat. They observed that lowest diversity and evenness values in the Mumbai Port area, where the *P. pinnata* and *Polydora giardi* were the dominant species. Similarly based on the diversity index, Sukumaran et al. (2011) observed an unhealthy condition in Ratnagiri bay; where during PRM and PM season the Shannon diversity varied from 1.4 to 2.4 and 0.6 to 1.6 respectively. Similarly the Margalef richness index during both the seasons such as PRM and PM season varied from 1.5 to 2.3 and 1 to 2 respectively. Martin et al. (2011) mentioned that invasion of opportunistic polychaete species revealed a stressed environment in Cochin estuary. In the present study richness value in the northern most station represented a healthy condition where, majority of the polychaete species are opportunistic which are associated with the aggregation of tube dwelling polychaetes, *D. neapolitana*. The tube dwelling polychaete species *D. cuprea* stabilize the bottom mud to form tubes and various materials are accumulated between them. So it creates new microhabitats with increased species richness (Rhoads, 1974). In the present investigation, during both the years 'Diopatra' zone having maximum species richness. It was contrasting with the observation of lowest species richness value in the fine sediment. According to Gray et al. (2002), the structurally uniform muddy sediments of increased organic matter formed oxygen depletion and also increased the toxic materials in the sediment column. Apart from this species in the muddy substratum was related to the silt/clay ratio, in which elevated level of clay content formed a lowest richness, diversity and abundance of benthic infauna (Fresi et al., 1983). According to Ourives et al. (2010), the difference in richness, abundance and distribution of macrofauna were primarily connected to salinity gradient of the system and secondarily the sediment parameters such as organic matter and fine sediments, along with the feeding type of benthos. Sivadas et al. (2010) noted that the polychaete species richness value ranged from 1.24 to 2.49, evenness ranged from 0.6 to 0.8 and the diversity ranged between 1.59 and 2.1 in the port area of Mormugoa. In the Mormugoa port area, from the 71 polychaete taxa, *Mediomastus* sp. (77 ind./m^2 to 453 ind./m^2) and *P. pinnata* (58 ind./m^2 and 100 ind./m^2) formed the most dominant species contributing 14 % both to the total abundance of polychaetes (Sivadas et al., 2010). In the polluted zone of Visakhapatnam harbour area, the polychaete species richness value ranged from 0.31 to 1.29 and diversity ranged between 1.18 and 2.79 (Raman and Ganapati, 1983). Increased abundance of opportunistic species was found in the organic carbon rich areas (Pearson and Rosenberg 1978). However sediment of higher organic carbon content supports only a few species (Cocito et al., 1990; Albayrak et al., 2006) and maximum diversity was observed in the moderate organic carbon area (Sivadas et al., 2011). In the retting zones of Kadinamkulam estuary, the polychaete species were represented by *Ceratonereis* sp., *Branchiocapitella* sp., *Nereis chingrihattensis* and *P. cirrifera* (Bijoy Nandan and Abdul Azis, 1995).

Polychaetes, the abundant taxon in benthic samples have been most often utilized as indicator species of environmental conditions (Dean, 2008). Polychaetes contain both sensitive and tolerant species indicating the environmental disturbance, from pristine to greatly distressed habitats (Pocklington and Wells, 1992). Bio-indicators were used for evaluating the ecosystem status of impacted area for taking up suitable management measures for the conservation (Sukumaran and Sarala Devi, 2009). The species belonging to genera Prionospio, Capitella and Mediomastus were more abundant in places where higher organic enrichment with reduced oxygen (Pearson and Rosenberg, 1978). *Mediomastus* sp. is dominant in a moderately disturbed environment (Rivero et al., 2005). The species such as P. *pinnata* and *Cossura* sp. are considered as pollution indicators (Grassle et al., 1974). In the present investigation the polychaete species, Spionids, Capitellids, Cossurids and Ciratulids were more abundant in the northern most stations especially in Aroor region. The mature and larval stages of species of Spionid, Cirratulids, Cossurids, Capitellids are known for its significant tolerance to hydrocarbon and other toxic pollutants, that are toxic to most of the other fauna (Levin et al., 1996). The Aroor region was highly impacted with polluted effluents from numerous shell fish industries situated on the banks of estuary. At the same time the urban and industrial discharges from Cochin metro city also enhances the pollution load. It was reported that the species of polychaete family Cirratulidae and Cossuridae were able to withstand in stressful condition and established in both healthy and unhealthy environmental conditions (Jayaraj et al., 2007). Among the nineteen species of polychaetes observed during the present investigation, majority of the species are indicators of organic enrichment. In station 1, *N. indica* and *Dendronereis aestuarina* were the abundant polychaete species. During 2011-2012 periods, *Dendronereis aestuarina* were abundant in station 4 and during 2012-2013 period it was abundant in station 7. Both the years *N. indica* were abundant at station 2. In Cochin backwater Sarala Devi et al. (1991a) considered *Dendronereis aestuarina* as a pollution resistant species. Similarly species such as *Prionospio polybranchiata*, *N. indica* and *P. tenuis* considered as pollution tolerant species. In the polluted inner harbour area of Visakhapatnam, Raman and Ganapati (1983) observed the assemblages of *Capitella capitata*, *Nereis glandicincta*, *Diopatra neapolitana*, *Nephtys oligobranchia*, *Polydora ciliata*, *Prionospio cirrifera*, *Branchiocapitella singularis* and *Cirratulus cirratus*. Heavy metal analysis revealed that, Vembanad estuary was moderately polluted with Zn, Cu, Cd and Pb. During the study period, station 1 was heavily polluted with Cu, Zn and Pb in all the seasons except during PM where Zn was moderately polluted. *Namalycastis indica* may also take heavy metals with high organic matter in the southern region. Ahn et al. (1995) observed that the polychaetes *Heteromastus filiformis* and the Nereid, *Nereis aibuhitensis* were able to live in the heavy metal polluted area (Pb, Cd, Zn, and Cr) with high organic matter from industrial sewage. According to Berthet et al. (2003) polychaetes consume large amount of heavy metals coupled with their food and converted to less toxic forms in the gut. Heavy metal analysis (Cu, Pb, Cr, Ni, Zn, Cd, Hg) showed that *Glycera longipinnis* act as a biological indicator for heavy metal pollution along the southwest coast of India. The capitellid, *Parheteromastus tenuis* were abundant in the northern most stations, station 9 and station 10. Generally capitellids, are usually identified as indicators for stressed environments in harbours (Belan, 2003). Remani et al. (1983) observed *P. tenuis* in the coconut husk retting area whereas *P. polybranchia* abundant in the municipal discharge area of Cochin estuary. Sukumaran and Sarala Devi, (2009) found that *P. tenuis* was dominated in the Prince's Docks area of Mumbai Port, which facilitate major jetties for sailing vessels, country crafts and fishing trawlers. In the anoxic environment of retting zone of Kadinamkulam estuary was represented by *Prionospio cirrifera* and *Ceratonereis* sp. (Bijoy Nandan and Abdul Azis, 1995). In the present investigation *Prionospio cirrifera* was more abundant in station 9. In the present investigation, increased levels of organic carbon in the northern most stations trigger the proliferation of species of Spionidae (*P. cirrifera*) and Capitellidae (*P. tenuis*) families. The species in these families are mainly opportunistic, having higher reproduction and dispersal ability. Increasing dominance of opportunistic species especially species of

Capitellidae indicated the destruction of aquatic ecosystem (Zajac and Whitlatch, 2001). In the sewage discharge site Cabral-Oliveira and Pardal (2016) observed the high abundance and colonization of species from Capitellidae or Spionidae families. According to Elias et al. (2006), the opportunistic species are pollution tolerant increasing in abundance in the impacted areas due to the accessibility of large amount of organic food. Occurrence of organic pollution indicators such as *Prionospio cirrifera*, *Prionospio pinnata*, *Cossura coasta*, *Glycera alba*, *Scoletoma impatiens, Sigambra parva, Micronephthys oligobranchia* and *Diopatra neapolitana* in station 10 suggested that the sampled area is negatively affected with organic enrichment. The stress indicators such as Prionospio, Nepthys, Lumbriconeries, Glycera, Neriedae spp. were found in the port area of Karwar (Bandekar et al., 2016). The species such as *Paraprionospio pinnata*, Polydora, *Cossura costa*, *Nephtys polybranchia*, *Capitella capitata*, Mediomastus, *Megalona longicornis, Terebellides stroemi* were dominant there. Polychaete species such as *Glycera alba* and *Lumbrinereis* spp. were described as indicators of unpolluted conditions and *Polydora* spp., were highly abundant at the most polluted sites and were therefore designated as indicators of polluted conditions (Dean, 2008). The spionid, *Polydora websteri* were dominant near the sewage outfall (Surugiu, 2009). According to Losovskaya (1978) the proliferation of detritus-feeding *Polydora cornuta* and *Heteromastus filiformis* was observed in the eutrophicated and organic enriched Black Sea. Similarly in the central part of Cochin estuary high abundance of *C. capitata* was noted by Martin et al. (2011). In station 10, most of the observed species are categorized as opportunistic deposit feeders. Organic rich sediments are directly ingested by deposit feeders either selectively or non-selectively. The suspension feeding organisms decreases when organic load in the sediment increasing and at the same time deposit feeders increases. Normally the deposit feeders are colonized in soft muddy substratum with higher organic content and suspension feeders in areas having sand or firm mud (McLusky, 1989). According to Hopkinson and Smith (2006) organic matter inputs was related with benthic remineralization process. Polychaetes such as spionids, ampharetids, terebellids, and some nereidids reduce the amount of the organic matter accumulated in the sediment and transformed and incorporated to its biomass through deposit-feeding. So polychaetes fasten the circulation of the organic matter from the water column and the sediments into the food chain, without being mineralized (Surugiu, 2005).

The variation of species composition in the southern and northern zone of Vembanad estuary depicted by the analysis of cluster and MDS, was supported by SIMPER analysis. In the Vembanad estuarine system, the polychaete species assemblages are important bioindicator to differentiate between southern and northern stations. The analyses showed that the grouping was mainly responsible for the variation in density of polychaete species and also the presence and absence of some taxa. Bray-Curtis similarity index was used for grouping the stations according to polychaete species abundance. Cluster analysis showed two main clusters, in which southern most stations formed cluster one and northern most stations such as stations 9 and 10 formed cluster two. In the present investigation, compared to southern stations, a clear pattern of species distribution of polychaete was observed in the northern most stations, where species such as *S. parva*, *M. oligobranchia, S. impatiens, A. filiformis, Pectinaria* sp., *S. pectinata* were present only in station 9 and 10. In the first subgroup of cluster one represented by southern most stations such as station 1, 2 and 3 had higher organic carbon (> 3 %), where only two species of polychaetes was present, *D. aestuarina* and *N. indica*. High organic carbon areas were avoided by polychaetes, which denoted that increased organic matter negatively affected the abundance and distribution of polychaetes (Musale and Desai, 2011). The annual monsoonal flooding in the Kuttanad (the unique below sea level farming system in India) region brings fertile water with high organic debris including the remaining of straw of harvest and also residues of chemical fertilizers into the estuary, which may enhanced the organic enrichment in the southern zone. In the Kuttanad paddy soils, the average organic carbon was 4.63 %, which is higher than the other low land agriculture fields due to the increased organic input and long-term flooding events (Kannan et al., 2014). In the present investigation, the annual monsoonal flooding in the Kuttanad region may greatly impacted the existence of less number of polychaete species in the southern zone. Cyrus et al. (2011) noted that, the periodic flooding caused an unstable nature of the sediments in the Mfolozi–Msunduzi estuarine system of South Africa, which prevent the occurrence of higher densities of benthic fauna there. The organic fertilizers and plant residues forms the major source of carbon in the paddy fields (Tanji et al., 2003). In the Nethravathi estuary, *D. aestuarina* was the dominant polychaete species recorded throughout the year and found that higher organic carbon retained high polychaete population density (Gowda et al., 2009). Similarly higher density of *Namalycastis abiuma* was found where the organic matter content was between 3.0 % and 3.8 % in the Shenzhen Bay Mudflat, China (Cai et al., 2013). The polychaetes *Dendronereis pinnaticirris*, *Neanthes glandicincta* and *N. abiuma* formed the major part of the endobenthos and act as an important food resource for migrating waterfowl in the mudflat in Shenzhen Bay, in which some polychaetes are preferred to ingest sediment deposits, at the same time it can be able to graze plant material and also able to prey upon small invertebrates (Cai et al., 2001). The southern sector of Vembanad estuary was famous for its large number of migratory as well as resident water fowl (Narayanan et al., 2011), which may be preferred habitat for the benthic polychaetes. According to Desrina et al. (2013), in the traditional ponds in Indonesia, Dendronereis spp. was the major natural prey of penaeid shrimps and also found that *Dendronereis* spp. act as a propagative carrier for White spot syndrome virus (WSSV), the causative agent of white spot disease (WSD) in penaeid shrimps.

When comparing the overall distribution of polychaete species during both the years it was found that, most of the species having a wide range of salinity tolerance, representing euryhaline forms (Table 6.12). *D. aestuarina* present in all stations having a salinity range of fresh water to 33.1 ppt. So the salinity tolerance of this species was found to be higher. Misra (1999) noted a narrow range of salinity tolerance (8–12 ‰) of *D. aestuarina* in the Munda River, India. Similar to *D. aestuarina,* the polychaete species *N. indica* found in all stations. The species *Ceratonereis* sp. present in northern most stations, such as station 8, 9 and 10, where the salinity ranged from 0.03 to 33.1 ppt. *Sigambra parva, M. oligobranchia, S. impatiens, A. filiformis, Pectinaria* sp.*, S. pectinata* were present only in station 9 and 10, whereas the species such as *D. neapolitana, D. flava and P. pinnata* was present only in station 10. In stations 9 and 10 the salinity value ranged between 0.04 to 32.9 ppt and 0.08 to 33.1 ppt respectively. Based on the annual average salinity pattern these two stations were classified as mesohaline to polyhaline zones. On the other hand based on the monthly salinity value these stations were categorized as mixoeuhaline. During both the years the average salinity pattern in station 1 to 5 denoted as an oligohaline zone. During first year stations 6 to 8 included in the oligohaline zone, whereas stations 9 and 10 were grouped as a mesohaline zone. During second year, station 7 to 9 denoted as mesohaline and station 10 as polyhaline zone. Salinity formed an important physical characteristic of an estuarine and marine ecosystem, which has often played a major role in ecological zonation (Onwuteaka, 2016). Based on the salinity preference Sunil Kumar (1993) grouped *P. tenuis* in the highly tolerant euryhaline forms, preferred the salinity variation of 0.2 to 29.76 ppt. The species *D. aestuarina* and *P. cirrifera,* were included into a moderately tolerant form capable to survive salinity as low as 20 ppt. The species such as *G. alba, D. neapolitana, Dipolydora* sp. and *Scoletoma* sp. were grouped into species able to tolerate small variation of salinity. These species were considered as stenohaline, mainly found in the high saline areas where the salinity was above 24 ppt. Hirst (2004), reported that variations in salinity can promote marked shifts in species representation and community structure because marine organisms have different salinity tolerances. According to Giangrande and Gravina (2015), based on the salinity variation polychaete species categorized different ecological zone as; oligohaline zone represented by few species (*Hediste diversicolor*, *Alitta succinea*, *Ficopomatus enigmaticus*, *Polydora ciliate*; mesohaline zone composed of large number of species able to live in brackish waters as well as marine coastal waters (*Nainereis laevigata*, *Cirroforus furcatus*, *Capitella capitata*) and polyhaline zone represented by large number of taxa, mainly marine species (*Glycera tridactyla*, *Lumbrinereis latreilli*, *Heteromastus filiformis*). Similar condition was observed in the present study, where the oligohaline southern stations dominated by *D. aestuarina* and *N. indica.* Large number of marine species such as *D. neapolitana*, *P. tenuis*, *M. oligobranchia*, *S. impatiens*, *P. cirrifera* and *C. coasta* were found in the meso to polyhaline stations. During first year *Ceratonereis* sp. (r = 0.51), *M. oligobranchia* (r = 0.44), *P. tenuis* (r = 0.47), (r = 0.44), *S. parva* (r = 0.55), *D. flava* (r = 0.48), *G. alba* (r = 0.49). *S. impatiens* (r = 0.63), *A. filiformis* (r = 0.44), *C. coasta* (r = 0.54), *S. pectinata* (r = 0.46) and *D. neapolitana* (r = 0.37) showed significant positive correlation ($P \leq 0.01$) with salinity. Similarly during second year period, polychaete species such as *P. tenuis* (r = 0.56), *P. cirrifera* (r = 0.43), *G. alba* (r = 0.53), *S. impatiens* (r = 0.48) and *D. neapolitana* (r = 0.46) showed significant positive correlation ($P \leq 0.01$) with salinity.

The unscientific operation of the Thaneermukkom barrage, the salinity barrier to safeguard rice cultivation appears to have blocked the connectivity of the Vembanad estuary to the Arabian Sea for a good part of the year. This has severely affected the ecological and saline water and thereby interfering with the natural cleansing mechanisms of the estuarine system. During the closure period, the stagnant nature of water column and absence of tidal flushing all enhances the settlement of dead and decaying organic matter in the sediment and boost the tolerant species of spionids and capitellids. A complete dominance of opportunistic species of Capitellidae showed an indicator of potential degradation of aquatic system by human activities (Zajac and Whitlatch, 2001). In the south, the abundance of opportunistic species *D. aestuarina* was decreased during the closure period of barrage. In this time a low salinity pattern and high organic load in the southern region may not support the larval recruitment of *D. aestuarina.* At the same time another opportunistic species such as *Mediomastus* sp. and *P. tenuis* start to colonizes in the southern zone. The development of benthic communities is depending upon the supply of colonizers such as larvae and its spatial and temporal variation (Pacheco et al., 2010). *Mediomastus* sp. quickly colonizes in a newly distressed or defaunated areas (Hyland et al., 1991). It was the ability of opportunistic polychaetes species to repopulate rapidly in a disturbed substratum (Gray, 1981; Diaz, 1984). In a disturbed habitat, usually pioneer colonizers with opportunistic life history traits appear first (eg. spionid polychaetes) and these species adapted to colonize quickly and also prepared the habitat for subsequent colonizers (Lu and Wu, 2007). In the non retting zones, the dominance and succession of benthos were primarily depend upon the variations in temperature, salinity and rainfall and river discharge, whereas in the polluted retting zone the high organic content, temperature, hydrogen sulfide, dissolved oxygen controlled the dominance and succession of species (Bijoy Nandan, 1991). Generally climatic features affect the water quality and quantity and it ultimately changed the benthic biodiversity. According to Brady and Somero (2006) the decreased salinity in estuarine habitats formed an important factor than temperature in controlling distributions of species. Dietz et al. (1996) noted that the invaded brackish water species in the freshwater habitat regularly appeared severe physiological alterations in the low saline environments. The climate change effects especially sea level variation altered the salinity gradient in estuaries and it ultimately affect the spatio-temporal variability in the standing stock of benthic macrofauna (Fujii, 2012). Climate change and benthic interactions study noted that salinity variation greatly increased the osmotic stress of benthos (Birchenough et al., 2015). Over the last 30 years, decreasing salinity and increasing organic matter in the southern zone may reduce the polychaete species diversity, where only *Namalycastis indica* and *D. aestuarina* were abundant. These two species are commonly found in the fresh to brackish water environment (Misra, 1995).

Table 6.12 The ranges of salinity preference for polychaete species in Vembanad estuary

Polychaete species (CCA code)	**Salinity range (ppt)**
Sigambra parva (SP)	0.08 – 31.2
Namalycastis indica (NI)	0.01 – 19.7
Dendronereis aestuarina (DA)	0.01 – 33.1
Nereis sp. (NS)	0.08 – 20.8
Ceratonereis sp. (CS)	0.17 – 20.8
Micronephthys oligobranchia (MO)	0.08 – 33.1
Glycera alba (GA)	0.02 – 33.1
Diopatra neapolitana (DA)	0.08 – 33.1
Scoletoma impatiens (SI)	0.14 – 33.1
Dipolydora flava (DF)	0.14 – 33.1
Prionospio cirrifera (PC)	0.02 – 33.1
Paraprionospio pinnata (PP)	0.08– 33.1
Aphelochaeta filiformis (AF)	0.14 – 30.9
Cossura coasta (SP)	0.14 – 33.1
Mediomastus sp. (MS)	0.02 – 2.31
Parheteromastus tenuis (PT)	0.03 – 33.1
Owenia sp. (OS)	0.01 – 29.5
Pectinaria sp. (PS)	2.31 – 20.8
Sabellaria pectinata (SP)	0.04 – 33.1

According to Ysebaert and Herman (2002), environmental variation is the key factor for disparity in the macrobenthic structure and its variation was observed in the deviation of richness and abundance of the taxa. The grain size and organic carbon content of sediment clearly separate the species assemblages (Sarda et al., 1995). According to Yu et al. (2012) BIO-ENV analyses in the Han River estuary, Korea showed that environmental variables such as benthic salinity, total organic carbon at the bottom, dissolved oxygen in bottom water, and silt and sorting values of the sediment showed the maximum correlations with macrofaunal composition ($\rho = 0.7$, $P < 0.01$). He also observed that lower salinity due to the freshwater input from the Han River was an important environmental variable affecting macrobenthic community structure during the rainy season. Generally the soft bottom polychaete fauna formed a significant role in distinguishing the system in accordance with granulometry, salinity and sediment organic matter (Raman and Ganapati, 1983; Pardal et al., 1993; Mendez, 2002). On the other hand, several studies point out that depth, water movement and dissolved oxygen in the water column formed the major factors controlling the polychaete distribution than the sediment granulometry and salinity etc. (Nicolaidou and Papadopoulou, 1989; Lardicci et al., 1993; Mistri et al., 2002). According to Day et al. (1989) the spatial variation of benthic communities along the estuarine gradient was closely related to the salinity variation, depth, sediment grain size and organic content. These results are in agreement with that of the present study. BIO-ENV analysis was performed to know the grouping of environmental parameters that best related with the polychaete community structure in a consistent manner. In the present study, oxygen concentration, temperature, and salinity were the major parameters controlling the distribution of polychaete species, while sediment particle size and organic carbon content also regulate the community structure of polychaetes. At the same time heavy metals such as Cu, Zn, Cd and Pb did not have any strong relation on the polychaete community structure. According to Sivadas et al. (2010) the absence of any important connection between the biotic and abiotic parameters was mainly due to the low bioavailability of the contaminants. The factors such as bioturbation, salinity, temperature and redox potential and pH were controlling the bioavailability of contaminants (Bryan and Langston, 1992). In Ratnagiri bay polychaete distribution was best explained by sand and silt (Sukumaran et al., 2011). Depth, temperature, dissolved oxygen and sand forming the best combination of variables explaining the distribution of polychaetes in the south west coast of India (Joydas and Damodaran, 2009). In Vasishta Godavari estuary a close relationship between polychaete species patterns and dissolved oxygen, sand, organic matter, salinity, clay and silt content were observed by Rao et al. (2009). Similarly, based on the BIO-ENV analysis, Guerra-Garca and Garca-Gmez (2004) found that the polychaete assemblages in the harbour area of Ceuta, northern Africa was elucidated by a combination of five abiotic parameters in the reduced environment (r = 0.42) such as Cu, phosphorous, Ca, Mn and percentage of sand.

CCA analysis clearly represented the influence of environmental parameters on the polychaete community structure in Vembanad estuary. In the CCA triplot, sediment parameters such as sand, silt, available nitrogen and organic carbon were having longest vector length. Large vector length of these parameters exhibited their role in controlling the community structure of polychaete species in Vembanad estuary. The CCA plot represented that the polychaete species, *P. cirrifera, P. tenuis* was mostly associated with the sand fraction and also the sand fraction was dominant in stations 6, 8 and 9. Normally the polychaete species *P. cirrifera* preferred the sandy substratum (Joydas and Damodaran, 2009). Estuarine sediments may vary due to particle size and organic carbon content of the sediment. It affects the distribution and abundance of benthic organisms in that particular environment. The bottom characteristics in estuaries and near shore region are linked to the source and texture of sediment and also the topography of the particular region (Sesamel et al., 1986). All the benthic fauna especially the distribution of polychaetes were greatly influenced by the sediment texture (Jayaraj et al., 2008; Manoharan et al., 2011). According to Ansari (1974) in Vembanad estuary, polychaetes are restricted to the fine deposits of sand, silt and clay. In the shallow regions of Cochin estuary fine sand and silt were dominant, whereas in the upper reaches and barmouth area, sandy mud and coarse were observed (Kurian, 1972). According to Feebarani et al. (2016) silty-sand was the predominant sediment type in Cochin estuary, in which southern part of Cochin estuary was sandy, whereas in the central and barmouth region it was either muddy (silt-clay) or mix of mud and sand type which sustain more organic matter in the sediment. The species *P. tenuis* generally observed in the areas was affected with sewage pollution (Das et al., 2009). In the Vembanad estuarine system sewage pollution was a major threat, in which there was no proper measure to control the dumping of municipal solid and sewage wastes and also from industrial, tourism and hospital wastes etc. into the water body (Anon, 2008). The CCA analysis of polychaete species with environmental parameters in the Visakhapatnam Harbour area on the east coast of India observed that organic carbon, salinity, DO, sand and clay were major environmental variables which were related to the polychaete distribution (Musale et al., 2015). Similarly species such as *D. neapolitana*, *M. oligobranchia*, *Nereis* sp., *Ceratonereis* sp., and *S. impatiens* were linked with sediment sodium and potassium in station 10. In the shelf waters of Arabian Sea the species such as *P. pinnata*, *Lumbrineris latrielli* and *D. neapolitana* were preferred sandy as well as muddy substrata (Joydas and Damodaran, 2009). In the present investigation the distribution of species such as *P. pinnata, D. neapolitana, M. oligobranchia, A. filiformis, C. coasta* and *D. flava* were mainly confined in station 10 where the sediment substratum had higher percentage of silt and clay. Pearson correlation analysis showed that *M. oligobranchia* ($r = 0.5$), *P. tenuis* ($r = 0.47$), *Dipolydora flava* ($r = 0.51$), *G. alba* ($r = 0.64$), *S. impatiens* ($r = 0.71$) and *D. neapolitana* ($r = 0.6$) showed significant positive correlation ($P \leq 0.01$) with clay. Mandal and Harkantra

(2013) noticed that the deposit feeding polychaetes *P. pinnata, C. coasta,* and *H. similis* preferred the silty substratum. Similarly Ganesh and Raman (2007) also reported such type of selection of silty-clay/clayey substratum by those polychaete species in the northeast Indian shelf. Available nitrogen, organic carbon, total carbon and low saline condition relate well with the distribution of *N. indica* in stations 1 to 4. Pearson correlation analysis showed that *Namalycastis indica* showed significant ($P \leq 0.01$) positive correlation with organic carbon (r = 0.49), silt (r = 0.48), available nitrogen (r = 0.6) and negative correlation with salinity (r = - 0.3). There were several reports that indicated the preference of *N. indica* towards organic carbon (Glasby, 1999; Magesh et al., 2012). *N. indica* can withstand low salinity conditions and this genus was distributed from a drinkable freshwater to marine waters (Glasby et al., 2003) and also in polluted waters. Similarly sediment pH has great impact upon the distribution of *G. alba, S. parva* and *Sabellaria pectinata.* In the CCA ordination it was also observed that, the distribution of *D. aestuarina* was closely related to station 7. Based on the CCA analysis in Vasishta Godavari estuary, Rao et al. (2009) noted that the distribution of polychaete species such as *Magelona cincta*, *P. cirrifera*, *C. coasta*, *S. parva* and *Glycera longipinnis* were controlled by salinity and the distribution of *Ceratonereis erythraeensis*, *Nereis lamellosa* and *Glycinde oligodon* were related to sand and temperature. Similarly organic matter, silt and dissolved oxygen have a great role in the distribution of *Prionospio krusadensis*, *Dendronereis arborifera, Polydora kempi*, *P. pinnata*, *Prionispio saldanha*, *Glycera tesselata* and *G. alba*. Similarly the distribution of *P. cirrifera, H. similis*, *Indonereis gopalai*, *Nephthys oligobranchia*, *Nereis neanthes capensis* and *Nectoneanthes ijimai* was controlled by clay fraction. Ngqulana et al. (2010) noted that from the Mfolozi–Msunduzi system, among the polychaetes *Ceratonereis* sp., *Dendronereis arborifera* and *C. capitata*, were dominated. Oxygen concentration, temperature, the state of the mouth and salinity were the main factors influencing the distribution, while particle size and organic content of the substratum that also influenced community structure. Higher abundance of polychaetes in the west coast of India was mainly related to the sandy substratum of the region, which provides more interstitial space for polychaetes to grow (Musale and Desai, 2011).

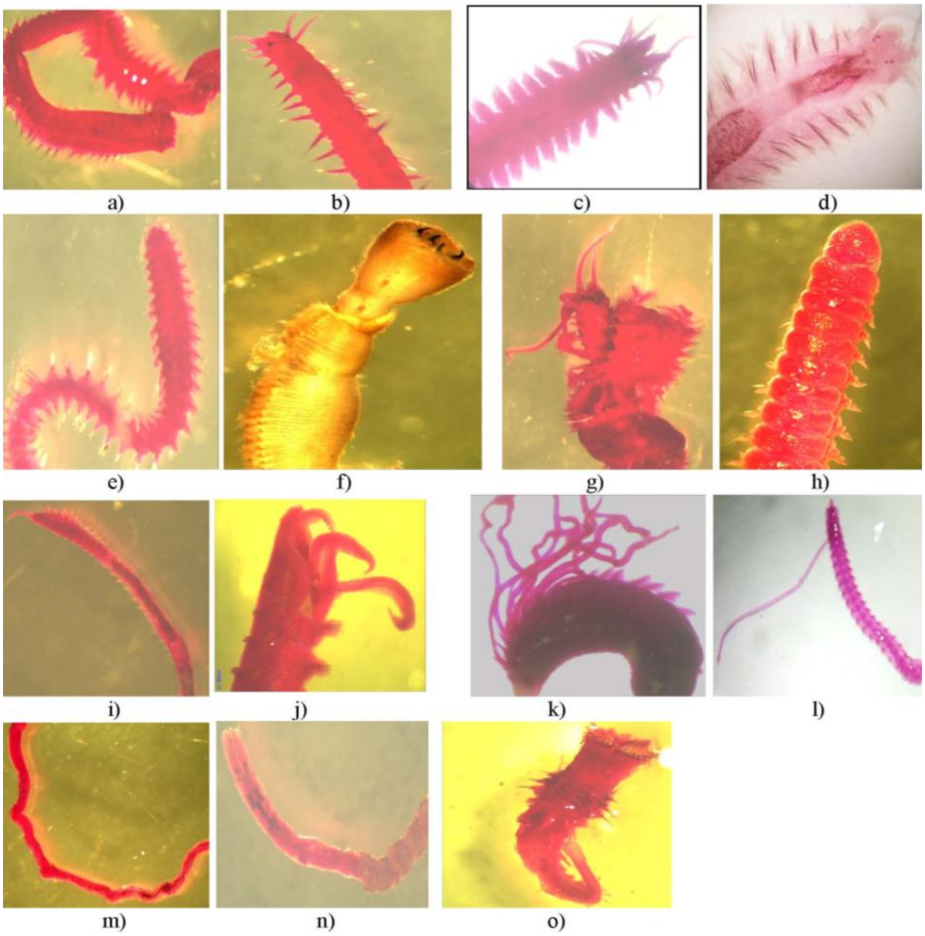


Plate 6.1 (a-o) Major polychaete species observed in Vembanad estuary during 2011-2013 period (Magnification– X4) a) *Sigambra parva;* b) *Namalycastis indica*; c) *Dendronereis aestuarina*; d) *Ceratonereis* sp.; e) *Micronephthys oligobranchia*; f) *Glycera alba*; g) *Diopatra neapolitana*; h) *Scoletoma impatiens*; i) *Prionospio cirrifera*; j) *Paraprionospio pinnata* k) *Aphelochaeta filiformis*; l) *Cossura coasta*; m) *Parheteromastus tenuis*; n) *Owenia* sp.; o) *Sabellaria pectinata*

Plate 6.2 Aggregation of tube building polychaetes in the Aroor region

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Chapter 7

COMMUNITY STRUCTURE AND ECOLOGY OF PERACARID CRUSTACEANS IN THE VEMBANAD ESTUARY

Contents

7.1 Introduction

Among the soft-bottom macrofaunal communities, crustaceans are the most diverse and abundant groups (Cunha et al., 1999; Lourido et al., 2008) and accepted as a most sensitive group towards environmental perturbation especially sediment contamination (Dauvin, 2008; Sanchez-Moyano and Garcia-Asencio, 2010). Traditionally, the crustaceans have been the most used in associated fauna studies to establish the environmental guides that control epifaunal communities (Russo, 1989; Aoki and Kikuchi, 1990). Most of the benthic faunal studies are concentrated on the infauna and a few has been made on benthic crustaceans. Crustaceans have increased mobility over other infaunal members. So with any alterations in the environmental conditions, the crustacean community not only reacts with its disappearance or recruitment, but quicker response happened to the mobile individuals by moving in or out of an area (Chou et al., 1999). Peracarids are the most diverse and dominant soft-bottom benthic crustaceans actively involved in the benthic community assemblages (Duffy and Hay, 2000). Peracarids composed of small benthic crustaceans found from the littoral to hadal regions of the oceans, and also found in terrestrial and freshwater habitats

(Jaume and Boxshall, 2008). It formed a major source of food for other benthic organisms especially for commercially important fishes (Beare and Moore, 1996) and also supports the major share of benthic production (Mancinelli and Rossi, 2002). It was noted that the peracarids effectively populated in the brackish water environment (Cunha et al., 1999). The functional characteristic such as feeding behaviour, reproductive strategy and mobility adapt the crustaceans to flourish in the highly dynamic brackish water habitat. Due to its various kinds of feeding strategies, majority of the peracarids are flexible in their diets and utilize different types of food resources (Cartes et al., 2003).

In marine sediments, distribution of peracarids related to abiotic features especially hydro dynamism, sediment characteristics, food availability, organic content and biotic relation such as competition and predation (De Grave, 1999). The Superorder Peracarida includes orders, such as Amphipoda, Bochusacea, Cumacea, Isopoda, Lophogastrida, Mictacea,Mysida, Pygocephalomorpha, Spelaeogriphacea, Stygiomysida, Tanaidacea, Thermosbaenacea (http://www.marinespecies.org). In which the most species rich orders are Amphipoda, Isopoda, Tanaidacea and Cumacea. The majority of the members of these orders are found in all major marine habitats. These are of ancient marine origin, with specific subgroups having successfully colonized brackish and freshwater habitats (Parr et al., 2007). Among the peracarids, amphipods are dominant in terms of number of species and individuals in soft bottom habitats (Zettler, 2001; Moreira et al., 2008). Amphipods characterized the most diverse taxa of crustaceans. More than 10,000 species of amphipods have been identified so far (Lowry, 2013). In the marine benthic communities sometimes it may exceed all other macrofaunal groups in terms of numerical abundance, species diversity and also in biomass. Niche specificity is the major characteristic feature of many amphipod species so it can be used for understanding the spatio-temporal changes in environmental conditions

(Chintiroglou et al., 2004). Many amphipod species have high ecological tolerances to different environmental conditions and exhibit wide distribution both latitudinal and vertical. Due to its increased abundance and biomass sometimes it becomes the dominant animal groups in the biocenosis. Many fishes, seabirds and marine invertebrates directly consume amphipods and it plays an important role in the food web (Bryazgin, 1997). Tanaids, the important peracarid malacostraca crustaceans formed an important constituent of coastal benthic communities and distributed in wide variety of marine and brackish water soft bottom habitats (Gambi et al., 1992). These are excellent microscavengers, consuming detritus and other food particles, which proliferate in abundance and forms a major food source for the higher trophic level (Priya, 2015). In the benthic realm, isopods formed a major contributor to biodiversity and biotic resources and are actively involved in the recycling of organic matter and providing energy towards higher trophic level (Lopez et al., 2012). Cumaceans are also an essential component of the benthic fauna found in marine and brackish water habitats (Brandt et al., 1999).

7.1.1 Significance of the study

The Vembanad estuarine system and associated coastal aquatic ecosystems are repositories for several diverse groups or species of plankton, benthos, shell fish and fin fish resources. The interdependence of these organisms and their energy transfer in the trophic environment is of paramount importance for maintaining the productivity of the ecosystem. According to Menon et al. (2000), the Vembanad estuary supports as much biological productivity and diversity as that of a tropical rainforests and also dependable for the rich fisheries potential of the state. Among the soft-bottom macrobenthos crustaceans especially peracarids were important for its diversity and abundance. Many groups of this taxon are ecologically sensitive towards environmental contamination and many are used for assessing different types of environmental impacts. Moreover, it formed a major source of food for other crustaceans and fishes (Stal et al., 2007). Vembanad estuary is famous for its shell fish and finfish resources. But the prosperous fishery in this region is seriously declining mainly due to the unscientific operation of TMB and various environmental issues (Asha et al., 2014). Benthic fauna constitutes a major share of crustaceans which has been totally neglected with scanty information reported for the Vembanad estuary during the pre or post barrage phase. Further, these macro invertebrates are strong drivers in portioning and cycling of organic matter in the benthic region. Since Vembanad estuary is fed by marine and riverine inputs, these peracarids have an important role in structuring the energy requirements of benthic-pelagic region. So, this chapter elaborates the distribution and diversity pattern of peracarids in relation to their spatio-temporal characteristics in the Vembanad estuary. However, there was no detailed information available about the species distribution, diversity and community structure of peracarid crustaceans in Vembanad estuary, especially after the commissioning of the barrage. In this context, understanding the spatial distribution patterns of soft bottom crustaceans along the environmental parameter might help to know the relation between benthic community distribution and its ecological interactions and ultimately the functioning status of the aquatic ecosystem.

7.1.2 Review of Literature

The peracarid crustaceans include amphipods, tanaids, cumaceans, mysids, ostracods and isopods which are distinguished by their direct development of young in a brood pouch. Majority of the benthic community studies are focused on the infauna and a few work has been made on benthic crustaceans (Wu, 1982). The nocturnal vertical migration of benthic amphipods, isopods, cumaceans and mysids were investigated by Stearns and Dardeau (1990). The research demonstrated that the tidal vertical migration of these oligohaline-mesohaline organisms is behavioural mechanisms which retain them in the upper reaches of the estuaries with minimal exposure to visual predation in the water column. Brandt (1995) studied the composition, abundance, and diversity of peracarids (Crustacea) in the Northeast water Polynya, Off Greenland. Diversity, distribution and ecology of benthic amphipods especially Gammaridea in the sublittoral region of Barents Sea were studied by Bryazgin (1997). Benthic crustacean communities in the waters of southwestern Taiwan and their relationships to environmental characteristics were investigated by Chou et al. (1999). They found that benthic crustacean communities are influenced by water depth, bottom substratum quality, and water circulation patterns etc. Cumaceans are an essential component of the benthic fauna, forming an important source of food for benthic macrofauna and demersal fishes (Brandt et al., 1999). Semeniuk (2000) gave details about some benthic crustaceans of the Leschenault Inlet estuary, Australia. The study noted that the spatial distribution of species is linked to salinity patterns or to the presence of aquatic vegetation, and few species had a preference for particular substrates and depths. It was observed that the seasonal variation of crustacean distribution was varied in summer and winter. Similarly, changes in population density and distribution were likely linked to adult migration and juvenile recruitment. The spatial–temporal variation in crustacean assemblages of the Odiel–Tinto estuary, one of the most polluted areas in the world, was studied by Ssnchez-Moyano and Garcia-Asencio (2010). The study found that based on the water and sediment characteristics such as dissolved oxygen, salinity, granulometry and organic content the crustacean assemblages were distinguished from estuarine to marine environment.

Amphipods have increased importance in pollution studies as natural indicators of contaminated conditions. It can tolerate wide range of salinity and sediment types (Marsden et al., 2000). Some amphipod species used as biomonitors of toxin concentration within sediments (Rainbow, 1995) and some species are utilized for bioassay studies (De Witt et al., 1988). Aikins and Kikuchi (2001) conducted field studies on habitat selection and community development of four gammarid amphipod species in a brackish lagoon at Nanakita River Estuary, Japan. The study demonstrated that amphipods mostly selected inhabited substrates with accessible space that give them sufficient protection. Peracarid crustaceans associated with the red alga *Corallina elongata* in Rafraf, Tunisia (Mediterranean Sea) were studied by Zakhama-Sraieb et al. (2011). Tanaids have high sensitivity towards organic matter in the soft bottom sediments and the species *Apseudopsis latreillii* formed dense populations in the Mediterranean Sea and the eastern Atlantic Ocean (de la Ossa Carretero et al., 2011). The species distribution of *Apseudopsis latreillii* in different environmental conditions was controversial, it considered as a tolerant species (Sanz-Lazaro and Marin, 2006; de Juan et al., 2007), but several studies noted that this species does not have any preference to the polluted sediment (Lourido et al., 2008; Moreira et al., 2008). Cruz-Garcia et al. (2013) studied about the abundance of isopods in the coral reef environment in Pacific Mexico. Blazewicz-Paszkowycz et al. (2012) investigated the diversity of Tanaidacea in the World's Oceans. The study found that the diversity of tanaidomorphs remains to be discovered; several smaller taxa are largely understudied; and many cryptic species remain to be identified. The response of tanaid, *Apseudopsis latreillii* to environmental variables in the coastline of Dardanelles, Turkey was investigated by Ates et al. (2014). The study found that gravel size in sediment was the major factor affecting abundance of tanaids. Apart from this, other environmental parameters such as water temperature, currents, and freshwater input from inlands was changing its abundance. Vieira et al. (2016) prepared a DNA barcode reference library for the superorder Peracarida mainly for Amphipoda, Isopoda and Tanaidacea from the Southern European Atlantic coast.

Gammarid and caprellid amphipods of Krusadai Islands in the Gulf of Mannar were studied by Gravely (1927) and Raj (1927) respectively. Several amphipod studies were conducted in various parts of India by Chilton (1920, 1921), Barnard's (1935), John (1955), Nayar (1959), Krishna Pillai (1957, 1961) etc. A preliminary note on the Tanaidacea and Isopoda of Travancore was given by Krishna Pillai (1954). Similarly the description of isopods of family Anthuridae from Kerala was given by Krishna Pillai (1966). The details of 132 species of intertidal Gammarid amphipods from the Indian coasts were given by Surya Rao (1974). Similarly a detail of Caprellid amphipods from India was given by Guerra-Garcia et al. (2009). Das and Roy (1984) gave details about the marine borers of mangroves of little Andaman. Four species of isopods and one species of amphipods were identified during the period, in which one species was amphipod borer *Melita zeylanica* and another species of isopod borer, *Sphaeroma terebrans* were observed. A new species of cymothoid isopod, *Nerocila madrasensis* is described by Ramakrishna and Venkata Ramaniah (1978). Ghatak and Misra (1983) described the occurrence and hermaphroditic nature of the parasitic isopod *Nerocila madrasensis* from the Hooghly Matla estuary. First report of parasitic isopod *Norileca indica* Milne-Edwards, 1840 from northern part of East coast of India was made by Ray et al. (2016). Developmental stages of the tanaidacean, *Ctenapseudes chilkensis* Chilton, and a common forage organism in the brackish water of Kerala were described by Vengayil et al. (1988). Mass culture of *Ctenapseudes chilkensis* was done for other aquaculture experiments in waters of Cochin. Experimental studies on the feeding of *Penaeus indicus* and *Metapenaeus dobsoni* showed that both voraciously feed on *Ctenapseudes chilkensis* (Tanaidacea) and *Eriopsa chilkensis* and *Mera orthonides* (amphipods) as food (Anon, 1977). Patel and Desai (2009) studied the animal-sediment relationships of two benthic communities (crustaceans and polychaetes) around Mandvi coast in the Gulf of Kachchh, Western India. These organisms create assemblages of biogenic structures which helped to understand the various substrate conditions of the micro-geomorphic units such as dune, beach, ridge, runnel and lagoon of the intertidal zone. Diversity of amphipods in the continental shelf sediments of southeast coast of India was investigated by Raja et al. (2013). Seven species of intertidal amphipod crustaceans from Pondicherry mangroves, Southeast Coast of India was described by Satheeshkumar (2011). Dev Roy (2013) dealt with the marine and estuarine isopod fauna of India. In this study a total of 232 species under 101 genera and 25 families have been recorded from different maritime states of India including Andaman and Nicobar Islands and Lakshadweep. The infestation of an isopod crustacean, *Cirolana fluviatius* in the Kumbalangi-Perumpadappu area of the Cochin backwaters were investigated by Mathew et al. (1994). This voracious carnivore attacks the living resources especially the fish and prawns caught in nets particularly in stake net catches. The study observed that the increased population of isopods was mainly associated with the tidal flushing in the region. Due to the construction of bunds, the tidal flushing in the region reduced and the water of the area was linked with increased stagnation and sedimentation, finally varying the benthic composition of the area. The soft bodied animals such as polychaetes, nemertines and nematodes in the regions were consumed by these isopods. Among the benthic faunal studies on the southwest and southeast coast of India, Sarala Devi et al. (1999) encountered six species of amphipods. Other crustacean group includes crabs, mysids, sergestids, cumacea, tanaidacea and anthuridae. In Cochin estuary, Sheeba (2000) noted that among the macrofauna polychaetes were the dominant group consists of 54 species and amphipods were the second dominant groups includes 7species. Other groups include two species of tanaids and isopods; three species of decapods, one species of mysids and cumaceans. Among the macrofauna Anon (2001) observed four species of amphipods in the southern zone of TMB. Similarly among the macrofauna, John (2009) identified nine species of peracarids in Cochin estuary. The life history and population dynamics of *Eriopisa chilkensis* Chilton from Cochin estuary were studied by Aravind (2008). This kind of life cycle study under laboratory conditions was the first time of gammarid amphipod from the south west coast of India. Ecology, diversity, and abundance of macrobenthic crustaceans in the Cochin estuary were investigated by Geetha and Bijoy Nandan (2014). They observed that among the benthic crustaceans, amphipods, isopods and tanaids were the abundant groups. Ecobiology, culture and live feed capability of tanaids of Pulicat lake was investigated by Priya (2015). A total of seven species were identified, *Apseudes gymnophobia*, *Halmyrapseudes killaiyensis*, *Apseudomorpha* sp., *Saltipedis achondroplasia*, *Pakinstanapseudes* sp., *Kalliapseudes* sp. and *Ctenapseudes chilkensis.*

7.2 Results

7.2.1 Composition and spatial distribution of crustacean fauna

Among the crustacean groups, insects, ostracods, harpacticoid copepods, amphipods, tanaids decapods, mysids, cumaceans and isopods were the most abundant groups. During first year, isopods accounted for 42.11% of the total crustacean fauna followed by amphipods (36.79 %), tanaids (12.18 %), insects (4.22 %) and decapods (2.11 %) (Fig.7.1). Spatial percentage compositions of crustacean groups in Vembanad estuary during the first year were given in figure 7.2. While observing the spatial distribution of crustacean groups in station 1, the highest percentage contribution was given by insects (dipterans) (64.78 %) followed by isopods (12.68 %) and tanaids (12.67 %). Similar to station 1, insects were the dominant crustacean fauna contributing 35.56 % to the total population. Isopods and amphipods were the next dominant group contributing 33.33 % and 31.11% in station 2. In station 3, the dominant group of amphipods contributed 79.37 %, followed by isopods (17.47 %) to the total crustacean population. Isopods were the dominant taxon in station 4 contributing 67.65 %, followed by tanaids (21.27 %) and amphipods (10.78 %). In station 5 amphipods, isopods and harpacticoid copepods were equally contributing 24.14 % to the total crustacean fauna. Similar to station 4, isopods were the dominant group contributing 42.38 % in station 6. Amphipods (35.59 %) and tanaids (13.56 %) were formed the next dominant taxa in station 6. Similar to station 6, in station 7 isopods were dominant taxa contributing 49.74 %, followed by amphipods (24.87 %) and tanaids (14.41 %). In station 8 and 9 isopods were the dominant taxon contributing 48.87 % and 70.69 % respectively. In station 10, the major group, amphipods contributed 59.88 % % followed by decapods (12.98 %) and tanaids (10.91 %). Among insects, one *Chironomus* sp. and one species from family Culcidae were identified. Decapods include juvenile of one *Macrobrachium* sp., one *Alpheus* sp., two *Metapenaeus* sp. and one crab *Portunus pelagicus* were observed and among these *Alpheus* sp. were more abundant during the study period. Among the peracarid crustacean species, from tanaids; *Ctenapseudes chilkensis*, from isopods; *Cirolana fluviatilis*, *Apanthura sandalensis*, *Xenanthura orientalis*; from amphipods; *Eriopisa chilkensis*, *Eriopisa* sp., *Cheiriphotis geniculata*, *Gammarus* sp., from cumacean; *Bodotria* sp. were identified (Plate 7.1).

In contrast with first year, among the crustacean fauna amphipods were the dominant group (55.53 %) followed by isopods (36.39%), tanaids (5.06 %) and insects (2.34 %) in second year (Fig. 7.3). Other taxon such as ostracods, harpacticoid copepods, decapods and mysids contribute < 0.5 % to the total crustacean population. By observing the spatial distribution of crustacean fauna during second year (Fig. 7.4), it was noted that in each station the percentage contribution of each taxon varied annually. In contrast with first year in station 1 and 2, amphipods were the dominant group contributing 93.52 % and 96.24 % in station 1 and 2 respectively. Similar to first year in station 3, amphipods were the dominant taxon contributed maximum 81.72 % to the crustacean density. In station 4 isopods were the dominant taxon that contributing 81.96 %, followed by amphipods (13.82%)

and tanaids (2.06 %). From station 5 to station 9 onwards isopods were the dominant taxon followed by amphipods and tanaids. Isopods contributing 57.92 %, 51.51 %, 57.65 %, 46.37 % and 51.61% to stations 5, 6, 7, 8 and 9 respectively. Amphipods were the dominant group in station 10, contributing 54.06 % followed by isopods (20.63 %) and tanaids (9.38 %). Among insects (dipterans), two *Chironomus* sp. and one species from family Culcidae were identified. Decapods include juvenile of one *Macrobrachium* sp., one *Alpheus* sp., one *Metapenaeus* sp. and one crab *Portunus pelagicus* were observed. Similar to first year, *Alpheus* sp. were more abundant among decapods. Among the peracarid crustacean species, from tanaids; *Ctenapseudes chilkensis*, from isopods; *Cirolana fluviatilis*, *Apanthura sandalensis*, *Xenanthura orientalis*; from amphipods; *Eriopisa chilkensis*, *Eriopisa* sp., *Cheiriphotis geniculata*, *Gammarus* sp., were identified.

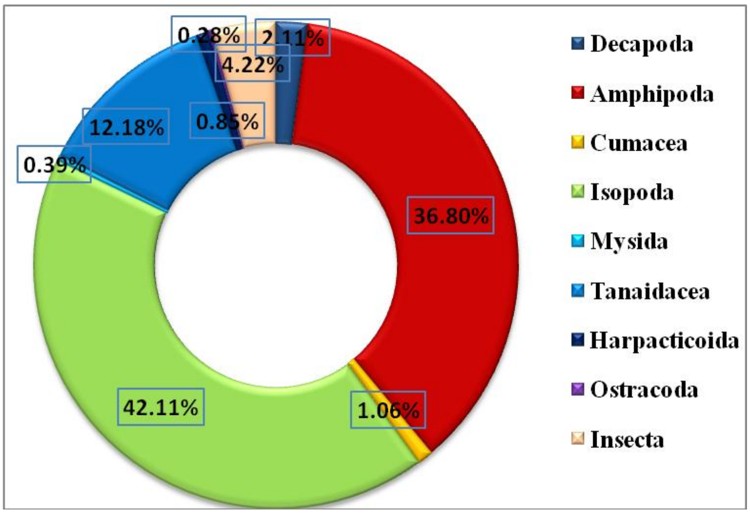


Fig. 7.1 Percentage composition of crustacean groups in Vembanad estuary during 2011 – 2012 period.

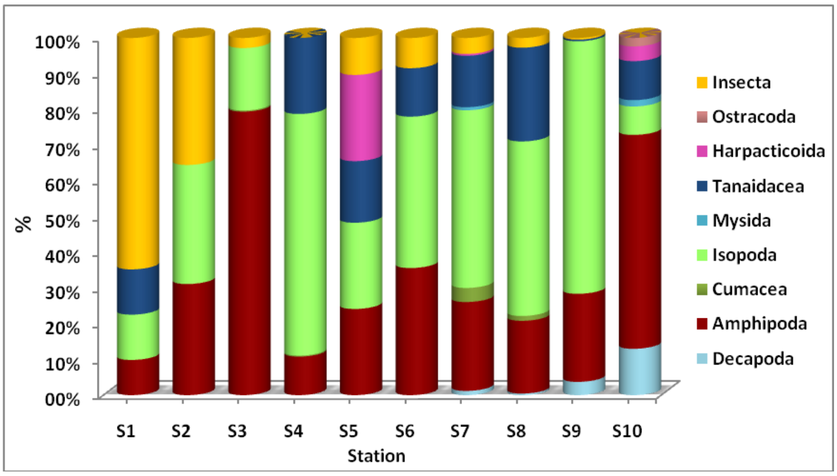


Fig. 7.2 Spatial percentage composition of crustacean groups in Vembanad estuary during 2011 – 2012 period.

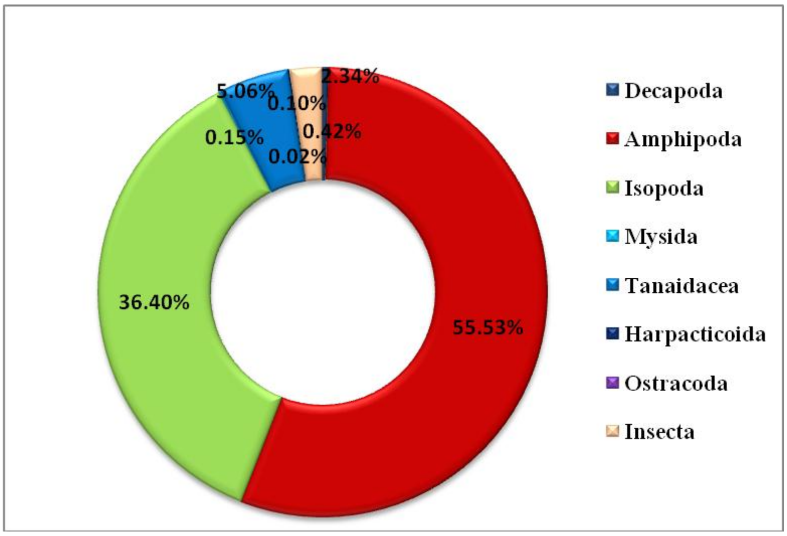


Fig. 7.3 Percentage composition of crustacean groups in Vembanad estuary during 2012 – 2013 period.

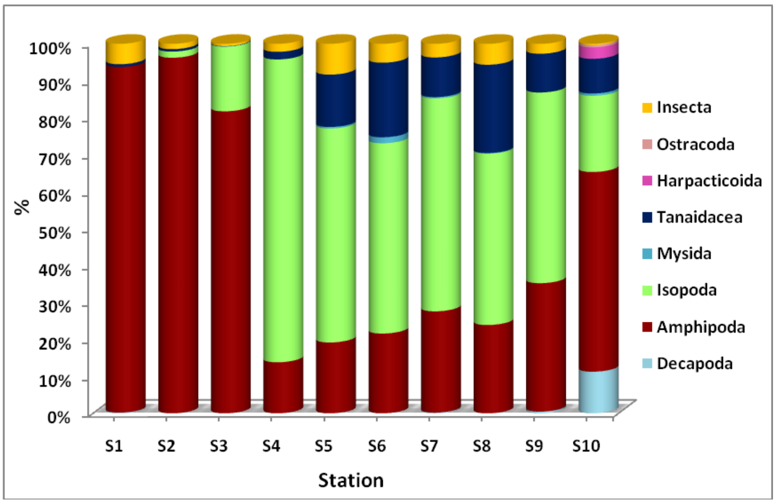


Fig. 7.4 Spatial percentage composition of crustacean groups in Vembanad estuary during 2012 – 2013 period.

7.2.2 Spatio-temporal variation of Peracarids in the Vembanad estuarine system

Ctenapseudes chilkensis, C. geniculata and *X. orientalis* were the three peracarid species present in station 1. In which the annual average density for *C. chilkensis, C. geniculata* and *X. orientalis* were 17.03 ± 42.33 ind./m^2, 13.24 ± 24.59 ind./m^2 and 17.03 ± 59.01 ind./m^2 respectively. The species *C. chilkensis* were present only during April 2011 (68 ind./m^2) and September 2011 (136 ind./m^2), whereas *X. orientalis* were present only during April 2011 (204 ind./m^2). In the case of *C. geniculata*, it was present during April 2011(45 ind./m^2), January 2012 (68 ind./m^2) and February 2012 (45 ind./m^2). During 2012-2013 period, *C. chilkensis* and *C. geniculata* were the two peracarids present in station 1. The *C. chilkensis* were observed only during October 2012 (91 ind./m^2) and *C. geniculata* were observed throughout the period except November and December 2012. Its maximum density was observed during August 2012 (3450 ind./m^2), followed by January 2013 (2588 ind./m^2) with an average density of 931 ± 1106 ind./m^2.

During 2011-2012 period, two isopod species *A. sandalensis* and *X. orientalis* and one amphipod *C. geniculata* were observed in station 2. The species *A. sandalensis* were present only during March 2012 (91 ind./m^2) and *X. orientalis* were observed during March (204 ind./m^2) and May 2012 (45 ind./m^2). The maximum density of *C. geniculata* were observed during January 2013 (159 ind./m^2) and its annual average was 27 ± 54 ind./m^2. During second year four peracarids were observed in station 2. *C. geniculata* showed maximum annual average density (3292 ± 2992 ind./m^2) followed by *X. orientalis* (49.18 ± 139 ind./m^2), *C. chilkensis* (23 ± 66 ind./m^2) and *A. sandalensis* (10 ± 20ind./m^2).

During the first year, four species of peracarids were observed in station 3, in which *C. geniculata* have maximum average density (902 ± 1739 ind./m^2), followed by *X. orientalis* (155 ± 304 ind./m^2), *A. sandalensis* (44 ± 84 ind./m^2) and *Bodotria* sp. (4 ± 9 ind./m^2). While observing the monthly distribution, most of the species were observed only a few months. The maximum density of both *A. sandalensis* and *X. orientalis* were observed during September 2011. In the case of *C. geniculata* maximum density was observed during April 2011 (5198 ind./m^2), followed by February 2012 (3904 ind./m^2). During second year five species of peracarids were observed in station 3. Similar to first year, *C. geniculata* have maximum average density (4254 ± 6833 ind./m^2), followed by *X. orientalis* (777 ± 1310 ind./m^2) and *A. sandalensis* (132 ± 201 ind./m^2). The species *X. orientalis* and *C. geniculata* were observed in most of the months and its maximum density was observed during August 2012 (4245 ind./m^2) and May 2012 (19340 ind./m^2).

During 2011-2012 period, five species of peracarids were observed in station 4. In contrast with station 2 and 3, the species *C. chilkensis* were observed in this station. Its maximum density was observed during November 2011 (1771 ind./m^2) followed by March 2011 (476 ind./ m^2). The maximum density of *A. sandalensis* (863 ind./m^2) and *X. orientalis* (2088 ind./m^2) were observed during February 2011 and March 2011 respectively. Similar to *X. orientalis*, the maximum density of *C. geniculata* (1022 ind./m^2) were observed during March 2011. Comparing the annual average density, maximum density was observed in *X. orientalis* (602 ± 697 ind./m^2) followed by *C. chilkensis* (272 ± 504 ind./m^2), *A. sandalensis* (264 ± 339 ind./m^2) and *C. geniculata* (138 ± 287 ind./m^2). During second year period *X. orientalis* (2408 ± 2571 ind./m^2) showed maximum average density, followed by *A. sandalensis* (738 ± 918 ind./m^2), *C. geniculata* (534 ± 867 ind./m^2) and *C. chilkensis* (79 ± 181 ind./m^2).

During the first year, compared to other southern stations, the density of peracarids were very low in station 5. The maximum value of annual average density was observed among *C. geniculata* (13 ± 31 ind./m^2), followed by *C. chilkensis* (10 ± 33 ind./m^2). In contrast with first year, the density of peracarids was higher in second year. The annual average density was maximum for *X. orientalis* (218 ± 307 ind./m^2) followed by *C. geniculata* (87 ± 188 ind./m^2) and *C. chilkensis* (23 ± 66 ind./m^2).

Similar to station 6 in first year, the annual average density of peracarids were also very lower in second year. *A. sandalensis* showed maximum average density (36 ± 57 ind./m^2) followed by *Gammarus* sp. (32 ± 43 ind./m^2). In the case of *A. sandalensis* the maximum monthly abundance was observed during June 2011 (182 ind./m^2) and of *Gammarus* sp. maximum abundance (113 ind./m^2) observed during April 2011 and September 2011. During second year, the maximum abundance of *C. chilkensis* (612 ind./m^2) was observed during May 2012 and also it was not encountered in many months. Its annual density (166 ± 194 ind./m^2) was low to *X. orientalis* (367 ± 477 ind./m^2). In the case of *A. sandalensis,* the monthly maximum abundance was observed during October 2012 (274 ind./m^2).

During the first year in station 7, among the peracarids, the maximum average density was shown by *X. orientalis* (518 ± 914 ind./m^2), *Gammarus* sp. (204 ± 379 ind./m^2), *C. chilkensis* (159 ± 459 ind./m^2), *E. chilkensis* (70 ± 149 ind./m^2) and *A. sandalensis* (30 ± 52 ind./m^2). *C. chilkensis* were found only in few months with its maximum density was observed during September 2011 (1612 ind./m^2). Similarly *A. sandalensis* were observed only in few numbers in limited months with its maximum density that was observed during September 2011 (136 ind./m^2). In the case of *X. orientalis*, it was present in most of the months except April 2011 and October 2011 and its maximum density was observed during September 2011 (3269 ind./m^2). The maximum density of *Gammarus* sp. was observed during August 2011 (1158 ind./m^2). During second year, similar to first year the distribution of *C. chilkensis* was present only in few months with in station 7. Its maximum density was observed during November 2012 (477 ind./m^2) with annual average of 115 ± 159 ind./m^2. Among other species, *X. orientalis* (489 ± 595 ind./m^2) having maximum average density followed by *Gammarus* sp. (257 ± 331 ind./m^2) and *A. sandalensis* (131± 218 ind./m^2). The maximum density of *X. orientalis* were observed during December 2012 (1748 ind./m^2) and August 2012 (1407 ind./m^2). In this station, *Gammarus* sp. was observed throughout the period except during November 2012. Its maximum density was observed during May 2012 (1226 ind./m^2), followed by March 2012 (477 ind./m^2). In this station *C. geniculata* was not observed during both the years.

During the first year in station 8, among the peracarid species the maximum annual density was observed in *X. orientalis* (157 ± 299 ind./m^2), followed by *C. chilkensis* (110 ± 158 ind./m^2), *Gammarus* sp. (61 ± 75 ind./m^2) and *A. sandalensis* (32 ± 78 ind./m^2). Similar to first year the annual average density of peracarids was higher in *X. orientalis* (411 ± 561 ind./m^2), followed by *C. chilkensis* (255 ± 359 ind./m^2), *Gammarus* sp. (210 ± 152 ind./m^2) and

A. sandalensis (85 ± 160 ind./m^2). The *Gammarus* sp. was observed all the moths except December 2012. Its maximum density was observed during March 2012 (477 ind./m^2) and followed by May 2012 (386 ind./m^2) and July 2012 (386 ind./m^2). Similar to *Gammarus* sp., the isopod *X. orientalis* was observed all the moths except December 2012. Its maximum density was observed during February 2013 (1975 ind./m^2) and followed by April 2012 (976 ind./m^2).

During the first year, the density of peracarid species was comparatively lower in station 9. The maximum average density was shown by *Gammarus* sp. (53 ± 79 ind./m^2) and followed by *E. chilkensis* (47 ± 80 ind./m^2) and *X. orientalis* (2 ± 7 ind./m^2). Compared to first year, the density of peracarid species was increasing during second year, in which the density of *X. orientalis* increased to 1089 ± 3521 ind./m^2 and that of *Gammarus* sp. to 373 ± 344 ind./m^2. In this station the *Gammarus* sp. was observed most of the period, except February 2013. Its maximum abundance of 976 ind./m^2 was observed during July 2012. In contrast to first year, the species *Eriopisa* sp. was present in this station with an annual average of 64 ± 85 ind./m^2 and its monthly maximum density was observed during July 2012 (272 ind./m^2). The monthly abundance of *C. chilkensis* was maximum during December 2012 (1362 ind./m^2), followed by May 2012 (409 ind./m^2).

During the first year in station 10, the maximum average density was shown by *E. chilkensis* (246 ± 280 ind./m^2) subsequently *Gammarus* sp. (79 ± 141 ind./m^2), *C. chilkensis* (70 ± 145 ind./m^2), *Eriopisa* sp.(57 ± 95 ind./m^2), *Cirolana fluviatilis* (51 ± 109 ind./m^2) and mysids (11 ± 15 ind./m^2). During second year the annual density was maximum for *Gammarus* sp. (327 ± 713 ind./m^2) and subsequently *Cirolana fluviatilis* (125 ± 180 ind./m^2) and *C. chilkensis* (57 ± 163 ind./m^2). The monthly distribution of peracarids noted that most of the species were observed only a few months in a year. The maximum density of *Cirolana fluviatilis* was observed during February

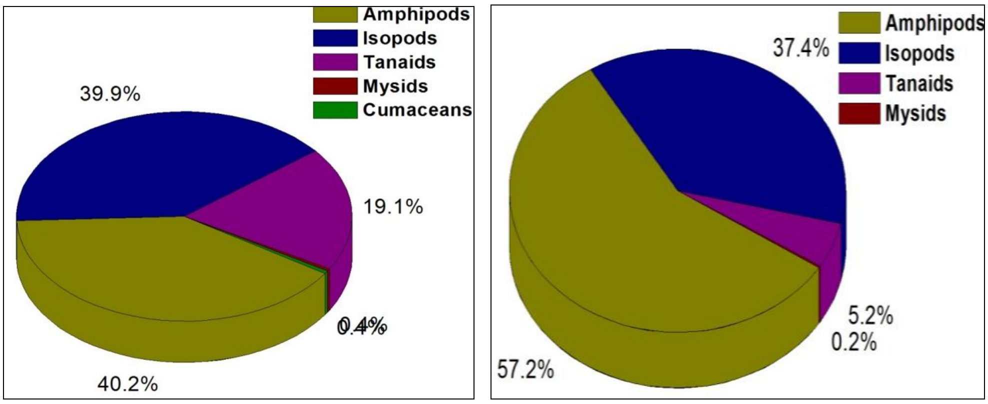
2013 (636 ind./m^2) and that of *Gammarus* sp. was observed during May 2012 (2497 ind./m^2). Similar to *Gammarus* sp., the maximum density of *C. chilkensis* was observed during May 2012 (568 ind./m^2).

7.2.3 Overall spatial and temporal variation of Peracarids

During first year, amphipods accounted for 40.17 % of the total peracarids followed by isopods (39.9 %), tanaids (19.12 %), whereas during second year amphipods contributed 57.23 %, followed by isopods (37.4 %) and tanaids (5.2 %) (Fig. 7.5). During 2011-2012 period, from the isopod species *X. orientalis* contributed 75.9 % followed by *A. sandalensis* (21.5 %) and *C. fluviatilis* (2.6 %) and from amphipods *C. geniculata* contributed 55.75 %, followed by *Gammarus* sp. (21.7 %), *E. chilkensis* (19.6 %) and *Eriopisa* sp. (2.9 %). Similar to first year, during second year among isopods *X. orientalis* was the dominant species (81.5 %) followed by *A. sandalensis* (16.7 %) and *C. fluviatilis* (1.8 %). Similar to first year, among amphipods *C. geniculata* showed maximum percentage contribution (84.3 %) followed by *Gammarus* sp. (11.4 %), *E. chilkensis* (3.7 %) and *Eriopisa* sp. (0.6 %) during second year. During first year, among the total peracarid species, *X. orientalis* was the dominant species contributing 30.3 % followed by *C. geniculata* (22.4 %) and *C. chilkensis* (19.1 %), whereas during second year *C. geniculata* contributed the major share (48.2 %) followed by *X. orientalis* (30.48 %) (Fig.7.6). Temporal variation in the species abundance of peracarids is given in Table 7.1 and Table 7.2. The bubble plot denotes the average spatial abundance of peracarid crustacean species present in the Vembanad estuarine system during first year (Fig. 7.7 (a-h)) and second year (Fig.7.8 (a-h)). During first year, *C. chilkensis* was present in most of the stations except station 2 and 3. Its maximum abundance was observed in station 4 (1771 ind./m^2) during November 2011, whereas during second year *C. chilkensis* observed in all the stations but its frequency of observation varied with a single observation in station 1 and 3. Comparing the spatial average density, the maximum was observed in station 9 (288 ind./m^2) during first year and station 8 (255 ind./m^2) during second year. The temporal variation of *C. chilkensis* noted that during first and second year the maximum annual average was observed during September 2011 (281 ind./m^2) and May 2012 (291 ind./m^2) respectively. The ANOVA analysis showed that the spatial variation of *C. chilkensis* were significant at both first year ($P \leq 0.05$) and second year ($P \leq 0.05$). Both the years, *C. fluviatilis* was observed only in station 10, but it was not regularly found in all the months. The ANOVA result demonstrated that, the spatial variation of *C. fluviatilis* was significant during first year at 1% level ($P \leq 0.01$) and second year at 5 % level ($P \leq 0.05$). In both the years, the isopod species *A. sandalensis* were observed in station 2 to 8 but was not observed in station 1, 9 and 10. During both the years the average maximum density of 265 ind./m^2 and 738 ind./m^2 of *A. sandalensis* was observed in station 4 during first and second year respectively. During first and second year, its temporal maximum was observed during March 2011 (107 ind./m^2) and July 2012 (361 ind./m^2) respectively. The ANOVA result showed that the spatial variation of *A. sandalensis* were significant at both first year ($P \leq 0.01$) and second year ($P \leq 0.01$) at 1% level. During first year, *X. orientalis* was observed in all stations except station 10 and its maximum density was observed in station 4 (602 ind./m^2) and station 7 (518 ind./m^2). During second year *X. orientalis* was not observed in station 1 and 10, with its maximum spatial abundance was noted in station 4 (2408 ind./m^2) and station 9 (1090 ind./m^2). Its temporal variation noted that maximum average density was observed in monsoon months during first year (September 2011 - 434 ind./m^2) and second year (July 2012 - 1085 ind./m^2) respectively. The ANOVA result showed that the spatial variation of *X. orientalis* was significant in both first year ($P \leq 0.01$) and second year ($P \leq 0.01$) at 1% level. In the case of mysids, during first year it was observed only in station 7 and 10. But the frequency of observation was very few in these stations.

During second year, it was present in stations 3, 5, 6, 7, 8 and 10. But in majority of the stations it was observed only in a single time. Similar to mysids the observation of *Bodotria* sp. was very few in stations 3, 4, 7 and 8 during first year, whereas during second year period, it was not encountered. In the case of amphipod species *E. chilkensis,* its distribution was confined in the northern stations only. During first year it was observed in station 7 to 10 with maximum spatial average was observed in station 10 with a value of 246 ind./m^2 and minimum was observed in station 8 (25 ind./m^2). But during second year *E. chilkensis* were observed in three stations such as stations 7, 8 and 9. Its maximum abundance was observed in station 9 (325 ind./m^2) and minimum during station 7 (36 ind./m^2). ANOVA results revealed that the variation of *E. chilkensis* were significant at 1% level both during first year ($P \leq 0.01$) and second year ($P \leq 0.01$). Contrast to the distribution of *E. chilkensis,* the species *C. geniculata* was present only in the southern stations. During first year the maximum density of *C. geniculata* was observed in station 3 (4254 ind./m^2), followed by station 2 (3292 ind./m^2) and minimum was observed in station 5 (87 ind./m^2). It was also noted that this particular species rarely forms tubes in station 3. During both the years, the ANOVA analysis showed that the spatial variation of *C. geniculata* were significant at 1% level during first year ($P \leq 0.01$) and second year ($P \leq 0.01$). Similar to the spatial distribution pattern of *E. chilkensis,* the *Gammarus* sp. was distributed in all the northern stations and also in station 6. During first year its maximum spatial density was observed in station 7 (204 ind./m^2) and minimum in station 6 (32 ind./m^2). Whereas during second year, the maximum density of *Gammarus* sp. was observed in station 9 (372 ind./m^2) followed by station 10 (327 ind./m^2) and station 7 (257 ind./m^2). Similar to first year, its minimum density was observed in station 6 (78 ind./m^2). In the case of *Eriopisa* sp. distribution, it was present only in station 10 during first year and station 9 during second year. During first year the spatial variation of *Gammarus* sp. ($P \leq 0.01$) and *Eriopisa* sp. ($P \leq 0.01$) were significant at

1% level. Similar to first year, the spatial variation of *Gammarus* sp. ($P \leq 0.01$) and *Eriopisa* sp. ($P \leq 0.01$) were significant at 1% level.



2011 – 2012 **2012 – 2013**

Fig.7.5 Mean percentage contribution of peracarids in Vembanad estuary during 2011 – 2012 and 2012 – 2013 period

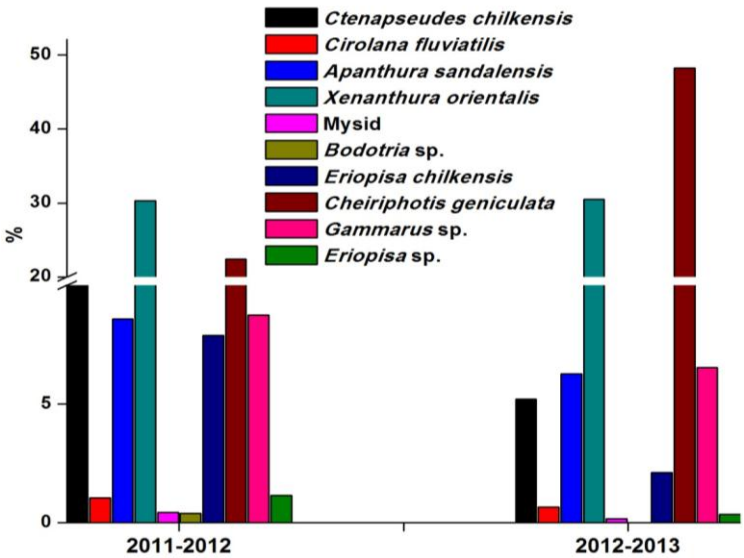
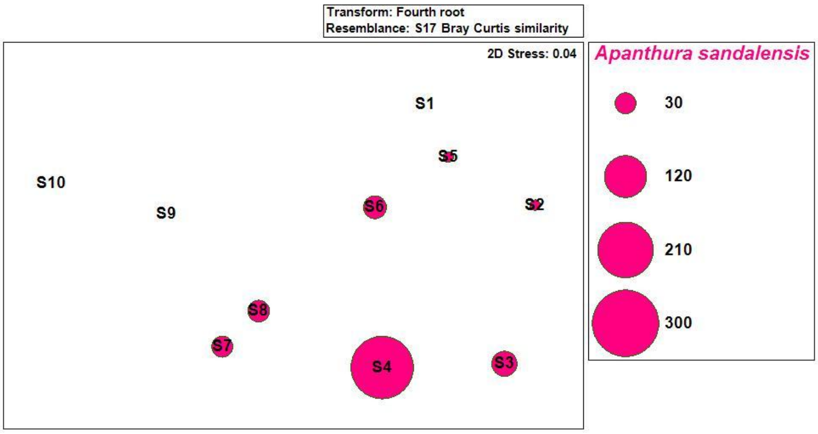


Fig. 7.6 Mean percentage contribution of peracarid species in Vembanad estuary during 2011 – 2013 period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2D Stress: 0.04
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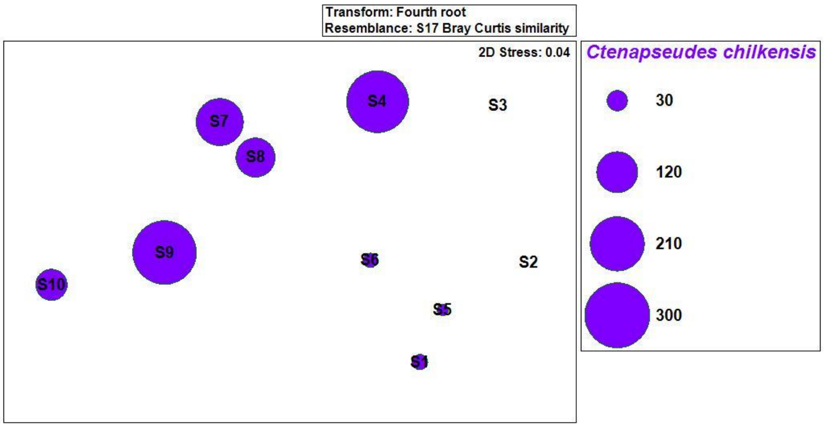
30
120
210
300
S1
S10
S9
S6
S8
S7
S4
S3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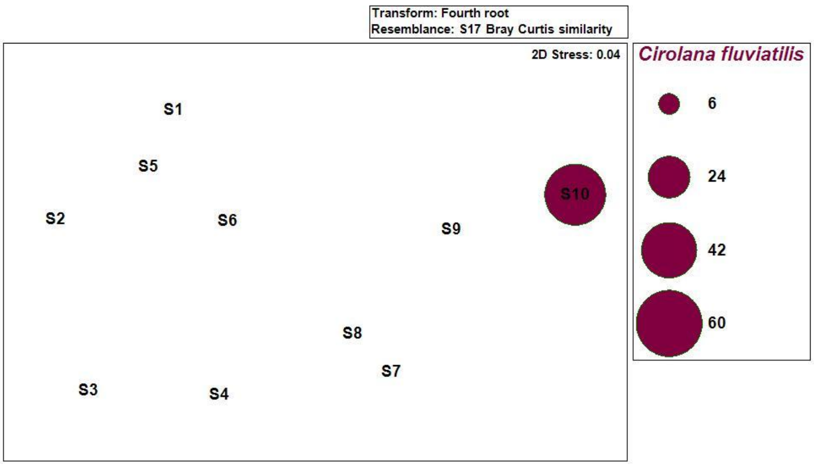
Resemblance: S17 Bray Curtis similarity
2D Stress: 0.04
Ctenapseudes chilkensis
30
120
210
300
S4
S3
S7
S8
S9
S2
S10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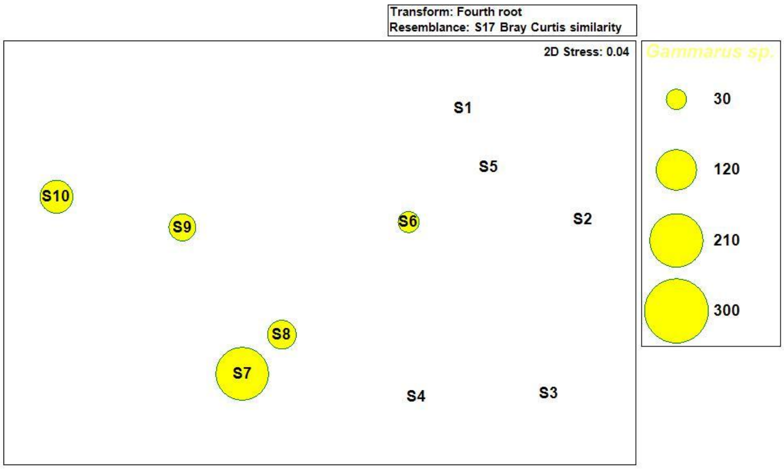
2D Stress: 0.04
Cirolana fluviatilis
6
24
42
60
S1
S5
S10
S2
S6
S9
S8
S7
S3
S4


Transform: Fourth root
Resemblance: S17 Bray Curtis similarity
2D Stress: 0.04
Eriopisa chilkensis
30
120
210
300
S3
S4
S7
S8
S2
S6
S9
S10
S5
S1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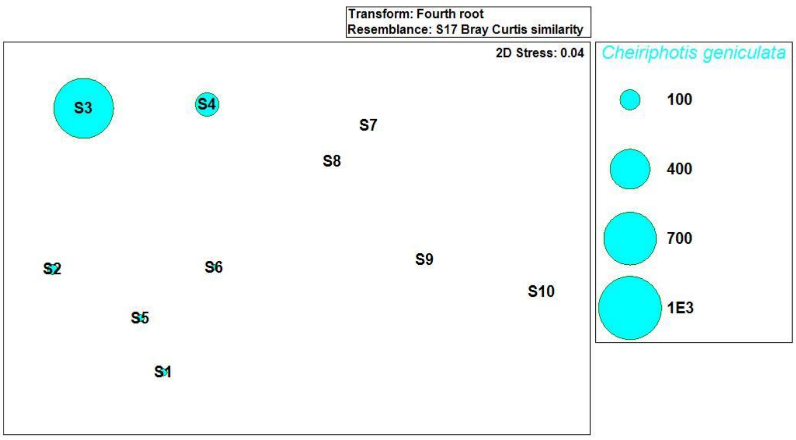
2D Stress: 0.04
Gammarus sp.
30
120
210
300
S1
S5
S10
S9
S6
S2
S8
S7
S4
S3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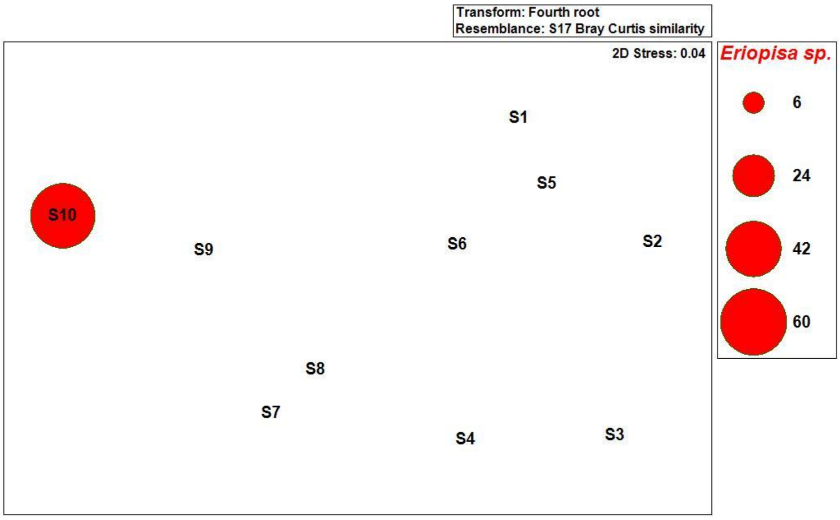
Resemblance: S17 Bray Curtis similarity
2D Stress: 0.04
Cheiriphotis geniculata
100
400
700
1E3
S3
S4
S7
S8
S2
S6
S9
S10
S5
S1



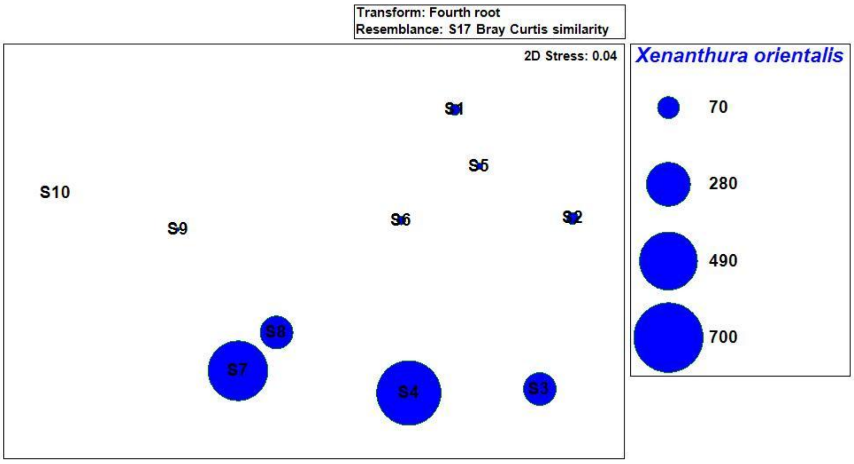
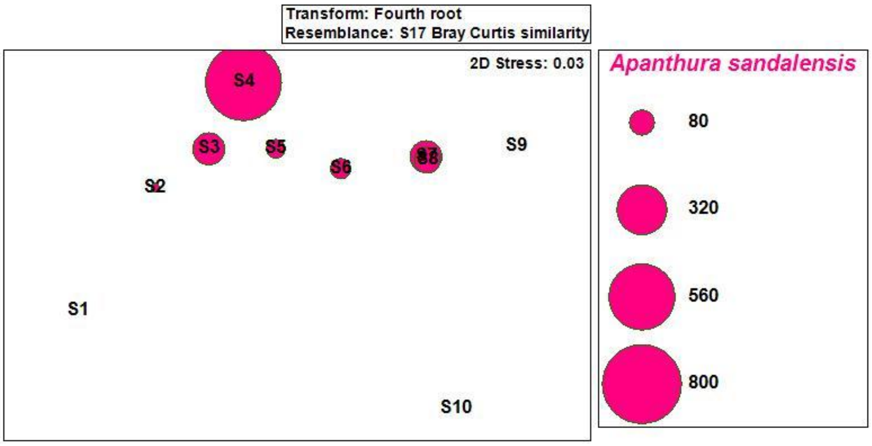
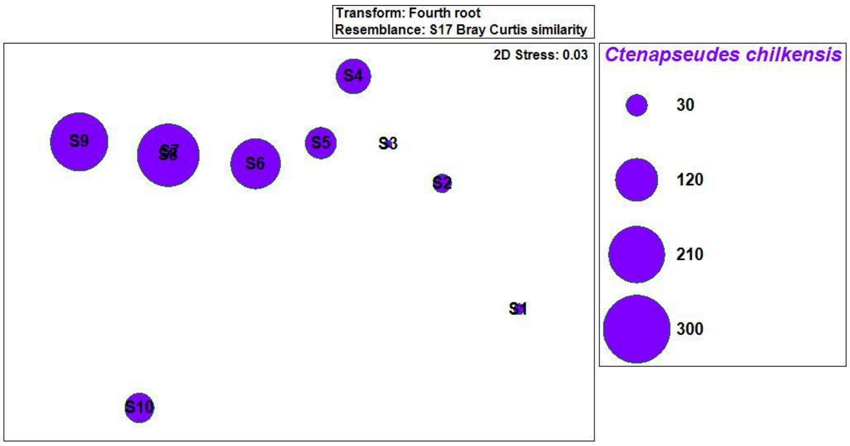
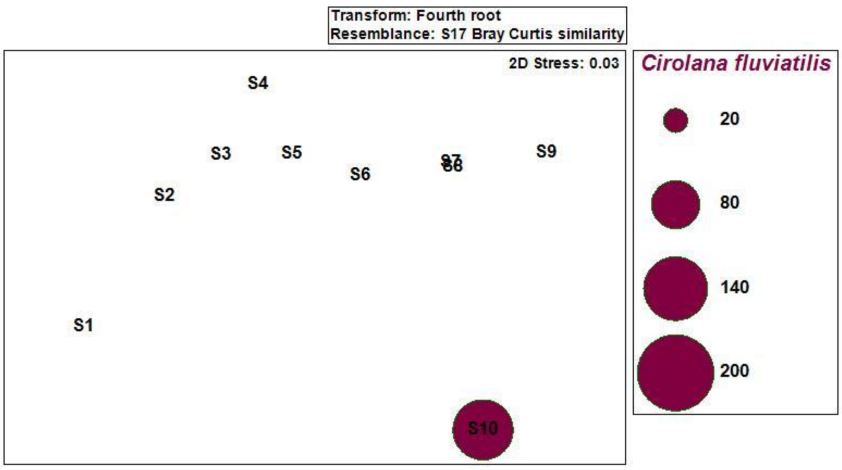


Fig. 7.7 a-h Bubble plots showing the spatial variation of peracarid crustaceans in Vembanad estuary during 2011 – 2012 period.


Transform: Fourth root
Resemblance: S17 Bray Curtis similarity
2D Stress: 0.03
Apanthura sandalensis
80
320
560
800
S1
S2
S3
S4
S5
S6
S9
S10


Transform: Fourth root
Resemblance: S17 Bray Curtis similarity
2D Stress: 0.03
Ctenapseudes chilkensis
30
120
210
300
S4
S9
S6
S5
S10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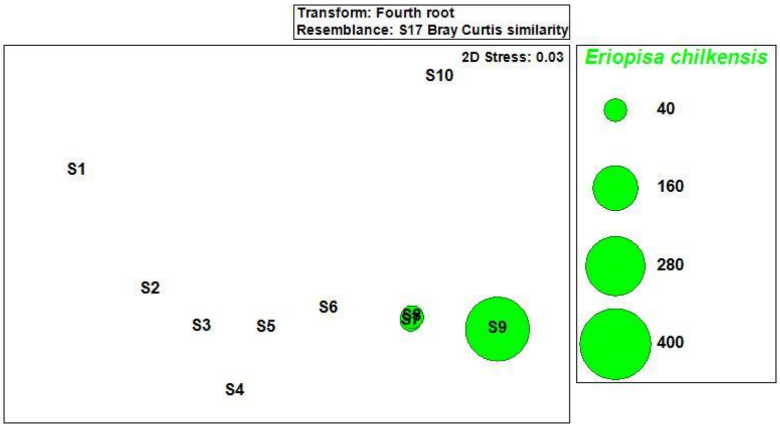
Resemblance: S17 Bray Curtis similarity
2D Stress: 0.03
Cirolana fluviatilis
20
80
140
200
S4
S3
S5
S6
S9
S2
S1
S10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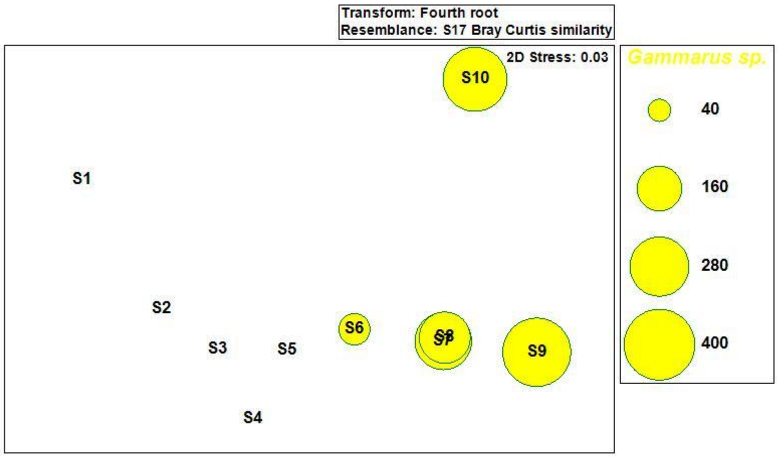
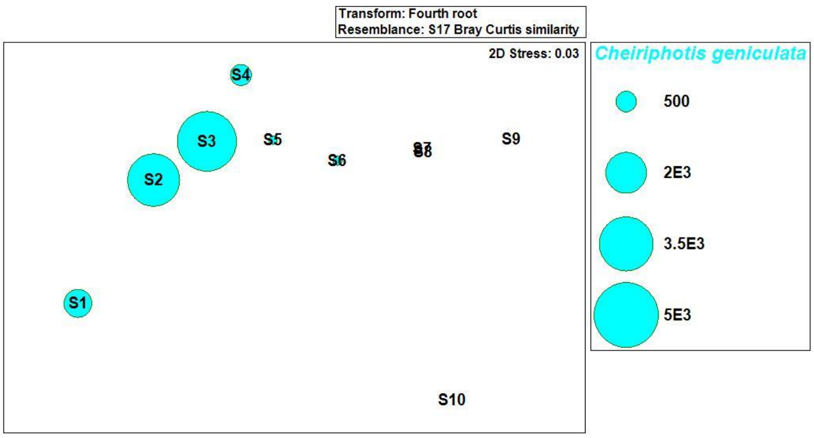
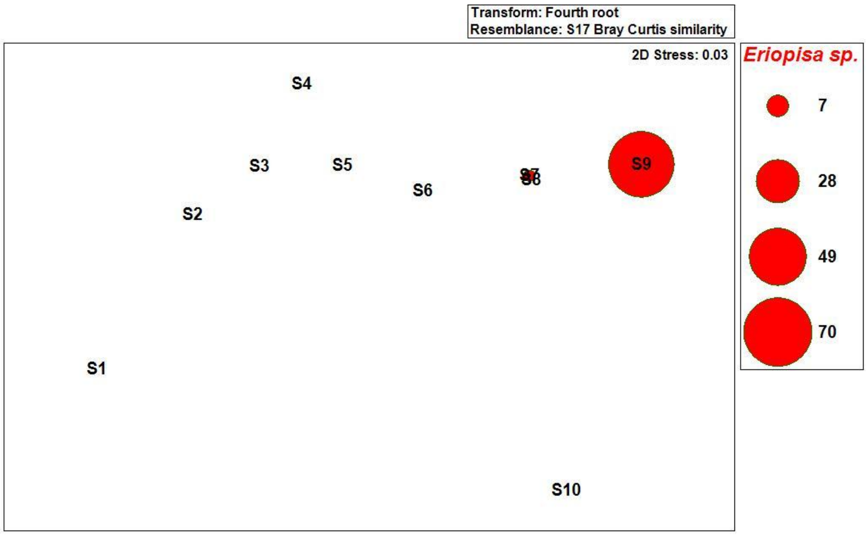
Resemblance: S17 Bray Curtis similarity
2D Stress: 0.03
S10
S1
S2
S3
S5
S6
S7
S8
S9
S4
Eriopisa chilkensis
40
160
280
400


Transform: Fourth root
Resemblance: S17 Bray Curtis similarity
2D Stress: 0.03
S10
S1
S2
S3
S5
S6
S7
S8
S9
S4
Gammarus sp.
40
160
280
400


Transform: Fourth root
Resemblance: S17 Bray Curtis similarity
2D Stress: 0.03
S4
S3
S5
S6
S7
S8
S9
S2
S1
S10
Cheiriphotis geniculata
500
2E3
3.5E3
5E3



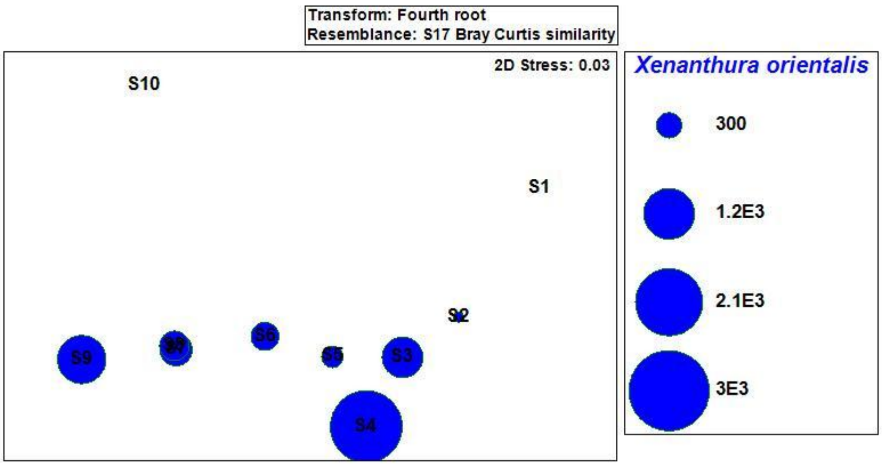


Fig.7.8 a-h Bubble plots showing the spatial variation of peracarid crustaceans in Vembanad estuary during 2012 – 2013 period.

Table 7.1 Temporal variation of peracarids (ind./m^2)in Vembanad estuary during 2011-2012 period

	Ctenapseudes chilkensis	*Cirolana fluviatilis*	*Apanthura sandalensis*	*Xenanthura orientalis*	**Mysida**	***Bodotria* sp.**	*Eriopisa chilkensis*	*Cheiriphotis geniculata*	*Gammarus* **sp.**	***Eriopisa* sp.**
MAR'11	91	5	107	227	0	0	0	75	2	0
APR' 11	52	0	82	250	0	7	59	640	59	0
MAY'11	27	0	2	95	0	0	39	52	48	27
JUN'11	2	0	48	152	0	2	0	25	2	0
JULY'11	36	0	11	57	2	0	27	16	41	20
AUG'11	0	0	5	7	0	2	54	11	154	0
SEPT'11	281	0	41	434	0	5	18	36	39	5
OCT'11	127	5	20	123	2	2	54	0	111	0
NOV'11	279	0	98	209	14	5	98	0	34	14
DEC'11	93	39	70	145	2	0	34	14	2	0
JAN'12	41	7	18	70	5	0	41	57	7	0
FEB'12	98	7	5	18	0	0	41	395	16	2

Table 7.2 Temporal variation of peracarids (ind./m^2) in Vembanad estuary during 2012-2013 period

	Ctenapseudes chilkensis	*Cirolana fluviatilis*	*Apanthura sandalensis*	*Xenanthura orientalis*	**Mysida**	***Bodotria* sp.**	*Eriopisa chilkensis*	*Cheiriphotis geniculata*	*Gammarus* **sp.**	***Eriopisa* sp.**
MAR'12	77	11	5	152	2	0	95	572	243	18
APR' 12	9	16	127	513	0	0	9	93	45	5
MAY'12	291	18	27	179	0	0	59	2377	502	7
JUN'12	107	0	163	690	2	0	48	1276	116	5
JULY'12	93	0	361	1085	2	0	32	860	168	27
AUG'12	16	0	132	697	2	0	16	370	50	0
SEPT'12	73	0	100	565	0	0	0	354	43	7
OCT'12	50	5	116	400	14	0	41	341	32	0
NOV'12	125	0	86	400	2	0	0	795	18	0
DEC'12	213	18	84	1625	2	0	129	1037	136	9
JAN'13	0	18	102	213	9	0	52	2604	89	0
FEB'13	136	64	129	452	0	0	2	354	52	2

7.2.4 Seasonal variation of Peracarids

By observing the overall seasonal variation of peracaridan species, its percentage contribution was distinct in each of the seasons. During first year the maximum population density was found during PRM (42.8 %) followed by PM (31 %) and MN (26 %), whereas during second year, the maximum density was observed during PM (38 %) followed by MN (32.6 %) and PRM (29.1 %) (Fig.7.9). During first year, the percentage contribution of *C. geniculata* (46 %) was maximum followed by *X. orientalis* (23.4 %), *C. chilkensis* (10.6 %), *A. sandalensis* (7.7 %) and *E. chilkensis* (5.5 %) during PRM season, whereas during MN, *X. orientalis* was the dominant species (42.3 %) followed by *C. chilkensis* (20.86 %) (Fig. 7.10). The third dominant density was shown by *Gammarus* sp. that contributed 15.38 %. Compared to PRM, the percentage abundance of *C. geniculata* was decreased during MN (5.8 %) and PM (3.8 %). Similar to MN, the percentage abundance of *X. orientalis* was rather higher during PM (29.8 %). During PM, the second and third dominant density was observed by *C. chilkensis* (29.4 %) and *E. chilkensis* (12.4 %). Similar to first year in PRM, among the peracarids *C. geniculata* (51.1 %) and *X. orientalis* (19.5 %) have higher density during second year. During MN season the percentage contribution of *X. orientalis* increased upto 40.7 % and become the dominant species among peracarids. In contrast with the MN and PM of first year, the density of *C. geniculata* was higher during both seasons in the second year. During MN, it becomes the second dominant species that contributed 38.3 % and at the same time during PM, it became the dominant species (54.5 %).

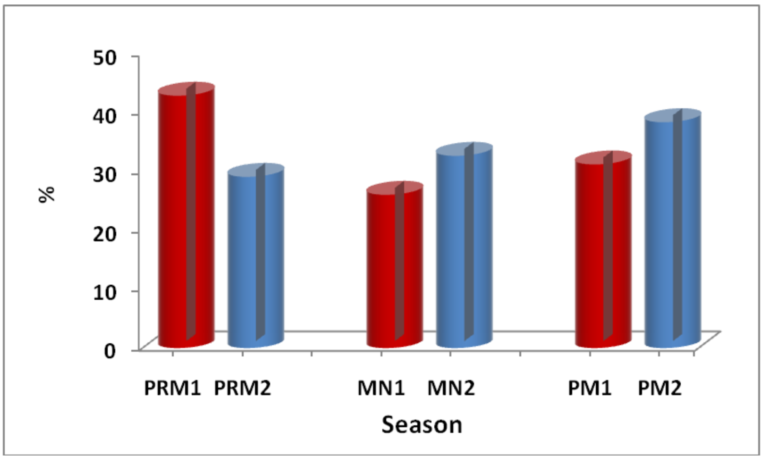


Fig. 7.9 Seasonal mean percentage contribution of peracarids in Vembanad estuary during 2011 – 2013 period

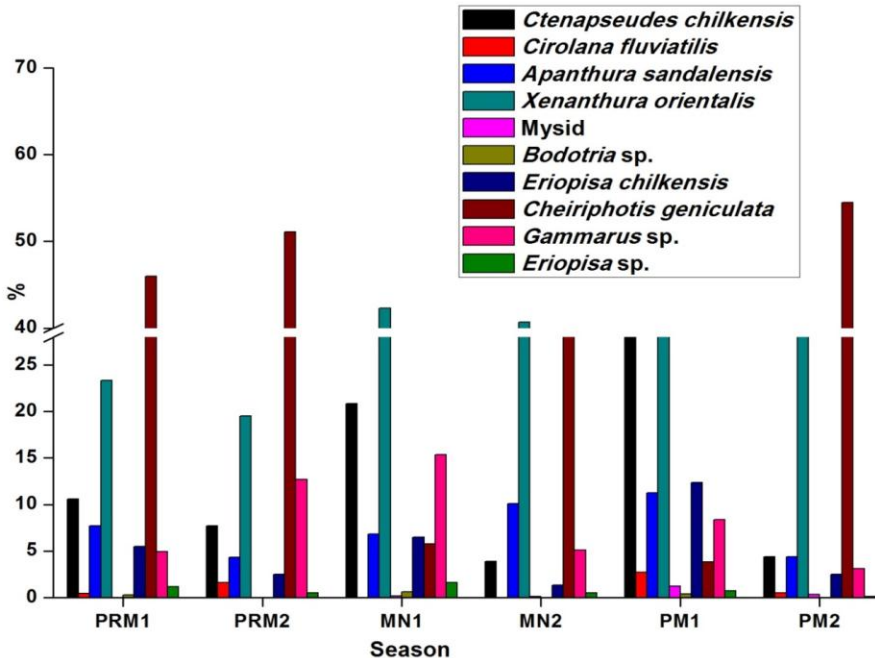


Fig. 7.10 Seasonal variation of percentage composition of peracarids in Vembanad estuary during 2011-2013 period

7.2.5 Data Analysis

7.2.5.1 Variation in diversity pattern and taxonomic distinctness of peracarids in Vembanad estuary

The annual spatial abundance of peracarid crustacean species was used for the analysis of diversity indices**.** During first year, the Shannon diversity index was maximum in station 10 (2.12) and minimum in station 3 (0.85) with annual spatial average of 1.69 (Fig. 7.11**)**. The Margalef index fluctuated between 0.43 in station 3 and 0.87 in station 7 with an annual average of 0.78. The Pielou's evenness was maximum in station 1 (0.99) and minimum in station 3 (0.42). The Simpson dominance index ranged from 0.25 to 0.69 with a minimum value in station 5 and maximum value in station 3 with an annual average of 0.37. In contrast with first year, during second year the Shannon diversity index was maximum in station 6 (2.06) and minimum in station 1 (0.07) with annual spatial average of 1.34 (Fig. 7.12). The richness index showed that maximum species richness value of 0.86 was observed in station 7 and minimum value in station 1 (0.15). Its annual spatial average value was comparatively lower in second year (0.53). In contrast with first year, the Pielou's evenness was minimum in station 1 (0.07) and maximum in station 9 (0.79), with an annual average value of 0.57. While observing the diversity pattern of annual temporal abundance of peracarids, during first year the maximum diversity value was observed during July 2011 (2.69) and minimum in June 2011 (1.41) (Fig. 7.13**)**, whereas during second year, the maximum diversity was observed in the month of February 2013 (2.21) and minimum in January 2013 (0.96) (Fig. 7.14**)**. During first year, the highest richness value was observed during the month of July 2011(1.31) and lowest in March 2011 (0.8). During second year the maximum richness value was noted during March 2012 (1.13) and minimum during November 2012 (0.69). During first year, the highest Pielou's evenness index was found during July 2011 (0.89) and lowest during February 2012

(0.51). The temporal variation of dominance index was just opposite to that of evenness index, in which the maximum value of 0.49 was observed during February 2012 and minimum of 0.17 observed during July 2011. Similar to temporal variation of diversity index during second year, the highest evenness index was observed during February 2013 (0.74) and lowest in January 2013 (0.32). Similar to first year the temporal variation of dominance index was opposite to evenness index, in which the maximum value was observed during January 2013 (0.72) and minimum during February 2013 (0.26).

During the first year taxonomic diversity index (Δ) of peracarids varied from 23.11 (station 3) to 54.28 (station 5) and total phylogenetic diversity index (sPhi+) varied from 225 (station 2) to 500 (station 7). Average Taxonomic Distinctness (Δ+) and variation in Taxonomic Distinctness (Λ+) ranged between 66.67 (station 2) to 75 (station 1) and 0 (station 1) to 205.6 (station 10) respectively (Table 7.3). The 95% confidence funnel generated for the Δ+ and Λ+ values of all the stations is shown in Figure 7.15 and Figure 7.16. As indicated by the funnel plot, values of Δ+ for peracarids in all stations except stations 1 and 7 in Vembanad estuary fall on or within the expected mean for Δ+, within its 95% probability limits. Values of Δ+ for station 1 and 7 lie in the borderline of funnel denoting significantly different diversity of peracarids. The funnel plot for peracarids showed a reverse pattern of the Δ+. During second year taxonomic diversity index (Δ) of peracarids varied from 1.201 (station 1) to 51.57 (station 8) and total phylogenetic diversity index (sPhi+) varied from 175 (station 1) to 450 (station 7). Average Taxonomic Distinctness (Δ+) and Variation in Taxonomic Distinctness (Λ+) ranged between 65 (station 9) to 75 (station 10) and 0 (station 1 and 10) to 275 (station 9) respectively (Table 7.3). The variation in taxonomic distinctness value and average taxonomic distinctness value of peracarids was superimposed on the funnel to find out the deviation from the normal distribution (within the 95% confidence limit). The funnel plot for Δ+ value indicated that station 9 lying lower the 95% probability limit of the funnel, station 1 and 10 in the borderline and stations 2 to 8 except station 7 lying above the 95% probability limit of the funnel (Fig. 7.17). The funnel plot of Λ+ for all stations were not come together within the 95% confidence level represented that peracarid diversity was varied significantly among the stations (Fig. 7.18).

7.2.5.2 K- Dominance Plot

The k-dominance plot is plotted according to both spatial average and seasonal percentage dominance of peracarid species. In the plot, the shallow curves denote communities with high dominance, whereas steep curves denote a more balanced and diverse community. Based on the spatial diversity, station 3 and station 9 representing highly disturbed situation with the dominance curve that lie at the top representing very low diversity and high dominance during the first year (Fig. 7.19). Whereas during second year, the dominance curve in station 1 to 3 was representing a highly distressed condition (Fig. 7.20). Seasonally, during first year the curve represented during the PM season lies at the top denoting a reduced diversity and the curve of PRM season at the bottom was indicating a higher diversity (Fig. 7.21). The shape of the graph was denoting that there was some disturbances occurred in the communities. In contrast with first year, during second year the curve of PRM and PM season was denoting lower and higher diversity respectively (Fig. 7.22).

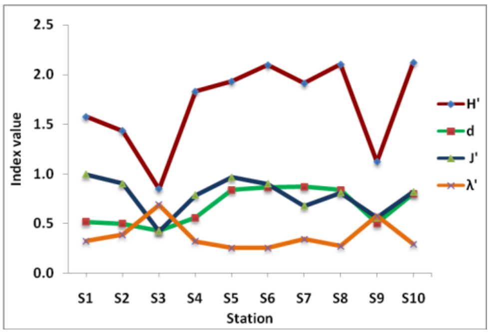


Fig. 7.11 Spatial variation of peracarid crustacean diversity indices in Vembanad estuary during 2011 – 2012 period

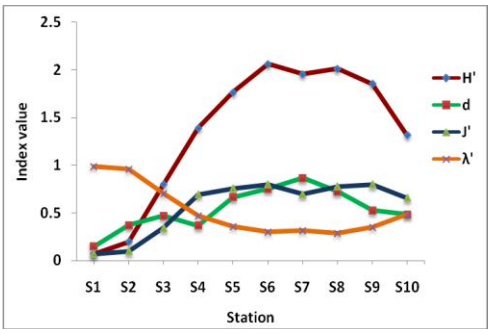


Fig. 7.12 Spatial variation of peracarid crustacean diversity indices in Vembanad estuary during 2012 – 2013 period

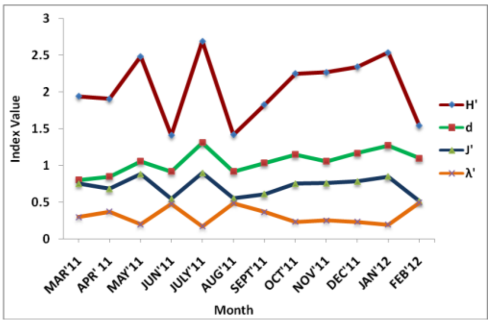


Fig. 7.13 Temporal variation of peracarid crustacean diversity indices in Vembanad estuary during 2011 – 2012 period

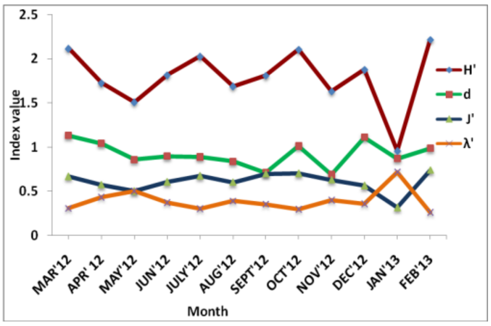


Fig. 7.14 Temporal variation of peracarid crustacean diversity indices in Vembanad estuary during 2012 – 2013 period

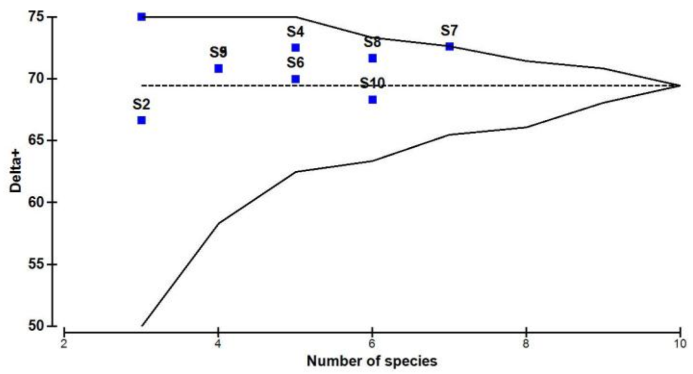


Fig. 7.15 The 95% confidence funnel for the average taxonomic distinctness (Delta+) values of peracarid diversity in Vembanad estuary during 2011 – 2012 period

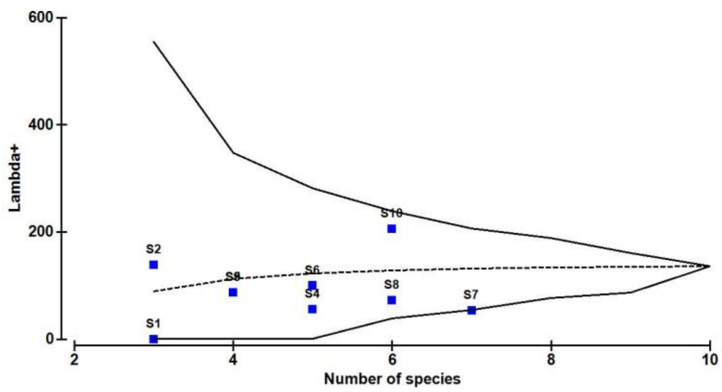


Fig. 7.16 The 95% confidence funnel for variation in taxonomic distinctness values (lambda+) of peracarid diversity in Vembanad estuary during 2011–2012 period

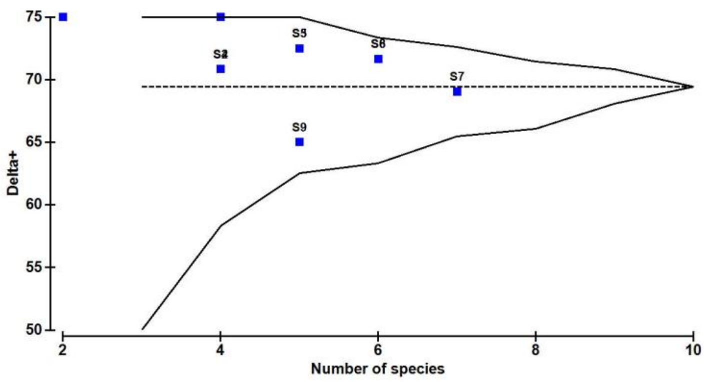


Fig. 7.17 The 95% confidence funnel for the average taxonomic distinctness (Delta+) values of peracarid diversity in Vembanad estuary during 2012 – 2013 period

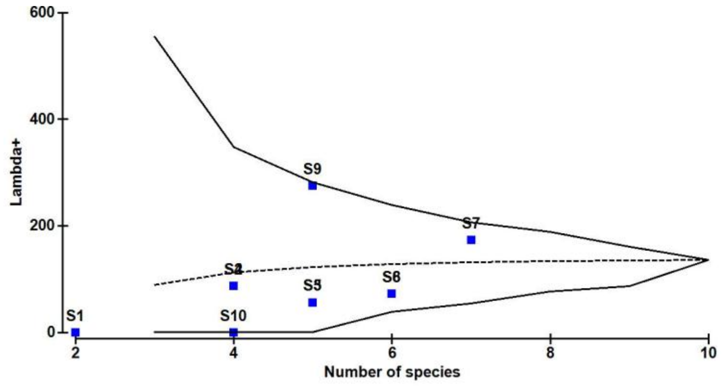


Fig. 7.18 The 95% confidence funnel for variation in taxonomic distinctness values (lambda+) of peracarid diversity in Vembanad estuary during 2012 – 2013 period

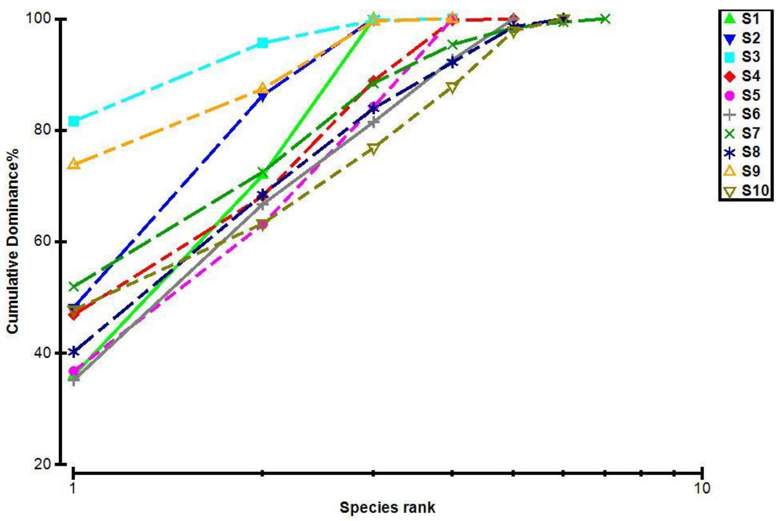


Fig.7.19 K-dominance plot showing the spatial dominance of peracaridan species in Vembanad estuary during 2011 – 2012 period.

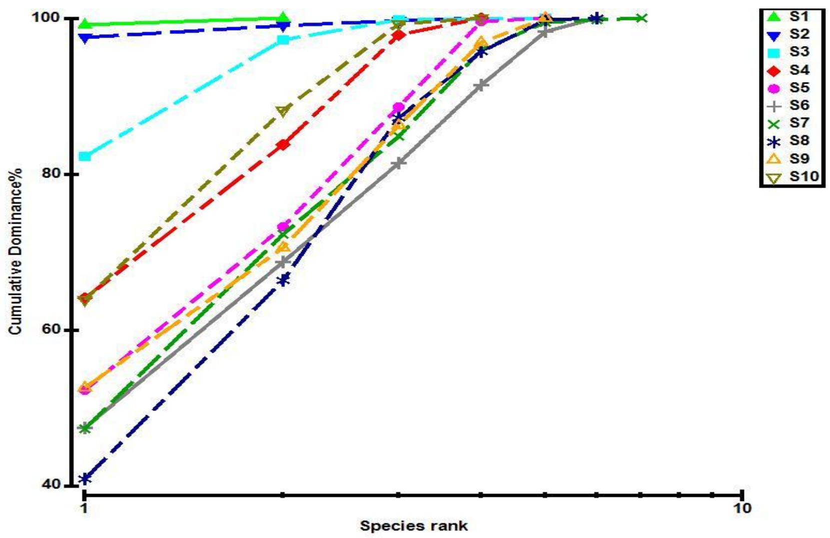


Fig. 7.20 K-dominance plot showing the spatial dominance of peracaridan species in Vembanad estuary during 2012 – 2013 period

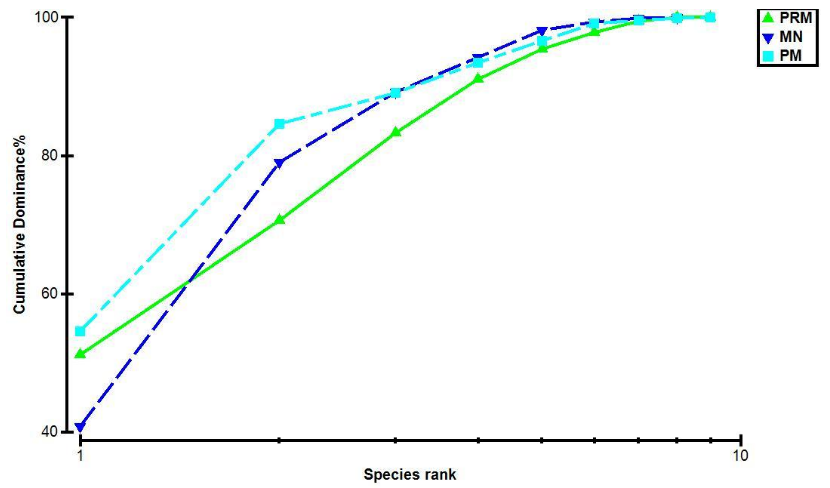


Fig. 7.21 K-dominance plot showing the seasonal percentage dominance of peracaridan species in Vembanad estuary during 2011 – 2012 period

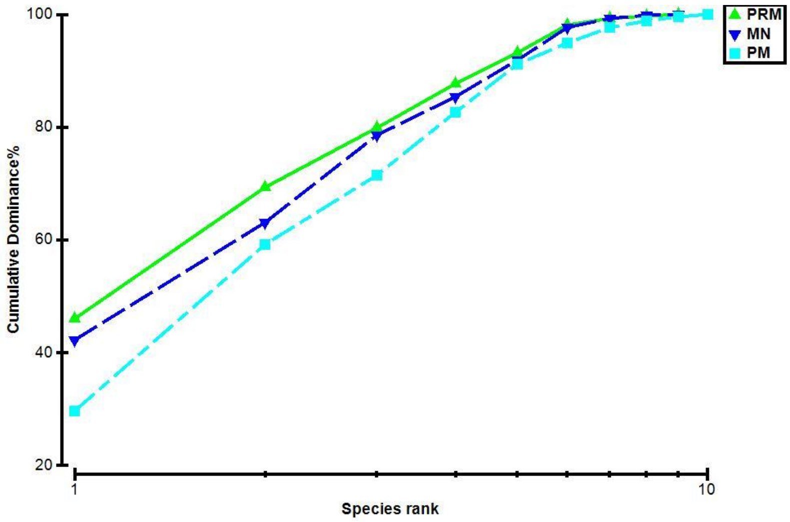


Fig. 7.22 K-dominance plot showing the seasonal percentage dominance of peracaridan species in Vembanad estuary during 2012 – 2013 period

7.2.5.3 MDS analysis

During 2011-2012 period, MDS plots showed a clear separation of the samples of station on the southern and northern zone of TMB and the low stress value of 0.04 representing good ordination of MDS plot (Fig. 7.23). Both stations on the southern and northern zone showed an overall similarity of 40%. It was clear that the southern zone was highly dominated with amphipod *C. geniculata* whereas in the northern zone it was completely absent. In contrast with *C. geniculata*, *E. chilkensis* were not found in the southern zone. During 2012-2013 period MDS plot indicated a good ordination with a stress value of 0.03 denoting that there was a strong interrelationship between the peracarids of each station (Fig. 7.24) and a significant variation was observed in the peracarids distribution between stations. MDS plots showed a clear differentiation of peracarids of station 1 and 10 from other stations. Among the peracarid species, only *C. geniculata* and *C. chilkensis* were present in station 1. Whereas *C. chilkensis*, *Cirolana fluviatilis,* Mysida and *Gammarus* sp. were the peracarid species present in station 10.

7.2.5.4 Comparison of distribution of peracarids in the southern and northern zone of Vembanad estuary

The peracarid species distribution on either side of the southern and northern zone of TMB was given in Table 7.4. One-way ANOSIM (analysis of similarities) was carried out to observe the variation of peracarid species between the southern and northern zone and were identified using the SIMPER (similarity percentages) analysis. During first year, the ANOSIM analysis showed that the differences in species composition between the southern and northern zone was significant (ANOSIM: Global R = 0.794, $P < 0.005$). The SIMPER analysis showed that the species that most contributed to the differences among two zone was *E. chilkensis* (17.71 %), *C. geniculata* (16.18 %) and *Gammarus* sp. (16.1 %) (Table 7.5). Similar to first year during second year, the ANOSIM analysis revealed that the differences in species composition between the southern and northern zone was significant (ANOSIM: Global R = 0.627, $P < 0.005$). The SIMPER results showed that the distinction between southern and northern zone was mainly in the dissimilarity of percentage contribution of *C. geniculata* (27.41 %) and *Gammarus* sp. (18.84 %) (Table 7.6).

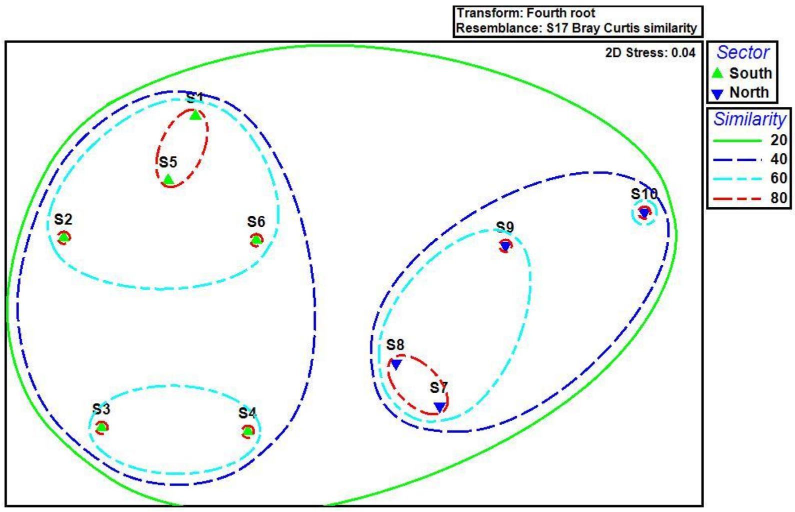


Fig. 7.23 MDS plot showing the spatial similarity of peracarid species in Vembanad estuary during 2011-2012 period

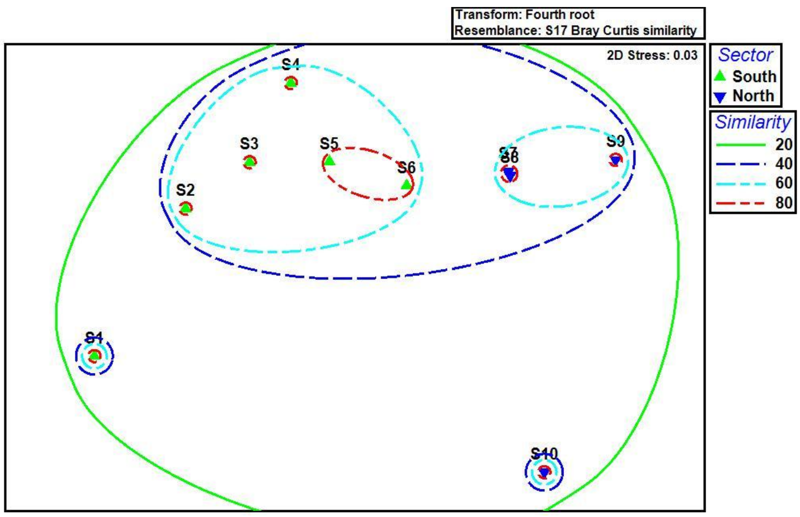


Fig. 7.24 MDS plot showing the spatial similarity of peracarid species in Vembanad estuary during 2012-2013 period

Table 7.3 Diversity indices and taxonomic distinctness values of peracarid species in Vembanad estuary during 2011 – 2013 period

Station	Delta (Δ)		Δ* (Delta*)		Δ^+ (Delta+)		Λ^+ (Lambda+)		sPhi+	
	2011-2012	2012-2013	2011-2012	2012-2013	2011-2012	2012-2013	2011-2012	2012-2013	2011-2012	2012-2013
S1	50.75	1.201	75	75	75	75	0	0	250	175
S2	43.48	3.553	70.67	74.96	66.67	70.83	138.9	86.81	225	300
S3	23.11	22.36	74.11	74.36	70.83	72.5	86.81	56.25	300	375
S4	46.16	33.47	67.85	63.13	72.5	70.83	56.25	86.81	375	300
S5	54.28	45.8	72.71	70.59	70.83	72.5	86.81	56.25	300	375
S6	52.69	50.2	70.77	71.76	70	71.67	100	72.22	350	425
S7	47.76	47.93	72.7	69.96	72.62	69.05	53.85	172.9	500	450
S8	52.23	51.57	72.02	71.98	71.67	71.67	72.22	72.22	425	425
S9	30.93	46.93	73.05	71.68	70.83	65	86.81	275	300	325
S10	43.28	39.14	61.12	75	68.33	75	205.6	0	400	325

Table 7.4 Peracarid species distribution in the south and north of TMB in Vembanad estuary during 2011-2013 period

Peracarid species	South of TMB	North of TMB
Eriopisa chilkensis (Chilton, 1921)	-	+
Cheiriphotis geniculata (K.H. Barnard, 1916)	+	-
Gammarus sp. (Fabricius, 1775)	+	+
Eriopisa sp. (Stebbing, 1890)	-	+
Bodotria sp.(Goodsir, 1843)	+	+
Cirolana fluviatilis (Stebbing, 1902)	-	+
Apanthura sandalensis (Stebbing, 1900)	+	+
Xenanthura orientalis (Barnard, 1935)	+	+
Mysida (Haworth, 1825)	+	+
Ctenapseudes chilkensis (Chilton, 1924)	+	+

(Note: '+' denote present; '-' denote absent)

Table 7.5 SIMPER test results showing the dissimilarity of peracarid species between the southern and northern sector of TMB during 2011-2012 period

2011-2012	**Average dissimilarity = 65.90 %**				
	Group South	**Group North**			
Species	**Average Abundance**	**Average Abundance**	**Average Dissimilarity**	**Percentage Contribution (%)**	**Cumulative percentage (%)**
Eriopisa chilkensis	0	2.93	11.67	17.71	17.71
Cheiriphotis geniculata	2.78	0	10.66	16.18	33.89
Gammarus sp.	0.4	3.06	10.61	16.1	49.99
Ctenapseudes chilkensis	1.64	3.45	8.51	12.91	62.89
Xenanthura orientalis	2.67	2.37	7.3	11.08	73.97
Apanthura sandalensis	2.06	1.18	5.91	8.97	82.94
Mysida	0	0.9	3.19	4.84	87.78
Bodotria sp.	0.46	0.77	2.93	4.45	92.24

Table 7.6 SIMPER test results showing the dissimilarity of peracarid species between the southern and northern sector of TMB during 2012-2013 period

2012-2013	**Average dissimilarity = 60.32 %**				
	Group South	**Group North**			
Species	**Average Abundance**	**Average Abundance**	**Average Dissimilarity**	**Percentage Contribution (%)**	**Cumulative percentage (%)**
Cheiriphotis geniculata	5.36	0	16.54	27.41	27.41
Gammarus sp.	0.49	4.11	11.36	18.84	46.25
Xenanthura orientalis	3.86	3.74	7.84	12.99	59.24
Eriopisa chilkensis	0	2.31	6.56	10.87	70.11
Apanthura sandalensis	2.61	1.6	5.97	9.89	80
Ctenapseudes chilkensis	2.44	3.47	3.56	5.9	85.9
Cirolana fluviatilis	0	0.84	3.17	5.25	91

7.2.5.5 Succession of peracarids in the southern zone of Vembanad estuary

In 2011-2012 period, the average abundance of peracarid species was highest during the opening period of TMB than the closure period, whereas contrasting results were obtained during 2012-2013 period. During the open period of barrage *C. geniculata* (av. 393 ind./m^2) formed the copious species followed by *X. orientalis (*av.201 ind./m^2*)*. During the closure period of barrage, the average numerical abundance of *C. geniculata* decreased to 32 ind./m^2 and that of *X. orientalis* also decreased to 92 ind. /m^2 (Table 7.7). But slight increase in the abundance of *C. chilkensis* was observed during the open period of barrage (70 ind./m^2) than the closure period (54 ind./m^2). During 2012-2013 period, the abundance of species such as *C. geniculata*, *X. orientalis*, *A. sandalensis* and *C. chilkensis* were increased from closure period than the open period of TMB.

7.2.5.6 Relation between peracarid distribution and environmental factors

7.2.5.6.1 Correlation analysis

The Pearson's correlation co-efficient was used to study the relationship between polychaete species distribution and environmental variables. The results of correlation analysis for first year and second year are given in Table 7.8 and Table 7.9 respectively.

Table 7.7 Comparison of average peracarid species abundance (ind./m^2) during the open and close period of TMB in the southern zone of Vembanad estuary

	2011-2012		2012-2013	
	open	**close**	**open**	**close**
Ctenapseudes chilkensis	70	54	94	17
Apanthura sandalensis	40	92	175	110
Xenanthura orientalis	92	201	721	362
Mysida	0	0	5	2
Bodotria sp.	2	0	0	0
Cheiriphotis geniculata	32	393	1647	1553
Gammarus sp.	5	5	23	1

Table 7.8 Table showing the correlation between physico-chemical parameters and peracarids in Vembanad estuary during 2011-2012 period

	Salinity	**Dissolved Oxygen**	**Sediment Tempera ture**	**Sediment pH**	**Organic Carbon**	**Clay**	**Silt**	**Sand**
Ctenapseudes chilkensis	.185*	0.00	0.14	0.13	-0.17	-0.15	-.329**	.327**
Cirolana fluviatilis	.459**	-.245**	0.08	0.17	0.05	.669**	0.016	-.248**
Apanthura sandalensis	-0.128	0.07	-.181*	-0.10	-0.07	-0.13	0.084	-0.025
Xenanthura orientalis	0.057	0.08	-0.12	-0.01	-0.13	-0.17	-0.023	0.077
Mysida	-0.062	0.05	0.03	0.07	-0.16	-0.05	-0.092	0.095
Eriopisa chilkensis	.325**	0.04	0.05	0.18	-.200*	-0.14	-.350**	.342**
Cheiriphotis geniculata	-.204*	0.01	-0.03	-0.05	0.16	0.04	.279**	-.247**
Gammarus sp.	.269**	-0.01	0.08	.358**	-0.11	0.17	-.321**	.211*
Eriopisa sp.	0.145	0.06	-0.08	.230*	-0.16	-0.09	-.280**	.264**

**. Correlation is significant at the 0.01 level ;*. Correlation is significant at the 0.05 level

Table 7.9 Table showing the correlation between physico-chemical parameters and peracarids in Vembanad estuary during 2012-2013 period

	Salinity	**Dissolved Oxygen**	**Sediment Temperature**	**Sediment pH**	**Organic Carbon**	**Clay**	**Silt**	**Sand**
Ctenapseudes chilkensis	0.06	-0.02	-.209*	0.02	-.231*	-0.09	-0.05	0.08
Cirolana fluviatilis	.528**	-0.11	0.04	0.11	0.00	0.02	0.17	-0.15
Apanthura sandalensis	-0.09	-0.15	-0.04	-0.07	-0.11	0.13	-0.02	-0.04
Xenanthura orientalis	-0.12	0.01	-0.13	-0.04	-0.14	-0.01	-0.06	0.05
Mysida	0.18	-.210*	-0.08	0.08	-0.02	-0.01	0.02	-0.01
Bodotria sp.	-0.10	-0.01	-0.05	-0.01	-0.04	-0.03	-0.08	0.08
Eriopisa chilkensis	.529**	0.01	0.00	0.14	-0.07	.297**	0.07	-0.18
Cheiriphotis geniculata	-0.05	0.08	0.06	-0.03	-0.02	0.01	0.04	-0.03
Gammarus sp.	-0.01	.229*	-0.05	0.03	-.180*	0.06	-0.11	0.07
Eriopisa sp.	.218*	-0.03	0.04	0.12	-0.05	.262**	0.16	-.241**

**. Correlation is significant at the 0.01 level; *. Correlation is significant at the 0.05 level

7.2.5.6.2 BIOENV analysis of peracarid community

The BIOENV analysis was used to determine the major environmental variables that best explaining the distribution of peracarids in the estuarine system. During first year, the result of BIOENV analysis demonstrated a strong correlation between water quality parameters and peracarids ($\rho = 0.717$; $P < 0.005$). Among the water quality parameters, that controlling the distribution of pericarids was including water temperature, alkalinity, salinity, depth etc. (Table7.10 and Fig. 7.25). Overall, the BIOENV analyses showed a strong significant correlation between sediment parameters and peracarid crustaceans in Vembanad estuarine system during first year ($\rho = 0.655$; $P < 0.005$) (Table7.11 and Fig. 7.26). The single sediment variable that best describes the peracarid community distribution is clay. A combination of other sediment parameters such as sand, sediment temperature, sediment pH, moisture content and available phosphorus also matched with the assemblage pattern of peracarids in the estuary. Results from the BIOENV analysis of water quality and sediment parameters between peracarids during second year are summarized in Table 7.12 and 7.13; Fig.7.27 and 7.28. BIOENV analysis revealed that variations in peracarids were best matched by a combination of five water quality variables such as pH, salinity, alkalinity, temperature and dissolved inorganic nitrogen ($\rho = 0.859$; $P < 0.001$). During second year, BIOENV analysis revealed that the best significant correlation was exhibited between peracarids and sediment parameters such as sediment pH, silt, clay and available phosphorus ($\rho = 0.816$; $P < 0.001$).

7.2.5.6.3 CCA analysis of peracarid community

Canonical Correlation Analysis (CCA) gives additional insight into the relationship between spatial variation in the community assemblages of peracarids and environmental variables in Vembanad estuary. The CCA analysis demonstrated that the selected water quality parameters had strong control over the spatial distribution of peracarids (Fig. 7.29). The first three CCA axes could explain a total variability of 91.39 % of the relationship between peracarids and water quality parameters (eigen value: axis 1 = 0.69, for axis 2 = 0.45 and for axis 3 = 0.15). TDS (r = 0.89), alkalinity (r = 0.88), conductivity (r = 0.85), salinity (r = 0.83) and bottom water temperature (r = 0.80) showed a strong negative correlation with axis 1, indicating the influence of sea water on the peracarid assemblages. During first year, the first three axes explained about 92.6 % of the total variance between peracarids and sediment parameters. The first axis was negatively correlated with sediment temperature (r = 0.81) and potassium (r = 0.58). The second axis mostly positively correlated with calcium (r = 0.91), potassium (r = 0.87), sodium (r = 0.82), available phosphorous (r = 0.66) and also the sediment particles clay (r = 0.79) and silt (r = 0.59). CCA plot showed that the isopod species *X. orientalis* and *A. sandalensis* assembled mainly in station 4; the amphipod *C. geniculata* preferred mainly available nitrogen and organic carbon and the species mainly confined in station 3 (Fig. 7.30). The tanaid, *C. chilkensis* were preferred by sandy substratum and it mainly located in station 9 and 10. Similar to first year, the CCA analysis conducted on peracarid species and environmental parameters including water (Fig. 7.31) and sediment variables (Fig. 7.32) during second year. The arrows denoting the environmental variables point out the direction of maximum variation of that parameter and the length of the arrow corresponds to the rate of variation. Two axes on canonical ordination plots together explained 89.49 % of variation in species abundance data. Axis 1 of CCA analysis exhibited 53.17 % of the total variance and was strongly positively influenced by sediment pH (r = 0.71) and sand (r = 0.58). At the same time available nitrogen (r = -0.76), silt (r = -0.73) and organic carbon (r = -0.70) showed a negative correlation with axis 1. The CCA analysis showed that *C. fluviatilis* was favored by clay fraction of sediment. The CCA plot also signifies that species *X. orientalis* have high preference towards sandy substratum and

C. geniculata have inclination towards silt and organic carbon content. During second year a total of 11 bottom water parameters were used for CCA analysis. Eigen value of axis 1 (0.69), 2 (0.39) and 3 (0.15) explained a total of 97.69 % variation. Salinity (r = -0.85) and bottom water temperature (r = -0.67) showed negative correlation towards axis 1.

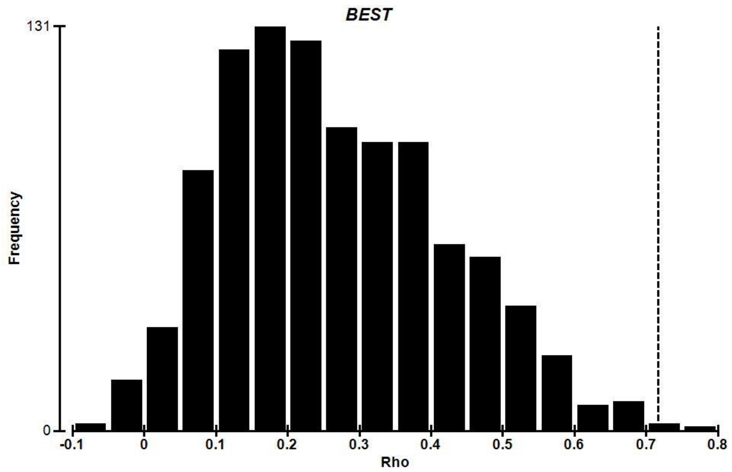


Fig. 7.25 Histogram showing the results of BEST analysis of peracarid species with water quality parameters in Vembanad estuary during 2011-2012 period

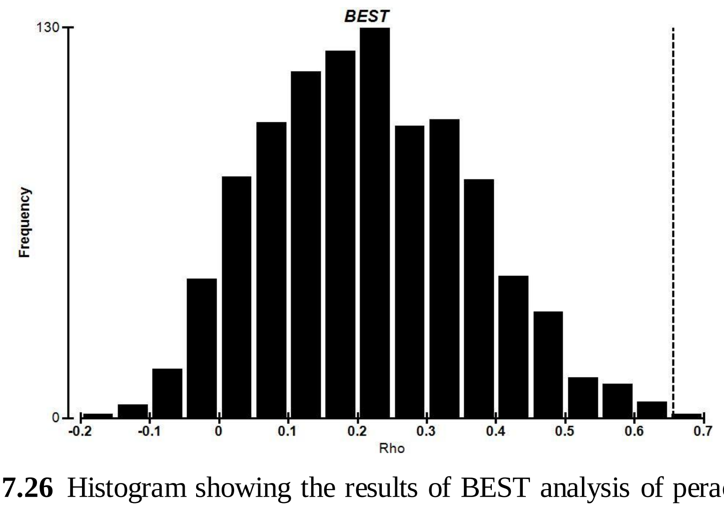


Fig. 7.26 Histogram showing the results of BEST analysis of peracarid species with sediment parameters in Vembanad estuary during 2011-2012 period.

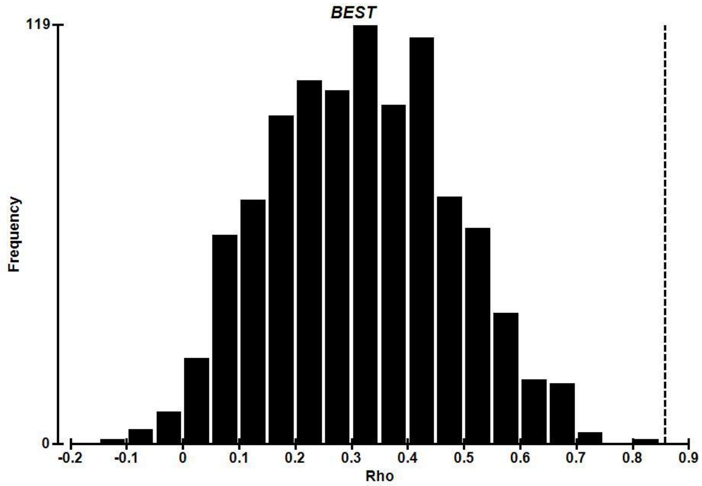


Fig. 7.27 Histogram showing the results of BEST analysis of peracarid species with water quality parameters in Vembanad estuary during 2012-2013 period.

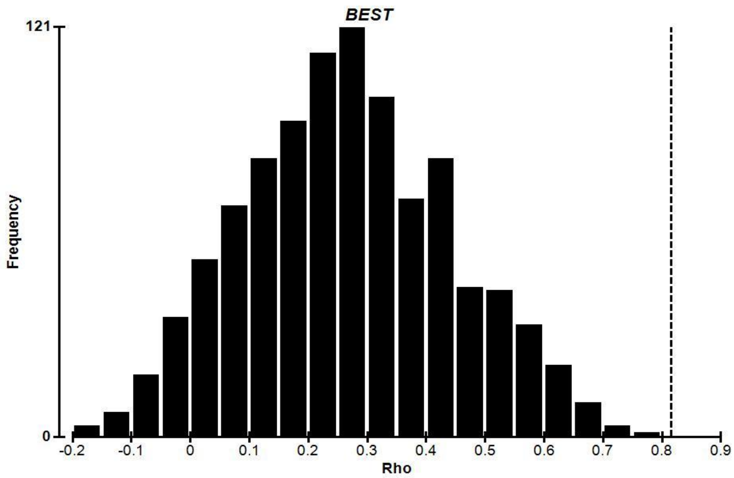


Fig. 7.28 Histogram showing the results of BEST analysis of peracarid species with sediment parameters in Vembanad estuary during 2012-2013 period.

Table 7.10 Results of BIOENV analysis for peracarid species and water quality parameters in Vembanad estuary during 2011 – 2012 period

Sl. No.	Variables	Variables Selected	Correlation Values (Rho)
1	Water pH	2, 7	0.717
2	Water temperature	2, 3, 7	0.706
3	Salinity	2, 3	0.691
4	Dissolved oxygen	2, 3, 7, 9	0.674
5	Silicate-Silicon	2, 3, 6	0.673
6	Phosphate-phosphorus	2, 3, 6, 7	0.671
7	Alkalinity	2-4, 7	0.666
8	Chlorophyll *a*	2, 6, 7	0.654
9	Depth	2, 3, 6, 7, 9	0.653
10	Dissolved inorganic nitrogen	2, 3, 9	0.653

Table 7.11 Results of BIOENV analysis for peracarid species and sediment parameters in Vembanad estuary during 2011 – 2012 period

Sl. No.	Variables	Variables Selected	Correlation Values (Rho)
1	Sediment temperature	8	0.655
2	Sediment pH	6, 8	0.616
3	Moisture content	1-3, 8	0.596
4	Organic carbon	1-3, 6, 8	0.588
5	Available nitrogen	1, 2, 8, 9	0.570
6	Sand	1, 2, 6, 8	0.568
7	Silt	1, 6, 8	0.567
8	Clay	2, 6, 8	0.561
9	Available phosphorus	2, 3, 6, 8	0.560

Table 7.12 Results of BIOENV analysis for peracarid species and water quality parameters in Vembanad estuary during 2012 – 2013 period

Sl. No.	Variables	Variables Selected	Correlation Values (Rho)
1	Water pH	1, 2, 5, 9, 10	0.859
2	Salinity	1, 2, 9, 10	0.854
3	Dissolved oxygen	2, 5, 9, 10	0.851
4	Phosphate-phosphorus	2, 9, 10	0.844
5	Alkalinity	1, 2, 5	0.836
6	Chlorophyll *a*	1, 2, 5, 9	0.833
7	Depth	2, 5, 8-10	0.832
8	Silicate-silicon	1, 2, 8-10	0.831
9	Water temperature	1, 2, 5, 8, 9	0.828
10	Dissolved inorganic nitrogen	1, 2, 5, 10	0.823

Table7.13 Results of BIOENV analysis for peracarid species and sediment parameters in Vembanad estuary during 2012 – 2013 period

Sl. No.	Variables	Variables Selected	Correlation Values (Rho)
1	Sediment temperature	2, 7 – 9	0.816
2	Sediment pH	2, 6 – 9	0.801
3	Moisture content	2, 5, 7-9	0.795
4	Organic carbon	7-9	0.786
5	Available nitrogen	2, 5, 7, 8	0.779
6	Sand	5, 7, 8	0.775
7	Silt	5, 7 – 9	0.775
8	Clay	2, 4, 7 – 9	0.773
9	Available phosphorus	2, 3, 7 - 9	0.770

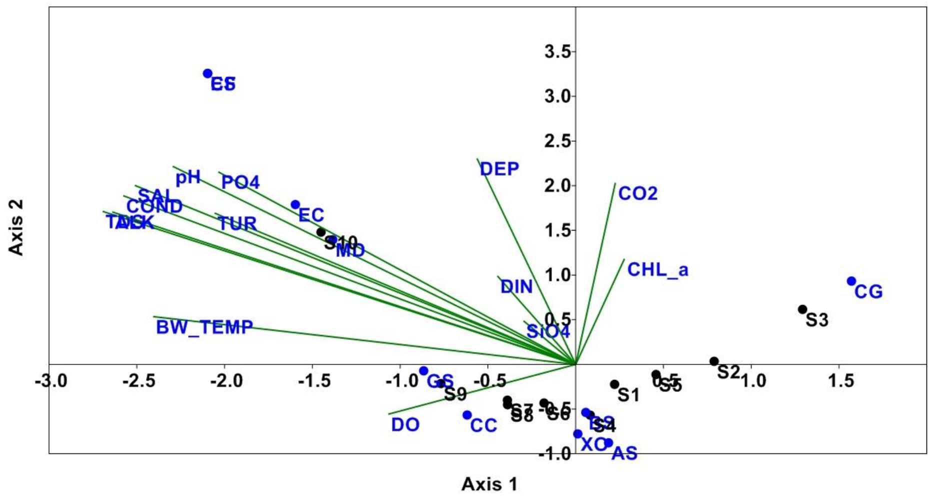


Fig. 7.29 CCA analysis of peracarid species and water quality parameters in Vembanad estuary during 2011 – 2012 period

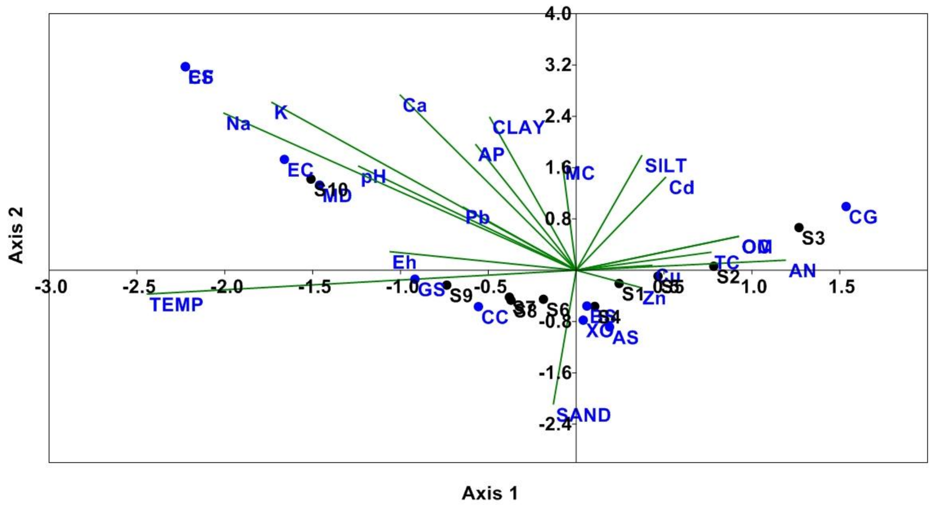


Fig. 7.30 CCA analysis of peracarid species and sediment parameters in Vembanad estuary during 2011 – 2012 period

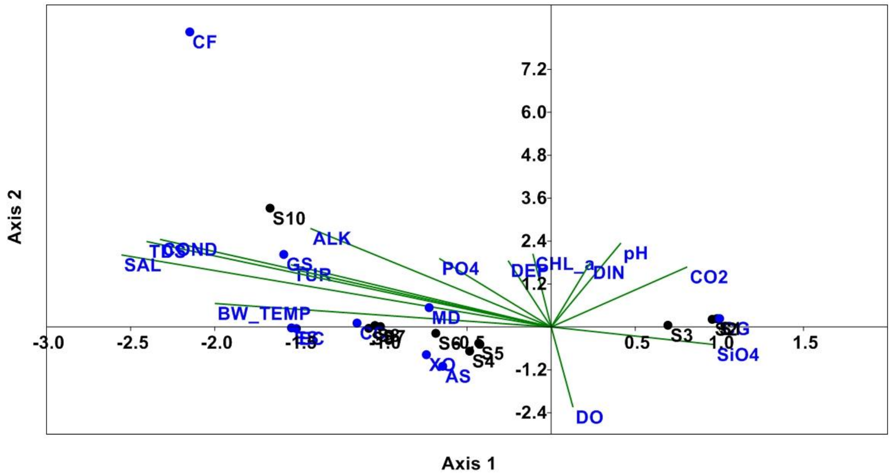


Fig. 7.31 CCA analysis of peracarid species and water quality parameters in Vembanad estuary during 2012 – 2013 period

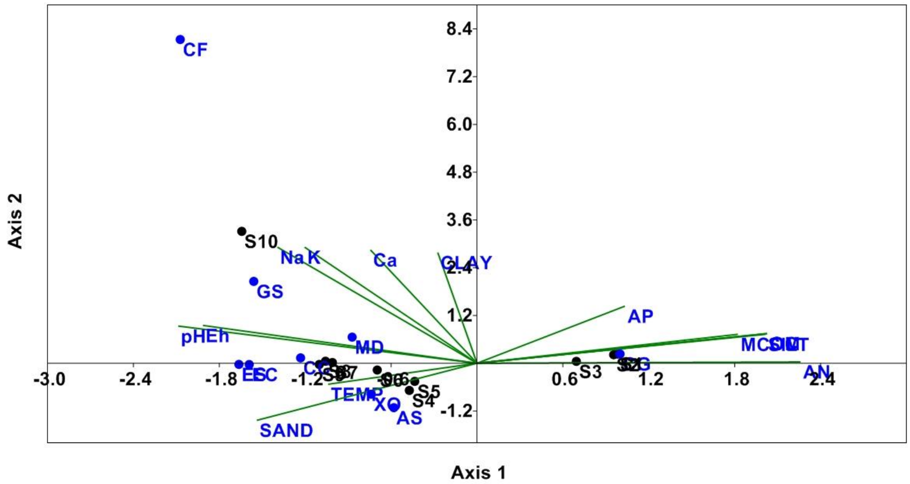


Fig. 7.32 CCA analysis of peracarid species and sediment parameters in Vembanad estuary during 2012 – 2013 period

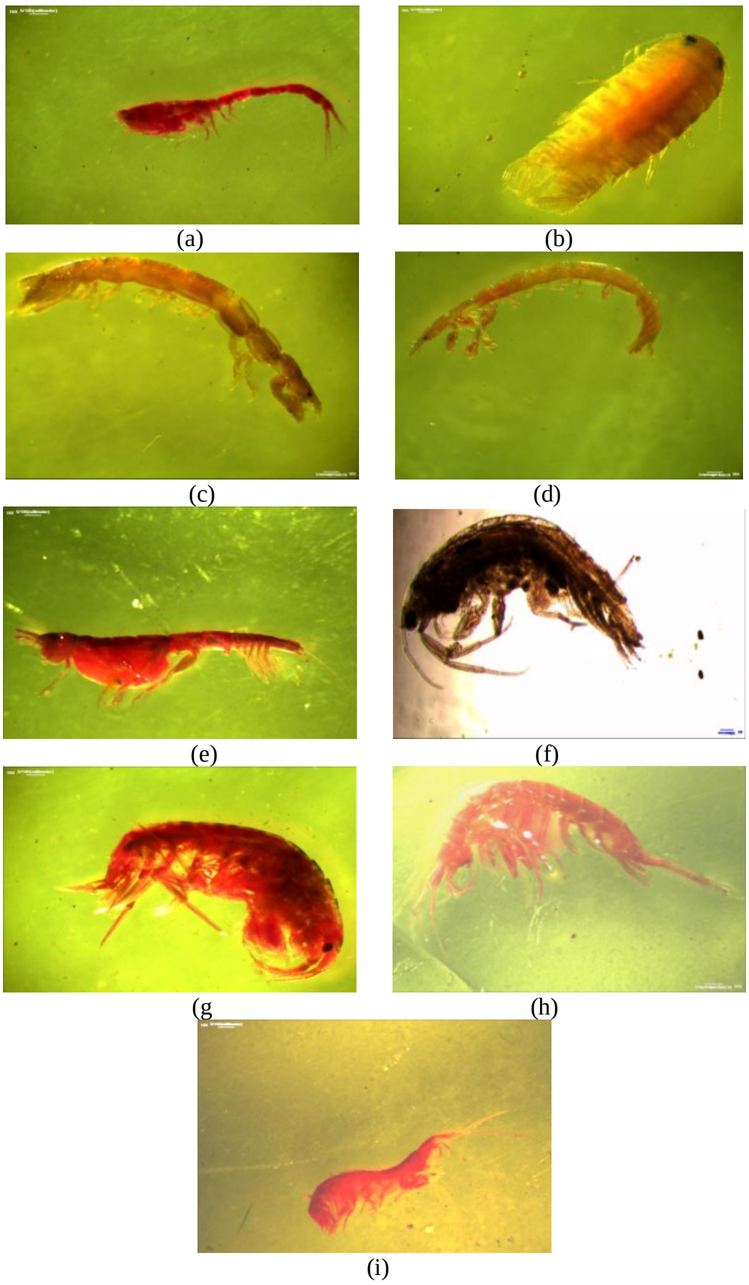


Plate 7.1 (a-i) Major peracarid species observed in Vembanad estuary during 2011-2013 period (Magnification– X4). [a) *Bodotria* sp. ; b) *Cirolana fluviatilis*; c) *Apanthura sandalensis*; d) *Xenanthura orientalis*; e) *Ctenapseudes chilkensis*; f) *Cheiriphotis geniculata*; g) *Gammarus* sp.; h) *Eriopisa chilkensis*; i) *Eriopisa* sp.]

7.3 Discussion

The details of spatial pattern of organism distribution were an important aspect in ecological studies, as it gives indirect details on biological relations (Vargas-Zamora et al., 2012). The density of peracarid crustacean assemblages in the Vembanad estuarine system exhibited distinct spatial and temporal variations. It is well known that peracarid crustaceans showed dial migration due to its definite behavioural characteristics such as endogenous rhythms, escape from fish predation and also due to the influence of light, currents and accessibility food etc. (Kaartvedt, 1985). During the night, the swimming movement of epibenthic and suprabenthic peracaridan increases and in the case of mysids which migrated towards surface and endobenthic groups such as amphipods and cumaceans come out from sediment (Cunha et al., 1999). From the crustacean groups, peracarid crustaceans formed the abundant group in Vembanad estuarine system. Among the peracarid crustaceans, amphipods formed the most dominant group with the species *Cheiriphotis* becoming the chief contributor. Among the peracaridans, amphipods were dominant in numbers of individuals and species in shallow subtidal soft bottoms of Galician Rias estuarine system, Spain (Moreira et al., 2008). This was attributed by the euryphagic nature of most of the amphipod species, which were able to utilize various microhabitats (Chintiroglou et al., 2004). Amphipods are among the most common micro-crustaceans and its abundance related to their ability for extensive parental care (Thiel, 1998). The pelagic larval stage was absent in amphipods and young ones developed in the marsupium and the parent directly released the young into the environment and so its distribution is used for the zoogeographical studies (Watling, 1979).

Among the marine fouling communities amphipods formed the dominant groups on artificial substrates such as parts of ship wrecks, navigational materials, fish cages, foundations of harbours and offshore constructions

(Zintzen et al., 2006; Greene and Grizzle, 2007). According to Rehm et al. (2007) variations in the results of community structure of peracarids were related to the differences in the gear. In the South Shetland Islands peracarids taken with an epibenthic sledge at depth between 200 to 400 m showed mean proportions of fauna as Amphipoda (62%), Isopoda (16%), Tanaidacea (13%), Cumacea (11%) and Mysidacea (< 1 %) (Lorz and Brandt, 2003). However, samples from southern Weddell Sea and off King George Island in depths of 200 to 400 m peracarids showed different proportions, as in the Weddell Sea, Isopoda (60%), Amphipoda (26%), Mysidacea (10%), Cumacea (4%) and Tanaidacea (1%) were abundant and in the off King George Island Cumacea (39%), Amphipoda (31%), Mysidacea (17%), Isopoda (13%), and Tanaidacea (1%) were abundant (Linse et al., 2002). Comparatively lowest Shanon diversity value was observed in Vembanad estuary. The species *Cheiriphotis geniculata* was identified from Vatta Kayal (round lake) region of Alleppey by H. S. Rao and M. Sharif during January 1928 (Barnard, 1935). Further it was reported in the Sagar Island, Sundarbans, West Bengal (Bhunia and Choudury, 1991). The dominant amphipod species *C. geniculata* was widely distributed in the southern most stations. This was reflected with the increased dominance and low diversity index in this region. Similarly the dominance curve of the region showed a disturbed condition with the curve is not ‘S’ shaped. This denoted that the habitat is not free from any type of impacts. At the same time, the curves showing steep rise representing a less diverse community with higher dominance. The southernmost stations were characterized with higher organic carbon in the sediment. Similarly thick deposits of decaying water hyacinth on the surface of the sediment were also common among these stations. This type of substratum may provide a shelter for this species and also acts as a major source of food. It was reported that Seagrass meadows support the amphipods by providing food and shelter and also overall forms a diverse area for peracarid assemblages (Nagle, 1968; Nienhuis and Van Ierland, 1978). Among amphipods species, *Corophium triaenonyx* was the most widespread and dominant species in Cochin estuary (Nair et al., 1983). Similar observation was reported by Geetha and Bijoy Nandan (2014) in the Cochin estuary. But the species *C. triaenonyx* was not recorded from the present investigation. It was also reported that salinity below 4 ppt was lethal to *C. triaenonyx* (Shyamasundari, 1972) but it has capacity to withstand a wide range of salinity varying from 0.1 – 27.7 ppt (Nair et al., 1983). Wang et al. (2010) noted that the composition, abundance and diversity of peracaridan varied in different vegetation. Mangrove harboring seaweeds favored the diversity of peracaridan by giving protection and emergent plants also support its diversity. Amphipod crustaceans are significant for various pollution studies because it act as natural indicators of contamination. It can also withstand a wide range of salinity and sediment types (Marsden et al., 2000). Many of the species were used in bioassays to estimate the toxicity effects in sediments (De Witt et al., 1988). *E. chilkensis* tolerated a wide range of salinity (5.22-35.35 ppt) and temperature (27.5-34 °C) conditions (Aravind et al., 2007). Nair et al. (1983) noted that *E. chilkensis* were absent during the month of February where salinity (33.61 ppt) and organic matter (72.74 mg g^{-1}) were high. In the present study period, *E. chilkensis* were not observed during the month of August 2011 and January 2013, but any type of disparity in the value of salinity and organic matter was not observed at that time. Its absence was either due to the predation by demersal fishes or it was accidental for not including in the grab sample. Raja et al. (2013) observed that in the continental shelf area of south east coast of India, 44 species of amphipods were identified, in which *Ampelisca* sp. formed the dominant group. They pointed out that based on the availability of food items demersal fishes primarily feed on amphipods and polychaetes and bivalves were consumed as secondary item.

Time scale study found that, in Vembanad estuary the diversity of amphipod species were varied over the years. Earlier seven species of amphipods were identified from the northern part of Vembanad estuary (Sheeba, 2000), now only 4 species were recorded. Previous studies noted that in Cochin backwater, *Grandidierella megnae* (Pillai, 1977) and *Corophium triaenonyx* (Nair et al., 1983) were the dominant amphipod species but now *C. geniculata* was more abundant especially in the southern zone. The southern stations were characterized with oligohaline zone, having higher organic carbon in the sediment, may act as a major source of food for *C. geniculata*. Similar observation of habitat preference of amphipods was earlier reported from the southern zone of Vembanad estuary (Anon, 2001). During 1994-1996 period, Caprillidae, Gammaridae and Talitridae were the three amphipod family reported from the southern zone of Vembanad estuary, in which caprillids were more abundant. From Gammaridae, two species such as *Eriopisa chilkensis* and *Quadrivisio bengalensis* and from Talitridae, *Parhyale hawaensis* formed the amphipod population (Anon, 2001). But in the present investigation caprillid species, *Quadrivisio bengalensis* and *Parhyale hawaensis* were not encountered from the whole Vembanad estuary. Similarly other common amphipod species *Grandidierella bonneri, Grandidierella gilesi* and *Melita zeylanica* were not observed in the present investigation (Table 7.14**)**. All these species, *Caprellid* sp., *Quadrivisio bengalensis*, *Grandidierella bonneri*, *Grandidierella gilesi*, *Melita zeylanica* and *Parhyale hawaensis* are considered as exclusively marine species (http://www.marinespecies.org). The decreased salinity pattern (oligohaline and mesohaline zone) in the Vembanad estuary over the years may possibly affect the distribution of these amphipod species. In Vembanad estuary, the isopod species *C. fluviatilis* were commonly observed over the years, but the species from Anthuridae were not mention in the earlier studies. Similar to *Cheiriphotis geniculata* the isopod species *A. sandalensis* and *X. orientalis* dominated in the southernmost stations especially in station 4. Apart from other stations, the sediment surface of this station was marked with the presence of large amount of filamentous algae. Similar to amphipods, these isopods may either prefer it as food or a place for protection and also the substratum may also provide habitat for other organisms. According to

Buschmann (1990), amphipods can utilize the algal resources as a refuge and also as a food source.

Table 7.14 Time scale changes in peracarid species from 1975 to 2011 - 2013 in Vembanad estuary especially in Cochin estuarine area.

	Study period	**1974-1975**	**1981**	**1996-1997**	**1994-1996**	**2002-2004**	**2005**	**2011-201**
Sl. No.	**Species**	**(Remani, 1979)**	**(Sarala Devi et al., 1991a)**	**(Sheeba, 2000)**	**(Anon, 2001) (South of TMB)**	**(John, 2009)**	**(Martin et al., 2011)**	**(Present study)**
I	**Amphipods**							
1	*Grandidierella bonneri*	+	+	+	-	-	+	-
2	*Grandidierella gilesi*	+	+	+	-	-	+	-
3	*Grandidierella* sp.	-	-	-	-	+	-	-
4	*Hyalehawaiensis*	+	-	-	+	-	-	-
5	*Eriopisa chilkensis*	+	+	+	+	+	+	+
6	*Photis digitata*	+	-	-	-	-	-	-
7	*Corophium triaenonyx*	-	+	+	-	+	+	-
8	*Quadrivisio bengalensis*	-	+	+	+	-	-	-
9	*Melita zeylanica*	-	+	+	-	-	+	-
10	*Caprillid* sp.	-	+	+	+	+	+	-
11	*Eriopisa* sp.	-	-	-	-	-	-	+
12	*Cheiriphotis geniculata*	-	-	-	-	-	-	+
13	*Gammarus* sp.	-	-	-	-	-	-	+
II	**Tanaidaeans**							
14	*Tanaid* sp.	-	+	-	-	-	-	-
15	*Ctenapseudes chilkensis*	+	-	+	-	+	+	+
16	*Apseudes gymnophobia*	+	-	+	-	+	-	-
III	**Isopods**							
17	*Cirolana fluviatilis*	+	+	-	-	+	-	+
18	*Synidotea variegata*	+	-	-	-	-	-	-
19	*Asellus* sp.	-	-	+	-	-	-	-
20	*Anthurid* sp.	-	+	+	-	+	-	-
21	*Apanthura sandalensis*	-	-	-	-	-	-	+
22	*Xenanthura orientalis*	-	-	-	-	-	-	+
23	*Isopod* sp.	-	-	+	-	-	-	-
IV	**Cumacea**							
24	*Cumacea* sp.	-	+	+	-	-	-	-
25	*Iphinoe* sp.	-	-	-	-	+	-	-
26	*Bodotria* sp.	-	-	-	-	-	-	+
	Total	**9**	**11**	**13**	**4**	**9**	**7**	**9**

From Vembanad lake, the species *A. sandalensis* Stebbing, 1900 was identified by H. S. Rao and M. Sharif during January 1928 and it was widely distributed in *Loyalty* Islands, Chilka Lake, South India, Madagascar, Mozambique, Mauritius, Egypt, Sumatra and South Africa (http://www.marinespecies.org/). *X. orientalis* Barnard, 1935 described form the Vattakayal region of Vembanad lake by H. S. Rao and M. Sharif during January 1928 and distributed in the Western Indo-Pacific region. The species preferred shallow depth range and the type specimen was described from India (http://www.marinespecies.org/). Anthuridean isopods are mainly inhabited in the littoral or shallow coastal environment. At the same time some are found in the freshwater to deep marine environment. In accordance with the narrow and elongate bodies of majority of the Anthurideans, most are resided in sediment burrows or tubes or in the algal mats (Cadien and Brusca, 1993). It categorized as carnivores, preyed upon tiny worms and different kinds of arthropods (Morton, 2003) and also most of the species are recognized as protogynous sequential hermaphrodites (Carlton, 2007). Overall, the feeding strategy of isopod ranged broadly from live and dead plant matter to animal matter. The marine species usually preferred algae, diatom, live and dead vegetation and a few utilize the flesh of dead animals (Mertz, 2004). According to Nienhuis and Van Ierland (1978) the herbivorous isopod species, *Idotea chelipes* mainly live in the Zostera meadows and usually feeding the eel grass leaves. The biting and scraping action breakdown the eelgrass leaves and it helps the bacterial decomposition rates of leaves. Similarly the isopod faecal pellets include undigested parts of eelgrass and formed a major source of food for detritus feeders and also help to increase its decomposition. It was also noted that the third isopod *Cirolana fluviatilis* observed only in the northern most station, usually associated with the polychaete tube structures whereas *A. sandalensis* and *X. orientalis* mainly found in the low saline areas and not observed in station 10. The habitat preferences of these three isopods were entirely different from each other. It was earlier reported that the predatory carnivorous species *C. fluviatilis* causes disturbance to the fishes and prawns and it feeds upon the live or dead wood borers, foulers, polychaetes, nematodes, prawns and fishes and also the catches trapped in nets especially in stake nets (Mathew et al., 1994). Similarly it can simply sense the availability of food items at long distances. It was observed that during sieving of the grab samples for macrofauna isolation, the retained *C. fluviatilis* in the sieve attacked the polychaetes (personal observation). Among the isopods, Cirolanids were voracious carnivores, functioning both as predators and scavengers (Carlton, 2007). Remani et al. (1983) point out that *C. fluviatilis* were sensitive and not observed in the polluted sites. Compared to other stations salinity was more in station 10, mainly due to the seawater intrusion from the nearby barmouth. In the present investigation, Pearson's correlation analysis revealed that *C. fluviatilis* showed significant ($P \leq 0.01$) positive correlation with salinity (r^2 = 0.528). *C. fluviatilis* was able to tolerate a wide range of salinity and the seasonal variation of salinity in Cochin estuary also supported suitable habitat for this species (Mathew et al., 1994). Similarly they pointed out that this particular isopod species were associated with the formation of mud heaps and tube forming habit, where the tube dwelling benthic amphipod (*Corophium triaenonyx*) were associated with the isopod and there was no association observed among polychaetes. This was contrasting to the present observation. Massive aggregation of tube dwelling polychaetes were observed in the Aroor region, where the isopods were associated more with tube dwelling polychaetes and no tube dwelling amphipods were present. According to Kensley et al. (1995) the knowledge about the small-scale ecological necessities of several isopod species are very less. Many species of isopods were observed in the calm, low saline, clayey, organic carbon rich areas of Indian River lagoon. They noted that the species *Xenanthura* *brevitelson* very common in the inlet whereas it was not observed in the lagoon area. Similarly the decapod diversity also acts as a major factor controlling the isopod variation (Gore et al., 1981). The Vembanad estuarine system act as a special biogeographic habitat where the fresh water zone on the south, brackish water on the middle and estuarine/marine zone on the north influencing the distribution of fauna. It is to be estimated that the isopod distribution was controlled by this transitional behaviour of the water body. Such type of habitat occurrence and transitional quality of isopod species were observed in the Indian River lagoon (Kensley et al., 1995). Isopods are suitably large, to form a significant source of food for juvenile especially the demersal fishes. So any alterations in the habitat may miserably affect the isopod population and in turn also influenced the fishery potential. In Vembanad estuary, tanaids have low diversity and represented by only one species but higher abundance was observed during the study period. Such type of observation of low diversity and higher abundance was found in the shallow water of tidal marsh (Kneib, 1992). The high density of tanaid species, *C. chilkensis* were mainly observed in the central and northern part of estuary whereas species of amphipods *E. chilkensis* and *Gammarus* sp. dominated in the northern most stations. Remani (1979), observed high densities *C. chilkensis* in the polluted regions and also found this pollution favouring species in the sewage polluted Cochin estuary. Time scale study found that, *C. chilkensis* were commonly observed in Vembanad estuary but the second dominant species *Apseudes gymnophobia* was not encountered in the present investigation.

In the Pearl River estuary, the distribution of tanaids were restricted to small regions and only very few species were observed (Bamber, 1997). The information of Tanaidacea from the Indian province is relatively very low (Venkataraman, 2005). *E. chilkensis* were nourished by detritus deposited on the surface sediment and considered as a major food for many fin fish and shell fishes of Chilka Lake and Cochin estuary (Chilton, 1924; George, 1974; Balasubramanian et al., 1979). The major shell fishes including *Penaeus indicus*, *P. monodon*, *P. semisulcatus*, *Metapeneus dobsoni* preferred crustacean diet along with other food items especially amphipods (Anon, 1978). Tanaids become a suitable live feed for fin and shell fishes due to its small size, cosmopolitan distribution, short generation time and moderately high fecundity (Priya, 2015). Experimental study proved that feeding of *P. indicus* and *M. dobsoni* showed that both voraciously feed on *C. chilkensis* and *E. chilkensis* and *Mera orthonides* (amphipods) as food (Anon, 1977). Among the benthic crustacean, amphipods were the most dominant group and from this *Corophium triaenonyx*, *Melita zeylanica* and *Quadrivisso bengalensis* were the dominant species and tanaids were represented by *C. chilkensis* and *A. gymnophobia* and isopod represented by *Cirolana* sp. in the prawn culture fields and ponds near Cochin area (Preetha, 1994).

In the present investigation, increased abundance of peracarids during PRM and PM season may be supported by the young ones. Harkantra and Parulekar (1985) noted that, increased abundance of macrofauna during PRM and PM season was mainly due the increased reproduction rate of macrofauna during the seasons. Seasonal variations in the density pattern and community structure of soft bottom crustaceans were related to its temperature-dependent life cycle and reproductive strategies (Moreira et al., 2008). Similarly the biotic characteristics such as competition, predation and migration also played a major role (Thorson, 1966). Temporal distribution of amphipods noted that their maximal density was observed during non-monsoon months. This may be due to the coincidence of breeding of amphipods in that period. In the tidal channels of Ria de Aveiro (NW Portugal) Cunha et al. (1999) observed that the maximum density of most of the amphipod species documented during warmer months and it was mainly supported due to its breeding peaks in that period. During the study period, both during PRM and PM season *C. geniculata* were the dominant species contributing more than 51 % of the sample. Similar type of situation also occurred during the second year period. But the percentage dominance of corresponding species was varied. During MN of both the year, *X. orientalis* formed the major dominant contributor among peracarids. This may be due to the high adaptability of *X. orientalis* to grow both in the fresh water and brackish water habitat. By observing the seasonal pattern of the dominance curve of peracarids, the curve is not an exact ‘S’ shaped curve denoting some disturbances. During both the years, there was no clear spatial and temporal density pattern of amphipods and isopods were observed. The density of peracaridan assemblages in the Ria de Aveiro exhibited significant spatial and temporal changes. Amphipods dominated the upper reach, whereas mysids were dominant in the lower and middle reaches. Throughout the year, isopods formed a dominant group in the upper reaches whereas tanaids showed maximum density in the middle reaches only during autumn and spring season (Cunha et al., 1999). Compared to isopod density, amphipods were denser in the southern region, where increased organic enrichment and oil seepage from the house boat and navigational boat make the region more polluted. According to Bonsdorff and Nelson (1981), in association with the isopod crustaceans, amphipods were more sensitive to different chemical disturbances especially oil pollution. But this observation was contrasting with the present amphipod and isopod density. Compared to polychaetes, amphipods were dominant during second year in stations 2, 3 and isopods in station 4. Sometimes it was due to the peculiar life characteristics of peracarids, it pursue the life cycle of k-strategy, having large body size, reduced metabolic rate, decreased fecundity, late maturation and extended reproductive rates (Luxmore, 1984; Arntz et al., 1994). It was also due to the good swimming behaviour of amphipods and isopods to get the food sources earlier than the other slow moving taxa like polychaetes and molluscs (Brandt, 1999). The small body size of amphipods and isopods were observed in the present investigation that may become an adjustment to escape from the predators. Such type of small body size adaptation to escape from fish predators was found in the sublittoral amphipods of soft sediments in the bay of Fundy (Wildish and Dadswell, 1985). During low tide, shore birds especially herons, egrets, sandpipers, plover etc. are mainly preyed on benthos in the estuarine shore areas and also the mangrove patches located in the northern part of estuarine bank (personal observation). The shore birds usually preferred organically rich areas have polychaetes such as *Nereis diversicolor* and amphipods *Corophium volutator* and *Hydrobia ulvae* and also in the freshwater inlets adjacent to estuaries where having local prey items are more through organic enrichment (Yates et al., 1993; Vanermen et al., 2006). Similarly benthos of the estuarine region was highly predated by migratory birds and also due to its predation pressure caused decrease in density of invertebrate preys in La Plata river estuary-Argentina (Martin et al., 2004). According to Wang et al. (2008) among the macrobenthic studies, most of the studies were concentrated on larger macrobenthic fauna and studies of smaller fauna particularly peracarids were usually avoided in the Pearl River estuary. Based on the diversity pattern, the details of diversity index of peracarids from the various aquatic systems are very limited. At the same time, the knowledge about the distributional pattern and habitat preference of peracarids in the soft bottom habitats of Vembanad estuary has been very less due to the scarcity of taxonomic literature and also the specialists in the groups. A generalized comparison of peracarids from different water bodies of the world with our findings was made (Table 7.15). It was noted that, among the peracarids amphipods have high abundance in majority of the region but its density and dominant species was highly variable. In the present investigation the spatial average value of diversity, richness and evenness index of peracarids was comparatively low. Such type of observation was found in Pearl River Estuary, China (Wang et al., 2010). They found that the resulting low diversity was mainly attributed to the dominance of a few species of tanaids from the peracaridan assemblages. Similarly heavy pollution of rivers by industrial wastewater sewage and intense aquaculture also formed a main reason for the low biodiversity. The unsuccessful operation of sewage treatment plants has consistently impacted the water quality of the Vembanad estuary (Asha et al., 2016). Similarly municipal wastewater and septic tank runoff into estuaries and modification of natural water flow can also result in increased eutrophication. Peracaridans act as a significant source of food for other benthic organisms and fishes of commercial importance (Woods, 2009). According to Stal et al. (2007) peracarids play a significant role as trophic resources for other crustaceans and fish populations. The feeding behavior of golden catfish, *Horabagrus brachysoma* from Vembanad estuary found that crustaceans formed a major food items and its percentage composition varied from negligible level to 37.5 % of their diet (Padmakumar et al., 2009). They also noted that due to various illegal exploitation and degradation of their natural habitats, the species become a rarity in Vembanad estuary. *Etroplus suratensis* formed a major fishery in Vembanad estuarine system (Asha et al., 2014). The food of juveniles of *E. suratensis* noted that, apart from filamentous algae, detritus, insect larvae and plant materials, micro crustaceans formed a major food item (Devaraj et al., 1975). Peracarids formed a key component in the diet of soft -bottom demersal fishes such as *Mullus barbatus*, where its diet contains upto 70% of crustaceans (Aguirre Villasenor, 2000). The food and feeding habits of *Tachysurus subrostratus* from Vembanad estuary observed that its food mainly consists of amphipods, tanaids and isopods (Ambily, 2016). The studies of food and feeding habits of demersal catfish *T. tenuispinis* observed that food of the fish is composed of a variety of organisms mainly including crustaceans. Among the 35 % contribution of crustacean groups, amphipods contribute a major share (Mojumder and Dan, 1979). According to Pillai (1979), in the estuarine food web of Vembanad estuary, macrofaunal groups such as amphipods, isopods. tanaids and cumaceans are key factors in the food chain. It consumed the mud and organic matter and which in turn preyed by demersal fishes and prawns. There were several reports that the finfish and shell fish fishery production of Vembanad estuary was declined over the years (Anon, 2001; Padmakumar et al., 2002; Asha et al., 2014). Among the total 80 species of fin fishes identified from the Vembanad estuary, 44 % were represented by demersal and 31 % utilizing bentho-pelagic environment (Asha et al., 2014). In this context, the protection of community assemblages of peracarid crustaceans was important for maintaining the energy transfer towards higher trophic level and ultimately conserve the demersal fishery production in Vembanad estuary.

The one-way ANOSIM test was used to test for differences among the peracarids from the south and north zone of TMB and found that there was a significant differences observed between the two zones. Based on the SIMPER analysis it was found that *E. chilkensis, C. geniculata* and *Gammarus* sp. played a major role in the maximum discrimination between these two zones, in which peracarids were more abundant in the northern zone. In the Odiel–Tinto estuary, South-Western Spain SIMPER analysis of crustacean species distribution indicated that, in the inner stations of estuary was represented by only few species such as isopod *Cyathura carinata*, the amphipods *Maera hamigera* and *Corophium acherusicum* or the decapods *Crangon crangon* and in the estuarine mouth area represented by large number of species (Sanchez – Moyano and Gracia – Asenio, 2010). Literature review revealed that *C. geniculata* was recorded only in the Vembanad estuary. In the present investigation, *C. geniculata* was recorded only in the southern zone of Vembanad estuary. It can be assumed that *C. geniculata* originated and established earlier in the Arabian Sea probably at the southern zone of Vembanad estuary, but later due to the geological process of Arabian Sea (Menon, 1913), formation of oligohaline southern zone, marsupium characteristics and absence of the planktonic larval stages and K-strategy life pattern of *C. geniculata* restricts its distribution in the southern zone. The species *E. chilkensis* were not observed in the southern zone of the estuary as the salinity was characterized by oligohaline zone where the annual average salinity was below 5 ppt. In the present investigation, Pearson's correlation analysis revealed that *E. chilkensis* ($r^2 = 0.529$) showed significant ($P \leq 0.01$) positive correlation with salinity, whereas *C. geniculata* showed significant ($P \leq 0.05$) negative correlation with salinity ($r^2 = 0.2$). According to Aravind et al. (2007) *E. chilkensis* can tolerate a wide range of salinity that varied from 5.22 - 35.35 ppt. It was also more abundant in the organically rich sediments of Cochin estuary and also considered as a tolerant species of organic pollution (Aravind, 2008). Ysebaert et al. (2002) point out that natural variation in salinity, sediment texture and organic content have been considered as the key factors in explaining the distribution and abundance of macrobenthic communities in various estuarine ecosystems. Salinity was master environmental parameter in the Vembanad estuary determining the distribution of macroinvertebrates (Asha et al. 2016). According to Menon et al. (2000) salinity variation in the estuary supports diverse species of biota based on their capacity to tolerate oligohaline, mesohaline or marine conditions. During the closure period of TMB the tidal water from the Arabian Sea was blocked towards the southern zone and an oligohaline zone prevailed during the whole year. These can have adverse effects on estuarine biota both in terms of its diversity and abundance.

Table 7.15 A generalized comparison of peracarids from different water bodies of the world

Study Locations	Dominant groups and species	H'	J'
In Western Mediterranean Sea- (Fernandez Gonzalez and Sanchez Jerez, 2011)	Amphipods (64.97%), tanaids (20.20%), cumaceans (14.41%) and isopods (0.42%). **Dominant species** – *Ampelisca* spp. (30.65% of total abundance)	NM	NM
An estuarine ecosystem of O Grove inlet, NW Iberian Peninsula - (Esquete et al., 2011)	Tanaids (52%), amphipods (35.8%) and cumaceans (3.12%) **Dominant species** - *Apseudes latreillii*	3.2	0.62
Northeast Water Polynya off Greenland - (Brandt, 1995)	Cumaceans (33.1%), followed by amphipods (31.4%), isopods (21.6%), mysids (10.6%), and tanaids (3.3%). **Dominant species** - *Leptostylis villosa*	3.1	0.74
Victoria Land (Ross Sea, Antarctica)- (Rehm et al., 2007)	Amphipods (66%), isopods (18%), tanaids (8%), cumaceans (7%), and mysids (<1%)	NM	NM
Tarifa Island, Strait of Gibraltar - (Guerra-Garcia et al., 2010)	Amphipods (89% represented by 32 species), isopods (11%; 12 species) and tanaids (1%; 2 species).	NM	NM
Tidal channels ofthe Ria de Aveiro (NW Portugal- (Cunha et al., 1999)	Amphipods 48 %, mysids 33% and isopods 17 %, cumaceans and tanaids 2 %	NM	NM
Pearl River Estuary, China – (Wang et al., 2010)	Tanaids (88.1%), amphipods (11%) and isopods (0.9%) **Dominant species** - *Discapseudes mackiei*	0.47	0.21
Soft bottoms at the Ensenada de Baiona (Galicia, NW Spain) – (Moreira et al., 2008)	Amphipods (59.3% - 67 species), Tanaids (29.7% - 3 species), isopods (3.8% - 10 species) and cumaceans (7.1% - 9 species). **Dominant species** – *Siphonoecetes kroyeranus*	2.6	0.69
Vembanad estuary – Present investigation (2011-2013)	2011-2012- Amphipods (40. 17 %, 4 species), isopods (39.9 %, 3 species) and tanaids (19.12 %, 1 species); **Dominant species -** *Xenanthura orientalis* (30.28%) 2012-2013- Amphipods (57.23 %, 4 species), isopods (37.4 %, 3 species) and tanaids (5.2 %, 1 species); **Dominant species –** *Cheiriphotis geniculata* **(48.2 %)**	1.69 1.34	0.78 0.57

*NM- Not Mentioned in the article; H' – Shanon diversity; J' -Pielou's evenness

The crustacean assemblage were mainly recognized according to the natural gradient from estuarine to marine environment, based on water and sediment characteristics such as dissolved oxygen, salinity, granulometry or organic content. This was confirmed by the BIOENV and canonical correspondence analyses. Factors such as temperature, substratum stability, particle size, food supply, organic enrichment, burrowing capacity, pollutants, predation and endogenous rhythms all together controlled the peracaridans (Weisshappel and Svavarsson, 1998; Cunha et al., 1999). The sediment type and organic carbon content formed the major abiotic factors (Robertson et al., 1989; De Grave, 1999) whereas predation become the major biotic feature (Nelson, 1979) influencing their distribution and abundance. In the present investigation BIOENV analysis showed that environmental factors including both water and sediment quality parameters were best explaining the peracaridan assemblages in Vembanad estuary. From the environmental factors water temperature, pH, depth, alkalinity, salinity, dissolved inorganic nitrogen, sediment temperature, sediment pH, moisture content, available phosphorus, sand, silt and clay have the major role in the distribution and abundance of peracarids in Vembanad estuary. Similarly CCA analysis also demonstrated the relationship of environmental parameters and peracaridan assemblages. Based on the BIOENV analysis Moreira et al. (2008) observed that sediment size formed a major factor influencing the distribution of peracarid crustaceans in Galician Rias estuarine system, Spain. Median grain size has highest correlation value (0.64), subsequently by fine sand (0.56) and very fine sand (0.50). Nair et al. (1983) observed that salinity was the prominent factor controlling the abundance of gammarid amphipods, followed by temperature and dissolved oxygen in Cochin estuary. Geetha and Bijoy Nandan (2014) observed that a combination of environmental factors including bottom water parameters such as temperature, pH, salinity and dissolved oxygen and sediment parameters such as silt, clay and sand fraction influenced the crustacean species distribution in Cochin estuary. Among the environmental parameters salinity and temperature were the key factors influencing the community structure of peracarid crustaceans in various aquatic systems (Yamada et al., 2007) and decisive factors include food availability and sediment texture (Bakanov, 2003). Wang et al. (2010) observed that temperature formed a primary decisive factor for the composition, abundance and diversity of peracarids in Qi'ao Island in the Pearl River Estuary, China. Based on the BIO-ENV analysis Esquete et al. (2011) noted that a combination of salinity, gravel, fine sand and silt/clay had the highest correlation (0.734) with Peracarid assemblages in the Zostera meadows in an estuarine ecosystem of O Grove inlet, NW Iberian Peninsula, in which silt/clay showed the best correlation (0.603). Sanders et al. (1965) noted that peracarid crustaceans were abundant in the deep sea environment, where amphipod was very less. *X. orientalis* and *A. sandalensis* assembled mainly in station 4 whereas amphipod *C. geniculata* preferred mainly available nitrogen and organic carbon. The species *X. orientalis* and *C. geniculata* were considered as endemic species mainly restricted its distribution in the southern zone of Vembanad estuary. Similarly *C. chilkensis* were preferred by sandy substratum and it mainly located in station 9 and 10. There are several reports that the distribution and density of tanaids were influenced by grain size of sediment (Lourido et al., 2008; Bakalem et al., 2009; de la Ossa Carretero et al., 2010; Ates et al., 2014). During 2012-2013 period, the distribution of *C. chilkensis* showed significant ($P \leq 0.01$) positive correlation with sand ($r^2 = 0.33$) and negative correlation with silt ($r^2 = 0.33$). The tanaid, *Apseudopsis latreillii* was more abundant in the bottom having softsand (Lourido et al., 2008; de la Ossa Carretero et al., 2010). Bakalem et al. (2009) observed that dominance of *Apseudopsis latreillii* in the sandy and muddy bottoms with depths ranging between 0 and 138 m in Algires coast (the southern Mediterranean). Riggio (1996) observed that *A. latreilli* was generally observed in the sediments with detritic deposits of Atlantic and

Mediterranean coastal waters. During 2012-2013 period, the distribution of *C. fluviatilis* showed significant ($P \leq 0.01$) positive correlation with clay ($r^2 = 0.67$) and negative correlation with silt ($r^2 = 0.23$). Similarly the clayey substratum preference of *C. fluviatilis* was earlier reported by Mathew et al. (1994). Lourido et al. (2008) suggest that in the subtidal sediments of Ria de Pontevedra (Galicia, NW Spain) the sedimentary composition and a number of environmental gradients such as hydrodynamism, sedimentation, carbonates, organic matter and the presence of seaweeds formed the major factors controlling the peracaridans spatial distribution. They noted that very fine sand was the variable that alone showed the highest correlation followed by depth and fine sand. In the present investigation, *E. chilkensis* showed significant negative correlation ($P \leq 0.05$) with organic carbon ($r^2 = 0.2$). In northeast Chukchi Sea, Alaska the sediment C:N values were correlates with the abundance distribution of amphipods and high abundance of benthic amphipods coincident with gray whale sightings (Schonberg et al., 2014). Spatial distribution patterns of *E. chilkensis*, *Gammarus* sp. and *Eriopisa* sp. indicated that its distribution was related to the grain size. During 2012-2013 period, the distribution of *E. chilkensis*, *Gammarus* sp. and *Eriopisa* sp. showed significant ($P \leq 0.01$) positive correlation with sand and negative correlation with silt. In the present investigation, species such as *E. chilkensis* and *Eriopisa* sp. were restricted its distribution in the northern zone of TMB, where the sand fraction was dominant. Similarly the CCA analysis revealed that median grain size, silt/clay, sorting coefficient and depth was the major variables explaining most of the peracaridan species variation in the Vembanad estuary. The CCA analysis revealed that factors such as pH, dissolved oxygen, salinity, percentage of sand, silt, organic matter, and pollutants influence the distribution of peracarids in the Vembanad estuary.

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Chapter 8

ECOLOGICAL INTEGRITY AND FUNCTIONALITY OF BENTHIC MACROFAUNA IN VEMBANAD ESTUARY

Contents

8.1 Introduction

Due to human pressures, the deterioration of ecological quality of coastal marine ecosystem was intense in recent years, in which some of these areas were considered as most distressed region in the biosphere (Lotze et al., 2006; Martinez-Crego et al., 2010). Variations in the diversity, abundance and biomass of macrofauna have been documented as responsive indicators of marine environmental disturbances either due to natural or anthropogenic origin (Quintino et al., 2006). As a result, in the benthic ecological quality measurements a lot of improvement has occurred in the methodologies and indices (Birk et al., 2012). Different univariate and multivariate methods have formed a major tool in the evaluation and monitoring of benthic community variations associated with the marine pollution. The major drawback for the application of univariate measures such as diversity, evenness and other metrics were related with the reduction of large information into a single index. Similarly, it was also difficult to distinguish between the diversity patterns formed due to the natural or anthropogenic factors (Warwick and Clarke, 1993). By comparing the accessibility of multivariate and univariate methods for the evaluation of community variations, it was noted that multivariate methods are more sensitive and its findings were difficult to understand by the non-scientists (Warwick and Clarke, 1991). Pearson and Rosenberg (1978) established a theoretical concept of the influence of organic enrichment on benthic communities. Based on this concept several benthic biotic indices were employed to analyze the natural and anthropogenic impacts on the estuarine and coastal waters (Hily, 1984; Rygg, 1985; Majeed, 1987; Smith and Rule, 2001). It was found that around 90 benthic indices were available to apply in marine and estuarine ecosystems (Martinez-Crego et al., 2010). Primarily by utilizing soft-bottom benthos a marine Biotic Index (BI) was developed by Glemarec and Hily (1981) and Hily (1984).

AZTI's Marine Biotic Index (AMBI) was a commonly used univariate index, developed to find out the impacts and the ecological quality status in soft-bottom marine benthic communities (Borja et al., 2000) and effectively used in various geographical areas having diverse impact sources (Borja et al., 2003 a, b, 2004 a). It was based upon the ecological models of Glemarec and Hily (1981) and Hily (1984) and theoretically depends upon the concept of ecological strategies (Pianka, 1970). Primarily it was created to monitor the ecological status of coastal and estuarine waters of Europe. Later, it was widely used in various impact sources, signifying its importance in measuring the particular or wide spread pollution effects (Borja et al., 2003a). In the multifaceted and inconsistent nature of marine ecosystem and the control of external pressures and also impact of environmental variables, the selection of the biotic indices was important in the independent evaluation of the ecological quality status (Zettler et al., 2007). According to Borja et al. (2003a) the applicability of indices in areas must be checked with the environmental pressure gradient data and both were appraised to confirm its ecological significance. According to Muxika et al. (2003) the assessment by using AMBI was not much influenced by the natural dynamism of the ecosystem. In comparison with various other indices, AMBI has numerous benefits. Apart from testing the various human impacts, it was capable to identify the spatio-temporal gradient of various impacts that occurred world wide (Borja and Tunberg, 2011). In a coastal ecosystem, the application of single index was not suitable for assessing the ecological status (Martinez-Crego et al., 2010) and so most of the researchers used more than one indices, mainly concentrating in the region where it was applied (Salas et al., 2004). In estuarine and coastal waters, AMBI and multivariate AMBI (M-AMBI) index have been extensively used to measure the ecological quality status of benthic communities impacted with various types of stress due to eutrophication, organic enrichment, oil pollution, heavy metal pollution, hypoxia, and harbor and dam construction (Xianxiang et al., 2016). Muxika et al. (2005) effectively established the link between various types of impact problems, such as discharges from drill cutting, submarine outfalls, harbour development activities, heavy metal loads, eutrophication, and various pollutant inputs. Similarly, the recovery from polluted systems such as the impact of sewage schemes, dredging activities, mud disposal, oil spill out, aquaculture etc. These indices are categorized as standard methods for water resources management in European coastal and estuarine waters (Borja et al., 2007).

In addition, for assessing the ecological quality of the environment, trophic group has been used as one of the parameters explaining the ecological functioning of the communities (Paganelli et al., 2012). The trophic structure analysis of benthic communities was valuable for understanding the energy flow at the ecosystem level (Paiva, 1993). But in the tropical region such study was more intricate due to the high diversity with low abundance of species. Due to this high diversity, it was difficult to find out the exact food webs largely contributing the energy flow. According to Pearson and Rosenberg (1978, 1987) the feeding mechanism is important for the study of functional diversity, which was considered as one of the chief mechanism structuring the marine ecosystem. Simultaneous evaluation of structural and functional mechanism of an ecosystem was important for obtaining an overall idea about the ecosystem health (Gessner and Chauvet, 2002). Based on the feeding habit/trophic guilds, macrobenthic communities were grouped differently by different authors. They were Carnivores (C), omnivores (O), filter feeders (FF), surface deposit feeders (SDF), and sub-surface deposit feeders (SSDF) (Diaz and Rosenberg, 1995; Rakocinski et al., 2000); planktophagous (Pl), phytophagous (Ph), detritivorous (D) groups (Baoming et al., 2008); Herbivorous (H), filter feeders/detritivores (F/D) varying the feeding type between filter feeders and detritivores, and carnivores/ detritivores (C/D) (Gaudencio and Cabral, 2007).

The factors which control the distribution of trophic groups put the community structure of fauna and its characteristics of functional groups were correlated with physical factors of the environment (Anger, 1975; Committo and Ambrose, 1985). In the benthic marine habitat, trophic structure analysis was used as an indicator of physical characteristics (Lastra et al., 1991) and the morphological characteristics of feeding organs of species also give information about the physical nature of the environment. According to Roth and Wilson (1998), feeding type and availability of food have a great role in controlling the establishment of macrobenthos in an environment. Similarly, environmental variables such as stability of water, dissolved oxygen, salinity, sediment texture, organic carbon, microbiomass were played a significant role on trophic composition of benthic fauna (Gaston et al., 1998). Benthic suspension feeders depend upon the tidal currents for the availability food through turbulent diffusion, permitting pelagic production to be accessible for benthic suspension/filter feeders (Putro et al., 2014). According to Olabarria et al. (2002) the soft bottom habitat was able to distinguish easily based on the relationship of resident trophic groups and environmental parameters. Sanders (1958) noted that suspension feeders were generally observed in the high energy conditions having coarse sediments, whereas deposit feeders were high in low energy situation having fine particles. According to Rakocinski et al. (2000), the observation of impact of pollution on trophic composition in estuaries was not easy because of different opportunistic taxa established facultative resistance to pollution pressure. Gaston et al. (1998) noted that due to the existence of relationship between benthic trophic structure, sediment contaminants and environmental parameters, the environmental disturbances were reflected on the variation of trophic structure, in which for suspension-feeders the quality and quantity of food formed a limiting factor. Several studies noted the dominance of opportunistic species in the organically-enriched and chemically-contaminated sediments, having reduced trophic complexity (Diaz and Rosenberg, 1995; Rakocinski et al., 2000).

Among the population of same species, the diet specificity was varied each other (Fauchald and Jumars, 1979). In the case of deposit feeders, particle size selection was significant characteristics (Fauchald and Jumars 1979; Word, 1990), in which it selects only a specific type or size of the particle or there was no specificity towards sediment particle features. Similarly the scales of preference of organism towards substratum characteristics, organic content etc. have high value (Taghon, 1982). Apart from the size preference of particle, specific gravity or surface texture characteristics was preferred by some taxa. So fondness of grains was not simply distinguished from each taxa (Lopez and Levinton, 1987). Based on the quality and quantity of food and also the environmental characteristics, the feeding guild composition was varied among the macrofaunal communities (Morin et al., 1985). In soft bottom habitat, deposit feeders influence the faunal distribution by destabilizing the sediment (Rhoads and Young, 1970). The surface deposit feeders were generally observed in the regions with very low hydrodynamic action on the bottom, because the currents control its feeding and movements (Wildish and Kristmanson, 1997).

8.1.1 Significance of the study

The Vembanad estuarine system is the largest coastal lagoon on the south west coast of India. Because of its special geographic position and their status as a connecting link between six major rivers and the Arabian Sea, the Vembanad estuary formed a habitat for various rare species of flora and fauna and its fishery resources ranged from freshwater to marine species. Similarly it acts as an important place for migratory water birds. However, the prosperous status of the Vembanad estuary has declined since the construction of Thaneermukkom Barrage during 1976. The operation of TMB has been a great impact on the estuarine morphology, biogeochemistry that overall affected the ecology, nutrient transport, primary production and fishery resources etc. (Padmakumar et al., 2002; Anon, 2001; Bijoy Nandan and Unnithan, 2004; Asha et al., 2016). As a result, these disparities may certainly influence upon the macrofaunal communities and benthic habitat. Apart from this, in recent years, the Vembanad estuarine system received large amount of pollution pressure from tourism, industrial, urban sewage and agriculture etc. The municipal wastewater discharges from Alappuzha town and also from the Cochin metro city has resulted in habitat degradation and affected the quality of water and sediment. Anthropogenic nutrient loading related with eutrophication were considered as a major threat to the structure and functioning of benthic communities (Veber et al., 2009). Trophic status studies revealed that Vembanad estuary was in a eutrophic state. These may impact the habitat and also the food conditions of benthic communities. Ultimately it affected the distribution of macrofaunal communities in the Vembanad estuarine system both spatially and temporally. Variation of quantity and food accessibility have a great role in controlling the metabolic reactions and movement of macrobenthic organisms and also indirectly changing their community structure, biodiversity and also interrelationships of species and its trophic groups. Thus, it is important to assess the ecological quality status of Vembanad estuary by means of benthic community indices and also its trophic guild. On the basis of AMBI and M-AMBI index, there was no study conducted in Vembanad estuary till now. Hence, the macrobenthic data from the monthly sampling of 24 cruises conducted during 2011 to 2013 were used for evaluating the ecological quality status of Vembanad estuarine system.

8.1.2 Review of Literature

By utilizing soft-bottom benthos, primarily a marine Biotic Index (BI) was developed by Glemarec and Hily (1981) and Hily (1984). Salen-Picard (1983) categorized four successive steps in the stressed environments of soft - bottom macrobenthic communities, first stage as unpolluted condition with high species diversity, second stage indicated as slightly disturbed condition with reduced species diversity and increase of tolerant species and appearance of new species. The third stage denoted as disturbed condition with decreased diversity and dominance of pollution tolerant indicator species and the fourth stage was grouped as azoic substrata. Similarly based on the relationship of soft-bottom macrofauna to the sensitivity towards rising stressed conditions especially due to the enrichment of organic matter, Hily (1984) and Glemarec (1986) proposed the possibility of five groups of soft-bottom macrofauna and it was abbreviated by Grall and Glemarec (1997). In Group I, the species are sensitive to organic enrichment and present in unpolluted environment (initial state). These are mainly represented by specific carnivores and some deposit feeding tubicolous polychaetes. In Group II category includes species which are unresponsive towards enrichment and observed in low densities with its temporal variations was not significant (from the initial state to slight unbalance). These include suspension feeders, less selective carnivores and scavengers. Group III category includes species tolerant to excess organic matter enrichment. These species are sometimes present in usual situations, but their population is enhanced by organic enrichment. This group mainly include surface deposit – feeders especially tubiculous spionids. Group IV comprise second order opportunistic species. Majority of the species are small sized polychaetes and includes mainly subsurface deposit-feeders. In Group V, contain first – order opportunistic species. Most of the species are deposit feeders, flourished in reduced conditions. Based on the status of these soft-bottom macrofaunal communities towards the pollution, gives a BI ranged from 0 to 7 (Hily, 1984; Hily et al., 1986; Majeed, 1987) (Fig. 8.1). But the application of the Hily model and calculation of each BI had some restrictions. So, based upon the percentage abundance of each ecological group Borja et al. (2000) proposed a single formula to get a continuous index, the Biotic Coefficient (BC). In Chesapeake Bay, USA three indices such as AZTI’s Marine Biotic Index (AMBI), Multivariate-AZTI’s Marine Biotic Index(M-AMBI) and Benthic-Integrity of Biotic Index (B-IBI) have been used (Borja et al., 2008c). Muniz et al. (2005) tested the use of AMBI index for evaluating the ecological quality of soft-bottom benthic communities in a new geographical region, the South America Atlantic region. It was found that the results obtained in the new region were in accordance with those obtained from the European estuarine and coastal environments, where AMBI was created. Similarly Reiss and Kroncke (2005) examined the utility of various benthic indices, mainly AMBI for the ecosystem quality assessment. Borja et al. (2003b) effectively used AMBI index for the assessment of marine ecological status of the region affected by a submarine outfall. Rullet and Dauvin (2007) investigated upon the five EcoQ (Ecological Quality) classes documented by the European Water Framework Directive (WFD) for the most used benthic indicators and diversity indicators (AMBI, BENTIX, BOPA, BQI, H', ITI and M-AMBI) were calculated to test the type of ecological classifications for transitional waters using these different indicators. It was found that, there was no chance to obtain a similar ecological classification with the selected indicators. Putro (2011) used the macrobenthic assemblage data of Southern Spencer Gulf, South Australia for the assessment of sensitivity of biotic indices for finding out the symptoms and the range of disturbances. It was found that AMBI can more able to find out the large scale spatial variation and less able to differentiate small variations the macrobenthic assemblages between sites, as have been revealed by Ecological Quality Ratio (EQR). In Hong Kong waters, Forde et al. (2013) used M-AMBI for evaluating the impact of seasonal hypoxia, salinity variation and organic enrichment formed by human origin. The ecological status of anthropogenically polluted and eutrophicated coastal waters of Yantai, Yellow Sea was estimated by using AMBI and M-AMBI (Li et al., 2013). Similarly the human impacted Huanghe (Yellow River) Estuary and its adjacent areas were evaluated by using AMBI and M-AMBI (Luo et al., 2014).

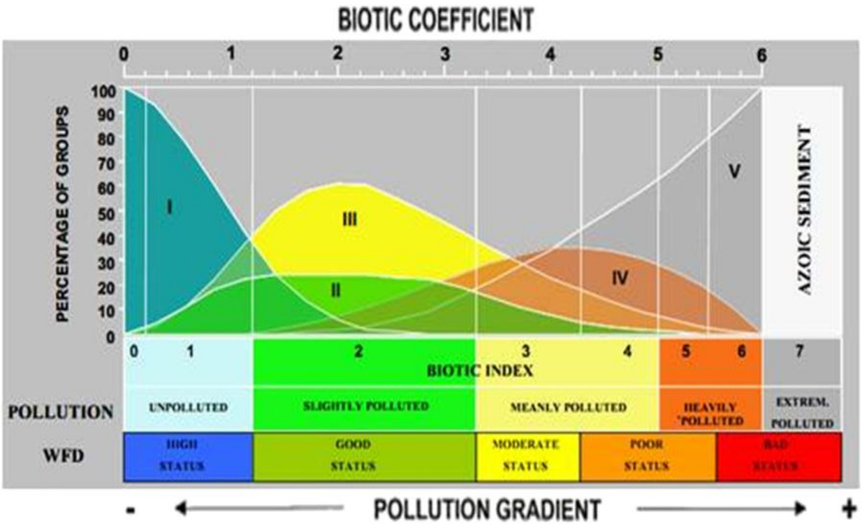

Fig. 8.1 Theoretical model, indicating the threshold values for AMBI modified from Hily (1984), Hily et al. (1986) and Majeed (1987), five ecological groups of soft-bottom macrofauna (EG I-V), based on their sensitivity towards pollution. The relative proportion of abundance of each group gives BI with eight levels (0 ± 7) and an equivalent continuous BC (values 0 to 6).

Ganesh et al. (2014) applied benthic opportunistic polychaetes amphipods index (BOPA) to investigate health condition of the benthic habitat of Visakhapatnam coast, India. Assessment of ecological quality of Vellar and Uppanar estuaries, southeast coast of India was done by Ajmal Khan et al. (2014) by using AMBI and M-AMBI. Based on these tools it was noted that Uppanar estuary showed polluted condition whereas the Vellar estuary was in a healthy condition. Similarly, Sivaraj et al. (2014) utilize AMBI and M-AMBI indices for evaluating the effluent stressed ecosystem in the coastal waters of Maharashtra, India. They recommended that AMBI indices were used as an acceptable tool for the estimation of quality of the entire coastal waters of India.

Sukumaran et al. (2014) used the BOPA index to assess the impact of 'Chitra' oil spill on the tidal pool macrofaunal communities of a tropical rocky shore of Mumbai coast, India, where the low BOPA index indicated that the area was good with few opportunistic species. Anu Pavithran (2015) observed the ecological quality of Fort Cochin and Dharmadam beach in Kerala, where AMBI and BOPA index showed an undisturbed to moderately disturbed condition. Sigamani et al. (2015) also assessed the ecological quality of Vellar–Coleroon estuarine system by adopting AMBI indices. Sivadas et al. (2016) monitored the ecological status of coastal waters of northwest and southeast coast of India by utilizing the temperate benthic indices. They also observed its effect of seasonal dynamism on the performance of benthic indices and recommended to use various indices for the correct evaluation of the environmental status. Feebarani et al. (2016) assessed the ecological status of macrobenthos in Cochin estuary by using AMBI and the multivariate-AMBI.

The trophic group approach was enhanced for the polychaetes with the "feeding guild" concept made by Fauchald and Jumars (1979). This concept was covering everything than the earlier classifications, as it was included the details of the movements and feeding approach of the worm in the guild determination. Later feeding guild classification of some polychaetes was modified by Dauer et al. (l98l) and Gaston (1987) but its basic pattern was not modified. The concept of feeding guild permits different species with related position in the food webs, which work as 'ecological units', and could substitute species in functional studies (Paiva, 1993). Majority of the trophic structure of macrofaunal studies were inclined to major grouping of fauna into: suspension feeders, deposit feeders (surface and subsurface), carnivores, herbivores and omnivores (Chardy and Clavier, 1988; Gaston and Nasci, 1988; Ricciardi and Bourget, 1999; Garcia-Arberas and Rallo, 2002; Coyle et al., 2007; Albano and Obenat, 2009; Cacabelos et al., 2009; Dolbeth et al., 2009). But many macrofauna showed more than one feeding mode (Okamura, 1990; Brown et al., 2000; Gaudencio and Cabral, 2007; Gray and Elliott, 2009). Earlier the correlation of distribution and abundance of functional groups with physical factors of the environment were noted by Rhoads (1974); Anger (1975); Committo and Ambrose (1985). Buchanan and Longbottom (1970) observed that the deposit feeders were more abundant in silty sediments containing more organic matter and nitrogen content than sandy sediments. Morin et al. (1985) noted that among the macrofaunal communities differences in the feeding guild composition were observed in relation to the food quality, quantity and also the environmental characteristics. The benthic macrofauna and the nature of sediment texture and its influence on the burrowing ability and filter feeding situations were investigated by Gerlach (1972).

Analysis of the distribution of trophic groups in relation to sedimentary features such as silt, clay and organic matter was investigated by Chardy and Clavier (1988). It was noted that white-sand community was preferred by surface deposit-feeders, in muddy substratum were chosen by filter-feeders, which take up the organic particles reaching towards the benthos either straight or through its resuspension. Grizzle and Monn (1989) highlighted the major role of tidal currents in food accessibility for benthic suspension feeders through turbulent diffusion, permitting pelagic production to be accessible for benthic suspension/filter feeders. Cocito et al. (1990) observed the relationships between trophic organization of benthic communities and organic matter content in Tyrrhenian Sea sediments. It was noted that the relationships between benthic consumers and organic matter do not have a specific model: where a mixture of conditions formed, in which the energetic input of sediment organic matter could in turn act as a feeding or stress source, determined by its quantity and quality. Gaston et al. (1998) observed the most important association between benthic trophic structure, sediment contaminants and environmental variables and also found that variation in the trophic structure was used an indicator of stress. Roth and Wilson (1998) studied the functional analysis by trophic guilds of macrobenthic community structure in Dublin Bay, Ireland. They found that trophic guild analysis easily helped to distinguish multivariate analysis of taxonomic basis of communities to explain the origin of pollution such as natural and anthropogenic, which was not possible with general multivariate analysis. Several authors noted the typical fine grained, muddy sediments with high organic content supported by deposit-feeders, whereas coarse-sandy sediments with low organic content and the high energy environments supported by suspension feeders and carnivores (Diaz and Rosenberg, 1995; Gaston et al., 1998; Rakocinski et al., 2000). Trophic relationships of the macrobenthic communities in Ubatuba Bay, southeastern Brazilian coast, were investigated seasonally by Santos and Pires-Vanin (2004), where among the benthic fauna carnivore and surface deposit-feeder polychaetes were dominant and the dominance of trophic groups was connected with factors such as mean grain size, fine sand, very fine sand, silt and clay contents. They mentioned that due to the survival of surface deposit feeders in the surface layer of the substratum, these were the first organisms more prone to the environmental distress caused by pollution.

Based on the functional guild approach, the trophic structure of benthic communities in the Tagus estuary and adjacent coastal shelf was characterized by Gaudencio and Cabral (2007). They found that surface deposit feeders were abundant in the Tagus estuary, while in the adjacent coastal shelf it was dominated by both surface- deposit feeders and filter feeders, in which, salinity, depth and sediment composition were the main factors affecting the spatial distribution of macrobenthos with particular trophic structure there. Dolbeth et al. (2009) observed the spatial variation in the feeding guild composition of subtidal macrobenthic community in accordance with the depth variation and also related to the hydrodynamic feature of the sea floor in Ancao Peninsula, southern Portugal. Taxonomic and feeding guild classification for the marine benthic macroinvertebrates of the Strait of Georgia, British Columbia was done by Macdonald et al. (2010). They made a trophic coding system, which grouped macrobenthos according to its feeding pattern and functional roles in the community for evaluating the carbon pathways. Trophic and functional organization of the benthic macrofauna in the lagoon of Boughrara – Tunisia (South West Mediterranean Sea) was investigated by Khedhri et al. (2015). They noted that the trophic structure of the macrobenthic community was generally dominated by selective deposit feeders, and found that communities were more connected to the availability of trophic resources than to disturbance. A comparison of trophic groups of macrobenthos in the organically enriched sediments of temperate and tropical regions was studied by Putro et al. (2014). They found that organic carbon content in the sediment was related with the species diversity of deposit feeders and sub-surface deposit feeders but reaction of trophic groups of macrobenthos to environmental disturbance was almost similar at both tropical and temperate regions.

The details of feeding guild composition of macrobenthos in the Indian coasts are very few. From the macrobenthic studies of shelf waters of the west coast of India, Joydas (2002) found that the macrofaunal biomass was more towards the southern region such as off Cape Comorin and off Thiruvananthapuram, where the filter feeding organisms were more. The feeding guilds of benthic polychaete species from the deltaic Sundarban Biosphere Reserve on the northeast coast of India was listed by Sarkar et al., 2005. Jayaraj et al. (2008) noted that in the North West Indian shelf region macrobenthic filter feeders were usually chosen medium grain sized sediment with low organic matter. It was also noticed that deposit feeding polychaetes dominated in shallow depths while carnivore species in the middle depths. The feeding types of macrofauna in Cochin estuary were studied by Aravind (2008), where deposit feeders were more abundant. It was also observed that during early premonsoon the deposit feeders were more abundant (44.01 %), followed by pre-monsoon (37.88%) and monsoon (40.66%) in the central part of Cochin estuary. The feeding guild of macrofaunal polychaete community in the western Indian continental margin was studied by Abdul Jaleel (2012). The whole region was dominated by surface deposit feeding polychaetes mainly in the shelf edge and at higher depths subsurface deposit feeders, carnivores and omnivores were more. The feeding guild type of polychaetes from the continental shelf region of southeast coast of India was investigated by Manokaran et al. (2013). They found that among the polychaete species, 65% composed of surface deposit feeders, 31% carnivores, 3% filter feeders and less than 1% were subsurface deposit feeders and omnivores. During the study of soft-bottom macrofauna from the ports of Mumbai, India, Mandal et al. (2013) observed the dominance of deposit feeders indicating a stressed condition throughout the period. Macrofaunal functional diversity in the Kalbadevi Bay along the west coast of India was studied by Sivadas et al. (2013) and found that temporal environmental changes have great impact upon the macrofaunal functional diversity pattern.

8.2 Results

8.2.1 Ecological quality of soft-bottom macrobenthos

The list of macrobenthic species/ groups with their pollution gradient of ecological groups is given in Table 8.1. The AMBI values for the macrobenthic species ranged from1.08 to 4.79 and 0.48 to 3.63 during first and second year respectively. During first year, after the classification in station 1, 42.5 % of macrofauna were in EG V category and the station belonged to moderately disturbed status with an AMBI value of 4.79 (Fig. 8.2 and Fig. 8.3). In station 2, 95.3 % of the macrofaunal species were in EG- IV category and the station was classified as moderately disturbed. The EG-I were dominated in station 3 (58.4 %), 4 (56.7 %), 7 (61 %) and 8 (42 %) and the AMBI value showed the slightly disturbed status, where the ecological group V were contributing only less percentage (< 10 %). The AMBI value of station 5 (4.0), 6 (3.31) and 9 (4.47) were indicating that these sampling stations were classified as moderately disturbed sites with an unbalanced benthic community status. Low AMBI value of station 10 was classified as undisturbed sites, where 70.9 % of the macrofaunal species were in EG- I category. In contrast to first year, during the second year in station 2, 5, 6, 9 and 10 were representing slightly disturbed condition (Fig. 8.4 and Fig. 8.5). At the same time, station 3 and 4 were classified as undisturbed sites where AMBI value was < 1. Based on the M-AMBI values, during first year the ecological grade of Vembanad estuary varied between poor and high status. The current study indicated that station 2 and 1were classified as poor (0.24) and moderate (0.4) status respectively. At the same time, stations 3 to 9 were in good condition and station 10 as high quality status (Table 8.2 and Fig. 8.6). During second year, M-AMBI values were representing high (0.89) to moderate (0.45) condition (Table 8.3 and Fig. 8.7). During first year BENTIX index varied from poor to high ecological status. The poor benthic status was observed in station 1 (2.2), station 2 (2.1) and station 9 (2.2) and high benthic status was observed in station 7 (4.5) and station 10 (5.2). Moderate status was observed in station 5 (2.7) and station 6 (3) (Fig. 8.8). At the same time during second year the BENTIX index obtained ranged between 3.1(station 1) to 5.5 (station 4) (Fig. 8.9) and get three ecological condition ranging from moderate to high.

Table 8.1 List of macrofauna and their ecological group values (EG) from the Vembanad estuary during 2011 – 2013 period

Sl. No.	Macrobenthos	Ecological groups	Sl. No.	Macrobenthos	Ecological groups
1	Nemertea	III	17	*Scoletoma impatiens*	II
2	Turbellaria	II	18	*Owenia* sp.	I
3	Nematoda	III	19	*Pectinaria* sp.	I
4	Oligochaeta	V	20	*Diopatra neapolitana*	I
5	*Namalycastis indica*	IV	21	*Nereidae* sp.	III
6	*Ceatonereis* sp.	II	22	Apseudidae	III
7	*Micronephthys oligobranchia*	II	23	Cirolana sp.	II
8	*Parheteromastus tenuis*	V	24	Anthuridae	I
9	*Mediomastus* sp.	III	25	Mysida	II
10	*Sigambraparva*	III	26	Cumacea	I
11	*Prionospio cirrifera*	IV	27	*Eriopisa* sp.	I
12	*Prionospio pinnata*	IV	28	*Photis* sp.	I
13	*Dipolydora flava*	IV	29	*Gammarus* sp.	I
14	*Glycera alba*	IV	30	*Alpheus* sp.	II
15	*Cossura costa*	IV	31	*Chironomus* sp.	III
16	*Aphelochaeta filiformis*	III	32		

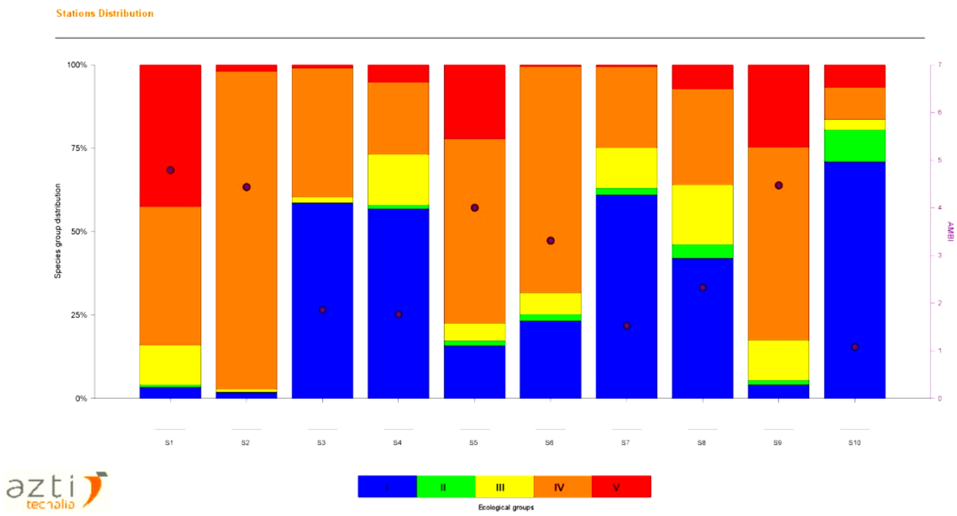

Fig. 8.2 Spatial distribution of percentage contribution of ecological groups in Vembanad estuary during 2011 – 2012 period

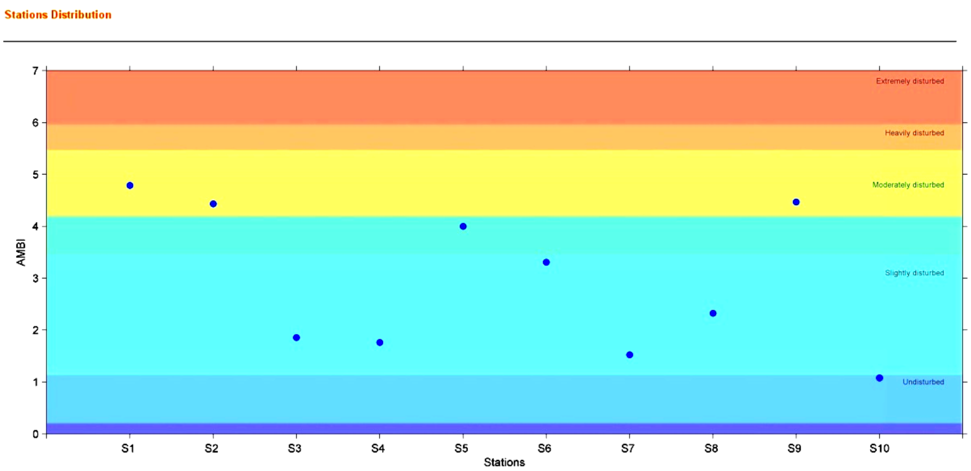

Fig. 8.3 Spatial distribution and AMBI status in Vembanad estuary during 2011 – 2012 period

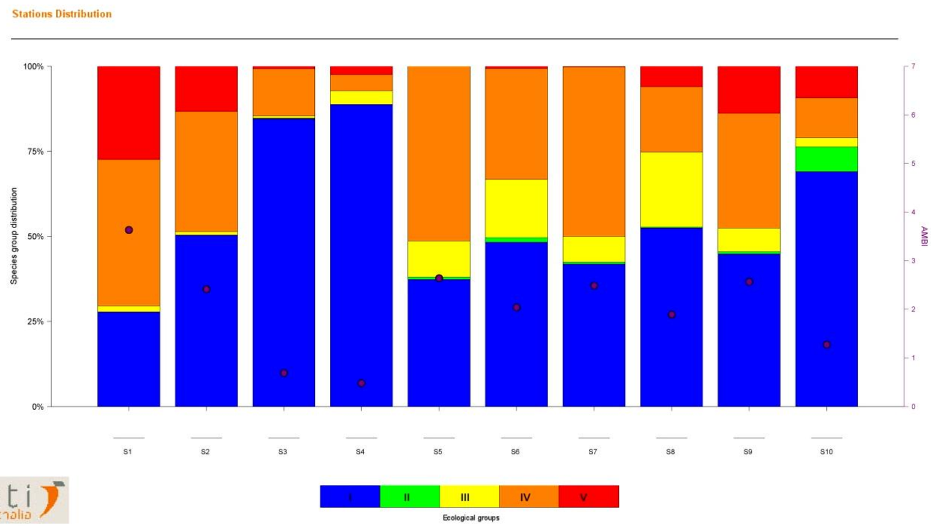

Fig. 8.4 Spatial distribution of percentage contribution of ecological groups in Vembanad estuary during 2012 – 2013 period

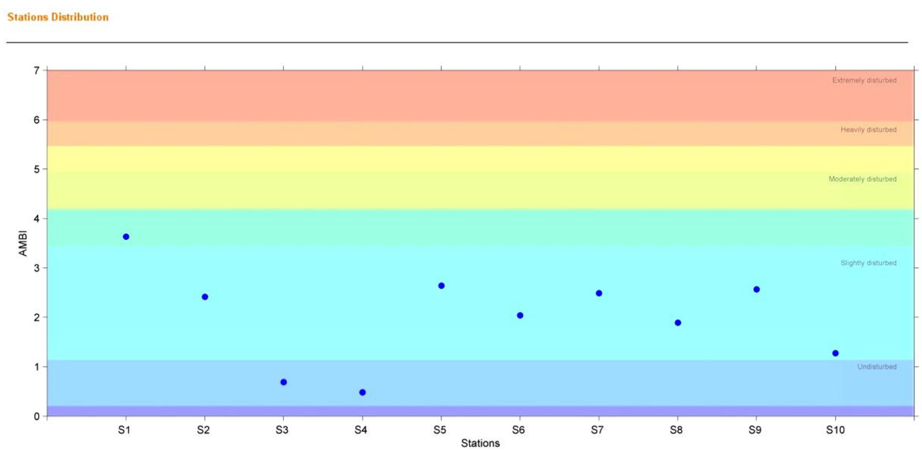

Fig. 8.5 Spatial distribution and AMBI status in Vembanad estuary during 2012 – 2013 period

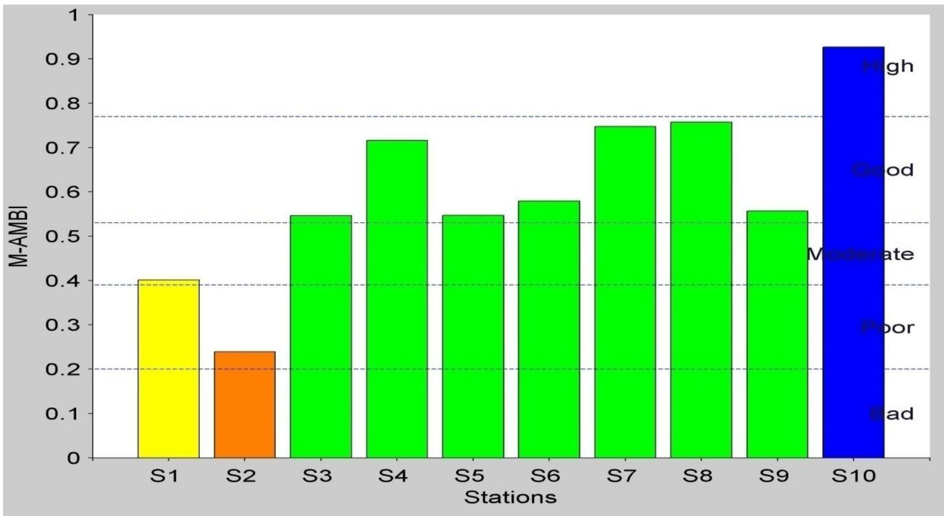

Fig. 8.6 Spatial distribution of M-AMBI values indicating the ecological status of Vembanad estuary during 2011 – 2012 period

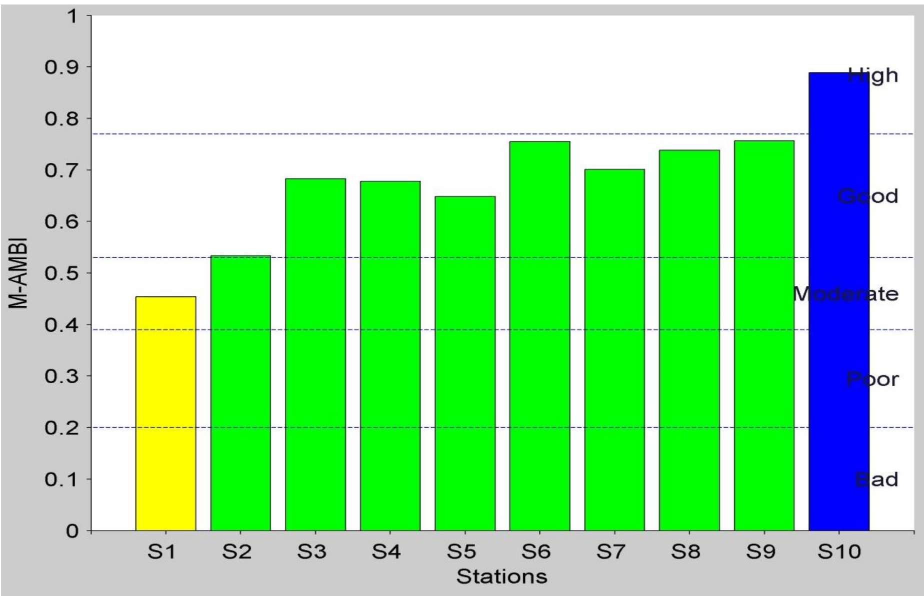

Fig. 8.7 Spatial distribution of M-AMBI values indicating the ecological status of Vembanad estuary during 2012 – 2013 period

Table 8.2 AMBI, BI classification and M-AMBI status in Vembanad estuary during 2011 – 2012 period

Stations	AMBI	BI	Disturbance classification	M-AMBI	Status
S1	4.8	4	Moderately disturbed	0.40	Moderate
S2	4.4	4	Moderately disturbed	0.24	Poor
S3	1.9	2	Slightly disturbed	0.55	Good
S4	1.8	2	Slightly disturbed	0.72	Good
S5	4.0	3	Moderately disturbed	0.55	Good
S6	3.3	3	Moderately disturbed	0.58	Good
S7	1.5	2	Slightly disturbed	0.75	Good
S8	2.3	2	Slightly disturbed	0.76	Good
S9	4.5	4	Moderately disturbed	0.56	Good
S10	1.1	1	Undisturbed	0.93	High

Table 8.3 AMBI, BI classification and M-AMBI status in Vembanad estuary during 2012 – 2013 period

Stations	AMBI	BI	Disturbance classification	M-AMBI	Status
S1	3.6	3	Moderately disturbed	0.45	Moderate
S2	2.4	2	Slightly disturbed	0.53	Good
S3	0.7	1	Undisturbed	0.68	Good
S4	0.5	1	Undisturbed	0.68	Good
S5	2.6	2	Slightly disturbed	0.65	Good
S6	2.0	2	Slightly disturbed	0.76	Good
S7	2.5	2	Slightly disturbed	0.70	Good
S8	1.9	2	Slightly disturbed	0.74	Good
S9	2.6	2	Slightly disturbed	0.76	Good
S10	1.3	2	Slightly disturbed	0.89	High

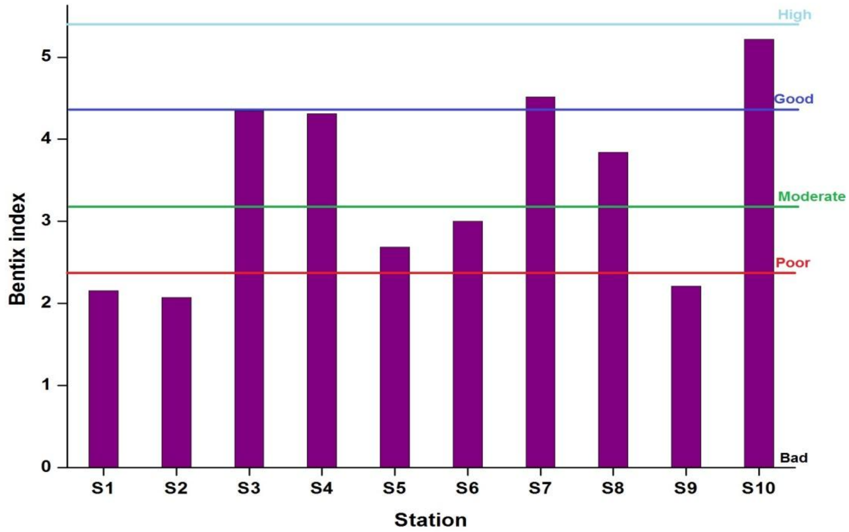

Fig. 8.8 Spatial distribution of Bentix index indicating the ecological status of Vembanad estuary during 2011 – 2012 period

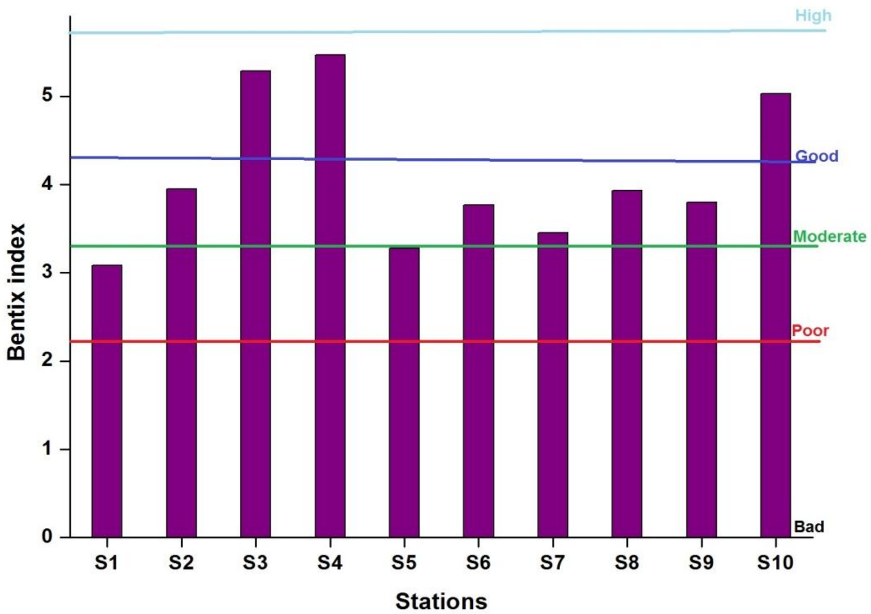

Fig. 8.9 Spatial distribution of Bentix index indicating the ecological status of Vembanad estuary during 2012 – 2013 period

In this study, BOPA index denoted that the ecological status of Vembanad estuary was varied from ‘bad’ to ‘high’ status during the first year and ‘bad’ to ‘good’ status during second year. During first year low BOPA value indicated that the ecological status of station 10 (0.04) was in a pristine condition during first year and the high value of station 2 (0.26) represented an impacted macrobenthic community structure (Table 8.4). Whereas during the second year, majority of the stations except station 6 (0.17) and 7 (0.14) were in moderate condition and station 5 (0.25) was observed as an impacted site (Table 8.5).

8.2.1.1 Data Analysis

8.2.1.2 Correlation analysis

During both the years, the Pearson correlation coefficients were calculated to assess the relationship between AMBI, M-AMBI, BOPA and BENTIX for all the stations (Table 8.6 and Table 8.7).

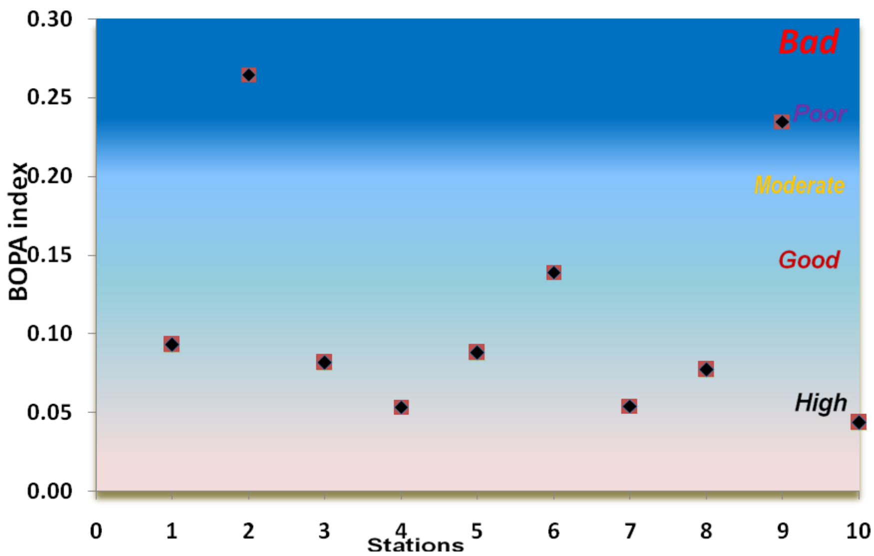

Fig. 8.10 Spatial distribution of BOPA index indicating the ecological status of Vembanad estuary during 2011 – 2012 period

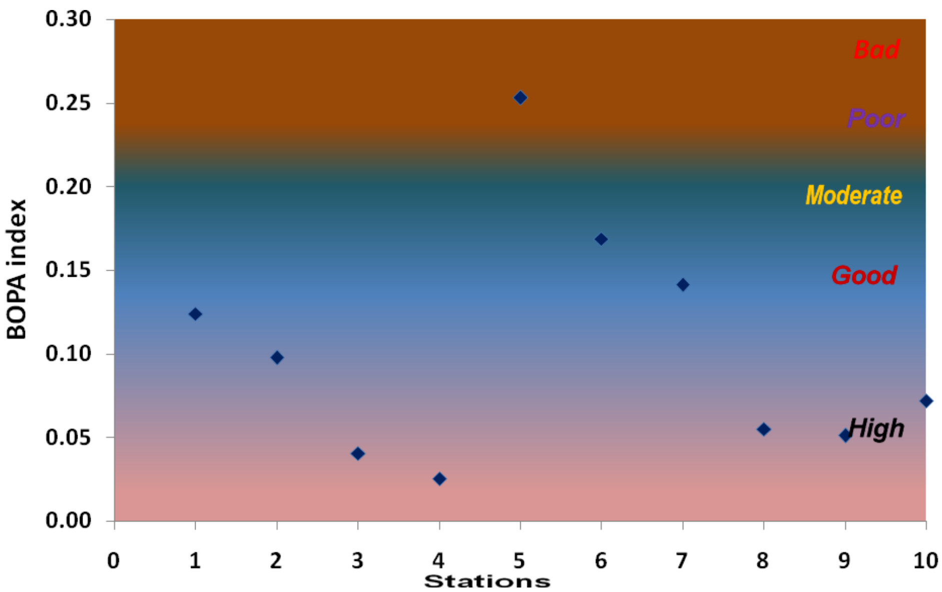

Fig. 8.11 Spatial distribution of BOPA index indicating the ecological status of Vembanad estuary during 2012 – 2013 period

Table 8.4 BENTIX and BOPA index and ecological status in Vembanad estuary during 2011 – 2012 period.

Stations	BENTIX index	BENTIX-Ecological classification	BOPA index	BOPA-Ecological classification
S1	2.2	Poor	0.09	Good
S2	2.1	Poor	0.26	Bad
S3	4.3	Good	0.08	Good
S4	4.3	Good	0.05	Good
S5	2.7	Mode	0.09	Good
S6	3.0	Mode	0.14	Moderate
S7	4.5	High	0.05	Good
S8	3.8	Good	0.08	Good
S9	2.2	Poor	0.23	Poor
S10	5.2	High	0.04	High

Table 8.5 BENTIX and BOPA index and ecological status in Vembanad estuary during 2012 – 2013 period.

Stations	**BENTIX index**	**BENTIX-Ecological classification**	**BOPA index**	**BOPA-Ecological classification**
S1	3.1	Moderate	0.12	Good
S2	4.0	Good	0.10	Good
S3	5.3	High	0.04	Good
S4	5.5	High	0.03	Good
S5	3.3	Moderate	0.25	Bad
S6	3.8	Good	0.17	Moderate
S7	3.5	Good	0.14	Moderate
S8	3.9	Good	0.05	Good
S9	3.8	Good	0.05	Good
S10	5.0	High	0.07	Good

Table 8.6 Correlation coefficient between different biotic indices during 2011-2012period.

	AMBI	**M-AMBI**	**BOPA**	**BENTIX**
AMBI	1	-.823**	.724*	-.991**
M-AMBI		1	-.719*	.843**
BOPA			1	-.772**
BENTIX				1

Table 8.7 Correlation coefficient between different biotic indices during 2012-2013period.

	AMBI	**M-AMBI**	**BOPA**	**BENTIX**
AMBI	1	-0.525	0.553	-.953**
M-AMBI		1	-0.212	0.416
BOPA			1	-.673*
BENTIX				1

8.2.2 Trophic group analysis of macrofauna

As a whole, six feeding guilds were identified from the macrofaunal community in Vembanad estuarine system. During 2011-2012 period among the trophic categories, carnivores were the most abundant and contributing 46.16 % of the trophic guild (Fig. 8.12). Other trophic guild such as filter feeders (25.36 %), surface deposit feeders (11.25 %) and subsurface deposit feeders (10.62 %) were in decreasing order of abundance. In contrast with 2011-2012 period, during 2012 – 2013 period, the filter feeders were the most abundant (52.25 %) among the trophic categories followed by carnivores (29.36 %), subsurface deposit feeders (8 %) and surface deposit feeders (7.08 %) in decreasing order (Fig. 8.13).

Spatial distribution of macrofaunal feeding guilds in Vembanad estuary is given in figure 8.13 and 8.14. During 2011-2012 period, in station 1, C constituted 35.38 %, followed by SSDF (34.75 %) and FF (21.23 %). Similar to station 1, C showed the highest percentage in station 2 (95.33 %). In station 3, FF (44.53 %) and C (44.3 %) were constituted the major share. In station 4, C contributed 61.2 %, followed by O (12.55 %). SSDF and SDF formed the most dominant group in station 5 (43.75 %) and station 6 (40.84 %) respectively. Both station 7 (42.25 %) and 8 (26.4 %), C dominant over other feeders and followed by FF. SDF and SSDF were the most dominant group and constituting 52.32 % and 27.1 % respectively. In station 10, C (45.76 %) and FF (38.97 %) were the abundant group contributing the major share among the trophic groups. Similar to 2011-2012, during 2012-2013 period, C was dominant over other feeders (41.88%), followed by SSDF (29.14 %) and FF (27.11 %) in station 1. In station 2 (47.99 %) and 3 (67.65 %), FF were the dominant forms followed by C (34.86 % (station 2) and 27.9 % (station 3)). In station 4 (75.96 %) and 6 (30.17 %) C constituted the major share over other trophic groups. FF were dominant at station 7 to 9 and contributing 49.2 %, 63.85 % and 77.38

% respectively. In station 10, C (44.83 %) and FF (41.24 %) were the dominant forms.

Based on the overall trophic group distribution, spatially it was noted that during 2011-2012 period, filter feeders (61.9 %), carnivores (39.9 %) and omnivores (27.5 %) were dominant at station 10. Both surface deposit feeders (51.3 %) and subsurface deposit feeders (28.1 %) were dominant at station 9, in the case of grazers, which were present only in station 9 (100 %). During 2012-2013 period, overall the filter feeders were dominant at station 9 (38.8 %) followed by station 10 (18.8 %). In the case of carnivores, the overall maximum percentage was observed in station 10 (36.46 %) followed by station 4 (17.76 %). Surface deposit feeders and subsurface deposit feeders were dominant at station 9 (29.86 %) and station 2 (21.7 %) respectively. Both omnivores (33.36 %) and grazers (56 %) were dominant at station 9.

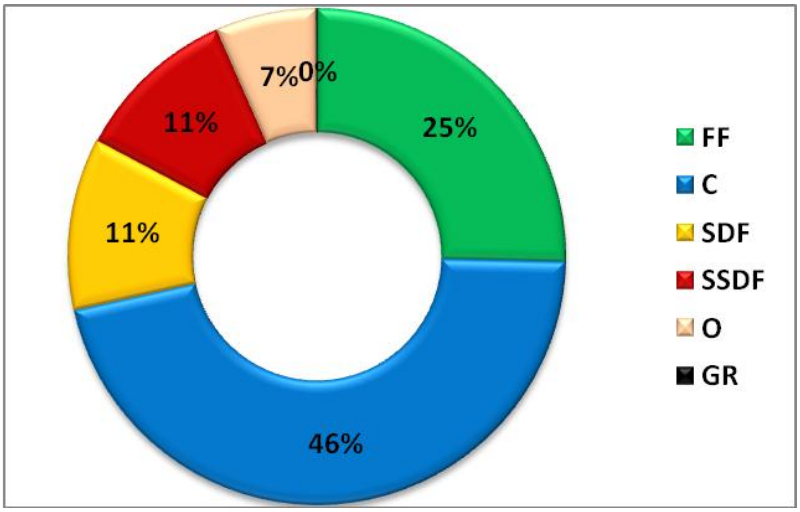

Fig. 8.12 Percentage composition of trophic groups of macrofauna in Vembanad estuary during 2011 – 2012 period

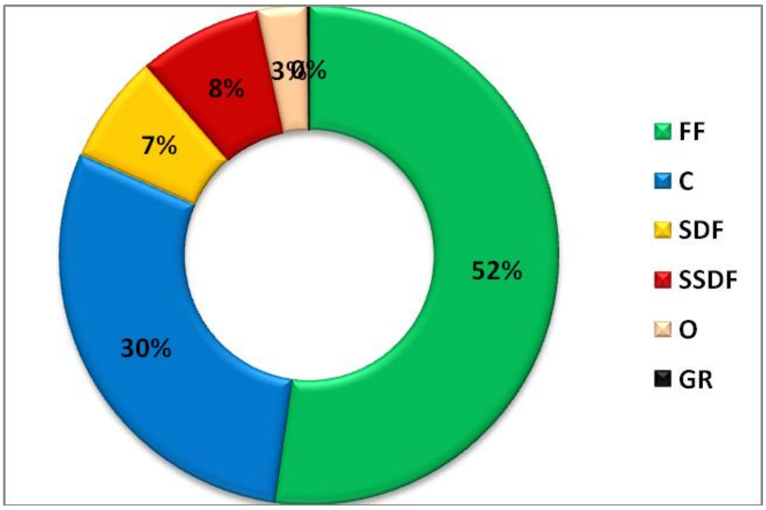

Fig. 8.13 Percentage composition of trophic groups of macrofauna in Vembanad estuary during 2012 – 2013 period

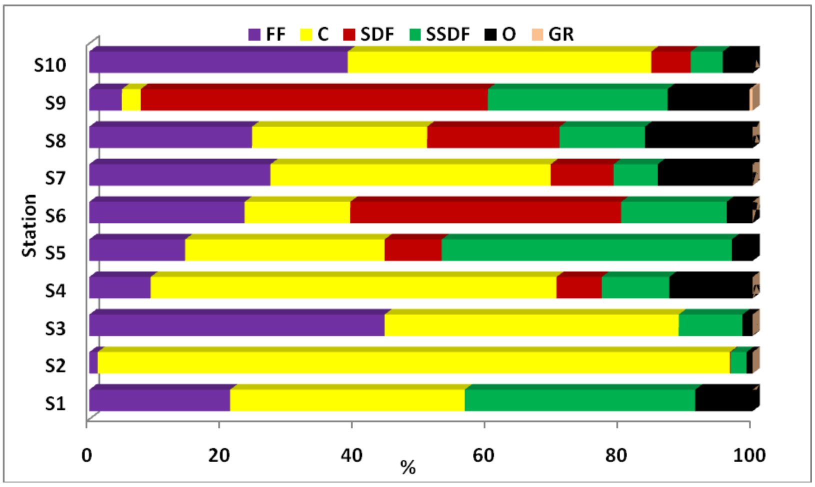

Fig. 8.14 Spatial distribution of macrofaunal feeding guilds in Vembanad estuary during 2011 – 2012 period

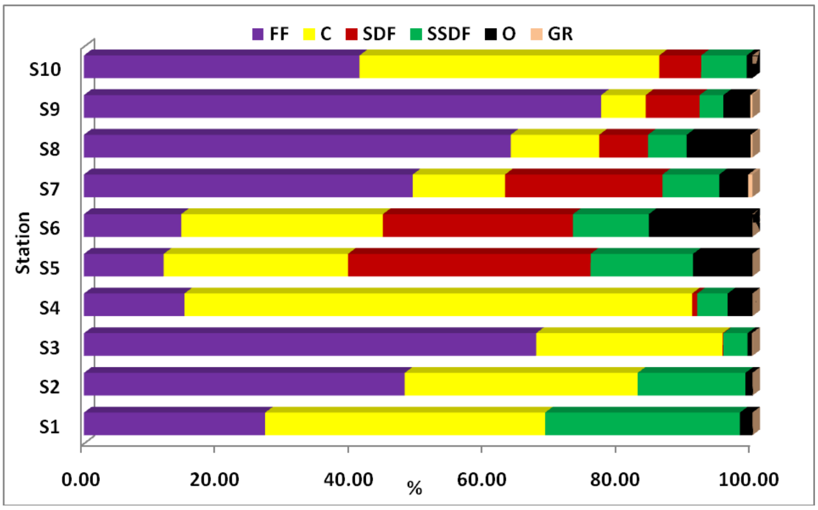

Fig. 8.15 Spatial distribution of macrofaunal feeding guilds in Vembanad estuary during 2012 – 2013 period

8.2.2.1 Data Analysis

8.2.2.2 Cluster analysis and ANOSIM

Based on the abundance data of trophic groups, the Bray-Curtis similarity cluster indicated that there was a distinct distribution of macrobenthic feeding guilds in Vembanad estuary. During 2011-2012 period, dendrogram showed two main clusters, in which cluster 1 consisted of majority of the stations, including station 1 to 8 (Fig. 8.16). In cluster I, station 2 stands apart from other stations and it was characterized by high abundance of carnivores (95.3 %). Dendrogram of the abundance of trophic categories showed, high similarities between the FF and C (89.57 %) and between SSDF and O (87.22 %). The GR type of feeding category has very low similarity percentage (8.71 %) towards the other groups (Fig. 8.17). ANOSIM revealed that there was no significant spatial variation of macrofaunal feeding guilds among southern and northern zones (Global R= 0.15, $P > 0.01$). During 2012 – 2013 period, two main clusters were obtained from the Bray-Curtis similarity analysis, in which southern most stations formed cluster I (Fig.

8.18). Cluster II divided into two, where southern stations such as station 5 and 6 formed a group and all northern stations grouped together. ANOSIM revealed that there was a significant spatial variation of macrofaunal feeding guilds among southern and northern zone (Global R= 0.333, $P < 0.05$). During second year the dendrogram of the abundance of trophic categories showed similar type of clustering but its percentage similarity varied (Fig. 8.19).

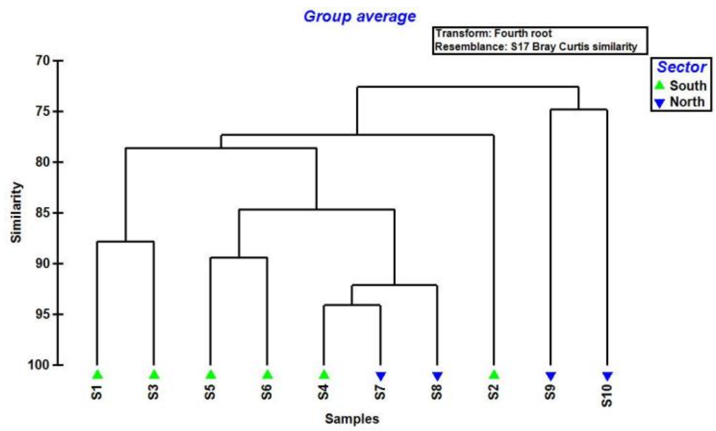

Fig. 8.16 Dendrogram showing the hierarchial clustering of the spatial similarities of macrofaunal trophic group abundance in Vembanad estuary during 2011-2012 period

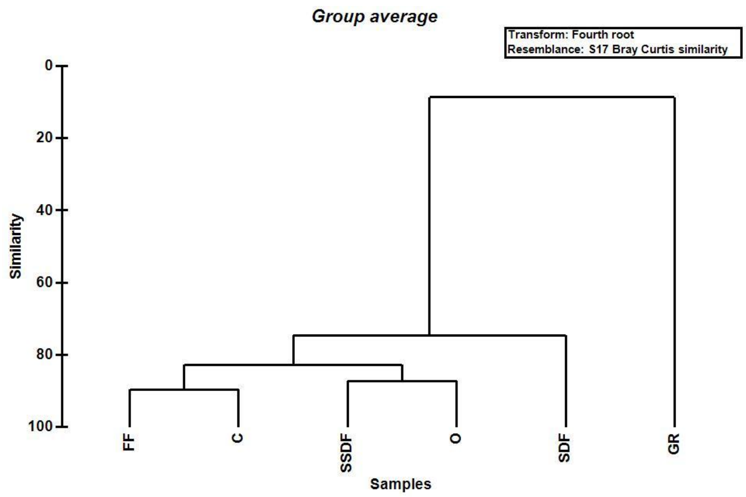

Fig. 8.17 Dendrogram showing the hierarchial clustering of the similarities of different trophic group in Vembanad estuary during 2011-2012 period

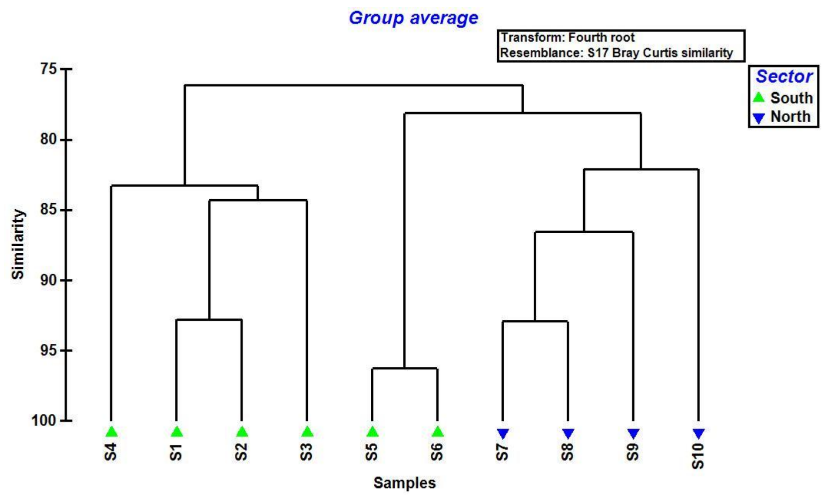

Fig. 8.18 Dendrogram showing the hierarchial clustering of the spatial similarities of macrofaunal trophic group abundance in Vembanad estuary during 2011-2012 period

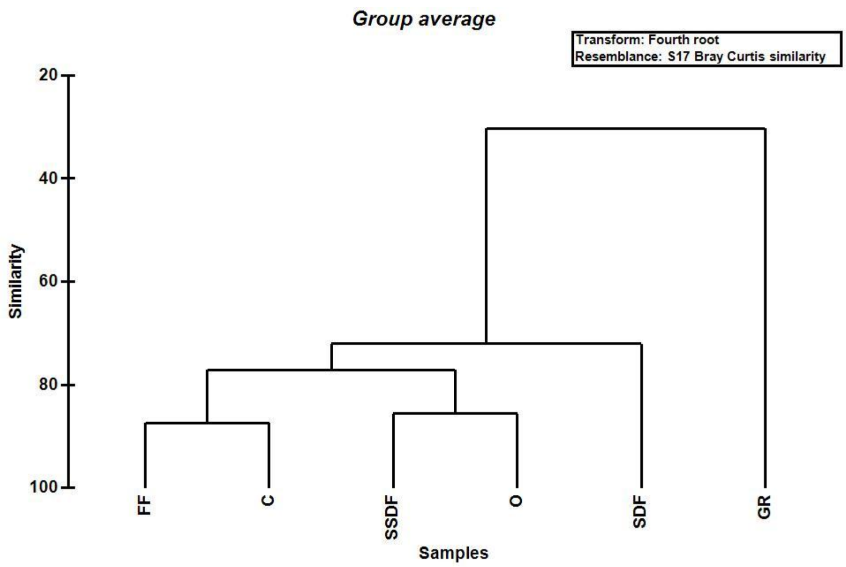

Fig. 8.19 Dendrogram showing the hierarchial clustering of the similarities of different trophic group in Vembanad estuary during 2012-2013 period

8.2.2.3 Relationships between trophic structure and environmental variables

8.2.2.4 Canonical Correspondence Analysis (CCA)

CCA analysis was used to evaluate the relationships between the trophic groups per sampling site and environmental conditions. During the first year the first two axis explained 67.8 % and 23.1 % of total variance with eigenvalues of 0.35 and 0.12 respectively. The SDF and SSDF showed a strong relationship with sandy substratum of station 9 (Fig. 8.20). During second year, the CCA analysis showed that the first and second ordination axis accounted for 65.92 % and 18.6 % of the total variance respectively. Here the O and GR showed strong affinity towards sand and temperature respectively (Fig. 8.21). Similarly FF showed close affinity towards salinity in station 8 and 9. It was also noted that the distribution of SSDF and C were related to station 1 and station 4 respectively.

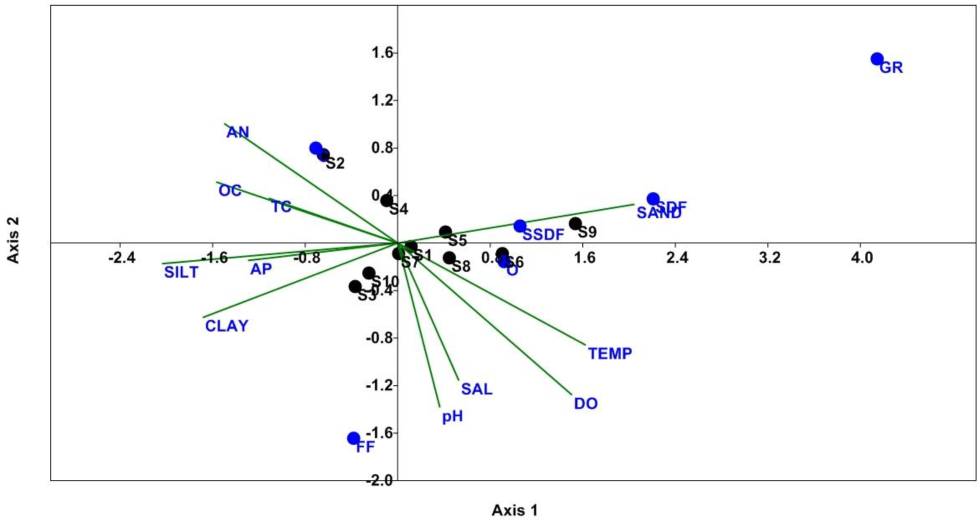

Fig. 8.20 CCA analysis of macrofaunal trophic groups and environmental parameters in Vembanad estuary during 2011 – 2012 period

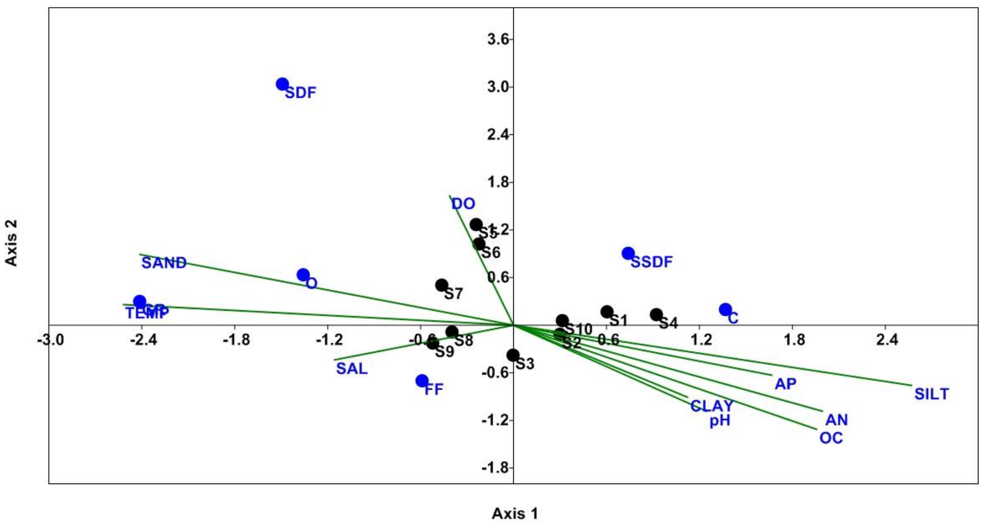

Fig. 8.21 CCA analysis of macrofaunal trophic groups and environmental parameters in Vembanad estuary during 2012 – 2013 period

8.3 Discussion

For assessing the health of benthic ecosystems, based on organic matter input either from urban effluents or from eutrophication, AMBI index has been used widely for many years (Borja et al., 2000). According to Muxika et al. (2003) AMBI and M-AMBI were important to represent both patial and temporal gradients. It doesn’t require any reference region for the study (Labrune et al., 2006) and also its analysis are not influenced by the sampling effort (Fleischer et al., 2007). Besides this, it doesn’t vary with time, because AMBI indices are also not influenced by the species abundance (Salas et al., 2004). In the present investigation AMBI value clearly indicated that the benthic environment in Vembanad estuary varied between undisturbed to moderately disturbed condition. A wide range of AMBI values denoting good to poor ecological status for the coastal and estuarine waters of India have been reported by (Sivadas et al., 2016). They found that AMBI values were high in the harbours, Mandovi estuary and coastal waters of Ratnagiri coast; moderate status in Mumbai and in Kochi and Mangalore indicating a good condition. Feebarani et al. (2016) observed that the AMBI value in Cochin estuary denoted a 'moderate' disturbance class and signified that estuary was facing stress from anthropogenic activities and natural stress during monsoon rainfall. In Fort Cochin beach AMBI value showed undisturbed (0.87) to moderately disturbed (3.5) condition (Anu Pavithran, 2015). The AMBI value in the Vellar estuary represented an undisturbed nature, whereas in the Uppanar estuary had a polluted status due to the industrial and municipal discharges (Ajmal Khan et al., 2014). The AMBI indices in the Vellar - Coleroon estuarine system, Sigamani et al. (2015) observed that stations located near to the discharge point of shrimp farms and dredging sites had a moderately disturbed condition (AMBI between 3.45 and 3.72). The ecological classification of sampling stations during the second year was contrasting with first year. During both the years, in Vembanad estuary, station 1 showed moderately disturbed condition with transition to pollution. It was noted that in the Punnamada region increase in organic matter input due to sewage discharges from tourism amenities, where intensive house boat tourism activities were rising. The leakages of oil from house boat, navigational boat and fishing operation may also enhance the petroleum hydrocarbon input into the station. Besides, urban discharges from Alappuzha town and also from the agricultural wastes and degradation of water plants increase the organic load in the region. The pollution due to increase in organic load, will lead to the proliferation of opportunistic species mainly those belonging to EG-IV and V, and in turn increase in AMBI value as 4 (Sigamani et al., 2015). It was observed that the opportunistic polychaetes (*Namalycastis indica*- EG-IV) and oligochaetes (EG-V) were more responsible for the high AMBI value. Similar condition was also observed in station 2. There are several reports that, organic pollution modified the composition of benthic fauna (Graves et al., 2004; De Paz et al., 2008). According to Cardell et al. (1999) the muddy substratum leads to anoxic situations, where the opportunistic or r-resistant species such as *Mediomastus capensis* (Capitellidae) and Spionid species were dominant. Similarly the dominance of opportunistic and/or resistant species in the organic matter and petroleum hydrocarbons enriched area with muddy substratum and anoxic condition due to effluent discharge in the coastal waters of Nandgaon, Maharashtra India were observed by Sivaraj et al. (2014), where the AMBI value was very high (average 3.2). A similar, situation in the Punnamada region, where the maximum AMBI value and annual average sediment organic carbon was observed. According to Arasaki et al. (2004), the organic enrichment and muddy substratum lead to an elevated value of AMBI. This is helped by the sedimentological and physico-chemical behavior of fine sediments to keep more organic matter than the coarser sediments. So the muddy sediments were considered as stressed habitats and several tolerant species also occurred here. The utilization of AMBI index was not restricted to organic pollution estimation alone but also, reveals the distresses caused due to the hydrocarbons, engineering works and harbour dredging etc. (Muxika et al., 2005).

Station 3, 4, 7 and 8 in the present study were in the category of slightly polluted with AMBI score value 2 and station 5, 6 and 9 were showing a moderately disturbed condition. The negative correlation of AMBI values with the percentage of coarse sand signified this aspect. The present results were in good agreement with station 7 and 8 where AMBI low with more than 70 % sandy sediment. Station 5, dominated by first and second order opportunistic species such as oligochaetes (EG- V), *Namalycastis indica* (EG- IV) and *Prionospio cirrifera* (EG- IV). The moderately disturbed station 6 represented that the benthic community health that was transitional to pollution which was dominated by the second order opportunistic species *Prionospio cirrifera* (EG- IV). This could be due to the closeness of these stations to the Thaneermukkom fish market situated on the estuarine bank, where the fish wastes directly disposed into the estuary. Similarly the dredging operation of clam fishers may cause disturbances to the bottom sediment substratum. In station 9, the dominance of first and second order opportunistic species such as *Parheteromastus tenuis* (EG-V) and *Prionospio cirrifera* (EG- IV) made the station moderately disturbed with BI value of 4. The discharges of effluents from the constructional activities in the nearby area may enhance these opportunistic species. Even though station 10 was located close to the estuarine mouth also it receives considerable amount of organic matter from the nearby shell fish processing unit, fish landing center and the sewage discharges from the urban Kochi area. The station is indicating an undisturbed condition with dominance of members of EG-I (*Diopatra neapolitana*). This was contrasting with the observations of several researchers. Large amount of organic matter was observed in the stations located close to the estuarine mouth area from the aquaculture farms, fish landing center and other various sewage disposal systems which cause the proliferation of opportunistic species (Dauer, 1993; Muniz, 2003; Bouchet and Sauriau, 2008). At the same time in station 10, the sensitive species (EG- I) was contributing more than 65 % of abundance and the percentage of opportunistic species of ecological groups IV and V were less than 10 %. This type of observation was found in the Bahrekan coast in Persian Gulf (Shokat et al., 2010). The benthic organisms respond similar to various impacts, generally they transformed from sensitive groups (I and II) to lower succession stages(the opportunistic IV and V groups) and also the percentage of each EG in the community related to the intensity, duration and the distance of source of impact (Muxika et al., 2005).

In contrast to AMBI analysis, the M-AMBI denoted that station 3 to 9 were in good condition. This type of contrasting results of AMBI and MAMBI was observed in same stations of Vellar–Coleroon estuarine system (Sigamani et al., 2015). They noted that the absence of comparison of reference condition and also the spatial variation of macrofaunal community structure and the type of habitat was responsible for this. The M-AMBI distinguished the benthic habitat health from the degraded conditions from the unpolluted areas where H' and AMBI could not distinguish (Xianxiang et al., 2016). In the Huanghe estuary and its adjacent areas the ecological quality status evaluated by these three indices showed obvious differences and M-AMBI was more effective. Ajmal Khan et al. (2014) noted that the M-AMBI value of Vellar estuary showed a high/good (0.9 ± 0.06) ecological quality, whereas in the Uppanar estuary, M-AMBI value represented a poor/bad condition (0.22 ± 0.04). Feebarani et al. (2016) observed that the M-AMBI value in Cochin estuary did not show any ‘high’ ecological status and overall represented a moderate to poor ecological status. Sivaraj et al. (2014) noted that in the effluent affected region of Nandgaon Coastal waters, Maharashtra India, the M-AMBI value denoted that the ecological quality was varied between moderate to high status (0.51 to 0.96). Earlier it was stated that AMBI and M-AMBI indices were able to discriminate different anthropogenic distresses such as anoxia and hypoxia, eutrophication, nutrient loading, oil and industrial pollution and aquaculture effluent impacts etc. (Borja et al., 2009; Pinto et al., 2009).

The M-AMBI and BENTIX index denoted a poor ecological condition in station 2 and the ecological condition of station 10 represented an undisturbed (AMBI) or high status (M-AMBI and BENTIX) during first year. During second year in station 1, the AMBI, M-AMBI and BENTIX index value showed a moderately disturbed nature. According to Borja et al. (2000) benthic fauna react to progress in environment quality by three means such as increase in abundance, diversity and variation of dominant species from tolerant to sensitive to pollution. The increasing organic matter input due to sewage discharges from tourism amenities; leakages of oil from house boat, navigational boat and fishing operation; urban discharges from Alappuzha town and also from the agricultural wastes and degradation of water plants was increases the organic load. This was evident in the proliferation of opportunistic polychaetes (*Namalycastis indica)* and oligochaetes in the southern zone. Due to the operation of TMB, the flow pattern of water in the southern most stations may be altered and it possibly changes the benthic assemblage structure. The absence of sensitive benthic invertebrate taxa and the dominance of tolerant depositional groups such as oligochaetes in Fox River impoundments in Illinois, USA are typical responses of aquatic invertebrates to impoundment in temperate rivers (Stanley et al., 2002). The BOPA index was broadly used for assessing the impact of oil spill on benthic communities in coastal waters (Dauvin and Ruellet, 2007; Joydas et al., 2012). It was based on the idea that an oil spill usually caused high mortalities of amphipods subsequently by the growth and development of opportunistic polychaetes (Sukumaran et al., 2014). By observing the impact of '*Chitra'* oil spill on the tidal pool macrobenthic communities of rocky shore of Mumbai, Sukumaran et al. (2014) found that oil spill did not have any negative impact on the macrobenthic biota and also the low BOPA values in the area indicated that the environmental quality status was good with few opportunistic species. They implicit that, the rocky shore along with sandy sediment, warm temperatures, monsoonal wave action, low amount of spilled oil (800 t) was responsible for the minimum damage to the macrobenthic communities. During the present investigation opportunistic polychaetes *Namalycastis indica* (Nereidae) and *Prionospio cirrifera* (Spionidae) of organically enriched sediments may be responsible for the bad ecological condition of station 2 and 5. The Vembanad estuary receives a wide range of wastes, such as sewage from the continuous backwater tourism activities mainly by the houseboat tourism including oil wastes and other wastes from repairing works of houseboat and navigational boat, wastes from resorts located on the estuarine bank, municipal, agricultural and industrial wastes that were subjected to play a significant role. In the pollution impacted southern islands of Singapore, the opportunistic polychaetes families of Eunicidae, Flabelligeridae, Nereidae and Glyceridae were dominated (Chou et al., 2004). In the inshore area of east coast of India, Ganesh et al. (2014) observed the intensity of organic pollution near the outfall point of sewage treatment plant (STP) by the BOPA index. Increased abundance and low biomass of Cirratulidae and Spionidae in the STP outfall point denoted a typical of sewage pollution where the BOPA index revealed poor–bad ecological conditions. At the same time an unaffected BOPA condition (high to good) was observed in the stations far from the STP out fall point. So the dependability of BOPA index was mainly for the preference of organic enriched sediments by the opportunistic polychaetes and the sensitivity of amphipods to pollution than any other marine species (Cesar et al., 2004; Riba et al., 2004). The major advantage of BOPA index was that the requirement of taxonomic expertise was very limited for this analysis. At the same time, it was blamed that the pollution sensitivity of species from the same taxonomic group had different response (Afli et al., 2008; Andrade and Renaud, 2011). In the tourism active Fort Cochin beach area, Anu Pavithran (2015) noted the BOPA index as high ecological status (0.01) and moderate ecological status (0.15). According to Sivadas et al. (2016) BOPA was not a suitable indicator for assessing the ecological condition of the Indian coast mainly because of the naturally low abundance or absence of amphipods in the coastal regions of India. Such type of concurrence in the applicability of BOPA was observed in the Canary Islands and Tunisian Coast (Afli et al., 2008; Riera and de-la-Ossa-Carretero, 2014). Riera and de-la-Ossa-Carretero (2014), pointed out that BOPA was more suitable in the heavily impacted regions such as harbours than the other anthropogenically disturbed regions. This was contrasting with the present investigation during first year in station 2. Here the BOPA denoted as a bad ecological condition, at the same time AMBI and

M-AMBI denoted as moderately polluted and poor ecological status respectively, whereas during second year the BOPA value of station 5 indicated as bad status but the AMBI and M-AMBI value represented as slightly polluted and good condition respectively. During the study period, a significant positive correlation was observed between AMBI and BOPA and also between M-AMBI and BENTIX. Such type of correlation was observed in the studies of impact of oil rigs in the northern Adriatic Sea by Spagnolo et al. (2014). By evaluating the ecological condition of Mediterranean coastal lagoons by using various benthic classification indices, Reizopoulou et al. (2014) found that biotic indices such as AMBI, M-AMBI, BENTIX, and BOPA were not effectively applicable due to the dominance of tolerant and opportunistic species and also due to the correlation between species diversity and natural stress. By comparing the five benthic indicators (H', AMBI, M-AMBI, BENTIX, and BOPA) for the evaluation of macrofaunalstress levels Spagnolo et al. (2014) noted some conflicting observations. They found that BENTIX, H' and BOPA indices were useful to evaluate stress levels better than the AMBI and M-AMBI indices.

Distribution of major trophic groups of benthic fauna related to the accessibility of their particular food resources, which in turn associated with the bottom topography and also the nutrient status (Peres, 1982). Carnivores and filter feeders become the dominant trophic group observed during first and second year in Vembanad estuarine system. Apart from Grazers, other feeding groups such as FF, SDF, SSDF, C and O were observed in all stations except station 1. So the presence of different feeding types in each station showed the availability of different food sources. The organic matter from the water column was utilized by filter feeders, though deposit feeders consumed the sediment detritus. The organic matter gets available for SSD of the bottom sediments by the influence of bioturbating macrofauna and carnivores helped to recycle the nutrients from the deposit feeders to the water column (Sivadas et al., 2013). According to Jayaraj et al. (2008) the togetherness of different feeding types in a region denoting the ability of organisms to utilizing different food resources in a particular niche and also the interaction of different species support the use of different feeding niches in a specific habitat. He also noted the dominance of deposit feeders in shallow depth of continental margin of southwest coast of India denoting its preference to the fine-grained sediment. So different trophic structure of macrofaunal community in Vembanad estuarine system easily utilizes the organic matter which is transformed into benthic biomass and become the major food for higher trophic level organisms. Sivadas et al. (2013) point out that different functional groups in an ecosystem resulted in more efficient exploitation of organic matter. In the Tagus estuary, seven distinct feeding categories were observed among the macrofaunal communities, in which surface deposit feeders were dominant (52%) (Gaudencio and Cabral, 2007). After analyzing the trophic structure of soft bottom macrobenthos in the inlet of Rıa de Vigo, Spain Cacabelos et al. (2009) observed the dominance of herbivores in the inner part of the inlet and surface-deposit feeders were dominated at the oceanic - influenced area. They observed that sediment parameters such as organic matter content, silt-clay fractions interrelated with the trophic composition, abundance and distribution of benthic macrofauna. By observing the soft bottom macrobenthic communities of the ports of Mumbai, India Mandal and Harkantra, (2013) noted the dominance of deposit-feeders, representing the stressed situation of the harbour region. In all the seasons, surface or sub-surface deposit feeding polychaete species, *C. coasta* and *P. pinnata* were recorded throughout the period and showed preference towards silty sediment (Mandal and Harkantra, 2013). In the present investigation these species showed similar type of preference towards the sediment substratum. These species were more abundant in station 10, where the substratum was clayey silt. Due to the sediment destabilizing activity of deposit feeders, the filtration mechanisms of suspension feeders were changed and it detrimentally affects its abundance (Rhoads and Young, 1970). In the sandy substratum, suspension feeders were dominant and at the same time surface deposit-feeders predominate in muddy bottoms (Maurer et al., 1979). The feeding type patterns of macrobenthos in the Northern Adriatic Sea found that filter feeders were chosen for silty-clayey substratum having high organic content, whereas deposit feeders selected sandy substratum with poor organic content (Mancinelli et al., 1998).

During first year, the carnivorous polychaete, *Namalycastis indica* and *Diopatra neapolitana* were more abundant over other macrofaunal species in Vembanad estuarine system. The increased abundance of C in station 1 and 2 was mainly due to the dominance of *Namalycastis indica,* whereas in station 7 and 8 was mainly attributed by the carnivorous isopods *Apanthura sandalensis* and *Xenanthura orientalis*. In station 10 polychaete abundance was dominated by *Diopatra neapolitana. D. neapolitana* feeds on algae, sponges, bryozoans and crustaceans (Fauchald and Jumars, 1979). During first year in station 6 and 9 surface deposit feeding polychaete species *Prionospio cirrifera* was predominant. According to Dolbeth et al. (2009), abundance of carnivorous macrobenthic communities was associated with environmental and anthropogenic impacts. Pearson and Rosenberg (1978) observed that the increasing abundance of carnivores over the filter-feeding bottom invertebrates species were characteristic of perturbed areas exposed to strong organic in puts. Carnivores are able to transport the nutrients preserved in the tissues of detritivores into the water column and available for primary producers (Ngai and Srivastava, 2006). According to Ronn et al. (1988) carnivores act as a major factor in structuring of brackish water systems, mainly for its predatory behavior on different bottom fauna species. At the same time during second year, the FF *Villorita cyprinoids* become more abundant. The black clam, *Villorita cyprinoids*, is an important fishery of the Vembanad estuary, where dense and vast mussel-beds were common. During second year, the filter feeding *Villorita cyprinoids* and amphipod, *Gammarus* sp. were more pronounced in station 7 to 9. The benthic bivalve filter feeders were established in shallow waters with suitable substrate and sufficient phytoplankton supply. So it can be dominant in water due to its short food chain, long life spans with high accumulated biomass to improve the ecosystem over a long period of time with various environmental cycles (Dame, 1996). It also plays an important role in controlling the eutrophication and so maintaining the water quality. Generally FF is involved for channelizing the organic material between the plankton and the benthos (benthic – pelagic coupling) (Holt et al., 1998). Continuous phytoplankton production and organic matter dynamics supported the filter feeding amphipod *Amplesica* sp., and bivalves in the Kalbadevi Bay, west coast of India (Sivadas et al.,2013). According to Dauwe et al. (1998) the fresh detritus matter in the water column was filtered by filter feeders and the sedimented matter was utilized by deposit feeders. Occurrence of filter feeders generally increased the higher trophic status of the lake (Grimas, 1969) and the filtering activity of bivalves generally improves water quality (Wilkinson et al., 1996).

Several authors reported the feeding guild distribution pattern in estuaries (Ysebaert et al., 1998, Garcia-Arberas and Rallo, 2002; Gaudencio and Cabral, 2007). Generally FF and C species were dominated in the outer and middle regions close to the estuary mouth, and DF and O were more in the inner habitats (Mancinelli et al., 1998). The feeding guild composition of polychaetes of southeast coast of India, Manokaran et al. (2013) noted that suspension feeders and carnivores were dominant in sandy and silty sediment respectively. Among the macrobenthic communities of Kalbadevi Bay, west coast of India Sivadas et al. (2013) observed the dominance of subsurface and surface deposit feeders and it showed a temporal variation correlated to the environmental parameters. In the present investigation, the distributional pattern of the feeding guilds along the Vembanad estuarine system indicates that overall among the feeding guilds C and FF were distributed in majority of the stations. The sediment textural analysis of Vembanad estuarine system revealed that both the years, the substratum was silty sand in nature. Carnivores and filter feeding macrobenthos were dominant in sandy sediments (Fonseca et al., 2001). Several authors observed the dominance of suspension feeders and carnivores in the coarse-sandy sediments of low organic content with high energy environments (Diaz and Rosenberg, 1995; Gaston et al., 1998; Rakocinski et al., 2000). When carnivores occurred throughout the coastal shelf that indicates that carbon dynamics were capable of maintaining the presence of the highest trophic levels (Wijsman et al., 1999).

The trophic structure of an uncontaminated area related to the current situation of the environment because the communities chiefly depend on the available food and space. At the same time in the polluted regions such type of relationship was not seen because the important species characteristics such as resistance, tolerance and opportunism have important role (Afli et al., 2008). During both the years the maximum percentage of SSDF were noted in station 1.The closeness of SSDF in station 1 in the Vembanad estuarine system may be related to the increasing organic carbon content (av. 7.2 %) in the sediment. The high faunal abundance of SSDF, oligochaetes possibly exploit the organic rich detritus settled on sediments. The SSDF feeding mode considered as the excellent approach for utilizing organic resources buried in the sediment column (Stora et al., 1999). Increasing organic carbon content in the sediment makes more labile organic matter and bacterial biomass for the SSDF. Similar type of observation was found in the shallow subtidal area off Kunsan, west coast of Korea, where SSDF were associated with muddy sediment with high organic matter (Choi and

Koh, 1992). According to Beukema, (1991) in the eutrophicated areas deposit feeders were more abundant. In the present investigation, SSDF were more abundant in areas with higher organic content (station 1), that in agreement with their feeding requirements. According to Gaston (1998), SSDF were dominant in the metal contaminated area. Similar situation existed in station 1, where the sediment was heavily polluted with Zn, Cu and Pb. According to Byren et al. (2002), fresh settled organic matter was utilized by SDF, whereas old organic matter was consumed by SSDF. The CCA analysis of the trophic structure of macrobenthos in the Tagus estuary showed that FF had strong affinity towards sandy areas and also found that SSDF were related to sediment of high mud content (Gaudencio and Cabral, 2007). Generally deposit feeders were more abundant in muddy substratum whereas suspension feeders in sediment with low content of fine fractions (Garcia-Arberas and Rallo, 2002). It was also noted that, the tidal currents in the southern sector of TMB had limited influence on the trophic structure of macrobenthos, because reduced tidal currents characterize the dominance of surface and subsurface deposit feeders as compared to suspension feeders and carnivores. According to Pearson and Rosenberg (1978), the increased accessibility of particulate organic matter in the sandy areas supported more FF. Greater proportions of carnivores and filter feeders in the Vembanad estuary associated with the occurrence of sandy substratum. Vaughn and Tylor (1999) noted that in the organically enriched environments several filter feeding bivalves grow well. The occurrence of suspended particulate matter in the upper Fitzroy river estuary, Queensland had a major role on controlling the density and distribution of FF bivalve (Currie and Small, 2005). Similarly Carrasco and Carbajal (1998) pointed out that among the various feeding groups, communities in the sandy sediments formed a balanced proportion due to the presence of diverse microhabitats in coarse sediments and also due to the occurrence of more food particles in the sediment interspaces (Muniz and Pires, 1999). In the high energy environments with coarse sediments generally support suspension feeders, while low energy condition supports high amount of fine particles (Sanders, 1958). The variations in the macrobenthic community and different trophic groups reflected different food resources with a complex food web in Vembanad estuarine system that was greatly getting influenced by drastic environmental and habitat modification.

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Chapter 9

SUMMARY AND CONCLUSION

Wetlands are unique, productive ecosystems where terrestrial and aquatic habitats meet, sustaining many natural cycles and also supporting wide range of biodiversity entities. Vembanad wetland ecosystem, the largest Ramsar site on the south west coast of India, is a massive and vibrant coastal ecosystem contributing to over 50 % of the total area of backwaters in the State of Kerala. The wetland complex consist of Vembanad estuary (backwater) bordered by Kuttanad on south - the 'rice bowl of Kerala' situated 1.5 to 2 m below mean sea level, and Kol lands on north, which are interlinked by river estuaries and mangrove marshes, well interconnected by a complex network of natural and manmade channels spreading over 1512.5km^2.

However over the past several years, the Vembanad estuary is passing through a phase of rapid ecological modifications mainly due to the unrestricted human interference for heterogeneous purpose. The commissioning of Thanneermukkom barrage (TMB) (regulator), across the backwater system in 1976, to prevent saline water intrusion, has adversely affected the ecological functions of the water body. Annual closure of barrage during December to April for paddy farming has transformed the estuary into two distinct zones, a fresh water dominant southern zone, and a predominately brackish water zone on the northern side. Scientific information on the ecosystem based analysis of the environment coupled with the benthic status on a comprehensive basis is severely lacking from the estuarine system after the commissioning of the barrage. Apart from this, sewage pollution with increasing faecal coliforms, in conjunction with intense tourism related activities and waste dumping, oil and other contaminants has seriously afflicted the estuary. So in this prevailing situation, understanding the trophic character and the organismal dynamics is all the more relevant. Benthic fauna form an important link in the transfer, mineralization and re mineralization of particulate and dissolved organic matter in different levels of trophic food web maintaining the energy balance and carbon structuring of the aquatic system. But, the ecological imbalances, as noticed in Vembanad estuary can inflict long term damage on the ecology of the benthic fauna and associated communities possibly having its adverse repercussions on the trophic stability possibly even affecting the climate regime of the ecosystem. So the present study tried to understand the benthic ecology and community dynamics of the macro and meiobenthic invertebrates, in this transforming trophic environment of Vembanad estuary. This study also made an effort to understand the benthic ecology of the estuary after the commissioning of the TMB.

Chapter 1- elaborates the general description of wetlands, coastal estuaries, degradation of coastal wetlands and structural and functional characteristics of benthic fauna. The chapter also highlights the significance and objectives of the study.

Chapter 2- provides the details of study area, study stations, field sampling, methods of analysis and statistical techniques employed.

Chapter 3- gives the details of physico-chemical factors structuring the trophic environment of Vembanad estuary. During both the years, the maximum rainfall was recorded during the MN period and minimum during PRM period. According to the present study, Vembanad estuary showed drastic decrease in depth especially between Varanadu to Perumbalam region. Appreciable decrease in the depth of the estuary from an average of 4.4 m during 1980 to 4.1 m during the current study period. The pH was neutral to slightly alkaline with maximum and minimum value observed in stations 10 and 1. Spatial distribution of conductivity, TDS, turbidity and alkalinity values showed a progressively increasing trend towards the northern high saline stations. The average salinity pattern in the southern part of estuary showed a limnetic to oligohaline condition. The northern stations exhibited a limnetic to oligohaline during MN to mesohaline to polyhaline condition during PRM season. While comparing the major ecological parameters from pre barrage and post barrage phase of Vembanad estuary, salinity was found to be the major factor that showed a drastic alteration. During 2011-2012 and 2012-2013 period, DO ranged between 5.69 to 9.76 mg $L^{-1}$and 3.2 to 11.2 mg L^{-1} respectively. The annual average of SiO_4-Si observed was 19.32 ± 1.52 µmol L^{-1} and 20.91± 1.49 µmol L^{-1} for first and second year respectively. Both the years the average phosphate concentration was higher in northern zone than southern zone. Time scale study noted that increased human induced sources such as partially treated and untreated sewage, run off from agriculture sites of Kuttanad mainly the fertilizers and other discharges all trigger the phosphate load in Vembanad estuary. The stoichiometric study revealed that the mean N: P ratio was below the normal Redfield ratio (16:1), so a nitrogen limiting condition existed in the entire estuarine system. TRIX analysis indicated an impacted water quality condition in the estuary leading to an accelerated eutrophication in the system. When comparing the TRIX value in the opened and closed period of barrage, the southern zone showed little higher TRIX value in the open phase (6.7) than the closure period (6.4). Due to the variations of hydrological parameters, the southern and northern stations of Vembanad estuary were grouped distinctly and it was clearly seen in the cluster analysis. During both the years in PCA, high loadings were obtained for pH, alkalinity, conductivity, salinity, TDS, turbidity, PO_4-P and NO_2-N. Salinity and fresh water inputs were the master factors influencing the physico-chemical variations and processes in the estuary during both the years of the study.

Chapter 4- gives the sediment characteristics of the Vembanad estuarine system. Sediment temperature for the 2011-2012 period ranged from 23 to 32 ^{0}C with an average of 28.6 ^{0}C.When comparing the south and north zone of TMB, sediment pH was minimum in southern zone than the northern zone. This was possibly due to the decreased saline water flushing in the southern zone, where the tidal inflow completely cut-off during the closure period of barrage. The redox potential (oxidation reduction) of the sediments remained reduced in the entire estuarine region indicating a stressed condition due to various factors. The southern zone was showing more reduced condition than northern zone. This was mainly due to the accumulation of large amount of decomposable organic material in this zone. During 2011-2012 period sediment organic carbon varied from 0.03 to 9.4 % with an average value of 2.69 ± 2.23 %. The ANOVA of organic carbon showed that variations between stations and seasons were significant ($P \leq 0.01$). During 2012 – 2013, the organic carbon varied from 0.31 to 9.8 % (av. 2.97 ± 2.25 %). The results of organic carbon and organic matter revealed that, over the three decades period, a threefold increase in sediment organic carbon and organic matter was observed. The organic matter enrichment was more severe in the southern most region especially station 1, Punnamada. The annual average of available nitrogen was 0.03 ± 0.03 % and 0.05 ± 0.05 % for first and second year respectively. The highest spatial average of available phosphorus was observed in station 1 and station 10. The increased concentration of available phosphorus and available nitrogen in the estuarine sediment of the southern zone indicated the indiscriminate and unscientific application of fertilizers, increasing house boat tourism activities, sewage inputs from Alappuzha town etc. Closure of TMB during the summer months (December to April) lead to a significant reduction in tidal flushing lead to a stagnation of the water mass in the southern zone. This suggested an accumulation of nutrients in the sediment of stagnant portion of the estuary over the years. During 2011-2012 period, sodium concentration in sediment showed a wide variation from 1 to 55200 mg/Kg and during second year it varied from 3 to 32500 mg/Kg. The sediment potassium varied from 165 to 3976 mg/Kg (av. 706.58 ± 820.42 mg/Kg) and 2 to 2984 mg/Kg (av. 400.69 ± 616.38 mg/Kg) during 2011-2012 and 2012–2013 period respectively. In both the years the highest average value of sodium, potassium and calcium were observed in station 10, where the tidal influence from Arabian Sea is maximum. Sediment texture of the entire estuary revealed that sand fraction was dominant over silt and clay. But its percentage contribution was varying in each station. Comparing the southern and northern zone sand fraction was predominant in northern zone whereas it was silt in the southern zone. This also revealed the impact of TMB on the spatial distribution of sediment particles, where the terrigenous silt from the rivers may accumulate in the southern zone over the years. Heavy metal analysis revealed that overall the estuary was moderately polluted with, Zn, Cu, Cd and Pb. Station 1 was heavily polluted with Zn, Cu and Pb in all the seasons except during PM, where Zn was moderately polluted. The PCA analysis also supports that input of silt fraction, organic carbon, organic matter, available nitrogen, available phosphorus from rivers coupled with the discharge of organic wastes from different industrial, urban, agricultural and tourisms sectors that leads to the settlement of organic matter in the estuarine sediments of southern region. But the association and inputs of sodium, potassium, calcium, pH and clay fraction accomplished their strong association towards seawater and tides. Therefore it is established that the TMB strongly influenced the diverging sedimentological characteristics of southern and northern zone of the Vembanad estuarine system.

Chapter 5 – deals with the ecology and distribution of soft bottom benthic communities of Vembanad estuary. During 2011-2012 period, the meiofauna taxa appeared to be well represented with a total of 12 groups. It includes foraminiferans, kinorhynchs, nematodes, oligochaetes, polychaetes, chironomids, ostracods, harpacticoid copepods, amphipods, tanaids, isopods and bivalves. In both the years the spatial mean abundance of meiofauna was maximum in station 5, (1617/10 cm^2) for 2011-2012 and (3819/10 cm^2) for 2012-2013 period. Seasonal analysis of meiofauna showed different trends in both years having highest abundance of meiofauna in MN (41.9 %) in 2011-2012 and PRM (67.4 %) in 2012-2013 period respectively. In all the seasons, nematodes were the dominant group. Its maximum percentage was noted in MN (75.2 %), followed by PRM (50.4%) and PM (47.4 %). During PRM foraminiferans formed the second dominant group (9.4 %) followed by bivalves (8.8 %) and harpacticoid copepods (8.6 %). Whereas during MN and PM harpacticoid copepods formed the second dominant group, which contributed 11.4 % and 20.3 % respectively. Similar to 2011-2012 period, nematodes were the dominant group in all the seasons and maximum in MN (66.6 %), followed by PRM (51.8 %) and PM (42.8 %) in 2012-2013 period. SIMPER analysis revealed that nematodes and harpacticoid copepods were responsible for the dissimilarity between two sectors and its highest average abundance observed in the northern zone of TMB. The oligohaline condition in the southern zone probably leads to decrease in abundance of nematodes and harpacticoid copepods in the southern zone. BIOENV analysis also revealed that environmental parameters had a major role in the distribution and abundance of meiofauna in the Vembanad estuarine system.

A total of 18 faunal taxa were encountered representing the macrofaunal community structure of Vembanad estuarine system. The macrofauna were represented by 5 major groups, namely, oligochaetes, Polychaetes, amphipods, tanaids and bivalves. During 2011-2012 period, polychaetes formed the most dominant group (av. 16580 ind./m^2, 59.8 %) followed by bivalves (4328 ind./m^2, 15.6 %) isopods (2262 ind./m^2, 8.2 %), amphipods (1977 ind./m^2, 7.1 %) and oligochaetes (706 ind./m^2, 2.5 %) and spatially maximum average abundance was observed in station 10 (5317 ind./m^2), followed by station 2 (2223 ind./m^2) and station 4 (1959 ind./m^2). During 2012-2013 period, bivalves formed the most abundant group (20059 ind./m^2, 36.09 %) followed by polychaetes (13684 ind./m^2, 24.62 %), amphipods (10883 ind./m^2, 19.58 %) and isopods (7134 ind./m^2, 12.84 %) and spatially maximum average abundance was observed in station 9 (8964 ind./m^2) followed by station 10 (6376 ind./m^2) and station 2 (4138 ind./m^2). Seasonal analysis of macrofauna showed different trends in both years having highest abundance in PRM (40.9 %) in 2011-2012 and in PM (37.1 %) in 2012-2013 respectively. The study found that there was a clear variation in the distribution of macrofaunal groups over the years. In the present investigation, increasing abundance of oligochaetes was noted in the southern zone, where the organically enriched sediments probably were responsible for the oligochaetes abundance. This also indicated the fresh water dominance in the ecosystem. Comparing to previous studies, the abundance of amphipod was decreased over the years, especially in the southern zone. During both the years, from the macrofaunal biomass, bivalves contributed the major share followed by the polychaetes. Abundance biomass comparison (ABC) curve indicated that half of the stations such as station 1, 2, 3, 9 and 10 were under environmental stress, but the other stations were in good condition. Based on the Shannon diversity (1.6) and Margalef richness (1.1) index macrofaunal community indicated an unhealthy condition in Vembanad estuary. During the study period, a clear spatial difference was observed in benthic macrofaunal abundance in the south and north of TMB. Based on the SIMPER analysis, the macrofauna in the southern and northern zone of TMB was dissimilar by

33.55% for first year and 35.8% for the second year. The average abundance of decapods, oligochaetes and bivalves made a major dissimilarity between southern and northern sector of TMB. During first year meio and macrofaunal diversity, richness and evenness indices were higher in northern zone than southern zone of TMB. Overall the BIOENV analysis revealed that, a combination of parameters such as bottom water pH, salinity, sediment potassium content, organic carbon, available nitrogen and sand fraction determined the abundance and distribution of macrofauna in Vembanad estuarine system. It has been confirmed that salinity, bottom water pH, silt and clay showed positive correlation with the spatial distribution of macrofauna during first year and during second year organic carbon, available nitrogen, silt fraction and bottom water pH depicting a strong positive correlation, confirming to the CCA analysis. Therefore, it is established that TMB was strongly influencing the physico-chemical parameters in the estuary which in turn had a strong bearing on the distribution, abundance and diversity patterns of both meio and macrofauna.

Chapter 6 – deals with the community structure and ecology of polychaetes in the Vembanad estuary. A total of 19 species of polychaetes belonging to 13 families and 19 genera were observed during 2011-2012 period, in which species such as *Namalycastis indica* (34.22 %), followed by *Diopatra neapolitana* (27.98 %), *Prionospio cirrifera* (17.15 %), *Parheteromastus tenuis* (7.18 %) and *Dendronereis aestuarina* (6.03 %) formed the most abundant species. At the same time during 2012-2013 period, only 14 species of polychaetes belonging to 11 families and 13 genera were observed, where *Diopatra neapolitana* (31.3 %), followed by *Namalycastis indica* (25.41 %), *Prionospio cirrifera* (22.58 %), *Dendronereis aestuarina* (8.02 %) and *Parheteromastus tenuis* (7.6 %) formed the most abundant. During first year the maximum polychaete species density was observed during PRM (74.05 %), followed by PM (20.7 %) and MN (5.25 %). In contrast to first year, the second year, had the maximum seasonal density of polychaetes during PM (45.83 %) followed by PRM (31.86 %) and MN (22.31 %). During both the years the minimum density was observed during MN. Higher rainfall and riverine runoff during MN season makes estuarine water becomes fresh may greatly impacted the existence polychaete species. Time scale study found that since 1981, there was a drastic reduction in the polychaete diversity. Earlier Pillai (1977) reported 32 species of polychaetes in the northern part of Vembanad estuary, now only 19 species were recorded. During 1996-1997 period there were 10 polychaete species were recorded in the southern zone, where it reduced to seven species. Relatively lower salinity pattern and higher organic carbon content in the southern zone resulted in a stressed condition, supporting only seven species of polychaetes in which *N. indica* were the governing species. Changes in the species distribution indicate that the polychaete fauna has been impacted with various stress especially because of various interventions in the aquatic system. Based on the diversity status of polychaete community structure, it was noted that Vembanad estuary was in an unhealthy condition. Dominance of pollution tolerant species such as *N. indica* and *D. aestuarina* in the southern most stations indicating that the area was heavily polluted. Unregulated sewage discharges from houseboats, resorts and municipal wastes; exhaust from motor boats, fuel leakage, agricultural runoff etc. add the hydrocarbon and other organic pollutants to the southern part of Vembanad estuary may trigger the pollution load. Similarly occurrence of organic pollution indicators such as *Prionospio cirrifera*, *Prionospio pinnata*, *Cossura coasta*, *Glycera alba*, *Scoletoma impatiens*, *Sigambra parva*, *Micronephthys oligobranchia* and *Diopatra neapolitana* in station 10 suggested that the sampled area is negatively affected with organic enrichment. Higher concentration of heavy metals such as Zn, Cu, Cd and Pb in the sediments indicates that most of the polychaete species were relatively resistant to these metals. The ANOSIM analysis showed that the spatial variation of polychaete species on the south and north of TMB were significant. The variation in the average abundance of polychaete species such as *N. indica, P. cirrifera,* and *P. tenuis* were responsible for the maximum dissimilarity between south and north of TMB. The average abundance of *N. indica* was maximum in the southern sector whereas species such as *P. cirrifera,* and *P. tenuis* were more abundant in the northern sector. The organic matter enrichment and low saline condition in the southern zone relate well with the distribution of *N. indica* in stations 1 to 4 and the sandy substratum in the northern stations 8 and 9 were preferred by *P. cirrifera,* and *P. tenuis.* Pearson correlation analysis showed that *N. indica* showed significant ($P \leq 0.01$) positive correlation with organic carbon (r = 0.49), available nitrogen (r = 0.6) and negative correlation with salinity (r = - 0.3). During the closure period of TMB, the abundance of polychaete species *P. tenuis* and *Mediomastus* sp. were increased in the southern zone. The tolerant species of spionids and capitellids were indicators of potential degradation of aquatic system by human activities. During the closure period of barrage, stagnant nature of water column and absence of tidal flushing all enhances the settlement of dead and decaying organic matter in the sediment that may boost the prevalence of tolerant species. It was also observed that majority of the polychaete species observed during the study period were euryhaline, capable of tolerating a wide range of salinity. Overall BIO-ENV analysis revealed a strong correlation among the environmental parameters and polychaetes. From the parameters a combination of bottom water pH, water temperature, dissolved oxygen, salinity, alkalinity, sediment temperature, sand, silt and clay fraction were most responsible for structuring the polychaete assemblages in Vembanad estuary. At the same time, salinity formed a single parameter that showed a strong correlation (ρ = 0.883) with polychaete community structure. Canonical correspondence analysis also confirmed that parameters such as water temperature, TDS, conductivity, salinity, dissolved oxygen, sediment temperature, sediment pH, Eh, moisture content, organic carbon, available nitrogen, sodium, silt, and sand fraction have influenced the community structure of polychaete in the Vembanad estuary.

Chapter 7 -deals with the community structure and ecology of peracarids in the Vembanad estuary. Among the crustacean, insects, ostracods, harpacticoid copepods, amphipods, tanaids decapods, mysids, cumaceans and isopods were the most abundant groups. During first year, isopods accounted for 42.11% of the total crustacean fauna followed by amphipods (36.79 %), tanaids (12.18 %), insects (4.22 %) and decapods (2.11 %). In contrast to first year, in the second year amphipods were dominant (55.53 %) followed by isopods (36.39%), tanaids (5.06 %) and insects (2.34 %). Among the peracarid crustacean species, from tanaids; *Ctenapseudes chilkensis*, from isopods; *Cirolana fluviatilis*, *Apanthura sandalensis*, *Xenanthura orientalis*; from amphipods; *Eriopisa chilkensis*, *Eriopisa* sp., *Cheiriphotis geniculata*, *Gammarus* sp., were identified. Time scale study found that, in Vembanad estuary the diversity of amphipod species were varied over the years. Earlier seven species of amphipods were identified from the northern part of Vembanad estuary, now only 4 species were recorded. Previous comparative studies noted that in Cochin backwater, *Grandidierella megnae* and *Corophium triaenonyx* were the dominant amphipod species but now *C. geniculata* was more abundant especially in the southern zone. The southern stations was characterized with oligohaline zone, having higher organic carbon in the sediment, along with thick deposits of decaying water hyacinth providing a shelter and also acting as a major source of food for *C. geniculata*. The early reported (1994-1996 period) abundant species of amphipods, *Caprillid* sp., *Parhyale hawaensis* and *Quadrivisio bengalensis* in the southern zone of Vembanad estuary were not encountered in the present investigation. Among the peracarid crustaceans, the tanaid species, *C. chilkensis* were present in most of the stations except station 2 and 3 during first year and all the stations during second year. In both the years the average maximum density, of *A. sandalensis* were observed in station 4 during first year (265 ind./m2) and second year (738 ind./m2) respectively. The ANOVA results showed that the spatial variation of *A. sandalensis* was significant in both the years ($P < 0.01$). During first year, *X. orientalis* was observed in all stations, except station 10 and its maximum density was observed in station 4 (602 ind./m2) and station 7 (518 ind./m2). During second year *X. orientalis* was not observed in station 1 and 10, with its maximum spatial abundance that was noted in station 4 (2408 ind./m2) and station 9 (1090 ind./m2). Both the years, the isopod species *C. fluviatilis* was observed only in station 10, where it preferred the specific clayey-silt substratum. Pearson's correlation showed that the distribution of *C. fluviatilis* showed significant ($P \leq 0.01$) positive correlation with clay ($r^2 = 0.67$). By observing the overall seasonal variation of peracarid species, its percentage contribution was distinct in each season. During first year the maximum population density was found during PRM (42.8 %) followed by PM (31 %) and MN (26 %), whereas during second year, the maximum density was observed during PM (38 %) followed by MN (32.6 %) and PRM (29.1 %). The k-dominance plot of peracarid species denoted that, station 3 and station 9 were representing highly disturbed situation with low diversity and high dominance during first year. During second year, the dominance curve of station 1 to 3 was representing highly stressed condition. The resulting low diversity was attributed to the dominance of *C. geniculata* in stations 1 to 3 and *C. chilkensis* in station 9. Based on the SIMPER analysis it was found that, *E. chilkensis, C. geniculata* and *Gammarus* sp. were the species that played a major role in the maximum discrimination between the south and north zone of TMB. However, peracarids were more abundant in the northern zone. The results of BIOENV analysis showed that environmental factors such as water temperature, pH, alkalinity, salinity, sediment pH, silt, clay and available phosphorus were forming the best combination to explain the distribution and abundance of peracarids. CCA analysis gives additional insight on the environmental parameters that played the most significant role in structuring the community assemblages of peracarids in Vembanad estuary.

Chapter 8 – deals with the ecological integrity and functionality of benthic macrofauna in Vembanad estuary. The AMBI values for the macrobenthic species ranged from1.08 to 4.79 and 0.48 to 3.63 during first and second year respectively. During first year, station 1 (4.8), 2 (4.4), 5 (4), 6 (3.3) and 9 (4.5) were representing a moderately disturbed condition, whereas wtation 3 (1.9), 4 (1.8), 7 (1.5) and 8 (2.3) representing a slightly disturbed condition. Low AMBI value of station 10 was classified as undisturbed sites, where 70.9 % of the macrofaunal species were in EG I category. In contrast with first year during second year in stations 2, 5, 6, 9 and 10 were representing slightly disturbed condition. At the same time station 3 and 4 were classified as undisturbed sites where AMBI value was < 1. Based on the M-AMBI values, during first year the ecological grade of Vembanad estuary was varied between poor and high status. In which stations 1 and 2 were classified as poor (0.24) and moderate (0.4) status respectively. At the same time station 3 to 9 were in good condition and station 10 as high quality status. During second year M-AMBI values representing high (0.89) to moderate (0.45) condition. BENTIX index varied from poor to high ecological status during first year. The poor benthic status was observed in station 1 (2.2), station 2 (2.1) and station 9 (2.2) and high benthic status was observed in station 7 (4.5) and station 10 (5.2). Moderate status was observed in station 5 (2.7) and station 6 (3). At the same time during second year the BENTIX index ranged between 3.1 (station 1) to 5.5 (station 4) and get three ecological condition ranging from moderate to high. BOPA index denoted that the ecological status of Vembanad estuary was varied from bad to high status during first year and bad to good status during second year.

During first year low BOPA value indicated that the ecological status of station 10 (0.04) was in a pristine condition during first year and the high value of station 2 (0.26) represented an impacted macrobenthic community structure, whereas during second year, majority of the stations except station 6 (0.17) and 7 (0.14) were in moderate condition and station 5 (0.25) was observed as an impacted site.

As a whole, six feeding guilds were identified from the macrofaunal community in Vembanad estuarine system. During 2011-2012 period among the trophic categories, carnivores were the most abundant and contributing 46.16 % of the trophic guild followed by filter feeders (25.36 %), surface deposit feeders (11.25 %) and subsurface deposit feeders (10.62 %). In contrast with 2011-2012 period, during 2012 – 2013 period, among the trophic groups the filter feeders were the most abundant (52.25 %) followed by carnivores (29.36 %), subsurface deposit feeders (8 %) and surface deposit feeders (7.08 %). The increased abundance of C in station 1 and 2 was mainly due to the dominance of *Namalycastis indica*, whereas in station 7 and 8 was mainly attributed by the carnivorous isopods *Apanthura sandalensis* and *Xenanthura orientalis*. At the same time during second year, the FF *Villorita cyprinoids* become more abundant and play an important role in maintaining the water quality. In the present investigation, SSDF were more abundant in areas with higher organic content (station 1), that in agreement with their feeding requirements. The closeness of SSDF in station 1 may be related to the increasing organic carbon content (av. 7.2 %) in the sediment. The high faunal abundance of SSDF oligochaetes in station 1 possibly exploit the organic rich detritus settled on sediments. Bray-Curtis similarity cluster indicated that there was a distinct distribution of macrobenthic feeding guilds in Vembanad estuary. CCA analysis showed that during first year SDF and SSDF showed a strong relationship with sandy substratum whereas during second year O and GR showed strong affinity towards sand and temperature. It was also noted that, the tidal currents in the southern sector of TMB had limited influence on the trophic structure of macrobenthos, because reduced tidal currents characterize the dominance of surface and subsurface deposit feeders as compared to suspension feeders and carnivores. Overall, the trophic composition of macrobenthic community seems to reflect the varying impact of hydrodynamics in the estuary. The abundance of filter feeders regulated the trophic production in the Vembanad estuary to a certain extent by their filtering activity, thus checking the eutrophication problems in the system. So in conclusion, variations in the macrobenthic community and different trophic groups reflected different food resources with a complex food web in Vembanad estuarine system that was greatly getting influenced by drastic environmental and habitat modification.

Various studies in the estuarine area indicate that the estuary is being imposed with major geomorphic changes at several segments, resulting in reduction in the extent. Alterations in the geomorphic characteristic in the estuary occurred over the decades modifying estuarine flows, which ultimately affect the dynamic processes within the estuary (Dinesh Kumar et al., 2014; Dipson, 2012). Marked fluctuations in water quality conditions in both sectors of Vembanad estuary is influenced by hydrological factors, salinity regime and eutrophication. The study also observed that, unscientific developmental activities, dredging and other anthropogenic activities are rampant in the study area that could have seriously affected the biodiversity treasure of benthic fauna – the basic food source for fishes and other organisms in the trophic system. This type of recognition brings to light the conservation requirements of habitats and specific protection measures must be fixed. Based on the study the following recommendations are put forth for proper management of the Vembanad estuary.

- The study has for first time chronicled the current status and etiology of benthic fauna in the Ramsar site -Vembanad estuary, under different episodic conditions (including climate related issues), especially the interventions created by Thanneermukkom barrage. The benthic realm regulates the grazing and detritus food chain sustaining the productivity and overall nourishment in an ecosystem. However, the study has reaffirmed that the southern and northern stations of the estuary witnessed serious benthic impoverishment and low ecological integrity, exemplified more in the southern zones creating isolated microhabitats with contrasting ecological properties nearly threatening the very existence of organisms. The changes recorded in benthic fauna in Vembanad estuary now, can also portend for long term impacts on the fauna of other backwaters and lagoons, which form a networked system along the south west coast of the state. Thus, to maintain the ecological integrity of Vembanad and associated aquatic systems along the west coast, an integrated BENTHIC MONITORING AND MANAGEMENT PROGRAM is to be mooted for the long term trophic sustainability of the coastal region.

- The trophic index, nutrient profile pattern depicted a contrasting and diverging trend in the different sectors of estuary, creating isolated microhabitats in the estuary. These microhabitats symbolized the depleting environmental conditions – nutrient – organic enrichment and salinity pattern arising from various stress factors like liquid and solid waste disposal from various sources, decaying weeds and plant materials, industrial activity, domestic discharges and related factors. As all these ecobiological alterations are interlinked, it has to be monitored on a regular basis for understanding and resolving the nutrient load and eutrophication related issues.

- An alarmingly high concentration of Zn, Cu, Pb has been observed in the sediments of Vembanad lake, which could have serious health issues related to biomagnification. Therefore it is high time to implement bioremediation solutions to reduce the environment load, preferably at the site of generation.

- The results of organic carbon and organic matter revealed that, over the three decades period, a threefold increase in sediment organic carbon and organic matter was observed. This was evident in the proliferation of opportunistic polychaetes and oligochaetes in the southern zone. Overall assessment of ecological quality of soft-bottom benthic communities in the estuary established an impacted condition especially in the southern zone. So the Vembanad estuary should be conserved for long term conservation objectives of the state. Otherwise Vembanad estuary will possibly disappear from the list of world famous tourist location.

- As Thanneermukkom barrage plays a pivotal role in determining the physico-chemical and biotic character of the entire Vembanad estuarine system, effective steps are to be initiated for co-ordinating its proper operation and maintaining the continuity of the water body, so as to resolve the burgeoning environmental and social issues in the designated Ramsar site of southern India. It is also important to note that, after our study period, as a recent development in 2014, Govt. of Kerala initiative, the coffer dams and other obstructions along the TMB are being suitably replaced by sluice gates (barrages) for overall rejuvenation of the ecology of the estuary, for the continuity of the water body on the south and northern sides and also the effective management for salinity regulation in the low lying paddy fields of the wetland. However further studies are necessary to understand the effects of such renewed changes on the ecology of estuary on a long term basis.

- Unauthorized sand mining is common in the whole estuarine system. The uncontrolled mining of live and dead shells (*Villorita cyprinoids*) from the lake is also posing a threat to the ecosystem. Large scale industrial mining of shells which take place in the estuary for the white shells is done using mechanized dredgers. Large scale mechanized dredging of white shells by the Travancore Cements Ltd. and undersized live clam exploitation has badly creates many environmental implications. Based on the trophic group analysis, filter feeders were more abundant in the estuary. The abundance of filter feeders regulated the trophic production in the Vembanad estuary to a certain extent by their filtering activity, thus checking the eutrophication problems in the system. So periodic monitoring of the estuary is essential for the formulation of viable management options for the sustainable utilization of this vital environmental resource.
- Since tourism activity is increasing in Vembanad estuary due to its international attraction, there is an urgent need to restrict various pressures associated with this on the wetland and its resources especially the rampant increase in house boats. A carrying capacity based model needs to be developed on the impact of tourism and related pressures in the wetland. Action for protecting Pathiramanal Island and Kumarakom bird sanctuary should be implemented. Special attention should be taken by Tourism Department for restricting nature tourism without involving any ecological modification of the islands and promote mangrove afforestation. Strengthening water bird habitat assessment and monitoring network through training, awareness and participation programmes is to be enforced.

- Urbanization and developmental activities in the Punnamada, Perumbalam and Aroor region resulted in the encroachment of the estuary by violating and regulating the coastal regulation zone (CRZ) norms. National Green Tribunal can enforce laws to prevent the reclamation in the estuaries.
- Kuttanad region encompassing the Vembanad estuary lying about average 2 m below mean sea level, supports indigenous species and farming technologies being propagated by local population that has been developed through indigenous knowledge. Such practices have been followed for decades and are being protected on a sustainable basis. In the present changing climatic regime, with impending gradual sea level rise and associated issues the farming techniques being implemented here would be useful for the overall benefit to the population. So more research and development oriented works should be evolved in Vembanad and used as a model in other parts of the country for coping (adaptive) with possible climate change issues in future.
- It has been observed that the Kuttanad package (Dr. M.S. Swaminathan Research Foundation) being implemented by the Govt. of India at ₹ 1840 crore for mitigating the agrarian issues, is not a successful scheme. It needs to be suitably amended and reworked for improving the quality of life in the entire estuarine system. It is even appropriate that a national level committee consisting of economists, bureaucrats, researchers and ecologists review the progress of the implementation and also suggest ways for its augmentation.
- Among the various conservation measures and system, Ramsar Convention is the important initiatives. Even though Vembanad estuary is a Ramsar site, there were no strict conservation measures and proper management of this degrading estuary in any Central

and State Governmental level. So Government should reform the existing laws in a scientific manner and take measures to solve the environmental problems. It should include habitat protection; prevent the unfriendly activities in the ecosystem avoiding further degradation. So establishment of protected areas has been considered as the most effective means to mitigate the impacts of deterioration in the estuary. Having the unique ecological characteristics and presence of endemic indigenous species, the Vembanad and associated estuarine system needs to be considered as UNESCO world Heritage site for conservation and sustainable utilization.

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Annexure 3.1 Mean monthly variations in physico-chemical parameters in Vembanad estuary during 2011 – 2012 period

	Rainfall (mm)	Tide and tidal height (cm)	Temperature (°C)	pH	Depth(m)	Transparency (m)	TDS (ppm)	Turbidity (NTU)	DO (mg L^{-1})	BOD (mg L^{-1})	Salinity (ppt)
Mar '11	69.78	low tide (35)	30.95 ± 0.79	6.9 ± 2.3	4.14 ± 1.89	0.89 ± 0.38	2.93 ± 1.86	2 ± 1.09	7.45 ± 0.76	2.58 ± 1.29	7 ± 4.76
Apr '11	96.6	high tide (87)	31.05 ± 0.59	6.7 ± 0.34	3.97 ± 1.44	0.75 ± 0.22	3.03 ± 2.08	2.64 ± 1.02	7.89 ± 1.08	4.15 ± 1.35	1.23 ± 1.2
May '11	228.06	low tide (48)	31.45 ± 0.44	7.2 ± 0.3	3.87 ± 2.2	0.94 ± 0.34	5.25 ± 2.10	1.74 ± 0.63	8.46 ± 0.78	4.72 ± 1.66	2.89 ± 2.84
Jun '11	442.2	low tide (54)	27.1 ± 0.8	6.8 ± 0.2	4.59 ± 2	0.79 ± 0.2	0.46 ± 0.6	2.27 ± 0.8	7.64 ± 0.9	3.01 ± 1.6	0.0 ± 0.2
Jul '11	339.96	high tide(82)	27 ± 0.91	6.7 ± 0.25	4.39 ± 2.4	0.96 ± 0.25	2.08 ± 0.81	2.15 ± 0.64	7.72 ± 0.88	2.12 ± 0.42	0.0 ± 0
Aug '11	373.92	high tide(80)	29.6 ± 1.17	7.14 ± 0.39	4.44 ± 1.9	1.08 ± 0.23	1.58 ± 1.35	2.55 ± 0.66	8.13 ± 0.86	1.47 ± 0.52	0.0 ± 0
Sep '11	322.41	high tide (96)	31.3 ± 0.86	6.7 ± 0.42	4.06 ± 1.9	1.23 ± 0.33	1.03 ± 0.83	2.51 ± 0.9	7.72 ± 0.88	3.90 ± 0.92	0.33 ± 0.68
Oct '11	331.65	high tide (76)	28.45 ± 0.76	6.9 ± 0.31	4.28 ± 1.8	1.08 ± 0.15	5.03 ± 6.35	1.46 ± 1.08	7.64 ± 0.69	3.42 ± 0.75	0.4 ± 0.76
Nov '11	161.91	high tide (61)	28.5 ± 1.13	7.32 ± 0.18	4.35 ± 2.4	0.96 ± 0.09	2.88 ± 6.7	0.92 ± 0.64	6.5 ± 0.53	2.03 ± 1.39	4.45 ± 9.7
Dec '11	12.21	high tide (78)	27.3 ± 1.81	7.59 ± 0.3	3.89 ± 2	1.12 ± 0.27	5.06 ± 6.37	1.56 ± 1.11	6.83 ± 0.57	2.44 ± 0.5	9.2 ± 0.64
Jan '12	5.16	high tide (75)	29.15 ± 0.94	7.65 ± 0.33	3.9 ± 2.1	1.08 ± 0.32	5.42 ± 6.8	1.29 ± 0.48	6.75 ± 0.77	3.01 ± 1.02	6.32 ± 6.5
Feb '12	28.68	high tide (86)	29.6 ± 0.94	7.97 ± 0.47	3.95 ± 1.9	1.0 ± 0.47	4.59 ± 6.18	2.09 ± 1.35	7.67 ± 0.59	2.44 ± 0.8	6.85 ± 6.5

Annexure 3.2 Mean monthly variations in major nutrient parameters, chlorophyll *a* and TRIX in Vembanad estuary during 2011 – 2012 period

	PO_4 – P (µmol L^{-1})	Si O_4 – Si (µmol L^{-1})	NO_2 – N (µmol L^{-1})	NO_3 – N (µmol L^{-1})	NH_4 – N (µmol L^{-1})	Chl. a (mg m^{-3})	N:P	TRIX
Mar '11	1.07 ± 1.32	29.48 ± 9.1	0.57 ± 0.9	0.79 ± 0.42	3.56 ± 4.28	12.43 ± 2.08	5.29 ± 4.6	6.5 ± 0.63
Apr '11	1.08 ± 0.77	21.93 ± 9.4	0.18 ± 0.11	1.06 ± 0.73	4.33 ± 5.37	12.93 ± 1.94	6.02 ± 5.47	6.35 ± 0.79
May '11	1.66 ± 0.48	3.19 ± 1.5	0.15 ± 0.083	1.08 ± 0.34	2.68 ± 1.62	9.84 ± 2.27	2.34 ± 0.83	6.62 ± 0.39
Jun '11	1.28 ± 0.43	17.07 ± 3.33	0.13 ± 0.07	2.52 ± 0.56	12.26 ± 5.27	6.86 ± 1.66	12.14 ± 3.7	6.96V0.36
Jul '11	0.36 ± 0.23	7.5 ± 4.07	0.27 ± 0.13	1.83 ± 0.45	3.18 ± 1.22	7.66 ± 1.58	21.08 ± 15.93	6.29 ± 0.25
Aug '11	0.62 ± 0.21	22.8 ± 1.36	0.29 ± 0.12	1.7 ± 0.88	1.92 ± 1.28	4.58 ± 1.52	6.73 ± 3.17	6.06 ± 0.51
Sep '11	1.23 ± 0.38	45.32 ± 2.88	0.91 ± 1.63	1.73 ± 0.78	4.00 ± 3.6	4.45 ± 1.26	5.74 ± 3.35	6.52 ± 0.62
Oct '11	1.09 ± 0.94	8.65 ± 6.67	0.15 ± 0.09	1.41 ± 0.85	25.02 ± 6.19	5.92 ± 7.78	37.71 ± 23.46	7.09 ± 0.41
Nov '11	1.75 ± 0.85	6.88 ± 2.38	0.1 ± 0.09	2.18 ± 0.89	4.35 ± 1.53	18.55 ± 4.16	4.62 ± 2.41	7.03 ± 0.38
Dec '11	1.24 ± 0.59	26.06 ± 6.7	0.17 ± 0.09	2.29 ± 1.03	17.8 ± 10.73	5.39 ± 3.57	19.75 ± 13.03	6.88 ± 0.56
Jan '12	1.43 ± 0.56	21.98 ± 6.23	0.24 ± 0.16	0.51 ± 0.34	1.23 ± 1.77	1.1 ± 0.95	1.46 ± 1.38	5.46 ± 0.85
Feb '12	1.28 ± 0.95	20.91 ± 4.55	0.26 ± 0.17	1 ± 0.29	7.15 ± 4.57	2.97 ± 2.51	8.64 ± 4.79	6.18 ± 0.51

Annexure 3.3 Mean monthly variations in physico-chemical parameters in Vembanad estuary during 2012 – 2013 period

	Rainfall (mm)	Tide and tidal height (cm)	Temperature (°C)	pH	Depth (m)	Transparency (m)	TDS (ppm)	Turbidity (NTU)	DO (mg L^{-1})	BOD (mg L^{-1})	Salinity(ppt)
Mar '12	7.71	high tide (94)	31.7 ± 0.6	7.36 ± 0.1	4.3 ± 2	0.76 ± 0.3	6.77 ± 7.8	7.13 ± 6.4	5.9 ± 1.	2.34 ± 1	11.64 ± 1
Apr '12	286.89	-	31.8 ± 1.32	7.19 ± 0.24	4.4 ± 2.13	0.96 ± 0.33	6.19 ± 5.3	9.65 ± 9.28	6.16 ± 0.93	3.15 ± 1.26	11.81 ± 8.82
May'12	189.06	high tide (73)	32.4 ± 0.66	7.19 ± 0.29	3.56 ± 2.15	0.81 ± 0.25	1.48 ± 2.28	2.8 ± 1.5	6.8 ± 0.57	2.00 ± 1.39	2.89 ± 4.45
Jun '12	269.91	high tide (60)	29.7 ± 0.48	6.92 ± 0.15	3.93 ± 1.8	0.75 ± 0.23	1.19 ± 1.12	1.75 ± 0.87	8.8 ± 0.99	3.03 ± 1.35	1.44 ± 1.05
Jul '12	167.19	low tide (55)	29.7 ± 0.82	6.8 ± 0.24	3.69 ± 2.12	0.99 ± 0.31	0.44 ± 0.4	2.23 ± 1.16	6.32 ± 0.45	2.09 ± 1.27	0.49 ± 0.59
Aug '12	280.92	low tide (52)	30.6 ± 1.17	6.63 ± 0.14	4.26 ± 2.04	1.03 ± 0.3	0.29 ± 0.48	1.99 ± 0.86	7.04 ± 0.83	2.49 ± 1.51	0.59 ± 0.65
Sep '12	119.52	high tide (76)	31.6 ± 0.6	6.89 ± 0.1	3.34 ± 2	1.06 ± 0.5	1.85 ± 4.3	2.11 ± 0.98	8.08 ± 0.96	4.01 ± 1.23	3.28 ± 3.93
Oct '12	555.45	high tide (76)	31.3 ± 0.67	6.73 ± 0.23	3.97 ± 2.16	0.95 ± 0.28	0.66 ± 1.4	2.92 ± 1.04	5.84 ± 0.93	3.08 ± 1.11	6.35 ± 4.2
Nov '12	105.54	high tide (98)	32 ± 1.05	6.72 ± 0.17	3.94 ± 1.9	1.14 ± 0.28	1.26 ± 2.85	2.4 ± 1.4	5.36 ± 1.0	2.19 ± 0.83	9.31 ± 8.53
Dec '12	8.04	high tide (82)	31.55 ± 0.95	6.97 ± 0.25	4.05 ± 1.95	1.2 ± 0.57	4.8 ± 4.26	0.49 ± 0.4	6.16 ± 0.93	1.77 ± 0.77	11.95 ± 1.13
Jan '13	6.06	high tide (77)	31.65 ± 0.94	7.49 ± 0.22	3.76 ± 1.9	10.32 ± 2.79	7.96 ± 6.6	2.18 ± 1.68	6.16 ± 0.66	2.34 ± 0.86	14.27 ± 1.03
Feb '13	53.22	low tide (54)	32.2 ± 1.03	7.58 ± 0.24	4.04 ± 2.09	1.15 ± 0.65	8.99 ± 7.47	12.83 ± 1.27	6.24 ± 1.35	2.09 ± 1.08	15.72 ± 9.46

Annexure 3.4 Mean monthly variations in major nutrient parameters, chlorophyll *a* and TRIX in Vembanad estuary during 2012 – 2013 period

	PO_4 - P(µmol/l)	Si O_4 - Si(µmol/l)	NO_2 - N(µmol/l)	NO_3 - N(µmol/l)	NH_4 - N(µmol/l)	Chl. a (mg m^{-3})	N : P	TRIX
Mar '12	2.65 ± 2.55	17.87 ± 1.04	0.28 ± 0.24	1.31 ± 0.81	6.41 ± 4.76	11.22 ± 6.13	4.54 ± 3.29	7.07 ± 0.73
Apr '12	0.79 ± 0.59	39.69 ± 9.97	0.7 ± 0.68	1.02 ± 1.17	7.5 ± 4.83	11.04 ± 8.39	14.59 ± 7.32	6.69 ± 0.54
May '12	1.45 ± 0.89	4.11 ± 0.89	0.16 ± 0.09	3.8 ± 4.86	5.57 ± 3.48	17.09 ± 9.43	7.69 ± 5.28	7.05 ± 0.47
Jun '12	1.97 ± 1.44	8.55 ± 1.97	0.15 ± 0.06	3.86 ± 5.19	6.04 ± 5.8	12.5 ± 6.77	8.11 ± 6.59	6.86 ± 0.56
Jul '12	1.7 ± 1.4	19.15 ± 6.12	0.15 ± 0.05	1.95 ± 0.96	1.73 ± 0.64	0.62 ± 0.49	3.01 ± 1.4	5.75 ± 0.65
Aug '12	1.14 ± 0.84	18.41 ± 5.87	0.16 ± 0.05	2.08 ± 0.43	2.22 ± 1	0.37 ± 0.35	6.32 ± 4.58	5.32 ± 0.89
Sep '12	2.18 ± 1.1	28.82 ± 9.32	0.3 ± 0.2	1.62 ± 0.49	1.81 ± 2.29	8.88 ± 1.16	1.74 ± 0.55	6.43 ± 0.75
Oct '12	0.93 ± 0.78	20.25 ± 1.14	0.14 ± 0.06	1.17 ± 0.27	2.36 ± 0.87	4.81 ± 2.39	6.5 ± 5.17	6.34 ± 0.46
Nov '12	6.73 ± 1.35	30.04 ± 2.71	0.12 ± 0.09	1.8 ± 0.89	5.78 ± 7.59	4.96 ± 3.63	1.51 ± 2.19	7.23 ± 0.49
Dec '12	0.99 ± 0.42	14.12 ± 8.7	0.14 ± 0.12	0.56 ± 0.34	3.34 ± 3.82	9.12 ± 5.23	4.05 ± 2.85	6.45 ± 0.49
Jan '13	1.42 ± 0.69	18.02 ± 7.6	0.35 ± 0.25	0.84 ± 0.88	3.83 ± 3.9	7.42 ± 4.67	3.92 ± 2.07	6.33 ± 0.38
Feb '13	1.85 ± 1.05	31.92 ± 1.74	0.62 ± 0.63	0.49 ± 0.16	6.38 ± 7.29	17.6 ± 2.59	15.68 ± 38.11	6.77 ± 0.64

Annexure 3.5 Seasonal variation in physico-chemical parameters in Vembanad estuary during 2011-2012 and 2012 – 2013 period

Parameter	2011 - 2012 period			2012 - 2013 period		
	PRM	MN	PM	PRM	MN	PM
Rainfall (mm)	394.44 ± 84.71	1478.49 ± 52.89	510.93 ± 153.97	483.66 ± 141.657	837.54 ± 78.82	675.09 ± 261.93
River discharge (Mm^3)	5297.17 ± 1096.6	27225.64 ± 1925.1	8686.28 ± 3864.5	-	-	-
Temperature (°C)	30.78 ± 0.99	28.75 ± 2.04	28.39 ± 1.37	32.03 ± 0.96	30.4 ± 1.13	31.63 ± 0.92
pH	7.14 ± 0.55	6.82 ± 0.36	7.41 ± 0.44	7.33 ± 0.27	6.82 ± 0.19	6.98 ± 0.38
Depth(m)	3.88 ± 1.72	4.37 ± 2	4.19 ± 2.1	4.08 ± 2.1	3.82 ± 1.9	3.92 ± 1.9
Transparency (m)	0.9 ± 0.36	1.02 ± 0.3	1.04 ± 0.23	0.92 ± 0.42	0.96 ± 0.34	
TDS (ppm)	3.55 ± 2.5	1.29 ± 1.09	4.96 ± 6.7	5.85 ± 6.49	0.95 ± 2.2	3.67 ± 5.05
Turbidity	2.04 ± 0.96	2.37 ± 0.74	1.39 ± 1.05	7.98 ± 8.85	2.02 ± 0.95	1.99 ± 1.49
DO (mg L^{-1})	7.88 ± 0.89	7.8 ± 0.86	6.94 ± 0.75	6.28 ± 1.03	7.56 ± 1.25	5.88 ± 0.92
BOD (mg L^{-1})	3.50 ± 1.62	2.62 ± 1.32	2.72 ± 1.07	2.40 ± 1.24	2.90 ± 1.48	2.34 ± 0.99
Salinity (ppt)	4.04 ± 4.1	0.12 ± 0.35	5.48 ± 8.4	10.51 ± 1	1.45 ± 2.3	10.47 ± 9.13
Chl a (mg/m^3)	9.76 ± 4.39	5.87 ± 2.03	7.58 ± 2.12	14.24 ± 1.45	5.59 ± 8.38	6.58 ± 4.36
PO_4 – P (µmol L^{-1})	1.23 ± 0.89	0.87 ± 0.5	1.42 ± 0.8	1.69 ± 1.57	1.77 ± 1.24	2.52 ± 2.6
Si O_4 –Si (µmol L^{-1})	18.99 ± 1.1	23.19 ± 2.09	15.86 ± 9.94	23.4 ± 1.76	18.73 ± 9.5	20.61 ± 1.63
NO_2 – N (µmol L^{-1})	0.29 ± 0.48	0.4 ± 0.84	0.16 ± 0.29	0.44 ± 0.52	0.19 ± 0.13	0.19 ± 0.17
NO_3 – N (µmol L^{-1})	0.98 ± 0.48	1.95 ± 0.74	1.51 ± 0.88	1.66 ± 2.75	2.38 ± 2.71	1.09 ± 0.79
NH_4 – N (µmol L^{-1})	4.22 ± 4.2	5.34 ± 5.2	12.11 ± 11.4	6.47 ± 5.11	2.95 ± 3.56	3.83 ± 4.69
N:P	5.59 ± 4.79	11.42 ± 10.23	15.61 ± 19.33			

Annexure 3.6 ANOVA table showing the spatial and seasonal variation of physico-chemical parameters in the Vembanad estuary during 2011-2012 period

Variable	Source	df	Mean Square	F	Variable	Source	df	Mean Square	F
Temperature	Station	9	7.36	3.382**	**DCO_2**	station	9	14.249	1.462**
	Season	2	68.327	31.396		Season	2	622.491	63.866**
pH	Season	2	3.463	15.562**	**Salinity**	station	9	111.994	6.842**
Depth	Station	9	42.437	95.256**		Season	2	282.916	17.284**
Conductivity	station	9	233.244	5.536**	**$PO_4 - P$**	station	9	2.189	5.455**
	Season	2	757.128	17.97**		Season	2	2.854	7.112**
TDS	station	9	90.514	23.783**	**$NO_3 - N$**	Season	2	9.152	15.856**
	Season	2	116.295	30.557**	**$NH_4 - N$**	Season	2	723.875	10.67**
Turbidity	station	9	3.288	5.047**	**TN**	Season	2	735.243	10.17**
	Season	2	10.501	16.119**					

Degree of freedom (df); F statistic (F); ** Variation is significant at 1% level; *Variation is significant at 5% level

Annexure 3.7 ANOVA table showing the spatial and seasonal variation of physico-chemical parameters in the Vembanad estuary during 2012-2013 period

Variable	Source	df	Mean Square	F	Variable	Source	df	Mean Square	F
Temperature	station	9	2.769	3.13*	**BOD**	station	9	7.704	7.117**
	Season	2	28.675	32.411**		Season	2	3.852	3.558*
pH	Season	2	2.768	31.124**	**Salinity**	station	9	439.335	18.103**
Depth	station	9	44.266	96.735		Season	2	1090.906	44.952**
Alkalinity	station	9	1255.126	12.854**	**Chl. a**	Season	2	895.567	11.629**
	Season	2	736.133	7.539**	**$NO_2 - N$**	station	9	0.336	4.229**
Conductivity	station	9	737.439	10.953**		Season	2	0.851	10.701**
	Season	2	977.284	14.515**	**$NO_3 - N$**	Season	2	16.591	3.207*
TDS	station	9	143.424	10.567**	**$NH_4 - N$**	station	9	92.454	7.784**
	Season	2	241.773	17.813**		Season	2	134.26	11.303**
Turbidity	Season	2	484.372	21.646**	**DIN**	station	9	102.269	5.434**
CO_2	Season	2	31.852	12.939**		Season	2	142.944	7.595**
DO	Season	2	30.805	25.113**	**TRIX**	station	9	1.667	3.752**
BOD	station	9	7.704	7.117**		Season	2	6.578	14.804**
	Season	2	3.852	3.558*					

Degree of freedom (df); F statistic (F); ** Variation is significant at 1% level; *Variation is significant at 5% level

Annexure 3.8 Correlation matrix between physico-chemical parameters in Vembanad estuary during 2011-2012 period

	Water Temperature	pH	Transparency	Conductivity	TDS	PO_4-P	Salinity	Turbidity	DO	Alkalinity	DCO_2	NH_4-N	NO_3-N	NO_2-N	TN	TRIX
Water Temperature	1															
pH	-0.054	1														
Transparency	-0.033	-0.028	1													
Conductivity	**.360****	**.441****	-0.121	1												
TDS	**.288****	**.319****	-0.151	**.752****	1											
PO_4-P	**.214***	**.248****	**-.263****	**.460****	**.536****	1										
Salinity	**.229***	**.433****	-0.072	**.804****	**.747****	**.449****	1									
Turbidity	**.220***	-0.077	**-.304****	0.095	0.061	0.023	0.046	1								
DO	**.285****	-0.164	0.089	-0.098	-0.1	-0.142	**-.264****	**.192***	1							
Alkalinity	**.308****	0.022	**-.244****	**.197***	**.226***	**.207***	**.373****	0.154	-0.035	1						
DCO_2	**-.294****	**.227***	0.141	0.091	0.177	**.250****	0.141	**-.281****	**-.294****	**-.291****	1					
NH_4-N	**-.414****	0.057	-0.078	-0.122	0.069	0.153	-0.078	0.013	-0.093	-0.091	**.547****	1				
NO_3-N	**-.366****	0.005	-0.157	-0.167	-0.021	0.172	-0.058	0.15	-0.037	-0.102	**.226***	**.314****	1			
NO_2-N	**.225***	-0.125	0.149	0.045	0.039	**.194***	0.093	**.224***	0.068	**.208***	0.006	-0.05	0.116	1		
TN	**-.422****	0.048	-0.082	-0.132	0.067	0.178	-0.075	0.043	-0.089	-0.084	**.550****	**.992****	**.412****	0.028	1	
TRIX	**-.195***	-0.098	**-.229***	0.028	0.121	**.446****	0.039	0.014	**-.228***	0.091	**.351****	**.525****	**.469****	0.163		1

Annexure 3.9 Correlation matrix between physico-chemical parameters in Vembanad estuary during 2012 – 2013 period.

	Water Temperature	pH	Conductivity	TDS	PO_4-P	Salinity	Turbidity	DO	Alkalinity	DCO_2	NH_4-N	NO_3-N	NO_2-N	TN	TRIX
Water Temperature	1														
pH	**.278****	1													
Conductivity	**.408****	**.446****	1												
TDS	**.388****	**.472****	**.967****	1											
PO_4-P	0.079	-0.111	0.051	0.054	1										
Salinity	**.515****	**.405****	**.889****	**.905****	0.129	1									
Turbidity	**.216***	**.269****	**.475****	**.571****	0.121	**.451****	1								
DO	**-.256****	-0.126	**-.248****	**-.274****	**-.226***	**-.345****	**-.228***	1							
Alkalinity	**.221***	**.326****	**.590****	**.593****	**.197***	**.630****	0.151	**-.256****	1						
DCO_2	-0.171	**.209***	0.132	0.134	-0.02	0.035	0.08	0.079	**.314****	1					
NH_4-N	0.138	**.369****	**.272****	**.257****	0.151	.268**	0.129	-0.099	**.511****	**.499****	1				
NO_3-N	-0.127	-0.107	-0.123	-0.124	0.141	**-.200***	-0.024	**.197***	-0.086	0.017	-0.004	1			
NO_2-N	**.270****	**.351****	**.673****	**.710****	-0.043	**.619****	**.712****	-0.133	**.484****	**.268****	**.248****	-0.06	1		
TN	0.084	**.302****	**.231***	**.219***	**.192***	**.191***	0.145	-0.011	**.447****	**.466****	**.900****	**.428****	**.258****	1	
TRIX	0.165	**.366****	**.257****	**.237****	**.506****	**.228***	**.198***	**-.217***	**.326****	0.133	**.544****	0.127	0.135	**.546****	**1**

Annexure 3.10 Comparison of physico- chemical parameters on south and north of TMB during 2011-2012 and 2012-2013 period

Parameters	2011-2012		2012-2013	
	South of TMB	**North of TMB**	**South of TMB**	**North of TMB**
pH	7.06±0.06	7.23±0.22	7.03±0.05	9.81±5.64
Depth (m)	4.27±1.42	4.06±2.72	4.12±1.66	3.65±2.52
Transparency (m)	1.05±0.21	0.91±0.20	1.17 ±0.26	0.89±0.22
Water temperature (^{0}C)	28.78±0.45	30.05±0.37	31.07±0.37	31.77±0.28
Turbidity (NTU)	1.74±0.40	2.19±0.65	2.54±0.62	5.61±2.46
TDS (pp)	1.45±0.72	6.02±3.24	1.24±0.52	6.86±3.18
Conductivity (mS cm^{-1})	2.81±0.65	9.85±5.32	2.36±1.00	14.63±7.90
Salinity (ppt)	1.30±0.50	6.11±3.80	3.28±1.47	13.78±4.24
Alkalinity (mg L^{-1})	38.17± 5.22	56.67 ±14.36	24.28 ±4.12	35.39 ±12.63
DO (mg L^{-1})	7.35±0.38	7.66±0.22	6.66±0.25	6.45±0.46
BOD (mg L^{-1})	2.94 ± 0.25	2.94± 0.26	2.29 ±0.53	2.94 ±1.06
Chl a (mg m^{-3})	7.35±0.56	6.63±1.02	6.79±1.94	8.57±1.51
Si O_4-Si (µmol L^{-1})	19.65±3.24	18.60±2.25	22.44±3.36	18.63±1.62
PO_4-P (µmol L^{-1})	1.00±0.24	1.43±0.57	1.77±0.39	2.31±0.63
NH_4-N (µmol L^{-1})	7.38±2.58	7.16±1.19	4.19±3.23	4.89±2.30
NO_3-N (µmol L^{-1})	1.45±0.29	1.61±0.40	1.51±0.50	2.00±0.79
NO_2-N (µmol L^{-1})	0.24±0.18	0.35±0.16	0.17±0.04	0.44±0.15
DIN (µmol L^{-1})	9.06±2.83	9.12±1.74	5.72±2.99	7.42±2.88
TRIX	6.53 ±0.25	6.48 ±0.39	6.44 ± 0.46	6.59± 0.38

Annexure 3.11 Comparison of physico-chemical parameters in the southern zone of Vembanad estuary during the open and closure period of TMB

Parameters	Open period	Closure period
pH	6.84	7.53
Transparency (m)	1.04	0.98
Turbidity (NTU)	1.77	2.76
TDS (ppm)	1.52	1.72
Salinity (ppt)	0.92	3.30
CO_2 (mg L^{-1})	5.80	6.20
DO (mg L^{-1})	7.38	6.77
BOD (mg L^{-1})	2.97	2.67
Si O_4-Si (µmol L^{-1})	18.61	25.97
PO_4-P (µmol L^{-1})	1.00	1.03
NH_4-N (µmol L^{-1})	6.38	8.92
NO_3-N (µmol L^{-1})	1.49	1.11
NO_2-N (µmol L^{-1})	0.26	0.19
TRIX	6.66	6.40
Chl. a (mg m^{-3})	8.47	7.15

Annexure 3.12 Mean monthly river discharge data (Mm^3) from major five rivers in to the Vembanad estuary

	Muvattupuzha	Meenachil	Pamba	Achenkovil	Manimala	Average
MAR'11	2946.6	39.2	0	0	55.3	608.22
APR' 11	3598.3	984.2	1101.7	337.2	652.7	1334.82
MAY'11	3818.8	909.1	713.4	242.9	492.1	1235.26
JUN'11	9939	5829.6	9085.8	2637.1	4648.5	6428
JULY'11	9670.2	3389.7	7019.6	1699.5	3278.7	5011.54
AUG'11	10137	3896.6	10571	1866.7	440.17	5382.29
SEPT'11	8727.8	2654.2	8486	1773	3152.4	4958.68
OCT'11	7586.8	1508.7	3522.1	922.9	1378	2983.7
NOV'11	4402.9	1305.3	2987	764.3	1469.8	2185.86
DEC'11	2823.3	482.8	1348.6	461.1	700.8	1163.32
JAN'12	1917.3	131	557.1	120.6	354.7	616.14
FEB'12	2805	0	0	0	8	562.6
Total	68373	21130.4	45392.3	10825.3	16631.2	2705.87

Annexure 4.1 Mean monthly variation of sediment parameters in Vembanad estuary during 2011-2012 period

Month	Temperature (°C)	pH	Eh (mV)	Water content (%)	Organic Carbon (%)	Organic matter (%)	Available Phosphorus (mg/Kg)	Available Nitrogen (%)	Total Carbon (g/Kg)
Mar '11	30.45 ± 0.60	5.54 ± 0.97	0.70 ± 62.25	4.49 ± 1.41	2.59 ± 2.29	4.46 ± 3.94	20.66 ± 17.68	0.01 ± 0.01	30.84 ± 24.38
Apr '11	30.75 ± 0.49	6.12 ± 0.27	-118.50 ± 60.54	7.13 ± 1.72	2.88 ± 2.54	4.97 ± 4.39	28.28 ± 28.38	0.01 ± 0.01	30.50 ± 32.95
May '11	31.28 ± 0.57	6.38 ± 0.29	-148.80 ± 74.76	4.96 ± 0.99	2.90 ± 2.60	4.99 ± 4.48	24.38 ± 20.92	0.01 ± 0.01	28.36 ± 23.82
Jun '11	30.90 ± 0.46	6.41 ± 0.32	-171.10 ± 117.82	9.50 ± 2.78	3.88 ± 3.02	6.68 ± 5.21	32.60 ± 36.17	0.01 ± 0.01	31.04 ± 17.71
Jul '11	28.10 ± 0.52	6.46 ± 0.41	-100.50 ± 75.56	8.18 ± 1.77	3.15 ± 2.41	5.43 ± 4.16	30.94 ± 17.16	0.03 ± 0.03	30.01 ± 25.32
Aug '11	29.75 ± 1.25	6.46 ± 0.32	-160.70 ± 82.35	10.95 ± 2.99	2.58 ± 2.02	4.44 ± 3.49	34.72 ± 30.37	0.03 ± 0.02	30.69 ± 26.30
Sep '11	25.25 ± 0.92	6.65 ± 0.37	-223.00 ± 130.53	5.43 ± 1.54	3.56 ± 2.99	6.13 ± 5.15	44.28 ± 39.30	0.03 ± 0.03	29.66 ± 26.63
Oct '11	23.35 ± 0.47	6.70 ± 0.30	-202.10 ± 113.67	5.79 ± 1.01	1.74 ± 1.82	2.99 ± 3.14	72.04 ± 88.77	0.05 ± 0.05	34.59 ± 32.64
Nov '11	27.05 ± 0.60	6.81 ± 0.37	-252.00 ± 102.87	5.57 ± 1.90	2.02 ± 1.64	3.48 ± 2.82	35.88 ± 40.79	0.05 ± 0.05	30.06 ± 29.92
Dec '11	27.75 ± 1.32	6.99 ± 0.28	-227.40 ± 128.22	8.82 ± 2.09	2.02 ± 1.64	4.10 ± 2.60	81.08 ± 70.82	0.04 ± 0.04	25.41 ± 19.37
Jan '12	29.00 ± 0.91	7.05 ± 0.16	-223.10 ± 138.97	6.39 ± 1.70	2.38 ± 1.51	4.55 ± 3.13	34.06 ± 24.04	0.04 ± 0.04	30.68 ± 26.20
Feb '12	29.53 ± 0.54	7.16 ± 0.19	-128.50 ± 84.94	7.55 ± 1.62	2.64 ± 1.82	2.76 ± 2.66	24.36 ± 31.59	0.04 ± 0.04	30.20 ± 24.91

Annexure 4.2 Mean monthly variation of sediment parameters in Vembanad estuary during 2012-2013 period

Month	Temperature (°C)	pH	Eh (mV)	Water content (%)	Organic Carbon (%)	Organic matter (%)	Available Phosphorus (mg/Kg)	Available Nitrogen (%)
Mar ’12	31.44 ± 0.54	7.25 ± 0.27	-255.20 ± 56.05	4.13 ± 0.53	3.21 ± 2.53	5.53 ± 4.37	50.86 ± 49.38	0.04 ± 0.04
Apr ’12	32.30 ± 0.48	7.36 ± 0.20	-103.20 ± 59.44	5.51 ± 1.01	2.93 ± 2.80	5.06 ± 4.82	25.72 ± 26.06	0.03 ± 0.01
May ’12	32.87 ± 0.54	7.42 ± 0.54	-170.00 ± 68.38	5.55 ± 1.70	3.55 ± 2.53	5.29 ± 3.69	22.36 ± 21.46	0.07 ± 0.06
Jun ’12	29.75 ± 0.63	7.35 ± 0.21	-169.10 ± 105.18	5.39 ± 1.27	3.91 ± 2.92	6.75 ± 5.03	9.32 ± 9.85	0.05 ± 0.04
Jul ’12	29.30 ± 0.48	6.97 ± 0.34	-218.90 ± 122.61	5.86 ± 2.22	3.71 ± 3.06	6.39 ± 5.28	10.25 ± 10.31	0.05 ± 0.05
Aug ’12	30.50 ± 0.97	7.26 ± 0.21	-168.40 ± 167.35	3.91 ± 1.14	2.94 ± 2.26	5.08 ± 3.89	19.12 ± 38.40	0.07 ± 0.06
Sep ’12	31.30 ± 1.25	7.13 ± 0.26	-285.40 ± 89.56	5.51 ± 1.01	3.04 ± 1.89	5.24 ± 3.25	35.82 ± 37.13	0.07 ± 0.06
Oct ’12	31.38 ± 0.49	7.01 ± 0.15	-88.90 ± 122.13	4.11 ± 0.96	2.49 ± 2.15	4.30 ± 3.71	60.92 ± 145.03	0.06 ± 0.05
Nov '12	32.20 ± 0.63	7.06 ± 0.14	-256.00 ± 70.62	4.96 ± 1.54	2.30 ± 1.80	3.97 ± 3.10	31.50 ± 41.18	0.06 ± 0.04
Dec ’12	31.40 ± 0.52	7.01 ± 0.32	-253.10 ± 53.36	4.91 ± 0.95	2.90 ± 1.31	5.00 ± 2.26	37.26 ± 30.81	0.06 ± 0.05
Jan ’13	31.25 ± 0.42	7.11 ± 0.24	-275.20 ± 73.09	4.45 ± 0.69	2.39 ± 1.77	4.11 ± 3.05	47.70 ± 43.49	0.05 ± 0.04
Feb ’13	31.50 ± 0.85	6.96 ± 0.27	-312.20 ± 118.70	5.59 ± 1.37	2.22 ± 1.77	3.83 ± 3.06	28.74 ± 22.45	0.05 ± 0.04

Annexure 4.3 Mean seasonal variations of sediment parameters in Vembanad estuary during 2011-2012 and 2012 – 2013 period

	2011-2012 period			2012-2013 period		
Parameter	**PRM**	**MN**	**PM**	**PRM**	**MN**	**PM**
Temperature (°C)	30.48 ± 0.83	28.50 ± 2.30	26.79 ± 2.30	32.03 ± 0.85	30.21 ± 1.15	31.56 ± 0.63
pH	6.30 ± 0.78	6.50 ± 0.36	6.88 ± 0.31	7.25 ± 0.38	7.18 ± 0.29	7.05 ± 0.22
Eh (mV)	-98.78 ± 90.55	-163.83 ± 109.36	-226.15 ± 118.31	-210.15 ± 111.39	-210.45 ± 129.18	-218.30 ± 110.82
Water content (%)	6.03 ± 1.95	8.51 ± 3.06	6.64 ± 2.11	5.20 ± 1.34	5.17 ± 1.62	4.61 ± 1.10
Organic Carbon (%)	2.75 ± 2.25	3.29 ± 2.59	2.04 ± 1.60	2.98 ± 2.39	3.40 ± 2.51	2.52 ± 1.73
Organic matter (%)	4.29 ± 3.89	5.67 ± 4.46	3.78 ± 2.88	4.92 ± 3.94	5.86 ± 4.33	4.35 ± 2.98
Available Phosphorus (mg/Kg)	24.42 ± 24.43	35.64 ± 31.09	55.77 ± 62.83	31.92 ± 32.71	18.63 ± 28.66	44.35 ± 77.65
Available Nitrogen (%)	0.02 ± 0.03	0.03 ± 0.02	0.05 ± 0.04	0.04 ± 0.04	0.06 ± 0.05	0.06 ± 0.04
Total Carbon (g/Kg)	29.97 ± 25.75	30.35 ± 23.32	30.19 ± 26.62	-	-	-
Sand (%)	54.31 ± 26.42	55.83 ± 26.08	54.45 ± 28.95	48.76 ± 34.82	48.59 ± 34.86	48.59 ± 34.59
Silt (%)	26.18 ± 20.87	23.93 ± 20.27	29.79 ± 23.88	41.06 ± 27.18	45.38 ± 31.71	42.18 ± 27.71
Clay (%)	19.51 ± 10.63	20.24 ± 13.27	15.76 ± 9.37	10.18 ± 13.68	6.08 ± 4.97	9.23 ± 14.98

Annexure 4.4 ANOVA table showing the spatio- temporal variation of sediment parameters in the Vembanad estuary during 2011-2012 period

Parameters	Source	df	F	Parameters	Source	df	F
Temperature	Season	2	28.688**	Total Carbon	Season	2	0.017
	station	9	0.427		station	9	86.046**
	Season * station	18	0.11		Season * station	18	0.779
pH	Season	2	12.824**	Sodium	Season	2	2.514
	station	9	1.249		station	9	17.195**
	Season * station	18	0.89		Season * station	18	1.433
Eh (mV)	Season	2	21.272**	Potassium	Season	2	0.908
	station	9	6.432**		station	9	56.869**
	Season * station	18	1.486		Season * station	18	2.612*
Water content	Season	2	10.98*	Calcium	Season	2	0.342
	station	9	1.169		station	9	0.337
	Season * station	18	0.684		Season * station	18	2.247
Organic Carbon	Season	2	11.038**	Sand	Season	2	0.125
	station	9	29.393**		station	9	29.601**
	Season * station	18	1.959*		Season * station	18	1.565
Organic matter	Season	2	7.903**	Silt	Season	2	1.891
	station	9	25.642**		station	9	19.951**
	Season * station	18	1.571		Season * station	18	1.652
Available Phosphorus	Season	2	7.052**	Clay	Season	2	2.684
	station	9	6.004**		station	9	7.804**
	Season * station	18	0.343		Season * station	18	0.595
Available Nitrogen	Season	2	14.926**				
	station	9	11.614**				
	Season * station	18	1.17				

Degree of freedom (df); F statistic (F); ** Variation is significant at 1% level; *Variation is significant at 5% level

Annexure 4.5 ANOVA table showing the spatio- temporal variation of sediment parameters in the Vembanad estuary during 2012 -2013 period

Parameters	Source	df	F	Parameters	Source	df	F
Temperature	Season	2	43.553*	Available Nitrogen	Season	2	5.03*
	station	9	1.104		station	9	41.548
	Season * station	18	0.915		Season * station	18	0.704**
pH	Season	2	5.948 *	Sodium	Season	2	7.635**
	station	9	4.958**		station	9	12.385**
	Season * station	18	1.026		Season * station	18	0.875
Eh (mV)	Season	2	0.064	Potassium	Season	2	0.394
	station	9	2.09*		station	9	85.792**
	Season * station	18	0.666		Season * station	18	4.078**
Water content	Season	2	2.623	Calcium	Season	2	6.214*
	station	9	3.238*		station	9	61.136**
	Season * station	18	0.609		Season * station	18	4.32**
Organic Carbon	Season	2	7.702**	Sand	Season	2	0.007
	station	9	50.404**		station	9	281.025*
	Season * station	18	2.134**		Season * station	18	1.173*
Organic matter	Season	2	7.616**	Silt	Season	2	3.133*
	station	9	47.374**		station	9	152.43**
	Season * station	18	2.083		Season * station	18	3.859**
Available Phosphorus	Season	2	4.457*	Clay	Season	2	14.99**
	station	9	10.55**		station	9	113.2399**
	Season * station	18	1.284		Season * station	18	14.8939**

Degree of freedom (df); F statistic (F); ** Variation is significant at 1% level; *Variation is significant at 5% level

Annexure 4.6 Correlation matrix between sediment parameters in Vembanad estuary during 2011-2012 period

Correlations															
	Temperature	pH	Water content	Organic Carbon	clay	silt	sand	Eh	Available Nitrogen	Organic matter	potassium	Calcium	Sodium	Total Carbon	Available Phosphorus
Temperature	1														
pH	-.306**	1													
Water content	0.151	0.129	1												
Organic Carbon	0.044	-0.13	0.171	1											
Clay	0.089	-0.119	0.169	0.078	1										
Silt	-.214*	0.07	0.14	.522**	.264**	1									
Sand (%)	0.135	-0.007	-.183*	-.453**	-.631**	-.915**	1								
Eh	.365**	-.413**	-.252**	-.221*	0.051	-.284**	.208*	1							
Available Nitrogen	-.371**	.289**	0.109	.407**	0.053	.501**	-.425**	-.553**	1						
Organic matter	0.026	-0.15	0.174	.944**	0.068	.489**	-.422**	-.266**	.394**	1					
Potassium	0.071	0.135	.198*	0.099	.490**	.365**	-.499**	0.051	-0.031	0.097	1				
Calcium	0.065	0.174	.215*	.234*	.468**	.404**	-.521**	-0.02	0.018	.246**	.801**	1			
Sodium	0.069	.188*	0.092	0.02	.371**	.298**	-.394**	0.133	-0.012	-0.006	.694**	.217*	1		
Total Carbon	-0.109	-0.131	.194*	.762**	0.125	.596**	-.532**	-.432**	.598**	.769**	0.051	-0.034	-0.01	1	
Available Phosphorus	-.336**	.236**	0.156	.337**	.190*	.489**	-.473**	-.301**	.425**	.300**	.423**	-0.034	.337**	.422**	1

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

Annexure 4.7 Correlation matrix between sediment parameters in Vembanad estuary during 2012-2013 period

	Temperature	pH	Water content	Organic Carbon	clay	silt	sand	Eh	Available Nitrogen	Organic matter	potassium	Calcium	Sodium	Available Phosphorus
Temperature	1													
pH	.204*	1												
Water content	-0.044	-0.095	1											
Organic Carbon	-0.17	-0.162	.354**	1										
Clay	0.04	0.093	0.163	.325**	1									
Silt	-.221*	-.270**	.391**	.640**	.309**	1								
Sand	0.17	.193*	-.384**	-.648**	-.609**	-.943**	1							
Eh	0.099	.296**	-0.147	-.186*	0.059	-.216*	0.16	1						
Available Nitrogen	-0.116	-.307**	.263**	.700**	0.168	.608**	-.566**	-.382**	1					
Organic matter	-.204*	-0.172	.319**	1.000**	.330**	.637**	-.648**	-.190*	.685**	1				
Potassium	-0.085	0.112	0.104	0.127	.669**	.286**	-.474**	0.071	-0.056	0.136	1			
Calcium	-0.172	0.045	0.15	.199*	.583**	.384**	-.525**	0	0.049	.204*	.907**	1		
Sodium	0.12	0.048	0.123	0.04	.684**	0.093	-.318**	-0.111	-0.04	0.044	.699**	.620**	1	
Available Phosphorus	-0.008	-0.129	0.033	.396**	.332**	.361**	-.418**	-.249**	.483**	.396**	.257**	.260**	.244**	1

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

Annexure 4.8 Comparison of sediment parameters on south and north of TMB during 2011-2012 and 2012-2013 period

Parameters	2011-2012		2012-2013	
	South of TMB	North of TMB	South of TMB	North of TMB
Temperature (°C)	28.33 ± 2.39	28.93 ± 2.52	31.15 ± 1.10	31.43 ± 1.28
pH	6.50 ± 0.67	6.65 ± 0.39	7.08 ± 0.32	7.27 ± 0.26
Eh (mV)	-193.03 ± 131.24	-117.75 ± 75.87	-232.29 ± 121.30	-183.98 ± 103.50
Water content (%)	7.20 ± 2.51	6.86 ± 2.80	5.23 ± 1.44	4.64 ± 1.21
Organic Carbon (%)	3.20 ± 2.34	1.93 ± 1.81	3.46 ± 2.48	2.22 ± 1.61
Organic matter (%)	5.47 ± 4.04	3.25 ± 3.15	5.87 ± 4.20	3.83 ± 2.78
Available Phosphorus (mg/Kg)	39.19 ± 41.30	37.73 ± 49.24	34.70 ± 62.02	27.02 ± 31.75
Available Nitrogen (%)	0.04 ± 0.04	0.02 ± 0.01	0.07 ± 0.05	0.03 ± 0.02
Total Carbon (g/Kg)	37.50 ± 29.07	19.17 ± 10.34	-	-
Sand (%)	49.43 ± 22.52	63.01 ± 31.00	36.05 ± 24.69	67.54 ± 38.46
Silt (%)	31.06 ± 20.80	20.00 ± 21.50	57.01 ± 20.90	21.67 ± 25.84
Clay (%)	19.51 ± 8.97	16.98 ± 14.03	6.97 ± 6.12	10.79 ± 17.45

Annexure 5.1 Spatial variation of mean macrofaunal groups (ind./m^2) during 2011-2012 period in Vembanad estuary

Marofaunal groups	S1	S2	S3	S4	S5	S6	S7	S8	S9	S10
Hydrozoans	0	0	0	0	0	0	0	0	0	516
Nemerteans	23	0	0	0	0	0	0	0	0	0
Turbellarians	6	6	4	23	4	8	17	26	17	19
Nematodes	0	4	0	9	0	4	9	0	0	26
Oligochaetes	388	64	19	98	62	0	8	2	0	64
Polychaetes	783	3532	1438	1795	507	496	815	568	2418	4230
Insects	87	30	32	0	6	9	49	11	2	0
Ostracods	0	0	0	0	0	0	0	0	0	15
Harpacticoids	0	0	0	0	13	0	6	0	0	26
Amphipods	13	26	902	138	13	40	274	85	100	384
Cumaceans	0	0	4	4	0	0	44	6	0	0
Tanaids	17	0	0	272	9	15	159	110	2	70
Decapods	0	0	0	0	0	0	13	2	15	83
Mysids	0	0	0	0	0	0	9	0	0	11
Isopods	17	28	199	866	13	47	549	204	288	51
Bivalves	240	15	32	59	44	108	265	157	49	3360
Gastropods	0	0	0	0	0	0	0	0	13	0
Fishes	0	0	0	0	0	0	0	0	0	6

Annexure 5.2 Spatial variation of mean macrofaunal groups (ind./m^2) in Vembanad estuary during 2012 - 2013 period

Marofaunal groups	S1	S2	S3	S4	S5	S6	S7	S8	S9	S10
Hydrozoans	0	0	0	0	0	0	0	0	0	57
Nemerteans	0	0	4	0	0	0	0	0	0	0
Turbellarians	0	0	6	2	4	4	9	2	21	0
Nematodes	0	0	8	8	2	4	11	0	0	13
Oligochaetes	916	880	42	102	0	0	0	0	0	40
Polychaetes	1793	2571	1035	323	236	159	384	115	115	6952
Insects	55	49	32	83	38	42	40	61	57	2
Ostracods	0	0	0	0	0	0	0	0	0	4
Harpacticoids	0	0	0	0	0	0	0	0	0	19
Amphipods	931	3292	4254	532	87	176	295	255	734	327
Cumaceans	0	0	0	0	0	0	0	0	0	0
Tanaids	8	23	4	79	64	165	115	255	221	57
Decapods	2	0	0	0	0	0	2	0	9	68
Mysids	0	0	6	0	2	13	4	2	0	4
Isopods	0	57	910	3153	263	420	620	496	1090	125
Bivalves	4	26	30	131	53	30	1999	2189	12640	2957
Gastropods	0	0	4	0	0	0	28	9	53	0
Fishes	0	0	0	0	0	0	0	0	0	4

Annexure 5.3 ANOVA table showing the spatio- temporal variation of macrofauna in the Vembanad estuary during 2011-2012 and 2012-2013 period

		2011-2012	2012-2013			2011-2012	2012-2013
Source	**df**	**F**	**F**	**Source**	**df**	**F**	**F**
Polychaetes				**Nematodes**			
Season	2	0.668	1.087	Season	2	0.695	1.983
station	9	4.148**	6.557**	station	9	1.785	0.91
Season x station	18	1.71	3.613**	Season x station	18	1.559	1.376
Oligochaetes				**Turbellarians**			
Season	2	0.621	0.696	Season	2	1.334	0.176
Station	9	5.820**	4.965**	station	9	0.575	2.871**
Seas x station	18	0.982	0.861	Season x station	18	1.006	0.905
Amphipods				**Harpacticoid copepods**			
Season	2	3.241*	0.4	Season	2	2.431	2.214
Station	9	3.609**	4.357*	station	9	3.837**	7.143**
Season x station	18	3.463**	1.118	Season x station	18	3.346**	2.214**
Tanaids				**Ostracods**			
Season	2	0.573	0.704	Season	2	3.500*	
Station	9	2.151*	2.092*	station	9	8.000**	
Seas x station	18	1.088	0.547	Season x station	18	3.500**	
Isopods				**Nemerteans**			
Season	2	0.136	1.056	Season	2	0.839	1
station	9	4.199*	3.824*	station	9	1.339	1
Season x station	18	1.658	1.261	Season x station	18	0.829	1
Insects				**Mysids**			
Season	2	0.342	0.347	Season	2	2.893	1.981
Station	9	1.795	0.344	station	9	2.365*	0.906
Season x station	18	0.816	1.25	Season x station	18	1.306	0.72
Bivalves				**Gastrapods**			
Season	2	1.166	0.148	Season	2	0.677	0.656
station	9	1.264	5.407	station	9	4.742**	6.314*
Season x station	18	0.813	0.949	Season x station	18	0.677	0.48
Decapods				**Fishes**			
Season	2	5.992**	4.496*	Season	2	1.286	3
station	9	5.523**	11.191**	station	9	3.857**	3.000*
Season x station	18	4.828**	4.574*	Season x station	18	1.286	3.000**

(Degree of freedom (d.f); F statistic (F); ** Variation is significant at 1% level; *Variation is significant at 5% level

Annexure 5.4 Mean spatial variation of macrofaunal biomass in Vembanad estuary during 2011-2012 period

	S1	S2	S3	S4	S5	S6	S7	S8	S9	S10
Turbellarians	0.01 ± 0.03	0.00±0.01	0	0.35 ± 1.21	0.01 ± 0.04	0.02 ± 0.05	0.27 ± 0.93	0.00 ± 0.01	0.01 ± 0.03	0.00 ± 0.01
Nematodes	0.01 ± 0.02	0	0	0.03 ± 0.10	0	0	0	0	0	0.04 ± 0.13
Oligochaetes	0.90 ± 1.78	0.30 ± 0.71	0.74 ± 2.57	0.25 ± 0.65	0.00 ± 0.01	0.39 ± 1.37	0	0	0	0.10 ± 0.18
Polychaetes	0.70 ± 0.66	11.02 ± 14.22	2.18 ± 1.71	2.33 ± 2.59	1.57 ± 2.83	0.72 ± 1.06	1.27 ± 1.65	0.64 ± 1.11	2.47 ± 2.66	15.63 ± 24.91
Insects	0.10 ± 0.14	0.02 ± 0.04	0.01 ± 0.02	0	0.00 ± 0.01	0.01 ± 0.02	0	0	0	0
Harpacticoid copepods	0	0	0.01 ± 0.03	0	0	0	0.01 ± 0.02	0.01 ± 0.02	0.01 ± 0.04	0.18 ± 0.63
Amphipods	0.00 ± 0.01	0.01 ± 0.02	0.26 ± 0.59	0.42 ± 0.95	0.03 ± 0.09	0.04 ± 0.07	0.11 ± 0.26	0.06 ± 0.12	0.04 ± 0.07	0.47 ± 0.76
Cumaceans	0	0	0	0.03 ± 0.11	0	0.00 ± 0.01	0	0.19 ± 0.63	0	0
Tanaids	0	0	0	0.33 ± 0.45	0.00 ± 0.00	0.02 ± 0.06	0.09 ± 0.18	0.23 ± 0.63	0.00 ± 0.00	0.02 ± 0.06
Decapods	0	0	0	0	0	0	0	0	0.32 ± 0.74	0.65 ± 1.33
Mysids	0.02 ± 0.08	0	0	0.03 ± 0.06	0	0	0.05 ± 0.18	0	0.01 ± 0.03	0
Isopods	0	0.22 ± 0.67	0.02 ± 0.04	0.25 ± 0.44	0	0.01 ± 0.01	0.56 ± 1.41	0.04 ± 0.12	0.07 ± 0.11	0.05 ± 0.09
Bivalves	0.34 ± 1.19	0.99 ± 3.36	1.52 ± 2.62	19.04 ± 62.86	27.94 ± 86.40	3.50 ± 6.94	42.79 ± 107.91	46.35 ± 86.90	3.27 ± 8.20	14.89 ± 38.31
Gastropods	0	0	0	0	0	0	0	0	1.41 ± 3.89	0.00 ± 0.00
Fishes	0	0	0	0	0	0	0	0	0	0.34 ± 0.67

Annexure 5.5 Mean spatial variation of macrofaunal biomass in Vembanad estuary during 2012-2013 period

	S1	S2	S3	S4	S5	S6	S7	S8	S9	S10
Oligochaetes	0.58 ± 0.64	2.40 ± 7.12	0.03 ± 0.10	0.13 ± 0.47	0	0	0	0	0	0.02 ± 0.06
Polychaetes	4.63 ± 3.98	31.08 ± 39.70	3.95 ± 4.53	3.96 ± 7.28	1.50 ± 1.72	1.94 ± 3.50	3.51 ± 5.16	1.28 ± 1.33	6.01 ± 8.07	23.50 ± 32.10
Insects	0.15 ± 0.41	0.07 ± 0.16	0.04 ± 0.12	0.04 ± 0.12	0.01 ± 0.02	0.00 ± 0.00	0.05 ± 0.11	0.07 ± 0.25	0.12 ± 0.36	0
Amphipods	0.40 ± 0.56	1.26 ± 1.19	4.72 ± 9.33	0.17 ± 0.29	0.06 ± 0.14	0.13 ± 0.24	0.14 ± 0.10	0.31 ± 0.56	0.44 ± 0.37	0.50 ± 1.15
Tanaids	0.01 ± 0.03	0.10 ± 0.32	0.02 ± 0.07	0.03 ± 0.07	0.14 ± 0.34	0.28 ± 0.39	0.10 ± 0.24	0.15 ± 0.27	0.19 ± 0.36	0.01 ± 0.02
Decapods	0.01 ± 0.04	0	0	0	0	0	0	0	0.01 ± 0.03	0.23 ± 0.41
Isopods	0	0.13 ± 0.44	0.64 ± 1.24	1.82 ± 2.49	0.22 ± 0.39	0.26 ± 0.46	0.38 ± 0.72	0.25 ± 0.43	1.07 ± 3.67	0.18 ± 0.39
Bivalves	0	17.78 ± 30.07	55.85 ± 129.97	166.41 ± 253.68	42.92 ± 61.31	41.96 ± 97.65	143.52 ± 257.02	112.37 ± 160.45	153.09 ± 128.36	56.83 ± 80.46
Gastropods	0	0	1.69 ± 5.86	0	0	0	8.96 ± 24.39	0.19 ± 0.66	3.91 ± 5.88	0
Fishes	0	0	0	0	0	0	0	0	0	0.14 ± 0.39

Annexure 5.6 ANOVA table showing the spatio- temporal variation of macrofaunal biomass in the Vembanad estuary during 2011-2012 and 2012-2013 period

2011-2012				2012-2013	
Source	**df**	**Mean Square**	**F**	**Mean Square**	**F**
Polychaetes					
Season	2	51.276	0.668	213.375	1.087
station	9	318.594	4.148**	1286.928	6.557**
Season x station	18	131.363	1.71	709.119	3.613**
Insects					
Season	2	0.005	2.678	0.027	0.653
Station	9	0.011	5.372**	0.031	0.745
Season x station	18	0.003	1.647	0.049	1.181
Tanaids					
Season	2	0.108	1.726	0.045	0.703
Station	9	0.162	2.601*	0.093	1.459
Season x station	18	0.064	1.018	0.065	1.011
Decapods					
Season	2	0.828	4.457*	0.029	2.08
station	9	0.574	3.091*	0.064	4.523**
Season x station	18	0.397	2.136*	0.031	2.210**
Amphipods					
Season	2	0.124	0.635	8.62	0.929
Station	9	0.371	1.906	24.026	2.590*
Season x station	18	0.194	0.998	8.061	0.869
Bivalves					
Season	2	8466.97	2.699	38397.53	1.948
Station	9	3720.01	1.186	42796.44	2.171*
Season x station	18	3060.75	0.976	25626.95	1.3
Fishes					
Season	2	0.04	0.877	0.023	1.76
station	9	0.139	3.058*	0.023	1.76
Season x station	18	0.04	0.877	0.023	1.760*

(Degree of freedom (d.f), F statistic); ** Variation is significant at 1% level; *Variation is significant at 5% level

Annexure 6.1 Mean spatial abundance of polychaetes (ind./m^2) during 2011-2012 period in Vembanad estuary

Sl. No.	Polychaete species (2011-2012)	S1	S2	S3	S4	S5	S6	S7	S8	S9	S10
	Pilargidae										
1	*Sigambra parva* (Day, 1963)	0	0	0	0	0	0	0	0	25	49
	Nereididae										
2	*Namalycastis indica* (Day, 1963)	380	3131	732	433	127	45	178	21	11	2
3	*Dendronereis aestuarina* (Southern, 1921)	30	15	195	85	149	98	110	68	125	15
4	*Nereis* sp. (Linnaeus, 1758)	0	0	0	0	0	0	0	2	0	23
5	*Ceratonereis* sp. (Kinberg, 1865)	0	0	0	0	0	0	0	2	2	15
	Nephtyidae										
6	*Micronephthys oligobranchia* (Southern, 1921)	0	0	0	0	0	0	0	0	11	363
	Glyceridae										
7	*Glycera alba* (O F Muller, 1776)	0	0	0	6	2	0	0	0	11	25
	Onuphidae										
8	*Diopatra neapolitana* (Delle Chiaje, 1841)	0	0	0	0	0	0	0	0	0	4137
	Lumbrineridae										
9	*Scoletoma impatiens* (Claparede, 1868)	0	0	0	0	0	0	0	0	2	74
	Spionidae										
10	*Dipolydora flava* (Claparede, 1870)	0	0	0	0	0	0	0	0	0	44
11	*Prionospio cirrifera* (Wirén, 1883)	0	0	0	0	25	242	153	176	1487	454
12	*Paraprionospio pinnata* (Ehlers, 1901)	0	0	0	0	0	0	0	0	0	62
	Cirratulidae										
13	*Aphelochaeta filiformis* (Keferstein, 1862)	0	0	0	0	0	0	0	0	2	30
	Cossuridae										
14	*Cossura coasta* (Kitamori, 1960)	0	0	0	0	0	0	0	0	2	28
	Capitellidae										
15	*Mediomastus* sp. (Hartman, 1944)	0	0	0	28	0	0	0	0	0	2
16	*Parheteromastus tenuis* (Monro, 1937)	0	0	0	6	0	2	0	47	643	363
	Oweniidae										
17	*Owenia* sp. (Delle Chiaje 1841)	0	0	0	138	17	11	4	6	2	2
	Pectinariidae										
18	*Pectinaria* sp. (Lamarck, 1818)	0	0	0	0	0	0	0	0	2	28
	Sabellariidae										
19	*Sabellaria pectinata* (Fauvel, 1932)	0	0	0	0	0	0	0	0	25	53

Annexure 6.2 Mean spatial abundance of polychaetes (ind./m^2) during 2012-2013 period in Vembanad estuary

Sl No.	Polychaete species (2012-2013)	S1	S2	S3	S4	S5	S6	S7	S8	S9	S10
	Pilargidae										
1	*Sigambra parva* (Day, 1963)	0	0	0	0	0	0	0	0	0	176
	Nereididae										
2	*Namalycastis indica* (Day, 1963)	1443	2351	847	199	57	4	4	0	0	0
3	*Dendronereis aestuarina* (Southern, 1921)	89	219	187	98	180	153	371	132	44	74
	Nephtyidae										
4	*Micronephthys oligobranchia* (Southern, 1921)	0	0	0	0	0	0	0	0	2	450
	Glyceridae										
5	*Glycera alba* (O F Muller, 1776)	0	0	0	0	0	0	0	0	6	23
	Onuphidae										
6	*Diopatra neapolitana* (Delle Chiaje, 1841)	0	0	0	0	0	0	0	0	0	6042
	Lumbrineridae										
7	*Scoletoma impatiens* (Claparede, 1868)	0	0	0	0	0	0	0	0	4	53
	Spionidae										
8	*Dipolydora flava* (Claparede, 1870)	0	0	0	0	0	0	0	0	0	17
9	*Prionospio cirrifera* (Wirén, 1883)	0	0	0	0	424	399	1069	274	1347	846
10	*Paraprionospio pinnata* (Ehlers, 1901)	0	0	0	0	0	0	0	0	0	85
	Cossuridae										
11	*Cossura coasta* (Kitamori, 1960)	0	0	0	0	0	0	15	0	0	110
	Capitellidae										
12	*Parheteromastus tenuis* (Monro, 1937)	0	0	0	0	0	8	4	85	554	819
	Oweniidae										
13	*Owenia* sp. (Delle Chiaje 1841)	0	0	0	26	0	0	0	0	9	0
	Sabellariidae										
14	*Sabellaria pectinata* (Fauvel, 1932)	0	0	0	0	0	0	0	0	4	2

Annexure 6.3 Correlation matrix between polychaete species and environmental parameters in Vembanad estuary during 2011-2012 period

Correlations										
	Salinity	**Dissolved oxygen**	**Sediment temperature**	**Sediment pH**	**Moisture content**	**Organic carbon**	**Clay**	**Silt**	**Sand**	**Eh**
Dendronereis aestuarina	-.055	.173	.185*	-.230*	-.117	-.144	-.100	-.173	.179	.168
Namalycastis indica	-.158	-.164	-.173	.051	.084	.306**	.039	.186*	-.165	-.217*
Ceratonereis sp.	**.505****	-.063	-.029	.167	.173	-.008	.245**	.301**	-.345**	-.043
Micronephthys oligobranchia	**.444****	-.022	.102	.109	-.052	-.042	.356**	.224*	-.330**	.147
Parheteromastus tenuis	**.473****	-.069	.060	.097	-.068	-.160	-.022	-.099	.095	.111
Mediomastus sp.	-.051	-.009	-.224*	.043	-.027	-.040	-.025	.121	-.086	-.069
Sigambra parva	**.550****	-.063	-.031	.110	.042	.000	.285**	.106	-.203*	-.051
Prionospio cirrifera	.347**	-.126	.072	.045	-.068	-.170	-.089	-.113	.133	.129
Paraprionospio pinnata	.329**	-.063	.096	.134	.182*	.030	.167	.157	-.195*	.041
Dipolydora flava	**.477****	-.055	-.013	.176	.015	-.065	.303**	.262**	-.338**	.009
Glycera alba	**.491****	-.092	.064	.183*	.037	-.122	.114	.080	-.109	.129
Cossura coasta	**.539****	-.050	.026	.158	.232*	-.026	.359**	.252**	-.353**	-.005
Aphelochaeta filiformis	**.439****	-.040	-.136	.134	.040	.020	.241**	.213*	-.271**	-.077
Scoletoma impatiens	**.631****	-.052	-.011	.200*	.177	.000	.346**	.254**	-.349**	-.006
Sabellaria pectinata	**.455****	-.080	.076	.126	.007	-.113	.117	.182*	-.194*	.057
Owenia sp.	-.079	-.350**	-.176	-.018	-.082	-.091	.031	.030	-.051	-.139
Pectinaria sp.	.306**	-.071	.023	.089	.089	.026	.117	.214*	-.222*	-.043
Diopatra neapolitana	**.373****	-.007	-.137	.171	.070	-.055	.153	.272**	-.283**	.016
Nereis sp.	.318**	-.006	-.059	.179	.075	-.037	.198*	.313**	-.335**	.015

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

Annexure 6.4 Correlation matrix between polychaete species and environmental parameters in Vembanad estuary during 2012-2013 period

Correlations										
	Salinity	**Dissolved oxygen**	**Sediment temperature**	**Sediment pH**	**Organic carbon**	**Clay**	**Silt**	**Sand**	**Eh**	**Aailable nitrogen**
Micronephthys oligobranchia	.291**	-0.1	-0.168	0.107	0.07	**.496****	0.174	-.320**	0.072	-0.081
Parheteromastus tenuis	**.557****	-0.16	0.052	.213*	-0.063	**.473****	-0.106	-0.078	0.004	-0.152
Sigambra parva	.199*	-0.12	-0.153	0.041	0.023	.386**	0.152	-.262**	0.07	-0.084
Prionospio cirrifera	**.426****	-0.07	.248**	.270**	-.231*	0.07	-.354**	.273**	-0.033	-.269**
Paraprionospio pinnata	.307**	-0.04	0.165	0.177	-0.116	.192*	-0.17	0.075	0.176	-0.13
Dipolydora flava	.407**	-.216*	-0.006	0.13	0.032	**.510****	0.068	-.235**	-0.118	-0.033
Glycera alba	**.528****	-.236**	0.051	0.177	-0.008	**.636****	-0.054	-0.178	-0.004	-0.055
Cossura coasta	-0.012	0.021	-0.161	-0.03	0.006	.186*	0.108	-0.156	0.117	-0.069
Scoletoma impatiens	**.482****	-.268**	0.035	0.166	0.026	**.710****	0.047	-.289**	0.001	-0.081
Sabellaria pectinata	0.096	-0.09	0.046	0.083	-0.042	0.161	-0.141	0.061	0.137	-0.09
Owenia sp.	-0.008	-0.04	-0.152	-0.038	-0.021	-0.056	0.045	-0.018	-.207*	0.084
Diopatra neapolitana	**.464****	-.260**	0.024	0.155	0.043	**.602****	0.06	-.261**	-0.076	-0.052
Dendronereis aestuarina	-0.108	0.026	0.109	0.027	-0.067	-0.066	-0.066	0.079	-0.004	-0.022
Namalycastis indica	-.277**	0.098	-0.145	-.240**	**.487****	0.092	**.475****	-.428**	-.229*	**.599****

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

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List of Publications

Paper Publications (Peer Reviewed)

[1] **Asha C V**., Retina I C., Suson P. S and Bijoy Nandan S., 2016. Ecosystem analysis of the degrading Vembanad wetland ecosystem, the largest Ramsar site on south west coast of India – measures for its sustainable management, *Regional studies in Marine Science*., 8, 408 – 421. Elsevier Publishers.

[2] Cleetus R I., **Asha C V.,** Suson P S and Bijoy Nandan S., 2016. Mesozooplankton Abundance and Community Structure in Vembanad-Kol Wetland Ecosystem, Kerala, India, *Indian J. Geo-Mar. Sci*., 45:4, 533-545. Publisher, National Institute of Science Communication and Information Resources (NISCAIR)

[3] Jayachandran P. R., Bijoy Nandan, S., Sanu, V. F., Jima, M., Anu, P. R., Don Xavier, N. D., Joseph, P., Midhun, A.M and **Asha C.V. 2016**. Authentication of Nassodonta insignis H. Adams, 1867 (Gastropoda: Nassariidae) from the Kodungallur - Azhikode backwater, southwest coast of India using mitochondrial COI marker. *Indian J. Geo-Mar. Sci*., Ref:IJMS/MS3413. Publisher, NISCAIR

[4] **Asha C V**., Cleetus R I., Suson P S and Bijoy Nandan S., 2015, Environmental factors structuring the fish assemblage distribution and production potential in Vembanad estuarine system, India, *International Journal of Marine Science*, Vol.5, No.23 1-13, BioPublisher.

[5] **Asha C V**., Suson P S., Retina C I and Bijoy Nandan S., 2014. Decline in Diversity and Production of Exploited Fishery Resources in Vembanad Wetland System: Strategies for Better Management and Conservation. *Open Journal of Marine Science*, 4, 344-357.Scientific Research Publisher.

Book Chapter

[1] **Asha C V**., Cleetus R I., Suson P S and Bijoy Nandan S., 2015, Tubiculous Worm Aggregates in A Tropical Estuary, Its Importance in Conservation of Wetland in Ecosystems. *In Biodiversity and Evaluation: In proceedings of International Conference on Perspectives and Paradigm shifts* (eds.) S. Bijoy Nandan, Sampath K. S, Mini K D, Revathy Babu. Held from 2nd to 3rd of December 2015. Published by Cochin University of Science and Technology, Cochin and Sree Sankara College, Kalady.

First pages of each publications are enclosed:

JOURNAL PUBLICATIONS

Regional Studies in Marine Science 8 (2016) 408–421

Contents lists available at ScienceDirect

Regional Studies in Marine Science

journal homepage: www.elsevier.com/locate/rsma

Ecosystem analysis of the degrading Vembanad wetland ecosystem, the largest Ramsar site on the South West Coast of India — Measures for its sustainable management

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HIGHLIGHTS

- The study presents an over view on a degrading coastal wetland system of India and signifies the need for its conservation.
- Thanneermukkom Barrage plays a crucial role in influencing the salinity pattern of Vembanad estuarine system.
- Trophic index (TRIX value) indicated an impacted water quality condition leading to an accelerated eutrophication in the estuarine system.
- Decline in fishery diversity and production reflects considerable ecological impacts on lower trophic levels.

ARTICLE INFO

Article history:
Received 15 December 2015
Received in revised form
15 June 2016
Accepted 16 June 2016
Available online 23 June 2016

Keywords:
Vembanad estuary
Ramsar site
Thanneermukkom barrage
Ecological degradation

ABSTRACT

Vembanad wetland ecosystem — the largest Ramsar site on the South West Coast of India, located in the most populous coastal segments of Kerala, serves as the rice bowl of the state that forms a major hotspot of biological diversity. Alterations in natural hydrologic regime of Vembanad wetland started with the commissioning of Thanneermukkom barrage (TMB) across the backwater system in 1976, to prevent saline water intrusion, which adversely affected the eco-biology of the water body. The study was conducted during March 2011 to February 2012 on a monthly basis to evaluate the eco-hydrological status of this multifunctional ecosystem. Reclamation of estuarine areas for agriculture and other interventions has led to drastic decline in water holding capacity (2.4 to 0.6 km^3) and depth (av. 4.02 m) of the estuary over the years. Water quality on an average reflects a neutral (7.11) and well oxygenated (7.4 mg L^{-1}) nature; the salinity varying from 0.01 to 31.8 ppt. TMB plays a crucial role in influencing the salinity pattern of Vembanad wetland; with an oligohaline (0.5–5 ppt) in southern whereas meso and polyhaline (5–18 ppt) condition prevailing in northern stations. The estuary is nutrient rich with nitrate-nitrogen and phosphate-phosphorus ranging between 0.05–5.9 μmol L^{-1} and 0.18–6.49 μmol L^{-1} respectively. Trophic index (TRIX value 6.39 $\pm$ 0.75) indicated an impacted water quality condition leading to an accelerated eutrophication in the system. Bacillariophyceae (83.55%), Chlorophyceae (8.98%), Cyanophyceae (6.92%) and Zygnemophyceae (0.55%) were the major phytoplankton. Zooplankton community was composed of calanoids (63%), rotifers (22.5%), copepodites (2%), amphipoda (2%), crustacean naupli (1.5%) and cladocerans (1.2%) showing a declining trend in post barrage phase. Benthic polychaetes like Spionidae and Capitellidae were opportunistic bio-indicators of organic enrichment. Decline in fishery diversity and production (4387.31 t), reflects considerable ecological impacts on lower trophic levels. Thus the study presents an over view on a degrading coastal wetland system of India and signifies the need for enhanced restoration programmes for long term management and conservation objectives.

1. Introduction

Estuaries are facing threats of degradation from multiple human impacts such as sewage discharge, pollution from pesticides, fertilizers and municipal wastes, conversion of land for agriculture, land reclamation for tourism activities etc. As the carbon flux

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http://dx.doi.org/10.1016/j.rsma.2016.06.003
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Indian Journal of Geo-Marine Sciences
Vol. 45(4), April 2016, pp. 533-545

Mesozooplankton abundance and community structure in vembanad-kol wetland ecosystem, kerala, India

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Received 01 June 2015; revised 16 July 2015

Abundance and community structure of mesozooplankton were studied (March 2012-February 2013) in the context of prevailing environmental parameters in the Vembanad estuarine system, a Ramsar site on the southwest coast of India. Of the sixteen taxa of mesozooplankton, 16 calanoid copepods and 26 rotifers were identified upto species level. Copepods was the most dominant taxon where calanoids (63%), followed by rotifers (23%), copepodites (2 %), amphipodes (2%), crusteacean naupli (1.5 %) and others contributing in minor proportions. Freshwater diaptomid species, including *Heliodiaptomus cintus* and *Allodiaptomus mirabilipes* contributed around 52 % of calanoids in the southern stations (1, 2 and 3) and *Bestiolinas imilis* (19%) and *Acartia southwelli* (12.3%) formed the major share of calanoid copepods in northern stations (8, 9 & 10). Seasonal distribution of zooplankton showed bimodality, with a primary peak in premonsoon (av. 25073 ind. m^{-3}) followed by secondary peak in post monsoon (av. 21093ind. m^{-3}) coinciding with maximum of copepod species. Multi-Dimensional Scaling analysis reveal 80% similarity in abundance of plankton between southern (1 to 7) and northern stations (8, 9 and 10). Phosphate, nitrite, temperature, dissolved oxygen, and salinity best determined the abundance and distribution of mesozooplankton in Vembanad estuary based on multivariate BIOENV analysis.

[**Key words**: Meszooplankton, abundance, distribution, Vembanad estuary, copepods, rotifers]

Introduction

Mesozooplankton contribute significantly to biological production and are important contributors and modifiers of vertical flux of organic matter to depth and play a crucial role in modeling the carbon dioxide dynamics of the system[1]. The relationship between the composition and abundance of zooplankton and the trophic state of lakes has been studied in both temperate [2,3,4&5] and tropical ecosystems[6&7]. In most subtropical and tropical part of the world's oceans copepods dominate the mesozooplankton communities[8], where they comprise about 75% of total abundances[9&10] with strong seasonality in accordance with the changes in salinity[11&12]. Subtropical and tropical regions are also known to be colonized by diverse groups of species[13]. As zooplankton is at the base of the aquatic ecosystem, understanding their distribution and abundance in space and time with prevailing hydrographic parameters is inevitable for understanding and modelling community dynamics, and is essential to understand physical and biological processes that structure marine ecosystems[14]. Recent information relating to mesozoopankton abundance, distribution and community structure in Vembanad estuary extending from south (Alappuzha) to (Cochin) north is very scarce and are limited to the prebarrage phase of Thanneermukkom barrage (TMB). Construction of TMB in 1976, to prevent the saline intrusion during summer months, so as to make possible paddy cultivation in the agrarian zone of Kuttanad, interfered the natural cleansing mechanism of the estuary along with accelerated loss of habitats and biodiversity[15]. Even though some information on the zooplankton ecology and abundance pattern are reported from the south of the estuary during the post barrage phase, a comprehensive study on mesozooplankton involving both southern and northern parts of the estuarine system is severely lacking. Hence the present study has been undertaken in the Vembanad estuarine system to understand the spatio-temporal distribution, abundance and community structure of mesozooplankton (MZ; 20-200 μm) with special reference to calanoid copepods and rotifers of the estuary.

Investigations on distribution and diversity of plankton in Vembanad backwater were made on south of the barrage[16,17&18]. Zooplankton composition and abundance in Cochin estuary

Indian Journal of Geo-Marine Sciences (IJMS) Ref: IJMS/MS 3413 Date: 09/12/2016

Authentication of *Nassodonta insignis* H. Adams, 1867 (Gastropoda: Nassariidae) from the Kodungallur - Azhikode backwater, southwest coast of India using mitochondrial COI marker

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Abstract

The genus, *Nassodonta* in the family Nassariidae comprises two accepted species; *Nassodonta insignis* and *Nassodonta dorri*. The type locality of *N. insignis* was indicated by Adams as "Peiho river, China . Smith (1895), Preston (1916) and Cernohorsky (1984) confirmed the occurrence of *N. insignis* in Indian brackish waters. The locality in China has been ignored because of lack of confirming data. Cernohorsky (1984) synonymized *Nassodonta gravelyi* described from Ennur (Ennore) backwater, Madras as *N. insignis*. The present study identifies *N. insignis* from brackishwater environments of Kodungallur - Azhikode backwater and confirms the occurrence of the species in these localities with molecular data. There are no molecular data available for the other species in the genus: *N. dorri*. Mitochondrial DNA COI sequences (KT985460, KT985461, KT985462, KT985463, KT985464) with 663 base pair length were developed for *N. insignis*. GenBank BLAST analysis shows 89% similarity with *Nassarius (Varicinassa) variciferus* and 44% bootstrap value in the Maximum Likelihood tree analysis.

Key words. Gastropoda, Nassariidae, *Nassodonta*, Cochin (Kochi) - Kodungallur - Azhikode backwaters, India.

International Journal of Marine Science 2015, Vol.5, No.23 1-13
http://ijms.biopublisher.ca

Research Article Open Access

Environmental factors structuring the Fish assemblage distribution and Production potential in Vembanad estuarine system, India

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International Journal of Marine Science, 2015, Vol.5, No.23 doi: 10.5376/ijms.2015.05.0023
Received: 03 Jan., 2015
Accepted: 10 Mar., 2015
Published: 09 Apr., 2015

Preferred citation for this article:
Asha et al., 2015, Environmental factors structuring the fish assemblage distribution and production potential in Vembanad estuarine system, India, International Journal of Marine Science, Vol.5, No.23 1-13 (doi: 10.5376/ijms.2015.05.0023)

Abstract Vembanad wetland system, a Ramsar site on the West coast of India is the largest estuarine system, renowned for its rich fishery diversity and endemicity. In 1976, to regulate salinity intrusion into the Vembanad estuarine system, the Thannermukkom barrage was constructed across the estuary dividing it into fresh water dominated southern and a northern region dominated with brackish water, which has grossly altered the eco-biology of the region. In this context, the influence of environmental factors on fish assemblage structure has been poorly recognised in Vembanad estuarine system mainly for effectively implementing proper management and conservation measures. Information on fish assemblage in relation to environmental parameters collected from February 2012 to January 2013 is reported in this contribution. Estimates of annual average fishery production indicated a catch of 4774.46 t in the estuary, in which 10.1% and 89.9% was contributed by southern and northern zones respectively. Seventy four species of finfishes, eleven species of shell fishes were identified for the study period. Multidimensional scaling and canonical correspondence analysis significantly demarcated the spatio – temporal variation of fish species distribution and observed a strong correspondence between fish assemblage and environmental variables. Depth, temperature, pH and salinity were the most important parameters explaining the variation in fish assemblage composition and abundance in Vembanad estuarine system. The aging Thaneermukkom barrage and its faulty operation reduced the tidal flushing altering the circulation and mixing process, especially in the southern part of estuary. This influenced the hydrological characteristics leading to a decline in estuarine fishery and its production potential.
Keywords Environmental factors; Fish assemblage; Ramsar site; Vembanad wetland system

Introduction

In an aquatic environment, understanding the dynamics of fish population and species distribution in response to environmental factors are essential. Variation in the abundance and composition of fish species in an ecosystem provide the basic idea for implementing the ecosystem – based fisheries management. For the evaluation of environmental variables on organisms, fishes are important because they participate in multiple trophic levels in aquatic community and are having long life span and are easily sampled. Estimation of fish assemblage structure has high value in estuarine quality assessment. Species assemblage is an important ecological unit that interact trophically and respond to environmental and habitat conditions (Craig and Bosman, 2013). Fish communities of coastal estuarine environment possess a mixture of euryhaline species adapted to brackish environments, along with true marine and fresh water species. Therefore, the diversity of fish species in estuaries are very high and are composed of marine, estuarine, freshwater and migrating species (Henderson, 1988; Lobry et al., 2003). Biological condition of aquatic environment is important because fishes react with external aquatic environmental parameters. It leads to shift in the assemblage composition of fish species. Fish assemblages are recognized as sensitive indicators of habitat degradation, environmental contamination and overall ecosystem productivity. The fish assemblages in the estuarine systems are typically dynamic, reflecting the changing environmental conditions in which they are exposed (Tremain and Adams, 1995; Able and Fahay, 1998; Idelberger and Greenwood,

Open Journal of Marine Science, 2014, 4, 344-357
Published Online October 2014 in SciRes. http://www.scirp.org/journal/ojms
http://dx.doi.org/10.4236/ojms.2014.44031

Scientific Research

Decline in Diversity and Production of Exploited Fishery Resources in Vembanad Wetland System: Strategies for Better Management and Conservation

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Received 15 July 2014; revised 3 September 2014; accepted 16 September 2014

Abstract

This paper describes the fishery catch structure of Vembanad wetland system during August 2012 to July 2013. The estimates of fishery production indicated an annual landing of 4387.31 t, in which 480.98 t and 3906.33 t contributed by southern and northern zone of Vembanad respectively. Eighty species of finfishes, five species of penaeid shrimps, three species of palaemonid prawns and two species of crabs were identified from the study period. The catch per unit effort (CPUE) value was maximum for gill net (3.04 kg·h^{-1}) followed by stake net (2.43 kg·h^{-1}), Chinese dip net (2.01 kg·h^{-1}), seines (1.2 kg·h^{-1}), cast net (0.72 kg·h^{-1}) and hook and line (0.34 kg·h^{-1}). Biological integrity and fishery production of estuaries in the country are declining mainly due to various man induced activities. Thaneermukkom barrage, the salinity barrier, constructed across the Vembanad wetland system in 1976, transforming the water body into two distinct ecosystems, a fresh water zone on the south and a brackish water zone on the north, resulting in gross changes in physical, chemical and biological entity of the aquatic ecosystem. In the southern zone of Vembanad the marine fish species were less available with the closure of the barrage period. There are signs of decline of the Vembanad fishery resources, evident in the lesser number of species and decline of fishery production. This will lead to biodiversity loss, fish stock reduction and will ultimately affect the livelihood support of the traditional fishers to a large extent, besides affecting other ecological services. A better conservation measure must be implemented for maintaining the sustainable fishery resources in Vembanad.

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How to cite this paper: Asha, C.V., *et al.* (2014) Decline in Diversity and Production of Exploited Fishery Resources in Vembanad Wetland System: Strategies for Better Management and Conservation. *Open Journal of Marine Science*, **4**, 344-357.
http://dx.doi.org/10.4236/ojms.2014.44031

BOOK CHAPTER

Biodiversity & Evaluation: Perspectives and Paradigm shifts(2015)

TUBICULOUS WORM AGGREGATES IN A TROPICAL ESTUARY, ITS IMPORTANCE IN CONSERVATION OF WETLAND ECOSYSTEMS

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Abstract

Benthos formed the largest habitat on earth and plays an important role in the aquatic ecosystems as primary and secondary consumers. In shallow estuarine habitats they are actively involved in more diverse energy pathways and greater exchange of matter through bioturbation and benthic-pelagic coupling. Dense aggregations of tube dwelling polychaetes can stabilize sediments and give rise to large structures that are considered to be biogenic reefs. It provides habitat for wide ranging benthic communities that are different, more abundant and diverse than the surroundings. By considering its ecological functioning, biogenic reefs are of specific interest to wetland conservation. The study described the tubiculous worm aggregates of polychaetes along the Vembanad estuary on the south west coast of India. The aggregates were analyzed on a monthly basis and field sampling was done from March 2011 to February 2012. Bottom water quality and sediment characteristics were measured using standard procedures. Sediment and macrofauna samples were collected using a standard Van Veen grab (area of 0.04 m^2), sieved through 0.5mm sieves and identified to species level as far as possible. Bottom water temperature varied from 28 to 32^0C and dissolved oxygen ranged between 6.5 to 8.1 mg/l. The salinity varied between 0.08 to 31ppt with a mean value 11.32 ± 1.18ppt. The sediment temperature and pH ranged from 23 to 31.5^0C and 6.2 to 7.48 respectively. Sediment organic carbon content was maximum (6.8%) during April 2011. Texture of the sediment was clayey silt and the average silt content (50.8%) was more in the area followed by clay (35.44%) and sand (13.8%). Polychaete of the genus Diopatra, are key ecosystem engineers of marine sediments building biogenic reefs. The mean abundance of polychaetes ranged from 250 ind./m^2during August 2011 to 20180 ind./m^2 during October 2011. The most abundant polychaetes were tube dwelling polychaet es, Diopatra neapolitana and its abundance varied from 91 to 18,978 no/m^2. A total of 18 species of polychaetes were associated with the aggregates of D. neapolitana along with 11 groups of other macrofauna. The bivalve Musculista sp. also forms thick mat along with the aggregation of D. neapolitana tubes. The study indicated that, there was a strong temporal effect on the aggregates of Diopatra with the onset of southwest monsoon during which its density decreases tremendously. Diversity indices of polychaetes showed that Shannon-Weiner index (0.53-2.7) and Margalef richness (0.63 - 1.7) indicated a poor to moderate environment. The study also observed that, unscientific fishing; dredging and other anthropogenic activities are rampant in the study area that could have seriously affected the aggregations, a biodiversity treasure of benthic fauna – the basic food source for fishes and other organisms in the trophic system.
Keywords: *Benthos, Diopatra, Diversity indices, Polychaetes, Vembanad.*

Introduction

Biogenic reefs are "Solid, massive structures which are created by accumulations of organisms, usually rising from the seabed, or at least clearly forming a substantial, discrete community or habitat which is very different from the surrounding seabed. Structure of the reef is mainly composed of reef building organism and its tubes, shells, sediments, stones etc." [1]. It plays an important role in the marine environment providing substrata for sessile organisms and acts as an important source of food for other organisms. Biogenic reefs are important due to its role in the ecological functioning of the habitats and place in which it is present are of particular interest to nature conservation [2]. Reef building tube worms extend its geographic range from hundreds of kilometers of coastline [3] and play an important role in the coastline development mainly because its capacity to withstand in high energy breaker condition and spreads its colonial massive structure upward and seaward by fusion of solitary floating things [4]. The faeces and pseudofaeces of the sediment act as main source of food for other sessile organisms, which are using this substratum. In Western Europe, Diopatra go through its spreading and over the 80 years it moved towards 350 km northward direction from France to the southern Brittany Peninsula [5].

Ecology of benthic fauna in Cochin estuary was studied by several authors [6, 7, 8, 9 and 10]. There were no detailed studies on the polychaete aggregation and its characterization in Cochin estuarine area. Therefore the study aims to describe the aggregates of tube dwelling polychaete species and bivalve along with the associated macrofauna.

Materials and Methods

Cochin estuary is a tropical positive estuary situated on the south west coast of India (9° 40' & 10° 12'N and 76° 10' & 76° 30'E) with two permanent openings to the Arabian Sea. Annual rainfall was 3500.4 mm, in which 75.4% obtained during southwest monsoon. During the onset of monsoon, the hydrographic conditions in the estuary varied considerably. Aroor, the southern part of Cochin estuary was distinguished with

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