BIOECOLOGICAL STUDY OF BENTHIC COMMUNITIES IN THE KODUNGALLUR-AZHIKODE ESTUARY, SOUTH WEST COAST OF INDIA

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Bioecological study of benthic communities in the Kodungallur-Azhikode Estuary, South West Coast of India

Ph.D. Thesis under the Faculty of Marine Sciences

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Certificate

This is to certify that the thesis entitled "Bioecological Study of Benthic Communities in the Kodungallur-Azhikode Estuary, South West Coast of India" is an authentic record of research work carried out by Mr. Jayachandran P.R. (Reg. No. 4278), under my scientific 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, School of Marine Sciences, Cochin University of Science and Technology under the faculty of Marine sciences and that no part thereof 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 have been incorporated in the thesis.

Kochi - 682 016
December, 2017

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DECLARATION

I hereby declare that the thesis entitled **"Bioecological Study of Benthic Communities in the Kodungallur-Azhikode Estuary, South West Coast of India"** is an authentic record of research work done by me under the supervision of Dr. S. Bijoy Nandan, Professor, Department of Marine Biology, Microbiology and Biochemistry, School of Marine Sciences, Cochin University of Science and Technology, in partial fulfilment of the requirements for the Degree of Doctor of Philosophy in Marine Biology, School of Marine Sciences, Cochin University of Science and Technology under the faculty of Marine sciences and no part of this has been presented for any other degree or diploma earlier.

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Jayachandran

Dedication

To my parents,
In consideration of love and affection

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LIST OF ACRONYMS & ABBREVIATIONS

%	percentage
$<$	less than
$>$	greater than
$\approx$	approximately
°C	degree Celsius
µm	micrometre
µmol L^{-1}	micromoles per litre
ANOSIM	analysis of similarities
ANOVA	analysis of variance
BOD	biological oxygen demand
CAP	canonical analysis of Principal coordinate
CCA	canonical correspondence analysis
Chl-*a*	chlorophyll-*a*
cm	centimetres
day^{-1}	per day
DO	dissolved oxygen
et al.	et alli (Latin word, meaning 'and others)
etc	et cetera (Latin word, meaning and
$g.m^{-2}$	grams per square meter
$g{\cdot}kg^{-1}$	grams per kilogram
ha	hectares
hrs	hours
$ind.m^{-2}$	individuals per square metre
KAE	Kodungallur-Azhikode estuary
km^{-2}	square kilometre
L	litres
m	meter
m^2	square metre
$m^3\ s^{-1}$	cubic metre per second
mg L^{-1}	milligram per litre
mg	milligram
ml	millilitre
MLD	million litres per day
mm	millimetre
Mon.	monsoon

mV	millivolts
N	north
no.m^{-2}	number per square metre
NTU	nephelometric turbidity units
OC	organic carbon
OM	organic matter
PCA	principal component analysis
PCO	principal Coordinates Analysis
Pos.	post-monsoon
Pre.	pre-monsoon
PSU	practical salinity unit
SD	standard deviation
SIMPER	similarity percentage
SIMPROF	similarity profile analysis
sp.	species (singular)
spp.	species (plural)
SW	southwest
t	tonnes
v	version
Vis-à-vis	in relation to
viz	videlicet (Latin word, meaning 'namely')
wt	weight
y^{-1}	per year

Chapter 1
Introduction

Estuaries and coastal marine ecosystems encompass diverse habitats such as coastal lakes, coastal floodplains, mudflats, dune swamps, sedimentary habitats, algal beds, mangroves, saltmarsh swamps, seagrass meadows, and coral reefs which support rich species assemblages (Costanza *et al.*, 1997; Gray, 1997). Among the 25 biodiversity hotspots identified in the world, 23 of them are at least partially within the coastal environment, of which 10.45 percentage of coastal zones are designated as protected (Singh *et al.*, 2006). These ecosystems are also considered as one of the most productive and complex natural aquatic ecosystems on earth, the primary production rate of these ecosystems are comparable to the rainforests (Bijoy Nandan *et al.*, 2014; Jayachandran *et al.*, 2013; McLusky and Elliott, 2004).

According to Costanza *et al.* (1997), among the global ecosystem services provided by earth, marine systems alone contributes more than 63 percent (US$20.9 trillion yr^{-1}) with a significant contribution from estuaries and coastal marine systems. Among these, estuarine ecosystems are acting as critical reproductive, and nursery ground for biological components mediated by constant nutrients supply from autochthonous and allochthonous sources and supports significantly to marine fisheries. They also function as sinks and transformers of nutrients, by changing the quality and quantity of their transport from the land to the sea (Ketchum, 1951). Estuaries further provide ecosystem services by acting as a filtration and detoxification mechanism for terrestrial pollutants and as a flood controller (Barbier *et al.*, 2011). However, these critical ecosystems are experiencing a wide variety of disturbances from human activities and such impacts even threat to their integrity and sustainable exploitations (Bijoy Nandan, 2008). While these crucial habitats are also being lost by 2 to 3 times quicker than those in tropical forests (Diaz and Rosenberg, 2008; Lotze *et al.*, 2006). The estuarine ecosystems are the most complex aquatic system that acts as intermediate transition zones or ecotones. They form a link between the freshwater and marine environment (McLusky, 1971; Nybakken, 1993) or simply an area where rivers meet or enter, the sea (Lauff, 1967; Levinton, 1995b; Pritchard, 1967). This integrative processes of tying together terrestrial, freshwater and marine biomes, weave a web of complexity far higher than that of their three contributor ecosystems, which differ in the abiotic and biotic conditions. The abiotic components along the water column fluctuate on a spatial and temporal scale and reach extremes in estuarine waters than they do at sea or in the riverine zone. The salinity distribution and its behaviour under various conditions control the physical, chemical, biological, and ecological position of the estuary (Vijith *et al.*, 2009; Vinita *et al.*, 2015). The short-term variabilities of estuarine environment driven by tidal cycle and the seasonal changes driven by the regional climate make this environment a unique environment that brims with the life of all kinds and supports a plethora of animals (Iriarte and Purdie, 1994). Even though estuarine flora and fauna are adapted to survive in intermittently varying physical and chemical conditions. However, the increasing rate of anthropogenic pressures in these systems prompt extreme fluctuations in the environmental variables, and it may eliminate organisms that are failing to adapt with such extreme conditions.

The biological productivity of the estuarine environments is mainly controlled by ecological factors such as light, nutrients, and salinity (Nair *et al.*, 1983b). Tide driven changes in salinity and nutrient distribution, which potentially influence the spatial distribution of biological components. However, estuaries are protected from the full force of the ocean waves, winds, and storms by its geomorphological landscape (McLusky and Elliott, 2004). While, the seasonal changes in an estuary can exhibit significant variations in the distribution of physicochemical as well as biological components rather than short-term changes caused by tide (Bijoy Nandan, 2008; Sooria *et al.*, 2015). A continuum of assemblages along the salinity gradient from the freshwater river to the sea, with shifts in the ranges of organisms, appears in response to changes in freshwater flow (Attrill and Rundle, 2002). Along with salinity gradient, there are clear associated changes in sedimentary conditions from coarse sediment (sand or gravel) to fine sediments (muds) have been invariably found

(Thrush *et al.*, 2013). The freshwater flux to estuaries carries substantial amounts of suspended particles, including sediment from erosion of surrounding catchments, stream, and riverbanks. The fine fractions of deposits are easily transported and have a significant influence on sediment texture and water column turbidity of the receiving environments (Thrush *et al.*, 2013).

In general, the high algal biomass and continual re-suspension of sediments controlling the vertical light attenuation coefficient (Kd) and turbidity of water column (Cho, 2007). Other changes relate to alterations in turbidity of the water column or in the chemical composition including changes in nutrient concentrations, dissolved gases, and trace metal distribution (McLusky and Elliott, 2004). The primary fate and accumulation of terrestrial inputs, as well as the exchange of nutrients between an estuary and the coastal ecosystems, profoundly influenced by the hydrodynamics of the area, including freshwater flow, salinity, wind and tidal action (Jickells *et al.*, 2014). The dissolved macronutrients (N and P) and sediment nutrients have more significant roles in the primary production and energy flow in the estuarine environments. Since these zones are most productive and active, the surplus amount of primary production descends to the bottom (Qasim, 1977). This surplus component of carbon facilitates a good source of food for benthic fauna in the depositional environments dominated by soft bottom communities (Eyre, 1998; Qasim, 1977). These benthic assemblages have an essential role in the overall functioning of the entire estuarine ecosystem such as organic matter mineralisation and nutrient recycling. They also form as food for a diverse array of higher trophic levels in the estuary; such as coastal birds, fishes, and larval forms marine species, etc. (Bijoy Nandan, 2008).

Importance of benthic bioecological study

The term 'benthos' was originally used by Haeckel, as derived from the Greek word for 'depths of the sea' (Haeckel, 1891). It belongs collectively to all aquatic lifeforms that live in, on or near the benthic biotope. Benthos comprises a diverse number of life forms, ranging from microscopic bacteria to larger megafauna and they exhibit different feeding mode and distributional pattern

(Cowie and Levin, 2009). They usually divided into three functional groups, infauna, epifauna and hyperbenthos, i.e., those organism living inside the substratum, on the surface of the substratum and just above it, respectively (McLusky, 1999; Pohle and Thomas, 2001). According to their size, benthic animals are classified into three groups, the macro, meio, and microbenthos (Mare, 1942). This classification is based on the mesh size of the strainers used to separate them, which varies arbitrarily in different studies. The macrobenthos defined as organism retained in the sieve having mesh size between 0.5 mm (500 μm) and 1 mm, while in recent years, the use of 0.3 mm sieves (instead of 0.5 mm) is becoming popular. The major taxonomic groups represented among macrofauna are the polychaetes, crustaceans, and molluscs, along with hydrozoans, cirripedians, echinoderms, etc. (Eleftherioo and Mc Intyre, 2005; Snelgrove, 1999). However, meiobenthos, the lowest size attributed is 63 μm, and the upper limit depends on the mesh size of the strainer used for separating macrobenthos from meiobenthos (Giere, 2008). Meiofaunal communities mainly represented by nematodes, harpacticoid copepods, foraminiferan etc. The smallest size group, microbenthos, include those organisms that are not retained in the finest strainer used for meiobenthos collection and that includes the bacteria, most protozoans and larvae/juveniles of macro and meiofauna. Within the sediment matrix, the vertical extent of benthos is quietly limited, with organisms occupying only the top few centimeters. The practical differences in the sampling procedures adopted have led to the differentiation of benthos into soft bottom benthos and hard bottom benthos (Holme and Mc Intyre, 1971).

Benthos forms a direct source of energy for higher trophic levels, which includes the economically critical demersal fishes and indirectly supports the pelagic forms by transferring the energy (Parulekar *et al.*, 1982). Many of the benthic organisms have pelagic larvae, and they influence considerably on the planktonic food web by forming a component of planktonic community (Richard, 1973). It is also well-established fact that there is always a link between the benthic standing crop and the production of the exploited demersal fishery (Feebarani, 2009; Parulekar *et al.*, 1980; Waters, 1977). Thus, benthos regulates the physical, chemical, and biological environment of the estuary and link the sediment to the aquatic food web, through their burrowing and feeding activities (Coull, 1999; Covich *et al.*, 2004). Suspension feeders in the benthic community pump a significant amount of water through their body for food; they clean the water by removing sediments and organic matter (Dame, 1993). The unutilised organic matter that left from the water-column is being deposited on the bottom sediment (Solan *et al.*, 2004). Then deposit-feeding populations in the sediment re-mineralize them into nutrients, which was later transferred back into the water column. These remineralised organic materials form a vital source of nutrients to the aquatic environment and form a critical factor for maintenance of high primary production rates in the estuaries (Giere, 2008; Levinton, 2013). They also influence in remineralisation of other nutrients, dispersion, and burial of sediments and secondary production (Snelgrove, 1998). Benthic organisms have a direct connection with the physical nature of the substratum, which acts as a significant controlling factor to a greater extent (Sanders, 1958). The burrowing activities of deposit feeding populations benefit the bottom environment by enhanced sediment oxygenation, vertical flux of sediment particles, repacking of sediments and change of sediment stability and such process is termed as bioturbation. The detritus feeders in the benthic community along with their predator form a channel for the transfer of energy back into the pelagic environment (Snelgrove, 1998). While suspension feeders capture large quantities of particles and might directly regulate primary production and indirectly regulate the secondary production in the littoral food chain (Gili and Coma, 1998).

Benthos are also treated as sensitive indicators of organic matter pollution and related stress in the sediments (Bordovskiy, 1964). The variations brought about by the deposition of pollutants on the bottom sediment significantly affect the benthic fauna and flora. In general, pollution affects benthic community structure, predominantly by reducing species diversity by altering the reproductive success, prey-predator relationship and various interactions between species. Benthic populations are structural communities with numerous connecting links and disturbance on these communities from an external source can affect the entire food web structure. The constant supply of industrial effluents into the water body endangers the health of aquatic life, and it can even reach the human through the food chain. Therefore, benthos is a critical component of shallow water estuarine and coastal marine environments. A healthy benthic community is imperative in the long-term healthy functioning of aquatic ecosystems (Rosenberg *et al.*, 2004). The identification of factors responsible for distribution patterns of macrofaunal assemblages, especially those which help to differentiate between natural and man-induced changes is a crucial factor in mitigation and adaptation strategies for multiple threats in these environments (Borja and Dauer, 2008; Dauvin, 2007). These changes may include an increase in dissolved nutrient concentration, decrease or increase in the level of dissolved oxygen, increase in turbidity level, or variance in nature of the estuarine bottom. The degree or intensity of the impact of these changes on the estuarine life varies with the type and quantity of contaminant with the character of the biota. Over the past few decades, attention in ecosystem diversity and rising anthropogenic pressure have led to development of applied ecological research and impact studies on the benthic communities of coastal and estuarine environments (Bilyard, 1987; Flint and Younk, 1983; Giere, 2008; Wilson and Fleeger, 2012). Therefore, benthic assemblage pattern can be used as good indicators of the understanding state of the estuarine environment by taking advantage of their sessile or sedentary nature and different tolerances to environmental stress (Dauer, 1993; Kennedy and Jacoby, 1999).

History and development of estuarine benthic studies

Investigations concerning benthos advanced well only in the late 18th and early 19th centuries when the use of various dredging devices became popular. A new era in benthic studies started during the early 1900's. It was connected with the detailed investigations (Petersen, 1915; Petersen, 1918; Petersen and Jensen, 1911; Peterson, 1913; Peterson, 1979) along Danish waters. Their works mainly focused on community structure and standing crop of benthic animals and followed by many scientific investigations on benthic fauna that have been initiated in different parts of the world. Most of these studies were restricted to macrobenthos owing to the relative ease in the investigation. The works of Remane (1933), Mare (1942), Weiser (1953), Weiser (1954), Weiser (1956), Weiser (1959) and Weiser (1960) on meiobenthos have been regarded as pioneer studies in the field of meiobenthology. However, no precise starting point can specify for the studies in estuarine science, but three investigations that have been undertaken in the 1930s point to the future direction of estuarine benthic ecology in Europe. Remane (1934) published a major review of the brackish water fauna (Die Brackwasserfauna), which particularly emphasised the physiological responses of brackish water organisms to gradients of salinity. Thamdrup (1935) while describing the ecology of animals from estuarine sediments, led to a detailed study of the Tees estuary in northern England by Alexander *et al.* (1935). To a great extent the three themes provided by the earlier studies are the physiological responses of estuarine organisms, ecology, and their responses to pollution, have provided the foundation for much of what has become the estuarine benthic science. Remane and Schlieper (1958) reviewed the existing knowledge of the brackish water environment, especially on physiological studies at that time on the effects of salinity on estuarine organisms. This subject has taken further, as reviewed by Kinne, in several publications, which emphasised the role of salinity as the 'ecological master factor', culminating in the 'Salinity' (Kinne, 1978). Yonge (1953) described 'Aspects of life on muddy shores' with interestingly little mention of estuaries. By 1958, a realisation had arrived that estuarine scientists need to define their terminology more accurately, which led to the Venice symposium on the classification of brackish waters, which described the zones of an estuary, or brackish water, in term of salinity zones (Venice system, Anonymous (1958). An increasing appreciation of the existence of the estuary as a habitat, distinct from either the sea or a river, led to an outstanding conference held at Jekyll Island, Georgia, the USA in 1964 and the subsequent publication of the proceedings of that symposium, edited by Lauff (1967). These studies ensue in the current status and developments of coastal and estuarine benthic ecological research.

Benthic studies in Indian estuaries

Studies on taxonomic aspects of brackish water benthic fauna were carried out along the Indian estuaries by Annandale (1907), Annandale and Kemp (1915), Preston (1916) in an early 20th century. Further various scientific investigations were initiated by the researchers on taxonomy and ecological aspects of benthic fauna along the coastal and estuarine waters of India. Fauvel (1953) recorded 283 species of marine and estuarine polychaetes from different parts of India, among these 47 species were estuarine. Later, Hartman (1974) prepared a bibliography of polychaetes from India that included 59 families, 315 genera, and 860 species. Misra (1998) reported 167 polychaete species belonging to 38 families from different brackish water bodies in India. Ajmal Khan and Murugesan (2005) studied polychaete diversity in Indian estuaries and recorded 153 species of polychaetes representing about 37.46 percent of the total polychaetes present in Indian estuaries.

Many studies have been carried out along the India water to understand isopod diversity (Dev Roy, 2012; Dev Roy *et al.*, 2012; Kensely, 2001; Pillai, 1954; Stebbing, 1911). A study by Dev Roy and Nandi (2010) recorded 299 species of isopods belonging to 131 genera and 38 families from marine waters of India that contributed 2.7 percent of the global isopod fauna. The total diversity of molluscs recorded from India is 5,169 species (MoEF, 2014), representing around seven percent of the overall global molluscan diversity. However, there is no consensus among various authors on the total number of marine molluscs from India. According to Venkataraman and Wafar (2005), in India waters, 3,370 species of marine molluscs were recorded while Tripathy and Mukhopadhyay (2015) accounted for 2,300 species. Subba Rao *et al.* (1992) recorded 48 species of molluscs from Rushikulaya estuary, of these only 13 species are of estuarine species, and further, Subba Rao *et al.* (1995) reported 120 species from Hooghly-Malta estuary, Kolkatta. Similarly, in Krishna estuary, nearly 91 species molluscs were recorded that for the Godavari estuaries was 62 species (Mahapatra, 2001, 2008) however, the majority of the species collected are represented by death shells. Gurumayum (2015) recorded

29 molluscan species in estuarine zone of Penner river in the Karnataka coast, among them, 14 species were collected in live condition.

On the east coast of India, several scientific investigations were carried out by different scientists. The benthic fauna of the brackish water environments of Madras was examined by Panikkar and Aiyar (1937). Balasubrahmanyan (1964) and Rajan (1964) conducted similar studies in the Vellar estuary and Chilka Lake respectively. Ganapathi and Raman (1970) assessed the potential of indicator species, *Capitella* sp. in the Vishakapatnam harbour. Further Raman and Ganapati (1983) studies focused on effects of pollution on eco-biology of benthic polychaetes in coastal environments of the east coast of India. Fernando *et al.* (1984) made observations on the distribution of benthic fauna in Vellar estuary, and later Chandran (1987) studied the relationship of benthic fauna to physicochemical parameters and sediment composition in the same estuary. Vijayakumar *et al.* (1991) have made observations on the macro and meiofauna from Kakinada bay and backwaters. Murugan and Ayyakkannu (1991) have given an account of benthic macrofauna in Cuddalore-Uppanar backwaters of Tamil Nadu. Manikandavelu and Ramdhas (1994) have worked on the bioproduction dynamics of mangrove-bordered brackish water along Tuticorin coast of Tamil Nadu. Chandra Mohan *et al.* (1997) have given an account of the role of Godavari mangroves in the production and survival of prawn larvae. The ratio of carbon and nitrogen stable isotope in the benthic invertebrates in the Coringa Wildlife Sanctuary area was carried out by Bouillon *et al.* (2002). Sigamani *et al.* (2015) made an attempt on biotic indices based approach to assess the ecological health of the Vellar-Coleroon estuarine system of the east coast of India.

In the west coast of India, benthic assemblages of Malabar and Trivandrum coasts were studied by Kurian (1953) and Seshappa (1953). Kurian (1967) has later given an account of the benthos of south-west coast of India. At the same time, Desai and Krishnan Kutty (1967) made a comparative study of the marine and estuarine fauna of nearshore region of the Arabian Sea. Damodaran (1973) carried out work on the benthos of the mud banks of Kerala coast. Harkantra (1975) examined seasonal variation in the benthic production of the Kali estuary. The benthic population of the estuarine region of Goa was studied by Parulekar and Dwivedi (1974). Harkantra *et al.* (1980) have worked on the benthos of shelf region along the west coast of India. Parulekar *et al.* (1980) have observed the benthic macrofauna annual cycle of distribution, production and trophic relations in Goa estuaries. Harkantra and Parulekar (1981) attempted to rule out the qualitative and quantitative differences in distribution and production strategies of benthic macrofauna in the coastal zone of Goa. Effect of high organic enrichment of benthic polychaete population in west coast estuary of India assessed by Ansari *et al.* (1986). Ansari *et al.* (1994) have worked on macrobenthos of Marmagao harbour. Gopalan *et al.* (1987) undertook some of the investigations on the benthic fauna extending right from Cochin to Alappuzha coast. Harkantra and Parulekar (1994) have studied the macroinvertebrates of Rajpur bay. Jagtap *et al.* (1994) examined benthic fauna in the mangrove environment of Maharashtra coast. Wafar *et al.* (1997) investigated the benthic fauna of mangrove environments in the Mandovi-Zuari estuaries on the central west coast of India. Mascarenhas and Chauhan (1998) studied the ancient mangrove of Goa. Phytoplankton and macrobenthos in the nearshore coastal waters of an oil terminal at Uran (Maharashtra) were studied by Ram *et al.* (1998). The examination of estuarine and nearshore benthos of Vashishti estuary in Maharashtra was reported by Vijayalakshmi *et al.* (1998). Ingole and Parulekar (1998) examined the role of salinity in structuring the intertidal meiofauna of a tropical estuarine beach in the Goa. Sivadas *et al.* (2016) tested the efficiency of various temperate benthic biotic indices in assessing the ecological status of a tropical ecosystem and recommended the complementary use of different indices for accurate assessment of the environmental condition. Murugan *et al.* (1980) deliberated the benthic community of the Ashtamudi estuary. The ecology and distribution of benthic fauna of Ashtamudi estuary were further carried out by Divakaran *et al.* (1981). Nair and Abdul Azis (1987a) have made observations on the benthic polychaetes of the retting zone in the Kadinamkulam backwaters. The fish mortality from anoxic and sulphide pollution in the estuaries of Kerala was investigated by Bijoy Nandan and Abdul Azis (1995a). Studies on the benthic fauna of the Veli estuary, Kerala state have made by Bijoy Nandan and Abdul Azis (1995a). Asha Nair and Abdul Aziz (1995) have given an account of the water quality and benthic fauna of Kayamkulam backwaters. Bijoy Nandan (2008) made a review on the status and biodiversity modification including bottom fauna of Kerala coastal wetlands.

Ecology of Vembanad-Kol Wetland

The term wetlands is defined by International Convention on Wetlands as an "areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six meters" (Article 1.1). Besides in Article 2.1 appends that wetlands "may incorporate riparian and coastal zones adjacent to the wetlands, and islands or bodies of marine water deeper than six meters at low tide lying within the wetlands"(Federal Geographic Data Committee, 2013). India has nearly 25 well-defined estuaries having a water-spread area of 2.7 x 104 km^2 located along 7500 km coastline spread over nine coastal states. Of these eight estuarine systems on the east coast and 17 are in the west coast (Qasim, 2003). There are 14 major, 44 medium and 162 minor rivers that collectively discharge about 1.56x1012 m^3 yr^{-1}, influencing the physicochemical and biological activity of these estuaries. However, the intensity of biological productivity in these waters has strongly affected the seasonal variability of salinity. The salinity of estuaries in the south-west coast of India is profoundly influenced by the Indian summer monsoon (ISM) or south-west monsoon (June-September), so these waters are referred as monsoonal estuaries (Vijith *et al.*, 2009). In the spatial salinity gradient of Indian monsoonal estuaries, primary and secondary production are observed maximum at low saline areas preferred by the variety of planktonic organisms of marine, brackish and freshwater origin (Sooria *et al.*, 2015; Vineetha *et al.*, 2015). Along the coastal zone of India that is inhabited by approximately 560 million people, producing about 20542 million liters of sewage per day (MLD) (Central Pollution Control Board, 2015b). These areas are also preferred destinations for developments, may it be an industry, urban settlement or a port (Heip and Herman, 1995). The continuous anthropogenic activities resulted in degradation of these critical habitats. In this scenario, the benthic secondary production estimation and diversity documentation of these vital habitats are the useful tools to understand the state of the environment (Dolbeth *et al.*, 2012).

Kerala State in the south-west coast of India having 41 rivers flowing towards west coast and emptying into the Arabian Sea through the backwaters /estuaries /coastal wetlands (locally called as Kayals) spreading over 590 km coastline. The main rivers in this zone are Chalakkudypuzha, Muvattupuzha, Pampa, Chaliyar, Bharathapuzha, Kallada, and Achankovil together discharge about 45060 x 106 m^3 of water annually into the Arabian Sea (Anonymous, 1974; Anu *et al.*, 2014). The State has an average human population density of 860 people per sq. km against the national average density of 382 people per sq. km (Census of India, 2011). People of the State depend on water bodies such as rivers, ponds, and wells for their daily requirements about more than 85 percent. Increasing discharge of industrial effluents with high BOD, toxic chemicals and suspended solids results in many rivers unsuitable for fishing and recreational use. Industrialisation along with support facilities and associated township developments also place demands on natural water resources. The State every day being discharging about 2399.03 million litres per day (MLD) untreated sewage to water bodies (Central Pollution Control Board, 2015).

Among the complex aquatic systems of Kerala, the International Convention on Wetlands designed three wetland ecosystems as ‘Ramsar sites’ for the conservation of biological diversity for supporting human life by the ecological and hydrological roles they perform (Anonymous, 2003; Bijoy Nandan, 2008; Gardner and Davidson, 2011). Among these, the Sasthamkotta Lake (Ramsar site no. 1204) is a freshwater ecosystem, while the Ashtamudi Wetland (Ramsar site no. 1204) and the Vembanad-Kol Wetland (Ramsar site no. 1214) are brackish water coastal wetlands. The Vembanad-Kol Wetland (09°00'-10°40'N and 76°00'-77°30'E) form the third most significant humid brackish wetland in India that has an area of 1521.50 km^2 (152150 ha) and defined as a Ramsar site in November 2002 (Gardner and Davidson, 2011). Moreover, it covers about 2.5 percent of the geographic area of Kerala state (1521.5 sq.km). This wetland complex includes Vembanad Lake, Kuttanadu marshy areas, and Kol wetlands that are extending from Kuttanad of Alappuzha district on the south to Kol wetlands of Thrissur district on the north. The Vembanad-Kol wetland ecosystem is fed by Periyar, Karuvannur, Chalakudy, Muvattupuzha, Meenachil, Manimala, Pamba, Achancoil, Keecheri and Puzhakkal rivers originating from the Western Ghats.

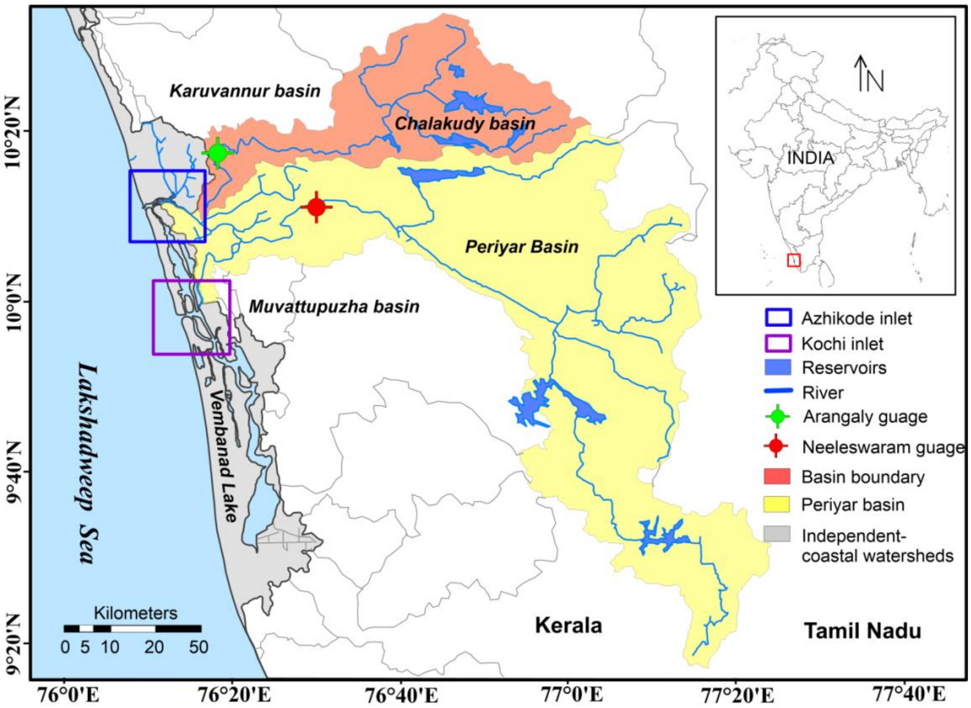


Figure 1. Location of study region in the Vembanad-Kol wetland Ecosystem-modified after Sreeja *et al.* (2016)

The Vembanad-Kol Wetland is typically divided into two distinct segments, the freshwater dominant southern zone, and the salt-water dominant northern zone. It has two permanent opening to the Arabian Sea (Ramamirtham and Muthusamy, 1986), one is at Kochi (Cochin estuary) with 450 m wide mouth and an average depth of 10 to 12 m in the main channel, and it receives Periyar, Pamba, Achankovil, Manimala, Meenachil and

Muvattupuzha rivers. Another permanent opening is at Azhikode with a 180 m wide mouth (Kodungallur-Azhikode estuary) and depth range of 7 to 8 m [Figure 1]. The region under the present study is Kodungallur-Azhikode estuarine complex (Azhikode inlet) forms a confluence zone of Periyar (70 % discharges through this channel), Chalakudy and Karavannur rivers (connected to estuary through human-made Canoli canal) (Revichandran and Abraham, 1998; Sreeja *et al.*, 2016). The tract, therefore, depends on these river systems and affected by all upstream activities that occur in the basins. The wetland also has a temporary opening at Thottappally in the southern zone of Vembanad wetland ecosystem, and it is active only during the southwest monsoon period. This channel is regulated by a spillway across the mouth of the estuary (Ramamirtham and Muthusamy, 1986).

The Periyar and Chalakudy stretch has a total catchment area of 6800 sq. km that are the most heavily developed Western Ghat river basins with 22 reservoirs for irrigation, power generation and water diversion in the upstream catchments. The abundant water resources of these high rainfall tropical basins have resulted in several water diversion projects into the arid eastern plains from the late nineteenth century onwards (Sreeja *et al.*, 2016). The Chalakudy and Periyar rivers originating in the southern Western Ghats after flowing west across forested hills, agricultural valleys and wetlands for a distance of 130 and 244 km respectively join together 10 km inland from the sea. The hydrological boundary of the Chalakudy basin is limited up to the confluence with Periyar whereas that of the Periyar is confined to the southern coastal tract where the southern arm of the Periyar spreads out to form the Vembanad Kol-wetland. The main branch of Periyar River joins the Chalakudi River at Punthenvelikara and then expands into a broad area of water in the Kodungallur-Azhikode estuary. The Karuvannur River originating from the Western Ghats takes a southwesterly direction up to Panamkulam and then a westerly direction. Just before it joins the backwater, it bifurcates, and one branch flows towards the south to join the Kodungallur-Azhikode backwater through Canoli canal while the other section flows towards northwards and enters the Lakshadweep Sea at Chettuva. The coastal stretch beyond the Chalakudy-Periyar confluence forming the Kodungallur-Azhikode estuary has therefore found to be technically outside the purview of any particular river basin. It is observed that surface drainage from these coastal tracts flows into the joined Periyar-Chalakudy River or directly into the estuary. Besides, several streams drain that straight into the sea forming small independent drainage areas. The shifts in the floodplain boundaries between the Chalakudy and the Periyar further complicate the drainage delineation (Sreeja *et al.*, 2016).

Mixed semidiurnal tides influence both Kodunagllur-Azhikode estuary (Azhikode inlet) and Cochin estuary (Kochi inlet) with an average tidal range of 1 m, which is referred to as microtidal estuary (Qasim and Gopinathan, 1969). Constant mixing with seawater through tidal exchanges in these opening has given the characteristics of a tropical estuary (Ajith and Balchand, 1996; Balchand and Nair, 1994). During the south-west monsoon or Indian summer monsoon (June-September) receives the most rainfall thus defining the peak of the "wet" season (Shivaprasad *et al.*, 2013). In most years, pre-monsoon (March-May) experiences the lowest recorded rainfall, thus defining the peak of the "dry" season. The cumulative runoff in the rivers exceeds the estuarine volume during the south-west monsoon, and the entire estuary assumes riverine condition (Revichandran *et al.*, 2012; Sarma *et al.*, 2012). Since the river discharge concentrated for only a couple of months in the Indian estuaries (flushing time < 10 days), the complete microbial decomposition of organic matter in these the waters are less compared to estuaries of Europe and USA (flushing time > 40 days) (Vijith *et al.*, 2009). Thus, after the wet rainy season, the invasion of seawater can be traced up to 15-20 km upstream during the inter-monsoon period (Revichandran, 1993).

After industrialisation era, the Vembanad-Kol Wetland ecosystem has undergone a wide array of anthropogenic alterations in the environment, leading to an estimated reduction of its extent by about 35 percent because of the installation of bunds and reclamation for agriculture, harbour, and urban development. Since 1970, an area covering 176 hectares has been reclaimed for harbour and urban growth. The increasing effluent discharge from industrial, agricultural, domestic and retting sources compound to its deterioration. The decreased volume of backwaters and limited exchange with the sea reduces the diluting capacity of the backwaters. Several ecological studies have been carried out in the southern zone of wetland especially after the construction of Thannermukkom Salt Water Barrage (at least 1250 m long) in 1976. The barrage was constructed in the lake aimed to prevent saltwater incursion during dry seasons and to promote cultivation in the low-lying paddy fields however it is still in debated on its ecological impact (Asha *et al.*, 2016; Shivaprasad *et al.*, 2012). However, rest of the previous studies in the Vembanad-Kol wetland is confined to the central part (Cochin backwater) of wetland due to proximity to the booming city of Cochin (Kochi). It has a population of nearly 1.5 million (Stephenson *et al.*, 2004) and 60 percent of the chemical industries of Kerala is located in the Cochin area of Wetland. It discharges approximately 0.104 M m^3 d^{-1} of effluent which containing nearly 260 t d^{-1} of organic wastes (Balachandran *et al.*, 2003). The river discharges of 19,000 M m^3 y^{-1} also carry a fertiliser load (20000 t y^{-1}) and which could facilitate the disposal of several chemical agents, with a consequent degradation in the water quality causing severe health hazards to the aquatic organisms. During the past 50 years, the effluent discharge from the industrial city of Kochi has increased to 6.5 million m^{-3} d^{-1} (Vinita *et al.*, 2015).

Since, these zones are most productive and active environment (Qasim, 1977), support enormously to benthic secondary production (Qasim, 1977), those are capable of organic matter mineralisation and nutrient recycling (Pratihary *et al.*, 2009). Their structure and function are strongly influenced by various anthropogenic pressures (Griffiths *et al.*, 2017). Therefore, they form better indicators of overlying water mass and being food for higher trophic levels of the backwater. Many scientific investigations are made on the benthic ecology of these estuarine complexes. Majority of the previous benthic studies in the wetland concentrated to nearby areas of Cochin backwaters (Ansari, 1974; Ansari, 1977; Desai and Krishnankutty, 1967; Kurian, 1972; Pillai, 1978; Unnithan *et al.*, 1977). The bottom fauna of Cochin backwaters was investigated by Preston (1916) in the early 20th century. The incidence of fish mortality due to industrial pollution from the upper reaches of Cochin backwater was reported by Unnithan *et al.* (1975). Kurian *et al.* (1975) documented the effect of organic pollution due to industrial pollution on water quality parameters in Cochin backwaters. Later, the impact of pollution on benthos was made by Pillai (1977) and Remani (1979), and further Venugopal *et al.* (1980) investigated the effects of physical alteration on abundance and distribution of flora and fauna in the backwater. The entire area of wetland is susceptible to shrinkage owing to reclamation and other anthropogenic activities (Gopalan *et al.*, 1983) and eventually lead to the variations in the population, the structure of trophic webs and even altered the overall functioning of this ecosystem (Menon *et al.*, 2000). It is noticed that due to intense eutrophication process, globally about 150 coastal ecosystems are reported as oxygen deficient (Joseph and Ouseph, 2010; The United Nations Educational-Scientific and Cultural Organization, 2003). Similarly, persistent stresses imposed by an increased level of chemicals in the industrial effluents and organic load from the sewage will also affect biotic communities, especially the benthos (Remani *et al.*, 1983; Sarala Devi and Venugopal, 1989).

The study by Nair *et al.* (1983a) estimated annual consumption of primary production by the herbivores zooplankton in the brackish water zone of the wetland as about 25 percent of the total production. This study hypothesised that rest of the unconsumed basic food supports a detritus food chain ultimately to benthic fauna in the backwater. Another study by Nair *et al.* (1983b) focused on indicator species of organic pollution in the Cochin backwaters and followed by series of benthic ecological studies were conducted in the different brackish water zones of wetland (Anvar Bachan, 1984; Aravindakshan *et al.*, 1992; Arun, 2005; Asha *et al.*, 2016; Feebarani, 2009; Feebarani *et al.*, 2016; Geetha and Bijoy Nandan, 2014; Geetha *et al.*, 2015; Geetha *et al.*, 2010; Gopalan *et al.*, 1983; Martin *et al.*, 2011; Menon *et al.*, 2000; Pillai, 2001; Prabhadevi *et al.*, 1996; Rasheed, 1997; Rehitha *et al.*, 2017; Remani *et al.*, 1983; Sarala Devi, 1986; Sarala Devi *et al.*, 1991; Sarala Devi and Venugopal, 1989; Sheeba, 2000; Sunil Kumar, 1993, 1995, 1999; Sunil Kumar, 2002; Sunil Kumar and Antony, 1994). However, no comprehensive ecological studies have evolved from the Kodungallur-Azhikode estuary regarding such matters, especially on anthropogenic influence on benthic ecology. In this scenario, the detailed study of the benthic fauna of Kodungallur-Azhikode estuary become worth to identify the major anthropogenic activities affecting estuary and to recommend remedial actions for improving the health and viability of this system. The importance of early detection of human-induced alteration of estuarine environments cannot overstate because the success of cost-effective remedial measures depends on addressing the problem expeditiously before it becomes intractable. In this regard, the present study will give comprehensive information on the distribution of benthic fauna about various ecological conditions in the Kodungallur-Azhikode estuary.

The main objectives of the present investigation are:

- To understand population dynamics of relevant species to calculate benthic secondary production and assess the energy flow to higher trophic levels in the Kodungallur-Azhikode estuary (KAE).
- To find out the major functional groups in the benthic assemblages and its characteristics in prevailing environmental conditions in the estuary.
- To assess the ecological status of the estuary by using different benthic marine biotic indices which are capable of detecting anthropogenic disturbances.

Chapter 2
Study area, sampling design and analysis

II. 1. Study site: Kodungallur-Azhikode estuary

The Kodungallur-Azhikode estuarine (10°11'-10°12'N and 76°10'-76°13'E) system spread over an area of about 700 ha. It extends into the Cochin backwaters in the south and connected to the Karuvannur River through the man-made Canoli canal to the north (Sreeja *et al.*, 2016). It faces the Arabian Sea on its west and bounded by the Pullut backwaters on its east. The coastal region of the study site (10°-10°20'N and 76°10'-76°20'E) has an area of 300 sq.km and a high average population density of 1850 persons per sq. km. It has a shallow water table and is speckled with numerous small freshwater tanks and water channels that crisscross the landscape to join either the backwaters, the main estuary further downstream or the sea directly (Sreeja *et al.*, 2016). The estuary having a 180 m wide mouth with a depth range of 7 to 8 m forms opening to the Arabian Sea at Munambam-Azhikode region. Tides in the estuary are semidiurnal with microtidal tidal range, and tidal effects extend to approximately 25 km landward of Azhikode. Average annual rainfall in the area is 310 cm (Revichandran and Abraham, 1998). The estuary resembles the positive type of estuary, and freshwater input varied from 10 m^3s^{-1} to 21 $m^3 s^{-1}$ during pre-monsoon to 123 m^3s^{-1} to 387 m^3s^{-1} during south-west monsoon (Revichandran and Abraham, 1998), and it can be referred as a monsoonal estuary (Vijith *et al.*, 2009). This estuarine, coastal wetland is a biologically productive region, where the hydrographic and biological processes show strong seasonal variations. This inlet and related backwater contribute a substantial share to the fishery of the area. In such productive estuarine ecosystem, benthic fauna plays a vital role in remineralisation, biogeochemistry, and food web dynamics. In recent years, the Kodungallur-Azhikode estuary has undergone deterioration due to land reclamation, illegal encroachments, dredging, mangrove deforestation, waste disposal, ballast water discharge, fish processing plants, domestic sewage yards etc. (Jayachandran and Bijoy Nandan, 2012). These critical systems are highly sensitive to changing climatic variables, such as river discharge of water and sediments, urbanisation, temperature rise, rainfall, wind, wave etc. (Anu *et al.*, 2014; Kennish, 2002; Unnithan *et al.*, 1975). However, no significant comprehensive scientific information has evolved on the ecology and trophic status benthic fauna of Kodungallur-Azhikode estuarine system [Figure 2]. Thus, the present study forms a substantial contribution to the bioecology of benthic fauna in the region.

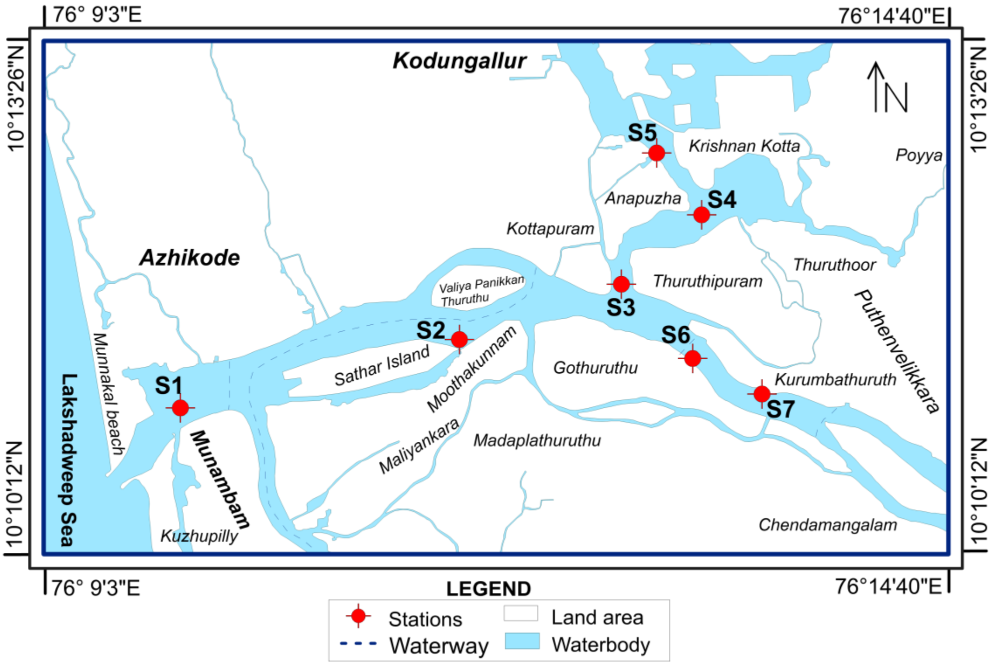
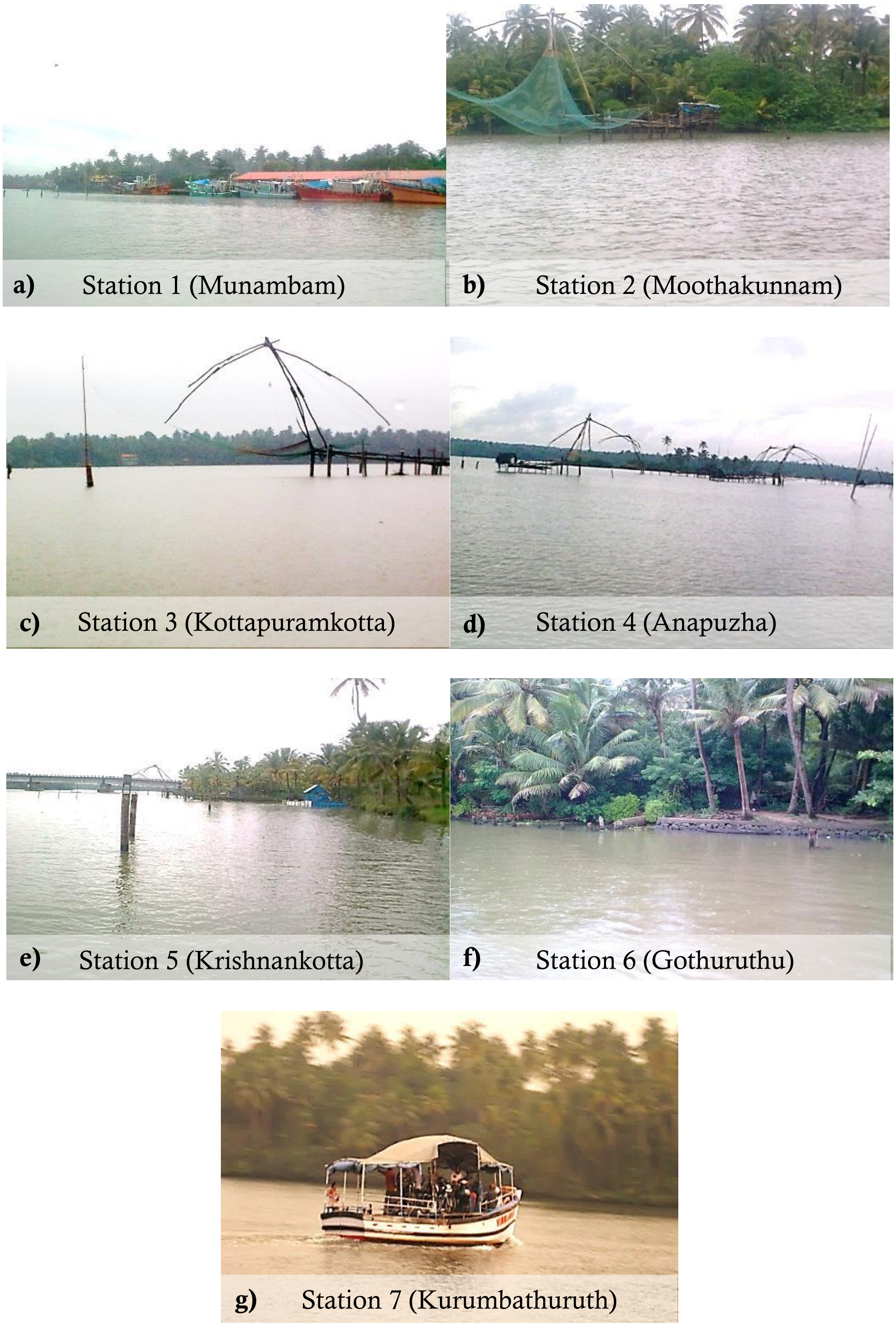


Figure 2. Map of the sampling sites in Kodungallur-Azhikode estuary

It was in this context, seven selected study stations with various ecological characteristics have been selected in the estuary for monthly field sampling of water, sediment and benthic faunal parameters. The details of the chosen locations are as follows. The station 1 was Munambam harbour (10°11'2.65"N-76°10'10.30"E), located at the estuarine mouth where the estuary permanently is connected to the Lakshadweep Sea, and the station was under the influence zone of the sea. At this zone, a narrow canal on the south side of inlet connects to the northern arm of Cochin backwaters. The area was also noted for series of stake-nets and batteries of Chinese dip-nets fishing activities.



a) Station 1 (Munambam)

b) Station 2 (Moothakunnam)

c) Station 3 (Kottapuramkotta)

d) Station 4 (Anapuzha)

e) Station 5 (Krishnankotta)

f) Station 6 (Gothuruthu)

g) Station 7 (Kurumbathuruth)

Figure 3 a-g. Study sites in Kodungallur-Azhikode estuary

The station 2 was Moothakunnam (10°11'29.38"N-76°11'48.77"E), extensive sand mining has been noticed near to this zone, however, this zone also witnessed for Chinese dip net fishing. The water column between Sathar Island and Moothakunnam was characterised by cage farming of finfishes, rope culture of green mussel *Perna viridis* and backwater oyster *Crassostrea bilineata*. The station 3 spread over an area between the landmass of Kottapuram and Turuthipuram (10°11'47.63"N-76°12'49.21"E), here estuary splits into two arms towards the north and south. This area is a confluence zone of Periyar and Karavannur River where dip net fishing activity was seen. Station 4 was Anapuzha (10°12'13.99"N-76°13'20.93"E), in the northern arm of the estuary has noticed for comparatively good fishing activities by dip net, gill nets and other traditional harvesting methods. The area between station 3 and 4 also had occasional sand mining activities. The station 5 was Krishnankotta (10°12'42.98"N-76°13'1.85"E), this zone was the northernmost station for the present study, and this area was influenced by cage culture of finfish, *Lates calcarifer*. The traditional aquaculture ponds were also noticed in this area. The station 6 was near to Gothuruthu (10°11'27.12"N-76°13'5.29"E), the zone was at the southern arm and under the influence zone of heavy river discharge from Periyar and Chalakudy. This zone was also noted for ferry service, and station 7 was Kurumbathuruth (10°11'9.01"N-76°13'36.88"E), the southernmost station in the estuary with high freshwater influence, the zone is near to confluence zone of Periyar River and Chalakudy River. The ferry service has also been noted in the area. The average depth of the estuary was 3.63 m, and among all station's, station 6 had the maximum mean depth (4.22 m) while station 5 depicted lowest mean depth (2.21 m) [Figure 3 a-g].

II. 2. Sampling design

The Research Vessel "King Fisher" was used as a conveyance for the sample collection during field sampling. Monthly field sampling was conducted during the early morning hours for 24 months, from July 2009 to June 2011. For seasonal analysis calendar year was divided into three distinct seasons; (1) monsoon (June - September), (2) post-monsoon (October - January) and (3) pre-monsoon (February - May) for further analysis (Qasim and Gopinathan, 1969).

Sampling was carried out under the scientific research project implemented in the Department of Marine Biology, Microbiology and Biochemistry, Cochin University of Science and Technology by Kerala State Council for Science, Technology and Environment (KSCSTE), Government of Kerala (KSCSTE sanction order no. (T) 060/SRS/2009/CSTE dated. 21.05.2009) entitled "Ecology and fish production potential of the Kodungallur-Azhikode backwater ecosystem".

II. 3. Hydrographic methods and analysis

The hydrographic data were collected at monthly intervals for the two-year period in the study. The rainfall data for the river basin was obtained from the resources of the Hydrometeorology division of the Indian Meteorological Department (www.imd.gov.in). While the daily river discharge data gathered from standard gauging stations such as the Neeleswaram gauge of Periyar basin and the Arangaly gauge of Chalakudy river basins was collected from the Central Water Commission (CWC), Govt. of India website (www.india-wris.nrsc.gov.in). Water samples for hydrographic data have been collected in replicates at monthly intervals along the water body. The bottom samples were collected using Niskin Water Sampler (General Oceanics) of 2.5 litre capacity. The samples for dissolved oxygen (DO) and biological oxygen demand (BOD) have taken in a 125 ml stoppered glass containers taking care that no air bubbles have trapped in the sample. The samples for dissolved oxygen were fixed immediately with manganous chloride solution (Winkler A) followed by alkaline potassium iodide (Winkler B) solution. Water samples for the analysis of nutrients and salinity were collected in pre-cleaned polythene bottles of 1 litre capacity and kept in iceboxes. The analyses of the following parameters have done using standard procedures. The nutrients were analysed immediately after filtering through Whatman No: 1 filter papers, following standard procedures (Grasshoff *et al.*, 1999; Grasshoff *et al.*, 1983; Strickland and Parsons, 1972) and using a spectrophotometer (Systronics UV-VIS spectrophotometer, Model No.117), after proper calibration. The nutrient values have expressed in the unit of micromole per litre ($\mu mol\ L^{-1}$).

The physical aspects of water quality obtained during the study and the depth of the water column have measured by hand-held calibrated lead sounding line and expressed in meters (m) (Food and Agriculture Organization, 2010). Digital nephelo turbidity meter was used to measure turbidity (Model. Systronics 132). A Secchi disc of 30 cm diameter designed as black and white in alternative sectors with ballast and that attached to a graduated string marked in centimetres has used for the measurement of transparency of water (American Public Health Association, 2005). The temperature of the water column was recorded using a mercury thermometer (0 - 100 ± 0.01°C) immediately after the collection of samples. The water column salinity was recorded in the field using an optical refractometer (Atago, Japan) and cross-checked in the laboratory by employing Mohr-Knudsen method (Strickland and Parsons, 1972; The United Nations Educational, Scientific and Cultural Organization, 1985).

The chemical property of water quality such as dissolved oxygen was determined by the modified Winkler method, as recommended in Strickland and Parsons (1968). The principles of the determination and the potential sources of systematic errors addressed by Grasshoff (1983) have also been noted. This method depends on the oxidation of manganese dioxide by the oxygen dissolved in the samples, resulting in the formation of a tetravalent compound, which on acidification liberates iodine equivalent to the dissolved oxygen present in the sample. The amount of iodine released was determined by titration with sodium thiosulphate. The results were expressed in the unit, milligrams per litre (mg L^{-1}). Biochemical oxygen demand (BOD) was measured by the method recommended by American Public Health Association (Eaton *et al.*, 2005). The principle is to measure the molecular oxygen used during a specified incubation period for the biochemical degradation of organic matter and that used for oxidising inorganic material such as ferrous iron and sulphides. It also measures the amount of oxygen used to oxidise reduced forms of nitrogen (nitrogenous demand) in the presence of an inhibitor and expressed as mg L^{-1}. Water column pH was measured in the field by a portable pH meter (Systronics model No. 371; accuracy ± 0.01) having a glass electrode and a calomel electrode as a reference.

Inorganic dissolved nutrients concentration in the water column is a critical parameter in the estuarine systems and such parameters were analysed during the study. The ammonia-nitrogen was determined according to the indophenol blue method (Koroleff, 1969, 1970). In a moderately alkaline medium, ammonia responds with hypochlorite to form monochloramine which in the presence of phenol, the catalytic amount of nitroprussideions and excess hypochlorite forms indophenol blue and measuring the absorbance spectrum of indophenol at λ = 630 nm. This method estimates the sum of NH_{4+} ion and NH_3, and the result is denoted here as NH_4-N (Grasshoff *et al.*, 1999; Grasshoff *et al.*, 1983). Nitrite-nitrogen concentration has measured by the method of (Bendschneider and Robinson (1952); Strickland and Parsons, 1968). In this technique, the nitrite in the water samples was allowed to react with the sulphanilamide and the process known as diazotisation. Then the same solution was permitted to respond with N-(1-Naphthyl) ethylenediamine dihydrochloride. The absorbance of the resultant azo dye was then measured at 543 nm in a spectrophotometer (Grasshoff *et al.*, 1999; Grasshoff *et al.*, 1983).

Nitrate-nitrogen was estimated using the resorcinol method (Jia Zhong and Fischer, 2006). The technique is primarily based on nitration of resorcinol (Benzene-1, 3-diol) in acidified water, resulting in a colour product (Nitrosophenol). The absorption range obtained for the reaction product shows maximum absorption at 505 nm. Dissolved inorganic phosphate-phosphorus was measured using the ascorbic acid method (Grasshoff *et al.*, 1999; Grasshoff *et al.*, 1983; Strickland and Parsons, 1972). In an acid solution with a molybdic acid, ascorbic acid, and trivalent antimony, inorganic phosphate forms a reduced phosphomolybdenum complex, which is blue and absorbance measured at 882 nm. The dissolved silicate-silicon in the water was estimated using the molybdosilicate method. The analysis of dissolved silicate in seawater is based on the formation of a yellowish silicomolybdic acid when an acid sample treated with a molybdate solution, and further, it was reduced by ascorbic acid in the presence of oxalic acid (Grasshoff *et al.*, 1999; Grasshoff *et al.*, 1983).

The water column Chlorophyll-*a* is a measure of productivity in a system and estimated by following the method of The United Nations Educational, Scientific and Cultural Organization (1966). A known volume of water samples was filtered through a Millipore membrane filter (0.45 μm pore size) with $MgCO_3$ suspension. Then the filter dissolved in 10 ml of 90 percentage acetone, and the pigments extracted by placing the tube in a refrigerator for about 10 - 20 hrs with complete darkness afterwards tube centrifuged for 10 minutes at 3000 - 4000 rpm. The extinction of the supernatant solution was measured in the spectrophotometer against a cell containing 90 percentage acetone at 480, 510, 630, 645, 665, and 750 nm. The concentration of pigment has calculated using standard equations.

II. 4. Sediment collection and analysis

Sediment samples have collected at monthly intervals by standard van Veen grab having mouth area of 0.044 m^2. Sediment temperature has recorded through mercury in glass thermometer (zero - 100 ± 0.01 °C) immediately after the collection of sediment samples in the grab. Sediment pH was measured in the field by a portable pH meter (Systronics model No. 371; accuracy ± 0.01) having a glass electrode and a calomel electrode as a reference. The redox potential (Eh) was measured on the field using a portable Eh meter and expressed in mV (model 318). Sediment samples for further analysis were collected (~500 g) in pre-cleaned plastic covers and kept in icebox during transportation to the laboratory. The sediment samples were dried in a hot air oven at 95 °C before sediment texture analysis. Sediment grain size determined through combined sieving and International Pipette method (Carver, 1971; Lewis, 1984). The percentage composition of each grade (sand, silt and clay) was calculated and plotted on triangular graphs based on the terminology suggested by Shepard (1954). Another portion was dried to a constant weight around 60 °C utilised for estimation of organic carbon, by modified wet oxidation method (El-Wakeel and Riley, 1957; Nelson and Sommers, 1982; Trivedy and Goel, 1986). Organic matter content of sediment was calculated by multiplying organic carbon values by Van Bemmelen factor of 1.724 (Trask, 1939). Organic matter expressed as a grams per kilogram ($g.kg^{-1}$) and for comparison, sometimes expressed as a percentage of dry sediment weight examined.

II. 5. The collection, processing, and identification of macrobenthos

The soft-bottom benthic communities can be sampled relatively well by retrieving quantitative samples of sediment and sieving them to separate the fauna. During the two-year period of study, a standard van Veen grab having a bite area of 0.044 m^2 was used to collect sediment samples on a monthly basis in duplicates. After each grab haul, samples were emptied into a plastic tray and well mixed with water. Samples for sediment quality parameters (grain size, sediment pH, Eh, temperature etc.) was treated separately. The larger animals were removed to separate container and rest of the sediment samples sieved through a 0.5 mm (500 μm) mesh sieve until all fine sediments become wash away. The material retained on the sieve has fixed with 5 to 7 percentage of neutral formalin in a plastic bottle containing eight percentages of narcotics such as $MgCl_2$ and stored in a labelled plastic container.

In the laboratory, the sediment was dyed with Rose Bengal biological stain (0.1 g in 100 mL of distilled water) for easy identification of transparent organisms and re-sieved using 0.5 mm mesh sieve to remove the residual sediment, excess stain, and formalin (Holme and Mclntyre 1984, Eleftherious and Anastarious, 2005). The deposit in the sieve was then transferred into Petri dishes. For qualitative enumeration, each sample was then examined under a binocular microscope (Leica DM). The photographs of selected specimens have been taken on a microscope camera (CatCan300) and digital camera (Sony DSC W830). The organisms were separated into different taxonomic groups (malacostracans, polychaetes, molluscs, and other groups) and preserved in 5 to 7 percentage neutral formalin for further analysis. Later each specimen was subjected to detailed identification up to the lowest possible taxonomic level. The number of each organism was enumerated. The numerical abundance was expressed in individuals per meter square (ind.m^{-2}), the live specimens were only considered for the numerical count of the faunal group. Many of the bivalves and gastropods were cut open to confirm staining of biological tissue.

Numerous taxonomic references were used for identification of macrofaunal species along with consultation with experts from various parts of the globe (Beesley *et al.*, 2000; Böggemann, 2002; Böggemann, 2005; Day, 1967; Fauchald, 1977; Fauvel, 1953; Imajima, 1990a, b, c, 1992a; Imajima, 1992b; Maciolek, 1985). Majority of macrofaunal specimens have been identified up to species level, and rest of the fauna has been identified up to possible lowest taxonomic level. Validity and taxonomic status of species have also checked and updated from the World Register of Marine Species (WoRMS, www.marinespecies.org). Before identification, the wet weight of each macrofaunal group was determined by using a high precision electronic balance (Sartorius AG–ME215P, Germany with a precision of 0.01 mg). The biomass of macrofauna was expressed in $g.m^{-2}$. The particular organisms possessing wet weight more than 0.5 g per 0.044 m^{-2} were not extrapolated into 1 m^{-2} instead, taken as such to avoid a biased picture. The wet weight of molluscs was taken with shell, but larger specimens (> 3 g) were not included when considering mean values. The juveniles of epibenthic species such as gobioid fishes, shrimps, molluscs (bivalves, clams and oysters) were also considered as a macrofaunal group in the study. In this study, the term macrofauna was synonymously used for macrobenthos.

II. 6. The collection, processing, and identification of meiobenthos

The meiobenthic sub-samples were collected from the topmost layer (5 cm) of sediment in the van Veen grab hauls using 15 cm long graduated glass corer with an inner diameter of 2.5 cm (Eleftherioo and Mc Intyre, 2005; Giere, 2008). Duplicate core samples were taken at each sampling station from separate grab hauls. Samples were then transferred into separately labelled plastic containers containing eight percentage of $MgCl_2$, four percentage of neutral formalin and transported to the laboratory (Giere, 2008).

In the laboratory, sediment containing the meiobenthos was stained with biological stain Rose Bengal (0.1 %) before sieving for ease of identification of transparent organisms then sieved through two layers of sieves, in the top one with a mesh size of 500 μm and the bottom one with 63-μm mesh size. The filtrate retained in the 63 μm mesh was then transferred into Petri dishes containing water. The animals were classified and enumerated using a binocular microscope (Leica DM) to higher taxon levels (Giere, 2008) and preserved in 4 percentage neutral formalin (Giere, 2008). The numerical abundance of organisms has been extrapolated into number per ten-centimetre square (no.10 cm^{-2}) and their dry weight biomass obtained by multiplying a factor of 0.00045 with a total number of taxa recorded on each sampling site (Ansari, 1989; Ansari *et al.*, 2001). The meiofaunal organisms were identified only up to group level. The animals appearing in small numbers were pooled and designated as 'others group'. In this study, the term meiobenthos has also synonymously been used for meiofauna.

II. 7. Feeding guild analysis

The feeding guilds and functional groups in the benthic communities can be used as an indicator of ecosystem status. In this study, species identified from the estuary has been assigned to different feeding guilds based on the information available on the feeding mode of each taxon with an examination of taxonomic features of the feeding mechanism (Jumars *et al.* 2015). In general, the feeding guild of macrofaunal assemblages is classified into macrophagous and microphagous. Further, the macrophagous group were subdivided into two sub-modes herbivores (HVR) and carnivores (CVR), the microphagous have been subdivided into three sub-modes such as suspension feeders or filter feeders (SF), deposit feeders (DF) and omnivorous (OVR). In this study, deposit feeders were again subdivided into sub-surface deposit feeder (SSDF) and surface deposit feeder (SDF) (Fauchald and Jumars, 1979; Gaston, 1987; Macdonald *et al.*, 2010; MarLIN, 2017). In case of missing information, they were add-on by using information referring to closely related species.

11. 8. Marine biotic indices

Ecological status of the estuary was assessed using the benthic macrofauna based biotic indices such as AZTI's Marine Biotic Index (AMBI), Benthic Opportunistic Polychaetes Amphipods (BOPA), Multivariate AMBI (M-AMBI) and BENTIX.

a) AZTI-Marine Biotic Index (AMBI): The biotic index AMBI was calculated using the software packages AMBI *v*5.0 freely available on the AZTI's website (http://www.azti.es), and it is developed based on the proportion of five ecological groups in the benthic community that was assigned by their sensitivity to disturbances. The five ecological groups (EG) were assigned based on the sensitivity of each species to an increasing gradient of stress in the benthic environment (Borja *et al.*, 2000; Borja *et al.*, 2007). From the 79 taxa identified in the Kondungallure-Azhikode estuary, nine species have been not included in the AMBI list. They were EG1 (species highly sensitive to organic matter enrichment), EG2 (species indifferent to enrichment), EG3 (species tolerant to high organic matter enrichment), EG4 (second-order opportunistic species favoured by excess organic matter enrichment) and EG5 (first-order opportunistic species preferred by excess organic enrichment). Since some of the species identified from the estuary have not been included in the species list of AZTI, the procedure described by Borja *et al.* (2007) has been followed when assigning new species. After the update on species assignment, the percentage of unassigned individuals ranged from zero to 15.9 percentage, with a mean value of 2.79 percentage. This result means that all samples could take into account in the analysis (Borja and Tunberg, 2011). The AMBI index was calculated using following formula:

$$AMBI = 0EG_1 + 1.5EG_2 + 3EG_3 + 4.5EG_4 + 6EG_5$$

The AMBI index can vary from zero (high ecological status) to seven (bad ecological status). The values between 0 to 1.2 represent the undisturbed condition and that for the slightly disturbed condition was 1.2 to 3.3, moderately disturbed ranged from 3.3 to 5, heavily disturbed was between 5 to 6 and extremely disturbed conditions denote value between 6 to 7 in AMBI index (Borja *et al.*, 2000).

b) Benthic opportunistic polychaetes amphipods index (BOPA): The benthic opportunistic polychaetes amphipods index (BOPA) index is an improved version of the benthic opportunistic polychaetes amphipods ratio proposed by Gesteira and Dauvin (2000). It takes into account the total number of individuals collected, the frequency of opportunistic polychaetes, and the frequency of amphipods, except the genus *Jassa* because they are part of the EG_5 on the AZTI list (Dauvin and Ruellet, 2007). The BOPA index was calculated using the following formula:

$$\text{BOPA index} = \log\left(\frac{f_P}{f_A}+1\right)$$

Where, f_P is the opportunistic polychaete frequency (ratio of the total number of opportunistic polychaete individuals to the total number of individuals in the sample) and f_A, the amphipod frequency (ratio of the total number of amphipod to the total number of individuals in the sample). The values can be varied from 0.30103 (when $f_A = 0$) to zero (when $f_P = 0$). The values indicate the status of an environment into five different classes from extremely polluted or azoic sites to unpolluted condition. The values between 0.30103 to 0.25512 indicate bad ecological status while from 0.25512 to 0.19884 indicate poor, that for moderate disturbance was 0.19884 to 0.13002, good condition by 0.13002 to 0.02452 range and less than 0.02452 indicate the high ecological status of the soft bottom macrobenthic communities (Dauvin and Ruellet, 2007; De-La-Ossa-Carretero and Dauvin, 2010).

c) BENTIX: The BENTIX index has been designed for the assessment of the impact caused by general stress factors and does not discriminate amongst natural and anthropogenic disturbances (Simboura and Zenetos, 2002). To calculate the BENTIX index, the same ecological groups were used with some proportional difference, EG_1 and EG_2 were placed in G_I, and EG_3, EG_4, and EG_5 were in G_2 (Simboura and Zenetos, 2002).

The BENTIX index was calculated using the following formula:

$$\text{BENTIX} = 6G_I + 2G_{II}$$

Where, $G_I = EG_1 + EG_2$ and $G_2 = EG_3 + EG_4 + EG_5$. The results for the BENTIX index can vary from zero (bad ecological status) to six (high ecological status). The value less than 2 indicate the bad ecological condition of an ecosystem while between 2 to 2.5 poor and that for moderate condition ranged between 2.5 to 3.5 while good condition among 3.5 to 4.5 and normal or pristine environment indicated by the value between 4.5 to 6 in the soft bottom macrobenthic communities (Simboura and Zenetos, 2002).

d) M-AMBI (Multivariate AMBI): The multimetric M-AMBI index was calculated using AMBI, species richness and the Shannon index, combined with the use in further development of factor analysis together with discriminant analysis (Bald *et al.*, 2005; Muxika *et al.*, 2007). This method compared monitoring results with reference situation (Borja and Tunberg, 2011) and was computed using AMBI software *v*5.0. As M-AMBI needs reference conditions to be calculated, and the KAE shows salinity gradient, that can determine the benthic communities living at each salinity stretch (Muxika *et al.*, 2007), each station was assigned to a salinity gradient to assess their reference conditions. Hence, in the absence of pristine areas, the reference conditions were determined by increasing 15 percentages upon the highest diversity and richness values of all replicate (Borja and Tunberg, 2011). As for the bad status, the references based upon the azoic situation (diversity and richness equal to 0 and AMBI equal to 6. Taking into account, the salinity stretches, and the pressures, the reference conditions for M-AMBI were set as follows: (i) in the station seven and six (limno-oligo-meso-polyhaline stretch) with a diversity of 3.55 and richness 14.95; (ii) in the stations five, four, three and two (oligo-meso-polyhaline stretch) with a diversity of 3.66 and richness 19.55 and (iii) in the stations one (meso-poly-euhaline stretch) with diversity 3.42 and richness 13.80. In all cases, AMBI was zero.

II. 9. Statistical methods and analysis

Study on benthic biocoenosis requires detection of specific patterns of statistical interactions in data sampled during various periods. Therefore, it forms a complicated process because benthic assemblages and population size vary in space and time. In this regard, the different kinds of diversity indices have widely been used in the bioecological study of benthic communities and environmental monitoring programs. The efficiency of these indices varies, depending on the type of research and other aspects of the sampling strategies. In the present study, different approaches were adopted for the analysis of diversity and community pattern. Which includes univariate (species richness, Shannon diversity index, species dominance and evenness, taxonomic diversity index / taxonomic distinctness index), multivariate (multi-dimensional scaling) and graphical (species-area plots, K-dominance curves, Canonical Correspondence Analyses, Abundance-Biomass Comparison (ABC) curves) methods. The software packages such as PRIMER *v*6.1.9 (Plymouth Routines in Multivariate Ecological Research) with add-on package PERMANOVA+*v*1 (Permutational Multivariate Analysis of Variance), SPSS/PASW Statistics *v*18.0 (Statistical Programme for Social Sciences version *v*18.0), CANOCO *v*4.5 and AMBI *v*5.0 (AZTI Marine Biotic Index), ORIGIN *v*8.0, Golden Software Surfer *v*11.0, MapInfo Professional *v*11.0 has been used for statistical analysis and graphical representation of data collected during the study. The one-way analysis of variance (ANOVA) was used to determine significant differences between the groups.

To verify, whether the number of species collected was adequate for the description of community structure in the estuary, species accumulation curve was plotted, which gives the cumulative number of species recorded as a function of sampling effort of grab. The different species estimators in PRIMER *v*6.1.9 such as Sobs (Curve of observed species counts), Chao1 (Chao's estimator based on number of rare species), Chao2 (Chao's estimator using just presence-absence data), Jackknife1 (based on species that only occur in one sample), Jackknife2 (Second-order jackknife estimator), Bootstrap (based on proportion of quadrats containing each species), MM (Michaelis-Menton-Curve fitted to observed S curve) and UGE (Calculated species accumulation curve) were used to predict the true number of species that would be observed as the numbers of samples that tend to be infinity (Clarke and Gorley, 2006).

Univariate diversity indices have sometimes been treated as much effective as multivariate methods and in the present study, the statistical package PRIMER *v*6 was used for univariate measurement such as Shannon diversity index (H’), Margalef’s index (d) Pielou’s index (J’), Simpson's index (λ’), average taxonomic distinctness index (Δ+) and variation in taxonomic distinctness (Λ+) of benthic communities in the estuary. The Shannon diversity index (H') was calculated from $\log_2$ transformed data on benthic assemblages in the estuary which explains both abundance and evenness of species present in the community (Shannon and Weaver, 1949). The index value will be high in samples that have large numbers of unique species or have greater species evenness. The species richness was tested by Margalef's index (d), and it measures the number of species present for a given number of individuals (Margalef, 1958). While species equitability was tested by Pielou's index (J'), species equitability or evenness shows how evenly the individuals have been distributed among the different species, and species dominance shows the dominance of particular species among a given number of individuals (Pielou, 1966). The Simpson's index (λ') is a measure of both the richness and proportion (percentage) of each species (Simpson, 1949).

Multivariate statistical analysis such as Bray-Curtis similarity index and nonmetric multidimensional scaling (nMDS), similarity percentages (SIMPER), similarity profile test (SIMPROF), Analysis of Similarity (ANOSIM), K dominance plot, Abundance Biomass Comparison (ABC) curve, BIO-ENV Analysis, Principal Component Analysis (PCA). PCO (Principal Coordinates Analysis), RELATE, CAP (Canonical Analysis of Principal coordinates) in PERMANOVA+*v*1 has also been performed (Anderson *et al.*, 2008). Statistical analysis was conducted on square root transformed data to reduce the impact of dominant groups of before analysis in the PRIMER *v*6.

Bray-Curtis similarity analysis (PRIMER *v*6.1.9) was calculated with suitable transformation (square root) for the species-abundance data to group the samples with similar community composition following the procedure described by Clarke and Warwick (1994). Bray-Curtis similarity index and average group linkage were used for cluster analysis and non-metric multi-dimensional scaling (n-MDS) ordination (Ludwig & Reynolds 1988). To compare the biodiversity between the stations and seasons, a stress value of < 0.2 gives a useful representation of results (Clarke and Warwick, 2001). The dominance plot (in PRIMER *v*6.1.9) was drawn by ranking the species in decreasing order of their abundance. Relative abundance expressed as 'percentage of abundance' in the sample was plotted, against the increasing rank in the x-axis, the latter on a log scale. This is used to find out the pollution effects on macrobenthos (Clarke, 1990); the J-shaped curve representing the dominance of opportunistic species (disturbed condition) whereas S-shaped curve indicates the occurrence of conservative species (undisturbed state).

Similarity Profile Analysis (SIMPROF) test was carried out for detecting statistically significant clusters (Clarke and Gorley, 2006). It conducts a series of permutation tests to determine whether groups in the dendrogram have statistically significant structure. Analysis of Similarity (ANOSIM) significance test was performed to provide proof of substantial differences between two or more groups of sampling units. Here, the significance level was calculated by referring to 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, denotes the clear distinction between samples (Clarke *et al.*, 2006). Similarity Percentage Analysis (SIMPER) assesses the average percentage contribution of individual variables to the dissimilarity between the objects in a Bray-Curtis dissimilarity matrix. That allows observing the variables that are important in contributing any similarity/difference between groups detected by methods such as ANOSIM.

Abundance Biomass Comparison curve (ABC curve): used to evaluate disturbances based on the trend of Abundance Biomass Comparison 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 strictly 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) was used to statistically define the relationship between trajectories 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 towards zero.

Principal component analysis (PCA) was also conducted on environmental data to detect trends of variation of ecological characteristics across the study area (Jolliffe, 2002). This analysis also uses an ordination plot to project the points of higher similarities closer together while samples more dissimilar are further apart. Unlike biological data, environmental data have mixed estimation scales, and similarity methods, such as normalised Euclidean distance used in PCA, are more suited for environmental data (Clarke and Gorley, 2006). A useful exercise before performing PCA is to examine the environmental data in a Draftsman's scatter plot to ascertain whether there are variables that are highly correlated with one another, which may then be omitted from the PCA. In this study, significant environmental variables measured have been included for the PCA. The Kaiser Rule was used to selecting the number of components from PCA. Similarly, attempts to represent the distances between samples were plotted using principal coordinates analysis (PCO) in the in PERMANOVA +*v*1 (Anderson *et al.*, 2008). In particular, it maximises the linear correlation between the distances in the distance matrix and the distances in the space of low dimension. Although PCO is based on a distance matrix, the solution can be found by eigen-analysis, and when the distance metric is Euclidean, PCO is equivalent to principal component analysis. Spatial variability of macrofaunal groups was further analysed using canonical analysis of principal coordinates (CAP) on the sum of squared canonical correlations as a constrained ordination and discrimination method, to determine whether there was any significant difference between station and season groups according to environmental conditions.

Canonical correspondence analysis (CCA) was conducted using the software package CANOCO *v*4.5 based on the subset of environment parameters identified by Pearson correlation analysis (ter Braak and Smilauer, 2002). The CCA is a linear function of the two sets of variables (abiotic and biotic) so that the correlation between the two functions maximised (Poore and Mobley, 1980; ter Braak and Smilauer, 2002). Geometrically, the method looks at the relative positioning of the subjects in the two-dimensional space, the variables with the highest coefficients in each of these linear functions have assumed to define that function. Hence, the key features relating the two data sets may assess from a pair of coefficient vectors (Poore and Mobley, 1980). The CCA plot was useful in determining which environmental factor that influenced the distribution of the selected macrofaunal species. Monte Carlo permutation test (with forward selection) was used to test the significance of environmental variables that explained the variance of species distribution and abundance ($p < 0.05$ level).

Chapter 3

Hydrography and sediment characteristics

III.1. Introduction

Estuaries are the critical transition zones that connect land, river, and the sea, subject to unpredictable hydrological, morphological and chemical conditions (Day *et al.*, 2012). These are the most complex and dynamic aquatic ecosystems comprising of interacting physicochemical and biological component, whose dynamics are often combined (Dutertre *et al.*, 2012). Estuaries are also characterised by having low salinities, shallow depths, high turbidities, excess nutrients and increased productivity (Donald and Michael, 2004). The highly variable physicochemical conditions due to the mixing of marine and freshwater create stress to most organisms in the estuary. Thus, many species have developed adaptations to live in estuarine conditions. Several studies suggest that estuarine species distributions are mainly driven by salinity gradient (McLusky and Elliott, 2004). In addition to this, other environmental features such as sediment particle size, organic matter content, depth, dissolved oxygen, hydrodynamic conditions and vegetation cover may also determine species distribution (van Houte-Howes *et al.*, 2004). The interaction of an organism with these environmental circumstances determines the size of its population and distribution, and the further existence of different communities in this complex ecosystem. Therefore, a comprehensive knowledge of various physicochemical parameters is imperative to document the biocenosis of estuarine flora and fauna.

In general, abiotic factors control the broad distributional patterns of benthic communities at a larger scale, while abiotic and biotic factors drive together at a minimum level. However, such a non-isolated marine ecosystems like estuaries are also susceptible to the influence of human activities (Dutertre *et al.*, 2012). The distribution and diversity of estuarine benthic communities are primarily influenced by physicochemical factors including turbidity, temperature, salinity, oxygen concentrations, current energy, substrate composition, sedimentation rates, bathymetry and food supply (Gogina *et al.*, 2010; Harkantra and Parulekar, 1981; McLusky, 1999). These factors influence the functional biology of the benthic organisms by individually or in combination (Kinne, 1963). In many studies, sediment properties are recognised as important factors responsible for distribution of benthic organisms (Hily *et al.*, 2008). However, on broad spatial scales, other natural environmental determinants such as the hydrodynamic conditions and physicochemical properties of the water column influence directly or indirectly in presence and abundance of benthic species assemblages (Bolam *et al.*, 2008). Because, these are the principal controlling factors for transport and distribution of sediment, food supply, larval dispersion and metabolism of benthic organisms (Pearson and Rosenberg, 1987). The hydrodynamic factors and physicochemical properties of water vary over shorter temporal scales while sediment characteristics and depth are relatively stable over time.

Estuarine benthic communities are often restricted to particular sections of environmental gradients, resulting in well-developed distribution patterns (Wolff, 1983). The spatial heterogeneity of macrobenthos along the estuarine gradient was traditionally defined in connection with salinity and sediment composition (Ysebaert *et al.*, 1998). Warwick *et al.* (1991) pointed out the importance of both dynamic processes (tidal range and wave fetch distance) and unvarying factors (sediment grain size and organic content), in determining the community structure of macrobenthos. Other studies also highlighted the significance of hydrodynamic processes resulting from currents and waves (such as bed shear stress) for the transportation and disposal of sediment, food and juvenile macrobenthos (Alf *et al.*, 2001). Therefore, the information on habitat characteristics linked with a species is a fundamental factor to understand the bioecology of benthos (Johnson, 1971; Kurbjeweit *et al.*, 2000). Recent studies have shown a complex interaction between hydrodynamics, sediment dynamics and benthic biology in structuring distribution patterns of benthos (Herman *et al.*, 2001; Nebra *et al.*, 2016). Therefore these complex factors such salinity, dissolved oxygen, grain size, organic content, food supply, trophic interaction and so many factors can affect the distribution of benthos, but there are no single mechanisms to predict these complex benthic assemblages (Snelgrove and Butman, 1994). In this regard, this chapter describes the importance of abiotic features in detail to relate them with the bioecology of benthic communities in the Kodungallur-Azhikode estuary (KAE) supported by respective tables and figures.

III. 2. Hydrography and Meteorology

The estuaries are highly complex aquatic systems that are heavily influenced by the precipitation and rivers discharge pattern, particularly in monsoonal estuaries, the riverine discharge pattern shaping the entire bioecology. The data on the hydrographic conditions of Kodungallur-Azhikode estuary (KAE) is described [Table 1 a-b].

III. 2. 1. Rainfall

The mean rainfall in the catchment area of KAE was 297 ± 281 mm during the entire study period (2009-2011) [Figure 4]. Annual mean precipitation was highest during the second year (2010-2011) period (306 ± 270 mm) as compared to the first year (2009-2010) period (289 ± 29mm). Seasonally, monsoon season of both years showed peak rainfall, and it was 652 ± 257 mm in the first-year period and that for the second year was 557 ± 197 mm, followed by 100 ± 63 mm in post-monsoon of the second year and the first year (98 ± 88 mm). The pre-monsoon seasons showed least precipitation rate, and that was 134 ± 97 mm in the first year and 244 ± 224 mm in the second year. A marked difference was recorded in rainfall between the seasons (ANOVA F (5,162) = 57.98, p =0.000).

III. 2. 2. River discharge

River discharge data was collected from the gauges situated in the Chalakudy and Periyar River, which nourish the KAE such as Neeleswaram gauge located in northern distributary channel of Periyar river basin and Arangaly gauge in the Chalakudy basin. The mean river discharge in the entire study period was 252 ± 267 $m^3 S^{-1}$, and it was highest during the first year (261 ± 292 $m^3 S^{-1}$) period as compared to the second year (244 ± 240 $m^3 S^{-1}$) period [Figure 4].

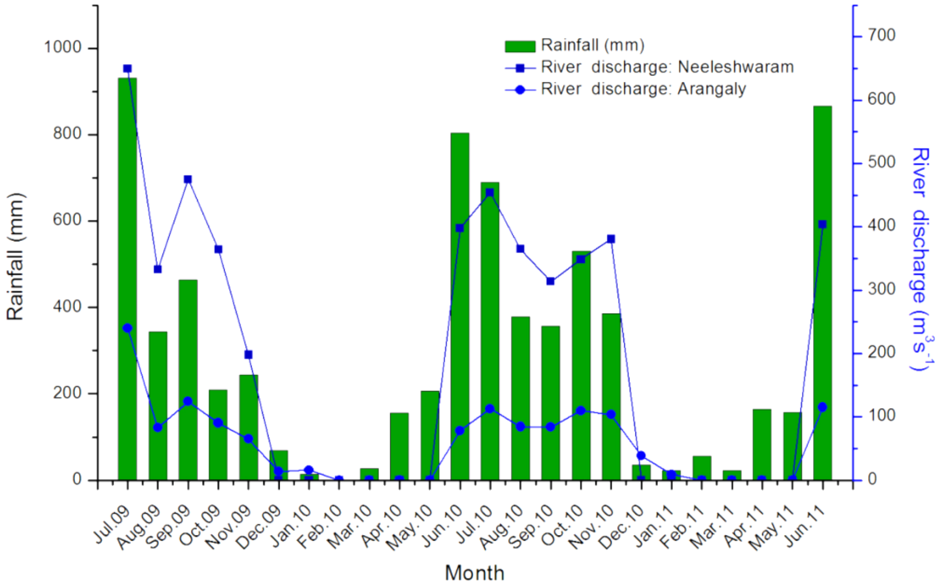


Figure 4. Mean monthly rainfall and river discharge in Kodungallur-Azhikode estuary (KAE) during the study period (2009 to 2011)

Seasonally, monsoon seasons showed large discharge pattern in relation with precipitation rate in the upper basins, and that was 606 ± 180 m^3 S^{-1} for the first-year and 473 ± 62 m^3 S^{-1} for the second year period. The post-monsoon season showed intermediate values of 248 ± 228 m^3 S^{-1} during the second year, and that was 187 ± 188 m^3 S^{-1} in the first year. However, no significant river discharge pattern was observed during the pre-monsoon seasons. Inter-annual mean river discharge from Neeleswaram gauge showed much higher value (195 ± 209 m^3 S^{-1}) when compared to Arangaly gauge (57 ± 60 m^3 S^{-1}). Annual mean discharge from the Neeleswaram gauge was 201 ± 226 m^3 S^{-1} during the first year, and that was 189 ± 192 m^3 S^{-1} during the second year. Similarly, in the mean discharge of Arangaly gauge was 59 ± 69 m^3 S^{-1} during the first year and that for the second year period was 55 ± 49 m^3 S^{-1}. Seasonal mean value of river discharge depicted a maximum quantity in monsoon seasons at both Neeleswaram (465 ± 120 m^3 S^{-1} in 2009-2010 & 383 ± 52 m^3 S^{-1} in the second year) and Arangaly gauges (141 ± 61 m^3 S^{-1} in 2009-2010 and 90 ± 14 m^3 S^{-1} in 2010-2011). However, in the post-monsoon period, an intermediate discharge pattern was recorded for Neeleswaram (141 ± 155 m^3 S^{-1} in 2009-2010 and 182 ± 186 m^3 S^{-1} in 2010-2011) and Arangaly gauges (46 ± 33 m^3 S^{-1} in 2009-2010 and 65 ± 44 m^3 S^{-1} in 2010-2011). The significant variation observed in river discharge pattern between seasons (ANOVA F (5,162) = 82.41, p =0.000).

III. 2. 3. Depth

The mean depth of selected seven stations in the KAE during the entire period of study was 3.63 m, and it varied from the highest depth of 4.43 ± 0.75 m in the station one (estuarine mouth) to lowest of 2.23 ± 0.51 m in the station five (northernmost station). The mean depth of other stations were 4.12 ± 0.46 m in station two, four ± 0.41m in station three, 2.91 ± 0.57 m in station four, 4.23 ± 0.41 m in station six and 3.53 ± 0.39 m in station seven [Figure 5]. A significant variation in was depth observed between the stations (ANOVA F (6,161) = 58.67, p =0.000).

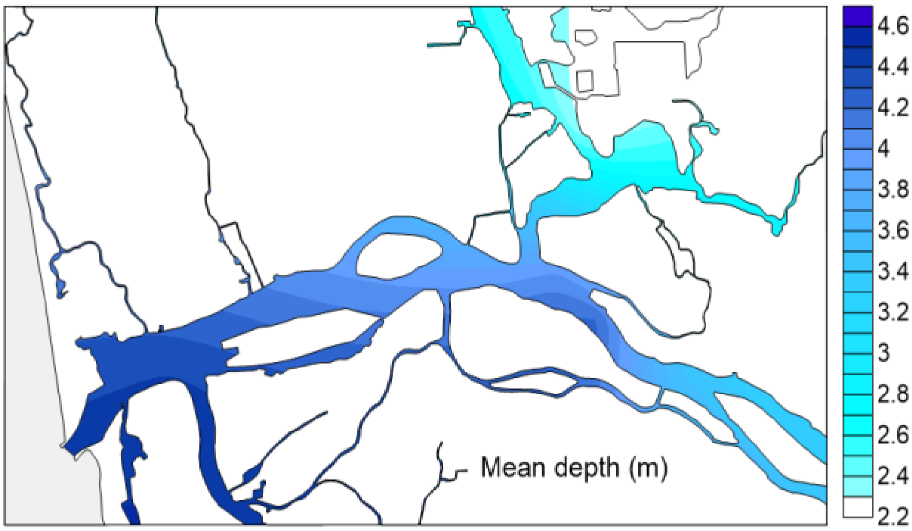


Figure 5. Mean depth (m) gradient in KAE during 2009 to 2011 period

III. 2. 4. Temperature

The mean bottom water temperature during the entire study was 28.45 ± 1.87 °C and that for the first year was 28.78 ± 2.16 °C and 28.15 ± 1.49 °C in the second year period [Figure 6 a-b]. Seasonal mean temperature showed a peak value during pre-monsoon periods such as 30.32 ± 1.12 °C in the first year and 29.64 ± 0.71 °C in the second year. The mean lowest seasonal value was recorded in post-monsoon of the second year (27.14 ± 1.08 °C). That for other seasons were 27.16 ± 2.15 °C in the monsoon of the first year, 28.04 ± 1.79 °C in the monsoon of the second year, and 28.38 ± 1.44 °C in post-monsoon of the first year. Station-wise highest value of 28.90 ± 1.84 °C was recorded at station 5 followed by 28.65 ± 1.94 °C at station six, 28.53 ± 1.98 °C at station seven. The mean lowest bottom water temperature of 27.83 ± 20 °C was recorded at station one and that for other stations were 28.30 ± 1.78 °C at station two, 28.46 ± 1.90 °C at station three and 28.48 ± 1.71 °C at station four. Significant variations were observed in water temperature between seasons (ANOVA F (5,162) = 22.29, p =0.000).

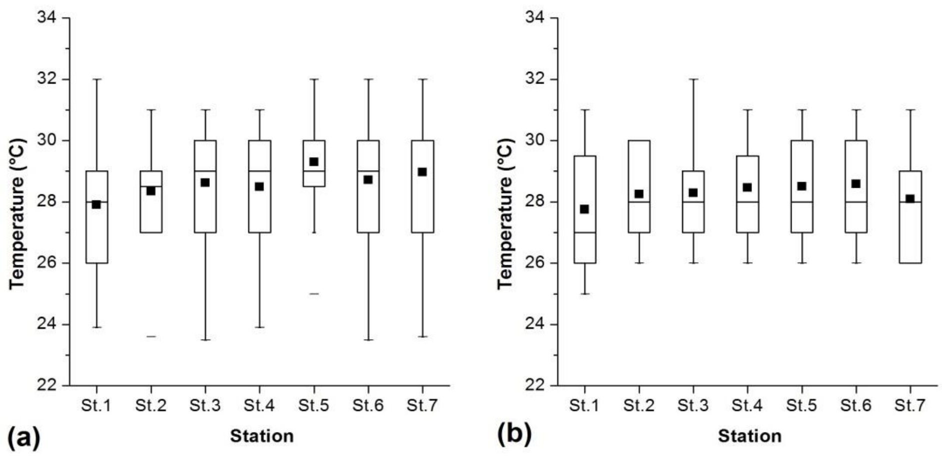


Figure 6 a-b. Box plot of bottom water temperature in different stations of KAE during (a) 2009-2010 and (b) 2010-2011 period, (whisker: range, box: interquartile range, square: mean)

III. 2. 5. Turbidity

The turbidity values expressed a wide variation in the bottom waters. The inter-annual mean of bottom turbidity value for KAE was 9.60 ± 11.46 NTU, and during the first year it was 11.65 ± 14.95 NTU and that for the second year it was 7.55 ± 5.70 NTU [Figure 7 a-b & c-h]. Seasonally, monsoon seasons showed highest mean values of 21.11 ± 20.58 NTU during the first year and that for the second year was 12.56 ± 9.20 NTU. Similarly, post-monsoon showed lowest at values of 3.04 ± 1.33 NTU in the first year, and that was 5.00 ± 3.00 NTU in the second year of study. Station-wise data depicted higher values in station one (13.38 ± 13.94 NTU), station five (10.46 ± 9.98 NTU) and station six (10.06 ± 9.11 NTU).

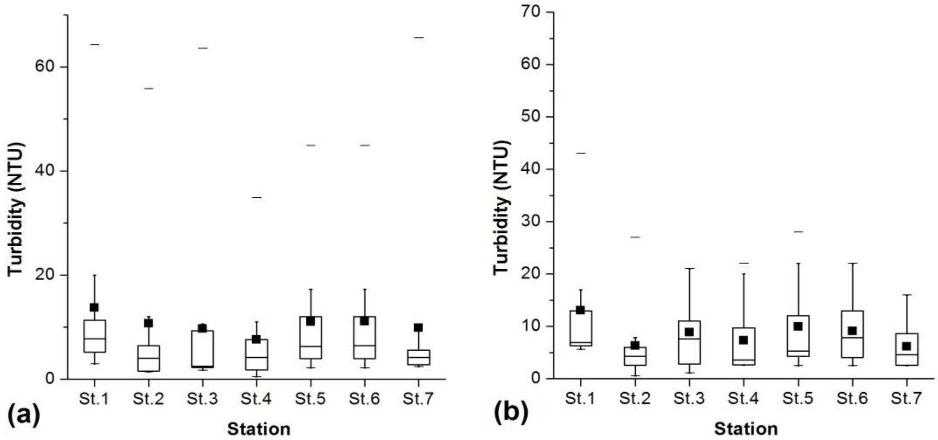


Figure 7 a-b. Box plot of bottom water turbidity in different stations of KAE during (a) 2009-2010 and (b) 2010-2011 period, (whisker: range, box: interquartile range, square: mean)

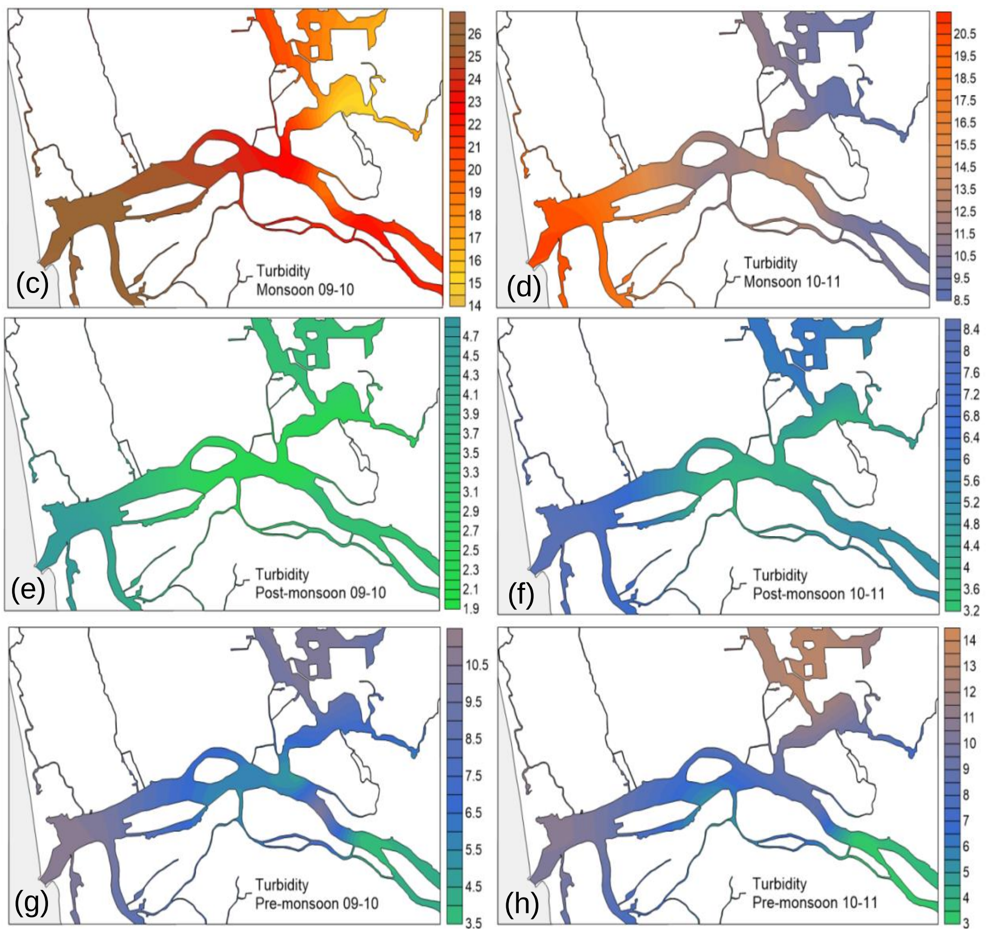


Figure 7 c-h. Spatial variation of bottom water turbidity (NTU) in KAE during the different seasons of entire study period (2009-2011)

The comparatively low turbidity values were recorded at station four (7.45 ± 7.97 NTU), station seven (8.02 ± 12.72 NTU) and station two (8.52 ± 12.66 NTU). The significant variation was observed between seasons (ANOVA $F(5,162) = 12$, $p = 0.000$).

III. 2. 6. Transparency (Secchi Depth)

The mean water column transparency in the estuary was 0.98 ± 0.44 m, and it was 0.99 ± 0.46 m in the first year and that for the second year was 0.97 ± 0.42 m [Figure 8 a-b]. Seasonally, mean lowest transparency values have been recorded during the monsoon seasons, and it was 0.66 ± 0.34 m in the first year and 0.72 ± 0.31 m during the second year period. Comparatively, higher transparency values were observed during the post-monsoon season of the first year (1.32 ± 0.38 m) and the second year (0.96 ± 0.38 m). During the pre-monsoon period of the first year depicted value of 1.04 ± 0.37 m and that for the second year was 1.19 ± 0.46 m. Spatially, station five exhibited lowest mean water transparency of 0.79 ± 0.35 m followed by 0.90 ± 0.46 m at station four, 0.91 ± 0.39 m at station one, 1 ± 0.50 m at station three. Highest values were recorded at station six (1.12 ± 0.43 m), station two (1.08 ± 0.39 m) and station one (1.06 ± 0.49 m). The significant variations were noticed between seasons (ANOVA $F(5,162) = 13.40$, $p = 0.000$).

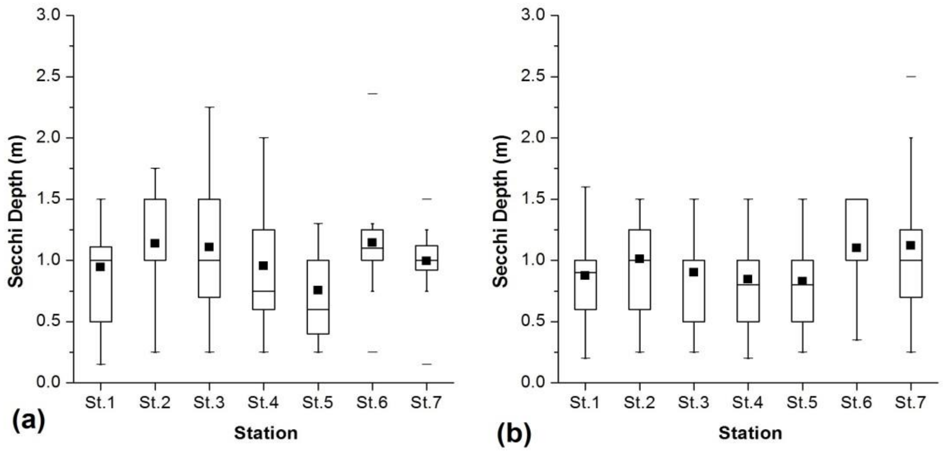


Figure 8 a-b. Box plot of Secchi depth in different stations of KAE during (a) 2009-2010 and (b) 2010-2011 period, (whisker: range, box: interquartile range, square: mean)

III. 2. 7. Salinity

The mean bottom salinity value for the entire estuary was 15.78 ± 9.90 PSU [Figure 9 a-f]. Annually, the first year period exhibited slightly high mean salinity value (16.90 ± 9.35 PSU), when compared to the second year period (14.66 ± 10.35 PSU).

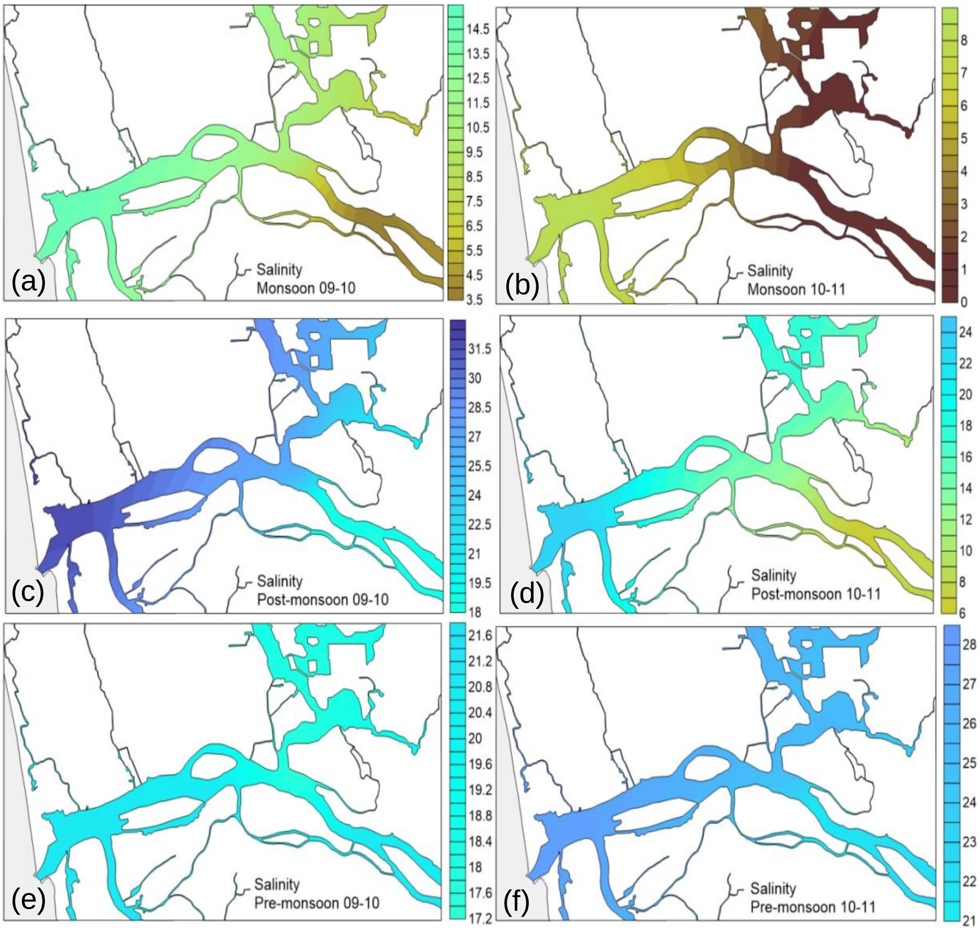


Figure 9 a-f. Spatial variation of bottom water salinity (PSU) in KAE during the different seasons of entire study period (2009-2011)

Seasonally, monsoon seasons showed a decreasing trend with the lowest mean value of 2.60 ± 4.47 PSU during the monsoon of the second year period and that for the first year period was 9.41 ± 6.42 PSU. The relatively high salinity values were recorded during post-monsoon of the second year (24.60 ± 5.49 PSU) and pre-monsoon of the first year (24.55 ± 5.78 PSU). Station-wise data depicted a mean lowest value of 11.78 ± 9.90 PSU at station seven and 12.52 ± 9.26 PSU at station six while highest at station one (21.15 ±

9.75 PSU), station two (17.61 ± 9.50 PSU) and station five (16.37 ± 9.77 PSU). A marked significant variation was observed in salinity between stations (ANOVA $F_{(6,161)} = 2.49$, p =0.025) and seasons (ANOVA $F_{(5,162)} = 59.94$, p =0.000) [Figure 9 a-f & g-h].

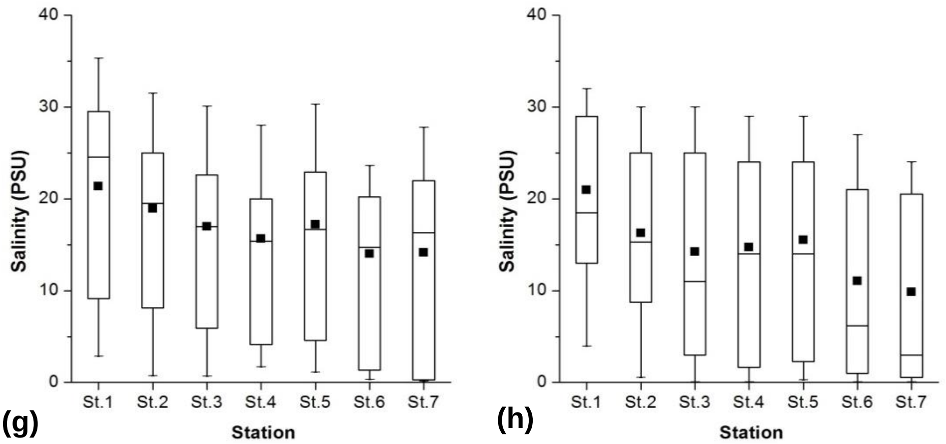


Figure 9 g-h. Box plot of bottom water salinity in different stations of KAE during (g) 2009-2010 and (h) 2010-2011 period, (whisker: range, box: interquartile range, square: mean)

III. 2. 8. pH

The mean bottom water pH for the entire estuary was 7.35 ± 0.54, and it was 7.41 ± 0.45 during the first year and that for the second year period was 7.29 ± 0.62 [Figure 10 a-b].

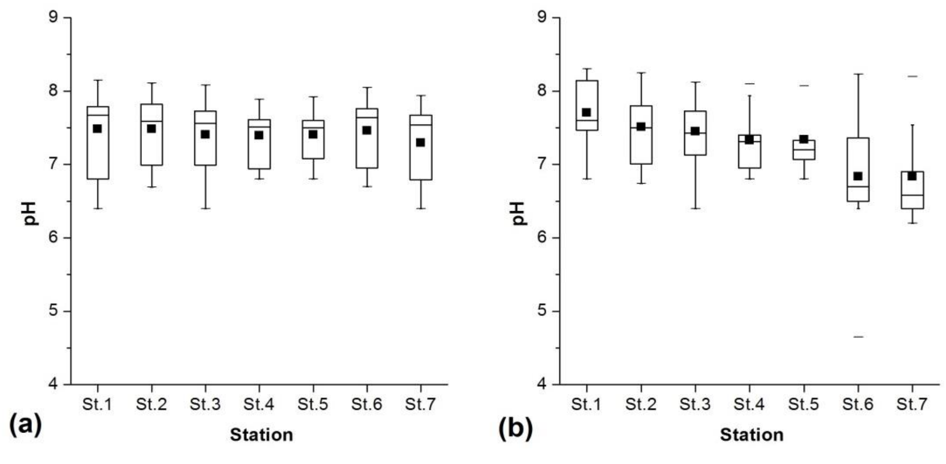


Figure 10 a-b. Box plot of bottom water pH in different stations of KAE during (a) 2009-2010 and (b) 2010-2011 period, (whisker: range, box: interquartile range, square: mean)

Seasonally, monsoon season exhibited a slightly acidic condition such as 6.88 ± 0.61 in the second year, and it was 6.89 ± 0.29 in the first year while highest value of 7.78 ± 0.27 was recorded during post-monsoon of the first year and followed 7.58 ± 0.09 in pre-monsoon of the same year. Spatially, station seven (7.07 ± 0.60) and station six (7.14 ± 0.78) exhibited lowest pH values, and that was comparatively higher in station one (7.59 ± 0.49) and station two (7.49 ± 0.48) due to the influence of the sea. A clear significant variation of water column pH was noticed between stations (ANOVA $F\,(5,161) = 3.04$, $p = 0.008$) (ANOVA $F_{2,\,168} = 3.044$, $p = 0.008$) and seasons (ANOVA $F\,(5,162) = 20.07$, $p = 0.000$).

III. 2. 9. Redox potential (Eh)

The mean bottom water redox potential (Eh) value for KAE was -15.99 ± 35.67 during the entire study and it was -4.76 ± 37.61 mV in the first year period and that for the second year was -27.21 ± 29.85 mV [Figure 11 a-b].

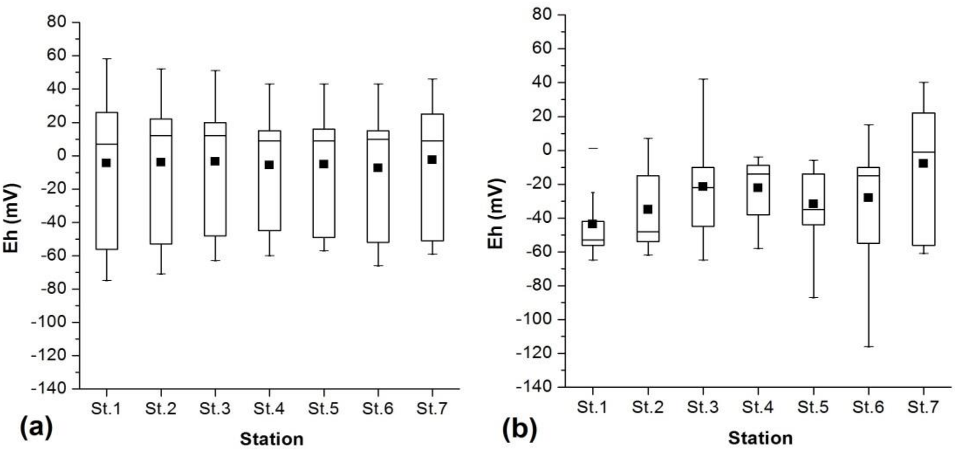


Figure 11 a-b. Box plot of bottom water Eh in different stations of KAE during (a) 2009-2010 and (b) 2010-2011 period, (whisker: range, box: interquartile range, square: mean)

Seasonally, pre-monsoon (-43.43 ± 25.36 mV), post-monsoon (-30.59 ± 23.34) of the second year and pre-monsoon (-26.75 ± 44.52) of the first-year recorded comparatively reduced condition when compared to the monsoon of the first year (15.86 ± 11.35 mV) second year (-5.500 ± 30.40 mV) and post-monsoon of the first year (-5.50 ± 35.29 mV. Spatially, station one (-24.00 ±

39.08 mV), station two (-19.58 ± 37.74 mV), station five (-18.56 ± 32.10 mV) showed comparatively reduced water column that compared to station 7 (-5.25 ± 39.07 mV), station three (-12.63 ± 34.74 mV) and station four (-14.08 ± 28.21 mV). Bottom water redox potential showed a significant variation between stations (ANOVA $F(5,162) = 13.42$, $p = 0.000$).

III. 2. 10. Dissolved oxygen

The average bottom water dissolved oxygen (DO) for the entire estuary was 4.94 ± 1.11 mg L^{-1} and that for the first year was 4.67 ± 1.08 mg L^{-1} and in the second year it was 5.21 ± 1.09 mg L^{-1} [Figure 12 a-b & c-h].

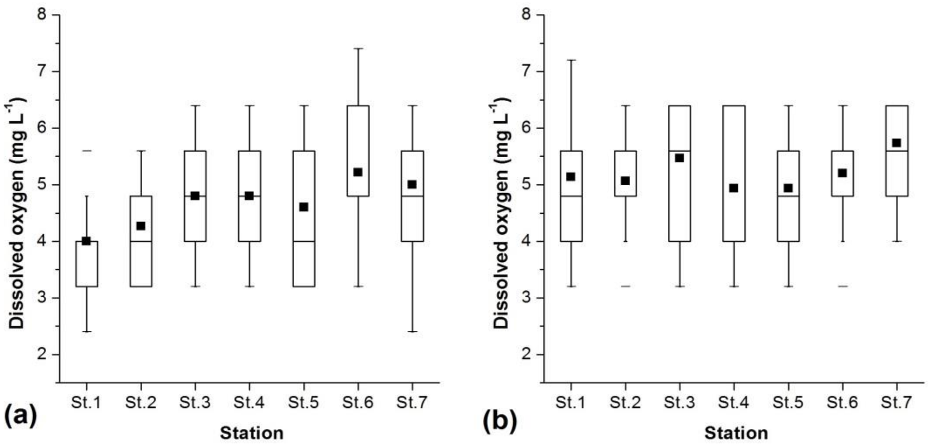


Figure 12 a-b. Box plot of bottom water DO in different stations of KAE during (a) 2009-2010 and (b) 2010-2011 period, (whisker: range, box: interquartile range, square: mean)

Seasonally, monsoon season (5.66 ± 0.87 mg L^{-1}) and post-monsoon of the second year (5.043 ± 0.88 mg L^{-1}) showed peak dissolved oxygen values while lowest values recorded at pre-monsoon (4.20 ± 0.83 mg L^{-1}) and post-monsoon (4.43 ± 0.86 mg L^{-1}) of the first year period. Spatially, comparatively low mean value was observed at station one (4.57 ± 1.30 mg L^{-1}), station two (4.67 ± 0.90 mg L^{-1}), station five (4.77 ± 1.07 mg L^{-1}) and station four (4.87 ± 1.05 mg L^{-1}). However, relatively high dissolved oxygen content has observed at riverine station seven (5.37 ± 1.11 mg L^{-1}), station six (5.21 ± 1.10 mg L^{-1}) and station three (5.13 ± 1.15 mg L^{-1}). A significant seasonal variability on DO was observed (ANOVA $F(5,162) = 11.21$, $p = 0.000$).

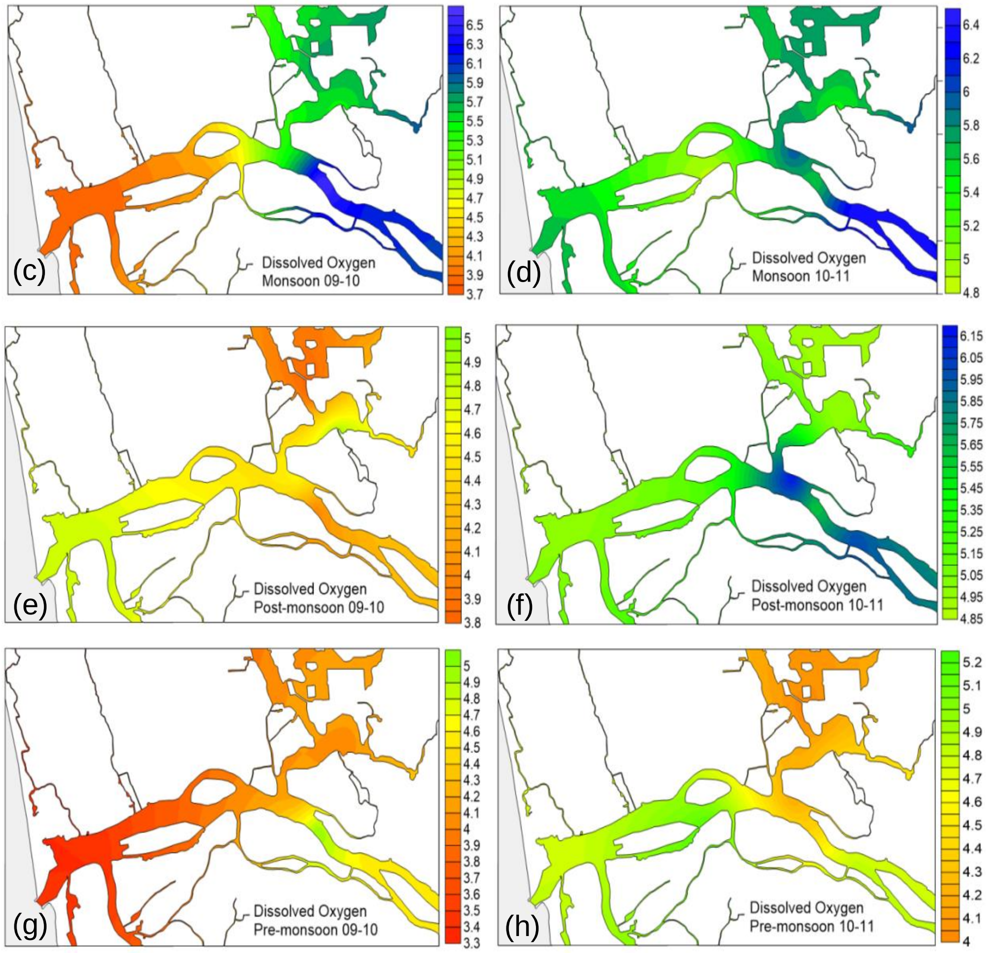


Figure 12 c-h. Spatial variation of bottom water dissolved oxygen (mg L^{-1}) in KAE during the different seasons of entire study period (2009-2011)

III. 2. 11. Biological oxygen demand

The average bottom water biological oxygen demand (BOD) value for the entire estuary was 2.54 ± 1.42 mg L^{-1} and that for the first year was 2.54 ± 1.22 mg L^{-1} and in the second year was 2.53 ± 1.62 mg L^{-1}. The monsoon seasons of both years exhibited lowest BOD value, such as 1.86 ± 1.36 mg L^{-1} in the second year and 2.20 ± 1.06 mg L^{-1} in the first year. However, post-monsoon of the second year (3.11 ± 1.14 mg L^{-1}), monsoon (2.77 ± 1.20 mg L^{-1}), and post-monsoon (2.66 ± 1.34 mg L^{-1}) of the first year demonstrated comparatively high BOD level. The lowest values of BOD has recorded at station six (2.23 ± 1.73 mg L^{-1}), and station five (2.33 ± 1.18 mg L^{-1}), while highest values at station one (2.83 ± 1.29 mg L^{-1}) and station seven (2.80 ± 1.73 mg L^{-1}). There was no significant difference in BOD value observed between months and seasons [Figure 13 a-b].

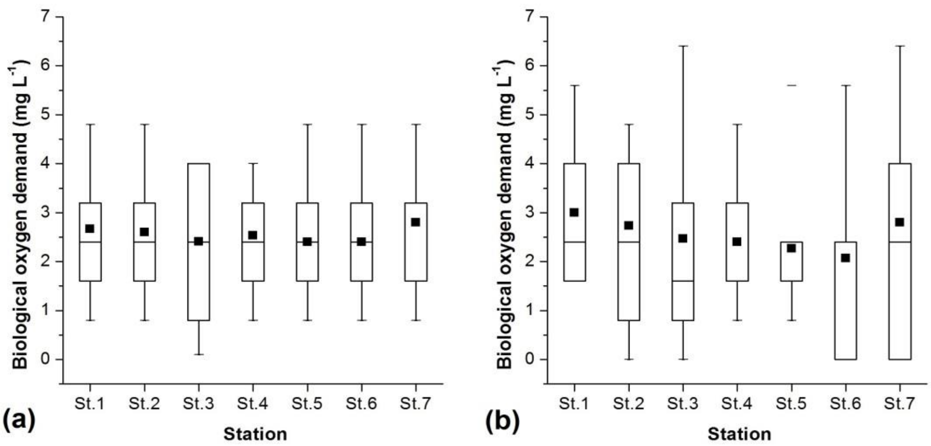


Figure 13 a-b. Box plot of bottom water BOD in different stations of KAE during (a) 2009-2010 and (b) 2010-2011 period, (whisker: range, box: interquartile range, square: mean)

III. 2. 12. Dissolved nutrients

Dissolved nutrients such as ammonia, nitrite, nitrate, phosphate, and silicate were monitored during the study and described below.

a) Dissolved ammonia-nitrogen

The average bottom water dissolved ammonia-nitrogen (NH_4-N) in the entire estuary was 3.89 ± 2.92 µmol L^{-1}, and it was 4.26 ± 3.10 µmol L^{-1} in the first year and 3.53 ± 2.71 µmol L^{-1} in the second year. The pre-monsoon seasons of the first year period was 5.74 ± 2.56 µmol L^{-1} and 5.18 ± 3.76 µmol L^{-1} in the second year that exhibited highest values. While in the post-monsoon period of the first year (2.87 ± 2.46 µmol L^{-1}) and the monsoon of the second year (2.94 ± 2.13 µmol L^{-1}) depicted the lowest concentration. Spatially, station one (4.47 ± 3.31 µmol L^{-1}) and station six (4.38 ± 4.48 µmol L^{-1}) showed highest peak value. However, stations seven (2.70 ± 2.12 µmol L^{-1}) and station three (3.88 ± 2.87 µmol L^{-1}) exhibited the lowest concentration [Figure 14 a-b]. The values between seasons (ANOVA F (5,162) = 5.95, p =0.000) showed a significant variation.

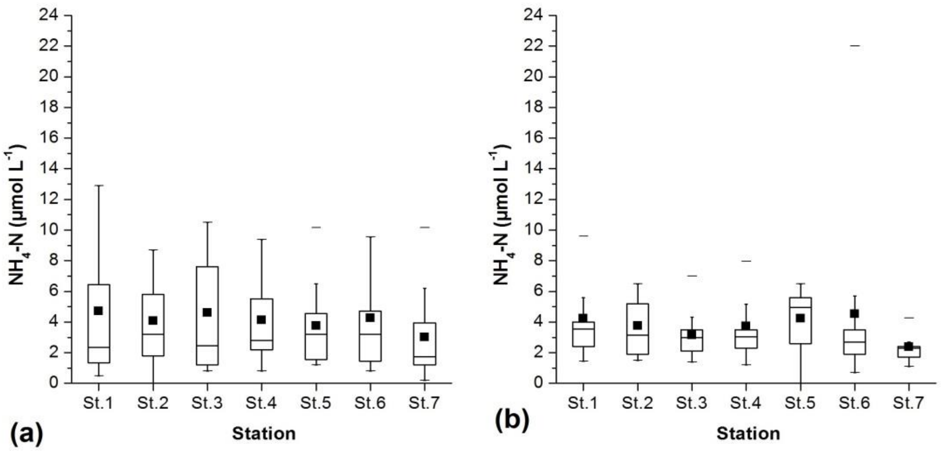


Figure 14 a-b. Box plot of bottom dissolved ammonia in different stations of KAE during (a) 2009-2010 and (b) 2010-2011 period, (whisker: range, box: interquartile range, square: mean)

b) Dissolved nitrite-nitrogen

The mean dissolved nitrite-nitrogen (NO_2-N) concentration for the entire estuary was 0.38 ± 0.30 µmol L^{-1}, and that was 0.36 ± 0.30 µmol L^{-1} in the first year and 0.40 ± 0.31 µmol L^{-1} in the second year [Figure 15 a-b].

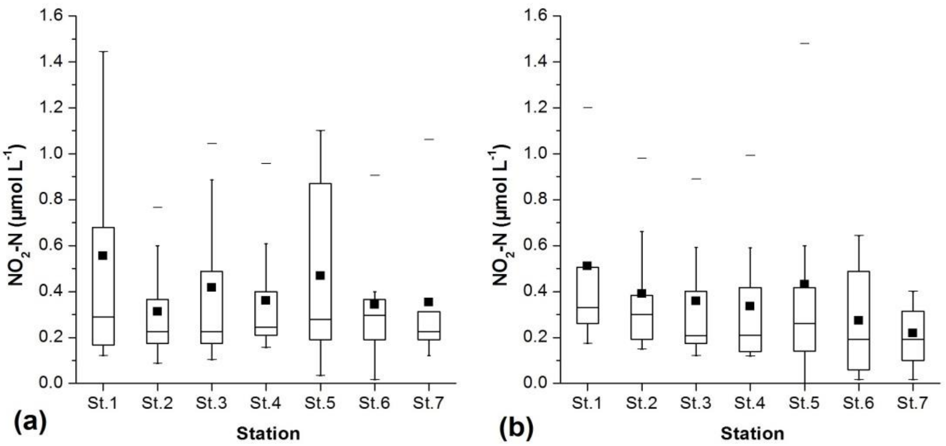


Figure 15 a-b. Box plot of bottom dissolved nitrite in different stations of KAE during (a) 2009-2010 and (b) 2010-2011 period, (whisker: range, box: interquartile range, square: mean)

Seasonally, pre-monsoon of the second year (0.57 ± 0.33 µmol L^{-1}) and first year (0.50 ± 0.32 µmol L^{-1}) showed the highest concentration while post-monsoon of the first year (0.23 ± 0.06 µmol L^{-1}) and the monsoon of the second year (0.24 ± 0.13 μmol L^{-1}) recorded the lowest concentration. In the spatial scale, station one (0.53 ± 0.40 μmol L^{-1}) and station five (0.45 ± 0.39 μmol L^{-1}) exhibited higher concentration in the same time station seven (0.29 ± 0.24 μmol L^{-1}), and station six (0.31 ± 0.23 μmol L^{-1}) was marked least. A significant variation was observed in the concentration between seasons (ANOVA F (5,162) = 8.24, p =0.000).

c) Dissolved nitrate-nitrogen

The overall mean of bottom water dissolved nitrate-nitrogen (NO_3-N) concentration for the estuary was 9.55 ± 11.67 μmol L^{-1}, and it was 10.79 ± 15.59 μmol L^{-1} during the first year and 8.32 ± 5.28 μmol L^{-1} during the second year of study [Figure 16 a-b]. The monsoon seasons of the first year (22.22 ± 22.16 μmol L^{-1}) and the second year (9.40 ± 4.78 μmol L^{-1}) exhibited the highest concentration while pre-monsoon of the first year (3.98 ± 3.82 μmol L^{-1}) and post-monsoon of the second year (6.52 ± 4.27 μmol L^{-1}) depicted the lowest concentration. Spatially, station seven (11.36 ± 18.95 μmol L^{-1}) records the highest value while that was lowest in the station 2 (7.57 ± 7.23 μmol L^{-1}). Bottom water dissolved nitrate concentration in the estuary showed a significant variation between seasons (ANOVA F (5,162) = 11.32, p =0.000).

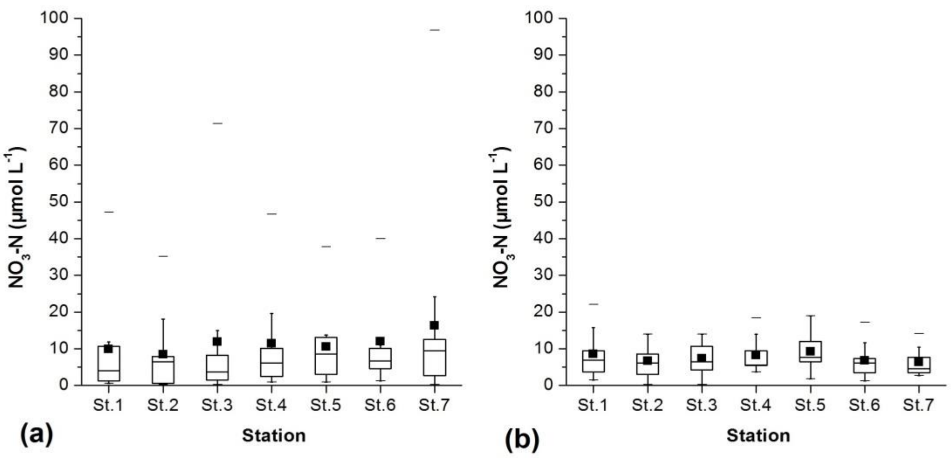


Figure 16 a-b. Box plot of bottom dissolved nitrate in different stations of KAE during (a) 2009-2010 and (b) 2010-2011 period, (whisker: range, box: interquartile range, square: mean)

d) Dissolved phosphate - phosphorus

The mean dissolved phosphate-phosphorus (PO_4-P) concentration in the estuary was 1.03 ± 1.67 µmol L^{-1} and that for the first year was 1.28 ± 2.26 µmol L^{-1} and that for the second year was 0.79 ± 0.59 µmol L^{-1} [Figure 17 a-b]. The pre-monsoon seasons of both first year (2.33 ± 3.64 µmol L^{-1}) and second year (1.15 ± 0.72 µmol L^{-1}), exhibited the highest concentration. Monsoon of the second year (0.52 ± 0.32 µmol L^{-1}), post-monsoon of both first year (0.54 ± 0.25 µmol L^{-1}) and second year (0.55 ± 0.37 µmol L^{-1}) recorded least concentrations. Spatially, station five (1.55 ± 3.61 µmol L^{-1}), and station one (1.32 ± 0.80 µmol L^{-1}) depicted the higher concentrations while station 7 (0.51 ± 0.28 µmol L^{-1}) and station four (0.71 ± 0.53 µmol L^{-1}) recorded least values. The concentration of phosphate showed a significant difference between seasons (ANOVA F (5,162) = 5.56, p =0.000).

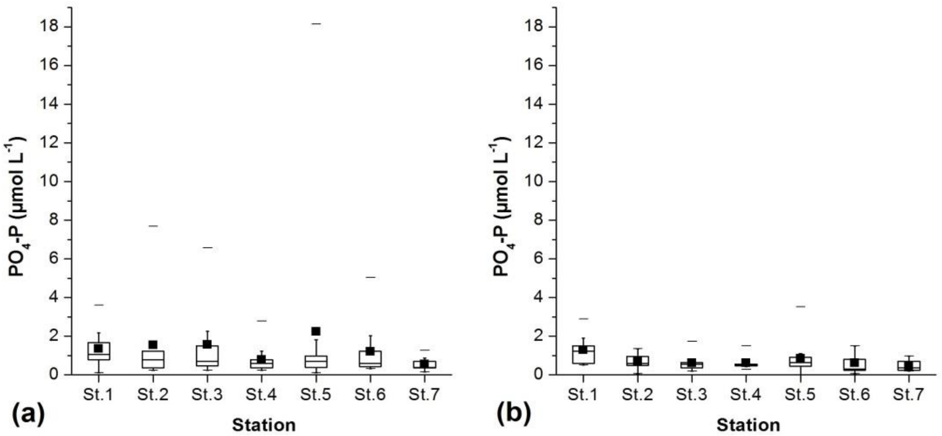


Figure 17 a-b. Box plot of bottom dissolved phosphate in different stations of KAE during (a) 2009-2010 and (b) 2010-2011 period, (whisker: range, box: interquartile range, square: mean)

e) Silicate-silicon

The mean bottom silicate concentration for the estuary was 47.42 ± 25.73 µmol L^{-1}, and it was 43.44 ± 28.79 µmol L^{-1} during the first year and 51.40 ± 21.70 µmol L^{-1} in the second year period [Figure 18 a-b]. The monsoon season first year (60.68 ± 43.34), and the second year (56.74 ± 22.52) depicted the maximum concentrations while post-monsoon of the second year (34.62 ±

14.45 µmol L^{-1}), post-monsoon of the first year (47.60 ± 22.42 µmol L^{-1}) was recorded with least values. In spatial scale, station five (51.49 ± 27 µmol L^{-1}) and station two (48.80 ± 28.91 µmol L^{-1}) were recorded with higher concentrations while station 1 (45.16 ± 25.19) and station seven (45.69 ± 22.72 µmol L^{-1}) exhibited low concentrations. A significant variation of bottom water silicate was observed between seasons (ANOVA F(5,162) = 6.54, p =0.000).

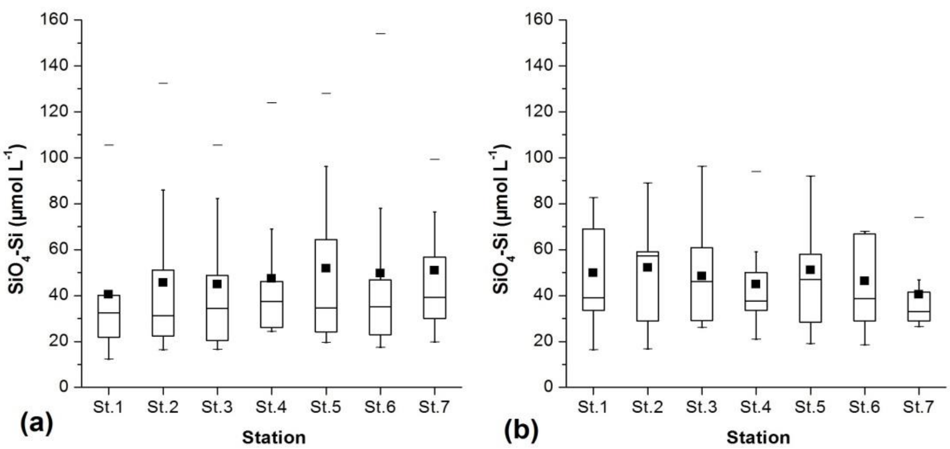


Figure 18 a-b. Box plot of bottom dissolved silicate in different stations of KAE during (a) 2009-2010 and (b) 2010-2011 period, (whisker: range, box: interquartile range, square: mean)

III. 2. 13. Chlorophyll-*a*

The mean bottom water Chlorophyll-*a* (Chl-*a*) in the Kodungallur-Azhikode estuary was 5.99 ± 4.48 mg m^{-3}, and it was 6.54 ± 5.15 mg m^{-3} during the first year and 5.42 ± 3.64 mg m^{-3} for the second year period of study [Figure 19 a-b, c-h]. The pre-monsoon season of both first year (9.04 ± 4.72 mg m^{-3}) and second year (8.16 ± 4.26) mg m^{-3}) period noticed a high concentration of Chl-*a*, while post-monsoon (3.81 ± 1.50 mg m^{-3}) and monsoon (4.27 ± 3.37 mg m^{-3}) of the second-year depicted least values. In a spatial scale, station five (7.73 ± 5.30 mg m^{-3}) and station seven (6.47 ± 5.47 mg m^{-3}) has recorded with high concentrations and that for station 2 (4.99 ± 3.24 mg m^{-3}) and station four (5.16 ± 3.46 mg m^{-3}) were having least value. Chlorophyll-*a* concentration in the bottom water showed a significant variation between seasons (ANOVA F (5,162) = 8.05, p = 0.000).

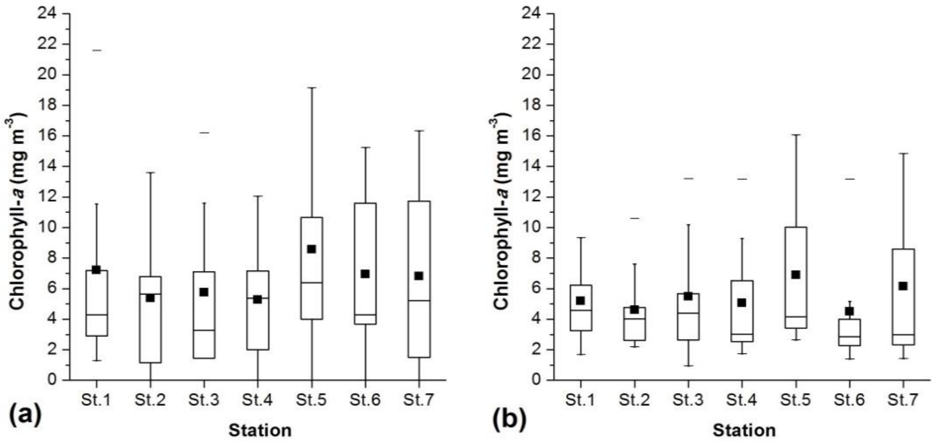


Figure 19 a-b. Box plot of bottom chlorophyll-*a* in different stations of KAE during (a) 2009-2010 and (b) 2010-2011 period, (whisker: range, box: interquartile range, square: mean)

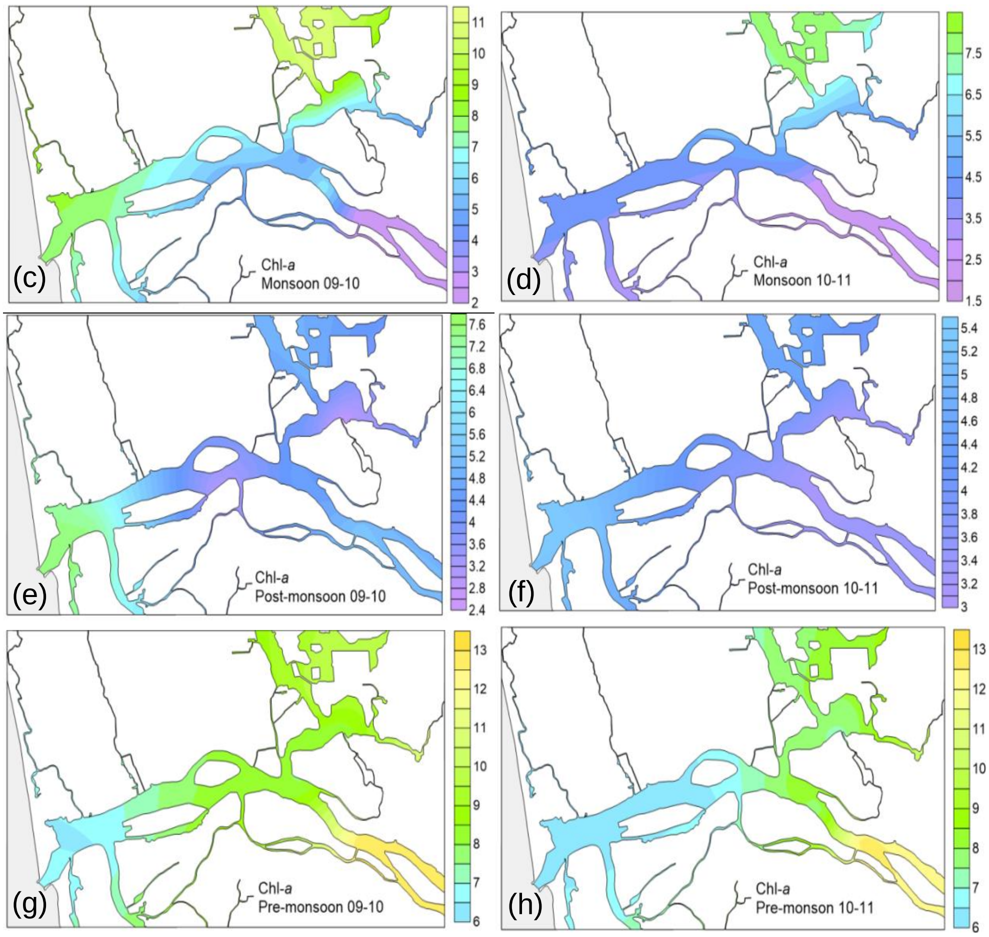


Figure 19 c-h. Spatial variation of bottom water chlorophyll-*a* (mg m^{-3}) in KAE during the different seasons of entire study period (2009-2011)

Table 1 a. Seasonal mean of hydrographic parameters (± SD) during the 2009 to 2010 period

Parameters	Monsoon	Post-monsoon	Pre-monsoon
Salinity (PSU)	9.41 ± 6.42	24.55 ± 5.78	19.38 ± 4.04
Temperature (°C)	27.16 ± 2.15	28.38 ± 1.44	30.32 ± 1.12
Transparency (m)	0.66 ± 0.34	1.32 ± 0.38	1.04 ± 0.37
Turbidity (NTU)	21.11 ± 20.58	3.04 ± 1.33	7.42 ± 4.64
pH	6.89 ± 0.29	7.78 ± 0.27	7.58 ± 0.09
DO (mg L^{-1})	5.38 ± 1.16	4.43 ± 0.86	4.2 ± 0.83
BOD (mg L^{-1})	2.77 ± 1.2	2.66 ± 1.34	2.2 ± 1.06
Eh (mV)	15.86 ± 11.35	-5.5 ± 35.29	-26.75 ± 44.52
Nitrate (µmol L^{-1})	22.22 ± 22.16	8.35 ± 8.01	3.98 ± 3.82
Nitrite (µmol L^{-1})	0.47 ± 0.4	0.23 ± 0.06	0.5 ± 0.32
Phosphate (µmol L^{-1})	1.11 ± 0.78	0.54 ± 0.25	2.33 ± 3.64
Silicate (µmol L^{-1})	60.68 ± 43.34	47.6 ± 22.42	33.52 ± 8.49
Ammonia (µmol L^{-1})	3.63 ± 3.39	2.86 ± 2.46	5.74 ± 2.56
Chl-*a* (mg m^{-3})	6.06 ± 4.64	4.58 ± 4.92	9.04 ± 4.72

Table 1 b. Seasonal mean of hydrographic parameters (± SD) during the 2010 to 2011 period

Parameters	Monsoon	Post-monsoon	Pre-monsoon
Salinity (PSU)	2.6 ± 4.47	14.11 ± 8.1	24.6 ± 5.49
Temperature (°C)	28.04 ± 1.79	27.14 ± 1.08	29.64 ± 0.71
Transparency (m)	0.72 ± 0.31	0.95 ± 0.38	1.19 ± 0.46
Turbidity (NTU)	12.56 ± 9.2	5 ± 3	8.46 ± 7.27
pH	6.89 ± 0.61	7.42 ± 0.56	7.55 ± 0.5
DO (mg L^{-1})	5.66 ± 0.87	5.43 ± 0.88	4.54 ± 1.17
BOD (mg L^{-1})	2.63 ± 2.02	3.11 ± 1.14	1.86 ± 1.36
Eh (mV)	-5.5 ± 30.4	-30.59 ± 23.34	-43.43 ± 25.36
Nitrate (µmol L^{-1})	9.4 ± 4.78	6.52 ± 4.27	6.86 ± 3.78
Nitrite (µmol L^{-1})	0.24 ± 0.13	0.28 ± 0.24	0.57 ± 0.33
Phosphate (µmol L^{-1})	0.52 ± 0.32	0.55 ± 0.37	1.15 ± 0.72
Silicate (µmol L^{-1})	56.74 ± 22.52	34.62 ± 14.45	51.38 ± 16.33
Ammonia (µmol L^{-1})	2.94 ± 2.13	3 ± 1.39	5.18 ± 3.76
Chl-*a* (mg m^{-3})	4.27 ± 3.37	3.81 ± 1.5	8.16 ± 4.26

III. 3. Sediment Characteristics

The data on sediment characteristics such as temperature, pH, Eh, texture, organic carbon and organic matter are described below [Table 2 a-b].

III. 3. 1. Temperature

The mean sediment temperature for the entire study period was 28.55 ± 1.70 °C, and that was 28.60 ± 1.99 °C in the first year and that was 28.50 ± 1.36 °C in the second year [Figure 20 a-b and Table 2 a-b]. The pre-monsoon seasons of both years depicted maximum value, such as 30.14 ± 1.18 °C in the first year and 29.82 ± 0.93 °C in the second year. While monsoon of the first year (27.31 ± 1.93 °C), post-monsoon of the second year (27.46 ± 0.97 °C) and post-monsoon first year (28.21 ± 1.45 °C) has exhibited least values. Spatially, station five (29.06 ± 1.43 °C) and station seven (28.77 ± 1.88 °C) depicted the higher values and station 1 (28.90 ± 1.68 °C) and station three (28.48 ± 1.83 °C) at lower side. A significant difference was observed in the sediment temperature between seasons (ANOVA $F(5,162) = 22.61$, $p = 0.000$).

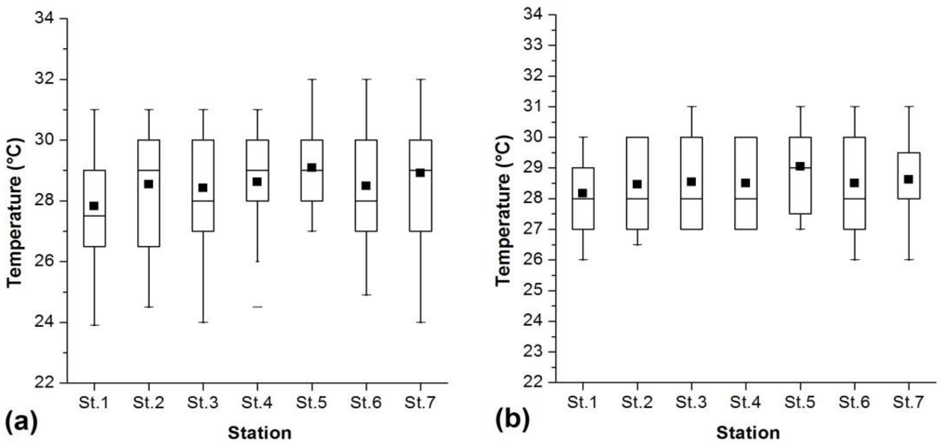


Figure 20 a-b. Box plot of sediment temperature in different stations of KAE during (a) 2009-2010 and (b) 2010-2011 period, (whisker: range, box: interquartile range, square: mean)

III. 3. 2. pH

The sediment pH was generally on an alkaline side with the overall mean of 7.81 ± 0.61, and that was 7.94 ± 0.71 in the first year and 7.69 ± 0.45 for the second year [Figure 21 a-b and Table 2 a-b].

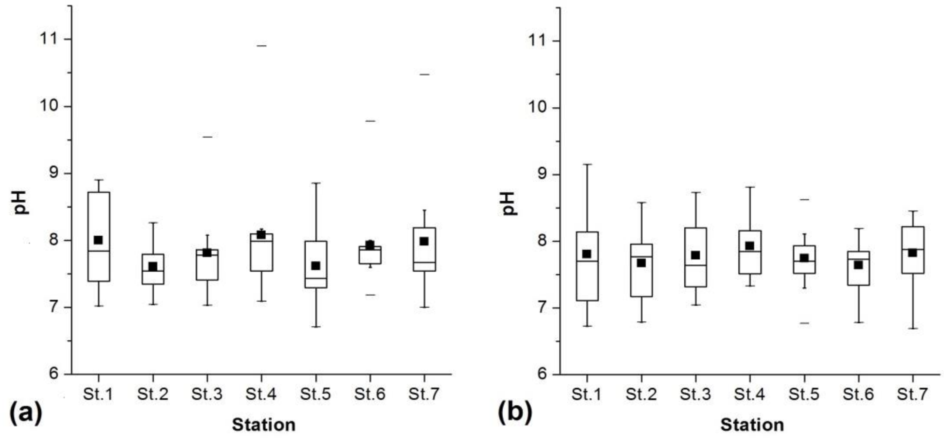


Figure 21 a-b. Box plot of sediment pH in different stations of KAE during (a) 2009-2010 and (b) 2010-2011 period, (whisker: range, box: interquartile range, square: mean)

Seasonally monsoon season of the first year (8.31 ± 0.91) and second year (8.18 ± 0.41) depicted higher values. Post-monsoon of the first year (7.51 ± 0.48), pre-monsoon (7.56 ± 0.51), and post-monsoon of the second year (7.57 ± 0.33) were at the lower side. Spatially, station four (8.0 ± 0.73), station one (7.9 ± 0.67) and station seven (7.9 ± 0.72) were in higher side while station two (7.64 ± 0.44), station five (7.68 ± 0.53) and station six (7.78 ± 0.54) depicted lower pH values. There was a significant differences observed between seasons (ANOVA $F\,(5,162) = 11.84$, p =0.000) and stations (ANOVA $F\,(6,161) = 1.08$, p =0.000).

III. 3. 3. Redox potential (Eh)

The sediment redox potential (Eh) values showed a reducing trend in all stations with a mean value for the entire study area was -92.55 ± 78 mV and was -93.32 ± 81.54 in the first year and -92.58 ± 76.51 for the second year period of study [Figure 22 a-b and Table 2 a-b]. Seasonally, comparatively higher oxidised sediment was noticed in the monsoon period both years, such as -53.43 ± 54.67 in the second year and -71.25 ± 8.50 in the first year of monsoon and that followed by post-monsoon of the second year (-83.7 ± 85.93). However, post-monsoon of the first year demonstrated highly reduced sediment with a value of -135.71 ± 54.16 and that followed by pre-monsoon of the first year (-107.64 ± 94.17). Spatial variation of sediment Eh showed highly reduced environment in station 6 (-120.38 ± 74.43), station seven (-118.63 ± 70.29), station two (-116.33 ± 89.47) and station five (-91.75 ± 76.78). However, station four (-81.45 ± 73.78) and station three (-82.0 ± 74) depicted relatively low reduced condition of sediment. A significant variation in sediment Eh was observed between seasons (ANOVA F (5,162) = 4.18, p =0.001) and stations (ANOVA F(6,161) = 3.51, p =0.003).

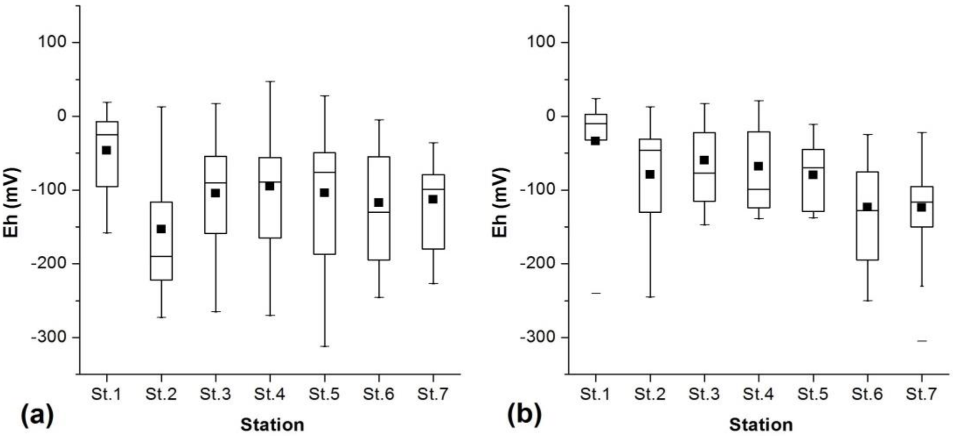


Figure 22 a-b. Box plot of sediment Eh in different stations of KAE during (a) 2009-2010 and (b) 2010-2011 period, (whisker: range, box: interquartile range, square: mean)

III. 3. 4. Sediment texture

The Kodungallure Azhikode estuary (KAE), which receives 70 percent of river discharge from Periyar River and the sediment load from these rivers have an important role in the textural composition of the sediment in the estuary. Sediment texture in the KAE was characterised by the abundance of sand and silt with a minor fraction of clay [Figure 23]. In the present study, sand content in sediment depicted overall mean of 81.61 ± 16.59 percent and that for the first year was 85.81 ± 13.20 percent while 77.40 ± 18.54 percent in the second year. The monsoon season for both years exhibited highest sand fraction in sediments such as 85.62 ± 13.73 percent in the first year and 85.30 ± 13.10 percent for the second year. The sand fraction of sediment reduced during pre-monsoon (74.52 ± 20.07 %) and post-monsoon of the second year (75.00 ± 19.52 %) period of the study. Spatially, station four (91.15 ± 5.45 %), station three (88.18 ± 11.74 %), station one (87.50 ± 12.39 %) and station five (86.90 ± 4.06 %) were exhibiting comparatively higher sand fraction when compared to other stations such as station six (58.10 ± 18 %), station two (76.99 ± 16.55 %) and station seven (82.43 ± 15.08 %). While silt fraction of sediment depicted a mean value of 10.55 ± 12.79 percent for the entire Kodungallure Azhikode estuary and that was 7.14 ± 10.59 percent in the first year while 13.96 ± 13.92 percent in the second year period.

Figure 23. The mean percentage composition of sediment texture and its spatial distribution pattern in KAE during the study

Seasonally, the post-monsoon season of the first year (16.36 ± 14.90 %) and pre-monsoon of the second year (15.92 ± 15.56 %) depicted comparatively high silt content in sediment. Monsoon (6.71 ± 9.06 %) and pre-monsoon (7.76 ± 12.62 %) of the first year showed least silt fraction. Spatially, silt content was high at station six (26.05 ± 15.82 %) and station two (13.13 ± 14.4 %) whereas lowest fraction at station four (3.73 ± 2.95 %) and station three (5.99 ± 10.17 %) [Figure 24 a-b, 25a-b and Table 2 a-b]. Clay fraction of sediment depicted mean value of 7.85 ± 6.76 percent with 7.05 ± 5.77 percent in the first year and that for the second year was 8.64 ± 7.56 %. Pre-monsoon (9.55 ± 6.48 %) and post-monsoon (8.64 ± 8.27) season of the second year demonstrated a high percentage of clay whereas post-monsoon (7.25 ± 3.87 %) and monsoon (7.67 ± 6.88 %) of the first year showed the least fraction of clay [Figure 24 a-i, 25a-b and Table 2 a-b].

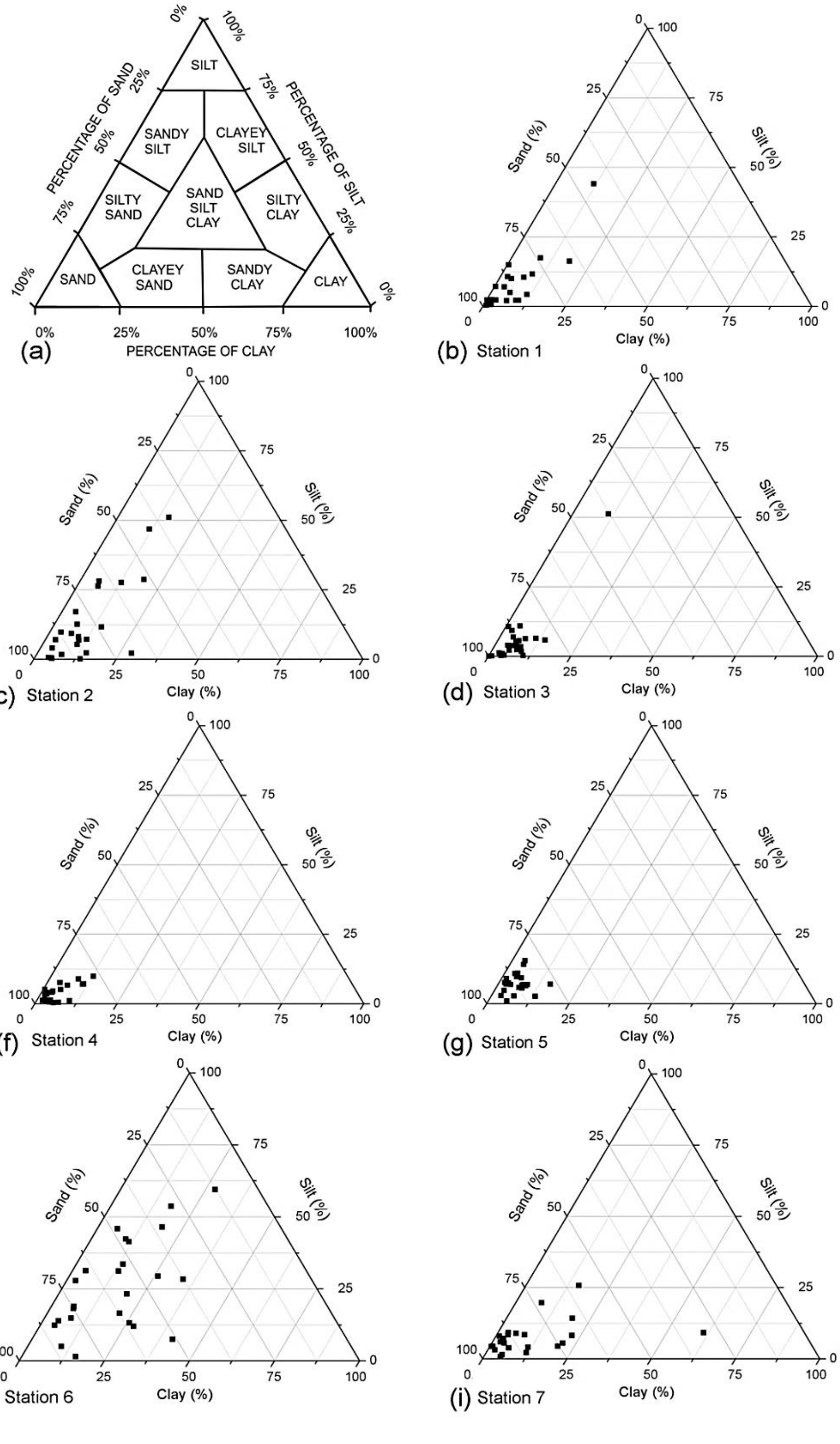


Figure 24 a-i. (a) Schematic plot showing textural classes according to Sheppard's classification (b-i) Ternary plot for the seven stations in KAE during the study period (2009-2011)

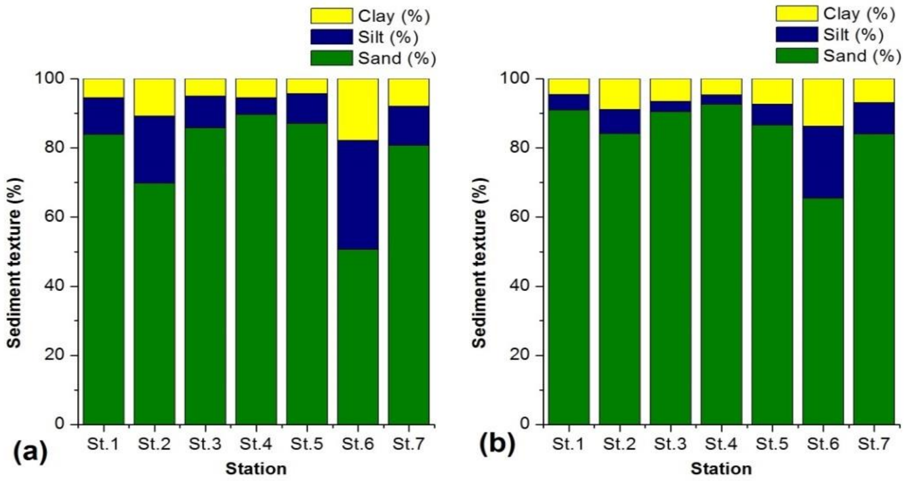


Figure 25 a-b. Station-wise mean percentage composition of sediment texture in KAE during (a) 2009-2010 and (b) 2010-2011 period

In spatial scale, station six (15.75 ± 10.47 %) and station two (9.87 ± 6.05 %) depicted comparatively higher percentage of clay whereas, station four (5.1 ± 8.35 %), station three (5.83 ± 3.57 %) were recorded low fractions in sediment [Figure 24 a-i, 25 a-b and Table 2 a-b]. The significant differences observed in the sand (ANOVA F (6,161) = 18.10, p =0.000), silt (ANOVA F (6,161) = 11.32, p =0.000) and clay (ANOVA F (6,161) = 1.063, p =0.000) fractions of sediment between the stations. Silt fraction also showed seasonal variability (ANOVA F(5,162) = 3.30, p =0.007).

III. 3. 5. Organic carbon

The average sediment organic carbon (OC) content in the of KAE during the entire study was 9.30 ± 7.10 g·kg^{-1}, and it was 8.80 ± 7.40 g·kg^{-1} first year and that for the second year was 9.9 ± 6.9 g·kg^{-1} [Figure 26 a-b and Table 2 a-b]. Seasonally, monsoon season of the first year (11.9 ± 8.2 g·kg^{-1}), pre-monsoon (10.5 ± 7.3 g·kg^{-1}), and post-monsoon of the second year (10.3 ± 6.9) depicted the highest concentration. However, pre-monsoon (5.6 ± 5.0 g·kg^{-1}) and post-monsoon (8.7 ± 7.7 g·kg^{-1}) of the first year and monsoon of the second year (9.1 ± 6.6 g·kg^{-1}) depicted lowest values. In spatial scale, station 6 (21.1 ± 6.3 g·kg^{-1}), station 2 (12.5 ± 5.7 g·kg^{-1}) and station 7 (10.2 ± 4.7 g·kg^{-1}) were noticed with higher organic carbon content as compared to station four (3.8 ± 2.2 g·kg^{-1}), station one (5.0 ± 4.1 g·kg^{-1}), station three (6.3 ± 4.7 g·kg^{-1}) and station five (6.4

± 3.1 g·kg^{-1}). The significant difference of organic carbon in sediment was noticed between stations (ANOVA F (6,161) = 41.28, p = 0.000) and a weak variation between seasons (ANOVA F (5,162) = 2.59, p = 0.028).

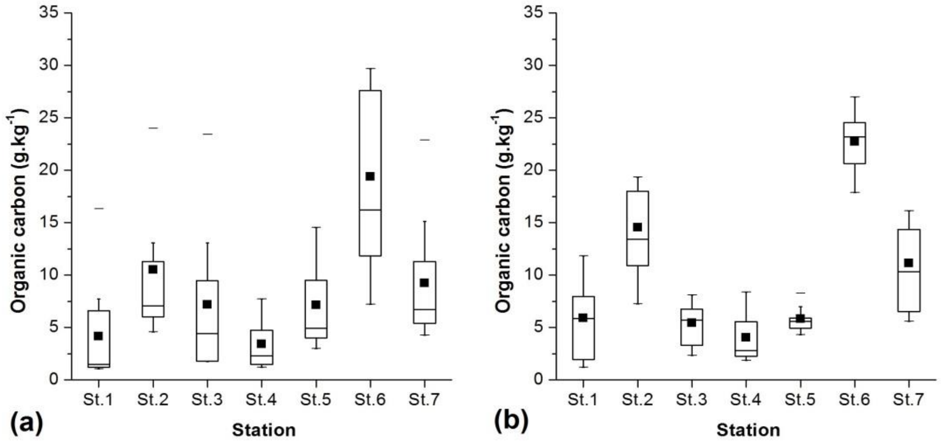


Figure 26 a-b. Box plot of sediment organic carbon in different stations of KAE during (a) 2009-2010 and (b) 2010-2011 period, (whisker: range, box: interquartile range, square: mean)

III. 3. 6. Organic matter

The mean sediment organic matter in the estuary was 16.10 ± 12.30 g·kg^{-1} while that was 15.2 ± 12.8 g·kg^{-1}during the first year and for the second year was 17.0 ± 11.8 g·kg^{-1}.

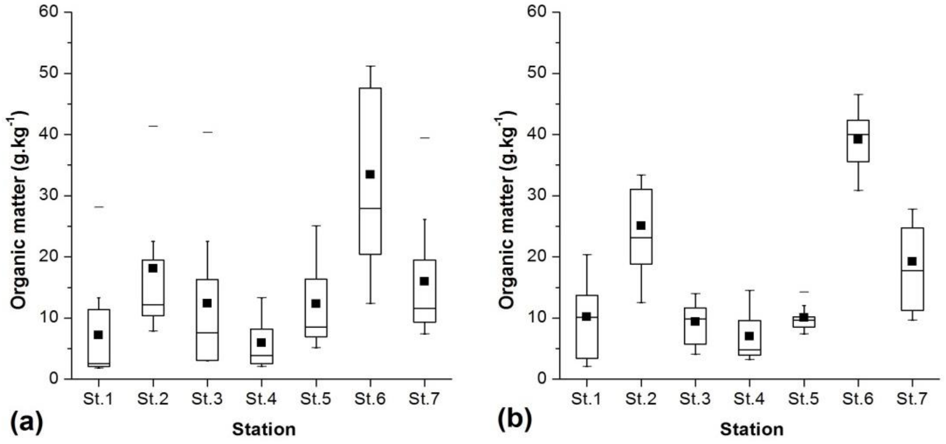


Figure 27 a-b. Box plot of sediment organic matter in different stations of KAE during (a) 2009-2010 and (b) 2010-2011 period, (whisker: range, box: interquartile range, square: mean)

Table 2 a. Seasonal mean of sediment parameters (± SD) and meteorological parameters (± SD) during 2009 to 2010 period (OC: organic carbon, OM: organic matter, RF: rainfall, RD NE: river discharge of Neeleswaram, RD AR: River discharge of Arangaly)

Parameters	Monsoon	Post-monsoon	Pre-monsoon
OC (%)	1.19 ± 0.83	0.87 ± 0.74	0.56 ± 0.5
OM (%)	2.04 ± 1.42	1.50 ± 1.27	0.97 ± 0.85
Temperature (°C)	27.31 ± 1.93	28.21 ± 1.45	30.14 ± 1.18
Eh (mV)	-71.25 ± 87.5	-135.71 ± 54.16	-107.64 ± 94.17
pH	8.31 ± 0.91	7.51 ± 0.48	7.76 ± 0.25
Sand (%)	85.62 ± 13.73	84.65 ± 12.11	84.55 ± 16.28
Silt (%)	6.71 ± 9.06	8.10 ± 10.63	7.76 ± 12.62
Clay (%)	7.67 ± 6.88	7.25 ± 3.87	7.69 ± 8.07
Depth (m)	3.47 ± 0.91	3.64 ± 0.83	3.4 ± 0.9
RF (mm)	651.57 ± 256.67	134.14 ± 97.43	97.52 ± 87.99
RD NE. ($m^3 S^{-1}$)	464.92 ± 119.81	140.61 ± 155.23	-
RD AR. ($m^3 S^{-1}$)	140.58 ± 60.59	46.24 ± 33.23	-

Table 2 b. Seasonal mean of sediment parameters (± SD) and meteorological parameters (± SD) during 2009 to 2011 period

Parameters	Monsoon	Post-monsoon	Pre-monsoon
OC (%)	0.91 ± 0.66	1.03 ± 0.69	1.05 ± 0.73
OM (%)	1.56 ± 1.14	1.78 ± 1.18	1.8 ± 1.27
Temperature (°C)	28.36 ± 1.25	27.46 ± 0.97	29.82 ± 0.93
Eh (mV)	-53.43 ± 54.67	-83.75 ± 85.93	-105.93 ± 64.11
pH	8.18 ± 0.41	7.57 ± 0.33	7.56 ± 0.51
Sand (%)	85.3 ± 13.10	75 ± 19.52	74.52 ± 20.07
Silt (%)	8.44 ± 9.74	16.36 ± 14.9	15.92 ± 15.56
Clay (%)	6.26 ± 6.14	8.64 ± 8.27	9.55 ± 6.48
Depth (m)	3.72 ± 0.91	3.95 ± 0.86	3.62 ± 0.95
RF (mm)	557.43 ± 197.46	243.84 ± 224.32	100.04 ± 63.01
RD NE. ($m^3 S^{-1}$)	382.73 ± 51.87	182.33 ± 186.02	-
RD AR. ($m^3 S^{-1}$)	89.79 ± 13.68	65.36 ± 43.53	-

Seasonally, monsoon season of the first year (20.4 ± 14.2), pre-monsoon (18.0 ± 12.7 g·kg^{-1}) and post-monsoon of the second year (17.8 ± 11.8 g·kg^{-1}) were exhibiting higher organic matter in sediment when compared to pre-monsoon (9.7 ± 8.5 g·kg^{-1}) and post-monsoon 15.0 ± 12.7 g·kg^{-1} of the first year and monsoon (15.6 ± 11.4 g·kg^{-1}) of the second year. Spatially, station six (36.3 ± 10.9 g·kg^{-1}), station two (21.6 ± 9.8 g·kg^{-1}) and station seven (17.6 ± 8.1) was recorded with high organic matter than station 4 (6.5 ± 3.8 g·kg^{-1}), station one (8.7 ± 7.0 g·kg^{-1}), station three (10.9 ± 8.0 g·kg^{-1}) and station five (11.2 ± 5.3 g·kg^{-1}) [Figure 27 a-b, 28 and Table 2 a-b]. Significant differences were observed in organic matter content of sediments between stations (ANOVA F (6,161) = 41.29, p = 0.000) and seasons (ANOVA F (5,162) = 2.60, p = 0.027).

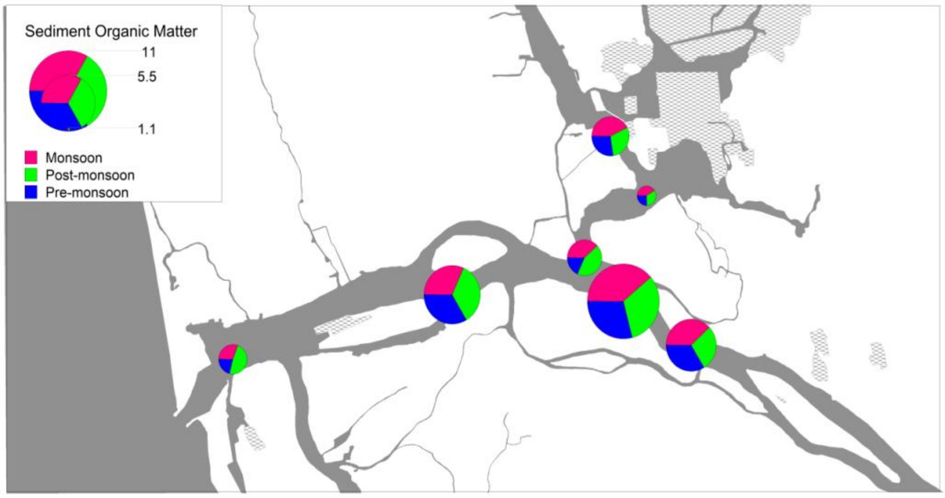


Figure 28. The mean percentage composition of sediment organic matter in KAE during the study (2009-2011) with its spatial distribution pattern (g·kg^{-1})

III. 4. Principal component analysis

The pattern of variation in environmental parameters in related to seasons and stations were made clear in the principal component analysis (PCA) ordination [Table 3]. The first five principal components accounted for 71.4 percentage of variability in environmental conditions over the seven stations [Figure 29 & 30]. Among this, the first two principal components accounted for 46.3 percentage of variability in environmental conditions, with 27.9 on axis 1 (eigenvalue value 4.86) and that for axis 2 was 18.4 percent (eigenvalue value 3.27).

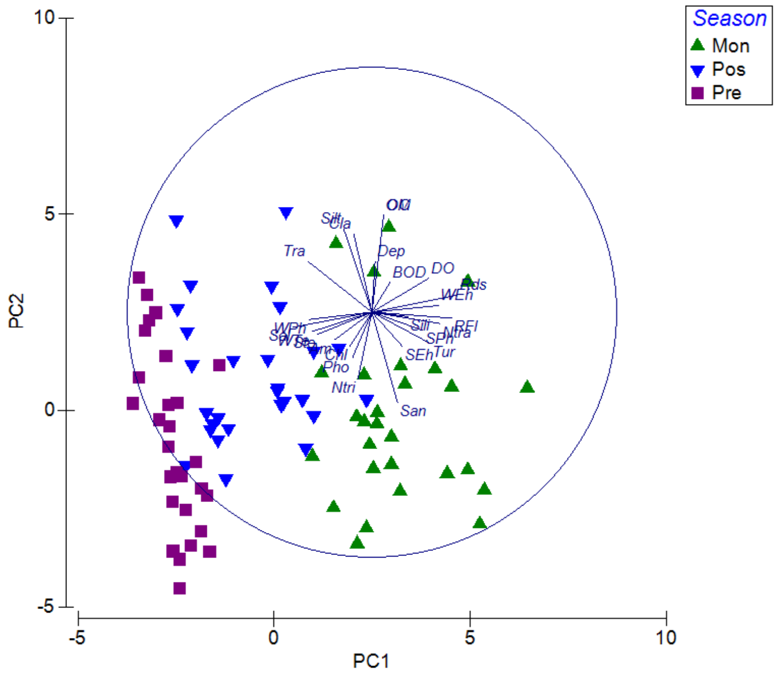


Figure 29. Two-dimensional principal component analysis (PCA) ordination of selected normalised environmental variables of stations chosen in KAE on a seasonal basis*

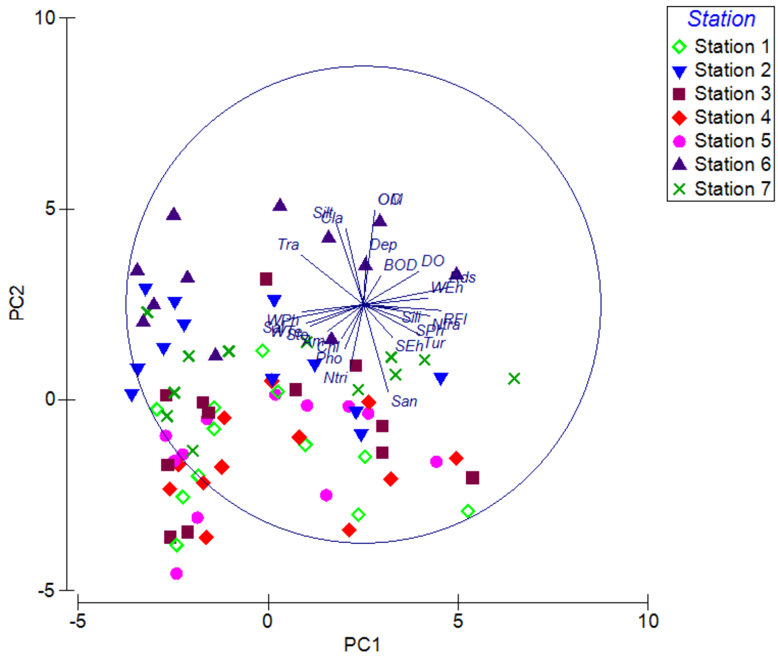


Figure 30. Two-dimensional principal component analysis (PCA) ordination of selected normalized environmental variables of stations chosen in KAE on a spatial basis*

*(Mon: Monsoon, Pos: post-monsoon, Pre: pre-monsoon, Sal: bottom water salinity, WTe: bottom water temperature, WEh: bottom water Eh, Tra: water transparency, Tur: bottom water turbidity, WPh: bottom water pH, DO: bottom water dissolved oxygen, BOD: bottom water Biological oxygen demand, Ntra: dissolved nitrate, Ntri: dissolved nitrite, Pho: dissolved phosphate, Sili: dissolved silicate, Am: dissolved ammonia, OC: sediment organic carbon, OM: sediment organic matter, Ste: sediment temperature, SEh: sediment redox potential (Eh), SPh: sediment pH, San: sand, Silt: silt, Cla: clay, RFl: rainfall, RDi: river discharge, Dep: depth)

Table 3. Two-dimensional principal component analysis (PCA) of environmental conditions at each sampling stations in KAE

PCA axis	1	2	3	4	5
Eigenvalues	6.98	4.59	3.19	1.59	1.51
%Variation	27.9	18.4	12.7	6.3	6
Cum. %Variation	27.9	46.3	59	65.4	71.4
Eigenvectors					
Rainfall	0.326	-0.024	-0.199	0.111	0.033
River discharge	0.350	0.069	0.010	0.107	0.006
Salinity	-0.323	-0.062	0.075	0.102	0.235
Nitrate	0.278	-0.047	-0.095	0.071	0.234
Water Eh	0.272	0.027	-0.009	0.145	-0.187
Transparency	-0.261	0.207	0.137	0.087	0.080
Water pH	-0.257	-0.030	0.180	0.063	0.203
Water temperature	-0.245	-0.079	-0.272	0.122	-0.312
Turbidity	0.237	-0.127	-0.263	-0.179	0.238
Dissolved oxygen	0.232	0.139	-0.010	-0.048	-0.209
Sediment temperature	-0.222	-0.090	-0.292	0.040	-0.388
Sediment pH	0.208	-0.074	-0.079	-0.463	-0.151
Silicate	0.147	-0.018	-0.14	0.477	-0.032
Sediment Eh	0.123	-0.138	0.034	-0.155	0.168
Sand	0.104	-0.369	0.222	0.134	-0.043
Ammonia	-0.154	-0.114	-0.173	-0.486	-0.155
Silt	-0.116	0.341	-0.194	-0.139	0.139
Chl-*a*	-0.092	-0.141	-0.36	0.197	0.038
Clay	-0.075	0.321	-0.194	0.068	-0.118
BOD	0.074	0.123	0.255	-0.233	-0.278
Nitrite	-0.057	-0.273	-0.291	-0.011	0.188
Organic Matter	0.048	0.396	-0.209	0.010	0.060
Depth	0.013	0.207	0.102	-0.136	0.394
Phosphate	-0.080	-0.188	-0.329	-0.149	0.265

Rainfall, river discharge, bottom water salinity, dissolved nitrate, waterh, transparency, water pH, water temperature, turbidity, dissolved oxygen, sediment temperature and sediment pH were most important determinants of differences between seasons along the first axis, whereas organic matter, sand, silt, clay, bottom water nitrite and depth were influential along axis 2.

III. 5. Discussion

The multivariate technique, principal component analysis (PCA) was used to emphasise the environmental variation and bring out significant patterns in the dataset from KAE. Hydrographic and sediment parameters were analysed simultaneously in the PCA and the dataset has mainly separated into two principal components accounted for 46.3 percent of the variability. The first two components have been mainly divided based on the spatial and temporal pattern in distribution. Seasonal variability of parameters was evident on the axis 1, such as rainfall-induced hydrographic changes like river discharge, bottom water salinity, nitrate, Eh, transparency, pH, temperature, turbidity, dissolved oxygen and linked sediment characteristics like sediment temperature and pH. However, sediment parameters were more varied on a spatial scale than that of hydrographic features such as organic matter, sand, silt, clay content along with related hydrographic parameter like bottom water dissolved nitrite, and it was distributed along the axis 2 (PCA 2).

In general, hydrography in the estuary is controlled by the freshwater influx, tides, meteorological forcing, and density currents (McLusky and Elliott, 2004; Talke and Stacey, 2008). However, rainfall received during the southwest and northeast monsoon is regarded as one of the significant factors that control overall ecology of Kerala coast. The average annual rainfall of Kerala is about 300 cm, and of which, 75 percent occurs during the southwest monsoon (Ananthakrishnan *et al.*, 1979). Similar trend was experience in the river basins of Kodungallur-Azikode estuary (KAE) with a peak rainfall during the south-west monsoon season. Since the rate of precipitation is a primary factor regulating the riverine discharge pattern of KAE, that controls the overall hydrography of this estuarine system. Kodungallur-Azikode estuary located north to Cochin (Kochi) estuary, through which 70 percent of the Periyar River discharges into the Arabian Sea and the rest through the Cochin (Kochi) estuary (Shivaprasad *et al.*, 2012). The Kodungallur-Azikode estuary was shallow, depth at mouth region was 4.43 ± 0.75 m, and it decreased towards the Riverhead. Therefore, they are convergent, i.e., the width decreases rapidly from mouth to head, and such monsoon-driven estuaries can be referred to as "monsoonal estuaries" (Vijith *et al.*, 2009). As runoff decreases, the salinity in the estuary migrates upstream.

The temperature of water column have control on the rate of fundamental biochemical processes in organisms and consequently, that influence organismal, population, and community level processes in estuaries (Jayachandran *et al.*, 2015; Levinton, 2014; O'Connor *et al.*, 2007). Global warming and increasing temperature may also influence interactions between the physical and biological process in estuaries (Gabler *et al.*, 2017; Uncles and Stephens, 2001). The bottom water temperature in the Kodungallur-Azkiode estuary (KAE) depicted an increasing trend from estuarine mouth to middle zone of estuary, and it could be due to the intrusion of comparatively cooler water mass through the estuarine mouth by the tidal cycle. Towards the northern arm of the estuary, the values were found to be increased. However, Riverhead of southern wing showed slightly low temperature due to the mixing of cold water from the upper riverine zone. The penetration of freshwater into the estuarine system is not the single factor affecting the water temperature in the estuary (Sankaranarayanan and Qasim, 1969), but the introduction of cold water from the sea (Ramamirthan and Jayaraman, 1963), atmospheric interaction with surface water and other localised factors may also be a significant factor (Uncles and Stephens, 2001). In the present study, the seasonal mean of bottom water temperature was higher during pre-monsoon when compared to monsoon and post-monsoon periods. A similar trend was also observed in other estuaries of the south-west coast of India (Bijoy Nandan and Abdul Azis, 1995b; Joseph, 1988; Qasim, 2003).

Turbidity levels in water column have control over phytoplankton biomass and overall productivity in estuarine systems (Cloern, 1987). In addition to that, turbidity influences the development of community pattern and feeding guild in the estuarine gradient (Sallenave and Barton, 1990). Turbidity level in the estuary was also enormously increased during monsoon season due to the high influx of silt content, agricultural runoff, sewage, and other allochthonous organic matters from upstream rivers. The observed low transparency values in the monsoon season, especially during substantial freshwater influx from the rivers attributed to the increase of turbidity due to increased supply of inorganic matter, soluble coloured organic compounds, and organic matter and low intensity of solar radiation (Jayachandran *et al.*, 2012; Qasim *et al.*, 1968; Sarala Devi and Venugopal, 1989; Saraladevi *et al.*, 1983). Qasim *et al.* (1968) also observed a reduction in the light penetration during rainfall ($r = 0.586$, $p < 0.01$) and subsequent river discharge ($r = 0.517$, $p < 0.01$). Transparency (Secchi depth) values were relatively low at stations towards the riverine side, and that mainly attributed to high turbidity level due to mixing up and drags during massive riverine influx. The lower water column transparency reported to be affecting the photosynthetic activity in phytoplankton and further productivity of water column (Cloern *et al.*, 2014; Nair *et al.*, 1975; Qasim, 1973).

Salinity is one of the critical factors affects functional and structural responses of marine organisms to changes in total osmotic concentration, the relative proportion of solutes, coefficient of absorption and saturation of dissolved gases (Sakamoto *et al.*, 2015; Vineetha *et al.*, 2015). The salinity distribution in coastal and estuarine waters was mainly controlled by freshwater influx through rivers, rainfall, evaporation rate, tidal variation, and coastal circulation (Revichandran and Abraham, 1998; Vijith *et al.*, 2009). In the present study, extreme drop in salinity was observed during monsoon ($r = -0.636$, $p < 0.01$) due to the dilution by a significant amount of freshwater influx ($r = -0.635$, $p < 0.01$), while the comparatively higher values in bottom salinity could be due to the outflowing riverine waters giving a two-layered structure (Revichandran, 1993). Similarly, a distinct spatial gradient of salinity that was declining from estuarine mouth to the riverine zone was observed during the present investigation, and it was similar to observations of Vijith *et al.* (2009) in other monsoonal estuaries of India. Therefore, the estuaries are critical examples where conditions at the mouth are entirely different from the inner part of the estuary, that varying from polyhaline and mesohaline sections in the estuarine mouth region to freshwater dominant oligohaline conditions in the river head (Little *et al.*, 2016; McLusky and Elliott, 2004). Due to this strong salinity gradient, physicochemical conditions in the estuaries show various properties through its extension (Bald *et al.*, 2005). This spatiotemporal variability in salinity distribution also affects the functional and structural responses of organisms to variations in total osmotic concentration, the relative proportion of solutes, coefficient of absorption and saturation of dissolved gases (Sakamoto *et al.*, 2015). The increase in salinity influenced the water column pH (r =0.566, $p < 0.01$) and depicted a relatively higher Chl-*a* concentration (r =0.248, $p < 0.01$) in the estuary.

The bottom water pH of KAE remained slightly alkaline in almost all months. Whereas, during the peak monsoon which marked by heavy rain, values that have tended to fall in all the stations due to the acidic state by higher freshwater discharge. The river discharge carries large quantities of humic material in colloidal suspension, which is frequently slightly acidic, but when it meets seawater, the colloidal particles are coagulated and the pH shifts towards the alkaline side (Ried, 1961). Photosynthetic removal of CO_2 through bicarbonate degradation process, dilution of seawater by freshwater mixing, variation in salinity with temperature and respiration, mineralisation of organic matter, industrial effluent discharge, and nature of dissolved materials were also observed as an important factor influencing the pH of water column (Paramasivam and Kannan, 2005). Thus on a spatial scale, a clear gradation of pH was observed with salinity gradient, an alkaline condition in the estuarine mouth to relatively low pH values in the Riverhead. However, human-induced variations in pH due to chemical and other industrial discharges render a waterbody unsuitable not only for recreational purposes but also for the healthy life of aquatic organisms in the estuary (Jayachandran *et al.*, 2012; Webb, 1982).

Under such extreme conditions, the survival of biota becomes a severe problem (Bijoy Nandan and Abdul Azis, 1995c), because tolerance level of most of the organisms to pH is quite narrow and critical (George, 1979). Most of these animals are adapted to an optimum pH and cannot withstand abrupt changes (Asha *et al.*, 2016; Jayachandran *et al.*, 2012). In general, aquatic organisms will be under stress when exposed to pH levels above nine and below five. The fluctuating pH also influences life processes such as metabolism, growth, and distribution of individuals. The industrial effluents discharged into the aquatic system may significantly lower or elevate the pH of water depending on the nature of the effluents. In extreme conditions, the survival of the organisms becomes a severe problem in estuaries (Bijoy Nandan and Abdul Azis, 1995a).

Oxidation Reduction Potential (Eh) measurement of the water column will also give a good idea of the state of water column (Bijoy Nandan and Abdul Azis, 1995c; ZoBell, 1946). That also is regarded as an integrated parameter, which is triggered by the activity of living microbial communities. The bottom water oxidation-reduction potential (Eh) of KAE depicted a comparatively reduced condition in the estuarine mouth region as compared to Riverhead. In the present observation, values tend to moderately reduced circumstances in the pre-monsoon and post-monsoon periods; however, monsoon period exhibited a comparatively oxidised water column. According to Reddy *et al.* (2000), microbial aerobic activities reflect oxidising conditions above an Eh of 300 mV; facultative reducing microbes are active from Eh 300 to -50 mV, or moderately reducing conditions. Remarkably decreasing situations does not support aquatic life especially sessile or sedentary forms such as benthic fauna. Thus, measured Eh values can be regarded as an integrated parameter, which triggered by the activity of living microbial communities. In spatial scale, Eh values in the KAE was showed a moderately reduced water column towards the estuarine mouth and Kottapuram region in the northern arm of the estuary, where active cage farming of finfishes are prominent. Periyar River was with colossal river flux depicting a comparatively oxidised water column. Thus, changes in external conditions, such as precipitation and river discharge, temperature, and availability of organic matter can affect the redox potential. Such conditions have also been observed in other Indian estuaries (Bijoy Nandan and Abdul Azis, 1995a&c).

Dissolved oxygen (DO) is an essential factor for aquatic organisms, for to keep their cells alive and to meet their respiratory needs (Boyd, 2015; Kusum *et al.*, 2011). In general, DO concentrations in estuaries are varied from near zero to over 6 mg L^{-1}. In the present study, comparatively high bottom water DO concentration was observed in the KAE during monsoon and post-monsoon; mixing by the turbulence of river discharge, and rainfall has great influence on the distribution of DO concentration. It is evident that oxygen enters to estuarine waters mainly through two natural processes such as diffusion from the atmosphere and photosynthesis by primary producers like phytoplankton and other macroalgae. The mixing of surface waters by wind and waves enhances the rate of oxygen dissolution or absorption by water column (Boyd, 2015). The river discharge accompanied by tidal effects could be the reason for the comparatively high DO values noticed towards Riverhead in the southern arm (r = -0.432, p <0.01). The relatively good DO levels noted in KAE were well comparable with the other estuaries in India. The DO concentration in the different estuaries of India as follows, 4.5 ml L^{-1} in the Vasishta Godavari estuary (Srinivasa Rao *et al.*, 2009), 5.26 mg L^{-1} in Gautami-Godavari estuary (Tripathy *et al.*, 2005), 4.78 mg L^{-1} in Dhamara estuary (Mahapatro *et al.*, 2011), about 6 mg L^{-1} in Chilka lake (Nayak *et al.*, 2004), 4.64 mg L^{-1} in Rushikulya estuary (Paikaray *et al.*, 2012), and 4.4 mg L^{-1} in Mandovi and Zuari estuary (Harkantra and Rodrigues, 2004). The prolonged exposure of DO level just below 2.8 mg L^{-1}, such as 'hypoxia' in coastal waters found to physiologically stress to most marine organisms and expected to affect the overall health of that water body (Bijoy Nandan and Abdul Azis, 1995a; Breitburg *et al.*, 2009). Benthos are mostly sedentary or sessile forms; therefore they could be more vulnerable to such hypoxia-related stress (Bijoy Nandan and Abdul Azis, 1995a; Diaz and Rosenberg, 1995). Johannesse and Dahl (1996) have reported that increased nutrient load reduces the dissolved oxygen content in the water column. The decomposition of organic waste and oxidation of inorganic waste reduces the dissolved oxygen level to extremely low, especially in the subsurface water column of the estuary (Bijoy Nandan and Abdul Azis, 1995b; Nair *et al.*, 1988; Ordoñez *et al.*, 2015). Thus, the partial utilisation of DO by organic-rich sediments may also influence the bottom water DO concentration. The spatial and temporal variation of DO concentration could also attribute to the seasonal and tidal fluctuations of water column (Vijayan *et al.*, 1976).

Oxygen consumption in the water column measured as biochemical oxygen demand (BOD) (Eaton *et al.*, 2005; MacPherson *et al.*, 2007) and considered as an important indicator in pollution assessment (Brill *et al.*, 1984; USEPA, 2002). The variation is dependent on the amount of suspended, dissolved organic matter in the water column. In the present study, BOD values for KAE was comparatively higher during monsoon, and post-monsoon period, that depicted a weak correlation with river discharge. In spatial scale, values tend to link with nutrient loading in the estuary. However, comparatively low biological oxygen demand was exhibited by KAE when compared to values recorded from other estuaries of the south-west coast of India (Remani *et al.*, 1981). This comparatively low BOD value of less than 5 mg L^{-1} in the estuary indicates periodic removal rate of organic matter from the floor of the estuary (Bhargava 1977). Similarly, noted comparatively low values in the estuary could be due to the intense semidiurnal tidal flushing and dilution by freshwater discharge from the Periyar River. However a water body with BOD_5 of 2 to 8 mg L^{-1} is considered to be moderately polluted and above that is regarded as contaminated condition (Bijoy Nandan and Abdul Azis, 1995a; Martin, 1970; Sarala Devi, 1986).

In estuarine systems, the sources of nutrients have varied from rivers, atmosphere, sediments, groundwater, adjacent wetlands and rainwater (Domingues *et al.*, 2005; Jayachandran *et al.*, 2012; Menon *et al.*, 2000; Sankaranarayanan and Qasim, 1969; Sarma *et al.*, 2010; Vollenweider *et al.*, 1998). Similarly, in the KAE, the primary source of nutrients input was associated with river discharge during the southwest monsoon and runoff from the adjacent coastal zone. The concentration and composition of various macro and micronutrient in the water column have considerable importance since they can modify the phytoplankton community structure (Sarma *et al.*, 2009). The rate of primary production, area, depth, the volume of a water body, influx rate, tidal exchange, water residence time, vertical mixing and stratification could also affect the nutrient transport and cycle in the estuary. Hence, average nutrient concentrations of the KAE was well below those of the San Francisco Bay (Cloern, 1996), Chesapeake Bay (Ward and Twilley, 1986), Pearl River estuary (Yin *et al.*, 2001) and the Guadiana estuary (Domingues *et al.*, 2005; Jayachandran *et al.*, 2012).

Nitrogen is the major macronutrient that controls phytoplankton production in tropical waters (Myers and Iverson, 1981), especially in Indian monsoonal estuaries like Mandovi-Zuari (Ram *et al.*, 2003), Cochin estuary (Gupta *et al.*, 2009), Godavari estuary (Sarma *et al.*, 2010), Hooghly estuary (Mukhopadhyay *et al.*, 2006) and Kodungallur-Azhikode estuary (Jayachandran and Bijoy Nandan, 2012). Dissolved nitrogen concentration in the estuary represented in different forms such as ammonia, nitrite and nitrate. Ammonia is the first inorganic product produced during regeneration of nitrogen from organic compounds (Domingues *et al.*, 2005). Therefore, during the degradation of nitrogenous compounds, dissolved ammonia concentration increases with increasing pH of estuarine sediment, and it was shown to be the common favoured form of nitrogen for planktonic absorption, and it inhibits the utilisation of other forms of nitrite and nitrate in its presence (Domingues *et al.*, 2005; Jayachandran *et al.*, 2012). Thus, phytoplankton production appears to influence the ammonia concentrations (Cloern *et al.*, 2014; Lallu *et al.*, 2014). However, the dissolved ammonia concentration in the estuary was comparatively lower than that reported from other estuaries of India by Aravindakshan *et al.* (1992), Nair *et al.* (1988) and Venugopal *et al.* (1980). Comparatively high concentrations in some zone of water column could be partly due to death and subsequent decomposition of phytoplankton and partly due to terrigenous input during monsoon flushing (Segar and Hariharan, 1989). Besides, excretion by planktonic organisms induces higher levels of ammonia.

Nitrite-nitrogen is unstable in the presence of oxygen and hence occurs mainly as an intermediate between ammonia and nitrate. An increase in nitrate concentration in the northern arm and estuarine mouth of KAE is attributed to the quantum of effluent released from the harbour and agriculture-related activities and further bacterial decomposition of detritus in the area. The values of nitrite concentration are related to concentration of ammonia and effluent discharge coupled with phytoplankton abundance during the preceding month (Rajendran and Venugopalan, 1977a; Rajendran and Venugopalan, 1977b). However, the increase of nitrite in the bottom water could be due to increased bacterial activity, which is expected in a silty-clay substratum compared to the sandy substrate. Seasonally, concentration in the KAE was higher during the pre-monsoon period, and this was attributed to the variation in the phytoplankton excretion, oxidation of ammonia and reduction of nitrate (Rajendran and Venugopalan, 1975). Among the three inorganic forms of nitrogen, dissolved nitrate-nitrogen was the most abundant at all the stations of KAE. Because nitrate is the most stable oxidation level of nitrogen in the presence of oxygen in seawater (Jayachandran *et al.*, 2012; Rajendran and Venugopalan, 1977b) and could accumulate if left unutilized. Seasonally, the concentration of nitrate in the KAE was abundant during the monsoon period. The dissolved nitrate concentration in the estuary mainly depends on the allochthonous source, external additions of some effluents loaded with nitrogenous compounds into the estuary, by the agricultural runoff and sewage. High values of nitrate at all stations in the present study preceded by allochthonous source nitrogenous compounds. The oxidation process of ammonia to nitrite and then to nitrate may take place photo-chemically or chemically in this surface layer or biologically in and near the bottom (Cooper, 1937). However, high values of nitrate might have been due to bacterial oxidation rather than the photochemical oxidation of the high level of ammonia.

Phosphorus considered as the vital macronutrient regulating the growth and production of phytoplankton and its concentration helps to predict the total biomass of phytoplankton in the estuary, and moreover, it stimulates secondary production. The study revealed that dissolved inorganic phosphate in the KAE was comparatively lower than other Indian estuaries (Jayachandran *et al.*, 2012; Sarma *et al.*, 2009). The river discharges are the significant sources of phosphorus input to estuaries (Balchand and Nair, 1994). The riverine influx of phosphorus in estuaries may be considerably altered by precipitation or dissolution causing changes in the concentration of phosphorus. The weathering of insoluble calcium and ferric phosphate rock and land drainage especially from agricultural runoff also delivers phosphorus to estuaries. Phytoplankton takes up phosphates, nitrates in relatively constant proportion, and releases these elements during their decomposition. Abundant phosphate availability in water stimulates undesirable plankton bloom. Phosphate concentration in the estuary was comparatively lower during the monsoon period, and such phosphorous limited conditions have found in several estuaries (Glé *et al.*, 2008). The low values of phosphate during monsoon period explained by the combined effect of dilution of estuarine water by fresh riverine water containing low phosphate and removal by adsorption caused by the influx of silt-laden fresh water. The low salinity during monsoon by the increase in river discharge favours the removal of phosphorus from the overlying water by the sedimentary particle. The concentration of phosphate in KAE was influenced by effluent discharges and agricultural runoff, and it is typical in estuaries waters (Nair *et al.*, 1988; Sankaranarayanan and Qasim, 1969). High concentration of phosphate leads to the abundance of phytoplankton and further, the subsequent decrease in the level due to its uptake. During pre-monsoon, high values of dissolved phosphate in the water column can be attributed to the leaching of phosphate from sediments to the water column. In the present study, dissolved phosphate concentration positively correlated with the Chl-a content.

Dissolved silicate-silicon is a key macronutrient form major composition for formation of dominant phytoplankton community such as diatoms; its frustules are made up of particulate amorphous silica or biogenic silica (Martin-Jézéquel *et al.*, 2000). In the present study, the dissolved bottom water silicate concentration in KAE showed wide fluctuations on a temporal scale. It was high during peak river discharge period south-west monsoon, and such observations were noticed in other Indian estuaries (Anirudhan and Nambisan, 1990; Asha, 2017; Mukhopadhyay *et al.*, 2006; Sankaranarayanan and Qasim, 1969). The mixing of fresh water with seawater (Anirudhan and Nambisan, 1990) and the intermittent summer showers affect the distribution pattern of silicate in the estuary. The increased silicate concentration in the mixing zone of estuary mainly attributed to continuous resuspension and riverine influence on silicate distribution in the estuary. Nitrogen/phosphorus ratio (N:P) in KAE was well above Redfield ratio during the southwest monsoon, which indicates that the estuary was under phosphorus limited during the monsoon period. In the same way, nitrogen limited during post-monsoon and pre-monsoon periods (Jayachandran *et al.*, 2012; Ptacnik *et al.*, 2010). Nitrogen-limited conditions were also observed in many of the Indian estuaries such as Cochin backwaters (Gupta *et al.*, 2009), Godavari estuary (Sarma *et al.*, 2010), Mandovi-Zuari estuarine ecosystem (Ram *et al.*, 2003), Hooghly estuary (Mukhopadhyay *et al.*, 2006) and Ashtamudi estuary (Nair and Abdul Azis, 1987b).

The measurement of photosynthetic pigments, particularly chlorophyll-a (Chl-a) is used as an index of phytoplankton productivity and biomass (Falkowski and Kiefer, 1985). The standing crop of phytoplankton indicates the availability of food for aquatic organisms including their larval forms (Jyothibabu *et al.*, 2006; Madhu *et al.*, 2007). In the present study, stations in the northern limb registered higher Chl-a compared to other areas due to comparatively higher residence time and less water column disturbances. Indiscriminate disposal of sewage and industrial wastes have been a significant cause for the nutrient enrichment in the estuary. Such situations resulted in declining phytoplankton diversity and increased biomass by promoting some opportunistic algal species that dominate and suppress others (Anu *et al.*, 2014; Dederen, 1992; Ramaiah and Ramarah, 1998). Such eutrophication process in the estuarine environment leads to degradation of water quality, water column hypoxia/anoxia, and harmful algal bloom events. That eventually results in loss of habitat and species diversity in the estuarine environment (Asha *et al.*, 2016;

Jayachandran and Bijoy Nandan, 2012; Nair *et al.*, 1984a; Qasim, 1973; Sreedevi, 2017; Thasneem, 2016; Vollenweider, 1992). Seasonal averages of Chl-a in the estuary showed high peak concentration during pre-monsoon period but that it varies depending upon rainfall and river discharge in the area. However, the nutrient requirement is known to differ with the phytoplankton, and that high concentrations of nutrients alone may not be conducive for a substantial increase in productivity (Qasim, 1973). Since the rate of regeneration of nitrogen is slower than that of phosphorus, the readily available ammoniacal form of nitrogen could have been responsible for high phytoplankton production. It is natural for Chl-*a* to fluctuate over time, it showed an increase followed by an intense rainfall; it could be due to flushing of nutrients by river discharge. High Chl-*a* levels were also common in summer months due to stable water column with favourable water temperature and light conditions. The tidal cycles in the estuaries have also formed as a crucial factor in controlling the algal biomass (Monbet, 1992). Hence, intense tidal mixing lowers Chl-*a* concentrations because the residence time of algae in the photic zone become short (Monbet, 1992). Tidal mixing also causes fine sediment to suspend, and the elevated turbidity levels that reduce the amount of light available for photosynthesis.

It is evident from this present study, the river discharge followed by an episode of high rainfall (precipitation) rates in the KAE during SW monsoon period was an important controlling factor for seasonal dynamics of hydrographic conditions. Such as water column turbidity and bottom water dissolved oxygen, BOD, nitrate, silicate, water Eh, salinity, water temperature, pH, dissolved ammonia, and transparency. The trophic index (TRIX) analysis (Vollenweider *et al.*, 1998) in the estuary depicted that, KAE experiencing higher productivity by the influence of the high degree of eutrophication (Jayachandran and Bijoy Nandan, 2012). An annual mean of 6.91 TRIX value was noticed in the KAE, and seasonal highest was observed during pre-monsoon period (7.15) and lowest during the post-monsoon period (6.51). Results indicate that eutrophication was predominant in the area where comparatively higher water residence time was observed. In the spatial scale, the northern arm of estuary showed higher trophic value and relatively low values in the southern division where freshwater discharge is higher. Intensive sand mining, poor agricultural practices and failures in sewage and other effluent discharge resulted in a significant amount of allochthonous and autochthonous nutrients and sediment transfer to KAE. The eutrophication process in the estuary could accelerate it. Perhaps, its appropriately scaled and parameterised regulations are the only realistic options for controlling eutrophication in the estuary.

In the case of estuarine benthos, nature of the sediments determines the lifestyle challenges that are widely considered as the principal factor controlling composition and abundance (Dutertre *et al.*, 2012; Sheeba, 2000). The condition of sediment is the indicator of the quality of overlying water mass, and hence their study is useful in the assessment of environmental pollution (Bijoy Nandan and Abdul Azis, 1995c; Burton, 2002; Dauer *et al.*, 2000). The sediments in the estuaries also indicate the equilibrium between the erosional and depositional strength of the ecosystem (Baker, 1978). The supply and source of these materials and the sites of deposition mainly depend on the type of estuaries, river discharge, currents, tidal regime, and wave action regime (Nichols, 1986). In an estuarine system, the sediment acts as the storage reservoir of nutrient materials in waters. The replenishment of these nutrients in time of need and their consequent removal dramatically helps in the biological cycle of the system. Such an exchange of nutrients depends upon the characteristics of the sediments and the hydrographic features of the estuary (Pomero *et al.*, 1965). The regeneration and mineralisation processes at the sediment-water interface significantly enhance the primary production by releasing nutrients (Martin, 1970). Among the sediment parameters, the temperature is one of the critical factors for spatial and temporal distribution pattern of species. It acts as a determinant parameter that controls the reproductive cycle and duration of the planktonic larval phase and also regulates the benthic ecosystem dynamics in temperate areas (Jayachandran *et al.*, 2015; Kinne, 1978). It will also influence the organic carbon mineralisation and CO_2 production in sediment (Malinverno and Martinez, 2015). In the present study, sediment temperature in the KAE varied on a spatiotemporal scale. Seasonally, pre-monsoon showed comparatively high temperature due to heat transfer from overly waters and biochemical process in the sediment. Spatial variability depended on biochemical process in sediment and temperature of overlying water mass, and such a condition was also observed in estuaries like Ashtamudi (Nair and Abdul Azis, 1987a). Higher temperature accelerates the decomposition of organic materials, which leads to oxygen depletion and release of total organic carbon (Malinverno and Martinez, 2015).

The sediment pH plays a crucial role in recycling of nutrients, and it is also significant in other chemical processes in sediments (Hou *et al.*, 2013). Hence, accumulation of organic matter and deposition may reduce the soil pH. In the present study, lowest pH levels noticed in sediment with a high organic matter. Das and Mangwani (2015), states that the reduced sediment pH could be due to the liberation of biogenic carbon dioxide by bacterial breakdown of organic matter. Sediment in the estuarine mouth area demonstrated comparatively higher pH by the influence of seawater. The sediment pH can move to alkaline nature owing to high sodium ion in overlying water mass. According to Miao *et al.* (2006), the higher value of pH can also be due to the redox changes in the sediment and water column apart from the influence of fresh water. The redox potential (Eh) is a quantitative measure of reducing power, which provides an idea of the degree of anoxic condition (Fiedler *et al.*, 2007). Thus, changes in external conditions, such as precipitation and river discharge, temperature, and availability of organic matter, can all lead to changes in Eh values (Bijoy Nandan and Abdul Azis, 1995c). Anoxic sediments have redox potential (Eh) on a negative side, while typical oxygenated sediment has positive values. The decomposition process of organic materials reduces the Eh (ZoBell, 1946). It depends on several factors, including diffusion from the surface of the sediment and infaunal activities. Therefore it is used as a surrogate measure of benthic conditions (Weissberger *et al.*, 2009). A reduced state of sediment was a common occurrence in KAE represented by negative values. In the recent study, redox values of -11 to -645 mV have been observed in Cochin estuary (Geetha *et al.*, 2010) and -34 to -400 mV in coconut husk retting areas of Kerala (Bijoy Nandan and Unnithan, 2007). According to Reddy *et al.* (2000), microbial aerobic activities can reflect in oxidising conditions above an Eh of 300 mV; facultative reducing microbes are active from Eh 300 to -50 mV, or moderately reducing conditions. Eh values around -200 mV, indicate high sulphide activity and presence of hydrogen sulphide. In such conditions, 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 (Miao *et al.*, 2006; ZoBell, 1946).

The floor of Kodungallur-Azhikode estuary (KAE) exhibits the different textural type of sediment, with a mixture of sand, silt, and clay and various combinations. Such differing combination of deposits was associated with the variation in tidal currents pattern and river flux (Murty *et al.*, 1976). The transport of bed load material becomes more common during the monsoon flushing time, and that mainly determines the sediment compositions in the estuary. In the present study, finer sediment types have been observed primarily in low energy zones in the KAE, and such low energy conditions significantly influence the sediment texture in the coastal and estuarine zones (Satyanaranyana Murty and Rao, 1959). Grain size co-varies with the sedimentary organic matter content, pore water chemistry and microbial abundance and composition, all of which are influenced by the near-bed flow regime. These variables could directly or indirectly affect distribution pattern of benthic fauna. Snelgrove and Butman (1994) stated that organic content of bottom sediments might be a more likely crucial factor than sediment grain size in determining the infaunal distribution. It is mainly because the organic matter in sediments is a dominant source of food for deposit feeders and indirectly for suspension feeders (Sanders, 1958; Snelgrove and Butman, 1994). The distributional pattern of organic carbon was closely related to sediment texture (Chaplot and Poesen, 2012) and concentration increases with decreasing particle size of the sediment (Bordovskiy, 1965). Thus, the spatial variation of organic carbon is in the KAE entirely agrees with this widely accepted observation. In the present investigation, comparatively higher organic carbon content in sediment was observed during monsoon seasons that could be attributed to the influx of land runoff containing the considerable amount of terrigenous matter. It is in general behaviour of tropical monsoonal estuaries like Mandovi estuary Alagarsamy (1991), and the present observations are also comparable to Mandovi and Zuari estuaries (Botto and Iribarne, 2000; Flemming, 2000). While was lower than that reported by Chanda *et al.* (1996) in the Mandovi estuary, coconut retting areas of Cochin (Remani *et al.*, 1981) and Ashtamudi backwaters (Nair *et al.*, 1984a). Estuarine sediments also display a highly structured small-scale organisation (Watling, 1989). The morphological standpoint of deposit, most fine sediments show a matrix forming a bridge between individual grains (Frankel and Meade, 1973), but completely encompassing all grains in muddy sediments (Watling, 1989).

Microorganisms are one of the primary agents producing the organic matrix that binds sediment (Frankel and Meade, 1973), although meiofauna and macroinfauna contribute. The nematodes (Riemann and Schrage, 1983) and turbellarians (Klause, 1986) secrete mucus by feeding, and other organisms produce linings of organic matter in burrow walls (Aller, 1983), or trails of slime. Burrowing activity may also change sediment characteristics (Jones and Jago, 1993). In both cases, the action of burrow construction should likely have a substantial effect on the structure of the sediment as the cohesive nature of the matric will be disrupted (Jumars and Nowell, 1984) and interstitial water will mix (Aller and Dodge, 1974). Active burrower, therefore, particularly at high densities can increase erosion rates and sediment mobility (Posey, 1987). Sediment transport depends on grain size, hydrodynamics, chemical and biological influences (Williamson and Ockenden, 1996). Movement of the substrate during wind, river current, and tidal forcing are believed to be an essential mechanism controlling community structure and function of shallow, soft-sediment benthos (Emerson, 1991). Sediment transport and its depositional pattern also affect by mixing of fresh and saline waters in the estuaries, and fine sediment particles get flocculated due to saline water and vary the sediment transport pattern (Baker, 1978). Therefore, the capacities of sediment loading within estuaries are related to the sedimentation rate and the energy available for transport. The bedload transport, the movement of particles in continuous or near-continuous contact with the bed, may affect the benthos by controlling their food availability (Luckenbach *et al.*, 1988), increasing mortality by abrasion (Miller, 1989) or predation (Grant, 1981), or rising dispersion (Palmer, 1988).

Organic carbon content in the sediments of the estuarine and riverine systems is of significant interest as possible food for the benthic fauna (Herman *et al.*, 2001). The primary sources of organic carbon in the estuarine sediment are estuarine primary production (autochthonous) and from the external supply (allochthonous) (Kelly and Levin, 1986; Nixon *et al.*, 1986). The dead planktonic matter in the estuary sinks to the bottom and get oxidised and on settling its decomposition releases organic matter into the interstitial water, part of which is then diffused into the overlying water (Bijoy Nandan and Abdul Azis, 1994; Smith and Hollibaugh, 1993; Wollast, 1998). The state of preservation of organic matter in sediment is depended partly on its texture as well as microbial and redox potential of the deposit. The dissolved and particulate material input through large rivers has a significant influence on the sediment characteristics of adjacent estuarine, coastal system (Herman *et al.*, 1999). The river plume has an essential control on estuarine sediments texture and enhancing the benthic production. River plumes supply sediments with phytodetritus that influence the benthic communities located beneath the plume, and it may enter benthic food webs (Hermand *et al.*, 2008). Moreover, the seaward margin of the plume can also act as a physical barrier to larval dispersal of benthic species with meroplanktonic development. The estuarine fronts may distribute with short-lived fauna compared to oceanic fronts, their periodically repetitive nature may ensure that they can induce geochemical and ecological responses from the benthic system (Hermand *et al.*, 2008). Sanders (1958) state that the association between infauna and sediments vary in food supply by the domination of sandy habitats by suspension feeders and muddy habitat by deposit feeders. Variation in colour and texture of sediments have brought about by changes in the grain size and state of oxidation of organic matter. Several factors such as oxygen exposure, the supply of reactive organic matter, sorptive preservation, mineral composition, winnowing, and redeposition may also control sedimentary organic matter. The shallow nature water estuary, higher temperature, and oxygenated environment seem to encourage oxidation of organic matter (Martin, 1970). Therefore, the seasonal variation in the organic carbon content in the sediments may be related to organic production in the overlying water, the humic material brought in from land and also to the oxidation of organic matter by benthos (Macnae, 1969). However, an overabundance of organic matter may lead to declines in species richness, abundance, and biomass due to oxygen depletion and accumulation of toxic by-products (ammonia and sulphide) connected with the decay of these materials (John *et al.*, 2002; Pearson and Rosenberg, 1978; Sankaranarayanan and Panampunnayil, 1979). The high organic matter in sediment may also lead to increasing amounts of physiological stress to organisms; from oxygen deficiency (due to BOD) and related by-products of the organic decomposition process (ammonia and sulphides). The shallow nature, higher temperature, and oxygenated environment seem to encourage oxidation of organic matter (Diaz and Rosenberg, 1995). Concurrently, higher levels of organic matter often correlate with increasing concentrations of other potential co-varying stressors (e.g., chemical contaminants). Therefore, benthic fauna in muddy depositional environments must often cope with multiple co-occurring stressors. Typically, benthic assemblages under such conditions dominated by a few pollution-tolerant, r-selected opportunistic species (Borja *et al.*, 2000). Species richness typically will show a gradual decline over the intermediate organic matter range, as increasing numbers of sensitive species fail to survive (Borja and Tunberg, 2011; Feebarani *et al.*, 2016). The heavy disturbances that are too severe may eliminate even the heartiest species, resulting in azoic conditions (Bijoy Nandan and Abdul Azis, 1995a; Sivadas *et al.*, 2010). Thus, benthic fauna, especially those in muddy depositional environments, must often cope with multiple, co-occurring stressors. Grain size co-varies with the sedimentary organic matter availability, interstitial water chemistry and microbial abundance and composition, however, all of which are influenced by the near-bed flow regime. These variables could be directly or indirectly affecting distribution (Botto and Iribarne, 2000; Flemming, 2000).

Chapter 4
Benthic standing crop of macrobenthos

IV.1. Introduction

Benthos are ecologically important sessile or sedentary organisms that are directly exposed to environmental changes in the overlying water mass (Brown *et al.*, 2004). The information on benthos is crucial in understanding the actual state of an aquatic ecosystem (Chuks Chindah, 1998) and it cannot be achieved by a simple measurement of water quality (Li *et al.*, 2010). Benthic organisms have a vital link in energy flow, from the standpoint of benthic secondary production and recycling of organic matter (Crisp, 1984). Therefore, it is a prerequisite for estimation of benthivorous fish production potential (Parulekar *et al.*, 1982). Such assessments are in turn necessary for devising reasonable management measures, not only of fisheries but also for the entire ecosystem.

The quantitative study on marine benthos was introduced by Hensen (1880s), and further Petersen (1915) initiated such studies in shallow waters. The series of scientific studies were further carried out in various parts of the world that mainly focused on estimating the fish production potential (Parker, 1975). Later, Thorson (1957) explained the concepts of marine benthic communities and afterwards many studies had been carried in the marine environments of different parts of the world. Annandale (1907) and Annandale and Kemp (1915) has initiated the quantitative ecology study on benthos in India by the investigations in the Gangetic delta and Chilka Lake. The Preston (1916) further described many new species of benthic molluscs from brackish waters of India, but his studies are restricted to taxonomic investigations. Subsequently, many benthic studies focused on coastal and estuaries waters of India. The important previous studies were carried out by Ansari (1978) in Karwar estuary, Varshney *et al.* (1981) in Narmada estuary, Ansari and Parulekar (1993) in Mandovi estuary of Goa (Ansari and Parulekar, 1998), Kumary (2008 in Poonthura estuary, Chinnadurai and Fernando (2006) in mangroves of Parangipettai, Santhanam *et al.* (1995) in Pullavali brackish water, Abdul Azis and Nair (1983) in Edava-Nadayara and Paravur backwaters of Kerala, and Nair *et al.* (1984a) in Kadinamkulam and Ashtamudi estuaries. Many studies were also carried in the Vembanad-Kol wetland ecosystem of south-west coast India by Desai and Krishnankutty (1967), Kurian (1972), Kurian *et al.* (1975), Ansari (1977), Pillai (1977), Pillai (1978), Batcha (1984), Anvar Bachan (1984), Sarala Devi (1986), Sunil Kumar (1993), Sheeba (2000), Sunil Kumar (2002), Feebarani (2009), John (2009), Geetha *et al.* (2010), Asha *et al.* (2016), and Rehitha *et al.* (2017). These studies revealed the pattern of distribution in the major macrofaunal groups that are contributed to diversity, density and biomass. However, there is no comprehensive study has evolved in the benthic ecology of the Kodungallur-Azhikode estuary.

In this scenario, the present study will provide quantitative ecology of benthos in the Kodungallur-Azikode estuary (KAE) in undeniable terms. The data presented here will provide valuable information against which further changes in the benthic community can assess. Especially the data on different benthic community and its production strategies will give a better idea on the status of the estuarine system, and it can use for proper monitoring and management activities.

IV. 2. Results

IV. 2. 1. Benthic standing stock of macrofauna

About nine diverse taxonomic groups (class) were encountered among the macrofauna during the study period (July 2009 to June 2011). The numerical density of macrofauna in study area varied between 23 ind.m^{-2} (station 7, November 2011) to 87568 ind.m^{-2} (station 3, September 2011) with an overall mean of 3887 ± 10083 ind.m^{-2}. Spatial and temporal variations have observed in the numerical density and biomass of macrofauna. A total of 18846 organisms collected in the grab samples, of which, 60 percentage was malacostracan crustaceans. They were the dominant group during the entire study, followed by polychaetes (20 %), molluscs (9 %) and the sporadic representatives were pooled together as 'others group' (11 %). They represented by hydrozoans, cirripedians, insects, nemerteans, benthic fishes and ophiuroideans. During the first year period (2009-2010), the numerical density of macrofauna was 2616 ± 4253 ind.m^{-2}. The malacostracan crustaceans formed 51 percentage to the total numerical density of macrofauna, followed by polychaetes (29 %), molluscs (11 %) and other (9 %). However in the second year (2010-2011), that was 5157 ± 13535 ind.m^{-2}. The malacostracans crustaceans formed 64 percentage to total macrofaunal density followed by polychaetes (15 %), molluscs (9 %) and other (12 %). Mean density of macrofauna collected during the study is provided in Tables 4 a-c.

Biomass of macrofauna was estimated on a wet weight basis, after sorting them into four major groups such as malacostracan crustaceans, polychaetes, molluscs (bivalves and gastropods) and 'other group', which included all faunal groups represented in few numbers (hydrozoans, cirripedians, insects, nemerteans, benthic fishes and ophiuroideans). Station-wise mean macrofaunal biomass ranged from lowest of 3.68 ± 4.45 g.m^{-2} at station 7 to highest of 90.93 ± 76.79 g.m^{-2} in station 3 with an overall mean of 27.92 ± 49.09 g.m^{-2} for seven stations in KAE [Tables 5 a-c]. During the entire study, malacostracans (crustaceans) contributed 18 percentage to total biomass of macrofauna that was 17 percentage for polychaetes, 63 percentage for molluscs, and 2 percentage for other group. In the first year (2009-2010), mean macrobenthic biomass was 24.72 ± 41.72 g.m^{-2} and malacostracans formed 15 percentage to total biomass followed by polychaetes (27 %), molluscs (57 %) and other group (1 %). During the second year (2010-2011), mean macrobenthic biomass was 31.13 ± 55.56 g.m^{-2} with the dominance of malacostracans (21 %) that followed by polychaetes (10 %), molluscs (67 %) and others group (2 %). A statistical tool such as one-way ANOVA was performed to test the significant differences in density and biomass of macrofauna and faunal groups during the study.

a) Variation in macrofaunal density

In all surveys, no significant differences in macrofaunal numerical density have observed between both years of study (ANOVA $F\,(1,166) = 2.696$, $p = 0.103$).

The mean density was 3887 ± 10083 ind.m^{-2} for the entire study and values ranged between 1039 ± 928 ind.m^{-2} (station 6) and 10337 ± 3896 ind.m^{-2} (station 3). The mean density of macrofauna during the first year period was 2616 ± 4253 ind.m^{-2} and that for the second year was 5157 ± 13535 ind.m^{-2} [Figure 31]. Similarly, no significant difference was observed between seasons (ANOVA $F(5,162) = 1.056$, $p = 0.387$). Seasonally, the highest mean numerical density of 6574 ± 21092 ind.m^{-2} was observed during the monsoon season of the second year, followed by post-monsoon of the second year (5771 ± 15921 ind.m^{-2}) and the first year (3677 ± 8474 ind.m^{-2}). The lowest density was observed during pre-monsoon (1473 ± 2468 ind.m^{-2}) and monsoon (2687 ± 5319 ind.m^{-2}) of the first year [Table 4 a-c]. The overall monthly mean density of macrofauna in KAE showed the highest during November 2010 (16188 ± 22099 ind.m^{-2}) and lowest during February 2011(1127 ± 856 ind.m^{-2}).

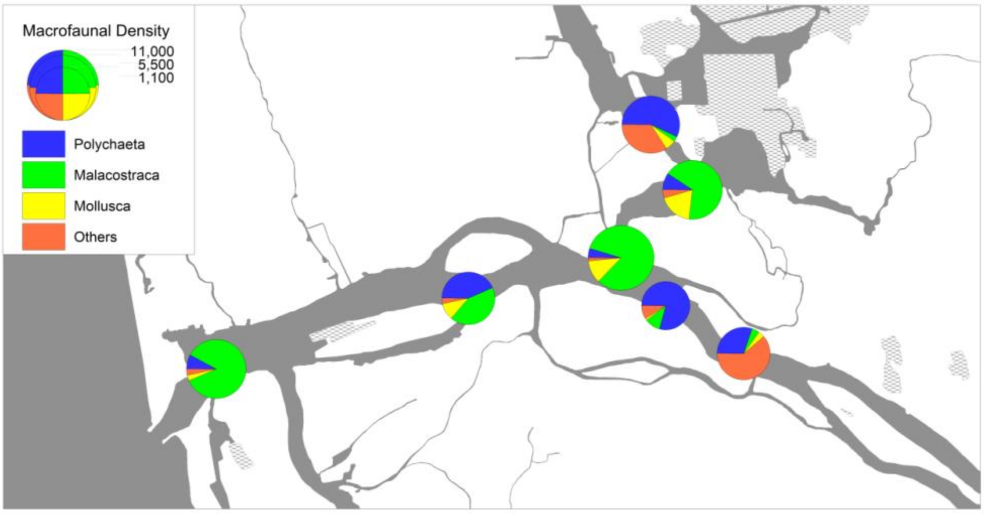


Figure 31. Mean percentage contribution of faunal groups to density of macrofauna at each station in KAE during the 2009-2011 period, size of each pie charts depicts the variation between stations

A significant variation was observed in density between stations (ANOVA F (6,161) = 16.087, p = 0.000). Among the seven stations, mean lowest density has recorded at riverside stations towards the southern arm of the estuary [Table 4 a-c], such as station seven (1039 ± 928 ind.m^{-2}) and station six (1686 ± 4898 ind.m^{-2}). However, the highest mean density was observed at mixing zone of the estuary, mainly in station 3 (10337 ± 3896 ind.m^{-2}) and station four (4652 ± 6227) while an intermediate density was observed at the estuarine mouth as station one (4090 ± 12052 ind.m^{-2}).

Table 4 a. Season-wise macrofaunal abundance (ind.m^{-2}) in KAE during 2009-2010 and 2010-2011 period

Seasons	Polychaeta	Malacostraca	Mollusca	Others	Total
Mon.09-10	1006 ± 1404	1261 ± 3205	409 ± 667	12 ± 43	2687 ± 5319
Pos.09-10	656 ± 756	2231 ± 6244	276 ± 478	515 ± 1007	3677 ± 8474
Pre.09-10	619 ± 553	494 ± 1073	203 ± 403	158 ± 439	1473 ± 2468
Mon.10-11	552 ± 751	5575 ± 19274	375 ± 718	76 ± 349	6574 ± 21092
Pos.10-11	1285 ± 3181	4169 ± 12230	312 ± 499	6 ± 11	5771 ± 15921
Pre.10-11	485 ± 501	204 ± 444	688 ± 1174	1740 ± 5384	3116 ± 7503

Table 4 b-c. Station-wise macrofaunal abundance (ind.m^{-2}) in KAE during 2009-2010 and 2010-2011 period

Table 4 b. Macrofaunal abundance (ind.m^{-2}) during 2009-2010 period

Station	Polychaeta	Malacostraca	Mollusca	Others	Total
St.1	496 ± 618	1991 ± 4166	167 ± 260	295 ± 709	2951 ± 4269
St.2	1013 ± 1916	123 ± 108	62 ± 64	131 ± 305	1333 ± 1932
St.3	576 ± 666	2693 ± 5005	706 ± 802	342 ± 899	4320 ± 4757
St.4	570 ± 510	4114 ± 8236	756 ± 793	358 ± 1015	5801 ± 8178
St.5	1208 ± 918	95 ± 78	250 ± 368	249 ± 592	1805 ± 1158
St.6	879 ± 959	157 ± 205	23 ± 32	194 ± 655	1256 ± 1028
St.7	578 ± 460	127 ± 156	108 ± 110	29 ± 64	845 ± 555

Table 4 c. Macrofaunal abundance (ind.m^{-2}) during the 2010-2011 period

Station	Polychaeta	Malacostraca	Mollusca	Others	Total
St.1	150 ± 217	5008 ± 16864	72 ± 80	5 ± 13	5237 ± 16812
St.2	667 ± 884	1523 ± 5111	341 ± 810	7 ± 14	2540 ± 5216
St.3	375 ± 453	14331 ± 28684	1648 ± 1207	5 ± 13	16362 ± 28281
St.4	299 ± 220	2155 ± 3244	975 ± 878	79 ± 221	3512 ± 3366
St.5	2720 ± 4542	102 ± 129	150 ± 154	2113 ± 4535	5087 ± 6077
St.6	761 ± 806	61 ± 61	2 ± 7	3 ± 7	830 ± 803
St.7	445 ± 605	32 ± 48	21 ± 26	2033 ± 7011	2534 ± 6950

b) Variation in macrofaunal biomass

There were no significant differences observed between annual surveys on macrofaunal biomass (ANOVA $F\,(1,166) = 0.714$, $p = 0.399$. The mean biomass was 27.92 ± 49.09 $g.m^{-2}$ during the present study and that for the first year was 24.72 ± 41.72 $g.m^{-2}$ and 31.13 ± 55.56 $g.m^{-2}$ in the second year of study. Temporarily, highest mean biomass was observed during the monsoon season of the second year (37.62 ± 68.04 $g.m^{-2}$) and the first-year (31.66 ± 44.71 $g.m^{-2}$). However, lowest mean biomass depicted during pre-monsoon of the first year (16.11 ± 28.79 $g.m^{-2}$) and the second year (25.13 ± 43.21 $g.m^{-2}$) [Table 5 a-c]. Similarly, no significant difference was observed between seasons (ANOVA $F\,(5,162) = 0.605$, $p = 0.696$) [Figure 32].

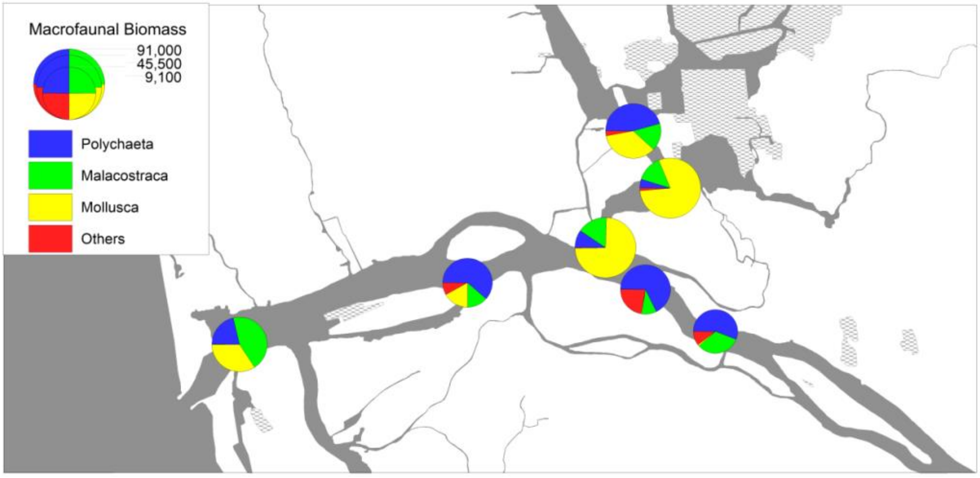


Figure 32. Mean percentage contribution of faunal groups to the biomass of macrofauna at each station in KAE during the 2009-2011 period, the size of each pie charts depicts the variation between stations

The highest monthly mean of macrofaunal biomass was observed during July 2010 (73.41 ± 100.58 $g.m^{-2}$) and followed by November 2010 (56.89 ± 82.29 $g.m^{-2}$), while lowest biomass was observed during March 2010 (7.83 ± 4.80 $g.m^{-2}$), November 2009 (8.93 ± 6.71 $g.m^{-2}$) and January 2011 (9.02 ± 7.06 $g.m^{-2}$). In the spatial scale, the highest mean of 90.93 ± 76.79 $g.m^{-2}$ was recorded at station 3 with the maximum of 270.12 $g.m^{-2}$, followed by station 4 (52.52 ± 52.46 $g.m^{-2}$) with the maximum of 223.66 $g.m^{-2}$. Similarly, moderate values were recorded at station 1 (18.42 ± 36.95 $g.m^{-2}$) and station five (16.43 ± 27.99

g.m^{-2}). However, the lowest mean biomass of 3.68 ± 4.45 g.m^{-2} was noticed at station seven, that for station six was 5.62 ± 5.99 g.m^{-2} and 7.86 ± 9.35 g.m^{-2} at station two. The mean biomass of macrofauna varied between stations and found to be statistically significant (ANOVA F (6,161) = 2.519, p =0.023), but the variation between monthly surveys was marginal.

Table 5 a. Season-wise macrofaunal biomass (g.m^{-2}) in KAE during 2009-2010 and 2010-2011 period

Seasons	Polychaeta	Malacostraca	Mollusca	Others	Total
Mon.09-10	6.62 ± 7.03	7.04 ± 15.31	17.79 ± 36.56	0.21 ± 0.81	31.66 ± 44.71
Pos.09-10	5.26 ± 12.33	1.93 ± 5.98	18.47 ± 41.42	0.74 ± 3.3	26.4 ± 48.93
Pre.09-10	6.38 ± 8.84	2.2 ± 5.19	6.93 ± 22.91	0.59 ± 2.67	16.11 ± 28.79
Mon.10-11	2.89 ± 4.74	9.82 ± 30.6	24.37 ± 58.7	0.54 ± 1.72	37.62 ± 68.04
Pos.10-11	3.25 ± 4.06	7.44 ± 16.37	19.71 ± 49.35	0.47 ± 2.36	30.87 ± 54.55
Pre.10-11	4.02 ± 6.07	2.2 ± 5.28	18.12 ± 37.48	0.79 ± 2.81	25.13 ± 43.21

Table 5 b-c. Station-wise macrofaunal biomass (g.m^{-2}) in KAE during 2009-2010 and 2010-2011 period

Table 5 b. Macrofaunal biomass (g.m^{-2}) during 2009-2010 period

Station	Polychaeta	Malacostraca	Mollusca	Others	Total
St.1	5.08 ± 6.1	1.2 ± 2.26	8.76 ± 27.35	0.07 ± 0.12	15.1 ± 28.74
St.2	4.82 ± 5.75	0.31 ± 0.53	0.05 ± 0.08	-	5.18 ± 5.75
St.3	12.5 ± 20.35	10.11 ± 10.67	43.06 ± 53.89	0.24 ± 0.6	65.91 ± 65.84
St.4	2.93 ± 3.25	10.98 ± 20.99	37.34 ± 40.32	1.58 ± 4.11	52.82 ± 48.75
St.5	10.34 ± 7.98	2.45 ± 7.83	11.44 ± 39.08	0.11 ± 0.39	24.34 ± 37.6
St.6	4.14 ± 5.14	0.28 ± 0.66	0.04 ± 0.12	1.46 ± 5.05	5.92 ± 6.65
St.7	2.83 ± 3.17	0.72 ± 1.8	0.07 ± 0.13	0.15 ± 0.22	3.77 ± 3.58

Table 5 c. Macrofaunal biomass during 2010-2011 period

Station	Polychaeta	Malacostraca	Mollusca	Others	Total
St.1	2.68 ± 5	14.19 ± 40.1	3.54 ± 10.14	-	20.41 ± 43.13
St.2	5.09 ± 5.73	1.77 ± 5	2.42 ± 6.16	1.19 ± 3.96	10.47 ± 11.09
St.3	4.75 ± 8.27	19.09 ± 27.9	91.8 ± 77.56	0.32 ± 1.11	115.95 ± 81.41
St.4	1.92 ± 1.52	3.9 ± 4.6	46.41 ± 58.1	-	52.23 ± 58.1
St.5	4.72 ± 5.45	2.82 ± 8.97	0.08 ± 0.08	0.91 ± 2.35	8.53 ± 9.39
St.6	3.47 ± 3	0.8 ± 1.77	-	1.06 ± 3.67	5.32 ± 5.53
St.7	1.29 ± 1.41	1.65 ± 5.09	0.01 ± 0.02	0.63 ± 1.38	3.58 ± 5.35

IV. 2. 2. Macrofaunal communities

a) Malacostraca

Malacostracan crustaceans in the present investigation were primarily represented by the amphipods and isopods, with significant representation from decapods, cumaceans, tanaids, stomatopods and mysids. They were the most dominant group concerning the number of individuals, representing 60 percentage of all macrofaunal organisms collected during the entire study. While their biomass represented about 18 percentage that of total macrofauna.

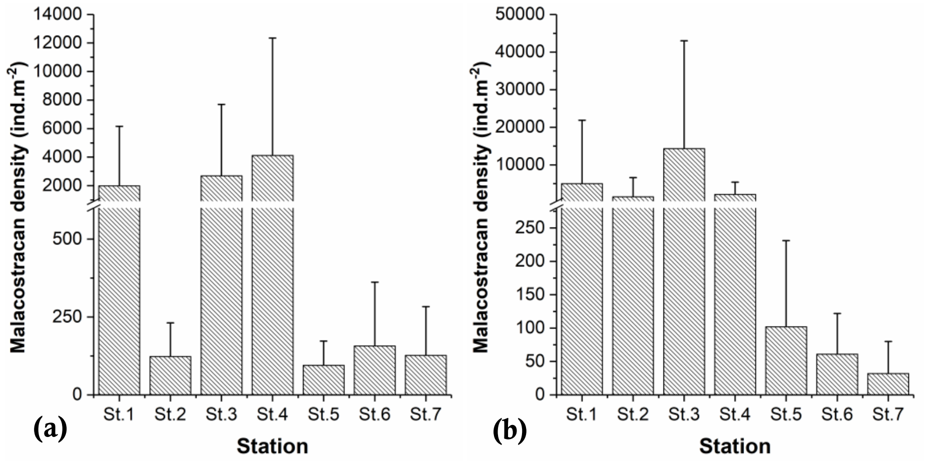


Figure 33 a-b. Mean (±SD) malacostracan crustacean density in different stations of KAE during (a) 2009-2010 and (b) 2010-2011 period

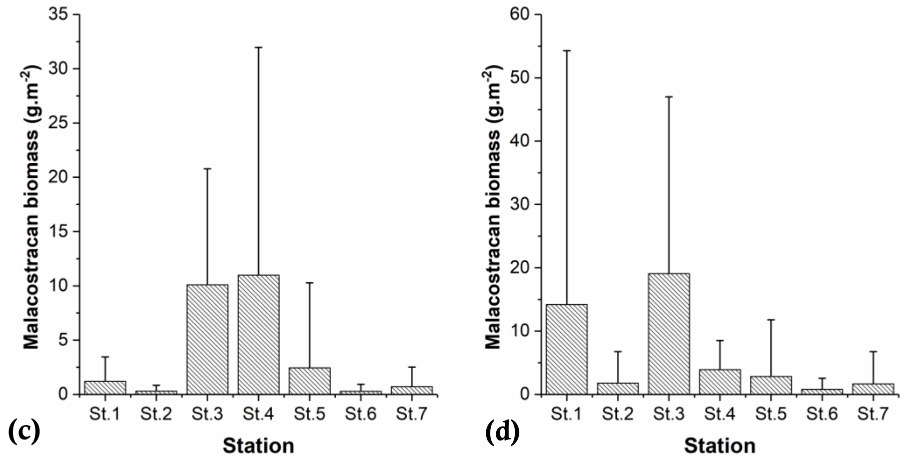


Figure 33 c-d. Mean (±SD) malacostracan crustacean biomass in different stations of KAE during (c) 2009-2010 and (d) 2010-2011 period

Mean density of malacostracan for the entire study period was 2322 ± 9812 ind.m^{-2}, and that was 1328 ± 4112 ind.m^{-2} during the first period and 3316± 13221 ind.m^{-2} during the second year. The mean density varied appreciably with stations, and that was found to be highest at station 3 (8512 ± 20996 Ind.m^{-2}) and lowest at station 7 (80 ± 123 ind.m^{-2}), with an intermediate value at station 1 (3499 ± 12112 ind.m^{-2}) and station 4 (3134 ± 6203 ind.m^{-2}). A significant difference in density was noticed between stations (ANOVA F (6,161) = 2.516, p =0.023). The relative abundance of this group was a maximum of 85.55 percentage at station one. While that was 42.61 percentage at station two, 82.35 percentage for station three, 67.38 percentage for station four, 2.88 percentage for station five, 10.58 percentage for station six and least value of 2.09 percentage at station seven [Figure 33 a-b]. In a seasonal scale, the mean density was highest during post-monsoon season of the second year survey (5576 ± 19274 ind.m^{-2}) and lowest during pre-monsoon season of the first year (204 ± 444 ind.m^{-2}). Over 15440 amphipods collected in 168 grab samples accounted for 55 percentage of the total density of macrofauna. Numerical density of other crustaceans were 1544 isopods (5.52 %), 234 decapods (0.42 %), 47 tannaids (0.17 %), 12 cumenceans (0.04 %), 5 mysids (0.03 %) and 1 stomatopod (0.004 %) during the entire period of study. Similarly, the mean biomass of malacostracans in the study period was 5.09 ± 15.90 g.m^{-2}, and that was 3.40 ± 9.98 g.m^{-2} for the first year and 6.78 ± 20.09 g.m^{-2} for the second year survey [Figure 33 b-d]. A significant variation was observed between stations (ANOVA F (6,161) = 2.751, p =0.014).

b) Polychaeta

Polychaetes were the second dominant group in terms of a number of individuals, representing 20 percentage of all organisms collected during the study, while polychaete biomass represented about 17 percentage that of total macrofauna. A total of 5669 polychaetes were collected during the entire study; subclass Errantia contributed 1811 organism (6.67 %) and 3858 individuals (13.80 %) of subclass Sedentaria by 168 grab haul [Figure 35]. Mean density of polychaetes during the entire study period was 767 ± 1520 ind.m^{-2} and that for the first year period was 760 ± 979 ind.m^{-2} and the second year was 774 ± 1921 ind.m^{-2}. There was no significant difference in polychaete density was observed between years (ANOVA $F(1,166) = 0.003$, $p = 0.953$) and seasons (ANOVA $F(5,162) = 1.069$, $p = 0.380$), but a significant differences was observed between stations (ANOVA $F(6,161) = 3.596$, $p = 0.002$).

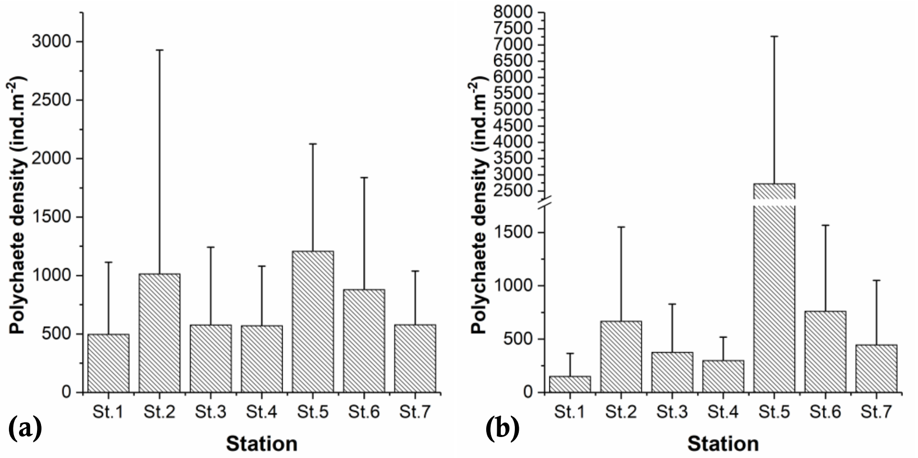


Figure 34 a-b. Mean (±SD) polychaete density in different stations of KAE during (a) 2009-2010 and (b) 2010-2011 period

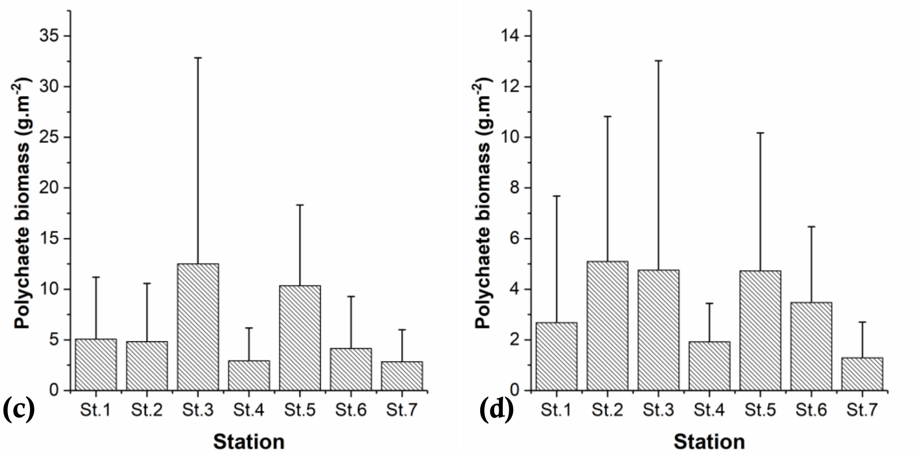


Figure 34 c-d. Mean (±SD) polychaete biomass in different stations of KAE during (c) 2009-2010 and (d) 2010-2011 period

The mean density of polychaetes varied appreciably with stations and mean value was found to be highest at station five (1965 ± 3296 ind.m^{-2}) and lowest at station one (324 ± 486 ind.m^{-2}), with an intermediate value at station two(840 ± 1470 ind.m^{-2}). The relative abundance of this group was 7.92 percentage for station one that for station two was 43.49 percentage, 4.60

percentage for station three, 9.37 percentage for station four, 57.07 percentage for station five, 78.94 percentage for station six and 13.18 percentage for station seven [Figure 34 a-b]. The mean density of polychaetes was highest observed during the post-monsoon season of the second year (1241 ± 3133 ind.m^{-2}) and similarly lowest during pre-monsoon season of the second year (485 ± 501 ind.m^{-2}). Mean biomass of polychaetes in the entire study was 4.74 ± 83 g.m^{-2} and that for the first year was 6.19 ± 66 g.m^{-2} ind.m^{-2} and 3.23 ± 100 g.m^{-2} for the second year [Figure 34 c-d]. There was a significant difference in polychaete biomass observed between years (ANOVA F (1,166) = 5.270, p =0.023), and stations (ANOVA F (6,161) = 2.629, p =0.023), however, no significant variation was observed between seasonal biomass (ANOVA F (5,162) = 1.195, p =0.314).

Figure 35. Mean percentage contribution of different subclasses to the total density of polychaetes at each station in KAE during the 2009-2011 period, the size of each pie charts depicts the variation between stations

c) Molluscs

Among the molluscs, bivalves constituted a significant group with few representations from gastropods. In all samples, 2788 clams were collected during the entire study period, making molluscans as one of the numerically dominant taxonomic groups (9.71 % of total macrofauna). The contribution of bivalves was 8.81 percentage, and that for gastropods was 0.90 percentage to overall macrofaunal density in the estuary. The mean density of molluscs in the

KAE was 377 ± 711 ind.m^{-2} during the entire period study and that for the first year was 296 ± 528 ind.m^{-2} and 458 ± 851 ind.m^{-2} for the second-year survey. Stations in the mixing zone of the estuary were distinct in their faunal composition from the other sampling sites.

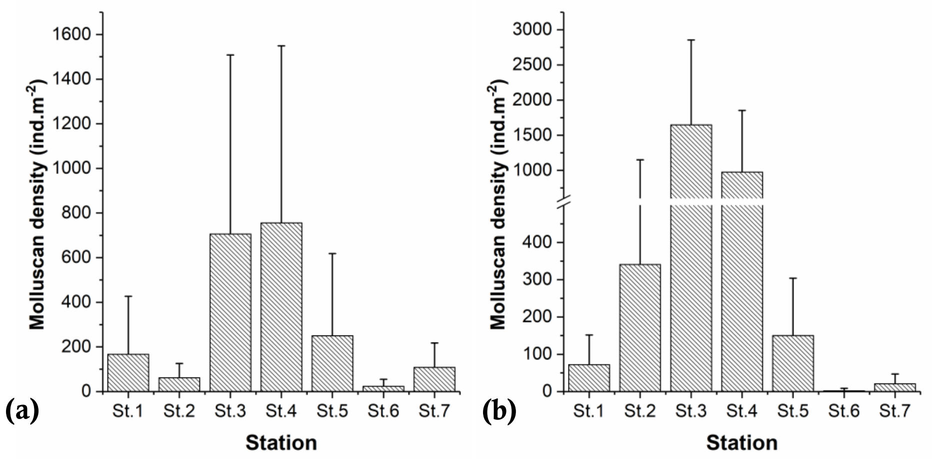


Figure 36 a-b. Box plot of molluscan density in different stations of KAE during (a) 2009-2010 and (b) 2010-2011 period

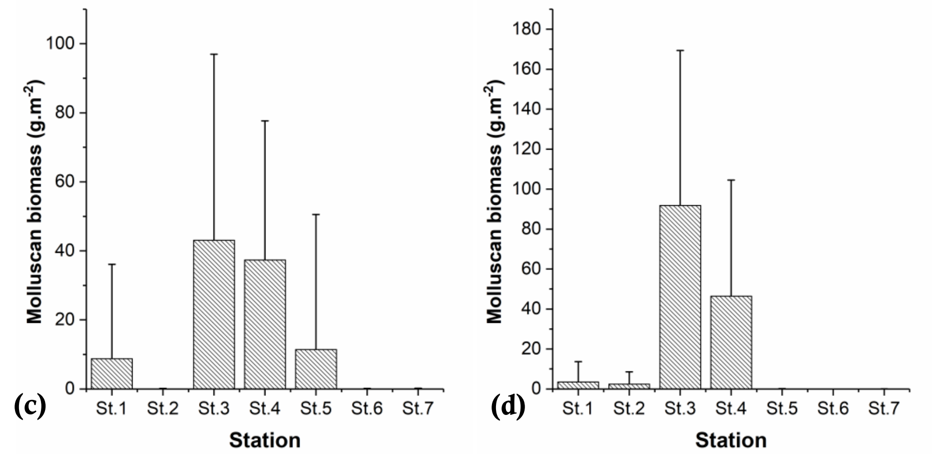


Figure 36 c-d. Mean (±SD) molluscan biomass in different stations of KAE during (c) 2009-2010 and (d) 2010-2011 period

The highest abundance of molluscs was obtained at stations three (1177 ± 1112 ind.m^{-2}) and lowest at the riverine side, station six (12 ± 25 ind.m^{-2}) and station seven (64 ± 90 ind.m^{-2}) while an intermediate density depicted at station four (866 ± 826 ind.m^{-2}). The relative abundance of the molluscan group varied as 2.93 percentage in station one, 10.03 percentage in station two, 11.39 percentage in station three, 18.63 percentage in station four, 5.84 percentage in station five, 1.25 percentage in station six and 1.67 percentage in station seven [Figure 36 a-b]. The wet weight of molluscs was taken without shells, but larger specimens (>3 g) have not included when considering mean values. Molluscs represented about 63 percentage of the total biomass. Mean biomass of mollusc in the grab samples varied from 0.02 ± 0.08 $g.m^{-2}$ in station six to 67 ± 70 $g.m^{-2}$ in station 3 with an average of 17 ± 54 $g.m^{-2}$ for the study. The mean biomass of molluscs at each sampling site is given in Figure 36 c-d. Variations in density and biomass of molluscs were not statistically significant at yearly and monthly surveys, while station wise variation was significant (density: (ANOVA F (6,161) = 14.570, p =0.00); biomass: (ANOVA F (6,161) = 13.907, p =0.023).

d) Other Groups

Other groups represented in the samples were hydrozoans (order Leptothecata), cirripedians (order: Sessilia), insects (order: Diptera), Pisces (order perciforms), ophiuroideans (order: Ophiurida) and nemerteans. They contributed 11 percentage of the total mean numerical density of macrofauna in KAE.

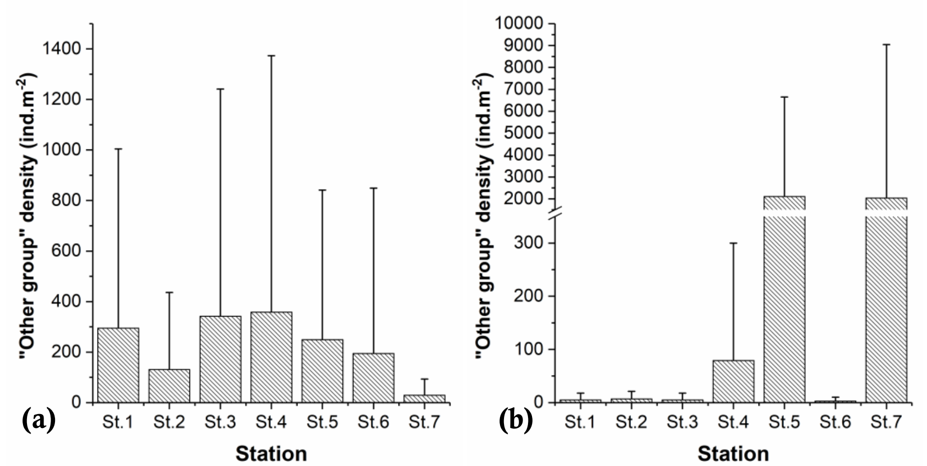


Figure 37 a-b. Mean (±SD) "other group" density in different stations of KAE during (a) 2009-2010 and (b) 2010-2011 period

Of these, hydrozoans were well represented seasonally at some stations, and these contributed more than 10 percentage to the total density in the study area, and this increased representation is mainly due to colony formation of hydrozoan. Pisces (0.49 %) and cirripedians (0.24 %) were also contributed to the numerical density of macrofauna in some stations in the estuary. Hydrozoan (92.61 %) were the most representing members in "other group" followed by Pisces (4.50 %), cirripedians (2.18 %), nemerteans (0.36 %), ophiuroideans (0.23 %) and insects (0.13 %) to the total density of "other groups" [Figure 37 a-b]. The combined biomass of these groups varied from nil to 17.50 g.m^{-2}, they accounted for two percentage of the total macrofaunal biomass in the entire study with mean biomass of 0.56 ± 132 g.m^{-2}. That was 0.56 ± 2.47 g.m^{-2} during the first year and 0.60 ± 2.33 g.m^{-2} for the second year [Figure 37 c-d].

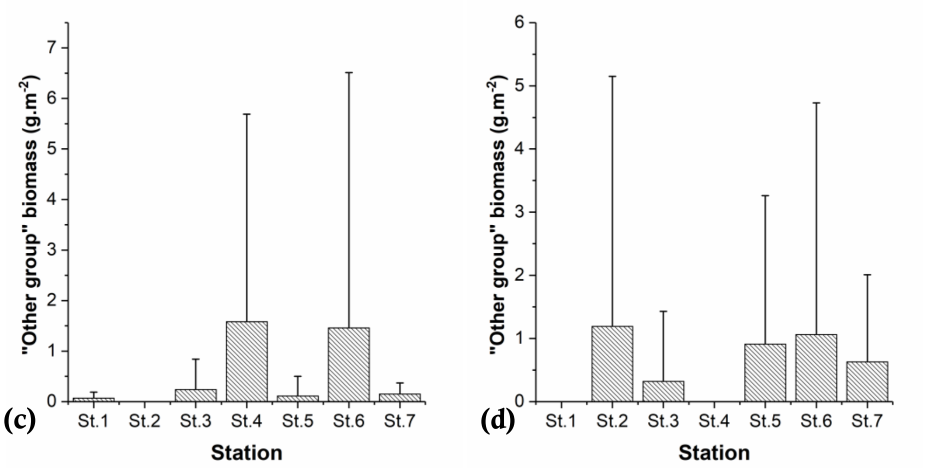


Figure 37 c-d. Mean (±SD) "other group" biomass in different stations of KAE during (c) 2009-2010 and (d) 2010-2011 period

IV. 2. 3. Trophic support of macrofauna to fishery

The benthic secondary production in the estuary supported a diverse number of fish species and their larval development (Parulekar *et al.*,1980). The average macrofaunal biomass for the study area was found to be 27.65 g.m^{-2} (27645 ± 48375 kg.km^{-2}). It was 24.72 g.m^{-2} (24723 ± 41720 kg.km^{-2}) during first year (2009-2010) and 30.57 g.m^{-2} (30570 ± 55030 kg.km^{-2}) for the second year (2010-2011). Most species of macrobenthos have a lifespan of about one year, and if the suggestion of Sanders (1956) that the annual production of twice the standing crop for these organisms holds true, then the annual average macrobenthic output in the study area is about 4696 ± 8218 kg.C.km^{-2}.yr^{-1}. For the first year it was 4200 ± 7088 $kg.C.km^{-2}.yr^{-1}$ and that for the second year was 5288 ± 9449 $kg.C.km^{-2}.yr^{-1}$. Using the conversion factor of Brey *et al.* (2010), according to which dry weight is equivalent to 23.4 percent of the wet weight and organic carbon is 36.3 percent of dry weight, the average organic carbon value for the study area was 33875 ± 57527 $kg.C.yr^{-1}$ (total area: 7 km^{-2}), that was 29400 ± 49613 $kg.C.yr^{-1}$ for the first year and 36353 ± 66441 $kg.C.yr^{-1}$ for the second year. According to the laws of energy transfer, 15 percentage of total organic carbon is expected to be assimilated by the next trophic level in the coastal waters (Gulland, 1971; Ryther, 1969). For coastal waters, 60 percent of the live weight is supposed to fish, and for offshore waters, only 40 percent is considered to represent by fish (Steel, 1974). This value is converted to live weight by multiplication by a factor of 10. By this calculation, macrofauna of the study area alone contributes on average of 29588 ± 51774 kg of fish biomass during the study period. It was 26460 ± 44652 kg fish for the first year and about 32718 ± 58897 kg for the second year [Figure 38].

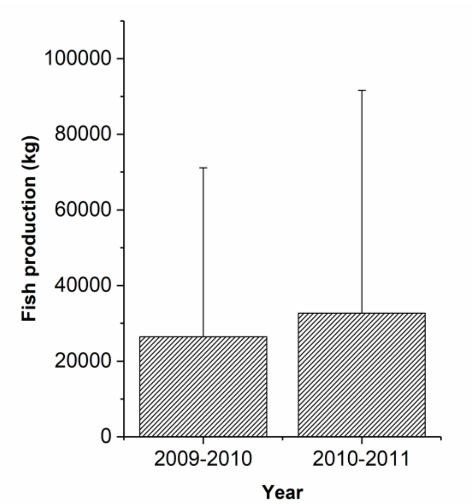


Figure 38. The estimated fish production (±SD) from the macrofaunal standing crop in the KAE

A multi-parameter artificial neural network model developed by Thomas Brey (2012) was used to estimate the somatic production-to-biomass ratio (P/B) in macrobenthic populations. The estimated macrobenthic P/B ratio for the estuary was 35.29 yr^{-1} during the first year and 36.21 yr^{-1} for the second year. Among the macrobenthic group, malacostracans exhibited relatively high P/B ratio, and it was 13.49 yr^{-1} for the first year and 12.81 yr^{-1} for the second year. P/B ration for polychaetes were 6.52 yr^{-1} in the first year and 7.13 yr^{-1} in the second year and that for molluscs were 3.32 yr^{-1} and 3.25 yr^{-1} respectively. The maximum P/B value was noticed during post-monsoon period of first year (41.40 yr^{-1}) followed by pre-monsoon of the second year (46.65 yr^{-1}), pre-monsoon of the first year (37.27yr^{-1}), monsoon of the second year (33.53yr^{-1}), post-monsoon of the second year (28.46 yr^{-1}) and least value was recorded at monsoon of first year (27.20 yr^{-1}). The estimated macrobenthic community production for the study area was 2554.19 J m^{-2} y^{-1} during the first year period and that was 3538.70 J m^{-2} y^{-1} during the second year. Molluscs exhibited maximum somatic production rate of 1397 75 J m^{-2} y^{-1} in the first year and 1515.51 J m^{-2} y^{-1} in the second year. Similarly, it was 165.67 J m^{-2} y^{-1} for malacostracans during the first year and 201.05 J m^{-2} y^{-1} for the second year, and that for polychaetes were 423.80 J m^{-2} y^{-1} for the first year and 270.20 J m^{-2} y^{-1} for the second year. The maximum macrobenthic secondary production was observed during the post-monsoon of the second year period followed by the monsoon of the first year of study [Table 6].

Table 6. Estimated production-to-biomass (P/B) ratio of macrobenthic populations (yr^{-1}) in the KAE during the 2009-2011 period

Period	Malacostraca	Polychaeta	Mollusca	Others
2009-2010	13.49	6.52	3.32	11.96
2010-2011	12.81	7.13	3.25	13.02

IV. 3. Benthic standing stock of Meiofauna

The meiofaunal organisms collected from the estuary has identified up to group level and the mean numerical depicted as 836 ± 1840 ind.10 cm^{-2}. Of these, 91.86 percentage were nematodes, which were the dominant group, followed by harpacticoid copepods (6.47 %), amphipods (1.26 %), foraminiferan (0.05 %), polychaete larvae (0.30 %) and the sporadic representatives were pooled together as 'others group' (0.06 %). The other group of meiofauna in the KAE represented by turbellarians, tardigrade, gastrotrichs, crustacean nauplii, and oligochaetes. During the first year period (2009-2010), the numerical density of meiobenthos was 797.66 ± 1772 ind.10 cm^{-2}. The nematodes formed 89.09 percentage of the total numerical abundance of meiofauna, followed by harpacticoid copepods (8.32 %), amphipods (2.05 %), foraminiferan (0.06 %), polychaete larvae (0.39 %) and other accounted for 0.09 percentage [Figure 39]. While, in the estuarine mouth, relatively high harpacticoid copepod population was observed. Similarly, during the second year period (2010-2011), it was 875.19 ± 404 ind.10 cm^{-2}. The major contribution from nematodes was 94.38 percent followed by harpacticoid copepods (4.79 %), amphipods (0.55 %), foraminiferan (0.04 %), polychaete larvae (0.21 %) and others (0.03 %). Seasonally, lowest density was observed during pre-monsoon of the first year (493.51 ± 1129 ind.10 cm^{-2}) and the monsoon of the second year (575 ± 2394 ind.10 cm^{-2}). However, a highest meiofaunal density was observed during pre-monsoon of the second year (1369 ± 1940 ind.10 cm^{-2}) and monsoon the first year (1208 ± 1817 Ind.10 cm^{-2}).

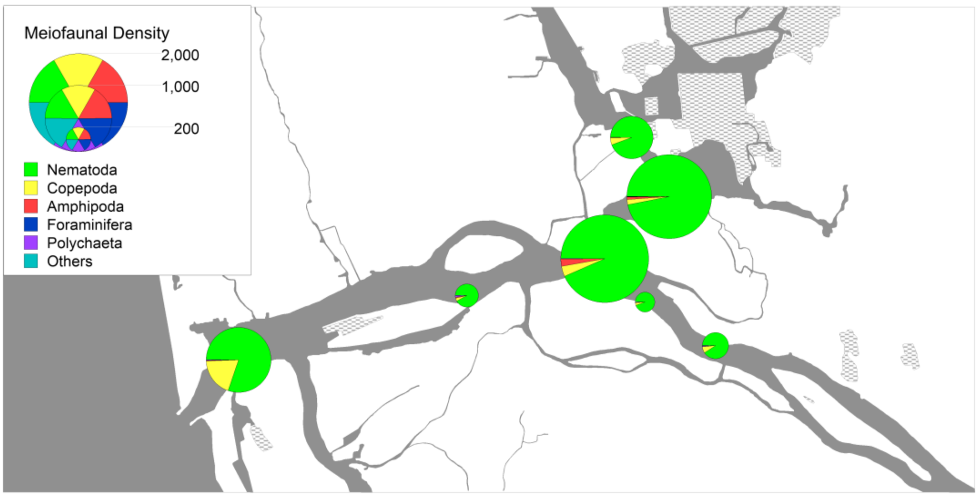


Figure 39. Mean percentage contribution of faunal groups to the density of meiofauna at each station in KAE during the 2009-2011 period, the size of each pie charts depicts the variation between stations

Spatial meiobenthic numerical density was lowest at station 6 (158 ± 393 ind.10 cm^{-2}), station two (167 ± 300 ind.10 cm^{-2}), station seven (212 ± 466 ind.10 cm^{-2}) and station five (506 ± 877 ind.10 cm^{-2}) while highest density was towards station three (1917 ± 3604 ind.10 cm^{-2}), station four (1796 ± 2600 ind.10 cm^{-2}) and station one (1099 ± 1618 ind.10 cm^{-2}). The significant difference in numerical density was observed between stations (ANOVA F (6, 161) = 420, p = 0.001). In the entire study period, nematodes dominated with 90.99 percentage of the total biomass of meiofauna, that for harpacticoid copepods was 6.41 percentage. However contribution of amphipods (1.25 %), foraminiferan (0.05 %), polychaete larvae (0.29 %) and other group (0.06 %) were negligible. During the first year period (2009-2010), mean meiobenthic biomass was 0.36 ± 0.70 mg 10 cm^{-2} and the second year period (2010-2011) mean meiofaunal biomass was 0.39 ± 1.02 mg 10 cm^{-2} with an overall mean of 0.38 ± 0.87 mg 10 cm^{-2}. Seasonal low meiofaunal biomass has recorded during pre-monsoon of the first year (0.22 ± 0.51) and the monsoon of the second year (0.26 ± 1.08 mg 10 cm^{-2}). Thus high biomass was observed towards monsoon of the first year (0.54 ± 0.82 mg 10 cm^{-2}), and pre-monsoon the second year (0.62 ± 1.21 mg 10 cm^{-2}). Station-wise mean meiofaunal biomass ranged from lowest of 0.01 ± 0.18 mg 10 cm^{-2} at station six to highest of 0.86 ± 1.62 mg 10 cm^{-2} at station three.

IV. 4. Discussion

Distribution of estuarine benthic communities shows the variability on the spatial and temporal scale, understandings on such changes in ecological communities are essential in assessing the environmental status of an estuary. Macrobenthic density and abundance in the estuaries are corresponding to successional dynamic in response to changes in surface sediment characteristics, total organic carbon, salinity, depth, current velocities, turbidity front, and dissolved oxygen (Ansari et al., 1986; Barros et al., 2008; Coull, 1999; Ingole and Parulekar, 1998; Rutledge and Fleeger, 1993; Ysebaert et al., 2003; Yu et al., 2012). Those factors are directly influencing the succession and recovery mechanisms of macrofauna to disturbances (Hermand *et al.*, 2008); therefore, macrofaunal population dynamics have a critical role in ecosystem perturbation studies. In the present study, spatiotemporal variation in macrofaunal distribution was evident in Kodungallur-Azhikode estuary (KAE), in which station wise changes were highly significant. According to Ysebaert and Herman (2002), local ecological variables such as 'spatial' components especially mud content, Chl-*a* and bed level height were the major factor in macrobenthic variation. While no significant variation was explained by 'temporal' components for several macrobenthic species in the estuaries. In the present stud, macrofaunal biomass showed a spatial variability; the middle zone contributed significantly to total abundance and biomass. However, estuarine mouth region depicted a moderately low numerical density and biomass of macrofauna, and it could be due to lack of stable and suitable substratum for larval settlement in this high energy zone, such conditions are prevalent in estuarine environment (Griffiths *et al.*, 2017; Madhu *et al.*, 2010; Sheeba, 2000; Ysebaert et al. (1993).

Malacostracan crustaceans were the most dominant group in the estuary that contributed significantly to the total numerical density of macrofauna (r = 0.961, $p < 0.01$), which is also comparable to other estuaries (Knox, 2006; Kumar and Khan, 2013). According to Nair *et al.* (1983a), several species of amphipods and isopods in the estuaries are tolerant to wide range of variation in salinity. Therefore such species become dominant in some of these waters. The second dominant group of macrofauna in the KAE was polychaetes, and they also contributed significantly to total macrofaunal biomass. However, many studies pointed out that, polychaetes are the most dominant benthic communities in the estuarine environment (Pocklington and Wells, 1992; Qasim, 2003; Venkataraman *et al.*, 2013), due to their high degree of tolerance to pollution and environmental perturbations (Sigovini *et al.*, 2013; Soniya and Sarala Devi, 2009). Polychaetes are also exhibiting diverse type feeding strategies thereby a varied number of polychaete species benefited from various sources food materials (Fauchald, 1977; Shields and Blanco-Perez, 2013). Such adaptations of polychaetes favour them to be an essential component of estuarine benthos. The mean macrofaunal biomass in the KAE was highly correlated with molluscan biomass (r = 0.932, $p < 0.01$), that principally dominated with filter-feeding bivalves. They play a significant role in sustaining water column transparency by continuous removal of suspended particles. They also exhibit an essential link between primary producers and consumers by forming a necessary intermediate in the flow of energy through the estuarine environment (Wang *et al.*, 2015).

The malacostracan crustaceans were the third dominant group in macrofaunal biomass next to molluscs and polychaetes. Malacostracans of the estuary was mostly represented by amphipods and isopods. They contributed significantly to communities in the mixing zone to the estuarine mouth region. However, their density depicted a decreasing trend towards the Riverhead. Among all species of malacostracans in the estuary, amphipod species *Americorophium triaeonyx* and isopod species *Cirolana fluviatilis* were contributed significantly to total macrofaunal density. These species were tolerant to a wide range of salinity fluctuation, and it could be the reason for their high dominance in the KAE (Shyamasundari, 1973), similarly, deposit feeding amphipod, *Corophium volutator* population has increased with increasing Chl-*a* content of water column (Ysebaert and Herman, 2002). Similarly, the relative abundance of malacostracans in the estuary has risen by well-oxygenated sandy sediment, and they avoided organic matter accumulated sediment. While other groups such as decapods, cumaceans, tanaids, stomatopods, mysids have represented in few numbers, a similar observation made in the southern part of Cochin backwater by Asha (2017). Because of their relatively small size, representation of malacostracans in macrofaunal biomass has shown relatively low biomass when compared to large polychaete and molluscan communities.

In the investigation, polychaete communities in the KAE depicted a wide spatial variation. Their population described a clear dominance towards the northern arm and other mud dominant stations where relatively low malacostracan density was noted. It could be due to the control of substrate type and different environmental variations or by intraspecific competition in macrofaunal communities. Macrofaunal polychaetes play a crucial role in the estuarine benthic food chain (Beesley *et al.*, 2000; Hutchings, 1998), by forming one of the most critical primary consumers in estuaries. They are the primary food source for different life stages of higher trophic levels (Ysebaert *et al.*, 1998). Among the density of all polychaetes collected in the KAE, members of tube-dwelling or tubicolous polychaetes under the subclass Sedentaria has dominated in all the stations and contributed about 13.80 percentage to total macrofaunal density. Most tubicolous polychaetes were either deposit feeders or filter feeding form capable of building tubes of unadorned mud, sand, and parchment or hardened calcium carbonate and often decorated with sand, shell, algae, and hydroids. They are often found in dense mats, and later it forms a refuge substrate for many other organisms (Day, 1967; O'Clair and O'Clair, 1998). Biomass of polychaetes in the KAE contributed reasonably similar to that of their density. Biomass of polychaetes was comparatively higher during the first year of observation, but the frequency was relatively high during the second year, and it could be due to the dominance of small-sized species in the second year. The benthic invertebrates like polychaetes and molluscs, they reproduce via complex planktonic stages, variation in the delivery of larvae to the suitable substratum, and habitats are fundamental determinant for recruitment rates and population structure (Roegner, 2000). The lack of suitable substratum and favourable environmental conditions in the estuary prevent settlement of larval stages of such species in high-energy zones in the estuary along the different salinity gradient. While polychaete species like *Prionospio cirrifera* and *Heteromastus filiformis* were dominant in organically enriched sediment that was attributed to its resistance and preference to organic matter, it forms as an essential source of food material to these sub-surface deposit feeders (Ajmal Khan et al., 2004; Herman et al., 1999). According to Diaz and Rosenberg (1995) polychaetes are the most tolerant group to low oxygen and increasing organic enrichment in sediment.

Mollucans formed one of the critical ecological group in the KAE, which constitute about 63 percentage to total macrofaunal biomass ($r = 0.932$, $p < 0.01$). Among them, bivalves form a principal component, with a dominance of three significant species such as *Arcuatula senhousia* (37.55 %), *Marcia recens* (27.72 %), *Villorita cyprinoides* (13.16 %). Among these three, *M. recens* and *V. cyprinoides* were the significant contributors to macrofaunal biomass. The contribution of *A. senhousia* to total biomass was relatively low due to their smaller size, but they exhibited high numerical density. In the entire Vembanad Lake, the black clam *V. cyprinoides* form a single dominant bivalve species that contributes significantly to ecology and economy (Sheeba, 2000). Spatial distribution of estuarine molluscan communities in the KAE was restricted to the middle zone of the estuary with intermediate salinity profile. Their density was decreased towards estuarine mouth and Riverhead. Estuary-dependent molluscs are reproduced primarily through larvae forms, and the slow-swimming veliger larva is the primary dispersal stage for most of these bivalves and gastropods. Therefore recruitment and population structure of these molluscs are also dependent on the larval retention time, circulation patterns, salinity, temperature, pressure, horizontal velocity and substrate type in the estuary (Roegner, 2000). The sporadic occurrence of other macrofaunal members in the estuary was represented by hydrozoans, cirripedians, insects, Pisces, ophiuroideans and nemerteans. Their representation of biomass was relatively small. The numerical density of hydrozoan colony of *Obelia bidentata* has dominated in almost all the stations except the riverine head region. The relatively high occurrence was observed towards northern arm with intermediate salinity range. They are generally distributed in brackish and marine environments, and they found attached to the hard substrate such as wood, shells, wrecks, sandy bottoms and rarely found in intertidal pools (MarLIN, 2017).

Due to the varying and unpredictable hydrological, morphological and chemical conditions in the estuary made natural stress to sessile or sedentary forms of benthic fauna. These natural stresses to benthic fauna are sometimes overwhelmed by intensified human activities like land reclamation, drainage of waste from domestic, industrial and agricultural operations, harbour and dredging (Feebarani *et al.*, 2016; Rehitha *et al.*, 2017; Ysebaert *et al.*, 1998). This sensitivity to natural and anthropogenic stress disturbs the species, the population as well as on community-level distribution pattern yet sediment characteristics forms the crucial factor determining the abundance and biomass of macrofauna. However, the ecological factors like variability in quality, quantity, and availability of food materials may also influence the faunal density, biomass and composition. In the present study, macrofaunal biomass was decreased with increasing organic matter, silt and clay content in sediment. Similarly, macrofauna plays a crucial role in the energy flow of the benthic ecosystem (Parulekar *et al.*, 1980) and the role of benthos in the fishery is well understood. Many studies found that macrofauna makes sizable contributions to energy flow in the estuarine ecosystem (McLusky and Elliott, 2004; Parulekar *et al.*, 1980). Among the macrofaunal groups, the estimated production to biomass ratio (P/B) was relatively high for malacostracans in the estuary followed by polychaetes and molluscs. However, macrobenthic molluscs exhibited a relatively high somatic production rate. The maximum macrobenthic secondary production was observed during post-monsoon. The macrofaunal communities in the KAE are estimated to contribute the production of about 29588 ± 51774 kg fish biomass during the study. The annual fish production in the estuary is about 908.6 t y^{-1} (Jayachandran *et al.*, 2013). It is evident that benthic production substantially contributes to total fish production in the estuary. The dominant finfish species noticed in the study were *Gerres erythrourus*, *Mugil cephalus*, *Lisa parsia*, *Lisa macrolepis*, *Valamugil speigleri*, *Plicofollis dussumieri*, *Etroplus suratensis*, *Etroplus maculatus*, *Ambassis ambassis*, *Eubleekeria splendens*, *Leiognathus berbis*, *Oreochromis mossambicus* and *Photopectoralis bindus* (Jayachandran *et al.*, 2013). Many of these species are depends on the benthic secondary production in the estuary. So it is evident in the present study that secondary benthic output contributes substantially to total fish biomass in the estuary.

Meiofauna, the most diversified element of the marine biota represented as many as 24 members of the 35 animal phyla, either showing a permanent life or just temporarily (Balsamo *et al.*, 2010) and classically used as indicators of energy transfer and overall health of aquatic system (Giere, 2008). In the present study, meiofauna density was characterised by nematodes, harpacticoid copepods, amphipods, foraminiferans, polychaete larvae with sporadic occurrence of turbellarians, tardigrades, gastrotrichs, crustacean nauplii and oligochaetes. The nematodes were the dominant community in all the stations and seasons in the estuary (El-Serehy et al., 2015; Zeppilli et al., 2015), in which deposit feeders and epi-growth feeders were abundant in the fine mud and sandy substratum of the estuaries (Ansari and Parulekar, 1993). John (2009) reported a total of 14 meiofaunal taxa in Cochin backwaters and was dominant with nematode population with polychaetes, copepods, and foraminifera. Some of these meiofaunal organisms are only possessed meiofaunal life as a part of their life cycle like larval stages of molluscs and polychaetes and temporary meiofauna (taxa that grow later into the macrofaunal size class). However, most of the nematodes and foraminifera are permanent meiofauna (taxa that complete their entire life cycle in meiofaunal size classes) (Desai and Krishnankutty, 1967). Hakenkamp and Morin (2000) stated that meiofaunal populations and composition is depending on permanent meiofauna further, Stead *et al.* (2005) observed that, temporary meiofauna contributed 51 percentage of the total secondary production. In the present study, permanent meiofaunal members dominated in all stations.

Meiofaunal distribution in the KAE depicted a similar trend that of macrofauna (Gopalan et al., 1987; Jayalakshmy and Kameswara Rao, 2004). In mesoscale, abiotic factors such as sediment texture, salinity and hydrodynamics were the most important factors controlling the meiofauna (Giere, 2008). Though, in micro-scale biological factors such as inter and intra-specific relationships, occurrences of biogenic structures, accessibility of food and reproductive strategies were shaping the meiofaunal community (Giere, 2008). Their abundance in the estuarine systems mainly depends on the source and availability of food; they aggressively consume diatoms, bacteria, protozoans, detritus, and dissolved organic carbon (Giere, 2008). A study by Schratzberger and Warwick (1998) compared meiofaunal communities from an organic-poor sandy estuary, and an organic-rich muddy estuary, a marked change in community structure observed based on the quantity of organic matter in the sediment. Similar to the present study, they found an increase of organic matter causes a reduction in diversity by declining abundances of dominant nematode species. Rao and Sarma (1990) also state that the reduced abundance of meiofauna was mainly influenced by the decrease in salinity, an absence of tidal inputs, other natural disturbance and resultant ending of meiofaunal recruitment in the neritic end of Gosthani estuary on the east coast of India. The presence of nematodes as a community is independent of the sediment structure (Vanaverbeke *et al.*, 2000), but, in general, nematodes said to be highly dominant in sand finer than 300 µm. While harpacticoid copepods become more relevant in sediments coarser than 350 µm (McLachlan and Brown, 2006). According to Ingole and Parulekar (1998), harpacticoid density was closely related to salinity fluctuations. Rao and Sarma (1994) pointed out that low salinity in the estuary results in a reduction of harpacticoid copepod density. Similarly, the copepod population in the KAE depicted maximum at high energy zone of estuarine mouth with relatively high sand content. According to Murray (1991), the benthic foraminiferal distributions may vary according to any combination of factors such as substrate type, light intensity, water temperature, food availability, oxygen, salinity, depth and current energy. Many species of foraminifera possess well-defined salinity and temperature making them particularly useful for predicting environmental conditions (Levin, 1992; Thrush et al., 1999). Similar to present observation, Asha *et al.* (2016) noticed the occasional appearance of meiofaunal foraminifera in the southern part of Cochin backwater with the highest density of 15826 ind.10 cm^{-2}.

Chapter 5
Community structure of macrobenthos

V.1. Introduction

Estuaries are being one of the highly productive aquatic systems serving as breeding and nursery grounds for a diverse array of organisms (Griffiths *et al.*, 2017; McLusky and Elliott, 2004). However, the complex hydrodynamic process in the estuaries leads to the establishment of an environmental gradient expressed by the gradual changes in salinity, variability of sediment composition and amount of organic matter (Medeiros *et al.*, 2016). This highly variable environment naturally stresses the ecosystem and guides the distribution of various species in estuarine gradient (Elliott and Quintino, 2007). In addition to natural stressors, estuaries are also subjected to a high degree of anthropogenic impact, which also exerts an influence on the species distribution in the estuarine environment (Borja *et al.*, 2000; Nybakken and Bertness, 2005; Remane, 1934).

Estuarine benthic biotope harbouring numerous species of different ecological communities (McLusky and Elliott, 2004). Their response to environmental variables can be monitor through understanding the changes in individual population or functional groups by using different univariate and multivariate analysis (Pearson and Rosenberg, 1978). Environmental stress, in broad terms considered to affect species abundance, richness and diversity. When an area undergoes a disruption or disturbance, a rapid re-colonisation of opportunistic species may occur, which leads to a spike in the abundance of small, opportunistic and rapidly growing species (Pearson and Rosenberg, 1978). These species typically have a low functionality to ecosystem processes and therefore have little ecosystem value (Thrush *et al.*, 2013). However, the loss of larger long-living infauna species can represent potential long-term degradation in benthic condition in the estuary (McLusky and Elliott, 2004; Pearson and Rosenberg, 1978; Thrush *et al.*, 2006).

Many macrofaunal communities developed through free-living, dispersive larval stages and thus the larval settlement is a crucial biological process influenced by a myriad of factors that structuring the biocoenosis of macrofauna (Jan, 1999). However, benthic macrofaunal communities presumed advantages of larval dispersal by avoiding competition for resources with adults, temporary reduction of benthic mortality, decreased inbreeding and increased the ability to withstand in local extinction. The sedentary soft-sediment macrofaunal species often have a period of mobility as juveniles, so settlement may not have as much influence on zonation. Therefore, understanding the processes that lead to estuarine benthic diversity is a complicated process and a challenge as they exposed to high natural biophysical variability (de Juan and Hewitt, 2014). According to Mäkelä *et al.* (2017), macrofaunal species richness decrease in an area where abundant food supports high benthic standing stock by the dominance of a single or few species. Whereas, in the food-limited condition the faunal abundance decreases with increase in taxonomic evenness. Many studies suggested that the communities with high species diversity and moderate abundances of each taxon are characteristic of communities with prolonged stability and minor or small-scale disturbances (Pearson and Rosenberg, 1978). So far, more studies that are recent have found that communities with highest species richness are a sign of intermediate disturbance through time (Connell, 1978; Nybakken and Bertness, 2005).

The distribution of various function feeding groups or guilds in a macrobenthic communities depicts the ecological status of that particular benthic biotope (Gallagher, 2008). Their distribution is related to food particle size and composition, food-intake mechanism and the motility patterns associated with feeding (Fauchald and Jumars, 1979; Gaston, 1987). According to Thomson (1982), high primary production combined with higher current speeds enables efficient filter feeding, which has been thought to be the contributing factor to the high bivalve biomass and density. However in low current condition, the biomass of filter feeding bivalves decreases in favour of facultative filter feeders and deposit feeders. These functional feeding groups of benthic communities also play a pivotal role in estuarine ecosystems, by acting as conduits for the carbon cycling. They enhance bioturbation and burrowing activities that result in the degradation or redistribution of organic matter (Fauchald and Jumars, 1979; Gaston, 1987; Schaffner *et al.*, 1992).

Estuarine macrofaunal assemblage pattern is a standard tool utilised in estuarine management, with predictive responses to different anthropogenic stressors catalogued by a multitude of studies over the years (Borja *et al.*, 2000). Tolerance values of different species can assess through a variety of techniques including the field-based observations, knowledge of life history, ecotoxicology experiments and best professional judgement (Borja *et al.*, 2000). Species that make the most useful indicators will have narrow and specific environmental tolerances, which would give a clear benchmark of ecological condition in the estuary (Pelletier *et al.*, 2010). This chapter discusses the bioecology of macrofauna concerning their species composition, community structure, distribution pattern and functional feeding groups in the Kodungallure-Azhikode estuary on spatial and temporal scales.

V. 2. Results

V. 2. 1. Diversity and species composition

V. 2. 1. 1. Univariate indices

Faunistic examination of macrofauna collected during 2009 to 2011 period at selected seven stations in the Kodungallur-Azhikode estuary (KAE) yielded a total of 79 species in 71 genera belonging to 49 families. Among the 79 species of macrofauna collected, polychaetes constituted the primary component with 33 species, among this subclass Errantia contributed the highest number of species (19 spp.), whereas subclass Sedentaria comprises 14 species. Class Malacostraca (Crustacea) formed the second dominant group with 26 species belonging to seven Order. They were Amphipoda (9 spp.), Decapoda (8 spp.), Isopoda (4 spp.), Tanaidacea (2 spp.), Stomatopoda (1 spp.), Cumacea (1spp.), and Mysida (1 sp.). The Class Bivalvia (9 spp.) and Gastropoda (2 spp.) in the Phylum Mollusca formed third position (11 spp.) in total macrofaunal species diversity. The sporadic representatives were pooled together as 'others group' and represented by different taxonomic Class such as Hydrozoa (1 sp.), Cirripedia (1 sp.), Insecta (1 sp.), Nemertea (1 sp.), Actinopterygii (4 spp.) and Ophiuroidea (1 sp.).

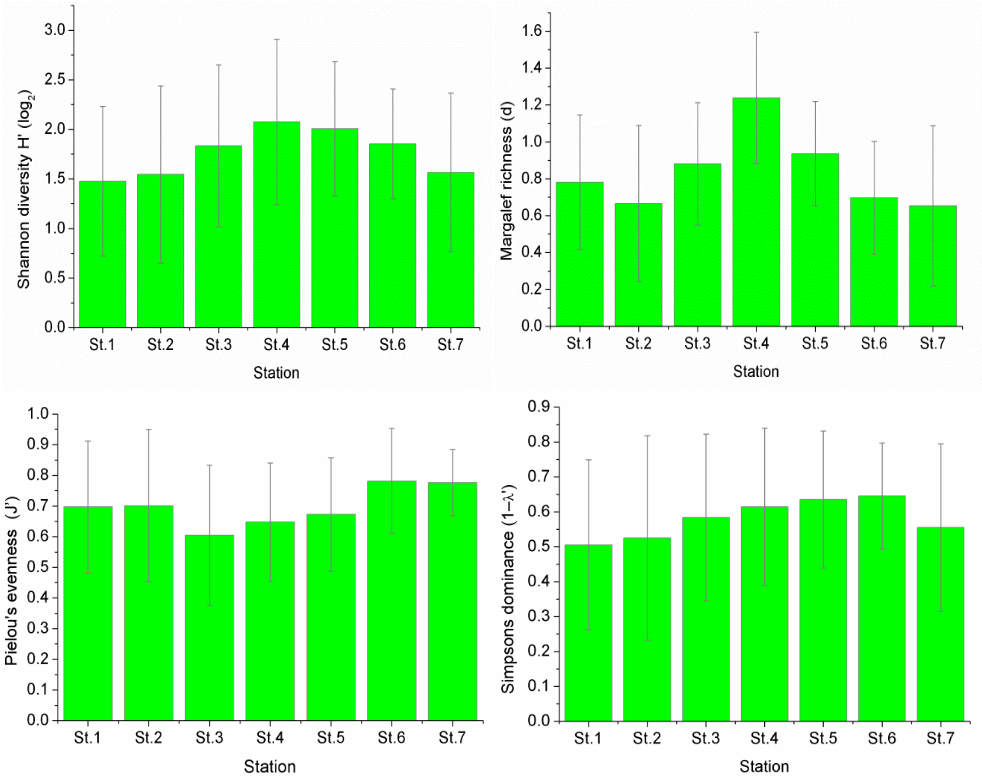


Figure 40 a-d. Mean macrofaunal species diversity (H' [log_2]), richness (d), evenness (J') and dominance (1-λ') for each station in the KAE during 2009-2011 period

The macrofaunal Margelef species richness (d) varied from 0.65 ± 0.43 in station seven to 1.24 ± 0.36 in station four with an overall mean of 0.84 ± 0.40 [Figure 40 a-d]. Seasonally, it varied from 0.78 ± 0.42 in the monsoon of second-year (2010-2011) to 0.86 ± 0.40 in post-monsoon of the first year (2009-2010). A relatively low Pielou's evenness index (J') was recorded during the study with mean value of 0.70 ± 0.20, and it varied from 0.60 ± 0.23 in station three to (0.78 ± 0.17) in station six. Seasonally, it varied from 0.61 ± 0.22 in post-monsoon to 0.80 ± 0.12 in pre-monsoon of the first year. The mean value of Shannon index [H'(log_2)] was highest during the first year (mean 1.93 ± 0.68) that compared to second year period (mean 1.60 ± 0.86). Seasonally lowest value was recorded during post-monsoon of the second year (mean 1.50 ± 0.89) and highest during pre-monsoon of the first year (mean 2.11 ± 0.60) with the overall mean of 1.77 ± 0.79. Spatial diversity varied from the lowest value in the station one (mean 1.50 ± 0.75) to highest in station four (mean 2.07 ± 0.84). The mean Simpson dominance (1-λ') index for macrofaunal communities in the study area was varied from 0.51 ± 0.24 in station 1 to 0.65 ± 0.15 in station six. Seasonally, it varied from 0.50 ± 0.27 during post-monsoon of the second year to 0.69 ± 0.13 during pre-monsoon of the first year. The macrofaunal species number (S), richness (d), diversity (H' [log_2]), evenness (J'), and dominance (1-λ') at each station and season during the two-year study is given in Tables 7 a-c.

Table 7 a. Diversity indices (±SD) of macrofauna during 2009-2010 period

Station	**d**	**J'**	**H'(log_2)**	**1-λ'**
Station 1	0.82 ± 0.30	0.66 ± 0.15	1.73 ± 0.31	0.58 ± 0.11
Station 2	0.79 ± 0.42	0.73 ± 0.27	1.89 ± 0.74	0.64 ± 0.23
Station 3	0.81 ± 0.32	0.65 ± 0.24	1.87 ± 0.84	0.61 ± 0.24
Station 4	1.2 ± 0.38	0.67 ± 0.21	2.2 ± 0.77	0.65 ± 0.20
Station 5	0.93 ± 0.28	0.71 ± 0.20	2.07 ± 0.74	0.65 ± 0.21
Station 6	0.73 ± 0.34	0.75 ± 0.20	1.87 ± 0.63	0.64 ± 0.18
Station 7	0.82 ± 0.45	0.75 ± 0.11	1.85 ± 0.64	0.63 ± 0.13
Mean	0.87 ± 0.38	0.7 ± 0.2	1.93 ± 0.68	0.63 ± 0.18

Table 7 b. Diversity indices (±SD) of macrofauna during 2010-2011period

Station	**d**	**J'**	**H'(log_2)**	**1-λ'**
Station 1	0.73 ± 0.45	0.75 ± 0.28	1.22 ± 0.98	0.43 ± 0.32
Station 2	0.54 ± 0.41	0.66 ± 0.22	1.2 ± 0.93	0.41 ± 0.31
Station 3	0.95 ± 0.34	0.56 ± 0.22	1.8 ± 0.83	0.56 ± 0.24
Station 4	1.28 ± 0.34	0.62 ± 0.18	1.94 ± 0.91	0.58 ± 0.25
Station 5	0.94 ± 0.30	0.64 ± 0.17	1.94 ± 0.64	0.62 ± 0.19
Station 6	0.66 ± 0.28	0.81 ± 0.14	1.84 ± 0.49	0.65 ± 0.13
Station 7	0.47 ± 0.34	0.81 ± 0.1	1.28 ± 0.87	0.48 ± 0.30
Mean	0.8 ± 0.43	0.69 ± 0.21	1.6 ± 0.86	0.53 ± 0.26

Table 7 c. Diversity indices (±SD) of macrofauna during different seasons

Station	**d**	**J'**	**H'(log_2)**	**1-λ'**
Mon.09-10	0.91 ± 0.37	0.72 ± 0.16	2.01 ± 0.57	0.66 ± 0.15
Pos.09-10	0.86 ± 0.40	0.61 ± 0.22	1.67 ± 0.79	0.54 ± 0.23
Pre.09-10	0.85 ± 0.37	0.80 ± 0.12	2.11 ± 0.60	0.69 ± 0.13
Mon.10-11	0.78 ± 0.43	0.72 ± 0.23	1.55 ± 0.93	0.52 ± 0.30
Pos.10-11	0.81 ± 0.44	0.64 ± 0.21	1.50 ± 0.89	0.50 ± 0.27
Pre.10-11	0.80± 0.44	0.72 ± 0.19	1.76 ± 0.75	0.58 ± 0.22
Mean	0.84 ± 0.40	0.70 ± 0.20	1.77 ± 0.79	0.58 ± 0.23

Table 8 a. Spatial mean density of polychaete species (ind.m^{-2}) in KAE during 2009- 2011 period

Species	St.1	St. 2	St. 3	St.4	St.5	St.6	St.7
Lumbrineris latreilli	24	0	0	0	0	2	0
Lumbrineris simplex	10	9	3	7	0	1	2
Ninoe notocirrata	23	24	9	18	51	79	31
Lumbrineris heteropoda	0	0	2	2	5	0	3
Diopatra neapolitana	1	2	21	13	17	0	3
Namalycastis indica	19	37	29	25	0	23	5
Dendronereis arborifera	17	0	0	9	0	27	0
Dendronereides heteropoda	2	0	0	8	2	5	0
Dendronereis aestuarina	38	103	38	32	123	51	48
Perinereis cavifrons	0	4	18	6	0	2	0
Neanthes glandicincta	0	15	8	0	10	43	0
Ceratonereis costae	0	0	0	12	432	0	0
Nephtys oligobranchia	0	0	0	0	0	2	0
Nephtys polybranchia	0	0	0	0	13	0	0
Phyllodoce sp.	0	0	0	0	9	0	0
Glycinde bonhourei	3	0	5	6	9	0	0
Glycera alba	0	0	0	1	2	5	0
Glycera tridactyla	9	34	1	30	28	1	12
Sigambra constricta	0	0	0	1	0	0	0
Owenia fusiformis	7	0	0	3	7	0	0
Ficopomatus sp.	0	0	3	0	0	0	0
Paraprionospio sp.	0	0	0	0	20	0	0
Prionospio cirrifera	47	238	43	93	466	284	135
Prionospio polybranchiata	0	65	0	3	6	0	0
Pseudopolydora kempi	0	0	0	0	33	8	0
Pista indica	3	9	130	36	0	0	1
Ophelia capensis	2	0	0	0	0	2	0
Cossura sp.	3	74	27	3	45	7	0
Capitella sp.	58	113	106	68	317	214	108
Heteromastus similis	0	0	5	0	0	1	9
Heteromastides bifidus	0	0	0	0	0	1	2
Parheteromastus tenuis	0	0	6	0	19	15	1
Notomastus sp.	61	113	25	65	350	49	161

a) Polychaete community

The polychaete families in the estuary have best been represented regarding the number of species, and they were Nereididae (6 spp.), Capitellidae (6 spp.), Lumbrineridae (4 spp.), Spionidae (4 pp.), Nephtyidae (2 spp.), Glyceridae (2 spp.), Onuphidae (1sp.), Syllidae (1 sp.), Phyllodocidae (1 sp.), Goniadidae (1 sp.), Pilargidae (1 sp.), Oweniidae (1 sp.), Serpulidae (1 sp.), Terebellidae (1s p.) and Opheliidae (1 sp.). The numerically dominant polychaete species in the study area were *Prionospio cirrifera* (24.22 % of total polychaetes), *Capitella* sp. (18.29 %), *Notomastus* sp. (15.26 %), *Ceratonereis* (*Composetia*) *costae* (8.27 %), *Dendronereis aestuarina* (8.04 %), *Ninoe notocirrata* (4.34 %), *Pista indica* (3.33 %), *Cossura* sp. (2.93 %), *Glycera tridactyla* (2.17 %), *Namalycastis indica* (2.56 %), *Diopatra neapolitana* (1.06 %) and *Neanthes glandicincta* (1.41 %). The data on species composition and abundance of polychaetes collected during the study were processed to identify the number of species present in each station. The study found that the relatively low species count was recorded at station seven (14 spp.) while moderately high species counts have been obtained at station four (21 spp.), station five (21 spp.) and station six (21 spp.) during the study. Annually, highest species count was recorded in station four (16 spp.) and lowest at station two (10 spp.) during the first year period (2009-2010). During the second year (2010-2011), station six obtained highest species count (17 spp.) and lowest at station seven (6 spp.) [Table 8 a].

During the study, the Margelef species richness (d) of polychaete varied from 0.19 ± 0.21 in station three to 0.42 ± 0.23 in station four with the overall mean of 0.29 ± 0.24. Seasonally, it varied from 0.21 ± 0.19 during the monsoon of the second year to 0.40 ± 0.24 during pre-monsoon of the second year. The relatively low Pielou's evenness index (J') was recorded and the mean value was 0.60 ± 0.38, and it varied from 0.51 ± 0.41 in station seven to 0.75 ± 0.36 in station six. Seasonally, it varied from 0.53 ± 0.38 during the monsoon of the first year to 0.72 ± 0.29 during pre-monsoon of the second year. However, Shannon diversity [H' ($\log_2$)] was comparatively similar during the first year (mean 1.02 ± 0.78) and the second year (mean 1.03 ± 0.72). Seasonally, lowest value was recorded during monsoon period of the second year (mean 0.84 ± 0.72) and highest during pre-monsoon of the second year (mean 1.25 ± 0.64) with the overall mean of 1.02 ± 0.75. Spatial diversity varied from station one (mean 0.73 ± 0.57) to station five (mean 1.41 ± 0.70). The mean Simpson dominance (1-λ') index for polychaetes ranged from 0.28 ± 0.29 in station three to 0.52 ± 0.25 in station five. Seasonally, it varied from 0.35 ± 0.28 during the monsoon to 0.49 ± 0.23 in the pre-monsoon of the same year.

b) Malacostracan community

The malacostracan crustacean families such as Penaeidae, Corophiidae, Eriopisidae and Apseudidae were contributed the highest number of species (2 spp.). While, other families such as Hymenosomatidae, Leucosiidae, Portunidae, Alpheidae, Diogenidae, Luciferidae, Maeridae, Melitidae, Ischyroceridae, Gammaridae, Aoridae, Bodotriidae, Anthuridae, Idoteidae, Sphaeromatidae, Cirolanidae, Squillidae and Mysidae were represented by single species. The numerically dominant malacostracan crustacean species in the study area were *Americorophium triaeonyx* (85.31 % of total malacostracan), *Cirolana fluviatilis* (7.54 %), *Grandidierella taihuensis* (2.69 %), *Victoriopisa chilkensis* (1.41 %), *Cyathura indica* (1.39 %) *Ctenapseudes chilkensis* (0.26 %), *Corophium volutator* (0.21 %), *Neorhynchoplax alcocki* (0.21 %), *Melita zeylanica* (0.16 %), *Philyra malefactrix* (0.11 %), *Metapenaeus affinis* (0.11 %), *Penaeus indicus* (0.10 %) etc. The data on species composition and abundance of malacostracan collected during the study were processed to identify the number of species present in each station. The study found that there is no variability in species count recorded at the different stations of the study area. The station two, five and seven were recorded with 14 species while station one, three, four and six recorded with 15 species. During the first year of study (2009-2010), station five (9 spp.) showed relatively low species count and station four showed highest (14 spp.) and similarly, during the second year (2010-2011), that was lowest at station 7 (6 spp.) and highest at station four (13 spp.) [Table 8 b]. During the study, the Margelef species richness (d) of malacostracan crustaceans varied from 0.91 ± 0.24 in station 3 to 0.14 ± 0.22 in station seven with the overall mean of 0.22 ± 0.23. Seasonally, values ranged from 0.12 ±

0.17 during pre-monsoon of the second year to 0.33 ± 0.44 during post-monsoon of the first year. A relatively low Pielou's evenness index (J') recorded during the study and the mean value recorded was 0.45 ± 0.42, and it varied from 0.34 ± 0.45 in station seven to 0.59 ± 0.34 in station four.

Table 8 b. Spatial mean density of malacostracan crustacean species (ind.m^{-2}) in KAE during 2009- 2011 period

Species	St.1	St. 2	St. 3	St.4	St.5	St.6	St.7
Metapenaeus affinis	0	1	2	7	3	3	3
Penaeus indicus	1	0	1	0	1	0	14
Neorhynchoplax alcocki	1	2	12	9	2	4	5
Philyra malefactrix	0	0	10	3	0	4	1
Scylla serrata	2	0	1	8	0	0	0
Alpheus malabaricus	0	1	2	0	0	1	5
Diogenes alias	3	0	0	0	0	0	0
Lucifer hanseni	0	0	0	0	2	0	0
Quadrivisio bengalensis	0	0	0	2	0	0	0
Melita zeylanica	36	1	0	5	5	0	0
Americorophium triaeonyx	6594	720	7428	2599	27	24	28
Corophium volutator	23	0	6	0	3	2	1
Victoriopisa chilkensis	86	32	35	37	14	23	2
Jassa falcata	0	0	0	0	0	1	0
Psammogammarus	0	8	0	0	0	0	0
Gammarus tigrinus	0	0	12	1	0	3	0
Grandidierella megnae	132	17	132	154	27	7	5
Iphinoe sp.	3	5	0	3	0	0	1
Cyathura indica	46	4	87	74	11	3	1
Synidotea variegata	0	1	1	4	0	2	0
Sphaeroma annandalei	1	0	0	0	0	0	0
Cirolana fluviatilis	214	16	779	304	2	28	12
Ctenapseudes chilkensis	13	14	4	2	1	5	4
Pagurapseudopsis gymnophobia	0	2	0	0	0	0	0
Miyakella nepa	0	0	0	0	0	0	1
Gastrosaccus dunckeri	3	0	0	0	1	1	0

Seasonally, it varied from 0.35 ± 0.44 during pre-monsoon of the second year to 0.57 ± 0.38 during post-monsoon of the first year. However, the mean Shannon diversity was highest during the second year period (mean 0.82 ± 0.73) and that for the first year was 0.58 ± 0.65. Seasonally the lowest values were recorded during pre-monsoon period of the second year (mean 0.43 ±

0.0.57) and highest during post-monsoon of the first year (mean one ± 0.73) with an overall mean of 0.70 ± 0.70. Spatial diversity varied from station seven (mean 0.47 ± 0.66) to station four (mean 1.23 ± 0.64). The mean Simpson dominance (1-λ) index for polychaetes in the study area was varied from 0.20 ± 0.27 in station seven to 0.40 ± 0.24 in station four. Seasonally it varied from 0.19 ± 0.24 during pre-monsoon of the second year to 0.39 ± 0.27 during post-monsoon in the first year.

c) Molluscan community

The molluscan families were best represented in the study area regarding the number of species belonging to Veneridae (3 spp.) and Ostreidae (spp. 2). Others families were represented by a single species such as Mytilidae, Psammobiidae, Cyrenidae and Cyrenidae of the Class Bivalvia, and Muricidae and Nassariidae of the Class Gastropoda. The family represented by single species with high abundance was Mytilidae. Muricidae was rarely observed in the estuarine mouth region. The numerically dominant molluscan species in the study area were *Arcuatula senhousia* (37.55 %), *Marcia recens* (27.72 %), *Villorita cyprinoides* (13.16 %), *Dosinia* sp. (2 %), *Nassodonta insignis* (7.42 %), *Meretrix casta* (7.31 %), *Crassostrea bilineata* (2.26 %), *Murex trapa* (1.87 %), *Hiatula* sp. (1.30 %), *Cuneocorbula cochinensis* (1.26 %) and *Saccostrea cucullata* (0.04 %). The juveniles oysters, *Crassostrea bilineata* and *Saccostrea cucullata* attached to benthic hard substratums with a size range of <1cm were only considered for the study. The molluscan species such as *Nassodonta insignis* and *Murex trapa* were the only representation from class Gastropoda and rest of the species were from class Bivalvia. The data on species composition and abundance of molluscans collected during the study were processed to identify the number of species present in each station. The species count observed between different stations in the estuary varied from 6 species in station six to 11 species in the station four and seven. In the first year period (2009-2010) species count varied from 5 species at station six and highest number of 10 species at station four and seven. In the second year (2010-2011) period, it ranged from 1 species at station six to 10 species at station four [Table 8 c].

Table 8 c. Spatial mean density of molluscan species (ind.m^{-2}) in KAE during 2009- 2011 period

Species	St.1	St. 2	St. 3	St.4	St.5	St.6	St.7
Villorita cyprinoides	2	0	22	265	63	1	9
Marcia recens	47	19	477	106	46	0	1
Dosinia sp.	6	4	11	17	7	3	7
Meretrix casta	33	0	72	65	23	2	2
Hiatula sp.	3	2	3	1	21	3	5
Cuneocorbula cochinensis	0	9	0	6	1	2	15
Arcuatula senhousia	1	143	458	389	0	0	3
Crassostrea bilineata	5	4	19	10	14	2	7
Saccostrea cucullata	0	0	0	1	0	0	0
Nassodonta insignis	11	13	104	35	18	0	16
Murex trapa	1	0	0	0	0	0	0

During the study, the Margelef species richness (d) of molluscs ranged from 0.04 ± 0.11 in station six to 0.43 ± 0.26 in station four with an overall mean of 0.19 ± 0.23. Seasonally, it varied from 0.15 ± 0.21 in the post-monsoon in the second year to 0.27 ± 0.25 in the monsoon of the first year. A mean value for Pielou's evenness index (J') was 0.19 ± 0.23, and it varied from 0.12 ± 0.33 in station six to 0.56 ± 0.42 in station five. Seasonally, it varied from 0.27 ± 0.36 during post-monsoon the second year to 0.54 ± 0.43 during monsoon the first year. However, Shannon index [H' (log_2)] was highest during the first year period (mean 0.69 ± 0.72) compared to the second year period of study (mean 0.51 ± 0.0.62). Seasonally, lowest values recorded during post-monsoon periods of the second year (mean 0.42 ± 0.58) and highest during monsoon the first year (mean 0.87 ± 0.74) with an overall mean of 0.60 ± 0.67. Spatial diversity varied from low value at station six (mean 0.15 ± 0.40) to highest value at station seven (mean 1.13 ± 0.74). The mean Simpson dominance (1-λ') index for molluscs in the study area varied from 0.07 ± 0.19 in station 6 to the highest value of 0.41 ± 0.27 in station four. Seasonally it ranged from 0.17 ± 0.23 during post-monsoon of the second year to a maximum of 0.35 ± 0.29 in the monsoon of the first year.

d) Other communities

Among other species identified, families represented from Gobiidae fishes of class Actinopterygii (4 spp.), Campanulariidae of class Hydrozoa (1 sp.), Balanidae of Infraclass Cirripedia (1 sp.), Chironomidae of class Insecta (1 sp.) and Ophiotrichidae of class Ophiuroidea (1 sp.). Small benthic fishes under the family Gobiidae was represented by *Callogobius mannarensis* from the estuarine mouth (station 1), *Trypauchen vagina* from stations two, three and four, *Psammogobius biocellatus* from stations four, five and six and *Acentrogobius viridipunctatus* from riverine station six. The other species recorded in the study area were species of *Obelia bidentata* (Hydrozoa), *Amphibalanus improvises* (Cirripedia), *Ophiothrix* sp. (Ophiuroidea) and insect (*Chironomidae*) [Table 8 d].

Table 8 d. Spatial mean density of 'other group' of species (ind.m^{-2}) during 2009- 2011 period in KAE

Species	St.1	St. 2	St. 3	St.4	St.5	St.6	St.7
Trypauchen vagina	1	23	2	22	1	1	2
Psammogobius biocellatus	0	0	0	1	76	1	0
Acentrogobius viridipunctatus	0	0	0	0	0	1	0
Callogobius mannarensis	1	0	0	0	0	0	0
Ophiothrix sp.	2	0	0	0	0	0	0
Obelia bidentata	143	38	168	180	1064	95	0
Balanus improvisus	1	0	1	14	38	0	9
Chironomus larvae	0	0	2	0	0	0	2
Nemertea	0	7	0	1	1	0	2

V. 2. 1. 2. Graphical Methods

a) Species-Area Plot and Species Estimator

The species accumulation plot for the macrofaunal grab samples of Kodungallur-Azhikode estuary was prepared using PRIMER *v*6. It can be used to determine whether the species collected during the survey was adequately described the actual species composition of the study area or not. The plot approached the upper asymptote, indicating that the study area was sampled sufficiently [Figure 41]. During the end of first year period of monthly sampling, 68 species was obtained, and during the second year, 11 species was added, it indicates that required sample size attained with the second year survey.

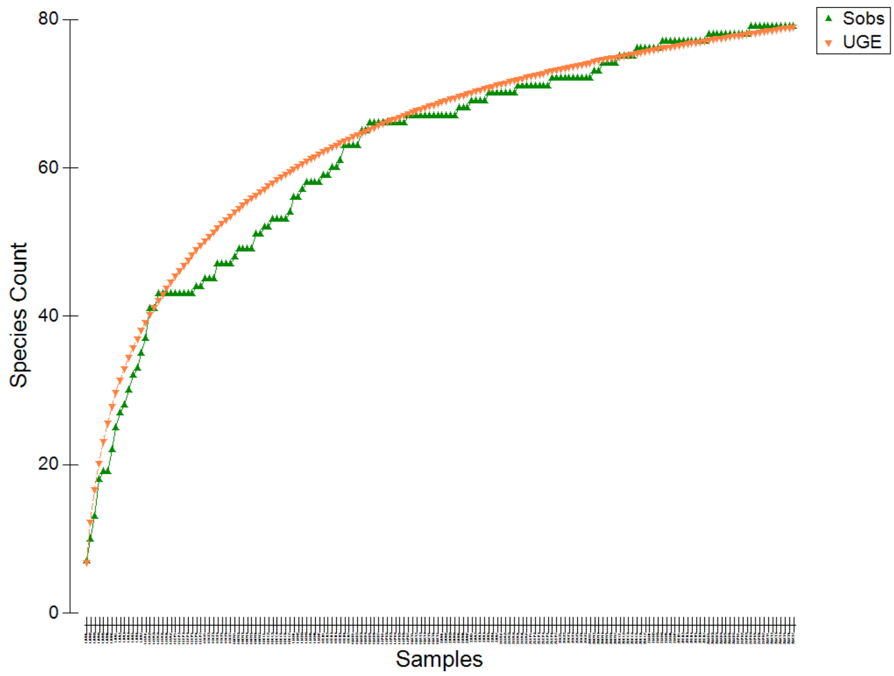


Figure 41. Species-area plot for macrofaunal species in KAE

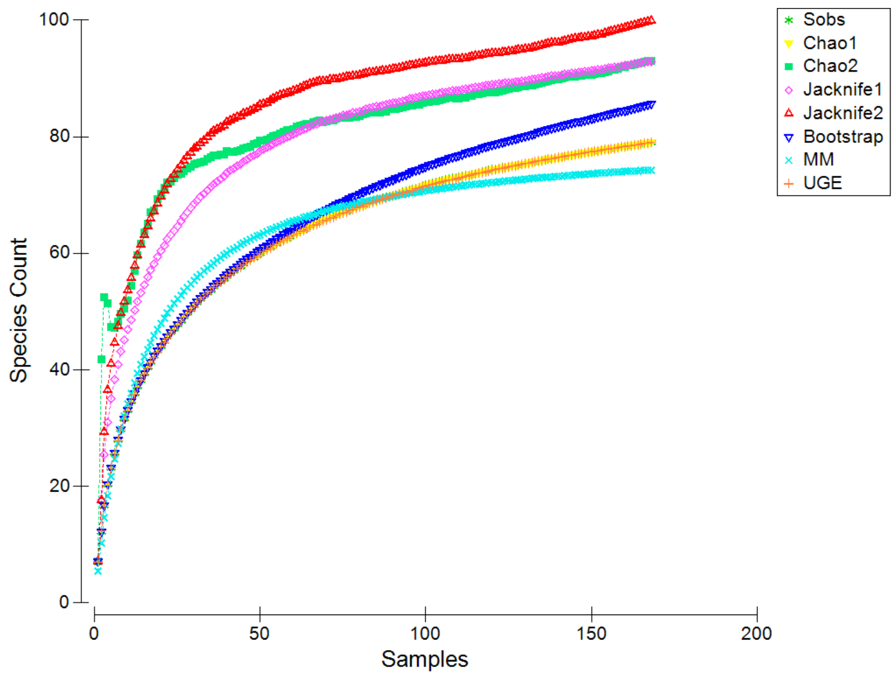


Figure 42. Species estimators for macrofaunal species in KAE

Species estimators were used to predict the actual number of species that would observe as the number of samples tends to be infinity. The total number of species estimated by the species estimators varied from 74 to 100 species [Figure 42]. While the minimum estimate given by MM (74 spp.), the maximum rating was given by Jacknife2 (99 spp.). The number of macrofaunal species estimated by MM (Michaelis-Menten) was 74, and that was 79 for Sobs (Observed number of species), UGE, Chao1, whereas 79 in Bootstrap, 86 in Jacknife1, 93 in Chao2 and 100 in Jacknife2.

b) k-dominance curve

In the present study, k-dominance plots were constructed for the annual, seasonal, and spatial pattern of macrofauna using statistical software package PRIMER *v*6.1.9.

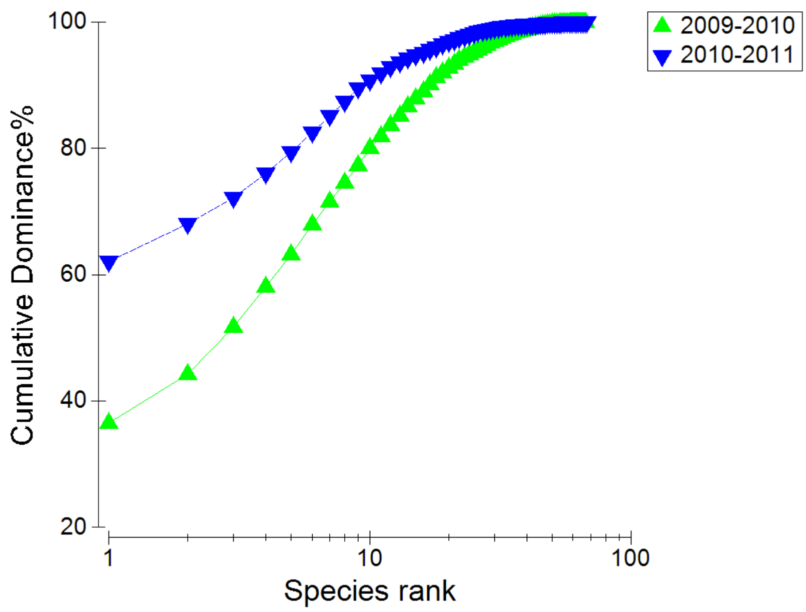


Figure 43. Annual variation of k-dominance curve of macrofauna in KAE

In this graphical plot, species are ranked in order of importance along the horizontal axis while the cumulative contribution of each of these to the total macrofaunal density is plotted along the vertical axis [Figures 43]. The k-dominance curve measures the intrinsic diversity, and in this plot, the lower lines represent samples with higher diversity. During the entire study, second year (2010-2011) had the highest species dominance in macrofaunal abundance. A single species of opportunistic amphipod, *Americorophium triaeonyx* contributed more than 62.05 percent to total numerical abundance. The other species such as hydrozoan *Obelia bidentata* (5.95 %), bivalve *Arcuatula senhousia* (4.21 %), isopod *Cirolana fluviatilis* (3.77 %), polychaete *Prionospio cirrifera* (3.45), *Capitella* sp. (3.05 %), *Ceratonereis costae* (2.61 %), *Notomastus* sp. (2.30 %), bivalve *Marcia recens* (2.07 %), and *Villorita cyprinoides* (1.28 %) were also significantly contributed to total abundance of macrofauna during the second year. However during the first year (2009-2010), more than 36.44 % was contributed by a amphipod species *A. triaeonyx* followed by polychaete *P. cirrifera* (7.83 %), *O. bidentata* (7.41 %), *C. fluviatilis* (6.44 %), *Capitella* sp. (5.08 %), *Notomastus* sp. (4.70 %), *M. recens* (3.58 %), *Arcuatula senhousia* (3.03 %), amphipod *Grandidierella megnae* (2.82 %) and polychaete *Dendronereis aestuarina* (2.64 %).

The k-dominance curve of species abundance data pooled for each season in the entire period of study is presented in Figure 44. Seasonal analysis depicted relatively high dominance of macrofauna during the monsoon of the second year followed by post-monsoon in the second year and pre-monsoon of the first year. Seasonal macrofaunal abundance showed high dominance during the monsoon of the second year (2010-2011) by *A. triaeonyx* contributed more than 80.61 percent followed by *C. fluviatilis* (2.30 %), *Capitella* sp. (2.12 %), *O. bidentata* (1.77 %), *Pista indica* (1.40 %), *P. cirrifera* (1.32 %), *D. aestuarina* (1.21 %), *M. recens* (1.06 %), *Victoriopisa chilkensis* (1 %) were dominant. During the post-monsoon of the second year (2010-2011) more than 64.17 % of macrobenthic diversity was dominated with *A. triaeonyx* followed by *Capitella* sp. (6.61 %), *C. fluviatilis* (5.43 %), *M. recens* (4.99), *D. aestuarina* (3.83 %), *G. megnae* (3.61 %), *Nassodonta insignis* (2.10 %), *P. cirrifera* (1.86 %), *Psammogobius biocellatus* (1.13). However, during post-monsoon (2009-2010), *A. triaeonyx* contributed 51.25 percent to total macrofaunal abundance followed by *P. cirrifera* (12.68 %), *Notomastus* sp. (4.16 %), *C. fluviatilis* (3.81 %), *A. senhousia* (3.61 %), *Capitella* sp.(2.94 %), *M. recens* (2.74 %), *G. megnae* (2.50 %)

and *Cyathura indica* (2.26 %). The other seasons exhibited relatively high macrofaunal diversity such as value of 4.42 during pre-monsoon of the first year, 3.76 for the monsoon of the first year and 3.08 during the pre-monsoon of the second year.

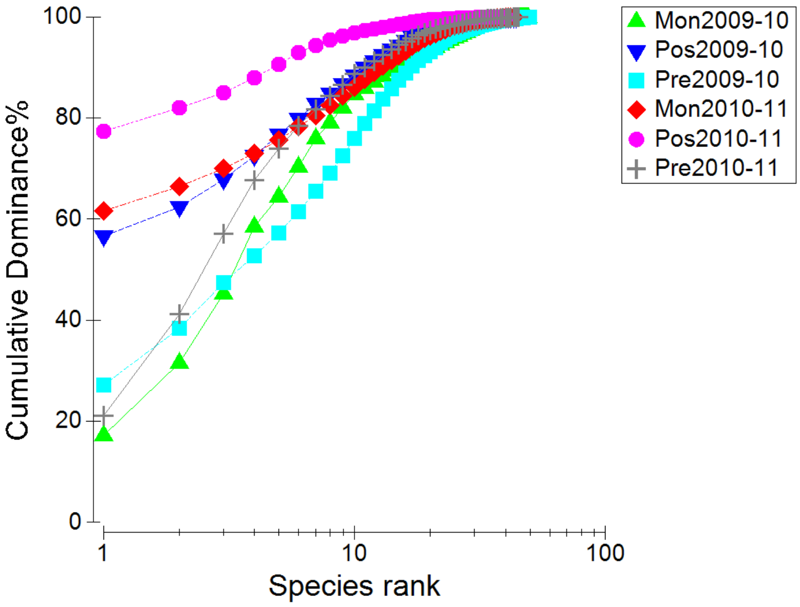


Figure 44. Seasonal variation of k-dominance curve of macrofauna in KAE

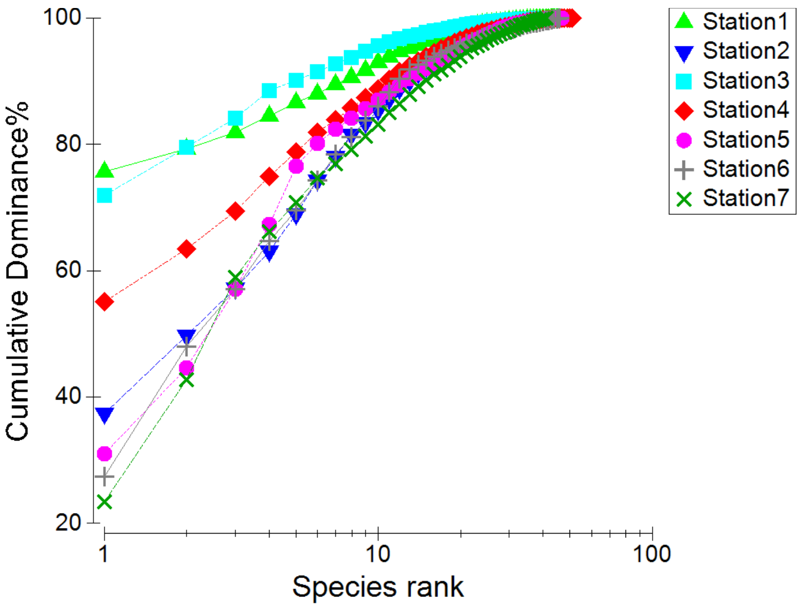


Figure 45. Spatial variation of k-dominance curve of macrofauna in KAE

The k-dominance curve of species abundance data pooled for each station is presented in Figure 45. The station-wise analysis demonstrated that relatively high dominance with a low diversity of macrofauna at stations one, three, and four. It inferred that, these stations would have a weak species diversity index. However, other stations such as station two (H'=3.48), station fiive (H'=3.49), station six (H'=3.60) and station seven (H'=3.70) showed comparatively high diversity. The dominance of opportunistic amphipod, *A. triaeonyx* in the stations one contributed 75.71 percent to total abundance of benthic communities in KAE during the entire period of study; similarly, the high dominance of this amphipod was observed in station three (71.93 %), and station four (55.09 %). At station one, the dominant species observed were *A. triaeonyx* (75.71 %), *O. bidentata* (3.51 %), *C. fluviatilis* (2.65 %), *G. megnae* (2.60 %), *V. chilkensis* (2.11 %), *Notomastus* sp. (1.49 %), *Capitella* sp. (1.42 %), *M. recens* (1.16 %), *P. cirrifera* (1.16 %) and *C. indica* (1.14 %). Similarly, station 3 was dominated with *A. triaeonyx* (71.93 %), *C. fluviatilis* (7.56 %), *M. recens* (4.62 %), *A. senhousia* (4.44 %), *O. bidentata* (1.62 %), *G. megnae* (1.27 %), *P. indica* (1.26 %), *Capitella* sp. (1.03 %), *N. insignis* (1 %), *C. indica* (0.84 %) and Station four dominated with *A. triaeonyx* (55.09 %), *A. senhousia* (8.33 %), *C. fluviatilis* (6.06 %), *V. cyprinoides* (5.41 %), *O. bidentata* (3.88 %), *G. megnae* (3.10 %), *P. cirrifera* (2 %), *M. recens (*1.90 %), *C. indica* (1.59 %), and *Capitella* sp. (1.43 %).

V. 2. 1. 3. Multivariate analyses of macrofauna

a) Family level

In the present study, macrofaunal species abundance from each stations were aggregated to the family level, and cluster analysis, non-metric multidimensional scaling (MDS) and analysis of similarity (ANOSIM) were carried out using the PRIMER *v*6.1.9 package. Bray-Curtis similarity was applied to construct a similarity matrix on square root transformed data. For classify the spatial assemblage pattern of macrofaunal families, the similarity matrix was subject to hierarchical agglomerative classification, employing group-average linkage. Figure 46 (a-b) shows a clear separation of stations into two groups, at 45.91 percent similarity level ($p < 0.5$).

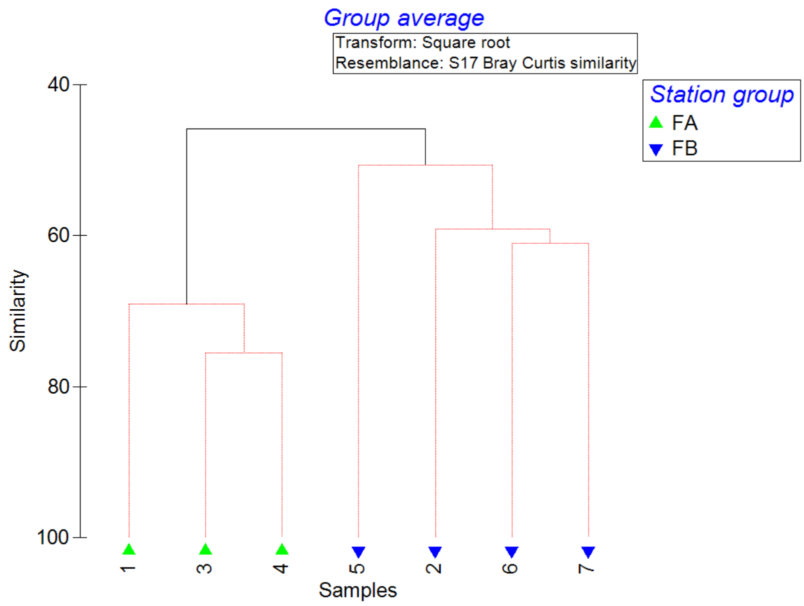


Figure 46 a. Dendrogram for macrofaunal families in each station of study area (FA group: station 1, 3, & 4; FB group: station 5, 2, 6, & 7)

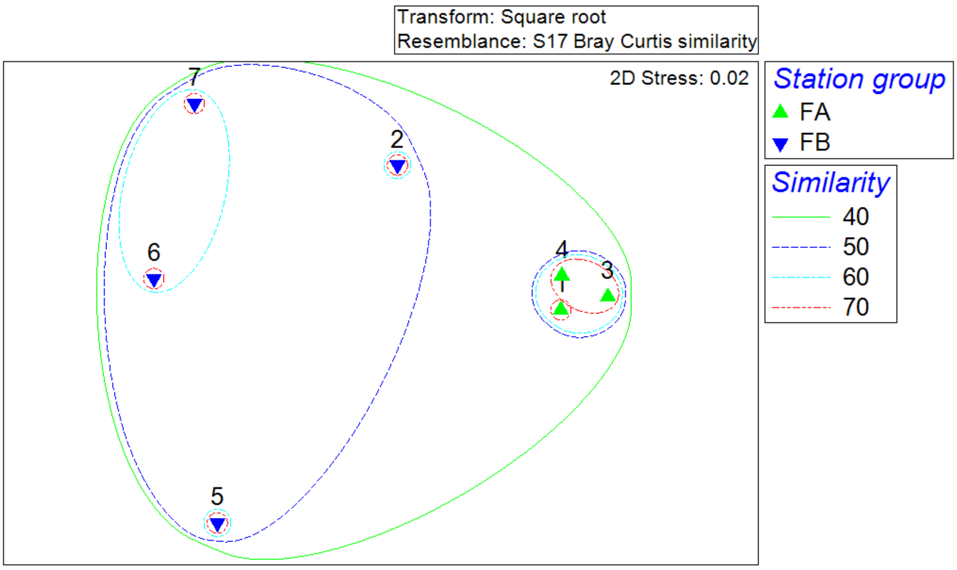


Figure 46 b. nMDS for macrofaunal families in each station of KAE (FA group: station 1, 3, & 4; FB group: station 5, 2, 6, &7)

The evaluation of R-values revealed that a significant differences in family wise distribution of macrofauna between stations (ANOSIM, R = 0.272, *p* = 0.1 %); whereas the faunal differences at family level from station two and six (ANOSIM, R = 0.07, *p* = 1.1 %), station two and seven (ANOSIM, R=0.03,

p = 11.9 %), station five and seven (ANOSIM, R = 0.108, p = 0.3 %). At this similarity level, two main spatial groups (FA & FB) were formed (Figure 45 a & b). Stations grouped under FA were stations 1, 3 and 4 with a similarity of 75.59 % ($p < 0.5$) and another group under FB were station 2, 5, 6 and 7 with a similarity of 50.69 percent ($p < 0.5$). Cluster FA and FB depicted significant variation. One-way ANOSIM (identified through SIMPROF, $p < 0.5$) revealed that highly significant differences exist among infaunal assemblages at the family level between these two groups (ANOSIM, R = 0.304, p = 0.1 %).

The stations grouped under FA were mainly from mixing zone of the estuary with high sandy substrates, the only exception was station 1, which is estuarine mouth region with the sandy substrate, and this station separated into another tail by forming a cluster (69.12 % similarity). In the FA group, forty-nine families of macrofauna were observed, and their abundance in these stations was contributed by a single family of malacostracan, Corophiidae (73.79 %). SIMPER (similarity percentage) analysis was also conducted in the macrofaunal data of KAE to identify the classes and families responsible for the defined clustering pattern. Three classes of macrofauna accounted for almost 19.20 percent of the average similarity within group FA with 91.41 percent abundance of macrofauna, which includes Bivalvia (8.46 %), Malacostraca (6.35 %) and Polychaeta (3.28 %). The family-wise analysis depicted that 10.72 percent of average similarity within eleven families of macrofauna contributed 90.50 percent cumulative abundance. The overall similarity of stations in FA was mainly contributed by dominant families such as Corophiidae (3.01 %), Cirolanidae (2.39 %), Mytilidae (1.16 %), Veneridae (1.13 %), Aoridae (0.59 %), Corbiculidae (91.54 %), Glyceridae (0.22 %), Campanulariidae (0.22 %), Terebellidae (0.21 %), Syllidae 0.20 %), Capitellidae 0.15 %).

In the group FB, forty-six families of macrofauna were observed and it was characterised by the dominance of Campanulariidae (31.05 %), Corophiidae (20.63 %), Spionidae (8.10 %), Capitellidae (6.89 %), Lumbrineridae (4.79 %), Mytilidae (3.78 %), Nereididae (3.74 %), Syllidae (1.67 %), Cirolanidae (1.52 %), Aoridae (1.45 %), Eriopisidae (1.02 %), Glyceridae (1.97 %) Corbiculidae (1.89 %), Gobiidae (1.38 %), Nassariidae (1.20 %), Veneridae (0.97 %), Psammobiidae (0.79 %), Cyrenidae (0.71 %) that contributed more than 94.79 percent of the total macrofaunal abundance. SIMPER analysis depicted three classes of macrofauna accounting for almost 24.83 percent of the average similarity within Group FB with 91.35 percent abundance of macrofauna, which include Polychaeta (16.85 %), Gastropoda (3.02 %), and Malacostraca (2.81 %). Family wise analysis depicted that 12.68 % of average similarity with nine families of macrofauna gave 90.18 percent cumulative abundance. Dominant families contributed to overall similarity of stations in group FB and were Capitellidae (4.14 %), Spionidae (3.11 %), Nereididae (0.97 %), Lumbrineridae (0.84 %), Nassariidae (0.76 %), Glyceridae (0.49 %), Corophiidae (0.43 %), Corbiculidae (0.35 %), and Aoridae (0.34 %).

Among the groups FA and FB, 85.12 percent average dissimilarity with 92 percent cumulative abundance was observed in five classes of macrofauna were Malacostraca (21.52 %), Bivalvia (20.63 %), Polychaeta (13.91 %), Hydrozoa (12.04), Gastropoda (10.54 %). The family-wise analysis depicted that 94.32 percent of average dissimilarity with eighteen families of macrofauna with 90.64 percent cumulative abundance. Major families showed average dissimilarity of stations in the Group FA and FB were Corophiidae (21.82 %), Cirolanidae (9.64 %), Mytilidae (8.24 %), Campanulariidae (7.21 %), Spionidae (4.82 %), Capitellidae (4.57 %), Corbiculidae (3.95 %), Aoridae (3.73 %), Veneridae (3.67 %), Lumbrineridae (3.03 %), Nassariidae (2.66 %), Terebellidae (2.48 %), Syllidae (2.42 %), Glyceridae (2.05 %), Nereididae (1.70 %), Anthuridae (1.40 %), Eriopisidae (1.27 %), and Onuphidae (0.84 %). Cluster FB consisted mainly of stations six, seven, two and five (50.69 % similarity, $p < 0.5$), among these stations, highest similarity was observed between station six and seven, where it was fresh water dominant (61.11 % similarity, $p < 0.5$)). However, station two and five joined to this group with 59.2 percent ($p < 0.5$) and 50.69 percent ($p < 0.5$) similarity respectively. Stations in the estuary have been separated into two significant groups based on their macrofaunal distribution with a 60 percent similarity. The nMDS derived from the pooled data, also clearly shows the salinity gradient and substrate difference in macrofaunal distribution at the family level. The year based differences between the faunal assemblages were not significant (ANOSIM, R = 0.013, p = 6.7 %) and so, the family level data for each year pooled.

b) Species-level

Before computing the similarity matrices, the entire data collected from the seven stations were square root transformed to reduce the impact of the species with high abundance on the assessment of the community similarities (Clarke and Gorley, 2006). The evaluation of R-values revealed that significant differences in species-wise distribution of macrofauna between stations (ANOSIM, Global R = 0.254, p = 0.1 %), whereas the faunal differences at species level from station 1 and 3 (ANOSIM, R = 0.168, p = 0.2 %), station two and five (ANOSIM, R = 0.127, p = 0.2 , station two and six (ANOSIM, R = 0.067, P = 1.3 %), two and seven (ANOSIM, R = 0.046, p = 6.2 %), five and six (ANOSIM, R = 0.122, p = 0.2 %), five and seven (ANOSIM, R = 0.071, p = 2.2 %) and five and seven (ANOSIM, R = 0.081, p = 1.5 %) were not statistically significant. The hierarchical clustering analysis based on Bray-Curtis similarity was then carried out to test the similarity among the stations. Hierarchical cluster analysis and SIMPROF test on the full set of data revealed that of the seven stations, grouped into two significant clusters (STA and STB) (47.19 %, p < 0.05) [Figure 46 a & b]. Of which STA exhibit higher similarity between stations station three, four and one (68.12 %, p < 0.5). While in group STB showed the overall similarity between stations six, seven, two and five of 50.25 percent (p < 0.5). SIMPER (similarity percentage) analysis was conducted to identify the species responsible for the defined clustering pattern of macrofauna. Thus, two significant (p <0.05) macrofaunal assemblages distinguished in the study area (STA and STB) [Figure 47 a-b]. The presence or absences of some unique species or the variation in abundance of predominant species were the basis for similarities and dissimilarities between assemblages. ANOSIM has applied to test the null hypothesis that there was no significant difference in faunal composition among these groups. One-way ANOSIM (identified through SIMPROF, $p < 0.5$) revealed that highly considerable differences exist in the faunal assemblages at the species level between these two groups (ANOSIM, R = 0.282, p = 0.1 %).

The stations of group STA formed a distinct cluster with a mean abundance of 6349 ± 3457, mean species richness of 5.37 ± 0.91, and the mean diversity of 2.18 ± 0.57. Stations in the group consisting of 67 species of macrofauna. The eight dominant and frequent species identified from the estuary were responsible for the formation of group STA, accounted for almost 62.20 % of the average similarity within the group STA. Which includes *Americorophium triaeonyx* (45.23 %), *Cirolana fluviatilis* (2.58 %), *Obelia bidentata* (2.50 %), *Grandidierella megnae* (1.89 %), *Arcuatula senhousia* (1.73 %), *Capitella* sp. (1.00 %), *Marcia recens* (0.97 %) and *Cyathura indica* (0.90 %). The amphipod, *A. triaeonyx* contributed highest macrofaunal abundance to Group STA (72.71 %). While stations in the group STB consist of 69 species of macrofauna with a mean abundance of 1767 ± 1231, the mean species richness 5.79 ± 0.69 and mean diversity of 3.57 ± 0.10 and its species composition and abundance differ from group STA. Twelve species accounted for almost 39.33 % of the average similarity within group STB, which include *Prionospio cirrifera* (15.81 %), *Capitella* sp. (7.93 %), *Notomastus* sp. (5.30 %), *Dendronereis aestuarina* (3.55 %), *Ninoe notocirrata* (1.96 %), *A. triaeonyx* (1.61 %), *O. bidentata* (1.37 %), *C. fluviatilis* (0.62 %), *Victoriopisa chilkensis* (0.53 %), *Glycera tridactyla* (0.47 %), *G. megnae* (0.42 %), *Nassodonta insignis* (0.41 %). *Prionospio cirrifera* (29.29 %), *Capitella* sp. (20.17 %) and *Notomastus* sp. (13.48 %) were the dominant species that contributed to average similarity of macrofaunal abundance to Group STB. The abundance of twelve species that contribute extensively to the average dissimilarity (80.99 %) between two groups STA and STB. They were *A. triaeonyx* (49.76 %), *C. fluviatilis* (4.08 %), *O. bidentata* (3.81 %), *A. senhousia* (3.25 %), *P. cirrifera* (2.80 %), *M. recens* (1.88 %), *G. megnae* (1.58 %), *Capitella* sp. (1.52 %), *Notomastus* sp. (1.48 %), *Villorita cyprinoides* (1.41 %), *Ceratonereis costae* (1.23 %), and *Cyathura indica* (0.85 %). Relatively high species richness (d = 0.97 ± 0.40), and diversity (H' = 1.79 ± 0.83) was noticed in the STA group and while species evenness (J' = 0.73 ± 0.19) and species dominance value (1-λ' = 0.57 ± 0.24) has been recorded maximum at STB group of stations [Figure 47 a-b].

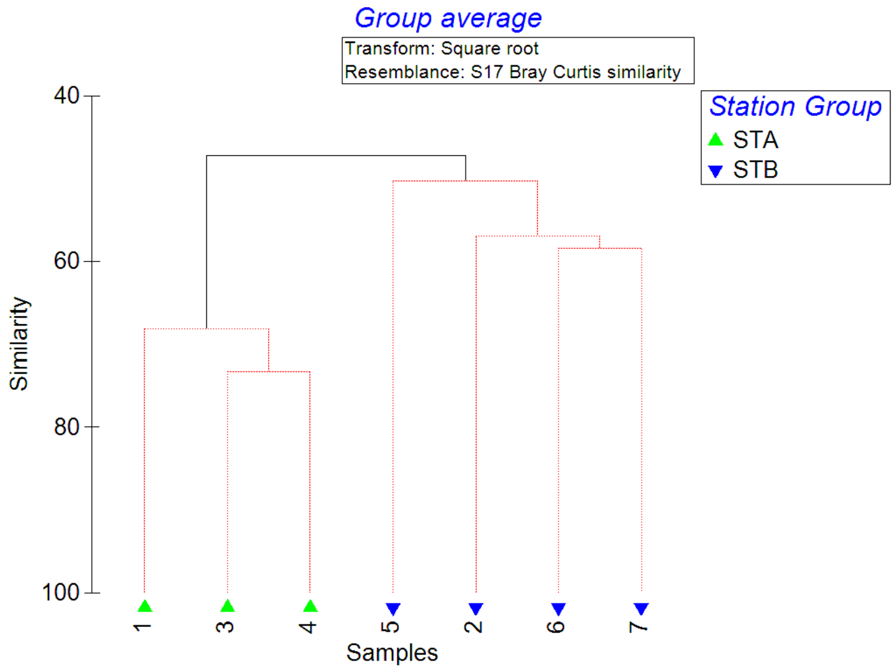


Figure 47 a. Dendrogram for macrofaunal species in each station in KAE (STA group: station 1, 3, & 4; STB group: station 5, 2, 6, & 7)

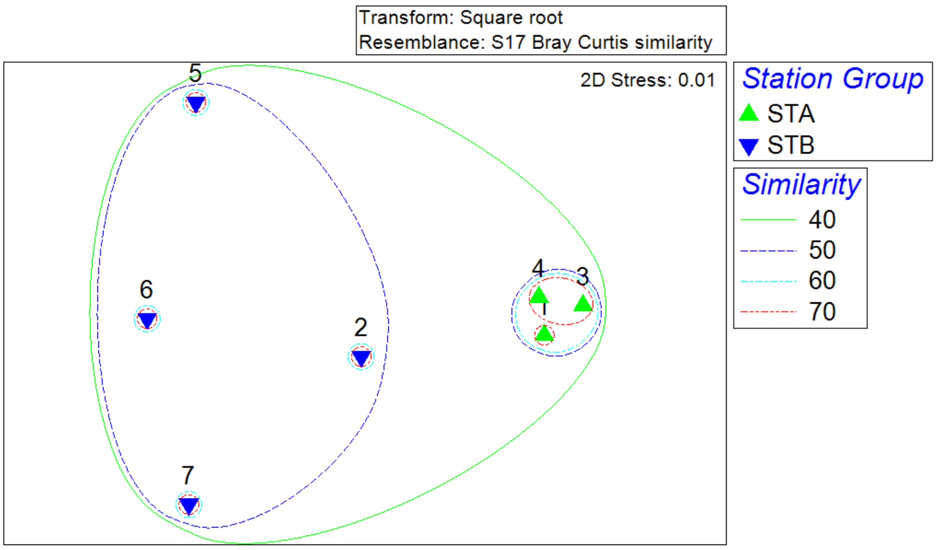


Figure 47 b. nMDS for macrofaunal species in each station in KAE (STA group: station 1, 3, & 4; STB group: station 5, 2, 6, & 7)

During the entire study period, thirteen species has formed with 8.66 percent average similarity in distribution at station one. The significant species were *A. triaeonyx, M. recens, D. aestuarina, Namalycastis indica, Meretrix casta, Glycera tridactyla, V. chilkensis, Crassostrea bilineata, C. fluviatilis, Capitella* sp.*, Ninoe notocirrata, Lumbrineris simplex.* In station two, 11 species were the primary contributor to distribution similarity, and they were *P. cirrifera, N. insignis, D. aestuarina, Capitella* sp.*, Cossura* sp.*, Glycera tridactyla, V. chilkensis, Ninoe notocirrata, Ctenapseudes chilkensis, Arcuatula senhousia, G. megnae* with 12.85 percent average similarity in distribution. Station three showed 27.82 percent in average distributional similarity with nine species, and they were *C. fluviatilis, M. recens, A. senhousia, A. triaeonyx, M. casta*, *G. megnae, N. insignis, Pista indica, D. aestuarina.* At station four, 13 species depicted average similarity of 25.62 percent of distributional similarity and species were *A. triaeonyx, V. cyprinoides, C. fluviatilis, A. senhousia, M. casta, M. recens, G. megnae, Cyathura indica, V. chilkensis, N. insignis, G. tridactyla, D. aestuarina, Capitella* sp. That for station 5 was nine species with 26.24 percent similarity, and they were *P. cirrifera, Capitella* sp.*, D. aestuarina, Notomastus* sp.*, V. cyprinoides, N. insignis, G. tridactyla, G. megnae* and *O.bidentata.* In station six, 7 species have formed with 18.67 percent similarity they were *Capitella* sp.*, P. cirrifera, D. aestuarina, Ninoe notocirrata, C. fluviatilis, V. chilkensis, Notomastus* sp. The station 7 depicted a similarity of 15.06 percent with 8 species such as *Notomastus* sp.*, Capitella* sp.*, D. aestuarina, N. insignis, P. cirrifera, A. triaeonyx, V. cyprinoides* and *G. tridactyla.*

The affinities among the seasons were established using non-metric multi-dimensional scaling. The stress value (Kruscal) measures the degree of coupling between sample distances and actual distances in the ordination. Ordination by multidimensional scaling (MDS), based on the season-wise species level similarity matrix, displayed a similar pattern, i.e. the same groups has been identified with a stress value of 0.01. The further analyses for understanding the seasonal variations have been carried out while considering the fauna of these seasons separately. The R-value depicted a significant seasonal variation of macrofaunal distribution in the KAE (ANOSIM, Global R

= 0.089, p = 0.1 %). The variations were primarily evident in the species level distribution between seasons such as monsoon and pre-monsoon of the first year (ANOSIM, R=0.129, p =0.1 %), monsoon of the first year and pre-monsoon of the second year (ANOSIM, R=0.158, p = 0.1 %), post-monsoon of the first year and pre-monsoon of the second year (ANOSIM, R= 0.189, p = 0.1 %), and post-monsoon of second year and pre-monsoon of the second year (ANOSIM, R=0.134, p = 0.1 %) showed statistically significant variation. Hierarchical cluster analysis and SIMPROF test on a full set of data revealed that of the three seasons for both first year and second year period were grouped into two clusters ($p < 0.05$) with the overall similarity of 56.6 percent, of which group SSA exhibited higher similarity between pre-monsoon seasons of the study period (2009-2010 and 2010-2011) with a likeness of 65.69 percent. The other group (SSB) has formed with rest of the seasons with an overall similarity of 64.9 percent [Figure 48]. Groups SSB was established with the highest similarity of monsoon (2009-2010) and post-monsoon (2010-2011) of the study period (69.05 %). R-values showed no significant differences between years in species distribution of macrofauna of the estuary (ANOSIM, Global R= 0.014, p = 5.4 %).

The seasons of group SSA formed a distinct cluster with a mean macrofaunal abundance of 1849 ± 544 and species richness of 0.82 ± 0.40 while that for diversity was 1.93 ± 0.67. Stations in the group consisting of 57 species of macrofauna. Fifteen species accounted for almost 42.84 percent of the average similarity within group SSA, which include *Obelia bidentata* (6.15 %), *Cirolana fluviatilis* (5.22 %), *Prionospio cirrifera* (4.43 %), *Capitella* sp. (3.95 %), *Pista indica* (3.12 %), *Marcia recens* (2.90 %), *Arcuatula senhousia* (2.77 %), *Americorophium triaeonyx* (2.41 %), *Dendronereis aestuarina* (1.98 %), *Ninoe notocirrata* (1.32 %), *Villorita cyprinoides* (1.23 %), *Cyathura indica* (1.10 %), *Grandidierella megnae* (0.70 %), *Meretrix casta* (0.70 %), *Notomastus* sp. (0.66 %). While *Obelia bidentata* (14.34 %), *Cirolana fluviatilis* (12.19 %), *Prionospio cirrifera* (10.35 %) were the dominant species that contributed to the overall similarity of macrofaunal abundance to group SSA. Group SSB consists of 72 species of macrofauna with a mean abundance of 4671 ± 1802, mean species richness 0.84 ± 0.41 and mean diversity of 1.8 ± 0.79 and its species composition and abundance differ from group SSA. Eight species accounted for almost 54.66 percent of the average similarity within Group SSB. Which include *A. triaeonyx* (32.70 %), *C. fluviatilis* (3.42 %), *P. cirrifera* (2.96 %), *Capitella* sp. (2.63 %), *M. recens* (2.50 %), *Notomastus* sp. (2.42 %), *G. megnae* (1.50 %), *D. aestuarina* (1.19 %). *Americorophium triaeonyx* that formed a primary contributor to numerical abundance (59.83 %) of macrofaunal species in SSB group.

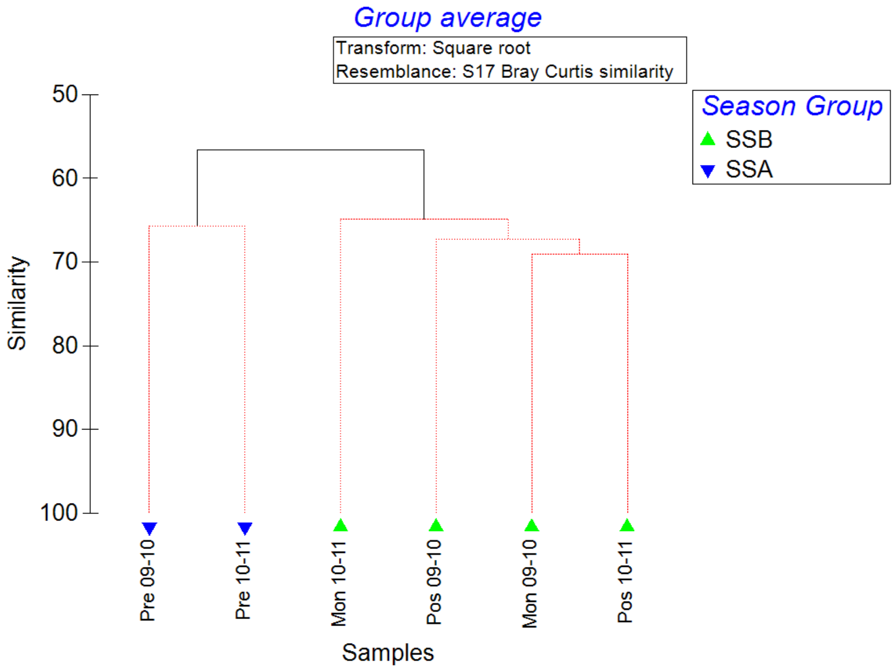


Figure 48. Dendrogram for macrofaunal species in each season in KAE (SSB group: season pre-monsoon 2009-2010, pre-monsoon 2010-2011; SSA group: monsoon 2010-2011, post-monsoon 2009-2010, monsoon 2009-2010, post-monsoon 2010-2011)

The abundance of 14 species that contributed extensively to the average dissimilarity (69.03 %) between two groups (SSA and SSB) formed between the seasons of the study area. They were *A. triaeonyx* (38.80 %), *O. bidentata* (7.21 %), *A. senhousia* (3.98 %), *P. cirrifera* (2.45 %), *Notomastus* sp. (2 %), *Pista indica* (1.28 %), *Ceratonereis costae* (1.25 %), *C. fluviatilis* (1.16 %), *M. recens* (0.93 %), *Capitella* sp. (0.88 %), *G. megnae* (0.81 %), *Cyathura indica* (0.48 %), *Ninoe notocirrata* (0.47 %), *V. cyprinoides* (0.45 %). The *A. triaeonyx* (56.20 %) formed the dominant species that prompted for average dissimilarity between SSA and SSB. Relatively high species diversity (H'=1.93 ± 0.67) and evenness (J'=0.76 ± 0.15) was observed in the SSA group (pre-monsoon) that compared species diversity (H'=1.68 ± 0.79) and evenness (J'=0.66 ± 0.21) to SSB group (monsoon and post-monsoon). Average similarity of 11.66 percent was noticed in the monsoon season of the first year with the dominance of twelve species with a cumulative contribution of 90.60 percent, and they were *P. cirrifera*, *C. fluviatilis*, *Notomastus* sp., *A. triaeonyx*, *D. aestuarina*, *Capitella* sp., *N.insignis*, *M. recens*, *G. megnae* and *A. senhousia.* In monsoon of the second year, *Capitella* sp., *A. triaeonyx*, *D. aestuarina*, *Ninoe notocirrata*, *P. cirrifera*, *M. recens*, *G. tridactyla*, *G. megnae*, *M.casta*, *N. insignis*, *Notomastus* sp., *C. fluviatilis* and *V. chilkensis* were contributed in decreasing order but with a cumulative abundance of 92.34 percent in the monsoon of the second year. That for post-monsoon in the first year, *O. bidentata*, *A. triaeonyx*, *P. cirrifera*, *Notomastus* sp., *G. megnae*, *D. aestuarina, C. fluviatilis*, *Capitella* sp., *N. insignis, M. recens*, *V. chilkensis* and *Cyathura indica* with 9.44 percent average similarity and cumulative abundance of 91.20 percent for twelve species. However, during post-monsoon in the second year, *Capitella* sp., *A. triaeonyx*, *D. aestuarina*, *N. notocirrata*, *P. cirrifera*, *M. recens*, *N. insignis, Notomastus* sp., *C. fluviatilis* and *V. chilkensis* had 6.52 percent average similarity for thirteen species with a cumulative abundance of 92.34 percent during post-monsoon of the second year period. In pre-monsoon of the first year, macrofaunal species *Capitella* sp., *D. aestuarina*, *V. chilkensis*, *G. tridactyla*, *N. notocirrata*, *C. fluviatilis*, *Pista indica*, *P. cirrifera*, *G. megnae*, *Cossura* sp., *N. insignis*, *Diopatra neapolitana*, *V. cyprinoides* and *M. recens* were depicted an average similarity of 11.33 percent with a cumulative abundance of 91.37 percent for fourteen species. At pre-monsoon of the second year period, species dominant with a average similarity of 10.83 percent (thirteen species) had a cumulative abundance of 91.51 percent and they were *A. senhousia*, *P. cirrifera*, *Capitella* sp., *D. aestuarina*, *Notomastus* sp., *P. indica*, *Crassostrea bilineata*, *O. bidentata*, *Lumbrineris simplex*, *M. recens*, *Ninoe notocirrata*, *C. fluviatilis* and *G. tridactyla*. The affinities among the seasons were established using non-metric multi-dimensional scaling. The stress value (Kruscal) measures the degree of coupling between sample distances and actual distances in the ordination. Ordination by multidimensional scaling (MDS), based on the season-wise species level similarity matrix, displayed a similar pattern, i.e. the same groups were identified with a stress value of 0.01. For analysing the different faunal assemblages of each season in the estuary, further analyses was carried out while considering the fauna of these seasons separately as SSA and SSB [Figure 48].

V. 2. 1. 4. Macrofaunal communities with environment

The data on macrofauna collected during the study was subjected to various statistical analyses. In the principal coordinate analysis (PCO), the first two axes explained about 24.6 percent of the total variability. PCO1 explained 16 percent variability of macrofaunal distribution due to the influence of rainfall, river discharge salinity, water temperature, transparency, turbidity, water column Eh and dissolved oxygen levels in the estuary. PCO2 explained 8.6 percent variability due to the variability of organic matter, organic carbon, sand, silt, clay, and transparency of water column in the estuary. Among these variables, the highest correlation was represented with organic matter, sand, silt clay and river discharge pattern. The direction of the vectors expressing organic matter, clay, silt, and sediment temperature increased towards station group STB. While, the depth, percentage of sand, salinity and sediment pH increased towards STA. The seasonal group SSA and SSB were separated mainly by the influence of environmental parameters such as river discharge, rainfall, DO, Chl-*a*, and transparency [Figure 49].

Canonical analysis of principal coordinates (CAP) performed to confirm the above pattern, the canonical correlation values of first two axes were 0.7354 and 0.4261 respectively. The advantage of using CAP analysis is that it prepares a model projecting the points (samples) on two axes that minimises the residual variation. To estimate the goodness of fit, taking one sample out of it analyses was done with the other samples. Then, the left out samples placed in the canonical space produced by the others. Then, finding out whether that sample is falling in the same group (getting allocated) to which it belongs, done for all the samples and allocation (classification) was verified. In the present study, the percentage of samples allocated to the correct groups STA, STB, SSA, and SSB was 57.143 percent (96 out of the 168 samples). CAP was also performed to confirm the grouping pattern of stations (STA and STB), the canonical correlation value obtained was comparatively high (0.7138). The percentage of samples allocated to the correct group in CAP was 80.952 percent (136 out of the 168 samples). As the allocation success of samples is substantially higher around 81 percent, the CAP explained a higher percentage of variability in organic carbon. The seasonal groping pattern was analysed using CAP (SSA and SSB) and the canonical correlation value obtained was 0.4057. The percentage of samples allocated to the correct group in CAP was 67.8570 percent (114 out of the 168 samples).

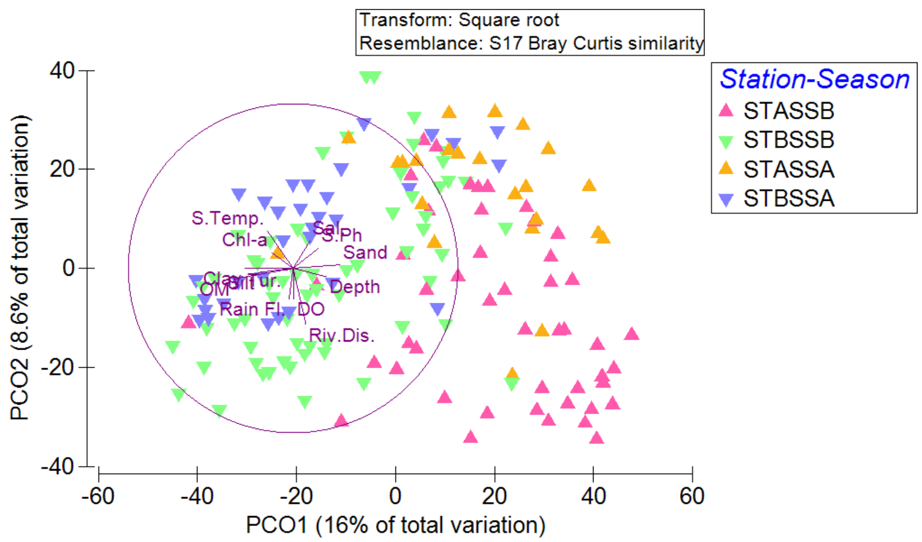


Figure 49. Principal coordinate analysis (PCO) ordinations of environmental data superimposed with macrofauna in KAE (STA-SSA: stations 1, 3 & 4 during pre-monsoon; STB-SSA: stations 2, 5, 6 & 7 during pre-monsoon; STA-SSB: stations 1, 3 & 4 during monsoon and post-monsoon; STB-SSB stations 2, 5, 6 & 7 during monsoon and post-monsoon)

The canonical analysis of principal coordinates (CAP) routine was used to identify members of the benthic assemblage that associated with the seasonal assemblages (SSA and SSB) distinguished in the clustering and MDS visualisation along with sample interactions associated with plot position relative to the spatial variability between stations [Figure 50]. The CAP analysis for macrofaunal abundance evidenced significant relationship between the biotic and abiotic matrices (p = 0.001).

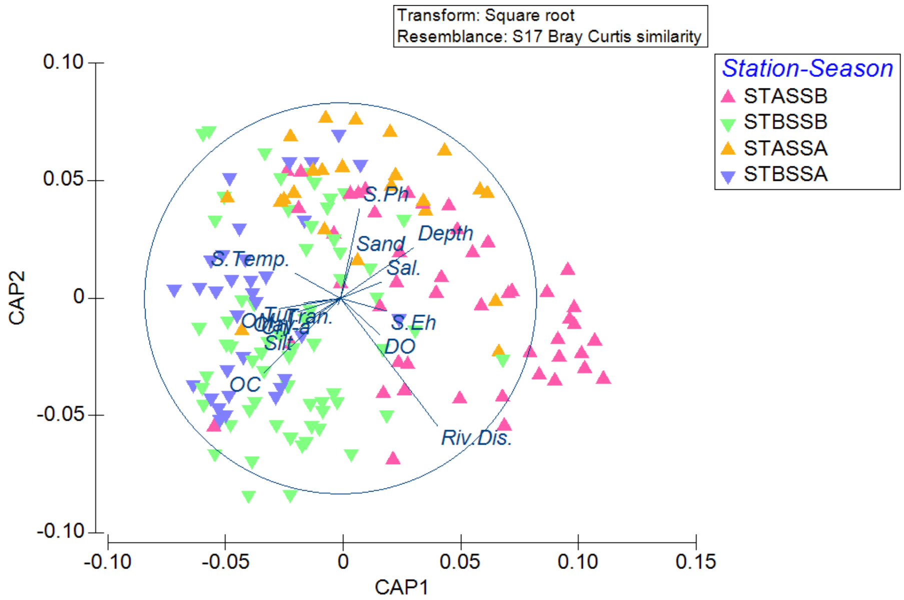


Figure 50. Canonical analysis of principal coordinates (CAP) showing direction of vector representing environmental factors towards spatio-temporal grouping of macrofaunal abundance in KAE (STASSA: stations 1, 3 & 4 during pre-monsoon; STBSSA: stations 2, 5, 6 & 7 during pre-monsoon; STASSB: stations 1, 3 & 4 during monsoon and post-monsoon; STBSSB stations 2, 5, 6 & 7 during monsoon and post-monsoon) Riv.Dis.: river discharge, S.ph: Sediment pH, S.Eh: sediment redox potential, S. Temp: sediment temperature, Tur: turbidity, Tran: transparency, Sal.: salinity, Chl-*a*: chlorophyll-*a*, depth, DO, OC, sand, silt, clay

The first two canonical correlations were 0.6001 and 0.5136 respectively, and the canonical axes explained 36.01 percent of the distribution of samples. CAP correlation of environmental variables such as organic carbon (0.391), organic matter (0.315), salinity (-0.212), sediment temperature (0.234), dissolved oxygen (-0.203), sediment Eh (-0.237), silt (0.226), clay (0.221), river discharge (-0.495), depth (-0.374) was significantly related with distribution of macrofauna along the CAP 1 and while sediment pH (0.459), sand (0.209), river discharge (-0.653), depth (0.26) were prominent along the CAP 2. Percentage of samples allocated to the correct station groups (STA and STB), with 80.95 percent (136 out of the 168 samples). While that for season group (SSA and SSB) was 67.86 percent (114 out of the 168 samples) and the overall, classification of SSA-STA-SSB-STB was 57.143 percent (96 out of the 168 sample). The CAP analysis for seasons evidenced significant correlations between the biotic and abiotic matrices (p = 0.0002). The two first canonical correlations were 0.6760 and 0.6327 respectively, and the canonical axes explained 40.03 percent of the distribution of samples. The CAP analysis for stations evidenced significant correlations between the biotic and abiotic matrices (p = 0.0002). The two first canonical correlations were 0.8247 and 0.6592 respectively, and the canonical axes explained 43.45 percent of the distribution of samples.

The Canonical Correspondence Analysis (CCA) was carried out to determine which environmental factors influence the distribution of the macrofaunal species [Table 9 & Figure 51]. Monte Carlo permutation test (with forward selection) was used to find out significant environmental variables responsible for the variance of species distribution ($p < 0.05$).

Table 9. Canonical Correspondence Analysis (CCA) results

Axes	**1**	**2**	**3**	**4**	**Total inertia**
Eigenvalues	0.65	0.21	0.19	0.14	12.71
Species-environment correlations	0.89	0.59	0.60	0.64	
Cumulative percentage variance					
Species data	5.1	6.7	8.2	9.3	
Species-environment relation	39.3	52	63.7	72.3	
Sum of all eigenvalues	12.71				
Sum of all canonical eigenvalues	1.64				

The CCA axes 1 and 2 explained 39.3 and 12.7 percent of the species variation respectively. Axis 1 of the CCA ordination plot (eigenvalue, 0.646) separated the river discharge (r = -0.557), depth (r = -0.519), salinity 0.338, turbidity (r = 0.3117), DO (r =-0.346), sediment temperature (r = 0.371), sediment Eh (r = -0.330), Chl-*a* (r = 0.394). Axis 2 of the CCA ordination plot (eigenvalue, 0.208) separated further by organic carbon (r = 0.397), organic matter (r = 0.396), RD (r = 0.282), clay (r = 0.231), silt (r=0.108), sand (r = -0.167) and transparency (r = -0.115).

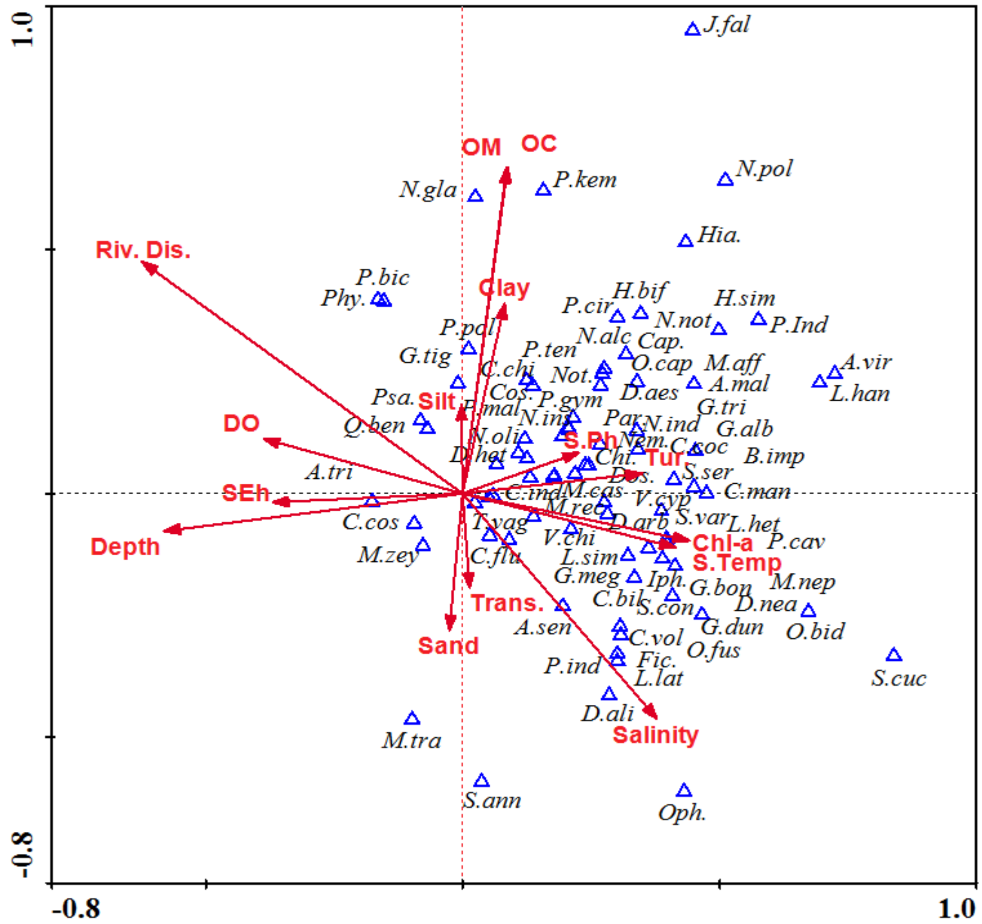


Figure 51. Canonical Correspondence Analysis (CCA) showing scatter plot for 79 macrofaunal species (SEh: sediment Eh, Sed.Temp.: sediment temperature, Cla-*a*: chlorophyll-*a*, S.Ph: sediment pH, Trans.: transparency, Riv.Dis: river discharge, salinity, sand, silt, clay, OM: organic matter, OC: organic carbon)
* List of species & abbreviation given in Table 13

The stations with comparatively higher salinity, turbidity, sediment Eh, and sandy substrate (stations 1, 3, & 4 in STA) were found to be the strongly linked parameters in the distribution of macrofaunal species. Species such as *Murex trapa*, *Callogobius mannarensis*, *Scylla serrata*, *Ophiothrix* sp., *Diogenes alias*, *Ficopomatus* sp., *Sphaeroma annandalei*, *Marcia recens*, *Saccostrea cucullata*, *Meretrix casta*, *Melita zeylanica*, *Cyathura indica*, *Grandidierella megnae* which well distributed at these stations.

Table 10. Subset of macrofaunal species used for Canonical Correspondence Analysis (CCA)

Species name (Abbreviation)	**Species name (Abbreviation)**
Metapenaeus affinis (M.aff)	*Phyllodoce* sp. (Phy.)
Penaeus indicus (P.Ind)	*Glycinde bonhourei* (G.bon)
Neorhynchoplax alcocki (N.alc)	*Glycera alba* (G.alb)
Philyra malefactrix (P.mal)	*Glycera tridactyla* (G.tri)
Scylla serrata (S.ser)	*Sigambra constricta* (S.con)
Alpheus malabaricus (A.mal)	*Owenia fusiformis* (O.fus)
Diogenes alias (D.ali)	*Ficopomatus* sp.(Fic.)
Lucifer hanseni (L.han)	*Paraprionospio* sp. (Par.)
Quadrivisio bengalensis (Q.ben)	*Prionospio cirrifera* (P.cir)
Melita zeylanica (M.zey)	*Prionospio polybranchiata* (P.pol)
Americorophium triaeonyx (A.tri)	*Pseudopolydora kempi* (P.kem)
Corophium volutator (C.vol)	*Pista indica* (P.ind)
Victoriopisa chilkensis (V.chi)	*Ophelia capensis* (O.cap)
Jassa falcata (J.fal)	*Cossura* sp. (Cos.)
Psammogammarus (Psa.)	*Capitella* sp. (Cap.)
Gammarus tigrinus (G.tig)	*Heteromastus similis* (H.sim)
Grandidierella megnae (G.meg)	*Heteromastides bifidus* (H.bif)
Iphinoe sp. (Iph.)	*Parheteromastus tenuis* (P.ten)
Cyathura indica (C.ind)	*Notomastus* sp. (Not.)
Synidotea variegate (S.var)	*Villorita cyprinoides* (V.cyp)
Sphaeroma annandalei (S.ann)	*Marcia recens* (M.rec)
Cirolana fluviatilis (C.flu)	*Dosinia* sp. (Dos.)
Ctenapseudes chilkensis (C.chi)	*Meretrix casta* (M.cas)
Pagurapseudopsis gymnophobia (P.gym)	*Hiatula* sp. (Hia.)
Miyakella nepa (M.nep)	*Cuneocorbula cochinensis* (C.coc)
Gastrosaccus dunckeri (G.dun)	*Arcuatula senhousia* (A.sen)
Lumbrineris latreilli (L.lat)	*Crassostrea bilineata* (C.bil)
Lumbrineris simplex (L.sim)	*Saccostrea cucullata* (S.cuc)
Ninoe notocirrata (N.not)	*Nassodonta insignis* (N.ins)
Lumbrineris heteropoda (L.het)	*Murex trapa* (M.tra)
Diopatra neapolitana (D.nea)	*Trypauchen vagina* (T.vag)
Namalycastis indica (N.ind)	*Psammogobius biocellatus* (P.bic)
Dendronereis arborifera (D.arb)	*Acentrogobius viridipunctatus* (A.vir)
Dendronereides heteropoda (D.het)	*Callogobius mannarensis* (C.man)
Dendronereis aestuarina (D.aes)	*Ophiothrix* sp. (Oph.)
Perinereis cavifrons (P.cav)	*Obelia bidentata* (O.bid)
Neanthes glandicincta (N.gla)	*Balanus improvises* (BalImp)
Ceratonereis costae (C.cos)	*Chironomus sp.* (Chi.)
Nephtys oligobranchia (N.oli)	*Nemertea* (Nem.)
Nephtys polybranchia (N.pol)	

While, *Hiatula* sp., *Psammogobius biocellatus*, *Cuneocorbula cochinensis*, *Lucifer hanseni*, *Nephtys polybranchia*, *Paraprionospio* sp., *Phyllodoce* sp., *Pseudopolydora kempi*, *Obelia bidentata*, *Glycinde bonhourei*, *Owenia fusiformis*, *Hiatula* sp., *Diopatra neapolitana* and *Villorita cyprinoides* were dominant in stations with high bottom Chl-*a* content, comparatively high sediment temperature with shallow depth, especially at station five (STB). However, distribution of nemerteans, Chironomus larvae, *Pagurapseudopsis gymnophobia*, *Ctenapseudes chilkensis*, *Ophelia capensis*, *Namalycastis indica*, *Prionospio polybranchiata*, *Psammogammarus* sp., *Synidotea variegata*, *Iphinoe* sp., *Dosinia* sp., *Victoriopisa chilkensis*, *Dendronereides heteropoda*, *Dendronereis arborifera*, *Alpheus malabaricus* and *Neanthes glandicincta* were found strongly related to organic matter enriched clayey silt sediment with well-oxygenated bottom water, paticularly in station five (STB). Similarly, clayey silt with organic matter enriched sediment has possibly stimulated the population of *Heteromastides bifidus*, *Heteromastus similis*, *Acentrogobius viridipunctatus*, *Nephtys oligobranchia*, *Miyakella nepa*, *Jassa falcata*, *Glycera alba*, *Capitella* sp., *Pista indica*, *Ninoe notocirrata*, *Prionospio cirrifera*, *Dendronereis aestuarina*, *Metapenaeus affinis* and *Ceratonereis costae* in station six and seven of STB group.

V. 2. 2. Feeding guild composition of macrofauna

Many species exploit the same class of resources similarly within an assemblage of macrofaunal groups, and in general, their feeding guilds are divided into macrophagous and microphagous modes. Although macrophagous are again subdivided as two sub-modes such as herbivores (HVR) and carnivores (CVR), while the microphagous are classified to three sub-modes that are filter feeders or suspension feeders (SF), deposit feeders (DF) and omnivorous (OVR). In a total of 18846 macrofaunal organisms collected, malacostracan group represented the dominant component of the benthic population and accounted for 60 percent of the total organisms recorded. Polychaetes constituted 20 percent, bivalves and gastropods 9 percent, while other groups recorded a total of 11 percent of the macrobenthic population. Suspension feeding amphipod *Americorophium triaeonyx* under the family Corophiidae was the most represented macrofaunal species. Other dominant macrofaunal members were suspension feeders (SF) from the hydroid colony of *Obelia bidentata*, surface deposit feeders (SDF) of sedentarian polychaete like *Prionospio cirrifera*, suspension feeder (SF) bivalve-like *Arcuatula senhousia*, carnivores (CVR) isopod like *Cirolana fluviatilis* and sub-surface deposit feeder (SSDF) sedentarian polychaete like *Capitella* sp. [Figure 52].

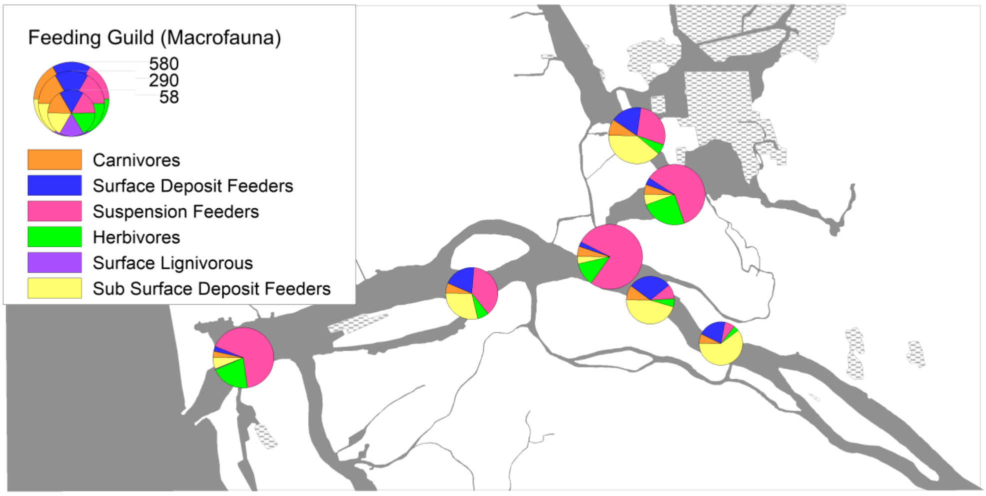


Figure 52. Mean percentage contribution of different feeding guild in KAE during 2009-2011 period, size of each pie diagram depicting spatial variation (ind.m^{-2})

During the first year period, macrofaunal assemblage showed the dominance of suspension feeders (SF) with 41.56 percent followed by 20.58 percent of sub-surface deposit feeder (SSDF), 20.23 percent of HVR, 10.51 percent SDF and 7.13 percent CVR. During the second year period, SF dominated with 66.14 percent of total density, 12.35 percent of SSDF, 9.34 percent of HVR, 6.92 percent of SDF and 5.16 percent CVR with sporadic occurrence of wood boring isopods. Based on hierarchical cluster analysis and SIMPROF test, the stations 3, 4 and 1 grouped as STA and rest of the stations 2, 5, 6, and 7 were grouped as STB. The stations in group STA dominated with 70.41 percent of SF followed by 17.09 percent HVR, 4.81 percent SSDF, 4.76 percent CVR and the rare occurrence of 0.32 percent SRL. While group STB dominated with 39.74 percent SSDF, 25.27 percent SF, 20.62 percent SDF, 8.55 percent CVR and 5.82 percent HVR. In the seasonal pattern, monsoon and post-monsoon periods were grouped as SSB in the SIMPROF test and pre-monsoon seasons formed as SSA. SSB group of stations in the estuary was dominated with 57.79 percent SF, 14.69 percent SSDF, 13.69 percent HVR, 8.30 percent SDF and 5.46 percent CVR, while SSA group dominated with 50 percent SF, 20.02 percent SSDF, 13.23 percent HVR, 8.49 percent SDF and 8.27 percent CVR [Figure 53 a-f].

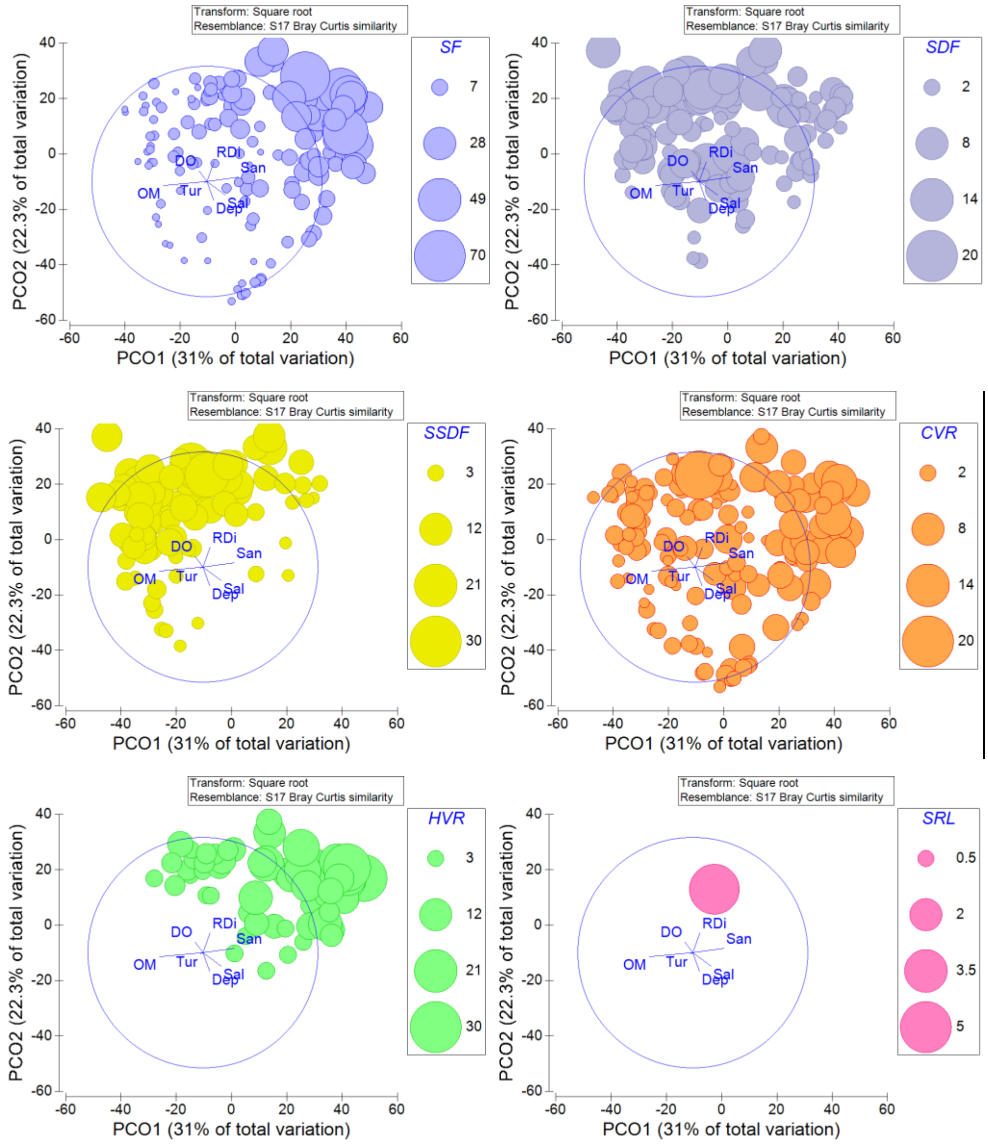


Figure 53 a-f. PCO ordinations of environmental data superimposed with bubble plots of estimated different feeding guild (SF, SDF, SSDF, CVR, HVR & SRL) of macrofauna in KAE

a) Feeding guild of Polychaetes

In this investigation, following Fauchald and Jumars (1979), the feeding guilds were classified as carnivores (CVR), surface deposit feeders (SDF), subsurface deposit feeders (SSDF), filter feeders or suspension feeders (SF) and omnivores (OVR). As Fauchald and Jumars (1979) have furnished information at the family level, the assigning of all the species encountered in the study to a particular feeding guild confirmed after examination of the mouthparts. The available food resources in the sediments are thereby utilised and partitioned between these trophic guilds [Figure 54].

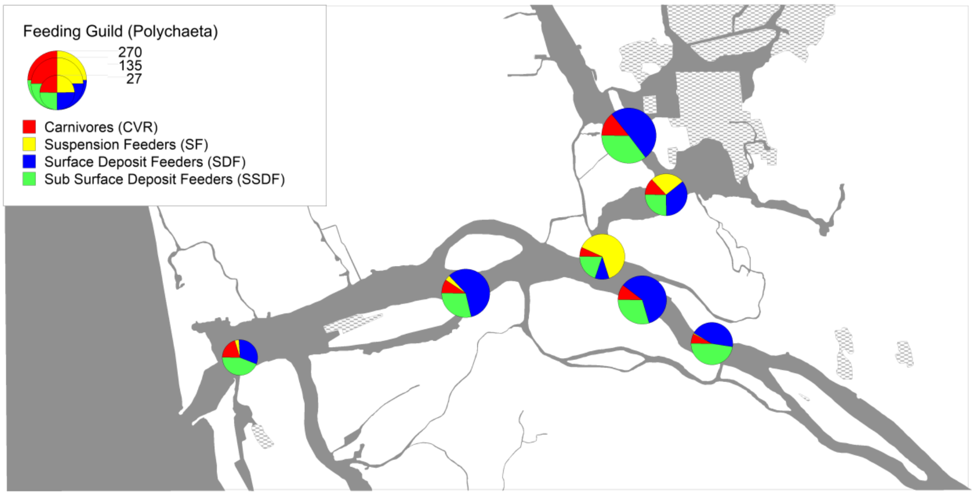


Figure 54. Mean percentage contribution of different feeding guild of polychaetes in KAE during 2009-2011 period, size of each pie diagram depicting spatial variation (ind.m^{-2})

The polychaete species represented in the present study was assigned to one of four feeding guilds such as carnivores (CVR), surface deposit feeders (SDF), sub-surface deposit feeders (SSDF), and suspension feeders (SF). The composition and structure of the polychaete feeding guilds were investigated using the species abundance data for understanding the spatio-temporal changes. Annual data showed a dominance of SDF polychaetes during both years followed by SSDF. Seasonally, group SSA (pre-monsoon) was dominated with SDF (51.95 %) and followed by SSDF (34.57 %), CVR (12.23 %) and SF (0.25 %). In seasonal group, SSA (pre-monsoon) was dominated with SF (38.63 %), SDF (28.56 %), SSDF (23.51 %) and CVR (9.25 %). Stations in the group

STA (station 1, 3 & 4) were dominated with SF (52.49 %), SSDF (33.28 %), SDF (28.42 %), and CVR (14.23 %). In spatial group STB (station 2, 5, 6 & 7) was dominated with SDF (53.40 %), SSDF (34.25 %), CVR (11.46 %) and SF (0.89 %). More clearly, stations one (43.83 %) and seven (47.38 %) was dominated with SSDF, station three with SF (63.21 %) and stations two (58.32 %), station four (35.27 %), station five (50.39 %), and station six (59.95 %) were dominated with SDF [Table 11 a].

Table 11 a. Feeding guild of polychaetes in KAE

Family	Feeding guild
Lumbrineridae	Carnivores (CVR)
Lumbrineridae	Carnivores (CVR)
Onuphidae	Carnivores (CVR)
Syllidae	Carnivores (CVR)
Nereididae	Carnivores (CVR)
Nephtyidae	Carnivores (CVR)
Phyllodocidae	Carnivores (CVR)
Goniadidae	Carnivores (CVR)
Glyceridae	Carnivores (CVR)
Glyceridae	Carnivores (CVR)
Pilargidae	Carnivores (CVR)
Oweniidae	Sub-surface deposit feeders (SSDF)
Serpulidae	Suspension feeders (SF)
Spionidae	Surface deposit/detritus feeders (SDF)
Terebellidae	Suspension feeders (SF)
Opheliidae	Sub-surface deposit feeders (SSDF)
Capitellidae	Sub-surface deposit feeders (SSDF)

b) Feeding guild of malacostracan crustaceans

Following the classification of Macdonald *et al.* (2010) and Fauchald and Jumars (1979), the feeding guilds in malacostracans in the present study were represented by carnivores (CVR), surface deposit or detritus feeders (SDF), suspension feeders (SF), herbivores (HVR), and sporadic occurrence of wood boring isopod (Lignivorous; LIG). Among the malacostracan communities, suspension feeders dominated in all stations with spatial variation. STA group of stations and station 2 in the STB highly dominated with SF. In the riverine stations, SF was comparatively less in number (Station 5, 6, & 7), station five (STB) dominated with HVR, and other stations (station 6 and 7) depicted increased population of CVR and HVR [Table 11 b].

Table 11 b. Feeding guild of malacostracan crustaceans in KAE

Family	Feeding guild
Penaeidae	Carnivores (CVR)
Hymenosomatidae	Surface deposit/detritus feeders (SDF)
Leucosiidae	Surface deposit/detritus feeders (SDF)
Portunidae	Carnivores (CVR)
Alpheidae	Carnivores (CVR)
Diogenidae	Carnivores (CVR)
Luciferidae	Suspension feeders (SF)
Maeridae	Surface deposit/detritus feeders (SDF)
Melitidae	Surface deposit/detritus feeders (SDF)
Corophiidae	Suspension feeders (SF)
Corophiidae	Suspension feeders (SF)
Eriopisidae	Suspension feeders (SF)
Ischyroceridae	Suspension feeders (SF)
Gammaridae	Carnivores (CVR)
Aoridae	Herbivores (HVR)
Bodotriidae	Surface deposit/detritus feeders (SDF)
Anthuridae	Carnivores (CVR)
Idoteidae	Herbivores (HVR)
Sphaeromatidae	Surface ligniorous (SRL)
Cirolanidae	Carnivores (CVR)
Apseudidae	Surface deposit/detritus feeders (SDF)
Squillidae	Carnivores (CVR)
Mysidae	Suspension feeders (SF)

c) Feeding guild of molluscs

In the present study, all bivalve species collected were suspension feeders (SF) group. The single dominant gastropod, *Nassodonta insignis* was surface deposit feeder (SDF). Carnivore (CVR) gastropod, *Murex trapa* was collected twice from estuarine mouth region. Among the molluscan communities, SF dominated in almost all stations along with SDF such as *N. insignis* [Table 14 c].

Table 11 c. Feeding guild of bivalve and gastropod molluscs in KAE

Family	Feeding guild
Cyrenidae	Suspension feeders (SF)
Veneridae	Suspension feeders (SF)
Psammobiidae	Suspension feeders (SF)
Cuspidariidae	Suspension feeders (SF)
Mytilidae	Suspension feeders (SF)
Ostreidae	Suspension feeders (SF)
Nassariidae	Surface deposit/detritus feeders (SDF)
Muricidae	Carnivores (CVR)

d) Feeding guild of 'Others groups'

Suspension feeders dominated in abundance of 'other group' by the suspension feeding hydroid colony of *Obelia bidentata*, echinoderm *Ophiothrix* sp., *Balanus improvises*, along with four carnivore fishes under the family Gobiidae, nemerteans and surface deposit feeding chironomid larvae [Table 14 d].

Table 11 d. Feeding guild of 'other group' in KAE

Family	Feeding guild
Gobiidae	Carnivores (CVR)
Gobiidae	Carnivores (CVR)
Ophiotrichidae	Suspension feeders (SF)
Campanulariidae	Suspension feeders (SF)
Balanidae	Suspension feeders (SF)
Chironomidae	Surface deposit/detritus feeders (SDF)
Nemertea	Carnivores (CVR)

V. 3. Discussion

In general, estuaries are less diverse than other marine systems and such conditions observed in the soft bottom of Kodungallur-Azhikode estuary (KAE). However, benthic diversity pattern of KAE is comparable to other tropical estuaries (Gray *et al.*, 1997). According to Asha (2017), Shanon diversity value for southern Vembanad Lake was relatively low (1.78) that compared to marine influenced middle zone (1.85). While central zones of the KAE depicted relatively higher values than southern and middle zones of Vembanad Lake. However, these tropical estuaries are much diverse than boreal estuaries (Sanders,1968). The comparatively low diversity in the estuaries is ensued by the wide range of biophysical forcing (Gray *et al.*, 1997; Herman *et al.*, 1999; McLusky, 1999).

Biocoenosis of macrofauna in the Kodungallur-Azhikode estuary (KAE) displayed a significant variation in diversity, density and pattern of distribution. These variables are influenced by the physical, chemical and biological response, as well as anthropogenic influences on the estuarine system, which varied on a spatial and temporal scale. These factors are potential explanatory variables for mesoscale spatiotemporal patterns on the local benthic ecology (John *et al.*, 2002; Kristensen, 1988; Ortega Cisneros *et al.*, 2011). The diversity of macrofaunal species in the estuary attained a maximum at pre-monsoon period and it was relatively low at the post-monsoon season. Similarly, the post-monsoon period dominated with opportunistic species like *Americorophium triaeonyx, Cirolana fluviatius,* and *Prionospio cirrifera*. The dominance of such species are mostly seen in the ecologically stressed environments (Ajmal Khan *et al.*, 2004; Fernando et al., 1984; Hermand *et al.*, 2008). In a spatial scale, macrofauna species richness was maximum observed at middle zone with sandy substratum, while low species richness was recorded in the Riverhead. Seasonally, moderate species richness was observed during the period of monsoon with heavy river discharge. Species evenness in the estuary depicted a relatively low value at the middle zone, but seasonally it was high at pre-monsoon period. In the present observation, polychaetes were the most diverse group in term of macrobenthic species diversity, while malacostracan crustaceans dominated in the numerical density and bivalve molluscs led the biomass.

The polychaetes were the most diverse community in the KAE, they found ubiquitously in almost all marine and estuarine sediments, with high abundance and diversity (Fauchald and Jumars, 1979; Glasby and Timm, 2008; van der Linden *et al.*, 2017). More than 12,000 polychaete species, belonging to 83 families have been described in this class so far (Hutchings, 1998; Rouse and Pleijel, 2001) and various estimates have made as to the total polychaete fauna ranging from 25,000 to 30,000 (Snelgrove, 1997). Among them, more than 1000 species of polychaetes have been reported from Indian waters that contribute 8.66 percent to global polychaete diversity (Geetha *et al.*, 2015; Venkataraman and Wafar, 2005). According to Ajmal Khan and Murugesan (2005), polychaete diversity studies in Indian estuaries are comparatively less, especially in west coast estuaries. From the available literature, they catalogued 153 species from the Indian estuaries, and east coast estuaries registered highest polychaete diversity. The highest polychaete species count observed from Vellar estuary had 98 species, while 69 species in Hoogli-Malta estuary, 44 species in Vasista-Godavari estuary, 37 species in Coleroon estuary and 33 species in Mahanadhi along the east coast of India. In west coast, they record the highest count from Cochin backwater with 19 species and that for Mandovi and Zuari estuarine system was ten species while, seven species from Ashtamudi estuary and five species from Mulki estuary (Ajmal Khan and Murugesan, 2005). The present study recorded 33 species of polychaetes out of 79 species of macrofaunal species identified from Kodungallur-Azhikode estuary.

The various observations on polychaete species diversity of Vembanad ecosystem depicted considerable variation from 11 to 53 species (Ansari, 1974; Feebarani, 2009; Geetha *et al.*, 2015; John, 2009; Martin *et al.*, 2011; Pillai, 1977; Pillai, 2001; Rehitha *et al.*, 2017; Remani *et al.*, 1983; Sarala Devi *et al.*, 1991; Sheeba, 2000; Sunil Kumar, 1993). The dominant polychaete species that was reported from the Vembanad ecosystem are belonging to capitellids (*Capitella* sp., *Scyphoproctus armatus*, *Heteromastides bifidus* & *Parheteromastus tenuis*), spionids (*Prionospio polybranchiata*, *Prionospio cirrifera*), nereids (*Dendronereis aesturina* & *Namalycastis indica*), lumbrinerids (*Lumbriconereis simplex*), pilargids (*Sigambra constricta*) and nephtyids (*Micronephthys oligobranchia*). In the Vembanad Lake, *P. cirrifera* was depicted maximum numerical density of 1068±1412 ind.m^{-2} at middle zone (Asha, 2017). According to Sarala Devi *et al.* (1991), studies in the northern extension of Cochin backwater system and recorded 30 species of polychaetes in which *Capitella* sp., *D. aestuarina, N. indica* and *P. tenuis* were common. Similarly, *P. cirrifera*, *Capitella* sp., *Notomastus* sp., *Ceratonereis costae* and *D. aestuarina* were dominated the soft bottom of KAE. Maximum species aggregation was observed in the northern zone of the estuary with sandy sediment. Seasonally, monsoon discharge significantly affected the polychaete diversity. They exhibited relatively high diverse at pre-monsoon due to less bottom disturbance.

The malacostracan-crustaceans were the most dominant communities in the macrofaunal numerical density of KAE. They also formed as the second most diverse group in the macrofaunal community, and that consists of amphipods, decapods, isopods, tanaids, stomatopods, cumaceans and mysids. Among them, amphipods dominated in the numerical density of soft bottom habitats; their global diversity exceeds 10,000 species (Feebarani, 2009; Geetha and Bijoy Nandan, 2014; Lowry and Myers, 2013; Moreira *et al.*, 2008). In Cochin backwater, malacostracan crustaceans have significantly contributed to benthic diversity and their species number varied in time and space from 12 to 34 species (Cheriyan, 1977; Feebarani, 2009; Geetha and Bijoy Nandan, 2014; Rao, 1968; Sheeba, 2000). According to Geetha and Bijoy Nandan (2014), amphipod species *Corophium volutator*, *Gammarus tigrinus* and *Victoriopisa chilkensis* were the dominant species in Cochin estuary. While the southern zone of Vembanad Lake dominated by *Cheiriphotis geniculata* (Asha 2017). However, in the KAE, *Americorophium triaeonyx*, *Grandidierella taihuensis* and *V. chilkensis* were dominated the amphipod community. Other dominant malacostracan group in the estuary was isopods, which mainly dominated by *Cirolana fluviatilis*.

The recent estimates showed that, the extant species diversity of phylum Mollusca is around 45000 to 50000 spp. of marine, 25000 spp. of terrestrial and 5000 spp. of freshwater habitats (Appeltans *et al.*, 2012; Biju Kumar and Ravinesh, 2016; WoRMS, 2017). However, they are less diverse in the estuarine environment. In the present study, species richness of molluscan group was least observed at the riverine zone and relatively higher at the northern and middle region of the estuary that is dominate with sand and less organic matter. The mussel *Arcuatula senhousia*, clams *Marcia recens*, *Meretrix casta*, *Villorita cyprinoides* and gastropod *Nassodonta insignis* were the dominant species of molluscan communities in the KAE. The species diversity of molluscs in Cochin backwaters varied from 2 to 18 species in several studies, which includes marine species from the estuarine mouth (Asha, 2017; Feebarani, 2009; Sheeba, 2000). Many species of estuarine bivalves demonstrates high endemism; the present study also observed an endemic species, *Cuneocorbula cochinensis* (Oliver *et al.*, 2016). However, many of brackish water species are still in the taxonomic ambiguities; comprehensive molecular studies that combined with conventional taxonomic investigations are highly recommended to resolve such problems. The juveniles of brackish water oyster species, *Crassostrea bilineata*, was collected from the hard bottom substrates like the clamshell. The juveniles organisms (<1cm) were only considered as macrofauna. They were massively colonised in the concrete, rock and wood substrates along the shores of KAE.

Gastropod diversity in the estuary was restricted to *Nassodonta insignis* with a sporadic live occurrence of juvenile of marine species, *Murex trapa* near at fishing harbour. The distribution of *N. insignis* is limited to brackish waters of southern India (Jayachandran *et al.*, 2018). Even though they are abundant in brackish waters of Kerala, records are limited (Preston; 1916). At the same time, many unusual records on *Littorina* sp. found. The genus of *Nassodonta* comprises of three accepted species such as *Nassodonta dorri* (Wattebled, 1886), *Nassodonta insignis* (H. Adams, 1867) and *Nassodonta annesleyi* (Strong *et al.*, 2017). *Nassodonta* species live in brackish water environments, and they are reported from India and Vietnam, with an unconfirmed report from China (Smith, 1895). Previously, *Nassodonta gravelyi* (Preston, 1916) described from the Cochin backwaters was synonymised as *N. insignis* by Cernohorsky (1984). The recent study by Strong *et al.* (2017) again transferred them and synonymised as *N. annesleyi*. The original description of *N. annesleyi* was under the name *Clea annesleyi*, collected from “a tank between the sea and the canal which communicates with Cochin to the north of Quilon” in Kerala (Benson, 1861). Since both *N. insignis* and *N. annesleyi* have been reported from the brackish waters of Kerala, a comprehensive molecular study is recommended for a better understanding on the species.

The diverse species from various “other group” classes contributed to the overall benthic macrofaunal diversity of estuary. The most dominant community in the “other group” was hydroid colonies of *Obelia bidentata*. They are reported from the southern and middle zone of Vembanad Lake and known to form massive colonies in the benthic biotope (Asha, 2017). In addition to this, four species of benthic gobioid fishes were also collected, such as *Callogobius mannarensis*, *Trypauchen vagina*, *Psammogobius biocellatus,* and *Acentrogobius viridipunctatus*. They also function as benthic predators. The sporadic occurrences of barnacle (*Amphibalanus improvises*), ophiurid, nemertean and chironomid species were also noticed. Ophiurids are marine in origin and occasionally found in estuarine mouth region, it prefers high saline conditions, while chironomid larvae in the “other group” prefer fresh water zones.

V. 3. 2. Patterns of macrofaunal species assemblages

According to Ourives *et al.* (2011), the difference in richness, abundance and distribution of macrofauna mainly related to salinity gradient of the estuarine system. Similarly, the sediment parameters such as organic matter and fine sediments, along with the feeding type of benthic assemblages. Muddy substratum with the elevated level of clay content will also result in the lowest richness, diversity, and abundance of macrofauna (Fresi *et al.*, 1983). In the present study, many macrofaunal species found in the oxidised sandy sediment that exposed to relatively high salinity and turbidity. They were *Murex trapa*, *Callogobius mannarensis*, *Scylla serrata*, *Ophiothrix* sp., *Diogenes alias*, *Ficopomatus* sp., *Sphaeroma annandalei*, *Marcia recens*, *Saccostrea cucullata*, *Meretrix casta*, *Melita zeylanica*, *Cyathura indica* and *Grandidierella megnae*. However, some species like *Psammogobius biocellatus*, *Cuneocorbula cochinensis*, *Lucifer hanseni*, *Nephtys polybranchia*, *Paraprionospio* sp., *Phyllodoce* sp., *Pseudopolydora kempi*, *Obelia bidentata*, *Glycinde bonhourei*, *Owenia fusiformis*, *Hiatula* sp., *Diopatra neapolitana*, *Villorita cyprinoides* and *Nassodonta insignis* were dominant in stations having muddy sediment with high bottom Chl-*a* content. Some of the species strongly linked with oxygenated clayey-silt substratum that enriched with organic matter. They were *Pagurapseudopsis gymnophobia*, *Ctenapseudes chilkensis*, *Ophelia* *capensis*, *Namalycastis indica*, *Prionospio polybranchiata*, *Psammogammarus* sp., *Synidotea variegata*, *Iphinoe* sp., *Dosinia* sp., *Victoriopisa chilkensis*, *Dendronereides heteropoda*, *Dendronereis arborifera*, *Alpheus malabaricus,* and *Neanthes glandicincta* with some species of nemertean and chironomids. Clayey silt content has stimulated the distribution of *Heteromastides bifidus*, *Heteromastus similis*, *Acentrogobius viridipunctatus*, *Nephtys oligobranchia*, *Miyakella nepa*, *Jassa falcata*, *Glycera alba*, *Capitella* sp., *Pista indica*, *Ninoe notocirrata*, *Prionospio cirrifera*, *Dendronereis aestuarina*, *Metapenaeus affinis* and *Ceratonereis costae* with the high organic matter supplied by the riverine discharge. The distribution of gastropod *N. insignis* populations preferred organically enriched sandy-silt sediment.

Macrofaunal assemblages in the estuary significantly separated into two groups based on the spatial distribution pattern and species composition. Changes in the distribution of *Americorophium triaeonyx*, *C. fluviatilis*, *O. bidentata*, *A. senhousia*, *P. cirrifera*, *M. recens*, *G. megnae*, *Capitella* sp., *Notomastus* sp., *V. cyprinoides*, *C. costae* and *C. indica* were responsible for the observed grouping of macrofaunal assemblages. The high dominance of peracarid amphipod *A. triaeonyx* and isopod *C. fluviatilis* in the estuary can, therefore, be likely associated with a broader variety of benthic habitats of heterogenous sediment in the STA group, than comparatively homogenous fine sediment in the STB group (Cattrijsse *et al.*, 1993; Deyzel, 2012). Relatively high species richness, diversity, and evenness were noticed in the group STB as compared to STA, it was mainly due to the dominance of *A. triaeonyx* in STA. These benthopelagic representatives of amphipods and isopods are predominantly hyperbenthic by nature, occasionally migrating to surface waters at night. *A. triaeonyx* display high mobility and opportunistic behavior. They quickly colonise in areas that exposed to high siltation and salinity fluctuations (Bryazgin, 1997; Robertson and Stevens, 2010). Similarly, many of other amphipod species are also exhibit high ecological tolerances to different environmental conditions and wide distributional pattern along with niche specificity (Aravind *et al.*, 2007; Chintiroglou *et al.*, 2004; Shyamasundari, 1973). Their distribution in the estuarine sediments are related to abiotic factors like salinity, hydrodynamic, sediment characteristics, food availability, aquatic vegetation, organic matter, water depth and water circulation pattern and biotic factors such as competition and predation (Brandt *et al.*, 1999; De Grave, 1999). According to Semeniuk (2000), seasonality in malacostracan population density and distribution are linked to adult migration and juvenile recruitment.

The population of *A. triaeonyx* in the KAE reached the maximum during monsoon season, and further, it declines with other population of amphipod, *Victoriopisa chilkensis*. The various studies on *A. triaeonyx* revealed that they survive in the vast salinity range of 0.1 to 27.7 ppt and temperature range of 26.9 to 30.9°C (Shyamasundari, 1973). Their high salinity tolerance level and capacity of the pelagic and benthic mode of life support them for being an opportunistic species in the estuary (Shyamasundari, 1973). The amphipod *V. chilkensis* is known to tolerate high organic content, and their population size is stimulated by organic matter enriched in sediment (Aravind *et al.*, 2007). According to Nair *et al.* (1983a), *V. chilkensis* survive in brackish water condition within the salinity range of 0.10 to 19.7 ppt. They consume a wide variety of plant and animal material in addition to scavenging dead and decaying organisms and even exhibit cannibalistic behaviour. According to Aravind *et al.* (2007), in the presence of *A. triaeonyx* population, the number of *V. chilkensis* decreased drastically due to resource partitioning and intraspecific competition.

The other dominant species in the estuary was cirolanid isopod, *Cirolana fluviatilis* (7.54 %), they are voracious scavengers, reported to being infesting in fishes and shrimps caught by gill net in the Cochin backwaters (Mathew *et al.*, 1994). Similarly, they caused high mortalities in fish cages of Asian seabass, *Lates calcarifer* in the study area (Sanil *et al.*, 2009). According to Newman *et al.* (2009), these euryhaline species demonstrated good survival rate in salinity between 7 to 35 ppt in laboratory condition. They also prefer to live in the muddy sediment of estuaries (Poore and Bruce, 2012). In general, benthic isopods are a significant contributor to biodiversity and biotic resources of malacostracan fauna in the estuaries, and they are actively involved in the recycling of organic matter and form food for many aquatic species (Lopez *et al.*, 2012).

The burrow-dwelling amphipod in the estuary *Corophium volutator* act as bioturbator in sediment and play the significant role in the nutrient cycle. They are common in the benthic biotope of Cochin backwater and significantly contributing to the numerical density of malacostracan crustaceans (Geetha and Bijoy Nandan, 2014; Pelegrí and Blackburn, 1994). Aravind *et al.* (2007) state that, brackish-water amphipod in the study area, *Melita zeylanica* breeds throughout the year in Indian estuaries, so they are capable of surviving in varying salinity conditions. Other malacostracans such as tanaidacean (*Ctenapseudes chilkensis*) and cumacean (*Iphinoe* sp.) species in the estuary are also excellent micro scavengers, which proliferate in abundance by consuming detritus and other minute food particles (Brandt *et al.*, 1999; Gambi *et al.*, 1992; Priya, 2015). Cumaceans live partially or entirely buried in the sediment whose grains lie within a narrow size range, and many cumacean species are known to migrate into the plankton community especially at night (Watling, 1979). However, sediment texture is the primary factor affecting the abundance of tanaids (tanaidaceans) and environmental factors such as temperature, currents, and freshwater discharge are also responsible for their density variation (Ates *et al.,* 2014). Juveniles of commercially valuable species like *Penaeus indicus* also contributed to macrofauna. They are unique euryhaline species that breed offshore, while post-larvae and juveniles inhabit in the estuarine environment (Kuttyama, 1973). They are tolerant to temperature ranges from 18 to 34.5 °C, and salinities that range between 5 to 50 ppt with an optimal salinity of 10 to 15 ppt for juvenile Garcia *et al.* (1981). The studies suggest that development stages of tanaid (tanaidaceans) species *C. chilkensis* and amphipod *V. chilkensis* in the estuary form food for different organisms like *Penaeus indicus* and *Metapenaeus dobsoni* (Vengayil *et al.*, 1988). The cryptogenic hydrozoan species, *Obelia bidentata* colonies also significantly contributed the ecological process of benthic communities. They are distributed from shallow estuaries to 200 m coastal marine environments, attached to hard substratum like wood, shells, algae, sandy bottoms (MarLIN, 2017).

The Asian nest mussel *Arcuatula senhousia* is considered as an invasive species with a wide range of distribution around the globe and are capable of surviving in a wide range of salinity, dissolved oxygen, and temperature (Sreedhar, 1991). Their distribution in the estuary was mainly ensued by clay sediment with organic matter. They form a nest or bag-shaped structures in the soft bottom habits by secreting fibrous threads that attach to sediment particles by aggregation of individuals that range from 2,500 to 126,000 per square meter (Crooks, 2001). This complex biogenic structure provides shelter for various benthic organisms like amphipods, tanaids, small snails and polychaete worms. Consequently, the other suspension-feeding communities would be declined due to intraspecific completion and resource partitioning (Crooks, 1998; Mistri, 2002). The venerid clam, *Marcia recens* in the estuary was also recorded in the sandy substratum of various estuaries of India (Pati and Panigrahy, 2013).

The black clam, *Villorita cyprinoides* (Cyrenidae) is the third dominant molluscan population in KAE. Their distribution is restricted to the brackish water environments of southern India (Madhyastha, 2011). They contributed significantly to the ecology and economy of Vembanad Lake by forming more than 70 percent of the total shellfish production. These species are tolerant to a broad range of salinity from 3 to 16 ppt and dissolved oxygen content of 2.83 to 6.5 ml L^{-1} (Laximilatha *et al.*, 2005; Suja and Mohamed, 2010). The venerid clam, *Meretrix casta* (7.31 %) also contributed significantly to molluscan density. According to Modassir (1990), high mortality of *M. casta* was observed due to increased siltation and sudden salinity variation in the estuaries. The brackish water species of oyster, *Crassostrea bilineata*, and *Saccostrea cucullata* are provided good ecosystems services by filtration of the turbid estuarine water column. However, salinity variation in the estuary has a significant role in their population structure (Nair *et al.*, 1984b). Their juveniles were also contributed to the macrofaunal diversity of estuary. The abundance of *Nassodonta insignis* population in the estuary was controlled by organic matter, and they found with black clam, *V. cyprinoides* population.

The distribution of polychaete communities has been mainly controlled by factors such as sediment type, salinity regimes, historical disturbances, organic content, microbial associations and food availability. These variables are significantly influence the total number of species and individuals present as well as the species composition (Ajmal Khan *et al.*, 2004; Ajmal Khan and Murugesan, 2005; Ansari *et al.*, 1986; Bijoy Nandan and Abdul Azis, 1995a; Butman, 1987; Crimaldi *et al.*, 2002;; Giangrande *et al.*, 2005; Hutchings, 1998). The many researchers around the globe suggest that dominant polychaete families in the KAE, capitellids, and spionids polychaetes are indicators of the stressed environment (Heip and Herman, 1995). Cabral-Oliveira and Pardal (2016) observed the high abundance and assemblage of capitellid and spionid species in the sewage discharge site. The polychaete species *Parheteromastus tenuis* was also observed in the area of sewage pollution (Das *et al.*, 2009). Moreover, they are good deposit feeders so considered as indicators of organic pollution (Fauchald, 1984; Fauchald and Jumars, 1979; Kristensen, 1988; Rouse and Fauchald, 1997). Musale and Desai (2011) also stated that the polychaete species *Prionospio* sp., *Capitella* sp., *Mediomastus* sp., and *Cossura* sp. are deposit feeders and indicators of organic pollution. According to Elias *et al.* (2006), such opportunistic pollution tolerant species show high numerical abundance in the disturbed areas due to the accessibility of a significant amount of organic matter as a food source.

Sivadas *et al.* (2010) observed the dominance of opportunistic deposit feeding species like *Prionospio* sp., *Magelona* sp., *Tharyx* sp. and *Cossura* sp. in the ecologically disturbed marine environment on the west coast of India. Similarly, *P. cirrifera* and *Ceratonereis* sp. were recorded from the highly reduced sediment of retting zones of Kadinamkulam estuary (Bijoy Nandan and Abdul Azis, 1995a). Studies also suggest that the concentration of clay fraction controls the polychaete species, such as *P. cirrifera*, *Heteromastus similis*, *Indonereis gopalai*, *Nephthys oligobranchia*, *Neanthes willeyi* and *Nectoneanthes ijimai*. Some polychaetes prefer silty substratum with organic matter, especially *Prionospio krusadensis*, *Dendronereis arborifera*, *Polydora kempi*, *Prionispio* sp., *P. saldanha*, *Glycera tesselata*, *G. alba* and *H. similis*. In Cochin estuary, Pillai (1977) observed the seasonal increase in abundance of *Prionospio* sp., *P. polybranchiata* and *Parheteromastus tenuis* due to its annual recruitment. Their population size increased during pre-monsoon and considerably decreased at monsoon period. The high rainfall and freshwater runoff during monsoon modifies substratum type in estuaries and result in recruitment failure owing to disturbed larval settlement in suitable substratum (Prabhadevi, 1994). However, capitellid worm, *Mediomastus* species was dominant in a moderately disturbed environment (Boudreau et al., 1991; Rivero *et al.*, 2005; Schwinghamer, 1983).

The series of studies in the backwater of Kerala recorded the dominance of *Dendronereis aestuarina* and *D. arborifera* at riverine zones with a good quantity of organic matter deposits (Jayachandran *et al.*, 2015). According to Sheeba (2000), an assemblage of *D. aestuarina* and *Namalycastis indica* are depended on the dissolved oxygen concentration and particulate organic carbon. These species found in the littoral or supralittoral zones in association with decaying organic materials in areas on or close to the shore. They also adapted to live in the semi-terrestrial habitat with low saline condition (Magesh *et al.*, 2012). According to Desrina *et al.* (2013), the abundant genus in the KAE, *Dendronereis* species act as a propagative carrier for white spot syndrome virus (WSSV), the causative agent of white spot disease (WSD) in penaeid shrimps of traditional ponds in Indonesia. Opportunistic tube dwelling species such as *Diopatra neapolitana* in the sediment play a crucial role in the distribution of other sub-surface feeding species by changing the sediment properties (Gaston, 1987). Their tube structure provides shelter from predation and supports diverse invertebrate communities around them by forming a stable environment. Sarkar *et al.* (2005) observed the formation of mudflats in the Sundarban Biosphere by continuous siltation in the 'Diopatra' zone. In the present observation, species diversity and richness of polychaetes was high during pre-monsoon season, while decreased in monsoon period. In spatial scale, the richness of polychaetes was highest in sandy sediment with low sediment organic matter and high dominance in sandy sediment with moderate organic matter content.

The climate change effects, especially sea level variation altered the salinity gradient in estuaries and ultimately influence macrofaunal assemblage pattern (Fujii, 2012). The stenohaline macrofaunal communities are unable to tolerate such variability in salinity. Euryhaline marine organisms adapted to wide fluctuations in salinity, and they found at varying distances along the tidal gradient of an estuary or close to areas of freshwater input. Brackish water organisms, considered as real estuarine organisms, can be found within the mid-tidal zone of an estuary and can tolerate neither marine waters nor freshwater (Nybakken & Bertness, 2005). Some species show the upward shift of communities in the estuaries and such invading estuarine species in the freshwater habitat regularly appeared severe physiological alterations in the low saline environments Dietz *et al.* (1996). In the present study, the riverine region was dominated with *D. aestuarina*, *Notomastus* sp., *Capitella* sp., *Nassodonta insignis*, *Prionospio cirrifera*, *Americorophium triaeonyx*, *Villorita cyprinoides* and *Glycera tridactyla*. The polychaete worm, *D. aestuarina* exhibited a massive reproductive swarming and mass mortality in the riverine environment of the estuary (Jayachandran *et al.*, 2015). Estuarine organisms are also able to evade temporarily unfavourable salinity fluctuations by burying within the sediment or by movement with tidal inundation, making it difficult to assign tolerance levels or estimate distribution patterns based on salinity. The amphipod species like *A. triaeonyx* are well adapted to salinity fluctuations in the estuary with moderate sedimentation. They also exhibit the benthopelagic mode of life. The amphipod species like *Corophium volutator* is burrow-dwelling forms that can avoid temporary fluctuations in salinity by hiding inside the burrow. The amphipod *Melita zeylanica* even breeds throughout the year in the estuaries irrespective of salinity fluctuations. However, the abrupt changes in salinity cannot tolerate the majority of estuarine species. Pearson and Rosenberg (1978) noted that identification of species occurring in areas which are subject to varying salinity and which is experiencing pollution especially organic enrichment is possible, as they frequently differ from species exposed to contamination in the purely marine environment. According to Glasby *et al.* (2003), the genus *Namalycastis* found in polluted coastal regions is capable of surviving in low salinity conditions and this genus is distributed from freshwater to marine waters (Glasby *et al.*, 2003). The estuarine species of gastropod *Nassodonta insignis* exhibited broad salinity tolerance towards low salinity gradient in the estuary. Many studies related to climate change and benthic interactions and state that variability in salinity significantly increases the osmotic balance of benthic communities (Birchenough *et al.*, 2015). According to Rao *et al.* (2009), polychaete communities represented by *Magelona cincta*, *Prionospio cirrifera*, *Cossura* sp., *Sigambra parva* and *Glycera longipinnis* greatly influenced by salinity fluctuations, oxygen depletion and accumulation of toxic materials owing to organic enrichment in the sediment column of Godavari estuary. According to (Musale *et al.*, 2015) the *Cirratulus* sp. and *Cossura coasta* are common and dominant in sandy-silt substratum with the moderate organic matter. At the same time, the pollution tolerant species of *Cirratulus* was more in the clayey and sandy substratum with the high organic matter. Sandy substratum provides more interstitial space for polychaetes to move and hide (Musale and Desai, 2011). The distribution of polychaete species like *Ceratonereis erythraeensis*, *Nereis lamellosa,* and *Glycinde oligodon* linked with sand and temperature of the benthic environment.

V. 3. 3. Functional feeding groups of macrofauna

The feeding guild of benthic communities are assigned based on their feeding characteristics and such grouping of taxa by a particular function is useful for addressing ecosystem level questions (Pagliosa, 2005). It enables transfer of information on the food source, food type, and feeding mode of a particular community to simplified taxonomic data. It is useful for addressing questions regarding carbon flow in the estuarine environment (Word, 1978). In the present study, suspension feeding communities were dominated in the estuarine system. It was mainly dominated with amphipod *Americorophium triaeonyx*, mussel *Arcuatula senhousia*, and hydrozoan *Obelia bidentata*. They are usually abundant in an area with low biological oxygen demand (BOD). Suspensions feeders are ecologically relevant organisms that provide various ecosystem services like water column filtration and removal of turbidity. However, the population of surface deposit feeders (*Prionospio cirrifera*), sub-surface deposit feeder (*Capitella* sp.) and carnivores (*Cirolana fluviatilis*) dominated in some zones of the estuary depending on the suitable substrate type and other ecological factors. Deposit feeders are more abundant in sediment with the high organic matter. They are taking part in the biomineralisation process of nutrients. The burrowing population of isopod, *Sphaeroma annandalei* was also observed in the benthic samples (SRL) and they are potentially harmful to wooden structures like conventional fishing boats.

Many species of suspension feeding communities like molluscs and malacostracan crustaceans are less tolerant to degraded environmental conditions, and they avoid such environments linked to eutrophication process (Cartes *et al.*, 2003; Cunha *et al.*, 1999; Dauvin, 2008). In a spatial scale, sandy sediment environments (STA) dominated by suspension feeders, while sub-surface deposit feeders (SSDF) were abundant in muddy sediments (STB). Sessile suspension feeders like bivalves were abundant in middle zones of the estuary. Their increased distribution attributed to the availability of suitable substratum type and a good source of food materials. However, consistent siltation and sudden salinity fluctuations induced by heavy river discharge period seriously affected the population. The massive amount of suspended silt and clay in water column found to be detrimental to suspension feeding populations like bivalves. It affects filtration mechanism and interferes the particle selection (Gattuso *et al.*, 1998; Granek *et al.*, 2010; Thrush *et al.*, 2004). Similarly, suspension feeder like *O. bidentata* colonised in sandy substratum with a clear water column. Surface deposit feeders like *Nassodonta insignis* found tolerant to low saline conditions in the organically enriched muddy and sandy substrates. Suspension feeders (SF) dominated in the estuary with maximum population intensity during monsoon. However, other dominant communities like SSDF increased their relative abundance during pre-monsoon. It could be due to the formation of comparatively stable benthic biotope in that period with increased organic matter deposit supplemented by heavy river discharge during the monsoon. BOD levels in the water column found to increase with suspended particles or surficial detritus loading in the estuary. However many of these surface deposit feeders (SDF) and sub-surface deposit feeders can survive within such increased BOD levels. Species which are tolerant to such high level of BOD are considered as an indicator species of pollution such as species of polychaetes and oligochaetes. Thus, the assemblage of SDF increased in proximity to wastewater discharge points and sub-surface deposit feeders that dominated at the site with high organic enrichment. They were also responding to high concentrations of hydrogen sulphide other than high organic matter, which is an unacceptable condition for other species (Word, 1978). Ecologically stable environments support a more diverse range of feeding groups than disturbed conditions (Word, 1978). Similarly, benthic biotope with heterogeneous sediment supports diverse communities that compared to homogenous substrate type (Pearson and Rosenberg, 1978). In the present study, SDF polychaetes dominated in the estuary with other groups of SSDF. Disturbance of the sediment by active burrower and deposit feeders, which ingest sediment and cause high volumes of turnover of the sediment, may inhibit or promote colonisation by other animals (Braeckman *et al.*, 2011; Peterson, 1991). They are considered ecosystem engineers because they play a vital role in the maintenance of regional bio-diversity by adding heterogeneity to the structure of benthic habitats (Levinton, 1995a; Patel and Desai, 2009; Stephen *et al.*, 2004)

In KAE, the most diverse feeding guild was exhibited by polychaete assemblages, and they depicted an apparent variation on the seasonal pattern. They possess different living strategies to adapt to various habitats such as significant variations in morphology, several feeding and reproductive modes (Jayachandran *et al.*, 2015). The pre-monsoon season was dominated by SDF and SSDF polychaetes species, while SF polychaetes dominated the population for other periods (SSA). In KAE, sandy sediment (STA) substratum dominated with SF, while muddy sediment (STB) controlled by SDF and SSDF communities. In Cochin estuary, among the polychaete feeding guild, SSDF group dominated with SDF and CVR (Geetha et al., 2015). However, the present study depicted the least contribution from SF communities to entire macrofaunal polychaete feeding guild in the STB group of stations, while SF and SDF communities dominated in sandy sediments of STA group. The opportunistic species can change their feeding mode depending on prevailing environmental conditions that which results in higher abundance and dominance of such species (Maurer *et al.*, 1999). In a gradient of organic matter enrichment in the estuarine sediments, high levels of organic matter result in a shift from suspension feeding to deposit feeders communities along with some carnivorous communities (Pearson and Rosenberg, 1978). However, the further increase in the eutrophication process also leads to the complete removal of suspension feeding communities (Lenihan, 1999). In such conditions, some of the species were found unaffected over a wide geographical range of polluted systems, and they are regarded as an indicator species (Pearson and Rosenberg, 1978). In organically enriched estuarine sediment, SSDF like capitellid polychaetes commonly seen, with some spionid polychaetes species like *Prionospio cirrifera*. They considered as indicators of organic pollution.

The distinct number of macrofaunal species in estuarine environment possesses characteristics of indicator species for the anthropogenic disturbances. They are continuously exposed to contaminations and the diverse responses relative to the invertebrates tolerances, feeding modes and trophic interactions (Pearson and Rosenberg, 1978). In addition, macrofaunal species are sedentary or sessile, and they cannot evade adverse conditions from overlying water mass. So they are getting chronic exposure to unfavourable conditions in their short lifespan which enables them a rapid response to environmental variations that may be otherwise imperceptible (Word, 1978). Among other macrofaunal communities, polychaetes exhibit the maximum tolerance level to ecosystem perturbations. They are the most tolerant taxa in oxygen-depleted conditions than other macrofaunal suspension feeding taxa like bivalves and malacostracan crustaceans (Diaz and Rosenberg, 1995). According to Sánchez-Moyano and García-Asencio (2010) massive polychaete reefs considered as an indicator to increase sewage contamination and chlorination process. In the present study, the increased assemblage of tube-dwelling polychaete like *D. neapolitana* was noticed in the organically enriched sediment. According to Bailey-Brock (1984), the packing of sediments provides substratum strength for tube dwellers and burrower, and the high organic matter of the trapped material serves as a food source for selective and burrowing detritivores. However, the structurally uniform muddy sediments of increased organic matter promote accumulation of toxic metals and depletion of oxygen content and ultimately results in the exclusion of species form that area (John *et al.*, 2002). In many studies, maximum polychaete diversity recorded in the sediment with moderate organic carbon content (Sivadas *et al.*, 2011). However, low abundance of polychaetes was seen in an area has a high level of silt and clay content that enriched by >3 % organic matter. It is attributed to the avoidance of polychaetes to high abundant of organic matter and suboxic levels in the sediment (Harkantra *et al.*, 1982). Similarly, the reduction of polychaete diversity and increased abundance of deposit feeders indicates the deterioration of estuarine ecosystem health (Geetha *et al.*, 2015).

Many of the polychaete species are lived in the stressed environments through their inherent ability to adapt to environmental changes (Ansari *et al.*, 1986). These opportunistic species dominate the polluted environments. They referred to as pollution indicator species (Pearson and Rosenberg, 1978; Rygg, 1985). Such unhealthy benthic habitats are also characterised by reduced macrofaunal species diversity, abundance and biomass with increased dominance of small-bodied pollution tolerant species (Albayrak *et al.*, 2006; Dauer, 1997). All the STB group of stations dominated by opportunistic small-bodied polychaetes like *P. cirrifera* and *Capitella* sp. The present study suggested that the conservative species in the estuary was largely replaced by opportunistic species. They are small-bodied organisms with short generation time. They dominated the benthic biotope which resulted in relatively low species diversity (Warwick, 1986). The aggregation of tube-dwelling polychaetes like *D. neapolitana* and *Pista indica* in the estuary support number of opportunistic species around their colonies. They provide microhabitats through the accumulation of organic matter and stabilizing sediment with a mat like structures (McCave, 1976). Studies suggest that use of ecological communities as whole to characterise the degree of pollution found to be much more effective than single species approach.

Chapter 6
Ecological status of Kodungallur-Azhikode estuary

VI. 1. Introduction

Marine systems contribute more than 63 percentages to the total global ecosystem services provided by earth, in which a significant contribution is from coastal marine systems (Costanza *et al.*, 1997). However, this crucial habitat is also being lost by 2 to 3 times faster than those in tropical forests, which are exhibiting the similar rate primary production (Diaz and Rosenberg, 2008; Lotze *et al.*, 2006). The recent studies also suggest that human activities threaten all the marine systems and among them, 41 percentages are being affected by multiple anthropogenic drivers, especially in nearshore habitats like estuaries and intertidal areas (Halpern *et al.*, 2008). The estimated coastal population within 100 km of shoreline and 100 meters of sea level is around 1.2×10^9 people with average densities nearly three times higher than that of global average density (Small and Nicholls, 2003; Stauber *et al.*, 2016). Consequently, almost all coastal systems are influenced in some way by anthropogenic activities. Which will likely to become more intense and acute during the next 25 years, because the coastal population is expected to approach six billion people by 2025 (Kennish, 2002).

The primary impact drivers on these coastal marine systems are land reclamation, dredging, over-exploitation of resources, unmanaged tourism, pollution, the introduction of invasive species and climate change due to human interventions (Bijoy Nandan, 2008; Costanza, 1999; Gray, 1997; Halpern *et al.*, 2008). These threats directly or indirectly lead to reduced biodiversity, alteration of biotic community structure, massive mortalities, imbalanced food webs, declined harvestable fisheries, loss of vital habitat and ultimately depreciate ecosystem resilience (Cardoso *et al.*, 2004; Dolbeth *et al.*, 2005; Griffiths *et al.*, 2017; Kennish and Townsend, 2007). Therefore, understanding of changes in these critical habitat forms a foundation for proper management.

Assessments of ecological state in the ecosystems are evolved from the traditional monitoring methods focused on physicochemical characters of water quality or determining the specific pollutants with the sign of contamination in aquatic systems (Karr, 1991). Further studies realised that analysis of sediment chemistry tend to be more conservative than the water quality monitoring and found that variability in the sediment chemistry has high inferences in the biogeochemical processes of the particular ecosystem. However, sediment chemistry only provides the information on contaminants, which will not give any knowledge of its effect in an ecosystem (Chapman, 2007). Therefore, monitoring of biological communities can be an integral part of many modern assessment techniques owing to their capability to integrate multiple stress factors and response to an unusual change in the ecosystem (Burton Jr, 2002). Since the use of environmental indicators become a routine method in monitoring programme, the discussion over the critical properties of the ecological indicator has been widely revisited, developed and inter-calibrated (Borja and Tunberg, 2011; Dauvin and Ruellet, 2007; De-La-Ossa-Carretero and Dauvin, 2010; Muxika *et al.*, 2007).

Many ecological indices are developed for assessing the aquatic ecosystem health, and among them, indicator species based indices are common (Johnson, 2008; Pinto *et al.*, 2009). Such indices are using information on the sensitivity of community or focusing on species composition with their presence, absence or dominance pattern data (Diaz *et al.*, 2004). While, ecological strategies based indices concentrate on environmental stress effects on environmental strategy such as different feeding modes, functional group or behaviour of different taxonomic groups (Dauvin and Ruellet, 2007; Worm *et al.*, 2006). The diversity based indices are widely used concepts in pollution monitoring, along with other popular index that measure species richness, species abundance, the proportional abundance of different species and dominance indices (Clarke and Gorley, 2006). Many other indices are using taxonomic, numerical, ecological, genetic and phylogenetic aspects of diversity (Clarke and Gorley, 2006). Indices are also developed based on species biomass and abundance approaches where the centre on the energy variation in the ecosystem accounts for the change of organism's biomass and abundance as a measure of environmental disturbances (Clarke and Gorley, 2006). The multimetric indices attempt to integrate information regarding different aspects of the ecosystem, incorporating metrics that span ecological levels from an individual through the population to community, ecosystem and landscape (Karr, 1991). Sometime multimetric indices also include data on physicochemical factors, diversity measures, specific richness, taxonomic composition and the system's trophic structure (Muxika *et al.*, 2007; Vollenweider *et al.*, 1998). Several biotic indices are developed to assess benthic community concerning regional reference conditions. Many of them have proved to be reliable and sensitive indicators to evaluate the marine and estuarine ecosystem health (Borja *et al.*, 2000; Borja and Muxika, 2005; Dauvin and Ruellet, 2007; Muxika *et al.*, 2007; Simboura and Zenetos, 2002).

The present study focused on four widely used benthic biotic indices such as the Benthic Opportunistic Polychaetes Amphipods [BOPA] Index (Dauvin and Ruellet, 2007), AZTI's Marine Biotic Index [AMBI] (Borja *et al.*, 2000), multivariate-AMBI [M-AMBI] (Muxika *et al.*, 2007) and BENTIX (Simboura and Zenetos, 2002) index to translate benthic community composition into an environmental quality classification in the estuary. Abundance biomass curves (ABC) were also plotted to understand the general nature of benthic communities in the estuary (Warwick, 1986; Clarke and Warwick, 2001a; Lambshead et al., 1983). Based on these indices, this chapter discusses the overview of the benthic ecosystem health in the Kodungallur-Azhikode estuary [KAE] to provide an "action plan" to protect and enhance the ecosystem quality.

VI. 2. Results

VI. 2. 1. Abundance biomass comparison (ABC) curves

The abundance biomass curves (ABC) for Kodungallure Azhikode estuary was plotted for each sampling sites separately as well combined for the entire estuary [Figure 55 a-f]. In ABC plot, W-values can be ranges between -1 to +1 for macrofaunal assemblages.

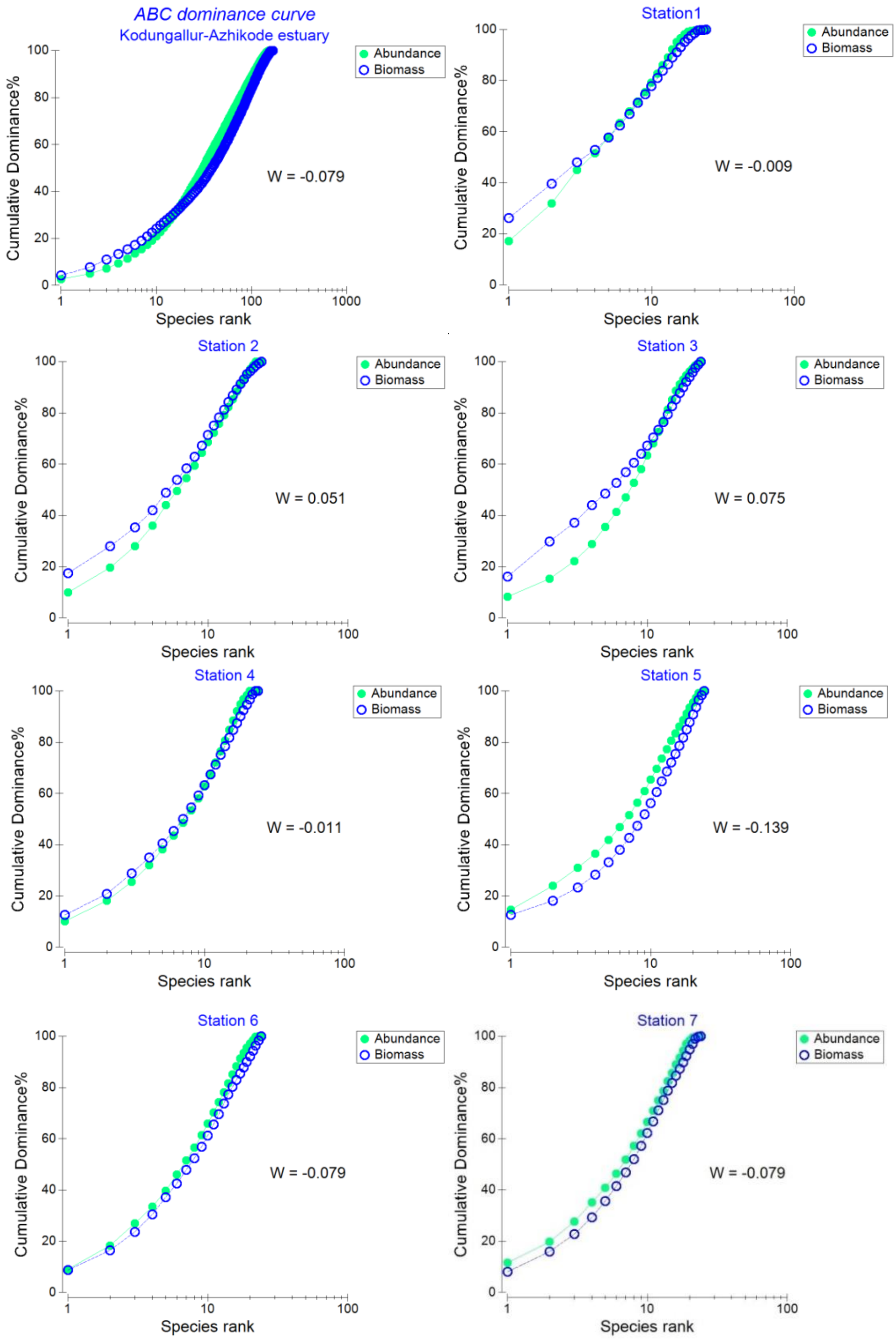


Figure 55 a-f. (a) Abundance biomass comparison (ABC) curves of macrofaunal assemblage in Kodungallur-Azhikode estuary and (b-f) each station plotted separately. The W statistic is positive when abundance curve remains below biomass curve and is negative when the two curves cross

When the biomass curve lies above the abundance curve of ABC plot, that will give positive values which indicate undisturbed benthic communities with the dominance of K-selected species. Comparatively good ecological conditions were noted in the middle zones of the estuary with positive W values, such as station three (W=0.075) and station two (W=0.051). When communities are characterised as moderately stressed, the abundance and biomass curves become very close or intersect, represented by W-values close to zero. Such conditions were observed in the station one (W=-0.009), station four (W=-0.011), station six (W=-0.079) and station seven (W=-0.079). In case of negative W-values, abundance curve lies above the biomass curve which depicts the disturbed benthic community status with the dominance of r-selected species. In the present study, the relatively high negative trend was observed in the station five (W= -0.139) which indicates the potential for high ecological disturbance. However, the W-value for the entire estuary was -0.079 that indicates the moderate disturbance in the estuary when considering the entire ecosystem as a whole. All the above analyses were performed by using statistical package PRIMER 6.1.13 (Clarke and Gorley, 2001).

VI. 2. 2. Marine biotic indices

a) Benthic Opportunistic Polychaetes Amphipods Index (BOPA)

The benthic opportunistic polychaetes amphipods index (BOPA) ranges from 0 to 0.30103 and the relatively low values representing a better ecosystem health. The mean BOPA index for the Kodungallur-Azhikode estuary during the present study (2009-2011) was 0.10880 ± 0.10893. Annual mean showed the higher value of 0.11544 ± 0.11495 for the second year (2010-2011) when compared to the value of 0.10217 ± 0.10282 in the first year (2009-2010). Seasonally, BOPA was relatively higher during the monsoon of the first year (0.12882 ± 0.11731) followed by post-monsoon (0.12011 ± 0.12426), pre-monsoon (0.11318 ± 0.11042), monsoon of the second year (0.11304 ± 0.11379), pre-monsoon (0.09264 ± 0.09555) and post-monsoon in the first year (0.08504 ± 0.09206). Spatially, STA group of stations depicted the relatively high BOPA values such as station six (0.18529 ± 0.09007), station five (0.16571 ± 0.09855), station seven (0.14758 ± 0.11508) and station two (0.13592 ±

0.11349), that compared to STB group station one (0.06083 ± 0.09239), station four (0.03475 ± 0.05995) and station three (0.03154 ± 0.06248). The index values of STA sites indicated good to moderate ecological condition while those in the STB sites ranged from a moderate to poor condition. Among all the sampling locations, station six, five, seven and two exhibited relatively high BOPA values, and the values even reached > 0.30 indicating the poor ecological conditions at these sites [Figure 56 & Table 12 a-c].

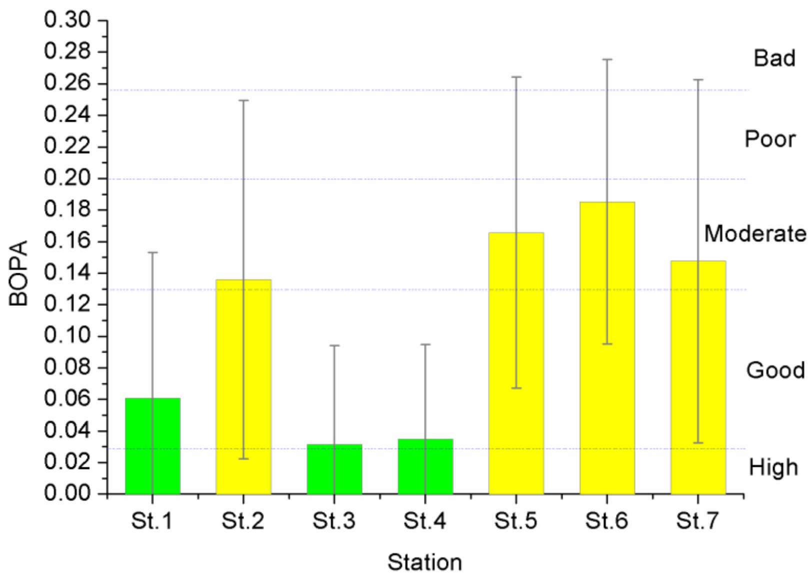


Figure 56. Benthic Opportunistic Polychaetes Amphipods Index (BOPA) showing the mean value of each station in KAE during 2009-2011, the BOPA index values 0.30103 - 0.25512 indicate bad ecological status while between 0.25512 - 0.19884 poor, 0.19884 - 0.13002 moderate, 0.13002 - 0.02452 good and <0.02452 indicate the high ecological status of soft bottom macrobenthic communities (modified from Dauvin and Ruellet, 2007; De-La-Ossa-Carretero and Dauvin, 2010)

b) AZTI's Marine Biotic Index (AMBI)

The value of AZTI's Marine Biotic Index (AMBI) ranges from 0 to 7, with the lower value representing the better ecological health of an estuary. The mean AMBI index for KAE during the present study was 2.69 ± 1.23. Relatively high annual mean value for AMBI was depicted in the second year (2.88 ± 1.14) that compared to the first year (2.51 ± 1.30) period. Seasonally, AMBI index was relatively higher during post-monsoon of the second year (3.14 ± 1.19) followed by the monsoon of the first year (2.85 ± 1.18), pre-monsoon (2.80 ± 0.97) and monsoon of the second year (2.71 ± 1.24), post-monsoon (2.39 ± 1.45) and pre-monsoon of the first year (2.28 ± 1.22). On a spatial scale, AMBI values was higher in STB group of stations, such as station six (3.21 ± 1.17), station two (3.13 ± 1.33), station seven (3.09 ± 1.42) and station five (2.78 ± 1.22), when compared to STA group of stations like station three (2.33 ± 0.79), station one (2.26 ± 1.21), and station four (2.06 ± 0.92). The AMBI index values ranged from 3.3 to 4.3 that refer to the moderate condition of pollution of the water body. When values range from 4.3 to 5.5 is considered as poor, and > 5.5 indicates the bad ecological state of the water body. The index values of STA sites reported undisturbed to the slightly disturbed condition while those in the STB sites range from a slightly disturbed to moderately disturbed status. Among all the sampling locations, station six, two, seven, and five exhibited relatively high AMBI index indicating the poor environmental condition [Figure 57 & Table 12 a-c].

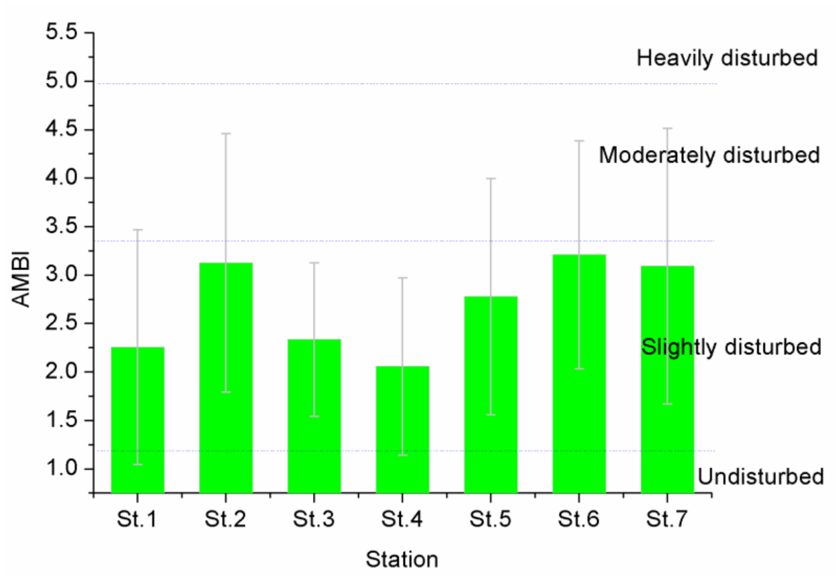


Figure 57. AZTI's Marine Biotic Index (AMBI) showing the mean value of each station in KAE during 2009-2011, the values between 0 to 1.2 represent the undisturbed condition and that for the slightly disturbed condition was 1.2 to 3.3, moderately disturbed ranged from 3.3 to 5, heavily disturbed was between 5 to 6 and extremely disturbed conditions were denoted by a value between 6 to 7 in AMBI index (Borja *et al.*, 2000)

c) Multivariate-AMBI (M-AMBI)

The value of M-AMBI (Muxika *et al.*, 2007) ranges from 0 to 1, the higher values representing better ecological health. The mean M-AMBI index for the Kodungallur-Azhikode estuary during the entire study was 0.4825 ± 0.1465. Annual variation was minimal during the study, and that was 0.4540 ± 0.1515 for the second year and 0.5107 ± 0.1364 for the first year. Temporarily, M-AMBI was relatively lower during the post-monsoon (0.4352 ± 0.1481), pre-monsoon (0.4629 ± 0.1470) and monsoon of the second year (0.4640 ± 0.1561) followed by the post-monsoon (0.4961 ± 0.1604), monsoon (0.5092 ± 0.1252) and pre-monsoon (0.5268 ± 0.1252) of the first year. Spatially, lowest M-AMBI values were recorded in the STB group of station two (0.3892 ± 0.1588), station seven (0.4291 ± 0.1760), station six (0.4551 ± 1106) and station five (0.4941 ± 0.1038), when compared to the STA group station four (0.6050 ± 0.1104), station three (0.5039 ± 0.12451) and station one (0.5003 ± 0.1395) [Figure 58 & Table 12 a-c].

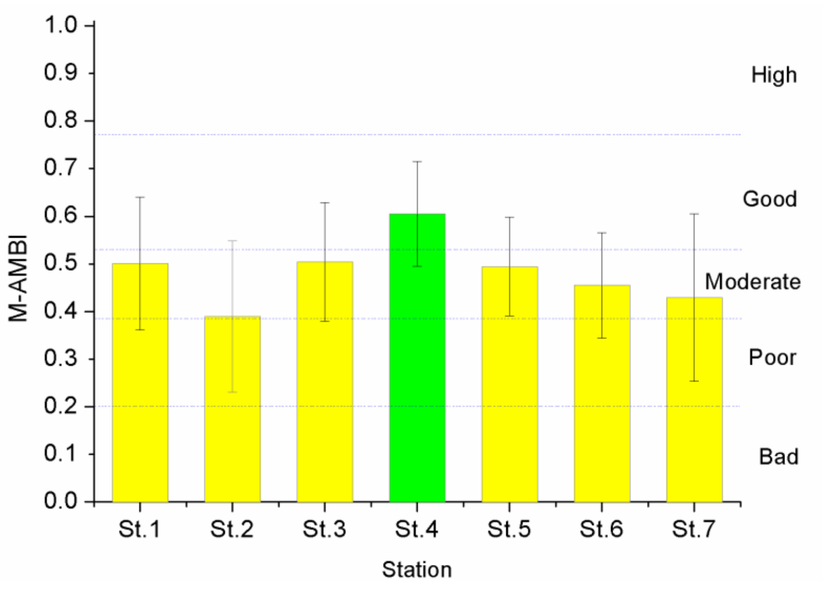


Figure 58. Multivariate-AMBI (M-AMBI) showing the mean value of each station in KAE during 2009-2011, The M-AMBI index values <0.20 indicate bad ecological status while between 0.20–0.38 poor, 0.38–0.53 moderate, 0.53–0.77 good and >0.77 indicate high ecological status of soft bottom macrobenthic communities (Muxika *et al.*, 2007).

d) BENTIX

The value of BENTIX index values are ranging from 0 to 6, with the higher value representing better ecological health. The mean BENTIX index for the Kodungallur - Azhikode estuary (KAE) was 3.62 ± 1.36. Annually, a comparatively low value was observed during the second year (3.38 ± 1.34) of study when compared to the first year (3.86 ± 1.35). Temporarily, BENTIX depicted a relatively low value during the post-monsoon of the second year (3.11 ± 1.46) followed by monsoon of the first year (3.29 ± 1.12), monsoon (3.44 ± 1.32) and pre-monsoon the second year (3.58 ± 1.22), post-monsoon (3.84 ± 1.57) and pre-monsoon (4.45 ± 1.10) of the first year. Spatially, BENTIX values were higher in the STA group station of stations such as station seven (3.24 ± 1.44), station two (3.27 ± 1.39), and station five (3.77 ± 1.28), and station six (3.40 ± 1.10), when compared to the STB group stations such as station one (3.82 ± 1.60), station three (3.84 ± 1.28) and station four (3.97 ± 1.35) [Figure 59 & Table 12 a-c].

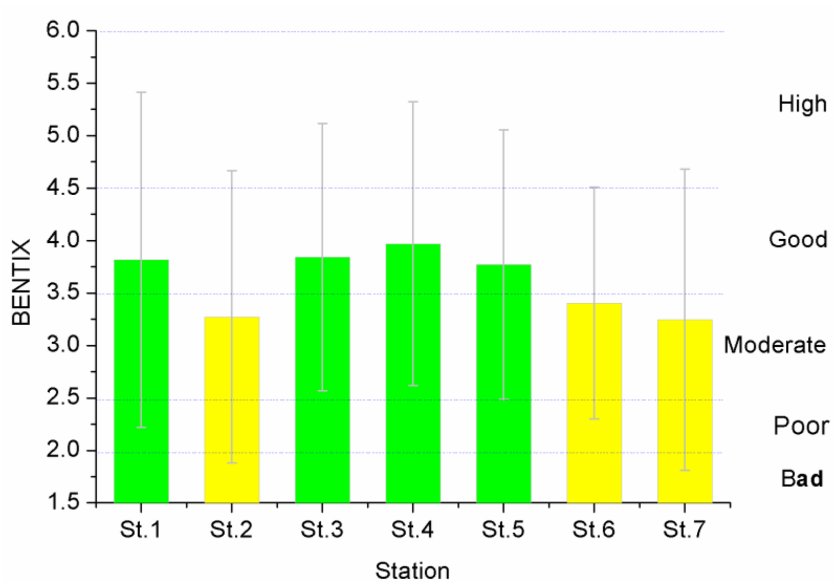


Figure 59. BENTIX showing the mean value of each station in KAE during 2009-2011, the BENTIX index values indicate less than 2 in bad ecological condition while between 2–2.5 poor, 2.5–3.5 moderate, 3.5–4.5 good and 4.5–6 for indicate normal/pristine ecological status of soft bottom macrobenthic communities (Simboura and Zenetos, 2002)

Table 12 a. Biotic indices (±SD) during 2009-2010 period in KAE

Station	BOPA	AMBI	BENTIX	M-AMBI
Station 1	0.05525 ± 0.09152	1.87 ± 1.13	4.24 ± 1.42	0.5625 ± 0.0966
Station 2	0.13658 ± 0.10756	3.18 ± 1.54	3.37 ± 1.51	0.4327 ± 0.1734
Station 3	0.04283 ± 0.07409	2.32 ± 0.85	4.21 ± 1.40	0.4910 ± 0.1275
Station 4	0.04767 ± 0.07220	2.06 ± 1.06	3.96 ± 1.49	0.5982 ± 0.1099
Station 5	0.12642 ± 0.09763	2.24 ± 1.32	4.2 ± 1.27	0.5193 ± 0.1149
Station 6	0.16217 ± 0.10154	2.97 ± 1.48	3.59 ± 1.19	0.4764 ± 0.1170
Station 7	0.14425 ± 0.10960	2.89 ± 1.25	3.45 ± 1.15	0.4949 ± 0.1607
Mean	0.10217 ± 0.10282	2.51 ± 1.30	3.86 ± 1.35	0.5107 ± 0.1364

Table 12 b. Biotic indices (±SD) during 2010-2011 period in KAE

Station	BOPA	AMBI	BENTIX	M-AMBI
Station 1	0.06642 ± 0.09698	2.64 ± 1.21	3.40± 1.72	0.4381 ± 0.1514
Station 2	0.13525 ± 0.12395	3.07 ± 1.16	3.18 ± 1.33	0.3456 ± 0.1360
Station 3	0.02025 ± 0.04894	2.35 ± 0.77	3.48 ± 1.07	0.5168 ± 0.1257
Station 4	0.02183 ± 0.04401	2.06 ± 0.79	3.99 ± 1.27	0.6117 ± 0.1153
Station 5	0.20500 ± 0.08608	3.31 ± 0.87	3.34 ± 1.2	0.4689 ± 0.0891
Station 6	0.20842 ± 0.07406	3.44 ± 0.75	3.22 ± 1.03	0.4338 ± 0.1044
Station 7	0.15092 ± 0.12511	3.29 ± 1.61	3.04 ± 1.7	0.3634 ± 0.1718
Mean	0.11544 ± 0.11495	2.88 ± 1.14	3.38 ± 1.34	0.4540 ± 0.1515

Table 12 c. Biotic indices (±SD) during different seasons in KAE

Station	BOPA	AMBI	BENTIX	M-AMBI
Mon.09-10	0.12882 ± 0.11731	2.85 ± 1.18	3.29 ± 1.12	0.5092 ± 0.1238
Pos.09-10	0.08504 ± 0.09206	2.39 ± 1.45	3.84 ± 1.57	0.4961 ± 0.1604
Pre.09-10	0.09264 ± 0.09555	2.28 ± 1.22	4.45 ± 1.10	0.5268 ± 0.1252
Mon.10-11	0.11304 ± 0.11379	2.71 ± 1.24	3.44 ± 1.32	0.4640 ± 0.1562
Pos.10-11	0.12011 ± 0.12426	3.14 ± 1.19	3.11 ± 1.46	0.4352 ± 0.1481
Pre.10-11	0.11318 ± 0.11042	2.80 ± 0.97	3.58 ± 1.22	0.4629 ± 0.1538
Mean	0.10880 ± 0.10893	2.69 ± 1.23	3.62 ± 1.36	0.4824 ± 0.1465

Table 13. Correlation coefficient between different biotic indices in KAE

	Diversity (H')	BOPA	AMBI	BENTIX	M-AMBI
S	**0.673**	-0.018	-0.110	0.086	**0.752**
N	-0.272	**-0.206**	0.012	*-0.175*	-0.093
d	**0.778**	0.001	-0.151	0.133	**0.878**
J	**0.657**	**0.224**	-0.113	**0.267**	**0.314**
H	1	*0.193*	-0.142	**0.261**	**0.779**
1-λ'	**0.955**	**0.254**	-0.136	**0.276**	**0.661**
BOPA	*0.193*	1	**0.654**	-0.494	*-0.157*
AMBI	-0.142	**0.654**	1	**-0.832**	**-0.503**
BENTIX	**0.261**	**-0.494**	**-0.832**	1	**0.470**
M-AMBI	**0.779**	*-0.157*	*-0.503*	**0.470**	1

Bold: $p < 0.01$, *Italics*: $p < 0.05$

VI. 3. Discussion

The preliminary assessment of benthic ecological disturbance in the Kodungalur-Azhikode estuary depicted a moderately disturbed condition on abundance biomass curve (ABC). This abundance biomass curve based assessment method was described initially by Warwick (1986), by plotting separate k-dominance curves for species abundances and species biomass on the single graph for comparison. The disturbed ecological communities are usually dominated by small sized r-selected or opportunistic species with a short lifespan often become the biomass dominants as well as the numerical dominants (Clarke, 1990). The detailed analysis in ABC plot for each station separately demonstrated relatively good conditions in the middle zone of the estuary. The moderately disturbed ecological state was observed in the northern station in Kottapuram region, characterised by comparatively high water residence time and cage farming activities of carnivore fishes. The northernmost station in the estuary was dominated with small-bodied opportunistic polychaete species such as *Prionospio cirrifera* and *Capitella* sp. However, the majority of other stations in the estuary exhibited moderately disturb condition in the ABC plot. Similarly, Asha (2017) reported slightly disturbed conditions in the southern part of the Vembanad wetland ecosystem with ABC plot analysis.

The benthic opportunistic polychaetes amphipods (BOPA) index was mainly based on the preference of opportunistic polychaetes in the organically enriched sediments. The sensitivity of amphipods to pollution was widely used in ecological monitoring programmes around the globe including Indian waters (Dauvin and Ruellet, 2007; Sivadas *et al.*, 2016). BOPA index was applied to seven stations in the estuary to understand different perturbations (cage farming of fishes, aquaculture and agriculture activities in the adjacent ponds, sewage discharge etc.) in the KAE. The value for the entire estuary was depicted an ecologically good condition with a spatial variation of good to moderate disturbance level. The pattern of disturbance closely matched with the spatial grouping pattern of macrofaunal assemblages in the estuary such as STA and STB group of stations. The STA group (stations 1, 3, & 4) of stations with sandy substrate and moderate organic content described good conditions while STB group (stations 2, 5, 6, & 7) demonstrated moderately disturbed condition. Among them, station six and five exhibited maximum disturbance. However, many scientific communities challenge the pollution sensitivity of species from the same taxonomic group that had a different response (Andrade and Renaud, 2011). Riera and de-la-Ossa-Carretero (2014), state that, BOPA index is more suitable for the heavily impacted environments than other areas of anthropogenic disturbances. The relatively high ecological status was noticed during the monsoon season in KAE and assumed that, could be due to the removal of organic matter deposits and well oxygenation of sediments in the estuary during massive monsoon-related river discharge. BOPA index has been widely used in oil spill monitoring studies based on the concept that an oil spill usually caused high mortalities of sensitive amphipods communities and subsequent proliferation and assemblage of opportunistic polychaetes (Dauvin and Ruellet, 2007).

The most widely accepted indices such as AZTI's Marine Biotic Index (AMBI) also applied to the estuary, and it is widely used for the coastal monitoring programmes of India (Ajmal Khan *et al.*, 2004; Sivadas *et al.*, 2016). In this biotic index, assessing the health of ecosystem-based on organic matter input either from urban effluents or eutrophication and subsequent assemblage pattern of a different ecological group of macrofaunal communities was cardinal (Borja *et al.*, 2000). The advantage of AMBI is that it doesn't require a reference site for comparison. In the present study, the AMBI index showed a similar pattern of BOPA index, but values for the entire estuary denoted slightly disturbed condition. Accordingly, STA group of stations exhibited relatively fewer disturbances compared to STB group. The proliferation of opportunistic species belonging to ecological group IV and V were more responsible for the high AMBI value (Borja and Muxika, 2005). In the similar kind of study, AMBI value depicted moderately disturbed condition in the Cochin estuary, Vellar estuary was undisturbed, and Uppanar estuary had polluted status (Ajmal Khan *et al.*, 2004; Feebarani *et al.*, 2016). In the Vellar Coleroon estuarine system, the moderately disturbed conditions (AMBI between 3.45 and 3.72) was observed at the discharge point of shrimp farms and dredging sites (Sigamani *et al.*, 2015). On a seasonal basis, relatively low ecological status was observed in the post-monsoon period of the study.

The biotic index BENTIX developed based on the relative percentages of two ecological groups of species that grouped according to their sensitivity or tolerance to disturbance factors and weighted proportionately to obtain a formula rendering a five-step numerical scale of environmental quality classification (Simboura and Zenetos, 2002). According to Simboura *et al.* (2007), BENTIX stands as the best index for assessing the long-term trends of decline or recovery status of ecosystem health. BENTIX biotic index value for the entire KAE depicted the good ecological state. Spatial variation of BENTIX in the estuary maintained the trend of other indices employed. In this index, STA a group of stations kept their good ecological status. However, one more site in the STB group in the northern arm (station 5) also included in the good condition of BENTIX. Other STB group stations were depicted having moderate disturbance as noticed in the different indices. However, some of the STB group of stations even occasionally displayed bad ecological status. On a seasonal basis, the values for the entire estuary depicted relatively high disturbance in post-monsoon (moderate condition) while least disturbance at pre-monsoon. BENTIX depicted a positive relation with diversity (r = 0.261, p

< 0.01) and M-AMBI ($r = 0.470$, $p < 0.01$), while a negative relation was observed with BOPA ($r = -0.494$, $p < 0.01$) and AMBI ($r = -0.832$, $p<0.01$).

According to Borja *et al.* (2000), benthic communities react to changes in environmental quality by three means such as the increase in species abundance, diversity and variation of dominants species from tolerant to sensitive to pollution. In multivariate-AMBI (M-AMBI) pressure gradients in the system were also taken into consideration before assigning the ecological status. According to Borja *et al.* (2009), biotic indices AMBI and M-AMBI are discriminating the different anthropogenic distresses in an ecosystem such as aquaculture effluent impacts, nutrient loading, eutrophication, anoxia and hypoxia, oil and industrial pollution etc. (Borja *et al.*, 2009; Pinto *et al.*, 2009). In the present study, ecological status of entire KAE has been depicted as a moderated condition in M-AMBI index based on the factorial analysis of Shannon's diversity, richness, and AMBI index. This index also has been validated as effective ecological status assessment tools for benthic communities of diverse geographical areas, ranging from tropics to high latitudes, under different human pressures (Borja and Tunberg, 2011; Feebarani et al., 2016; Sigovini *et al.*, 2013; Spagnolo *et al.*, 2014). Many studies were carried out on M-AMBI based ecological status assessment in the estuarine systems of India and observations were varied in place and time (Ajmal Khan *et al.*, 2014; Sivadas *et al.*, 2016). The high to good ecological status (0.9 ± 0.06) observed in Vellar estuary and that for Uppanar estuary was poor to bad (0.22 ± 0.04) (Ajmal Khan *et al.*, 2014). While moderate condition was reported in Cochin estuary (Feebarani *et al.*, 2016). Similar to all other three indices used in the present study, M-AMBI has also depicted the same pattern in spatial scale and noticed a relatively good ecological status in STA group of stations. However, the ecologically good status only observed in the STA group station with sandy sediment, and low organic matter (station 4) and all other stations in the estuary was displayed moderate ecological status. It also noticed that station two in the middle zone of an estuary with muddy substratum depicted relatively low ecological status (moderate) with the potential of poor status (0.3892 ± 0.1588).

In the present observation, all indices tested in the KAE have behaved in the same pattern with slight variation in the status assignment. BOPA index depicted a significant relationship with all other indices tested with a strong positive correlation on AMBI (r = 0.654, $p < 0.01$) and weak negative association with species diversity (r = 0.193, $p < 0.05$) and M-AMBI (r = -0.157, p <0.05). Species richness also described a healthy positive relationship with biotic index M-AMBI; however, at the same time, AMBI revealed the strong significant positive correlation with BOPA (r = 0.654, $p < 0.01$) and a negative relation with BENTIX (r = -0.832, $p < 0.01$) and M-AMBI (r = -0.503, $p <$ 0.01). However, the M-AMBI exhibited a positive relation with species diversity(r = 0.779, $p < 0.01$), and BENTIX index (r = 0.470, $p < 0.01$) and that for BOPA (r = -0.157, $p < 0.01$) and AMBI (r = -0.503, $p < 0.01$) were the negative correlation. The species evenness, diversity, and dominance positively related to all indices tested in the KAE, except AMBI index. While the present study demonstrated the similar pattern in spatial scale distribution of indices, however many studies indicate that these biotic indices failed when the communities with the dominance of tolerant and opportunistic species or correlation between species diversity and natural stress were high (Reizopoulou *et al.*, 2014). Some studies also noticed opposite result when comparing the indices like Shannon diversity index (H'), AZTI's Marine Biotic Index (AMBI), multivariate-AMBI (M-AMBI), benthic opportunistic polychaetes amphipods (BOPA) and BENTIX indices (Spagnolo *et al.*, 2014). According to Sivadas *et al.* (2016), these temperate benthic indices are efficient in Indian coastal waters however complementary use of different index recommended to an accurate assessment of ecological quality.

Chapter 7

Summary and conclusion

Kerala state of the south-west coast of India has few wetlands of international or national importance. Of these, Vembanad Kol wetland ecosystem is the largest humid tropical brackish coastal wetland ecosystem in the south-west coast of India. The Vembanad Lake possesses dual openings to the Arabian Sea; one is at Kochi (Cochin estuary) another at Munambam-Azhikode (Kodungallur-Azhikode estuary). The region under the present study is Kodungallur-Azhikode estuary (KAE), the confluence of Chalakudy River, Karuvannur River and Periyar River with an area of 700 ha. These coastal habitats considered as cradle grounds for biological components mediated by constant nutrients supply from autochthonous and allochthonous sources. They also function as sinks and transformers of nutrients, by altering the quantity and quality of nutrients transported from land to the sea. They also form an essential zone of human use for fisheries, transportation, aquaculture, and recreational activities. Thus by its nature and easy accessibility, estuaries are vulnerable to anthropogenic effects. They exhibit a dynamic environment, primarily due to short-term changes caused by the tide and the seasonal changes driven by the regional climate. Interaction of environmental characteristics determines the population dynamics and biocoenosis of these waterbodies. In this critical environment, benthic fauna plays an important role in the food web dynamics and bio-mineralisation processes and forms a better indicator of environmental status. Therefore, this PhD work creates a better database for understanding and managing these crucial habitats by studying the essential functional groups in the Kodungallur-Azhikode estuary.

Chapter 1 describes the general description importance and features of estuaries and coastal habitats along an introduction and developmental history of benthic studies in worldwide and Indian estuaries. This chapter also provides an outline of significant studies in the adjacent coastal and estuarine areas. This part also highlights the significance and objectives of the study.

Chapter 2 provides the details of the study area with the detailed description of each station, sampling strategies, methods of analysis and statistical tools used for the study.

Chapter 3 elaborates the hydrographic and sediment characteristics of the study area. The pattern of variation in hydrological and sediment quality parameters was made clear in the PCA ordination; dataset has mainly separated into two principal components accounted for 46.3 percentage of the variability. The first two components have been mainly divided based on the spatial and temporal pattern in distribution. Seasonal variability of parameters was evident on the axis 1, such as rainfall-induced hydrographic changes like river discharge, bottom water salinity, nitrate, Eh, transparency, pH, temperature, turbidity, dissolved oxygen and linked sediment characteristics like sediment temperature and pH. However, sediment parameters were more varied on a spatial scale than that of hydrographic features such as organic matter, sand, silt, clay content along with related hydrographic parameter like bottom water dissolved nitrite was distributed along the axis 2 (PCA 2). The heavy rainfall and subsequent river discharge during monsoon season control the entire biogeochemical and ecological process in the estuary, along with other anthropogenic influence. Sediment texture and organic matter content of sediment play a significant role in the distribution pattern of benthic communities in the estuary.

Chapter 4 discussed the benthic secondary production and its contribution to fish biomass in the estuary. The numerical density of macrobenthos varied from 23 ind.m^{-2} (Station 7, November 2011) to 87568 ind.m^{-2} (Station 3, September 2011) with an overall mean of 3887 ± 10083 ind.m^{-2}. Spatio-temporal variability was observed in the numerical density and biomass of macrofauna. A total of 18846 macrofaunal organisms collected, 60 percentage were malacostracan crustaceans, which was the dominant group, followed by polychaetes (20 %), molluscs (9 %) and the sporadic representatives were pooled together as 'others group' (11 %). They represented by hydrozoans, cirripedians, insects, nemerteans, benthic fishes and ophiuroideans. Biomass of macrofauna was estimated on a wet weight basis. In the entire study period, malacostracan crustaceans contributed about 18 percentage to total biomass, while polychaete formed 17 percentage, which for molluscs 63 percentage and other groups with 2 percentage. The estimated macrobenthic P/B ratio for the estuary was 35.29 yr^{-1} during the first year and 36.21 yr^{-1} for the second year. Based on the calculation, the macrofauna of the study area alone can be said to contribute to about 29588 ± 51774 kg of fish biomass during the study period. It was 26460 ± 44652 kg for the first year and about 32718 ± 58897 kg for the second year. The mean numerical density of meiofauna in the study area was 836 ± 404 ind.10 cm^{-2}. Of these, 91.86 % were nematodes, which were the dominant group that followed by harpacticoid copepods (6.47 %), amphipods (1.26 %), foraminiferan (0.05 %), polychaete larvae (0.30 %) and the sporadic representatives were pooled together as 'others' (0.06 %).

Chapter 5 describes the biocenosis of benthic macrofauna. In the faunistic examination of macrofauna yielded a total of 79 species in 71 genera belonging to 49 families. Among the 79 species of macrofauna, polychaetes constituted the major component with 33 species. The class Malacostraca formed the second dominant group with 26 species belonging to seven orders. The class Bivalvia and Gastropoda of phylum Mollusca all together formed the third position by the representation of 11 species to macrofaunal species count. The class Bivalvia contributed nine species while class Gastropoda represented with two species. Other groups were also contributed to the overall diversity of macrofaunal communities. The numerically dominant polychaete species in the study area were *Prionospio cirrifera*, *Capitella* sp, and *Notomastus* sp. and that for malacostracans were *Americorophium triaeonyx* and *Cirolana fluviatilis*. The numerically dominant molluscan species were *Arcuatula senhousia*, *Marcia recens* and *Villorita cyprinoides*.

In a spatial scale, macrofauna species richness was maximum observed at middle zone with sandy substratum, while low species richness was recorded in the Riverhead. Seasonally, moderate species richness was observed during the monsoon period with heavy river discharge. Species evenness in the estuary depicted a relatively low value at the middle zone, but seasonally it was high at pre-monsoon period. In the present observation, polychaetes were the most diverse group in term of macrobenthic species diversity, while malacostracan crustaceans dominated in the numerical density and bivalve molluscs led the biomass. In the present study, suspension feeding communities were dominated by the estuarine system. They were amphipod *Americorophium triaeonyx*, mussel *Arcuatula senhousia* and hydrozoan *Obelia bidentata*. They are usually abundant in an area with low biological oxygen demand (BOD). Suspensions feeders are ecologically relevant organisms that provide various ecosystem services like water column filtration and removal of turbidity. However, the population of surface deposit feeders (*Prionospio cirrifera*), sub-surface deposit feeder (*Capitella* sp.) and carnivores (*Cirolana fluviatilis*) dominated in some zones of the estuary depending on the suitable substrate type and other ecological factors. Deposit feeders were more abundant in sediment with the high organic matter. They are taking part in the biomineralisation process of nutrients. The burrowing population of isopod, *Sphaeroma annandalei* was also observed in the benthic samples (SRL).

Hierarchical cluster analysis and SIMPROF test on the full set of data revealed that of the seven stations, grouped into two significant clusters (STA and STB) of which STA exhibit higher similarity between stations station three, four and one, while the in-group STB showed the overall similarity between stations six, seven, two and five. The Canonical Correspondence Analysis (CCA) was carried out to determine which environmental factors influence the distribution of the macrofaunal species. The CCA axes 1 and 2 explained 39.3 and 12.7 percentage of the species variation respectively. In the stations-environment CCA biplot, the stations one, three, and four strongly influenced by sand content, salinity, sediment Eh and turbidity. While, stations six and seven correlated with organic matter and clayey sediment (%). Station two was related with oxygenated clayey sediment and organic matter. While at station five, Chl-*a*, sediment temperature with shallow depth noticed as controlling factor. In the Principal Coordinate Analysis (PCO), the first two axes explained about 24.6 % of the total variability. The direction of the vector representing organic matter, clay, silt, sediment temperature increased towards station group STB. While, depth, the percentage of sand, salinity and sediment pH increased towards STA. While SSA and SSB were separated mainly by the influence of environmental parameters river discharge, rainfall, DO, Transparency, water temperature, dissolved phosphate and ammonia. Canonical Analysis of Principal Coordinates (CAP) performed to confirm the above pattern, the canonical correlation values of the first two axes were 0.7354 and 0.4261, respectively.

Chapter 6 focused on the ecological status of Kodungallur-Azhikode estuary. The preliminary assessment of macrofaunal disturbance in the Kodungalur-Azhikode estuary depicted a moderately disturbed condition in abundance biomass curve (ABC). The detailed analysis of ABC plot demonstrated moderately disturbed ecological state in the northern station of Kottapuram region, where we can see cage farming of carnivore fishes with relatively high water residence time. BOPA index was applied to seven stations in the estuary to understand different perturbation (cage farming of fishes, aquaculture and agriculture activities in the adjacent ponds, sewage discharge etc.) in the KAE. The pattern of disturbance closely matched with the spatial grouping pattern of macrofaunal assemblages in the estuary and it varied from ecologically good to moderate condition. AMBI index also showed a similar pattern of BOPA index and noticed a slightly disturbed condition. BENTIX biotic index for the entire KAE varied from good to moderate ecological status. Spatial variation of BENTIX in the estuary maintained the trend of other index employed. Stations with sandy substratum and moderate organic carbon content depicted good ecological status. M-AMBI depicted the same pattern with relatively good ecological status in the area of low organic carbon content with sandy sediment. It also noticed that a station in the middle zone of the estuary with muddy substratum depicted relatively low ecological status (moderate) with the potential of poor condition. In the present observation, all indices tested in the KAE have behaved in the same pattern with slight variation in the status assignment. The species evenness, diversity, and dominance positively related to all index tested in the KAE, except AMBI index. The complimentary use of these indices is recommended for an accurate ecological quality assessment.

The several alterations observed along the estuarine gradient, such as harbour activities, estuarine mouth alteration, transportation, fixing of fishing gears like dip nets and stake nets, aquaculture including cage farming, sand mining, modification of the water body for infrastructure development such as bridge, harbour, boat maintenance yards and construction of concrete belt along the estuarine shore for protecting soil erosion. Such geomorphic changes in the estuary were resulting in a reduction of natural water flow, periodic removal of organic matter and other contaminants in the estuary. These changes along with anthropogenic and natural disturbance in the estuary directly influence the hydrological factors, salinity regime and eutrophication process, and ultimately loss of natural habitats for several estuarine communities. The ecosystem perturbations in the estuary primarily affect the benthic biotope and its communities. These communities mainly sessile or sedentary forms which continuously exposed to these changes and contamination. The loss of such healthy benthic communities will seriously affect the overall health of estuary because these communities are essential components in carbon and other nutrient recycling in the estuary as well they form food for diverse fauna in and around estuarine system including birds, larval forms of many marine organisms along with true estuarine communities. Thereby this observation brings the importance of conservation and management requirements of critical habitats and implementation of specific protection measures strictly implemented by the law. In this regard, finding from this study suggests some recommendations to put forth for proper sustainable management of the Kodungallur-Azhikode estuary.

- ❖ The study has for first time chronicled the current status and aetiology of benthic fauna in the northern part of Vembanad Kol, Ramsar site - the Kodungallur-Azhikode estuary (KAE) experiencing different episodic conditions, especially the ecosystem modifications along the waterbody. The benthic biotope regulates the grazing and detritus food chain sustaining the productivity and overall nourishment in an aquatic ecosystem. Therefore, benthic system based studies are comparatively long-standing and well-accepted method to understanding the potential risk assessment. The present research reaffirmed that some zones in the estuary witnessed moderately disturbed benthic ecological status with potential for high ecological disturbances, exemplified more in the area of low river current and high water residence time. The changes documented on benthic fauna in the 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 India. Thus, to maintain the ecological integrity of Vembanad Kol wetland (including KAE) 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 the estuary, creating isolated microhabitats in the ecosystem. These microhabitats symbolised the changes in the hydrological regime combined with depleting environmental conditions - nutrient - organic enrichment, salinity pattern arising from various stress factors like liquid and solid waste disposal from multiple sources, decaying weeds and plant materials, industrial and harbour activity, domestic discharges, fish cage operations and related factors. Comparison of the different locations of KAE suggests that the Kottapuram region is experiencing relatively high ecological risk due to exposure from various disturbances altering the estuarine geomorphology. However, the bar mouth region of the estuary is experiencing multiple pressures from the adjoining fishing boat movement, effluent discharge from the adjacent processing and harbour-related accomplishments and fishing boat maintenance activities along the southern arm that is connected to the Cochin channel. The southern channel in the barmouth region is also polluted with plastic litter and other wastes that lead to severe benthic loss and low ecological integrity. As all these eco-biological alterations are interlinked, the benthic zone and the organisms are to be monitored on a regular basis or even employed as monitoring tools for understanding the health of an aquatic system. Thus, benthic monitoring should be made mandatory for all EIA (environmental impact assessment) and coastal zone monitoring programmes.

- ❖ Overall assessment of the ecological quality of soft-bottom benthic communities in the estuary established a moderately disturbed condition with potential for high disturbance, especially in the area where organic matter accumulation from effluent discharge and various anthropogenic activities with high water residence time. Such as in Kottapuram, Gothuruth, Moothakunnam and Munambam harbour area are witnessing enrichment of sediment organic matter. This was evidenced by a proliferation of opportunistic polychaetes in the regions. The pattern of organic matter accumulation is related to riverine organic matter loading, the source of domestic effluent, cage farming, sediment type, and regional circulation pattern. The enhanced accumulation of organic matter in the sediment leads to increased oxygen demand that results in the formation of hypoxic or anoxic zones and induces the accumulation of inorganic pollutants like heavy metals in the sediment. That eventually leads to the loss of habitat for a diverse number of aquatic fauna and reduces the resilience capacity of an ecosystem to perturbations. The effective management measures like imposing restrictions in these impacted zones are to be implemented for a specific period liaisoning with local self-government agencies and stakeholders so that they get rejuvenated for their wise use.
- ❖ Similarly, the continuous sand mining activities near to Moothakunnam region of KAE resulted in deepening of that area which induces the organic matter accumulation along shadow zone by changed circulation pattern. The uncontrolled mining of live shells (*Marcia recens*, *Meretrix*

casta and *Villorita cyprinoids*) from the water body is also posing a threat to the ecosystem. Based on the trophic guild analysis, filter feeders were the most abundant group in the estuary. The abundance of filter feeders regulated the turbidity levels and eutrophication problems in the system. So periodic monitoring of the estuary is essential for the formulation of viable management options for the sustainable utilisation of these vital ecological habitats. Similarly, scientific methods should be adopted for shoreline protection activities along the estuarine zones. In many occasions, concrete seawall constructed for preventing soil erosion has negatively affected the ecology of coastal zone.

❖ Even though several national and international schemes like the Ramsar Convention for conservation, Swaminathan Commission for agrarian reforms and several other states and central Government sponsored action plans have been implemented in Vembanad and associated backwaters, no serious action has emanated at the grass-roots level for the overall social benefits and development of these unique ecosystems. So appropriate and timely action is to be effected soon by the Government agencies. Otherwise, these backwater systems will get wiped out from the face of the earth. Considering the importance and value of the backwater, the KAE and associated systems can be considered as heritage sites under the United Nations Educational, Scientific, and Cultural Organization (UNESCO) World Heritage site for conservation and sustainable utilisation.

References

Abdul Azis, P.K., Nair, B.N., 1983. Meiofauna of the Edava-Nadayara Paravur backwater system of south west coast of India. *Mahasagar - Bulletin National Institute of Oceanography* 16, 55-66.

Ajith, J.K., Balchand, A.N., 1996. Morpho dynamic behaviour of a tropical coastal sea-inlet-estuary system. in *Proceedings of the 8th international conference on physics of estuaries and coastal seas*, Netherlands. pp. 42-43.

Ajmal Khan, S., Manokaran, S., Lyla, P.S., 2014. Assessment of ecological quality of Vellar and Uppanar estuaries, southeast coast of India, using Benthos. *Indian Journal of Geo-Marine Sciences* 43, 1989-1995.

Ajmal Khan, S., Murugesan, P., 2005. Polychaete diversity in Indian estuaries. *Indian Journal of Geo-Marine Sciences* 34, 114-119.

Ajmal Khan, S., Murugesan, P., Lyla, P.S., Jaganathan, S., 2004. A new indicator macro invertebrate of pollution and utility of graphical tool and diversity indices in pollution monitoring studies. *Current Science* 87, 1508-1510.

Albayrak, S., Balkis, H., Zenetos, A., Kurun, A., Kubanç, C., 2006. Ecological quality status of coastal benthic ecosystems in the Sea of Marmara. *Marine Pollution Bulletin* 52, 790-799.

Alexander, W.B., Southgate, B.A., Bassindale, R., 1935. Survey of the River tees. D.S.I.R. Water Pollution Research Technical Paper. 5, 1-171.

Alf, N., Vonda, J.C., Simon, F.T., Judi, E.H., Terry, H., 2001. Local dispersal of juvenile bivalves: implications for sandflat ecology. *Marine Ecology Progress Series* 212, 131-144.

Aller, R.C., 1983 The importance of the diffusive permeability of animal burrow linings in determining marine sediment chemistry. *Journal of Marine Research* 41, 299-322.

Aller, R.C., Dodge, R.E., 1974 Animal-sediment relations in a tropical lagoon Discovery Bay, Jamaica. *Journal of Marine Research* 32, 209-232.

American Public Health Association, 2005. Standard methods for the examination of water and wastewater. 21st Edition, American Public Health Association/American Water Works Association/Water Environment Federation, Washington, DC, pp. S-1 to S-16.

Ananthakrishnan, R., Parthasarthy, B., Pathan, J.M., 1979. Meteorology of Kerala. Contributions to Marine Sciences, 60, 123-125.

Anderson, M.J., Gorley, R.N., Clarke, K.R., 2008. PERMANOVA+ for PRIMER: guide to software and statistical methods. Plymouth, UK. pp. 214.

Andrade, H., Renaud, P.E., 2011. Polychaete/amphipod ratio as an indicator of environmental impact related to offshore oil and gas production along the Norwegian continental shelf. *Marine Pollution Bulletin* 62, 2836-2844.

Anila Kumary, K.S., 2016. Distribution of meiofauna in the Poonthura Estuary, Thiruvananthapuram, Kerala *International Journal of Pure and Applied Bioscience* 4:5, 121-126.

Anirudhan, T.S., Nambisan, N.P.K., 1990. Distribution of salinity and silicon in Cochin estuary. *Indian Journal of Geo-Marine Sciences* 19, 137-139.

Annandale, N., 1907. Fauna of the Brakish ponds of Port Canning, Lower Bengal, Introduction and Preliminary account of fauna. 1, 46-74.

Annandale, N., Kemp, S., 1915. Fauna of the Chilka lake. *Memories of Indian Museum* 5, 28.

Anonymous, 1958. Venice system. Supplement to the Symposium on the classification of brackish waters. Venice, April 8-14, 1958. *Archivio di Oceanografia e Limnologia* 11 (Suppl.), 1-248.

Anonymous, 1974. Water resources of Kerala, Government of Kerala Publication. Public Works Department (Kerala), Trivandrum.

Anonymous, 2003. Survey and inventory of wetlands of Kerala for conservation and sustainable management of resources, Project Report, Kerala Forest Department,Thiruvananthapuram, Kerala. Centre for Environment and Development.

Ansari, Z.A., 1974. Macrobenthic production in Vembanad Lake. *Mahasagar- Bullettin of the National Institute of Oceanography* 7, 197-200.

Ansari, Z.A., 1977. Macrobenthos of the Cochin backwaters. *Mahasagar- Bullettin of the National Institute of Oceanography* 10, 169-171.

Ansari, Z.A., 1978. Meiobenthos from the Karwar region (Central west coast of India). *Mahasagar- Bullettin of the National Institute of Oceanography* 11, 163-167.

Ansari, Z.A., 1988. Ecology of meiobenthos in two estuaries of Goa, Ph.D. Thesis, University of Bombay pp.224.

Ansari, Z.A., Ingole, B.S., Parulekar, A.H., 1986. Effect of high organic enrichment of benthic polychaete population in an estuary. *Marine Pollution Bulletin* 17, 361-365.

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

Ansari, Z.A., Parulekar, A.H., 1998. Community structure of meiobenthos from a tropical estuary. *Indian Journal of Geo-Marine Sciences* 27, 362-366.

Ansari, Z.A., Rivonkar, C., Sangodkar, U., 2001. Population fluctuation and vertical distribution of meiofauna in a tropical mudflat at Mandovi Estuary, west coast of India. *Indian Journal of Geo-Marine Sciences* 30, 237-245.

Ansari, Z.A., Sreepada, R.A., Kanti, A., 1994. Macrobenthic assemblage in the soft sediment of Marmagoa harbour, Goa (Central west coast of India). *Indian Journal of Geo-Marine Sciences* 23, 225-231.

Anu, P.R., Jayachandran, P.R., Sreekumar, P.K., Bijoy Nandan, S., 2014. A Review on Heavy Metal Pollution in Cochin Backwaters, Southwest Coast of India *International Journal of Marine Science* 4, 92-98.

Appeltans, W., Ahyong, Shane T., Anderson, G., Angel, Martin V., Artois, T., Bailly, N., Bamber, R., Barber, A., Bartsch, I., Berta, A., Błażewicz-Paszkowycz, M., Bock, P., Boxshall, G., Boyko, Christopher B., Brandão, Simone N., Bray, Rod A., Bruce, Niel L., Cairns, Stephen D., Chan, T.-Y., Cheng, L., Collins, Allen G., Cribb, T., Curini-Galletti, M., Dahdouh-Guebas, F., Davie, Peter J.F., Dawson, Michael N., De Clerck, O., Decock, W., De Grave, S., de Voogd, Nicole J., Domning, Daryl P., Emig, Christian C., Erséus, C., Eschmeyer, W., Fauchald, K., Fautin, Daphne G., Feist, Stephen W., Fransen, Charles H.J.M., Furuya, H., Garcia-Alvarez, O., Gerken, S., Gibson, D., Gittenberger, A., Gofas, S., Gómez-Daglio, L., Gordon, Dennis P., Guiry, Michael D., Hernandez, F., Hoeksema, Bert W., Hopcroft, Russell R., Jaume, D., Kirk, P., Koedam, N., Koenemann, S., Kolb, Jürgen B., Kristensen, Reinhardt M., Kroh, A., Lambert, G., Lazarus, David B., Lemaitre, R., Longshaw, M., Lowry, J., Macpherson, E., Madin, Laurence P., Mah, C., Mapstone, G., McLaughlin, Patsy A., Mees, J., Meland, K., Messing, Charles G., Mills, Claudia E., Molodtsova, Tina N., Mooi, R., Neuhaus, B., Ng, Peter K.L., Nielsen, C., Norenburg, J., Opresko, Dennis M., Osawa, M., Paulay, G., Perrin, W., Pilger, John F., Poore, Gary C.B., Pugh, P., Read, Geoffrey B., Reimer, James D., Rius, M., Rocha, Rosana M., Saiz-Salinas, José I., Scarabino, V., Schierwater, B., Schmidt-Rhaesa, A., Schnabel, Kareen E., Schotte, M., Schuchert, P., Schwabe, E., Segers, H., Self-Sullivan, C., Shenkar, N., Siegel, V., Sterrer, W., Stöhr, S., Swalla, B., Tasker, Mark L., Thuesen, Erik V., Timm, T., Todaro, M.A., Turon, X., Tyler, S., Uetz, P., van der Land, J., Vanhoorne, B., van Ofwegen, Leen P., van Soest, Rob W.M., Vanaverbeke, J., Walker-Smith, G., Walter, T.C., Warren, A., Williams, Gary C., Wilson, Simon P., Costello, Mark J., 2012. The Magnitude of Global Marine Species Diversity. *Current Biology* 22, 2189-2202.

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

Aravindakshan, P.N., Balasubramanian, T., Lalithambika Devi, C.B., Nair, K.K.C., Gopalakrishnan, T.C., Jayalakshmy, K.V., Krishnan Kutty, M., 1992. Benthos arid substratum characteristics of prawn culture fields in and around the Cochin backwater. *Journal of the Marine Biological Association of India* 34, 203-217.

Arun, A.U., 2005. Impact of artificial structures on biodiversity of estuaries: a case study from Cochin estuary with emphasis on clam beds. *Applied Ecology and Environmental Research* 4, 99-110.

Asha, C.V., Retina, I.C., Suson, P.S., Bijoy Nandan, S., 2016. Ecosystem analysis of the degrading Vembanad wetland ecosystem, the largest Ramsar site on the South West Coast of India- Measures for its sustainable management. *Regional Studies in Marine Science* 8, 408-421.

Asha Nair, M., Abdul Azis, P.K., 1995. Studies on the benthos of the Veli estuary, Thiruvananthapuram, Proceedings of the Seventh Kerala Science Congress, pp. 102-104.

Ates, A.S., Katagan, T., Sezgin, M., Acar, S., 2014. The Response of *Apseudopsis latreillii* (Milne-Edwards, 1828) (Crustacea, Tanaidacea) to Environmental Variables in the Dardanelles. *Turkish Journal of Fisheries and Aquatic Sciences* 14, 113-124.

Attrill, M.J., Rundle, S.D., 2002. Ecotone or Ecocline: Ecological Boundaries in Estuaries. *Estuarine, Coastal and Shelf Science* 55, 929-936.

Bailey-Brock, J.H., 1984. Ecology of the tube-building polychaete *Diopatra leuckarti* Kinberg, 1865 (Onuphidae) in Hawaii: community structure, and sediment stabilizing properties. *Zoological Journal of the Linnean Society* 80, 191-199.

Baker, C.A., 1978. A study of estuarine sedimentation in South Slough, Coos Bay,Oregon. Dissertations and Theses. Paper 2867. 1-117.

Balachandran, K.K., Thresiamma, J., Maheswari, N., Sankaranarayanan, V.N., Kesavadas, V., Sheeba, P., 2003. Geochemistry of surficial sediments along the central southwest coast of India: Seasonal changes in regional distribution. *Journal of Coastal Research* 19, 664-683.

Balasubrahmanyan, K., 1964. Studies in the ecology of the Vellar estuary-the intertidal and estuarine polychaeta. *Journal of the Annamalai University* 25, 101- 104.

Balchand, A.N., Nair, S.M., 1994. Fractionation of phosphorus in the sediments of a tropical estuary. *Environmental Geology* 23, 284-294.

Bald, J., Borja, A., Muxika, I., Franco, J., Valencia, V., 2005. Assessing reference conditions and physico-chemical status according to the European Water Framework Directive: A case-study from the Basque Country (Northern Spain). *Marine Pollution Bulletin* 50, 1508-1522.

Balsamo, M., Albertelli, G., Ceccherelli, V.U., Coccioni, R., Colangelo, M.A., Curini-Galletti, M., Danovaro, R., D'Addabbo, R., De Leonardis, C., Fabiano, M., Frontalini, F., Gallo, M., Gambi, C., Guidi, L., Moreno, M., Pusceddu, A., Sandulli, R., Semprucci, F., Todaro, M.A., Tongiorgi, P., 2010. Meiofauna of the Adriatic Sea: present knowledge and future perspectives. *Chemistry and Ecology* 26, 45-63.

Barbier, E., Hacker, S.D., Kennedy, C., Koch, E.W., Stier, A.C., Silliman, B.R., 2011. The value of estuarine and coastal ecosystem services. *Ecological Monographs* 81, 169-193.

Barros, F., Hatje, V., Figueiredo, M.B., Magalhães, W.F., Dórea, H.S., Emídio, E.S., 2008. The structure of the benthic macrofaunal assemblages and sediments characteristics of the Paraguaçu estuarine system, NE, Brazil. *Estuarine, Coastal and Shelf Science* 78, 753-762.

Batcha, A.S.M., 1984. Studies on the bottom fauna of north Vembanad Lake. Ph.D. Thesis, Cochin University of Science and Technology, India. pp.157.

Beesley, P.L., Ross, G.J.B., Glasby, C.J., 2000. Polychaetes and Allies: The Southern Synthesis. Environment Australia, Australia. pp.465.

Bendschneider, K., Robinson, R.J., 1952. A new spectrophotometric method for the determination of nitrite in sea water. *Journal of Marine Research* 11, 87-93.

Benson, W.H., 1861. Descriptions of freshwater shells collected in southern India by Lieut. Charles Annesley Benson, 45th MNI. *Annals and Magazine of Natural History* 3, 257-260.

Bijoy Nandan, S., 2008. Current status and biodiversity modification in the coastal wetland ecosystems of India with objectives for its sustainable management, Proc. conserve-vision conference, The University of Waikato, www.waikato.ac.n/ wtass/conserve-vision

Bijoy Nandan, S., Abdul Azis, P.K., 1994. Primary production of the retting zones in the Kadinamkulam estuary, South West coast of India. *Mahasagar- Bullettin of the National Institute of Oceanography* 27, 103.

Bijoy Nandan, S., Abdul Azis, P.K., 1995a. Benthic polychaetes in the anoxic sulphide biomass of the retting zones in the Kadinamkulam Kayal. *International Journal of Environmental Studies*, 257-267.

Bijoy Nandan, S., Abdul Azis, P.K., 1995b. Ecology of the Kadinakulam backwater, Kerala. *Journal of the Inland Fisheries Society of India* 27, 18-32.

Bijoy Nandan, S., Abdul Azis, P.K., 1995c. pH and Eh in the Kadinakulam estuary a comparison of the retting and non retting environments. *Marine Research* 4, 93-112.

Bijoy Nandan, S., Jayachandran, P.R., Sreedevi, O.K., 2014. Spatio-temporal pattern of primary production in a tropical coastal wetland (Kodungallur-Azhikode estuary), south west coast of India. *Coastal Development* 17, 392.

Bijoy Nandan, S., Unnithan, V.K., 2007. Ecology of Vembanad lake with investigations in Vembanad lake, in: 107, B.N. (Ed.). CIFRI (ICAR), Govt. of India, p. 38.

Biju Kumar, A., Ravinesh, R., 2016. Taxonomy of Marine Molluscs of India: Status and Challenges Ahead, in: Bijoy Nandan, S., Graham Oliver, P., Jayachandran, P.R., Asha, C.V. (Eds.), Training manual -1st International Training Workshop on Taxonomy of Bivalve Molluscs. Directorate of Public Relations and Publications, Cochin University of Science and Technology Kochi, pp. 67-88.

Bilyard, G.R., 1987. The value of benthic infauna in marine pollution monitoring studies. *Marine Pollution Bulletin* 18, 581-585.

Birchenough, S.N.R., Reiss, H., Degraer, S., Mieszkowska, N., Borja, Á., Buhl-Mortensen, L., Braeckman, U., Craeymeersch, J., De Mesel, I., Kerckhof, F., Kröncke, I., Parra, S., Rabaut, M., Schröder, A., Van Colen, C., Van Hoey, G., Vincx, M., Wätjen, K., 2015. Climate change and marine benthos: a review of existing research and future directions in the North Atlantic. *Wiley Interdisciplinary Reviews: Climate Change* 6, 203-223.

Böggemann, M., 2002. Revision of the Glyceridae Grube 1859 (Annelida: Polychaeta). *Abhandlungen der Senckenbergischen naturforschenden Gesellschaft* 555, 1-249.

Böggemann, M., 2005. Revision of the Goniadidae. *Abhandlungen des Naturwissenschaftlichen Vereins in Hamburg* (Neue Folgen) 39, 1-354.

Bolam, S.G., Eggleton, J., Smith, R., Mason, C., Vanstaen, K., Rees, H., 2008. Spatial distribution of macrofaunal assemblages along the English Channel. *Journal of the Marine Biological Association of the United Kingdom* 88, 675-687.

Bordovskiy, O.K., 1964. Accumulation of organic matter in bottom sediments. *Marine Geology* 3, 34-79.

Bordovskiy, O.K., 1965. Accumulation of organic matter in bottom sediments. *Marine Geology* 3, 33-82.

Borja, A., Dauer, D.M., 2008. Assessing the environmental quality status in estuarine and coastal systems: Comparing methodologies and indices. *Ecological Indicators* 8, 331-337.

Borja, Á., Dauer, D.M., Grémare, A., 2012. The importance of setting targets and reference conditions in assessing marine ecosystem quality. *Ecological Indicators* 12, 1-7.

Borja, A., Franco, J., Pérez, V., 2000. A marine biotic index to establish the ecological quality of soft-bottom benthos within European estuarine and coastal environments. *Marine Pollution Bulletin* 40, 1100-1114.

Borja, A., Josefson, A.B., Miles, A., Muxika, I., Olsgard, F., Phillips, G., Rodríguez, J.G., Rygg, B., 2007. An approach to the intercalibration of benthic ecological status assessment in the North Atlantic ecoregion, according to the European Water Framework Directive. *Marine Pollution Bulletin* 55, 42-52.

Borja, A., Muxika, I., 2005. Guidelines for the use of AMBI (AZTI's Marine Biotic Index) in the assessment of the benthic ecological quality. *Marine Pollution Bulletin* 50, 787-789.

Borja, A., Ranasinghe, A., Weisberg, S.B., 2009. Assessing ecological integrity in marine waters, using multiple indices and ecosystem components: Challenges for the future. *Marine Pollution Bulletin* 59, 1-4.

Borja, A., Tunberg, B.G., 2011. Assessing benthic health in stressed subtropical estuaries, eastern Florida, USA using AMBI and M-AMBI. *Ecological Indicators* 11, 295-303.

Botto, F., Iribarne, O., 2000. Contrasting effects of two burrowing crabs (*Chasmagnathus granulata* and *Uca uruguayensis*) on sediment composition and transport in estuarine environments. *Estuarine, Coastal and Shelf Science* 51, 141-151.

Boudreau, P.R., Dickie, L.M., Kerr, S.R., 1991. Body-size spectra of production and biomass as system-level indicators of ecological dynamics. *Journal of Theoretical Biology* 152, 329-339.

Bouillon, S., Raman, A.V., Dauby, P., Dehairs, F., 2002. Carbon and nitrogen stable isotope ratios of subtidal benthic invertebrates in an estuarine mangrove ecosystem (Andhra Pradesh, India). *Estuarine, Coastal and Shelf Science* 54, 901-913.

Boyd, C.E., 2015. Dissolved Oxygen and Other Gases, in: Boyd, C.E. (Ed.), Water Quality: An Introduction. Springer International Publishing, Cham, pp. 113-137.

Braeckman, U., Provoost, P., Moens, T., Soetaert, K., Middelburg, J.J., Vincx, M., Vanaverbeke, J., 2011. Biological vs. physical mixing effects on benthic food web dynamics. *PLoS ONE* 6, e18078.

Brandt, A., Muhlenhardt Siegel, U., Schmidt, A., 1999. Density, diversity and community patterns of selected peracarid taxa (Malacostraca) in the Beagle Channel, South America. In Schramm FR, von Vaupel Klein JC (eds.) Crustaceans and the Biodiversity Crisis, Proceedings of the fourth international crustacean congress, Amsterdam, The Netherlands, pp. 541-558.

Breitburg, D.L., Hondorp, D.W., Davias, L.A., Diaz, R.J., 2009. Hypoxia, nitrogen, and fisheries: integrating effects across local and global landscapes. *Annual Review of Marine Science* 1, 329-349.

Brey, T., Müller-Wiegmann, C., Zittier, ZMC., Hagen, W., 2010. Body composition in aquatic organismsa global data bank of relationships between mass, elemental composition and energy content. *Journal of Sea Research* 64, 334340.

Brey, T., 2012. A multi-parameter artificial neural network model to estimate macrobenthic invertebrate productivity and production. *Limnology* and *Oceanography*: *Methods* 10, 581–589.

Brill, E.D., Eheart, J.W., Kshirsagar, S.R., Lence, B.J., 1984. Water quality impacts of biochemical oxygen demand under transferable discharge permit programs. *Water Resources Research* 20, 445-455.

Brown, J.H., Gillooly, J.F., Allen, A.P., Savage, V.M., West, G.B., 2004. Toward a metabolic theory of ecology. *Ecology* 85, 1771-1789.

Bryazgin, V., 1997. Diversity, distribution and ecology of benthic amphipods (Amphipoda, Gammaridea) in the Barents Sea sublittoral *Polish Polar Research* 18, 89-106.

Burton Jr, G.A., 2002. Sediment quality criteria in use around the world. *Limnology* 3, 65-76.

Butman, C.A., 1987. Larval settlement of soft-sediment invertebrates: the spatial scales of pattern explained by active habitat selection and the emerging role of hydrodynamical processes. *Oceanography and Marine Biology - An Annual Review* 25, 113-165.

Cabral-Oliveira, J., Pardal, M.A., 2016. Sewage discharges in oceanic islands: effects and recovery of eulittoral macrofauna assemblages. *Journal of Coastal Conservation* 20, 307-314.

Cardoso, P.G., Pardal, M.A., Lillebø, A.I., Ferreira, S.M., Raffaelli, D., Marques, J.C., 2004. Dynamic changes in seagrass assemblages under eutrophication and implications for recovery. *Journal of Experimental Marine Biology and Ecology* 302, 233-248.

Cartes, J.E., Jaume, D., Madurell, T., 2003. Local changes in the composition and community structure of suprabenthic peracarid crustaceans on the bathyal Mediterranean: influence of environmental factors. *Marine Biology* 143, 745-758.

Carver, R.E., 1971. Procedures in sedimentary petrology. Wiley-Interscience, New York; Sydney. pp.653.

Cattrijsse, A., Mees, J., Hamerlynck, O., 1993. The Hyperbenthic Amphipoda and Isopoda of the Voordelta and the Westerschelde Estuary. *Cahiers De Biologie Marine* 34 187-200.

Census of India, 2011. Population size of Kerala, retrieved December 6, 2004, from Census of India online via access: http://censusindia.gov.in/.

Central Pollution Control Board, 2015a. Directions Under Section 18(1)(b) of the Water (Prevention And Control Of Pollution) Act, 1974 regarding treatment and utilization of sewage, retrived on December 6, 2017 from Census of India online via access: http://cpcb.nic.in/kerala_swg_18(1)(b)_2015.pdf.

Central Pollution Control Board, 2015b. Status of Sewage Treatment in India Central Pollution Control Board Delhi, India. p.101.

Chanda, N.M., Shirodkar, P.V., Singbal, S.Y.S., 1996. Studies on organic carbon nitrogen and phosphorous in the sediments of Mandovi estuary, Goa. *Indian Journal of Geo-Marine Sciences* 25, 120-124.

Chandra Mohan, P., Rao, R.G., Dehairs, F., 1997. Role of Godavari mangroves (India) in the production and survival of prawn larvae. *Hydrobiologia* 358, 317-320.

Chandran, R., 1987. Hydrobiological studies in the gradient zone of the Vellar estuary. I Physicochemical parameters. *Mahasagar- Bullettin of the National Institute of Oceanography* 17, 69-78.

Chaplot, V., Poesen, J., 2012. Sediment, soil organic carbon and runoff delivery at various spatial scales. *Catena* 88, 46-56.

Chapman, P.M., 2007. Do not disregard the benthos in sediment quality assessments! *Marine Pollution Bulletin* 54, 633-635.

Cheriyan, C.J., 1977. Studies on some boring and fouling crustaceans. Ph.D. thesis, Cochin University of Science and Technology, Kochi, India. pp. 158.

Chinnadurai, G., Fernando, O.J., 2006. New records of free-living marine nematodes from on artificial mangrove of India. *Journal of the Marine Biological Association of India* 48, 105-107.

Chintiroglou, C.C., Antoniadou, C., Baxevanis, A., Damianidis, P., Karalis, P., Vafidis, D., 2004. Peracarida populations of hard substrate assemblages in ports of the NW Aegean Sea (eastern Mediterranean). *Helgoland Marine Research* 58, 54-61.

Cho, H.J., 2007. Effects of prevailing winds on turbidity of a shallow Estuary. *International Journal of Environmental Research and Public Health* 4, 185-192.

Chou, W.R., Lai, S.H., Fang, L.S., 1999. Benthic crustacean communities in waters of Southwestern Taiwan and their relationships to environmental characteristics *Acta ZoologicaTaiwanica* 10, 25-33.

Chuks Chindah, A., 1998. The effect of industrial activities on the periphyton community at the upper reaches of New Calabar river, Niger delta, Nigeria. *Water Research* 32, 1137-1143.

Clarke, K.R., 1990. Comparisons of dominance curves. *Journal of Experimental Marine Biology and Ecology* 138, 143-157.

Clarke, K.R., Gorley, R.N., 2001. PRIMER *v*5: user manual/tutorial. PRIMER-E: Plymouth, U.K. pp.91.

Clarke, K.R., Gorley, R.N., 2006. PRIMER *v*6: user manual/tutorial. PRIMER-E, Plymouth, U.K. pp.192.

Clarke, K.R., Warwick, R.M., 1998. A taxonomic distinctness index and its statistical properties. *Journal of Applied Ecology* 35, 523-531.

Clarke, K.R., Warwick, R.M., 2001a. Change in marine communities: an approach to statistical analysis and interpretation Natural Environment Research Council, Plymouth, UK. pp.175.

Clarke, K.R., Warwick, R.M., 2001b. A further biodiversity index applicable to species lists: variation in taxonomic distinctness. *Marine Ecology Progress Series* 216, 265-278.

Cloern, J.E., 1987. Turbidity as a control on phytoplankton biomass and productivity in estuaries. *Continental Shelf Research* 7, 1367-1381.

Cloern, J.E., 1996. Phytoplankton bloom dynamics in coastal ecosystems: A review with some general lessons from sustained investigation of San Francisco Bay, California. *Reviews of Geophysics* 34, 127-168.

Cloern, J.E., Foster, S.Q., Kleckner, A.E., 2014. Phytoplankton primary production in the world's estuarine-coastal ecosystems. *Biogeosciences* 11, 2477-2501.

Connell, J.H., 1978. Diversity in tropical rain forests and coral reefs. *Science* 199, 1302-1310.

Cooper, L.H.N., 1937. The nitrogen cycle in the Sea. *Journal of the Marine Biological Association of the United Kingdom* 22, 183-204.

Costanza, R., 1999. The ecological, economic, and social importance of the oceans. *Ecological Economics* 31, 199-213.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M., 1997. The value of the world's ecosystem services and natural capital. *Nature* 387, 253-260.

Coull, B.C., 1999. Role of meiofauna in estuarine soft-bottom habitats. *Australian Journal of Ecology* 24, 327-343.

Covich, A.P., Austen, M.C., BÄRlocher, F., Chauvet, E., Cardinale, B.J., Biles, C.L., Inchausti, P., Dangles, O., Solan, M., Gessner, M.O., Statzner, B., Moss, B., 2004. The role of biodiversity in the functioning of freshwater and marine benthic ecosystems. *BioScience* 54, 767-775.

Cowie, G.L., Levin, L.A., 2009. Benthic biological and biogeochemical patterns and processes across an oxygen minimum zone (Pakistan margin, NE Arabian Sea). *Deep Sea Research Part II: Topical Studies in Oceanography* 56, 261-270.

Crimaldi, J.P., Thompson, J.K., Rosman, J.H., Lowe, R.J., Koseff, J.R., 2002. Hydrodynamics of larval settlement: The influence of turbulent stress events at potential recruitment sites. *Limnology and Oceanography* 47, 1137-1151.

Crisp, D.J., 1984. Energy flow measurements, in: Holme, N.A., McIntyre, A.D. (Eds.), Methods for the study of marine benthos. Blackwell Scientific Publications, Oxford, pp. 197-279.

Crooks, J.A., 1998. Habitat alteration and community-level effects of an exotic mussel, *Musculista senhousia*. *Marine Ecology Progress Series* 162, 137-152.

Crooks, J.A., 2001. Assessing invader roles within changing ecosystems: Historical and experimental perspectives on an exotic mussel in an urbanized lagoon. *Biological Invasions* 3, 23-36.

Cunha, M.R., Sorbe, J.C., Moreira, M.H., 1999. Spatial and seasonal changes of brackish peracaridan assemblages and their relation to some environmental variables in two tidal channels of the Ria de Aveiro (NW Portugal). *Marine Ecology Progress Series* 190, 69-87.

Dame, R.F., 1993. The role of bivalve filter feeder material fluxes in estuarine ecosystems, in: Dame, R.F. (Ed.), Bivalve Filter Feeders: in Estuarine and Coastal Ecosystem Processes. Springer Berlin Heidelberg, Berlin, Heidelberg, pp. 245-269.

Damodaran, R., 1973. Studies on the benthos of the mud banks of Kerala coasts. Bulletin of the Department of Marine Sciences, University of Cochin VI, 1-126.

Das, S., Mangwani, N., 2015. Ocean acidification and marine microorganisms: responses and consequences. *Oceanologia* 57, 349-361.

Dauer, D.M., 1993. Biological criteria, environmental health and estuarine macrobenthic community structure. *Marine Pollution Bulletin* 26, 249-257.

Dauer, D.M., 1997. Dynamics of an estuarine ecosystem; long-term trends in the macrobenthic communities of Chesapeake Bay. *Oceanologica Acta* 20, 291-298.

Dauer, D.M., Ranasinghe, J.A., Weisberg, S.B., 2000. Relationships between benthic community condition, water quality, sediment quality, nutrient loads, and land use patterns in Chesapeake Bay. *Estuaries* 23, 80-96.

Dauvin, J.-C., 2007. Paradox of estuarine quality: Benthic indicators and indices, consensus or debate for the future. *Marine Pollution Bulletin* 55, 271-281.

Dauvin, J.C., 2008. Effects of heavy metal contamination on the macrobenthic fauna in estuaries: the case of the Seine estuary. *Marine Pollution Bulletin* 57.

Dauvin, J.C., Ruellet, T., 2007. Polychaete/amphipod ratio revisited. *Marine Pollution Bulletin* 55, 215-224.

Day, J.H., 1967. A monograph on the polychaeta of Southern Africa, Part 1 (Errantia) and II (Sedentaria). Trustees of the British Museum (Natural History), London. pp. 1-878.

Day Jr., J.W., Kemp, W.M., Yáñez-Arancibia, A., Crump, B.C., 2012. *Estuarine Ecology*, 2nd ed. Wiley-Blackwell. pp.1-18.

De-La-Ossa-Carretero, J.A., Dauvin, J.C., 2010. A comparison of two biotic indices, AMBI and BOPA, for assessing the ecological quality status (EcoQS) of benthic communities. *Water Transitional Bulletin* 4, 12-24.

De Grave, S., 1999. The influence of sedimentary heterogeneity on within maerl bed differences in infaunal crustacean community. *Estuarine, Coastal and Shelf Science* 49, 153-163.

de Juan, S., Hewitt, J., 2014. Spatial and temporal variability in species richness in a temperate intertidal community. *Ecography* 37, 183-190.

Dederen, L.H.T., 1992. Marine eutrophication in Europe: similarities and regional differences in appearance, in: Vollenweider, R.A., Marchetti, R., Viviani, R. (Eds.), Marine Coastal Eutrophication. Elsevier, Amsterdam. pp. 663-672.

Desai, B.N., Krishnan Kutty, M., 1967. A comparison of the marine and estuarine benthic fauna of the nearshore regions of the Arabian Sea, Symposium on Indian Ocean. *Bulletin of National Institute of Sciences of India*. 38, pp. 677-683.

Desai, B.N., Krishnankutty, M., 1967. Studies on the benthic fauna of Cochin backwater. pp. 123-142.

Desrina, Verreth, J.A.J., Prayitno, S.B., Rombout, J.H.W.M., Vlak, J.M., Verdegem, M.C.J., 2013. Replication of white spot syndrome virus (WSSV) in the polychaete Dendronereis spp. *Journal of Invertebrate Pathology* 114, 7-10.

Dev Roy, M.K., 2012. An updated systematic list of isopod fauna of India. *Journal of Environment and Sociobiology* 9, 163-175.

Dev Roy, M.K., Mithra, S., Pati, S., 2012. Crustacea: Isopoda. Fauna of Maharashtra State Fauna Series. 20, 387-393.

Dev Roy, M.K., Nandi, N.C., 2010. Checklist and distribution of marine and estuarine isopod crustaceans from India, Proceedings of Professor P. N. Ganapati Centennial Symposium on "Landmark Researches in Marine Biology". Marine Biological Association of India and Department of Zoology, Andhra University. p. 23.

Deyzel, H.P., 2012. Mesozooplankton dynamics in a Biogeographical Transition Zone Estuary. Nelson Mandela Metropolitan University. p. 322.

Diaz, R.J., Rosenberg, R., 1995. Marine benthic hypoxia: a review of its ecological effects and the behavioral responses of benthic macrofauna *Oceanography and Marine Biology: An Annual Review* 33, 245-303.

Diaz, R.J., Rosenberg, R., 2008. Spreading dead zones and consequences for marine ecosystems. *Science* 321, 926-929.

Diaz, R.J., Solan, M., Valente, R.M., 2004. A review of approaches for classifying benthic habitats and evaluating habitat quality. *Journal of Environmental Management* 73, 165-181.

Dietz, T.H., Shawn, J.W., Byrne, R.A., Lynn, J.W., Silverman, H., 1996. Osmotic and ionic regulation of north american zebra mussels (*Dreissena polymorpha*). *American Zoologist* 36, 364-372.

Divakaran, O., Murugan, T., Nair, N.B., 1981. Distribution and seasonal variation of the benthic fauna of the Ashtarnudi Lake, South west coast of India. *Mahasagar- Bullettin of the National Institute of Oceanography* 14, 167-172.

Dolbeth, M., Cusson, M., Sousa, R., Pardal, M.A., 2012. Secondary production as a tool for better understanding of aquatic ecosystems. *Canadian Journal of Fisheries and Aquatic Sciences* 69, 1230-1253.

Dolbeth, M., Lillebø, A.I., Cardoso, P.G., Ferreira, S.M., Pardal, M.A., 2005. Annual production of estuarine fauna in different environmental conditions: An evaluation of the estimation methods. *Journal of Experimental Marine Biology and Ecology* 326, 115-127.

Domingues, R.B., Barbosa, A., Galvão, H., 2005. Nutrients, light and phytoplankton succession in a temperate estuary (the Guadiana, south-western Iberia). *Estuarine, Coastal and Shelf Science* 64, 249-260.

Donald, M.S., Michael, E., 2004. The estuarine ecosystem: ecology, threats, and management. Oxford OX26DP, Great Clarendon Street. pp. 224.

Dutertre, M., Hamon, D., Chevalier, C., Ehrhold, A., 2012. The use of the relationships between environmental factors and benthic macrofaunal distribution in the establishment of a baseline for coastal management. *ICES Journal of Marine Science* 70, 294-308.

Eaton, A.D., Franson, M.A.H., American Public Health Association, American Water Works Association, Water Environment Federation, 2005. Standard methods for examination of water and wastewater: Centennial edition. American Public Health Association.

El-Serehy, H.A., Al-Misned, F.A., Al-Rasheid, K.A., 2015. Population fluctuation and vertical distribution of meiofauna in the Red Sea interstitial environment. *Saudi Journal of Biological Sciences* 22, 459-465.

El-Wakeel, S.K., Riley, J.P., 1957. The determination of organic carbon in marine muds. *ICES Journal of Marine Science* 22, 180-183.

Eleftherioo, A., Mc Intyre, A., 2005. Methods for the study of marine benthos. 3rd Edition, Blackwell Science Publishers Oxford pp.418.

Eleftherioo, A., 2013. Methods for the study of marine benthos. 4th Edition, John Wiley & Sons, Ltd. pp.496.

Elias, R., Rivero, M.S., Palacios, J.R., Vallarino, E.A., 2006. Sewage-induced disturbance on Polychaetes inhabiting intertidal mussel beds of *Brachidontes rodriguezii* off Mar del Plata (Southwestern Atlantic, Argentina). *Scientia Marina* 70, 187-196.

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

Emerson, C.W., 1991 A method for the measurement of bedload sediment transport and passive faunal transport on intertidal sandfats. *Estuaries* 14, 361-371.

Eyre, B., 1998. Transport, retention and transformation of material in Australian estuaries. *Estuaries* 21, 540-551.

Falkowski, P., Kiefer, D.A., 1985. Chlorophyll a fluorescence in phytoplankton: relationship to photosynthesis and biomass. *Journal of Plankton Research* 7, 715-731.

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

Fauchald, K., 1984. Polychaete distribution patterns, or: can animals with Palaeozoic cousins show large-scale geographic patterns, In: Proceedings of the First International Polychaete Conference, Sydney, Australia. pp. 1-6.

Fauchald, K., Jumars, P.A., 1979. The Diet of Worms: A Study of Polychaete Feeding Guilds, in: Barnes, M. (Ed.), Oceanography and Marine Biology - An Annual Review. Aberdeen University Press. pp. 193-284.

Fauvel, P., 1953. The Fauna of Indian including Pakistan, Ceylon, Burma and Malaya : Annelid Polychaeta. The Indian Press Ltd. pp.507.

Federal Geographic Data Committee, 2013. Classification of wetlands and deepwater habitats of the United States, 2nd ed. Wetlands Subcommittee, Federal

Geographic Data Committee and U.S. Fish and Wildlife Service, Washington, D.C. pp.86.

Feebarani, J., 2009. Meiobenthos of Cochin backwaters in relation to macrobenthos and environmental parameters. Ph.D. Thesis, Cochin University of Science and Technology, Kochi. pp. 238.

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

Fernando, S.A., Ajmal Khan, S., Kasinathan, R., 1984. Observations on the distribution of benthic fauna in Vellar estuary, Porto Novo. *Mahasagar - Bulletin of the National Institute of Oceanography* 16, 341-348.

Fiedler, S., Vepraskas, M.J., Richardson, J.L., 2007. Soil redox potential: Importance, field measurements, and observations. *Advances in Agronomy* 94, 1-54.

Flemming, B.W., 2000. A revised textural classification of gravel-free muddy sediments on the basis of ternary diagrams. *Continental Shelf Research* 20, 1125-1137.

Flint, R.W., Younk, J.A., 1983. Estuarine benthos: Long-term community structure variations, Corpus Christi Bay, Texas. *Estuaries* 6, 126-141.

Food and Agriculture Organization, 2010. Hydrographic surveys, in: Sciortino, J.A. (Ed.), Fishing harbour planning, construction and management. Food and Agriculture Organization of the United Nations Rome. pp. 49-64.

Frankel, L., Meade, D.J., 1973. Mucilaginous matrix of some estuarine sands in Connecticut. *Journal of Sedimentary Petrology* 43, 1090-1095.

Fresi, E., Gambi, M.C., Focardi, S., Bargagli, R., Baldi, F., Falciai, L., 1983. Benthic community and sediment types: A structural analysis. *Marine Ecology* 4, 101-121.

Fujii, T., 2012. Climate change, sea-level rise and implications for coastal and estuarine shoreline management with particular reference to the ecology of intertidal benthic macrofauna in NW Europe. *Biology* 1, 597.

Gabler, C.A., Osland, M.J., Grace, J.B., Stagg, C.L., Day, R.H., Hartley, S.B., Enwright, N.M., From, A.S., McCoy, M.L., McLeod, J.L., 2017. Macroclimatic change expected to transform coastal wetland ecosystems this century. *Nature Climate Change* 7, 142-147.

Gallagher, E.D., 2008. Benthic feeding guilds and functional groups. http://ocw.umb.edu/environmental-earth-and-ocean-sciences/eeos-630-biological-oceanography/course-schedule.html.

Gambi, M.C., Lorenti, M., Russo, G.F., Scipione, M.B., Zupo, V., 1992. Depth and seasonal distribution of some groups of the vagile fauna of the *Posidonia oceanica* leaf stratum: structural and trophic analyses. *Marine Ecology* 13, 17-39.

Ganapathi, P.N., Raman, A.V., 1970. *Capitella capitata* (Fabricius, 1780) (Polychaete : capitellidae) : An indicator of pollution in Visakhapatnam Harbour. *Indian Journal of Geo-Marine Sciences* 5, 251.

Garcia, S., le Reste, L., Rome, 1981. Life cycles dynamics, exploitation and management of coastal penaeid shrimp stocks, FAO Fisheries Technical Paper 203, pp. 215

Gardner, R.C., Davidson, N.C., 2011. The Ramsar Convention, in: LePage, B.A. (Ed.), Wetlands: Integrating Multidisciplinary Concepts. Springer Netherlands, Dordrecht. pp. 189-203.

Gaston, G.R., 1987. Benthic polychaeta of the Middle Atlantic Bight: feeding and distribution. *Marine Ecology Progress Series* 36, 251-262.

Gattuso, J.P., Frankignoulle, M., Wollast, R., 1998. Carbon and carbonate metabolism in coastal aquatic ecosystems. *Annual Review of Ecology and Systematics* 29, 405-434.

Geetha, P.N., Bijoy Nandan, S., 2014. Ecology, diversity, and abundance of macrobenthic crustaceans in Cochin estuary, India. *Research Journal of Recent Sciences* 3, 137-148

Geetha, P.N., Bijoy Nandan, S., Thasneem, T.A., 2015. Community ecology and taxonomy of polychaetes from a tropical estuary, south west coast of India, in: Gupta, V.K., Anil, K.V. (Eds.), Animal Diversity, Natural History and Conservation Daya Publishing House: A Division of Astral International Pvt. Ltd. pp. 15-40.

Geetha, P.N., Thasneem, T.A., Bijoy Nandan, S., 2010. Macrobenthos and its relation to ecosystem dynamics in the Cochin estuary, in *proceedings of Lake 2010: Wetlands, Biodiversity and Climate Change* 1-12.

George, T., 1979. Waste water engineering treatment disposal reuse- Metcalf and Eddy. Inc. Publication, New Delhi. pp.920.

Gesteira, J.L.G., Dauvin, J.C., 2000. Amphipods are good bioindicators of the impact of oil spills on soft-bottom macrobenthic communities. *Marine Pollution Bulletin* 40, 1017-1027.

Giangrande, A., Licciano, M., Musco, L., 2005. Polychaetes as environmental indicators revisited. *Marine Pollution Bulletin* 50, 1153-1162.

Giere, O., 2008. Meiobenthology: The microscopic motile fauna of aquatic sediments, 2nd ed. Springer. Pp.527.

Gili, J.-M., Coma, R., 1998. Benthic suspension feeders: their paramount role in littoral marine food webs. *Trends in Ecology and Evolution* 13, 316-321.

Glasby, C., Timm, T., 2008. Global diversity of polychaetes (Polychaeta; Annelida) in freshwater, in: Balian, E.V., Leveque, C., Segers, H., Martens, K. (Eds.), Freshwater Animal Diversity Assessment. Springer Netherlands. pp. 107-115.

Glasby, C.J., Mogi, M., Takahashi, K., 2003. Occurrence of the polychaete *Namalycastis hawaiiensis* Johnson, 1903 (Nereididae: Namanereidinae) in Pandanus leaf axils on Palau, West Pacific. The Beagle. *Records of the Museums and Art Galleries of the Northern Territory* 19, 97-99.

Glé, C., Del Amo, Y., Sautour, B., Laborde, P., Chardy, P., 2008. Variability of nutrients and phytoplankton primary production in a shallow macrotidal coastal ecosystem (Arcachon Bay, France). *Estuarine, Coastal and Shelf Science* 76, 642-656.

Gogina, M., Glockzin, M., Zettler, M.L., 2010. Distribution of benthic macrofaunal communities in the western Baltic Sea with regard to near-bottom environmental parameters. 1. Causal analysis. *Journal of Marine Systems* 79, 112-123.

Gopalan, U.K., Meenakshikunjamma, P.P., Vengayil, D.T., 1987. Macrobentos of Vembanad estuary in relation to the deposition of degraded water fern Salvinia and other macrophytes, in: Nair, N.B. (Ed.), Proceedings of national seminar on Estuarine Management, Trivandrum, pp. 410-418.

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

Granek, E.F., Polasky, S., Kappel, C.V., Reed, D.J., Stoms, D.M., Koch, E.W., Kennedy, C.J., Cramer, L.A., Hacker, S.D., Barbier, E.B., Aswani, S., Ruckelshaus, M., Perillo, G.M.E., Silliman, B.R., Muthiga, N., Bael, D., Wolanski, E., 2010. Ecosystem services as a common language for coastal ecosystem-based management los servicios del ecosistema como un lenguaje común para el manejo de costas basado en ecosistemas. *Conservation Biology* 24, 207-216.

Grant, J., 1981. Dynamics of competition among estuarine sandburrowing amphipods *Journal of Experimental Marine Biology and Ecology* 49 255-265.

Grasshoff, K., 1983. Determination of oxygen, in: Grasshoff, K., Ehrhardt, M., Kremling, K. (Eds.), Methods of sea water analysis. Verlag Chemie, Weinheim pp.419.

Grasshoff, K., Ehrhard, M., Kremling, K., 1999. Methods of seawater analysis. VCH publishers, Wainheins, Germany pp.632.

Grasshoff, K., Ehrhardt, M., Kremling, K., 1983. Methods of sea water analysis. Weinheim: Verlag Chemie pp.419.

Gray, J.S., 1997. Marine biodiversity: patterns, threats and conservation needs. *Biodiversity and Conservation* 6, 153-175.

Gray, J.S., Poore, G.C.B., Ugland, K.I., wilson, R.S., Olsgard, F., Johannessen, O., 1997. Coastal and deep-sea benthic diversities compared. *Marine Ecology Progress Series* 159, 97-103.

Griffiths, J.R., Kadin, M., Nascimento, F.J.A., Tamelander, T., Törnroos, A., Bonaglia, S., Bonsdorff, E., Brüchert, V., Gårdmark, A., Järnström, M., Kotta, J., Lindegren, M., Nordström, M.C., Norkko, A., Olsson, J., Weigel, B., Žydelis, R., Blenckner, T., Niiranen, S., Winder, M., 2017. The importance of benthic–pelagic coupling for marine ecosystem functioning in a changing world. *Global Change Biology* 23, 2179-2196.

Gulland, J.A., 1971. The fish resources of the ocean England. Food and Agriculture Organization, Fishing News (Books) Ltd., England. pp.255.

Gupta, G.V.M., Thottathil, S.D., Balachandran, K.K., Madhu, N.V., Madeswaran, P., Nair, S., 2009. CO_2 supersaturation and net heterotrophy in a tropical estuary (Cochin, India): influence of anthropogenic effect. *Ecosystems* 12, 1145-1157.

Gurumayum, S.D., 2015. Estuarine molluscs of Pennera River, Andhra Pradesh. *Advances in Applied Science Research* 6, 207-210.

Haeckel, E., 1891. Plankton Studien, *Jena Z. Naturwis*. 25, 232–336.

Hakenkamp, C.C., Morin, A., 2000. The importance of meiofauna to lotic ecosystem functioning. *Freshwater Biology* 44, 165-175.

Halpern, B.S., Walbridge, S., Selkoe, K.A., Kappel, C.V., Micheli, F., D'Agrosa, C., Bruno, J.F., Casey, K.S., Ebert, C., Fox, H.E., Fujita, R., Heinemann, D., Lenihan, H.S., Madin, E.M.P., Perry, M.T., Selig, E.R., Spalding, M., Steneck, R., Watson, R., 2008. A global map of human impact on marine ecosystems. *Science* 319, 948-952.

Harkantra, S.N., 1975. Benthos of the Kali estuary, Karwar. *Mahasagar- Bullettin of the National Institute of Oceanography* 8, 53-58.

Harkantra, S.N., A.H. Parulekar, 1981. Ecology of benthic production in the coastal zone of Goa. *Mahasagar- Bullettin of the National Institute of Oceanography* 14, 135-139.

Harkantra, S.N., Ayyappan Nair, Anzari, Z.A., Parulekar, A.H., 1980. Benthos of the shelf regions along the west coast of India. *Indian Journal of Geo-Marine Sciences* 9, 106-110.

Harkantra, S.N., Rodrigues, C.L., Parulekar, A.H., 1982. Macrobenthos of the shelf off northeastern Bay of Bengal. *Indian Journal of Geo-Marine Sciences* 11, 115-121.

Harkantra, S.N., Rodrigues, N.R., 2004. Environmental influences on the species diversity, biomass and population density of soft bottom macrofauna in the estuarine system of Goa, west coast of India. *Indian Journal of Geo-Marine Sciences* 33, 187-193.

Hartman, O., 1974. Polychaetous annelids from the Indian Ocean including an account of species collected by members of the International Indian Ocean Expendition,1963-64 and a catalogue and bibilography of species from India. *Journal of the Marine Biological Association of India Part I&II* 16(11), 191-252 and 116(192), 609-644.

Heip, C., Herman, P.M.J., 1995. Major biological processes in European tidal estuaries: a synthesis of the JEEP-92 Project. *Hydrobiologia* 311, 1-7.

Herman, P.M.J., Middelburg, J.J., Heip, C.H.R., 2001. Benthic community structure and sediment processes on an intertidal flat: results from the ECOFLAT project. *Continental Shelf Research* 21, 2055-2071.

Herman, P.M.J., Middelburg, J.J., Koppel, J.V.D., Heip, C.H.R., 1999. Ecology of estuarine macrobenthos, Estuaries. *Advances in Ecological Research*, pp. 195-240.

Hermand, R., Salen-Picard, C., Alliot, E., Degiovanni, C., 2008. Macrofaunal density, biomass and composition of estuarine sediments and their relationship to the river plume of the Rhone River (NW Mediterranean). *Estuarine, Coastal and Shelf Science* 79, 367-376.

Hily, C., Le Loc'h, F., Grall, J., Glémarec, M., 2008. Soft bottom macrobenthic communities of North Biscay revisited: long-term evolution under fisheries-climate forcing. *Estuarine, Coastal and Shelf Science* 78, 413-425.

Holme, N.A., Mc Intyre, A.O., 1971. Methods of study of marine benthos. Blackwell Scientific Publications. pp.346.

Hou, D., He, J., Lü, C., Sun, Y., Zhang, F., Otgonbayar, K., 2013. Effects of environmental factors on nutrients release at sediment-water interface and assessment of trophic status for a typical shallow lake, northwest China. *The Scientific World Journal* 1-16. http://dx.doi.org/10.1155/2013/716342

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

Imajima, M., 1990a. Spionidae (Annelida, Polychaeta) from Japan: II. The Genus Prionospio (Aquilaspio). *Bulletin of the National Science Museum, Tokyo, Series A* 8(1), 1-13.

Imajima, M., 1990b. Spionidae (Annelida, Polychaeta) from Japan: III. The Genus Prionospio (Minuspio). *Bulletin of the National Science Museum, Tokyo, Series A* 16(2), 61-78, 1990-06.

Imajima, M., 1990c. Spionidae (Annelida, Polychaeta) from Japan: IV. The Genus Prionospio (Prionospio). *Bulletin of the National Science Museum, Tokyo, Series A (Zoology).* 16(3): 105-140.

Imajima, M., 1992a. Spionidae (Annelida, Polychaeta) from Japan: IX. The Genus Aonidella. *Bulletin of the National Science Museum. Series A, Zoology* 18(2), 57-61, 1992-06

Imajima, M., 1992b. Spionidae (Annelida, Polychaeta) from Japan: VIII. The Genus Scolelepis. *Bulletin of the National Science Museum. Series A, Zoology* 18(1), 1-34, 1992-03

Ingole, B.S., 2004. Bottom dwelling animala - Benthos, in: Untawale, A.G. (Ed.), Know our shore. National Institute of Oceanography, Goa, India, pp. 83-95.

Ingole, B.S., Parulekar, A.H., 1998. Role of salinity in structuring the intertidal meiofauna of a tropical estuarine beach: Field evidence. *Indian Journal of Marine Sciences* 27 (3&4) 356-361

Iriarte, A., Purdie, D.A., 1994. Size distribution of chlorophyll *a* biomass and primary production in a temperate estuary (Southampton Water): the contribution of photosyntheticpicoplankton. *Marine Ecology Progress Series* 115, 283-297.

Jagtap, T.G., Untawale, A.G., Inamdar, S.N., 1994. Study of mangrove environment of Maharashtra coast using remote sensing data. *Indian Journal of Geo-Marine Sciences* 23, 90.

Jan, A.P., 1999. On the advantages and disadvantages of larval stages in benthic marine invertebrate life cycles. *Marine Ecology Progress Series* 177, 269-297.

Jayachandran, P.R., Bijoy Nandan, S., 2012. Assessment of trophic change and its probable impact on tropical estuarine environment (the Kodungallur-Azhikode estuary, India). *Mitigation and Adaptation Strategies for Global Change* 17, 837-847.

Jayachandran, P.R., Bijoy Nandan, S., Sanu, V.F., Anu, P.R., Jima, M., Midhun, A.M., Don Xavier, N.D., Philomina, J., Asha, C.V., 2018. Authentication of *Nassodonta insignis* H. Adams, 1867 (Gastropoda: Nassariidae) from the Kodungallur-Azhikode backwater, southwest coast of India using mitochondrial COI marker. *Indian Journal of Geo-Marine Sciences* 47 (3), 623-628.

Jayachandran, P.R., Bijoy Nandan, S., Sreedevi, O.K., 2012. Water quality variations and nutrient characteristics of Kodungallur-Azhikode Estuary, Kerala, India. *Indian Journal of Geo-Marine Sciences* 41, 180-187.

Jayachandran, P.R., Bijoy Nandan, S., Sreedevi, O.K., Sanu, V.F., 2013. Influences of Environmental Factors on Fish Assemblage in the Tropical Estuary of South West Coast of India, A Case Study of Kodungallur-Azhikode Estuary. *International Journal of Marine Science* 3, 4-16.

Jayachandran, P.R., Prabhakaran, M.P., Asha, C.V., Akhilesh Vijay, Bijoy Nandan, S., 2015. First report on mass reproductive swarming of a polychaete worm, *Dendronereis aestuarina* (Annelida, Nereididae) Southern 1921, from a freshwater environment in the south west coast of India. *International Journal of Marine Science* 5(3) 1-7.

Jayalakshmy, K.V., Kameswara Rao, K., 2004. Macrobenthos in relation to sediment characteristics of nearshore waters of Chitrapur, west coast of India receiving industrial effluents. Asian Fisheries Society, Manila, Philippines. 17, 21-28.

Jia Zhong, Z., Fischer, C.J., 2006. A simplified resorcinol method for direct spectrophotometric determination of nitrate in seawater. *Marine Chemistry* 99, 220-226.

Jickells, T.D., Andrews, J.E., Parkes, D.J., Suratman, S., Aziz, A.A., Hee, Y.Y., 2014. Nutrient transport through estuaries: the importance of the estuarine geography. *Estuarine, Coastal and Shelf Science* 150, 215-229.

Johannesse, T., Dahl, E., 1996. Declines in oxygen concentrations along the Norwegian Skagerrak coast, 1927-1993: A signal of ecosystem changes due to eutrophication? *Limnology and Oceanography* 41, 766-778.

John, F., 2009. Meiobenthos of Cochin back waters in relation to macrobenthos and environmental parameters Cochin university of Science and Technology pp. 1-204.

John, S.G., Rudolf Shiu-sun, W., Ying Ying, O., 2002. Effects of hypoxia and organic enrichment on the coastal marine environment. *Marine Ecology Progress Series* 238, 249-279.

Johnson, D., 2008. Environmental indicators: their utility in meeting the OSPAR Convention's regulatory needs. *ICES Journal of Marine Science* 65, 1387-1391.

Johnson, R.G., 1971. Animal-sediment relations in shallow water benthic communities. *Marine Geology* 11, 93-104.

Jolliffe, I.T., 2002. Principal Component Analysis. Springer. pp.487.

Jones, S.E., Jago, C.F., 1993. In situ assessment of modification of sediment properties by burrowing invertebrates. *Marine Biology* 115, 133-142.

Joseph, K.J., 1988. Studies on some aspects of phytoplankton. Cochin University of Science and Technology, Cochin, India. p. 217.

Joseph, S., Ouseph, P.P., 2010. Assessment of nutrients using multivariate statistical techniques in estuarine systems and its management implications: a case study from Cochin Estuary, India. *Water and Environment Journal* 24, 126-132.

Jumars, P.A., Nowell, A.R.M., 1984. Fluid and sediment dynamic effects on marine benthic community structure. *American Zoologist* 24, 45-55.

Jumars, P.A., Dorgan, K.M., Lindsay, S.M., 2015. Diet of worms emended: an update of polychaete feeding guilds. *Annual Review of Marine Science* 7, 497-520.

Jyothibabu, R., Madhu, N.V., Jayalakshmi, K.V., Balachandran, K.K., Shiyas, C.A., Martin, G.D., Nair, K.K.C., 2006. Impact of fresh water influx on micro zooplankton mediated food web in a tropical estuary (Cochin backwaters), India. *Estuarine, Coastal and Shelf Science* 69 (3&4) 505-518.

Karr, J.R., 1991. Biological integrity: a long-neglected aspect of water resource management *Ecological Applications* 1, 66-84.

Kelly, J.R., Levin, S.A., 1986. A comparison of aquatic and terrestrial nutrient cycling and production processes in natural ecosystems, with reference to ecological concepts of relevance to some waste disposal issues, in: Kullenberg, G. (Ed.), The Role of the Oceans as a Waste Disposal Option. Springer Netherlands, Dordrecht. pp. 165-203.

Kennedy, A.D., Jacoby, C.A., 1999. Biological indicators of marine environmental health: meiofauna- a neglected benthic component? *Environmental Monitoring and Assessment* 54, 47-68.

Kennish, M.J., 2002. Environmental threats and environmental future of estuaries. *Environmental Conservation* 29, 78-107.

Kennish, M.J., Townsend, A.R., 2007. Nutrient Enrichment and Estuarine Eutrophication1. *Ecological Applications* 17, S1-S2.

Kensely, B., 2001. Biogeography of the marine isopoda of the Indian ocean with a checklist of species and records in Isopod systematics and evolution, Crustacean Issue. pp. 205-264

Ketchum, B.H., 1951. The exchange of fresh and salt waters in tidal estuaries. *Journal of Marine Research* 10, 18-38.

Kimor, B., 1991. Changes and stress signs in plankton communities as a result of man-induced perturbations in enclosed coastal seas (Mediterranean, Baltic). *Marine Pollution Bulletin* 23, 171-174.

Kinne, O., 1963. The effects of temperature and salinity or marine and brackish water animals. *Oceanography and Marine Biology: Annual Review* 1, 301-340.

Kinne, O., 1978. Marine Ecology (5 volumes) John Wiley, New York.

Klause, M.D., 1986. Mucous secretions of the acoel turbellarian *Convoluta* sp. Orsted: an ecological and functional approach. *Journal of Experimental Marine Biology and Ecology* 97, 123-133.

Knox, G.A., 2006. Biology of the Southern Ocean, 2nd ed. CRC Press pp.640.

Koroleff, F., 1969. International Council for the Exploration of the Sea, C.M. I969/C, 9, 4.

Koroleff, F., 1970. International Council for the Exploration of the Sea, Interlab. Rep., 3,19.

Kristensen, E., 1988. Benthic fauna and biogeochemical processes in marine sediments: microbial activities and fluxes, in: Blackburn, T.H., Ssrensen, J. (Eds.), Nitrogen cycling in coastal marine environments. John Wiley and Sons Ltd., Chichester. pp. 275-299

Kumar, P.S., Khan, A.B., 2013. The distribution and diversity of benthic macroinvertebrate fauna in Pondicherry mangroves, India. *Aquatic Biosystems* 9, 15-15.

Kurbjeweit, F., Schmiedl, G., Schiebel, R., Hemleben, C., Pfannkuche, O., Wallmann, K., Schäfer, P., 2000. Distribution, biomass and diversity of benthic foraminifera in relation to sediment geochemistry in the Arabian Sea. *Deep Sea Research Part II: Topical Studies in Oceanography* 47, 2913-2955.

Kurian, C.V., 1953. Preliminary survey of the bottom fauna and bottom deposits of the Travancore coast within the 15 fathom line, *Proceedings of the National Academy of Sciences, India*, pp. 746-775.

Kurian, C.V., 1967. Studies on the benthos of the Southwest coast of India. *Bulletin of National Institute of Sciences of India* 30, 649-656.

Kurian, C.V., 1972. Ecology of benthos in a tropical estuary. *Proceedings of the Indian National Science Academy* 38, 156-163.

Kurian, C.V., Damodaran, R., Antony, A., 1975. Bottom fauna of the Vembanad Lake. Bulletin of the Department of Marine Sciences, University of Cochin. 7, 987-994.

Kusum, K.K., Vineetha, G., Raveendran, T.V., Muraleedharan, K.R., Nair, M., Achuthankutty, C.T., 2011. Impact of oxygen-depleted water on the vertical distribution of chaetognaths in the northeastern Arabian Sea. *Deep Sea Research Part I: Oceanographic Research Papers* 58, 1163-1174.

Kuttyama, V.J., 1973. Observation on the food and feeding of some penaeid prawns of Cochin Area. *Journal of the Marine Biological Association of India* 15, 189-194

Lallu, K.R., Fausia, K.H., Vinita, J., Balachandran, K.K., Naveen Kumar, K.R., Rehitha, T.V., 2014. Transport of dissolved nutrients and chlorophyll a in a tropical estuary, southwest coast of India. *Environmental Monitoring and Assessment* 186, 4829-4839.

Lambshead, P.J.D., Platt, H.M., Shaw, K.M., 1983. The detection of differences among assemblages of marine benthic species based on an assessment of dominance and diversity. *Journal of Natural History* 17, 859-874.

Lauff, G.H., 1967. Estuaries. American Association for the Advancement of Science 83, 1-757.

Laximilatha, P., Velayudhan, T.S., Kripa, V., Sharma, J., Alloycious, P.S., 2005. Biology of the black clam, *Villorita cyprinoides* (Gray) in the backwater of Vembanad Lake. *Indian Journal of Fisheries* 52, 361-366.

Lenihan, H.S., 1999. Physical-biological coupling on oyster reefs: How habitat structure influences individual performance *Ecological Monographs* 69, 251-275.

Levin, S.A., 1992. The problem of pattern and scale in ecology: the Robert H. Mac Arthur award lecture. *Ecology* 73, 1943-1967.

Levinton, J., 1995a. Bioturbators as ecosystem engineers: control of the sediment fabric, inter-individual interactions, and material fluxes, in: Jones, C.G., Lawton, J.H. (Eds.), Linking Species and Ecosystems. Springer US, Boston, MA. pp.29-36.

Levinton, J.S., 1995b. Marine biology: function, biodiversity, ecology. Oxford University Press, New York. pp.420.

Levinton, J.S., 2013. Deposit feeding and coastal oceanography, Ecology of Marine Deposit Feeders. Springer-Verlag. pp. 1-23.

Levinton, J.S., 2014. Marine biology: function, biodiversity, ecology. Oxford University Press, New York. pp.516.

Lewis, D., 1984. Practical scientology, Huctchinson Ross publishing company.New York. pp.227.

Li, L., Zheng, B., Liu, L., 2010. Biomonitoring and bioindicators used for river ecosystems: Definitions, approaches and trends. *Procedia Environmental Sciences* 2, 1510-1524.

Little, S., Wood, P.J., Elliott, M., 2016. Quantifying salinity-induced changes on estuarine benthic fauna: The potential implications of climate change. *Estuarine, Coastal and Shelf Science* 198 (B) pp. 610-625

Lotze, H.K., Lenihan, H.S., Bourque, B.J., Bradbury, R.H., Cooke, R.G., Kay, M.C., Kidwell, S.M., Kirby, M.X., Peterson, C.H., Jackson, J.B.C., 2006. Depletion, degradation, and recovery potential of estuaries and coastal seas. *Science* 163, 1806-1809.

Lowry, J.K., Myers, A.A., 2013. A phylogeny and classification of Senticaudata subord. nov. (Crustacea: Amphipoda). *Zootaxa* 3610 1-80.

Luckenbach, M.W., Huggett, D.V., Zobrist, E.C., 1988. Sediment transport, biotic modifications, and selection of grain size in a surface deposit-feeder. *Estuaries* 11, 134-139.

Ludwig, J.A., Reynolds, J.F., 1988. Statistical ecology. John Wiley & Sons, New York pp.337.

Macdonald, T.A., Burd, B.J., Macdonald, V.I., van Roodselaar, A., 2010. Taxonomic and feeding guild classification for the marine benthic macroinvertebrates of the Strait of Georgia, British Columbia, *Canadian Technical Report of Fisheries and Aquatic Sciences.* 2874: 1-63.

Maciolek, N.J., 1985. A revision of the genus *Prionospio* Malmgren, with special emphasis on species from the Atlantic Ocean, and new records of species belonging to the genera *Apoprionospio* Foster and *Paraprionospio* Caullery (Polychaeta, Annelida, Spionidae). *Zoological journal of the Linnean Society* 84, 325-383.

Macnae, W., 1969. Adventures in Marine Biology. Academic press London and New York 6, 73.

MacPherson, T.A., Cahoon, L.B., Mallin, M.A., 2007. Water column oxygen demand and sediment oxygen flux: patterns of oxygen depletion in tidal creeks. *Hydrobiologia* 586, 235-248.

Madhu, N.V., Jyothibabu, R., Balachandran, K.K., 2010. Monsoon induced changes in the size fractionated phytoplankton biomass and production rate in the estuarine and coastal waters of south west coast of India. *Environmental Monitoring and Assessment* 166, 521-528.

Madhu, N.V., Jyothibabu, R., Balachandran, K.K., Honey, U.K., Martin, G.D., Vijay, J.G., Shiyas, C.A., Gupta, G.V.M., Achuthankutty, C.T., 2007. Monsoonal impact on planktonic standing stock and abundance in a tropical estuary (Cochin backwaters, India). *Estuarine, Coastal and Shelf Science* 73, 54-64.

Madhyastha, A., 2011. *Villorita cyprinoides*, The IUCN Red List of Threatened Species 2011: e.T175076A7112167, http://dx.doi.org/10.2305/IUCN.UK.2011-1.RLTS.T175076A7112167.en.

Magesh, M., Kvist, S.B., Glasby, C.J., 2012. Description and phylogeny of Namalycastis jaya sp. n. (Polychaeta, Nereididae, Namanereidinae) from the southwest coast of India. *ZooKeys* 238, 31-43.

Mahapatra, A., 2001. Molluscan fauna of Godavari estuary: Estuarine ecosystem series, Zoological Survey of India. 4, 52-82.

Mahapatra, A., 2008. Mollusca, fauna of Krishna estuary: Estuarine ecosystem series, Zoological Survey of India. 5, 105-173.

Mahapatro, D., Panigrahy, R.C., Naik, S., Pati, S.K., Samal, R.N., 2011. Macrobenthos of shelf zone off Dhamara estuary, Bay of Bengal. *Journal of Oceanography and Marine Science* 2, 32-42.

Mäkelä, A., Witte, U., Archambault, P., 2017. Benthic macroinfaunal community structure, resource utilisation and trophic relationships in two Canadian Arctic Archipelago polynyas. *PLoS ONE* 12, e0183034.

Malinverno, A., Martinez, E.A., 2015. The effect of temperature on organic carbon degradation in marine sediments. 5, 17861.

Manikandavelu, D., Ramdhas, V., 1994. Bioproduction dynamics of a mangal bordered brackishwater in Tuticorin coast. *Indian Journal of Geo-Marine Sciences* 23, 108-109.

Mare, M.F., 1942. A study of a marine benthic community with special reference to the micro-organisms. *Journal of the Marine Biological Association of the United Kingdom* 25, 517-554.

Margalef, D.R., 1958. Information theory in ecology. *International Journal of General Systems* 3, 36-71.

MarLIN, 2017. Marine Life Information Network. Marine Biological Association of the UK. Available online http://www.marlin.ac.uk.

Marques, J.C., Maranhão, P., Pardal, M.A., 1993. Human impact on the subtidal macrobenthic community structure in the Mondego estuary (western Portugal). *Estuarine, Coastal and Shelf Science* 37, 403-419.

Martin-Jézéquel, V., Hildebrand, M., Brzezinski, M.A., 2000. Silicon metabolism in diatoms: Implications for growth *Journal of Phycology* 36, 821-840.

Martin, D.F., 1970. Marine Chemistry. Theory and application. Marcel Dekker Inc., New York. pp.280.

Martin, G., Nisha, P., Balachandran, K., Madhu, N., Nair, M., Shaiju, P., Joseph, T., Srinivas, K., Gupta, G., 2011. Eutrophication induced changes in benthic community structure of a flow-restricted tropical estuary (Cochin backwaters), India. *Environmental Monitoring and Assessment* 176, 427-438.

Mascarenhas, A., Chauhan, O.S., 1998. A note on the ancient mangroves of Goa, central west coast of India. *Indian Journal of Geo-Marine Sciences* 27, 473-476.

Mathew, K.J., Selvaraj, G.S.D., T.S., N., Varghese, M., Sridhar, N., 1994. Studies on the infestation of an isopod crustacean, *Cirolana fluviatius* in some parts of the Cochin backwaters, Kerala. Marine Fisheries Information Service, Technical and Extension Series 131, 1-10.

Maurer, D., Nguyen, H., Robertson, G., Gerlinger, T., 1999. The Infaunal Trophic Index (ITI): Its Suitability for Marine Environmental Monitoring. *Ecological Applications* 9, 699-713.

McCave, I.N., 1976. Organism-Sediment Relationships, in: McCave, I.N. (Ed.), The Benthic Boundary Layer. Springer US, Boston, MA, pp. 273-295.

McLachlan, A., Brown, A.C., 2006. Interstitial Ecology, The Ecology of Sandy Shores (Second Edition). Academic Press, Burlington. pp. 181-196.

McLusky, D.S., 1971. Ecology of estuaries, Heinemann Educational, London pp.401.

McLusky, D.S., 1999. Estuarine benthic ecology: A European perspective. *Australian Journal of Ecology* 24, 302-311.

McLusky, D.S., Elliott, M., 2004. The estuarine ecosystem: ecology, threats, and management, 3rd ed. Oxford University Press, Oxford, New York. pp.214.

Medeiros, C.R., Hepp, L.U., Patrício, J., Molozzi, J., 2016. Tropical estuarine macrobenthic communities are structured by turnover rather than nestedness. *PLoS ONE* 11, e0161082.

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

Miao, S., DeLaune, R.D., Jugsujinda, A., 2006. Influence of sediment redox conditions on release/solubility of metals and nutrients in a Louisiana Mississippi River deltaic plain freshwater lake. *Science of The Total Environment* 371, 334-343.

Miller, D.C., 1989. Abrasion effects on microbes in sandy sediments. *Marine Ecology Progress Series* 55, 73-82.

Misra, A., 1998. Polychaeta of West Bengal. State Fauna Series 3: Fauna of West Bengal, Part 10 Zoological Survey of India. pp.125-225.

Mistri, M., 2002. Ecological characteristics of the invasive Asian date mussel, *Musculista senhousia*, in the Sacca Di Goro (Adriatic Sea, Italy). *Estuaries* 25, 431-440.

Modassir, Y., 1990. Ecology and production of benthic bivalve *Meretrix casta* (Chemnitz) in Mandovi estuary, Goa. *Indian Journal of Geo-Marine Sciences* 19, 125-127.

MoEF, 2014. India's fifth national report to the convention on biological diversity. Ministry of Environment and Forests, New Delhi. pp.100.

Monbet, Y., 1992. Control of phytoplankton biomass in estuaries: A comparative analysis of microtidal and macrotidal estuaries. *Estuaries* 15, 563-571.

Moreira, J., Lourido, A., Troncoso, J.S., 2008. Diversity and distribution of peracarid crustaceans in shallow subtidal soft bottoms at the ensenada de baiona (Galicia, N.W. Spain). *Crustaceana* 81, 1069-1089.

Mukhopadhyay, S.K., Biswas, H., De, T.K., Jana, T.K., 2006. Fluxes of nutrients from the tropical River Hooghly at the land–ocean boundary of Sundarbans, NE Coast of Bay of Bengal, India. *Journal of Marine Systems* 62, 9-21.

Murray, J.W., 1991. Ecology and palaeoecology of benthic foraminifera. Harlow, Essex, England: Longman Scientific and Technical; New York: Co-published in the U.S. with Wiley. pp.397.

Murty, C.S., Das, P.K., Nair, R.R., Veerayya, M., Varadachari, V.V.R., 1976. Circulation and sedimentation processes in and around the Aguada bar. *Indian Journal of Geo-Marine Sciences* 5, 9-17.

Murugan, A., Ayyakkannu, K., 1991. Ecology of benthic macrofauna in Cuddalore-Uppanar backwaters, southeast coast of India. *Indian Journal of Geo-Marine Sciences* 20, 200-203.

Murugan, T., Divakaran, O., Balakrishnan Nair, N., Padmanabhan, K.G., 1980. Distribution and seasonal variation of benthic fauna of the Veli Lake, South west coast of India. *Indian Journal of Geo-Marine Sciences* 9, 184-188.

Musale, A.S., Desai, D.V., 2011. Distribution and abundance of macrobenthic polychaetes along the South Indian coast. *Environmental Monitoring and Assessment* 178, 423-436.

Musale, A.S., Desai, D.V., Sawant, S.S., Venkat, K., Anil, A.C., 2015. Distribution and abundance of benthic macroorganisms in and around Visakhapatnam Harbour on the east coast of India. *Journal of the Marine Biological Association of the United Kingdom* 95, 215-231.

Muxika, I., Borja, Á., Bald, J., 2007. Using historical data, expert judgement and multivariate analysis in assessing reference conditions and benthic ecological status, according to the European Water Framework Directive. *Marine Pollution Bulletin* 55, 16-29.

Myers, V.B., Iverson, R.I., 1981. Phosphorus and nitrogen limited phytoplankton productivity in northeastern Gulf of Mexico Coastal Estuaries, in: Neilson, B.J., Cronin, L.E. (Eds.), Estuaries and Nutrients. Humana Press, Totowa, NJ, pp. 569-582.

Nair, K.K.C., Gopalakrishnan, T.C., Venugopal, P., George Peter, M., Jayalakshmi, K.V., Rao, T.S.S., 1983a. Population dynamics of estuarine amphipods in Cochin backwaters. *Marine Ecology Progress Series* 10, 289-295.

Nair, K.K.C., Sankaranarayanan, V.N., Gopalakrishnan, T.C., Balasubramanian, T., Lalithambika Devi, C.B., Aravindakshan, P.N., Krishnankutty, M., 1988. Environmental conditions of some paddy cum prawn culture fields of Cochin backwaters, southwest coast of India. *Indian Journal of Geo-Marine Sciences* 17, 24-30.

Nair, N.B., Abdul Azis, P.K., Dharmaraj, K., Arunachalam, M., Krishnan Kumar, K., Balasubramanian, N.K., 1984a. Ecology of Indian estuaries - primary production of the Ashtamudi estuary, South-West coast of India, *Proceedings of the National Academy of Sciences (Animal Science)*, pp. 11-23.

Nair, N.B., Abdul Azis, P.K., 1987a. Ecology of the Ashtamudi estuary, southwest coast of India. *Journal of the Marine Biological Association of India* 29, 177-194.

Nair, N.B., Abdul Azis, P.K., 1987b. Hydrobiology of the Ashtamudi estuary a tropical backwater system in Kerala, *Proceedings of national seminar on estuarine management* (ed.) N.B. Nair, National Committee on Science, Technology and environment, Government of Kerala, Trivandrum. pp. 268-280.

Nair, N.B., Dharmaraj, K., Abdul Azis, P.K., Arunachalam, M., Krishna Kumar, K., Balasubramanian, N.K., 1983b. Nature of primary production in tropical back water of the southwest coast of India. *Proceedings of the Indian Academy of Science* 6, 581-597.

Nair, N.B., Dharmaraj, K., Abdul Azis, P.K., Arunachalam, M., Kumar, K.K., 1984b. Ecology of biofouling on *Crassostrea madrasensis* (Preston) (Mollusca: Bivalvia) in a tropical backwater. *Proceedings: Animal Sciences* 93, 419-430.

Nair, P.V.R., Joseph, K.J., Balachandran, V.K., Pillai, V.K., 1975. A study on the primary production in the Vembanad Lake. *Bulletin of the Department of Marine Sciences, University of Cochin* 7, 170.

Nayak, B.K., Acharya, B.C., Panda, U.C., Nayak, B.B., Acharya, S.K., 2004. Variation of water quality in Chilka lake, Orissa. *Indian Journal of Geo-Marine Sciences* 32, 164-169.

Nebra, A., Alcaraz, C., Caiola, N., Muñoz-Camarillo, G., Ibáñez, C., 2016. Benthic macrofaunal dynamics and environmental stress across a salt wedge Mediterranean estuary. *Marine Environmental Research* 117, 21-31.

Nelson, D.W., Sommers, L.E., 1982. Total carbon, organic carbon and organic matter. in *Methods of Soil Analysis*, Part 2. Chemical and microbiological properties. ASA Monograph Number 9. American Society of Agronomy, Madison, Wisconsin, USA. pp.539-579.

Newman, B.K., Wooldridge, T.H., Cockcroft, A.C., 2009. Aspects of the biology and ecology of the estuarine cirolanid isopod, *Cirolana fluviatilis*. *African Zoology* 42, 12-22.

Nichols, M.M., 1986. Effects of Fine Sediment Resuspension in Estuaries, in: Mehta, A.J. (Ed.), Estuarine cohesive sediment dynamics: Proceedings of a workshop on cohesive sediment dynamics with special reference to physical processes in estuaries, Tampa, Florida, November 12–14, 1984. Springer New York, New York. pp. 5-42.

Nixon, S.W., Oviatt, C.A., Frithsen, J., Sullivan, B., 1986. Nutrients and the productivity of estuarine and coastal marine ecosystems. *Journal of the Limnological Society of Southern Africa* 12, 43-71.

Nybakken, J.W., 1993. Marine Biology: an ecological approach Harpercollins College Div, New York. pp.496.

Nybakken, J.W., Bertness, M.D., 2005. Marine Biology: An Ecological Approach. Pearson Education, Inc., Benjamin Cummings publishers, San Francisco. pp.597.

O'Clair, R.M., O'Clair, C.E., 1998. Southeast Alaska's rocky shores : animals. Plant Press, Auke Bay, Alaska.

O'Connor, M.I., Bruno, J.F., Gaines, S.D., Halpern, B.S., Lester, S.E., Kinlan, B.P., Weiss, J.M., 2007. Temperature control of larval dispersal and the implications for marine ecology, evolution, and conservation. *Proceedings of the National Academy of Sciences* 104, 1266-1271.

Oliver, P.G., Jayachandran, P.R., Bijoy Nandan, S., 2016. Cuspidaria cochinensis Preston, 1916 transferred to the Corbulidae (Mollusca, Bivalvia) and assigned to the genus *Cuneocorbula*. *Marine Biodiversity*, 1-5.

Ordoñez, C., de la Fuente, A., Díaz-Palma, P., 2015. Modeling the influence of benthic primary production on oxygen transport through the water–sediment interface. *Ecological Modelling* 311, 1-10.

Organization, T.U.N.E.-S.a.C. USEPA, 2002. Methods for measuring the acute toxicity of effluents and receiving waters to freshwater and marine organisms. U.S. Environmental Protection Agency, Office of Water (4303T), Washington, DC 266.

Ortega Cisneros, K., Smit, A.J., Laudien, J., Schoeman, D.S., 2011. Complex, dynamic combination of physical, chemical and nutritional variables controls spatio-temporal variation of sandy beach community structure. *PLoS ONE* 6, e23724.

Ourives, T.M., Rizzo, A.E., Boehs, G., 2011. Composition and spatial distribution of the benthic macrofauna in the Cachoeira River estuary, Ilhéus, Bahia, Brazil. *Revista de Biología Marina y Oceanografía* 46, 17-25.

Pagliosa, P.R., 2005. Another diet of worms: the applicability of polychaete feeding guilds as a useful conceptual framework and biological variable. *Marine Ecology* 26, 246-254.

Paikaray, S., Srichandan, S., Baliarsingh, S.K., Sahu, B.K., Panigrahy, R.C., 2012. Meiobenthos distribution in estuarine and beach sediments of rushikuly, estuary, east coast of India- a case study. *Asian Journal of Experimental Biological Sciences* 3, 391-396.

Palmer, M.A., 1988 Dispersal of marine meiofauna: a review and conceptual model explaining passive transport and active emergence with implications for recruitment. *Marine Ecology Progress Series* 48, 81-91.

Panikkar, N.K., Aiyar, R.G., 1937. The brackishwater fauna of Madras, *Proceedings of the Indian Academy of Science*, pp. 284-337.

Paramasivam, S., Kannan, L., 2005. Physico-chemical characteristics of Muthupettai mangrove environment, southeast coast of India. *International Journal of Ecology and Environmental Sciences* 31, 273-278.

Parker, 1975. The study of benthic communities: A model and a review. Elsevier Scientific Publishing Company. pp. 278.

Parulekar, A.H., Dhargalkar, V.K., Singbal, S.Y.S., 1980. Benthic studies in Goa estuaries, annual cycle of macrofaunal distribution, production and trophic relation. *Indian Journal of Geo-Marine Sciences* 9, 189-200.

Parulekar, A.H., Dwivedi, S.N., 1974. Benthic studies in Goa estuaries. Part I - Standing crop and faunal composition in relation to bottom salinity distribution and substratum characteristics in the estuary of Mandovi river. *Indian Journal of Geo-Marine Sciences* 3, 41-45.

Parulekar, A.H., Harkantra, S.N., Ansari, Z.A., 1982. Benthic production and assessment of demersal fishery resources of the Indian seas. *Indian Journal of Geo-Marine Sciences* 11, 107-114.

Patel, S.J., Desai, B.G., 2009. Animal-sediment relationship of the crustaceans and polychaetes in the Intertidal zone around Mandvi, Gulf of Kachchh western India. *Journal of the Geological Society of India* 74, 233-259.

Pati, P., Panigrahy, R.C., 2013. On some mollusca collectlons from different beaches of south Odisha coast of India. *Records of the Zoological Survey of India* 113, 229-254.

Pearson, T., Rosenberg, R., 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. *Oceanography and Marine Biology - An Annual Review* 16, 229-311.

Pearson, T.H., Rosenberg, R., 1987. Feast and famine: structuring factors in marine benthic communities, Organization of Communities Past and Present. Wiley-Blackwell, Oxford. pp. 588.

Pelegrí, S.P., Blackburn, T.H., 1994. Bioturbation effects of the amphipod *Corophium volutator* on microbial nitrogen transformations in marine sediments. *Marine Biology* 121, 253-258.

Pelletier, M.C., Gold, A.J., Heltshe, J.F., Buffum, H.W., 2010. A method to identify estuarine macroinvertebrate pollution indicator species in the Virginian Biogeographic Province. *Ecological Indicators* 10, 1037-1048.

Petersen, C.G.J., 1915. On the animal communities of the sea bottom in the Skagerak, the Christiania Fjord and the Danish waters. *Reports of the Danish Biological Station* 23, 1-28.

Petersen, C.G.J., 1918. Report XXV from the Danish Biological Station. pp. 66.

Petersen, C.G.J., Jensen, P.B., 1911. Valuation of the sea. I. Animal life of sea bottom, its food and quantity, in: Station, R.f.t.D.B. (Ed.), pp. 81.

Peterson, C.G.J., 1913. Valuation of the sea II. The animal communities of the seabottom and their importance for marine zoogeography, *Report of the Danish Biological station to the Board of agriculture*. pp. 1-44.

Peterson, C.H., 1979. Predation, competitive exclusion, and diversity in the softsediment benthic communities of estuaries and lagoons, in: Robert J. Livingston (ed.) *Ecological Processes in Coastal and Marine Systems*, Plenum, New York. Vol. 10, pp. 233-264

Pielou, E.C., 1966. The measurement of diversity in different types of biological collections. *Journal of Theoretical Biology* 13, 131-144.

Pillai, N.G.K., 1977. Distribution and seasonal abundance of macrobenthos of the Cochin backwaters. *Indian Journal of Geo-Marine Sciences* 6, 1-5.

Pillai, N.G.K., 1978. Macrobenthos of a tropical estuary. Ph.D. Thesis, University of Cochin. pp.133.

Pillai, N.G.K., 2001. On some benthic polvchaetes from Cochin Estuary. *Journal of the Marine Biological Association of India* 43, 120 - 135.

Pillai, N.K., 1954. A preliminary note on the Tanaidacea and Isopoda of Travancore. Bulletin of the Central ResearchInstitute, University of Travancore. 3, 1-21

Pinto, R., Patrício, J., Baeta, A., Fath, B.D., Neto, J.M., Marques, J.C., 2009. Review and evaluation of estuarine biotic indices to assess benthic condition. *Ecological Indicators* 9, 1-25.

Pocklington, P., Wells, P.G., 1992. Polychaetes Key taxa for marine environmental quality monitoring. *Marine Pollution Bulletin* 24, 593-598.

Pohle, G.W., Thomas, M.L.H., 2001. Monitoring protocol for marine benthos: Intertidal and subtidal macrofauna. in: St.Andrews (ed.), Report for Atlantic Maritime Ecological Science Cooperative, New Brunswick, Canada. pp.28.

Pomero, L.R., Smith, E.E., Grant, C.M., 1965. The exchange of phosphate between estuarine water and sediments. *Limnology and Oceanography* 10, 167-172.

Poore, G.C.B., Bruce, N.L., 2012. Global diversity of marine isopods (except Asellota and Crustacean symbionts). *PLoS ONE* 7, e43529.

Poore, G.C.B., Mobley, M.C., 1980. Canonical correlation analysis of marine macrobenthos survey data. *Journal of Experimental Marine Biology and Ecology* 45, 37-50.

Posey, M.H., 1987. Influence of relative mobilities on the composition of benthic communities. *Marine Ecology Progress Series* 39, 89-93.

Prabhadevi, L., 1994. Ecology of Coleroon estuary: studies on benthic fauna *Journal of the Marine Biological Association of India* 36, 260-266.

Prabhadevi, L., Natarajan, P., Saraswathy Ammal, G., Abdul Azis, P.K., 1996. Water quality and benthic fauna of the Kayamkulam backwaters and Arattupuzha coast along south west coast of India. *Indian Journal of Geo-Marine Sciences* 27, 264-267.

Pratihary, A.K., Naqvi, S.W.A., Naik, H., Thorat, B.R., Narvenkar, G., Manjunatha, B.R., Rao, V.P., 2009. Benthic fluxes in a tropical Estuary and their role in the ecosystem. *Estuarine, Coastal and Shelf Science* 85, 387-398.

Preston, H.B., 1916. Report on a collection of Mollusca from the Cochin and Ennur backwaters. *Records of the Indian Museum* 12, 27-39.

Pritchard, D.W., 1967. What is an estuary: physical viewpoint, in: GH, L. (Ed.), Estuaries. American Association for the Advancement of Science, Washington DC. pp. 3-5.

Priya, A., 2015. Assessment of tanaids in pulicat coastal lake south east India. Manonmaniam Sundaranar University. pp. 214.

Ptacnik, R., Andersen, T., Tamminen, T., 2010. Performance of the Redfield ratio and a family of nutrient limitation indicators as thresholds for phytoplankton N vs. P Limitation. *Ecosystems* 13, 1201-1214.

Qasim, S.Z., 1973. Productivity of backwaters and estuaries, in: Bernt Zeitzschel, S.A.G. (Ed.), The Biology of the Indian Ocean. Springer-Verlag Berlin, Heidelberg. New York. pp. 143-154.

Qasim, S.Z., 1977. Biological productivity of the Indian Ocean. *Indian Journal of Marine Science* 6, 122-137.

Qasim, S.Z., 2003. Indian estuaries. Allied Publication, Mumbai pp.259.

Qasim, S.Z., Bhattathiri, P.M.A., Abidi, S.A.H., 1968. Solar radiation and its penetration in a tropical estuary. *Journal of Experimental Marine Biology and Ecology* 2, 87-103.

Qasim, S.Z., Gopinathan, C.K., 1969. Tidal cycle and the environmental features of Cochin backwater (A tropical estuary). 336-353.

Rajan, S., 1964. Environmental studies of the Chilka Lake: (2) Benthic animal communities. *Indian Journal of Fishereis* 12, 492-499.

Rajendran, A., Venugopalan, V.K., 1975. Recent researches in estuarine biology. In : R. Natarajan (ed.), Hindustan Publishing Corporation Ltd., New Delhi pp.247.

Rajendran, A., Venugopalan, V.K., 1977a. Microbiology of nutrient regeneration: Part-II Distribution of nitrifying bacteria in Vellar estuary. *Indian Journal of Geo-Marine Sciences* 6, 6-9.

Rajendran, A., Venugopalan, V.K., 1977b. Microbiology of nutrient regeneration: Part - III Experimental studies on marine nitrification. *Indian Journal of Geo-Marine Sciences* 6, 10-14.

Ram, A.S.P., Nair, S., Chandramohan, D., 2003. Seasonal shift in net ecosystem production in a tropical estuary. *Limnology and Oceanography* 48, 1601-1607.

Ram, M.J., Mehta, M.P., Govindan, K., 1998. Phytoplankton pigments and macrobenthos in nearshore waters off an oil tenninal at Uran (Maharashtra), west coast of India. *Indian Journal of Geo-Marine Sciences* 27, 317-322.

Ramaiah, N., Ramarah, N., 1998. Phytoplankton characteristics in a polluted Bombay Harbour-Thana-Bassein creek estuarine complex. *Indian Journal of Geo-Marine Sciences* 27, 281-285.

Ramamirtham, C.P., Muthusamy, S., 1986. Estuarine oceanography of the Vembanad lake Part II : the region between Cochin and Azhikode. *Indian Journal of Fisheries* 33, 218-224.

Ramamirthan, C.P., Jayaraman, R., 1963. Some aspects of the hydrographical conditions of the backwaters around Willington Island (Cochin). *Journal of the Marine Biological Association of India* 2, 199-207.

Raman, A.V., Ganapati, P.N., 1983. Pollution effects on ecobiology of benthic polychaetes in Visakhapatnam Harbour (Bay of Bengal). *Marine Pollution Bulletin* 14, 46-52.

Rao, D., Rao, M., Annapurna, C., 2009. Polychaete community structure of Vasishta Godavari estuary, east coast of India. *Journal of the Marine Biological Association of India* 51, 137-144.

Rao, D.G., Sarma, A.L.N., 1994. Seasonal abundance and breeding cycles of meiobenthic copepods at Parikud Island in Chilka lagoon (Bay of Bengal). *Indian Journal of Geo-Marine Sciences* 23, 217-220.

Rao, G.S., Sarma, D.V.R., 1990. Meiobenthos of the Gosthani estuary. *Indian Journal of Geo-Marine Sciences* 19, 171-173.

Rao, P.V., 1968. Maturation and spawning of the penaeid prawns of the south west coast of India. *FAO Fisheries Report* 52, 285-302

Rasheed, K., 1997. Studies on Dredging Impact Assessment (DIA) at Cochin- A tropical Estuarine harbour. Ph.D. Thesis, Cochin University of Science and Technology, Cochin, India pp.159.

Reddy, K.R., D'Angelo, E.M., Harris, W.G., 2000. Biogeochemistry of wetlands, in: Sumner, M.E. (Ed.), Handbook of Soil Science. CRC Press, Boca Raton, FL, pp. G89-G119.

Rehitha, T.V., Ullas, N., Vineetha, G., Benny, P.Y., Madhu, N.V., Revichandran, C., 2017. Impact of maintenance dredging on macrobenthic community structure of a tropical estuary. *Ocean and Coastal Management* 144, 71-82.

Reiss, H., Kröncke, I., 2005. Seasonal variability of benthic indices: An approach to test the applicability of different indices for ecosystem quality assessment. *Marine Pollution Bulletin* 50, 1490-1499.

Reizopoulou, S., Simboura, N., Sigala, K., Barbone, E., Aleffi, F., Kaisakis, G., Rosati, I., Basset, A., Nicolaidou, A., 2014. Assessing the ecological status of Mediterranean coastal lagoons using macroinvertebrates. Comparison of the most commonly used methods. *Mediterranean Marine Science* 15(3), 602-612.

Remane, A., 1933. Verteilung und Organisation der benthonischen Mikrofauna der Kleler Bucht, Wiss Meeresunters. Abt. Kiel. NF 21, 161-221.

Remane, A., 1934. Die Brackwasserfauna. Verhandlungen der Deutschen Zoologischen Gesellschaft 36, 34-74.

Remane, A., Schlieper, C., 1958. Die biologie des brackwassers. In: Die Binnengewasser, A. Thienamann (ed.), Stuttgart, E. Schweizerbart'sche, Verlagsbuchhandlung 22, pp.348.

Remani, K.N., 1979. Studies on the effect of pollution, with special reference to benthos in Cochin backwaters. Unviersity of Cochin.

Remani, K.N., Sarala Devi, K., Venugopal, P., Unnithan, R.V., 1983. Indicator organisms of pollution in Cochin backwaters *Mahasagar - bulletin of the National Institute of Oceanography* 6, 199-207.

Remani, K.N., Venugopal, P., Saraladevi, K., Vnnithan, R.V., 1981. Sediments of a retting Yard. *Indian Journal of Geo-Marine Sciences* 10, 41-44.

Revichandran, C., 1993. Studies on the dynamics and suspended sediment transport in the Azhikode estuary (south-west coastof India). Ph.D. Thesis, Cochin University of Science and Technology, National Institute of Oceanography, Regional Centre, Kochi. pp.100.

Revichandran, C., Abraham, P., 1998. Mixing and flushing time scale in the Azhikode estuary, south west coat of India. *Indian Journal of Geo-Marine Sciences* 27, 163-166.

Revichandran, C., Srinivas, K., Muraleedharan, K.R., Rafeeq, M., Shivaprasad, A., Vijayakumar, K., Jayalakshmy, K.V., 2012. Environmental set-up and tidal propagation in a tropical estuary with dual connection to the sea (SW coast of India). *Environmental Earth Sciences* 66(4), 1031–1042.

Richard, R.V., 1973. On reproductive strategies in marine benthic invertebrates. *The American Naturalist* 107, 339-352.

Riemann, F., Schrage, M., 1983. The mucus-trap hypothesis on feeding of aquatic nematodes and implications for biodegradation and sediment texture. *Oecologia* 34, 75-88.

Riera, R., de-la-Ossa-Carretero, J.A., 2014. Response of benthic opportunistic polychaetes and amphipods index to different perturbations in coastal oligotrophic areas (Canary archipelago, North East Atlantic Ocean). *Marine Ecology* 35, 354-366.

Rivero, M.S., Elías, R., Vallarino, E.A., 2005. First survey of macroinfauna in the Mar del Plata Harbor (Argentina), and the use of polychaetes as pollution indicators. *Revista de biología marina y oceanografía* 40, 101-108.

Robertson, B., Stevens, L., 2010. Haldane Estuary. Fine scale monitoring 2009/10. Nelson, prepared for environment southland by wriggle coastal managemnt. pp.35.

Roegner, G.C., 2000. Transport of molluscan larvae through a shallow estuary. *Journal of Plankton Research* 22, 1779-1800.

Rosenberg, R., Blomqvist, M., C Nilsson, H., Cederwall, H., Dimming, A., 2004. Marine quality assessment by use of benthic species-abundance distributions: a proposed new protocol within the European Union Water Framework Directive. *Marine Pollution Bulletin* 49, 728-739.

Rouse, G.W., Fauchald, K., 1997. Cladistics and polychaetes. *Zoologica Scripta* 26, 139-204.

Rouse, G.W., Pleijel, F., 2001. Polychaetes. Oxford University Press, New York. pp. 354.

Rutledge, P.A., Fleeger, J.W., 1993. Abundance and seasonality of meiofauna, including harpacticoid copepod species, associated with stems of the salt-marsh cord grass, Spartina alterniflora. *Estuaries* 16, 760.

Rygg, B., 1985. Effect of Sediment Copper on Benthic Fauna. *Marine Ecology Progress Series* 25, 83-89.

Ryther, J.H., 1969. Photosynthesis and fish production in the sea. *Science* 166, 72-76.

Sakamoto, T., Ogawa, S., Nishiyama, Y., Akada, C., Takahashi, H., Watanabe, T., Minakata, H., Sakamoto, H., 2015. Osmotic/ionic status of body fluids in the euryhaline cephalopod suggest possible parallel evolution of osmoregulation. *Scientific Reports* 5, 14469.

Sallenave, R.M., Barton, D.R., 1990. The distribution of benthic invertebrates along a natural turbidity gradient in Lake Temiskaming, Ontario-Quebec. *Hydrobiologia* 206, 225-234.

Sánchez-Moyano, J.E., García-Asencio, I., 2010. Crustacean assemblages in a polluted estuary from South-Western Spain. *Marine Pollution Bulletin* 60, 1890-1897.

Sanders, H.L., 1956. Oceanography of long Island, Sound. 1952-54 X, Biology of marine bottom. *Bulletin of the Bingham Oceanographic Collection* 15, 345-414.

Sanders, H.L., 1958. Benthic studies in Buzzard's Bay. I. Animal - sediment relationships. *Limnology and Oceanography* 3, 245-258.

Sanders, H.L., 1968. Marine benthic diversity: A comparative study. *American Naturalist* 102, 243-282.

Sanil, N.K., Vikas, P.A., Ratheesh, T.B., George, K.C., Vijayan, K.K., 2009. Mortalities caused by the crustacean isopod, *Cirolana fluviatilis*, in tropical, cage-cultured Asian seabass, *Lates calcarifer*: a case study from the southwest coast of India. *Aquaculture Research* 40, 1626-1633

Sankaranarayanan, V.N., Panampunnayil, S.U., 1979. Studies on organic carbon, nitrogen and phosphorus in sediments of the Cochin backwaters. *Indian Journal of Geo-Marine Sciences* 8, 27-30.

Sankaranarayanan, V.N., Qasim, S.Z., 1969. Nutrients of the Cochin backwaters in relation to environmental characteristics. *Marine Biology* 2, 236-247.

Santhanam, R., Srinivasan, A., Devaraj, M., 1995. Trophic model of an estuarine ecosystem at the southeast coast of India., *ICLARM Conference Proceedings*, pp. 230-233.

Sarala Devi, K., 1986. Effect of industrial pollution on a benthic community of a tropical estuary. Ph.D. Thesis, Cochin University of Science and Technology, Cochin, India. pp.380.

Sarala Devi, K., Jayalakshmy, K.V., Venugopal, P., 1991. Communities and coexistence of benthos in northern limb of Cochin backwaters. *Indian Journal of Geo-Marine Sciences* 20, 249-254.

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

Saraladevi, K., Venugopal, P., Remani, K.N., Zacharias, D., Unnithan, R.V., 1983. Nutrients in some estuaries of Kerala. *Mahasagar* 16, 161-174.

Sarkar, S.K., Bhattacharya, A., Giri, S., Bhattacharya, B., Sarkar, D., Nayak, D.C., Chattopadhaya, A.K., 2005. Spatiotemporal variation in benthic polychaetes (Annelida) and relationships with environmental variables in a tropical estuary. *Wetlands Ecology and Management* 13, 55-67.

Sarma, V.V.S.S., Gupta, S.N.M., Babu, P.V.R., Acharya, T., Harikrishnachari, N., Vishnuvardhan, K., Rao, N.S., Reddy, N.P.C., Sarma, V.V., Sadhuram, Y., Murty, T.V.R., Kumar, M.D., 2009. Influence of river discharge on plankton metabolic rates in the tropical monsoon driven Godavari estuary, India. *Estuarine, Coastal and Shelf Science* 85, 515-524.

Sarma, V.V.S.S., Prasad, V.R., Kumar, B.S.K., Rajeev, K., Devi, B.M.M., Reddy, N.P.C., Sarma, V.V., Kumar, M.D., 2010. Intra-annual variability in nutrients in the Godavari estuary, India. *Continental Shelf Research* 30, 2005-2014.

Sarma, V.V.S.S., Viswanadham, R., Rao, G.D., Prasad, V.R., Kumar, B.S.K., Naidu, S.A., Kumar, N.A., Rao, D.B., Sridevi, T., Krishna, M.S., Reddy, N.P.C., Sadhuram, Y., Murty, T.V.R., 2012. Carbon dioxide emissions from Indian monsoonal estuaries. *Geophysical Research Letters* 39, n/a-n/a.

Satyanaranyana Murty, P., Rao, K.S., 1959. *Journal of the Geological Society of India* 13, 38.

Schaffner, L.C., Jonsson, P., Diaz, R.J., Rosenberg, R., Gapcynski, P., 1992. Benthic communities and bioturbation history of estuarine and coastal systems: effects of hypoxia and anoxia in: Vollenweider, R.A., Marchetti, R., Viviani, R. (Eds.), Marine Coastal Eutrophication. Elsevier, Amsterdam. pp. 1001-1016.

Schratzberger, M., Warwick, R.M., 1998. Effects of the intensity and frequency of organic enrichment on two estuarine nematode communities. *Marine Ecology Progress Series* 164, 83-94.

Schwinghamer, P., 1983. Generating ecological hypotheses from biomass spectra using causal analysis: a benthic example. *Marine Ecology Progress Series* 13, 151-166.

Segar, K., Hariharan, V., 1989. Seasonal distribution of nitrate, nitrite, ammonia and plankton in effluent discharge area of Mangalore, West coast of India. *Indian Journal of Geo-Marine Sciences* 18, 170-173.

Semeniuk, T.A., 2000. Small benthic crustacea of the Leschenault Inlet estuary. Journal of the Royal Society of Western Australia. 83 429-441.

Seshappa, G., 1953. Observation of the physical and biological features of the inshore sea bottom along the Malabar coast. *Proceedings of the National Academy of Sciences, India* 19, 257-279.

Shannon, C.E., Weaver, W., 1949. The mathematical theory of communication. University of Illinois Press, Urbana. pp.117.

Sheeba, P., 2000. Distribution of benthic infauna in the Cochin backwaters in relation to environmental parameters, Ph.D. Thesis, NIO-RC, Cochin University of Science and Technology, Kochi. pp. 241.

Shepard, F.P., 1954. Nomenclature based on sand-silt-clay ratios. *Journal of Sedimentary Research* 24, 151-158.

Shields, M.A., Blanco-Perez, R., 2013. Polychaete abundance, biomass and diversity patterns at the Mid-Atlantic Ridge, North Atlantic Ocean. *Deep Sea Research Part II: Topical Studies in Oceanography* 98, 315-325.

Shivaprasad, A., Vinita, J., Revichandran, C., Manoj, N.T., Srinivas, K., Reny, P.D., Ashwini, R., Muraleedharan, K.R., 2012. Influence of Saltwater Barrage on Tides, Salinity, and Chlorophyll-*a* in Cochin Estuary, India. *Journal of Coastal Research* 29 (6), 1382-1390.

Shivaprasad, A., Vinita, J., Revichandran, C., Reny, P.D., Deepak, M.P., Muraleedharan, K.R., Naveen Kumar, K.R., 2013. Seasonal stratification and property distributions in a tropical estuary (Cochin estuary, west coast, India). *Hydrology and Earth System Sciences* 17, 187-199.

Shyamasundari, K., 1973. Studies on the tube-building amphipod Corophium triaenonyx stebbing from Visakhapatnam Harbor: effect of salinity and temperature. *The Biological Bulletin* 144, 503-510.

Sigamani, S., Perumal, M., Arumugam, S., Preetha Mini Jose, H.M., Veeraiyan, B., 2015. AMBI indices and multivariate approach to assess the ecological health of Vellar–Coleroon estuarine system undergoing various human activities. *Marine Pollution Bulletin* 100, 334-343.

Sigovini, M., Keppel, E., Tagliapietra, D., 2013. M-AMBI revisited: looking inside a widely-used benthic index. *Hydrobiologia* 717, 41-50.

Simboura, N., Papathanassiou, E., Sakellariou, D., 2007. The use of a biotic index (Bentix) in assessing long-term effects of dumping coarse metalliferous waste on soft bottom benthic communities. *Ecological Indicators* 7, 164-180.

Simboura, N., Zenetos, A., 2002. Benthic indicators to use in Ecological Quality classification of Mediterranean soft bottom marine ecosystems, including a new Biotic Index. *Mediterranean Marine Science* 3(2), 77-111.

Simpson, E.H., 1949. Measurement of diversity. *Nature* 163, 688.

Singh, A., Pathirana, S., Shi, H., 2006. Assessing coastal vulnerability: developing a global index for measuring risk. In: Ashbindu, S., Sumith, P., Hua, S. (ed.), United Nations Environment Programme, Nairobi, Kenya. pp.54.

Sivadas, S., Ingole, B., Nanajkar, M., 2010. Benthic polychaetes as good indicators of anthropogenic impact. *Indian Journal of Geo-Marine Sciences* 39, 201-211.

Sivadas, S., Ingole, B., Nanajkar, M., 2011. Temporal variability of macrofauna from a disturbed habitat in Zuari estuary, west coast of India. *Environmental Monitoring and Assessment* 173, 65-78.

Sivadas, S.K., Nagesh, R., Gupta, G.V.M., Gaonkar, U., Mukherjee, I., Ramteke, D., Ingole, B.S., 2016. Testing the efficiency of temperate benthic biotic indices in assessing the ecological status of a tropical ecosystem. *Marine Pollution Bulletin* 106, 62-76.

Small, C., Nicholls, R.J., 2003. A global analysis of human settlement in coastal zones. *Journal of Coastal Research* 19, 584-599.

Smith, E.A., 1895. *Nassodonta insignis*. *Proceedings of the Malacological Society of London* 1, 257-258.

Smith, S.V., Hollibaugh, J.T., 1993. Coastal metabolism and the oceanic organic carbon balance. *Reviews of Geophysics* 31, 75-89.

Snelgrove, P.V.R., 1997. The Importance of marine sediment biodiversity in ecosystem processes. *Ambio* 26, 578-583.

Snelgrove, P.V.R., 1998. The biodiversity of macrofaunal organisms in marine sediments. *Biodiversity and Conservation* 7, 1123-1132.

Snelgrove, P.V.R., 1999. Getting to the bottom of marine biodiversity: Sedimentary habitats: ocean bottoms are the most widespread habitat on earth and support high biodiversity and key ecosystem services. *BioScience* 49, 129-138.

Snelgrove, P.V.R., Butman, C.A., 1994. Animal-sediment relationships revisited: cause versus effect. *Oceanography and Marine Biology: An Annual Review* 32, 111-177.

Solan, M., Cardinale, B.J., Downing, A.L., Engelhardt, K.A.M., Ruesink, J.L., Srivastava, D.S., 2004. Extinction and ecosystem function in the marine benthos. *Science* 306, 1177-1180.

Soniya, S., Sarala Devi, K., 2009. Polychaete diversity and its relevance in the rapid environmental assessment of Mumbai Port. *Current Science* 97, 1439-1444

Sooria, P.M., Jyothibabu, R., Anjusha, A., Vineetha, G., Vinita, J., Lallu, K.R., Paul, M., Jagadeesan, L., 2015. Plankton food web and its seasonal dynamics in a large monsoonal estuary (Cochin backwaters, India)- significance of mesohaline region. *Environmental Monitoring and Assessment* 187, 1-22.

Spagnolo, A., Punzo, E., Santelli, A., Scarcella, G., Strafella, P., Grati, F., Fabi, G., 2014. Offshore platforms: comparison of five benthic indicators for assessing the macrozoobenthic stress levels. *Marine Pollution Bulletin* 82, 55-65.

Sreedevi, O.K., 2017. Assessment of trophic change and production potential of Kodungallur-Azhikode estuary, India. Ph.D. Thesis, Cochin University of Science and Technology, pp. 480.

Sreedhar, K.S., 1991. Studies on the biology of bivalve *Musculista senhousia* (Benson) from Cochin backwaters. Ph.D. Thesis, Cochin University of Science and Technology, Kochi, India.

Sreeja, K.G., Madhusoodhanan, C.G., Eldho, T.I., 2016. Coastal zones in integrated river basin management in the West Coast of India: Delineation, boundary issues and implications. *Ocean and Coastal Management* 119, 1-13.

Srinivasa Rao, D., Srinivasa Rao, M., Annapurna, C., 2009. Polychaete community structure of Vasishta Godavari estuary, east coast of India. *Journal of the Marine Biological Association of India* 51, 137-144.

Stauber, J.L., Chariton, A., Apte, S., 2016. Chapter 10 - Global Change A2 - Blasco, Julián, in: Chapman, P.M., Campana, O., Hampel, M. (Eds.), Marine Ecotoxicology. Academic Press. pp. 273-313.

Stebbing, T.R.R., 1911. Indian Isopods. *Records of the Indian Museum* 6, 179-191

Steel, J.H., 1974. The structure of marine ecosystems. Harvard University Press, Cambridge, Massachusetts. x + pp.128.

Stephen, W., Melanie, C.A., Michael, A.K., Frode, O., Morten, T.S., Sarah, L.D., Hazel, R.N., 2004. Importance of bioturbators for biodiversity maintenance: indirect effects of fishing disturbance. *Marine Ecology Progress Series* 275, 1-10.

Stephenson, D., Shemang, E.M., Chaoka, T.R., 2004. Water resources of arid areas: Proceedings of the international conference on water resources of arid and semi-arid regions of Africa, Gaborone, Botswana, 3-6 August 2004. Taylor & Francis. pp.588.

Strickland, J.D.H., Parsons, T.R., 1968. Practical handbook of seawater analysis. *Bulletin of the Fisheries Research Board of Canada* 167, pp.311.

Strickland, J.D.H., Parsons, T.R., 1972. A practical handbook of seawater analysis. *Bulletins of the Fisheries Research Board of Canada*, Canada 167, pp.311.

Strong, E.E., Galindo, L.A., Kantor, Y.I., 2017. Quid est Clea helena? Evidence for a previously unrecognized radiation of assassin snails (Gastropoda: Buccinoidea: Nassariidae). *PeerJ* 5, e3638.

Subba Rao, N.V., Dey, A., Barua , S., 1992. Estuarine and Marine molluscs of West Bengal. *State Fauna series*, 3: Fauna of West Bengal, Zoological Survey of India (Part-9): 129-268.

Subba Rao, N.V., Dey, A., Maitra, S., Barua, S., 1995. Mollusca, Hooghly-Malta Estuary. *Estuarine Ecosystem series*, Zoological Survey of India. 2, 41-91.

Suja, N., Mohamed, K.S., 2010. The Black Clam, Villorita cyprinoides, Fishery in the State of Kerala, India. *Marine Fisheries Review* 72, 48-61.

Sunil Kumar, R., 1993. Studies on the benthic fauna of the mangrove swamps of Cochin area. Cochin University of Science and Technology.

Sunil Kumar, R., 1995. Comparative study on the community structure and distributional ecology of benthos in the two mangrove swamps of Cochin estuary, *Proceedings of the Seventh Kerala Science Congress* pp. 121-122.

Sunil Kumar, R., 1999. New Record of five annelids (Class-Polychaeta) from the mangrove habitat of the south west coast of India. *Journal of the Marine Biological Association of India* 41, 116-118.

Sunil Kumar, R., 2002. Soil macro-invertebrates in a prawn culture field of a tropical estuary. *Indian Journal of Fishereis* 49, 451-455.

Sunil Kumar, R., Antony, A., 1994. Impact of environmental parameters on polychaetous annelids in the mangrove swamps of Cochin - south west coast of India. *Indian Journal of Geo-Marine Sciences* 23, 137-142.

Talke, S.A., Stacey, M.T., 2008. Suspended sediment fluxes at an intertidal flat: The shifting influence of wave, wind, tidal, and freshwater forcing. *Continental Shelf Research* 28, 710-725.

ter Braak, C.J.F., Smilauer, P., 2002. CANOCO reference manual and CanoDraw for Windows user's guide : software for canonical community ordination (version 4.5). Biometris, Wageningen. pp. 41-50.

Thamdrup, H.M., 1935. Bietrage zur Okologie der watternfauna auf experimenteller Grundlage. *Meddelelser fra Kommissionen for Danmarks Fiskeri- og Havundersøgelser. Serie Fiskeri* 10, 1-125.

Thasneem, T.A., 2016. Productivity pattern and its dynamics in a tropical estuary, southwest coast of India. Cochin University of Science and Technology, Kochi, pp. 477.

The United Nations Educational - Scientific and Cultural Organization, 1985. The international system of units (SI) in oceanography. IAPSO Publication Scientific No. 32, Paris, France pp. 124.

The United Nations Educational - Scientific and Cultural Organization, 1966. Determination of photosynthetic pigments in sea water, Monographs on Oceanographic methodology. pp.69.

Thomson, D.H., 1982. Marine benthos in the eastern Canadian high Arctic: multivariate analyses of standing crop and community structure. *Arctic* 35 (1), 61-74.

Thorson, G., 1957. Bottom communities (sublittoral or shallow shelf) Geol. Soc. to the micro organisms in: Hedgpeth, J.W. (Ed.), In: Treatise on marine ecology and paleoecology. Geological Society of America, New York. pp. 461-534.

Thrush, S.F., Hewitt, J.E., Gibbs, M., Lundquist, C., Norkko, A., 2006. Functional role of large organisms in intertidal communities: community effects and ecosystem function. *Ecosystems* 9, 1029-1040.

Thrush, S.F., Lawrie, S.M., Hewitt, J.E., Cummings, V.J., 1999. The problem of scale: uncertainties and implications for soft-bottom marine communities and the assessment of human impacts, in: Gray, J.S., Ambrose, W., Szaniawska, A. (Eds.), Biogeochemical Cycling and Sediment Ecology. Springer Netherlands, Dordrecht. pp. 195-210.

Thrush, S.F., Townsend, M., Hewitt, J.E., Davies, K., Lohrer, A.M., Lundquist, C., Cartner, K., 2013. The many uses and values of estuarine ecosystems. Manaaki Whenua Press, Landcare Research, Lincoln. pp. 226-237.

Trask, P.D., 1939. Recent marine sediments. Tulsa, Oklohama, USA: American Association of Petroleum Geologists Publication. pp.736.

Tripathy, B., Mukhopadhyay, A.K., 2015. Marine Molluscan Diversity in India, in: Venkataraman, K., Sivaperuman, C. (Eds.), Marine faunal diversity in India, taxonomy, ecology and conservation. Academic Press is an imprint of Elsevier, 32 Jamestown Road, London NW1 7BY, UK. pp. 39-74.

Tripathy, S.C., Ray, A.K., Patra, S., Sarma, V.V., 2005. Water quality assessment of Gautami - Godavari mangrove estuarine ecosystem of Andhra Pradesh, India during September 2001. *Journal of Earth System Science* 114, 185-190.

Trivedy, P.K., Goel, P.K., 1986. Chemical and biological methods for water pollution studies, Series in Methodology. Environmental Publications, Karad. pp.220.

Uncles, R.J., Stephens, J.A., 2001. The annual cycle of temperature in a temperate estuary and associated heat fluxes to the coastal zone. *Journal of Sea Research* 46, 143-159.

UNEP, 2003. Global Environmental Outlook Year Book 2003, in: United Nations Environment Programme (UNEP), GEO Section, Nairobi, Kenya. pp.208.

Unnithan, R.V., Vijayan, M., Radhakrishnan, E.V., Remani, K.N., 1977. Incidence of fish mortality from industrial pollution in Cochin backwaters. *Indian Journal of Geo-Marine Sciences* 6, 81-83.

Unnithan, R.V., Vijayan, M., Remani, K.N., 1975. Organic pollution in Cochin backwaters. *Indian journal of Geo-Marine Sciences* 4, 39-42.

van der Linden, P., Marchini, A., Smith, C.J., Dolbeth, M., Simone, L.R.L., Marques, J.C., Molozzi, J., Medeiros, C.R., Patrício, J., 2017. Functional changes in polychaete and mollusc communities in two tropical estuaries. *Estuarine, Coastal and Shelf Science* 187, 62-73.

van Houte-Howes, K.S.S., Turner, S.J., Pilditch, C.A., 2004. Spatial differences in macroinvertebrate communities in intertidal seagrass habitats and unvegetated sediment in three New Zealand estuaries. *Estuaries* 27, 945-957.

Vanaverbeke, J., Gheskiere, T., Vincx, M., 2000. The meiobenthos of subtidal sandbanks on the belgian continental shelf (Southern Bight of the North Sea). *Estuarine, Coastal and Shelf Science* 51, 637-649.

Varshney, P.K., Govindan, K., Desai, B.N., 1981. Benthos of the Narmada estuary. *Mahasagar - Bulletin National Institute of Oceanography* 14, 141-148.

Vengayil, D.T., Gopalan, U.K., Krishnankutty, M., 1988. Development of *Apseudes chilkensis* (Tanaidacea, Crustacea), a forage organism in estuaries. *Mahasagar - Bulletin National Institute of Oceanography* 21, 95-103.

Venkataraman, K., Sivaperuman, C., Raghunathan, C., 2013. Ecology and conservation of tropical marine faunal communities. Springer, Heidelberg. pp.481.

Venkataraman, K., Wafar, M., 2005. Coastal and marine biodiversity of India. *Indian Journal of Geo-Marine Sciences* 34, 57-75.

Venugopal, P., Remani, K.N., Saraladevi, K., Unnithan, R.V., 1980. Fish kill in Chitrpuzha. *Seafood Export Journal* 12 (8), 15-21.

Vijayakumar, R., Anzari, Z.A., Parulekar, A.H., 1991. Benthic fauna of Kakinada bay and backwaters, east coast of India. *Indian Journal of Geo-Marine Sciences* 20, 195-199.

Vijayalakshmi, R.N., Mustafa, S., Mehta, P., Govindan, K., Ram, M.J., Gajbhiye, S.N., 1998. Biological characteristics of the Vashishti estuary, Maharashtra (West coast of India). *Indian Journal of Geo-Marine Sciences* 27, 310-316.

Vijayan, M., Remani, K.N., Unnithan, R.V., 1976. Effect of organic pollution on some hydrographical features of Cochin backwaters. Indian *Journal of Geo-Marine Sciences* 5, 190-200.

Vijith, V., Sundar, D., Shetye, S.R., 2009. Time-dependence of salinity in monsoonal estuaries. *Estuarine, Coastal and Shelf Science* 85(4), 601-608.

Vineetha, G., Jyothibabu, R., Madhu, N.V., Kusum, K.K., Sooria, P.M., Shivaprasad, A., Reny, P.D., Deepak, M.P., 2015. Tidal Influence on the Diel Vertical Migration Pattern of Zooplankton in a Tropical Monsoonal Estuary. *Wetlands* 35, 597-610.

Vinita, J., Shivaprasad, A., Revichandran, C., Manoj, N.T., Muraleedharan, K.R., Binzy, J., 2015. Salinity response to seasonal runoff in a complex estuarine system (Cochin estuary, west coast of India). *Journal of Coastal Research*, 869-878.

Vollenweider, R.A., 1992. Coastal marine eutrophication: principles and control, Marine Coastal Eutrophication. Elsevier, Amsterdam. pp. 1-20.

Vollenweider, R.A., Giovanardi, F., Montanari, G., Rinaldi, A., 1998. Characterization of the trophic conditions of marine coastal waters with special reference to the NW Adriatic Sea: proposal for a trophic scale, turbidity and generalized water quality index. *Environmetrics* 9, 329-357.

Wafar, S., Untawale, A.G., Wafar, M., 1997. Litter fall and energy flux in a mangrove ecosystem. *Estuarine, Coastal and Shelf Science* 44, 111-124.

Wang, S., Jin, B., Qin, H., Sheng, Q., Wu, J., 2015. Trophic Dynamics of Filter Feeding Bivalves in the Yangtze Estuarine Intertidal Marsh: Stable Isotope and Fatty Acid Analyses. *PLoS ONE* 10, e0135604.

Ward, L.G., Twilley, R.R., 1986. Seasonal distributions of suspended particulate material and dissolved nutrients in a coastal plain estuary. *Estuaries* 9, 156-168.

Warwick, R.M., 1986. A new method for detecting pollution effects on marine macrobenthic communities. *Marine Biology* 92, 557-562.

Warwick, R.M., Clarke, K.R., 1998. Taxonomic distinctness and environmental assessment. *Journal of Applied Ecology* 35, 532-543.

Warwick, R.M., Goss-Custard, J.D., Kirby, R., George, C.L., Pope, N.D., Rowden, A.A., 1991. Static and dynamic environmental factors determining the community structure of estuarine macrobenthos in SW Britain: why is the severn estuary different? *Journal of Applied Ecology* 28, 329-345.

Waters, T.F., 1977. Secondary production in inland waters 11Paper No. 9959, scientific journal series, minnesota agricultural experiment station, St Paul, Minnesota 55108. *Advances in Ecological Research* 10, 91-164.

Watling, L., 1979. Marine flora and fauna of the northeastern United States. Crustacea: Cumacea, NOAA Technical Report NMFS Circular 423. National Oceanic and Atmospheric Administration. pp.24.

Watling, L., 1989. Small-scale features of marine sediments and their importance to the study of deposit feeding, in: Lopez, G., Taghon, G., Levinton, J. (Eds.), Ecology of Marine Deposit Feeders. Springer New York, New York. pp. 269-290.

Webb, A.H., 1982. Weak and concentrations and river chemistry in the Tovdal river, Southern Norway. *Water Research* 16 641-648.

Weiser, W., 1953. Free living marine nemaodes Enoploidea. *Acta Universitatis Lundensis* (N.F.2) 49, 1-155.

Weiser, W., 1954. Free living marine nematodes II Chromaroroidea. *Acta Universitatis Lundensis* (N.F.2) 50, 1-148.

Weiser, W., 1956. Free living marine nemaodes III Axonolainoidea and Monhysteroidea *Acta Universitatis Lundensis* (N.F.2) 53, 1-115.

Weiser, W., 1959. The effect of grain size and the distribution of small invertebrates inhabiting the beaches of Puget Sound. *Limnology and oceanography* 4, 181-194.

Weiser, W., 1960. Benthic studies in Buzzards Bay. II. The meiofauna. *Limnology and Oceanography* 5, 121-137.

Weissberger, E.J., Coiro, L.L., Davey, E.W., 2009. Effects of hypoxia on animal burrow construction and consequent effects on sediment redox profiles. *Journal of Experimental Marine Biology and Ecology* 371, 60-67.

Williamson, H.G., Ockenden, M.C., 1996. ISIS: and instrument for measuring erosion shear stress in situ. *Estuarine Coastal and Shelf Science* 42, 1-118.

Wilson, J.G., Fleeger, J.W., 2012. Estuarine Benthos, Estuarine Ecology. John Wiley and Sons, Inc. pp. 303-325.

Wolff, W.J., 1983. Estuarine benthos. In: Ketchum, B. H. (ed.). Ecosystems of the world. Estuaries and enclosed seas. New York: Elsevier. pp.121-132.

Wollast, R., 1998. Evaluation and comparison of the global carbon cycle in the coastal zone and in the open ocean, in: Brink, K.H., Robinson, A.R. (Ed.), The Sea. Wiley, New York. pp.213-252.

Word, J.Q., 1978. The infaunal trophic index. SCCWRP 1978 Annual Report, Southern California Coastal Research Project. El Segundo, CA. pp. 13-39.

Worm, B., Barbier, E.B., Beaumont, N., Duffy, J.E., Folke, C., Halpern, B.S., Jackson, J.B.C., Lotze, H.K., Micheli, F., Palumbi, S.R., Sala, E., Selkoe, K.A., Stachowicz, J.J., Watson, R., 2006. Impacts of Biodiversity Loss on Ocean Ecosystem Services. *Science* 314, 787-790.

WoRMS, 2017. World Register of Marine Species Available from http://www.marinespecies.org at VLIZ. Accessed on 20-06-2017. doi:10.14284/170.

Yin, K., Qian, P., Wu, M., Chen, J., Huang, L., Song, X., Jian, W., 2001. Shift from P to N limitation of phytoplankton biomass across the Pearl River Estuarine plume during summer. *Marine Ecology Progress Series* 221, 17-28.

Yonge, C.M., 1953. Aspects of life on muddy shores, in: Marshall, S.M., Orr, A.P. (Eds.), Essays in Marine Biology Oliver and Boyd, Edinburgh. pp. 29-49.

Ysebaert, T., Herman, P.M.J., 2002. Spatial and temporal variation in benthic macrofauna and relationships with environmental variables in an estuarine, intertidal soft-sediment environment. *Marine Ecology Progress Series* 244, 105-124.

Ysebaert, T., Herman, P.M.J., Meire, P., Craeymeersch, J., Verbeek, H., Heip, C.H.R., 2003. Large-scale spatial patterns in estuaries: estuarine macrobenthic communities in the Schelde estuary, NW Europe. *Estuarine, Coastal and Shelf Science* 57, 335-355.

Ysebaert, T., Meire, P., Coosen, J., Essink, K., 1998. Zonation of intertidal macrobenthos in the estuaries of Schelde and Ems. *Aquatic Ecology* 32, 53-71.

Yu, O.H., Lee, H.-G., Lee, J.-H., 2012. Influence of environmental variables on the distribution of macrobenthos in the Han River Estuary, Korea. *Ocean Science Journal* 47, 519-528.

Zeppilli, D., Sarrazin, J., Leduc, D., Arbizu, P.M., Fontaneto, D., Fontanier, C., Gooday, A.J., Kristensen, R.M., Ivanenko, V.N., Sørensen, M.V., Vanreusel, A., Thébault, J., Mea, M., Allio, N., Andro, T., Arvigo, A., Castrec, J., Danielo, M., Foulon, V., Fumeron, R., Hermabessiere, L., Hulot, V., James, T., Langonne-Augen, R., Le Bot, T., Long, M., Mahabror, D., Morel, Q., Pantalos, M., Pouplard, E., Raimondeau, L., Rio-Cabello, A., Seite, S., Traisnel, G., Urvoy, K., Van Der Stegen, T., Weyand, M., Fernandes, D., 2015. Is the meiofauna a good indicator for climate change and anthropogenic impacts? *Marine Biodiversity* 45, 505-535.

ZoBell, C.E., 1946. Studies on redox potential of marine sediments. *Bulletin - American Association of Petroleum Geologists* 30, 477-513.

Annexure

ECOLOGICAL GROUPS OF MACROFAUNA IN THE

KODUNGALLUR-AZHIKODEESTUARY

JULY 2009-JUNE 2011

TABLE I

Ecological groups classification of benthic polychaetes in KAE

Polychaete species	EG	G
Lumbrineris latreilli	II	I
Lumbrineris simplex	II	I
Ninoe notocirrata	II	I
Lumbrineris heteropoda	II	I
Diopatra neapolitana	I	I
Namalycastis indica	IV	II
Dendronereis arborifera	IV	II
Dendronereides heteropoda	I	I
Dendronereis aestuarina	I	I
Perinereis cavifrons	I	I
Neanthes glandicincta	III	II
Ceratonereis costae	II	I
Nephtys oligobranchia	II	I
Nephtys polybranchia	II	I
Phyllodoce sp.	II	I
Glycinde bonhourei	IV	II
Glycera alba	IV	II
Glycera tridactyla	II	I
Owenia fusiformis	II	I
Ficopomatus sp.	III	II
Paraprionospio sp.	IV	II
Prionospio cirrifera	IV	II
Prionospio polybranchiata	IV	II
Pseudopolydora kempi	III	II
Pista indica	II	I
Ophelia capensis	I	I
Cossura sp.	IV	II
Capitella sp.	V	II
Heteromastus similis	IV	II
Heteromastides bifidus	IV	II
Parheteromastus tenuis	V	II
Notomastus sp.	III	II

TABLE II

Ecological groups classification of benthic malacostracans in KAE

Malacostracan species	EG	G
Metapenaeus affinis	II	I
Penaeus indicus	II	I
Neorhynchoplax alcocki	III	II
Philyra malefactrix	II	I
Scylla serrata	I	I
Alpheus malabaricus	II	I
Diogenes alias	II	I
Lucifer hanseni	III	II
Quadrivisio bengalensis	I	I
Melita zeylanica	I	I
Americorophium triaeonyx	III	II
Corophium volutator	III	II
Victoriopisa chilkensis	I	I
Jassa falcata	V	II
Psammogammarus	II	I
Gammarus tigrinus	II	I
Grandidierella megnae	I	I
Iphinoe	I	I
Cyathura indica	III	II
Sphaeroma annandalei	III	II
Cirolana fluviatilis	II	I
Ctenapseudes chilkensis	III	II
Pagurapseudopsis gymnophobia	III	II
Miyakella nepa	II	I
Gastrosaccus dunckeri	II	I

Table III

Ecological groups classification of macrofaunal molluscs in KAE

Molluscan species	EG	G
Villorita cyprinoides	I	I
Marcia recens	I	I
Dosinia sp.	I	I
Meretrix casta	I	I
Hiatula sp.	I	I
Arcuatula senhousia	III	II
Crassostrea bilineata	III	II
Saccostrea cucullata	III	II
Murex trapa	I	I

Table IV

Ecological groups classification of ‘other group’ of macrofaunal in KAE

Other groups	EG	G
Ophiothrix fragilis	I	I
Obelia bidentata	II	I
Balanus improvisus	III	II
Chironomid larva	III	II
Nemertea	III	II

List of publications

1. **Jayachandran P.R.**, S. Bijoy Nandan, V. F. Sanu, M. Jima, P.R. Anu, N. D. Don Xavier, Philomina Joseph, A.M. Midhun, C.V. Asha (2018) Authentication of *Nassodonta insignis* H. Adams, 1867 (Gastropoda: Nassariidae) from the Kodungallur - Azhikode backwater, southwest coast of India using mitochondrial COI marker. *Indian Journal of Marine Science* (Accepted for publication Ref. No. IJMS/MS 3413 Date: 09/12/2016) ISSN: 0379-5136 (Print) 0975-1033 (Online)

2. **Jayachandran P.R.**, M.P. Prabhakaran, C.V. Asha, Akhilesh Vijay, S. Bijoy Nandan (2015) First report on mass reproductive swarming of a polychaete worm, *Dendronereis aestuarina* (Annelida, Nereididae) Southern 1921, from a freshwater environment in the south west coast of India, *International Journal of Marine Science*, Vol.5, No.3 1-7. DOI: 10.5376/ijms.2016.06.0054) ISSN 1927-6648

3. **Jayachandran, P.R.**, S. Bijoy Nandan, O.K. Sreedevi, V.F. Sanu, 2013. Environmental influence on fish assemblage in a tropical estuary (Kodungallur-Azhikode Estuary), south west coast of India. *International Journal of Marine Science* 3(2) 4-16. DOI: 10.5376/ijms.2016.06.0054) ISSN: 1927-6648

4. **Jayachandran, P.R.**, S. Bijoy Nandan 2012. Assessment of trophic change and its probable impact on tropical estuarine environment (the Kodungallur-Azhikode estuary, India). *Mitigation and Adaptation Strategies for Global Change.* 17, 837-847. *ISSN:* 1381-2386 (print version) ISSN*:* 1573-1596 (online)

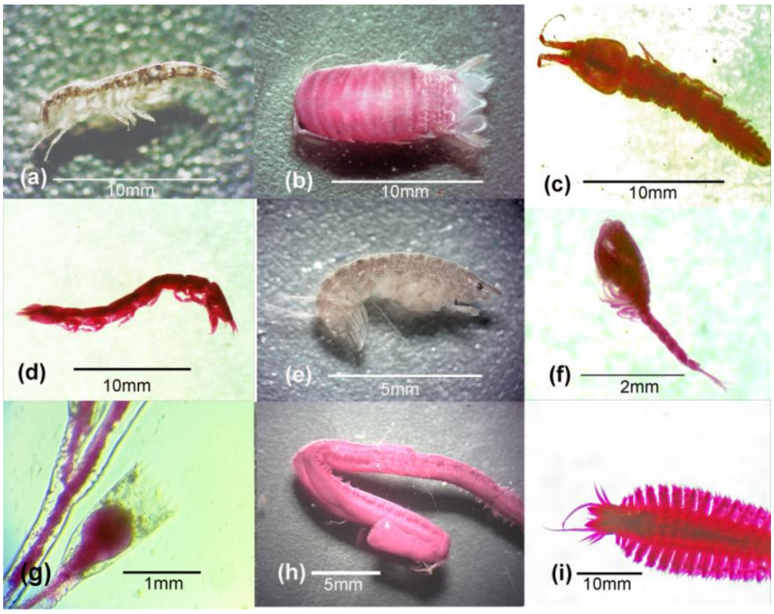
5. **Jayachandran, P.R.**, S. Bijoy Nandan, O.K.Sreedevi (2012). Water quality variations and nutrient characteristics of Kodungallur-Azhikode estuary, Kerala, *Indian Journal of Marine Science* 41, 180-187. ISSN: 0379-5136 (Print) 0975-1033 (Online)

6. Oliver P.G., **P.R. Jayachandran**, S. Bijoy Nandan (2016) *Cuspidaria cochinensis* Preston, 1916 transferred to the Corbulidae (Mollusca, Bivalvia) and assigned to the genus *Cuneocorbula*. *Marine Biodiversity.* DOI: 10.1007/s12526-016-0578-y. ISSN: 1867-1616 (Print) 1867-1624 (Online)

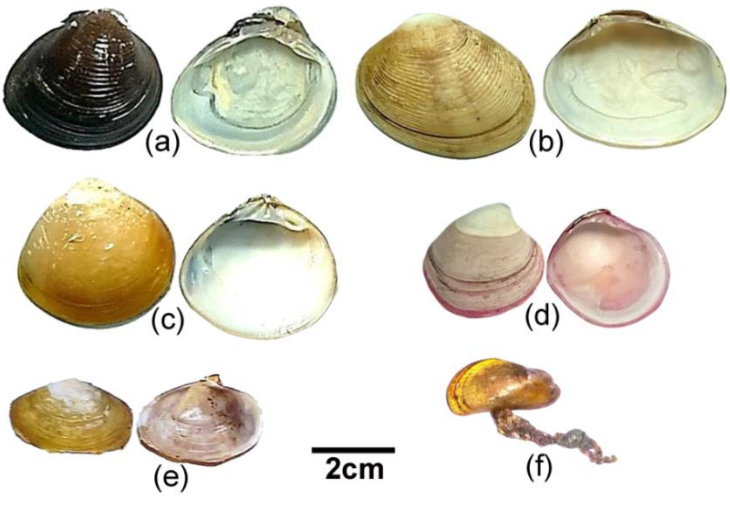
7. Bijoy Nandan, S., **P.R. Jayachandran**, O.K. Sreedevi (2012). Temporal pattern of fish production in a microtidal tropical estuary in the south-west coast of India. *Indian Journal of Fisheries* 59, 17-26. ISSN: 0970-6011

8. Bijoy Nandan S., **P.R. Jayachandran**, O.K. Sreedevi (2014) Spatio-temporal pattern of primary production in a tropical coastal wetland (Kodungallur-Azhikode Estuary), south west coast of India. *Journal of Coastal Development* 17: 392. doi: 10.4172/1410-5217.1000392. ISSN: 1410-5217

PLATE I



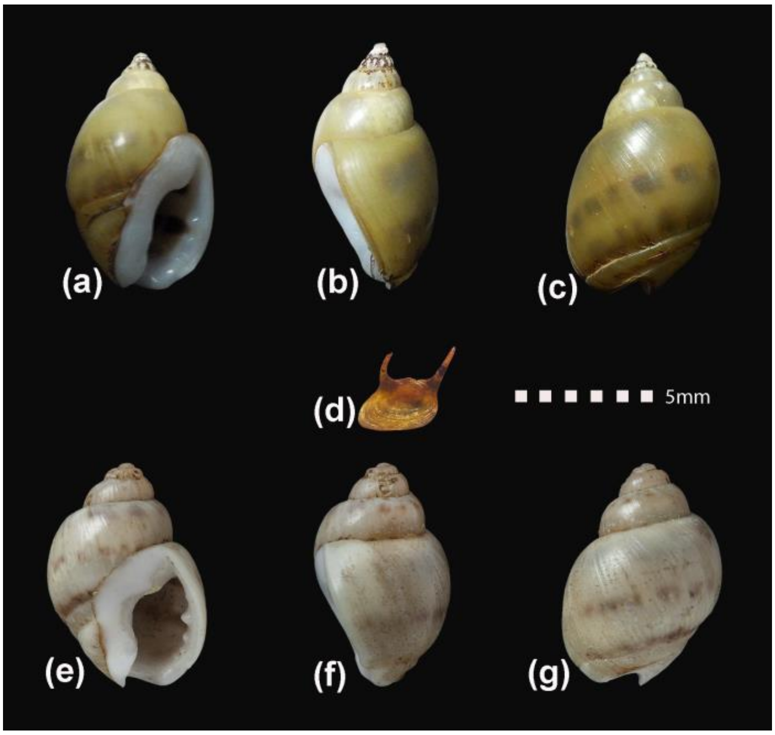
Macrofaunal species (a) *Americorophium triaeonyx,* (b) *Cirolana fluviatilis,* (c) *Ctenapseudes chilkensis,* (d) *Cyathura indica,* (e) *Grandidierella megnae*, (f) *Iphinoe* sp., (g) *Obelia bidentata,* (h) *Glycera* sp., and (i) *Dendronereis aestuarina* collected from the Kodungallur-Azhikode estuary



Macrofaunal molluscs (a) *Villorita cyprinoides,* (b) *Marcia recens,* (c) *Meretrix casta,* (d) *Dosinia* sp., (e) *Hiatula* sp., and (f) *Arcuatula senhousia* collected from the Kodungallur-Azhikode estuary

PLATE 2

The specimen of *Cuneocorbula cochinensis* collected from the Kodungallur-Azhikode estuary



(a-g) The specimen of *Nassodonta insignis* collected from the Kodungallur-Azhikode estuary (d) Operculum of specimen (e-g) *Nassodonta insignis* H. Adams, 1866, lectotype registration number NHMUK 1878.1.28.428.