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GROFLO Final Report Part 2

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COORDINATOR

NETHERLANDS INSTITUTE OF ECOLOGY
CENTRE FOR ESTUARINE AND COASTAL ECOLOGY
P.O. BOX 140
4400 AC YERSEKE
THE NETHERLANDS

PROF. DR. HEMMINGA MARTEN
E-M : hemminga@cemo.nioo.knaw.nl
TEL : 31-113-577472
FAX : 31-113-573616

CONTRACTORS

KENYA MARINE AND FISHERIES RESEARCH INSTITUTE
P.O. BOX 81651
MOMBASA
KENYA

DR. KAZUNGU JOHNSON
E-M : jkazungu@recoscix.com
TEL : 254-11-475151
FAX : 254-11-472215

FREE UNIVERSITY OF BRUSSELS
INSTITUTE OF ENVIRONMENTAL RESEARCH
PLEINLAAN 2
1050 BRUSSELS
BELGIUM

PROF. DR. TACK JURGEN
E-M : jtack@vub.ac.be
TEL : 32-2-6293410
FAX : 32-2-6293403

UNIVERSITY OF DAR ES SALAAM
INSTITUTE OF MARINE SCIENCES
P.O. BOX 668
ZANZIBAR
TANZANIA

DR. MTOLERA MATERN
E-M : mtolera@zims.udsm.ac.tz
TEL : 255-54-30741
FAX : 255-54-33050

STOCKHOLM UNIVERSITY
DEPARTMENT OF ZOOLOGY

10691 STOCKHOLM
SWEDEN

DR. JOHNSTONE RON
E-M : ron.johnstone@awtensight.nsw.gov.au
TEL : 61-2-93340735
FAX : 61-2-93340948

EDUARDO MONDLANE UNIVERSITY
DEPARTMENT OF BIOLOGICAL SCIENCES
P.O. BOX 257
MAPUTO
MOZAMBIQUE

MR. GOVE DOMINGOS
E-M : idomio@biologia.uem.mz
TEL : 258-1-490009
FAX : 258-1-492176

UNIVERSITY OF LISBON
GUIA MARINE LABORATORY
ESTRADA DO GUINCHO
2750 CASCAIS
PORTUGAL

DR. PAULA JOSE
E-M : jpaula@fc.ul.pt
TEL : 351-1-4869211
FAX : 351-1-4869720

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Kenya Marine and Fisheries Research Institute

Groundwater outflow dynamics and circulation at Diani and Nyali mesitidal beaches in Kenya

J. U. Kitheka
Kenya Marine and Fisheries Research Institute, Mombasa, Kenya

Introduction

Groundwater is a precious and the most widely distributed resource of the earth and unlike any other mineral resource, it gets annual replenishment from the meteoric precipitation. It is the largest source of freshwater on the planet earth excluding the polar icecaps and glaciers. The amount of groundwater within 800m from the ground surface is over 30 times the amount in all freshwater lakes and reservoirs and about 3000 times the amount in stream channels at any one time (Raghunath, 1990). However, despite the enormous groundwater reservoir and supply into the world oceans, little research has been conducted to understand the dynamics of ground-water outflow in Eastern Africa coastal zones.

Groundwater outflow in coastal areas is important in terms of moderation of lagoonal salinity and also boosting of nutrient supply in coastal areas. It is therefore of great importance in sustaining coastal nearshore ecosystems. The height of the water table along the coast influences the stability of structures founded on soils or sand and may be a limiting factor for agricultural land use because of saltwater intrusion. Groundwater outflow may also stratify the water column in tidal creeks with low rates of water-exchange (Kitheka, 1998). Bokuniewicz (1980) observed that, the submarine outflow of groundwater across the seafloor is an integral part of the coastal hydrography and it is usually the most poorly documented components of the freshwater supply and rarely measured directly. In areas where stream flow is small, groundwater seepage may dominate the freshwater discharge controlling the distribution of salinity in the coastal zone. It is also that fraction of the groundwater discharge that maintains the position of the freshwater/saltwater interface in coastal aquifers (Bokuniewicz, 1990). There is at present paucity of information and data on the patterns of groundwater outflow into the ocean in most parts of the world and the scenario is worse in tropical parts of Africa.

Although studies of groundwater phenomena in unconfined and confined aquifers are numerous and a voluminous body of scientific literature dealing with the beach environment exists (Duncan, 1964; Holman and Guza, 1994; Packwood, 1983; IGBP, 1994; Gillian et al, 1974), only a few studies concern the dynamics of the beach groundwater system (Baird & Horn, 1996; Copper, 1959; Dominick &Wilkins, 1971). Water-table fluctuations and associated beach-face processes have not been studied extensively although importance of water-table fluctuation as a mechanism affecting beach-face deposition and erosion is well recognized (Grant, 1946 & 1948; Emery & Foster, 1948; Lanyon, et al, 1982).

Objectives

The objectives of the present study are twofold, (1) establish the patterns of groundwater discharge in representative coastal beaches of Kenya and (2) establish its influence on beach matrix and lagoon temperature and salinity changes. Special attention is paid on the dynamics of groundwater outflow and tide and corresponding variations on the rates of groundwater outflow into the sea. This study was carried out under the auspices of the EU-INCO GROFLO project titled 'Anthropogenically-induced changes in groundwater outflow and quality, and functioning of the Eastern Africa nearshore ecosystems.

Description of study areas

This study was conducted in two (2) locations in the South and North Coast region of Kenya (Figures 1). Nyali beach (04° 03.052 S and 039° 42.371 E) is located approximately 7km from Kenya Marine and Fisheries Research Institute Headquarters in Mombasa. It occurs within

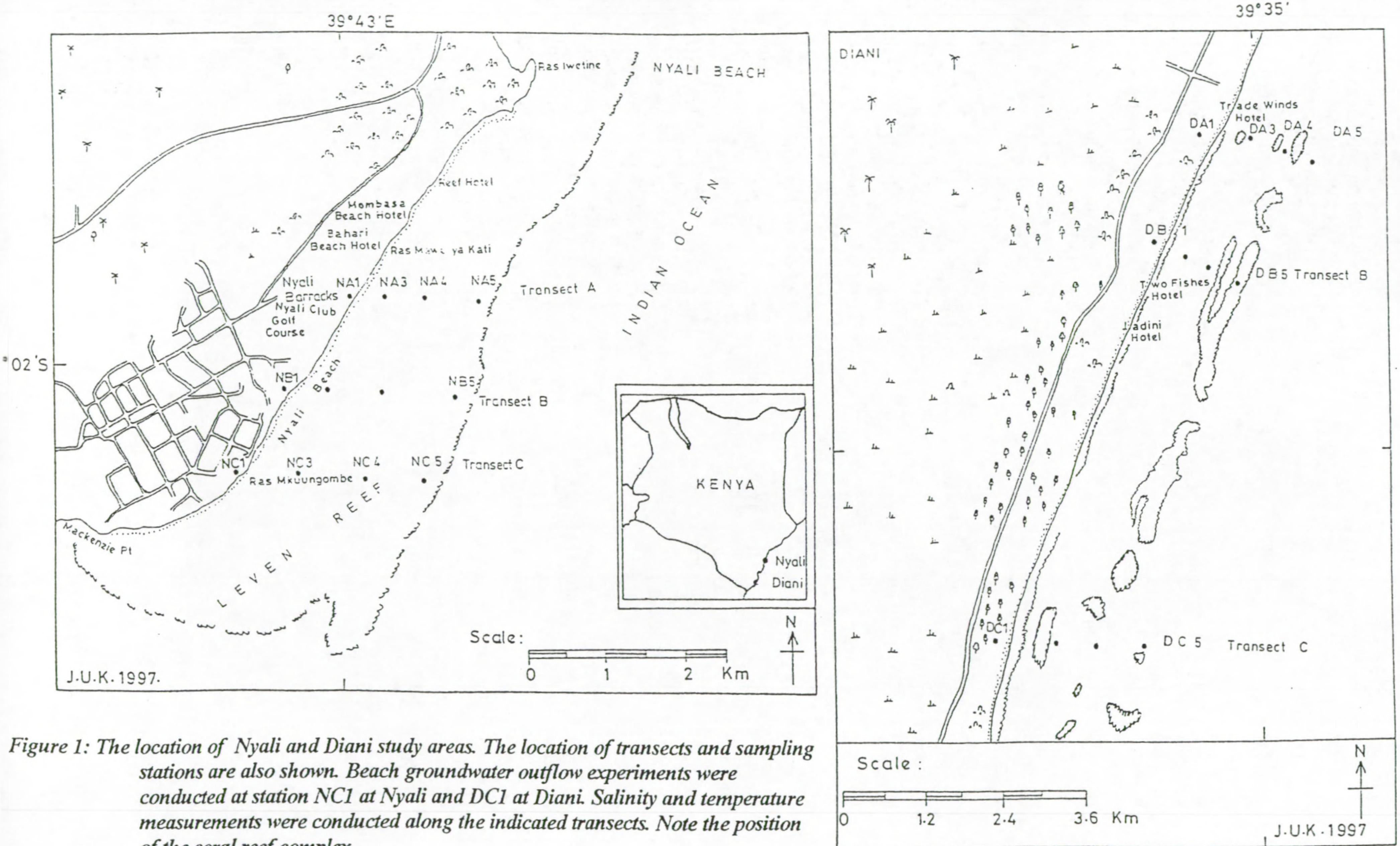


Figure 1: The location of Nyali and Diani study areas. The location of transects and sampling stations are also shown. Beach groundwater outflow experiments were conducted at station NC1 at Nyali and DC1 at Diani. Salinity and temperature measurements were conducted along the indicated transects. Note the position of the coral reef complex.

the Coastal belt and geology is characterized by the presence of Pleistocene coral limestone complex running parallel along the beach. Carbonate sand consists of fine to medium course sand and in some places pebbles and gravels are found lying on top of partially cemented coral limestone rock base. Fine to very fine silt component is usually about 0.25m thick while that of course sand to very coarse sand is roughly 0.20m. The vertical profile of sand distribution shows that the distribution is usually complex and layers of fine sand are sometimes embedded on layers of coarse to vary coarse sand. Groundwater outflow occurs mainly at the lower margins of the beach just below the saturated zone, a zone in which sediment thickness is <0.10m. The Thickness of coarse and medium fine sand is often < 0.1m. The beach gradient is in the order of 2 to 3% and changes as a result of alongshore transport of sediments. The aquifer supplying groundwater to Nyali beach extends into the North Mainland region. Main population concentration zones in Mombasa occur in areas, which have high groundwater potential. These zones are covered by porous Pleistocene coral limestone rock and permeable Kilindini sands. Enormous groundwater potential is evidenced by extensive community-based exploitation of groundwater.

Diani beach (040° 19.222'S and 39° 34.579' E) is located in the south coast region approximately 50 Km from Mombasa. As in case of Nyali zone, Diani region is dominated by Pleistocene coral limestone rocks and therefore sand found along the beach is mainly carbonate type. The medium fine sand is variable but usually 0.3 - 0.2m thick while the coarse sand is <0.2m thick. As in case of Nyali Beach, thickness of sand varies depending on the intensity longshore sediment transport. Diani beach gradient is roughly 2.0%, but that of the beach berm can be as high as 29% due to beach erosion. As in Nyali, groundwater aquifer consists mainly of Pleistocene carbonate rocks and sands.

The extents of saturated and unsaturated zones within both beaches are spatially non-uniform. At Diani beach, the unsaturated zone at low tide extends 50m from the beach berm while the saturated zone is usually about 30m wide, but the zone of intensive groundwater outflow is usually narrow, often < 10m.

The Diani catchment area has one of the most high yielding groundwater aquifer in the whole of coastal region of Kenya. With the financial assistance from donor community, in particular SIDA/SAREC, there has been extensive drilling of boreholes in the south coast region of Kenya. Water supply in the southern zone of Mombasa Municipality is depended on withdrawal from Tiwi boreholes found near Diani. Semi-diurnal tides, wind-set up waves as well as various hydroclimatic forcing related to beach swash dynamics influence Beaches. There is no river discharge in either of the study areas.

Coastal region of Kenya is influenced by seasonally reversing North- and SouthEast monsoons. NorthEast monsoon (NEM) occurs in the period between September and March and is characterized by low and shorter rainy season. South- East monsoon (SEM) occurs in the period between March and September and is characterized by longer rainy season and higher rainfall. Mean annual rainfall is in the order of 700 and 1000mm and there are large inter-annual variations related El Nino and La Nina Southern Oscillations. Highest monthly rainfall (~ 250mm) in SEM and NEM occur in May and November respectively. The period between October and November 1997 experienced very heavy rains associated with the El Nino Southern Oscillation Phenomena. The lowest temperature is recorded in July (24°C) and the highest (31°C) occur in February. Main uses of groundwater include agricultural (mainly irrigation), industrial and domestic uses particularly in homes, hotels, hospitals and schools.

Materials and methods

Measurement on beach ground water table were conducted using perforated piezometers located along a transect running perpendicular to the beach. The piezometers were set at an interval of 10 to 15m apart. Figure 2 depicts the experimental set-up. This approach is based on pioneering work of Duncan (1964) and by Emery & Foster (1948). The piezometers consist of 0.05m diameter PVC tubes approximately 2.5m long designed with modifications according to Dominic (1971). The lower 0.3m is perforated with many tiny holes, which allow free exchange of water, and prevents sediments from entering the tubes. The piezometers were inserted when groundwater level was >0.05m. Location of transects dependent on the extent of groundwater outflow. Three (3) piezometers were usually installed along the transect as soon as the saturated zone of the beach was exposed during ebb tide. Water level was then measured after every 30

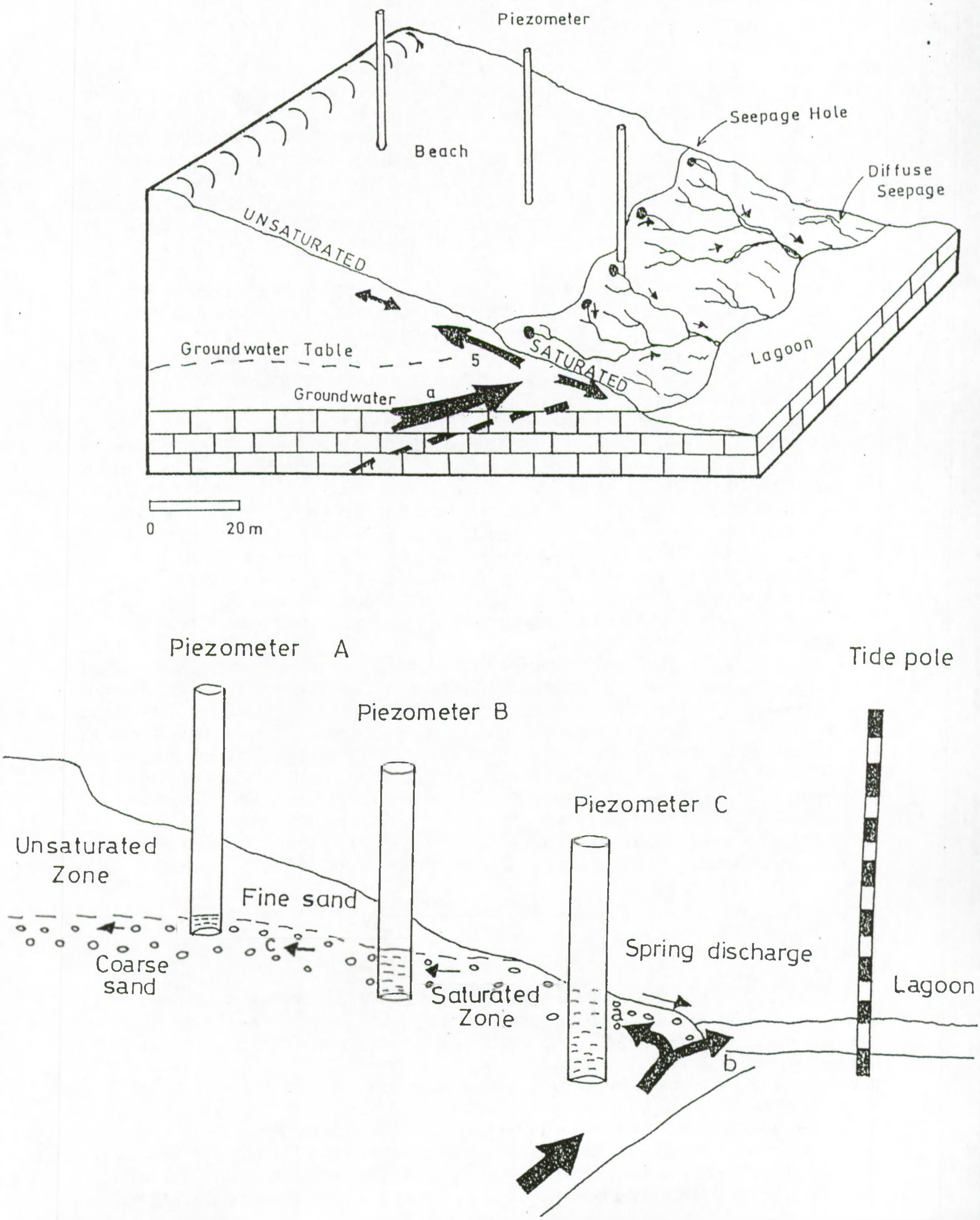


Figure 2: The experimental set-up and location of piezometers in different positions of the beach. The modes of groundwater seepage are illustrated in the upper diagram.

minutes by sounding using a thin wire. Apart from level, water salinity and temperature were also measured at an interval of 30 minutes using Anderaa S-T meter. The Aanderaa salinometer has a long cable, which enables sensor to be inserted into the 0.03m opening of the piezometer. In order to minimize water displacement as a result of insertion of salinity and temperature sensor and cable into the piezometers, water level was determined before salinity and temperature were measured. The measurement process usually started once the saturated zone at the base of the beach was exposed during low tide and it usually took 4 to 6 hours to complete measurements before flood tide covers the entire beach. Measurements were stopped once the flood tide water covered the lower piezometer. Tidal elevation was measured using a tide pole installed at the lowest water point in the lagoon. Measured tidal elevations were compared with the predicted water levels and found to be in agreement. Measurements were also conducted on run-up dynamics and wave heights according to Baird & Horn (1996). Beach sediment samples were taken along the transect and particle–size distribution were determined by dry sieving using sieves of different mesh sizes. Sediment samples were carefully taken at the upper, middle and bottom layers. Beach profiles and gradients were determined using standard surveying techniques using quick set level and surveying staff with foresight and backsight measurements being taken every 5m from a known elevation above the beach berm to the lowest point at the base of the beach.

With a given stretch of the beach, numerous groundwater springs occur and for monitoring purpose only a few were chosen depending on the representativeness of the groundwater spring to the overall outflow along the entire stretch of the beach. Discharge rates in selected groundwater springs were determined at an interval of 30 minutes by measuring spring hydraulic gradient, width and flow velocities. The springs were exposed for upto 4 hrs during low tide so measurement process took about the same time. Simultaneously, water salinity and temperature changes in spring water were also determined. Sediment concentrations were determined for one (1) litre water-sediment mixture sampled at an interval of 30 minutes. Later as the floodwater covered the saturated zones, water samples were taken in uprush and backwash according to Hegge & Masselink (1991). During this time, measurements on wave-height and speed were determined.

Within the lagoons, water depths, salinity, temperature and conductivity were measured along three transects established in the upper, middle and lower regions. Along each transect, 3 stations were chosen; (1) situated along the shore close to the beach, (2) located in the middle seagrass zone and (3) located in the coral reef region just below the reef flat. These measurements were conducted once a month using a rubber dinghy and both neap and spring tides were sampled. Most measurements were conducted during low water (Kitheka, 1997).

A broad-based surveys on groundwater salinity, temperature, total dissolved solids concentration were conducted in wells and boreholes distributed in the North and South coast region in both dry and wet season. Other variables measured include well depth, rock type, land use and water demand in relation to population distribution. Divers dataloggers specially designed for water level measurements were programmed to measure groundwater level every 10 minutes. These were installed at the North and South Bamburi Nature Trail for a period of one (1) month. Data were processed using Matlab software and compared with tidal measurements at the Bamburi lagoon. Data obtained in the field were supplemented with secondary information on rainfall, air temperature, evaporation, water demand and landuse.

Results

Groundwater outflow

At Nyali Beach, measurements on tide and groundwater outflow were conducted on August 20, September 17, November 13 and in December 2, 1997 while at Diani Beach they were conducted on December 3, 1997, January 21 and February 12, 1998. Measurements conducted in November represent condition in rainy season while those conducted in August; September-February and January-February period represented typical dry season. Tables 1 to 4 present data on minimum and maximum groundwater discharges, salinity and temperature. Mean values and standard deviations are also shown.

Groundwater outflow depends on the stage of the tide. In periods when tide does not go beyond 0.5m, there was a tendency for groundwater discharge to increase during low tide at both beaches (Figures 3 & 7). However in periods when the tidal elevation was upto 1.5m, there was a tendency for groundwater discharge to decrease during the low tide. However there is an initial rapid increase in discharge, which is followed by a decrease (Figures 4 & 8). In all cases lowest discharge occurred 1 to 1.5 hrs after lowest tide. The rapid initial rise in discharge occurs about 1 hr before the lowest tide.

Table 1. The descriptive statistics of variables associated with groundwater seepage streams at Nyali beach. There is wide salinity variations at Nyali as compared to Diani beach and this is related to the groundwater use and recharge inland.

Variable	Valid N	Mean	Minimum	Maximum	Std. Div.
Discharge (m^3/s)	10	0 .0142	0.00400	0.0240	0.0066
Salinity (psu)	10	19.1490	5.20000	29.7400	7.7809
Temperature (°C)	10	29.1690	27.25000	32.6300	1.5892
Rainfall (mm)	13	230.5154	0.0000	802.0000	263.1672

Table 2. Descriptive statistics of variables associated with groundwater seepage streams at Diani beach. Note that salinity at Diani is higher than that at Nyali beach due to greater seawater intrusion at Diani beach. Salinity variations show a narrow range at Diani beach.

Variable	Valid N	Mean	Minimum	Maximum	Std. Dev.
Discharge (m^3/s)	8	0. 0328	0. 01100	0.0580	0. 0184
Salinity (psu)	8	27.2875	21.71000	33.3000	3. 7128
Temperature (°C)	8	27.6575	21.83000	32. 6300	2. 6240
Rainfall (mm)	13	230.5154	0.00000	803.0000	263.1672

Beach groundwater outflow, salinity and temperature

A strong relationship exists between rainfall and groundwater discharge at Nyali beach ($r = 0.64$; $N=10$, $P<0.05$). The relationship between rainfall and changes in groundwater salinity is also strong ($r = 0.77$, $n=10$, $p< 0.05$). These results are indications of the fact that rainfall replenishes groundwater aquifers in the region. An increase in ground water discharge is associated with a decrease in salinity as a result of increased dilution.

Typical dry season measurements are those conducted on September 17, 1997 at Nyali beach. During this period groundwater discharge was low and varied from 0.012 m^3/s at low tide to 0.02 m^3/s about 2 hrs after low tide with tidal elevation = 0.7m (Figure 3). There was a corresponding decrease in salinity from 25.7 psu at low tide when tide = 0.1 to 18.6 psu when tide was 1.8m. This decrease in salinity is due to progressive discharge of brackish water from the aquifer adjacent the beach as the tide falls. Temperature increased from 28.3°C at low tide (at tide = 0.1m) to 29.8°C during peak groundwater discharge. The lowest salinity of 18.6 psu occurred at lowest groundwater flow rate, 1.5 to 2 hours after lowest tide.

On November 13, 1997 measurements represents typical conditions in rainy season at Nyali beach. Groundwater discharge also showed an increasing trend from $1.8 \times 10^3 m^3/s$ to $4.2 \times 10^3 m^3/s$ about 2 hrs after low water (Figure 4). However salinity was much lower and declined from 6.2 psu at low tide (tide = 0.2m) to 5.2 psu (tide = 0.2m) psu about 2.5hrs after low water (Figures 4). Similar patterns were observed in January 20, 1998, but the salinity (varied from 15.2 psu to 12.7 psu) showed a progressive increase from the lowest recorded in November. Increased recharge in rainy season increases the volume of groundwater in storage but continued withdrawal and discharge to the sea, reduces this volume and leads to seawater intrusion in dry season. This explains the relatively higher salinity groundwater noted in dry season.

There was a general observation that in rainy season, there is an increasing trend in groundwater discharge. This is followed by a salinity decrease. The salinity attain the minimum not at the lowest tide, but 1.5 to 2.0 hrs after low water, a period during which the tide starts rising. In dry season, there is a decreasing trend in groundwater discharge.

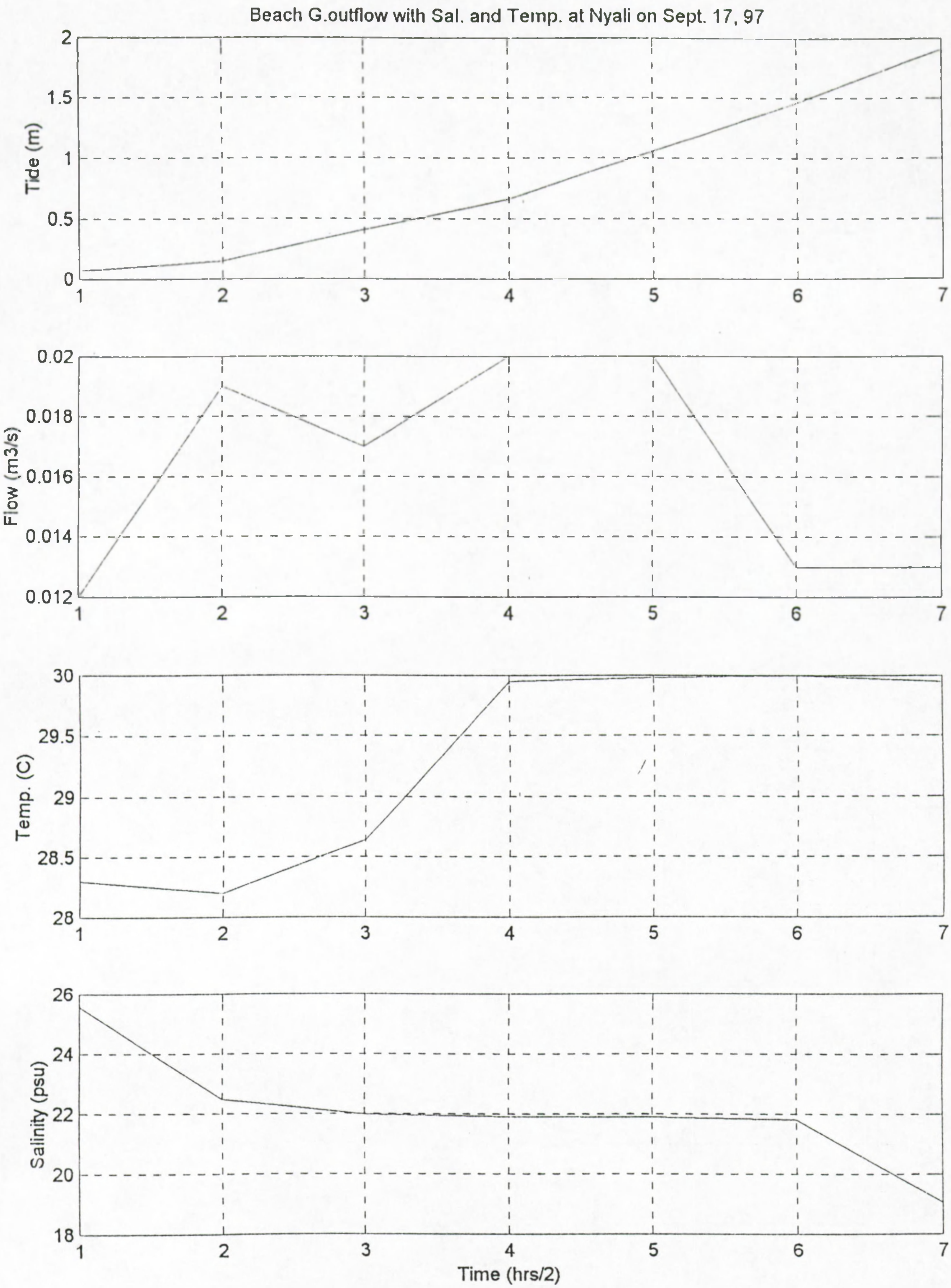


Figure 3: The relationship between groundwater discharge and variations in salinity and temperature at Nyali beach in dry season. Note the relatively low salinity of groundwater. Groundwater discharge increases as the tide falls while temperature and salinity decreases.

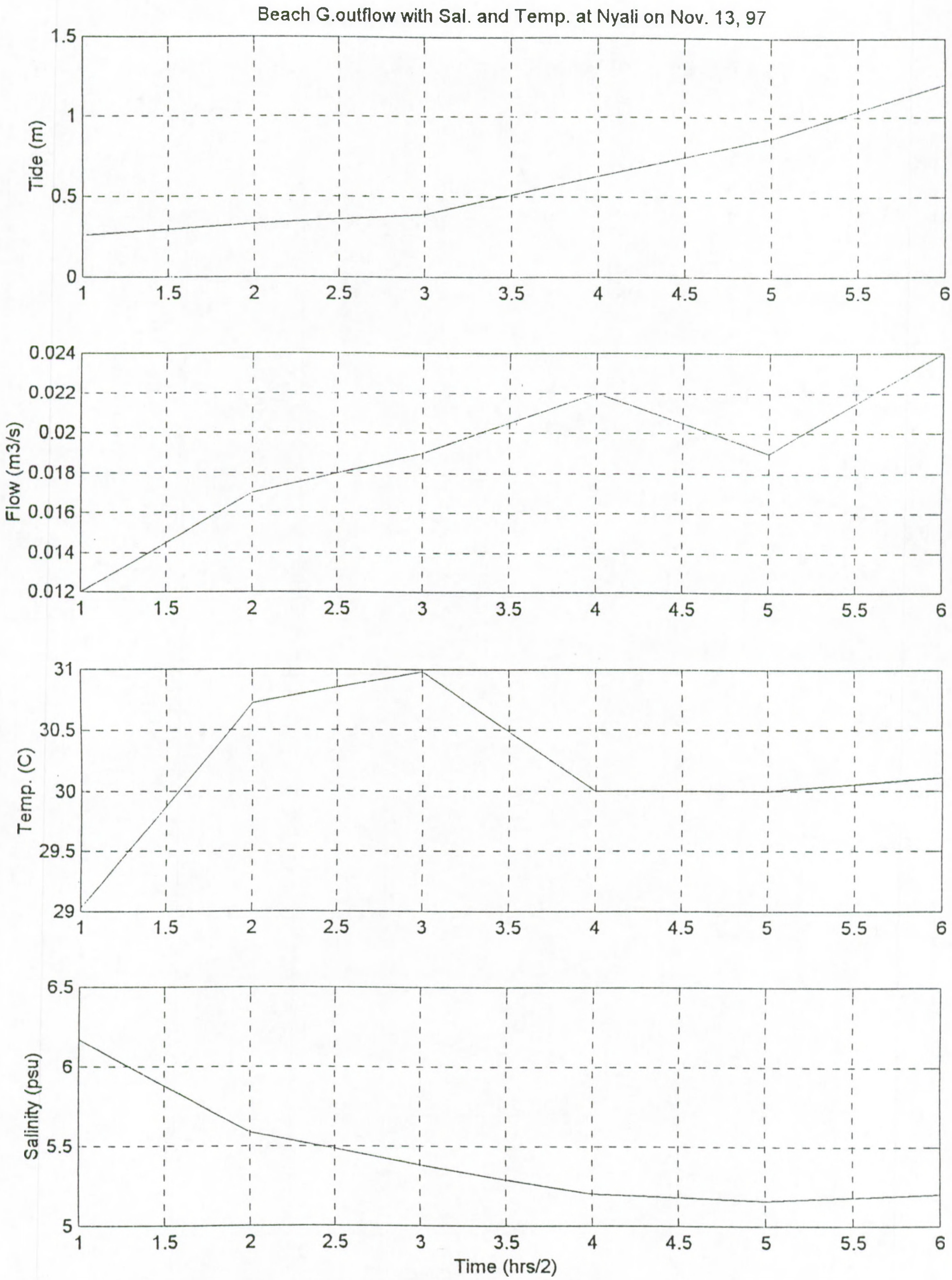


Figure 4: The relationship between groundwater discharge and variations of salinity and temperature at Nyali beach in rainy season. An initial low discharge is followed by a rapid increase and then a decrease. Salinity decreases while temperature increases.

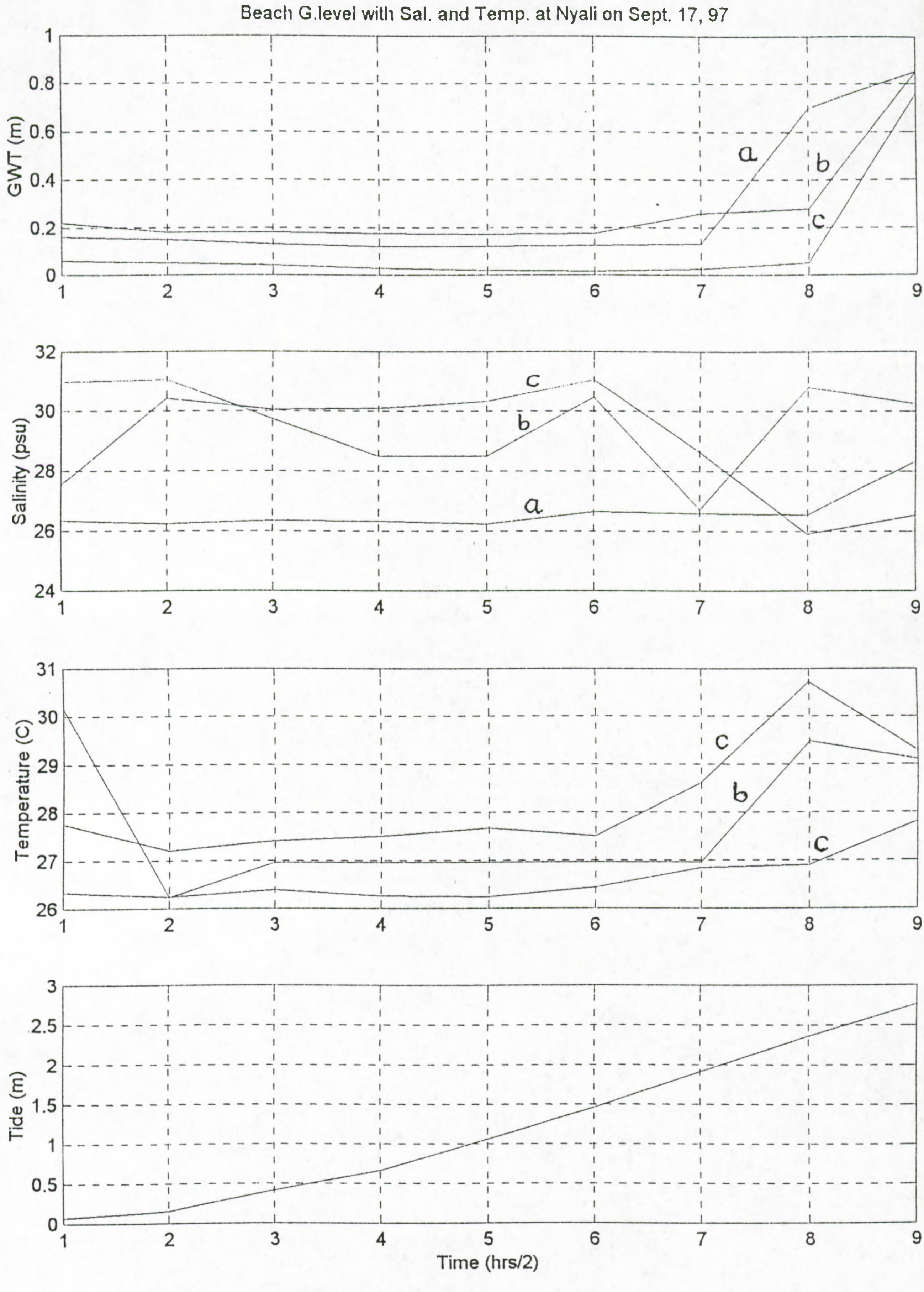


Figure 5: The relationship between the tide and beach groundwater table level at Nyali beach in dry season. There is a phase lag in the changes in groundwater level between the lower and upper zone of the beach. There is a rapid decline in salinity as the tide begins to rise.

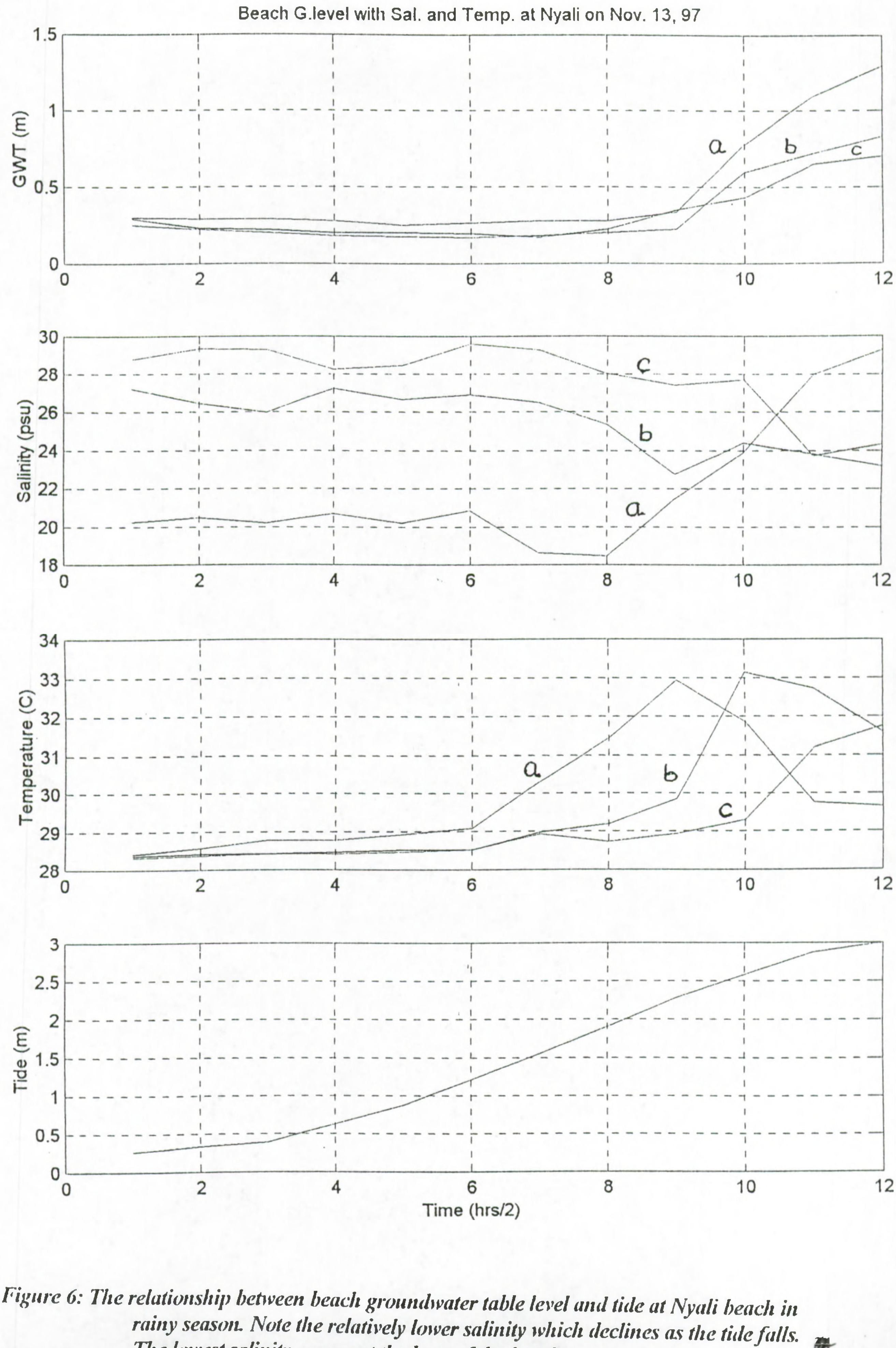


Figure 6: The relationship between beach groundwater table level and tide at Nyali beach in rainy season. Note the relatively lower salinity which declines as the tide falls. The lowest salinity occurs at the base of the beach.

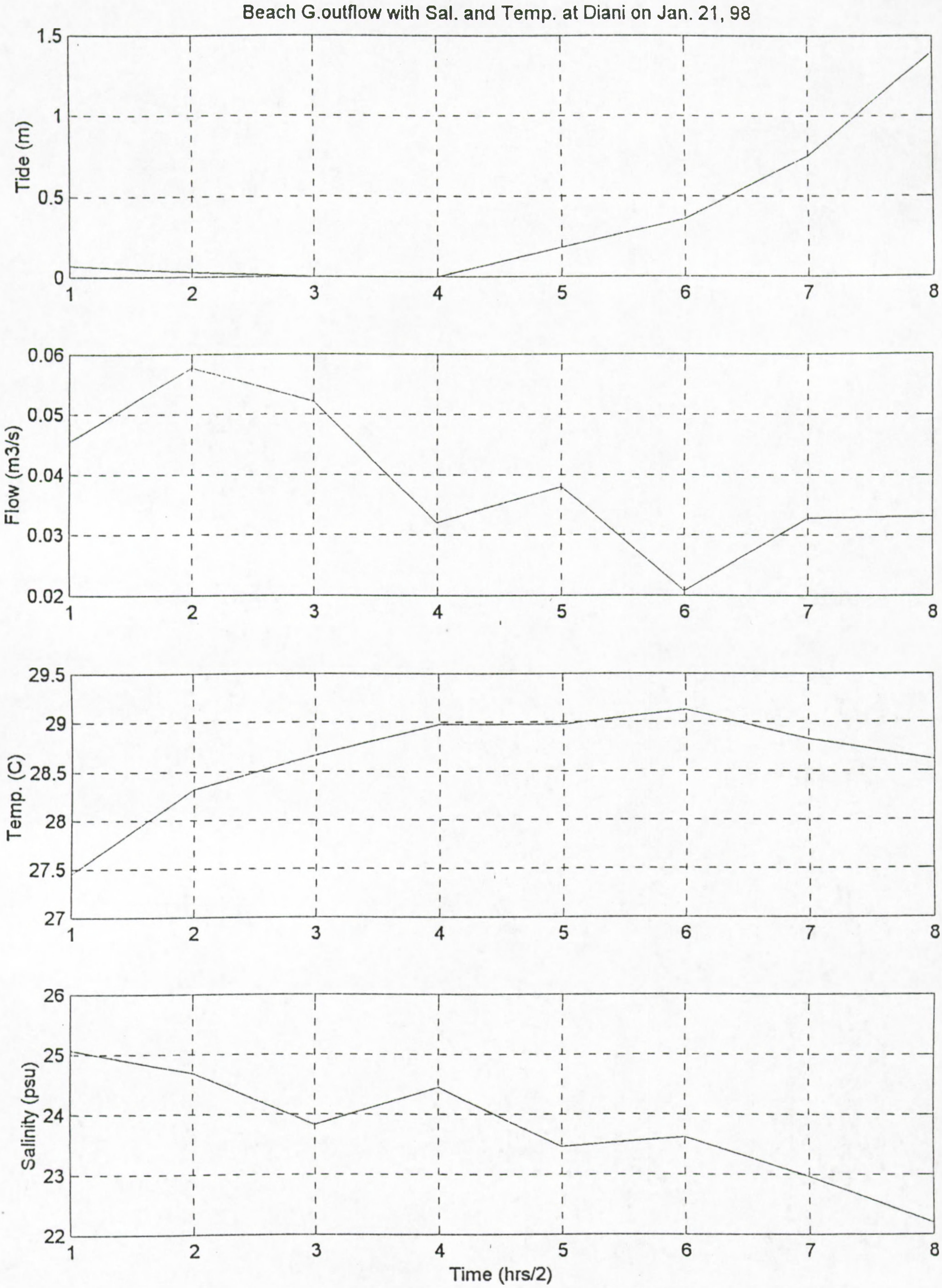


Figure 7: The relationship between groundwater discharge and variations in salinity and temperature at Diani beach in dry season. Note the relatively low salinity. Groundwater discharge increases as the tide falls while temperature and salinity decreases.

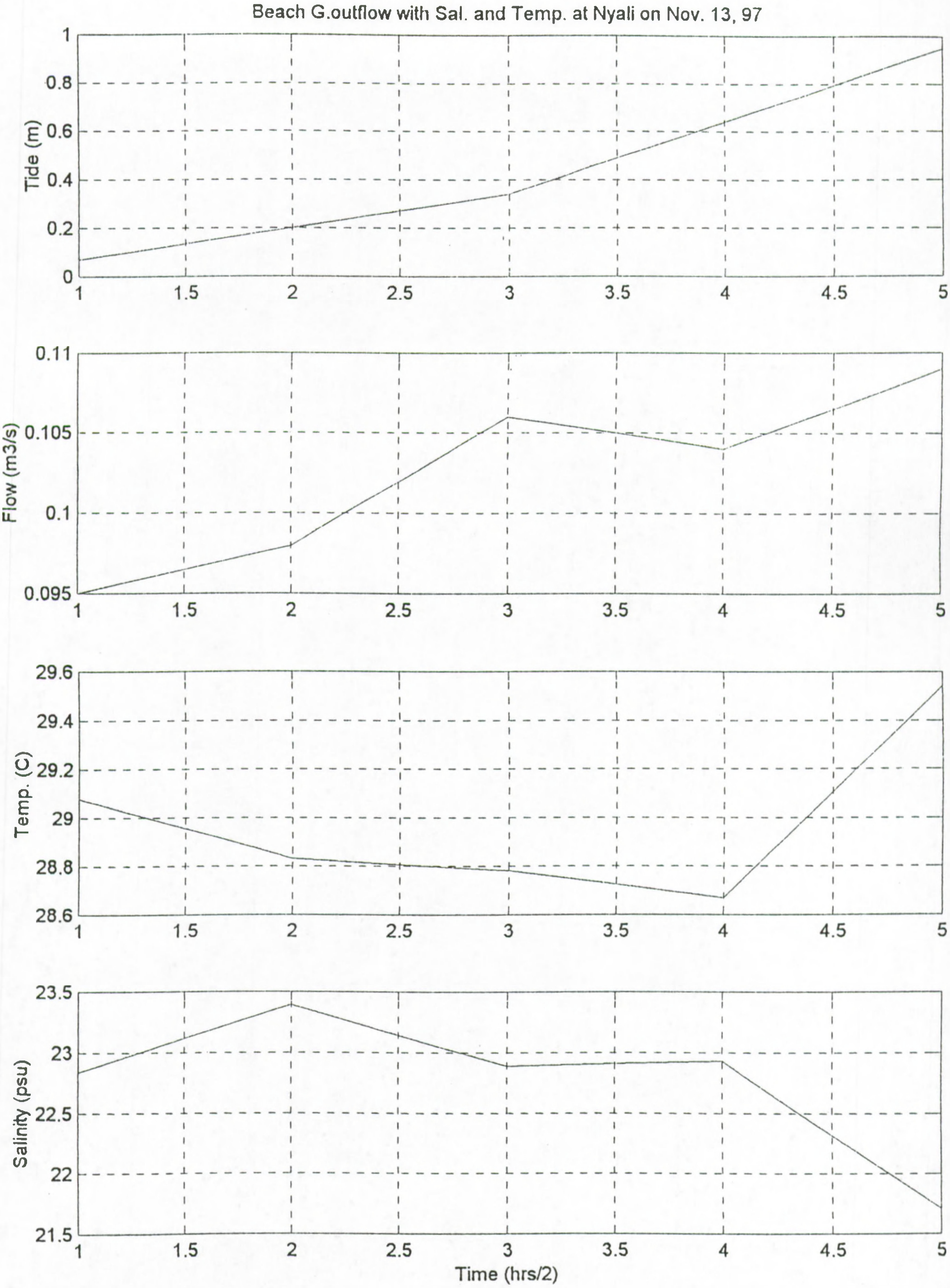


Figure 8: The relationship between groundwater discharge and variations of salinity and temperature at Diani beach in rainy season.

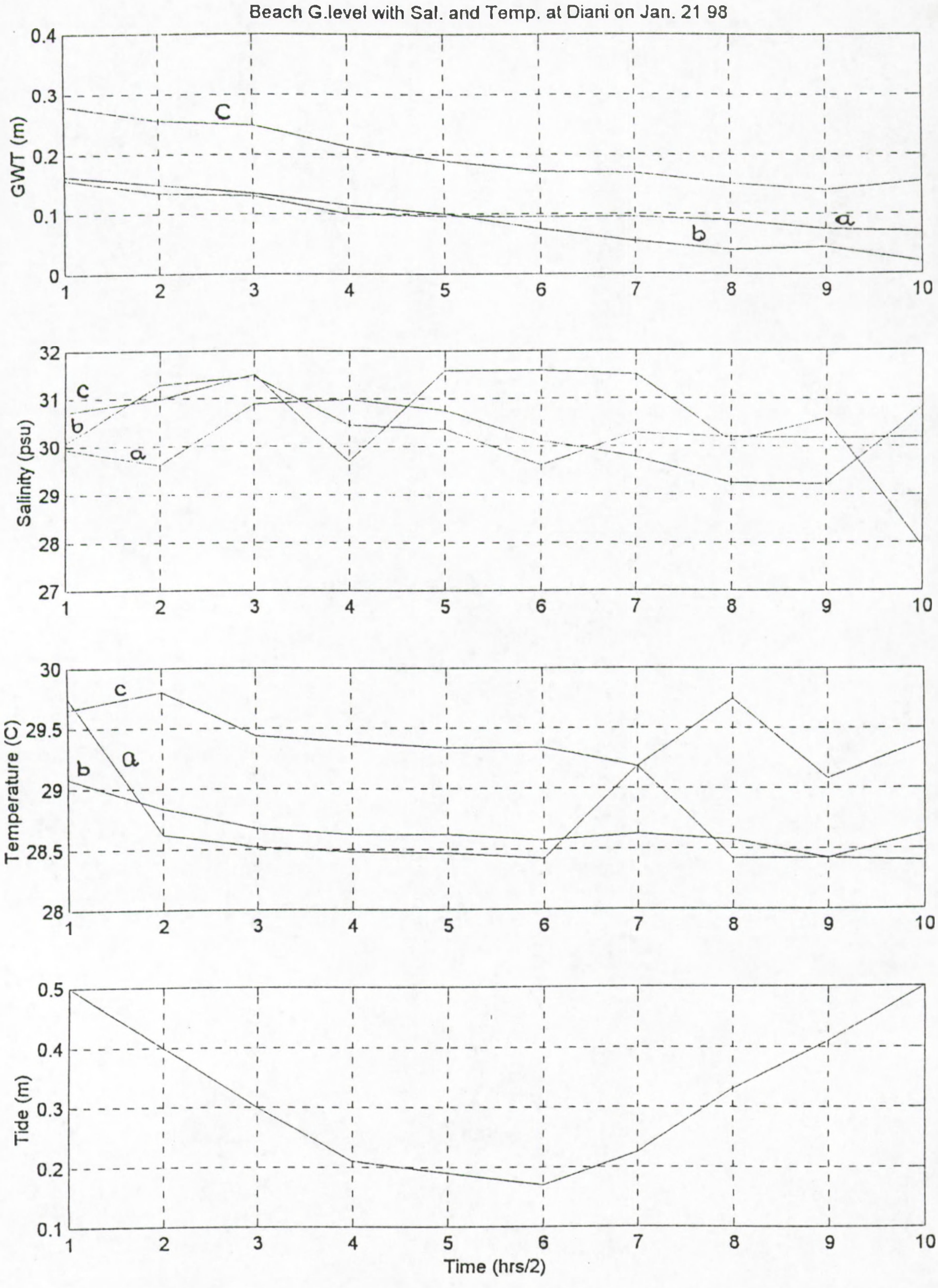


Figure 9: The relationship between the tide and groundwater table level at Diani beach in dry season. There is a phase lag in changes in groundwater level between the lower and upper zone of the beach. There is a rapid decline in salinity as the tide begins to rise.

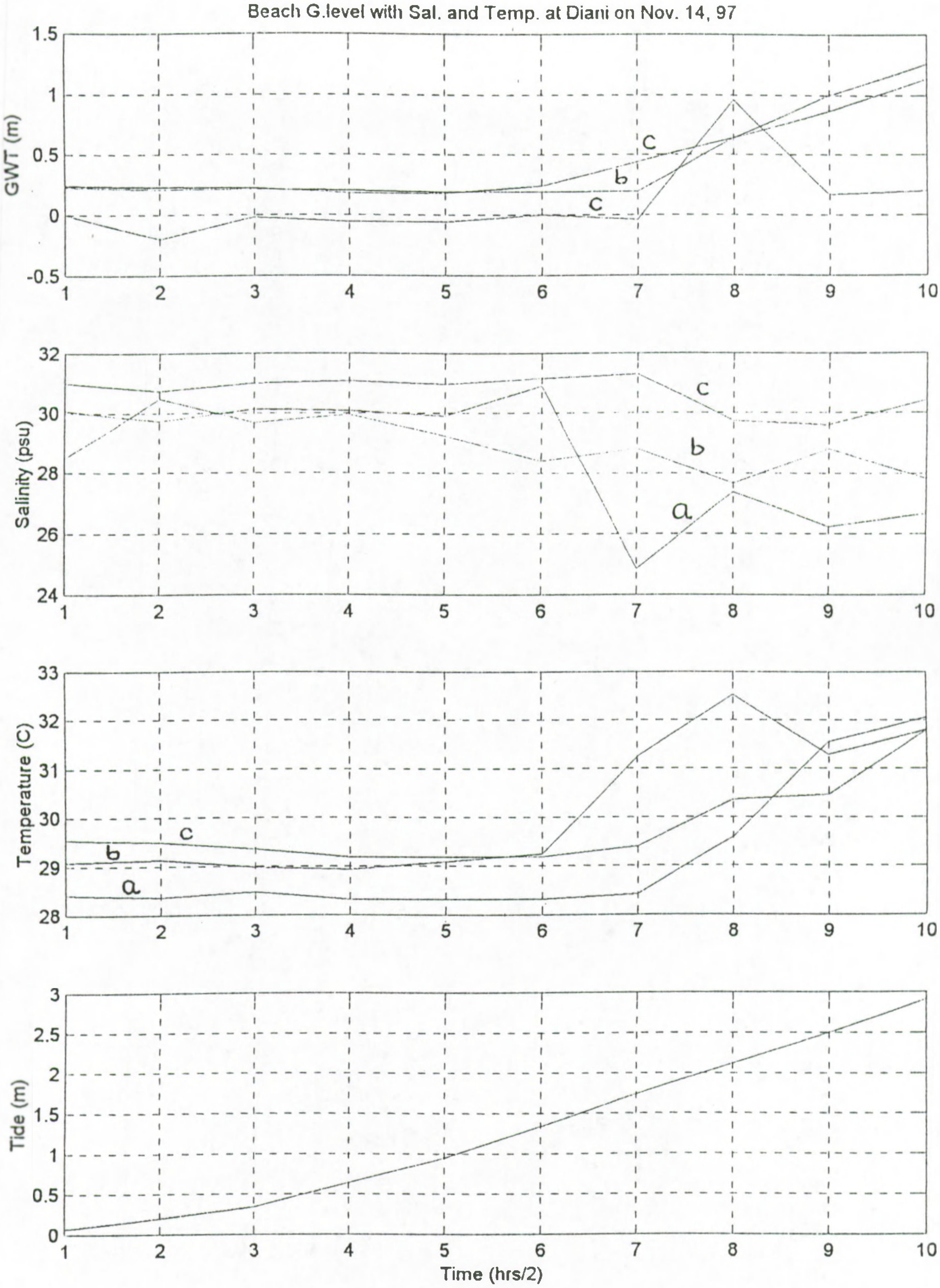


Figure 10: The relationship between beach groundwater table level and tide at Diani beach in rainy season. Note the relatively lower salinity which declines as the tide falls. The lowest salinity occurs at the base of the beach.

The results for Diani beach are also interesting. Typical rainy season results are presented for 14th November 1997 when lowest salinity recorded varied from 22.84 to 21.71 psu and groundwater discharge showed an increasing trend varying from 0.095 m^3/s to 0.10m^3/s. Considering typical dry season period, such as January 21, 1998, it was observed that groundwater discharge starts at a low rate of ≅ 0.01m^3/s when tide is 0.5 m high, and increases to 0.04 m^3/s as the tide falls further during low tide with elevation of 0.2m. There was a decrease in discharge reaching a minimum of 0.03m^3/s at the lowest tide (tide = 0.16m). As in Nyali beach, salinity was higher and showed a declining trend varying from 29.5psu to 28.4 psu (similar trend as in Nyali). The lowest salinity was recorded 1.5 to 2 hours after low tide, a period during which the tide had already begun rising. Temperature showed a declining trend at low tide (tide = 0.3m).

Attempt was made to establish the association between groundwater discharge, salinity and temperature. The relationship between groundwater outflow and temperature at Nyali beach was inverse ($r = -0.21$, $n = 42$, $p < 0.05$) while that between groundwater flow and salinity was very weak ($r = -0.01$). However in case of Diani Beach, the relationship between groundwater outflow and salinity was strongly inverse ($r = -0.60$, $n = 27$, $p < 0.05$). That between groundwater flow and temperature was positive but weaker ($r = 0.35$, $n = 27$). The variations in water temperature and salinity were closely correlated at Diani beach ($r = -0.88$).

Table 3. Descriptive statistics for groundwater outflow variables measured at Diani Beach using piezometers. This is based on measurements conducted on groundwater spring discharges at the base of the beach. Note the low salinity range.

Variable	Valid N	Mean	Minimum	Maximum	Std. Dev.
Tide (m)	27	0.378	0.00000	1.400	0.3362
Depth (m)	27	0.061	0.01500	0.095	0.228
Width (m)	27	1.282	0.39000	2.100	0.5034
Area (m^2)	27	0.084	0.00819	0.189	0.0555
Velocity (m/s)	27	0.510	0.32000	1.020	0.1450
Outflow(m^3/s)	27	0.044	0.00400	0.109	0.0316
Temp. (°C)	27	26.205	21.42000	29.540	3.2528
Cond(mS/cm)	19	34.568	28.50000	43.700	5.7960
Salinity (psu)	27	25.869	21.71000	29.440	2.6376
Sediment (g/l)	17	3.331	0.23000	10.163	2.9105

Table 4. Descriptive statistics for variables measured at Nyali Beach. Note relatively lower salinity as compared to Diani Beach. Note the high salinity range.

Variable	Valid N	Mean	Minimum	Maximum	Std. Dev.
Width (m)	42	0.710	0.2600	1.300	0.3097
Depth (m)	42	0.028	0.0100	0.070	0.0161
Area (m2)	42	0.022	0.0036	0.055	0.0164
Outflow (m^3/s)	42	0.011	0.0018	0.024	0.0070
Velocity (m/s)	42	0.460	0.0360	0.780	0.1884
Salinity (psu)	42	16.365	5.1600	26.570	5.9581
Temp. (°C)	42	30.015	26.2400	32.630	1.1956
Cond (mS/cm)	35	23.467	2.6300	39.900	9.9816
Tide (m)	42	0.504	0.0000	1.910	0.4760
Sediments (g/l)	14	10.880	1.0331	22.780	7.9164

Beach groundwater table dynamics

Tables 5 to 6 presents maximum and minimum groundwater level, salinity and temperature measured in piezometers located in the lower, middle and upper margins of the beach at Nyali and

Diani Beaches. The relationship between groundwater level change at the base of the Nyali beach and tide was quite weak and inverse as confirmed by low correlation ($r = -0.029$, $n = 70$, $p < 0.05$). This weak relationship was also confirmed by ANOVA test. However the relationship between groundwater table variations and tide at the middle section of the beach was inverse and moderately strong ($r = -0.41$, $n = 65$, $p < 0.05$). In the upper zones of the beach, groundwater variations are poorly correlated to the tide ($r = -0.11$, $n = 51$ $p < 0.05$). However groundwater level variations at the base of the beach are closely related to variations at the middle zone of the beach ($r = 0.81$). Surprisingly there is also a strong relationship ($r = 0.88$) between groundwater level variations at the base of the beach and that at the upper zone implying movement of water through the beach matrix.

Table 5. The descriptive statistics for Nyali beach based on experiments conducted using three piezometer. A: Piezometer is located at the lower beach margins, B: Piezometer is located at the middle beach section and C: Piezometer is located at the upper margins. The distance between piezometer is 10m. GWRT refers to groundwater fall or rise time (m/hr) while TIDERISE refers to rise or fall of the tide. Note that groundwater rise time is different from tiderise time.

Variable	Valid N	Mean	Minimum	Maximum	Std. Dev.
GWTA (m)	70	0.303	0.0200	1.300	0.2697
GWRTA (m/hr)	69	0.085	-0.3000	1.140	0.2736
TEMPA (°C)	71	29.012	25.8800	32.940	1.5528
SALA (psu)	71	24.938	13.7400	32.060	4.9489
GWTB (m)	65	0.281	-0.0200	1.400	0.2677
GWRTB (m/hr)	65	0.090	0.1200	1.140	0.2615
TEMPB (°C)	65	28.663	26.2400	33.150	1.4922
SALB (psu)	61	28.666	21.3000	31.410	2.5109
GWTC (C)	51	0.147	0.0020	0.830	0.2124
GWRTC (m/hr)	51	0.065	-0.1300	1.420	0.2712
TEMPC (°C)	43	28.446	26.2400	31.750	1.1806
SALC (psu)	36	27.856	23.7500	30.720	1.6790
TIDE (m)	76	1.141	0.0000	4.000	1.0613
TIDERISE (m/hr)	51	0.383	-0.7000	2.000	0.5121

Table 6. The descriptive statistics for Diani beach. The location of piezometers is as in Nyali. Symbols used are described in Table 5 above.

Variable	Valid N	Mean	Minimum	Maximum	Std. Dev.
GWTA (m)	29	0.342	0.140	1.260	0.3085
GWRTA (m/hr)	29	0.097	-0.100	1.160	0.2598
TEMPA (°C)	29	29.370	27.050	32.530	1.4048
SALA (psu)	29	28.972	24.030	30.980	1.8639
GWTB (m)	33	0.282	0.070	1.200	0.2852
GWRTB (m/hr)	32	0.070	-0.150	0.890	0.2364
TEMPB (°C)	33	28.635	26.490	31.800	1.2543
SALB (psu)	33	29.938	26.230	32.010	1.4824
GWTC (C)	33	0.106	-0.200	0.960	0.2109
GWRTC (m/hr)	32	0.056	-0.200	1.320	0.2955
TEMPC (°C)	24	28.930	28.320	32.060	0.9742
SALC (psu)	24	30.406	27.700	31.450	0.7799
TIDE (m)	33	0.896	0.070	2.910	0.8160
TIDERISE (m/hr)	19	614	0.060	0.840	0.2547

The variation of groundwater level at the upper and base of the beach were moderately related ($r = 0.57$) at Diani while that between the middle and lower zones as in Nyali beach, was very strong ($r = 0.97$ at 95% confidence level). This is similar to what was observed at Nyali beach.

Attempt was made to establish the relationship between beach groundwater level change and changes in water temperature and salinity. The influence of tide on groundwater salinity variations at the base of the Diani beach was found to be inverse and moderately strong ($r = -0.50$). Variations of groundwater salinity within the upper zones of the beach were weakly and inversely related to the tide ($r = -0.14$), but in the middle zone, the relationship is inverse and slightly stronger ($r = -0.36$). In all above cases, ANOVA tests confirmed significant relationships at $p < 0.05$. The relationship between groundwater level variations at the base of Nyali beach and temperature was positive ($r = 0.31$), while at the middle zone it is lower ($r = 0.24$). These relationships are shown in figures 9 to 10. Figure 11 shows the scatterplot with histograms of the relationship between tide and groundwater level at different positions of the beach.

Different zones of the Diani beach responds differently to tidal forcing. There is a 30 minutes phase lag between levels at different positions. Main variations in groundwater salinity occur within the lower margins of the beach possibly due to poor mixing within the sediment matrix. Of importance to note is the fact that groundwater salinity is low even when flood tide (tide <1.3m) engulf the beach. Groundwater temperature change is usually constant at low waters (between tide of 0.0m and 1.3m), but rapid increases with delayed responses occur as flood tide elevation is ≥1.3m. The lowest groundwater salinities were recorded during this time. There are minor variations in groundwater salinity at the middle and upper zones but variations at the base of the beach were characterized by a declining trend with the lowest salinity being recorded 1 to 2 hrs after lowest water level in the lagoon. Rapid groundwater level changes occurred when tide was 2.0m high. The lowest salinity was also recorded at the lower margins of the beach and it progressively increased up the beach. There was also a difference in the variations of groundwater temperature with the highest being recorded at the base of the beach.

Beach run-up and seasonal variations

There are seasonal variations in the levels of groundwater springs salinity and temperature at both Diani and Nyali Beaches. The lowest and highest salinity at Diani is 21.71 psu and 29.4 psu respectively. At Nyali Beach, the lowest and highest salinity is 5.2 psu and 26.6 psu respectively. The lowest salinities were measured in November just after heavy rains, while the highest were measured in February-March periods in dry seasons. The lowest temperature was recorded in July and highest in March.

Measurements conducted on lagoon water salinity and temperature variations also showed there are seasonal variations as is shown in Figures 12 and 13. Salinity variations are however less dramatic inside the lagoon, but there was a rapid decline along the shore in November 1997. There is an increase in water temperature from the minima in June (24.0°C) to maximum (32°C) in November 1997. The lowest salinity at Diani was recorded in November while at Nyali, it occurred in December. The difference could be due to differences in groundwater transmission rates. In both Nyali and Diani beaches, a narrow zone along to the shore experience lowest salinity as compared to zones located in the coral reef and central region of the lagoon. The salinity differences between these three zones were more distinct in rainy season as compared to dry season. In the latter periods minimal variations were recorded. Similarly, the zones located along the shore (closer to the beach) experienced relatively higher water temperature as compared to those located in the central and coral reef complex region of the lagoon. Major differences in lagoon water salinity are noted in rainy season (November-December 1997), and minor differences occurred in dry seasons (March-February). These patterns are closely related to the variability in the fluxes of groundwater, the effect of evaporation and short-wave radiation influx. The water salinity starts declining in July and reach minimum in December. After December, there is a salinity increase, which is then followed by the salinity minimum in May. Water temperature starts increasing June and reach peak in March.

On September 17th and 18th, 1997, surveys on the salinity along the entire stretch of both beaches was carried out. It was established, within the entire stretch of Diani beach, there is seepage of groundwater but rates of discharge are spatially non-uniform. The zone with the highest discharge is located in the southern parts adjacent Robinson Hotel and highest is in the northern sections towards Forty Thieves hotel. There is a progressive decline in seepage as one moves northward. The saturated seepage band is greater at the southern zones ranging between 30 to 40m while in the northern section it is narrow, rarely exceeding 15m. In the southern zone, groundwater springs have also relatively lower salinity ranging from 27.20 to 29.95 psu while in the northern region salinity ranged from 31.32 to 34.94 psu. While in the lower southern zones the spring discharges are predominantly point-based, but there is also diffuse discharge in some places. At Nyali beach, greatest outflow occur next to Nyali Beach Hotel and around Ras Mkuu Ng'ombe.

Groundwater Seepage occurs in three dominant modes. These are (1) diffuse seepage, (2) point-based seepage and (3) a combination of the two modes mentioned above. Diffuse seepage occurs in the saturated zone in the lower portions of the beach and presents mirror-like appearance with water oozing gently from the sediments. This water may collect into small rivulets and flow to the lagoon. Point-based seepage occurs in holes and opening in the coral limestone base. Water exudes at high velocity and presents boiling appearance. Large volume discharged quickly form channels whose width range between 1.3 and 0.5m. These channels cut through the carbonate sediments. Sediments within these holes are usually very coarse since fine particles are removed by high velocity of flow. The diameter of the holes range from a few centimeters to a couple of meters. Groundwater seepage in the Central Diani zone is mainly a combination of diffuse and point-based flow while in the south, point-based seepage is dominant. At Nyali, most groundwater seepage occurs as a combination of the modes described above. In some locations within the Diani lagoon, point-based seepage occurs right into the lagoon as is illustrated by a 2m-diameter seepage hole situated next to Robinson hotel <100m from the beach.

Tidal forcing, groundwater variations and coastal processes

Tidal forcing partly drives groundwater level changes in areas located < 1000m from the ocean. The tide has a range of 3.2m but there are variations in spring and neap tides. There is however significant filtering of the tidal energy so that tidal amplitudes are significantly reduced in the groundwater system inland. Thus, groundwater level depicts changes, which is closely related to tidal forcing. Groundwater level variations are seldom > 0.20m compared to tidal variations, which are often in the order of 2.0m. There is no asymmetry in groundwater level variations (Figure 14). These variations are synonymous with high variability in the flux of groundwater from nearby coastal aquifer. During spring tides, the groundwater levels reached a maximum of between 0.14 and 0.16m while in neap tide the levels were hardly more than 0.09m. These levels have important implications on the exchange of groundwater and ocean water. In spring tide, there is an enormous flow of seawater (during flood tide) into the aquifer adjacent the beach. At the same time, there is also increased discharge of brackish water from the aquifer (in ebb tide) into the sea. In neap tide, a relatively lower volume of seawater flows into the aquifer during flood tide and the resultant low hydraulic gradients may limit the flow of brackish water into the sea at low tide. The variations in groundwater level do not occur at the same pace as the changes in semi-diurnal tide. There is a phase lag (30–50 minutes) between the two.

Discussion

Computations on the rates of groundwater discharge at both Nyali and Diani beaches were attempted using data derived on the spatial extent of saturated zones as well as on the average seepage velocity. The Diani beach covers a distance of 7.4 Km if one considers the beach stretch between the headland and Trade winds hotel to the North (Figure 1b). The total beach area calculated using a mean beach width of 0.83km is roughly 6.14km^2. Groundwater discharges in a band with an average width of 20m, thus, the total seepage area is 148,600m^2. With mean depth of 0.06m and mean seepage velocity of 0.30m/s, this yields groundwater flux of 133m^3/s. In case of Nyali Beach, if the area between Ras Mkuu Ng'ombe headland upto the Leave Camp to the North is considered, the beach covers a distance of 3.1 Km with a mean width of 1.8km. Thus the total beach area is roughly 5.7km^2. Groundwater discharges in a 3,100m by 20m area thus seepage band can safely be put at 62,000m^2 This is relatively a smaller than that considered at Diani beach. The seepage band at Nyali beach yields groundwater discharge of 43m^3/s, which is lower than that at Diani Beach. The difference is merely due to differences in the spatial extent of the two areas. From experiments conducted, it was noted that groundwater outflow occurring during low tide

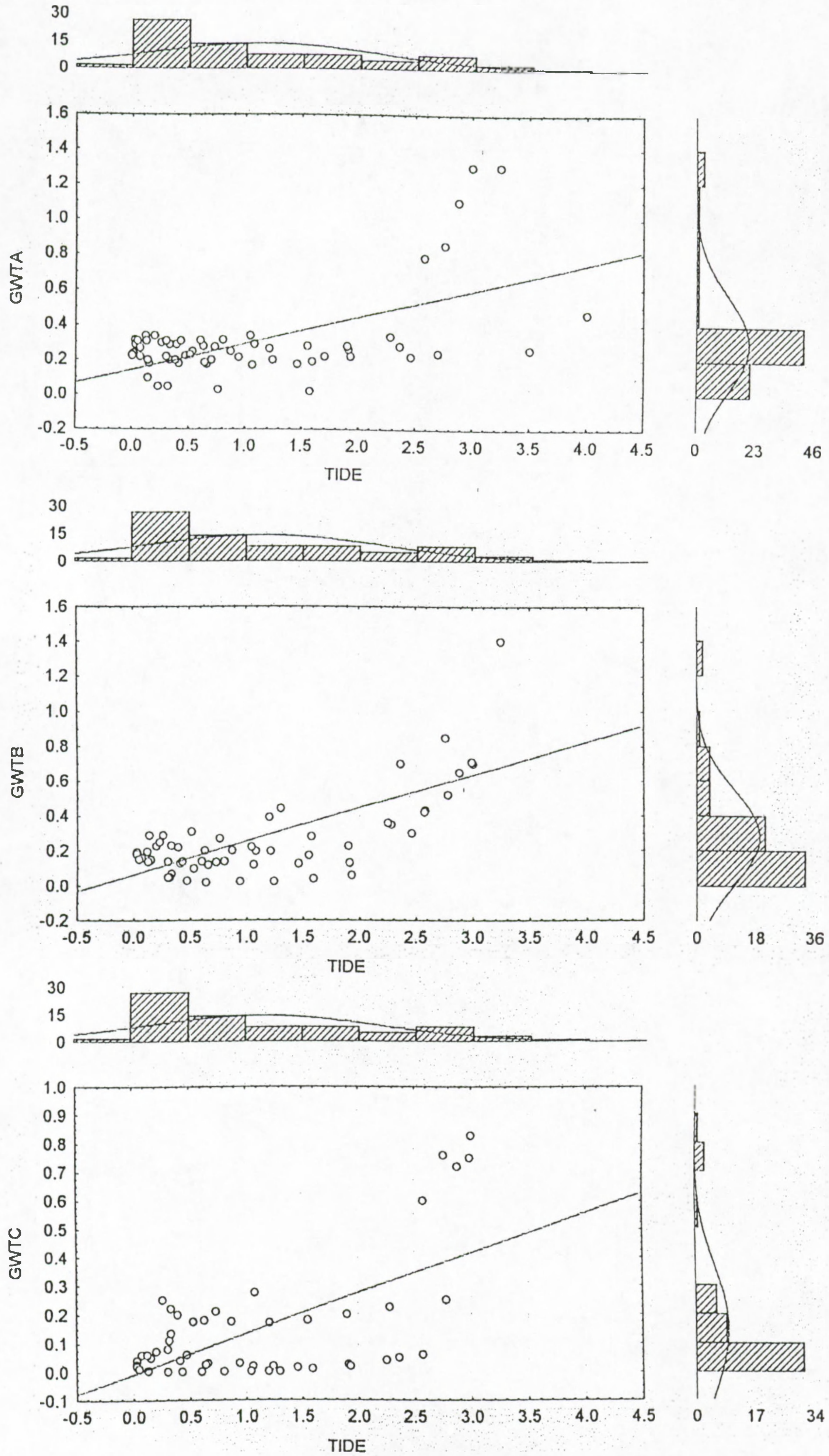


Figure 11: The scatterplots with histograms showing the relationship between groundwater level and tide at Nyali beach. Note the scatter of the points and linear relationship. The GWTA, GWTB and GWTC refer to groundwater level at the base, middle and upper zones of the beach respectively.

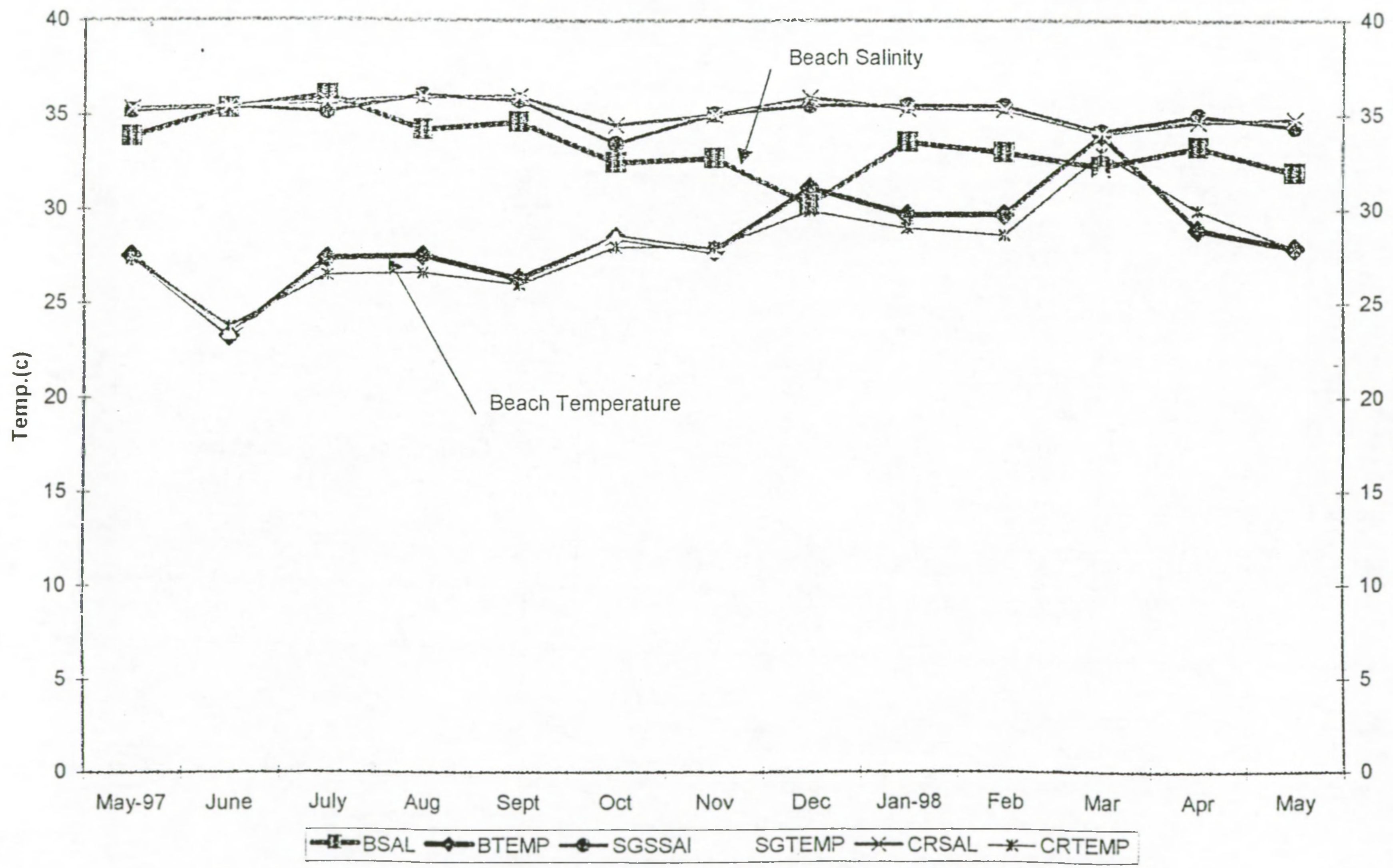


Figure 12: The seasonal variations of water temperature and salinity at Nyali lagoon. There was an increase in water temperature from June with the maximum in March. Salinity shows a decline from July and minimum occurs in November-December period after the El Nino Southern Oscillation related rains. The zone along the shore experience low water salinity throughout the year.

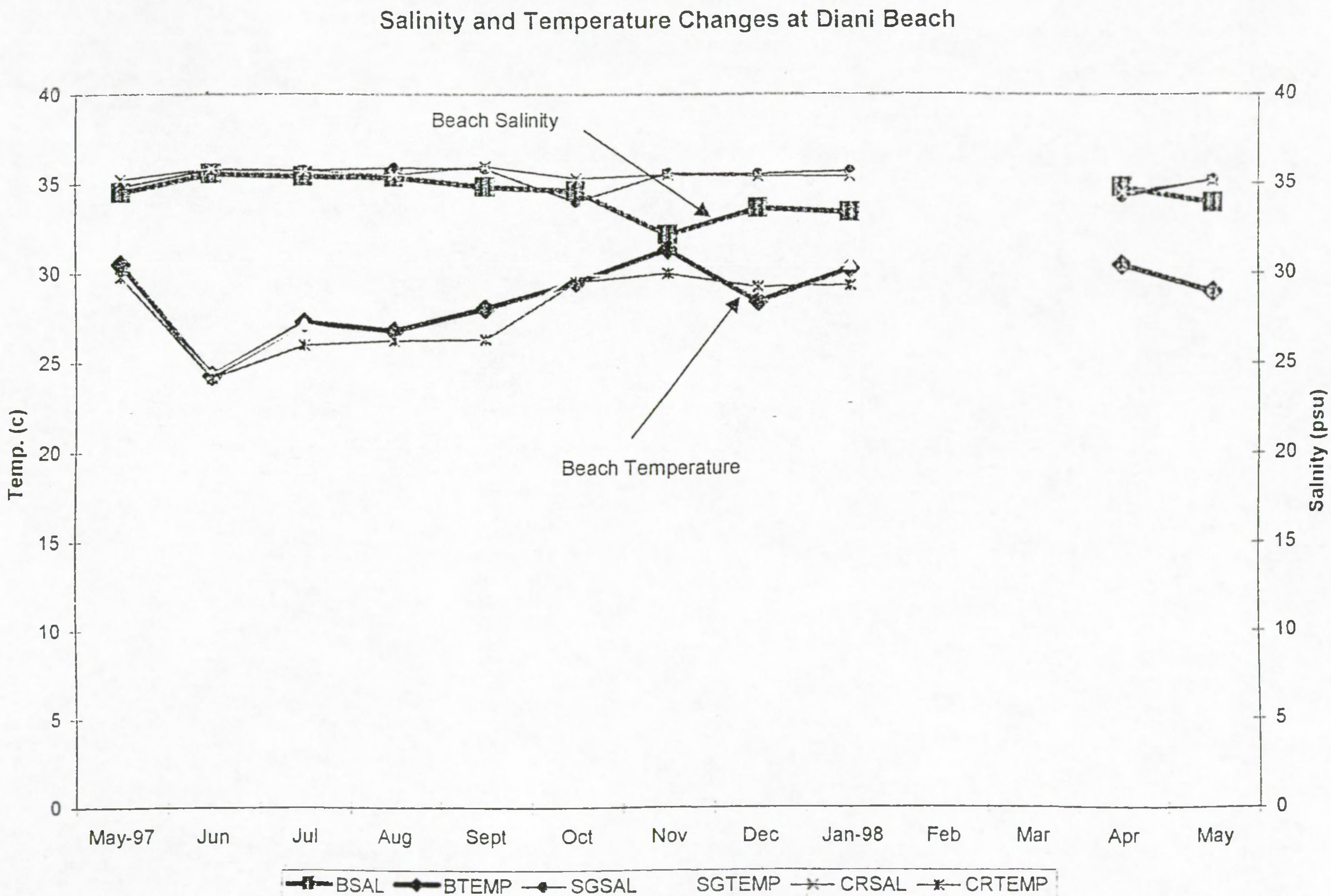
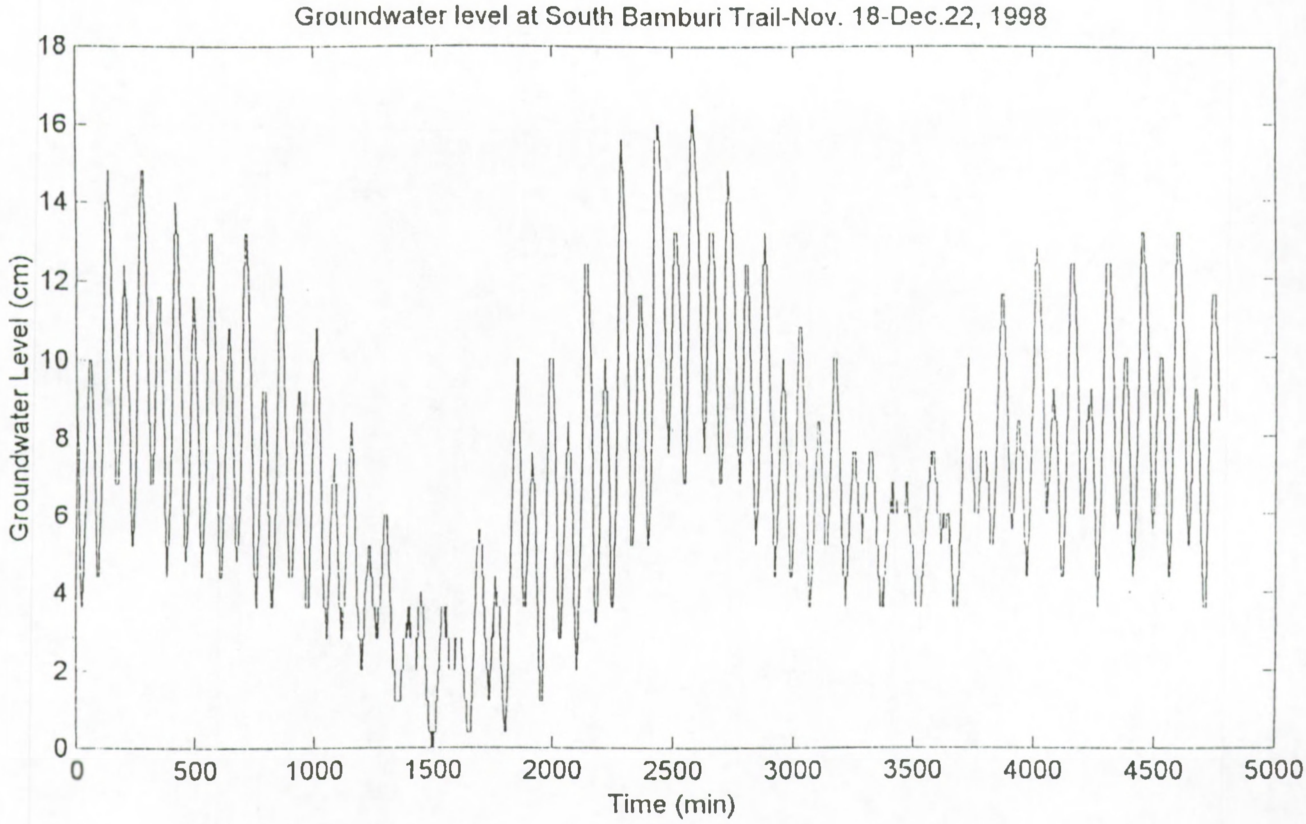


Figure 13: The seasonal variations of water temperature and salinity at Diani lagoon. As at Nyali lagoon a zone along the shore experience low water salinity throughout the year. Lowest salinity occurred in November-December period.



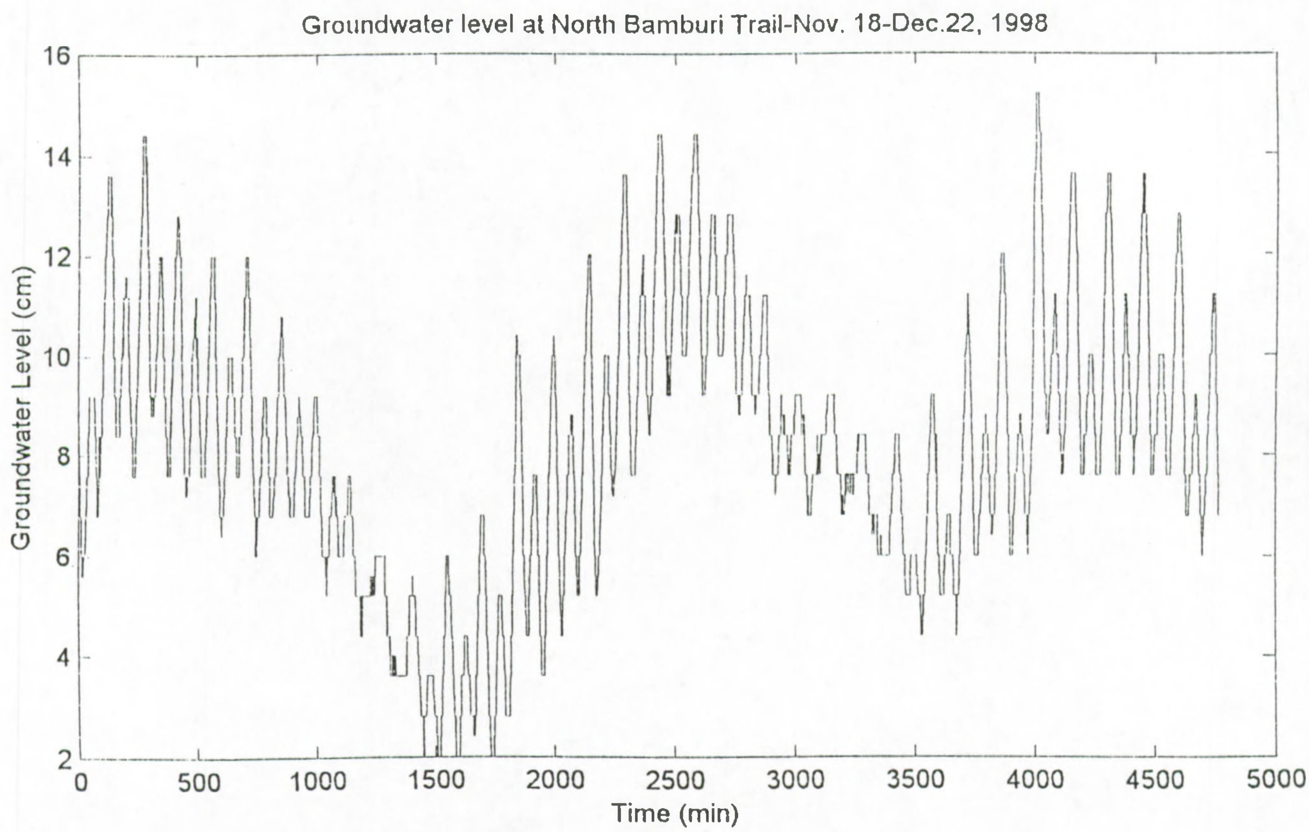


Figure 14: The variations of groundwater level at Bamburi nature trail in the period between 18th November-22nd December 1998 period. Below is the pattern of semi-diurnal tide measured in the period between November and December (not starting on November 18th). There is considerable filtering of the tidal wave in the groundwater aquifer so that only minor variations in groundwater are detected

continues for a period of 3.5 to 6hrs- a period when beach is usually exposed (Figures 3 to 10). If we take the maximum duration of 6 hours, the total groundwater discharge into the lagoons on daily basis can be estimated. This at Nyali beach is put at 93 x 10^4 m^3 per a ½ day. Since the beach is usually exposed twice a day as a result of semi-diurnal tides, the total groundwater outflow at Nyali Beach is 186 x $10^4 m^3$/day while that at Diani is 575 x 10^4 m^3/day. The mean lagoon volume at Nyali is 6.5 x $10^6 m^3$ while that at Diani is 4.9 x 10^6 m^3 which when compared to groundwater volume it is possible to establish ratios. Groundwater contributes between 2 and 10% of the total volume of the Nyali and Diani lagoons respectively. There is therefore a substantial portion of groundwater in these two coastal lagoons and it is expected that this will influence salinity and nutrient distribution. This could however be an over-estimation since there are certain locations along the beach where there is no significant flow, and in some zones, the seepage velocity is too low as compared to the mean we used in this study. But could also be within the actual volume flux since substantial groundwater outflow also occurs in form of less obvious point-based submarine discharge within the lagoons and this contribution was not taken into account in our calculations.

Tidal forcing essentially controls the exchange of groundwater between the coral limestone aquifer and the ocean, but depending on the characteristics of beach sediments and hydraulic pressure gradient created by groundwater recharge inland, groundwater outcrops at the lower margins of the beach as illustrated in figure 2. This is basically related to the fact that groundwater does not flow directly through the beach sediment matrix, but is ejected almost vertically from below in a zone of interaction between saline seawater and brackish groundwater. As it has been reported by Emery & Foster (1948), Nielsen, (1990), Duncan, (1964), Cooper (1959) this occurs as a result of hydraulic interaction between the intruding seawater and groundwater flowing from inland. This observation explains the salinity gradients noted in the piezometers. In piezometers located at the lower margins of the beach, the salinity was often much lower as compared to that in the middle and upper sections. However, as the tide rises, the low salinity groundwater is pushed through the beach matrix into the middle and upper zones of the beach so that relatively lower salinity is recorded as the flood tide commences. During low tide, this water is gradually released and at the later stages of the ebb tide, fresh groundwater starts discharging with increased vigor from below at the base of the saturated zone. However, these fluxes are depended on the beach sediment characteristics such as porosity and permeability. Complications arise when flow takes place in limestone caverns.

As a result of nigh porosity and permeability of coral limestone rock, some tidal constituents still manages to pass through the beach matrix so that tidal signals are detected in inland groundwater aquifer located mostly within 2Km from the beach (Figure 14). However since tidal amplitudes are reduced by filtering action of the rock matrix (Turner et.al., 1997; Hegge & Masselink, 1991), ground water level variations due to tides are <0.20m.

Groundwater discharge at the base of the beach starts at a relatively low rate as the tide falls since low hydraulic gradient reduces outflow. Effective discharge occurs once seawater has been drained from the beach and adjacent aquifer during ebb tide and favorable hydraulic gradient therefore exists. This is followed by rapid discharge of brackish groundwater until the backlog of high salinity water pumped into aquifer and beach by flood tide is drained after which groundwater of much lower salinity starts flowing from the aquifer.

Beach groundwater salinity is a good indicator of the degree of groundwater recharge and seawater intrusion in coastal aquifers. After heavy rainfall, coastal aquifers are recharged so that increased groundwater volume inland prevents seawater intrusion and dilution lowers salinity. In dry seasons, lack of rainfall recharge and abstraction reduces groundwater volume in storage and this increases intrusion of seawater into the aquifer and therefore salinity increases. Thus when favorable hydraulic gradient exists as tide falls, relatively high and low salinity groundwater is discharged from the aquifer in dry and wet seasons respectively. Patterns confirming these observations were evident at both Nyali and Diani Beaches. Relatively lower groundwater salinity at Nyali than at Diani is an indication of relatively low seawater intrusion in the former than in the later. The mean beach groundwater salinities are 16.4 PSU and 25.9 PSU at Nyali and Diani beaches respectively. Rapid change in groundwater salinity and discharge with rainfall as in November 1997 is an indication that groundwater recharge is controlled by local replenishment from rainfall falling on the permeable rocks and soils within the coast belt.

There is no complete mixing of water discharged from the groundwater aquifer into the lagoons as is exemplified by clear spatial differences between zones located closer to the beach and those located in the middle and the coral reef zones (Figures 12&13). Existence of large storage capacity of the coastal aquifers and relatively good recharge makes groundwater to flow throughout the year, albeit at different rates. Zones closer to the beach (along the shore) thus experience relatively lower salinity throughout the year. This cannot be explained by rainfall or surface runoff influx since it was even noted during drought periods. Pleistocene sediments in the region prevent generation of surface runoff since they promote rapid infiltration and absence of river drainage precludes any possibility of supply from river systems. The existence of low salinity water even in dry season is a clear evidence of the influence of groundwater in moderating salinity. The low salinity zone is 5 to 25 m wide and runs parallel along the beach. This zone is more prominent at low tide since large waves capable of mixing different water types are dissipated in the coral reef zone and do not enter the lagoon. At high tide, the reef is submerged and progressive waves enter the lagoon and effectively mix the water column so that the low salinity band disappears. But this band re-appears during low waters. At Diani lagoon wave heights are seldom more than 1.2m high and wave speed varies from 3.3 to 4.6 m/s. At high tide, wave energy is high varying from 1000 to 1800 Joules/m^2 with wave momentum in the order of 246 to 381 J/m/s and therefore waves have high mixing capability. At low tides, waves are smaller with low energy ranging between 13 and 314 Joules/m^2 with a mean of 146 Joules/m^2.

Variations in lagoon water temperature are not related to influx of groundwater but to the influx of short-wave radiation. Water is effectively heated in shallow zones along the coral reef complex and along the beach, so that highest water temperature is observed in shallow zones along the shore. Were it not for moderating effect of groundwater outflow, salinity along the shore would rise to near hyper saline conditions. The main factors sustaining the low salinity band along the coast seems to be onshore wind (both NEM, SEM, SWM winds) which generate longshore currents trapping water along the beach. Wave breaking at the coral reef complex also generates currents, which prevents low salinity water from reaching the reef complex at high tide. Residence time of water is seldom more than 6hrs and with currents at low tide being <0.35m/s, there is less effective mixing of the water column during both low and high tides (see also Nielsen, 1988). Wave breaking is important in mixing processes. These hydrodynamic factors have also being dealt with by Kitheka (1997) in case of Gazi bay lagoon in Southern Kenya.

Conclusions

Groundwater outflow in coastal zones and its impact on nearshore ecosystems remains an amorphous subject. In this study an attempt was made to understand the patterns of groundwater outflow in Kenya's carbonate beaches. The most important conclusion is that groundwater flows along carbonate beaches of Kenya, however, the rates of outflow are spatially non-uniform being influenced by the underlying geology and morphology as well as rainfall recharge. Groundwater outflow occurs at the base of saturated zone of the beach at low tide; the outflow is either diffuse or point-based, and in some areas, a combination of the two modes of seepage exists. Beach sediment matrix acts as a filter filtering tidal wave energy so that there is a delayed change in groundwater level inland as the tide rises and falls. While the lowest salinity occurs at the base of the beach there is an increase up the beach during low tide, but during high tide, the salinity progressively increases as groundwater flowing at the base of the beach is pushed upward into the beach matrix. Due to relatively poor mixing at low tide, a band of relatively low salinity water occurs along the shore throughout the year. This moderates the lagoon water salinity so that hypersalinity is never reached. Circulation characterized by low current speed <0.35m/s in low tide, and onshore wind which generates longshore current helps in trapping low salinity water along the coast. But intensive mixing occurs at high tide when large waves penetrate through the submerged coral reef complex and enter the lagoon to break at the beach. During this period, low salinity band disappears.

Tidal wave is filtered by coral limestone rocks so that tidal forcing on groundwater level is non-asymmetric and <0.20m <2Km inland. Further inland possibly a distance >3 Km, the tidal frequencies may be completely filtered out. From estimation of groundwater volume fluxes, it was established that groundwater volume is roughly equivalent to 2 and 10% of the total lagoon volume. By any other comparisons, this shows that there is enormous supply of groundwater to coastal beaches and lagoons. Groundwater is therefore an important component of the coastal nearshore environment. However in urban areas, rapidly increasing demand for freshwater is necessitating increased exploitation the effect of which will be increased seawater intrusion and reduced outflow into the ocean exploitation increases above safe-yield value. This will not only interfere with the present patterns of water-use but it will also impose negative repercussions since increased salinity will preclude the use of groundwater in agricultural, touristic, domestic and industrial sectors. Reduced discharge into the lagoons will reduce the salinity moderation and nutrient boosting functions with negative impacts on fisheries, recreation and biodiversity conservation. There is need for a comprehensive programme on groundwater management in the coastal zone of Kenya, perhaps, in the framework of integrated coastal zone management.

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Groundwater-associated anthropogenic influence on coastal lagoons: Diani and Nyali Beach, Kenya

B. M. Mwashote, S. N. Mwangi & J. M. Kazungu
Kenya Marine and Fisheries Research Institute, Mombasa, Kenya

Introduction

Everywhere along the coast a dynamic balance exists between the seaward outflow of groundwater and salt water intrusion into the coastal freshwater aquifers. The quality of groundwater which is a resource of enormous importance, is dependent upon numerous factors but most importantly has a direct bearing on the anthropogenic and economic development of the area.

On the other hand, the undisturbed flow of groundwater may be a prerequisite for the persistence of species and even entire ecosystems in the coastal zone. For instance this is found to be the case with many mangrove species and some seagrasses.

The constant withdrawal of groundwater along the E. African coastal zone occurs in many places to supply an increasing number of beach hotels and settlements of the local population, subsequently resulting to decrease in the coastal groundwater levels. It has been documented that where groundwater withdrawal is heavy and concentrated, such that it greatly exceeds local recharge, water levels may continue to decline over many years and the area affected spreads out producing major changes in head distribution within the aquifer system and significant reversal of groundwater flow paths (BGS/ODA/UNEP/WHO, 1996).

It is also likely that these waters exhibit elevated nutrient and pollutant concentrations. This is mainly attributed to poor conservation and agricultural practices as well as inappropriate disposal of effluent into pits or sink holes as they offer cheap means of sewage disposal. Indeed the majority of decline in ground water levels reported elsewhere (Wu *et al.* 1993, Krishnasamy, 1997; Foster, 1976; NEPA, 1992) has been accompanied by a deterioration in groundwater quality and /or subsidence.

As a result of this situation there is not only contamination of groundwater but the effect of this in its potability through elevation of nutrient levels and anthropogenic toxins, and this effect may also be transmitted to the shallow coastal waters which receive groundwater outflow. Little information however exists concerning the interactions between groundwater and seawater in the coastal zone within the E.African region.

Recent demographic changes have resulted in a lot of pressure being exerted on coastal resources. One of the consequences of these changes for the marine ecosystem has been the constant input of anthropogenic materials either directly, mainly through discharges, river and surface runoffs or indirectly, through groundwater. An important health concern related to human use of groundwater and shallow coastal lagoons arises from the potential presence of human pathogens in the waters. Potential sources of pathogenic microorganisms for the marine environment include untreated or poorly treated municipal waste and industrial effluents, sanitary wastes from waterfront residences, swimmers and stormwater runoff. Microbiological agents from these sources include pathogenic bacteria, viruses, protozoa and several more complex multi-cellular organisms which can cause gastrointestinal diseases (Geldreich, 1989) such as infectious hepatitis, dysentery, vibrio infections and cholera. Those aspects of the aquatic environment may affect water quality and exert severe influence upon human health, in terms of severity of effects as well as in terms of numbers of people affected.

With the massive increase in population and coastal tourism, the actual and perceived risk of contracting disease from swimming in polluted seawater has been equally augmented during the past decade. Although a variety of infections and diseases can be contracted from swimming in polluted seawater, it is predominantly the gastrointestinal disorder which result from sewage contamination of coastal bathing waters. The assumption that discharges of animal and human faecal waste into water bodies used primarily for recreational purposes are a source of potential public health hazards has developed historically into the search for a microbial indicator capable of characterizing the risk involved in the recreational use of the water. Whilst it is possible to examine water for the presence of a specific pathogen, a more sensitive test employs the use of indicator organisms. *Escherichia coli,* which is a normal inhabitant of the human intestine and is excreted in large numbers, is one of such indicators. Its presence in water thus indicates human excretal contamination and the sample is therefore potentially dangerous in that pathogenic faecal bacteria might also be present.

Faecal coliforms and particularly *E. coli* are considered to be practical indicators of the degree of sewage pollution, and subsequently proposed for routine monitoring of coastal water quality. For public health protection however, faecal streptococci is an additional water quality indicator (USEPA, 1979). The faecal coliform/faecal streptococci (FC/FS) ratio has been used to evaluate the origin of faecal pollution; for instance domestic sewage versus industrial sewage (Martens *et al.*, 1984).

The study areas, Nyali and Diani, are characterized by a number of beach hotels and high population in the nearby residential estates. There is high demand and consumption of freshwater by tourists and the local population with a consequence of generation of high volumes of wastewater and sewage, which are potential sources of pollution to the marine environment. The preferred method of sewage disposal in these areas is the use of pit latrines and septic tank - soakage pit systems. Domestic waste and sewage from the areas are likely to contaminate groundwater and thus the lagoonal waters adjacent to the areas. It has been shown that *Escherichia coli*, whose presence is usually used to indicate bacterial pollution from sewage, thrives in nitrogen rich environments, such as sewage contaminated seawater (Lim & Flint, 1989). Cases of outbreaks of acute gastroenteritis associated with swimming in sewage contaminated marine waters are not uncommon. For example this was reported from New York State in 1982 and 1983 (Cabelli, 1986). In this study therefore, bacterial indicators were used to assess the level of contamination ground and lagoonal waters in the study sites.

It is with the preceding background knowledge in mind that the present work was initiated with the aim of studying the impacts of anthropogenic influences on the groundwater quality and the effect of groundwater outflow in the Kenyan nearshore coastal waters and the adjacent lagoons.

Objectives

The general aim of the study was to elucidate the importance of groundwater as a vector of anthropogenic inputs into the coastal zone. In order to achieve this, research work was planned and approached in two fronts: (a) nutrient dynamics studies and (b) microbiological water quality studies.

The study was conducted in two main areas viz. Nyali and Diani. Although it had been expected that one of these areas would be a groundwater outflow area and the other an intrusion area, they both turned out to be outflow areas. A third station, Kenyatta beach had to be included to serve as an intrusion area for comparative purposes. The specific objectives of the study were to:

1) Determine the dissolved inorganic nutrient levels ($NO_2^- + NO_3^-$, NH_4^+, PO_4^{3-} and $Si(OH)_4$) of the nearshore coastal waters and sediment where groundwater outflow occurs,
2) Determine the dissolved inorganic nutrient levels in the adjacent wells and boreholes where groundwater is being utilized,
3) Determine the level of bacterial contamination in ground and lagoonal waters in the study areas,
4) Determine the influence of groundwater quality on lagoonal water quality,
5) Determine the spatial and temporal variability in the level of microbiological contamination in the study areas,
6) Determine the potability of borehole/well water in the study area,
7) Use the above assessments to determine the state of groundwater quality and the effect of this on the nearshore coastal ecosystems.

Materials and methods

Sampling

Regular sampling started since May 1997. Sampling was done at least once a month at all sampling stations at spring low tides. In particular, for microbiological samples, sterile techniques were used in sample collection and all the samples were kept cool before analysis commenced within 12 hours. At both the two main sampling stations (Nyali and Diani), three transects were set (A, B and C) perpendicular to the shoreline each of which had four transect stations (N1, N3, N4 and N5 for Nyali and D1, D3, D4 and D5 for Diani). For the case of the comparative station, Kenyatta beach, only one transect (with transect stations K1, K3, K4, K5) was selected. In all cases while transect stations 3, 4 and 5 were completely within the lagoonal waters, transect station 1 was located on the upper part of the beach. For the case of Diani and Nyali transect stations (DB1, DC1, NB1 and NC1 were stations which clearly had groundwater outflow. Fig. 1 on page 6 of this report gives an idea of where the indicated sampling stations were located along the coastline.

Dissolved nutrients analysis

The concentration of dissolved inorganic nutrients viz. nitrite + nitrate [(NO_2^-+NO_3^-)-N], ammonium (NH_4^+-N), phosphate (PO_4^{3-}-P) and silicate $Si(OH)_4$ -Si, and other associated measurements were determined according to methods described by Parsons *et al.* (1984) and APHA (1995).

Microbiological analysis

The classical MPN method was used: 5-tube, 3-dilution techniques (FAO, 1979; UNEP/WHO/IAEA, 1985a,b). Sterilized glass bottles were used to collect water samples. In the marine environment, sub-surface (30cm below surface) water samples were collected at the sampling sites in the lagoons. On the beach, groundwater was collected as it flowed from the ground. For wells/boreholes, untreated water samples were collected directly from taps. All samples were kept cool in an icebox before analysis within 12 hours of sampling.

As indicators of the microbial contamination levels, faecal coliforms, *E. coli* and faecal *Streptococci* were enumerated in water samples by the use of the 5-tube 3-dilution techniques.

For faecal coliforms, positive coliform tubes were used to inoculate fresh tubes of MacConkey broth and incubated at 44-45°C to give faecal coliform counts. *E. coli* was biochemically determined by indole production while faecal *Streptococci* were enumerated by use of azide dextrose broth and confirmed with ethyl violet broth.

Statistical analysis

Spatial and temporal variations in dissolved nutrients were carried out using Analysis of Variance (two-way ANOVA). All statistical analyses were based on the significance level at $p = 0.05$ and critical values of F at $\alpha = 0.05$ (Yule and Kendall, 1993).

Results and discussion

Nutrient dynamics

The nutrient distribution in the water column for Diani, Nyali and Kenyatta beach sampling stations is depicted in Figures 2a-5b and 10-12.

The general trend observed in the transect stations for Diani and Nyali are such that the A1, B1, and C1 stations show higher nutrient concentrations compared to the rest of the transect stations with B1's and C1's depicting the highest values (though statistically there was no within stations difference, $p>0.05$). This is a strong indication that the main source of nutrients in the area within stations is coming in through groundwater outflow which is predominantly experienced in these stations. Groundwater is characteristically more laden with nutrients as compared to the oceanic water.

With regard to seasons, for both Diani and Nyali, the highest concentration of most nutrients was observed during the dry season (though not statistically different, $p>0.05$). The relatively lower nutrient concentrations during the wet season is most likely due to the dilution effect arising from large volumes of rain water which infiltrate the groundwater. The observed phenomenon seems to differ with what has been observed for the case of river (and surface runoff) nutrients within the region (Mwashote 1997, Ohowa *et al.* 1997, Kazungu 1989). During the wet seasons there is usually an observed general elevation of their nutrient levels.

Fig. 2a DIANI WATER COLUMN: Nitrate+Nitrite seasonal variations (1997-98)

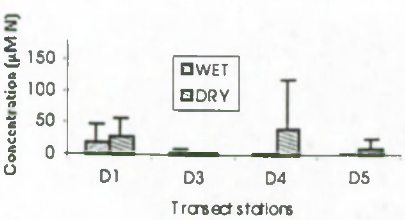


Fig. 2b: NYALI WATER COLUMN: Nitrates+Nitrites seasonal variations (1997-98)

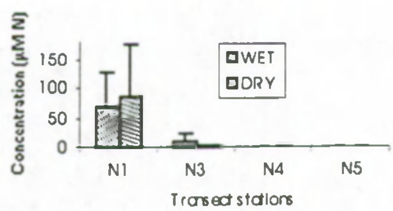


Fig. 3a DIANI WATER COLUMN: Ammonium seasonal variations (1997-98)

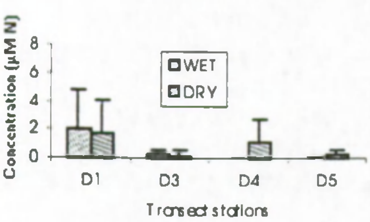


Fig. 3b: NYALI WATER COLUMN: Ammonium seasonal variations (1997-98)

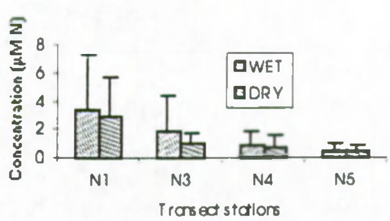


Fig. 4a DIANI WATER COLUMN: Phosphate seasonal variations (1997-98)

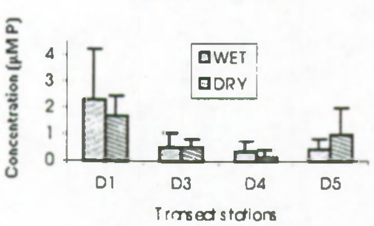


Fig. 4b: NYALI WATER COLUMN: Phosphate seasonal variations (1997-98)

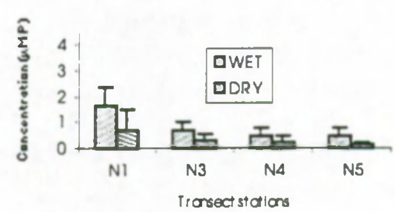


Fig. 5a DIANI WATER COLUMN: Silicate seasonal variations (1997-98)

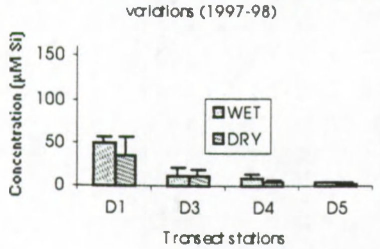


Fig. 5b: NYALI WATER COLUMN: Silicate seasonal variations (1997-98)

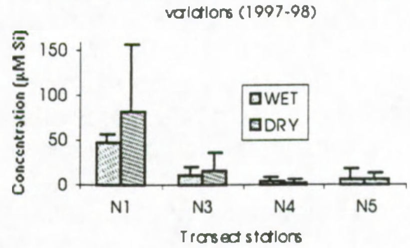


Fig. 9a: DIANI SEDIMENTS: Silicate seasonal variations profile (1997-98)

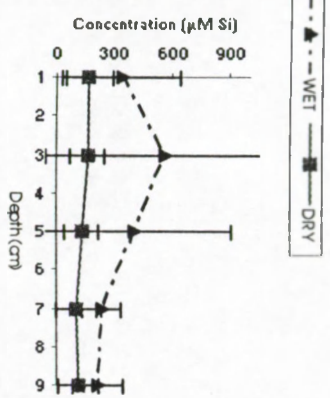


Fig. 8a: DIANI SEDIMENTS: Phosphate seasonal variations profile (1997-98)

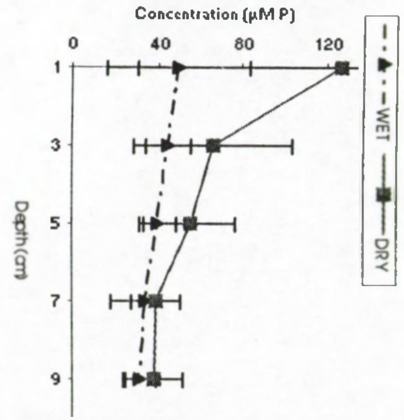


Fig. 7a: DIANI SEDIMENTS: Ammonium seasonal variations profile (1997-98)

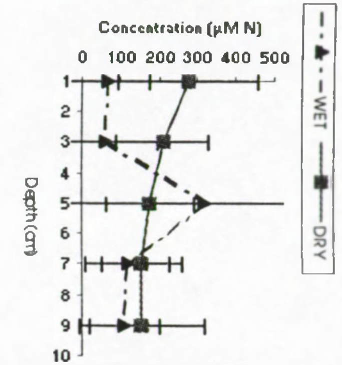


Fig. 6a: DIANI SEDIMENTS: Nitrate+Nitrite seasonal variations profile (1997-98)

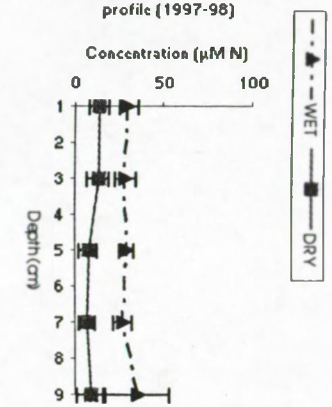


Fig. 9b: NYALI SEDIMENTS: Silicate seasonal variations profile (1997-98)

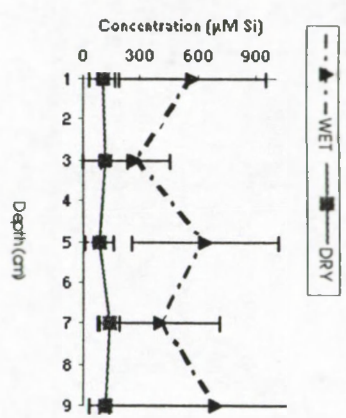


Fig. 8b: NYALI SEDIMENTS: Phosphate seasonal variations profile (1997-98)

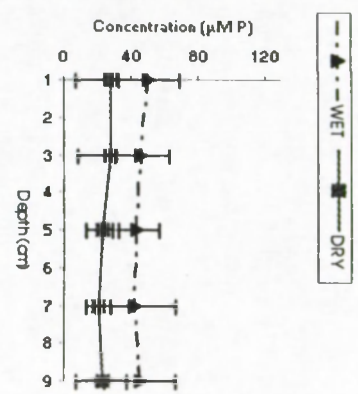


Fig. 7b: NYALI SEDIMENTS: Ammonium seasonal variations profile (1997-98)

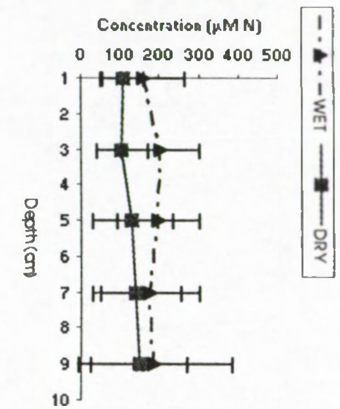


Fig. 6b: NYALI SEDIMENTS: Nitrate+Nitrite seasonal variations profile (1997-98)

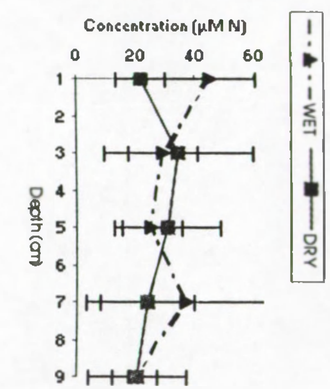


Fig. 10: Mean Ammonium concentration levels in the water column in KenyattaBeach

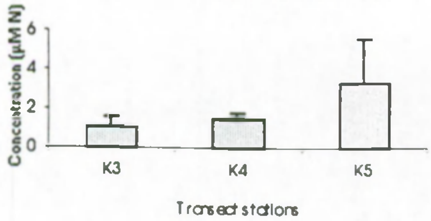


Fig. 11: Mean Phosphate concentration levels in the water column in KeyattaBeach

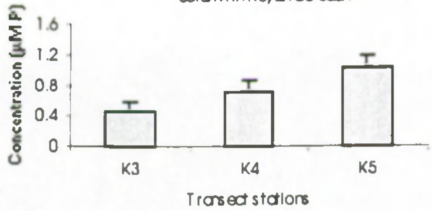
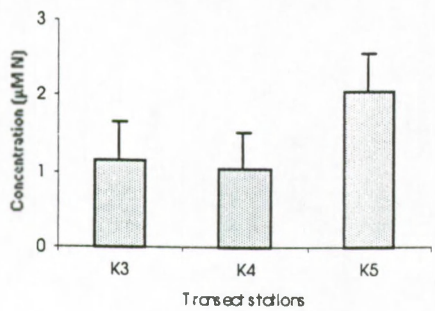


Fig. 12: Mean concentration levels of Nitrates+Nitrites in the water column in Kenyatta Beach



A rather interesting phenomenon also seems to be evident for the case of Diani water column nutrients at transect stations D4's and D5's, which is found within the coral reef area (Figs. 2a-4b). Some rather unusual elevation of nutrient levels seem to occur in these stations which is unlike the case with the corresponding Nyali stations. This observation strongly suggest that there is likelihood of localized discharge of nutrients, perhaps due to groundwater outflow occurring within these stations which gave rise to the observed elevation. The results show that the effect is more enhanced during the dry season as compared to the wet season. However, this notwithstanding, statistical analysis did not reveal any seasonal or spatial significant differences ($p>0.05$).

The foregoing information allows us to prudently assert that the nutrients coming into the ocean through groundwater outflow, impact mainly the nearshore areas especially those closest to the beach (stations 1 and 3). The impact rapidly diminishes toward the coral reef area (stations 4 and 5). When Diani and Nyali sampling areas are compared, it is found that the relative water column nutrient concentrations in the groundwater outflow area of Nyali are higher than those of Diani area (though not statistically, $p>0.05$). These relative differences in nutrients are most probably due to the fact that Nyali area is far more densely populated than Diani since it is found within an urban environment where there is more anthropogenic influence on the groundwater quality. For instance, in the area around Nyali beach, many pit latrines and septic systems are found per unit area as compared to the Diani area. The general anthropogenic impact on groundwater is evidenced by the levels of nutrients (and microbiota) observed in the boreholes/wells found around the sampling areas (Tables 1-2) as well as the sediment nutrient profiles of the sampling areas (Figs. 6a-9b).

In general the nutrient levels in the water column for Kenyatta (Figs. 10-12) beach sampling station were much lower compared to either Diani and Nyali. Kenyatta beach was chosen to represent a comparative station as it is found in an intrusion area unlike the other two sampling areas which were both outflow areas. It is evidently clear that the characteristics of water in this station are basically oceanic.

Previous similar studies elsewhere show that septic systems can potentially be significant point sources of nutrients especially NO_3^--N to groundwaters. Several of them have reported on the extent of NO_3^--N contamination of shallow ground water aquifers adjacent to septic tank beds. For instance, Walker *et al.* (1973) conducted *in situ* monitoring of soil profiles below sub-surface disposal beds of five septic systems. Their results indicated that essentially complete nitrification of NH_4^+-N from septic tank effluent occurred in the area of unsaturated flow in well-aerated soil below the crusted seepage bed. Nitrate removal by denitrification was highly unlikely and significant local groundwater contamination may be anticipated. The results of the same work also suggested the density of homes in a certain area is particularly important in judging the relative impact of nutrients derived from septic tanks on the nutrient status of groundwater stream and lakes in watersheds dominated by sandy soils. These observations are in close agreement with those of the present work.

Table 1. Mean Nitrate+Nitrite concentration levels in some boreholes/wells

AREA	BOREHOLE/WELL	CONCN. (μM N)
DIANI	Kitasa	59.37
	Bahati	117.00
	Mkwakwani	226.51
	Bondeni	148.41
	Solola	148.41
	Maweni	278.10
	Saidia	537.46
	Subira	234.87
	Bongweni	134.01
	Kibuyuni	148.41
	MEAN FOR DIANI	203.3±133.7
NYALI	FSH	643.00
	SBH	130.92
	NBH	420.77
	SOS	560.87
	FRT	184.06
	KSOKO	1451.70
	MWAND	219.81
	BERSH	606.28
	JER	2707.70
	KISIM	751.21
	KONG	1765.70
	MEAN FOR NYALI	858.4±799.3

Table 2. Some mean phosphate concentration levels in some boreholes/wells

AREA	BOREHOLE/WELL	CONCN. (μM P)
DIANI	Kitasa	1.794
	Bahati	1.950
	Mkwakwani	0.905
	Tangulia	1.061
	Bondeni	1.794
	Solola	1.584
	Maweni	1.166
	Saidia	1.794
	Subira	2.055
	Bongweni	1.271
	Kibuyuni	1.324
	MEAN FOR DIANI	1.52±0.39
NYALI	SBHdesal	0.225
	SBH	4.155
	SAF	0.378
	FSH	2.245
	NBH	1.058
	ZLN	0.542
	SOS	0.542
	FRT	2.400
	KSOKO	0.594
	MWAND	1.368
	BERSH	1.729
	JER	0.387
	KONG	1.523
	MEAN FOR NYALI	1.32±0.74

Microbiological water quality

<u>Nyali and Diani lagoons</u>

Tables 3 and 4 below are summaries of indicator organisms enumerated in the study areas between June 1997 and May 1998.

Table 3. Mean number of microbial indicators of pollution enumerated in Nyali area.

	Faecal coliforms no/100ml (±SD)	*E. coli* no/100ml (±SD)	Faecal *Streptococci* no/100ml (±SD)
Beach	32 ± 102.4	11 ± 23.6	537 ± 712.4
Lagoon	6 ± 10.1	4 ± 5.2	290.2 ± 551.8
Dry season	7 ± 7.5	7 ± 12.9	378 ± 632.3
Wet season	25.3 ± 90.4	7.0 ± 16.4	398.5 ± 635.8

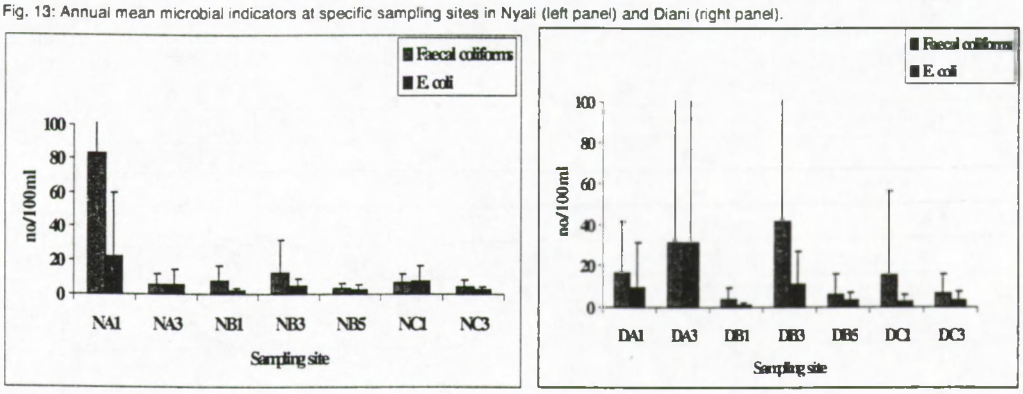
Table 4. Mean number of microbial indicators of pollution enumerated in Diani area.

	Faecal coliforms no/100ml (±SD)	*E. coli* no/100ml (±SD)	Faecal *Streptococci* no/100ml (±SD)
Beach	12 ± 27.8	5 ± 12.8	331 ± 611.8
Lagoon	20 ± 61.1	12 ± 50.5	202 ± 474.2
Dry season	20 ± 45.8	7 ± 14.9	289 ± 562.5
Wet season	13 ± 52.0	10 ± 51.3	234 ± 525.8

In Nyali area the level of contamination of the groundwater from the beach was generally higher than in seawater from the lagoon (Table 3) but no statistical difference was found between the number of faecal coliforms or faecal *Streptococci* enumerated in the beach and the lagoon. However the number of *E. coli* were significantly higher in the beach than in the lagoon ($p<0.05$). On average, the level of contamination (faecal coliforms, *E. coli* and faecal *Streptococci*) was higher in the NA transect followed by NB and then NC transect but the differences were not significant statistically.

In Diani, the trends were somewhat different with an indication of more microbial indicators enumerated in the lagoon than in the beach (Table 4). However, like in Nyali, these differences were not statistically significant. The number of indicator organisms enumerated in the dry season were slightly higher than in the wet season but again, not statistically so. In Diani, comparison among transects indicated that the number of these organisms enumerated in DA was higher than DB which in turn was higher than DC but the differences among transects were not statistically significant.

Fig. 13: Annual mean microbial indicators at specific sampling sites in Nyali (left panel) and Diani (right panel).



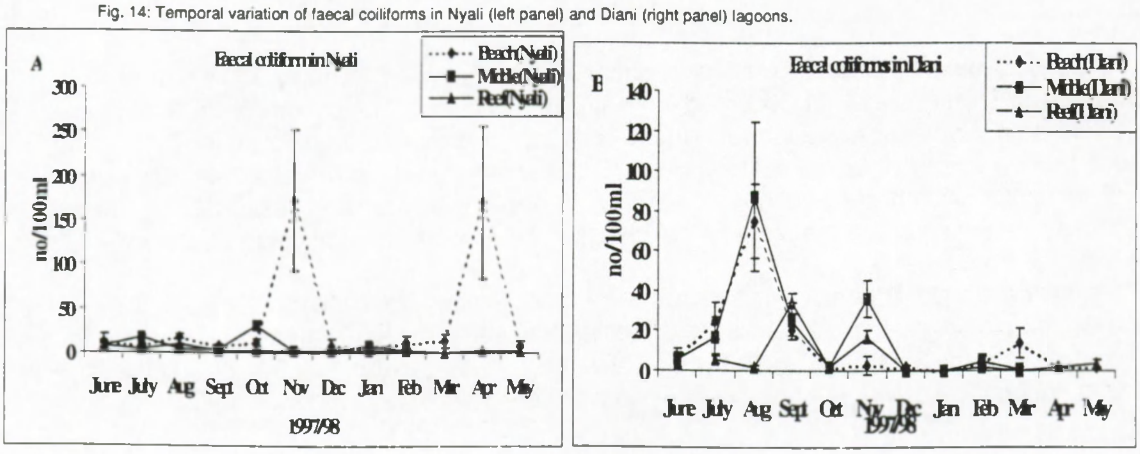
When considering the wet and dry seasons, all months in which at least 100mm of rainfall was recorded were considered wet and the others dry. Using this criterion, there was no significant difference found between the wet and dry seasons in both Nyali and Diani. However, there was a rise in the number of microbial indicators enumerated from groundwater at the beach in Nyali during the very wet months of April and November (Fig. 14a).

In both Nyali and Diani, there was no significant positive correlation between the number of microbial indicators enumerated from groundwater seeping along the beaches and that enumerated in the lagoons.

Temporal variation of faecal coliforms are graphically illustrated in Figs. 14a and 14b. As illustrated in Figs. 14a and 14b, there were some differences in the month to month level of microbial contamination in the two areas but these were not striking.

In Nyali area there was a general increase in the number of all microbial indicators near the beach during the wet periods (April, May and November). However, there were no statistical differences in the number of bacteria enumerated between the dry and wet seasons in the lagoon. This was not observed in Diani.

Fig. 14: Temporal variation of faecal coiliforms in Nyali (left panel) and Diani (right panel) lagoons.



Boreholes and wells

Microbiological water quality of water from eleven (11) wells and two (2) boreholes in Nyali area and one (1) well and ten (10) boreholes in Diani was investigated. Table 5 shows the results of the levels of faecal coliforms and *E. coli* in these water source points. Except for the well at Ziwa la ng'ombe in Nyali, all the other wells were covered.

Table 5. Levels of microbial contamination of water from boreholes /wells in Nyali and Diani.

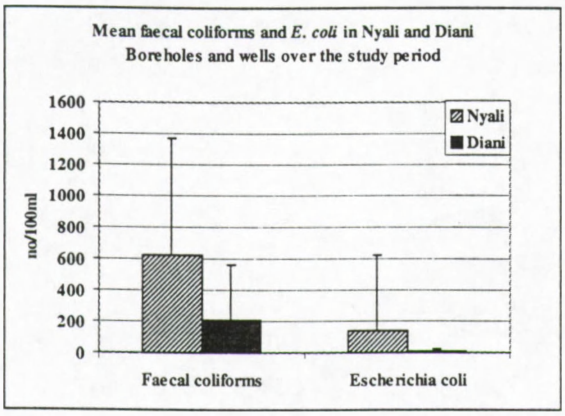
Sampling area	Name of site	Water sampled	Mean faecal coliforms (no/100ml)	SD	Mean *E. coli* (no/100ml)	SD
Nyali	Barsheba	Borehole	817	1108	55	7.1
Nyali	Four seasons	Well	1123	825.6	184	273.7
Nyali	Freetown	Well	825	1096	8	6.4
Nyali	Jerusalem	Well	142	195.2	30	5.7
Nyali	Kisauni Soko Mjinga	Borehole	1		2	2.8
Nyali	Kisimani/VOK	Well	817	1107.3	456	627.9
Nyali	Bahari club	Well	34	0	86	118.8
Nyali	Mkomani	Well	1600		34	
Nyali	Mwandoni	Well	230	169.7	26	12
Nyali	Nyali Beach	Well	47	46.7	7	6.4
Nyali	Silver beach	Well	1072	913.9	546	912.9
Nyali	SOS	Well	34	34.2	18	27.7
Nyali	Ziwa la Ng'ombe	Well	803	1127.1	12	14.1
Diani	Bondeni	Borehole	2	2.8	0	0
Diani	Kibuyuni	Borehole	170		8	
Diani	Kirora	Borehole	2		2	
Diani	Kitasa	Borehole	4	4.9	1	1.4
Diani	Maweni	Borehole	95	63.6	9	4.2
Diani	Mpwakwani	Borehole	801	1130	17	24
Diani	Safari Beach	Well	20	13.2	4	3.5
Diani	Solola	Borehole	2		0	
Diani	Subira	Borehole	800	1131.4	17	24
Diani	Ukunda Police	Borehole	26		17	

Results showed a clear difference in water quality between Nyali and Diani (Fig. 15). Highest faecal coliform counts were recorded in Nyali area. In terms of faecal coliforms, water from boreholes/wells in Nyali area was about four times as contaminated as that in Diani. The differences in the number of faecal coliforms and *E. coli* enumerated were statistically significant. The level of contamination showed no seasonality (between wet and dry seasons). Again, no obvious trend was observed on the relationship between the level of contamination of groundwater and the distance from the sea.

Using the Kenya Bureau of standards safe limits, water from the boreholes/wells was classified as potable or not. Fig. 16 shows the potability of water in the study areas.

Results in both Diani and Nyali are indicative of a certain extent of contamination of the ground and lagoonal waters. Within the sampling period, there was variation in the level of microbial indicators of pollution (faecal coliforms, *E. coli* and faecal *Streptococci*) in both Diani and Nyali.

Fig. 15: Mean faecal coliforms and *E. coli* in Nyali and Diani boreholes and wells over study period.



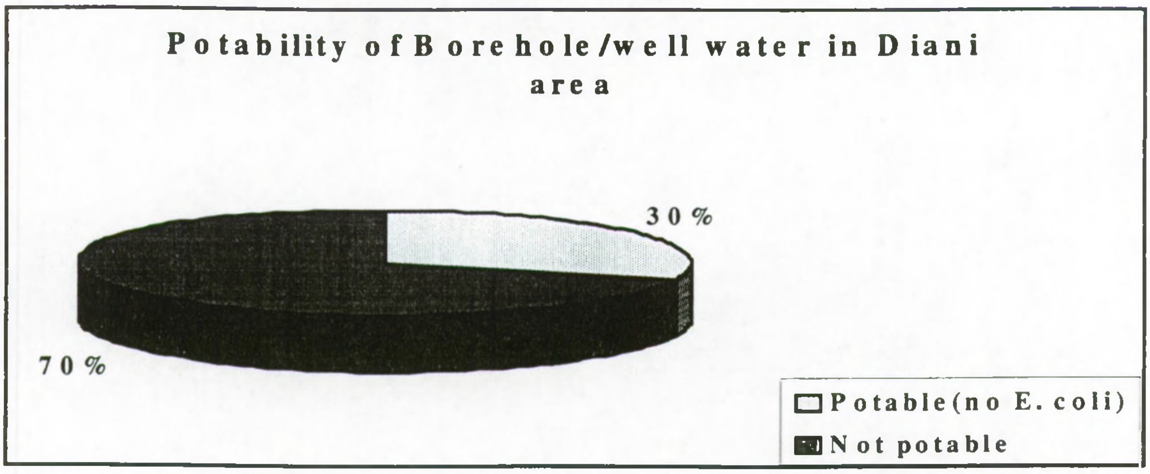
In the two study areas in general, higher number of indicator organisms were enumerated in the groundwater collected from the seepage points (on the beach before mixing with the seawater) than in the seawater in the lagoons. This is an indication that groundwater is generally more contaminated than lagoonal water but once groundwater mixes with seawater, indicator organisms are exposed to the antiseptic properties of seawater leading to the die-offs of these organisms. In Both Nyali and Diani, the A and B transects were found to be relatively more contaminated than the C transects. More groundwater seepage was however notable in the B and C transects in Nyali and C in Diani. This is an indicator that groundwater may not be the only source of contamination of the lagoonal waters and that discharges from beach establishments adjacent to these transects may be significant sources of faecal or raw sewage pollution as compared to the groundwater sources. Lack of obvious seasonal trends in the level of microbial indicators in both areas is a further pointer that groundwater or/and perhaps direct discharge from beach establishments could be responsible for determining the level of contamination of water in the lagoons.

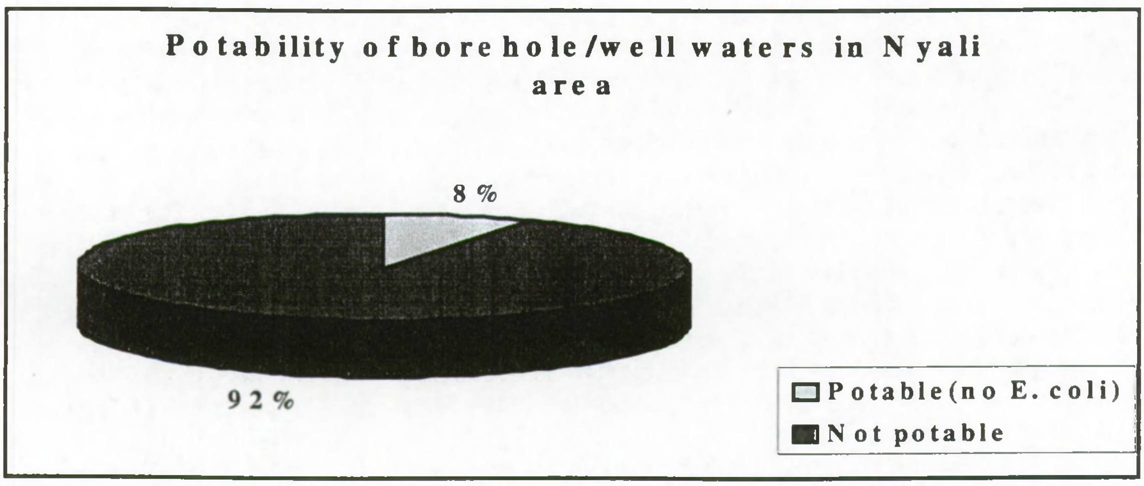
Results from the study period are indicative of a clear difference between Nyali and Diani. Nyali is a relatively more contaminated area than Diani. Though there was no direct relationship between groundwater and lagoonal water quality, there is evidence of groundwater influence on the marine water quality. High groundwater contamination in Nyali was coupled with relatively higher contamination in the lagoonal waters.

In general, the level of contamination was higher during the rainy season (like in April, May and June). During these seasons, storm and surface water runoffs become additional sources of bacterial contaminants. The level of bacterial contamination was especially higher in Nyali, an indication that the surface runoff in this area emanates from a more contaminated grounds than in Diani.

In both Nyali and Diani, faecal coliform levels in the lagoon waters were less than 100/100ml for more than 95 per cent of all samples analyzed. There are no existing Kenyan standards for marine or estuarine water quality. However in terms of microbial indicator levels, water quality in the lagoons in the two areas meet the EEC guideline on bathing water standards of less than 100/100ml faecal coliforms for 90% of the samples (Tebutt, 1992). In both Nyali and Diani, the EEC mandatory limit of 2000 faecal coliforms/100ml was never reached (Council of European communities, 1975, European economic commission, 1976). This was an indication that the lagoonal waters in these areas meet the EEC guide limit for recreational water (faecal coliforms and *Streptococci* of 100/100ml) and so the water is fit for recreational use.

Fig. 16: Potability of water in Nyali and Diani.





An attempt was made to use the faecal coliform and faecal *Streptococci* (FC:FS) ratio in the two locations to identify pollution sources (Martens *et al.*, 1984). It is a valuable test because a ratio of faecal coliform to faecal S*treptococci* of 4.0 or higher typically indicates domestic waste (human faecal pollution) and ratios of 0.6 or lower are typical for discharges from farm animals or stormwater runoff. Most microbial indicators have poor survival in many of the circumstances encountered in the marine environment.

Results showed that in most cases the ratio was less than 1. This was due to the fact that though the faecal coliform levels were less than 100/100ml, those of faecal *Streptococci* were high and variable. In marine waters, the die off rate of coliforms is higher than that of faecal *Streptococci* and though the faecal coliform/faecal streptococci (FC/FS) ratio has been used to evaluate the origin of faecal pollution, chlorination lowers this ratio and thus renders the method unreliable under such circumstances (Rosser & Sartory 1982). Thus, there is a likelihood that chlorinated swimming pool water discharged on the beach and lagoons may interfere with the FC/FS ratio and therefore caution may be required in the use of the ratio in predicting pollution type or source.

Based on the results, the groundwater from the boreholes and wells in these two areas are not potable as the levels of faecal coliforms, *E. coli* and faecal *Streptococci* exceed those recommended by the Kenya Bureau of Standards (<10 faecal coliforms/100ml and 0 *E. coli*/100ml) and EEC guide limit of 0/100 ml of these organisms for drinking water (EEC member states recommend a maximum of 20 faecal coliforms or 20 faecal *Streptococci* for surface water intended for abstraction or drinking).

Within each area, there were differences in groundwater quality in the boreholes and wells. Differences were observed even between wells/boreholes that were within short distances from each other. This is an indication that the water in the wells/boreholes may be derived from different groundwater aquifers receiving different anthropogenic inputs from the adjacent population. The level of groundwater contamination is related to the population density and mode of sewage disposal.

According to the population projection, the population of Mombasa district was to reach 654783 persons by 1996. With a projected density of 3118 persons per square kilometer, it makes Mombasa one of the most densely populated district in the country (Republic of Kenya, 1996). Expected density in Kisauni division, where all the boreholes sampled in Mombasa are located, was 1539 persons per square kilometer.

Kwale District 1996 projected population was 465011 persons with Msambweni being the most populous in the district (190422 persons and a density of 57 persons per kilometre). Diani area has been growing steadily due to booming tourist industry with hotels and recreational facilities (Republic of Kenya, 1996). The difference in water quality between the two areas is directly related to population densities. High population density in Nyali has had the impact of lowering the groundwater quality in this area as opposed to the relatively lower anthropogenic influence in Diani. In both areas, there is no centralized sewage treatment and septic tanks/soakage pits and latrines are a common phenomena. Large volumes of sewage from the population eventually find itself in the groundwater and eventually in the marine environment. However judging from the level of microbiological contamination of both ground and marine waters, there seems to be some filtration process that somehow minimizes the entry of potential pathogens into the marine environment. Otherwise, the high groundwater contamination especially in Nyali would have resulted in commensurate lowering of marine water quality in the adjacent Nyali lagoon through underground seepage.

Conclusions

The above results with respect to nutrient concentrations within the Nyali and Diani lagoonal waters clearly indicate that there is a considerable additional contribution of nutrients from streams associated with ground water outflow. At Nyali study sites, these underground water streams feeding the lagoonal waters were found to have salinity values of between 25 and 27 PSU (Kitheka, this report). (NO_3^-+NO_2^-)-N was found to be the most dominant form of dissolved nitrogen contributed by these low salinity ground water streams. While concentrations of (NO_3^-+NO_2^-)-N associated with these streams varied between 5 and 200 µM, those of ammonium were $\leq$ 16 µM. At Diani study stations, the streams feeding the lagoonal waters were found to have salinity values of between 26 and 28 PSU (Kitheka, this report). The maximum (NO_3^- +NO_2^-)-N concentration associated with these streams was ca. 65 µM which is relatively lower than the maximum observed at Nyali study station.

In both the study stations a general gradient of nutrient concentrations was observed within the lagoonal water column, with those stations in the offshore deep lagoonal waters having relatively lower nutrient concentrations compared to those stations next to the beach except for Diani which showed a slightly different trend for offshore stations due to a possibility of a localized source of nutrient input most probably arising from groundwater outflow within the area.

Nutrient enrichment of the oceanic waters takes place mainly in the nearshore areas for groundwater outflow areas. The highest nutrient levels in Diani and Nyali were generally observed during dry season. Diani groundwater outflow area extends much further into the ocean than the Nyali case.

Nyali nearshore area has higher nutrient levels than Diani area mainly due to the greater anthropogenic influence experienced in the former. Kenyatta beach nutrient characteristics are basically oceanic since there is no evidence of groundwater outflow within the beach.

Data generated from this study is indicative of some extent of contamination of lagoonal water by groundwater, surface runoff and occasional direct discharge of raw sewage and waste from beach establishments. In general, lagoonal waters in both Nyali and Diani appear safe for contact recreation. However, though the situation is not serious for now, the continuing rise in population and tourist industry may eventually lead to increased contamination of groundwater and recreational beaches with faecal pathogens resulting in higher health risk to tourists and other recreationists. The level of bacterial contamination of groundwater is higher in Nyali than in Diani and the question of public safety should be addressed. Groundwater from Diani can be made potable by simple physical treatment and disinfection like boiling, rapid filtration and disinfection while that in some wells/boreholes may require physical and chemical treatment and disinfection including chlorination. In Nyali, there is high risk of using untreated groundwater and both physical and chemical treatment of the water is necessary before use.

In general it is evident that groundwater outflow areas impact strongly on nearshore coastal ecosystems with regard to elevation of anthropogenic associated inputs such as nutrients and microorganisms.

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Influence of groundwater discharge on community structure in Kenyan coastal lagoons

J. M. Mwaluma, P. O. Wawiye, J. N. Uku & S. N. Mwangi
Kenya Marine and Fisheries Research Institute, Mombasa, Kenya

a) Influence on plankton communities

Introduction

Along the Kenyan coast, there exists a number of places with ground water discharge and such may be reflected by the rich and diverse mangrove fauna. There is an increasing awareness that groundwater is a resource of enormous significance. However due to increase in local population settlements along the coast, construction of beach hotels for tourism, that pour their effluent directly into the ground, there is an increase in anthropogenic pressure on the groundwater reservoirs both in quantity and quality. The ground water together with all the effluents and nutrients eventually seeps out into various parts of the lagoons and creeks. The nutrients and the fresh water input may alter the abundance community structure of nearshore plants and animals, however very little information exist on the extent. Previously, studies conducted in inshore waters of Tudor, Gazi and Mida creeks (Mwaluma *et al* 1993, Osore 1994, Okemwa 1990, and Mwaluma *et al* 1998) revealed that salinity changes often affected zooplankton species diversity. They observed that low salinity often resulted to low species diversity and high salinity associated with high zooplankton species diversity. The explanation given to that observation was that, when salinity was low, only a few organisms are able to tolerate that salinity, hence only few species thrive, bringing the diversity index down. With higher salinites, a wider range of zooplankton organisms are able to thrive hence resulting in higher diversities. Few of these organisms that have been associated with low salinities are Mysiids and the copepod, *Pseudodiaptomus stuhlmani.* In Gazi bay Kenya, Osore (1994), reported *Pseudodiaptomus stuhlmani* to occur in brackish waters, whereas *Pseudodiapomus hessei* and Mysiid, *Rhopalophthalmus terranatalis* (Mysiid) has also been found to flourish in low salinities (15 -30 ppt) in South African estuaries (Wooldridge & Smith 1979, Wooldridge & Webb, 1988).

Along the coast a dynamic balance and interaction exists between seaward outflow of ground water and salt water intrusion into coastal freshwater aquifers. These interactions and the implications they may have on the ecological processes, structure and functioning of the nearshore environment still remain scantily known

This project was therefore initiated with an overall aim of investigating the significance of changing inputs of ground water discharge and the effects of anthropogenic substances it may contains to the near shore ecosystems.

Objective

The specific objective, for the plankton studies was, to investigate the effects of ground water discharge on the abundance, diversity and community structure of plankton in Nyali and Diani lagoons.

Materials and methods

Plankton samples were collected once a month from May 1997 to June 1998 at each lagoon using a 25μm (Phytoplankton) and 332μm (Zooplankton) mesh size plankton nets during low tide. Collected samples were preserved in 5% formalin labeled and reserved for laboratory analysis.

Phytoplankton species composition was analysed using an inverted microscope and abundance expressed as cell/l. Chlorophyll-a and B.O.D samples were collected and analysed according to Parsons et al (1984) and expressed as μg/l and mgO_2 / l respectively. Zooplankton samples were scanned using a Wild Heerbrugg stereo microscope. This initial step of scanning through the whole sample was to ensure that all possible zooplankton groups present in the sample was observed. Each sample was analysed to the lowest taxonomic level possible using identification keys of Giesbrecht (1892), Sars (1901), Scott (1909), Owre and Foyo (1967), Wickstead (1965), and Wickstead (1976). After listing all zooplankton identified, the sample was then subsampled. Five subsamples of 5 ml each were then pipette from the sample and counted. In cases where organisms in the sample were too sparse, the entire sample was examined. The abundance of zooplankton groups was obtained by dividing the total number of organisms counted by the volume of water filtered (m^{-3}) to obtain number of individuals per cubic meter (no.m^{-3}). Mean abundance of zooplankton was calculated for all the transects at Nyali (NA, NB and NC) and Diani (DA, DB and DC) for the year. Shannon diversity index (H) was also calculated for Phytoplankton and zooplankton communities from the beach stations to the reef stations with the given formula:

$$H = \{n \log n - \sum_{i=1}^{k} f_i \log f_i\}/n$$

Where H is species diversity, k is the number of categories, n is sample size and fi is the number of observations in category i. (Zar, 1974) . Two factor ANOVA without replication was used to check for significant differences in zooplankton abundance between transects, stations (Beach, Middle and Reef) and between the two lagoons.

Additional samples were collected for Kenyatta beach lagoon at stations K3, K4, and K5 in June and October 1998.

Results

During the wet season, phytoplankton cell abundance was higher in the middle stations in Nyali lagoon (Figure 1). The high standard deviation was due to the predominance of *Oscillatoria* spp in one sample which formed upto 82% of total population. Other species found in both lagoons were *Biddulphia* spp, Zooxanthellae, *Synedra filiformis*, and *Thalassiothrix frauenfeldii*.

There were however, very few dinoflagellates recorded in both lagoon systems.
In both lagoons, abundance values tended to decrease towards the reef. The blue green algae *Oscillatoria* spp was the dominant species in both lagoons. During the dry season, however, abundance was lower in the middle station in both Nyali and Diani lagoons (Figure 1a) but was not the case in the wet season (Figure 1b).

Chlorophyll-a values were higher in Nyali lagoon than in Diani, with an annual mean of 1.46µg/l as compared to 1.13µg/l in Diani though not statistically significant ($p > 0.05$). No significant differences in Chlorophyll-a were observed among stations as well. In both lagoons higher chlorophyll- a values were recorded during the rainy season in November, April and May (figure 2a). Meanwhile, chlorophyll-a values obtained at nearshore beach stations (3) were always higher than reef stations (5) in both lagoons (figure 2b). Nyali lagoon had comparatively higher values with the peak station mean of 3.4 ± 1.5 µg/l (NB3) compared to 1.2 ± 1.0 µg/l (DB3) at Diani (Figure 2b).

The B.O.D levels in Nyali were statistically higher ($p < 0.05$) than Diani lagoon with a mean of 1.5 mgO_2/l compared to 1.00 mgO_2/l for the year (figure 3). Higher B.O.D levels were obtained during the rainy season as compared to the dry though not statistically significant. No significant differences were observed between stations. Results obtained from Bamburi beach (thought to be a salt intrusion area) revealed chlorophyll–a values of less than 0.5µg/l and B.O.D values of less than 1.0 mgO_2/l.

Shannon diversity index (H) calculated for phytoplankton species, revealed higher values during the wet season as compared to the dry for both lagoons. Diversity in Nyali lagoon during the wet season varied between 0.4 – 1.1, and during the dry it varied between 0.05 – 0.5 (Figure 4). In Diani diversity varied between 0.2 – 1.00 during the wet season while during the dry it varied between 0.1 – 0.4 (Figure 4).

Zooplankton species composition in both Nyali and Diani lagoons was similar. Common zooplankton groups observed in both sites were Copepoda, Brachyuran larvae, Brachyuran

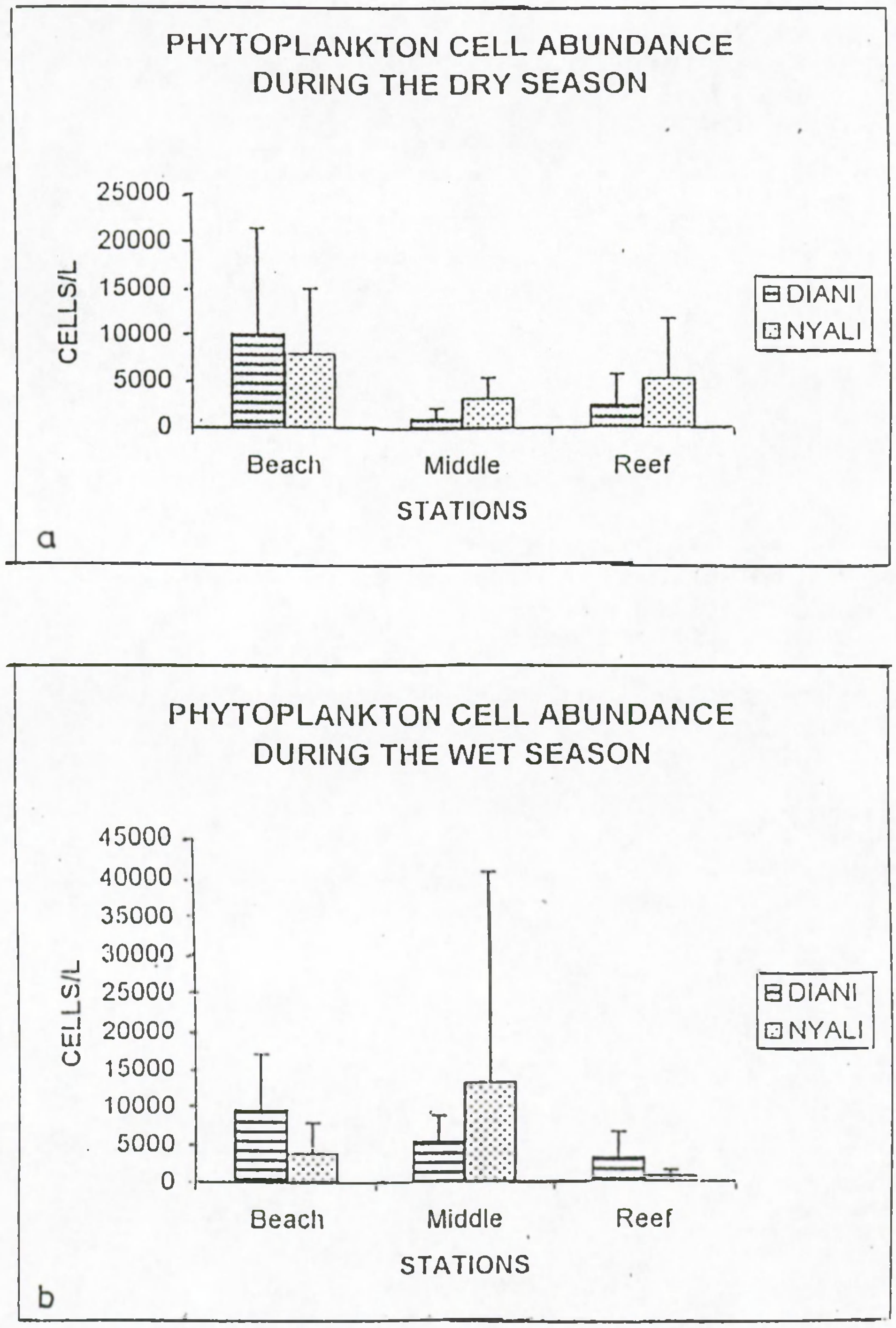


Fig.1 Phytoplankton cell abundance during the dry and wet season in Nyali and Diani lagoons.

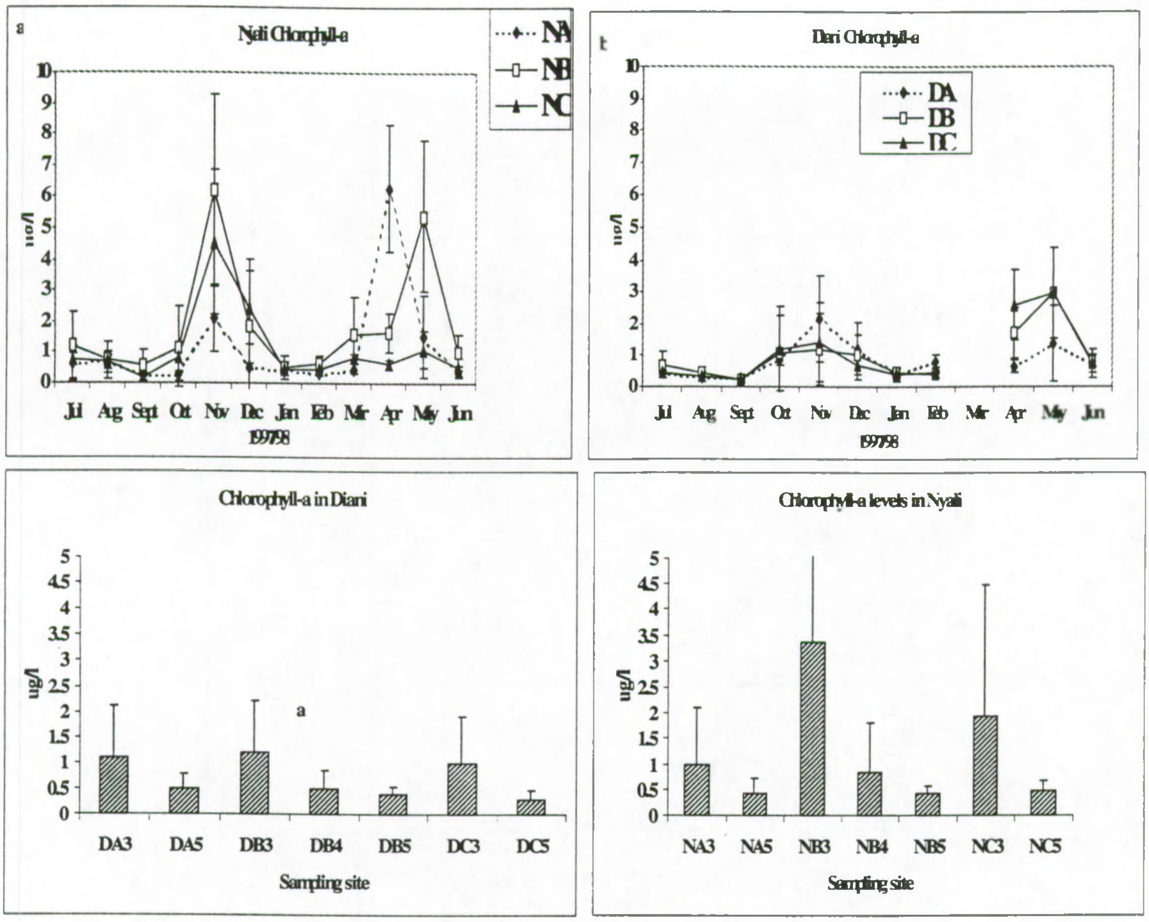


Fig. 2. Temporal changes (top panels) and spatial distribution (bottom panels) in chlorophyll-a levels in Nyali and Diani lagoons.

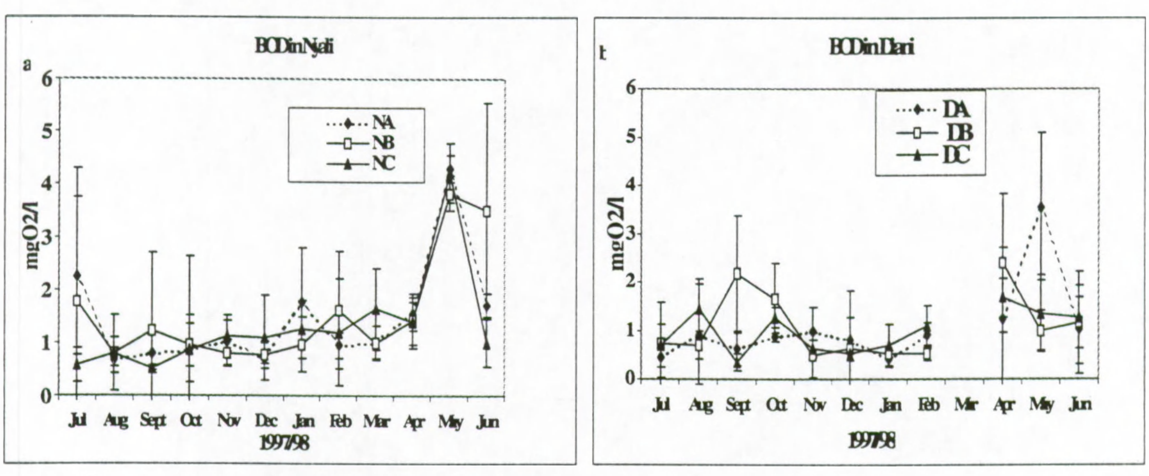
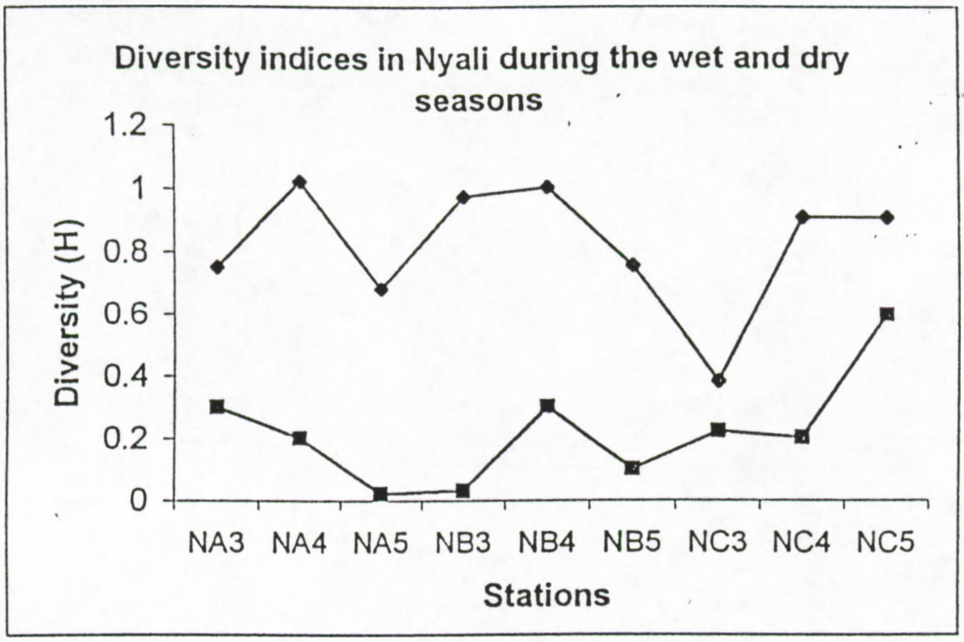


Fig. 3. Temporal changes in BOD in Nyali and Diani lagoons.



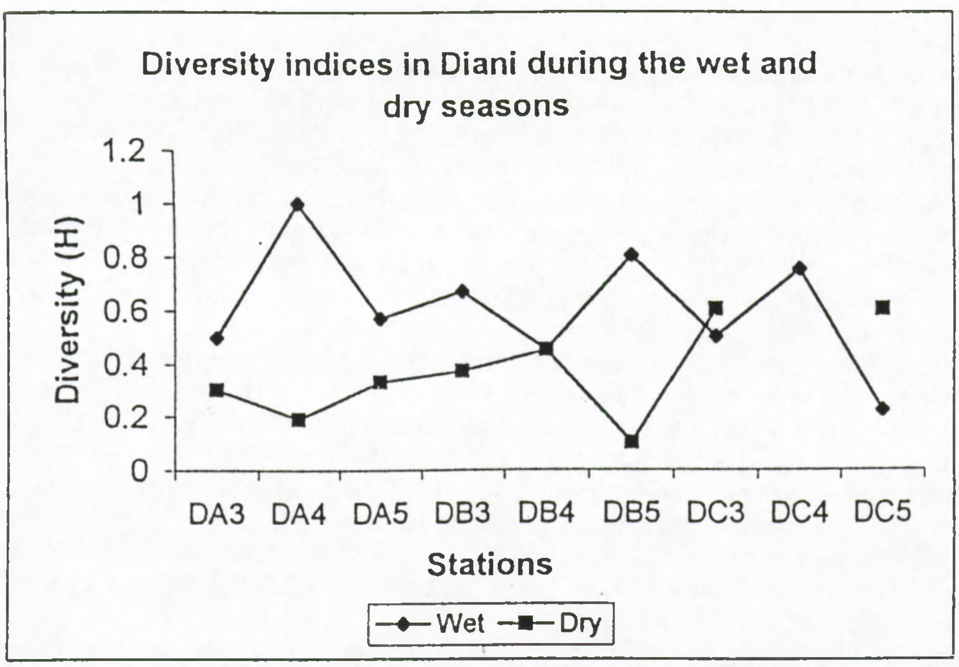


Fig. 4. Phytoplankton species diversity during the dry and wet season in Nyali and Diani lagoons.

meagalopa, Mollusca (Gastropod larvae, Bivalve larvae and Squid larvae) Fish eggs and larvae, (Gobiidae, Bleniidae, Lutjanidae, Apogonidae) Amphipoda (Gammarid, Hyperiids & Carprellid), Appendicularia (*Oikopleura* spp & *Flitillaria* spp), Mysiids, Cumacean (*Diastylis* spp), Sergestidae (*Lucifer* spp) and Caridea (Table 1).

Table 1. Common zooplankton groups in Nyali and Diani and their distribution.

	Stn 3 (Beach)	**Stn 4 (Middle)**	**Stn 5 (Reef)**
Copepoda	XXX	XXX	XXX.
Amphipoda	XXX	XX	X
Fish eggs	X	XX	XX
Fish larvae	X	XX	XX
Mollusca			
Gastropod larvae	XXX	XX	X
Bivalve larvae	X	X	X
Squid larvae		X	X
Cumacean	X	X	
Caridea		XX	XX
Chaetognatha		XX	XX
Sergestidae			X
Appendicularia		X	X
Polychaeta larvae	X	X	X
Brachuyran larvae	X	XXX	XX
Brachyuran megalopa		X	X

XXX Above 100 no.m^{-3}
XX Between 50- 100 no.m^{-3}
X Below 50 no.m^{-3}

Some groups were present throughout all stations whereas others were found at particular stations in the lagoon (Table 1). Dominant copepod species in the lagoons were *Paracalanus* spp, *Tortanus murrayi, T. gracilis, Temora turbinata, Acartia* spp, *Oithona* spp, *Labidocera* spp, *Acrocalanus* spp, *Centropages orsinii, Corycaeus gibbulus,* and *Corycaeus* spp. Harpacticoid copepods were also common. Species composition at Kenyatta beach transect was similar in composition to the other lagoons. Dominant zooplankton species at station K3 were *Oithona* spp, Harpacticoid copepods, and Amphipods (Gammarids). Dominant zooplankton at K4 were, *Paracalanus* spp, Brachyuran larvae, Fish eggs and fish larvae. At station K5, Brachyuran larvae, Gastropod larvae and *Paracalanus* spp were dominant.

Mean zooplankton abundance for the year was calculated for Nyali and Diani Beach. Zooplankton abundance in Nyali ranged between 75 - 175 no.m^{-3}. Zooplankton abundance was highest in the middle belt (NA4, NB4 and NC4) in all the transects (Figure 5) The reef station (N5) had lowest abundance at NA and NB transects. Mean zooplankton abundance between transects (NA, NB & NC) however was not significant ($p > 0.05$). Zooplankton abundance in Diani ranged from 20 - 100 no.m^{-3}, with higher abundances occurring at DB4, and DB5 (Figure 5). Mean zooplankton abundance between transects (DA, DB & DC) was nonetheless not significant ($p > 0.05$). Between the two lagoons however, Nyali had significantly ($p = 0.0003$) higher zooplankton abundance as compared to Diani.

Mean zooplankton diversity for the study for Nyali and Diani is summed up in Table 2. At Nyali lagoon, the lowest diversity recorded was 0.34 at NB3 during the rainy season in May. Highest diversity was 1.41 at NC5 in October during the dry season. Zooplankton diversity was generally found to increase from the beach to reef stations (Table 2). Diversity between transects and between transects and between stations was however not significantly different ($p > 0.05$). In Diani lagoon, lowest diversity for the period was 0.1 at DC4 in December during the rainy season. Highest diversity was 1.44 recorded in August during the dry spell. Like Nyali lagoon, diversity tended to increase from beach to reef stations in all transects. However, no significant ($p > 0.05$) differences were observed between transects and among stations.

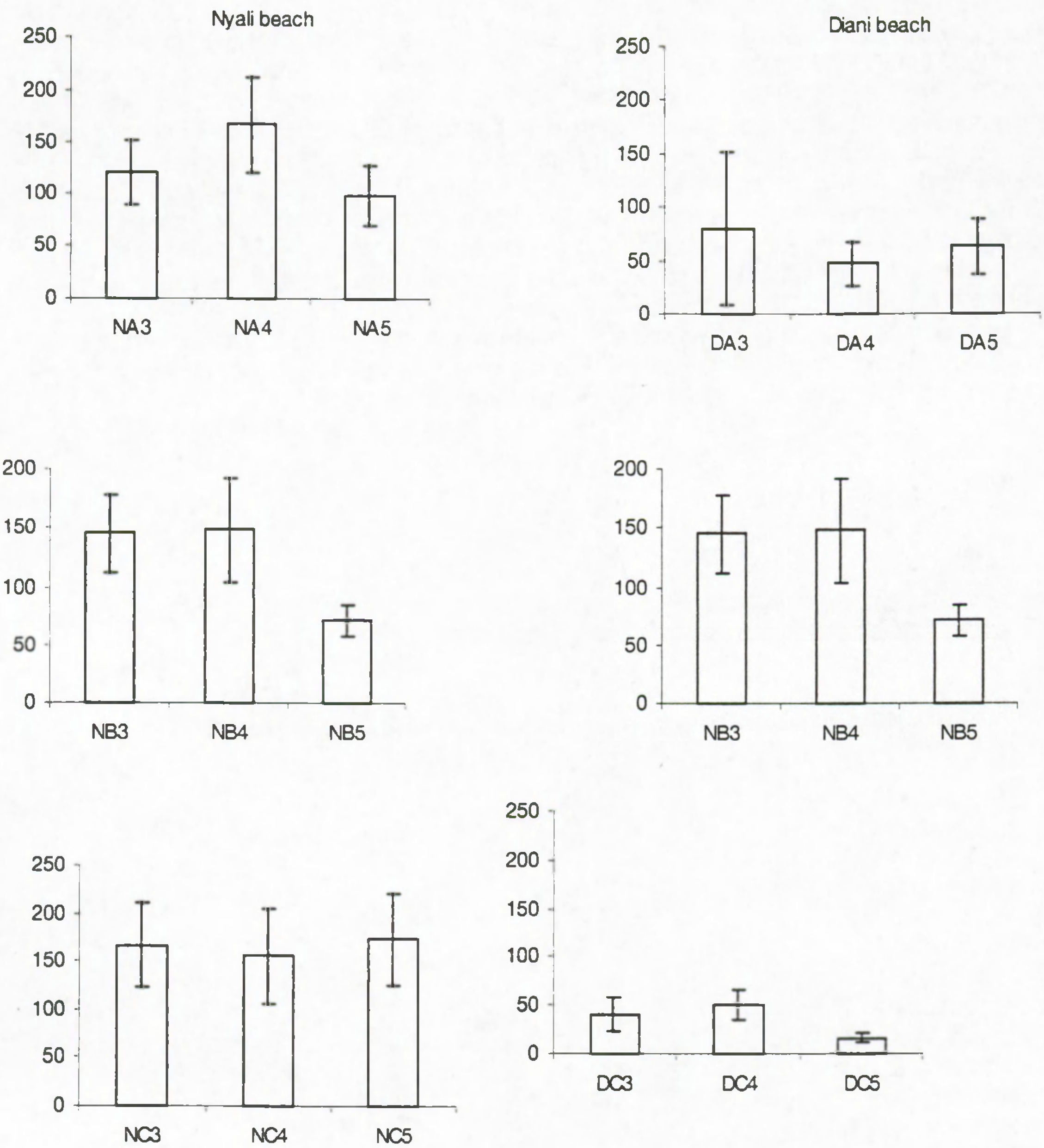


Fig. 5. Mean zooplankton abundance at Nyali and Diani Beach lagoons.

Table 2. Mean diversity (H) of zooplankton in Nyali and Diani lagoons May 1997 – June 1998.

Nyali	Beach (3)	Middle (4)	Reef (5)
NA	**0.87** ± 0.32	**0.89** ± 0.37	**1.00** ± 0.30
NB	**0.74** ± 0.34	**0.93** ± 0.15	**0.96** ± 0.41
NC	**0.77** ± 0.26	**1.07** ± 0.24	**0.93** ± 0.41
Diani			
DA	**0.80** ± 0.2	**0.90** ± 0.27	**0.95** ± 0.29
DB	**1.00** ± 0.33	**0.92** ± 0.27	**0.95** ± 0.21
DC	**0.63** ± 0.31	**0.82** ± 0.38	**1.00** ± 0.19

Effects of ground water seepage caused the emergence of two " indicator" sps species of zooplankton associated with low salinities. These were copepod *Pseudodiaptomus* spp and a

Mysid spp. In Nyali beach, *Pseudodiaptomus* spp and Mysids showed two peaks corresponding to the two rainy seasons May - June 1997 and during the El-nino Floods in November- December 1997 (Figure 6a). A third smaller peak was observed for *Psedodiaptomus* spp in the rainy period of April 1998. Likewise in Diani beach, three peaks were observed associated with fresh water influx. A peak abundance of Mysids occurred in November during the El nino followed by smaller a peak of both *Pseudodiaptomus* spp and Mysids in February 1998 and finally a third peak in May 1998 during the long rains.(Figure 6b). However, the distribution of these two species in both lagoons in relation to ground water discharge was somewhat different. In Nyali beach, the two species were found distributed nearshore (N3 stations), and the population dwindled towards the reef (Figure 7a). In Diani, the population of these two species increased away from the beach stations with most of them occurring at D5 (Figure 7b).

In the additional stations sampled at Kenyatta beach, few (three individuals) Mysids were reported at the K3 in October. This however did not bear any significance as not many samples were collected for this transect. *Pseudodiaptomus* spp was not reported.

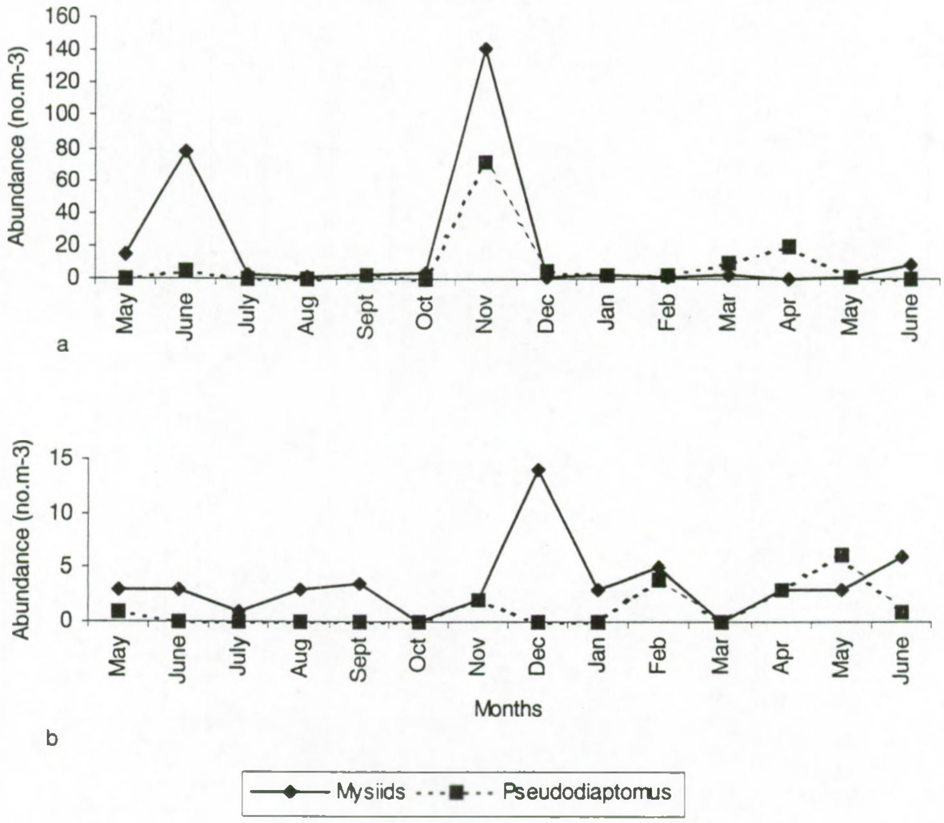


Fig. 6. Mean abundance of Mysids and Pseudodiaptomus *spp. in Nyali (a) and Diani Beach (b).*

DISCUSSION

The dominance and diversity of the phytoplankton community was greatly affected by alternate levels of nutrient input and may have had far reaching consequences on the biota at higher trophic levels as they are directly or indirectly dependant on these primary producers. Phytoplankton abundance was influenced by groundwater discharge in both lagoons. Higher abundances were encountered at nearshore stations due to high amounts of nutrients at the beach (Figure 1). Seasonality did not seem to influence phytoplankton abundance however during the rainy season *Oscillatoria* spp was found to flourish due to increased amount of nutrients especially at Nyali beach. Phytoplankton species diversity was affected by seasonality. In Nyali lagoon their was a wider variation in species diversity between the wet and dry season as compared to Diani (figure 4). This was associated to a wider seasonal salinity variation in Nyali lagoon (5.2 PSU - 29

PSU) as compared to Diani (26 PSU - 30 PSU) (Kitheka 1998) which influenced diversity. In both lagoons, higher phytoplankton diversities were observed during the wet season as compared to the dry.

The higher Chlorophyll–a values observed during the rainy season was attributed to a combination of increased nutrient rich surface runoff into the lagoon in addition to ground water outflow. The higher values of B.O.D and Chlorophyll-a obtained in Nyali lagoon as compared to Diani were indicative of higher nutrient enrichment of water in Nyali lagoon. The higher diversities obtained during the wet season as compared to the dry season in both lagoons may be linked to the higher nutrients being received in the wet season allowing diverse phytoplankton species to thrive.

Zooplankton species composition in Nyali and Diani lagoons are comparable. Distribution of zooplankton in Nyali and Diani lagoons was influenced by differences in salinity and habitat selectivity. Differences in species distribution along transects (from beach to reef) was noted for some species like; Amphipods, fish eggs, fish larvae, Caridea, Gastropod larvae, Brachyuran larvae, Sergestidae, and Chaetognaths (Table 1). Copepoda group displayed this trend best, with *Oithona* spp, *Acartia* spp, and Harpacticoids dominating the nearshore stations (3), *Labidocera* spp, *Tortanus murrayi*, *T. gracilis*, *Calanopia elliptica*, *Temora turbinata* occupying the middle stations (4) while, *Eucalanus* spp , *Copilia mirabilis*, *Euchaeta marina* and *Candacia truncata* being found at the reef station (5). This distribution however, was not always strictly encountered as sometimes due to wave action these species were found distributed all parts of the lagoon. The presence of seagrass cover in the mid - lagoons provided a suitable habitat for gammarid amphipods, and sphaeromatid Isopods.

The higher abundance of zooplankton observed in Nyali lagoon as compard to Diani was associated to higher nutrients being received in Nyali as Diani lagoon (Mwashote and Kazungu, 1998).

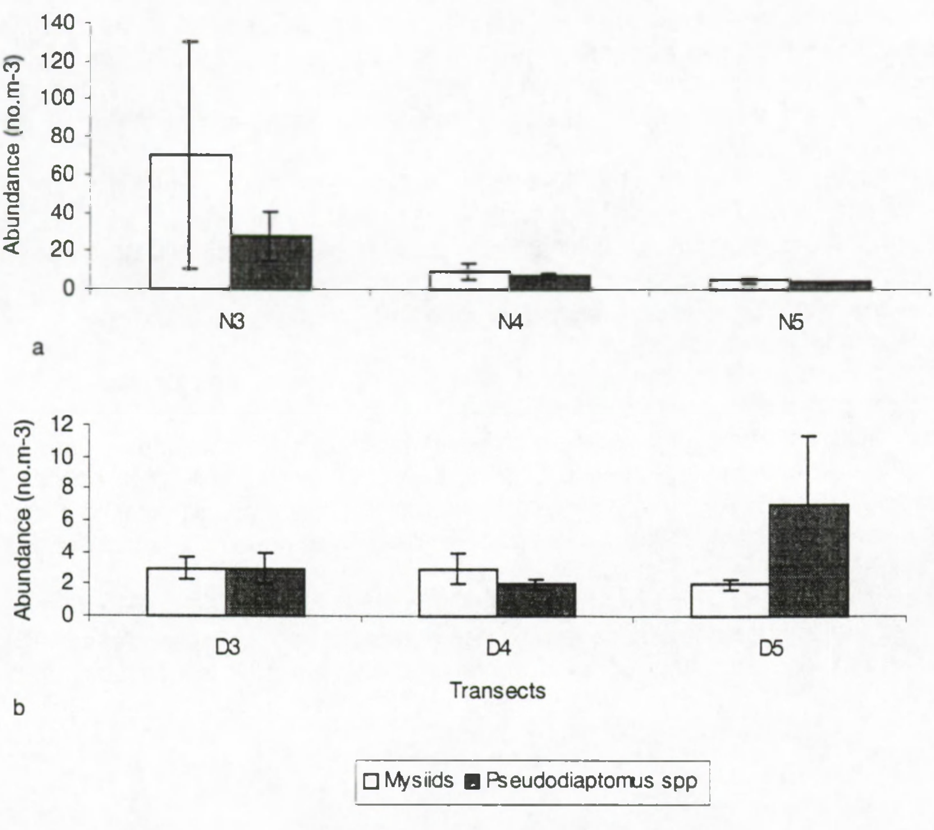


Fig. 7. Mean abundance of Mysids and Pseudodiaptomus *spp at Nyali (a) and Diani Beach (b).*

In both beaches (Nyali & Diani) the occurrence of *Pseudodiaptomus* spp and Mysiids throughout the year though varying in abundance signified fresh water influx. This was further shown by the consequent increase in abundance of the two species during the rainy season (figure

6a) in November-December 1997. The distribution of these organisms was only localised to areas of fresh water influx. Density of indicator species in Nyali occurred in the beach and mid- lagoon stations due to greater amounts of seepage experienced at the beach (Figure 7a). However in Diani, the situation was reverse. The organisms occurred in higher abundances in stations located further away from the beach (Figure 7b), and may thus have implied the occurrence of groundwater seepage close to the reef. Nyali beach had higher abundances of both Mysiids (70 no.m^{-3}) and *Pseudodiptomus* spp (20 no.m^{-3}) as compared to Diani (Mysiids 3.0 & *Pseudodiaptomus* spp 6.0 no.m^{-3}) though not significantly ($p > 0.05$).

Zooplankton diversities in Nyali and Diani lagoons followed a distinct pattern related to salinity. The beach sites which had lower salinities due to ground water seepage recorded lower diversities as compared to the middle and reef station that had less influence of freshwater influx. Only species able to tolerate the low salinites were able to thrive, hence the low diversities. Similarly during the rainy season, lowest diversities were recorded at Nyali in May with 0.34 and Diani 0.1 was recorded in December.

Conclusions

Phytoplankton species composition, cell abundance and diversity were affected by different levels of anthropogenic input into the two lagoons. The species *Oscillatoria* was associated with high nutrients and was abundant in nearshore stations receiving higher nutrients. Though phytoplankton cell abundance was more or less similar in both lagoons during wet and dry seasons, a gradient in abundance was observed from the beach to reef stations with higher values being obtained from the beach and middle stations as compared to the reef which were located away from the beach. Diversity of phytoplankton was higher in the wet months due to increased nutrients enabling more species to thrive.

Chlorophyll-a and B.O.D values were directly influenced by groundwater seepage into the lagoon. Higher values were obtained at nearshore beach stations as compared to the reef stations due higher amounts of nutrients at the beach sites. Nyali lagoon had higher values of Chlorophyll-a possibly linked to slightly higher nutrient input in Nyali as compared to Diani. Increased nutrients through groundwater and runoff during the wet season were responsible for higher Chlorophyll-a and B.O.D levels as compared to the dry.

The effects of ground water into the lagoons of Nyali and Diani was reflected in zooplankton species composition . The presence of "indicator species" throughout the year and at particular spots was indicative of groundwater discharge. The higher numbers of indicator species observed near the beach was linked to higher discharge rates near the beach as compared to the reef . In Diani, the reverse was true. The possibility of groundwater seepage occurring at the reef in Diani could be subject for further investigation

The effects of ground water discharge on zooplankton community and particularly the indicator species was magnified during the rainy season (May - June & November - December). This was mainly associated with increased volumes of fresh water during this season. Greater abundance of zooplankton observed at Nyali beach as compared to Diani could be reflected to the higher amount of nutrients being received in Nyali as compared to Diani lagoon (Mwashote and Kazungu,1998). The higher productivity in the middle parts of both Nyali and Diani lagoons could be due associated to higher nutrients and POM in this area.

High density of fish eggs and larvae occurring at stations N4 &5 and D4 &5 (mid lagoon) in Nyali and Diani reflected the importance of these lagoons as nursery grounds. These areas which also had seagrass cover provided good habitats for meroplankters like; amphipods., isopods, squid larvae and juvenile fish.

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b) Influence on Macroalgal Communities

Introduction

The importance of groundwater resources in coastal ecosystems has long been recognized. Along the East African coastline, there has been concern about the effect of nutrient inputs into these groundwater systems due to poor effluent disposal systems. This concern led to the formulation of a study to evaluate the impacts that nutrient inputs may have on the lagoons that border the East African coast. The survey of macroalgae, in coastal lagoons in Kenya, forms part of a multi-disciplinary project aimed at investigating the effects of groundwater influx on the submerged aquatic vegetation found in the lagoons. Macroalgae have been used worldwide as pollution indicators as there is usually an excessive growth of algae in response to sewage effluents. Borowitzka (1972) reported that studies based on benthos tend to reveal the cumulative effects of pollution and as algae are sedentary they tend to integrate the effects of long term exposure to adverse conditions. Pollution has also been cited as a cause for the reduction in the number of species in polluted areas particularly those belonging to the Phaeophyta and Rhodophyta groups (Hardy *et al.* 1983). Hence the overall objective of this study was to evaluate the species composition associated with the presence of ground water outflow in two Kenyan lagoon ecosystems.

Objectives

1. To compare the benthic algae in two Kenyan lagoons in order to evaluate the effect of anthropogenic inputs via the ground water outflow on the species composition and biomass of different algae.
2. To compare the epiphytic load on the seagrasses in two Kenyan lagoons

Materials and methods

The study was conducted in Nyali and Diani, two lagoons influenced by different levels of groundwater outflow. Three permanent line transects were established, perpendicular to the beach in the two areas and sampling was undertaken during the low spring tide periods. In Nyali, the transects were located at the Army Barracks (Transect NA), at the Nyali Beach Hotel (Transect NB) and at the rocky outcrop of Ras Iwatine (Transect NC). In Diani the transects were located at the Ali Barbour Restaurant (Transect DA), at the University of Nairobi Moana Field Station (Transect DB) and at the rocky outcrop at Robinsons Hotel (Transect DC). Replicate quadrats were laid at the beach, middle and reef zones of each transect.

In June 1998, one additional transect was surveyed in Jomo Kenyatta Public Beach. The survey of macroalgae included the free growing macroalgae as well as the epiphytic algae that grow attached on the stems of the seagrass *Thalassodendron ciliatum* (Forsk.) den Hartog . *T. ciliatum* has a stem which functions as an ideal substrate for epiphytic attachment. Hence, this species was chosen for the survey of epiphytic loading.

In the case of the free growing macroalgae, percentage cover of algae in the field was obtained using a 0.25m^2 quadrat. Algae were cropped for biomass estimates from an area of 0.0625m^2 placed in plastic bags and transported to the laboratory. In the laboratory, the algae were sorted into species and dried at 80°C for 5 days to obtain the dry weight biomass of the different species.

Additionally, the line intercept transect method, that is described by English *et al.* (1977), was used to evaluate algal composition in transects NC, DC and one transect in Jomo Kenyatta Beach. A total of 145 m, from the high water mark at the beach, were surveyed in each transect. Transects NC and DC had point based groundwater seepage through holes opening on coral limestone rocks (Kitheka, 1999). This evaluation was conducted at these sites to give an overview of the algae found in the beach end, as these areas were most impacted by groundwater outflows.

The epiphytic load, in this study, was considered to be an estimate of the dry weight of epiphytes per sampling area. All the epiphytes in a quadrat (sampling area) of 0.0625 m^2 were lumped together to represent the epiphytic load as the individual epiphytic species were too small to weigh adequately. Once the sample was collected from the field it was transported to the laboratory. In the laboratory, the seagrass (in this case *T. ciliatum*) and epiphytes were separated and dried for 5 days at 80°C. The epiphyte/seagrass ratio was calculated using the method outlined by Neveraukas (1987). *T. ciliatum* was found in the middle sections of the transects hence the data collected for this parameter is based on samples obtained from the middle of each transect.

Sampling was conducted monthly from May 1997 to June 1998. However, Diani was not sampled in January 1998, March 1998 and May, 1998 due to unavoidable circumstances.

Statistical analysis

The biomass data was log transformed before statistical analysis was undertaken. The data was then analyzed using the Two-Factor Analysis of Variance (ANOVA) with replication. The data collected in the transects of the two lagoons were considered to be the replicates of each lagoon.

Results

Species composition of marine flora in Nyali, and Diani and Jomo Kenyatta Beach.The species distribution of Chlorophyta (Green algae) in the Nyali and Diani transects is shown in Table 3. This table represents the overall species found through out the study period.

Table 3. The species composition of Chlorophyta in the study areas

	NYALI			DIANI		
TRANSECTS	NA	NB	NC	DA	DB	DC
SPECIES						
Ulva pertusa Kjellman*	*M*R	BR	B		B	BR
Ulva reticulata Forskaal	B*M*R	B*M*R	B*M*R			B
Ulva fasciata Delile		B	B*M*			B
Ulva rigida C. Ag. F.		B	B			B
Ulva lactuca					B	
Ulva pulchra Jaasund	R	R	B			
Enteromorpha prolifera	R					B
Enteromorpha flexuosa			B		B	B
Enteromorpha kylnii Bliding sensu Dawson		B	B		B	
Chaetomorpha crassa (Ag.) Kutz*	BR	B	BR			B*M*
Chaetomorpha indica Kutz		B	B		B	
Caulerpa lentillifera J. Ag.	R					
Caulerpa taxifolia (Vahl) C. Agardh						M
Caulerpa sertularioides (Gmelin) Howe	R					
Halimeda opuntia (L.) Lamouroux	BR	R	BR	*M*	B*M*R	B*M*R
Halimeda tuna (Ellis & Sol.) Lamouroux	R	R		*M*R	*M*R	R
Halimeda renschii Hauck		R	M			
Halimeda incrassata	R	R		*M*R	*M*	
Halimeda macroloba Decaisne	R	R		R		
Codium dwarkense Borgesen		R	R	*M*R	R	R
Codium arabicum Kutzing					R	
Dictyospheria carvenosa (Forskal) Borgesen	R	R		R*M*	R	R
Neomeris van bosse Howe		R				R
Valonia sp.					R	
Anadyomene wrightii Gray*	R	R	B*M*R			
Udotea palmetta Decaisne	R			B	R	
Microdictyon montagnei Harvey			*M*R			
Rhizoclonium grande Borgesen		B				
Boodlea composita (Harvey) Brand*		R				
Cladophora mauritana Kutzing			B			

Legend: **B**: Beach sites R: Reef sites *M*: Middle sites * Species found as epiphytes.

Ulva spp. occurred in all the Nyali transects and extended to the reef sites in all the transects. *Enteromorpha prolifera* and *Chaetomorpha crassa* was also found in the reef in Nyali. In Diani, the there was only one occurrence of *Ulva pertusa* at the reef in transect DC. However, all the Ulva spp. were confined to the beach sites of the transects. The middle zones in Nyali did not have as much *Halimeda* sp compared to Diani. However, the reef zones in Nyali, especially transect NB, had a variety of *Halimeda* spp.

The Rhodophytes (Red algae) were widespread in both Nyali and Diani (Table 4). *Amphiroa fragillisima* was the most common red algae as it occurred in all transects at the all the sampling sites. Some red algae like *Gelidiella acerosa*, *Chondria collisina*, *Galaxura tenera*, *Amansia glomerata* and *Champia* sp. were confined to the beach sites in the study area.

Table 4. The species composition of Rhodophyta in the study areas

	NYALI			DIANI		
TRANSECTS	**NA**	**NB**	**NC**	**DA**	**DB**	**DC**
SPECIES						
Hypnea cornuta (Lamour.) J. Agardh*	**B**R	**B**R	**B**R	R	**B**	**B***M*R
Hypnea nidifica J. Agardh		**B**	**B**		R	
Hypnea nidulans Setchell			**B**		R	
Hypnea hamulosa (Turn.) Montagne					R	
Amphiroa rigida Lamouroux	**B**R	R	R	**B***M*R	R	*M*R
Amphiroa fragilissima (L.) Lamouroux*	**B***M*R	*M*R	**B***M*R	**B***M*R	**B***M*R	**B***M*R
Amphiora ancepes Lamouroux*	R	R	*M*R			***M***
Ceramium sp.*		**B**	**B**	***M***		**B**
Ceramium brevizonatum Petersen						**B**
Laurencia papillosa. (Forsk.) Greville	R			R		**B**
Gracilaria sp.	R		*M*B		R	
Gracilaria corticata J. Agardh*	***M***	***M***	**B***M*R			***M***
Gracilaria edulis (J. Ag.) Silva			**B**			
Gracilaria salicornia (J. Ag.) Dawson	R	**B**				*BR*
Jania adherens Lamouroux*	**B***M*R	*M*R	***M*R**	**B***M*	**B**	*M*R
Gelidiella acerosa (Forsk.) Feldmann et Hamel				R	R	R
Gelidiella myrioclada (Borgs.) Feldmann et Hamel						
Gelidiopsis intricata (Ag.) Vickers*			R	**B**R		***M***
Chondria sp.	R		**B**R	***M***	R	**B**
Chondria collisina Howe		R		R	R	R
Spyrida filamentosa (Wulfen) Harvey					R	
Galaxura tenera Kjellman				R		R
Amansia glomerata C. Agardh		R	R	R		R
Amansia dietrichiana Grunow						R
Desmia pulvinata J. Agardh		**B**R	***M***			
Gelidium sp.		**B**R				
Wurdamania minata (Draparnaud) Feldmann et Hamel	**B***M*	***M***	***M*R**	**B**	R	***M***
Galaxura sp.						
Leveilla jungermannioides (Mart. & Her.) Harvey*	R		R			
Centroceras sp.		**B**	*B*			**B**
Haliptylon subulata (Ell. Et Sol.) Johansen*	***M***	*M*	*M*R	**B**M		***M***
Sarcoidea montagnena J. Agardh						**B**
Champia sp.	R	R			R	
Actinotrichia fragilis (Forsk) Borgesen				*M*R		R

Legend: **B**: Beach sites *M*: Middle sites R: Reef sites * Species found as epiphytes

The Phaeophyta (Brown algae) were confined to the reef and middle sites in Nyali whereas in Diani they were found at the beach sites in some instances. Nevertheless, there were more species present in Diani compared to Nyali (Table 5). The brown algae *Sphacelaria furcigera* was epiphytic on *T. ciliatum* stems.

Table 5. The species composition of Phaeophyta in the study areas

	NYALI			DIANI		
TRANSECTS	**NA**	**NB**	**NC**	**DA**	**DB**	**DC**
SPECIES						
Cystoseria myrica (Gemlin) C. Agardh					**B**R	
Padina gymnospora (Kutz.) Vickers		R			*M*R	
Padina boryana Thivy	R	R	R	*M*R	R	R
Dictyota adnata Zanardini sensu Web. V. Bosse*	*M*R	*M*R	*M*R	**B***M*R	R	R
Turbinaria ornata (Turn.) J. Agardh		R	R	R	R	*M*R
Turbinaria kenyaensis Taylor					R	
Turbinaria tanzanensis Jassund					R	
Turbinaria crateriformis Taylor				R	R	
Sargassum sp.		R	R	R	**B**R	*M*R
Sargassum ilicifolium (Turn.) J. Agardh		R	*M*	R	R	R
Sargassum binderi Sonder		R				
Sphacelaria furcigera Kutzing*	*M*	*M*	*M*	*M*	*M*	*M*
Hormophysa triquetra (L.) Kutzing				R		

Legend: **B**: Beach sites *M*: Middle sites R: Reef sites * Species found as epiphytes

The most common seagrass in the study areas was *Thalassia hemprichii* followed by *Thalassodendron ciliatum* (Table 6).

Table 6. The species composition of Seagrasses in the study areas

	NYALI			DIANI		
TRANSECTS	**NA**	**NB**	**NC**	**DA**	**DB**	**DC**
SPECIES						
Thalassia hemprichii (Ehrenb.) Aschers	**B***M*R	*M*R	**B***M*R	**B***M*R	**B***M*R	**B***M*R
Cymodocea rotundata (Ehrenb.) Hempr. Ex Aschers	**B**	*M*	**B**	**B**	**B**	**B**
Cymodocea serrulata (R. Br.)Aschers & Magnus	**B***M*	*M*	**B**R	**B***M*		
Thalassodendron ciliatum (Forsk.) den Hartlg	**B***M*R	*M*R	**B***M*R	**B***M*	**B***M*	*M*R
Halophila stipulacea (Forsk.) Aschers				**B***M*	**B**	**B***M*
Halophila ovalis (R. Br.) Hook. f.			**B**R	**B**		
Halodule wrightii (Aschers)	**B**	**B**	**B**R	**B**	**B**	**B**R
Halodule uninervis (Forsk.) Aschers	**B**R	**B***M*	**B**	**B***M*	**B**	**B**R
Syringodium isoetifolium (Aschers.) Dandy	**B**	**B***M*		**B***M*	**B***M*	**B**

Legend: **B**: Beach sites *M*: Middle sites R: Reef sites

However, other seagrassses occurred in different sites in the study areas. The macro-epiphytes found on the stems of *T. ciliatum* were *Ulva pertusa, Chaetomorpha crassa, Anadyomene wrightii, Boodlea composita, Hypnea cornuta, Amphiroa spp., Ceramium spp., Gracilaria corticata, Jania adherens, Gelidiopsisi intracata, Leveilla jungermannioides, Haliptylon subulata, Sphacelaria rigudula* and *Dictyota adherens.* Some of these species also occurred as free growing macroalgae in the study area.

Jomo Kenyatta Beach had *Halimeda opuntia* and *Hypnea cornuta* at the beach sites. However, there were no dense algal accumulations at the reef sites. Only small quantities of turf algae were found covering the reef in this particular transect. The seagrasses *Cymodocea rotundata* and *Thalassia hemprichii* was found at the beach, middle and reef sites of the transect studied. *Thalassodendron ciliatum* was found in the middle sites of this transect.

Spatial and temporal distribution of macroalgal species in Nyali and Diani

A comparison of green algae at the beach sites in both Nyali and Diani revealed that the overall biomass of green algae was higher in Nyali (Figs. 1a & 1b). There was a statistically significant difference between the sites (Table 7) in terms of green algae at the beach sites. The highest biomass was recorded in transect NB in Nyali in July 1997, which was a rainy month and this was contributed by the calcareous green algae *H. renschii.* In Diani, the peak biomass was recorded in transect DB in October 1997, which was a dry month (Fig. 1b). This peak was due to the dominance of *Ulva pertusa*. However, there was no significant seasonality in the two areas over the study period. Green algae at the reef sites were also significantly higher in Nyali (Table 7)

Fig. 1a: Green algae in beach sites in Nyali

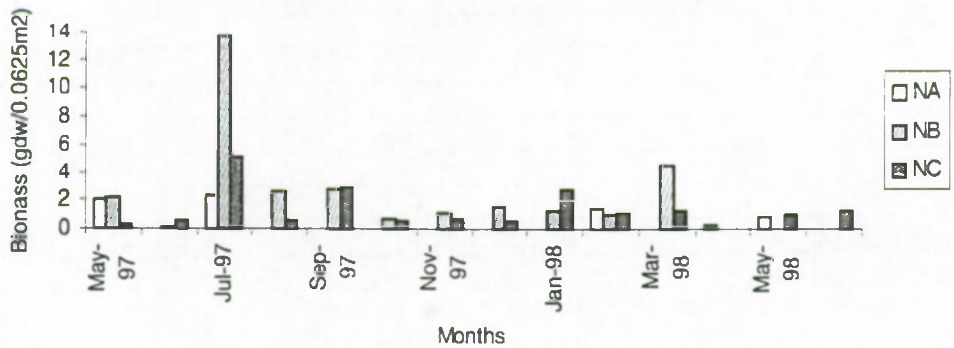


Fig. 1b: Green algae at beach sites in Diani

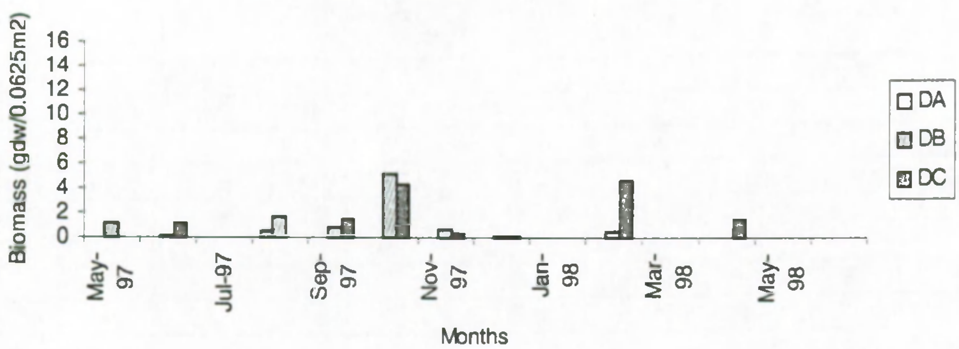


Table 7. Summary of the ANOVA analysis

VARIABLE	Mean Biomass ± SEM		Time		Sites		Interaction	
	Nyali	Diani	F	P	F	P	F	P
Green algae (Beach)	1.41 ± 0.36	0.76 ± 0.24	1.48	0.15	9.51	0.00*	2.25	0.02*
Green algae (Reef)	2.81 ± 1.40	0.57 ± 0.30	1.06	0.40	6.69	0.01*	1.21	0.29
Red algae (Beach)	1.45 ± 0.36	0.70 ± 0.24	0.85	0.59	5.23	0.03*	1.55	0.13
Red algae (Reef)	1.68 ± 0.48	0.47 ± 0.11	0.98	0.49	9.02	0.00*	1.05	0.42
Brown algae (Reef)	0.28± 0.15	2.38 ± 0.65	2.53	0.00*	21.65	0.00*	3.00	0.00*
E:L ratio	0.13 ± 0.04	0.09 ± 0.03	2.36	0.01*	2.13	0.15	0.70	0.75

(* significance at p ≤ 0.05)

with peaks recorded in September 1997 at transect NA (Fig. 2a). This was contributed by the calcareous green algae *H. opuntia.* In Diani the peak was earlier in the year, June, 1997 which was a rainy month, and this was also contributed by the calcareous algae *Halimeda opuntia* in the transect DC (Fig. 2b).

Fig. 2a: Green algae at reef sites in Nyali

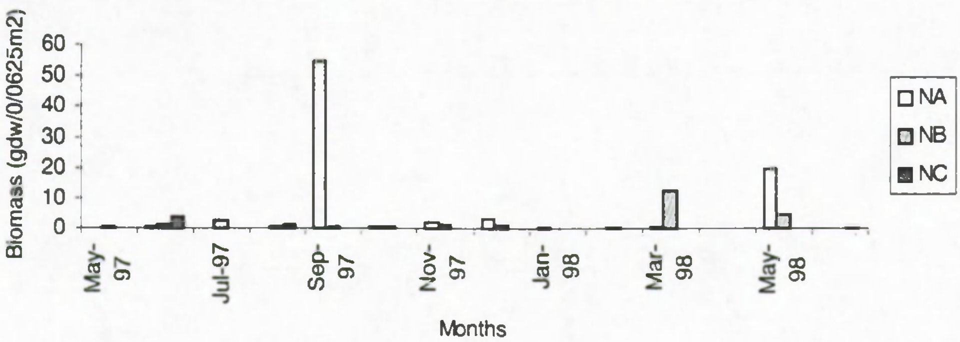
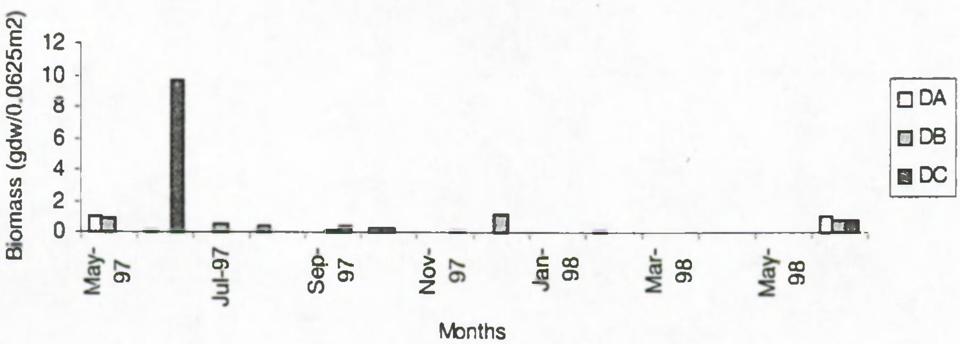


Fig. 2b: Green algae at reef sites in Diani



Red algae at the beach sites also varied significantly in the two study areas (Table 5). Denser accumulations of red algae were found in the beach zones of the Nyali transects particularly in Transect NB (Fig. 3a) whereas in Diani, Transect DC had the highest amount of red algae (Fig. 3b). The red algae at the beach sites were not affected significantly by the seasonal variations (Table 7). Red algae at the reef sites were significantly higher in Nyali compared to Diani (Table 7). The red algae at the reef sites in Nyali were higher in biomass compared to Diani. (Fig. 4a &Fig. 4b). In Diani, different transects had different quantities of red algae, with the highest biomass recorded in June 1998 in transect DB in Diani (Fig. 4b).

In contrast to the green and red algae, brown algae were dominant at the reef site. The biomass of brown algae recorded in Diani was significantly higher than that recorded in Nyali (Table 7). There was a peak of the brown algae in Diani in February in Transect DC and this was contributed by *Padina gymnospora* (Fig. 5a & Fig. 5b). The brown algae were the only group of algae that was influenced significantly by seasonality (Table 7).

The results of the line intercept transect method reveal that Nyali had the highest number of algae at the beach sites compared to Diani and Jomo Kenyatta Beach (Fig 6). The abundance of *Ulva* spp. was higher at the beach in Nyali compared to the other areas.

The epiphyte:seagrass ratio.

The epiphyte:seagrass ratio is shown in Figures 6a and 6b. In Nyali, transect NC had the highest ratio over the wet months of December, 1997, to January 1998 (Fig 7a & 7b) whereas in Diani transect DA had the highest E:L ratio followed by transects DC. There was a significant variation with the seasons (Table 7).This seasonality variation is clearly seen when the highest E:L ratio was recorded between November, 1997 and January, 1997 which were rainy months in both study areas.

Fig. 3a: Red algae at beach sites in Nyali

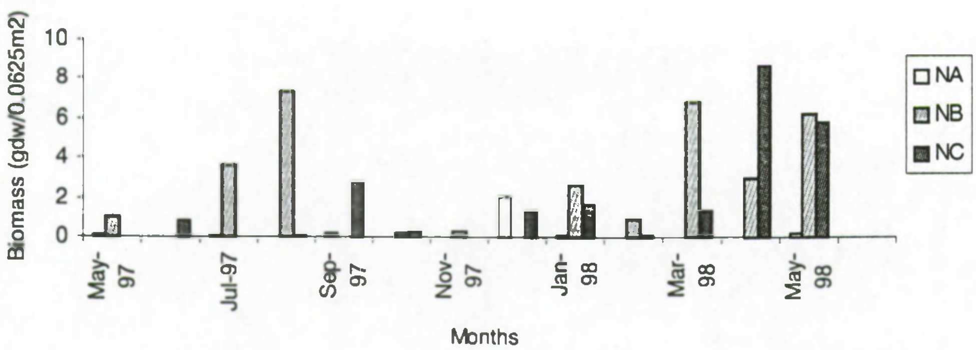


Fig. 3b: Red algae at beach sites in Diani

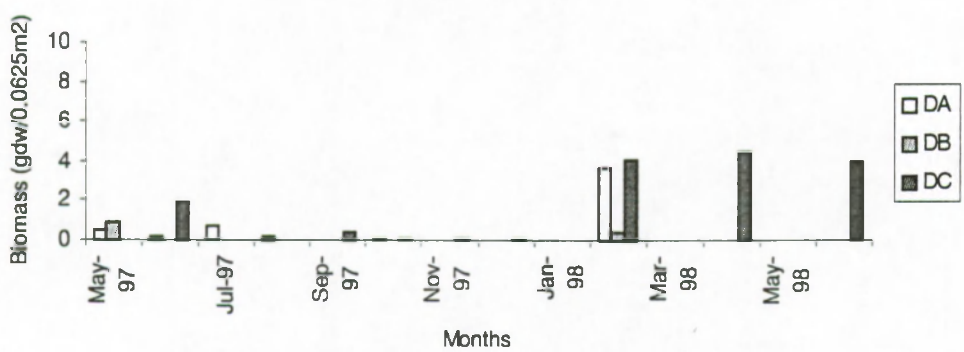


Fig. 4a: Red algae at reef sites in Nyali

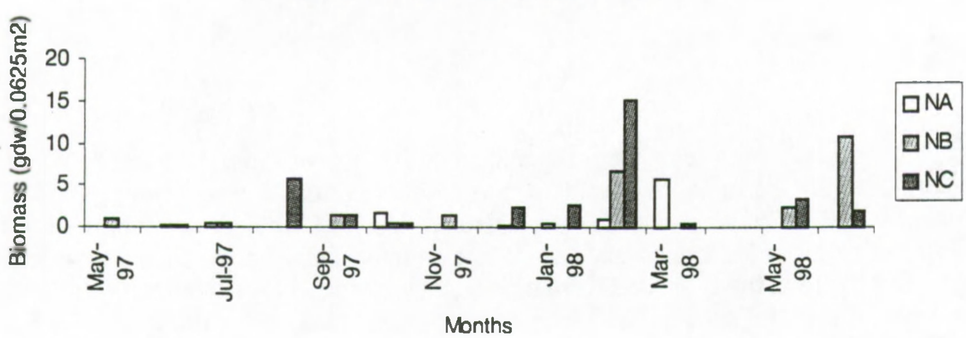


Fig. 4b: Red algae at reef sites in Diani

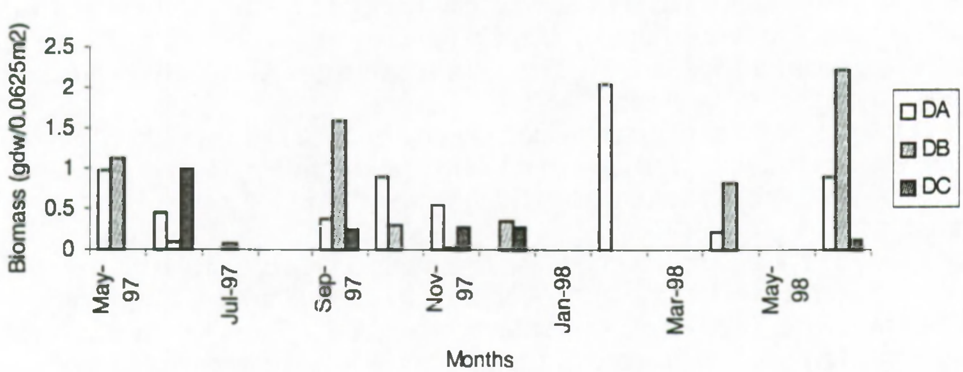


Fig. 5a: Brown algae at reef sites in Nyali

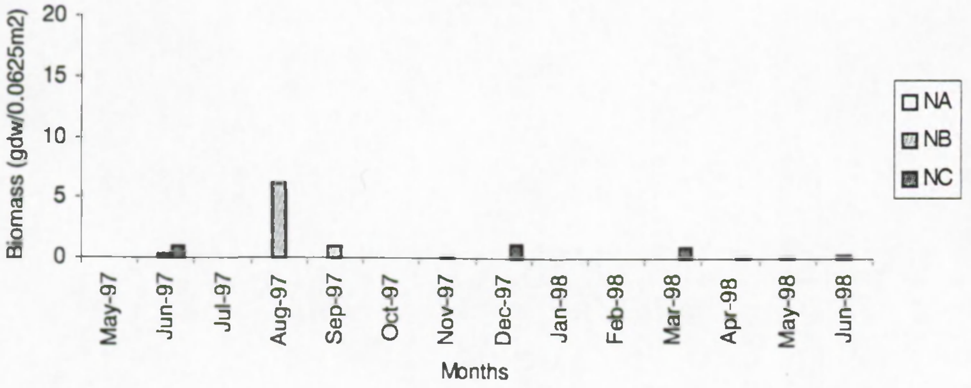


Fig. 5b: Brown algae at reef sites in Diani

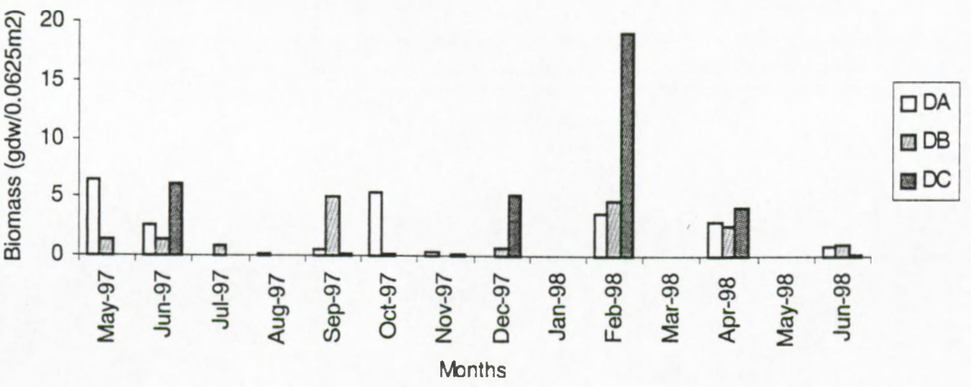


Fig. 6: Cover of macroalgae in the most impacted transects of the study areas

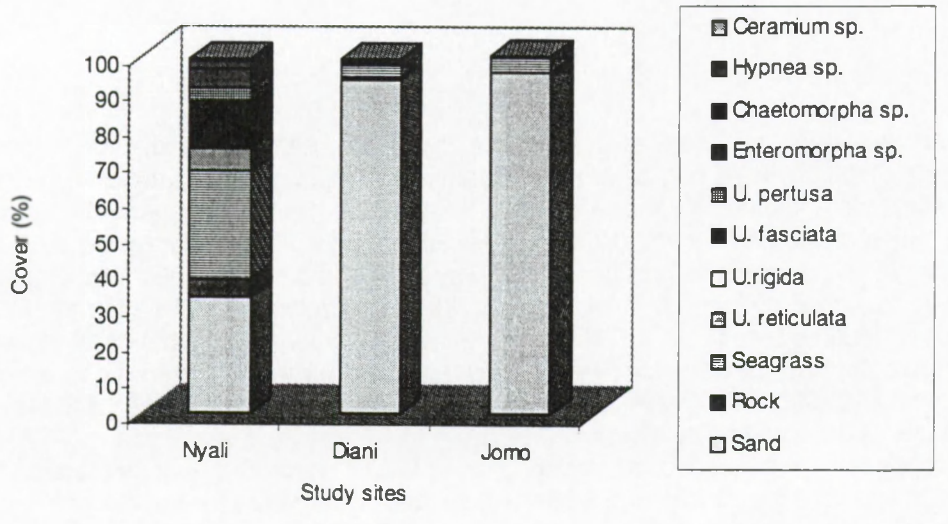


Fig. 7a: Epiphyte:Leaf ratio in Nyali

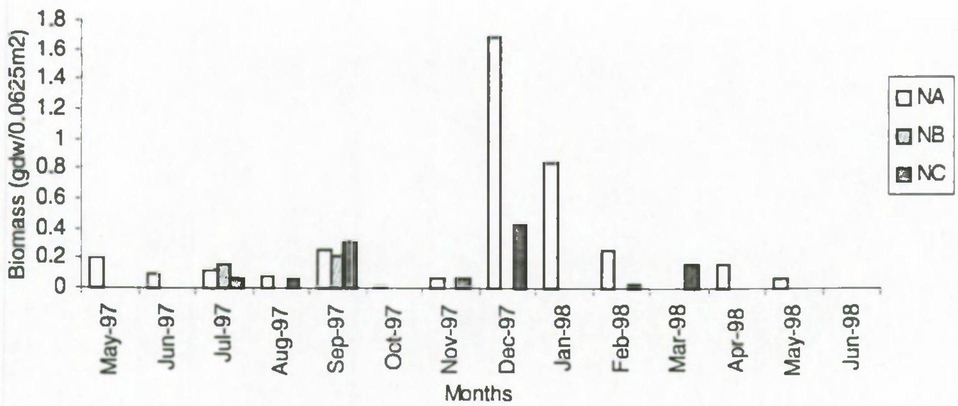
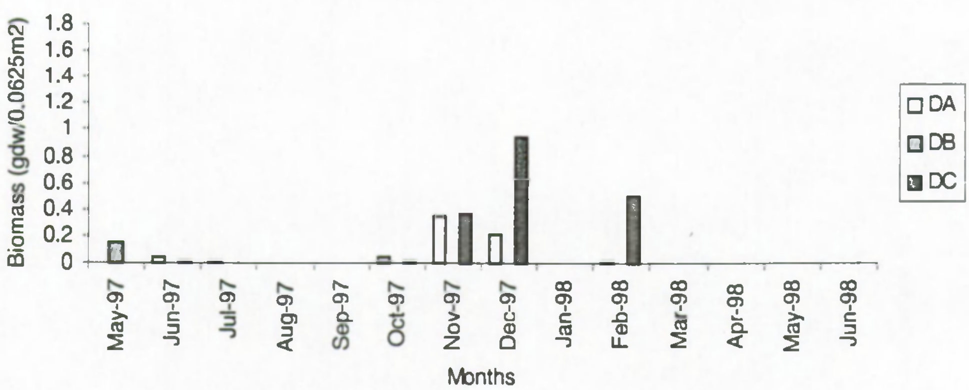


Fig. 7b: Epiphyte:Leaf ratio in Diani



The epiphytic load in Nyali was generally higher than that found in Diani especially during the month of December. However, statistically there was no significant difference between the two sites.

Discussion and conclusions

Previous studies along the Kenyan coast have indicated seasonality patterns in the distribution of algae. Chlorophyta (Green algae) and Cyanophyta (Blue-green algae) have been found to be abundant at the end of the NE monsoon while Phaeophyta (Brown algae) are abundant at the end of the SE monsoon (Moorjani, 1977). Most Rhodophytes (Red algae) are found throughout the year. In this study seasonality patterns were only seen with the brown algae. However the peak in abundance especially in Diani was in the NE monsoon rather than in the SE monsoon as had been found by Moorjani (1977). Nevertheless, in Diani the distribution of algae was found to be consistent with previous studies conducted in the area where brown algae were found to be dominant (Uku, 1996). The red algae were found throughout the study period and they conformed to the patterns documented by Moorjani (1977).

Brown macroalgae have been documented to be the climax flora of East African reefs and large canopy forming brown algae such as *Sargassum* sp and *Turbinaria* sp. are the species associated with the climax stage of the algal succession process (McClanahan, 1997). The higher biomass of brown algae at reef sites in Diani suggests that this community has been able to attain the succession climax whereas in Nyali this process appears to have been interfered with. However, there is no background data of work done in Nyali to indicate if the macroalgal composition of the reef sites in this area may have been different in the past.

McClanahan (1997) showed that turf algae dominated reefs with high sea urchin densities. During the survey of Jomo Kenyatta transect, the field observations revealed a total of 17 sea urchins/m at the reef sites whereas in Nyali only 4 sea urchins/m were counted at the reef in transect NB. In Diani, 32 sea urchins/m were estimated in the reef site of transect DB. The herbivory effects of the sea urchins were obvious in Jomo Kenyatta where the reef site, of the transect that was surveyed, was dominated by small turf algae. However, in Diani, where the number of sea urchins was high, brown algae were still abundant. Yet in Nyali the cover of brown algae was low inspite of a lower sea urchin density. This leads to the speculation that other factors such as nutrient loading could be responsible for this. There could be far reaching effects of groundwater towards the reef zones in Nyali. However, the salinity at the middle and reef sites, in both study areas, remained at 35 psu throughout the study period (Kitheka, 1999) indicating the absence of groundwater loading in areas towards the reef especially in Nyali. Thus one can only speculate about possible factors that may contribute to the differences seen in the algal composition in the Nyali reef zones.

The beach zones in Nyali, especially transects NB and NC had persistent quantities of green algae throughout the year unlike the beach zones in Diani. The results of the line intercept transect method show clearly that the number of green algal species in the beach zones of transect NC were higher compared to the similarly impacted transect DC in Diani. *Ulva* spp., which occurred in abundance at the beach sites of transect NC, have been known to be pollution indicators (Edwards, 1972) hence its occurrence could be an indicator of nutrient rich groundwater outflow into the lagoon in from the beach in Nyali.

Salinity levels recorded in the beach sections of the study areas by Kitheka (1999) indicate that the lowest level recorded in Nyali was 5.2 psu in November 1997,after the rains. The highest level was 26.6 psu during the dry months of February to March 1998. In Diani, the salinity was higher with 21.71 psu recorded at the beach during the wet months of November and December, 1997, and 29.4 psu recorded during the dry months. These levels indicate the general salinity levels at the beach sites in Nyali are lower than those in Diani and this may also be a cause for the proliferation of green algae at the beach sites

The epiphytic load did not give distinctly significant differences between Nyali and Diani, however, the trend indicates the occurrence of a slightly higher loading on the *Thalassodendron ciliatum* stems in Nyali especially during the rainy months. This peak in the rainy months suggests the influence of nutrient rich water particularly in Nyali in transect NC where the groundwater outflow was the highest.

In view of the foregoing results of macroalgae, we can conclude that the impacts of groundwater are obvious in Nyali with the proliferation of green algae at the beach sites where the influence of freshwater is high. However the occurrence of green algae at the reef sites in Nyali does not seem to be adequately explained by the salinity levels. However, the high diversity of other algal groups like the red algae indicate that any effects of eutrophication by the groundwater in Nyali and Diani may be termed as mild eutrophication.

Eutrophication is usually marked by the reduction in the number of species found in an area (Borowitzka, 1972) hence if the eutrophication levels were high then the dominance of one group or species would have been obvious in Nyali.

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Netherlands Institute of Ecology
Centre for Estuarine and Coastal Ecology

Groundwater effects on diversity and abundance of lagoonal seagrasses in Kenya and on Zanzibar Island (East Africa)

Pauline Kamermans, Marten A. Hemminga, Miguel A. Mateo, Nuria Marbà, Johan Stapel
Netherlands Institute of Ecology, Yerseke, The Netherlands
Matern Mtolera
Institute of Marine Sciences, Zanzibar Island, Tanzania

Objectives

The second objective of the GROFLO project is to elucidate differences in nearshore community structures and ecosystem functions in relation to groundwater outflow. In March 1997 and February 1998 two seagrass surveys were carried out. In the framework of the GROFLO project, models of groundwater outflow along the coasts of Kenya and Zanzibar Island were constructed (see contribution of VUB in this report). This made it possible to select study sites with contrasting groundwater-outflow rates. The aim of our surveys was to relate the rate of coastal groundwater outwelling to productivity and vitality of lagoonal seagrasses. The objective of the present study is to relate the rate of coastal groundwater outwelling to the abundance and species diversity of lagoonal seagrasses in East Africa. For the dominant species *Thalassodendron ciliatum*, supplemental data on nitrogen content and natural abundance of nitrogen isotopes in leaves were also collected. In addition, leaf-production rates, shoot demography, and flowering frequency of *T. ciliatum* were determined.

Introduction

Submarine groundwater discharge is a widespread phenomenon throughout the world (Johannes 1980). The outflow in coastal areas is driven by the elevation of the land-based groundwater system that causes a positive hydraulic head. Seepage from unconfined aquifers may occur at various locations, ranging from above the water line at low tide, to as far as 14 km from the shore (Johannes 1980), and at depths of more than 30 m (Simmons 1992). The majority of the outflow, however, occurs close to the shore (Johannes, 1980). The flux of groundwater into coastal waters can be substantial. By using the distribution of radium (^{226}Ra) as a tracer of freshwater input to near-shore waters of the South Atlantic Bight, Moore (1996) estimated that groundwater flow was of the order of 40 % of the river flow to the bight.

The two main ecological effects of submarine groundwater discharge are the supply of nutrients to primary producers and the reduction of osmotic stress. The discharged groundwater can be enriched with nutrients from fertilisers, manure and septic tanks that leach into the groundwater on land (Johannes 1980, Capone & Bautista 1985). Several studies indicate a response of primary producers to the elevated supply of nutrients by groundwater. When comparing three sub-estuaries in Waquoit Bay USA, Valiela et al. (1992) observed highest phytoplankton and macroalgal biomass in the estuary receiving the highest nitrate input by groundwater. Paerl (1997) couples the occurrence of harmful algal blooms in coastal waters to increased nitrogen input by submarine groundwater discharge that bypasses estuarine filters.

In rooted vegetation, a close contact with groundwater that percolates through the sediment is expected. Lodge et al. (1989) employed seepage meters and sampled macrophytes in a freshwater lake. They found a positive relation between macrophyte abundance and distribution and groundwater-outflow rates. They suggested enhanced concentrations of nutrients in the pore water as a possible explanation for this observation (Lodge et al. 1989). On its route to, and through, the groundwater table, a large part of the ammonium is nitrified or adsorbed to surfaces of soil particles (De Haan & Zwerman, 1978). Furthermore, phosphate is retained, although the amount depends on the type of soil (De Haan & Zwerman, 1978). These processes result in a predominance of nitrate in groundwater (Johannes 1980, Capone & Bautista 1985). Macrophytes preferentially utilise ammonium (Short & McRoy 1984). Since in most cases the main nutrient supplied by groundwater is nitrate, it can be questioned whether the extra resource is beneficial to macrophytes. There are indications that groundwater input of nitrate can induce elevated nitrate reductase activities in eelgrass (Maier & Pregnall 1990). However, above a certain level, macroalgae more successfully exploit nitrate than eelgrass (Maier & Pregnall 1990). This suggests that chronic nitrate enrichment from groundwater outflow may result in overgrowth by macroalgae and displacement of seagrass.

A few studies have used the natural abundance of ^{15}N as a tracer to link the nitrogen input by groundwater to the nitrogen content of macrophytes. When sampling a salt marsh with spatial variability in groundwater outflow, Page (1995) found highest $\delta^{15}N$ values in porewater and *Salicornia virginica* tissue at the groundwater-outflow locations. Within the salt marsh, Page (1995) found a decrease in *Salicornia virginica* biomass away from locations of groundwater outflow. McClelland & Valiela (1998) demonstrated a strong correlation between $\delta^{15}N$ in groundwater delivered to three estuaries with varying levels of urbanisation and $\delta^{15}N$ values in several species of macroalgae, the cordgrass *Spartina alterniflora* and eelgrass *Zostera marina* collected in these estuaries. Corbett et al. (1999) found a relation between groundwater flux and $\delta^{15}N$ signatures of seagrass and macroalgae in Florida Bay.

In marine environments, groundwater outflow can reduce porewater salinity (Capone & Bautista 1985, Simmons 1992, Matson 1993). Rooted vegetation in the coastal zone shows different salinity optima for growth and survival. These different preferences can have effects on the distribution of salt-marsh plants and mangrove trees (Smart & Barko 1978, Clough 1992). Furthermore, seagrass species respond differently to salinity (McMillan & Moseley 1967, Zieman 1975, Walker 1985, Walker & McComb 1990, Bird et al. 1993, Adams & Bate 1994, Hillman et al. 1995, Kamermans et al 1999). Species diversity may thus be affected by differences in groundwater outflow. Continuous salinity reductions can result in the dominance of species that tolerate lower salinities.

The coastal zone of East Africa supports an increasing number of beach hotels and settlements of the local population. This development affects the groundwater that is discharged into the coastal waters. The widespread practice of waste disposal into pit latrines and larger sewage pits promotes leaching of nutrients into aquifers. Consequently, both indigenous-population growth, and the rapidly expanding tourist industry enhance the release of nutrients into the coastal lagoons. In tropical environments, coastal waters are usually oligotrophic. Under such conditions, discharge of nitrate-rich groundwater can be a major source of nutrients. Several studies showed the importance of groundwater in the nitrogen supply to tropical bays and lagoons (D'Elia et al. 1981, Lewis 1987, Lapointe et al. 1990, Dollar & Atkinson 1992, Matson 1993). Corbett et al. (1999) calculated that nutrient supplied by groundwater into Florida Bay may be as much as provided by surface freshwater sources in the area. Input of these extra nutrients may significantly affect the ecosystem. Observed responses of tropical primary producers include enhanced growth of phytoplankton (Marsh 1977, Lapointe & Clark 1992) and macro-algae (Lapointe & O'Connell 1989).

Not only the quality of the groundwater that flows into East African coastal waters is susceptible to change, the amount of freshwater that is discharged is affected as well. To sustain the freshwater demand, increasing amounts of groundwater are withdrawn in many places. Tack & Polk (1999) developed a model of groundwater outflow along the Kenyan coast and showed that augmented groundwater depletion for domestic and agricultural use can lead to enhanced seawater intrusion. This can have profound effects on coastal ecosystems. For example, Tack & Polk (1999) observed that the distribution pattern of mangrove forests along the Kenyan coast is closely related to groundwater outflow. Reductions in groundwater-outflow rates may endanger the mangroves, because absence of freshwater would lead to destruction of the forests.

In many tropical coastal zones, mangrove forests are bordered by seagrass meadows. This is also the case in East Africa. There, the coast is protected by a nearly continuous fringing reef which lies between 0.5 and 3 km offshore. The bottom of the back reef lagoons is composed of coral sand, vast seagrass meadows and occassional patches of hard coral. The limited exchange of coastal lagoons with the ocean make these environments very suitable for studying effects of groundwater discharge. With its extensive root system seagrass is a likely candidate for possible effects of changes in groundwater discharge. Tack & Polk (1999) suggest that the present complete absence of groundwater flow in the southeast part of the Everglades in Florida, USA can explain the local seagrass die-off. It is unknown in what way the East African seagrass meadows are affected by groundwater outflow. Observations on possible relations between tropical seagrasses and groundwater are very scarce. To our knowledge, the only report in the literature is of Kohout & Kolipinski (1967) who noted that *Halodule wrightii* Aschers. surrounded a submarine groundwater well, while the dominant species *Thalassia testudinum* Banks ex König was absent from these locations.

Materials and methods

Study area

Based on the information provided by the groundwater flow models of Jurgen Tack (see the VUB contribution in this report), 8 back-reef lagoons with different groundwater-outflow rates were selected in southern Kenya and on Zanzibar Island (Fig. 1). At low tide, the depth of the lagoons can range from very shallow with exposed parts to approximately 10 m deep. The tidal range in the area is 4 m at spring tide and 1 m at neap tide. All selected lagoons were situated on the East coast to ensure similar exposure to ocean swell and the prevailing easterly winds. We expected effects of groundwater outflow to be largest for seagrass species which have rhizomes and roots extending deepest into the sediment. Of the 12 species occurring in East Africa, *Enhalus acoroides* (L. f.) Royle and *Thalassodendron ciliatum* (Forsk.) Den Hartog root most deeply (upto 20 cm into the sediment; Den Hartog, 1970). In the East African lagoons, *T. ciliatum* is more common than *E. acoroides*. Therefore, the study area was restricted to the zone in which *T. ciliatum* occurred. As the depth distribution of this species is from mean low-water spring to about 10 m depth, only lagoons with large subtidal areas were selected.

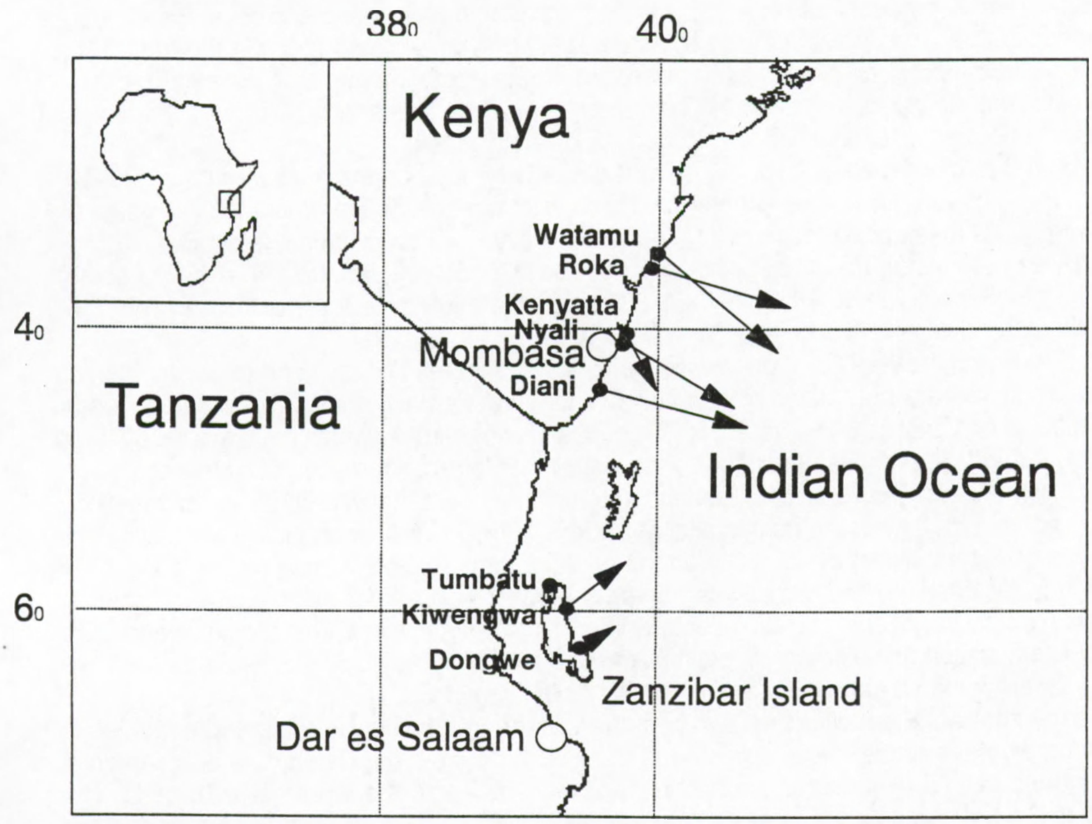

Fig. 1. Location of 8 sampling transects and groundwater flow as calculated by the model. Arrows show the direction of the groundwater flow. The lengths of the arrows represent the size of the flow (for values see Table 1).

Sample collection and processing

For optimal underwater light conditions, fieldwork was conducted before the rainy season which starts in April. In March 1997, 5 lagoons along the southern part of the Kenyan coast were sampled and in February 1998 sampling was carried out in 3 lagoons on Zanzibar Island (Fig. 1). At each site, seagrass sampling was carried out along a 480-m transect. Since depth may influence seagrass species composition, the transects were laid out parallel to the shore at the same depth (1.5 m below 0 Chart Datum). The distance to the beach varied from 100 to 500 m. For each sampling occasion, the actual sampling depth was calculated from tide-table data. Samples were collected SCUBA diving around low-water slack tide. Every 15 m, a picture was taken with a Sea & Sea Motomarine II underwater camera equipped with a flash and a 35-mm lens. The camera was positioned 1.5 m above the bottom and in the same plane as the bottom, to photograph a surface area of 1 m^2. A 1.5-m long stick was used to ensure the right angle and distance to the bottom for each picture. Per transect 33 pictures were taken. Fuji 100 ASA slide film was used in all surveys. For each picture, seagrass species composition and the occurrence of sand, hard and soft coral, rock and benthic macro-algae was noted in the field. Percentage coverage of the individual seagrass species and other components on the photographed plots was later estimated from the pictures. The slides were projected onto paper and the outlines of the different components traced. The surface area of the different patches was determined with an electronic area meter (LI-COR). Hard coral and rock are unsuitable substrates for seagrass growth. In addition, benthic macro-algae and soft corals are attached to hard substrates. Therefore, these components were not included in the calculation of percentage seagrass coverage, i.e. the percentage of the available substrate that is covered by seagrass. The coverage data obtained from analysing the slides were used to calculate the Shannon (or Shannon-Wiener) diversity index (Shannon & Weaver, 1949):

$$-\sum_{i=1}^{s}(p_i)(\log p_i)$$

in which s is the total number of species and p_i the proportion of the total seagrass coverage taken up by the ith species. Even in highly diverse seagrass meadows such as those found in the tropics, the chance to encounter more than 1 seagrass species within the photographed surface area of 1 m^2 is relatively small. Therefore, the mean of the coverage data of all slides with suitable substrate for seagrass growth per transect was used to calculate the diversity index.

Every 120 m along the transect, a 0.06 m^2 frame was placed on the seafloor at the location where a picture had been taken. The number of shoots within the frame were counted per species and all above-ground material enclosed by the frame was harvested for biomass determination. Dry weight (DW) was determined after 2 days drying at 60 °C. The coordinates of the 5 sampling spots per transect were marked with a GARMIN GPS 38 to be able to indicate the exact location of the transect on the groundwater-outflow map.

When studying the effect of groundwater outflow on seagrass diversity and abundance in the field, other factors, that are not related to groundwater, but that can affect diversity and abundance, should not show large variability. The selection of lagoons was carried out in such a way that average light, temperature and water movement conditions were similar for each transect. Although sediments in back reef lagoons usually consist of coral sand, the sediment type showed some variation among transects. Sediment type affects nutrient availability and is known to influence plant productivity and community composition (Giesler et al. 1998). In the present study, grain size and carbonate content were used to characterise the sediment. At the 5 stations per transect where biomass was sampled, sediment was collected by pushing a 50 ml container 5 cm into the sediment. The samples were dried at 60°C prior to estimation of the sediment grain size with a Malvern Particle Sizer. Pieces of shell and coral fragments larger than 1 mm were first removed from the samples.

To detect possible reductions in salinity of the water surrounding the roots and rhizomes of seagrasses, porewater samples were collected at the five same sites per transect. A 50-ml syringe was fit with a modified tip. A series of small holes was punched over the length of a 10-ml tip of a Finn pipette with a hot needle and the end of the tip was closed by melting. This tip was connected to the syringe and porewater was withdrawn over a depth of 10 cm in the sediment. Salinity was determined with a WTW conductivity meter. Salinity of the porewater samples was corrected for the 10 ml of water that was already present in the tip at the sampling time.

To study possible enhanced nutrient availability at groundwater outflow sites, total nitrogen of *T. ciliatum* leaves was measured with a Carlo Erba NA 1500 CN element analyser. The importance of groundwater as a source of nutrient input was studied by determining $\delta^{15}N$ values of *T. ciliatum* leaves with a Finnigan Mat Delta-S isotope-ratio mass spectrometer coupled to a Fisons NA 1500 CN element analyser. The 3rd leaves were cut at the break point with the sheath, and cleaned with paper to remove epiphytes.

Information on methods used to obtain leaf production rates, shoot demography, and flowering frequency for *T. ciliatum* are presented in Annex II.3.

Statistical analysis

The significance of relationships of seagrass variables, porewater salinity and sediment characteristics, with groundwater outflow rates was determined with linear regression analyses (Sokal & Rohlf, 1995). To avoid the issue of pseudoreplication within a transect, regression lines were calculated from mean values per transect, instead of using the individual data obtained on each transect. A significance level of 0.05 was set in all tests. Prior to each analysis, the data were tested for heteroscedacity with a Bartlett's test for homogeneity of variances (Sokal & Rohlf, 1995). Data that scored as significant were log-transformed, which yielded non-significant results in Bartlett's test. Transformation did not improve some data. Therefore, the significance of relationships of these data with groundwater-outflow rates were tested using the non-parametric Kendall's coefficient of rank correlation (τ). For the diversity index only one value was obtained for each transect. As the assumptions of independence and normality could not be tested in this case, Kendall's correlation coefficient was used here as well to determine the significance of the observed relation. The statistical analyses were conducted using the STATISTICA programme (StatSoft Inc., Tulsa, Oklahoma).

Results

A total of 10 seagrass species was observed on all transects together (Table 1). Total seagrass coverage and above-ground biomass did not show significant relationships with groundwater outflow (Table 1; coverage τ = 0.43, p > 0.05 and biomass R^2 = 0.47, p > 0.05). *Thalassodendron ciliatum* occurred on all transects and dominated in coverage on 6 transects. At 3 transects it formed mono-specific meadows of more than 500 m long. The next most frequently observed species were *Thalassia hemprichii* (Ehrenb.) Aschers., *Syringodium isoetifolium* (Aschers.) Dandy, and *Halodule uninervis* (Forsk.) Aschers.. They were found on the same 5 transects and *T. hemprichii* was the dominant species on 1 transect. *Cymodocea serrulata* (R. Br.) Aschers. & Magnus. occurred on 3 transects and had the highest coverage on 1 transect. *Halophila stipulacea* (Forsk.) Aschers. was recorded on 4 of the 8 transects, *Halodule wrightii* and *Cymodocea rotundata* Ehrenb. & Hempr. ex Aschers. were found on 2 transects and *Halophila ovalis* (R. Br.) Hook. F. and *Enhalus acoroides* only on 1 transect. These latter 5 species all attained coverages lower than 10 %. None of the 10 seagrass species were restricted to either groundwater-outflow or non-outflow sites. Thus, the presence of a single species could not be considered indicative of groundwater outflow, or the lack of it. However, low numbers of species were generally found in the presence of high groundwater outflow (Table 1). Furthermore, the coverage was less evenly spread among species at those locations. This is indicated by a significant decrease in species diversity index with an increase in groundwater-outflow rate (Fig. 2). Apparently, not all seagrass species have the same tolerance for conditions caused by groundwater outflow.

Groundwater outflow commonly causes reduced porewater salinity and enhanced nitrogen supply. Indeed, in the present study porewater salinities were significantly lower at locations with high groundwater-outflow rates (Fig. 3). Stable nitrogen isotope signature of the only species present at all transects can give information on the nitrogen source at the different transects. *T. ciliatum* leaves showed a significant increase with increased groundwater-ouflow rates (Fig. 4). This indicates that groundwater was the source of the nitrogen supplied to those plants. However, the nitrogen content of *T. ciliatum* leaves did not show a significant relation with groundwater outflow (Table 1, R^2 = 0.003, p > 0.05). Thus, an indication of increased nitrogen supply at groundwater-outflow sites was not found.

Table 1. Groundwater outflow, seagrass coverage (Watamu n = 33, Diani n = 33, Roka n = 7, Nyali n = 33, Kiwengwa n = 30, Kenyatta n = 33, Dongwe n = 21, Tumbatu n = 33), above-ground biomass (n = 5, except for Roka n = 2 and Dongwe n = 4), sediment median grain size, fraction > 1 mm, carbonate content (n = 5 except for Dongwe n = 4) and total nitrogen content of *T. cilatum* leaves (n = 5, except for Kenyatta and Roka n = 2 and Tumbatu, Kiwengwa and Dongwe n = 3) on the different transects (mean with SE; K = Kenya, Z = Zanzibar Island).

Transect	Watamu (K)	Diani (K)	Roka (K)	Nyali (K)	Kiwengwa (Z)	Kenyatta (K)	Dongwe (Z)	Tumbatu (Z)
outflow (m^2 day^{-1})	17.9	15.8	15.0	12.8	10.2	6.1	2.0	0.4
Thalassodendron ciliatum (%)	87.1 (4.4)	68.0 (7.1)	97.4 (1.5)	98.6 (0.6)	7.6 (3.1)	19.8 (6.3)	32.5 (9.4)	40.2 (6.1)
Thalassia hemprichii (%)		4.2 (2.8)			21.1 (3.1)	7.6 (3.7)	15.4 (3.8)	16.3 (2.5)
Cymodocea serrulata (%)		4.0 (2.4)				51.7 (7.2)	1.0 (1.0)	
Cymodocea rotundata (%)						0.2 (0.2)	11.8 (3.50)	
Syringodium isoetifolium (%)		7.1 (3.3)			0.4 (0.3)	0.7 (0.5)	8.1 (1.9)	4.1 (1.8)
Halodule uninervis (%)		1.9 (1.9)			0.3 (0.2)	0.4 (0.3)	3.0 (1.6)	0.4 (0.4)
Halodule wrightii (%)					0.1 (0.1)	0.2 (0.2)		
Halophila ovalis (%)								0.4 (0.2)
Halophila stipulacea (%)					0.3 (0.1)	0.5 (0.3)	2.3 (0.7)	0.2 (0.2)
Enhalus acoroides (%)					1.8 (1.2)			
total seagrass coverage (%)	87.1 (4.4)	85.2 (4.1)	97.4 (1.5)	98.6 (0.6)	31.6 (5.2)	81.1 (3.8)	74.1 (5.9)	61.6 (4.1)
total seagrass biomass (g DW m^{-2})	457 (61)	430 (67)	644 (55)	604 (44)	115 (43)	223 (63)	224 (102)	222 (91)
median grain size sediment (μm)	218 (3)	251 (36)	292 (4)	229 (9)	115 (13)	151 (9)	121 (44)	230 (13)
coarse fraction sediment (%)	4.6 (1.3)	41.4 (8.9)	48.0 (12.7)	7.5 (2.5)	7.7 (2.3)	9.9 (6.9)	26.6 (10.9)	8.0 (1.5)
$CaCO_3$ sediment (%)	35.1 (2.2)	47.8 (9.3)	89.0 (1.1)	8.5 (2.3)	90.3 (0.6)	61.0 (10.0)	84.0 (1.5)	91.1 (0.9)
N content *T. ciliatum* leaves (%)	1.7 (0.2)	2.0 (0.2)	1.8 (0.1)	1.6 (0.1)	2.1 (0.1)	1.4 (0.1)	2.0 (0.1)	1.8 (0.1)

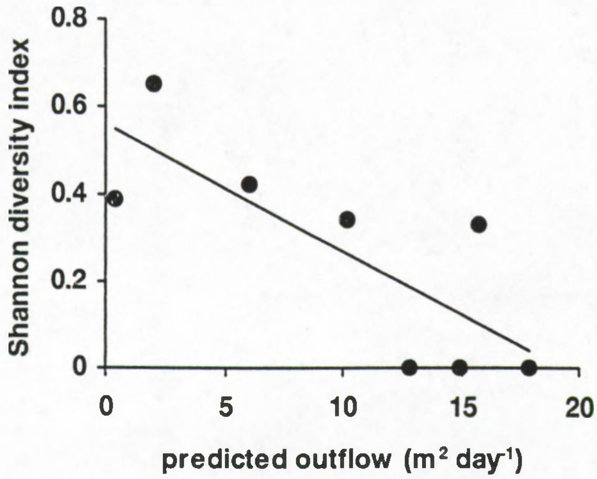

Fig. 2. Groundwater outflow as predicted by the model related to the Shannon diversity index per transect (n = 1). Line of best fit is indicated, τ = -0.64, p = 0.026.

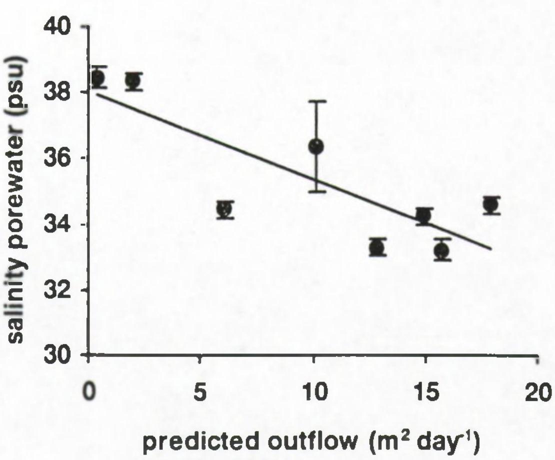

Fig. 3. Groundwater outflow as predicted by the model related to porewater salinity (mean with SE; n = 5 except for Nyali and Dongwe where n = 4). Line of best fit is indicated, τ = -0.57, p = 0.048.

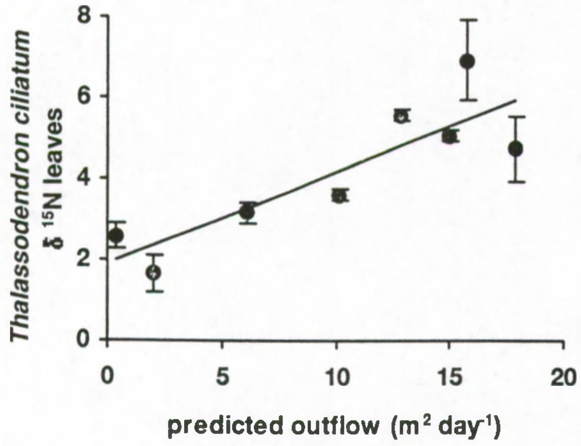

Fig. 4. Abundance of $\delta^{15}N$ in Thalassodendron ciliatum leaves at the different transects. (mean with SE; n = 5, except for Kenyatta and Roka where n = 2 and Tumbatu, Kiwengwa and Dongwe where n = 3). Line of best fit is indicated, $\tau = 0.64$, $p = 0.026$.

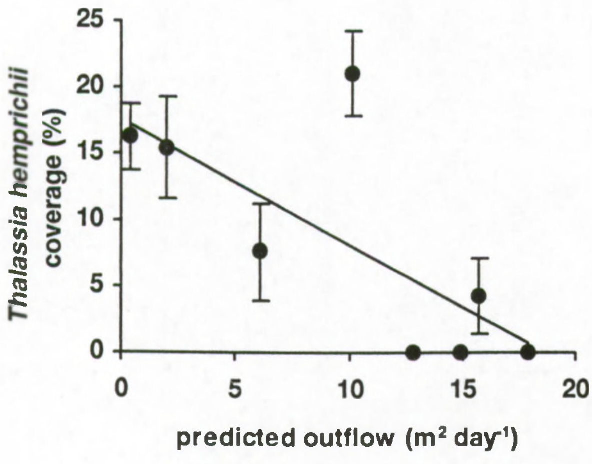

Fig. 5. Groundwater outflow as predicted by the model related to the coverage of Thalassia hemprichii (mean with SE; Watamu n = 33, Roka n = 7, Nyali n = 33, Tumbatu n = 33, Dongwe n = 21, Diani n = 33, Kenyatta n = 33, Kiwengwa n = 30). Line of best fit is indicated, $\tau = -0.57$, $p = 0.049$.

Of all species encountered, only *T. hemprichii* showed a significant relation with groundwater outflow (Fig. 5). Thus, growth and development of *T. hemprichii* may be hampered at locations with high groundwater outflow. A negative correlation was found between coverage of *T. ciliatum* and *T. hemprichii* (Fig. 6). The relation suggests competition between the two species, *T. hemprichii* being the stronger one at sites with low groundwater outflow, while *T. ciliatum* can dominate at sites with high groundwater outflow. There is one transect (Kenyatta) at which the coverages of both species are lower than expected (Fig. 6). At this transect, *C. serrulata* is the dominant species (Table 1). Interactions with this third species may influence competition between the two others.

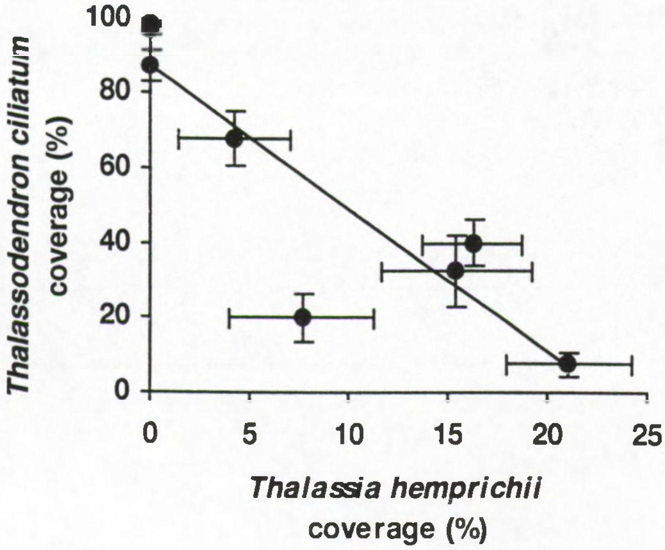

Fig 6. Relation between coverage of Thalassodendron ciliatum and Thalassia hemprichii per transect (mean with SE; Watamu n = 33, Roka n = 7, Nyali n = 33, Tumbatu n = 33, Dongwe n = 21, Diani n = 33, Kenyatta n = 33, Kiwengwa n = 30). Line of best fit is indicated, τ = -0.72, p = 0.013.

Median grain size of the sediment on the different transects varied from approx. 100 µm to approx. 300 µm and did not show a relation with groundwater outflow (Table 1, $\tau = 0.21$, $p > 0.05$). Coarse sediments, with more than 25 % particles larger than 1 mm, were found on 3 transects. These transects did not differ consistently from the other transects in terms of groundwater-outflow rate (Table 1, R^2 of log-transformed data = 0.01, $p > 0.05$). Carbonate content of the sediment was generally high (on 5 transects more than 60%) and did not show a significant relation with groundwater-outflow rate (Table 1, $\tau = -0.50$, $p > 0.05$). Hence, sediment characteristics did not co-vary with predicted groundwater-outflow rates. For the present groundwater study, it is sufficient to note that the unexplained variance of the observed relations between seagrass variables and groundwater-outflow rates may be related to potential effects of sediment type on seagrass diversity and abundance.

T. ciliatum leaf production rates and recruitment and mortality rates did not show any relation with groundwater outflow rates. Those results are presented in Annex II.3.

Discussion

The present study strongly suggests that groundwater outflow influences seagrass species diversity along the East African coast. The influence of groundwater outflow is thus not restricted to mangrove forest on the shore as shown by Tack & Polk (1999), but extends to rooted vegetation in the back-reef lagoons. Sites with high groundwater outflow displayed a lower species diversity than sites with low groundwater outflow. The decline in species diversity indicates a negative effect of groundwater outflow on growth and persistence of some species. A possible factor that negatively influences seagrasses at groundwater-outflow sites is reduced porewater salinity. The present study showed a reduction in porewater salinity of upto 5 psu at groundwater-outflow sites. Little information exists about the relation with salinity in the 10 seagrass species encountered on the transects. Experimental studies, in which exact salinity optima were determined, are lacking for most of these species. In Table 2, the species are divided into 3 groups according to laboratory measurements and data on occurrence in the field. From this separation, one would expect and *Thalassia hemprichii*, *Cymodocea serrulata*, *Halodule wrightii* and *Halophila ovalis* to dominate at locations with little groundwater outflow, while *Cymodocea rotundata* is expected to dominate at groundwater-outflow sites. For *C. rotundata* the expected relation was not observed, as the species was not found at sites with high groundwater-outflow rates. However, *H. wrightii* and *H. ovalis* were found at the four transects with the lowest groundwater outflow. In addition, *T. hemprichii* and *C. serrulata* were absent or did not attain high abundances at the four transects with high groundwater-outflow rates. Thus, some of the observed differences in species diversity between transects may be explained by salinity effects.

Table 2. Salinity tolerance of seagrass species. Source is indicated: (1) Den Hartog 1970, (2) Jagtap 1991, (3) Mitchell 1987, (4) Hillman et al. 1995, (5) Brouns & Heijs 1985, (6) Aioi & Pollard 1993, (7) Erftemeijer & Herman 1994.

Tolerant to low salinity	tolerant to high salinity	no specific tolerance reported
Cymodocea rotundata (1)	*Thalassia hemprichii* (2)	*Thalassodendron ciliatum* (1,5)
	Cymodocea serrulata (1,2)	*Syringodium isoetifolium* (5,6)
	Halodule wrightii (3)	*Halodule uninervis* (1)
	Halophila ovalis (1,4)	*Halophila stipulacea* (1)
		Enhalus acoroides (1, 7)

Nitrogen stable isotope signatures of *Thalassodendron ciliatum* indicate that groundwater is a major source of nitrogen input into the back-reef lagoons. Page (1995) observed that groundwater outflow can cause increased inorganic nitrogen concentrations in the root/rhizome area. Therefore, it is expected that rooting species such as seagrasses will benefit from extra nutrients supplied by groundwater seepage. Due to logistic difficulties, the present study did not include determination of dissolved inorganic nitrogen concentrations in the sediment porewater. However, data on nitrogen content of *T. ciliatum* leaves were obtained during this study. Results did not show an increase in nitrogen content with increasing groundwater-outflow rates. The roots of *T. ciliatum* are stongly lignified and show limited absorption and transport of nutrients, which suggests that their main function is anchorage and support (Barnabas 1991). The existence of limited nutrient transport between below- and above-ground biomass could explain why a relation between groundwater outflow and nitrogen content of the leaves was not found.

Groundwater supplies relatively more nitrate than ammonia (Johannes 1980, Capone & Bautista 1985). Thus, changes in seagrass species diversity at groundwater outflow sites could be a result of the type of nitrogen supplied. Uptake of ammonium is less costly than uptake of nitrate and is thus favoured by seagrasses (Maier & Pregnall 1990, Hemminga et al. 1994). Seagrass species can differ in their capability to sequester nitrate. In order to assimilate nitrate, a plant must first reduce it to ammonium via inducible enzymes. The activity of these enzymes (nitrate reductase activity) can be used as an indicator of nitrate utilization by plants. By using this method, Maier & Pregnall (1990) demonstrated elevated nitrate reductase activities in *Zostera marina* growing at a location with groundwater discharge. On the other hand, Doddema & Howari (1983) showed a very low nitrate reductase activity for *Halophila stipulacea*, which was independent of the ambient nitrate concentration. Low nitrate reductase activities are expected in species occurring at sites with low groundwater-outflow rates, while higher activities may be found in species dominating at sites with high groundwater-outflow rates. Indeed, *H. stipulacea* was found at the four transects with the lowest groundwater outflow. These transects most probably have the lowest nitrate porewater concentrations. Information on the ability for uptake of nitrate are lacking for the other species that were encountered.

In the present study, transects with high and low outflow rates were located at populated sites near villages or beach hotels, and at pristine sites without any habitation. Valiela et al. (1992) showed that the nitrate concentration in groundwater is closely linked to the density of buildings in an area. In addition, McClelland & Valiela (1998) reported highest $\delta^{15}N$ values for *Zostera marina* in the estuary with the highest antropogenic nitrogen input. The increase in proportion of waste-water derived nitrate contributing to the total nitrogen pool is believed to cause the elevated $\delta^{15}N$ values (McClelland et al 1997). Hence, differences in population density between the transects of the present study may affect $\delta^{15}N$ signatures of the seagrass. Indeed, highest $\delta^{15}N$ values were found at Nyali and Diani, two groundwater-outflow locations with a high population density and a large number of hotels. The $\delta^{15}N$ values measured were around 6 ‰, which was similar to the highest value measured in *Zostera marina* by McClelland & Valiela (1998). This suggests that those Kenyan lagoons receive antropogenic nitrogen in comparable inputs as the estuarine system near Cape Cod, USA. The use of nitrogen stable isotope signatures is a promising tool to elucidate the contribution of antropogenic nitrogen sources in the functioning of an ecosystem. Recent developments along the lagoons in East Africa have increased antropogenic nitrogen input considerably. In the lagoonal ecosystem, seagrasses are primary producers with a long life span, that can integrate nitrogen input over long time periods. Thus, a tracer study with seagrasses in this area could provide an early detection method for the level of nutrient enrichment of the lagoons.

The results on coverage of *T. ciliatum* and *T. hemprichii* suggest competition between the two species. *T. ciliatum* dominated at sites with high groundwater outflow, while *T. hemprichii* showed higher coverage at sites with low groundwater outflow. From this observation we hypothesize that reduced salinity and enhanced nutrient supply favour development of *T. ciliatum*, while normal salinity and low nutrient supply give *T. hemprichii* competitive advantage. Long-term experiments of Fourqurean et al (1995) with the closely related species *Thalassia testudinum* support the hypothesis on *T. hemprichii*. The authors demonstrated that *H. wrightii* colonized *T. testidunum*-dominated seagrass beds and out-competed *T. testidunum* when nutrient availability was increased. *T. ciliatum* has a high canopy of more than 50 cm (Kamermans et al submitted). Thus, the species can effectively shade smaller species such as *T. hemprichii*. Comparison of relative growth rates and nutrient contents of different species can give information on their respective nutrient demand (Fourqurean et al 1992). Leaf production rates of *T. ciliatum* in the East African lagoons ranged from 5 to 8 mg DW per shoot per day (Kamermans et al submitted). Using nitrogen content data for the same locations from Table 1, it was calculated that nitrogen demand of *T. ciliatum* ranged from 9 to 17 mg per shoot. Similar data on *T. hemprichii* in the lagoons are not available. Erftemeijer et al (1994) determined a leaf production rate for *T. hemprichii* in Indonesia of 1.7 mg AFDW per shoot per day and a nitrogen content of 2.7 % N. A conversion factor from DW to AFDW of 0.65 was established by Stapel et al (submitted). Thus, leaf production of *T. hemprichii* was 2.6

mg DW per shoot per day and nitrogen demand 7 mg per shoot. This is lower than the values obtained for *T. ciliatum*. The calculations suggest that *T. ciliatum* can only outcompete *T. hemprichii* when nutrient availability is high. The observed reduction in species diversity with increased groundwater outflow may also be explained by competition. With 0.2 g DW per shoot, *T. hemprichii* had a size which was in the range of the other species (0.2 - 0.001 g DW per shoot) except *T. ciliatum* (0.9 g DW per shoot) and *Enhalus acoroides* (9.6 g DW per shoot). Thus, although *T. hemprichii* may have competitive advantage in sequestering nitrogen, it is probably less effective in competing for light. This may prohibit development of large mono-specific meadows such as found for *T. ciliatum*.

All *T. ciliatum* populations in the present study appeared to be either expanding or in steady state (see Annex II.3). This suggests that the environmental quality of the back-reef lagoons is still suitable for seagrass development, which is indicative of a healthy ecosystem. Flowering frequencies were generally low (see Annex II.3). In addition, seedlings were not found in our study. These results indicate that sexual reproduction is of minor importance for the permanently submerged *T. ciliatum* populations, which reduces the ability to adapt to changes. The present study suggests that changes in the amount or quality of groundwater outflow along the East African coast will affect the seagrass species composition in lagoons. Specifically, the occurrence of *T. hemprichii* and *T. ciliatum* are likely to be affected. At present, information on the function of these species in the ecosystem of back-reef lagoons is absent, which impedes predictions of possible consequences.

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Free University
Institute of Environmental Research

Objective of the research

(1) To construct a model of groundwater flow along the Eastern African coast. Existing data on aquifers and the water balance in Eastern Africa coastal areas and hydrological field data will be collected, with the aim to construct a model of groundwater outflow that will allow a prediction of the response of the hydrological system to anthropogenic disturbances.

(2) To detect historical changes in chemical characteristics of molluscs in relation to changes in groundwater flow in recent years.

Groundwater flow in the coastal zone influences mangrove distribution

Tack, J.F., O. Batelaan, F. De Smedt, and P. Polk
Free University, Brussels, Belgium

Abstract

Mangroves are the only protection in the tropics against coastal erosion, are nursery grounds for a variety of marine animals and increase sedimentation of particles brought to the coast by rivers. Mangrove distribution in the tropics and subtropics is often linked with the presence of estuaries and creeks. There is a consensus that the brackish water micro environment is caused by river discharges into the oceans. However all over the world mangrove areas are found where no rivers or estuaries are in the immediate neighbourhood. An explanation is reported here for a number of those exceptions. Using a mathematical groundwater flow model it is shown how human activities as far as several hundreds of kilometres inland can destroy vast areas of mangroves by changing the groundwater flow. The model predicts and/or confirms the destruction of large mangrove areas in Kenya and in Florida (USA).

Introduction

Walsh (1974) suggested the existence of extensive mangal depended upon five basic factors. Chapman (1975, 1977, 1984) believed there are seven: (1) air temperature, (2) ocean currents, (3) protection from wave action, (4) shallow shores, (5) salt water, (6) tidal range, and (7) substrate.

Mangrove distribution in the tropics and subtropics is often linked with the presence of estuaries and creeks (Macnae, 1968; Barth, 1982; Blasco, 1991). There is a consensus that the brackish water micro environment, which is the key factor for the development of mangroves, is caused by river discharges into the oceans (Macnae, 1968; Barth, 1982; Snedaker, 1982).
However all over the world mangrove areas are found where no rivers or estuaries are in the immediate neighbourhood. In this study an explanation is presented for a number of those exceptions making use of a mathematical groundwater flow model. Figure 1 shows the distribution of mangroves along the Kenyan coast. Most of the mangrove areas are in the proximity of one or more rivers. However, the rivers north of Mombasa are perennial while the rivers south of Mombasa are drying up after the rainy season. On the other hand there are a number of mangrove areas with no rivers in the immediate neighbourhood. The mangrove forest of Mida Creek (between Kilifi and Malindi, Kenya) is a clear example of a mangrove forest growing in an area without a visible freshwater source.

For regions whose groundwater flow pattern is not known, the use of mathematical models can be a powerful tool in predicting unknown variables, e.g. the impact of a changing groundwater flow pattern on the mangrove ecosystem. The model used is able to explain mangrove distribution in two different areas (Kenyan coast and the Everglades National Park in Florida (USA)). Also the usefulness of the model in predicting mangrove destruction in the above mentioned mangrove areas is shown.

This study describes the groundwater flow of two different regions by a mathematical groundwater model (Ituli, 1984; Dapaah-Siakwan, 1986). Such a model consists of a set of mathematical differential equations with their boundary and initial conditions, and a numerical solution procedure.

Groundwater flow takes place according the gradient of piezometric heads. This means that by solving the model equation for piezometric heads and by knowing their distribution in the study areas the groundwater flow pattern can be established.

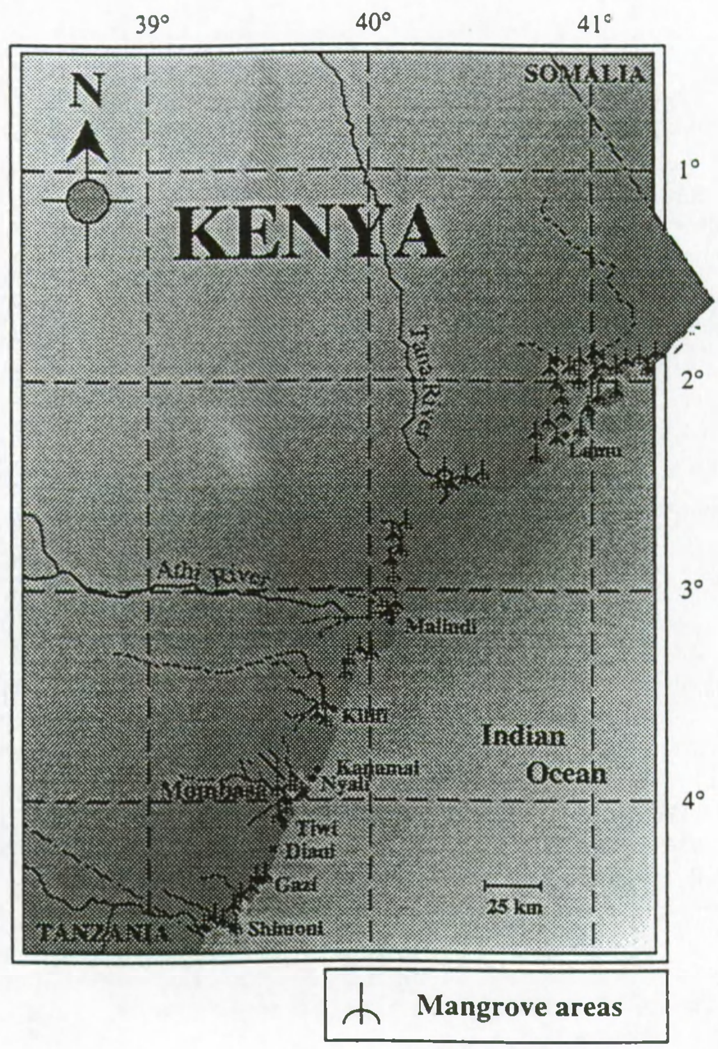

Figure 1. Distribution of mangrove forests along the Kenyan coast.

Material and methods

Groundwater model

The mathematical model used in this study was developed by the Laboratory of Hydrology of Brussels Free University (De Smedt and Bronders, 1985). The model used is designed to simulate the response of a phreatic, semi-confined or confined aquifer to an imposed stress. The model allows homogeneous or heterogeneous aquifers with irregular boundaries. The model also allows constant/variable recharge, constant/variable and surface inflow or outflow. The groundwater flow is considered horizontal in the model. Because of the large surface of the study areas the aquifers vary in geological composition from place to place. This implicates that the transmissivity also varies with the geologic composition of the aquifer.

To describe the groundwater flow through porous media, the model is based on Darcy's Law and the Law of Conservation of Mass. The model considers steady-state two-dimensional, horizontal flow through a non-homogeneous isotropic aquifer of variable thickness, including source and sink terms, as groundwater recharge and withdrawal.

To formulate the governing equation for groundwater flow, the Continuity equation is applied to an elemental volume within the aquifer (Fig. 2):

$$\frac{\partial q_x}{\partial x}\Delta x(d\Delta y)+\frac{\partial q_y}{\partial y}\Delta y(d\Delta x)=R(x,y)\Delta x\Delta y-Q(x,y)\Delta x\Delta y \qquad \text{(equation 1)}$$

where

x and y are the horizontal Cartesian co-ordinates;
q_x is the flux in the x-direction;
q_y is the flux in the y-direction;
d is the thickness of the aquifer;
R(x,y) is the recharge; and
Q(x,y) is the surface outflow.

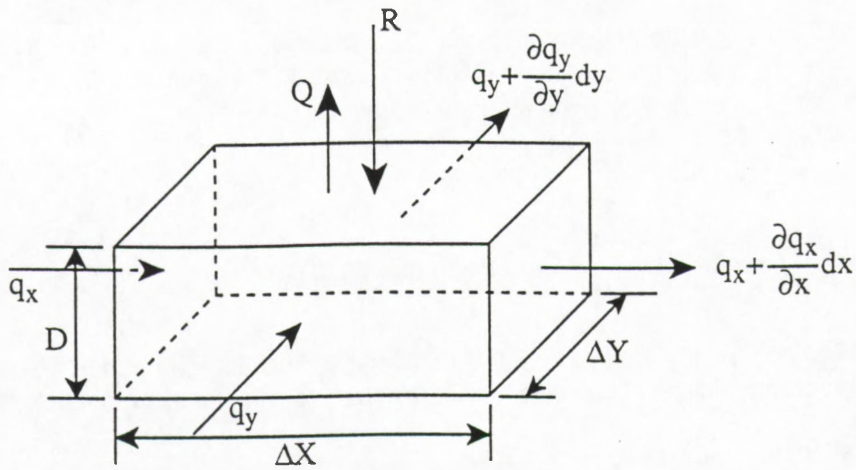

Figure 2. An elemental volume within the aquifer.

The components of the groundwater flux are obtained by Darcy's law:

$$q_x=-K\frac{\partial h}{\partial x} \qquad \text{(equation 2)}$$

$$q_y = -K\frac{\partial h}{\partial y} \qquad \text{(equation 3)}$$

where
K is the hydraulic conductivity; and
h is the piezometric head or groundwater elevation.
Equation 1, after substitution of equitions 2 and 3 and after dividing by DxDy, becomes:

$$\frac{\partial}{\partial x}(-Kd\frac{\partial h}{\partial x}) + \frac{\partial}{\partial y}(-Kd\frac{\partial h}{\partial y}) = R(x,y) - Q(x,y) \qquad \text{(equation 4)}$$

The transmissivity is defined as T=Kd in case of a homogeneous aquifer, or as

$$T = \int_0^d Kdz \qquad \text{(equation 5)}$$

in case of a layered medium, such that equation 4 can be written as:

$$\frac{\partial}{\partial x}(T\frac{\partial h}{\partial x}) + \frac{\partial}{\partial y}(T\frac{\partial h}{\partial y}) + R(x,y) - Q(x,y) = 0 \qquad \text{(equation 6)}$$

In the model recharge and discharge zones are considered:
A. a recharge zone: if the piezometric head h is less than the topographic level h_t, the surface outflow Q(x,y) is taken to be equal to zero:
Q(x,y)= 0 for $h<h_t$
Equation 6 becomes:

$$\frac{\partial}{\partial x}(T\frac{\partial h}{\partial x}) + \frac{\partial}{\partial y}(T\frac{\partial h}{\partial y}) + R(x,y) = 0 \qquad \text{(equation 7)}$$

B. a discharge zone: if the piezometric head h is equal to or tends to be greater than the topographic level h_t, the surface outflow Q(x,y) is taken to be greater than zero, so that the piezometric head becomes equal to the topographic level:
Q(x,y)>0 for $h=h_t$
Equation 6 becomes:

$$\frac{\partial}{\partial x}(T\frac{\partial h_t}{\partial x}) + \frac{\partial}{\partial y}(T\frac{\partial h_t}{\partial y}) + R(x,y) = Q(x,y) \qquad \text{(equation 8)}$$

If surface outflow occurs, the model considers the water to flow downwards to the adjoining areas with lower topographic levels. This water is than added to the normal recharge input in these areas.
The flow equation now becomes:

$$\frac{\partial}{\partial x}(T\frac{\partial h_t}{\partial x}) + \frac{\partial}{\partial y}(T\frac{\partial h_t}{\partial y}) + R(x,y) - Q(x,y) + Q_s(x,y) = 0 \qquad \text{(equation 9)}$$

where Q_s is the surface inflow from the neighbouring areas.

Excessive withdrawals from wells can have adverse effects on the groundwater storage in the aquifer. This effect can be studied when a well, located at a point (x,y), is pumped at a rate of $Q_W(x,y)$ and this withdrawal is incorporated in the flow equation as follows:

$$\frac{\partial}{\partial x}(T\frac{\partial h_t}{\partial x})+\frac{\partial}{\partial y}(T\frac{\partial h_t}{\partial y})+R(x,y)-Q_w(x,y)-Q(x,y)+Q_s(x,y)=0 \qquad \text{(equation 10)}$$

where Q_W is the withdrawal from the well.

In order to solve the groundwater flow equation, we need to specify the boundary and the initial conditions. A groundwater flow domain can be defined by several types of boundary conditions. In the regions studied, use is made of two boundary conditions:

Potential boundary conditions

In this type of boundary conditions the piezometric heads are known: $h=h^*(x,y)$ where h^* is a known piezometric head for all points along the boundary. In the model this type of boundary represents that part of the aquifer where the piezometric head would not change in time. In natural conditions, such boundary conditions occur as recharge boundaries or areas beyond the influence of hydraulic stresses and are defined by known equi-potential lines.

No flow boundary conditions

These boundaries are defined by a line across which no flow is occurring, thus

$$\frac{\partial h}{\partial x}=0 \qquad \text{(equation 11) or}$$

$$\frac{\partial h}{\partial y}=0 \qquad \text{(equation 12)}$$

This means that perpendicular to the boundary no flow is occurring. This kind of boundary can be defined in nature by two situations: the existence of an outcropping of impervious rock, or a groundwater divide.

Initial conditions

The groundwater flow equation describes a steady state situation so no initial condition is required. However, the topographic levels are required to calculate the piezometric heads in the study area, and to identify the discharge and recharge zones.

The central finite difference approximation method is used to solve the partial differential equation describing the groundwater flow. The model equations describing the regional groundwater flow are solved by a computer programme originally written in FORTRAN IV (Ituli, 1984; Dapaah-Siakwan, 1986). For this application it has been rewritten in C^{++}, and also a number of graphic outputs were added.

Input

The basin characteristics that serve as inputs for the computer to solve the model equations are the transmissivity values T, the areal net precipitation R, the topographic levels h_t, the aquifer thickness d and the porosity of the aquifer material n.

The transmissivity values are obtained by the product of hydraulic conductivity and the thickness of the aquifer: T=KD, where K is the hydraulic conductivity and d is the thickness of the aquifer. The flow domains are divided into several zones having different transmissivity values. Those zones correspond to the geological units distinguished in the flow domains. Transmissivity data of Kenya were obtained from Ituli (1984) and are shown in Figure 3. Transmissivity data of Florida were obtained from the Water Resources Research Center (Florida). In this study the transmissivity data for Florida were kept constant at a mean value of 0.242 10^4 $m^2.day^{-1}$ for the whole study area.

Similarly to the transmissivity the flow domains are divided into zones of equal net precipitation.

The areal net precipitation has been estimated from:

$R=P - E_t$

where

R is the net precipitation which may be available for infiltration;

P is the mean annual precipitation; and

E_t is the mean annual evapotranspiration.

Areal net precipitation values were obtained from Ituli (1984) for Kenya and from the Water Resources Research Center for Florida. Areal net precipitation values of Kenya are shown in Figure 4. Areal net precipitation values of Florida range between 130 mm/year and 690 $mm.year^{-1}$.

The topographic levels are obtained by averaging the topographic levels on each element of an 33 by 38 grid system imposed on a topographic map of the study areas. The square grids are measuring respectively 10.0 km by 10.0 km for the Athi-Tana River Basin, and 7.9 km by 7.9 km for Southern Florida. The topographical data of Kenya are shown in Figure 5. Topographical data of Florida range between 0 and 45 m.

Well withdrawals

To provide Mombasa (Kenya) with the necessary drinking water a pipeline was built between Mzima Springs and Mombasa. In 1995 35.000 $m^3.day^{-1}$ were pumped up at Mzima springs. Presently, plans exist to multiply the pumping capacity at Mzima springs up to 350.000 $m^3.day^{-1}$. We will compare the groundwater flow taking into account both pumping capacities.

In Florida tourism (e.g. Miami) and agricultural activities have lead to an explosion of the use of fresh water. While in 1900 no groundwater was pumped up, at least 2.500.000 $m^3.day^{-1}$ is pumped up today. We will compare the groundwater flow in southern Florida making use of a pumping capacity of respectively 0 $m^3.day^{-1}$ and 1.000.000 $m^3.day^{-1}$.

Additional measurements

Salinity and distance to the mean high water line (spring tide) were measured for 104 boreholes along the Kenyan coast. Those salinities and distances were related to the groundwater flow estimated by the mathematical model. Individual relations between groundwater flow and respectively salinity and distance to the mean high water line (spring tide) were studied making use of a Spearman rank correlation coefficient.

The possibility to use groundwater outflow to predict the presence/absence of mangroves along the coastal zone of Kenya was tested with a c^2-test comparing the number of squared grids with groundwater flow higher than a certain value (expected) with the number of squared grids with as well a high groundwater flow as with the mangrove ecosystem being present (observed).

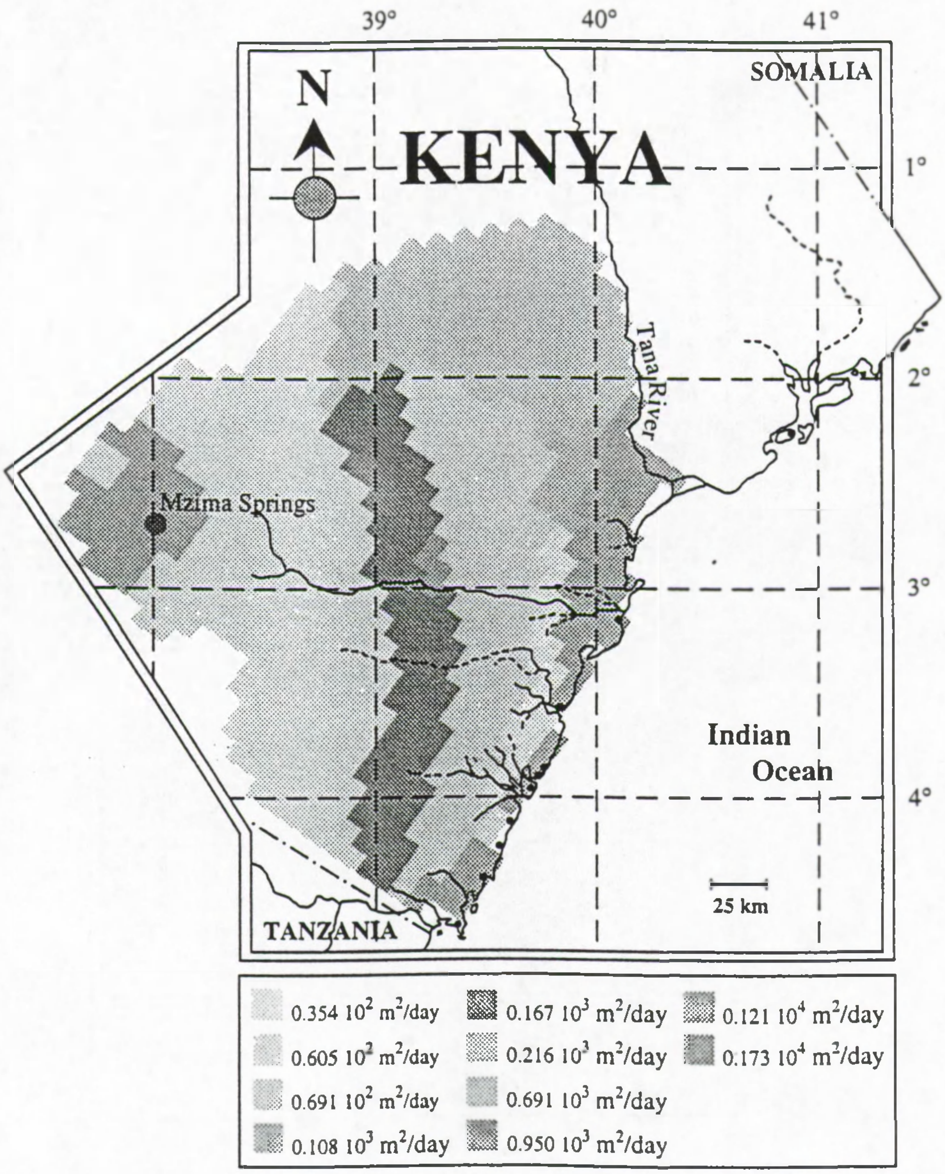

Figure 3. Zones with equal mean transmissivity (m^2/day) in the study area (Kenya).

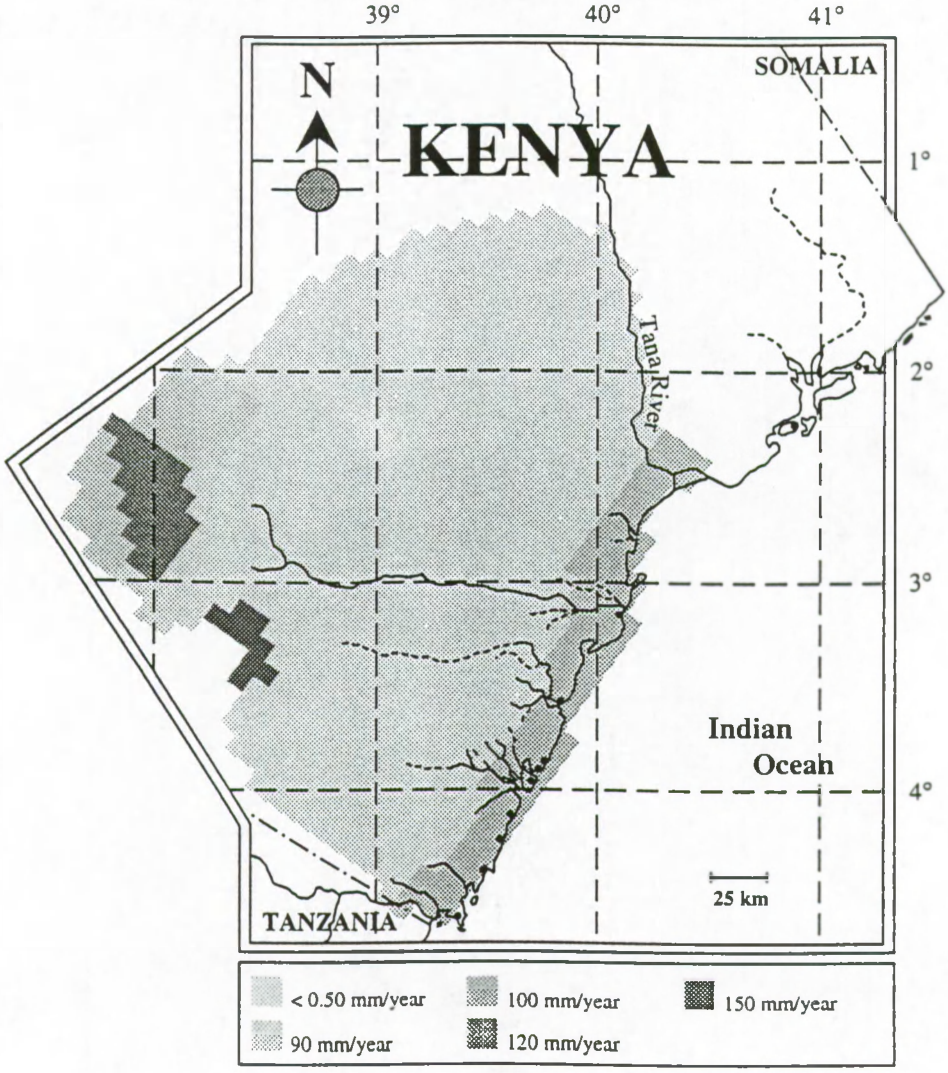

Figure 4. Zones with equal mean areal net precipitation (mm/year) in the study area (Kenya).

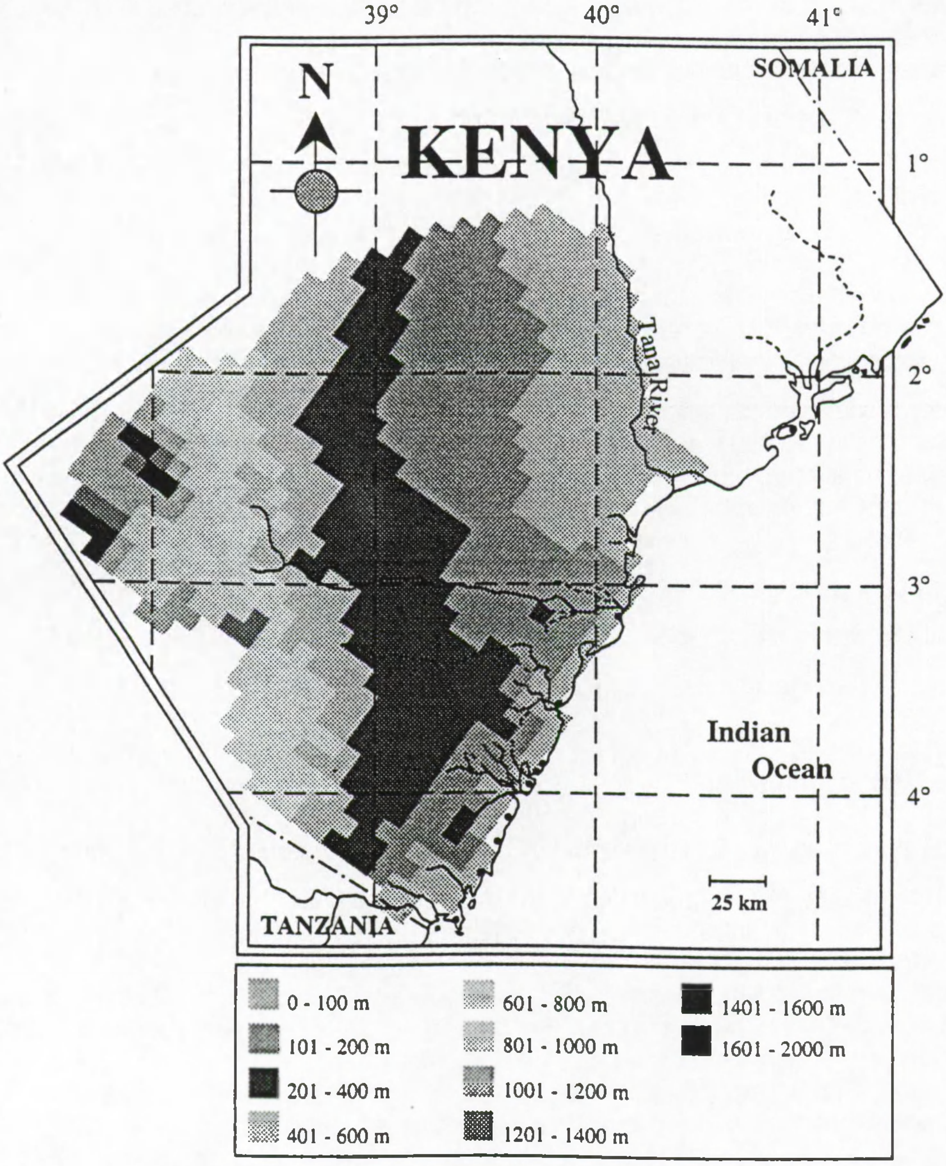

Figure 5. Zones with equal mean topographic levels (m) in the study area (Kenya).

Results

Figure 6 gives the results of the mathematical model for the groundwater flow in Kenya when 35.000 $m^3.day^{-1}$ is pumped up at Mzima springs. The arrows show the direction of the groundwater flow, while the colours represent the magnitude of the groundwater flow, which is expressed as the total flux over the aquifer, i.e. flux times aquifer tickness. A vector representation of the groundwater flow along the coastal zone is given in Figure 7. Only groundwater flow vectors with a groundwater flow = 1 $m^2.day^{-1}$ are shown. Figure 8 shows groundwater flow vectors along the Kenyan coast when every day 350.000 m^3 of groundwater is pumped up at Mzima springs.

Figure 9 and 10 show respectively groundwater flow in South Florida at the beginning of the century and in 1994.

Figure 11 shows the relationship between salinity, distance to the mean high water line (spring tide) and groundwater flow. Boreholes with a salinity lower than 1‰ were not used in the graph. With one exception all salinities higher than 1‰ were measured at locations where the groundwater model predicts a groundwater flow smaller than 1 $m^2.day^{-1}$. When the distance is kept constant, salinities show a clear Spearman rank correlation with groundwater flow (Table 1). Also salinities show very high correlations (Spearman rank correlation coefficient) with the distance to the mean water line (spring tide), when measured in a region with constant groundwater flow.

Comparing the number of cells with a groundwater flow higher than 1 $m^2.day^{-1}$ with the number of cells where the groundwater flow is higher than 1 $m^2.day^{-1}$ and where the mangrove forest is present, a c^2 test shows no significant differences.

Discussion and conclusions

The coastal zone of Kenya is characterised by moderately high groundwater flow ranging between 0.31 and 12.8 $m^2.day^{-1}$ (average value: 4 $m^2.day^{-1}$). The high groundwater flow is due to high potential gradients, high infiltration capacity of the geologic formations and high precipitation received by the area.

The elevation of the coastal belt ranges between 0 and 76 m. Piezometric heads as calculated from the model range between 0 and 48 m. In general those piezometric heads reflect the field situation: wells in the coastal belt strike water between 5 and 15 m.

High discharge of fresh groundwater into the sea is in conformity with Cashwell and Bakers 's (1953) assertion that close to the shores, in some points, seepage of freshwater occurs. Seawater seepage is reported in several areas along the Kenyan coast (Isaac and Isaac, 1968; Knutzen and Jasuund (1979); Ruwa and Polk, 1986; Ruwa, 1993).

Figure 7 shows a very clear correlation between groundwater flow and the distribution of the mangroves along the coast. This correlation is confirmed by the result of the c^2 test comparing the number of cells with a groundwater flow higher than 1 $m^2.day^{-1}$ with the number of cells where the mangrove forest is present and the groundwater flow is higher than 1 $m^2.day^{-1}$. The value of 1 $m^2.day^{-1}$ was chosen on the basis of Figure 11. Only 1 out of 104 boreholes had a salinity larger than 1‰ in case the groundwater flow was more than 1 $m^2.day^{-1}$. Figure 11 also shows that places with groundwater flow lower than 1 $m^2.day^{-1}$ are highly susceptible to seawater intrusion. This is much higher than the theoretical value of 0.22 $m^2.day^{-1}$ mentioned by

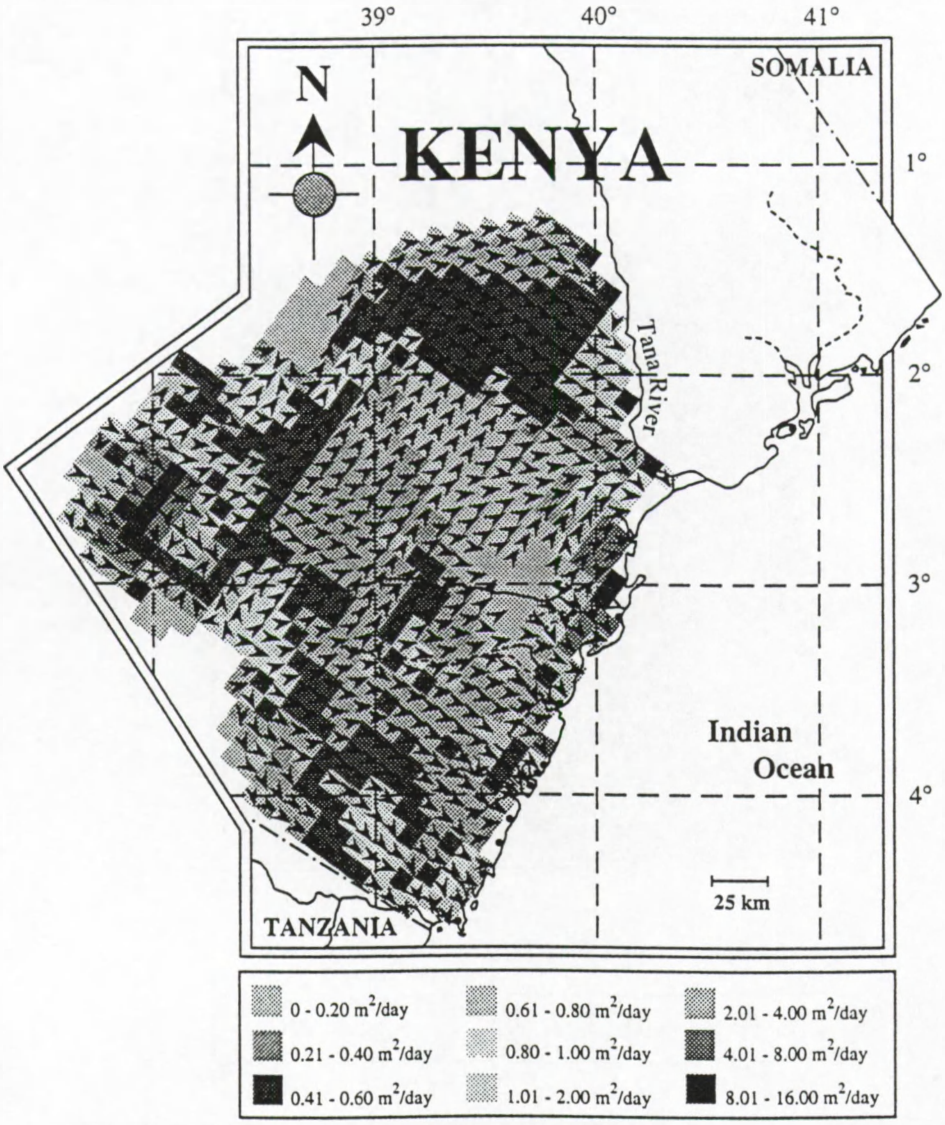

Figure 6. Groundwater flow: graphical output of the model. Arrowheads show the direction of the groundwater flow. Background colours indicate the size of the groundwater flow (m^2/day).

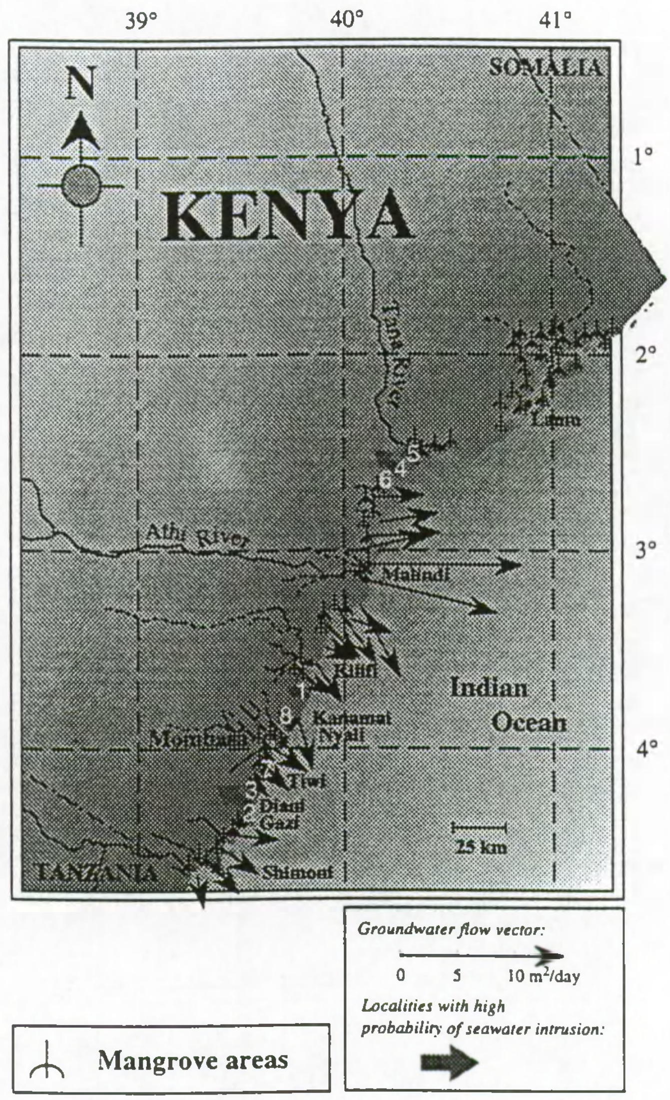

Figure 7. Groundwater flow: graphical output (vectorial) of the actual situation along the Kenyan coast between the Tanzanian border and Tana River. Only those areas with groundwater flow higher than 1 m^2/day and regions with a high probability of seawater intrusion are indicated. Numbers 1 to 8 refer to salinity measurements at different distances from the coast (see Fig. 11).

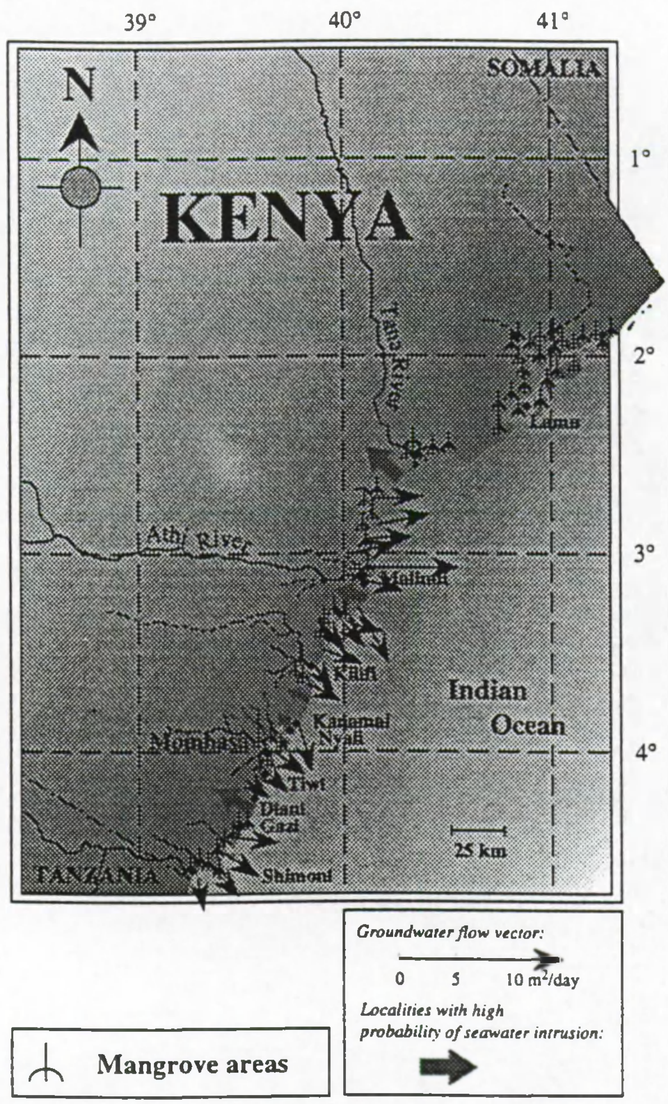

Figure 8. Groundwater flow: graphical output (vectorial) of the modelled situation along the Kenyan coast between the Tanzanian border and Tana River when 350.000 m^3/day is pumped at Mzima springs. Only those areas with groundwater flow higher than 1 m^2/day and regions with a high probability of seawater intrusion are indicated.

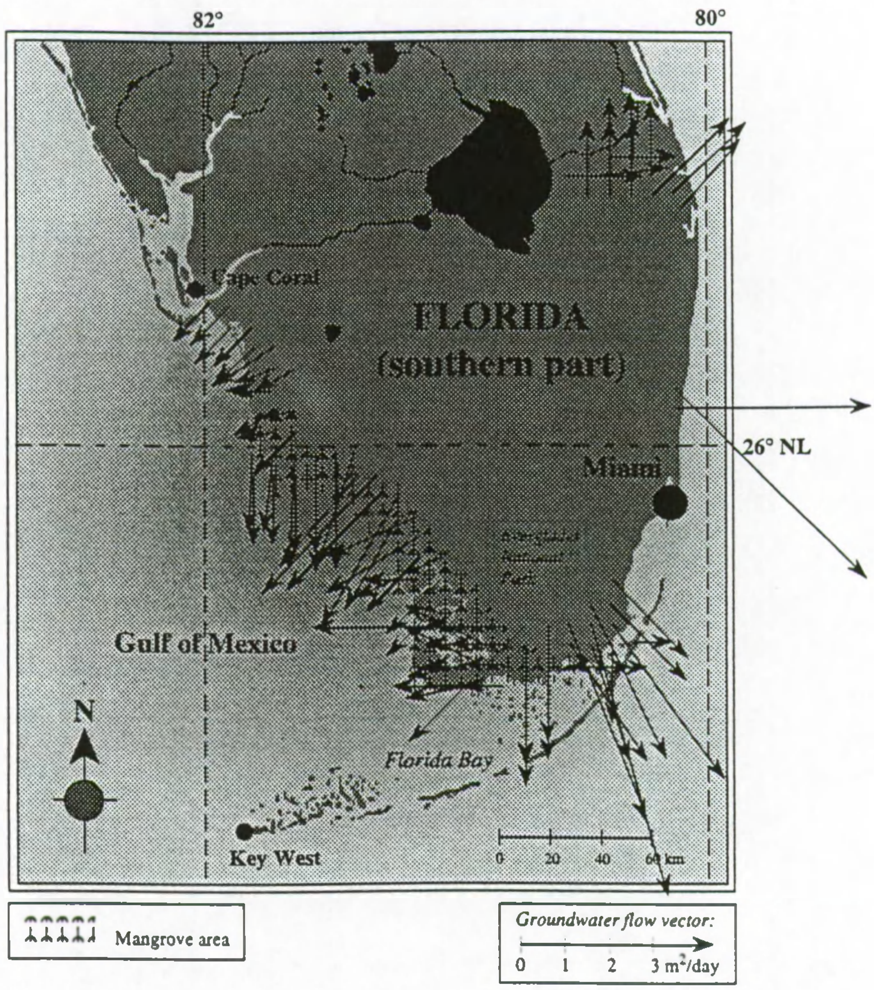

Figure 9. Groundwater flow: graphical output (vectorial) of the groundwater flow situation in the southern part of Florida in the beginning of the 19th century. In the western part of the study region groundwater flow is high in the direction of the Gulf of Mexico. In the eastern part there are only a few areas with high groundwater flow.

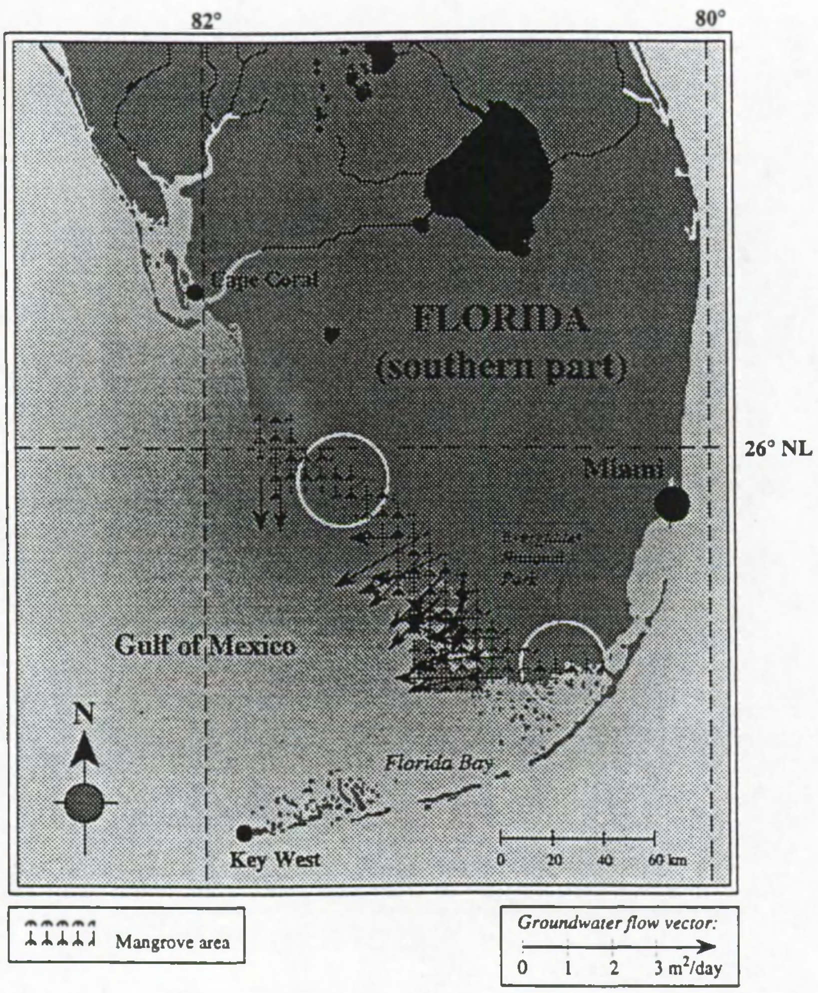

Figure 10. Groundwater flow: graphical output (vectorial) of the groundwater flow in the southern part of Florida based on actual data (Feb. 1994).

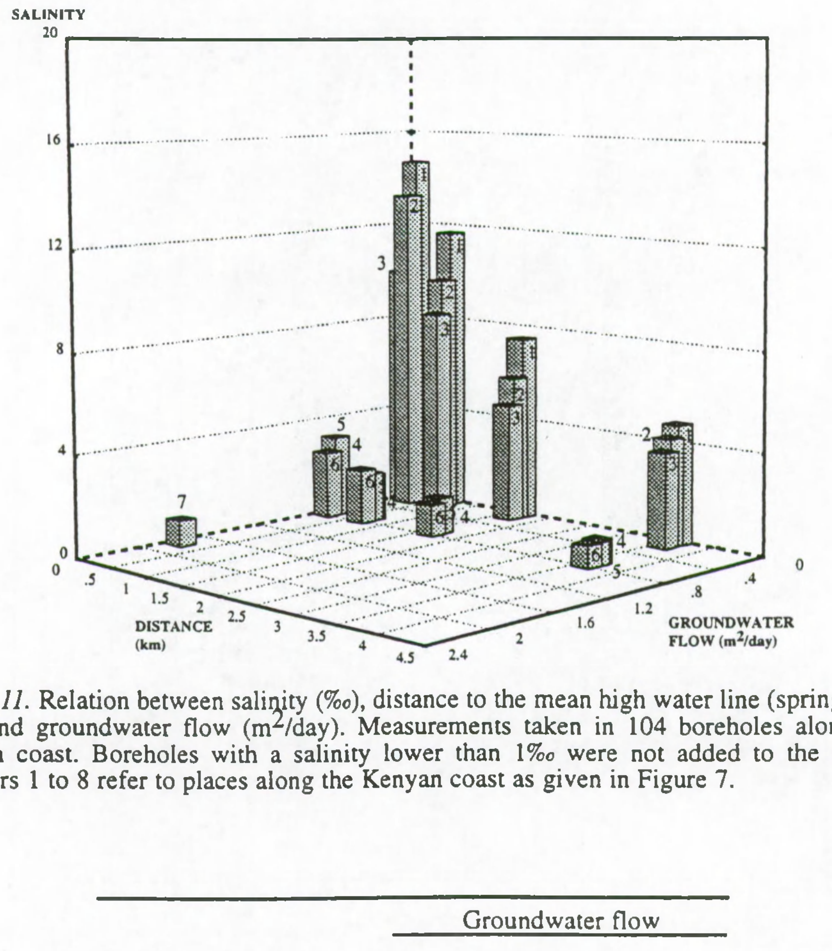

Figure 11. Relation between salinity (‰), distance to the mean high water line (spring tide) (km) and groundwater flow (m^2/day). Measurements taken in 104 boreholes along the Kenyan coast. Boreholes with a salinity lower than 1‰ were not added to the graph. Numbers 1 to 8 refer to places along the Kenyan coast as given in Figure 7.

	Groundwater flow	
	Spearman R	N
Salinity (0.5 km)	-.78*	26
Salinity (1 km)	-.73*	26
Salinity (2 km)	-.74*	26
Salinity (4 km)	-.74*	26

Table 1. Spearman Rank Correlation Coefficient between groundwater flow and salinity (distance to the mean high water line at spring tide kept constant).

Ituli (1984). Areas with a groundwater flow lower than 1 $m^2.day^{-1}$ are indicated in Figure 7 by red arrows.

Figure 11 shows the relation between salinity, or seawater intrusion, groundwater flow and the distance to the coast. Those three variables are clearly correlated with each other (Table 1 and 2). Salinity decreases exponentially with an increasing distance from the coast and with an increasing groundwater flow. Numbers 1 to 7 on Figure 11 correspond with identical numbers on the map of Kenya (Fig. 7). Figure 8 indicates a region with local seawater intrusion, probably caused by pumping up large volumes of groundwater by a local cement factory. However, we were not able to get the necessary information from the local responsibles to confirm this hypothesis.

	Distance	
	Spearman R	N
Salinity (0.242 m^2/day)	-1*	4
Salinity (0.312 m^2/day)	-1*	4
Salinity (0.354 m^2/day)	-1*	4
Salinity (0.886 m^2/day)	-1*	4
Salinity (0.904 m^2/day)	-1*	4
Salinity (0.963 m^2/day)	-1*	4
Salinity (2.030 m^2/day)		1

Table 2. Spearman Rank Correlation Coefficient between distance to the mean high water line at spring tide and salinity (groundwater flow kept constant).

To solve the drinking-water problem of Mombasa, plans exist to pump up more water at Mzima Springs. Pumping of huge amounts of water will lead to an alteration of the regional groundwater flow along the coast. At the moment groundwater is pumped at a rate of 35.000 $m^3.day^{-1}$. If this amount would be raised to the planned 350.000 $m^3.day^{-1}$ the groundwater flow in the region of Malindi and in the region south of Tana River would decrease drastically (Fig. 8). As a consequence, the probability of seawater intrusion would increase in both regions, and the existing mangrove forests would be endangered, because seawater intrusion would lead to the destruction of the ecosystem.

To prove the capability of the model to predict the destruction of the mangrove forest due to lower groundwater flows, we tested the model for the southern part of Florida, with a special interest to the Everglades National Park. The groundwater flow in the beginning of the century was compared to the situation in 1994. Figure 9 (beginning of this century) shows large groundwater outflow in the southern and western part of the region. In Figure 10 (situation 1994) the groundwater flow in the whole area has dropped drastically, because of urbanisation, tourism and agriculture. In those places where groundwater flow fell below 1 $m^2.day^{-1}$, mangroves are dying, as indicated by the yellow circles in Figure 10.

The complete absence of groundwater flow in the south-eastern part of the park together with the diminished surface water flow (Holloway, 1994; Fennema *et al.*, 1994), also explanains the doubling of the salinity in Florida Bay and the sea grass die-off in this bay (Mairson, 1994).

Hence, this study shows a clear relation between the distribution of the mangrove ecosystem and groundwater flow. Groundwater modellisation seems to be a proper tool to evaluate the effects of massive pumping of groundwater. The model also shows how human activities as far as several hundreds of kilometres inland can destroy vast areas of mangroves by changing the groundwater flow. The model predicts and/or confirms the destruction of large mangrove areas in Kenya and the Everglades (Florida-USA). The present authors believe at least one variable must be added to the list of seven basic conditons (Chapman, 1975, 1977, 1984) for the existance of extensive mangrove forests: presence of fresh water.

Acknowledgements

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The ecological importance of groundwater modellisation making use of different grid sizes

Tack, J.F., A. Verheyden, T. Van Daele & P. Polk
Free University, Brussels, Belgium

Introduction

Changes to inland watersheds are quite often manifested and even amplified at the margin between land and estuary (Simenstad *et al.*, 1992; Bjork & Powell, 1993). The high productivity of undisturbed coastal regions depends on the adaptation of organisms to gradients generated by freshwater inflow into the sea. Nichols *et al.* (1986) described the catastrophic decreases in valuable fisheries and other biotic processes after altering patterns and amounts of freshwater inflow influencing the balance between plant and animal communities.

Several studies illustrate the conflicts between water resource management and the quality of estuarine habitats. Nichols *et al.* (1986) describe how, in California (U.S.A.), the flow from two major tributaries to San Francisco Bay (San Joaquin and Sacramento rivers) has been reduced to less than 40% of historic levels. Most of the historic flow upstream in the watershed is now hold in reservoirs for use by agricultural and urban consumers during dry California summers. The projects upstream have resulted in three major ecological impacts: (1) collapse of the salmoid fisheries which is directly attributable to the construction of the Shasta Dam, (2) reduced freshwater inflow together with overfishing resulted in the decline in abundance of sturgeon, sardines, flatfish, crabs, and shrimp, and (3) the diminished freshwater inflow is probably responsible for the reduction of the capacity of this estuary to dilute, transform, or flush contaminants that are diluted into San Francisco Bay (Nichols *et al.*, 1986).

Even when changes in volumes of groundwater discharge are small, alteration of timing of freshwater flow can result in negative impacts in the estuarine ecosystems. Since 1900 twenty-eight dams have been constructed on the Columbia River (U.S.A.). Management policies have reduced springtime freshwater flows to 50% of former levels, while fall discharges have been artificially increased by 10-50% (Simenstad *et al.*, 1992). These changes had a negative effect on salmon fisheries in the region.

Other examples of how changes in freshwater flow affect the estuarine ecosystems located downstream can be found throughout the whole world. Completion of the Aswan High Dam completely eliminated flows from the Nile River to the Mediterranean Sea during the dry season. As a result, the sardine fishery in the area was reduced by 95% due to elimination of the nutrient plume from the river delta (Alleem, 1972). As a result of greatly reduced sediment supply from operation of the Aswan High Dam, coastal erosion now exceeds deltaic sedimentation. This situation, coupled with delivery of nutrient-laden agricultural runoff and urban wastewater to the lower delta , threatens ecologically critical coastal lagoons and the wetland species they support (Stanley & White, 1993).

Mangrove losses in Kenya, India/Bangladesh and Florida (Everglades) can be explained by changes in groundwater flow (Tack & Polk, 1999). They showed how tropical catchments, sometimes several hundreds of kilometers inland, influence the mangrove areas along the coastal lines of the continents. Those changes, either caused by human interference or by nature, can have serious consequences.

Material and methods

Study areas

Three catchment areas along the East African coast were studied: (1) the Athi-Tana River Catchment area in Kenya, (2) Zanzibar Island in Tanzania, and, (3) Inhaca Island in Mozambique.

All catchment areas are bordered by coastal areas with mangrove forests and seagrasses present on certain sites along the coast. The major difference between the three catchment areas is the scale (see maps).

Groundwater model

The mathematical model used in this study was developed by the U.S. Geological Survey. This modular finite-difference groundwater flow model, commonly called MODFLOW, can simulate groundwater flow in a three dimensional heterogeneous and anisotropic medium provided that the principal axes of hydraulic conductivity are aligned with the coordinate directions. A constant density fluid is also assumed. MODFLOW was originally documented in McDonald & Harbaugh (1984). The MODFLOW version used in this study was MODFLOW-96. As graphical interface we run Visual Modflow, developed by Waterloo Hydrogeologic.

The basin characteristics that serve as inputs for the computer to solve the model equations are identical to those used in the model described in the former article. We made use of the transmissivity values, T, the areal net precipitation, R, the topographic levels, h_t, the aquifer thickness, d, and the porosity of the aquifer material, n.

The transmissivity values were obtained by the product of hydraulic conductivity and the thickness of the aquifer, $T=KD$, where K is the hydraulic conductivity and d is the thickness of the aquifer. The flow domains were divided into several zones having different transmissivity values. Those zones correspond to the geological units distinguished in the flow domains. Similarly the flow domains were divided into zones of equal transmissivity and of equal net precipitation.

The areal net precipitation was estimated from:

$R=P-E_t$

Where R is the net precipitation which may be available for infiltration, P is the mean annual precipitation, and E_t is the mean annual evapotranspiration.

Visual MODFLOW has a maximum of 499 x 499 cells. To reach a resolution of less then 200x200 m as foreseen in the project proposal a special procedure was developed for the Athi-Tana River basin. The Kenyan coastal zone was modeled to this resolution between the coastal line and a parallel line 20 km inland. At the same time the Kenyan coast was divided in three sub-basins: (1) between Tana River and Malindi, (2) between Malindi and Mombasa, and, (3) between Mombasa and Vanga. Those sub-basins are bordered by no flow boundaries (rivers). On the landside we used the flow conditions as calculated by the model described in the former article. In the cases of Zanzibar island (Tanzania) and Inhaca island (Mozambique) the maximum of 499 rows and 499 columns used by MODFLOW was sufficient to model the groundwater flow up to a resolution of less than 200x200 meters.

In each of the three study sites groundwater flow was measured in 4 grid systems with different resolution. In the three Kenyan sub-basins we used the following resolutions: (a) between Tana River and Malindi: 150x200m, 750x750m, 2x2km, and, 5x5km; (b) between Malindi and Mombasa: 150x250m, 750x750m, 2x2km, and, 5x5km; (c) between Mombasa and Vanga: 150x150m, 750x750m, 2x2km, and, 5x5km. On Inhaca island we made use of 100x100m, 750x750m, 2x2km, and, 5x5km. On Zanzibar island we made use of 100x150m, 750x750m, 2x2km, and, 5x5km.

Results

We should make the difference between two kinds of results. First of all we have the outcome of the groundwater flow modellisation for the different basins and the different grid sizes. Those results are of special interest to validate the groundwater modellisation. Secondly we have the groundwater flow along the coastal strip and the way this flow interacts with the ecology of the estuarine ecosystems. Those figures are of immediate importance for the project partners since they can connect their ecological studies to the outcome.

The following figure shows how different groundwater flow can be at specific sites along the Kenyan coast when different grid sizes are used.

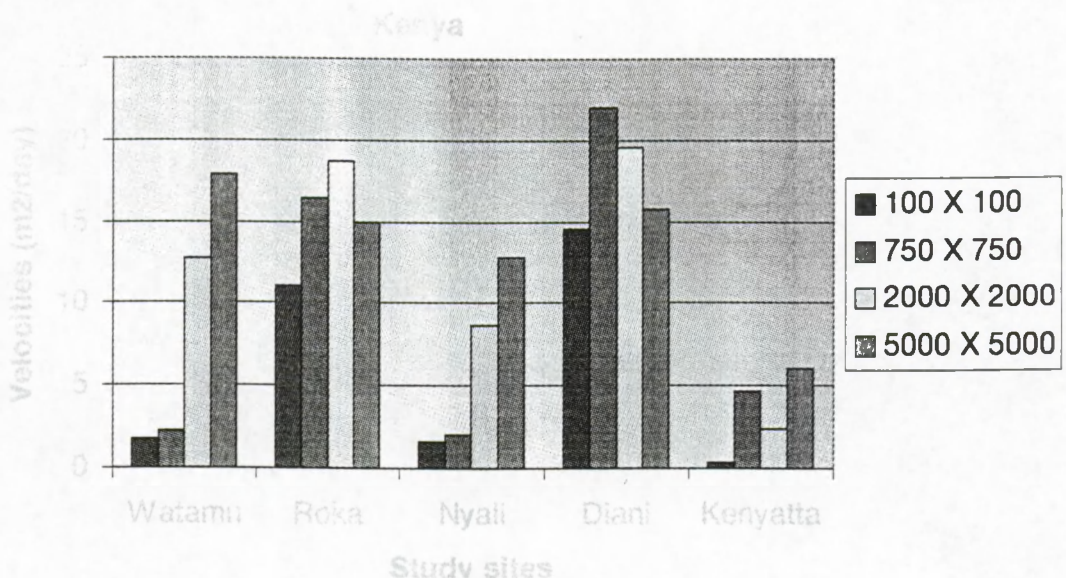

Figure 1: *Groundwater flow velocities ($m^2.day^{-1}$) at different study sites along the Kenyan coast in function of different grid sizes.*

Figure 1 shows significant differences in groundwater flow at the same site in function of the grid size used. Similar differences were found for the different study sites on Zanzibar island and Inhaca island. Figure 2 – 6 show groundwater flow for respectively Kenya - Tana River to Malindi (5x5 km), Kenya – Malindi to Mombasa (5x5 km), Kenya - Mombasa to Vanga (5x5 km), Zanzibar island (2x2 km), and, Inhaca island (100x100 m). A full overview of all results is given on the CD-ROM with interactive groundwater flow maps of the regions studied. This CD-ROM will be available in August 1999.

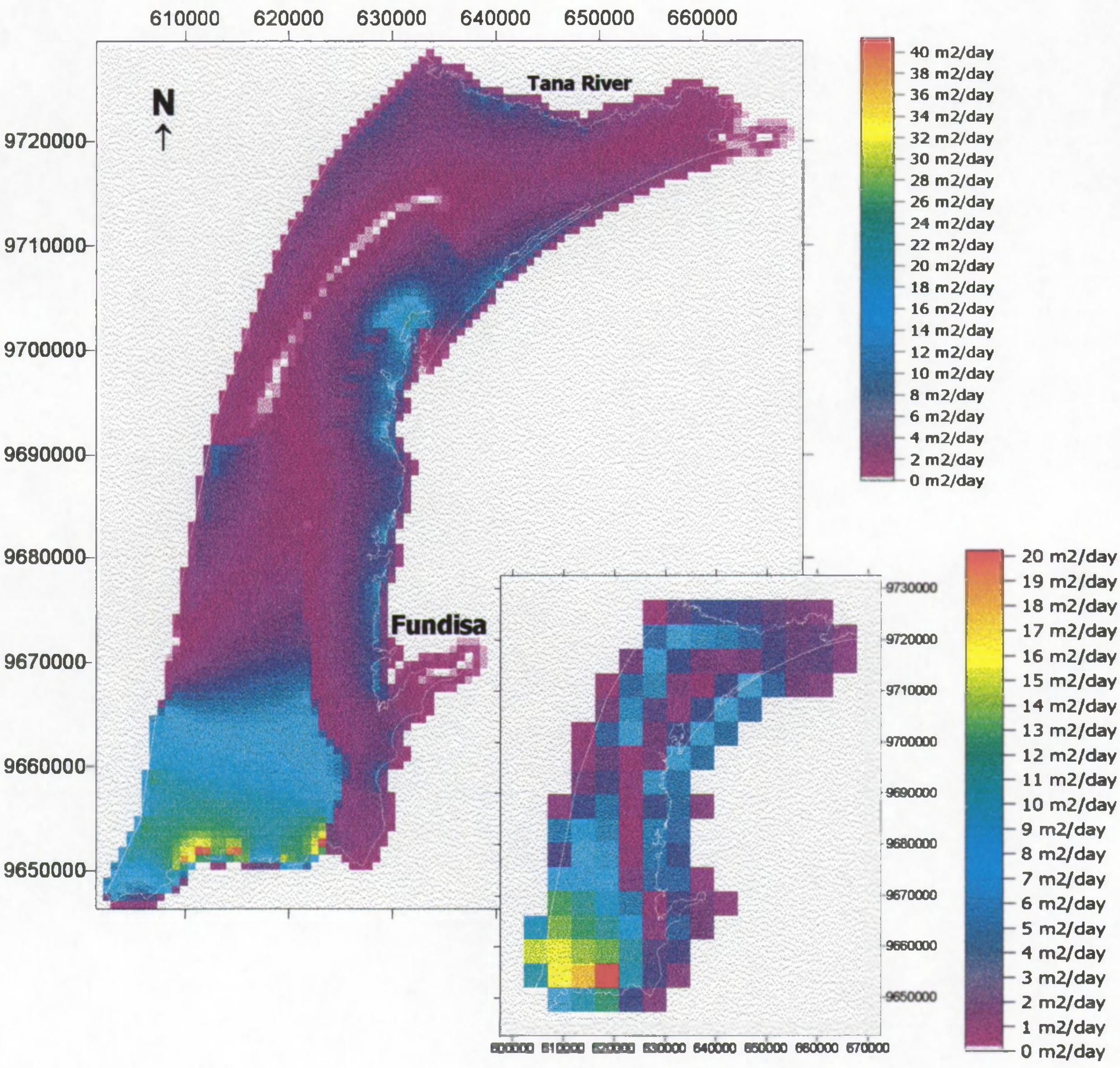

Figure 2: Kenya: groundwater flow velocities ($m^2.day^{-1}$) in the area between Tana River and Malindi (grid sizes: 150x200 m and 5x5 km).

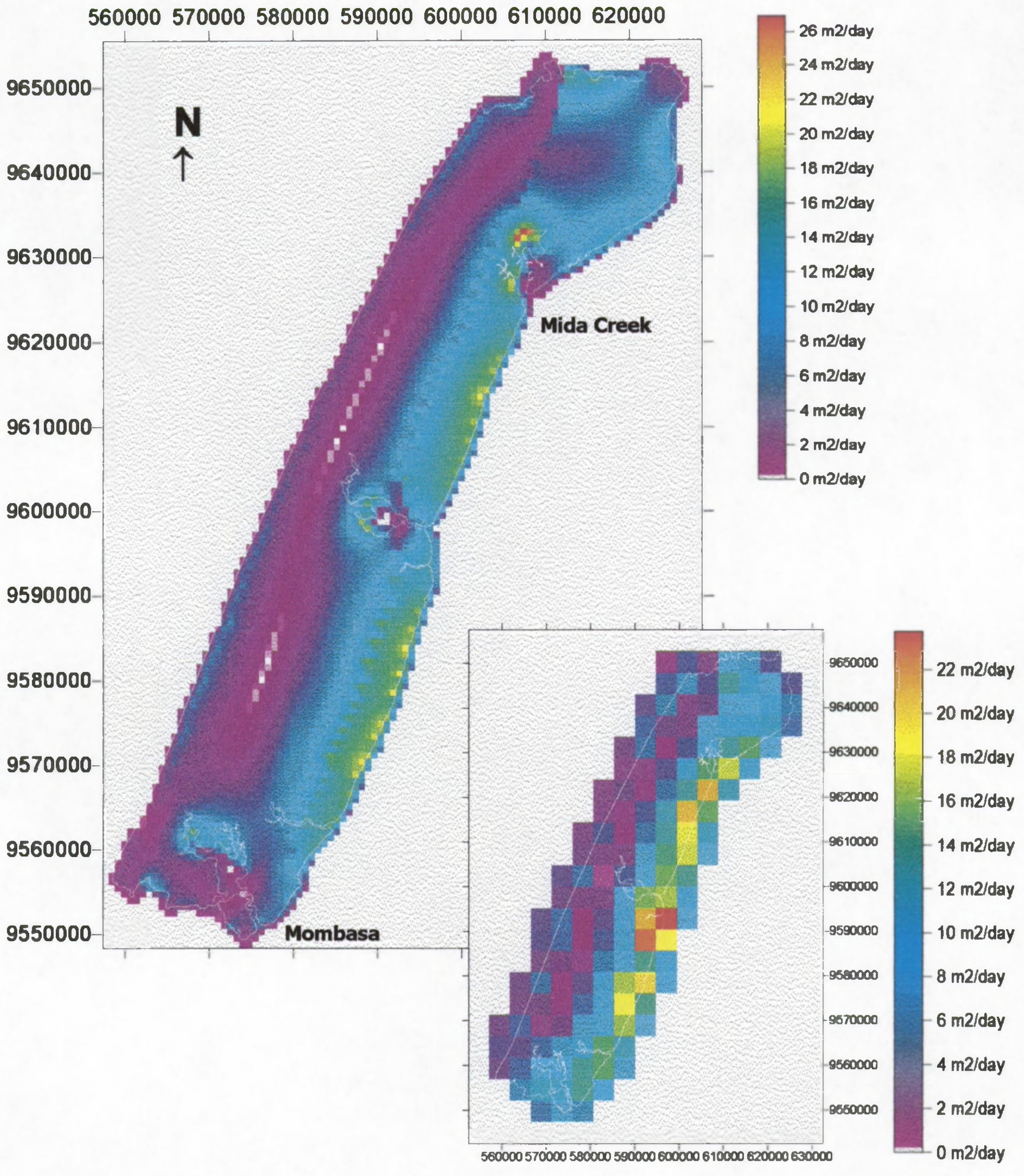

Figure 3: Kenya: groundwater flow velocities ($m^2.day^{-1}$) in the area between Malindi and Mombasa (grid sizes: 150x250 m and 5x5 km).

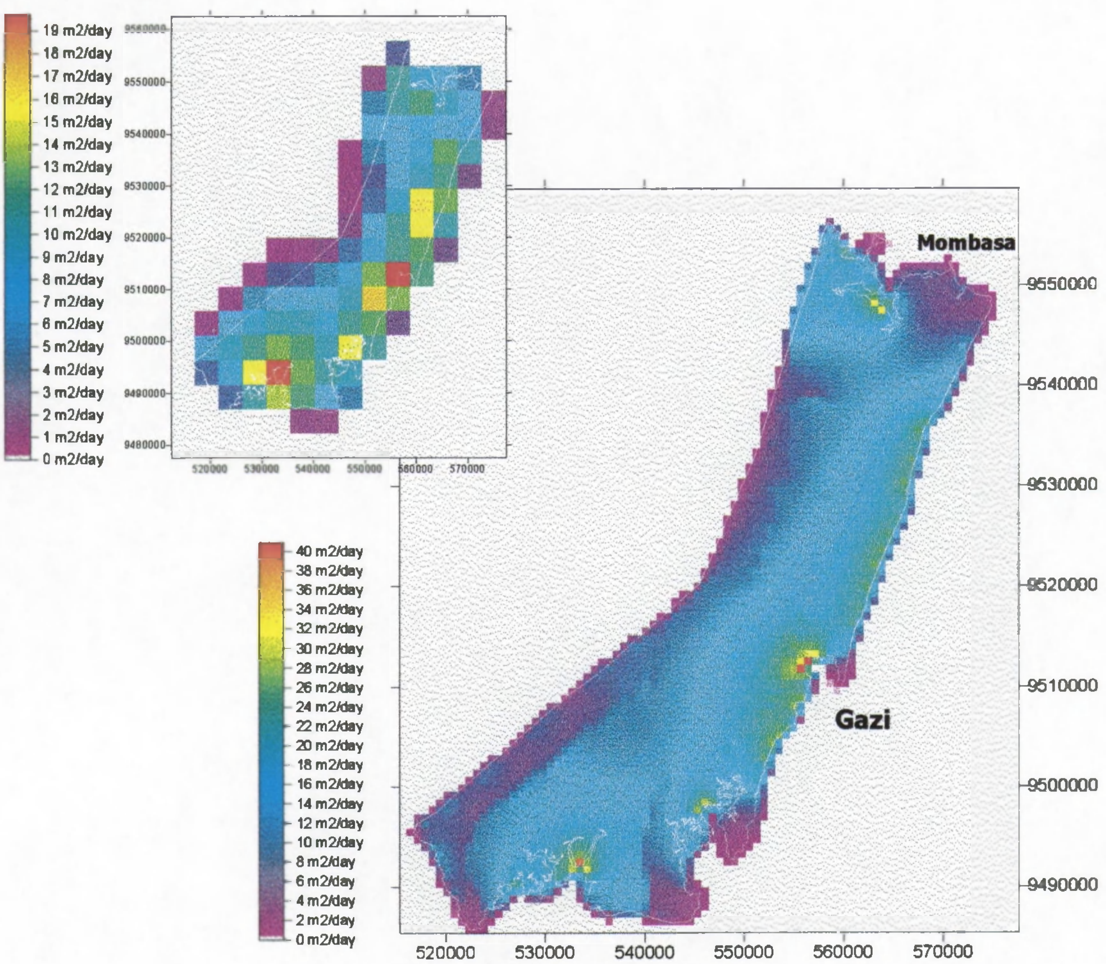

Figure 4: Kenya: groundwater flow velocities ($m^2.day^{-1}$) in the area between Mombasa and Vanga (grid sizes: 150x150 m and 5x5 km).

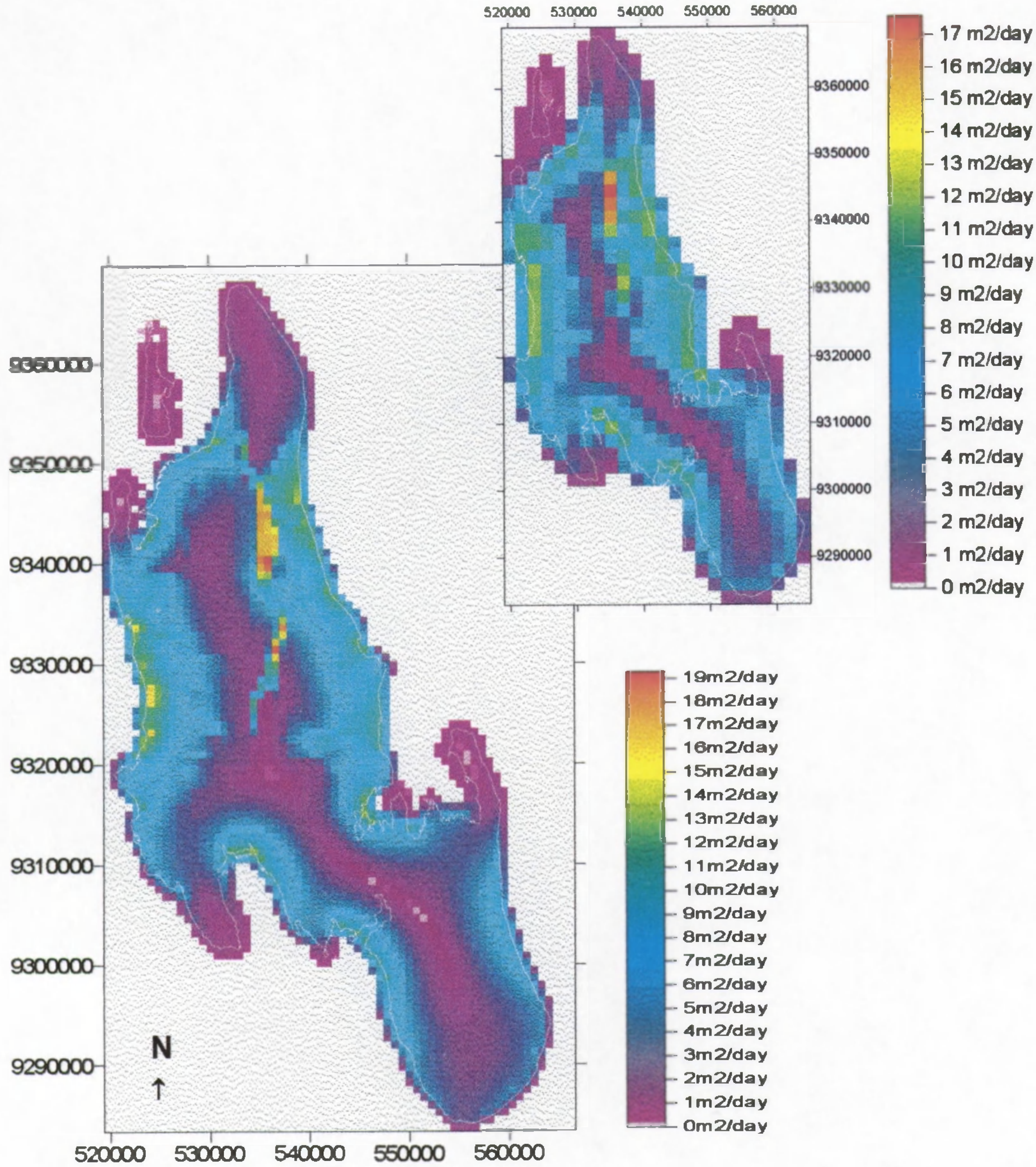

Figure 5: Tanzania: groundwater flow velocities ($m^2.day^{-1}$) on Zanzibar island (grid sizes: 100x150 m and 2x2 km).

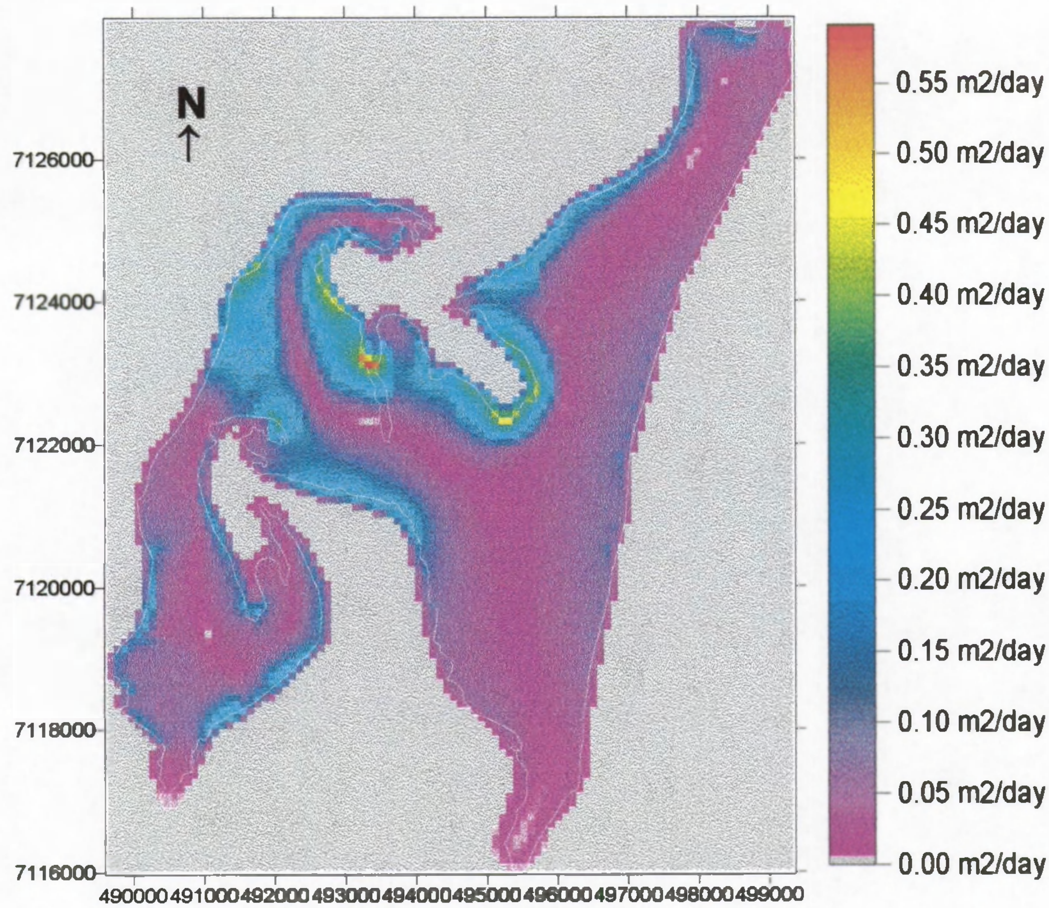

Figure 6: Mozambique: groundwater flow velocities ($m^2.day^{-1}$) on Inhaca island (grid sizes: 100x100 m).

Discussion and conclusions

Figure 1 shows significant differences in groundwater flow at the same site in function of the grid size used. Those differences can be explained by a change in the mean groundwater flow at a certain point when the grid size is changing. We give an example to explain this situation. Along the Kenyan coast we have several places with a relative low groundwater flow surrounded by areas with a relative large groundwater flow. When you take a small grid size into account the place with the small groundwater flow is detectable. By increasing the grid size the place with the small groundwater flow is taken together with the areas with relative larger groundwater flow. The result is an increase in groundwater flow at that specific location.

The opposite situation, a place with a relative high groundwater flow surrounded by areas with relative low groundwater flow, does also exist but is less common. This explains the differences shown in figure 1.

More important is the question how to link our results to ecological phenomenons. The ecology of estuarine ecosystems is especially influenced by groundwater outflow. Groundwater flow and groundwater outflow are two complete different processes. However groundwater flow can be an indication of the quantity of groundwater outflow. Groundwater outflow is most often very localized. However, due to current and tidal movement the groundwater outflow at one point is spread over a much bigger area.

When we compared the different grid sizes with the distribution of the mangrove ecosystem and with seagrass distribution (see contribution of NIOO-CEMO in this report) we came to optimal grid sizes to explain ecological processes: 5x5 km for the Kenyan coast, 2x2 km for Zanzibar island and 750x750 m for Inhaca island. The differences in grid size used are comparable with the differences in catchment areas of the three study areas. Whether there is indeed a positive correlation between those variables should be the focus of future research.

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Search for changes in groundwater flow as recorded by elemental changes in biogenic carbonates

F. Dehairs, C. Lazareth, E. Keppens,
Free University, Brussels, Belgium
L. André,
Royal Museum for Central Africa, Tervuren, Belgium
J. Kazungu
Kenya Marine and Fisheries Research Institute, Mombasa, Kenya

Objective

To elucidate differences in nearshore community structures and ecosystem functions in relation to groundwater flow.

Introduction

Shells of gastropods and bivalves from different sites along the Kenyan coast were analysed for major, minor, trace element and isotopic composition. The aim was to assess the potential of shells to archive short term fluctuations of the physico-chemical characteristics of the environment in which they were thriving and to document differences between sites eventually related with differences in regime of groundwater and surface water flow. Focus was on the bivalve *Isognomon* sp. common to the area, but other species of bivalves and gastropods were analysed as well.

Methodology

Sampling

Shells were collected from several sites differing in groundwater and surface water flow regime (groundwater flow data from VUB partner ECOL): (1) Mida mangrove ecosystem, situated north of Mombasa; freshwater input is essentially by groundwater flow with little or no surface flow (GW flow = 0.119 m d^{-1}); (2) Gazi mangrove ecosystem, located south of Mombasa; freshwater input is largely by groundwater flow (GW flow = 0.164 m d^{-1}), complemented by a seasonally flowing small river (River Kidogoweni); (3) Diani Beach, south of Mombasa (GW flow = 0.134 m d^{-1}); (4) Tudor Estuary; adjacent to Mombasa city with freshwater input mainly via surface water flow from River Kombeni (GW flow = 0.067 m d^{-1}); (5) Fort Jesus, old harbour of Mombasa town in Tudor Estuary (GW flow = 0.084 m d^{-1}); (6) English Point, at the mouth of Tudor Estuary (GW flow = 0.107 m d^{-1}); (7) Nyali Beach, just north of Mombasa (GW flow = 0.107 m d^{-1}); (8) Kenyatta Beach in Mombasa area (GW flow = 0.121 m d^{-1}). In addtion the following sites were sampled: Vipingo (GW flow = 0.164 m d^{-1}; Roka (GW flow = 0.170 m d^{-1})

Analysis

Total element content by ICP-MS

Shells were treated overnight with warm (60°C) concentrated peroxide (Merck P.A.) to remove all associated organic matter. Treatment was continued till no further visible reaction. Then, shells (whole gastropod shell or one valve of bivalve shell) were rinsed, dried, weighed and digested *in toto,* using concentrated HCl (10%; Merck Suprapur). Elements were finally recovered in HNO_3 solution (Merck Suprapur). Solutions were brought to volume and, if necessary, diluted with Mili-Q grade water. Multi-element standards were prepared in acidified water. To both samples and standards internal standards (^{99}Ru, ^{115}In, ^{185}Re, ^{209}Bi) were added to account for matrix effects and instrumental drift. Concentrations of the following isotopes were measured by Inductively Coupled Plasma - Mass Spectrometry (ICP-MS; Fisons-VG PlasmaQuad II+): ^{59}Co, ^{65}Cu, ^{66}Zn, ^{114}Cd, ^{138}Ba, ^{208}Pb, ^{238}U as well as the Rare Earth Elements (REE) ^{139}La, ^{140}Ce, ^{141}Pr, ^{146}Nd, ^{151}Eu, ^{152}Sm, ^{157}Gd, ^{163}Dy, ^{166}Er, ^{172}Yb, ^{175}Lu and also ^{232}Th.

Laser Ablation ICP-MS

A thin slice (2 to 3 mm thick) was cut out of the shell along the axis of maximum growth using a diamond saw. The LA-ICP-MS analysis was carried out with a frequency quadrupled Nd-YAG Fisons-VG Microprobe (wavelength: 266 nm) coupled to a Fisons-VG PlasmaQuad II+ mass spectrometer. The laser was operating in the Q-switched mode with a power of 2 mJ, a frequency of 10 Hz and without aperture. The pre-ablation and acquisition times were respectively set to 10 and 20 s (all conditions are summarised in Table 1). With these conditions the craters produced by laser ablation have a diameter of approximately 30 μm. In order to obtain quantitative LA-ICP-MS results, the analyses were calibrated with the fused glass international standard (NBS) NIST610 (nominal elemental concentrations of ~ 500 ppm).
Element distribution was investigated in the calcite layer of the shell, from umbo towards the edge along the growth axis. Crater spacing was about 200 μm (Figure 1). Per shell between 140 and 185 craters / analyses were perfomed

Table 1. Operating conditions of the laser probe ICP-MS.

Laser Probe (l = 266 nm)	
Laser mode	Q-switched
Laser power (mJ)	2
Frequency (Hz)	10
Preablation time (s)	10
ICP-MS	
Argon flow rate (l min-1)	
Carrier gas	1
Auxiliary gas	1.4
Cooling gas	13.7
Acquisition mode	Peak jump
Points per peak	3
Dwell time (ms)	200
Acquisition time (s)	20

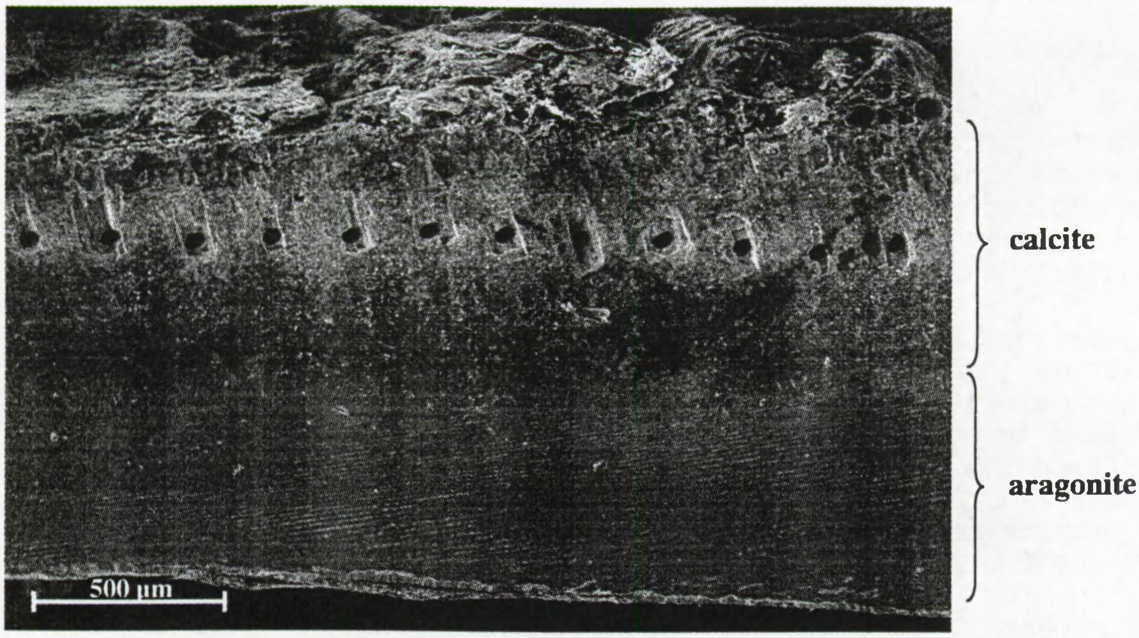


Fig. 1. Scanning Electron Microscope picture of part of the section along the Isognomon *shell's growth axis. The craters arranged in the calcite layer size about 60 mm and are the result of ablation by the laser. Every crater represents a point analysis.*

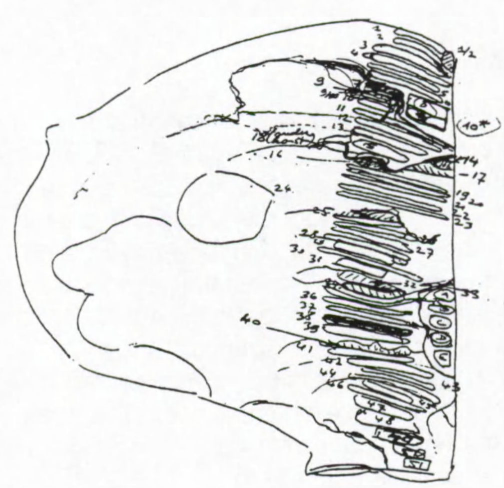


Fig. 2. Sketch of the sampling procedure of the calcite layer of an Isognomon *shell for $d^{18}O$ and $d^{13}C$ analysis. Successive bands indicate the subsampling of the calcite layer. 51 aliquots were sampled. Aliquots A to E repeat sampling of aliquots 34 to 40.*

Determination of stable $^{18}O/^{16}O$ and $^{13}C/^{12}C$ composition of Isognomon shell

About fifty, ca. 1 mg, calcium carbonate samples were taken with a dental drill, from umbo to outer rim of the shell, along the external calcitic layer (roughly about every 0.5 to 1 mm), taking care not to contaminate with underlying aragonitic shell carbonate (Figure 2). Previous investigations (Vander Putten et al. submitted) have confirmed that roasting the aliquots to eliminate organic matter prior to carbonate dissolution was not necessary. Organic matter still attached to the shell does not affect the isotopic signal associated with the carbonate.

Three surface seawater samples from the vicinity of the bivalve's habitat were sampled in August after the collection of the specimens.

Carbonate samples were digested at 25 °C by anhydrous orthophosphoric acid to release CO_2, using standard techniques. Water samples were isotopically equilibrated at 25 °C with CO_2, using standard techniques. Isotope measurements were performed on a Finnigan MAT Delta E stable isotope ratio mass spectrometer.

Results are reported in d notation relative to PDB standard for carbonates and to SMOW standard for waters. Reproducibility (at 2 s level) is better than 0.1 ‰ for $\delta^{18}O$ and better than 0.05 ‰ for $\delta^{13}C$.

Temperature and rainfall data

Monthly averaged values of sea surface temperature (SST) were obtained from the web site as blended from ship, buoy and bias-corrected satellite data (Reynolds and Smith, 1994). Monthly averaged values of precipitation were also obtained from the web as blended from data of NOAA, National Centers for Environmental Prediction, Climate Prediction Center). Data shown are representatitve for a geographical setting encompassing the Mombasa area with its coastal waters.

Results

Trace metals Pb, Cu, Zn, Co

Metal contents in the shells are in general very low, not exceeding of few ppm. *Isognomon* sp shells from Fort Jesus and English Point locations adjacent to Mombasa city show the highest enrichments with average concentrations reaching 5.6, 4.1, 2.9 and 1.55 ppm (mg g^{-1} of shell material) for Pb, Zn, Cu and Co respectively (Figure 3; data for Co not shown). Average Cu contents are also relatively high for Diani and Nyali sites (2.6 and 2.4 ppm respectively). Pb content in shells at Fort Jesus is about a factor 10 higher than at other sites. Least enriched site for Pb, Zn and Co appears to be Diani, south of Mombasa and for Cu, Kenyatta site.

Cypraea sp. shells are most enriched in Cu and Co (up to 1.47 ppm) at Nyali relative to Diani and Vipingo, with Vipingo showing the lowest concentrations (Figure 4). Vipingo site on the contrary shows highest enrichments of Zn for *Cypraea* shells. Lead is very low in general for *Cypraea* (< 0.1 ppm), but Nyali has lowest values. Cadmium values (not shown) were in general low (< 0.1 ppm) but quite variable from site to site.

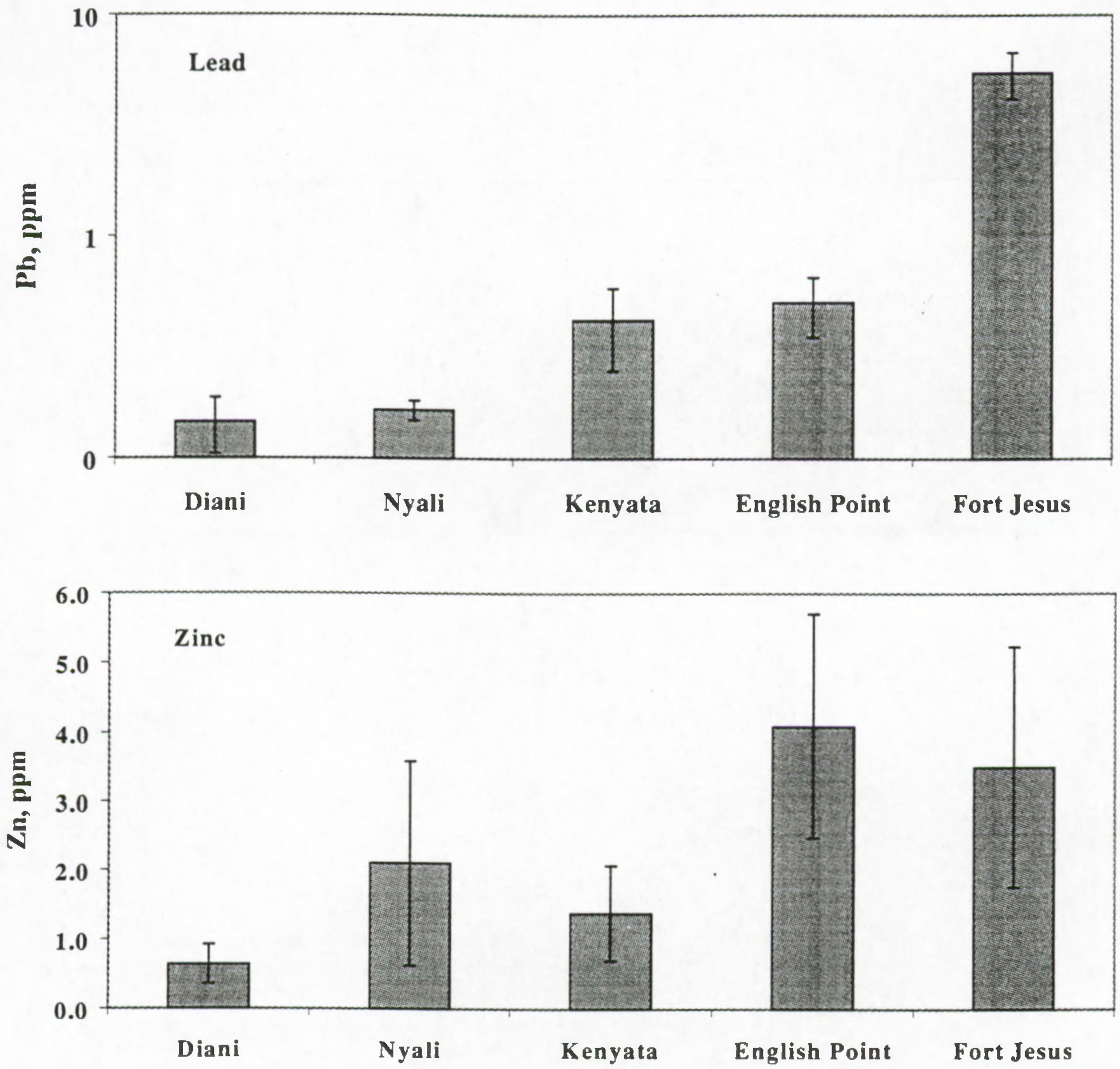
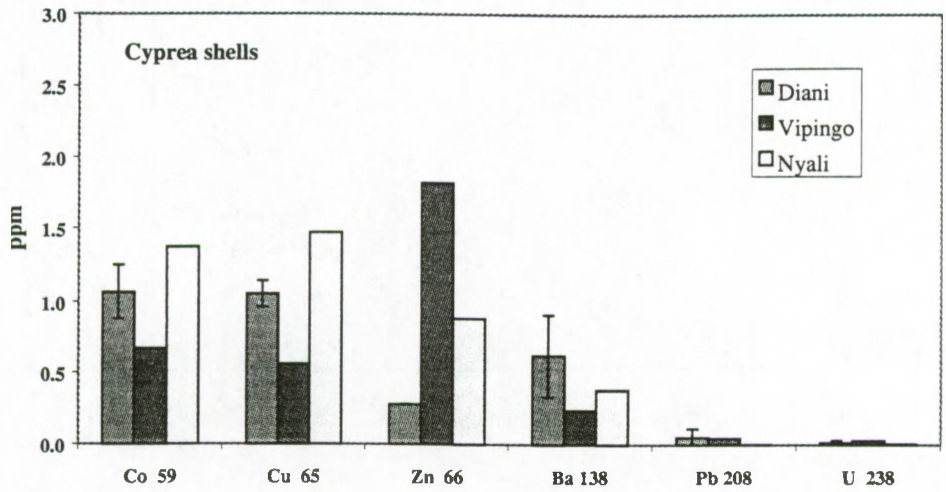


Fig. 3. Pb, Zn and Cu content of small sized (1.5 to 2 cm) whole Isognomon *shells from Diani, Nyali, Kenyatta, English Point and Fort Jesus. For Pb logarithmic scale.*



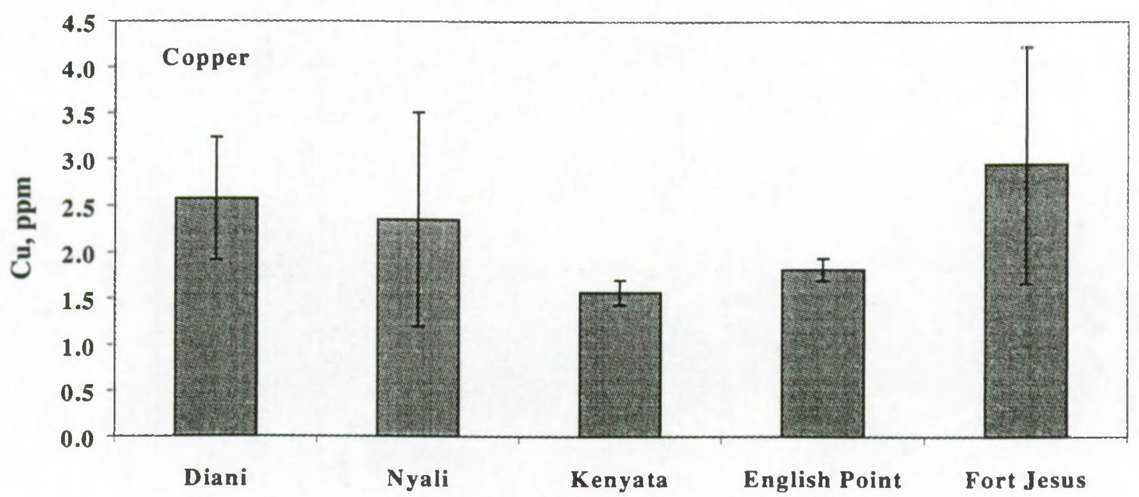
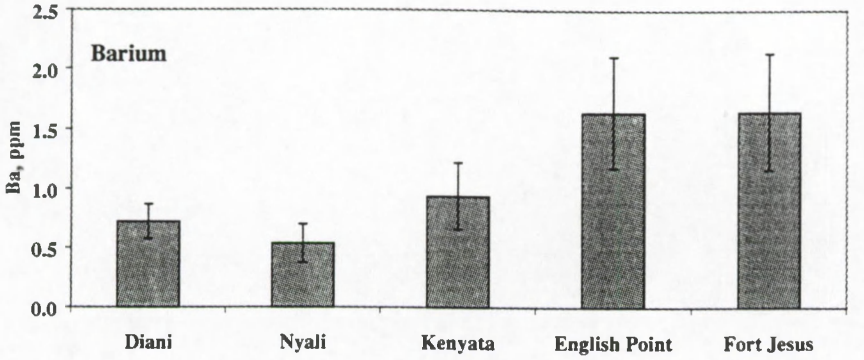
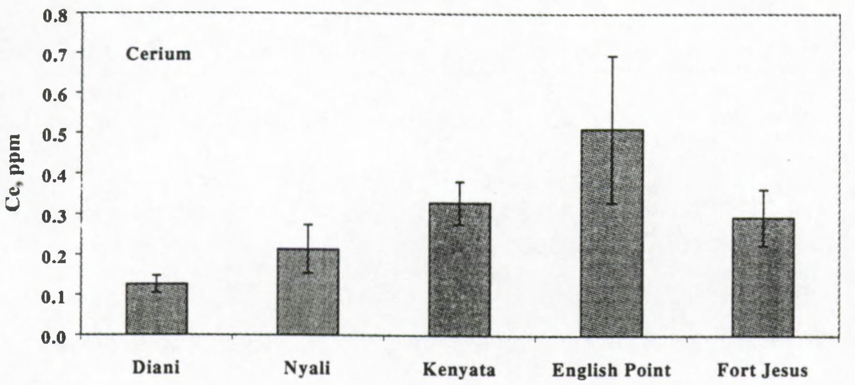


Fig. 4. Co, Cu, Zn, Pb, Ba and U content in Cypraea *shells from Diani, Vipingo and Nyali*

Barium

Highest average Ba concentrations are observed for a series of large sized (2.5 to 4 cm) *Isognomon* shells from Tudor (up to 6.5 ppm; see later Figure 7). Non-identified lamellibranch specimens collected at Nyali had second highest Ba values (up to 2.5 ppm). For a series of small sized (1.5 to 2 cm) *Isognomon* shells (Figure 5) highest Ba concentrations are observed for English Point (1.7 ppm) and Fort Jesus (1.5 ppm), at the entrance of Tudor Estuary, and lowest for Nyali and Diani (0.5 and 0.7 respectively). For large sized *Isognomon* shells lowest Ba values are observed for Mida specimens (1.3 ppm).





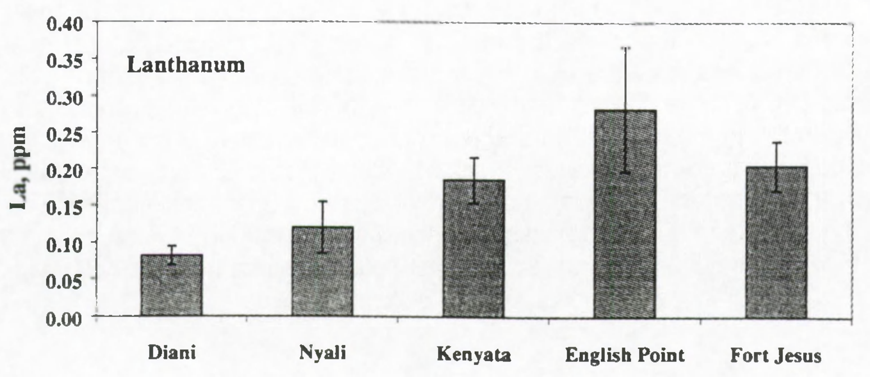


Fig. 5. Ba, Nd and Ce content of small sized (1.5 to 2 cm) whole Isognomon *shells from Diani, Nyali, Kenyatta, English Point and Fort Jesus.*

Rare Eearth Elements (REE)

In general, concentrations of REE Ce, Nd, La, Gd, Pr, Sm, Dy Er are highest (up to 0.51 ppm for Ce) for the *Isognomon* sp. shells (small and large) collected at English Point and Tudor, both sites close to Mombasa (Figure 5; only Ce and Nd shown). When comparing the small *Isognomon* shells, English Point clearly has the highest REE enrichments relative to PAAS reference and Diani the lowest (Figure 6).

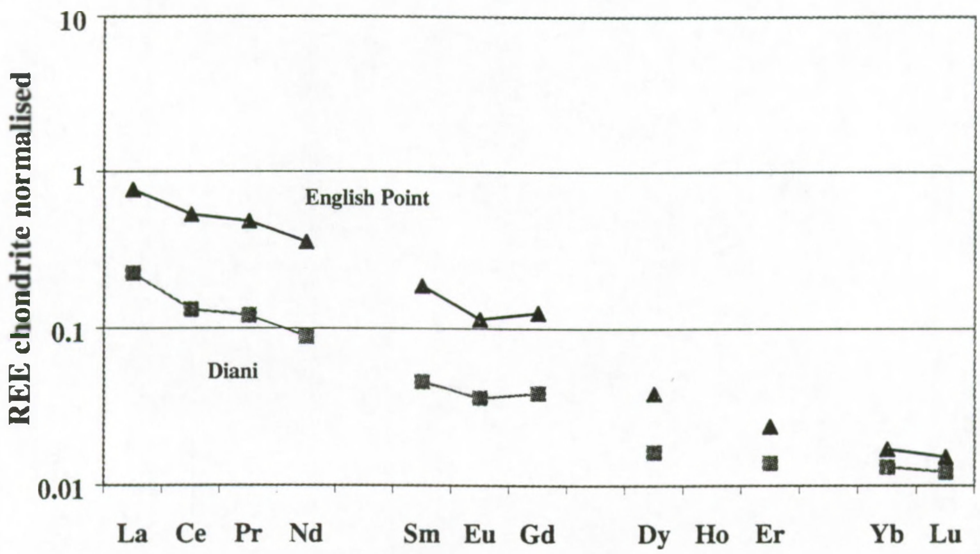


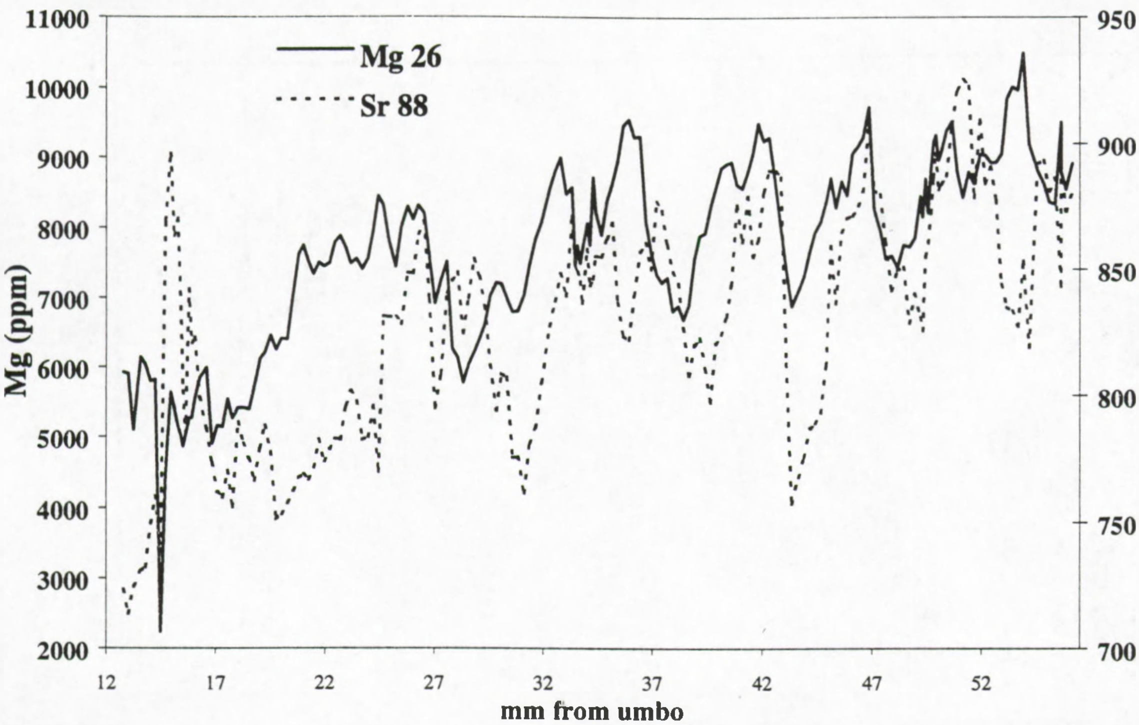
Fig. 6. REE profiles of Isognomon *shells from English Point and Diani; data were normalised against Post Archaean Australian Shale (PAAS) composition*

Temporal evolution of metal concentration as archived by the calcitic layer of Isognomon shells

LA-ICP-MS analysis is less sensitive than total ICP-MS analysis, since much less sample is analysed per ablation. For LA-ICP-MS we obtained reliable concentration data for Mg, Sr, Ba and Mn. Trace metals zuch as Zn and Pb could be detected as well. For Mg, Sr, Ba and Mn we observe some cyclicity of element distribution along the growth axis of the shell, suggesting a seasonal pattern. While for Mg a pattern can be discerned for both the Isognomon shell from Tudor and Mida, for Ba, Sr and Mn, cyclicity is only discernable for the Tudor specimen. For the Tudor specimen we observed that Mg and Sr on the one hand and Ba and Mn, on the other, co-vary rather well, while this is much less so for the Mida specimen (Figures 7 and 8). While Mg and Sr contents are similar for Tudor and Mida specimen Ba and Mn are much more enriched in the Tudor specimen.

Mg profiles are characterised by broad peaks with decreasing width when approaching the date of shell collection. The Ba profile on the contrary (for the Tudor specimen) is characterised by rather sharp, recurrent peaks occurring when the Mg peak is decreasing.

Tudor, T1



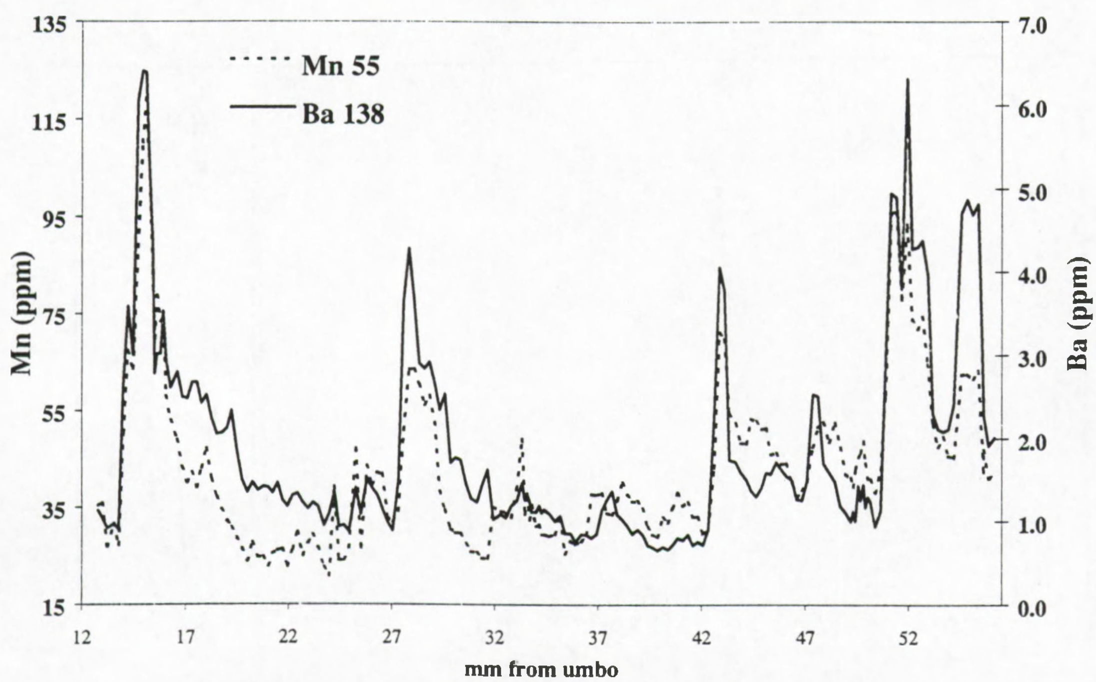
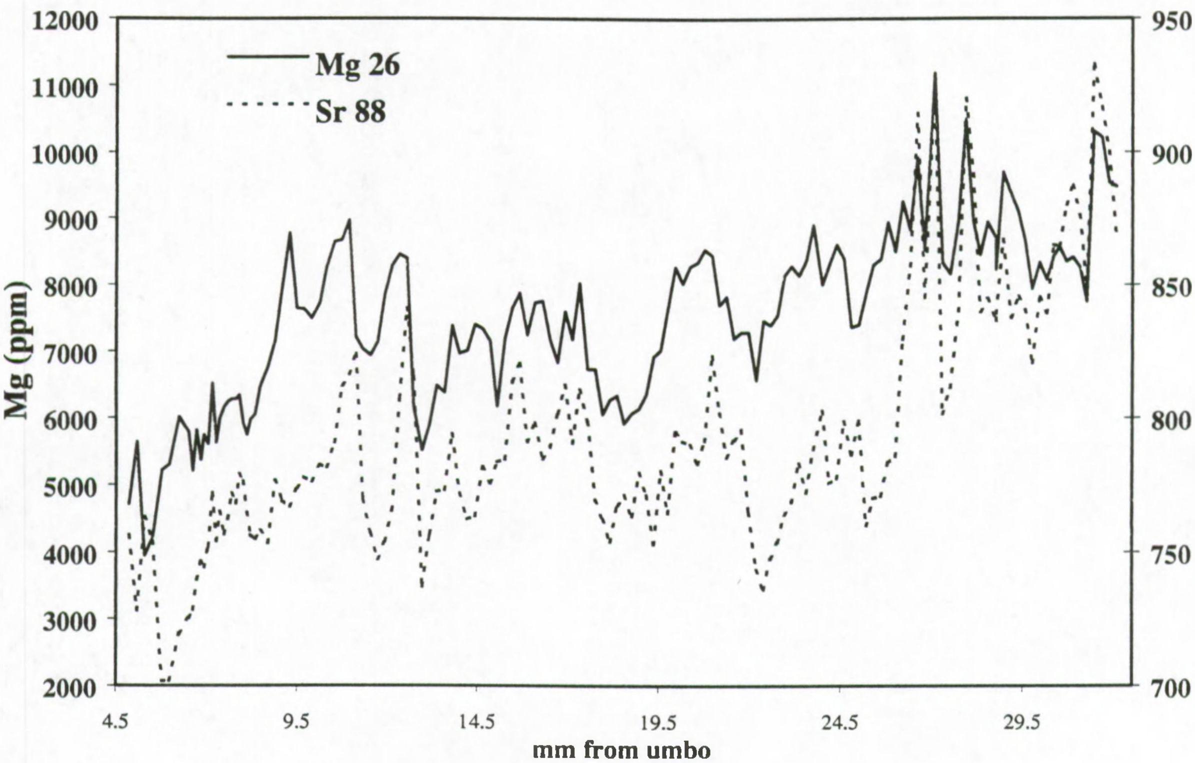


Fig. 7. Mg, Sr, Ba and Mn distribution in the calcitic layer of Isognomon shell T1 from Tudor, analysed by LA-ICP-MS along maximum growth axis

Mida, M2



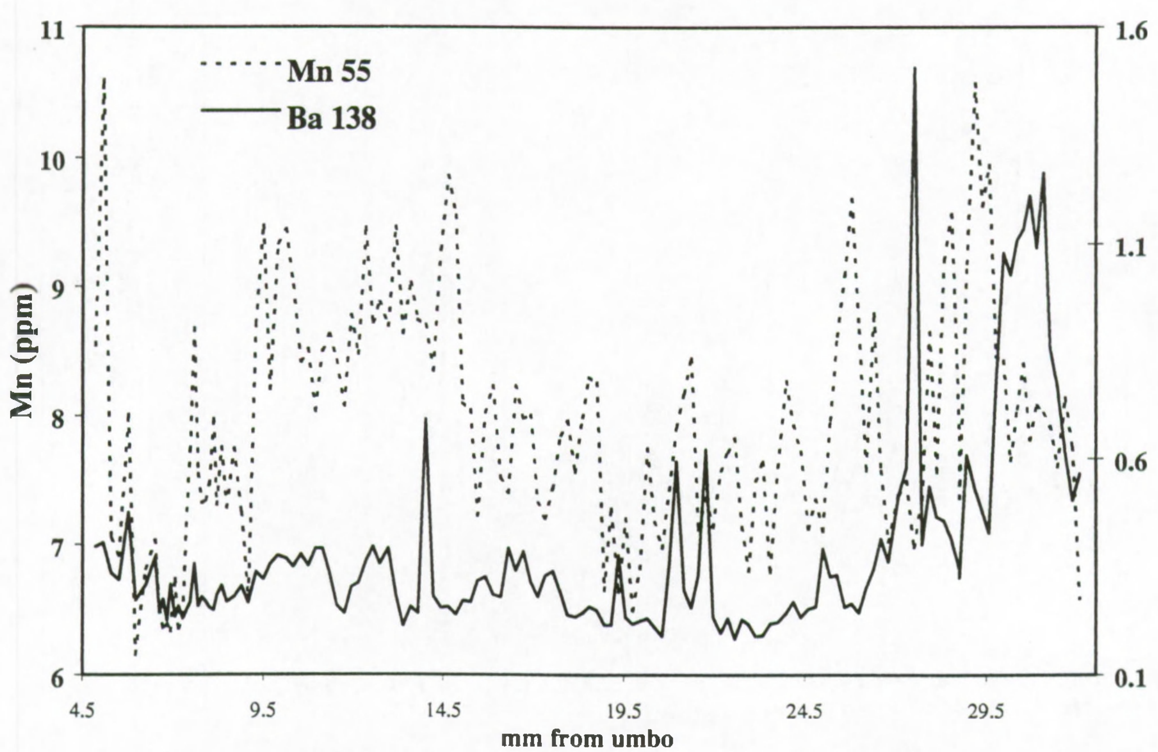


Figure 8: Mg, Sr, Ba and Mn distribution in the calcitic layer of Isognomon *shell M2 from Mida, analysed by LA-ICP-MS along maximum growth axis*

It is well known that the Mg/Ca ratio in biogenic calcite is temperature controlled, with Mg content increasing with increasing temperature, while Sr content would essentially be controlled by changing salinity due to freshwater discharge (e.g. Curtiss and Hodell, 1993). What about the Ba pattern ? Sharp Ba peaks were also observed in Mytilus edulis (blue mussel) from the Scheldt Estuary (Vander Putten et al., 1999 and submitted). For the Scheldt Estuary these Ba peaks appeared to occur in spring and to coincide with the early stages of the spring phytoplankton bloom. In the Kenya setting we expect enhanced input of Ba with enhanced freswater run-off during southeast monsoon. During southeast monsoon rains (April-May), Tudor system behaves truely as an estuary with salinities ranging between < 1 ‰ and 35 ‰ (Okemwa, 1990 and Kazungu pers. commun.). Nutrient input via river discharge and surface run-off is high and induces a characteristic phytoplankton bloom (Okemwa, 1990; M.-H. Daro, pers. commun.). Seasonality of this feature is more pronounced for Tudor Estuary with proportionally larger surface flow of freshwater than for Gazi and Mida, which are characterised by a larger contribution of groundwater flow. It can be expected that in systems with predominance of groundwater flow, freswater input is more constant over time and shows less seasonal variability.

We compared the Mg profiles in the shells of Isognomon specimens from Tudor and Mida with data for SST and precipitation for the period prior to June 1998, the date of shell collection (Figures 9 and 10). This fitting was performed by forcing the SST minima to coincide with the minima in the Mg profile, taking the date of collection as the reference time point. The fitting between Mg profile and SST is clearly much better for the Tudor specimen than for the Mida one. This approach also allows to calculate a year averaged shell growth rate. It can be seen from Figures 9 and 10 that calculated growth rate decreases with age as expected (from 12.8 to 3 mm y^{-1} and from 5.3 to 2.5 mm y^{-1} for Tudor and Mida specimens respectively). From this comparison we deduce that both specimens probably have about seven and a half years of age. Comparing now the Ba profile in the Tudor shell with SST and precipitation, we observe that the sharp Ba maximum generally occurs at the end of the rain period associated with southeast monsoon. Thus, there seems to be a delay of a few weeks between increased run-off of freshwater carrying high contents of dissolved Ba (e.g. Coffey et al., 1997) and the appearance of high Ba concentrations in the shell. It is therefore likely that the Ba is deposited in the shell rather as a result of enhanced ingestion of phytoplankton that has incorporated Ba, than by direct incorporation from solution. This scenario also fits with findings for other estuarine environments (see above).

Why is the seasonal signal so salient for the Tudor specimen and less so for the Mida one? We explain these differences as resulting from the difference in freshwater flow regimes between both systems. Tudor has relatively less groundwater flow but has a stronger surface flow discharge. Mida has no direct freshwater input from a river. With predominance of surface flow over groundwater flow it is expected that seasonal effects are enhanced. Signal of freshwater outflow in Mida is probably much more attenuated because of remoteness of the source region, while for Tudor the source region is closeby. Furthermore Mida is a relatively shallow lagoon type system, as is also Gazi. In such systems we expect significant, relatively short term excursions of the SST and salinity inland, because of shallow water column.

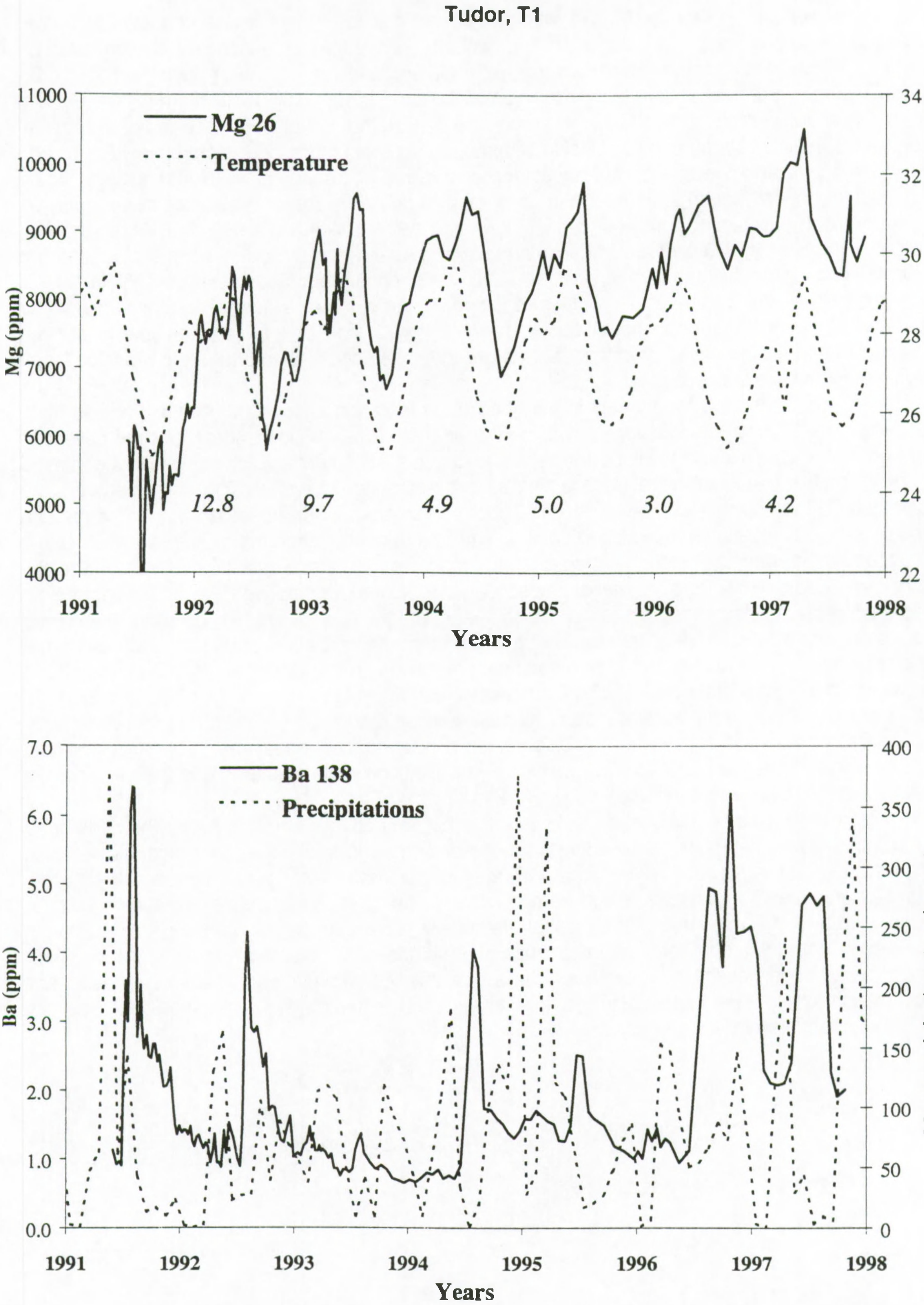


Fig. 9. Isognomon *shell T1 from Tudor; Comparison of Mg profile with Sea Surface Temperature and Ba profile with precipitation. Italicized numbers are estimated growth rates in mm y^{-1} (see text).*

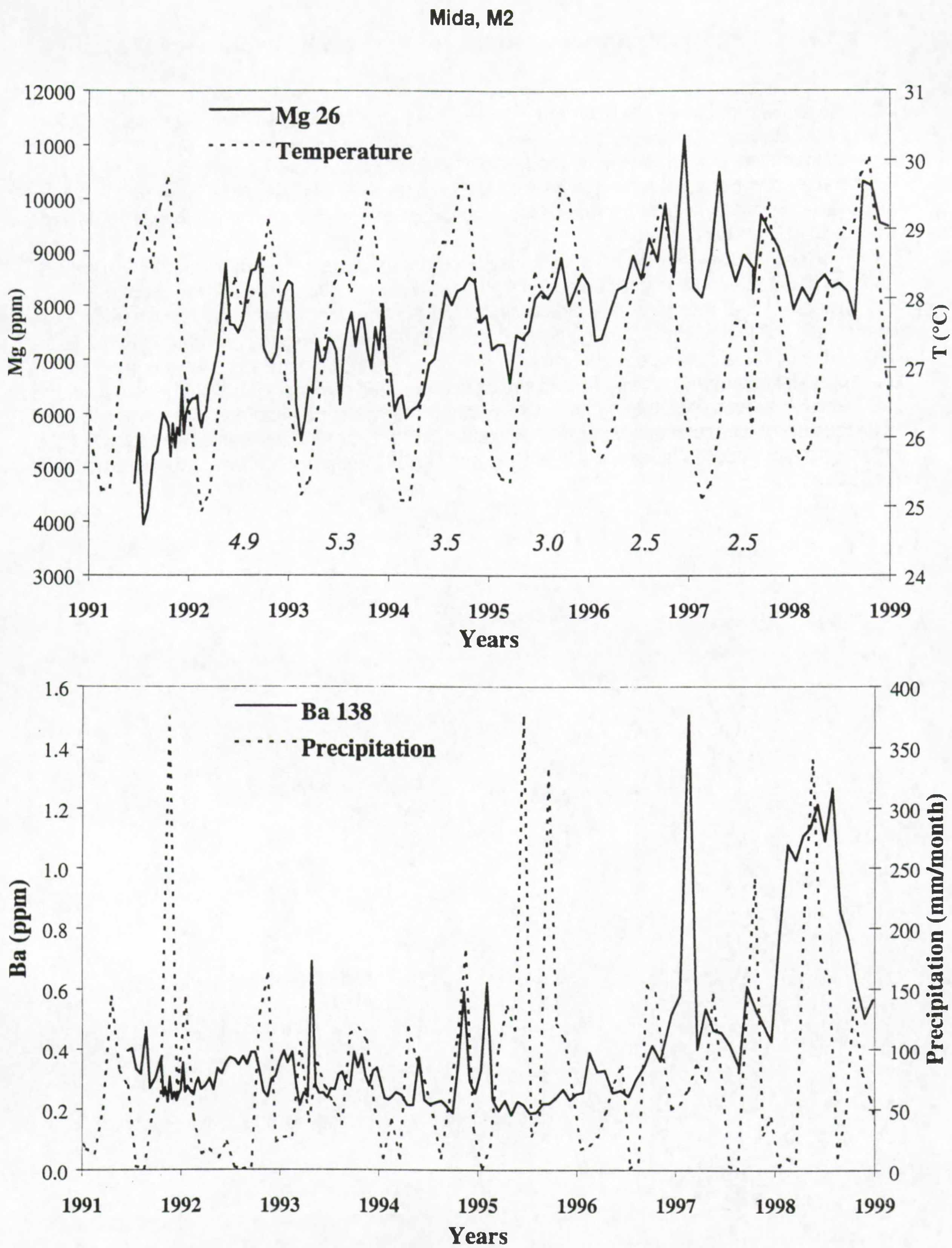


Fig. 10. Isognomon *shell M2 from Mida; Comparison of Mg profile with Sea Surface Temperature and Ba profile with precipitation. Italicized numbers are estimated growth rates in mm y*$^{-1}$ *(see text).*

Temporal evolution of oxygen and isotopic composition in the calcitic layer of an Isognomon shell

The shell of a single *Isognomon* specimen, collected in Gazi (June 1992), south of Momabasa, was analysed for $\delta^{13}C$ and $\delta^{18}O$ in an attempt to visualize the seasonal variations along a growth axis from umbo to rim. The aim of this excercise was to obtain (1) a chronological calibration of the growth of the shell and (2) information on the isotopic record of salinity, temperature and biological activity. The $\delta^{18}O$ of the shell's carbonate is determined by the $\delta^{18}O$ of the seawater and by the temperature dependence of the isotope fractionation factor between water and calcium carbonate.

The $\delta^{18}O$ of the seawater is, to a very good approximation, linearly related to the salinity. Both vary with evaporation and/or dilution with fresh water. On the contrary, the carbon isotopic composition of the shell is barely directly influenced by the water temperature. A temperature change of +1°C makes the $\delta^{13}C$ of carbonate about 0.03 ‰ more negative, compared to 0.2 ‰ for $\delta^{18}O$ in the considered temperature range. The $\delta^{13}C$ the shell's carbonate is mainly determined by the control that biological activity has on the carbon isotopic composition of DIC. Photosynthesis preferentially pumps isotopically light CO_2 out of the water leaving les negative DIC and consequently shell carbonate, whereas respiration and any other oxidation of (isotopically light) organic matter produces isotopically light CO_2, shifting DIC and shell carbonate to more negative $\delta^{13}C$ values.

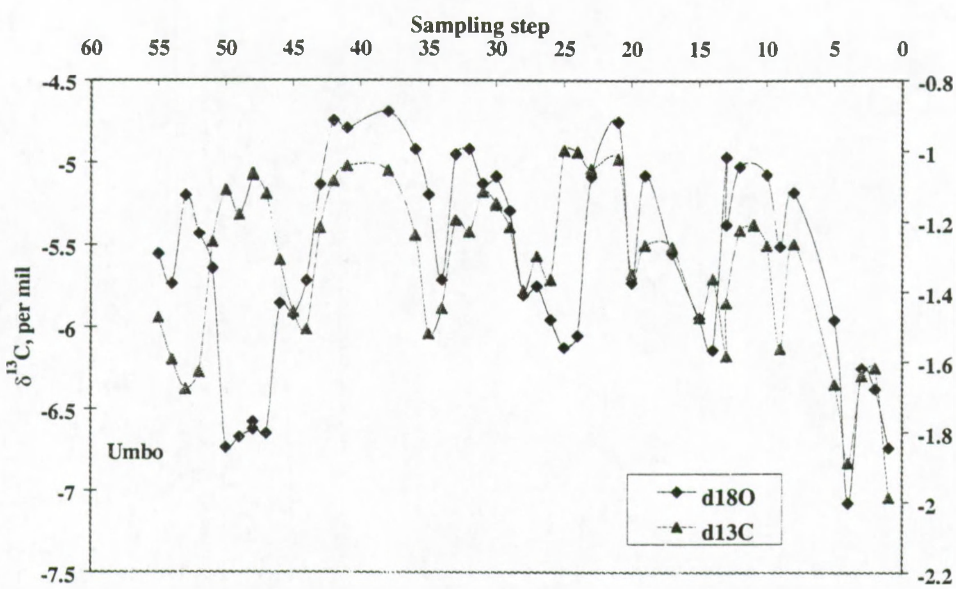


Fig. 11. Evolution of $\delta^{18}O$ and $\delta^{13}C$ in the calcitic layer of a Gazi Isognomon *specimen sampled along its growth axis.*

$\delta^{13}C$ and $\delta^{18}O$ results are reported in Figure 11. They show clear wiggles within a maximum range from about –1.9 ‰ to –0.9 ‰ for $\delta^{18}O$ and from –7.1 ‰ to –4.6 ‰ for $\delta^{13}C$. There is little doubt that these wiggles correspond to seasonal environmental variations. Observed $\delta^{18}O$ values agree with possibly expected values. Taking into account salinities between 35 ‰ and 40 ‰, i.e.

excluding possible temporal and local important fresh water contribution, $\delta^{18}O$ values of seawater can be estimated to fall between 0‰ and +2.2‰, based on observed salinity - $\delta^{18}O$ relations in sea surface waters, as discussed in Railsback et al. (1989). Three water samples from Mombasa, Gazi and Mida Creek had values of +1.07 ‰, +0.35 ‰, and +1.68 ‰ respectively. Carbonates formed in waters with $\delta^{18}O$ between 0 ‰ and +2.2 ‰ at temperatures between 25 °C and 30 °C must yield $\delta^{18}O$ values between +0.3 ‰ and –2.9 ‰ , based on the temperature dependence of $\delta^{18}O$ H_2O (SMOW) - $\delta^{18}O$ $CaCO_3$ (PDB), as established by Friedman and O'Neil (1977).

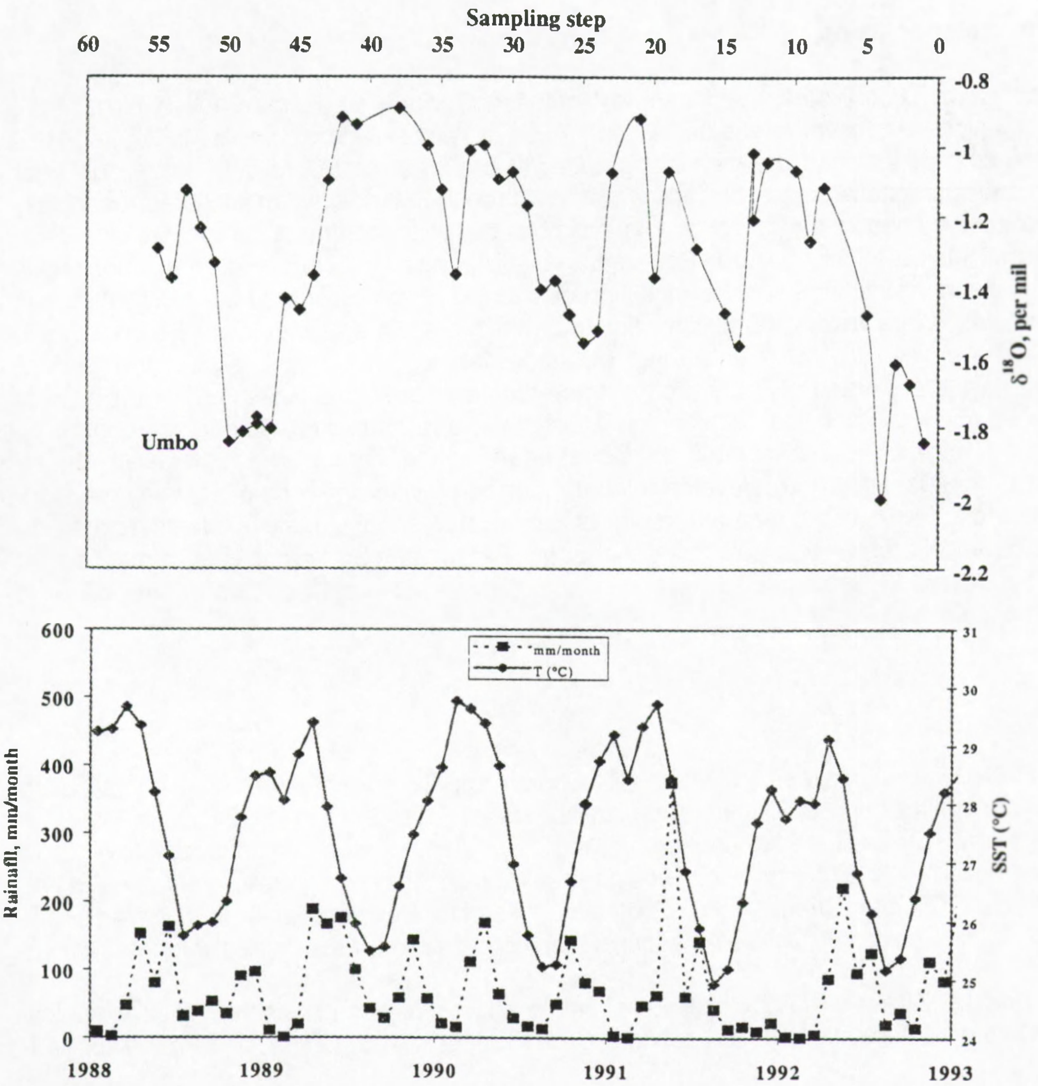


Fig. 12. SST, rainfall and shell (calcitic layer) $\delta^{18}O$ for the years preceding the date of sampling of Gazi Isognomon *specimen collected in June 1992.*

Moreover, the changes may be mainly attributed to the yearly SST changes of 4°C to 5°C (see Figure 12). Indeed, in the concerned temperature range the $\delta^{18}O$ $CaCO_3$ change is of 0.20 ‰ for a 1°C temperature change (Friedman and O'Neil, 1977). We observe that the largest seasonal $\delta^{18}O$

$CaCO_3$ shift spans 1‰ corresponding to the maximum temperature jump of 5°C. Substantially smaller seasonal $\delta^{18}O$ $CaCO_3$ shifts (corresponding to < 4°C temperature changes) may be explained by enhanced evaporation, producing isotopically heavier surface water during warm seasons. This effect reduces the temperature induced $\delta^{18}O$ $CaCO_3$ change.

The seasonal $\delta^{13}C$ shift of roughly 1 ‰ may be explained by increase of respiration and/or aerobic breakdown of organic matter after the main rain season. An intriguing observation is the co-variation of $\delta^{13}C$ and $\delta^{18}O$ throughout the shell (Figure 10), with the exception of the very first part (close to umbo) where $\delta^{13}C$ and $\delta^{18}O$ are anti-correlated. We have no explanation for this observation, which deserves further attention.

Discussion and conclusion

Shells from Tudor Estuary, adjacent to Mombasa (Tudor; Fort Jesus; English Point sites), show typically highest concentrations of Ba and REE. These greater enrichments in Ba and REE occur for the system that has the smallest groundwater outflow (0.067 m d^{-1}). However, Tudor system has a proportionally greater surface water flow than the other sites investigated (especially during southeast monsoon period) and it is possible that part of the observed Ba and REE enrichments is related to this feature. Fort Jesus shells, furthermore, are the most enriched in trace metals, clearly reflecting impact of pollution due to the proximity of Mombasa city. Although Mida system, which has lower groundwater flow than Gazi system also has lower Ba contents in *Isognomon* shells, we do not find strong evidence for a close link between intensity of groundwater flow and storage of Ba and other trace elements in shells. However, for Ba there is evidence from our laser ablation data for an *Isognomon* specimen from Tudor, for control by surface run-off. However, this control on Ba appears to be indirect via uptake of Ba by phytoplankton that is known to develop bloom conditions with the onset of rainy season. Whenever surface run-off is the main route of freshwater supply, bivalve shells appear to represent a potential archive of environmental change. Parameters and processes that appear to be efficiently recorded are sea surface surface temperature surface water run-off and phytoplankton biomass.

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University of Dar es Salaam
Institute of Marine Sciences

Anthropogenically induced changes in groundwater outflow and quality, and the fuctioning of Zanzibar nearshore ecosystems Part I

Mmochi, A.J., Mtolera, M.S.P., Shunula J.P. & Ndaro, S.G.M.
Institute of Marine Sciences, Zanzibar Island, Tanzania

Objectives

The objectives of this study were three fold named:

- To study nutrient and pesticide quantities in sites receiving groundwater.
- To find out the effect of groundwater on macrophyte growth performance.
- To find out the effect of groundwater on both macrofauna and meiofauna.

Methods

Study sites

Study sites were established at four coastal villages named Paje and Chwaka, located on the east coast as well as Fumba and Makoba, located on the west coast of Unguja Island (Zanzibar). The characteristics of the identified sites with regard to possible sources of nutrient and pesticide pollutants were as follows:

Chwaka: The Mapopwe creek and some areas close to the shore in Chwaka bay were identified with sites having groundwater bore-holes. Cheju rain-fed and irrigation rice farms are very close to the Mapopwe Creek. Sites along this creek were of interest as pesticides and nutrients used in rice farms may be affecting the productivity in Chwaka Bay. There is no surface water flow from Cheju to Chwaka.

Paje: Paje beach was also identified with areas having groundwater bore-holes. As the areas surrounding Bwejuu, Paje and Jambiani villages are coral rag areas, with very little agricultural activities, the concentrations of nutrients here were expected to show background concentrations. There are no streams/rivers in the area.

Fumba: A site with vivid ground water bore-hole was not identified. As sites with luxurious growth of *Avicenia* spp. is usually associated with the presence of freshwater sources, a rocky beach adjacent the *Avicenia* spp. growth was taken to represent a site with groundwater source. At this station, samples were taken from a station established in the vicinity of the mangrove stands and away from mangrove in the intertidal region.

Makoba: Makoba bay is at the end of the Mahonda-Makoba drainage basin. The basin has rice farms, a sugar cane plantation and a sugar factory. According to reports from villagers and fishermen, there are occasional fish mortalities in the river. Mahonda sugar factory had a store of expired pesticides some of which have already lost their labels. These were noted to be leaking out and thus it was assumed that during cleaning of the factory some pesticides may reach the nearby river through the tunnel.

Another reason for selecting this site is that the Prisons Department in Zanzibar owns some 10 ha of salt pans at Makoba bay. Recently, the amount of freshwater in the bay has made salt mining inefficient. Some of the salt pans have now been transformed into experimental aquaculture ponds. There are acceptable maximum residue levels recommended for food. It is necessary to ensure that concentrations of pesticides in fish and shellfish at Makoba are below these limits before commercial scale aquaculture commences. Samples were taken from old salt pans, new salt pans and a creek receiving Zingwezingwe and Mwanakombo river waters.

The sampling schedule was as summarised in Table 1.

Table 1. Sampling dates for different study components

MONTHS	NUTRIENT & PESTICIDES	MACROPHYTES	MEIOBENTHOS	MACROBENTHOS
DEC. - FEBR. (DRY)	FEBRUARY 98 P, M, F & C SAMPLED	FEBRUARY 98 P, F & C SAMPLED		FEBRUARY 98 P, M, F & C SAMPLED
MAR-MAY (LONG RAINS)	MAY 98 P, M, F & C SAMPLED	MAY 98 P, F & C SAMPLED	MAY 98 P, M, F & C SAMPLED	MAY 97& 98 P, M, F & C SAMPLED
JUNE-SEPT. (DRY)	JULY-SEPT. 97 P, M, F & C SAMPLED	JUNE 97 P, F & C SAMPLED SEPTEMBER 98 P, F & C SAMPLED	JUNE 97 P, M, F & C SAMPLED SEPTEMBER 97 P, F, M & C SAMPLED	JUNE 97 P, M, F & C SAMPLED
OCT.-NOV. (SHORT RAINS)	NOVEMBER 97 P, F, M & C SAMPLED	NOVEMBER 97 & 98 P, F & C SAMPLED		NOVEMBER 98 P, F, M & C SAMPLED

NB: The letters P, F, M and C stands for Paje, Fumba, Makoba and Chwaka, respectively.

Pesticide and nutrient pollution study methods

Sample collection

Water samples for pesticide analyses were collected in 1 litter teflon capped glass sampling bottles and preserved with 3% dichloromethane. Sediments samples were collected by grab sampler in to 250 l teflon capped glass bottles. Samples for dissolved inorganic nutrients analyses were collected in 250 ml plastic bottles and fixed by using 3 drops of chloroform.

Samples for dissolved oxygen were taken by shallow water, water sampler in the bays or directly by sampling bottles in the rivers and streams. The waters were taken carefully to avoid trapping of air bubbles and splashing. The samples were taken in 125 ml oxygen samples and fixed by using 0.5 ml manganese chloride and alkaline iodide solution. Samples for pH analysis were taken in 20 ml plastic vials. Temperature and salinity were measured by using normal thermometer and refractometer (ATAGO, Japan) respectively. Except for samples meant for the oxygen analysis which were just kept in a dark box, the rest of the samples were put in an ice box immediately after sampling and during transport to the laboratory. In the laboratory the samples were kept in a refrigerator at 4°C when it was not possible to extract or analyse immediately.

Sample analysis

Pesticides: Pesticides from water samples were extracted by shaking with dichloromethane. For the sediment samples, samples were dried and ground with sodium sulphate until the mixture floated freely. The resulting powder was extracted by shaking with several portions of a mixture of cyclohexane and acetone. Pesticide residue analysis was done as shown in Åkerblom (1995), with modifications shown below. All laboratory work was continuously validated by frequent recovery and blank tests. For analysis of pesticides, water samples were injected on SE-30 column and electron capture detector (ECD) and OV-1701 nitrogen phosphorus detector (NPD).

Nutrients and pH: Ammonia concentrationwas determined as described in Strickland & Parsons (1972) and phosphate was determined as shown in Parsons et al. (1984). Oxygen analysis was done by titration (Winkler's method) and pH was recorded using Beckman 34 pH metre, which was standardised at 4, 7 and 10 depending on the range observed. The water volume and ammonia volume transport at Cheju and Mapopwe were estimated by using data from Wolanski (1989) and the mean salinity of the creek at low tide. The water volume and ammonia volume transport in Mahonda-Makoba drainage basin were estimated by measuring water volume in the rivers and the mean salinity at Kiwani creek.

Macrophyte and microphyte study methods

Sample collection

In each area samples were taken in the vicinity of the freshwater bore-hole and at a control site where the seawater had normal salinity. Ten quadrats, each measuring 0.25 m^2 were randomly thrown and the macrophyte composition and biomass assessed. Sampling was limited to a diameter of about 20 m around the freshwater bore-hole or control site. Control sites were established 250-500 m away from the bore-hole. For purposes of assessing the influence of the groundwater to the macrophyte communities, additional sampling had to be done on a permanent transect. In this case sampling was guided by the current direction. Standing stocks (biomass (for dominant species) and species composition) were analysed. Photosynthetic measurements were used to assess the response of selected macrophytes to the suspected stresses.

The species representation has also been assessed 1.5 - 2 km away from the permanent transects and towards the reef and along the reefs using a 0.0625 m^2 quadrat randomly thrown around the sites.

Thus a total of five types of sampling were conducted in each of the stations, that is Chwaka, Fumba and Paje.

Sample analysis

Photosynthetic measurements: Areas receiving groundwater were established to have low pH and salinity. It was therefore of interest to assay the effect of different pH and salinities on the productivity of macrophytes. Thus freshly collected samples were allowed to stay in natural seawater with salinity 10, 20 or 35 or pH regime of around 7.3, 8.2 and 8.5 for a maximum of 2 hours before it was analysed for its photosynthetic response. During such incubations, plants received normal solar radiation ranging between 2000 and 2500 µmol photon m^{-2} s^{-1}. Natural seawater has pH 8.2 ± 0.2 and salinity of around 35‰ and therefore plants subjected to such conditions were considered as controls. *Gracilaria salicornia, Eucheuma denticulatum, Ulva reticulata, Ulva fasciata, U. rigida* and *U. pertusa* were the macroalgae used and seagrass species were *Thalassia hemprichii* and *Thalassodendron ciliatum*.

Assay of photosynthetic activity by oxygen evolution method: A dose response curve of photosynthetic oxygen evolution was measured as a function of photon flux densities using the Illuminova light dispensing system (Illuminova, Uppsala, Sweden) capable of supplying 0 - 2870 µmol photon m^{-2} s^{-1} (PAR, 400 - 700 nm). Algal thalli or 2nd/3rd tender seagrass leaf tips were incubated in 3 ml seawater. When the salinity was varied, the pH was maintained at around 8.2 and when the pH was varied, salinity was kept at around 35‰. Light exposure started with 2 minutes of darkness, then light intensity was increased exponentially for 3-4 minutes to 2879 µmol photon m^{-2} s^{-1} depending upon the light saturation value of a plant. 600 data points were sampled during 11-12 minutes. Maximal photosynthetic activities or light saturation values were calculated as the interception of a regression line calculated from 30 data points around the estimated maximal photosynthetic output and light saturation value, respectively. Total dissolved inorganic carbon was around 2.2 mM. Photosynthetic O_2 evolution was measured at 25ºC.

Chlorophyll fluorescence measurements. In vivo chlorophyll fluorescence was measured at room temperature with a Plant Efficiency Analyser (PEA, Hansatech Instruments Ltd., England). Thereafter, algal/seagrass tips were dark-adapted for 5 min before measurements took place. Maximum light intensity at the thalli surface was >3000 µmol photon m^{-2} s^{-1}. On completion of the measurements, the parameters minimal fluorescence (Fo), maximal fluorescence (Fm), variable fluorescence (Fv) and Fv/Fm were automatically calculated from the recorded data.

Determination of the growth rate of *Eucheuma denticulatum* in the field: To study the effect of groundwater on the productivity of an economically important macrophyte, *Eucheuma denticulatum* farms owned by the villagers were used. DGR for each monoline was determined and the mean for ten monolines was recorded. Measurements for the season were recorded for a month. DGR as a percentage, was calculated using the formula: $DGR = [(W_t/W_0)^{1/t} - 1] \times 100$ where W_0 and W_t were initial and final biomass at day t, respectively (Lignell *et al.*, 1987). The fresh weights were determined following a brief gentle shaking to get rid of excess water. Variation in weight values using this approach was ±0.5% and ±5%.

Benthic microalgae study methods

<u>Sample collection and analysis</u>

Biomass: The biomass was estimated by determining the amount of chlorophyll 'a' in sediment. Sediment samples were collected into 20 ml plastic vials using a corer made out of a 60 ml plastic syringe. Only the upper 2 cm sediment layer was taken. In the laboratory the sediment samples were freeze-dried and homogenised. Then 0.5 gm of dry homogenised sediment was transferred into a glass centrifuge tube in which 10 ml of 90% acetone was added. The tube was left to extract chlorophyll over night (approximately 12 hrs) in a refrigerator. Prior to measurement, the samples were centrifuged at a speed of 10,000 rpm for 10 – 15 minutes to ensure an absorption at 750 nm of less than 0.005. The supernatant was decanted into a glass cuvette (1 cm path length) and the extinction of the extract was measured at 664 and 630 nm. The amount of chlorophyll in the original sample was then obtained using the relationship by Jeffrey and Humphrey (1975). Chl 'a' (µg / g dwt sediment) = $11.93 * A_{664} - 0.4 * A_{630}$

Species composition: Samples for microalgae species composition analysis were collected in a similar way as those intended for biomass analysis. These were however immediately fixed with 4% formalin. Species identification was done using a light microscope.

Macrobenthos study methods

The study was aimed at deploying the abundance, biomass (AB Curves) comparison of macrobenthos in the study sites. It is understood that abundance and biomass vary quite markedly as a function of water quality.

<u>Sample collection</u>

Sampling was conducted during spring low tides.

1. Each study site was divided into two sub-zones, A and B 50m apart.

2. Four replicate substrate samples were taken at each sub-zone using a metallic corer of size 20x20x25cm. The core was pressed into the substrate to a depth of 20 cm, and using a spade, a substrate core/block of size 20x20x20 cm was extracted.
3. The substrate core was wet sieved in sea water with a sieve of mesh size l mm.
4. The residue (retained material) was stored in a plastic bag, with a label indicating the site reference.
5. Measurement of temperature and salinity of the water in the hole created by the extraction of the substrate core was taken using an ordinary thermometer and a refractometer (ATAGO, Japan).
6. Two substrate samples were also taken from each sub-zone for (a) granulometric analysis, (taken from the lower, middle and surface of the core) and (b) for the determination of organic matter content (by loss on ignition method). For this purpose samples were taken from superficial sediment, l cm deep, using a small plastic petri dish to include a volume of 5 mls.

Sample analysis

1. At the laboratory, the field sieved samples were dyed with a small amount of Bengal-Rose and fixed with 4% formalin, by adding a little quantity to each plastic bag and then letting the samples to stay overnight.
2. a) After fixation, each sample was washed carefully over a l mm sieve with tap water to remove formalin.
   b) To each sample, Ludox colloidal silica was added.
   c) The remaining residue/gravel was then manually sorted to remove all possibly remaining animals, and placing them in the relevant vial each time.
3. The animals recovered from the respective substrate samples were then identified and assigned to family groups, and their abundance and biomasses worked out.

Meiobenthos study methods

Sample collection

Sampling was conducted on three dates (June, September, 1997 and May 1998). These dates were devised such that they cover both the wet and the dry seasons in Zanzibar. Each area (Paje, Chwaka, Fumba and Makoba) was visited once during spring low tide on each sampling date for collection of samples for meiobenthos counts, sediment organic content and mean grain size. At each of the locations a freshwater site and a normal saline site were established. Three random samples of sediment were taken at each site for meiobenthos, organic content and mean grain size analysis. A polycarbonate tube with cross-sectional area 10 cm^2 was pressed into sediment 5 cm deep (during the first two sampling dates but up to 10 cm deep on later sampling dates after the Yerseke Workshop recommendations), corked and retrieved. Sampling deeper than 5 cm was only possible in Paje and Fumba because in Chwaka Bay only a shallow sediment layer covers a rocky bottom. The samples for meiobenthos were preserved in 5 % formalin.

Sample analysis

Samples for meiobenthos counts were washed over a sieve of mesh size 40 μm with tap water and then the animals were extracted from sediment by Ludox MT 40 (colloidal silica) maintained at specific gravity 1.15. Animals were sorted out and identified to major taxa, and counted under a stereo dissecting microscope (Magnification. x 25). No identification to lower levels was attempted. Animal counts (densities) were compared for different locations, sites and sampling dates using 2- way analysis of variance ANOVA. Multidimensional scaling ordination was done to distinguish between sites receiving fresh water and those of normal saline conditions.

Results

Surface-, ground- and coastal-water qualities

Nutrients

The results of analysis of nutrients found in the water column in Paje, Makoba, Chwaka, and Fumba are summarised in Figure 1. Whereas, phosphate concentrations were generally low in all the sites, ammonia concentrations were significantly high is some of the sites.

Paje: The concentrations of ammonia in the well waters in Paje are higher than the freshwater sites with salinities below 10 ‰ (Fig. 1 a). This is an indication that the groundwater is a source of ammonia to the sea. The well water ammonia concentrations ranged from 6.42 - 0.24 Fg.at.N.l^{-1} in the wells depending on the season of sampling, highest concentrations being in the dry season and the lowest in the late wet season. The concentrations of ammonia in Paje are higher than expected of background levels in Zanzibar. It is possible that aquifers or underground water streams/river are inter-linked to agricultural areas. The booming tourist development in the area, which has been shown to have a significant influence on the groundwater budget and quality (data not shown) is also a possible source of higher nutrient levels in the groundwater.

Makoba: In the Kiwani creek, where the rivers meet in the Makoba bay, the concentrations of ammonia were variable (Fig. 1 b). In November 1997, the ammonia concentration in the creek was 31 Fg.at.N.l^{-1}. The upper limit for ammonia concentrations in domestic water supplies (Ministry of Lands, Water, Housing and Urban Development, 1986) is 30 Fg.at.N.l^{-1}. In a brackish water pool in Makoba the ammonia concentration recorded in November, 1997 was around 78 Fg.at.N.l^{-1}. Mean ammonia concentration was 16 Fg.at.N.l^{-1}.

In parallel studies of water quality in the rivers in the Mahonda-Makoba drainage basin, it has been shown that the concentrations of ammonia in the Zingwezingwe tributaries and Mwanakombo rivers ranged from 9.0 Fg.at.N.l^{-1}, in Zingwezingwe river to 0.69 Fg.at.N.l^{-1}, in Mwanakombo. The ammonia concentration in the tunnels was on average of 8.8 Fg.at.N.l^{-1}. The ammonia concentrations in the Zingwezingwe main river was 6.0 Fg.at.N.l^{-1}, indicating that the water here is a mixture of tunnel waters and the tributaries. Ammonia concentration of 22.0 Fg.at.N.l^{-1} was measured in rain water pools in the sugar cane plantations.

In both studies, the concentrations of ammonia were highest in November. The exceptionally high ammonia concentrations recorded during November indicates an anthropogenic source other than rice farms where chemicals are applied in March and April. The source may be industrial wastes from Mahonda sugar factory. The concentrations in a freshwater well water in Makoba ranged from 1-5 Fg.at.N.l^{-1}.

Chwaka: Samples analysed from Mapopwe creek of the Chwaka bay, which is supposed to receive effluents from Cheju rice farms have shown consistently low levels of ammonia ($\leq$ 3.31 Fg.at.N.l^{-1}) (Fig. 1 d). In Cheju rice farms, the ammonia concentration recorded was 108 Fg/at N/l. Although this reading was recorded in November, high readings were recorded in the Cheju irrigation farm throughout the period. While it is necessary to check the other species of nitrogen, it is suggested here that the mangroves abundant in the area are trapping nutrients and other wastes from the rice farms. This is contrary to the popular belief that mangrove stands are sources of nutrients to the neighbouring ecosystems (Mmochi, 1993; see also the estimation of ammonia load to Mapopwe creek in Table 3).

Fumba: The concentration of ammonia were consistently high in Fumba (14.0 Fg.at.N.l^{-1}) compared to Chwaka (1.5 Fg.at.N.l^{-1}) and Paje (2.8 Fg.at.N.l^{-1}) (Fig. 1 c). The salinities in Fumba were also high. Fumba stations had no obvious anthropogenic sources for nutrients.

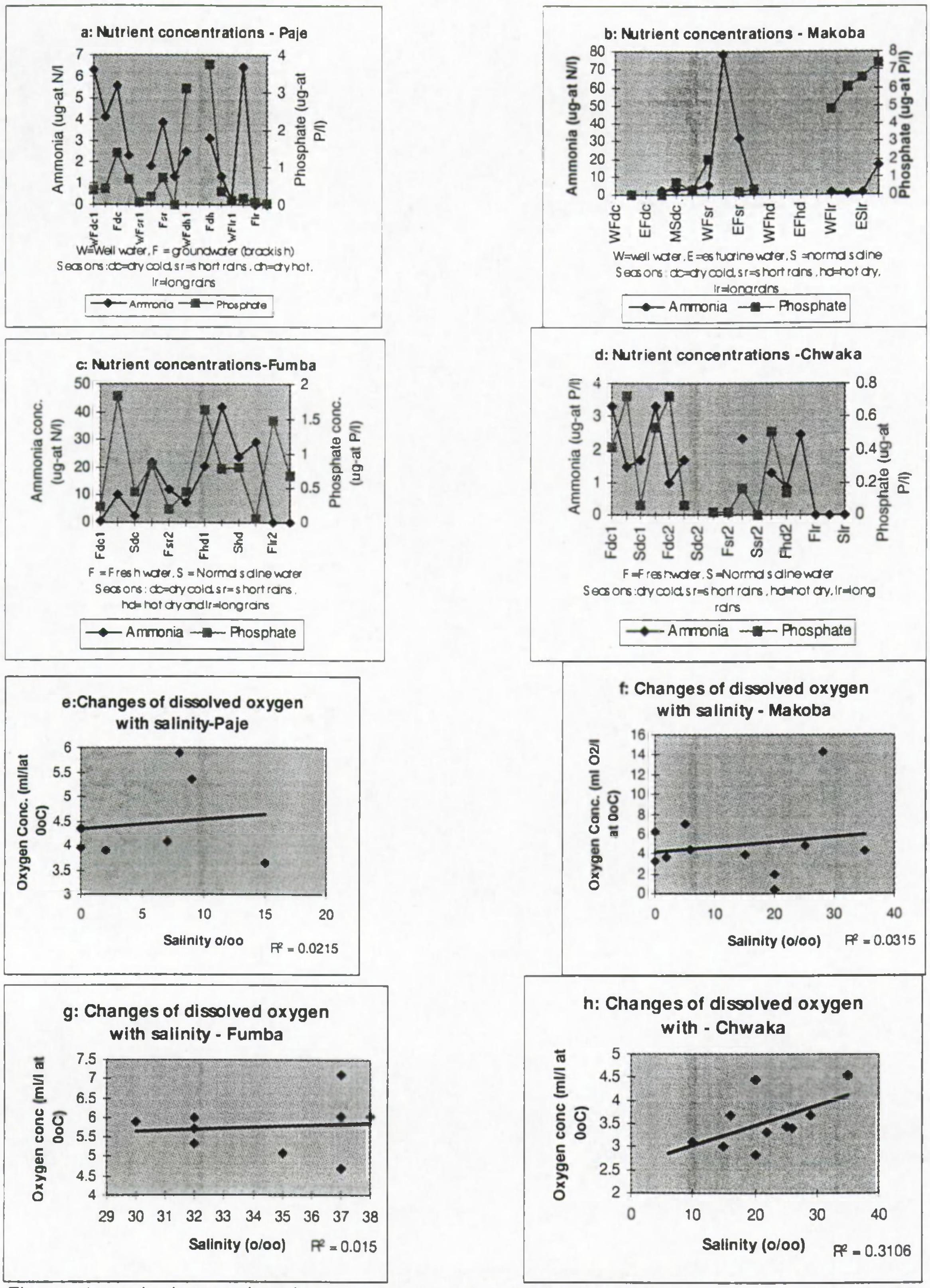


Figure 1: a-d shows phosphate and ammonia distribution patterns in groundwater sites, freshwater wells and normal saline sites in different seasons. e-h show correlation between dissolved oxygen and salinity.

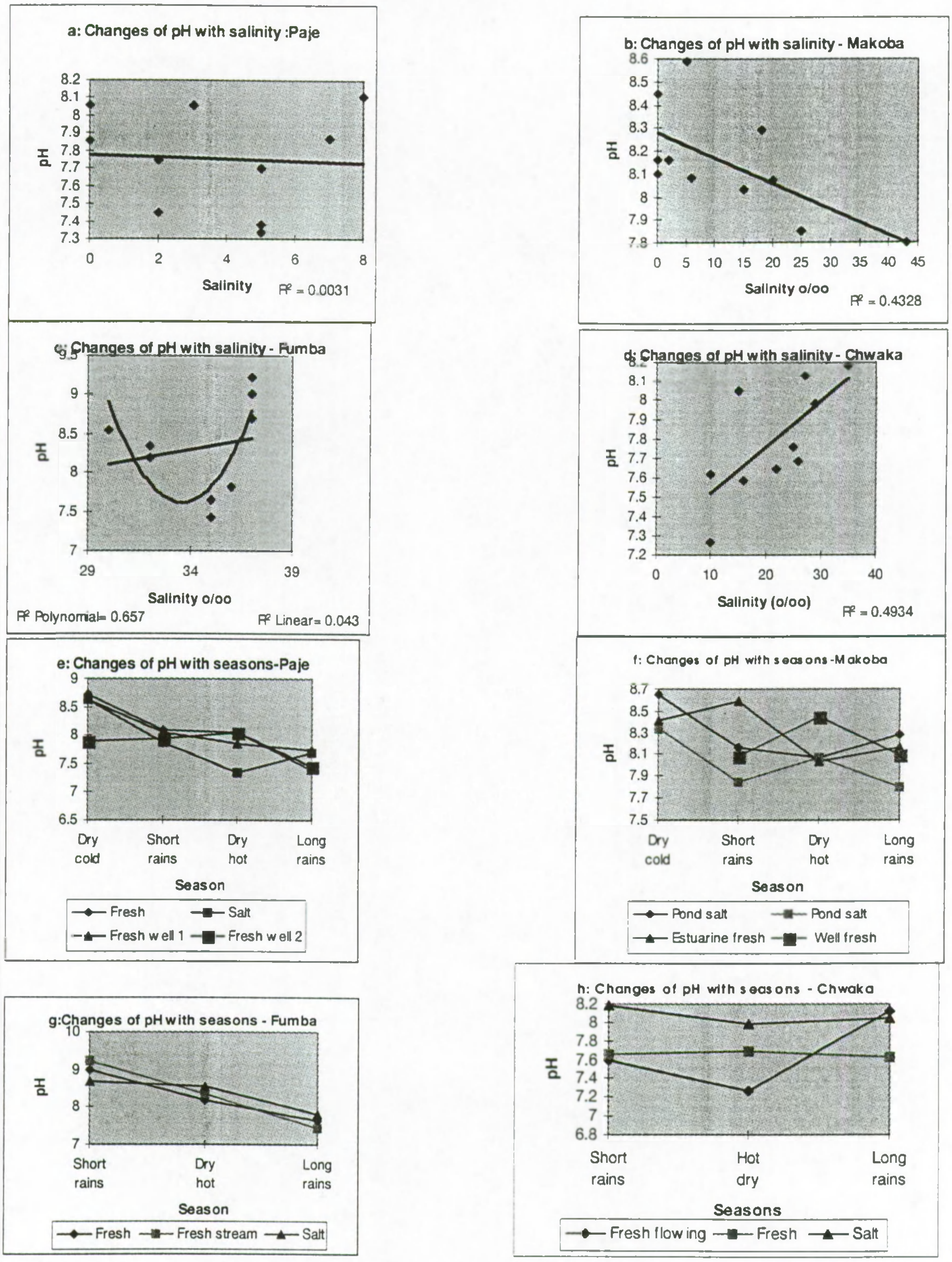


Figure 2: a-d show correlation between pH and salinity. e-h show variations of pH between sites and seasons.

Oxygen

Oxygen concentration at Makoba was highly variable ranging from 14 in the normal saline ponds to 0.4 $ml.l^{-1}$ at 0°C in the normal saline mangrove waters. In the Kiwani estuarine waters, it ranged from 7 to 3 $ml.l^{-1}$ at 0°C (Fig. 1 f). The values at Chwaka ranged from 5 to 3 $ml.l^{-1}$ at 0°C (Fig. 1 h) while at Fumba they were consistently high, ranging from 7 to 5 $ml.l^{-1}$ at 0°C (Fig. 1 g). This is interesting considering that the salinity at Fumba was the highest of all the stations measured. Solubility of oxygen in water decreases with increase in salinity and temperature.

Paje and Chwaka have been shown to have the lowest concentration of oxygen. These areas were also found to have the highest amount of organic matter (see section on sediment analysis). Organic matter decomposition leads to low oxygen levels (Anderson, 1987).

pH

The results for pH are indicated in Figure 2. a-h pH values ranged from the highest of 9.21 in Fumba to 7.27 in Chwaka. High pH values were recorded in Cheju irrigation farm. The values in Cheju decreased downstream starting from the irrigation farm to the lowest in the rain-fed farm in April and May but was consistently high in February and irregular in November. The farming season for rice is in March, April and May. Agrochemicals are used in the rice farms during this period. The pH values in Cheju rain-fed rice farms were very low (pH 6.62) in this period probably due to herbicides (Satunil) that were sprayed during the season. Relatively lower values of pH were obtained in Chwaka when compared with Paje, Makoba and Fumba. The acceptable pH values for coastal waters are 6.5-8.5 (DENR, 1990).

Pesticides

Pesticides that have been identified from different sites are shown in Table 2. The pesticide lindane seems to be consistently present in all the study sites. Furthermore, DDT seems to be more closely associated with townships rather than agriculture and may be a result of fumigation.

With the view to estimating the potential of mangroves as nutrient trapping system in Chwaka and Makoba, ammonia-N concentration measured at ground water sites were compared with values measured at their possible sources. The estimation of the nutrient loads shown in Tables 3 A and B took into consideration the volume of freshwater flowing into the relevant near-shore ecosystems or through an established reference point.

(i) Volume of water at Mapopwe creek at low tide

Mapopwe creek, in Chwaka consists of the cell area numbers A18, A19, A20 and A31 (Wolanski, 1989). Taking the mean width of the Mapopwe creek being around 50 m, the volume of Mapopwe creek at mean sea level (MSL) is around $8.8x10^5$ m^3

(ii) The volume of freshwater in Mapopwe creek:

The salinity of Mapopwe creek at low water is 19.25‰. Salinity difference between low water and high water is around 15.75‰. Thus the fresh water inflow into Mapopwe creek is about $4x10^5$ m^3.

(iii) Volume of river water in Mahonda-Makoba basin

The volume of freshwater in the rivers was estimated by taking the length of the bridge and measuring the speed with which a floating plastic crosses the bridge. The water level on the bridge was taken at 10 cm intervals and the average water volume is estimated at 814,320 tons $.(12\ hrs)^{-1}$

Table 2. Pesticide values in water in selected sites in Unguja

Location/Season	Salinity (ppt)	Pesticide	Concentration (ppb)
PAJE			
Fresh water well (short rains)	2	HCH gamma	0.03
		HCH delta	0.08
Fresh water site 1 (dry season)	3	HCH gamma	0.07
		HCH-delta	0.19
		DDD pp'	0.41
		DDT pp'	0.25
Fresh water site 2 (dry season)	5	DDT pp'	0.68
Fresh water well 1 (dry season)	0	HCH gamma	0.2
		HCH delta	0.07
		DDD pp'	0.44
		DDT pp'	1
MAKOBA			
Salt pond (dry season)		HCH-alfa	0.13
		HCH-gamma	0.08
		HCH-delta	0.42
		DDT pp'	0.28
Salt pond (dry season)		HCH alfa	0.05
		HCH gamma	0.04
		HCH delta	0.03
Salt pond (short rains)		HCH alfa	0.37
		HCH gamma	0.64
		HCH delta	0.05
FUMBA			
Freshwater site 1 (dry season)	32	HCH-gamma	0.15
		HCH-delta	0.17
Normal saline site (dry season)		HCH-alfa	0.06
CHWAKA			
Freshwater site 2 (dry season)	26	HCH-delta	0.07
		HCH-alfa	0.19
		DDD pp'	0.75
Fresh water spring (long rains)	27	HCH-alfa	0.05
		HCH-gamma	0.02
		HCH-delta	0.1

Note: In all the sites, sampling has been taken from both fresh water sites and normal saline sites. Where any of the two has not been shown indicates that no pesticide has been detected.

Table 3 A: Estimation of ammonia load trapped in mangrove stands in Chwaka freshwater site

AREA	AMMONIA ug-at N/l)	AMOUNT OF WATER (TONS)	AMMONIA LOAD (g-at)
Cheju	30	400,000	12,000
Chwaka	1.5	880,000	1,320
Trapping in the mangroves			10,680

Table 3 B: Estimation of ammonia load brought to Makoba bay by underground-water. The additional sources for ammonia may be Mahonda sugar factory and/or the sugar cane plantation.

AREA	AMMONIA ug-at N/l)	AMOUNT OF WATER (TONS)	AMMONIA LOAD (g-at)
Mahonda rivers	5	814320 tons/12 hrs	4071
Mahonda sugar factory & sugar cane farm			+ 7,929
Makoba bay	11.92	1,1x 10^5 tons (low tide)	12,000

The difference in the ammonia concentrations that have been estimated at various sources and sinks may be due to the ability of mangroves to trap such nutrients. It is becoming increasingly accepted that mangroves are a sink rather than a source of nutrients to the other marine ecosystems. The volume of ammonia into Makoba bay is higher than values measured in rivers. The difference (7929) per tide cycle suggests an input from the sugar factory and/or sugar cane farm through underground-water.

Sediment analysis

The results for the sediment analysis are summarised in Table 4 and Figure 3. All the sites have very poorly sorted soils/sediments. This implies that this similarity in sediment type would similarly affect the infauna in all the sites. Chwaka freshwater point appear to have the highest percentage of organic matter.

Table 4: Grain size statistics for different sites

SITE	DATE	MEAN	SORTING	SKEWNESS	KURTOSIS
Makoba GW site	Febbruary	1.308	0.994	-0.1107	1.116
Makoba NS site	Febbruary	1.346	1.078	-0.0113	1.107
Chwaka GW syte	May	1.022	1.980	-0.382	0.860
Chwaka NS site	May	0.327	1.175	0.002	1.942
Fumba GW site	May	0.907	1.574	-0.652	1.425
Fumba NS site	November	0.774	1.133	0.014	1.290
Paje GW site	Febbruary	2.857	1.577	-0.628	1.305
Paje NS site	Febbruary	3.039	1.374	-0.496	1.643

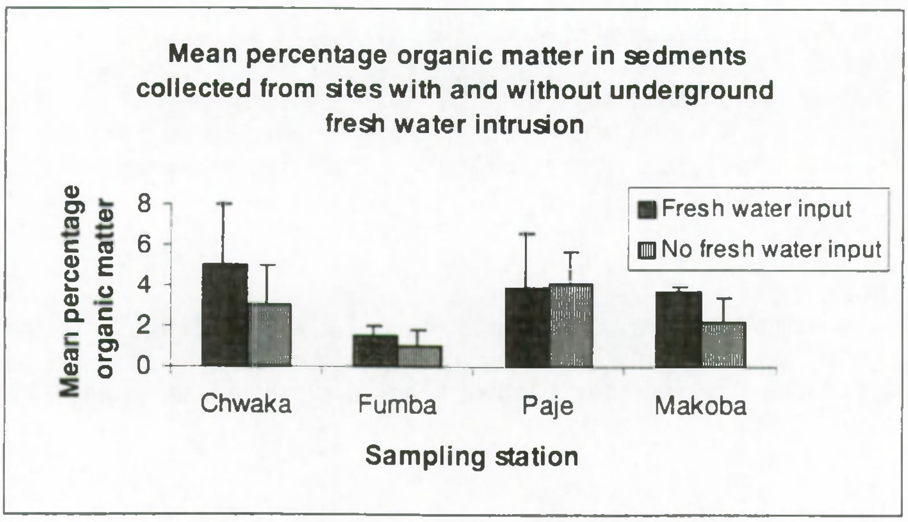


Figure 3. The status of the organic matter content for the different sites

Macrophytes and benthic microalgae

Macroalgal species composition in the different sites

A total of 86 macroalgal species comprised of Chlorophyta (58%), Phaeophyta (23%) and Rhodophyta (18) have been identified in all the sites (See Appendix 2.1). Ground water sites in Chwaka and Paje had the lowest species composition (see Table 5 A & B).

Table 5A: Shown in (A) and (B) are the Chlorophyta (C), Rhodophyta (R), and Phaeophyta (P) identified from the different study sites in Chwaka, Fumba and Paje. Sampling was done as detailed in the materials and methods. A total of 60, 30, and 76 species have been identified in Chwaka, Paje and Fumba, respectively during the whole duration of the study. The number of macrophyte species identified in sites located in the vicinity of the ground water bore hole (GW) and normal saline (SW) sites. The transect here includes both GW and NS sites.

Study sites	**Algae:**			**Comment**
	C	P	R	
Chwaka	21	5	4	Represented here are nearly 50% of all the algal species identified from Chwaka, and that is about 30% of all the species identified from Chwaka, Paje and Fumba are represented here.
Groundwater	5	0	1	Largest biomass made up of *Ulva spp.* 3.1 kg (DW) m^{-2} and *G. salicornia* 2.0 kg (DW) m^{-2} recorded during May 1998. In both cases, dry seasons had quantities ≤1.0 kg (DW) m^{-2}
Normal saline	8	3	2	Largest biomass made up of *Ulva spp.*, 0.7 kg (DW) m^{-2}, and *G. salicornia*, 0.5 kg (DW) m^{-2}, recorded during May 1998. Dry seasons had quantities below 0.3 kg (DW) m^{-2}
Transect	14	4	3	Areas located besides the groundwater had significant quantities of green algae and the red alga *G. salicornia* but the quantities decrease as the normal saline site was approached.
Paje	6	3	2	Represented here is less than 40% of all the species identified from Paje and that is only 12% of all the algal species identified from Chwaka, Paje and Fumba are represented here.
Groundwater	3	0	0	Species of the genus *Ulva* were the only macroalgae found around the bore hole (0.2± 0.03 kg (DW) m^{-2}). Differences between seasons were not significant as most of the area is sandy and macroalgae could not easily attach themselves.
Normal saline	4	3	2	No peculiar quantities of macroalgae recorded.
Transect	4	3	2	No peculiar quantities of macroalgae recorded.
Fumba	25	9	6	Nearly 50% of all the species identified from Fumba are represented here and that is about 45% of all the species identified from Chwaka, Paje and Fumba.
Groundwater (transect)	20	9	5	Species of *Ulva* (1.0 kg (DW) m^{-2}) dominated the first 10 m around the suspected groundwater outlet. Thereafter the species richness increases tremendously.
Normal saline site (transect)	17	8	6	With exception of tidal pools, the species richness is almost evenly distributed.

The difference in the species representation between Fumba freshwater and normal saline sites was not significant. The salinity difference between the Fumba sites was also insignificant. The species composition increased with distance from the freshwater site in the order: reef transect >1-5-2 km transect > ground-water site transect (Compare transects in Table 5 A& B.).

Table 5 B: The number of macrophyte species identified from transects found in the reefs and 1.5-2 km away from the transects established besides the groundwater bore-hole.

Study sites	Algae:			Comment
	C	P	R	
Chwaka	42	10	8	67% of all the algal species identified from Chwaka, Paje and Fumba are represented here.
1.5-2.0 km away from the transect.	15	6	5	The area is dominated by seagrasses, mainly *Enhalus ecoroides*. Sedimentation is high. The rocky shore is frequently visited by fishermen.
Reef	38	9	8	In terms of fisheries, this is one of the most productive reefs in Zanzibar. Fisheries catch here comparable to Fumba (personal communication with Department of Fisheries, Zanzibar)
Paje	13	8	9	Only 33% of all the algal species identified from Chwaka, Paje and Fumba are represented here.
1.5-2.0 km away from the transect.	7	6	6	Most of the algae attached to sticks and monolines used for seaweed farming. Some attached to rocks.
Reef	12	5	8	No peculiar quantities of macroalgae recorded. An interesting aspect here is that the local '*Eucheuma'* species strives here as the case of Chwaka and Fumba. However the fisheries here is not as productive as Chwaka or Fumba.
Fumba	43	19	14	85% of all the algal species identified from Chwaka, Paje and Fumba are represented here.
1.5-2.0 km away from the transect.	35	19	13	Species richness comparable to the reef as this part of the beach is rocky. There is also a rocky pool. 'Local *Eucheuma*' also strives fairly well here especially during the dry season as seaweeds are frequently exposed and the effect of rain water to such species could be serious. See also results for photosynthetic measurements for discussion.
Reef	37	16	11	Species richness is almost evenly distributed. In terms of fisheries, this is one of the most productive reefs in Zanzibar. Fisheries catch here comparable to Chwaka (Personal communication with Department of Fisheries, Zanzibar)

Note: A total of 86 species have been identified from different sites. One interesting aspect is that, the reefs of Chwaka, Paje and Fumba are all characterised by good growth of local '*Eucheuma*' species.

The species composition in the different study sites were also analysed in relation to seasons (see Table 6). Results shows that remarkably few species were encountered during the long rainy season. During the rainy season, species of the genus *Ulva* grew more luxuriously in all the study sites and in fresh water sites they had significantly high biomass (see table 5 A & B and further discussion below). The highest green algal blooms were encountered in Chwaka fresh water site. Paje fresh water site did not have an equally high green algal blooms. It is possible that the sandy beach in Paje was not stable enough to be colonised.

Table 6: Seasonal variation in the number of macroalgae species identified from different study sites. The letters C, P and R represents Chlorophyta, Phaeophyta and Rhodophyta species.

Study sites	Dry cold season (June-September) Mean for 97 & 98			Short rainy season (October-November) Mean for 97 & 98			Dry hot season (December-February) 97-98			Long rainy season (March-May) 98		
	C	P	R	C	P	R	C	P	R	C	P	R
Chwaka												
Groundwater	5	0	1	3	0	1	4	0	1	5	0	1
Normal saline	8	2	2	7	2	2	8	2	2	6	3	1
Transect	13	4	3	11	3	2	14	4	3	9	3	2
1.5-2,0 km	14	5	5	8	2	3	13	4	5	7	1	2
Reef	33	5	6	32	4	5	30	8	8	23	2	2
Paje												
Groundwater	3	0	0	3	0	0	3	0	0	3	0	0
Normal saline	4	2	2	3	2	1	3	2	2	2	1	0
Transect	4	3	2	4	2	2	4	2	2	2	2	2
1.5-2.0 km	6	3	5	5	3	5	7	5	4	4	4	4
Reef	11	5	5	8	6	4	12	6	6	8	4	5
Fumba												
suspected g.w. site	19	6	3	15	6	3	18	8	4	15	2	1
Normal saline	15	5	5	17	7	4	16	6	5	12	3	2
1.5-2.0 km	32	14	11	30	10	9	33	16	11	25	5	6
Reef	33	13	11	30	11	9	34	17	9	28	7	6

Note: Details on how the sites were located is indicated in the materials and methods. Letters (C), (P) and (R) stands for Chlorophyceae, Phaeophyceae and Rhodophyceae, respectively. g.w. = ground water.

Seagrass species composition in the different sites

A total of 11 species of seagrasses, 3 species belonging to the Hydrocharitaceae and 8 Potamogentonaceae, have been identified in all the sites (see Appendix 2.2). All the seagrass species are represented at Chwaka where the species composition and biomass were highest in the transect located around the bore hole followed by the transect located 1.5-2 km away. The reefs had the least number of species. Species of the genera *Halophila* and *Thalassia* (Hydrocharitaceae), believed to have re-colonised the sea originating from the freshwater sources, were the ones which were found to flourish around the freshwater bore-hole. Species of the genus *Enhalus* was not found around the bore hole, possibly because it requires silt mud. However, *Enhalus acoroides* and *Thalassodendron ciliatum* were among the species which grow most abundantly in areas receiving GW.

Table 7: Seagrasses (family Hydrocharitaceae (H) and Potamogentonaceae (P)) identified from different study sites. In A are shown seagrasses that are located in the vicinity of the bore hole i.e. freshwater and normal saline sites and permanent transects used to monitor the influence of groundwater to macrophyte biomass and species composition. A total of 11, 5, and 6 species have been identified in Chwaka, Paje and Fumba, respectively during the whole duration of the study. Sampling was done as detailed in the materials and methods.

A

Study sites	**Seagrasses**		**Comment**
	H	P	
Chwaka	3	8	Largest percentage cover made up of *Thalassia hemprichii. Thalassodendron ciliatum* with longer inter-nodes than in Paje
Groundwater	2	2	Largest biomass made up of Thalassia hemprichii
Normal saline	2	2	
Transect	3	8	
Paje	2	3	Largest percentage cover made up of *Thalassia hemprichii.*
Groundwater	1	1	Less than 5% cover of *Halodule* and *Halophila*
Normal saline	2	2	
Transect	2	3	
Fumba	2	2	Largest percentage cover and biomass made up of *Thalassia hemprichii*
Transect (suspected g.w.)	2	2	
Transect (normal saline)	2	2	

The status of major macrophytes found in the groundwater sites

The status of the major macrophytes found in the transect located around the ground-water bore-hole in Paje and Chwaka during the el-nino rains is shown in Figures 4 A-H. The productivity of seagrass meadows in areas receiving groundwater flow shows the presence of high densities of macrophytes, notably of the genera *Gracilaria* (Rhodophyta), *Enteromorpha* and *Ulva* (Chlorophyta).

The highest macrophyte biomass were recorded during April-May 1998, during heavy rains (see Appendix 1). Sites around groundwater bore hole had low biomass of seagrasses and highest biomass of green algal blooms. Nevertheless, for Chwaka a few metres away from the bore hole, the seagrass densities were found to also increase. As earlier noted, the seagrass species diversity and biomass were highest in Chwaka, where the groundwater flow is possibly the highest. The groundwater in Chwaka is possibly originating from Cheju where fertilisers are applied in the rice farms. Nutrient data shown above indicate that Chwaka receives groundwater with low nutrients. Among other reasons it has been urged that low nutrients could be a result of mangrove filtering effect. It is likely that seagrasses and macroalgae benefit indirectly from additional nutrients possibly from decomposing N-rich mangrove leaf litter. Sediment pore water in Chwaka has been shown have 100-200 μg.atN.l^{-1}. (Mohammed, 1998)

The macroalgal densities around groundwater bore-hole were highest in Chwaka. In Paje the productivity of *Eucheuma* farms located in an area where the groundwater is swept to is low (see section on photosynthetic responses for further discussion).

Table 7: Seagrasses (family Hydrocharitaceae (H) and Potamogentonaceae (P)) identified from different study sites. In B are shown seagrasses identified from reefs and 1.5-2 km away from freshwater sites. A total of 11, 5, and 6 species have been identified in Chwaka, Paje and Fumba, respectively during the whole duration of the study. Sampling was done as detailed in the materials and methods.

B

Study sites	**Seagrasses**		**Comment**
	H	P	
Chwaka	3	8	Largest percentage cover made up of *Thalassia hemprichii. Thalassodendron ciliatum* with longer inter-nodes than in Paje
1.5-2.0 km from GW.	3	8	Largest percentage cover made up of *Enhalus ecoroides, Thalassodendron ciliatum* and *Thalassia hemprichii*
Reef	2	4	Largest percentage cover made up of *Thalassia hemprichii*
Paje	2	3	Largest percentage cover made up of *Thalassia hemprichii.*
1.5-2.0 km from GW.	2	2	Largest percentage cover and biomass made up of *Thalassia hemprichii.* Thalassodendron ciliatum grows fairly well as you approach the reef.
Reef	2	3	Largest percentage cover and biomass made up of *Thalassia hemprichii*
Fumba	3	3	Largest percentage cover and biomass made up of *Thalassia hemprichii*
1.5-2.0 km from 'GW'.	2	2	Largest percentage cover and biomass made up of *Thalassia hemprichii*
Reef	2	2	Largest percentage cover and biomass made up of *Thalassia hemprichii*

Note: Zanzibar has a total number of 11 seagrass species belonging to two families. Family Hydrocharitaceae *Halophila* (1 species) *Thalassia* (1 species) *Enhalus* (1 species); Family Potamogentonaceae- *Zostera* (1 species) *Haludule* (2 species) *Cymodocea* (3 species) *Syringodium* (1 species) *Thalassodendron* (1 species).

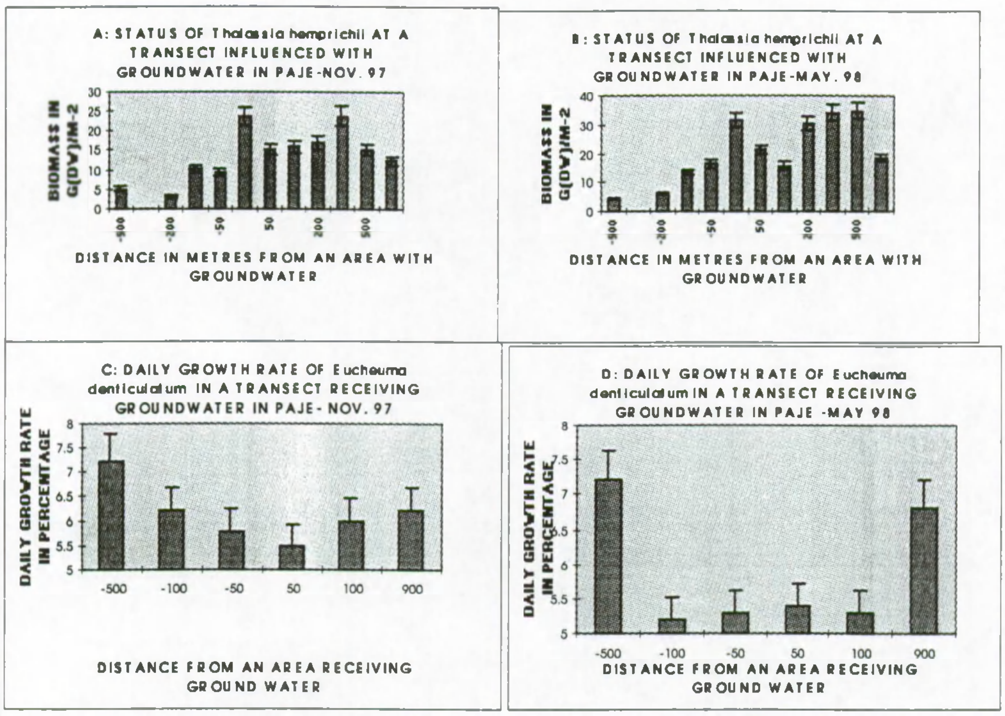


Figure 4 A-D: The status of the macrophytes at a transect influenced by groundwater in Paje. As the area besides the groundwater nearly bare (no seagrass meadow) the transect was located at the middle of the channel (250 m away). A distance marked 0 m is opposite the first major fresh water bore-hole. In the figures negative and positive distance indicates distance to and from the first groundwater bore-hole, respectively. Growth rate of E. denticulatum *was determined from farms located at respective points as shown in the Materials and Methods. Macrophytes around the bore hole include* Ulva rigida, U. reticulata, U. fasciata *and* Chaetomorpha indica *and* U. rigida *and* U. reticulata *were the dominating species.*

Photosynthetic responses of selected macrophytes found to flourish or languish in groundwater sites

Low pH and salinity were among factors associated with the groundwater. It was of interest to assess their influence on the productivity of some algae and seagrasses found to flourish or languish in sites with groundwater as shown in section on seagrass species composition. Dose response curves such as those shown in Figure 5 were used to estimate the macrophyte responses summarised in Tables 8 A-C.

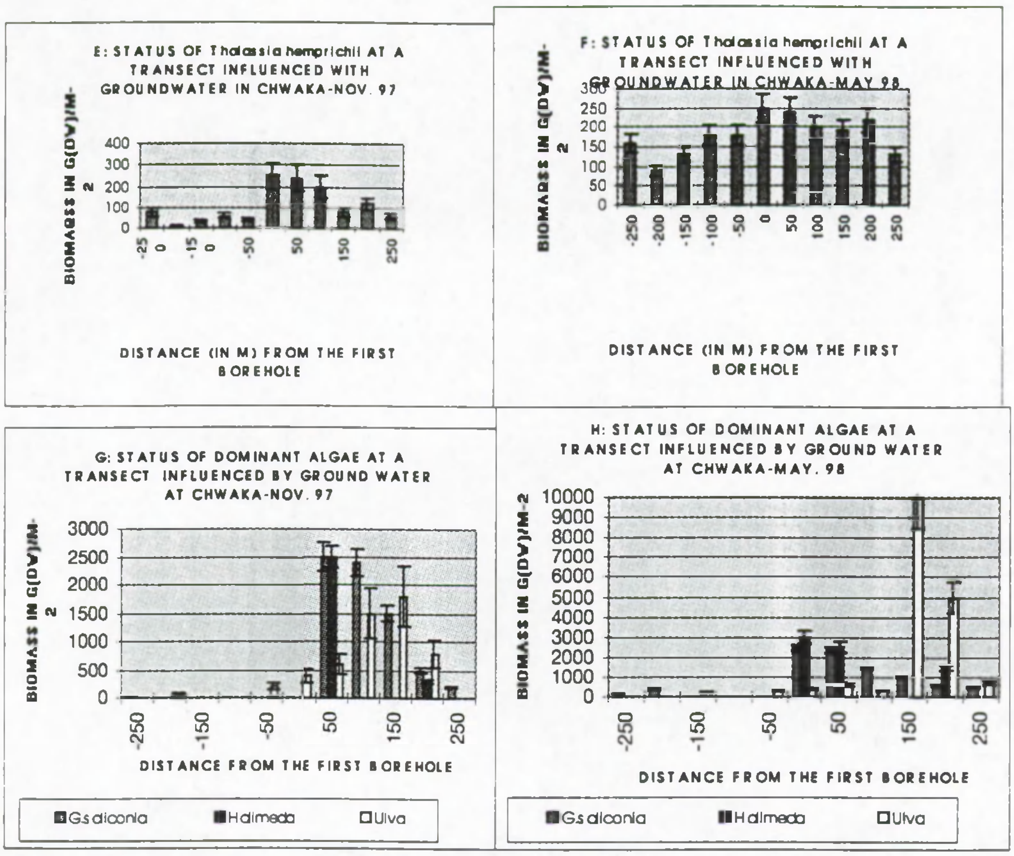
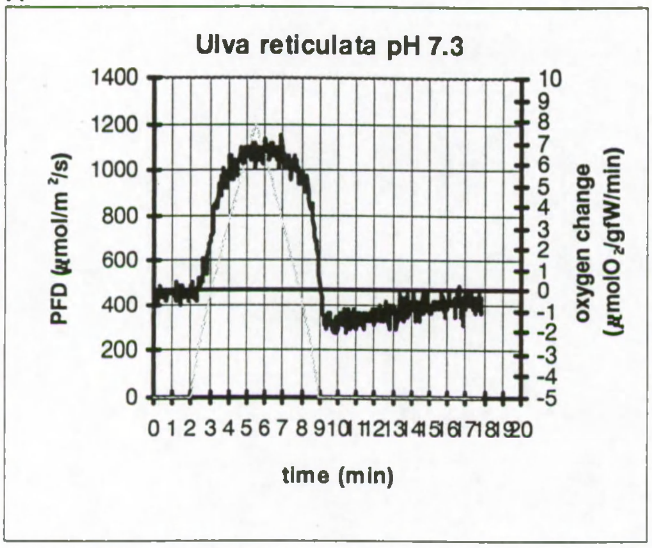


Figure 4 E-H: Macrophyte status at a transect located by the groundwater bore-hole in Chwaka. Positive and negative numbers at the x-axis refers to the distance to (from up stream) and from the first groundwater bore-hole, respectively. Ground water streams into the transect at 0 m and 120 m. In November, around the bore holes 1 and 2 the Ulva species measured around 2.0 and 1.5 kg(DW) m^{-2}, respectively. In May they were 2.0 and 3.1 kg(DW) m^{-2}, respectively. The highest biomass shown in May (around 10 kg(DW) m^{-2}) includes macrophytes deposited by the channel. G. saliconia = Gracilaria salicornia; *HALIMEDA=* Halimeda tuna, H. macroloba, *and* H. opuntia.

Low pH (pH 7.3) appeared to favour the photosynthetic rates of all the macrophytes tested, as compared to the normal pH of the sea water (pH 8.2) or higher pH (pH 8.5). The reason behind that has not been established. However the tendency of pH 7.3 favouring the availability of higher concentrations of carbon dioxide (CO_2) (Skirrow, 1975) in the seawater could be one of the reasons. The CO_2 may be more favoured by macrophytes than HCO_3^- because it freely diffuses through membranes and is the inorganic carbon species used by the carbon-fixing enzyme, ribulose-1,5-bisphosphate carboxylase/ oxygenase (RUBISCO). Utilisation of HCO_3^- may be facilitated by energy requiring direct uptake and/or carbonic anhydrase (CA)-mediated dehydration to CO_2 (Lucas, 1983). Low CO_2 around the RUBISCO site may favour the photorespiration (Warburg effect), the Mehler reaction (Mehler, 1951), photoinhibition and photodamage (Krause & Weis, 1991, Kyle, 1987). The efficiency of light utilisation in plants varies between 2-36% (Asada, 1992) depending upon the availability of factors favouring the good productivity of plants.

A



B

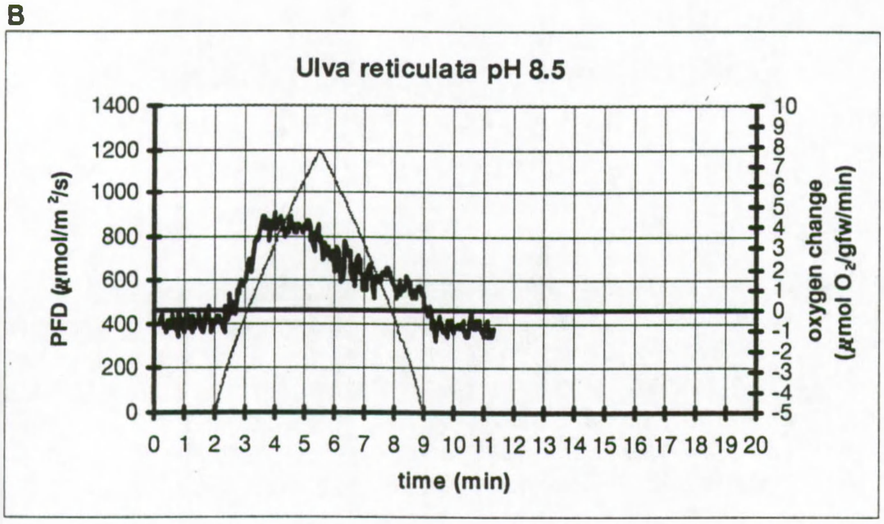


Figure 5 Typical dose response curves for photosynthetic oxygen evolution in macrophytes. Shown here are the effect of pH on the light saturation values and photosynthetic rate at light saturation value.

Table 8 A-D: Photosynthetic responses of the macroalgae Gracilaria salicornia, Eucheuma denticulatum and Ulva reticulata and seagrass Thalassia hemprichii and Thalassodendron ciliatum subjected to different pH or salinity. Respiration was measured before photosynthesis commenced. Shown in the tables A-C are means and standard deviations (SD) for three experiments. N.Photo. = net photosynthetic rate. Shown in Table D are the estimated quantum efficiencies at different pH and salinities.

A: Photosynthetic light saturation values in μmole photon m^{-2} s^{-1}. Mean and SD shown in the nearest whole number. Respir. =only respiratory activity was measured.

Macrophyte	pH 7.3	pH 8.2	pH 8.5	10‰	20‰	35‰
Ulva reticulata	1053±94	803±65	851±103	402±61	702±52	903±47
Gracilaria salicornia	851±46	450±20	403±31	253±22	352±21	500±61
Eucheuma denticulatum	1020±151	610±50	450±45	Respirat.	350±15	501±31
Thalasia hemprichii	2020±205	2040±250	2050±105	820±44	2201±102	2100±352
Thallassodendron ciliatum	1502±100	1450±131	1450±115	605±80	1501±70	1600±121

B: Maximal photosynthetic oxygen evolution (in μmole O_2 mg(fresh weight)$^{-1}$ min^{-1}) at saturating light intensity. Mean and SD shown in the nearest one decimal point.

Macrophyte	pH 7.3 N.Phot.	pH 7.3 Respir.	PH 8.2 N.Phot.	pH 8.2 Respir.	pH 8.5 N.Phot.	pH 8.5 Respir.
Ulva reticulata	6.5±1.0	0.3±0.1	5.2±0.8	1.0±0.2	4.5±0.3	1.0±0.1
Gracilaria salicornia	3.5±0.4	3.0±1.0	2.4±0.3	4.1±1.1	2.0±0.5	3.2±0.1
Eucheuma denticulatum	4.0±0.5	2.0±0.7	3.0±0.1	4.0±0.5	2.5±0.1	4.7±0.3
Thalasia hemprichii	5.5±1.0	0.2±0.1	5.0±0.2	0.3±0.1	5.0±1.0	2.0±0.4
Thallassodendron ciliatum	5.0±0.8	0.3±0.2	5.0±0.3	0.3±0.1	6.0±1.1	1.3±0.2

Low salinity, particularly 10‰, appeared to be detrimental to all the Rhodophytes used (see Table 8 A-D). *Ulva reticulata, Thalasia hemprichii* and *Thallassodendron ciliatum* maintained relatively high photosynthetic rates and quantum efficiencies. Other species sharing this tendency include *Ulva fasciata, U. rigida* and *U. pertusa*. These results explains the reason why the *Ulva* spp and *Thalassia* sp. flourishes around the freshwater bore-holes. *Thalassodendron ciliatum* was equally resistant to low salinity. It is likely that grown up plants of this species of seagrass are excluded in areas around the freshwater bore hole for reasons other than salinity fluctuations. Most seagrass species seem to tolerate salinities of approximately 15-55 ‰ depending upon other variables such as temperature. Salinity fluctuation could act as a selection pressure through favouring recruitment of some seagrasses.

C: Maximal photosynthetic oxygen evolution (in μmole O_2 mg(fresh weight)$^{-1}$ min^{-1}) at saturating light intensity. Mean and SD shown in the nearest one decimal point.

Macrophyte	10‰ N.Phot.	10‰ Respir.	20‰ N.Phot.	20‰ Respir.	35‰ N.Phot.	35‰ Respir.
Ulva reticulata	2.5±0.1	3.0±0.1	3.2±0.6	1.0±0.2	4.6±1.1	0.5±0.2
Gracilaria salicornia	0.0±0.0	18.0±6.1	2.4±0.2	10.1±1.1	2.0±0.6	3.2±0.3
Eucheuma denticulatum	0.0±0.0	0.0±0.0	0.9±0.1	14.0±2.1	2.5±0.7	4.7±0.5
Thalasia hemprichii	3.7±1.1	3.2±0.1	4.0±0.5	2.5±0.1	5.4±1.0	0.8±0.1
Thallassodendron ciliatum	3.0±0.7	2.6±0.4	3.5±0.3	3.2±0.3	5.0±0.7	1.3±0.1

D: Ratio of the variable fluorescence to maximal fluorescence (Fv/Fm) for plants subjected to different pH and salinity regimes. Mean and SD shown in the nearest two decimal points.

Macrophyte	pH 7.3	pH 8.2	pH 8.5	10‰	20‰	35‰
Ulva reticulata	0.88±0.11	0.85±0.13	0.80±0.11	0.70±0.11	0.80±0.12	0.75±0.11
Gracilaria salicornia	0.83±0.12	0.76±0.21	0.65±0.05	0.11±0.03	0.35±0.10	0.89±0.21
Eucheuma denticulatum	0.90±0.32	0.65±0.14	0.50±0.01	0.03±0.1	0.27±0.07	0.72±0.12
Thalasia hemprichii	0.90±0.21	0.80±0.23	0.81±0.11	0.43±0.03	0.76±0.01	0.85±0.03
Thallassodendron ciliatum	0.90±0.27	0.88±0.21	0.89±0.21	0.54±0.21	0.67±0.11	0.82±0.14

Benthic Microalgae

Biomass and species composition for the benthic microalgae found in the different study sites are shown in Table 9. With exception of Paje, freshwater sites in the other

Table 9: Benthic microalgal biomass and species composition in different study sites.

Site	Station	µgChl a (g dwt)$^{-1}$	Species identified
PAJE	Freshwater site	2.7	*Gyrosigma spp. Amphora spp. (dominant), Hantzschia spp. Navicula spp.* and *Nitzschia spp.*
	Normal Saline site	1.45	*Nitzschia longissima, Pleurosigma spp., Pleurosigma angulatum (dominant), Gyrosigma spp. Ampora wisei (cpmmon) Tropidoneis approximata, Microcystis spp. Hantzschia spp.* and *Diploneis spp.*
CHWAKA	Freshwater site	6.58	*Navicula spp. Hantzschia spp. Amphora spp. Biddulphia pulchella, Pleirosigma spp.* and *Oscillatoria spp. Gyrosgma spp. Diploneis spp. Coscinodscus spp., Tropidoneis spp* and, *Prorocentrum spp.*
FUMBA	Freshwater	6.78	*Hantzschia spp., Gyrosigma spp. (dominant), Oscillatoria spp., Nostoc spp., Navicula spp. Amphora spp.*, and *Pleurosigma spp.*,
	Normal saline site	3.97	*Amphora spp., Hantzschia spp.* and *Protoperidinium spp.*
MAKOBA	Freshwater site	2.67	*Hantzschia spp. Amphora spp. Asterolampra spp., Diploneis spp, Nitzschia longissima* and *Navicula spp.*
	Normal saline site	2.75	*Pleurosgma spp., Tropidoneis spp.* and *Diploneis spp.*

locations (Chwaka, Fumba and Makoba) had higher benthic microalgal species representation than normal saline sites. The biomass were also highest in freshwater sites. It is difficult to conclude anything here as the sampling was done only once after the Yerseke Workshop.

Macrobenthos

The major groups of organisms featuring in the macrobenthos collection included polychaetes, gastropods, bivalves, crustacean crabs, shrimps and nematodes with varying densities. The substrate constitution broadly included fine mud, sand, shells, calcareous algal remnants and organic debris. The data on the macrobenthos in various sites is presented in the Appendix 3.1. Polychaetes were the most prevalent and were taken as the most reliably representative macrobenthon. A comparison of polychaete densities from Paje, Chwaka and Fumba for February, May and November 1998 seems to indicate no significant differences between sea water and fresh water influenced areas (Table 10). However, when the freshwater influenced sites are compared, Chwaka appears to have the lowest density. As shown in section on water quality of this report, the underground water reaching this area drains from an intensively rice cultivated area which contains agrochemicals which could be responsible for the low polychaete numbers at the outpour points. The highest amount of DDT, possibly originating from house fumigation, has been measured in Chwaka freshwater site (Table 4.1.1).

Table 10: A comparison of the macrobenthos densities (Number/m^2) of the major group, ie Polychaete, from Paje, Chwaka and Fumba for February, May and November 1998. GW and NS represents freshwater and normal saline sites.

Season	Paje GW	Paje NS	Chwaka GW	Chwaka NS	Fumba GW	Fumba NS	Makoba GW	Makoba NS
Dry (February 1998)	175 350 250 0 0	875 275 475 300 450	175 25 50 0 0	100 600 50 350 0	100 225 225 0 0	25 150 25 0 0		
Mean	**155±69.1**	475±108	49±32.6	**220±112**	183±50.4	39±28.1		
Long rains (May 1998)	100 0 0 0	275 100 325 0	50 0 0 0	700 1150 575 550	100 25 75 0	150 325 325 275		
Mean	**25±25**	175±75.7	12.5±12.5	474±139	**50±22.8**	**269±41.3**		
Short rains (November, 1998)	25 575 200	200 0 0	25 0 0	25 50 0	125 25 0	175 25 100	25 175 0	0 0 0
Mean	266.7±162	66.7±66.7	8.33±8.33	25±14.4	50±38.2	100±43.3	**66.6±54.6**	**0**

In the May 1998 sampling period, the freshwater outpour points appear to have the lowest polychaete densities,when compared to normal saline sites. Again when the fresh water outpour points are compared, the Chwaka site apears to have the lowest density. this could indicate the presence of a supresive agent as suggested in the February samples.

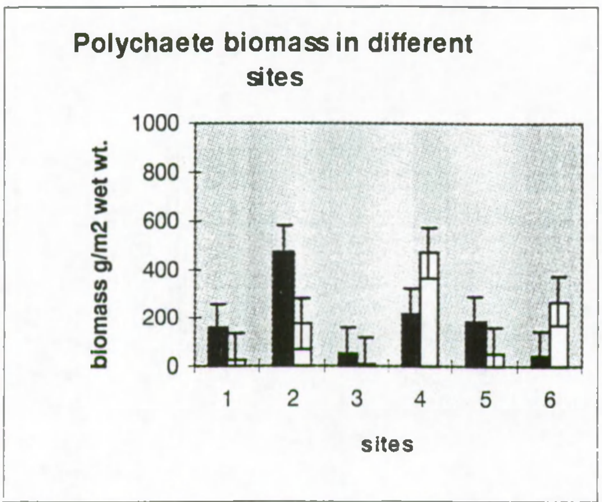


Figure 6: Polychaeta biomass in the different sites for the representative dry season (February 1998, filled bars) and wet season (May 1998, open bars). Numbers on the x-axis refers to the different sites as follows: 1= Paje freshwater site (Pf), 2= Paje normal saline site (Ps), 3=Chwaka freshwater site (Cf), 4= Chwaka normal saline site (Cs), 5= Fumba freshwater site (Ff), 6= Fumba normal saline site (Fs), 7= Makoba freshwater site (Mf), and 8=Makoba normal saline site (Ms).

Both Makoba sites had zero values with respect to polychaetes.

The Makoba site was consistently having extremely low numbers or no macrobenthos at all. This site is basically a large freshwater outpour area and drains from sugarcane plantation and rice farms. The hypersalinity of the interstitial waters in this area (60-100‰) may also contribute to the low macrobenthos numbers. It is possible that the polychaete and generally macrobenthic infaunal communities here are stressed.

A comparison of the biomasses for the polychaete populations at the salt water and the freshwater sites at Paje,Chwaka and Fumba is given in Figure 6. In all the sites, Paje, Chwaka and Fumba had significant differences in their biomasses between freshwater/ groundwater and normal saline areas. As in the case of density, biomasses are consistently lower at the freshwater sites, and of the three places, Chwaka has the lowest.

Table 11: The diversity of the groups in each of the study sites for February, May and November, 1998. FW and NS represents freshwater and normal saline sites. Macrobenthos features in different sites include polychaetes, gastropods, bivalves, crustacean crabs, shrimps and nematodes.

	Paje FW	Paje NS	Chwaka FW	Chwaka NS	Fumba FW	Fumba NS	Makoba FW	Makoba NS
June 97	4	-	3	5	5	2	2	0
Feb. 98	3	2	3	5	2	3	1	0
May. 98	3	2	2	4	4	3	0	3
Nov. 98	1	1	2	1	1	2	1	0
Mean	2.5	1.25	2.5	3.75	3	2.5	1	0.75

Note: - = no samples taken

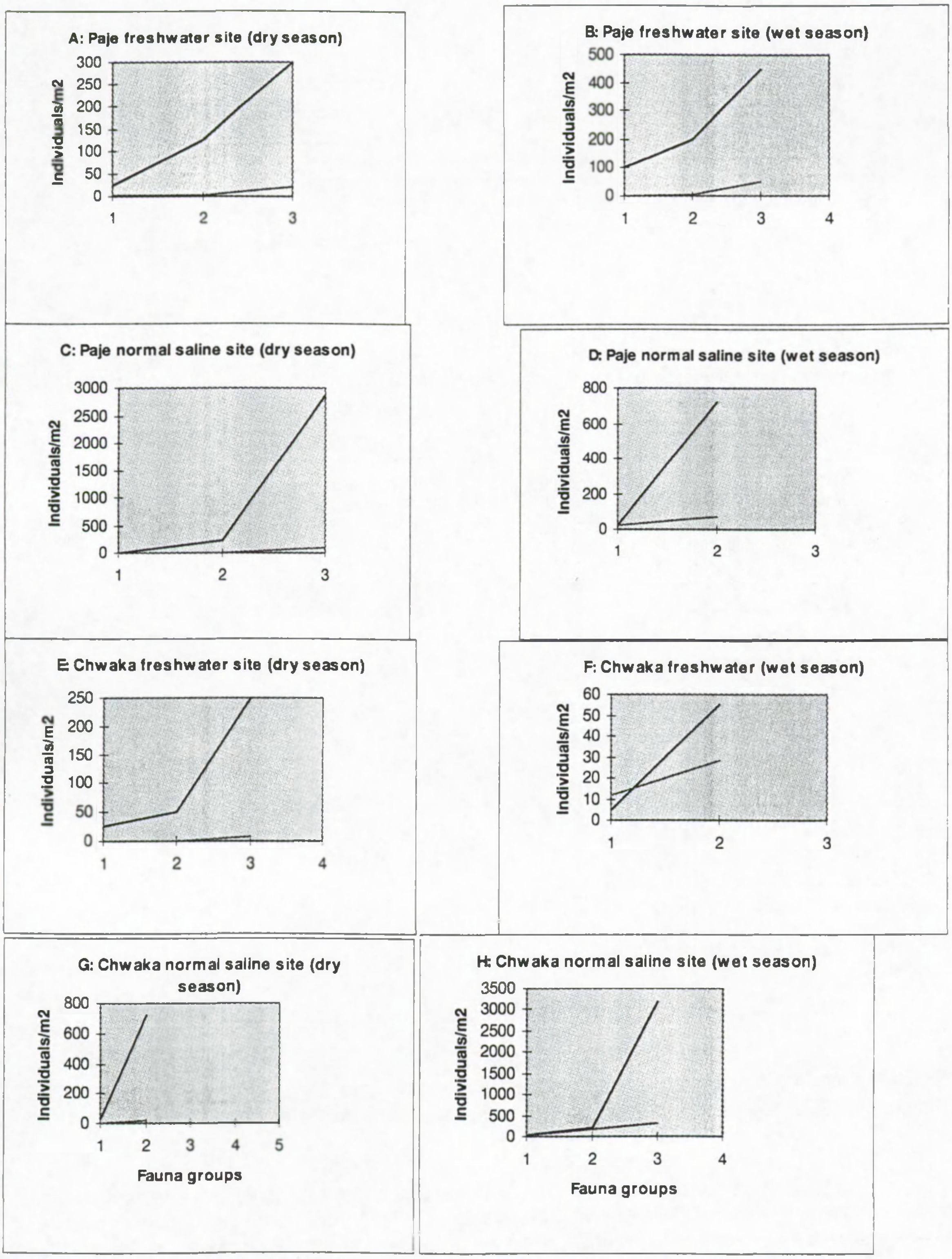


Figure 7. For legend see next page.

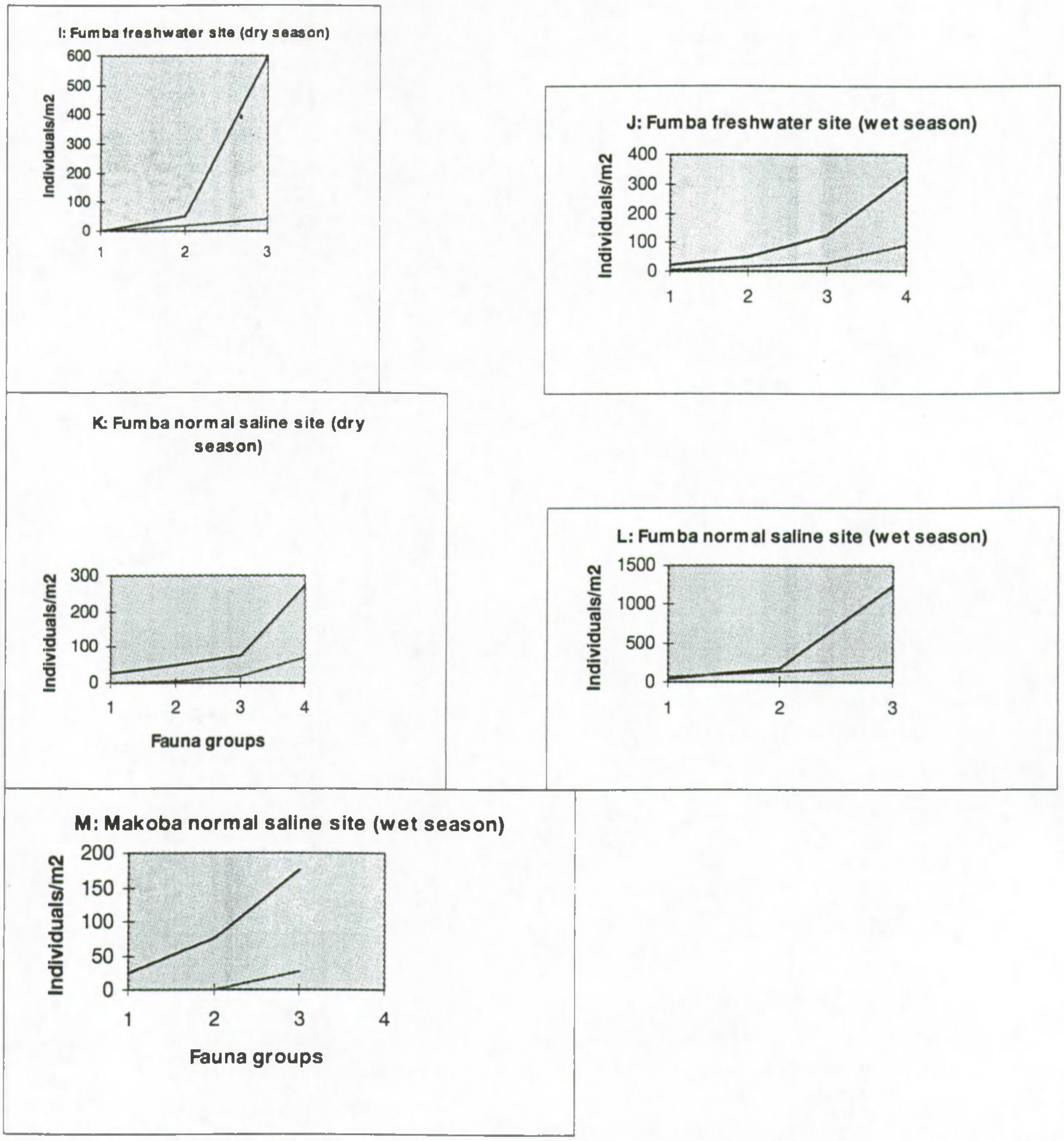


Figure 7 A-M: The abundance biomass curves (Boesch & Rosenberg, 1981 in Barrett et al., 1990; Wolf et al., 1982) for the various sites using the major infauna element in each case.

The mean fauna diversity is lowest in the Makoba sites, highest in Chwaka normal saline and Fumba freshwater sites. Otherwise there was no obvious trend. Whereas the low diversity at Makoba sites may be associated with suppressive agrochemicals the Paje normal saline area with little agrochemicals residues, could be affected by a different factor.

The abundance biomass curves for the major fauna found in the different sites are shown in Figure 7. During the dry season, there was no indication of stress for macrobenthos in Paje groundwater (Figure 7A), Chwaka groundwater (Figure 7E) and normal saline (Figure 7G) and Fumba normal saline sites (Figure 7K). The ABC curve in Chwaka groundwater site during this season diverge in a way typical for the unstressed populations (Jose Paula, verbal communication). The crossing ABC curves in Paje normal saline site (Figure 7C) indicates stress. Another site showing the same feature in the dry season is Fumba freshwater site (Figure 7I). It is possible that lindane that has been detected in Fumba freshwater site (see section on water quality) was stressful to macrobenthos.

During the wet season, stress has been indicated in both Chwaka sites (Figure 7F and H) and Fumba normal saline site (Figure 7L). In the other sites, macrobenthos stress could not be detected.

From these results there is evidence of stress during the rainy season in both of the Chwaka sites. However there is no direct evidence of stress difference between freshwater and normal saline sites in the other locations during both wet and dry season. The case of Chwaka may be associated with the influx of freshwater, agricultural and household waste and sediments from land.

Meiobenthos

Ten major taxa were found during the surveys. These included Harpacticoida, Polychaeta, Turbellaria, Nematoda, Oligochaeta, Ostracoda, Amphipoda, Isopoda, Gnathostomulida, Insecta, and Zoea. Only four of the major taxa were found to be relatively abundant (Nematoda, Harpacticoida, Polychaeta and Turbellaria) while the rest were rare, encountered only very occasionally. These were grouped together as 'others'. Nematoda was the most abundant taxa constituting on average between 33 % and 95 % of the total count in the different locations. This was followed by Harpacticoida, constituting between 1 % and 57 % of the total counts. Table 12 illustrates the results of meiobenthos counts

Table 12 Variation in total number of meiobenthos between sites with normal saline conditions and sites receiving freshwater input in Zanzibar during surveys: June and September 1997 and in May, 1998.

	FUMBA		PAJE		CHWAKA		MAKOBA	
	NS	FW	NS	FW	NS	FW	NS	FW
JUNE	2846	2461	2120	2100	2050	784	791	787
SEPTE MBER	1053	1717	906	110	1896	457	138	88
MAY	891	818	1216	623	1509	864		

between normal saline sites and sites receiving freshwater. Meiobenthos counts between sites in Fumba do not seem to show clear difference on all sampling dates. In both Paje and Chwaka there was a clear difference in total counts between sites during September, 97 and May 98. In the June 97 survey the difference in counts was not great between sites in all locations and total counts were highest of all sites and locations. Makoba does not appear clear regarding the difference in meiobenthos counts during the surveys.

Results of ANOVA and multi-dimensional scaling ordination MDS

The results of analysis of variance (ANOVA) are shown in Table 13. These results, generally indicate lack of a clear pattern. The results of multi-dimensional scaling ordination

(MDS) are shown in Appendix 1 in Part II by the Stockholm University. The lack of a clear pattern between sites is again illustrated in the MDS maps. In many cases the fresh water sites do not clearly separate from the normal saline sites except for the case of Paje.

Overall, the data generally lack any clear pattern. This is considered to be due to the nature of the freshwater input in the intertidal areas where the tidal waters dilute the impact of the incoming groundwater for a large part of the day. In the case of Chwaka bay mangrove forest sites, the groundwater had a mean salinity of 32‰ so salinity was not significantly different to tidal waters.

Table 13: Summary of the 2-way ANOVA for the four major taxa from three sampling occasions LxS-interaction between Location and Site, ns-not significant,=P<0.05, ***=P<0.001.*

	June 1997			**September 1997**			**May 1997**		
	Locatn	**Sites**	**LxS**	**Locatn**	**Sites**	**LxS**	**Locatn**	**Sites**	**LxS**
Nematoda	ns	ns	ns	***	***	***	ns	ns	ns
Harpaticoida	**	ns	ns	ns	ns	ns	ns	ns	ns
Turbellaria	**	ns	*	ns	ns	ns	*	ns	**
Polychaeta	*	ns	ns	***	**	ns	ns	**	***
Others	*	ns	ns	ns	ns	ns	*	ns	ns

June 1997: The results of ANOVA for June 1997 indicate that Nematoda shows no significant difference neither among locations nor between sites (i.e. Fresh water and normal saline sites). Harpacticoida, Turbellaria and Polychaeta show results similar to that of Nematoda, but in addition Turbellaria show interaction effect between factors (locations, sites and interaction). The remaining taxa (others) show significant difference only for locations. MDS reflects this i.e. no obvious pattern for the F (fresh water) and N (normal salinity).

September 1997: Nematoda shows significant difference both among locations, sites (i.e. fresh water and normal salinity sites) and interactions. Hence it is difficult to attribute the difference to a single factor. For Polychaeta there seems to be clear difference both among locations and sites. No interaction is exhibited. The remaining taxa show no significant difference. The MDS shows also that some of the freshwater sites are clearly separated from the rest.

May 1998: Both Nematoda and Harpacticoida do not show significant difference between locations, sites and interaction. Turbellarians show significant difference among locations and also interactions but not between sites. Polychaeta show significant difference between sites and interactions. The remaining taxa show significant difference only for locations. The MDS show practically no pattern.

The above results indicate that only certain meiobenthos taxa show significant difference among locations, sites and interactions. In some cases certain taxa show significant difference both among locations and sites plus interaction. Such results make the interpretation very difficult because it is not easy to attribute the difference to a single factor. This is indicated by lack of obvious pattern even in the MDS. Experience has shown that meiobenthos distribution is influenced by a suite of factors. Freshwater carries with it nutrients and other dissolved substances that may promote algae and diatoms growth. Certain groups of meiobenthos feed on algae, diatoms and detritus. Therefore increase in algae and diatoms can mean increase in quantities of food available for some meiobenthos. However salinity has been found to alter meiobenthos populations in some areas. In the tropics the distribution of meiobenthos has been observed to vary along salinity gradients. There appear to be a distinct relationship between salinity and meiobenthos assemblages. The relationship is reflected in species composition, abundance and species diversity. Lower salinity regions have preponderance of freshwater forms (Warwick, 1971; Brenning, 1973). Associated with a switch in fauna, there is usually a decrease in the number of species as brackish water is approached. There is also a general decrease in number of animals per unit area as lower salinity is experienced (Van Damme et al., 1980; Alongi, 1987). Dilution of sea water resulting from an addition of freshwater to the sea as for instance, during heavy down pour has often lead to mass mortality of certain groups of meiobenthos leading to low population densities (Gnapati & Rao, 1962; Harkantra & Parulekar, 1985). The lack of obvious pattern as indicated by the MDS in the current study can only be explained by the relatively high salinity at fresh water sites i.e. the fresh water sites were for the most cases not so fresh (<0.1 ‰). Also animals remained exposed to low salinity for only brief periods. For most of the time, seawater influence is so great that freshwater effect is masked. However the effects of sea water enrichment with nutrients has not been investigated to any extent. Other factors, e.g. temperature (Gnapati & Rao, 1962; Broom, 1982; Alongi, 1988), sediment mean grain size (Aller & Aller, 1986; Giere et al., 1988; Alongi & Christoffersen, 1992), level of shore and biotic factors (see Fleeger & Decho, 1987) are known to influence meiobenthos populations. Assuming that all these other factors remained constant in all the sites, the only other factor responsible for the changes in some meiobenthos populations was probably the quality and quantity of freshwater added to the near-shore regions. The differences in meiobenthos densities between locations would be governed by a suite of factors as well as the time of sampling. The differences in densities of certain meiobenthos taxa between sites is however likely to be governed by quantities of freshwater added to the sea. Changes in the number of Nematoda and Harpaticoida observed between the dry season (June and September) and wet season (May) is probably a reflection of seasonality.

Seasonal Variation in the Chemical and Biological Parameters

The seasonal variation in the chemical and biological parameters investigated in Paje, Chwaka, Fumba and Makoba is summarised in Table 14.

Table 14 Seasonal Variation in the Chemical and Biological Parameters at different sampling sites.

SEASON	NUTRIENT & PESTICIDES	MACROPHYTES SPECIES & BIOMASS	MEIOBENTHOS TOTAL NUMBERS	MACROBENTHOS TOTAL NUMBERS
JUNE-SEPT., 1997 (DRY)	Mf and Ms-highest O_2 levels in the season; Ff-Highest nutrients conc. in the season. Generally low nutrient conc. in the season.	Pf & Cf species number (3-5); Ff had 22. Ps, Cs and Fs had 7, 11, & 25 respectively. GROUND-WATER sites dominated by green algae. Cf had the highest biomass of *Ulva* spp.	Ff-highest; Pf-high, Cf & Mf-low, Fs-highest, Ps & Cs high, Ms-lowest.	Pf & Ff -high, Cf-low, Mf-lowest; Cs-high, Fs-low.
OCT.-NOV. 1997 (SHORT RAINS)	Oxygen conc. are normal in all stations. Ms had the highest NH_4^+ in the year. Mf & Ff also high NH_4^+. PO_4^{2-} levels were normal.	Pf & Cf species number (3-4) Ff had 20). Ps, Cs and Fs had 5, 10, & 25. Cf & Pf had 1.5-2.0 & 0.1 kg(DW) m^{-2} of *Ulva* spp. respectively.	Ff-highest, Cf-high, Pf-low; Mf-lowest, Cs-highest, Fs-high, Ps-low, Ms-lowest.	No sampling done
DEC.-FEBR. 1997/98 (DRY SEASON)	Ff & Fs had high NH_4^+. O_2 was high in Ff and Fs. Lowest pH in Cf for the year.	Pf & Cf species number (3-5); Ff had 30. Ps, Cs and Fs had 7, 12, & 27 respectively. Ground-water sites dominated by green algae.	No sampling done	Pf-high, Ff-high, Cf-lowest, Nf-low; Ps-highest, Cs-high, Fs-low, Ms-lowest.
MARCH-MAY, 1998 (LONG RAINS)	Mf and Ms high PO_4^{2-} levels. Cf&Cs had low nutrients. Pf,Pw,Ps,Ff,Fs had low pH.	Pf & Cf species number (3-6); Ff had 18. Ps, Cs and Fs had 3, 10, & 17 respectively. Cf & Pf had 2.0-3.1 & 0.6 kg(DW) m^{-2} of Ulva spp. respectively..	Cf-highest numbers, Ff-high numbers, Pf-lowest numbers; Cs-highest numbers, Ps-high numbers, Fs-lowest numbers	Pf-highest, Ff & Cf-low, Mf-lowest; Cs-highest, Ps & Fs-high, Ms-lowest.
JUNE-SEPT. 1998 (DRY)	No sampling done.	Pf & Cf species number (3-6); Ff had 24. Ps, Cs and Fs had 8, 12, & 23 respectively. Ground-water sites dominated by green algae. Ff had the highest biomass of *Ulva* spp.	No sampling	No sampling done
OCT. - NOV. 98 (SHORT RAINS)	No sampling done.	Pf & Cf species number (3-4) Ff had 23). Ps, Cs and Fs had 7, 12, & 31. Cf & Pf had 0.7-1.2 & <0.1 kg(DW) m^{-2} of *Ulva* spp. respectively.	No sampling	Pf-highest, Ff-high, Cf-low, Mf-lowest; Fs-highest, Ps-high, Cs-low, Ms-lowest.

NB: The capital letters P, F, M and C stands for Paje, Fumba, Makoba and Chwaka, respectively; The small letters (f) and (s) stands for freshwater and normal saline sites.

Conclusions

This study has established that groundwater outflowing into the Zanzibar near-shore ecosystems from agricultural areas and townships is a potential vector of nutrient and pesticide pollution on such ecosystems and that species diversity for macrophytes, benthic microphytes, macrobenthos and meiobenthos are threatened by such groundwater. Monitoring programs should be established to ensure that the anthropogenic substances carried by the groundwater does not exceed acceptable levels.

Pesticides (lindane and DDT) and ammonia originating from agricultural areas and/or sewage systems were among the anthropogenic substances detected in the groundwater. Whereas lindane has been detected in all the study sites receiving groundwater, DDT has been associated with townships. It is likely that the lower levels detected in normal saline sites were as a result of dilution of the groundwater.

The groundwater has been established as a vector of land based ammonia to near-shore ecosystems in Paje, Fumba and Makoba. Chwaka which receives groundwater from Cheju rice farms, where the highest ground-water ammonia (108 μg. at.N.l^{-1}), has been measured, had very low ammonia ($\leq$3.31 μg. at.N.l^{-1}). Further research is required to establish if mangrove stands here act as a filter system for land based nutrients.

Variations in macrobenthos densities appear to be influenced by fresh water input as lower densities were observed in the majority of freshwater sites. Application of the abundance biomass curves (ABC) indicate stress in Chwaka at both fresh water and salt water sites, particularly during the long rainy season. This may be attributed to low pH and agrochemicals residues (pesticides and nutrients) at the groundwater sites.

Variations in the meiobenthos numbers between sites has only been shown in Chwaka and Paje. Chwaka groundwater sites had lower numbers/ densities of Meiobenthos compared to normal saline sites. This is probably due to the influence of agrochemical residues brought by the freshwater input. The lowest numbers of meiobenthos observed in Makoba were due to a preponderance of groundwater input, low pH and high organics.

Areas receiving ground water in Paje and Chwaka had low macrophyte species representation. Among the algae, opportunistic species of the family Ulvaceae, e.g. *Ulva reticulata, U. rigida, Enteromorpha spp.,* and *Chaetomorpha spp.* were dominant on such areas. Among the sites investigated, the mentioned green algae were most dominant in Chwaka site receiving groundwater. Also dominant in Chwaka was the Rhodophyte, *Gracilaria salicornia.* Among the seagrasses, *Thalassia hemprichii* was the dominant seagrass on sites receiving freshwater. However, *Thallasodendron ciliatum* patches grew more luxuriously in freshwater points.

Whereas low pH measured in freshwater sites favours the photosynthetic rate of all the macrophyte tested, low salinity has been shown to limit the growth of *E. denticulatum* and *G. salicornia* in sites receiving groundwater. The seagrasses *Thalassodendron ciliatum* and *Thalassia hemprichii* were as tolerant to low salinity as the green algae *Ulva reticulata, U. fasciata, U. rigida* and *U. pertusa.*

A total of 86 macroalgae species have been identified in all the study sites. The species representation was lowest (12-45%) in sites receiving groundwater and highest at the reefs (29-74%). Species representation was highest in sites found in Fumba (74%), high in Chwaka (63%) and lowest in Paje (33%).

A total of eleven seagrass species have been identified in all the sites. Species representation and biomass were however highest around the ground-water sites and lowest by the reef. Chwaka groundwater site was represented by all the seagrass species and their biomass was highest.

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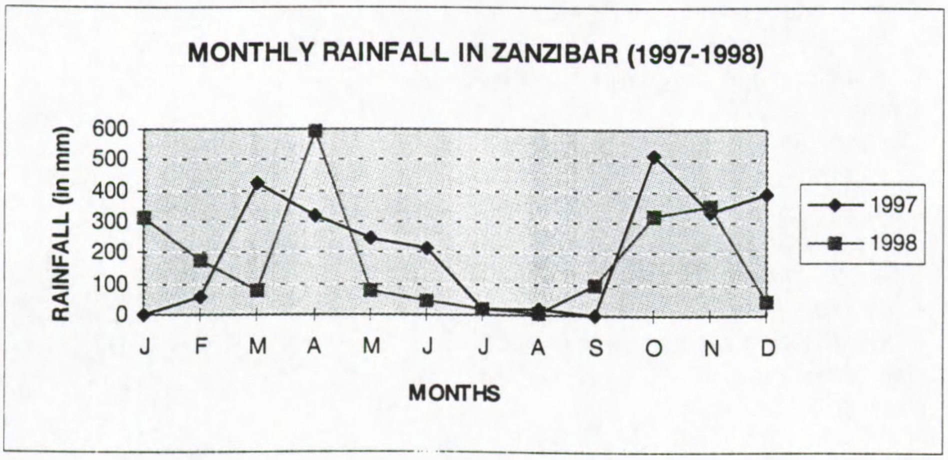
Appendix 1 Rainfall

The status of the rainfall during 1997-1998. The rainfal experienced during October 1997 - February 1998 is believed to be influenced by el-nino.



Appendix 2.1 Macroalgae

Macroalgae species identified from transects located in the visinity of the groundwater borehole in Paje, Fumba and Chwaka are indicated by (*). Species that have been identified from reefs and site located 1.5-2 km away from the transect estabilished besides the groundwater in Paje, Fumba and Chwaka are indicated by (√). A total of 86 species (Chlorophtes 50 species, 20 Phaeophytes and 16 Rhodophytes) have been recorded at Chwaka Bay (C), Paje (P) and Fumba (F).

Family	Species	C	P	F
Chlorophyceae:	*Anadyomene wrightii* Gray	√		√
	Avrainvillea erecta (Berkeley) Gepp.	√		
	Boergesenia forbesii (Harvey) Feldmann			√
	Boodlea composita (Harvey) Braud	√		√
	Bornetella oligospora Solms-Laubach			√
	Bryopsis sp.	√		√
	Bryopsis sp.	√		√
	Bryopsis sp	√		√
	Chaetomorpha crassa (Ag.) Kütz	√	√*	√*
	Cladophora sp.	√*		
	C. sibogae Reinbold			√
	C. fascicularis (mert.) Kutzing	√		√
	C. mauritiana Kutzing	√		√
	C. patentiramea (Mont.) Kutz.	√		√
	Caulerpa brachypus Harvey	√		√
	C. cupressoides (West) C. Agardh			√
	C. fastigiata Montagne			√
	C. lentillifera J. Ag.	√*		
	C. mexicana (Sonder) J. Agardh	√*		√*
	C. occidentalis (J. Ag.) Jaasund	√		√
	C. racemosa (Forsk.) J. Ag.	√*	√	√*
	C. pickeringii Harvey and Bailey	√		√
	C. serrulata (Forsk.) J.Ag. emend. Borgesen	√		√*
	C. taxifolia (Vahl) C. Agardh		√	√
	Caulerpa sp.	√*		
	Caulerpa sp.	√*		√*
	Codium sp	√*	√	√*
	C.dwarkense Borgesen	√	√	√
	Dictyosphaeria cavernosa (Forsk.) Børgesen	√*		√
	Enteromorpha ramulosa (J.E. Smith) Hooker.	√*	√*	√*
	E. clathrata (Roth) J. Ag.	√*	√	√*
	E. kylinii Bliding sensu Dawson	√		√*
	Halimeda tuna (Ellis & Sol.) Lamouroux	√*	√*	
	H. macroloba Decaisne	√*	√	√
	H. opuntia (L.) Lamouroux	√*	√	
	Halimeda renschii Hauck	√*	√	
	Microdictyon montgnei Hervey	√		√
	Neomeris sp.			√
	Rhizoclonium sp.	√		√
	Spongocladia vaucheriaeformis Areschoug			√
	Ulva reticulata Forskaal	√*	√*	√*
	Ulva pulchra Jaasund	√*		√*

Appendix 2.1: Cont.

Family	Species	C	P	F
Chlorophyceae:	*Ulva pertusa* Kjellman	√*	√*	√*
	U. rigida C. Ag.	√*	√*	√*
	U. fasciata Delile	√*		√*
	Valonia aegrophila C. Ag.	√*		√*
	V. fastigita Harvey	√*		√
	V. macrophysa Kutzing	√		√
	V. ventricosa J.Ag.	√		√
	Valoniopsis pachynema (Martens) Borgesen			√
Phaeophyceae:	*Cystoseira myrica* (Gmelin) C. Agardh	√	√	√*
	C. trinoidis (Forsk.) C. Agardh	√		√*
	Dictyota sp.	√		√*
	D. bartayresii Lamouroux sensu Vickers	√		√
	D. ciliolata Kutzing			√
	D. cervicornis Kutzing	√*	√*	√
	D. friabilis Setchell			√
	Hormophysa triquetra (L.) Kutzing	√*	√	√
	Hydroclathrus clathratus (Bory) Howe	√	√	√
	Padina gymnospora (Kütz) Vickers	√		√*
	P. boryana Thivy	√*		√*
	P. tetrastromatica Hauck		√	√
	Sargassum sp.	√*	√*	√*
	Sargassum sp.	√*	√*	√*
	S. aquifolium (Turn.) J. Agardh	√		√*
	S. binderi Sonder			√*
	Spatoglossum asperum J. Ag.	√*		
	Turbinaria conoides (J. Ag.) Kützing		√	√*
	T. crateriformis Taylor			√
	T. ornata (Turn) J. Ag.			√
Rhodophyceae:	*Amphiroa anceps* Lamouroux.			√
	A. fragilissima (L.) Lamouroux		√*	
	Acanthophora spicifera (Vahl) Bôrgesen		√	√
	Digenia simplex (Wulfen) C. Agardh			√
	Eucheuma denticulatum (Burman) Collins & Hervey	√	√	√
	E. platyclada (Schmitz)	√	√	√
	Galaxaura sp.			√
	Gracilaria corticata J. Agardh	√*	√*	√*
	G. millardeti J. Agardh			√*
	Gracilaria salicornia (J. Ag.) Dawson	√*		√*
	Hypnea musciformis (Wulfen) Lamouroux			√*
	Jania sp.	√*		√
	Kappaphycus alvarezii (Doty) Doty)	√	√	√*
	Laurencia papillosa (Forsk.) Greville	√	√	√*
	L. elata (C. Agardh) Harvey	√	√	√
	Liagora sp.	*	√	

Note: √ = present

Appendix 2.2: Seagrasses

Species of sea grasses (Division Anthophyta, Class Monocotyledonaceae, Order Helobiae) that have been identified from the study sites in Zanzibar. A total of 11 species have been recorded at Chwaka Bay (C), Paje (P) and Fumba (F).

Family	Species	C	P	F
Potamogentonaceae:				
	Cymodocea ciliata (Forsk) Ehrenb.ex Aschers.	√		
	Cymodocea serulata (R.Br.) Aschers and Magnus	√		
	Cymodocea rotundata Aschers and Schweinf	√		
	Halodule wrightii Aschers	√	√	√
	H. uninervis(Forsk) Aschers	√	√	√
	Syringodium isoetifolium Aschers) Dandy.	√		
	Zostera capensis Setchell	√		
	Thalassodendron ciliatum (Forsskål) den Hartog	√	√	√
Hydrocharitaceae:				
	Halophila ovalis(R.Br.)Hook.f.	√	√	
	Thalassia hemprichii (Ehrenb.) Aschers	√	√	√
	Enhalus acoroides (Linnaeus f.) Royle	√		√

Note: √ = present

Appendix 3.1. Macrobenthos

June 1997
Total number of different types/species of organisms in fresh water(F) influenced areas and sea water(S) influenced areas for the different sampling sites.

GROUP	PAJE	CHWAKA		MAKOBA	FUMBA	
Crabs	F 0	F 0	S 6	F 0	F I	S 0
Polychaeta	248	1	50	18	39	9
Nematode	8	0	11	05	02	02
Bivalves	13	0	01	0	05	0
Gastropod	11	0	0	0	07	0
Shrimps	0	22	0	0	0	0
Eels	0	3	1	0	0	0
TOTAL	280	26	68	23	53	11

Note: Paje freshwater site had the highest total number of organisms

June 1997
Densities of the Various organisms per metre squared in the different sampling locations.

	PAJE	CHWAKA		MAKOBA	FUMBA	
	FW	F	S	F	F	S
Crabs	0	0	37	0	6	0
Polychaeta	775	6.2	312	56	243	56
Nematode	25	0	68	15	12	12
Bivalves	40.6	0	6	0	31	0
Gastropod	34.37	0	0	0	43	0
Shrimps	0.0	137.5	0	0	0	0
Eels	0.0	0	6	0	0	0

Note: The polychaetes were quite frequent and often had higher densities in the majority of the sites except at Chwaka freshwater groundwater point.

Appendix 3.2. Subsites prevalent in fauna during February, May and November, 1998.

February, 1998:-

Site:	Habitat:	Group:	Total:	F/weight(g):	Individuals.m^{-2}:	g.m^{-2}:
Paje	fresh:	plc	7	1.026	175	25.65
		nmt	4	0.068	100	1.70
		spc	1	0.660	25	16.50
		plc	14	0.704	350	17.60
		plc	10	0.118	250	2.95
Paje	salt	plc	35	1.262	875	31.55
		plc	11	0.250	275	6.25
		plc	19	0.568	475	14.20
		plc	12	0.402	300	10.05
		plc	18	0.870	450	21.75
		nmt	9	0.082	225	2.05
		plc	11	0.332	275	8.30
Chwaka	fresh:	plc	7	0.032	175	0.80
		crb	1	0.010	25	0.25
		biv	1	0.242	25	6.05
		plc	1	0.006	25	0.15

Site:	Habitat:	Group:	Total:	F/weight(g):	Individuals.m^{-2}:	g.m^{-2}:
Chwaka	salt	plc	4	0.070	100	1.75
		biv	1	0.122	25	3.05
		plc	24	0.398	600	9.95
		nmt	1	0.050	25	1.25
		crb	3	0.128	75	3.20
		plc	2	0.028	50	0.70
		nmt	4	0.018	100	0.45
		cf	1	0.008	25	0.20
		plc	14	0.380	350	9.50
		crb	1	0.500	25	12.50
		ces	2	0.024	50	0.60
		nmt	4	0.012	100	0.30
Fumba	fresh:	plc	4	0.064	100	1.60
		crb	1	0.408	25	10.20
		plc	9	0.272	225	6.80
		crb	1	0.436	25	10.90
		plc	9	0.418	225	10.45
Fumba	salt:	plc	1	0.030	25	0.75
		cf	1	0.086	25	2.15
		plc	6	0.316	150	7.90
		biv	1	2.078	25	51.95
		plc	1	0.174	25	4.35
		crb	1	0.066	25	1.65
Makoba	fresh:	plc	4	0.001	100	0.025

Makoba salt: (No macrobenthos). Generally this site was poor in macrobenthos).

Appendix 3.2. Cont.

May 1998:

Site:	Habitat:	Group:	Total:	F/weight(g):	Individuals.m^{-2}:	g.m^{-2}:
Chwaka	salt:	plc	28	1.416	700	35.40
		plc	46	2.378	1150	59.45
		biv	1	2.644	25	66.10
		crb	2	0.190	50	4.75
		crb	2	0.408	50	10.20
		plc	23	0.906	575	22.65
		biv	1	2.214	25	55.35
		crb	3	1.144	75	28.60
		plc	22	0.514	550	12.85
Chwaka	fresh:	gst	2	0.468	50	11.70
		plc	2	0.670	50	16.75
Paje	salt:	biv	1	1.828	25	45.70
		plc	11	0.512	275	12.80
		plc	4	0.220	100	5.50
		plc	13	0.286	325	7.15
Paje	fresh:	nmt	10	0.066	250	1.65
		spc	4	1.860	100	46.50
		plc	4	0.126	100	3.15
Fumba	salt:	plc	6	0.056	150	1.40
		plc	13	0.198	325	4.95
		crb	1	2.142	25	53.55
		plc	13	1.602	325	40.05
		spc	5	2.816	125	70.40
		plc	11	0.586	275	14.65
		crb	1	0.576	25	14.40
Fumba	fresh:	plc	2	0.092	100	2.30
		plc	1	0.222	25	5.55
		biv	1	2.384	25	59.60
		crb	1	0.426	25	10.65
		gst	3	0.240	75	6.00
		plc	3	0.182	75	4.55

Site:	Habitat:	Group:	Total:	F/weight(g):	Individuals.m^{-2}:	g.m^{-2}:
Makoba	salt:	nmt	4	0.002	100	0.05
		crb	1	1.018	25	25.45
		plc	2	0.022	50	0.55

Makoba fresh: (No macrobenthos)

Key: (plc=polychaete; biv=bivalve; crb=crab; gst=gastropod; spc=sipunclid; nmt=nematode; sq=squid; ces=cestode; cf=crayfish/shrimp).

Appendix 3.2. Cont.

November, 1998:-

Site:	Habitat:	Group:	Total:	F/weight(g):	Individuals.m^{-2}:	g.m^{-2}:
Chwaka	salt	plc	1	0.006	25	0.15
		plc	2	0.028	50	0.70
Chwaka	fresh	nmt	1	0.05	25	1.25
		plc	1	0.006	25	0.15
		nmt	1	0.05	25	1.25
Fumba	salt	plc	7	0.032	175	0.80
		crb	1	0.50	25	12.50
		plc	1	0.006	25	0.15
		plc	4	0.07	100	1.75
Fumba	fresh	plc	5	0.312	125	7.80
		plc	1	0.006	25	0.15
Paje	fresh	plc	1	0.006	25	0.15
		plc	23	1.61	575	40.25
Paje	salt	plc	8	0.298	200	7.45
Makoba	salt	-	0	0	0	0
Makoba	fresh	plc	1	0.006	25	0.15
		plc	7	0.02	175	0.50

Key:(plc=polychaete;biv=bivalve;crb=crab;gst=gastropod;spc=sipunclid;nmt=nematode;sq=squid; ces=cestode;cf=crayfish/shrimp).

Stockholm University
Department of Zoology

Anthropogenically induced changes in groundwater outflow and quality, and the functioning of Zanzibar nearshore ecosystems Part II

Ron Johnstone, Stefan Gössling, Petra Lundgren, Erik Hansson, Martin Ekman, & Johan Hast
Stockholm University, Stockholm, Sweden

Introduction and Background

Tropical marine ecosystems characteristically have low levels of dissolved nutrients. In spite of this, however, tropical marine ecosystems also typically exhibit a high level of productivity due to an efficient recycling of nutrients (Mann, 1982). In conjunction with this, these ecosystems are generally considered to be limited by either nitrogen, or phosphorous, (Carpenter and Capone, 1983; Lunghurst 1981). Further, as noted by Kinsey (1978), deviations in the ambient nutrient regime can have considerable effects on, for example, biotope primary production.

At another scale, and in line with the recycling and internal conservation of nutrients, it is important to note that the different biotopes comprising most tropical coastal areas all play a significant role in the maintenance of the nutrient balance of the larger ecosystem (D'Elia and Webb 1977). These biotopes include mangroves, seagrass meadows, and coral reefs. Consequently, the different biotopes are of great importance for both the functioning of the ecosystem, and for the integrity and economy of the human populations which often depend on them for sustenance (Saenger et. al., 1983; Munro and Williams, 1985; Pointer et. al., 1989).

Given the inter-linked nature of coastal biotopes and the intrinsic character of the ecosystems they comprise, various human activities have been shown to have a significant impact at both a biotope level (Smith et. al.1973; Johnson & Johnstone, 1995), as well as at a larger ecosystem level (Ngoile and Horrill, 1993). In particular, the use of destructive or non-sustainable resource extraction methods (e.g. dynamite fishing and coral mining), and the release of pollutants into coastal areas are perhaps the two key areas where human activities are presently most damaging for coastal marine ecosystems (Ngoile and Horrill, 1993; Baalsrud, 1967). Specifically, the impact of pollutants and effluent has been shown to have a major impact on certain coastal biotopes (Smith et. al.1973; Linden, 1990). Clearly, point source inputs are a major problem in many areas, however, low level, more diffuse inputs such as might occur through, for example, groundwater inputs, may also be a potential problem (Dollar and Atkinson, 1991).

Unguja Island (Zanzibar), off the East Coast of Tanzania, is a limestone based island which is comprised of a number of aquifers that underlie almost the entire island (Zanzibar Hydrological Survey, 1987). These aquifers provide virtually all of the potable water on Zanzibar, and also form the freshwater base for virtually all of the major mangrove forests along the Zanzibar coastline (Zanzibar Hydrological Survey, 1987). In recent years, Zanzibar has undergone a significant population growth with an indigenous population growth rate of approximately 3% (Bureau of statistics, 1993) and a rapidly expanding tourist industry. The number of visitors to the island has increased by 12.5% per year from 1992 to 1995 (Bureau of statistics, 1993). As a result, the economy of Zanzibar is becoming increasingly dependent on money derived from tourism, which accounts for an estimated 70% (Toni, 1996; *pers. comm.*) of all major investments made on the island. Also, associated with this heavy influx of tourists, there has been a boom in development with large numbers of hotels and hostels being constructed along the island foreshores.

Despite the attention that some impacts of tourist development are receiving, one in particular, is still poorly addressed. Little, if any, data exists related to the effects that increased coastal habitation may have on ground based fresh water supply, and the marine environment. The wide spread practice of channeling waste materials into pit latrines and larger sewage pits (ODA survey, 1992; Johnstone *pers. comm.*) may adversely affect the quality of fresh water when these waste products leach into sub-terrainean water sources. As virtually all fresh water in these coastal areas is obtained from wells supplied by such aquifers, any contamination of them could lead to the release of nutrients, such as nitrates and phosphates, at abnormally high levels into the surrounding lagoons. The significance of this is amplified by an interim survey conducted by the Institute of Marine Sciences (IMS) on Zanzibar (*pers. comm.*); which indicated that there are significant groundwater inputs into the coastal lagoons and biotopes along the Zanzibar coastline. Consequently, the contamination of aquifers by effluent from hotels and other coastal habitations may have a considerable impact on the nutrient dynamics of the surrounding marine environment.

One apparent example of this, which warrants verification, are the villages of Bwejuu and Paje on the East Coast of Zanzibar. Intermittent samples taken by researchers associated with IMS on Zanzibar have shown a substantial increase in nitrate/nitrite levels in ground water taken from wells in the Bwejuu area. Bwejuu has experienced a several fold increase in the number of visitors over the past years, it historically has the largest population in the area, and large hotels are being constructed along the beaches north of the village. This construction has increased the local job opportunities and so the population of Bwejuu has also consequently increased due to immigration from other areas. By comparison, a southern neighbouring village, Paje, has not experienced the same increase in tourist related activities and it has the smallest population in the area. The third village in the vicinity, Jambiani, is somewhat between the other two villages in terms of population and tourism. Whilst it has more tourist facilities, and its population is between that of Paje and Bwejuu, the tourist facilities are essentially small guest houses and the village covers a much larger area than either Bwejuu or Paje. It should also be noted that the basal population in most coastal areas of Zanzibar is simultaneously increasing at a fast rate.

In view of this background, initial studies showed there to be considerable concern among inhabitants of the coastal villages regarding the supply and quality of their water supply. The bulk of the information obtained from these initial surveys is summarised below in table 1.

Village	Population	Major Income sources	Mean Water Use (Litres)	Supply System	Number of Guest houses	Perceived Problems
Jambiani S	2795	Fishing, agriculture & seaweed farming.	34.8	Piped from cave approx 2km away. 14 wells, 4 for drinking water.	6	Pipe: smell (10%), illness (7%). Well: Salt (100%), smell (40%), Illness (33%).
Jambiani N	3442	Fishing, seaweed farming.	31.4	As above. 30 wells in use.	6 (3)	Pipe: supply (7%). Well: Salt (100%), smell (60%).
Paje	1750	Fishing, agriculture, & seaweed farming.	33	20 wells of different depths and use.	4 (2)	Well: Salt (20%), Supply (6%)
Bwejuu	2620	Fishing, seaweed farming.	31.1	16 wells of different use.	8	Well: Salt (94%), Illness (6%), Smell (6%)
Chwaka	1820	Fishing, seaweed farming.	36.8	Piped water from Ufufuma (3km away). 10 wells supplement.	1Hotel, 1 guesthouse	Pipe: Smell (7%), Salt (100%). Well: Salt (100%), smell (83%), Illness (43%).
Uroa	3000	Fishing, seaweed farming.	31.9	Piped from a well west of village. Approx. 8 useful wells.	5	Pipe: Salt (97%). Well: Supply (34%), smell (7%), Salt (100%).

Table 1. Summary of the initial socio-economic information gathered on groundwater use and quality from villages on the Eastern coast of Unguja Island, Tanzania

In the light of the above material, data has subsequently collected on groundwater release rates in a number of areas adjacent to villages and at more remote locations. As is presented below, samples were analysed from all of these sites to determine water chemistry characteristics. The preliminary data showed NH_4^+ concentrations to generally be ≤1.0µM, and NO_x (NO_3^-+NO_2^-) concentrations to be between 4 to 7µM.. Also, soluble reactive phosphate was consistently ≤1.0µM. It was further observed that the waters being released from aquifers in the Mapopwe Creek mangroves, Chwaka Bay, were often sulphitic, and had a much higher salinity in the dry season than groundwater's being released in the lagoon at Paje; on the south Eastern shore of the island.

At the same time, some large areas of semi-diffuse groundwater release were identified and mapped in the lagoon at Paje. The largest area covered approximately 1.1km^2 and approximately 60% of this area showed varying levels of groundwater release. The mean output measured in the initial period was approximately 4,200litres.m^{-2}.d^{-1}. This value was obtained during the short dry season, and so measurements were also necessary during the wet seasons.

Objectives

In line with the initial background data presented above, the ensuing research set out to investigate the chemical character and socioeconomic's of the ground water extracted from wells in the villages of Bwejuu, Paje and Jambiani; Chwaka and Uroa. All of which are located on the East Coast of Unguja Island. In addition to groundwater associated with villages, investigation was also made of the groundwater being released within the Chwaka Bay mangrove forest, and in the lagoon at Paje. The overall study set out to investigate the following:

1. The level of dissolved inorganic nutrients in groundwater as reflected by well water, and by groundwater released from direct discharge points.
2. To examine the level of dissolved inorganic nitrogen (DIN) and soluble reactive phosphorous (SRP) released into coastal waters so that an assessment can be made of the possible contribution that groundwater's may make to the nutrient budget of coastal aquatic ecosystems.
3. To examine alterations in the benthos adjacent to groundwater discharges as defined by meiofauna populations, and microphytobenthos communities. The meiofauna have been used elsewhere as an indicator of benthic perturbation. Also this study includes an examination of nitrogen fixation rates since this process can be critical in delivering new nitrogen to the otherwise nitrogen limited tropical waters.
4. The socioeconomic aspects of groundwater use and how this might impinge on the long-term sustainability and quality of groundwater resources.

Material and Methods Nutrient Studies

In order to estimate both the nutrient load carried by groundwaters, and the possible contribution this could make to coastal nutrient budgets, nutrient concentrations were measured in well water samples from six villages[1]. Also, samples were taken from groundwater release points in the Chwaka Bay mangrove forest and beach area, as well as from a large area of direct release in Paje Lagoon. In the later case, 4 replicate samples were taken randomly from the larger area of groundwater release. This was particularly the case for Paje, but also applied to the beach location in Chwaka Bay. The mangrove forest locations were all discrete holes with groundwater discharge.

[1] It should be noted that although the two sites associated with Jambiani bear the same name, the northern and southern locations are well separated by a break in population and human activity. The two sites are approximately 2km apart.

Well location

The well sampling was conducted in the villages of Bweju, Paje and Jambiani (north and south), Chwaka, and Uroa (Figure 1).

The wells used for sampling were randomly selected to represent the range of well conditions in each village. This, therefore, included wells within densely populated parts of the village, as well as wells from intermediate and sparsely populated village areas. In total, 40 wells were sampled.

It should also be noted that the areas studied all fall within the geological zone 300 metres from mean sea level. This would imply that they would be most affected by the ocean and its tidal regimes (see Figure 1). To elucidate any effect the tide may therefore have on nutrient concentrations, water samples were taken during three different low tides and three different high tides.

Well Water Sampling and Analysis

All water samples were collected directly from the wells using a pre-rinsed bucket attached to a rope that was lowered into the wells. Water samples were then taken with a pre-washed 50ml syringe and the sample was immediately filtered through a pre-washed, 45mm, glass

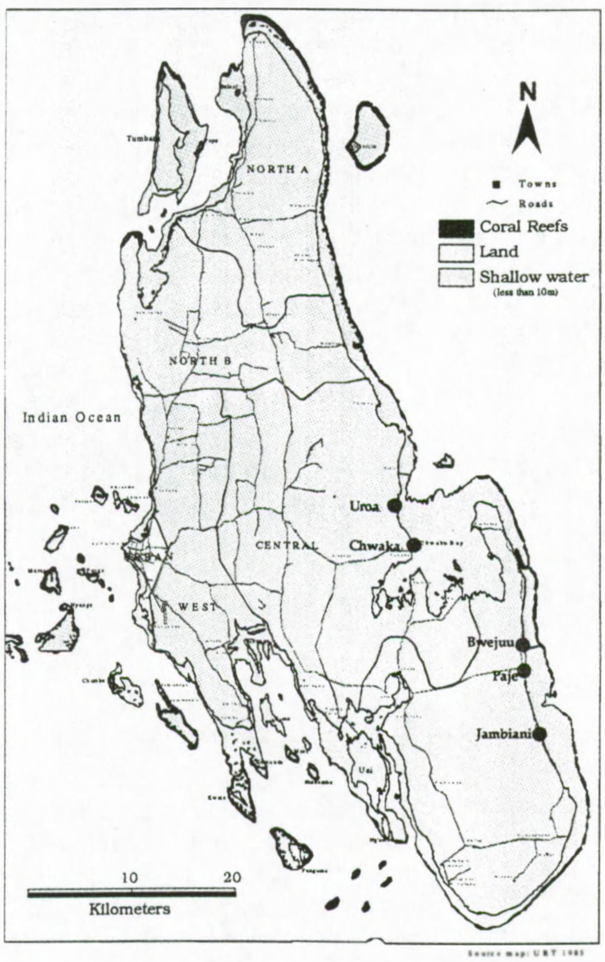


Fig. 1 Map of Unguja Island showing the location of the study villages Paje, Bwejuu, and Jambiani on the South-East coast.

fiber filter. To avoid contamination of the sample, all collection apparatus was also rinsed three times with the water from the particular well being sampled prior to sample collection. Each sample was stored in an acid washed 25ml plastic vial; total volume of 20ml. A total of 3 samples were taken from each well on each occasion, and these were immediately frozen until analyses could be conducted. In the case of ammonium analysis samples, the samples were only chilled on ice after filtration and the initial reagent in the analysis (phenol) was added as a preservative. These samples were then analyzed within 6 hours of collection.

All chemical analyses were conducted at the IMS laboratory in Zanzibar town. The concentration of ammonium and phosphate in the collected water was determined using the methods described by Parson et al (1984). The concentration of nitrate+nitrite (NO_3^-+NO_2^-) was determined using a Technicon AAII auto-analyzer.

Groundwater Release Point Sampling

In the case of direct discharge sites for groundwater, sampling was conducted differently depending on location and the mode of water discharge. In Paje lagoon, the largest area of consistent groundwater discharge existed at the interface between mean sea level and the lagoon proper. This site covered an approximate total area of 1.1km^2, and consisted of a range of large (approx. 1m dia.), and small (≤ 10cm dia.) release points. All of the release points were filled with sand that was more or less suspended by the outflow of water.

Water samples were taken with a 50ml disposable syringe and all samples were filtered directly through a pre-washed and rinsed Whatman GF/F filter. A total of 8 release points were sampled within the larger area, and triplicate samples were taken on each occasion.

A similar situation existed at in the case of Chwaka Bay intertidal zone, however the number of release points was much lower, and they were spread further apart. Further, the release of water was far less consistent and the release occurred more often as a slow flowing seepage without enough pressure to suspend the sand as was observed at Paje.

By comparison, the third location in Chwaka Bay mangrove forest consisted of very discrete holes in the sub-tidal coral rag. Again, these holes were very few in numbers and ranged in size from approximately 80cm dia., to ≤ 15cm dia.

Nutrient analysis was conducted as for well samples.

Flow Estimation

Groundwater flow was only estimated for direct outlet points and was conducted by a combination of volume displacement and volumetric methods. In the case of surface release through channels, the volume of the channel was estimated and then the flow of water was timed through the volume using vegetable dyes in the water. The volume exchange over time was then used to calculate the volume released.

In the second method, 4 replicate small chambers of 70cm diameter were placed over the release point and the rate at which they were filled was measured over time. Again, the flow was calculated over time and normalised in this case to the area over which the release occurred.

Flow rates estimated using these methods gave unreliable results for the sites with very low flow rates. In sites with stronger rates, however, the methods gave repeatable results.

Mean flow rates for the Chwaka beach locations were below accurate detection for the methods available.

In contrast, the locations in Chwaka Bay mangrove forest gave a mean rate of 23 $m^3.d^{-1}$, and the larger intertidal area near Paje village gave a mean rate of 43.4 $m^3.d^{-1}$. Over the total area of release in Paje this corresponded to approximately 281,853 $m^3.d^{-1}$.

Material and methods micro- and macrophytobenthos and macrozoobenthos studies

After an initial pilot study, a more comprehensive study was conducted in June, July and August of 1997 on a 1,2 km long coastal beach area in Paje lagoon and in the mangrove area at Mapopwe Creek in the inner reaches of Chwaka Bay on the East coast Zanzibar.

Paje lagoon is an open shallow lagoon with a fringing coral reef approximately 2km from the shore and a tidal fluctuation of about 2 meters. The bottom is sand covered and has only very sparse seagrass or macrophyte cover for the bulk of the lagoon. Some areas are more consolidated seagrass beds but these are limited in distribution.

The nearby coastal village of Paje has a population of about 1000 people and there is a presence of a large number of both man-made wells and pit latrines. Therefore the likelihood of human impact was considered high, especially in view of the numerous outlets of groundwater into the lagoon which are obvious and easy to define during low tide.

By comparison, Mapopwe Creek in Chwaka, is surrounded by mangrove forest. The bottom is bedrock, largely covered with mud and there are no adjacent human settlements or associated sources of impact, though there is a large agricultural area that backs on to the forest approximately 8 km to the west. The agricultural area also uses groundwater for irrigation although this is limited (Zanzibar Dept. of Land and Environment). There are few clear groundwater outputs but since there are no freshwater streams connecting to the creek, groundwater is considered to be the only permanent freshwater source to the mangrove forest apart from precipitation. It was therefore considered that any change or fluctuation in quality or quantity of the groundwater may have an effect on the forest.

In this study, the following parameters were measured since they have been shown elsewhere to be potentially sensitive to groundwater input (Lapoint et al. 1990): Total organic nitrogen (TON)/ total nitrogen (TN), algal biomass, chlorophyll and pheaopigments in surface sediments, macrofauna, salinity, pH, temperature, soluble reactive phosphate (SRP), ammonia and nitrate/nitrite.

As indicated earlier for the pilot study, groundwater discharge rate was also measured in an effort to estimate the output of groundwater into the respective aquatic ecosystems.

In the case of Paje Lagoon, a randomly selected 33 meter by 33 meter area in the intertidal tidal zone area was mapped where groundwater output existed. The area was completely covered with seawater at high tide and totally "exposed" during low tide. In this area, 3 larger holes of output were randomly selected. Water samples were collected from the output of each hole and filtered as described above. Salinity, pH and temperature of groundwater output were measured and triplicate samples were taken and analysed for SRP, ammonia and nitrate/notrite.

Also, a 1 x 1 meter quadrat was placed over 3 randomly selected groundwater release points and then also over 3 other randomly selected spots 3 meters away from any groundwater release points. In addition, at a minimum distance of 30 meters away from any groundwater release point, 3 more 1 x 1 meter quadrants were also put down.

In each quadrant 10 random samples of 5cm^3 of sediment were taken in the top 1.5 cm of sediment and analysed for chlorophyll and phaeopigments. In addition, 5 randomly distributed samples of 5 cm of top soil (approx. 110cm^3) were also collected, pooled and examined for macrofauna. Finally, in each 1m^2 quadrat all macrophytes were taken for identification and their biomass estimated. A survey of the 33x33m^2 area was also conducted, to assess whether the macrophytes (essentially just *Ulva* sp) were associated with outputs of groundwater and whether they reflected the distribution of the groundwater outputs.

For the estimation of water discharge rates, the flow rates of three large groundwater holes were measured at low tide using a seepage meter as described above.

In the case of Chwaka Bay, 2 holes with groundwater seepage were selected in different parts of Mapopwe Creek, however as described later in "Results", one of them was later identified as a subterranean channel carrying saline water and not freshwater. As described earlier, water samples were collected from the output of each hole, and analysed for salinity, pH and temperature. Triplicate water samples were also taken and analysed for SRP, ammonia and nitrate/nitrite.

For macrophyte and macrofauna estimates, the two holes were covered with a 1 x 1 meter quadrat and these were compared with 3 randomly placed quadrats situated at a distance of 3m from of any visible source of groundwater output. A further 3 quadrats were randomly placed at a distance of at least 30m from any visible source of output. In each quadrat 10 random samples of $5cm^3$ of sediment were taken from the top 1.5cm of mud and analysed for chlorophyll and phaeopigments. Five randomly distributed samples of the upper 5 cm of surface sediment (approx. $110cm^3$) were also collected, pooled and examined for macrofauna from each quadrant. The biomass of macrophytes was also measured.

Results Nutrient Studies

Standing Stocks of Nutrients in Groundwater

From the overall study of nutrients in groundwater, nutrient concentrations showed considerable variation between wells and between villages. There was, however, no clear correspondence between the concentration of each nutrient type and the level of other nutrients. At a village scale, Jambiani showed the highest mean concentration for ammonium and soluble reactive phosphorous (SRP), but Bwejuu gave a significantly higher mean for $NO_3^-+NO_2^-$ (Table 1). The range of values for each nutrient was 0.16 to 2.70 $M.l^{-1}$ SRP, 3.99 to 11.70 $M.l^{-1}$ $NO_3^-+NO_2^-$, and 0.28 to 1.3 $M.l^{-1}$ NH_4^+. As is shown in Figure 2, these concentrations changed with the tide, with the greatest change being observed for $NO_3^-+NO_2^-$ at Bweju. It should be noted here that a technical failure lead to the loss of some nitrate samples. This included all of the samples for Paje and so the difference shown for Paje in Figure 2 is due to a lack of data for high tide.

It should also be noted here that, in some instances, some wells dried up at low tide and so it was not possible to obtain an accurate sample.

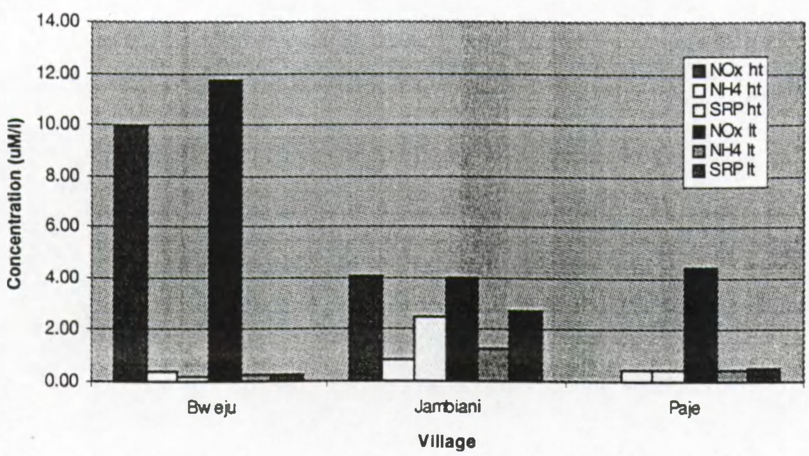


*Fig 2 Graph of mean nutrient concentrations at high **(ht)** and low tide **(lt)** for each of the villages showing clear tidal influence.*

Follow up sampling in the wet season showed significantly elevated levels of NO_x in well water in some locations. This was particularly so in several wells in Chwaka Village and Bwejuu village where concentrations were as high as 295 $\mu M.l^{-1}$. As is discussed in the socioeconomic section below, both of these villages had more consistent problems with water quality and potable water was sometimes limited to only a few wells.

In terms of groundwater outflow into the adjacent aquatic environment, there was considerable difference between the groundwaters from Chwaka Bay, and those from Paje village.

After initial sampling, it was realised that in the case of the Paje study area, there was no significant difference between mean nutrient concentrations for the different outlet points examined. Because of this, the samples from all outputs there were pooled together for the nutrient analysis. By comparison, the Chwaka sites differed significantly from each other.

As shown in figure 3, compared to weel samples, the levels of SRP and NH_4^+ were low in the outlet waters from Paje with a mean of 0.3 and 0.15$\mu M.l^{-1}$ respectively. These values are slightly higher than observed in the initial sampling undertaken but this is considered to be because the initial sampling did not cover many tidal cycles and days. In the initial study, mean values were from non-detected to 0.1 $\mu M.l^{-1}$ for SRP, and between 0.1 and 1.2 $\mu M.l^{-1}$ for NH_4^+. By far the dominant nutrient species in Paje groundwater was NOx with a mean concentration of 5.8 $\mu M.l^{-1}$.

Compared to Paje, the groundwater at the Chwaka Bay site showed generally low nutrient concentrations with SRP being the dominant nutrient species (Figure 3).

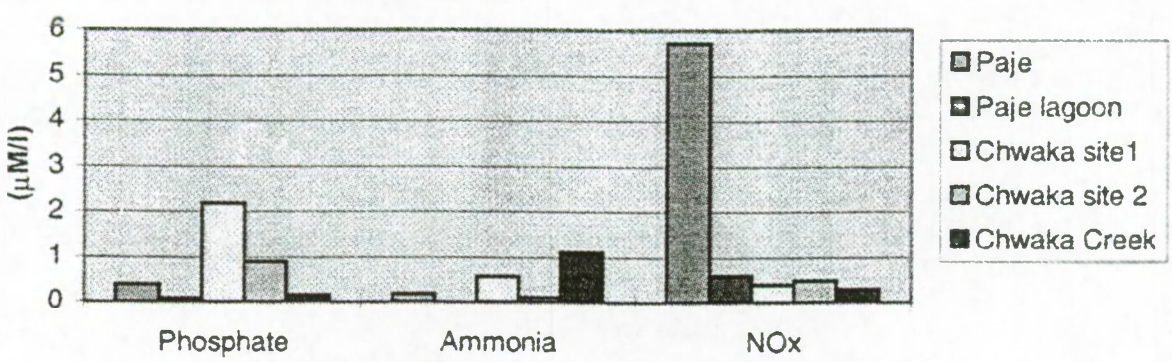


Figure 3. Mean nutrient concentrations in groundwater outlets at Paje Village and Chwaka Bay.

Also of note, for the Paje site, the levels of nitrate/nitrite in the groundwater tended to vary a lot between subsequent low tides and also between different release points depending on time of measurement (Figure 4). This study was unable to determine a reason for this but considered it to be due to possible variations in the flow from the different contributing aquifer areas that are themselves probably connected to the different well and latrin areas with differing contamination levels.

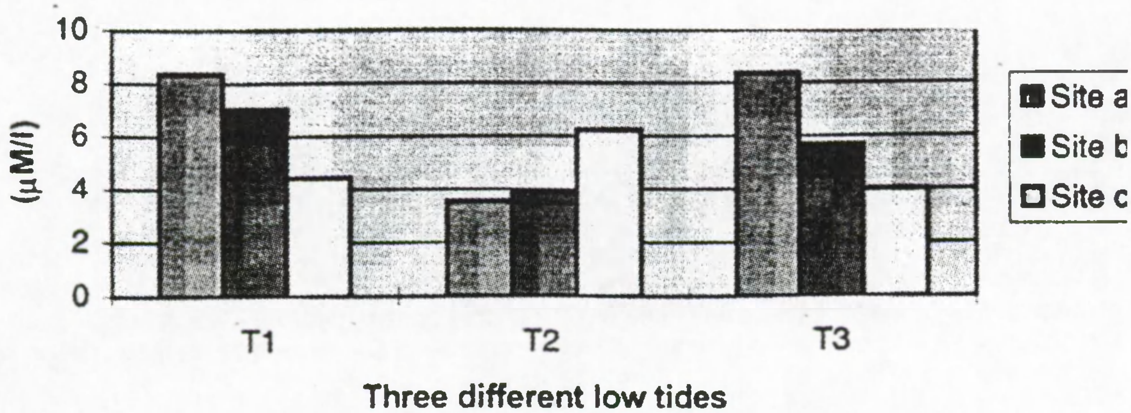


Fig 4. Variations in the mean concentration of NOx in groundwater samples taken from outlet points adjacent to Paje village.

Results macrophytes and microphytobenthos associated with groundwater outlets

A significant difference was observed in the apparent relationship between algal distribution and groundwater release between the Paje site and the Chwaka Bay site. In the case of Paje, there was consistently a large quantity of macrophytes in the vicinity of the groundwater release points whilst in Chwaka Bay there were no macrophytes associated with the outlet points. In Paje, the macrophyta consisted only of *Ulva pertusa* and this was only found at the groundwater release points. Biomass estimates showed that the mean dry weight of *Ulva* within each quadrat on the outlet points was 95g. m^{-2}. Further, the survey conducted over the entire in the 33m x33m ($999m^2$) area showed that there were no U. *pertusa* growing anywhere without a close (< 1 m) connection to a groundwater release or seepage point (n=31).

No macrophytes were found in the equivalent area at Chwaka Bay suggesting that other factors are also involved in the distribution of *U. pertusa* in that area.

In addition to macrophytes, an examination of chlorophyll *a* levels in the upper 1.5cm of sediment showed there to be a well represented microphytobenthos however no significant difference was found between different distances from the groundwater sources. It is worth mentioning, however, that in close proximity of the groundwater points in Paje, the dense macrophyte growth would shadow the benthos and hence may retard microphytobenthos growth in these areas. The results of Chlorophyll *a* analyses for Paje are summarised in Figure 5 below.

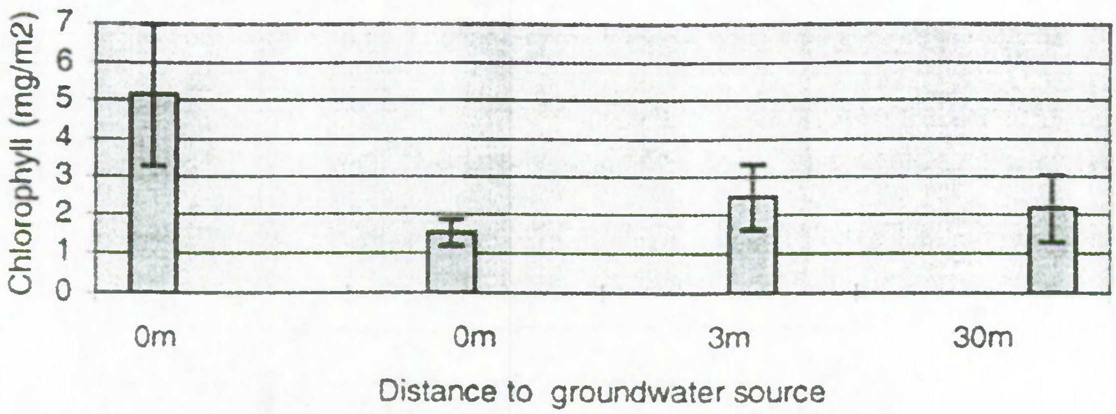


Fig 5. Chlorophyll a concentrations in surface sediments taken at different distances from groundwater outlet points in Paje Lagoon. Values are in mg Chl a.m^{-2} with standard deviation bars.

Compared to Paje, sediment samples from the Chwaka site showed significantly less chlorophyll *a* with mean values approximately half of that observed for Paje sediments (Figure 6). Also, there was no significant difference between locations in Chwaka however the variation was much higher at site 1 than site 2.

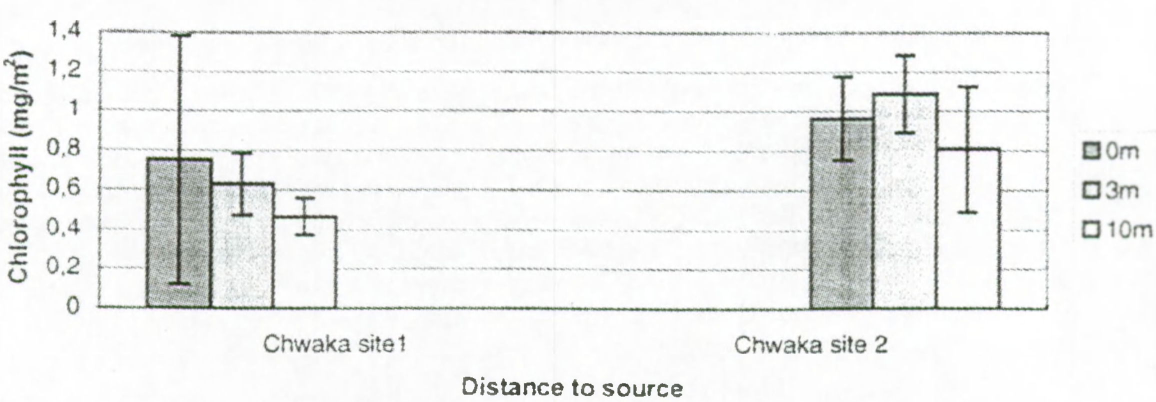


Fig 6. Mean chlorophyll a levels in surface sediments taken at different distances from groundwater outlet points in Chwaka Bay. Values are in mg Chl a.m^{-2} with standard deviation bars.

Results macrofauna studies

The macrofauna investigation showed that there were no macrofauna associated with groundwater outlets at the Chwaka Bay site, however more macrofauna (Polychaetae) were found closer to the groundwater outlet points in Paje than at sites further away. This difference between distances was not statistically significant (Figure 7), however, this can be largely attributed to the high variance in numbers at the zero distance locations.

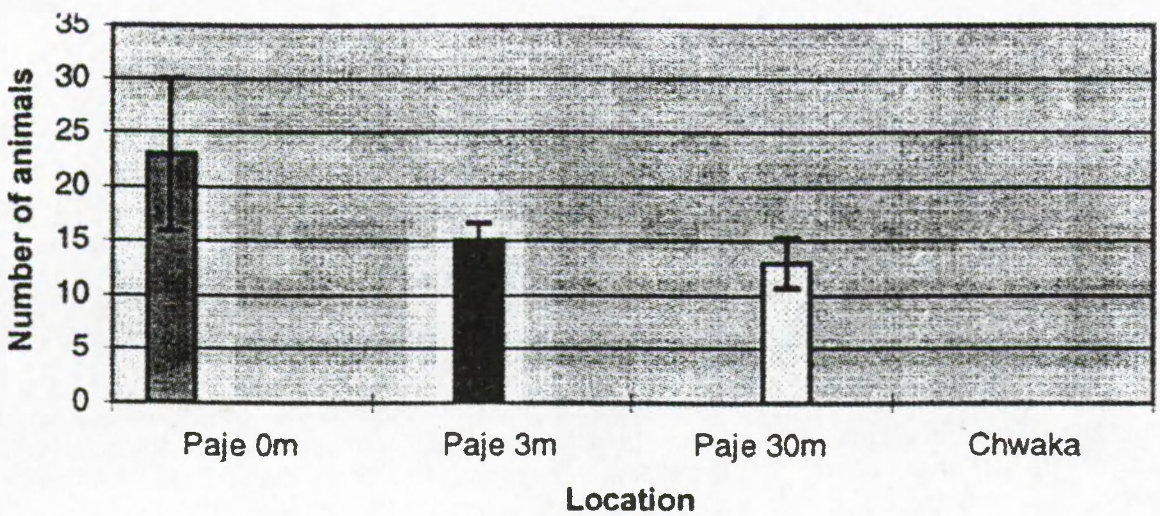


Fig 7. The mean number of macrofauna found in sediments at different distances from groundwater outlets in Paje Lagoon and Chwaka Bay. All Chwaka Bay samples gave a zero result, and the bars represent standard deviations for the mean.

Methods meiofauna studies

Meiofauna was sampled on three occasions, June and September 1997 and May 1998 from three different locations: Chwaka Bay, Paje and Fumba. At each location samples were taken in the vicinity of a freshwater output and control site without fresh water influence. At each site 3 replicate 5 cm deep meiofauna cores (of 10 cm^2) were retrieved and brought back to the laboratory. The samples were washed through 500 and 40mm sieves and meiofauna extracted from the 40 mm fraction using Ludox (colloidal silica polymer) at a specific gravity of 1.15. The meiofauna was enumerated and identified to major taxa in a petri dish under a stereo dissecting microscope.

Results meiofauna studies

Altogether 13 major meiofauna taxa were recorded from the samples of which nematodes dominated. Other important taxa were harpacticoid copepods, turbellarians and polychaetes. As shown in Table 1 the results of ANOVA demonstrate that the only difference in sites for taxa (fresh water and controls), occurred with the polychaetes in September 1997. However, there were often several significant differences among locations (Table 2).

Also, the multi-dimensional scaling ordination (MDS) did not reveal any obvious pattern in the community structure among the sites (Appendix 1) at any of the sampling dates.

Table 2 Summary of the ANOVA on meiofauna data.

Summary of the 2-way ANOVA for the four major taxa from three sampling occasions
LxS=interaction between Location and Site, ns=not significant, *=P<0.05, **=P<0.01, ***=P<0.001

	Jun-97			Sep-97			May-98		
	Location	*Sites*	*LxS*	*Location*	*Sites*	*LxS*	*Location*	*Sites*	*LxS*
Nematoda	ns	ns	ns	***	***	***	ns	ns	ns
Harpacticoida	**	ns	ns	ns	ns	ns	ns	ns	ns
Turbellaria	**	ns	*	ns	ns	ns	*	ns	**
Polychaeta	*	ns	ns	***	**	ns	ns	**	***
Others	*	ns	ns	ns	ns	ns	*	ns	ns

Overall, the data generally lack any clear pattern. This is considered to be due to the nature of the freshwater input in the intertidal areas where the tidal waters dilute the impact of the incoming groundwater for a large part of the day. In the case of Chwaka Bay mangrove forest sites, the groundwater had a mean salinity of 3.2% so salinity was not significantly different to tidal waters.

Materials and methods nitrogen fixation studies

As indicated earlier, in addition to examining any direct impacts that groundwater inputs might have on the biota such as meiofauna, it was also undertaken to examine the possible impact that groundwater inputs may have on the overall nutrient dynamics of the recipient areas. Accordingly, studies were also made on the possible effects that groundwater inputs may have on nitrogen fixation. Since most tropical aquatic ecosystems are considered to be nitrogen limited a key factor in their nitrogen dynamics is the contribution of new nitrogen that enters via nitrogen fixation (N-fixation). Any significant increase or loss of this process may potentially have a significant impact on the local aquatic environment.

The effect of groundwater on N-fixation was examined at three different sites. Two of these were located in the mangrove forest of Chwaka (Chwaka-1, Chwaka-2) and one in the coral lagoon of Paje.

At Paje, a large area was covered by many groundwater sources. Samples (sediment cores) were taken at three different distances from six sources of different sizes; 1) One sample was taken at a distance of less than 50 cm from each of the six different sources (N=6). 2) One sample was taken at approximately 3 m from each of the 50 cm samples and not closer than 3 m from any other source (N=6). 3) Six samples were taken at about 100 m from the area of groundwater sources (N=6).

In the Chwaka area the sources that were found were single (no other source found within 100 m from the source). Two sources were chosen (Chwaka-1, Chwaka-2) and samples were taken at three different distances as above. Six samples were taken at each distance. Chwaka-1 was a considerably bigger source than Chwaka-2.

Sediment cores of the top 10-15 cm of sediment were taken using plastic core tubes (30 cm length, 4,5 cm dia.). After the sediment samples had been taken the sediment cores were carefully pushed up to the same height in all tubes. Any water above the core surface was then removed and after the cores had been sealed with rubber stoppers each core had an headspace of 100 ml of air.

Nitrogen fixation was assayed using the acetylene-reduction technique. Firstly, approximately 0.5 cm of sea water containing acetylene gas (20% v/v) was placed on top of the sediment surface. Acetylene gas was then injected with a syringe and needle through an air-tight septum in the core tube, to a final volume of about 20% (v/v). The cores were incubated *in situ* at *in situ* water temperatures for approximately two hours. After incubation, gas samples were taken with a syringe and injected into pre-evacuated 10-ml vacuum containers. The gas samples were brought back to lab and analyzed for ethylene production by gas chromatography.

Peak height responses for unknowns were compared to a calibration standard and the molar concentration of ethylene in the samples could thus be determined.

The calculated rate of ethylene production was converted into a N2 fixation rate by using the theoretical ratio of 3:1 for acetylene reduced compared to nitrogen fixed. This ratio was not experimentally verified so the calculated N2 fixation rates may not be entirely correct (rates have been observed ranging from 1.9-6.2:1 but mostly they are between 3-4:1).

Results nitrogen fixation studies

As summarised in Figures 8 and 9, there was a general trend toward decreased nitrogen fixation rates as you moved further away from the source of groundwater.

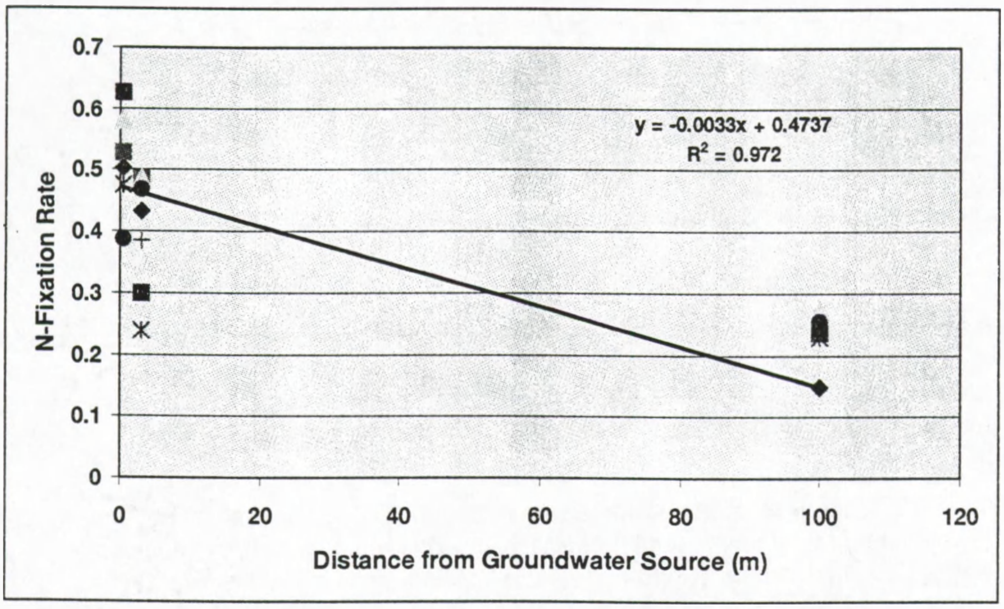


Fig 8 Nitrogen fixation rates ($nmolN_2.cm^{-2}.h^{-1}$) at different distances from the groundwater source at Paje Lagoon.

This trend is clearest in the Paje site where the decrease continued over 100m distance and the correlation in the data gave $r^2 = 0.97$. By contrast, the Chwaka Forest site showed a marked decrease between the edge of the source and 3m distance. This may be due to an acute impact of the groundwater but since the groundwater being released has a high salinity (mean = 3.2%), the effect may be due to other factors such as low oxygen tensions in the groundwater ($\leq$ 2ppm dO_2), or sulphidic compounds in the groundwater. This site consistently gave a strong sulphidic smell during sampling.

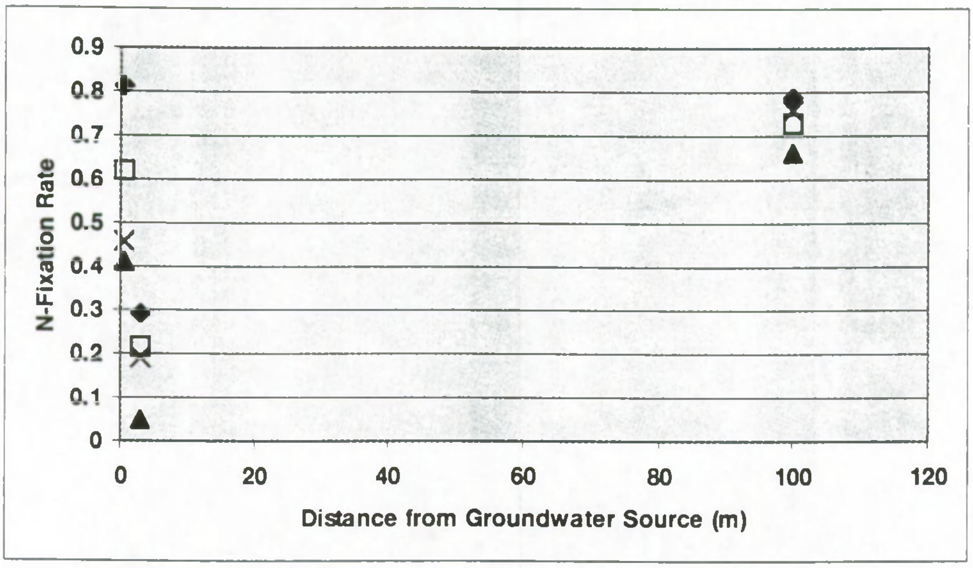


Fig 9. Nitrogen fixation rates ($nmolN_2.cm^{-2}.h^{-1}$) at different distances from the groundwater source in Chwaka Bay Forest.

In all cases, the mean nitrogen fixation rates observed were equal to or higher than background levels observed in random samples taken in similar sediments remote to the sources involved.

Socio-Economic Issues of Groundwater Utilisation

Aims and Background Information

As with the nutrient and biota work, a pilot study was conducted initially to assess the scope of the issue involved and the best method of addressing them. In addition to the scientific and information value of the work undertaken, it was also intended that the socio-economic information generated should particularly examine the potential management and long-term aspects of groundwater use.

Initially, population data was collected for each village in the pre-defined south coast area where local perceptions had indicated potential for environmental problems associated with groundwater contamination and use. A summary of the initial population data is given in Figure 10.

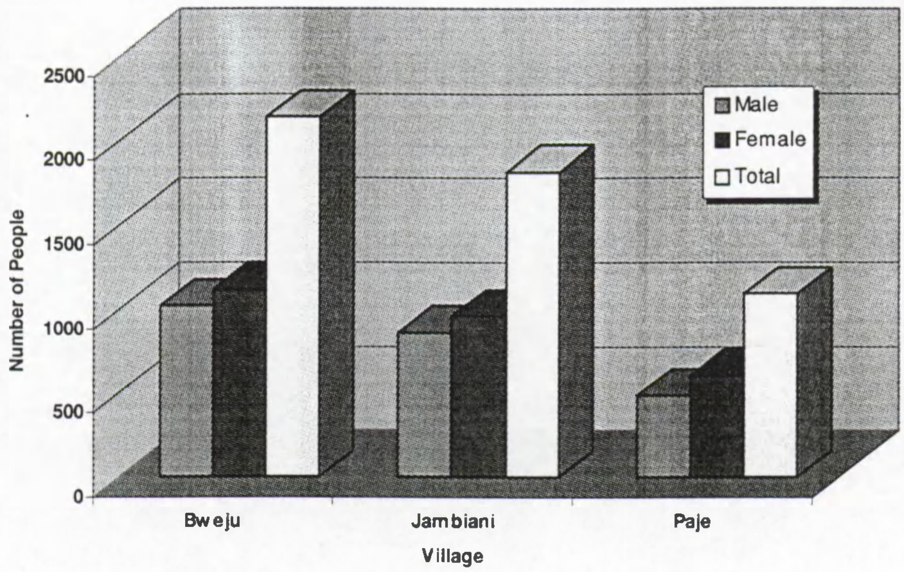
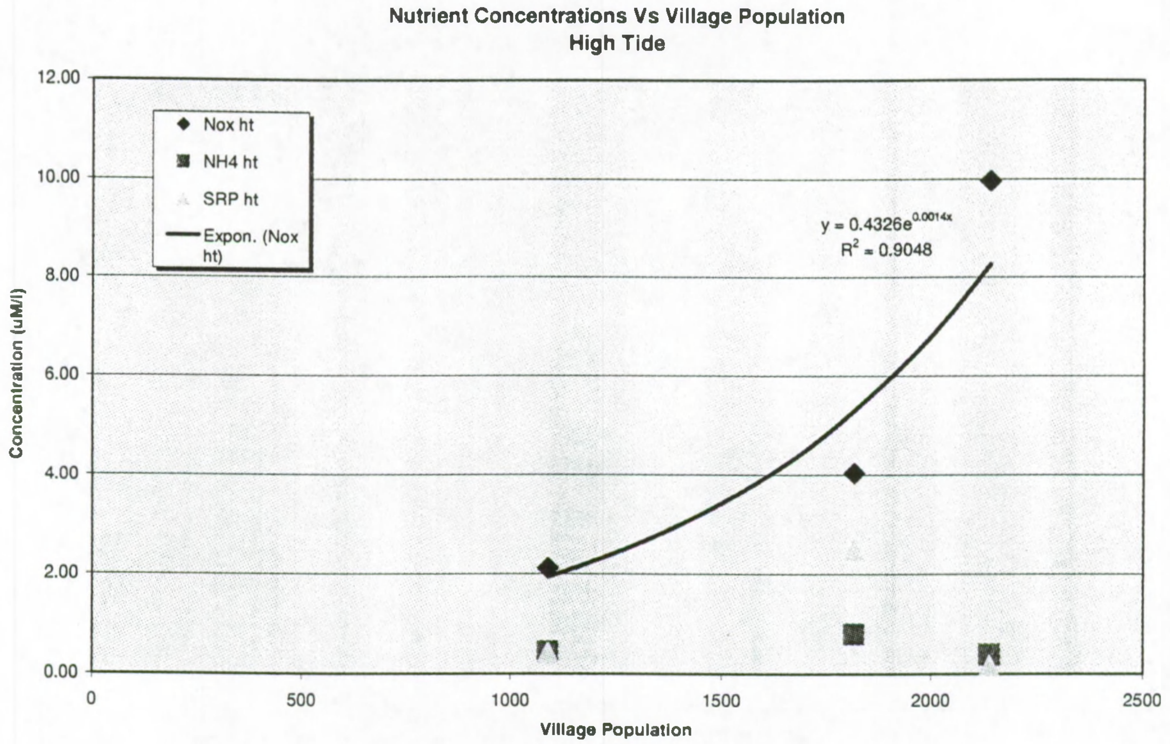


Fig 10 Village populations along the East Coast of Unguja Island.

As was suspected at the outset, Bwejuu had the largest population with Jambiani and Paje lying sequentially lower in number. Unfortunately, it was not possible to obtain an official definition of the borders for each village, so an estimate of area was made based on the approximate corner bearings given by village elders. This gave a total approximate area of 3km^2 for Jambiani, 2 km^2 for Paje, and 3km^2 for Bwejuu. These estimates give an approximate population density for the respective villages of 605 people/km2 in Jambiani, 544 people/km2 in Paje, and 713 people/km2 in Bwejuu.

Given the observed increase in NO_3^-+NO_2^- in some wells, it was deemed likely that this nitrogen was being transferred into the adjacent lagoon and hence may have an effect on the nutrient dynamics of the recipient environment. Ongoing hydrological modelling of the groundwater on Unguja Is. already indicates that there is a significant input of groundwater into most of the coastal zone around eastern Unguja Is., and this input appears to support a range of biotopes including the mangroves of Chwaka Bay (S. Mohammed, IMS, pers. com.).

In addition, the initial data on nutrient concentrations in wells appeared to indicate a relationship to human population number as presented below in figure 11.



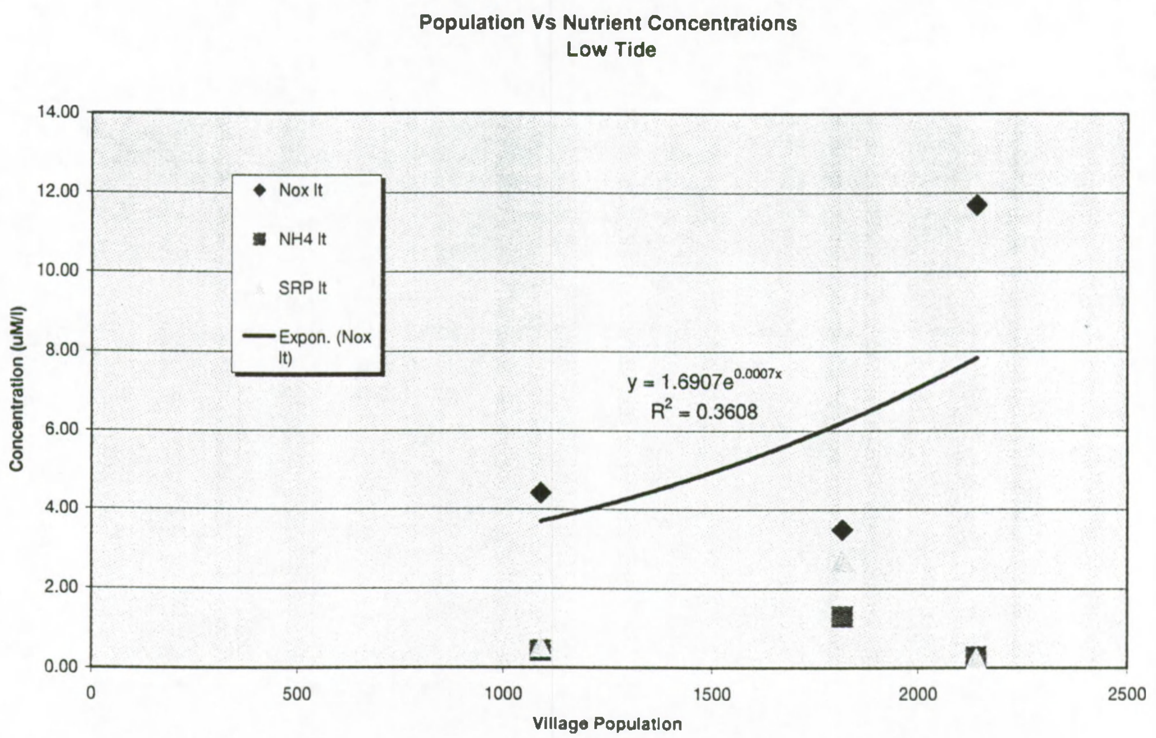


Figure 11a,b: Well water nutrient concentrations at high and low tide plotted against village population size.

In the light of this, the present study examines the potential role that human activities may have in modifying the nutrient dynamics of coastal groundwater and, subsequently, the biotopes receiving it.

The present investigation aimed to identify and, where possible, quantify the use of groundwater resources and the impact this might have in different areas.

Patterns of groundwater use - villages

Water use in 22 villages from Nungwi to Jambiani was investigated. Data was collected on population size, water supply systems, water quality, sewage, and use patterns of both locals and hotels/ guesthouses.

In 1988, the population of Unguja Island was about 375,500, and may today have risen to 530,000 (9). The coral rag areas have a comparably low population density. At present, approximately 35,000 people live along the coast between Nungwi and Jambiani, including all major villages and settlements.

Generally, some villages receive piped water, while others use the local wells. Piped water from springs and caves is available in Nungwi, Kigomani, Tundangaa, Matemwe, Mbupurini, Klima Juu, Pwani Mchangani Ndogo, Pwani Mchangani, Kiwengwa, Marumbi, Chwaka, and Jambiani. Uroa, Michamvi and Pingwe receive as well piped water, but in contrast to the other villages, it is pumped from boreholes or wells in proximity to the villages. Pongwe, Ndudu Mkubwa, Charawe, Ukongoroni, Bwejuu and Paje are dependent on well water.

Problems attributed to water are mainly the continuity of supply and the salinity. Basically, piped water has a good quality, but due to breakdowns or repairs, even villages connected to the pipe system have to use wells up to one month per year.

In general, the situation of the wells is different in each village, even though the number of freshwater-wells has decreased in almost all of them. Cholera cases due to well pollution occurred in Pingwe, Ndudu Mkubwa, and Pwani Mchangani. Water shortages occur at low tide, and in dry season (December, January, and February). Today, some villages have become very dependent on piped water.

The majority of water in the villages is used for sanitary purposes, cooking, and washing dishes or cloths, only little is used for cleaning or feeding animals. To assess the total amount of water used, and the percentage of different uses, a representative number of local inhabitants (n=107) was interviewed. Data was collected from adults, who were also interviewed on water use by their children. This way, additional data for 34 children was gained.

It turned out that some water uses are individual (taking showers, toilet purposes, drinking), while others are family based (cooking, washing dishes and clothes, cleaning the house and feeding the animals). To calculate *per capita* averages for the latter, the amount of water used per family was divided by the number of family members and added to amounts used individually. This possibly leads to a certain incorrectness, but a more weighted calculation did not seem feasible. Uses that were too seldom (e.g. washing bicycles) were not considered in calculations. To interpolate the total amount of water used on the East Coast, it was necessary to distinguish between adults and children. For this reason, water use in different age groups was compared, and children afterwards defined as aged 0-14, adults as 15 or older.

The amount of water used on average was calculated at 33.46 $l\ d^{-1}$ for children and 54.54 $l\ d^{-1}$ for adults (Fig. 12).

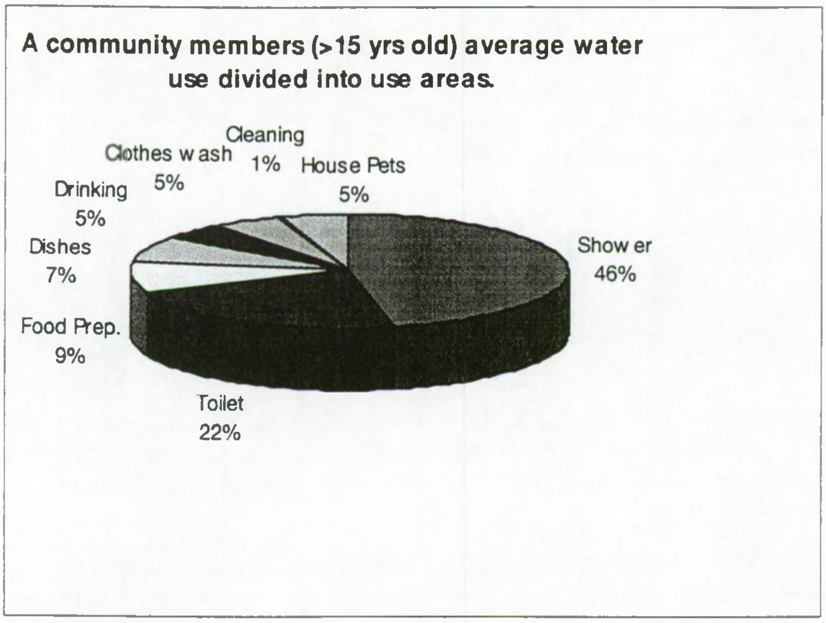


Fig 12: Daily Water Use people in villages on Unguja Island, Zanzibar.

In detail, this amount consists of 12.87 l (39%) for taking showers, 4.9 l (15%) for cooking, 4.62 l (14%) for toilet purposes 4.04 l (12%) for washing dishes, 2.59 l (8%) for washing clothes, 2.64 l (8%) for animals, 1.15 l (3%) for drinking, and 0.29 l (1%) for cleaning the house (children). For adults, water use amounted to 25.21 l for taking showers (46%), 11.92 l (22%) for toilet purposes, 4.9 l (9%) for cooking, 4.04 l (7%) for washing dishes, 2.64 l (5%) for feeding the animals, 2.59 l (5%) for drinking, 2.59 l (5%) for washing clothes, and 0.29 l (1%) for cleaning the house.

In total, about 35,000 people live in the coastal zone investigated during this survey. Also, 44.5% of the island's population belong to the age group of 0-14 years (10). Applying this percentage to the East Coast shows daily water demands of approximately 1,059,440 l (adults) and 521,140 l (children). In total, 1,580,580 l are used per day, or 45.16 l per capita. This sum includes both well and piped water.

Public water demand (mosques, schools, etc.) was not included into calculations. This might account for a 10% surplus. The overall water demand on the East Coast of Zanzibar is therefore in the order of 1,738,638 l d^{-1} (Fig.5).

Sewage- villages

Pit latrines are only common in a few villages. In general, there are more latrines when the soil is sandy, and less when it is rocky. Rocky grounds make it a major investment to built a pit latrine, with costs ranging between TSh 30,000-60,000 (US$ 50-100). There is no village with more than 75% of houses having pit latrines, and on average it might be only 30%. Where pit latrines are less common, residents frequently use the beaches for toilet purposes, and showers are taken inside a fenced area that belongs to the house. Clothes are washed inside this area as well or brought to the next well or tap. Water from washing dishes is thrown outside for religious believes. Overall, a major proportion of the sewage is soaked into the ground. As wells are often located close to houses, the amount of nutrients leaching into the groundwater and/or into the ocean could therefore be substantial.

Future Development- villages

In the future, population will grow significantly. As is presented in the attached manuscripts, there are number of significant managerial issues that will face future generations on Zanzibar (Annex II and III).

The 'medium growth rate decline' scenario (9) indicates a population doubling time of about 25-28 years. In the past, shortages of arable land on the East Coast have led to migration from the coral rag to the deep soil areas (3, 9), but there is evidence that tourism will reverse this trend and lead to additional population growth. Moreover, the number of partly huge private houses along the coastline has risen dramatically in the 90s.

This development, possibly in combination with a higher per capita demand of water, will increase water withdrawal substantially.

Assuming an average population growth rate of 2.5% until 2015 (to a total of 53,000), and adding a surplus of 20% for public uses, migration, private houses, and rising per capita demand, the total amount of water used on the East Coast will be in the order of about 2,872,200 $l\ d^{-1}$ by 2015 (scenario Business as Usual, Fig. 5).

A new phenomenon: Tourism

Since 1985, tourist arrivals in Zanzibar have grown consistently at around 20% per year, and this trend is expected to continue (11, 12). In 1997, international tourist arrivals accounted for 86,495 (12) (Fig. 8). Out of a 156 hotels and guesthouses on the island, 58 with about 2,256 beds are located in the study area.

Guesthouses are mainly found in the villages, and water is abstracted from local or private wells. Guesthouses in Nungwi, where water is imported by truck are an exception. Hotels have often resort character and are built outside the villages on uninhabited stretches of beach. To meet water demands, local caves or wells have been developed. Withdrawal happens without any form of overall planning or control. Water is mostly pumped all day around from a single well or cave at rates of up to 80 $l\ s^{-1}$.

Occupancy rates of hotels and guesthouses vary significantly. While guesthouses are bound to the tourist season, most hotels have signed contracts with tour operators in Europe, allowing for high annual occupancy rates of up to 80%. In general, high season is in July and August with a second peak in December.

The average high seasonal occupancy rate for all guesthouses and hotels on the East Coast was calculated at 73%, and the daily number of tourists staying there at 1,874.

Per capita water use by tourists varies extremely, ranging between 125-2,000 $l\ d^{-1}$. Averages were calculated by dividing the total amount of water used in a guesthouse or hotel by the number of tourists. The use by guesthouse owners, etc. were excluded this way, as data on this issue was basically not available.

The overall weighted average water use of a tourist on the East Coast was calculated at 674.5 $l\ d^{-1}$. This is 15 times the average demand of a local resident (Fig. 13).

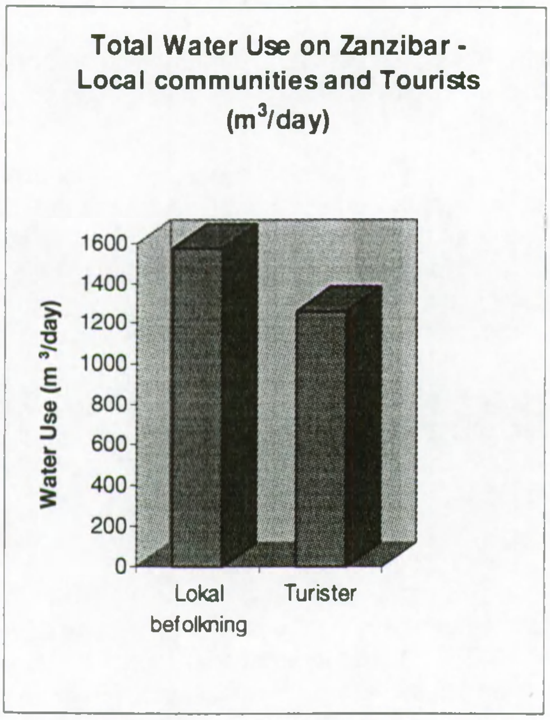


Fig.13: Water use by local residents and tourists

The total water demand of all tourists on the East Coast was estimated at 1,264,000 l d^{-1}. 826,000 l (65%) of this amount are on account of the 8 biggest hotels, hosting 55% of the guests (1,055). Except for one hotel, which is desalinating ocean water, all hotels abstract their water from the local aquifers.

Purposes of use are difficult to quantify and change both in total and as a percentage with hotel categories. High class hotels with expansive gardens use most water for continuous irrigation (up to 80%), because soils have a poor storage capacity, evaporation is high, planted species are not naturally growing on the East Coast, and have a high water demand. In comparison, small guesthouses use little or no water at all for irrigation purposes. Swimming-pools consume large amounts of water, both due to high evaporation and water renewal. Moreover, additional showers are taken frequently at pools, and extra towels handed out to guests. Pools can account for up to 30% of the total water demand of a hotel.

Cleaning and washing are other important uses. Together, they might cause up to 20% of the total water demand. Restaurants use 5-10% of the water. All values given vary significantly between hotels and guesthouses.

Water used by the tourists themselves for taking showers, flushing toilets and other sanitary purposes constitutes a rather small proportion of water used in high-class hotels (10-25%), but can account for up to 80% in small guesthouses.

Sewage-hotels

All hotels and guesthouses are expected to have closed concreted septic tanks. However, 9 of 22 hotels and guesthouses interviewed on this question stated that they had open tanks leaking into fissures and caves. One hotel pumped sewage into a cave, and another one directed its effluent into a former well. This way, a major proportion of sewage is possibly washed out into the sea.

Future Development-hotels

Zanzibar has built up an extensive tourist infrastructure in the last decade. In the future, this trend is expected to continue: In 2015, 550,000 tourist arrivals are expected to account for 3.8 million bed nights (See Annex II and III).

At present, 2,556 beds exist on the East Coast, but this number is planned to be extended to a total of 8,060 beds in 2015 (11). Until then, the average length of stay on the island is expected to rise from 2.5 days (1993) to 7 days.

Calculated at the average high seasonal occupancy rate found in this survey (73%), 5,884 tourists would stay on the East Coast per day, using a total amount of 3,968,758 l of water (674.5 per capita). This Business as Usual scenario (BAU) has to be considered as rather conservative, as the previous calculation was based on a mixture of hotels and guesthouses, while new buildings will be almost entirely hotel resorts. As described above, these have a higher average demand. Pumping and piping the water to the hotels entails losses, which were calculated in the order of 30% (4). These are not included here. Moreover, the government intends to increase the standard of all guesthouses and hotels. Presently, 65% do not fullfil the prerequisites of the Tourist Commission. As requirements include for example private bathrooms, per capita water demand might increase.

Discussion

It has been shown that groundwater abstraction on the East Coast of Zanzibar is substantial, and has possibly led to problems like vertical and lateral salt water intrusion, and the release of nutrients into the groundwater and the ocean. In turn, this has caused a low and decreasing water quality in many wells, and increasing dependence on piped water. Community problems related to this development are:

- Long walking distances to the remaining freshwater wells (especially in Nungwi).
- Hard working conditions (carrying water, which is foremost done by women).
- Economic losses due to time spent for fetching water.
- Illnesses (entailing economic losses).

Further consequences could be:

- Lowering of the water table, which might be a potential threat to rice plantations further inland, and the Jozani Forest.
- Negative impacts on ecosystems due to effluent leaching into the ocean.
- Negative effects on the tourist industry.

Future development will worsen the situation: In 2015, population on the East Coast will have grown to a minimum of 53,000, not including migration. Private houses, and increased per capita demand will possibly contribute to enforced water abstraction. The tourist industry is already contributing to withdrawal, and will by 2015 become the major user. Moreover, per capita demand in tourism might increase, as the government intends to establish a high-class tourism on Zanzibar. Finally, water demand is not equally high throughout the year. Most tourists visit the island during dry season when precipitation is lowest (Fig. 11), and both tourists' and locals' per capita use is higher in this period. This causes the highest demands in the driest season.

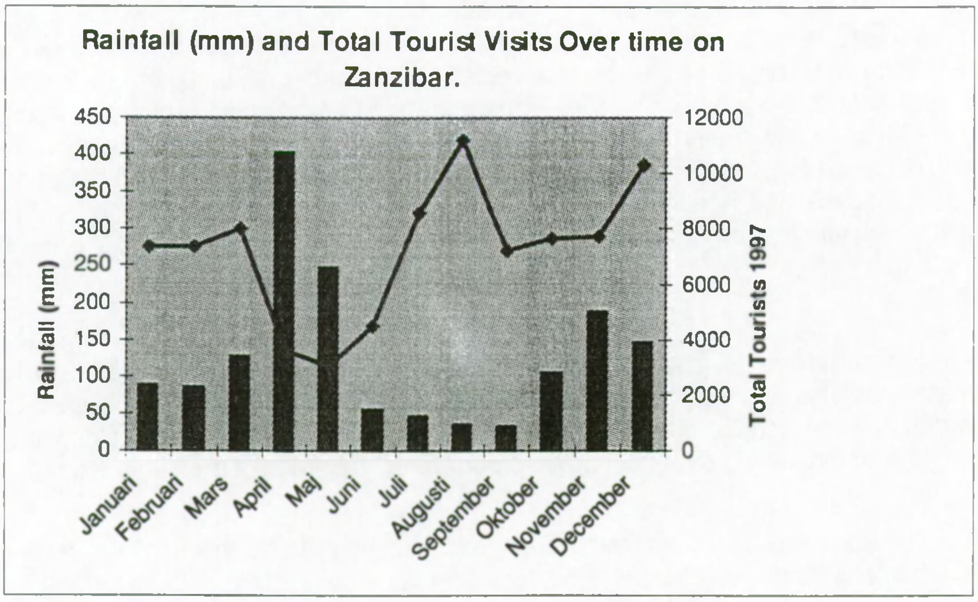


Fig 14: Precipitation and tourist arrivals per month in 1997

An additional risk factor is the El Niño phenomenon, which can severely influence rainfall, as has been shown in 1998, when March and April were extremely dry.

Therefore, the main problem connected to water abstraction is to estimate the amount of water that can be withdrawn without causing intrusion or upconing. Basically, water resources on the island as a whole are sufficient to meet the anticipated demands of the population and the tourist industry over the next 20 years, but current abstractions are concentrated in a few areas and on single places (4). A number of hotels have already reported slightly salty water, even though in some cases being abstracted from caves 1-4 kms away (inland). This might be a first sign of upconing. Basically, the karstic nature of the coral rag makes it virtually impossible to predict with confidence the safe yield from any one particular source (4).

As water is most scarce when the highest amounts are needed, an important strategy is to reduce water demand during this period. For the local population, present consumption patterns are considered to be close to minimum standards, and could possibly be reduced by no more than 20%. Assuming that water scarcity will make this a necessity, water demand may be reduced to 36.13 l d^{-1} per capita by 2015, or a total of 1,914,890 l d^{-1}. Allowing for a 20% surplus (public uses, migration, etc.), the overall amount used on the East Coast would then amount to 2,297,868 l d^{-1}. Note that this sum represents a minimum, and is considered less realistic.

As stated above, most villages receive piped water. According to the Department of Water (personal message), the amount pumped to the coastal zone is by far higher than the water demand calculated above. This leads to the assumption that leakages and losses are massive, which was confirmed by own observations, and commentaries of local inhabitants. Continuous control and repair of the pipes is therefore a primary measure.

Hotels and guesthouses should be able to reduce their consumption by far. It was estimated (4) that resorts should manage to keep the average daily consumption down to the equivalent of 200 litres per day per bed by close control, a figure that is reached by many guesthouses. This would help to reduce the total amount used by the hotel industry to an equivalent of 1,176,800 l per day in high season (Fig. 10, scenario Management). This would help to reduce pressure on local aquifers, and to improve the situation.

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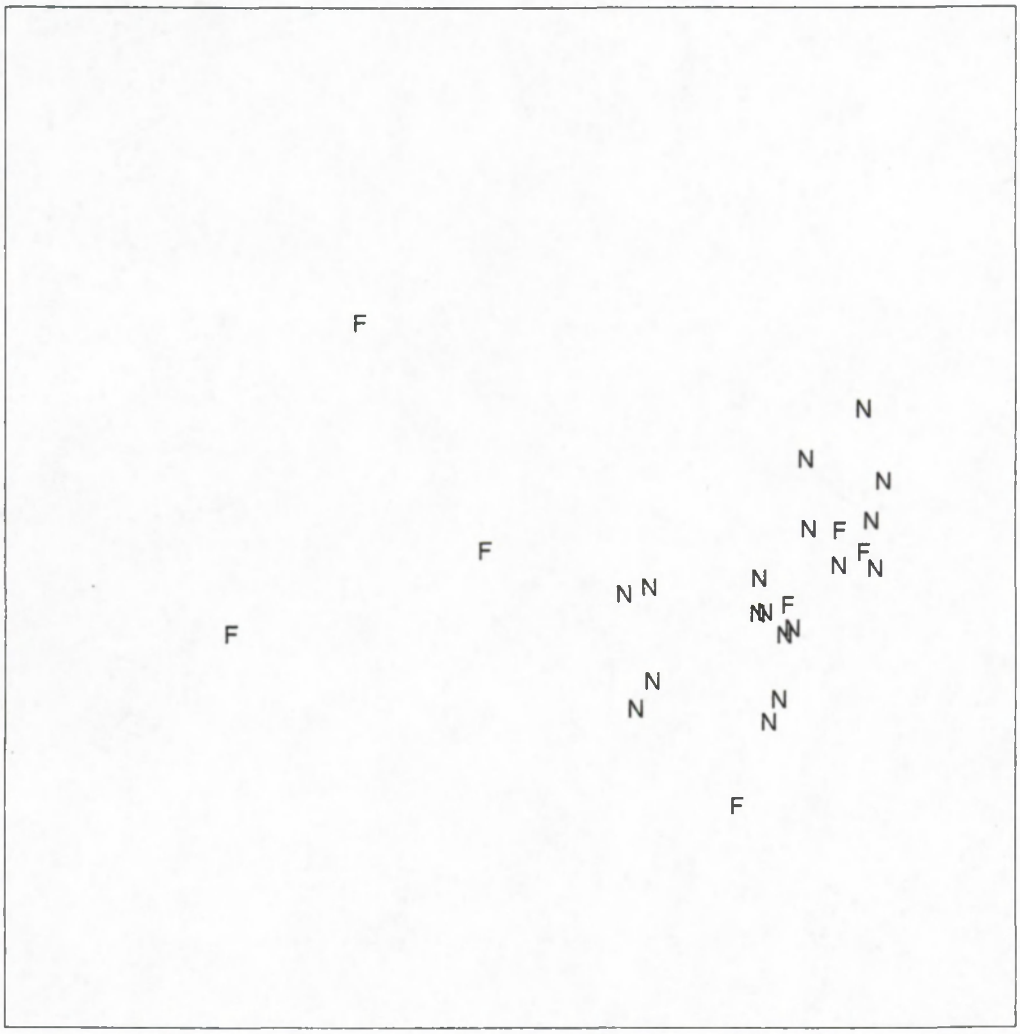
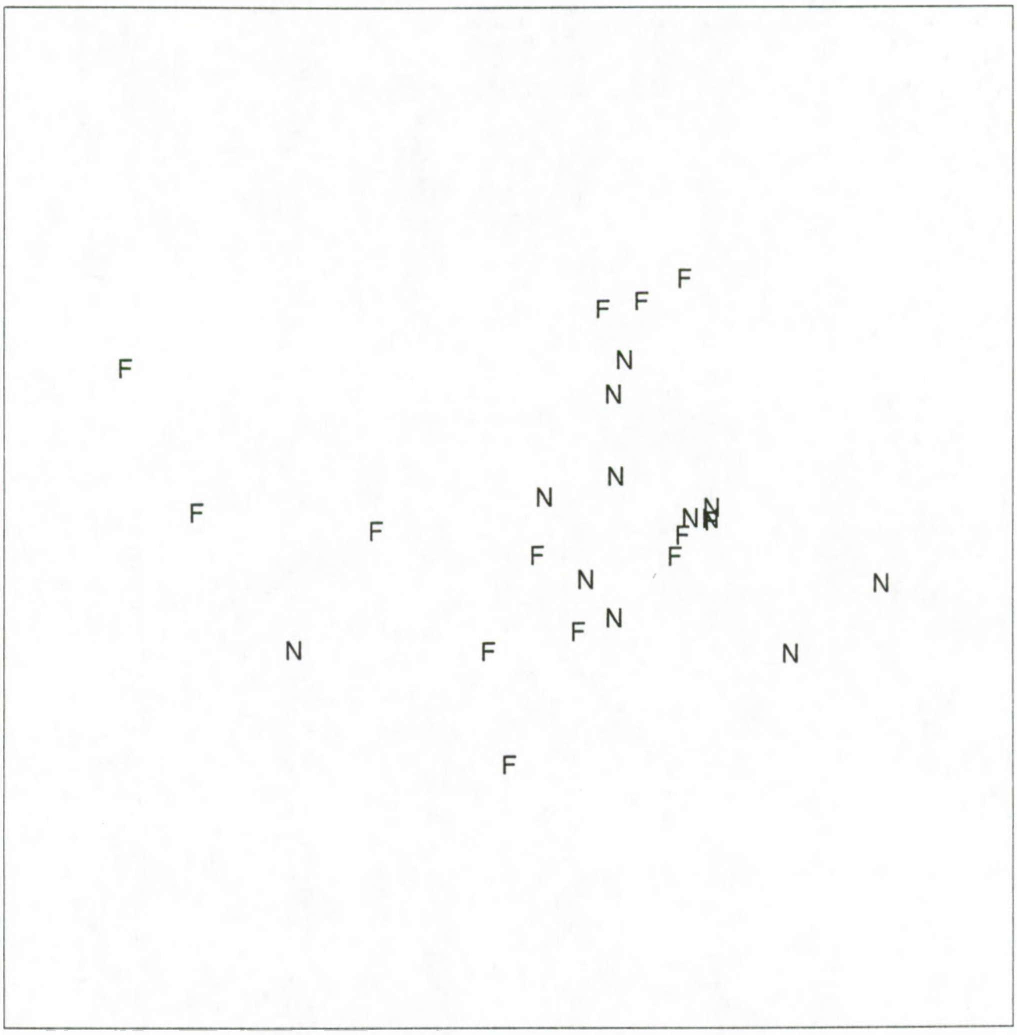
Appendix 1

Multi-dimensional scaling ordination (MDS) for Meiofauna Data.

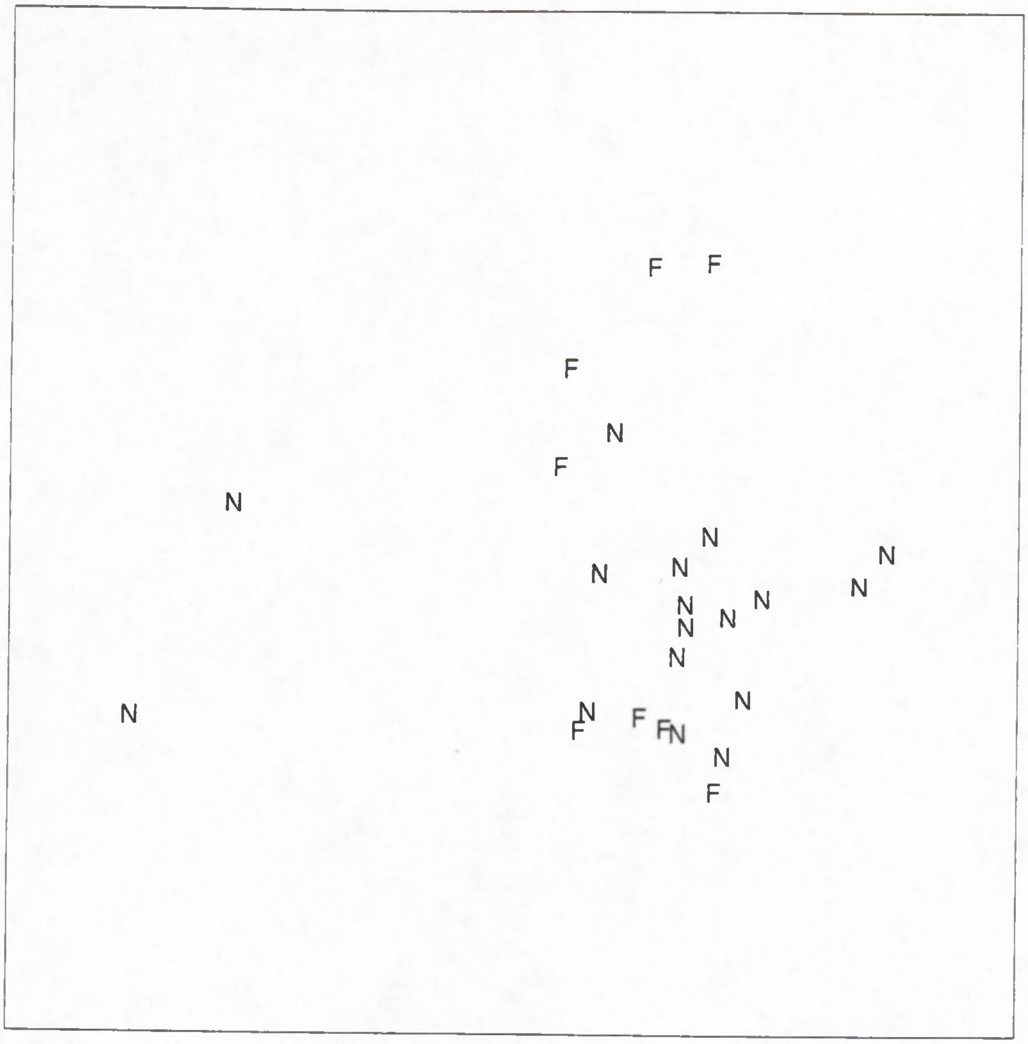
Meiofauna, Sep 1997, Stress = 0.06



Meiofauna, May 1998, Stress = 0.05



Meiofauna, June 1997, Stress = 0.04



University of Lisbon
Guia Marine Laboratory

General introduction

Inhaca island is particularly difficult to study in what concerns groundwater outflow and its effects on nearshore ecosystems. The dimensions of the island and its sandy constitution do not allow the formation of clear water bodies. However, the model of groundwater flows developed (see somewhere else in this report) seem to correspond to the heterogeneity pattern of the island. In fact mangrove and freshwater swamps correspond to maximum flow areas in the model.

Mangroves, being one of the most conspicuous cases of freshwater source dependence, were the first approach to study the groundwater outflow effects. The philosophy of the approach was to define contrasting areas of inflow and outflow, both derived from the early model results, and also by biological evidence. This evidence was based on the major development of *Avicennia marina*, supposed to reach bigger dimensions in outflow areas. An attempt was made to apply the ABC curve method, developed to study community dysfunction in pollution studies. The development of the model of flows throughout the programme has proved some of the early site definitions as inadequate, up to a certain degree, for the original purposes. The full results of the studies relating to mangroves were presented in the 2nd year report, thus here we include an abstract.

Other productive nearshore areas at Inhaca are seagrass meadows, and several dominant associations occur in extensive intertidal and shallow subtidal areas. Included are large areas of populations of *Thallassodendron ciliatum*, supposed to tolerate lower salinity, and being a key species for other components of Groflo project. In this context was decided to expand the studies to the seagrass meadows, having has primary goal the detection of evidence of groundwater outflow, even if diffuse, and to assess its effects on the nutrients levels of pore water and biological parameters at different levels. The approach was a nested design with decreasing scales of variability, for a set of different seagrass associations. Full results are presented in this report.

The workload of these studies was large, and results were obtained in full just prior to the writing of the present report, thus publication during the project period was not possible. Another aspect of the seagrass studies was the deeper cross involvement of both IMAR (University of Lisbon) and UEM (University Eduardo Mondlane), where all field work and laboratory analysis were performed jointly. The part of this report concerning seagrass meadows thus also constitutes part of report of UEM – Mozambique.

Groundwater effects on the macrobenthic communities at Saco mangrove

(summary of the second annual progress report)

The main objectives of the work carried out on Saco mangrove at Inhaca Island were to determine the groundwater effects on the macrobenthos community, the study of its composition (at family level), abundance and the respective biomass. The localization of the sites was performed with intent to separate sampling areas with different groundwater influences, three with groundwater inflow and other three with groundwater outflow. The selection of sampling sites was made according to field observations, as Avicenia size, and also the groundwater modellisation map.

The results obtained showed that on dry season the temperature varied on a 6ºC range, recording a minimum of 18ºC for S6A and S6B, and a maximum of 24ºC for S1A. On the sampling done for wet season we recorded a range of 12ºC, with a minimum of 26ºC for S5B and S8A and a maximum of 38ºC for S6B.

The recorded salinity values present a minimum of 35‰ and a maximum of 44‰, so a range of 9‰ on dry season. During wet season the minimum and the maximum recorded was 26‰ and 45‰ respectively (a range of 19‰).

In relation to the sediment analysis, the most abundant classes are medium and fine sands respectively with $1 < \Phi < 2$ ($0{,}5 > \varnothing > 0{,}25$mm) and $2 < \Phi < 3$ ($0{,}25 > \varnothing > 0{,}125$mm) grain sizes. The percentages of organic matter content varies between 0,3% (S4B) and 1.2% (S6B).

According to density and biomass values obtained during dry and wet seasons the most representative taxonomic groups were selected and presented below. The maximum value recorded for each group is stressed in bold.

Dry season

Samples	% Density	% Biomass
S1	**Gastropoda (94 %)** Decapoda Rep. (3 %) Bivalvia (1 %)	**Bivalvia (65 %)** Gastropoda (20%) Decapoda Rep. (12 %)
S3	Gastropoda (86 %) **Decapoda Rep. (12 %)** Polychaeta (1 %)	**Decapoda Rep. (74 %)** **Nemerta (15 %)** Gastropoda (7 %)
S4	Gastropoda (91 %) Decapoda Rep. (7 %) Polychaeta (2 %)	Decapoda Rep. (60 %) **Polychaeta (33 %)** Gastropoda (5 %)
S5	Gastropoda (90 %) Polychaeta (5 %) Decapoda Rep. (2,5 %)	Decapoda Rep. (42 %) **Gastropoda (29 %)** Polychaeta (25 %)
S6	Gastropoda (74 %) **Polychaeta (12 %)** **Bivalvia (7 %)**	Bivalvia (41 %) Decapoda Rep. (35 %) Decapoda Nat. (9 %)
S8	**Gastropoda (94 %)** Polychaeta (4 %) Decapoda Rep. (0,5 %)	**Decapoda Nat. (41 %)** Decapoda Rep. (27 %) Gastropoda (19 %)

Wet season

Samples	% Density	% Biomass
S1	Gastropoda (66 %) **Decapoda Rep. (27 %)** Bivalvia (4 %)	**Bivalvia (67 %)** Decapoda Rep. (25 %) Gastropoda (4 %)
S3	**Gastropoda (92 %)** Decapoda Rep. (5 %) Polychaeta (2 %)	Decapoda Rep. (61 %) Gastropoda (25 %) Polychaeta (12 %)
S4	Gastropoda (57 %) **Polychaeta (35%)** Bivalvia (3 %)	**Polychaeta (40 %)** **Gastropoda (38 %)** Bivalvia (16 %)
S5	Gastropoda (79 %) Decapoda Rep. (11 %) Bivalvia (7 %)	**Decapoda Rep. (65 %)** Polychaeta (19 %) Bivalvia (8 %)
S6	Gastropoda (56 %) Polychaeta (17 %) **Bivalvia (15 %)**	**Bivalvia (67 %)** Decapoda Rep. (17 %) Gastropoda (8 %)
S8	Gastropoda (83 %) Decapoda Rep. (7 %) Polychaeta (5 %)	Decapoda Rep. (40 %) **Nemerta (33 %)** Gastropoda (17 %)

The ABC curves (Abundance/Biomass Comparison) done with the data from dry season showed high levels of ecological disturbance, even greater than obtained on wet season. We considered that ecological disturbance presented by communities sampled on different seasons on Saco mangrove were not related with a anthropogenic pollution factors. The natural disturbance was result of high stress due to low oxygen, high salinity and temperature levels mainly on dry season. Biological interactions has an important influence and should be take into account.

In the hierarchical analysis, the characterisation of the stations was based in the following parameters: biomass total/station, density total/station, temperature, salinity, organic matter and granulometry of the sediment. The stations were grouped as function of exposition degree to the groundwater or not, showing the importance of this factor concerning the community structure.

In this analysis, involving the six stations sampled on dry season and the six sampled on wet season, were obtained two distinct clusters. One cluster includes samples 3, 4, 5 and 8 (WET) and the other includes samples 3, 4 and 5 (DRY) indicating that in these stations the season of sampling was an important factor for their characterisation. Sample DRY_8 presents a greater linkage distance than the rest of the group, probably due to its superior biomass and density values recorded on this season, resulting of the increasing number and size of the sampled organisms.

In regards to stations 1 and 6, it was shown less linkage distance between WET_1 and DRY_1, and between WET_6 and DRY_6 respectively. The results suggest that on these two cases the intrinsic parameters of the stations are more relevant than the season in which sampling was made.

As a conclusion, with the integration of results obtained by different analysis and also by the observation *in situ*, we consider that ecological disturbance presented by the different communities sampled on dry and wet season on Saco mangrove were not related with a anthropogenic pollution factors. The natural disturbance was result of high stress due to low oxygen, high salinity and temperature levels mainly on dry season. Biological interactions has an important influence and should be take into account. The degree of disturbance detected on these communities seems to be not related to the level of groundwater influence in the studied stations.

Standing crop of seagrass and associated infaunal communities at Inhaca Island, Mozambique

José Paula & Pedro Fidalgo e Costa
Guia Marine Laboratory,Cascais, Portugal
Angelina Martins & Domingos Gove
Eduardo Mondlane University, Maputo, Mozambique

Introduction

The distribution of seagrasses at Inhaca island was studied by Bandeira (1995), and further considered by Kalk (1997). There are 9 species of seagrasses at Inhaca island: *Thalassodendron ciliatum, Thalassia hemprichii, Halodule uninervis, H. wrightii, Cymodocea rotundata, C. serrulata, Halophila ovalis, Syringodium isoetifolium* and *Zostera capensis*. However, a few form associations that constitute extensive meadows around the island. On the west coast there is a dense population of *T. ciliatum*, which dominates an association that also includes *C. serrulata* and patches of *H. ovalis*. This meadow starts close to spring low water, and continue subtidally, and distributes north-south acompanying the development of banks parallel to the coast. The intertidal part is the main area for oyster catching (*Pinctada capensis*), and the meadow presents a high diversity of conspicuous associated fauna, such as the sea stars of the genera *Pentaceraster* and *Protoreaster*. Also fish and crustaceans are exploited on these banks, which shows the importance of the meadow for a variety of resources during the high tide period. The meadow never dries up although exposed at spring low tides, and a layer of water of 5 to 10 cm high is always present above the sediment. The dead leaves of *T. ciliatum* induce high bacterial activity and consequent nutrient recycling, and constitute the main detritus feeding the trophic chains of the western shores of the island.

On the southern bay (see Fig. 1), the flat sides of the channel between Ponta Torres and Saco mangrove are covered with a seagrass association dominated by *Thalassia hemprichii* and *Halodule wrightii*. This association is also characterised by a number of conspicuous faunal elements as fan shells (genera *Pinna* and *Atrina*), a variety of sea urchins (*Diadema* spp, *Salmacis bicolor*, etc), synaptid holothrians (*Synapta maculata, Opheodesoma serpentina*), among others. This flat keeps a layer of water of around 5 cm at low tides, and the sediment never dries out. At the lower border an extensive growth of solitary corals occurs, and commensal gobids nest throughout the flat. This type of meadow also occurs in great part of the northern bay.

The third main association of seagrasses occur on the island southern banks, and is dominated by the southern species *Zostera capensis*. The populations of this seagrass are a consistent coverage but are more patchy than the previous two seagrass associations. The sediment retains less water, and the major part of the flat remains dry at low tide, or with a very thin layer of water (1-2 cm) that gets very hot and saline during warm periods. In deeper pools (around 10 cm deep) can be observed dense populations of *Halodule wrightii* and *Thalassia hemprichii* where *Holothuria* spp are very abundant. Also common in these flats is the giant anemone *Stichodactyla haddoni*.

Objectives

The main objectives of this work were to assess the biomass of seagrasses and respective associated infaunal communities at a seasonal basis and in various scales, and relating it to substrate parameters as grain size, phytopigments, organic matter, salinity and nutrient contents. The hypothesis is that biological parameters reflect the magnitude of groundwater outflow, through evidence from pore water salinity and nutrient load.

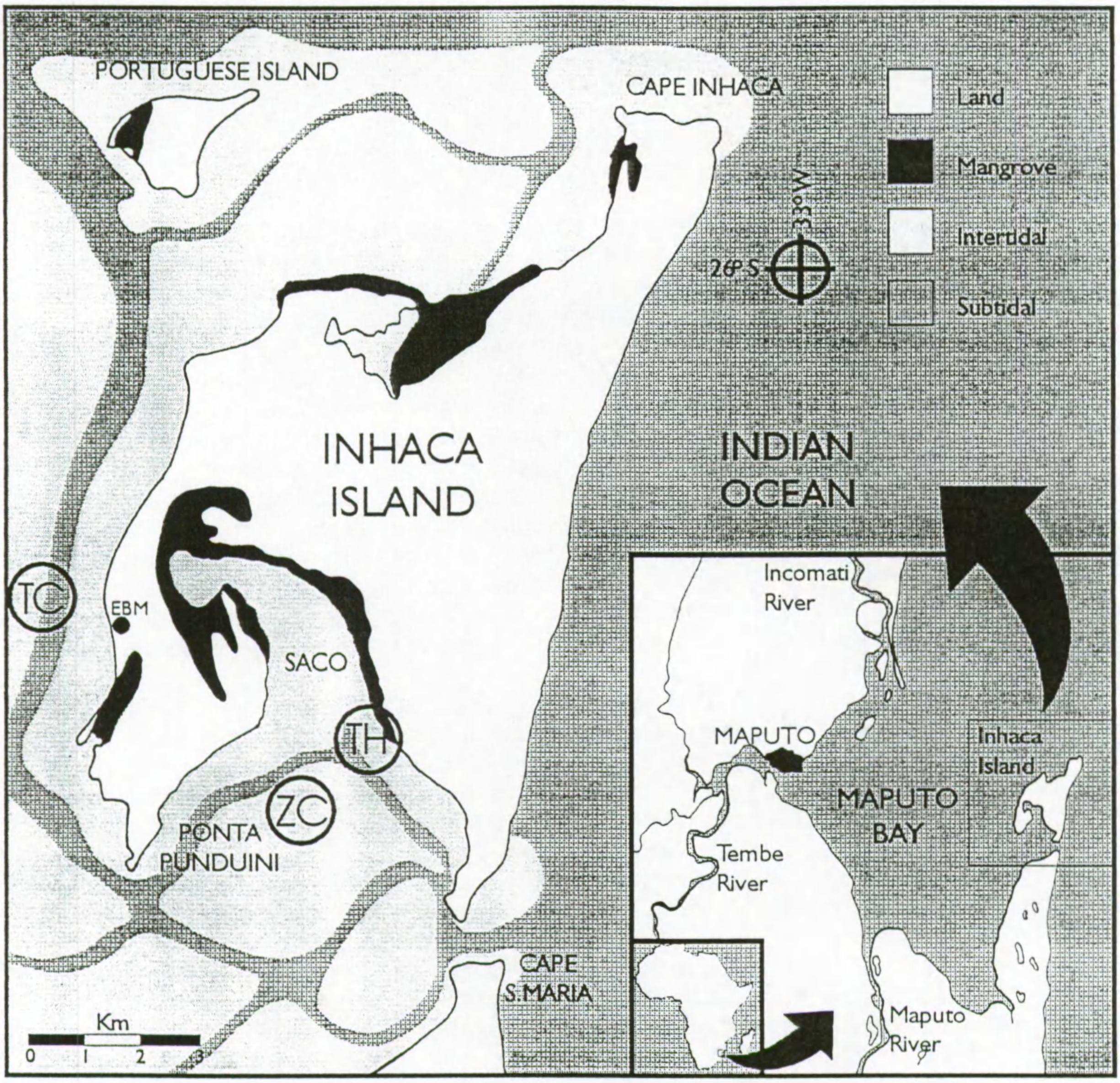


Fig. 1 - Map of Inhaca island showing location of the sampled seagrasses. ZC - *Zostera capensis*, TC - *Thalassodendron ciliatum*, TH - *Thalassia hemprichii*.

Material and methods

Study sites

Three seagrass beds were chosen due to their extension around Inhaca island, and for constituting main areas for local fisheries activities with high productivity (see Introduction and Fig. 1). On the west coast a meadow dominated by *Thalassodendron ciliatum* was chosen (TC). On the southern bay the *Thalassia hemprichii/Halodule wrightii* association was sampled (TH), as well as the *Zostera capensis* meadows on the south banks (ZC). The importance of each seagrass species in the respective association is presented in the results section.

Sampling strategy

Strategy for sampling procedure followed a nested design with different levels and scales of variability, and focused on the referred 3 different seagrass associations (see Fig. 2). First level is fixed and refers to the season. The seagrass meadows constitute also an orthogonal level, and sites were chosen according to banks of major intertidal abundance. All other levels were chosen randomly, according to decreasing spatial scales, namely 2 stations (100-500 m apart), 2 samples (10-50 m apart) and 2 replicates (1-5 m apart). Independent sampling was performed, except for water parameters which were taken from the same water sample per replicate. Each sampling station was sampled on a different day, at low tide, thus the entire sampling for each season took 6 consecutive days of field work, around a spring tidal period. Stations were assigned randomly to sampling days.

Sampling methods

Studied parameters are summarised in Table I, and included physico-chemical parameters of the water column and pore water, and sediment biological and physical parameters.

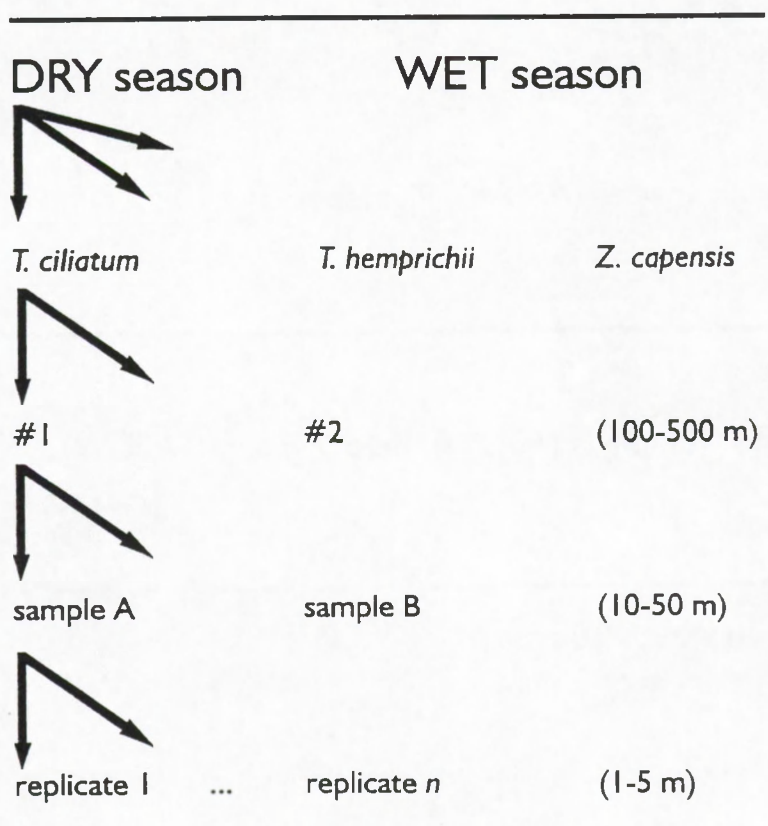


Fig. 2 - Diagrammatic view of global nested sampling strategy.

Table I - Summary of sampled parameters at Inhaca island

Water column

upper layer or tide pools when only available

Parameters:
Temperature, Salinity, pH, Nutrients (ammonia, nitrites, nitrates)

Pore water

extracted from sediment down to 20 cm

Parameters:
Temperature, Salinity, pH, Nutrients (ammonia, nitrites, nitrates)

Sediment

extracted from sediment down to 20 cm

Parameters:
Grain size structure
1st centimeter - Organic matter (AFDW), Pigments

Seagrasses

Species composition

Above ground (AFDW), Epiphyts (AFDW), Below ground (AFDW)

Associated fauna

Meiofauna (AFDW) - down to 10 cm
Macrofauna (AFDW) - down to 25 cm

Pore water was extracted from a block of sediment down to 25 cm, using a manual vacuum pump connected to a Millipore filter. Each replicate water sample was 200ml. Another sample from surface water was taken, in the case of presence of a thin water layer over the bank. If no water was covering the flat, samples were taken from the nearest available tide pools. Using a Ysi30M salinometer and a pH probe, immediate register of temperature, salinity and pH was made. Samples were then preserved cold until reaching the laboratory.

Organic matter and phytopigments were sampled in the superficial centimetre with cores of 6 cm in diameter. These were immediately kept in the dark and frozen. Meiofauna used cores of the same diameter and to a depth of 10 cm. Samples were sieved through a mesh with 0.5 mm to extract macrofauna and then through a mesh with 0.063 mm and fixed with 4% formalin. For seagrass and macrofauna independent cores of 15 cm in diameter were used up to a depth of 25 cm. Samples were sieved in place with a pore of 0.5 mm and fixed with 4% formalin. A special sample of the sediment was also taken for granulometric analysis.

Laboratory analysis

In the laboratory water samples were immediately analysed for nutrients, namely ammonia, nitrites and nitrates, using a Hach 2010 spectrophotometer. Procedures followed strictly the respective protocols. For granulometric determination Udden-Wentworth scale was used (Kramer *et al.*, 1994), and separation of mud (>62μ), sand (62μ-2000μ) and gravel (>2000μ) was done by wet sieving method (Buller and McManus, 1979). of following the scale Organic matter content of the sediment was obtained by loss of ignition (450ºC/24h). Phytopigments (chlorophyll a and phaeopigments) were determined after sediment water content estimation (%H^2O) by 24hr drying at 65ºC. Evaluation of phytopigments concentrations (g/g dry sediment) were obtained by spectrophotometry after 24hr, cool extraction in 90% acetone and calculated by modified Lorenzen equations (Plante-Cuny, 1974).

Epibiose attached to seagrasses was removed and dry weighted (AFDW). Seagrass species were sorted out from the samples and decomposed on above-ground and below-ground fractions, and dry weighted (AFDW). Meiofauna and macrofauna were extracted from sediment using a saturated saccharose solution. Remaining sediment was also checked with the aid of a binocular microscope. Biomass was determined by loss of ignition (AFDW).

To calculate ash-free dry-weight (AFDW), all samples were dried at 65ºC to constant weight (48 or 36hr), after which the samples were placed in a desiccator and allowed to cool. The samples were weighed, incinerated at 450ºC in a muffle furnace for 24h. After cooling, the residue was stored in a desiccator. After weighing, ash-free dry-weight (AFDW) was calculated by subtraction (Crisp, 1984).

Data treatment

Nested ANOVA was performed for all parameters for the different hierarchic levels. Season and seagrass meadows were treated as orthogonal fixed factors, and lower levels as random factors. Pearson correlation was used to correlate different biotic and abiotic parameters with pore water salinity and between all variables.

Results

Temperature

Temperature patterns varied between both sampled seasons (see Fig. 3). During dry season temperature was more uniform (ranging approximately from 23 to 26 ºC) , with no significant differences between seagrass meadows or even when comparing WC with PW values (minimum at *T. hemprichii* with 23.0 ºC and maximum at *T. ciliatum* with 26.7 ºC). It was however, and as expected, consistently slightly lower in pore water when compared with the water column. This pattern can be explained by the good and stable climatic conditions during the season sampling period, with no rain, no major cloud coverage or excess heat.

During wet season temperature was less stable with major differences between pore water and water column, and also between the different considered seagrass meadows (ranging approximately from 24 to 33 ºC). In fact these differences can be attributed solely to punctual climatic conditions of sampling days, and not true meadow differences (maximum at *Z. capensis* with 33.1 ºC, and minimum at *T. ciliatum* with 24.6 ºC). Weather is not very stable in this period of the year, and we could observe a direct relation between conditions (as insulation for instance) and gradual increase of temperature over the flats. Pore water temperature was consistently lower, from 1 to 3 ºC in relation to water column values.

Salinity

Salinity has been the major factor providing evidence of groundwater outflow. As can be seen in Fig. 4, the global pattern is opposite in relation to the temperature pattern. In fact, during wet season the salinity was more stable, and more homogeneous between water column and pore water (range approximately between 32 and 36 ‰ during dry season, and between 35 and 37 ‰ during wet season).

In pore water during dry season, the lowest values were registered at *Thalassodendron ciliatum* meadow (minimum 28.6 ‰) and maximum values at *Zostera capensis* meadow (maximum 35.5 ‰).

pH

pH registered values seem to support also the trends showed by salinity (Fig. 5). Values are consistently lower at dry season, when compared with wet period (range between 7.50 and 7.78 during dry season, and between 7.71 and 9.21 during wet season). Also, as expected, pore water values are much lower than water column, and this is very marked during wet season.

Nutrients

Water column

Studied nutrients showed higher values consistently during dry period. Again we have evidence that freshwater influence is maximal during the initial part of dry season. This is particularly evident for ammonia and nitrates. Nitrites have shown a higher degree of variability, although means reflect the common trend. Figs. 6 to 8 present results from respectively ammonia, nitrites and nitrates.

During dry season ammonia ranged from 0.09 mg l^{-1} at *T. ciliatum* and 0.21 mg l^{-1} at *T. hemprichii*, and during wet season ranged between 0 (obtained at all sampled meadows) and 0.06 mg l^{-1} at *Z. capensis* meadow. During dry season nitrites ranged from 0.012 mg l^{-1} at *T. ciliatum* and 0.200 mg l^{-1} also at *T. ciliatum*, and during wet season ranged between 0.002 at *T. hemprichii* and 0.076 mg l^{-1} also at *T. hemprichii* meadow. During dry season nitrates ranged from 3.1 mg l^{-1} at *T. ciliatum* and 4.4 mg l^{-1} at *T. hemprichii*, and during wet season ranged between 1.0 at *T. hemprichii* and 2.5 mg l^{-1} at *Z. capensis* meadow.

Pore water

Pore water nutrient concentration, as expected, has shown higher values when compared to water column. The exception is again nitrite concentration, which during wet season has no significant difference between pore water and water column. Figs. 6 to 8 present results from respectively ammonia, nitrites and nitrates.

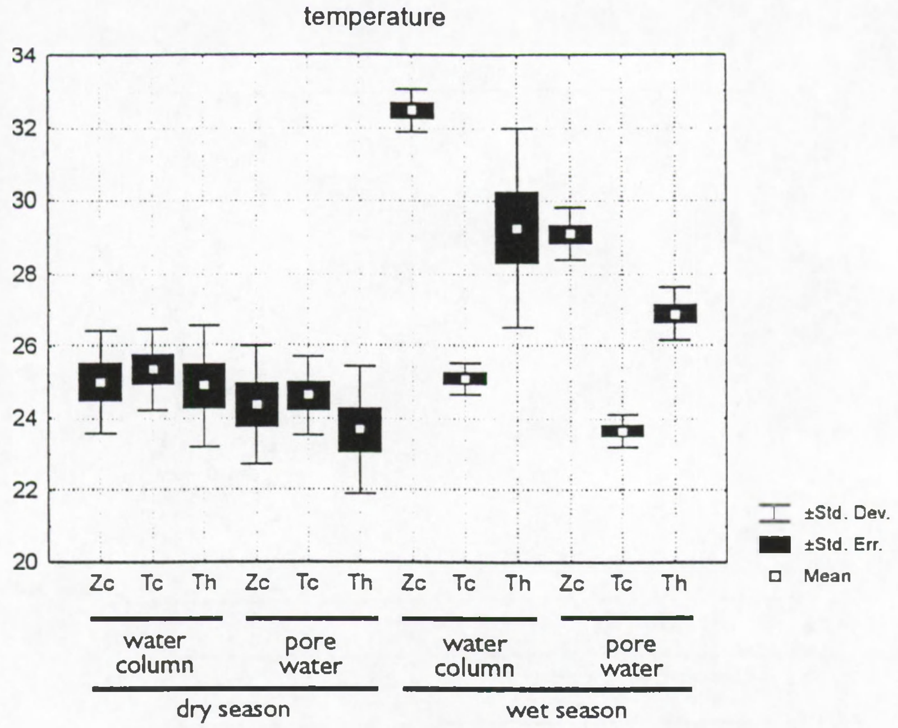


Fig. 3 - Temperature of water column and pore water at different seagrass associations, during both seasons. Zc - *Zostera capensis*, Tc - *Thalassodendron ciliatum*, Th - *Thalassia hemprichii*.

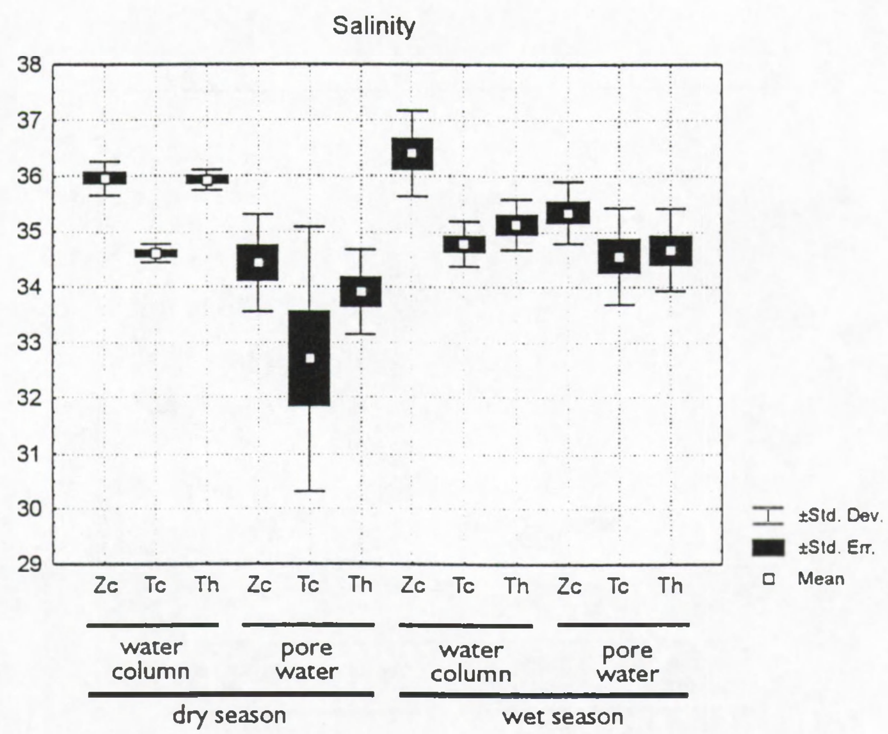


Fig. 4 - Salinity (ppt) of water column and pore water at different seagrass associations, during both seasons. Zc - *Zostera capensis*, Tc - *Thalassodendron ciliatum*, Th - *Thalassia hemprichii*.

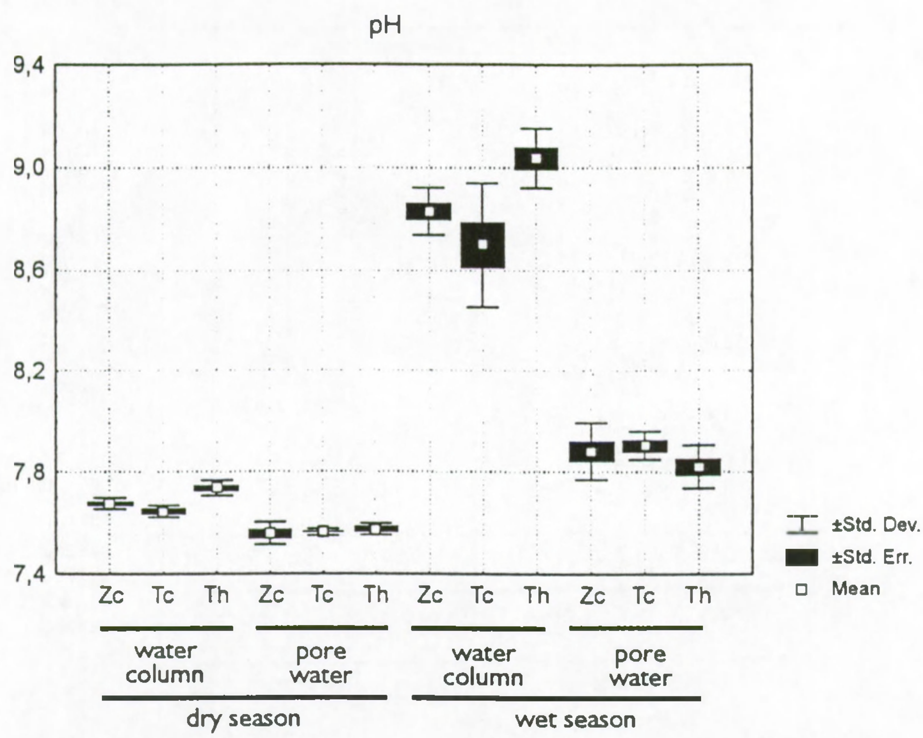


Fig. 5 - pH of water column and pore water at different seagrass associations, during both seasons. Zc - *Zostera capensis*, Tc - *Thalassodendron ciliatum*, Th - *Thalassia hemprichii*.

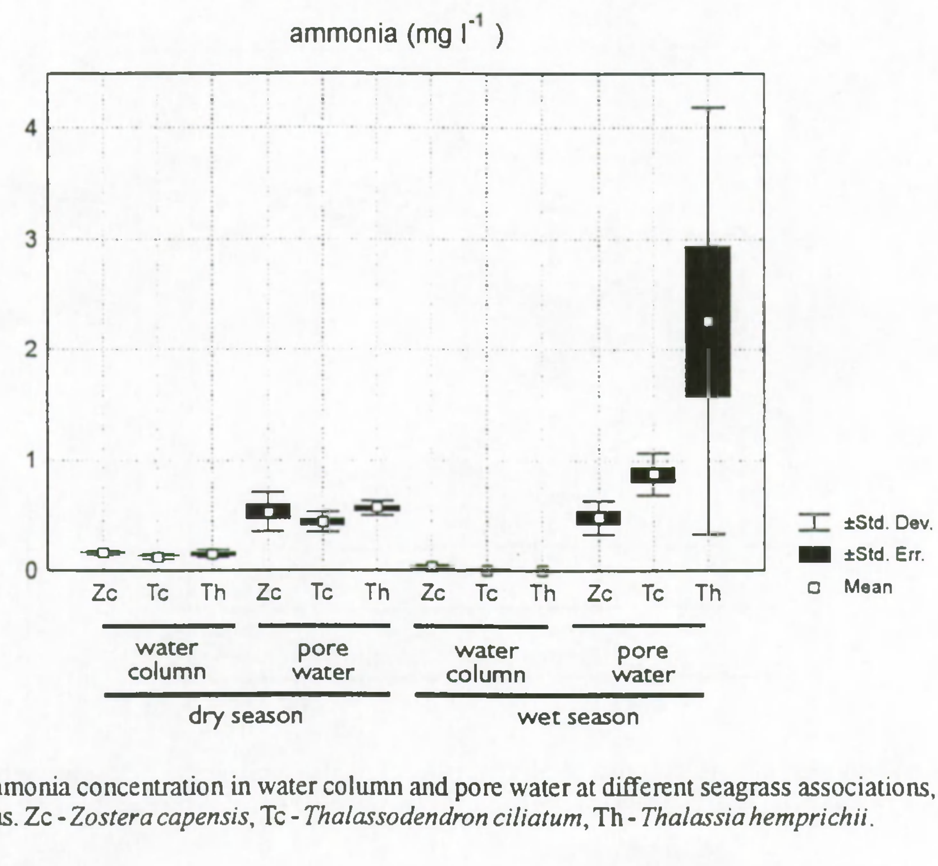


Fig. 6 - Ammonia concentration in water column and pore water at different seagrass associations, during both seasons. Zc - *Zostera capensis*, Tc - *Thalassodendron ciliatum*, Th - *Thalassia hemprichii*.

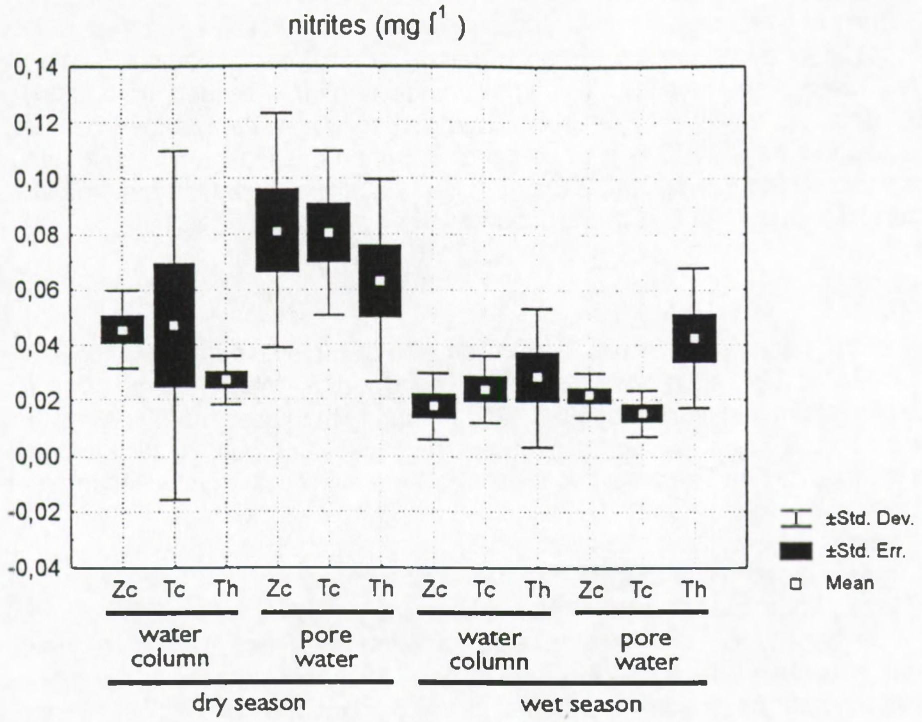


Fig. 7 - Nitrite concentration of water column and pore water at different seagrass associations, during both seasons. Zc - *Zostera capensis*, Tc - *Thalassodendron ciliatum*, Th - *Thalassia hemprichii*.

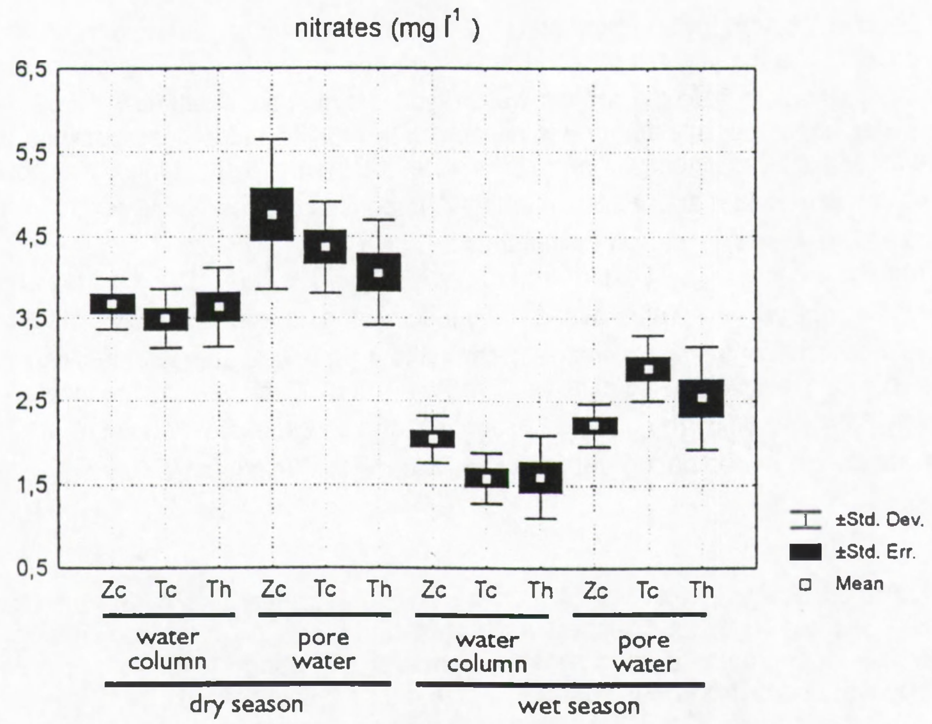


Fig. 8 - Nitrate concentration of water column and pore water at different seagrass associations, during both seasons. Zc - *Zostera capensis*, Tc - *Thalassodendron ciliatum*, Th - *Thalassia hemprichii*.

During dry season ammonia ranged from 0.32 mg l^{-1} at *T. ciliatum* and 0.84 mg l^{-1} at *Z. capensis*, and during wet season ranged between 0.40 at *Z. capensis* and *T. hemprichii* and 5.00 mg l^{-1} at *T. hemprichii* meadow. During dry season nitrites ranged from 0.044 mg l^{-1} at *T. hemprichii* and 0.121 mg l^{-1} at *Z. capensis*, and during wet season ranged between 0.000 at *T. ciliatum* and 0.074 mg l^{-1} at *T. hemprichii* meadow. During dry season nitrates ranged from 3.2 mg l^{-1} at *T. hemprichii* and 6.6 mg l^{-1} at *Z. capensis*, and during wet season ranged between 1.7 at *T. hemprichii* and 3.8 mg l^{-1} at *T. ciliatum* meadow.

Granulometry

Table II presents the results of granulometric distribution from sampling station during dry and wet season. The grain size distribution analysed at Inhaca Island showed mainly the presence of medium and fine sand (0.5-0.125mm) during both seasons. *Thalassodendron ciliatum* meadow presented higher fraction of fine sand but also a coarser fraction (>2mm), composed specially by coral debris. The other two meadows were slightly different in what regards grain size distribution.

Organic matter

Organic matter (Fig. 9) in the 1st sediment centimetre, as other parameters in this thin sediment layer, is more affected by water column than by pore water. Globally, mean values of OM were higher during dry season, also showing a higher degree of variability (range approximately between 4 and 14 % during dry season, and between 3 and 8 % during wet season). At *Z. capensis* meadow are present the higher and the lower values. During wet season, range of OM in the sediment was narrower, with higher values at the *T. ciliatum* meadow. At *T. hemprichii* meadow higher variability occurred. The maximum value was obtained at *Z. capensis* meadow during dry season (16.0 %), and the minimum value at *T. hemprichii* during wet season (2.3 %).

Pigments

Sediment chlorophyll a contents (Fig. 10) has shown a different trend in relation to phaeopigments. The former presented more uniform average values during dry season (ranging approximately from 5 to 8 $\mu g\ g^{-1}$), than at wet season (ranging approximately from 3 to 11 $\mu g\ g^{-1}$). Variability was high, specially during wet season, with marked differences between the different seagrass meadows. Consistently lower values were obtained at *T. hemprichii* meadow sites. The highest values were found at one station in the *Z. capensis* meadow (maximum 18.6 $\mu g\ g^{-1}$), and the lowest at *T. hemprichii* meadow (minimum 1.2 $\mu g\ g^{-1}$).

Phaeopigments (Fig. 11), as referred, have shown a different pattern in relation to Chl a. In fact, values were more variable during dry season (ranging approximately from 3 to 6 $\mu g\ g^{-1}$), than at wet season (ranging approximately from 1.5 to 4 $\mu g\ g^{-1}$). Average values were consistently higher at dry season, with maximum at *T. ciliatum* and *T. hemprichii* meadows (maxima of respectively 6.64 and 6.95 $\mu g\ g^{-1}$), and minimum at the *Z. capensis* meadow (2.75 $\mu g\ g^{-1}$). The lower values were obtained during wet season at Z. capensis (minimum 0.76 $\mu g\ g^{-1}$).

Seagrasses

Obtained biomass values for the different seagrass meadows during both seasons are summarised in Figs. 12 and 13 (respectively above and below ground standing crop). There no net differences during both sampled seasons, however at meadow level major differences were found. Above ground biomass was consistently lower than below ground.

Thalassodendron ciliatum above ground biomass ranged between 53.1 and 170.7 gr m^{-2} and below ground ranged between 0.04 and 1,471.1 gr m^{-2}. *Thalassia hemprichii* above ground biomass ranged between 41.2 and 291.1 gr m^{-2}, and below ground ranged between 9.21 and

Table II - Cumulative frequency of granulometric distribution from sampled stations, during dry and wet seasons.

Dry season													
	gran(øscale)	TC 1A	TC 1B	TC 2A	TC 2B	TH 1A	TH 1B	TH 2A	TH 2B	ZC 1A	ZC 1B	ZC 2A	ZC 2B
>2mm	< -3 ø	2.92	7.63	10.49	25.31	13.30	3.64	3.9	1.94	0.93	0.43	3.26	2.52
2-1mm	-2...-1 ø	4.37	5.47	10.82	12.90	5.42	5.19	4.2	5.96	1.2	2.94	5.9	4.12
1-0.5mm	-1 ...0 ø	4.69	6.75	9.13	11.41	12.28	12.60	11.9	12.18	12.69	8.24	8.75	7.59
0.5-0.25mm	0 ... 1 ø	7.80	9.70	10.71	12.87	29.83	33.41	25.48	25.71	31.29	27.92	25.84	26.2
0.25-0.125mm	1 ... 2 ø	75.5	65.46	50.74	32.75	37.34	43.93	52.76	52.41	51.29	56.44	53.52	56.83
0.125-0.063mm	2 ... 3 ø	4.17	4.29	6.41	3.54	1.05	0.79	1.12	1.282	2.3	2.75	2.07	2.22
<0.063mm	3 ... 4 ø	0.49	0.62	1.56	1.16	0.60	0.34	0.59	0.46	0.45	1.13	0.54	0.49
Wet season													
	gran(øscale)	TC 1A	TC 1B	TC 2A	TC 2B	TH 1A	TH 1B	TH 2A	TH 2B	ZC 1A	ZC 1B	ZC 2A	ZC 2B
>2mm	< -3 ø	2.10	3.35	14.03	19.48	2.77	5.91	2.26	2.15	1.80	7.68	2.48	0.00
2-1mm	-2...-1 ø	3.87	4.71	15.24	14.18	2.77	4.11	3.44	2.95	3.74	4.90	1.81	1.87
1-0.5mm	-1 ...0 ø	5.60	4.56	9.27	9.23	6.68	11.00	10.73	14.66	5.96	8.42	4.56	3.80
0.5-0.25mm	0 ... 1 ø	12.91	33.91	48.93	12.28	73.96	72.69	32.70	77.08	57.12	67.15	35.68	24.99
0.25-0.125mm	1 ... 2 ø	71.51	51.76	11.77	40.87	11.38	4.65	49.20	2.82	28.31	10.66	52.82	63.95
0.125-0.063mm	2 ... 3 ø	3.76	1.46	0.37	2.22	0.31	0.10	0.74	0.06	0.89	0.44	2.14	3.45
<0.063mm	3 ... 4 ø	0.58	0.60	0.10	0.34	0.01	0.03	0.17	0.00	0.26	0.14	0.34	0.42

Tc - *Thalassodendron ciliatum* , Th - *Thalassia hemprichii* , Zc - *Zostera capensis*

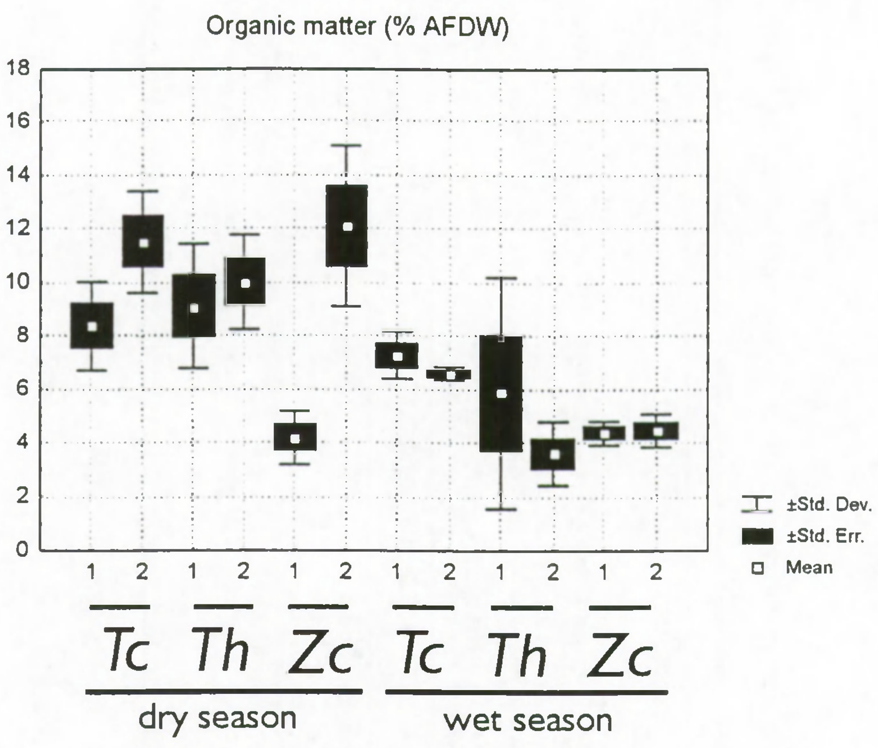


Fig. 9 - Organic matter in sediment (1st centimeter) at different collecting stations in the considered meadows, during both seasons. Zc - *Zostera capensis*, Tc - *Thalassodendron ciliatum*, Th - *Thalassia hemprichii*.

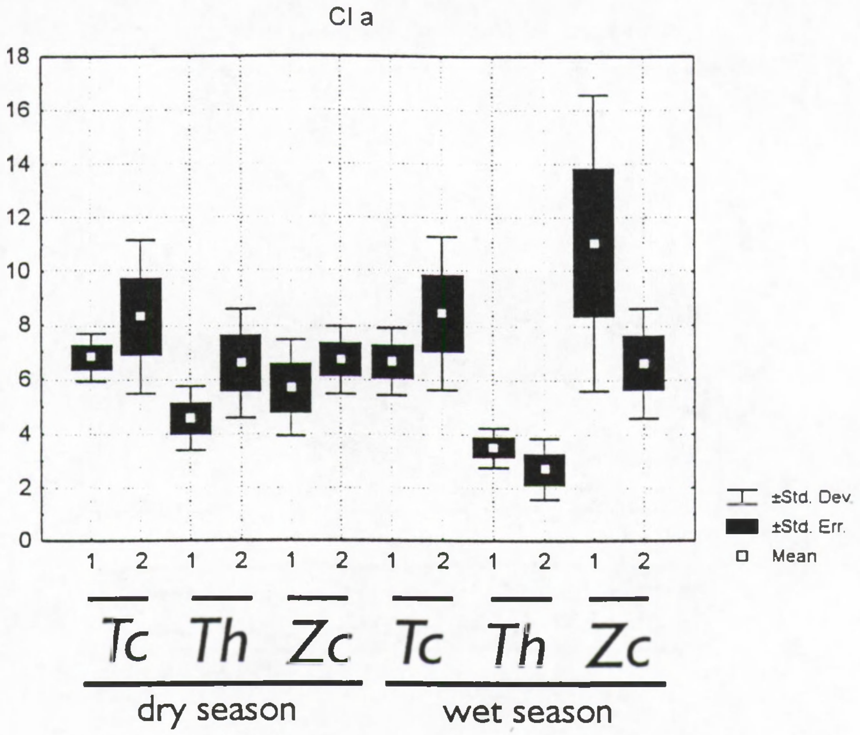


Fig. 10 - Chlorophyll a (sediment 1st centimeter, $\mu g\ g^{-1}$) at different collecting stations in the considered meadows, during both seasons. Zc - *Zostera capensis*, Tc - *Thalassodendron ciliatum*, Th - *Thalassia hemprichii*.

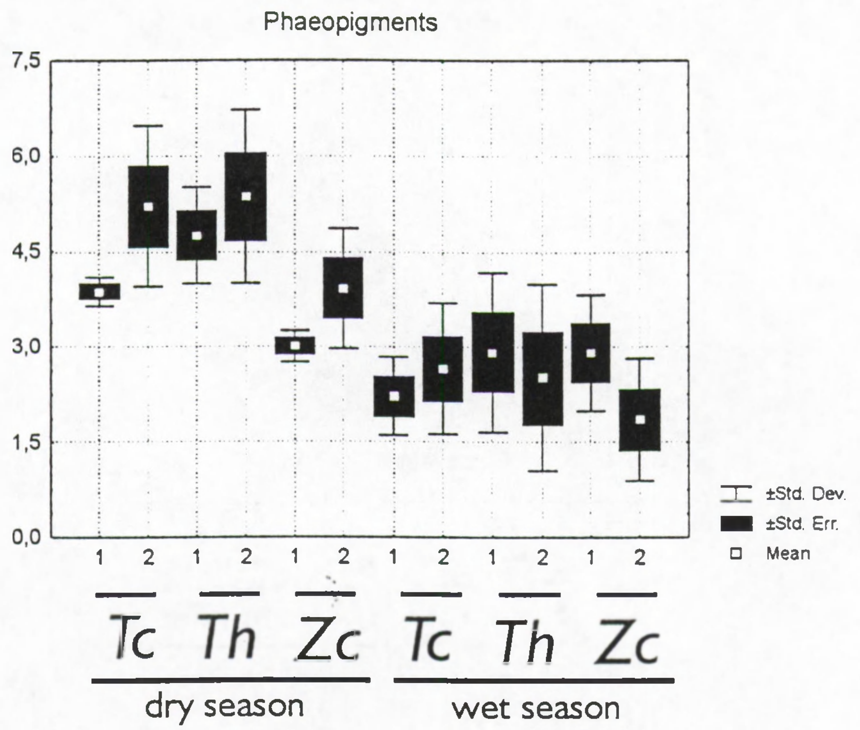


Fig. 11 - Phaeopigments (sediment 1st centimeter, $\mu g\ g^{-1}$) at different collecting stations in the considered meadows, during both seasons. Zc - *Zostera capensis*, Tc - *Thalassodendron ciliatum*, Th - *Thalassia hemprichii*.

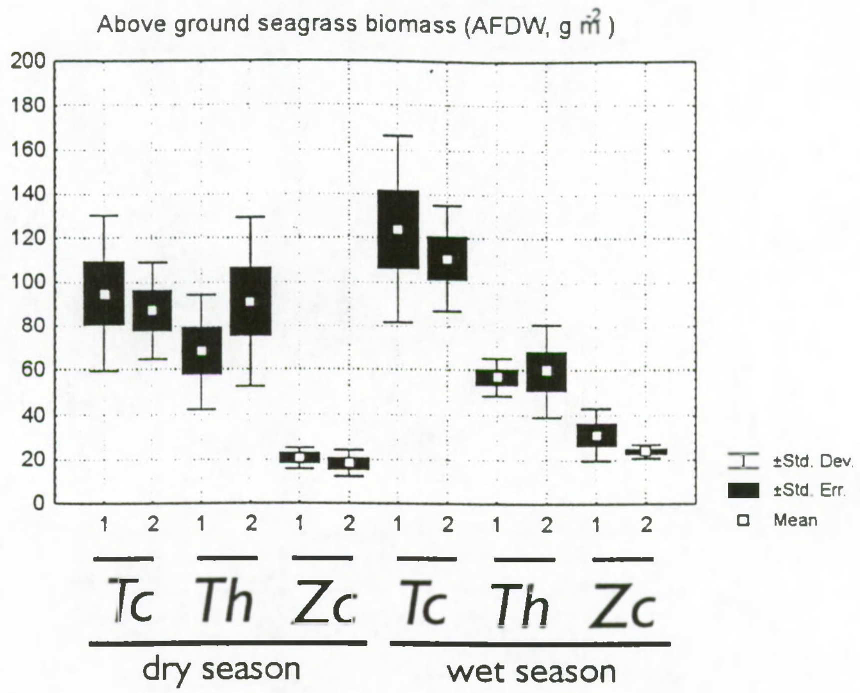


Fig. 12 - Total seagrass biomass (above ground) at different collecting stations in the considered meadows, during both seasons. Zc - *Zostera capensis*, Tc - *Thalassodendron ciliatum*, Th - *Thalassia hemprichii*.

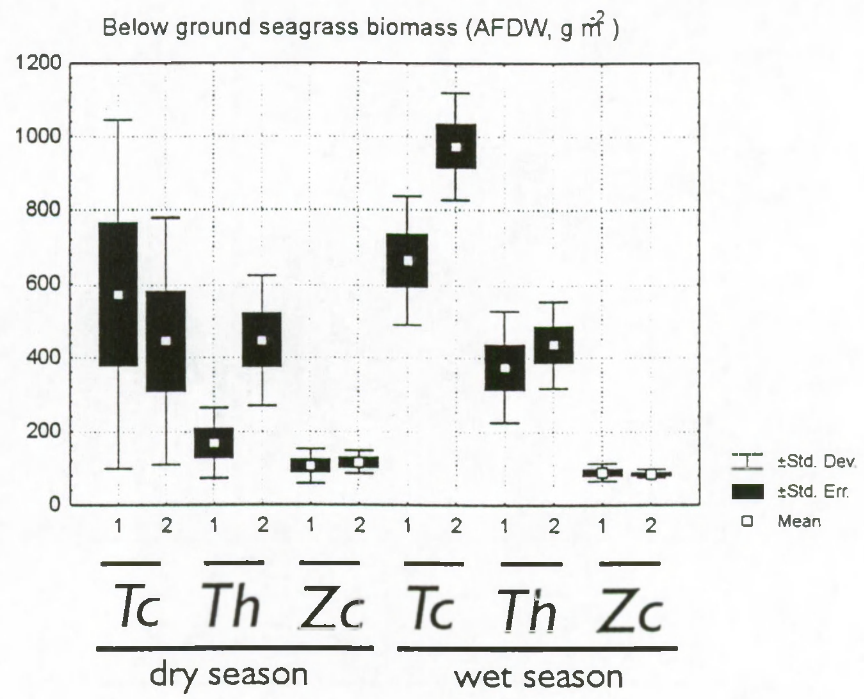


Fig. 13 - Total seagrass biomass (below ground) at different collecting stations in the considered meadows, during both seasons. Zc - *Zostera capensis*, Tc - *Thalassodendron ciliatum*, Th - *Thalassia hemprichii*.

1,307.6 gr m^{-2}. *Zostera capensis* above ground biomass ranged between 7.9 and 51.3 gr m^{-2}, and below ground ranged between 66.0 and 195.5 gr m^{-2}. In spite of this ranges, mean values have shown that higher biomass was obtained at *T. ciliatum* meadow, followed by *T. hemprichii*.

Figs. 14 and 15 present the biomass contribution of all sampled seagrasses. Seven species were sampled, however it can be seen that at most sampling stations the respective dominant seagrass contributes for a very high biomass proportion. The exceptions are during wet season, at *T. hemprichii* meadow station 1 showed a high biomass of *T. ciliatum* and *Halodule wrightii*, and at *Zostera capensis* meadow station 2 with a mixture of *H. wrightii*.

Epiphytic biomass

In what concerns epiphytic biomass on seagrasses, *Thalassodendron ciliatum* showed an epiphytic biomass ranging between 1.9 and 9.9 gr m^{-2}. In *Thalassia hemprichii* the biomass ranged between 0.0 and 0.4 gr m^{-2}, and in *Zostera capensis* biomass ranged between 0.0 and 1.4 gr m^{-2}. Mean values have shown that higher biomass was obtained at *T. ciliatum* meadow, the remaining meadows presented very low values (Fig. 16).

Meiofauna

Meiofauna biomass showed some fluctuation between seagrass meadows, stations and also between seasons (Fig.17). On the *T. ciliatum* meadow meiofauna had an improve in biomass on wet season but only in station 1 (ranged approximately between 4.21 and 53.20 g m^{-2} during dry season, and between 7.62 and 58.05 g m^{-2} during wet season). For *T. hemprichii* meadow, biomass presented during dry season a strong difference between stations with a high value on station 2 (mean value $\pm$ 110 g m^{-2}) due to a clear increase in density (nematodes and copepods) (ranged approximately between 5.29 and 184.15 g m^{-2} during dry season, and between 4.27 and 22.70 g m^{-2} during wet season). On the *Z. capensis* meadow both sampling station reached high values of biomass during dry season, when compared to wet season (ex. station 1 attained on dry season almost four times the biomass value) (ranged approximately between 10.55 and 56.68 g m^{-2} during dry season, and between 2.30 and 29.35 g m^{-2} during wet season.

Macrofauna

Macrofauna biomass showed marked fluctuations between seasons (Fig.18), being values more homogeneous during dry season and a very high variability during wet season. On the *T. ciliatum* meadow macrofauna biomass ranged approximately between 2.08 and 42.01 g m^{-2} during dry season, and between 6.30 and 208.87 g m^{-2} during wet season. For *T. hemprichii* meadow, biomass ranged approximately between 5.36 and 38.68 g m^{-2} during dry season, and between 10.77 and 243.41 g m^{-2} during wet season). On the *Z. capensis* meadow biomass ranged approximately between 1.08 and 18.27 g m^{-2} during dry season, and between 3.19 and 53.19 g m^{-2} during wet season.

Anova results

Global nested ANOVA results are presented in table III. Detailed results are presented in the tables X to XIII. Parameters have showed different behaviour towards the nested analysis, suggesting different spatial and temporal patterns and correlations.

At season level, some water column and pore water parameters present significant differences. In the water parameters (both water column and pore water), temperature, pH, ammonia and nitrates. In the sediment, significant differences were found for organic matter, phaeopigments, epiphyts and macrofauna.

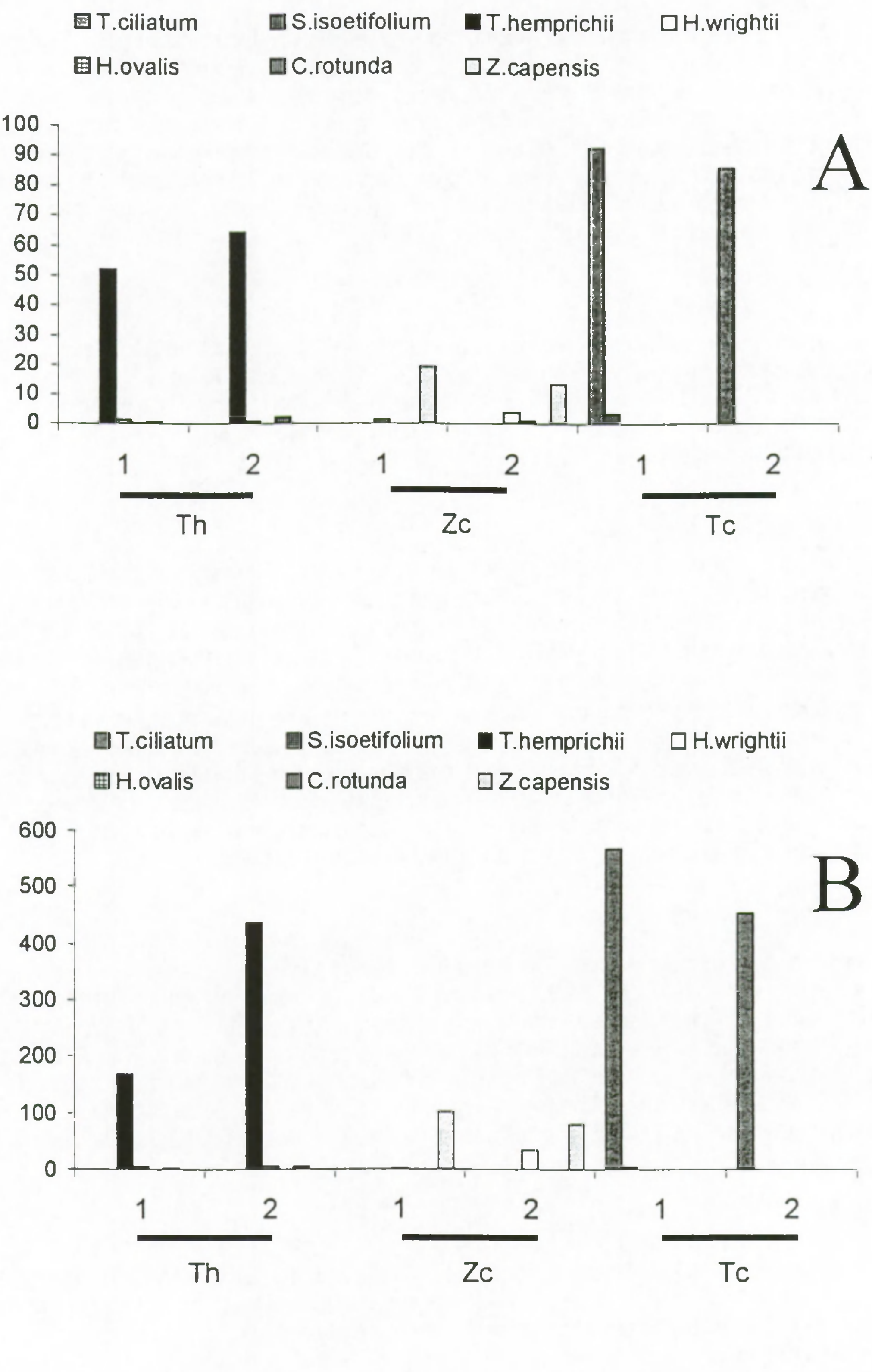


Fig. 14 – Biomass of different species of seagrass (AFDW g m^{-2}), for both above (A) and below (B) ground, during dry season.

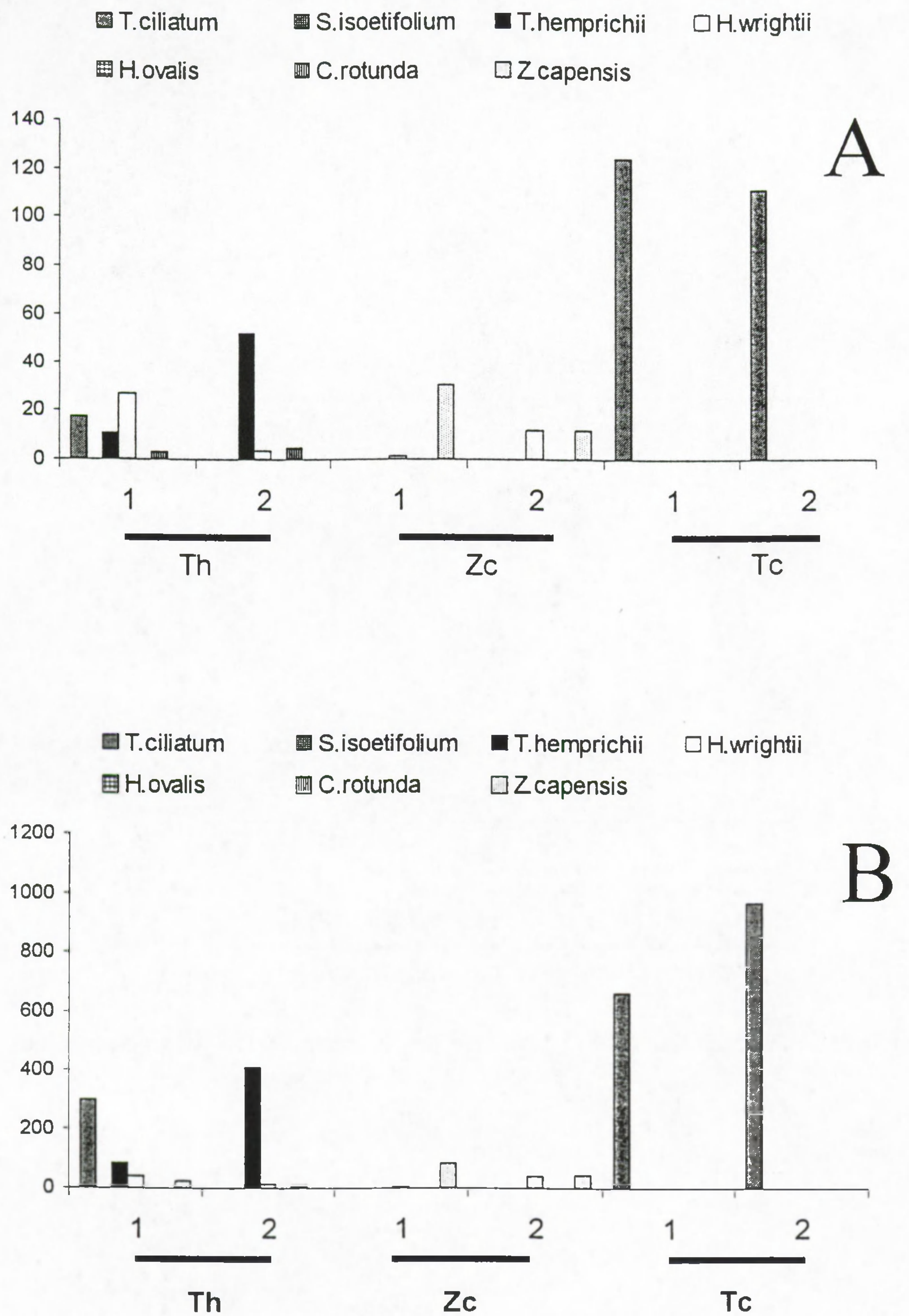


Fig. 15- Biomass of different species of seagrass (AFDW g m^{-2}), for both above (A) and below (B) ground, during wet season.

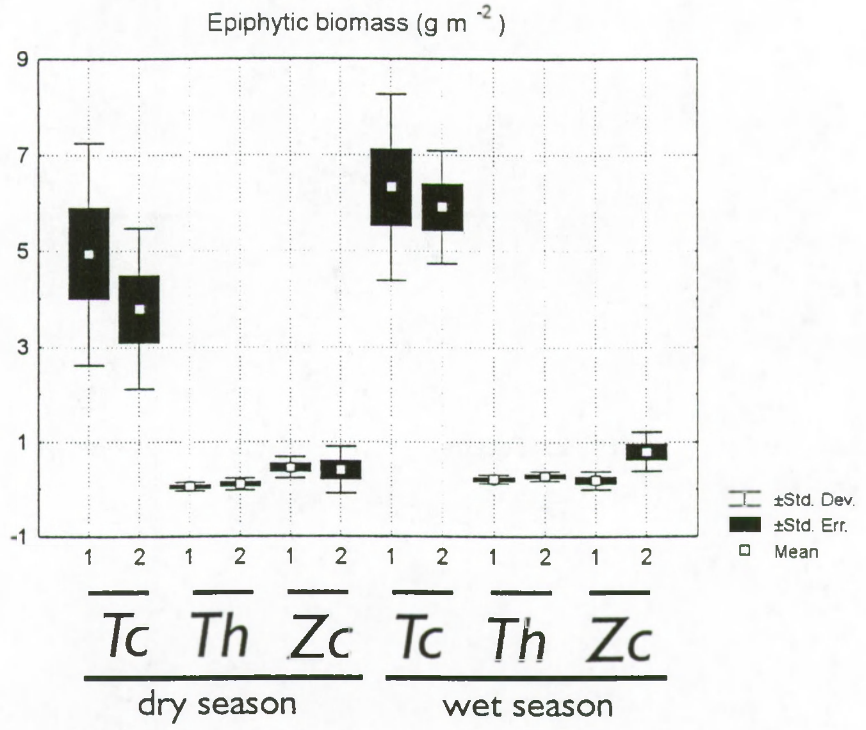


Fig. 16 - Epiphytic biomass at different collecting stations in the considered meadows, during both seasons. Zc - *Zostera capensis*, Tc - *Thalassodendron ciliatum*, Th - *Thalassia hemprichii*.

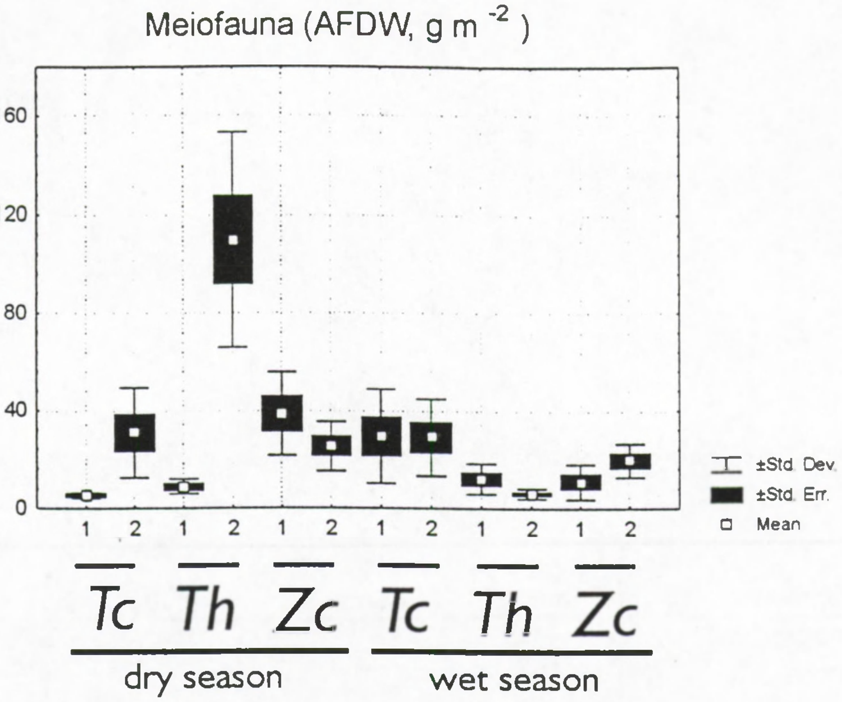


Fig. 17 - Total meiofaunal biomass (AFDW) at different collecting stations in the considered meadows, during both seasons. Zc - *Zostera capensis*, Tc - *Thalassodendron ciliatum*, Th - *Thalassia hemprichii*.

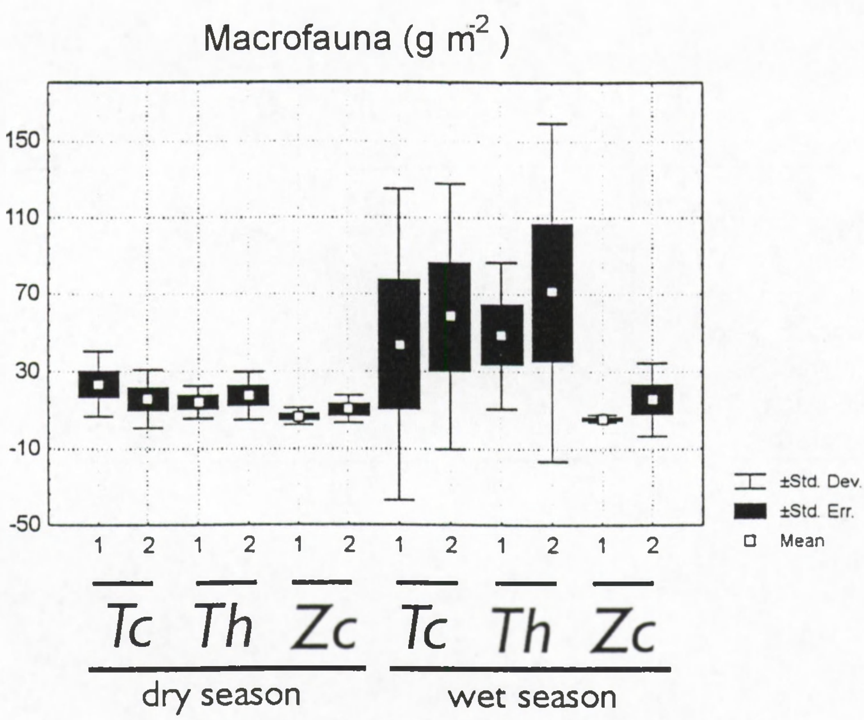


Fig. 18 - Total macrofaunal biomass (AFDW) at different collecting stations in the considered meadows, during both seasons. Zc - *Zostera capensis*, Tc - *Thalassodendron ciliatum*, Th - *Thalassia hemprichii*.

Table III - Global significance of nested ANOVA for all sampled parameters.

	Parameter	Season	Seagrass	Station	Sample	Season x Seagrass
Water column	Temperature	*	-	***	***	-
	Salinity	-	**	-	***	-
	pH	***	-	**	***	-
	Amonia	***	**	-	-	-
	Nitrites	-	-	-	-	-
	Nitrates	***		*		
Pore water	Temperature	*	-	***	**	-
	Salinity	-	-	***	-	-
	pH	***	-	-	***	-
	Amonia	-	-	***	-	-
	Nitrites	**	-	-	-	-
	Nitrates	***	-	-	-	-
Sediment	OM	*	-	*	*	-
	Chl a	-	*	-	*	-
	Phaeopigments	**	-	-	-	-
	Seagrass above	-	***	-	-	-
	Seagrass below	-	**	*	-	-
	Epiphyts	*	***	-	-	*
	Meiofauna	-	-	***	-	-
	Macrofauna	**	*	-	-	*

* $p<0.05$, ** $p<0.01$, *** $p<0.001$

At seagrass level, pore water did not present significant differences for any of the studied parameters. In the water column salinity and ammonia were significant. In the sediment Chl a, seagrass above and below, ephiphytic biomass and macrofauna were significantly different.

At station level, water column and pore water parameters presented significant differences. In the water parameters temperature, pH and nitrates were significant. In the pore water significant differences were found for temperature, salinity and ammonia. In the sediment, organic matter, seagrass below and meiofauna were significantly different.

At sample level, water column presented significant differences for temperature, salinity and pH. In the pore water parameters, temperature and pH were significant. In the sediment organic matter and Chl a were significantly different.

The only interaction effects between fixed factors (season x seagrass), were for epiphytic and macrofaunal biomasses.

Correlation between parameters

Correlation between pore water salinity and remaining parameters is presented in table IV. In relation to pore water salinity the only significant positively correlated parameter was pH (at $p<0.01$ level). All remaining significantly correlated parameters had negative coefficients, namely amonia, nitrates and phaeopigments (at $p<0.01$ level) and organic matter (at $p<0.001$ level). Table V shows correlations among all studied parameters.

Discussion

The seasonal effects are to be treated carefully, especially when trying to assign conditions to typical dry or wet seasons. The dry sampling period was performed during the month of July 1998, thus already within the dry period in the geographical area (May to October), but being in the initial phase some of the measured parameters may in fact reflect the cumulative character of the wet season. This is suggested for instance by the pore water salinity, which is generally lower at the "dry" season period. The opposite is also true, as wet season was performed in November/December, which may in fact correspond to the cumulative condition of dry season.

Water column parameters reflected markedly the influence of climacteric conditions on the different sampling dates. This is particularly evident in temperature and salinity values. In a narrow layer of water, as is the case of the water covering the seagrass meadows at low tide, atmospheric conditions as temperature, precipitation and insulation affect the water characteristics and its evolution along the tidal cycle.

Pore water parameters reflected mainly biogeochemical influences, with high concentrations of nutrients (NH_4+, NO_2- and NO_3-), when compared to the water column values. Although it has implications to both the productivity of seagrass meadows as well as being a pathway for exchange between bellow-ground pools and tidal waters. In what concerns salinity, values obtained on pore water in all meadows were lower than water column due to the fact that this surface layer suffers a greater influence from atmospheric conditions. Temperature presented almost the same values on dry season with few differences, specially on wet season due to weather conditions. pH values were always low on pore water both dry and wet season.

The higher values of salinity were obtained during wet season, reflecting the occurrence of "insulation windows". However, at the studied scale, no evidence of groundwater based on salinity can be viewed in this period. As referred above, the small catchment area of Inhaca island does not allow large water bodies to be formed. During the dry season period, salinity behaved more heterogeneously. Lower values, specially at the *T. ciliatum* meadow were registered when compared to wet season. Also a more pronounced difference was found between pore water and water column. This differences between dry and wet seasons can in fact reflect that "dry is wet" and "wet is dry", that is, the cumulative effects of one season are reflected in the period of following season.

Table IV - Correlation significance between pore water salinity and other parameters.

Parameter	r	p
pH	0.391	0.006045***
Ammonia	-0.382	0.007379**
Nitrites	-0.211	0.150051
Nitrates	-0.453	0.001215**
Seagrass above	-0.186	0.204836
Seagrass below	-0.020	0.891891
Epiphyts	-0.190	0.193735
Chl a	-0.042	0.776210
Phaeopigments	-0.420	0.002971**
Organic matter	-0.516	0.000172***
Meiofauna	-0.016	0.913168
Macrofauna	0.102	0.490559

* p<0.05, ** p<0.01, *** p<0.0001

Table V - Correlation matrix (Pearson) for all variables.

	Salinity	Seagrass above	Seagrass below	Epiphyts	Meiofauna	Macrofauna	Chlorophyll a	Phaeopigments	Organic matter	pH	Ammonia	Nitrite	Nitrate
Salinity	1.0000												
Seagrass above	-0.1863	1.0000											
Seagrass below	-0.0201	0.7139***	1.0000										
Epiphyts	-0.1909	0.5288***	0.4777**	1.0000									
Meiofauna	-0.0162	0.5297***	0.2815	-0.0904	1.0000								
Macrofauna	0.1019	0.0975	0.3081*	0.1121	-0.1496	1.0000							
Chlorophyll a	-0.0421	0.0604	0.0171	0.1944	0.1307	-0.1414	1.0000						
Phaeopigments	-0.4198**	0.2136	0.0793	-0.1569	0.3680*	-0.0352	0.3330*	1.0000					
Organic matter	-0.5164***	0.1929	0.0527	0.1638	0.1840	-0.1419	0.0696	0.5526***	1.0000				
pH	0.3907**	0.0244	0.1712	0.1770	-0.2272	0.2649	0.0045	-0.6190***	-0.5039***	1.0000			
Ammonia	-0.3820**	-0.1204	-0.2758	-0.2271	0.2556	-0.3704*	-0.0116	0.5575***	0.5552***	-0.8600***	1.0000		
Nitrite	-0.2110	0.1119	-0.0132	-0.0847	0.2278	-0.1820	-0.0126	0.4083**	0.3661*	-0.6069***	0.5651***	1.0000	
Nitrate	-0.4533**	0.1090	-0.0747	0.0340	0.3405*	-0.2742	0.0048	0.4929***	0.6090***	-0.7354***	0.7761***	0.7013***	1.0000

* $p<0.05$, ** $p<0.01$, *** $p<0.001$

Table VI - Data from water column parameters (Tc - *Thalassodendron ciliatum* , Th - *Thalassia hemprichii* , Zc - *Zostera capensis*).

Season	Seagrass	Station	Sample	Replicate	Temperature	Salinity	pH	Ammonia	Nitrite	Nitrate
Dry	Tc	1	A	1	23.5	34.6	7.63	0.12	0.012	3.4
Dry	Tc	1	A	2	24.0	34.6	7.62	0.09	0.012	3.3
Dry	Tc	1	B	1	25.0	34.8	7.65	0.11	0.025	3.3
Dry	Tc	1	B	2	25.6	34.8	7.63	0.13	0.025	3.1
Dry	Tc	2	A	1	25.4	34.6	7.62	0.11	0.041	3.5
Dry	Tc	2	A	2	26.2	34.3	7.67	0.12	0.200	3.6
Dry	Tc	2	B	1	26.3	34.5	7.66	0.14	0.042	4.3
Dry	Tc	2	B	2	26.7	34.7	7.68	0.17	0.019	3.5
Dry	Th	1	A	1	26.3	35.7	7.78	0.21	0.030	2.9
Dry	Th	1	A	2	26.2	35.7	7.71	0.12	0.042	3.3
Dry	Th	1	B	1	26.9	36.0	7.73	0.15	0.020	3.8
Dry	Th	1	B	2	26.3	36.2	7.77	0.12	0.019	3.2
Dry	Th	2	A	1	23.0	36.0	7.70	0.20	0.018	4.4
Dry	Th	2	A	2	23.4	35.8	7.71	0.13	0.037	3.7
Dry	Th	2	B	1	23.4	36.1	7.72	0.15	0.025	3.8
Dry	Th	2	B	2	23.5	36.0	7.76	0.13	0.026	4.0
Dry	Zc	1	A	1	23.5	35.6	7.68	0.14	0.033	3.7
Dry	Zc	1	A	2	23.4	36.2	7.69	0.13	0.027	3.4
Dry	Zc	1	B	1	23.5	36.2	7.63	0.17	0.029	3.7
Dry	Zc	1	B	2	24.7	36.4	7.67	0.16	0.060	3.7
Dry	Zc	2	A	1	25.9	35.9	7.70	0.18	0.055	3.4
Dry	Zc	2	A	2	25.7	36.0	7.69	0.14	0.050	4.0
Dry	Zc	2	B	1	26.3	35.5	7.68	0.16	0.047	3.3
Dry	Zc	2	B	2	27.0	35.9	7.66	0.18	0.063	4.2
Wet	Tc	1	A	1	24.7	34.2	8.35	0.00	0.018	1.6
Wet	Tc	1	A	2	25.6	34.4	8.49	0.00	0.010	1.3
Wet	Tc	1	B	1	25.4	34.6	8.45	0.00	0.020	1.2
Wet	Tc	1	B	2	25.3	34.6	8.66	0.00	0.023	1.9
Wet	Tc	2	A	1	25.6	35.2	8.90	0.00	0.024	1.6
Wet	Tc	2	A	2	24.7	35.4	8.90	0.00	0.034	1.4
Wet	Tc	2	B	1	24.7	34.9	8.87	0.00	0.048	2.1
Wet	Tc	2	B	2	24.6	35.0	8.96	0.00	0.015	1.5
Wet	Th	1	A	1	26.5	34.7	8.92	0.00	0.076	1.9
Wet	Th	1	A	2	26.8	34.8	8.96	0.00	0.002	1.5
Wet	Th	1	B	1	26.8	34.9	9.07	0.04	0.015	2.4
Wet	Th	1	B	2	26.8	35.0	9.10	0.00	0.057	2.1
Wet	Th	2	A	1	31.1	34.7	8.95	0.00	0.021	1.5
Wet	Th	2	A	2	31.3	35.5	8.91	0.00	0.022	1.3
Wet	Th	2	B	1	32.5	35.7	9.16	0.00	0.021	1.1
Wet	Th	2	B	2	32.2	35.8	9.21	0.00	0.012	1.0
Wet	Zc	1	A	1	32.0	35.8	8.78	0.04	0.023	1.6
Wet	Zc	1	A	2	31.4	35.8	8.74	0.03	0.010	1.9
Wet	Zc	1	B	1	32.8	37.8	8.85	0.01	0.039	1.9
Wet	Zc	1	B	2	33.1	37.0	8.75	0.03	0.022	2.5
Wet	Zc	2	A	1	32.3	35.5	8.75	0.01	0.022	2.3
Wet	Zc	2	A	2	32.3	36.1	8.82	0.04	0.009	1.9
Wet	Zc	2	B	1	32.7	36.5	8.96	0.00	0.000	2.0
Wet	Zc	2	B	2	33.1	36.8	8.97	0.06	0.017	2.2

Table VII - Data from pore water parameters (Tc - *Thalassodendron ciliatum*, Th - *Thalassia hemprichii*, Zc - *Zostera capensis*).

Season	Seagrass	Station	Sample	Replicate	Temperature	Salinity	pH	Ammonia	Nitrite	Nitrate
Dry	Tc	1	A	1	23.0	34.4	7.54	0.53	0.093	4.2
Dry	Tc	1	A	2	23.1	34.9	7.57	0.32	0.058	3.8
Dry	Tc	1	B	1	24.3	34.8	7.56	0.33	0.052	3.6
Dry	Tc	1	B	2	24.5	34.8	7.55	0.46	0.074	4.3
Dry	Tc	2	A	1	25.3	30.9	7.58	0.56	0.072	4.4
Dry	Tc	2	A	2	25.6	30.9	7.57	0.49	0.146	5.3
Dry	Tc	2	B	1	25.7	32.3	7.57	0.36	0.066	4.3
Dry	Tc	2	B	2	25.4	28.6	7.57	0.46	0.081	4.9
Dry	Th	1	A	1	25.5	33.5	7.61	0.60	0.048	4.0
Dry	Th	1	A	2	25.1	32.5	7.60	0.59	0.044	3.7
Dry	Th	1	B	1	25.7	34.0	7.56	0.48	0.048	3.2
Dry	Th	1	B	2	24.7	34.4	7.56	0.50	0.044	3.8
Dry	Th	2	A	1	22.6	33.5	7.59	0.62	0.050	4.0
Dry	Th	2	A	2	22.5	34.1	7.57	0.65	0.152	5.4
Dry	Th	2	B	1	21.7	35.1	7.55	0.59	0.066	4.3
Dry	Th	2	B	2	21.5	34.2	7.56	0.48	0.052	4.0
Dry	Zc	1	A	1	22.2	34.6	7.54	0.57	0.160	5.2
Dry	Zc	1	A	2	22.4	35.5	7.51	0.59	0.088	5.0
Dry	Zc	1	B	1	23.6	35.2	7.50	0.84	0.059	4.2
Dry	Zc	1	B	2	23.5	35.3	7.53	0.41	0.034	3.9
Dry	Zc	2	A	1	25.6	34.3	7.59	0.35	0.055	4.9
Dry	Zc	2	A	2	25.5	33.6	7.61	0.33	0.046	4.3
Dry	Zc	2	B	1	26.5	33.0	7.59	0.70	0.121	6.6
Dry	Zc	2	B	2	25.6	34.0	7.60	0.46	0.086	3.9
Wet	Tc	1	A	1	23.4	33.4	7.85	1.15	0.026	2.9
Wet	Tc	1	A	2	23.6	34.2	7.85	1.15	0.000	3.8
Wet	Tc	1	B	1	23.2	33.4	7.84	0.75	0.009	2.8
Wet	Tc	1	B	2	23.4	35.1	7.99	0.95	0.023	3.0
Wet	Tc	2	A	1	23.5	34.6	7.91	0.75	0.014	2.5
Wet	Tc	2	A	2	23.5	34.7	7.91	0.75	0.019	2.6
Wet	Tc	2	B	1	24.0	35.2	7.95	0.65	0.018	2.8
Wet	Tc	2	B	2	24.6	35.9	7.96	0.85	0.014	2.8
Wet	Th	1	A	1	26.3	35.2	7.75	3.68	0.073	2.7
Wet	Th	1	A	2	26.0	35.4	7.71	5.00	0.048	2.8
Wet	Th	1	B	1	26.2	35.4	7.88	4.80	0.074	3.7
Wet	Th	1	B	2	26.4	35.3	7.96	1.80	0.058	2.6
Wet	Th	2	A	1	27.8	34.5	7.75	0.70	0.020	1.7
Wet	Th	2	A	2	27.4	33.6	7.80	0.80	0.008	1.8
Wet	Th	2	B	1	27.4	33.8	7.82	0.40	0.019	2.5
Wet	Th	2	B	2	27.6	34.2	7.91	0.90	0.040	2.5
Wet	Zc	1	A	1	28.6	35.1	7.85	0.45	0.035	2.1
Wet	Zc	1	A	2	29.9	35.8	7.74	0.50	0.031	1.9
Wet	Zc	1	B	1	28.4	35.3	8.06	0.70	0.020	2.3
Wet	Zc	1	B	2	28.7	36.4	8.00	0.70	0.023	2.3
Wet	Zc	2	A	1	28.5	35.0	7.91	0.40	0.012	1.9
Wet	Zc	2	A	2	29.1	35.6	7.92	0.45	0.014	2.3
Wet	Zc	2	B	1	30.4	34.9	7.85	0.25	0.018	2.7
Wet	Zc	2	B	2	29.2	34.7	7.74	0.40	0.021	2.2

Table VIII - Data from sediment biological parameters (AFDW m^{-2}). Tc - *Thalassodendron ciliatum*, Th - *Thalassia hemprichii*, Zc - *Zostera capensis*.

Season	Seagrass	Station	Sample	Replicate	Seagrass above	Seagrass below	epiphytic biomass	meiofauna biomass	macrofauna biomass
Dry	Tc	1	A	1	137.07	718.11	8.457	4.59	2.93
Dry	Tc	1	A	2	58.77	229.25	3.731	4.65	39.49
Dry	Tc	1	A	3	53.09	288.36	2.189	4.21	4.35
Dry	Tc	1	B	1	101.49	1471.09	3.333	4.38	36.28
Dry	Tc	1	B	2	131.44	371.70	6.567	6.05	37.52
Dry	Tc	1	B	3	93.09	357.96	5.373	7.24	20.72
Dry	Tc	2	A	1	78.56	1044.48	4.329	7.54	3.50
Dry	Tc	2	A	2	55.47	0.40	1.891	53.20	2.08
Dry	Tc	2	A	3	85.57	358.47	3.583	48.85	21.95
Dry	Tc	2	B	1	84.63	13.23	3.184	17.40	42.01
Dry	Tc	2	B	2	123.88	349.50	6.816	20.70	16.68
Dry	Tc	2	B	3	86.42	358.47	2.935	37.30	7.91
Dry	Th	1	A	1	42.99	258.06	0.149	5.29	5.36
Dry	Th	1	A	2	105.00	323.48	0.000	8.18	26.64
Dry	Th	1	A	3	87.16	9.21	0.000	6.82	5.67
Dry	Th	1	B	1	47.06	216.96	0.000	9.20	11.11
Dry	Th	1	B	2	79.30	251.59	0.199	9.50	14.96
Dry	Th	1	B	3	48.61	148.98	0.000	14.20	20.28
Dry	Th	2	A	1	78.56	340.55	0.000	79.80	8.35
Dry	Th	2	A	2	291.14	1307.56	0.199	184.15	9.31
Dry	Th	2	A	3	60.50	353.58	0.139	96.70	25.51
Dry	Th	2	B	1	90.38	471.14	0.000	102.60	11.17
Dry	Th	2	B	2	125.32	642.19	0.000	133.05	38.68
Dry	Th	2	B	3	45.22	211.81	0.298	61.20	10.33
Dry	Zc	1	A	1	14.58	105.08	0.548	22.52	9.65
Dry	Zc	1	A	2	23.98	94.18	0.398	45.37	5.27
Dry	Zc	1	A	3	22.58	83.38	0.597	56.68	3.25
Dry	Zc	1	B	1	15.87	66.22	0.249	47.45	7.99
Dry	Zc	1	B	2	18.56	85.27	0.199	46.50	1.08
Dry	Zc	1	B	3	27.02	195.52	0.792	12.18	13.12
Dry	Zc	2	A	1	7.91	79.40	0.249	36.90	18.27
Dry	Zc	2	A	2	22.97	127.56	0.199	18.90	16.70
Dry	Zc	2	A	3	25.23	151.00	1.393	19.85	2.54
Dry	Zc	2	B	1	19.45	97.56	0.398	10.55	3.79
Dry	Zc	2	B	2	17.56	149.16	0.097	32.40	5.95
Dry	Zc	2	B	3	16.37	90.05	0.099	31.80	15.35
Wet	Tc	1	A	1	83.33	835.82	4.677	25.30	208.87
Wet	Tc	1	A	2	136.52	413.04	9.851	18.20	6.90
Wet	Tc	1	A	3	115.37	548.56	5.672	7.62	6.43
Wet	Tc	1	B	1	170.65	578.36	4.527	53.15	26.04
Wet	Tc	1	B	2	168.21	824.93	6.468	52.90	6.30
Wet	Tc	1	B	3	69.55	774.43	6.766	18.65	8.91
Wet	Tc	2	A	1	138.11	1068.16	6.716	58.05	9.98
Wet	Tc	2	A	2	110.40	1093.18	7.264	33.90	11.93
Wet	Tc	2	A	3	132.59	1034.93	5.771	22.38	28.62
Wet	Tc	2	B	1	112.59	1065.27	4.378	17.20	155.42
Wet	Tc	2	B	2	99.45	814.03	6.667	25.55	138.63
Wet	Tc	2	B	3	71.94	759.55	4.677	16.05	7.16
Wet	Th	1	A	1	62.79	488.01	0.249	22.70	34.19
Wet	Th	1	A	2	58.56	339.20	0.172	14.05	50.21
Wet	Th	1	A	3	41.19	125.18	0.249	13.90	14.91
Wet	Th	1	B	1	64.03	543.48	0.095	7.50	111.24
Wet	Th	1	B	2	59.20	447.86	0.149	7.90	71.02
Wet	Th	1	B	3	56.47	310.95	0.348	6.30	10.58
Wet	Th	2	A	1	66.97	474.88	0.398	3.62	243.41
Wet	Th	2	A	2	98.71	478.76	0.149	4.27	10.77
Wet	Th	2	A	3	54.27	247.36	0.249	4.60	78.03
Wet	Th	2	B	1	51.74	531.19	0.279	7.60	46.99
Wet	Th	2	B	2	42.54	542.09	0.199	7.15	14.51
Wet	Th	2	B	3	46.32	328.91	0.398	8.65	33.26
Wet	Zc	1	A	1	15.82	71.89	0.189	7.05	5.37
Wet	Zc	1	A	2	33.88	121.94	0.149	2.30	4.37
Wet	Zc	1	A	3	32.74	68.71	0.000	6.00	6.51
Wet	Zc	1	B	1	23.33	67.07	0.547	11.05	4.04
Wet	Zc	1	B	2	51.29	87.71	0.199	18.35	4.47
Wet	Zc	1	B	3	31.60	112.99	0.050	20.25	9.55
Wet	Zc	2	A	1	24.28	82.89	0.796	11.13	12.21
Wet	Zc	2	A	2	22.49	97.02	1.393	17.55	7.63
Wet	Zc	2	A	3	22.29	83.78	1.144	14.67	4.47
Wet	Zc	2	B	1	20.70	96.32	0.597	25.65	3.19
Wet	Zc	2	B	2	29.85	65.97	0.299	19.25	12.38
Wet	Zc	2	B	3	23.73	86.17	0.547	29.35	53.19

Table IX - Data from sediment 1st centimetre parameters (Tc - *Thalassodendron ciliatum*, Th - *Thalassia hemprichii*, Zc - *Zostera capensis*).

Season	Seagrass	Station	Sample	Replicate	%O.M.	Chl a ug/g	Phaeop. ug/g
Dry	Tc	1	A	1	7.2	8.1	4.11
Dry	Tc	1	A	2	6.7	6.0	3.79
Dry	Tc	1	B	1	9.6	6.5	3.96
Dry	Tc	1	B	2	10.0	6.7	3.59
Dry	Tc	2	A	1	10.2	8.4	5.91
Dry	Tc	2	A	2	9.6	11.8	6.64
Dry	Tc	2	B	1	13.1	4.8	4.12
Dry	Tc	2	B	2	13.2	8.2	4.15
Dry	Th	1	A	1	10.0	4.2	4.66
Dry	Th	1	A	2	6.7	5.7	5.16
Dry	Th	1	B	1	7.8	3.1	3.72
Dry	Th	1	B	2	11.9	5.3	5.46
Dry	Th	2	A	1	9.6	4.0	3.61
Dry	Th	2	A	2	7.8	6.7	5.47
Dry	Th	2	B	1	12.0	6.6	5.40
Dry	Th	2	B	2	10.6	8.9	6.95
Dry	Zc	1	A	1	5.3	7.3	3.23
Dry	Zc	1	A	2	3.0	7.0	3.20
Dry	Zc	1	B	1	4.6	4.9	2.75
Dry	Zc	1	B	2	3.8	3.6	2.84
Dry	Zc	2	A	1	8.7	6.4	3.64
Dry	Zc	2	A	2	11.7	6.4	3.05
Dry	Zc	2	B	1	16.0	5.5	3.71
Dry	Zc	2	B	2	12.0	8.4	5.26
Wet	Tc	1	A	1	6.0	7.1	2.54
Wet	Tc	1	A	2	7.9	4.8	1.54
Wet	Tc	1	B	1	7.3	7.6	2.90
Wet	Tc	1	B	2	7.9	7.1	1.93
Wet	Tc	2	A	1	6.4	12.1	3.30
Wet	Tc	2	A	2	6.3	5.3	1.17
Wet	Tc	2	B	1	6.9	9.0	3.38
Wet	Tc	2	B	2	6.8	7.5	2.80
Wet	Th	1	A	1	12.1	3.1	3.98
Wet	Th	1	A	2	5.3	4.2	1.77
Wet	Th	1	B	1	3.7	2.6	1.90
Wet	Th	1	B	2	2.4	4.0	4.01
Wet	Th	2	A	1	5.2	3.8	4.53
Wet	Th	2	A	2	2.3	2.5	2.51
Wet	Th	2	B	1	3.4	1.2	1.09
Wet	Th	2	B	2	3.6	3.2	1.95
Wet	Zc	1	A	1	4.9	11.8	2.69
Wet	Zc	1	A	2	4.0	18.6	4.25
Wet	Zc	1	B	1	3.9	7.1	2.36
Wet	Zc	1	B	2	4.1	6.8	2.33
Wet	Zc	2	A	1	5.2	4.3	0.76
Wet	Zc	2	A	2	4.4	5.6	1.36
Wet	Zc	2	B	1	4.4	8.6	2.80
Wet	Zc	2	B	2	3.7	7.9	2.50

Table X - Results of nested ANOVA for water column parameters.

Effect	df effect	MS effect	df error	MS error	F	p - level
Temperature						
season	1*	178.6408*	6*	14.65125*	12.19287*	0.012954
seagrass	2	49.7494	6	14.65125	3.39557	0.103211
station	6*	14.6512*	12*	.64833*	22.59833*	0.000007
sample	12*	.6483*	24*	.13333*	4.86250*	0.000489
replicate	24	0.1333	0	0	-	-
season x seagrass	2	60.4765	6	14.65125	4.12773	0.074561
Salinity						
season	1	0.040833	6	0.34625	0.11793	0.743004
seagrass	2*	8.8975*	6*	0.34625*	25.69675*	.001143*
station	6	0.34625	12	0.36875	0.93898	0.502567
sample	12*	0.36875*	24*	0.055*	6.70455*	.000041*
replicate	24	0.055	0	0	-	-
season x seagrass	2	1.725833	6	0.34625	4.98436	0.53045
pH						
season	1*	16.38003*	6*	0.062975*	260.1038*	.000004*
seagrass	2	0.18661	6	0.062975	2.9632	0.127328
station	6*	0.06297*	12*	0.011871*	5.305*	.006921*
sample	12*	0.01187*	24*	.002267*	5.2371*	0.000284
replicate	24	0.00227	0	0	-	-
season x seagrass	2	0.06201	6	0.062975	0.4268	0.426769
Amonia						
season	1*	.213333*	6*	.000279*	764.1791*	.000000*
seagrass	2*	.003756*	6*	.000279*	13.4552*	.006060*
station	6	0.000279	12	0.000417	0.67	0.676209
sample	12	.000417*	24	0.000529	0.7874	0.658540
replicate	24	0.000529	0	0	-	-
season x seagrass	2	0.00054	6	0.000279	1.9328	0.224944
Nitrite						
season	1	0.003284	6	0.001385	2.371594	0.174489
seagrass	2	0.000244	6	0.001385	0.17634	0.842525
station	6	0.001385	12	0.000759	1.82509	0.176615
sample	12	0.000759	24	0.000773	0.981615	0.492230
replicate	24	0.000773	0	0	-	-
season x seagrass	2	0.000959	6	0.001385	0.69258	0.536257

* p<0.05, ** p<0.01, *** p<0.001

Table XI - Results of nested ANOVA for pore water parameters.

Effect	df effect	MS effect	df error	MS error	F	p - level
Temperature						
season	1*	65.33334*	6*	7.887500*	8.28315*	.028130*
seagrass	2	27.17021	6	7.8875	3.44472	0.100868
station	6*	7.8875	12*	0.563333*	14.00148*	.000085*
sample	12*	.56333*	24*	0.13875*	4.06006*	.001700*
replicate	24	0.13875	0	0	-	-
season x seagrass	2	34.87521	6	7.8875	4.42158	0.06605
Salinity						
season	1	16.68521	6	7.343125	2.27222	0.182434
seagrass	2	6.28083	6	7.343125	0.86895	0.466212
station	6	7.34312	12	0.338125	21.71719	0.000009***
sample	12	0.33812	24	0.536042	0.63078	0.795746
replicate	24	0.53604	0	0		
season x seagrass	2	1.42333	6	7.343125	0.19383	0.828758
pH						
season	1	1.119352	6	0.004123	271.4952	0.000003***
seagrass	2	0.005527	6	0.004123	1.3406	0.330158
station	6	0.004123	12	0.010240	0.4026	0.863428
sample	12	0.01024	24	0.001523	6.7237	0.000040***
replicate	24	0.001523	0	0		
season x seagrass	2	0.010415	6	0.004123	2.5260	0.160002
Amonia						
season	1	5.782408	6	3.289158	1.75802	0.233113
seagrass	2	3.767652	6	3.289158	1.14548	0.379001
station	6	3.289158	12	0.111417	29.52124	0.000002***
sample	12	0.111417	24	0.238387	0.46738	0.914601
replicate	24	0.238387	0	0		
season x seagrass	2	3.248952	6	3.289158	0.9878	0.425766
Nitrite						
season	1	0.027937	6	0.001185	23.57946	0.002835**
seagrass	2	0.000105	6	0.001185	0.08832	0.916632
station	6	0.001185	12	0.001037	1.14197	0.396132
sample	12	0.001037	24	0.000574	1.80605	0.105253
replicate	24	0.000574	0	0		
season x seagrass	2	0.002340	6	0.001185	1.97540	0.219220
Nitrate						
season	1	40.33333	6	0.715417	56.37740	0.000289***
seagrass	2	0.44146	6	0.715417	0.61706	0.570552
station	6	0.71542	12	0.267917	2.67030	0.069667
sample	12	0.26792	24	0.305000	0.87842	0.578322
replicate	24	0.30500	0	0		
season x seagrass	2	1.49146	6	0.715417	2.08474	0.205380

* p<0.05, ** p<0.01, *** p<0.001

Table XII - ANOVA results for benthic biological parameters.

Effect	df effect	MS effect	df error	MS error	F	p - level
Seagrass above						
season	1	5.95	6	1268.581	0.00469	0.947605
seagrass	2	40000.57	6	1268.581	31.53174	0.000656***
station	6	1268.58	12	914.878000	1.38661	0.295979
sample	12	914.88	48	1192.737	0.76704	0.680085
replicate	48	1192.74	0	0	-	-
season x seagrass	2	5636.18	6	1268.581	4.4429	0.065484
Seagrass below						
season	1	255617	6	136041.7	1.87896	0.219517
seagrass	2	1765433	6	136041.7	12.97715	.006620**
station	6	136042	12	34418.6	3.95257	.020491*
sample	12	34419	48	56796.8	0.60599	0.826317
replicate	48	56797	0	0	-	-
season x seagrass	2	252185	6	136041.7	1.85373	0.236122
Epiphyts						
season	1	7.7934	6	0.938919	8.3003	0.02802
seagrass	2	194.9117	6	0.938919	207.5915	0.000003***
station	6	0.9389	12	0.531098	1.7679	0.188733
sample	12	0.5311	48	1.319025	0.4026	0.955639
replicate	48	1.319	0	0	-	-
season x seagrass	2	5	6	0.938919	5.77290	0.039988*
Meiofauna						
season	1	6199.179	6	5547.526	1.11747	0.331157
seagrass	2	900.048	6	5547.526	0.16224	0.853843
station	6	5547.526	12	240.858000	23.03235	.000006***
sample	12	240.858	48	295.298000	0.81156	0.633442
replicate	48	295.29800	0	0	-	-
season x seagrass	2	5650.042	6	5547.526	1.01848	0.416082
Macrofauna						
season	1	12166.62	6	451.709	26.93461	.002035**
seagrass	2	3786.68	6	451.709	8.383	.018306*
station	6	451.71	12	2320.059	0.19470	0.972195
sample	12	2320.06	48	1678.674	1.38208	0.207312
replicate	48	1678.67000	0	0	-	-
season x seagrass	2	4906.02	6	451.709*	10.86101	.010139*

* $p<0.05$, ** $p<0.01$, *** $p<0.001$

Table XIII - ANOVA results for sediment 1st centimetre parameters.

Effect	df effect	MS effect	df error	MS error	F	p - level
Organic matter						
season	1	180.2103	6	26.286	6.855752	0.39674*
seagrass	2	19.6744	6	26.286	0.748475	0.512625
station	6	26.286	12	6.3122	4.164317	.017074*
sample	12	6.3122	24	2.64537	2.386128	.033684*
replicate	24	2.6454	0	0	-	-
season x seagrass	2	3.1834	6	26.286	0.121106	0.888051
Chlorophyll a						
season	1	0.0342	6	10.443	0.003275	0.956225
seagrass	2	54.66239	6	10.443	5.234357	.048359*
station	6	10.443	12	9.15494	1.11407	0.39673
sample	12	9.15494	24	3.49584	2.61881	.021562*
replicate	24	3.49584	0	0	-	-
season x seagrass	2	26.435	6	10.443	2.53136	0.159539
Phaeopigments						
season	1	40.46028	6	1.486542	27.21771	.001982**
seagrass	2	3.76528	6	1.486542	2.53291	0.159405
station	6	1.48654	12	1.463434	1.01579	0.459664
sample	12	1.46343	24	0.763229	1.91743	0.084446
replicate	24	0.76323	0	0	-	-
season x seagrass	2	1.77508	6	1.486542	1.1341	0.365972

* $p<0.05$, ** $p<0.01$, *** $p<0.001$

The difference of pH between seasons can be attributed to the stronger influence of freshwater during dry season. In the water column, the lower values were obtained over *the T. ciliatum* meadow, where estuarine influence is maximal (it seems that river discharge into Maputo bay has anyway a stronger effect when compared to groundwater outflow, at least for water column values).

For nitrites and nitrates, maximal values occurred during dry season, thus corroborating the picture observed for other parameters. The exception is ammonia concentration, which tends to be higher during wet season, specially at *T. ciliatum and T. hemprichii* meadows (in this last meadow concentration is up to x10 and could be due to the massive presence of bivalve moluscs).

Organic matter will reflect the deposition and decomposition of mainly decaying seagrass blades and also small animals, and is expected to fluctuation according the production of the biomass on each sampling areas. Consistent is the correlation between this parameter and phaeopigments (at $p<0.001$ level), which may indicate the decay of seagrass detritus.

In shallow habitats such seagrass meadows provide a rich environment which support a large variety of microscopic organisms. Regularly present in this type of habitat, benthic microalgae are found in the top few cm (Cariou-Le Gall and Blanchard, 1995) and its spacial distribution is affected on different spatial scales (Saburova *et al.*, 1995). The chlorophyll a and phaeopigment determination allows to quantify quickly microalgal biomass. Differences on chlorophyll a concentrations were found between all studied stations and seasons, except on *T. ciliatum* meadow. The observed differences in chlorophyll a concentration among stations could be caused by heterogeneity of the environment, with increasing area allowing microalgae species to disperse in space according to a complex of environmental physical and chemical requirements. As referred by Sundbäck and Granéli (1988) there is a correlation between light intensity, microphytobenthic activity and nutrient flux. The microphytobenthos influences the nutrient flux both directly through nutrient uptake from water and sediment, and indirectly by affecting the oxygen concentration at sediment-water interface through photosynthesis and respiration.

The chlorophyll a degradation products, known as phaeopigments, form an important part of the whole pigments often found in aquatic ecosystems. Its concentration depends on phytoplankton sinking, seagrass decomposition, periphyton and microphytobenthos. From phaeopigment analysis carried out on seagrass meadows was clear that on dry season were found the highest values at all stations.

Both above and below ground components of seagrasses are, as expected, correlated. Meiofauna, which may depend of organic matter originated by the decaying leafs of seagrasses, was correlated to above ground seagrass biomass as well as to phaeopigment contents of the sediment. Epiphytic biomass was correlated to seagrass biomass. Macrofauna was correlated to seagrass below ground biomass, suggesting the dependence on microhabitat provided by the roots of seagrasses and associated microfauna.

Intertidal vegetated habitats are thought to provide greater densities of food and greater degrees of refuge to attract mobile organisms (Sheridan, 1997). Their high productivity have received much attention from estuarine researchers attempting to ascertain the value of these habitats to fishery and foraging organisms (Ansari *et al.*, 1991; Sheridan, 1997). As observed by Ansari *et al.* (1991) the habitat complexity, as measured by benthic macrophyte biomass, seems to play an important role in structuring macrobenthic community in the seagrass meadows. Benthic macrofauna assemblages on the three seagrass meadows presented at Inhaca island showed a similar behaviour during both seasons. Nevertheless, during wet season the biomass was higher mainly on *T. ciliatum* and *T. hemprichii,* with net differences between replicates. For *Z. capensis* the biomass fluctuation obtained between seasons were sensibly lower. In our study, macrofauna biomasses followed the same pattern as observed for seagrasses, corroborating the strong relation between them. The dominance of suspension feeders and detritivores like polychaets, nematodes and bivalves were observed during the sampling period in all stations. This fact could be related to the high concentration of organic particulate matter proceeding from breakdown of seagrass detritus.

There seems to be no strong evidence of groundwater outflow at the study areas. As referred above the catchment area of Inhaca Island is to small to produce large aquifers. The main supporting evidence for goundwater outflow are the mangrove areas around the island corroborated by the model flux developed within GROFLO scope. Nevertheless, there seems to be an indication of diffuse groundwater outflow at the *Thalassodendron ciliatum* meadow. This evidence is supported by the distribution of the salinity values on pore water which have a significant variability at the scale of hundred meters (station level, see nested ANOVA results). The influence of lower salinity is significantly correlated to nutrient load and organic matter contents in the sediments. The lack of negative correlation of pore water salinity with seagrass and infaunal biomass may reflect only the narrow range of salinity found in the meadows.

Conclusions

As a conclusion, it seems that there is no strong evidence of groundwater outflow at the study areas, due to the fact that catchment area of Inhaca Island is to small to produce large aquifers. Nevertheless, there are some indications of diffuse groundwater outflow at the *Thalassodendron ciliatum* meadow. Differences between seagrass meadows specially in what concerns benthic macrophyte biomass, macro and meiobenthic community associated could be attributed to complex interactions inducing heterogeneity of the environment at different spatial and temporal scales.

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Eduardo Mondlane University
Department of Biological Sciences

Human impact on the quantity and quality of groundwater of Inhaca Island, Southern Mozambique

Domingos Z. Gove & Helena Chavale
Eduardo Mondlane University, Maputo, Mozambique

Introduction

Inhaca Island is located in Southern Mozambique, at 26º 00' S and 36º 00' E, being part of the barrier that separates Maputo Bay from the Indian Ocean. It has an area of 42 Km^2, and it is in the transition zone between wet tropical climate to warm temperate climate with wet and warmer summers and dry and fresh winters (Macnae and Kalk, 1969). The mean annual precipitation is 874 mm, being January and February, the most wet months (Kalk, 1985).

The topography of the island is irregular and its soils are sandy, alkaline, and with low humidity (Hernroth and Gove, 1995). There is no river within the island, and the freshwater lagoons which existed in the past, now are dry, in part due to agricultural practices.

The human population is composed by 5,504 inhabitants, with a population increase of 3% per year (Anon., 1996), and is distributed as follow: Nhaquene (), Ribjene (), and Inguane (). In average, there are seven people per family. Thirty per cent of the island is terrestrial reserve, which is managed by the Marine Biological Station of Inhaca.

Socio-economic infrastructures include one hotel, with 110 beds; one restaurant; several small bars; one hospital; one fishing centre; some fishing camps; and the Marine Biological Station, with capacity for 60 people. There is also a project for installing one more hotel, in the southern part of the island, with a capacity for 300 beds.

The great dependence of population from the island on groundwater implies the existence of impacts on this resource. Very little information on groundwater exists in Mozambique, however, it is known that in the southern part of the country sedimentary formations determine important underground basins which can be widely exploited for supplying water to villages, urban centres and irrigation (Casadei, 1980). The present work is intended to determine the present status in the island regarding the amount of water used and its quality.

Objectives

- To estimate the amount of groundwater used by the human population at Inhaca island;
- To analyze the use patterns; and
- To determine its quality.

Material and Methods

Sampling

Estimation of the amount of water used by the population and water use pattern

For estimating the quantity of water used by the population and water use pattern, thirty families in each of the three villages were inquired (see the questionnaire). The quantity of water collected per day by each of the families was obtained using the following formula:

Quant = Vol x Freq.
Quant – Quantity of water
Vol – Volume of containers used in collection of water

Freq – Frequency of use of the containers per day

Quality of water

Twelve (12) wells in each of the three villages were sampled in order to determine the quality of water, including its color. This consisted of determination of salinity (a small amount of water was taken and its salinity determined with the use of a refractometer), NO_3^-, NO_2^-,NH_4^+, Cl^- and bacterial counting. The smell was determined directly.

For bacterial contamination, samples were incubated and then observed under microscope. The total coliforms were determined after incubation of samples at 37°C for 48 hours, and the fecal coliforms after incubation at 44°C for 24 hours.

Questionnaire:

1. Which containers do you use to collect water?
2. How often do you use them per day?
3. How many people are in the family?
4. What do you do with the water?
4. Is there a lack of water in wells? When?
5. Is there a toilet in the family? Which is the distance from the well?

Data analysis

Kruskal-Wallis non parametric test was used to compare the chemical levels and fecal coliforms among the three villages.

Results and Discussion

Number of wells

Fifty-nine wells were counted at Inhaca Island (table 1). At Nhaquene, there were five taps, two cemented wells with cover, five cemented wells without a cover and three artisanal wells. From the 26 wells found at Ribjene, 10 were taps, five cemented with cover, five cemented without cover, 4 not cemented and 2 artisanal. At Inguane there were five taps, six cemented with cover, four cemented without cover and three artisanal wells. The village of Ribjene showed a relatively larger number of wells since it has got the majority of the inhabitants (table 2).

Quantity of groundwater taken in liters per day

At the village of Inguane, from 223 interviewed inhabitants was calculated an amount of 3,805 l of water used per day, which gives a value of 17.1 l per person. At Nhaquene, 214 inhabitants used 3,177, which gives a value of 14.5 l per person, and at Ribjene, excepting the Hotel, has a value of 16.7 l/day/person (table 3).

Table 1. Number of wells per village

Village	Number of wells
Nhaquene	15
Ribjene	26
Inguane	18
Total	59

Table 2. Number of inhabitants and families per village (from Anon., 1990)

Village	Inhabitants	Families
Nhaquene	1,871	300
Ribjene	2,480	565
Inguane	1,153	206
Total	5,504	1,071

The values are similar among the three villages. The average from the three villages is 16.2 l per person per day, which is smaller when compared to Unguja Island (Zanzibar), where they have a value of 32.5 l per person per day, that is, twice the amount consumed at Inhaca. The amount of water used per day at Hotel Inhaca is 7000 l/day (Manager, inform). Taking in account that in average there are 97 people at the Hotel, including staff and tourists, the amount per person per day is 72.2 l.. If we include the Hotel, the amount of water used per person per day in the island increases to 23.4 l/person/day.

Table 3. Number of users and the volume of water used

Village	Number of users	Total volume of water used (l)/day	Volume of water (l)/day/person
Inguane	223	3805.0	17.1
Nhaquene	214	3177.0	14.5
Ribjene	225	3757.5	16.7
Total	662	10739.5	16.2

Multiplying the mean value of water used per person per day (table 3) by the number of inhabitants (table 2), we have the following (table 4):

Table 4. Amount of water used per day per village

Village	Inhabitants	Amount of water used per day (l)
Nhaquene	1871	29936
Ribjene	2480	40176
Inguane	1153	18448
Hotel Inhaca	97	7000
Total	5601	95560

The total amount of water used per day at Inhaca Island by the local people and tourists is 95560 l.

Color of water/village/well

Generally, the water taken from the taps is clean and uncolored, while that taken from wells without cover, or artisanal is yellowish, and sometimes dirty (table 5). This fact is related to the protection of water from external contamination. Some of covered wells present yellowish water, because are not very well protected.

Smell

Water taken from the majority of the wells did not have smell.

Table 5. Color of the water

Village	Well	Color of water	
A. **Nhaquene**	1	Uncolored	Tap
	2	Yellowish	Uncovered well
	3	Yellowish	Covered well
	4	Yellowish	Uncovered well
	5	Uncolored	Tap
	6	Uncolored	Tap
	7	Yellowish	Covered well
	8	Uncolored	Tap
	9	Uncolored	Tap
B. **Ribjene**	1	Uncolored	Tap
	2	Uncolored	Tap
	3	Uncolored	Covered well
	4	Yellowish	Tap
	5	Yellowish	Uncovered well
	6	Yellowish	Covered well
	7	Yellowish	Tap
	8	Uncolored	Tap
C. **Inguane**	1	Uncolored	Tap
	2	Yellowish	Uncovered well
	3	Yellowish	Uncovered well
	4	Uncolored	Covered well
	5	Uncolored	Tap
	6	Yellowish	Uncovered well
	7	Uncolored	Covered well

Water Use

Water is used for drinking, cooking, shower and laundry. Almost nothing is used for watering the plants, since the main type of agriculture practiced in the island relies only on rainwater. The majority of water is used for laundry (42%), followed by shower (29%), cooking and drinking (20%), and washing dishes and others (9%). These results are somewhat different from those of Unguja Island (Zanzibar), where laundry and shower had the same value (30%).

Salinity

It ranges from 0.5 to 4 ppt, although the majority is between 1 to 2 ppt, that is, is suitable for human consumption. Higher values are found near the coast.

Chemical content

The level of chemicals (NO_3^-, NO_2^-, NH_4^+ and Cl^-) in the three zones can be seen in appendix 1. All values are far below the acceptable limits for human consumption, namely, 50 mg/l NO_3^-, 3 mg NO_2^-. Regarding Cl^- , there are two wells in Nhaquene with higher values than the maximum acceptable (250 mg/l) (WHO, 1994, 1985, 1984). This means that regarding the chemical component, the groundwater of Inhaca Island is potable.

There are no significant differences among the three villages concerning the chemical composition of the water (appendix 2). Taking in account that these villages are separated by more than 5-8 Km, and that Ribjeni has more inhabitants than the other two villages (almost double), similarity in chemical composition may indicate that human impact in chemical terms is low.

Level of bacterial contamination of water

From 31 wells sampled in the three villages, 83.9% (appendix 1) showed values higher than 3 fecal coliforms/100 ml, the maximum acceptable value (WHO, 1994, 1985, 1984), that is, only 16.1% of the wells have suitable water for human consumption. This result shows that one of the main human influences on the groundwater is bacterial contamination. The reason for this can be because the majority of families in the island do not have toilets, defecating in the bush and on the mangroves and inadequate location of the wells in relation to the toilets.

There are significant differences in contamination with fecal coliforms among the three villages, being less in Ribjene (appendix 2). This is because Ribjene is the economic center of the island, being more urbanised, with latrine for every house, in opposite to the other two villages which a more rural. The existence of latrines diminishes significantly the fecal contamination of the water, through runoff.

Conclusions

- The amount of water used by the population is the same throughout the island and is estimated at 16.2 l/person/day, excluding the Hotel.
- Water is used mainly for laundry, shower, cooking and drinking.
- Chemical content of the water is still far below the maximum acceptable levels.
- Chemical contamination is very low at Inhaca Island.
- There is a great bacterial contamination of the wells.
- One of the main human impacts on groundwater is fecal contamination.

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Appendix 1: Chemical and bacteriological data

Village	Well N°	Total Coliform	Fecal Coliform	NO_3^-	NO_2^-	Cl^-	NH_4^+
Nhaquene	1	≥2400	≥2400	13.1	0.12	46	0.81
	2	≥2400	120	6.8	0.12	46	2.96
	3	120	64	11.6	0.05	158.2	0.5
	4	≥2400	≥2400	12.86	0.37	773.6	0.57
	5	≥2400	≥2400	0.8	0.04	91.1	0.56
	6	75		2.35	0.04	245.8	0.15
	7	≥2400	1100	4.75	0.11	498.6	0.49
	8	≥2400	20	0.24	0.05	61.42	0.95
	9	≥2400	1100	0.24	0.05	61.42	0.62
	10	≥2400	28	0.36	<0.03	122.68	0.89
	11	23	<3	0.15	<0.03	61.43	0.36
Ribjeni	1	≥8	<3	5.4	0.07	100.3	0.28
	2	≥2400	23	6.6	<0.03	139.5	0.34
	3	≥2400	75	23.8	0.08	110.6	0.37
	4	≥2400	150	9.5	<0.03	148	0.34
	5	<3	-	1.12	<0.03	56	0.08
	6	≥2400	460	4.09	0.11	60.1	0.08
	7	<3	-	2.86	0.05	63.6	0.24
	8	460	240	-	-	-	-
	9	23	4	0.04	<0.03	122.8	0.73
	10	≥2400	1100	0.13	0.12	131.6	1.44
	11	42	<3	0.28	<0.03	61.42	0.41
	12	<7	<3	0.30	0.04	122.8	1.28
Inguane	1	≥2400	≥2400	5.4	<0.03	46	0.82
	2	9	4	2.4	<0.03	168.4	0.21
	3	≥2400	1100	15	0.12	103.7	0.32
	4	≥2400	1100	15.2	0.07	148.0	0.79
	5	≥2400	210	4.6	0.02	137.8	0.66
	6	<3	-	0.78	<0.03	67	0.31
	7	≥2400	23	9	0.14	160	0.98
	8	15	<3	3.64	<0.03	56.7	0.08
	9	75	23	2.64	<0.03	158.1	0.32
	10	≥2400	93	0.13	0.87	49.16	1.84
	11	<3	-	0.90	<0.03	133.4	0.43
	12	≥2400	210	0.26	0.26	140.38	1.22
	13	≥2400	≥2400	0.32	<0.03	289.5	1.66

Appendix 2: Results of Kruskal-Wallis tests

Kruskal-Wallis for NO_3

Wells	Nhaquene	Ribjeni	Inguane
1	13.1 (33)	5.4 (25.5)	5.4 (25.5)
2	6.8 (28)	6.6 (27)	2.4 (17)
3	11.6 (31)	23.8 (36)	15 (34)
4	12.86 (32)	9.5 (30)	15.2 (35)
5	0.8 (13)	1.12 (15)	4.6 (23)
6	2.35 (16)	4.09 (21)	0.78 (12)
7	4.75 (24)	2.86 (19)	9 (29)
8	0.24 (5.5)	4.44 (22)	3.64 (20)
9	0.24 (5.5)	0.04 (1)	2.64 (18)
10	0.36 (11)	0.13 (2.5)	0.13 (2.5)
11	0.15 (4)	0.28 (8)	0.90 (14)
12		0.30 (9)	0.26 (7)
13			0.32 (10)
Average	**4.86**	**4.44**	**4.64**
n	11	12	13
R	203	216	247
R^2	41209	46656	61009
R^2/n	3746.3	3888	4693

$K = [\Sigma(R^2/n) \times 12/N(N+1)] - 3(N+1) = 12327.3 \times 12/36(36+1) - 3(36+1) = 0.057$

$P > 0.05$

NB: The values in the brackets are ranks

Kruskal-Wallis for NO_2

Wells	Nhaquene	Ribjeni	Inguane
1	0.12 (30.5)	0.07 (24.5)	<0.03 (8.5)
2	0.12 (30.5)	<0.03 (8.5)	<0.03 (8.5)
3	0.05 (21.5)	0.08 (26)	0.12 (30.5)
4	0.37 (35)	<0.03 (8.5)	0.07 (24.5)
5	0.04 (17.5)	<0.03 (8.5)	0.02 (1)
6	0.04 (17.5)	0.11 (27.5)	<0.03 (8.5)
7	0.11 (27.5)	0.05 (21.5)	0.14 (33)
8	0.05 (21.5)	0.04 (17.5)	<0.03 (8.5)
9	0.05 (21.5)	<0.03 (8.5)	<0.03 (8.5)
10	<0.03 (8.5)	0.12 (30.5)	0.87 (36)
11	<0.03 (8.5)	<0.03 (8.5)	<0.03 (8.5)
12		0.04 (17.5)	0.26 (34)
13			<0.03 (8.5)
n	11	12	13
R	240	207.5	217.5
R^2	57600	43056.25	47306.25
R^2/n	5236.4	3588.0	3638.9

$K = [\Sigma(R^2/n) \times 12/N(N+1)] - 3(N+1) = 1.28$

$p > 0.05$

NB: The values in the brackets are ranks

Appendix 2 Cont.

Kruskal-Wallis for Cl^-

Wells	Nhaquene	Ribjeni	Inguane
1	46 (3)	100.3 (16)	46 (3)
2	46 (3)	139.5 (25)	168.4 (32)
3	158.2 (30)	110.6 (18)	103.7 (17)
4	773.6 (36)	148 (27.5)	148.0 (27.5)
5	91.1 (15)	56 (6)	137.8 (24)
6	245.8 (33)	60.1 (8)	67 (14)
7	498.6 (35)	63.6 (13)	160 (31)
8	61.42 (10)	36.6 (1)	56.7 (7)
9	61.42 (10)	122.8 (20.5)	158.1 (29)
10	122.68 (19)	131.6 (22)	49.16 (5)
11	61.43 (12)	61.42 (10)	133.4 (23)
12		122.8 (20.5)	140.38 (26)
13			289.5 (34)
n	11	12	13
R	206	187.5	262.5
R^2	42436	35156.25	68906.25
R^2/n	3857.8	2929.7	5300.5

$K = [\Sigma(R^2/n) \times 12/N(N + 1)] - 3(N + 1) = -2.0173$

$p > 0.05$

NB: The values in the brackets are ranks

Kruskal-Wallis for faecal coliforms

Wells	Nhaquene	Ribjeni	Inguane
1	≥2400 (29)	<3 (3)	≥2400 (29)
2	120 (16)	23 (10)	4 (6.5)
3	64 (13)	75 (14)	1100 (24)
4	≥2400 (29)	150 (17)	1100 (24)
5	≥2400 (29)	460 (21)	210 (18.5)
6	1100 (24)	240 (20)	23 (10)
7	20 (8)	4 (6.5)	<3 (3)
8	1100 (24)	1100 (24)	23 (10)
9	28 (12)	<3 (3)	93 (15)
10	<3 (3)	<3 (3)	210 (18.5)
11			≥2400 (29)
12			
13			
n	10	10	11
R	187	121.5	187.5
R^2	34969	14762.25	35156.25
R^2/n	3496.9	1476.225	3515.625

$K = [\Sigma(R^2/n) \times 12/N(N + 1)] - 3(N + 1) = 6.69$

$p < 0.05$

NB: The values in the brackets are ranks

Ecology of sea-grass communities in South Bay, at Inhaca Island, Southern Mozambique

Angelina Martins and Domingos Gove
Eduardo Mondlane University, Maputo, Mozambique

Introduction

Sea-grasses are superior plants which are adapted to marine environment, and having various important functions in the surface coastal waters of tropical and temperate zones, especially in relation to productivity (Dawes, 1981).

There are 12 genus and approximately 60 species of sea-grasses distributed on surface waters along almost all coasts of the world (Warne, 1994). Twelve of those species occur in Mozambique, which correspond to 21% of the world biodiversity (Bandeira, 1995). Inhaca Island has 9 sea-grass species, namely, *Cymodocea rotundata, C. serrulata, Halodule uninervis, H. wrightii, Halophila ovalis, Syringodium isoetifolium, Thalassia hemprichii, Thalassodendron ciliatum* and *Zostera capensis* (Bandeira, 1995).

According to Warne (1994), and Lent and Verschuure (1994), environmental factors like the light, availability of nutrients, temperature, salinity and type of sediments influence the distribution, growth and biomass of sea-grasses.

The present study has a general aim of verifying the effect of variation of environmental factors, in part due to groundwater discharge, on the sea-grass communities of South Bay.

Objectives

- Determine and compare the percentage cover, species composition and biomass of sea-grasses of South Bay within areas with and without groundwater effects.
- Determine the seasonal variation of the biomass of sea-grasses
- Determine the growth rate of the most abundant species.
- Relate the parameters above mentioned with environmental factors, namely salinity, organic matter, temperature and soil texture.

Study area

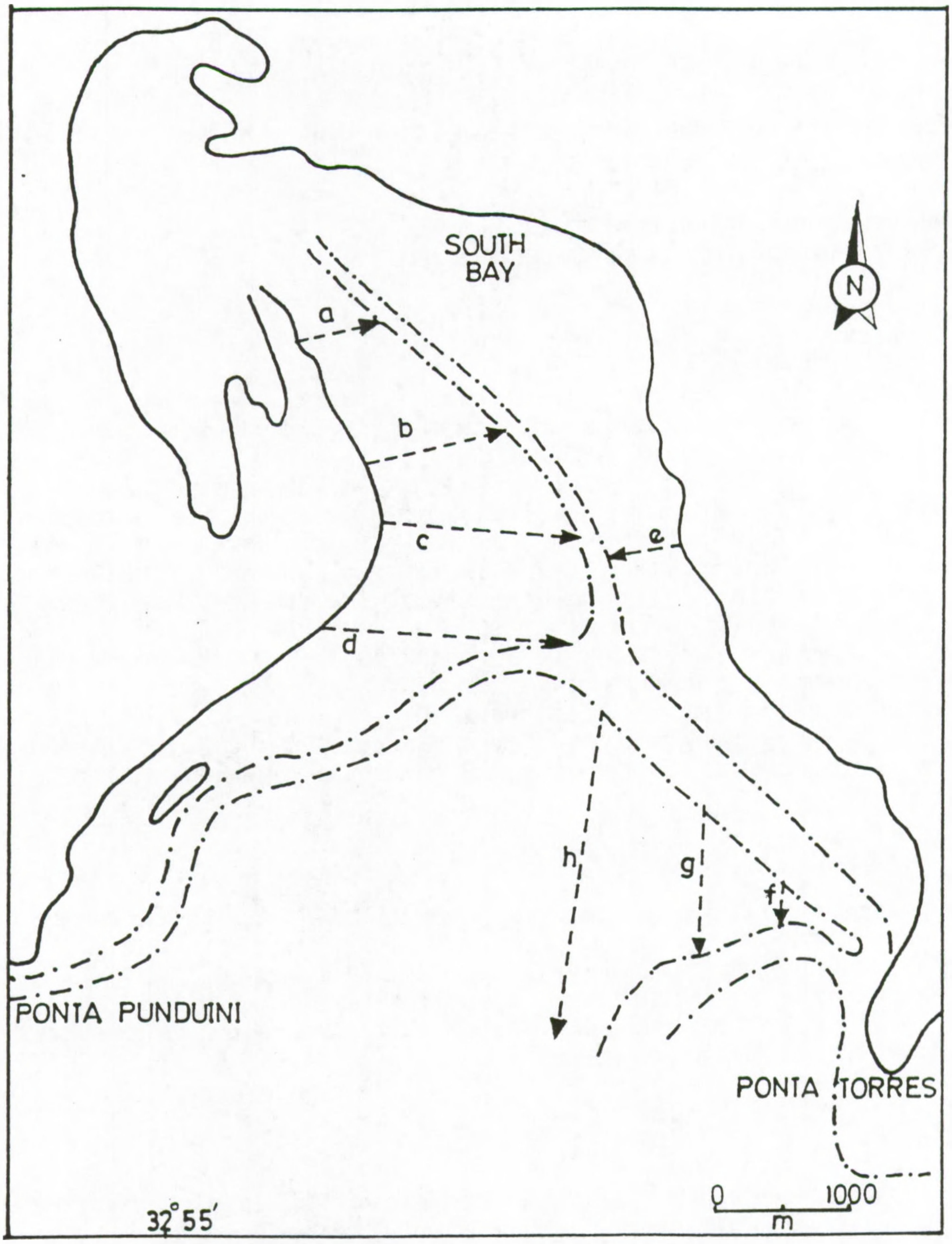
Inhaca Island, with an area of 42 Km^2, is part of the eastern barrier, together with Machangulo Peninsula, that separates Maputo bay from the Indian Ocean.

Mean water temperature at Inhaca Island is 24.5 ºC, with a slight increase on inter tidal flats, during low tide (André, 1995). Salinity varies from 30 p.p.t. to 35.5 p.p.t. (Macnae and Kalk, 1958).

Inter-tidal zone around Inhaca Island is characterised by two kind of beaches with implications on the distribution of sea-grasses (Bandeira, 1991). The eastern side is exposed to open sea, and with a rapid increase of depth near the coastline, while the western side, facing Maputo Bay is relatively protected with a smooth topography (Bandeira, 1991). According to Bandeira (1991) and Kalk (1995), sea-grasses are mainly confined to the western side of the island.

In the southern side of the island there is a long bay, called South Bay, with a triangular form, and with 10 Km of length and 6 Km of width (Kalk, 1995). This study was carried out in three zones of South Bay, at Inhaca Island (fig. 1). Zones 1 and 2 are influenced by the groundwater, coming respectively from the western and eastern sand dunes surrounding the South Bay

Fig. 1. South Bay. Location of study area.



LEGEND

a---Transect	1	ZONE 1
b---Transect	2	ZONE 1
c---Transect	3	ZONE 1
d---Transect	4	ZONE 1
e---Transect	5	→ ZONE 2
f---Transect	6	ZONE 3
g--Transect	7	ZONE 3
h--Transect	8	ZONE 3

Canal

(according to the groundwater outflow map). These two zones are separated by a small channel. Zone 3 is surrounded by channels and is suffering a high influence of oceanic waters coming from the strait of Santa Maria. According to Bandeira (1991), the predominant species in the three zones are the following: zone 1 - *Thalassia hemprichii, Halodule wrightii* and *Zostera capensis;* zone 2 - *T. hemprichii, H. wrightii*; and zone 3 - *Z. capensis.*

Methodology

Sampling

Percentage cover and species composition

Eight transects were fixed in the three zones between September and December 1997. In each 30m along transect, quadrants of 1m2 were installed, in which percentage cover, species composition, water salinity and temperature were determined. It was recorded also if the sea grasses were immersed or emerged. Percentage cover was estimated using the point transect methodology (Bonhan, 1989):

% Cover = sum of positives x 100/total number of points used

Biomass and growth rate

Sampling started in January 1998 until December 1998, and samples were collected bimonthly. Ten (10) samples per zone were taken randomly every two months using a quadrant of 0.2m X 0.2m (Ott, 1990). In the lab, the material was rinsed, treated with diluted HCl to remove calcareous and epiphytes. The different species present in each sample were separated, and then for each species the material was left 48 hours at 80°C, weighted and then left 2 hours at 550°C to obtain ash free dry weight (Ott, 1990).

For growth rate, five (5) shoots for each of the most abundant species (*Halodule wrightii and Zostera capensis*) were marked bimonthly with varnish (Dennison, 1990) at the spring tides. At the end of spring tides, the plants were collected to the lab, where a separation of the material produced before and after marking per shoot was made for measuring and weighting. It was not possible to study the growth of *Thalassia hemprichii*, although it was also abundant, since the marks were not remaining in the shoots. Furthermore, it was not possible to separate this study by the three zones, since the number of marks recovered by zone was insignificant. The growth rate was expressed, using the following formulae:

a) mm/shoot/day = length produced after marking / time needed to produce that length

b) mg/shoot/day = weight of material produced after marking / time needed to produce that material

Specific growth rate (SGR) is the percentage of dry weight produced after marking (DW) by the total weight of the shoot (TW) X time of experience - TE (Larkum *et al*, 1989):

SGR = DWx100/TWxTE

Environmental parameters

- Water salinity and temperature

Water salinity and temperature were measured along transects, using 30 meters intervals. In order to study the seasonal variation of these two parameters, 10 random measurements per zone were made per two months. Salinity was measured using a refractometer. Temperature was obtained using a thermometer.

- Organic matter and sediment grain size

Five sediment samples were taken randomly per zone. Water content was determined by the difference between fresh and dry weight. Organic matter was obtained using the method of combustion. Ten grams from each sediment sample were weighted, and then kept 3 hours at 105 ºC; after that, the sample was weighted, and then kept 2 hours at 550 ºC, and weighted again (Stewart and Allen, 1989). For grain size determination, samples were dried at 105 ºC, for 24 hours, and then separated by sieves of different sizes (6 μm to 2 mm). The different size material was weighted. The sediment retained by 6 μm sieve was considered sand, and the one that passed, was considered lime and silt.

Data Analysis

The relation between water salinity and percentage cover for each of the species along transects was tested using Linear Regression. This test was also used to verify the relation between total biomass with water temperature and salinity.

ANOVA single factor was used to compare the mean temperature and salinity in the eight transects, and to compare the biomass of each of the species of sea grass in the three zones.

ANOVA two factors was used to compare mean monthly biomass in the three zones, and mean monthly water temperature and salinity in the three zones; and also to compare the growth of the species along the year.

Results

Species composition and percentage cover

Zone 1

Zone 1 was covered by the following species: *Halodule wrightii*, *Zostera capensis*, *Thalassia hemprichii*, *Cymodocea rotundata*, *Syringodium isoetifolium* and *Halophila ovalis*. The most important species were *Halodule wrightii* and *Zostera capensis* (Appendix 1 – 4).

According to the linear regression test, the relation between salinity and percentage cover in transect two was not significant for *Z. capensis*, p=0.55, r^2=0.014; *T. hemprichii*, p=0.33, r^2=0.038, and *C. rotundata*, p=0.572, r^2=0.013, but it was for *H. wrightii*, p=0.001, r^2=0.33.

On transect three, salinity showed relation with *S. isoetifolium*, p=0.01, r^2=0.30, but no relation was found with *Z. capensis*, p=0.13, r^2=0.116, and *H. wrightii*, p=0.65, r^2=0.011.

Salinity and percentage cover did not show any relation in transect four *(Z. capensis*, p=0.05, r^2=0.154; *H. wrightii*, p=0.1, r^2=0.11; H. *ovalis*, p=0.67, r^2=0.008).

Zone 2

The following species were observed in Zone 2: *Halodule wrightii*, *Thalassia hemprichii* and *Halophila ovalis*. *H. wrightii* was the predominant species (transect 5, appendix 5).

No relation was obtained between the salinity and percentage cover (*Z. capensis*, p=0.56, r^2=0.034, *H. wrightii*, p=0.09, r^2=0.256, H. *ovalis*, p=0.17, r^2=0.179, *T. hemprichii*, p=0.558, r^2=0.035).

Zone 3

In zone 3, the species observed were *Halodule wrightii*, *Z. capensis*, *Thalassia hemprichii*, *Cymodocea rotundata*, *Halophila ovalis* and *H. uninervis*. The most important species were *H. wrightii* and *Z. capensis* (transects 6, 7 and 8, appendix 6,7,8).

On transect 6, salinity was related to *H. wrightii*, p=0.03, r^2=0.55, but it was not for *Z. capensis*, p=0.22, r^2=0.238; *C. rotundata*, p=0.48, r^2=0.09; and *H. ovalis*, p=0.35, r^2=0.145.

On transect 7, salinity was related to *T. hemprichii*, p=0.03, r^2=0.38, but not to *Z. capensis*, p=0.78, r^2=0.007; *C. rotundata*, p=0.19, r^2=0.136; *H. uninervis*, p=0.96, r^2=0.0003; and *H. wrightii*, p=0.20, r^2=0.133.

Percentage cover was not related to salinity on transect 8 (*Z. capensis*, p=0.22, r^2=0.06; *H. wrightii*, p=0.17, r^2=0.08; H. *ovalis*, p=0.63, r^2=0.009; T. *hemprichii*, p=0.6, r^2=0.011).

According to the linear regression test, in the three zones, salinity, which could be an indication of groundwater, was not related to the percentage cover of the sea-grass species, apart from *H. wrightii*, which sometimes showed some correlation.

<u>Comparison of species composition among the three zones</u>

Results of species composition in the three zones agree with Bandeira (1991). Some small differences are due to the differences in the methods used, where Bandeira used only direct observation method, while in this study we used a quantification method.

From table 1, it is obvious that there is no difference in species composition among the three zones, except the occurrence of *S. isoetifolium* only in zone 1, *H. uninervis* in zone 3, and the absence of *C. rotundata* in zone 2. In relation to zone 2 it is important to note that it was only constituted by one transect, in opposite to the other two, which had equal or more than three transects.

Apart from that, the most important species, in terms of percentage cover were the same in the three zones, namely *H. wrightii* and *Z. capensis*.

From these results, it is possible to say that the potencial effect of the differences in groundwater influence between zones 1 - 2, and zone 3 it is not reflected in species composition and percentage cover.

Table 1. Species composition in the three zones. Underlined species are the most important in terms of percentage cover.

Zone 1	**Zone 2**	**Zone 3**
Cymodocea rotundata	----------------	*Cymodocea rotundata*
Halophila ovalis	*Halophila ovalis*	*Halophila ovalis*
----------------	----------------	*Halodule uninervis*
<u>Halodule wrightii</u>	***<u>Halodule wrightii</u>***	***<u>Halodule wrightii</u>***
Syringodium isoetifolium	----------------	----------------
Thalassia hemprichii	*Thalassia hemprichii*	*Thalassia hemprichii*
<u>Zostera capensis</u>	*Zostera capensis*	***<u>Zostera capensis</u>***

Salinity and temperature variation along the transects

According to *ANOVA single factor test*, there were significant differences in the salinity of transects (p=1.3 X 10^{-10}). Higher values of salinity were observed in transects 1 and 3, both in zone 1. Again, *ANOVA* test showed differences in temperature (p=3.3 X 10^{-38}). These differences can be due to differences in exposure period (higher in zone 1) and sampling time, which varied during the study, rather than the impact of groundwater.

Table 2. Mean salinity and temperature along the eight transects

Transect	Salinity (p.p.t)	Temperature (°C)
1	37.3 ± 1.4	30.0 ± 0.0
2	35.5 ± 1.6	31.8 ± 0.5
3	37.5 ± 1.0	30.7 ± 0.7
4	35.5 ± 1.2	32.0 ± 1.1
5	36.3 ± 1.4	34.8 ± 8.4
6	35.4 ± 0.5	28.3 ± 1.0
7	35.5 ± 0.5	30.1 ± 1.9
8	35.1 ± 0.4	29.5 ± 0.6

Biomass

a) The most important species in zone 1 were *H. wrightii* and *T. hemprichii* (fig.3a-f). According to *ANOVA single factor*, there are no significant differences among the biomass of the various species in zone 1, all the year round (p=0.963).

b) In zone 2, the predominant species was *T. hemprichii* followed by *H. wrightii* (fig.4a-f). There are no significant differences in the biomass of the different species during the year (p=0.881, *ANOVA single factor*).

c) *T. hemprichii* was the most important species, followed by *Z. capensis* (fig. 5a-f). There are no significant differences among the biomass of the various species, during the year (p=0.626, *ANOVA single factor*).

<u>Seasonal variation of total biomass in the three zones</u>

Higher values of biomass in the three zones were observed in winter (May-October) – fig. 6-8, which correspond to the dry season (Kalk, 1995); however, according to *ANOVA two factors*, no significant differences exist among the total biomass along the year in the three zones (p=0.624). Furthermore, there are no differences among the three zones (p=0.105).

The absence of seasonal variation in the biomass of the different species agrees with the results of Coles *et al* (1989), who made a study in northwest Australia, with *Cymodocea serrulata*, *Syringodium isoetifolium*, *Zostera capricorni*, *Halophila ovalis* and *Halodule uninervis*. The results also agree with Walker and McComb (1988), who studied in Australia, species from temperate zones, namely *Amphibolis antartica* and *Pasidonia australis*; and Brouns (1985 and 1987) in Papua New Guinea, with *T. Hemprichii*, *C. serrulata*, *C. rotundata*, *H. uninervis* and *S. isoetifolium*. According to Hillman *et al* (1989), these results are due to the small variations of environmental factors in tropical and sub-tropical zones, especially light and temperature. In fact, our salinity values did not show a variation along the year. This implies that the potencial seasonal variation of discharges of groundwater in this area, due to variation of rainfall was not reflected in the salinity. Although water temperature was different (table 3), taking in account that the amplitude of the difference is narrow, probably it was not enough to be reflected in variation of biomass of the different species.

The absence of differences in the total biomass among the three zones is because there are no differences in the environmental parameters, namely salinity (which could indirectly indicate groundwater discharge), temperature (table 3), light and sediment type (table 4).

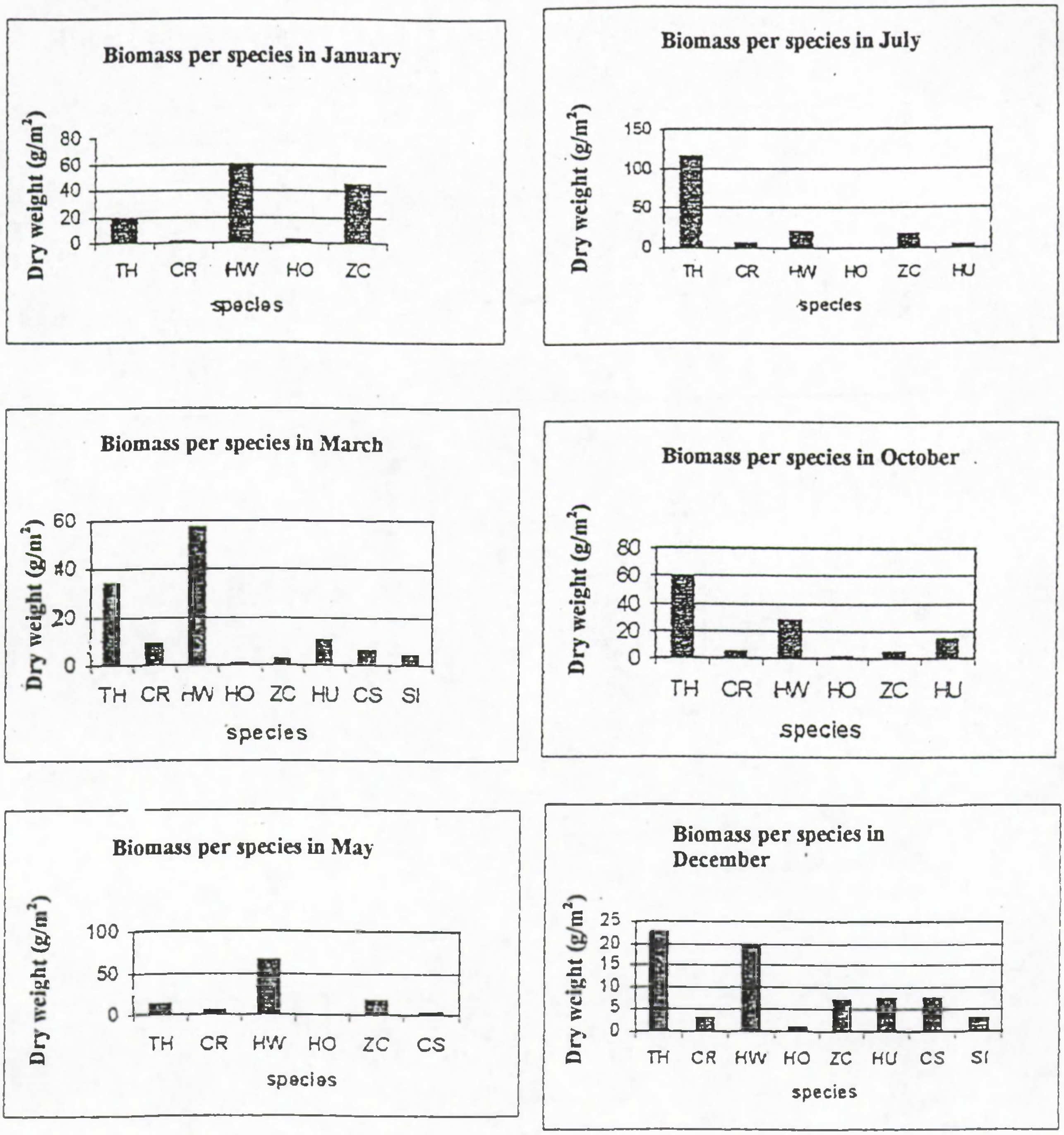


Fig.3 (a-f). Biomass in zone 1 along the year

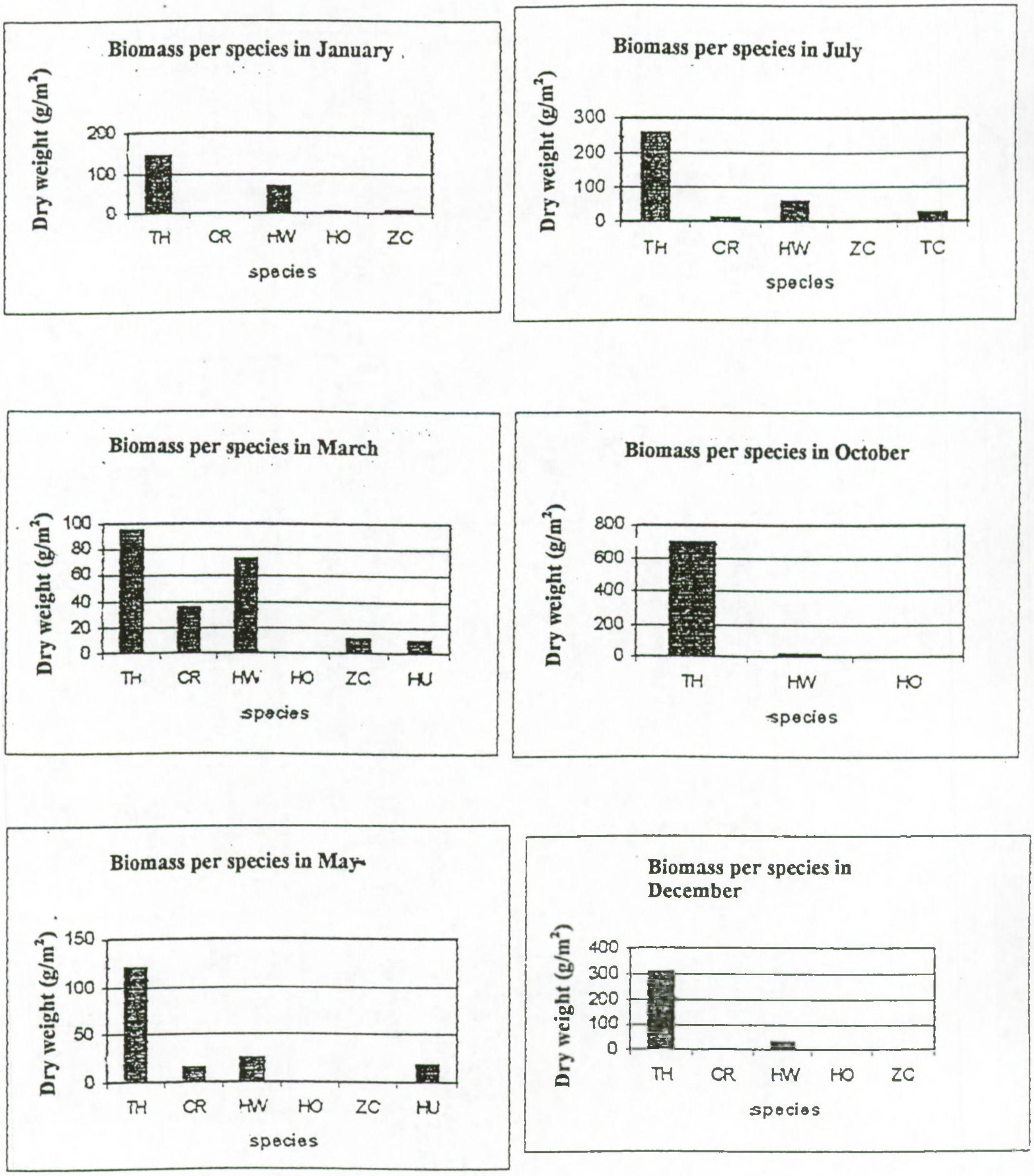


Fig.4 (a-f). Biomass in zone 2 along the year

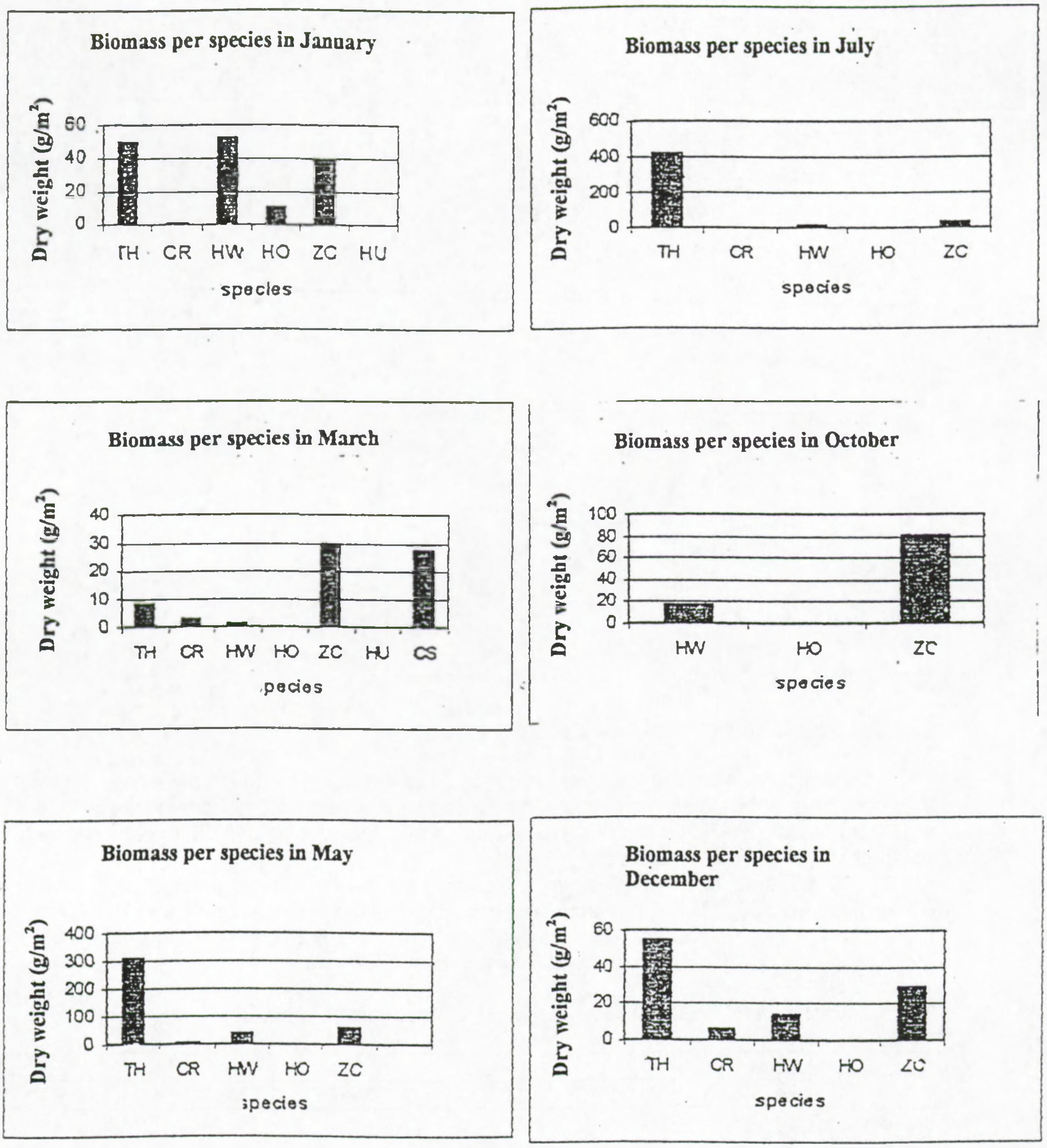


Fig. 5 (a-f). Biomass in zone 3 along the year

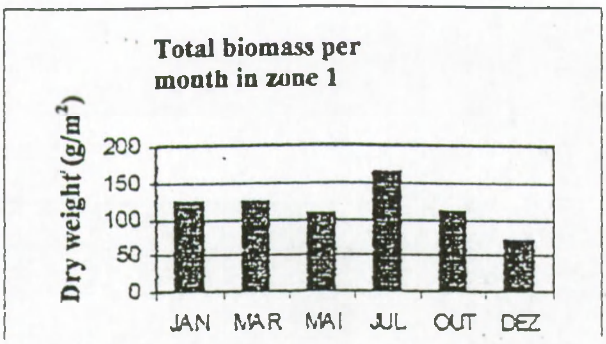


Fig. 6. Total biomass in zone 1

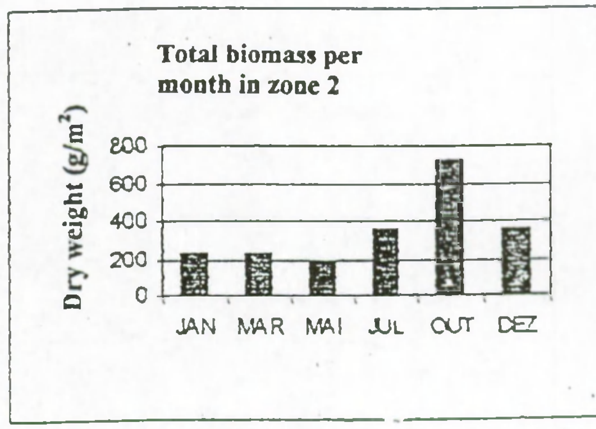


Fig. 7. Total biomass in zone 2

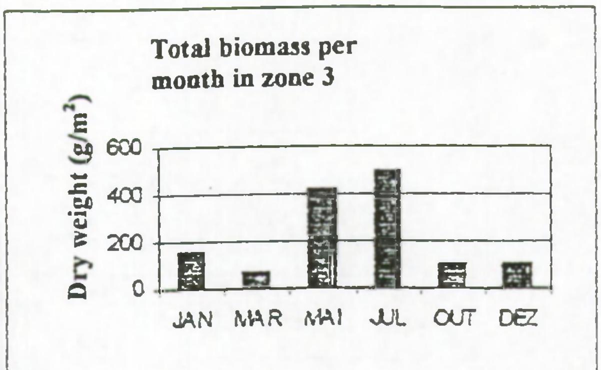


Fig. 8. Total biomass in zone 3

Environmental parameters

Temperature and salinity variation in the three zones

Water temperature showed a decrease during winter (May-July) in the three zones, reflecting the decrease of atmospheric temperature in this period of the year. Monthly differences were very significant (p=0.003, *ANOVA two factors*), however, the differences among the three zones were not (p=0.106, *ANOVA two factors*).

Salinity did not show a clear pattern in the three zones. There were no significant monthly differences (p=0.133, *ANOVA two factors*), and no significant differences among the three zones (p=0.051, *ANOVA two factors*). This may indicate that there is not a significant difference in groundwater discharge all the year round, and among the three areas.

Table 3. Mean monthly values of temperature and salinity in the three zones

	Zone 1		Zone 2		Zone 3	
	Sal. (‰)	Temp (ºC)	Sal. (‰)	Temp (ºC)	Sal. (‰)	Temp (ºC)
January	36.6	34	35	33	35	33
March	35	33	34.5	33	35	31
May	35	31	35	28	34.5	29
July	36	31	35	28	35	29
October	37.1	33	34.5	30	35	33
December	34	33	33	33	35	34
Annual Mean	**35.6**	**32.5**	**34.5**	**30.8**	**34.9**	**31.5**

Organic matter and grain size

Sand is the predominant grain size in the sediment of the three zones. From table 4, there is no difference among the three zones. This result agrees with Torre de Vale (1996) in Inhaca Island, and Moriarty and Boom (1989) in Moreton Bay, Australia. The low levels of organic matter can be explained by rapid decomposition of roots and rhizomes and complete exudation of organic matter (Moriarty and Boom, 1989).

Table 4. Mean percentages of sand, lime + silt and organic matter

	Sand	Lime + Silt	Organic matter
Zone 1	99.2	0.85	1.5
Zone 2	99.1	0.90	1.5
Zone 3	99.2	0.80	0.14

Relation between environmental parameters and the biomass

According to Linear Regression test, there is no any relation between the salinity and the biomass in the three zones (zone 1: r^2=0.29, p=0.27; zone 2: r^2=0.166, p=0.423; zone 3: r^2=0.268, p=0.293). There is, however, a relation between water temperature and the biomass (zone 1: r^2=0.121, p=0.0351; zone 2: r^2= 0.15, p=0.00815; zone 3: r^2=0.757, p=0.06).

According to Hillman *et al* (1989), water temperature is one of the most important factors, which influence productivity of sea-grasses.

Growth rate

Specific growth rate

The values are similar all the year round for each of the two species (p=0.26, ANOVA two factors), however, specific growth rate of *H. wrightii* is higher than that of *Z. capensis* (p=0.0003, ANOVA two factors).

Specific growth rate of *Z. capensis* is higher than that of *Z. marina* and *Z. capricorni* obtained in Australia (Hillman *et al*, 1989), and the results of *H. wrightii* agree with those of Australia for the same species.

According to Hillman, et al (1989) there are two categories of sea-grasses: the smaller colonizers species, frequently with a specific growth rate higher than 4%/day, and the bigger climax species, usually less than 4%/day. In this context, both *H. wrightii* and *Z. capensis* can be considered colonizers, however, the higher specific growth rate of *H. wrightii* may indicate that it is the first to colonize the area. This agrees with Phillips and Menez (1988) and Warne (1994) who consider the genus *Halodule* to be composed by pioneer species, tolerant to a great variation of temperature, salinity and substrate.

The absence of seasonal variation indicates that the variation of environmental factors is also not significant along the year.

Table 5. Mean percentage of specific growth rate.

Species	**January (%/day)**	**March (%/day)**	**May (%/day)**	**July (%/day)**	**October (%/day)**	**December (%/day)**
H. wrightii	5.9±2.1	5.8±0.9	6.0±1.5	5.7±2.1	6.0±2.6	6.4±2.5
Z. capensis	4.5±1.9	3.9±1.5	4.4±2.7	4.3±2.0	5.2±1.9	4.5±1.8

Seasonal variation of length growth

Z. capensis has a higher length growth than *H. wrightii* (p=0.04, *ANOVA two factors*). Seasonal variation of length growth of the two species is not significant (p=0.14, *ANOVA two factors*).

Results of length growth are similar to that from *Podosinia oceanica* (Bay, 1984) and *T. hemprichii* (Brouns, 1985). Higher values of length growth of *Z. capensis* compared to *H. wrightii* can be because *Z. capensis* is living in a relatively more stressful conditions.

The stability of growth all the year round can be due to the stability of environmental conditions.

Conclusions

The main conclusion is that no groundwater effect was detected both in the environmental parameters and in the species composition, percentage cover, biomass and growth of sea-grass from South Bay, Inhaca Island. The reason for this can be due to the small area of the island, which implies a small catchment area and small reservoir of groundwater, so that the outflow is not relevant in the physical and biological components of the South Bay.

Other conclusions are the following:

Species composition and percentage cover

- Species composition is the same in the three areas.
- The most important species in terms of percentage cover in South Bay are *H. wrightii* and *Z. capensis.*
- Salinity is not related to species composition and percentage cover.

Biomass

- There is no seasonal variation of biomass in South Bay
- Biomass values are the same for the whole South Bay

Environmental parameters

- Values of environmental parameters (Water temperature and salinity, and grain size) are similar in the whole South Bay.
- Water salinity does not vary all the year round.
- Water temperature decreases in winter and increases in summer.

Growth rate

- Specific growth rate of *H. wrightii* is higher that that of *Z. capensis.*
- Specific growth rate does not vary seasonally.

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Appendix 1. Transect one

On transect one no sea grass was found. Along of the transect, salinity and temperature were stable.

Distance from the coast (m)	Species composition and percentage of cover	Salinity	Temperature	Submerged or Emerged
0	-	40	30	Emerged
30	-	37	30	Emerged
60	-	37	30	Emerged
90	-	37	30	Emerged
120	-	37	30	Emerged
150	-	36	30	Emerged

Appendix 2. Transect two

No sea grass was found in the first meters of transect two. From 180 m, the dominant species was *Halodule wrightii*, which sometimes was associated with *Zostera capensis. Thalassia hemprichii* was observed around 510 m from the coast associated with the other two previously mentioned species. *T. hemprichii* was associated with *Cymodocea rotundata* and a small population of *H. wrightii*, at the end of transect. Salinity was high in areas without vegetation. Near the coast, the values of salinity were slightly lower in vegetated areas. Water temperature was constant along of transect.

Distance from the coast (m)	Species composition and percentage of cover	Salinity	Temperature	Submerged or Emerged
0	-	40	31	Submerged
30	-	35	31	Submerged
60	-	40	31	Submerged
90	-	37	31	Submerged
120	-	38	32	Submerged
150	-	37	32	Submerged
180	Halodule wrightii-76.2	34	32	Submerged
210	H.wrightii-76.2	35	32	Submerged
240	H.wrightii-90.5	34	31.5	Submerged
270	H.wrightii-85.7	34	33	Submerged
300	H.wrightii-57.1	35	32	Submerged
330	H.wrightii-57.1 Zostera capensis-14.3	35	31.5	Submerged
360	H.wrightii-57.1 Z.capensis-14.3	34	31.5	Submerged
390	H.wrightii-57.1 Z.capensis-28	36	32	Submerged
420	Thalassia hemprichii-66.7	35	32	Submerged
450	H.wrightii-80.9	34	32	Submerged
480	H.wrightii-76.2 Z.capensis-4.8	36.	32	Submerged
510	H.wrightii-42.9 Z.capensis-14.3 T.hemprichii-33	35	32	Submerged
540	H.wrightii-47.6 Z.capensis-23.8 T.hemprichii-28.6	35	32	Submerged
570	H.wrightii-9.5 Z.capensis-57.1	35	31	Submerged
600	H.wrightii-23.8 Zostera capensis-4.8 T.hemprichii-33.3	35	32	Submerged
630	H.wrightii-57.1 T.hemprichii-28.6	35	32	Submerged
660	H.wrightii-14.3	35	32	Submerged
690	H.wrightii-38.1 Thalassia hemprichii-4.8	35	32	Submerged
720	H.wrightii-33.3 Cymodocea rotundata-28.6 T.hemprichii-23.8	35	31	Submerged
750	C.rotundata-19.4 T.hemprichii-19.4	35	31.5	Submerged
780	H.wrightii-9.5 C.rotundata-19 T.hemprichii-52.4	35	31	Submerged

Appendix 3. Transect three

The first 420 m from the coast, in the transect three, showed no vegetation, which was only observed from 450 m and was composed by *Zostera capensis* and *Halodule wrightii.* Along of the transect these two species alternate with intervals without vegetation. The first meters of the channel, at the end of transect, are covered by *Syringodium isoetifolium.* Salinity was high in areas without vegetation, and lowering slightly in vegetated areas.

Distance from the coast (m)	Species composition and percentage of cover	Salinity	Temperature	Submerged or Emerged
0	-	40	32	Emerged
30	-	37	31	Emerged
60	-	38	31.5	Emerged
90	-	38	31	Emerged
120	-	38	31	Emerged
150	-	38	31.5	Emerged
180	-	39	30.9	Emerged
210	-	38	30.9	Emerged
240	-	38	30.9	Emerged
270	-	37	30.9	Emerged
300	-	38	30	Emerged
330	-	38	30.5	Emerged
360	-	37	30	Submerged
390	-	37	30.1	Submerged
420	-	37	30	Submerged
450	Zostera capensis-76.2	36	30	Submerged
480	-	37	30.5	Emerged
510	Z.capensis-8.9	37	30	Emerged
540	H.wrightii-57.1	37	30	Emerged
570	-	37	30	Emerge
592 (channel)	Syringodium isoetifolium-50	35	30	Emerged

Appendix 4. Transect four

In the transect four, up to 60 m from the coast no sea grass was observed. Between 90 and 270 m *Zostera capensis* was the dominant species, sometimes associated with a small population of *Halodule wrightii*. From 300 to 390m, there was a co-dominance of *Z. capensis* and *H. wrightii*. From 420 m onwards there was no sea grass.

Distance from the coast (m)	Species composition and percentage of cover	Salinity	Temperature	Submerged or Emerged
0	-	40	30	Submerged
30	-	34	30	Submerged
60	-	36	30	Submerged
90	Zostera capensis-85.7	35	30	Submerged
120	Z.capensis-100	35	30	Submerged
150	Z.capensis-66.7 H.wrightii-19	35	31	Submerged
180	Z.capensis-95.2 H.wrightii-4.8	35	32	Submerged
210	Z.capensis-90.4	35	31.5	Submerged
240	Z.capensis-76.2 H.wrightii-9.5	35	32	Submerged
270	Z.capensis-61.9 Halophila ovalis-23.8 H.wrightii-4.8	35	32.5	Submerged
300	Z.capensis-42.9 H.wrightii-47.6	34	32.5	Submerged
330	Z.capensis-66.7 H.wrightii-47.6	35	32.5	Submerged
360	Zostera capensis-47.6 H.wrightii-33.3	35	32	Emerged
390	Z.capensis-42.9 H.wrightii-33.3	35	32	Emerged
420	-	35	33	Emerged
450	-	35	33	Emerged
480	-	35	33	Emerged
510	-	36	33	Emerged
540	-	35	33	Emerged
570	-	36	33	Emerged
600	-	37	33	Emerged
630	-	38	33	Emerged
660	-	36	32	Emerged
690	-	36	32	Emerged
720	-	36	32	Emerged
750	-	35	32	Emerged

Appendix 5. Transect five

In the transect 5, sea grasses were observed from 120 m. From this point up to 240 m there was a dominance of *Thalassia hemprichii. Halodule wrightii* was dominant at 270 m and was associated with *Halophila ovalis, Zostera capensis* and *Thalassia hemprichii.* Salinity was high in the first meters of transect and reduced slightly in the last meters of transect.

Distance from the coast (m)	Species composition and percentage of cover	Salinity	Temperature	Submerged or Emerged
0	-	36	34	Emerged
30	-	35	35	Submerged
60	-	39	35	Emerged
90	-	38	35	Emerged
120	Thalassia hemprichii-19	37	35	Emerged
150	T.hemprichii-71.4	37	34.5	Submerged
180	T.hemprichii-85.7	37	35	Submerged
210	T.hemprichii-80.9	36	35	Submerged
240	T.hemprichii-80.9	36	34	Submerged
270	T.hemprichii-14.3 Halodule wrightii-61.9 Halophila ovalis-9.5 Zostera capensis-9.5	35	35	Submerged
300	H.wrightii-57.1 H.ovalis-4.8 T.hemprichii-38	34	35	Submerged
330	Z. capensis-19 H.ovalis-9.5 H.wrightii-66.7 T.hemprichii-48	36	35	Submerged

Appendix 6. Transect six

On transect six, the first meters from the coast are occupied by three species, *namely C. rotundata*, *H. wrightii* and *H.ovalis*. The first two species are dominant. There is an area without vegetation between 30 and 60 meters. From 90 until 200 meters (the end of transect) occur a new species, *Z. capensis*, and *C. rotundata* no longer occur. The dominant species are *H. wrightii* and *Z. capensis*. Water salinity and temperature were stable along of transect.

Distance from the coast (m)	Species composition	% cover	Salinity	Temperature	Submerged/ emerged
0-30	C.rotundata H.wrightii H.ovalis	42.9 42.9 9.5	35	28	Emerged
30-60	-	-	35	28	Submerged
90-200	H.wrightii H.ovalis Z.capensis	23.8-61.9 19 33.3	35-36	29	Submerged

Appendix 7. Transect seven

On transect seven, the most important species were *Z. capensis* and *H. wrightii*. This last species was more important on the part of transect far from the coast. Salinity was stable, however, water temperature showed some variations, being lower next and far from the coast, but higher in between.

Distance from the coast (m)	Species composition	% cover	Salinity	Temperature	Submerged/ emerged
0-30	Z.capensis H.wrightii	57 42.9	35	27	Emerged
30-270	H.uninervis H.wrightii T.hemprichi, C.rotundata H.ovalis Z.capensis	52.4 9.5-71.4 14.3-61.9 4.8-28.6 9.5-14.3 23.8-81	35-36	29-32	Submerged
300-360	H.wrightii Z.capensis H.ovalis	52.4-100 14.3-28.6 4.8	35	28-29	Emerged

Appendix 8. Transect eight

No vegetation was observed in the first 180 meters of the transect eight. *H. wrightii* and *Z. capensis* were the dominant species and occurred in the all transect. Salinity did not vary. Water temperature was highly variable in the first 180 meters, however, was stable from here until the end of transect.

Distance from the coast (m)	Species composition	% cover	Salinity	Temperature	Submerged/ emerged
0-180	-	-	34-36	20-31	Emerged
210-300	H.wrightii Z.capensis	9.5-42.9 19-28.6	35	29-30	Emerged
330-360	H.wrightii Z.capensis H.ovalis	23.8-33.3 23.8-57.2 4.8	35	29-30	Emerged
390-540	H.wrightii Z.capensis H.ovalis T.hemprichii	23.8-42.7 14.3-52.4 4.8-14.3 9.5-57.1	35	29-30	Emerged
570	-	-	35	30	Emerged
600-720	H.wrightii Z.capensis	4.8-33.3 19-1.4	35	29-30	Emerged
750-780	H.wrightii Z.capensis T.hemprichii	23.8-38.1 9.5-28.6 14.3-28.6	35	29	Submerged

Influence of groundwater on the zooplankton from South Bay, Inhaca Island (Mozambique)

Domingos Z. Gove & Emília Fumo
Eduardo Mondlane University, Maputo, Mozambique

Introduction

Plankton studies are scarce in eastern Africa in general, and in Mozambique, in particular. Several planktonic studies have been carried out at Inhaca Island, namely the studies of Silva (1960), Gove and Cuamba (1989), and Paula *et al*, (1998). All these studies did not include the South Bay. The waters in South Bay are coming from the Indian Ocean through the Santa Maria aperture, and from Maputo Bay, through Ponta Punduini (fig. 1).

The aim of this work was to describe the potencial impact of groundwater in zooplankton communities, namely species composition and abundance.

Objectives

- Determine the species composition.
- Estimate the abundance.
- Relate the biological data with environmental factors.

Material and Methods

Sampling

Three sampling stations were selected along a transect which started in Saco da Inhaca and ended in Ponta Torres, just in the opening to the open ocean (fig. 1). Stations 1 and 2 were located in a place where there was relatively more groundwater influence, while station 3 was in an area with a great oceanic influence.

Samples were taken bimonthly and twice a month, through a year (August 1997- July 98). The timing of sampling was during the neap tides, in order to minimize the dislocation of the water mass and maintain similar physical conditions throughout the collecting stations. The high-tide period was chosen because during low tide the low depth of the water column did not permit effective zooplankton trawls to be performed.

Samples were taken using a small boat with outboard engine. Taking into account the very small depth of the sampling stations, and the great amplitude of tides, no stratification should occur, therefore samples were taken horizontally at subsurface water using two plankton nets (125 and 330 micrometers) with flow meters (Hydro-Bios Kiel) attached. Samples were fixed and preserved in buffered 4% formaldehyde.

Water filtered by the net was calculated through the following formulae: WF = AxVx0.3, where WF, is the water filtered in m^3, A, area of the net mouth, V, the number of flow meter counts, and 0.3 the constant of flow meter calibration. The area of both nets was the same and was $3.14 \times 0.125^2 = 0.0490625$.

Sampling using 125 µm net took 2 minutes and with 330 µm, 5 minutes. Maximum and Sechii depths were obtained using the Sechii disc. Water temperature was measured using thermometer with 0.1 °C precision, and salinity, using a refractometer. (ATAGO; 0-100ppt)

Displacement volume was determined with a conical and graduated jar. Samples were observed under Olympus stereomicroscope. Identification was made only to major taxonomic groups, since local species are still not described.

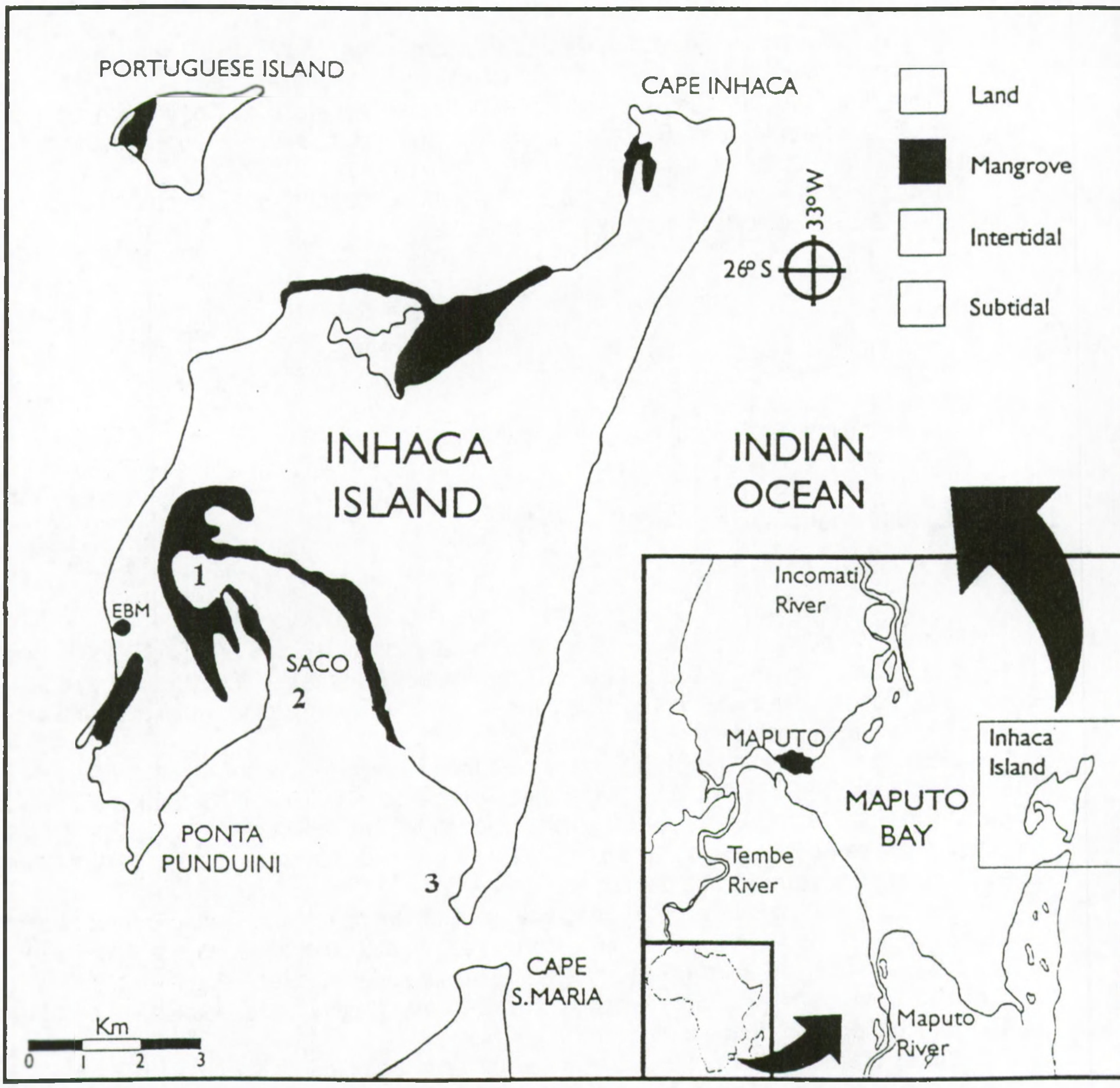

Figure 1. Location of the study area (from Paula et al, 1998).

1: Station 1
2: Station 2
3: Station 3

Data Analysis

Monthly results of water temperature (°C) and salinity (p.p.t.), water depth (m), Sechii depth (m), displacement volume (ml/m^3) and density (n°/m^3) were the average of the two samplings made per month.

Results (hydrographic and abundance) from the three stations were compared using the no parametric test, Kruskal-Wallis. Possible relation between environmental factors and the abundance were tested using the Spearman Rank Correlation Coefficient (r_s).

Results and Discussion

Hydrography

Water temperature varied between 22 to 29.5 °C, with a mean value of 24.6 °C (tab. 1). Lower values occurred in winter (May-July) and higher values in summer (December-March), reflecting the subtropical conditions of Maputo Bay (Paula *et al*, 1998). According to Kruskal-Wallis test, these variations were significant (k=14.25731, p<0.05; tab. 17). Water temperature was similar in all three stations (k=0.2836245, p>0.05; tab. 16).

Water salinity varied between 34 to 38 p.p.t., with an average of 36 p.p.t. Its variation pattern along the year was not obvious, however, there were higher values in March and October compared to December and May (tab. 1). According to Paula *et al* (1998), salinity fluctuations reflect a complex set of factors, namely the circulation in the bay, in which five rivers discharge, and evaporation due to sunshine and dilution due to rainy periods, over a short water column (at sampling sites between 0.45 and 3.60 m). These differences were significant (k=14.017544, p<0.05; tab. 19). Salinity was similar in all the three stations (k=1.0555557, p>0.05; tab. 18).

Similarity of water temperature and salinity in all the three stations indicates that the hydrographic conditions are homogenous in all South Bay. This situation can be because of good mixing of water, and insignificant impact of groundwater discharge in the bay. In relation to this last point, salinity values are purely saline not indicating any freshwater discharge in the area.

Table 1. Hydrographic parameters

Month	Temperature			**Salinity**			Water depth			**Sechii depth**		
	St. 1	**St. 2**	**St. 3**	**St. 1**	**St. 2**	**St. 3**	**St. 1**	**St. 2**	**St. 3**	**St. 1**	**St. 2**	**St. 3**
August	25.2	23.3	22.7	36.5	36	36	0.45	0.48	2.3	0.45	0.48	2.3
October	23.5	23	24.3	37	37.5	38	0.45	0.68	1.4	0.45	0.68	1.4
Decemb.	26.8	26.5	26	36	35	35.5	0.56	0.65	2.8	0.56	0.65	2.3
March	27.8	28.8	29.5	38	38	37.5	0.45	1.03	3.3	0.45	0.9	2.6
May	23.1	22.9	22.5	36	34	35	0.50	1.20	4.10	0.50	1.20	5.0
July	22	22	23	35.9	35.5	35.9	0.45	1.05	3.60	0.45	1.05	3.60
Average	24.7	24.4	24.7	36.6	36.0	36.3	0.48	0.85	2.92	0.48	0.83	2.87

Plankton

<u>Species composition</u>

The groups of zooplankton were similar in all the three stations, except Amphipoda and Radiolarian, which occurred at station 3 (tabs. 4 – 15). Important species were copepods, cirripede, gastropod and brachyura larvae.

Abundance

- Displacement volume

It was similar along the year (for 125 µm, k=5.2333328, p>0.05, tab. 21; for 330 µm, k=2.23508, p>0.05, tab. 23), and among the three stations (for 125 µm, k=1.52, p>0.05, tab. 20; for 330 µm, k=5.7631579, p>0.05, tab. 22). The lack of variations in plankton abundance among the stations shows that the potencial impact of groundwater in this area is not reflected in zooplankton.

Table 2. Displacement volume (ml/m3): 125 µm mesh size plankton net

Month	Station 1	Station 2	Station 3
August 97	1.99	2.21	1.66
October 97	1.87	11.61	2.82
December 97	1.96	5.31	2.54
March 98	6.83	2.43	5.64
May 98	1.25	-	0.36
July 98	0.25	1.38	4.36
Average	2.36	4.59	2.9

Table 3. Displacement volume (ml/m^3): 330 µm mesh size plankton net

Month	Station 1	Station 2	Station 3
August 97	0.41	0.75	0.48
October 97	1.01	1.52	0.37
December 97	5.17	0.34	0.23
March 98	1.34	0.67	0.51
May 98	0.31	4.14	0.31
July 98	0.62	0.52	0.36
Average	1.48	1.32	0.38

- Mean density (n°/m3) of planktonic organisms

a) 125µm mesh size plankton net

There were no differences along the year (k=2.333332, p>0.05, tab. 25), however, there were differences among the stations (k=8.96, p<0.05, tab. 24), station three showing lower values and one, higher. These differences were provoked by copepods and cirripede larvae, which were very abundant a station 1.

b) 330 µm mesh size net

There were no differences among the stations (k=3.3099086, p>0.05, tab. 26), however, there were differences along the year (k=49.016666, p<0.01, tab. 27), being higher in August and October and lower in December-March. Brachyura larvae and copepods, followed by cirripede, Caridea and gastropod larvae, which were very abundant in August and October, mainly provoked these differences.

Mean density (n°/m3) of planktonic organisms using 125μm mesh size plankton net

	Station 1	Station 2	Station 3
August	38246.1	6878	3361.5
October	88255.2	15913	9890.6
December	72936.6	5806.4	5009.8
March	31840.1	4011.9	8591
May	53620.6	-	1548.6
July	12470.5	8648.2	4123.8
Average	49561.5	8251.5	5421.1

Mean density (n°/m3) of planktonic organisms using 330μm mesh size plankton net

	Station 1	Station 2	Station 3
August	492.81	646.2	349.74
October	1172	623.5	331
December	263.4	233.5	183.3
March	327.8	157.5	201.36
May	464.02	1682.7	317.7
July	336.6	326.43	184.3
Mean	509.5	611.6	261.2

Relation between environmental factors and abundance

<u>Displacement volume</u>

a) with 125 μm mesh size

There was a relation between water temperature and displacement volume in station one (r_s = 0.9428571, $p=0.02$, tab. 28), but no relation was found between temperature and displacement volume at station three (r_s = 0.7714285, $p > 0.05$, tab. 29). Regarding water salinity no relation was found with abundance (r_s = 0.7857143, $p > 0.05$, station 1, tab. 30; r_s = 0.6, $p > 0.1$, station 3, tab. 31).

b) with 330 μm mesh size

There was no relation between water temperature and salinity with abundance (r_s = 0.6571428, $p > 0.1$ for temperature at station 1, tab. 32; r_s = 0.3142857, $p > 0.1$ for temperature at station 2, tab. 33; r_s = 0.2571428, $p > 0.1$ for temperature at station 3, tab. 34; r = 0.3, $p > 0.1$ for salinity at station 1, tab. 35; r_s = - 0.0285714, $p > 0.1$ for salinity at station 2, tab. 36; r_s = 0.6857142, $p > 0.1$ for salinity at station 3, tab. 37).

Density

a) with 125 μm mesh size

There was no any relation between water temperature and salinity with the density ($rs = 0.1428571$, $p > 0.1$, for temperature at station 1, tab. 38; $rs = 0.8285714$, $p = 0.1$, for temperature at station 3, tab. 39; $rs = 0.2428571$, $p > 0.1$, for salinity at station 1; $rs = 0.7714285$, $p > 0.1$, for salinity at station 2, tab. 41).

b) with 330 μm mesh size

There was no any relation between water temperature and salinity with the density ($rs = -0.4285714$, $p > 0.1$, for temperature at station 1, tab. 42; $rs = -0.6$, $p > 0.1$, for temperature at station 2, tab. 43; $rs = 0.4857142$, $p > 0.1$, for temperature at station 3, tab. 44; $rs = 0.2428571$, $p > 0.1$, for salinity at station 1, tab. 45; $rs = -0.4285714$, $p > 0.1$, for salinity at station 2, tab. 46; $rs = 0.4285714$, $p > 0.1$, for salinity at station 3, tab. 47).

The lack of relations between environmental factors with the abundance (displacement volume and density) indicates that although South Bay is located in subtropical area the variation of water temperature between winter and summer is still not enough to influence the abundance of zooplankton. Regarding salinity, this also shows that it does not vary in order to provoke any impact on the abundance. This may indicate that the groundwater discharge in insignificant in this area, however it should be pointed out that there is in the area a well established mangrove forest, which is normally related to freshwater discharge.

Conclusions

- Hydrographic conditions (water temperature and salinity) are homogeneous throughout the South Bay, which may indicate good mixing and very low impact of groundwater discharge in the bay.
- There are variations of water temperature and salinity along the year.
- Zooplankton abundance and composition is similar through the South Bay.
- Zooplankton abundance is generally similar all the year round, except for brachyura and some gastropod larvae.
- There is no relation between water temperature and salinity with zooplankton abundance.
- Potential impact of groundwater is not reflected in zooplankton.

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Appendices

Table 4. Mean density in August ($n°/m^3$): 125 µm mesh size plankton net

Taxa	Station 1	Station 2	Station 3	Total
Amphipoda	0	0	1.5	1.5
Appendicularia	2847.3	377.4	447.6	3672.3
Caridea	0	0	85.9	85.9
Chaetognatha	22.8	34.7	360.9	418.4
Cirripede larvae	2260.9	239	156.7	2656.6
Fish larvae	15.1	7.0	9.0	31.1
Foraminifera	0	0	1.5	1.5
Harpacticoida	5982	359.1	127.5	6468.6
Larvae of bivalve	156.3	212.2	112.6	481.1
Larvae of brachyura	1677.4	361	91.2	2129.6
Larvae of coelenterata	4.4	0	112.8	117.2
Larvae of gastropoda	68.4	1434.6	198.9	1701.9
Larvae of prawn	0	7.7	0	7.7
Lucifer	0	0	9	9
Mysid	0	12.2	0	12.2
Nauplius of copepods	12490.7	543.9	179.4	13214
Ostracoda	38.9	30.8	3.9	73.6
Other copepods	12343.3	3034.1	1143	16520.4
Polychaeta larvae	338.6	159.4	21.8	519.8
Tintinnid	0	64.9	298.3	363.2
Total	**38246.1**	**6878**	**3361.5**	**48485.6**

Table 5. Mean density in October ($n°/m^3$): 125 µm mesh size plankton net

Taxa	Station 1	Station 2	Station 3	Total
Amphipoda	0	0	3.8	3.8
Appendicularia	630.4	246.5	193.5	1070.4
Caprella	0	3.1	0	3.1
Caridea	10.7	19.2	345.4	375.3
Chaetognatha	46	5.2	262.1	313.3
Cirripede larvae	24654.8	5275.6	2212	32142.4
Cladocera	0	3.1	0	3.1
Fish larvae	89.5	3.1	0	92.6
Harpacticoida	10566.8	1427	1070.5	13064.3
Larvae of bivalve	214.8	52.3	41.2	308.3
Larvae of brachyura	543.2	83.6	1194.8	1821.6
Larvae of coelenterata	12.2	0	10.2	22.4
Larvae of gastropoda	761.1	837	444.8	2042.9
Lucifer	0	3.1	13.1	16.2
Medusa	4.4	24.7	23.9	53
Nauplius of copepods	27445.2	3417.8	726.7	31589.7
Not identified 1	131.5	54.6	16.1	202.2
Not identified 2	0	514.5	0	514.5
Not identified 3	73.3	0	0	73.3
Ostracoda	48.9	25.5	6.5	80.9
Other copepods	21835.1	3599.7	3271	28705.8
Polychaeta larvae	1156.7	105.4	30.8	1292.9
Tintinnid	30.6	212	24.2	266.8
Total	**88255.2**	**15913**	**9890.6**	**114058.8**

Table 6. Mean density of December (n°/m³): 125 μm mesh size plankton net

Taxa	Station 1	Station 2	Station 3	Total
Amphipoda	0	0	1.9	1.9
Appendicularia	176.7	51.5	59.7	287.9
Caridea	6.3	32.7	18.9	57.9
Chaetognatha	0	91.4	82.4	173.8
Cirripede larvae	2273.2	134.3	79.8	2487.3
Fish larvae	0	2.8	1.5	4.3
Foraminifera	0	0	1.9	1.9
Harpacticoida	9929.4	1073.3	1113.2	12115.9
Larvae of bivalve	262.7	461.8	142.9	867.4
Larvae of brachyura	18.2	76.7	55.1	150
Larvae of gastropoda	1803.8	1550.9	1832.1	5186.8
Lucifer	0	5.6	0	5.6
Medusa	0	179.6	9.4	189
Nauplius of copepods	30589.6	513.3	582.5	31685.4
Not identified 1	82.4	68	0	150.4
Not identified 2	10203.3	0	0	10203.3
Ostracoda	21.3	0	7.5	28.8
Other copepods	17489.5	1482.5	926.4	19898.4
Polychaeta larvae	80.2	69.9	68	218.1
Tintinnid	0	12.1	26.6	38.7
Total	**72936.6729 36.6**	**5806.4**	**5009.8**	**83752.8**

Table 7. Mean density of March (n°/m³): 125 μm mesh size plankton net

Taxa	Station 1	Station 2	Station 3	Total
Amphipoda	0	0	0	0
Appendicularia	0	0	0	0
Caridea	3.1	12.5	24.1	39.7
Chaetognatha	17.2	174.4	12	203.6
Cladocera	1.6	0	0	1.6
Cirripede larvae	5627.4	566.9	0	6194.3
Copepods	10181.7	0	0	10181.7
Fish larvae	1.6	0	0	1.6
Foraminifera	0	0	0	0
Harpacticoida	4423.2	1264.6	478.8	6166.6
Larvae of bivalve	0	71.6	1011.8	1083.4
Larvae of brachyura	62.5	0	60.2	122.7
Larvae of gastropoda	604.4	398.7	5498.5	6501.6
Lucifer	0	3.1	0	3.1
Medusa	0	0	9	9
Nauplius of copepods	10728.4	242.95	105.4	11076.75
Not identified 1	0	165.1	150.6	315.7
Not identified 2	0	0	557.1	557.1
Ostracoda	3.1	0	0	3.1
Other copepods	0	756.9	647.4	1404.3
Polychaeta larvae	21.9	56.1	9	87
Tintinnid	1.6	0	0	1.6
Tunicate	162.4	299	27.1	488.5
Total	**31840.1**	**4011.85**	**8591**	**44442.95**

Table 8. Mean density of May (n°/m³): 125 μm mesh size plankton net

Taxa	Station 1	Station 2	Station 3	Total
Amphipoda	0	-	0	0
Appendicularia	0	-	0	0
Caridea	10	-	0	10
Chaetognatha	8	-	4.3	12.3
Cirripede larvae	4161.7	-	0	4161.7
Fish larvae	0	-	0	0
Foraminifera	0	-	0	0
Harpacticoida	7067.9	-	211.4	7279.3
Larvae of bivalve	0	-	106.4	106.4
Larvae of brachyura	64.2	-	18.7	82.9
Larvae of gastropoda	1835.2	-	614	2449.2
Lucifer	0	-	0	0
Medusa	0	-	0	0
Nauplius of copepods	22378.8	-	33.1	22411.9
Not identified 1	186.5	-	24.4	210.9
Not identified 2	62.2	-	0	62.2
Ostracoda	0	-	0	0
Other copepods	17150.2	-	509	17659.2
Polychaeta larvae	86.2	-	15.8	102
Tintinnid	0	-	1.4	1.4
Tunicate	609.7	-	10.1	619.8
Total	**53620.6**	**-**	**1548.6**	**55169.2**

Table 9. Mean density of July (n°/m³): 125 μm mesh size plankton net

Taxa	Station 1	Station 2	Station 3	Total
Amphipoda	0	0	0	0
Appendicularia	0	0	0	0
Caridea	5.5	3	2.5	11
Chaetognatha	5.5	81.2	7.5	94.2
Cirripede larvae	748.6	1020.9	0	1769.5
Fish larvae	0	1	0	1
Foraminifera	0	0	0	0
Harpacticoida	1358.8	1821.1	279.2	3459.1
Larvae of bivalve	0	127.4	1047.2	1174.6
Larvae of brachyura	28.1	4	7.5	39.6
Larvae of gastropoda	361.5	293.8	1668	2323.3
Lucifer	0	50.1	0	50.1
Medusa	1.5	14	0	15.5
Nauplius of copepods	5728.1	2154.1	177	8059.2
Not identified 1	96.3	201.6	0	297.9
Not identified 2	0	151.4	22.4	173.8
Ostracoda	0	0	0	0
Other copepods	3974.2	2424.8	832.7	7231.7
Polychaeta larvae	5.5	48.1	2.5	56.1
Tintinnid	8	43.1	22.4	73.5
Tunicate	148.9	208.6	54.9	412.4
Total	**12470.5**	**8648.2**	**4123.8**	**25242.5**

Table 10: Mean density in August (n°/m³): 330 μm mesh size plankton net

Taxa	Station 1	Station 2	Station 3	Total
Amphipoda	0.41	0	0.34	0.75
Appendicularia	56.5	33.3	35.6	125.4
Caridea	28.1	33.5	32.9	94.5
Chaetognatha	2.1	2.9	28.5	33.5
Cirripede larvae	47.2	9.5	19.6	76.3
Copepods	24.8	127.6	56.9	209.3
Euphasid	2.5	0	0	2.5
Fish larvae	0	0.8	3	3.8
Harpacticoida	17.8	11.	13.2	42
Larvae of bivalve	1.7	16.6	9.1	27.4
Larvae of brachyura	140.6	365.8	99.6	606
Larvae of coelenterata	3.3	4.7	25	33
Larvae of gastropoda	27.4	6.9	8.8	43.1
Larvae of Polychaeta	15.8	0	2.7	18.5
Lucifer	0	0	2.4	2.4
Mysid	1.7	4.5	0	6.2
Nauplius of copepods	95.6	16.2	0	111.8
Ostracoda	2.5	1.9	2.7	7.1
Tintinnid	24.8	11	9.4	45.2
Total	**492.81**	**646.2**	**349.74**	**1488.75**

Table 11: Mean density in October (n°/m³): 330 μm mesh size plankton net

Taxa	**Station 1**	**Station 2**	**Station 3**	**Total**
Amphipoda	0	0	0.6	0.6
Appendicularia	59.1	24.9	5.6	89.6
Caprella	31.7	0	0	31.7
Caridea	40.3	66	27.5	133.8
Chaetognatha	0	21.5	12.2	33.7
Cirripede larvae	83	144.3	32.8	260.1
Fish larvae	2.1	0.6	0.3	3
Harpacticoida	4.4	29.2	12.2	45.8
Larvae of bivalve	0.7	0	0	0.7
Larvae of brachyura	359.1	160.9	154.9	674.9
Larvae of gastropoda	113	15.9	0	128.9
Larvae of prawn	0	0	20.1	20.1
Lucifer	11.6	1.3	7.7	20.6
Medusa	3.2	10.6	4.3	18.1
Megalopa	1.1	0	0	1.1
Nauplius of copepods	34.5	11.4	1.5	47.4
Not identified 1	3.7	8.7	0	12.4
Not identified 2	0	2.1	0	2.1
Not identified 3	0	6.9	0	6.9
Ostracoda	0	0.9	0.8	1.7
Other copepods	407.4	109.2	47.4	564
Polychaeta larvae	10	4.6	1.2	15.8
Tintinnid	7.7	4.5	1.9	14.1
Total	**1172.6**	**6235**	**331**	**2127.1**

Table 12. Mean density in December ($n°/m^3$): 330 µm mesh size plankton net

Taxa	Station 1	Station 2	Station 3	Total
Appendicularia	7	9.1	4.9	21
Caprella	0	0	0.8	0.8
Caridea	1.1	6.3	9.7	17.1
Chaetognatha	2.7	15.5	15.1	33.3
Cirripede larvae	22.7	9.4	7.3	39.4
Euphasid	0	0.7	0.2	0.9
Harpacticoida	9.1	10	25.2	44.3
Larvae of bivalve	4.9	14	2.7	21.6
Larvae of brachyura	17.4	30.3	22.7	70.4
Larvae of gastropoda	35.9	27.8	15.5	79.2
Lucifer	0	1.1	1.7	2.8
Medusa	0.6	50.3	3.9	54.8
Nauplius of copepods	97	4.5	6.7	108.2
Not identified 1	0	12	0	12
Not identified 3	0	0	1.9	1.9
Not identified 4	0	0	0.2	0.2
Other copepods	53.8	35.9	47.9	137.6
Polychaeta larvae	0	5	8.8	13.8
Tintinnid	11.2	1.6	8.1	20.9
Total	**263.4**	**233.5**	**1833**	**680.2**

Table 13. Mean density in March ($n°/m^3$): 330 µm mesh size plankton net

Taxa	Station 1	Station 2	Station 3	Total
Appendicularia	0	0	0	0
Caprella	0	0	0	0
Caridea	5.9	17.2	31.7	54.8
Chaetognatha	2.95	2.5	5.5	10.95
Cirripede larvae	36.5	24.3	27.1	87.9
Euphasid	0	0	0	0
Harpacticoida	0	0	0	0
Larvae of bivalve	13.3	5.1	5.2	23.6
Larvae of brachyura	25	16.7	18.6	60.3
Larvae of gastropoda	26.5	8.6	21.8	56.9
Lucifer	46.4	27.4	6.1	79.9
Medusa	3.3	2.5	0.9	6.7
Megalopa	0.4	0	0	0.4
Nauplius of copepods	0	0	0	0
Not identified 1	9.2	9.1	13.96	32.26
Not identified 3	0	0	0	0
Not identified 4	0	0	0	0
Other copepods	144.7	43.1	66.6	254.4
Polychaeta larvae	2.95	1	3.2	7.15
Tintinnid	0	0	0.6	0.6
Tunicate	10.7	0	0	10.7
Total	**327.8**	**157.5**	**201.26**	**686.56**

Table 14. Mean density in May (n°/m³): 330 µm mesh size plankton net

Taxa	Station 1	Station 2	Station 3	Total
Appendicularia	0	0	0	0
Caprella	0	0	0	0
Caridea	2.3	11.6	7.3	21.2
Chaetognatha	2.7	30.7	6.2	39.6
Cirripede larvae	94	125.1	14.1	233.2
Euphasid	0	0	0	0
Harpacticoida	19.3	304.9	6.2	330.1
Larvae of bivalve	4.4	14.1	2.7	21.2
Larvae of brachyura	59.5	82	19.7	161.2
Larvae of gastropoda	58	90.3	1.5	149.8
Lucifer	0.6	4.1	3.1	7.8
Medusa	0	3.3	0	3.3
Megalopa	0.21	0	0	0.21
Nauplius of copepods	37.5	0	0	37.5
Not identified 1	0	45.6	8.3	53.9
Not identified 3	2.9	34.8	13.1	50.8
Not identified 4	0.21	0	0	0.21
Other copepods	157.2	918	222.7	1297.9
Polychaeta larvae	1.7	14.1	4.1	19.9
Tintinnid	2.4	4.1	1.7	8.2
Tunicate	21.1	0	7	28.1
Total	**464.02**	**1682.7**	**317.7**	**2464.12**

Table 15. Mean density in July (n°/m³): 330 µm mesh size plankton net

Taxa	Station 1	Station 2	Station 3	Total
Amphipoda	9.7	0	0	9.7
Appendicularia	0	0	0	0
Caprella	0	0	0	0
Caridea	2.9	9.1	2.9	14.9
Chaetognatha	9.7	9.1	2.9	21.7
Cirripede larvae	29.4	31.5	24.2	85.1
Euphasid	0	0	0	0
Harpacticoida	2.5	25.7	20.5	48.7
Larvae of bivalve	14.5	6.2	1.5	22.2
Larvae of brachyura	7	22.8	9.7	39.5
Larvae of gastropoda	19.9	14.9	4.6	39.4
Lucifer	20.3	0.83	0.4	21.53
Medusa	1.2	3.7	0.6	5.5
Nauplius of copepods	11.6	23.2	31.3	66.1
Not identified 1	20.3	46.8	52.8	119.9
Not identified 3	0	2.5	0	2.5
Not identified 4	0	0	0	0
Other copepods	147.9	103.6	20.7	272.2
Polychaeta larvae	3.7	5.8	1.2	10.7
Tintinnid	5.8	3.3	2.7	11.8
Tunicate	30.2	17.4	8.3	55.9
Total	**336.6**	**326.43**	**184.3**	**847.33**

Table 16. Kruskal-Wallis test to compare water temperature among the three stations

Month	Station 1 (ranks)	Station 2 (ranks)	Station 3 (ranks)
August	12	9	4
October	10	6.5	11
December	15	14	13
March	16	17	18
May	8	5	3
July	1.5	1.5	6.5
n	6	6	6
R	62.5	53	55.5
R^2	3906.25	2809	3080.25
R^2/n	651.04167	468.16667	513.375

$K = [\Sigma(R^2/n) \times 12/N(N+1)] - 3(N+1) = 0.2836245$
$P > 0.05$

Table 17. Kruskal-Wallis test to compare water temperature along the year

	August (ranks)	October (ranks)	December (ranks)	March (ranks)	May (ranks)	July (ranks)
Station 1	12	10	15	16	8	1.5
Station 2	9	6.5	14	17	5	1.5
Station 3	4	11	13	18	3	6.5
n	3	3	3	3	3	3
R	25	27.5	42	51	16	9.5
R^2	625	756.25	1764	2601	256	90.25
R^2/n	208.33333	252.08333	588	867	85.333333	30.083333

$K = [\Sigma(R^2/n) \times 12/N(N+1)] - 3(N+1) = 14.25731$
$P < 0.05$

Table 18. Kruskal-Wallis test for water salinity in the three stations

Month	Station 1 (ranks)	Station 2 (ranks)	Station 3 (ranks)
August	12	9.5	9.5
October	13	14.5	17
December	9.5	2.5	4.5
March	17	17	14.5
May	9.5	1	2.5
July	6.5	4.5	6.5
n	6	6	6
R	67.5	49	54.5
R^2	4556.25	2401	2970.25
R^2/n	759.375	400.16667	495.04167

$K = [\Sigma(R^2/n) \times 12/N(N+1)] - 3(N+1) = 1.0555557$
$P > 0.05$

Table 19. Kruskal-Wallis test to compare water salinity along the year

	August (ranks)	October (ranks)	December (ranks)	March (ranks)	May (ranks)	July (ranks)
Station 1	12	13	9.5	17	9.5	6.5
Station 2	9.5	14.5	2.5	17	1	4.5
Station 3	9.5	17	4.5	14.5	2.5	6.5
n	3	3	3	3	3	3
R	31	44.5	16.5	48.5	13	17.5
R^2	992	1980.25	272.25	2352.25	169	306.25
R^2/n	330.66667	660.08333	90.75	784.08333	56.333333	102.08333

$K = [\Sigma(R^2/n) \times 12/N(N+1)] - 3(N+1) = 14.017544$
$P < 0.05$

Table 20. Kruskal-Wallis test to compare displacement volume within the three stations with 125 μm mesh size.

Month	Station 1 (ranks)	Station 2 (ranks)	Station 3 (ranks)
August	6	7	3
October	4	15	10
December	5	12	9
March	14	8	13
July	1	2	11
n	5	5	5
R	30	44	46
R^2	900	1936	2116
R^2/n	180	387.2	423.2

$K = [\Sigma(R^2/n) \times 12/N (N + 1)]-3(N + 1) = 1.52$
$P > 0.05$

Table 21. Kruskal-Wallis test to compare displacement volume with 125 μm mesh size along the year.

	August (ranks)	October (ranks)	December (ranks)	March (ranks)	July (ranks)
Station 1	6	4	5	14	1
Station 2	7	15	12	8	2
Station 3	3	10	9	13	11
n	3	3	3	3	3
R	16	29	26	35	14
R^2	256	841	676	1225	196
R^2/n	85.333333	280.33333	225.33333	408.33333	65.333333

$K = [\Sigma(R^2/n) \times 12/N (N + 1)]-3(N + 1) = 5.2333328$
$P > 0.05$

Table 22. Kruskal-Wallis test to compare displacement volume within the three stations with 330 μm mesh size.

Month	Station 1 (ranks)	Station 2 (ranks)	Station 3 (ranks)
August	7	13	8
October	14	16	6
December	18	4	1
March	15	12	9
May	2.5	17	2.5
July	11	10	5
n	6	6	6
R	67.5	72	31.5
R^2	4556.25	5184	992.25
R^2/n	759.375	864	165.375

$K = [\Sigma(R^2/n) \times 12/N (N + 1)]-3(N + 1) = 5.7631579$
$P > 0.05$

Table 23. Kruskal-Wallis test to compare displacement volume with 330 μm mesh size along the year.

	August (ranks)	October (ranks)	December (ranks)	March (ranks)	May (ranks)	July (ranks)
Station 1	7	14	18	15	2.5	11
Station 2	13	16	4	12	17	10
Station 3	8	6	1	9	2.5	5
n	3	3	3	3	3	3
R	28	36	23	36	22	26
R^2	784	1296	529	1296	484	676
R^2/n	261.33333	432	176.33333	432	161.33333	225.33333

$K = [\Sigma(R^2/n) \times 12/N (N + 1)]-3(N + 1) = 2.23508$
$P > 0.05$

Table 24. Kruskal-Wallis test to compare density within the three stations with 125 μm mesh size.

Month	Station 1 (ranks)	Station 2 (ranks)	Station 3 (ranks)
August	13	6	1
October	14	11	9
December	15	5	4
March	12	2	7
July	10	8	3
n	5	5	5
R	64	32	24
R^2	4096	1024	576
R^2/n	819.2	204.8	115.2

$K = [\Sigma(R^2/n) \times 12/N (N + 1)]-3(N + 1) = 8.96$
$P < 0.05$

Table 25. Kruskal-Wallis test to compare density with 125 μm mesh size along the year.

	August (ranks)	October (ranks)	December (ranks)	March (ranks)	July (ranks)
Station 1	13	14	15	12	10
Station 2	6	11	5	2	8
Station 3	1	9	4	7	3
n	3	3	3	3	3
R	20	34	24	21	21
R^2	400	1156	576	441	441
R^2/n	133.33333	385.33333	192	147	147

$K = [\Sigma(R^2/n) \times 12/N (N + 1)]-3(N + 1) = 2.2333332$
$P > 0.05$

Table 26. Kruskal-Wallis test to compare density within the three stations with 330 μm mesh size.

Month	Station 1 (ranks)	Station 2 (ranks)	Station 3 (ranks)
August	14	16	12
October	17	15	10
December	6	5	2
March	9	1	4
May	13	18	7
July	11	8	3
N	6	6	6
R	70	63	38
R^2	4900	3969	1444
R^2/n	816.66667	661.5	240.66667

$K = [\Sigma(R^2/n) \times 12/N (N + 1)]-3(N + 1) = 3.3099086$
$P > 0.05$

Table 27. Kruskal-Wallis test to compare density with 330 μm mesh size along the year.

	August (ranks)	October (ranks)	December (ranks)	March (ranks)	May (ranks)	July (ranks)
Station 1	14	17	6	9	13	11
Station 2	16	15	5	1	18	8
Station 3	12	10	2	4	7	3
n	3	3	3	3	3	3
R	42	42	13	14	38	22
R^2	1764	1764	169	196	1444	484
R^2/n	588	588	56.333333	65.333333	481.33333	161.33333

$K = [\Sigma(R^2/n) \times 12/N (N + 1)]-3(N + 1) = 49.016666$
$P < 0.01$

Table 28. Spearman Rank Correlation Coefficient (r_s) to relate water temperature and displacement volume (125 μm mesh size)

Station 1

	Temperature (ranks)	Biovolume (ranks)	d	d^2
August	4	5	-1	1
October	3	3	0	0
December	5	4	1	1
March	6	6	0	0
May	2	2	0	0
July	1	1	0	0

$rs = 1 - [6\Sigma d^2/(n^3 - n)] = 0.9428571$
$p = 0.02$

Table 29. Spearman Rank Correlation Coefficient (r_s) to relate water temperature and displacement volume (125 μm mesh size)

Station 3

	Temperature (ranks)	Biovolume (ranks)	d	d^2
August	2	2	0	0
October	4	4	0	0
December	5	3	2	4
March	6	6	0	0
May	1	1	0	0
July	3	5	-2	4

$rs = 1 - [6\Sigma d^2/(n^3 - n)] = 0.7714285$
$p > 0.1$

Table 30. Spearman Rank Correlation Coefficient (r_s) to relate water salinity and displacement volume (125 μm mesh size)

Station 1

	Salinity (ranks)	Biovolume (ranks)	d	d^2
August	4	5	-1	1
October	5	3	2	4
December	2.5	4	-1.5	2.25
March	6	6	0	0
May	2.5	2	0.5	0.25
July	1	1	0	0

$rs = 1 - [6\Sigma d^2/(n^3 - n)] = 0.7857143$
$p > 0.1$

Table 31. Spearman Rank Correlation Coefficient (r_s) to relate water salinity and displacement volume (125 μm mesh size)

Station 3

	Salinity (ranks)	Biovolume (ranks)	d	d^2
August	4	2	2	4
October	6	4	2	4
December	2	3	-1	1
March	5	6	-1	1
May	1	1	0	0
July	3	5	-2	4

$rs = 1 - [6\Sigma d^2/(n^3 - n)] = 0.6$; $p > 0.1$

Table 32. Spearman Rank Correlation Coefficient (r_s) to relate water temperature and displacement volume (330 μm mesh size)

Station 1

	Temperature (ranks)	**Biovolume (ranks)**	**d**	**d^2**
August	4	2	2	4
October	3	4	-1	1
December	5	6	-1	1
March	6	5	1	1
May	2	1	1	1
July	1	3	-2	4

$rs = 1 - [6\Sigma d^2/(n^3 - n)] = 0.6571428$; $p > 0.1$

Table 33. Spearman Rank Correlation Coefficient (r_s) to relate water temperature and displacement volume (330 μm mesh size)

Station 2

	Temperature (ranks)	**Biovolume (ranks)**	**d**	**d^2**
August	4	4	0	0
October	3	5	-2	4
December	5	1	4	16
March	6	3	3	9
May	2	6	-4	16
July	1	2	-1	1

$rs = 1 - [6\Sigma d^2/(n^3 - n)] = -0.3142857$; $p > 0.1$

Table 34. Spearman Rank Correlation Coefficient (r_s) to relate water temperature and displacement volume (330 μm mesh size)

Station 3

	Temperature (ranks)	**Biovolume (ranks)**	**d**	**d^2**
August	2	5	-3	9
October	4	4	0	0
December	5	1	4	16
March	6	6	0	0
May	1	2	-1	1
July	3	3	0	0

$rs = 1 - [6\Sigma d^2/(n^3 - n)] = 0.2571428$; $p > 0.1$

Table 35. Spearman Rank Correlation Coefficient (r_s) to relate water salinity and displacement volume (330 μm mesh size)

Station 1

	Salinity (ranks)	**Biovolume (ranks)**	**d**	**d^2**
August	4	2	2	4
October	5	4	1	1
December	2.5	6	-3.5	12.25
March	6	5	1	1
May	2.5	1	1.5	2.25
July	1	3	-2	4

$rs = 1 - [6\Sigma d^2/(n^3 - n)] = 0.3$ $p > 0.1$

Table 36. Spearman Rank Correlation Coefficient (r_s) to relate water salinity and displacement volume (330 μm mesh size)

Station 2

	Salinity (ranks)	Biovolume (ranks)	d	d^2
August	4	4	0	0
October	5	5	0	0
December	2	1	1	1
March	6	3	3	9
May	1	6	-5	25
July	3	2	1	1

$rs = 1 - [6\Sigma d^2/(n^3 - n)] = -0.0285714$; $p > 0.1$

Table 37. Spearman Rank Correlation Coefficient (r_s) to relate water salinity and displacement volume (330 μm mesh size)

Station 3

	Salinity (ranks)	Biovolume (ranks)	d	d^2
August	4	5	-1	1
October	6	4	2	4
December	2	1	1	1
March	5	6	-1	1
May	1	2	-2	4
July	3	3	0	0

$rs = 1 - [6\Sigma d^2/(n^3 - n)] = 0.6857142$; $p > 0.1$

Table 38. Spearman Rank Correlation Coefficient (r_s) to relate water temperature and density (125 μm mesh size)

Station 1

	Temperature (ranks)	Density (ranks)	d	d^2
August	4	3	1	1
October	3	6	-3	9
December	5	5	0	0
March	6	2	4	16
May	2	4	-2	4
July	1	1	0	0

$rs = 1 - [6\Sigma d^2/(n^3 - n)] = 0.1428571$; $p > 0.1$

Table 39. Spearman Rank Correlation Coefficient (r_s) to relate water temperature and density (125 μm mesh size)

Station 3

	Temperature (ranks)	Density (ranks)	d	d^2
August	2	2	0	0
October	4	6	-2	4
December	5	4	1	1
March	6	5	1	1
May	1	1	0	0
July	3	3	0	0

$rs = 1 - [6\Sigma d^2/(n^3 - n)] = 0.8285714$; $p = 0.1$

Table 40. Spearman Rank Correlation Coefficient (r_s) to relate water salinity and density (125 µm mesh size)

Station 1

	Salinity (ranks)	Density (ranks)	d	d^2
August	4	3	1	1
October	5	6	-1	1
December	2.5	5	-2.5	6.25
March	6	2	4	16
May	2.5	4	-1.5	2.25
July	1	1	0	0

$rs = 1 - [6\Sigma d^2/(n^3 - n)] = 0.2428571$; $p > 0.1$

Table 41. Spearman Rank Correlation Coefficient (r_s) to relate water salinity and density (125 µm mesh size)

Station 3

	Salinity (ranks)	Density (ranks)	d	d^2
August	4	2	2	4
October	6	6	0	0
December	2	4	-2	4
March	5	5	0	0
May	1	1	0	0
July	3	3	0	0

$rs = 1 - [6\Sigma d^2/(n^3 - n)] = 0.7714285$ $p > 0.1$

Table 42. Spearman Rank Correlation Coefficient (r_s) to relate water temperature and density (330 µm mesh size)

Station 1

	Temperature (ranks)	Density (ranks)	d	d^2
August	4	5	-1	1
October	3	6	-3	9
December	5	1	4	16
March	6	2	4	16
May	2	4	-2	4
July	1	3	-2	4

$rs = 1 - [6\Sigma d^2/(n^3 - n)] = - 0.4285714$; $p > 0.1$

Table 43. Spearman Rank Correlation Coefficient (r_s) to relate water temperature and density (330 µm mesh size)

Station 2

	Temperature (ranks)	Density (ranks)	d	d^2
August	4	5	-1	1
October	3	4	-1	1
December	5	2	3	9
March	6	1	5	25
May	2	6	-4	16
July	1	3	-2	4

$rs = 1 - [6\Sigma d^2/(n^3 - n)] = - 0.6$; $p > 0.1$

Table 44. Spearman Rank Correlation Coefficient (r_s) to relate water temperature and density (330 μm mesh size)

Station 3

	Temperature (ranks)	Density (ranks)	d	d^2
August	2	6	-4	16
October	4	5	-1	1
December	5	1	4	16
March	6	3	3	9
May	1	4	-3	9
July	3	2	1	1

$rs = 1 - [6\Sigma d^2/(n^3 - n)] = -0.4857142$; $p > 0.1$

Table 45. Spearman Rank Correlation Coefficient (r_s) to relate water salinity and density (330 μm mesh size)

Station 1

	Salinity (ranks)	Density (ranks)	d	d^2
August	4	5	-1	1
October	5	6	-1	1
December	2.5	1	1.5	2.25
March	6	2	4	16
May	2.5	4	-1.5	2.25
July	1	3	-2	4

$rs = 1 - [6\Sigma d^2/(n^3 - n)] = 0.2428571$ $p > 0.1$

Table 46. Spearman Rank Correlation Coefficient (r_s) to relate water salinity and density (330 μm mesh size)

Station 2

	Salinity (ranks)	Density (ranks)	d	d^2
August	4	5	-1	1
October	5	4	1	1
December	2	2	0	0
March	6	1	5	25
May	1	6	-5	25
July	3	3	0	0

$rs = 1 - [6\Sigma d^2/(n^3 - n)] = -0.4285714$; $p > 0.1$

Table 47. Spearman Rank Correlation Coefficient (r_s) to relate water salinity and density (330 μm mesh size)

Station 3

	Salinity (ranks)	Density (ranks)	d	d^2
August	4	6	-2	4
October	6	5	1	1
December	2	1	1	1
March	5	3	2	4
May	1	4	-3	9
July	3	2	1	1

$rs = 1 - [6\Sigma d^2/(n^3 - n)] = 0.4285714$; $p >0.1$

Study of reproductive aspects of *Holothuria scabra* (Jaeger) at Inhaca Island, Southern Mozambique

Rabia Abdula
Fisheries Research Institute, Maputo, Mozambique.
Domingos Gove
Eduardo Mondlane University, Maputo, Mozambique

Introduction

At Inhaca Island, marine resources are the basis for survival of the local population either directly as food, or indirectly as source of income for purchase of other goods (Anon., 1990).

Fish is the most important economical resource; however, marine invertebrates have also a significant role as source of animal protein and minerals for the local population. In the recent years they have became relevant as source of income.

In this context, one has to highlight the sea cucumbers, also known as magajojo in Mozambique (Fisher *et al*, 1990), whose exploitation was started at Inhaca Island at the beginning of 60s by the Chinese, and used only for trade to Asia (Montecino, 1989).

According to Fisher *et al*, (1990), there are six species of sea cucumbers with commercial value, being *Holothuria scabra*, one of the most exploited in the country.

A rapid increase of collectors was observed at Inhaca in late 80s, leading to overexploitation of this resource. This was provoked by a number of factors, namely, the high commercial value of sea cucumbers; the depletion of its stocks in many places within the country; and because the fuel wood used for its preparation is free in the island. In this context, there was a strong necessity for implementation of sustainable management of its exploitation, but to achieve that, a basic knowledge of its ecology and biology was required.

Costa and Montecino (1990) and Gujral (1995) respectively carried out some studies, respectively to describe the situation of *H. scabra* at Inhaca and to study some biological and ecological aspects of *H. scabra*. These works, however, were not able to determine the reproductive season of this species, mainly due to the failure of the methods used.

This work is to determine the reproductive period as well as the related environmental factors. South Bay (fig. 1 of Martins & Gove and Gove & Fumo in this report) is an area, within the island, where there is a significant discharge of groundwater (according to the map of Belgium team) and at the same time the most important area of occurrence of *H. scabra* (Gujral, 1995). This could suggest that salinity variations might affect this species, and probably its reproduction. Salinity influence could be not only direct, but could be an indication of groundwater discharge, which would imply the availability of nutrients, with consequences on primary production and organic matter.

Furthermore, the island is located at subtropical zone, where thermal annual variation would show significant differences between the two seasons (summer and winter) with consequences on biological resources.

Objectives

- To determine the spawning periods of *Holothuria scabra*
- To determine the minimum length for reproduction; and
- To relate the spawning with environmental parameters (temperature and salinity)

Material and Methods

Sampling

This study was carried out from February 1998 to January 1999. Sampling took place monthly at low tide of spring tides, since this is the time where the sandbanks, where the sea cucumbers occur, are easily accessible. Twenty individuals (Connand, pers. comm.) were collected randomly every month and the following items were measured or determined: total length, fresh and drained weights, sex, weight of the gonads and their maturation stage (according to the table of Conand, 1990, appendix 6). The gonads were observed under microscope. Salinity and temperature of the sediment were also measured, using, respectively, a refractometer (ATAGO) and a thermometer. The study area was divided by substrate type, and for each section, water temperature and salinity were measured randomly in four places.

Data analysis

The sexual cycle was expressed in terms of sex ration and gonad-somatic index:
Sex ratio = N° of males/N° of females
Gonad-somatic index (GSI) = G x 100/Dw
Where:
G – Weight of the gonads
Dw – Drained weight of the whole individual

The study of monthly variation of gonad-somatic index was based on drained weight, since it is less variable, compared to fresh weight (Conand, 1990).

The first maturation size was estimated graphically through cumulative curve of the percentage of individuals close to maturation, mature and after spawning stages (stages III, IV and V) in relation to their length classes. The point in the curve where 50% of individuals were considered mature was the size of the first maturation (Conand, 1990).

X^2 test was used to compare the proportions of males and females along the year. Spearman Rank Correlation Coefficient (r_s) was used to correlate the environmental parameters (Water temperature and salinity with the gonad-somatic index and percentage of mature individuals).

Results and Discussion

Sexual Cycle

This parameter was described in terms of sex ratio and monthly evolution of gonad-somatic index.

Sex ratio

Although there was always slightly more males than females (table 1) during the study period, the proportion was 1:1 (X^2 = 19.1, $p > 0.05$, appendix 1).

Gonad-somatic index

The average gonad-somatic index (GS I) varied from 1.16 to 8.72. According to table 2, the maturation period, where the gonad-somatic index is high, was during the early summer (November-January), which correspond to the period where the temperature was high (26 ºC), however, there was no correlation between temperature and gonad-somatic index (r_s= 0.552, $p > 0.05$, appendix 2).

In Gulf of Mannar, India, *Holothuria scabra* shows two spawning peaks, one in March-April, and the other in September-October (James, 1989). In New Caledonia, mature individuals have been found almost all the year round, although inter annual variations seems to occur (Conand, 1990). Costa and Montecino (1990) report in a study carried out in Cabo Delgado, northern Mozambique, that this species showed two spawning peaks, namely in March/April and September/October.

No correlation exists between water salinity and the gonad-somatic index (r_s = -0.282, $p > 0.1$, appendix 3). This may indicate that there is no groundwater influence in the reproduction of H. scabra in South Bay, Inhaca Island.

Table 1. Number of females and males per month

Month	**Number of individuals collected**	Number of females	**Number of males**
February 98	20	8	12
March	20	5	15
April	20	6	14
May	20	9	11
June	20	10	10
July	20	10	10
August	20	6	14
September	20	5	15
October	20	10	10
November	20	11	9
December	20	8	12
January 99	20	14	6
Average	20	8.5	11.5

Table 2. Monthly values of gonad-somatic index

	N° of Individ.	**Drained weight (g)**	**Gonad weight (g)**	**Gonad Index**	**Temper ature**	**Salinity (p.p.t)**
February 98	20	158.73	1.84	1.16	26.0 °C	35.2
March	20	196.45	3.22	1.64	25.7 °C	35.0
April	20	164.20	4.19	2.55	24.6 °C	35.1
May	20	144.38	4.97	3.44	24.5 °C	35.3
June	20	188.35	4.39	2.33	23.3 °C	35.0
July	20	124.20	2.40	1.93	23.5 °C	35.1
August	20	127.40	1.86	1.46	23.1 °C	35.0
September	20	141.33	2.18	1.54	23.5 °C	35.0
October	20	124.20	2.41	1.94	24.0 °C	34.0
November	20	162.75	8.93	5.49	26.0 °C	34.9
December	20	193.81	6.70	3.46	26.0 °C	35.0
January 99	20	310.40	27.09	8.72	27.0 °C	34.3

Relation among environmental parameters and percentage of mature individuals

As was the case with the gonad-somatic index, there was no any relation among water temperature (r_s = 0.36014, $p > 0.1$, appendix 4) and salinity (r_s = 0.164, $p > 0.1$, appendix 5) with the percentage of mature individuals. This also may indicate that the seasonal variation of water temperature in South Bay is not enough to influence the maturation of this species. Groundwater discharge in this are, which may be indicated by salinity values, does not also influence the maturation of this species.

Table 3. Percentage of mature individuals per month.

Month	% of mature individuals	Temperature (°C)	Salinity (‰)
February	30	26.0 °C	35.2
March	20	25.7 °C	35.0
April	15	24.6 °C	35.1
May	20	24.5 °C	35.3
June	35	23.3 °C	35.0
July	15	23.5 °C	35.1
August	15	23.1 °C	35.0
September	20	23.5 °C	35.0
October	10	24.0 °C	34.0
November	10	26.0 °C	34.9
December	40	26.0 °C	35.0
January	45	27.0 °C	34.3

Length at first maturity

According to the fig. 2, the length at first maturity was estimated to be between 17 – 18 cm. This value is similar with that obtained by Conand (1990) in New Caledonia (16 cm), however, is slightly smaller than that from Gulf of Mannar, India, 21 cm (James *et al*, 1994).

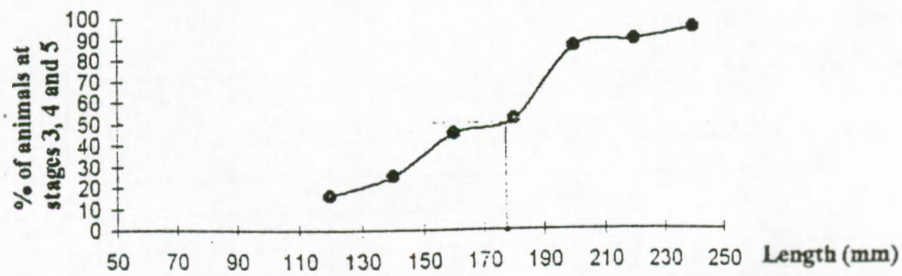

Fig.2. Relation between length of individuals and the percentage of individuals at stages III, IV and V.

Conclusions

The influence of groundwater, indicated here by salinity values, was not observed in the reproduction of *H. scabra*.

- The proportion of females and males of H. scabra in South Bay is 1 : 1.
- Maturation is in the early summer (November – January)
- There is no relation between the temperature and salinity with the maturation of individuals.
- The length at first maturation is between 17 and 18 cm in South Bay.

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Appendix 1

Results of X^2

Month	N^0 of individuals collected	N^0 of females	O-E	$(O–E)^2$	$(O-E)^2/E$	N^0 of males	O-E	$(O–E)^2$	$(O-E)^2/E$
February 98	20	8	- 0.5	0.25	0.029412	12	+ 0.5	0.25	0.02941
March	20	5	- 3.5	12.25	1.4412	15	+ 3.5	12.25	1.4412
April	20	6	- 2.5	6.25	0.7353	14	+ 2.5	6.25	0.7353
May	20	9	+ 0.5	0.25	0.029412	11	- 0.5	0.25	0.02941
June	20	10	+ 1.5	2.25	0.26471	10	- 1.5	2.25	0.26471
July	20	10	+ 1.5	2.25	0.26471	10	- 1.5	2.25	0.26471
August	20	6	- 2.5	6.25	0.7353	14	+ 2.5	6.25	0.7353
September	20	5	- 3.5	12.25	1.4412	15	+ 3.5	12.25	1.4412
October	20	10	+ 1.5	2.25	0.26471	10	- 1.5	2.25	0.26471
November	20	11	+ 2.5	6.25	0.7353	9	- 2.5	6.25	0.7353
December	20	8	- 0.5	0.25	0.029412	12	+ 0.5	0.25	0.02941
January 99	20	14	+ 5.5	30.25	3.55883	6	- 5.5	30.25	3.5588
AVERAGE	20	**8.5**			**9.529496**	**11.5**			**9.5294**

$X^2 = 9.529496 \times 2 = 19.06$. Degrees of freedom $= (12 – 1)(2 – 1) = 11$. $p > 0.05$

Appendix 2

The Spearman Rank Correlation Coefficient r_s

	Gonad Index	Ranks	Temperature	Ranks	d	d^2
February 98	1.16	1	26.0 °C	10	-9	81
March	1.64	4	25.7 °C	8	-4	16
April	2.55	8	24.6 °C	7	1	1
May	3.44	9	24.5 °C	6	3	9
June	2.33	7	23.3 °C	2	5	25
July	1.93	5	23.5 °C	3.5	1.5	2.25
August	1.46	2	23.1 °C	1	1	1
September	1.54	3	23.5 °C	3.5	-0.5	0.25
October	1.94	6	24.0 °C	5	1	1
November	5.49	11	26.0 °C	10	1	1
December	3.46	10	26.0 °C	10	0	0
January 99	8.72	12	27.0 °C	12	0	0
Total					0.0	137.5

$r_s = 1 – [6\sum d^2/(n^3 – n)] = 1 – [6 \times 137.5/(12^3 – 12)] = 0.481$. $n = 12$. $p > 0.1$

Appendix 3

The Spearman Rank Correlation Coefficient r_s

	Gonad Index	Rank	Salinity (p.p.t)	Rank	d	d^2
February 98	1.16	1	35.2	11	-10	100
March	1.64	4	35.0	6	-2	4
April	2.55	8	35.1	9.5	-1.5	2.25
May	3.44	9	35.3	12	-3	9
June	2.33	7	35.0	6	1	1
July	1.93	5	35.1	9.5	-4.5	20.25
August	1.46	2	35.0	6	-4	16
September	1.54	3	35.0	6	-3	9
October	1.94	6	34.0	1	5	25
November	5.49	11	34.9	3	8	64
December	3.46	10	35.0	6	4	16
January 99	8.72	12	34.3	2	10	100
Total					0.0	366.5

$r_s = 1 - [6\Sigma d^2/(n^3 - n)] = -0.282.$ $n = 12.$ $p > 0.1$

Appendix 4

The Spearman Rank Correlation Coefficient r_s

Month	**% of mature individuals**	**Rank**	**Temperature (°C)**	**Rank**	**d**	d^2
February	30	9	26.0 °C	10	-1	1
March	20	7	25.7 °C	8	-1	1
April	15	4	24.6 °C	7	-3	9
May	20	7	24.5 °C	6	1	1
June	35	10	23.3 °C	2	8	64
July	15	4	23.5 °C	3.5	0.5	0.25
August	15	4	23.1 °C	1	3	9
September	20	7	23.5 °C	3.5	3.5	12.25
October	10	1.5	24.0 °C	5	-3.5	12.25
November	10	1.5	26.0 °C	10	-8.5	72.25
December	40	11	26.0 °C	10	1	1
January	45	12	27.0 °C	12	0	0
Total					0.0	183

$r_s = 1 - [6\Sigma d^2/(n^3 - n)] = 1 - [6 \times 183/(12^3 - 12) = 0.36014.$ $n = 12.$ $p > 0.1$

Appendix 5

The Spearman Rank Correlation Coefficient r_s

Month	**% of mature individuals**	**Rank**	**Salinity (p.p.t)**	**Rank**	**d**	**d^2**
February	30	9	35.2	11	-2	4
March	20	7	35.0	6	1	1
April	15	4	35.1	9.5	-5.5	30.25
May	20	7	35.3	12	-5	25
June	35	10	35.0	6	4	16
July	15	4	35.1	9.5	-5.5	30.25
August	15	4	35.0	6	-2	4
September	20	7	35.0	6	1	1
October	10	1.5	34.0	1	0.5	0.25
November	10	1.5	34.9	3	-1.5	2.25
December	40	11	35.0	6	5	25
January	45	12	34.3	2	10	100
Total					0.0	239

$r_s = 1 - [6\Sigma d^2/(n^3 - n)] = 1 - [6 \times 239/123 - 12)] = 0.16444$

$n = 12$

$p > 0.1$

Appendix 6

Sea-cucumbers – Maturation scale (Conand, 1990)

Stage/Sex	Morphology	Microscopic characteristics
Not determined Stage I (immature) Stage II (rest)	Few transparent tubes with small and narrow ramifications	Spherical germinative cells with a diameter smaller than 20 µm
Stage III (growth)		
Females		Spherical and opaque ovum with a diameter between 20 and 120 µm
Males	Whitish and ramified tubes. There is an increase on the length and diameter of the tubes	Development of some spermatozoid
Stage IV (mature)		
Females	Transparent, translucid rose tubes. Mature ovum	Multi-modal distribution of the diameters of ovum. The main mode is approximately at 150-200 µm. free ovum or attached to follicular membrane through follicular appendices
Males	White and dilated tubes, with the maximum volume. Sperm can be present in the duct of the gonads	Many moving spermatozoids
Stage V (post-spawning)		
Females	Some tubes are equal to stage IV; some are small, brownish and loose	Few mature ova dispersed around the tubes. Empty follicular membranes.
Males		Spermatozoids aggregated or sperical