RESULTS

Results

I. Ecological parameters

Hydrographic conditions:

Table 1 shows the seasonal variations of temperature, salinity, pH and dissolved oxygen (DO) at the different surveyed locations during 1998 and 1999.

1. *Water temperature*:

In general, the seasonal variations of water temperatures follow those of the prevailing climate conditions.

As expected, air temperatures are usually higher than water temperatures during the middle of the day but lower at night and early in the morning. Water temperature in the area of study fluctuated between 17 °C and 30 °C. The lowest value was measured during winter season at location II, while the highest value was measured during summer season at locations I, II and VII (Table 1). The highest annual average of 26.27 ± 2.45 °C was obtained at location VII, while the lowest annual average of 23.89 ± 2.31 °C was obtained at location IV (Table 1).

2. *Salinity*:

Salinity is expressed as ppt (or ‰S). Table 1 indicates that the salinity values in the area of study decreases from the northern part (Gulf of Suez) towards the southern part of the study area (northern Egyptian Red Sea proper). The maximum salinity of 43.1 ppt was recorded during summer at locations I, whilst the minimum salinity of 39.34 ppt was recorded during winter at location VII. Location I showed the highest

Table 1. Hydrographic data of seawater in the area of study during 1998 and 1999.

Parameter	Season	Location						
		I	II	III	IV	V	VI	VII
Temperature (°C)	Spring	26.00	25.00	25.16	24.72	24.66	24.73	27.36
	Summer	30.00	30.00	29.20	28.75	29.10	29.00	30.00
	Autumn	27.00	27.50	26.55	24.49	26.22	25.23	28.60
	Winter	17.20	17.00	17.70	17.60	17.50	17.60	19.10
	Ann. Av.	25.05	24.88	24.65	23.89	24.37	24.14	26.27
	± SE	± 2.75	± 2.82	± 2.46	± 2.31	± 2.47	± 2.38	± 2.45
Salinity (ppt)	Spring	42.30	42.26	40.23	40.12	39.90	39.96	39.60
	Summer	43.10	42.57	40.35	40.10	39.85	39.60	40.20
	Autumn	42.61	42.40	40.32	40.12	39.87	40.03	39.40
	Winter	42.60	42.50	40.29	40.20	39.90	40.05	39.34
	Ann. Av.	42.65	42.43	40.30	40.14	39.88	39.91	39.64
	± SE	± 0.17	± 0.07	± 0.03	± 0.02	± 0.01	± 0.11	± 0.20
pH	Spring	8.24	8.18	8.26	8.24	8.22	8.22	8.19
	Summer	8.26	8.26	8.12	8.21	8.18	8.12	8.14
	Autumn	8.10	8.10	8.20	8.17	8.23	8.20	8.15
	Winter	8.30	8.30	8.24	8.25	8.24	8.23	8.20
	Ann. Av.	8.23	8.21	8.21	8.22	8.22	8.19	8.17
	± SE	± 0.04	± 0.04	± 0.03	± 0.02	± 0.01	± 0.02	± 0.01
Dissolved oxygen ($mg\ O_2\ L^{-1}$)	Spring	6.00	5.22	6.15	6.17	6.42	6.41	7.00
	Summer	5.00	4.71	5.68	5.43	5.11	5.95	6.18
	Autumn	5.10	5.61	5.41	4.82	4.72	5.14	6.45
	Winter	5.50	5.00	4.85	5.53	4.66	5.14	6.49
	Ann. Av.	5.40	5.14	5.52	5.49	5.23	5.66	6.53
	± SE	± 0.23	± 0.19	± 0.27	± 0.28	± 0.41	± 0.31	± 0.17

Ann. Av. = Annual average. SE = Standard error

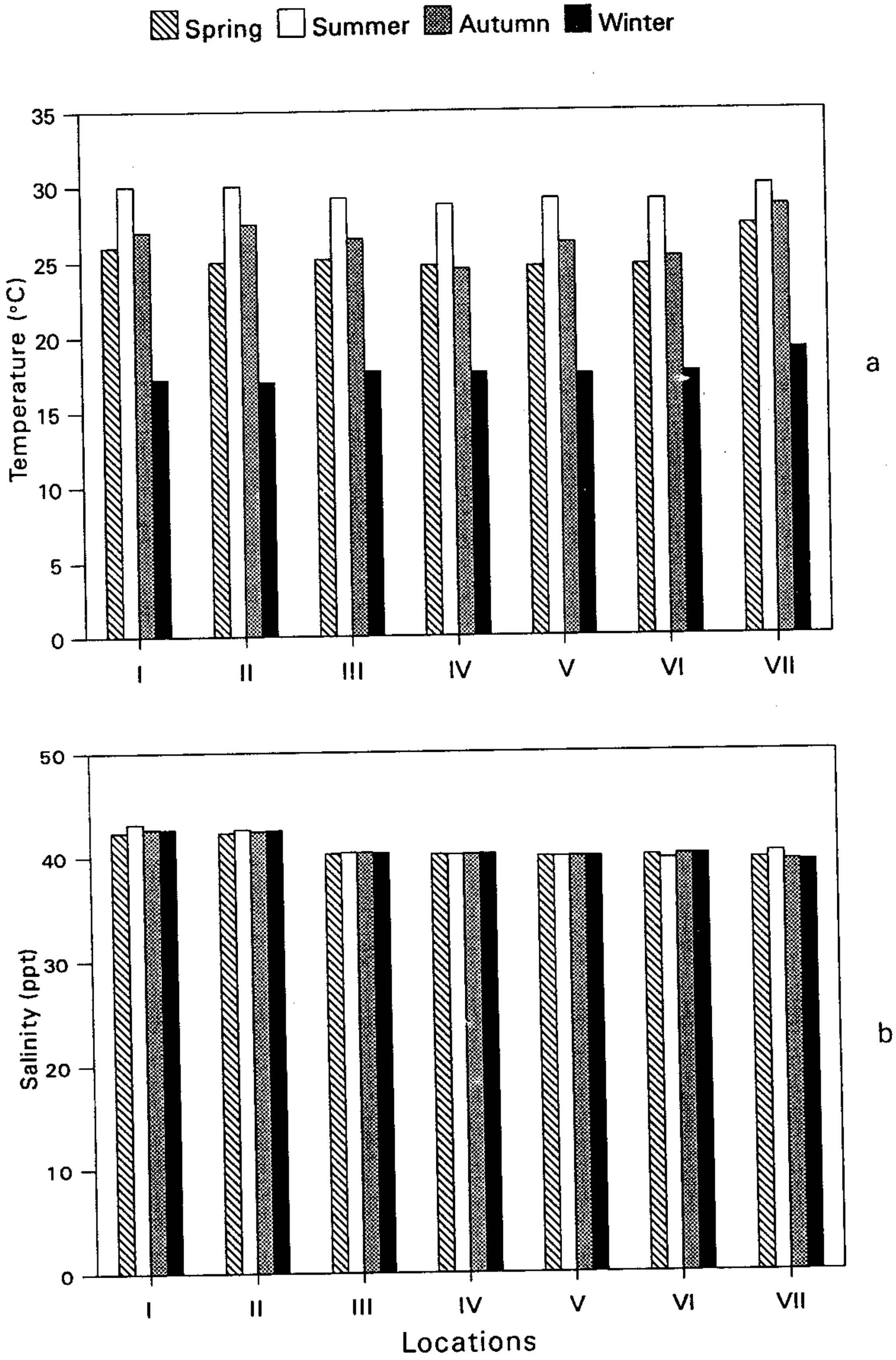


Fig. 3. Seasonal variation of seawater temperature (**a**) and salinity (**b**) at the investigated locations during 1998 and 1999.

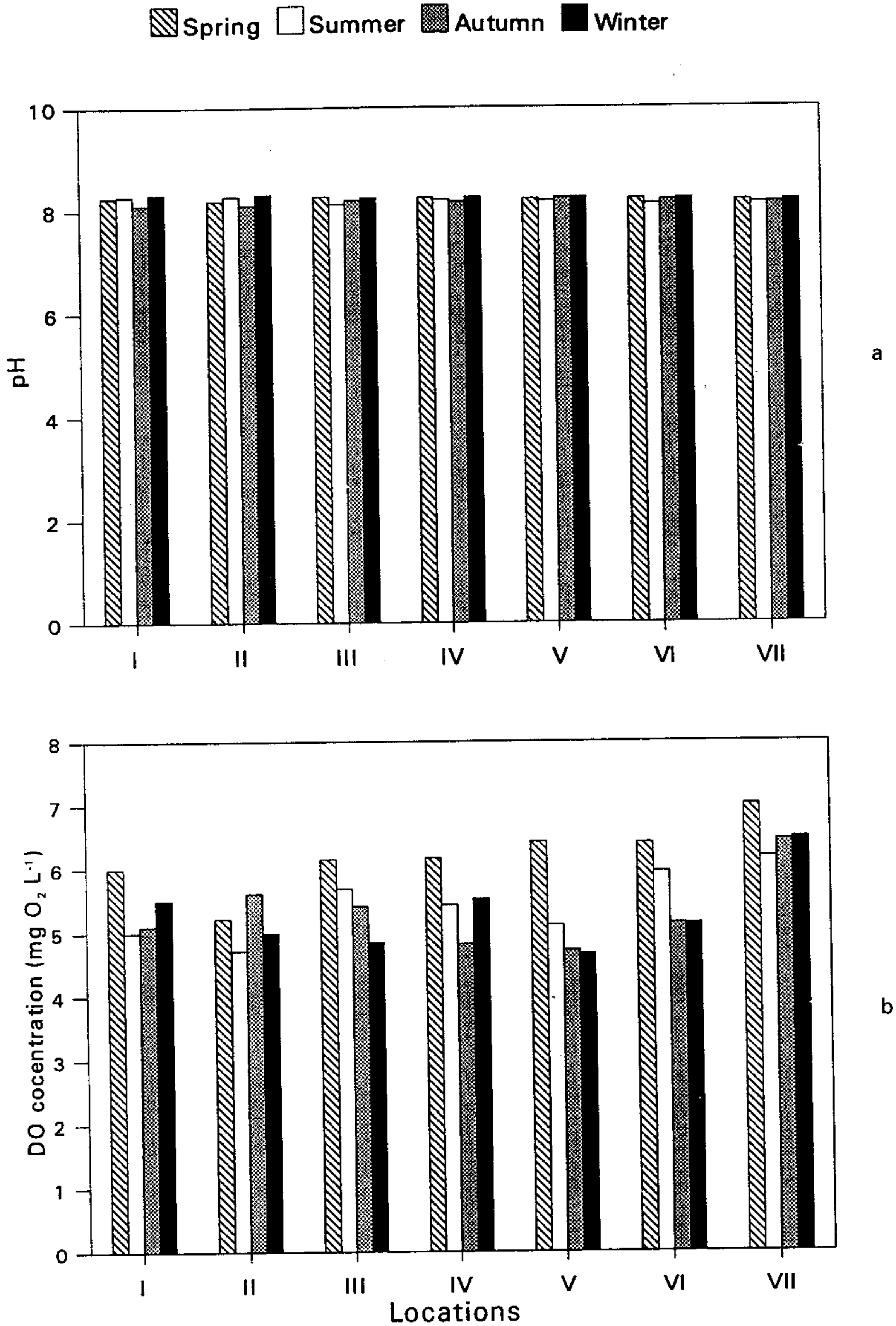


Fig. 4. Seasonal variation of seawater pH (**a**) and dissolved Oxygen (**b**) at the investigated locations during 1998 and 1999.

annual average of salinity (42.65 ± 0.17 ‰S), whilst location VII showed the lowest value (39.64 ± 0.20 ‰S) (Table 1).

3. *pH measurements*:

Table 1 reveals that pH in the area of investigation varied within a narrow limit. Locations II & I showed the highest pH value of 8.3 during winter as well as the lowest pH value of 8.1 during autumn. The annual average of pH was slightly higher (8.23 ± 0.04) at location I and slightly lower (8.17 ± 0.01) at location VII than other locations (Table 1).

4. *Dissolved oxygen (DO)*:

Dissolved oxygen is expressed as mg O_2 per liter of water (or ppm). Concentration of dissolved oxygen is affected by several factors such as temperature, the amount of organic matter, water pollution, respiration of aquatic animals and the photosynthetic activity of aquatic flora. Concentration of dissolved oxygen in sea water of the surveyed area reached to its maximum value of 7 mg O_2 L^{-1} during spring at location VII, and reached to its minimum value of 4.71 mg O_2 L^{-1} during summer at location II (Table 1). Location VII showed the highest annual average of dissolved oxygen (6.53 ± 0.17 mg O_2 L^{-1}), whereas the lowest value (5.14 ± 0.19 mg O_2 L^{-1}) was recorded at location II (Table 1).

Nutrients:

Coral reefs are typically occurring in warm and clear waters with very low nutrient concentrations. High concentrations of nutrients can cause phytoplankton or algal blooms. Table 2 shows seasonal variation of dissolved ammonia, nitrite, nitrate and phosphate, and total dissolved inorganic nitrogen (DIN) concentrations in seawater of the investigated sites during 1998 and 1999. The mean values and standard errors at each site were calculated (Table 3).

Table 2. Seasonal variation of ammonia, nitrite , nitrate, total dissolved inorganic nitrogen (DIN), Phosphate and chlorophyll *a* concentrations in seawater at the studied reef sites during1998 and 1999.

Parameter	Season	Location						
		I	II	III	IV	V	VI	VII
Ammonia (NH_3-N) (μ**M**)	Spring	3.00	5.57	6.36	1.71	1.16	1.36	2.28
	Summer	2.11	1.71	5.14	5.86	2.50	3.00	2.43
	Autumn	2.01	3.36	3.36	0.93	1.14	2.36	2.07
	Winter	5.07	4.21	2.14	1.78	1.36	1.57	2.50
Nitrite (NO_2-N) (μ**M**)	Spring	0.48	0.09	0.17	0.09	0.15	0.18	0.06
	Summer	0.08	0.06	0.08	0.22	0.12	0.22	0.01
	Autumn	0.07	0.03	0.12	0.01	0.01	0.04	0.05
	Winter	0.09	0.07	0.18	0.15	0.13	0.20	0.03
Nitrate (NO_3-N) (μ**M**)	Spring	1.34	0.62	0.88	0.21	0.60	0.26	0.34
	Summer	0.15	0.13	0.41	0.36	0.33	0.49	0.11
	Autumn	0.12	0.14	0.36	0.05	0.03	0.63	0.53
	Winter	0.13	0.65	0.87	0.22	0.75	0.43	0.61
DIN (μ**M**)	Spring	4.82	6.28	7.41	2.01	1.91	1.80	2.68
	Summer	2.34	1.90	5.63	6.44	2.95	3.71	2.55
	Autumn	2.20	3.53	3.84	0.99	1.18	3.03	2.65
	Winter	5.29	4.93	3.19	2.15	2.24	2.20	3.14
Phosphate (PO_4-P) (μ**M**)	Spring	0.32	0.18	0.23	0.07	0.02	0.17	0.20
	Summer	0.22	0.27	0.33	0.18	0.12	0.03	0.15
	Autumn	0.12	0.76	0.71	0.10	0.03	0.05	0.05
	Winter	0.15	0.41	0.43	0.27	0.08	0.12	0.07
Chlorophyl *a* (µg L^{-1})	Spring	0.22	0.24	0.31	0.13	0.02	0.07	0.16
	Summer	0.24	0.24	0.28	0.34	0.42	0.34	0.13
	Autumn	0.23	0.25	0.29	0.32	0.24	0.12	0.23
	Winter	0.29	0.34	0.22	0.01	0.03	0.05	0.04

Table 3. Average values of ammonia, nitrite, nitrate, total dissolved inorganic nitrogen (DIN), phosphate and chlorophyll *a* (chl. *a*) concentrations in seawater of the investigated locations. Standard errors are presented in parenthesis.

Parameter	**Location**						
	I	**II**	**III**	**IV**	**V**	**VI**	**VII**
Ammonia (μM)	3.05 [0.71]	3.71 [0.81]	4.25 [0.93]	2.57 [1.11]	1.54 [0.32]	2.07 [0.38]	2.32 [0.09]
Nitrite (μM)	0.18 [0.10]	0.06 [0.01]	0.14 [0.02]	0.12 [0.04]	0.10 [0.03]	0.16 [0.04]	0.04 [0.01]
Nitrate (μM)	0.43 [0.30]	0.38 [0.14]	0.63 [0.14]	0.21 [0.06]	0.43 [0.16]	0.45 [0.08]	0.40 [0.11]
DIN (μM)	3.66 [0.81]	4.16 [0.94]	5.02 [0.95]	2.90 [1.21]	2.07 [0.37]	2.68 [0.43]	2.75 [0.13]
Phosphate (μM)	0.20 [0.04]	0.40 [0.13]	0.42 [0.10]	0.15 [0.04]	0.06 [0.02]	0.09 [0.03]	0.12 [0.03]
Chl. *a* ($\mu g\ L^{-1}$)	0.25 [0.02]	0.27 [0.02]	0.28 [0.02]	0.20 [0.08]	0.18 [0.09]	0.14 [0.07]	0.14 [0.04]

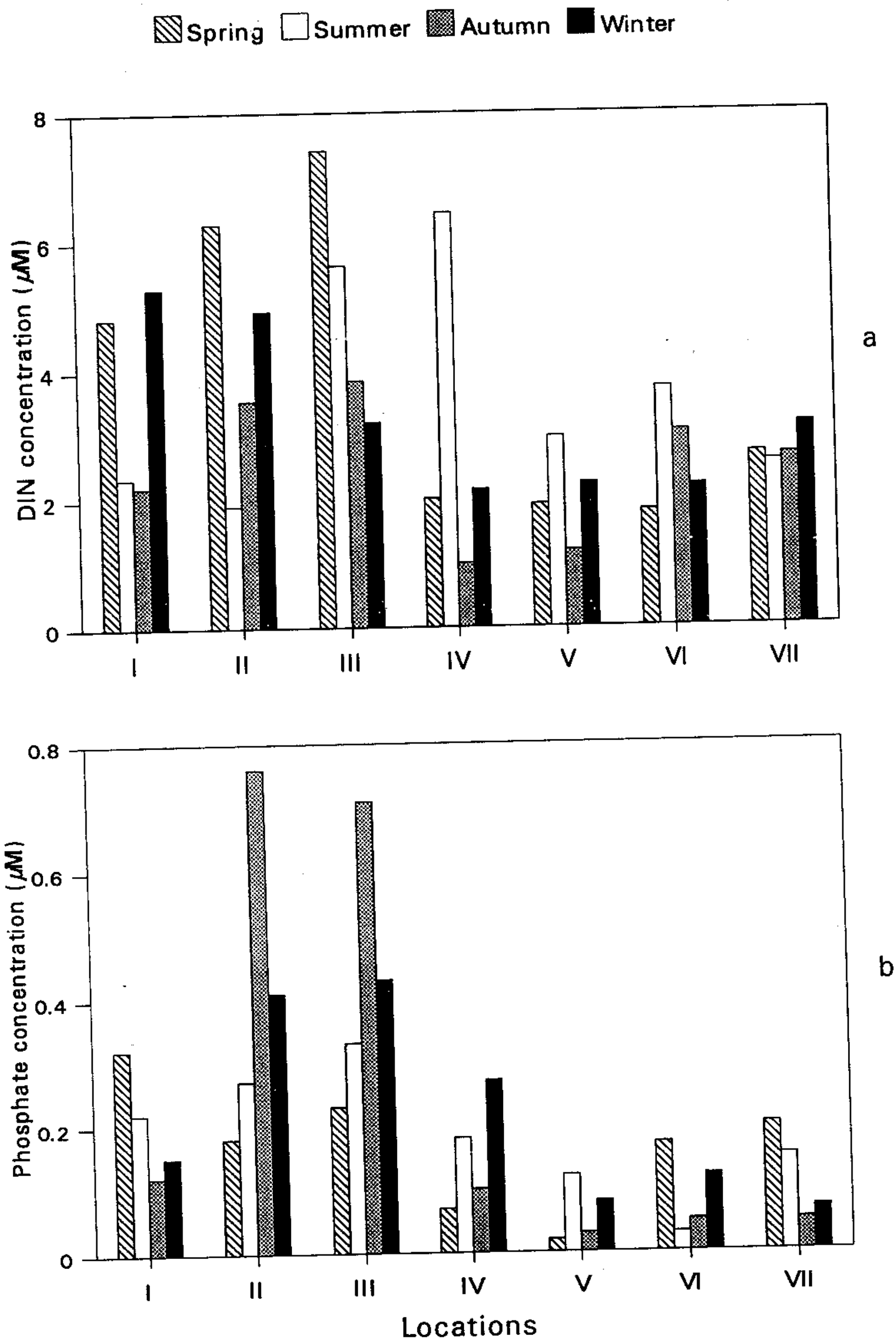


Fig. 5. Seasonal variation of total dissolved inorganic nitrogen (DIN) (**a**) and dissolved phosphate (**b**) concentrations (μM) in seawater at the studied locations during 1998 and 1999.

1. *Ammonia*:

Ammonia is biologically an active compound present in most waters as normal biological degradation product of nitrogenous organic matter. Also, it reaches to seawater through sewage runoff and other human activities. The maximum concentration of ammonia 6.36 μ**M** was measured during spring season at site III, while the minimum concentration 0.93 μ**M** was measured during autumn season at site IV. The highest mean level of ammonia 4.25 ± 0.93 μ**M** was obtained at site III, whereas the lowest mean level 1.54 ± 0.32 μ**M** was obtained at site V (Table 3). Concentration of dissolved ammonia was significantly ($P<0.05$) positively correlated with percentage cover of macroalgae and chlorophyll *a* concentration in seawater (r^2 = 0.75 and 0.75 respectively, Fig. 7).

2. *Nitrite*:

Concentrations of nitrite in the area of study were relatively low. The highest concentration of 0.48 μ**M** was registered during spring at site I, while the lowest concentration of 0.01 μ**M** was registered during summer at site VII and autumn at sites IV & V (Table 2).

Site I supported the largest mean concentration of nitrite 0.18 ± 0.1 μ**M**, while site VII supported the lowest mean value 0.04 ± 0.01 μ**M** (Table 3).

3. *Nitrate*:

Nitrate concentration showed a distinct seasonal variation in each sampled site. The highest nitrate level 1.34 μ**M** was recorded during spring at site I, while the lowest level 0.03 μ**M** was recorded during autumn at site V (Table 2). Site III had the greatest mean concentration of nitrate 0.63 ± 0.14 μ**M**, whereas site IV had the lowest mean 0.21 ± 0.06 μ**M** (Table3)

4. *Total dissolved inorganic nitrogen* (DIN):

It is the summation of dissolved ammonia, nitrate and nitrite. Trends in DIN concentrations were similar to those in ammonia, which is the major contributor of nitrogen to DIN (Tables 2 and 3). Site III showed higher mean concentration of DIN 5.02 ± 0.95 μ**M** and site V showed lower mean concentration 2.07 ± 0.37 μ**M** (Table 3and Fig. 3). DIN showed high significant (P<0.05) positive correlation with macroalgal cover and concentration of chlorophyll *a* in seawater (r^2 = 0.78 and 0.74 respectively, Fig. 7).

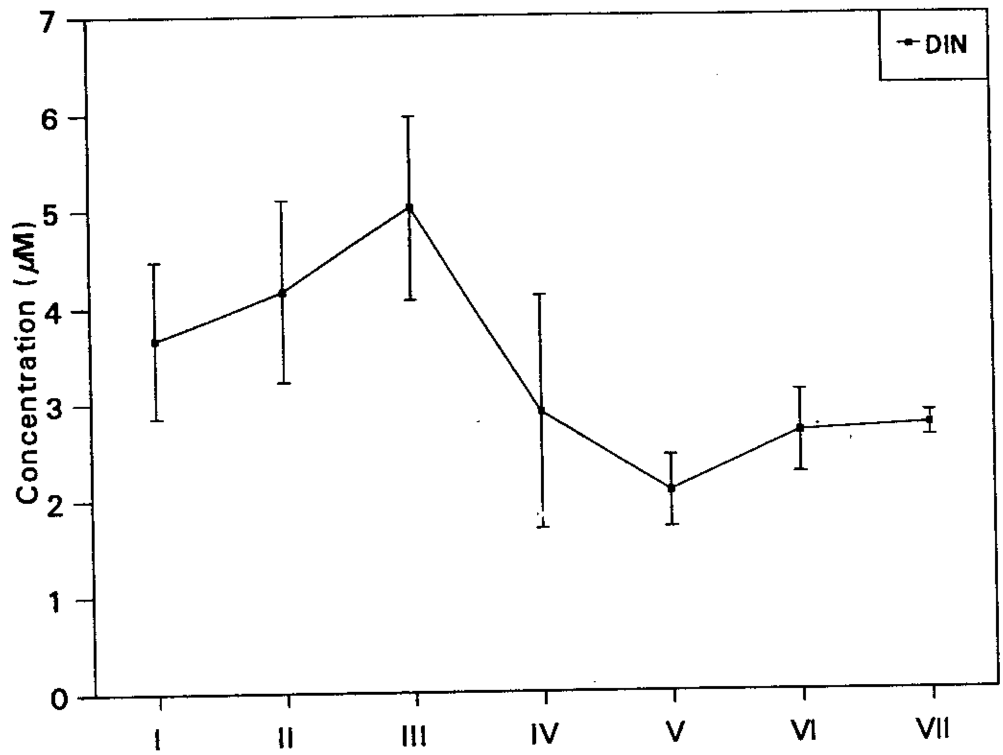
5. *Phosphate*:

Phosphate eutrophication of reef areas may be considered as one of a significant factors in creating better conditions for algal growth and thus, increasing competition for space between algae and corals.

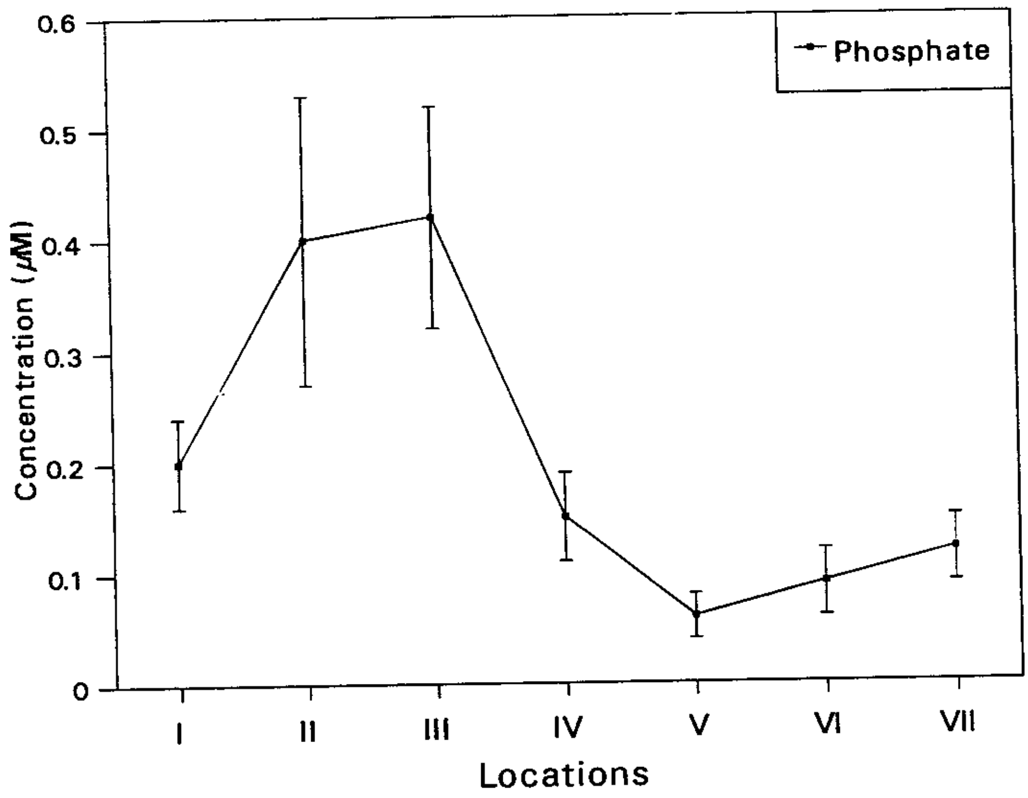
Dissolved phosphate concentration in the surveyed locations varied between a maximum of 0.76 μ**M** during autumn season at location II and a minimum of 0.02 μ**M** during spring at location V (Table 2).

Phosphate concentration was significantly different among the studied reef locations (ANOVA, F = 4.54, df = 21, P<0.01; Table 6). Mean concentration of phosphate showed a general decline towards the offshore reefs. The highest mean level of phosphate 0.42 ± 10 μ**M** was obtained at location III and the lowest value 0.06 ± 0.02 μ**M** was obtained at location V (Table 3 and Fig. 6).

Figure 7 illustrates a significant (P<0.05) positive correlation between dissolved phosphate concentrations and both of percentage cover of macroalgae (r^2 = 0.78) and concentration of chlorophyll *a* in seawater (r^2 = 0.77).



a



b

Fig. 6. Mean concentrations (μM) of total dissolved inorganic nitrogen (DIN) (**a**) and dissolved phosphate (**b**) in seawater at the studied locations. Vertical bars indicate standard error.

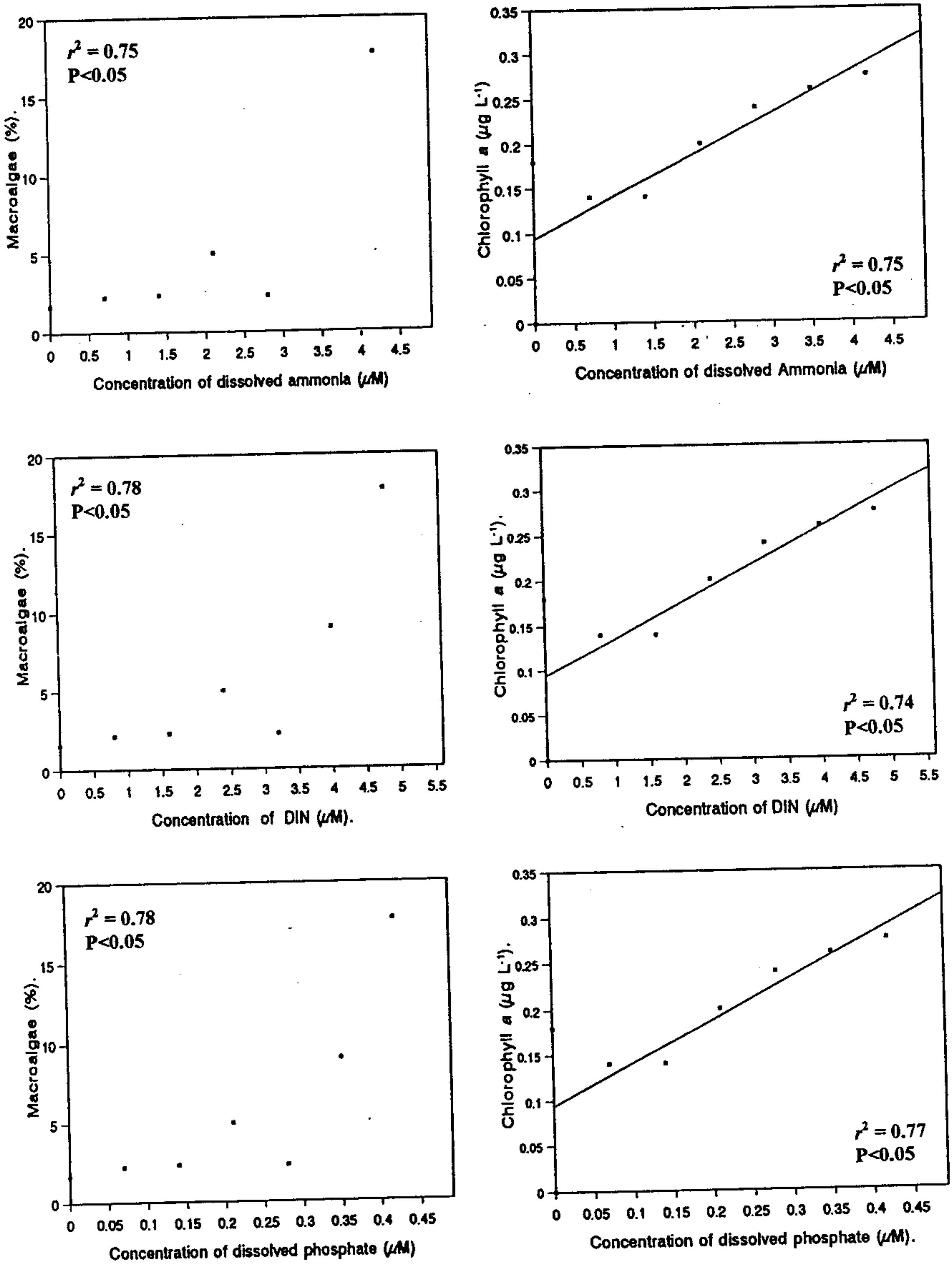


Fig. 7. Relationships between dissolved nutrient concentrations and both of chlorophyll *a* concentration in seawater and percentage cover of macroalgae.

Chlorophyll *a*:

By measuring the amount of chlorophyll *a* in a water sample, the concentration of phytoplankton can be estimated. Because a nutrient influx may cause a phytoplankton blooms, an increase in chlorophyll *a* might also indicate an increase in nutrients (Fig. 7).

Table 2 demonstrates the seasonal variation of chlorophyll *a* concentration at the sampling locations during 1998 and 1999. Levels of chlorophyll *a* showed obvious fluctuations from one season to another in locations IV, V, VI and VII. The highest value of 0.42 μg L^{-1} was recorded during summer season at location V, while the lowest value of 0.01 μg L^{-1} was recorded during winter at location IV. Chlorophyll *a* concentration did not significantly differ among the studied locations (ANOVA, $F = 0.98$, $df = 21$, $P>0.05$; Table 6). The maximum average value of chlorophyll *a* concentration 0.28 ± 0.02 μg L^{-1} was found at location III and the minimum average value 0.14 ± 0.07 μg L^{-1} was found at locations VI and VII (Table 3 and Fig. 9). This indicates that location III was the most productive area for flourishing phytoplankton.

Suspended particulate matter (SPM):

The seasonal variation of SPM concentration during 1998 and 1999 is shown in Table 4. The maximum concentration of SPM (15 mg L^{-1}) occurred in winter season at site I and the minimum concentration (5 mg L^{-1}) occurred at sites V and VII during summer and autumn seasons, respectively. The results of one-way ANOVA revealed a significant variations in SPM concentration among the investigated sites (ANOVA, $F = 6.56$, $df = 21$, $P<0.01$; Table 6). Site I reflected the highest mean concentration of SPM 12.52 ± 1.18 mg L^{-1}, whereas site V reflected the lowest mean concentration 5.52 ± 0.31 mg L^{-1} (Table 5 and Fig. 10). The

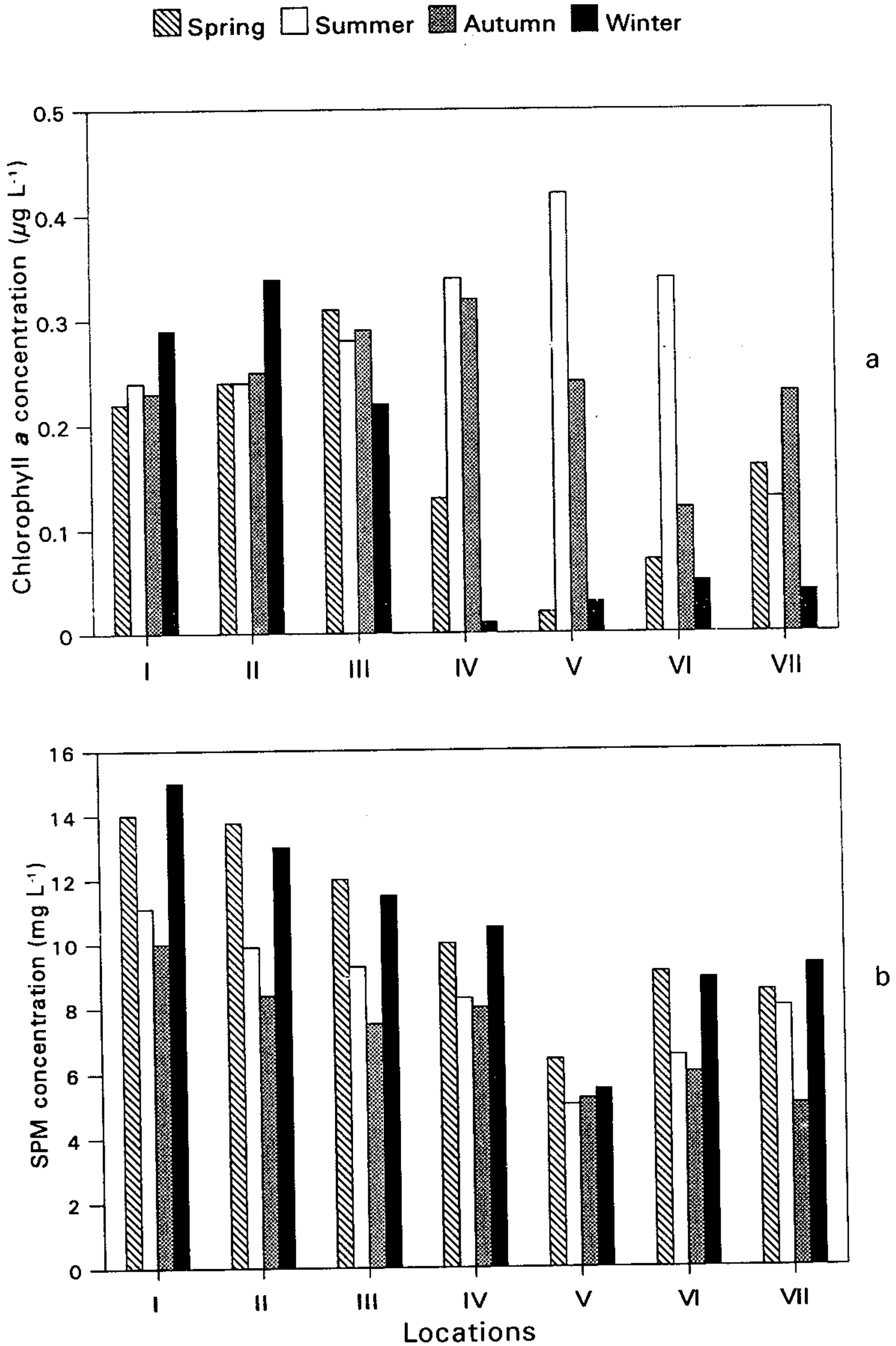


Fig. 8. Seasonal variation of chlorophyll *a* (**a**) and total suspended particulate matter (SPM) (**b**) concentrations in seawater at the investigated locations during 1998 and 1999.

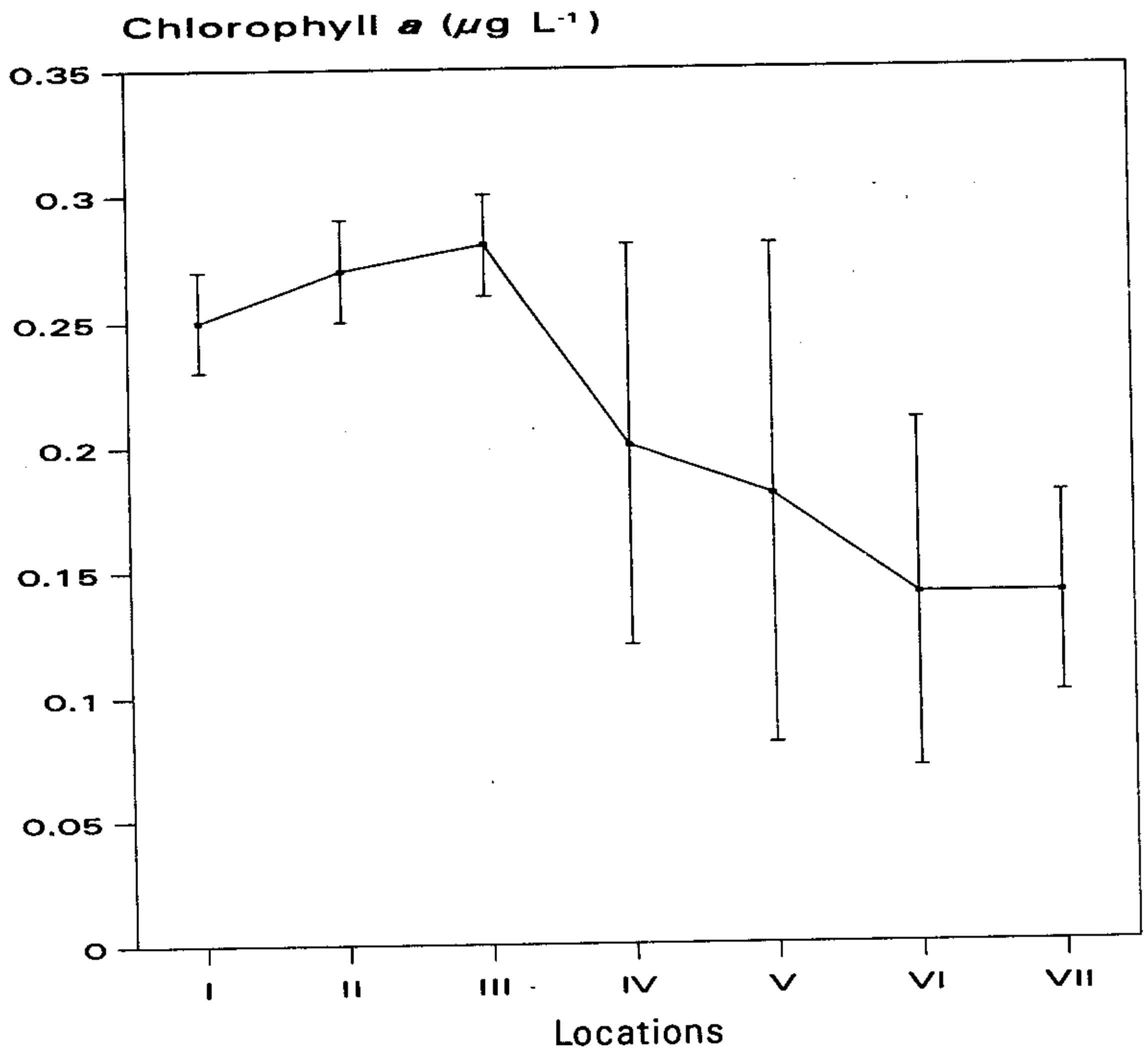


Fig. 9. Mean concentrations ($\mu g\ L^{-1}$) of chlorophyll *a* in seawater at the investigated locations. Vertical bars indicate standard error.

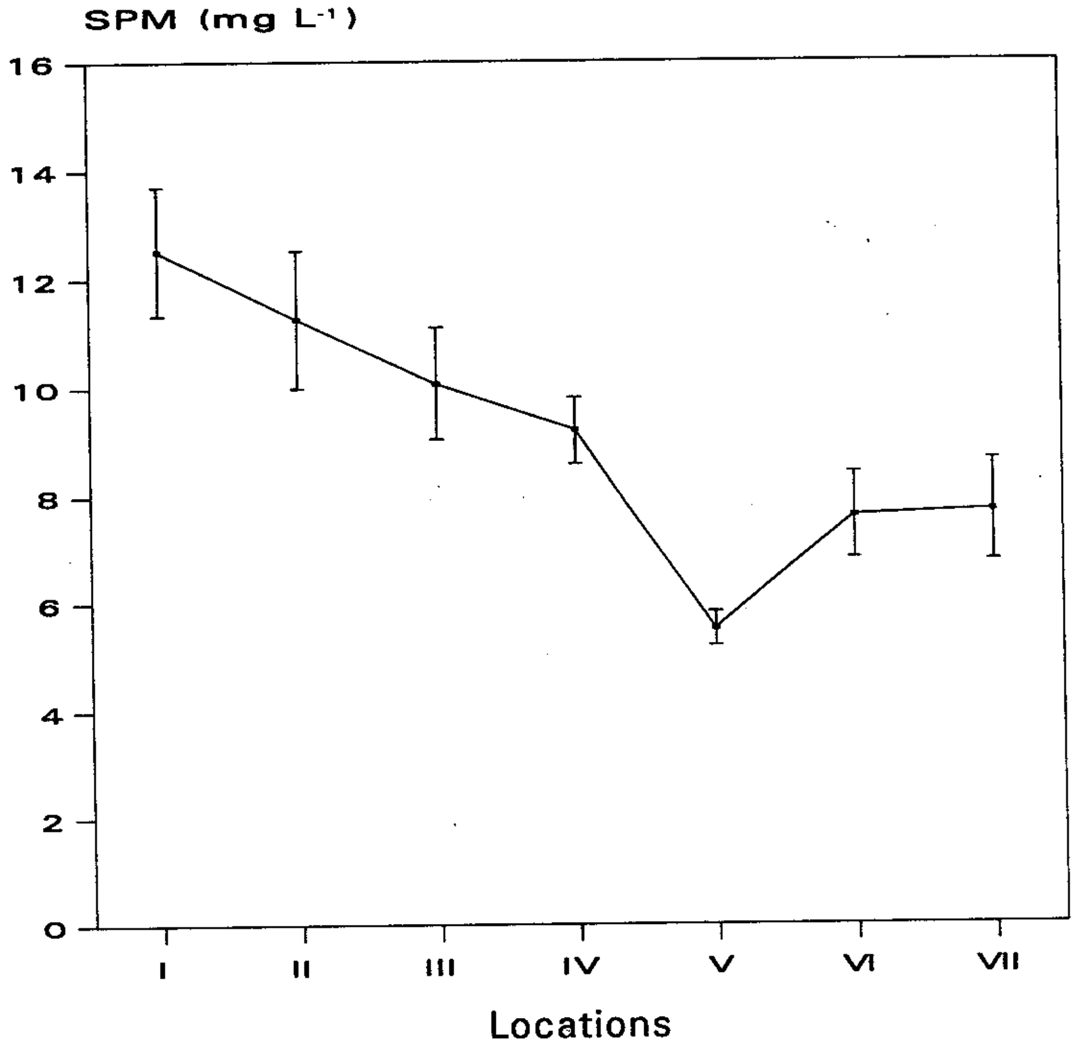


Fig. 10. Mean concentrations (mg L^{-1}) of total suspended particulate matter (SPM) in seawater at the surveyed locations. Vertical bars indicate standard error.

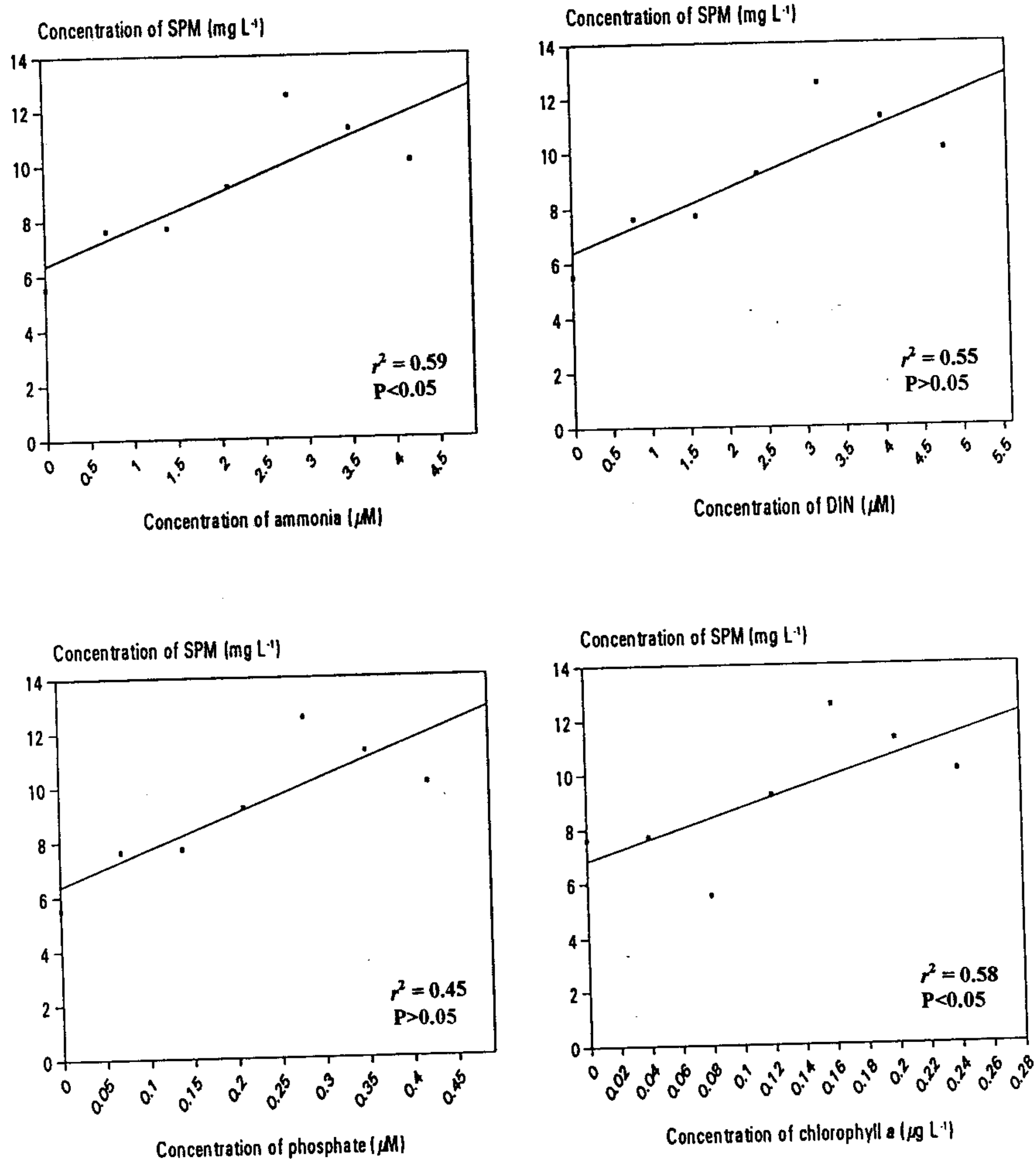


Fig. 11. Relationships between concentrations of dissolved nutrients and total suspended particulate matter (SPM).

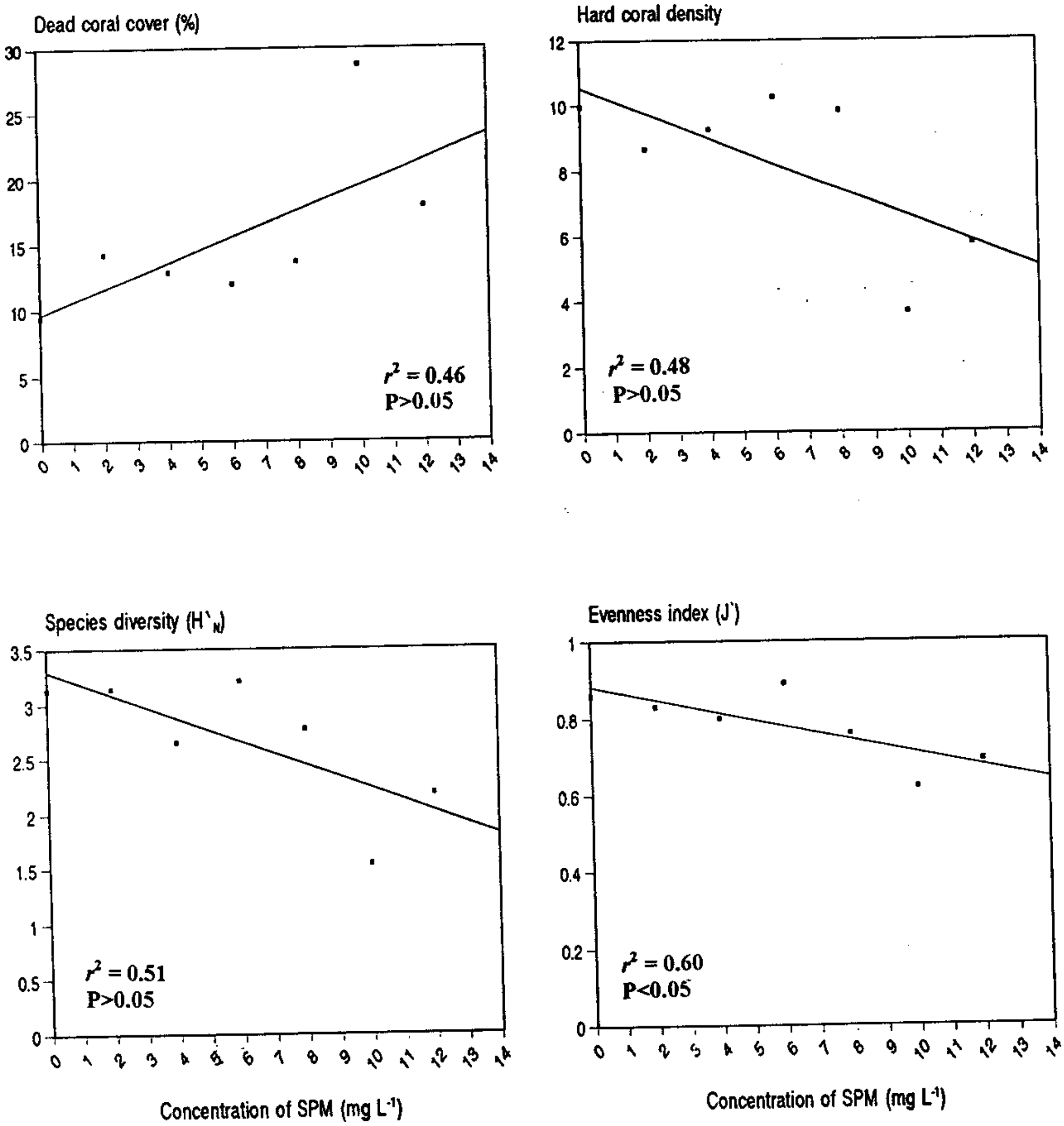


Fig. 12. Relationships between concentration (mg L^{-1}) of total suspended particulate matter (SPM) in the seawater and some coral community variables.

data in Table 4 demonstrates that SPM concentration was higher during spring and winter seasons than summer and autumn seasons in the surveyed areas. The strong winds, which occurred during winter and spring, stir up reef sediments and increase the rate of runoff from land.

SPM concentration showed a general decrease from north to south and from nearshore to offshore reefs in the area of present study (Table 5 & Fig. 10). SPM concentration was significantly and positively correlated with concentrations of dissolved ammonia and chlorophyll *a* (r^2 = 0.59 and 0.58 respectively, P<0.05; Fig. 11). On the other hand, SPM concentration showed strong positive correlation with concentration of DIN (r^2 = 0.55, P>0.05) and moderate positive correlation with concentration of phosphate (r^2 = 0.45, P>0.05, Fig. 11).

The relationship between SPM concentration and some of coral community parameters is illustrated in Figure 12. Evenness index was significantly negatively correlated with SPM concentration (r^2 = 0.60, P<0.05), whereas hard coral density and species diversity showed considerable inverse correlations with SPM concentration (r^2 = 0.48 and 0.51 respectively, P>0.05). There was a moderate positive correlation between SPM concentration and percentage of dead coral cover (r^2 = 0.46, P>0.05).

Sedimentation rate (SDR):

Data on sedimentation rate are especially important for coral reefs which are vulnerable to sedimentation from various sources either natural or anthropogenic. Given information about the amount of sediment introduced into the reef environment, the coral community structure and coral health; we should be able to predict the consequences of a high rate of terrestrial sediment input. The given Data in Table 4 illustrates the

Table 4. Seasonal variation of total suspended particulate matter (SPM) and total dissolved/dispersed petroleum hydrocarbons (DDPH) concentrations in seawater, sedimentation rate (SDR), and percentages of total carbonates and total organic matter (TOM) in reef sediments at the investigated reef sites during 1998 and 1999.

Parameter	Season	Location						
		I	II	III	IV	V	VI	VII
SPM (mg L^{-1})	Spring	14.00	13.75	12.00	10.00	6.40	9.10	8.50
	Summer	11.10	9.90	9.29	8.30	5.00	6.50	8.00
	Autumn	10.00	8.40	7.50	8.00	5.20	6.00	5.00
	Winter	15.00	13.00	11.50	10.50	5.50	8.90	9.33
DDPH (μg L^{-1})	Spring	4.50	6.20	3.42	4.05	5.50	6.66	12.42
	Summer	44.08	114.45	6.33	7.50	14.66	11.33	20.00
	Autumn	19.08	6.75	15.50	9.42	11.83	10.42	5.25
	Winter	12.58	11.92	8.66	8.00	5.72	8.58	15.12
SDR (mg cm^{-2} day^{-1})	Spring	64.34	257.75	8.30	5.36	5.30	3.80	8.50
	Summer	62.83	220.08	13.40	4.20	4.00	3.70	6.80
	Autumn	66.93	537.41	12.00	2.30	5.40	4.00	7.40
	Winter	68.50	255.80	12.70	3.00	4.20	4.50	8.30
Carbonate (%)	Spring	68.80	38.75	70.22	86.25	94.60	87.84	87.90
	Summer	73.27	43.40	76.73	92.00	92.85	90.90	90.60
	Autumn	71.35	47.40	72.23	88.30	86.55	90.50	88.50
	Winter	65.70	40.20	74.67	89.40	88.65	87.20	86.30
TOM (%)	Spring	13.77	4.17	5.58	3.51	1.82	5.09	4.90
	Summer	14.92	4.46	2.27	1.56	1.60	1.50	5.05
	Autumn	9.72	4.96	5.18	2.35	3.15	3.20	5.20
	Winter	10.17	4.70	7.46	5.50	5.19	6.03	5.70

Table 5. Mean values of total suspended particulate matter (SPM) and total dissolved / dispersed petroleum hydrocarbons (DDPH) concentrations in seawater, total petroleum hydrocarbon concentrations in reef sediments (TPH), sedimentation rate (SDR), and percentages of total carbonates and total organic matter (TOM) in reef sediments at the investigated locations. Standard errors are presented in parentheses.

Parameter	Location						
	I	II	III	IV	V	VI	VII
SPM (mg L^{-1})	12.52 [1.18]	11.26 [1.27]	10.07 [1.04]	9.20 [0.62]	5.52 [0.31]	7.62 [0.80]	7.71 [0.94]
DDPH (μg L^{-1})	20.06 [8.54]	34.83 [26.57]	8.48 [2.57]	7.24 [1.14]	9.43 [2.28]	9.25 [1.03]	13.20 [3.08]
TPH (μg g^{-1} dry wt)	21.79 [11.07]	52.85 [8.33]	4.10 [0.90]	3.24 [0.56]	1.40 [0.01]	4.77 [0.14]	17.60 [10.65]
SDR (mg Cm^{-2} day^{-1})	65.65 [1.27]	317.76 [73.73]	11.60 [1.14]	3.71 [0.67]	4.72 [0.36]	4.00 [0.18]	7.75 [0.40]
Carbonate (%)	69.78 [1.64]	42.44 [1.92]	73.46 [1.42]	88.99 [1.20]	90.66 [1.85]	89.11 [0.93]	88.32 [0.89]
TOM (%)	12.15 [1.29]	4.57 [0.17]	5.12 [1.07]	3.23 [0.86]	2.94 [0.82]	3.95 [1.01]	5.21 [0.17]

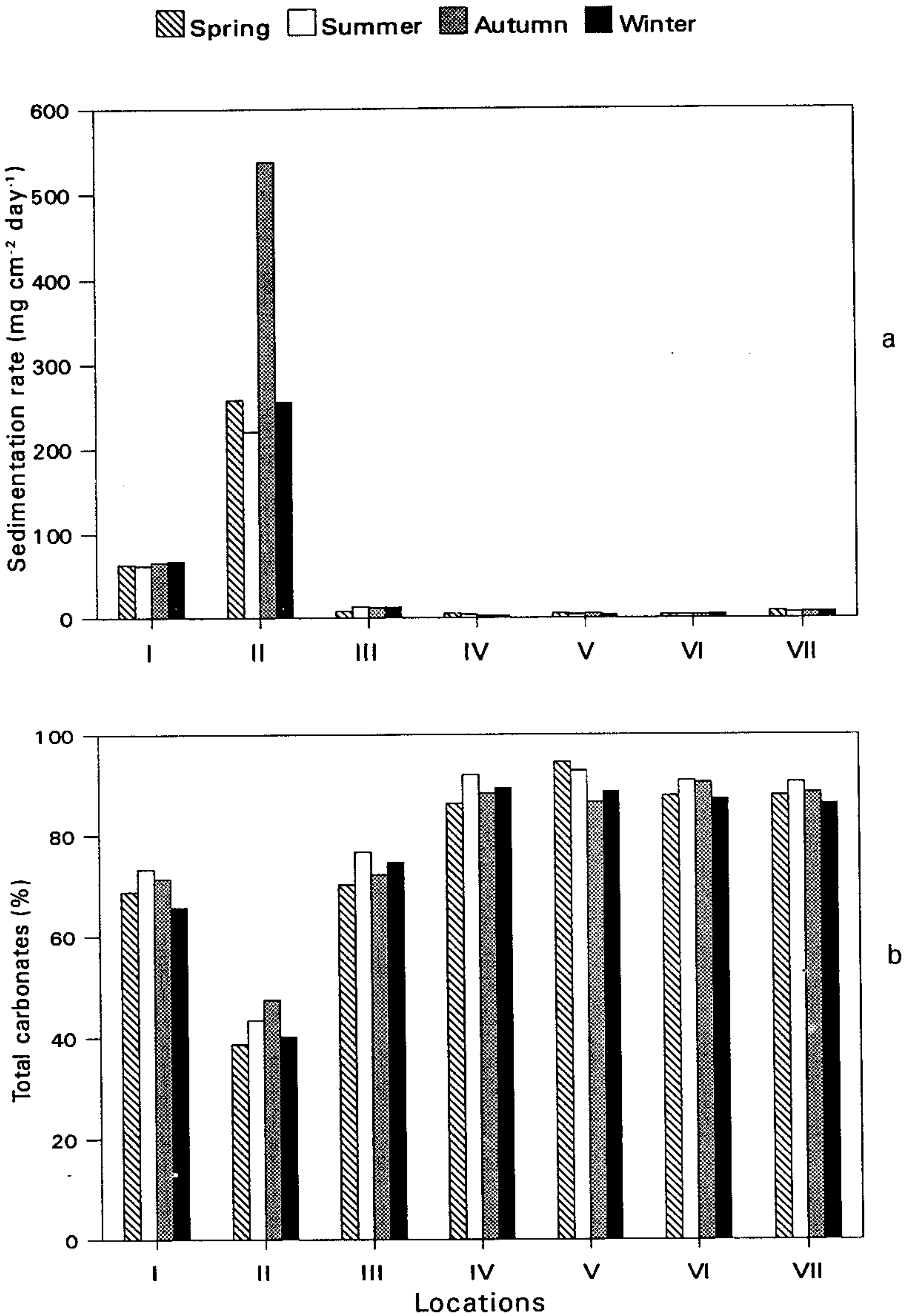


Fig. 13. Seasonal variation of sedimentation rate (**a**) and percentage of total carbonates in reef sediment (**b**) at the investigated locations during 1998 and 1999.

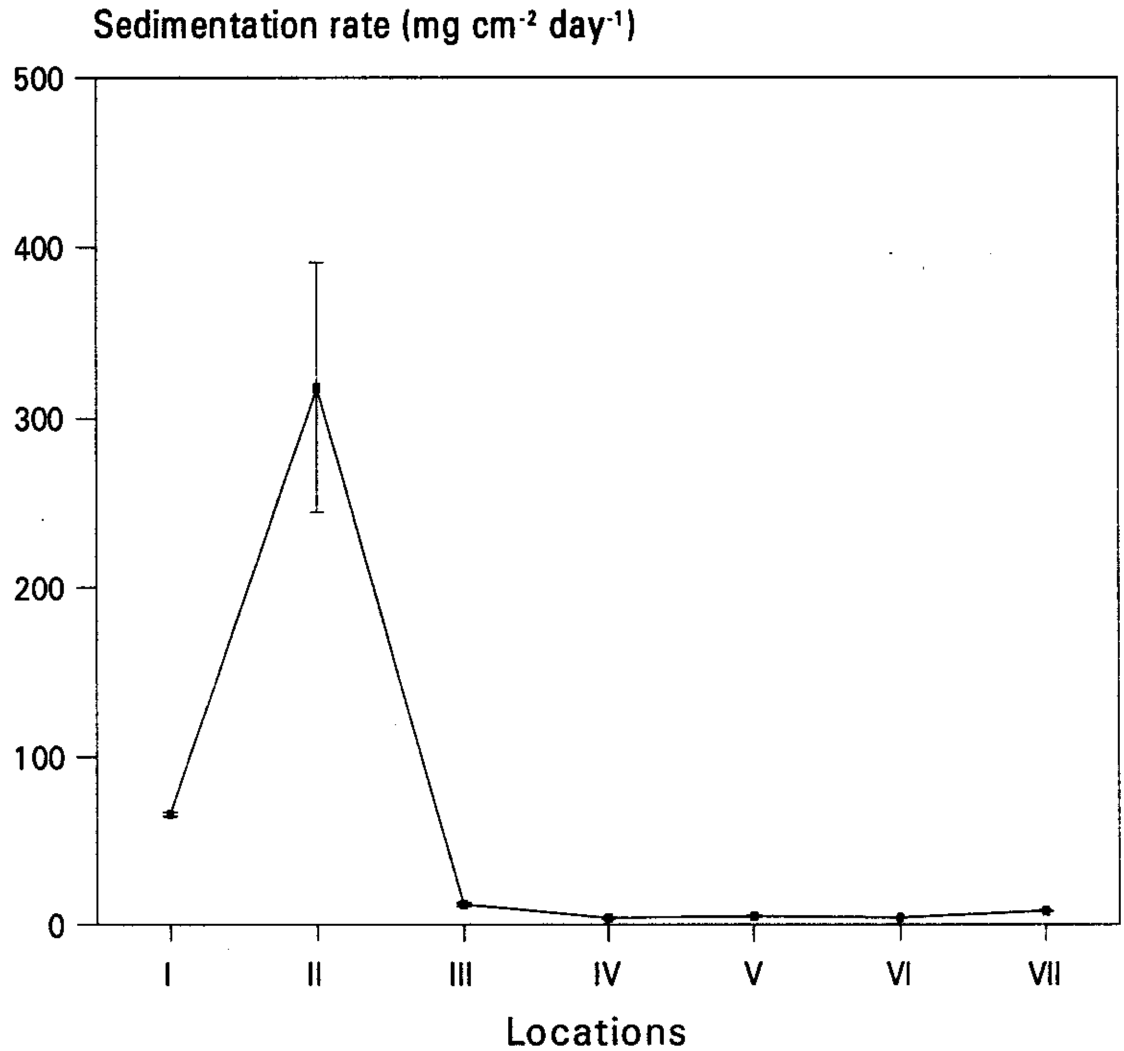


Fig. 14a. Mean sedimentation rates (mg cm^{-2} day^{-1}) at the studied locations. Vertical bars indicate standard error.

Table 6. Analysis of variance of environmental variables among the studied locations. The appropriate F statistics, degree of freedom (df) and probability of a significant difference are shown : * = P<0.05, * * = P<0.01 and NS = no significant difference at P>0.05.

Variable	df	F	P	Significance
Hydrocarbons in sediments	10.00	6.73	0.00	* *
Hydrocarbons in seawater	21.00	0.85	0.54	NS
Sedimentation rate	21.00	17.35	0.00	* *
Suspended particulate matter	21.00	6.56	0.00	* *
Dissolved ammonia	21.00	1.78	0.15	NS
Dissolved inorganic nitrogen	21.00	1.70	0.17	NS
Dissolved phosphate	21.00	4.54	0.00	* *
Chlorophyll *a* in seawater	21.00	0.98	0.46	NS
Total carbonates (%)	21.00	145.73	0.00	* *
Total organic matter (%)	21.00	12.97	0.00	* *

Table 7. Size distribution of deposited sediments at the investigated locations. Data are expressed as percentage.

Particle size	Location						
	I	II	III	IV	V	VI	VII
≥ 2000 µm	0.03	0.26	9.51	3.43	0.20	2.30	0.00
1000-1999 µm	0.06	0.21	15.41	7.19	1.21	9.01	0.01
500-999 µm	1.37	1.19	20.90	12.54	3.80	19.97	16.00
250-499 µm	2.57	4.54	21.43	20.25	10.17	26.47	41.15
125-249 µm	7.91	64.46	13.40	25.12	19.03	22.75	21.78
63-124 µm	60.30.	25.74	11.04	29.02	52.25	15.77	15.77
< 63 µm	27.76	3.60	8.31	2.45	13.34	3.73	5.29

seasonal variation of sedimentation rates at the different surveyed locations. The highest sedimentation rate of 537.41 mg cm^{-2} day^{-1} occurred during autumn at location II, whilst the lowest value of 2.30 mg cm^{-2} day^{-1} has been recorded also during autumn but at location IV. Analysis of data on the sedimentation rate at the different studied locations using one-way ANOVA revealed that a very high significant difference was found among the investigated locations (ANOVA, F = 17.35, df = 21, P<0.01; Table 6). For the entire study period, location II represented the maximum mean deposition rate 317.76 ± 73.73 mg cm^{-2} day^{-1}. In contrast, location IV represented the minimum value 3.71 ± 0.67 mg cm^{-2} day^{-1} (Table 5 and Fig. 14a).

Sedimentation rate was significantly correlated with coral community variables (Fig. 15). Live hard coral cover (%) was significantly decreased with increased sedimentation rate (r^2 = 0.65, P<0.05). In contrast, dead coral cover (%) showed a very high significant positive correlation with the rate of sediment accumulation (r^2 = 0.92, P<0.05). Thus, coral mortality would be strongly accelerated with the excessive sediment deposition on the surveyed reefs. Species diversity, evenness index and species richness exhibited a high significant reduction with enhanced sedimentation rate (r^2 = 0.79, 0.68 and 0.71 respectively, P<0.05). Density of hard corals was also showed a high significant negative correlation with sedimentation rate (r^2 = 0.79, P<0.05).

Population density of zooxanthellae was significantly negatively correlated with the rate of sedimentation (r^2 = 0.62, P<0.05; Fig. 16). Consequently, number of zooxanthellae per cm^2 of coral surface area was conspicuously reduced in the heavily-sedimented locations.

Table 7 and figure 14b indicate the percentage of deposited sediment sizes at the different studied reef sites. At site I (El-Ain Al-Sukhna) 88.06% of deposited sediment sizes is less than 125 µm (60.30%

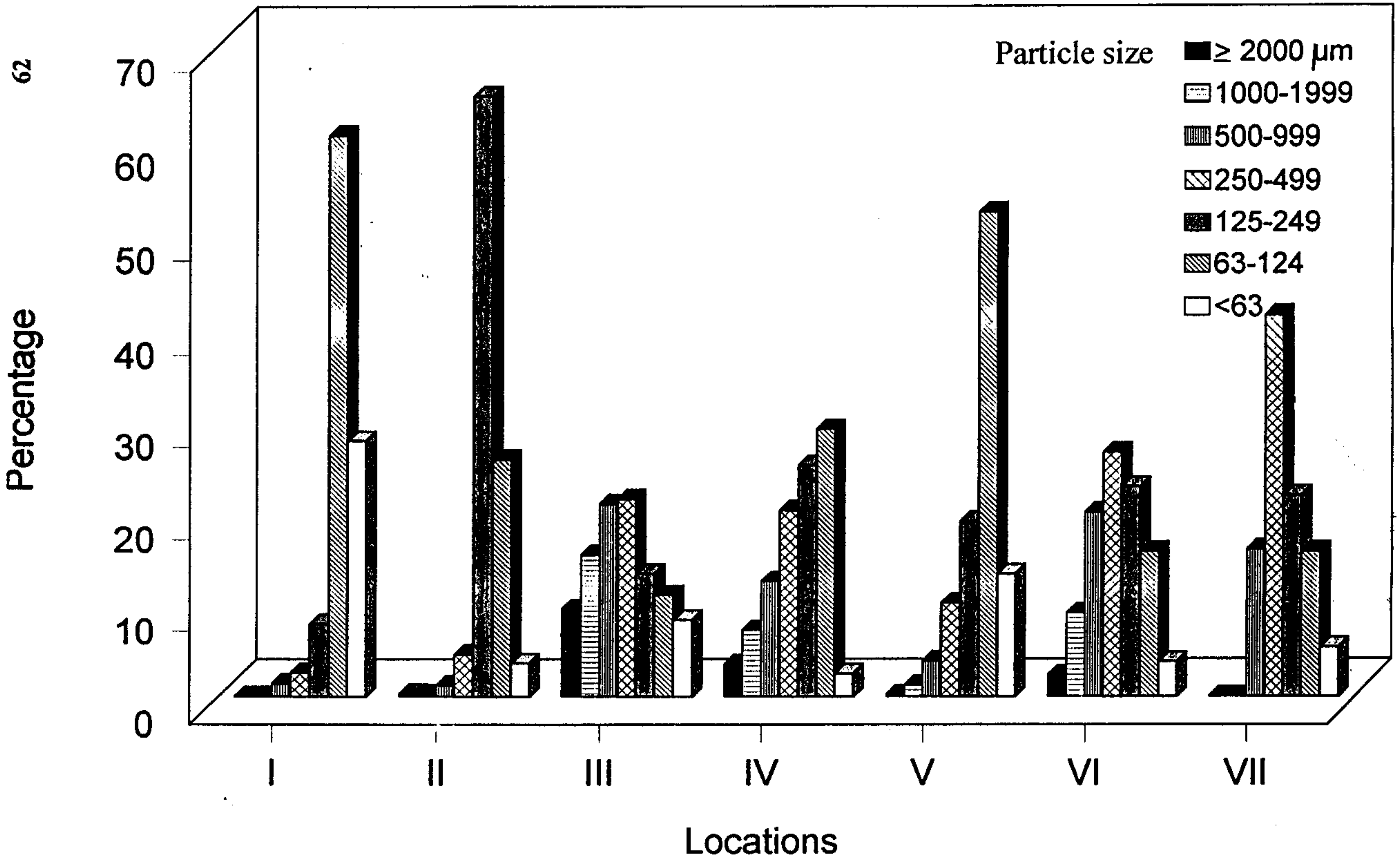


Fig. 14b. Size distribution of deposited sediments at the investigated locations.

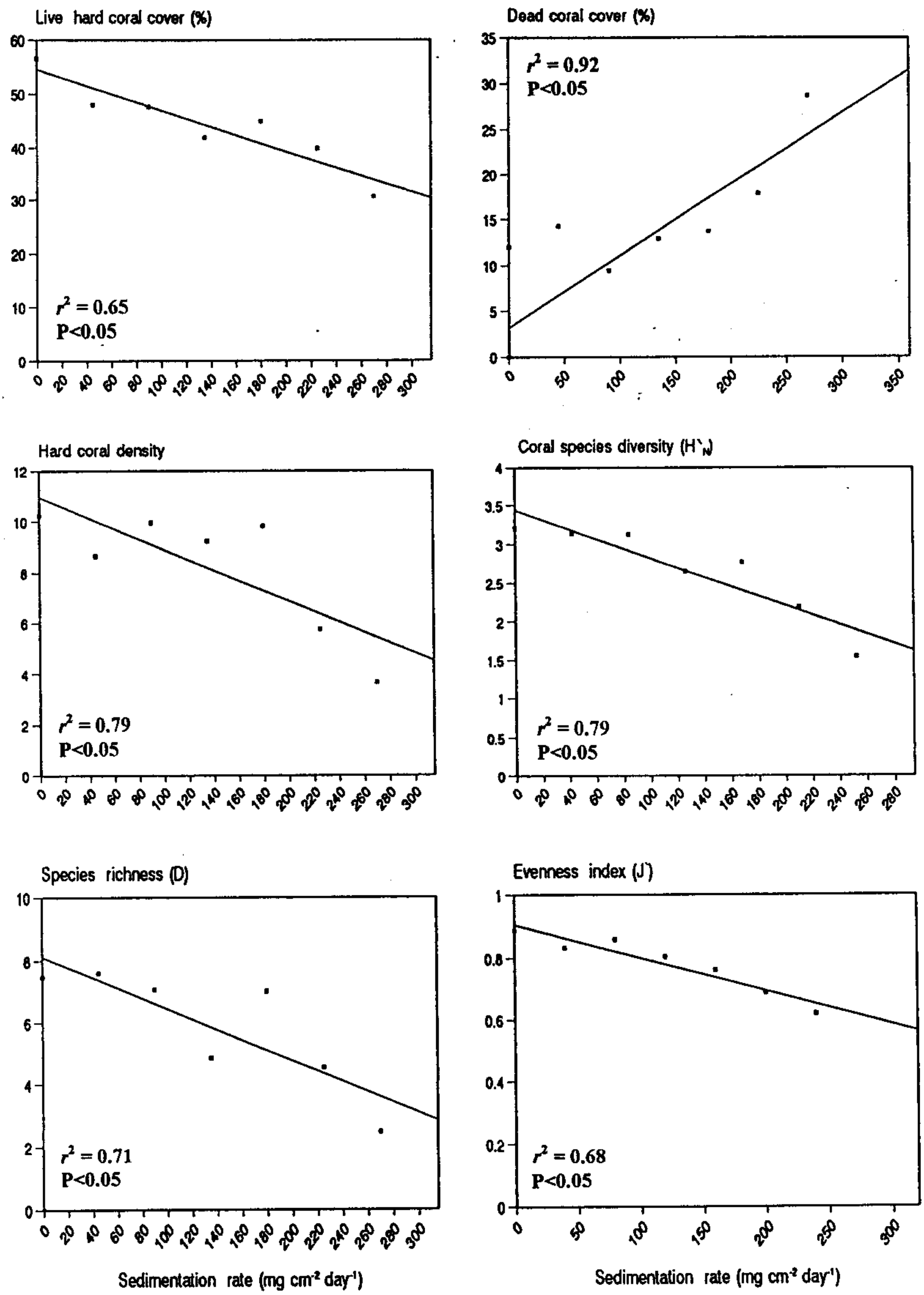


Fig. 15. Relationships between coral community variables and sedimentation rate.

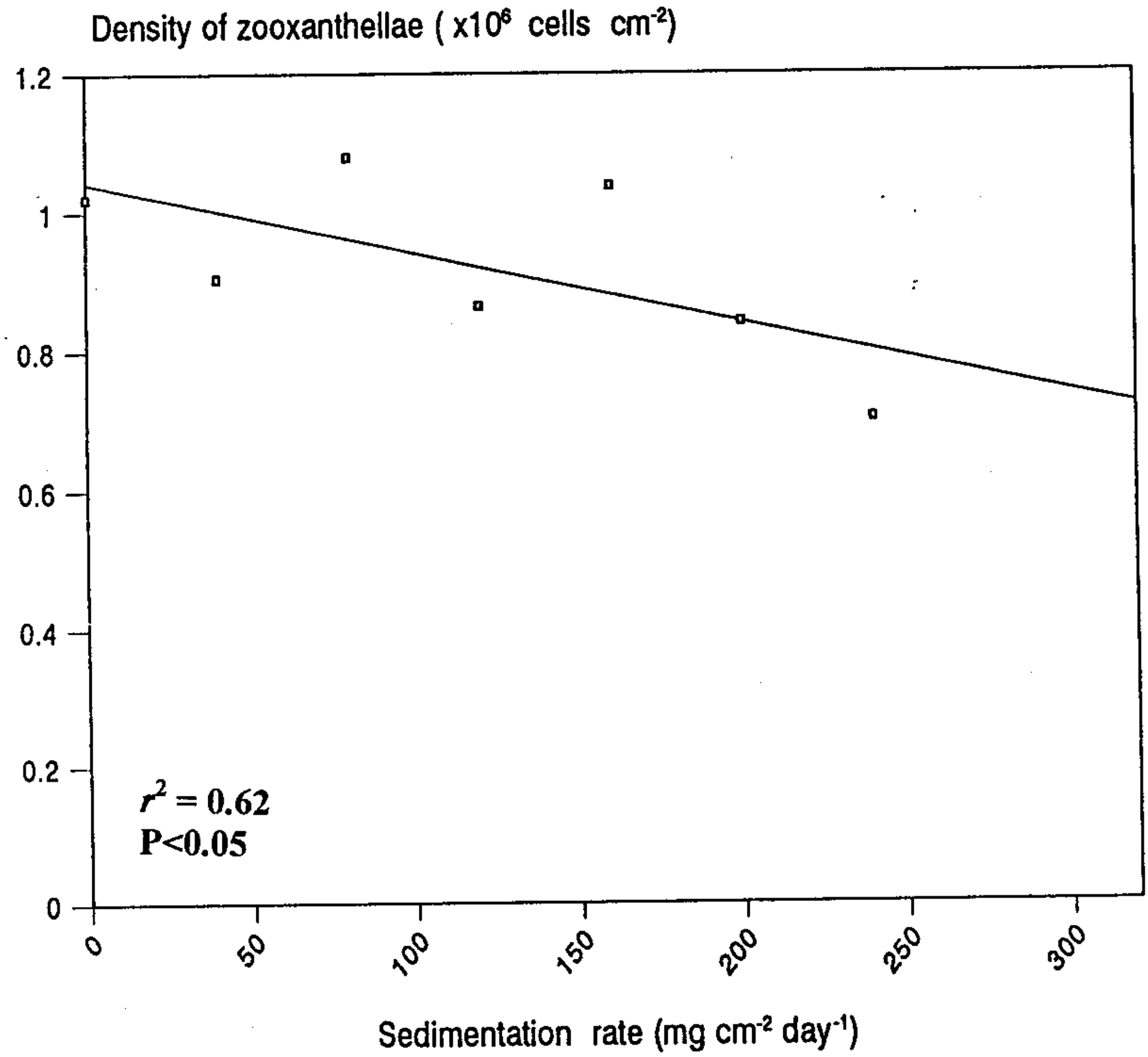


Fig. 16. Relationship between sedimentation rate and zooxanthellae density ($\times 10^6$ cells cm^{-2}).

in the size class 63-124 μm and 27.76% is <63 μm). The results also show that 90.20% of the deposited sediment sizes at Ras Za'farana (Site II) is less than 250 μm and larger than 63 μm, where 64.46% and 25.74% are in the size classes 125-249 μm and 63-124 μm respectively. On the contrary to sites I and II, 57.74% of the sediment sizes deposited at the Marine Biological Station (Site III) falls in the size ranges 250-499 μm, 500-999 μm and 1000-1999 μm (21.43, 20.90 and 15.41% respectively). At site IV (Sha'b Saad) 74.39% of the deposited particle sizes is restricted to the size classes 63-124 μm (29.02%), 125-249 μm (25.12%) and 250-499 μm (20.25%). The highest percentage (52.25%) of the deposited sediment sizes at site V (Sha'b Abu-Galawa) falls in the size range 63-124 μm followed by the size classes 125-249 μm (19.03%) and <63 μm (13.34%). At site VI (Sha'b El-Fanadir) most of the deposited particle sizes are in the size classes 250-499 μm, 125-249 μm, 500-999 μm and 63-124 μm (26.74, 22.75, 19.97 and 15.77% respectively). The highest percentage (41.15%) of sediment sizes deposited at site VII (Gasus1) is 250-499 μm followed by the size class 125-249 μm (21.78%). It is observed that the highest percentage (27.76%) of sediment sizes <63 μm was deposited at site I, while the highest percentage (9.51%) of sediment sizes ≥2000 μm was deposited at site III.

Total carbonate content (%) in reef sediments:

Percentage of carbonate contents in reef sediments may provide information about the sources of these sediments. The seasonal variation of carbonate content (%) in the accumulated sediments at the different surveyed locations is illustrated in table 4. The percentage of total carbonates was ranged between 38.75% during spring at site II and 94.60% during spring at site V. The mean value and standard error for total carbonate content at each site was calculated (Table 5).

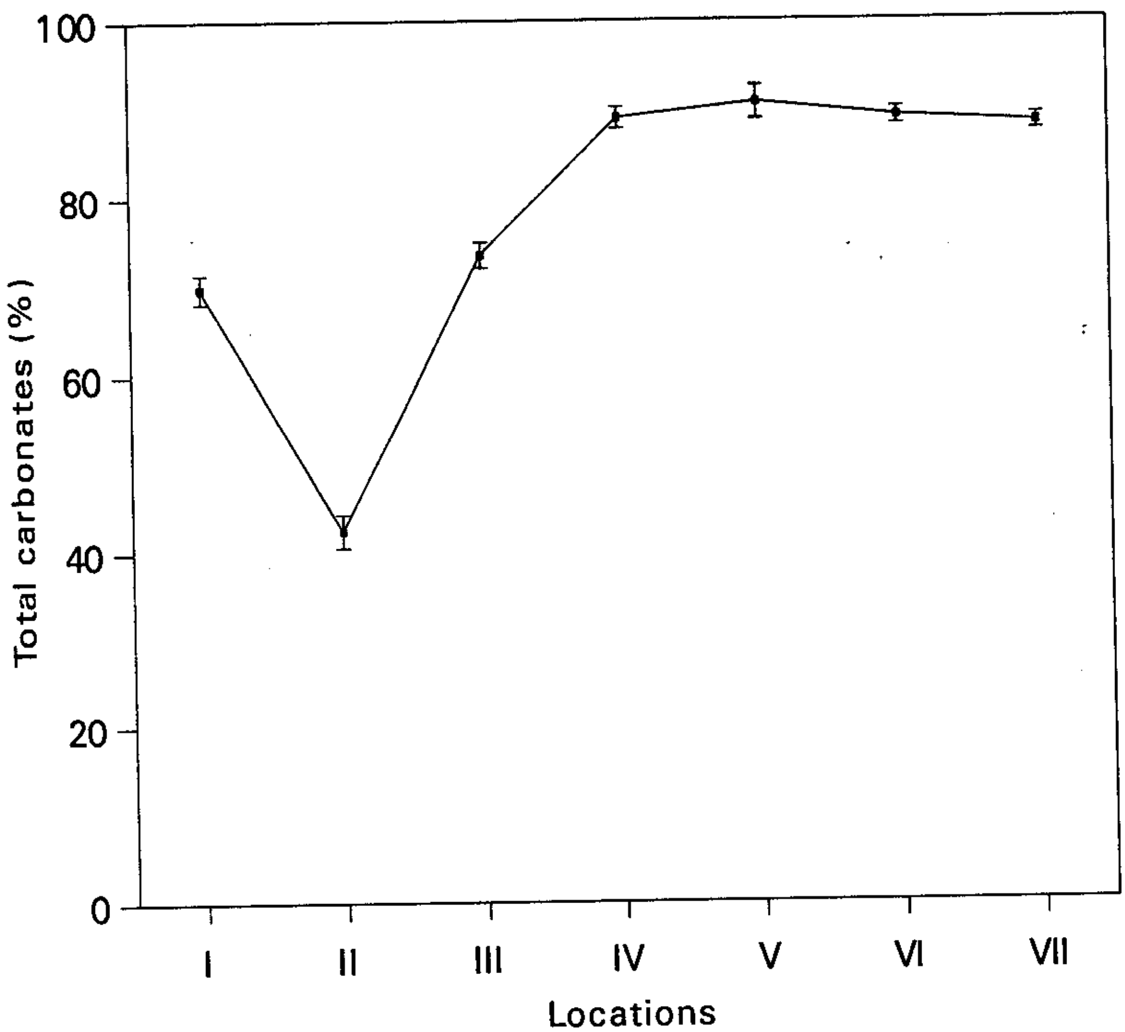


Fig. 17. Mean percentages (±SE) of total carbonates in reef sediments at the different sampled locations.

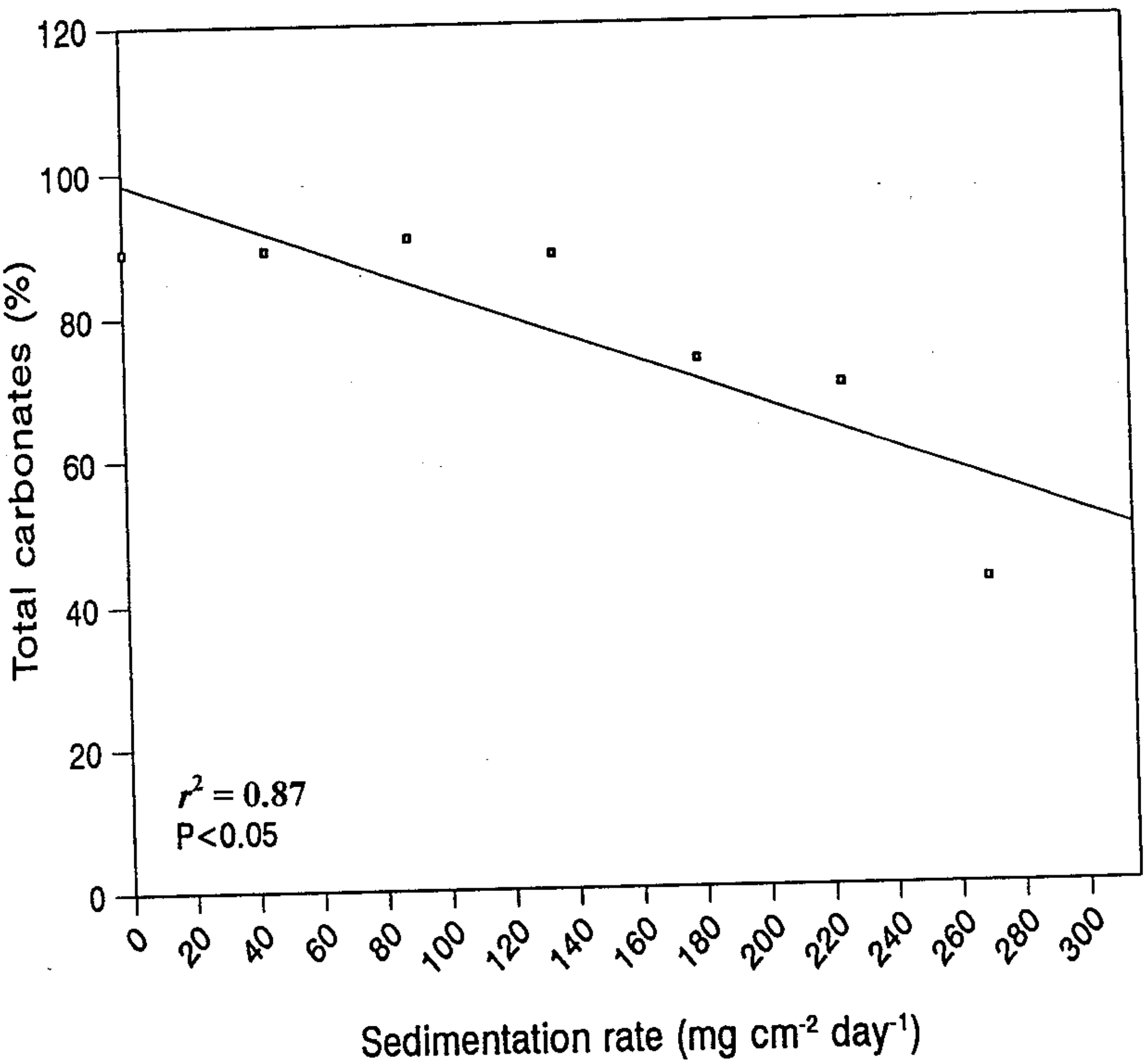


Fig. 18. Relationship between the total carbonate content (%) in reef sediments and sedimentation rate.

The percentage of total carbonate content showed a very high significant difference among the investigated reef areas (ANOVA, F = 145.73, df = 21, P<0.01; Table 6). The maximum mean of total carbonate content (90.66 ± 1.85%) was registered at site V, while the minimum mean (42.44 ± 1.92%) was registered at site II (Table 5 and Fig. 17). The results in table 5 exhibited a general trend of increasing total carbonate content in sediments away from the shore, and the northern part of the study area had lower values than southern part. A very high significant inverse correlation existed between total carbonate content (%) in reef sediments and sedimentation rate ($r^2 = 0.87$, P<0.05; Fig. 18).

Total organic matter (TOM) in reef sediments:

Seasonal variation of total organic matter content (%) in bottom sediments at the surveyed reefs is shown in table 4. The greatest percentage of TOM (14.92%) was recorded during summer at location I, whereas the lowest percentage (1.50%) was obtained at location VI during summer. Percentage of TOM was also differed significantly among the studied areas (ANOVA, F = 12.97, df = 21, P<0.01; Table 6). Location I (El-Ain Al-Sukhna) had the highest mean value of TOM content (12.15 ± 1.29%). On the other hand, location V (Sha'b Abu-Galawa) had the lowest mean 2.94 ± 0.86% (Table 5).

In contrast to the total carbonate content, TOM in reef sediments showed a general decline towards the offshore reefs (Fig. 20). Investigations of relationships between TOM percentages and the concentrations of studied trace metals in reef sediments indicated the presence of positive correlations (Fig. 21). These correlations were high and significant (P<0.05) in cases of Cu ($r^2 = 0.74$), Zn ($r^2 = 0.72$), Cr ($r^2 = 0.59$), Ni ($r^2 = 0.70$), Co ($r^2 = 0.76$), Fe ($r^2 = 0.80$) and Mn ($r^2 = 0.74$). On the other hand, the correlations between TOM percentages and

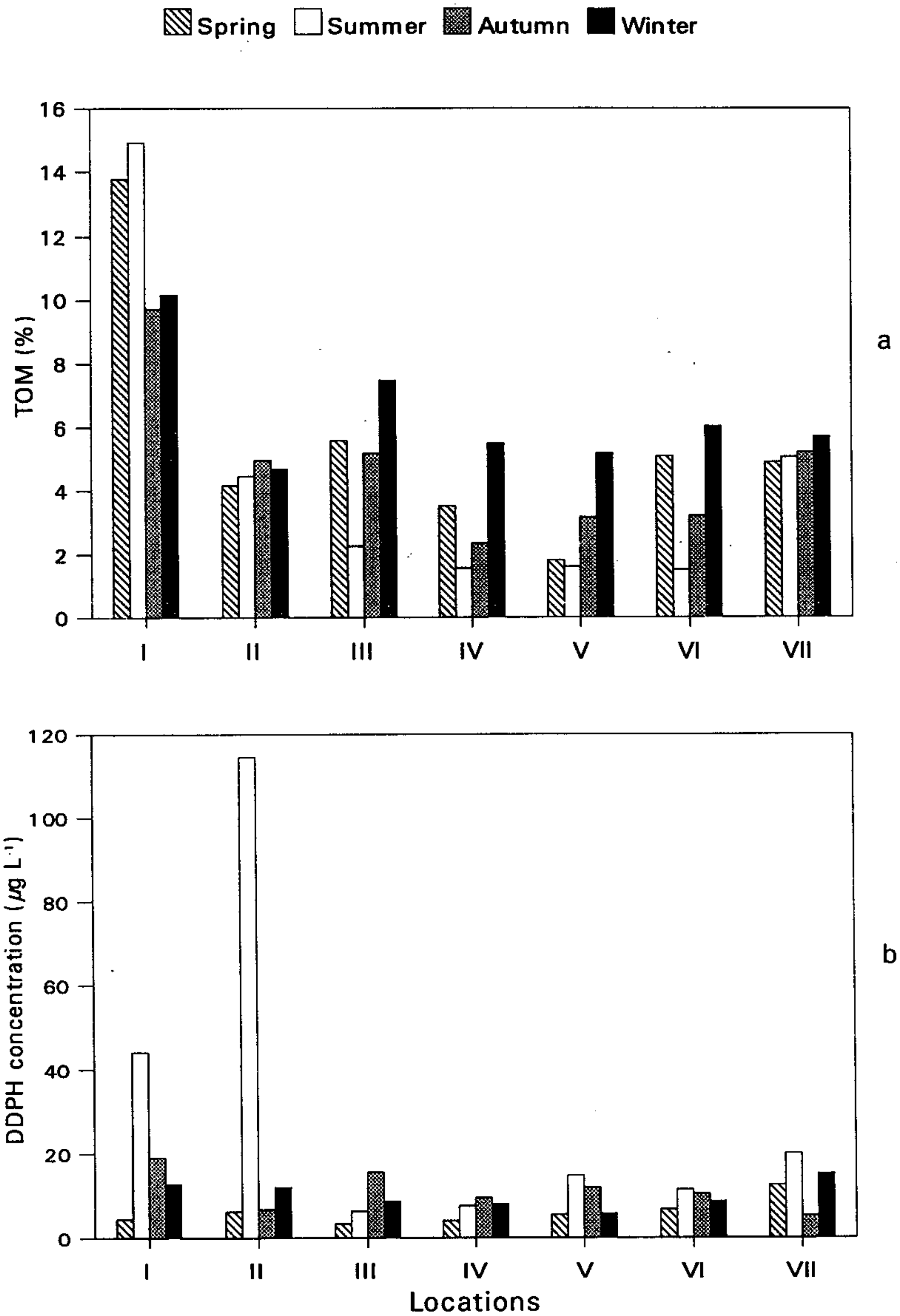


Fig. 19. **a:** Seasonal variation of total organic matter (TOM) percentage in reef sediments; and **b:** Seasonal variation of oil concentration in seawater (DDPH) at the investigated locations during 1998 and 1999.

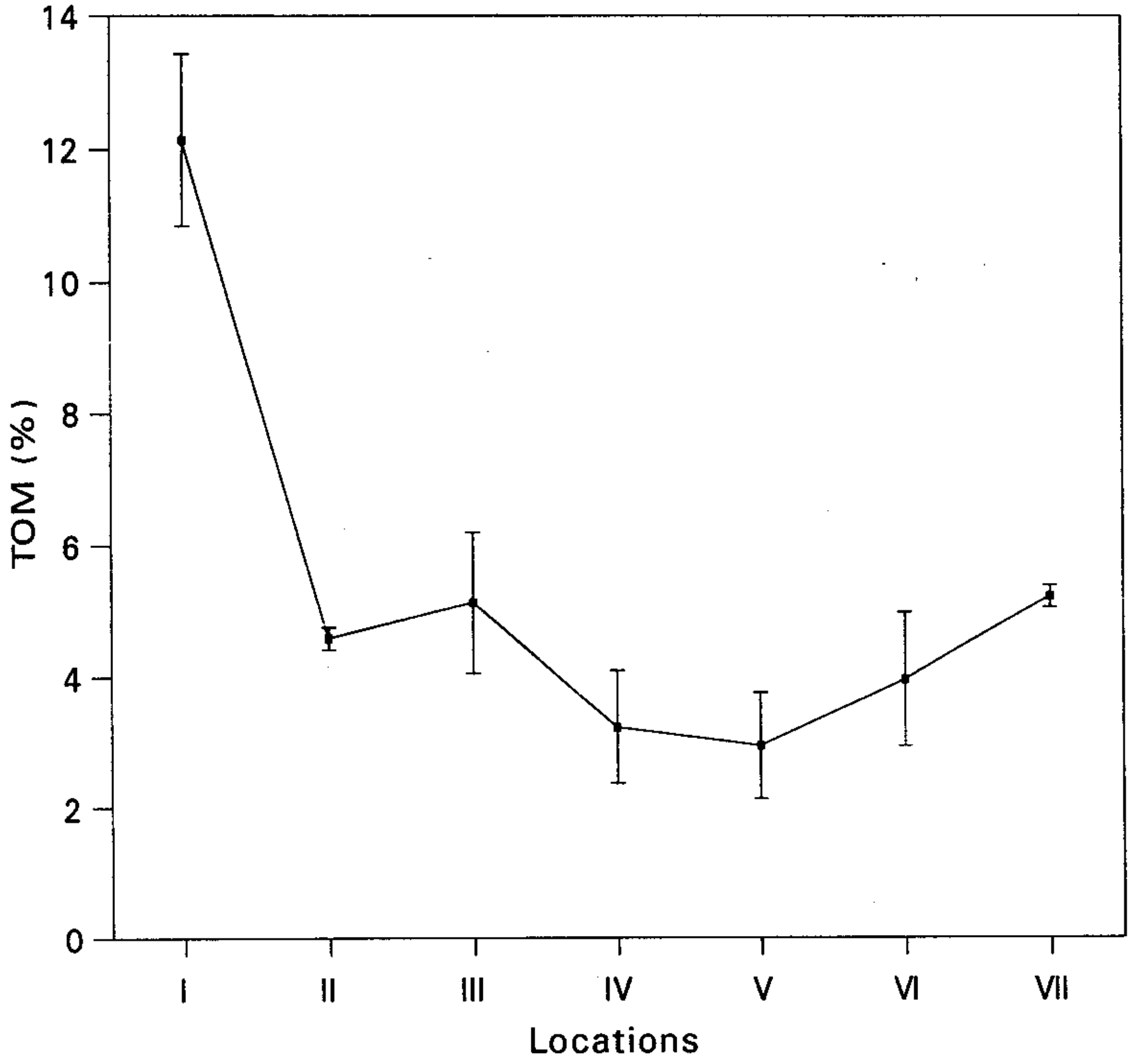


Fig. 20. Mean percentages (±SE) of total organic matter (TOM) in reef sediments at the different sampled locations.

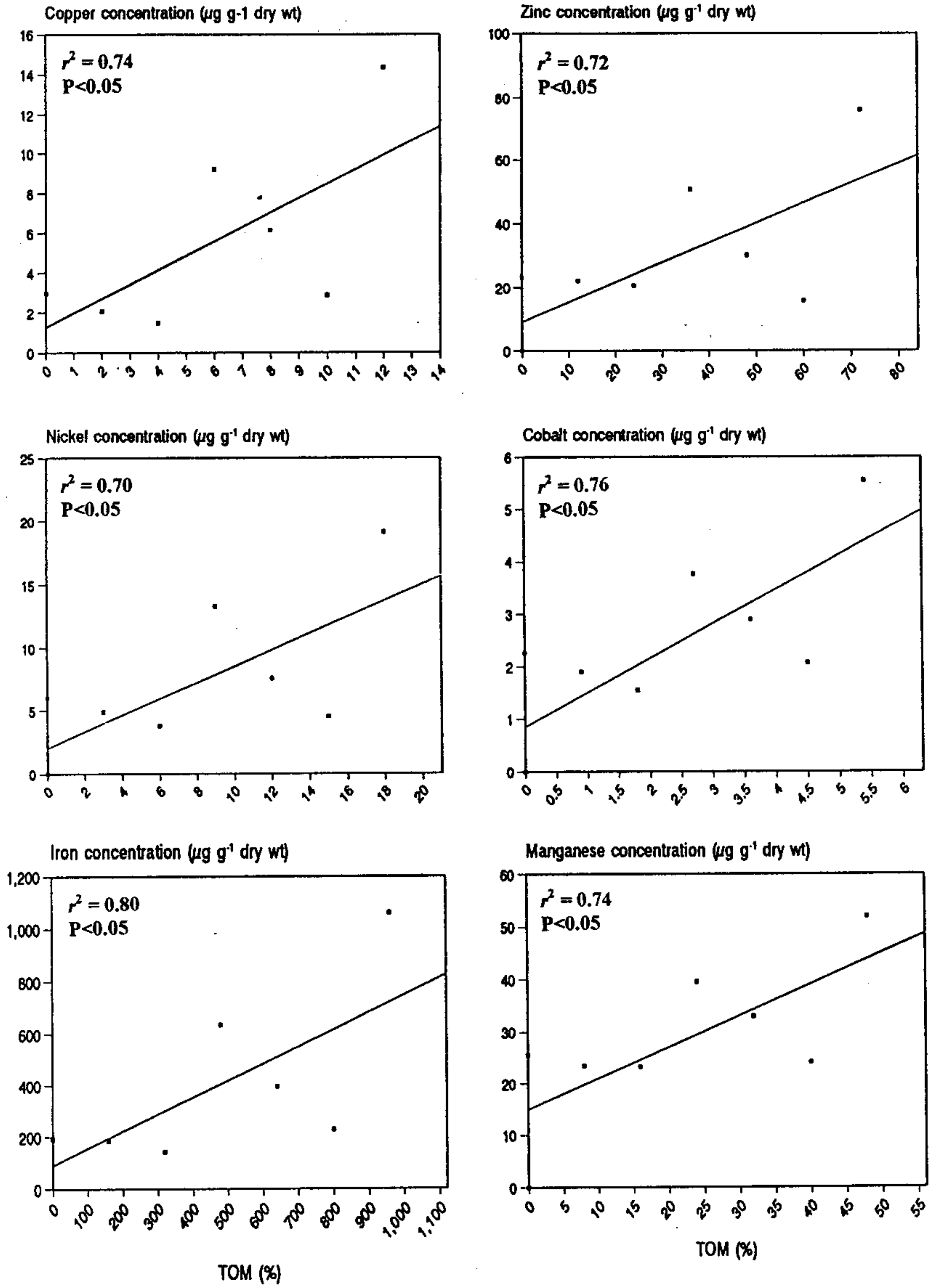


Fig. 21. Relationships between concentrations of different trace metals and percentage of total organic matter (TOM) in reef sediments.

concentrations of Pb and Cd were strong (r^2 = 0.55 and 0.51 respectively) but not significant (P>0.05).

Petroleum hydrocarbons:

First of all, some field observations was made during sample collection. The beaches were drastically fouled with residual oil, not only at the chosen locations but also along the coast from El-Ain Al-Sukhna (location I) to Gasus 1 (location VII). There were numerous oil slicks either on beaches or visible in the nearshore water. Although some beaches appeared clean, removal of the sandy superficial layer revealed a bed of accumulated residual oil, which sometimes extended to a depth of tens of centimeters. Indeed, the rocks of the few rocky beaches were usually completely covered with oil slicks. There were many spots containing considerable quantities of tar balls and tar clusters most of which were fresh. Analysis of total petroleum hydrocarbon contents in reef environments was made in seawater and reef sediments for assessment the extent of oil contamination in the area under investigation.

1. ***In seawater*** :

Table 4 shows obvious seasonal variation of total dissolved and dispersed petroleum hydrocarbons (DDPH) concentration in seawater of the investigated reef sites during 1998 and 1999. The maximum concentration of DDPH (114.45 $\mu g\ L^{-1}$) was measured at site II during summer, whilst the minimum concentration (3.42 $\mu g\ L^{-1}$) was measured at site III during spring.

The mean concentration of DDPH at each site is demonstrated in table 5 and figure 22. Site II showed the highest mean concentration 34.80 ± 26.57 $\mu g\ L^{-1}$, whereas site IV showed the lowest mean concentration 7.24 ± 1.14 $\mu g\ L^{-1}$.

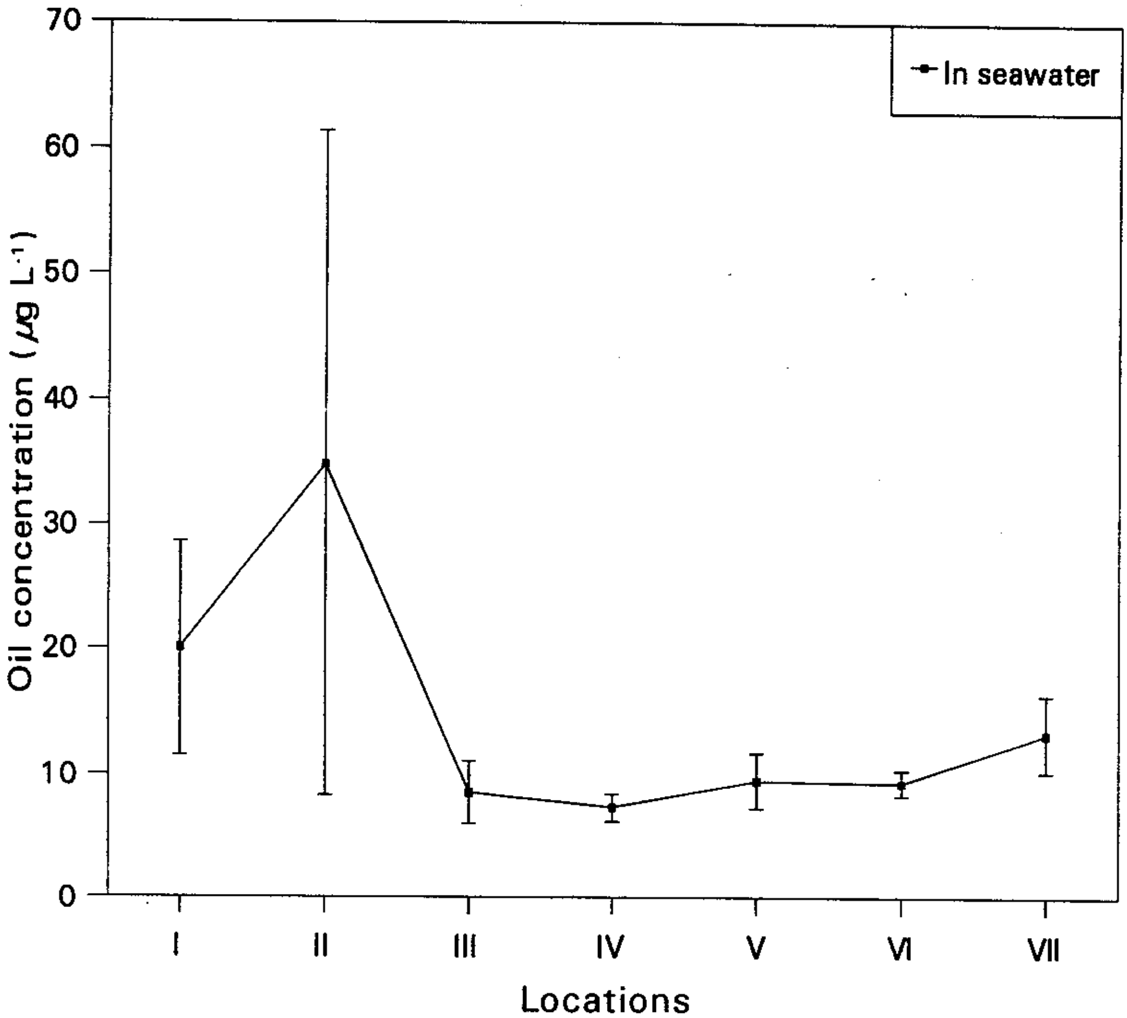


Fig. 22. Mean concentrations ($\mu g\ L^{-1}$) of oil in seawater at the surveyed locations. Vertical bars indicate standard errors.

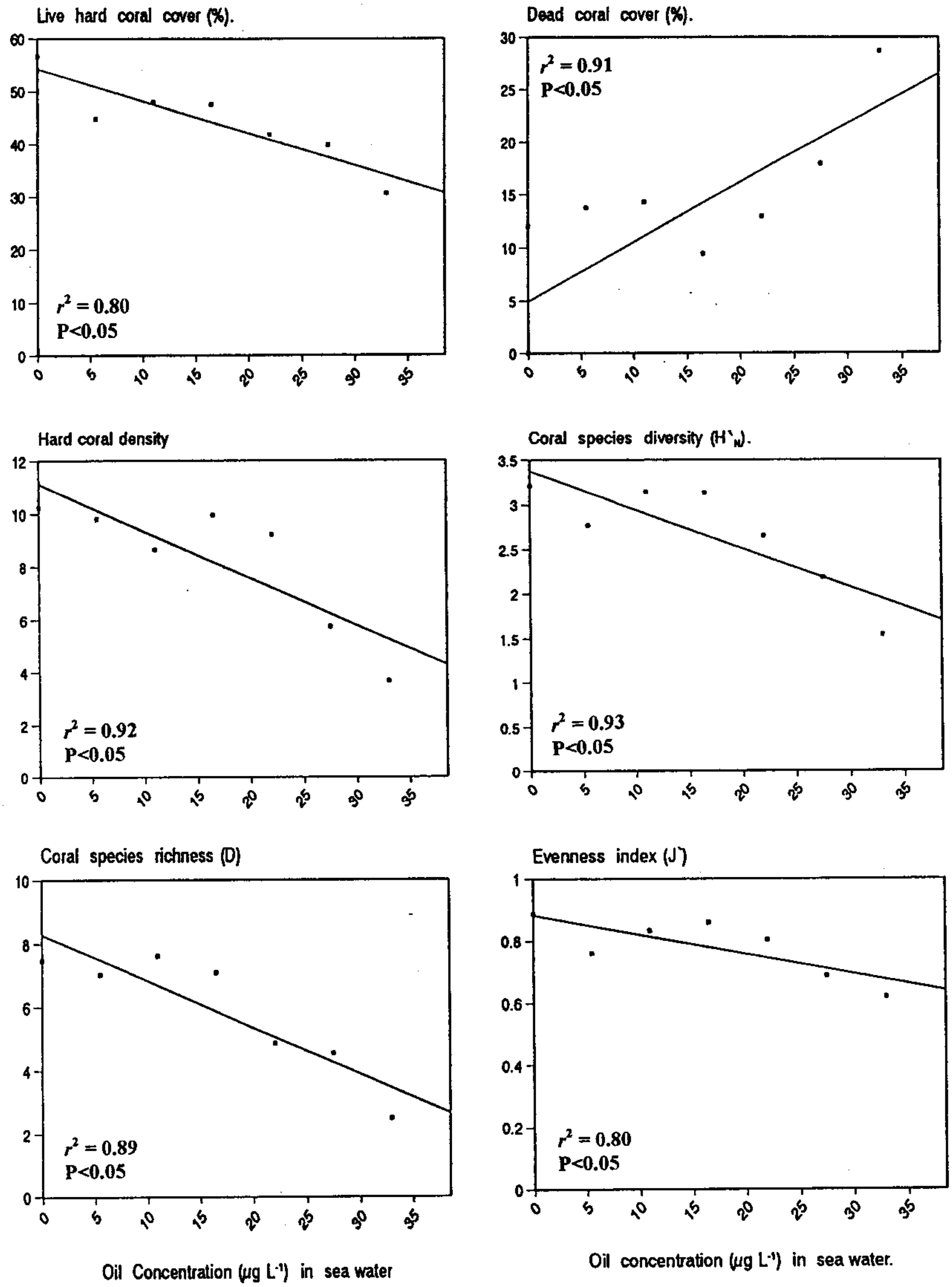


Fig. 23. Relationships between oil concentration in seawater and coral community variables.

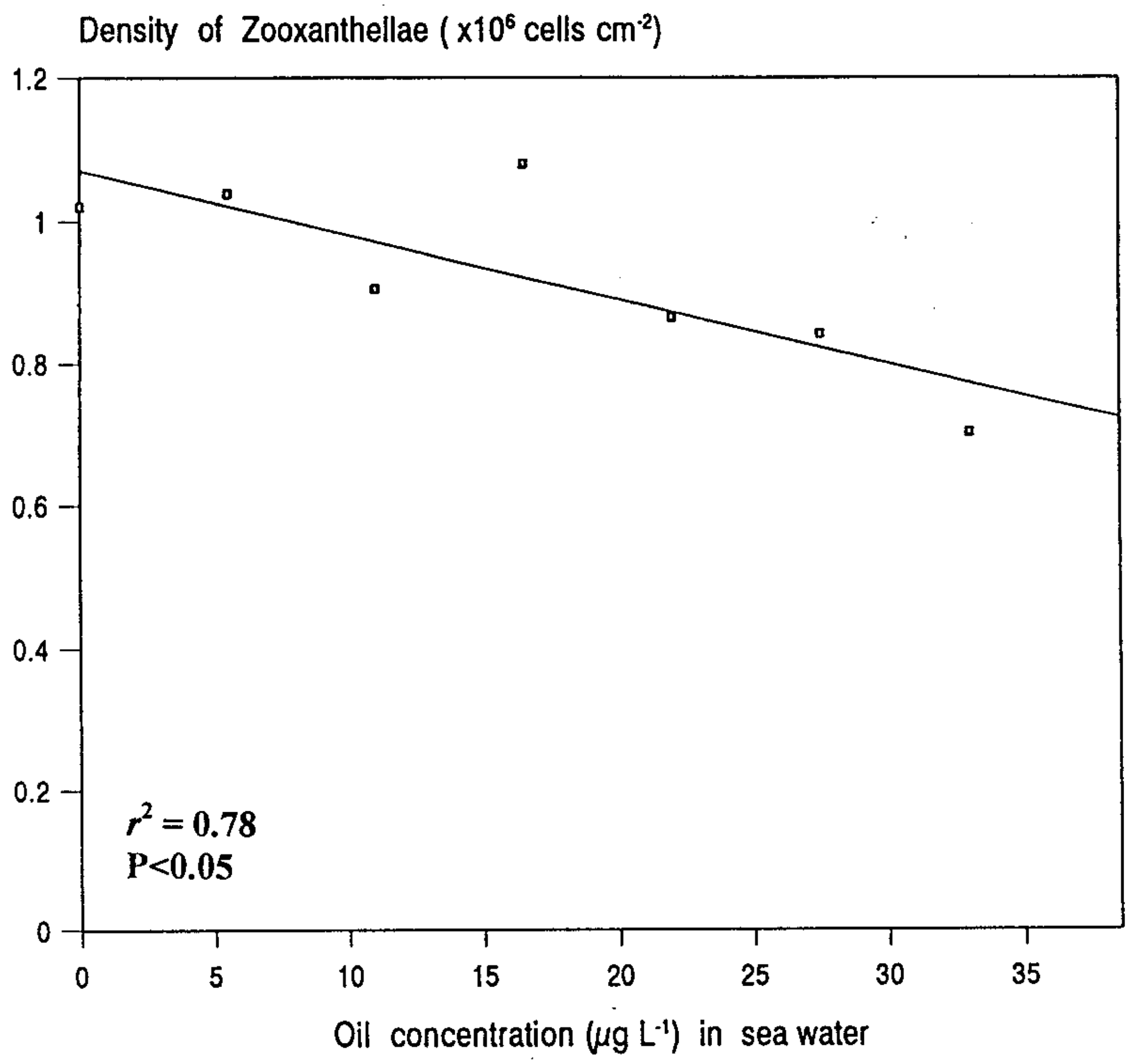


Fig. 24. Relationship between oil concentration in seawater and zooxanthellae density ($\times 10^6$ cells cm^{-2}).

The increasing DDPH (oil) concentration displayed a very high and significant (P<0.05) inverse correlations (Fig. 23) with increasing live hard coral cover (r^2 = 0.80), hard coral density (r^2 = 0.92), species diversity (r^2 = 0.93), species richness (r^2 = 0.89) and evenness index (r^2 = 0.80). On the contrary, the percentage of dead coral cover displayed a very high significant positive correlation with increasing concentration of DDPH (r^2 = 0.91, P<0.05; Fig. 17). In addition, population density of zooxanthellae showed high significant reduction with increasing levels of oil in seawater (r^2 = 0.78, P<0.05; Fig. 24).

2. *In reef sediments* :

Analysis of total petroleum hydrocarbons in reef sediments (TPH) can give a clearer indication of oil inputs in reef environment. There were a significant difference in TPH concentration between the surveyed reef sites (ANOVA, F = 6.73, df = 10, P<0.01; Table 6). The maximum mean concentration of TPH 52.85 ± 8.33 $\mu g\ g^{-1}$ was recorded at site II, while the minimum mean concentration 1.4 ± 0.01 $\mu g\ g^{-1}$ was recorded at site V (Table 5 and Fig. 25). The present data revealed a general decrease in the TPH concentration away from the shore (Fig. 25).

High significant inverse relationship (r^2 = 0.78, P<0.05) existed between oil concentration in reef sediments and percentage of living hard coral cover (Fig. 26). In contrast, a very highly significant positive correlation (r^2 = 0.90, P<0.05) was obtained between oil concentration in reef sediments and percentage of dead coral cover (Fig. 26).

Species diversity, species richness, evenness index and hard coral density were highly significantly and negatively correlated with oil concentration in reef sediments (r^2 = 0.91, 0.92, 0.76, and 0.85 respectively, P<0.05; Fig. 26). Similarly, population density of zooxanthellae showed high significant negative correlation with increasing oil concentration in reef sediments (r^2 = 0.83, P<0.05; Fig.

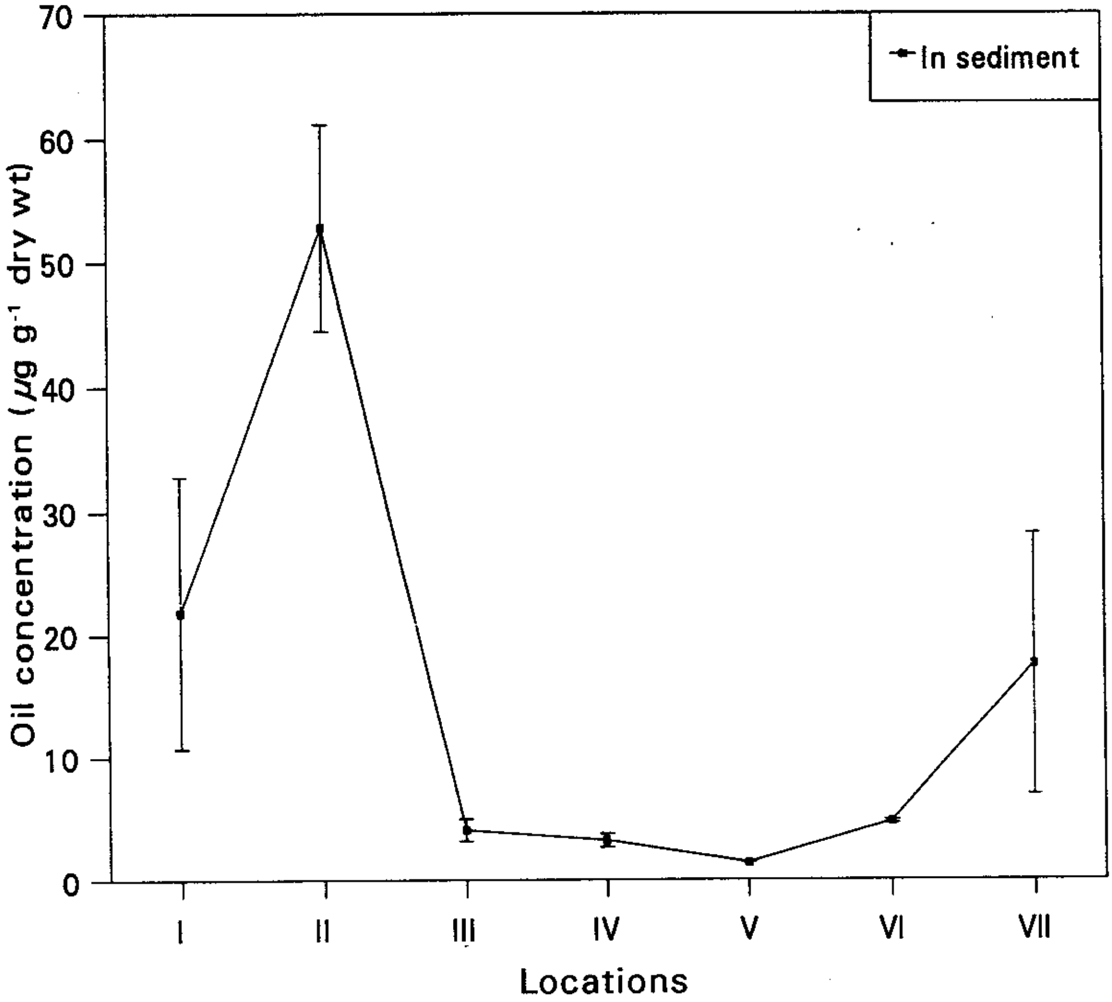


Fig. 25. Mean concentrations ($\mu g\ g^{-1}$ dry weight) of oil in reef sediments at the surveyed locations. Vertical bars indicate standard errors.

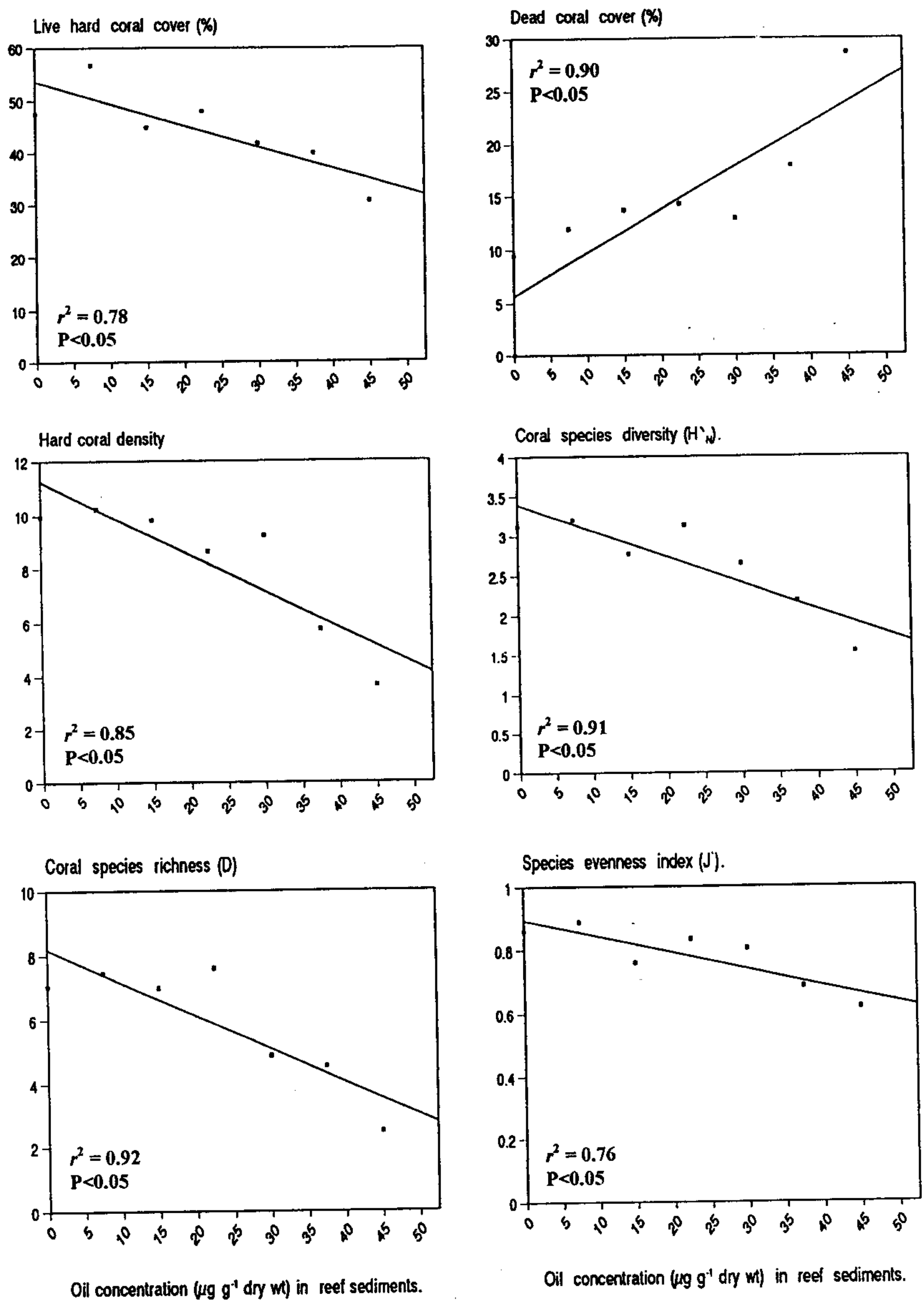


Fig. 26. Relationships between oil concentration in reef sediments and coral community variables.

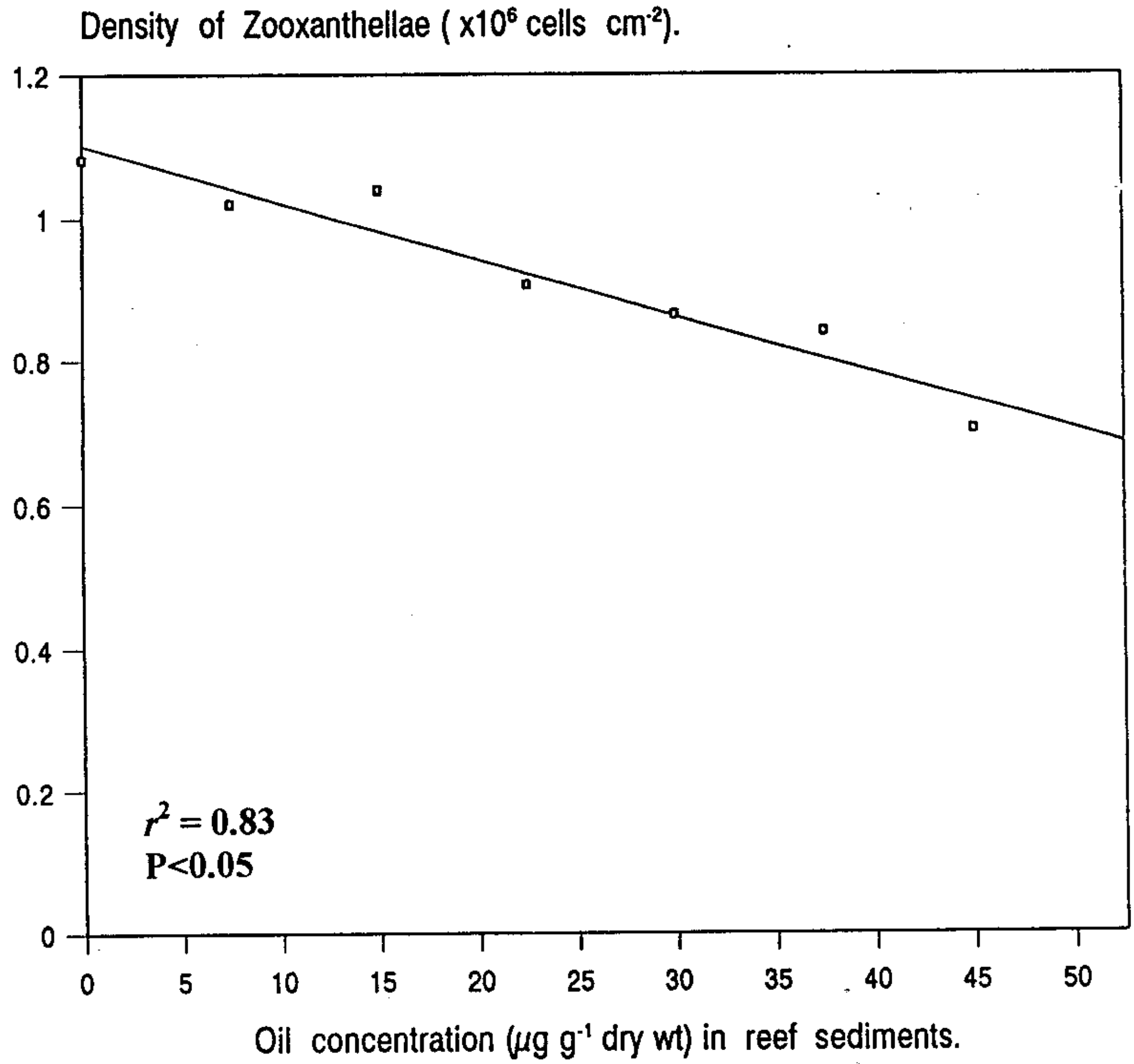


Fig. 27. Relationship between oil concentration in reef sediments and zooxanthellae density ($\times 10^6$ cells cm^{-2}).

27). Consequently, number of zooxanthellae per cm^2 of coral surface area was substantially fewer in heavily oiled reefs than less oiled reefs (Table 10).

Heavy metals:

Heavy metals in the area of study were analyzed in seawater and reef sediments. Mean and standard errors for heavy metal concentrations at each location were calculated.

1. *In seawater* :

Table 8a and Figure 28 presents the average concentrations of ten dissolved trace metals viz.: Cu, Zn, Pb, Cd, Cr, Ni, Co, Fe, Mn and Hg in the different surveyed locations. Concentration of Hg was very low as compared with other metals. Five metals, namely Cu, Zn, Ni, Fe and Mn showed considerable concentrations at each location.

The highest average concentrations for all analyzed dissolved trace metals were found at location I. On the other hand, the lowest concentrations of Cu, Zn, Pb, Fe and Mn were obtained at location VII. Locations IV and V showed the lowest mean concentrations of Cd and Cr, whereas location VI showed the minimum mean concentrations of Ni and Co. In general, the northern nearshore reef areas (locations I & II) had much higher concentrations of dissolved trace metals than the southern nearshore reef areas (locations III & VII). Also, the nearshore reefs, with the exception of site VII, had higher concentration of trace metals than offshore reefs.

Hard coral density, species diversity and evenness index showed a high significant negative correlation with increasing concentration of dissolved Cu ($r^2 = 0.67$, 0.61 and 0.72 respectively, $P<0.05$; Fig. 29).

On the same manner, hard coral density and evenness index were significantly negatively correlated with increasing concentration of

Table 8a. Mean and standard error (in parentheses) of dissolved trace metal concentrations (µg L^{-1}) in seawater at the investigated locations.

Element	Location						
	I	II	III	IV	V	VI	VII
Cu	12.20	9.42	5.35	2.55	3.55	2.55	1.68
	[0.61]	[0.57]	[0.82]	[0.25]	[0.45]	[0.35]	[0.06]
Zn	20.01	14.39	12.69	9.52	10.22	9.59	6.75
	[0.60]	[1.42]	[1.05]	[0.56]	[0.98]	[1.03]	[0.09]
Pb	0.94	0.62	0.51	0.24	0.32	0.22	0.16
	[0.05]	[0.06]	[0.07]	[0.01]	[0.04]	[0.08]	[0.02]
Cd	0.32	0.22	0.32	0.08	0.10	0.10	0.19
	[0.03]	[0.02]	[0.15]	[0.01]	[0.01]	[0.00]	[0.03]
Cr	0.17	0.13	0.10	0.06	0.06	0.07	0.07
	[0.01]	[0.01]	[0.01]	[0.00]	[0.01]	[0.01]	[0.00]
Ni	4.39	3.20	2.64	2.30	2.20	1.90	3.02
	[0.51]	[0.30]	[0.22]	[0.50]	[0.20]	[0.30]	[1..48]
Co	0.36	0.30	0.25	0.18	0.21	0.17	0.59
	[0.06]	[0.04]	[0.03]	[0.03]	[0.03]	[0.01]	[0.08]
Fe	33.38	23.55	13.72	10.24	10.32	9.85	8.12
	[2.44]	[2.18]	[1.41]	[1.94]	[1.56]	[2.85]	[0.20]
Mn	4.07	3.01	2.34	2.03	2.03	2.11	1.48
	[0.15]	[0.16]	[0.21]	[0.39]	[0.25]	[0.61]	[0.04]
Hg	0.04	0.03	0.02	0.02	0.02	0.02	———
	[0.00]	[0.00]	[0.00]	[0.00]	[0.00]	[0.00]	

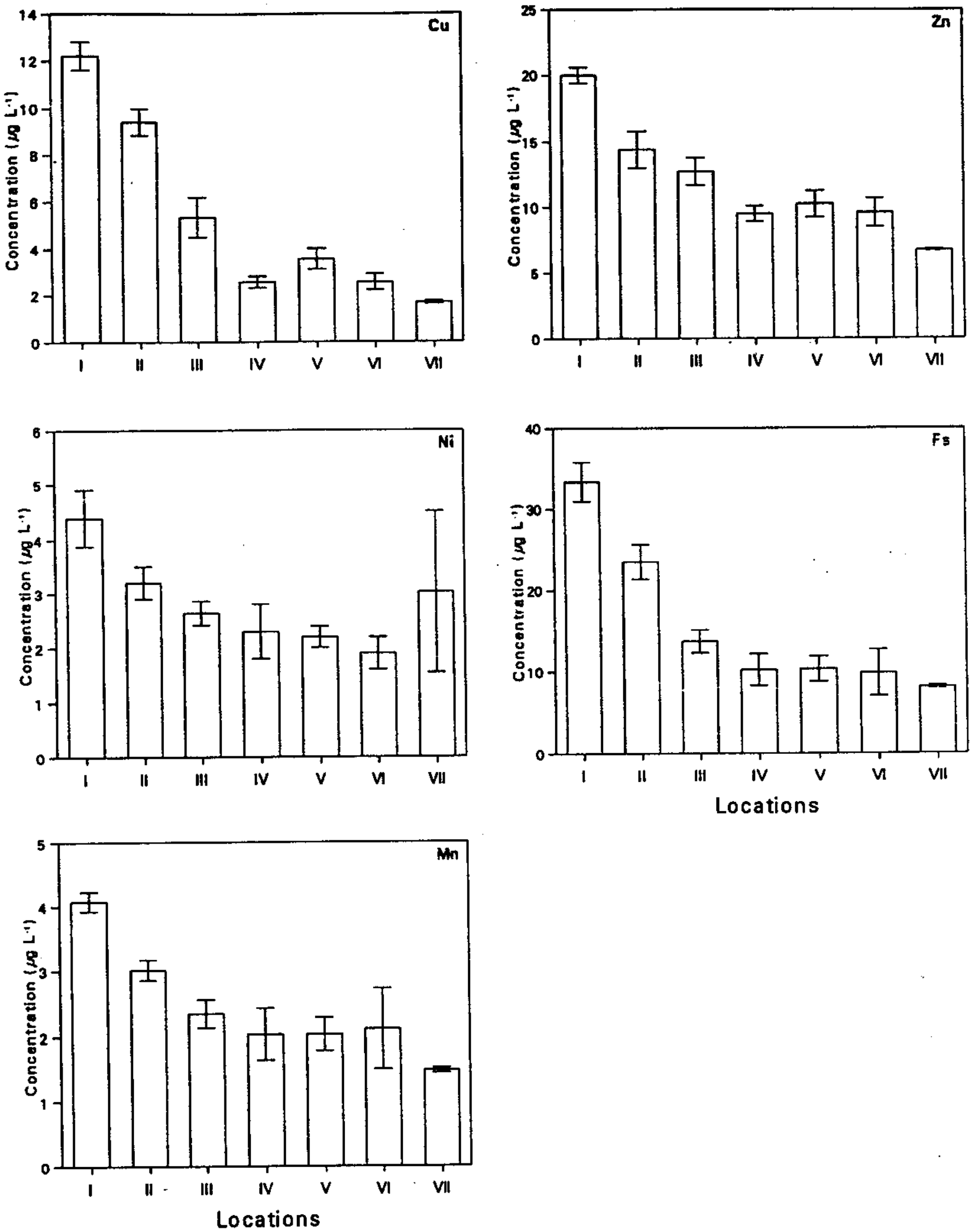


Fig. 28. Mean concentrations ($\mu g\ L^{-1}$) of dissolved copper, zinc, nickel, iron and manganese in seawater at the studied locations. Error bars indicate standard error.

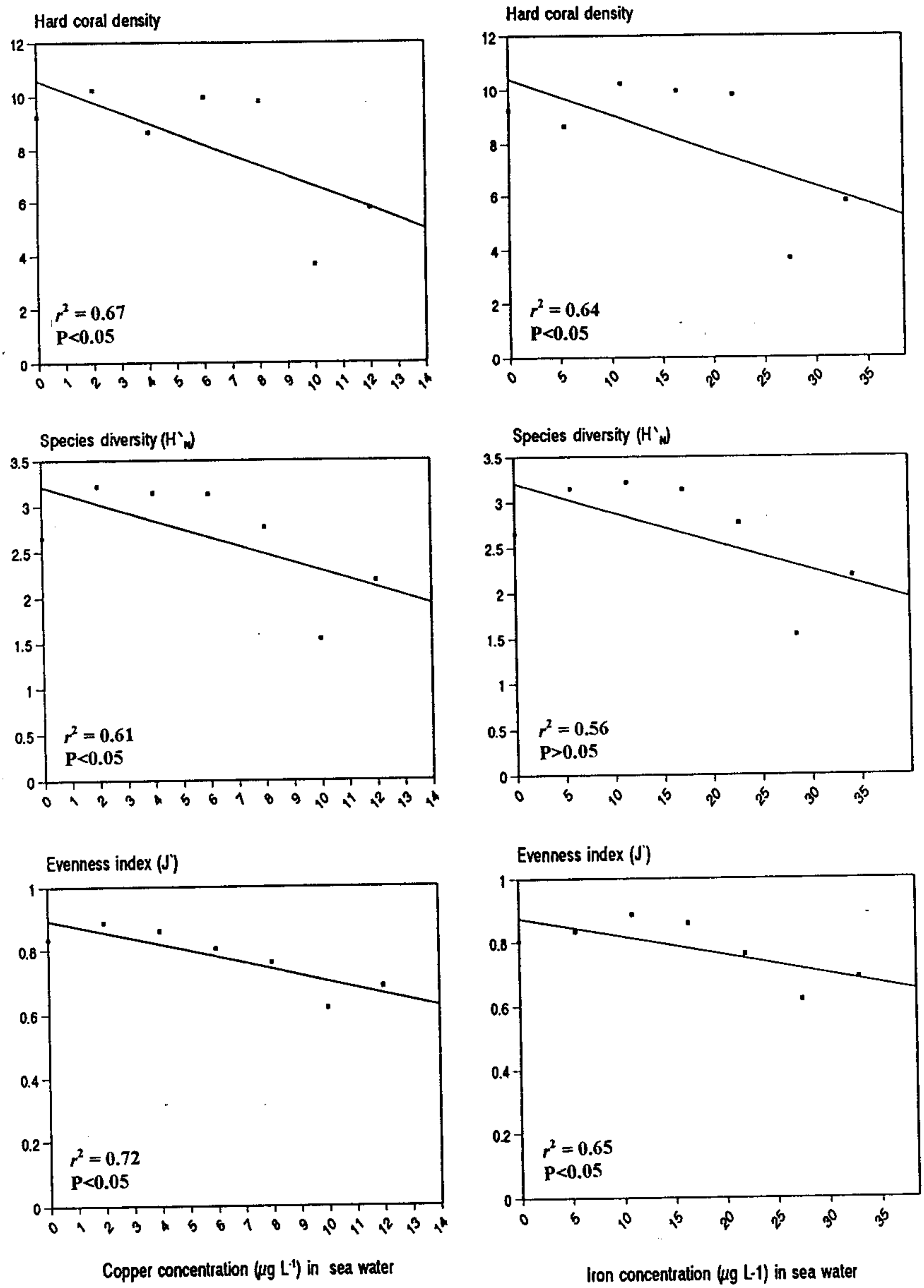


Fig. 29. Relationships between hard coral density, species diversity and evenness index, and concentrations of dissolved copper and iron in seawater.

dissolved Fe (r^2 = 64 and 65 respectively, $P<0.05$; Fig. 29), whilst species diversity showed strong negative correlation with Fe concentration but not significant (r^2 = 0. 56, $P>0.05$; Fig. 29).

2. *In reef sediments* :

Analyses of trace metals in reef sediments provide significant indications about heavy metal pollution in reef environment. Trace metals concentration in reef sediments reflected the same general trend observed in seawater (Table 8b and Fig. 30). Location I represented clearly higher average levels of all determined trace metals than other sampled locations. On the other side, the lower average levels for Zn, Pb, Cd and Cr were recorded at location VII, while the lower average values for Cu, Ni, Co, Fe and Mn were recorded at location VI. The present data indicates that the concentrations of trace metals in reef sediments showed clearer variations between the sampled locations than in seawater. Concentration of Hg was very low and showed no marked variation among the surveyed locations (Table 8b). Concentrations of trace metals in reef sediments were generally found to be higher in the northern nearshore reefs than in the southern nearshore reefs. With the exception of location VII, Figure 30 reveals that a general decrease in heavy metal levels towards the offshore reefs.

Figures 31, 32, 33 and 34 demonstrate the relationships between concentrations of eight heavy metals in reef sediments and some coral community parameters. Cu, Cr and Ni concentrations were significantly negatively correlated with species diversity (r^2 = 0.59, 0.67 and 0.58 respectively, $P<0.05$). On the other hand, the correlation between concentrations of Zn, Pb, Cd, Fe and Mn and species diversity were obvious and inverse but not significant (r^2 = 0.51, 0.54, 0.54, 0.55 and 0.55 respectively, $P>0.05$). The hard coral density exhibited high significant negative correlations with concentrations of Zn, Pb, Cr and Ni

Table 8b. Mean and standard error (in parentheses) of trace metal concentrations ($\mu g\ g^{-1}$) in reef sediments at the investigated reef sites. Concentrations are expressed in a dry weight.

Element	Location						
	I	II	III	IV	V	VI	VII
Cu	14.35	9.20	6.14	2.06	2.97	1.47	2.87
	[1.59]	[1.05]	[0.88]	[0.56]	[0.57]	[0.22]	[0.32]
Zn	76.08	50.62	29.90	21.89	23.06	20.38	15.68
	[1.44]	[3.82]	[2.42]	[2.04]	[2.14]	[2.68]	[1.86]
Pb	8.92	7.05	4.77	3.00	3.60	2.52	1.60
	[0.39]	[1.00]	[0.70]	[0.60]	[0.80]	[0.67]	[0.37]
Cd	3.57	2.95	2.55	1.22	1.66	1.20	0.98
	[0.33]	[1.47]	[0.30]	[0.22]	[0.33]	[0.04]	[0.13]
Cr	7.43	6.21	4.27	2.43	3.02	2.48	2.42
	[0.50]	[0.40]	[0.47]	[0.48]	[0.48]	[0.33]	[0.36]
Ni	19.11	13.25	7.57	4.87	6.00	3.75	4.50
	[1.10]	[1.31]	[0.54]	[1.12]	[1.50]	[0.75]	[0.62]
Co	5.54	3.77	2.89	1.90	2.26	1.55	2.07
	[0.56]	[0.48]	[0.38]	[0.30]	[0.31]	[0.31]	[0.23]
Fe	1061.72	635.15	394.52	183.40	192.04	142.04	226.65
	[77.25]	[62.70]	[21.58]	[13.88]	[10.54]	[3.46]	[13.85]
Mn	51.94	39.50	32.92	23.40	25.60	23.20	24.05
	[3.05]	[2.90]	[1.68]	[3.40]	[2.40]	[0.40]	[2.65]
Hg	0.08	0.07	0.05	0.04	0.05)	0.03	———
	[0.00]	[0.00]	[0.00]	[0.00]	[0.01]	[0.00]	

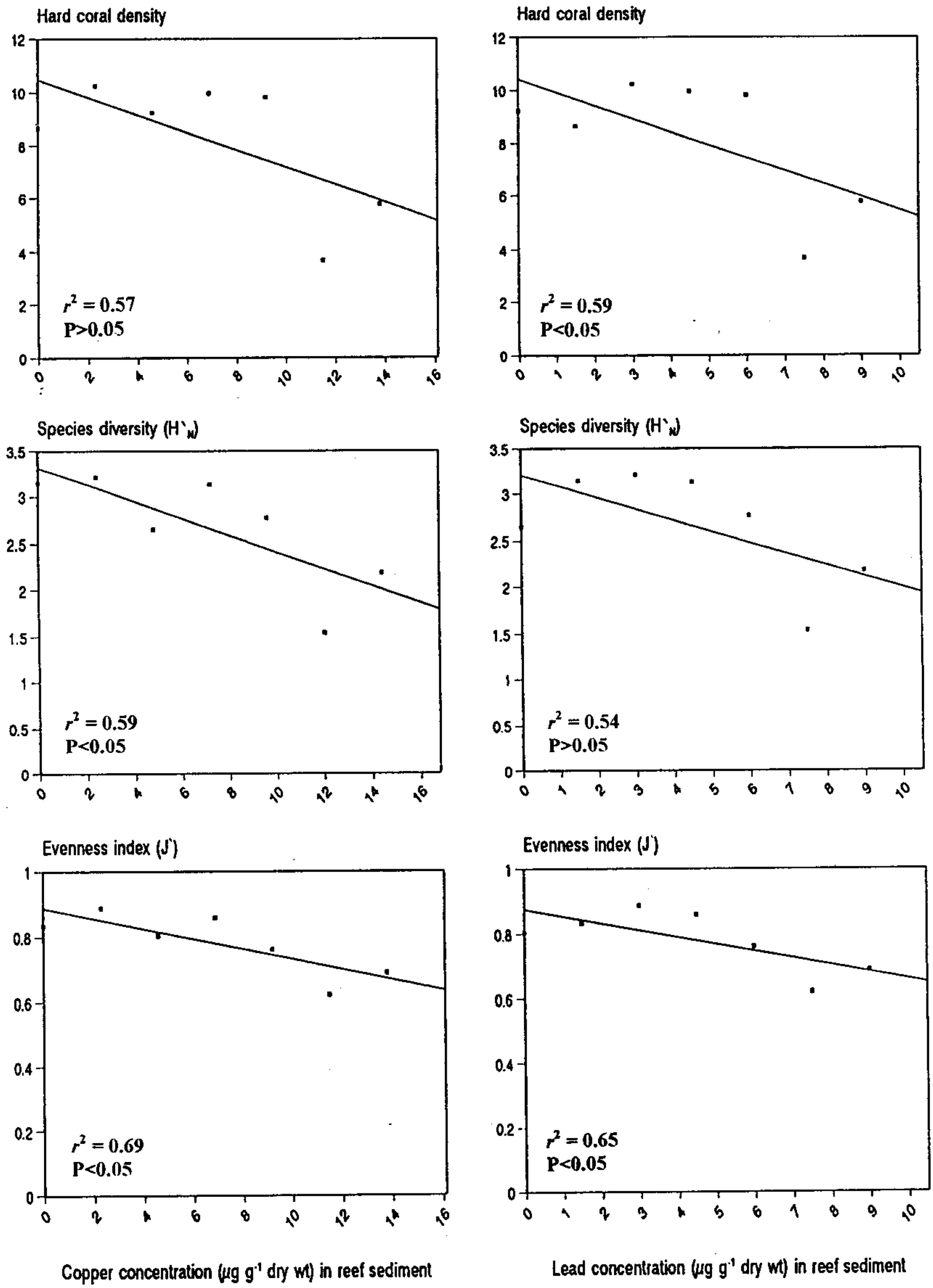


Fig. 31. Relationships between hard coral density, species diversity and evenness index, and concentrations of copper and lead in reef sediments.

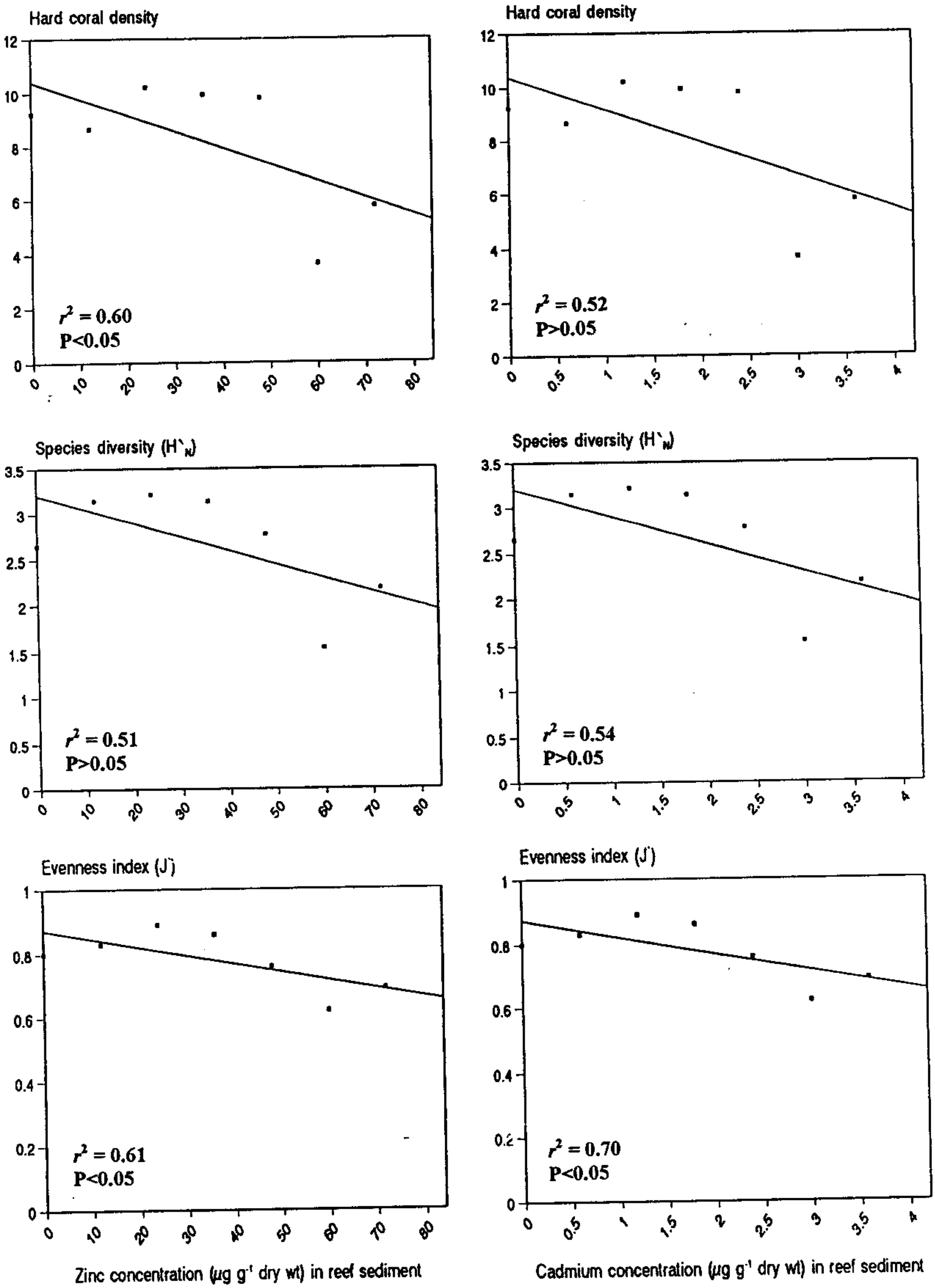


Fig. 32. Relationships between hard coral density, species diversity and evenness index, and concentrations of zinc and cadmium in reef sediments.

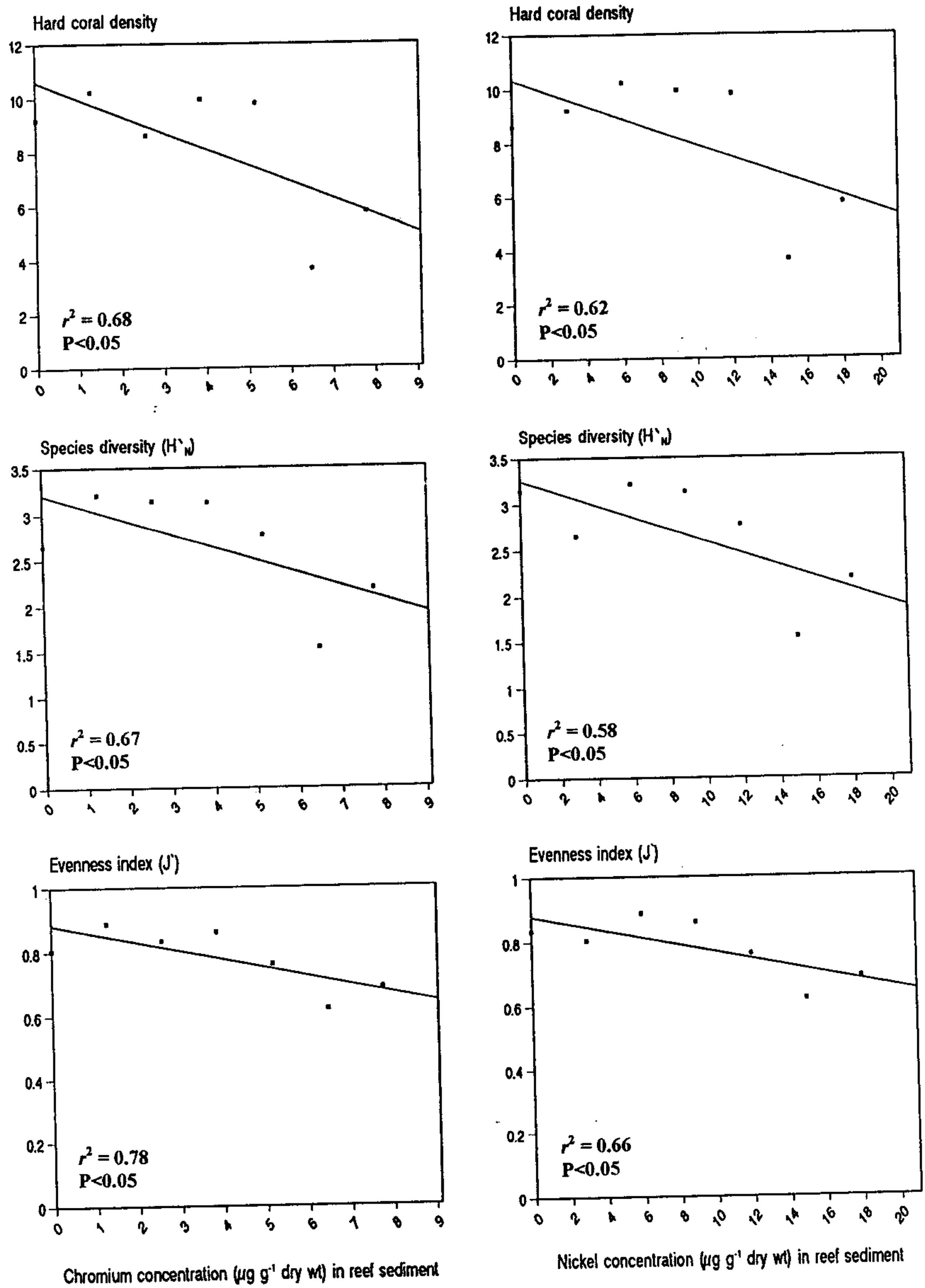


Fig. 33. Relationships between hard coral density, species diversity and evenness index, and concentrations of chromium and nickel in reef sediments.

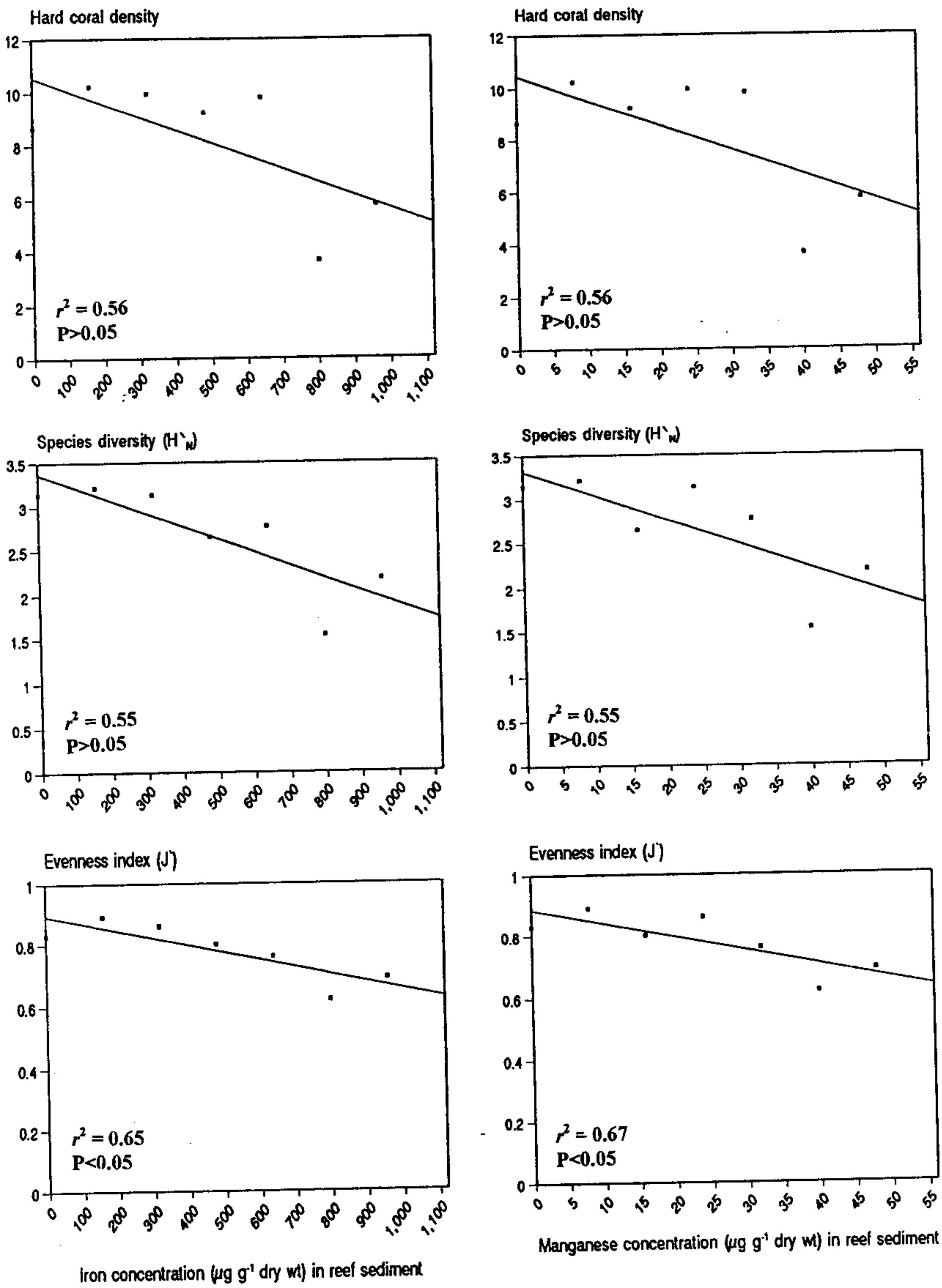


Fig. 34. Relationships between hard coral density, species diversity and evenness index, and concentrations of iron and manganese in reef sediments.

(r^2 = 0.60, 0.59, 0.68 and 0.62 respectively, $P<0.05$), whereas its negative correlations with concentrations of Cu, Cd, Fe and Mn were not significant but strong (r^2 = 0.57, 0.52, 0.56 and 0.56 respectively, $P>0.05$).

Concerning the relationships between evenness index and heavy metal concentrations, there are a high significant negative correlations between evenness index and concentrations of Cu, Zn, Pb, Cd, Ni, Fe and Mn (r^2 = 0.69, 0.61, 0.65, 0.70, 0.66, 0.65 and 0.67 respectively, $P< 0.05$). This correlation was very high in case of Cr (r^2= 0.78, $P<0.05$).

II. Biological parameters

Coral community structure:

Number of species, Species richness, Evenness index and Species diversity:

Table 9 lists 83 hermatypic coral species belonging to 35 coral genera and six alcyonacean (soft) corals were encountered during the survey in the whole study area, together with their percentage covers at each sampled site. The number of hard coral species and species richness were greatly varied between the surveyed sites (Table 10 & Fig. 36a). Site VI showed a greater number of species (43) and species richness (7.60), whereas the fewer number of species (12) and lower species richness (2.48) were registered at site II.

Similarity in species composition between the studied reef sites was suggested by cluster analysis using presence/absence data (Fig. 35).

Pielou's evenness index (J`) was assessed to give information about the distribution of individuals (coral colonies) among the species. Evenness index reaches to its maximum value of 1.00, when all coral colonies are equally distributed in the reef community. Therefore, it is essentially considered as a measure of dominance in the coral reef community. Evenness index in the area of study ranged between 0.62 at site II and 0.89 at site IV (Table 10 & Fig. 36b). The lowest evenness index found at site II indicates that the clear dominance of a few numbers of species in this area.

Comparing the Shannon and Weaver's species diversity ($H`_N$) in the area of study (Table 10 & Fig. 36a) revealed that site IV supported higher species diversity (3.21), while site II supported lower value (1.54). Species diversity in the present work is controlled not only by number of

Table 9. List of all hard coral species and soft corals encountered during the study, together with their percentage covers at different surveyed reef sites.

Species	Location						
	I	II	III	IV	V	VI	VII
Hard corals							
Psammocora profundacella	0.00	0.00	0.00	0.00	0.66	0.03	0.00
Psammocora haimeana	0.00	0.00	0.00	0.00	0.00	0.03	0.66
Stylophora pistillata	3.16	9.35	0.65	1.54	1.23	0.78	1.40
Seriatopora hystrix	0.00	0.00	1.76	2.20	0.12	0.05	0.00
Pocillopora damicornis	0.00	0.00	0.00	0.60	5.76	1.74	6.33
Pocillopora verrucosa	0.00	0.00	0.00	0.00	2.40	1.03	1.00
Montipora verrucosa	0.11	0.00	0.78	0.00	0.00	0.00	0.00
Montipora monasteriata	0.97	0.22	1.43	0.00	0.00	0.00	0.50
Montipora stilosa	0.00	0.00	0.00	0.77	0.16	0.07	0.00
Montipora circumvallata	0.00	0.00	0.07	0.00	0.00	0.00	0.00
Montipora spongiosa	1.16	0.00	0.00	1.81	0.00	0.00	0.00
Montipora tuberculosa	1.08	0.00	0.00	0.00	0.00	0.00	0.00
Montipora informis	0.00	0.00	0.00	1.33	0.00	0.00	0.00
Montipora danae	0.00	0.00	0.65	0.00	0.00	0.00	0.00
Montipora meandrina	0.00	0.00	0.00	0.40	0.00	0.00	0.00
Acropora hyacinthus	0.00	0.00	1.56	1.52	3.34	6.60	2.12
Acropora cytherea	0.00	0.00	0.00	0.00	2.20	1.08	1.64
Acropora humilis	0.54	0.00	0.52	0.00	1.80	0.75	0.33
Acropora valida	0.00	0.86	0.26	0.66	0.49	1.27	2.63
Acropora hemprichi	2.37	0.56	0.20	1.67	2.57	2.29	3.21
Acropora pharaonis	7.04	6.74	3.55	0.80	2.20	0.46	0.00
Acropora corymbosa	0.00	0.00	0.52	0.00	0.00	0.00	3.04
Acropora granulosa	0.00	0.00	2.02	0.33	0.00	0.02	0.00
Acropora capillaris	0.00	0.00	0.00	0.00	0.11	1.24	0.00
Acropora nobilis	0.43	0.00	0.00	0.53	1.83	4.08	0.33
Acropora digitifera	0.00	0.00	0.00	0.00	0.16	0.00	0.00
Acropora eurystoma	0.00	0.00	0.65	0.00	0.00	0.81	0.00
Acropora forskali	0.00	4.40	0.00	0.47	0.00	1.14	0.00
Acropora nasuta	0.00	0.00	0.40	0.00	0.00	0.00	0.00

Table 9. continued

Species	Location						
	I	II	III	IV	V	VI	VII
Pavona cactus	2.83	0.00	0.78	0.00	0.00	0.00	0.00
Pavona explanulata	0.00	1.09	0.00	0.00	0.00	0.00	0.00
Leptoseris foliosa	0.00	0.00	0.52	0.00	0.00	0.00	0.00
Leptoseris explanata	0.00	0.00	0.00	0.00	0.00	0.00	0.50
Gardineroseris planulata	0.00	0.00	0.39	0.00	0.28	0.05	0.92
Pachyseris rugosa	0.00	0.00	0.13	1.10	2.10	0.00	0.00
Porites solida	2.26	2.61	0.00	12.20	0.40	4.28	4.61
Porites lutea	2.15	2.40	1.56	7.10	0.00	0.21	1.48
Porites lobata	0.86	0.00	0.00	0.00	0.00	0.00	0.00
Porites compressa	0.00	0.00	0.00	3.20	0.00	0.00	0.00
Goniopora planulata	0.00	0.00	1.34	0.00	0.00	2.07	0.16
Goniopora stokesi	0.22	0.00	0.52	0.00	0.00	2.76	0.00
Alveopora spongiosa	0.00	0.00	0.00	0.63	0.00	0.00	0.00
Siderastraea savignyana	0.22	0.17	0.00	0.00	0.00	0.11	0.00
Galaxea fascicularis	4.90	0.00	8.10	1.60	0.53	0.34	0.00
Echinophyllia aspera	0.00	0.00	0.78	1.36	0.37	0.00	0.00
Oxypora lacera	0.00	0.00	1.30	0.50	0.00	0.00	0.00
Mycedium elephantotus	0.00	0.00	0.00	1.60	0.00	0.00	0.00
Acanthastrea echinata	0.00	0.00	0.00	0.20	0.00	0.07	0.00
Symphyllia erythraea	0.05	0.00	0.00	0.00	0.00	0.34	0.00
Lobophyllia corymbosa	3.20	0.00	2.86	6.13	0.00	1.52	0.00
Lobophyllia hemprichi	0.00	0.00	1.04	0.00	0.00	0.00	0.00
Hydnophora exesa	0.00	0.00	0.00	0.00	0.00	0.21	0.00
Hydnophora microconos	0.00	0.00	0.65	0.00	0.00	0.00	0.00
Favia stelligera	0.00	0.00	0.00	1.07	0.50	0.43	0.49
Favia pallida	0.00	0.00	0.00	0.85	0.63	0.00	0.08
Favia favus	0.00	0.00	0.00	0.54	1.22	0.00	0.00
Favia speciosa	0.00	0.00	0.00	0.45	0.00	0.00	0.00
Favia amicorum	0.00	0.00	0.00	0.36	0.00	0.00	0.00
Favia laxa	0.00	0.00	0.00	0.00	0.00	0.00	0.08
Favites abdita	0.00	0.00	0.13	0.20	0.00	0.10	0.00

Table 9. continued

Species	Location						
	I	II	III	IV	V	VI	VII
Favites halicora	0.00	0.00	0.00	0.00	0.65	0.00	0.00
Favites complanata	0.00	0.00	2.51	0.00	0.33	0.00	0.00
Favites peresi	0.00	0.00	0.26	0.27	0.11	0.11	0.00
Favites flexuosa	0.00	0.00	0.00	0.00	1.26	1.45	0.82
Goniastrea retiformis	0.00	0.00	0.00	0.90	2.97	2.53	0.00
Goniastrea pectinata	0.00	0.00	0.00	0.20	0.16	1.03	0.00
Platygyra daedalea	0.00	0.00	1.30	0.36	0.33	0.76	0.00
Platygyra lamellina	0.43	0.86	1.56	0.00	1.31	0.00	0.00
Platygyra sinensis	1.62	1.30	0.26	0.00	0.00	0.00	0.00
Leptoria phrygia	0.00	0.00	0.00	0.00	1.32	0.86	0.00
Montastrea curta	0.00	0.00	1.95	0.00	0.00	0.00	0.25
Leptastrea purpurea	0.00	0.00	0.00	0.00	0.11	0.00	0.00
Leptastrea transversa	0.00	0.00	0.00	0.00	0.00	0.10	0.00
Cyphastrea serailia	0.22	0.00	0.00	0.00	0.00	0.00	0.16
Cyphastrea microphthalma	0.00	0.00	0.52	0.00	0.25	0.00	0.00
Echinopora gemmacea	3.45	0.00	0.33	0.80	1.84	0.60	0.00
Echinopora lamellosa	0.00	0.00	0.00	0.40	0.00	0.21	0.50
Echinopora fruticulosa	0.11	0.00	1.04	0.00	0.00	0.00	3.29
Plerogyra sinuosa	0.00	0.00	0.00	0.00	0.00	0.45	0.00
Tubastraea aurea	0.00	0.00	0.00	0.00	0.16	0.00	0.00
Turbinaria mesenterina	0.43	0.00	0.00	0.00	0.00	0.00	0.00
Millepora dichotoma	0.00	0.00	0.00	0.00	4.64	3.18	4.20
Millepora platyphylla	0.00	0.00	0.00	0.00	1.04	0.71	1.07
Soft corals							
Lobophytum pauciflorum	4.77	7.28	0.00	0.00	1.63	0.07	1.37
Lithophyton arboreum	9.60	8.11	1.17	0.00	0.51	1.32	0.64
Sarcophyton trocheliophorum	2.80	0.00	1.58	1.28	0.26	0.62	0.00
Sinularia polydactyla	0.00	0.00	0.00	0.00	3.53	1.23	1.11
Dendronephthya klunzingeri	0.00	0.00	0.00	0.00	1.65	0.00	0.00
Xenia spp.	1.34	2.74	1.02	4.54	1.24	1.35	0.25

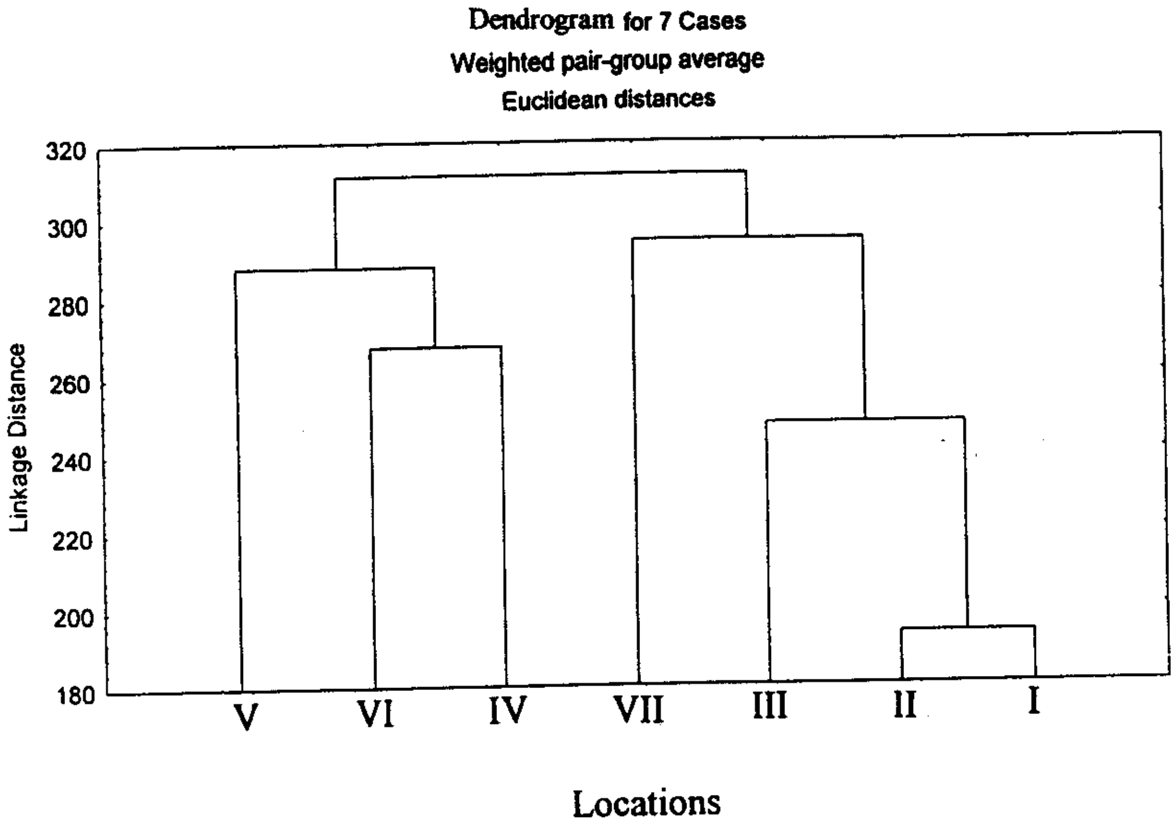


Fig. 35. Similarity in species composition between the surveyed reef sites. Dendrogram drawn using weighted pair-group average cluster analysis. Two main clusters are distinguished, nearshore (I, II, III and VII) and offshore reef locations (IV, V and VI).

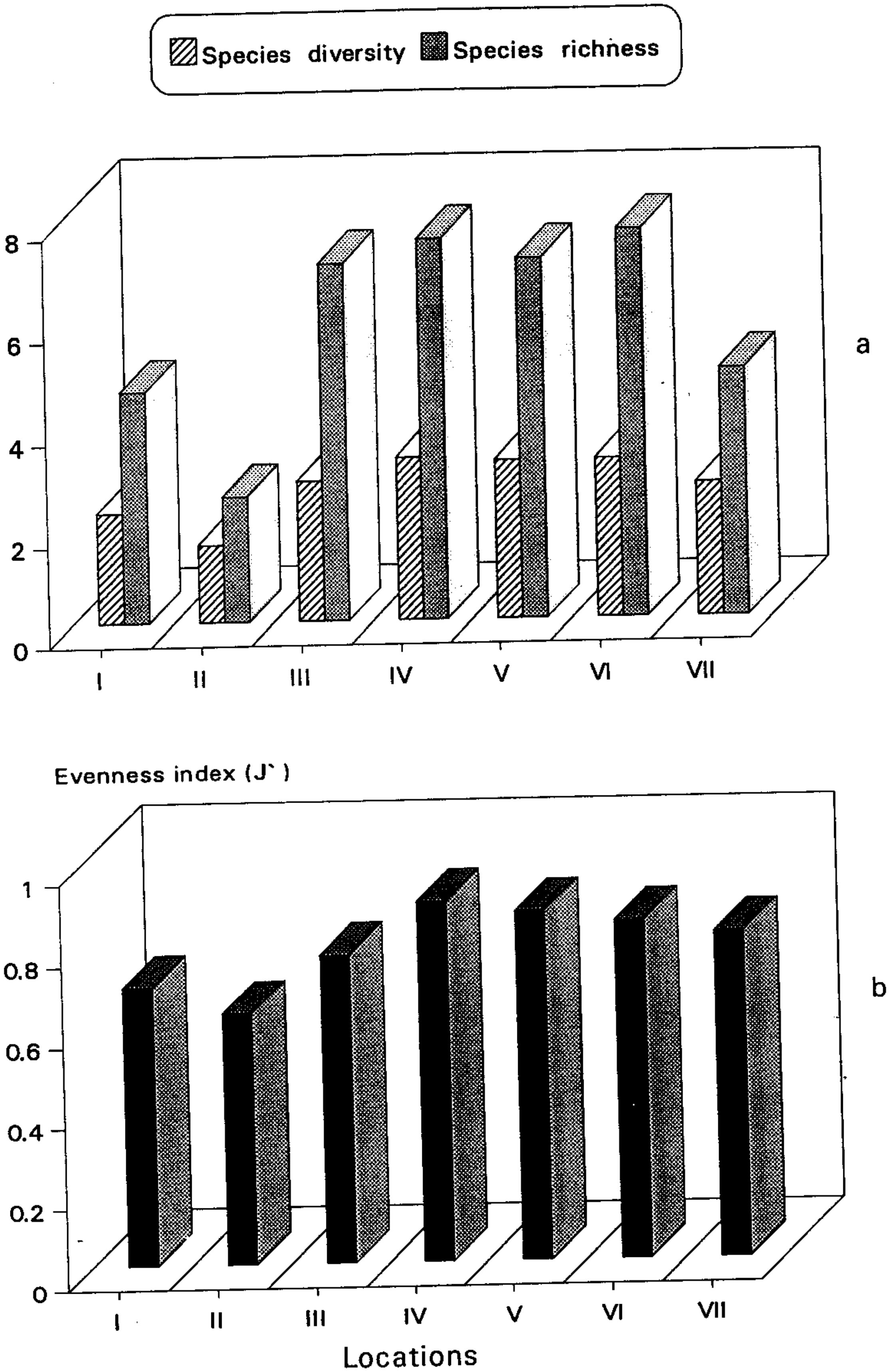


Fig. 36. a: Species diversity and species richness; and **b:** evenness index at the surveyed locations.

species but also evenness index. The higher values of species diversity in sites IV, V & VI (Table 10), revealing the importance of even distribution of individuals among coral species and number of species. Thus, Shannon and Weaver's species diversity depends not only on number of species, but also species evenness index.

The data presented in Table 10 and illustrated in Figure 36 display a general trend of increasing species richness, species diversity and evenness index from nearshore reefs to offshore reefs.

***Hard coral density*:**

Hard coral density is the number of hard corals per square meter in each surveyed reef site. The mean and standard error for hard coral density at each investigated area were calculated. The maximum mean for hard coral density 10.23 ± 1.88 was obtained at site IV (Sh'ab Saad), whilst the minimum mean 3.65 ± 0.9 was obtained at site II (Table 10 & Fig. 37a).

***Population density of sea urchins* :**

It is the number of sea urchins per square meter of reef substrate at each location (# m^{-2}). Location I supported the highest density (10.93 ± 2.87 urchins m^{-2}) followed by locations II and III (Table 10 & Fig. 37b), while location VII supported the lowest population density of sea urchin (1.40 ± 0.40 urchins m^{-2}). Two species of sea urchins were encountered during this survey, short stout-spined urchin, namely *Echinometra mathaei*; and long black-spined urchin, namely *Diadema setosum*. The former one was the most abundant sea urchin in the Gulf of Suez (locations I & II), whereas the latter one was probably the most abundant sea urchin in the area of Hurghada (locations III, IV, V & VI) and Safaga (location VII). *Diadema setosum* generally conceal (or partially conceal) themselves in holes or crevices during the day time and emerge at night to browse on algae covering coral surfaces or undermined coral skeleton.

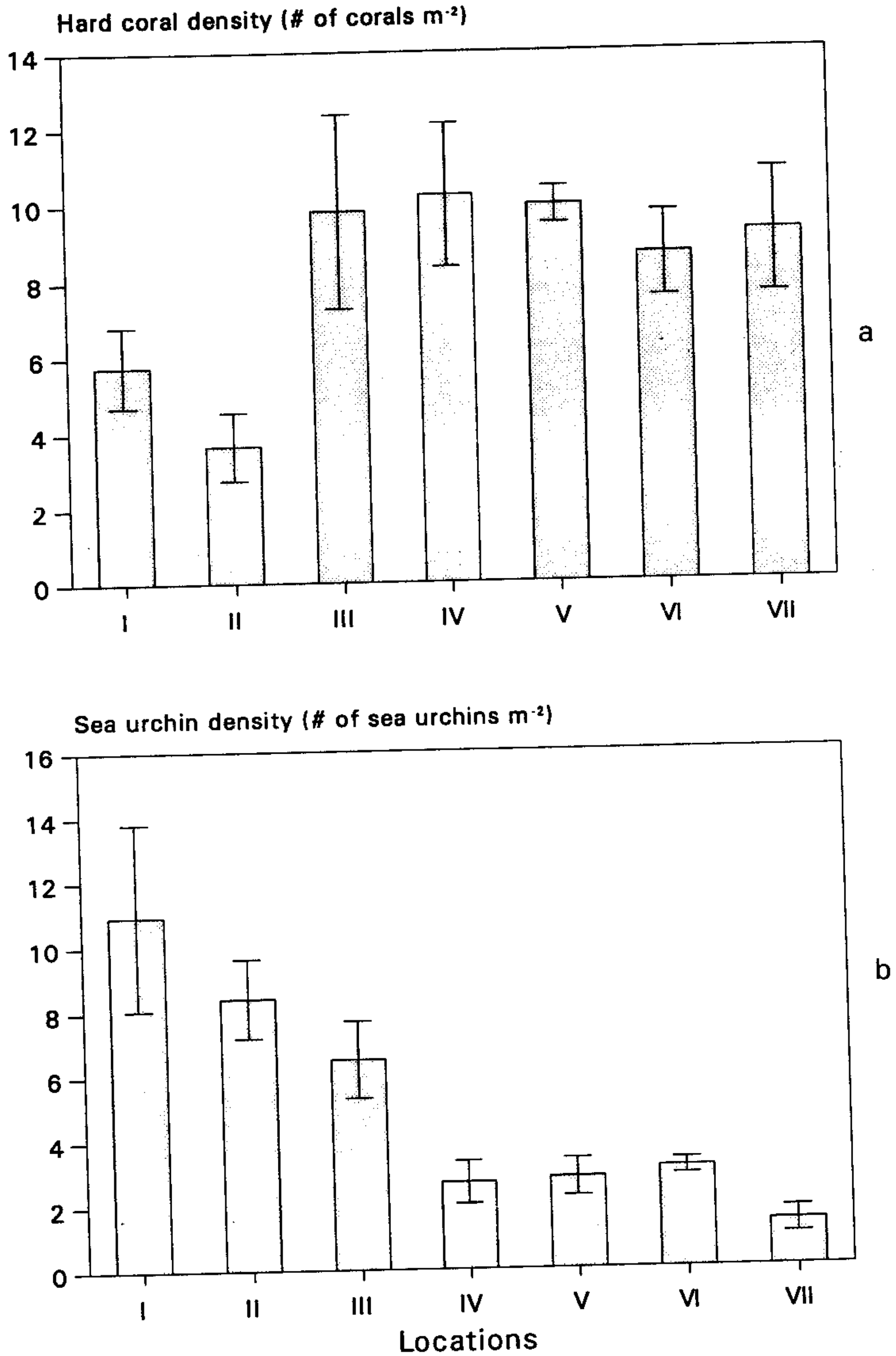


Fig. 37. Mean densities (±SE) of hard corals (**a**) and sea urchins (**b**) at the surveyed locations.

Coral cover :

The term cover refers to the amount of substrate occupied by an organism, often expressed as a percent. The live hard coral cover (%) has significantly differed among the studied sites (ANOVA, $F = 2.45$, $df = 149$, $P<0.05$; Table 11). Site IV had the highest percentage of living hard coral cover (56.65 ± 9.17%), while site II (Ras Za'farana) had the lowest percentage (30.56 ± 4.71%, Table 10 & Fig. 38). Live hard coral cover in the area of study showed similar general trend as for species diversity, being higher in the offshore reef sites than that in the nearshore reef sites (Fig. 38).

Soft coral cover was also significantly different among the surveyed locations (ANOVA, $F = 6.21$, $df = 149$, $P<0.01$; Table 11). Location I reflected the highest percentage of soft coral cover (18.51 ± 1.35%), which was relatively similar to that recorded at location II (18.13 ± 4.79%). On the other hand, location VII reflected the lowest percentage of soft coral cover (3.37 ± 1.39%) which was nearly identical with that obtained at location III (Table 10 & Fig. 38). Soft coral cover did not show the same trend as for species diversity and live hard coral cover, but it was higher in the northern nearshore reefs (Gulf of Suez) than in the southern nearshore reefs (northern Red Sea proper). Figure 39 shows high significant negative correlation between soft coral cover (%) and live hard coral cover (%) at locations I, II, IV, V and VI ($r^2 = 0.78$, 0.64, 0.82, 0.70 and 0.65 respectively; $P<0.05$).

The dominant hard coral species in terms of percentage cover at each surveyed site are displayed in Figure 40. The hermatypic coral species which contributed more than 5% of the total live hard coral cover in each investigated site were listed with the dominant species.

Acropora pharaonis (which occupied 17.68% of the total live hard coral cover) was the most dominant species in site I, while the second,

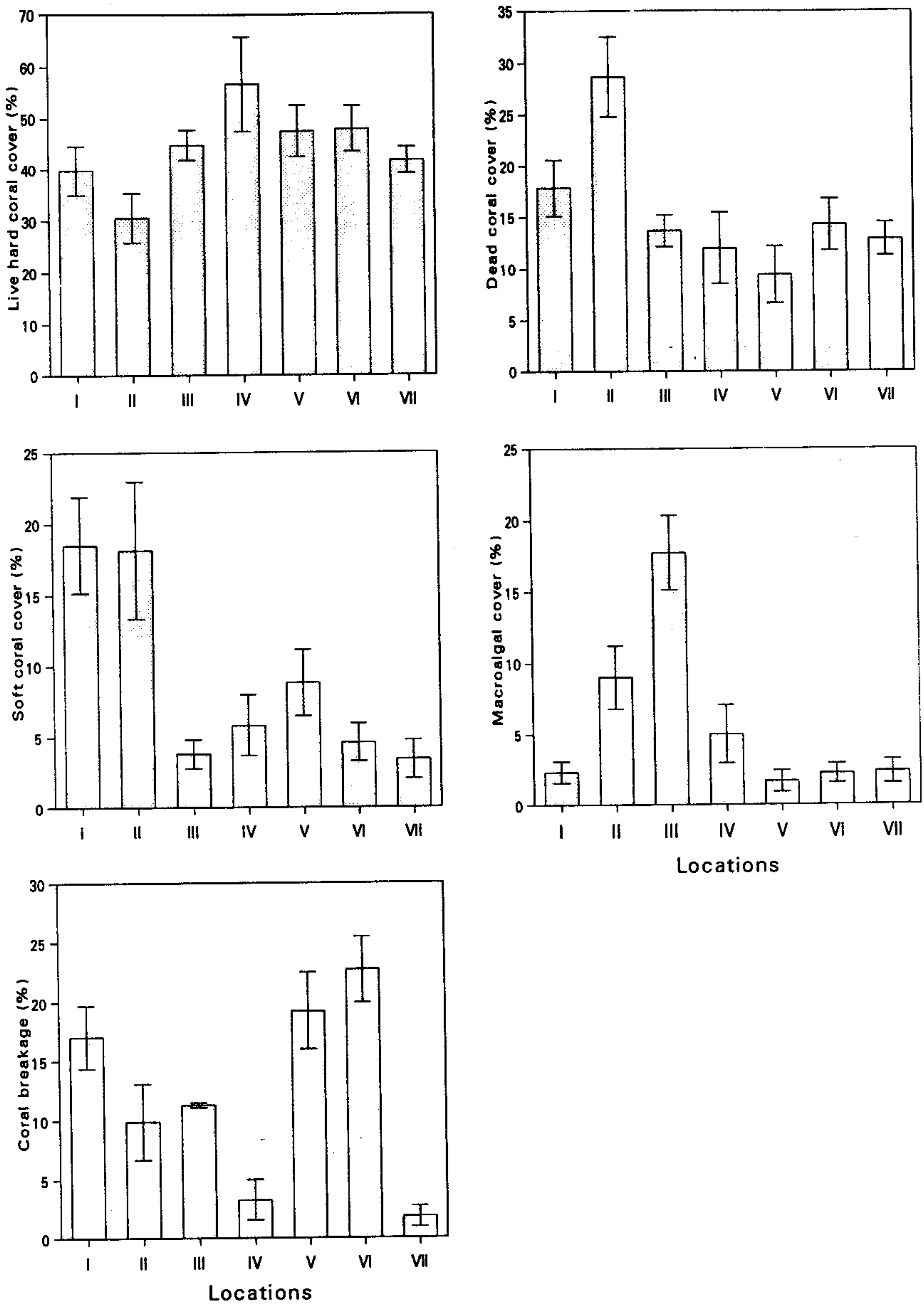


Fig. 38. Mean percentages (±SE) of live hard coral cover, soft coral cover, dead coral cover, coral breakage and macroalgal cover at the surveyed locations.

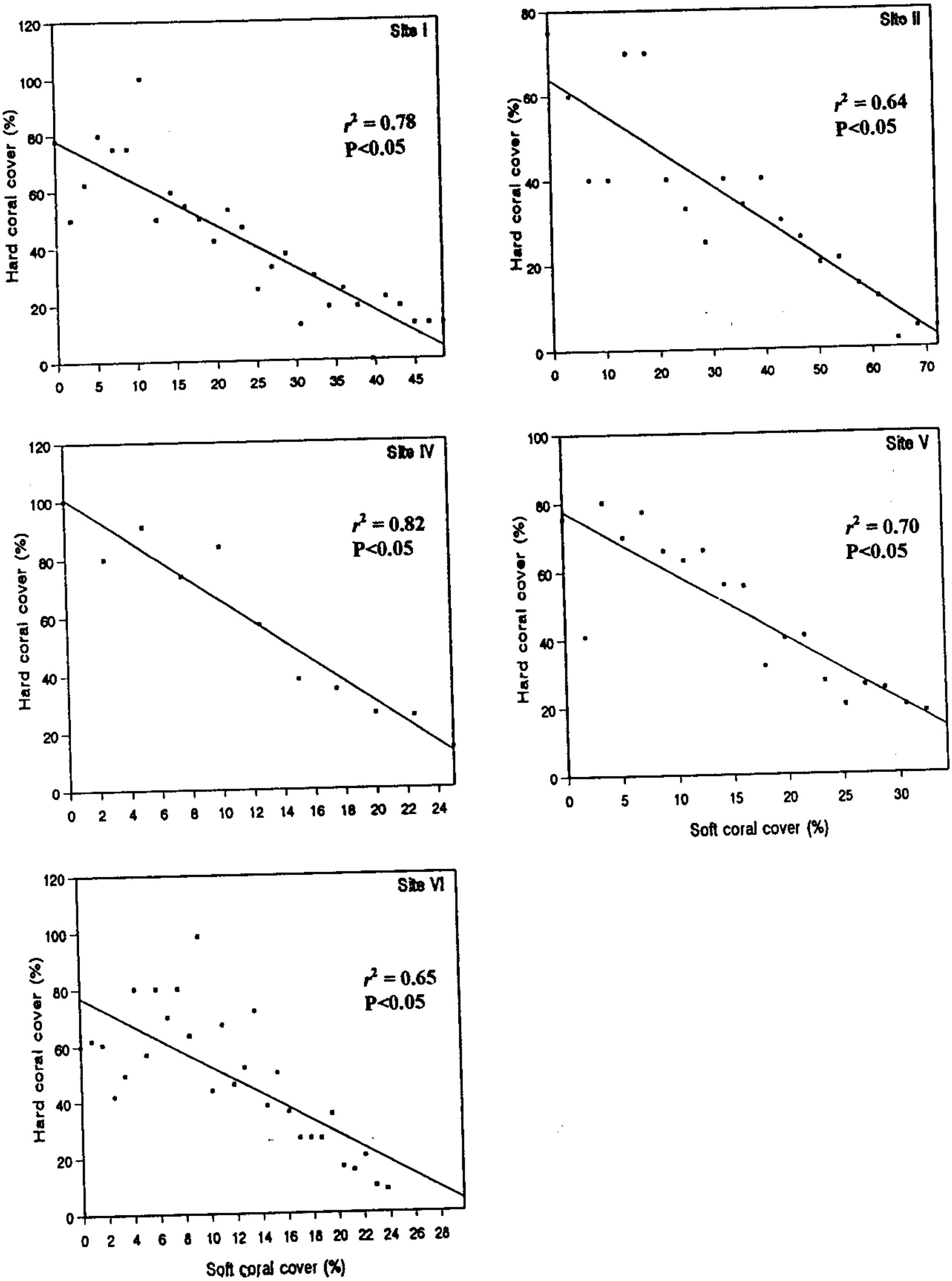


Fig. 39. Relationship between live hard coral cover (%) and soft coral cover (%) at locations I, II, IV, V and VI.

third and fourth most dominant species were *Galaxea fascicularis* (12.13%), *Echinopora gemmacea* (8.67%) and *Lobophyllia corymbosa* (8.04%), respectively. The most dominant soft corals in this site were *Lithophyton arboreum*, which constituted 51.86% of total soft coral cover (or 9.60% of total cover), whereas *Lobophytum pauciflorum* ranked the next dominant soft corals, constituting 25.77% of the total soft coral cover (or 4.77% of the total cover, Table 9).

Site II was clearly dominated by *Stylophora pistillata* (30.59% of total live hard coral cover), *Acropora pharaonis* (22.05%) and *Acropora forskali* (14.40%). The three species are accounted for more than two-thirds of total live hard coral cover in this area. The dominant soft corals in site II were also *Lithophyton arboreum* and *Lobophytum pauciflorum*, contributing 44.73 and 40.15% of the total soft coral cover, respectively.

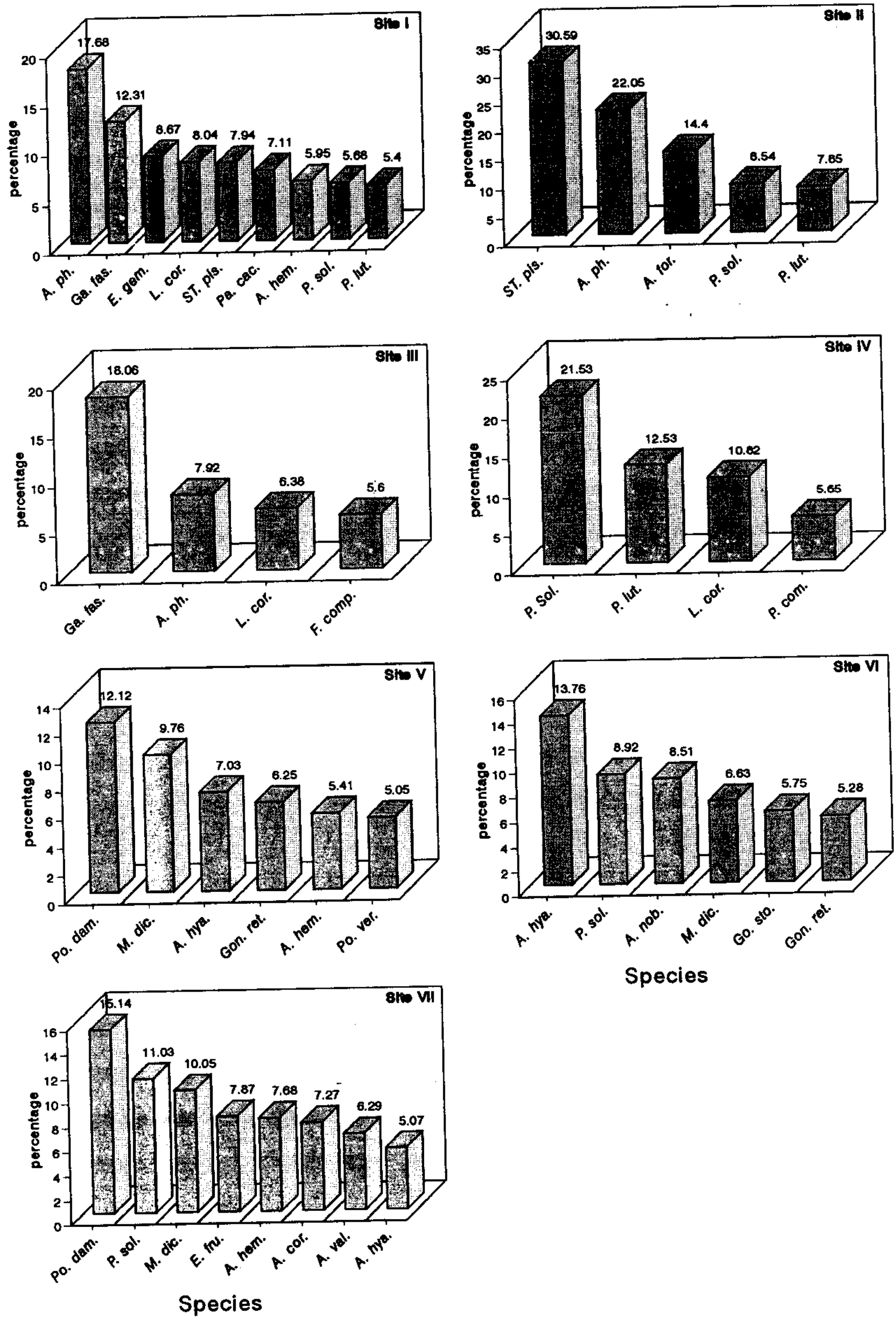
Site III was dominated by *Galaxea fasicularis*, which was accounted for 18.06% of the total live hard coral cover. Although, *Acropora pharaonis* ranking next to *Galaxea fascicularis*, it constituted only 7.92% of the total live hard coral cover. The common soft corals in this site were *Sarcophyton trocheliophorum*, *Lithophton arboreum* and *Xenia* spp. (constituting 41.91, 31.03 and 27.05% of the total soft coral cover, respectively, Table 9).

Site IV was characterized by a high share of *Porites* spp. (occupying 39.71% of the total live hard coral cover), where *Porites solida* had the greatest percentage cover (21.53% of total live hard coral cover), followed by *Porites lutea* (12.53%) and *Lobophyllia corymbosa* (10.82%). *Xenia* spp. were the most abundant soft corals in terms of percentage cover at site IV, they are accounted for 78.01% of the total soft coral cover and 4.54% of the total cover at this site (Table 9).

Fig. 40. Percentage covers of hard coral species, which occupied more than 5% of the total live hard coral cover in each surveyed location.

Abbreviations:

A. ph. = *Acropora pharaonis*	*A. hem.* = *Acropora hemprichi.*
A. for. = *Acropora forskali.*	*A. hya.* = *Acropora hyacinthus.*
A. nob. = *Acropora nobilis.*	*A. cor.* = *Acropora corymbosa.*
A. val. = *Acropora valida.*	*St. pis.* = *Stylophora pistillata.*
Po. dam. = *Pocillopora damicornis.*	*Pa. Cac.* = *Pavona cactus.*
Po. ver. = *Pocillopora verrucosa.*	*P. sol.* = *Porites solida.*
P. com. = *porites compressa.*	*L. cor.* = *Lobophyllia corymbosa.*
P. Lut. = *Porites lutea.*	*E. gem.* = *Echinopora gemmacea.*
E. fru. = *Echinopora fruticulosa.*	*Go. sto.* = *Goniopora stokesi.*
M. dic. = *Millepora dichotoma.*	*F. comp.* = *Favites complanata.*
Gon. ret. = *Goniastrea retiformis.*	*Ga. fas.* = *Galaxea fascicularis.*


Site I
percentage
17.68
12.31
8.67
8.04
7.94
7.11
5.95
5.68
5.4
A. ph.
Ga. fas.
E. gem.
L. cor.
ST. pis.
Pa. cac.
A. hem.
P. sol.
P. lut.
Site II
percentage
30.59
22.05
14.4
8.54
7.65
ST. pis.
A. ph.
A. for.
P. sol.
P. lut.
Site III
percentage
18.06
7.92
6.38
5.6
Ga. fas.
A. ph.
L. cor.
F. comp.
Site IV
percentage
21.53
12.53
10.82
5.65
P. Sol.
P. lut.
L. cor.
P. com.
Site V
percentage
12.12
9.76
7.03
6.25
5.41
5.05
Po. dam.
M. dic.
A. hya.
Gon. ret.
A. hem.
Po. ver.
Site VI
percentage
13.76
8.92
8.51
6.63
5.75
5.28
A. hya.
P. sol.
A. nob.
M. dic.
Go. sto.
Gon. ret.
Species
Site VII
percentage
15.14
11.03
10.05
7.87
7.68
7.27
6.29
5.07
Po. dam.
P. sol.
M. dic.
E. fru.
A. hem.
A. cor.
A. val.
A. hya.
Species

Pocillopora damicornis had the greatest percentage cover at site V (contributed 12.12% of the total live hard coral cover), whilst the second and third positions were occupied by *Millepora dichotoma* (contributed 9.765%) and *Acropora hyacinthus* (contributed 7.03%), respectively. The dominant soft corals in this site was *Sinularia polydactyla* which constituted 39.98% of the total soft coral cover, followed by *Dendronephthya klunzingeri* which was accounted for 18.69% of the total soft coral cover.

In site VI the hard coral species with the highest percentage cover was *Acropora hyacinthus*, accounted for 13.76% of total live hard coral cover, while the second (8.92%) and third (8.5%) highest percentages were contributed by *Porites solida* and *Acropora nobilis*, respectively. The most dominant alcyonacean corals in this site were *Xenia* spp., *Lithophyton arboreum* and *Sinularia polydactyla* (representing, 29.41, 28.76 and 26.8% of total soft coral cover or 1.35, 1.32 and 1.23% of total cover, respectively; Table 9).

Pocillopora damicornis, *Porites solida* and *Millepora dichotoma* were the main reef builders in site VII (constituted 15.14, 11.03 and 10.05% of total live hard coral cover, respectively; Fig. 33). The soft coral cover (%) in this site was low and commonly represented by *Lobophytum pauciflorum* (constituted 40.65% of total soft coral cover) and *Sinularia polydactyla* (constituted 32.9% of total soft coral cover).

***Dead coral cover* :**

The present results showed a general decrease in dead coral cover from the nearshore reefs towards the offshore reefs, with the exception of location VI (Table 10 & Fig. 38). Also, dead coral cover was greater in the northern part of the study area than in the southern part. There were a significant variations in the percentage of dead coral cover between the surveyed reef sites (ANOVA, $F = 5.15$, $df = 149$, $P<0.01$; Table 11).

Table 10. Percentage covers of total live hard corals (LCOV), soft corals, macroalgae, dead corals and broken corals; number of hard coral species (S); Shannon and Weaver's species diversity (H'_N); Pielou's evenness index (J'); species richness (D); hard coral density (# of corals m^{-2}); population density of zooxanthellae (x 10^6 zoox. cm^{-2}); and population density of sea urchins (# of urchins m^{-2}) at the surveyed reef sites. The variables are presented as mean values with standard errors in parentheses, except for S, H'_N, J' and D.

Parameter	Location						
	I	II	III	IV	V	VI	VII
LCOV (%)	39.81	30.56	44.84	56.65	47.52	47.95	41.80
	[4.86]	[4.71]	[2.97]	[9.17]	[4.96]	[4.40]	[2.62]
Soft corals (%)	18.51	18.13	3.77	5.82	8.83	4.59	3.37
	[3.35]	[4.79]	[1.04]	[2.17]	[2.32]	[1.37]	[1.39]
Macroalgae (%)	2.27	9.00	17.73	5.00	1.64	2.18	2.31
	[0.75]	[2.22]	[2.6]	[2.10]	[0.75]	[0.67]	[0.84]
Dead corals (%)	17.85	28.70	13.71	12.00	9.43	14.27	12.90
	[2.74]	[3.89]	[1.51]	[3.47]	[2.78]	[2.48]	[1.61]
Corals breakage	17.04	9.87	11.30	3.27	19.23	22.73	1.85
(%)	[2.68]	3.22	[0.20]	[1.73]	[3.25]	[2.76]	[0.90]
S	24.00	12.00	38.00	37.00	38.00	43.00	27.00
H'_N	2.18	1.54	2.77	3.21	3.13	3.14	2.65
J'	0.69	0.62	0.76	0.89	0.86	0.83	0.80
D	4.54	2.48	7.00	7.46	7.06	7.60	4.87
Hard coral density	5.75	3.65	9.81	10.23	9.95	8.65	9.23
	[1.05]	[0.90]	[2.55]	[1.88]	[0.48]	[1.10]	[1.62]
x 10^6 zoox. Cm^{-2}	0.84	0.70	1.04	1.02	1.08	0.91	0.86
	[0.04]	[0.03]	[0.03]	[0.13]	[0.03]	[0.04]	[0.03]
Sea urchin density	10.93	8.40	6.50	2.67	2.80	3.10	1.40
	[2.87]	[1.21]	[1.20]	[0.66]	[0.58]	[0.24]	[0.40]

Table 11. Analysis of variance of some coral community variables among the studied locations. The appropriate F statistics, degree of freedom (df) and probability of a significant difference are shown : * = P<0.05, * * = P<0.01.

Variable	df	F	P	significance
Live hard coral cover (%)	149.00	2.45	0.03	*
Soft coral cover (%)	149.00	6.21	0.00	* *
Macroalgal cover (%)	149.00	15.06	0.00	* *
Dead coral cover (%)	149.00	5.15	0.00	* *
Coral breakage (%)	149.00	7.60	0.00	* *
Density of zooxanthellae	29.00	5.81	0.00	* *

Location II supported the largest mean percentage of dead coral cover (28.70 ± 3.89%), while location V supported the least mean value (9.43 ± 2.78%, Table 10). The highest occurrence value of dead corals (about 95%) at each location was observed within the first 10m depth.

***Coral breakage*:**

In all surveyed reefs the majority of broken corals (90%) was found within the first 10m depth. About 65% of all broken corals were observed in the shallowest 5m and 25% at depth range 6-9m. Coral breakage showed a high significant difference among reef areas (ANOVA, F = 7.60, df = 149, $P<0.01$; Table 11). The highest mean percentage of coral breakage 22.73 ± 2.76% occurred at location VI. On the other hand, the lowest value 1.85 ± 0.9% occurred at location VII (Table 10 & Fig. 38). Significantly fewer corals ($P<0.01$) were broken on the reefs with low visitor frequency (locations II, IV & VII) than on reefs with high visitor frequency (locations I, V & VI). Moreover, the exposed reefs with low visitor frequency (locations II & VII) showed also fewer broken corals than less exposed reefs with high visitor frequency (locations I, V & VI). The most frequently broken corals in the area of study were *Acropora* spp., *Millepora dichotoma* and *Stylophora pistillata*.

In location I, the coral species with highest breakage were *Acropora pharaonis* (34.40% of total breakage value), *Stylophora pistillata* (25.50%) and *Acropora hemprichi* (10.70%).

The most frequently broken corals in location II were *Acropora pharaonis* (40.50% of total broken corals), *Stylophora pistillata* (31.70%) and *Acropora forskali* (12.20%).

The higher percentages of coral breakage in location III were contributed by *Acropora* spp., *Lobophyllia corymbosa* and *Galaxea* *fascicularis* (32.50, 19.20 and 17.50% of total coral breakage, respectively).

Millepora dichotoma, Acropora hyacinthus and *Pocillopora damicornis* had greater percentages of total coral breakage in location V (36.40, 23.50 and 18.20% of total coral breakage value, respectively) than other species.

Again, *Acropora hyacinthus* and *Millepora dichotoma* were the most frequently broken corals in location VI, in addition to *Acropora nobilis* and *Acropora hemprichi*. *Acropora hyacinthus* alone accounted for 40% of total broken corals, whereas *Millepora dichotoma, Acropora nobilis* and *Acropora hemprichi* were accounted for 18.90, 15.30 and 11.12%, respectively.

Macroalgal cover (algal overgrowth):

It was observed that the macroalgae readily colonized dead parts of coral colonies and overgrown adjacent living coral tissue, particularly at sites II & III. There was highly significant difference in the percentage of macroalgal cover among studied reef sites (ANOVA, $F = 15.06$, $df = 149$, $P<0.01$; Table 11). The greatest mean percentage of macroalgal cover, $17.73 \pm 2.6\%$ was found in site III, whereas the lowest mean value, $1.64 \pm 0.75\%$ was found in site V (Table 10 & Fig. 38). Percentage of macroalgal cover showed high inverse significant correlation with living hard coral cover (%) at sites II, III and IV ($r^2 = 0.70$, 0.92 and 0.75 respectively, $P<0.05$; Fig. 41). It is noted that this correlation was very high at site III (Marine Biological Station).

Distribution of macroalgae in the study area is presented in Table 12. The most abundant macroalgae, which competed and overgrown corals in location III, were *Padina pavonia, Cystoseira myrica, Turbinaria elatensis, Ulva lactuca, Caulerpa serrulata* and *Caulerpa racemosa*. The dominant macroalgae in location II (Gulf of Suez) were

Table 12. Distribution of macroalgae among the investigated reef sites.

Macroalgae	Location						
	I	II	III	IV	V	VI	VII
Phaeophyceae							
Padina pavonia	0	++	+++	0	0	0	0
Cystoseira myrica	0	+	+++	0	0	0	0
Turbinaria elatensis	0	++	+++	0	+	++	0
Colpomenia sinuosa	+	+++	++	0	0	0	0
Hydroclathrus clathratus	+	+++	+	0	0	0	0
Chlorophyceae							
Caulerpa racemosa	0	0	+++	0	0	0	0
Caulerpa serrulata	0	+	+++	0	0	0	0
Valonia ventricosa	0	0	+	0	0	0	0
Ulva lactuca	+	++	+++	+	0	0	0
Halmida tuna	0	+	+++	+	+	+	+
Enteromorpha spp.	+	+	++	+	0	+	0
Rhodophyceae							
Laurencia obtusa	0	0	0	0	0	0	++
Laurencia papilosa	0	0	0	0	0	0	++
Hypnea valintae	0	0	0	0	0	0	+
Galaxura oblongata	0	+	0	+	0	0	0

0 = Not found + = rare

++ = moderate +++ = abundant

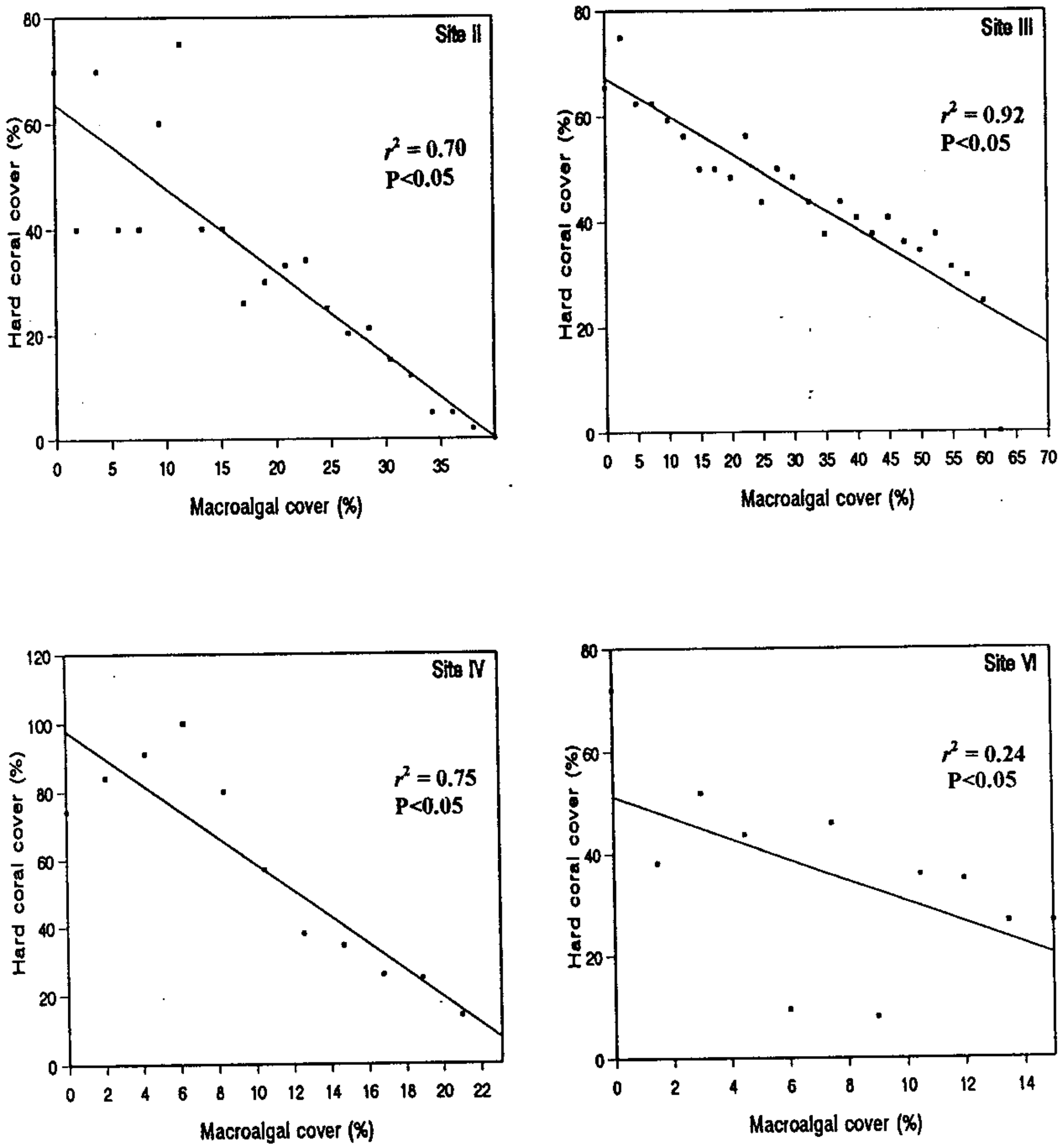


Fig. 41. Relationships between live hard coral cover (%) and macroalgal cover (%) at locations II, III, IV and VI.

Colpomenia sinuosa, *Hydroclathrus clathratus*, *Turbinaria elatensis* and *Ulva lactuca*. In location VII most of macroalgae were belonging to *Rhodophyceae* (red algae), *Laurencia obtusa*, *Laurencia papilosa* and *Hypnea valintae*. *Acropora* species and *Stylophora pistillata* were the most affected corals with algal overgrowth.

Population density of Zooxanthellae:

Zooxanthellae are endosymbiotic single-cell dinoflagellate algae, which normally give the coral tissue a brownish coloration. They are living principally in the gastrodermal tissues (within cells) of hermatypic corals and have been identified as *Gymnodinium microadriaticum*. The mean and standard error for population density of zooxanthellae in the tissue of *Stylophora pistillata* at each location was calculated (Table 10 and Fig. 42). Statistical analysis for population density of zooxanthellae indicated the presence of a significant difference among the studied locations (ANOVA, $F = 5.81$, $df = 29$, $P<0.01$; Table 11). Location V displayed the highest mean value of zooxanthellae density $1.08 \times 10^6 \pm 0.03 \times 10^6$ cells cm^{-2}, whereas location II displayed the lowest mean value $0.70 \times 10^6 \pm 0.03 \times 10^6$ cells cm^{-2}.

Coral diseases:

The occurrences and distribution of four coral diseases in the investigated sites are given in Table 13.

Coral Bleaching or Tissue Bleaching (TBL) is a temporary or permanent loss of the photosynthetic zooxanthellae (endosymbiotic microalgae) and/or their pigments, leaving whitened (bleached) coral colonies.

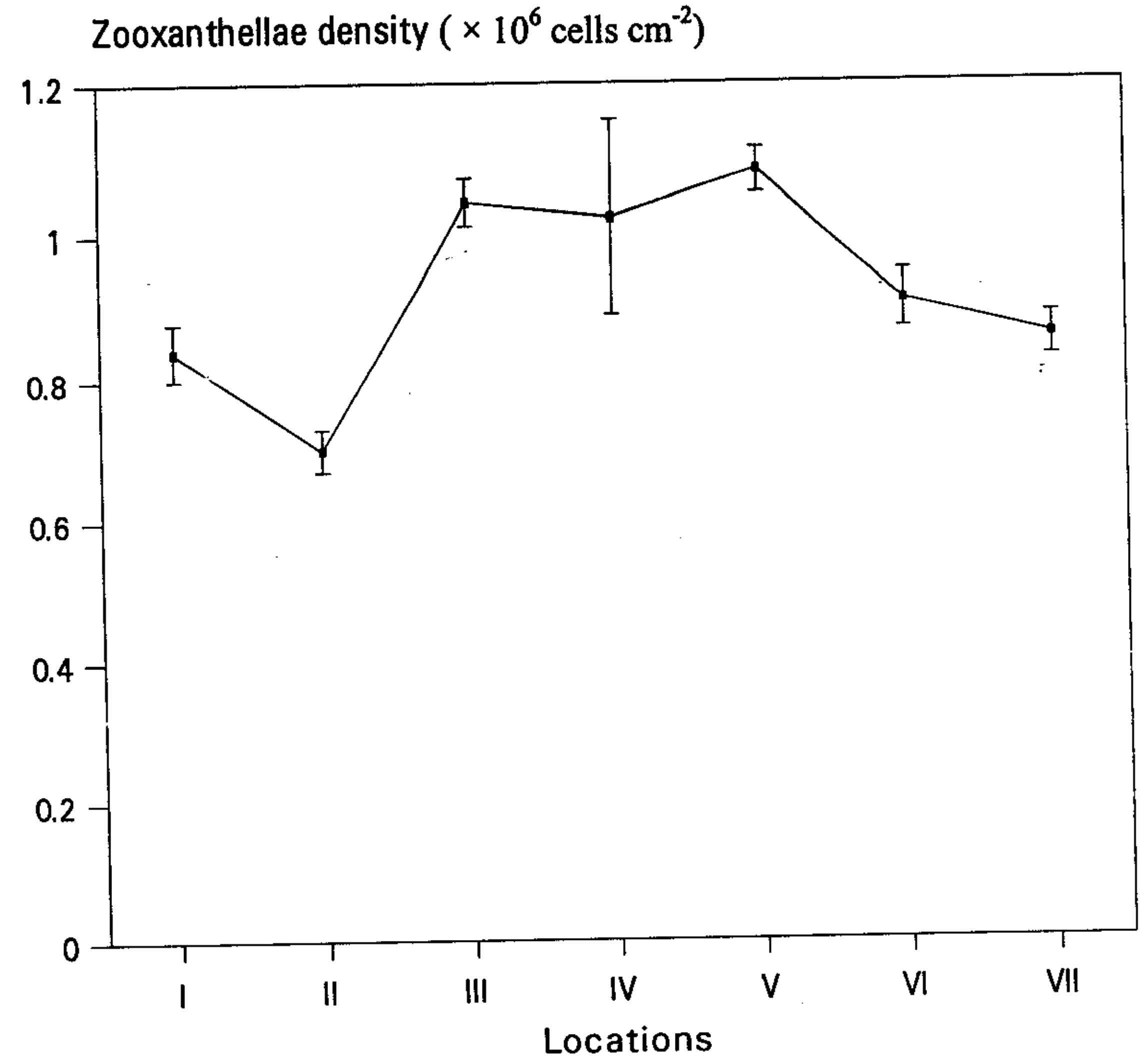


Fig. 42. Population density (±SE) of zooxanthellae (× 10^6 cells cm^{-2}) in tissue of *Stylophora pistillata* at the investigated locations.

The second disease was White Band Disease (WBD) which is a band of brilliant white coral skeleton always visible in the wake of a moving front of tissue destruction. It appeared as a simple interface across a coral colony, separating living tissue from denuded skeleton. On the branching coral growth forms, WBD almost without exception started from the base of the branches and proceeded to the tips. On hemispherical or platy growth forms, WBD is usually originated from an overhang, a hole, a crack, or an otherwise shady spot that show some algal overgrowth.

Black Band Disease (BBD) was found in the form of black algal ring around a center of tissue-stripped coral skeleton. It was killing more and more coral tissue, thus constantly enlarging the area of denuded coral skeleton (or they are usually found in the form of a dark band of algal material, surrounding and constantly enlarging an area of denuded coral skeleton). The origin of BBD seems to be dependent on some preceding necrotic process. It may also be settled at the exact borderline of an active WBD where coral tissue is disintegrated (or settled on already existing WBD). In this way a disease may change from WBD to BBD. Thus, WBD seems to be a precondition for the establishment of BBD.

Bacterial Infection (BIN) was found in the form of a web-like film over the layer of mucus.

Site I was characterized by a high occurrence of TBL and frequent occurrence of WBD. TBL was found abundantly on *Favites* species, *Lopophyllia corymbosa* and also on species of *Porites* and *Acropora*. WBD occurred mostly on *Echinopora gemmacea*, *Favites* species, *Lobophyllia corymbosa*, *Platygyra lamellina* and genus *Acropora*. BIN was in the intermediate range.

Site II showed frequent occurrences of TBL and BIN, while the occurrence of WBD fell consistently into the intermediate range.

Table 13. Summary of occurrence and frequency of four coral diseases at the investigated locations.

Location	Disease			
	TBL	WBD	BBD	BIN
I	4	3	1	2
II	3	2	1	3
III	2	3	3	3
IV	1	2	1	1
V	2	2	1	1
VI	2	2	1	2
VII	2	2	0	1

WBD = White Band Disease.
BIN = Bacterial Infection
TBL = Tissue Bleaching.
BBD = Black Band Disease.

Condition 0 = not found.
1 = rare (1-3 cases per scan)
2 = moderate (4-12 cases per scan)
3 = frequent (13-25 cases per scan)
4 = abundant (26-50 cases per scan)

Platygyra species, *Porites* species and *Siderastraea savignyana* were the most susceptible corals to the TBL in this site. BIN was observed mostly on *Acropora* species, *Stylophora pistillata* and *Pavona explanulata*.

Site III was characterized by frequent occurrence of Black Band Disease (BBD) which commonly afflicted *Platygyra lamellina*, *Goniastrea retiformis*, *Goniastrea pectinata* and *Hydnophora microconos*. Also, BIN and WBD displayed frequent occurrence. *Echinopora gemmacea*, *Oxypora lacera* and *Seriatopora hystrix* were the most commonly afflicted species with BIN. WBD was found mostly on *Acropora* species, *Montastrea curta*, *Oxypora lacera*, *Seriatopora hystrix* and *Platygyra lamellina*.

The incidence of coral diseases in site IV was considerably lower than other sites. The impact of WBD seemed to be moderate, whereas the occurrence of BBD, BIN and TBL were very scarce. WBD was recorded on *Porites solida*, *Porites lutea*, *Goniastrea retiformis* and *Acropora hyacinthus*.

Site V showed moderate occurrences of WBD and TBL. TBL was common on genus *Acropora*, *Pocillopora damicornis*, whilst WBD was found mostly on *Acropora hyacinthus*, *Acropora cytherea*, *Goniastrea retiformis* and *Millepora dichotoma*.

Site VI exhibited moderate occurrences of TBL, WBD and BIN. WBD was found mostly on *Acropora hyacinthus*, *Acropora hemprichi*, *Millepora dichotoma* and *Goniastrea retiformis*, whereas TBL was observed on species of *Acropora*, *Goniopora* and *Porites*. *Goniastrea retiformis*, *Porites solida* and *Platygyra daedalea* were found the most affected species by BIN.

The occurrences of TBL and WBD in site VII were in the intermediate range. TBL was occurred widely on *Acropora* of various species, *Pocillopora damicornis* and *Porites solida*. WBD was registered mostly on *Acropora* species, *Porites solida, Millepora dichotoma* and *Pocillopora damicornis*.

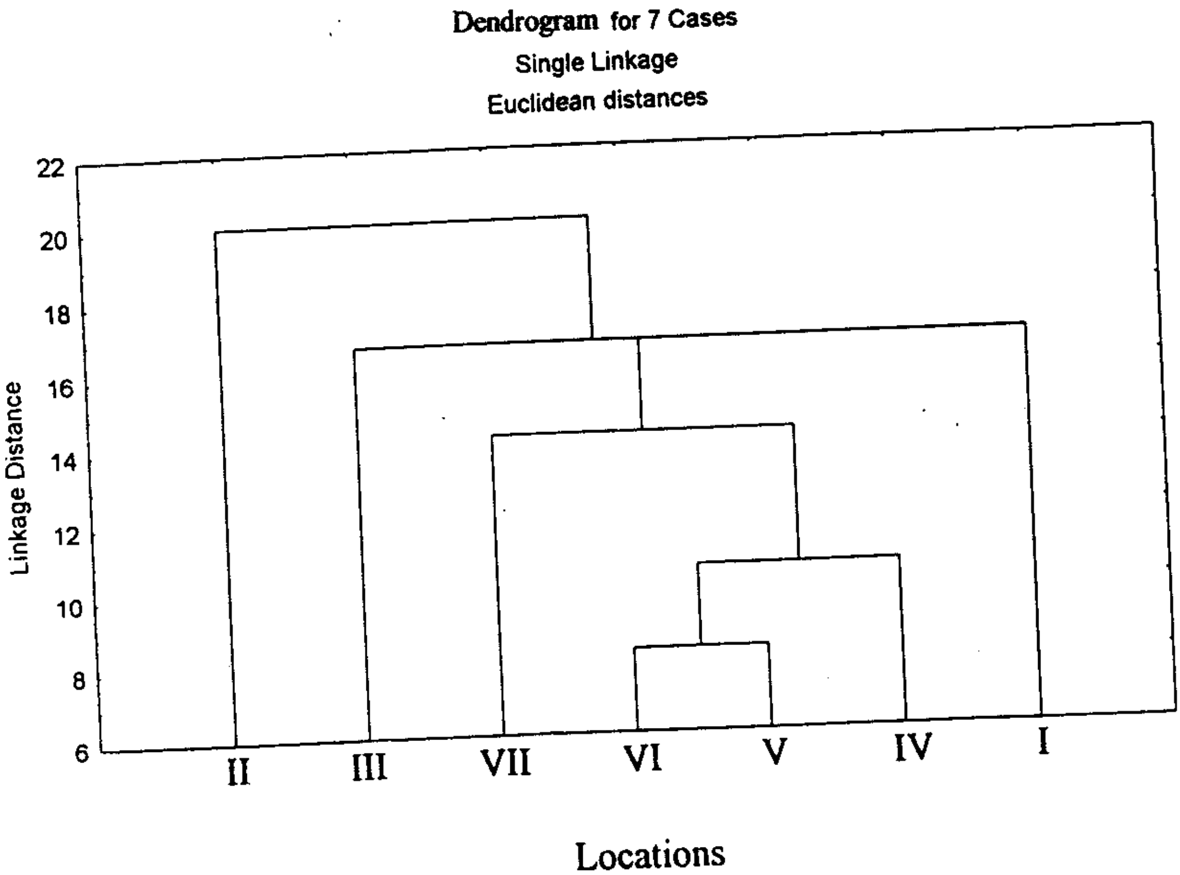


Fig. 43. Cluster analysis dendrogram showing the similarity in coral community variables among the surveyed locations. Dendrogram shows also the higher similarity between locations I, II and III.

DISCUSSION

Discussion

The unique marine ecosystem of coral reefs expresses varying levels of degradation as a result of increasing anthropogenic pressures of different amplitudes. The degradation of coral reefs in the Egyptian Red Sea coast has increased dramatically over the last two decades. Considering the amount and the distribution of the observed damage over the reefs, the questions arise whether it is caused by humans or not, and whether it has a deleterious effect on the viability of the coral communities. Since the oceanographic and climatic conditions in the area under investigation are constant (Edwards, 1987; Sheppard *et al.*, 1992), the main causes of coral reef damage are predominantly anthropogenic. Reefs in the northern Red Sea are especially vulnerable to degradation due to limited water circulation and high temperature extremes (Bryant *et al.*, 1998). The most frequent kinds of damage observed in the area of study are coral breakage, coral death, algal overgrowth, and enhanced coral diseases (Tables 10 & 13). The major anthropogenic disturbances leading to coral reef devastation in the investigated locations include: excessive sedimentation arising from coastal development and construction activities, mainly for tourism development (such as construction of hotels, recreational resorts, artificial lagoons); oil pollution due to petroleum activities (oil exploration, production and transportation); increased sewage run-off; inshore and offshore recreational activities (such as SCUBA diving, snorkelling, trampling on reef flats, tourist boat anchoring, ... etc.); and heavy collection of reef fauna, like shells, ornamental reef fishes and corals to be sold as souvenirs. This study pointed out also to the combined effect of several anthropogenic factors in some locations, as well as the interaction between man-made and natural impacts. The synergism between natural and man-made disturbances leads to extensive deterioration to coral reefs (Johannes, 1975; Loya, 1975; Zann, 1994). Brown (1997) stated that the impact of multiple stresses, both natural and human-caused, can have a multiplicative effect on reef ecosystem. The present study agree with many workers (Wells, 1992; Glynn, 1994; Peters & McCarty, 1996; Sebens, 1994; Bryant *et al.*, 1998), who believe that reef degradation frequently occurs through the interaction of a combined human-caused environmental factors, which leaves reef communities less resistant to periodic natural disturbances. Natural stresses such as normal occurrence of coral diseases, temperature extremes, and severe low tide periodically devastate corals. However, healthy corals are resilient and will recover with time.

Hydrographic conditions:

Seawater temperature, salinity, pH, and dissolved oxygen were measured in the area of study (Table 1) because the organisms that build coral reefs (mainly hermatypic corals) generally survive only within a narrow ranges of temperature, salinity, and pH, with high concentration of dissolved oxygen (Rogers *et al.*, 1994). Peters *et al.* (1997) reported that corals are very susceptible to changes in environmental conditions including temperature, outside the range that they normally experience. Such changes can affect the coral-algal symbiosis and, ultimately, calcification and the entire reef community. Air temperature increases of 0.2-0.3°C/decade will induce slower increases in sea-surface temperatures, which may cause localized, or regional increases in coral bleaching (Wilkinson, 1996). Glynn (1996) suggested that temperature-induced coral bleaching is the only global climate change impact that may directly damage reefs. The ideal temperature range for the coral fauna in the Red Sea is between 20 °C in winter and about 28 °C in summer (Scheer, 1984). The range of water temperature measured by the author during summer season in the area of present study (28.75-30 °C, Table 1) was higher than that previously reported by other authors during summer at the northern part of Red Sea. The water temperature measured during summer, 1967 in the Gulf of Suez was varied between 27.65 and 28.40 °C (Meshal, 1967). Abdel-Salam (1981) reported that water temperature recorded during summer, 1981 in the area of Hurghada, Red Sea was varied from 26.00 to 28.00 °C. The water temperature of the northern Red Sea water measured during summer, 1995 by Hamed (1996) was ranged from 28.00 to 28.25 °C. The present data indicated the general decrease of water temperature away from the shore. It is clear that the increases in water temperature in the surveyed locations may play a role in the enhanced coral bleaching, especially if it interacts synergistically with the man-made disturbances (Sebens, 1994; Done, 1999). Chadwick-Furman (1996) noted that, the global climate change, particularly temperature raising may interact with regional anthropogenic processes (e.g. pollution and sedimentation) to create additional impacts on coral diversity and exacerbate the rate of coral species extinctions. Wilkinson (1996) stated that the greatest impact of climate change (stimulated by anthropogenic increase in greenhouse gases) will be a synergistic enhancement of direct anthropogenic stresses (excessive sedimentation; pollution from the land; over-fishing, especially via destructive methods), which currently cause most damage to coral reefs. Furthermore, healthy reefs may be able to withstand the negative impacts of global warming more successfully than reefs under stress from anthropogenic factors (Wells, 1992).

The maximum annual mean of salinity (42.65 ± 0.17 ppt) was recorded at location I, while the minimum value (39.64 ± 0.20 ppt) was recorded at location VII (Table 1). This condition may be ascribed to the high evaporation rate and low wave action at location I, which is a shallow basin and is protected from the prevailing northerly winds by Ataqa mountain. In contrast, the low salinity reported at location VII may be attributed to the high wave action and good water circulation in this area. This investigation indicated also that the average salinity values in the area of study decreases from north (Gulf of Suez) to south (Gasus I, northern part of the Red Sea proper). This is consistent with the results of Abdel-Salam (1981); El-Sabh & Beltagy (1983); and Hamed (1996) obtained from the Gulf of Suez and the northern part of the Red Sea proper. The influence of evaporation and perhaps dissolution of some evaporite beds from the bottom are the main causes responsible for increasing salinity in the Gulf of Suez (Beltagy, 1983; El-Sabh & Beltagy, 1983). Grasshof (1969) attributed the increase of the surface water salinity from south to north in the Red Sea, to the evaporation and mixing of surface water with the more saline deep water through turbulence. Morcos (1970) suggested that high salinity in the Gulf of Suez and its rapid increase towards the north explained by not only the high rate of evaporation, but also the influence of high saline water of the Suez Canal. The range of salinity in the area of study (39.34-43.10 ppt) is higher than that reported from the Gulf of Aden (36-40 ppt; Scheer, 1984). High salinity (>41 ppt) is amongst the factors that causing a reduction in the number of coral species, whereas few number of species can survive with increased salinity (Sheppard, 1988). Consequently, it is suggested that the high salinity values recorded at sites I & II (Gulf of Suez) probably participated with other factors (such as sedimentation and chemical pollution) in reducing species richness and species diversity in this area. This is coincided with observations of Sheppard (1988), who indicated that the low number of coral species in the most northerly reefs near Suez (11 species) arises from increased sedimentation, high salinity (over 41 ppt) and raised temperature. Scheer (1984) reported that the salinity in the Gulf of Suez is even higher than Red Sea proper, and depresses the number of coral genera and species which can survive there. The present data are in disagreement with observation of Scheer (1984) because the high salinity levels is not the only ecological factor accounted for depressing number of coral genera and species in the Gulf of Suez. However, the present results agree with his finding in the respect of the salinity in the Gulf of Suez is higher than Red Sea proper.

The pH of reef waters (ranging from 7.5 to 8.4) does not vary much over time, but may be valuable to record for long-term monitoring as changes in pH may indicate that the reef is being affected by pollution (Rogers *et al.*, 1994). The pH values of reef waters in the present work were not obviously varied between the sampled locations and fell in the normal range of reef waters (Table 1).

The highest annual average concentration of dissolved oxygen measured at location VII (6.53 ± 0.17 mg O_2 L^{-1}; Table 1) may be reflected the importance of high gas exchange at the sea-air interface, due to high wave action and good water circulation in this area (El-Sabh & Beltagy, 1983). Beltagy (1983) demonstrated that the dissolved oxygen concentrations recorded at stations facing the open sea near Al-Ghardaqa, Red Sea were usually high, which reflected the effect of turbulence produced by wave agitation and tidal currents. Hamed (1996) ascribed the high concentration of dissolved oxygen (6.45 mg O_2 L^{-1}) near the Strait of Gubal (entrance of Red Sea proper from Gulf of Suez) to the active mixing and gas exchange in this area. He also attributed the low levels of dissolved oxygen (4.77 mg O_2 L^{-1}) at the northern part of Gulf of Suez to the effect of wastewater disposal in this area.

Eutrophication:

The dramatic increase in eutrophication during the past 20 years has clearly posed a potential threat to coral reefs along the Egyptian Red Sea coast (Walker & Ormond, 1982; Riegl & Velimirov, 1991). Eutrophication gradient is clearly a function of increased nutrient concentration, an increase of phytoplankton biomass, as indicated by high chlorophyll *a* concentrations (Tomascik & Sander, 1985; Bell, 1992; Bell & Elmetri, 1995). The principal source of nutrient enrichment or eutrophication in the area of investigation is the sewage pollution coincided with increased tourism development and expansion of coastal urbanization, especially in the area of Hurghada and the northern part of Gulf of Suez (Riegl & Velimirov, 1991; Hawkins & Roberts, 1994; Hamed, 1996). Heavy terrestrial sedimentation due to land reclamation and construction activities may also be considered as an important source of nutrients in some sampling locations (Rogers, 1977; Bell, 1992; Hunte & Wittenberg, 1992). It is suggested that the enhanced nutrient loading from anthropogenic sources in seawater can affect on reef building corals by stimulating the growth of both phytoplankton and macroalgae (Fishelson, 1973b; Bell, 1991, 1992; Wilkinson, 1996). This view is substantiated by the high positive significant ($P<0.05$) correlations between the concentrations of dissolved nutrients (NH_3-N, DIN and PO_4-P) and both of macroalgal cover (%) and chlorophyll *a* concentration (Fig. 7). Maragos (1972) found that, increases in nitrogen and phosphorus from sewage at Kaneohe Bay, Hawaii, have instigated the takeover by algae, *Dictyosphaeria cavernosa*. Walker & Ormond (1982) reported that algal overgrowth on corals in the northern Gulf of Aqaba was in response to a fertilization effect of phosphate pollution. Thus, the significant ($P<0.05$) highest percentage cover of macroalgae ($17.73 \pm 2.6\%$, Table 10 and Fig. 38) registered at location III (Marine Biological Station), may be ascribed to the highest nutrient concentrations (Table 3) resulted largely from increased sewage discharges. This area is directly impacted by the sewage disposal from urban Hurghada, seaside recreational resorts and hotels (Riegl & Velimirov, 1991; Hawkins & Roberts, 1994). Seepage of wastewater from septic trenches and hotel gardens into the shallow water of the sea constituted another source of nutrient enrichment. A more localized source of nutrient input into the reef at location III (Marine Biological Station) is the domestic sewage that is dumped into the sea from the population living in the Marine Biological Station itself. The elevated levels of nutrients recorded at location III may also be directly responsible for the increased phytoplankton (as indicated by chlorophyll *a*) concentration measured in this area (Table 3 & Fig. 9). Previous studies (Bell, 1992; Bell & Elmetri, 1995) have indicated that the suggested threshold concentrations of dissolved nutrients associated with the onset eutrophication in coral reef communities are in the range of 0.1-0.2 μM for PO_4-P and 1-2 μM for DIN. The high levels of dissolved nutrients (PO_4-P, NH_3-N and DIN) registered at location II (Table 3 & Fig. 6) probably explain the second highest mean values of macroalgal cover (Table 10 and Fig. 38) and chlorophyll *a* concentration (Table 3) reported in this area. The high concentrations of nutrients in this area have probably resulted from a combination of increased sewage pollution and terrestrial sediment run-off. This reef is affected by the sewage pollution derived from the recreational resorts that lie to the north of Ras Za'farana and reaches to location II in the course of wind induced southerly currents (Roberts, 1985; Edwards, 1987). Also, the heavy terrestrial sediment input recorded in this reef (Table 5) may provide a significant source of nutrients for algal growth (Bell, 1992). The third highest mean concentration of dissolved nutrients measured at location I (Table 3 and Fig. 6) was probably a direct result of sewage discharged from the recreational resorts adjacent to this reef and the high terrestrial sedimentation rate.

It seems that the increased nutrient enrichment from anthropogenic sources and the associated algal growth may have negative impacts on coral reefs of the investigated area via a number of complex and interacting routes. These impacts can cause a reduction in the diversity and percentage cover of reef building corals, and eventually leading to the replacement of coral community with various flora and fauna (Smith *et al.*, 1981; Tomascik & Sander, 1987a; Kinsey, 1988). Nutrient enrichment enables the macroalgae to colonize dead parts of coral colonies, overgrow living coral tissue and kill corals (Fishelson, 1973a; Walker & Ormond, 1982; Hawkins & Roberts, 1996). Since the increased eutrophication from anthropogenic sources stimulated the growth of macroalgae, it can favor the macroalgae in their competition with reef building corals for space and light, resulting in physical smothering of corals (Benayahu & Loya, 1977; Grigg & Dollar, 1990; Wittenberg & Hunte, 1992). The high inverse significant ($P<0.05$) relationship established between macroalgal cover (%) and hard coral cover (%) at locations II, III and IV (r^2 = 0.70, 0.92 and 0.75 respectively, Fig. 41), may indicate the strong competition between them for space and light, as well as the detrimental effects of enhanced macroalgal overgrowth on hermatypic corals. Fishelson (1973b) noticed a marked increase in algae with a simultaneous decline in the coral colonies in the Gulf of Aqaba. He thought that the phosphate eutrophication of the shallow lagoon at Eilat might be a significant factor in creating better conditions for algal growth and, thus, increasing competition for space between algae and corals. Marsalak (1981) found that the most pronounced effects of sewage effluents on coral morbidity and mortality were not directly related to effluent toxicity, but were the result of competition with algae for space and light. Riegl & Velimirov (1991) suggested that the high algal overgrowth on living and dead corals in the fringing reef at Hurghada, Red Sea may be caused by waste discharges and additional sediment input due to land reclamation and building activities in the lagoon. Wilkinson (1996) reported that excess inorganic nitrogen and phosphorus favor the growth of algae over symbiotic corals: phytoplankton growth results in light shielding to corals; and enhanced macroalgal growth increases the competition with corals and results in physical smothering. The most frequently overgrown corals by macroalgae in the study area were the branching ones *Acropora* spp., *Pocillopora* spp., *Stylophora pistillata* and *Seriatopora hystrix*. The genus *Acropora* had the most algal overgrowth. This may in part be caused by the fact that the species of this genus are very common and proportionally more often affected by tissue loss and breakage, their damaged surfaces offering a favorable substrate for fouling algae in the area of high nutrient concentration. In this context, the present observations agree with those of Fishelson (1973a); and Riegl & Velimirov (1991). The results presented in tables 3 and 13 clarify that increased water eutrophication can also enhance the diseases in corals, especially the Black Band Disease (BBD). This is because the elevated levels of dissolved nutrients from anthropogenic sources can trigger the growth of the Cyanophyte algae *Phormidium corallyticum*, which is the main cause of BBD (Antonius, 1985, 1988a; Rutzler *et al.*, 1983; Brown & Howard, 1985; Santavy & Peters, 1997; Edmunds, 2000). According to Banner (1974); Smith *et al.* (1981); Maragos *et al.* (1985) and Bell (1992), the enhanced growth of macroalgae stimulated by nutrient enrichment may reduce the ability of reef building corals to reproduce by inhibiting the settlement of planulae larvae and recolonization of corals. Collins (1978) noted that increased algal growth associated with increased eutrophication make the conditions unsuitable for the settlement of coral larvae on inshore reefs in the Townsville, Australia. Wilkinson (1996) indicated also that increased turf algal growth caused by elevated levels of dissolved nutrients (N and P) cover surfaces, thereby preventing coral recruitment.

It is possible to suggest that the enhanced algal growth promoted by anthropogenic nutrient enrichment in the water column of the studied locations can greatly aggravate the devastating effect of increased sedimentation by their sediment trapping ability. Sediments trapped by the larger algal mat, growing on already damaged parts of corals, may spread or fall onto the coral tissues, placing it under further stress and hastening its death (Walker and Ormond, 1982; Bell, 1992). Bell & Elmetri (1995) attributed the lack of the coral reef recovery in the inner GBR lagoon to the combined affects of increased eutrophication and sedimentation.

There are some evidences that excess of nutrient concentrations have been found to reduce the rate of coral calcification (Kinsey & Davies, 1979; Walker & Ormond, 1982; Hawkins *et al.*, 1991). This is probably attributed to the increased nutrient concentrations may enhance the concentration of phytoplankton that have been implicated in the reduction of available light for zooxanthellae photosynthesis, which required for coral calcification. In addition, the high concentration of PO_4-P may interfere with calcium carbonate formation (Kinsey & Davies, 1979). Kinsey (1988) found that coral calcification was dramatically reduced by 50-60% in response to nutrient enhancement. Tomascik & Sander (1985) indicated that high PO_4-P concentrations (0.21 μM) might play a role in suppressing coral calcification rates. Therefore, the only locations in the present study, where the PO_4-P concentrations may be high enough to play a role in suppressing coral calcification rates were III and II (0.42 ± 0.10 and 0.40 ± 0.13 μM respectively, Table 3).

Suspended particulate matter:

There were obvious negative impacts on the surveyed coral reefs that resulted from increased suspended particulate matter (SPM) concentrations in the water column (Fig. 10).

It is suggested that SPM is considered one of the most important measurements of stress on corals, since it reflects the total concentration of suspended particles in the water column (Tomascik & Sander, 1985). Previous studies (Chalker, 1981; Tomascik & Sander, 1985; Cortes & Risk, 1985; Wittenberg & Hunte, 1992) have suggested that increased SPM adversely affecting the growth rates of reef building corals because it is involved in the reduction of coral calcification through reduction of available light for zooxanthellae photosynthesis. This because light is scattered by suspended particles in the water column and hence illumination, a vital source of energy, is reduced. The enhanced SPM concentrations in the area of study may be occurred directly in consequence of terrigenous sediment inputs and resuspension of bottom sediments (Rogers, 1977; Neil, 1990; Woolfe & Larcombe, 1999). The high turbidity observed during the periods of winter and spring (Table 4) was probably a direct result of bottom sediment resuspension caused by high current velocity associated with wind speed over 10 knots (Edwards, 1987) and the activities of reef visitors in some sampled locations. In most of months, winds force of 4 knots or less occur between 60% and 70% of the time in the region to the north of 26°N of the Red Sea, including Gulf of Suez. So, the mean annual wind speed is a little over 10 knots with winter and spring being the windiest seasons (Edwards, 1987). The high nutrient concentrations was also contributed to the increased SPM by enhancing the phytoplankton concentration in the water column (Kinsey, 1988; Rogers *et al.*, 1994; Bell & Elmetri, 1995). This is supported by the strong positive correlation existed between SPM concentration and the concentrations of nutrients and chlorophyll *a* (Fig. 11). Tomascik & Sander (1985) investigated the effects of water quality on *Montastrea annularis* growth rates off Barbados over a gradient of increasing eutrophication. They found that mean concentration of SPM was the strongest single estimator of growth rate, with the most rapid coral growth occurring at the site farthest from the primary pollution source. Contribution of total organic matter to the SPM concentration was confirmed by the positive relationship found between them (r^2 = 0.51, P>0.05).

Rogers (1990) suggested that the mean concentrations of SPM for reefs, which are not subject to stresses from human activities, are generally less than 10 mg L^{-1}. In other words, the natural concentration of SPM for coral reefs is generally less than 5 mg L^{-1} and hence the reef is considered under the effects of anthropogenic stresses when the concentration of SPM is greater than 5 mg L^{-1} (Cortes & Risk, 1985).

It is important to note that concentration of SPM (Fig. 10 & Table 5) was generally higher at the northern part of the study area (Gulf of Suez) than that at the southern part (northern Red Sea proper). Such trend revealed that the Gulf of Suez is subjected to higher levels of sewage pollution and sedimentation than Red Sea proper. Hamed (1996) found that the concentration of SPM at the northern part of Gulf of Suez was due to the sewage effluent from the nearby dwellings.

The highest mean concentration of SPM (12.52 ± 1.18 mg L^{-1}) measured at location I (El-Ain Al-Sukhna) may be attributed principally to increased sewage disposal from the adjacent hotels and recreational resorts, high terrestrial sediment runoff resulting from intensive construction of recreational resorts and expansion of Suez-Hurghada high way at El-Ain Al-Sukhna, and the high percentage of total organic matter contained in sediments (12.15 ± 1.29%, Table 5). In addition, this area is recognized as a popular summer resort and is subjected to high visitor frequency by publics who stirring up bottom sediments whilst walking on reef flats, thereby increasing the SPM concentration in the water column. Increased sewage pollution and excessive terrigenous sedimentation were the main causes responsible for the second highest mean concentration of SPM (11.26 ± 1.27 mg L^{-1}) found at location II (Ras Za'farana). The third highest concentration of SPM (10.07 ± 1.04 mg L^{-1}) determined at location III (Marine Biological Station) may be a result of high concentrations of nutrients associated with sewage discharges reported in this area and the resuspended bottom sediments caused by wave action and boat anchoring. On the other hand, The lowest mean concentration of SPM (5.52 ± 0.31 mg L^{-1}) occurred at location V (Sh'ab Abu-Galawa) was probably ascribed to the fact that it lies farther from the land-based pollution caused by human activities.

Figure 12 demonstrated that high levels of SPM in the water column may be involved in the reduction of coral species diversity (r^2 = 0.51, P>0.05), evenness index (r^2 = 0.60, P<0.05) and hard coral density (r^2 = 0.48, P>0.05), and in increasing the percentage of dead coral cover (r^2 = 0.46, P>0.05). This is consistent with the results of Loya (1976a); Cortes & Risk (1985); Rogers (1990); Neil (1990); and Bell (1992). It is suggested that, the decreasing hard coral density accompanied with increasing SPM concentration in the present study is probably as a result of the lower light levels due to increasing SPM concentration in the water column may inhibit the development of coral larvae by reducing the amount of energy available to maturing ova or embryo (Neil, 1990). Thereby reducing the coral recruitment. This is supported by the observations of Tomascik & Sander (1987b), who found that reduced number of larvae from colonies of *Porites porites* growing on reefs polluted by nutrients and SPM. The reduction in evenness index and species diversity with increasing SPM concentration may be explained by the dominance of coral species which are more tolerant to high levels of SPM and reduced light intensity than other species (Woesik & Done, 1997). Rogers (1979) showed experimentally that reduction of incident light by suspended matter has a greater effect on some coral species than others.

Sedimentation rate:

Sedimentation from terrestrial runoff and increased coastal construction as a result of human activities in the area of investigation constitutes one of the biggest potential sources of reef degradation. Landfilling concomitant with construction of seaside recreational resorts and hotels, beach enhancement, coastal development, land reclamation, and protruded constructions inside the sea are the most important causes of increased sedimentation in the present study. Landfilling was proceeded probably to create prime but inexpensive land and improve the access to the sea for tourists or to the deep waters for SCUBA divers and snorkellers. The depositional hydrodynamic pattern has been affected as a result of blocking littoral currents by protruded constructions in the front of hotels and recreational resorts, subsequently, some coastal segments have been subjected to local downdraft erosion (Frihy *et al.*, 1996), thus increasing the rate of sedimentation on the nearby reefs. Land reclamation in the area of study can cause the change in current direction and erosion of the beaches, which will also affect the reef.

Site II (Ras Za'farana) is characterized by a very high sedimentation rate (317.76 ± 73.73 mg cm^{-2} day^{-1}, Table 5 & Fig. 14a) which was significantly ($P<0.01$) higher than those reported at the other sites (Table 6). This increase in sedimentation may be a result of uncontrolled massive construction of recreational resorts and hotels in the vicinity of this reef in addition to the high weathering activities including high precipitation rate and wind speed which are characteristic for Ras Za'farana area (Edwards, 1987). Moreover, the recreational resorts under construction accumulated huge amounts of fine sand and sediments on the shore to enhance and improve their beaches, as well as to create artificial shallow lagoons. Much of these fine sand and sediments are transported to the reef with the prevailing northerly winds. This assumption is confirmed by the higher percentage (90.20%) of fine (125-249 μm in diameter, 64.46%) and very fine (63-124 μm, 25.74%) sand deposited at Ras Za'farana (Table 7 & Fig. 14b). Also, the lowest percentage of total carbonate content (42.44 ± 1.92%) in reef sediments assessed at site II (Table 5) may indicate that the majority of sediments accumulated on this reef are terrigenous in origin (*i.e.* derived from land) due to human activities and high wind speed (Beltagy *et al.*, 1986; Cortes, 1990).

The principal causes responsible for the second highest sedimentation rate (65.65 ± 1.27 mg cm^{-2} day^{-1}) registered at site I (Table 5 & Fig. 14a) are: 1) active construction of recreational resorts and related beach replenishment and improvement, 2) widening and improving the coastal way between Suez and Hurghada cities at El-Ain Al-Sukhna through blasting the Ataqa hillsides, thereby increasing the runoff from Ataqa mountain, 3) increased coastal urbanization near reef and the high intensity of reef use by public. As a result of these activities, it was observed that most (88.06%, Table 7 & Fig. 14b) of partricles deposited in this area are very fine (60.36%) and silt-clay (<63 μm in diameter, 27.76%).

Despite the sedimentation rate at site III (11.60 ± 1.14 mg cm^{-2} day^{-1}) was greatly and significantly ($P<0.01$) lower than those found at sites II & I (Table 5 & Fig. 14a), it is considered severe and may also cause local damage to coral reefs (Loya, 1976a; Rogers, 1983; Cortes & Risk, 1985; Pastorok & Bilyard, 1985). This level of sedimentation rate was largely due to the landfilling concomitant with construction of hotels and recreational resorts for tourism development in this area. Also, the artificial lagoons and protruded constructions initiated in front of recreational resorts and hotels contributed to the sedimentation problem in this area. Hawkins & Roberts (1994) indicated that tourism development has already caused substantial damage to inshore reef near Hurghada from infilling and sedimentation. Sediment size analysis showed that 67.25% of sediment sizes deposited at site III is larger than 250 μm (Table 7 & Fig. 14b).

From Figure 14a, it is evident that the offshore coral reefs (sites IV, V & VI) which are situated farther from the sources of terrestrial sediment input, are exposed to limited sedimentation stress. In contrast, the nearshore reefs, except site VII, are subjected to heavy terrigenous sediment runoff resulted from the detrimental human activities. The higher percentage of total carbonates in the bottom sediments at sites IV, V, VII and VII (88.99 ± 1.20, 90.66 ± 1.85, 89.11 ± 0.93 and 88.32 ± 0.89% respectively, Table 5 & Fig. 17) demonstrated that these reefs were not exposed to high rates of terrigenous sediment inputs which are poor in carbonates (Beltagy *et al.*, 1986; Cortes, 1990). The very high significant negative relationship (r^2 = 0.87, $P<0.05$) established between enhanced sedimentation rate and percentage of total carbonate content in reef sediments (Fig. 18) supports the view that, increased sediment deposition and associated high turbidity would probably reduce the net production of carbonate by corals (Woolfe & Larcombe, 1999). This may explain the lowest carbonate content in reef sediments at location II, which is subjected to higher levels of terrestrial sedimentation.

The normal sedimentation rate on coral reefs which are not subjected to stresses from human activities, are generally less than 10 mg cm^{-2} day^{-1} (Rogers, 1983 and 1990; Neil, 1990; Te, 1992). Loya (1976a) showed a decline in coral diversity and cover due to an increase of sedimentation from 3 to 15 mg cm^{-2} day^{-1}. Pastorok & Bilyard (1985) have assembled a general classification of level of sedimentation impact on coral communities. They found sedimentation rates <10 mg cm^{-2} day^{-1} have slight to moderate impact, deposition of 10 to 50 mg cm^{-2} day^{-1} implies moderate to severe impact.

The high significant positive relationship (r^2 = 0.92, $P<0.05$) existed between increasing sedimentation rate and percentage cover of dead corals (Fig. 15) indicated the lethal effect of high sedimentation levels on reef corals. Heavy sedimentation can kill corals physically by smothering and burial (Rogers, 1979 and 1990; Cortes, 1990; Sammarco, 1991; Hawkins & Roberts, 1994; Woolfe & Larcombe, 1999). In a study of the response of Puerto Rican reefs to sedimentation, Rogers (1977) found that sediments can bury and kill corals, and also change substrate characteristics, inhibiting larval settlement. Öhman *et al* (1993) stated that increased sedimentation on coral reefs in Srilanka smothers the reefs and causes their degradation. Wesseling *et al* (1999) studied the damage and recovery of four Philippinine corals from short-term sediment burial. They concluded that complete burial caused considerable whole-colony mortality, and thus may result in a permanent loss of coral taxa from reefs that are subjected to such intense sedimentation events. The corals respond to intensive sedimentation by continuous secretion of mucus. This active cleaning process consumes part of energy required for skeletal growth (Bak & Elgershuizen, 1976; lasker, 1980; Abdel-Salam & Porter, 1988; Rogers, 1990).

The results of regression analysis (Fig.15) demonstrated also that increasing sedimentation rate was associated with a significant ($P<0.05$) reduction in live hard coral cover, hard coral density, species richness, species diversity and evenness index ($r^2 = 0.65, 0.79, 0.71, 0.79$ and 0.68 respectively). These results are in agreement with findings of Pastorok & Bilyard (1985); and Hubbard (1986) who found that the higher levels of sedimentation (>10 mg cm^{-2} day^{-1}) have a severe to catastrophic effect on coral communities, reducing coral diversity and cover to low levels. Our data are supported also by the following observations: Loya (1976a) reported fewer coral species and less live coral cover on a reef receiving high sedimentation rates than on a nearby reef with less sedimentation. At Cahuita reef, which is characterized by heavy terrigenous sedimentation rates, the species diversity and live coral coverage were greatly reduced (Cortes & Risk, 1985). In southern Pureto Rico, Acevedo *et al* (1989) found that live coral cover and species diversity increased away from a terrigenous sediment source. In a study of the coral reefs of Golfo Dulce, Costa Rica, Cortes (1990) demonstrated that high sediment loads around coral reefs can cause a reduction in coral species diversity and live coral cover, alteration in coral species composition and distribution, and ultimately, leading to coral death.

The possible reasons for reducing living coral cover and hard coral density with increasing sedimentation rate in the present study are probably enhancing coral mortality, reducing coral larvae settlement and depressing coral growth rate (Sheppard, 1982; Cortes & Risk, 1985; Stafford-Smith, 1993; Woesik & Done, 1997). The adverse effects of exceesive sedimentation on the settlement of coral larvae and coral recruitment have been previously investigated by several authors (Maragos *et al.*, 1985; Hodgson, 1990; Sammarco, 1980 and 1991; Babcock & Davies, 1991; Te, 1992) who stated that sediments coating hard substrates affect the composition and density of coral population by interfering with the settlement of coral larvae (by changing the substrate characteristics) and reducing early post-settlement survival. Bell & Elmetri (1995) revealed that the reduction in hard coral cover can partly be explained by a reduction in hard substrate (required for planulae settlement) which has resulted from the deposition of sediment and sand.

Previous studies (Bak & Elgershuizen, 1976; Bak, 1978; Lasker, 1980; sheppard, 1982; Abdel-Salam & Porter, 1988; Rogers, 1990; McClanahan & Obura, 1997; Ammar *et al.*, 2000) indicated that the different coral species have different capabilities of clearing themselves from sediments and surviving the condition of lower light levels. Thus, we can suggest that the reduction in species diversity, species richness and eveness index associated with increasing sedimentation rate in the present study may be explained by the greater abundance of coral species which had greater resistance to higher levels of sediment accumulation and reduced light intensity. Rogers (1990) concluded that reef with heavy sedimentation showed fewer coral species, less live hard coral cover, lower coral growth rates, reduced coral recruitment, and greater abundance of species with higher resistance to heavy sedimentation and reduced light levels. Woesik & Done (1997) pointed out that high sediment loading in the water column influences species presence or absence, growth form, growth rates, and survival of established corals.

Moreover, our results suggested that higher levels of sedimentation rate might lead to a decline in the population density of zooxanthellae (Rogers, 1979; Cortes & Risk, 1985; Riegl, 1995; Woolfe & Larcombe, 1999). This view is confirmed by the strong inverse significant relationship (r^2 = 0.62, P<0.05) established between increasing sedimentation rates and population density of zooxanthellae (Fig. 16). Neil (1990) indicated that loss of symbiotic zooxanthellae resulted from enhanced sedimentation rates (>10 mg cm^{-2} day^{-1}). Grigg & Dollar (1990) reported that prolonged exposure to siltation (sediment loading) and high turbidity resulted in loss of zooxanthellae and abnormal mucus secretion.

Oil pollution:

Oil pollution is one of the main potential causes of coral reef demise along the Egyptian Red Sea coast (UNEP/IUCN, 1988). The oil pollution in the area of the present study is a normal and direct consequence of the rapid expansion of petroleum activities (including exploration, production and transportation) in the Gulf of Suez and the northern part of the Red Sea proper. The higher mean levels of petroleum hydrocarbons (34.83 ± 26.57 µg L^{-1} and 52.85 ± 8.33 µg g^{-1} for seawater and sediments respectively, Table 5 and Figs. 22 & 25) recorded at site II demonstrated a serious oil pollution problem existed in this region. The possible chronic oil pollution sources in this region include: 1). Inevitable oil spills resulted from SUMED pipeline terminals at El-Ain Al-Sukhna during loading and unloading operations; 2). Discharge of oil contaminated ballast and/or bilge water from tankers waiting in the offshore transit area in the vicinity of El-Ain Al-Sukhna. The oil pollution from these two sources reaches to site II by the predominantly southerly currents induced by the prevailing northerly winds (Mancy, 1983; Awad *et al.*, 1983; Edwards, 1987). Furthermore, this area may be affected by the oil pollution derived from oil fields centered at Ras Gharib and Ras Sukhier due to the occasionally reversed currents caused by the sudden strong southerly winds, which is called "Aziab" (Edwards, 1987). Similarly, oil pollution in the vicinity of SUMED pipeline company terminals including both floating and land-based receiving terminals, and discharges of dirty ballast water from tankers in the offshore transit area are probably the very important causes of high oil pollution levels (20.06

± 8.54 μg L^{-1} and 21.79 ± 11.07 μg g^{-1} for seawater and sediments respectively) measured at site I (El-Ain Al-Sukhna). In addition, sites I & II were greatly affected by the offshore accidental oil spills, mainly due to tanker collision and the accidental oil spills due to pipeline damage (Mancy, 1983; Said, 1996). In the last years, these two areas were exposed to repeated accidental crude oil spillage from tankers and pipeline damage (Said, 1996). More than twenty-one accidental oil spills caused by tankers, shipping and pipeline ruptures occurred in these areas from 1995-1999 (personal commnication with environmental protection officials in Suez governrate). For example: during may 1995 an accidental crude oil spill of 7 km in length and 3 km in width occurred at El-Ain Al-Sukhna and spread to Fanar Abu El-Darag (79 km south of Suez city) and Ras Za'farana. This spill resulted from the ships working in this region and the routine petroleum operations (personal communication with Nasr Petroleum Company). In September 1996, a large amount of crude oil spilled into the sea from ruptured loading and unloading oil piplines at El-Ain Al-Sukhna. In January 1998, a large oil spill took place during decreasing the load of Liberian tanker "Temryuk" by the help of Egyptian tankers "Karnak1 and Karnak2". In the same month 1998, a crude oil spill occurred in the area of El-Ain Al-Sukhna due to the collision between the Egyptian tanker "Gehad" and the platform of SUMED company. In March 1999, there also occurred an oil spill of 12 km length and 2 km width from the Moraco tanker "Al-Faraby". The third highest mean levels (13.20 ± 3.08 μg L^{-1} and 17.60 ± 10.65 μg g^{-1} for seawater and sediments respectively) of oil pollution recorded at site VII originate mainly from various types of ships travelling freely (regularly) between Safaga and Quseir ports and oil coming from the north oilfields with currents travelling southward (Hanna, 1983; UNEP/IUCN, 1988). Also, a number of oil exploration activities are performed near Safaga constituting additional source of oil pollution in this area. The low levels of petroleum hydrocarbons (Table 5 and Figs. 22 & 25) determined at nearshore (site III) and offshore reefs (sites IV, V and VI) in the area of Hurghada indicated a limited oil pollution problem in this area. This is because the area of Hurghada is especially protected from the prevailing southerly currents, which presumably carries the oil with it, by projecting headlands (such as the projecting coastline of Ras Ghanim and Ras-Gemsa) and by the cluster of islands including Gubal and Shadwan (Hanna, 1982 and 1983; UNEP/IUCN, 1988). The limited oil pollution (Table 5) around Hurghada may be related to the localized discharge of used oil and/or oil mixed with water from fishing and tourist boats in the vicinity of coral reefs. The discharges from recreational resorts, hotels and other tourist centers may also contribute to the localized oil pollution in this area.

The present data obtained from regression analysis (Fig. 23) revealed that live hard coral cover, hard coral density, species diversity, species richness and evenness index were significantly ($P<0.05$) decreased with increasing levels of petroleum hydrocarbons (r^2 = 0.80, 0.92, 0.93, 0.89 and 0.80 respectively), while the percentage of dead coral cover was significantly increased (r^2 = 0.91). These findings are in accordance with a number of previous studies (Fishelson, 1973b; Rinkevich & Loya, 1979; Loya & Rinkevich, 1987; Guzman *et al.*, 1993; Guzman *et al.*, 1994). In a study of the Gulf of Aqaba reefs, Loya (1975) found that number of coral species, percent coral cover and species diversity all decreased after stress from oil pollution. At the south west of Aruba Island, southern Caribbean Sea, the spatial structure of the reefs, which are present in front and downcurrent of a large oil refinery have deteriorated. Polluted reefs there have much lower cover of live hard corals and lower numbers of coral species than do unpolluted reefs (Bak, 1987). Jackson *et al.* (1989) reported that oiling can lead to the increased incidence of mortality and continued partial mortality of coral colonies in Panamanian coastal communities. Guzman *et al* (1991) studied the harmful effects of a major oil spill, which has occurred in the Caribbean coast of Panama in 1986, on reef corals. They found a significant reduction in numbers of corals, total living coral cover and species diversity with increased levels of oil. Also, frequency and size of recent injuries on massive corals increased with high levels of oil.

Oil incorporated into reef sediments pose a serious and persistent threat to the reef corals, because these sediments have remained as reservoir of oil and a source of chronic reoiling of the reef. Long time must be required to flush this persistent oil pollution from fine sediments (Rogers, 1990; Corredor *et al.*, 1990; Guzman *et al.*, 1991; Guzman & Holst, 1993; Guzman *et al.*, 1994). Our results showed that the increasing concentration of petroleum hydrocarbons in reef sediments was associated with a significant ($P<0.05$) reduction in living hard coral cover, hard coral density, species diversity, species richness and evenness index ($r^2 = 0.78$, 0.85, 0.91, 0.92 and 0.76 respectively, Fig. 26). On the other hand, dead coral cover was significantly increased ($r^2 = 0.90$, $P<0.05$) with increasing concentration of petroleum hydrocarbons in reef sediments. These findings indicate the negative impacts of increasing oil levels in reef sediments at the studied locations. Five years after a major oil spill in Panama, Guzman & Holst (1993) found that extensive reef areas are chronically threatened by oil and large amounts of sediments containing toxic hydrocarbons. They also demonstrated that the increased number of injuries on reef building coral *Siderastrea siderea* and associated reduction in colony size can ultimately reduce the population survival. Guzman *et al* (1994) studied the effect of oil pollution on injury of Caribbean reef corals in Panama. They found that corals exhibited much higher levels of injury on heavily oiled reefs, and concentration of hydrocarbons in reef sediments was significantly positively correlated with amounts of coral injury. They suggested that the probable cause of persistently high levels of coral injury is chronic exposure to sediments mixed with oil.

From the foregoing, we could suggest that the significant reduction in live hard coral cover and hard coral density with high levels of oil pollution in the present study may be attributed to enhanced coral mortality and injury in response to increased oil pollution.

Several authors (Bak & Elgershuizen, 1976; Brown & Howard, 1985; Bak, 1978 and 1987; Guzman *et al.*, 1991; Guzman *et al.*, 1993) have been previously indicated that the oil pollution affected the various coral species very differently, where the species showing a greater tolerance to extensive oil pollution will dominate the impacted reefs at the expense of the other species, which had a lower resistance to oil pollution. Therefore, it is clear that the possible explanation for decreased species diversity, species richness and evenness index with increased oil pollution in the present study is the different abilities of coral species to tolerate high levels of oil pollution.

Because of the close relationships between reef corals and zooxanthellae, which is essential for photosynthesis, any detrimental effect on these endosymbiotic dinoflagellates due to high levels of oil pollution could cause indirect damage to corals (O'Brien & Dixon, 1976; Rinkevich & Loya, 1983a; Loya & Rinkevich, 1987). From the high significant ($P<0.05$) inverse relationship that existed between population density of zooxanthellae and increasing concentrations of petroleum hydrocarbons in seawater and reef sediments (r^2 = 0.78 and 0.83 respectively, Figs. 24 & 27), it appears that the intensive oil pollution may also damage and kill the zooxanthellae resulting in a decrease in the number of these symbiotic algal cells (normally found in the coral gastrodermis) which are necessary to maintain the coral in good condition and to increase coral calcification. This is consistent with Peters *et al* (1981), who stated that the pathological responses of corals to petroleum hydrocarbons included degradation and loss of symbiotic zooxanthellae. Also, Fucik *et al.* (1984) found that the effects of oil on individual coral colonies range from tissue death to loss of the symbiotic algae, zooxanthellae (bleaching). Thus, the reduction in density of zooxanthellae due to increased oil pollution may lead to a decrease in photosynthesis in corals (O'Brien & Dixon, 1976; Cook & Knap, 1983; Loya & Rinkevich, 1987). Rinkevich & Loya (1983a) indicated a marked decrease in zooxanthellae photosynthesis of reef coral *Stylophora pistillata* in response to higher concentrations of petroleum hydrocarbons.

Heavy metals pollution:

Persistent pollutants such as heavy metals can remain in marine environment unchanged for years and may thus pose a threat to marine organisms and man. All the problems associated with heavy metal pollution will increase considerably in the coming years if measures for control and management are not created. Almost all heavy metals reported in this study (Tables 8a & 8b and Figs. 28 & 30) are normal components of oil, sewage and wastewater, terrigenous sediments, and antifouling and anticorrosive paints which are used for the protection of ship hulls. Hence, high concentrations of heavy metals might be used as indicators for the different sorts of pollution (Matson, 1989; Guzman & Jimenez, 1992; Machiwa, 1992). Petroleum-related heavy metals reported in the present study were Pb, Ni, Cr and Co (Hamed, 1996; Metwally *et al.*, 1997). Urban sewage and wastewater disposal have been accompanied by an increase in the levels of Zn, Cu, Pb, Cr, Cd and other metals (Abel-Salam, 1989; Al-Abdali *et al.*, 1996; Shriadah, 1998). Copper and cupric oxides are used as antifouling agents in marine paints for protection of ship hulls from the attachment and subsequent growth of marine organisms (Al-Abdali *et al.*, 1996). This paint releases most of its copper to the sea. The high concentrations of Fe and Mn reported here are indicators of terrigenous sediment contamination (Matson, 1989; Guzman & Jimenez, 1992; Machiwa, 1992). The accidental spills of crude oil or fuel oil contaminated with Fe from tankers would act as an additional source of Fe pollution in the area of study (Al-Abdali *et al.*, 1996).

The high significant positive correlations found between increasing trace metal concentrations and increasing total organic matter content in reef sediment (Fig. 21), indicating that complexation of these metals with organic matter plays an important role in their distribution patterns (Barsdate, 1970; Bell, 1992; Shriadah, 1998). Therefore, organic matter is considered as a significant concentrator of most trace metals. Particulate suspended matter (SPM) showed high positive and significant ($P<0.05$) correlation with concentrations of the following trace metals in reef sediments: Cu (r^2 = 0.71), Zn (r^2 = 0.69), Pb (r^2 = 0.67), Cd (r^2 = 0.66), Cr (r^2 = 0.71), Ni (r^2 = 0.67), Co (r^2 = 0.66), Fe (r^2 = 0.73) and Mn (r^2 = 0.71). Also, SPM showed high positive and significant ($P<0.05$) correlation with concentrations of dissolved Cu, Zn, Fe and Mn in seawater (r^2 = 0.69, 0.65, 0.71 and 0.66 respectively). These findings reveal that SPM may act as a significant source of heavy metal pollution in the area of study, whereas the major portion of metals input enters the marine systems in the form of metal-rich particulate matter (Al-Abdali *et al.*, 1996; Shriadah, 1998).

The highest levels of heavy metals measured at site I (Tables 8a and 8b; Figs. 28 and 30) were probably as a result of high levels of oil pollution, high terrestrial sediment input and sewage disposal from the recreational resorts, and the highest levels of SPM and total organic matter in this area. Also, the vessels waiting in the transit area at El-Ain Al-Sukhna (site I) painted with antifouling and anticorrosive coatings could be another source of Cu pollution in the area (Hamed & Mohamed, 1999). The limited water currents and the slow rate of water exchange with the coastal waters tend to enhance conditions suitable for the accumulation of organic matter (Fig. 20) in this area (Shriadah, 1998).

Similarly, the elevated heavy metal concentrations found at site II (Ras Za'farana) may also be due to the high levels of oil pollution, enhanced eutrophication and SPM, and excessive terrigenous sediment runoff estimated in this region. The high levels of iron and manganese on the northern part of Gulf of Suez may also be attributed to Fe ores, which is dominant in carbonate rocks of Northern Galala flanking at the western side of the Gulf of Suez.

The major anthropogenic sources accounted for the third highest concentrations of trace metals recorded at site III (Marine Biological Station) were sewage and wastewater discharge, high levels of SPM, terrestrial sediment inputs and antifouling paints coating the fishing and tourist boats. The localized oil pollution in this area acts as an additional source of trace metals. The higher concentrations of heavy metals in the northern part of the area under study than those in the southern part (Figs. 28 & 30) reflected a serious pollution problem in the northern part as a result of increased unplanned detrimental human activities (Hamed & Mohamed, 1999).

The effects of heavy metal pollution on coral reef communities are still unclear (Brown, 1987; Guzman & Jimenez, 1992) and not isolated from the possible effects of other stressors (Scott, 1990; Peters *et al.*, 1997). The present study is focusing on the effect of heavy metals on coral reef population. Results of the present study indicated, therefore, that increased heavy metal pollution was accompanied by a reduction of hard coral density, species diversity and evenness index (Figs. 29, 31, 32, 33 and 34). Corals in a polluted estuary in Hong Kong also showed clear signs of stress resulting from increasing exposure to heavy metals, nutrients, sewage and turbidity, where the growth rates, species abundance, diversity and cover were significantly declined (Scott, 1990).

In addition, the percentage of dead coral cover was positively correlated with concentrations of the following heavy metals in reef sediments: Cu, Zn, Pb, Cd, Cr and Ni (r^2 = 0.36, 0.36, 0.41, 0.38, 0.49 and 0.38 respectively, $P>0.05$). On the other hand, the percentage of dead coral cover was also positively correlated with concentrations of dissolved Cu and Fe in seawater (r^2 = 0.47 and 0.41 respectively, $P>0.05$). Despite these relationships are not significant, It can be suggested that high levels of heavy metals may partially be contributed to the coral death. This observations is substantiated by the results of Brown & Holley (1982) who found that the dead coral cover on the reef in the vicinity of a tin smelter in Thailand was high, although it was not significantly different from values observed on reefs several kilometers away from the smelter, which were not apparently under the influence of increased metal loads associated with tin smelting and dredging including copper, zinc, iron and tin.

Abdel-Salam (1989) experimentally studied the effects of different concentrations of copper sulphate on two species of Atlantic reef corals, namely *Montastrea annularis* and *Diploria strigosa*. He found that high concentrations (1.0 mg L^{-1}) of copper sulphate killed the two species, though *Diploria strigosa* showed greater resistance to copper sulphate than *Montastrea annularis*.

Heyward (1988) has investigated the effect of heavy metal pollution on coral fertilization. He exposed gametes from colonies of three coral species *Goniastrea aspera*, *Favites chinensis* and *Platygyra ryukyuensis* to different concentrations of copper and zinc sulphates. He found that higher concentrations of copper and zinc sulphates led to complete inhibition of fertilization for all three species.

Consequently, it is clear that the reduction of hard coral density with high concentrations of heavy metal may be attributed mainly to inhibition of coral fertilization. Moreover, the decrease in species diversity and evenness index with increased heavy metal concentrations may be related to the different abilities of various coral species to tolerate the heavy metals contamination, whereas the polluted reefs can be dominated by the more resistant species (Brown & Holley, 1982; Abdel-Salam, 1989; Peters *et al.*, 1997).

Variation of biological parameters between the studied reef sites: Responses to human impacts.

Coral cover, hard coral density, species diversity and evenness index:

The present analysis showed a substantial difference in coral community's structure among various surveyed reefs. It is difficult to isolate and identify the specific cause leading to a particular coral community structure and degradation, and to attribute this to either a natural or a human influence. In attempting to develop a unified theory about natural and anthropogenic disturbance on coral reefs, Grigg & Dollar (1990) reported that there is little qualitative difference between anthropogenic and natural stress to coral reefs, and that both sources of disturbance are important in controlling reef community structure. However, our findings indicate that human activities are the major factors behind changes in coral reef structure in the area of study (Riegl & Velimirov, 1991; Hawkins & Roberts, 1994; Rinkevich, 1995). Human impacts are more apparent on the coastal habitats and would be the stronger influence on the nearshore coral reef communities (Chou & Yamazato, 1990; Ali, 1993). In most of surveyed locations, the existing reef structure and deterioration have resulted from a combination of different factors. Coral community structure and reef deterioration at sites I, II & III are the best examples of the multiplicative effects of several combined anthropogenic stresses on a coral community structure. The lowest percentage of live hard coral cover, hard coral density, species richness, species diversity and evenness index, and the highest percentage of dead coral cover reported at site II (Table 10) could be attributed to the combined effects of different anthropogenic impacts. The most common and serious of these influences were heavy sedimentation rate, high levels of oil pollution, sewage discharge and high levels of SPM in this area (Table 5). Similarly, site I exhibited the second lowest percentage of live hard coral cover, hard coral density, species richness (or number of species), species diversity and evenness index, and the second highest percentage of dead coral cover (Table 10). This may also be attributed to the synergistic effects of high sedimentation rate, high levels of oil pollution and sewage discharge, in addition to the highest levels of heavy metals, SPM and TOM measured in this area (Tables 3, 5, 8a & 8b). The synergistic effects of combined anthropogenic disturbances on coral reef structure have been previously reported in the following studies: Maragos (1972) estimated that 80% of the coral communities in the lagoon in Kaneohe Bay, Hawaii was died because of a combination of dredging, increased sedimentation and swage discharge. Loya (1975) suggested that synergistic effect of phosphate eutrophication induced algal growth on the one hand, and chronic oil pollution on the other hand are the major man-made disturbances that preventing coral colonization at Eilat, Red Sea. Walker & Ormond (1982) suggested that increased algal growth stimulated by increased nutrient concentrations due to sewage disposal might be important in greatly increasing the sediment load experienced by corals. Maragos *et al* (1985) reported that the sewage pollution had delayed or prevented recolonization of corals on heavily sedimented surfaces by stimulating growth of algae, which competed with corals for substrate space. Tomascik & Sander (1985) concluded that the reduced growth rates of *Montastrea annularis* in Barbados reefs, West Indies are a direct result of increased SPM concentrations (7.32 ± 2.86 mg L^{-1}) brought about by the increased eutrophication processes (PO_4-P = 0.21 ± 0.11 μM and DIN = 8.06 ± 0.96 μM). Rogers (1990) stated that increased sedimentation, chemical pollution and thermal pollution are all contributing to the demise of the Florida's reefs. Thus, it is suggested that the combined effects of enhanced sewage discharge (as indicated by highest concentrations of nutrients and chlorophyll *a*), high concentration of SPM and considerable levels of terrestrial sediment runoff found at site III (Tables 3 & 5) were probably the main causes accounted for the considerable percentage of dead coral cover, and the moderate values of species diversity and evenness index in this reef. This is also in coincidence with the findings of Kinsey (1988); Bell (1992); and Bell & Elmetri (1995). Interacting and synergistic effects of oil pollution in conjunction with increased sedimentation resulting in extensive damage to coral reefs in the area under investigation, particularly at sites II and I. This is because the increased sedimentation may complicate the problem of oil pollution, where sediments act as a reservoir for oil and a source of chronic reoiling (Rogers 1990; Guzman *et al.*, 1991). Corals subjected to high amount of oil mixed with sediments clean themselves by continuous secretion of large amounts of mucus. This cleaning process and regeneration of damaged tissues necessarily reduce the amounts of energy and materials available for other functions such as skeletal growth and reproduction (Lasker, 1980; Bak, 1983; Stearns, 1992).

Notwithstanding site VI (El-Fanadir) is an offshore reef and mostly not affected by the land-based pollution, it has reflected the third highest percentage of dead coral cover (Table 10). This may mainly be as a result of intensive diving (number of divers ranging from 160-300 per day) and fishing activities in this area, as well as the localized oil pollution resulted from diving and fishing boats. In addition, El-Fanadir reefs are used by local people for collection of reef invertebrates and reef fishes for aquariums and curio trade. Bryant *et al* (1998) indicated that destructive fishing practices and overfishing pose great threats to the integrity of coral reefs. Heavy collection of reef fishes and invertebrates adversely affected the coral reefs through disturbing the ecological balance inside the reef ecosystems (Rogers, 1988b; Sudara & Nateekarnchanalap, 1988; Hawkins & Roberts, 1994).

The relatively low species richness and percentage of live hard coral cover, in addition to a considerable percentage of dead coral cover and moderate species diversity had been reported at site VII (Table 10) are probably attributed to the relatively high levels of oil pollution existed in this area (Table 5 & Figs. 22 and 25). This is consistent with the findings of Fishelson (1973b); Loya (1975); Rinkevich & Loya (1979); Loya & Rinkevich (1980); Bak (1987); and Guzman *et al* (1991).

In contrast to site II (Ras Zafarana), site IV (Sha'b Saad) displayed the highest values of live hard coral cover, hard coral density, species diversity and evenness index (Table 10). This possibly because site IV appears to be comparatively free to some extent from most of deleterious anthropogenic impacts, especially heavy terrestrial sedimentation, increased oil pollution and recreational activities.

Population density of zooxanthellae:

It is suggested that high oil pollution and sedimentation rate seemed to be the major anthropogenic factors that are responsible for the lowest and second lowest densities of zooxanthellae recorded at sites II & I, respectively (Table 10 and Fig. 42). Conversely, the highest density of zooxanthellae obtained at site V may be due mainly to the lower concentrations of oil pollution and decreased sedimentation rate found at this reef (Table 5). This is in agreement with Rogers (1979, 1990); Peters *et al* (1981); Cortes & Risk (1985); and Riegl (1995). The relatively high levels of oil pollution measured at site VII may also explain the low density of zooxanthellae at this reef (O'Brien & Dixon, 1976; Peters *et al.*, 1981; Cook & Knap, 1983; Fucik *et al.*, 1984).

Coral breakage:

The physical damage to coral reefs in the present study was indicated largely by the percentage of coral breakage. The greater percentage of coral breakage reported at sites V & VI (19.23 ± 3.25 and 22.73 ± 2.76% respectively, Table 10) resulted from high intensity of recreational diving and snorkeling, anchoring and grounding of diving boats, and destructive fishing activities (Riegl & Velimirov, 1991; Hawkins & Roberts, 1992a and b; Ali, 1993). These two sites are recognized as popular diving places, heavily visited by divers and tourists using boats (number of diving boats visited each site ranging from 20-30 per day). Divers and snorkellers physically damage corals by kicking, bumping into them, kneeling or standing on them and also by resuspending bottom sediments (Rogers, 1988b; Hawkins & Roberts, 1994). This may be due to inadequate buoyancy control by divers. Hawkins & Roberts (1992b) investigated the effects of recreational SCUBA diving on the fore-reef slopes of coral reefs near Sharm

El-Sheikh, a popular resort in Egypt, Red Sea. They noticed that there were significantly more damaged coral colonies, loose fragments of live corals, fragments of corals reattached to the substratum, partially dead and abraded corals in areas heavily used by divers than in control areas. Therefore, the sites which had experienced the greatest increase in diving appeared to have accumulated physical damage (coral breakage), whereas the others did not.

The trampling by SCUBA divers and snorkellers on reef flats caused additional physical damage to coral reefs in the area of study. In previous study, Hawkins & Roberts (1993) investigated the effects of trampling by SCUBA divers and snorkellers on reef flat communities of coral reefs near Sharm El-Sheikh. They reported that there were significantly more damaged coral colonies and loose fragments of live corals in heavily-trampled than in little-trampled areas; percentage cover of bare rocks and rubbles was also significantly greater; conversely, number of hard coral colonies and total percentage of live hard coral cover were lower. Since site I is a popular summer resort in the Gulf of Suez, it is subjected to high visitor frequency by publics and some tourists. So, the high percentage of broken corals in this area (17.04%, Table 10) may be ascribed to heavy trampling by publics and high density of sea urchins. Antonius (1984) indicated that the trampling of non-swimming public on coral reef in the area of Jeddah, Red Sea causes substantial physical destruction (60-70% mortality) to corals. Rogers (1988b) showed that residents and tourists, who walk on shallow reef corals of Virgin Island do considerable damage to corals, whereas approximately 95% of the shallow reef areas has been destroyed. Diving and trampling appeared to increase coral abrasion. However, the most damaging effect of abrasion might occur some time later. Broken and abraded corals are likely to be more susceptible to invasion by pathogens, possibly increasing mortality (Peters, 1984; Rogers, 1988a; Hawkins & Roberts, 1992b, 1994).

Anchor damage from boats is also a serious problem, where many coral colonies were broken or overturned due to boat anchoring (Smith, 1988; Price, 1993; Rajasuriya *et al.*, 1995; Hawkins & Roberts, 1996). Many broken coral colonies do not remain in place but are often completely detached and dragged over the reef when an anchor is lifted by force. Some overturned coral colonies and others slide down the reef slope and come to rest in position and light conditions, where regeneration speed is slowed down or inhibited (Causey, 1990; Riegl & Velimirov, 1991; Glynn, 1994). Boat diving represents two additional risks to reefs: grounding and anchoring. The latter has been a major cause of mechanical damage to corals in many areas (Rogers, 1988a; Hawkins & Roberts, 1992a and b). The considerable percentage of coral breakage (11.30%, Table 10) recorded at site III could be as a result of intensive fishing activities of the residents and high population density (6.50 $\pm$ 1.20 m^{-2}) of sea urchins (mainly *Diadema setosum*) in this area. The resident people took this area as an anchorage zone for their fishing boats, causing intensive coral breakage (personal observation). By repeated anchoring, however, the reef crest's carbonate rock basis itself is broken, leading to the formation of loose boulders. This unstable material does not offer a good settling substrate for planulae larvae and renders coral reattachment difficult. This leads to the assumption that these damaged reef areas may remain unsuitable for resettlement for extended periods as observed on sites III, V and VI (Riegl & Velimirov, 1991; Price, 1993). Thus, reef damage through breakage is not only restricted to coral colonies, but reef rock is also affected, leading to partial reduction of substrates suitable for recolonization. Viewing coral reefs through glass-bottom boats is a popular recreational activity in the area of Hurghada.

This may also cause a considerable mechanical damage to the reef as they may break corals by running over shallow reef flats (causey, 1990; Price, 1993). In order to give a better view of corals to the visitors, the operators often stop their glass-bottom boats over shallow coral patches and stand on the reef to keep the boats in place as well as dropping anchor on live corals. In addition to causing biological damage, anchoring and trampling reduced the aesthetic appeal of the coral reefs for tourists (Hawkins & Roberts, 1992a, 1993).

In contrast to site VI, site VII (Gasus 1) had the lowest percentage of coral breakage (1.85 ± 0.90%, Table 10) because it is little-visited reef. This area is not suitable for shore divers and snorkellers because it is exposed to heavy wave action and strong currents in most of the year (Edwards, 1987). The low percentage of broken corals (3.27 ± 1.73%) found at site IV might also be owing to the fact that this area is not popular for divers and snorkellers, and hence less visited by tourist boats.

The present data indicated that exposed reefs with low visitor frequency (site VII) showed significantly fewer broken corals than less exposed reefs with higher visitor frequency (sites I, V and VI). Therefore, wave action cannot be considered the main cause of the observed breakage in the area of study. This is consistent with the observations of Riegl & Velimirov (1991); and Hawkins & Roberts (1992a). Moreover, previous research suggests that corals in exposed areas can have denser skeletons and hence may be more resistant to breakage than those in more sheltered areas (Brown *et al.*, 1985; Hawkins & Roberts, 1992b and 1993). Chamberlain (1978) and Bottjer (1980) showed that wave action could increase skeletal density in corals. In all surveyed reef sites, most of coral breakage had occurred in the upper ten meters, where most of human interference with the coral reefs takes place. Thus, it can be assumed that the coral breakage in the area of study caused by human activities.

The present results demonstrated that physical damage to corals varied with coral growth form, the fragile branching growth forms were the most vulnerable to breakage (Kay & Liddle, 1989; Hawkins & Roberts, 1992a and 1993). Thus, reefs with a high proportion of delicate branching corals are more easily physically damaged than those dominated by species with robust growth forms. The corals underwent highest amount of breakage in most of the surveyed reef sites were *Acropora* species, *Millepora dichotoma and Stylophora pistillata*, because they are fragile and delicate branching corals. However, it is interesting to note that the most abundant coral species in terms of percentage cover at heavily dived reef sites (sites V and VI) were mostly branching growth forms (Table 9 and Fig. 40). These species were *Pocillopora damicornis*, *Millepora dichotoma* and *Acropora hyacinthus* in site V, while *Acropora hyacinthus*, *Millepora dichotoma* and *Acropora nobilis* in site VI. This because the branching corals generally have high growth rates, often regenerate rapidly after fragmentation (i.e."r" strategy) and are swift colonizers of bare substrata (Endean, 1976; Loya, 1976c; Pearson, 1981; Highsmith, 1982; Rinkevich & Loya, 1983b; Ross, 1984). Hence, although a branching morphology may be disadvantageous with respect to physical damage resistance, their life history strategies ("r" strategies) are adapted to existence in disturbed habitats (Loya, 1972; Highsmith, 1982). This can mean that branching corals are able to persist and flourish in heavily dived and trampled areas (Fig. 40). Hence, the "r" strategist branching coral *Acropora pharaonis* was the most dominant and broken species in site I, which is heavily trampled area. In previous study, Riegl & Velimirov (1991) indicated that the branching coral genera, namely *Acropora*, *Millepora* and *Stylophora* were the most frequently broken corals in the area of Hurghada and the most dominant in shallow reef areas. They are relatively fast growing and hence to a certain extent can tolerate repeated breakage. It was observed that despite *Lobophyllia corymbosa* and *Galaxea fascicularis* are massive (robust) corals, they displayed the second and third highest percentages of broken corals, respectively in site III after genus *Acropora*. This may be due to that most of coral breakage in this area caused by anchoring and grounding of fishing boats, and the destructive fishing activities, rather than the activities of divers and snorkellers.

Dominant species of hard corals:

The lowest species diversity and evenness index recorded at site II (Table 10 & Fig. 36) reflected the high dominance of *Stylophora pistillata*, *Acropora pharaonis* and *Acropora forskali* in terms of percentage cover (Fig. 40). It is suggested that these branching coral species are an advantage in areas of high sedimentation (Loya, 1972; Dodge & Vains, 1977; Walker & Ormond, 1982) and their fast growth rates enable them to cover extensive areas of reef (Rogers, 1977; Highsmith, 1982). Sheppard (1985) demonstrated that branching coral forms tend to dominate in areas of high sedimentation because their shapes prevent the accumulation of sediment on the colony surface. In addition, the most dominant species *Stylophora pistillata* in site II has been regarded as an opportunistic species which colonizes relatively harsh and unpredictable environments (Loya, 1976b, c; Chou & Yamazato, 1990). Walker & Ormond (1982) showed that *Stylophora pistillata* was a successful colonizer of polluted habitats.

At site I (El-Ain Al-Sukhna), the dominance of branching coral *Acropora pharaonis* in terms of percentage cover (Fig. 40) may also be related to its tolerance to high sedimentation, and its fast growth rate and rapid regeneration of damaged parts.

Galaxea fascicularis was the dominant species at site III and occupied the second rank at site I in terms of percentage cover (Fig. 40). A possible explanation is that, this species can tolerate high levels of sedimentation and adapted to withstand low tide which is occasionally severe on the reef flats of sites I & III (Fishelson, 1973a; Ali, 1993). It is evidenced therefore that, *Galaxea fascicularis* has some cleaning mechanisms, which enables this species to withstand heavy sedimentation (Loya, 1972; Abdel-Salam, 1988). McClanahan & Obura (1997) indicated that *Galaxea* was one of the sediment-tolerant genera in Kenyan coral reef community. Furthermore, *Galaxea* spp. appeared to be more resistant to the smothering effects of rapid sponge growth and the toxic effect of some sponges such as *Terpios* spp. than others (Vine, 1986).

It is important to note that *Porites solida* and *Porites lutea* were the most dominant species in site IV in terms of percentage cover (Fig. 40). Moreover, *Porites solida* showed the second highest percentage of total live hard coral cover in sites VI & VII. This may be ascribed mainly to genus *Porites* is a successful re-colonizer (Loya, 1975), and characterized by fast growth rate, large adult size and long life history (Highsmith, 1982; Kay & Liddle, 1987; Chou & Yamazato, 1990), in addition to its hard massive growth form.

The high abundance of *Pocillopora damicornis* and *Millepora dichotoma* in terms of percentage cover in site VII (in addition to *Porites solida*) may also be attributed to their fast growth rates (Pearson, 1981; Highsmith, 1982; Ross, 1984; Hawkins & Roberts, 1992a and 1993), and their ability to tolerate the high wave action in exposed reefs (Vine, 1986; Head, 1987). Riegl & Velimirov (1994) concluded that *Millepora dichotoma* apparently has a wider ecological tolerance than anticipated.

The similarity in species composition between sites I, II and III which are heavily affected by the land-based pollution, and the higher similarity between the offshore reef sites (IV, V and VI) suggesting the greater influence of anthropogenic factors on species composition in the studied reefs (Fig. 35).

Sea urchins:

The main bio-eroders of coral skeleton in the studied reefs were sea urchins. Vine (1986) reported that scraping of coral rock surfaces by urchins is a major source of bio-erosion on the Red Sea reefs. *Echinometra mathaei* was clearly the most abundant species at sites I and II, while *Diadema setosum* was the most abundant species at the other sites. These grazing echinoids actively undermine corals causing skeletal abrasion and remove the newly settled juvenile corals or coral spat (Sammarco, 1980; Sheppard, 1982; Ormond, 1982; Wilkinson, 1996; Bryant *et al.*, 1998). The abraded or excavated skeleton may be easily invaded by pathogens (Sheppard, 1982; Hawkins & Roberts, 1992b). Glynn *et al.* (1979) reported heavy grazing on *Pocillopora* species by the echinoid, *Eucidaris*.

The high population density of sea urchins (mainly *Echinometra mathaei*) in sites I and II (10.93 ± 2.87 and 8.40 ± 1.21 urchins m^{-2} respectively,Table 10 and Fig. 37) may have partially contributed to the coral damage in these two sites by continuous scraping the coral skeleton with its short spine and destroying of coral spat. Tsuchiya & Nishihira (1986) have shown that this echinoid (*Echinometra mathaei*) to be an active grazer on coral skeletons.

The great density of sea urchins (*Diadema setosum*) in site III have caused extensive reef erosion and damaging of coral spat whilst feeding on algal films growing on the reef surfaces (Ormond, 1982; UNEP/IUCN, 1988; Hawkins & Roberts, 1994). In her review of biological destruction of reefs, Hutchings (1986) mentioned that echinoids are major reef eroders when occurring in high densities on Caribbean and some eastern pacific reefs. The long black spines of *Diadema setosum* provide an effective defense against most predators, except for several persistent trigger and puffer fishes which has been observed to attack them despite suffering numerous spine wounds (Vine, 1986).

It is evident that the high density of sea urchins at sites I, II and III appeared to be a direct response to the anthropogenic disturbances of various sorts. The most important of which are overfishing of sea urchin predators (UNEP/IUCN, 1988; Hawkins & Roberts, 1994), increased terrestrial sediment inputs due to land reclamation and coastal building activities (Chou & Yamazato, 1990), sewage discharge (Walker & Ormond, 1982), and installation of breakwaters (Chou & Yamazato, 1990).

Overfishing removes ecological controls on reefs and can lead to massive phase shifts in population from reef dominated by corals to dominance by macroalgae, soft corals or echinoids (Sudara & Nateekarnchanalap, 1988; Done, 1992; Wilkinson, 1996). In previous study, Ormond (1982) suggested that the population explosions of sea urchins *Diadema setosum*, *Echinometra mathaei* and *Heterocentrotus mamillatus* in the area of Hurghada in the early 1980s were partly due to removal of their predators: large trigger fish *Pseudobalistes fuscus* and *Balistoides viridescens*, and large puffer fish *Arothron hispidus* and *Diodon histrix,* which are popular in souvenir trade. Population explosions of sea urchins after removal of these fishes have been demonstrated on Kenyan coral reef community (McClanahan & Muthiga, 1988). In another study of Kenyan coral reefs, McClanahan & Obura (1995) indicated that a high abundance of sea urchins in unprotected reefs result from reduction in sea urchin predators largely due to overfishing. They also showed that high sea urchin populations are associated with reefs had lower coral cover, topographic complexity and reduced calcium carbonate deposition rates.

Walker and Ormond (1982) showed a higher population density of sea urchins *Diadema setosum* in the sewage area (mean 6.6 m^{-2}) than in the control area (2.1 m^{-2}). This is consistent with our findings at site III (6.50 $\pm$ 1.20 urchins m^{-2}) that is affected by sewage pollution. Heavy sedimentation caused by land reclamation and building activities has resulted in a very soft silty bottom, which is unsuitable substrate for coral larvae settlement. However, this condition favors the growth of many species of sea urchins particularly, *Echinometra mathaei*. The man-made breakwaters could have altered water current conditions resulting in a more sheltered habitat, which also favors the growth of sea urchins (Chou & Yamazato, 1990).

Soft corals:

The ecological studies dealing with soft corals are scarce (Peters *et al.*, 1997). In spite of their high abundance in Indopacific coral reef regions, soft corals constitute a minor part in general descriptions of coral reef surveys (Nishihira & Yamazato, 1974; Veron *et al.*, 1974; Dai, 1988). Soft corals and macroalgae are the major competitors to stony corals for substrate space on the surveyed reefs (Benayahu & Loya, 1981). The high significant ($P<0.05$) inverse correlation existed between soft coral cover and live hard coral at sites I, II, IV, V and VI (r^2 = 0.78, 0.64, 0.82, 0.70 and 0.65 respectively, Fig. 39) interpreted the strong competition between them for substrate space as previously reported by Fishelson (1970); Schuhmacher & Mergner (1985); Vine (1986); Dai (1988). In addition, the rapid growth of soft corals can favor them in their competition with hard corals for substrate space (Endean, 1976). Hodgson (1983) showed that alcyonacean corals (soft corals) are adventitious invaders of areas where reef building corals have been damaged. Nishihira & Yamazato (1974) have shown that a soft coral community was replacing some of Okinawan reefs affected by *Acanthaster planci* devastation. Also, the fusion of neighboring soft coral colonies reduce the substrate space available for the settlement of hard coral planulae larvae, thereby reducing hard coral recruitment (Chou & Yamazato, 1990). Thus, the deterioration of reef building corals due to the anthropogenic activities can promote the growth of soft corals at the expense of hard corals. It is suggested that increasing sedimentation and sewage pollution are the significant anthropogenic factors that favor the soft corals in their competition with hard corals (Smith *et al.*, 1981; Bell, 1992). According to Sheppard *et al* (1992), soft corals cover was declined significantly in the southern part of the investigated area.

The higher percentages of soft coral cover (Table 10 and Fig. 38) at El-Ain Al-Sukhna (site I) and Ras Za'farana (site II) may partly be attributed to high sedimentation rates and sewage discharge in these two areas. The possible explanation is that the soft corals showed a greater resistance to increased sedimentation than reef building corals which are more sensitive to increased sedimentation and changes in environmental conditions (Benayahu, 1985). Bell & Elmetri (1995) reported that many of the nearshore hard corals at the Great Barrier Reef lagoon have been displaced by other benthos such as algae and soft corals as a result of high deposition of sediments, which reduces the hard substrate available for settlement of hard coral larvae. McClanahan & Obura (1997) found that percentage covers of soft corals and sponge were higher at increasing levels of sediment influence in Kenyan reef community. In addition, higher levels of organic matter (12.15 ± 1.29%) recorded at site I may also be contributed to the highest percentage of soft coral cover (18.51 ± 3.35%) in this area. Wilkinson (1996) indicated that organic matter, particularly from sewage enhances the growth of filter-feeding animals: ascidians, sponges and soft corals, which either outcompete corals or result in increased bioerosion.

It has been noted that although site V (Abu-Galawa) is an offshore reef that is mostly not affected by high terrestrial sediment runoff and sewage discharge from urban Hurghada, it had a considerable percentage of soft corals (8.82 ± 2.32%, Table 10 and Fig. 38). This may be due mainly to the high proportion of broken corals caused by diving and snorkeling activities in this popular diving spot was replaced by soft corals.

Coral diseases:

Coral pathology is introduced as an approach to the diagnosis of coral reef health. Four different coral diseases, namely Tissue Bleaching or Coral Bleaching (TBL), White Band Disease (WBD), Black Band Disease (BBD) and Bacterial Infection (BIN) were investigated in the area of present study (Table 13). No infections appeared to be responsible for the first two diseases, which seemed to be purely physiological response to the environmental circumstances (Antonius, 1981a, 1995a; Peters, 1993; Antonius & Riegl, 1997). In contrast, the latter two are dependent on the presence of distinct pathogens (Mitchell & Chet, 1975; Antonius, 1981b, 1985 and 1988a; Rutzler *et al.*, 1983; Peters, 1997; Santavy & Peters, 1997). All diseases and pathologic syndromes or reactions dealt with here occur but rarely under normal conditions and they do not seem to pose a threat to healthy reefs (Antonius, 1995a). In addition to their remarkable destructive effects on coral reefs in the present study, the anthropogenic disturbances such as increased sedimentation, eutrophication, oil pollution, thermal pollution and the direct physical damage may also considerably enhance the incidence of natural diseases (Antonius, 1981a, 1988a, 1995a; Peters, 1984 and 1993). Thus, there can be no doubt that the greatest threats to coral reefs in the area of study are the synergistic effects of man-made stresses and natural diseases. It is suggested that exposure of coral reefs to adverse anthropogenic activities can alter the resistance of some coral species which are found to be resistant to diseases under normal conditions, rendering them more susceptible to coral diseases (Antonius, 1988a; Peters & McCarty, 1996; Santavy & Peters, 1997).

Coral bleaching events appear to have increased significantly in the past two decades (Glynn, 1993). Antonius (1988a) reported the Tissue Bleaching (TBL) in the eastern Red Sea and attributed this phenomenon to the unfavorable natural conditions and seawater pollution. No pathogen is involved in the progress of TBL, but it is a physiological response to the environmental stresses (Antonius, 1995a; Glynn, 1996). TBL is not necessarily lethal and corals are usually able to recover their symbiont algae (zooxanthellae) once environmental conditions return back to normal state (Brown & Suharsono, 1990; Peters *et al.*, 1993). On the other hand, severe and prolonged bleaching can cause partial to total colony death, resulting in diminished reef growth, transformation of reef building coral communities to alternate non-reef building coral communities and ultimately the disappearance of reef structures (Glynn, 1996). Some evidence pointed out to elevated seawater temperature associated with the global climate change as the main cause of coral bleaching (Glynn, 1983, 1984; Williams *et al.*, 1987; Brown & Suharsono, 1990). The general rise in the atmospheric CO2 concentration and the increase in ultraviolet radiation (UVR) are suspected to be the most important factors of global temperature rising. On the other side, Antonius (1995a) indicated that seawater contamination acts as further possible culprit of coral bleaching, while Riegl (1995) showed that increased sedimentation resulted in TBL. The present results suggested that enhanced TBL in the area of study (Table 13) might be due to the interaction between anthropogenic factors (particularly, high sedimentation rates and chemical pollution) and natural factors such as elevated seawater temperature (seawater temperature in the area of study during summer ranged from 28.75-30°C). This is coincided with Bryant *et al* (1998), who stated that coral bleaching is a frequent symptom of pollution-induced stress, as well as a response to natural factors such as changes in water temperature. Consequently, high abundance of TBL at site I (El-Ain Al-Sukhna) and its frequent occurrence at site II (Ras Zafarana) were probably ascribed to high levels of sedimentation, oil and heavy metal pollution, and increased seawater temperature during summer season. These factors may act jointly to enhance the severity of TBL (Wilkinson, 1996). In the last years, world's oceans have warmed about 0.5°C over normal summer seawater temperatures (Reid, 1991) and are expected to increase by 1-2°C more, in the next few years (Hoegh-Guldberg, 1999). Brown & Howad (1985); and Williams & Bunkley-Williams (1990) attributed the enhanced coral bleaching to exposure of reef corals to high turbidity and sedimentation resulting in reduced light levels, chemical pollution and temperature extremes. The frequent occurrence of TBL recorded at site III (Marine Biological Station) may be related to the relatively high sedimentation and high turbidity in this area, in addition to elevated seawater temperature during summer season. It was observed that the most susceptible corals to TBL in most of the sampled reefs were species of genera *Favites, Acropora* and *Platygyra*. The nature and extent of bleaching vary between individuals and among species at the same location during a bleaching event and have been attributed to different physiological tolerances of coral hosts and strains of zooxanthellae (Peters, 1997).

The White Band Disease (WBD) is still the most mysterious of all coral diseases because no specific agent is known to be responsible for the disease (Antonius, 1981b; Santavy & Peters, 1997). WBD is neither infectious nor contagious and can not be transmitted from afflicted to healthy corals by any means (Antonius, 1981a, b, 1985). The progress of this disease is not influenced by anthropogenic seawater eutrophication or sunlight (Antonius, 1981b, 1985). On the other hand, severity and progress of WBD may depend on variables such as species specific resistance, water temperature and pollution (Antonius, 1981b, 1988a). Thus, WBD seems to be a physiological response of the coral tissue to certain traumata, seawater temperature and pollution (Antonius, 1981a; Peters, 1984). High levels of oil and heavy metal pollution found at site I, as well as the activities of reef visitors and increased seawater temperature (30°C during summer) were probably the main causes accounted for the frequent occurrence of WBD in this area (Table 13). Also, the intermediate occurrence of WBD at Ras Za'farana (site II) were probably ascribed to high levels of oil and heavy metal pollution in addition to increased seawater temperature during summer (30°C). The increased seawater temperature during summer season over normal summer seawater temperatures in the last years (Reid, 1991) may be implicated in widespread occurrence of WBD in the surveyed locations. There are some evidence that revealed the importance of increased seawater temperature and algal overgrowth in triggering of WBD (Antonius, 1981a, b, 1985a and 1995a; Peters, 1993). Therefore, the frequent occurrence of WBD existed at site III (Marine Biological Station) may be triggered by enhancement of algal cover, together with increased seawater temperature during summer (29.20°C) in this area.

Antonius (1988a) found that frequency of WBD in the eastern Red Sea was generally higher in summer and lower in winter, and the occurring algal overgrowth appeared to be triggering WBD. The intermediate occurrence of WBD at Sh'b Abu-Galawa (site V) and Sh'b El-Fanadir (site VII) may be related to the recreational and fishing activities in these areas. Oil pollution problem and increased seawater temperature (30°C during summer) were possibly the main reasons accounted for intermediate occurrence of WBD at site VII (Gasus1). It was observed that WBD occurred widely throughout the order of Scleractinia, but did not afflict all species with the same severity. The effects of WBD were most severe among family Favidae and family Acroporidae, especially genus *Acropora*. Among family Favidae, species of *Goniastrea*, *Platygyra*, *Favites* and *Echinopora* were most conspicuous for their susceptibility to WBD. This is in agreement with the findings of Antonius (1985); and Coles (1994) in the Red Sea and Pacific Ocean.

The Black Band Disease (BBD) may be the best known of the coral diseases and the main pathogen is the Cyanophyte alga *Phormidium corallyticum* (Rutzler *et al.*, 1983; Antonius, 1985, 1988b; Bell, 1992; Santavy & Peters, 1997). BBD was previously proved to be highly infectious and contagious, and could be transferred among the susceptible species from afflicted to healthy ones by various means (Antonius, 1981b, 1985 and 1988b; Peters, 1993). As previous studies (Antonius, 1981a, 1985, 1995a; Littler & Littler, 1996) pointed out, BBD respond positively to nutrient concentrations (N & P). Thus, high nutrient concentrations (caused by enhanced urban sewage effluent from Hurghada city and recreational resorts) measured at Marine Biological Station (site III) may explain the frequent occurrence of BBD in this area (Table 13). Antonius (1988a) attributed the high occurrence of BBD in the area of Jeddah, Red Sea, to the high eutrophication resulted from sewage outfall. The present data indicated that susceptibility to BBD seems to be restricted to family Faviidae, with *Platygyra*, *Goniastrea* and *Hydnophora* species were the most heavily afflicted corals. This is consistent with previous findings both in the Caribbean Sea (Antonius, 1981b) and the Indo-Pacific region (Antonius, 1985; Coles, 1994). On the other hand, despite Ras Za'farana (site II) showed the second highest concentrations of dissolved nutrients (Table 3), the occurrence of BBD in this reef was rare. This may be primarily due to the fact that the dominant corals in this area were genus *Acropora* and *Stylophora pistillata* which are highly resistant to BBD (Antonius, 1981a, 1985; Rutzler *et al.*, 1983; Peters, 1997). This also is confirmed by the observation of Antonius (1981b) who found that *Acropora* species were not only never found with BBD in the field, but also experimentally found to be totally immune to BBD infections. It is interesting to note that on the coral species susceptible to BBD, the infection with BBD was probably made by a preceding WBD. Therefore, WBD seems to be a precondition for the establishment of BBD. Thus, it is suggested that the existing WBD among the species susceptible to BBD may be considered as pathological condition that exposes these species long enough to successful penetration by *Phormidium corrallyticum* (Antonius, 1981a, b, 1985; Santavy & Peters, 1997). The occurrence of BBD at sites I, IV, V and VII (rare, 1-3 cases per scan; Table 13) is considered in natural range (Peters, 1993; Antonius, 1995a).

Bacterial Infections (BIN) is dependent on the coral's mucus production to exert their impact (Antonius, 1981a; Peters, 1997). The main defense of coral against all kinds of deleterious external influences is the production of mucus. It forms a protective layer around a colony, shielding the sensitive tissues against the outside detrimental influences. But sometimes it can lead to an unwanted effect; since mucus being a glycopeptide, it attracts bacteria and can be used by a wide variety of marine bacteria as a source of carbon and nitrogen (Ducklow & Mitchell, 1979; Antonius, 1981a; Peters, 1993). The resulting tremendously high rate of microbial activity soon reduces the level of dissolved oxygen at the coral surface to zero, causing coral death within few days (Mitchell & Chet, 1975; Antonius, 1995a). Consequently, any kind of pollution or other man-made disturbances in the studied areas will certainly increase the incidence of BIN. According to the nature of this influence, the amount of mucus produced varies considerably. A much stronger mucus production can be observed in response to sedimentation (Bak, 1978; Lasker, 1980; Abdel-Salam & Porter, 1988). Here, mucus is used to trap sediment because it seems that sediment-saturated sheets of mucus can be removed by cilia much easier than sediment alone (Antonius, 1981a). Thus, the enhanced sedimentation rate, increased oil and heavy metal pollution, and high suspended particulate matter concentrations were probably the principal factors that are responsible for the frequent occurrence of BIN at Ras Za'farana (site II) and the intermediate occurrence of BIN at El-Ain Al-Sukhna (site I). The frequent occurrence of BIN at Marine Biological Station (site III) may be as a result of increased eutrophication and relatively high sediment loading in this region. The intermediate occurrence of BIN reported at Sha'b El-Fanadir (site VI) may be attributed to the stresses caused by diving, snorkelling and fishing activities observed in this area.