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Sedimentation Threats to Red Sea Corals: An Ecological Study of Reefs in the Hurghada Region, Egypt

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by

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Abbreviations

ABH	Adaptive Bleaching Hypothesis
CBD	Convention on Biological Diversity
CCA	Crustose coralline algae
DOS	Dissolved organic substances
EEAA	Egyptian Environmental Affairs Agency
EEPP	Egyptian Environmental Policy Program
GBO	Global Biodiversity Outlook
GEOHAB	Global Ecology and Oceanography of Harmful Algal Blooms
IPCC	Intergovernmental Panel on Climate Change
SPM	Suspended Particulate matter
SSTs	Sea surface temperatures
TON	Total organic nitrogen
UNFCCC	UN Framework Convention on Climate Change

ABSTRACT

Hurghada coastal reefs have encountered a huge modification in the last three decades, associated with an increase in sediment input from coastal development. The aim of this study was to examine the impact of sediment on reef health at a number of sites along the coast of Hurghada encountering differing sedimentation loads. A range of physico-chemical parameters were measured in the field and related to a variety of reef health indicators. This was supported by laboratory based experiments examining the direct impact of sediment on coral bleaching and mucus production.

Annual and seasonal patterns of sedimentation were investigated along the Hurghada coastal area of the Red Sea using sediment traps. Suspended particulate matter (SPM) and the percentage of non-carbonate sediment in bottom sediment were sampled. Physico-chemical parameters measured in the field included temperature, salinity, depth, pH, specific conductivity SPC, dissolved oxygen DO, total dissolved salts TDS, percentage of dissolved oxygen DO% and turbidity using a multprobe. Inorganic phosphates, silicate, ammonia, nitrite and nitrate concentrations were determined.

Reefs were surveyed to determine a number of reef health indicators including coral cover, percentage of live and dead coral, new recruits, abundance, species richness, the percentage of r-strategist, Diversity Index, Deterioration Index (DI) and disturbance. Three species of Acropora were transplanted in this study and their survival was investigated. Six fish families were surveyed using belt transects to examine changes in community structure.

Zooxanthellae density was measured in transplanted corals in field and laboratory conditions to determine the effect of sedimentation on coral bleaching. In addition, direct feeding experiments were performed using fluorescein-isothiocyanate sediment to assess coral ability to cope with higher sedimentation. Mucous secretion by corals was measured in field and laboratory to test variations under different sedimentation condition.

Significant differences between sites in sedimentation rate, SPM and the percentage of non-carbonate sediment were observed. Sedimentation and SPM were also shown to reduce coral cover, species richness, diversity, mean colony size of branched corals and the abundance of algal feeding fish. Other parameters such as non-carbonate sediment, turbidity and percentage of mud in bottom sediment affected corals and fish to various degrees.

Sedimentation did not appear to reduce the number of live or dead corals or new recruits. In addition, it did not affect the distribution of r-strategist as a pioneer group of corals. It did not affect transplant survival or macroborer distribution, although it did reduce zooxanthellae density and increase mucus secretion and sediment uptake by *Lobophyllia hemprichii.* Coral abundance, mean colony size of massive corals and coral feeder abundance were not reduced under the observed sedimentation conditions. The Deterioration Index did not provide a strong tool to gauge coral condition in this study.

Although many indicators did not show significant correlations with sedimentation, SPM, turbidity or non carbonate sediment, it was found that sites with the highest readings of these parameters has the lowest biological quality. These finding support early studies that showed that low sedimentation levels do not have significant impact on coral health. Some sites showed continuous degradation and increased level of sedimentation from land

sources and need urgent mitigation measures to be followed by coral restoration and transplantation.

Arabic summary الملخص العربى

يتمتع البحر الاحمر بمنظومة بيئية فريدة من الشعاب المرجانية تميزة عن باقى بحار العالم. هذه الميزة النسبية تجعلة محل جذب للسياحة العالمية و ما يترتب عليها من تزايد التنمية الساحلية التى تنمو بصورة مطردة على طول الساحل المصرى للبحرالاحمر .تعتبر منطقة الغردقة واحدة من اعلى مناطق العالم من حيث معدل التنمية الساحلية و ما يترتب عليها من تدهور للبيئة البحرية المجاورة. لقد تعرض ساحل منطقة الغردقة بالبحر الاحمر الى تغير هائل على مدار العقود الثلاثة الاخيرة بالتزامن مع عمليات البناء و الحفر و الردم التى تمت خلال تلك الفترة. و قد تزامن مع هذا التغيير زيادة كبيرة فى الرواسب التى دخلت الى المياة الساحلية و ذلك من الشاطئى كنتيجة لعمليات الحفر و الردم.

تناقش هذة الدراسة تأثير التنمية الساحلية و ما ترتب عليها من زيادة فى الارساب على بيئة الشعاب المرجانية الساحلية بمنطقة الغردقة.

تنقسم الدراسة الى مكونين رئيسيين و هما الدراسة الحقلية و الدراسة المعملية, و كلا الجزئين يكملان بعضهما لتوضيح مدى تأثير الارساب على صحة الشعاب المرجانية.

الجزء الاول و هو المسوحات و البحث الحقلى:

من خلال هذة الدراسة تم قياس معدلات الارساب على طول ساحل الغردقة على مدار عامين, كما تم قياس عدة متغيرات للشعاب المرجانية بالمنطقة لايجاد العلاقة بين هذة المتغيرات و الحالة الصحية للشعاب بالمنطقة من ناحية, و معدلات الارساب من ناحية اخرى.

اختير سبعة مواقع على طول ساحل الغردقة لتمثل معدلات ارساب و ظروف بيئية و حيوية مختلفة, عدة متغيرات تمت قياسها لتعبر عن زيادة معدل الرواسب عموما فى المنطقة الساحلية و هى, معدل الرواسب العالقة و المترسبة ايضا نسبة الرواسب ذات الاصول البرية فى القاع.

عدة متغيرات اخرى تم قياسها مثل العناصر المغذية و مقاييس جودة المياة مثل العكارة و الاس الهيدروجينى و الملوحة و نسبة الاكسجين الذائب و الاملاح الذائبة و معامل التوصيل.

متغيرات للشعاب المرجانية التى تم قياسها كانت كالاتى, نسبة الغطاء المرجانى و نسبة المستعمرات الحية و الميتة و الناشئة حديثا و الوفرة المرجانية و الغنى النوعى و مقياس التنوع و مقياس التدهور و مستوى الاضطراب.

اظهرت النتائج تباين ملموس بين المواقع و لكن ليس بين العامين اللذين اجريت فيهم الدراسة. كما اظهرات الدراسة تزامن عكسى بين معدلات الارساب و نسبة الغطاء المرجانى و مقياس التنوع فى المواقع التى اجريت بها الدراسة.

خلال الدراسة تم ايضا استزراع ثلاثة انواع من شعب الاكروبورا فى مواقع الدراسة و تم قياس معدل الفقد فى كل نوع. اظهرت النتائج عدم وجود تزامن ملموس بين معدلات الارساب و الفقد فى الشعب المزروعة. كم تم اختبار معدل استقرار يرقات الشعاب على بلاطات استزراعية, و التى اظهرت ندرة عامة فى جميع المواقع.

تم ايضا قياس معدلات افراز المخاط من الشعاب المرجانية الكتلية و المتفرعة فى جميع المواقع, . اظهرت النتائج وجود تزامن ملموس بين معدلات الارساب و معدلات افراز المخاط من الشعاب المرجانية المتفرعة.

تم تقدير نسب الزوزانثلا (الطلائعيات ثنائية الاسواط المتكافلة مع المرجان) فى احد انواع الشعاب و ذلك بعد ان تم استزراعها فى مواقع الدراسة, اظهرت النتائج تفاوت فى كثافة الزوزانثيلا بين المواقع و المدة الزمنية للاستزراع, اظهرت الدراسة وجود تزامن ملموس بين معدلات الارساب و كثافة الزوزانثيلا.

تم قياس وفرة الاسماك لستة عائلات مشهورة من اسماك الشعاب المرجانية ثلاثة تتغذى على الشعاب و ثلاثة تتغذى على الطحالب و ذلك فى نفس مواقع الدراسة, كما تم قياس وفرة الاسماك لكل نوع من مجموعة اسماك الفراشات, كمؤشر على صحة الشعاب. لم تظهر الدراسة وجود فرق ملموس بين عامى الدراسة بينما كان هناك فرق بين المواقع و ايضا بين المجموعتين من الاسماك. كما دلت على تأثير ملموس لمعدلات الارساب على كثافة الاسماك بالمنطقة و خاصة الاسماك التى تتغذى على الطحالب.

خلال الدراسة تم ايضا دراسة تأثير الارساب على التأكل الحيوى(Bioerosion) للشعاب المرجانية, حيث تم مسح اربع مجموعات من الكائنات التى تنشط داخل الشعاب و تعمل على تاكلها. دلت نتائج الدراسة على و جود فروق

ملموسة بين المواقع و المجموعات فى كثافة الثقوب, اظهرت الدراسة وجود تزامن ملموس بين معدلات الارساب و نشاط بعض هذة المجموعات.

الجزء الثانى و هو التجارب المعملية:

و هو يتلخص فى اجراء ثلاث تجارب, قياس كثافة الزوزانثيلا فى ثلاث انواع من الشعاب بعد تعرضها لمستويات مختلفة من الارساب لمدد متفاوتة و قياس معدل افراز المخاط من ثلاث انواع من الشعاب بعد تعرضها لمستويات مختلفة من الارساب و قياس معدل تغذية الشعاب على الرواسب تحت ظروف مختلفة من نوع الرواسب و حالة الشعاب و كذا كمية الرواسب و فترة التعرض لها.

قياسات معدل افراز المخاط اظهرت وجود تزامن ملموس بين معدلات الارساب و معدل افراز المخاط لاحد الانواع الثلاث التى اجريت عليها التجربة

نتائج قياسات كثافة الزوزانثيلا فى المعمل اظهرت وجود فروق ملموسة بين مستويات الارساب المختلفة و الثلاث انواع التى اجريت عليها التجربة و كذا مدة التعرض للارساب. كما اظهرت الدراسة وجود تزامن ملموس بين معدلات الارساب و كثافة الزوزانثيلا فى الثلاث انواع التى اجريت عليها التجربة, و كذا مع مدة التعرض للرواسب. اى انة كلما زادت كثافة الرواسب و مدة التعرض لها كلما قلت كثافة الزوزانثيلا فى انسجة الشعاب المرجانية.

نتائج قياسات معدلات تغذية احد انواع الشعاب على الرواسب اظهرت انها تقوم بتغذية نشطة على هذة الرواسب و ان هناك علاقة طردية بين زيادة الرواسب و معدلات التغذية عليها الى مستوى معين تبدا بعدها عملية التغذية فى النقصان.

اظهرت الدراسة ان كلا من الرواسب العالقة و المترسبة تؤثر سلبا على كل من كثافة الغطاء المرجانى, تنوع الشعاب, الغنى النوعى للشعاب, متوسط حجم المستعمرة للشعاب المتفرعة و كذا وفرة الاسماك أكلة الطحالب.

هناك عوامل اخرى اثرت فى صحة الشعاب المرجانية بدرجات متفاوتة مثل نسبة الرواسب ذات الاصول البرية و درجة العكارة و نسبة الطين فى الرواسب القاعية. كما لوحظ ان عمليات الارساب لم تؤثر فى عدد مستعمرات الشعاب سواء الحية او الميتة او المتوالدة حديثا, و لم توثر ايضا على توزيع الانواع البادئة ذات النمو السريع و التحمل للظروف البيئية الغير مواتية.

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CHAPTER 1

INTRODUCTION

1.1 Red Sea environment, coral reefs and sedimentation

Given the global and local concerns relating to the impact of sedimentation on coral reefs, this study focuses on anthropogenic sedimentation stress on Hurghada reefs. The last twenty years has seen a progressive development in human activities along the Egyptian Red Sea coast. Driven directly by the tourist industry, this has indicated intensive urbanization and consequently a high rate of construction along the whole coast. The development area along the Egyptian Red Sea coastal strip is about 135,000 km² (About 1/8 of Egypt's area). This extends along the Red Sea coast for about 1080 km between Latitudes 29ºN and 22ºN in the eastern Egyptian coast (Red Sea Governorate, 2004). Most of the visitors to the Red Sea participate in water sports, including snorkeling and SCUBA diving, which both depend on and impact the state of the reefs (Hawkins and Roberts, 1994). Along the Red Sea coast large-scale development has already taken place. This development is considered to be unsustainable without the conservation of the coral reefs upon which tourism development is based (Winer, 1999).

Hurghada developed after the oil discoveries along the Red Sea coast in 1913. Later it became the capital of the Red Sea Governorate and underwent huge development activities. Ongoing development and tourism activities are continuing this expansion and affect the adjacent marine environment. The Red Sea area is known worldwide for its unique coral reefs, marine life and other natural resources. Consequently, most of the tourist development areas are located within its coastal zone (Mansour, 2003). Diversity and beauty of the natural environment is the major element of attraction for tourists in the area. The contrast provided by the biodiversity of the coral reefs and the adjacent desert is a unique attribute valued by tourists visiting the Red Sea. Egypt's current plan is a 13-fold increase in coastal tourism development, which is the major source of impact for Red Sea corals (Hawkins and Roberts, 1993). According to the Red Sea Governorate database (2004) the total number of working hotels in the Red Sea region is 188 with 36,135 rooms, including 117 hotels in Hurghada alone. A further 39 hotels are under construction, of which 22 are in Hurghada city.

1.1.1 Red Sea Reefs

The basic type of reef in the Red Sea is the fringing reef, which occurs along most of the length of the Red Sea, on both coasts but tend to be well developed in the central and northern Red Sea (Alasdair and Head, 1987). Although the zooxanthellate coral diversity in the Red Sea is high, it does not reach the diversity of most central Indo-Pacific areas. According to Alasdair and Head (1987), there are 53 genera and 177 species of zooxanthellate corals so far known in the Red Sea, the largest genera are *Acropora* and *Montipora*, with 15 species each. Also *Stylophora, Pavona, Leptoseris*, *Cycloseris, Fungi,*

Porites, Favia and *Favites* are all well represented. In contrast Sheppard (1981) lists 64 genera and over 200 species from the Chagos area of the Indian Ocean and Nemenzo (1981) recorded 78 genera and 488 species of corals in the Philippines. In the most recent publications, Veron and Stafford-Smith (2002) identified 340 species from the Red Sea compared with 634 species from central Indo-Pacific, 430 species from the north Indian Ocean and 659 species from eastern Australia. Rosen (1981) has shown that water temperature is a good predictor of coral generic diversity. The great diversity in these areas may be due to rapid local speciation, warmer waters, or simply because of the large reef areas within the dispersion range of coral larvae (Veron and Stafford Smith, 2002). The Red sea is usually warm with seawater temperature between 21.2°C at the tip of the Gulf of Aqaba and 24.5°C in the central Red Sea. However, there are seasonal patterns, especially towards the north with water temperature higher than the minimum inhibiting temperature of 10°C (reviewed in Alasdair and Head, 1987) (Figure 1.1).

Egyptian Red Sea coral cover reaches about 55.5% in exposed areas; while in sheltered areas this average increases to about 85% (Kotb et al, 2003). According to Kotb et al. (2003), the reefs in the north have the higher diversity indices, whilst the southern reefs have the highest percentage cover. Egypt's National Strategy and Action Plan for Biodiversity Conservation concluded that many plant and animal species in Egypt are on the periphery of their geographical and ecological range of distribution in the world. Under these conditions, such species or group of species has limited tolerance for ecological stress. Perhaps the best example of such a precarious existence for organisms is the case of coral reefs and mangrove swamps in the Red Sea, Gulf of Suez and Gulf of Aqaba; where these localities represent the northernmost latitudinal limit of their distribution in the world (EEAA, 1998). Environmental threats, notably from habitat destruction, over-exploitation and pollution, are increasing rapidly, requiring immediate action to protect the near pristine coastal and marine environment of the Red Sea (Wilkinson, 2000a). There has been a decline in coral cover at most sites in the Egyptian Red Sea between 1987 and 1996 from 20 to 30% (Jameson et al, 1997). Egypt has developed a network of 25 protected areas, which cover more than 15% of the Egyptian land area, with nearly all the Red Sea reefs under protection.

1.1.2 Development and Sedimentation in the Red Sea

Impacts on Red Sea coral reefs include but are not restricted to, physical breakage by divers and anchorage, excessive fishing and exploitation of species threatened with extinction, habitat destruction for developmental purposes and sedimentation. Boat anchors and anchor chains also cause significant damage to reefs at diving sites by physically breaking and destroying entire coral colonies which needs several years to re-grow (Rouphel, 2003). The grounding of daily and safari diving vessels are more frequent and contribute to local coral reef breakage. The carrying capacity of coral reefs in some Red Sea diving sites seems in no doubt to be exceeded with widespread reef degradation the most likely result (Hawkins and Roberts, 1993). Current levels of tourist diving in the Red Sea reefs directly cause high levels of damage to reef-building corals, and are above the ecological carrying capacity of the reef (Zakai and Chadwick-Furman, 2002). Outbreaks of Crown of Thorns were recorded in the Red Sea in many locations in 1998 and the four following years causing severe damage to many reefs (EST/EEAA, 2002). In the Egyptian part of the Red Sea, a decline of 20-30% in coral cover has been recorded at most sites, and this corresponds with increases in the cover of recently dead coral, and Crown of Thorns starfish (COTS) outbreaks (Wilkinson, 2000a).

As defined by Rogers (1990), sedimentation is an anthropogenically-influenced process that has been acknowledged as a major threat to coral reefs and is inferred to be responsible for local reductions in live coral cover and reef diversity. The effects of sediments on fringing reef biota have been of particular concern in the Hurghada coastal reefs, where tourism developments and recurrent dredging of a coastal shipping channel and swimming pools as well as land filling of the shallow inshore reefs have called for the need to establish extensive research programs. In Hurghada, the reefs have undergone huge dredging and land filling processes with subsequent sedimentation impacts. According to the Red Sea Governorate database (2004), in Hurghada alone, about 2.9 millions square metres of reef flat was buried and occupied with huge tourist facilities. In addition, construction of artificial beaches and private marinas, within each hotel complex adds to the sedimentation problem. The beaches regularly become eroded and consequently have to be regularly re-charged.

Although tourism development has already caused substantial damage to inshore reefs near Hurghada from land filling of reef flats, sedimentation, over fishing and shell fishing for marine curios, the intensity of impacts on reefs is likely to increase much more in the future (EEAA, 2005). This highlights the challenge for the conservation of nature in this part of the world. Elsewhere, new constructions are also beginning to modify reef habitats. Sedimentation in the Red Sea is therefore considered one of the major impacts for coral reefs. However, natural sedimentation is known along the Red Sea coast as an important factor in developing coastal and nearshore reefs (Mansour, 2003). The source of natural sediment input is runoff from adjacent wadis and flood plains, which create inland bays with sandy or muddy bottoms. Further research is needed on the response of individual reef organisms and the reef system as a whole to sedimentation, and the threshold levels for individual reef species (hard corals, soft corals and others) and for the reef ecosystem (Rogers, 1990).

1.2 Aims and Objectives of the Study

The aim of this study is to determine the impact of the recent coastal developments in Hurghada as a source of sediment to the marine environment and its consequences on the adjacent reefs.

The specific objectives were:

- To determine the magnitude of sedimentation and other marine environmental parameters including SPM, turbidity, nutrients as a major source of impact for Hurghada coastal reefs.

- To identify the major source of sediment input in the area. Tests for correlations between the beach and bottom percentage non-carbonate sediment, will help to define the relationship between landfilling activities and both sedimentation rate and SPM concentration

- To determine correlations between sedimentation and the biological status of the reefs.

- To determine the current reef health status in the area under the observed environmental condition. The following reef health parameters were examined: coral cover, abundance, diversity, richness, mortality, survival of transplanted corals, coral deterioration, recruitment, mucus production, zooxanthellae density and macroborer

distribution at seven sites that were exposed to varying levels of sedimentation. Correlations were tested between each of the health parameters and sedimentation rate, SPM concentration, percentage non-carbonate sediment, turbidity, percentage of gravel, sand and mud of bottom sediment.

- To identify the most appropriate reef health indicators which give a clear signal of the reef status in the study area.

- To define the reef health trends and the future expectation for the area.

- To determine the effect of sedimentation and reef deterioration on fish population structure. Fish surveys were carried out for 6 fish families (three coral feeders and three algal feeders). In addition, abundance and species richness of Butterfly fish were examined in the study area as a reef health indicator.

- To explore more precisely the relationship between sedimentation and individual coral health parameters, laboratory experiments were carried out to measure mucus production rates at four different sedimentation rates, and zooxanthellae cell density of three common coral species after sedimentation treatment. In addition, laboratory based experiments were performed to test the ability of one species of common coral in the region to feed on sediment and gain advantage from living in a high sedimentation area. Results of feeding laboratory experiments will provide information on the ability of certain coral species to survive or replace the dead ones under the same sedimentation conditions. The results will reveal the correlations between sedimentation rate and sediment uptake rate by this coral species.

Sedimentation rates, SPM concentration, percentage of non-carbonate sediment and turbidity were measured at a number of sites along the coast of Hurghada. The study also examined sediment composition (grain size analysis and percentage of non-carbonate sediment of both bottom and beach) and the percentage of gravel, sand and mud of bottom sediment were all measured along the study area. Two years data were collected for each of these parameters at each site. Sites were selected to represent all environmental and sedimentation condition in the area of the reef facing the Hurghada. The sites from north to south were as follow; NIOF (1), Abu Sadaf (2), Shedwan (3), Arabia (4), Abu Minkar (5), Holidays (6) and S.Hashish (7) (Figure 2.1).

Water quality parameters measured in the field included water temperature, salinity, pH, specific conductivity (SPC), dissolved oxygen (DO), total dissolved salts (TDS), percentage of dissolved oxygen (DO%) and turbidity. Nutrients analysis for major nutrients, inorganic phosphate, Silicate, Ammonia, Nitrite and Nitrate were carried out in the laboratory. Correlations were tested between salinity, PH, SPC, DO, TDS, DO% and turbidity with sedimentation rate, SPM concentration, non-carbonate sediment percentage and mud percentage of bottom sediment. Correlations also were tested between all of the nutrients; NH_4, NO_2, NO_2, PO_3 and SIO_3; and turbidity, sedimentation rate, SPM concentration, non-carbonate sediment percentage and mud percentage of bottom sediment.

The first chapter of the thesis includes aims and objectives, introduction and literature review of the study subject. Study methods and materials are described in Chapter 2. The results of the study are presented in the next 4 chapters. Environmental parameters including sedimentation rate, SPM concentration, turbidity, sediment type and the percentage of non-carbonate sediment, nutrients and physical-chemical parameters are all incorporated in Chapter 3. Chapter 4 reports on all coral parameters measured in the field

including; coral cover, abundance, Species Richness, percentage r-strategists, Diversity Index, Deterioration Index, Disturbance, transplanted coral survival, settlement and recruitment. Abundance of 6 fish families, butterfly fish species richness and the intensity of bioerosion by four macroborers are reported in Chapter 5. Chapter 6 covers the results from mucus measurement for field and laboratory experiments, the results of two feeding experiments and zooxanthellae measurements both in the field and in the laboratory. Chapter 7 covers the discussion of the study outcome, conclusions and suggestions for further work.

Results from this study will serve to identify whether coastal development has increased the turbidity and sedimentation process in the study areas. Furthermore, if this is the case, whether there has been a corresponding depression in the reef health of the area. In regard to the future of the area; the study will help to determine if these impacts are increasing or decreasing in the long run. The study will also consider which of the used reef health parameters are more appropriate and sensitive indicators of sedimentation stress. By comparing reef health parameters over two years it will be possible to determine whether there is a chance for coral reefs under this impact to be restored naturally or whether certain mitigating process will be required. Recruitment and survival intensity may be a useful measure to inform managers whether physically damaged reefs may require coral transplantation or whether they can be left to recover naturally.

1.3 Coral Reefs

1.3.1 Coral distribution

The coastal euphotic waters of tropical and subtropical seas are often dominated by coral reefs. Coral reefs are important constituents of the coastal benthic community and cover about 1.2% of the world's continental shelf area (Spalding et al., 2001). The ecosystem is associated with high biodiversity and is of economic and environmental importance. Coral reef ecosystems provide many functions, goods and services. Costanza et al. (1997) defined functions as habitat, biological or system properties or processes; goods such as food and services such as waste assimilation. They are essentially tropical, shallow water ecosystems largely restricted to the area between latitudes 30°N and 30°S and are most abundant in shallow, well flushed marine environments characterized by clear, warm, low-nutrient waters that are of average oceanic salinity (Global Biodiversity Outlook, 2004). Coral reefs are known as the most diverse and productive marine ecological system and dominate the coastal water of the warm seas (Veron and Stafford Smith, 2002).

Although coral reefs are estimated to cover only 2% of earth's surface area, they encompass a substantial proportion of marine biodiversity (Kleypas, 1997). There are two groups of coral animal present, reef building or hermatypic corals and non-reef building ahermatypic corals. Hermatypic coral is the common name for organisms of the Order Scleractinia. These have a hard calcareous skeleton and are inhabited by symbiotic microalgae called zooxanthellae. Non-reef building, ahermatypic corals do not have intracellular symbiotic algae and therefore are not dependent on light. There is a clear link between ocean temperature and both zooxanthellate coral distribution and the formation of highly consolidated reef. Coral have the capacity to disperse to all latitudes, but it can only develop a well-developed reef where water temperature does not fall below 18ºC for

extended periods of time (Veron and Stafford Smith, 2002). Species diversity reaches a peak, for any given region on fringing reef protected from strong wave action where the water is slightly turbid. This is probably because *Acropora* does not dominate this habitat, giving a chance for other groups to propagate (Veron and Stafford Smith, 2002).

It is difficult to measure the global area of coral reefs, especially under continuous changing environmental condition and related reef degradation. Titlyanov and Titlyanova (2002) estimate reefs to exceed 600,000 km², and the islands built by coral to occupy more or less the same area. Near-surface reefs, which are the most productive and diverse ecosystem, are estimated to cover around 255,000km² (Global Biodiversity Outlook, 2004).

Although there are variations in the estimates of reef area, there is great consensus that coral reef are among the most productive and diverse of all natural ecosystems on the earth, second only after the terrestrial analogue the tropical wet forests (Global Biodiversity Outlook, 2004). Coral reefs support large coastal human populations, but their existence is threatened by the economic activities they support (Cesar et al., 1997; Berg et al., 1998; Edinger et al., 1998; White et al., 2000). Coral reefs worldwide are subject to extensive anthropogenic damage and many studies refer to and quantify anthropogenic impacts on reef (Wilkinson, 1992; Sebens, 1994; Hodgson, 1999).

1.3.2 Reef Topography

In regards of reef topography and structure, three basic reef types have been recognized: fringing reefs which lie close to shore and line the continental edge, barrier reefs arise at the edge of an offshore shelf, and atolls which have an ovoid reef structure which emerge in mid ocean areas and are characterized by a central lagoon. Reefs are made of two types of materials, consolidated limestone and unconsolidated rubble (Veron and Stafford Smith, 2002). Consolidated reefs are constructed of solid limestone pulled together by waves and cemented by coralline algae. In the Red Sea the basic reef type is the fringing reef, which varies in width and physical shape and grows slowly seaward at the seaward crest where water exchange is good. The Red Sea's extensive coastline incorporates a considerable range of coral reef formations including fringing reefs that fringe the inland coast as well as offshore islands and some offshore submerged reef patches.

1.3.3 Coral Forms

There are two forms of coral animals, hermatypic and ahermatypic. The terms are derived from ''herma'' meaning mound or reef. The early definition, ''hermatypic'' refers not only to corals that build reefs but also to those species possessing zooxanthellae. Ahermatypic corals, on the other hand, neither build reefs nor possess zooxanthellae (e.g. are azooxanthellate). It is usually regarded that ahermatypic corals as those taxa living in cold and deep water, below the photic zone, while hermatypic corals inhabit shallow, tropical to subtropical zones. Most scleractinian corals have the ability to calcify rapidly and their success as reef builders is related to the symbiotic association with zooxanthellae (reviewed in Stanley and George, 2003).

The hermatypic corals have attracted a lot of scientific attention mostly because of their symbiosis with unicellular algae, zooxanthellae. The presence of zooxanthellae in corals allows them to create a dual pattern of feeding (autotrophy and heterotrophy) and to inhabit nutrient poor (oligotrophic) waters (Leletkin, 2000a). This pattern of feeding has a profound importance to reef ecology and distribution. These unicellular algal symbiotants

live in the endodermic tissues of their coral host in great numbers, and are thought responsible for promoting calcification and then coral skeleton construction. The evolutionary significance of this symbiosis and the implications it holds for explaining the success of corals is of vital importance. The symbiotic relationship between living corals and zooxanthellae stands behind the reef phenomena of calcification and light-depth adaptation (Veron and Stafford Smith, 2002). Coral animals are however equipped with stinging cells (nematocysts), and are capable of voracious zooplankton feeding.

The modern hermatypic scleractinian corals can be classified according to their life strategy into three groups; *r-strategist*, *k-strategist* and an intermediate group between the two (Sorokin, 1995). The *r-strategist* corals are opportunistic, have small to medium size colonies and reach sexual maturity early. This group is characterized by short life duration, high growth rate and intensive breeding (Sorokin, 1995). The opportunistic corals can survive different kinds of stress such as low salinity, exposure, water warming and pollution. The most common examples are *Stylophora pistillata, Pocillopora damicornis, Seriatopora histrix, Psammocora contigua* and many species of *Montipora, Acropora* and *Pavona*.

The *k-strategist* corals are conservative, use most of their energy for growth, have an annual breeding cycle, and long life span and have unlimited growth. This group may live for hundreds of years. Examples include the massive corals *Porites* and *Montastrea*. The third group has an intermediate life strategy (*t-strategist)* between the two contrasting groups and includes the rest of coral groups. This group is less specialized and found in a wide range of environments, forming a multitude of adaptive ecomorphs. Examples of this group include some species of *Acropora, Pavona, Hydronophora, Galaxea* and *Goniopora* (reviewed in Sorokin, 1995).

It has been argued that the early definition of hermatypic coral should be abandoned as none of the fossil coral possessed zooxanthellae or built a reef (Stanley and George, 2003). While the association with symbiotic zooxanthellae restricts many corals to shallow-water reefs, not all zooxanthellate corals build reefs and many zooxanthellate corals inhabit non-reef environments. There also exist living thickets and reef-like mounds characterized by substantial amounts of coral bioconstruction growing in the aphotic zone, in water as deep as 1500m (reviewed in Stanley and George, 2003).

In one approach to overcome these problems, Rosen (1981) proposed that the terms ‘‘zooxanthellate’’ and ‘‘nonzooxanthellate’’ be divorced from reefdwelling or reef building. Rosen (2000) reviewed living zooxanthellate coral taxa, which he designated ‘‘z-corals’’, and living azooxanthellate taxa, which he designated ‘‘az-corals’’, and he acknowledged the novel metabolic capacity enjoyed by z-corals, which usually also are associated with rapid calcification. Living scleractinian species are considered either z-corals or az-corals and in terms of diversity among the 1314 species currently known, they are about equally split with 48.2% of the genera and 49% of the species being z-corals. There is a small number of living species that can and do exist perfectly well in either of the two states. In other words, they are species having the capacity to switch back and forth and are called apozooxanthellate. Such species are facultatively zooxanthellate corals (reviewed in Stanley and George, 2003).

1.3.4 Coral Symbioses

Scleractinian corals have five different feeding patterns; (1) the majority comes from zooxanthellae as photoassimilates, (2) Ingestion of suspended particles of animal or

bacterial origin directly from water, (3) dissolved organic substances (DOS) of sea water obtained by the coral using an osmotrophic feeding pattern, (4) predation, which provides, on average, 10–40% of the overall biomass of feeding; and (5) the digestion of the plant symbionts proper (zooxanthellae) (Titlyanov and Titlyanova, 2002). The relationship between animal and plant cells through the exchange of substances and the input of heterotrophic and autotrophic functions to the energy budget of the coral organism has always been and still is the subject of intensive experimental investigation (Leletkin, 2000b). Although the dynamics of this relationship and exactly how it promotes rapid calcification are currently debated (Marshall, 1996; Goreau et al., 1996), this symbiotic relationship is crucial to gaining a full appreciation of why living corals are so successful on reefs. This relationship helps explain the rapid calcification phenomenon among corals, their restriction to tropical, shallow water settings, and the success of coral reefs as ecosystem (Stanley and George, 2003).

Many studies have summarized the nature of the interactions between corals and their symbiotic algae and analyzed the benefits and costs of the symbiosis (Muller-Parker and D'Elia, 1997). The zooxanthellae were traditionally classified as a single species *Symbiodinium microadriadicum*, but recent molecular studies revealed their taxonomic diversity (Trench, 1997). Symbiotic dinoflagellates comprise 25 species belonging to eight genera and four orders of algae. However, scleractinian corals comprise only one species of symbiotic dinoflagellates of the genus *Symbiodinium.* This species is however genetically heterogeneous and is represented in nature by different morphophysiological, genetically determined types (taxons) of algae (Titlyanov and Titlyanova, 2002). The fact that the coral–zooxanthellae relationship is more random and not as phylogenetically coherent among particular coral species is supported by the discovery that genetically diverse populations of these endosymbionts may inhabit the tissues of the same coral species and that genetically differentiated populations of *Symbiodinium* may inhabit different coral hosts, at different depths and in different levels of light (Stanley and George, 2003).

Hallock (2001) reviewed the advantages of this association and emphasized the fact that symbiotic zooxanthellae provide their hosts with a magnitude of energy, more than normally available to other heterotrophic organisms. This is accomplished through the production of photosynthate products consisting of carbohydrates and lipids. Zooxanthellae provide corals with the bulk of their food (30–90%) as photoassimilates (Bil et al, 1991). Early studies demonstrated that, on average, the gross photosynthesis in reef-building corals of the Indo-Pacific at midday ranged within 10–50μg O_2/cm^2 of the colony surface per hour, sometimes reaching a value of 100μg/O_2/(cm^2/h) (Edmunds and Davies, 1986). This energetic advantage is really what permits zooxanthellate corals to prosper in nutrient-limiting environments that otherwise could not support them (Stanley and George, 2003).

As a by-product of the association, the zooxanthellae also promote the calcification process (Pearse and Muscatine, 1971). Among their other attributes, scleractinian corals are considered hypercalcifiers, that is, they have the ability to extract large amounts of $CaCO_3$ from seawater and secrete it as skeleton (Stanley and Hardie, 1999). The products of photosynthesis by zooxanthellae are the primary source of energy used by corals for calcification and skeletal growth; the loss of zooxanthellae reduces the amount of energy available for accretion of calcium carbonate and subsequently growth (Souter and Linden, 2000). The great ability to secrete $CaCO_3$ among scleractinians, calcified algae and other marine organisms, largely results in the calcium carbonate structure called coral reef (Stanley and George, 2003).

1.3.5 Reef Productivity

Coral reefs are one of the most productive ecosystems in the world, which often develop, in an oligotrophic oceanic environment. This paradox is not a recent concept and it is now accepted as an oversimplification (Hatcher, 1997). Reef building corals provide a significant contribution to the production of organic matter and to the development of the carbonate of the coral reef (Titlyanov and Titlyanova, 2002). The zooxanthellae function *in situ* as primary producers and much of the fixed carbon is translocated to the cnidarian host, and provides a significant portion of the coral's daily carbon requirement for maintenance respiration (Ogden and Gladfelter, 1983).

Nitrogen sources for a coral include both organic nitrogen from ingested food (Johannes et al, 1970; Johannes, 1974) and inorganic nitrogen in the form of nitrate (Franzisket, 1974; D'Elia and Webb, 1977; Webb and Wiebe, 1978) and ammonia (Muscatine and D'Elia, 1978). Coral has a high ability to absorb nitrogen and phosphorus compounds from seawater, and has a role as producer and consumer of detritus (Lewis, 1977). Through the activities of zooxanthellae, corals efficiently absorb inorganic nitrogen from the environment, showing the ability to take up both ammonia and nitrate, and also retain rather than release excretory ammoni (Ogden and Gladfelter, 1983).

Photosynthetic products are translocated from the zooxanthellae to the animal host (Trench, 1979; Muscatine, 1980). Since these products are assimilated and metabolized by host tissue, the coral can be viewed as an herbivore, functioning as a primary consumer (Ogden and Gladfelter, 1983). Corals are also active carnivores, feeding by a variety of mechanisms utilizing tentacles, ciliary mucus tracts, and extracoelenteric extrusion of gastric filaments (Lewis and Price, 1975). In summary, corals probably do not depend on secondary consumption for their total daily energy needs. They do, however, probably rely on feeding to supply nitrogen and perhaps phosphorus to the system; they might also feed heavily when food is available and then store excess food in the form of lipids. Reef corals have a fine recycling mechanism which enables them to exist under a variety of nutritional regimes; i.e. higher or lower irradiances; more or fewer zooplankters; higher or lower nutrient concentration (Ogden and Gladfelter, 1983).

Zooxanthellae are the primary producers of the coral symbiosis; they behave as oligotrophic phytoplankton (Muscatine, 1980). Primary productivity in coral is expressed as the amount of carbon fixed per unit chlorophyll a per unit time (Ogden and Gladfelter, 1983). Lewis (1977) estimated reef production levels as high as 10kg $C.m^{2}.y^{-1}$ in contrast with those poor waters of the tropical reefs where productivity is only 20-50gm $C.m^{-2}.y^{-1}$ by the pelagic community. High reef productivity seems to be related to the ability of the reef community to retain and recycle all nutrients coming to it (Alasdair and Head, 1987). Photosynthetic production in the ecosystem of the coral reef has been estimated to constitute approximately 2-5gm of organic carbon for $1m^{2}$ of bottom per day (Sorokin, 1990). An early study of daily productivity showed that generally shallow water corals produce more carbon than they consume, but this pattern changes with depth (Ogden and Gladfelter, 1983). Many studies have measured coral growth rates; Stoddart (1969) stated that reef upward growth is about 0.2-0.7cm per year.

1.3.6 Reproduction and Recruitment

Many studies have focused on coral reproduction and regeneration. Corals reproduce either by brooding or broadcast spawning. Brooding coral release fully formed planula larvae after internal fertilization of eggs, in a process called planulation (McGuire, 1998).

Broadcasting coral release gametes into the water, where fertilization and larval development take place. Many broadcast-spawning corals release their yearly reproductive investment on a single evening (Harrison et al. 1984; Gittings et al. 1992). During these synchronous and brief spawning events, gametes are released simultaneously in the form of gamete bundles. Broadcast-spawning massive coral species have low reproductive success, as evidenced by low recruitment rates and comparatively high adult survivorship (Hughes and Tanner, 2000; Meesters et al. 2001).

Seasonal changes in water temperature may determine the timing of annual reproductive cycles in corals, while tidal cycles and lunar periodicity may influence the timing of gamete release (review in Richmond and Hunter 1990; Szmant, 1991). Larval release by corals appears to be correlated with the average water temperature over the 3 days prior to the new moon, peaking when water temperatures are in the range 24.5–27.5°C (McGuire, 1998). In the western Atlantic, broadcast spawning corals typically release gametes several days after the full moon in the late summer, when water temperatures are maximal (Szmant 1991; Wyers et al. 1991; Acosta and Zea, 1997). On western Australian reefs, corals spawn in late summer (Simpson et al. 1993), while on the Great Barrier Reef mass spawning by corals occurs in the late spring and early summer (Willis et al. 1985; Oliver et al. 1988). Lunar cycles seem to regulate planula release by the brooding *Pocillopora damicornis* (Jokiel et al. 1985).

In recent years, scientists have recognized that broadcast spawning is the more common pattern (Richmond and Hunter 1990), and that the timing of spawning for some coral species can be accurately predicted from the lunar cycle (Harrison et al. 1984; Szmant, 1991). These broadcast spawning events typically occur once or twice per year for a given species. In contrast to broadcast spawning corals, many brooding corals have several reproductive cycles throughout the year, usually with monthly periodicity (McGuire 1998). However, several brooding species of coral have been observed to release larvae with some sort of lunar periodicity in both the Atlantic and Pacific oceans (Szmant-Froelich et al. 1985). Reproductive periodicity may vary with location, both latitudinally and from one reef area to another (Stimson, 1978; Rinkevich and Loya, 1979; Oliver et al. 1988) but may still have an underlying seasonal pattern (Stimson 1978; Harriott, 1983; Stoddart and Black, 1985). Larvae usually settle in four to ten days and some scientists believe that most larvae settle within 600m of the parent reef while others argue that some larvae travel longer distances (Seaworld, 2007). After a time planula larvae become 'competent' and can settle on a solid surface, metamorphose into a polyp, and secrete their corallite (Anderson, 2003).

Recruitment is one important component of coral life history that can compensate mortality and tissue loss processes (Ammar et al., 2000). Many factors have been shown to influence recruitment and propagation including, the biotic composition of communities, local oceanography, substratum complexity and herbivore grazing intensity (Baggett and Bright, 1985; Carlton and Sammarco, 1987). Recruitment patterns also vary between fore-reef and back-reef sites (Harriot and Fisk, 1987).

Low recruitment rates may be due to low availability of larvae or low settlement success (including larval preference), or both (Dikou and van Woesik, 2006). Recovery periods for damaged reefs vary and depend largely on the nature of the disturbance. However, it may be accelerated by stabilizing substrate and removing loose sand or rubble (Miller et al., 1993), or consolidating rubbles (Wulff, 1984), deploying artificial structures to serve as areas for coral settlement or stable sites for transplantation (Clark and Edwards, 1994), and transplantation of corals to the damaged areas (Yap et al., 1992).

Coral under harsh conditions might favor energy allocation strategies emphasizing growth above reproduction to cope with limited food resources and reproductive characters such as fecundity, reproductive allocation, and reproductive effort might be expected to diminish with increasing stress on corals (Villinski, 2003). Growth, tissue repair and the production of the germ cell require one major resource, the stem cells. Trade-offs for stem cells between tissue repair and sexual reproduction should be considered as a relevant factor shaping reproductive activities during regeneration in reef corals (Rinkevich, 1996). Lang da Silveira & Van't Hof (1977) maintained that repeated cycles of injuries and regenerations in *Plexaura flexousa* can inhibit future regeneration by depleting critical population of stem cells. Wahle (1983) found that regeneration was independent of either colony size or reproductive phase. Rinkevich (1996) suggested that any extensive use of these reserve cells may reduce their numbers and significantly affect one or more biological functions, he proposed that a trade-off for stem cells between tissue repair and sexual reproduction should be considered when studying changes in reproductive activities during regeneration in reef corals.

Many studies have clearly demonstrated the adverse effects of increased sediment load on various stages of coral recruitment, including gamete production, egg fertilization, embryo development, larval settlement and survival, and juvenile growth and survival (Dikou and van Woesik, 2006). Since most corals are mass spawners and produce floating gametes, pollutants and toxins in the surface water layer can effect coral reproduction and development to a large extent. Contamination by pesticides, heavy metals, hydrocarbons or other human-made pollutants can significantly affect the health of reefs at local scales (Guzman and Holst, 1993). For example, heavy metals such as copper and zinc and some hydrocarbons have been linked to reduced fertilization, fecundity and growth in adult corals (Heyward, 1988; Brown, 1987).

1.4 Conservation of Biological Diversity

The human population has increased greatly leading to a rise in the consumption of biological diversity components in the last few decades (UNEP/IUCN, 1988). As the result of urbanization and expansion of industrial production, threats to these natural systems have increased and affect biodiversity in many ways. Ozone depletion and global warming, as universal threats, have been added to significant local threats such as eutrophication, sedimentation, chemical pollution, and overfishing which are occurring worldwide (Hallock, 2005). The rates of eradication and loss of biodiversity are high and, in some cases, the situation has reached a critical level in regard of species and habitat loss as stated by the Convention on Biological Diversity (CBD, 1992a).

At the global level, rising patterns of unsustainable consumption of marine resources are usually correlated with high levels of poverty and the lack of awareness of the consequences (CBD, 1992b). Human activities consistently amplify the impacts of naturally occurring stresses. For example, hurricanes are natural events whose impact on coastal ecosystems is hugely aggravated by deforestation, agriculture, and coastal development (Hallock et al., 2004). Unsustainable use of the components of biological diversity will continue the deterioration of biodiversity at both species and habitat levels. In the last few decades human activities have expanded to alter the earth’s atmospheric and oceanic chemistry, with profound impacts on climate and the biosphere; rather than local scale environmental degradation.

At the Earth Summit in Rio de Janeiro in 1992, world leaders agreed on a strategy for sustainable development, which meets current human needs and ensures sustainability for future generations. One of the key agreements adopted at Rio was the Convention on Biological Diversity (CBD, 1992a). The CBD is the first global agreement on the conservation and sustainable use of biological diversity. The Convention established three main goals: the conservation of biological diversity, the sustainable use of its components, and the fair and equitable sharing of the benefits from the use of genetic resources (CBD, 1992b). The World Summit also recognized the importance of biodiversity conservation for overall sustainable development and effective poverty eradication programs.

The 2nd conference of the parties of the CBD adopted the "Jakarta Mandate on Marine and Coastal Biological Diversity" in 1995. This focuses on the conservation and sustainable use of marine and coastal biodiversity (Jakarta Mandate, 1995). Seas and coastal areas are considered under threat because of many reason includes pollution, over-exploitation and ill-planned coastal development. Many reef areas in the world have been degraded because of human impacts, and the world's fishery resources are threatened by depletion because of overfishing (Jakarta Mandate, 1995). Other living resources, such as mangroves, seagrasses and their habitats are also over-exploited in many areas around the world. Coral reef ecosystems are increasingly being degraded and destroyed worldwide by a variety of human activities in addition to global warming (Jakarta Mandate, 1995).

The CBD through the Jakarta Mandate program, and in order to achieve its objectives, has developed five program elements. The work program on marine and coastal biodiversity aims to assist the implementation of the Jakarta Mandate at the national, regional and global level. It identifies key operational objectives and priority activities within the five key program elements, namely: implementation of integrated marine and coastal area management, marine and coastal living resources, marine and coastal protected areas, mariculture and alien species. The parties of the convention agreed upon and commit themselves to a more effective and coherent implementation of the three objectives of the Convention, to achieve by 2010 a significant reduction of the current rate of marine biodiversity loss (Zedan, 2003). MPAs were recognized among the most applicable way to manage overexploited fisheries; a recent worldwide summary of MPAs produced an impressive list of over 1300 MPAs with subtidal components (McClanahan, 1999).

Generally, establishing Marine Protected Areas (MPAs) is the action urgently required to mitigate the direct human impacts of land-based pollution, sediment release, and over-exploitation. Permanent marine protected areas are an essential element in the management of coastal resources (McClanahan, 1999).

1.5 Threats to Coral Reefs Resources

1.5.1 General Introduction

Nearly all of the world reefs are undergoing various degrees of impacts. Stachowitsch (2003) maintains that research in undamaged marine ecosystems has become difficult, and that most of the future insights into marine habitats will stem from knowledge gained by examining various missed functions of those systems rather than their functions. Future reef research will therefore differ from the past in many aspects; the aims, range of topics, the selection and funding of those topics, the validity of its conclusions, and in its urgency (Stachowitsch, 2003). The declaration of the 10th international coral reef symposium

(2004) acknowledged that coral reefs worldwide are becoming degraded to a critical level. It highlighted the urgency for conservation and restoration and advocated scientific research and rigorous monitoring, management-tool development, and appropriate measures for conservation and sustainable use of coral reefs.

There is a serious concern that coral reefs are being rapidly degraded by a wide range of human activities (Wilkinson, 1993; Ginsburg, 1994; Birkeland, 1996; Bryant et al., 1998). Bryant et al. (1998) developed a risk index analysis based on the effect of the major threats to the health of coral reef systems namely, coastal development, overexploitation, inland and marine pollution and erosion. Overall, about 58% of the world's reefs are under medium to high threat, with regional values reaching over 80% in some sites as in Southeast Asia (Stachowitsch, 2003). Damage to coral reef ecosystems comes from a wide range of direct and indirect human activities; for instance, unsustainable fishing practices can lead to dramatic change in dominant species (Jackson, 1997). Impacts such as increased rates of eutrophication and sedimentation, bleaching associated with increased water temperatures, damage by anchors or cyclones all appears to be increasing. In addition, events such as Crown of Thorns plagues, all contribute to death of coral colonies (reviewed in Hutchings and Peyrot-Clausade, 2002).

Environmental stress is usually magnified by many factors such as: poor water circulation, nutrient rich and fresh water runoff, widely fluctuating salinity, high levels of suspended solids, elevated water temperatures and widely varying dissolved oxygen levels (reviewed in Linton and Warner, 2003). Other sources of stress include sediments, organic loading, and direct physical damage. Direct sedimentation onto the reef or an increase in the turbidity of the water due to eutrophication, both leads to decreases in the amounts of light reaching corals and may cause bleaching (Brown and Ogden 1993). In addition, increases in the amounts of nutrients enhance the growth of other reef organisms such as sponges and algae, which may compete with corals for space. The indirect effects of human exploitation of other reef resources (e.g. fish and shellfish) may also be deleterious to corals. For example, fishing of parrotfishes has a disproportionately large and deleterious impact on the ability of coral recruits to escape overgrowth by macroalgae (Mumby, 2004).

Coral colonies are capable of recovering from small lesions; however, when a damaged area is large, or physical damage occurs frequently, recovery may be difficult (Bak et al., 1977; Bak and Steward-Van Es, 1980). Corals are less likely to recover from chronic disturbance or disturbance that alters the physical environment even at low levels than from acute, intense disturbance that leaves the habitat intact or less offended (Connell et al., 1997). Examples of the first kind of disturbances include coral dynamiting and ship grounding which leave movable rubble and coral fragments incompatible for coral larvae settlement. On the other hand Crown of Thorns starfish outbreaks, which cause instantaneous death for huge areas of coral reefs, do leave coral skeleton as suitable substrate for settlement progression.

1.5.2 Human Population and Development of the Coastal Zone

The coastal area is the region of interaction between land and sea; it is a dynamic and highly complex area (Kuijper, 2003). The coast is affected not only by local conditions but also by events and conditions long distances away. The coastal zone supports the majority of the planet's human population (World Bank, 1992) and contains some of the most

productive ecosystems with rich biodiversity reserves (GESAMP, 1997). It was reported that 30% (1.2 billion) of the world human population lived at or near the coast in 1990 (Small and Nicholls, 2003), while this percentage rose to 41% (2.5 billion) by 2002 (UN, 2005). During this period the coastal population increased by 919 million, which represents 56% of the initial population in 1992. By contrast, in the same time period the global population rose from 5.4 to 6.2 billions (a total of 783 million) (NationMaster, 2005), which represents 14%. These figures show that the coastal population is growing at a high rate, probably as a result of the combination of both population growth and migration (Martinez et al, 2007). Within the next three decades, 75 percent of the world's population could reside in coastal areas (NOAA, 2007).

Travel and tourism is the world's largest industry. As reported by the World Tourism Organization, travel and tourism involved more than 528 million people internationally and generated $322 billion in receipts in 1994. In 1995, travel and tourism generated an estimated $3.4 trillion in gross output creating employment for 211.7 million people (NOAA, 2006). More than twenty percent of Egypt's population live in the coastal zone, nearly forty percent of industrial development is concentrated along the coasts, more than eighty-five percent of the nation's oil and gas is located in the coastal areas (EEAA, 2006). In addition, more than eight million tourists visit Egypt annually, coastal tourism is the major sector of the Egyptian tourism market, with estimated annual reef recreational value of about USD142 million (EEAA, 2006). There are currently ten coastal protected areas in Egypt receiving more than 1.5 million visitors annually (EEAA, 2006).

Some of the currently recognized human impacts on the coast and coastal ecosystems include first, habitat and shoreline modification which has altered currents and sediment delivery, enhancing coasts in some areas and inducing erosion and receding beaches in others (Psuty, 2004), coastal habitats are being polluted, modified by development and replaced by artificial structures.

Second is over-exploitation of resources. A preliminary estimate of endangered littoral species indicates a total of 85 species at risk (Burke et al., 2001). As the extent and functionality of coastal ecosystems declines, the capacity to deliver ecosystem services will become depleted and, eventually, be lost.

Third, invasive species are one of the most globally pervasive threats to natural ecosystems worldwide (Primack, 1993). Human vectors have aggravated the natural movements of species from one region to another (Martinez et al, 2007). Multiple experiences have demonstrated that removal of invasive species may be an extremely expensive and time-consuming task that is not always successful (reviewed in Martinez et al, 2007) (Table 1.1).

Table 1.1: The major impacts to coastal and marine environment and its main sources.

Impacts	causes	reference
Changes in currents and sediment delivery, enhancing coasts in some areas and inducing erosion in others.	Habitat and shoreline modification	Psuty, 2004
Decline in functionality of coastal ecosystems, increase in endangered species	over-exploitation of resources	Burke et al., 2001
Change of community and genetic structure, disturb individuals (internal biological pollution by parasites or	invasive species and biological pollution	Primack, 1993, Elliott, 2003

pathogens), a population (by genetical change) or a community (by increasing or decreasing the species complement)		
Coral reefs degradation	Increase human population in coastal areas	Owen et al, 2005; UNEP/IUCN, 1988; Wilkinson, 2002
Increase of macro-algae, reduced grazing fishes and sea urchins, increase phytoplankton growth, coral diseases.	nutrient pollution	Wilkinson, 2004

Major anthropogenic threats to coral reefs are linked to high human population densities in coastal areas (UNEP/IUCN 1988). Coral reefs continue to deteriorate in all areas where human activities are concentrated, with approximately 30% of the world's coral reefs already considered to be seriously degraded (Wilkinson, 2002). Globally coral damage is obvious in many areas, notably along the coast of eastern Africa, all of continental South Asia, throughout southeast and East Asia and across the wider Caribbean region. Where both warming and direct human impacts occur together, each may exacerbate the effects of the others (Westmacott et al, 2000).

Human activities are playing a major role in coral reef demise, contributing to; the conversion of habitats dominated by corals to those dominated by algae, coral bleaching, disease, mortality, erosion and reef loss (Owen et al., 2005). All reefs near human populations or adjacent to large land masses suffer degradation from nutrient pollution (Wilkinson, 2004). Reefs are damaged by excess nutrients in many ways; as a consequence of the growth of macro-algae when the populations of grazing fishes and sea urchins are reduced, or increase phytoplankton growth in seawater which reduce light energy penetration to the light-dependent corals; leading to the growth of other competitors of corals, especially those that bore into coral skeletons such as sponges, molluscs, worms and burrowing algae; and probably excess nutrients make corals more susceptible to disease (Wilkinson, 2004).

1.5.3 Overexploitation, Overfishing and Destructive Fishing Practices

Dalzell (1998) concluded that, subsistence artisinal fishing in the Pacific has had no impact in coral reef fish communities over the last thousand years, as indicated from the archeaological records. However, the direct anthropogenic threats to coral reefs are increasingly being magnified by the impact of fisheries in reef habitats (Dayton et al., 1995). On the one hand, overfishing involves overexploitation to the point of possible ecological extinction (Jackson et al., 2001). According to Erdmann (2000), destructive fishing practices have been considered to be the largest immediate threat to coral reef ecosystems in some countries. In addition to the potential for over-fishing, fishery operations can have a destructive physical impact on the seabed, and affect population levels of non-target species through incidental catch, such problems being of particular significance for reef fishes, cetaceans, sea turtles and seabirds (GBO, 2004). Fishing gear used in coral reef areas is known to cause direct physical damage to the reef substratum

(Jennings and Kaiser, 1998). Gill nets, fish traps, and anchors all break coral physically or cause direct coral death through entanglement and abrasion (Saila et al., 1993; McManus et al., 1997; Edinger et al., 1998; Jennings and Kaiser, 1998). Lost or discarded fishing lines, along with other marine debris such as fishing nets and plastic ropes, have been reported to entangle marine wildlife such as seals, sea lions, dolphins, turtles and sea birds (Jones, 1995).

Another form of devastating fishing practice, blast fishing has been well documented (Alcala and Gomez, 1987; Edinger et al., 1998; McManus et al., 1997; Pauly et al., 1989). Blast fishing causes dramatic devastation to the reef framework and tremendous loss to society using the reef, furthermore the recovery processes is likely to be so slow (Cesar et al., 1997; Pet-Soede et al., 1999). The removal of large numbers of reef fish has affected the coral reef ecosystem balance and allowed more competitive organisms, such as algae, to become dominant on reefs in many regions.

As a consequence of decreased yields, fishermen change their methods in order to catch adequate fish to sustain their needs. This includes the use of small mesh size nets, which catch even the small juvenile fish, and the use of explosives or poisons (Richmond, 1994). These practices not only kill all fish in the affected areas, but they also severely damage the corals and all marine life in these areas. The removal of fish has been likened to removing the immune system of coral reefs. Coral reefs without fish are far more susceptible to overgrowth by macro-algae, plagues of coral predators, and increases in disease (Wilkinson, 2004).

Corals and shells were also collected for curio trade and this leads to selective damage to good looking or may be rare corals and shells. Corals of the orders; Scleractinia (stony corals), Stolonifera (Organ-pipe corals), Antipatharia (Black corals), Milleporina (Fire corals) and Stylasterina (Lace corals) are listed in Appendix II of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). This lists species in which trade is controlled through export and import quotas to reduce threats to their survival. Approximately 5,000 species of animals and 28,000 species of plants are protected by CITES against over-exploitation through international trade. The species covered by CITES are listed in three Appendices, (I, II and III) according to the degree of protection they need. Generally CITES prohibits commercial international trade in specimens of Appendices I, II. However, it may be allowed under exceptional circumstances, such as for scientific research (CITES, 1973). Appendix II of CITES lists nearly all of the species of the scleractinian coral families; Tubiporidae, Astrocoeniidae, Pocilloporidae, Acroporidae, Poritidae, Siderastreidae, Agariciidae, Fungiidae, Oculinidae, Mussidae, Merulinidae, Favidae, Trachyphylliidae, and Dendrophylliidae in Egypt's marine waters (CITES, 1973). The Appendix includes species that are not necessarily threatened with extinction, but in which trade must be controlled in order to avoid incompatible utilization with their survival. The convention is among the conservation agreements with the largest membership, with now 169 Parties.

1.5.4 Pollution and Eutrophication

Pollution including sewage and heavy metals may contribute to the collapse of reef ecosystems. A wide spectrum of factors such as chemical pollutants, oil, radioactivity, heavy metals, eutrophication, global climate change, sedimentation and marine debris; plus the probable synergistic effects of their combinations are all sources of deterioration to coral reef ecosystem (ICRS, 2004). Moreover, the line between marine use and abuse is a fine one, with many factors such as coastal and pelagic fisheries, mariculture, marine

mining, coastal engineering measures (beach replenishment, breakwaters) and tourism all contributing to the deterioration of marine ecosystem. Nutrients from agricultural areas or septic systems, as well as many other pollutants such as petroleum products, spills of insecticides also occur in parallel with coastal development. Outflows from desalination, sewage treatment plants and large power plants are examples of point source pollution, which are considered the cause of much damage to coral reefs in their locations (Jameson et al, 1998). As with all these factors, the basis for the continued degradation of coral reefs is the increasing size of the coastal human population, which in addition leads to the destructive harvest of marine resources for human uses.

Globally, sewage is a major component of marine pollution from land-based activities, which account for approximately three-fourths of all pollutants entering the world's oceans (Rapaport, 1995). Land-based sources of marine pollution are contributing to an alarming decline in the health of the world's marine ecosystems and their ability to provide for human needs. Sewage along with other forms of pollution from land-based activities is blamed for the decline and collapse of fisheries and tourism, and represents a severe threat to public health in various regions around the world (Rapaport, 1995). Although there are marine species that can thrive under relatively high metal concentrations the difference in species-specific response due to biochemical utility or toxicity may prove deleterious to other life stages in which low metal concentrations can significantly affect the fertilization process (Reichelt-Brushett and Harrison, 1999, 2000).

Another source of pollution results from introduction of alien species to the marine environment or what is called biological pollution. Biological pollution was defined as the effects of introduced, invasive species sufficient to disturb an individual (internal biological pollution by parasites or pathogens), a population (by genetical change) or a community (by increasing or decreasing the species complement); including the production of adverse economic consequence (Elliott, 2003). However, the terms biological pollutants and biological pollution have been used to discuss the problems caused by such invasive species (e.g. Boudouresque and Verlaque, 2002). As another form of biological pollution, genetic pollution may be regarded as occurring if the natural genetic structure has changed as the result of invasions (Elliott, 2003).

Three major anthropogenic factors are contributing to coral reef decline worldwide; these factors are eutrophication (reviewed in Koop et al., 2001), physical damage from either snorkelers and SCUBA divers or by boats anchoring and grounding (Chadwick- Furman, 1997; Hawkins and Roberts, 1997; Schleyer and Tomalin, 2000; Zakai and Chadwick-Furman, 2002), and sedimentation, which may be enhanced by anthropogenic activities (Loya, 1976; Rogers, 1990).

Stachowitsch (2003) stated that, eutrophication, in particular, is increasingly being recognized as the form of pollution most capable of leading to the collapse of entire coastal systems. The symptoms of eutrophication include increased turbidity, excessive plankton blooms, marine snow, oxygen deficiency, and mass mortalities of benthic organisms, typically in shallow seas with fine-grained sediments and water body stratification (Stachowitsch and Avcin, 1988). Eutrophication-related hypoxia and associated benthic mortalities have become more than the imagined ‘‘future nuisance’’ to a serious threat worldwide (Rosenberg, 1985). Eutrophication also promotes other phenomena such as harmful algal blooms (HABs) and their associated syndromes including ciguatera fish poisoning and paralytic, diarrheic, neurotoxic, and amnesic shellfish poisoning (GEOHAB, 2001).

Many studies have demonstrated the detrimental effects of anthropogenic input of excess nutrients (Smith et al. 1981; Tomascik & Sander, 1985; Cuet et al. 1988; Bell & Tomascik, 1993), and alterations in reefs from coral dominance to algae dominance have been attributed to eutrophication (Littler & Littler 1984). The majority of the world's coral reefs thrive in relatively nutrient-poor waters, although corals in aquaria can survive under high nutrient concentrations (Atkinson et al. 1995). Early studies demonstrated the link between eutrophication, which increases algal growth, and water-column anoxia and the devastating effects on coral reef ecosystem in the long-term. The well-established macro algae can outcompete and overgrow coral colonies (Birkeland, 1977; Hughes 1994; Jompa and McCook 2002). Holmes et al (2000) maintained that the levels of bioerosion in both live massive corals and in branching coral rubble were significantly higher on reefs subject to eutrophication than on reference reefs.

1.5.5 Coral Reef Predators

Many organisms feed on corals, either by selectively picking off the soft tissues or by crunching whole chunks of skeleton. Parrotfish are among the most important of the latter, and the specialized butterfly fish are important polyp feeders (Ormond, 1980). Crown of Thorns starfish (*Acanthaster planci*) is one of the famous coralivores, which digests coral tissues leaving bare skeleton. Outbreaks of Crown of Thorns starfish (eg. 1000 countable in 20 minutes) cause severe damage to coral reefs and change the nature of the reef from coral to algal dominated (reviewed in Alasdair and Head, 1987). In the Indo-Pacific reefs, Crown of Thorns starfish have done the equivalent damage to coral reefs as bleaching events (Veron and Stafford Smith, 2002). On the other hand coral compensate their tissue loss either by asexual reproduction by fragmentation or sexual reproduction by releasing; either large brooded planula larvae or many small eggs, which are fertilized and developed in the plankton (Fadlallah, 1983).

Crown of Thorns starfish (*Acanthaster planci*) and *drupella* gastropods, in "outbreak" situations have caused severe reef destruction on many reefs throughout the Pacific (reviewed in Jameson et al, 1998). Outbreaks *of Acanthaster planci* populations, under certain environmental conditions are known to cause up to a 90% decline in the scleractinian coral population (Sammarco, 1996; Souter and Linden, 2000).

A relationship between human activities and periodic outbreaks of the Crown of Thorns sea star (COTS) has been suggested (reviewed in Hodgson, 1999). These predators have had serious effects on the coral populations in many regions of the Pacific and its outbreakout have been linked to regions of increased development and eutrophication (Birkeland, 1989). New research further strengthens the evidence of the relation between higher outbreak frequencies of Crown of Thorns and terrestrial runoff; and recognized the effect of removal of Crown of Thorns predators in the likelihood of outbreaks (Fabricius, 2005). Crown of Thorns sea star (COTS) population outbreaks are assumed to be initiated by enhanced larval survivorship due to phytoplankton blooms on which the larvae feed. The phytoplankton blooms themselves may be the result of increased nutrient runoff from adjacent catchments due to anthropogenic modification (Devlin and Brodie, 2005).

Crown of Thorns starfish feed on coral tissues by extruding their stomach out onto the coral to digest the living tissue layer (Birkeland, 1989). Lethrinid fish feed on Crown of Thorns at biologically significant rates, although there is no evidence of a relationship to reproductive status of fish (Hugh, 1997). Many other predators such as fish and gastropods are also known to cause damage to coral colonies, but these generally do not weigh against the severe effects, which *Acanthaster planci* outbreaks have on coral populations.

1.5.6 Coral disease

Anthropogenic activities result in loss of diversity of corals and an increase in diseased or damaged corals (Dustan and Halas, 1987; Harvell et al., 1999; Porter et al., 1999). In addition, stressors including disease can cause bleaching (Westmacott et al, 2000). Diseases can lead to the death of parts of coral living tissue in a process called partial mortality (Nugues and Roberts, 2003a), and reduce coral diversity but not coral cover (Kuta and Richardson, 2002). In a recent study of coral disease, it was noted that 97% of coral disease in the Caribbean region was documented in reefs moderately to highly impacted by human activities (Green and Bruckner, 2000). Page (2004) maintained that diseases that affect both hard and soft corals have been found on reefs in many regions including the Caribbean, the Red Sea, the Pacific and the Indo-Pacific. Coral diseases are now known to affect more than 150 coral species in the Caribbean and Indo-Pacific reefs; and new diseases are continuously being added to the 29 already described (Wilkinson, 2004).

Accumulation of micro organisms has been detected at significantly elevated levels at the coral surface relative to the overlying water column, where overlying water column conditions may promote rapid die-off of bacteria and viruses from wastewater (Lipp et al., 2001). The study of these microorganisms on coral tissue could help in the detection of faecal contamination and the relation with coral disease itself. Lipp et al., (2002) hypothesized that the surface microlayer of coral heads (mucus) might accumulate microbial indicators of waste and human viruses, and could thereby provide more direct evidence for human impacts on reef environments. Coral diseases are caused by microbial pathogens including a variety of bacteria, algae, and fungi and often result in small amounts of coral mortality on healthy coral reefs. These diseases include shut-down-reaction, yellow band, yellow blotch, red band, rapid wasting disease, dark spot disease, white pox, and skeleton eroding band (Page, 2004). Correlations have been reported between increased sea temperature and the incidence of coral diseases. According to Ben-haim and Rosenberg (2002) at high water temperature (29ºC), coral tissue lysis begins as small white spots, rapidly spreading so that the entire tissue was destroyed, leaving only the intact bare skeleton. They noticed that coral disease was contagious and transfers from infected coral to the in direct contact healthy one.

Nutrient input, sedimentation, and runoff have also been implicated as potential factors that may promote the incidence of coral disease. For example, increase in black band disease activity on reefs that were close to sewage outflows or exposed to high sediment deposition; has been reported (Kuta and Richardson, 2002). Physical contact with the macroalga can trigger coral disease in the coral in many ways; the close proximity of algae may cause an abnormal physiological stress or trauma to corals that facilitates invasion by the pathogen; organisms associated with the alga may give off chemicals toxins to corals; and the alga itself may damage coral tissue through physical abrasion or shading or by allelochemical effects (Nugues et al., 2004).

1.5.7 Climate Change

The Intergovernmental Panel on Climate Change (IPCC) was established in 1988, to provide at regular intervals an assessment of the state of knowledge on climate change. The IPCC also prepares special reports and technical papers where independent scientific information and advice is deemed necessary and it supports the UN Framework Convention on Climate Change (UNFCCC) through its work on methodologies for National Greenhouse Gas Inventories (IPCC, 2001a).

Climate change as defined by the UNFCCC is a change of climate that is attributed directly or indirectly to human activity that alters the composition of the global atmosphere and which is additional to natural climate variability observed over comparable time periods (CBD, 1992a). Global mean surface temperature increased by 0.6°C over the 20th century which is greater than during any other century in the last 1,000 years (IPCC, 2001b). The warming rate since 1976 (0.17°C/decade) has been slightly larger than the rate of warming during the 1910 to 1945 period (0.14°C/decade) and temperature increases of 1–2°C in sea temperature can be expected by 2100 (IPCC, 2001b). Although these are seemingly small changes, they translate into an increased likelihood that, during the warmer periods of normal seasonal fluctuations, temperatures will exceed the tolerance levels of most coral species (Westmacott et al, 2000).

Human activities have increased the atmospheric concentrations of greenhouse gases and aerosols since the pre-industrial era. The atmospheric concentrations of key anthropogenic greenhouse gases (i.e., carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), and tropospheric ozone (O_3) reached their highest recorded levels in the 1990s, primarily due to the combustion of fossil fuels, agriculture, and land-use changes (IPCC, 2001b). Global climate change is likely to have six main impacts on coral reefs: sea level rise, temperature increase, reduced calcification rates, altered ocean circulation patterns, increased frequency of severe weather events and coral bleaching. (Westmacott et al, 2000).

The coral bleaching event in 1998 was a one in a 1000-year event in many regions with no past history of such damage in official government records or in the memories of traditional cultures of the affected coral reef countries. Increasing sea surface temperatures and CO_2 concentrations provide clear evidence of global climate change in the tropics, and current predictions are that the extreme events of 1998 will become more common in the next 50 years, i.e. massive global bleaching mortality will not be a 1/1000 year event in the future, but a regular event (Wilkinson, 2004). Large scale bleaching episodes can usually be attributed to fluctuations in sea surface temperatures (SSTs). If average temperatures continue to increase due to global climate change, corals will likely be subjected to more frequent and extreme bleaching events in the future. Thus, climate change may now be the single greatest threat to reefs worldwide (Westmacott et al., 2000). Nearly 100% of areas with coral reefs should experience bleaching when the regional SST inconsistency increases to just over +0.9°C, which is close to the +1.0°C threshold frequently reported as the critical limit for coral reefs (Wilkinson, 2004). Human activities are causing the acceleration of global climate change to rates that may make it difficult for coral reefs to adapt (Westmacott et al., 2000).

1.**5.7.1 Coral bleaching**

Scleractinian corals have a narrow range of temperatures, within which they can survive, grow and reproduce (22-28°C). When exposed to water temperatures above (or sometimes below) this range, many coral species bleach (lose their zooxanthellae and some of their tissue pigments) (Sammarco, 1996). Experiments and observations indicate that coral bleaching results primarily from elevated seawater temperatures under high light conditions. An increase in coral bleaching has been noted on a global basis and this is believed to be due to global warming (Sammarco, 1996). The majority of large-scale coral bleaching episodes over the last two decades have been linked to the presence of increased sea surface temperatures (SSTs) (Wilkinson, 2004).

Thermal stress results in a general weakening of the coral animal, its tissues are probably starved and then die due to reduced translocation of photoassimilates, which in turn is

caused by a decrease in photosynthesis by the algal symbionts (Fine et al., 2002). Hoegh-Guldberg (1999) maintains that coral mortality as a result of El Nino event was 80–90% on some parts of the Great Barrier Reef, and some of the corals killed were at least 700 years old. Simply when reef-building corals are exposed to abnormally high temperatures or other stresses, they turn white because they lose their zooxanthellae (photosynthetic microalgae). These bleached corals die when thermal stresses are prolonged and intense, in part because zooxanthellae photosynthesis contributes most of the carbon budget of the coral (Sotka and Thacker, 2005). Bleaching is reversible when the increases are short-term and of no more than 1-2°C. However, sustained increases in water temperatures of 3-4°C above normal maxima can cause significant coral mortality (CBD, 2001).

It has been recognized that other factors besides elevated SSTs could be involved in coral bleaching, such as wind, cloud cover and rainfall, although climate change may now be the single greatest threat to reefs worldwide (Wilkinson, 2004). The influence of irradiance on the density of zooxanthellae (number of cells.cm^{-2} of coral tissue) has been examined in a number of experimental studies. Two transplantation studies reported significant changes in the density of zooxanthellae with a decrease in depth (Dustan, 1979; Gleason and Wellington, 1993). Dustan (1979) reported a decrease in density with an increase in depth. Gleason and Wellington (1993) however, observed a decrease in the density of zooxanthellae in *Montastrea annularis* 21 days after transfer to shallower waters, and experimentally demonstrated that this decrease was specifically in response to increased UV radiation. In the laboratory, bleaching can be triggered by multiple factors: extremes of temperature (heat shock and cold shock), high irradiance, prolonged darkness, heavy metals and pathogenic microorganisms (Douglas, 2003).

Corals which appear completely bleached to the naked eye have experienced a 70 to >90% reduction in zooxanthellae density (Douglas, 2003). The long-term capacity of reef corals to survive bleaching episodes is likely to be dependent, at least initially on the diversity and specific identities of the symbiotic dinoflagellates 'zooxanthellae' they contain (Baker et al, 2004). The four major clades of *Symbiodinium* (labeled A, B, C, and D) that dominate all hard corals, are differentially adapted to high temperature stress. Although there is variation in environmental resistance among strains within a single clade (especially clade C), *Symbiodinium* from clades A and D tend to associate with corals in shallow, thermally stressful environments, whereas those from clade C are rare in such environments (Sotka and Thacker, 2005). In an experiment of the heat tolerance of *Symbiodinium* C and D in the host coral *Pocillopora verrucosa*; Rowan (2004) exposed coral to increases in water temperature and used chlorophyll fluorescence to measure changes in photosynthetic capacity. The results showed that quantum yield of photosystem II in *Symbiodinium* C decreased when water temperatures were raised from 28.5°C to 32°C. After returning water temperatures to 28.5°C for four days, it remained at the lower value. By contrast, the quantum yield of *Symbiodinium* D increased as temperature increased and reduced at the lower temperature. These data suggest that there are costs for corals that host *Symbiodinium* D at low temperatures and for corals that host *Symbiodinium* C at high temperatures (Sotka and Thacker, 2005). Complete recovery of reefs following severe bleaching is dependent on growth and fragmentation of remaining corals, and on recruitment from stocks in the area. The study of global climate change impacts on coral reefs and the need to reduce global emissions of greenhouse gases has been added as a major action to conserve coral reefs (Jakarta Mandate, 1995).

Buddemeier and Fautin (1993) proposed that bleaching is an ecologically risky but adaptive strategy of the animal partner to replace an inferior symbiont by an alternative superior form in a process called adaptive bleaching. The Adaptive Bleaching Hypothesis

(ABH) suggests that frequent and severe environmental stresses tend to favor stress resistant coral–zooxanthellae associations. The hypothesis predict that: symbionts acquired after bleaching provide greater heat tolerance than do those present before bleaching; and heat-sensitive coral–zooxanthellae associations have a competitive advantage in the absence of heat stress; that is, an association that is heat tolerant is also costly to the coral (Sotka and Thacker, 2005). This hypothesis explains the persistence of deleterious bleaching in *Symbiodinium* symbioses. It presupposes three processes; the displaced symbiont is inferior to the invading symbiont; Replacement of a resident *Symbiodinium* population in an animal requires depletion of the population; or an alternative (putatively superior) strain of *Symbiodinium* is available to populate the bleached animal host (reviewed in Douglas, 2003).

1.5.7.2 Storm Frequency and Intensity

Although much of the degradation to coral reefs is directly related to human impact, there are several natural disturbances that cause significant damage to coral reefs. The clearest example is hurricanes, or typhoons, which strike the tropics. These storms cause physical damage to fragile coral colonies, the reefs and coastal communities (Westmacott et al., 2000). In addition, these storms generally bring heavy rain, which increases runoff and sedimentation. Alterations to annual atmospheric patterns could result in changes in the frequency and intensity of storms and cyclones.

Alterations to annual atmospheric patterns, as a consequence of climate change, could result in changes in the frequency and intensity of storms and cyclones, which could cause increased damage not only to coral reefs but to coastal communities as well (Westmacott et al., 2000). Analyses of wave buoy data along the entire west coast of North America demonstrate that the heights of storm-generated waves have increased significantly during the past 3 decades (IPCC, 2001b). It is predicted that tropical storms could increase in frequency and severity, and that the major global ocean currents may change (Wilkinson, 2000b). Sea level will rise by 0.09–0.88 m between 1990 and 2100, higher mean sea level will increase the frequency of existing extreme levels associated with storm waves and surges (IPCC, 2001b).

1.5.7.3 Increased El-Nino Events

The El Nino-Southern Oscillation (ENSO) is a natural part of the Earth's climate, which might undergo changes in intensity and frequency as a result of global warming (Timmermann et al., 1999). The frequency and intensity of ENSO has been unusual since the mid-1970s compared with the previous 100 years. Warm phase ENSO episodes have been relatively more frequented, persistent, or intense than the opposite cold phase during this period (IPCC, 2001b). This recent behavior of ENSO is related to variations in precipitation and temperature over much of the global tropics and subtropics and some mid-latitude areas. The overall effect is likely to have made a small contribution to the increase in global surface temperature during the last few decades. Timmermann et al. (1999) suggested an increased frequency of El Nino-like conditions under future greenhouse warming and stronger "cold events" in the tropical Pacific Ocean. If temperature differences between the tropics and Polar Regions are reduced, however, a weakening of the atmospheric circulation patterns that cause upwelling could be expected. An association between CO_2 variability and El Nino in particular has been reported for over twenty years and has been confirmed by recent data analyses (IPCC, 2001b).

El Nino events result in extensive coral bleaching and mortality over large parts of the Indian Ocean and southeast and East Asia. The 1998 global coral bleaching event effectively destroyed 16% of the world's coral reefs, with most damage throughout the Indian Ocean and the Western Pacific (Wilkinson, 2000b). On some reefs, there were mortality levels greater than 90% leaving them almost bare of corals and with early indications of major shifts in the population structures (CBD, 2001a). Sea temperature increases associated with ENSO events have been implicated in reproductive failure in seabirds (IPCC, 2001b). In addition, changes in large-scale ocean circulation patterns could alter the dispersal and transport of coral larvae; this could have impacts on the development and distribution of reefs worldwide (Westmacott et al., 2000)

1.5.7.4 Sea Level Rise

Sea level is estimated to increase by 50cm by the year 2100 (IPCC, 2001b). Reef flats that are exposed at low water may benefit from such a rise. However low-lying islands will no longer be afforded the protection from wave energy and storm surges that their surrounding coral reefs currently provide (Westmacott et al., 2000). 20th century global warming has contributed significantly to observed sea-level rise, through thermal expansion of seawater and widespread loss of land ice. Global mean sea level has risen by about 0.1-0.2mm.yr^{-1} over the past 3,000 years and by 1-2mm yr^{-1} since 1900, with a mean value of 1.5mm yr^{1} (IPCC, 2001b).

Coastal zones under global warming and sea-level rise, will suffer from increased levels of inundation and storm flooding, increased coastal erosion, seawater intrusion into fresh groundwater, intrusion of tidal waters into estuaries and river systems and elevated sea-surface and ground temperatures (IPCC, 2001b). The impact of accelerated sea-level rise in coastal areas will depend on vertical buildup rates and space for horizontal migration. Over the past 100 years or so, about 70% of the world's sandy shorelines have been retreating, about 20–30% has been stable, and less than 10% have been advancing (IPCC, 2001b). Bird (1993) argues that with global warming and sea-level rise there will be tendencies for currently eroding shorelines to erode further, stable shorelines to begin to erode, and shorelines buildup to decline or stabilize. Sea-level rise could have dramatic changes in the reef-top zonation of corals, other biota, and abiotic reef substrata because these are strongly controlled by the energy of the waves that pass over the reef (Done, 1983). The tendency would be for the plunging point for waves to gradually move toward the back of the reef, causing changes in the benthic zonation at the new location of the breaker zone and for several metres to tens of metres either side (IPCC, 2001b).

1.6 Other Sources of Impact, Red Sea Specific

The coral reef system of the Red Sea supports an exceptional biodiversity, and is particularly vulnerable to the effects of pollution in many locations due to inland source of sediment (reviewed in Alasdair and Head, 1987). Oil pollution in the Red Sea does not appear to pose significant threat to coastlines at present and the most serious threat comes from passing tankers discharging ballast or oil. Up to 100 million tonnes per annum may pass through the area and more than 20 oil spills from tankers have occurred along the Red Sea in the period 1982-2000 (PERSGA, 2003). The oil companies operating in the area carry out their own investigation of surrounding seawater and beaches for any hydrocarbon content of their products and have contingency plans in the event of a spill or accident (reviewed in Hariri et al, 2000). Chronic oil pollution has already been observed

in the immediate vicinity of some major Red Sea ports as a result of operations at oil terminals or discharges from power plants in the area (Gerges, 2002). Oil may damage the reproductive system of corals, interfere with production of larvae or reduce larval viability and inhibit their normal settling (Alasdair and Head, 1987).

Desalination plants increase salinity levels by discharging warm brine. They also release maintenance chemicals at their discharge points (Wilkinson, 2000a; Gladstone et al, 1999). All desalination plants in the area dispose the rejected brine water back to sea either through onshore wells or directly to near shore waters. This brine effluent is of high salinity and temperature, both potentially damaging to the marine environment close to the outfalls (Awad and Shabara, 2003). Selected chemicals were estimated for 21 desalination plant locations in the Red Sea including the Gulf of Aqaba and Gulf of Suez. Their combined capacity exceeds 1.5million m^3/d, the daily chemical discharge amounts about 2,708 kg chlorine, 36kg copper and 9,478kg antiscalants, when effluent concentrations of 0.25ppm, 0.015ppm and 2ppm are assumed, respectively (Hoepner and Lattemann, 2002). The results indicate that low concentrations of chlorine can have adverse effects on aquatic life as it is a highly effective biocide and it reacts with organic compounds in seawater producing an immense number of chlorinated and halogenated by-products, many of which are carcinogenic or otherwise harmful to aquatic life (Hoepner and Lattemann, 2002).

Copper might only become toxic if excess amounts become biologically available; a low brine contamination level of 15ppb reduces the risk of toxic conditions for aquatic life. The daily load of 9.4 ton antiscalants seems high and far above the typical dosage in desalination plants, but the environmental risk of these substances is relatively low compared to chlorine and copper. Whilst antiscalants have a relatively low toxicity and their environmental fate is characterized by dilution, which further reduces any risk of negative effects; they have high metal ion binding capacity which reduce the biologically essential trace metal ions (reviewed in Hoepner and Lattemann, 2002).

In the Egyptian part of the Red Sea coast there are no point sources of land based sewage pollution. However non-point sources continue to occur from picnic boats using the area and affect some of the inland in low circulation bays. Most hotels and resorts either had build treatment plants of their own, or had other arrangements in place for sewage treatment off-site, often with disposal in designated open desert areas (Awad and Shabara, 2003).

Red Sea reefs also receive significant damage from both commercial and private vessels crossing coastal and offshore waters. These impacts include the leakage of fuel into the water, sewage discharge and the occurrences of oil spills by large tankers which are extremely dangerous to local reefs and islands beaches (Pilcher and Abuzaid, 2004). Boats also harm reefs indirectly by anti-fouling bottom paints which form toxic compounds harmful to corals and other reef creatures especially in the coastal low circulation area as in boat docks and marinas (Wilkinson, 2000a). Increased boating and sailing activities in coastal waters of Egypt has resulted in large amounts of plastic debris, which are suspended in the water column or sink to the bottom; and entangle coral colonies.

The Suez Canal provides an additional probability of threat to the Red Sea fauna and flora. Historically, it was first opened in 1869 and provides a direct connection between the Red Sea and the Mediterranean Sea. Such a connection allows some species, which succeed in crossing the salinity barrier, to invade new areas where they have not been previously recorded. Thus far, there has been a quite considerable flow of species from the Red Sea into the Mediterranean, but relatively few have made the reverse migration, and their

impacts on reefs are insignificant. Por (1978) recorded 128 species that have migrated from the Red Sea to the Mediterranean, and about 53 species have been suggested to migrate in the opposite direction. One negative aspects of such immigration, was the transport of parasites by their fish-hosts: the fish *Siganus rivulatus* and *S. luridus* have transported into the Mediterranean endo- and ectoparasites not formerly known from this sea (Diamant, 1996).

1.6.1 Conservation of the Red Sea Coral Reefs

The Red Sea and Gulf of Aden contain complex and unique tropical marine ecosystems, especially coral reefs, with high biological diversity and high endemism (Wilkinson, 2000a). The concentrations of small-range endemics in small area are called hotspots which are considered the major centres of endemism. The stable warm waters and lack of major fresh water runoff provide ideal conditions for coral reefs; there are more than 250 species of hard (scleractinian) corals in the Red Sea, the region being the highest diversity in any part of the Indian Ocean (Wilkinson, 2000a). Coral endemics with ranges $<5x10^5$ km^2 are numerous only in the Red Sea and Hawaii, whereas endemic fishes are distributed much more widely (Hughes et al, 2002).

Recent concerns about loss of biodiversity have led to calls for the preservation of hotspots as a priority; which is partly economic as protecting hotspots may be the most cost effective way to protect large numbers of species (Hughes et al, 2002). The major part of Egypt's coral reefs are protected according to Law Number 102 of 1983, including all those in the Gulf of Aqaba and all the fringing reefs around the Red Sea islands. The Red Sea Islands, coastal mangrove swamps and southern part of the coastal reefs was declared a protected area in 1986 as a part of the Elba protected area (Figure 1.1). Egypt future plan aims to increase the number of natural protectorates to 40 (EEAA, 1986). Law No 4/1994 set the rules for development including a set back area of 200 metres from the high water mark. Although, Environmental Impact Assessments (EIA) are now mandatory for any sort of development, to ensure environmental sustainability, the early land filling operations still contribute the major part of sediment deposition in the Hurghada region (Mansour et al, 1997).

Figure 1.1: The Red Sea, Gulf of Aqaba and Gulf of Suez, showing the Egyptian coastal area and islands.

Early studies indicate that the use of protected areas as a tool for conservation of coral reefs is inadequate to stop further damage even if they are supported by varying legislation. The reason is that often those conventional management plans attract more visitors, increase the accessibility to those areas and strongly enhance the impacts of tourism on reef habitats (reviewed in Rinkevich, 1995). The role of protected areas in conserving and enhancing fish stocks has received far greater attention; particularly in coral reef environments where increases of stocks inside protected areas have been demonstrated at several locations (Roberts et al., 2001).

1.7 Sedimentation Impacts on Reefs

1.7.1 Development and Sedimentation

Human population expansion and subsequent development of coastal areas is one of the greatest threats to coral reefs (Fabricius, 2005). The rate of sediment release into the

oceans is increasing, as more coastal lands are developed to accommodate rising urban populations and increases in agriculture (Wilkinson, 2004). In coastal areas, development continues to alter the landscape, coast and shore contour by dredging, land fill and artificial construction that enhances sediment input from land-clearing areas, and modified beaches. Coastal modification may also interfere with dominant water currents and lead to erosion and sedimentation in many sites.

At a global level, sedimentation is considered as a major cause of coral reef degradation worldwide (Ginsburg, 1994; Wilkinson, 2000a; Burke et al. 2002). Rogers (1990) maintain that sedimentation can affect corals in several ways; it can cause their death by smothering or burial; it can decrease adult coral growth by abrasion and shading; it can depress zooxanthellae densities and photosynthetic activity, and increase respiration and mucus production and it can reduce coral reproduction, coral larval settlement, and early survival. These effects are considered the most commonly reported by researchers; and they are thought to act together to limit coral reef development and distribution (Cortes and Risk, 1985; Hubbard, 1986). High sediment precipitation causes partial mortality in large colonies and full death of small colonies of reef corals. Runoff and sedimentation may prevent successful reproduction and recruitment in corals and other reef organisms (Richmond, 1994). Abundance and richness of zooxanthellate octocorals are negatively correlated with turbidity and low visibility (reviewed in Fabricius and McCorry, 2006). The consequences of sedimentation on reef organisms vary dramatically depending on grain size, exposure time and sediment composition; a recent study has shown that some coral species are more sensitive to fine grained sediments compared to coarser sediments (Harrington, et al., 2005). Coral reef communities respond variably to sedimentation and thus sedimentation effects should be evaluated on a case-by-case basis using a number of community-level variables (Brown, 1997).

The proposed sedimentation threshold rate is 10 $mg.cm^{-2}$ which have been expected to inhibit coral reefs growth and diversity (Rogers, 1990). The magnitude of sediment impact depends on the size of sediment particles, the intensity of load, the rate of sediment dispersal, the presence of other stressors, the geographic setting and coral community composition (reviewed in Dikou and van Woesik, 2005).

Coral damage appears to not only depend on the amount and duration of sedimentation, but also strongly depends on the sediment type, for example, tissue damage under a layer of sediment increases with increasing organic content and bacterial activity, and with decreasing grain sizes (Fabricius, 2005). Nugues and Roberts (2003a) suggest that the composition of bottom sediments in term of particle size may be more critical to sediment effects on corals than sedimentation rates. Removal rates depended on the sediment properties; sandy sediments were removed more efficiently than silty sediments, being rejected about three to four times more effectively than the nutrient-rich silts (Weber et al., 2006). Corals developed sediment removal mechanisms which can minimize sedimentation damage. Those mechanisms include mucus secretion, ciliary movements and capture by tentacles.

A few studies have addressed the hypothesis that variable sediment regimes represent variation in heterotrophic food resources. Hermatypic scleractinian corals function as phototrophs as well as heterotrophs (Anthony, 2000). They are able to capture and ingest a wide range of food types, including large sediment particles (Stafford-Smith and Ormond, 1992) and fine suspended particulate matter (Lewis, 1977; Anthony, 1999). Although phototrophy by reef corals may be impaired by shading under elevated particle concentrations, heterotrophy will potentially be enhanced by increased availability of particulate food (Dallmeyer et al. 1982; Te, 1997).

Despite high turbidity levels and sedimentation rates, which often exceed those described as lethal for corals, inshore reefs in the GBR lagoon generally sustain high coral cover and diversity suggesting that local adaptation to intense sediment regimes has occurred (e.g. Ayling and Ayling, 1991). In a study by Anthony (1999), results showed that scleractinian reef corals may achieve up to 50% of their predicted tissue growth by feeding on fine suspended sediment at high sediment concentrations (30mg dry weight). This suggests that corals in highly turbid conditions should make optimal use of the high particle availability despite its relatively low food value. This prediction is based on the idea that the rejection of an abundant, low quality food source is costly (reviewed in Hughes, 1980). Where anthropogenic eutrophication and enhanced sedimentation occur together, nutrient loading may boost coral metabolism and skeletogenesis compensating for reduced light penetration from high sedimentation, and yielding no net change in coral growth rates (reviewed in Edinger et al, 2000)

1.7.2 Sedimentation and Coral Mucus Secretion

Mucus production could be a protective mechanism for coral, since it often removes silt, prevents the settling of other organisms and perhaps prevents parasitism and predation (Small and Adey, 2001). Many studies refer to mucus secretion in coral as an important strategy to withstand sediment stress. Richman et al. (1975) suggested that the rate of mucus production by massive forms are much greater than branched ones, supporting the suggestion that massive corals use a mucus secretion mechanism to prevent burial in heavily sedimented reefs. Loya (1972) suggested that the branching species of *Acropora* are predominant in high sedimentation areas because they are less likely to be buried and the massive forms are likely to be higher mucus producers in order to withstand heavy sedimentation load. This leads the notion that different species also produce mucus at different rates in response to sedimentation stress.

During laboratory experiments on the effects of sediment on coral reefs, increased mucus production by *Montipora peltiformis* was observed (Weber et al., 2006). Some coral groups are abundant in turbid environments, although they have a low capacity to reject and get rid of sediment. For example, *Porites* may survive because of its ability to tolerate sediment deposition through heavy mucus secretion (reviewed in Dikou and van Woesik, 2006). The efficiency of sediment clearance by corals is determined by: size and shape of the entire colony, colony surface (rugged, smooth) including morphology of the calyx, a coral's behavior, and grain size distribution of the sediment, with mud being more deleterious than sand (reviewed in Diethard and Baron-Szabo, 2005).

Mucus secretion could also take place because of other reasons; it could be sign of other stresses such as warming and higher solar irradiance (Jokiel and Brown, 2004). Corals under minimum access to nitrogen and phosphorus (from captured zooplankton), cannot build sufficient biomass to match the glycerol received from zooxanthellae and are suggested to excrete the excess carbon as mucus (Small and Adey, 2001).

1.7.3 Sedimentation and Algal growth

Filamentous turf algae are commonly associated with relatively pristine reef systems and are a major contributor to their high productivity (Klumpp and McKinnon 1989). However, it can also persist in high sediment conditions where corals and other algal groups become rare (Sousa et al.1981; Umar et al. 1998). Their dominance in high sediment areas has been particularly attributed to their ability to accumulate large amounts of fine sediment by acting as sediment traps (Sousa et al.1981; Seapy and Littler 1982;

Stewart, 1989). This can indirectly increase coral mortality even further and can facilitate algal overgrowth onto coral colonies and increase competition (Dustan, 1994; Potts, 1977; Walker and Ormond 1982; Brown and Howard, 1985). By trapping sediments, turf algae have also been reported to reduce the cover of important settlement substrata for coral juveniles (Kendrick 1991; Belliveau and Paul, 2002). Coral–algal interactions are one aspect of sediment effects on corals that have been largely overlooked due to focusing on corals relative to other components of the benthos (Nugues and Roberts, 2003b). Higher rates of sedimentation can act directly and lead to increased macroalgal abundance, by enhancing macroalgal recruitment or survival; or indirectly by inhibiting competitors (e.g. corals) or herbivores (e.g. fish). Both direct and indirect impacts could involve effects of either suspended sediment (turbidity) or sediment deposition on the substratum.

Degradation or even simple disturbance of coral reefs, generally involves increased dominance by benthic algae, which are likely to have critical effects on coral settlement and recruitment, a recent study found that coral recruits were more often found in close proximity to filamentous algal turfs than any other major benthic group (Birrell et al., 2005).

1.7.4 Sedimentation and Reef Bioerosion

Reef calcium carbonate may be lost either through biological or physical processes. Biological mechanisms include both external erosion, e.g. the scraping and feeding actions of grazing fish such as scarids and acanthurids, and internal erosion caused by the colonization and growth of endolithic organisms such as bivalves, sponges, sipunculans, polychaetes and pyrgomatine barnacles (Sammarco, 1996). The persistence of coral reefs results from a delicate balance between construction and erosion, corals and calcareous algae contribute to reef building via biomineralisation, while macroborers, microborers, and grazers are the dominant agents of carbonate removal (Tribollet and Payri, 2001). In disturbed reefs, rates of bioerosion by macroborers exceed rates of reef growth leading to reef collapse (Hutchings and Peyrot-Clausade, 2002). In such situations bioerosion may be expected to weaken the framework of the reef, making it increasingly susceptible to damage from storm waves (Sammarco, 1996).

Macroborer groups include sponges, polychaete and sipunculan worms, bivalves, crustaceans, and Foraminifera. Modern boring communities (i.e. assemblages comprised of sponges, bivalves, and worms), are reported from reefs of the Miocene, Oligocene and even Jurassic age (reviewed in Dustan, 1994). Macroboring organisms are ubiquitous within a modern coral reef framework (Bromley, 1970; Macintyre, 1984; Hutchings, 1986). These groups are all readily found when breaking up reef rock and have been found to be useful indicators of sewage pollution (Brock and smith, 1983). In a study to assess the distribution of macro boring species and the degree of framework infestation, Perry (1998) found great variations in the distribution across the reefs. Relative abundances of the main groups of macroborers (sponges, bivalves, worms) illustrated clear distributional trends, sponges were dominant at fore-reef clear water sites, while sipunculan and polychaete worms are only of importance at back-reef, turbid lagoons and shallow fore-reef sites (Perry, 1998). Macdonald & Perry (2003) maintained that, increasing sedimentation rate alters the bioeroding community structure. In recent clear-water reefs, the most common macroborers are clionid sponges followed by lithophagid bivalves and by worms, and to less extent barnacles and shrimps (Sanders and Baron-Szabob, 2005).

Macroborer infestations and the enhanced effect of sedimentation is one of the parameters suggested to have a significant effect on reef degradation. Holmes et al (2000) maintain

that levels of bioerosion in both live massive corals and in branching coral rubble were significantly higher on reefs subject to eutrophication than on reference reefs. Disturbed, more polluted sites showed higher levels of macroborer infestations including sponges and polychaetes (Zubia and Perot-clausade, 2001). The increased amount of dead coral resulting from anthropogenic impacts increases substrate available for colonization by both macro- and microborers (Hutchings and Peyrot-Clausade, 2002). Coral rubble bioerosion was more sensitive to low levels of eutrophication and sedimentation stress than was massive coral bioerosion and can be used as a general indicator of eutrophication stress on coral reefs (Holmes et al., 2000).

Macroborer species display niche preferences, such as a preference for either live or dead substrates (Scofin and Bradshaw, 2000) and there is evidence that some coral species exhibit varying degrees of susceptibility to macroboring (Perry, 1998). The polychaete *Spirobranchus giganteus* is abundant in living coral, especially *Porites solida* and *Millipora sp*; while sponges of the genus *Cliona* live within rock or coral substrate, boring by chemical action and creating significant quantities of fine sediment (Futterer, 1974). *Cliona* is a major factor in the breakdown of reef rock and by burrowing into the coral skeleton can seriously weaken the structure and cause the death of the coral by fracture (Chamberlain, 1978). Polychaetes are a highly diverse and ecologically important group of macroborers abundant in shallow environment. Amoureux (1983) lists 81 species of Polychaetes from rock, coral and algal substrates in the Red Sea and Gulf of Aqaba.

Modes of substratum modification vary between borer groups from chemical dissolution of substrate with no sediment input as in the bivalve *Lithophaga sp*. (Kleeman, 1990), to physical borers like *clionid* sponges that produce fine-grained sediment. Macroboring organisms play a significant role in both; the direct dissolution and degradation of in-situ and rubble reef carbonate, and the production of abundant fine-grained sediment (reviewed in Perry, 1998). Bioeroding processes are implicated as a significant factor that influences styles of reef framework preservation; they promote both the removal of primary reef framework and may result in degradation of specific coral morphologies (Perry, 1998). In addition, erosion may influence a reef's structural integrity and thus increase the potential for wave over-topping (reviewed in Macdonald and Perry, 2003). Macroboring is, therefore, a significant factor influencing reef development, framework preservation, and carbonate accumulation (Edinger and Risk 1992; Perry, 1999).

1.7.5 Sedimentation and Coral Zooxanthellae Density

Stimson (1997) maintained that the density of zooxanthellae cells in the tissue of hermatypic coral is generally around 1×10^{6}cells.cm^{-2}, with considerable variation around this figure. Zooxanthellae density is presumably the net result of many factors such as; cell division, cell release from coral host, colonization of newly produced coral tissue, and possibly cell digestion (reviewed in Stimson, 1997). It is not constant throughout the year and varies according to; irradiance level, UV radiation, NH_4 enrichment and also elevated temperature can cause a zooxanthellae decrease in coral tissue (reviewed in Stimson, 1997).

The role of the zooxanthellae in the symbiotic relationship is to provide their coral hosts with a major part of their excess carbon, though most of this may ultimately be released as mucus (Small and Adey, 2001). When coral animals are stressed they expel the algae, causing them to bleach. Corals can survive short-term bleaching, but prolonged severe events are fatal for them. In a laboratory study of the effect of sediment on coral it was found that some species initially appeared unaffected directly after sediment exposure but

developed visible signs of bleaching within 24 to 48 hrs of recovery, possibly from the expulsion of damaged zooxanthellae (Weber et al, 2006).

The experience of the last two decades suggests that bleaching is generally deleterious to corals and other *Symbiodinium* symbioses and to reef communities (Douglas, 2003). Elevated nutrient levels can result in high phytoplankton biomass, reducing light intensity and altering spectral quality, thus influencing zooxanthellae photosynthesis essential for normal coral physiology (Winkler et al, 2004). Photosynthetic rates of zooxanthellae from well-fed corals were up to 1.7 times greater than those of zooxanthellae from starved corals which indicate nitrogen deficiency and reduced light- saturated photosynthesis (Simon et al, 2006). The effect of sediments on the photophysiological yield in corals increased with increasing concentrations of organic and nutrient-related matter in the sediment (Weber et al, 2006). Although this indicates a significant role for nutrients for photosynthesis, excess nutrients may upset the internal balance between the animal coral and its zooxanthellae further limiting calcification rate (Small and Adey, 2001).

1.7.6 Sediment Characterization

Important distinguishing characteristics of sediments include; chemistry and mineralogy (reflects origin), grain size (reflects energetics of transport, deposition) and degree of compaction and cementation (McDuff, 2001). Shallow water sediment of the Red Sea coast are a mixture of carbonates and non-carbonate minerals. Carbonate sediments are derived from the disintegration of recent and fossilized coral reefs and the associated communities; while non-carbonate sediments are derived from neighboring mountains (Abd-Elwahab, 1996).

The internal structures, the textures, and the composition of sediment may reflect interactions with the biosphere and lithosphere (Lewis and McConchie, 1994). In a study of the intertidal and seagrass bottom sediment of the Red Sea coast between Marsa Alam and El Gemsha (335km); Mansour et al (1997) found a high percentage of quartz and rock fragments which increased towards the north, ranging from 45% at Marsa Alam to 99% at El Gemsha. It varied from 63% at the intertidal zone to 56% in the seagrass bottom and its abundance was due to the incidence of nearby basement rock. This indicates that non-carbonates are mainly quartz, plagioclase was the next most predominant, whilst feldspars were very rare. Piller (1994) has categorized sediment constituents in the coastal area of the Red Sea into two main categories; 1- carbonate grains which include, foraminifera, sponges, corals, bryozoa, mollusks, worm tubes, crustaceans, ostracods, vertebrates, red algae, green algae, pellets and cryptocrystalline grains; and 2- non-carbonate grains which includes quartz, feldspar and rock fragments.

1.8 Coral Regeneration and Recovery

Coral species differ in morphology, size, depth distribution, regeneration capacity, recruitment, attractiveness to predators, and susceptibility to disease and environmental stress. This variation suggests interspecific and environment-related differences in vulnerability of corals (Meesters et al., 1997). Resilience in coral reefs and regeneration appears to be dependent upon the frequency and severity of the perturbation experienced. For example, repeated oil spills at Eilat in the Red Sea and in Panama have resulted in decimated reefs with a very slow recovery for scleractinian corals (Sammarco, 1996). Large scale impacts by several densely spaced blasts over a large portion of reef can

totally alter ecological and even environmental parameters and thereby make natural regeneration almost impossible (Riegl and Luke, 1998). Multiple stressors or chronic anthropogenic impacts such as blast fishing weaken the reef's ability to recover (Hughes and Connell, 1999).

Coral reefs are faced with many sources of disturbance and their resilience depends on the successful settlement and survival of reef dwelling corals (Harrington et al, 2005). Unfavorable larval transport, insufficient suitable substrate, low natural recruitment rates, and continued disturbance from illegal fishing activity may all impede reef regeneration (Brooke and Young, 2003). Coral recruitment is directly vulnerable to the effects of sedimentation and pollution. Crustose coralline algae (CCA) is thought to play a key role in facilitating coral recruitment, and live CCA is significantly more effective in inducing coral larvae to settle than dead CCA. Sediments have the potential to indirectly affect reef recovery and regeneration through covering CCA with a sediment barrier that reduces coral larvae settlement (reviewed in Harrington et al, 2005).

Large coral colonies have the ability to survive the death of parts of their living tissue, commonly known as partial mortality, the living parts of tissue grow again and repair the wound through the regeneration of tissue and skeleton (reviewed in Nugues and Roberts, 2003a). Partial mortality is seen more often than whole colony death in reefs under high sediment load and is responsible for most loss of living tissue by coral populations (Hughes and Jackson, 1985). If lesions cannot be regenerated, colony size is reduced and survival chances decrease (Meesters et al., 1997). Partial mortality patterns are influenced by many factors such as colony size, (total colony mortality is known to increase sharply with decreasing colony size), morphology, depth and species (Nugues and Roberts, 2003a). Juvenile colonies often cannot recover from partial mortality and subsequently die, possibly because the energy for regeneration is probably very limited in small colonies (Meesters et al., 1994). Fragmentation wounds in smaller coral pieces have larger surface: volume ratios than those in large fragments and thus, are energetically more costly to repair (Dizon and Yap, 2006). However, colony fission by mortality has some advantages: it increases the chances of long-term colony survival in many species and has an important role in reproduction and colonization of space in some species (Nugues and Roberts, 2003a).

Natural regeneration via the growth *in situ* of naturally occurring propagules is the best option for restoration with artificial regeneration a potential alternative if this appears to be inadequate (Yap, 2000). The natural recovery and regeneration of disturbed areas of reef is dependent on the influx of coral larvae that settle and grow. The number of larvae a disturbed reef will receive will depend on its occurrence with other undisturbed reefs (Souter and Linden, 2000). Coral recovery varies from as rapidly as five years in some damaged habitat to about 40-50 years or longer to retain its original states. In addition, the recovery process depends on the availability of stable and complex substrate and species assemblages adapted to high disturbance regimes (Dollar and Tribble, 1993). In the context of a reef's life span of thousands of years, this time frame is not considered a long time. Branching coral forms are expected to have the highest regeneration ability, although there seems to be a size threshold for this ability. (Dizon and Yap, 2006).

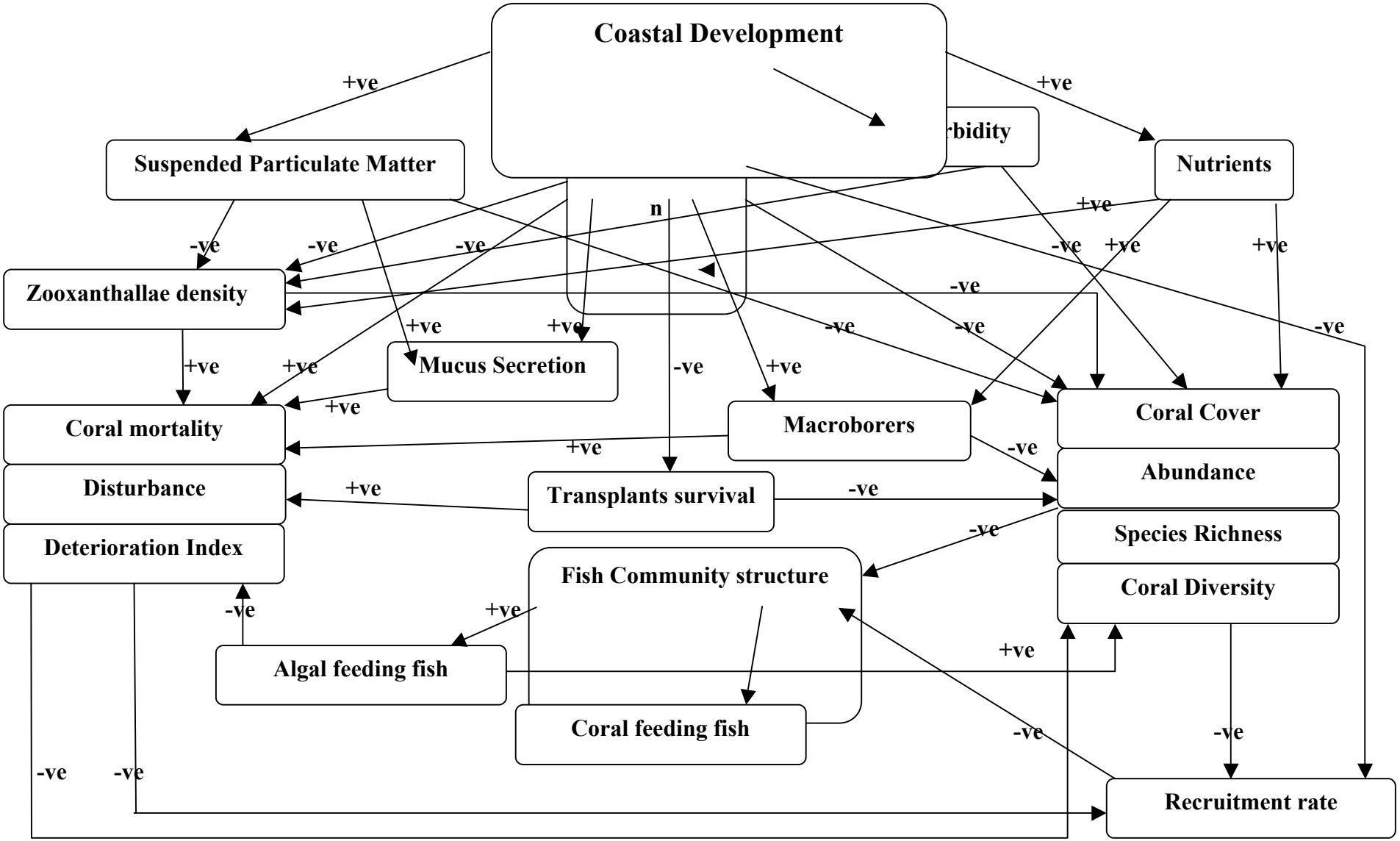
In an experimental study by Titlyanov et al. (2005), two species of scleractinian corals, the massive colonies *Porites lutea* and the branching colonies *Porites cylindrica*, were found to be able to completely regenerate tissue and skeleton from injuries with partial or complete tissue removal. In the long-term the regeneration rate dropped 2–3 times under 1.5–2-fold decrease in the growth rates of the corals probably caused by elevated

temperature in aquaria of up to 30–31.8C. It is assumed that regeneration of injuries has metabolic priority in corals even under extreme condition (Titlyanov et al., 2005). Probably, scleractinian corals are able to use the energy of heterotrophic metabolism as a result of predation of zooplankton or digestion of its own zooxanthellae for tissue recovery under low light (Titlyanov et al., 1996, 2000).

1.9 A Conceptual Model of the Impacts of Coastal Development on Reef Health

Coastal development in Hurghada has been anticipated to introduce a huge amount of sediment into the coastal waters and by the end increase, sedimentation rate, Suspended Particulate Matters, turbidity and nutrients contents. These items were expected to reduce coral cover, species richness, diversity, abundance, settlement and recruitment of hard corals. They also, with the exception of nutrients were anticipated to reduce transplanted coral survival and zooxanthellae density in coral tissues. Nutrients increase zooxanthellae and survival to certain level before the eutrophication condition attained. Decline of coral cover, species richness, diversity and abundance were though to change fish community structure and increase algal feeding fish and decrease coral feeding fish. Other processes were thought to increase in correlation to augment sediment input and by the end negatively impact coral cover, richness, diversity and abundance. These processes include, coral mortality, disturbance, deterioration, bioerosion and mucus secretion by corals (Figure 1.2).

Figure 1.2 A conceptual multidimensional schema of sedimentation impacts on coastal reefs of Hurghada.



1.10 Biological indicators.

1.10.1 The Need for Indicators

There is a need for efficient methods for monitoring the state of reef health currently as a consequence of the extensive reef deterioration observed in the last few decades (McKenna et al., 2001). Coral community monitoring is vital to provide information and data required for their conservation and management, and provide answers to many current questions about possible reef recovery. It is also essential to build a comprehensive data foundation for future research into effects of disturbance on reefs. The causes of change in an ecosystem are hard to be specifically linked to any one factor; for example, bleaching events in corals are thought to be a result of high temperature, fluctuating salinity, insulation, sedimentation or the combination of these factors (reviewed in Linton and Warner 2003). The consequences of reef deterioration for research is that ever fewer studies will be conducted in benthic ecosystems that are not deteriorated to some degree, and most research will concentrate on impacted systems (Stachowitsch, 1992).

Biological indicators are defined as that subset of environmental indicators in which the living component of the environment is used to reflect the state of the environment in response to human induced stresses (Linton and Warner, 2003). This group of indicators is also called bio-indicators, they are defined as the signs used to convey the complex status of the reef in a simple manner, and provide insights into trends of reef health that cannot be directly observed (reviewed Linton and Warner, 2003). Bioindicator species should be characterized by; natural abundance throughout the study area and should not be exploited. They should be easy to sample in an objective and quantitative manner and should show response specificity, so that a particular impact on the ecosystem can be identified. Ideally, they should indicate gradations in the response relative to the amount of stress (Linton and Warner, 2003).

Edinger et al. (2000) maintained that coral growth (extension) rates alone are poor indicators of coral reef health, particularly where anthropogenic eutrophication may play a role in reef degradation. Measures such as community structure and function, including species richness, species abundance and indicator species are used as indicators of reef health (USEPA, 1990). Coral rubble bioerosion is also used as an indicator of eutrophication stress on coral reefs, and forms a valuable rapid reef assessment technique (Holmes et al., 2000). The surface micro layer of coral heads (mucus) might accumulate microbial indicators of waste and human viruses, and could thereby provide more direct evidence for human impacts on reef environments. (Lipp et al, 2002). Partial mortality in some species of massive corals is a good indicator of sediment stress and could reflect temporal changes in coral communities (Nugues and Robert, 2003a). A number of studies have shown a positive correlation between Chaetodontid diversity and abundance and percent live coral cover or coral species richness (Jameson et al., 1998).

Biological criteria or biocriteria are defined as narrative expressions or numerical values that describe the biological integrity of natural community (Jameson et al., 1998). Development of Coral Reef Ecosystem Biocriteria includes the following steps:

1-Coral reef is classified into classes or groups based on physical and geographic characteristics not subject to human perturbation.
2-Both the biotic and physical habitat characteristics are surveyed using standardized methods within each group or class.

3-The preliminary classification is tested with the biological data to determine whether it consistently reflects the biological communities.
4-Potential metrics that have ecological relevance are identified and tested in this step. These measures should reflect biological properties which are shown to be sensitive to environmental impairment such as richness, diversity and dominance indices, biomass and mean individual size measurements, trophic shifts, health indices, abundance proportions of taxonomic groups, and the presence or dominance of tolerant (opportunistic) and sensitive species.
5-Biocriteria are then formulated in part from the metrics and index values developed from the population of reference sites for a given coral reef ecosystem class. The process of biocriteria assessment includes the conversion of monitoring data to biological indices for comparison with a minimally impaired reference site (reviewed in Jameson et al, 1998).

A list of bioindicators that has been developed to some degree were shown in Table 1.2. Although they appear to be useful and suitable for wide application, they do not all give unambiguous signals of stress. They require adapting to local conditions, and training for those gathering and analyzing the data (Jameson et al., 1998).

Table 1.2: Potential bioindicators identified for coral reefs ecosystems and the impacts they point to.

Bioindicators	Indicator of	Reference
-Phytoplankton biomass.	-Indicator of eutrophication - Phytoplankton diversity.	(reviewed in Linton and Warner, 2003).
	-As indicators of eutrophication.	
-Foraminifera species composition.	-Indicator of exposure to tributyltin (TBT).	
-Seagrass productivity and algal epiphyte loads.	-Indicator of stress and foundational to all coral reef assessment programme.	
-Gastropod imposex.	-Indicators of increasing eutrophication and/or organic loading on reefs.	
-Percent cover by hard corals.	An indicator of stress, but not accurate or specific.	
-Coral bioeroders, especially sponges.	-Indicators of past stress, But not a present signal.	
-Corals bleaching.	-Provides a past history, but not necessarily identify the cause.	
-Contaminant uptake by coral skeletons.	-For pollution detection.	
-Coral mortality and broken coral.	-Reef health monitoring.	
-Stomatopod biomonitoring.	-For detecting over-exploitation.	
-Chaetodontidm fish (butterflyfish).		
-Fisheries or fish survey data.		

Table 1.3: Other potential bioindicators of generalized environmental stress which require further research to determine whether they are effective and likely to provide clear signals of particular stresses.

Bioindicators	Indicator of	Reference
-Composition and structure of the sessile community on mangrove roots.	-Eutrophication	(reviewed in Linton and Warner, 2003).
-Coral disease frequency and intensity.	-Pollution	
-Growth rates of corals and other reef biota.	-Indicator of many stress sources	
-Productivity and calcification of reefs.	-Indicator of many stress sources	
-Measurements of reproductive and recruitment success of corals.	-Indicator of many stressors.	
-Various organisms, such as amphipods.		

1.10.2 Coral Indictors

Several parameters have been used as indicators of the state and health of the coral community. These parameters include the ratio between living and dead corals (Yap, 1986), the Mortality Index (Gomez et al., 1994) and the size–frequency distribution of corals, which may indicate recruitment rate to the reef (Bak and Meesters, 1998; Meesters et al., 2001). In healthy coral communities, the frequency of the smallest size groups is expected to be the highest. In addition percentage live coral cover, abundance, species richness, diversity index and degree of disturbance in the reef are very common and used as indicators of reef status and health (reviewed in Ben-tzvi et al, 2004). Percent hard coral cover is a widely used reef monitoring parameter; coral cover and colony number have traditionally been considered as an essential part of long-term monitoring programs (Nugues and Roberts, 2003a). Reduction in percent cover is thought to signal stressful environmental conditions for corals. Several other coral related measures have also been used as indicators of reef quality; these include coral growth rates, productivity, calcification, fecundity and recruitment (reviewed in Linton and Warner, 2003). Aronson et al. (1994) suggest the measurement of reef topographic complexity as a more relevant indicator of reef health than simple percentage live cover.

Brown (1988) reviewed a number of coral-focused parameters that may provide an indication of environmental stress and therefore is of particular use in pollution assessment studies. These include: measurement of coral growth (skeletal extension) rates; calcification and productivity rates; coral fecundity and recruitment; monitoring for zooxanthellae loss, coral disease and cyanobacterial blooms; and measurement of the bioaccumulation function of coral skeletons. Ben-Tzvi et al (2004) suggested a new approach to help in the detection of disturbed coral communities in one survey. This approach has developed the deterioration index (DI). This indicator measures the ratio between mortality and recruitment rates in the coral reef.

Physical damage to corals in the form of broken coral rubble is used by Jameson et al. (1997) as an indicator of diving over-capacity in the Red Sea. Recruitment intensity itself may be used as a useful measure for damaged reefs whether it may require coral transplantation or they can recover naturally (Kojis and Quinn, 2001). Tomascik and

Sander (1987) suggested that coral fertility is decreased on reefs subject to increased eutrophication. Recruitment could be used as indicator of sediment stress. Richmond (1994) maintains that runoff and sedimentation may prevent successful reproduction and recruitment in corals and other reef organisms. Low recruitment rates on ceramic tiles may be due to low availability of larvae or low settlement success (including larval preference), or both (Dikou and van Woesik, 2006). The frequency and severity of occurrences of coral diseases may indicate environmental stress, since stress may cause a decline in natural immunity. However, some coral diseases such as White Band Disease (WBD) appear to be novel so may indicate the introduction of exotic microorganisms (reviewed in Linton and Warner, 2003).

Many reef researchers suggest the use of coral "vitality" or "mortality" indices, which take into account the ratio of live and dead coral, cover in an estimation of reef "health" (Grigg and Dollar, 1990; Dustan, 1994; Gomez et al. 1994; Ginsburg et al. 1996). Considering that one hundred percent coral cover is not a standard to which most coral reefs can compare (reviewed in Jameson et. al, 1998), live coral cover, species richness, mortality and recruitment rates differ normally between different types of coral reefs; live coral cover may misleadingly indicate a healthy reef whereas, in fact, the reef may be undergoing deterioration due to either high mortality and/or low recruitment rates (Ben-Tzvi et al., 2004). In many cases, impact studies usually start after the insult began. In such a case, it must be assumed that sensitive species have already been eliminated, and that a stable state has been reached (Jameson el al, 1998).

1.10.3 Transplanted Coral as Indicators

Coral transplantation and its survival is another good indicator of suitability of reef to support a healthy coral community. According to Rinkevich (2000) several restoration experiments have revealed that the use of coral fragments may serve as a good tool for reef rehabilitation. The primary objectives of coral transplantation are to improve reef quality in terms of live coral cover, improve biodiversity and enhance topographic complexity and to accelerate rehabilitation of denuded reefs (Oren and Benayahu, 1997). Rinkevich (1995) further revealed that local conditions, the type of substratum and coral species chosen might significantly affect the results. The rate of stabilization of fragments is related to substrate type and distance from a patch of mature colonies, suggesting that standing colonies may protect regenerating fragments removed from the reef (Ammar et al, 2000).

Coral colonies transplanted to 1m depth grew faster than those transplanted to 10m, which is most likely due to the greater light intensities which enabled corals to maintain a high rate of photosynthesis and growth (Custodio and Yap, 1997). In a transplantation study by Clark and Edwards (1994) 530 coral pieces were transplanted by cementing them into the bottom, within 16 months most colonies had accreted naturally to the concrete mats and over a 28-month period most losses of transplants were mainly due to wave action. Transplantation of sensitive scleractinian coral indicator species to an impacted areas, and examination of several sub-lethal indicators (possibly growth rate, fecundity, etc), could help to determine at what distance these are no longer affected (Jameson el al, 1998). Transplant survival varies greatly between sites with general high survival, in transplantation study carried out by Rinkevich (2000). After one month monitoring of almost 2500 coral and octocoral specimens transplanted to denuded reefs using underwater containers; revealed that only 3% mortality was recorded (Rinkevich, 2000).

1.10.4 Fish as Indicators

Fish species that depend on live coral throughout their adult life, or species that use live coral at other critical life history stages will be negatively impacted by the broad-scale reduction in coral cover and changes to coral community (Munday, 2004). Reductions in herbivore fish populations caused by overfishing may enhance the likelihood of a coral–algal phase shift. However, according to McManus et al. (2000) the reduction in herbivore fish can lead to the proliferation of algae even in the absence of eutrophication. Previous studies reveal that heavily disturbed or overfished sites often undergo a shift to communities dominated by soft corals and macro algae, which limit recovery of hard coral colonies (Done, 1992; Hughes, 1994; Roberts, 1995). For widespread coral bleaching such as that associated with the 1997–1998 El Niño event, the combined effect of overfishing as well as pollution may prevent a return to coral dominance (McManus et al, 2000)

When using fish community composition as a bioindicator for changes in the health of the habitat, recording only the most common species will give satisfactory results with less effort and expertise (Brokovich et al., 2005). Reef fish abundance and distribution indicate the status and health of the reef they live within. Many species of the family Chaetodontidae are obligate corallivores, and thus depend on the live tissue of corals for their food, they are considered as excellent candidates of changes of reef conditions (Crosby et al, 1996). They are, therefore, thought to provide effective early warning signals of deterioration of coral reefs. The simple hypothesis behind their use as coral reef bioindicators is that they feed on corals and so if corals decline then butterfly fish populations should also decline or change their behavior in response (Linton and Warner, 2003).

Butterfly fish (Chaetodontidae) are widely used as bioindicators of environmental stress on coral reefs and many scientists have given data to support the butterfly fish bioindicator hypothesis (Jameson et. al, 1998). The butterfly fish bioindicator hypothesis proposes that for those species of butterfly fish which are obligate corallivores, a decline in the condition of a coral reef, manifested by decreasing food quality of the stressed coral polyps, will result in a decrease in the abundance and diversity of these species and an increase in territory size, feeding rate and struggle encounters as mated pairs attempt to maintain their nutritional intake by expanding their territories to include more coral colonies (Jameson et. al, 1998).

Many studies have revealed that live coral cover is generally positively correlated with both; the number of species and the abundance of Chaetodontids, and the abundance and diversity of fish assemblages as a whole. It is not clear; however, which aspect of coral cover is important for butterfly fishes (Bozec et al., 2005). Many studies provide evidence that a reef which is dead from sedimentation, Crown of Thorns or from some other factor, no longer has its characteristic assemblage of coral feeder Chaetodontids (Reese, 1981; Bouchon-Navaro et al., 1985; Hourigan et al., 1988). On the other hand, coral deterioration, causing a phase shift from coral to algal dominance on reefs, could be a result of chronic pressures, such as constant fishing for herbivores or organic pollution (Done, 1992). Between group competition could affect fish distribution as in the case of fishing of piscivorous fish, which results in herbivorous fish becoming more abundant (Miller and Hay, 1998).

There appears to be a threshold level of reef deterioration at which fish begin to leave, perhaps related to the decrease in both abundance and diversity of the coral on which they are feeding (Crosby et al, 1996). Results of early studies of chaetodontids did not quite

link changes in population density with declines in corals, possibly because they themseleves may be exploited either intentionally for the marine aquarium trade or accidentally in net fishing as a part of the fish catch (Nash, 1989; White, 1989). Early fish survey studies indicated that the highest percentage of sites in the Indo-Pacific was in the range of >4±6 fish per 100m^2 class, whereas in the Red Sea, the peak number of sites was in the >6±8 fish per 100 m^2 class (reviewed in Hodgson, 1999).

CHAPTER 2

MATERIALS AND METHODS

2-1 Field Sampling and Survey

A Pilot survey was carried out in July 2003 to choose a range of study sites that represented various sedimentation conditions in the Hurghada region. Using a speedboat, a large area of reef was surveyed by snorkeling to check the reef condition with regard to coral cover, turbidity, degree of exposure and dominant currents. In addition, distance from dredging and filling activities, impacts (whether natural or human induced) and exposure level whether the site was sheltered or exposed to strong currents and wave actions; were all taken into considerations in the selection process. Sites were chosen to represent inshore and offshore habitat conditions. Seven sites were selected after this rapid survey to carry out the study of sedimentation effect on reef health. The sites from the north to south were NIOF (1), Abu Sadaf (2), Shedwan (3), Arabia (4), Abu Minkar (5), Holidays (6) and Sahal Hashish (7) (Figure 2.1). Geographically the study sites covered a reef area extending to more than 30km latitudinal distant from north to south of the main development complex of Hurghada city, which is the largest city on the Egyptian Red Sea coast. Sahal Hashish (7) was taken as the control site in regard of sedimentation, fishing, diving and all anthropogenic impacts.

Fieldwork started In August 2003 by fixing sediment traps, settlement tiles and transplanted coral branches at each site. After one month, the first readings for sedimentation rate and survival of transplanted corals were taken. Sediment sampling and survey trips were performed seasonally in the period from September 2003 to October 2004 (five seasons) and monthly from January till December 2006. Coral samples used for feeding experiments were collected from A.Sadaf (2) reef and samples for the transplantation and zooxanthellae studies were taken from the NIOF (1) reef. All fieldwork was carried out using a small fibreglass speedboat. The Marine Park laboratory was used for sediment filtration, nutrient analysis, feeding tests and mucus sample analysis. Laboratory experiments for coral feeding, mucus secretion and zooxanthellae density under various levels of sediment were also carried out in the Marine Park laboratory. Sediment size class analyses were done in the laboratory of the National Institute of Oceanography and fisheries (NIOF). All field surveys and sampling processes were performed using SCUBA diving at depth range of 3m-5m on the reef slope. Data were recorded using PVC slates and pencil. The fieldwork was carried out during daylight hours (10:00am-05:00pm).

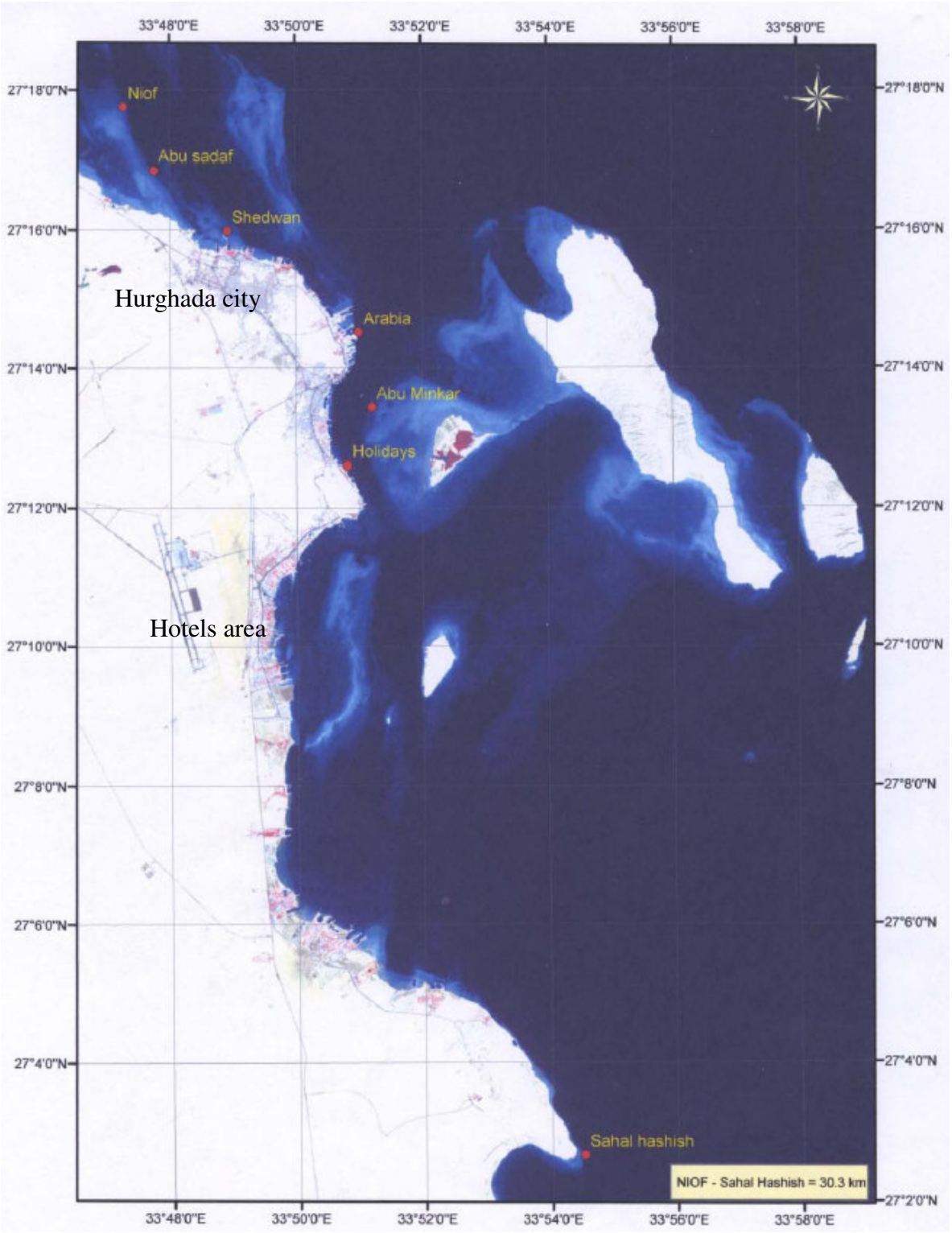


Figure 2.1- Map of the study sites along the reefs of Hurghada together with the relative position of each of the sites in relation to Hurghada main coastal development.

2.2 Description of study sites

The study was conducted at six sites along Hurghada coastline and one site on near shore island (site 5), Red Sea (Figure 2.1). Sites from north to south may be described as follow:

2.2.1 NIOF (1)

The National Institute of Oceanography and Fisheries (NIOF) was the most northern site in the study area (27º 17' 45" N, 33º 47' 16" E). This site is characterized by a long patchy reef, representing the front edge of a wide and shallow reef flat with many depressions and lagoons (Figure 2.2). Seaward of the reef edge was a shallow mostly sandy bottom area extending a long distance with few coral patches. The depth ranged from about 3m at the reef front with gentle slope towards deep water. The area was generally exposed to strong waves, and the currents follow the prevailing current direction in the Red Sea from north to south. Fishing is considered the major impact in this area, mainly net fishing on the reef flat and the lagoons (Table 2.1). The coastline facing this site does not undergo any modification through filling or dredging, although there has been old building development along the coast of this area.

Table 2.1: Impact matrix of the major impact sources in the study area for NIOF site, Hurghada, Egypt.

	High	Medium	Low
Fishing		*	
Diving			*
Snorkeling			*
Pollution			*
Litter		*	
Sedimentation			*
Turbidity		*	

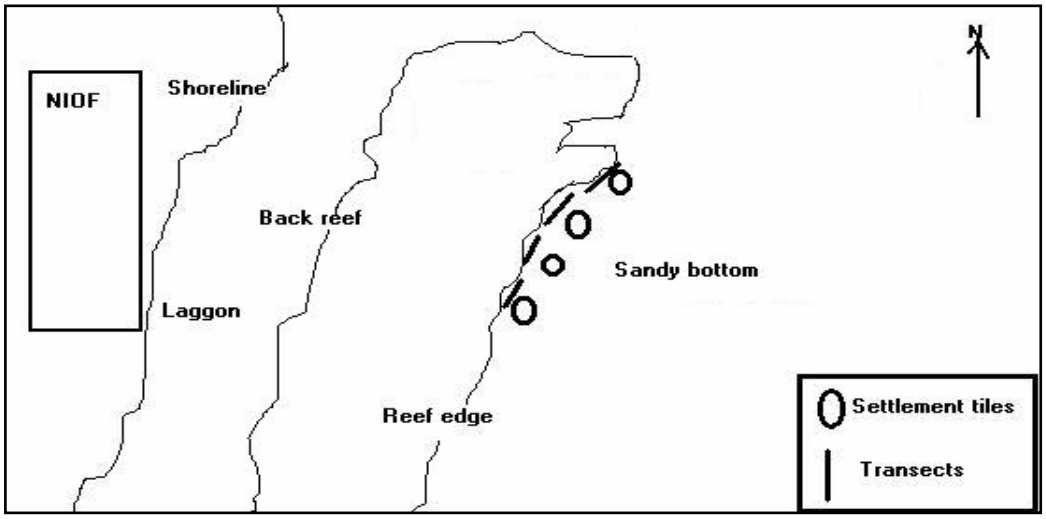


Figure 2.2: Reef topography and survey site position in relation to reef and shoreline, for the NIOF site, Hurghada, Egypt.

2.2.2 Abu Sadaf (2)

The second site to the north was Abu Sadaf (27º 16' 5"1N, 33º 47' 45" E), as it is called by local people (Figure 2.3). This means that it had a great amount of shells (sadaf), as it used to be inhabited by great populations of gastropod shells (*Strombus* and *Lambus*) on the reef flat. Although these are depleted due to overfishing, the site still attracts many fishermen for seasonal fishing. An extended reef flat with seagrass and seaweed beds

suitable for fish and shellfish breeding characterizes this site. There are nearshore sandy patches and lagoons that were usually used by local people for artisanal seasonal fishing of *Parupeneus* (Goatfish) (Table 2.2). The depth at the reef edge is about 1m. Seawards of the reef edge the depth starts at 5m with a sandy bottom that drops rapidly towards deep water. Currents are weak and mostly from north to south and strong waves breaks at the reef edge.

Table 2.2: Impact matrix of the major impact sources in the study area for A.Sadaf, site, Hurghada, Egypt.

	High	Medium	Low
Fishing		*	
Diving			*
Snorkeling			*
Pollution			*
Litter		*	
Sedimentation		*	
Turbidity		*	

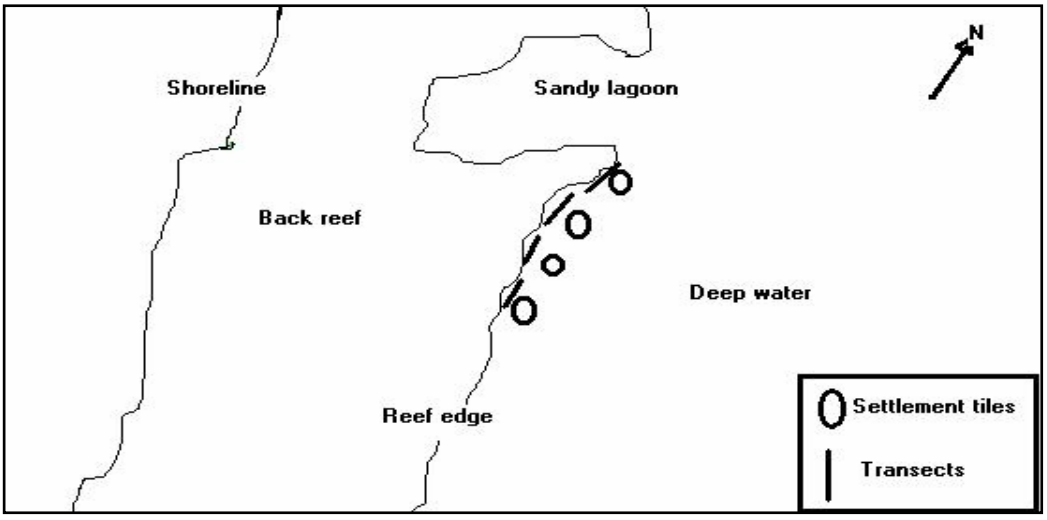


Figure 2.3: Reef topography and survey site position in relation to reef and shoreline, for the site A.Sadaf, Hurghada, Egypt.

2.2.3 Shedwan (3)

This reef faces one of the oldest resorts in Hurghada called Shedwan Resort (27º 15' 59" N, 33º 48' 55" E). The reef had undergone substantial filling of the entire back reef and an artificial beach was constructed with embayment and concrete barrier to protect the filled area (Figure 2.4). The site was chosen in front of the filled area with reef pinnacles and patches, which were scattered in a sandy bottom area at 4m depth and extend to 1m below the sea surface (Table 2.3). Weak currents and waves from north to south were the most prevailing in this reef throughout the year. As the site is protected from wave action by extended reef patches, it is used by resident tourist and visitors for snorkeling and rarely by fishermen for net fishing.

Table 2.3: Impact matrix of the major impact sources in the study area for Shedwan site, Hurghada, Egypt.

	High	Medium	Low
Fishing	*		
Diving			*
Snorkeling	*		
Pollution			*
Litter		*	
Sedimentation	*		
Turbidity	*		

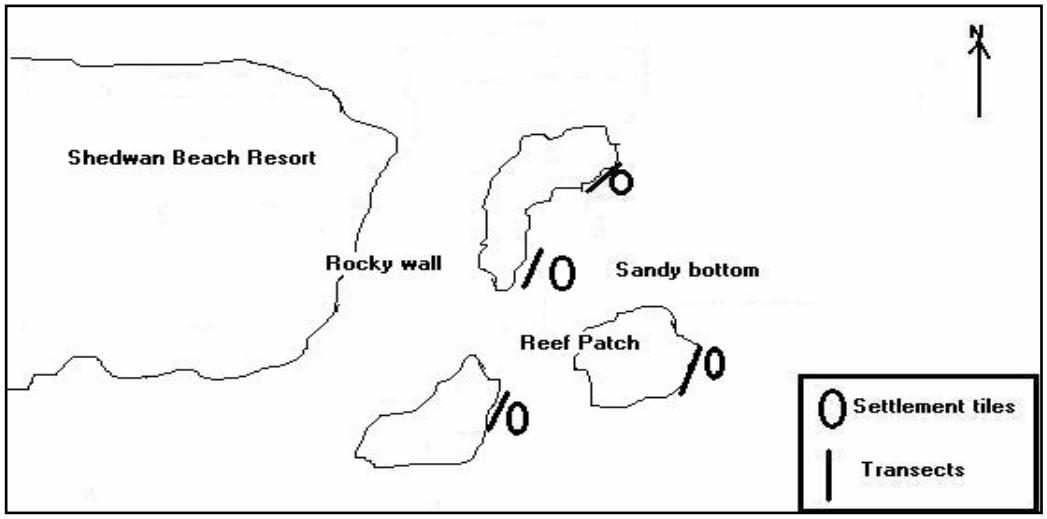


Figure 2.4: Reef topography and survey site position in relation to reef and shoreline, for the site Shedwan, Hurghada, Egypt.

2.2.4 Arabia (4)

The Arabia site was in front of the Arabia Resort complex (27º 14' 31" N, 33º 51' 00" E). The reef flat was filled to the reef edge with an embayment and artificial beach. Concrete blocks were used to fence the filled area at the reef edge (Figure 2.5). The reef edge was smashed a long distance with concrete blocks leaving coral rubble mixed with gravel and sand extending to deep water. The site was characterized by a sharp reef wall dropping to the bottom at 10m depth. The site was used by swimmers and snorkelers, but was not suitable for fishing activities (Table 2.4). The Arabia site was exposed to strong waves and currents as it represents a land protrusion into open deep waters.

Table 2.4: Impact matrix of the major impact sources in the study area for Arabia, site, Hurghada, Egypt.

	High	Medium	Low
Fishing			*
Diving		*	
Snorkeling	*		
Pollution			*
Litter		*	
Sedimentation	*		
Turbidity		*	

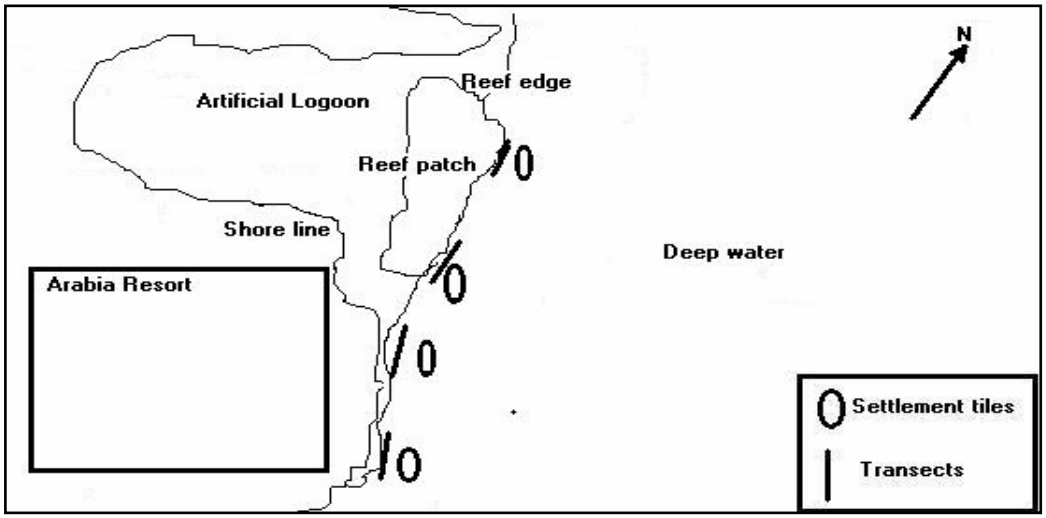


Figure 2.5: Reef topography and survey site position in relation to reef and shoreline, for Arabia site, Hurghada, Egypt.

2.2.5 Abu Minkar (5)

The study site was located on the north side of Abu Minkar island (27º 13' 26" N, 33º 51' 12" E). This site represents an offshore exposed condition, in a semi-sheltered shallow bay (Figure 2.6). The site has been under fishing and snorkeling pressure for a long time (Table 2.5). The reef edge comes up to the water surface during low tide. The bay depth was 4m with a sandy bottom and sparse coral patches and pinnacles. The site was exposed to strong wave action and weak current effects.

Table 2.5: Impact matrix of the major impact sources in the study area for A.Minkar site, Hurghada, Egypt.

	High	Medium	Low
Fishing	*		
Diving		*	
Snorkeling		*	
Pollution	*		
Litter		*	
Sedimentation			*
Turbidity		*	

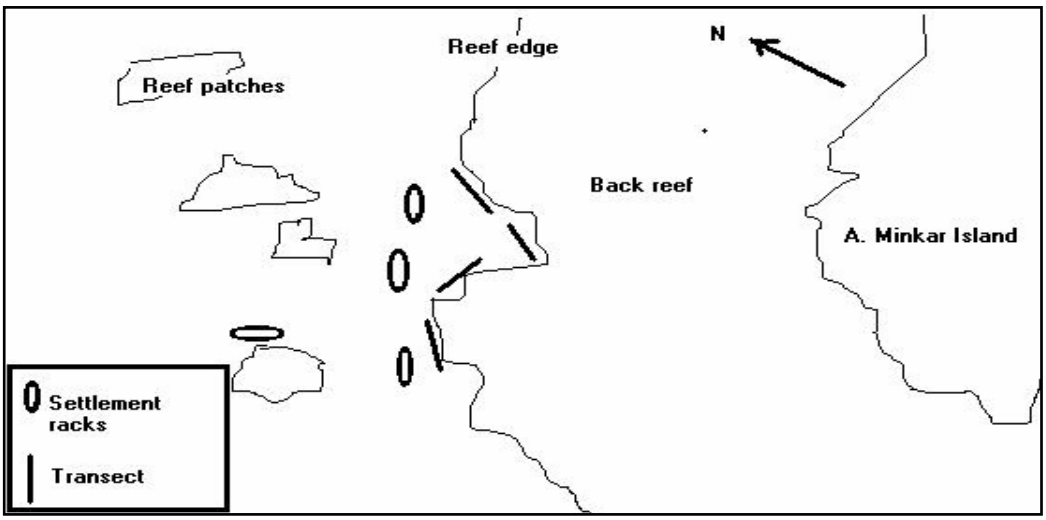


Figure 2.6: Reef topography and survey site position in relation to reef and shoreline, for the site A.Minkar, Hurghada, Egypt.

2.2.6 Holidays (6)

Holidays Resort was built on the filled back reef (27º 12' 36" N, 33º 50' 49" E). The reef edge had disappeared and was replaced with massive concrete blocks forming a protective wall against waves and provided a boat-mooring marina for the hotel (Figure 2.7). Water depth was 5m with a loose sandy bottom very poor in corals for a long distance from the shore. The artificial beach was usually weathered and beach nourishment usually takes place on a yearly basis. Concrete blocks form an ideal settlement substratum for coral larvae although there was high sedimentation in the site. Generally the site in not suitable for fishing or snorkeling activities and limited diving practices take place by hotel residents (Table 2.6).

Table 2.6: Impact matrix of the major impact sources in the study area for Holidays site, Hurghada, Egypt.

	High	Medium	Low
Fishing		*	
Diving	*		
Snorkeling	*		
Pollution		*	
Litter	*		
Sedimentation	*		
Turbidity	*		

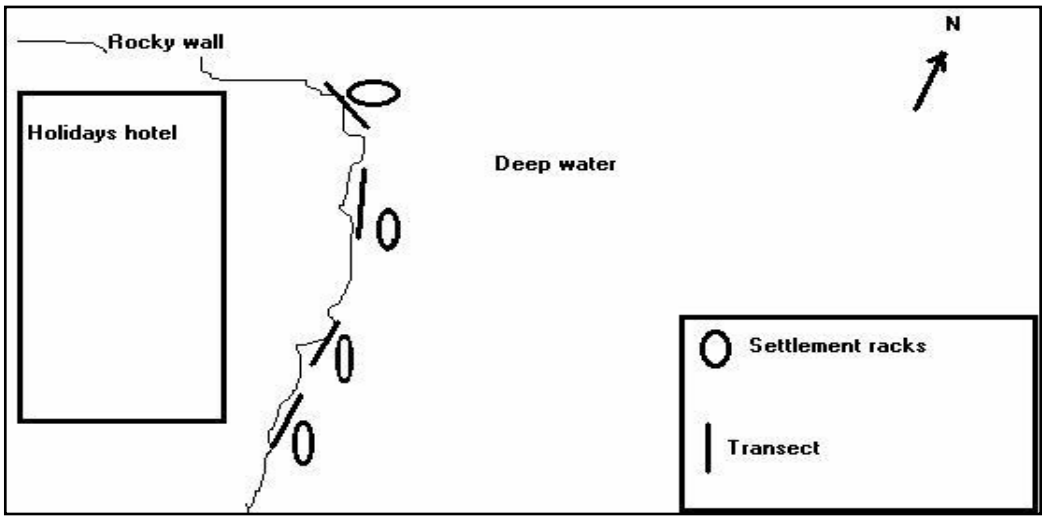


Figure 2.7: Reef topography and survey site position in relation to reef and shoreline, for the site Holidays, Hurghada, Egypt.

2.2.7 Sahal Hashish (7)

This site represents the most protected site, to the south of the development area (27º 02' 41" N, 33º 54' 31" E)., although a huge resorts complex is now under construction and will be completed in a few years. The reef is still in good condition so far; the back reef was narrow and shallow, less than 0.5m in depth, and usually uncovered at low tide. The reef edge was sharp and dropped down to about 12m depth towards a sandy bottom and then declined rapidly to great depths (Figure 2.8). Currents and waves are strong in the area most of the year, which make it inappropriate for diving, fishing and boating and thus protects the reefs from human usage (Table 2.7).

Table 2.7: Impact matrix of the major impact sources in the study area for S.Hashish site, Hurghada, Egypt.

	High	Medium	Low
Fishing			*
Diving		*	
Snorkeling			*
Pollution			*
Litter		*	
Sedimentation			*
Turbidity			*

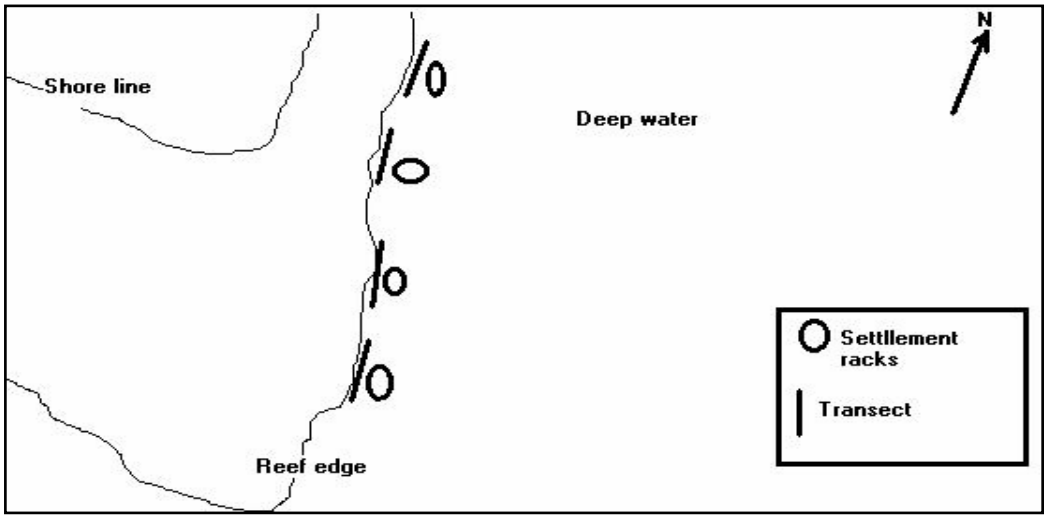


Figure 2.8: Reef topography and survey site position in relation to reef and shoreline, for the site S.Hashish, Hurghada, Egypt.

Table 2.8: Impact matrix for the whole study sites and the expected levels of impact influence each site, Hurghada, Egypt.

sites	1	2	3	4	5	6	7
Fishing	**	***	***	*	***	**	*
Diving	*	*	*	**	**	***	**
Snorkeling	*	*	***	***	**	***	*
Pollution	*	*	*	*	***	**	*
Litter	**	**	**	**	**	***	**
Sedimentation	*	**	***	***	*	***	*
Turbidity	**	**	***	**	**	***	*

Where: *= low, ** =medium and *** =high impact.

2.3 Sedimentation rates

Sedimentation rates were measured at the seven sites, seasonally from September 2003 to October 2004 and monthly from January 2006 till December 2006. Cylindrical sediment traps, 15 cm length and 5 cm diameter were deployed in sets of three as shown in Figure (2.10), diameter- length ratio is designated to avoid sediment washout from the trap under water turbulence. The design of the sediment trap is described in the survey manual for tropical marine resources (English et al., 1997). Five replicates were installed at each site, alongside the settlement racks, at 20ms intervals. The traps were fastened using steal pins and cable ties 40cm above the bottom, pins were fixed to the bottom using a hammer and

cement. Sediment traps were left in situ for 3months at the first survey (from September 2003 to October 2004) and for one month in the second survey (January 2006 till December 2006) then collected and replaced with new ones. The traps were sealed underwater, placed in plastic bags and transferred to the laboratory. Here the contents of each trap were filtered on the same day and dried for 24hrs at 80°C and weighed. Large objects like stones; shells and living algae or fish were excluded from weighing process. Used traps were then cleaned, washed and dried for reuse.

- ***Statistical Analyses***

Measured values were given as the means ± standard deviation (Stdev), with the number of observation either stated or shown in parentheses. Normal distribution test and homogeneity of variances test were carried out before proceeding with ANOVA. Group normality was tested using a One-Sample Kolmogorov-Smirnov Test, and homogeneity of variance was checked using Levene's Test. Data which fulfilled these criteria were compared using analysis of variance (ANOVA ($p<0.05$). If significant differences were detected, the post hoc Turkey's test (equal number of observations in each group) or Scheffé's test (unequal number of observations in each group) were used to identify where the differences were. Mean frequency histograms for seasonal sedimentation rate and mean total with standard deviation of mean were plotted. Data analysis for the whole study was carried out using SPSS statistical package.

Figure 2.9: Sediment accumulation on top of massive coral leads to partial mortality.

Figure 2.10: Sediment trap design used for sedimentation rate study.

2.4 Suspended Particulate Matter (SPM)

Suspended particulate matter (SPM) was measured by filtering one litre of seawater onto a pre-weighed glass fibre filter, which was subsequently oven-dried and weighed (Cortes and Risk, 1985). This represents a quantitative instantaneous measure of the concentration of particles suspended in the water column. It was used as another variable to describe the sediment regime in the area in addition to sediment traps. One litre of subsurface water was collected from four different locations at each site using clear plastic bottles. The samples were then transferred to the laboratory and filtered with pre-weighed glass fiber

filter paper, dried at 450°C for 3hrs and then reweighed. Suspended Particulate Matter (SPM) were measured along with sedimentation at each site.

- ***Statistical Analyses***

Group normality was tested using the One-Sample Kolmogorov-Smirnov Test, and homogenetiy of variance was checked using Levene's Test. Two-way ANOVA ($p<0.05$) was used to examine the differences between sites and years in SPM. Mean frequency Histogram with standard deviation of mean was plotted. Correlation was tested between sedimentation rate and SPM of the two years. Degree of significance varied from significant where $P> 0.01$ and <0.05 and marked with one asterisk (*), highly significant if $P<0.01$ and marked with two asterisk (**), very highly significant where $P< 0.005$ and marked (***) and not significant when $P>0.05$ and marked (ns).

2.5 Sediment Size Classes Analysis

The study of sediment textural parameters is used to differentiate between various environmental and depositional condition. It was also used here to define the sediment source in each site. Deposited sediment was sampled at all sites, four 500gm samples were cored (core diameter 5cm and length 20cm) from the bottom of the study sites and carried to the laboratory in plastic bags for further analysis. Shore sediment samples were also sampled from four locations in each site to compare with bottom one. Samples were washed with distilled water to remove soluble salts, and then oven dried and mixed well before testing. The grain-size distributions of these sediment samples were obtained using the sieving method (Folks and Ward, 1957). Subsamples of 100gm of each sample were taken by splitting the dry samples for mechanical analysis. These were sorted using a standard set of sieves ranges from –1 to 4Φ, shaken in a Ro-Tap shaker for 20 minutes. The collected sieves fractions were accurately weighed to define the percentage of each size class. The mean size, sorting and non-carbonate percentage were also determined. Samples were collected and examined once each year (2004, 2006). Mean size was calculated using the following equation (After Folks and Ward, 1957):

$$Mz = \frac{\Phi 16 + \Phi 50 + \Phi 84}{3}$$

Where the size range is: 1.0 -----0.0 Φ very coarse sand
0.0-------1.0 Φ coarse sand
1.0--------2.0 Φ medium sand
2.0 -------3.0 Φ fine sand
3.0-------4.0 Φ very fine sand
4.0---------5.0 Φ coarse silt
5.0----------6.0 Φ medium silt
6.0---------7.0 Φ fine silt

The sorting parameter σi is given by the following equation (After Folks and Ward, 1957):
$\sigma i = 0.25 (\Phi 84 - \Phi 16) + 0.1515 (\Phi 95 - \Phi 5)$
Where; < 0.35 Φ very well sorted;
0.35------0.5 Φ well sorted

0.5------0.71 Φ moderately well sorted
0.71-------1.0 Φ moderately sorted
1.0--------2.0 Φ poorly sorted
2.0-------4.0 Φ very poorly sorted
> 4.0 Φ extremely poorly sorted.

Two methods were applied to measure the percentage of non-carbonate sediment. The first method depends on dissolving the carbonate component of sediment using hydrochloric acid and the second method depends on the use of thin section technique and the microscopic investigation of sediment particles.

For the first method, five representative subsamples were treated with one normal hydrochloric acid (1N HCL) to dissolve the carbonate materials following the methods mentioned in Robinson (1980). Samples were then dried and reweighed, weight differences represent the percentage of biogenic or calcium carbonate component of the sediment sample in relation to the total sample weight.

Sieve fractions of the size 0.25mm and 0.125mm were used for the microscopic investigation of the relative percentage frequency of the two main constituents: carbonate and non-carbonates grains. Sediment was mounted onto slides (five slides for each location) using Canada balsam, dried at 70°C for 24hrs, then polished using sandpaper to remove all the extra dismounted sand fractions. The slides were examined under a binocular microscope to count the number of terrigenous rock fragments and biogenic fragments, at least 300 grains were counted to satisfy the statistical reliability (Ujiie, 1962). The grain number of each component was correlated to the total number of particles to determine the percentage of each component.

- ***Statistical analysis***

Two way ANOVA ($p<0.05$) was used to examine the differences between sites and location in the percentage of terrigenous sediment; and between the two years and sites. Mean frequency Histograms for seasonal sedimentation rate and mean total with standard deviation of mean were plotted. Group normality was tested using One-Sample Kolmogorov-Smirnov Test, and homogenetiy of variance was checked using Levene's Test. Descriptive statistics was carried for non-carbonate sediment, sorting and mean grain size to investigate standard deviation, kurtosis and skewness of these parameters from the mean values.

A Spearman's Correlation test was carried out to investigate the relationships between bottom and beach non-carbonate sediment percentage, and between non-carbonate sediment percentage of the two years (2004 and 2006), and both of sedimentation rates and SPM. All significant correlations are illustrated by their respective scatter graphs. Degree of significance varied from significant where $P <0.05$ and >0.01 and marked with one asterisk (*), highly significant if $P<0.01$ and marked with two asterisk (**), very highly significant where $P< 0.005$ and marked (***) and not significant when $P>0.05$ and marked (ns)

2.6 Coral Field Survey

Line intercept transects (LIT) (10m) were carried out using a plastic measuring tape (Figure 2.11), to measure the percentage of coral cover and all other substratum categories, following methods described in AIMS (English et al., 1997). The line was laid parallel to reef edge at 3-5m depth and depending in the reef topography, four transects separated by at least 5m, were surveyed each site.

Line belt transects (LBT) (10x0.5m) were also carried out to determine the number of living coral species and coral colonies per transect. Number of new recruits, recent dead colonies and maximum diameter for branched and massive coral colonies were also counted. A species list for hard corals was recorded using the same method of line belt transects (Figure 2.12). The survey was carried our along four transects separated by at least 5m, at each site.

These data were then used to calculate coral percentage cover or the percentage of substratum covered by living corals; level of disturbance, abundance and index of diversity. This data also used to measure species richness, species list and percentage of *r-strategist* corals species and Deterioration Index.

2.6.1 Coral percentage cover

Coral cover was estimated as the proportion of area covered with hard coral divided by the total area surveyed, using Line intercept transects (LIT). It was estimated for each transect and averaged for each site.

2.6.2 Coral Abundance

The abundance of hard coral is described in this study by number of hard coral colonies per unit area of reef slope. Number of hard coral colonies was counted at transect level for 4 transect at each site, as this provides representation of degree of presence at each reef.

2.6.3 Species Richness

Species richness is measured by counting the number of species present over a specified area (Dikou & Van Woesik, 2006). The number of hard coral species per transect ($5m^2$) was averaged over sites. The richness was calculated for each transect and averaged for the sites as species per square metre. Species richness is calculated as a density (number per unit area); therefore, the unit area must be approximately the same for all observations.

Number of species was also used to calculate the percentage of *r-strategist* coral groups. Corals of the *r-strategist* group are opportunistic, live in small to medium size colonies; reach sexual maturity early, their success is related to the intensive breeding and the ability to survive different kinds of stress (Sorokin, 1995). The *r-strategists* coral species include *Stylophora pistillata, Pocillopora damicornis, Seriatopora histrix, Psammocora contigua,* and many species of *Montipora, Acropora* and *Pavona*. Percentage of *r-strategist* coral species was determined as a percentage from the total number of coral species present in each site. The data were checked for normality of distribution and homogeneity of variance, and then a Kruskal Wallis test was used to check the difference between sites.

2.6.4 Diversity Index

The number of species present in site or species richness is a simple index of diversity, which does not describe how evenly the total number of individuals is distributed between each species. Shannon's diversity index is one of the most widely used indices to show the allocated proportion of each species. The Shannon-Weaver diversity index (H'_c) was used to further investigate species diversity of the reefs in this study. It is defined as:

$H'_c = -\sum^{g} p_i \ln p_i$, where p_i is the proportion of the i^{th} species in a sample which is multiplied by the natural logarithm of itself. The index, H'_c, is then derived by summing the product for all species, g, in the sample, the minus sign serves to make the final value of H'_c positive. In this study, H'_c was calculated for each transect and subsequently averaged over sites in order to make comparisons at site scale.

2.6.5 Deterioration Index (DI)

The Deterioration Index (DI), which is the ratio between mortality and recruitment rates of hard corals, described by Ben-Tazvi (2004), was also determined at each study site. DI is used to assess the status of the reef by estimating the proportion of small coral colonies (<3cm), which indicate recruitment rate to reef, in relation to the number of dead colonies.

Deterioration index DI is calculated using the following equation:

$$DI = \frac{DC}{DC+LC} \div \frac{SC}{LC}$$

Where DI: Deterioration index.

DC: Number of dead coral colonies.

LC: Number of living coral colonies.

SC: Number of smallest detectable living coral (up to 3cm).

2.6.6 Stability against disturbance

The scale of disturbance here is correlated to colony size for branched and massive corals. The size of the largest colony was measured by two variables: the length of the longest branch of a branching or digitate colony and the diameter of the biggest massive colony in each transect surveyed (Connell, 1978; Done, 1992). These data were then averaged over transects and reefs in order to make comparisons. The use of colony size as an indicator of disturbance emerges from the hypothesis that the less frequent catastrophe or impact, the increase in life span of coral colonies and increase in colony size. The smaller the mean colony size the higher the disturbance level, as the colonies are always destroyed in early stage of growth (Done & Potts, 1992). In this study the size of the largest colonies recorded in each transects represents the magnitude of coral growth occurring over time and the stability of the reef. Coral mortality on reefs is a normal biological process, which

could be accelerated by anthropogenic impacts leading to coral death. Consequently the more frequent the disturbance the smaller the coral colony sizes at a given site. Therefore the living surface area of a coral colony or colony size can be used as an integrated measurement for disturbance intensity and frequency (Connell, 1978; Done, 1992).

- ***Statistical analysis***

Normality was tested using the One-Sample Kolmogorov-Smirnov test, and homogenetiy of variance was checked using Levene's test. Three-way ANOVA ($p<0.05$) was used to examine the differences between years, site and growth form for coral cover and coral disturbance. Mean frequency histograms with standard deviation of mean were plotted. Spearman's correlation was carried out to test for correlation between coral cover and sedimentation, SPM, non- carbonate sand, turbidity, gravel sand and mud. The test was also used to check for correlation between disturbance level and sedimentation, SPM, non-carbonate sand, turbidity, gravel sand and mud.

Two way ANOVA ($p<0.05$) was used to examine the differences between years and sites for coral abundance, species richness, diversity index and DI. Mean frequency histograms with standard deviation of mean were plotted. Pearson correlation tests were carried out to test for correlations between coral abundance, species richness, diversity index and DI with sedimentation, SPM, non- carbonate sand, turbidity, gravel sand and mud. Degree of significance varied from significant where $P < 0.05$ and > 0.01 and marked with one asterisk (*), highly significant if $P<0.01$ and marked with two asterisk (**), very highly significant where $P< 0.005$ and marked (***) and not significant when $P>0.05$ and marked (ns).

Figure 2.11: Line intercept transect (LIT) used for coral cover survey.

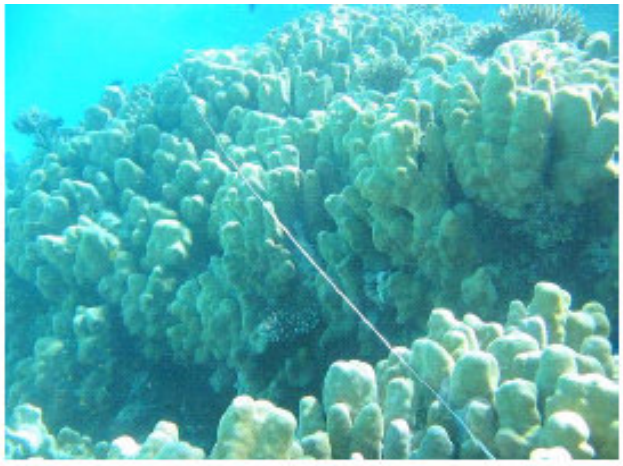


Figure 2.12: Line belt transect (LBT) used for survey number of species, living, dead, recruits colonies and colony diameter.

2.7 Coral Transplantation

Three species of *Acropora* were used for the transplantation study, as they are the most common in the study area. These were *Acropora arabensis, Acropora selago* and *Acropora tenuis* (Figure 2.15,16 and 17). Transplanted coral survival was used as an

indicator of the level of sediment impact, and compatibility of coral transplantation as a tool for rehabilitation of the degraded sites.

This study was carried out in 6 of the test sites. These sites were NIOF (1), Abu Sadaf (2), Shedwan (3), Arabia (4), Holidays (6) and S. Hashish (7). Mature and healthy specimens (Figure 2.13) were chosen from shallow waters (5 m depth) at NIOF. Branches were detached from large colonies at wide intervals. Pliers were used to detach two or three branches from each colony, removal or damage of most of the colony was avoided. The fragments were carefully transferred and placed in the new site where they were tied with plastic cable ties to galvanized steel mesh frames (Figure 2.14), at the same depth as they had been naturally growing. Metal mesh structures were used for artificial reef construction as recommended by Fitzhardinge and Bailey-Brock (1989). Specimens harmed by this procedure were excluded from transplantation. Six samples were fixed to each mesh frame; two branches of each species and five frame replicates were deployed at each site. Samples collected at NIOF and transplanted at NIOF were used as an internal control.

The transplantation process was carried out in April 2004 and survival readings were taken after one month of transplantation. Rinkevich (2000) maintained that transplanted corals survival after one month of transplantation were found to varies greatly between sites with general high indications of survival level. All underwater work was carried out by SCUBA diving. Transplanted corals were resurveyed after one month and number of survived branches was then counted for each species.

- ***Statistical analysis***

Data were checked for normality using One-Sample Kolmogorov-Smirnov test, and homogeneity of variance were checked using Levene's test. Statistical analysis using two way ANOVA ($p<0.05$) was carried out to examine the differences between sites on coral survival of the three species. Mean frequency histograms and standard error of mean were plotted. Spearman's rho correlation tests were carried out to test correlation between coral survival with sedimentation, SPM, non-carbonate sediment percentage, turbidity, and percentage of gravel, sand and mud. Degree of significance varied from significant where $P < 0.05$ and > 0.01 and marked with one asterisk (*), highly significant if $P<0.01$ and marked with two asterisk (**), very highly significant where $P< 0.005$ and marked (***) and not significant when $P>0.05$ and marked (ns)

Figure 2.13: *Acropora selago*, healthy colonies used for sampling.

Figure 2.14: Transplanted coral branches fixed to mesh frame.

Figure 2.15:*Acropora selago* (After Veron and Stafford- Smith, 2002).

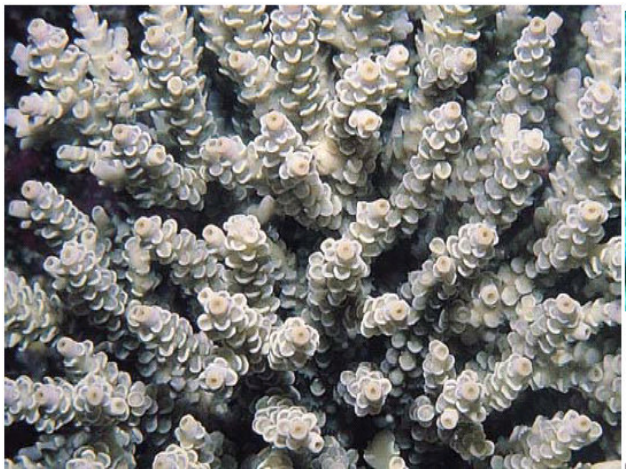


Figure 2.16:*Acropora tenuis* (After Veron and Stafford- Smith, 2002).

Figure 2.17:*Acropora arbensis* (After Veron and Stafford- Smith, 2002).

2.8 Settlement and Recruitment

Settlement tiles were use to measure recruitment rates at each of the study sites. Four 20x20cm cement tiles were deployed at five locations in each site and fixed to steel mesh racks at 20metres intervals. Figure 2.18 and 2.19 show the fixed tiles and racks in position facing the water current at an angle of 45°. The steel mesh rack method used in this study is considered the most common method used for recruitment studies (Reviewed in Mundy, 2000). All settlement plates were placed on the reef for the whole study period from August 2003 to September 2006. The plates were visited frequently to examine any coral recruits on the upper, lower and vertical sides of the plates.

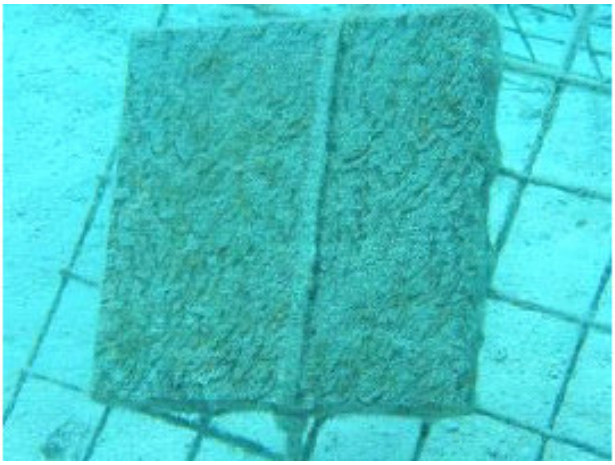


Figure 2.18: Settlement tiles after 6 months of deployment, showing dense algal mates and grazing scares.

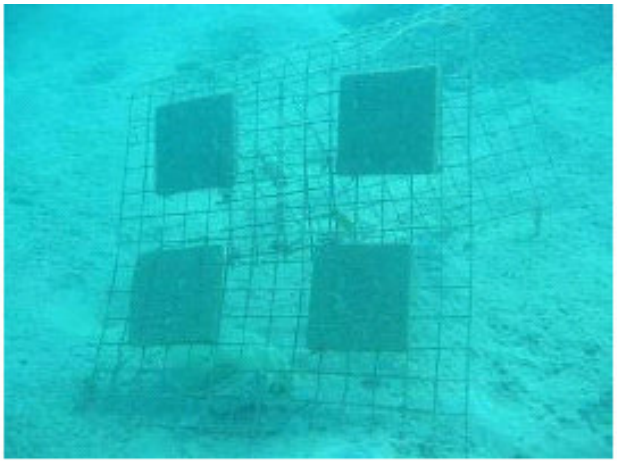


Figure 2.19: Settlement racks and tiles fixed at 45° angle facing the current.

2.9 Mucus Secretion Rates

Rate of mucus secretion under the different sedimentation levels at each site were measured for both branched and massive corals. Branched corals were measured at all of the seven sites, whilst massive corals were measured in all sites except site 6. Site 6 only contained small size massive colonies, which were not comparable with the other sites. The method depends on collecting mucus by enclosing each coral head within a plastic bag with fixed size and surface area (Figure 2.20). The total organic matter is then collected after 24hrs and determined (Richman et al, 1975). Care was taken while placing and removing the bags to avoid damaging the coral heads. Coral heads were selected for uniform size and a staked configuration to facilitate tying and removing the bags (Figure 2.21). much care should taken to avoid mucus dissolving in water. Ten colonies were sampled from each site, 5 samples of both branched and massive coral colonies. At the same time, a water sample was collected near the experimental corals to serve as a control to the coral heads samples. The contents of all of the experimental and control bags were filtered through a gelman type glass fiber precombusted at 450°C for 3hrs. The filters were rinsed with 5ml. of 0.5N H_2SO_4 to remove free carbonate, then with 5ml of distilled water and dried. Total particulate matter and percentage of ash were determined by combusting the filters at 450°C for 3hrs., and determining loss on ignition of the cooled filters using an analytical balance accurate to ±0.01mg (Richman et al, 1975). The amount of particulate matter in the control bags was used as a correction to the experimental bags for particulate matter initially present in the water.

Figure 2.20: Mucus sampling method, coral head enclosed for 24hrs in plastic bag.

Figure 2.21: Coral head selection for uniform size to facilitate tying and removing bags.

Mucus samples were also collected under laboratory condition from three coral species, *Acropora tenuis, Stylophora pistillata* and *Pocilopora damicornis* (Figure 2. 16, 23, 24). Coral colonies were collected at 3-5m depth and transferred to the laboratory in aerated water tank. Coral samples were exposed to 4 different levels of sedimentation in four different aquarium tanks ($5mg.cm^{-2}.day^{-1}$, $10mg.cm^{-2}.day^{-1}$, $20mg.cm^{-2}.day^{-1}$ and $30mg.cm^{2}.day^{-1}$). These sedimentation levels were selected to represent the whole range of sedimentation occur in the field. Each tank was filled with 80 litre of natural seawater, maintained within a temperature range of 22-24°C with a light: dark cycle of 12 hours, as seen in the field. Water circulation was carried out using a water pump to keep sediment suspended and continuous aeration was applied using air pumps (Figure 2.22).

Corals were kept in the aquarium for three days as an acclimatization period before adding sediment. Sediment was applied for 24hrs before enclosing coral colonies in plastic bags and water circulation was stopped while aeration continued. Coral colonies were enclosed with plastic bags for 24hrs to collect mucus. Four replicates of each species were sampled for mucus secretion at each of the four-sedimentation level. Control water samples were collected from each tank, in parallel with the mucus samples. Mucus content was determined using the same method used for the field mucus samples. The three coral species, *Acropora tenuis, Stylophora pistillata* and *Pocilopora damicornis* were shown in Figures 2. 16, 23 and 2. 24.

- ***Statistical analysis***

Field data were checked for Normality using One-Sample Kolmogorov-Smirnov test, and homogeneity of variance were checked using Levene's test. Two way ANOVA ($p<0.05$) was used to examine the differences between sites and growth forms for mucus secretion for field samples. Differences between species and sedimentation level were tested for laboratory mucus samples using two way ANOVA ($p<0.05$). Mean frequency histograms with standard deviation of mean were plotted.

Pearson correlation tests were carried to test the correlation between mucus production of branched and massive corals in the field and sedimentation, SPM, non-carbonate sand, turbidity, and mud. Pearson correlations were carried to test the correlation between mucus production in laboratory and sedimentation rate. Degree of significance varied from significant where $P < 0.05$ and > 0.01 and marked with one asterisk (*), highly significant if

P<0.01 and marked with two asterisk (**), very highly significant where P< 0.005 and marked (***) and not significant when P>0.05 and marked (ns)

Figure 2.22: Water tanks used for laboratory mucus, feeding and zooxanthellae sampling.

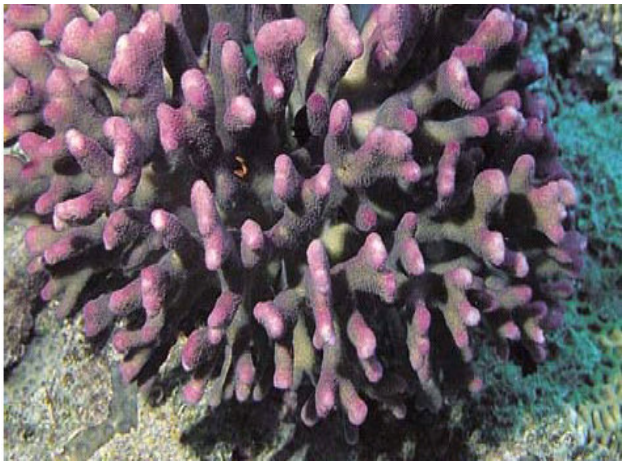


Figure 2.23: *Stylophora pistillata* (After Veron and Stafford-Smith, 2002).

Figure 2.24: *Pocillopora damicornis* (After Veron and Stafford-Smith, 2002).

2.10 Coral feeding trial

The feeding trial depended on using sediment labelled with fluorescein-isothiocyanate which consists of the fluorescent and the isothiocyanate components, making a covalent bond with proteins, peptides and amino acids in the sediment (Rosenfeld et al, 1999). The coral used was one of the common large polyp species, *Lobophyllia hemprichii* (Figure 2.25) in the area. The sediment used in this study was collected at 5m depths from the upper sediment patches at site 4. Half of the sediment was then sterilized, combusted at 450°C for 4 hours. All the sediment (fresh and sterile) then underwent the same labelling process. Sediment was incubated with fluorescein-isothiocyanate (7.7µM) for 12hrs and washed to remove free label. 5ml of sediment was then spread over the water tank, 4 times at 4hrs intervals.

The experiment was carried in two parts following the method of Rosenfeld et al. (1999): The first part depends on a comparison between different kinds of sediment applied (fresh and sterile) and different coral conditions (live and fixed). Samples were divided into six

treatment groups: live coral exposed to fresh labelled-sediment; live coral exposed to sterile, labelled-sediment; live coral with no sediment treatment; fixed coral exposed to fresh labelled-sediment; fixed coral exposed to sterile, labelled-sediment and fixed coral with no sediment treatment. Corals were fixed in 5% glutaraldehyde for 1hr to stop the active feeding by coral. This will rule out the possibility of passive diffusion of fluorescein label into the coral tissue. Four replicates of each treatment were used of both live and fixed corals. Sterile sediment was used to determine the percentage of physically attached fluorescein label to sediment particles in absence of organic matter.

The second part of the experiment depends on a comparison between different levels of sedimentation applied to live corals (10mg.cm^{-2}, 20mg.cm^{-2}, 30mg.cm^{-2} and 40mg.cm^{-2}). Fresh sediment was labeled and applied to live coral using the procedures described earlier. Sediment quantities varied between each treatment according to the concentration multiplied by the surface area of the experimental tank. Four replicates of each of the four treatments were sampled from the four tanks. The experiment was carried out in the laboratory of the Red Sea Protected Area, at the same water temperature of the sea (22-24°C).

Following exposure to sediment for 12hrs, coral tissue was scraped off the skeleton, milled and examine in a spectrofluom to detect the level of fluorescence in each treatment. All four of the replicates from the ten treatments together with blank and standard solutions were quantatively measured by Spectrofluormeter at the University of Hull laboratories and the Suez Canal University.

Figure 2.25: *Lobophyllia hemprichii* (After Veron and Stafford-Smith, 2002).

- ***Statistical analysis***

Data were checked for normality using a One-Sample Kolmogorov-Smirnov test, and homogeneity of variance was checked using Levene's test. One-way ANOVA ($p<0.05$) was used to examine the differences between treatments for each of the two experiments. Mean frequency histograms with standard deviation of means were plotted.

2.11 Zooxanthellae density

Zooxanthellae cells were measured under various sedimentation rates both in the field and in the laboratory. To examine the effects of sediment on zooxanthellae in the field, four colonies of *Acropora tenuis* were transplanted into the seven study sites at 5m depth. Colony samples were collected from site 1 at 5m depths, also control samples were taken from site 1 at the beginning of the experiment. Branch samples from each site were then collected after one and two months respectively. Four vertical branches were collected from the top of each colony and fixed directly in the field, in 10% formalin solution of seawater. Samples were labelled and kept for further decalcification.

In a follow up laboratory experiment coral zooxanthellae samples were collected from three coral species (*Stylophora pistillata, Pocilopora damicornis* and *Acropora tenuis*). Healthy coral colonies (4 colonies) of the three species were collected from the field and carried to the aquarium in the laboratory of the Red Sea Protectorates. Four levels of sedimentation (5mg/cm²/day, 10mg/cm²/day, 20mg/cm²/day, 30mg/cm²/day) were applied to four separate aquaria. Sediment used in this experiment was collected from Site 4 study site. Coral colonies were left for three days in the aerated tanks, before applying sediment. Sediment was applied for each treatment daily at the water surface; the amount of sediment was calculated according the surface area of each tank. Water in each aquarium was replaced at the rate of 25% every day. Temperature was kept around 23°C, salinity around 41 (as in natural sea) and light/dark illumination period of 12hrs was used to follow field conditions. The aquarium contained 80litres of natural seawater, circulation was carried out using a water pump to keep sediment suspended and continuous aeration was applied using an air pump. Coral branches were then collected (4 branches) at the beginning of the experiment (before sediment addition) and after one week, two weeks and four weeks of sediment addition. Sample branches were then preserved in 10% formalin solution of seawater for 24hrs before decalcification.

Coral branches were decalcified in the laboratory in formic acid/sodium citrate solution. 50% formic acid solution in distilled water was added to 40% sodium citrate solution in distilled water (400gm dissolved in 1litre of water), in 1:1 proportion to make the decalcification solution (Amer, 2005). This was added to the coral branches and left for 24hrs, then washed and preserved again in 25% opersol solution. The glove-like tissue from each branch was cut longitudinally, opened and laid flat in a Petri dish containing water. A flat $1cm^2$ of tissue was cut, 1-2cm below the tip of the branch in order to control the position at which algal densities were estimated. The tissue was then ground in an homogenizer with 2ml of preservation solution (opresol 25%). The zooxanthellae cells showed high concentration when examined under microscope; consequently the samples were diluted at 1:6 ratio by adding 12ml of preserving solution. The number of cells was counted on a haemocytom, five cells count of the number of zooxanthellae per 1.0 mm³ were performed (0.005 of the total haemocytom cell content). This process was repeated 6 times, averaged and then multiplied by 14 X 10³ to convert the haemocytom counts to cells per cm² of coral tissue. The counting process was repeated from the four different coral branches to take the mean number of cells as the actual zooxanthellae number per cm² of coral tissue.

- ***Statistical analysis***

A One-Sample Kolmogorov-Smirnov Test was carried out to check that the data were normally distributed, and homogeneity of variances was checked using Levene's Test. Two way ANOVA ($p<0.05$) was used to examine the differences between sites and period

(one and two months) in the field experiment and three way ANOVA ($p<0.05$) for the difference between species, period and sediment concentration in the laboratory experiment. Mean frequency histograms with standard deviation of the mean were plotted. Pearson correlation tests were carried between field data and sediment, SPM, non-carbonate sediment, turbidity, gravel sand and mud. Degree of significance varied from significant where $P < 0.05$ and > 0.01 and marked with one asterisk (*), highly significant if $P<0.01$ and marked with two asterisk (**), very highly significant where $P< 0.005$ and marked (***) and not significant when $P>0.05$ and marked (ns).

2.12 Bioerosion rates and Bioeroding community

This study was undertaken to assess the distribution and abundance of the four main macroboring groups (sponges, bivalves, polychaetes and sipunculids) and the degree of framework infestation across the reefs of the study sites, following the method of (Macdonald and Perry, 2003). Macroborer sampling was carried out along five 10m transects at 5m depth from points at fixed intervals as possible, together with sediment trap in all sites. Recently dead coral fragments were collected and carried to the laboratory for further investigation. 50 samples of dead in-situ coral and coral rubble were collected from each of the seven reef sites. Specimens were selected that appeared to have undergone relatively recent mortality and which were not heavily degraded. Each sample was cut into parallel slab sections. The cut surfaces were then traced to record the number of pores for each group. Identification of borers was based on published tissue and borehole descriptions for the Caribbean region (e.g. Perry, 1998). Prior to washing and tracing, and to facilitate sponge identification, each slab was examined to determine sponge tissue coloration.

- ***Statistical analysis***

Data were checked for normality using One-Sample Kolmogorov-Smirnov test, and homogeneity of variances were checked using Levene's test. Two way ANOVA ($p<0.05$) was used to examine the differences between sites and boring groups. Mean frequency histograms with standard deviations of means were plotted. Pearson correlations were carried out between each of the boring groups and sediment, SPM, turbidity, non-carbonate sediment, gravel sand and mud. Degree of significance varied from significant where $P < 0.05$ and > 0.01 and marked with one asterisk (*), highly significant if $P<0.01$ and marked with two asterisk (**), very highly significant where $P< 0.005$ and marked (***) and not significant when $P>0.05$ and marked (ns)

2.13 Abundance of 6 common reef fish families

Abundance of the major coral feeding and algal feeding reef fish groups, were surveyed in the seven study sites. Fish species composition is expected to change under increased sedimentation and subsequent coral deterioration. In order to identify the fish population composition of the study reefs, six common families were chosen, representing a wide range of species of different sizes and trophic affiliations. Fish groups were surveyed along four 50m× 5m transect at each site. The coral feeding (corallivorous) were the butterfly (Chaetodontidae), angel (Pomacanthidae), and wrasse (Labridae) and the three algal feeding were the damsel (Pomacentridae), surgeon (Acanthuridae) and rabbitfishes

(Siganidae). Survey was carried out in the day time from 11am to 4pm. The transect tape was laid on the reef and left for 15minutes before the survey to allow the fish to regroup. The number of individual fish of each family were recorded along the 50m transect. The surveys were carried out twice, in 2004 and 2006. The butterfly fish group was identified to the species level during 2006 survey.

- ***Statistical analysis***

Data were checked for normality using the One-Sample Kolmogorov-Smirnov test, and homogeneity of variances was checked using Levene's test. Three-way ANOVA for fish abundance as the variable against year, site and feeding habits (algal feeding and coral feeding) were performed. Pearson correlation tests were carried out between first year fish abundance and sediment, SPM, non-carbonate, turbidity, gravel, sand and mud. Mean frequency Histograms and standard deviation of means were plotted for each fish group. Degree of significance varied from significant where $P < 0.05$ and > 0.01 and marked with one asterisk (*), highly significant if $P < 0.01$ and marked with two asterisk (**), very highly significant where $P < 0.005$ and marked (***) and not significant when $P > 0.05$ and marked (ns). Cluster analysis was made for coral reef fish families surveyed in the two years (2004, 2006) using Multivariate Statistical Package (MVSP), to show the similarities between sites.

Figure 2.26: Butterfly fish, *Chaetodon fasciatus*, usually live close to coral reefs; photos were taken during the survey.

Figure 2.27: Two species of Butterfly fish, *Chaetodon paucifasciatus* and *Chaetodon austriacus*, during feeding.

2.14 Water Quality

Each of the seven sites was sampled for physico-chemical parameters and nutrients using a small boat and taking the readings close to the reefs of each study site.

2.14. a Physico-Chemical Characteristics

Physico-chemical characteristics of the water were measured in the field using a multprobe (HORIBA) at 3 depths 1, 3 and 5m. Parameters measured were; temperature (^{o}C), salinity (‰), depth in m, pH, specific conductivity (SPC) in (µmhos/cm), dissolved

oxygen (DO) in (mg/L), total dissolved salts (TDS), percentage of dissolved oxygen (DO%) and turbidity in (NTU, nephelometric turbidity unit). The readings were taken once a year, for the years 2004 at April and 2006 at January. The turbidity levels were related to SPM levels measured in the field.

2.14.b Nutrients Analysis

Five major nutrient elements were measured in seawater samples, inorganic phosphate, silicate, ammonia, nitrite and nitrate. Water samples were collected in the field using clean polyethylene stoppered bottles (500ml) that were rinsed with seawater from the site before sampling. Four subsurface water samples were collected from each site and then kept iced on the boat in an icebox and taken immediately to the laboratory within two hours, for analysis. Samples for inorganic nutrient analysis were filtered in the laboratory to remove any suspended matter. The methods used to study water nutrients are published in the UNEP series of reference methods for marine pollution studies (UNEP, 1991). The nutrient samples were collected and examined once, in January 2006.

2.14.b 1 Ammonia-Nitrogen (NH_4)

The procedure outlined here follow the methods described by Grasshoff and Johanson (1973) and by Koroleff (1983a). The method depends on the formation of the blue coloured indophenol by phenol and hypochlorite in the presence of NH_4 and NH_3. The reaction of alkaline phenol and hypochlorite with ammonia form indo-phenol blue that is proportional to ammonia concentration. The color was measured at 630nm and is stable for at least 30hrs. $2cm^3$ of phenol reagent, $1cm^3$ buffer solution and $2cm^3$ hypochlorite reagent were added to $50cm^3$ of the sample, mixed well and kept in the dark for 6hrs. The absorbance was measured after 6hrs at 630nm and distilled water was used as a reference.

2.14.b 2 Nitrite-Nitrogen (NO_2)

Nitrite-nitrogen was determined by following the methods of Bendschneirder and Robinson (1952) and Grasshoff (1983). The technique is based on the formation of a highly coloured azo dye, which is measured colorimetrically at 540nm. The photometric detection of nitrite is based on the reaction of nitrite with an aromatic amine (sulphanilamide) which leads to the formation of a diazonium compound at pH=1.5-2.0. this diazo compound coupled with a second aromatic amine (N-(1-naphthyl)-ethylene diamine) forms the azo dye.

$1cm^3$ of the sulphanilamide reagent was added to $50cm^3$ of the water sample and mixed well, then after one minute $1cm^3$ of the diamine solution was added. The flask was then shaken well and left for 30 minutes to allow the azo dye to develop. The absorbance was measured in a cell of suitable length at 540nm against distilled water as reference.

2.14.b 3 Nitrate-Nitrogen (NO_3)

Nitrate was determined using the Cadmium Reduction Method as described by Grasshoff (1983). This method depends on the quantitative reduction of nitrate to nitrite in the presence of cadmium. Commercially available cadmium granules treated with copper sulphate ($CuSO_4$) were used as a reducing agent. The NO_2 produced thus was determined colorimetrically. Determination of Nitrate (NO_3) is based on its reduction to nitrite; in a reduction column filled with copper-coated cadmium granules, which was then determined calorimetrically at 540nm. The method determines the sum of nitrite and

nitrate, therefore a separate determination of nitrite must be conducted, and the concentration subtracted from that obtained with this method. The analysis should start within 1hour of subsampling.

25cm^3 of the buffer solution was added to 25cm^3 of the sample and mixed well. About 20cm^3 of the mixture were passed through the reduction column to rinse the system and adjust the time of passage (3-5min). Another fraction was passed through the column until the 25cm^3 level was reached. To this sample 0.5cm^3 of the sulphanilamide reagent and 0.5cm^3 of the diamine solution were added in the same way as described for the analysis of nitrite. The azo dye colour was determined within about one hour (at 540nm) against distilled water as a reference.

2.14.b 4 phosphate (PO_4)

Using Ascorbic Acid, inorganic phosphate in seawater was analyzed by following the colorimetric method outlined by Koroleff (1983c). Samples were analyzed as soon as returned to the laboratory. 1.5cm^3 of ascorbic acid solution and then 1.5cm^3 of mixed reagent (Ammonium para-molybdate, ascorbic acid, sulphuric acid and potassium antimonyl tetrate) was added to 50ml sample. Another 50ml sample represents the turbidity blank, to which 1.5cm^3 of ascorbic acid solution was added. After 10 minutes the absorbance of the sample and the turbidity blank was measured at 882nm against acidified distilled water. A reagent blank from the same volumes of distilled water and reagents was prepared, the standard concentrations against the absorbance was plotted. The concentration of phosphate was determined from a standard linear curve as follow:

Absorbance=b*concentration. Where b is the slope of the calibration curve

Phosphate stock solution was prepared by drying potassium dihydrogen-phosphate at 110 ^oC, dissolving 1.361gm in distilled water and diluting to 1000cm^3. A working solution was prepared by diluting 10cm^3 of stock solution with distilled water to 1000cm^3. A series of working standard solution was then prepared by adding distilled water to a series of 0.5, 1.0, 2.0, 3.0, 4.0 and 5.0cm^3 of working solution to 100cm^3. These gave concentrations of 0.5, 1.00, 2.00, 3.00, 4.00 and 5.00µmol dm^{-3} PO_4^{3-}.

2.14. b 5 Soluble Reactive Silicate (SiO_3)

Reactive silicate was determined using silicomolybdate method according to standard methods (Koroleff, 1983b). To 50cm^3 of sample, a 1.5cm^3 of mixed reagent was added in a plastic reaction bottle and the well-mixed sample was allowed to stand for 15minutes. Then 1cm^3 of oxalic acid was added, followed immediately by 1cm^3 of ascorbic acid after well mixing. The sample was left to cool at room temperature for 30minutes then the absorbance was measured at 810nm against distilled water as a reference. Distilled water usually contains detectable amounts of silicate so the reagent blank for analysis of the sample was determined as follow. A reagent blank from the same volumes of distilled water and reagents was prepared, only 1cm of the mixed reagent were added, then after reduction 0.5cm^3 distilled water mixed with 1.0cm^3 reagent were then added. The standard concentrations against the absorbance were plotted.

Absorbance=b*concentration. Where b is the slope of the calibration curve

CHAPTER 3

SEDIMENT AND WATER QUALITY RESULTS

Introduction

Sedimentation impacts corals in several ways; it can cause coral death; it can decrease adult coral growth; it can depress zooxanthellae densities, and increase respiration and mucus production, it can reduce coral reproduction and coral larval settlement and survival (Rogers, 1990). These effects were considered the most common impacts descriped by researchers and they are thought to act together to limit coral reef development and growth (Cortes and Risk, 1985; Hubbard, 1986). Development in Hurghada's coastal area have been shown to continue and increase sediment input to sea water. Sedimentation and turbidity levels in the coastal area have seen a marked increase at many sites which coinside with reef deterioration. There are no detailed studies addressing sedimentation impact on reefs in the region and no sediment records for Hurghada reefs. The needs for the data about sedimentation rate, SPM, turbidity, nutrients in seawater and physico-chemical parameters was raised after the apparent decline in corals and fish in this area. This part of study tries to answer many questions including: what the sedimentation levels in this area are, whether it is increasing or decreasing, what the source of this sediment is and to what degree these sedimentation levels impact coral reefs. This study also examines correlations between sedimentation rate and SPM, turbidity and nutrients in seawater. The environmental parameters measured in this study include sedimentation rate, SPM, turbidity, non-carbonate sediment and percentage of gravel, sand and mud in bottom sediment. In addition, beach sediment and bottom sediment were microscopically and chemically examined for the percentage of biogenic and non-carbonate sediment. Major nutrients including NH_4, NO_3, NO_2, SIO_3 and PO_4 were measured of sea water at the seven study sites in January 2006, correlations with sedimentation rate, SPM, non-carbonate, turbidity and percentage of gravel sand and mud were tested. Water quality, includng SPC, DO, PH, TDS, DO%, turbidity and salinity were determined and tested for correlations with sedimentation rate, SPM, non-carbonate, turbidity, percentage of gravel sand and mud.

3.1 Sedimentation Rates

A wide range of sedimentation rates were determined between sites and between the two years of analysis 2004 and 2006 (Figure 3.1). The first year sedimentation readings showed that the mean highest sediment level was recorded at site 3 and the lowest at site 7. The mean values of sedimentation around the year (Figure 3.1) reveals that site 3 had the highest rate and was markedly higher than all of the other sites. Site 1, 2 and 5 also had close values with slight differences between each other. Site 7 showed a marked reduction in sedimentation compared with the rest of the study sites. Mean sedimentation values for the whole of the 2004 sampling period ranged from 0.4 $mg.cm^{-2}.day^{-1}$ at site 7 to 7.12 $mg.cm^{-2}.day^{-1}$ at site 3. Seasonal variation in 2004 sedimentation rates indicated that the highest rates were in Aug, principally in both site 3 and 4. Sites 2 and 5 had the higher

levels in October, and both sites 1 and 6 were higher in April than the rest of the year. The mean lowest sedimentation rate was found in June (Figure 3.2).

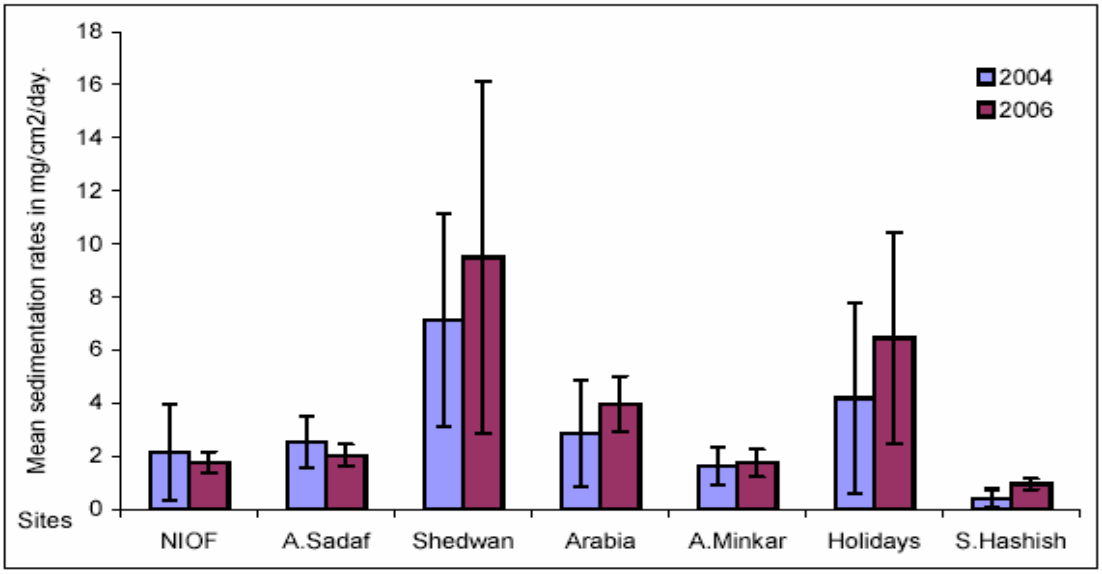
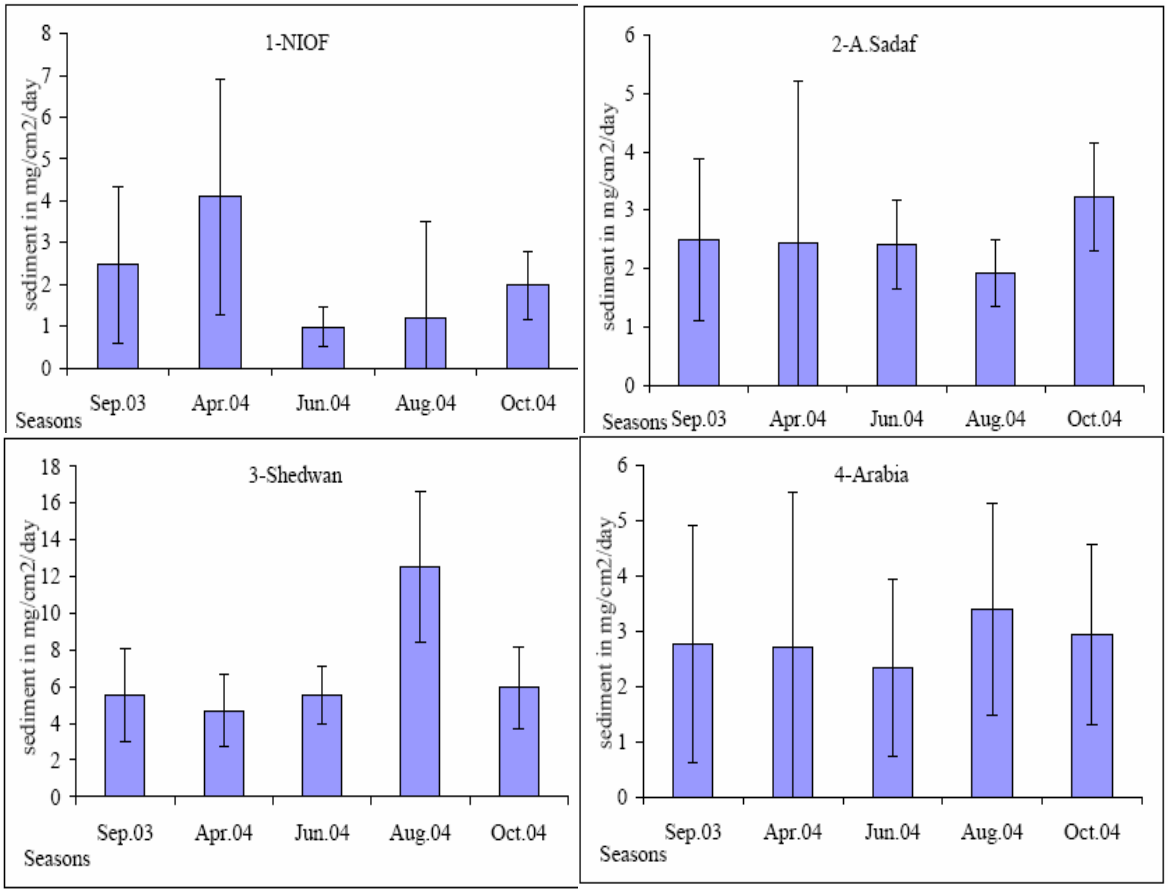


Figure 3.1: The mean sedimentation rate (in $mg.cm^{-2}.day^{-1}$) recorded between the seven study sites along the coast of Hurghada, Egypt, in 2004 and 2006 ± SD standard deviation.

Sedimentation readings in 2006 showed the same trend as those in the first year with little difference (Figure 3.1); the highest sediment level was recorded at site 3 and 6, the lowest was at site 7. There was a general increase in the rate of sedimentation in 2006 compared with 2004 at most sites. In contrast to 2004 the mean highest sedimentation rates were found in February and the lowest were in August. Figure 3.3 illustrates the differences in sediment level between the twelve months of sediment sampling for 2006.



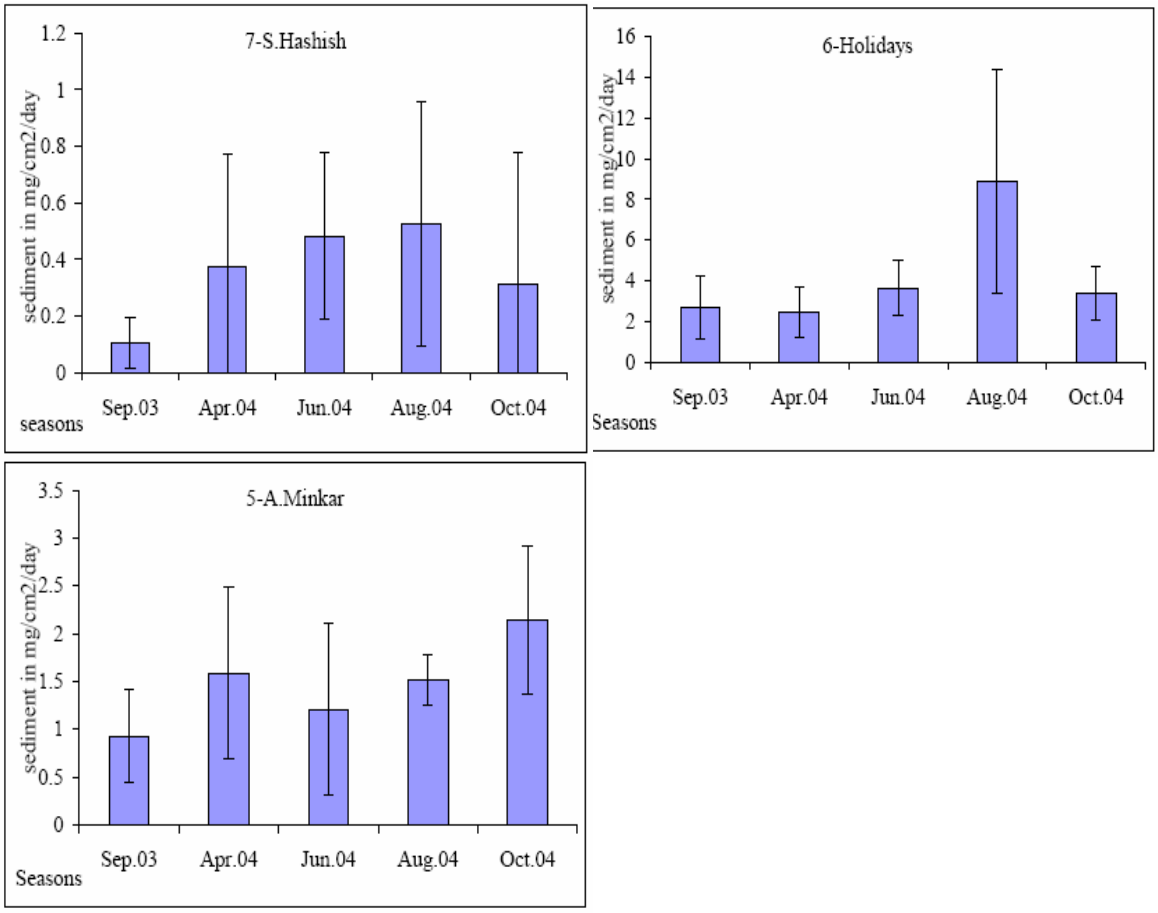
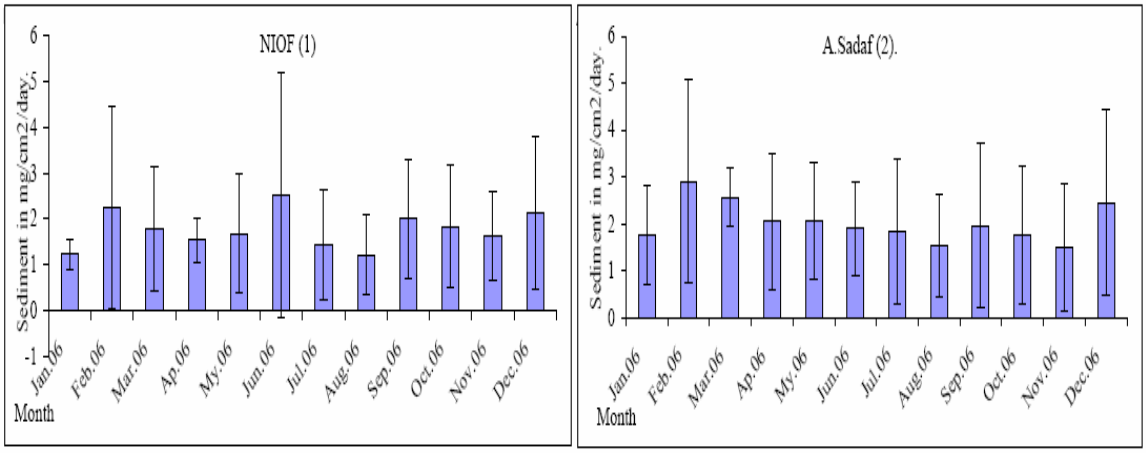


Figure 3.2: The mean sedimentation rate (mg.cm^{-2}.day^{-1}) recorded for the five seasons of 2004 survey (Sep. 2003, April 2004, June 2004, August 2004 and Oct. 2004) at the seven study sites, along the coast of Hurghada, Egypt.

Two way ANOVA for the sedimentation rate against both sites and years (Table 3.1) showed a significant difference between sites (F= 48.94, df=6, p=0.000) and years (F= 17.18, df=1, p=0.000). A post-hoc Scheffe test showed that site 1 and 2 were significantly different from sites 3, 4 and 6; sites 3, 4 and 6 were significantly different to all other sites. Site 5 and 7 were significantly different from sites 3, 4 and 6, as shown in Figure 3.1.



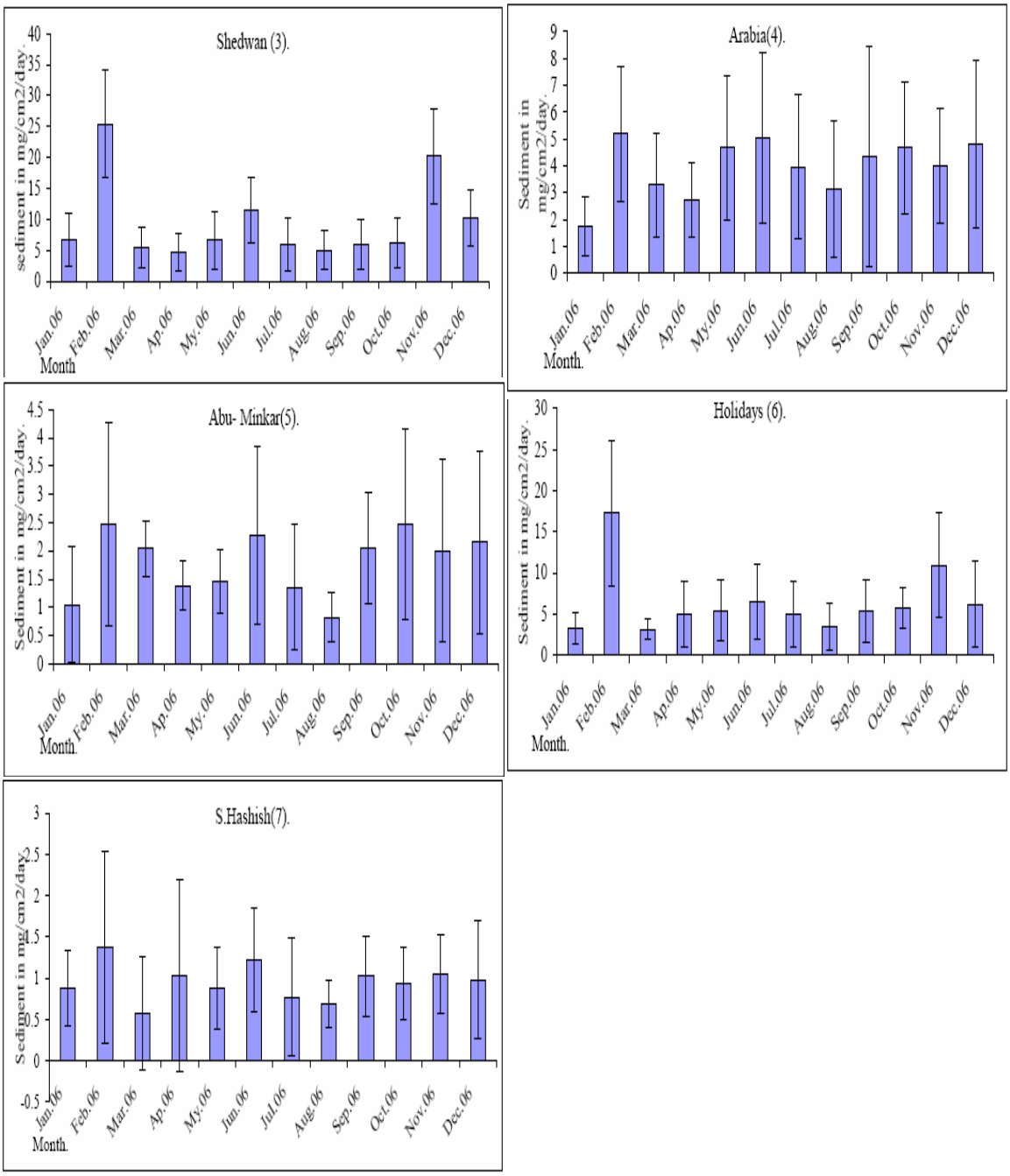


Figure 3.3: The mean sedimentation rate (mg.cm^{-2}.day^{-1}) between the monthly readings of 2006 survey (twelve months) at the seven study sites along the coast of Hurghada, Egypt ± SD standard deviation.

Table 3.1: The two way ANOVA test for sedimentation rate variation between sites and between the two years 2004 and 2006, for each sampling site; expressed in $mg.cm^{-2}.day^{-1}$ for the seven study sites, Hurghada, Egypt.

Source	S.S	d.f.	M. S.	F	P value
Site	3754.621	6	625.770	48.943	.000
Year	219.760	1	219.760	17.188	.000
Site * Year	567.347	6	94.558	7.396	.000
Error	15508.961	1213	12.786		
Total	37841.886	1227			

Table 3.2: The significance matrix between sites in the sedimentation rate, measured in 2004 and 2006 at the 7 study sites, Hurghada, Egypt.

Sites	1	2	3	4	5	6	7
1							
2							
3	x	x					
4	x	x	x				
5			x	x			
6	x	x	x	x	x		
7			x	x		x	

3.2 Suspended Particulate Matter (SPM)

Mean suspended particulate matter (SPM) readings recorded in 2004 and 2006 mainly followed the same trend of sedimentation between sites and the sites with the higher levels of sedimentation also has the higher levels of SPM (Figure 3.5). Site 3 has the highest SPM readings (19.18-21.11mg/l) and site 7 is the lowest (1.6-2.07mg/l) in both years (Figure 3.4). There was no huge difference between 2004 and 2006 readings, with some sites recording almost the same values for both years (sites 1, 5 and 7). Sites 2, 3, 5, 6 and 7 indicated increased SPM levels in 2006 than 2004 while sites 1 and 4 indicated lower SPM levels in 2006 than 2004.

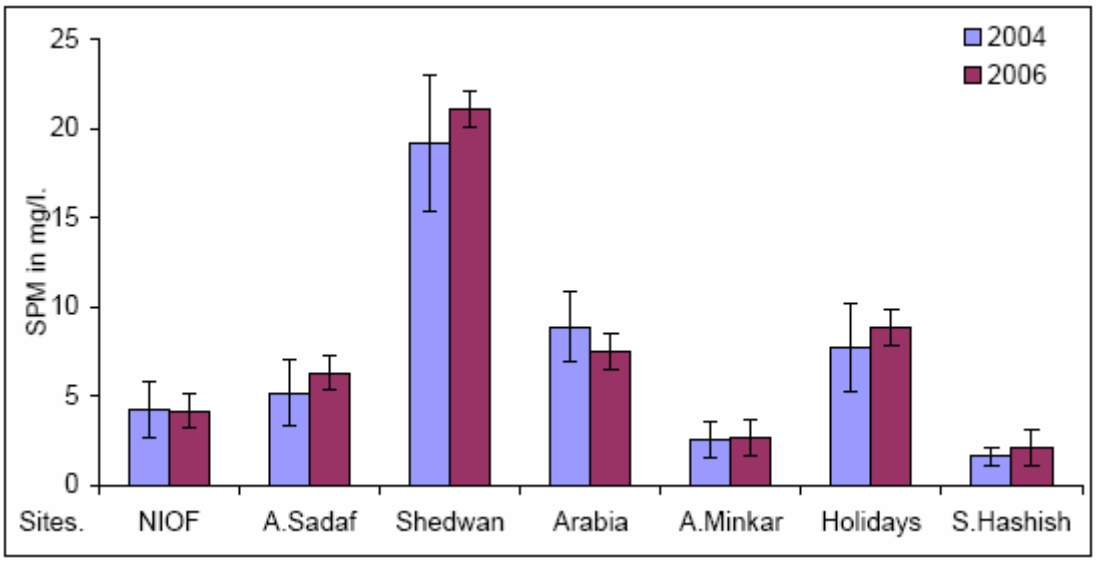


Figure 3.4: The mean SPM concentrations (in $mg.l^{-1}$) at the seven sites along the coast of Hurghada, Egypt in 2004 and 2006. Data were presented as mean ± SD standard deviation.

A Two way ANOVA test was carried out for SPM between sites and years. Results (Table 3.3) showed a strong significant difference between sites (F=17.35, df=6, P=0.000) but not between years (F=0.588, df=1, P=0.447). A post-hoc Scheffe test showed that site 1 was significantly different from site 3; site 2 was significantly different from sites 3 and 7; site 3 was significantly different from all other sites; site 4 was significantly different from sites 3 and 7; site 5 was significantly different from sites 3; site 6 was significantly different from sites 3 and 7; and site 7 was significantly different from sites 2, 3, 4 and 6 as showed in Table 3.4.

Table 3.3: Two way ANOVA testing SPM between sites and between the two years, 2004 and 2006 for each sampling site; in Hurghada, Egypt.

Source	S.S	d.f.	M. S.	F	P value
Site	1499.117	6	249.853	17.351	.000
year	8.474	1	8.474	.588	.447
Site * year	54.483	6	9.081	.631	.705
Error	604.813	42	14.400		
Total	5155.940	56			

Table 3.4: The significance matrix between sites in the SPM, measured in 2004 and 2006 at the 7 study sites, Hurghada, Egypt.

Sites	1	2	3	4	5	6	7
1							
2							
3	x	x					
4			x				
5			x				
6			x				
7		x	x	x		x	

The Pearson correlation tests between sedimentation and SPM (Table 3.5), showed significant positive correlations between 2004 sedimentation rate and SPM (R= 0.962, p=0.000), and between 2006 sedimentation rate and SPM (R= 0.937, p=0.001). The sites with highest SPM levels also had the highest sedimentation rates. However, it should be emphasized that the correlation or association does not necessarily mean that one variable causes the other. There might be some other factor that affects both of them. A scattergram relation between sediment and SPM (Figure 3.5a,b) was examined for the two years.

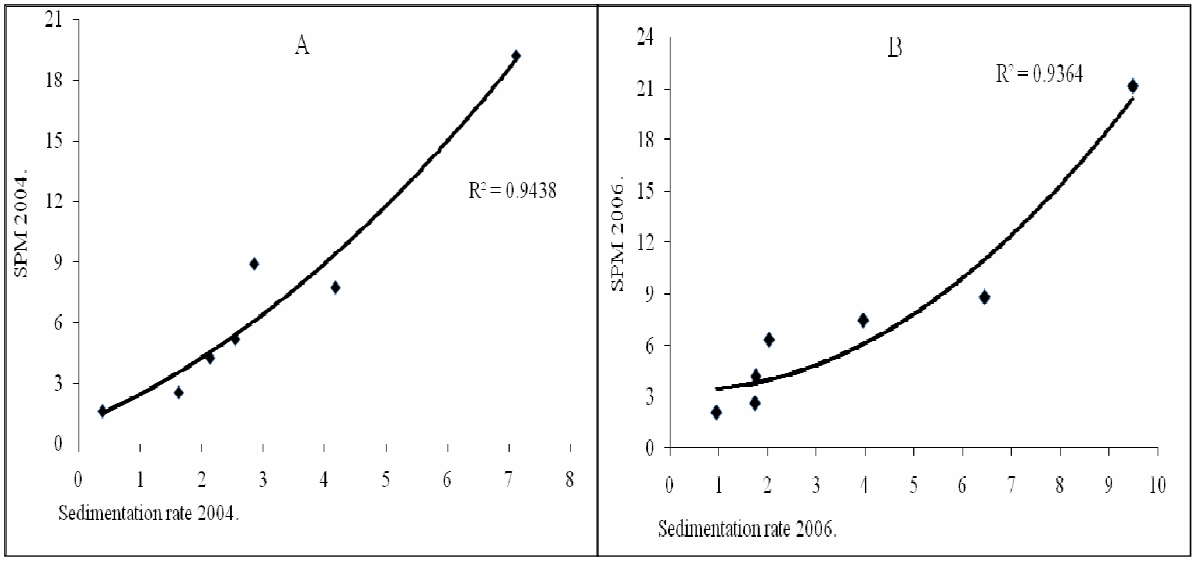


Figure 3.5: The degree of correlation between mean sedimentation rate and SPM for 2004 (A), and 2006 (B), and trend line and R^2 value for correlation as measured at the seven sites in Hurghada, Egypt.

Table 3.5: Pearson correlation test for correlations between sedimentation rate and SPM of the two years for each sampling site, Hurghada, Egypt.

		2004 SPM	2006 SPM
2004 Sediment	Pearson Correlation	.962	.973
	Sig. (1-tailed)	.000(***)	.000(***)
	N	7	7
2006 Sediment	Pearson Correlation	.937	.944
	Sig. (1-tailed)	.001(***)	.001(***)
	N	7	7

3.3 Sediment Type and Particle Size Analysis

A particle size analysis of deposited bottom sediment was performed to determine mean grain size, sorting and percentage, of gravel sand and mud (Table 3.6). In addition, beach sediment and bottom sediment were microscopically examined for the percentage of biogenic and non-carbonate sediment. Figure 3.6 showed great variations between sites in percentage of biogenic and non-carbonate sediment for both beach and bottom samples.

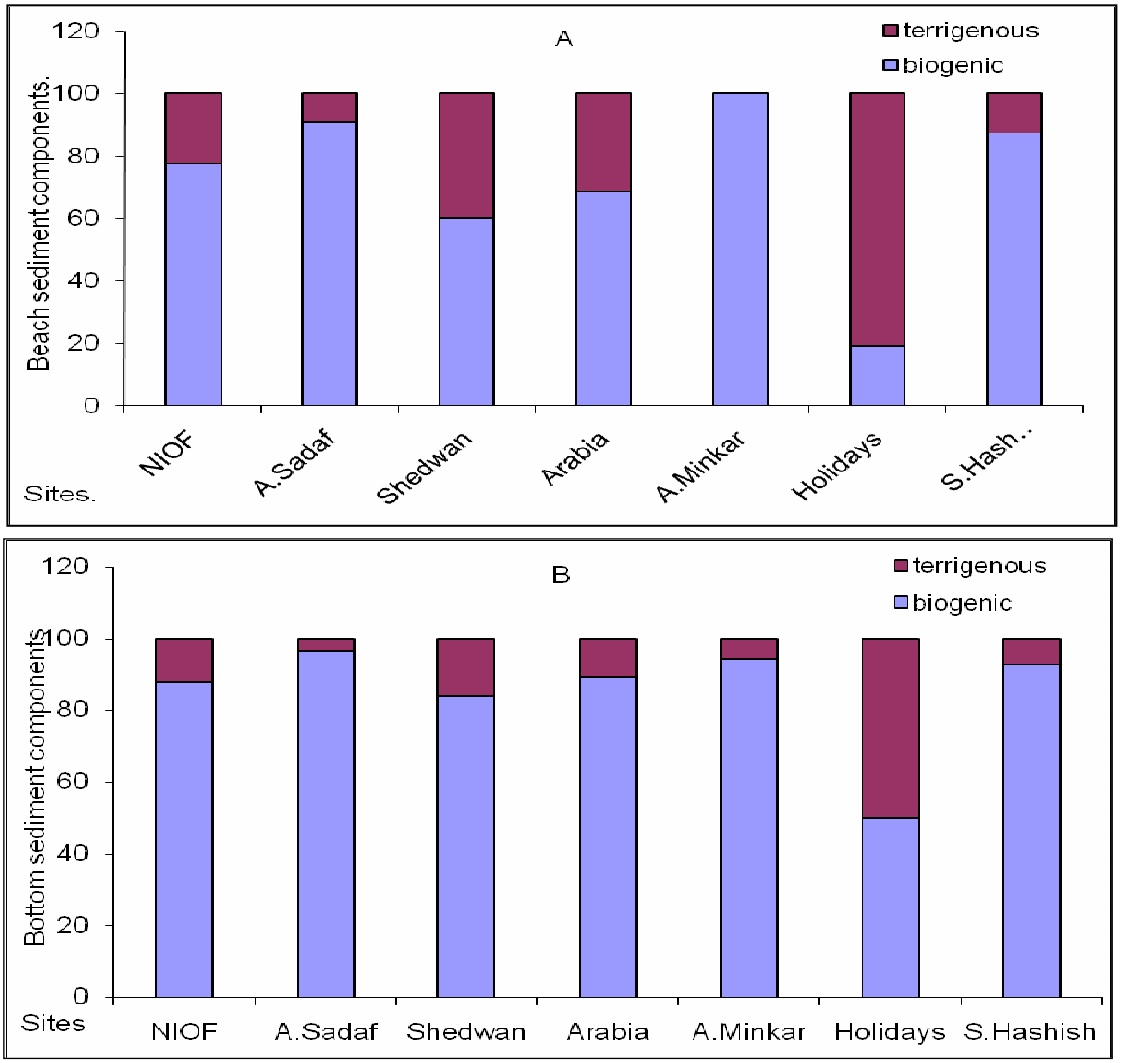


Figure 3.6: The ratio between carbonate and non-carbonate sediment components of both (A) beach and (B) bottom sediment, collected at 2006 from all sites, Hurghada, Egypt.

Sediment origin (biogenic or non-carbonate) identification using HCL showed big differences between sites in the percentage of non-carbonate sediment. Microscopic identification of sediment particles using thin section analysis indicated various kinds of particles including in both terrigenous and biogenic components. Terrigenous sediment was composed mainly of quartz and feldspar, while biogenic sediment was mainly foraminifera, corals and shell fragments. Other components such as sponge, crustacea worm tubes and algae were extremely rare. The microscopic test showed that some sites contained a high percentage of non-carbonate sediment reaching about 50% at site 6. The rest of the sites consisted mainly of biogenic sediment and the lowest readings of non-carbonate sediment were recorded at site 2 and 7.

The differences in sediment composition between beach and bottom sediment are shown in Figure 3.7. The ratio of carbonate to non-carbonate sediment in both beach and bottom sediment showed great variation between sites (Figure 3.6) and between the beach and bottom samples (Figure 3.7). The highest percentage of non-carbonate sediment (50.5%) was found at site 6 and the lowest was at sites 2 (3.6%) and site 7 (5.2%). The percentage of non-carbonate sediment of the two year (2004 and 2006) showed general higher levels at 2004 (Figure 3.8) for most of the sites (2, 3, 4, 5, 6), while sites 1 and 7 had higher percentage for 2006.

At all sites the sediment was mainly sandy with various degrees of sorting (Table 3.9). Site 6 had the highest percentage (23.25%) of fine sediment (mud), while site 1 has the lowest percentage (0.4%). Sediment grain size was mainly medium with various degrees of sorting varying from moderate at sites 1, 5 and 7; to bad at site 2, 3 and 4; with the most poorly sorted at site 6.

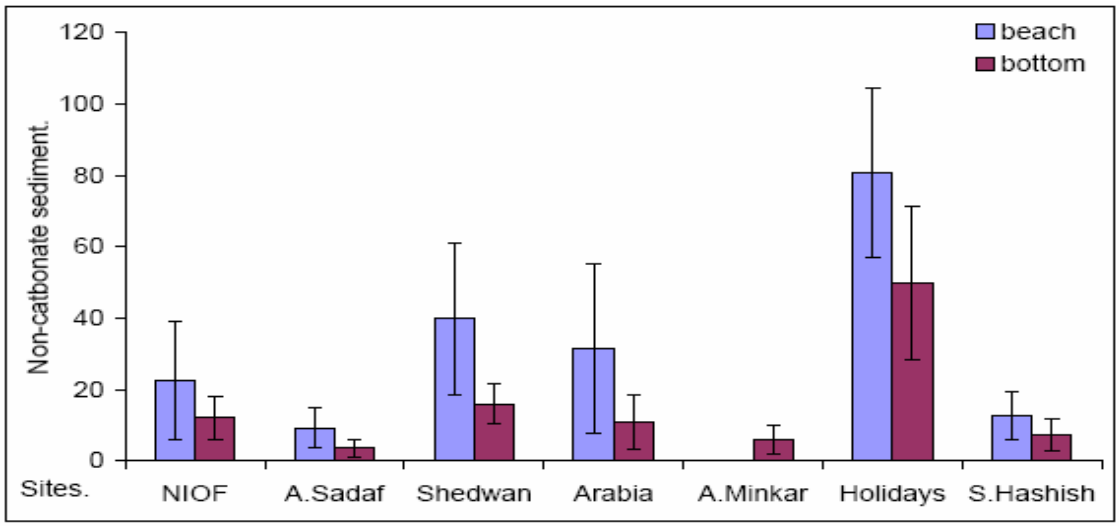


Figure 3.7: The difference between the mean percentage of non-carbonate sediment of both beach and bottom samples ± SD standard deviation, for each of the study sites, Hurghada, Egypt.

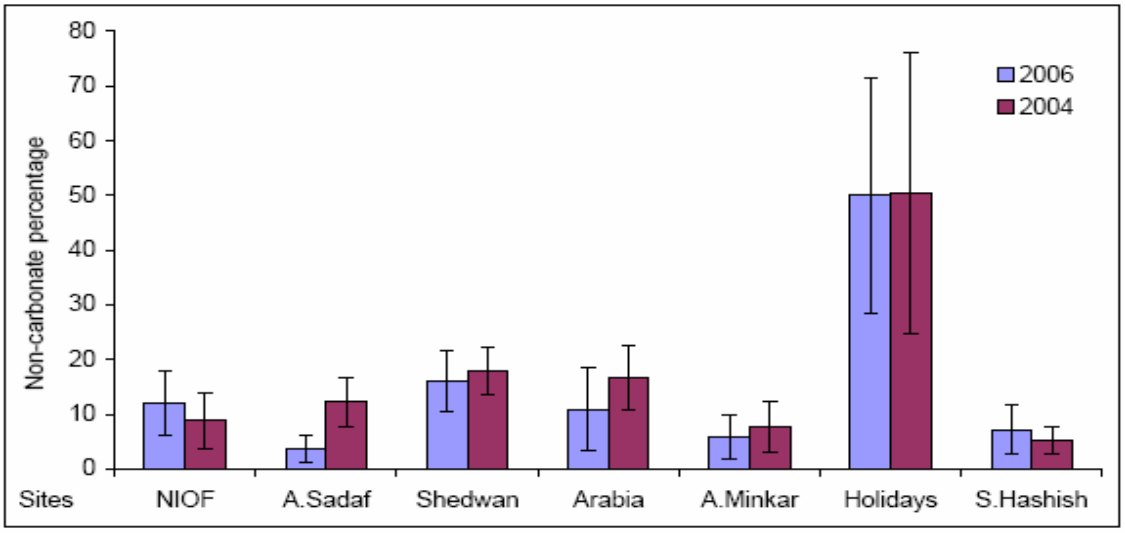


Figure 3.8: The difference between the mean percentage of non-carbonate sediment ± SD standard deviation of mean for the two years (2004 and 2006) samples, using chemical and microscopic test for sediment samples form each of the study sites, Hurghada, Egypt.

Two way ANOVA (Table 3.6) was also used to test differences between the two years (2004 and 2006) readings of non-carbonate sediment percentage. The results indicate a significant difference between sites (F=19.18, df=6, P=0.000) but not between the two years (F=0.54, df=1, P=0.46). A Post-hoc Tukey test showed that sites 1, 2, 3, 4, 5 and 7 were significantly different from sites 6 (Table 3.7). A Spearman's Correlation test (Table 3.8) indicated no significant correlation between non-carbonate sediment level at each site and both the mean sedimentation rate and SPM for 2006 readings. The same correlation test showed a significant positive correlation between 2004 non-carbonate sediment and all of sedimentation rate (R=0.964, P= 0.000) and mean SPM (R=0.893, P=0.003) as showed at figure 3.9. It is important to remember that a significant correlation does not imply a cause and effect relationship, only that there is an association between the two

variables and that there is a less than 5% probability that this association occurred by chance.

Table 3.6: Two way ANOVA test of non-carbonate sediment differences between the two years (2004 and 2006) for each of the seven study sites, Hurghada, Egypt.

Source	S.S	d.f.	M. S.	F	P value
Year	54.609	1	54.609	.540	.467
Site	11652.860	6	1942.143	19.187	.000
Year * Site	209.703	6	34.950	.345	.909
Error	4251.208	42	101.219		
Total	30603.750	56			

Table 3.7: The significance matrix between sites in the non-carbonate sediment percentage, measured at 2004 and 2006 at the 7 study sites, Hurghada, Egypt.

Sites	1	2	3	4	5	6	7
1							
2							
3							
4							
5							
6	x	x	x	x	x		
7						x	

Table 3.8: Spearman's rank correlation of non-carbonate sediment percentage of the two years (2004 and 2006), and both of sedimentation rate and SPM of the two study years (2004 and 2006), for the seven sites, Hurghada, Egypt.

			Sediment 04	SPM 04	Sediment 06	SPM 06
Spearman's rho	Non-carbonate 2006	Correlation Coefficient	.643	.536	.643	.643
		Sig. (1-tailed)	.06(ns)	.108(ns)	.06(ns)	.06(ns)
		N	7	7	7	7
	Non-carbonate 2004	Correlation Coefficient	.964	.893	.964	.964
		Sig. (1-tailed)	.000(***)	.003(***)	.000(***)	.000(***)
		N	7	7	7	7

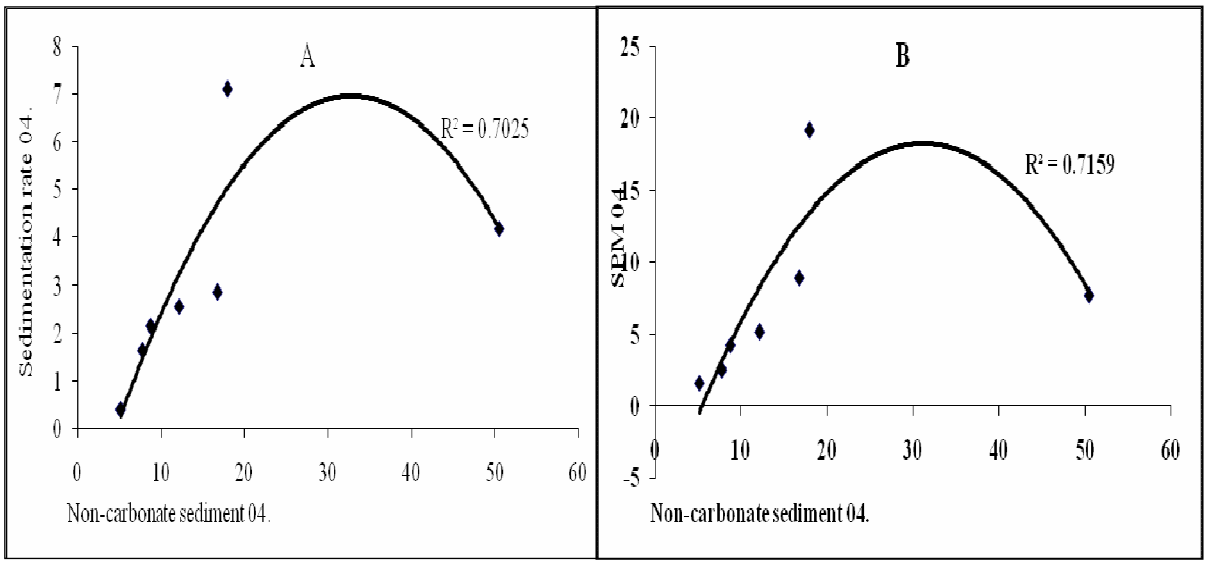


Figure 3.9: The correlation between 2004 non-carbonate sediment percentage and both of sedimentation rate (A) and SPM (B), for the seven study sites at Hurghada, Egypt.

Table 3.9: The mean of non-carbonate, sorting, mean grain size, sorting and size class percentage, and their skewness and kurtosis measured for bottom sediment collected from the seven sites at 2006, Hurghada, Egypt.

Parameters	Mean sediment analysis results for each site.						
	NIOF	A.Sadaf	Shedwan	Arabia	A.Minkar	Holidays	S.Hashish
Non-carbonate	8.8%	12.2%	18%	16.8%	7.8%	50%	5.2%
So	1.04	1.59	1.42	1.4	1.03	1.86	1.27
M_z	0.39	0.37	0.22	0.59	1.01	3.35	0.73
Gravel %(>2m)	9.3	25.5	22.7	14	1.68	0.48	8.88
Sand %(2-1/16m)	90.3	73.4	75.1	82.5	96.02	76.27	89.58
Mud % (<1/16m)	0.4	1.1	2.2	3.5	2.3	23.25	1.54
Skewness	1.14	1.01	2.75	1.24	.85	2.22	1.03
Kurtosis	2.3	0.9	1.8	1.12	1.55	3.25	1.96
	Sand-moderately sorting	Gravely sand bad sorting	Gravely sand bad sorting	Gravely sand bad sorting	Sand-moderately sorting	Silty sand very bad sorting	Sand-moderately sorting

Key: Mz = mean grain size, So =Sorting, Non-Carb =non-carbonate%, Sk= Skewness, Kr= Kurtosis.

Grain size analysis for bottom sediment samples indicated great variation between sites in the percentage of each components; gravel, sand and mud (Figure 3.10). Sand (2-1/16mm) was present at the highest percentage in all sites. Gravel (>2mm) varied from 22.7% at site 3 to 0.48% at site 6. Mud (<1/16mm) was at its highest (23.25%) at site 6 and the lowest reading (0.4%) was at site 1. Site 6, which has the highest mud percentage and the lowest gravel percentage, also has the highest non-carbonate terrigenous sediment percentage and the worst sorting (1.86) (Table 3.9).

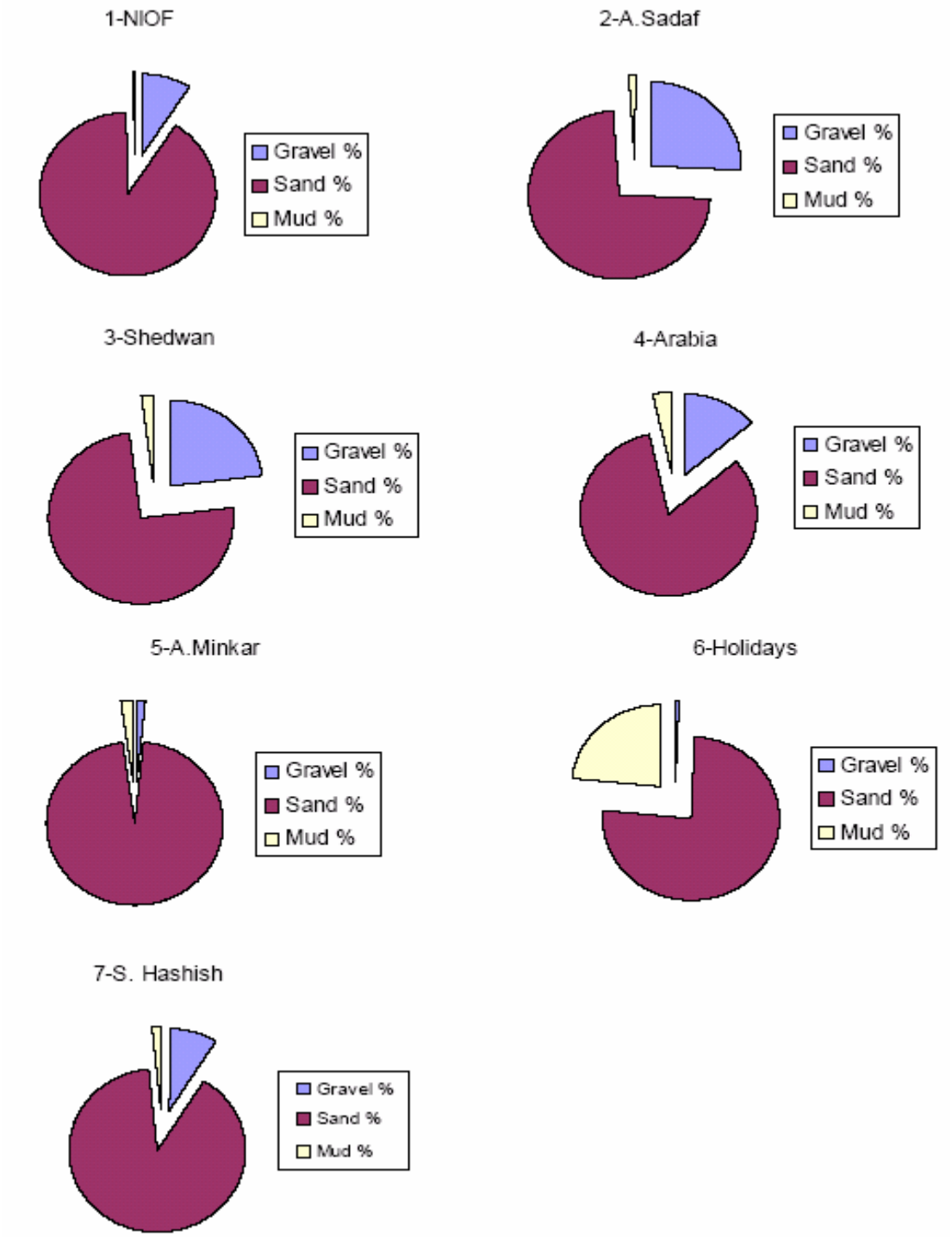


Figure 3.10: The mean percentage of gravels, sand and mud components of deposited sediment, collected from the seven study sites, Hurghada, Egypt.

Statistical analysis for the difference in non-carbonate sediment between sites and locations (beach or bottom) was carried using two way ANOVA (Table 3.10). The results indicate a significant difference between sites (F=21.62, df=6, P=0.000) and between locations (F=56.83, df=1, P=0.000). A Post-hoc Tukey test indicated that site 1 was significantly different from sites 5 and 6. Site 2 was significantly different from sites 3, 5 and 6. Site 3 was significantly different to sites 2, 5 and 6. Site 4 was significantly different from sites 5 and 6; Site 5 was significantly different from sites 1, 2, 3, 4 and 7. Site 6 was significantly different from sites 1, 2, 3, 4 and 7. Site 7 was significantly different from sites 5 and (6) (Table 3.12). A Spearman's Correlation test (Table 3.12) indicated a positive significant correlation (R= .821, P= .012) between beach and bottom non-carbonate sediment level at each site (Figure 3.11).

Table 3.10: Two way ANOVA test of non-carbonate sediment level between the two locations (beach and bottom) for each of the seven study sites, Hurghada, Egypt.

Source	S.S	d.f.	M. S.	F	P value
Site	23772.625	6	3962.104	21.621	.000
location	10414.959	1	10414.959	56.834	.000
Site * location	11552.273	6	1925.379	10.507	.000
Error	7696.572	42	183.252		
Total	99580.290	56			

Table 3.11: The significance matrix between sites in the non-carbonate sediment measured in beach or bottom at the 7 study sites, Hurghada, Egypt.

Sites	1	2	3	4	5	6	7
1							
2							
3		x					
4							
5	x	x	x	x			
6	x	x	x	x			
7					x	x	

Table 3.12: A Spearman's Correlation test between non-carbonate sediment level of beach and bottom samples, at each site, Hurghada, Egypt.

			Beach sediment
Spearman's rho	Bottom sediment	Correlation Coefficient	.821
		Sig. (1-tailed)	.012(*)
		N	7

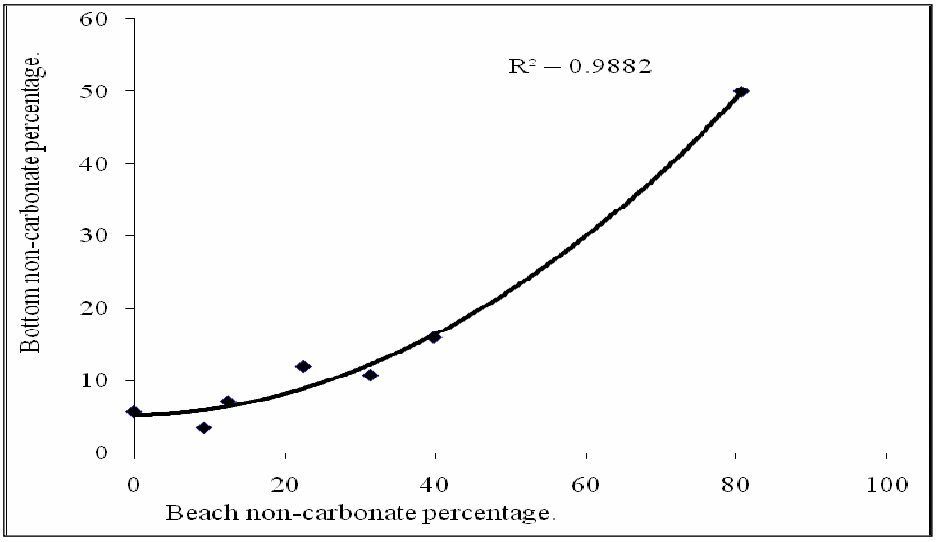


Figure 3.11: The correlation between beach and bottom non-carbonate sediment percentage, for the seven study sites at Hurghada, Egypt.

3.4 Nutrients Content in Seawater

NH_4, NO_3, NO_2, SIO_3 and PO_4 were measured of water samples from the seven study sites in January 2006. The mean concentration of all nutrients tested is shown in Figure 3.13. NH_4 concentrations ranged from 3.59µmol/l at site 3 to 0.96µmol/l at site 5 with most of the sites close to this value, the highest reading for NO_3 was 0.2µmol/l at site 6 and the lowest (0.06µmol/l) was at site 7. Site 3 had the highest value (0.52µmol/l) for NO_2 and site 7 was the lowest (0.4µmol/l). The highest level of SiO_3 (1.64µmol/l) was recorded at site 3 and the lowest reading of SiO_3 was (0.17µmol/l) recorded at site 5. PO_4 readings varied from 0.28µmol/l at site 3 to 0.04µmol/l at site 1 (Figure 3. 13).

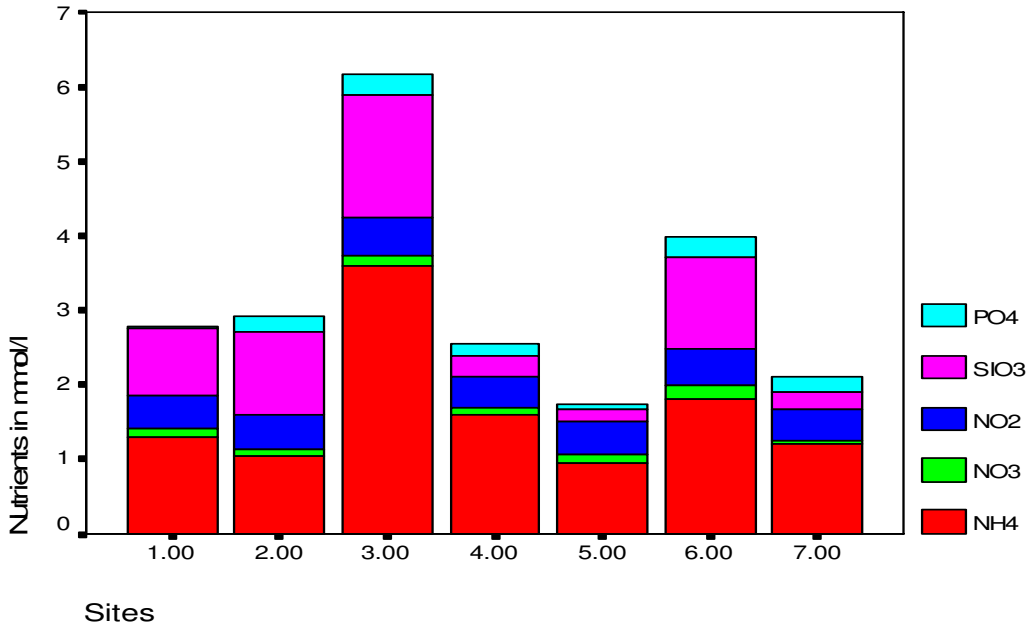


Figure 3.12: The mean of NH_4, NO_3, NO_2, SIO_3 and PO_4 for seawater samples of the seven study sites, based on sampling during January 2006, Hurghada, Egypt.

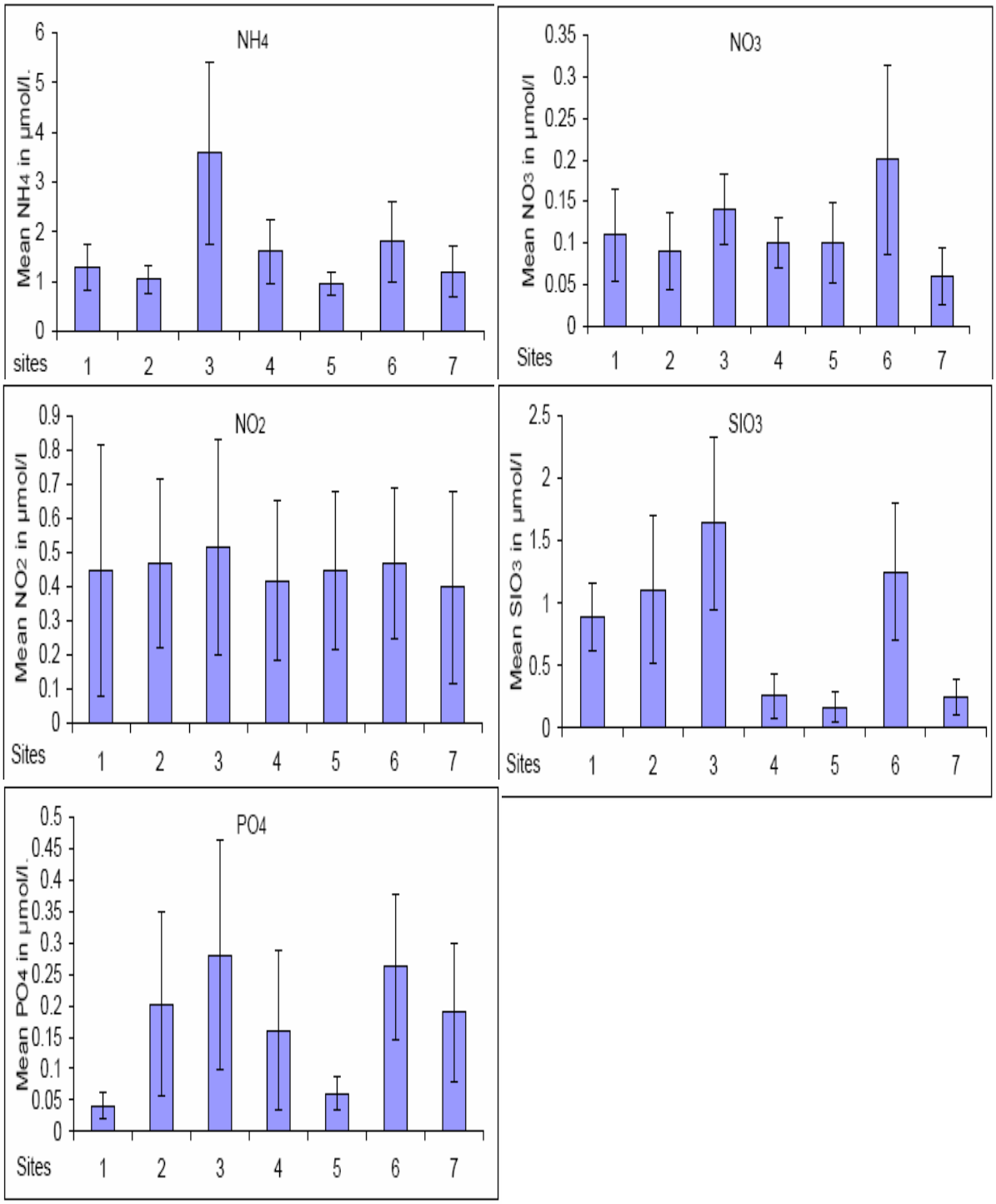


Figure 3.13: The differences between sites in the mean of (1) NH_4, (2) NO_3, (3) NO_2, (4) SIO_3 and (5) PO_4 in µmol/l ± SD standard deviation of mean for seawater samples from the seven study sites, based on January 2006 sampling, Hurghada, Egypt.

A one-Sample Kolmogorov-Smirnov Test indicated no significant differences from normal distribution. A Pearson Correlation test was carried between the five nutrients; NH_4, NO_3, NO_2, SIO_3 and PO_4; and the environmental parameters; sedimentation rate, SPM, non-carbonate sediment, turbidity and mud (Table 3.13). The results showed a significant positive correlation between NH_4; and both sedimentation rate (R=0.922, P=0.002), SPM (R=0.962, P=0.000) and turbidity (R=0.675, P=0.048) (Figure 3.14). NO_3

showed significant positive correlation with non-carbonate (R=0.922, P=0.002), turbidity (R=0.825, P=0.011) and mud (R=0.849, P=0.008) (Figure 3.15). NO_2 showed a significant positive correlation with sedimentation rate (R=0.874, P=0.005) and SPM (R=0.755, P=0.025) (Figure 3.16). SIO_3 showed a significant positive correlation with sedimentation rate (R=0.821, P=0.012), SPM (R=0.703, P=0.039) and turbidity (R=0.739, P=0.029) (Figure 3.17). PO_4 showed a significant positive correlation with turbidity (R=0.762, P=0.023) (Figure 3.18).

Table 3.13: The correlations between nutrients concentration in $\mu mol.l^{-1}$ in seawater of the study sites and sedimentation rate, SPM, non-carbonate sediment, turbidity and mud.

		Sediment	SPM	Non-carbonate	Turbidity	Mud
NH_4	Pearson Correlation	.922	.962	.289	.675	.108
	Sig. (1-tailed)	.002(***)	.000(***)	.265(ns)	.048(*)	.409(ns)
	N	7	7	7	7	7
NO_3	Pearson Correlation	.664	.477	.922	.825	.849
	Sig. (1-tailed)	.052(ns)	.140(ns)	.002(***)	.011(*)	.008(**)
	N	7	7	7	7	7
NO_2	Pearson Correlation	.874	.755	.345	.543	.165
	Sig. (1-tailed)	.005(**)	.025(*)	.224(ns)	.104(ns)	.362(ns)
	N	7	7	7	7	7
SIO_3	Pearson Correlation	.821	.703	.486	.739	.306
	Sig. (1-tailed)	.012(*)	.039(*)	.134(ns)	.029(*)	.252(ns)
	N	7	7	7	7	7
PO_4	Pearson Correlation	.637	.633	.566	.762	.458
	Sig. (1-tailed)	.062(ns)	.064(ns)	.093(ns)	.023(*)	.151(ns)
	N	7	7	7	7	7

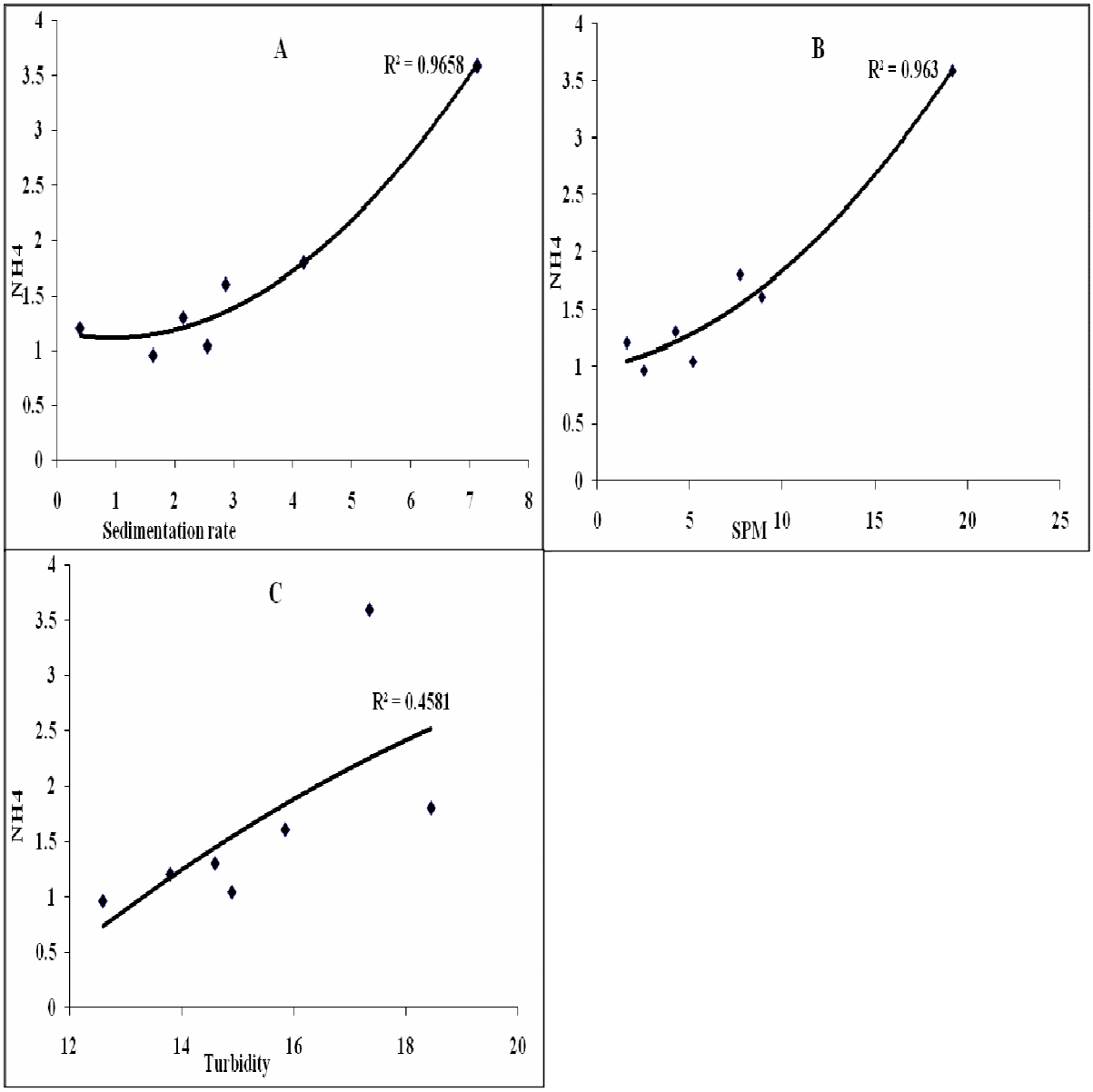


Figure 3.14: The correlation of NH_4 with sedimentation rate (A), SPM (B) and turbidity, for the seven study sites, Hurghada, Egypt.

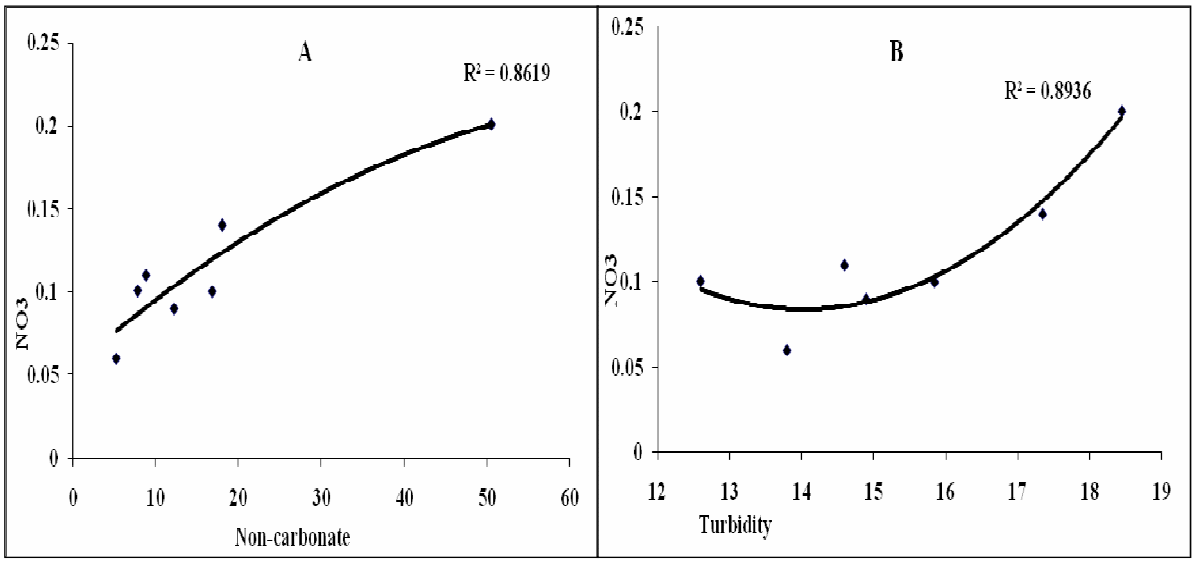


Figure 3.15: The correlation of NO_3 with non-carbonate (A) and turbidity (B) for the seven study sites, Hurghada, Egypt.

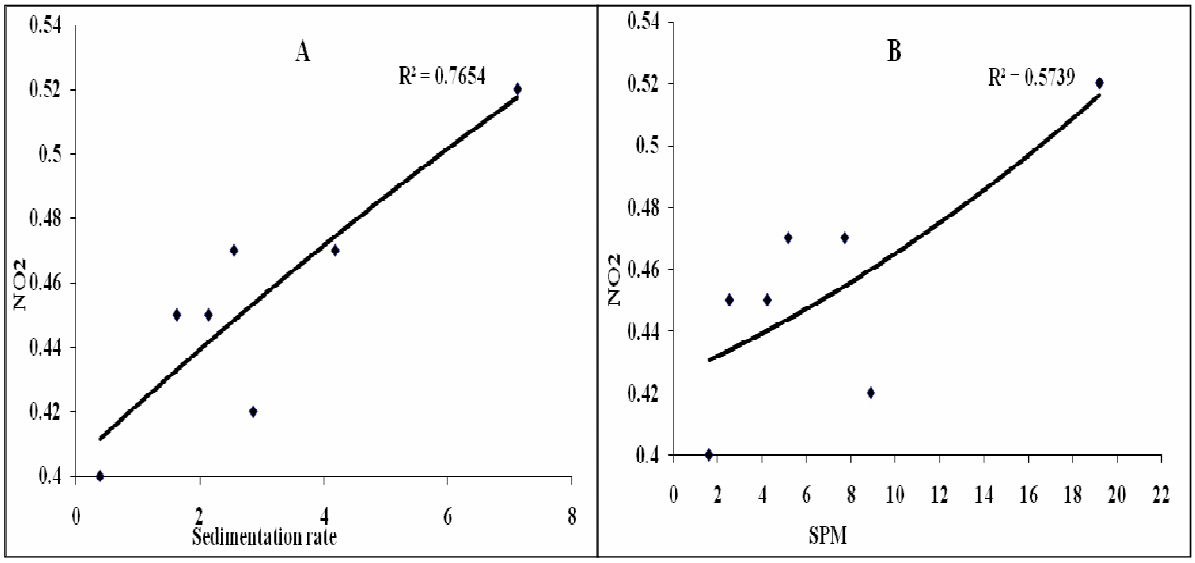
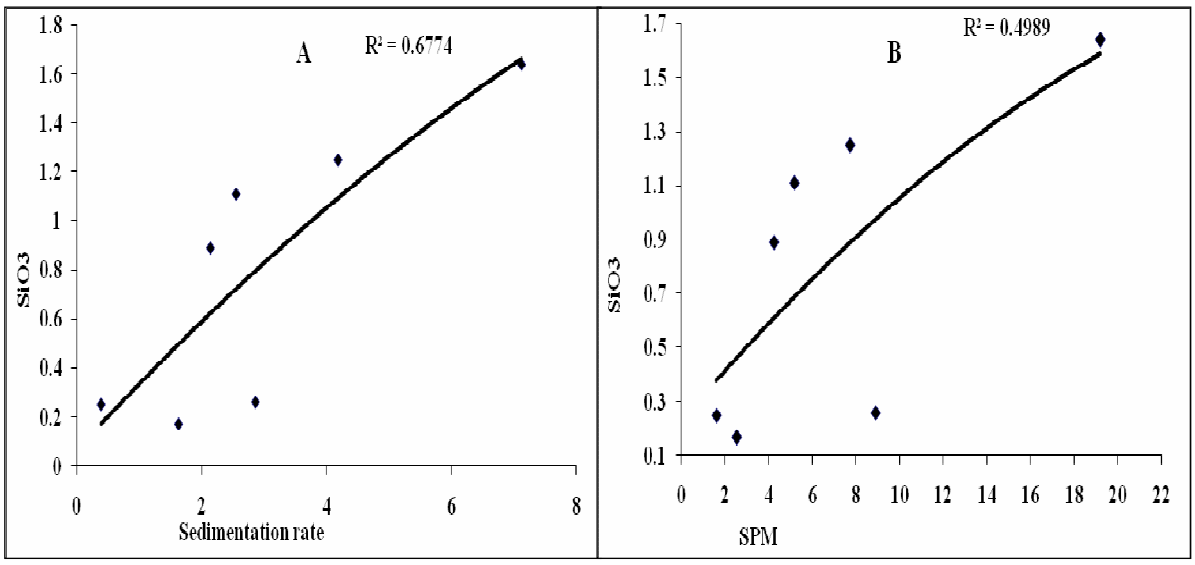


Figure 3.16: The correlation of NO_2 with sedimentation rate (A) and SPM (B) for the seven study sites, Hurghada, Egypt.



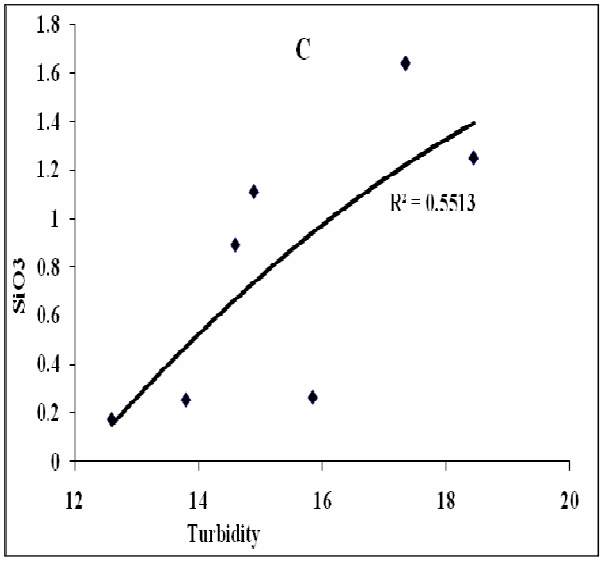


Figure 3.17: The correlation of SIO_3 with sedimentation rate (A), SPM (B) and turbidity (C) for the seven study sites, Hurghada, Egypt.

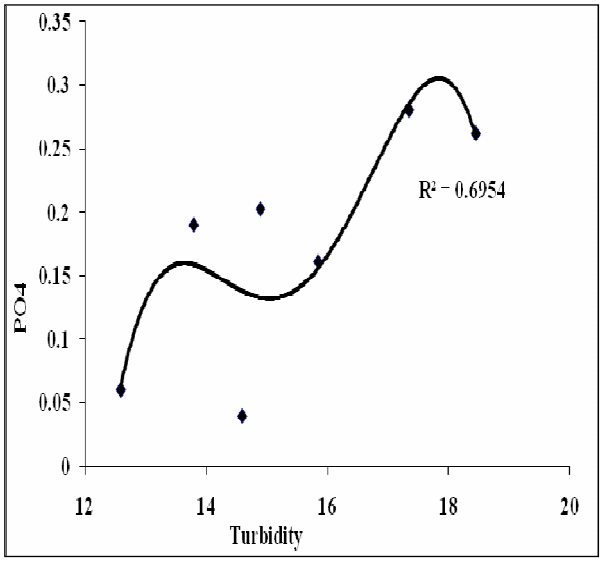


Figure 3.18: The correlation of PO_4 with turbidity for the seven study sites, Hurghada, Egypt.

3.5 Water Quality Parameters

Water quality readings did not indicates large differences between sites (Table 3.15). Specific conductivity (SPC) has the highest readings at site 2, 3 and 7 and the lowest were sites 4 and 5. Dissolved oxygen (DO) was at its highest at site 3 and the lowest was at site 6. PH varied from 9.06 at site 7 to 8.91 at site 5. Total dissolved solids (TDS) varied from 39.45 at site 7 to 38.9 at site 4. Percentage of dissolved oxygen (DO%) had the highest reading (95.3) at site 3 and the lowest reading (84.65) was at site 2. The highest turbidity (18.45) was found at site 6 and the lowest (12.6) was at site 5. Salinity ranged from 40.8 at site 7 to 41.55 at site 3 (Table 3.14).

Pearson correlation test between all of SPC, DO, PH, TDS, DO%, turbidity and salinity; and sedimentation rate, SPM, non-carbonate sediment, and mud showed various degree of correlations (Table 3.15). SPC showed a significant positive correlation with SPM (R= .681, P=0.046) (Figure 3.15). Turbidity showed significant positive correlation with

sedimentation rate (R= .781, P=0.019), SPM (R= .707, P=0.038), mud (R= .689, P=0.043) and percentage of non-carbonate sediment (R=0.839, P=.009) as shown in Figure 3.16.

Table 3.14: The means water quality parameters for the seven sampling sites, Hurghada, Egypt.

Sites	Tem.	Depth	SPC	DO	PH	TDS	DO%	Turbidity	S‰
NIOF	27	2m	61.65	5.75	8.98	39	86.8	14.60	41.4
A.Sadaf	27.1	2	61.7	5.55	9	39	84.65	14.90	41.5
Shedwan	26.6	2	61.7	5.9	9.02	39	95.3	17.35	41.55
Arabia	27.2	2	61.05	5.4	9.01	38.9	85.8	15.85	41.1
A.Minkar	27.15	2	61.05	5.55	8.91	38.95	85.85	12.60	41.3
Holidays	27.2	2	61.15	5.35	9.01	39.05	88.1	18.45	41.2
S. Hashish	27.5	2	60.7	5.5	9.06	39.45	88.75	13.80	40.8

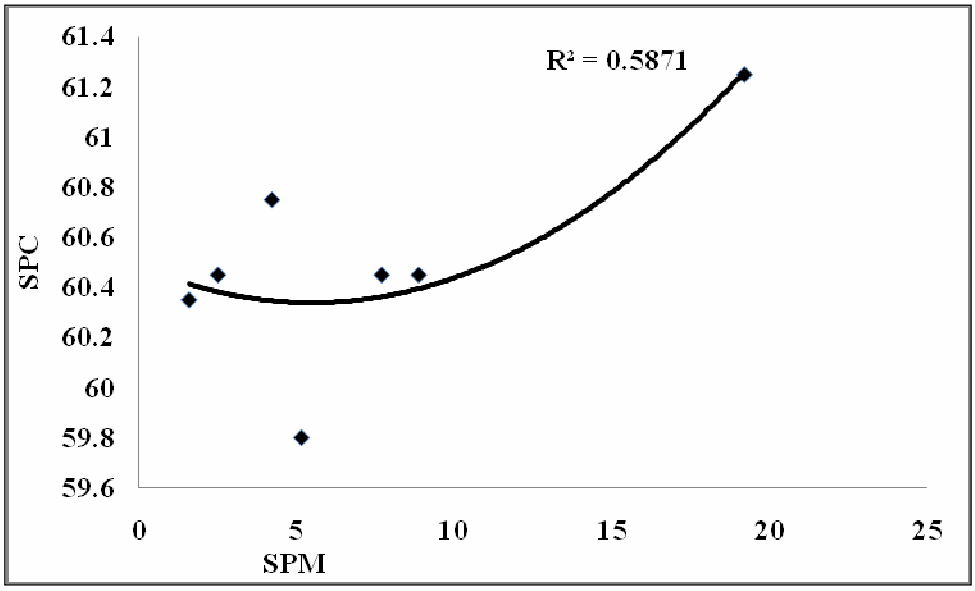


Figure 3.19: The correlations between SPC and SPM for the seven study sites, Hurghada, Egypt.

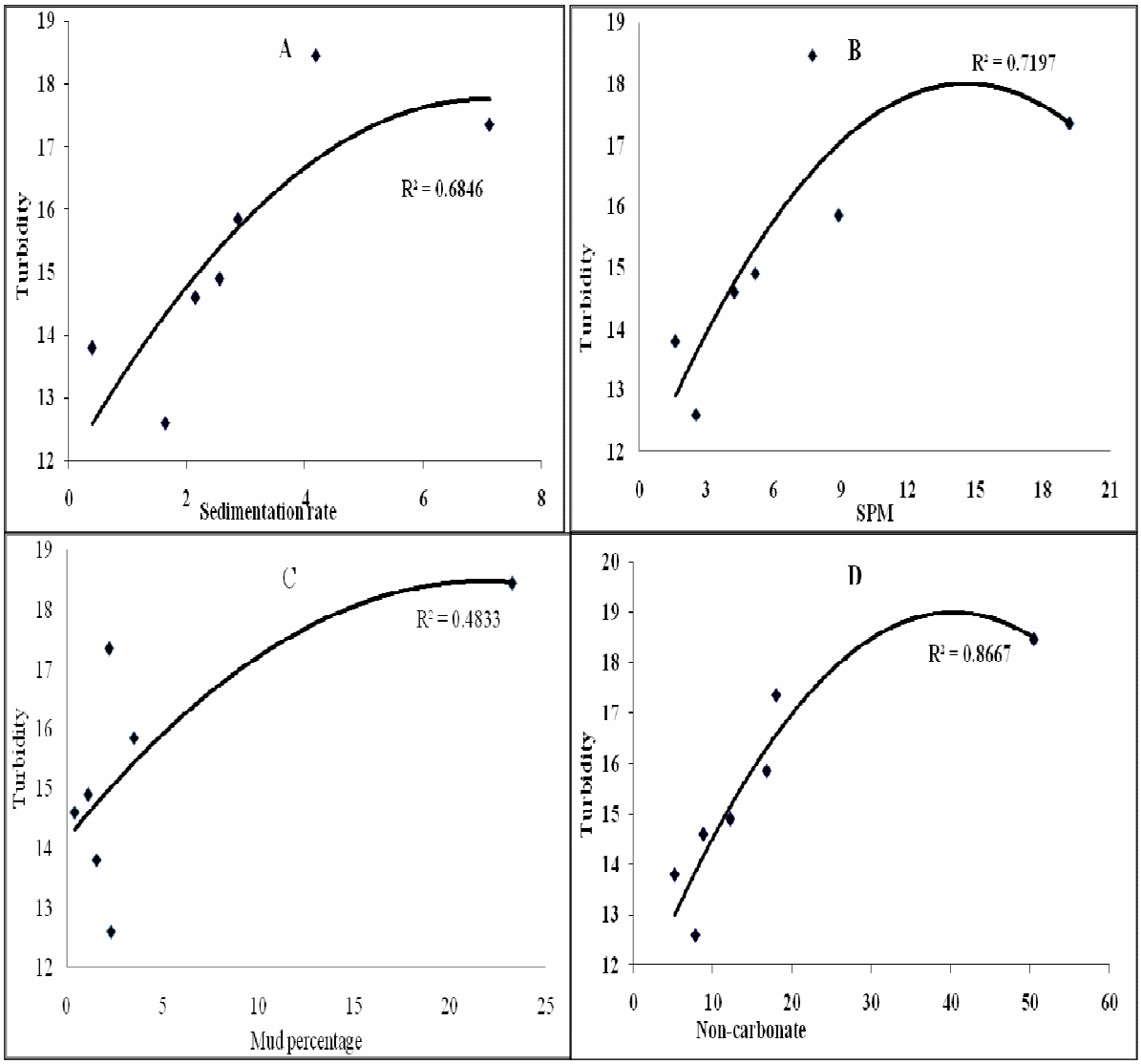


Figure 3.20: The correlations between turbidity and: sedimentation rate (A), SPM (B), mud percentage (C) and non-carbonate sediment (D), for the seven study sites, Hurghada, Egypt.

Table 3.15: The correlations between all of SPC, DO, PH, TDS, DO%, turbidity and salinity; and sedimentation rate, SPM, non-carbonate sediment and mud readings of the seven study sites, Hurghada, Egypt.

		Sediment	SPM	Mud	Non-carbonate
SPC	Pearson Correlation	.636	.681	-.033	.065
	Sig. (1-tailed)	.062(ns)	.046(*)	.472(ns)	.445(ns)
	N	7	7	7	7
DO	Pearson Correlation	.367	.343	-.264	-.236
	Sig. (1-tailed)	.209(ns)	.225(ns)	.284(ns)	.305(ns)
	N	7	7	7	7
PH	Pearson Correlation	.129	.246	.106	.149
	Sig. (1-tailed)	.391(ns)	.297(ns)	.411(ns)	.375(ns)
	N	7	7	7	7
TDS	Pearson Correlation	.490	.448	.314	.333
	Sig. (1-tailed)	.132(ns)	.157(ns)	.247(ns)	.233(ns)
	N	7	7	7	7
DO %	Pearson Correlation	.163	.119	-.556	-.499
	Sig. (1-tailed)	.363(ns)	.399(ns)	.097(ns)	.127(ns)
	N	7	7	7	7
Turbidity	Pearson Correlation	.781	.707	.689	.839
	Sig. (1-tailed)	.019(*)	.038(*)	.043(*)	.009(*)
	N	7	7	7	7
Salinity	Pearson Correlation	.445	.463	-.245	-.153
	Sig. (1-tailed)	.159(ns)	.147(ns)	.298(ns)	.372(ns)
	N	7	7	7	7

Discussion

Sedimentation rate results from this study indicated significant differences between sites and between the years 2004 and 2006, while SPM readings indicated significant differences between sites but not between years. The results showed that SPM, sedimentation and turbidity were all significantly and positively correlated with each other. This study has shown that mean highest sedimentation rates were 7.12-9.49mg.cm^{-2} day^{-1} and were recorded at site 3 in 2004 and 2006 respectively. Site 7 represented the lowest sediment deposition rate of all sites (0.4-0.95mg.cm^{-2}.day^{-1} in 2004 and 2006). Compared to a study of sedimentation during four seasons at 6 sites in Sharm El-Shiekh, (Amer, 2005) where sedimentation rates ranged from 0.067mg.cm^{-2}.day^{-1} to 4.8mg.cm^{-}

^{2}.day^{-1}, sedimentation rates in the current study were generally higher and varied from 0.4 to 9.49mg.cm^{-2}.day^{-1} along the Hurghada area. Rogers (1990) proposed a sedimentation threshold level of 10mg.cm^{-2}.day^{-1}, which was assumed to reduce cover and diversity of coral reefs. From the current study, results showed that this threshold level could be exceeded at certain sites during certain seasons, although most of the sites did not reach this level. However the sites which exceeded this proposed threshold level showed the lowest quality in relation to the biological indicators measured.

Suspended particulate matter SPM measurements demonstrated the same trend, with the highest levels (19.19-21.12mg.l^{-1} in 2004 and 2006) at site 3 and the lowest (1.6-2.07mg.l^{-1} in 2004 and 2006) at site 7. Sediment traps measure the total downward flux of suspended particles (Asper, 1996). Because short-term increases, which have a greater influence on trap data, are less deleterious to corals than chronic increases, SPM is considered a better expression of long-term sediment effects on coral reefs (Dodge & Vaisnys, 1977; Bak, 1978; Tomascik & Sander, 1985). SPM in seawater in the current study showed significant differences between sites but not between the two years, 2004 and 2006. A significant positive correlation between sedimentation rate and SPM was also found, indicating that sites with high SPM also had high sedimentation. However, this does not mean one of them causes the other. Other factors might cause both of them in the same time.

Bottom sediment characteristics were investigated in the current study to define the major sources of sediment in the study area. Bottom sediment analysis indicated a significant difference between sites and locations (beach and bottom) but not between the two years (2004 and 2006) for the percentage of non-carbonate sediment. Furthermore, there was a strong positive correlation between the percentage of beach and bottom non-carbonate sediment. A significant positive correlation between the percentage of non-carbonate sediment and both sedimentation rate and SPM was also found. This suggests the synchronized increase of sedimentation, SPM and non-carbonate sediment as a consequence of land filling processes.

Analysis of bottom sediment samples showed that sediment deposits varied considerably between reefs in the study area, ranging from a bare carbonate pavement with little sediment deposits of mainly biogenic origin at site 7, through coarse medium sediment and carbonate sands on offshore reefs at site 1 and 2, to mud on the inshore reef edge at site 6. Sediment sample analysis showed that some sites had a high percentage of non-carbonate sediments reaching about 50.5 % as in site 6 while the rest of the sites were mainly biogenic sediment. It was noticed that although some sites such as site 3 and 4 have undergone extensive filling, they do not bear high levels of non-carbonate sediment in their deposits if compared with site 6. These variations could be related to the materials used to fill the reef flats at these sites, which were biogenic limestone imported from near shore cliffs and mountains. The sediment profile at site 6 was silt sand characterized by the most poorly sorted (Sorting coefficient= 1.86) between all sites, and the largest mean grain size (M_Z= 3.35). The lowest percentage of non-carbonate sediment (3.6% and 5.2%) was found at sites 2 and 7 respectively. These sites were also considered to be far from land discharge and rarely receive land-based rainfall during the wet seasons. The best sorting was recorded at site 5 and 1 (sorting coefficient were = 1.03 and 1.04 and the smallest mean grain size (M_Z= 0.22) was at site 3. Turbidity levels varied from mean highest level 18.45 at site 6 to mean lowest 12.6 at site 5. In conclusion sedimentation rate, SPM levels, non-carbonate sediment and turbidity levels all act together to support the strong relationship between poor environmental quality and the proximity to coastal developments. Most, if not all of the above parameters, were at their highest levels at the sites which had undergone intensive dredging and filling disturbance in the back reef flat

and near to tourist developments in the area.

Physico-chemical parameters including temperature, salinity, pH, SPC (specific conductivity), DO (dissolved oxygen), TDS (total dissolved solids), DO% (percentage of dissolved oxygen) and turbidity were measure at the seven study sites as environmental parameters that could interact with reef health in one way or another. Major nutrients, inorganic phosphate, silicate, ammonia, nitrite and nitrate were also measured in seawater from the study sites. From this study NH_4^+, NO_3^-, NO_2^- and PO^{3-}_4 levels were generally lower than those reported from the Egyptian Environmental Policy Program (EEPP) water quality report for the Red Sea region (Awad and Shabara, 2003), Global Environmental Facility monitoring project carried out in the Red Sea region (GEF, 1997) and the nutrient enrichment study in coral reef of the Gulf of Aqaba, Red Sea (Rasheed et al, 2002).

GEF results for NO^-_2 ranged from 0.083µmol.l^{-1} to 0.79µmol.l^{-1}; from 1.53µmol.l^{-1} to 0.02µmol.l^{-1} in the EEPP study and from 0.52µmol.l^{-1} to 0.4µmol.l^{-1} in the current study. From the GEF results NO^-_3 ranged from 12.76µmol.l^{-1} to 59.18µmol.l^{-1}; and from EEPP it ranged from 1.13µmol.l^{-1} to 0.02µmol.l^{-1}. From this study NO^-_3 ranged from 0.14µmol.l^{-1} to 0.06µmol.l^{-1}. SiO_3 reading was around the mean of previous study (0.61µmol.l^{-1}) (Table 7.1). The variations between these studies could be attributed to the seasonal variation in the Red Sea water masses mixing in both coastal and offshore areas in response to the local climatic province. These variations affect the vertical transport as well as dispersion patterns of hydrographic parameters such as salinity, nutrient salts and dissolved oxygen. In addition to this, biological activities also influence these components especially in the coastal productive areas (Awad and Shabara, 2003). Major nutrients in seawater (NH^+_4, NO^-_3, NO^-_2, SiO_3 and PO^{3-}_4) showed various degrees of positive correlations with sedimentation rate, SPM, non-carbonate sediment, turbidity and percentage mud. Significant positive correlations were indicated between: NH_4, NO^-_2 and $SiO_3$with sedimentation rate and SPM. NO^-_3 had significant positive correlation with non-carbonate sediment, turbidity and mud, and PO^{3-}_4 had significant positive correlation with turbidity.

Water quality parameters; SPC, DO, PH, TDS, DO%, S‰ and turbidity did not indicate a large differences between sites. Turbidity showed significant positive correlations with sedimentation rate, SPM, mud and non-carbonate sediment. This study has shown that there was no significant correlation between any of the nutrients measured and coral health parameters. There were no significant differences in the mean values of all parameters between this study and the two studies. SPC, DO, PH, TDS, DO%, S‰ and turbidity were generally higher in the current study than the mean reading for the site of Hurghada port reported in the EEPP water quality report (Awad and Shabara, 2003). However, they are still around the mean values reported in the same report for the entire Red Sea survey.

In conclusion the environmental parameters measured in this study indicated various degrees of discrepancy from what is describe as normal levels. It also showed great variations between sites with clear trend for all the parameters. With exception of nutrients all these parameters showed a coincide variations between sites. The biological parameters were then expected to experience a wide range of impacts varied between seasons and between each environmental parameter.

CHAPTER 4

CORAL SURVEY RESULTS

Introduction

Many studies refer to sedimentation as one of the main impacts significantly reducing coral cover and diversity (Cortes & Risk, 1984; Hubbard, 1986; Hodgson & Dixon, 1988; Rogers, 1990). Sedimentation is also considered one the parameters, which affect coral recruitment (Birkeland, 1977, Bak & Engel, 1979, Birkeland et al. 1981, Rogers et al. 1984). Coral genera show various threshold level of sedimentation that can be endured (Sanders and Baron-Szabo, 2005). Sedimentation in the Hurghada area has shown an unambiguous disturbance to the reefs along the coastal area. However, the precise level of reef decline along the coastal zone has not been determined and none of the earlier studies provided enough information about the extent of deterioration or its probable source. This part of study tries to define the reef health status in the area and the level of sedimentation impact on the coral reefs using various coral parameters. Many parameters which have previously been widely used as reef health indicators, and have proved efficient to determine reef health status, were used in this study. These indicators include coral percentage cover, number of dead and new recruits, abundance, species richness, diversity index, deterioration index, disturbance, transplanted coral survival and recruitment, and settlement. New parameters were introduced such as Deterioration Index (DI) and the percentage of *r-strategist* corals. All these parameters were measured along line intercept transects (LIT) and line built transects (LBT). The study aims to investigate the relationship between the observed sedimentation and turbidity levels and coral health status.

4.1 Coral Percentage Cover

Survey results for hard coral cover indicated that there was a large variation in both branched and massive coral cover between sites (Figure 4.1). Total percentage cover in 2004 varied from a mean highest cover of 57.87% at site 1 to the lowest 8.17% at site 3 and from 64.8% at site 1 to 6.7% at site 3 in 2006. The highest massive coral percentage was recorded at site 1 and the highest branched coral cover was at site 5 (Figure 4.2). Whilst massive coral cover was at the lowest level at site 5 and the lowest branched coral cover was found at site 6 for both of the two years. Two-way ANOVA results (Table 4:1) for total coral cover (branched and massive) against year and site showed no significant difference between years ($F=3.76$, $df=1$, $P=0.059$). There was, however, a significant difference between sites ($F=41.79$, $df=6$, $P=0.000$) in coral cover. Post Hoc test showed that site 1 was significantly different from all the other sites; site 2 was significantly different from sites 1, 3, 6 and 7; site 3 was significantly different from sites 1, 2, 5 and site 7; site 4 was significantly different from sites 1 and 7; site 5 was significantly different from sites 1, 3 and 6; site 6 was significantly different from sites 1, 2, 5 and 7 and site 7 was significantly different from sites 1, 2, 3, 4 and 6 (Table 4.2).

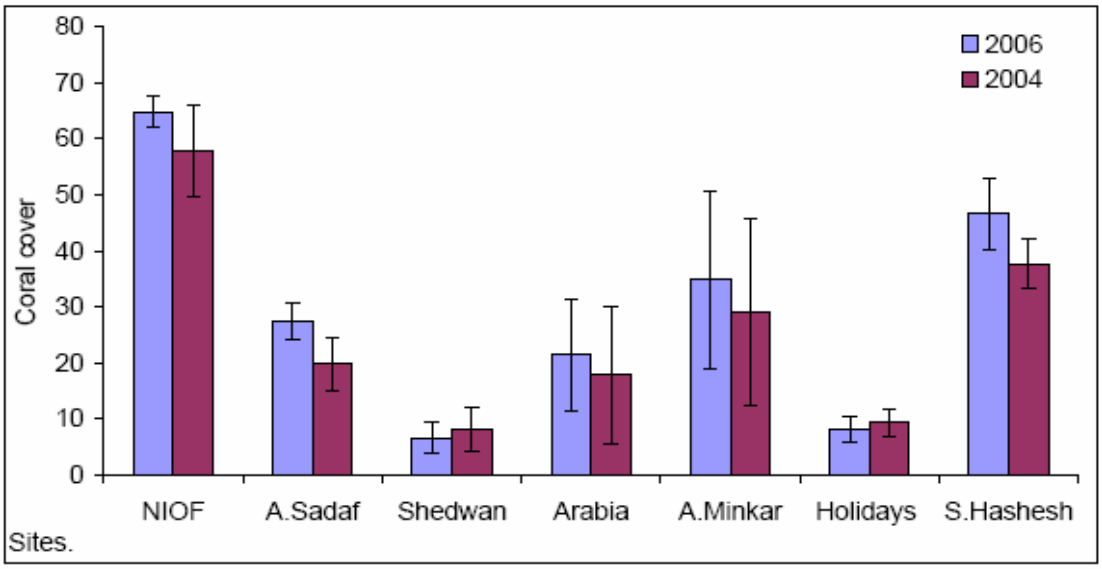


Figure 4.1: Mean percentage coral cover (n=4) ± SD standard deviation of the mean as indicated by line intersects transect carried out in 2004 and 2006 in the reef slope of the study sites, Hurghada, Egypt.

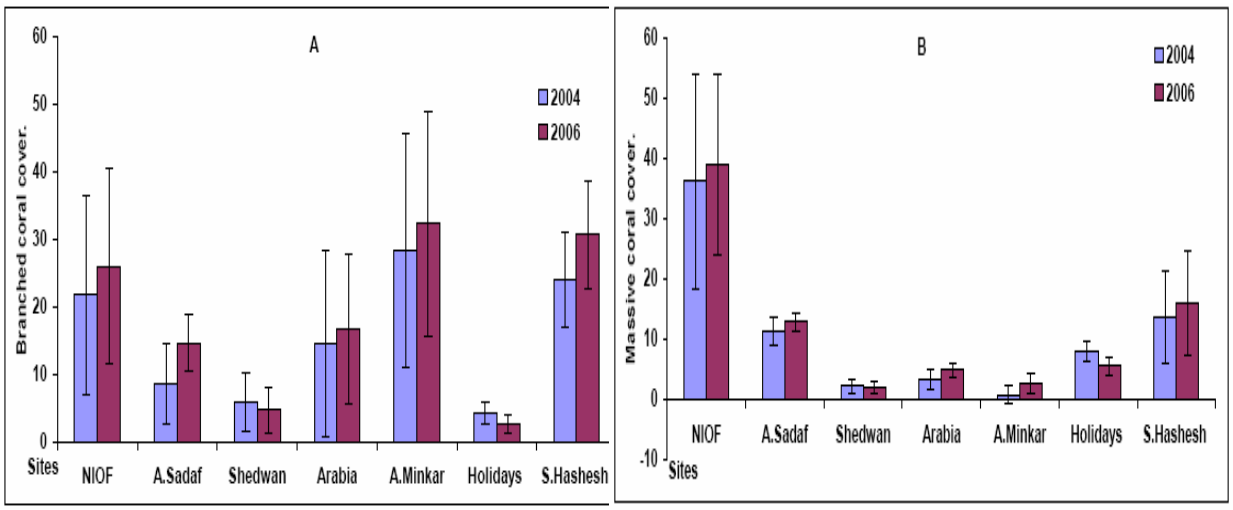


Figure 4.2: Mean percentage coral cover (n=4) ± SD standard deviation of the mean for branched forms (A), and massive forms (B) for the two study years 2004 and 2006 as indicated by line intersect transect carried out in 2004 and 2006 in the reef slope of the study sites, Hurghada, Egypt.

Table 4.1: Two-way ANOVA statistical analysis for coral cover variations between the two years, and between sites, data collected from the seven study sites in 2004 and 2006, Hurghada, Egypt.

Source of variation	S.S.	d.f.	M.S.	F	p-value
Years	265.350	1	265.350	3.761	.059
Sites	17691.547	6	2948.591	41.796	.000
Years * Sites	211.436	6	35.239	.500	.805
Error	2962.962	42	70.547		
Total	64672.050	56			

Table 4.2: The significance matrix between sites in the coral cover, measured in 2004 and 2006, as indicated by Post Hoc Tukey test, for the 7 study sites, Hurghada, Egypt.

Sites	1	2	3	4	5	6	7
1							
2							
3	x	x					
4	x						
5	x		x				
6	x	x			x		
7	x	x	x	x		x	

Univariate analysis of variance was also carried for coral cover of both branched and massive corals to check whether each variable differed significantly between sites and years. Three-way ANOVA (Table 4:3) for coral cover against year, site and growth forms (branched or massive) showed no significant difference between years (F=1.67, df=1, P=0.20). There was, however, a significant difference between sites (F=18.56, df=6, P=0.00) and between growth forms (F=10.81, df=1, P=0.001) in coral cover (Table 4.3). A Post Hoc test showed that site 1 was significantly different from all the other sites; site 2 was significantly different from site 1; site 3 was significantly different from sites 1, 5 and 7; site 4 was significantly different from sites 1 and 7; site 5 was significantly different from sites 1, 3 and 6; site 6 was significantly different from sites 1, 5 and 7; and site 7 was significantly different from sites 1, 3, 4 and 6 (Table 4.4).

Table 4.3: Three-way ANOVA statistical analysis for mean coral cover variation between year, site and growth form, data collected form the seven study sites at 2004 and 2006, Hurghada, Egypt.

Source of variation	S.S.	d.f.	M.S.	F	p-value
Years	132.675	1	132.675	1.671	.200
Sites	8845.773	6	1474.296	18.565	.000
Growth forms	858.589	1	858.589	10.811	.001
Years * Sites	105.718	6	17.620	.222	.969
Years * Growth forms	22.411	1	22.411	.282	.597
Sites * Growth forms	4399.811	6	733.302	9.234	.000
Years * Sites * Growth	23.732	6	3.955	.050	.999
Error	6670.823	84	79.415		
Total	42829.910	112			

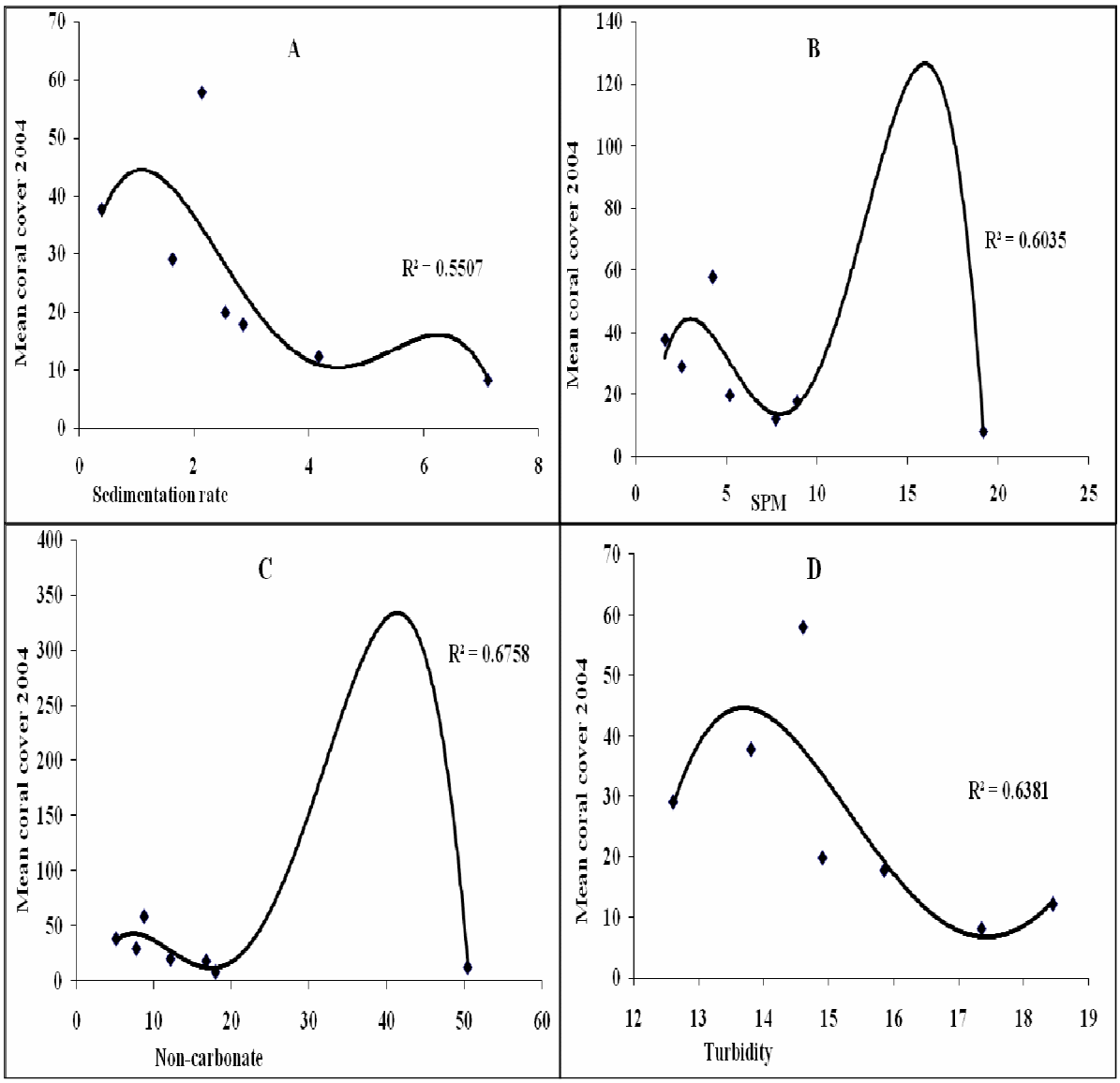
Table 4.4: The significance matrix between sites in coral cover, measured in 2004 and 2006, as indicated by Post Hoc Tukey test, for the 7 study sites, Hurghada, Egypt.

Sites	1	2	3	4	5	6	7
1							
2	x						
3	x						
4	x						
5	x		x				
6	x				x		
7	x		x	x		x	

Spearman's rho correlation was carried out to test the correlation of the 2004 mean coral cover with sedimentation, SPM, non- carbonate sand, turbidity, gravel sand and mud (Table 4.5). Results showed a significant negative correlation with sedimentation (R=-0.89, P=0.003), SPM (R=-0.85, P=0.007), non-carbonate sediment (R=-0.85, P=0.007), turbidity (R=-0.82, P=0.012) and sand (R=-0.67, P=0.047), which means that sites with high levels of sedimentation, SPM, non-carbonate sediment and turbidity had lower coral cover. Correlations with, gravel and mud did not indicate any significant correlation between coral cover and any of these parameters (Table 4.5). Significant correlations with sedimentation rate, SPM, Non-carbonate, turbidity and sand percentage were plotted in figure (4.3) with R^2 value.

Table 4.5: Spearman's correlation between 2004 mean readings of coral cover with sedimentation, SPM, non- carbonate sand, turbidity, gravel sand and mud for the seven study sites, Hurghada, Egypt.

		Sediment-ation	SPM	Non carbonate	Turbidity	Gravel	Sand	Mud
Coral cover 2004	Correlation Coefficient	-.893	-.857	-.857	-.821	-.179	.679	-.643
	Sig.(1-tailed)	.003(***)	.007 (**)	.007(**)	.012(*)	.351(ns)	.047(*)	.06 (ns)
	N	7	7	7	7	7	7	7



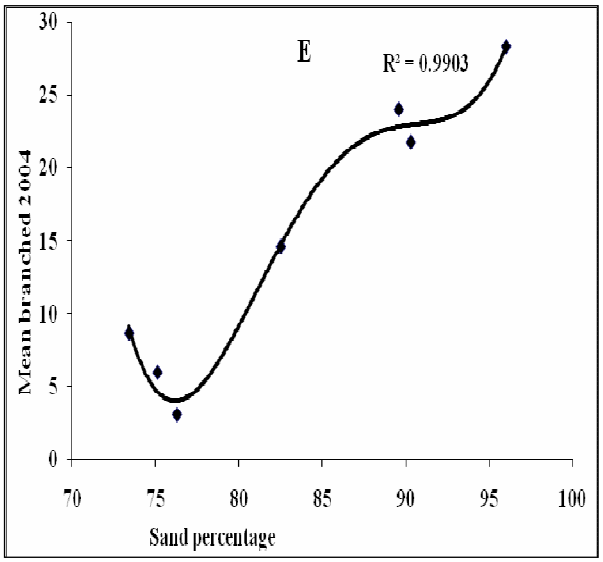


Figure 4.3: The correlation level between 2004 mean coral cover and sedimentation rate in mg.cm-^{2}.day^{-1} (A), SPM in mg.l^{-1} (B); percentage of non-carbonate sediment (C), turbidity in NTU units (D) and sand percentage (E) in all study sites, Hurghada, Egypt.

Spearman's rho Correlations were also carried out to test the correlation of 2006 coral cover with sedimentation, SPM, non- carbonate sand, turbidity, gravel sand and mud (Table 4:6). Results showed a significant negative correlation with sedimentation rate (R=-0.893, P=0.003), SPM (R=-0.893, P=0.003), turbidity (R=-0.821, P=0.012) and sand (R=-0.679, P=0.047). Correlation with sediment, SPM, turbidity and sand percentage were plotted in figure (4.4) with R^2 value.

Table 4.6: The correlation between 2006 mean readings of coral cover with sedimentation rate, SPM, non- carbonate sand, turbidity, gravel sand and mud for the seven study sites, Hurghada, Egypt.

		Sediment-ation	SPM	Non carbonate	Turbidity	Gravel	Sand	Mud
Coral cover 2006	Correlation Coefficient	-.893	-.893	-.464	-.821	-.179	.679	-.643
	Sig.(1-tailed)	0.003(***)	.003 (***)	.147(ns)	.012(*)	.351(ns)	.047(*)	.06 (ns)
	N	7	7	7	7	7	7	7

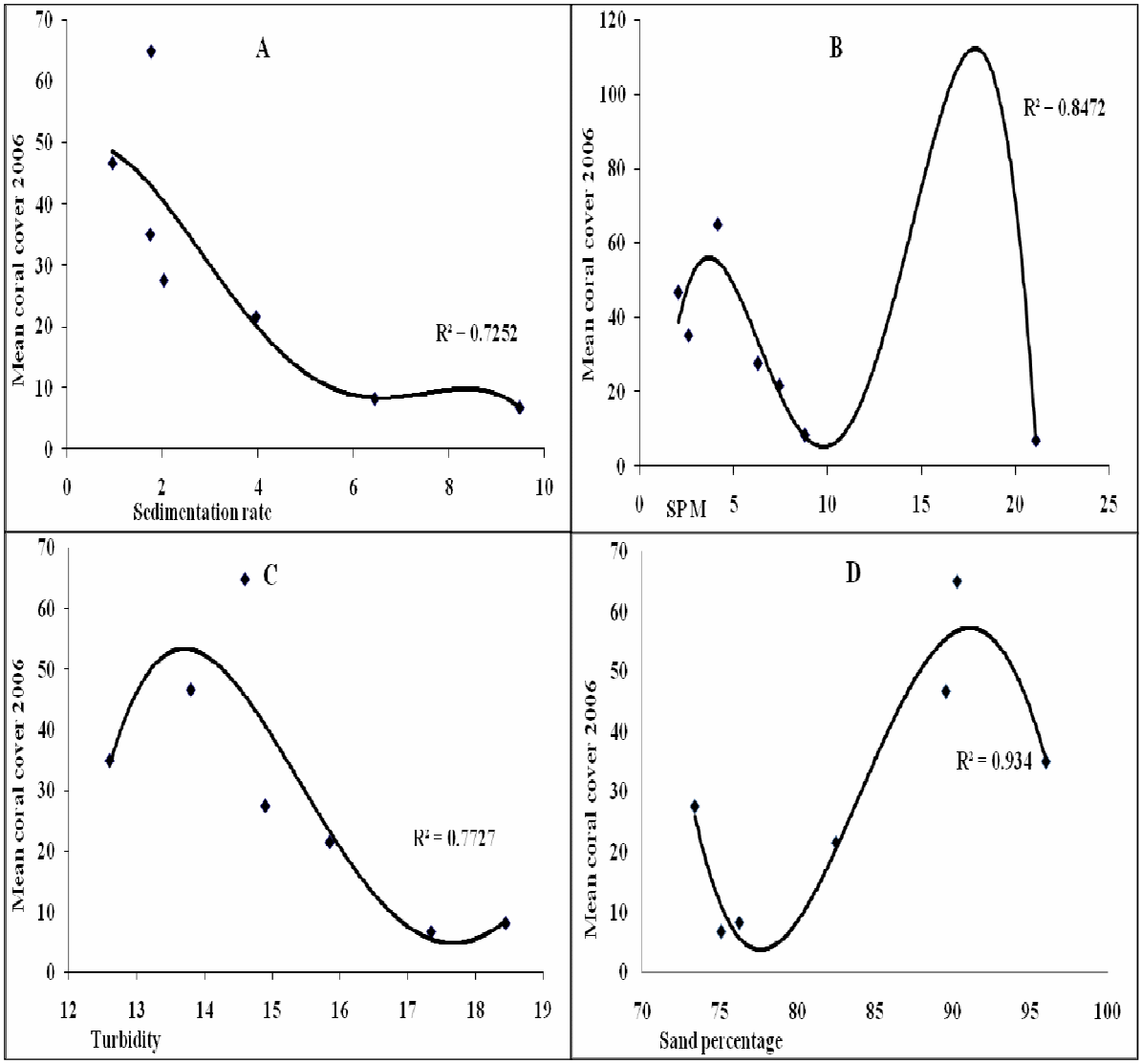
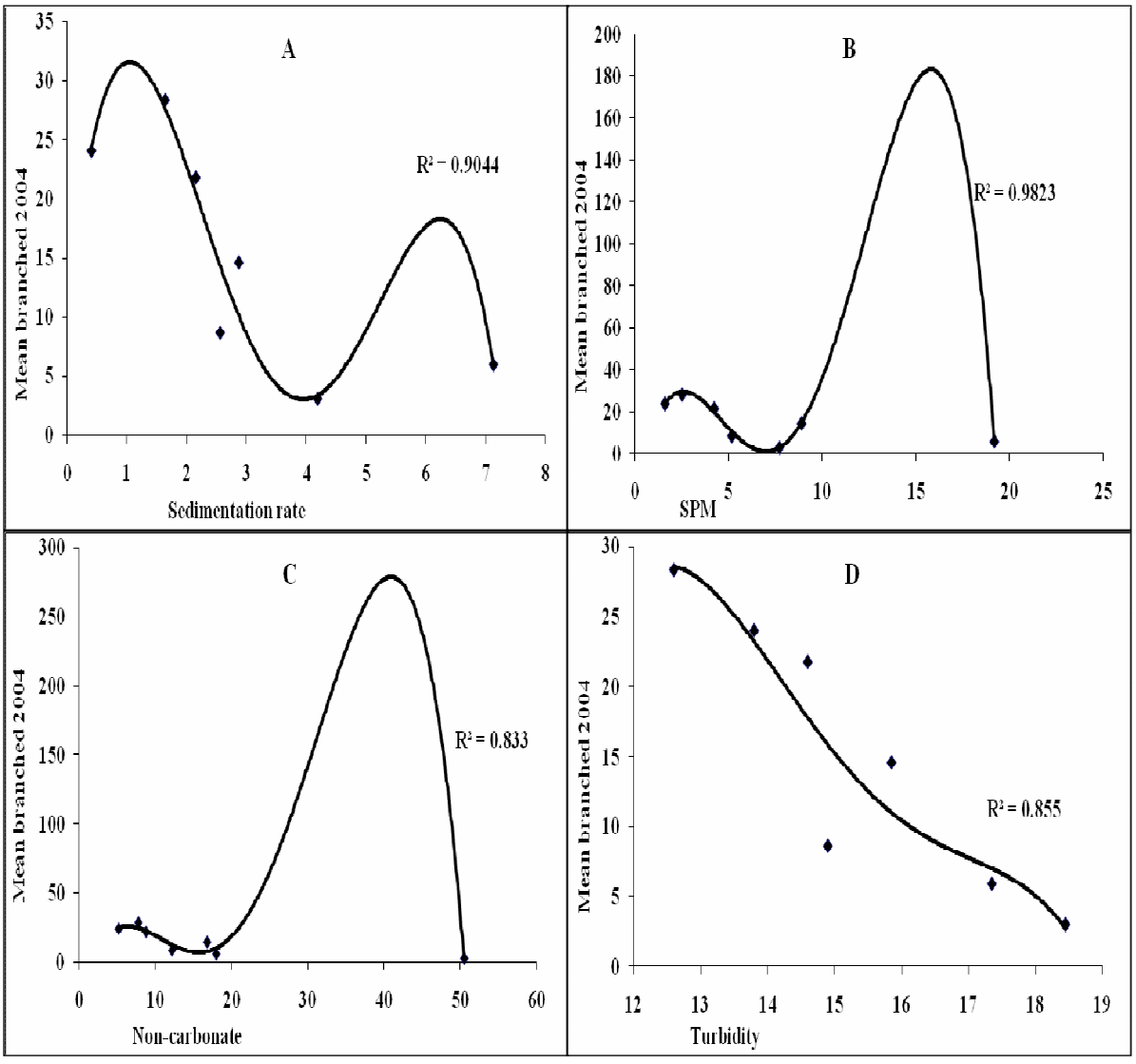


Figure 4.4: The correlation between 2006 coral cover and sedimentation rate in mg.cm^{-2}.day^{-1} (A), SPM in mg.l^{-1} (B), turbidity in NTU units (C) and percentage of sand (D), in all study sites, Hurghada, Egypt.

Spearman's correlations between 2004 mean branched and massive coral cover and sedimentation, SPM, non-carbonate sand, turbidity, gravel sand and mud were shown in table (4.7). The results indicated that branched coral cover had significant negative correlation with sedimentation rate (R=-0.893, P=0.003), SPM (R=-0.786, P=0.018), non-carbonate sand (R=-0.929, P=0.001) and turbidity (R=-0.964, P=0.000) (Figure 4.5). Branched coral cover showed a positive correlation with the percentage of sand (R=0.821, P=0.012). Massive coral cover in 2004 did not show any significant correlations with any of these parameters.

Table 4.7: Correlations between 2004 mean readings of branched and massive coral cover with sedimentation, SPM, non- carbonate sand, turbidity, gravel sand and mud for the seven study sites, Hurghada, Egypt.

		Sediment	SPM	Non carbonate	Turbidity	Gravel	Sand	Mud
Branched 2004	Correlation Coefficient	-.893	-.786	-.929	-.964	-.214	.821	-.321
	Sig. (1-tailed)	.003(***)	.018(*)	.001(***)	.000(***)	.322(ns)	.012(*)	.241(ns)
	N	7	7	7	7	7	7	7
Massive 2004	Correlation Coefficient	-.357	-.393	-.286	-.107	.071	.000	-.643
	Sig. (1-tailed)	.216(ns)	.192(ns)	.267(ns)	.41(ns)	.44(ns)	0.5(ns)	.06(ns)
	N	7	7	7	7	7	7	7



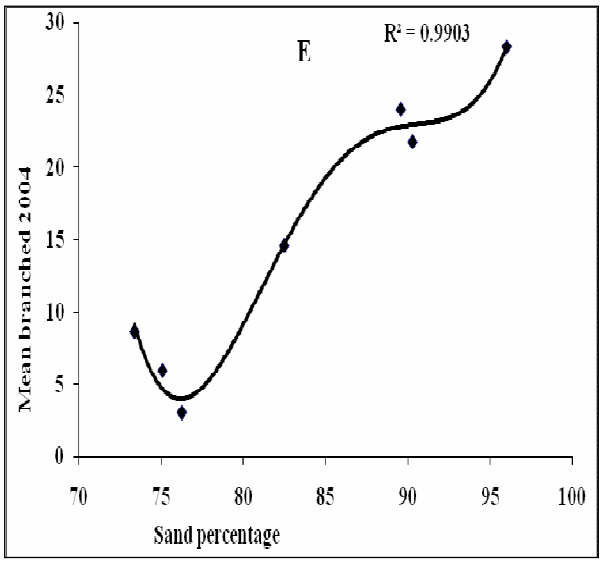


Figure 4.5: Correlations between 2004 branched coral cover and sedimentation rate in $mg.cm^{-2}.day^{-1}$ (A), SPM in $mg.l^{-1}$ (B), non-carbonate percentage (C), turbidity in NTU units (D); and sand percentage (E) in all study sites, Hurghada, Egypt.

Spearman's correlations between 2006 mean branched and massive coral cover and sedimentation, SPM, non-carbonate sand, turbidity, gravel sand and mud (Table 4.8), indicated various degrees of correlations. Significant negative correlations were found between 2006 mean branched cover and, sedimentation rate (R=-0.893, P=0.003), SPM (R=-0.893, P=0.003) and turbidity (R=-0.964, P=0.000). Branched coral cover showed a positive correlation with the percentage of sand (R=0.821, P=0.012) (Figure 4.6). Massive coral cover of 2006 did not show any significant correlations with any of these parameters. These results indicate nearly the same trend as seen in 2004 coral cover correlations.

Table 4.8: Spearman's correlations between 2006 mean readings of branched and massive coral cover with sedimentation, SPM, non- carbonate sand, turbidity, gravel sand and mud for the seven study sites, Hurghada, Egypt.

		Sediment-ation	SPM	Non carbonate	Turbidity	Gravel	Sand	Mud
Branched 2006	Correlation	-.893	-.893	-.607	-.964	-.214	.821	-.321
	Sig. (1-tailed)	.003(***)	.003(***)	.074(ns)	.000(***)	.322(ns)	.012(*)	.241(ns)
	N	7	7	7	7	7	7	7
Massive 2006	Correlation	-.536	-.536	-.143	-.286	-.071	.179	-.607
	Sig. (1-tailed)	.108(ns)	.108(ns)	.380(ns)	.267(ns)	.440(ns)	.351(ns)	.074(ns)
	N	7	7	7	7	7	7	7

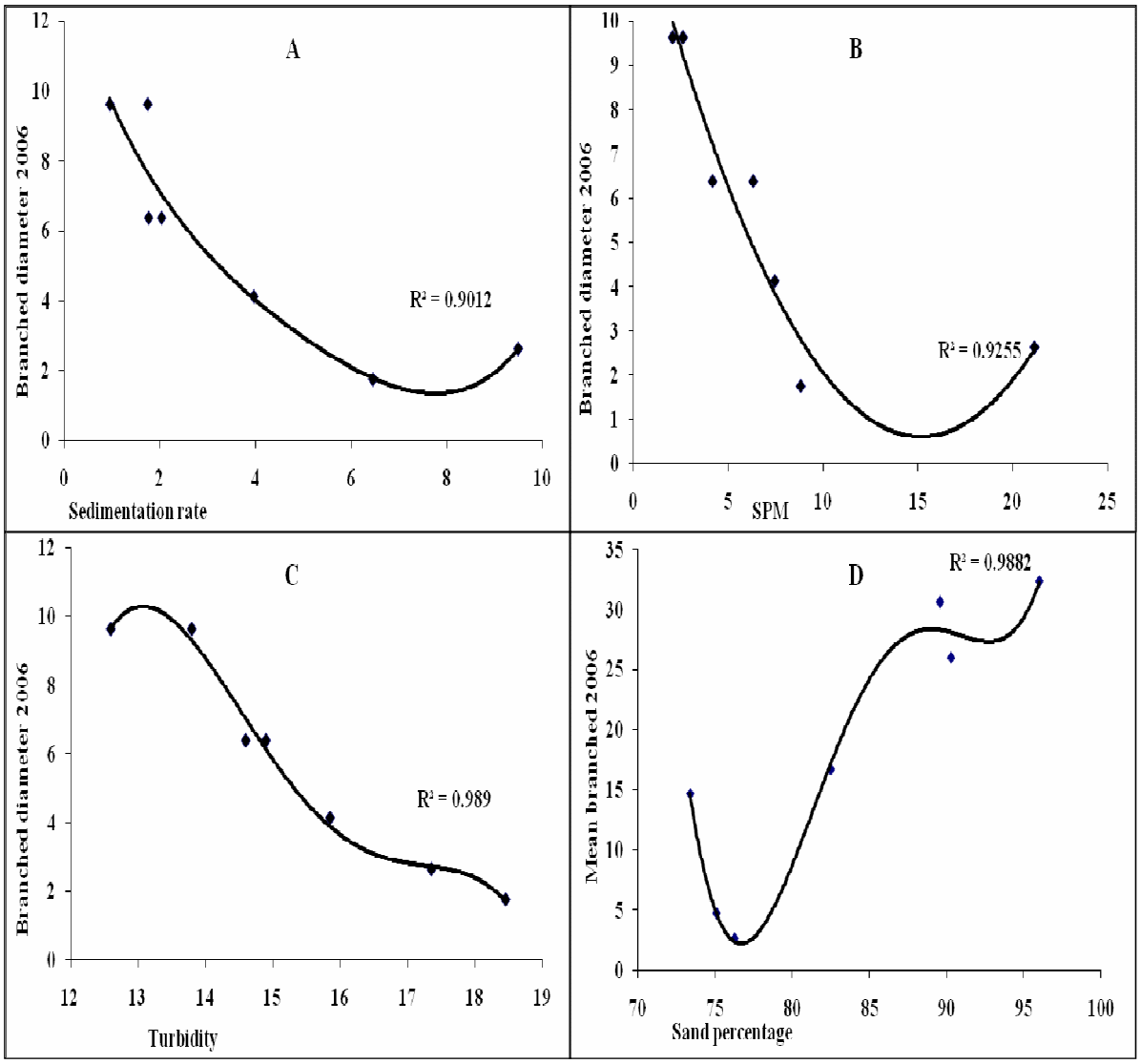


Figure 4.6: Correlations between 2006 branched coral cover and sedimentation rate in $mg.cm^{-2}.day^{-1}$ (A), SPM in $mg.l^{-1}$ (B), turbidity in NTU units (C); and sand (D) in all study sites, Hurghada, Egypt.

The number of coral colonies varied greatly between sites for the two surveys 2004 and 2006, as counted along the Line Belt Transect; for live, dead and recruitment at each site. The results are displayed in Figure (4.7) for the 2004 and Figure (4.8) for the 2006 survey. The mean number of live coral colonies per $5m^2$ was at its highest (106 colonies) at site 6 and the lowest (21 colonies) number was at site 3. Number of recruits did not show much difference between the two surveys, but between sites (Figure 4.9), the highest number (66.5) was found at site 6 while the lowest was (3.5) at site 3. The mean number of dead coral colonies varied from 7 at site 7 to 2.75 at site 2, also it show high difference between the two surveys (Figure 4.10).

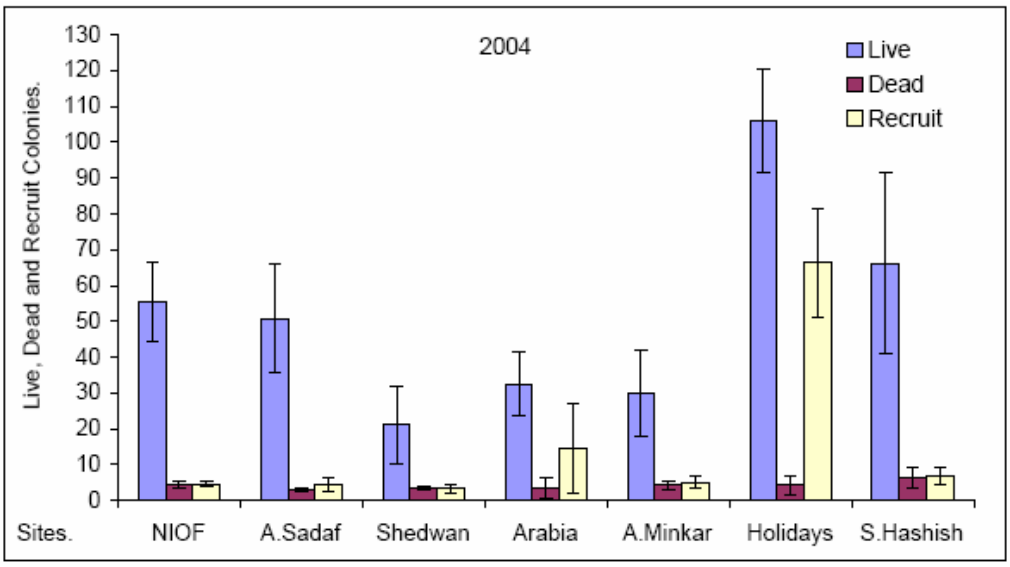


Figure 4.7: The mean numbers of coral colonies for live coral, dead coral and recruits as counted per five square metre for the year 2004 survey at the seven study sites, Hurghada, Egypt.

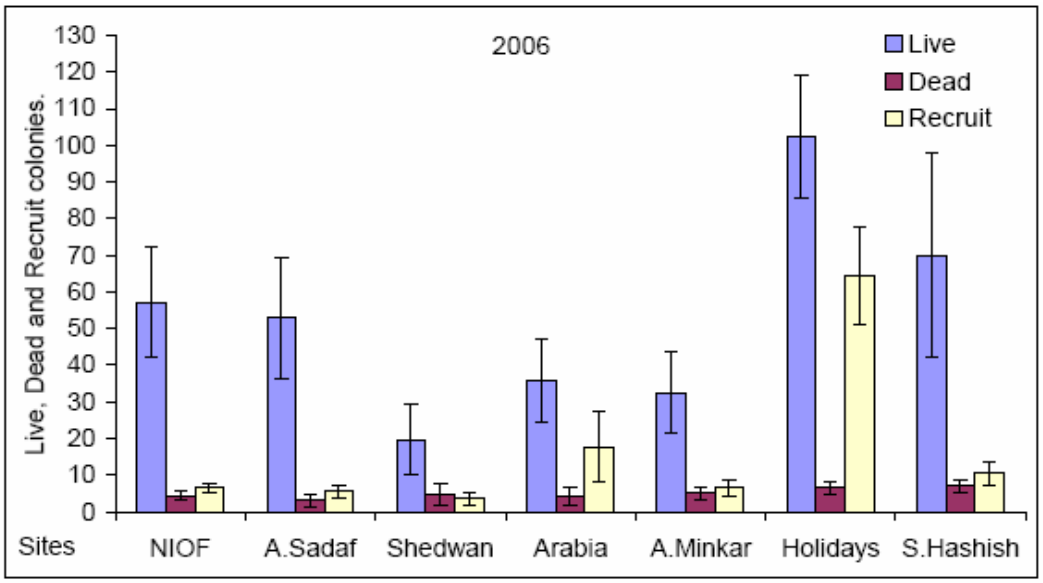


Figure 4.8: The mean numbers of coral colonies for live coral, dead coral and recruits as counted per five square metre for the year 2006 survey at the seven study sites, Hurghada, Egypt.

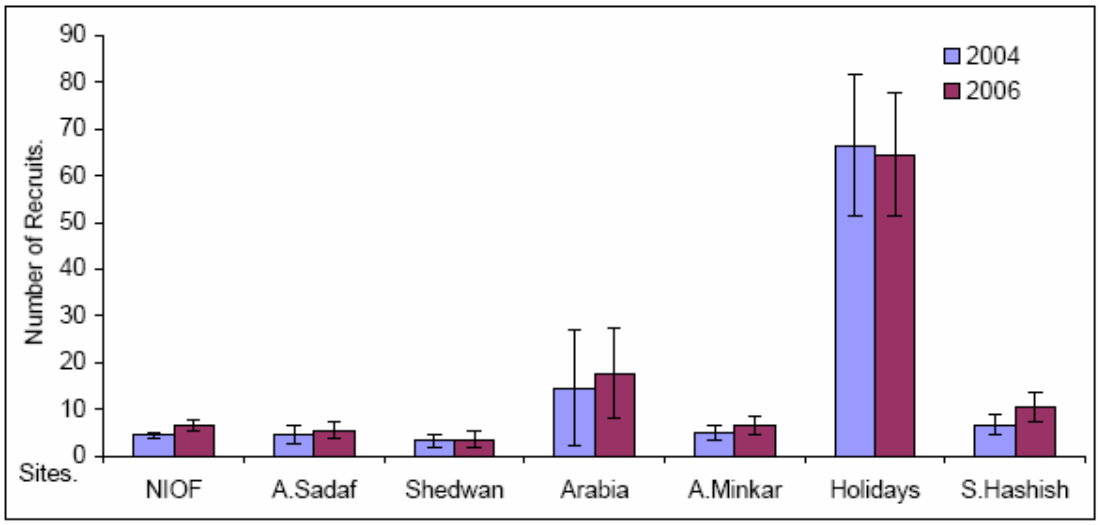


Figure 4.9: The mean number of coral recruits, as counted per five square metres for the two years 2004 and 2006, at the seven study sites, Hurghada, Egypt.

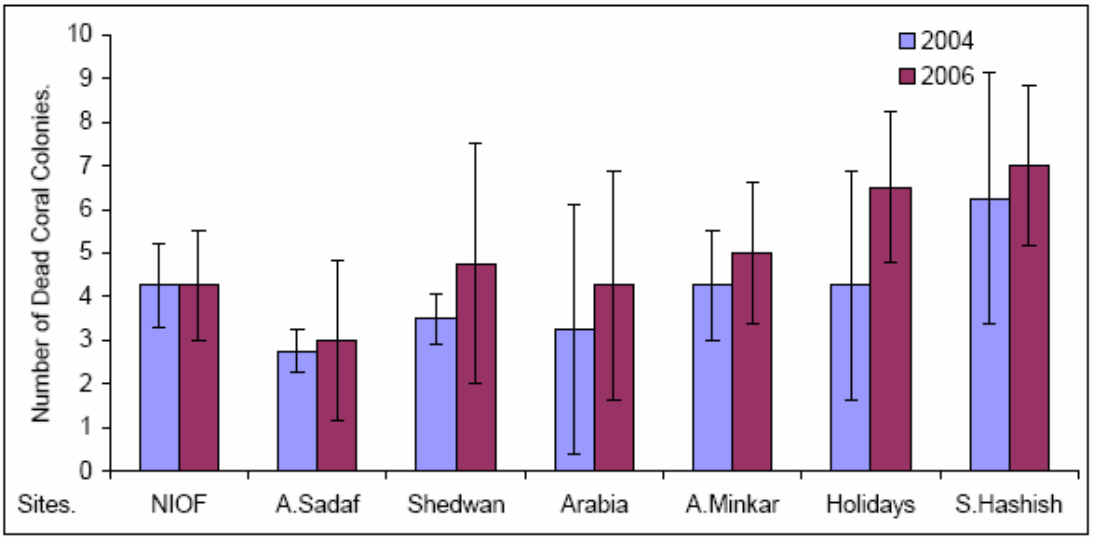


Figure 4.10: The mean number of dead coral colonies, as counted per five square metres for the two years 2004 and 2006 surveys at the seven study sites, Hurghada, Egypt.

Univariate analysis of variance was carried for number of live coral colonies, recruits number and dead colonies number to check whether each variable differed significantly between sites and years. Number of live coral colonies indicated a significant difference between sites (F=25.6, df=6, P=0.000) but not between years (F=0.088, df=1, P=0.768) (Table 4.9). A Post Hoc test showed that site 1 is significantly different from sites 3, 5 and 6; site 2 was significantly different from sites 3 and 6; site 3 is significantly different from sites 1, 2, 6 and 7; site 4 was significantly different from sites 6 and 7; site 5 is significantly different from sites 1, 6 and 7; site 6 was significantly different from all other sites; site 7 was significantly different from sites 3, 4, 5 and 6 (Table 4.10).

The number of recruits coral colonies indicated a significant difference between sites (F=81.2, df=6, P=0.000) but not between years (F=0.552, df=1, P=0.462) (Table 4.11). Post Hoc test showed that site 1 was significantly different from sites 6; site 2 and 3 were significantly different from sites 4 and 6; site 4 is significantly different from sites 2, 3 and 6; site 5 is significantly different from sites 6; site 6 was significantly different from all other sites; site 7 was significantly different from site 6 (Table 4.10).

The number of dead coral colonies indicated a significant difference between sites (F=2.95, df=6, P=0.017) but not between years (F=2.84, df=1, P=0.099) (Table 4.12). A Post Hoc test showed that sites 1, 3, 4, 5 and 6 were not significantly different from any other site; site 2 was significantly different from site 7 (Table 4.13).

Table 4.9: Two-way ANOVA test for the number of live coral colonies, between sites and year variations, recorded in 2004 and 2006 at the 7 study sites, Hurghada, Egypt.

Source of variation	S.S.	D.f.	M. S.	F	P-value
Year	21.875	1	21.875	.088	.768
Sites	38069.179	6	6344.863	25.609	.000
Year * Sites	83.750	6	13.958	.056	.999
Error	10405.750	42	247.756		
Total	201987.00	56			

Table 4.10: The significance matrix between sites in the number of live coral colonies (A), and the number of recruits to coral colonies (B), recorded in 2004 and 2006 at the seven study sites, Hurghada, Egypt.

	A							B						
Sites	1	2	3	4	5	6	7	1	2	3	4	5	6	7
1														
2														
3	x	x												
4									x	x				
5	x													
6	x	x	x	x	x			x	x	x	x	x		
7			x	x	x	x							x	

Table 4.11: Two-way ANOVA test for the number of recruits coral colonies, between sites and year differences, recorded in 2004 and 2006 at the 7 study sites, Hurghada, Egypt.

Source of variation	S.S.	D.f.	M. S.	F	P-value
Year	27.161	1	27.161	.552	.462
Sites	23997.714	6	3999.619	81.299	.000
Year * Sites	44.714	6	7.452	.151	.988
Error	2066.250	42	49.196		
Total	39933.000	56			

Table 4.12: Two-way ANOVA test for the number of dead coral colonies, between sites and year differences, recorded in 2004 and 2006 at the 7 study sites, Hurghada, Egypt.

Source of variation	S.S.	D.f.	M. S.	F	P-value
Year	11.161	1	11.161	2.845	.099
Sites	69.607	6	11.601	2.958	.017
Year * Sites	6.464	6	1.077	.275	.946
Error	164.750	42	3.923		
Total	1395.000	56			

Table 4.13: The significance matrix between sites in the number of dead coral colonies, recorded in 2004 and 2006 as indicated by Tukey test, for the 7 study sites, Hurghada, Egypt.

Sites	1	2	3	4	5	6	7
1							
2							
3							
4							
5							
6							
7		x					

Pearson correlation was carried out to test the correlation of the 2004 number of live coral colonies with sedimentation, SPM, non- carbonate sand, turbidity, gravel sand and mud (Table 4.14). Results showed a significant positive correlation with non-carbonate (R=0.672, P=0.049) and mud percentage (R=0.783, P=0.019). Spearman's rho correlation was carried out to test the relation between 2004 recruits and number of dead colonies with the same parameters. Test results indicated significant positive correlation between recruit number and the percentage of mud (R=0.703, P=0.039). Dead colonies number showed a significant negative correlation with gravel percentage (R=-0.778, P=0.02) (Table 4.15). Significant Correlations with non-carbonate, mud and gravel percentage were plotted in figure (4.11) with R^2 value.

Pearson correlation was carried also to test the correlation of the 2006 number of coral colonies with sedimentation, SPM, non- carbonate sand, turbidity, gravel sand and mud (Table 4.16). Results indicated that a significant negative correlation with non- carbonate (R=-0.686, P=0.044) and mud percentage (R=-0.739, P=0.029) were found. Spearman's rho correlation was carried out to test the correlations of the same parameters and 2006 recruits and dead colonies. Test results indicated significant negative correlations between recruits and gravel percentage (R=-0.685, P=0.045). Dead colonies number showed a significant negative correlation with gravel percentage (R=-0.775, P=0.02) (Table 4.17). Significant Correlations with gravels was plotted in figure (4.12) with R^2 value.

Table 4.14: The correlations between 2004 readings of number of live coral colonies, with sedimentation rate, SPM, non- carbonate sand, turbidity, gravel sand and mud for the seven study sites, Hurghada, Egypt.

		Sediment-ation	SPM	Non-carbonate	Turbidity	Gravel	Sand	Mud
Live 2004	Pearson Correlation	-.190	-.352	.672	.378	-.504	-.173	.783
	Sig. (1-tailed)	.341(ns)	.219(ns)	.049(*)	.201(ns)	.124(ns)	.355(ns)	.019(*)
	N	7	7	7	7	7	7	7

Table 4.15: The Spearman correlation between 2004 readings of recruits and dead coral cover with sedimentation, SPM, non- carbonate sand, turbidity, gravel sand and mud for the seven study sites, Hurghada, Egypt.

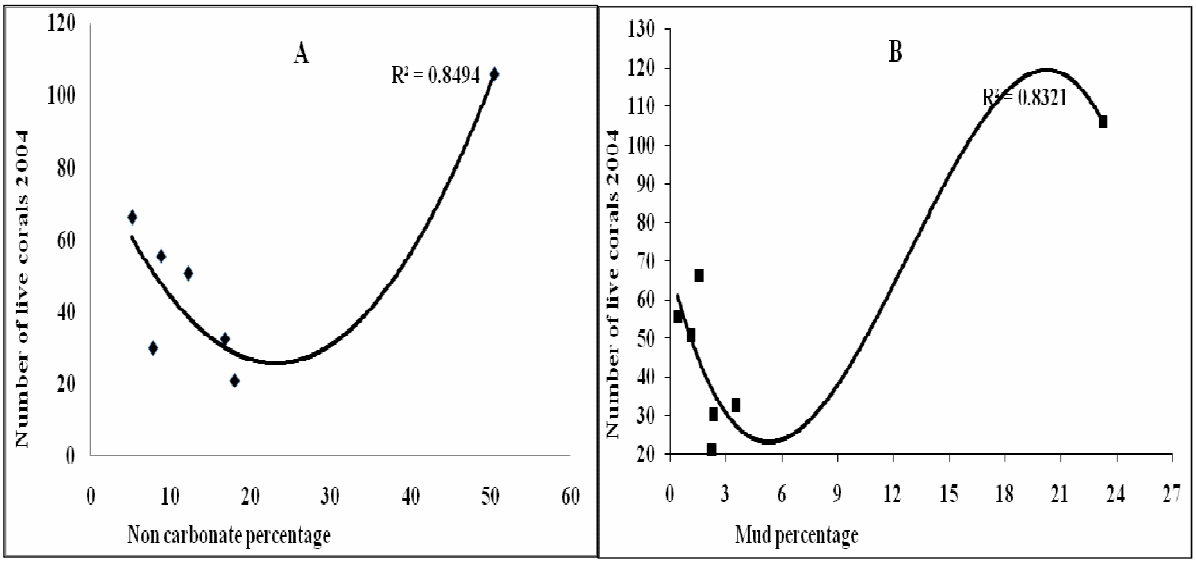
		Sediment-ation	SPM	Non-carbonate	Turbidity	Gravel	Sand	Mud
Recruits 2004	Correlation Coefficient	-.090	-.126	.126	.162	-.667	.198	.703
	Sig. (1-tailed)	.424(ns)	.394(ns)	.394(ns)	.364(ns)	.051(ns)	.335(ns)	.039 (*)
	N	7	7	7	7	7	7	7
Dead 2004	Correlation Coefficient	-.556	-.667	-.482	-.408	-.778	.667	.000
	Sig. (1-tailed)	.098(ns)	.051(ns)	.137(ns)	.182(ns)	.02(*)	.051(ns)	0.50(*)
	N	7	7	7	7	7	7	7

Table 4.16: The correlation between 2006 readings of number of live coral colonies with sedimentation, SPM, non- carbonate sand, turbidity, gravel sand and mud for the seven study sites, Hurghada, Egypt.

		Sediment-ation	SPM	Non-carbonate	Turbidity	Gravel	Sand	Mud
Live 2006	Pearson Correlation	-.145	-.366	.686	.313	-.509	-.127	.739
	Sig.(1-tailed)	.378(ns)	.210(ns)	.044(*)	.247(ns)	.121(ns)	.393(ns)	.029(*)
	N	7	7	7	7	7	7	7

Table 4.17: The Spearman's correlation between 2006 readings of number of recruits and dead coral cover with sedimentation, SPM, non- carbonate sand, turbidity, gravel sand and mud for the seven study sites, Hurghada, Egypt.

		Sediment-ation	SPM	Non-carbonate	Turbidity	Gravel	Sand	Mud
Recruit 2006	Correlation Coefficient	-.090	-.090	.306	.180	-.685	.270	.613
	Sig. (1-tailed)	.424(ns)	.424(ns)	.252(ns)	.350(ns)	.045(*)	.279(ns)	.072(ns)
	N	7	7	7	7	7	7	7
Dead 2006	Correlation Coefficient	-.252	-.252	.306	-.108	-.775	.360	.414
	Sig. (1-tailed)	.293(ns)	.293(ns)	.252(ns)	.409(ns)	.02(*)	.214(ns)	.178(ns)
	N	7	7	7	7	7	7	7



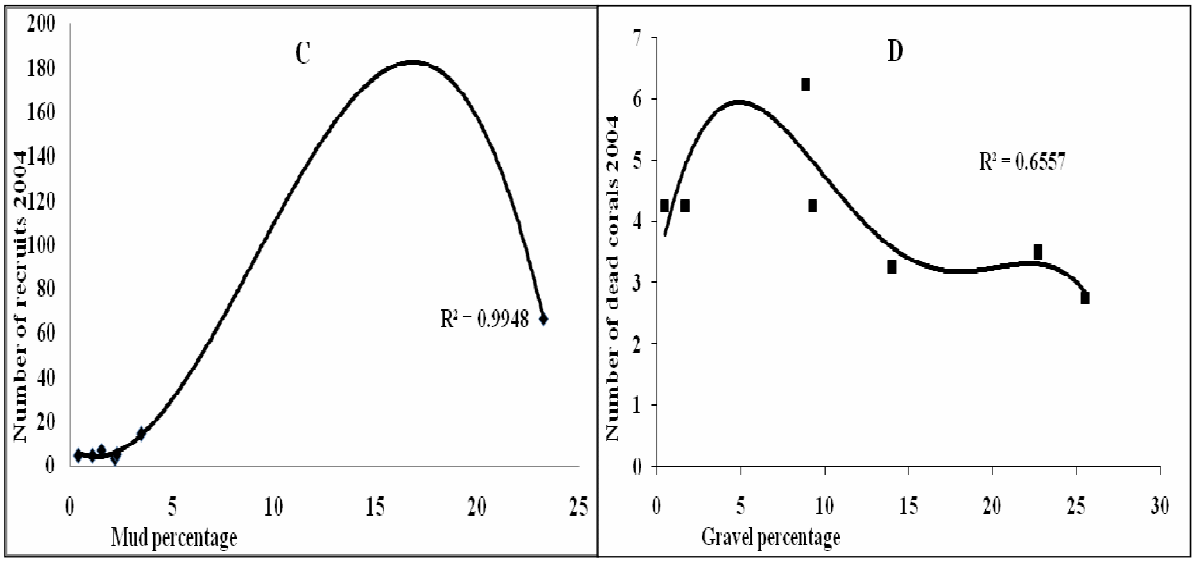
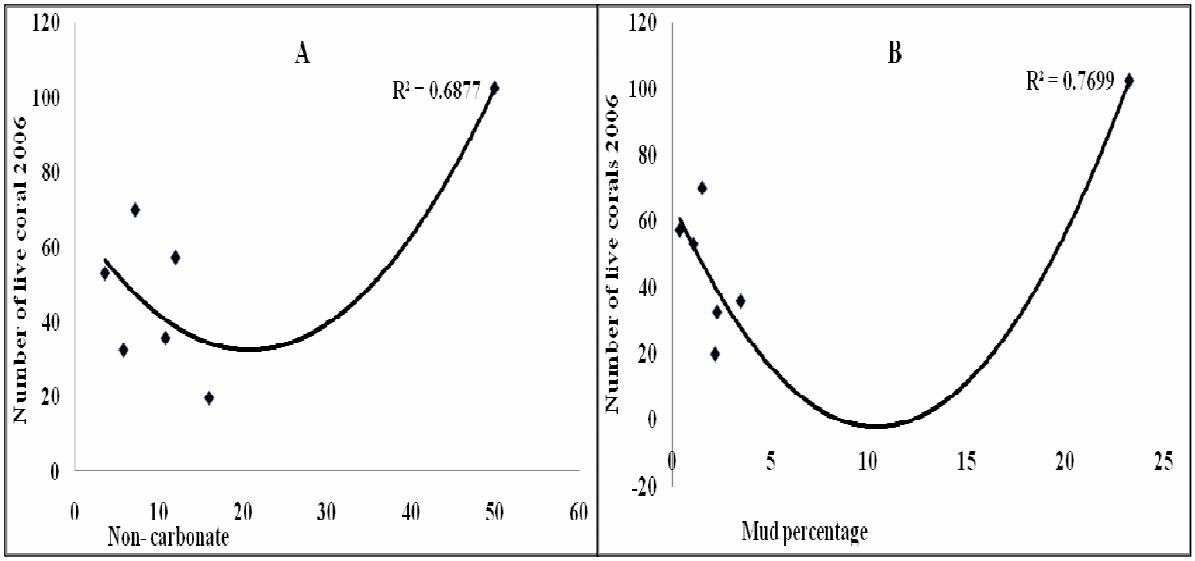


Figure 4.11: The correlations between 2004 number of coral colonies and percentage of non carbonate (A) and mud (B), and between recruits and the percentage of mud (C), and between dead colonies and gravel (D) in bottom sediment, in all study sites, Hurghada, Egypt.



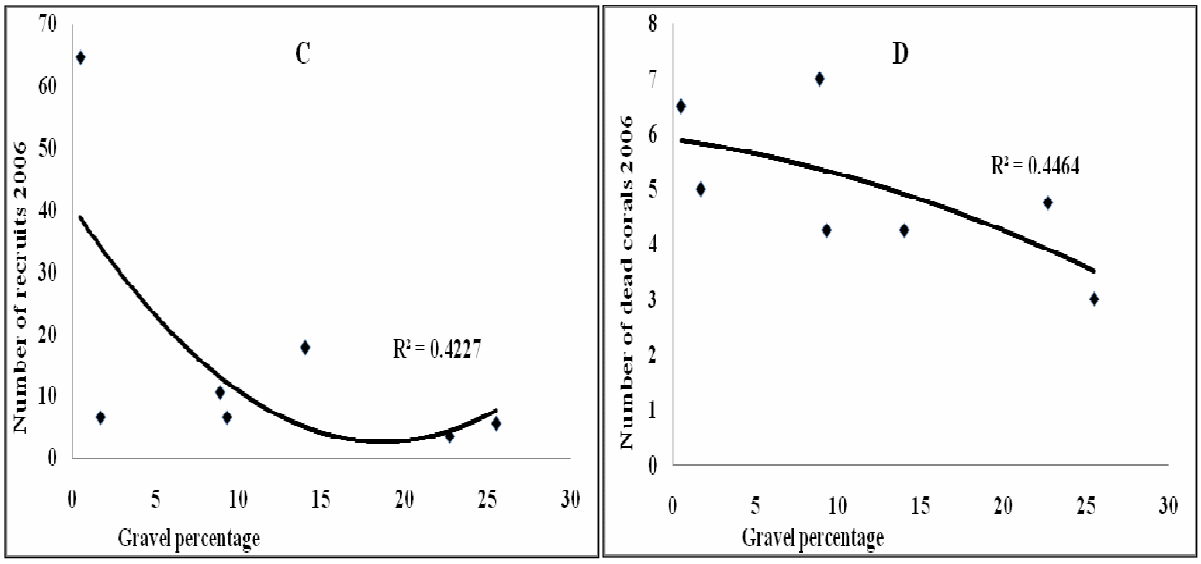


Figure 4.12: The correlations between 2006 number of live corals and non carbonate percentage (A), and mud percentage (B), recruit colonies and gravel percentage (C) and dead coral colonies and percentage of gravel (D) in bottom sediment, in all study sites, Hurghada, Egypt.

4.2 Coral Abundance

The coral abundance (number of colonies per unit area) was determined for each site. Results showed that site 6 had the highest coral abundance although it also had the lowest coral cover in both 2004 and 2006 (Figure 4.13). This paradox could be attributed to the high recruitment rate and the high number of small size colonies. The lowest abundance was recorded at site 3 and 5.

Statistical analysis using two-way ANOVA showed no significant difference (F=0.696,df=1, P=0.409) between the two surveys (2004 and 2006) in coral abundance as shown in Table 4.18. The results showed high significant difference between sites (F=18.6,df=6, P=0.000) and no significant effect of the combined year and site factor (F=0.29, df=6, P=0.93). The Post Hoc test showed that site 1 was significantly different from sites 3 and 6; site 2 was significantly different from site 6; site 3 was significantly different from sites 1, 6 and 7; sites 4 and 5 were significantly different from sites 6 and 7; site 6 was significantly different from all other sites; and site 7 was significantly different from sites 3, 4 and 5 and 6 (Table 4.19).

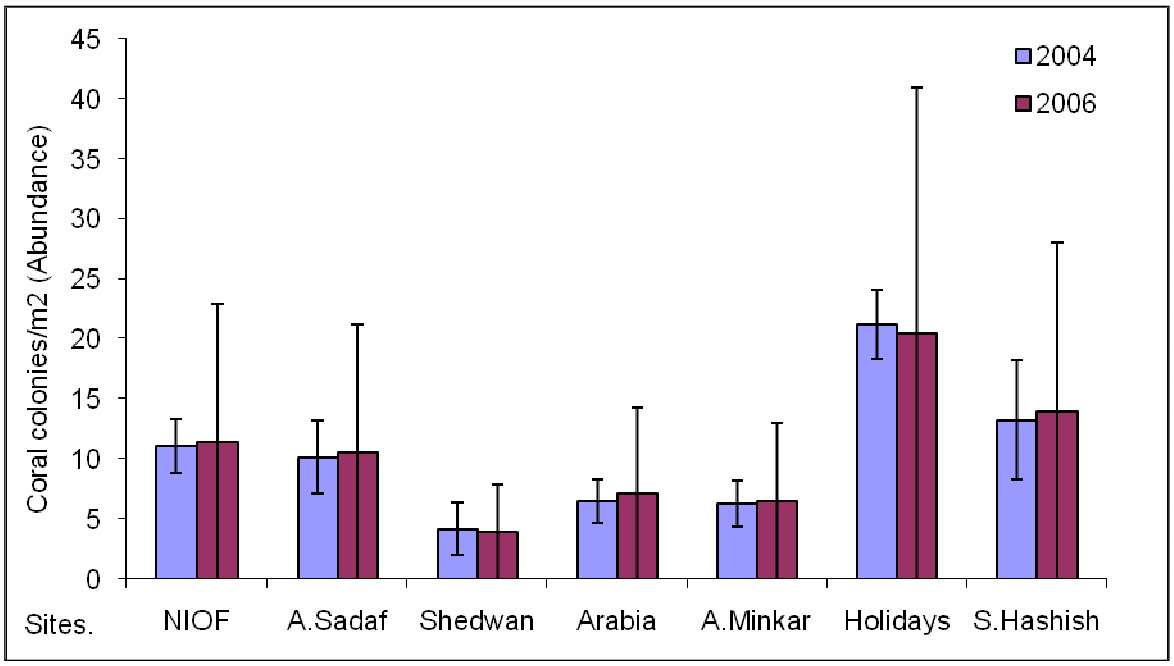


Figure 4.13: The mean (n=4) coral abundance ± standard deviation of mean for each of the study sites in Hurghada area, Egypt.

Table 4.18: The two-way ANOVA test for coral abundance of the 7 sites in Hurghada reef, Egypt, recorded in 2004 and 2006.

Source of variation	S.S.	D.f.	M. S.	F	P-value
Year	12.446	1	12.446	.696	.409
Sites	1997.424	6	332.904	18.606	.000
Year * Sites	31.504	6	5.251	.293	.937
Error	751.460	42	17.892		
Total	9411.360	56			

Table 4.19: The significance matrix between sites in coral abundance, recorded in 2004 and 2006 as indicated by Tukey test, for the 7 study sites, Hurghada, Egypt.

sites	1	2	3	4	5	6	7
1							
2							
3	x						
4							
5							
6	x	x	x	x	x		
7			x	x	x	x	

Pearson correlation tests (Table 4:20) indicated significant positive correlation between 2004 coral abundance and non-carbonate sediment (R=0.671, P=0.049) and mud percentage (R=0.785, P=0.018) (Figure 4.14). However there was positive significant correlation between coral abundance and mud (R=0.785, P=0.037) as shown in Figure (4.14A). Sediment, SPM, non-carbonate sediment, turbidity, sand and gravel did not have any significant correlation with 2006 abundance readings (Table 4.21). Abundance readings of the 2006 almost showed a positive significant correlation with mud (R=0.739, P=0.058) (Figure 4.14B).

Table 4.20: Correlations between 2004 coral abundance and sedimentation rate, SPM level, turbidity, percentage of non-carbonate, gravel, sand and mud in deposited sediment of the study sites, Hurghada, Egypt.

		Sediment ation	SPM	Non-carbonate	Turbidity	Gravel	Sand	Mud
Abundance 2004	Pearson Correlation	-.197	-.361	.671	.369	-.516	-.162	.785
	Sig. (1 tailed)	.336(ns)	.213(ns)	.049(*)	.208(ns)	.118(n s)	.364(ns)	.018(*)
	N	7	7	7	7	7	7	7

Table 4.21: Correlations between 2006 coral abundance and sedimentation rate, SPM level, turbidity, percentage of non-carbonate, gravel, sand and mud in deposited sediment of the study sites, Hurghada, Egypt.

		Sediment -ation	SPM	Non-carbonate	Turbidity	Gravel	Sand	Mud
Abundance 2006	Pearson Correlation	-.145	-.366	.686	.313	-.509	-.127	.739
	Sig. (1-tailed)	.378(ns)	.210(ns)	.044(*)	.247(ns)	.121(ns)	.393(ns)	.029(*)
	N	7	7	7	7	7	7	7

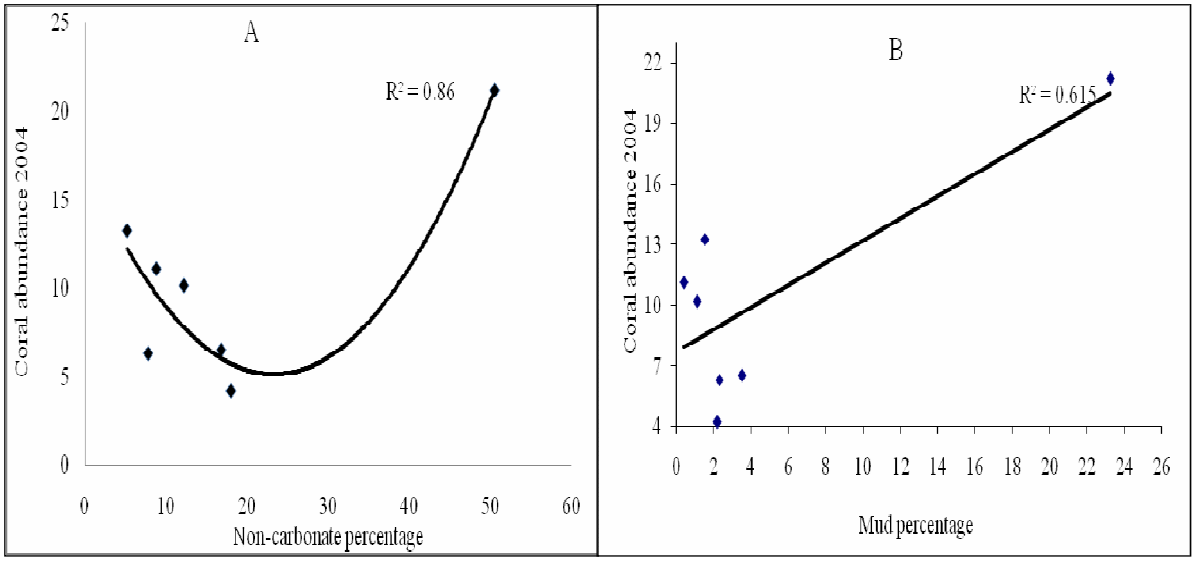


Figure 4.14: The correlations between 2004 coral abundance and non-carbonate (A) and percentage of mud (B), in all study sites, Hurghada, Egypt.

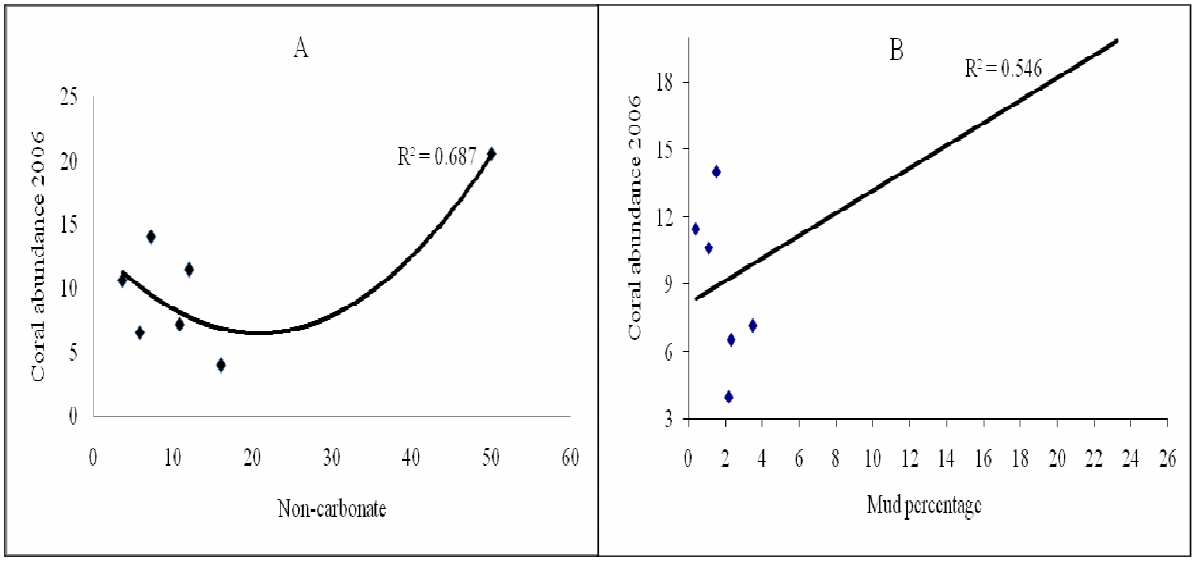


Figure 4.15: The correlations between 2006 coral abundance and non-carbonate (A) and percentage of mud (B), in all study sites, Hurghada, Egypt.

4.3 Species Richness

Number of hard coral species per unit area or species richness is used as a diversity and reef health measure. Figure 4.16 shows the mean species richness for the years 2004 and 2006 and standard deviation for the seven sites. The outcome of the study shows that site 7 had the highest coral species richness although it was not the highest in coral cover or coral abundance. The lowest species richness was recorded at site 3 which also recorded the lowest coral cover and coral abundance as well.

Two way ANOVA results showed a high significant difference (F=22.2, df=6, P=0.000) between sites for coral species richness as shown in Table 4.22, but no significant

difference between years (F=0.336, df=1, P=0.565) and the combined effect of year and site. A Post Hoc test indicated that sites 1 and 2 were significantly different from sites 3, 6 and 7; site 3 was significantly different from sites 1, 2, 4, 5 and 7; site 4 was significantly different from sites 3, 6 and 7; site 5 was significantly different from sites 3 and 7; site 6 was significantly different from sites 1, 2, 4 and 7 and site 7 was significantly different from all the other sites (Table 4.23).

Pearson correlations (Table 4:24) showed a significant negative correlation between 2004 species richness and sedimentation rate (R=-0.839(*), P=0.009) and SPM (R=-0.686(*), P=0.044) (Figure 4.17). The test did not indicate any significant correlations with turbidity, non-carbonate, gravel, sand and mud. Correlation tests were also carried between 2006 species richness and sedimentation rate, SPM, non-carbonate, turbidity, gravel, sand and mud. Pearson correlation results (Table 4:25) showed a significant negative correlation between sedimentation rate (R=-0.805, P=0.014) and 2006 species richness (Figure 4.18) and between species richness and SPM (R=-0.703, P=0.039). Overall, these correlations indicate that sites with high level of sedimentation also had reduced coral species richness and that few species could withstand a high level of sedimentation, while higher non-carbonate sediment was less likely to affect coral species richness.

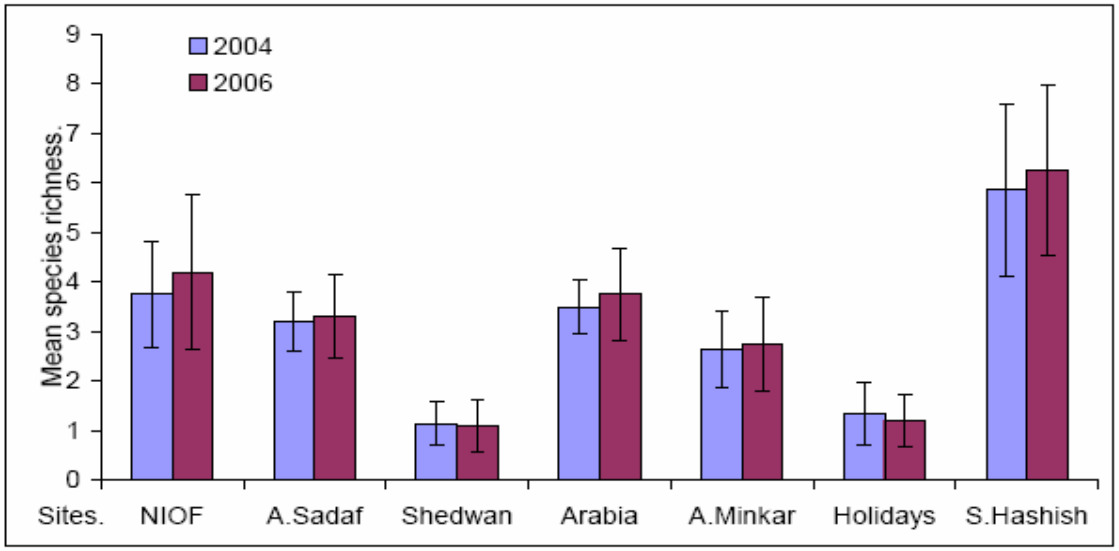


Figure 4.16: The mean species richness (n=4) ± standard deviation of mean in each study site, Hurghada, Egypt.

Table 4.22: Results of two way ANOVA test for the significance level of species richness variation between the seven study sites along Hurghada coast, Egypt.

Source of variation	S.S.	D.f.	M. S.	F	P-value
Year	.346	1	.346	.336	.565
Sites	137.157	6	22.860	22.204	.000
Year * Sites	.594	6	.099	.096	.996
Error	43.240	42	1.030		
Total	734.480	56			

Table 4.23: The significance matrix of species richness variations between sites, recorded in 2004 and 2006; as indicated by Tukey test, for the 7 study sites, Hurghada, Egypt.

Sites	1	2	3	4	5	6	7
1							
2							
3	x	x					
4			x				
5			x				
6	x	x		x			
7	x	x	x	x	x	x	

Table 4.24: The correlations between the 2004 coral species richness and sedimentation rate, SPM, non-carbonate sediment, turbidity, gravel, sand and mud.

		Sediment-ation	SPM	Non-carbonate	Turbidity	Gravel	Sand	Mud
Species richness 2004	Pearson Correlation	-.839	-.686	-.641	-.656	-.047	.512	-.498
	Sig. (1-tailed)	.009(**)	.044(*)	.06(ns)	.055(ns)	.460(ns)	.120(ns)	.128(ns)
	N	7	7	7	7	7	7	7

Table 4.25: The correlations between 2006 coral species richness and; sedimentation rate, SPM, non-carbonate sediment, turbidity, gravel, sand and mud.

		Sedimnt-ation	SPM	Non-carbonate	Turbidity	Gravel	Sand	Mud
Species richness 2006	Pearson Correlatio	-.805	-.703	-.573	-.663	-.044	.531	-.523
	Sig. (1-tailed)	.014(*)	.039(*)	.089(ns)	.052(ns)	.463 (ns)	.110 (ns)	.114 (ns)
	N	7	7	7	7	7	7	7

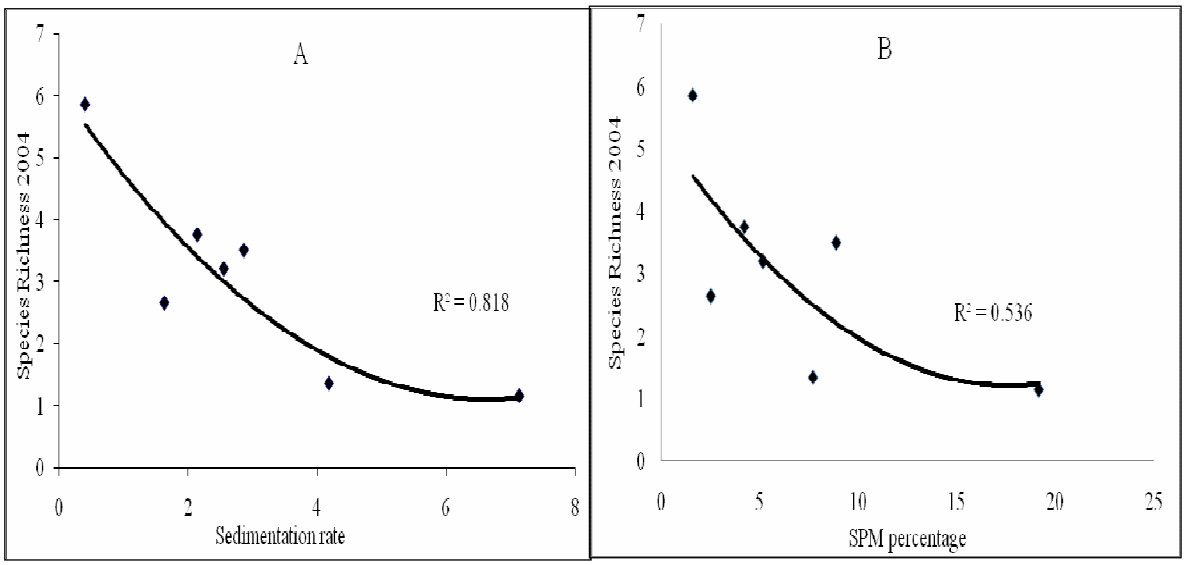


Figure 4.17: Species richness for 2004 and its correlation with mean sedimentation rate (A), and SPM (B), at the 7 study sites, Hurghada, Egypt.

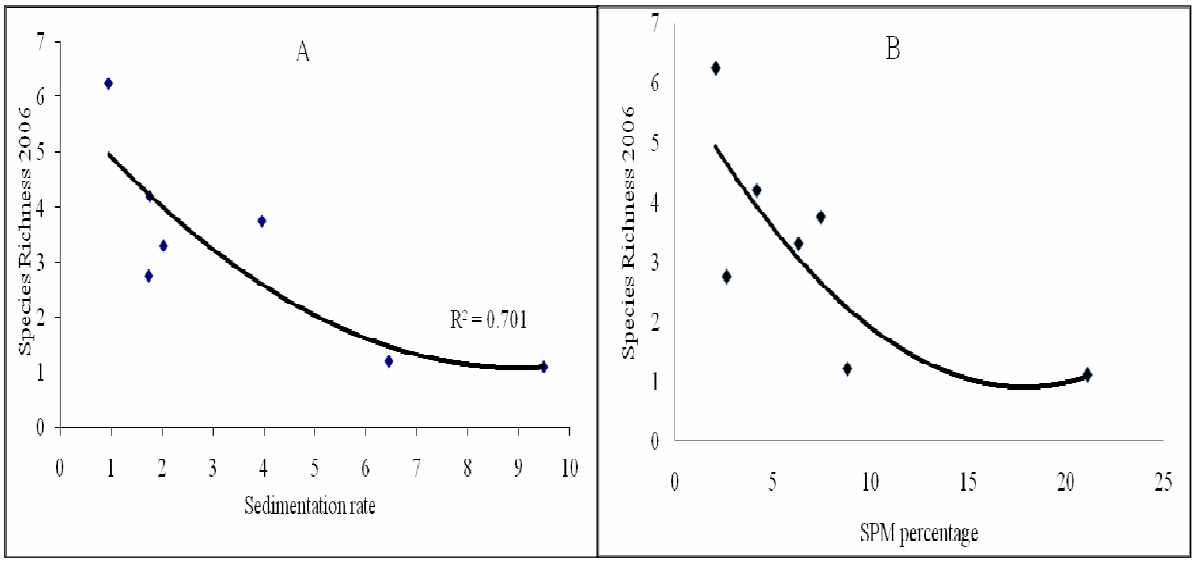


Figure 4.18: Species richness for 2006 and its correlation with mean sedimentation rate (A), and SPM (B), at the 7 study sites, Hurghada, Egypt.

4.4 Species List

Number of species of hard coral were counted along 10m transect and a separate percentage of R-strategist species were calculated (Figure 4.17). This includes all species of the following genera: *Acropora, Pavona, Pocillopora, Stylophora, Psammocora, Seriatopora* and *Montipora.* Results indicated no significant difference between sites in the percentage of r-strategist species (R=11.66, df=6, P=0.070) as shown using a Kruskal Wallis test (Table 4.26). Spearman's correlations were carried out between the percentage of r-strategist species and the environmental parameters; sediment, SPM, non-carbonate sediment, turbidity, gravel, sand and mud. The result indicates no significant correlation with any of these factors (Table 4.27).

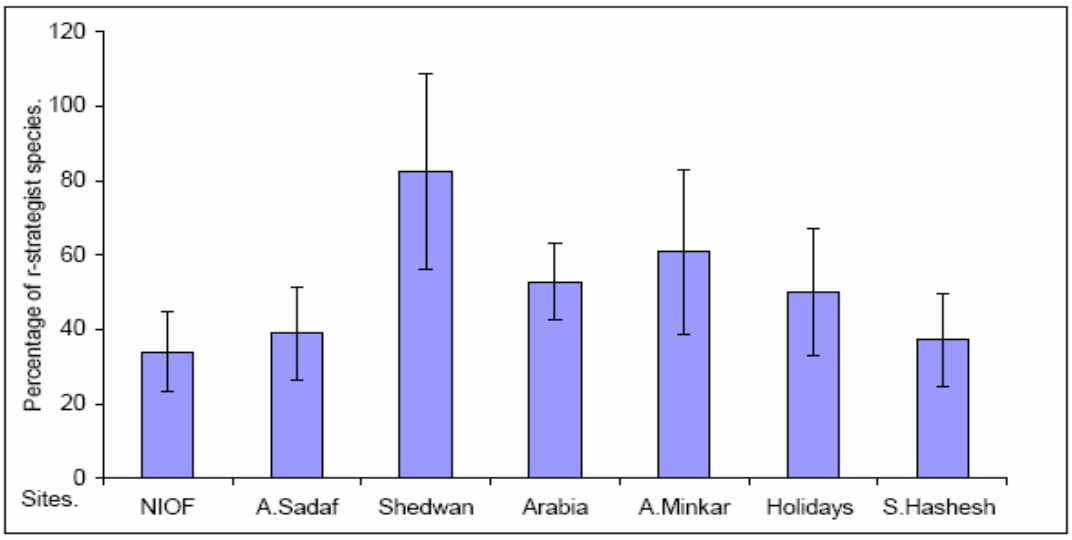


Figure.4.19: The mean (n=4) percentage of R-strategist species ± SD standard deviation of mean, measured at the 7 study sites, Hurghada, Egypt.

Table 4.26: The Kruskal Wallis test for the percentage of r-strategist species and differences between the seven study sites along Hurghada, Egypt.

Sites	N	Mean Rank		r-strategist
1	4	7.63	Chi-Square	11.662
2	4	10.88	df	6
3	4	23.75	Asymp. Sig.	0.070
4	4	17		
5	4	18.5		
6	4	14.63		
7	4	9.13		
Total	28			

Table 4.27: Spearman's correlations between the percentage of R-strategist species and the environmental parameters; sedimentation rate, SPM, non-carbonate sediment, turbidity, gravel, sand and mud.

		Sediment-ation	SPM	non-carbonate	Turbidity	Gravel	Sand	Mud
R-Strategist	Correlation Coefficient	.536	.536	.143	.286	.071	-.179	.607
	Sig. (1-tailed)	.108(ns)	.108(ns)	.380(ns)	.267(ns)	.440(ns)	.351(ns)	.074(ns)
	N	7	7	7	7	7	7	7

4.5 Shannon-Weaver Diversity Index

Using the number of species of hard coral and the number of individual colonies of each species, counted in the field survey, the diversity index for each site was determined. Figure (4.20) shows the mean diversity index and standard deviation of the mean for the two surveys (2004 and 2006) at each site. This showed that the highest diversity was at site 7, and site 1 was second, while lowest diversity was found at site 3.

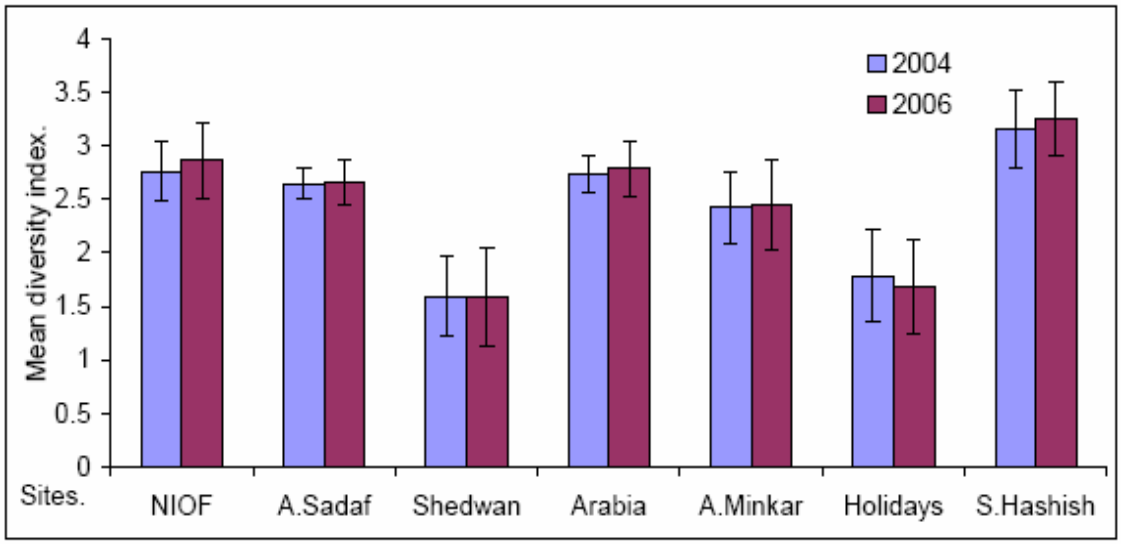


Figure 4.20: Shows means (n=4) of diversity index readings and standard deviation of mean for each site, Hurghada, Red Sea.

Two way ANVOA (Table 4.28) showed a highly significant difference between sites in diversity index (F=23.78, df=6, P=0.000) but not between years. A Post Hoc test showed that sites 1, 2 and 4 were significantly different from sites 3 and 6; site 3 was significantly different from sites 1, 2, 4, 5 and 7; site 5 was significantly different from sites 3, 6 and 7; site 6 was significantly different from sites 1, 2, 4, 5 and 7 and site 7 was significantly different from sites 3, 5 and 6 (Table 4.29).

Table 4.28: Two way ANOVA results for diversity index and significance level between years and study sites, Hurghada, Egypt.

Source of variation	S.S.	D.f.	M. S.	F	P-value
Year	.009	1	.009	.075	.785
Sites	16.713	6	2.785	23.785	.000
Year * Sites	.055	6	.009	.078	.998
Error	4.919	42	.117		
Total	359.365	56			

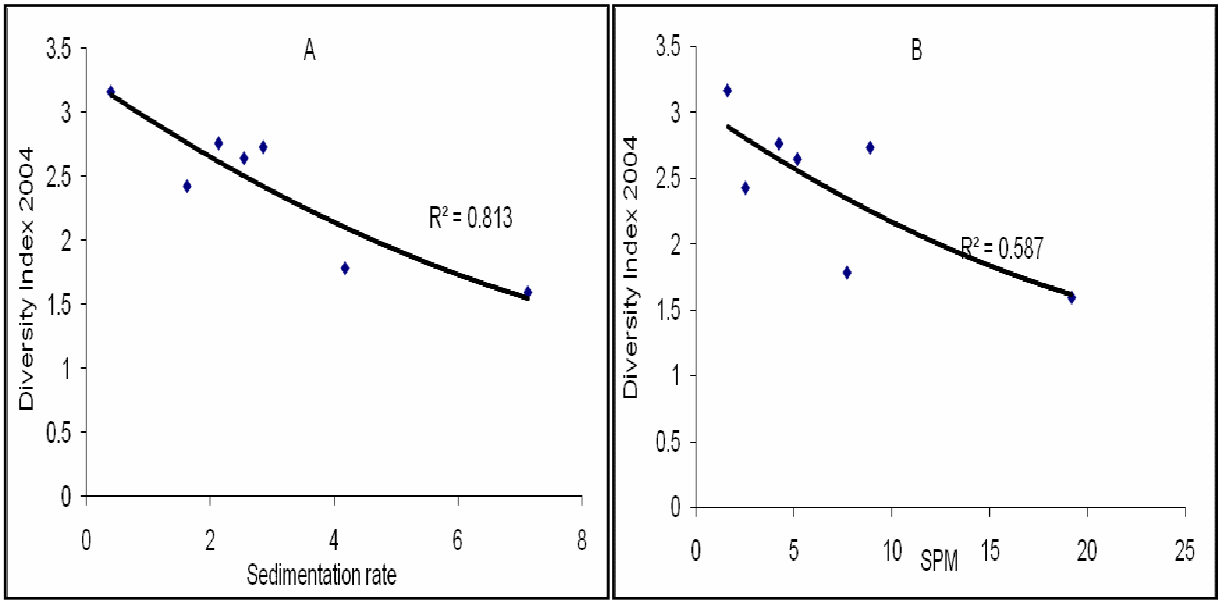
Table 4.29: Significance matrix of diversity index variations between sites, recorded in 2004 and 2006; as indicated by Tukey test, for the 7 study sites, Hurghada, Egypt.

Sites	1	2	3	4	5	6	7
1							
2							
3	x	x					
4			x				
5			x				
6	x	x		x	x		
7			x		x	x	

From the Pearson correlation coefficient calculations (Table 4.30), the diversity indices calculated for 2004 showed significant negative correlations with sedimentation rate (R=-0.896, P=0.003), SPM (R=-0.762, P=0.023), non-carbonate (R=-0.668, P=0.05) and turbidity (R=-0.731, P=0.032) (Figure 4.21). The 2006 diversity indices showed nearly the same trend of correlation indicated in 2004. It showed a significant negative correlation with sedimentation rate (R=-0.891, P=0.004), SPM (R=-0.777, P=0.03) and turbidity (R=-0.743, P=0.028) (Figure 4.22).

Table 4.30: Correlations between 2004 and 2006 diversity indices, and sedimentation rate, SPM, non-carbonate sediment, turbidity, gravel, sand and mud.

		Sediment	SPM	Non-carbonate	Turbidity	Gravel	Sand	Mud
Diversity Index 2004	Pearson Correlation	-.896	-.762	-.668	-.731	-.034	.535	-.538
	Sig. (1-tailed)	.003(***)	.023(*)	.05(*)	.031(*)	.471(ns)	.108(ns)	.107(ns)
	N	7	7	7	7	7	7	7
Diversity index 2006	Pearson Correlation	-.891	-.777	-.659	-.743	-.012	.551	-.581
	Sig. (1-tailed)	.004(***)	.030(*)	.054(ns)	.028(*)	.490(ns)	.100(ns)	.086(ns)
	N	7	7	7	7	7	7	7



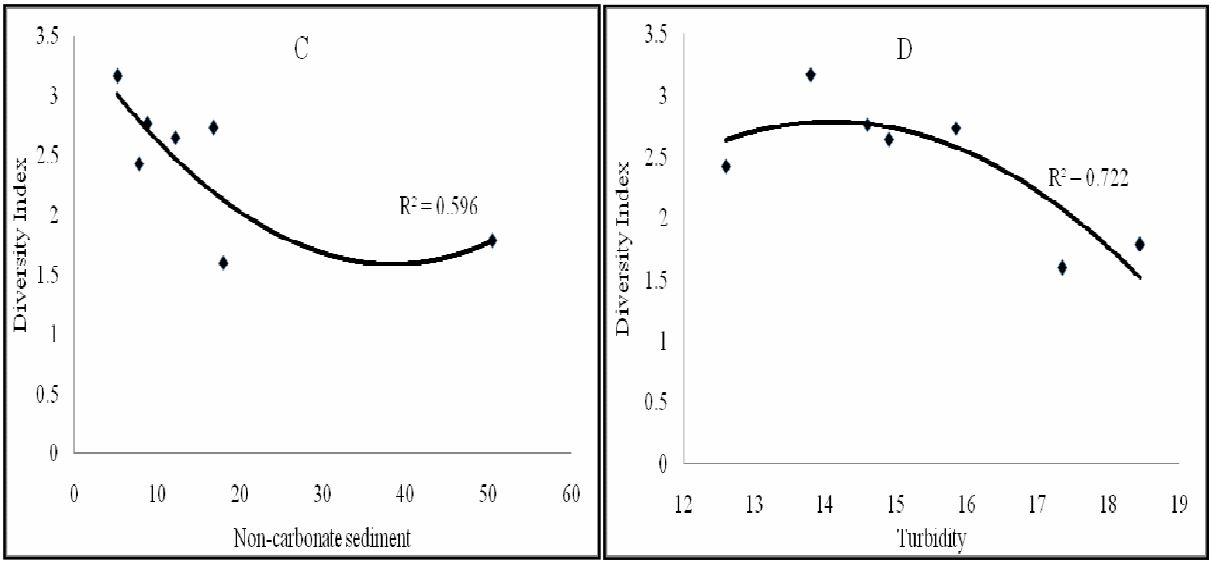
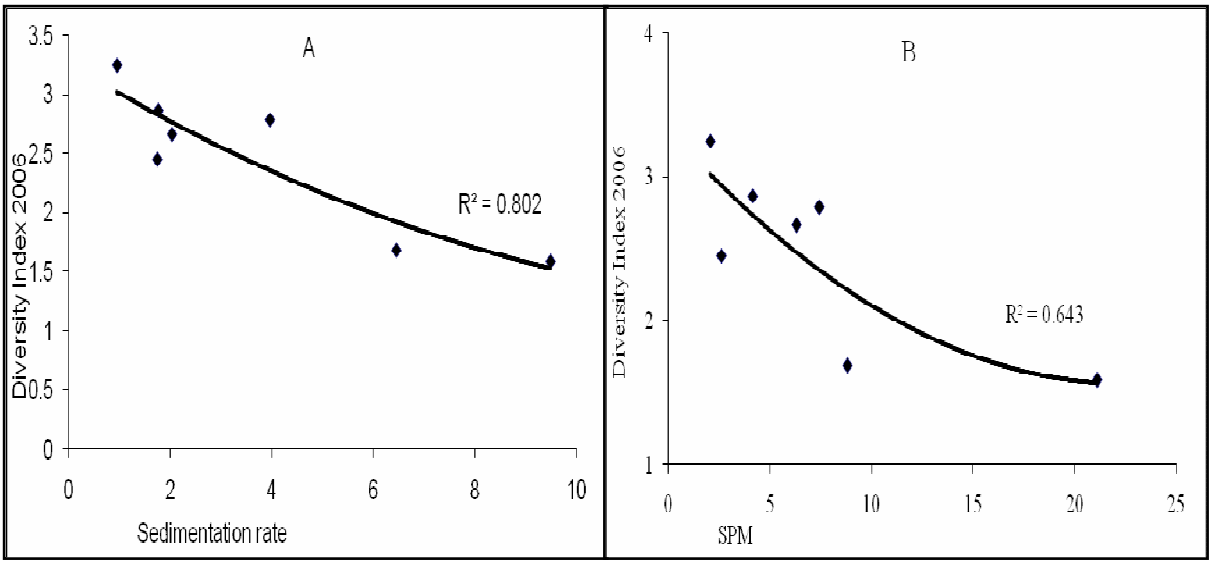


Figure 4.21: The correlation between 2004 coral diversity index and; sedimentation rate (A), SPM (B), non-carbonate (C) and turbidity (D) for each site, Hurghada, Egypt.



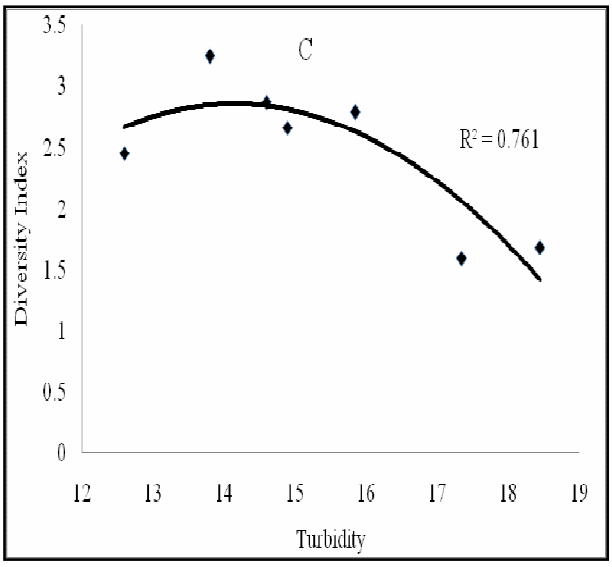


Figure 4.22: The correlation between 2006 coral diversity index and sedimentation rate (A) and SPM (B) for each site, Hurghada, Egypt.

4.6 Coral Deterioration Index (DI)

Site 3 was found to have the highest Deterioration Index score, indicating that it was the poorest of the seven sites (Figure 4.23). Sites 1 and 5 also showed high deterioration level using this index because of the low number of recruits found at each and despite their relative coral percentage cover. Site 6 had the lowest deterioration index although it has the lowest coral cover. Generally the rest of the sites were in or close to stable condition in relation to the ratio between dead coral and number of recruits.

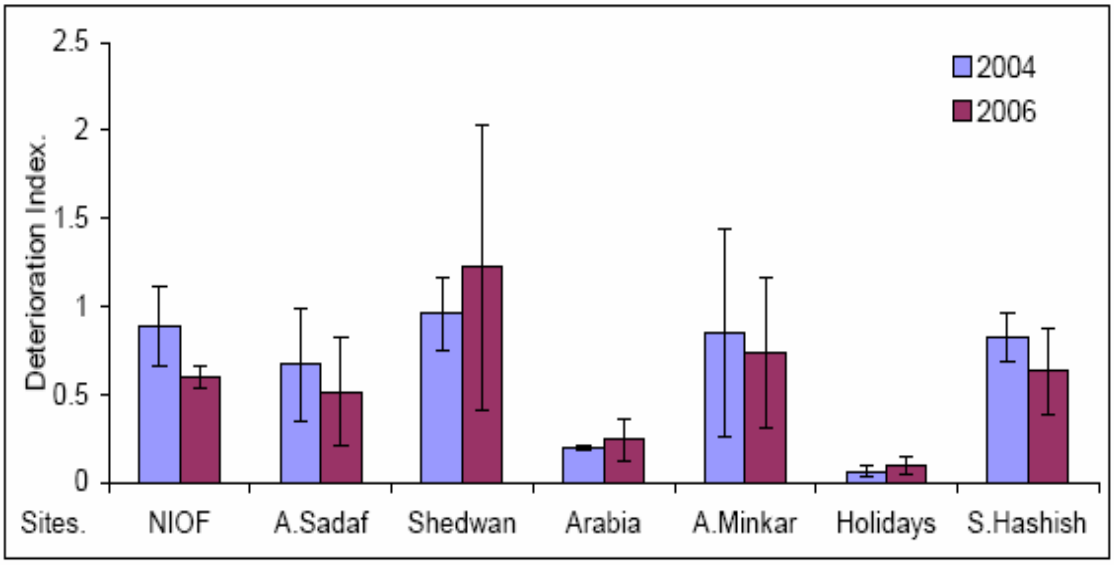


Figure 4.23: The mean (n=4) Deterioration Index (DI) and standard deviation of mean for each of the sites, Hurghada, Egypt.

Two-way ANOVA (Table 4.31) showed that there was a significant difference (F= 8.62, df=6, P=0.000) between sites but not between years in their deterioration indices. A Post

Hoc Tukey test showed that sites 1 and 3 were significantly different from sites 4 and 6; site 2 was not significantly different from any other sites; site 4 was significantly different from sites 1, 3 and 5; site 5 was significantly different from sites 4 and 6; site 6 was significantly different from sites 1, 3, 5 and 7; and site 7 was significantly different from sites 4 and 6 (Table 4.32). However, by comparing 2004 and 2006 data, it appears that sites 1, 2, 5, and 7 were showing slight improvement whilst sites 3, 4 and 6 have shown further deterioration.

Pearson correlation tests (Table 4.33) showed significant negative correlations between 2004 DI and both non-carbonate sediment (R=-.757(*), P=0.024) and mud (R=-0.755, P=0.025) (Figure 4.24). 2006 DI did not show any significant correlation with sedimentation rate, SPM, turbidity, non-carbonate, gravels, sand or mud (Table 4.34).

Table 4.31: Two-way ANOVA results for DI between study sites and between the two years, 2004 and 2006, Hurghada, Egypt.

Source of variation	S.S.	D.f.	M. S.	F	P-value
Year	.047	1	.047	.414	.523
Sites	5.839	6	.973	8.624	.000
Year * Sites	.410	6	.068	.605	.724
Error	4.739	42	.113		
Total	31.705	56			

Table 4.32: Significance matrix of Deterioration Index (DI) between sites, recorded in 2004 and 2006; as indicated by Tukey test, for the 7 study sites, Hurghada, Egypt.

Sites	1	2	3	4	5	6	7
1							
2							
3							
4	x		x				
5				x			
6	x		x		x		
7				x		x	

Table 4.33: Correlations between 2004 Deterioration Index DI and sedimentation rate, suspended particulate matter SPM, non-carbonate sediment, turbidity, gravel, sand and mud for the seven study sites, Hurghada, Red Sea.

		Sediment	SPM	Non-Carbonate	Turbidity	Gravel	Sand	Mud
DI 2004	Pearson Correlation	-.062	-.001	-.757	-.559	.299	.371	-.755
	Sig. (1-tailed)	.448(ns)	.499(ns)	.024(*)	.096(ns)	.257(ns)	.206(ns)	.025(*)
	N	7	7	7	7	7	7	7

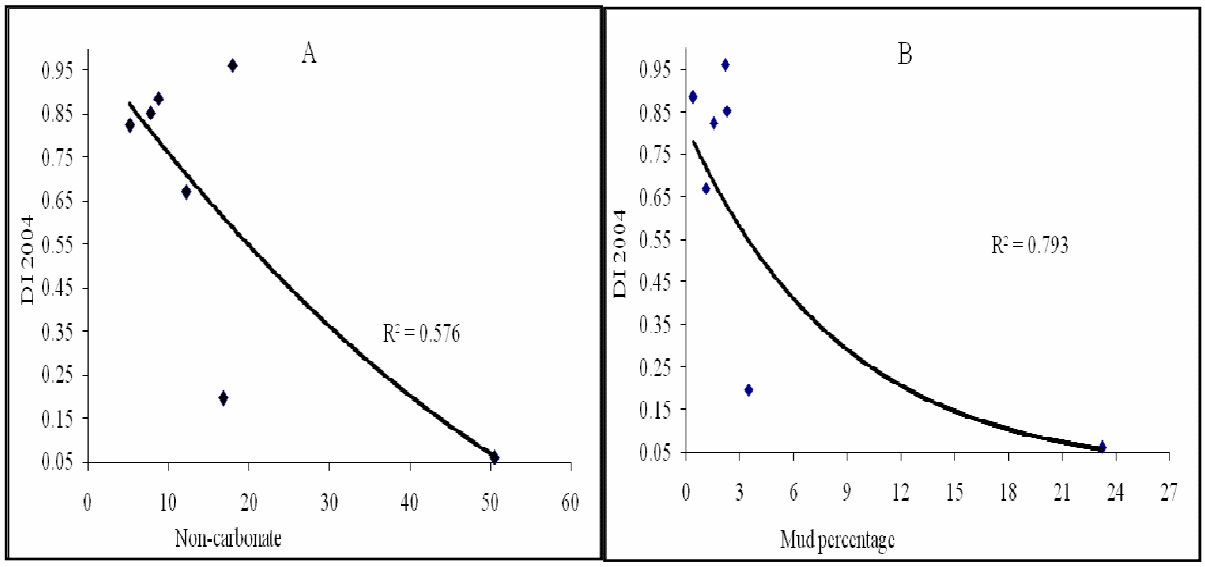


Figure 4.24: The correlation between 2004 coral Deterioration Index (DI); and non-carbonate (A) and mud (B) for each site, Hurghada, Egypt.

Table 4.34: Correlations of 2006 Deterioration Index (DI) with sedimentation rate, SPM, non-carbonate sediment, turbidity, gravel, sand and mud for the seven study sites, Hurghada, Egypt.

		Sediment	SPM	Non-carbonate	Turbidity	Gravel	Sand	Mud
DI 2006	Pearson Correlation	.284	.503	-.457	-.196	.430	.077	-.591
	Sig. (1-tailed)	.269(ns)	.125(ns)	.151(ns)	.337(ns)	.168(ns)	.435(ns)	.081(ns)
	N	7	7	7	7	7	7	7

4.7 Reef Stability against Disturbance

Site 1 had the largest mean diameter of massive coral colonies and site 7 had the mean largest diameter of branched coral colonies as shown in figure (4.25). Site 6 appeared the worst of the sites with the mean smallest diameter of branched and massive coral colonies. There appeared to be a very high frequency of death of the early coral recruits at site 6 which may explain the low coral cover, although it had a high recruitment rate.

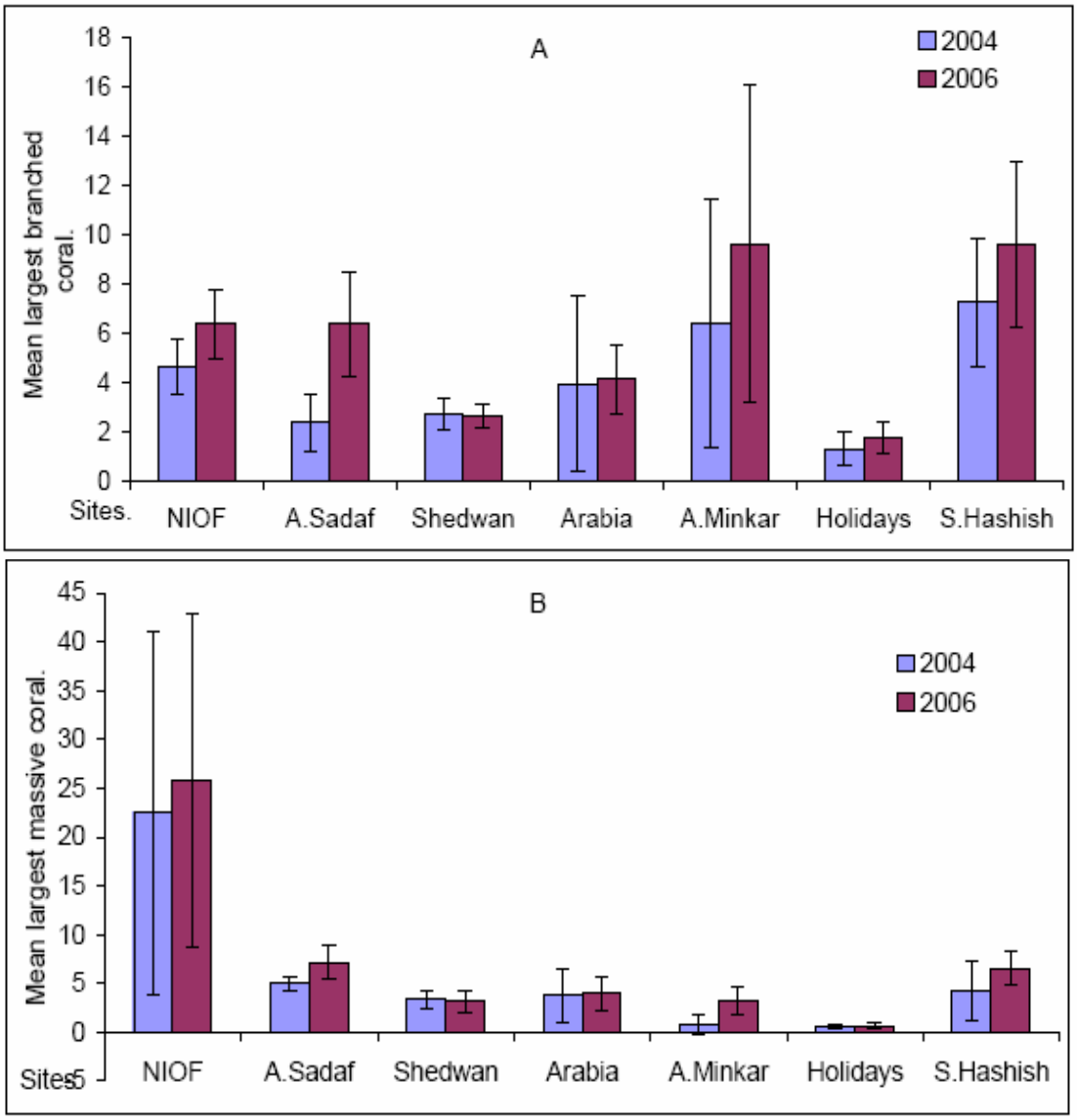


Figure 4.25: The means (n=4) largest colony size for branched (A) and massive corals (B) ± standard deviation of mean, in each of the study sites, Hurghada, Egypt.

Three-way ANOVA was carried out for coral colony size with year, site and growth form (Table 4.35) and the output indicates a high significant difference between sites (F=11.19, df=6, P=0.000) but not between years (F=2.48, df=1 P=0.119) or growth forms (F=2.38, df=1, P=0.126). The results of the ANOVA test indicate that the study sites are subjected to various degrees of disturbance, which has almost equal impact on both branched and massive corals and does not vary from one year to another. A Post Hoc Tukey test showed that sites 1 was significantly different from all other sites; sites 2, 3, 4 and 5 were significantly different from site 1 only; site 6 was significantly different from sites 1 and 7; and site 7 was significantly different from sites 1 and 6 (Table 4. 36).

Table 4.35: Three-way ANOVA results for mean colony size of branched and massive coral forms, years and between sites, for all the study sites, Hurghada, Egypt.

Source of variation	S.S.	D.f.	M. S.	F	P-value
Years	69.143	1	69.143	2.484	.119
Sites	1869.986	6	311.664	11.198	.000
Growth forms	66.343	1	66.343	2.384	.126
Years * Sites	48.350	6	8.058	.290	.940
Years * Growth forms	.413	1	.413	.015	.903
Sites * Growth forms	1518.559	6	253.093	9.094	.000
Years * Sites * Growth forms	6.112	6	1.019	.037	1.000
Error	2337.885	84	27.832		
Total	9567.080	112			

Table 4.36: Matrix of significant differences in Disturbance between sites, recorded in 2004 and 2006; as indicated by Tukey test, for the 7 study sites, Hurghada, Egypt.

Sites	1	2	3	4	5	6	7
1							
2	x						
3	x						
4	x						
5	x						
6	x						
7	x					x	

Correlation tests were carried out to investigate the degree of correlation between coral disturbance and sedimentation rate, SPM, non-carbonate sediment, turbidity, gravel sand and mud (Table 4.37). There was a significant negative correlation between the 2004 readings of branched corals disturbance and sedimentation rate (R=-0.708, P=0.037), non-carbonate sediment (R=-0.741, P=0.028), turbidity (R=-0.845, P=0.008) and a significant positive correlation with sand (R=0.879, P=0.005). Whilst massive corals disturbance did not show any significant correlations. Figure (4.26) showed the correlations between 2004 disturbance and sedimentation, turbidity, non-carbonate and sand.

Table 4.37: Correlations between the 2004 means of the largest colony size of both branched and massive corals; with sedimentation rate, SPM, non-carbonate sediment, turbidity and percentage of gravel, sand and mud, for the seven study sites, Hurghada, Egypt.

2004		Sediment -ation	SPM	Non-Carbonate	Turbidity	Gravel	Sand	Mud
Branched Disturbance	Pearson Correlation	-.708	-.574	-.741	-.845	-.327	.879	-.563
	Sig. (1-tailed)	.037(*)	.089(ns)	.028(*)	.008(**)	.237(ns)	.005(**)	.094(ns)
	N	7	7	7	7	7	7	7
Massive Disturbance	Pearson Correlation	-.192	-.188	-.345	-.193	.069	.271	-.375
	Sig. (1-tailed)	.340(ns)	.343(ns)	.224(ns)	.340(ns)	.441(ns)	.278(ns)	.204(ns)
	N	7	7	7	7	7	7	7

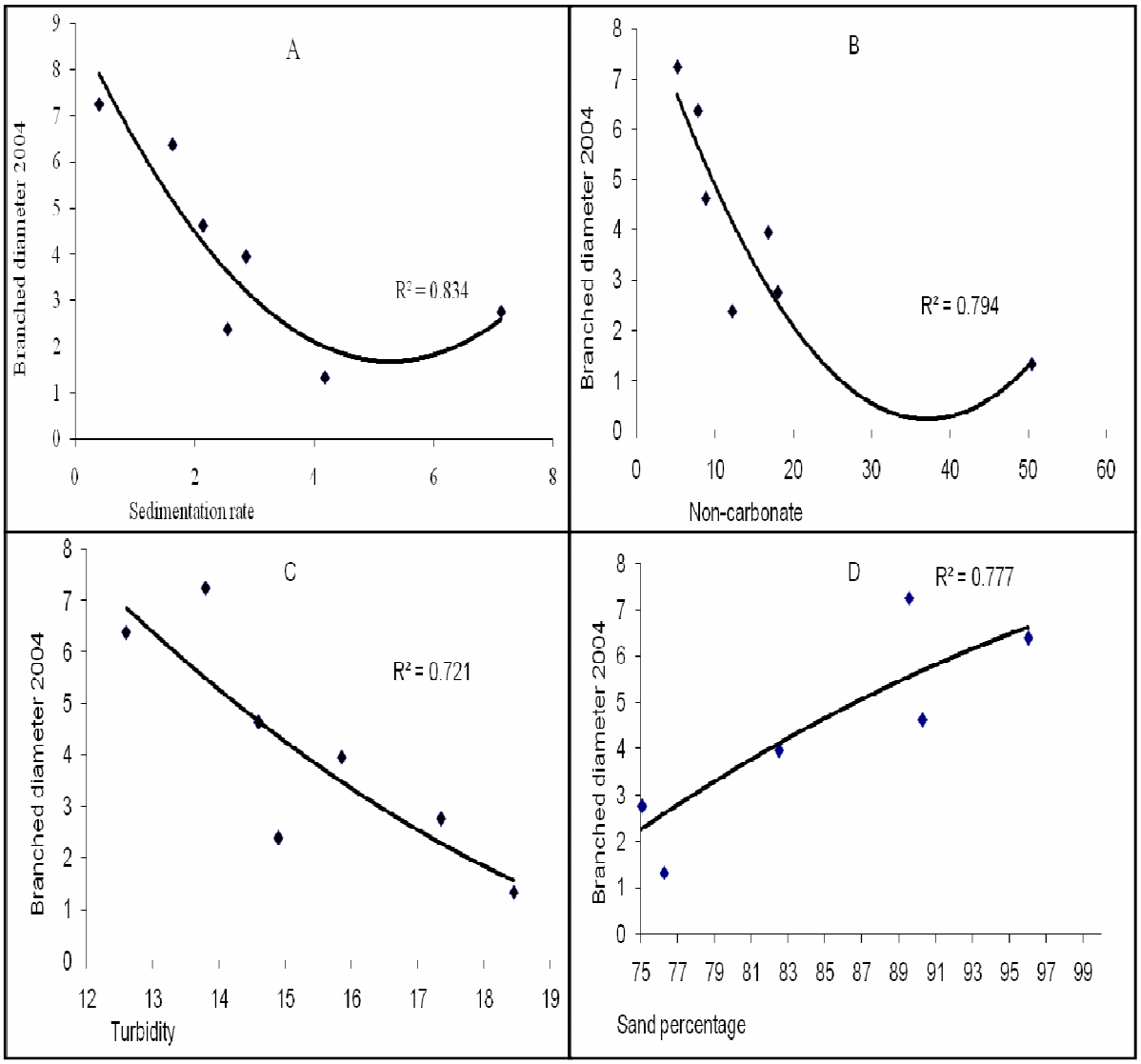
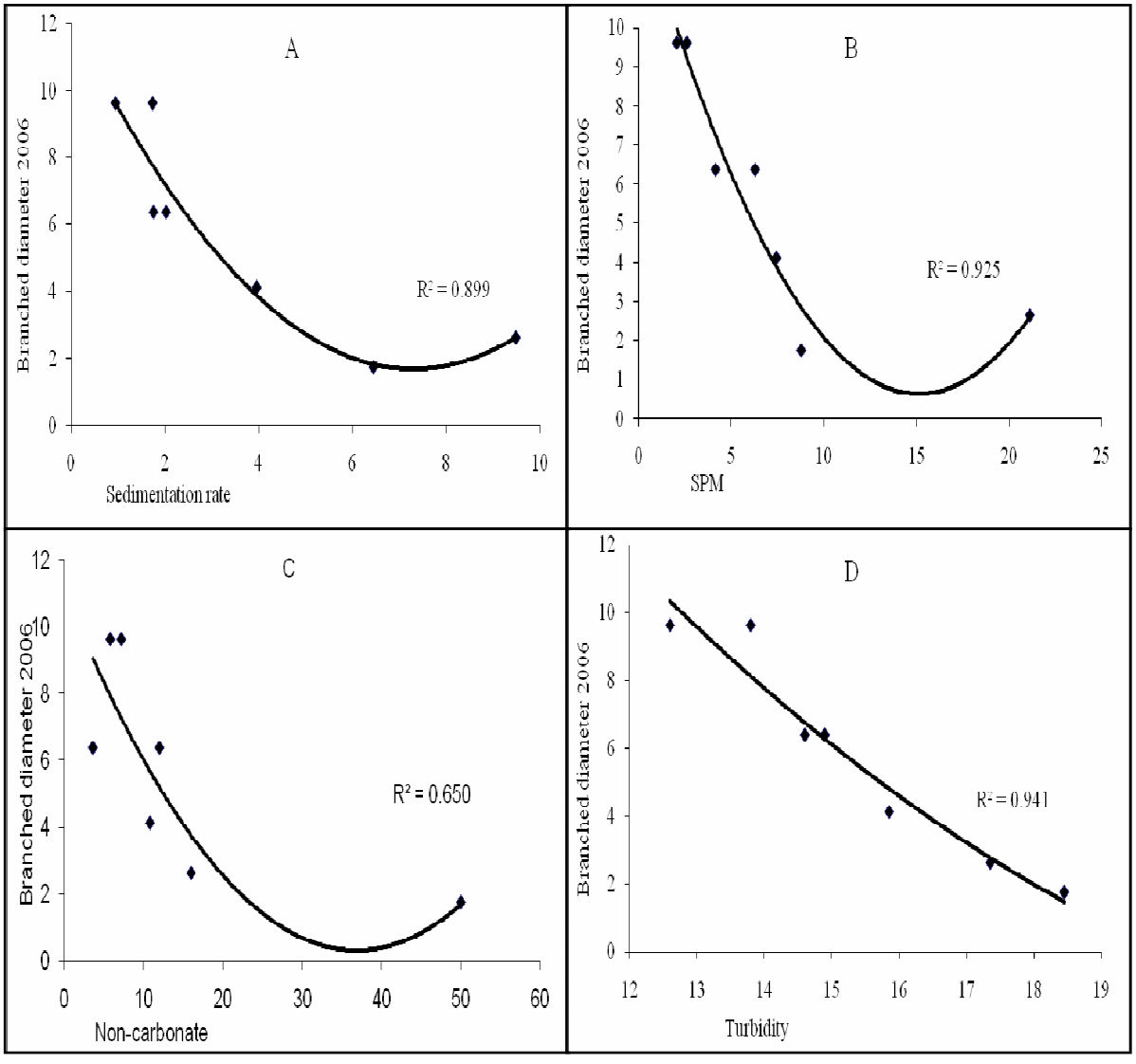


Figure 4.26: The correlations between 2004 means of the largest colony size of branched corals and sedimentation rate (A), non-carbonate (B), turbidity (C) and sand (D), in each of the study sites, Hurghada, Egypt.

Pearson correlations of 2006 mean colony size (Table 4.38) showed significant negative correlations between branched corals disturbance and sedimentation rate (R=-0.846, P=0.008), SPM (R=-0.739, P=0.029), non-carbonate (R=-0.702, P=0.039), turbidity (R=-0.967, P=0.000) and significant positive correlation with sand (R=-0.764, P=0.023). Massive coral mean largest colony size did not indicate any significant correlation with any of these parameters. Figure (4.27) illustrates the correlation between 2006 coral disturbance and sediment, SPM, non-carbonate, turbidity and sand.

Table 4.38: Correlations between the means of the largest colony size of both branched and massive corals of 2006; with sedimentation rate, SPM, non-carbonate sediment, turbidity and percentage of gravel, sand and mud; for the seven study sites, Hurghada, Egypt.

2006		Sediment-ation	SPM	Non-Carbonate	Turbidity	Gravel	Sand	Mud
Branched disturbance	Pearson Correlation	-.846	-.739	-.702	-.967	-.193	.764	-.592
	Sig. (1-tailed)	.008(**)	.029(*)	.039(*)	.000(***)	.335(ns)	.023(*)	.081(ns)
	N	7	7	7	7	7	7	7
Massive disturbance	Pearson Correlation	-.430	-.311	-.286	-.310	.035	.353	-.422
	Sig. (1-tailed)	.168(ns)	.249(ns)	.267(ns)	.249(ns)	.470(ns)	.219(ns)	.173(ns)
	N	7	7	7	7	7	7	7



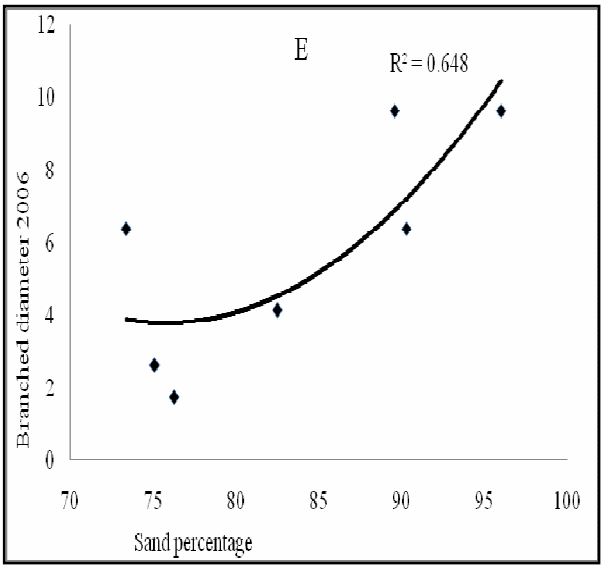


Figure 4.27: The correlations between 2006 means of the largest colony size of branched corals and all of sedimentation rate (A), SPM (B), non-carbonate (C), turbidity (D) and sand (E), for 2006 survey in each of the study sites, Hurghada, Egypt.

4.8 Transplanted Coral Survival

The transplantation study was carried out in six of the study sites namely NIOF (1), A.Sadaf (2), Shedwan (3), Arabia (4), Holidays (5) and S.Hashish (6). Survival trends for the three species *Acropora arabesis, Acropora selago* and *Acropora tenuis* after one month are shown in figure (4.28). Site 4 showed the highest mean survival rate of the three species (90%±22%) and site 6 was the lowest (50%±32%). The experiment indicated that *Acropora selago* had the highest survival rate (63.16%±40.66%) and *Acropora tenuis* the lowest (58.66±36.5%). More frequent mortality was observed among transplants of *Acropora tenuis* than the other two species. Generally there was better survival in the low sediment level sites than the high sediment sites.

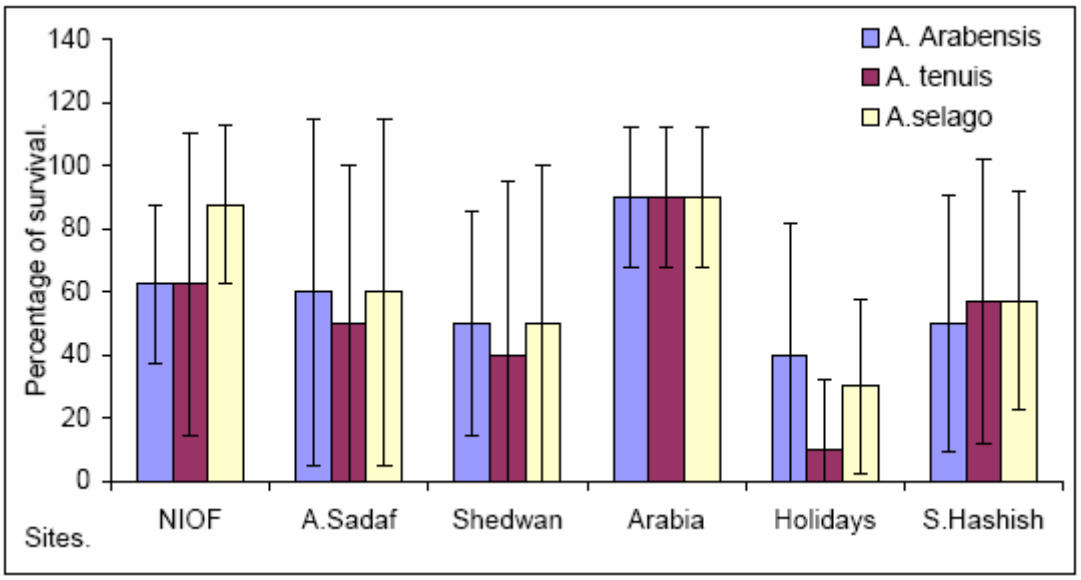


Figure 4.28: The means survival percentage of the three species transplanted; *Acropora arabesis*, *Acropora selago*, and *Acropora tenuis, after* one month of transplantation, at each of the six study sites, Hurghada, Egypt.

Two way ANOVA for coral transplant survival against sites and species, Table (4:39) showed significant differences between sites (F=4.42, df=5, P=0.001) but not between species (F=0.735, df=2, P=0.483). The interaction between sites and species had no significant effect on survival process (F=0.180, df=10, P=0.997). A Post Hoc Scheffe test showed that sites 1, 2, 3 and 6 were not significantly different from any other site. The only significant different was between site 4 and site 5 (Table 4. 40).

Spearman's rho correlation tests (Table 4:41) were carried out to investigate correlations between survival of the three species *Acropora arabesis, Acropora selago* and *Acropora tenuis* with sedimentation rate, SPM, non-carbonate sediment percentage, turbidity, and percentage of gravel, sand and mud. Results showed no significant correlation with any of these factors.

Table 4.39: Two-way ANOVA for transplanted coral survival, between the 6 sites and the 3 species used in the transplantation study after one month, Hurghada, Egypt.

Source of variation	S.S.	D.f.	M. S.	F	P-value
Site	3.446	5	.689	4.426	.001
Species	.229	2	.114	.735	.483
Site * Species	.281	10	2.808E-02	.180	.997
Error	11.677	75	.156		
Total	45.250	93			

Table 4.40: Matrix of significant differences in transplants survival between sites, recorded in 2004 and 2006; as indicated by Scheffe test, for the 7 study sites, Hurghada, Egypt.

Sites	1	2	3	4	5	6	7
1							
2							
3							
4							
5				x			
6							
7							

Table 4.41: The Spearman's correlation between mean survival of the three species tested; *Acropora arabensis, Acropora tenuis* and *Acropora selago,* and sedimentation rate, SPM, non-carbonate, turbidity, gravel, sand and mud; in sites 1, 2, 3, 4, 6 and 7, Hurghada, Egypt.

Spearman's		Sediment -ation	SPM	Non carbonate	Turbidity	Gravel	Sand	Mud
Acropora arabensis	Correlation Coefficient	.058	.261	.058	.058	.377	-.116	-.232
	Sig. (1-tailed)	.457(ns)	.309(ns)	.457(ns)	.457(ns)	.231(ns)	.413(ns)	.329(ns)
	N	6	6	6	6	6	6	6
Acropora tenuis	Correlation Coefficient	.143	.257	.257	.257	-.029	-.029	.086
	Sig. (1-tailed)	.394(ns)	.311(ns)	.311(ns)	.311(ns)	.479(ns)	.479(ns)	.436(ns)
	N	6	6	6	6	6	6	6
Acropora selago	Correlation Coefficient	-.029	-.200	-.086	.086	.257	-.086	-.143
	Sig. (1-tailed)	.479(ns)	.352(ns)	.436(ns)	.436(ns)	.311(ns)	.436(ns)	.394(ns)
	N	6	6	6	6	6	6	6

4.9 Settlement and Recruitment

Examination of larval settlement onto tiles showed a general poor coral recruitment and settlement in the whole study area after three years of deployment. There was a general increase in recruit numbers at all sites; number of recruits varied from zero at site 3 to four recruits at site 6 after the first year. After the third year of deployment, the number of recruits varied from zero at site 3 to 15 individual coral recruits at site 6 (Figure 4.27). The 2006 recruits were identified as less than one year old, which means that none of the 2004 recruits survived to 2006 at any site.

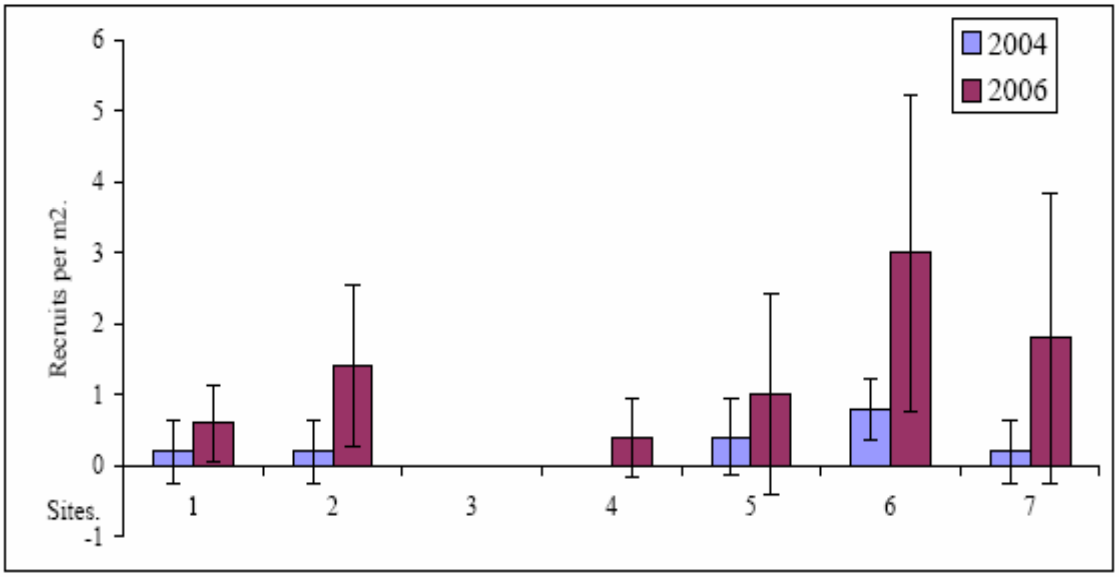


Figure 4.29: The mean number of recruits per m^2, found after one and three years of settlement tiles deployment, at the 7 study sites, Hurghada, Egypt.

Discussion

Coral percentage cover, number of dead and new recruits, abundance, species richness, diversity index, deterioration index, disturbance, transplanted coral survival and recruitment rates varied significantly between study sites. The highest coral cover for massive corals (36.12-38.87%) was at site 1 and the lowest (0.75-2.65%) was at site 5 while the highest branching coral cover (28.32-32.32%) was at site 5 and the lowest (4.25-2.67%) was at site 6. The mean hard coral cover at some of the study sites (1 and 7) exceeded the global mean (32%) described by Reef Check (Hodgson and Liebeler, 2002). The Reef Check report maintained that the percent hard coral cover was significantly higher on reefs having no anthropogenic impacts than on reefs with high levels of such impacts.

Coral cover of some of the current study sites (e.g. site 1) was higher than the mean cover of Careless reef (the highest coral cover in EEPP study, 49%) as indicated from coral reef monitoring in the Red Sea. In addition, species richness ranged from 1.1 to 6.25 species.m^{-2} and diversity index from 1.58 to 3.24 in the current study. Thus there was a wider range of species richness variation between study sites and lower diversity index range than that of the EEPP survey (Kotb et al, 2003). In a study of the use of coral as indicators of reef health at fifteen selected sites around Hurghada Red Sea, Evans (2006) found that generic richness varied from11.0 to 5.8. Results of coral reef monitoring in the Red Sea carried by EEPP (Kotb et al, 2003) indicated a coral cover ranging from 13% to 49%, species richness from 1.73 to 3.24species.m^{-2} and a diversity index from 3.7 to 2.2 at Hurghada diving sites.

Many studies have referred to sedimentation stress on coral parameters such as coral cover and diversity. There was no significant difference in coral cover between the two years. It did however, have a significant negative correlation with sedimentation rate, SPM, non-carbonate sediment and turbidity. These results agreed with what Sheppard (1980) found in a study of the dredging and blasting in Diego Carcia Lagoon (Indian Ocean) that showed a variable and low coral cover but no reduction in coral diversity. Also Hodgson & Dixon (1988) found that high rates of sediment deposition significantly reduced coral cover and diversity. Dredging caused heavy sedimentation on an intertidal reef in Thailand and decreased both living coral cover and species diversity (reviewed in Rogers, 1990). Cortes & Risk (1984) found a correlation of heavy river discharge with lower diversity. Live coral cover and species diversity was also reportedly much lower on sites where sedimentation rates were higher (Hubbard, 1986). Highest coral diversity is often associated with fore reef slopes (Porter 1972, Sheppard 1982), presumably at least partly because sediments do not accumulate as easily in these areas (Rogers, 1990). Reefs subject to stresses such as sewage, agricultural runoff, sedimentation and industrial effluent had approximately equivalent reductions in coral diversity (Edinger, 1988).

Number of live colonies had a significant positive correlation with percentage non-carbonate and mud content while dead coral colonies and number of recruits had significant negative correlations with percentage gravel. The number of recruits had significant negative correlations with sedimentation rate. Dead coral colonies ranged from 1.4 to 0.6 colonies.m^{-2} and recruits ranged from 12.9 to 0.7colonies.m^{-2}. Number of dead colonies did not show any significant correlation with sedimentation rate, SPM or turbidity. This result is supported by the outcome of Winkler et al. (2004) who maintained that coral death through disease and bleaching may be exacerbated by high sedimentation rates but may also be influenced by other external stressors. Sedimentation rate may not be the only factor accountable for the number of dead corals or recruits recorded in the current study. Especially in light of the fact that most sedimentation rates observed were

well below the estimated survival threshold rate of 10mg.cm^{-2}.day^{-1} (Dikou & Van Woesik, 2006; McClanahan & Obura, 1997; Rogers, 1990).

Abundance of coral colonies showed no significant differences between the two years; a significant positive correlation with percentage of mud and the percentage of non-carbonate sediment were indicated. In general, coral abundance as a whole did not exhibit significant relationships with other environmental parameters in this study. It could be suggested that the higher rate of sedimentation may have lead to the observed low coral cover on reefs within the coastal region through inhibiting coral growth.

The general low recruitment rates in the present study may be attributed not only to inhibition of settlement by sedimentation but other factors such as high juvenile mortality. In comparison to the earlier study by Rogers (1990) which referred to sedimentation as one factor responsible for reduced coral recruitment, current results did not detect sedimentation impact on the recruitment process. Sedimentation mortality thresholds for coral recruits were greatly lower than those for larger colonies, tens rather than hundreds of mg.cm^{-2} (Fabricius et al., 2003), which could explain the early death of coral recruits. It is implied that low recruits in this study resulted from the effects of both turbidity shading and sedimentation. It, therefore, seems that recruits can successfully settle onto the new artificial reefs despite the high sedimentation rate.

Species richness showed no significant differences between the two years and a significant negative correlation with sedimentation rate and SPM. These correlations may support the implications that species richness is higher on reefs where sedimentation rate is low. Diversity indices in the current study showed significant differences between sites, but not between years. There was also significant negative correlation between diversity index and sedimentation rate, SPM, non-carbonate and turbidity. The outcome of this study agreed with early studies which revealed that species diversity decreased as a result of high sedimentation rate (Loya and Slobodkin, 1971). Nugues and Roberts (2003b) maintained that sedimentation could cause reef degradation in two ways: by direct burial and smothering of corals, and by suppressing the regeneration of the adult colonies together with the settlement of new recruits through increased competition with algae. The differences in diversity between sites in the current study suggest that the growth of certain coral genera is inhibited by the high rate of sedimentation. It is possible that there is a different threshold level of sedimentation that can be endured by each genera (Sanders and Baron-Szabo, 2005); therefore, the higher the rate of sedimentation, the lower the number of species that are able to grow and survive, and the lower the species diversity.

There were no significant differences between sites in the percentage cover by pioneer coral species *r-strategists* and no significant correlations between *r-strategist*s and sedimentation rate, SPM, non-carbonate, turbidity, gravel, sand or mud. Thus there was no effect of sedimentation on *r-strategists* percentage cover and they occupy high sedimentation sites as well as low sedimentation ones. The *r-strategists* includes all species of the genera; *Acropora, Pavona, Pocillopora, Stylophora, Psammocora, Seriatopora* and *Montipora.*

The Deterioration Index (DI) study indicated significant difference between sites. However, significance differences were also found between sites in abundance, species richness and Diversity Index. The Deterioration Index showed no significant difference between years. However, there was a significant negative correlation between DI and the percentage of non-carbonate and mud. At high mortality and/or low recruitment, DI value will be high; on the other hand, if recruitment exceeds mortality, DI will be small, indicating recovery or improvement of the reef. Deterioration Index (DI) was first used by

Ben-Tzvi et al. (2004) as a reef health measure. Their results proved a significant difference between sites in contrast to other reef health parameters such as mortality rates, abundance, species richness, and species diversity. Their study suggested that DI might be a more sensitive indicator than those other parameters. The results of the current contrasted markedly with the conclusions of Ben-Tzvi et al. (2004). At site 6 the live coral cover was low whilst it had the highest recruitment rates (12.9 - 13.3 colonies per m^2) in both 2004 and 2006 surveys. As noted previously, if the number of recruits is greater than the number of dead colonies this may indicate that the reef is improving. However if the recruits do not survive for a long time and die under sediment or some other stress, this may give a false impression of the reef's health status.

In the current study, site 3 was ranked the highest for DI, while site 6 was the lowest. Whilst site 3 had the highest Deterioration Index, which concurs with highest sediment, and SPM at this site, the interpretation of the results infers that site 6 was the best site using this index. This case was the opposite to what might have been expected, but most of the recruits dies at an early stage. Furthermore, whilst site 6 had the lowest DI this deviated from expectation based on the other health indicators measured during this study. Site 6 was poor for coral cover, abundance, species richness and diversity index. As noted, the low DI value at site 6 comes from the high number of recruits to the site compared with the low number of dead colonies. However most of those recruits do not endure for a long time and die at an early stage as indicated from the 2006 survey, which showed low number of large recruited colonies and general low coral cover.

The colony size was used as indicator of disturbance, significant differences between sites but not between years and growth forms (branched or massive) were indicated. The study proved a significant negative correlation between branched coral disturbance and sedimentation rate, SPM, non-carbonate and turbidity, whilst a significant positive correlation with sand percentage was found. Massive corals showed significant negative correlations with the percentage of mud only. In this study the size of the largest colonies recorded in each transects represents the magnitude of coral growth occurring over time and the stability of the reef. Coral mortality on reefs is a normal biological process, which could be accelerated by anthropogenic impacts leading to coral death. Consequently the more frequent the disturbance the smaller the coral colony sizes at a given site. Therefore the living surface area of a coral colony or colony size can be used as an integrated measurement for disturbance intensity and frequency (Connell, 1978; Done, 1992). The results of the current study indicated that the smallest colony size was found at sites 3 and 6 which undergo high disturbance. This outcome agreed with the results of a study by Done & Potts (1992) which showed that the smaller the mean colony size the higher the disturbance level, as the colonies are always destroyed at early stage of growth.

The assessment of the level of disturbance indicated that the most frequent disturbance occurred at site 6, where mean largest colony diameter of branched and massive corals was the smallest and site 3 was the next most disturbed site. Whilst site 1 and 7 had the lowest level of disturbance based on the largest massive and branched coral colonies diameters. The deviation of massive corals from the anticipated general strong negative correlation with sedimentation rate is thought to be linked to the major physical disturbance (breakage), rather than steady sediment input. Branched coral diameter was inversely related to sedimentation rate, suggesting that higher rates of sedimentation inhibit colony growth in branched corals.

Transplanted coral survival of the three species used in this study ranged from 90% at site 4 to 30% at site 6. Earlier transplantation studies in Hurghada carried out by Ammar et al. (2000) indicated a mean coral survival of 70% at low sedimentation sites. In the current

study, the highest survival rate was shown by *Acropora selago* and the lowest was by *Acropora tenuis*, although no significant difference between the three species (*Acropora Arabensis, Acropora selago* and *Acropora tenuis*) was found. Ammar et al. (2000) suggested that high mortality of transplanted corals at the high sedimentation sites highlights the lethal effect of sediment and necessity for clear water for successful reef restoration. A significant difference in transplant survival between sites was found in the current study. Sites with highest sedimentation (3 and 6) showed the lowest survival rate. However, the sites with lowest sedimentation (7) did not have the highest survival, indicating a non-linear relationship between survival and sedimentation. There were no significant correlations with sedimentation rate, SPM, non-carbonate sediment, turbidity or percentage of gravel, sand or mud. At low sedimentation levels, as observed at site 4 there was no evidence of a negative effect of sediment on transplanted coral survival. Consequently it can be suggested that differences in sedimentation rate is one major factor affecting coral survival processes at higher sedimentation rates.

Results from the settlement study indicated a general low larval settlement in all sites after one and three years of tile deployment. Number of recruits varied from zero at sites 2 and 4 to 3 recruits.m^{-2} of substrate at site 7. In a study of recruitment at Ooshima and Satsuki Islands, Japan, Tioho et al, (2001) found that recruitment rates of *Pocillopora damicornis* ranged from 3.8 recruits.m^{-2} to 11.2 recruits.m^{-2}. Whilst other studies recorded lower rates (0.2 .m^{-2}) at Okinawa (Yeemin, 1991), higher rates (11.6 recruits.m^{-2}) were reported from Coconut Island, Hawaii (Fitzhardinge, 1989). The steel mesh rack method used in this study is considered the most common method used for recruitment studies (reviewed in Mundy, 2000). Variation in topographic complexity between tiles and natural reefs which could affect larval supply and settlement process was taken into consideration in choosing tiles size and fixing angle.

In conclusion coral the parameters measured in this part of the study indicated wide range of correlations with the environmental parameters measures at this study. Coral cover, species richness, diversity and disturbance indicated unambiguous decline under the effect of sedimentation rate, SPM, non carbonate and turbidity. While other parameters such as coral abundance, DI, percentage of *r-strategist*s and the number of recruits, dead and live colonies were insignificantly impacted by sedimentation rate, SPM, non carbonate and turbidity. Except for DI all the coral parameters showed the lowest values at sites which had the highest sedimentation rate, SPM, non carbonate and turbidity. Coral cover, species richness, diversity and disturbance had showed as important means of sediment impact recognition.

CHAPTER 5

FISH ABUNDANCE AND BIOEROSION INTENSITY

Introduction

Early studies have referred to fish abundance as an appropriate indicator of reef health. Coral destruction may lead to a marked reduction in fish species richness and abundance due to loss of shelter and living space (Kawasaki et al., 2003). The species of the family Chaetodontidae have previously been put forward as candidates for coral bioindicators, because many are obligate coral feeders (Hourigan et al. 1988). There is an apparent lack of information about coral reef fish community structure in the coastal reefs of Hurghada. Fish abundance is used in this study as a sign of reef health status. This part of the study tries to determine fish abundance of the most common fish families in the Red Sea and test the relationship with other reef health parameters and the impact of sedimentation. Underwater visual census was considered the most common technique used to record fish densities on reefs (Brock, 1954; Craik, 1981; McManus et al. 1981; Williams, 1982, Russ 1984), and for population ecology studies for management decisions (Carpenter et al. 1981; Harmelin-Vivien et al., 1985, Gomez et al. 1988). Species surveyed were three coral feeding fish, Butterfly (Chaetodontidae), Angel (Pomacanthidae), and wrasse (Labridae) and three algal feeding fish groups, Damsel (Pomacentridae), Surgeon (Acanthuridae) and Rabbitfishes (Siganidae). A Chaetodontidae family survey was also carried to species level. An earlier study by Gosline (1985) had maintained that chaetodontid fish were known to feed on living corals, and are considered the most specialized family of percoid fishes. The study also aims to test the correlation between fish abundance of chaetodontid fish and reef health parameters such as coral cover, diversity, richness and Deterioration Index.

Decreasing water quality and increasing sediment load may change the structure and composition of the initial macroboring community and lead to an enhanced bioerosion rate. Holmes et al. (2000) maintained that coral rubble bioerosion is sensitive to low levels of eutrophication and sedimentation stress and provides a valuable indicator of eutrophication stress on coral reefs. There were no data about macroborer distribution and abundance in the Red Sea in general and in the study area in particular. A Coral bioerosion study was carried out as a reef health indicator to examine the variation in the bioeroding community in relation to sedimentation rate. Bioerosion of coral rubbles was monitored along line belt transects (LIT) along the reef edge. Macroborer bore number were counted for the main four groups of macroborers; polychaetes, bivalves, sponges and sipunculids.

In an attempt to link the observed decline in reef health to the major causea, a reef quality matrix was applied. Sites were set from the highest to the lowest quality according to the values given for each parameter measured. Quality matrices for the study site in relation to measured environmental and biological parameters was set up and tested for correlations between biological and environmental parameters measured in this study.

5.1 Fish Abundance

The average number of individuals of each fish group (coral and algal feeding) per square meter for the two surveys 2004 and 2006 is shown in Figure (5.1). Average highest abundance of coral feeding fish was found at site 2 and algal feeding fish at site 7. Whilst the lowest abundance of both coral and algal feeding were recorded at site 6.

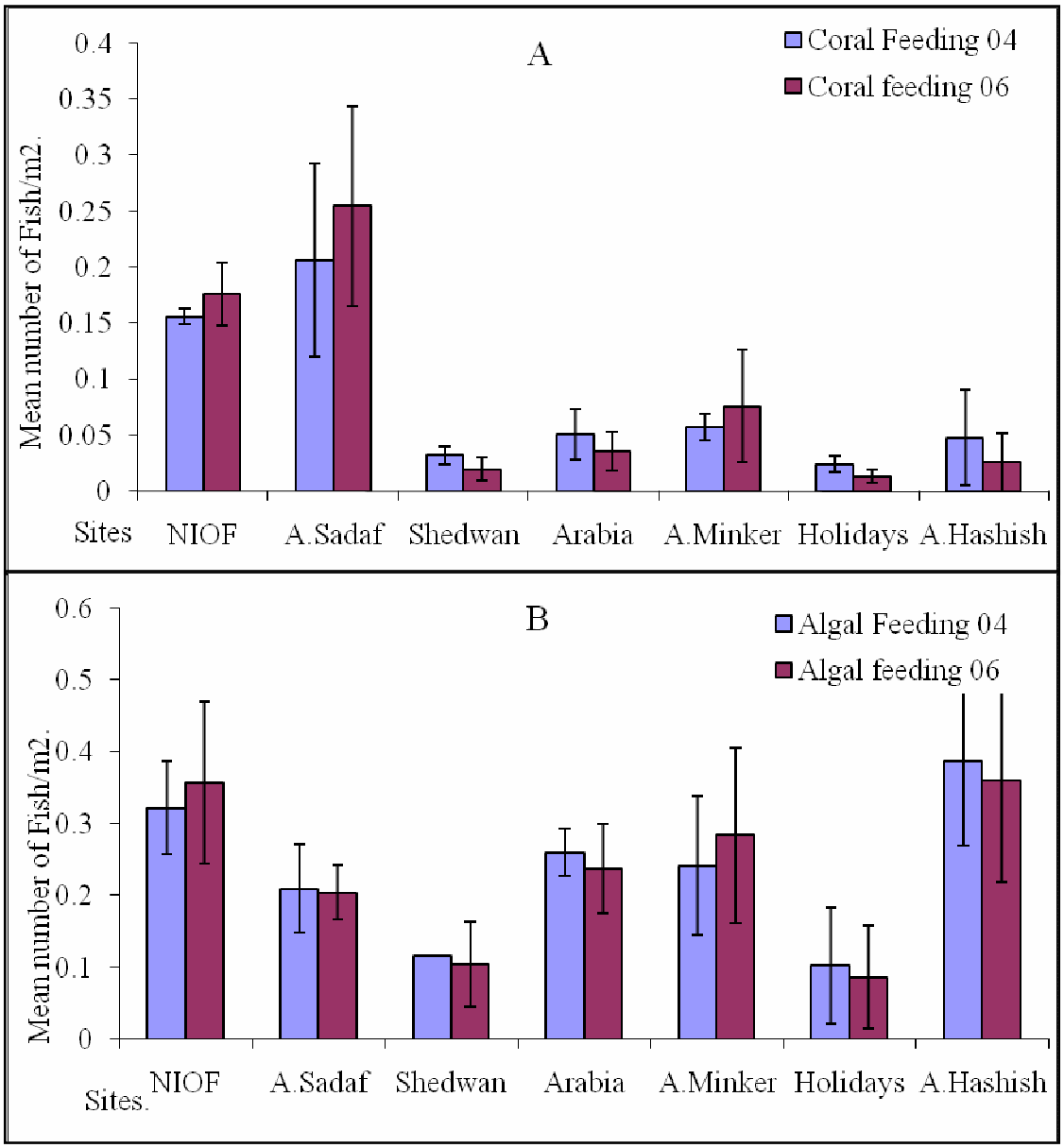


Figure 5.1: The mean abundance for coral feeders (A) and algal feeders (B) fish groups ± standard deviation of mean for the two surveys (2004, 2006) in each of the seven study sites, Hurghada, Egypt.

A three-way ANOVA for fish abundance against year, site and feeding habits (algal feeding and coral feeding) showed a high significant difference between sites, feeding groups but not between the two survey years (Table 5.1). The differences between the means of the two groups; coral feeders and algal feeders and between sites are very high and are unlikely to be the result of chance. A Post Hoc Tukey test showed that site 1 was significantly different from 3, 4, 5 and 6; site 2 was significantly different from 3 and 6;

site 3 was significantly different from 1, 2, 5 and 7; site 4 was significantly different from 1 and 6; site 5 was significantly different from 1, 3 and 6; site 6 was significantly different from 1, 2, 4, 5 and 7 and site 7 was significantly different from 3 and 6 (Table 5. 2).

Table 5.1: The three way ANOVA results for differences between years, sites and feeding group (coral and algal feeding) for 2004 and 2006 fish survey data from the seven study sites, Hurghada, Egypt.

Source of variation	S.S.	D.f.	M. S.	F	P-value
Years	25.190	1	25.190	.092	.763
Sites	25911.071	6	4318.512	15.746	.000
Feeding group	28306.714	1	28306.714	103.211	.000
Years* Sites	506.310	6	84.385	.308	.930
Years* Feeding group	37.333	1	37.333	.136	.714
Sites* Feeding group	13794.452	6	2299.075	8.383	.000
Years * Sites * Feeding group	239.500	6	39.917	.146	.989
Error	15358.667	56	274.262		
Total	219220.000	84			

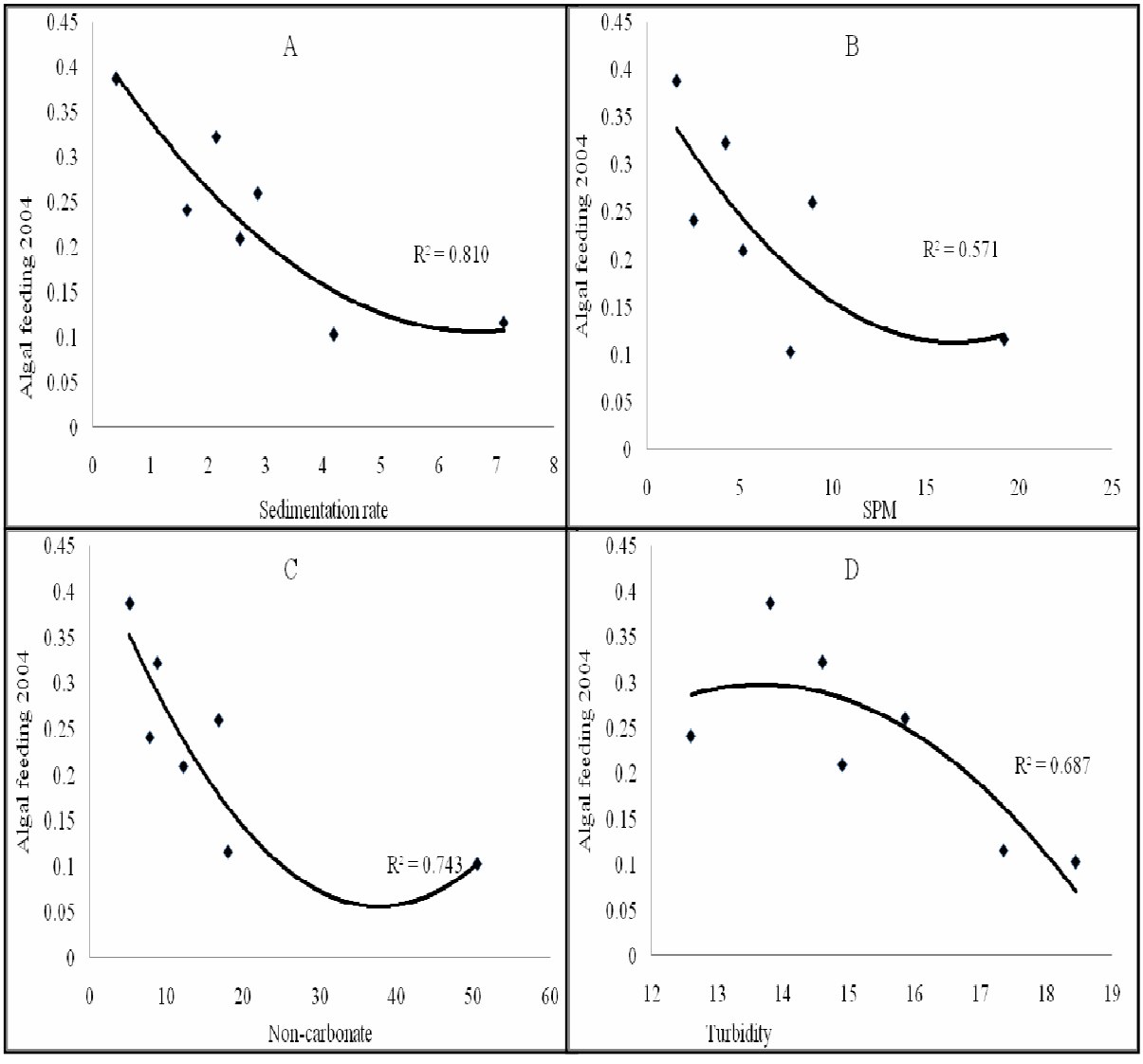
Table 5.2: Matrix of significant differences in fish abundance between sites, recorded in 2004 and 2006; as indicated by Tuky test, for the 7 study sites, Hurghada, Egypt.

sites	1	2	3	4	5	6	7
1							
2							
3	x	x					
4	x						
5	x		x				
6	x	x		x	x		
7			x			x	

Pearson correlation test (Table 5.3) between 2004 fish abundance and sedimentation rate, SPM, non carbonate, turbidity, gravel, sand and mud; showed a significant negative correlation between algal feeder abundance and sedimentation rate (R=-0.837, P=0.009), SPM (R=-0.690, P=0.043), non-carbonate (R=-0.731, P=0.031), turbidity (R=-0.752, P=0.026) and significant positive correlation with sand (R=0.695, P=0.042). The results indicated that sites with high sedimentation and SPM tend to support a low abundance of algal feeding fish (Figure 5.2). Coral feeders did not indicate any significant correlation with any of these parameters.

Table 5.3: The correlation test for 2004 mean number of fish per square meter for the three coral feeding fish families and the three algal feeding fish families; with sedimentation rate, SPM, non carbonate, turbidity, gravel, sand and mud; in the 7 study sites, Hurghada, Egypt.

2004		Sediment	SPM	Non carbonat	Turbidity	Gravel	Sand	Mud
Coral feeder	Pearson Correlatio	-.294	-.338	-.393	-.336	.471	-.108	-.440
	Sig. (1-tailed)	.261(ns)	.229(ns)	.192(ns)	.231(ns)	.143(ns)	.409(ns)	.161(ns)
	N	7	7	7	7	7	7	7
Algal feeder	Pearson Correlatio	-.837	-.690	-.731	-.752	-.138	.695	-.587
	Sig. (1-tailed)	.009(**)	.043(*)	.031(*)	.026(*)	.384(ns)	.042(*)	.083(ns)
	N	7	7	7	7	7	7	7



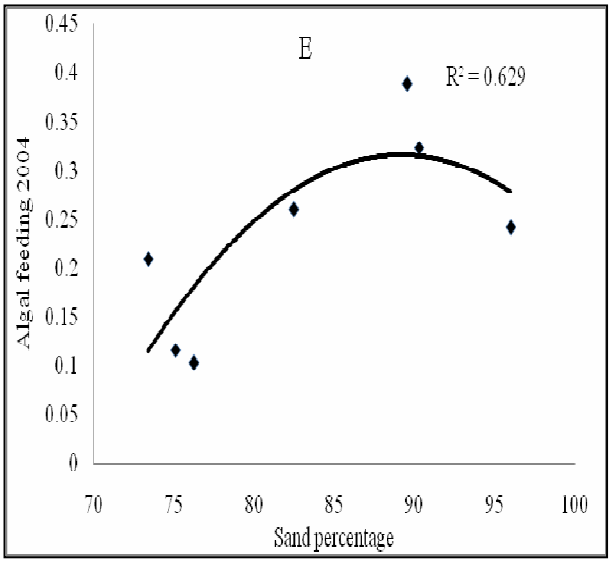


Figure 5.2: The correlation of the 2004 mean number of algal feeder families with sedimentation rate, SPM, non-carbonate, turbidity and percentage of sand at the seven study sites, Hurghada, Egypt.

Correlation between fish abundance from 2006 and sedimentation rate, SPM, non carbonate, turbidity, gravel, sand and mud (Table 5.3) showed significant negative correlation between algal feeding abundance and sedimentation rate (R=-0.862, P=0.006), SPM (R=-0.76, P=0.024) and turbidity (R=-0.84, P=0.009). Algal feeding had a significant positive correlation with sand (R=0.803, P=0.015) (Figure 5.3). Coral feeding from 2006 did not show any significant correlation with any of these parameters.

Table 5.4: Results of correlation between 2006 mean number of fish per square meter for the three coral feeding fish families and the three algal feeding fish families; and sedimentation rate, SPM, non carbonate, turbidity, gravel, sand and mud; in the 7 study sites, Hurghada, Egypt.

2006		Sediment -ation	SPM	Non carbonate	Turbidity	Gravel	Sand	Mud
Coral feeders	Pearson Correlation	-.577	-.380	-.503	-.427	.429	-.041	-.462
	Sig. (1-tailed)	.087(ns)	.200(ns)	.125(ns)	.169(ns)	.169(ns)	.465(ns)	.148(ns)
	N	7	7	7	7	7	7	7
Algal feeders	Pearson Correlation	-.862	-.760	-.641	-.840	-.209	.803	-.621
	Sig. (1-tailed)	.006(**)	.024(*)	.061(ns)	.009(**)	.326(ns)	.015(*)	.068(ns)
	N	7	7	7	7	7	7	7

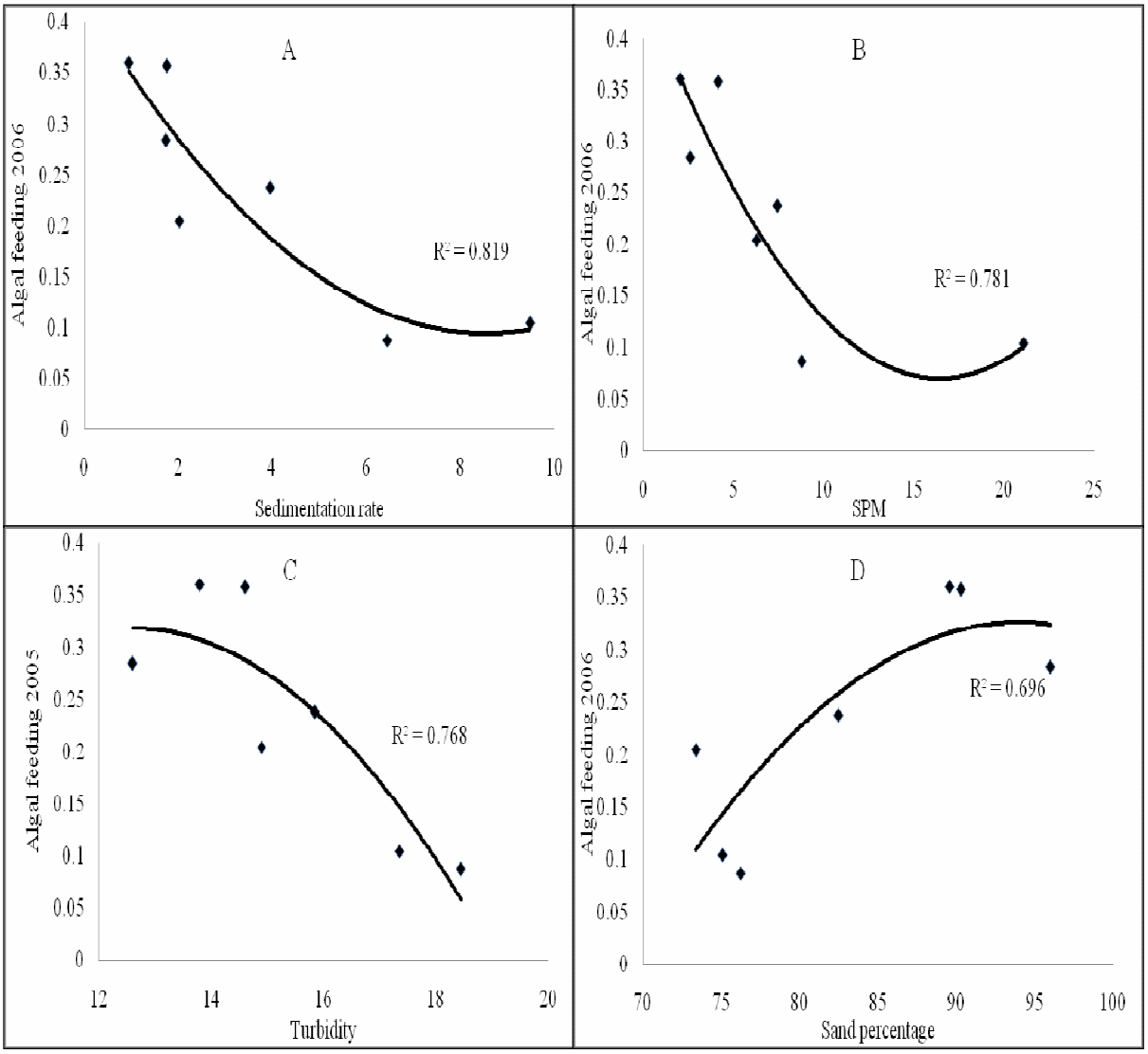


Figure 5.3: The correlation of the mean number of 2006 algal feeding fish families with sedimentation rate (A), SPM (B), Turbidity (C) and sand (D); at the seven study sites, Hurghada, Egypt.

-Cluster analysis

Cluster analysis was done for coral reef fish families surveyed in the two years (2004, 2006) using Multivariate Statistical Package (MVSP), to show the similarities between sites. First year cluster analysis (Figure .5.4) showed the highest similarity between site 6 (Holidays) and 3 (Shedwan) with distance 11.1, and between site 5 (Abu Minkar) and 7 (S.Hashish) with distance 25.5. The lowest similarities recorded between site 2 (A.Sadaf) and site 7 (A.Hashish) with the highest distance 252.2.

Cluster analysis for the second year fish families (Figure.5.5) shows high similarity between site 5 (Holidays) and 3 (Shedwan) with distance 10.14 and between site 4 (Arabia) and site 5 (Abu Minkar) with distance (33). The least similarity shown between site 1 (NIOF) and site 6 (Holidays) with distance 199.

When the two years done in one cluster test the results indicated that the highest similarity between was between site 5 (Holidays 2004) and 3 (Shedwan 2006) with distance 6.55 as

shown in figure (5.6). The least similarity between two sites was shown between site 1 (NIOF) 2006 and A.Sadaf 2004 reading with distance 250.2.

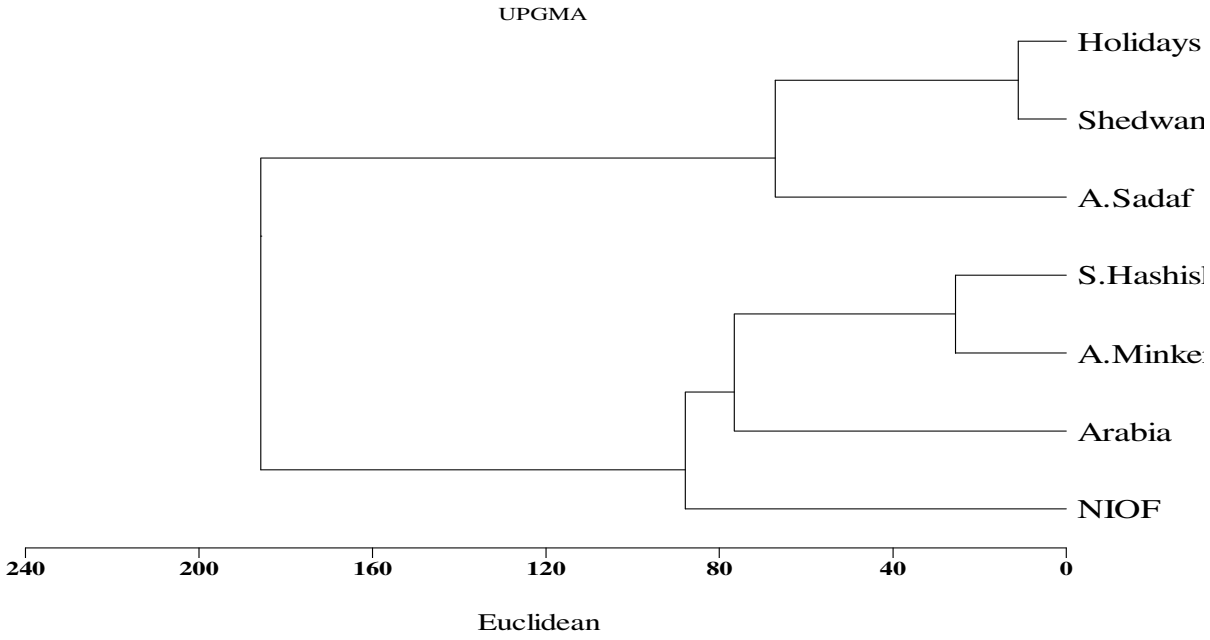


Figure .5.4: The similarity distance between sites in relation to 2004 fish data for Hurghada sites, Red Sea.

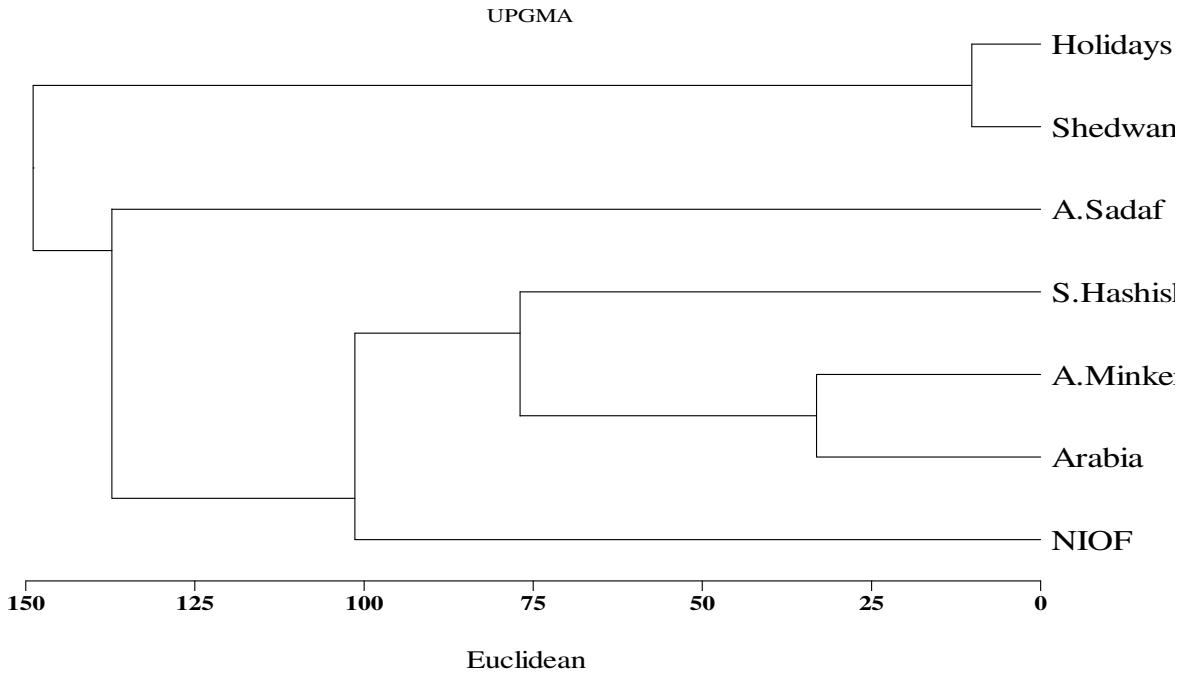


Figure .5.5: The similarity distance between sites in relation to 2006 fish data for Hurghada sites, Red Sea.

5.2 Butterfly fish *(*Chaetodontidae*)* Abundance

There are no important differences between the two years in the number of butterfly fish found in all sites (Figure 5.6). Survey results showed that site 5 had the highest number of butterfly fish (28 fish), site 2 had 26 fish and the lowest were sites 6, which had 2 fish, site 3 with 3 fish, and site 4 with 5 fish (Figure 5.7). The most abundant species was *Chaetodon fasciatus* with 23 fish in total. The highest number of *Chaetodon fasciatus* (9 fish) was found at site 5. *Chaetodon larvatus* was the lowest represented species, had 5 fish found in total. Six individuals of each *Chaetodon lineolatus* and seven of *Chaetodon auriga* were also found.

The species *C. semilarvatus, C. lineolatus* and *C. paucifasciatus* were the most widely represented in most of the sites (6 sites). *Megaprotodon trifascialis, C. larvatus, C. fasciatus* and *C. auriga* were the least distributed between the sites (found in 4 sites from the seven). *C. semilarvatus* had the highest number (6 fish) in site 5 and *C. paucifasciatus* had the highest number (4 fish) at sites 2. The three species found in site 3 were *C. semilarvatus, C. paucifasciatus* and *C. lineolatus;* and in site 6 there were *C. larvatus* and *C. lineolatus* only.

A two-way ANOVA for butterfly fish abundance against year and site was carried out to test variation between sites and years. The results showed high significant difference between sites (F=7, df=6, P=0.000) but not between the two years (F=1.07, df=1, P=0.308) (Table 5.5). A Post Hoc Tukey test showed that site 2 was significantly different from site 6, sites 3 and 4 were significantly different from site 5, and site 5 was significantly different from sites 6 and 7 (Table 5. 6).

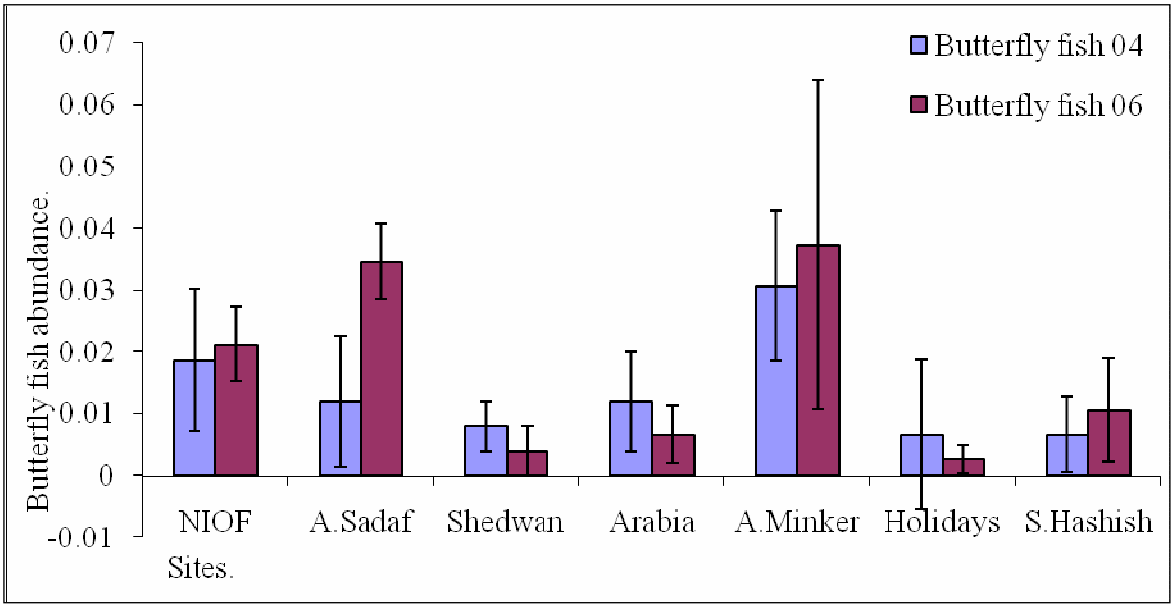


Figure 5.6: The abundance of butterfly fish (chaetodontid) per square meter ± standard deviation of mean for the two surveys (2004, 2006) at each of the seven sites, Hurghada, Egypt.

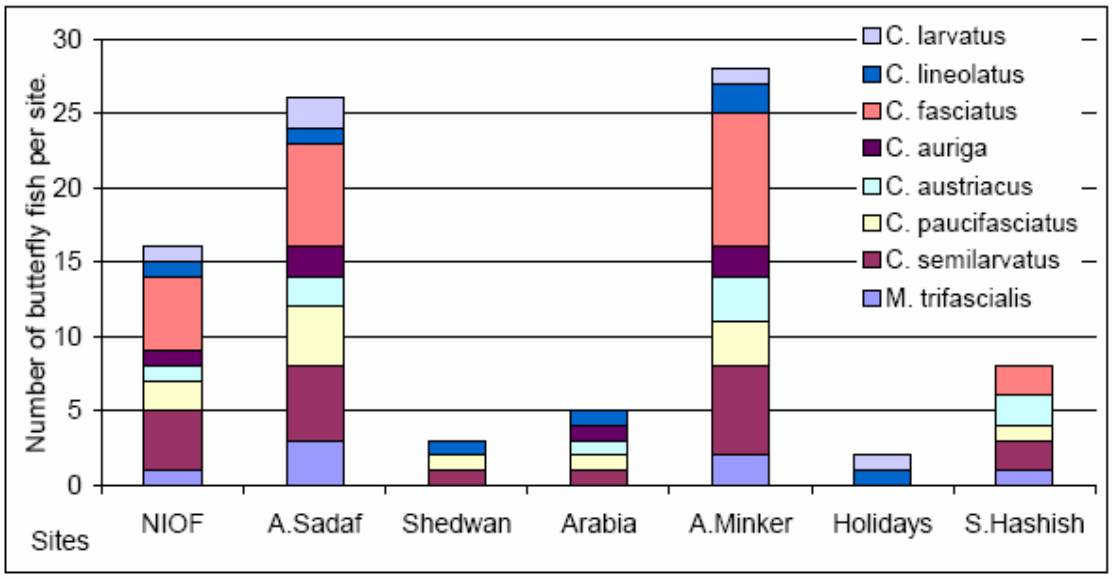


Figure 5.7: The total number of individuals for each chaetodontid fish species found at 2006 at each of the seven study sites, Hurghada, Egypt.

Table 5.5: The two-way ANOVA results for differences between years and sites in butterfly fish abundance for 2004 and 2006 fish survey data from the seven study sites, Hurghada, Egypt.

Source of variation	S.S.	D.f.	M. S.	F	P-value
Years	6.881	1	6.881	1.078	.308
Sites	268.238	6	44.706	7.006	.000
Years* Sites	53.286	6	8.881	1.392	.253
Error	178.667	28	6.381		
Total	1109.000	42			

Table 5.6: Significance matrix for butterfly fish abundance between sites as indicated by Tuky test, for the 7 study sites, Hurghada, Egypt.

Sites	1	2	3	4	5	6	7
1							
2							
3							
4							
5			x	x			
6		x			x		
7					x		

Spearman correlation test between butterfly fish abundance and Coral cover, Species Richness, Diversity Index and Deterioration Index (Table 5.7) indicate a significant positive correlation between 2006 butterfly fish abundance and coral cover (R=0.679, P=0.047).

Table 5.7: The correlation of number of butterfly fish per site with Coral cover, Species Richness, Diversity Index and Deterioration Index; at the 7 study sites, Hurghada, Egypt.

Spearman,s rho		Coral cover	Species Richness	Diversity Index	Deterioration Index
Butterfly fish 2004		.418	.037	.037	.364
	Sig. (1-tailed)	.175(ns)	.438(ns)	.438(ns)	.211(ns)
	N	7	7	7	7
Butterfly fish 2006		.679	.357	.357	.321
	Sig. (1-tailed)	.047(*)	.216(ns)	.216(ns)	.241(ns)
	N	7	7	7	7

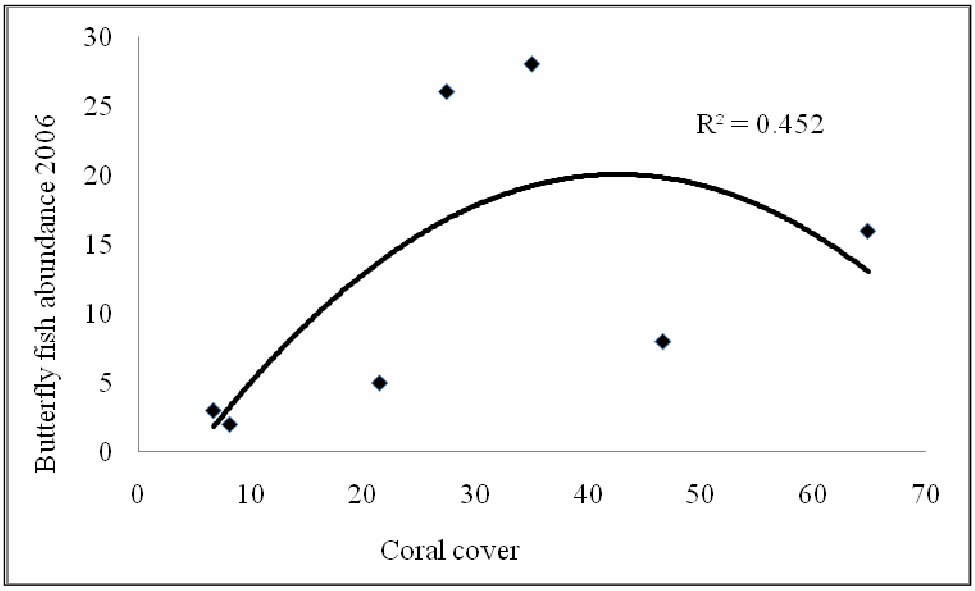


Figure 5.8: The correlation between coral cover and butterfly fish abundance, at the seven study sites, Hurghada, Egypt.

5.3 Bioerosion Intensity

Macroborer bores (Figure 5.9) were counted for the main four groups. Polychaetes had the highest number of bores (1.12±1.18) at site 6 and the lowest (0.2±0.2) at site 2 (Figure 5.10). The highest bivalve mean number of bores per square meter of reef (1.48±1.13) was recorded at site 1 and the lowest (0.24±0.32) at site 3 (Figure 5.11). Site 5 had the highest abundance of sponges (1.4±0.92) and site 3 the lowest (0.48±0.46) (Figure 5.12). Sipunculids showed the highest abundance of all groups of macroborers at all sites with the highest infestation (5.08±4.02) at site 3 and the lowest (1.12±0.8) at site 7 (Figure 5.13). The highest total abundance of all groups was found at site 3 and the lowest at site 7.

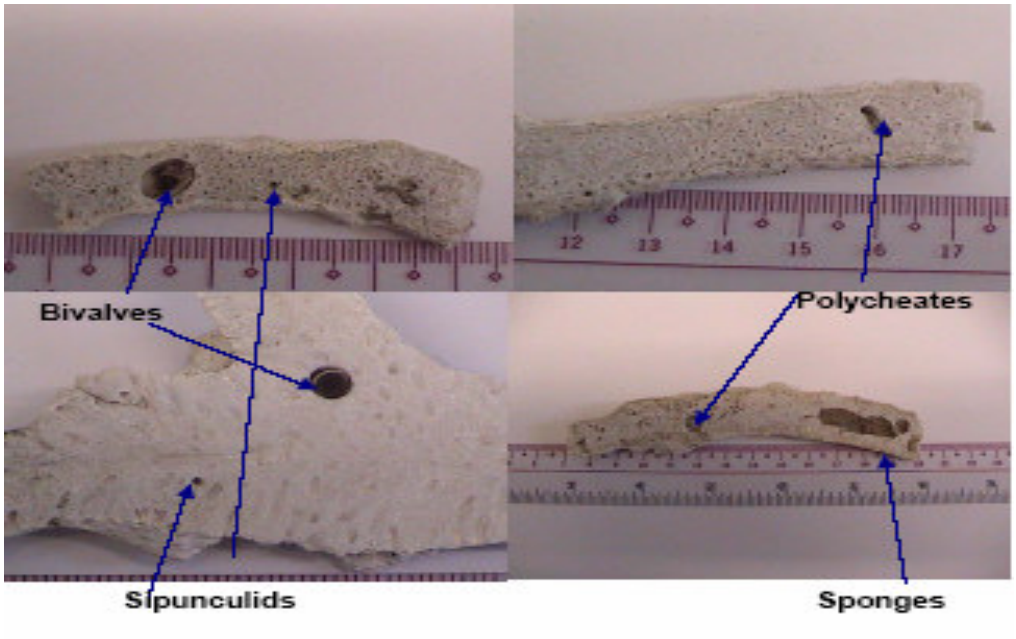


Figure 5.9: The bore shape for the four bioeroding groups: polychaetes, bivalves, sponges and sipunculids, as showed by coral rubble section.

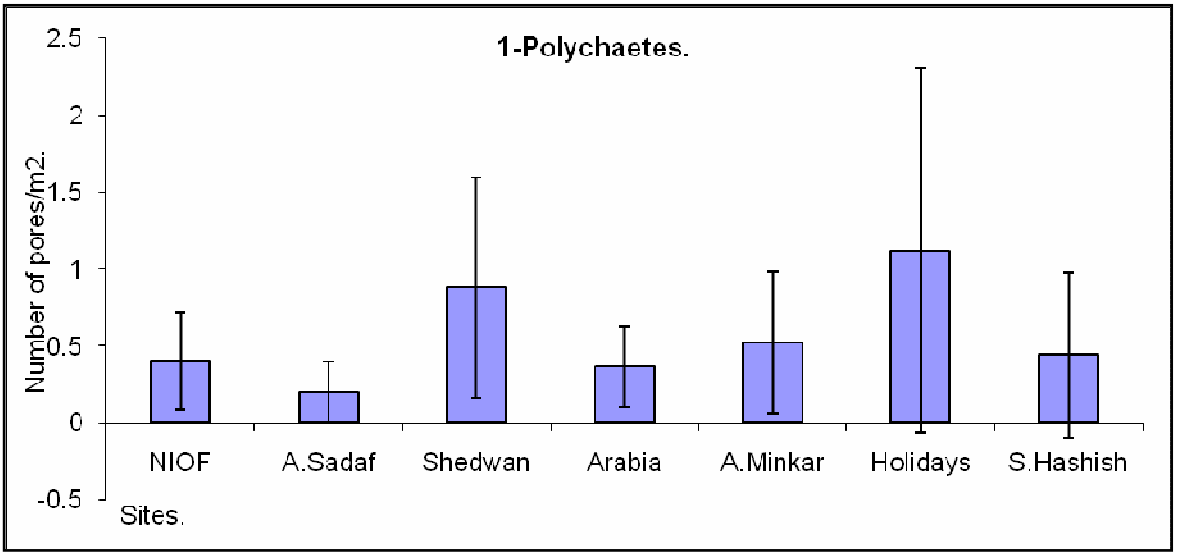


Figure 5.10: The mean number of polychaetes bores per square meter of reef ±standard deviation, surveyed along the reef edge of the seven study sites, Hurghada, Egypt.

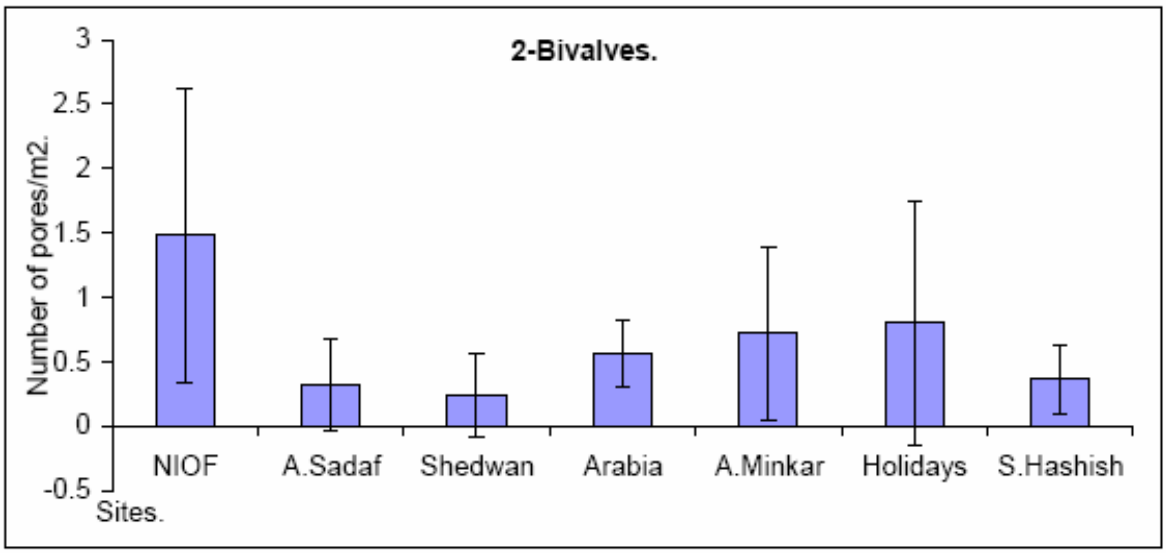


Figure 5.11: The mean number of bivalves bores per square meter of reef ±standard deviation, surveyed along the reef edge of the seven study sites, Hurghada, Egypt.

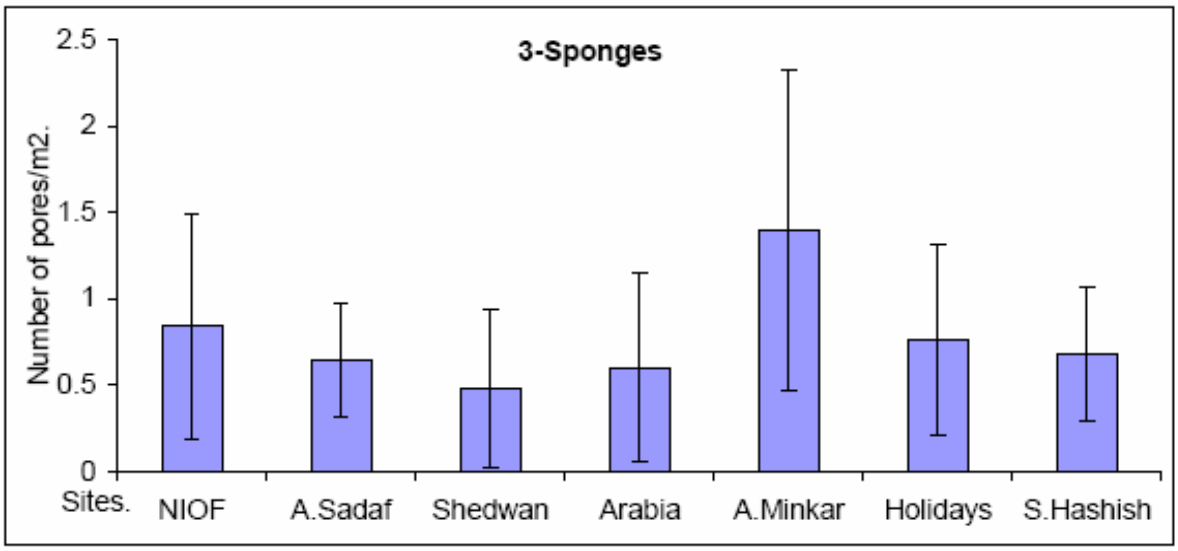


Figure 5.12: The mean number of sponge bores per square meter of reef ±standard deviation, surveyed along the reef edge of the seven study sites, Hurghada, Egypt.

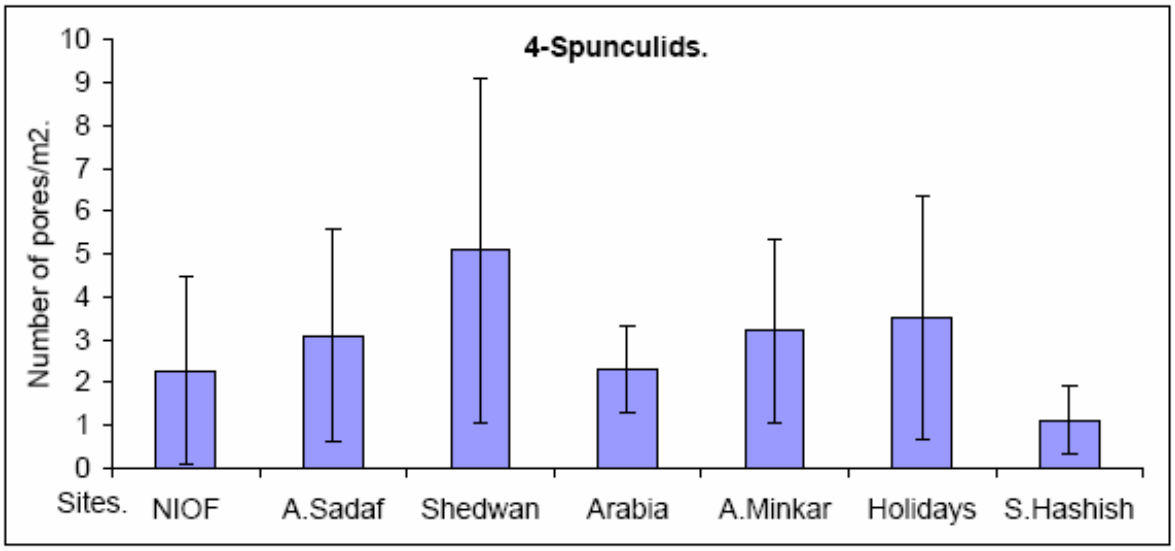


Figure 5.13: The mean number of spunculid bores per square meter of reef ±standard deviation, surveyed along the reef edge of the seven study sites, Hurghada, Egypt.

Two-way ANOVA statistical analysis was carried out to investigate the differences between sites and macroborer groups for the total number of bores per square meter of reef (Table 5.8). The results showed a significant difference between sites in number of bores (F=3.13, df=6, P=0.005) and between groups (F=64.61, df=3, P=0.000). A Post Hoc test showed that the only significant difference was between site 3 and site 7 (Table 5.9). There was a significant difference between sipunculids and polychaetes, bivalves and sponges; but not between polychaetes, bivalves and sponges (Table 5.10).

Spearman's correlations were carried out between each of the boring groups and sedimentation rate, SPM, turbidity, non-carbonate sediment, gravel sand and mud (Table 5.11). Polychaetes and bivalves did not show any significant correlation with any of these factors. Sponges showed significant negative correlation with SPM (R= -0.714, P= .036), gravels (R= -0.714, P= .036) and significant positive correlation with sand (R= 0.750, P= 0.026) (Figure.5.14). Sipunculids showed significant positive correlation with sedimentation rate (R= .750, P= .026) and non-carbonate sediment (R= .714, P= .036) (Figure 5.15).

Table 5.8: The Two-way ANOVA for difference between sites and bioeroding groups in bore number per square meter of reef at the seven study sites, Hurghada, Egypt.

Source of variation	S.S.	D.f.	M. S.	F	P-value
Sites	64.014	6	10.669	3.136	.005
Groups	659.529	3	219.843	64.618	.000
Sites * Groups	216.190	18	12.011	3.530	.000
Error	2354.308	692	3.402		
Total	4294.000	720			

Table 5.9: Matrix of significant differences between sites in bioerosion density, measured at the 7 study sites, as indicated by Tuky test, Hurghada, Egypt.

sites	1	2	3	4	5	6	7
1							
2							
3							
4							
5							
6							
7			x				

Table 5.10: Matrix of significant differences between bioeroding groups in bioerosion intensity at the seven study sites; Hurghada, Egypt.

	Polychaetes	Bivalves	Sponges	Spunculids
Polychaetes				
Bivalves				
Sponges				
Spunculids	x	x	x	

Table 5.11: Spearman's correlations between density of boring by each bioeroding group with sedimentation rate, SPM, non-carbonate sediment, turbidity, gravel, sand and mud; at each of the study sites, Hurghada, Egypt.

		Sediment -ation	SPM	Non carbonate	Turbidity	Gravel	Sand	Mud
Polychaetes	Correlation Coefficient	.321	.143	.357	.321	-.643	.107	.536
	Sig. (1-tailed)	.241(ns)	.380(ns)	.216(ns)	.241(ns)	.06(ns)	.410(ns)	.108(ns)
	N	7	7	7	7	7	7	7
Bivalves	Correlation Coefficient	-.250	-.32	-.071	-.143	-.679	.643	.107
	Sig. (1-tailed)	.294(ns)	.241(ns)	.44(ns)	.38(ns)	.047(ns)	.06(ns)	.410(ns)
	N	7	7	7	7	7	7	7
Sponge	Correlation Coefficient	-.607	-.71	-.464	-.571	-.714	.750	-.071
	Sig. (1-tailed)	.074(ns)	.036(*)	.147(ns)	.09(ns)	.036(*)	.026(*)	.44(ns)
	N	7	7	7	7	7	7	7
Sipunculids	Correlation Coefficient	.750	.643	.714	.571	.000	-.429	.500
	Sig. (1-tailed)	.026(*)	.060(ns)	.036(*)	.09(ns)	0.500(ns)	.169(ns)	.127(ns)
	N	7	7	7	7	7	7	7

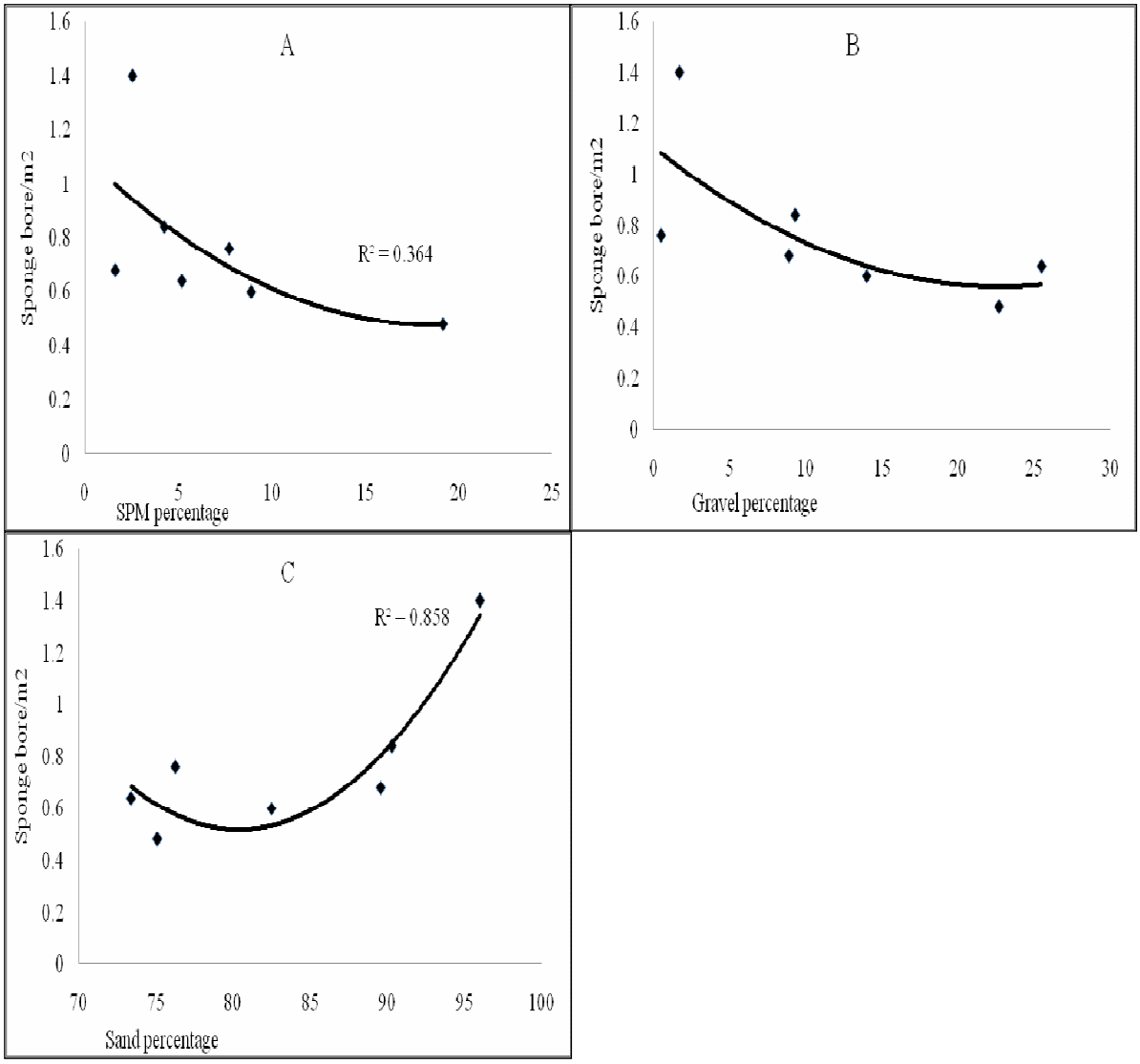


Figure 5.14: The correlation between the number of bores by sponges and SPM (A), gravel (B) and sand percentage (C) at the seven study sites, Hurghada, Egypt.

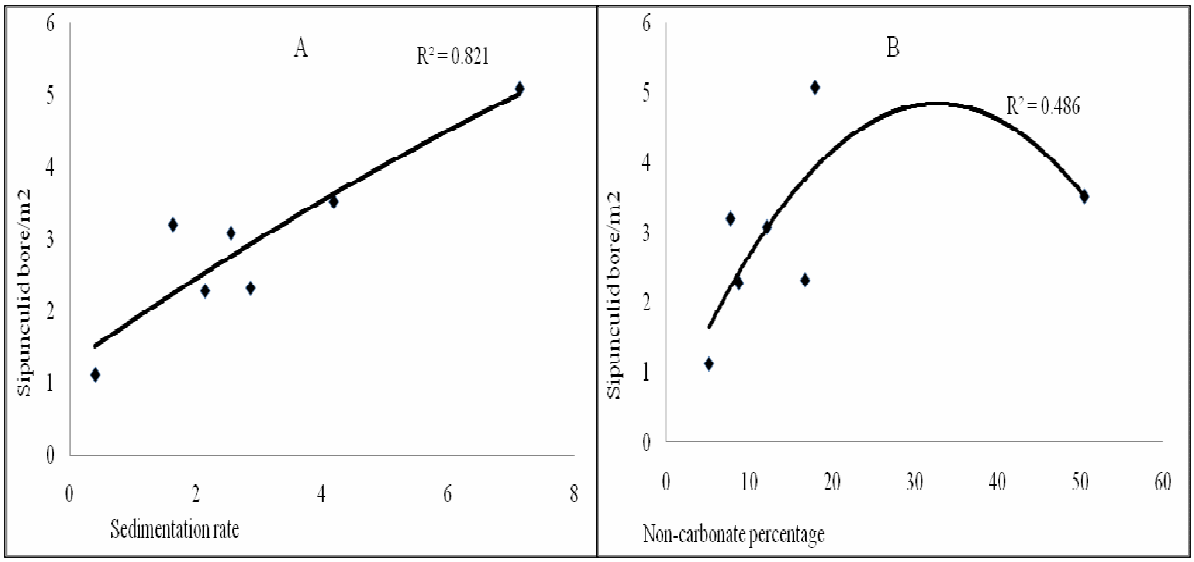


Figure 5.15: The correlation between the number of spunculid bores and sedimentation rate (A), and the percentage of non-carbonate sediment (B) at the seven study sites, Hurghada, Egypt.

5.4 Biological and Environmental Quality Matrix

Quality was estimated for each site according to biological and environmental parameters measured during this study. Biological parameters were Coral cover, abundance, Species Richness, Diversity Index, Deterioration Index (DI), disturbance branched, disturbance massive, mucous branched, mucous massive, transplants survival, recruitment rate, coral feeding fish and algal feeding fish abundance (Table 5.13). Sites were arranged from the best (7) to the worst (1) according to the biological parameters as follow; NIOF (7), S.Hashish (6), A.Sadaf (5), Arabia (4), A.Minkar (3), Holidays (2) and Shedwan (1). Environmental parameters used were sedimentation rate, SPM percentage, non-carbonate sediment percentage, NH_4, NO_3, NO_2, SIO_3, PO_4 and turbidity (Table 5.12). Sites were arranged from the best (7) to the worst (1) according to the environmental parameters as follow; A.Minkar (7), S.Hashish (6), NIOF (5), A.Sadaf (4), Arabia (3), Holidays (2) and Shedwan (1).

One-Sample Kolmogorov-Smirnov test indicated no significant difference from normal distribution for both biological and environmental parameters. Pearson Correlation was carried out between mean ranks of biological and environmental parameters, results showed positive significant correlation (R=0.764, P=0.046) between biological and environmental parameters (Table 5.14). And sites with high rank in relation to environmental parameters have a strong chance to have better biological conditions (Figure 5.13).

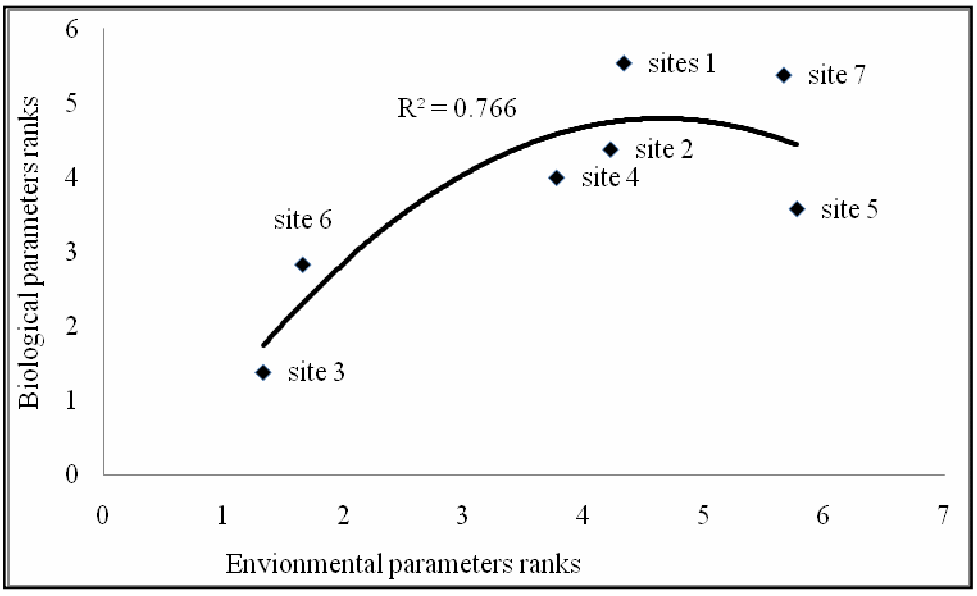


Figure 5.16: The correlation between the biological parameters and the environmental parameters measured at the seven study sites, Hurghada, Egypt.

Table 5.12: Site quality matrix according to environmental parameter ranks arranged from the highest (7) to the lowest (1) quality.

	NIOF	A.Sadaf	Shedwan	Arabia	A.Minkar	Holidays	S.Hashish
sedimentation	5	4	1	3	6	2	7
SPM	5	4	1	3	6	2	7
Non carbonate	3	7	2	4	6	1	5
NH_4	4	6	1	3	7	2	5
NO_3	3	5	2	4	4	1	6
NO_2	3	2	1	4	3	2	5
SiO_3	4	3	1	5	7	2	6
PO_4	7	3	1	5	6	2	4
Turbidity	5	4	2	3	7	1	6

Table 5.13: Quality matrix for sites, according to biological parameters ranks, arranged from the highest (7) to the lowest (1) quality.

Sites	NIOF	A.Sadaf	Shedwan	Arabia	A.Minkar	Holidays	S.Hashish
Coral cover	7	4	1	3	5	2	6
Abundance	5	4	1	3	2	7	6
Species Richness	6	4	1	5	3	2	7
Diversity Index	6	4	1	5	3	2	7
Deterioration Index (DI)	4	5	1	6	2	7	3
Disturbance branched	4	4	2	3	5	1	5
Disturbance massive	7	6	2	4	3	1	5
Mucous branched	7	5	1	3	4	2	6
Mucous massive	6	2	1	4	3	---	5
Transplants survival	5	4	2	6	--	1	3
Recruitment rate	3	5	1	2	4	7	6
Coral feeding fish	6	7	2	3	5	1	4
Algal feeding fish	6	3	2	5	4	1	7

Table 5.14: Pearson Correlations between the ranks of site according to biological and environmental parameters measured at each of the seven study sites, Hurghada, Egypt.

		Environmental parameters
Biological parameters	Pearson Correlation	.764
	Sig. (1-tailed)	.023(*)
	N	7

-Cluster analysis

Cluster analysis was done for the biological and environmental parameters measured in this study using Multivariate Statistical Package (MVSP), to show the similarities between sites. Biological parameters cluster analysis (Figure 5.17) showed the highest similarity between site5 (Abu Minkar) and 6 (Holidays) with distance 133.6, and between site 4 (Arabia) and 7 (S.Hashish) with distance 154.7. The lowest similarities recorded between site 1 (NIOF) and site 3 (Shedwan) with the highest distance 982.2.
Cluster analysis for the environmental parameters (Figure.5.18) indicated that site 5 (Abu Minkar) has the highest similarity with site 7 (S.Hashish) with distance 3.5. Site 1 (NIOF) was the second highest similarity with site 2 (A.Sadaf) with distance 4.1. The least similarity shown between site 5 (Abu Minkar) and site 6 (Holidays) with distance 44.4.

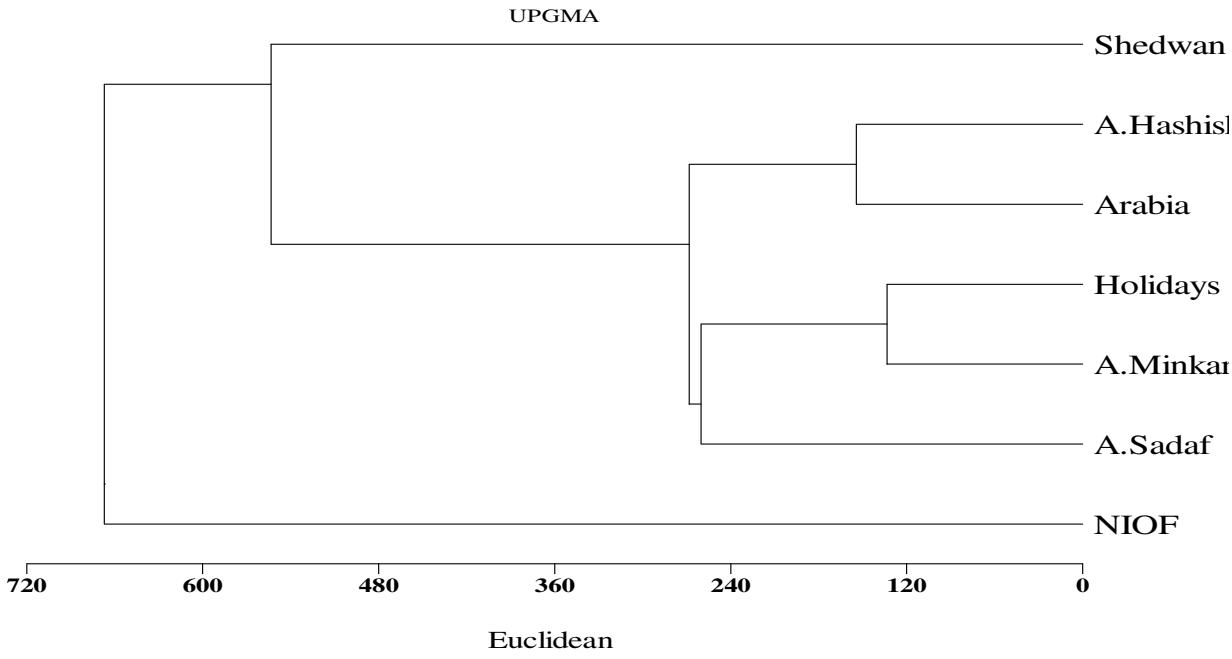


Figure 5.17: The similarity distance between sites in relation to biological parameters data for Hurghada sites, Red Sea.

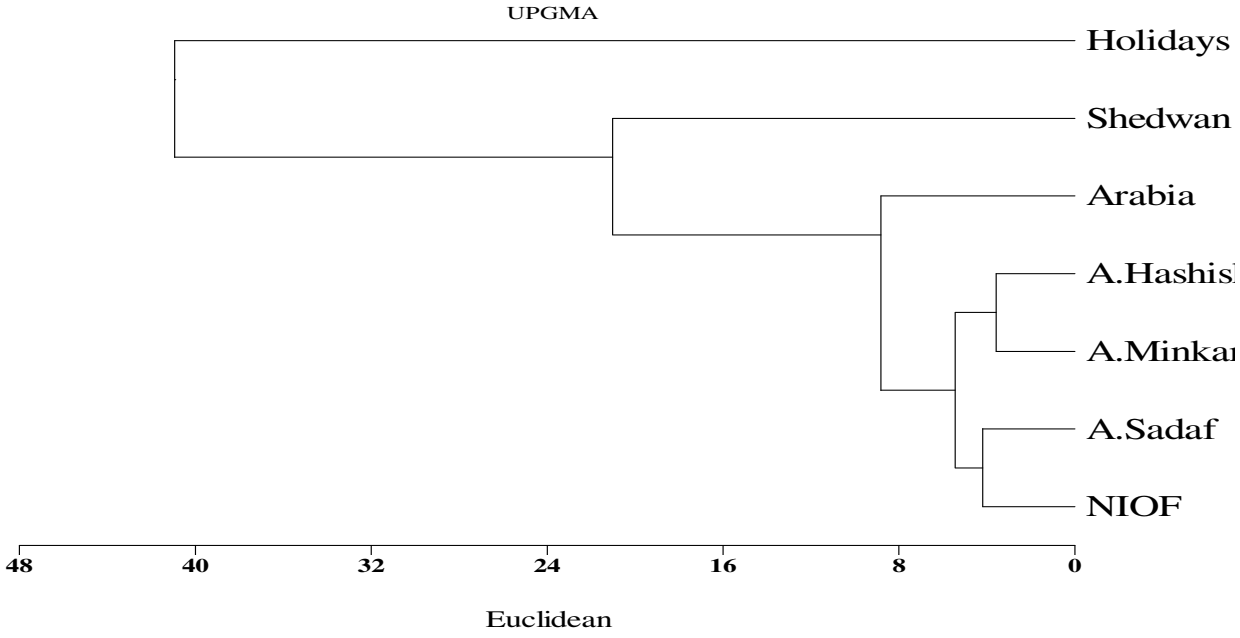


Figure 5.18: The similarity distance between sites in relation to environmental parameters measured in this study for Hurghada sites, Red Sea.

Discussion

A fish abundance survey was carried out for the most common coral feeder and algal feeder fish groups in the Red Sea. Six fish groups were surveyed in this study; three coral feeders (corallivorous), Butterfly (Chaetodontidae), Angel (Pomacanthidae), and wrasse (Labridae); and three algal feeders: Damsel (Pomacentridae), Surgeon (Acanthuridae) and Rabbitfishes (Siganidae). These fish groups are characterized by their dominance of the fish biota in the northern Red Sea on shallower reefs as they represented about 88 % of total fish population in this area (El-Alwany, 2003). They were also considered simple to distinguish and identify in the field. Fish abundance was used as an indicator of reef health and correlations with sedimentation rate, SPM, turbidity, non-carbonate sediment, gravel, sand and mud were tested. Evidence from early studies referred to reefs, which were dead from siltation or from crown-of-thorns or from some other casual factor, as having a poor assemblage of the coral feeder chaetodontids (Hourigan et al., 1988). Some fish species select corals particularly with the highest energy content and most butterfly fishes feed selectively on corals with high-energy contents (El-Alwany, 2003).

The fish abundance results from this study indicated a significant difference between sites and feeding group but not between the two surveys, 2004 and 2006. Algal feeders had significant negative correlations with sedimentation rate, SPM, non-carbonate sediment and turbidity; and a significant positive correlation with percentage of sand. Algal feeder abundance was inversely related to sedimentation rate, suggesting that higher rates of sedimentation inhibit algal feeder colonization, while coral feeder abundance was unrelated to sedimentation rate, indicating that rate of sedimentation had not affected coral feeder distribution. The results indicated that the lowest abundance of both coral feeder and algal feeder fish were recorded at sites 6 and 3 that were mainly those with the lowest coral cover.

The species of the family Chaetodontidae have previously been put forward as candidates

for coral bioindicators, because many are obligate coral feeders (Hourigan et al. 1988). Eight species were recorded during this study from ten species known to inhibit the Red. These species were; *Megaprotodon trifascialis, Chaetodon semilarvatus, Chaetodon paucifasciatus, Chaetodon austriacus, Chaetodon auriga, Chaetodon fasciatus, Chaetodon lineolatus* and *Chaetodon larvatus.* In a study to test for effects of coral disturbance on total numbers of fishes and species richness in 6 major taxonomic families: the Serranidae, Pomacentridae, Chaetodontidae, Apogonidae, Labridae, and Scaridae, the Chaetodontidae were the only family to show a significant reduction in abundance after the disturbance (Lewis, 1997).

In the current study, Chaetodontid fish abundance indicated significant differences between sites but not between the two years of survey. A significant positive correlation with coral cover was found in this study. Although the highest abundance was recorded at site 5 which is the highest in branched coral cover, and had medium species richness and Deterioration Index. The lowest abundance was recorded at site 6, which had the lowest coral cover and species richness. The results of this study agreed with a recent study by Sano (2004) who suggested that corals feeding fishes disappear at sites with high coral death, although other fish groups have no particular response to substantial coral mortality. Also Kawasaki et al (2003) maintained that coral destruction may lead to a marked reduction in fish species richness and abundance due to loss of shelter and living space.

There was a significant difference in the abundance of all macroborer between sites. There were also significant positive correlations between abundance of sipunculids and both sedimentation rate and the percentage non-carbonate sediment. These correlations suggest that sipunculid abundance was higher on reefs where sedimentation rate was high. Sipunculids are deposit feeders and may feed on small sediment particles trapped in coral rubble (Zubia and Peyrot-Clausade, 2001). This may explain the dominance of this group in all study sites in this study.

From the current study, sponge abundance had a strong negative correlation with SPM and the percentage of gravel and a significant positive correlation with the percentage of sand. The inverse relationship implies that sponges occur in lower abundance when sedimentation rate is high. Differences in abundance implied by these correlations suggested that the growth of sponge was inhibited by the high levels of SPM. Polychaetes and bivalves did not show any significant correlation with sedimentation rate, SPM, non-carbonate sediment, gravel, sand or mud, suggesting that higher rates of sedimentation do not have a direct effect on abundance of these two groups of macroborers. These results are supported by the findings of an earlier study on coral rubble which indicated an increased in abundance of bioeroders, especially boeroding sponges, along eutrophication and SPM gradients (Holmes, 1997). However, the study of macroborers in some massive coral species at Discovery Bay, Jamica, carried by Macdonald and Perry (2003) indicated that sponges dominate macroboring communities in low sedimentation sites. There was, however, an increase of the percentage of bivalves and worms as sedimentation increased. Sponges dominated the fore-reef boring community (low sedimentation sites) rather than the diverse assemblage of sponges, worms and bivalves in the back-reef where sedimentation was high (Perry, 1998).

The current survey found a higher percentage of sponge bores at site 5 which generally did not have the highest level of macroborer infestation and also had a low sedimentation rate. In contrast to Macdonald and Perry (2003) who found that sponges did not show a hierarchy of tolerance to sedimentation and turbidity, although it still dominated the boring community at high sedimentation levels, Sipunculid worms dominated all study sites. The highest percentage of bivalves was recorded at site 1 where low sedimentation

was also recorded. Sipunculid worms were higher at site 3 where the highest level of sedimentation was recorded; and site 6 had the highest percentage of polychaete borers and also a high level of sedimentation. Generally the highest level of bioerosion of all groups was found at site 3 and the lowest was at site 7. Early studies refer to change in the boring community over time (Hutchings and Peyrot-Clausade, 2002), which supports the opinion of the early modification of the boring community under continuous sedimentation stress. Reducing sediment input and maintaining water quality in coastal areas is therefore essential to improve reef quality further by reduce destructive effects of coral reef bioeroders.

In conclusion parameters measured in this chapter, fish abundance and bioerosion density were varied in their responses to the environmental parameters measures at this study. Algal feeding fish families indicated obvious reduction in relation to sedimentation rate, SPM, non carbonate and turbidity. These parameters did not decrease the abundance of coral feeding fish families to significant levels, even though the highest sites in sedimentation rate, SPM, non carbonate and turbidity were the lower in fish abundance of coral feeding fish families. Butterfly fish abundance was reduced in response to the decline in coral percentage cover although species richness, diversity index and DI did not indicate the same level of impact. Bioerosion density was generally higher where sedimentation rate and turbidity were higher. Sipunculids abundance was higher at higher sedimentation rate and non carbonate percentage, while sponge bioerosion were at its lowest levels at the highest SPM levels. Polychaetes and bivalves bioerosion intensity had not impacted significantly by these environmental parameters although the highest site in sedimentation rate, SPM and turbidity has the lowest bivalve and the second highest polychaetes bioerosion density. In general fish abundance, chaetodontids fish abundance, sipunculids and sponge bioerosion had showed as important tools for sediment impact detection.

CHAPTER 6

MUCUS, FEEDING AND ZOOXANTHELLAE EXPERIMENT RESULTS.

Introduction

Richman et al. (1975) maintained that corals use mucus secretion to prevent burial in heavy sedimentation reefs and that the rate of mucus production by massive corals is much greater than branched ones. Environmental stresses cause corals to secrete more mucus to coat their outer tissues (Kloeppel, 2001). There has been a lack of detailed study regarding the impact of sedimentation on Hughada reefs particularly in relation to the level of mucus production by corals as a protection mechanism. Mucus secretion by corals is used in this study as an indicator of sedimentation stress. This part of the study aimed to determine mucus secretion rates under various sedimentation rates. It also tries to clarify the correlation between mucus secretion and the variation in sedimentation levels. The laboratory experiment was designed to apply a wide range of sedimentation and aimed to explain what is occurring in the field.

Corals feed either by utilizing organic food as heterotrophs or by functioning as phototrophs, through their association with zooxanthellae (Muscatine, 1990). As heterotrophs coral are able to capture and ingest a wide range of food types, including large sediment particles (Stafford-Smith and Ormond, 1992) and fine suspended particulate matter (Lewis 1997; Anthony, 1999). Coral feeding on sediment is used as an indicator of sedimentation impact on coral reefs. The aim of the coral feeding study is to test the hypothesis that corals, which live in turbid areas, have the capacity to utilize suspended sediment as a food source and may flourish in such stressed conditions. The experiments also intended to find any correlations between sedimentation rate and sediment uptake level by corals. The hypothesis was tested on one common coral species in the study area (*Lobophyllia hemprichii)*, to see if its dominance was related to the ability to utilize this food source more than the other competent coral species.

Zooxanthellae density in coral tissue was inferred to diminish under stress condition such as high turbidity and sedimentation. Early studies had referred to sedimentation as a factor that increases the loss of zooxanthellae and is therefore considered to be detrimental to corals (e.g. Bak, 1978; Abdel-Salam et al, 1988; Hodgson, 1990; Philipp and Fabricius, 2003). However, there has been no detailed study about the effect of sedimentation on coral zooxanthellae density. Zooxanthellae density is used in this study as indication of sedimentation impact on coral reefs in Hurghada coastal area. This part of the study therefore aimed to examine zooxanthellae density and its correlation to sedimentation rate in the field. The laboratory experiment applies a wide range of sedimentation to reveals the mechanisms taking place in the field.

6.1 Mucus Secretion Rates

6.1.1 Mucus Secretion in the field

Coral mucus secretion in the field showed great variations between sites and between branched and massive corals. Mucus production readings indicated that site 3 had the highest mucus secretion rate for both branched (11.18±5.59mg/day) and massive corals (13.58±4mg/day), whilst site 7 was the lowest in mucus production by branched (1.68±0.72mg/day) and site 1 was the lowest in mucus secretion by massive corals (2.25±0.94mg/day) as shown in Figure 6.1.

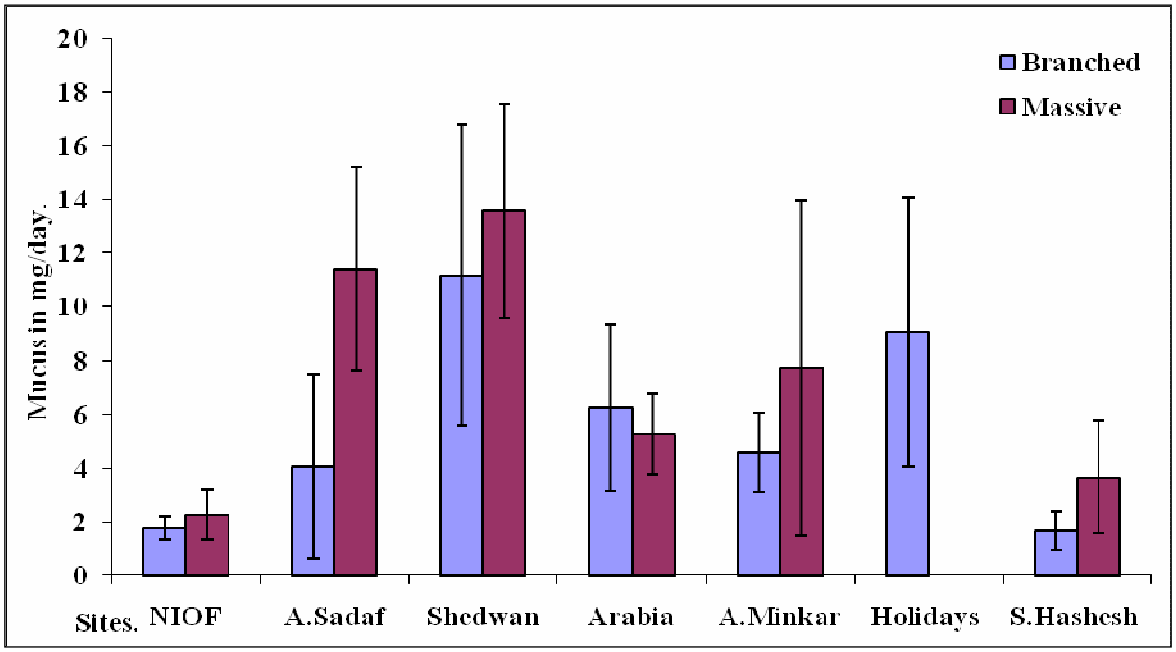


Figure 6.1: Mean mucus secretion rate in mg.day^{-1} ±SD of mean for field samples, between sites and growth forms, branched and massive corals, with standard deviation of mean, in each of the seven study sites, Hurghada, Egypt.

Table 6.1: Statistical analysis for field mucus data using two-way ANOVA, testing mucus secretion between sites and growth forms, for the seven study sites, Hurghada, Egypt.

Source	SS	d.f.	M S	F	P-value
Sites	773.075	6	128.846	9.835	.000
Growth Form	125.131	1	125.131	9.551	.003
Site * Growth Form	58.340	5	11.668	.891	.495
Error	655.058	50	13.101		
Total	4340.181	63			

Two-way ANOVA for mucus secretion between sites and growth forms showed a significant difference between sites (F=9.835, df=6, P=0.000), and between growth forms (F=9.551, df=1, P=0.003) as shown in Table 6.1. Post Hoc test showed that site 1 was significantly different from sites 3 and 6; site 2 was not significantly different from any

other site; site 3 was significantly different from sites 1, 4, 5 and 7; sites 4, 5 and 7 were significantly different from site 3; and site 6 was significantly different from site 1.

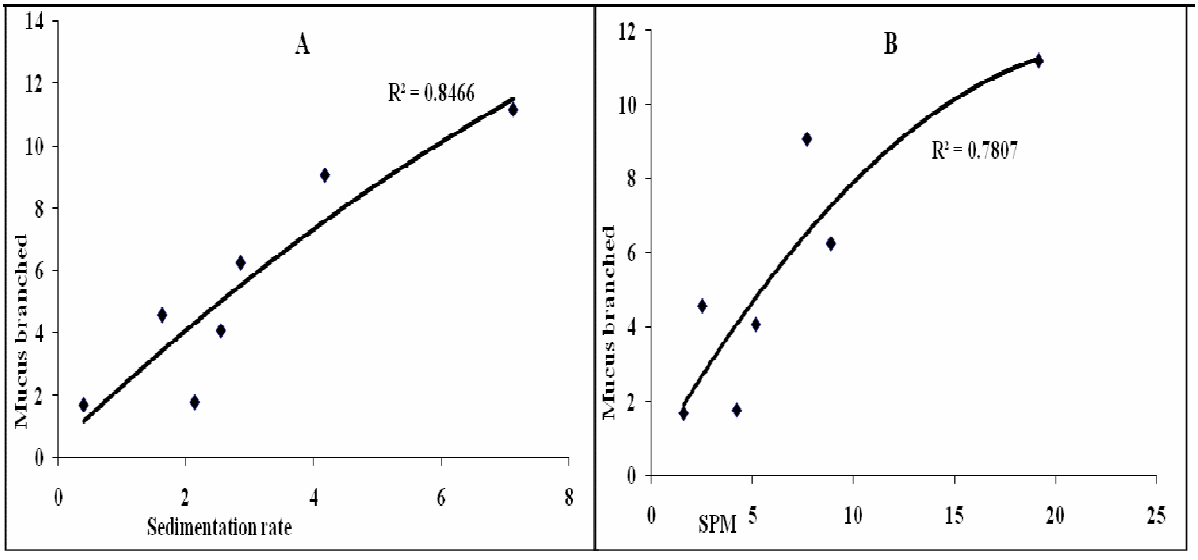
Table 6.2: The significance matrix for between sites differences in mucus production rates, as indicated by Post Hoc Tukey test, for the 7 study sites, Hurghada, Egypt.

Sites	1	2	3	4	5	6	7
1							
2							
3	x						
4			x				
5			x				
6	x						
7			x				

From Pearson correlation tests (Table 6.3), it was found that mucus production by branched corals has significant positive correlation with sedimentation rate (R=0.917, P=0.002), SPM (R=0.866, P=0.006) and turbidity (R=0.798, P=0.016). Mucus production by massive corals showed significant positive correlation with sedimentation rate (R=0.734, P=0.048).

Table 6.3: Pearson correlation tests between mucus secretion by branched and massive corals in mg/day and; sedimentation rate, SPM, non-carbonate, turbidity, gravel, sand and mud; for all of the study sites, Hurghada, Egypt.

		Sediment	SPM	Non carbonat	Turbidity	Gravel	Sand	Mud
Branched	Pearson Correlation	.917	.866	.650	.798	.161	-.634	.494
	Sig. (1-tailed)	.002(***)	.006(**)	.057(ns)	.016(*)	.365(ns)	.063(ns)	.130(ns)
	N	7	7	7	7	7	7	7
Massive	Pearson Correlation	.734	.667	.595	.492	.691	-.713	.185
	Sig. (1-tailed)	.048(*)	.074(ns)	.106(ns)	.161(ns)	.064(ns)	.056(ns)	.363(ns)
	N	6	6	6	6	6	6	6



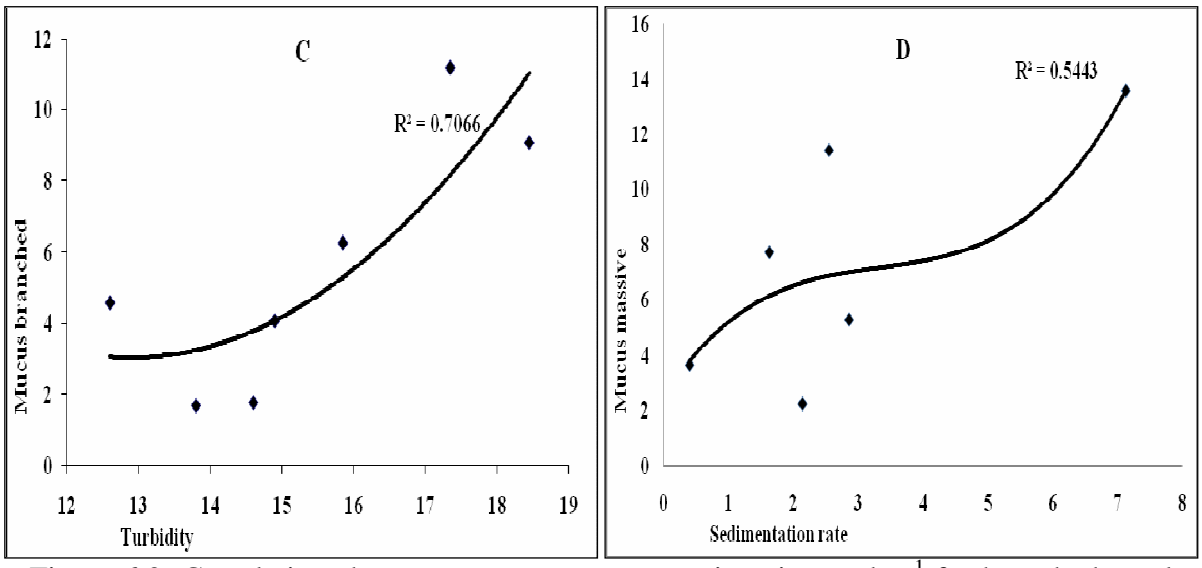


Figure 6.2: Correlations between mean mucus secretions in mg.day^{-1} for branched corals: sedimentation rate (A), SPM (B) and turbidity (C), and between mean mucus secretions by massive coral and sedimentation rate (C) at the seven sites, Hurghada, Egypt.

6.1.2 Laboratory Mucus Readings

Mucus production for the three species, *Acropora tenuis, Pocillopora damicornis* and *Stylophora pistillata,* under different sedimentation condition, showed great variations between species and treatments. *Stylophora pistillata* had the highest mucus production rate of the three species used in this experiment, *Acropora tenuis* came next and *Pocillopora damicornis* has the lowest mean mucus production. The highest level (13.07 mg.day^{-1}) was found at 30mg.day^{-1} sedimentation rate by *Stylophora pistillata,* whereas the lowest (2.85mg.day^{-1}) level was recorded at 5mg.cm^{-2} sedimentation rate from *Pocillopora damicornis* (Figure 6.3).

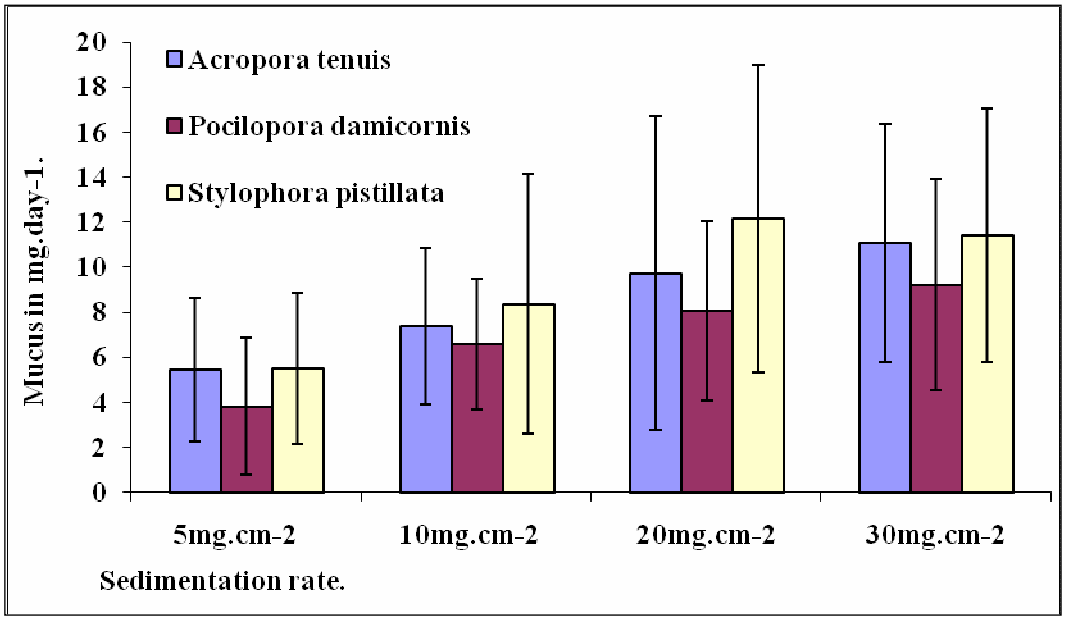


Figure 6.3: The mean mucus secretion rate ± SD of mean for the three species: *Acropora tenuis, Pocillopora damicornis* and *Stylophora pistillata* used in laboratory experiment at four different sedimentation levels.

Two-way ANOVA for mucus secretion at different sedimentation levels and different coral species (Table 6.4) showed a significant difference between different sedimentation levels (F=6.9, df=3, P=0.000), but not between species (F=2.1, df=2, P=0.12). Therefore it can be concluded that there is a significant difference between treatments in the mean mucus secretion at the four sediment treatments. A Post Hoc test showed that treatment 1 was significantly different from treatments 3 and 4, treatment 2 was not significantly different from any other treatment, and treatments 3 and 4 were significantly different from treatments 1.

Pearson correlations (Table 6.5) between mucus production by the three coral species and sedimentation rate showed a significant positive correlation between sedimentation rate and *Acropora tenuis* (R=0.980, P=0.01), but not between sedimentation rate and mucus production by *Pocillopora damicornis* (R=0.939, P=0.03).

Table 6.4: Statistical analysis using two way ANOVA for mucus secretion rates at different sedimentation levels, under laboratory condition for the three species, *Acropora tenuis, Pocillopora damicornis* and *Stylophora pistillata.*

Source	SS	d.f.	M S	F	P-value
Treatment	482.133	3	160.711	6.955	.000
Species	97.711	2	48.856	2.114	.127
Treatment* Species	20.502	6	3.417	.148	.989
Error	1941.045	84	23.108		

Table 6.5: Pearson correlation test between mucus secretion by *Acropora tenuis, Pocillopora damicornis* and *Stylophora pistillata* corals in mg.day^{-1} and sedimentation rate.

		Acropora sp.	*Pocillopora sp.*	*Stylophora sp.*
Sedimentation rate	Pearson Correlation	.980	.939	.872
	Sig. (1-tailed)	.010(*)	.030(*)	.064(ns)
	N	4	4	4

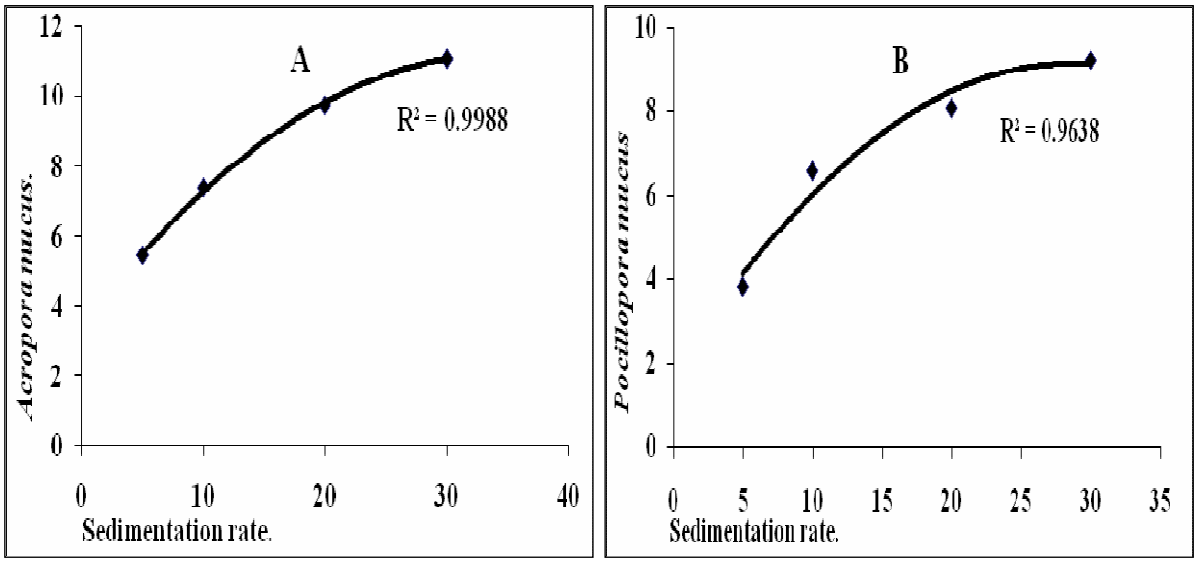


Figure 6.4: The correlation between sedimentation rate and mucus secretion rate for *Acropora tenuis* (A) *and Pocillopora damicornis* (B) as indicated by laboratory experiment at four different sedimentation levels.

6.2 Coral Feeding

6.2.1 Feeding at Different Sediment Qualities and Coral Conditions

A feeding experiment was carried out to test the ability of some dominant coral species to acclimatize to and withstand high sedimentation rates and to explore their ability to get benefit from living in this condition. The feeding experiment using the large polyp coral *Lobophyllia hemprichii* showed variation in sediment uptake levels. Figure 6.5 shows the mean of fluorescence uptake for each treatment. Fluorescence was quantitatively measured and calibrated against standard solution of fluorescein-isothiocyanate in µm/litre. The standard curve of spectrophotometer readings of known standard solutions as plotted, indicating a high accuracy in the concentrations used for sediment labelling in the feeding experiment (R=0.999) (Figure 6.6).

Statistical analysis using one way ANOVA for the five different treatments (Table 6.6) indicated a significant difference between treatments in fluorescence levels (F=7.314, df=4, P=0.002). A Post Hoc test showed that treatment 1 and 2 were significantly different from treatments 3 and 4; treatment 3 and 4 were significantly different from sites 1, 2 and 5; treatment 5 was significantly different from treatments 3 and 4 (Table 6.7).

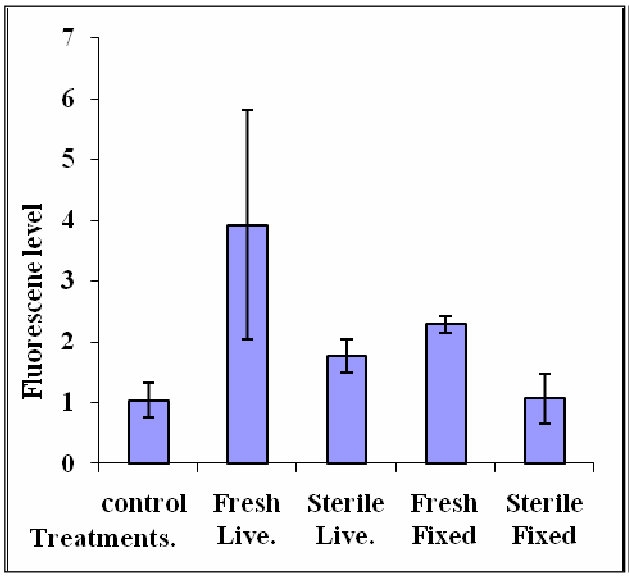


Figure 6.5: The mean fluorescence level for each of the five feeding treatments and standard deviation of mean.

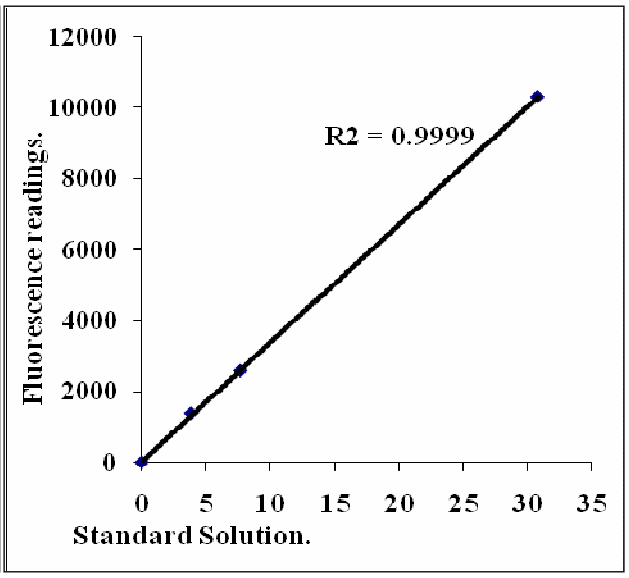


Figure 6.6: Calibration curve for fluorescein-isothiocyanate solution, used for sediment labelling in the feeding experiment.

Table 6.6: The one way ANOVA for fluorescence levels between the five treatments of coral feeding of *Lobophyllia hemprichii,* a common species in the fringing reef of the study sites, Hurghada, Egypt.

	S.S.	Df	M. S.	F	P value
Between Groups	59.026	4	14.757	20.465	.000
Within Groups	32.448	45	.721		
Total	91.474	49			

Table 6.7: The significance matrix for between different treatments (fresh and sterile sediment and control with no sediment) of live and fixed coral in fluorescene level, as indicated by Post Hoc Tukey test, for the 7 study sites, Hurghada, Egypt.

Treatments	1	2	3	4	5
1					
2					
3	x	x			
4	x	x			
5			x	x	

6.2.2 Feeding at Different Sedimentation Levels

A second feeding experiment was carried out to test the variations in feeding rates at different sedimentation levels. The large polyp coral *Lobophyllia hemprichii* showed an increase in sediment uptake up to a concentration of 20mg/l sediment. At higher sediment levels the coral started to decrease sediment uptake. Figure 6.7 shows the mean fluorescence uptake for each sediment level. Again, the standard curve of spectrophotometer readings of known standard solutions was plotted (Figure 6.8), indicating a high accuracy in the concentration used for sediment labelling in the feeding experiment (R=0.997).

Statistical analysis using one way ANOVA for the five different treatments (Table 6.8) indicated a significant difference between treatments in fluorescence levels (F=13.15, df=4, P=0.000). Fluorescence was quantitatively measured and calibrated against standard solution of fluorescein-isothiocyanate in µm/litre. A Post Hoc test (Table 6.9) showed that treatment 1 and 2 were significantly different from treatments 3 and 4; treatments 3 and 4 were significantly different from treatments 1, 2 and 5; treatment 5 was significantly different from treatments 3 and 4.

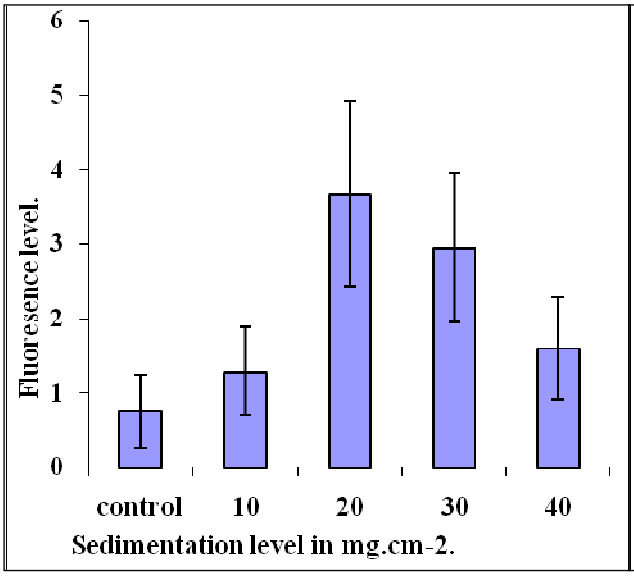


Figure 6.7: The mean feeding rate of *Lobophyllia hemprichii* at four levels of sediment treatments and Standard Deviation of mean.

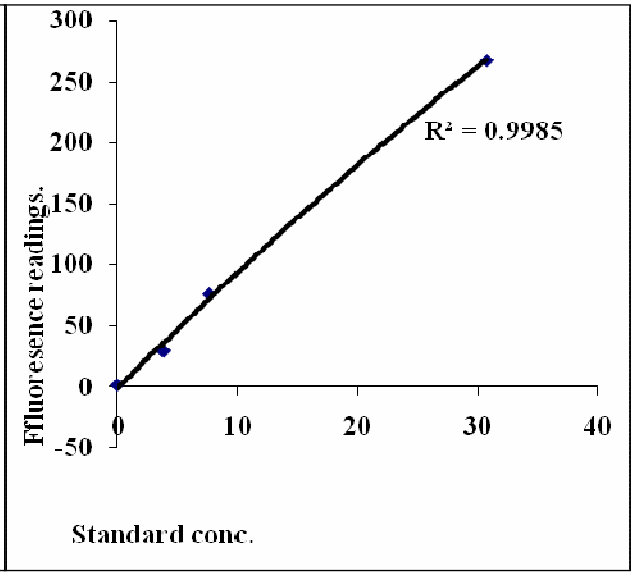


Figure 6.8: Calibration curve for fluorescein-isothiocyanate solution, which used for sediment tagging in the feeding experiment.

Table 6.8: Statistical analysis using one way ANOVA between sedimentation levels in *Lobophyllia hemprichii* feeding experiment.

	S. S.	df	M. S.	F	P value
Between Groups	19.749	4	4.937	13.153	.000
Within Groups	7.507	20	.375		
Total	27.257	24			

Table 6.9: The significance matrix between different sediment treatments (control, 10, 20, 30 and 40mg.cm^{-2}) in fluorescene level, as indicated by Post Hoc Tukey test, for the 7 study sites, Hurghada, Egypt.

Treatments	1	2	3	4	5
1					
2					
3	x	x			
4	x	x			
5			x	x	

6.3 Zooxanthellae Density

6.3.1 Field Zooxanthellae Density

Experiment of zooxanthellae density in transplanted *Acropora tenuis* indicated a great variation between sites and periods after transplantation. The mean number of zooxanthellae cells per square centimetre of coral tissue varied from 1281.4x10^3 at site 1 to 365.8x10^3 at site 3 after one month of transplantation, while the control samples collected at the beginning of the experiment had a mean of 1741.1x10^3 cells per square centimeter of coral tissue. Densities after 2 months varied from 850.1x10^3 at site 4 to 283.7x10^3 at site 3 (Figure 6.9). At two months, readings showed that transplants at three of the seven sites (4, 5 and 7) had increased zooxanthellae density, whilst the other four sites (1, 2, 3 and 6) suffered from a continuing decrease in zooxanthellae density. The differences between the two readings varied from as much as 220.6x10^3 increases at site 4 to as much as 653.33x10^3 decreases in zooxanthellae density at site 1.

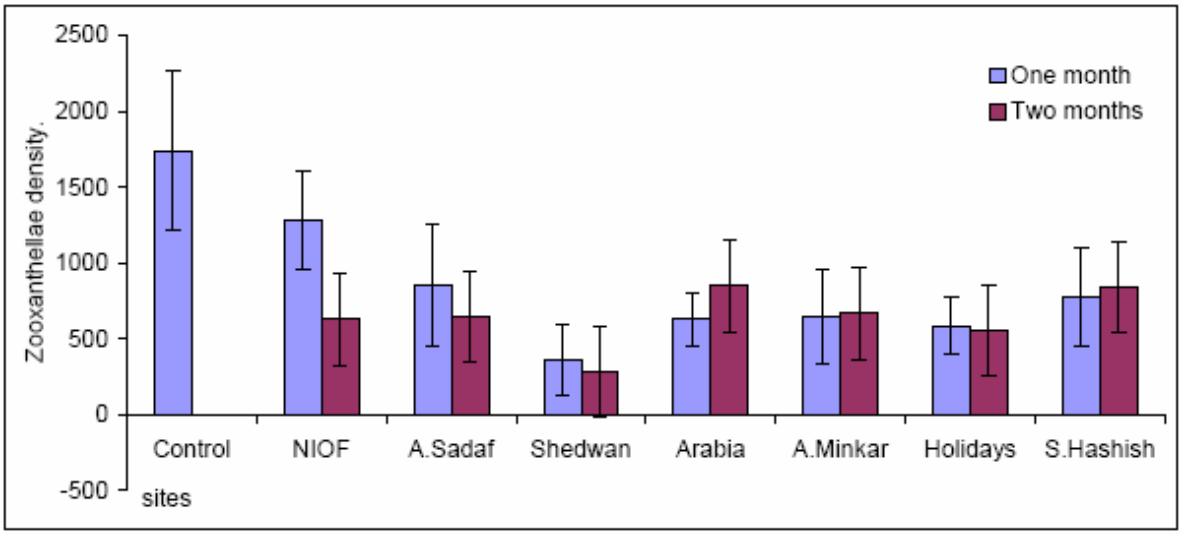


Figure 6.9: The mean (n=30) number of zooxanthellae cell per square cm² of coral tissue ± SD of *Acropora tenuis,* after one month and two months of transplantation in the seven study sites compared with a control sample in Hurghada reefs, Egypt.

Table 6.10: Statistical analysis using two way ANOVA for zooxanthellae cells count between sites and months after transplantation in the seven study sites, Hurghada, Egypt.

Source	S. S.	df	M S	F	P-value
Site	14302511.048	6	2383751.841	27.912	.000
Season	13299453.689	2	6649726.844	77.864	.000
Site* Season	7045045.067	6	1174174.178	13.749	.000
Error	37149847.600	435	85401.949		
	348438948.000	450			

Two way ANOVA (Table 6.10) was carried out to investigate the effect of sites and duration on zooxanthellae density in coral tissues. The outcome showed high significant differences between sites (F=27.91, df=7, P=0.00) and time (F=77.8,df=2, P=0.00) on cell number per unit area of tissue. A Post Hoc test showed that sites 1 and 3 were significantly different from all other sites; sites 2 and 4 were significantly different from sites 1, 3 and 6; site 5 was significantly different from sites 1 and 3; site 6 was significantly different from sites 1, 2, 3, 4 and 7; site 7 was significantly different from sites 1, 3, and 6 (Table 6.11).

Table 6.11: The significance matrix between sites in zooxanthellae cell density as indicated by Post Hoc Tukey test, for the 7 study sites, Hurghada, Egypt.

Sites	1	2	3	4	5	6	7
1							
2	x						
3	x	x					
4	x		x				
5	x		x				
6	x	x	x	x			
7	x		x			x	

Spearman's Correlations were carried between zooxanthellae density and sedimentation rate, SPM, non-carbonate sediment, turbidity, gravel, sand and mud (Table 6.12). One month readings showed a significant negative correlation with sedimentation rate (R=-0.714, P=0.035), SPM (R=-0.679, P=0.047), non-carbonate sediment (R=-0.679, P=0.047) and mud (R=-0.786, P=0.018) (Figure 6.10). Two month readings did not indicate any significant correlations with sedimentation rate, SPM, non-carbonate sediment, turbidity, gravel sand or mud.

Table 6.12: Spearman's Correlation for sedimentation rate, SPM, non-carbonate sediment, turbidity, gravel, sand and mud, with zooxanthellae cell density after one and two months of transplantation in the seven study sites, Hurghada, Egypt.

		Sediment -ation	SPM	Non carbonate	Turbidity	Gravel	Sand	Mud
One Month	Correlation Coefficient	-.714	-.679	-.679	-.643	.143	.321	-.786
	Sig. (1-tailed)	.035(*)	.047(*)	.047(*)	.060(ns)	.380(ns)	.241(ns)	.018(*)
	N	7	7	7	7	7	7	7
Two Months	Correlation Coefficient	-.607	-.429	-.571	-.536	-.036	.393	.071
	Sig. (1-tailed)	.074(ns)	.168(ns)	.090(ns)	.107(ns)	.470(ns)	.191(ns)	.439(ns)
	N	7	7	7	7	7	7	7

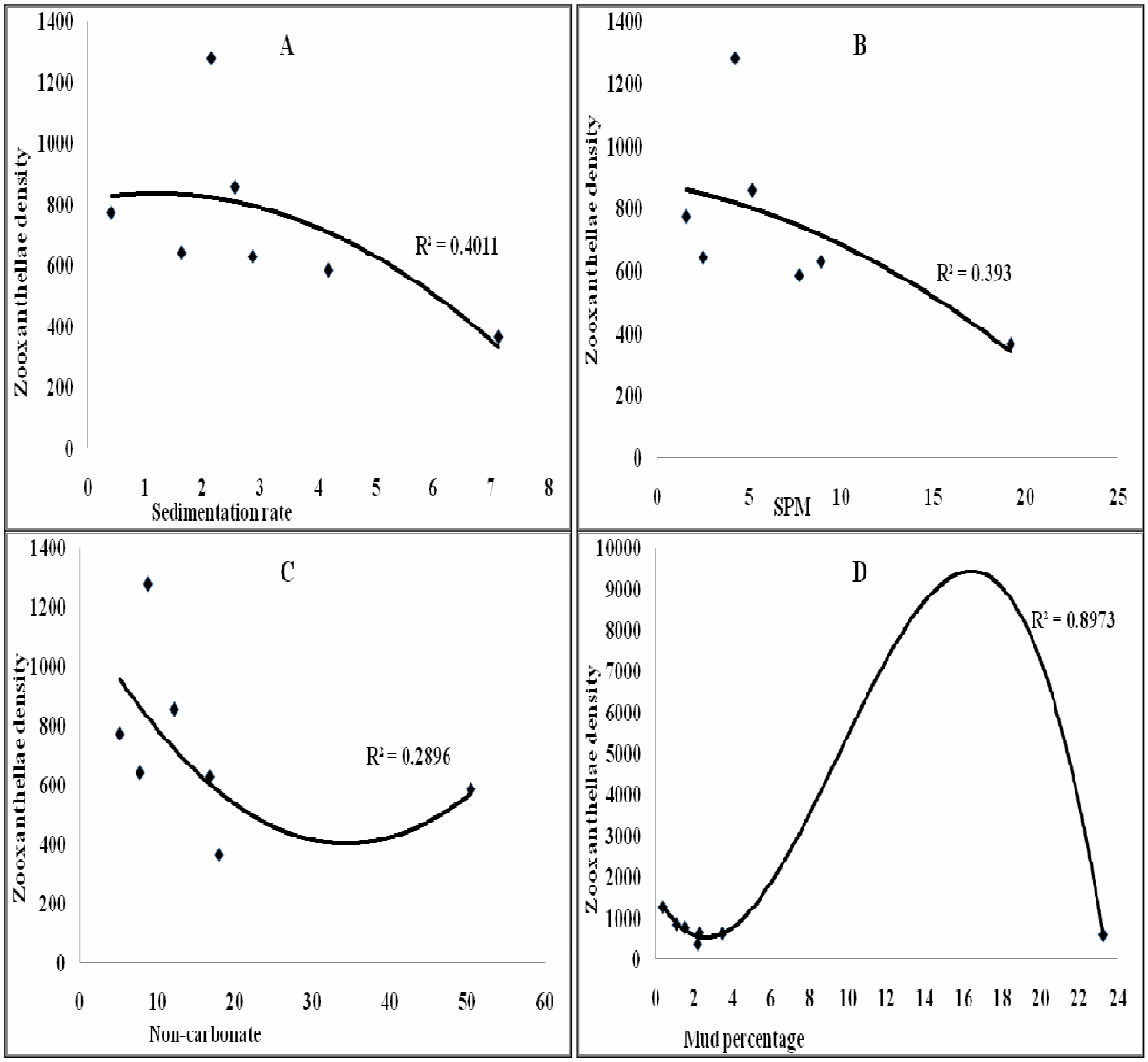


Figure 6.10: The correlations between the mean number of zooxanthellae cells per cm² in *Acropora tenuis*, after one month of transplantation; with sedimentation rate (A), SPM (B), non-carbonate sediment (C) and percentage of mud (D), for each of the study sites, Hurghada, Egypt.

6.3.2 Laboratory Readings of Zooxanthellae Density

The density of zooxanthellae cells were measured in the tissues of three coral species; *Acropora tenuis, Stylophora pistillata* and *Pocillopora damicornis* exposed to four levels of sedimentation; 5mg.cm^{-2}.day^{-1}, 10mg.cm^{-2}.day^{-1}, 20mg.cm^{-2}.day^{-1} and 30mg.cm^{-2}.day^{-1}, for a period of four weeks. Mean zooxanthellae cell density for *Acropora tenuis* showed continuous reduction during the four weeks of treatment (Figure 6.11). *Acropora tenuis* showed a decrease in zooxanthellae density from 1700.5x10^3 at the beginning of the experiment (control samples) to 807.5x10^3 after four weeks exposed at 30mg/day sedimentation. Zooxanthellae density in *Stylophora pistillata* tissues reduced from 1957x10^3 at the control samples to 636x10^3 after four weeks exposure to 30mg.cm^{-2} sedimentation (Figure, 6.12). The mean zooxanthellae density in *Pocillopora damicornis* reduced rapidly after two weeks of treatments (Figure 6.13) from 1104x10^3 in control samples to 404.5x10^3 after four weeks of exposure to sediment at 20mg/day. The samples started to bleach after 20 days and died under the higher sediment level (30mg.cm^{-2}.day^{-1}).

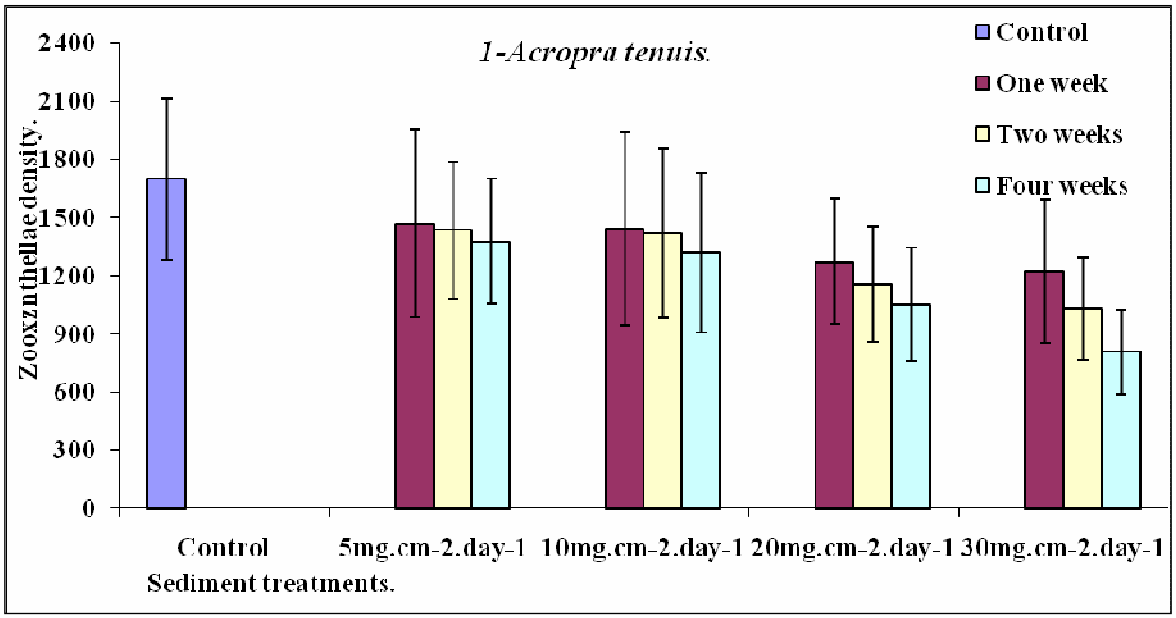


Figure 6.11: The mean number of zooxanthellae cell per cm² tissue of *Acropora tenuis* ± SD, after one, two and four weeks of sediment treatments at four levels of sedimentation; 5mg.cm^{-2}.day^{-1}, 10mg.cm^{-2}.day^{-1}, 20mg.cm^{-2}.day^{-1} and 30mg.cm^{-2}.day^{-1}, under laboratory conditions.

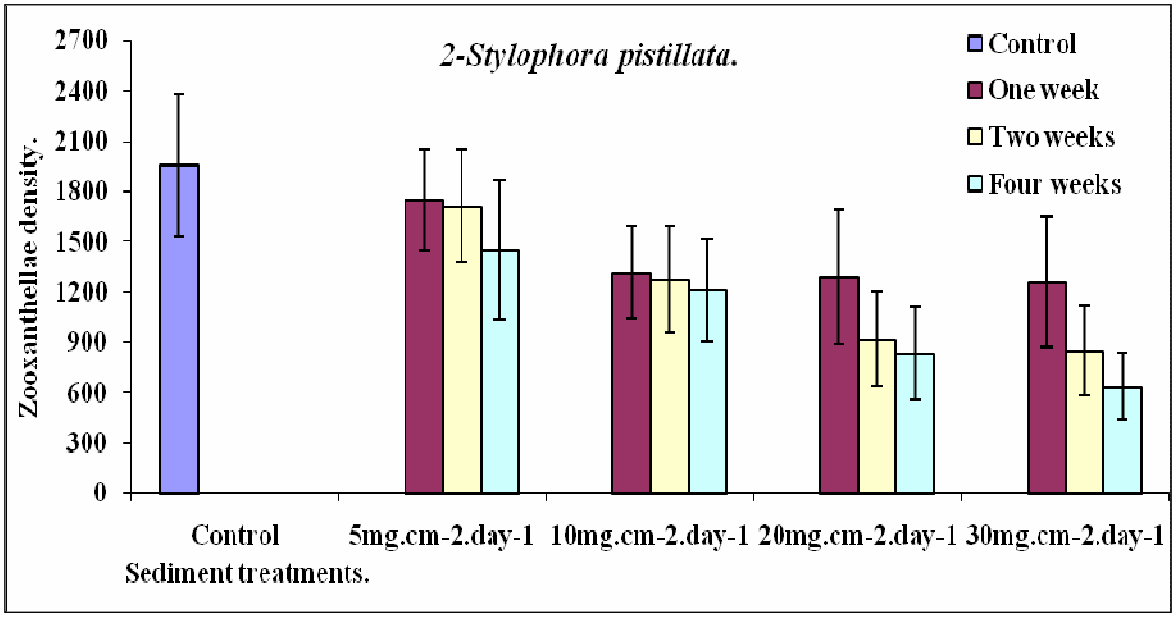


Figure 6.12: The mean zooxanthellae density in coral tissue of *Stylophora pistillata* ± SD, after one, two and four weeks of exposure at four levels of sedimentation; 5mg.cm^{-2}.day^{-1}, 10mg.cm^{-2}.day^{-1}, 20mg.cm^{-2}.day^{-1} and 30mg.cm^{-2}.day^{-1}, under laboratory conditions.

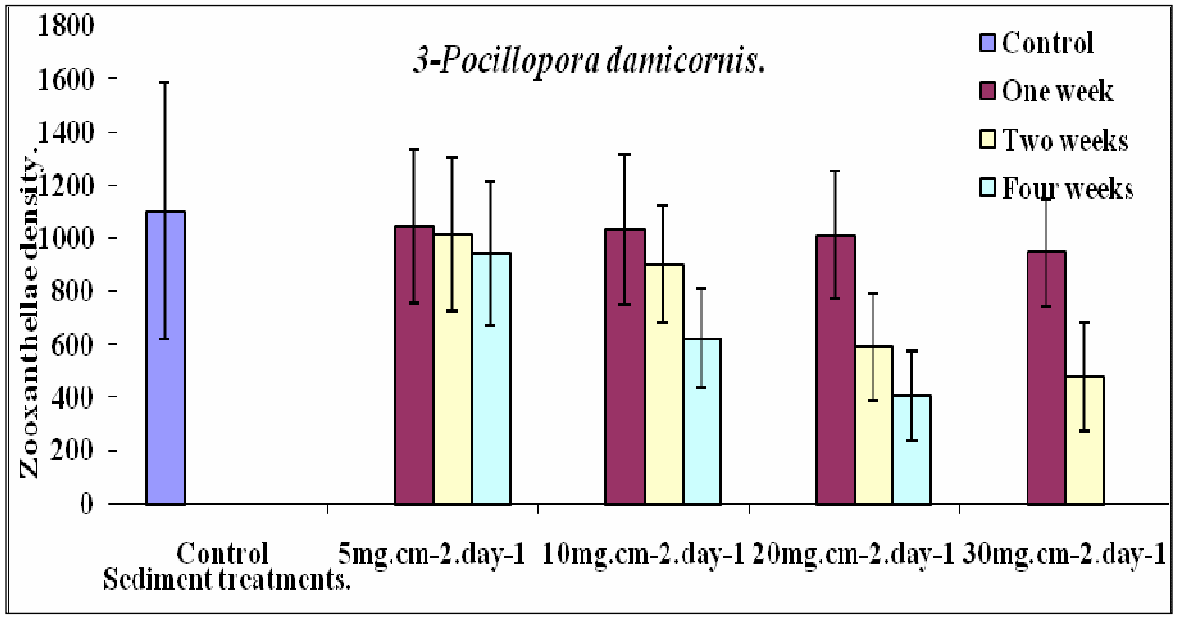


Figure 6.13: The mean number of zooxanthellae cell per cm² of coral tissue of *Pocillopora damicornis* ± SD; after one, two and four weeks of exposure at four levels of sedimentation; 5mg.cm^{-2}.day^{-1}, 10mg.cm^{-2}.day^{-1}, 20mg.cm^{-2}.day^{-1} and 30mg.cm^{-2}.day^{-1}, under laboratory conditions.

Three way ANOVA (Table 6.11) was carried out to investigate the difference between treatments, period and species on zooxanthellae density in coral tissues. The results indicated highly significant differences between treatments (F=83.81, df=3, P=0.000), period (F=60.35, df=2, P=0.000) and species (F=185.3,df=2, P=0.000) in zooxanthellae cell density per unit area of tissue. Post Hoc tests showed that control samples and treatments 1 and 2 were significantly different from all other treatments. Treatments 3 and 4 were significantly different from control, 1, and 2, but not from each other. Control samples, period 1, period 2 and period 3 were significantly different from each other in

zooxanthellae density. *Stylophora pistillata* and *Acropora tenuis* were significantly different from *Pocillopora damicornis* but not from each other.

Table 6.13: The three way ANOVA test for zooxanthellae cell density between the three species; *Stylophora pistillata, Acropora tenuis* and *Pocillopora damicornis,* the four treatments and time.

Source of variation	S.S.	D.f.	M. S.	F	P-value
Sediment level	254097.898	3	84699.299	78.502	.000
Period	121981.104	2	60990.552	56.528	.000
Species	364279.066	2	182139.533	168.813	.000
Sediment * Period	31220.660	6	5203.443	4.823	.000
Sediment* Species	30330.696	6	5055.116	4.685	.000
Period * Species	12860.007	4	3215.002	2.980	.019
Sediment * Period* Species	10124.754	11	920.432	.853	.587
Error	778995.450	722	1078.941		
Total	11599813.000	760			

Table 6.14: The significance matrix for differences between each of the four sediment treatments and control samples (before sediment addition) in zooxanthellae density as indicated by Post Hoc Tukey test, for the laboratory experiment, Hurghada, Egypt.

Treatment	Control	5mg.cm^{-2}.day^{-1}	10mg.cm^{-2}.day^{-1}	20mg.cm^{-2}.day^{-1}
Control				
5 mg.cm^{-2}.day^{-1}	x			
10 mg.cm^{-2}.day^{-1}	x	x		
20 mg.cm^{-2}.day^{-1}	x	x	x	
30 mg.cm^{-2}.day^{-1}	x	x	x	

Table 6.15: The significance matrix for differences between each of the three exposure periods to sediment treatments and control samples (before sediment addition) in zooxanthellae density as indicated by Post Hoc Tukey test, for the laboratory experiment, Hurghada, Egypt.

Period	Control	1st week	2nd week
Control			
1st week	x		
2nd week	x	x	
4th week	x	x	x

Table 6.16: The significance matrix for differences in zooxanthellae density between each of the three species (*Stylophora pistillata, Acropora tenuis* and *Pocillopora damicornis*) used in sediment treatment experiment, as indicated by Post Hoc Tukey test, for the laboratory experiment, Hurghada, Egypt.

Species	*Acropora tenuis*	*Stylophora pistillata*	*Pocillopora damicornis*
Acropora tenuis			
Stylophora pistillata			
Pocillopora	*x*	*x*	

Pearson correlation tests between zooxanthallae density and rate of sedimentation and period of treatment were carried out. The results indicated a significant negative correlation between zooxanthallae densities with all sedimentation treatments (Table 6.17). The least significant correlation was found between sedimentation rate and first week readings of zooxanthallae density in *Stylopohora pistillata* (R=-.843, P=0.036). Periods of treatment with zooxanthellae densities indicated a significant negative correlation between sedimentation rate and all treatments of *Acropora tenuis*, *Pocillopora damicornis* and *Stylophora pistillata* (Table 6.18).

Table 6.17: Pearson correlations between sedimentation rate and zooxanthellae density in the three species *Acropora tenuis, Stylophora pistillata* and *Pocillopora damicornis*; after the three treatment periods, one week, two weeks and four weeks; under laboratory conditions, Hurghada, Egypt.

		Ac. 1 week	Ac. 2 weeks	Ac. 4 weeks	Poc. 1 week	Poc. 2 weeks	Poc. 4 weeks	Sty. 1 week	Sty. 2 weeks	Sty. 4 weeks
Sediment -ation rate	Pearson Correlation	-.935	-.969	-.980	-.960	-.987	-.991	-.843	-.945	-.958
	Sig. (1-tailed)	.008(**)	.003(***)	.001(***)	.004(***)	.001(**)	.0005(***)	.036(*)	.007(**)	.005(**)
	N	5	5	5	5	5	5	5	5	5

Table 6.18: Pearson correlations between zooxanthellae density in the three species *Acropora tenuis, Stylophora pistillata* and *Pocillopora damicornis* and period of treatment with sediment; after the three treatment periods, one week, two weeks and four weeks under laboratory condition, Hurghada, Egypt.

		Ac. 5mg	Ac. 10mg	Ac. 20mg	Ac. 30mg	Pocil. 5mg	Pocil. 10mg	Pocil. 20mg	Pocil. 30mg	Sty. 5mg	Sty. 10mg	Sty. 20mg	Sty. 30mg
Period	Pearson Correlation	-.914	-.929	-.935	-.975	-.987	-.956	-.973	-.979	-.967	-.847	-.945	-.971
	Sig. (1-tailed)	.043(*)	.035(*)	.032(*)	.012(*)	.006(**)	.022(*)	.013(*)	.010(*)	.016(*)	.076(ns)	.027(*)	.014(*)
	N	4	4	4	4	4	4	4	4	4	4	4	4

Discussion

The mucus study showed that a significant difference was found between both sites and growth forms (massive and branching) in relation to mucus production. Site 3 had the highest mucus production by both branched and massive corals, whilst site 7 was the lowest for mucus production by branched and site 1 by massive. Most of the sites (sites 1, 2, 3, 5 and 7) proved to have a higher mucus production by massive corals than by branched corals. The outcome of the current study therefore supports the results of earlier studies that corals use mucus production as a protective technique against sedimentation and that massive corals are more reliant on this mechanism than branched species (Richman et al, 1975). There was a significant positive correlation between mucus production by branched corals and sedimentation rate, SPM and turbidity. Production of mucus by massive corals also showed a significant positive correlation with sedimentation rate. These correlations suggest that mucus production is higher on reefs where sedimentation rate is high. Corals use mucus secretion to prevent burial in heavy sedimentation conditions, branching corals withstand high sediment levels because they are less likely to become buried and massive corals overcome heavy sedimentation by high mucus production (Loya, 1972).

Results from the laboratory study on mucus production for the three *species Acropora tenuis, Pocillopora damicornis* and *Stylophora pistillata* showed a significant difference between the mucus secretions at different sedimentation rates, but not between the three species. Generally, *Stylophora pistillata* had the highest mucus production of the three species used in this study, especially at 20mg.cm^{-2} sedimentation rate but this started to decrease at higher sedimentation levels. Mucus secretion by *Acropora tenuis* and *Pocillopora damicornis* indicated a significant positive correlation with sedimentation rate. The positive relationships support the results of the field study, indicating that mucus production is related to sedimentation, suggesting that higher rates of sedimentation increase mucus production by *Acropora tenuis* and *Pocillopora damicornis.*

The results from the feeding experiment indicated a significant difference between the five treatments (live coral with fresh labeled-sediment, live coral with sterile labeled-sediment, fixed coral with fresh labeled-sediment, fixed coral with sterile labeled-sediment and live coral with no sediment) and a high uptake of fresh sediment by live coral. Live corals displayed a reasonable ability to reject low quality sterile sediment as indicated from the difference between fresh and sterile sediment treatments. The results agreed with a study carried out by Rosenfeld et al. (1999) which indicated that active ingestion of the organic matter in the sediment had occurred. This result is assumed to represent the pattern of heterotrophic ability of coral species that dominate high turbidity and sedimentation habitat. In the second experiment, in which different levels of sedimentation were applied, there was a significant difference between the four treatments. The results indicated that there was an increase in sediment uptake up to concentration of 20mg.l^{-1} sediment. At higher sediment levels sediment uptake by coral started to decrease. Rosenfeld et al. (1999) suggested that sediment could represent a source of nutrients in the oligotrophic reef environment, with various roles among coral species according to its organic content, quantity and grain composition.

Anthony (1999) maintained that scleractinian reef corals might achieve up to 50% of their predicted tissue growth by feeding on fine suspended sediment at high sediment concentrations (30mg dry weight). A previous study of sedimentation reported high coral cover and diversity in some reef lagoons despite high turbidity and sedimentation levels, which were considered lethal for corals (Ayling and Ayling, 1991). This supports the

suggestion that adaptation to intense sediment regimes may occur in the species that inhabit high sedimentation areas. Anthony (2000) argued that corals in turbid environment have a greater capacity to feed on suspended sediment than analogous corals from oligotrophic environments, suggesting local adaptation to sedimentation had occurred.

A common shallow reefs coral *Acropora tenuis* was selected to examine the response of the symbiotic zooxanthellae to sedimentation level variation. The results of the study showed a general decrease of zooxanthellae cells in transplants at all sites after one month of transplantation. After two months zooxanthellae counts showed a continuous decrease at some sites (site 1, 2, 3 and 6; while sites 4, 5 and 7 showed an increase in zooxanthellae density. The continuous decrease in zooxanthellae cells in some sites indicated that the poor conditions which could lead to coral bleaching, were being encountered. Zooxanthellae density in coral tissues of the transplanted coral *Acropora tenuis* showed significant differences between sites and between the two readings. Significant negative correlations between first month readings and sedimentation rate, SPM, the percentage of non carbonate and the percentage of mud were proven. This supports results from previous studies, which had found sediment load to increases the loss of zooxanthellae and is, therefore, considered to be detrimental to corals (e.g. Bak, 1978; Abdel-Salam et al, 1988; Hodgson, 1990; Philipp and Fabricius, 2003). First month readings of zooxanthellae density were inversely related to sedimentation rate, suggesting that higher rates of sedimentation inhibit the coral's ability to maintain abundant zooxanthellae cells. The second month's readings did not follow the same negative correlation because the corals at some sites started to retain higher zooxanthellae density.

Results from the laboratory study of zooxanthellae density in the three coral species; *Acropora tenuis, Stylophora pistillata* and *Pocillopora damicornis*; indicated significant differences between sediment treatments (5, 10, 20 and 30mg.cm^{-2}.day^{-1}), period (one, two and four weeks) and the three species used. All three species showed significant negative correlations between zooxanthellae density and sedimentation rate at the three sampling periods. Correlations of zooxanthellae density with the period of exposure to sedimentation showed various degrees of significant correlation. These ranged from highly significant for *Pocillopora* at 5mg.cm^{-2}.day^{-1} to non-significant for *Stylophora* at 10mg.cm^{-2}.day^{-1}.

Zooxanthellae occur in nature at densities of between 0.5 and 5x10^{6}.cm^{-2} of coral tissues and convey up to 95% of their photosynthate to their coral host (Muscatine, 1990). Any factor that reduces the efficiency of the symbiotic relationship between corals and the zooxanthellae will have a major effect on the scleractinian coral, and consequently the productivity of the entire coral reef (Glynn, 1996). In a study of zooxanthellae carried out by Stimson (1997), results showed that variation in irradiance, UV levels and skeletal coral tissue growth are important factors affecting the density of zooxanthellae in coral tissues. Bleached corals lose zooxanthellae and turn white, the colour of the underlying skeleton of calcium carbonate. The animal partner may survive and the symbiosis recover over a period of weeks to months or, alternatively, the animal tissues may die (Douglas, 2003). Corals and other symbiotic species on some reefs have suffered mass mortality as a consequence of bleaching events (Wilkinson, 1999). Two transplantation studies reported changes in the density of zooxanthellae with a change in depth (Dustan, 1979; Gleason and Wellington, 1993). One study reported a decrease in density with an increase in depth (Dustan, 1979). Gleason and Wellington (1993), however, observed a decrease in the density of zooxanthellae in *Montastrea annularis* after transfer to shallower waters, and showed that this decrease is specifically in response to increased UV radiation. These results should, however, be considered in regard to the results of Gladfelter et al (1989)

who stated that zooxanthellae density is not uniform through the coral colony and tissue.

In general conclusion mucus secretion, zooxanthellae density and feeding rates measured in this study had impacted by sedimentation and SPM. Sedimentation, SPM and non carbonate had increased mucus secretion while they had reduced zooxanthellae density. Sediment feeding by corals was increase to certain level where feeding start to decrease under the impact of sedimentation and turbidity shading. Mucus secretion rates were generally higher where sedimentation rates were higher. Zooxanthellae density decreased significantly under the effect of sedimentation rate, SPM and non carbonate sediment. Zooxanthellae density and mucus secretion had showed as important tools in sediment impact detection.

CHAPTER 7

GENERAL DISCUSSION AND CONCLUSIONS

Much information is available on the individual effects of sedimentation on corals mortality, but few studies have been made specifically on the effect of sedimentation on other reef health parameters such as diversity, species richness, abundance, coral disturbance, transplanted coral survival, mucous secretion, zooxanthellae density, species composition and deterioration index. The main studies concerning the effects of sedimentation on coral reefs were: Loya (1976), Hodgson (1990), Rogers (1990), Ayling and Ayling (1991), Richmond (1994), Torres et al. (2001), Airoldi (2003), Nugues and Roberts (2003a,b), Philipp and Fabricius (2003), Fabricius (2005), Dikou and van Woesik (2006), Fabricius and McCorry (2006) and Weber et al (2006). Further research is needed on the response of individual reef organisms and the reef system as a whole to sedimentation, and the threshold levels for individual reef species (hard corals, soft corals and others) and for the reef ecosystem (Rogers, 1990).

There are several general parameters that are commonly considered to reflect the condition or health of coral reefs. These include: live coral cover, diversity, deterioration and mortality indices, abundance, colony size, and fish abundance (Van Woesik and Done, 1997; Jameson et al, 1997; Ben-Tzvi et al, 2004 and Ruiz-Zárate and Arias-González, 2004). Sedimentation affects the coral community by reduction of coral cover, colony abundance, species richness, recruitment rates and mean colony size (Brown and Howard, 1985; Rogers, 1990; Brown, 1997; Fabricius, 2005, Dikou & Van Woesik, 2006). Philipp and Fabricius (2003) maintained that short-term exposure of coral reefs to sedimentation severely affected the quantum yield of the photosystem, chlorophyll a, CO_2 concentrations, and zooxanthellae densities. Their further field measurements also showed that foliose corals or corals with relatively small polyps such as the massive *Porites* were particularly sensitive to sedimentation.

Results from the current study in Hurghada provided direct measurements of sedimentation rate, SPM, turbidity and the percentage non-carbonate sediment, gravel, sand and mud at each site and their correlation with various reef health parameters. Correlation tests were used to confirm major associations between the variables measured and these environmental parameters. Sedimentation rates in the current study throughout the two years varied from mean highest 9.49 to mean lowest 0.4mg.cm^{-2}.day^{-1} along the Hurghada area. Compared to the threshold level of 10mg.cm^{-2}.day^{-1} which, proposed by Rogers (1990) and assumed to reduce cover and diversity of coral reefs, the mean measured sedimentation rate did not exceed this level. Although in many sites it reached higher levels, as high as 25.4mg.cm^{-2}.day^{-1} at Shedwan and 17.2mg.cm^{-2}.day^{-1} at Holidays in February 2006. In November 2006 sedimentation levels increased to 20.1 and 10.9 mg.cm^{-2}.day^{-1} at Shedwan and Holidays respectively. Compared to a study of sedimentation during four seasons in Sharm El-Shiekh (Amer, 2005) which showed sedimentation rates ranged from 0.067 to 4.8mg.cm^{-2}.day^{-1}, the current study throughout the two years were generally higher with mean range from 9.49 to 0.4mg.cm^{-2}.day^{-1} along two years.

Many studies refer to low level sedimentation impacts such as increased respiration and

reduced net photosynthesis (Abdel-Salam et al, 1988), reduced coral cover and coral species diversity, increased partial or total burial of colonies, bleaching and surface colonization by filamentous blue-green algae (Acevedo and Morelock, 1988). Sedimentation also results in loss of zooxanthellae and reduced calcification (Bak, 1978). Compared to the above studies, the current results indicated that the sites with highest sedimentation rates have the lowest recruitment, survival and zooxanthellae.

This study has shown that the sitrs with highest sedimentation rates had the lowest coral cover, species richness, diversity, fish abundance and the highest disturbance, mucus production and bioerosion. The results of this study therefore support earlier studies which maintained that sedimentation reduced coral cover (Loya, 1976; Cortes and Risk, 1985; Acevedo and Morelock, 1988; Brown et al, 1990; Chansang et al, 1981; and Morelock et al, 1983); changed coral community structure and life forms and reduced species richness (Loya, 1976; Morelock et al, 1983; Pastorok and Bilyard, 1985; Acevedo and Morelock, 1988; Rogers, 1990; Brown et al, 1990; Edinger et al, 1998; and West and Van Woesik, 2001).

Suspended particulate matter SPM measurements demonstrated the same trend, with the highest (19.19mg.l^{-1}) at site 3 and the lowest (1.6mg.l^{-1}) at site 7 and showed highly significant positive correlations with sedimentation rate. It has been indicated that SPM reduced coral cover, species richness, diversity, fish abundance, zooxanthellae density and sponge bioerosion, and increase disturbance, mucus production and sipunculid worm abundance. These results agreed with earlier studies which maintained that SPM reduced light levels and reduced zooxanthellae photosynthesis and resulted in small average colony size, but had no effect on partial mortality (Tomascik and Sander, 1985; Lewis, 1997). SPM and sedimentation increased the linear extension of corals at moderate SPM and reduced linear extension at high SPM due to smothering (Tomascik and Sander, 1985; Lewis, 1997). Although measured levels of sedimentation and SPM were well within the normal ranges defined by Rogers (1990) at most of the study sites, there was evidence that reefs in the area are extremely degraded to a critical level. This is supported by the outcome of a recent study by Dikou & Van Woesik (2006) which proved continuous reef deterioration along the west coast of the southern islands of Singapore under low sedimentation levels. Evidence such as colony fission, severe reduction in live coral cover and highly significant negative correlations between the percentage of both live coral cover and the dead coral component, and the thick layer of sediment and filamentous algae carpeting the sessile corals, suggested a continued reef deterioration (Dikou & Van Woesik, 2006).

The current study showed that the threshold level defined by Rogers (1990) could be exceeded at certain sites during certain seasons as mentioned above, although most of the sites did not reach this level. However the sites which exceeded this proposed threshold level showed the lowest reef quality in regard of the parameters measured in this study and agreed with many of early studies. At high sedimentation and SPM (>10mg.$cm^{-2}d^{-1}$ and >10mg.l^{-1}) coral reefs suffer reduction in coral species richness, live coral cover, coral growth rates, calcification, net productivity of corals, rates of reef accretion and increased proportion of branching forms (Cortes and Risk, 1985; Dodge et al, 1974; Bak, 1978; Van Woesik and Done, 1997). Reduction in linear extension and growth is inversely related to sedimentation (Cortes and Risk, 1985; Dodge et al, 1974). Sedimentation reduces mean colony sizes through stunted growth and/or reduced life expectancy (Van Woesik and Done, 1997). Sedimentation could also increase mean colony sizes through reduced recruitment (Wesseling et al, 2001; Cortes and Risk, 1985; and Tomascik and Sander, 1985). On the other hand some sites did not go beyond the threshold level defined by

Rogers (1990) and showed reduced reef health condition in relation to the parameters measured. This may explained by the outcome of Philipp and Fabricius (2003) who maintained that low levels of sedimentation for extended periods of time altered the photophysiology of corals as much as exposure to high amounts of sediment for short periods of time. The non significant correlation between sedimentation rate and some of the parameters measured are supported with the outcome of a study done by Wielgus et al. (2004) which revealed that, at low sedimentation rates there is no correlation between sedimentation rate and coral mortality. Low or brief sedimentation is also known to reduce photosynthetic yield and high or prolonged sedimentation results in loss of zooxanthellae, partial mortality, but increases species-specific tolerances (Philipp and Fabricius, 2003).

Mean turbidity levels from the current study ranged from 18.4 NTU at site 6 to 12.6 NTU at site 5. In contrast to a study carried out by Telesnicki and Goldberg (1995) the current levels were lower than the levels (28–30 NTU) reported to increase mucus production and depress the P: R ratio to below 1.0, possibly due to increased respiration. Turbidity is known to reduce growth, net primary productivity and alter community structure after bleaching and death in several coral species (Rogers, 1979). The results from this study showed a significant negative correlation between coral cover and turbidity. This agreed with the findings from a study carried out by Yentsch et al. (2002) which showed that water transparency and percent coral cover were significantly related. Decreases in water transparency affected corals in two ways; it forced corals to grow in more shallow waters, and forced corals to produce fragile thin skeletons vulnerable to damage (Cook et al., 1997). This study has shown that the most turbid reefs has the lowest coral cover, species richness, diversity, fish abundance and the highest disturbance, mucus production and bioerosion. These sites also have the lowest recruitment, survival and zooxanthellae density. Therefore mucus production was shown to be induced at much lower turbidity levels compared with that described by Telesnicki and Goldberg (1995).

Two factors appear to increase sediment suspension in the coastal area of Hurghada. First, beach re-nourishment activities are frequently carried out on the adjacent hotel resorts. Second, the area is exposed to turbulent currents and rough weather, which stir bottom sediment and strongly wash coastal sediment. It could also be suggested that the threshold sedimentation level proposed by Rogers was exceeded during the early stages of tourism development along the Hurghada coast, which resulted in the observed low coral cover and diversity in most of the coastal sites although there is no evidence for this. Alternatively the threshold proposed by Rogers (1990) has generally not been exceeded in Hurghada but that continuous low to medium levels of sedimentation my have a long term impact on coral cover. These results were coincided with the earlier mentioned study (Philipp and Fabricius, 2003) which maintained that low levels of sedimentation for extended periods of time altered the photophysiology of corals as much as exposure to high amounts of sediment for short periods of time.

A significant positive correlation between the percentage of non-carbonate sediment and both sedimentation rate and SPM was found. This suggests that the synchronized increase of sedimentation, SPM and non-carbonate sediment has occurred as a consequence of land filling processes undertaken in the region. High-suspended particulate matter (SPM) loads in the water column could be a result of sediment resuspension and may cause accumulation of sediment on coral tissues in return (Anthony & Fabricius, 2000). In conclusion sedimentation rate, SPM levels, non-carbonate sediment and turbidity levels all act together to support the strong relationship between poor environmental quality and the proximity to coastal developments. All of the above parameters were at their highest levels at the sites which had undergone intensive dredging and filling disturbance in the

back reef flat. These sites were also near to tourist developments in the area and still receive sediment from land. The results of this study have shown that the sites with the highest loads of non-carbonate sediment had the lowest coral cover, abundance, species richness, diversity, fish abundance, recruitment, survival and zooxanthellae density and the highest disturbance, mucus and bioerosion.

The nutrient study showed that NH^+_4, NO^-_3, NO^-_2 and PO^{3-}_4 levels were generally lower than those reported from the Egyptian Environmental Policy Program (EEPP) water quality report for the Red Sea region (Awad and Shabara, 2003), Global Environmental Facility monitoring project carried out in the Red Sea region (GEF, 1997) and the nutrient enrichment study in coral reef of the Gulf of Aqaba, Red Sea (Rasheed et al, 2002) as show in table 7.1. NH^+_4 levels in the study area were lower than what is considered eutrophic (at 15µM) and to affect zooxanthellae density and calcification. Early studies referred to enrichment with dissolved inorganic nutrients NH_4 plus PO_4 as increasing zooxanthellae density and increased protein synthesis by zooxanthellae (Muscatine et al, 1989). NH^+_4 did not alter the buoyant weight gain of corals at 10µM but reduced it by 60% at 20µM (Ferrier-Pages et al, 2000). It also increase zooxanthellae density, chlorophyll concentration and decreased linear extension in corals (Stambler et al. 1991). NH_4 (at 15µM) increased zooxanthellae density and chlorophyll in adult corals (Snidvongs and Kinzie, 1994).

NO^-_3 levels were also lower than eutrophic levels (>1µM) which were reported to increase zooxanthellae size, chlorophyll per zooxanthellae, photosynthesis and coral protein (through greater zooxanthellae biomass) and reduce respiration per unit protein (Marubini, 1996). NO^-_3 at 2µM had no effect on zooxanthellae density or rate of photosynthesis but reduced buoyant weight gain by 34% after 3 weeks (Ferrier-Pages et al, 2001). In addition, NO^-_3 at 15µM reduced primary production but did not alter zooxanthellae density or chlorophyll concentration after two weeks (Nordemar et al, 2003). Mean PO^{3-}_4 readings were lower than what is reported to cause eutophication (2µM), Kinsey and Davies (1979) maintained that at 2µM, PO^{3-}_4 increased photosynthesis and reduced calcification in adult corals, but reportedly had no effect on zooxanthellae density or their protein production (Muscatine et al, 1989). In a study in the northern Gulf of Aqaba, Red Sea, a 50% increase in nutrients resulted in 50% coral mortality from benthic algal blooms, 3–4 fold reduced reef calcification and 50% increased P/R ratio (Loya et al, 2004).

Table 7.1: The mean concentration of inorganic nutrients (NH^+_4, NO^-_3, NO^-_2, SiO_3 and PO^{3-}_4) in the Red Sea area, a comparison of early studies with the current study results.

Variable	EEPP, 2003.	GEF, 1997.	Rasheed et al, 2002.	Current study.
NO^-_3 (µmol/l)	0.69-0.34	12.76 - 59.18		0.2-0.06
NO^-_2 (µmol/l)	2.53-0.55	0.083 - 0.792		0.52-0.4
NH^+_4 (µmol/l)	13.2-28.8			3.56-0.96
Total Inorganic nitrogen			0.09-0.08	
PO^{3-}_4 (µmol/l)	1.37-1.45	0.191 - 4.140	0.01-0.01	0.28-0.04
SiO_3 (µmol/l)	0.61-0.69		0.14-0.10	1.64-0.25

The present study indicates various degrees of positive correlation between the nutrients measured and sedimentation rates, SPM and turbidity. The sites with highest nutrients have the highest sedimentation rate, SPM, non-carbonate sediment and turbidity. Edinger (1991) suggested that nutrient enhancement can increase coral growth rates up to a certain critical level, after which eutrophication becomes deleterious and growth rates decline. In a study of coral partial mortality, Wielgus et al. (2004) found that sites exposed to average total organic nitrogen (TON) between 0.4 and 0.6lm had significantly lower live coral cover and a higher abundance of colonies showing partial mortality than at sites exposed to lower TON levels. Whilst levels of nutrients determined in the present study fall within the ranges of those of previous studies in the Red Sea, and are not within the ranges of what may be considered eutrophic, they do appear to have a positive link to sediment level which itself was negatively correlated to coral cover. This outcome is supported by the study of Wielgus et al. (2004) which indicated a significant impact of such low TON on coral cover and partial mortality. However, proof of correlations does not prove cause and effect, the correlation between coral cover and sedimentation rate might be a result of nutrient loading, direct physical impact or a combination of both.

SPC, DO, PH, TDS, %DO and salinity did not show any significant correlation with sedimentation rate, SPM, non-carbonate and the percentage of mud. Turbidity showed significant correlations with sedimentation rate, SPM, non-carbonate and the percentage of mud. Thus sites with high sedimentation rates had higher turbidity, but it was not essential to have high SPC, DO, PH, TDS, DO% or salinity. Sediment input from the coast was shown to increase sedimentation, SPM, turbidity and nutrient contents in seawater as a result, although it did not affect other parameters such as SPC, DO, PH, TDS, DO% and salinity. All of these parameters by the end act together to impact reef quality by one mean or another. Nutrients up to these levels of sediment input did not reach eutrophication levels but increased and continuous sediment charge are expected to increase nutrients more. However the observed levels of nutrients from the EEPP water quality report (Awad and Shabara, 2003) and GEF project (GEF, 1997) were measure at Hurghada port which is considered the most polluted site in regards of shipping and low circulation conditions.

Coral cover from this study has declined under the effect of sedimentation rate, SPM, non-carbonate sediment and turbidity. Results have shown significant negative correlations with sedimentation rate, SPM, non-carbonate sediment and turbidity. Number of live colonies had a significant positive correlation with mud and non-carbonate sediment, dead colonies and number of recruits had significant negative correlation with percentage gravel. Number of recruits had a significant positive correlation with percentage of mud and non carbonate. No proof of significant correlations between dead corals and sedimentation rate or SPM was found. It could be suggested that the current rate of sedimentation may have led to the observed low coral cover through inhibiting coral growth, but not by upsetting the number of live, dead or recruited colonies. Abundance of coral colonies showed a significant positive correlation with percentage of mud and no correlation with sedimentation rate, SPM or turbidity. Sedimentation rate may not be the only factor accountable for the number of dead corals or recruits recorded in the current study. Especially in light of the fact that most sedimentation rates observed were well below the estimated survival threshold rate of $10mg.cm^{-2}.day^{-1}$ (Rogers, 1990; McClanahan & Obura, 1997; Dikou & Van Woesik, 2006). Dead coral colonies ranged from 1.4 to 0.6 $colonies.m^{-2}$ and recruits ranged from 12.9 to 0.7$colonies.m^{-2}$ which is indicating improving reefs. These results agreed with the earlier study by Winkler et al. (2004) who maintained that coral death through disease and bleaching may be exacerbated by high sedimentation rates but may also be influenced by other external stressors.

Species richness was also lower at sites which had higher sedimentation and SPM. Results showed significant negative correlations with sedimentation rate and SPM. These correlations support the implications that species richness is higher on reefs where sedimentation rate and SPM were low and agreed with the findings from many early studies. Early studies indicated that species diversity decreased as a result of high sedimentation rate (Loya and Slobodkin, 1971). Nugues and Roberts (2003b) maintained that sedimentation could cause reef degradation in two ways: by direct burial and smothering of corals, and by suppressing the regeneration of the adult colonies together with the settlement of new recruits through increased competition with algae. The differences in diversity between sites in the current study suggested that the growth of certain coral genera is inhibited by the high rate of sedimentation. It is possible that there is a different threshold level of sedimentation that can be endured by each genus (Sanders and Baron-Szabo, 2005). Therefore, the higher rate of sedimentation coincides with the lower number of species that are able to grow and survive and the lower the species diversity. Diversity indices in the current study showed significant differences between sites, but not between the two years 2004 and 2006. Sedimentation was considered responsible for reduce coral cover, species richness and diversity but not to affect the number of dead colonies, recruits and number of live colonies.

Sedimentation rate, SPM, non-carbonate sediment and turbidity did not reduce the levels of recruitment seen at the reefs. The general low recruitment in the present study may be attributed not only to inhibition of settlement by sedimentation but other factors such as high juvenile mortality. It has been shown that light affects both reproduction and recruitment, as coral fecundity decreases in low-light conditions, and coral larvae use light quantity and quality to choose their settlement site (Fabricius, 2005). At low light levels, corals prefer to settle on upper surfaces, where the risk of sedimentation damage is high, rather than on vertical of downward facing surfaces (Birkeland et al., 1981). Also the inter-annual variation in coral recruitment also affects recruitment rate (Miller et al, 2000). Sedimentation mortality thresholds for coral recruits were greatly lower than those for larger colonies, tens rather than hundreds of $mg.cm^{-2}$ (Fabricius et al., 2003). Results of the current study suggests that low recruit in this study resulted from the effects of both turbidity shading and sedimentation smothering. Early studies referred to sedimentation as one factor responsible for reduced coral recruitment (Birkeland, 1977, Bak & Engel, 1979, Birkeland et al. 1981, Rogers et al. 1984, Rogers, 1990). Miller et al (2000) suggested that the increased exposure to waves and coarser particle size causes greater abrasion stress to juveniles, resulting in a low recruitment number. Hixon (1986) noted that damselfish, which defend algal territories, directly influence coral recruitment and growth. Although site 6 experienced one of the highest sedimentation rates and SPM levels: it showed a striking observation, the largest mean number of recruits ($12.9.m^{-2}$). However, the replacement of the natural reef with concrete blocks has resulted in low percentages of both live coral cover and dead coral cover. It appears, therefore, that recruits can successfully settle onto the new artificial reefs despite the high sedimentation rate, SPM and turbidity but they did not survive for long time.

Results from the settlement study indicated a general low larval settlement in all sites after one and three years of tile deployment. The steel mesh rack method used in this study is considered the most common method used for recruitment studies (reviewed in Mundy, 2000). Variations in topographic complexity between tiles and natural reefs which could affect larval supply and the settlement process were taken into consideration in choosing tile size and fixing angle. Many studies have suggested that a conditioning period of 4 to 12 months is required prior of coral recruitment onto newly deployed substrata (Sammarco, 1982; Bailey-Brock, 1989; Wendt et al., 1989), but the current results did not

indicate important differences after this period. Sedimentation rate, SPM and turbidity did not reduce settlement while it did appear to inhibit successful recruitment survival for a long time.

Sedimentation rate, SPM, turbidity, and non-carbonate sediment did not impact on the percentage of *r-strategists* species. There were no significant differences between sites in the percentage of *r-strategists* species, and no significant correlations between *r-strategists* and sedimentation rate, SPM, non-carbonate, turbidity, gravel, sand or mud. The percentage of *r-strategists* varied from 34% at site 1 to 82.5% at site 3. The highest percentage was found at site 3 which had the lowest coral cover and diversity and the highest sedimentation, the second highest site was site 5 which had relatively high coral cover but not diversity and had low sedimentation. Thus there was no assumed effect of sedimentation indicated on *r-strategist* percentage at low sedimentation rates as in site 5 and these species occupied sites at no preference of sedimentation levels. At higher sedimentation levels as at site 3 total coral cover decreased with particular reduction in *k* and *t-strategists* in favor of *r-strategist* which were more tolerant to these levels of sedimentation. The *r-strategists* included all species of the genera *Pocillopora, Stylophora, Psammocora, Seriatopora* and many species of *Montipora, Acropora* and *Pavona*. The *k-strategists* include some species of *Acropora, Pavona, Hydronophora, Galaxea* and *Goniopora,* while the third group (*t-strategist)* includes the rest of coral groups (reviewed in Sorokin, 1995).

The Deterioration Index (DI) was not influenced by sedimentation rate, SPM or turbidity. However, the current study showed significant negative correlations between DI and the percentage of non-carbonate sediment and mud. These results indicated no effect of sedimentation and SPM on reef deterioration. In this study, site 3 was ranked the highest for DI, while site 6 was the lowest. Whilst site 3 had the highest DI, which concurred with the highest sedimentation rate and SPM at this site, site 6 had the lowest DI level although it was the second highest site in sedimentation rate. Interpretation of the results indicated that site 6 was the best site using this index which was not true, however, as most of the recruits were found to die at early stage. Furthermore, whilst site 6 had the lowest DI this deviated from expectation based on the other health indicators measured during this study. Site 6 was poor in coral cover, abundance, species richness and diversity index. As noted from this study, the low DI value at site 6 emerged from the high number of recruits at the site compared with the low number of dead colonies. However most of those recruits do not endure for a long time and die at an early stage an indicated from the 2006 survey, which showed a reduced number of large colonies. Deterioration Index (DI) was shown to be a more sensitive reef health parameter in contrast to other reef health parameters such as mortality rates, abundance, species richness, and species diversity (Ben-Tzvi et al. 2004).

At high mortality and/or low recruitment the DI value will be high. Conversely, if recruitment exceeds mortality, DI will be small, indicating recovery or improvement of the reef. As noted previously, if the number of recruits is greater than the number of dead colonies this may indicate that the reef is improving. However if the recruits do not survive for a long time and die under sedimentation or any other stress, this may give a false impression of the reef's health status. This is the case at site 6; the live coral cover was low whilst it also had the highest recruitment rates (13.3-12.9 colonies per m^2). At site 4 the situation was of a typically improving reef after physical destruction. It had a relatively high number of new recruits (2.9-3.55 colonies per m^2) if compared with coral cover (17.8-21.5) and dead colonies (0.65-0.85 colonies per m^2). In contrast to results of Ben-Tzvi et al. (2004) the Deterioration Index was not sensitive enough to explain the

status of reefs similar to Holidays site.

Disturbance seems to be increased under the effect of sedimentation rate, SPM, turbidity, and non-carbonate sediment. Study results indicated significant negative correlations between mean colony size of branched coral and sedimentation rate, SPM, turbidity, non-carbonate sediment and significant positive correlation with sand. The smallest mean colony size was found at sites 6 and 3 which had the highest sedimentation and SPM, whilst the largest colony size was recorded at site 1 and 5 which are low in sedimentation and SPM. The results agreed with an earlier study which maintained that the living surface area of a coral colony or colony size can be used as an integrated measurement for disturbance intensity and frequency (Connell, 1978; Done, 1992). Branched coral diameter was inversely related to sedimentation rate, suggesting that higher rates of sedimentation inhibit colony growth in branched corals. Massive coral diameter was, however, unrelated to sedimentation rate, indicating that rate of sedimentation does not inhibit growth of massive colonies. Transplanted coral survival of the three species used in this study did not show any significant correlations with sedimentation rate, SPM, non-carbonate sediment, turbidity or percentage of gravel, sand or mud. These results contradicted the results of study by Ammar et al. (2000) which suggested a lethal effect of sediment on transplanted corals results in high mortality at high sedimentation rates. Sedimentation rate, SPM and turbidity did not prove to affect DI or transplanted coral survival while they reduced coral cover, species richness, diversity and mean colony size of branched corals.

Algal feeding fish abundance was reduced at sites which had high sedimentation rates, SPM, non-carbonate sediment and turbidity. A site which has high sedimentation rate and low coral cover has the lowest coral feeding fish abundance. The fish abundance results from this study indicated that algal feeders had significant negative correlations with sedimentation rate, SPM, non-carbonate sediment and turbidity; and a significant positive correlation with percentage of sand in bottom sediment. Algal feeder abundance was inversely related to sedimentation rate, suggesting that higher rates of sedimentation, SPM, non-carbonate sediment and turbidity inhibit algal feeder colonization. While coral feeding abundance was not affected by sedimentation rate, SPM, non-carbonate sediment and turbidity, indicating that sedimentation had no effect on coral feeding distribution. Although, the results indicated that the lowest abundance of both coral feeder and algal feeder fish were recorded at sites 6 and 3 which had the lowest coral cover, diversity and species richness and the highest sedimentation rate. The outcome of the current study agreed with the early study of Lewis (1997) which revealed that experimental coral disturbance led to a significant decline in fish species richness on the Great Barrier Reef. However, disturbance by dredging in Moorea and Tahiti has led to a decrease in fish abundance at some sites, whereas other reefs showed no significant difference (Harmelin-Vivien, 1992). This supports the non significant correlation between coral feeding fish abundance and sedimentation rate, SPM and turbidity. It has also been found that there were more species and individuals of fish on a living reef than on a primarily dead reef, which had low structural relief; also a further decline in fish abundance and diversity was indicated as the dead reef collapsed into rubble (Rogers, 1990).

The current study suggested that both fish groups declined at sites where low coral cover and species richness were recorded; and indicate that this low coral cover and species richness push fish to leave the area. There is a threshold level of reef deterioration at which fish begin to leave, perhaps related to the decrease in both abundance and diversity of the coral on which they are feeding. An earlier study showed that disturbance of the coral reefs led to changes in the fish community through reduction of total fish abundance by 50%, decreases abundance of invertebrate and fish-feeders, increased abundance of

herbivorous, detritivorous, and relative abundance of planktivorous fishes (Khalaf and Kochzius, 2002). The highest abundance of both groups of fish was found at site 2 for coral feeders and site 7 for algal feeders, those sites have high coral cover, species richness and low sedimentation rate. The least damsel fish (*Pomacentridae)* abundance were found at sites 6 and 3, the same sites which have the lowest coral cover and diversity. This result supported by a previous study by Lirman (2001) which showed that reduced herbivorous damselfish and eutrophication could result in an increased rate of coral algal interaction and negatively affects coral growth and survival. Also it agreed with the results of Thacker et al. (2001) who stated that exclusion of herbivorous fish plays a greater role in algal biomass changes than nutrient enrichment, which indirectly affects the coral's ability to compete. It is suggested that increased sedimentation rate, SPM and turbidity decline coral cover, diversity, species richness, mean colony size and fish abundance, and at higher levels it increases nutrients and deterioration levels. The resulted decline in fish abundance contribute again to reef decline through the reduction in coral cover as maintained by Lirman (2001) and Thacker et al. (2001).

Similarity tests between sites according to fish abundance of the 6 groups showed that site 3 and 6 has the highest similarity in the two years with the lowest fish abundance, followed by site 5 and 7 in 2004 and 5 and 4 in 2006. The lowest similarity was found between site 2 and 7 in the first year and between 1 and 6 for 2006. Sites 3 and 6 were also the highest in sedimentation rate, SPM and turbidity indicating the indirect effects of these parameters in fish abundance of all studied groups.

Abundance of butterflyfish (chaetodontids) indicated a significant positive correlation with coral cover, but no significant correlation with species richness, diversity index or deterioration index. Sites which have high coral cover also have higher abundance of butterflyfish. This result agreed with the study by Lewis (1997) who referred to the family Chaetodontidae as the only family from 6 major taxonomic families: the Serranidae, Pomacentridae, Chaetodontidae, Apogonidae, Labridae, and Scaridae which showed a significant reduction in abundance after coral disturbance. It is suggested that increased sedimentation rate, SPM and turbidity decline coral cover, diversity, species richness, mean colony size which are finally reduce butterflyfish fish abundance.

The outcome from the macroborers study showed that macroborer community at all sites was dominated by sipunculid worms, which showed the highest bore number per unit area of reefs. However, the study of macroborers in some massive coral species at Discovery Bay, Jamica, carried by Macdonald and Perry (2003) indicated that sponges dominated macroboring communities in low sedimentation sites. Therefore, in contrast to the early study in the Caribbean reefs, Hurghada reefs were dominated by sipunculids at all sites to various degrees, sponge were the next most dominant group followed by polychaetes and bivalves with the lowest abundance. Hutchings and Peyrot-Clausade (2002) reported that polychaetes were the primary invaders followed by sipunculids, which dominated the reef after a short time, then sponges and bivalves colonized the reef and constituted, with the sipunculids, the major part of the bioeroding fauna. Zubia and Peyrot-Clausade (2001) maintain that sipunculid are deposit feeders and may feed on small sediment particles trapped in coral rubble, which may explain the dominance of this group in all study sites in this study. There was a significant difference in the abundance of all macroborers between sites.

Sedimentation rate, SPM, non-carbonate sediment and turbidity did not increase the percentage of polychaetes and bivalve macroborers. However they reduced sponge and increased sipunculids macroborers, which imply that sipunculids abundance was higher on reefs where sedimentation rate was high and the contrary for sponges. An earlier study by

Hutchings and Peyrot-Clausade (2002) indicated that decreasing water quality and increasing sediment load changed the structure and composition of the initial macroboring community and lead to an enhanced bioerosion rates. In contrast to the outcome of Holmes et al. (2000) who maintained that coral rubble bioerosion is sensitive to low levels of eutrophication and sedimentation stress and provides a valuable indicator of eutrophication stress on coral reefs, polychaetes and bivalves macroborers did not show sensitivity to sedimentation or SPM changes in this study. Sponges showed significant positive correlations with percentage of sand and negative correlation with SPM in seawater. Compared to a study by Holmes (1997) which indicated increase abundance of bioeroders, especially boeroding sponges, along an eutrophication and SPM gradient, the current study indicated increase sponge abundance with decrease SPM. The current results agreed with the outcome of a study by Perry (1998) which found that sponges dominated the fore-reef boring community (low sedimentation sites) rather than the diverse assemblage of sponges, worms and bivalves in the back-reef where sedimentation was high.

The mucus study showed that sedimentation rate, SPM and turbidity significantly increased mucus secretion by corals. These results agreed with finding of a study by Richman et al. (1975) who found that corals use a mucus secretion mechanism to prevent burial in heavy sedimentation reefs. Massive corals showed higher rates of mucus production than branched in this study, which agreed with the study by Loya (1972) which maintained that branching corals withstand high sediment levels because they are less likely to become buried, and massive corals overcome heavy sedimentation by high mucus production. Kloeppel (2001) referred to environmental stresses as a major factor that forces corals to secrete more mucus to coat their outer tissues. It was found that the dominant genus of hard corals (Acropora) in the Great Barrier Reef exudes up to 4.8 litres of mucus per square metre of reef area per day, between 56% and 80% of this mucus dissolves in the reef water (Wild et al, 2004). Corals at high sedimentation rates (site 3 and 6) may expend much of their energy as mucus to remove sediment, which reduce the coral's ability to grow and reproduce and finally decrease reef development and growth.

Zooxanthellae density in coral tissues of the transplanted coral *Acropora tenuis* showed significant reduction under the effect of sedimentation, SPM, non carbonate and mud percentage. This agreed with the results from previous studies, which had found sediment load to increases the loss of zooxanthellae and is considered to be detrimental to corals (e.g. Bak, 1978; Abdel-Salam et al, 1988; Hodgson, 1990; Philipp and Fabricius, 2003). Results from the laboratory study of zooxanthellae density in the three coral species; *Acropora tenuis, Stylophora pistillata* and *Pocillopora damicornis*; indicated significant reduction in zooxanthellae density with increasing sedimentation rate and increasing period of exposure to sedimentation. The results suggested that sedimentation lowered zooxanthellae density in coral colonies at all sedimentation levels and prolonged exposure to sedimentation also reduced zooxanthellae density in coral tissue as indicated by correlation with the periods of exposure. Corals under the observed sedimentation rates seemed to experience zooxanthellae loss, increase in mucus production and bioerosion, reduction in recruitment, survival and fish abundance. These entire factors act together to reduce coral cover, diversity, species richness, mean colony size and increase deterioration levels at sites which undergo high sedimentation levels. Such processes as zooxanthellae loss followed by reduction in coral cover and diversity could take a long time and the current recorded levels can be a result of earlier sedimentation and SPM impacts. However the current sedimentation, SPM and turbidity still has significant correlations with these parameters.

Feeding experiments showed that coral species *Lobophyllia hemprichii* actively feed on sediment and displayed a reasonable ability to reject low quality sterile sediment as indicated from the difference between fresh and sterile sediment treatments. The results agreed with the outcome of a study carried out by Rosenfeld et al. (1999) which indicated that active ingestion of the organic matter in the sediment had occurred. This result is assumed to represent the pattern of heterotrophic ability of coral species that dominate high turbidity and sedimentation habitat. The results also indicated that there was an increase in sediment uptake by increasing sedimentation up to 20mg.l^{-1} sediment. The outcome of this part of study suggested that long-term high sedimentation levels in these sites could allow some corals species to prevail over others, because of their ability to assimilate sediment. Species which have a low tolerance to sedimentation and are less able to feed on sediment, are expected to be dominated and replaced by ones which showed a strong ability to feed on and derive organic matter from sediment. Change of the coral community in favor of these species could be the most probable result of long-term sedimentation stress. In comparison to a study by (Ayling and Ayling, 1991) who reported high coral cover and diversity in some reef lagoons despite high turbidity and sedimentation levels, which were considered lethal for corals; the current study revealed a great reduction in coral cover and diversity at high sedimentation sites. In the long term, some coral species might develop a higher feeding capability on sediment and maintain high coral cover and diversity. One early study developed the same view and suggested that local adaptation to sedimentation had occurred and corals in turbid environment have a greater capacity to feed on suspended sediment than analogous corals from oligotrophic environments (Anthony, 2000).

Summary

The main mechanisms by which sediment is hypothesized to detrimentally impact coral reefs are: cause their death by smothering or burial; decrease adult coral growth by abrasion and shading; depress zooxanthellae densities and photosynthetic activity, increase respiration and mucus production; reduce coral reproduction, coral larval settlement, and early survival (Rogers, 1990). The results of this study highlight those parameters that are related to, and potentially affected by, the rate of sedimentation and other environmental parameters such as SPM, turbidity and percentage of non-carbonate sediment.

The findings of this study with respect to these propositions were:

Sedimentation and SPM seems to reduce coral cover as indicated from the negative correlations with coral percentage cover.

The current sedimentation levels did not appear to cause coral death across the study area since no significant correlations with percentage of dead colonies were found. A more likely cause of death is thought to be the large-scale physical disturbances such as reef filling and dredging.

Sedimentation appeared to reduce coral growth and diversity as indicated by the significant negative correlation with species richness, diversity index and mean branched colony size (disturbance).

Sedimentation, SPM and turbidity did not appear to reduce coral abundance, percentage of *r-strategists*, Deterioration Index, transplanted coral survival or massive colony size

(disturbance), in a significant fashion.

Sedimentation was found to reduce zooxanthellae densities as indicated by the significant negative correlation with zooxanthellae cell density in the field and laboratory.

Sedimentation, SPM and turbidity were found to increase coral mucus production as indicated by the significant positive correlation with mucus production rate.

Sedimentation did not appear to prevent the settlement of coral larvae; in fact, high recruits were counted on sites where sedimentation rate was high (site 6) and no evident of significant correlation with the number of coral recruits.

Sedimentation did not appear to reduce the abundance of coral feeding fish, although it strongly reduces algal feeding fish abundance, significant negative correlations were shown with sedimentation rate, SPM and turbidity.

Sedimentation appeared to enhance sipunculid macroborer abundance and reduces sponge abundance as indicated by the significant correlations. It did not, however, affect the distribution of the other groups of macroborers, polychaetes or bivalves.

During the period of the study the sedimentation rate had increased at six sites (1, 3, 4, 5, 6 and 7) in 2006, and only one of the seven sites (2) had a lower sedimentation rate than in 2004. SPM had increased in four of the study sites (3, 5, 6 and 7) and decreased at three sites (1, 2 and 4). The percentage of non-carbonate sediment increased in two of the seven study sites (1 and 7) and decreased at five sites (2, 3, 4, 5 and 6). Coral cover, abundance and species richness had increased at five sites (1, 2, 4, 5 and 7) and decreased in two sites (3 and 6). The Diversity index increased at 4 sites (1, 4, 5, 7), and had not changed at two sites (2 and 3) and had decreased at one site (6). Deterioration Index of the reefs increased at three sites (3, 4 and 6) and decrease at four sites (1, 2, 5 and 7), indicating a continuous impact at three of the sites. Mean colony size for branched and massive corals showed an increase in six sites (1, 2, 4, 5, 6 and 7) and decrease in one site (3). Abundance of coral feeding fish increased at four sites (1, 2, 5 and 7) and decreased at three sites (3, 4 and 6), while algal feeders increased at two sites (1 and 5) and decreased at five sites (2, 3, 4, 6 and 7).

In general, the results of this study indicated that there are some sites around the city of Hurghada that are still undergoing deterioration at various levels due at least in part to sedimentation. This was indicated from the health indicators used, although some parameters did not show significant correlations with sedimentation rate and SPM levels. Some other sites (such as site 4) seem to have suffered great deterioration in the past but have shown natural recovery coinciding with reduced sedimentation and SPM levels. Other sites (e.g. site 6 and 3) showed continuous deterioration and need human intervention to reduce sediment input in both sites and to begin restoration of the reefs in the site 3 whilst site 6 can be restored naturally as indicated from the high recruitment rates. A restoration project using coral transplantation is most likely required for restoration of site 3 and 4 whilst site 6 is assumed to be able to restore itself naturally if sediment input is reduced. The high survival of transplanted corals and low deterioration level of the reef at site 4 raised its biological rank although it had low coral cover and diversity. The high larval settlement at site 6 raised its biological rank and increase corals abundance and reduces Deterioration Index.

Site 3 represented the most chronic sedimentation impacted site in the study area. The high sedimentation and SPM in this site come from two sources; continuous erosion of

artificial beaches by currents and waves and from the early sediment input deposited in the bottom of the coastal shallow lagoons which is stirring up and maintains the high turbidity, SPM and sedimentation levels.

In general, the study sites could be arranged from the best to the worst according biological quality as determined from the indicators used in this study, as follow; NIOF (1), S.Hashish (7), A.Sadaf (2), Arabia (4), A.Minkar (5), Holidays (6) and Shedwan (3). Also they could be arranged according to sedimentation rate from the lowest to the highest as follow; S.Hashish (7), A.Minkar (5), NIOF (1), A.Sadaf (2), Arabia (4), Holidays (6) and Shedwan (3).

Conservation and restoration of coral reefs needs further scientific research and monitoring, developing effective management-tools, and appropriate measures for conservation and sustainable use of coral reefs and to stop further coral degradation. Actions must be taken over the longer term to reduce human induced climate change by reducing green-house gases, reduce immediate threats of declining water quality because of land-use changes and pollution, and mass exploitation of fish and corals. Coral transplantation is one tool used for reef rehabilitation to restrain the increasing degradation of coral reefs. The ICRS 2004 declaration recommended four key strategies for coral reef conservation; achieve sustainable fishery on coral reefs, increase effective marine protected areas on coral reefs, recover land-use change impacts, and develop technology for coral reef restoration (ICRS, 2004).

Critique and Suggestions for further work

The Red Sea coast extends for long distance and support a diverse ecosystem which interact with various human activities in the area. Based on the outcome from this study, many areas of research need more investigation in more detail and in the long term. Other new techniques need to be applied, also some study areas could be done better, these include:

- Sedimentation study needs investigation of the water current in each site and seasonal patterns which could explain various sedimentation regimes in the area.
- Data about the amount of sediment entering the coastal area every year or month, such as through hotel beach re-nourishment, will be useful to estimate the amount sediment and soil entering the sea.
- New sites which are expected to undergo coastal development in the near future should be included in any future study.
- Sediment re-suspension may be enhanced by diving, snorkelling and fishing activities, data about these parameters is important in some sites and could have a significant role in shaping the sedimentation regime in these areas.
- Transplantation could be done by using many species to test the ability of different species to survive under these conditions and using different techniques for fixing transplants to save effort and cover a larger area.
- Transplantation also needs to cover a larger area and include greater numbers of transplanted colonies and be investigated for longer time.
- Coral spawning in the area is one subject which needs to be explored in much greater detail. It could be depressed by sedimentation and turbidity and ultimately reduce the available larvae for settlement.

- Recruitment and settlement could be tested using various materials of various degree of preference for coral larvae such as steel, rubber and stones. More sites need to be included in this study to test a wider range of larvae source and current directions.
- Coral growth rates are an important factor that could be included in any future study of this area; it can give a clear signal in the long term changes in any area.
- The relative proportion of *r-strategists* pioneer coral groups need to be compared to many other sites free of disturbance and to be identified to species level. This may help to identify the more resistant species.
- Some of the indicators of this study could be impacted by environmental conditions that occurred a long time ago, so future measurements are required that could be most correlated to the current conditions. Examples are species richness and disturbance levels.
- Mucus secretion could vary between species and should not be compare different species, although it is not recommended by earlier studies. It is suggested here that species need to be identified for this study and then compared separately between sites.

Data from this study is considered to provide baseline information about the area for any further study of sedimentation, suspended matter, turbidity, bottom sediment composition and other parameters measured. Future research in the area can use reef health parameters in this study to compare and define ecosystem trends. The outcome of this study will support environmental managers in applying more restraint to refilling permits to the hotels and beaches in the area. It will also encourage the decision makers to take more steps towards reducing and stopping coastal modifications in the sites under construction. For the stakeholders and the users it will clarify the consequences of the abuse of marine resources and the fate of their investment are in no doubt unsustainable.

CHAPTER 8

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