

Marine pollution and coral reefs

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Abstract

Coral reefs are exposed to many anthropogenic stresses increasing in impact and range, both on local and regional scales. The main ones discussed here are nutrient enrichment, sewage disposal, sedimentation, oil-related pollution, metals and thermal pollution. The stress comprising the main topic of this article, eutrophication, is examined from the point of view of its physiological and ecological mechanisms of action, on a number of levels. Nutrient enrichment can introduce an imbalance in the exchange of nutrients between the zooxanthellae and the host coral, it reduces light penetration to the reef due to nutrient-stimulated phytoplankton growth, and, most harmful of all, may bring about proliferation of seaweeds. The latter rapidly outgrow, smother and eventually replace, the slow-growing coral reef, adapted to cope with the low nutrient concentrations typical in tropical seas.

Eutrophication seldom takes place by itself. Sewage disposal invariably results in nutrient enrichment, but it also enriches the water with organic matter which stimulates proliferation of oxygen-consuming microbes. These may kill corals and other reef organisms, either directly by anoxia, or by related hydrogen sulfide production. Increased sediment deposition is in many cases associated with other human activities leading to eutrophication, such as deforestation and topsoil erosion.

Realistically achievable goals to ensure conservation, and in some instances, rehabilitation of coral reefs are listed.

Keywords: coral reefs, eutrophication, pollution, sedimentation, sewage, zooxanthellae

Introduction

On a global scale there is a marked and alarmingly accelerated deterioration in the status of coral reefs (Wilkinson 1993, 1996; Wilkinson & Buddemeier 1994). This has led to concentrated efforts at national and international scales to evaluate the extent of the phenomenon, and to find means and measures to mitigate the results, and, possibly, to reverse the trend. The most high-profile of these has been the establishment, following the 1992 Earth Summit, of The United States Coral Reef Initiative, along with other such initiatives in Australia, Japan, Jamaica, the UK and France. The overall aim of these actions is the creation of a global partnership for the conservation and restoration of coral reefs at local levels (Crosby & Maragos 1995; Crosby *et al.* 1995). Wilkinson (1993; also see Wilkinson 1996) estimates that 30% of all coral reefs have reached the no-return, critical stage, another 30% are seriously threatened, and only less than 40% are stable and safe. However, upon careful examination of the global distribution of these different

reef categories, it is evident that the main factor in their demise is human pressure, to which virtually all reefs found in proximity to densely populated land are exposed. Almost all reefs which are considered in stable condition, at least for the foreseeable future, are found in remote areas and, as such, are less exposed to the different anthropogenic stresses discussed above. Over the last century there seems to have been virtually no global-scale increase in oceanic productivity (Falkowski & Wilson 1992), which would have been evident in the case of significant overall eutrophication. Therefore, it seems safe to conclude that so far, human impacts on reefs are on the local and regional, rather than global scale. Of course this statement does not include the two obvious exceptions to it, namely the release of different greenhouse gases leading to global warming, and the depletion of ozone causing increases in UV exposure of shallow-reef biota. These global aspects are the subject of other papers in this Special Issue (Glynn 1996; Shick & Jokiel 1996).

In this review we aim to describe the major anthropogenic inputs into reef ecosystems, other than atmospheric

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CO₂ (which is the topic of a separate paper in this issue) and to discuss the evidence for their damage, with special emphasis on their mechanisms of action.

As defined by the UNESCO Group of Experts on the Scientific Aspects of Marine Pollution (GESAMP), as cited by Grigg & Dollar (1990), 'pollution', in the marine domain includes 'the introduction by man, directly or indirectly, of substances or energy into the marine environment resulting in such deleterious effects as harm to living resources, hazards to human health, hindrance to marine activities including fishing, impairment of quality for use of sea water and reduction of amenities'. However, it is noteworthy that Grigg & Dollar (1990) themselves chose the following stricter definition, 'pollution might be considered any factor that causes man-induced mortality to be greater than zero.'

The *Report of the UNEP-IOC-ASPEI-IUCN Global Task Team on the Implications of Climate Change on Coral Reefs, Global Climate Change and Coral reefs: Implications for People and Reefs* (1994), prepared by C. R. Wilkinson and R. W. Buddemeier, states, in spite of its explicit original mission to evaluate the dangers posed to coral reefs by global climate change: 'Coral reefs are being seriously and increasingly stressed by exploitation and anthropogenic environmental changes, such as sedimentation, nutrient loading and pollution, physical destruction, and overfishing. These effects are distinct from, and unrelated to, climate change.' They further underscore: 'Our major finding is that human pressures pose a far greater immediate threat to coral reefs than climate change, which may only threaten reefs in the distant future.'

Eutrophication

Definition

Webster's Random House Dictionary defines eutrophication (of a lake) as 'characterized by an abundant accumulation of nutrients that support a dense growth of algae and other organisms, the decay of which depletes the shallow waters of oxygen in summer'. In more general terms, we should consider any increase, natural or anthropogenic, in nutrient concentrations to levels above their long-term averages as eutrophication. Of course, areas of seasonal upwelling are 'naturally eutrophied' for half of the year, but coral reefs seldom thrive in such localities. In this context we usually consider as nutrients any inorganic substances, compounds, elements, and ions, which are essential for plant growth. Furthermore, normally the lack of any one such compound will harm the plant, and is bound to interfere with its normal growth and development, and prevent it from completing parts of its life cycle. Such harm will begin to be manifest after the internal reserve, or cell

quota, of any such essential nutrient within the plant, or in our case the algal cell, is exhausted (Droop 1983). As we shall see below, corals, having presumably evolved in oligotrophic waters, manage to cope with low nutrient levels.

Although at some minute level many elements are indispensable, in most instances those elements which are required only in minute amounts, the micronutrients, are found at sufficient levels. The elements needed in more substantial quantities, the macronutrients, are either structural components of plant biomass, or are a part of essential molecules like abundant enzymes or pigments. In the open sea the most common growth-limiting nutrients are, in that order, nitrogen (D'Elia & Wiebe 1990 Smith 1984), phosphorus (D'Elia & Wiebe 1990; Smith 1984), carbon (Weis *et al.* 1992) and iron (Martin 1992; Cullen 1995), although in coastal zones, next to carbonate reefs, phosphorus may become limiting before nitrogen. All fertile oceanic regions other than reefs, are characterized by abundance of all necessary nutrients. These nutrients, even in the inclement Arctic and Antarctic weather lead to proliferation of phytoplankton, and the resulting rich fisheries, ultimately feeding upon them.

Corals, nutrients and eutrophication

In the tropical oceans, roughly between 30° N, and 30° S, year-round elevated surface temperatures ranging between 18 and 32 °C result in permanent thermal stratification of the water column. Under these conditions, the upper, illuminated, warm strata of the ocean become depleted of nutrients, through the continuous sinking of dead organisms to the cold, dark, heavy bottom waters, where they become released by the mineralizing activities of decomposing microbes. These nutrients are not available for phytoplankton growth in the surface waters of the barren 'blue deserts' of tropical seas. It is in the coastal margins of these nutrient-poor, or oligotrophic waters that coral reefs have evolved and can thrive (Muscatine & Porter 1977). Only in instances where these nutrients are brought up to the surface, in areas of upwelling, they form islands of high phytoplankton productivity, which support rich fishing grounds, but exclude coral reefs.

Virtually all reef-building, or hermatypic (Schumacher & Zibrowius 1985) corals, as well as numerous other tropical invertebrates harbour symbiotic microalgae in their cells (as is the case in corals), or in their tissues (giant clams). These unicellular dinoflagellate symbionts are known since their discovery by Brandt (1883) as zooxanthellae. It has been suggested (e.g. see review by Muscatine 1990) that they may have co-evolved with their hosts as a way of coping with the nutrient-poor waters which they monopolize, possibly while taking

ecological advantage of the calcification potential enhanced by the photosynthesis of the symbionts (Barnes & Chalker 1990). In their mutualistic symbiosis the zooxanthellae contribute their photosynthetic capability as the basis for meeting the needs for metabolic energy of the whole association, and eventually of a great part of the entire reef ecosystem. As a by product of their photosynthesis, they also provide oxygen to the hosts' tissues (Harland and Davies, in press). In return, the zooxanthellae gain access to the nutrient-rich metabolic wastes of their animal host. These include the products of the digestion of zooplankton, consumed by all corals, as well as respiratory CO₂ (Harland and Davies, in press).

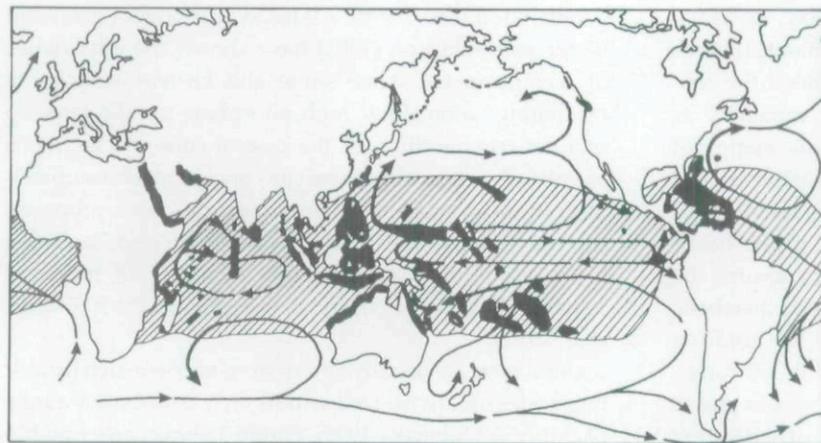
Zooxanthellae also have the capability of extracting from the water inorganic nutrients such as nitrate (D'Elia & Webb 1977; Webb & Wiebe 1978), ammonium (Muscatine & D'Elia 1978; Rahav *et al.* 1989) and phosphate (Jackson & Yellowlees 1990; Dubinsky *et al.* 1990). Most authors attribute the capability of assimilating inorganic nitrogen from seawater to the zooxanthellae alone (Rahav *et al.* 1989; Falkowski *et al.* 1993). Nevertheless, since enzymes involved in the assimilation of ammonium such as glutamate synthetase (GS) and glutamate dehydrogenase (GDH) were found in some coral animal tissues, the possibility of host involvement in ammonium acquisition has also been raised (Yellowlees *et al.* 1991). The correlation between the distribution of reefs and low nutrient levels has led to an almost generally accepted dogma, that the algal-coral association conserves and recycles nutrients under steady-state conditions, utilizing inorganic nutrient inputs, usually from large reservoirs of low concentration (D'Elia & Wiebe 1990). Kinsey (1991) states that the levels of primary production on reefs 'are as high or perhaps higher than any reported for other natural ecosystems', however, almost all of it is consumed by community respiration. This author estimates net new production at 3% of gross primary production, amounting to $\approx 0.1 \text{ g C m}^{-2} \text{ d}^{-1}$, a value which 'happens to be extraordinarily similar to the typical net production of the oligotrophic oceans' surrounding these reefs. Since reefs are not likely to obtain the nutrients needed to support that new production by advective input from below, a number of hypotheses were offered to account for it. Smith (1988) pointed out that due to their extreme Redfield (1934) ratios C:N:P 550:30:1 (Atkinson & Smith 1983), reefs can assimilate far more CO₂ for the same amount of nutrients, or, conversely, assimilate comparable carbon provided with only a fraction of the nutrients which would have been required by oceanic phytoplankton for equivalent fixation. Furthermore, since the oceanic waters reaching the reef are N depleted and, as such, contain excess P, they will support nitrogen fixation in the reef (Smith 1988; Kinsey 1991). An additional, possibly underestimated source of nutrients is predation on

zooplankton (Erez 1990). Atkinson and Bilger (1992) and Bilger and Atkinson (1992) have shown the importance of reef geometry, wave surge and current velocity in facilitating 'anomalous' high phosphate uptake rates by reef flat communities. In the case of some Pacific atolls of volcanic origin, the ingenious concept of 'endo-upwelling' of deep, nutrient laden waters has been proposed (Rougerie & Wauthy 1993) — and criticized, as being insignificant in relation to the quantities of nutrients required to support observed reef growth rates (Tribble *et al.* 1994).

Coral reefs are usually absent from nutrient-rich upwelling zones otherwise well within their distribution range (Achtuv & Dubinsky 1990). Figure 1 shows areas within the latitudinal range otherwise adequate for coral reef development, from where reefs are absent. At least in some cases that distribution may be the result of upwelling-induced high-nutrient concentrations. Sheppard *et al.* (1992) illustrate that trend in a number of localities in the Red Sea, and the Persian, Oman and Aden Gulf areas. However, one should remember that this interpretation is confounded by the low temperatures of the upwelling waters. This zoogeographic distribution pattern, damage to reefs following discharge, and their subsequent recovery after diversion of sewage (Grigg & Dollar 1990), and controlled field and laboratory experiments (Muscatine *et al.* 1989; Dubinsky *et al.* 1990; Stambler *et al.* 1991; Falkowski *et al.* 1993; Jokiel *et al.* 1994) led to the consensus that corals are adversely affected by elevated nutrient levels. However, there is no agreement on what should be defined as 'high' values. In the series of studies conducted at Elat, on the Gulf of Aqaba (Northern Red Sea) and at the Hawaii Institute of Marine Biology the concentrations which led to significant responses were 10–100 μM of ammonium and 1–10 μM of phosphate (Muscatine *et al.* 1989; Dubinsky *et al.* 1990; Stambler *et al.* 1994). It is noteworthy that the Red Sea corals showed a much stronger response than Hawaiian ones to similar levels of experimental nutrient enrichment. It may well be that these differences in sensitivity reflect the differences in nutrient levels between Hawaii (2 μM N, and 0.1 μM P) and Elat (< 1 μM N, and 0.01–0.02 μM P).

The remarkable vigour and high growth rates of numerous species of mostly Red Sea corals in the Monaco Marine Aquarium and the associated European Oceanographic Observatory (Dubinsky, personal observation) were achieved, according to the director J. Jaubert, by keeping nutrients at very low levels.

The general notion that reef corals are adversely affected by elevated nutrient levels was further strengthened by the putative relationship between the June 1991 eruption of Mount Pinatubo in the Philippines, the resulting cold winter which led to deeper mixing in the Gulf of Aqaba (Red Sea) resulting in an unusual 'natural



Main reef area Ocean currents



Fig. 2 Macroalgae overgrowing fringing reefs in Elat, Israel, Gulf of Aqaba, Northern Red Sea, during the 1990 winter, following the mount Pinatubo eruption in The Philippines. Photograph courtesy of Dr Nannette Furman-Chadwick, department of Life Sciences Bar-Ilan University, and The Inter-university Institute, Elat, Israel.

wintertime eutrophication' of the surface waters around the area's reefs. The ensuing proliferation of the Chlorophyte seaweed *Enteromorpha* sp. (Fig. 2) smothered some 25% of the coral colonies on these reefs (Genin *et al.* 1995)

In the face of all the considerable evidence and conjecture summarized above, stand some studies in which no adverse effects were evident at all. Grigg (1994, 1995) in studies from Hawaii, reports that treated sewage outfalls and urban non-point source nutrient and pollutant runoff had no demonstrable effect on reef communities. Meyer & chultz (1985) actually found growth enhancement of corals exposed to elevated nutrient concentrations. Meyer *et al.* (1983) points out the enhanced growth observed in corals in which there are resident fish schools, and attributes that effect to nutrients in their excretions. In a recent publication Atkinson *et al.* (1995) report that all of 57 coral species kept in well-above ambient nutrient concentrations, flourished at the Waikiki Aquarium in

Fig. 1 Oceanic currents and coral reef distribution. Dark areas show reefs, hatched ones show the 20 °C isotherm band. Reproduced by permission from: Wilkinson, C. R. and R. W. Buddemeier. Report of the UNEP-IOC-ASPEI-IUCN Global Task Team on the Implications of Climate Change on Coral Reefs: Global climate Change and Coral reefs: Implications for People and Reefs (1994). Note the predominance of reefs along the eastern margins of continents where nutrient levels are low, and their absence from the upwelling regions along the western continental shelves. In addition, reefs are thought to be restricted to the winter isotherms of 18–20 °C.

Honolulu. It may well be that the combination of high pH and elevated nutrient levels facilitated increased supply of CO₂ to meet the needs of a nutrient-stimulated zooxanthellae population. This carefully conducted study underscores the need for better understanding of the nutrient-coral relationship.

Physiological and ecological bases for deleterious effects of eutrophication on corals and coral reefs

There are three major ways by which increased nutrient supply may adversely affect corals and entire reefs. These range from physiological through organismic, to whole community consequences.

1 On the physiological level, there is considerable evidence that eutrophication destabilizes the functioning of the otherwise finely attuned symbiotic alga-host association. The way the increase in nutrients may affect the relationship has been postulated to be as follows. The zooxanthellae normally are N-limited, and have no use for most of the abundant photosynthate (C) that they produce in excess of their own respiratory requirement, at least under the high irradiance levels at the shallow depths where reefs thrive. This excess is translocated to the host. Translocation may amount to some 95% of the photosynthetic production of the zooxanthellae, and thereby provide more than the total metabolic needs of the host animal (Falkowski *et al.* 1984; Muscatine *et al.* 1984; Porter *et al.* 1984). Under such conditions the zooxanthellae are nitrogen- (and phosphorus-) limited and can multiply only extremely slowly, with doubling times as long as 70–100 days in the common Red Sea coral *Stylophora pistillata*. Once supplied with additional nutrients, either as inorganic compounds such as ammonium and phosphate, or via zooplankton consumption by the host animal (Muscatine *et al.* 1989; Dubinsky *et al.*

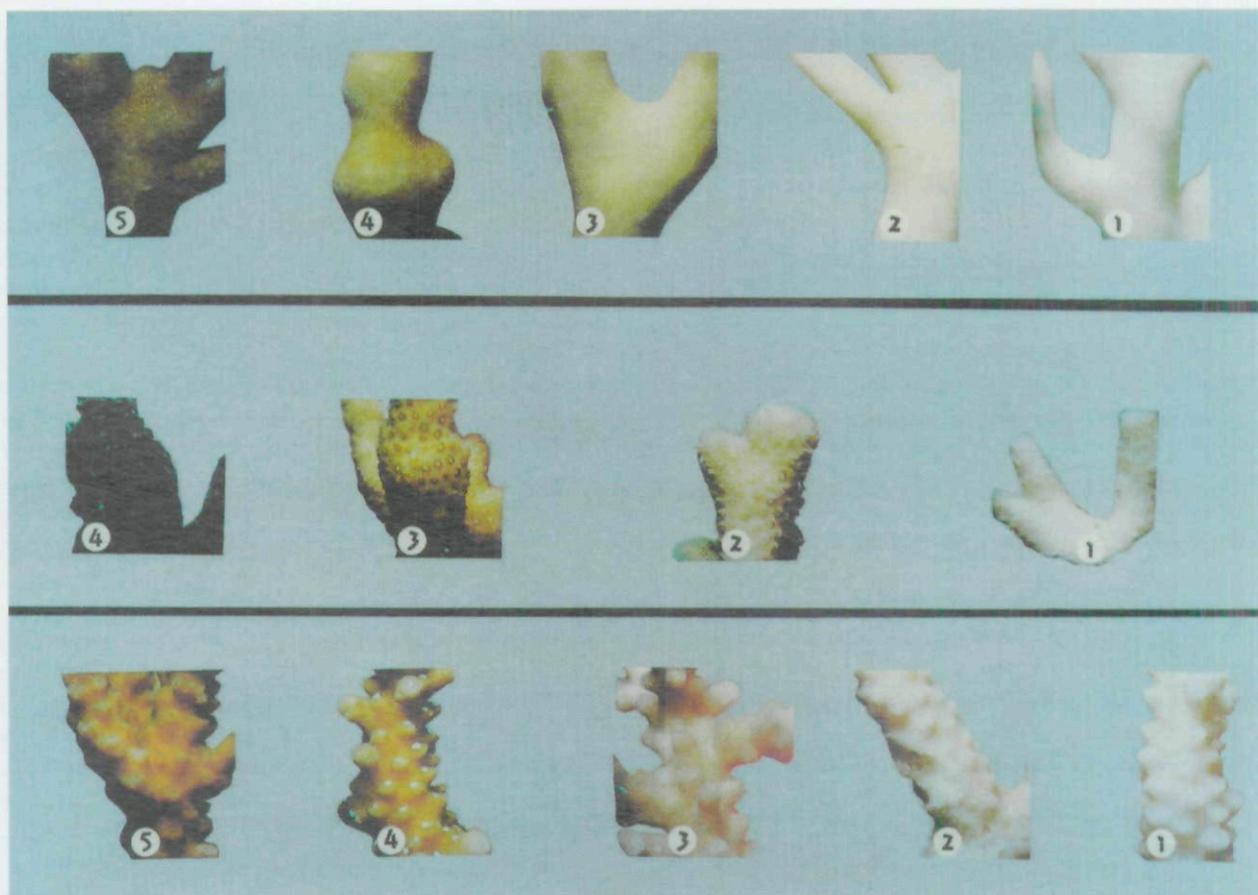


Fig. 3 Changes in colour of coral colonies harbouring different densities, and areal concentrations of pigments, from Thieberger *et al.* (1995).

1990; Falkowski *et al.* 1993; Dubinsky & Jokiel 1994), the zooxanthellae retain most of their photosynthate. That carbon is now utilized for synthesis of zooxanthellae biomass, accelerating their growth rates (Høegh-Guldberg 1994), and increasing their densities up to five fold (Dubinsky *et al.* 1990). These extremely high areal densities (up from 6×10^5 – 3×10^6 zooxanthellae per cm^2 , in a study with *S. pistillata*), are evident to the eye, since shallow water corals, normally ivory coloured (Dubinsky *et al.* 1990), turn almost black as the result of runaway algal multiplication (Fig. 3). This growth results in the following adverse effects on the overall carbon and energy flux within the association.

(a) The zooxanthellae now show significantly reduced photosynthetic rates per algal cell, due to carbon limitation in the super-dense, multilayered algal population (Fig. 4, Dubinsky *et al.* 1990). However, when multiplied by the increased algal numbers, photosynthesis on an areal basis increases by a factor of $\times 1.6$.

In addition to the above-described, measured effect, the following two effects were also inferred.

(b) The rapidly multiplying (Falkowski *et al.* 1985) algal

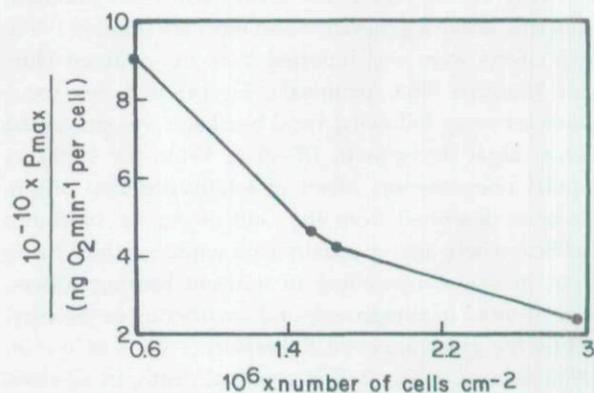


Fig. 4 The decrease in light saturated rates of photosynthesis (P_{max}) of zooxanthellae, $\text{ng O}_2 \text{ min}^{-1} \text{ cell}^{-1}$ (y-axis) as their areal density, $\text{cells} \times 10^6 \text{ cm}^{-2}$ (x-axis) increases. The increase in zooxanthellae density was caused by experimental nutrient enrichment, from Dubinsky *et al.* (1990).

cells respire a larger share of the reduced, photosynthetically acquired carbon.

(c) As a result of (a) and (b), the zooxanthellae population

retains a much higher fraction of photosynthate, rather than translocate it to the animal (Muscatine *et al.* 1989; Falkowski *et al.* 1993).

Snidvongs & Kinzie (1994), working with nitrogen- and phosphorus-enriched *Pocillopora damicornis*, suggested some negative feed-back mechanisms which enhance the effects of single-nutrient enrichments. They suggest that the augmented nitrogen levels did raise areal zooxanthellae densities, which in turn increased CO₂ demand for algal photosynthesis. At the same time, the increased phosphorus levels reduced skeletal calcification rates, thereby reducing the calcification-driven CO₂ supply. It is worthwhile underscoring that the varied, and sometimes contradictory results obtained in nutrient enrichment studies may reflect the opposite roles of nutrients like phosphorus. While phosphorus interferes with skeletal calcification, it is also a tissue growth stimulant.

2 The increase in available essential nutrients annuls the competitive edge of zooxanthellate corals over other benthic producers, most specifically, seaweeds. These macroalgae seem to have significantly lower affinities toward these nutrients than those of zooxanthellae. The inferior ability of macroalgae to extract nutrients may in part explain their paucity in tropical benthos dominated by coral reefs (Hatcher & Larkum 1983; Berner 1990). However, the onset of eutrophication removes the competitive advantage normally held by zooxanthellate coelenterates and opens the substratum to invasion by the much faster growing seaweeds, which now settle on any vacated substratum patch, dead coral, or even on wounds on living corals where the tissue has been abraded, hindering tissue regeneration and recovery (Ramati 1994). Such effects were also reported from the polluted Hurgada (Eastern Sinai peninsula, Egypt) Red Sea reefs, where recovery following coral breakage was prevented due to algal overgrowth (Riegl & Velimirov 1991). A parallel phenomenon, albeit of totally different origin, has been described from the Gulf of Aqaba, Northern Red Sea, where any unusually cold winters, which bring about increased upwelling of nutrient bearing waters, have resulted in overgrowth and smothering of the coral reef by the green seaweed *Enteromorpha* sp. (Genin *et al.* 1995), as seen in Fig. 2. The eventual death, in all cases when corals were overgrown by algae, results from the combination of shading of the colonies severely curtailing photosynthesis of the zooxanthellae, night-time depletion of oxygen due to impeded circulation, and evolution of toxic hydrogen sulfide from the decaying algae and anoxic sediment. In general, any pollution such as sewage or eutrophication is likely to lead to increased supply of food to benthic microbial communities, which reduce sulfate to hydrogen sulfide (Sorokin 1978), further stressing the corals.

It is important not to underestimate the impact of grazing, mostly by fishes and sea urchins in modifying, and even reversing nutrient effects on seaweed growth (Hatcher & Larkum 1983; Lewis 1986; Carpenter 1986, 1990; Liddell & Ohlhorst 1986; Morrison 1988). All of these authors cite cases where seaweed proliferation occurred as a result of removal or mitigation of grazing, while nutrient levels remained unchanged. Therefore, overfishing of herbivorous species may lead to replacement of corals by seaweeds (Hughes 1995; McClanahan & Shafir 1990).

3 Increase in phytoplankton populations in the waters surrounding the reef. Any increase in nutrient availability in oligotrophic waters invariably results in considerable increases in phytoplankton standing stocks. This is usually reported as increase in chlorophyll *a*, and causes a steep decrease in the light available to the underlying corals.

Indeed, increases in phytoplankton biomass, chlorophyll *a* (Genin *et al.* 1995) and turbidity, and concomitant decreases in irradiance, were reported in conjunction with nutrient enrichment episodes, whether of natural (Genin *et al.* 1995) or anthropogenic origin. Hunter & Evans (1995) reported dramatic decreases in nutrients, phytoplankton biomass and turbidity following the diversion of two large sewage outfalls in Kaneohe Bay Hawaii. Such decreases in available light mean a significant, potentially harmful decrease in photosynthesis of the zooxanthellae, and a corresponding decrease in the carbon translocation stream from symbiont to host (Muscatine *et al.* 1984; Porter *et al.* 1984; Falkowski *et al.* 1984). Tomascik & Sander (1987b) attributed depressed larval development and maturation to decreased light available for photosynthesis of the zooxanthellae in the coral *Porites porites*. Bell (1992) states that even phosphorus alone is sufficient to bring about harmful blooms of nitrogen-fixing cyanobacteria like *Trichodesmium* sp. Bell (1992) further links the devastating outbreaks of the corallivorous crown of thorns starfish, *Acanthaster planci*, to eutrophication-driven bloom of nanoplanktonic algae, on which the larval stages of the starfish feed. Bell (1992) also attributes the overall documented decline in the reefs of the inner Great Barrier Reef lagoon to various results of anthropogenic eutrophication.

In addition to the effects discussed above, a number of studies have reported reduced calcification and slower linear skeletal extension rates in nutrient-enriched corals (Stambler *et al.* 1991; Stimson 1992). Growth rates of both lobate and columnar morphotypes of the hermatypic coral *Montastrea annularis* in Barbados, were also inversely correlated with eutrophication along a gradient in Barbados (Tomascik 1990). However, moderate (3.9 mmol N m⁻² d⁻¹, 0.2 mmol P m⁻² d⁻¹) nutrient enrichment did stimulate growth in the same species (Meyer & Schultz

1985), probably due to an increase in suspended particulate food sources (Erez 1990), whereas at higher nutrient levels and suspended particulate matter, linear extension rates decreased (Tomascik & Sander 1985). Similar results were obtained by Davies (1990), also along a eutrophication gradient in Barbados, using small pieces (cores) of the same species. However, in nubbins from the branching species *Porites porites* that author found no such nutrient effect. It may well be that the nutrient-induced shifts in carbon allocation and translocation patterns interfere with the supply of carbon skeleton needed for skeletal matrix formation. Simkiss (1964) demonstrated that elevated phosphorus can interfere with CaCO_3 precipitation in seawater, acting as a 'crystal poison'. Walker & Ormond (1982) cite the enrichment of water of the Gulf of Aqaba, Red Sea to up to $20 \mu\text{g L}^{-1}$, due to inefficient loading of phosphate at the harbour, as a possible factor in the heavy coral mortality in nearby reefs. However, they point out that death may have also resulted from the enhanced growth of various algae, mainly *Ulva lactuca* and *Enteromorpha clathrata*, which in addition to competing with the corals for substratum (Benayahu & Loya 1977), also increase the amount of sediment trapped on the corals.

The overall growth and stability of coral reefs results from the dynamic balance between the rates of calcification and of bioerosion. Interestingly, Hallock (1988) and Highsmith (1980) postulate that eutrophication adversely affects that balance, not only by slowing down growth and calcification of corals, but also by stimulating the activities of various bioeroding organisms. That author proceeds to link widespread extinctions of coral reefs found in the geological record, to concomitant increases in deep sea mixing. Such episodes pump nutrient-rich waters to the surface, reducing the competitiveness of hermatypes vs. bioeroders. This increase in bioerosion is aggravated by the nutrient induced proliferation of various algae reported in recent natural and anthropogenic eutrophication events.

The overall effects of eutrophication alone, or when combined with sedimentation, seem to be more severe on juvenile corals at all stages. Tomascik (1991) reported significant differences in planulae settling along eutrophication gradients within individual reefs as well as among reefs. Not only did eutrophication reduce the overall number of surviving and settling planulae, but it also totally excluded corals like *Montastrea annularis* and *Diploria* sp. from affected sites. Hunte & Wittenberg (1992) exposed cement blocks on increasingly eutrophic reefs in west Barbados. After three years, they found that both the total number of coral recruits, and the number of recruiting species settling on the blocks diminished with nutrient concentrations. Survival was affected by a combination of competitive exclusion of settling planulae by

algae, which rendered the substratum unsuitable for settling, and by size-dependent smothering of small colonies by seaweed overgrowth (Wittenberg & Hunte 1992).

At least one study (Antonius 1985), reported a connection between a disease and eutrophication. The spread of the black band disease, caused by the highly infectious cyanobacterium *Phormidium corallyticum*, affecting many Faviid corals, is enhanced by high light and eutrophication.

Like in the case of most other environmental forcing functions acting on organisms, coral species differ in their sensitivity to eutrophication, as was documented by Tomascik & Sander (1987a) who showed that species diversity in coral assemblages in Barbados reefs was inversely correlated with eutrophication levels, and that the intensity of eutrophication controlled coral zonation as well. Tomascik (1991), documented a species-specific susceptibility to enrichment, a trend also reported by Walker & Ormond (1982). These authors found that in Red Sea reefs exposed to a combination of phosphorus dust and sewage, only the hardy, 'r strategist' *Stylophora pistillata* (Loya 1976) survived, although in much reduced abundance.

The instances when eutrophication alone occurs in the field are few, and result from natural causes (Lapointe *et al.* 1992), regular or catastrophic upwelling (Ganin *et al.* 1995). It is rather common for eutrophication to take place in association with pesticides, herbicides, organics and increased sedimentation resulting from hotel and resort gardening (Hawkins & Roberts 1994), and expanding agriculture (Lewis 1987). Therefore, all conclusions drawn from such studies and surveys should be regarded with caution, due to the difficulty in isolating experimental variables. These constraints underscore the unique importance of the few artificial eutrophication studies conducted *in situ* (Kinsey & Domm 1974; Larkum & Steven 1994; Steven 1994). In the ongoing ENCORE study a series of small atolls in the One Tree Island region of the Great Barrier Reef are being exposed to chronic nutrient enrichment by sophisticated automatic systems. That study aims at examining the effects of nitrogen and phosphorus enrichment through four sub-programs: (i) Organismic level responses, (ii) Population and community responses, (iii) Integrative processes, (iv) Modelling the nutrient effects.

Sedimentation

Corals are harmed by sediment due to outright burial, which leaves them in darkness and lacking in oxygen, with death ensuing within hours to within a few days. In less severe cases, the survival of coral colonies will depend on the amount of sediment and its size

distribution (Rogers 1990). Such sedimentation, in addition to increasing water turbidity and curtailing light available for the photosynthesis of the zooxanthellae, may also abrade the coral tissue. The susceptibility of corals to sedimentation depends on their colony morphology, since sediment will not stay on vertical surfaces and on finely branched species. Many massive species, which are prone to sediment accumulation on their horizontal upper faces, have developed mechanisms to cope with sedimentation. Removal may be accomplished by copious mucus production, coenosarc distention with water and ciliary transport of the particles away from the mouth (Dodge & Vaisnys 1977). The physiological mechanisms for sediment-induced damage certainly include severe anoxia in outright burial, through increased metabolic costs of sediment removal in chronic cases (Dodge *et al.* 1974). These authors show that small colonies fare better in terms of energetic cost/benefit terms when removing sediment, which may contribute to their higher survivorship. In addition to these effects, the loose sediment is avoided by settling planulae, who, facing constant sedimentation, prefer overhanging and vertical substrata to horizontal ones (Nannette Furman, personal communication). In the unique case of Fungids, these large non-attached single polyps may actively use their limited locomotion to disinter themselves after burial (Nannette Furman, personal communication). Not only does sedimentation harm adult colonies, but it also interferes with planula settling and reduces overall recruitment. Riegl (1995) summarizes his short (17 h)-, and long-term experimental exposure of scleractinian and alcyonacean corals to sedimentation. He found no effect of sand grain size, and described that scleractinians did actively remove sediment, whereas the soft corals relied on water motion. The alcyonaceans proved more sensitive to the continuous exposure to sediment which led to death in this group. In all corals sedimentation caused necroses, and death of permanently buried colony parts. In localities subject to chronic moderate sedimentation, zonation proceeds along sedimentation gradients, according to species susceptibility.

Many of the human activities described above, such as coastal resort development, poor agricultural practices, common in Sri Lanka (Rajasuriya & White 1995) and many other South East Asian countries, sewage discharge, forest clear-cutting in Sri Lanka (Rajasuriya & White 1995) and the Philippines (Gomez *et al.* 1994), dredging for marine structures, dumping of sand to provide beaches for tourists, mining, and oil-drilling, all cause varying degrees of increased sedimentation, in addition to increases in organic loadings and nutrient concentrations. A clear negative correlation between riverine sediment discharge and coral reef health was shown in a

study of the Malindi-Watamu reefs in Kenya (Van-Katwijk *et al.* 1993). The 1990 volcanic eruption of Mount Pinatubo in the Northwestern Philippines resulted in massive increases in sediment load, carried to reef areas by rivers, by volcanic mudflows and ash. The resulting situation reduced live reef cover from 60–70% to 10–20% in 10 days in affected areas which were concentrated in the 4 km next to the shoreline and 10 km each side of river mouths (Pajaro *et al.* 1992).

Thermal pollution

Johannes (1975) in his review of the harmful effects of high water temperatures on marine life, describes the responses of corals to sublethal temperatures. He cites interference with normal reproduction, depression of feeding, loss of zooxanthellae, increased mucus excretion and decrease in the photosynthesis/respiration ratio. The combination of these responses will undoubtedly lead to gradual, species-specific death of corals from affected sites. The adverse effects of elevated seawater temperatures on corals, putatively resulting from global warming, are the topic of another paper in the present issue (Glynn 1996).

In nature, virtually all instances of local seawater warming are associated with power generation. Roessler & Zieman (1969) reported killing of all corals in an extensive area surrounding the Turkey Point power plant heated water canals, in Biscayne Bay, Florida. Neudecker (1981) showed that elevated temperatures in the vicinity of two power plants in Guam led to higher mortality and slow growth in three coral species. The author suggests that susceptibility to elevated temperatures increases with growth rate. Coles (1984) monitored the damage caused by coolant discharge from the Kahe Power Plant, on Oahu, Hawaii, and the recovery resulting from the subsequent diversion of the same outfall. In all cases, a 4–5°C increase in temperature killed nearly all corals in the affected area. Huang *et al.* (1991) described a consistent decrease in the growth rates of corals due to an increase in water temperature to over 30 °C, resulting from the discharge of cooling water from a nuclear power plant in Southern Taiwan. This was interpreted by the authors as a distinct temperature-related effect, since all other power-plant related parameters, such as salinity, pH, dissolved O₂, BOD, oils & greases, were all within the Government recommended standards. The possibility that temperature may interact with parameters within government standards, as well as with other power station-generated effects not covered by the government regulations, in producing the reported damage can not be discounted. However, Meesters & Bak (1994) reported an apparent thermal acclimation process in corals exposed to power plant effluent in Curaçao.

Toxic chemicals

1 Pesticides and herbicides

With the expansion and intensification of agriculture and coastal gardening increasing areas of coral reefs become exposed to pesticide containing run-off. Experimental exposure of *Pocillopora damicornis* planulae to 0.01–100 p.p.m. concentrations of the pesticides carbaryl, naphthol and chlorpyrifos showed that the last one was by far the most toxic. Chlorpyrifos concentrations of 1 p.p.m. resulted in 50% mortality, whereas concentrations 10 times higher of the other two compounds had no effect on the larvae (Acevedo 1991). High (p.p.m.) levels of organochlorine pesticides, and heavy metals were found both in reefs receiving urban pollutants from Miami and those exposed to agricultural runoff from the Homestead centre. Corals from the two sites showed high incidence of abnormalities, blemishes, necrotic patches and high incidence of disease which were thought to result from these pollutants. (Glynn *et al.* 1988).

2 Oil and oil drilling

Unfortunately, from the point of view of coral reef health and conservation, many coral reefs are found in proximity to oil-rich areas. The activities related to oil drilling, production and transportation inevitably result in chronic and accidental exposure of reefs and their denizens to many harmful substances. Acute oil spills cause extensive death of corals and prolonged cessation of growth and reproduction in damaged, but recovering colonies, probably due to allocation resources to regeneration. Following a massive oil spill affecting Panamanian reefs these effects were still apparent five years after the spill, with no evidence of overall reef recovery (Guzman *et al.* 1994). Nevertheless, there are indications that the effect of the dispersants used in the clean-up operations following oil spills may in fact be even more harmful than the oil itself, depending on the coral species and duration of exposure (Wyers *et al.* 1986; Thorharg 1988).

Rinkevich & Loya (1979) found that exposure of colonies of the common Red Sea Coral *Stylophora pistillata* to sub-lethal concentrations of Iranian crude oil significantly decreased the number of female gonads per polyp. The effects of crude oil and its different fractions on corals, and their physiological basis of damage have been examined in a number of studies, reviewed by Loya & Rinkevich (1980, 1987). Among reported effects were impairment of reproduction due to planulae abortion, changes in tentacle motion in Xenids (a group of soft corals), and in mucus excretion and photosynthesis of the zooxanthellae. Harmful fractions of oil are bioaccumulated in the different organs and tissues, and are retained

there for long periods after transfer of the colonies to clean water. The resulting pathological responses included defective reproductive tissues, degeneration and loss of zooxanthellae, and degeneration of mucus secreting cells and muscle bundles (Peters *et al.* 1981). Kusmaro *et al.* (1994) reported inhibition of both settlement and metamorphosis of planulae of the Red Sea soft coral *Heteroxenia fuscescens* by crude oil. These authors quantified that response, proposing it as a convenient bioassay to assess oil damage to reef organisms.

One year after the Gulf war production and respiration (P/R) rates of coral reefs and other benthic communities exposed to massive oil spills in the course of the war were examined in a study based on oxygen exchange rates. Surprisingly, it was concluded that the affected biota on the Saudi Arabian coast had reduced production only in heavily polluted enclosed bays, while no increase in community respiration was evident (Burns *et al.* 1993). A similar conclusion is also reached by Vogt (1995) who summarizes his video-transect surveys as follows: 'Therefore, it has been concluded that corals in Saudi Arabia survived the largest oil spill on record remarkably unscathed, with no visible signs of immediate or late effects up to 3.5 years after the Gulf war'. Interestingly, even far smaller oil spills, if recurrent, may seriously reduce the resilience of coral reefs facing other stressors. Loya & Rinkevich (1980) carefully documented the recovery process of the Elat (Northern Red Sea) reefs, following a catastrophic low tide in 1970, which killed almost all corals on the reef flat. Recovery was delayed in areas affected by recurrent oil spills, as compared with unaffected areas (Loya & Rinkevich 1980). It seems plausible to conclude that chronic sub-lethal exposure to oil or its derivatives, may prove to be more harmful in the long run than isolated, even catastrophic spills. Bak (1987) documented deleterious changes in Aruba reefs which have been exposed for 60 years to chronic pollution from a nearby oil refinery. These included deterioration of the physical structure of the reef and decreased juvenile recruitment.

However, there seems to be no harmful effect of carefully managed oil production facilities on either coral reproduction, or subsequent recruitment. In the Gulf of Mexico, on the Flower Garden Bank oil platforms various coral species of different reproductive strategies, like broadcasters and brooders, hermaphrodite and dioecious, all reproduced normally and successfully settled on oil production equipment (Gittings *et al.* 1992). Colonization of corals on the pylons supporting oil platforms in the South China Sea is so heavy, that it compromises the integrity of the structures and their resistance to waves (Latypov 1991). Lush coral communities have also developed on the pylons of the oil terminal in Elat in the

course of the 20 years elapsed since its construction in 1968.

In addition to the rather extensively studied and documented effects of oil hydrocarbons on coral reefs and other marine life, the drilling fluids and mud may also have serious effects on reefs. When the colonies of the coral *Montastrea annularis* were exposed to increasing levels of oil and gas-drilling fluids under controlled conditions in flowing sea water, changes in free amino acids and plasmalogen phospholipids were detected. These were interpreted by the authors as symptoms of infection with anaerobic bacteria (Parker *et al.* 1984). Considerable reduction in photosynthesis, respiration and calcification rates were measured in the same coral, when chronically exposed to drilling mud at 1–100 p.p.m. concentrations (Szmant-Froelich *et al.* 1981). Other, non-specific effects of drilling on surrounding corals has also been reported. In an elliptical 85 × 115 m iron-stained area around a drilling site off Palawan Island, Philippines, a 70–90% reduction in the abundance of all non-massive, coral species was observed by Hudson *et al.* (1982).

3 Heavy metals

The current extensive deforestation processes taking place in so many regions adjacent to coral reefs, in addition to the resulting eutrophication and sediment loads, exposes organisms to an unprecedented flux of a variety of heavy metals. Guzman & Jimenez (1992) reported that in Central America, skeletons of the coral *Siderastrea siderea* were found to contain 12 heavy metals, carried to the reefs by the hundreds of rivers whose catchment areas are affected by the prevalent clear-cutting practices. In a heavily polluted Hong Kong estuary, declining species diversity, number of species, and growth rates of corals, were correlated with skeletal concentrations of the heavy metals aluminium, cadmium, copper, lead, uranium, vanadium and yttrium (Scott 1990). It is, however, likely that the skeletal concentrations themselves were not the cause of harm, but rather a correlated and conveniently 'documented' evidence of some primary tissue damage. Zinc, copper, cadmium, nickel and lead were also detected in different Great Barrier Reef scleractinians and octocorallians (Denton & Burton-Jones 1986). However, only few direct physiological effects of heavy metals have been described so far from controlled laboratory studies. Copper and zinc sulfates were found to interfere with fertilization in the three Japanese coral species *Goniastrea aspera*, *Favites chinensis* and *Platygyra ryukyuensis* (Heyward 1988). Low levels of (Ni^{++}), were found to be toxic to coral larvae. A 12 h exposure to nickel concentrations of 9 p.p.m. caused 50% mortality in larvae of the coral *Pocillopora damicornis*. Settlement of planulae proved even

more sensitive, and was reduced significantly even by 1 p.p.m. concentrations (Goh 1991).

It was found that colonies of the coral *Porites lutea* lost their zooxanthellae when experimentally exposed to high iron concentrations in the environment. Corals from sites where iron levels were always elevated seemed to be more resistant to the effect, presumably having developed tolerance (Harland & Brown, 1989). Howard & Brown (1987) found elevated metal concentration in tissues, not skeletons, of *Pocillopora damicornis* exposed to tin smelter effluent near Phuket, Thailand. These were thought to have reached the corals from their dissolved state in the water rather than from particulate food. The occurrence of metal in the tissues correlated with decreased growth rates and low abundance of branching species.

Sewage disposal

The most common cause of local and regional eutrophication is the disposal of domestic, urban and rural sewage, and agricultural wastewater (Hawkins & Roberts 1994).

This common abuse of coral reefs exemplifies the inherent difficulty in separating from each other the different possible causes for reef decline. Except in controlled experiments, only seldom does eutrophication take place alone, without other, related, potentially just as harmful processes taking place. The domestic and agricultural sewage as well as the effluent resulting from treatment to varying degrees of purification and legal discharge standards are all very rich in combined nitrogen and phosphorus, which bring about massive multiplication, or 'water blooms', of the otherwise sparse phytoplankton, as well as proliferation of benthic seaweeds, also normally covering only a small fraction of substrata in healthy coral reefs. These effects were dealt with in detail in the previous sections discussing 'pure' eutrophication, or nutrient enrichment.

Proliferation of the green 'bubble alga' *Dictyosphaeria cavernosa*, following sewage disposal in Kane'ohe Bay, in Hawaii, and the ensuing displacement of corals, has been followed and described in detail (Maragos, 1972; Smith *et al.* 1981; Grigg & Dollar 1990; Hunter & Evans 1995). That was followed by a trend reversal once sewage had been diverted. The course of changes in nutrients and the concomitant increases in phytoplankton standing stocks were also documented by these authors and re-evaluated by Coles & Ruddy (1995). The latter study confirmed the earlier findings, but also confirmed that recovery rates of the reef community were considerably slower than the reduction of nutrient levels and turbidity. Marszalek (1981), in a review of the effects of sewage effluents, concludes that the most serious damage to reefs

results from competition with high-nutrient stimulated algae, rather from effluent toxicity.

The effects of the increased nutrient load resulting from the oceanic disposal of raw sewage and untreated agricultural wastewater are exacerbated by the accompanying organic matter, varying greatly, depending on the source of the wastewater, and on the type and extent of its pre-disposal treatment. The amount of residual organic matter is given as BOD, the sanitary engineering term for biological oxygen demand, in p.p.m. O_2 consumed per litre, over five days at 25°C. In the case of the interaction between corals and sewage this is indeed most adequate. High BOD sewage is an ideal medium for the development of oxygen-demanding microbial populations (Pastorok & Bilyard 1985); therefore, the impact of sewage depends on circulation patterns and lagoonal flushing rates. In lagoons with long retention times the effects of sewage tend to be devastating. The depletion of oxygen due to the respiratory activities of the microbial communities thriving on the organic matter in the wastewater may severely harm coral communities. This aspect was examined in detail by Mitchell & Chet (1976), and brought forward as a contributing agent in the death of corals exposed to sewage and phosphorus in the Gulf of Aqaba, Red Sea (Walker & Ormond 1982).

A similar, though not directly related phenomenon is reported by Rajasuriya & White (1995) from Sri Lankan reefs, where extensive damage in the south of the island results from the common practice of seasoning coconut fiber in calm, near fringing-reef waters. The decomposition process depletes waters of oxygen, compounding other anthropogenic effects.

Furthermore, it has been suggested that some of the microorganisms present in sewage, or proliferating in sewage polluted seawater, may have been involved in the spread of some coral diseases (Antonius 1985).

Even the remaining effluent following sewage treatment, although most of the originally present organic carbon (BOD) has been eliminated in the course of the primary and secondary treatments, usually still contains most of the combined nitrogen and phosphorus. As such, any sewage, whether raw or treated to some extent is still the most common agent of eutrophication.

In the many instances where industrial sewage is not treated separately from domestic and agricultural wastewater, the problems described above are further compounded by the presence of various toxic substances.

Forest clear-cutting

Deforestation in the catchment areas of rivers reaching the sea in the proximity of coral reefs, due to enhanced soil runoff, may result in an increase in the levels of nutrients discharged to the water. Furthermore, much of

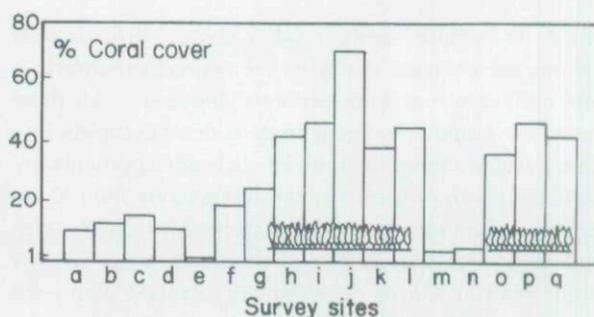


Fig. 5 The relation of coastal forest belts (indicated by trees) and percentage of coral reef cover at different sites (a-q) in the waters surrounding Ishigaki Island, Japan, from Kuhlman (1988).

the lesser branches and shrubs are burnt, eventually releasing the nutrients in the remaining ashes through the river into the waters of the reef. Even prior to the present rates of deforestation, it was thought that the nutrients and sediment carried by major rivers discharging into otherwise coral reef-covered shorelines, prevented their development over considerable distances. These effects may have been combined with the deleterious effects of the reduced salinities in these riverine outlets and estuaries. At the present time, deforestation covers extensive areas in the tropics, many of them in drainage systems terminating next to coral reefs. Not only does the washed down topsoil reach the reefs with its organic components and nutrient load, but these mature, nutrient-conserving and -recycling, tropical rain forest ecosystems (Odum 1969), are replaced by immature, leaky, stressed ecosystems, dependent on constant fertilizer subsidies (Odum 1985), which likewise end up in the sea.

Extensive deforestation affecting corals has been reported from numerous localities adjacent to coral reefs. Mustafa (1990), reports heavy sedimentation and siltation effects on South Andaman reefs, as was also reported from Sri Lanka (Rajasuriya & White 1995) and the Philippines (Gomez *et al.* 1994). Guzman & Jimenez (1992), working on Costa Rican and Panamanian reefs, documented extensive deposition of heavy metals in coral skeletons. These were demonstrated to originate from rivers whose catchment areas are affected by deforestation. The example of the reefs surrounding Ishigaki island in Japan is noteworthy. The wealth of coastal coral reefs was shown by Kuhlman (1988) to be correlated with the distribution of forest belts on shore (Fig. 5). Wherever these forests were cut down nearby reefs were damaged by sediment.

Coastal resort development

Because of its scale, waterfront development (such as the construction of hotel rooms, and the needed population

centres to service them, road systems, airfields and marinas for pleasure craft), in the nearest proximity to coral reefs deserves some separate discussion. All these related developments result in considerable inputs into nearby reef waters. Examples of such developments are the 40 000 room project in construction along the 100 km Cancun–Tulum touristic corridor in the Mexican Yucatan coral reefs. This is only part of the much larger GAMA (Gran Arrecife Maya) development planned along the reefs of four Mesoamerican countries (Mexico, Guatemala, Belize and Honduras). As a byproduct of the most welcome peace process in the Middle East, an almost uninterrupted hotel belt will be built by the year 2005 along the shores of the Gulf of Aqaba, with the addition of 92 630 rooms on the 1840 km Egyptian coastline, a growth of 1168%, 2000 rooms on the 27 km Jordanian strip, a 100% growth, and 6000 rooms, a 'scant' 43% increase in the already overcrowded Israeli 14 km corner (Hawkins & Roberts 1994). Sources of sewage are exponentially growing and intractable (e.g. wastewater discharges from pleasure boats moored next to reefs or cruising around them, and even more so, the sprawling hotel complexes being built wherever coral reefs may still be found and enjoyed). In most cases, the individual hotel operators are expected to treat their own sewage. With the existing and foreseeable levels of technology for small sewage treatment plants, awareness, legislation and enforcement, outright raw sewage dumping, seepage, and under the best scenarios, discharge of high-nutrient effluents are nearly inevitable. The lush gardens expected to be found around hotel complexes, even in regions as arid as the shores of the Northern Red Sea, where annual rainfall is in the range 25–50 mm, also lead to eutrophication resulting from fertilizer runoff to adjacent reefs. At least in some cases plans were drawn to somewhat mitigate this grim prognosis by using sewage effluent for resort gardening. The situation in the rainy tropics is not any better in this respect, since in places like the Hawaiian islands extensive golf courses are continuously being added. These green lawns require generous supply of fertilizers to counter their continuous wash away by rain.

Discussion and recommendations

In the face of the futility of trying to prevent oceanic scale eutrophication, it is important to reiterate that there is so far no evidence for such global fertilization of the oceans, as concluded by Falkowski & Wilson (1992).

Since human impacts on reef are rather localized, they are much more manageable. The potential effectiveness of protection policies is evident in the successful establishment and implementation of the Great Barrier Reef conservation policies (Craick *et al.* 1990). However, based

on currently available evidence, there is no assurance that degraded reefs will recover to their previous state. Done (1995) suggests that there are at least two prerequisites for such recovery to take place: 'Restoration of 'on-site ecological conditions', which will initiate a succession leading towards a coral-dominated and productive state', and 'availability of 'propagules' of key reef-building and reef associated taxa'. In many cases, one or both of these conditions may not be met any more. Pre-remediation human action may have irrevocably changed such ecological features as bottom sediment character, flow patterns and lagoonar residence times, as well as brought about a reduction in the supply of propagules carried to the area in question.

Eutrophication and other types of pollution can be minimized by a combination of several measures.

1 Control of deforestation in areas whose drainage patterns result in increase in nutrient and sediment loads on reefs. Reforestation policies should be initiated wherever possible.

2 Agricultural practices which reduce sediment, nutrient and pesticide runoff should be mandatory in areas affecting reefs.

3 Prevention of any sewage discharge. Treated effluent should also be prevented from reaching the sea, and be used in gardening and restricted agricultural uses. Modern, centralized government facilities should be preferred to owner-operated single-hotel scale plants, which are hard to control.

4 Damage resulting from oil drilling, production and shipping is well understood, and because of the scale of these operations, rather easy to regulate, supervise and manage in ways compatible with reef conservation.

5 Thermal pollution is localized, and by carefully selecting sites of power generating plants, their intakes and outfalls, damage to reefs may be very localized and minimal.

6 Dumping of sand to provide beaches on hotel water fronts should not be allowed. Such beaches, whenever inevitable, should be constructed in ways preventing any of the sand from reaching the reef.

7 Any metal mining and smelter activities should be constructed and operated in ways assuring that no metal or other related pollutants reach the sea, causing harm to reef organisms and turning seafood potentially toxic to consumers.

The economic benefits reaped from coral-reef based tourism are increasing rapidly. Therefore, there is hope that authorities will realize that the long-term benefits to be gained from healthy reefs, residing in crystal-clear waters, by far exceeds the immediate revenue resulting from their short-sighted destruction. Implementation of conservation measures will save the 'threatened reefs' and possibly in some cases initiate their recovery.

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