

1. A REVIEW OF STUDIES OF IMPACTS OF SEWAGE EFFLUENTS
ON MARINE ORGANISMS, and
2. PROJECT RATIONALE AND OBJECTIVES, from:

IMPACTS OF SEWAGE EFFLUENTS ON MARINE ORGANISMS

ENVIRONMENTAL COMPONENT of THE BIOLOGICAL NUTRIENT REMOVAL BARBADOS PILOT PLANT DEMONSTRATION

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1. A REVIEW OF STUDIES ON IMPACTS OF SEWAGE EFFLUENT ON MARINE ORGANISMS

1.1 THE ECOLOGICAL AND SOCIO-ECONOMIC VALUE OF CORAL REEFS

Coral reefs are tropical shallow water ecosystems that thrive at temperatures between 25°C and 29°C. They are restricted to a circumtropical belt between latitudes 30°N and 30°S. Coral reefs are widespread throughout the Caribbean, ranging from Panama and Tobago in the south to Florida and Bermuda in the north; with most Caribbean nations having reefs along at least parts of their coast. Since reef-building corals need light, significant reef development is restricted to < 30 m depth in clear water and much less in turbid water. Consequently, coral reefs are typically found relatively close to land. Depending on location, origin and form, reefs are classified as fringing reefs, bank reefs, barrier reefs and atolls. Fringing reefs are common in the Caribbean. They occur close to shore, extending seawards to about 100-200 m. Their location ensures that they are particularly vulnerable to land-based pollution. Bank reefs are also common in the Caribbean. They occur further offshore than fringing reefs, e.g. about 1 km offshore along the entire length of the south and west coasts of Barbados. Barrier reefs are also offshore reefs, differing from bank reefs in mode of formation. The world's second largest barrier reef occurs in the Caribbean, off Mexico and Belize. It is approximately 200 km long and lies 11-12 km offshore. There are no atolls in the Caribbean.

Coral reefs are among the most biologically productive and diverse of all natural ecosystems, and are arguably the most important coastal marine habitat in the Caribbean. Their socio-economic value rests on several characteristics:

- (1) Coral reefs are the most species rich marine community in the tropics. Their biodiversity gives them tremendous aesthetic value. This is increasingly used to lure tourists to the Caribbean, since most commercial and recreational water sports activities (e.g. glass bottom boats, snorkelling, SCUBA) are based on coral reefs.
- (2) The extreme biodiversity of coral reefs makes them more important than any other marine community as a reservoir for potentially valuable pharmaceutical compounds.
- (3) The traditional artisanal trap fishery in the Caribbean targets fish that live on relatively shallow reefs, and these reefs may also be important nursery grounds for juvenile snapper and

grouper. Adult snapper and grouper are fished by hand-line on the deeper slopes of offshore reefs and on offshore banks. Deterioration of near-shore reefs could therefore affect the offshore snapper/grouper fishery as well as the trap fishery. The lucrative lobster fishery in the Caribbean, and most spearfishing activity, is dependent on the maintenance of healthy coral reefs. In many eastern Caribbean countries, reef-based fisheries remain the single largest contributor to annual fish catch. It is estimated that Caribbean reefs may be capable of annual fishing yields of 750,000 tons.

(4) Corals reefs play a pivotal role in maintaining beaches and minimising coastal property damage due to waves. Physical and bioerosion of coral skeletons is the source of the sediment of which most beaches are made. Moreover, coral reefs physically protect beaches and coastal property from the impact of waves, thereby preventing beach erosion and minimising property damage during periods of high wave energy. The importance of stable high quality beaches to the tourism sector in the Caribbean is evident.

Corals have evolved under relatively constant environmental conditions and are therefore particularly sensitive to environmental changes in environmental quality. Coral reefs increase only slowly in size, those in the Caribbean growing at about 1 cm/yr. This slow rate of reef regeneration emphasizes the urgency of eliminating activities that lead to reef loss.

1.2 THE ECOLOGICAL AND SOCIO-ECONOMIC VALUE OF SEAGRASS BEDS

Seagrasses are rooted vascular plants that depend primarily on nutrient resources in the sediments for growth. A seagrass bed is dominated by such plants, but also contains benthic and epiphytic algae which utilise nutrients from the water column. Of the 12 genera of seagrasses known globally, 7 are tropical in distribution and 4 occur in the Caribbean; namely, *Halodule*, *Syringodium*, *Thalassia* and *Halophila*. Given their dependency on light, seagrass beds are restricted to shallow sub-tidal areas and are therefore particularly vulnerable to land-based pollution.

Tropical seagrass beds are among the most productive of natural ecosystems, seagrasses growing rapidly and producing up to 10 g of dry leaf/m²/day. They support a diverse and abundant fish and invertebrate fauna which typically exceeds that in unvegetated portions of the coastline in both biomass and numbers. Despite this, surprisingly little research effort has focused on seagrass ecology. It is only recently, against the background of an increasing perception of accelerating deterioration and loss of tropical seagrass beds, that efforts are increasing to improve our comprehension of their ecology in an effort to conserve them (Sheppard et al., 1989).

The high productivity and extensive root and rhizome systems of seagrasses make seagrass beds valuable coastal habitats in the Caribbean. The high productivity drives three major food webs; namely, direct grazing of seagrass blades, detrital food webs within the grass beds, and detrital food webs based on material exported from the grass beds. A fourth food web is based on the epiphytes which grow on the seagrass blades. The epiphytes may be comparable in biomass to the seagrasses themselves and are heavily grazed by invertebrates and fishes.

A number of species that are commercially fished are permanent residents of seagrass beds as adults. These include the queen conch *Strombus gigas* and the white sea urchin *Tripneustes ventricosus*, both of which support important coastal fisheries in the Caribbean. Green turtles and the West Indian manatee are direct grazers of seagrasses, but their numbers have declined dramatically in the Caribbean under extreme fishing pressure. Perhaps most importantly in the context of the importance of seagrass beds to coastal fisheries, the adults of many species of commercially harvested fish migrate from coral reefs to seagrass beds at night to feed. These include snappers, grunts, parrotfishes and squirrelfishes. Given this, considerable fishing effort is focused on seagrass beds by commercial fisherman. Fishing activities include the use of Antillean fish traps, spear-fishing, and hand-lining for larger species, particularly at night. Apart from supporting commercially important adults, seagrass beds are also important nursery habitats for juveniles of commercially harvested shrimps, crabs, lobsters and fish, the beds providing abundant detrital organic matter for food as well as shelter from predators (Menzies and Rowe, 1969).

The export of organic material from seagrass beds to food webs of other coastal communities, the migration of adult coral reef fish to seagrass beds to graze by night, and the use of seagrass beds as nursery habitats for early life stages of both invertebrates and fish which frequent coral reefs as adults, amply illustrate the important ecological linkages between seagrass beds and coral reef communities. Studies have frequently shown that coral reefs in close proximity to seagrass beds have larger reef fish populations than those in parts of the coasts devoid of beds (Randall, 1965; Starck and Davis, 1966; Ogden, 1972).

Apart from their role as food producers, and their consequent importance as a food source and nursery habitat for commercially fished species, the dense leaves characteristic of seagrass beds reduce current velocity near the sediment surface and promote sedimentation of organic and inorganic particles. The roots and rhizomes bind these sediments, hindering erosion of the sediment surface. Seagrass beds are therefore important stabilizers in the coastal zone (Scoffin, 1970). The role of seagrass beds in this context is illustrated by the extensive beach loss and formation of offshore bars that accompanied massive disease-induced loss of seagrass beds in the Atlantic in the 1930's (Hunte and Lark, 1990).

Despite recognition within the scientific community of the importance of seagrass beds as fish and invertebrate habitat and in coastline stability, no attempt has been made to protect and manage seagrass beds in the Caribbean. Little quantitative information exists on the distribution and abundance of seagrass beds in the region, and quantitative statements cannot therefore be made on rates of disappearance and/or deterioration. Nevertheless, in Barbados, consistent reports from divers whose observations span the last two decades suggest substantial loss of seagrass habitat, particularly in the St. Lawrence - Sandy Beach area of the south coast and the Spring Garden - Brighton area of the west coast.

1.3 ANTHROPOGENIC IMPACTS ON CORAL REEFS

1.3.1 Sedimentation

1.3.1.1 Sources of sedimentation

Increased sedimentation due to coastal and other land-based development is recognized as one of the most detrimental anthropogenic factors causing the degradation of coral reefs (Johannes, 1972, 1975; Dahl, 1985; Pastorok and Bilyard, 1985; Rogers, 1985, 1990). Potential sources of sediment include domestic sewage input, terrestrial runoff, dredging and filling activities, drilling muds, industrial effluent, bioerosion, and the resuspension of bottom sediments (Pastorok and Bilyard, 1985; Grigg and Dollar, 1990).

1.3.1.2 Mechanisms of impact

Sediment loading of the water column may directly or indirectly affect coastal marine organisms. Chronic rates of sediment deposition may overwhelm an organism's ability to shed sediment particles, leading to physiological stress and ultimately death due to smothering. Highly turbid water reduces subsurface light levels which diminishes the amount of light available for photosynthesis of benthic marine organisms (Pastorok and Bilyard, 1985; Rogers, 1990). Decreased photosynthetic rates of endosymbiotic zooxanthellae within coral tissues can affect overall productivity in the reef ecosystem. For example, Rogers (1979) showed that under partially shaded conditions (simulation of extreme turbidity), the hard coral *Acropora cervicornis* experienced bleaching, a reduction in growth rate, and a decrease in productivity.

Sedimentation may affect coral reef organisms, not only by affecting mortality rates and growth rates through the processes described above, but also by influencing coral recruitment. Recruitment has been shown to be affected by sedimentation in Guam (Birkeland, 1977; Birkeland et al., 1981), Curaçao (Bak and Engel, 1979), St. Croix (Rogers et al., 1984), and Barbados (Tomascik, 1991; Hunte and Wittenberg, 1992). Hodgson (1990) showed that settlement rates of *Pocillopora damicornis* planulae were inversely related to sediment thickness on the substrate. Increased sediment load on a reef can also significantly reduce the survival of

young recruits, which may lead to a shift in community structure if recruit mortality rate varies between species (Wittenberg and Hunte, 1992).

1.3.1.3 Species-specific variation in susceptibility

Studies by Marszlek (1981) off Miami have demonstrated that, whereas soft corals and some species of sponges are relatively tolerant of high sedimentation, scleractinian corals can only withstand a few days of heavy sediment loading before experiencing localised zooxanthellae loss and tissue deterioration. Studies in Grand Anse Bay in Grenada have also indicated that sponges are less susceptible to sedimentation than scleractinian corals (Hunte, 1987; Hunte et al. 1989). Moreover, within scleractinian corals, there is considerable between-species variation in susceptibility to sedimentation. This variation results primarily from species-specific differences in tendency to trap sediment, in ability to reject sediment particles from the surface of the colony, and in reliance on photosynthesis as a form of nutrition.

Colony shape and size are important in the context of trapping sediment. Plate-like and massive corals offer a large catchment area for particles to settle, whereas upright, branching, vertical-plate, and convex colonies are less likely to trap particles (Bak and Elgershuizen, 1976; Dryer and Logan, 1978; Lasker, 1980; Rogers, 1983). Colony size is also important in the context of sediment rejection. Smaller colonies are typically more efficient at rejection due to the smaller distance that particles must be moved to be shed from the colony (Dodge and Vaisnys, 1977). However, between-species variation in rejection ability may depend more on polyp size, polyp behaviour and polyp coordination than on colony size. Coral polyps have four mechanisms by which particulate matter can be shed; polyp distension through the uptake of water into the stomodeum, tentacular movement, ciliary action and sloughing off with mucus (Hubbard and Pocock, 1972). Lasker (1980) reported that tall polyps were more effective at sediment rejection; and laboratory studies by Hubbard and Pocock (1972) and by Bak and Elgershuizen (1976) indicate inter-specific variation in the polyp coordination necessary to remove sediments from the colony by the shortest available route.

Rogers (1983) studied the effects of varying sediment loads on several Caribbean coral species, representing most of the size/shape classes found in the region. Deterioration of the underlying tissue occurred under loads of 200 mg/cm² for the plate-like *Acropora palmata* and 800 mg/cm² for the massive coral *Montastrea annularis*, but less massive and less plate-like species were not significantly affected by any of the experimental doses. Note that this study measured only direct effects of sedimentation and direct effects would normally be synergistic with the energetic costs of sediment rejection and reduced photosynthesis resulting from light reduction.

The fact that susceptibility to sedimentation in the field may involve more than susceptibility to smothering and sediment rejection capability is indicated by contradictory results which have sometimes emerged between field and laboratory observations. For example, laboratory results indicated that *Porites asteroides* would be more inefficient than other species in removing sediments (Bak and Elgershuizen, 1976), but field results suggest that it can be the dominant coral in heavily sedimented areas (Morelock et al., 1983; Cortes and Risk, 1984; Tomascik and Sander, 1987; Mann pers. obs.).

Not only may susceptibility to sedimentation be influenced by species-specific variation in coral morphology, but exposure to sedimentation may influence phenotypic development of morphology within species. Foster (1979, 1980) showed that morphological plasticity in *Montastrea annularis* can be used as a bioindicator of increased sediment loading. Colonies transplanted from relatively pristine habitats to habitats with higher sedimentation rates showed skeletal morphological changes; and Hubbard et al. (1987) suggest that the columnar growth form in this species is a response to heavy sedimentation.

Inter-specific differences in susceptibility to sedimentation may also reflect inter-specific differences in reliance on photosynthesis as a form of nutrition. The metabolism of photosynthesizing organisms on a reef can be affected by increases in sediment particles in the water column, since the quantity and spectral quality of light available to reef organisms is lowered (Rogers, 1990). Corals, with their endosymbiotic zooxanthellae, rely on light to promote rapid deposition of calcium carbonate into the skeleton (Chalker, 1981). Rogers (1979) shaded a portion of the San Cristobal reef in Puerto Rico, and found that the simulated turbidity

reduced the productivity of the coral community. Edmunds and Davies (1986, 1989) calculated energy budgets for *Porites porites* off Discovery Bay, Jamaica, and reported that colonies experiencing lower light levels due to turbidity had lower growth rates. Dodge et al. (1974) demonstrated reduced growth rates in *Montastrea annularis* as sedimentation rates increased; and Tomascik and Sander (1985) found that, of 14 biotic and abiotic environmental variables measured, suspended particulate matter in the water column was the strongest predictor of growth rate in *Montastrea annularis* along the west coast of Barbados.

1.3.1.4 Sedimentation thresholds

Sedimentation levels leading to reef deterioration may vary considerably between reefs, due largely to the fact that physical factors can accentuate or diminish the effects of turbid water, e.g. strong currents will inhibit the settlement of particulate matter (Johannes, 1975). However, Rogers (1990) suggests that normal sedimentation rates range from less than 1 to about 10 mg/cm²/d, and that rates above the latter limit can be considered "high". Randall and Birkeland (1978) report sedimentation rates as high as 250 mg/cm²/d in Guam, but state that a negative impact on coral communities occurs at values well below 50 mg/cm²/d. Pastorok and Bilyard (1985) state that increasing negative impacts can be expected over the range 10-50 mg/cm²/d, and consider values in excess of 50 severe to catastrophic. Hunte et al. (1992) have proposed that suspended sediment values in the water column in excess of 5 mg/l should be considered stressful to corals.

1.3.2 Eutrophication

1.3.2.1 Sources of eutrophication

Eutrophication refers to nutrient enrichment, and is arguably the most serious problem affecting coastal marine communities globally. Many anthropogenic activities have contributed, either directly or indirectly, to this phenomenon. The most important of these in the Caribbean is probably increased domestic sewage release accompanying urbanisation, coastal development and tourism. Other sources include industrialisation and agricultural practices. Coral reefs are

typically found in oligotrophic (nutrient-poor) water, and have evolved intricate nutrient recycling mechanisms that enable them to maintain high productivity in the face of low nutrient levels. They are therefore particularly susceptible to even modest increases in nutrient concentrations of coastal waters.

1.3.2.2 Mechanisms of impact

While nutrients may be directly toxic to coral reef organisms, this is only likely to occur at extreme concentrations. Few coral reefs globally are exposed to nutrient concentrations high enough to have direct toxic effects. However, indirect negative effects of elevated nutrients are both subtle and diverse.

Corals are autotrophic (i.e. use solar sources of energy through a symbiotic relationship with zooxanthellae) as well as heterotrophic (i.e. feed on plants and animals). Their two forms of nutrition allow high productivity even in oligotrophic tropical waters. Nutrient enrichment of these waters facilitates growth and reproduction in benthic and planktonic algae, which are normally nutrient limited (Bell, 1991). Low levels of eutrophication can enhance benthic algal production without affecting the biomass, species composition, or trophic structure of coral reefs (Kinsey and Domm, 1974). However, moderate levels of eutrophication cause a substantial increase in benthic algal populations (Laws and Redalje, 1979; Smith et al., 1981), including benthic green or blue-green algae (Banner, 1974); and this has major implications for the abundance of corals and the structure of coral reefs. For example, Maragos (1972) described dense mats of the green bubble algae *Dicryosphaeria cavernosa* supported by eutrophication in Kaneohe Bay, Hawaii caused by sewage discharge. The extent of the algal growth was so severe that coral cover was reduced virtually to 0% (Maragos, 1972).

Phosphate pollution has been clearly implicated in the degradation of reefs in the Red Sea (Fishelson, 1973; 1977; Loya, 1975; 1976; Walker and Ormond, 1982), and in the Indo-Pacific (Kinsey and Domm, 1974; Kinsey and Davies, 1979). Kinsey and Domm (1974) and Kinsey and Davies (1979) experimentally increased nutrient levels on One Tree Island, Great Barrier Reef, Australia. They found that the elevated phosphate levels suppressed calcification by at

least 50%, while increasing productivity by about 50%, and increased benthic algal production by about 50% (Kinsey and Davies, 1979). The enhanced algal benthic growth accompanying eutrophication often attracts and supports higher densities of herbivorous fish (Johannes, 1975; Russo, 1982), but the increased abundance typically persists in spite of increased grazing pressure (Walker and Ormond, 1982).

Increased growth of planktonic algae resulting from eutrophication affects corals by increasing competition for sunlight. This constrains coral nutrition, growth and ultimately survival, through negative impacts on their endosymbiotic zooxanthellae (Smith et al., 1981). The increased primary production in the water column also favours filter-feeders such as sponges, bryozoans and tunicates, which in turn may outcompete corals for space on the reef (Maragos, 1972; Maragos and Chave, 1973; Birkeland, 1977; Wittenberg and Hunte, 1992). Tomascik and Sander (1985, 1987) showed that growth and reproduction in adult corals was negatively correlated with increasing nutrient levels in Barbados; and Wittenberg and Hunte (1992) and Hunte and Wittenberg (1992) reported higher mortality and lower settlement of juvenile corals on more eutrophic reefs.

Birkeland (1977) described the response of coral reef communities to natural nutrient inputs caused by upwelling off the Pacific coast of Panama. Shifts in community structure were similar to those described for reefs affected by anthropogenically-induced eutrophication. High densities of fouling organisms such as bryozoans, tunicates, and filamentous algae were found to rapidly colonise substrates during and immediately after upwelling episodes, but coral growth and recruitment rates recovered after nutrient levels subsided and the rate of benthic biomass accumulation declined (Birkeland, 1977).

Phytoplankton blooms resulting from eutrophication can also add to the sedimentation load of the reef ecosystem, particularly the organic fraction of the load (Bell, 1991). The resulting breakdown of decaying organic particles in the water column uses oxygen, and can reduce oxygen concentrations to critical levels, especially at night when photosynthesis (oxygen production) stops. Since tropical marine organisms live closer to their lower oxygen limits than

their temperate counterparts, even slight depressions of oxygen levels can cause mortality in reef organisms (Kinsey, 1973).

The most well documented case of reef destruction caused by anthropogenically-induced eutrophication is that of Kaneohe Bay, Hawaii. This bay was described by Edmondson (1928) as "one of the most favourable localities for the development of shallow water coral, with nearly all the reef-forming genera known in Hawaii being represented ... and many species growing luxuriantly". Sewage discharge rates into Kaneohe Bay increased from about $5.7 \times 10^3 \text{ m}^3/\text{d}$ to $3.2 \times 10^4 \text{ m}^3/\text{d}$ between 1950 and 1977, with secondary treatment commencing after 1963. By 1970, more than 99% of the original 880 hectares of reef in the southern portion of the bay were dead (Maragos, 1972). A principal cause of mortality was exposure to H_2S resulting from anaerobic conditions in the sediments (Maragos, 1972). In less devastated reef areas, the general response was that community structure shifted from coral and coral associates to green algae and filter-feeders such as sponges and zooanthids (Maragos, 1972; Smith et al., 1981).

The installation of a deep sea sewage outfall off Mokupa Point in 1977/1978 initiated recovery of the reefs in the southern end of the bay. Smith et al. (1981) monitored physical, chemical and biological variables to investigate the response of the marine communities to the reduced sewage input. The variables that showed the most significant decreases were phosphates, hard bottom algal biomass, hard bottom cryptofaunal biomass and lagoon floor biomass, the consequence being increased availability of benthic space for corals. While the recovery of the benthic community was substantial following sewage input reduction, the community structure did not return to its "pre-sewage" condition (Smith et al., 1981). Laws and Redalje (1982) showed that nutrients released from organic rich sediments continued to support high phytoplankton populations for at least 2 years after sewage reduction, presumably constraining the return to the original community structure.

1.3.2.3 Species-specific variation in susceptibility

Little is known about species-specific variation in susceptibility of corals to eutrophication. Preliminary work in Barbados suggests that species with finger-like colonies and heavy dependence on asexual reproduction through fragmentation (e.g. *Porites porites*) may retain higher abundance levels than other species under eutrophic conditions; and that, in general, corals whose natural life history strategy is based on low recruitment but low natural mortality are more rare on eutrophic reefs than corals whose life history strategy is based on high recruitment and high natural mortality (Wittenberg and Hunte, 1992; Hunte and Wittenberg, 1992).

1.3.2.4 Eutrophication thresholds

Based on values reported in case studies from generally oligotrophic areas around the world, Bell (1991) suggested that inorganic nitrogen, inorganic phosphate and chlorophyll-a have to exceed values of 1 μM , 0.1-0.2 μM , and 0.5 mg/l respectively, before changes in coral community structure can be detected. Based on preliminary correlations between water quality data and coral community structure, Hunte et al. (1992) have suggested that $\text{NO}_3\text{-NO}_2\text{-N}$ values greater than 0.7 $\mu\text{g at/l}$ and $\text{PO}_4\text{-P}$ values greater than 0.1 $\mu\text{g at/l}$ alter coral population biology and coral community structure in eastern Caribbean islands.

1.3.3 Toxins

Toxic contaminants such as PCB's, heavy metals, chlorine, pesticides and petroleum hydrocarbons occur in waste water from industrial, agricultural and urban sources. The many combinations of chemicals which may be found in such effluents can have various additive, synergistic, and/or antagonistic effects on reef organisms (Pastorok and Bilyard, 1985). The direct effects of these compounds on coral reef organisms have received little attention, but the few studies conducted indicate that effects may be severe.

The toxicity of pesticides to corals (*Montastrea annularis*, *Acropora cervicornis* and *Mirabilis madracis*) was investigated by McCloskey and Chesher (1971). They found no changes in feeding behaviour, polyp extension, sediment clearing, settling of coral associates, or crystal formation, but respiration rates (R) increased and photosynthetic rates (P) decreased. Light compensation levels (the light level at which O_2 used in respiration equals O_2 produced by photosynthesis) increased for all three species. The depression of P/R ratios below 1.0, and the increased compensation levels for all of the corals suggests that coral growth and maintenance may be compromised by exposure to chlorinated hydrocarbons (McCloskey and Chesher, 1971).

Trace metal levels are normally much lower in the carbonate sediments associated with reef ecosystems than in clay based sediments, due to the adsorption properties of clay particles. Heavy metal pollution should therefore be easily detectable in reef organisms (Segar and Pellenberg, 1973). Taylor and Bright (1973) investigated the occurrence of heavy metals in reef groupers in the Caribbean. They found mercury, cadmium, lead, copper and zinc in tissue samples, but at levels generally below those dangerous for human consumption. They detected arsenic levels which exceeded U.S.F.D.A. tolerance levels (0.5 ppm), but argued that the levels were a natural phenomenon in phosphate poor waters, where arsenic may be accumulated as a substitute in the phosphate transport system.

The effects of free residual chlorine on reef organisms have received little attention. Muchmore and Epel (1973) showed that concentrations as low as 0.05 mg/l can inhibit fertilisation in marine invertebrates. Alderson (1972) showed that marine fish are even more sensitive, with lethal doses being as low as 0.024 mg/l. Randall (1973) noted that, when the power plant in Guam ran out of chlorine, 9 fish species recolonized the area near the outfall. When normal operations were resumed, the fish disappeared. Davis (1971) found that planulae from 3 species of Indo-Pacific corals were tolerant of a chlorine concentration of 0.49 mg/l for up to 7 hours, but suggested that adult corals may have less resistance.

1.4 ANTHROPOGENIC IMPACTS ON SEAGRASS BEDS

1.4.1 Sources and Mechanisms of Impact

The nearshore location of seagrass beds makes them particularly vulnerable to coastal activities and land-based pollution. Dredging and filling for construction and land reclamation present a considerable threat to seagrass beds. Apart from direct physical damage, the heavy sedimentation associated with these activities negatively affects seagrass beds by physical smothering of the blades, by reduced light penetration, and by bacterial decomposition of the organic sediment which results in reduced O₂ levels and hence reduced photosynthetic rates of the seagrass plants. These effects were well illustrated by dredging and filling operations in Boca Ciega Bay, Florida, reported by Taylor and Saloman (1968). The activities destroyed 80% of the seagrass beds in the Bay, either directly (mechanical uprooting) or indirectly (sedimentation). The conservative estimate of economic loss through loss of fishing and recreational activities was \$1.4 million US per year.

Eutrophication resulting from the input of sewage or other nutrient enriched effluents is also known to negatively affect seagrass beds. The nutrients stimulate growth of epiphytic algae on the blades and increase phytoplankton abundance in the water column. Both processes decrease the light available to the seagrass plants and reduce photosynthesis. A 66% reduction of seagrass beds has been documented over a 17-year period in Christiansted Harbour, St. Croix in response to increasing sewage pollution and dredging in the bay (Nichols et al., 1972). In Cockburn Sound, Australia, Sheppard et al. (1989) found that epiphytes on seagrass blades and filamentous algae growth were greatly enhanced by nutrient enrichment and loose filamentous algae were commonly associated with thinning seagrass meadows. The blades of seagrass plants act as an inert substratum, but not as a nutrient source for the epiphytic algae. Sheppard et al. (1989) concluded that enhanced epiphyte growth in nutrient-enriched waters significantly impaired seagrass photosynthesis through shading and was primarily responsible for the elimination of extensive seagrass meadows in the Sound. Taylor and Bright (1973) documented a major decline of *Thalassia* beds in response to dredging and discharge of domestic and industrial effluents in Hillsborough Bay, Florida.

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2. PROJECT RATIONALE AND OBJECTIVES

The preceeding section reviews the growing number of studies indicating that the impact of anthropogenic activities on coastal marine habitats in the tropics is increasing, that coral reef communities may be particularly sensitive to such impacts, that a major cause of the deterioration of coral reefs is increasing eutrophication and sedimentation of coastal waters, and that a major cause of eutrophication and sedimentation is domestic sewage release. However, several factors continue to constrain attempts to demonstrate impacts of domestic sewage release on coastal marine organisms, and to understand the mechanisms of the impacts. A major constraint is that the typical approach used to date to investigate effects of eutrophication and sedimentation on coral abundance and on coral species numbers and diversity has been a simultaneous spatial comparison of reefs believed to differ in the quality of coastal water to which they are exposed. A problem with this approach is that there may be considerable natural variation within and between reefs in coral population dynamics and community characteristics, and the natural variation may confound attempts to quantitatively correlate coral characteristics with water quality characteristics. This problem can be countered by investigating long-term changes in coral reef characteristics at specific locations differing in exposure to water quality stress. The results of an investigation of long-term (1987-92) changes in coral reef characteristics on the south and west coasts of Barbados are presented in Section 3.

A second factor constraining attempts to understand effects of eutrophication and sedimentation, and hence of domestic sewage release, on corals is that sub-lethal effects of pollutants on coral life history characteristics, such as growth rates, have proved difficult to assess. A major problem is that coral growth rates are characteristically slow, and techniques have therefore not been available which can measure growth rate differences over short time periods. The problem can be countered by developing and using techniques which can investigate coral growth rates in the time frame of weeks rather than years. The results of an investigation of effects of eutrophication and sedimentation, and hence of domestic sewage release, on coral growth rates using a novel buoyant weight technique for short-term assessment of coral growth rates in the field are provided in Section 4.