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CHAPTER 2

POLLUTION AND DEGRADATION OF CORAL REEF COMMUNITIES

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INTRODUCTION

Long lines of offshore breakers, blue lagoons and white sand beaches are often spectacularly visible in the coastal tropics, but the coral reef communities which create them are seldom seen except by the skin diver. To be told that many of these communities are threatened by the pressures of civilization, and that some have already succumbed, is thus not liable to elicit the same concern as when one hears, for example, about the destruction of redwood forests. But there are good reasons for being concerned, reasons that make the fate of the redwoods, despite their beauty and value, seem minor by comparison.

Coral reefs are probably the most extensive shallow marine communities on earth (Goreau, 1961) and among the most biologically productive of all natural communities, marine or terrestrial (Odum, 1971). In the tropics, where man's terrestrial protein sources are often inadequate, reef fishes and shellfishes provide high-quality protein for people living near the sea. Reef communities also abound with species containing a wide range of pharmacologically active compounds (e.g., Baslow, 1969; Arehart, 1971; Hashimoto and Fusetani, 1972; Burkholder, 1974). Reef coelenterates alone afford "a mine of novel compounds of potential value as drugs or as tools for pharmacological research" (Ciereszko and Kams, 1974).

Fringing reefs are self-repairing breakwaters which permit the continued existence of about 400 atolls and numerous other low tropical islands, as well as preserving thousands of miles of continental coastlines. Munk and Sargent (1948) estimated that normal waves dissipate 500,000 horsepower against the windward side of Bikini Atoll. Ladd (1961) states, "the growth forms of the (reef) builders. . . enable them to construct an efficient baffle that, in addition to bringing a constant supply of refreshing sea water to all

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parts of the organisms, robs the incoming waves of much of their force by spreading the water in all directions... These structures are truly worthy of an engineer's respect".

Chave et al. (1972) estimate that the gross production of calcium carbonate by reef communities averages between 100 and 500 tonnes per acre per year. This production goes into the building and maintenance of both the reef and nearby beaches. The recreational values of coral reefs are well known, and their use in this connection is growing rapidly. The uniquely peaceful and beautiful vistas that reefs present to the human visitor is a psychological resource to which any coral reef diver will bear witness.

Corals occupy only a portion (often a minor portion) of reef communities. Large areas of reefs are often covered mainly by a thin layer of algae. This algal zone lacks the spectacular beauty of the coral community and has received comparatively little study. The layman may assume that these communities deserve a low conservation priority. They are, however, an important component of the reef community complex for several reasons. First, they are sometimes even more biologically productive than adjacent coral communities (Smith and Marsh, 1973), and are a source of food for fishes of the coral community who leave its shelter periodically to graze (e.g. Chartock, 1971). In addition, large quantities of algal fragments are removed from the algal community by currents and transported to adjacent coral and other communities where they serve as detrital food (Johannes and Gerber, 1974). Finally, some of these algal communities have been found to be capable of extremely high rates of nitrogen fixation (Webb et al., 1975). This newly fixed nitrogen is distributed through reef food chains via grazers and consumers of algal detritus. This mechanism appears to be important in maintaining the high productivity of reef communities in waters characteristically very low in fixed nitrogen.

To anticipate the impact of man on the reef community, we must have some knowledge of environmental tolerances of the component organisms. Such studies are few, with the exception of a number of investigations on corals themselves. These animals, which dominate the reef visually, do not appear to account for the major fraction of reef community biomass, or energy metabolism (Odum and Odum, 1955). Their role in the reef community is pivotal nonetheless. As White (1969) has said: "All forms of life modify their context. The most spectacular and benign instance is doubtless the coral polyp. By serving its own ends it has created an undersea world favorable to thousands of other kinds of animals."

So central are corals to the integrity of the reef community that when they are selectively killed, migration or death of much of the other reef fauna ensues (e.g., Barnes, 1966, p. 261; Chesher, 1969, 1970). Therefore, the environmental tolerances of the reef community as a whole cannot exceed those of its corals. Accordingly, our knowledge of the environmental limits of corals can provide us with convenient preliminary criteria for set-

ting standards of environmental quality for reef communities. In instances where other vital components of the reef community prove to have significantly narrower stress tolerances, such regulations would, of course, prove inadequate.

SEDIMENTATION

The problem

A number of workers, who will be cited below, have documented the destruction of coral communities by sedimentation. However, others have noted the existence of apparently healthy coral communities in the presence of generally turbid waters (Crossland, 1907; Marshall and Orr, 1931; Verwey, 1931a; Motoda, 1939; Roy and Smith, 1971). Consequently, some have concluded that sediments has been exaggerated as a threat to corals (e.g., Yonge, 1940; Maragos, 1972).

Recent work by Hubbard and Pocock (1972) helps reconcile these conflicting impressions. Working with 6 differentially stained size classes of sediments and 26 species of Caribbean reef corals, they showed that corals are "size-specific sediment rejectors" and that this specificity differs markedly in different species. These characteristics were previously known (e.g., Mayor 1918a; Edmondson, 1928; Marshall and Orr, 1931; Abe, 1937) but had only been demonstrated in a rough qualitative way, and the implications had not been pursued. Silt, the smallest sediment size class used by Hubbard and Pocock, was the only size removed effectively by all species tested. Unlike most larger particles, silt is light enough to be removed by ciliary action.

Differences in sediment rejection were shown to be a function of variation in the polyps' size and distentional capacity and the geometry of the calyx (see also Edmondson, 1928; Marshall and Orr, 1931; Hubbard, 1973). Thus the threat posed by sediment varies with particle size and species of coral. It is worth noting that in this single paper by Hubbard and Pocock, palaeontologists working on problems of interpretation of fossil coral distribution, have provided us with more information concerning sediment rejection by modern corals than all the marine biologists combined who have examined the problem in the past.

The existence of healthy coral communities in chronically turbid waters can also be explained in part if strong currents are present which help prevent the build-up of sediments on coral surfaces (Marshall and Orr, 1931; Verwey, 1931b). Most corals cannot live for long if heavily coated or buried (Wood-Jones, 1907; Mayor, 1918a; Edmondson, 1928; Umbgrove, 1930; Marshall and Orr, 1931; Motoda, 1940). Where corals do survive in turbid waters their growth rates and the depth to which they grow will be reduced

since calcification in corals is light-dependent (e.g. Goreau and Goreau, 1959).

Dodge et al. (1974) found an inverse trend between rates of resuspension of sediments and coral growth in Jamaica. They also noted that high sediment resuspension was related to decreased variability in growth rates. In a detailed field study in Kaneohe Bay, Hawaii, Maragos (1972) found that reduced light intensity due to sewage-induced phytoplankton blooms and suspended silt markedly reduced growth rates of *Pocillopora damicornis* and *P. meandrina*. Of the eight environmental factors Maragos studied, light intensity showed a higher correlation with the combined growth rate of six species of corals than all other factors combined, accounting for 57% of the observed variation. Wells et al. (1973) and J.W. Wells (personal communication, 1974) observed lowered numbers of zooxanthellae and abnormally low photosynthesis/respiration (P/R) ratios in corals living beneath a plume of polluted, highly turbid water leaving Biscayne Bay, Florida. Verwey (1931b) found a clear correlation between the maximum depth of reef growth and water clarity in Batavia (Jakarta) Bay. Thus turbidity, which may not kill corals in shallow water, will reduce their growth rates and bring about their eventual death in deeper water. Coral species diversity has also been observed to decrease with increasing turbidity (Brock et al., 1966; Roy and Smith, 1971). Sedimentation can also be inimical to algae growth in coral communities (Dahl, 1973).

Strong currents carrying sand and other coarse suspended material can inhibit coral growth by abrasion (Bourne, 1888; Crossland, 1907, 1927, 1928; Bertram, 1937; Wiens, 1962; Storr, 1964). Coral planulae cannot settle and survive on an unconsolidated substrate (e.g. Harrigan, 1972). Fine sediments subject to resuspension are inimical to suspension-feeding benthos in general (e.g. Rhoads and Young, 1970) and bring about the destruction of a variety of reef fauna in addition to corals (Bakus, 1968).

The rich and varied biota found in the myriad of caves, tunnels and crevices in the typical reef are undoubtedly more vulnerable to siltation than organisms on the reef surface. Sediments settling out in a reef community tend to fill or plug these cavities. The contribution of this extensive sub-community to the structure and function of the reef community as a whole, while poorly known, is probably large (e.g. Hiatt, 1958).

Near Okha in the muddy Gulf of Kutch, India, high tides produce strong currents which flush sediments from the raised surfaces of flourishing coral reefs. In the areas I examined the typical cavities characteristic of most coral reefs were not apparent; if they existed at all they were filled with sediments. DiSalvo (1969) found that the internal spaces in dead coral heads in Kaneohe Bay contained unusually large quantities of terrigenous sediment in areas influenced by runoff.

The impact of bad land management

Property damage on land is often used as a criterion of whether erosion induced by man's activities has exceeded legally permissible levels. It seems to be generally overlooked that property damage brought about by erosion can and frequently does extend into the sea. Natural erosion is an essential process without which we would have no soil, and the delivery by rivers of nutrient-laden soil to the ocean enhances marine productivity. In excess, however, siltation of erosion products leads to decreased productivity of coral reef communities and, ultimately, to their destruction.

Exposure of reefs to brackish, silt-laden water associated with flood runoff has probably been the single greatest cause of reef destruction historically. Undoubtedly some damage would have occurred in man's absence, but there can be no doubt that bad land management has greatly magnified the problem.

High islands and continental land areas fringed by reefs are frequently areas of high rainfall and lush vegetation, and are thus particularly susceptible to erosion. Kirby (1969) states: "Experiments have shown that the greater the rainfall and the thicker the vegetation (up to a point) the greater will be the acceleration of erosion consequent on stripping the vegetation."

R. Moberly Jr. (1963; and personal communication, 1971) has studied the effects of man's activities on reef sedimentation in Hawaii. Ancient Hawaiians cultivated taro in swampy lowlands, and raised pigs, chickens and dogs either penned in the villages or loose in the forests. After the arrival of Europeans, virtually all native plants below 1,000 m were replaced by invaders., Sheep, goats and cattle were introduced, and upland soils were plowed for sugar cane and pineapple. The impact of the resulting erosion on some reefs has been dramatic. On west Molokai for example, ranching started in 1897 and pineapple-growing in 1918. Maps of the U.S. Geological Survey and old photographs examined by Dr. Moberly show the advance of the shoreline as much as 2 km across the reef flat in this century. The process was still proceeding "rapidly" seaward when observations were made in 1962. In some areas 10—70 cm of red-brown silt now overlie the reef.

The coral reefs in Kaneohe Bay, Hawaii, have been increasingly exposed to sedimentation and low salinities associated with flooding. The bay watershed is scarred by many acres of exposed soil due to land clearing associated with housing and road construction. Laws requiring the replanting of land cleared of vegetation are inadequate and have been poorly enforced. The bay turns red and opaque after every heavy rain. During one rainstorm in 1969 one of the ten streams entering the bay was found to be transporting enough sediment into the bay in 24 hours to cover the entire bay bottom (about 41 km^2) with about 1 kg/m^2 of sediment if spread evenly (calculated from Fan and Burnett, 1969). Surveys of mean bay depth made in 1888 and 1927 yielded similar values. A third survey in 1969 showed that subsequent accel-

erated sedimentation had reduced the mean depth by more than 1.5 m (Roy, 1970).

Fine, clay-sized, terrigenous sediments have killed many corals growing below 6—9 m (Maragos, 1972), particularly in the southern end of the bay, where runoff is high and circulation is slowest (Bathen, 1968), and on reefs fringing the shoreline. Sediments here are trapped between coral heads on the reef slopes. Settling of planulae is thus poor or absent, growth of existing corals is slow, and only a few older corals remain alive in many areas. DiSalvo (1972) attributed pitting and fouling of crab carapaces in inner Kaneohe Bay to siltation. Sedimentation, sewage and dredging have combined to reduce viable reef slopes in the bay to 29% of their pre-1939 coverage (Maragos, 1972).

Fairbridge and Teichert (1948) suggested that because of "colossal soil erosion due to unplanned agriculture", continental sedimentation appeared to be gaining on coral growth in the vicinity of the Low Isles, Great Barrier Reef. On the seaward slopes of some of these reefs they reported that much of the coral was dead and covered with a fine coating of terrigenous sediments. Stephenson et al. (1958) note "the widely held opinion that floods have become increasingly severe in North Queensland owing to land clearance". "If so", they state, "a new element — human interference — has entered the ecological complex, and coral growth may never attain its previous luxuriance" (see also Thompson, 1971).

Van Eepoel and Grigg (1970) report that in large areas of Lindberg Bay, St. Thomas, Virgin Islands, most corals and other sessile animals have been destroyed and conditions remain unsuitable for their establishment due to sedimentation caused by bulldozing, construction and the surfacing of land which drains into the bay. A subsequent survey indicated that conditions were rapidly worsening (van Eepoel et al., 1971). Damage to reef communities due to accelerated terrigenous sedimentation has also been observed in Tanzania by Ray (1968), and in the Seychelles by Vine (1972).

Although sedimentation due to bad land management is very likely the biggest pollution threat to coral reefs, there seems to be very little that marine scientists can do about it in most cases. The terrestrial damage done by accelerated erosion is often even greater than that done to marine communities — yet efforts at soil conservation by terrestrial resource managers have, in the tropics, usually been dismal failures. It is very doubtful that significant progress can be made with this problem while burgeoning human populations create accelerating and irresistible pressures on tropical soils.

Dredging

Fine sediments carried into coastal waters by flooding are often concentrated initially in the upper low-salinity layer (Emery, 1962; personal observations). To some extent this facilitates diffusion and dilution before the

sediments settle. Such is not the case when sediments are suspended by dredging, a related source of reef pollution. Levin (1970) provides a useful review of the pertinent literature. Brock et al. (1966) give a detailed account of the destruction of corals and reduction of fish and echinoderm populations at Johnston Island due to siltation brought about by dredging. Over 2,800 ha of reef and lagoon were more or less affected by silty water and 440 ha of reef totally destroyed through dredging and filling. Deterioration of reef communities continued for at least a year after dredging ceased because of continual resuspension of sediments.

In Water Bay, St. Thomas, Virgin Islands, Grigg and Van Eepoel (1970) observed the destruction of hard corals due to sedimentation associated with the release of clays brought about by the dredge removal of overlying sand. Since no exploratory cores were taken, the presence of the clays was not detected in advance of dredging.

In this and related surveys, personnel at the Caribbean Research Institute have found in the U.S. Virgin Islands a total of "several miles of dead reefs and bay with impoverished fauna and flora as a result of ill-planned and ill-managed dredging operations" (Grigg, 1970; see also Grigg et al., undated). Grigg et al. (undated) report having observed dredge-suspended fines remaining in suspension or continually being resuspended for up to five years in the Virgin Islands. These workers have proposed "a total ban on further inshore dredging excepting for improvement and maintenance of harbors and channels" (Grigg, 1970).

In Castle Harbor, Bermuda, an airfield was built between 1941 and 1943 by dredging and filling. The bay's population of brain corals, *Diploria*, the dominant coral genus on Bermuda reefs, was destroyed (J. Burnette-Herkes, personal communication, 1971). Dead colonies up to 1 m in diameter can still be seen today. The sediments produced by dredging were still being resuspended in large quantities in 1969 and no significant recolonization by *Diploria* had occurred (personal observations).

In 1939, 29% of the coral reefs in Kaneohe Bay, Hawaii, were removed by dredging. Some partially dredged reefs have recovered as a result of the encroachment of corals from adjacent undamaged areas (Maragos, 1972). None of the patch reefs that were dredged completely shows signs of growing back to the surface however (Roy, 1970).

Siltation in Guam, brought about by dredging, destroyed reef flat communities in "much larger areas than were destroyed by the physical removal of substrate" (Marsh and Gordon, 1974).

The injury of destruction of reef communities due to dredging has also been observed in the port of Singapore (Johnson, 1970) and nearby straits (Wycherly, 1969), the Seychelles (Vine, 1972); Hawaii (Sullivan and Gerritsen, 1972; Tepley and Johannes, 1972), and American Samoa (Sverdloff, 1973).

Dredge-suspended sediments exert both biological and chemical oxygen

demands (e.g., Brown and Clark, 1968; Frankenberg and Westerfield, 1968). Since coral reef communities may not be able to tolerate oxygen levels much lower than the normal nighttime minima to which they are exposed (e.g. Kinsey and Domm, 1974; Johannes and Betzer, Chapter 1), the scavenging of oxygen by suspended sediments may constitute a significant stress. No research seems to have been done on this problem as it relates to coral reef communities.

Alteration of reef topography through dredging, filling or underwater construction will alter current velocities and directions. Since the zonation of corals (e.g. Hubbard, 1974) and other reef biota is clearly related to current regimes, current alterations will affect settling and survival patterns.

Ingle (1952) states that dredging should be "under the constant surveillance of persons capable of evaluating the conditions of the operation. Frequent checks by silt collectors placed at various distances from the dredge and simple biological checks would constantly be made of the deposits at various distances from the dredge... In this way it would be possible to acommodate the activities of the dredge to momentary conditions." Using this approach, dredging close to reef communities might be accomplished without causing serious damage — activity could be restricted to periods when the currents carried suspended materials away from the reef and dispersed them adequately.

The gathering of corals or dredging of reef substrate for use as construction material is common in some areas (Phelan, 1952 and references therein; McNeil, 1951; Beaven, 1962; Walker, 1962; Porter and Porter, 1973). In Malaysia coral mining is done by looping a cable around live coral heads of some tonnes and winching them across the reef front and down the reef slope until they are beneath a vessel from which they are winched aboard. On a "good day" 450 tonnes of coral heads per vessel are mined in this fashion. Each coral head thus removed destroys, on the average, an area 3 m wide and 7.6 m long. In a day 0.13—0.2 ha per vessel is destroyed (Lulofs, 1974).

Casciano (1973) describes a new dredging system which under some conditions should provide a more economical and less biologically destructive means of recovering offshore sand than conventional systems. The suction head is buried in the sand rather than held above it, thus reducing the turbidity accompanying its operation.

Sugar mill wastes

Sugar mill wastes are another source of sediment damage on reefs in Hawaii and parts of the Caribbean. In Hawaii harvesting of sugar cane is done using a modified bulldozer blade to break the cane stalks and roll them into bundles. The bundles contain quantities of soil which, along with some cane leaves, are washed from the cane and then often dumped directly into the

ocean. After the cane is cleaned and the juice pressed from it, the waste stalk and cane pulp, which is known as "bagasse", is also often dumped into the sea. Hundreds of tonnes of rocks, silt and bagasse have been dumped daily into Hawaiian coastal waters by sugar mills (Copeland and Wilkes, 1969; Burm and Morris, 1971). The authors noted one sludge bank which extended for 400 m radius from a mill outfall. The water was extremely turbid in the vicinity. Numbers and diversity of corals and fishes were reduced and there was an overgrowth of benthic algae.

Grigg (1972) found a considerable reduction in abundance and diversity of corals and fish in the vicinity of a number of sugar mill outfalls, although the fish were significantly larger than in control areas. Grigg employed an interesting method of measuring fish availability. He hired a commercial diver to spear commercial-sized fish as fast as he could for 30-min periods in polluted and control areas.

Yamaguchi (1950) described the effect on corals of siltation associated with sugar mill wastes on Kauai, Hawaii. Because of turbulence the silt did not settle out in waters less than 12 m deep. Below that depth siltation and the destruction of corals and associated fauna was pronounced.

SEWAGE

Kaneohe Bay

Although numerous reefs throughout the tropics are exposed to sewage, the problems this creates have gained attention only in the last few years. Kaneohe Bay, Hawaii, is the best studied example of the effect of sewage on reef communities. For a comprehensive summary of pollution research here see Smith et al. (1973).

Edmondson (1928) once described this bay as "one of the most favorable localities for the development of shallow water coral. Nearly all the reef-forming genera known in Hawaii waters are represented . . . and many species grow luxuriantly". It is also the most accessible reef complex in the central Pacific and has probably been the most valuable for recreational purposes.

The south end of the bay, which receives secondarily treated domestic sewage, is a shallow basin of about 880 ha in surface area and contains patch and fringing reefs. More than 99% of the corals in this basin are now dead. Living specimens of the formerly dominant reef-building species in this area now die within a few weeks when placed there (Maragos, 1972).

Survival of five out of six reef coral species transplanted to various locations in the bay showed a high negative correlation with sewage-related variables — turbidity and phosphate content of the water. The exact cause of death however was neither phosphate levels nor turbidity. Maragos (1972)

suggests that anaerobic conditions in the sediments leading to the release of hydrogen sulphide may have been responsible. Sorokin (1973) reported an expanding anaerobic layer, increasing sulphide levels, and dropping redox potentials in sediments with increasing proximity to the main sewage outfall in Kaneohe Bay (see also DiSalvo, 1971). The use of sediment characteristics such as these as a rapid means of tracing the extent of the impact of sewage on benthic communities deserves more attention.

Dictyosphaeria cavernosa is a reef alga distributed circumtropically but seldom constituting an important component of reef flora. In Kaneohe Bay, however, *D. cavernosa* has become "the major benthic organism on the lagoon slopes, having surpassed and caused the decline of corals in recent years" (Maragos, 1972). The accelerating growth of this alga in the bay has paralleled the increasing quantities of nitrogen and phosphorus entering the bay in sewage (Banner and Bailey, 1970). Between 1966 and 1970 dissolved phosphate levels near the main sewer outfall increased at an average rate of 0.75 μg-atom/l (Caperon et al., 1971).

D. cavernosa or "green bubble alga", first colonizes the bases and crevices of coral heads, then extends upward engulfing entire colonies in the bay. Finally, it spreads in thick, heavy sheets fusing with other colonies to blanket large areas of reef. Shut off from light, oxygen and food the enveloped corals and many of the plants and animals associated with them die. Wherever this alga comes into physical contact with corals, the skeletons erode by some unknown process (Banner and Bailey, 1970), and the reef surface begins to disintegrate. Reef fish populations are often strikingly reduced in such areas (personal observations).

Sewage nutrients have stimulated extremely dense growths of *D. cavernosa* in the middle and parts of the northern bay, some distance from the sewage outfalls, but not in the south basin nearer the outfalls. This may also be due to the release of hydrogen sulphide from sediments near the outfalls.

Sewage and oxygen levels

The connection between organic or nutrient enrichment and lowered, stressful oxygen levels is too well known to need elucidation. Johannes and Betzer (Chapter 1) have pointed out that tropical marine organisms live closer, on the average, to their lower oxygen limit than biota in colder waters. D.W. Kinsey (1973; and personal communication, 1974) observed that oxygen levels artificially depressed only slightly below normal levels on a healthy reef caused the death of many reef inhabitants. Depressed oxygen levels in reef and near-reef environments subjected to sewage effluents have been reported by Bathen (1968) and Wade et al. (1972).

Unfortunately oxygen has been measured only during the day in many pollution studies. As Vaughan and Wells (1943) pointed out, even under natural conditions in reef communities "the quantity of oxygen in the water

over reef communities varies greatly between day and night". In enriched waters, rapid photosynthesis may produce high daytime oxygen levels in the same community where high respiration rates produce abnormally low night-time levels. Thus, measurements limited to daylight hours can lead to false confidence in the state of the system.

Bacterial water quality

The bacterial standard for marine bathing waters suggested by the World Health Organization is less than 350 fecal coliforms or 1,000 total coliforms per 100 ml. Siren and Scheuring (1970) found fecal coliform bacteria in numbers averaging 11 million per 100 ml in inshore reef and lagoon waters near sewage outfalls in all six district centers in the Trust Territory of the Pacific Islands. According to Garner (1972), "certain lagoons *were* posted off limits after the report (Siren and Scheuring, 1970) came out — but the signs didn't last long and pollution continues". In one of these district centers, Koror, Palau, Hardy and Hardy (1972) found fecal coliform levels reached 34 million per 100 ml near a hospital outfall.

Barnes (1973) found coliform levels "grossly exceeding" World Health Organization standards in the vicinity of a number of hotel sewage outfalls in Jamaica. Total coliform counts consistently higher than 240,000 per 100 ml were reported in Kingston Harbor, Jamaica, in 1967 (Wade et al., 1972). Dong et al. (1972) reported high fecal coliform levels in Christiansted Harbor, St. Croix. Moreau et al. (1971) reported comparatively low levels of fecal coliforms in Tahiti lagoon.

Human sewage is often combined with sugar cane wastes. This practice results in high concentrations of sewage bacteria which thrive in the sugar-rich waste, according to Copeland and Wilkes (1969). They report fecal coliform levels of 10,000 per 100 ml off a sugar mill outfall in Hawaii.

Sewage treatment

Little research has been done on sewage treatment and disposal as they relate to tropical coastal areas. Laevastu et al. (1964) concluded that in the Hawaiian Islands "there may be a very few sites at which an offshore set of the current is so predominant that untreated sewage may safely be introduced through outfalls of normal length. In general, however, alongshore and seaward sets occur, especially in the surface layers due to wind action, with sufficient frequency as to make inadvisable large discharges of untreated sewage within a few miles in any direction from a shoreline used intensively for recreation or fishing."

Aquatic ecosystems can tolerate some nutrient enrichment without serious ecological dislocations (e.g. McAllister et al., 1972). Experimental enrichment of a coral reef community with low levels of inorganic phospho-

rus and nitrogen increased primary productivity without affecting trophic structure or biomass noticeably (Kinsey and Domm, 1974). The release, in moderate quantities, of sewage effluent which has received adequate secondary treatment could conceivably enhance reef productivity without altering community structure unacceptably. Acceptable levels of enrichment cannot be defined, however, without much more research.

Biological disc filtration units were tested successfully in Thailand by Pescod and Nair (1972). High tropical temperatures favored anaerobic conditions in the units. The system has the ability to absorb shock loading and nutrient-deficient wastes. It requires comparatively little land and capital and is simple to operate.

Shatrosky et al. (1972) and Grigg et al. (1971) discuss operating efficiencies of sewage treatment plants in the Virgin Islands. Rum distillery wastes present special problems for waste treatment because of high BOD, COD, low pH, high dissolved solids and nutrient imbalance, and are the cause of "severe coastal pollution" in the Caribbean (Burnett, in Shatrosky et al., 1972).

Chuck and Lum (1971) suggest that disposal of sewage effluent in coastal limestone aquifers on tropical islands may sometimes be a practical alternative to marine disposal.

Chan (1973) has described how copra farming in the Pacific islands has led to sewage pollution of reef lagoons. To leave the land free for coconut plantations, people who once lived scattered about now concentrate in villages. Whereas pit latrines once took care of human excreta, sewage must now be exported and is generally dumped into the lagoons. This appears to have led to serious decreases in fish and shellfish production as well as the removal of already scarce plant nutrients from the terrestrial ecosystem.

Chan (1973) and Marshall (1969) describe simple, inexpensive sewage treatment units for use on tropical islands. In each case fermentation produces methane which can be bled off and used for cooking or light. When fermentation is complete the residue is sanitary and useable as fertilizer. The effluent, Chan suggests, may be used in specially constructed ponds for aquaculture.

Recently, a number of proposals have been made for raising various marine animals commercially in atoll lagoons using the entire lagoon (e.g., Pinchot, 1966, 1970; Inoue, 1968; Isaacs and Schmitt, 1969). This can be expected to produce increases in lagoon nutrient levels either deliberately through fertilization or involuntarily through the increased production of animal excretions containing nutrients derived from food brought from elsewhere. The fate of Kaneohe Bay and the problems in Pacific lagoons noted above should be taken into consideration in this context.

Some of these plans require the closing of lagoon passes to prevent the escape of the animals being raised. Because most corals require fresh, well-circulated waters, this may have serious consequences. Restricting lagoon

circulation in this manner at Palmyra Atoll completely destroyed its lagoon reef communities (Dawson, 1959). At Kolonia, Ponape, Caroline Islands, a causeway built to an airfield in 1970 has blocked the normal flow of water in the bay and siltation is destroying the reef communities (L.G. Eldredge, personal communication, 1973).

In predicting the net value of lagoon aquaculture the possible loss of already productive reef communities must be considered among the costs. In some cases this cost may be acceptable, but this should not be assumed in the absence of careful evaluation.

Constructing special enclosures of limited size in tropical inshore water to utilize the nutrients in sewage and industrial effluents (e.g., Gundersen and Bienfang, 1972; Chan, 1973) should produce less ecological dislocation and allow simultaneously for greater control over the species being farmed.

Chlorination

The effect of free residual chlorine on marine organisms is poorly known. It is a potent fertilization inhibitor in marine invertebrates at concentrations as low as 0.05 mg/l available chlorine (Muchmore and Epel, 1973), and lethal to marine fish at levels as low as 0.024 mg/l (Alderson, 1972).

Sewage treatment workers are often inadequately trained and sometimes use chlorine well in excess of prescribed levels, particularly when sludge is discharged, during floods when sewage may bypass normal treatment, and during "shock chlorination", the periodic dosing of industrial effluents with chlorine to prevent biological fouling.

Marsh and Gordon (1973) noted a chlorine level of more than 5.0 mg/l in the water adjacent to a power plant outfall in Guam on one occasion. Observed fish kills in the vicinity appeared to be caused by such sporadic high chlorine levels. In the outfall plume of another Guam power plant, Jones and Randall (1973) noted that when the plant ran out of chlorine nine species of reef fishes began colonizing the reef flat and stilling well. When chlorination recommenced all nine species disappeared immediately.

Davis (1971) found that the planulae of three species of Hawaiian corals were more tolerant to chlorine than most marine forms previously tested. Exposure of all three species to 0.49 mg/l for up to 7 hours did not prove lethal. Davis points out that adult corals cannot be assumed to be as tolerant as planulae.

Eutrophication-loving organisms

Despite its ubiquity in coral reef communities, *Dictyosphaeria cavernosa* has not "taken over" in any area other than Kaneohe Bay, Hawaii (see above) to my knowledge. However, the increased dominance of a number of other algal species in response to sewage effluent has been observed.

Soegiarto (1972) noted the association of sewage pollution and heavy growth of *Acanthophora spicifera*, an alien species which arrived in Hawaii in about 1951 (Doty, 1961). (Soegiarto's work constitutes probably the first quantitative measurement made of the effect of water quality on standing crop and species composition of benthic algae in a tropical coastal area.)

Enteromorpha is a genus which thrives in tropical marine areas polluted by sewage (Dong et al., 1972; Tewari, 1972; personal observations), as is the genus *Ulva* (Doty, 1973a; personal observations). Littler (1973) cites evidence that sewage stimulates the growth of various coralline algae at the expense of coral growth.

The tubiculous polychaetes *Spiochaetopterus oculatus* and *Capitella* sp. were observed to be indicators of high organic pollution by Wade et al. (1972) in Kingston Harbor, Jamaica, and by McNulty (1970) in Biscayne Bay, Florida. Sponges appear to increase in response to particulate organic enrichment in the tropics. Dong et al. (1972) noted that the sponge *Chondrilla nucula* increased in abundance, with a concomitant reduction in the coral *Acropora palmata*, with increasing proximity to a sewage outfall in Christiansted Harbor, St. Croix. Burm and Morris (1971) found an increase in sponge density in the vicinity of a sugar mill waste outfall in Hawaii. In Kaneohe Bay, Hawaii, a variety of species of sponges have increased greatly in abundance on the reef slopes in the vicinity of sewage outfalls in recent years (personal observations).

The holothurian *Ophiodesoma spectabilis*, although common in Kaneohe Bay before the introduction of sewage (MacCaughey, 1918), increased considerably in numbers and range between 1959 and 1973 in areas affected by sewage (personal observations).

Sewage outfalls often attract fish because of enhanced plant growth and sewage particulates which serve as food. As a consequence sewage outfalls sometimes become favored fishing grounds. Some fishermen consume or sell the fish they catch in such areas. I know of no studies of the health hazards involved as they relate to shallow tropical marine waters. Hardy and Hardy (1972) reported high coliform concentrations in oysters and crabs near a seawage outfall in Palau.

OIL POLLUTION*

The problem

Moore et al. (1973) provide a useful summary of the composition and

* An exhaustive and important document dealing with oil pollution and coral reefs, which became available after this book went to press, is: *Royal Commissions Into Exploratory and Production Drilling for Petroleum in the Area of the Great Barrier Reef.* Volumes I and II. Australian Government Publishing Service, Canberra, A.C.T., 1050 pp.

structure of hydrocarbons, their solubility in seawater, their volatility and their susceptibility to biodegradation. Baker (1971) relates structure to toxicity. Generally the low-boiling aromatics in petroleum are most immediately toxic (Blumer, 1971). They are also the most volatile and so are among the first components of crude oil (by far the most abundant type of oil spilled) to evaporate. Floating oil thus rapidly decreases in toxicity.

Not all oil remains afloat however. Some oil compounds are soluble — among these are some of the most toxic fractions (e.g. Mironov, 1972). In addition, some heavier oil residues sink readily, while lighter oils may quickly absorb to suspended sediment particles (Hawkes, 1961) and enter the food chain via suspension- and deposit-feeders. Petroleum degrades very slowly when trapped in marine sediments (Blumer and Sass, 1972).

Blumer et al. (1973) describe the weathering history of a light paraffinic crude oil that stranded on a Bermuda beach. After thirteen months the spill residue was still "far from being inert asphalts", and was not depleted of its biologically most active higher molecular weight compounds. During World War II a tanker filled with marine fuel was wrecked on a reef near Japtan Island, Enewetak. Today (1974) many porous rocks and boulders near the remains of the ship are still heavily impregnated with tarry residues from the spill (personal observations).

Oil and corals

To date there appears to be no conclusive evidence that oil floating *above* reef corals damages them. Grant (1970) floated Moonie crude oil in a vessel over specimens of the coral *Favia speciosa*. The corals showed no visible signs of injury during six days of subsequent observation. Johannes, Coles and Maragos (unpublished) floated five different types of oil over groups of the Hawaiian corals *Porites compressa*, *Montipora verrucosa* and *Fungia scutaria* for 2.5 h. During 25 days of subsequent observation no visible evidence of damage was noted.

Lewis (1971) found that four species of coral exposed to crude oil or an oil spill detergent exhibited ruptured tissue and other signs of distress. These experiments were carried out in sealed vessels in order to prevent the escape of volatile oil fractions. Since the harmful effects may have been due to these volatiles it cannot be assumed that the results are representative of what might occur during an oil spill when the volatiles would have escaped to the atmosphere.

Published reports of oil spills in the vicinity of coral reefs also provide no conclusive evidence of injury to corals. In connection with a Panama spill where inshore littoral fauna suffered heavily, Rützler and Sterrer (1970) state that corals escaped readily observable damage, presumably because they were continually submerged. Spooner (1970) observed no damage to reef communities in Tarut Bay, Saudi Arabia, in an area of long-term chronic

oil pollution. Shinn (1972) states that staghorn corals grow "apparently unaffected" around loading terminals for oil tankers in the Persian Gulf and near the entrance to the Suez Canal in the Red Sea.

Unlike reef corals, the alcyonarian coral *Heteroxenia fuscescens* has been shown to be injured or killed by crude oil floated above it. This species proved more resistant to crude oil under conditions of continuous flow in large tanks than under static conditions in small jars (Cohen, 1973). The results of this work suggest that the farther specimens are held below the surface film of oil, the less effect the oil has on them. Cohen (1973) also found that an oil spill dispersant was at least 100 times as toxic to *H. fuscescens* as two species of crude oils tested. The tissue of this species incorporated small amounts of components of the crude oil floating above it.

In some areas reef surfaces are exposed during low spring tides. This phenomenon, which is common on Indo-Pacific reefs, was taken into account in an experiment carried out by Johannes et al. (1972). They exposed 22 species of corals, whose upper portions were above water, to Santa Maria crude for 1.5 hours. Oil adhered to portions of the surfaces of most species. Tissue death ensued within a few days in these areas but not on areas where the oil did not adhere. The most extensive experimental work to date on the impact of oil on various tropical marine species and communities is that of Birkeland et al. (1973). Although it will be described briefly here and in Odum and Johannes (Chapter 3), it is difficult to summarize, and the original publication should be obtained by anyone interested in the subject. Birkeland et al. (1973) observed that corals exposed directly to diesel or Bunker C fuel showed responses, depending on species and experimental conditions, that ranged from a mere decrease in growth rate, in some cases, to death after a few days to several weeks, in others. The work of these authors points up the desirability of long-term experiments; some of the effects on coral were first noted more than one month after exposure to oil.

Shinn (1972) noted that "*Montastrea annularis* can survive two hours total immersion in Louisiana crude while *Acropora cervicornis* exposed for two hours to a mixture of seawater containing one part crude to 6 to 12 parts seawater caused immediate retraction of polyps but complete recovery in 24 hours". He does not state whether subsequent observations were made on the condition of these corals. It seems premature to state, as Shinn does, that "it would seem safe to conclude then that crude oil spills do not pose a significant threat to Atlantic reef corals".

Other reef organisms

Although he makes no mention of corals, Gooding (1971) reports the extensive destruction of associated reef fauna in connection with the release by a tanker of 6 million gallons of assorted oils off the harbor entrance at Wake Island. Of the reef fishes that died, 2,500 kg washed ashore. Numerous reef invertebrates were also killed.

Diaz-Piferrer (1962) briefly described the effects of a large crude oil spill from a ship that ran aground on a reef off Guayanilla Harbor, Puerto Rico. Extensive mortality of marine fauna and flora occurred in sublittoral and intertidal zones and beach erosion was extensive. There was "obvious depletion of fisheries". Mohammad (1974) describes the effects of chronic oil pollution on a polychaete in Kuwait.

Birkeland et al. (1973) observed that algal diversity and productivity increased on artificial substrates and in intertidal reef flat communities exposed to Bunker C fuel. The damage caused to the intertidal fauna in meter square quadrats exposed to diesel or Bunker C fuel oil was relatively small and short-lived relative to observed responses in comparable temperate marine communities exposed to massive oil spills. Birkeland et al. (1973) suggest that this may be an artifact of the small areas exposed to oil in these experiments, and that realistic oil pollution experiments may entail exposing areas on the scale of hectares rather than square meters.

Communities may be damaged by oil, from man's viewpoint, even when their structure is unaltered, since commercial species can become tainted with oil and therefore unsaleable (Grant, 1969; Connell, 1971; Deshimaru, 1971; Sidhu et al., 1972).

It is well known that dispersants used to control oil spills are often toxic — sometimes more so than the oil on which they are used. Observations on the effects of oil dispersants on tropical marine organisms have been made by Cerame-Vivas (1968), Cohen (1973) and Lewis (1971). Tests run during the Ocean Eagle oil spill in Puerto Rico indicated that dispersants (emulsifiers) created more damage than they prevented by dispersing the oil through the water column and by causing serious beach erosion through reduction in the physical attraction of sand grains to one another. Adsorbants, which facilitated the collection and removal of the oil, proved more useful. Ekoperl-33 (volcanic ash treated with silicone to enhance absorbance and retard sinking) proved the most desirable of several absorbants tested. It spontaneously immobilized large quantities of oil and was susceptible to only slight leaching (Cerame-Vivas, 1968).

Considerable effort has gone into the planning of baseline studies in temperate marine areas so that, in the event of an oil spill, the state of the affected community prior to the spill is known. The availability of personnel to make such studies is very limited in most tropical areas. Accordingly it does not seem practical to devote much effort to this type of study except in very high risk areas. Efforts might better be spent equipping and training mobile teams who can respond within a few hours to an oil spill.

Oil and oxygen

As discussed in the section of this paper on sewage pollution, even a small reduction in dissolved oxygen levels may, under some circumstances, bring

about considerable destruction of reef fauna. Blumer (1971) points out: "the oxygen requirement in bacterial oil degradation is severe; the complete oxidation of 1 gallon of crude oil requires all the dissolved oxygen in 320,000 gallons of air saturated sea water." Therefore, bacterial degradation may cause additional ecological damage through oxygen depletion.

Oil films may also cut down oxygen transfer across the air—water interface. Kinsey (1973) reviews the general literature on this subject and describes the results of experiments he carried out on a coral reef with Moonie crude oil. Layers of this oil up to 0.7 mm thick did not interfere measurably with atmospheric exchange except insofar as they calmed the water surface somewhat, reducing the surface area available for gaseous exchange. As Kinsey (1973) states, it would be useful to study films of much greater thickness and films of higher viscosity.

Future problems

Extensive and expanding offshore drilling of oil is planned or underway in various parts of Southeast Asia, in the Persian Gulf, the Caribbean, Fiji, Tonga and in other reef areas, often with no prior environmental impact analysis. Australian public opinion forced the government to suspend oil exploration permits for the Great Barrier Reef until the threats these activities may pose can be evaluated.

Transport and refining of oil in the Caribbean is widespread and expanding rapidly. Here in 1971 there were 62 refineries situated in 17 different countries. A "spectacular increase" of movement of crude oil through the region is predicted (Kesterman and Towle, 1973). According to these authors, "from Bermuda to Anegada, to Aves to Trinidad, to Bonaire, even to remote beaches, these areas are plagued with oil drifting ashore from carelessly and illegally pumped tanker bilges".

THERMAL POLLUTION

Laboratory tolerance tests

The threat of destruction or alteration of marine communities by heated effluent from power plants and other industrial installations is greatest in the tropics. Unlike the biota of temperate and polar regions, tropical organisms characteristically live at temperatures only a few degrees below their upper lethal limit (Mayer, 1914). Edmondson (1928) showed that 11 of 13 species of Hawaiian corals survived less than 24 hours at 32°C — a temperature only five to six degrees higher than ambient summer water temperatures. Similarly narrow ranges exist between maximum ambient temperatures and lethal temperatures for corals in Guam (Jones and Randall, 1973), on the Great Barrier

Reef (Mayor, 1918a), the Caribbean (Mayor, 1918b), and Samoa (Mayor, 1924). In the tropics, in contrast to higher latitudes, there are few species capable of replacing a community eliminated by thermal stress.

Some corals which live naturally in exceptionally warm waters have been reported from the Persian Gulf (Kinsman, 1964) and the Laccadives (Gardiner, 1903). The ranges between maximum environmental temperatures and lethal temperatures are probably even smaller here.

Zieman and Johannes (Chapter 5) discuss temperature tolerance levels of tropical marine organisms other than corals.

Lethal temperatures as determined in the laboratory during short-term experiments must be extrapolated to nature with caution. Organisms may not be able to tolerate temperatures as high in nature as they do under controlled conditions in the laboratory (e.g., Read, 1967; North et al., 1972). Also as Mayer (1914) pointed out, the temperature at which the feeding reactions and normal metabolic processes cease are more significant than death temperatures, "for naturally an animal cannot survive long in water in which it can neither move nor function".

It has been established that sublethal heat stress is deleterious to corals for a number of reasons. Mayor (1918b) reported that three species of Caribbean corals ceased to feed at temperatures 1.5—3.0°C lower than their lethal temperatures. Edmondson (1928) reported similar results for Hawaiian corals and noted that reproductive rates also dropped at high sublethal temperatures. Yonge and Nicholls (1931) observed that corals extrude their zooxanthellae in strings of mucus at high sublethal temperatures. Coral planulae may not settle at high sublethal temperatures (Edmondson, 1946).

Gross photosynthesis generally exceeds respiration in hermatypic corals. With increasing temperature, photosynthesis would have to increase faster than respiration for maintenance of a stable photosynthesis/respiration (P/R) ratio. It does not: Coles (1973) found that P/R ratios usually decreased with increasing temperature in several species of corals from Hawaii in the temperature range of 18—31°C. (Similar results were obtained by Hohman and Tsuda (1973) working with the reef alga *Caulerpa racemosa.*) A direct relationship was indicated between the temperature effect on the P/R ratio and the upper thermal tolerance of the species. An inverse relationship between high temperature tolerance and respiration rate was also observed.

Coles found that high temperature tolerance in *Montipora verrucosa* decreases as the salinity was lowered below ambient, but did not decrease when the salinity was raised to 40% (see also Edmondson, 1928). Coles (1973) discusses a variety of observations suggesting that Hawaiian corals have lower upper thermal limits and metabolic optima than corals in warmer waters.

Field observation

In shallow tropical bays and back-reef areas where water is heated rapidly

by the sun, growth of various marine plants and animals may be prohibited by high temperatures even under normal conditions. Mayor (1918a), for example, showed that high temperatures within 150 m of shore inhibited coral growth in the Torres Straits. Wells (1952) drew similar conclusions concerning coral growth at Arno Atoll. Mass mortalities of intertidal and shallow subtidal species in the tropics due to high temperatures and desiccation associated with low midday tides are described by Mayer (1914), Yonge and Nicholls (1931), and Glynn (1968). Obviously *any* increase in summer water temperatures brought about by thermal effluent released at the water's edge in such environments will expand the area inimical to growth.

Thermal effluent from a power plant at Turkey Point, Biscayne Bay (Florida) has in the recent years killed virtually all plants and killed or greatly reduced animal populations in the bay in an area circumscribed closely by a +4°C isotherm (Roessler and Zieman, 1969; Thorhaug et al., 1973a; Zieman and Wood, Chapter 5). Mortalities extend for more than 1.5 km from the outfall.

Corals were among the most sensitive species involved, being killed by the effluent at greater distances from the outfall than most other organisms (Anonymous, 1970). Colonies of the corals *Solenastera hyades* and *Siderastrea siderea* have been eliminated in the area noticeably effected by the heated effluents. Even in areas of marginal influence on other biota many individuals of these two coral species were dead or visibly damaged. Here living *S. siderea* specimens were pale and appeared to be more susceptible to overgrowth by algae and encrustation by other forms (L.L. Purkerson, personal communication, 1972).

Heated effluent from a power plant on Guam led to the destruction of reef margin corals in a zone of 4320 m^2 and damage to a peripheral area including a total of 10,000 m^2 (Jones and Randall, 1973). Fishes, crustaceans, echinoderms and most species of benthic algae also disappeared. Determining the specific cause(s) of these mortalities was complicated by the fact that significant levels of chlorine were added periodically to the cooling water as a desliming and anti-fouling agent. The effluent also contained elevated levels of copper. Jones and Randall (1973) described the relative tolerance of many species of corals, fish and algae to the plant effluent. Large numbers of juvenile rabbitfish were also killed by entrainment in the cooling water.

Jokiel and Coles (1974) studied the impact of thermal effluent on corals in Hawaii. Nearly all corals in water 4—5°C above ambient were killed. A 30% increase in heated discharge resulted in an increase in the area of dead or bleached corals by 100%. The relative heat resistance of different coral species observed in this field study coincides with the results of Edmondson's (1928) laboratory studies. Jokiel and Coles (1974) data also support Mayer's (1917) suggestion that the ability of coral species to resist heat stress is inversely related to their metabolic rates.

Thermal plumes from power plants have been observed to attract reef fish (DeSylva, 1969; Clarke et al., 1970a,b; McCain and Peck, 1973; Jokiel and Coles, 1974). One might reasonably surmise that fish are not liable to seek such elevated temperatures if they are harmful. However, there is evidence that weight loss, lowered reproductive success, higher disease rates and increased mortality can result from prolonged exposure to temperatures only slightly higher than ambient (Coutant, 1970; see also Chesher, Chapter 6). Sharks seem particularly attracted to thermal plumes in tropical waters (DeSylva, 1969; personal observations); thermal effluent may thus sometimes constitute a distinct *human* health hazard.

ABNORMAL SALINITIES

Hypersaline effluents

Although some corals grow in ambient salinities as high as 48% (Kinsman, 1964), many corals are particularly sensitive to salinities slightly above normal oceanic values. Nine out of twelve Hawaiian species tested could not tolerate salinities of even 110% of normal for more than two weeks. Three species died within three days. Most species died within 24 hours when exposed to salinities 150% of normal (Edmondson, 1928). Goldberg (1974), who studied the effects of altered salinities on six species of gorgonians, concluded that they are more stenohaline than scleractinian corals.

Effluents from desalination plants are characterized by elevated levels of not only salinity but also temperature and toxic metals such as copper, zinc and nickel. They may also contain various chemicals added for pH and corrosion control (Zeitoun et al., 1969a). These effluents are often sufficiently saline that, despite their elevated temperature, they sink and flow along or near the bottom where there is no loss of heat directly to the atmosphere and where their potential for damage to benthic communities is greatest (e.g., Van Eepoel and Grigg, 1970; Chesher, Chapter 6).

Toxic metals are introduced in desalination effluents due to corrosion of metallic surfaces of the distillation system. Zinc and nickel ions, though toxic to marine organisms at elevated concentrations, are much less harmful than copper at equivalent concentrations; copper appears to be the toxic contaminant of greatest concern in such effluents (Zeitoun et al., 1969a). The mixture of brine blowdown and cooling water discharged from desalination plants may contain this element at concentrations 6–8 times as high as the recommended maximum concentration of 0.02 mg/l, and 30–40 times the natural copper concentration in coastal waters (Zeitoun et al., 1969a).

Studies of the effects of desalination effluents are greatly complicated by interactions between different stress factors. For example, zinc and copper can act synergistically on organisms, as can copper and elevated temperature.

Elevated salinities may reduce the toxicity of copper due to the presence of more ions to compete with copper for sorption sites (Zeitoun et al., 1969a).

Desalination effluent in Key West, Florida, appears to be responsible for the reduction or disappearance of some marine species from the outfall area and an increase in the numbers of others, including fish and shellfish of several edible species (Clarke et al., 1970a,b; Chesher, Chapter 6). Clarke et al. (1970a) reported that desalination effluent was detected, due to elevated temperatures and salinity, at greater distances from the discharge point than were gross changes in the biological communities. This suggests, they state, that the levels of effluent dilution necessary to produce minimal environmental modification should be amenable to prediction.

The use of combination power—desalination plants in the vicinity of marine communities may be expected to amplify ecological problems because of the elevated temperatures of the effluent. Van Eepoel and Grigg (1970) have made some preliminary observations near one such plant which discharges effluent into Lindberg Bay, St. Thomas, Virgin Islands. Salinities and temperatures near the outfall were only slightly higher than ambient on two occasions when surveys were made. On another occasion, however, divers could not measure the effluent temperature because it was above that which they could tolerate. Corals and other invertebrates were killed to a distance of 200 m from the discharge plant. Difficulty in studying the effects of the effluent beyond that point were encountered because the bay was also polluted with oil and soot from the plant as well as from excessive sedimentation due to land-clearing and associated erosion. Power and water output from the plant were expected to double within the following three years.

Desalination plants are proliferating as the demand for freshwater grows and the cost of desalination decreases. It is disturbing, therefore, to learn how little effort seems to have been made to examine the ecological consequences of brine disposal. Of the first 481 research and development reports based on work carried out under contract to the United States Office of Saline Water, only three specifically concern ecological problems attending effluent disposal in marine environments. It is also noteworthy that of the eight desalination plants surveyed by Zeitoun et al., (1969a) (five of which are located on coastlines in regions of extensive coral reef development), no governmental regulations existed with regard to effluent discharge, and in only one case (Nassau, Bahamas) was an oceanographic or engineering study of the effect of the effluent undertaken before the plant was built.

Low salinities

Poor land management leads not only to sediment stress on coastal communities but also the stress of lowered salinities due to accelerated freshwater runoff. Freshwater is clearly a pollutant when introduced into marine environment in excess due to man's carelessness.

Most species of Indo-Pacific and Atlantic corals that have been tested for tolerance to low salinities survived exposure to 50% seawater for less than two days (Mayor, 1918a; Vaughan, 1919; Edmondson, 1928; Wells, 1932). Eight out of fifteen Hawaiian species survived less than a week in seawater diluted by one-quarter with freshwater. Exposure to freshwater for 30 min killed most species (Edmondson, 1928). Coral planulae may not settle or grow at reduced salinities (Edmondson, 1929). Calcification in corals decreases with decreasing salinity (e.g. Yamazato, 1970). Observations by Maragos (1972) in Hawaii suggest, however, that the growth of some corals is unusually rapid in varying and slightly suboceanic salinities. Lewis et al. (1968) made similar observations in Jamaica.

There are a number of accounts of the destruction of reef biota by freshwater during heavy rains (Hedley, 1925; Rainford, 1925; Hiatt, 1958; Slack-Smith, 1960; Cooper, 1966; Banner, 1968). It was not always clear to what extent sedimentation rather than salinity caused the observed damage. Goreau (1964) documented the mass expulsion of zooxanthellae from a variety of species of Jamaican corals exposed to low-salinity waters associated with flooding. This loss of zooxanthellae may be expected to lower the rate of calcification of corals significantly (Goreau, 1964). It may also lower tissue growth rates since dissolved organic compounds released by zooxanthellae are of nutritive value to the coral host (e.g. Muscatine and Cernichiari, 1969). Recovery of zooxanthellar populations in corals which survived the incident in Jamaica was not complete after fourteen weeks (Goreau, 1964).

CHLORINATED HYDROCARBONS

Little is known about the distribution of chlorinated hydrocarbons in reef organisms and less about their effects. McCloskey and Chesher (1971) found that a reef coral, *Acropora cervicornis*, contained 3—12 ppb DDT, 260—320 ppt Dieldrin and 200—300 ppt DDE plus PCBs. *Astrangia danae*, a non-reef building coral from Woods Hole, showed detectable quantities of Dieldrin only. Both species rapidly concentrated experimentally introduced DDT, Dieldrin and Aroclor 1254.

Several species of reef corals exposed to an organochlorine mixture responded by increasing their respiration rates while decreasing their photosynthetic rates (McCloskey and Chesher, 1971), or, expressed in other terms, their P/R ratios decreased and their light compensation levels increased. The authors state: "if, as it appears, subtidal corals at the depth studied are maintaining a diurnal O_2 balance at very close to 1.0, then any alteration which drops this value to significantly less than 1.0 may result in profound alterations in the reef community."

Acanthaster collected from a variety of Pacific sites contained from 0.04

to 3.89 ppm DDT, 0.01—1.04 ppm Dieldrin (McCloskey and Deubert, 1972) and 0.01—0.21 ppm PCBs (McCloskey and Deubert, 1973). No marked correlation was found between pesticide levels and starfish abundance. For additional data on pesticide levels in *Acanthaster* from the Great Barrier Reef see Tranter in Walsh et al. (1971, appendix F).

Barracuda taken from the Flower Gardens reefs in the northern Gulf of Mexico were found to contain high levels of DDT. Significant amounts of Dieldrin and Endrin were also present (T. Bright, personal communication, 1973; Houston Post, December 12, 1970). Low levels of DDT were found in barracuda taken in the Florida Keys (Deichmann et al., 1972). Giam et al. (1973) found variable but generally low levels of chlorinated hydrocarbons in groupers in the Gulf of Mexico and the Bahamas. Tissue levels increased with increasing size in the area where the highest concentrations were found.

The danger of lagoon contamination by pesticides, sometimes used in large quantities on atolls, has recently been noted (Anonymous, 1971). Fifteen to twenty tonnes of fish died suddenly in Truk Lagoon on April 17, 1970, hospitalizing six people who ate some of them. The flesh was found to contain the pesticide Endrin in the highest concentrations ever recorded in fishes to that date (Bourns, 1970). People of the Cook Islands now use chlorinated hydrocarbon pesticides as a substitute for derris in killing lagoon food fishes (Hambuechen, 1973).

Seba (1970) refined bioassay procedures for Dieldrin using three tropical fish, including the tarpon *Megalops atlanticus*. Shultz (1971) found moderately low levels of chlorinated pesticide residues in the sediments, water and biota in a polluted Hawaiian estuary. Nunogawa et al. (1971) tested the resistance of two coastal Hawaiian fish to DDT, Dieldrin, Lindane and phenol. Lane (cited in Randall, 1972) found the tropical sea urchin *Echinometra* had a higher resistance to Dieldrin than any of the other vertebrate or invertebrate animal she tested.

HEAVY METALS

Trace metal pollution has been little studied in the tropics. Normal trace metal concentrations are much lower in carbonate sediments characteristic of reef areas than in clay sediments where they are strongly sorbed to clay particles. Thus, comparatively large fractional changes in sediment metal concentrations may be expected in reef areas polluted with heavy metals, (Segar and Pellenbarg, 1973).

Taylor and Bright (1973) analyzed the distribution of heavy metals in reef-dwelling groupers in the Gulf of Mexico and the Bahamas. They found mercury, cadmium, lead, copper and zinc levels below those generally thought dangerous to humans. Arsenic levels in the muscles of most groupers tested surpassed U.S. Food and Drug Administration tolerance level of 0.5

ppm. The highest concentrations were found in the Nassau grouper from the Bahamas where it is the main commercial fish. Taylor and Bright (1973) argue convincingly that these high arsenic levels found both in groupers and various other reef species in their stomachs are a natural phenomenon; phosphorus levels are characteristically very low in tropical waters, and reef organisms probably accumulate arsenate, which is chemically similar, as a substitute in the phosphate transport system.

Barker and Zeitlin (1972) found that the elemental enrichment and metal/ion ratios of sea surface microlayers can be changed and enhanced by orders of magnitude by pollutants in near-shore waters in Hawaii. While the water samples at 0.6 m depths were relatively normal, microlayer samples often contained metallic pollutants in high concentrations, strongly organically associated and available for assimilation into marine food chains.

Heavy-metal levels in sediments and marine biota have been determined by Evans et al. (1972) in Pearl Harbor, Hawaii. Chesher (Chapter 6) reports on copper toxicity tests on a variety of south Florida marine organisms. Heavy-metal pollution is also mentioned above in connection with desalination and power plant effluents.

MISCELLANEOUS

Nuclear weapons tests

The obliteration of coral reef communities in the vicinity of nuclear blasts is of minor significance for man compared with subsequent effects spread over much wider areas. The most obvious immediate biological impact outside the area of complete destruction is that of radiation burns to the biota. Donaldson, cited by Hines (1962), collected living fishes in the vicinity of a recent blast whose skin was missing from one side as if the animals had been "dropped in a hot pan". This also is a comparatively trivial phenomenon. It is the radioactive contamination of the food chain which has by far the greatest consequence for man.

The geographic distribution of radioactive contaminants has been difficult to predict in advance of nuclear explosions because of the complexity of the atmospheric and ocean currents on which they are carried. It is nonetheless well known that serious contamination may extend for great distances. Because of atmospheric circulation patterns, areas more than 240 km from a thermonuclear explosion at Bikini Atoll received four times as much radiation as parts of Bikini itself. Parts of Rongelap Atoll 160 km away received sufficient fallout within 48 hours that human survival in the absence of shelter would have been unlikely (Hines, 1962). It was not until four years later that all reef species at Rongelap were declared safe for human consumption (Hines, 1962). A single nuclear blast may thus render fish and shellfish

unsafe or unsaleable for thousands of square kilometers.

Generally it is the lower trophic levels, the algae and herbivores, which become most radioactive immediately after a blast. Later carnivorous reef fishes become more radioactive than omnivores or herbivores (Donaldson, 1960; Hines, 1962). As radioisotope levels in the community subsequently diminish, *Tridacna* and other edible reef bivalves are among the last reef animals to become safe for human consumption (Hines, 1962; see also Welander, 1969).

The ecological impact of radioactive contamination on reef communities is almost unknown. Gorbman and James (1963) discovered radiation damage to the thyroid glands of various coral reef fishes after a nuclear test at Enewetak Atoll. This appears to be the only observed damage attributable to the injurious effects of radioactive contamination of the reef food chain. Recent X-ray and autoradiographic studies of corals exposed to radioactive fallout during nuclear testing at Enewetak Atoll revealed "no micro- or macroscopic effects . . . which can be correlated with or attributed to the nuclear tests" (Knutson and Buddemeier, 1973).

Dying animals on the reef are liable to be consumed by reef scavengers and carnivores before they are observed (Donaldson, 1960). To evaluate the impact of any stress on a reef community, population sizes of various species should therefore be measured before and after the stress occurred. Unfortunately this does not seem to have been done in connection with nuclear tests in the Pacific in the 1940's and 1950's. The statement by Schultz, "the pressure of population from all sides into the damaged area is very great and soon replaces the losses" (cited with approval by Templeton et al., 1972), therefore seems premature. It is to be hoped that the French have made efforts to clarify this in connection with their recent tests in the South Pacific.

In January 1971, James Maragos and I had a brief opportunity to examine some shallow patch reef communities which had apparently been destroyed by a nuclear blast at Enewetak Atoll. The date of destruction could not be determined because several nuclear tests were carried out on different dates in the vicinity. The nearest blast created a crater several hundred meters away on the inter-island reef. The last year in which nuclear tests took place in the vicinity was 1958.

The reefs are located several hundred meters northwest of Yvonne (Runit) Island in an area where the inter-island reef begins to slope toward the lagoon. The tops of the reefs are covered by about 0.5 m of water at low tide. The communities had been dominated structurally by *Acropora corymbosa* ("table Acropora"). We could not tell whether its destruction was caused by heat or radiation. Since skeletons of *A. corymbosa* were still standing we assume the communities were not killed by the force of the blast.

Extensive regrowth of *A. corymbosa* had occurred. Some living specimens

were nearly 3 m in greatest diameter. Coral diversity was low however. Only 4 species of corals were common, whereas Maragos counted 9—10 common genera and estimated there were about 15 common species on typical *Acropora*-dominated inner reef communities at Enewetak. Diversity and abundance of other invertebrates, fishes and algae also appeared low on the affected reefs. Very little grew on the dead skeletons of *A. corymbosa* or the substrate. In short, although regrowth of *A. corymbosa* on this reef was impressive, the community had by no means recovered thirteen years after the last nuclear test in the vicinity. Patch reefs further from the test sites, several hundred yards south of Yvonne Island, showed no readily observable effects of the blasts.

Fishing with poisons and explosives

Though usually illegal, the use of dynamite and poisons to kill reef fishes is commonplace in the tropics (e.g., Ronquillo, 1950; Anonymous, 1952; Owen, 1969, Ramas, 1969; Powell, 1970; Sverdloff, 1973; Jones et al., 1973; Lulofs, 1973). Christian (1973) reviews past research on the effects of underwater explosions on fish and describes a new method for predicting the maximum ranges to which such explosions injure fish. The depth at which a charge is exploded has an important influence on the shape and extent of the surrounding region in which such damage occurs. The destruction of habitat by explosives in coral reef communities has never been adequately evaluated.

Ronquillo (1950) summarizes anatomical damage observable in fish killed by explosives. A knowledge of these characteristics should be useful to fisheries personnel investigating suspected dynamite fishing.

Ronquillo lists the following anatomical disarrangements:

Gross aspect

1. The air bladder, if present, is almost always ruptured and blood clots are found in its lumen.
2. The vertebral column may be fractured in any part along its length.
3. Localized hemorrhages are present around the area of fractured parts due to the destruction of the blood vessels and tearing of adjacent tissues.
4. Parts or all of the contents of the body cavity may be damaged or crushed with hemorrhages depending upon the size, shape, position and distance of the fish from the explosion.
5. Fracture and/or dislocation of the abdominal ribs from the vertebral column may be found especially in spiny fishes, with accompanying hemorrhages present in the area of the fracture.
6. The blood vessels below the vertebral column may break and cause hemorrhages of varying degrees along that region.
7. Rupture of the parietal peritoneum, especially that attached to the abdominal ribs.

Roentgenological aspect

1. Dislocation and/or fracture of the vertebral column and ribs, if present, are clearly shown in the negative.
2. The air bladder, if ruptured, will be filled with blood and will be obliterated in the negative. If not ruptured and, therefore, filled with air as in normal fish, it occupies a definite shape and position in the abdominal cavity.

In the Philippines, according to Ramas (1969), a "method used by blast fisherman is to find some large coral formation about the size of a regular house in about 100 to 150 feet of water. They then rig up some batteries, wires and light bulbs inside mason jars, and attach these underwater lights around the area. They also chum the area with conch and fish chunks. The area is completely wired with dynamite, with the detonator in the boat on the surface. Periodically, a scuba diver dives down to check on the fish gathering. When the big fish come around in enough numbers, the charge is detonated and scuba divers are used to retrieve the fish. So much fish is obtained this way that . . . the smaller fish under five pounds are not even picked up."

In Truk Lagoon a group of fishermen with fast boats are organized for obtaining dynamite, fishing with it and retailing the dynamited fish (Fuchs, 1968). Powell (1970), referring to Truk Lagoon, states: "An explosion echoing through a village is ample notice that fresh fish are on sale . . . People who are generally protein deficient in their diet do not protest or complain if the fish is mushy."

Lulofs (1973), commenting on widespread use of explosives to capture reef fish in Malaysia, states: "Although thorough policing of coastal waters is almost impossible, this form of fishing could be greatly reduced, if not entirely prevented, by the provision of legislation which not only prohibits the sale of blasted fish, but cancels without right of appeal the licenses of those dealing in blasted fish. Blasted fish can be identified, and once the sales outlets are removed, the operation becomes pointless."

Surplus 700-lb (ca. 20-kg) U.S. Navy bombs were used (25 at a time!) to clear a channel through the reef at Kayangel Atoll, Palau, in 1973. The concussions cracked every cement structure on an island more than 5 km away. Two months later underwater visibility was still greatly reduced up to several kilometers from the blast site and coral damage, apparently due to sedimentation, was still in progress (D. Faulkner, personal communication, 1974).

The use of poisons to kill marine fishes in Hawaii was outlawed in 1850. But here, as elsewhere, the practice remains common today. Bleach (sodium hypochlorite) was used by fishermen in a cave with which I am familiar off the south coast of Oahu, Hawaii. It contained dense schools of three species of fishes, populations of palinurid and homarid lobsters, and a rich, colorful

encrusting fauna of sponges and tunicates. After the cave was exposed to bleach almost no visible living macrofauna remained. Recolonization was far from complete a year later.

Ciguatera

Algae are generally among the first macroscopic organisms to colonize any fresh surface in reef areas, including the surfaces of blasted or dredged areas and of newly deposited sediments (Dawson, 1959; Brock et al., 1966), artificial reefs (Randall, 1963), skeletons of corals eaten by *A. planci* (Chesher, 1970), sunken ships, abandoned war materials and other human artifacts (Randall, 1958; Halstead, 1972).

The apparent connection between these new algal growths and fish poisoning was first noted by Dawson et al. (1955) and discussed in detail by Randall (1958) and Dawson (1959). These authors pointed to the relationship between the availability of new surfaces in the reef environment, the rapid growth of algae, and the development of toxicity, known as ciguatera, in normally edible species of reef fish in the immediate area. They theorized that toxic algae, possibly blue-greens, growing unusually rapidly on new reef surfaces, are at the base of the toxic food chain leading ultimately to man and causing illness and sometimes death. Yasumoto et al. (1971) found that the most toxic fraction of the food of highly ciguateric detritus-feeding surgeon fish consisted of small unidentified particles of low specific gravity.

Randall (1958) points out: "The importance of ciguatera goes beyond the purely medical aspects of treatment of patients suffering from the ingestion of toxic fish. Many nontoxic fish are denied to humans or domestic animals as food because of fear of their being poisonous. Sections of reef with a reputation of harboring poisonous fishes are not fished, whereas nontoxic areas may, as a result, be overfished."

Bagnis (1969) has provided a valuable account of the development of ciguatera in a reef community in the same areas and in the same chronological sequence as disruptive human activities. However, there are numerous examples of dredged or otherwise disturbed reef areas where ciguatera has not developed (e.g. Bagnis, 1973). At Johnston Island where ciguatera has occurred in the past, extensive dredging of reef areas in 1963 has not been followed by any new outbreaks (A.H. Banner, personal communication, 1973). Nonetheless, the relationship between the disruption of reefs by man and the subsequent development of ciguatera in the immediate area seems too frequent to be coincidental, and research continues on this recalcitrant problem.

Crown-of-thorns starfish

Whether large aggregations of coral-eating crown-of-thorn starfish, *Acan-*

thaster planci, are brought about by man's activities, including pollution, or are a natural phenomenon, is, at this writing, still vigorously debated. Some lay conservationists (e.g. Cropp, 1971) have inferred from those who suggest that they are a natural phenomenon,that outbreaks should be allowed to progress unchecked.

This reasoning is characteristic of those who attribute to man the distinction of being the only force in the biosphere capable of exerting "unnatural" influences on it, and who assume that all other influences being, by their definition "natural" are therefore somehow always desirable. Anyone really committed to this philosophy would presumably allow his house to burn down if lightning struck it, rather than interfere with a natural phenomenon.

Whether ecological disruptions caused by non-human agencies are desirable from man's viewpoint (and whom else can we poll?) depends upon the particular circumstances. Fire is disastrous to some types of communities but ultimately beneficial to others. Man tries to prevent fires in the former case regardless of their origin and sometimes sets them in the latter.

Similar reasoning should govern decisions concerning control of *Acanthaster* outbreaks. Unfortunately, we do not possess adequate information to make a final decision. We know *Acanthaster* outbreaks are harmful to reef communities on a short-term basis; we do not know whether they are ultimately beneficial over a longer period, as Dana et al. (1972) have suggested, and we may not find out for decades. An interim decision should be based on what we know rather than on what we do not know. Therefore, until new information dictates otherwise, serious *Acanthaster* outbreaks should be controlled, in my opinion, irrespective of their origin.

The esthetic fishery

The collection of shells, corals and tropical aquarium fish cannot be called pollution. Neither does it fall within the conventional purview of commercial fisheries biology. Thus, what might be called the esthetic fishery, which sometimes clearly constitutes an environmental threat in coral reef communities, has received virtually no scientific scrutiny. It seems justifiable, therefore, to mention it briefly here.

There are numerous reports throughout much of the Caribbean and Indo-Pacific of overharvesting of corals, gastropods and aquarium fish by amateur and professional collectors. The physical destruction of shallow reef flats due to trampling, and of individual coral heads due to the use on them of crowbars by shell collectors, are related problems. Since no systematic quantification of the effects have apparently ever been attempted it is not possible to say exactly how serious they are. But it is my impression, judging by reports from shell collectors and biologists and from my own observations, that the problem is widespread and growing.

Corals from the Florida Keys, for example, are sold by the truckload to

shell stores (e.g. Barada, 1974), but extensive collecting is not limited just to areas of high local demand. Adult specimens of the giant clam *Tridacna maxima* are almost gone from some Marshall Island atolls where they were extremely abundant twenty years ago. Exporters are reportedly rapidly depleting some reefs of aquarium fish in Malaysia and Indonesia (e.g. Lulofs, 1973). Cousteau (1972) claims that teams of Tahitians, using crow bars to collect shells, "destroy six miles of reef per week".

A number of species of reef fish and invertebrates which remove parasites from other fish are aquarium favorites. These species constitute a very small portion of total community biomass. But their importance to the reef fish community far outweighs their biomass according to several investigators. Limbaugh (1961) reported that when he removed all known cleaners from two small Caribbean patch reefs, reef fish populations were "drastically reduced" within a few days. Within two weeks almost all except the territorial fishes had disappeared. Many of the fishes remaining developed "fuzzy white blotches, swellings, ulcerated sores and frayed fins". On the basis of observations in the Red Sea, Slobodkin and Fishelson (1974) state: "it seems evident that the local diversity of coral reef fishes may be to a large extent determined by the presence of cleaner wrasse stations."

It is not difficult to completely remove some species of cleaners from a reef. If the results are as serious as the above observations suggest, then clearly the capture and sale of these species should be outlawed.

Reef communities can comfortably yield limited quantities of some species of fish and shells and small corals if gathered with care, just as they yield food fish. But until research is done on the optimum sustained yields of species sought by collectors, and on the physical impact of collecting on the environment, complete closure of some reefs to collecting is probably the only way to prevent serious excesses. Certainly the movement or breaking of coral to obtain the fish and shellfish within them should be prohibited.

Alien species

The importation of alien species has brought about numerous well-known ecological dislocations in terrestrial environments. Potentially hazardous introductions of various alien species into coral reef communities has been noted by various authors (e.g. Dickie, 1875; Doty, 1961, 1973b; Gilbert, 1962) but the identity and ecological impact of these species has received very little study.

As noted above, one species of benthic algae introduced to Hawaii around 1951 has become the dominant species in some reef areas in the vicinity of large human populations (Soegiarto, 1972; Doty, 1973b). There is a possibility of passage of crown-of-thorns starfish, sea snakes and other species of dubious value through the Panama Canal from one ocean to the other via seawater ballast (Chesher, 1968) or through the proposed sea level canal (Thomas, 1970).

THE FATE OF DAMAGED REEF COMMUNITIES

When a reef community is destroyed the ecological conditions that follow cannot be expected to coincide with those which preceded the initial development of the community. Thus, it cannot be taken for granted that the reef community will ever replace itself. Wood-Jones (1907) for example, reported that thirty years after the destruction of a coral reef by "foul water" released from a volcanic vent, only dead corals remained; there was no sign of regeneration. Rainford (1925) reporting 1918 flood damage to corals in the Whitsunday Group, Great Barrier Reef, doubted that recovery would ever occur. Seven years later Hedley (1925) found that one luxuriant reef subjected to this flood had been "planed away by the waves as if some huge razor had shaved off the coral growth down to low tide level". Except for the bivalve *Chama*, animal life was "very scarce". In 1953 areas still remained in the Whitsunday Group where negligible recolonization had occurred (Stephenson et al., 1958).

In 1965 corals and other invertebrates were killed by flooding on the Coconut Island reef in Kaneohe Bay, Oahu (Banner, 1968). An area on this reef where corals once covered 70—80% of the bottom (Gordon and Kelly, 1962) is now colonized largely by densely packed colonies of zoanthids (personal observation). Sites for settling of coral planulae are very limited and restoration of former conditions on most of this reef in the forseeable future seems unlikely.

When corals die their surfaces are rapidly colonized by algae (Moorhouse, 1936; Slack-Smith, 1960; Goreau, 1964; Banner, 1968; Chesher, 1969, 1970; Endean, 1969; Pearson and Endean, 1969; Stoddart, 1968; Weber, 1969; Fishelson, 1973b). Benthic algae are inimical to coral regrowth. They not only occupy substrate which might otherwise be used by settling coral planulae (Hedley, 1925), they also trap sediment and create shifting banks unsuitable for larval settlement or continued growth of adults (Wood-Jones, 1907; Moorhouse, 1936). In addition, they can overgrow young or injured coral colonies (Edmondson, 1928) and kill young coral colonies by current-induced abrasion (R.H. Randall, personal communication, 1973).

Subsequent to colonization by algae, dead coral surfaces may become colonized by encrusting animals including alcyonarians, hydroids, bivalves and zoanthids (Wood-Jones, 1907; Slack-Smith, 1960; Barnes, 1966; Banner, 1968; Chesher, 1970; Endean, 1969; Weber, 1969; Fishelson, 1973b). In Guam large numbers of the sea urchin *Echinometra mathaei* appeared and removed most of the growth of filamentous algae on coral recently killed by *Acanthaster planci* (R.H. Chesher, personal communication, 1973).

Although many faunal elements of the community disappear after corals are killed, the standing crop of herbivorous fishes may increase as a result of this increased food supply (Chesher, 1970). It has been suggested that the production of harvestable protein under such circumstances might equal or

exceed that of the original reef. However, my observations in Fiji and Enewetak in areas recently infested with *Acanthaster* indicate that an increase in herbivorous fish under such circumstances is not an invariable response. In these two areas herbivorous fishes were conspicuously *less* numerous than in adjacent areas with abundant living coral, despite a heavy overgrowth of algae on the dead coral skeletons. Perhaps the colonizing algae here were not acceptable as food. In areas where herbivorous fish populations increase after an *Acanthaster* invasion, it is likely that this is just a transient early stage of ecological succession on the recovering reef.

Rates of recovery of reef communities from damage differ widely and depend in part upon the extent of the damage and the species affected (e.g. Endean, 1971). If damage is extensive, complete recovery of the community appears often to require several to many decades (Stephenson et al., 1958; Chesher, 1970; Stoddart, 1968; Endean, 1971; Randall, 1973; Marsh and Gordon, 1974). Some of the larger coral heads killed by *Acanthaster* in Guam appeared to be over 200 years old (Goreau, in Chesher, 1970).

However Shinn (1972) has observed that recovery from extensive storm damage may require as little as five years in the case of the very rapidly growing staghorn coral *Acropora cervicornis*, in the Caribbean. In the two instances cited by Shinn "the majority of unburied fragments (of storm-damaged *A. cervicornis*) remained alive and within months began re-establishing discrete colonies". On the basis of his coral growth studies Shinn (1972) concludes: "complete recovery of a reef composed principally of head corals would require more than an average man's lifetime. On the other hand, a reef composed of staghorn corals has the capacity to regenerate in a matter of a few years."

Based on observations made in the wake of *Acanthaster* outbreaks on the Great Barrier Reef, Pearson (1974) states similarly that "it is conceivable that less than 10 years . . . large sections of reef could become almost completely covered by tabular species of *Acropora* and other fast growing species. However for a more diverse assemblage to develop it will be necessary for the slower growing species to compete successfully with them."

The recovery of coral communities in Hawaii after inundation by lava takes about twenty years in areas where exposure to sea and swell maintains coral communities in an early stage of ecological succession. In more sheltered areas where reefs are more nearly climax communities recovery is on the order of fifty years (Grigg and Maragos, 1974).

Recovery may be more rapid if only mobile organisms or organisms with shorter lifespans than corals are affected. For example, less than four years after the fish population was removed from a Bermuda patch reef recolonization was complete (J. Bardach, personal communication, 1974). In Hawaii recolonization of a patch reef from which all the fish were removed was well over half complete after 241 days (Wass, 1967). Doty (1973a) estimates that the establishment of climax populations of tropical benthic algae on under-

water lava flows takes five to ten years.

If, when corals are damaged, portions of living tissue remain, the tissue may regenerate and regrow over exposed skeleton (Fishelson, 1973b; Randall, 1973). Filamentous algae and invertebrate epifauna growing on the exposed skeleton may retard or completely inhibit this regrowth however. Fishelson (1973b) found that "bushy" forms with small polyps were most successful in encroaching upon the epibionts and repairing the breached surface.

Studies of recolonization of corals are of considerable interest, but to use coral cover or coral species diversity as simple indices of recovery of the reef *community*, as some have done, can be very misleading. Coral diversity on a recovering reef may rapidly reach, or even exceed, diversity in the original undamaged community. This is because large numbers of small colonies may develop on unoccupied substrate (Goreau et al., 1969; Grigg and Maragos, 1974).

Such colonies may rapidly occupy large areas of bottom, resulting in high figures for coral coverage. But full recolonization of the area by the many fish and invertebrates that live in, on, or in close proximity to corals, requires a greater volume, total surface area and complexity of coral architecture than these young colonies provide. These features of the coral colony are developed in rough proportion to colony height. Any index of community recovery which is based on corals alone could thus be improved if measurements of mean colony height were incorporated.

In addition it would be instructive to know how the original and present community compared in terms of similarity of coral (or other) species. Coral communities with similar diversity would nonetheless be quite distinct if species similarity were low. The use of similarity indices developed by plant sociologists would be appropriate here (e.g., Sorensen, 1948; Cole, 1949; Whittaker, 1967; Scheer, 1975; see also Chesher, Chapter 6).

CONCLUSIONS

The inadequacy of water column studies

An 18-month multidisciplinary study of pollution in Kaneohe Bay led to the publicly expressed opinion that it was not badly polluted and was "in exact balance" (Honolulu Star Bulletin, July 19, 1969). The results of this study, which was focussed mainly on water quality, did not hint at the disruption of the reefs described above. This error is instructive for it demonstrates two common shortcomings of pollution surveys.

First, the research was heavily weighted towards public health aspects of pollution. It is not always sufficiently recognized by pollution scientists with a public health background, that waters containing levels of pollutants which

do not threaten human health directly can be highly destructive to aquatic communities.

Secondly, no one surveyed or apparently even looked at bottom communities during the study. The continuous removal and replacement of water and its contents due to tidal flushing, and the high reproductive rates of planktonic organisms, makes water quality an inadequate indicator of pollution in shallow coastal waters (e.g. Chesher, Chapter 6). The bottom, in contrast, serves as a reservoir for pollutants, which accumulate there through settling and sorption. The impact of pollution is thus liable to be more profound in bottom communities. (See also Zieman and Wood, Chapter 5). For example, while reef communities were being destroyed by pollution in Kaneohe Bay, only "subtle" changes were observed initially in the plankton community (Clutter, 1972).

Bioassays

Pollution is likely to affect populations with rapid turnover times first. Thus benthic microorganisms should provide early warning signs of pollution in coral reef communities as they do elsewhere (e.g., Storrs et al., 1969; Chesher, Chapter 6). The increased activities of sulfate reducers in sediments exposed to sewage has been mentioned above.

A good biological index of pollution is one which is easy to measure, responds to pollutants rapidly, and leaves a permanent record of its response. Benthic foraminifers satisfy all three criteria. They are, furthermore, especially abundant in coral reef sediments (Nicholls, 1969). Seiglie (1968, 1971, 1974) describes changes in foram abundance and species composition due to pollution in Mayaguez Bay, Puerto Rico (see also Chesher, Chapter 6). Nicholls (1969) provides a useful general review of the relations between environmental conditions and foram distribution in estuaries.

Coral planulae are easily obtained and live well in aquaria. Since larval forms are often more susceptible than adults to environmental stress it might be thought that coral planulae constitute convenient and sensitive pollution bioassay organisms. However, coral planulae seem more tolerant than adults to extremes of salinity and temperature (Edmondson, 1946) and are unusually resistant to chlorine (Davis, 1971).

More useful, perhaps, would be very young corals obtained by allowing planulae to settle in laboratory tanks. The growth rates of such corals are very rapid and easy to measure (by counting polyps) and their small size and the large numbers easily obtainable (e.g. Harrigan, 1972) suggest they might prove ideal for bioassay work.

The growth and survival rates of transplanted corals have been used as environmental indices by Mayor (1924), Shinn (1966) and, more extensively, by Maragos (1972). Branching hermatypic corals are particularly useful because, as Shinn states, "they are extremely sensitive to environmental

changes; they grow rapidly; their shape makes distal growth easy to measure; and they do not swim away".

If one is to obtain a measure of the impact of pollutants in this manner it is necessary to use coral species which grow (or grew before the advent of pollution) naturally in the area. If reduced growth or survival rates are observed at sites where the transplanted species never grew, these effects cannot be attributed to pollution. Shinn (1966) observed growth rates of less than one-half the control values when he transplanted *Acropora cervicornis* to unpolluted areas where it did not normally grow.

Both Shinn (1966) and Maragos (1972) noted that growth was not only slower in corals transplanted to areas of suboptimal environmental conditions, but also more erratic than in control areas. Of the six Hawaiian species used by Maragos (1972) in transplant studies, *Fungia scutaria* was the most sensitive in terms of survival time.

The use of small transplanted communities for pollution research is being pioneered by coral reef ecologists at the Hawaii Institute of Marine Biology. Coral communities are transplanted to tanks about 1.5 m^2 in area and about 0.5 m deep, containing running seawater. The salinity and temperature of seawater are automatically controlled. In addition to measuring structural responses of these sub-communities to stress, it is possible to monitor metabolic responses by measuring differences in concentrations of oxygen, nutrients, etc., between inflow and outflow water.

This approach helps fill a major gap between single-species laboratory studies and field studies on whole communities. While laboratory studies facilitate refined control of physical environmental variables, one cannot predict community responses with much confidence on the basis of the responses of single-species components isolated in the laboratory; communities are more than the sum of their component species (e.g. Odum, 1971). Field studies, while providing information about real ecosystems, do not provide adequate opportunity to monitor the system continuously or to manipulate important physical and biological variables. In addition, the detailed prior environmental history of a community is seldom available in field studies. This shortcoming can be overcome in tank communities by acclimation prior to manipulation. The major limitations of such studies are the size (and therefore the complexity) of the community that can be transported and housed, and the lack of normal interaction with adjacent communities.

The use of measurements of species diversity as pollution indices often proves a snare and a delusion. Hedgepeth (1973) puts the problem in perspective: "The search for simple approaches and magic numbers that may be obtained with relative ease has led some pragmatic ecologists down the primrose path of theoretical ecology. Perhaps unaware of the highly speculative basis for such concepts as diversity and stability, they have oversimplified the idea of diversity indices to suggest that a high diversity index is ipso

facto indication of the well being or more natural condition of a biocoenosis and consequently the number can be interpreted as an abstract value of the state of pollution."

Retrospective pollution analysis

At least some reef corals contain daily, seasonal and annual growth bands in their skeletons. By using autoradiography or X-radiography of sections of coral skeletons it is possible to measure the width of these bands and construct a diary of the past growth history of the coral. Moore and Krishnaswami (1974) state: "If we accept the rather compelling evidence that . . . banding in corals is annual and that specific bands may be assigned definite ages . . . then corals become vast storehouses of information about chemical and physical changes of waters in which they grow. Recorded in every coral head is a record of changes in fallout isotopes (e.g., ^{14}C, ^{90}Sr), uranium series isotopes (e.g., ^{238}U, ^{226}Ra, ^{228}Ra, ^{227}Ac, and ^{210}Pb), stable isotopes (e.g., ^{13}C, ^{18}O), trace metals, and clay particles as well as significant data on small scale environmental fluctuations."

These techniques appear to have considerable potential for correlating coral growth rates retrospectively with pollution of known temporal extent, just as tree ring "calendars" are used to retrieve paleoclimatic information. For example, coral autoradiography has been used to reconstruct changes in the relative concentrations of bomb-produced isotopes in surrounding waters (e.g., Knutson et al., 1972; Knutson and Buddemeier, 1973; Moore and Krishnaswami, 1974).

Moore and Krishnaswami (1974) found a hiatus in coral growth which corresponded temporally with the operation of a cement plant nearby. They suggest that pollution associated with this plant may have caused the observed cessation of growth. Dodge et al. (1974), using X-radiography to measure past growth rates of coral in Jamaica, found an inverse trend between rates of resuspension of sediments and coral growth.

Restocking damaged reefs

The "restocking" of damaged reefs with corals has been suggested by various writers. Hubbard (1974) points out that species must be chosen which can tolerate the currents in the area in question. She provides information on polypal tolerance to currents of eighteen species of Caribbean corals.

Fast-growing species and ones that dominate early successional stages of reef communities seem like logical choices for restocking programs. Various species of *Acropora* and *Montipora* (Vaughan and Wells, 1943; Lewis et al., 1968; Shinn, 1972) have these qualities. These species are also among those which rapidly cement themselves to the bottom and establish new colonies when fragmented and strewn across the bottom (Shinn, 1972; personal ob-

servations). *Porites lobata* has similar recuperative powers (personal observations). Since reseeding any substantial area with adult corals is quite impractical, this ability is of considerable value.

Importing consultants

Frequently foreign consultants are called in, often for just a few days or weeks, with the expectation that they can help solve environmental problems in the tropics. Having played this role several times, I would like to implore those who ask in outsiders to do their homework first. Sometimes the consultant is forced to spend his entire time canvassing various laboratories and agencies to find out what is already known locally about the problem at hand. This job could be done much more easily in many instances by local scientists familiar with the language and with the appropriate people to be interviewed and the relevant reports to be located. This should be done *before the consultant arrives*. Otherwise he has little opportunity to make observations of his own and finds it very difficult to make a contribution consonant with the cost of bringing him in.

Pollution control

Effective policing of reefs in order to deter the individual polluter is frequently out of the question. The reefs around Truk, for example, are so extensive that dynamite has been used to kill reef fishes with little concern that the authorities will hear about it (Fuchs, 1968). The widespread use of poisons in reef communities is also carried out with little fear of detection by authorities. In many tropical areas, as Storer (1967) points out for the Philippines, the effort to stop such practices "has occupied a disproportionate amount of time and personnel of both municipal and federal officials. Such efforts are, of course, largely futile. There is no incentive to prosecute; the offender cannot pay a fine and if committed to jail, the burden of supporting his wife and children is merely shifted to the government. Furthermore, such efforts to control the problem are easily thwarted." Public censure is thus probably the only effective form of pollution abatement in cases of individual abuse of reef communities, but it is rarely operative in an ecologically unsophisticated public.

Although the corporate polluter is easier to detect, laws to prevent such pollution are generally inadequate and frequently non-existent in the tropics. Too often the public, including even that segment whose livelihood depends directly upon the reefs, does not realize the problem exists. The enactment and effective enforcement of appropriate laws, and the public censure of polluters can only be brought about when the public is made aware of their value. The most important step in deterring both the corporate and the individual polluter is thus education.

There are numerous examples of customs which served the purposes of conservation in pre-industrial cultures. Falanruw (1971) points out that "inasmuch as former conservation-valued practices were in the guise of religion, social stratification, taboos, etc., the young people of Micronesia today are not individually conservation minded". This is undoubtedly also true elsewhere, and is one of several reasons why the teaching of conservation becomes increasingly imperative as the seemingly inevitable westernization of these cultures proceeds.

Merton (1967) notes: "the teaching of ecology in the tropics and subtropics is not only of absorbing interest, but has a social relevance which is difficult to underestimate. I should like to see, therefore, a course in ecology as a necessary requirement for any degree or teacher's certificate in such countries." Since general biology—ecology texts for schools in tropical areas do not exist (publishers take note!), many tropical school systems have recently prepared their own in booklet form.

Teachers in the Caribbean and the tropical Pacific Islands often come from temperate regions and have little knowledge of local fauna and flora. The Bermuda Department of Education recognized this problem and took steps to remedy it. They engaged a number of foreign and local scientists to give to the islands' biology teachers (most of whom were trained in England) a series of lectures on the ecology of Bermuda, supplemented by field trips. The information was enthusiastically received by the teachers. It is now being reflected in their curricula and will undoubtedly lead to an increased concern of Bermudians for their environment. The program costs little and is a good example for others to follow.

Most coral reefs are situated in developing countries with very limited conservation and research funds. The allocation of money for coral reef research and management is therefore very small in relation to the importance of these communities to man, their vulnerability to pollution, and the amounts spent studying some less valuable but more visible environments in industrialized countries. New or accelerated programs of research on some of the problems discussed above are obviously needed.

But environmental crises develop faster than they can be completely assessed. In this context it is more important to make interim decisions *in time* than to make more scientifically satisfying decisions later. Pollution biologists sometimes inject too much traditional laboratory caution into matters of immediate practical concern. They hesitate because their information is not final. But information is never final in science, which proceeds by disproof rather than by proof.

The fact that biologists have sometimes erred in their warnings about environmental degradation is no justification for abdicating the responsibility to speak up — any more than the fact that scientists sometimes publish mistakes means we should all abandon publishing. Those who remain silent when their observations point to environmental decay are the undertakers of the environment; environmental post mortems become their stock and trade. "They measure and we weep."

CHAPTER 4

TROPICAL SEA GRASS ECOSYSTEMS AND POLLUTION

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TROPICAL SEA GRASSES

Until recently, studies in the nearshore tropical environment have largely concentrated on the spectacular and complex coral reefs, and the dense, productive mangrove forests. However as marine science focuses on the living resources of the shallow, coastal seas, the vast beds of sea grasses bordering the tropical coastlines are being recognized as important resources. The significance of sea grasses in coastal ecosystems was documented early in the classic studies of Petersen (1918), but little further work on the ecological importance of these highly productive and easily studied areas followed until recently.

Sea grasses are unique among submerged marine and estuarine plants due to the presence of an extensive root and rhizome system. Because of this system — high growth rates, and dense leaf development — sea grasses exert considerable influence over their environment. Wood et al. (1969) summarized the functions of sea grasses in estuarine and marine ecosystems as follows: (1) sea grasses have a rapid rate of growth and high organic productivity; (2) while alive, they act as food for only a limited number of organisms since the demise of the large sea turtles, but provide large quantities of detrital material which, along with its resident microbes, provide a major food source for the estuarine ecosystem; (3) the leaves support large numbers of epiphytic organisms, which under favorable conditions, may be comparable in biomass to the sea grass leaf weight, and which are grazed extensively; (4) sea grasses provide organic matter which initiates sulfate reduction and maintains an active sulfur cycle in the estuarine sediments; (5) the dense leaves reduce the current velocity near the sediment surface and promote sedimentation of organic and inorganic particles; and (6) the roots and rhizomes bind the sediments and hinder erosion of the surface of the sediment.

Sea grasses are widely distributed throughout the shallow coastal seas from the Arctic to the southern tips of Africa and New Zealand. Den Hartog (1969) recognizes 12 genera of sea grasses with 49 species, of which 7 are tropical in distribution. In the Hydrocharitaceae, *Enhalus*, *Thalassia*, and *Halophila* are tropical, and in the Cymodoceoiceae, *Halodule*, *Cymodocea*,

Syringodium, and *Thalassodendron* of the sub-family Cymodoceoideae are basically tropical genera, although several of the above genera have some species which extend into subtropical or warm temperate seas. As is common with many marine groups, the highest diversity in the sea grasses is found in the Indo-West Pacific, with all seven genera represented. By contrast the Caribbean has only *Halodule*, *Syringodium*, *Thalassia*, and *Halophila*. Only *Halodule wrightii* and *Halophila decipiens* are common to both areas.

Petersen (1918) was the first to recognize the overall importance of sea grasses in the coastal ecosystem, and in recent years it has become apparent that sea grasses contribute significantly to the maintenance of the coastal ecosystem of the tropics and near tropics in numerous ways (Wood et al., 1969; Zieman, 1974a).

Because sea grass beds attracted little attention scientifically until the late 1960's, there are large gaps in our knowledge of these systems. The best studied of the tropical sea grasses is the turtle grass, *Thalassia testudinum* Konig, of the Caribbean. Although this study is directed to tropical sea grasses, nearly all of the information is drawn from *Thalassia testudinum*, as there is unquestionably far more published material dealing with this species than with all other tropical sea grasses combined. As research progresses, undoubtedly differences in the structure and function of different grass beds will be found. However, judging from the experience of this author and other workers in the field, the similarities between different sea grass communities are more striking than the differences.

Much attention has been drawn to the importance of sea grass beds as nurseries, important in the rearing and sheltering of small fishes and invertebrates (Hutton et al., 1956; Moore, 1958; Tabb et al., 1962; Hoese and Jones, 1963; Wood et al., 1969; Jackson, 1972; Odum et al., 1973). Prominent among the organisms inhabiting the grass beds are commercially important species such as the pink shrimp *Penaeus duorarum* (Hoese and Jones, 1963), mullet (Wood et al., 1969), sea trout, and stone crab (Tabb et al., 1962).

Whereas some animals have been recorded as feeding directly on turtle grass (Randall, 1965a, 1967; Ogden, 1972), since the decimation of the green sea turtle by overfishing there has been little direct grazing pressure on these beds except in certain localized situations. Most of the green leaves remain untouched until they die and are cast off from the plant, when they decay and are converted to detritus (Fenchel, 1970; Odum et al., 1973; Zieman, 1974a). In *Thalassia* beds, at least 90% or better of the energy makes its way to the higher trophic levels via the detritus food chain. Some of this detritus may be transported offshore to provide food far from the site of growth (Menzies et al., 1967; Menzies and Rowe, 1969).

Tropical sea grass beds are frequently found in the vicinity of coral reefs and the interactions between these two communities appear important. Particularly at night, fish, including valuable commercial species, leave the

reef to forage on species living in the grass beds (e.g. Earle, 1971). Populations of fishes living in reef communities are larger where the reefs are adjacent to seagrass beds (Randall, 1965; Starck and Davis, 1966; Ogden, 1972).

Tropical sea grass beds are among the most productive of natural ecosystems. Productivities of 1,300 to about 3,000 g dry weight/m per year have been recorded (Odum et al., 1973; Zieman, 1975).

Grass beds also stabilize both organic (Wood et al., 1969) and inorganic sediments (Ginsburg and Lowenstam, 1958; Scoffin, 1970; Zieman, 1972) and offer a substrate for epiphytic flora and fauna. Scoffin (1970) observed that in dense, long-bladed beds of *Thalassia testudinum* currents up to 40 cm/sec measured immediately above the blades are reduced to zero at the sediment—water interface.

Extremely high rates of nitrogen fixation have been reported by Goering and Parker (1972) by the epiphytes of *Thalassia testudinum* and by Patriquin and Knowles (1972) by bacteria in association with the roots. This implies that sea grass communities may be important sources of newly fixed nitrogen in the characteristically nitrogen-poor waters of the coastal tropics. McRoy et al. (1973) found much lower values of nitrogen fixation than the previous authors however, and the importance of this process in sea grass beds cannot be determined without considerably more research on different species in different areas.

Thalassia has been used as a feed supplement for sheep (Bauersfeld et al., 1969) and chickens (Alvarez, personal communication, 1973) and as fertilizer (Van Breedveld, 1966).

Sea grass beds are especially vulnerable to degradation by human-induced stresses as they inhibit only the shallow marginal seas where the activities of man are the greatest. Their mode of growth is unique for marine plants, and they are subject to damage from stresses or pollutants both in the water column and in the sediments.

POLLUTANTS AND STRESSES OF TROPICAL SEA GRASSES

In the tropics, there are only a few scattered examples of pollution studies in sea grass communities to draw from, and much of the information is in the form of difficult-to-obtain reports. Thus much of the discussion here will, of necessity, be drawn from these studies.

Dredging and filling

Of all forms of man-made or induced disturbances of the estuarine and nearshore environments, dredging and filling presents the greatest potential for damage to the sea grass beds and has undoubtedly caused the destruction

of more desirable grass bed habitat than any other form of pollution. Dredge and fill operations, which remove or dredge material from one portion of an estuary to another area to be filled, damage sea grasses in a variety of ways.

The most obvious destruction comes from the physical removal of the dredged material whether by a hydraulic dredge or by dragline and the covering of other marine areas with the resulting fill material. These two joint processes cause the destruction of sea grass beds for an indeterminate, but often considerable length of time, as the resulting sediments are not initially suitable for colonization by sea grasses. In the Caribbean region, large areas of the highly productive turtle grass, *Thalassia testudinum*, have not recovered many years following dredging operations.

In addition to the direct physical damage, secondary effects can be pronounced. Sea grasses generally require high light levels to sustain populations, and the available light is greatly decreased in the vicinity of dredge operations due to the turbidity caused by suspended sediments, despite protective devices designed to contain the suspended material such as turbidity curtains or screens. The fauna associated with grass beds are highly susceptible to damage both from the effects of direct siltation and suffocation, as well as from low oxygen levels due to the high oxygen demand of the decomposing suspended organic matter.

An area where extensive dredging and filling operations have been used to create waterfront homesites is Boca Ciega Bay, Florida. Here Taylor and Saloman (1968), found that between 1950 and 1968, 1,400 acres (ca. 567 ha) of the bay had been filled, reducing its area by 20%. Sediments in undisturbed areas of the bay were firm and averaged 94% sand and shell and supported luxuriant beds of sea grass, primarily turtle grass. In contrast, the sediments in dredged canals averaged 92% silt and clay. Light penetration was highest in the southern bay away from the land fills, lowest in the bay near fill areas, and increased again in the dredged canals. The canals, however, were sufficiently deep that adequate light for sea grass photosynthesis did not reach the bottom.

The soft sediment of the canals is generally unsuitable for most macro fauna as well as for sea grasses. A variety of studies (Phillips, 1960; Springer and Woodburn, 1960; Taylor and Saloman, 1968) listed nearly 700 species of marine plants and animals from Boca Ciega Bay. Taylor and Saloman (1968) found the deeply dredged canals contained less than 20% of the species recorded from the bay and these remaining species were nearly all fishes which were highly motile and not restricted to the canals. Thus, in ten years following the disturbance of the bay bottom recolonization of the sediments by plants and animals was negligible. Using conservative and incomplete figures, Taylor and Saloman (1968) estimated an annual loss in fisheries and water sports of about $1.4 million per year from the bayfill operations.

Godcharles (1971) studied the effects of a commercial hydraulic clam

dredge on the benthic communities in the estuarine areas of Tampa and Tarpon Springs, Florida. He found the hydraulic dredge sampled clams effectively, but uprooted or severed the rhizomes of all sea grasses in its path. The dredged areas filled with fine-grained sediment, and at one station some recovery of *Halodule beaudettei* (*Diplanthera wrightii*) occurred, but no regrowth of *Thalassia testudinum* or *Syringodium filiforme* was ever observed.

Odum (1963) found that the chlorophyll *a* and productivity of turtle grass beds diminished following dredging operations in Redfish Bay, Texas. The year following dredging operations, chlorophyll *a* and productivity increased except where the communities had been covered with soft silt.

In addition, Odum reported that the spill from dredging destroyed grass beds when discharged into areas only 0.5 m deep as it caused the bottom to shoal critically to a point where plants could no longer survive. Odum attributed the increased productivity to the release of nutrients from the displaced sediments.

In Lindbergh Bay, St. Thomas, Virgin Islands, increased turbidity brought about by dredging resulted in a marked reduction in the distribution and abundance of benthic sea grasses (Van Eepoel et al., 1971). In 1968, "benthic grasses could be easily seen from the surface in water 10 meters in depth; the *Thalassia* community was lush, covering the major portion of the bay bottom". By 1971 *Thalassia* along with *Syringodium*, the second most common sea grass in the bay, had become extremely sparse throughout, with "only very few plants being found deeper than 2.5 meters. In the most turbid portions of the bay they were limited to 1.5 meters in depth." Similar destruction of seagrass beds by sedimentation and turbidity caused by dredging in Brewers Bay, St. Thomas, is described by Grigg et al. (1971). The light compensation point for *Thalassia testudinum* in clear waters is reported to be about 40 m (Pomeroy, 1960; Marmelstein et al., 1968; Gessner, 1971), although this figure may not be reliable due to oxygen storage problems.

Eutrophication

The input of sewage or other nutrient-enriched substances into water near sea grass beds produces a number of effects. Sea grasses may have the ability to take up nutrients through the leaves as well as through the roots, and moderate amounts of excess nutrients could stimulate growth. Enrichment also fosters the development of epiphytic algae and phytoplankton. One of the most luxuriant growths of turtle grass and assorted epiphytic algae this author has observed was just inshore of the outfall of the Miami sewage plant on Virginia Key, Florida. Sea grasses are known to have a high light requirement, and the excessive growth of epiphytic algae and phytoplankton decrease the light energy available to the plants.

Taylor et al. (1973) found that significant portions of Hillsborough Bay, Florida, had been damaged by dredging and domestic and industrial efflu-

ents, particularly phosphorus and nitrogen compounds and suspended solids. This resulted in heavy growths of phytoplankton and filamentous algae, marked fluctuations in oxygen concentration, and high turbidity. This resulted in low light transmission which reduced the density and coverage of the sea grass beds, primarily *Thalassia*. Catches of fish and crustaceans were low compared with other areas where environmental conditions were more natural and extensive beds of sea grasses persisted.

Concomitant with increasing sewage pollution and dredging in Christiansted Harbor, St. Croix, Virgin Islands, extensive beds of *Thalassia testudinum* have been overgrown and replaced by *Enteromorpha* (Dong et al., 1972). Over a 17-year period Nichols et al. (1972) documented a 66% reduction in grass beds in the bay.

McNulty (1970) found a lack of sea grasses in an area polluted by sewage effluents in Biscayne Bay, Florida. Only *Halophila decipiens* and *Halodule beaudettei* (*Diplanthera wrightii*) existed in low quantities within 1 km of the outfall. Past this point *Thalassia testudinum* began to occur in sparse quantities. Four years after the abatement of the sewage input the sea grasses had actually declined further, but the area had continued to be disturbed by heavy hydraulic dredging and the resultant increased turbidity, to which sea grasses are especially susceptible.

Sewage effluents (and suspended sediments) reduce dissolved oxygen levels. Hammer (1972a) reports that anaerobiosis depresses photosynthesis in *Thalassia testudinum* and *Halophila decipiens*. After 24 hours exposure to anaerobic conditions *Thalassia* recovered fully while *Halophila* did not. The greater tolerance of *Thalassia* is probably related to its greater ability to store oxygen in its tissues.

Temperature and salinity

Raising the temperature regime of tropical sea grasses can have far greater effects than a similar modification in a temperate environment since organisms in tropical waters live much closer to their thermal death point (Biebl, 1962; see also Zieman and Wood, Chapter 5).

Phillips (1960) reported that *Thalassia testudinum* preferred 20—30°C temperatures in south Florida. Zieman (1975) found that this species has an optimum temperature of between 28 and 30°C and net photosynthesis sharply declines on either side of this range.

Glynn (1968) observed that the leaves of *Thalassia* were killed in Puerto Rico by temperatures of 35—40°C, but that the rhizomes apparently were not affected.

Thorhaug and Stearns (1972) reported that *Thalassia* growing at artificially elevated temperatures produced flowers but no observed fruit. They also produced more new shoots than plants growing at normal temperatures.

Halodule beaudettei (*Diplanthera wrightii*) is an earlier successional

species than *Thalassia* in the Caribbean. It is usually the first sea grass colonizer in disturbed areas, as it is able to establish and spread rapidly.

Phillips (1960) found *Halodule beaudettei* to prefer temperatures of 20—30° C, as does *Thalassia*, but that *Halodule* is a more eurythermal tropical species. Manatee grass, *Syringodium filiforme*, is generally less abundant than *Thalassia* or *Halodule*, and its ecological position is less well defined. Phillips (1960) considers *Syringodium* a tropical plant but not a stenothermal one, although leaves of *Syringodium* begin to die at temperatures below 20° C.

Zieman (1970) found that sea grasses tend to be more resistant to thermal pollution stress than do the algae. The sea grasses tend to be more resistant to periodic acute temperature increases, and even if the leaves are killed off, rapid regeneration is often experienced from the horizontal rhizomes which lie several centimeters below the surface of the sediments. This regeneration occurs in various species because the rhizomes contain large starch reserves and are injured only if the temperature of the sediment reaches lethal levels for extensive periods of time. Phillips (1960) however, points out that frequent removal of leaves may so debilitate rhizomes that the capacity for recovery is impaired.

There is considerable variation in the observed salinity ranges of the Caribbean sea grasses. *Thalassia* has been found to tolerate salinities as low as 3.5‰ (Sculthorpe, 1967) and as high as 60‰ (McMillan and Moseley, 1967) for short periods of time, but usually undergoes defoliation at these extremes. Optimum salinities for *Thalassia* are in the range of 24—35‰ (Phillips 1960; McMillan and Moseley, 1967; Zieman, 1975). McMillan and Moseley (1967) found that *Halodule beaudettei* (*Diplanthera wrightii*) showed the greatest resistance to high-salinity conditions, *Thalassia* was intermediate, and *Syringodium* was the least resistant. This agreed with the observations of Phillips (1960) who found *Halodule* existing in freshwater for short periods of time. He found *Halodule* to be the most euryhaline species, but that it was outcompeted by *Thalassia* and *Syringodium* whenever they were present.

Chesher (Chapter 6) reports that, although the effluent of heated brine from the Key West desalination plant caused a marked reduction in biotic diversity in the vicinity of the discharge, nearby beds of *Thalassia* were unaffected.

Halodule beaudettei is more euryhaline than *Syringodium filiforme*, and tolerates salinities of 9—52.5‰, whereas *Syringodium* only survives to 35‰ (McMahan, 1968).

Hammer (1968) found the photosynthesis of sea grasses, including *Thalassia* and *Syringodium*, to reach a maximum in full strength seawater, and to decrease as the salinity decreased. Hammer ascribed part of the decrease to direct salinity effects, but pointed out that, with dilution, the available carbon supply was diluted and may also have become limiting.

Zieman (1970, 1974b) found that the seasonality of *Thalassia* could be ascribed largely to temperature and salinity effects. When temperature and salinity vary together, the combination of high temperature and low salinities was found to cause the greatest decline in the plant populations.

Tabb et al. (1962) state: "Most of the effects of man-made changes on plant and animal populations in Florida estuaries (and in many particulars in estuaries in adjacent regions of the Gulf of Mexico and South Atlantic) are a result of alterations in salinity and turbidity. . . High salinities (30—40 ppt) favor the survival of certain species like sea trout, redfish and other marine fishes, and therefore improve angling for these species. On the other hand these higher salinities reduce survival of the young stages of such important species as commercial penaeid shrimp, menhaden, oysters and others. It seems clear that the balance favors the low to moderate salinity situation over the high salinity. Therefore, control in southern estuaries should be in the direction of maintaining the supply of sufficient quantities of fresh water which would result in estuarine salinities of 18 to 30 ppt."

Wind and sea stress

Sea grasses occurring in the tropical marine environment are periodically subject to severe stress from tropical hurricanes and typhoons. Although man cannot generate the extreme winds and resultant wave stress on the beds as a major tropical cyclonic storm, man can influence the manner in which the waves interact with sea grasses by means of his modification of the nearshore sea bottom.

Of all marine communities which are affected by the storm-generated winds, subtidal sea grass beds appear to be among the least susceptible to severe damage by the storms.

Thomas et al. (1961) found that following Hurricane Donna in 1960, large quantities of *Thalassia testudinum* washed onshore. However, there was little quantifiable loss within the beds, although Tabb and Jones (1962) observed heavy mortality of fishes and invertebrates from the same storm in North Florida Bay. Mortalities were caused by mud suffocation, by stranding as the water retreated, and by severe oxygen depletion due to decomposition of suspended organic matter.

In Texas, Oppenheimer (1963) found that inshore turtle grass flats and bottom flora were undamaged, although major erosion occurred as Hurricane Carla passed near Port Aransas, Texas, in 1961.

Glynn et al. (1964) recorded that following Hurricane Edith, a relatively mild storm in September 1963, *Thalassia* beds were not noticeably damaged, although mangroves and beds of *Syringodium* suffered some effects and the coral reef community was severely disturbed.

Population explosions and overgrazing

Overgrazing of sea grasses occurs commonly in the immediate vicinity of patch reefs, leading to the formation of bare "halos" surrounding Caribbean patch reefs. These halos have been attributed to grazing reef fish, primarily surgeon fish (Acanthuridae) and parrot fish (Scaridae) (Randall, 1965) and more recently to the echinoid *Diadema antillarum* (Ogden, 1972). The grazing is localized and presents no threat to the overall grass beds, as the small grazers generally do not venture far from the shelter of the nearby patch reef (Mathison et al., 1971).

However, Camp et al. (1973) have observed dense concentrations of the regular urchin *Lythechinus variegatus* causing massive destruction of mixed sea grass beds, primarily composed of *Thalassia*, on the west coast of Florida. During these outbreaks, areas hundreds of meters long and meters across are completely denuded by the migrations of these urchins. It is doubtful if these outbreaks are of severe consequence compared to the grazing pressure exerted formerly during seasonal migrations of large numbers of sea turtles.

The cause for urchin outbreaks has not been well studied. It is possible that it is due simply to exceptionally successful spawning and recruitment. However, there is the possibility that these outbreaks are due to the reduced predation of the urchins in areas where the normal predators have been overfished.

Oil pollution

As sea grasses are generally subtidal, they are less susceptible to damage from oil than other organisms, especially those in the intertidal zone. Still, shallow beds of sea grasses can suffer destruction from severe oil spills. Diaz-Piferrer (1962) found beds of *Thalassia testudinum* were badly affected by crude oil on the south shore of Puerto Rico. The turtle grass deteriorated over a period of several months, and the normal algal flora was denuded and replaced by blue-green algae. Following the much publicized Santa Barbara oil spill in January 1969, *Phyllospadix torreyi* growing in the intertidal zone was severely damaged (Foster et al., 1971), although subtidal plants and those in the extreme low intertidal zone were largely undamaged as they were, to some degree, protected from direct contact with the oil.

In addition to direct mortality from the contact of the plants with the oil, a second and possibly longer-term damage is caused by the interaction of the sediment and oil. Sea grasses tend to stabilize the fine-grained sediments, but as oil seeps into sediment, it agglomerates into large lumps which offer a greater purchase for waves and currents to act on, and increase the buoyancy of the sediments, thus making removal of the sediments easier. Diaz-Piferrer (1962) recorded the loss of 3,000 m^3 of sand from Tamarindo Beach in less than a week due to this effect.

Metals

Thus far metal pollution has not been implicated in the decline or destruction of marine grass beds. As many sea grasses require a reducing environment around the roots (Wood et al., 1969) but an oxidized environment surrounding the leaves, they can pick up metals in a variety of forms.

Many types of metals are found in abundance in polluted sediments where they may become easily mobilized and be taken up by the roots of sea grasses. Parker et al. (1963) found the highest concentrations of cobalt and manganese in Texas bays in sea grasses, the concentrations being over an order of magnitude greater than the concentration in the sediments. Iron was in nearly the same concentration in sea grasses as the sediments, but was more concentrated in sea grasses than in any other plant or animal.

Parker (1962) found the zinc concentration of *Thalassia* and *Halodule beaudetti* (*Diplanthera wrightii*) was 5—10 times the concentration in the sediments. Thus, there is the possibility that excess metals in estuaries will be concentrated in sea grasses and available for passing up the food chain. This is of increasing concern as metals are released into the environment in large quantities from nuclear power plants (e.g. Segar and Pellenbarg, 1973) and desalination plants (e.g. Chesher, Chapter 6), both of which are being constructed at accelerating rates in the tropical inshore regions.

Physical disturbance and recolonization of sea grasses

Physical disturbance of grass beds due to boat traffic on shallow beds has been observed by students of sea grass ecology for quite some time (U.S. Dep. of Int., 1973), although there has been no systematic study of the effects of propeller cuts on grass beds. Phillips (1960) states that "in shallow bays perhaps the most common means of breaking the continuity of growth is the destruction caused by boat propellers". Phillips removed *Thalassia* rhizomes and shoots from two plots in April and May of 1958 in Boca Ciega Bay, and observed no regrowth into the areas after ten months, although the more rapidly growing *Halodule beaudetti* (*Diplanthera wrightii*) recolonized the denuded plots.

Scoffin (1970) also found that *Thalassia* does not spread its rhizome mat rapidly. In a 4-month experiment, he found trenches dug in the sediment were not recolonized.

In attempting to establish the rate of vegetative growth of *Thalassia*, Jones (1968) cut two trenches in grass beds in Bear Cut, Biscayne Bay, Florida, in August 1965. Re-excavation two years later, in August 1967, showed little regrowth into the 1 m × 30 cm × 20 cm deep cuts, although the trenches had filled with sand within a month of cutting. Jones also noted that *Halodule beaudetti* (*Diplanthera wrightii*) had recolonized the upper 2 cm of sediment with 270 cm of rhizome and about 20 growing rhizome tips.

Having heard from local fishermen that propeller cuts were persistent features and sometimes useful in relocating fishing holes and navigation channels, this author used boat tracks in grass beds as station location markers, in areas where it was impractical or imprudent to use marker buoys. Several of these cuts were used reliably for two years (1966—1968) in the Soldier Key region of Biscayne Bay and at least one was recognizable after three years (1966—1969).

Fuss and Kelly (1969) and Kelly et al. (1971) studied the survival of transplanted *Thalassia* and the feasibility of recolonizing turtle grass in disturbed areas. They found that sprigs of *Thalassia* require considerable time to initiate growth after disturbance. Fuss and Kelly (1969) found that ten months was required for new rhizome apices to develop, and Kelly et al. found that in a control bed 40% of the transplants had initiated new rhizome growth after thirteen months, whereas in an area with altered sediments only 15—18% of the transplants exhibited new growth.

If only the leaves of *Thalassia* are killed, or removed, regrowth is often rapid because of the starch reserves in the rhizomes. Too frequent leaf destruction will use up these reserves however. Taylor et al. (1973) found that *Thalassia* in Tampa Bay, Florida, tolerated harvesting twice a year. They speculate that this species might tolerate three harvests per year in more tropical areas.

Thorhaug (1974) has reported success in establishing *Thalassia* from seeds in south Florida.

Patriquin (1972) studied the nutrient sources and requirements of *Thalassia* in Barbados, and found that a reduced sediment layer is essential for good growth of turtle grass. Wood (1965) also noted this requirement and found that sea grasses stimulate the development of reducing conditions in the root layer.

Zieman (1974a) found a reduction of the finer components of the sediments in the disturbed *Thalassia* beds due to the lack of the stabilizing effect of the grass blades. This effect is even more pronounced in areas subject to continual boat traffic such as scoured ruts which are used by boaters as channels through the shallow beds and banks. Zieman also found a reduction in the pH and generally in the Eh of the disturbed sediments, yielding a more reducing environment. He suggested that *Thalassia* beds did not recover rapidly from physical disturbances such as boat traffic or dredging because: (1) the disturbed areas, whether sedimented in or scoured open, are not a suitable environment for rhizome growth; (2) when damaged, rhizomes normally require one year just to begin developing active new apices; and (3) rhizome apices, whether arising new from damaged *Thalassia* plants or simply growing from other portions of the grass bed, do not grow rapidly. The particular circumstances at each site determine which of these mechanisms is most important.

Recolonization of disturbed *Halodule beaudetti* (*Diplanthera wrightii*),

shoal grass, is not likely to be a serious problem. This plant does not have a well-developed deep rhizome system, grows well from seeds, and is capable of colonizing a damaged area in a short time. It does not seem as sensitive to sedimentation as *Thalassia*. The author has seen large mats of shoal grass attached to the substrate at only one edge, with the rest of the mat growing floating in the water column with roots and rhizomes unattached. *Halodule* can colonize an area in but a few weeks when conditions are correct, and exists as an annual plant in many estuarine areas. Once *Thalassia* becomes established, however, *Halodule* cannot compete, and is overgrown in all but the upper intertidal zones, or in areas in which turtle grass cannot become established. Den Hartog (1969) reported that in the tropical Indo-Pacific *H. uninervis* and *H. pinifolia* are the first sea grasses to appear following the disturbance of the sea bottom, and that they are gradually replaced by other species.

CONCLUSIONS

As knowledge of the structure and function of sea grass beds increases, it becomes evident that these areas are important components of the nearshore tropical marine ecosystem. A variety of studies have shown how grass beds act as nurseries, food producers, and stabilizers of the sea floor. As increasing development in the tropics begins to encroach upon these valuable resources, careful attention must be given to their fate, as many coastal people depend upon the resources of tropical grass beds for their food and sustenance, from the conch-fishermen of the Bahamas to the turtlers of the western Caribbean. In more developed areas, the grass beds are sources of sport-fishing and other forms of recreation.

Fortunately, recognition of the importance of grass beds to the ecology and ultimately to the economy is beginning to be recognized. The State of Florida, through its Coastal Coordinating Council, has declared grass beds to be preservation areas where no development is permitted. This is in part due to the recognition of the linkage between ecology and economy. An area that is dependent on its income from recreation, and primarily water-oriented recreation, must maintain high water quality standards and esthetic values.

Let us hope that other responsible decisions will result in the continued maintenance of the nearshore tropical marine environment.

CHAPTER 5

EFFECTS OF THERMAL POLLUTION ON TROPICAL-TYPE ESTUARIES, WITH EMPHASIS ON BISCAYNE BAY, FLORIDA

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GENERAL CONSIDERATIONS

Tropical estuarine regions are beginning to undergo the rapid development and industrialization which has damaged estuarine and neritic ecosystems of the temperate zone. Nearly all thermal pollution studies have been made in temperate waters and the body of quantitative information on the ecology of tropical marine and estuarine environments is still quite small.

The problem of thermal additions to the tropical oceans is similar qualitatively to that in temperate water. Any additions are potentially more harmful in the tropics, however, since tropical organisms live closer to their upper thermal limits. Usually, even in the tropics, the effluent will be warmer than the surface ocean water, and, if the intake is from an estuary, the salinity will normally be lower, so the heated water will tend to lie on the surface until mixed. Here it is subject to maximal evaporation, and mixing by wave and wind. Only in areas where there is a continual, onshore wind over long periods, is there likely to be any extensive heat damage to sedentary animal and plant communities, or near a high temperature—high salinity outfall from a combination power generation—desalination plant, where the effluent will sink to the bottom. Each ocean outfall will have its own properties, and should be the subject of a separate study; consideration will have to be given to wind, tide, and current patterns, distance from shore, relations of sandy beach and rocky headland, depth of water and the presence or absence of submarine canyons and upwelling.

Estuarine systems have been shown to be extremely important ecologically, but unfortunately engineers seem mesmerized by the possibility of discharging heated effluents into them, even where it is possible to put the waste into the ocean. One would think that thermal considerations alone would persuade those who desire to use seawater for cooling to direct their outfalls into the ocean so as to avoid any possibility of heating shallow areas, and raising the temperature of the intake water; however, this is often disregarded in practice. For example, in the original design and operation of the Turkey Point power station on South Biscayne Bay, Florida, the intake was situated to the north of the outfall in such a way that the warm water was

carried into the intake channel by prevailing northerly movements of the shallow water. This resulted in a virtual recirculation of the warmed water from the effluent in a small coastal lagoon less than 3 m deep. It would have been possible to site the power plant with easy access to an oceanic discharge point, with a strong current flowing past the outfall to insure quick mixing.

The term tropical-type estuaries has been used to signify those estuaries which may or may not be situated within the actual latitudes of the tropics but have the characteristics of tropical environments due to warm ocean currents moving close inshore. On the east coast of continents in both hemispheres, water circulation is such that tropical waters extend beyond the defined limits of the tropics (23°), so coasts of such regions as the northern half of New South Wales and southern Florida are still ecologically in the tropics. Thus, within the southernmost part of Florida, "there are many more species of West Indian marine life than in any one area of the Caribbean, with the exception of the islands of Hispaniola and Cuba and perhaps Jamaica" (Committee on Inshore and Estuarine Pollution, 1969), while Moreton Bay in southern Queensland has large numbers of species from the Coral Sea and the Solomon Islands over 2,000 km to the north. In shallow waters such as these, where there is high insolation and a low evaporation rate owing to high humidity, water temperatures will not be much, if any, lower than in truly tropical waters. Thus, in Australia, on the east coast, tropical species of sedentary plants and animals occur as far south as Long Reef, just north of Sydney, close to 34°S latitude, while Moreton Bay, at 27°S, is quite tropical in character. Florida waters have a still more tropical habitat due to the Florida Current derived from the Caribbean and to the prevailing onshore summer winds blowing over warm waters, though the latitude is north of 25°N.

Most tropical and subtropical estuaries are of the barrier lagoon type, though drowned valley estuarines do occur, such as those of Puerto Rico, Indonesia, and northern Australia. In these regions, coral reefs are often present. Because the terrain around the shores of drowned valleys tends to be somewhat rugged, industry prefers to build around the barrier lagoon type of estuary, where the hinterland tends to be flat. The world's largest rivers such as the Ganges, Congo, and Amazon are not the scene of much secondary industry which would occasion thermal pollution on any large scale, and possess a vast, though not unlimited, dilution capacity.

Industrial development in the true tropics, i.e. between latitudes 23°N and 23°S, has not resulted in studies of the effects of thermal pollution until recently. At present, therefore, we must be content with research carried out largely in quasitropical environments, and the most comprehensive studies thus far have been in South Biscayne Bay, Florida, which will be discussed below.

Let us consider first the ecological situation in several types of estuaries, and the likely effects of thermal pollution.

The barrier lagoon environment

Barrier lagoons are, in the main, shallow, elongated and narrow, with openings at intervals, and usually have short rivers coming in, though not necessarily opposite the openings to the ocean. The bottom may be silty or sandy, with, at times, ridges of more or less consolidated rock, generally parallel with the shoreline. At times some of these lagoons may be entirely out off from the sea and, when they break out again, the new opening may not be at the site of the previous entrance. These lagoons have usually a strong tendency to silt up, since suspended terrigenous sediments have restricted access to the sea. Lagoons of this type are exemplified by the Texas bays from the Laguna Madre at Port Isabel to Galveston Bay, and stretch from Florida Bay along much of the coast of the United States almost to the tip of Long Island, representing environments from tropical to cool-temperate. Similar strings of coastal lagoons, though not so continuous, stretch along the east coast of Australia, and are represented in Western Australia by Cockburn Sound and the shallow areas north of this.

In some lagoons, temperatures over 40°C may be encountered, as in Baffin Bay, near Corpus Christi, Texas, parts of Florida Bay, and Steward's Point on the Macleay River in New South Wales. Such temperatures are lethal to many algae, but are not sufficiently prolonged to kill the sea grass rhizomes, which persist in most cases, though exposed leaves often die off.

The shallow lagoons are rarely stratified, except during flash floods or heavy local rains, but salinities may vary considerably with the season. Parts of the Texas Bays may have salinities up to 60‰ (Oppenheimer and Wood, 1965), and Florida Bay also has very high salinities at times. After heavy rains, the salinity may drop to 0.1‰, so most of the organisms present are euryhaline. Thermal effluents, even in these shallow estuaries, produce stratification and, in areas with heavy summer rains, the cool, fresh water may flow over the top of the bay water, though the latter is heated by the effluent. This will cause a temperature inversion and insulation of the bottom water from the atmosphere and thus retard cooling.

Drowned valleys

In drowned valley estuaries, because of their greater depth, stratification is frequent, there is an annual turnover, and temperature and salinity gradients are not so steep. Many organisms can seek shelter from rapid changes by moving into the hypolimnion, so there is much greater chance of regeneration of species after local killing. One difficulty is that industrial cooling units usually take their water from the epilimnion and return it thereto with added heat so that stratification becomes more marked. Even so, there is much less risk of serious thermal pollution by industrial cooling in drowned

valleys than in coastal lagoons. However, the effects depend largely on the size of the industrial plant in relation to the area of the drowned valley or the portion of it available for the discharge. There is also the possibility of drawing cooler water from the hypolimnion, heating it, and returning it to the epilimnion at approximately the same temperature as the epilimnion. This must be considered carefully, however, using appropriate mathematical models.

River deltas

River deltas present special problems of a very complex nature, e.g. relations between depth, river volume and rate of flow, current and tide patterns at the mouth, flushing rates, the shape of the saltwater wedge and other factors. The degree of thermal pollution is most likely related to river flow rate.

BIOLOGICAL CONSIDERATIONS OF THERMAL POLLUTION IN THE TROPICS

Tropical environments often present quite different problems than the temperate ones. The first is that the organisms occurring in tropical waters are much closer to their thermal limits than those of temperate waters, especially in summer, may be either inhibitory or lethal, depending on their range, rate of change, and duration. Biebl (1962) has shown that the thermal death point of submerged algae in Puerto Rico is 32—35°C, only 4—6°C above the average summer maximum of 28°C, whereas for temperate algae from the north coast of France the upper limit is 27—30°C, which is 10.5—13.5°C above the average summer maximum of 16.5°C. The Committee on Inshore and Estuarine Pollution (1969) found that numerous "tropical marine organisms belonging to a widely divergent group of plants and animals all survive to temperatures of about 28°C, reduce their metabolism between 28° and 30°C, begin to die about 33—34°C. Mayer (1914, 1918b) was the first to recognize that tropical organisms normally live only a few degrees below their upper lethal temperatures, and Edmondson (1928) found that temperatures 5—6°C above summer temperatures killed 11 of 13 species of Hawaiian corals. Hohman and Tsuda (1973) found that above 28°C photosynthetic rates of *Caulerpa racemosa* decreased continuously, and that increasing the temperature from 28°C to 30°C doubled the respiration rate. Specimens of this algae incubated for 12.5 hours at 32°C had a P/R ratio much less than 1 and no plants at 34°C showed any net photosynthesis.

Biebl (1962) and Schwartz and Almodovar (1971) describe the effects of exposure and elevated temperature on a variety of coral reef algae. Thorhaug (1971) and Thorhaug and Fernandez (1973) report that changes in bioelectric potential across the cell membrane system of the giant celled macroalgae *Valonia* are a highly sensitive index of thermal stress. A variety of trop-

ical benthic green macro-algae tested in south Florida had abrupt thermal limits between 31° and 33°C (Bader et al., 1972). Temperature acclimation in these algae conferred adaptations of the order of only tenths of degrees.

Rupp (1973) found that temperatures of 34—36°C reduced fertilization and cleavage in a number of reef echinoderms. Early cleavage appeared to be more sensitive to elevated temperatures than fertilization. *Acanthaster planci* stop feeding and behave abnormally when the temperature reaches 33°C, and die within one week when held at 33—34°C (Yamaguchi, 1973).

Studies in Guam (Jones and Randall, 1973) have suggested the upper lethal limit for some corals is between 30° and 32°C (see Johannes, Chapter 2).

Graham (1971) found that the critical thermal maxima for three species of tropical Atlantic fish were higher than those of three closely related Pacific species.

In temperate waters, even lethal temperatures which kill off most of the existing local flora and fauna may not necessarily denude an area, as, under favorable conditions of currents and winds, the original association may be replaced by one derived from warmer regions, or by a resident population of warmer-water species living close to its temperature minimum, but restrained normally by competition with the local dominant populations (Naylor, 1965; North, 1966). Any artificial rise in temperature in tropical waters, however, will pose a greater threat to the biota, which, once eradicated, is largely not replaceable as long as the elevated temperature is sustained; there are very few replacement species able to colonize such environments, blue-green algae being an exception (see below).

This means that, if a thermal effluent is discharged into a tropical estuary, there is a strong possibility of permanent or long-term damage, the extent depending on the nature of the organisms, particularly the sessile plants and animals as opposed to the more motile forms, and on the relation between the area close to the discharge and the estuary as a whole.

Chadwick et al. (1950), Southward (1958), Singletary (1971), and Wood (1953) have shown that time is an important factor in defining the temperature tolerance of organisms. Most can stand considerable changes of temperature for a short time, but lesser changes over a longer period may be lethal. For this reason, diurnal fluctuations of temperature are usually of sufficiently short duration that the peak temperatures have little effect, except on the most vulnerable species.

Van Eepoel et al. (1971) reported that the winter—summer temperature range in parts of Lindberg Bay, St. Thomas, was reduced by about half at sampling stations in a plume of heated effluent. They noted that the environmental effects of limiting the seasonal water temperature range in the tropics is unknown.

With respect to the animals the sedentary species are likely to be more severely affected, as many, especially adult fishes and crustaceans, can

migrate. Jones and Randall (1973) found that fishes, crustaceans, and echinoderms characteristic of reef flat areas in Guam disappeared when heated water from a power plant was introduced. In addition there was considerable destruction of the sessile corals and marine algae.

Thorhaug et al. (1971) found upper temperature limits critical in the development of *Penaeus duorarum*. Maximum development rates were found between 25° and 31.5°C, survival was lessened at 33° to 34°C, and 18-h exposure to 37°C was lethal to all developmental stages.

Virnstein (1972) found that normal summer temperatures of 32—33°C in Tampa Bay restricted benthic fauna and caused a decrease in density and number of species, and that there was a large decrease in diversity and density of benthic infauna associated with the temperature of 34—37°C produced by a power plant. Glynn (1968) found high mortalities of echinoids, turtle grass, brachyuran crabs, chitons, and ophiuroids coincident with low-water conditions on shallow flats in Puerto Rico when temperatures rose to 35—40°C.

Some submerged algae such as *Gracilaria confervoides* (Wood, 1964), *Caulerpa* sp., *Cladophora* sp., and *Laurencia* sp. (Zieman, 1970), may be killed off in summer under natural conditions in shallow waters. Other species such as the coralline reds and calcareous green algal species, e.g., *Penicillus*, *Halimeda*, *Acetabularia* and *Melobesia*, are hardier, although Jones and Randall (1973) found heated effluents killed coralline algae.

Sea grasses are often less susceptible to thermal stress due to the rhizomes buried in the sediments (see Zieman, Chapter 6). The algae will be more easily killed by acute thermal stress and will disappear fairly rapidly from the scene at rates dependent on their thermal tolerances. Recolonization of algae from spores of plants which have been growing outside the thermal influence will occur as soon as the high temperature decreases. If the heating of the water in contact with the bottom is persistent enough and the temperature high enough to cause an increase in the temperature of the sediment, killing of the rhizomes can result (Zieman, 1970).

Sediments are poorer conductors of heat than seawater, so absorb heat more slowly. On the other hand, once the sediments become heated they will retain the heat much longer. Therefore, an intermittent heating of the water, if sufficiently great, can produce a lasting effect in the sediment, which will retain its heat after the water has cooled and act as a heat reservoir for the system. This fact is important for the sea grasses, but will also affect the rhizoids and spores of the seaweeds and will help to delay their regeneration after the water temperature has fallen again.

It is thus the duration of the heating as well as the temperature reached that will produce a given biological effect, and the heating, though intermittent, may have a cumulative result as sediment acts as an integrator of the fluctuating temperature. This is more likely to be critical for plant and animal populations in the tropics than in cooler waters and sediments. Legisla-

tion for regulating the effects of thermal effluents should include provisions for limiting the duration of heating as well as the temperature.

Reports of estuarine pollution more often describe the effects on plankton and larger, more obvious fish rather than on sedentary forms. In the case of phytoplankton, this emphasis is due to a basic misconception (see Johannes, Chapter 2). In studies which were made in two continents, Wood (1964) has found little evidence that phytoplankton is a major element in the food web of tropical estuaries. One rarely finds elements of phytoplankton in the gut of estuarine animals which prefer attached algae, either microscopic or macroscopic. Phytophagous fishes such as the mullets usually eat epiphytic algae, mainly diatoms and flagellates as well as some of the simpler macro-algae such as *Enteromorpha*.

These attached algae are, of course, much more abundant in the estuaries than the phytoplankton and, under ideal conditions, may present an annual production similar to that of the sea grasses (Wood, 1959, 1964), since the rate of reproduction, i.e. turnover, is far greater. Moreover, much of what is sometimes considered to be phytoplankton is frequently composed of detached epontic or benthic algae together with spores of macro-algae, many of which are motile. It is therefore of greater significance to study the benthic and epontic organisms than the phytoplankton when we are interested in the effects of thermal pollution.

Another difficulty in studying phytoplankton is that it fluctuates, both with regard to numbers and species, with time as well as with area (Wood, 1964). A bloom may occur in one part of an estuary while another bloom occurs several kilometers distant at the same time, and in stratified waters simultaneous blooms may occur one above the other.

The sedentary forms, of course, do not present a similar problem, and blooms of particular species tend to remain *in situ* for a week or more, and sometimes for months (Wood, unpublished). There are differences in the epontic community of estuaries from one part to another, and this has been reported by Wood (1964) from Lake Macquarie, New South Wales, and from the Texas bays. This must be taken into consideration in deciding the location of sampling stations and the provision of adequate controls.

In studying the structure of water at various temperatures Drost-Hansen (1969) found anomalies in the vicinity of 15, 30 and 45°C and noted abrupt changes in the activities of various organisms at these temperatures. Thorhaug (1969) showed a rapid decrease in the survival of *Valonia* and *Nitella* above 30°C, and Thorhaug et al. (1971) showed that the survival of several species of tropical algae and animals decreased in the range of 30—35°C.

With animals, the planktonic species, even if rapidly killed by heated water, may repopulate via larvae brought in from unheated areas, and from reproduction by survivors. Because of the haphazard chances of survival in a highly competitive environment, most marine animals produce a huge number of spawn, so that the death of a relatively large number may well result

in the survival of individuals that would normally die, and a very small diminution in the final population of the area. A heavy kill may, however, delay the reproductive peak of susceptible species, or allow the dominance of a competing species and thus upset the ecology of the system as a whole. The killing of certain ecologically important species can disrupt entire food chains as some species are selective in their feeding and may not readily accept as food other organisms which might appear to be quite similar to their favorite food.

Reeve and Cosper (1972) showed that *Acartia tonsa*, the dominant planktonic copepod in South Biscayne Bay, increased their tolerance to high temperature as the ambient temperature increased. The higher the ambient temperature to which the animal is adapted the less the mortality due to elevated temperatures. Reeve (1970) reported that when effluent temperatures exceeded 40°C the results were "immediate and catastrophic", with death of over 80% of the plankton. He also said that later work has "confirmed that the critical thermal maxima for many groups of plankton are in the mid-thirties".

Since the distribution of estuarine plankton is influenced strongly by circulation, a severe problem can develop if the circulation of a power plant is sufficiently large so as to dominate the normal estuarine circulation. In shallow bays and lagoons where this sometimes happens, the power plant can become a giant "predator" with the capacity to "consume" or destroy a large portion of the total plankton due to high temperature and recirculation.

In any estimation of the possibilities of damage to an estuary by thermal effluents, it is desirable, and should be considered essential, to make a study of the whole estuarine system before any industrial plant is constructed. However, except in remote areas, every estuary has, for many years, been subject to artificial stresses of one kind or another as long as there is any human population on its shores. There is no such thing as an unpolluted estuary in populated areas. It is still essential to fix a datum line for each estuary, and the sooner this is done the better.

Of course, such a program is wishful thinking, as few, if any, governments or other sponsors are willing to invest in a pollution research project which has no immediate effect on a recognized pollution problem. This is a tragedy, as the first defense of a polluter has often been that there is no adequate criterion of previous pollution, and therefore of how much he is contributing to the problem. He maintains that his pollution is minor when compared with the damage already done, and this may be very difficult to prove untrue.

A problem of still greater difficulty, and one that is almost completely unsolved, is the assessment of sub-lethal effects. Nearly all studies of necessity deal with acute, or short-term, severe effects. Studies dealing with chronic stresses, those that are sub-lethal in a short-time span but may prove lethal

if continued for a sufficient period, are scarce. One such effect, noted by Zieman (1970) for the sea grass *Thalassia*, will be discussed below, but studies of decreased swimming and feeding abilities, decreased reproductive capacity, and generally decreased competitive ability due to sub-lethal increases are few. These are exceptionally difficult problems to assess. Jones and Randall (1973), Mayer (1914), and Edmondson (1928) studied sub-lethal effects on corals and have found that sub-lethal temperature reduced the growth rate of corals in Guam (see Johannes, Chapter 2). Hohman and Tsuda (1973) found sub-lethal temperature increased respiration and reduced net photosynthesis in *Caulerpa racemosa* from Guam.

ASPECTS OF THERMAL POLLUTION IN A TROPICAL-TYPE ESTUARY — BISCAYNE BAY

The history of this study, mainly carried out by the University of Miami, is a classical example of frustration in science, especially when it is related to industrial expansion and the improvement of related facilities. A small but adequate proposal for an ecological study of South Biscayne Bay was submitted by the University of Miami as soon as the power plant at Turkey Point was projected, and before it was approved. This proposal was rejected after considerable delay, on the grounds that it was not extensive enough, and a much larger proposal was submitted to a federal agency, which then administered the desired funds. This was rejected because it was too large, and finally, a third proposal was accepted that included twelve senior research scientists instead of the three in the original proposal but with about a third of the funds originally asked for and was divided between several funding agencies.

This meant that a comprehensive, or even adequate study was impossible, and that one of the fossil fuel generators of 432 MW was already in operation when the study commenced. For this reason, no baseline information other than that available from aerial photography or obtainable from studying the history of recent sediments was available on the ecology of South Biscayne Bay prior to the starting of active operations by the Florida Power and Light Company.

As a result of the failure to establish a baseline, the scientists studying South Biscayne Bay have had to either select control areas well away from the part affected by the plume of hot water discharge or use gradient analysis.

Characteristics of South Biscayne Bay

Physical setting

Biscayne Bay is a shallow tropical estuarine lagoon situated on the extreme southwest portion of the Florida peninsula (Fig. 1). The bay as a

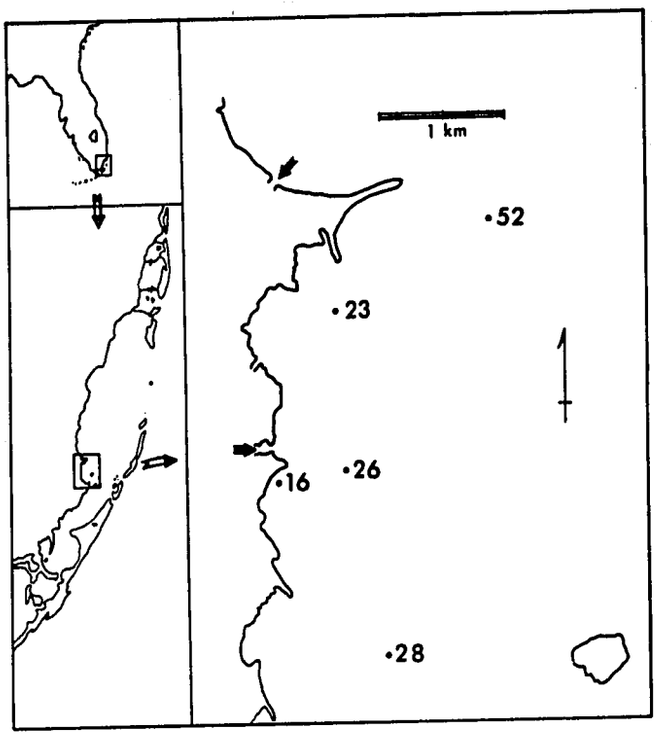

Fig. 1. Location map of south Florida showing Turkey Point and indicating several stations.

whole is about 56 km in length and averages about 8 km in width with a maximum width of about 16 km. Turkey Point is situated in Southern Biscayne Bay, which is divided from the northern and central portions by Featherbed Banks. Southern Biscayne Bay is about 16 by 20 km. It is bordered on the west by the mainland mangrove swamps and on the east by Elliot Key, which forms an unbroken barrier. Communication with the ocean is through Caesar's Creek at the southern end of Elliot Key, and through Boca Chica Cut at the north. There is some communication with the central portion of the bay around the west end of the Featherbed Banks, and across them at high tide, and with Card Sound to the south across Cutter Bank.

The local climate is tropical, due to warm currents moving north through the Straits of Florida from the Caribbean Sea. East of the Straits of Florida lie the Bahamas with shallow, warm waters, and east of them again is the northward-flowing Antilles Current derived from the North Equatorial Current. Thus, the trade winds blowing from the east are kept continually warm and near-saturated with moisture. The average summer air temperature is in the upper 20s and lower 30s, while the winter air temperatures are around 18–20°C. The summer humidity is consistently around 90–100%, and this

condition extends from May through October. In the summer, when the winds are from the east to southeast, the potential for extensive evaporative cooling from the surface of the bay is low. Precipitation is high, averaging about 150 cm per year, with 75—80% falling from June to November. The average at Homestead, just north of Turkey Point, is about 168 cm per year.

The tides are small, averaging 0.5 m. Circulation is generally tidal but may be overpowered by the wind. A large part of the subsequent problems surrounding Turkey Point was due to the failure to ascertain the true circulation in the vicinity of the power plant. It was originally believed that southern Biscayne Bay was well flushed because the tidal prism was 1/3 of the low-water volume of the bay, and the flow was predominantly NNE of the ebb and SSW of the flood (Richardson, 1966). Later studies, however, showed that this part of the bay is not well flushed.

Ambient temperatures in the inshore areas of southern Biscayne Bay average about 17°C in the winter and rise to 31°C in the summer. Minimum temperatures observed in the study area were 9—10°C during an extreme cold spell which caused considerable cold mortality. Maximum inshore temperatures were near 33°C but reached 36—37°C in shallow tide pools (Roessler and Zieman, 1969; Roessler, 1971). There was considerable modification of the normal thermal distribution by the effluent plume. Fig. 2 shows the average summer temperature distribution for the summer of 1967.

There is no large single source of freshwater input into the southern part of the bay. Freshwater comes from land runoff and numerous drainage canals. The drainage canals influence the near-shore salinity just north of Turkey Point, but there are none of these between Turkey Point and Card Sound. The salinity of inshore southern Biscayne Bay averages 27—33‰ in the summer and 31—34‰ in the winter; the salinity is often reduced following heavy rains to around 10‰ and salinities of 35—36‰ are reached during dry spells. The salinity distribution is one of longshore isohalines parallel to the shoreline and is not significantly altered by the Turkey Point effluent (Fig. 3).

Biological setting

The western shoreline of southern Biscayne Bay is a fringing mangrove swamp. The red mangrove *Rhizophora mangle* is dominant. Just seaward of the mangrove fringe, the sediments are 1—2 m thick, and are composed of a mixture of paralic and freshwater peats, calcitic mud, recent organic debris and detritus, shell hash and *Halimeda* plates. This shoreline forms a gradually sloping wedge which extends 350—400 m into the bay.

Among the algae, *Batophora oerstedi*, *Acetabularia crenulata*, and *Cladophora* sp. occur commonly throughout the zone. Occasional specimens of *Laurencia* sp., *Avrainvillea nigricans*, *Anadyomene stellata*, and *Digenia simplex* are found. During the winter, various ephemeral brown and red algae appear briefly and disappear.

Laurencia is frequently epiphytic, but is capable of independent existence,

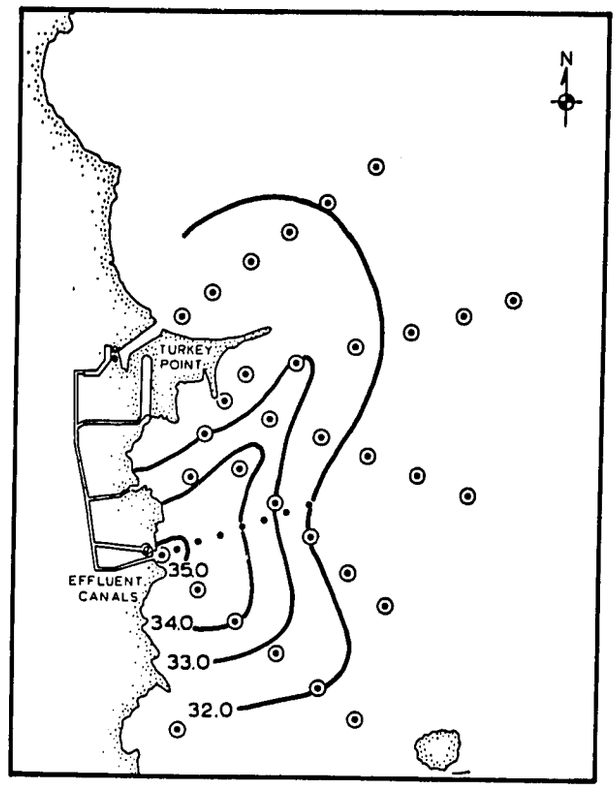

Fig. 2. Average summer temperature distribution off Turkey Point.

and tends to form parallel, generally longshore windrows and bundles when it is abundant. Its value in assessing pollution is very limited, as it is impossible to tell if a plant has existed in a particular locality or has recently been deposited there by the currents.

Beyond the inshore sediment wedge, two distinct community associations are found, which are regulated by the depth of the sediment. The first community consists of the *Thalassia* beds which are generally circular or oval in shape, occurring in areas of deep sediment (1—4 m) of remnant mangrove peat overlain with recent estuarine sediments (Zieman, 1972). These are often pure stands of *Thalassia* with few or no other higher plants in association, although some beds do have sporadic individuals of *Penicillus capitatus* associated with them. The *Thalassia* is also host to a number of epiphytes, including small macro-algae and large numbers of diatoms.

Other than the *Thalassia* beds, the area is nearly all Pleistocene limestone with little or no sediment. The thin veneer of sediment present is usually calcitic, with far less organic matter present than occurs in the inshore sediments. All of the algal species mentioned previously are present, though there is rarely a dominant species of fixed algae. *Udotea conglutinata*, *Rhipo-*

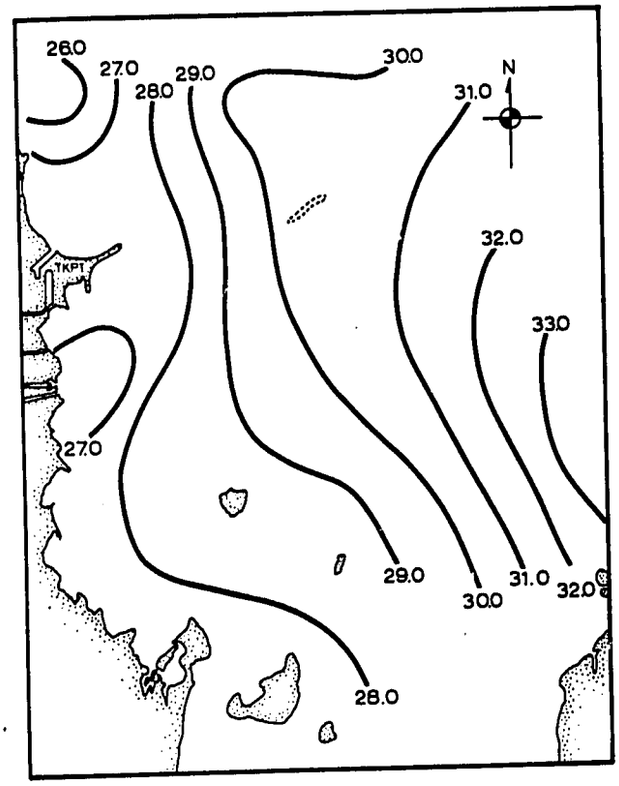

Fig. 3. Average salinity distribution off Turkey Point.

cephalus phoenix, *Sargassum pteropleuron*, and *Caulerpa paspaloides* appear abundantly. In addition there are more species of short-term or ephemeral reds and browns which appear primarily in the late winter months.

A detailed list of the plants and animals occurring in the area is given by Voss et al. (1969).

Thermal pollution studies at Turkey Point

Early operations

The Turkey Point power station is situated on the western shore of southern Biscayne Bay as shown in Fig. 1. The plant is designed to consist of two fossil fuel units, each generating 432 MW and requiring 5.9 m^3 /sec of cooling water heated about 5° C, and two nuclear units generating 760 MW each and requiring 13.8 m^3 /sec of water heated 6—8° C, in the original design. This design called for a discharge at full operations of an effluent of 39.6 m^3 /sec heated 6—8° C; however, as soon as the plant began operations with the small conventional units, it became obvious that there were some serious problems with the design of the cooling system.

As shown in Fig. 1, the intake for the power plant is just to the north of Turkey Point itself, in a barge canal. The initial discharge canals emptied most of the effluent about 1.5 km south of the intake at Turtle Point. This was the cooling system in use during the studies which will be discussed here. The ecologic damage occurred just off this discharge point. Initially this cooling-canal design was thought to be sufficient for the entire plant, including the nuclear units. However, insufficient attention was paid to the local current patterns until the oil-fired conventional generators were running, and it was discovered that the units were recirculating considerable amounts of their effluent.

Studies began in the Turkey Point vicinity about May of 1968. Early work was devoted largely to description of the area. It proved extremely important that studies commenced at this time. The first conventional unit began operation in the spring of 1967, with the second unit following about a year later. Although temperatures were beginning to rise with the onset of summer, and the increase in plant operations, the earliest surveys were made when the area was largely undisturbed. As opposed to some effects which will be related later, the benthic plant communities were initially the same just off the main discharge point as they were in areas to the immediate north and south. Damage at this time was limited to just a few hectares but increased throughout the summer and into the fall.

As the heat stress increased, the damage developed in a consistent pattern. Algae disappeared according to the thermal tolerances described below. The calcareous green algae and the sea grasses were the most heat-tolerant plants

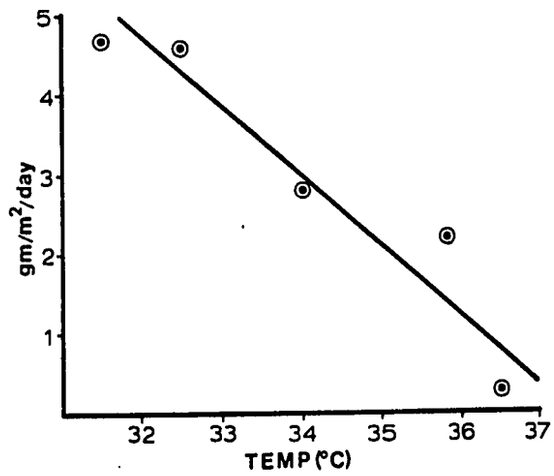

Fig. 4. Results of early studies at Turkey Point: decrease in productivity of *Thalassia* with increasing temperature.

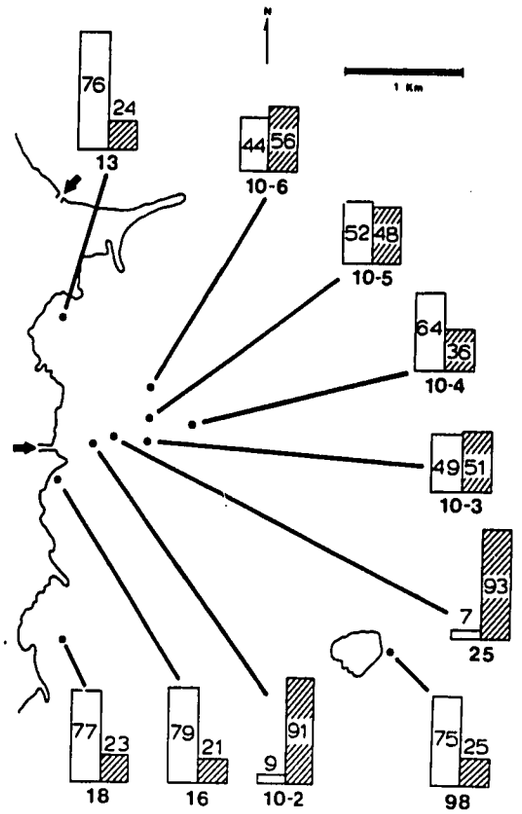

Fig. 5. Relationship between heat death of *Thalassia* leaves and the thermal effluent. Numbers on left (clear) are % of attached leaves which are alive, while numbers on right (slashed) are % of total leaves that are dead.

of the natural community. Ultimately, after much erosion due to the velocity of the effluent, the area was carpeted by a blue-green algal mat.

Preliminary studies showed the effect of temperature on the growth rate of *Thalassia* blades (Zieman, 1968; 1975). Growth rate decreased rapidly with temperatures above summer ambient (Fig. 4). There was also rapid increase in the death rate of *Thalassia* leaves with increases in temperature above ambient (Fig. 5), causing a decline in the abundance and density. By the end of September 1968, 14 ha of sea grasses had been destroyed.

Average summer temperatures for the Turkey Point region are shown in Fig. 2. By the summer of 1969 the area in which the average temperature was raised +5°C or more was 16—20 ha; 30 more ha were raised +4°C, 70 ha additional were raised +3°C and a further 120 ha were raised +2°C, in temperature (Fig. 6; Roessler and Zieman, 1969; Roessler, 1971).

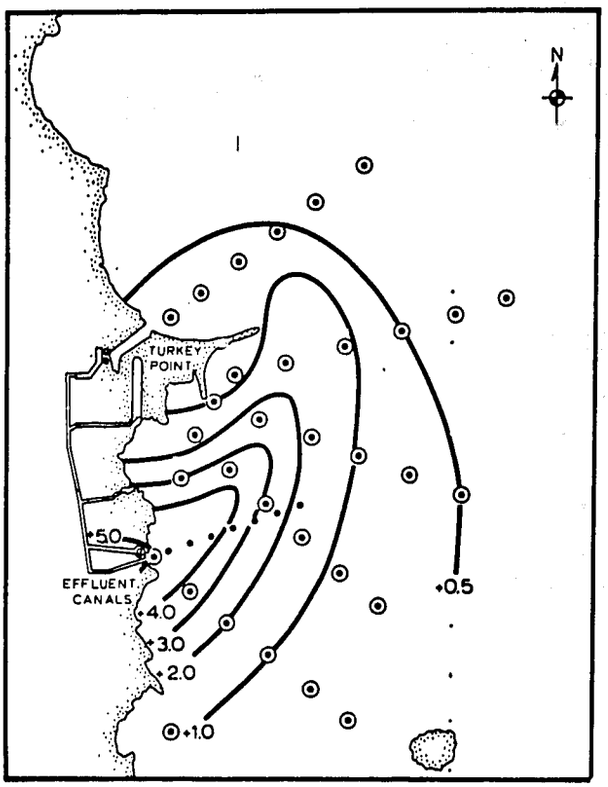

Fig. 6. Average temperature increase (ΔT) off Turkey Point.

The marine algae

The effects of pollution on primary producers especially is important, as they provide with food and shelter for the higher trophic levels. If they are severely damaged or destroyed, the entire trophic scheme collapses. In addition, the benthic plants are valuable indicator organisms, as they cannot avoid the stresses.

In the vicinity of Turkey Point, the distribution of algae (Roessler and Zieman, 1969; Zieman, 1970) was closely related to the distribution of the heated water (Fig. 6). Patterns of algal exclusion were closely correlated with the temperature elevation. Fig. 7A shows the distribution of algal species and Fig. 7B shows the pattern of the species diversity. In general, both algal species numbers and diversities increased until temperature reached about 30—31°C, and declined rapidly at higher temperatures. Blue-green algal distribution (Fig. 8) contrasted markedly with the above patterns. The areas where the natural communities were severely damaged by heat were replaced by blue-green algal mats consisting primarily of *Lyngbya majuscula* and *Schizothrix calcicola* (Hine, in DeSylva and Hine, 1972). Thus, the earlier findings of Cairns (1956) and Patrick (1969), that as temperatures increase

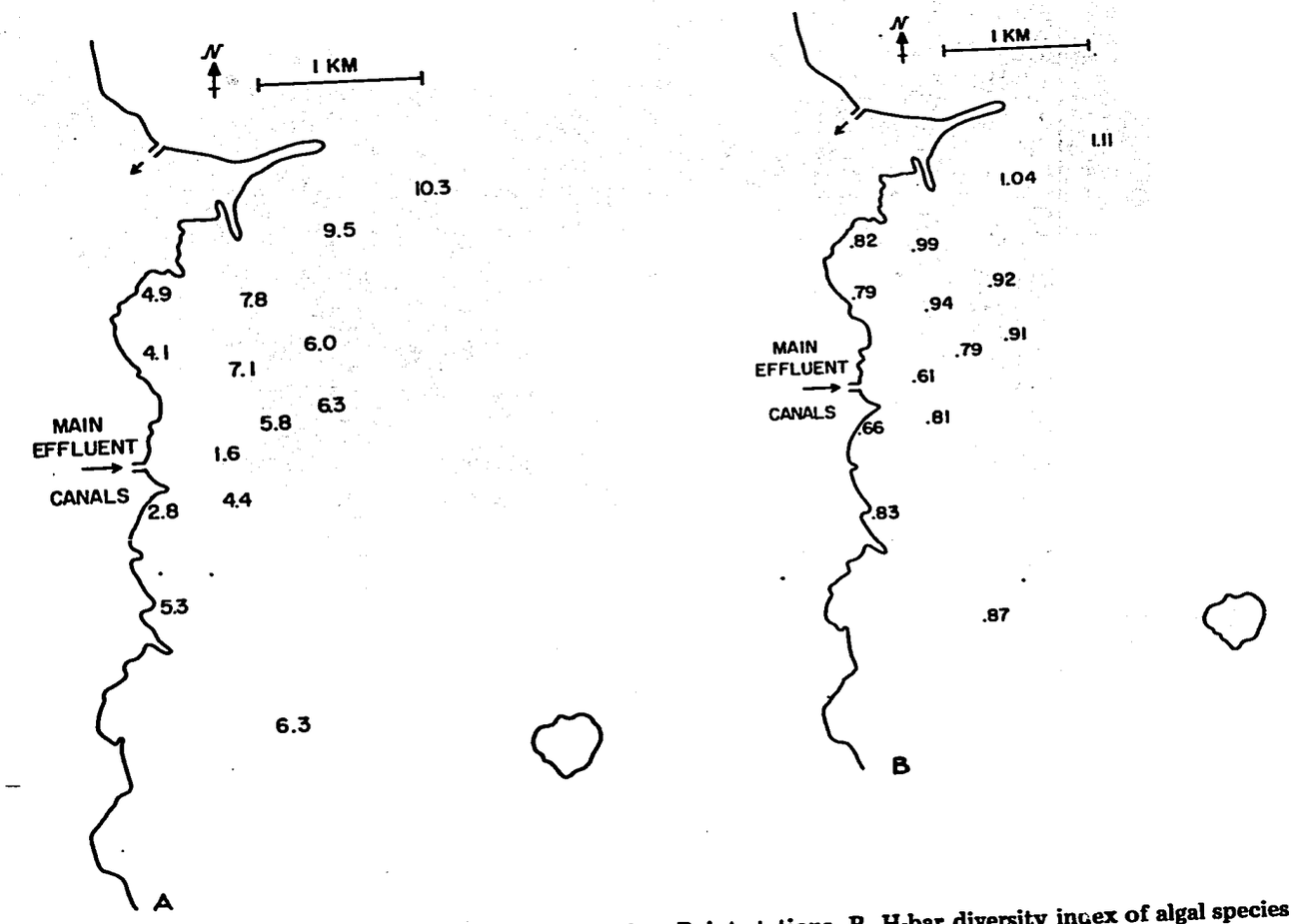

Fig. 7. A. Average numbers of algal species at Turkey Point stations. B. H-bar diversity index of algal species at Turkey Point stations.

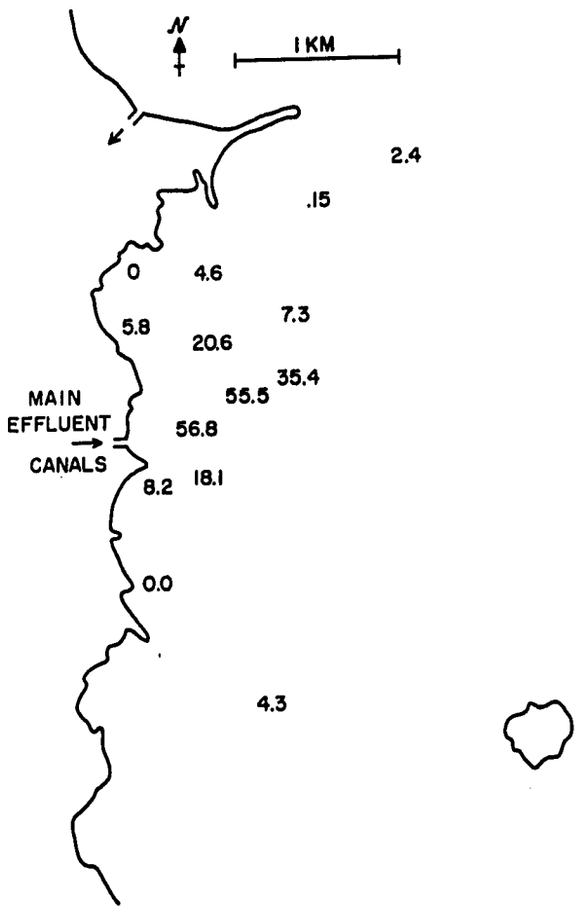

Fig. 8. Average yearly blue-green algal mat coverage off Turkey Point (% coverage).

normal algal communities are replaced by blue-greens, hold for marine as well as freshwater systems. Jones and Randall (1973) and Kolehmainen et al. (1973) also found blue-green algae replaced other species at elevated temperatures in Guam and Puerto Rico.

The shifts in distribution of four species of algae (*Penicillus capitatus*, *Halimeda incrassata*, *Digenia simplex*, and *Laurencia* spp.) are shown in Fig. 9. These clearly show the retreat of the algae due to heated water in the summer and incomplete recovery in the winter. Note the similarity of the pattern of retreat in the summer with the pattern of the raised temperatures (ΔT) from the effluent (Fig. 6). This also indicates the relative tolerance of heat by the algae, showing that *Penicillus* has a higher tolerance than the other species.

The relative tolerance of the algae, and the gradient of damage can be shown in another manner. Where Fig. 9 showed the spatial distribution of

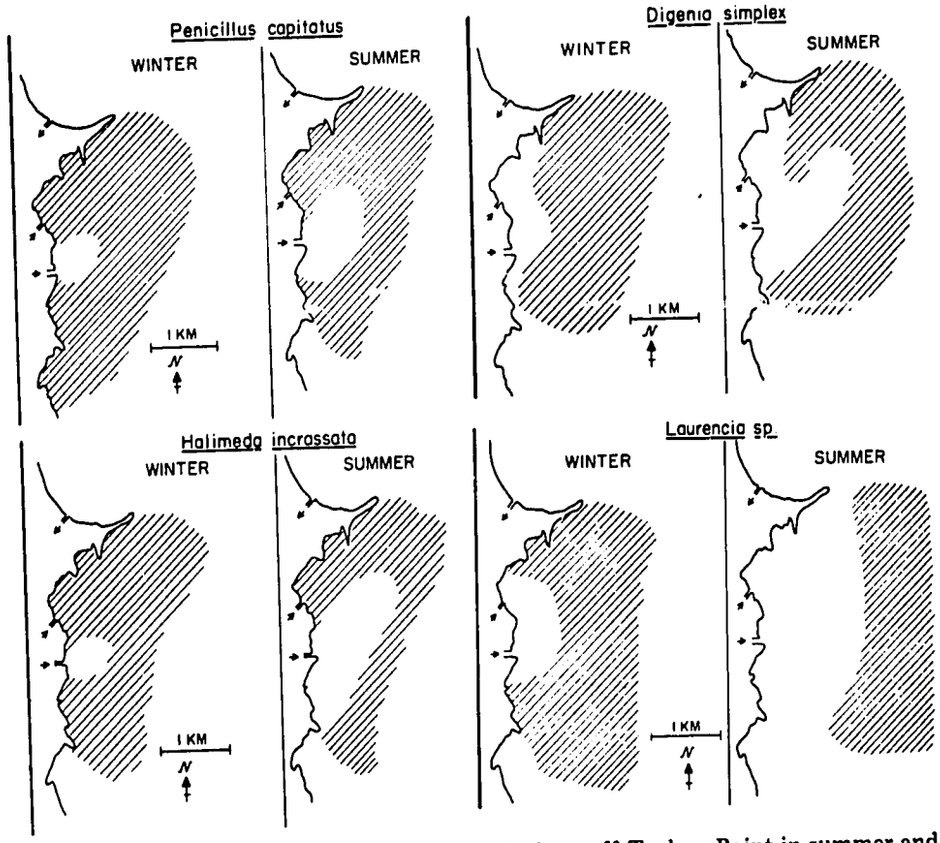

Fig. 9. Spatial distribution of four species of algae off Turkey Point in summer and winter.

certain algal species, Fig. 10 shows the temporal distribution of species at three stations at increasing distances from the mouth of the effluent canal. These represent the species present within a permanent 1-m^2 quadrat which was monitored every 2—3 weeks. At the station closest to the effluent (Station 26 — 600 m from effluent), all species of algae were killed during the summer of 1969, and only half recovered the following winter and half of these were killed the following summer. At Station 24, 900 m from the effluent along the axis of the plume, the distributional lines are more continuous, with several species persisting throughout the summer and others recovering the following winter. Station 42 is 2 km from the effluent canal and shows even less seasonality. Here the number of species which are present continually is considerably higher than at the closer stations.

From these studies it is possible to arrange the dominant algal species in order of thermal tolerance. *Acetabularia crenulata* is the most resistant algae and survived average temperatures 3.5—4° C above ambient. *Bataphora oerstedi* was nearly as resistant, surviving +3° C temperature increases, and *Peni-*

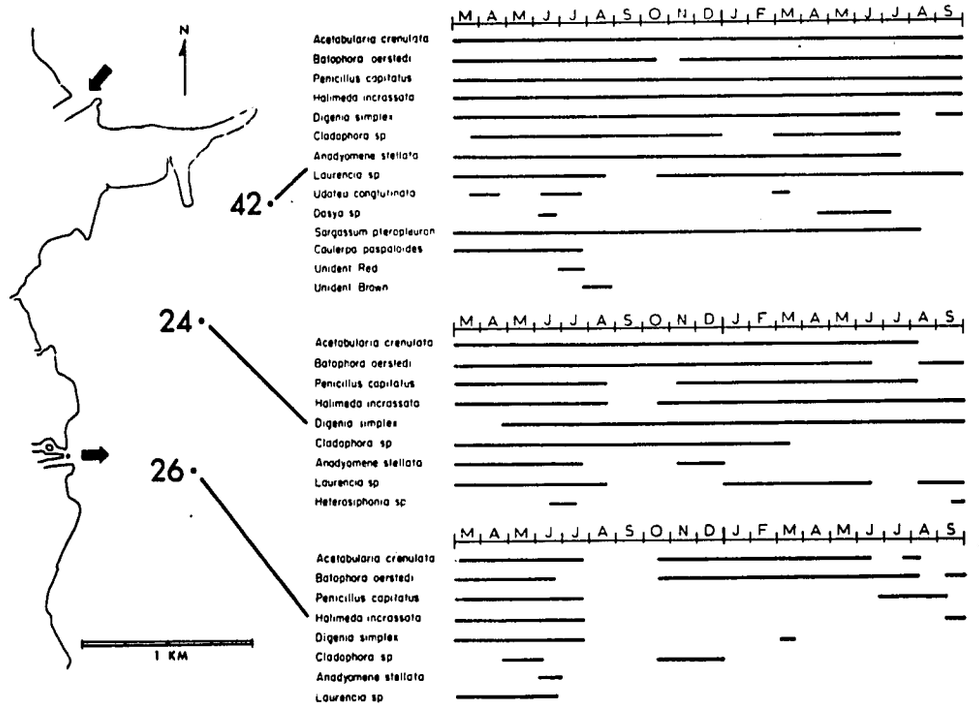

Fig. 10. Temporal distribution of algae at three stations off Turkey Point.

cillus capitatus disappeared with temperature increases of 2.5—3°C. *Halimeda incrassata*, *Digenia simplex*, and *Udotea conglutinata* disappeared at temperatures of +2°C, and *Anadyomene stellata* tolerated no more than 1°C temperature increases. *Laurencia* sp. and *Caulerpa* sp. were not resistant to temperature rises of even 1°C.

Sea grasses

Thalassia beds within the Turkey Point area tended to fluctuate between low values of 2,500 blades m^{-2} in the winter (January—March), and a high of 4,000 blades m^{-2} from May through October, after which they declined to their winter values. This corresponds approximately to average standing crops of leaves of 91 to 146 g dry weight per m^2 respectively. Leaf numbers increased until a temperature of about 30—31°C was reached, at which point they remained stable. Above these temperatures the density of the leaves declined. Net production by the leaves is also temperature-dependent (Fig. 4). *Thalassia* has a distinct temperature optimum in the immediate vicinity of 30°C.

The ratio of live/dead leaves on 10 September 1968 was 75%/25% at stations remote from the power plant, and decreased to approximately 50%/50% about 700—900 m from the mouth of the effluent canal, and to

9%/91% at a station 450 m from the canal mouth (Fig. 4). The initial decline of the sea grasses at this station was carefully documented. On 14 August 1968 there were approximately 2,300 healthy blades m^{-2}, by 4 September this was down to 400 blades m^{-2}, and by 24 September there were none. Studies of the rhizomes from this area showed that the plants had used all of their stored starch reserves. If stresses are alleviated before the starch reserves are exhausted, the plant can recover.

The sea grasses, which require high light levels, were further stressed in this area by the greatly increased turbidity of the effluent. This increased turbidity was due both to erosion at the canal mouth and to the washing out of material from the mangrove swamps in the areas adjacent to the effluent system on ebb tide. Because of the semipermanence of the robust rhizome system, which is 5—25 cm below the sediment surface, *Thalassia* is less subject to short-term fluctuations than other sea grasses or macro-algae, whose rhizomes and holdfasts are restricted to the upper few centimeters of the sediment. However, once the *Thalassia* is killed in an area, the recovery time can be years to decades.

Epiphytes

Early studies of epiphytes were made in the vicinity of Turkey Point utilizing wooden dowels placed in the main axis of the effluent at 100 m intervals. There was an increase in the thickness of the diatom mat outward from the canal mouth in summer, and a reverse in winter where the innermost sticks had the highest epiphytism. Wood found more then 240 species from 37 genera of diatoms alone.

Following these early attempts, studies by Wood (unpublished) and Sprogis (1971) showed a definite reduction in number of species colonizing glass slides with increasing temperature and closeness to the effluent (Fig. 11). They found the dominants were *Frustula saxonica* var. *rhomboides* and *Cocconeis placentula* with occasional high numbers of species of *Mastogloia* and *Synedra*. Sprogis (1971) found that the highest chlorophyll *a* values occurred within the effluent plume, but at the stations closest to the effluent canal, the chlorophyll dropped to extremely low values, possibly indicating that "the organisms are thriving at a temperature close to their critical level, and that any change of another parameter could drastically alter productivity".

Animals

Faunal responses to elevated temperature at Turkey Point were similar to floral patterns and showed similar patterns of temperature resistance, primarily in the sessile and less motile forms. There was a noticeable reduction in motile forms, such as fishes, due to avoidance of the high temperatures and the reduction of plant cover and food sources.

Nugent (1970) studied the fish population within the effluent canals and

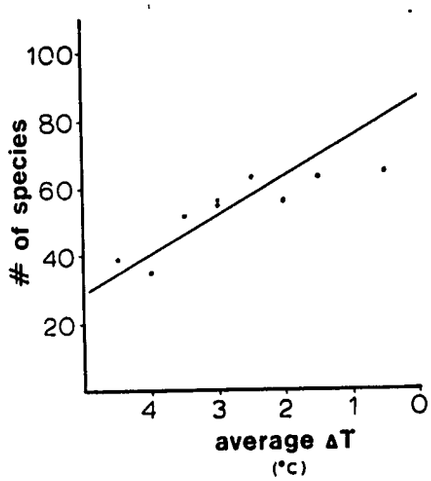

Fig. 11. Abundance of diatoms species vs. average station temperature off Turkey Point.

found that the grey snapper (*Lutjanus griseus*) and tarpon showed the least response to temperature increases. Fantail mullet (*Querimanaa trichodon*), white mullet (*Mugil curema*), and snook (*Centropomus undecimalis*) were not affected by the effluent during the winter, but decreased in abundance in the summer. Pinfish (*Lagodon rhomboides*) were abundant in the winter but decreased sharply in the summer when temperatures reached 40°C. In the intertidal zones, the settlement of the barnacle *Balanus eburneus* increased in the winter but was strongly inhibited in June and July.

Purkerson (1973; in U.S. Dep. of Int., 1970; Hagan and Purkerson, 1970) studied the effect of the heated effluent on the biota of the Turkey Point area, and recorded the effects of several particularly acute thermal stresses. He recorded the decline of the biota in 1968 and reported nearly complete destruction over an area of nearly 40 ha. Partial recovery was noted in the winter months.

Siderastrea sp. corals had expired as far as 540 m from the canals and some were dead at 725 m, although "small fish, shrimp, crabs, amphipods and brittle stars were noted as close as 185 m to the outfall. On 25 June temperatures reached 39.5°C 300 m off the canal and 36°C 1000 m off the canal and very turbid waters were found. Dead plants and animals were found 900 m east, 900 m north, and 450 m south of the canal. Dead animals included sponges (*Ircinia* and *Spheciosphongia*), mollusks (*Chione cancellata*, *Trachycardium egmontanum*, *Fasciolaria hunteria*, *Fasciolaria tulipa*, *Aquipecten givvus*, *Vermicularia spirata*, *Acmaea* sp., and others), blue crabs (*Callinectes sapidus*), stone crabs (*Menippe mercenaria*), other crustaceans, and "thousands of dead fish". This severe mortality was determined to cover at least 228 ha, and was also the cause of considerable sudden mortality of macro-algae (Zieman, 1970).

Roessler and Zieman (1969) found that temperatures of 4°C above ambient killed nearly all fauna normally present. In the area between the +4°C and +3°C isotherms there was a decrease in fish but some increase in mollusks and crustacea. This was found to be a transition zone with a temporary increase in detritus feeders due to the large amount of dead and decaying plant material.

Roessler (1971) later added that the optimum temperature for species diversity of animals and for maximum numbers of individuals was between 26° and 28°C. He found that the 50% exclusion temperature of invertebrates and fish caught by trawling was between 30° and 34°C, and the 75% exclusion temperature was between 35° and 37°C. Roessler found that the decline in abundance covered an area of about 120 ha, although he concluded that in about 50 ha the increased winter catches compensated for the summer decline.

CONCLUSIONS

The work described here has shown that thermal pollution is a real and immediate problem in tropical waters. Roessler and Zieman (1969) showed that heated water pumped into Biscayne Bay has reduced the diversity and abundance of algae and animals in a small area adjacent to the mouth of the effluent canals of Turkey Point. The region where virtually all plants and animals were killed or greatly reduced in numbers corresponded closely with the +4° isotherm and included an area of about 50 ha. A second zone in which algae were damaged and species numbers and diversity reduced corresponded generally to the +3°C isotherm and covered an area of about 68 additional ha. This was the extent of damage in the fall of 1969. Subsequent studies (Roessler et al., 1970; Zieman, 1970; Bader et al., 1972; Roessler, 1971) have shown it to be a just sub-equilibrium condition.

The figures for damage given by Roessler and Zieman (1969) and Zieman (1970) describe mean damage associated with averaged temperatures above ambient. The figures of Purkerson (1973) are maximum damage values associated with extreme short-term thermal maxima. Both are important. The average conditions poise the environment in the sense that they determine the potential components of which the community is composed. However, ultimately the realized community, that which is measured, is dictated by the short-term maxima and minima.

The damage described above was the result of the two conventional power generators at Turkey Point. It was feared that when the nuclear units became operational, the damage would increase dramatically. Added to this was the evident problem of recirculation of the heated effluent. To alleviate these situations, a 9-km canal was dug to Card Sound to the south; however, environmentalists and officials feared that this would simply shift the prob-

lem to a more distant location. As a result of this, construction was begun on a closed cooling system consisting of 270 km of 60 m wide cooling canals, covering some 2,800 ha of former mangrove and marsh area. Thus, it appears the bay has won a reprieve at the sacrifice of the mangroves and marshes.

Despite the fact that ambient temperatures are higher in tropical estuaries as opposed to temperate ones, the upper thermal lethal point of tropical estuarine organisms is not raised proportionally, so that there are often considerable mortalities due to high ambient temperatures caused by natural solar heating. This is coupled with high humidity, which greatly depresses evaporative cooling, and the fact that many tropical estuaries are shallow lagoonal environments which lose excess heat slowly.

Diverse studies in Florida (Roessler and Zieman, 1969; Zieman, 1970; Roessler, 1971), Guam (Jones and Randall, 1973), and Hawaii (Jokiel et al., quoted in Jones and Randall, 1973; Hohman and Tsuda, 1973) show temperatures of 2°C above ambient causing sub-lethal effects and some mortalities, and 4—6°C rises causing severe damage or total destruction of natural communities. These results indicate temperatures in excess of 30—32°C are likely to cause mortality in tropical environments.

They indicate that great care must be used when locating facilities which require extremely large amounts of cooling water, such as electric power plants, on tropical estuaries and lagoons.

In the future careful studies must be made when weighing alternatives of economic and engineering feasibility and potential environmental damage. In particular, more than perfunctory acknowledgement of alternate sites and plant configurations must be made, and they must be available for public scrutiny. In the past, prime consideration was given to the quiet acquisition of desirable land to avoid losses to speculators. Today other considerations must take precedence. In addition plant costs have risen so drastically — power plants often exceed $1 billion now — that land costs are no longer the concern they once were. To avoid repeats of the problems, delays, accusations, and recriminations of Turkey Point, and other power plants, such as the Pt. Aguirre power plant on the south coast of Puerto Rico, considerations must be made well in advance of *all* inputs: economic, engineering, ecological, and social.

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CHAPTER 2

POLLUTION AND DEGRADATION OF CORAL REEF COMMUNITIES

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INTRODUCTION

Long lines of offshore breakers, blue lagoons and white sand beaches are often spectacularly visible in the coastal tropics, but the coral reef communities which create them are seldom seen except by the skin diver. To be told that many of these communities are threatened by the pressures of civilization, and that some have already succumbed, is thus not liable to elicit the same concern as when one hears, for example, about the destruction of redwood forests. But there are good reasons for being concerned, reasons that make the fate of the redwoods, despite their beauty and value, seem minor by comparison.

Coral reefs are probably the most extensive shallow marine communities on earth (Goreau, 1961) and among the most biologically productive of all natural communities, marine or terrestrial (Odum, 1971). In the tropics, where man's terrestrial protein sources are often inadequate, reef fishes and shellfishes provide high-quality protein for people living near the sea. Reef communities also abound with species containing a wide range of pharmacologically active compounds (e.g., Baslow, 1969; Arehart, 1971; Hashimoto and Fusetani, 1972; Burkholder, 1974). Reef coelenterates alone afford "a mine of novel compounds of potential value as drugs or as tools for pharmacological research" (Ciereszko and Kams, 1974).

Fringing reefs are self-repairing breakwaters which permit the continued existence of about 400 atolls and numerous other low tropical islands, as well as preserving thousands of miles of continental coastlines. Munk and Sargent (1948) estimated that normal waves dissipate 500,000 horsepower against the windward side of Bikini Atoll. Ladd (1961) states, "the growth forms of the (reef) builders. . . enable them to construct an efficient baffle that, in addition to bringing a constant supply of refreshing sea water to all

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parts of the organisms, robs the incoming waves of much of their force by spreading the water in all directions... These structures are truly worthy of an engineer's respect".

Chave et al. (1972) estimate that the gross production of calcium carbonate by reef communities averages between 100 and 500 tonnes per acre per year. This production goes into the building and maintenance of both the reef and nearby beaches. The recreational values of coral reefs are well known, and their use in this connection is growing rapidly. The uniquely peaceful and beautiful vistas that reefs present to the human visitor is a psychological resource to which any coral reef diver will bear witness.

Corals occupy only a portion (often a minor portion) of reef communities. Large areas of reefs are often covered mainly by a thin layer of algae. This algal zone lacks the spectacular beauty of the coral community and has received comparatively little study. The layman may assume that these communities deserve a low conservation priority. They are, however, an important component of the reef community complex for several reasons. First, they are sometimes even more biologically productive than adjacent coral communities (Smith and Marsh, 1973), and are a source of food for fishes of the coral community who leave its shelter periodically to graze (e.g. Chartock, 1971). In addition, large quantities of algal fragments are removed from the algal community by currents and transported to adjacent coral and other communities where they serve as detrital food (Johannes and Gerber, 1974). Finally, some of these algal communities have been found to be capable of extremely high rates of nitrogen fixation (Webb et al., 1975). This newly fixed nitrogen is distributed through reef food chains via grazers and consumers of algal detritus. This mechanism appears to be important in maintaining the high productivity of reef communities in waters characteristically very low in fixed nitrogen.

To anticipate the impact of man on the reef community, we must have some knowledge of environmental tolerances of the component organisms. Such studies are few, with the exception of a number of investigations on corals themselves. These animals, which dominate the reef visually, do not appear to account for the major fraction of reef community biomass, or energy metabolism (Odum and Odum, 1955). Their role in the reef community is pivotal nonetheless. As White (1969) has said: "All forms of life modify their context. The most spectacular and benign instance is doubtless the coral polyp. By serving its own ends it has created an undersea world favorable to thousands of other kinds of animals."

So central are corals to the integrity of the reef community that when they are selectively killed, migration or death of much of the other reef fauna ensues (e.g., Barnes, 1966, p. 261; Chesher, 1969, 1970). Therefore, the environmental tolerances of the reef community as a whole cannot exceed those of its corals. Accordingly, our knowledge of the environmental limits of corals can provide us with convenient preliminary criteria for set-

ting standards of environmental quality for reef communities. In instances where other vital components of the reef community prove to have significantly narrower stress tolerances, such regulations would, of course, prove inadequate.

SEDIMENTATION

The problem

A number of workers, who will be cited below, have documented the destruction of coral communities by sedimentation. However, others have noted the existence of apparently healthy coral communities in the presence of generally turbid waters (Crossland, 1907; Marshall and Orr, 1931; Verwey, 1931a; Motoda, 1939; Roy and Smith, 1971). Consequently, some have concluded that sediments has been exaggerated as a threat to corals (e.g., Yonge, 1940; Maragos, 1972).

Recent work by Hubbard and Pocock (1972) helps reconcile these conflicting impressions. Working with 6 differentially stained size classes of sediments and 26 species of Caribbean reef corals, they showed that corals are "size-specific sediment rejectors" and that this specificity differs markedly in different species. These characteristics were previously known (e.g., Mayor 1918a; Edmondson, 1928; Marshall and Orr, 1931; Abe, 1937) but had only been demonstrated in a rough qualitative way, and the implications had not been pursued. Silt, the smallest sediment size class used by Hubbard and Pocock, was the only size removed effectively by all species tested. Unlike most larger particles, silt is light enough to be removed by ciliary action.

Differences in sediment rejection were shown to be a function of variation in the polyps' size and distentional capacity and the geometry of the calyx (see also Edmondson, 1928; Marshall and Orr, 1931; Hubbard, 1973). Thus the threat posed by sediment varies with particle size and species of coral. It is worth noting that in this single paper by Hubbard and Pocock, palaeontologists working on problems of interpretation of fossil coral distribution, have provided us with more information concerning sediment rejection by modern corals than all the marine biologists combined who have examined the problem in the past.

The existence of healthy coral communities in chronically turbid waters can also be explained in part if strong currents are present which help prevent the build-up of sediments on coral surfaces (Marshall and Orr, 1931; Verwey, 1931b). Most corals cannot live for long if heavily coated or buried (Wood-Jones, 1907; Mayor, 1918a; Edmondson, 1928; Umbgrove, 1930; Marshall and Orr, 1931; Motoda, 1940). Where corals do survive in turbid waters their growth rates and the depth to which they grow will be reduced

since calcification in corals is light-dependent (e.g. Goreau and Goreau, 1959).

Dodge et al. (1974) found an inverse trend between rates of resuspension of sediments and coral growth in Jamaica. They also noted that high sediment resuspension was related to decreased variability in growth rates. In a detailed field study in Kaneohe Bay, Hawaii, Maragos (1972) found that reduced light intensity due to sewage-induced phytoplankton blooms and suspended silt markedly reduced growth rates of *Pocillopora damicornis* and *P. meandrina*. Of the eight environmental factors Maragos studied, light intensity showed a higher correlation with the combined growth rate of six species of corals than all other factors combined, accounting for 57% of the observed variation. Wells et al. (1973) and J.W. Wells (personal communication, 1974) observed lowered numbers of zooxanthellae and abnormally low photosynthesis/respiration (P/R) ratios in corals living beneath a plume of polluted, highly turbid water leaving Biscayne Bay, Florida. Verwey (1931b) found a clear correlation between the maximum depth of reef growth and water clarity in Batavia (Jakarta) Bay. Thus turbidity, which may not kill corals in shallow water, will reduce their growth rates and bring about their eventual death in deeper water. Coral species diversity has also been observed to decrease with increasing turbidity (Brock et al., 1966; Roy and Smith, 1971). Sedimentation can also be inimical to algae growth in coral communities (Dahl, 1973).

Strong currents carrying sand and other coarse suspended material can inhibit coral growth by abrasion (Bourne, 1888; Crossland, 1907, 1927, 1928; Bertram, 1937; Wiens, 1962; Storr, 1964). Coral planulae cannot settle and survive on an unconsolidated substrate (e.g. Harrigan, 1972). Fine sediments subject to resuspension are inimical to suspension-feeding benthos in general (e.g. Rhoads and Young, 1970) and bring about the destruction of a variety of reef fauna in addition to corals (Bakus, 1968).

The rich and varied biota found in the myriad of caves, tunnels and crevices in the typical reef are undoubtedly more vulnerable to siltation than organisms on the reef surface. Sediments settling out in a reef community tend to fill or plug these cavities. The contribution of this extensive sub-community to the structure and function of the reef community as a whole, while poorly known, is probably large (e.g. Hiatt, 1958).

Near Okha in the muddy Gulf of Kutch, India, high tides produce strong currents which flush sediments from the raised surfaces of flourishing coral reefs. In the areas I examined the typical cavities characteristic of most coral reefs were not apparent; if they existed at all they were filled with sediments. DiSalvo (1969) found that the internal spaces in dead coral heads in Kaneohe Bay contained unusually large quantities of terrigenous sediment in areas influenced by runoff.

The impact of bad land management

Property damage on land is often used as a criterion of whether erosion induced by man's activities has exceeded legally permissible levels. It seems to be generally overlooked that property damage brought about by erosion can and frequently does extend into the sea. Natural erosion is an essential process without which we would have no soil, and the delivery by rivers of nutrient-laden soil to the ocean enhances marine productivity. In excess, however, siltation of erosion products leads to decreased productivity of coral reef communities and, ultimately, to their destruction.

Exposure of reefs to brackish, silt-laden water associated with flood runoff has probably been the single greatest cause of reef destruction historically. Undoubtedly some damage would have occurred in man's absence, but there can be no doubt that bad land management has greatly magnified the problem.

High islands and continental land areas fringed by reefs are frequently areas of high rainfall and lush vegetation, and are thus particularly susceptible to erosion. Kirby (1969) states: "Experiments have shown that the greater the rainfall and the thicker the vegetation (up to a point) the greater will be the acceleration of erosion consequent on stripping the vegetation."

R. Moberly Jr. (1963; and personal communication, 1971) has studied the effects of man's activities on reef sedimentation in Hawaii. Ancient Hawaiians cultivated taro in swampy lowlands, and raised pigs, chickens and dogs either penned in the villages or loose in the forests. After the arrival of Europeans, virtually all native plants below 1,000 m were replaced by invaders., Sheep, goats and cattle were introduced, and upland soils were plowed for sugar cane and pineapple. The impact of the resulting erosion on some reefs has been dramatic. On west Molokai for example, ranching started in 1897 and pineapple-growing in 1918. Maps of the U.S. Geological Survey and old photographs examined by Dr. Moberly show the advance of the shoreline as much as 2 km across the reef flat in this century. The process was still proceeding "rapidly" seaward when observations were made in 1962. In some areas 10—70 cm of red-brown silt now overlie the reef.

The coral reefs in Kaneohe Bay, Hawaii, have been increasingly exposed to sedimentation and low salinities associated with flooding. The bay watershed is scarred by many acres of exposed soil due to land clearing associated with housing and road construction. Laws requiring the replanting of land cleared of vegetation are inadequate and have been poorly enforced. The bay turns red and opaque after every heavy rain. During one rainstorm in 1969 one of the ten streams entering the bay was found to be transporting enough sediment into the bay in 24 hours to cover the entire bay bottom (about 41 km^2) with about 1 kg/m^2 of sediment if spread evenly (calculated from Fan and Burnett, 1969). Surveys of mean bay depth made in 1888 and 1927 yielded similar values. A third survey in 1969 showed that subsequent accel-

erated sedimentation had reduced the mean depth by more than 1.5 m (Roy, 1970).

Fine, clay-sized, terrigenous sediments have killed many corals growing below 6—9 m (Maragos, 1972), particularly in the southern end of the bay, where runoff is high and circulation is slowest (Bathen, 1968), and on reefs fringing the shoreline. Sediments here are trapped between coral heads on the reef slopes. Settling of planulae is thus poor or absent, growth of existing corals is slow, and only a few older corals remain alive in many areas. DiSalvo (1972) attributed pitting and fouling of crab carapaces in inner Kaneohe Bay to siltation. Sedimentation, sewage and dredging have combined to reduce viable reef slopes in the bay to 29% of their pre-1939 coverage (Maragos, 1972).

Fairbridge and Teichert (1948) suggested that because of "colossal soil erosion due to unplanned agriculture", continental sedimentation appeared to be gaining on coral growth in the vicinity of the Low Isles, Great Barrier Reef. On the seaward slopes of some of these reefs they reported that much of the coral was dead and covered with a fine coating of terrigenous sediments. Stephenson et al. (1958) note "the widely held opinion that floods have become increasingly severe in North Queensland owing to land clearance". "If so", they state, "a new element — human interference — has entered the ecological complex, and coral growth may never attain its previous luxuriance" (see also Thompson, 1971).

Van Eepoel and Grigg (1970) report that in large areas of Lindberg Bay, St. Thomas, Virgin Islands, most corals and other sessile animals have been destroyed and conditions remain unsuitable for their establishment due to sedimentation caused by bulldozing, construction and the surfacing of land which drains into the bay. A subsequent survey indicated that conditions were rapidly worsening (van Eepoel et al., 1971). Damage to reef communities due to accelerated terrigenous sedimentation has also been observed in Tanzania by Ray (1968), and in the Seychelles by Vine (1972).

Although sedimentation due to bad land management is very likely the biggest pollution threat to coral reefs, there seems to be very little that marine scientists can do about it in most cases. The terrestrial damage done by accelerated erosion is often even greater than that done to marine communities — yet efforts at soil conservation by terrestrial resource managers have, in the tropics, usually been dismal failures. It is very doubtful that significant progress can be made with this problem while burgeoning human populations create accelerating and irresistible pressures on tropical soils.

Dredging

Fine sediments carried into coastal waters by flooding are often concentrated initially in the upper low-salinity layer (Emery, 1962; personal observations). To some extent this facilitates diffusion and dilution before the

sediments settle. Such is not the case when sediments are suspended by dredging, a related source of reef pollution. Levin (1970) provides a useful review of the pertinent literature. Brock et al. (1966) give a detailed account of the destruction of corals and reduction of fish and echinoderm populations at Johnston Island due to siltation brought about by dredging. Over 2,800 ha of reef and lagoon were more or less affected by silty water and 440 ha of reef totally destroyed through dredging and filling. Deterioration of reef communities continued for at least a year after dredging ceased because of continual resuspension of sediments.

In Water Bay, St. Thomas, Virgin Islands, Grigg and Van Eepoel (1970) observed the destruction of hard corals due to sedimentation associated with the release of clays brought about by the dredge removal of overlying sand. Since no exploratory cores were taken, the presence of the clays was not detected in advance of dredging.

In this and related surveys, personnel at the Caribbean Research Institute have found in the U.S. Virgin Islands a total of "several miles of dead reefs and bay with impoverished fauna and flora as a result of ill-planned and ill-managed dredging operations" (Grigg, 1970; see also Grigg et al., undated). Grigg et al. (undated) report having observed dredge-suspended fines remaining in suspension or continually being resuspended for up to five years in the Virgin Islands. These workers have proposed "a total ban on further inshore dredging excepting for improvement and maintenance of harbors and channels" (Grigg, 1970).

In Castle Harbor, Bermuda, an airfield was built between 1941 and 1943 by dredging and filling. The bay's population of brain corals, *Diploria*, the dominant coral genus on Bermuda reefs, was destroyed (J. Burnette-Herkes, personal communication, 1971). Dead colonies up to 1 m in diameter can still be seen today. The sediments produced by dredging were still being resuspended in large quantities in 1969 and no significant recolonization by *Diploria* had occurred (personal observations).

In 1939, 29% of the coral reefs in Kaneohe Bay, Hawaii, were removed by dredging. Some partially dredged reefs have recovered as a result of the encroachment of corals from adjacent undamaged areas (Maragos, 1972). None of the patch reefs that were dredged completely shows signs of growing back to the surface however (Roy, 1970).

Siltation in Guam, brought about by dredging, destroyed reef flat communities in "much larger areas than were destroyed by the physical removal of substrate" (Marsh and Gordon, 1974).

The injury of destruction of reef communities due to dredging has also been observed in the port of Singapore (Johnson, 1970) and nearby straits (Wycherly, 1969), the Seychelles (Vine, 1972); Hawaii (Sullivan and Gerritsen, 1972; Tepley and Johannes, 1972), and American Samoa (Sverdloff, 1973).

Dredge-suspended sediments exert both biological and chemical oxygen

demands (e.g., Brown and Clark, 1968; Frankenberg and Westerfield, 1968). Since coral reef communities may not be able to tolerate oxygen levels much lower than the normal nighttime minima to which they are exposed (e.g. Kinsey and Domm, 1974; Johannes and Betzer, Chapter 1), the scavenging of oxygen by suspended sediments may constitute a significant stress. No research seems to have been done on this problem as it relates to coral reef communities.

Alteration of reef topography through dredging, filling or underwater construction will alter current velocities and directions. Since the zonation of corals (e.g. Hubbard, 1974) and other reef biota is clearly related to current regimes, current alterations will affect settling and survival patterns.

Ingle (1952) states that dredging should be "under the constant surveillance of persons capable of evaluating the conditions of the operation. Frequent checks by silt collectors placed at various distances from the dredge and simple biological checks would constantly be made of the deposits at various distances from the dredge... In this way it would be possible to acommodate the activities of the dredge to momentary conditions." Using this approach, dredging close to reef communities might be accomplished without causing serious damage — activity could be restricted to periods when the currents carried suspended materials away from the reef and dispersed them adequately.

The gathering of corals or dredging of reef substrate for use as construction material is common in some areas (Phelan, 1952 and references therein; McNeil, 1951; Beaven, 1962; Walker, 1962; Porter and Porter, 1973). In Malaysia coral mining is done by looping a cable around live coral heads of some tonnes and winching them across the reef front and down the reef slope until they are beneath a vessel from which they are winched aboard. On a "good day" 450 tonnes of coral heads per vessel are mined in this fashion. Each coral head thus removed destroys, on the average, an area 3 m wide and 7.6 m long. In a day 0.13—0.2 ha per vessel is destroyed (Lulofs, 1974).

Casciano (1973) describes a new dredging system which under some conditions should provide a more economical and less biologically destructive means of recovering offshore sand than conventional systems. The suction head is buried in the sand rather than held above it, thus reducing the turbidity accompanying its operation.

Sugar mill wastes

Sugar mill wastes are another source of sediment damage on reefs in Hawaii and parts of the Caribbean. In Hawaii harvesting of sugar cane is done using a modified bulldozer blade to break the cane stalks and roll them into bundles. The bundles contain quantities of soil which, along with some cane leaves, are washed from the cane and then often dumped directly into the

ocean. After the cane is cleaned and the juice pressed from it, the waste stalk and cane pulp, which is known as "bagasse", is also often dumped into the sea. Hundreds of tonnes of rocks, silt and bagasse have been dumped daily into Hawaiian coastal waters by sugar mills (Copeland and Wilkes, 1969; Burm and Morris, 1971). The authors noted one sludge bank which extended for 400 m radius from a mill outfall. The water was extremely turbid in the vicinity. Numbers and diversity of corals and fishes were reduced and there was an overgrowth of benthic algae.

Grigg (1972) found a considerable reduction in abundance and diversity of corals and fish in the vicinity of a number of sugar mill outfalls, although the fish were significantly larger than in control areas. Grigg employed an interesting method of measuring fish availability. He hired a commercial diver to spear commercial-sized fish as fast as he could for 30-min periods in polluted and control areas.

Yamaguchi (1950) described the effect on corals of siltation associated with sugar mill wastes on Kauai, Hawaii. Because of turbulence the silt did not settle out in waters less than 12 m deep. Below that depth siltation and the destruction of corals and associated fauna was pronounced.

SEWAGE

Kaneohe Bay

Although numerous reefs throughout the tropics are exposed to sewage, the problems this creates have gained attention only in the last few years. Kaneohe Bay, Hawaii, is the best studied example of the effect of sewage on reef communities. For a comprehensive summary of pollution research here see Smith et al. (1973).

Edmondson (1928) once described this bay as "one of the most favorable localities for the development of shallow water coral. Nearly all the reef-forming genera known in Hawaii waters are represented . . . and many species grow luxuriantly". It is also the most accessible reef complex in the central Pacific and has probably been the most valuable for recreational purposes.

The south end of the bay, which receives secondarily treated domestic sewage, is a shallow basin of about 880 ha in surface area and contains patch and fringing reefs. More than 99% of the corals in this basin are now dead. Living specimens of the formerly dominant reef-building species in this area now die within a few weeks when placed there (Maragos, 1972).

Survival of five out of six reef coral species transplanted to various locations in the bay showed a high negative correlation with sewage-related variables — turbidity and phosphate content of the water. The exact cause of death however was neither phosphate levels nor turbidity. Maragos (1972)

suggests that anaerobic conditions in the sediments leading to the release of hydrogen sulphide may have been responsible. Sorokin (1973) reported an expanding anaerobic layer, increasing sulphide levels, and dropping redox potentials in sediments with increasing proximity to the main sewage outfall in Kaneohe Bay (see also DiSalvo, 1971). The use of sediment characteristics such as these as a rapid means of tracing the extent of the impact of sewage on benthic communities deserves more attention.

Dictyosphaeria cavernosa is a reef alga distributed circumtropically but seldom constituting an important component of reef flora. In Kaneohe Bay, however, *D. cavernosa* has become "the major benthic organism on the lagoon slopes, having surpassed and caused the decline of corals in recent years" (Maragos, 1972). The accelerating growth of this alga in the bay has paralleled the increasing quantities of nitrogen and phosphorus entering the bay in sewage (Banner and Bailey, 1970). Between 1966 and 1970 dissolved phosphate levels near the main sewer outfall increased at an average rate of 0.75 μg-atom/l (Caperon et al., 1971).

D. cavernosa or "green bubble alga", first colonizes the bases and crevices of coral heads, then extends upward engulfing entire colonies in the bay. Finally, it spreads in thick, heavy sheets fusing with other colonies to blanket large areas of reef. Shut off from light, oxygen and food the enveloped corals and many of the plants and animals associated with them die. Wherever this alga comes into physical contact with corals, the skeletons erode by some unknown process (Banner and Bailey, 1970), and the reef surface begins to disintegrate. Reef fish populations are often strikingly reduced in such areas (personal observations).

Sewage nutrients have stimulated extremely dense growths of *D. cavernosa* in the middle and parts of the northern bay, some distance from the sewage outfalls, but not in the south basin nearer the outfalls. This may also be due to the release of hydrogen sulphide from sediments near the outfalls.

Sewage and oxygen levels

The connection between organic or nutrient enrichment and lowered, stressful oxygen levels is too well known to need elucidation. Johannes and Betzer (Chapter 1) have pointed out that tropical marine organisms live closer, on the average, to their lower oxygen limit than biota in colder waters. D.W. Kinsey (1973; and personal communication, 1974) observed that oxygen levels artificially depressed only slightly below normal levels on a healthy reef caused the death of many reef inhabitants. Depressed oxygen levels in reef and near-reef environments subjected to sewage effluents have been reported by Bathen (1968) and Wade et al. (1972).

Unfortunately oxygen has been measured only during the day in many pollution studies. As Vaughan and Wells (1943) pointed out, even under natural conditions in reef communities "the quantity of oxygen in the water

over reef communities varies greatly between day and night". In enriched waters, rapid photosynthesis may produce high daytime oxygen levels in the same community where high respiration rates produce abnormally low night-time levels. Thus, measurements limited to daylight hours can lead to false confidence in the state of the system.

Bacterial water quality

The bacterial standard for marine bathing waters suggested by the World Health Organization is less than 350 fecal coliforms or 1,000 total coliforms per 100 ml. Siren and Scheuring (1970) found fecal coliform bacteria in numbers averaging 11 million per 100 ml in inshore reef and lagoon waters near sewage outfalls in all six district centers in the Trust Territory of the Pacific Islands. According to Garner (1972), "certain lagoons *were* posted off limits after the report (Siren and Scheuring, 1970) came out — but the signs didn't last long and pollution continues". In one of these district centers, Koror, Palau, Hardy and Hardy (1972) found fecal coliform levels reached 34 million per 100 ml near a hospital outfall.

Barnes (1973) found coliform levels "grossly exceeding" World Health Organization standards in the vicinity of a number of hotel sewage outfalls in Jamaica. Total coliform counts consistently higher than 240,000 per 100 ml were reported in Kingston Harbor, Jamaica, in 1967 (Wade et al., 1972). Dong et al. (1972) reported high fecal coliform levels in Christiansted Harbor, St. Croix. Moreau et al. (1971) reported comparatively low levels of fecal coliforms in Tahiti lagoon.

Human sewage is often combined with sugar cane wastes. This practice results in high concentrations of sewage bacteria which thrive in the sugar-rich waste, according to Copeland and Wilkes (1969). They report fecal coliform levels of 10,000 per 100 ml off a sugar mill outfall in Hawaii.

Sewage treatment

Little research has been done on sewage treatment and disposal as they relate to tropical coastal areas. Laevastu et al. (1964) concluded that in the Hawaiian Islands "there may be a very few sites at which an offshore set of the current is so predominant that untreated sewage may safely be introduced through outfalls of normal length. In general, however, alongshore and seaward sets occur, especially in the surface layers due to wind action, with sufficient frequency as to make inadvisable large discharges of untreated sewage within a few miles in any direction from a shoreline used intensively for recreation or fishing."

Aquatic ecosystems can tolerate some nutrient enrichment without serious ecological dislocations (e.g. McAllister et al., 1972). Experimental enrichment of a coral reef community with low levels of inorganic phospho-

rus and nitrogen increased primary productivity without affecting trophic structure or biomass noticeably (Kinsey and Domm, 1974). The release, in moderate quantities, of sewage effluent which has received adequate secondary treatment could conceivably enhance reef productivity without altering community structure unacceptably. Acceptable levels of enrichment cannot be defined, however, without much more research.

Biological disc filtration units were tested successfully in Thailand by Pescod and Nair (1972). High tropical temperatures favored anaerobic conditions in the units. The system has the ability to absorb shock loading and nutrient-deficient wastes. It requires comparatively little land and capital and is simple to operate.

Shatrosky et al. (1972) and Grigg et al. (1971) discuss operating efficiencies of sewage treatment plants in the Virgin Islands. Rum distillery wastes present special problems for waste treatment because of high BOD, COD, low pH, high dissolved solids and nutrient imbalance, and are the cause of "severe coastal pollution" in the Caribbean (Burnett, in Shatrosky et al., 1972).

Chuck and Lum (1971) suggest that disposal of sewage effluent in coastal limestone aquifers on tropical islands may sometimes be a practical alternative to marine disposal.

Chan (1973) has described how copra farming in the Pacific islands has led to sewage pollution of reef lagoons. To leave the land free for coconut plantations, people who once lived scattered about now concentrate in villages. Whereas pit latrines once took care of human excreta, sewage must now be exported and is generally dumped into the lagoons. This appears to have led to serious decreases in fish and shellfish production as well as the removal of already scarce plant nutrients from the terrestrial ecosystem.

Chan (1973) and Marshall (1969) describe simple, inexpensive sewage treatment units for use on tropical islands. In each case fermentation produces methane which can be bled off and used for cooking or light. When fermentation is complete the residue is sanitary and useable as fertilizer. The effluent, Chan suggests, may be used in specially constructed ponds for aquaculture.

Recently, a number of proposals have been made for raising various marine animals commercially in atoll lagoons using the entire lagoon (e.g., Pinchot, 1966, 1970; Inoue, 1968; Isaacs and Schmitt, 1969). This can be expected to produce increases in lagoon nutrient levels either deliberately through fertilization or involuntarily through the increased production of animal excretions containing nutrients derived from food brought from elsewhere. The fate of Kaneohe Bay and the problems in Pacific lagoons noted above should be taken into consideration in this context.

Some of these plans require the closing of lagoon passes to prevent the escape of the animals being raised. Because most corals require fresh, well-circulated waters, this may have serious consequences. Restricting lagoon

circulation in this manner at Palmyra Atoll completely destroyed its lagoon reef communities (Dawson, 1959). At Kolonia, Ponape, Caroline Islands, a causeway built to an airfield in 1970 has blocked the normal flow of water in the bay and siltation is destroying the reef communities (L.G. Eldredge, personal communication, 1973).

In predicting the net value of lagoon aquaculture the possible loss of already productive reef communities must be considered among the costs. In some cases this cost may be acceptable, but this should not be assumed in the absence of careful evaluation.

Constructing special enclosures of limited size in tropical inshore water to utilize the nutrients in sewage and industrial effluents (e.g., Gundersen and Bienfang, 1972; Chan, 1973) should produce less ecological dislocation and allow simultaneously for greater control over the species being farmed.

Chlorination

The effect of free residual chlorine on marine organisms is poorly known. It is a potent fertilization inhibitor in marine invertebrates at concentrations as low as 0.05 mg/l available chlorine (Muchmore and Epel, 1973), and lethal to marine fish at levels as low as 0.024 mg/l (Alderson, 1972).

Sewage treatment workers are often inadequately trained and sometimes use chlorine well in excess of prescribed levels, particularly when sludge is discharged, during floods when sewage may bypass normal treatment, and during "shock chlorination", the periodic dosing of industrial effluents with chlorine to prevent biological fouling.

Marsh and Gordon (1973) noted a chlorine level of more than 5.0 mg/l in the water adjacent to a power plant outfall in Guam on one occasion. Observed fish kills in the vicinity appeared to be caused by such sporadic high chlorine levels. In the outfall plume of another Guam power plant, Jones and Randall (1973) noted that when the plant ran out of chlorine nine species of reef fishes began colonizing the reef flat and stilling well. When chlorination recommenced all nine species disappeared immediately.

Davis (1971) found that the planulae of three species of Hawaiian corals were more tolerant to chlorine than most marine forms previously tested. Exposure of all three species to 0.49 mg/l for up to 7 hours did not prove lethal. Davis points out that adult corals cannot be assumed to be as tolerant as planulae.

Eutrophication-loving organisms

Despite its ubiquity in coral reef communities, *Dictyosphaeria cavernosa* has not "taken over" in any area other than Kaneohe Bay, Hawaii (see above) to my knowledge. However, the increased dominance of a number of other algal species in response to sewage effluent has been observed.

Soegiarto (1972) noted the association of sewage pollution and heavy growth of *Acanthophora spicifera*, an alien species which arrived in Hawaii in about 1951 (Doty, 1961). (Soegiarto's work constitutes probably the first quantitative measurement made of the effect of water quality on standing crop and species composition of benthic algae in a tropical coastal area.)

Enteromorpha is a genus which thrives in tropical marine areas polluted by sewage (Dong et al., 1972; Tewari, 1972; personal observations), as is the genus *Ulva* (Doty, 1973a; personal observations). Littler (1973) cites evidence that sewage stimulates the growth of various coralline algae at the expense of coral growth.

The tubiculous polychaetes *Spiochaetopterus oculatus* and *Capitella* sp. were observed to be indicators of high organic pollution by Wade et al. (1972) in Kingston Harbor, Jamaica, and by McNulty (1970) in Biscayne Bay, Florida. Sponges appear to increase in response to particulate organic enrichment in the tropics. Dong et al. (1972) noted that the sponge *Chondrilla nucula* increased in abundance, with a concomitant reduction in the coral *Acropora palmata*, with increasing proximity to a sewage outfall in Christiansted Harbor, St. Croix. Burm and Morris (1971) found an increase in sponge density in the vicinity of a sugar mill waste outfall in Hawaii. In Kaneohe Bay, Hawaii, a variety of species of sponges have increased greatly in abundance on the reef slopes in the vicinity of sewage outfalls in recent years (personal observations).

The holothurian *Ophiodesoma spectabilis*, although common in Kaneohe Bay before the introduction of sewage (MacCaughey, 1918), increased considerably in numbers and range between 1959 and 1973 in areas affected by sewage (personal observations).

Sewage outfalls often attract fish because of enhanced plant growth and sewage particulates which serve as food. As a consequence sewage outfalls sometimes become favored fishing grounds. Some fishermen consume or sell the fish they catch in such areas. I know of no studies of the health hazards involved as they relate to shallow tropical marine waters. Hardy and Hardy (1972) reported high coliform concentrations in oysters and crabs near a seawage outfall in Palau.

OIL POLLUTION*

The problem

Moore et al. (1973) provide a useful summary of the composition and

* An exhaustive and important document dealing with oil pollution and coral reefs, which became available after this book went to press, is: *Royal Commissions Into Exploratory and Production Drilling for Petroleum in the Area of the Great Barrier Reef.* Volumes I and II. Australian Government Publishing Service, Canberra, A.C.T., 1050 pp.

structure of hydrocarbons, their solubility in seawater, their volatility and their susceptibility to biodegradation. Baker (1971) relates structure to toxicity. Generally the low-boiling aromatics in petroleum are most immediately toxic (Blumer, 1971). They are also the most volatile and so are among the first components of crude oil (by far the most abundant type of oil spilled) to evaporate. Floating oil thus rapidly decreases in toxicity.

Not all oil remains afloat however. Some oil compounds are soluble — among these are some of the most toxic fractions (e.g. Mironov, 1972). In addition, some heavier oil residues sink readily, while lighter oils may quickly absorb to suspended sediment particles (Hawkes, 1961) and enter the food chain via suspension- and deposit-feeders. Petroleum degrades very slowly when trapped in marine sediments (Blumer and Sass, 1972).

Blumer et al. (1973) describe the weathering history of a light paraffinic crude oil that stranded on a Bermuda beach. After thirteen months the spill residue was still "far from being inert asphalts", and was not depleted of its biologically most active higher molecular weight compounds. During World War II a tanker filled with marine fuel was wrecked on a reef near Japtan Island, Enewetak. Today (1974) many porous rocks and boulders near the remains of the ship are still heavily impregnated with tarry residues from the spill (personal observations).

Oil and corals

To date there appears to be no conclusive evidence that oil floating *above* reef corals damages them. Grant (1970) floated Moonie crude oil in a vessel over specimens of the coral *Favia speciosa*. The corals showed no visible signs of injury during six days of subsequent observation. Johannes, Coles and Maragos (unpublished) floated five different types of oil over groups of the Hawaiian corals *Porites compressa*, *Montipora verrucosa* and *Fungia scutaria* for 2.5 h. During 25 days of subsequent observation no visible evidence of damage was noted.

Lewis (1971) found that four species of coral exposed to crude oil or an oil spill detergent exhibited ruptured tissue and other signs of distress. These experiments were carried out in sealed vessels in order to prevent the escape of volatile oil fractions. Since the harmful effects may have been due to these volatiles it cannot be assumed that the results are representative of what might occur during an oil spill when the volatiles would have escaped to the atmosphere.

Published reports of oil spills in the vicinity of coral reefs also provide no conclusive evidence of injury to corals. In connection with a Panama spill where inshore littoral fauna suffered heavily, Rützler and Sterrer (1970) state that corals escaped readily observable damage, presumably because they were continually submerged. Spooner (1970) observed no damage to reef communities in Tarut Bay, Saudi Arabia, in an area of long-term chronic

oil pollution. Shinn (1972) states that staghorn corals grow "apparently unaffected" around loading terminals for oil tankers in the Persian Gulf and near the entrance to the Suez Canal in the Red Sea.

Unlike reef corals, the alcyonarian coral *Heteroxenia fuscescens* has been shown to be injured or killed by crude oil floated above it. This species proved more resistant to crude oil under conditions of continuous flow in large tanks than under static conditions in small jars (Cohen, 1973). The results of this work suggest that the farther specimens are held below the surface film of oil, the less effect the oil has on them. Cohen (1973) also found that an oil spill dispersant was at least 100 times as toxic to *H. fuscescens* as two species of crude oils tested. The tissue of this species incorporated small amounts of components of the crude oil floating above it.

In some areas reef surfaces are exposed during low spring tides. This phenomenon, which is common on Indo-Pacific reefs, was taken into account in an experiment carried out by Johannes et al. (1972). They exposed 22 species of corals, whose upper portions were above water, to Santa Maria crude for 1.5 hours. Oil adhered to portions of the surfaces of most species. Tissue death ensued within a few days in these areas but not on areas where the oil did not adhere. The most extensive experimental work to date on the impact of oil on various tropical marine species and communities is that of Birkeland et al. (1973). Although it will be described briefly here and in Odum and Johannes (Chapter 3), it is difficult to summarize, and the original publication should be obtained by anyone interested in the subject. Birkeland et al. (1973) observed that corals exposed directly to diesel or Bunker C fuel showed responses, depending on species and experimental conditions, that ranged from a mere decrease in growth rate, in some cases, to death after a few days to several weeks, in others. The work of these authors points up the desirability of long-term experiments; some of the effects on coral were first noted more than one month after exposure to oil.

Shinn (1972) noted that "*Montastrea annularis* can survive two hours total immersion in Louisiana crude while *Acropora cervicornis* exposed for two hours to a mixture of seawater containing one part crude to 6 to 12 parts seawater caused immediate retraction of polyps but complete recovery in 24 hours". He does not state whether subsequent observations were made on the condition of these corals. It seems premature to state, as Shinn does, that "it would seem safe to conclude then that crude oil spills do not pose a significant threat to Atlantic reef corals".

Other reef organisms

Although he makes no mention of corals, Gooding (1971) reports the extensive destruction of associated reef fauna in connection with the release by a tanker of 6 million gallons of assorted oils off the harbor entrance at Wake Island. Of the reef fishes that died, 2,500 kg washed ashore. Numerous reef invertebrates were also killed.

Diaz-Piferrer (1962) briefly described the effects of a large crude oil spill from a ship that ran aground on a reef off Guayanilla Harbor, Puerto Rico. Extensive mortality of marine fauna and flora occurred in sublittoral and intertidal zones and beach erosion was extensive. There was "obvious depletion of fisheries". Mohammad (1974) describes the effects of chronic oil pollution on a polychaete in Kuwait.

Birkeland et al. (1973) observed that algal diversity and productivity increased on artificial substrates and in intertidal reef flat communities exposed to Bunker C fuel. The damage caused to the intertidal fauna in meter square quadrats exposed to diesel or Bunker C fuel oil was relatively small and short-lived relative to observed responses in comparable temperate marine communities exposed to massive oil spills. Birkeland et al. (1973) suggest that this may be an artifact of the small areas exposed to oil in these experiments, and that realistic oil pollution experiments may entail exposing areas on the scale of hectares rather than square meters.

Communities may be damaged by oil, from man's viewpoint, even when their structure is unaltered, since commercial species can become tainted with oil and therefore unsaleable (Grant, 1969; Connell, 1971; Deshimaru, 1971; Sidhu et al., 1972).

It is well known that dispersants used to control oil spills are often toxic — sometimes more so than the oil on which they are used. Observations on the effects of oil dispersants on tropical marine organisms have been made by Cerame-Vivas (1968), Cohen (1973) and Lewis (1971). Tests run during the Ocean Eagle oil spill in Puerto Rico indicated that dispersants (emulsifiers) created more damage than they prevented by dispersing the oil through the water column and by causing serious beach erosion through reduction in the physical attraction of sand grains to one another. Adsorbants, which facilitated the collection and removal of the oil, proved more useful. Ekoperl-33 (volcanic ash treated with silicone to enhance absorbance and retard sinking) proved the most desirable of several absorbants tested. It spontaneously immobilized large quantities of oil and was susceptible to only slight leaching (Cerame-Vivas, 1968).

Considerable effort has gone into the planning of baseline studies in temperate marine areas so that, in the event of an oil spill, the state of the affected community prior to the spill is known. The availability of personnel to make such studies is very limited in most tropical areas. Accordingly it does not seem practical to devote much effort to this type of study except in very high risk areas. Efforts might better be spent equipping and training mobile teams who can respond within a few hours to an oil spill.

Oil and oxygen

As discussed in the section of this paper on sewage pollution, even a small reduction in dissolved oxygen levels may, under some circumstances, bring

about considerable destruction of reef fauna. Blumer (1971) points out: "the oxygen requirement in bacterial oil degradation is severe; the complete oxidation of 1 gallon of crude oil requires all the dissolved oxygen in 320,000 gallons of air saturated sea water." Therefore, bacterial degradation may cause additional ecological damage through oxygen depletion.

Oil films may also cut down oxygen transfer across the air—water interface. Kinsey (1973) reviews the general literature on this subject and describes the results of experiments he carried out on a coral reef with Moonie crude oil. Layers of this oil up to 0.7 mm thick did not interfere measurably with atmospheric exchange except insofar as they calmed the water surface somewhat, reducing the surface area available for gaseous exchange. As Kinsey (1973) states, it would be useful to study films of much greater thickness and films of higher viscosity.

Future problems

Extensive and expanding offshore drilling of oil is planned or underway in various parts of Southeast Asia, in the Persian Gulf, the Caribbean, Fiji, Tonga and in other reef areas, often with no prior environmental impact analysis. Australian public opinion forced the government to suspend oil exploration permits for the Great Barrier Reef until the threats these activities may pose can be evaluated.

Transport and refining of oil in the Caribbean is widespread and expanding rapidly. Here in 1971 there were 62 refineries situated in 17 different countries. A "spectacular increase" of movement of crude oil through the region is predicted (Kesterman and Towle, 1973). According to these authors, "from Bermuda to Anegada, to Aves to Trinidad, to Bonaire, even to remote beaches, these areas are plagued with oil drifting ashore from carelessly and illegally pumped tanker bilges".

THERMAL POLLUTION

Laboratory tolerance tests

The threat of destruction or alteration of marine communities by heated effluent from power plants and other industrial installations is greatest in the tropics. Unlike the biota of temperate and polar regions, tropical organisms characteristically live at temperatures only a few degrees below their upper lethal limit (Mayer, 1914). Edmondson (1928) showed that 11 of 13 species of Hawaiian corals survived less than 24 hours at 32°C — a temperature only five to six degrees higher than ambient summer water temperatures. Similarly narrow ranges exist between maximum ambient temperatures and lethal temperatures for corals in Guam (Jones and Randall, 1973), on the Great Barrier

Reef (Mayor, 1918a), the Caribbean (Mayor, 1918b), and Samoa (Mayor, 1924). In the tropics, in contrast to higher latitudes, there are few species capable of replacing a community eliminated by thermal stress.

Some corals which live naturally in exceptionally warm waters have been reported from the Persian Gulf (Kinsman, 1964) and the Laccadives (Gardiner, 1903). The ranges between maximum environmental temperatures and lethal temperatures are probably even smaller here.

Zieman and Johannes (Chapter 5) discuss temperature tolerance levels of tropical marine organisms other than corals.

Lethal temperatures as determined in the laboratory during short-term experiments must be extrapolated to nature with caution. Organisms may not be able to tolerate temperatures as high in nature as they do under controlled conditions in the laboratory (e.g., Read, 1967; North et al., 1972). Also as Mayer (1914) pointed out, the temperature at which the feeding reactions and normal metabolic processes cease are more significant than death temperatures, "for naturally an animal cannot survive long in water in which it can neither move nor function".

It has been established that sublethal heat stress is deleterious to corals for a number of reasons. Mayor (1918b) reported that three species of Caribbean corals ceased to feed at temperatures 1.5—3.0°C lower than their lethal temperatures. Edmondson (1928) reported similar results for Hawaiian corals and noted that reproductive rates also dropped at high sublethal temperatures. Yonge and Nicholls (1931) observed that corals extrude their zooxanthellae in strings of mucus at high sublethal temperatures. Coral planulae may not settle at high sublethal temperatures (Edmondson, 1946).

Gross photosynthesis generally exceeds respiration in hermatypic corals. With increasing temperature, photosynthesis would have to increase faster than respiration for maintenance of a stable photosynthesis/respiration (P/R) ratio. It does not: Coles (1973) found that P/R ratios usually decreased with increasing temperature in several species of corals from Hawaii in the temperature range of 18—31°C. (Similar results were obtained by Hohman and Tsuda (1973) working with the reef alga *Caulerpa racemosa.*) A direct relationship was indicated between the temperature effect on the P/R ratio and the upper thermal tolerance of the species. An inverse relationship between high temperature tolerance and respiration rate was also observed.

Coles found that high temperature tolerance in *Montipora verrucosa* decreases as the salinity was lowered below ambient, but did not decrease when the salinity was raised to 40% (see also Edmondson, 1928). Coles (1973) discusses a variety of observations suggesting that Hawaiian corals have lower upper thermal limits and metabolic optima than corals in warmer waters.

Field observation

In shallow tropical bays and back-reef areas where water is heated rapidly

by the sun, growth of various marine plants and animals may be prohibited by high temperatures even under normal conditions. Mayor (1918a), for example, showed that high temperatures within 150 m of shore inhibited coral growth in the Torres Straits. Wells (1952) drew similar conclusions concerning coral growth at Arno Atoll. Mass mortalities of intertidal and shallow subtidal species in the tropics due to high temperatures and desiccation associated with low midday tides are described by Mayer (1914), Yonge and Nicholls (1931), and Glynn (1968). Obviously *any* increase in summer water temperatures brought about by thermal effluent released at the water's edge in such environments will expand the area inimical to growth.

Thermal effluent from a power plant at Turkey Point, Biscayne Bay (Florida) has in the recent years killed virtually all plants and killed or greatly reduced animal populations in the bay in an area circumscribed closely by a +4°C isotherm (Roessler and Zieman, 1969; Thorhaug et al., 1973a; Zieman and Wood, Chapter 5). Mortalities extend for more than 1.5 km from the outfall.

Corals were among the most sensitive species involved, being killed by the effluent at greater distances from the outfall than most other organisms (Anonymous, 1970). Colonies of the corals *Solenastera hyades* and *Siderastrea siderea* have been eliminated in the area noticeably effected by the heated effluents. Even in areas of marginal influence on other biota many individuals of these two coral species were dead or visibly damaged. Here living *S. siderea* specimens were pale and appeared to be more susceptible to overgrowth by algae and encrustation by other forms (L.L. Purkerson, personal communication, 1972).

Heated effluent from a power plant on Guam led to the destruction of reef margin corals in a zone of 4320 m^2 and damage to a peripheral area including a total of 10,000 m^2 (Jones and Randall, 1973). Fishes, crustaceans, echinoderms and most species of benthic algae also disappeared. Determining the specific cause(s) of these mortalities was complicated by the fact that significant levels of chlorine were added periodically to the cooling water as a desliming and anti-fouling agent. The effluent also contained elevated levels of copper. Jones and Randall (1973) described the relative tolerance of many species of corals, fish and algae to the plant effluent. Large numbers of juvenile rabbitfish were also killed by entrainment in the cooling water.

Jokiel and Coles (1974) studied the impact of thermal effluent on corals in Hawaii. Nearly all corals in water 4—5°C above ambient were killed. A 30% increase in heated discharge resulted in an increase in the area of dead or bleached corals by 100%. The relative heat resistance of different coral species observed in this field study coincides with the results of Edmondson's (1928) laboratory studies. Jokiel and Coles (1974) data also support Mayer's (1917) suggestion that the ability of coral species to resist heat stress is inversely related to their metabolic rates.

Thermal plumes from power plants have been observed to attract reef fish (DeSylva, 1969; Clarke et al., 1970a,b; McCain and Peck, 1973; Jokiel and Coles, 1974). One might reasonably surmise that fish are not liable to seek such elevated temperatures if they are harmful. However, there is evidence that weight loss, lowered reproductive success, higher disease rates and increased mortality can result from prolonged exposure to temperatures only slightly higher than ambient (Coutant, 1970; see also Chesher, Chapter 6). Sharks seem particularly attracted to thermal plumes in tropical waters (DeSylva, 1969; personal observations); thermal effluent may thus sometimes constitute a distinct *human* health hazard.

ABNORMAL SALINITIES

Hypersaline effluents

Although some corals grow in ambient salinities as high as 48% (Kinsman, 1964), many corals are particularly sensitive to salinities slightly above normal oceanic values. Nine out of twelve Hawaiian species tested could not tolerate salinities of even 110% of normal for more than two weeks. Three species died within three days. Most species died within 24 hours when exposed to salinities 150% of normal (Edmondson, 1928). Goldberg (1974), who studied the effects of altered salinities on six species of gorgonians, concluded that they are more stenohaline than scleractinian corals.

Effluents from desalination plants are characterized by elevated levels of not only salinity but also temperature and toxic metals such as copper, zinc and nickel. They may also contain various chemicals added for pH and corrosion control (Zeitoun et al., 1969a). These effluents are often sufficiently saline that, despite their elevated temperature, they sink and flow along or near the bottom where there is no loss of heat directly to the atmosphere and where their potential for damage to benthic communities is greatest (e.g., Van Eepoel and Grigg, 1970; Chesher, Chapter 6).

Toxic metals are introduced in desalination effluents due to corrosion of metallic surfaces of the distillation system. Zinc and nickel ions, though toxic to marine organisms at elevated concentrations, are much less harmful than copper at equivalent concentrations; copper appears to be the toxic contaminant of greatest concern in such effluents (Zeitoun et al., 1969a). The mixture of brine blowdown and cooling water discharged from desalination plants may contain this element at concentrations 6–8 times as high as the recommended maximum concentration of 0.02 mg/l, and 30–40 times the natural copper concentration in coastal waters (Zeitoun et al., 1969a).

Studies of the effects of desalination effluents are greatly complicated by interactions between different stress factors. For example, zinc and copper can act synergistically on organisms, as can copper and elevated temperature.

Elevated salinities may reduce the toxicity of copper due to the presence of more ions to compete with copper for sorption sites (Zeitoun et al., 1969a).

Desalination effluent in Key West, Florida, appears to be responsible for the reduction or disappearance of some marine species from the outfall area and an increase in the numbers of others, including fish and shellfish of several edible species (Clarke et al., 1970a,b; Chesher, Chapter 6). Clarke et al. (1970a) reported that desalination effluent was detected, due to elevated temperatures and salinity, at greater distances from the discharge point than were gross changes in the biological communities. This suggests, they state, that the levels of effluent dilution necessary to produce minimal environmental modification should be amenable to prediction.

The use of combination power—desalination plants in the vicinity of marine communities may be expected to amplify ecological problems because of the elevated temperatures of the effluent. Van Eepoel and Grigg (1970) have made some preliminary observations near one such plant which discharges effluent into Lindberg Bay, St. Thomas, Virgin Islands. Salinities and temperatures near the outfall were only slightly higher than ambient on two occasions when surveys were made. On another occasion, however, divers could not measure the effluent temperature because it was above that which they could tolerate. Corals and other invertebrates were killed to a distance of 200 m from the discharge plant. Difficulty in studying the effects of the effluent beyond that point were encountered because the bay was also polluted with oil and soot from the plant as well as from excessive sedimentation due to land-clearing and associated erosion. Power and water output from the plant were expected to double within the following three years.

Desalination plants are proliferating as the demand for freshwater grows and the cost of desalination decreases. It is disturbing, therefore, to learn how little effort seems to have been made to examine the ecological consequences of brine disposal. Of the first 481 research and development reports based on work carried out under contract to the United States Office of Saline Water, only three specifically concern ecological problems attending effluent disposal in marine environments. It is also noteworthy that of the eight desalination plants surveyed by Zeitoun et al., (1969a) (five of which are located on coastlines in regions of extensive coral reef development), no governmental regulations existed with regard to effluent discharge, and in only one case (Nassau, Bahamas) was an oceanographic or engineering study of the effect of the effluent undertaken before the plant was built.

Low salinities

Poor land management leads not only to sediment stress on coastal communities but also the stress of lowered salinities due to accelerated freshwater runoff. Freshwater is clearly a pollutant when introduced into marine environment in excess due to man's carelessness.

Most species of Indo-Pacific and Atlantic corals that have been tested for tolerance to low salinities survived exposure to 50% seawater for less than two days (Mayor, 1918a; Vaughan, 1919; Edmondson, 1928; Wells, 1932). Eight out of fifteen Hawaiian species survived less than a week in seawater diluted by one-quarter with freshwater. Exposure to freshwater for 30 min killed most species (Edmondson, 1928). Coral planulae may not settle or grow at reduced salinities (Edmondson, 1929). Calcification in corals decreases with decreasing salinity (e.g. Yamazato, 1970). Observations by Maragos (1972) in Hawaii suggest, however, that the growth of some corals is unusually rapid in varying and slightly suboceanic salinities. Lewis et al. (1968) made similar observations in Jamaica.

There are a number of accounts of the destruction of reef biota by freshwater during heavy rains (Hedley, 1925; Rainford, 1925; Hiatt, 1958; Slack-Smith, 1960; Cooper, 1966; Banner, 1968). It was not always clear to what extent sedimentation rather than salinity caused the observed damage. Goreau (1964) documented the mass expulsion of zooxanthellae from a variety of species of Jamaican corals exposed to low-salinity waters associated with flooding. This loss of zooxanthellae may be expected to lower the rate of calcification of corals significantly (Goreau, 1964). It may also lower tissue growth rates since dissolved organic compounds released by zooxanthellae are of nutritive value to the coral host (e.g. Muscatine and Cernichiari, 1969). Recovery of zooxanthellar populations in corals which survived the incident in Jamaica was not complete after fourteen weeks (Goreau, 1964).

CHLORINATED HYDROCARBONS

Little is known about the distribution of chlorinated hydrocarbons in reef organisms and less about their effects. McCloskey and Chesher (1971) found that a reef coral, *Acropora cervicornis*, contained 3—12 ppb DDT, 260—320 ppt Dieldrin and 200—300 ppt DDE plus PCBs. *Astrangia danae*, a non-reef building coral from Woods Hole, showed detectable quantities of Dieldrin only. Both species rapidly concentrated experimentally introduced DDT, Dieldrin and Aroclor 1254.

Several species of reef corals exposed to an organochlorine mixture responded by increasing their respiration rates while decreasing their photosynthetic rates (McCloskey and Chesher, 1971), or, expressed in other terms, their P/R ratios decreased and their light compensation levels increased. The authors state: "if, as it appears, subtidal corals at the depth studied are maintaining a diurnal O_2 balance at very close to 1.0, then any alteration which drops this value to significantly less than 1.0 may result in profound alterations in the reef community."

Acanthaster collected from a variety of Pacific sites contained from 0.04

to 3.89 ppm DDT, 0.01—1.04 ppm Dieldrin (McCloskey and Deubert, 1972) and 0.01—0.21 ppm PCBs (McCloskey and Deubert, 1973). No marked correlation was found between pesticide levels and starfish abundance. For additional data on pesticide levels in *Acanthaster* from the Great Barrier Reef see Tranter in Walsh et al. (1971, appendix F).

Barracuda taken from the Flower Gardens reefs in the northern Gulf of Mexico were found to contain high levels of DDT. Significant amounts of Dieldrin and Endrin were also present (T. Bright, personal communication, 1973; Houston Post, December 12, 1970). Low levels of DDT were found in barracuda taken in the Florida Keys (Deichmann et al., 1972). Giam et al. (1973) found variable but generally low levels of chlorinated hydrocarbons in groupers in the Gulf of Mexico and the Bahamas. Tissue levels increased with increasing size in the area where the highest concentrations were found.

The danger of lagoon contamination by pesticides, sometimes used in large quantities on atolls, has recently been noted (Anonymous, 1971). Fifteen to twenty tonnes of fish died suddenly in Truk Lagoon on April 17, 1970, hospitalizing six people who ate some of them. The flesh was found to contain the pesticide Endrin in the highest concentrations ever recorded in fishes to that date (Bourns, 1970). People of the Cook Islands now use chlorinated hydrocarbon pesticides as a substitute for derris in killing lagoon food fishes (Hambuechen, 1973).

Seba (1970) refined bioassay procedures for Dieldrin using three tropical fish, including the tarpon *Megalops atlanticus*. Shultz (1971) found moderately low levels of chlorinated pesticide residues in the sediments, water and biota in a polluted Hawaiian estuary. Nunogawa et al. (1971) tested the resistance of two coastal Hawaiian fish to DDT, Dieldrin, Lindane and phenol. Lane (cited in Randall, 1972) found the tropical sea urchin *Echinometra* had a higher resistance to Dieldrin than any of the other vertebrate or invertebrate animal she tested.

HEAVY METALS

Trace metal pollution has been little studied in the tropics. Normal trace metal concentrations are much lower in carbonate sediments characteristic of reef areas than in clay sediments where they are strongly sorbed to clay particles. Thus, comparatively large fractional changes in sediment metal concentrations may be expected in reef areas polluted with heavy metals, (Segar and Pellenbarg, 1973).

Taylor and Bright (1973) analyzed the distribution of heavy metals in reef-dwelling groupers in the Gulf of Mexico and the Bahamas. They found mercury, cadmium, lead, copper and zinc levels below those generally thought dangerous to humans. Arsenic levels in the muscles of most groupers tested surpassed U.S. Food and Drug Administration tolerance level of 0.5

ppm. The highest concentrations were found in the Nassau grouper from the Bahamas where it is the main commercial fish. Taylor and Bright (1973) argue convincingly that these high arsenic levels found both in groupers and various other reef species in their stomachs are a natural phenomenon; phosphorus levels are characteristically very low in tropical waters, and reef organisms probably accumulate arsenate, which is chemically similar, as a substitute in the phosphate transport system.

Barker and Zeitlin (1972) found that the elemental enrichment and metal/ion ratios of sea surface microlayers can be changed and enhanced by orders of magnitude by pollutants in near-shore waters in Hawaii. While the water samples at 0.6 m depths were relatively normal, microlayer samples often contained metallic pollutants in high concentrations, strongly organically associated and available for assimilation into marine food chains.

Heavy-metal levels in sediments and marine biota have been determined by Evans et al. (1972) in Pearl Harbor, Hawaii. Chesher (Chapter 6) reports on copper toxicity tests on a variety of south Florida marine organisms. Heavy-metal pollution is also mentioned above in connection with desalination and power plant effluents.

MISCELLANEOUS

Nuclear weapons tests

The obliteration of coral reef communities in the vicinity of nuclear blasts is of minor significance for man compared with subsequent effects spread over much wider areas. The most obvious immediate biological impact outside the area of complete destruction is that of radiation burns to the biota. Donaldson, cited by Hines (1962), collected living fishes in the vicinity of a recent blast whose skin was missing from one side as if the animals had been "dropped in a hot pan". This also is a comparatively trivial phenomenon. It is the radioactive contamination of the food chain which has by far the greatest consequence for man.

The geographic distribution of radioactive contaminants has been difficult to predict in advance of nuclear explosions because of the complexity of the atmospheric and ocean currents on which they are carried. It is nonetheless well known that serious contamination may extend for great distances. Because of atmospheric circulation patterns, areas more than 240 km from a thermonuclear explosion at Bikini Atoll received four times as much radiation as parts of Bikini itself. Parts of Rongelap Atoll 160 km away received sufficient fallout within 48 hours that human survival in the absence of shelter would have been unlikely (Hines, 1962). It was not until four years later that all reef species at Rongelap were declared safe for human consumption (Hines, 1962). A single nuclear blast may thus render fish and shellfish

unsafe or unsaleable for thousands of square kilometers.

Generally it is the lower trophic levels, the algae and herbivores, which become most radioactive immediately after a blast. Later carnivorous reef fishes become more radioactive than omnivores or herbivores (Donaldson, 1960; Hines, 1962). As radioisotope levels in the community subsequently diminish, *Tridacna* and other edible reef bivalves are among the last reef animals to become safe for human consumption (Hines, 1962; see also Welander, 1969).

The ecological impact of radioactive contamination on reef communities is almost unknown. Gorbman and James (1963) discovered radiation damage to the thyroid glands of various coral reef fishes after a nuclear test at Enewetak Atoll. This appears to be the only observed damage attributable to the injurious effects of radioactive contamination of the reef food chain. Recent X-ray and autoradiographic studies of corals exposed to radioactive fallout during nuclear testing at Enewetak Atoll revealed "no micro- or macroscopic effects . . . which can be correlated with or attributed to the nuclear tests" (Knutson and Buddemeier, 1973).

Dying animals on the reef are liable to be consumed by reef scavengers and carnivores before they are observed (Donaldson, 1960). To evaluate the impact of any stress on a reef community, population sizes of various species should therefore be measured before and after the stress occurred. Unfortunately this does not seem to have been done in connection with nuclear tests in the Pacific in the 1940's and 1950's. The statement by Schultz, "the pressure of population from all sides into the damaged area is very great and soon replaces the losses" (cited with approval by Templeton et al., 1972), therefore seems premature. It is to be hoped that the French have made efforts to clarify this in connection with their recent tests in the South Pacific.

In January 1971, James Maragos and I had a brief opportunity to examine some shallow patch reef communities which had apparently been destroyed by a nuclear blast at Enewetak Atoll. The date of destruction could not be determined because several nuclear tests were carried out on different dates in the vicinity. The nearest blast created a crater several hundred meters away on the inter-island reef. The last year in which nuclear tests took place in the vicinity was 1958.

The reefs are located several hundred meters northwest of Yvonne (Runit) Island in an area where the inter-island reef begins to slope toward the lagoon. The tops of the reefs are covered by about 0.5 m of water at low tide. The communities had been dominated structurally by *Acropora corymbosa* ("table Acropora"). We could not tell whether its destruction was caused by heat or radiation. Since skeletons of *A. corymbosa* were still standing we assume the communities were not killed by the force of the blast.

Extensive regrowth of *A. corymbosa* had occurred. Some living specimens

were nearly 3 m in greatest diameter. Coral diversity was low however. Only 4 species of corals were common, whereas Maragos counted 9—10 common genera and estimated there were about 15 common species on typical *Acropora*-dominated inner reef communities at Enewetak. Diversity and abundance of other invertebrates, fishes and algae also appeared low on the affected reefs. Very little grew on the dead skeletons of *A. corymbosa* or the substrate. In short, although regrowth of *A. corymbosa* on this reef was impressive, the community had by no means recovered thirteen years after the last nuclear test in the vicinity. Patch reefs further from the test sites, several hundred yards south of Yvonne Island, showed no readily observable effects of the blasts.

Fishing with poisons and explosives

Though usually illegal, the use of dynamite and poisons to kill reef fishes is commonplace in the tropics (e.g., Ronquillo, 1950; Anonymous, 1952; Owen, 1969, Ramas, 1969; Powell, 1970; Sverdloff, 1973; Jones et al., 1973; Lulofs, 1973). Christian (1973) reviews past research on the effects of underwater explosions on fish and describes a new method for predicting the maximum ranges to which such explosions injure fish. The depth at which a charge is exploded has an important influence on the shape and extent of the surrounding region in which such damage occurs. The destruction of habitat by explosives in coral reef communities has never been adequately evaluated.

Ronquillo (1950) summarizes anatomical damage observable in fish killed by explosives. A knowledge of these characteristics should be useful to fisheries personnel investigating suspected dynamite fishing.

Ronquillo lists the following anatomical disarrangements:

Gross aspect

1. The air bladder, if present, is almost always ruptured and blood clots are found in its lumen.
2. The vertebral column may be fractured in any part along its length.
3. Localized hemorrhages are present around the area of fractured parts due to the destruction of the blood vessels and tearing of adjacent tissues.
4. Parts or all of the contents of the body cavity may be damaged or crushed with hemorrhages depending upon the size, shape, position and distance of the fish from the explosion.
5. Fracture and/or dislocation of the abdominal ribs from the vertebral column may be found especially in spiny fishes, with accompanying hemorrhages present in the area of the fracture.
6. The blood vessels below the vertebral column may break and cause hemorrhages of varying degrees along that region.
7. Rupture of the parietal peritoneum, especially that attached to the abdominal ribs.

Roentgenological aspect

1. Dislocation and/or fracture of the vertebral column and ribs, if present, are clearly shown in the negative.
2. The air bladder, if ruptured, will be filled with blood and will be obliterated in the negative. If not ruptured and, therefore, filled with air as in normal fish, it occupies a definite shape and position in the abdominal cavity.

In the Philippines, according to Ramas (1969), a "method used by blast fisherman is to find some large coral formation about the size of a regular house in about 100 to 150 feet of water. They then rig up some batteries, wires and light bulbs inside mason jars, and attach these underwater lights around the area. They also chum the area with conch and fish chunks. The area is completely wired with dynamite, with the detonator in the boat on the surface. Periodically, a scuba diver dives down to check on the fish gathering. When the big fish come around in enough numbers, the charge is detonated and scuba divers are used to retrieve the fish. So much fish is obtained this way that . . . the smaller fish under five pounds are not even picked up."

In Truk Lagoon a group of fishermen with fast boats are organized for obtaining dynamite, fishing with it and retailing the dynamited fish (Fuchs, 1968). Powell (1970), referring to Truk Lagoon, states: "An explosion echoing through a village is ample notice that fresh fish are on sale . . . People who are generally protein deficient in their diet do not protest or complain if the fish is mushy."

Lulofs (1973), commenting on widespread use of explosives to capture reef fish in Malaysia, states: "Although thorough policing of coastal waters is almost impossible, this form of fishing could be greatly reduced, if not entirely prevented, by the provision of legislation which not only prohibits the sale of blasted fish, but cancels without right of appeal the licenses of those dealing in blasted fish. Blasted fish can be identified, and once the sales outlets are removed, the operation becomes pointless."

Surplus 700-lb (ca. 20-kg) U.S. Navy bombs were used (25 at a time!) to clear a channel through the reef at Kayangel Atoll, Palau, in 1973. The concussions cracked every cement structure on an island more than 5 km away. Two months later underwater visibility was still greatly reduced up to several kilometers from the blast site and coral damage, apparently due to sedimentation, was still in progress (D. Faulkner, personal communication, 1974).

The use of poisons to kill marine fishes in Hawaii was outlawed in 1850. But here, as elsewhere, the practice remains common today. Bleach (sodium hypochlorite) was used by fishermen in a cave with which I am familiar off the south coast of Oahu, Hawaii. It contained dense schools of three species of fishes, populations of palinurid and homarid lobsters, and a rich, colorful

encrusting fauna of sponges and tunicates. After the cave was exposed to bleach almost no visible living macrofauna remained. Recolonization was far from complete a year later.

Ciguatera

Algae are generally among the first macroscopic organisms to colonize any fresh surface in reef areas, including the surfaces of blasted or dredged areas and of newly deposited sediments (Dawson, 1959; Brock et al., 1966), artificial reefs (Randall, 1963), skeletons of corals eaten by *A. planci* (Chesher, 1970), sunken ships, abandoned war materials and other human artifacts (Randall, 1958; Halstead, 1972).

The apparent connection between these new algal growths and fish poisoning was first noted by Dawson et al. (1955) and discussed in detail by Randall (1958) and Dawson (1959). These authors pointed to the relationship between the availability of new surfaces in the reef environment, the rapid growth of algae, and the development of toxicity, known as ciguatera, in normally edible species of reef fish in the immediate area. They theorized that toxic algae, possibly blue-greens, growing unusually rapidly on new reef surfaces, are at the base of the toxic food chain leading ultimately to man and causing illness and sometimes death. Yasumoto et al. (1971) found that the most toxic fraction of the food of highly ciguateric detritus-feeding surgeon fish consisted of small unidentified particles of low specific gravity.

Randall (1958) points out: "The importance of ciguatera goes beyond the purely medical aspects of treatment of patients suffering from the ingestion of toxic fish. Many nontoxic fish are denied to humans or domestic animals as food because of fear of their being poisonous. Sections of reef with a reputation of harboring poisonous fishes are not fished, whereas nontoxic areas may, as a result, be overfished."

Bagnis (1969) has provided a valuable account of the development of ciguatera in a reef community in the same areas and in the same chronological sequence as disruptive human activities. However, there are numerous examples of dredged or otherwise disturbed reef areas where ciguatera has not developed (e.g. Bagnis, 1973). At Johnston Island where ciguatera has occurred in the past, extensive dredging of reef areas in 1963 has not been followed by any new outbreaks (A.H. Banner, personal communication, 1973). Nonetheless, the relationship between the disruption of reefs by man and the subsequent development of ciguatera in the immediate area seems too frequent to be coincidental, and research continues on this recalcitrant problem.

Crown-of-thorns starfish

Whether large aggregations of coral-eating crown-of-thorn starfish, *Acan-*

thaster planci, are brought about by man's activities, including pollution, or are a natural phenomenon, is, at this writing, still vigorously debated. Some lay conservationists (e.g. Cropp, 1971) have inferred from those who suggest that they are a natural phenomenon,that outbreaks should be allowed to progress unchecked.

This reasoning is characteristic of those who attribute to man the distinction of being the only force in the biosphere capable of exerting "unnatural" influences on it, and who assume that all other influences being, by their definition "natural" are therefore somehow always desirable. Anyone really committed to this philosophy would presumably allow his house to burn down if lightning struck it, rather than interfere with a natural phenomenon.

Whether ecological disruptions caused by non-human agencies are desirable from man's viewpoint (and whom else can we poll?) depends upon the particular circumstances. Fire is disastrous to some types of communities but ultimately beneficial to others. Man tries to prevent fires in the former case regardless of their origin and sometimes sets them in the latter.

Similar reasoning should govern decisions concerning control of *Acanthaster* outbreaks. Unfortunately, we do not possess adequate information to make a final decision. We know *Acanthaster* outbreaks are harmful to reef communities on a short-term basis; we do not know whether they are ultimately beneficial over a longer period, as Dana et al. (1972) have suggested, and we may not find out for decades. An interim decision should be based on what we know rather than on what we do not know. Therefore, until new information dictates otherwise, serious *Acanthaster* outbreaks should be controlled, in my opinion, irrespective of their origin.

The esthetic fishery

The collection of shells, corals and tropical aquarium fish cannot be called pollution. Neither does it fall within the conventional purview of commercial fisheries biology. Thus, what might be called the esthetic fishery, which sometimes clearly constitutes an environmental threat in coral reef communities, has received virtually no scientific scrutiny. It seems justifiable, therefore, to mention it briefly here.

There are numerous reports throughout much of the Caribbean and Indo-Pacific of overharvesting of corals, gastropods and aquarium fish by amateur and professional collectors. The physical destruction of shallow reef flats due to trampling, and of individual coral heads due to the use on them of crowbars by shell collectors, are related problems. Since no systematic quantification of the effects have apparently ever been attempted it is not possible to say exactly how serious they are. But it is my impression, judging by reports from shell collectors and biologists and from my own observations, that the problem is widespread and growing.

Corals from the Florida Keys, for example, are sold by the truckload to

shell stores (e.g. Barada, 1974), but extensive collecting is not limited just to areas of high local demand. Adult specimens of the giant clam *Tridacna maxima* are almost gone from some Marshall Island atolls where they were extremely abundant twenty years ago. Exporters are reportedly rapidly depleting some reefs of aquarium fish in Malaysia and Indonesia (e.g. Lulofs, 1973). Cousteau (1972) claims that teams of Tahitians, using crow bars to collect shells, "destroy six miles of reef per week".

A number of species of reef fish and invertebrates which remove parasites from other fish are aquarium favorites. These species constitute a very small portion of total community biomass. But their importance to the reef fish community far outweighs their biomass according to several investigators. Limbaugh (1961) reported that when he removed all known cleaners from two small Caribbean patch reefs, reef fish populations were "drastically reduced" within a few days. Within two weeks almost all except the territorial fishes had disappeared. Many of the fishes remaining developed "fuzzy white blotches, swellings, ulcerated sores and frayed fins". On the basis of observations in the Red Sea, Slobodkin and Fishelson (1974) state: "it seems evident that the local diversity of coral reef fishes may be to a large extent determined by the presence of cleaner wrasse stations."

It is not difficult to completely remove some species of cleaners from a reef. If the results are as serious as the above observations suggest, then clearly the capture and sale of these species should be outlawed.

Reef communities can comfortably yield limited quantities of some species of fish and shells and small corals if gathered with care, just as they yield food fish. But until research is done on the optimum sustained yields of species sought by collectors, and on the physical impact of collecting on the environment, complete closure of some reefs to collecting is probably the only way to prevent serious excesses. Certainly the movement or breaking of coral to obtain the fish and shellfish within them should be prohibited.

Alien species

The importation of alien species has brought about numerous well-known ecological dislocations in terrestrial environments. Potentially hazardous introductions of various alien species into coral reef communities has been noted by various authors (e.g. Dickie, 1875; Doty, 1961, 1973b; Gilbert, 1962) but the identity and ecological impact of these species has received very little study.

As noted above, one species of benthic algae introduced to Hawaii around 1951 has become the dominant species in some reef areas in the vicinity of large human populations (Soegiarto, 1972; Doty, 1973b). There is a possibility of passage of crown-of-thorns starfish, sea snakes and other species of dubious value through the Panama Canal from one ocean to the other via seawater ballast (Chesher, 1968) or through the proposed sea level canal (Thomas, 1970).

THE FATE OF DAMAGED REEF COMMUNITIES

When a reef community is destroyed the ecological conditions that follow cannot be expected to coincide with those which preceded the initial development of the community. Thus, it cannot be taken for granted that the reef community will ever replace itself. Wood-Jones (1907) for example, reported that thirty years after the destruction of a coral reef by "foul water" released from a volcanic vent, only dead corals remained; there was no sign of regeneration. Rainford (1925) reporting 1918 flood damage to corals in the Whitsunday Group, Great Barrier Reef, doubted that recovery would ever occur. Seven years later Hedley (1925) found that one luxuriant reef subjected to this flood had been "planed away by the waves as if some huge razor had shaved off the coral growth down to low tide level". Except for the bivalve *Chama*, animal life was "very scarce". In 1953 areas still remained in the Whitsunday Group where negligible recolonization had occurred (Stephenson et al., 1958).

In 1965 corals and other invertebrates were killed by flooding on the Coconut Island reef in Kaneohe Bay, Oahu (Banner, 1968). An area on this reef where corals once covered 70—80% of the bottom (Gordon and Kelly, 1962) is now colonized largely by densely packed colonies of zoanthids (personal observation). Sites for settling of coral planulae are very limited and restoration of former conditions on most of this reef in the forseeable future seems unlikely.

When corals die their surfaces are rapidly colonized by algae (Moorhouse, 1936; Slack-Smith, 1960; Goreau, 1964; Banner, 1968; Chesher, 1969, 1970; Endean, 1969; Pearson and Endean, 1969; Stoddart, 1968; Weber, 1969; Fishelson, 1973b). Benthic algae are inimical to coral regrowth. They not only occupy substrate which might otherwise be used by settling coral planulae (Hedley, 1925), they also trap sediment and create shifting banks unsuitable for larval settlement or continued growth of adults (Wood-Jones, 1907; Moorhouse, 1936). In addition, they can overgrow young or injured coral colonies (Edmondson, 1928) and kill young coral colonies by current-induced abrasion (R.H. Randall, personal communication, 1973).

Subsequent to colonization by algae, dead coral surfaces may become colonized by encrusting animals including alcyonarians, hydroids, bivalves and zoanthids (Wood-Jones, 1907; Slack-Smith, 1960; Barnes, 1966; Banner, 1968; Chesher, 1970; Endean, 1969; Weber, 1969; Fishelson, 1973b). In Guam large numbers of the sea urchin *Echinometra mathaei* appeared and removed most of the growth of filamentous algae on coral recently killed by *Acanthaster planci* (R.H. Chesher, personal communication, 1973).

Although many faunal elements of the community disappear after corals are killed, the standing crop of herbivorous fishes may increase as a result of this increased food supply (Chesher, 1970). It has been suggested that the production of harvestable protein under such circumstances might equal or

exceed that of the original reef. However, my observations in Fiji and Enewetak in areas recently infested with *Acanthaster* indicate that an increase in herbivorous fish under such circumstances is not an invariable response. In these two areas herbivorous fishes were conspicuously *less* numerous than in adjacent areas with abundant living coral, despite a heavy overgrowth of algae on the dead coral skeletons. Perhaps the colonizing algae here were not acceptable as food. In areas where herbivorous fish populations increase after an *Acanthaster* invasion, it is likely that this is just a transient early stage of ecological succession on the recovering reef.

Rates of recovery of reef communities from damage differ widely and depend in part upon the extent of the damage and the species affected (e.g. Endean, 1971). If damage is extensive, complete recovery of the community appears often to require several to many decades (Stephenson et al., 1958; Chesher, 1970; Stoddart, 1968; Endean, 1971; Randall, 1973; Marsh and Gordon, 1974). Some of the larger coral heads killed by *Acanthaster* in Guam appeared to be over 200 years old (Goreau, in Chesher, 1970).

However Shinn (1972) has observed that recovery from extensive storm damage may require as little as five years in the case of the very rapidly growing staghorn coral *Acropora cervicornis*, in the Caribbean. In the two instances cited by Shinn "the majority of unburied fragments (of storm-damaged *A. cervicornis*) remained alive and within months began re-establishing discrete colonies". On the basis of his coral growth studies Shinn (1972) concludes: "complete recovery of a reef composed principally of head corals would require more than an average man's lifetime. On the other hand, a reef composed of staghorn corals has the capacity to regenerate in a matter of a few years."

Based on observations made in the wake of *Acanthaster* outbreaks on the Great Barrier Reef, Pearson (1974) states similarly that "it is conceivable that less than 10 years . . . large sections of reef could become almost completely covered by tabular species of *Acropora* and other fast growing species. However for a more diverse assemblage to develop it will be necessary for the slower growing species to compete successfully with them."

The recovery of coral communities in Hawaii after inundation by lava takes about twenty years in areas where exposure to sea and swell maintains coral communities in an early stage of ecological succession. In more sheltered areas where reefs are more nearly climax communities recovery is on the order of fifty years (Grigg and Maragos, 1974).

Recovery may be more rapid if only mobile organisms or organisms with shorter lifespans than corals are affected. For example, less than four years after the fish population was removed from a Bermuda patch reef recolonization was complete (J. Bardach, personal communication, 1974). In Hawaii recolonization of a patch reef from which all the fish were removed was well over half complete after 241 days (Wass, 1967). Doty (1973a) estimates that the establishment of climax populations of tropical benthic algae on under-

water lava flows takes five to ten years.

If, when corals are damaged, portions of living tissue remain, the tissue may regenerate and regrow over exposed skeleton (Fishelson, 1973b; Randall, 1973). Filamentous algae and invertebrate epifauna growing on the exposed skeleton may retard or completely inhibit this regrowth however. Fishelson (1973b) found that "bushy" forms with small polyps were most successful in encroaching upon the epibionts and repairing the breached surface.

Studies of recolonization of corals are of considerable interest, but to use coral cover or coral species diversity as simple indices of recovery of the reef *community*, as some have done, can be very misleading. Coral diversity on a recovering reef may rapidly reach, or even exceed, diversity in the original undamaged community. This is because large numbers of small colonies may develop on unoccupied substrate (Goreau et al., 1969; Grigg and Maragos, 1974).

Such colonies may rapidly occupy large areas of bottom, resulting in high figures for coral coverage. But full recolonization of the area by the many fish and invertebrates that live in, on, or in close proximity to corals, requires a greater volume, total surface area and complexity of coral architecture than these young colonies provide. These features of the coral colony are developed in rough proportion to colony height. Any index of community recovery which is based on corals alone could thus be improved if measurements of mean colony height were incorporated.

In addition it would be instructive to know how the original and present community compared in terms of similarity of coral (or other) species. Coral communities with similar diversity would nonetheless be quite distinct if species similarity were low. The use of similarity indices developed by plant sociologists would be appropriate here (e.g., Sorensen, 1948; Cole, 1949; Whittaker, 1967; Scheer, 1975; see also Chesher, Chapter 6).

CONCLUSIONS

The inadequacy of water column studies

An 18-month multidisciplinary study of pollution in Kaneohe Bay led to the publicly expressed opinion that it was not badly polluted and was "in exact balance" (Honolulu Star Bulletin, July 19, 1969). The results of this study, which was focussed mainly on water quality, did not hint at the disruption of the reefs described above. This error is instructive for it demonstrates two common shortcomings of pollution surveys.

First, the research was heavily weighted towards public health aspects of pollution. It is not always sufficiently recognized by pollution scientists with a public health background, that waters containing levels of pollutants which

do not threaten human health directly can be highly destructive to aquatic communities.

Secondly, no one surveyed or apparently even looked at bottom communities during the study. The continuous removal and replacement of water and its contents due to tidal flushing, and the high reproductive rates of planktonic organisms, makes water quality an inadequate indicator of pollution in shallow coastal waters (e.g. Chesher, Chapter 6). The bottom, in contrast, serves as a reservoir for pollutants, which accumulate there through settling and sorption. The impact of pollution is thus liable to be more profound in bottom communities. (See also Zieman and Wood, Chapter 5). For example, while reef communities were being destroyed by pollution in Kaneohe Bay, only "subtle" changes were observed initially in the plankton community (Clutter, 1972).

Bioassays

Pollution is likely to affect populations with rapid turnover times first. Thus benthic microorganisms should provide early warning signs of pollution in coral reef communities as they do elsewhere (e.g., Storrs et al., 1969; Chesher, Chapter 6). The increased activities of sulfate reducers in sediments exposed to sewage has been mentioned above.

A good biological index of pollution is one which is easy to measure, responds to pollutants rapidly, and leaves a permanent record of its response. Benthic foraminifers satisfy all three criteria. They are, furthermore, especially abundant in coral reef sediments (Nicholls, 1969). Seiglie (1968, 1971, 1974) describes changes in foram abundance and species composition due to pollution in Mayaguez Bay, Puerto Rico (see also Chesher, Chapter 6). Nicholls (1969) provides a useful general review of the relations between environmental conditions and foram distribution in estuaries.

Coral planulae are easily obtained and live well in aquaria. Since larval forms are often more susceptible than adults to environmental stress it might be thought that coral planulae constitute convenient and sensitive pollution bioassay organisms. However, coral planulae seem more tolerant than adults to extremes of salinity and temperature (Edmondson, 1946) and are unusually resistant to chlorine (Davis, 1971).

More useful, perhaps, would be very young corals obtained by allowing planulae to settle in laboratory tanks. The growth rates of such corals are very rapid and easy to measure (by counting polyps) and their small size and the large numbers easily obtainable (e.g. Harrigan, 1972) suggest they might prove ideal for bioassay work.

The growth and survival rates of transplanted corals have been used as environmental indices by Mayor (1924), Shinn (1966) and, more extensively, by Maragos (1972). Branching hermatypic corals are particularly useful because, as Shinn states, "they are extremely sensitive to environmental

changes; they grow rapidly; their shape makes distal growth easy to measure; and they do not swim away".

If one is to obtain a measure of the impact of pollutants in this manner it is necessary to use coral species which grow (or grew before the advent of pollution) naturally in the area. If reduced growth or survival rates are observed at sites where the transplanted species never grew, these effects cannot be attributed to pollution. Shinn (1966) observed growth rates of less than one-half the control values when he transplanted *Acropora cervicornis* to unpolluted areas where it did not normally grow.

Both Shinn (1966) and Maragos (1972) noted that growth was not only slower in corals transplanted to areas of suboptimal environmental conditions, but also more erratic than in control areas. Of the six Hawaiian species used by Maragos (1972) in transplant studies, *Fungia scutaria* was the most sensitive in terms of survival time.

The use of small transplanted communities for pollution research is being pioneered by coral reef ecologists at the Hawaii Institute of Marine Biology. Coral communities are transplanted to tanks about 1.5 m^2 in area and about 0.5 m deep, containing running seawater. The salinity and temperature of seawater are automatically controlled. In addition to measuring structural responses of these sub-communities to stress, it is possible to monitor metabolic responses by measuring differences in concentrations of oxygen, nutrients, etc., between inflow and outflow water.

This approach helps fill a major gap between single-species laboratory studies and field studies on whole communities. While laboratory studies facilitate refined control of physical environmental variables, one cannot predict community responses with much confidence on the basis of the responses of single-species components isolated in the laboratory; communities are more than the sum of their component species (e.g. Odum, 1971). Field studies, while providing information about real ecosystems, do not provide adequate opportunity to monitor the system continuously or to manipulate important physical and biological variables. In addition, the detailed prior environmental history of a community is seldom available in field studies. This shortcoming can be overcome in tank communities by acclimation prior to manipulation. The major limitations of such studies are the size (and therefore the complexity) of the community that can be transported and housed, and the lack of normal interaction with adjacent communities.

The use of measurements of species diversity as pollution indices often proves a snare and a delusion. Hedgepeth (1973) puts the problem in perspective: "The search for simple approaches and magic numbers that may be obtained with relative ease has led some pragmatic ecologists down the primrose path of theoretical ecology. Perhaps unaware of the highly speculative basis for such concepts as diversity and stability, they have oversimplified the idea of diversity indices to suggest that a high diversity index is ipso

facto indication of the well being or more natural condition of a biocoenosis and consequently the number can be interpreted as an abstract value of the state of pollution."

Retrospective pollution analysis

At least some reef corals contain daily, seasonal and annual growth bands in their skeletons. By using autoradiography or X-radiography of sections of coral skeletons it is possible to measure the width of these bands and construct a diary of the past growth history of the coral. Moore and Krishnaswami (1974) state: "If we accept the rather compelling evidence that . . . banding in corals is annual and that specific bands may be assigned definite ages . . . then corals become vast storehouses of information about chemical and physical changes of waters in which they grow. Recorded in every coral head is a record of changes in fallout isotopes (e.g., ^{14}C, ^{90}Sr), uranium series isotopes (e.g., ^{238}U, ^{226}Ra, ^{228}Ra, ^{227}Ac, and ^{210}Pb), stable isotopes (e.g., ^{13}C, ^{18}O), trace metals, and clay particles as well as significant data on small scale environmental fluctuations."

These techniques appear to have considerable potential for correlating coral growth rates retrospectively with pollution of known temporal extent, just as tree ring "calendars" are used to retrieve paleoclimatic information. For example, coral autoradiography has been used to reconstruct changes in the relative concentrations of bomb-produced isotopes in surrounding waters (e.g., Knutson et al., 1972; Knutson and Buddemeier, 1973; Moore and Krishnaswami, 1974).

Moore and Krishnaswami (1974) found a hiatus in coral growth which corresponded temporally with the operation of a cement plant nearby. They suggest that pollution associated with this plant may have caused the observed cessation of growth. Dodge et al. (1974), using X-radiography to measure past growth rates of coral in Jamaica, found an inverse trend between rates of resuspension of sediments and coral growth.

Restocking damaged reefs

The "restocking" of damaged reefs with corals has been suggested by various writers. Hubbard (1974) points out that species must be chosen which can tolerate the currents in the area in question. She provides information on polypal tolerance to currents of eighteen species of Caribbean corals.

Fast-growing species and ones that dominate early successional stages of reef communities seem like logical choices for restocking programs. Various species of *Acropora* and *Montipora* (Vaughan and Wells, 1943; Lewis et al., 1968; Shinn, 1972) have these qualities. These species are also among those which rapidly cement themselves to the bottom and establish new colonies when fragmented and strewn across the bottom (Shinn, 1972; personal ob-

servations). *Porites lobata* has similar recuperative powers (personal observations). Since reseeding any substantial area with adult corals is quite impractical, this ability is of considerable value.

Importing consultants

Frequently foreign consultants are called in, often for just a few days or weeks, with the expectation that they can help solve environmental problems in the tropics. Having played this role several times, I would like to implore those who ask in outsiders to do their homework first. Sometimes the consultant is forced to spend his entire time canvassing various laboratories and agencies to find out what is already known locally about the problem at hand. This job could be done much more easily in many instances by local scientists familiar with the language and with the appropriate people to be interviewed and the relevant reports to be located. This should be done *before the consultant arrives*. Otherwise he has little opportunity to make observations of his own and finds it very difficult to make a contribution consonant with the cost of bringing him in.

Pollution control

Effective policing of reefs in order to deter the individual polluter is frequently out of the question. The reefs around Truk, for example, are so extensive that dynamite has been used to kill reef fishes with little concern that the authorities will hear about it (Fuchs, 1968). The widespread use of poisons in reef communities is also carried out with little fear of detection by authorities. In many tropical areas, as Storer (1967) points out for the Philippines, the effort to stop such practices "has occupied a disproportionate amount of time and personnel of both municipal and federal officials. Such efforts are, of course, largely futile. There is no incentive to prosecute; the offender cannot pay a fine and if committed to jail, the burden of supporting his wife and children is merely shifted to the government. Furthermore, such efforts to control the problem are easily thwarted." Public censure is thus probably the only effective form of pollution abatement in cases of individual abuse of reef communities, but it is rarely operative in an ecologically unsophisticated public.

Although the corporate polluter is easier to detect, laws to prevent such pollution are generally inadequate and frequently non-existent in the tropics. Too often the public, including even that segment whose livelihood depends directly upon the reefs, does not realize the problem exists. The enactment and effective enforcement of appropriate laws, and the public censure of polluters can only be brought about when the public is made aware of their value. The most important step in deterring both the corporate and the individual polluter is thus education.

There are numerous examples of customs which served the purposes of conservation in pre-industrial cultures. Falanruw (1971) points out that "inasmuch as former conservation-valued practices were in the guise of religion, social stratification, taboos, etc., the young people of Micronesia today are not individually conservation minded". This is undoubtedly also true elsewhere, and is one of several reasons why the teaching of conservation becomes increasingly imperative as the seemingly inevitable westernization of these cultures proceeds.

Merton (1967) notes: "the teaching of ecology in the tropics and subtropics is not only of absorbing interest, but has a social relevance which is difficult to underestimate. I should like to see, therefore, a course in ecology as a necessary requirement for any degree or teacher's certificate in such countries." Since general biology—ecology texts for schools in tropical areas do not exist (publishers take note!), many tropical school systems have recently prepared their own in booklet form.

Teachers in the Caribbean and the tropical Pacific Islands often come from temperate regions and have little knowledge of local fauna and flora. The Bermuda Department of Education recognized this problem and took steps to remedy it. They engaged a number of foreign and local scientists to give to the islands' biology teachers (most of whom were trained in England) a series of lectures on the ecology of Bermuda, supplemented by field trips. The information was enthusiastically received by the teachers. It is now being reflected in their curricula and will undoubtedly lead to an increased concern of Bermudians for their environment. The program costs little and is a good example for others to follow.

Most coral reefs are situated in developing countries with very limited conservation and research funds. The allocation of money for coral reef research and management is therefore very small in relation to the importance of these communities to man, their vulnerability to pollution, and the amounts spent studying some less valuable but more visible environments in industrialized countries. New or accelerated programs of research on some of the problems discussed above are obviously needed.

But environmental crises develop faster than they can be completely assessed. In this context it is more important to make interim decisions *in time* than to make more scientifically satisfying decisions later. Pollution biologists sometimes inject too much traditional laboratory caution into matters of immediate practical concern. They hesitate because their information is not final. But information is never final in science, which proceeds by disproof rather than by proof.

The fact that biologists have sometimes erred in their warnings about environmental degradation is no justification for abdicating the responsibility to speak up — any more than the fact that scientists sometimes publish mistakes means we should all abandon publishing. Those who remain silent when their observations point to environmental decay are the undertakers of the environment; environmental post mortems become their stock and trade. "They measure and we weep."

CHAPTER 4

63

TROPICAL SEA GRASS ECOSYSTEMS AND POLLUTION

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TROPICAL SEA GRASSES

Until recently, studies in the nearshore tropical environment have largely concentrated on the spectacular and complex coral reefs, and the dense, productive mangrove forests. However as marine science focuses on the living resources of the shallow, coastal seas, the vast beds of sea grasses bordering the tropical coastlines are being recognized as important resources. The significance of sea grasses in coastal ecosystems was documented early in the classic studies of Petersen (1918), but little further work on the ecological importance of these highly productive and easily studied areas followed until recently.

Sea grasses are unique among submerged marine and estuarine plants due to the presence of an extensive root and rhizome system. Because of this system — high growth rates, and dense leaf development — sea grasses exert considerable influence over their environment. Wood et al. (1969) summarized the functions of sea grasses in estuarine and marine ecosystems as follows: (1) sea grasses have a rapid rate of growth and high organic productivity; (2) while alive, they act as food for only a limited number of organisms since the demise of the large sea turtles, but provide large quantities of detrital material which, along with its resident microbes, provide a major food source for the estuarine ecosystem; (3) the leaves support large numbers of epiphytic organisms, which under favorable conditions, may be comparable in biomass to the sea grass leaf weight, and which are grazed extensively; (4) sea grasses provide organic matter which initiates sulfate reduction and maintains an active sulfur cycle in the estuarine sediments; (5) the dense leaves reduce the current velocity near the sediment surface and promote sedimentation of organic and inorganic particles; and (6) the roots and rhizomes bind the sediments and hinder erosion of the surface of the sediment.

Sea grasses are widely distributed throughout the shallow coastal seas from the Arctic to the southern tips of Africa and New Zealand. Den Hartog (1969) recognizes 12 genera of sea grasses with 49 species, of which 7 are tropical in distribution. In the Hydrocharitaceae, *Enhalus*, *Thalassia*, and *Halophila* are tropical, and in the Cymodoceoiceae, *Halodule*, *Cymodocea*,

Syringodium, and *Thalassodendron* of the sub-family Cymodoceoideae are basically tropical genera, although several of the above genera have some species which extend into subtropical or warm temperate seas. As is common with many marine groups, the highest diversity in the sea grasses is found in the Indo-West Pacific, with all seven genera represented. By contrast the Caribbean has only *Halodule*, *Syringodium*, *Thalassia*, and *Halophila*. Only *Halodule wrightii* and *Halophila decipiens* are common to both areas.

Petersen (1918) was the first to recognize the overall importance of sea grasses in the coastal ecosystem, and in recent years it has become apparent that sea grasses contribute significantly to the maintenance of the coastal ecosystem of the tropics and near tropics in numerous ways (Wood et al., 1969; Zieman, 1974a).

Because sea grass beds attracted little attention scientifically until the late 1960's, there are large gaps in our knowledge of these systems. The best studied of the tropical sea grasses is the turtle grass, *Thalassia testudinum* Konig, of the Caribbean. Although this study is directed to tropical sea grasses, nearly all of the information is drawn from *Thalassia testudinum*, as there is unquestionably far more published material dealing with this species than with all other tropical sea grasses combined. As research progresses, undoubtedly differences in the structure and function of different grass beds will be found. However, judging from the experience of this author and other workers in the field, the similarities between different sea grass communities are more striking than the differences.

Much attention has been drawn to the importance of sea grass beds as nurseries, important in the rearing and sheltering of small fishes and invertebrates (Hutton et al., 1956; Moore, 1958; Tabb et al., 1962; Hoese and Jones, 1963; Wood et al., 1969; Jackson, 1972; Odum et al., 1973). Prominent among the organisms inhabiting the grass beds are commercially important species such as the pink shrimp *Penaeus duorarum* (Hoese and Jones, 1963), mullet (Wood et al., 1969), sea trout, and stone crab (Tabb et al., 1962).

Whereas some animals have been recorded as feeding directly on turtle grass (Randall, 1965a, 1967; Ogden, 1972), since the decimation of the green sea turtle by overfishing there has been little direct grazing pressure on these beds except in certain localized situations. Most of the green leaves remain untouched until they die and are cast off from the plant, when they decay and are converted to detritus (Fenchel, 1970; Odum et al., 1973; Zieman, 1974a). In *Thalassia* beds, at least 90% or better of the energy makes its way to the higher trophic levels via the detritus food chain. Some of this detritus may be transported offshore to provide food far from the site of growth (Menzies et al., 1967; Menzies and Rowe, 1969).

Tropical sea grass beds are frequently found in the vicinity of coral reefs and the interactions between these two communities appear important. Particularly at night, fish, including valuable commercial species, leave the

reef to forage on species living in the grass beds (e.g. Earle, 1971). Populations of fishes living in reef communities are larger where the reefs are adjacent to seagrass beds (Randall, 1965; Starck and Davis, 1966; Ogden, 1972).

Tropical sea grass beds are among the most productive of natural ecosystems. Productivities of 1,300 to about 3,000 g dry weight/m per year have been recorded (Odum et al., 1973; Zieman, 1975).

Grass beds also stabilize both organic (Wood et al., 1969) and inorganic sediments (Ginsburg and Lowenstam, 1958; Scoffin, 1970; Zieman, 1972) and offer a substrate for epiphytic flora and fauna. Scoffin (1970) observed that in dense, long-bladed beds of *Thalassia testudinum* currents up to 40 cm/sec measured immediately above the blades are reduced to zero at the sediment—water interface.

Extremely high rates of nitrogen fixation have been reported by Goering and Parker (1972) by the epiphytes of *Thalassia testudinum* and by Patriquin and Knowles (1972) by bacteria in association with the roots. This implies that sea grass communities may be important sources of newly fixed nitrogen in the characteristically nitrogen-poor waters of the coastal tropics. McRoy et al. (1973) found much lower values of nitrogen fixation than the previous authors however, and the importance of this process in sea grass beds cannot be determined without considerably more research on different species in different areas.

Thalassia has been used as a feed supplement for sheep (Bauersfeld et al., 1969) and chickens (Alvarez, personal communication, 1973) and as fertilizer (Van Breedveld, 1966).

Sea grass beds are especially vulnerable to degradation by human-induced stresses as they inhibit only the shallow marginal seas where the activities of man are the greatest. Their mode of growth is unique for marine plants, and they are subject to damage from stresses or pollutants both in the water column and in the sediments.

POLLUTANTS AND STRESSES OF TROPICAL SEA GRASSES

In the tropics, there are only a few scattered examples of pollution studies in sea grass communities to draw from, and much of the information is in the form of difficult-to-obtain reports. Thus much of the discussion here will, of necessity, be drawn from these studies.

Dredging and filling

Of all forms of man-made or induced disturbances of the estuarine and nearshore environments, dredging and filling presents the greatest potential for damage to the sea grass beds and has undoubtedly caused the destruction

of more desirable grass bed habitat than any other form of pollution. Dredge and fill operations, which remove or dredge material from one portion of an estuary to another area to be filled, damage sea grasses in a variety of ways.

The most obvious destruction comes from the physical removal of the dredged material whether by a hydraulic dredge or by dragline and the covering of other marine areas with the resulting fill material. These two joint processes cause the destruction of sea grass beds for an indeterminate, but often considerable length of time, as the resulting sediments are not initially suitable for colonization by sea grasses. In the Caribbean region, large areas of the highly productive turtle grass, *Thalassia testudinum*, have not recovered many years following dredging operations.

In addition to the direct physical damage, secondary effects can be pronounced. Sea grasses generally require high light levels to sustain populations, and the available light is greatly decreased in the vicinity of dredge operations due to the turbidity caused by suspended sediments, despite protective devices designed to contain the suspended material such as turbidity curtains or screens. The fauna associated with grass beds are highly susceptible to damage both from the effects of direct siltation and suffocation, as well as from low oxygen levels due to the high oxygen demand of the decomposing suspended organic matter.

An area where extensive dredging and filling operations have been used to create waterfront homesites is Boca Ciega Bay, Florida. Here Taylor and Saloman (1968), found that between 1950 and 1968, 1,400 acres (ca. 567 ha) of the bay had been filled, reducing its area by 20%. Sediments in undisturbed areas of the bay were firm and averaged 94% sand and shell and supported luxuriant beds of sea grass, primarily turtle grass. In contrast, the sediments in dredged canals averaged 92% silt and clay. Light penetration was highest in the southern bay away from the land fills, lowest in the bay near fill areas, and increased again in the dredged canals. The canals, however, were sufficiently deep that adequate light for sea grass photosynthesis did not reach the bottom.

The soft sediment of the canals is generally unsuitable for most macro fauna as well as for sea grasses. A variety of studies (Phillips, 1960; Springer and Woodburn, 1960; Taylor and Saloman, 1968) listed nearly 700 species of marine plants and animals from Boca Ciega Bay. Taylor and Saloman (1968) found the deeply dredged canals contained less than 20% of the species recorded from the bay and these remaining species were nearly all fishes which were highly motile and not restricted to the canals. Thus, in ten years following the disturbance of the bay bottom recolonization of the sediments by plants and animals was negligible. Using conservative and incomplete figures, Taylor and Saloman (1968) estimated an annual loss in fisheries and water sports of about $1.4 million per year from the bayfill operations.

Godcharles (1971) studied the effects of a commercial hydraulic clam

dredge on the benthic communities in the estuarine areas of Tampa and Tarpon Springs, Florida. He found the hydraulic dredge sampled clams effectively, but uprooted or severed the rhizomes of all sea grasses in its path. The dredged areas filled with fine-grained sediment, and at one station some recovery of *Halodule beaudettei* (*Diplanthera wrightii*) occurred, but no regrowth of *Thalassia testudinum* or *Syringodium filiforme* was ever observed.

Odum (1963) found that the chlorophyll *a* and productivity of turtle grass beds diminished following dredging operations in Redfish Bay, Texas. The year following dredging operations, chlorophyll *a* and productivity increased except where the communities had been covered with soft silt.

In addition, Odum reported that the spill from dredging destroyed grass beds when discharged into areas only 0.5 m deep as it caused the bottom to shoal critically to a point where plants could no longer survive. Odum attributed the increased productivity to the release of nutrients from the displaced sediments.

In Lindbergh Bay, St. Thomas, Virgin Islands, increased turbidity brought about by dredging resulted in a marked reduction in the distribution and abundance of benthic sea grasses (Van Eepoel et al., 1971). In 1968, "benthic grasses could be easily seen from the surface in water 10 meters in depth; the *Thalassia* community was lush, covering the major portion of the bay bottom". By 1971 *Thalassia* along with *Syringodium*, the second most common sea grass in the bay, had become extremely sparse throughout, with "only very few plants being found deeper than 2.5 meters. In the most turbid portions of the bay they were limited to 1.5 meters in depth." Similar destruction of seagrass beds by sedimentation and turbidity caused by dredging in Brewers Bay, St. Thomas, is described by Grigg et al. (1971). The light compensation point for *Thalassia testudinum* in clear waters is reported to be about 40 m (Pomeroy, 1960; Marmelstein et al., 1968; Gessner, 1971), although this figure may not be reliable due to oxygen storage problems.

Eutrophication

The input of sewage or other nutrient-enriched substances into water near sea grass beds produces a number of effects. Sea grasses may have the ability to take up nutrients through the leaves as well as through the roots, and moderate amounts of excess nutrients could stimulate growth. Enrichment also fosters the development of epiphytic algae and phytoplankton. One of the most luxuriant growths of turtle grass and assorted epiphytic algae this author has observed was just inshore of the outfall of the Miami sewage plant on Virginia Key, Florida. Sea grasses are known to have a high light requirement, and the excessive growth of epiphytic algae and phytoplankton decrease the light energy available to the plants.

Taylor et al. (1973) found that significant portions of Hillsborough Bay, Florida, had been damaged by dredging and domestic and industrial efflu-

ents, particularly phosphorus and nitrogen compounds and suspended solids. This resulted in heavy growths of phytoplankton and filamentous algae, marked fluctuations in oxygen concentration, and high turbidity. This resulted in low light transmission which reduced the density and coverage of the sea grass beds, primarily *Thalassia*. Catches of fish and crustaceans were low compared with other areas where environmental conditions were more natural and extensive beds of sea grasses persisted.

Concomitant with increasing sewage pollution and dredging in Christiansted Harbor, St. Croix, Virgin Islands, extensive beds of *Thalassia testudinum* have been overgrown and replaced by *Enteromorpha* (Dong et al., 1972). Over a 17-year period Nichols et al. (1972) documented a 66% reduction in grass beds in the bay.

McNulty (1970) found a lack of sea grasses in an area polluted by sewage effluents in Biscayne Bay, Florida. Only *Halophila decipiens* and *Halodule beaudettei* (*Diplanthera wrightii*) existed in low quantities within 1 km of the outfall. Past this point *Thalassia testudinum* began to occur in sparse quantities. Four years after the abatement of the sewage input the sea grasses had actually declined further, but the area had continued to be disturbed by heavy hydraulic dredging and the resultant increased turbidity, to which sea grasses are especially susceptible.

Sewage effluents (and suspended sediments) reduce dissolved oxygen levels. Hammer (1972a) reports that anaerobiosis depresses photosynthesis in *Thalassia testudinum* and *Halophila decipiens*. After 24 hours exposure to anaerobic conditions *Thalassia* recovered fully while *Halophila* did not. The greater tolerance of *Thalassia* is probably related to its greater ability to store oxygen in its tissues.

Temperature and salinity

Raising the temperature regime of tropical sea grasses can have far greater effects than a similar modification in a temperate environment since organisms in tropical waters live much closer to their thermal death point (Biebl, 1962; see also Zieman and Wood, Chapter 5).

Phillips (1960) reported that *Thalassia testudinum* preferred 20—30°C temperatures in south Florida. Zieman (1975) found that this species has an optimum temperature of between 28 and 30°C and net photosynthesis sharply declines on either side of this range.

Glynn (1968) observed that the leaves of *Thalassia* were killed in Puerto Rico by temperatures of 35—40°C, but that the rhizomes apparently were not affected.

Thorhaug and Stearns (1972) reported that *Thalassia* growing at artificially elevated temperatures produced flowers but no observed fruit. They also produced more new shoots than plants growing at normal temperatures.

Halodule beaudettei (*Diplanthera wrightii*) is an earlier successional

species than *Thalassia* in the Caribbean. It is usually the first sea grass colonizer in disturbed areas, as it is able to establish and spread rapidly.

Phillips (1960) found *Halodule beaudettei* to prefer temperatures of 20—30° C, as does *Thalassia*, but that *Halodule* is a more eurythermal tropical species. Manatee grass, *Syringodium filiforme*, is generally less abundant than *Thalassia* or *Halodule*, and its ecological position is less well defined. Phillips (1960) considers *Syringodium* a tropical plant but not a stenothermal one, although leaves of *Syringodium* begin to die at temperatures below 20° C.

Zieman (1970) found that sea grasses tend to be more resistant to thermal pollution stress than do the algae. The sea grasses tend to be more resistant to periodic acute temperature increases, and even if the leaves are killed off, rapid regeneration is often experienced from the horizontal rhizomes which lie several centimeters below the surface of the sediments. This regeneration occurs in various species because the rhizomes contain large starch reserves and are injured only if the temperature of the sediment reaches lethal levels for extensive periods of time. Phillips (1960) however, points out that frequent removal of leaves may so debilitate rhizomes that the capacity for recovery is impaired.

There is considerable variation in the observed salinity ranges of the Caribbean sea grasses. *Thalassia* has been found to tolerate salinities as low as 3.5‰ (Sculthorpe, 1967) and as high as 60‰ (McMillan and Moseley, 1967) for short periods of time, but usually undergoes defoliation at these extremes. Optimum salinities for *Thalassia* are in the range of 24—35‰ (Phillips 1960; McMillan and Moseley, 1967; Zieman, 1975). McMillan and Moseley (1967) found that *Halodule beaudettei* (*Diplanthera wrightii*) showed the greatest resistance to high-salinity conditions, *Thalassia* was intermediate, and *Syringodium* was the least resistant. This agreed with the observations of Phillips (1960) who found *Halodule* existing in freshwater for short periods of time. He found *Halodule* to be the most euryhaline species, but that it was outcompeted by *Thalassia* and *Syringodium* whenever they were present.

Chesher (Chapter 6) reports that, although the effluent of heated brine from the Key West desalination plant caused a marked reduction in biotic diversity in the vicinity of the discharge, nearby beds of *Thalassia* were unaffected.

Halodule beaudettei is more euryhaline than *Syringodium filiforme*, and tolerates salinities of 9—52.5‰, whereas *Syringodium* only survives to 35‰ (McMahan, 1968).

Hammer (1968) found the photosynthesis of sea grasses, including *Thalassia* and *Syringodium*, to reach a maximum in full strength seawater, and to decrease as the salinity decreased. Hammer ascribed part of the decrease to direct salinity effects, but pointed out that, with dilution, the available carbon supply was diluted and may also have become limiting.

Zieman (1970, 1974b) found that the seasonality of *Thalassia* could be ascribed largely to temperature and salinity effects. When temperature and salinity vary together, the combination of high temperature and low salinities was found to cause the greatest decline in the plant populations.

Tabb et al. (1962) state: "Most of the effects of man-made changes on plant and animal populations in Florida estuaries (and in many particulars in estuaries in adjacent regions of the Gulf of Mexico and South Atlantic) are a result of alterations in salinity and turbidity. . . High salinities (30—40 ppt) favor the survival of certain species like sea trout, redfish and other marine fishes, and therefore improve angling for these species. On the other hand these higher salinities reduce survival of the young stages of such important species as commercial penaeid shrimp, menhaden, oysters and others. It seems clear that the balance favors the low to moderate salinity situation over the high salinity. Therefore, control in southern estuaries should be in the direction of maintaining the supply of sufficient quantities of fresh water which would result in estuarine salinities of 18 to 30 ppt."

Wind and sea stress

Sea grasses occurring in the tropical marine environment are periodically subject to severe stress from tropical hurricanes and typhoons. Although man cannot generate the extreme winds and resultant wave stress on the beds as a major tropical cyclonic storm, man can influence the manner in which the waves interact with sea grasses by means of his modification of the nearshore sea bottom.

Of all marine communities which are affected by the storm-generated winds, subtidal sea grass beds appear to be among the least susceptible to severe damage by the storms.

Thomas et al. (1961) found that following Hurricane Donna in 1960, large quantities of *Thalassia testudinum* washed onshore. However, there was little quantifiable loss within the beds, although Tabb and Jones (1962) observed heavy mortality of fishes and invertebrates from the same storm in North Florida Bay. Mortalities were caused by mud suffocation, by stranding as the water retreated, and by severe oxygen depletion due to decomposition of suspended organic matter.

In Texas, Oppenheimer (1963) found that inshore turtle grass flats and bottom flora were undamaged, although major erosion occurred as Hurricane Carla passed near Port Aransas, Texas, in 1961.

Glynn et al. (1964) recorded that following Hurricane Edith, a relatively mild storm in September 1963, *Thalassia* beds were not noticeably damaged, although mangroves and beds of *Syringodium* suffered some effects and the coral reef community was severely disturbed.

Population explosions and overgrazing

Overgrazing of sea grasses occurs commonly in the immediate vicinity of patch reefs, leading to the formation of bare "halos" surrounding Caribbean patch reefs. These halos have been attributed to grazing reef fish, primarily surgeon fish (Acanthuridae) and parrot fish (Scaridae) (Randall, 1965) and more recently to the echinoid *Diadema antillarum* (Ogden, 1972). The grazing is localized and presents no threat to the overall grass beds, as the small grazers generally do not venture far from the shelter of the nearby patch reef (Mathison et al., 1971).

However, Camp et al. (1973) have observed dense concentrations of the regular urchin *Lythechinus variegatus* causing massive destruction of mixed sea grass beds, primarily composed of *Thalassia*, on the west coast of Florida. During these outbreaks, areas hundreds of meters long and meters across are completely denuded by the migrations of these urchins. It is doubtful if these outbreaks are of severe consequence compared to the grazing pressure exerted formerly during seasonal migrations of large numbers of sea turtles.

The cause for urchin outbreaks has not been well studied. It is possible that it is due simply to exceptionally successful spawning and recruitment. However, there is the possibility that these outbreaks are due to the reduced predation of the urchins in areas where the normal predators have been overfished.

Oil pollution

As sea grasses are generally subtidal, they are less susceptible to damage from oil than other organisms, especially those in the intertidal zone. Still, shallow beds of sea grasses can suffer destruction from severe oil spills. Diaz-Piferrer (1962) found beds of *Thalassia testudinum* were badly affected by crude oil on the south shore of Puerto Rico. The turtle grass deteriorated over a period of several months, and the normal algal flora was denuded and replaced by blue-green algae. Following the much publicized Santa Barbara oil spill in January 1969, *Phyllospadix torreyi* growing in the intertidal zone was severely damaged (Foster et al., 1971), although subtidal plants and those in the extreme low intertidal zone were largely undamaged as they were, to some degree, protected from direct contact with the oil.

In addition to direct mortality from the contact of the plants with the oil, a second and possibly longer-term damage is caused by the interaction of the sediment and oil. Sea grasses tend to stabilize the fine-grained sediments, but as oil seeps into sediment, it agglomerates into large lumps which offer a greater purchase for waves and currents to act on, and increase the buoyancy of the sediments, thus making removal of the sediments easier. Diaz-Piferrer (1962) recorded the loss of 3,000 m^3 of sand from Tamarindo Beach in less than a week due to this effect.

Metals

Thus far metal pollution has not been implicated in the decline or destruction of marine grass beds. As many sea grasses require a reducing environment around the roots (Wood et al., 1969) but an oxidized environment surrounding the leaves, they can pick up metals in a variety of forms.

Many types of metals are found in abundance in polluted sediments where they may become easily mobilized and be taken up by the roots of sea grasses. Parker et al. (1963) found the highest concentrations of cobalt and manganese in Texas bays in sea grasses, the concentrations being over an order of magnitude greater than the concentration in the sediments. Iron was in nearly the same concentration in sea grasses as the sediments, but was more concentrated in sea grasses than in any other plant or animal.

Parker (1962) found the zinc concentration of *Thalassia* and *Halodule beaudetti* (*Diplanthera wrightii*) was 5—10 times the concentration in the sediments. Thus, there is the possibility that excess metals in estuaries will be concentrated in sea grasses and available for passing up the food chain. This is of increasing concern as metals are released into the environment in large quantities from nuclear power plants (e.g. Segar and Pellenbarg, 1973) and desalination plants (e.g. Chesher, Chapter 6), both of which are being constructed at accelerating rates in the tropical inshore regions.

Physical disturbance and recolonization of sea grasses

Physical disturbance of grass beds due to boat traffic on shallow beds has been observed by students of sea grass ecology for quite some time (U.S. Dep. of Int., 1973), although there has been no systematic study of the effects of propeller cuts on grass beds. Phillips (1960) states that "in shallow bays perhaps the most common means of breaking the continuity of growth is the destruction caused by boat propellers". Phillips removed *Thalassia* rhizomes and shoots from two plots in April and May of 1958 in Boca Ciega Bay, and observed no regrowth into the areas after ten months, although the more rapidly growing *Halodule beaudetti* (*Diplanthera wrightii*) recolonized the denuded plots.

Scoffin (1970) also found that *Thalassia* does not spread its rhizome mat rapidly. In a 4-month experiment, he found trenches dug in the sediment were not recolonized.

In attempting to establish the rate of vegetative growth of *Thalassia*, Jones (1968) cut two trenches in grass beds in Bear Cut, Biscayne Bay, Florida, in August 1965. Re-excavation two years later, in August 1967, showed little regrowth into the 1 m × 30 cm × 20 cm deep cuts, although the trenches had filled with sand within a month of cutting. Jones also noted that *Halodule beaudetti* (*Diplanthera wrightii*) had recolonized the upper 2 cm of sediment with 270 cm of rhizome and about 20 growing rhizome tips.

Having heard from local fishermen that propeller cuts were persistent features and sometimes useful in relocating fishing holes and navigation channels, this author used boat tracks in grass beds as station location markers, in areas where it was impractical or imprudent to use marker buoys. Several of these cuts were used reliably for two years (1966—1968) in the Soldier Key region of Biscayne Bay and at least one was recognizable after three years (1966—1969).

Fuss and Kelly (1969) and Kelly et al. (1971) studied the survival of transplanted *Thalassia* and the feasibility of recolonizing turtle grass in disturbed areas. They found that sprigs of *Thalassia* require considerable time to initiate growth after disturbance. Fuss and Kelly (1969) found that ten months was required for new rhizome apices to develop, and Kelly et al. found that in a control bed 40% of the transplants had initiated new rhizome growth after thirteen months, whereas in an area with altered sediments only 15—18% of the transplants exhibited new growth.

If only the leaves of *Thalassia* are killed, or removed, regrowth is often rapid because of the starch reserves in the rhizomes. Too frequent leaf destruction will use up these reserves however. Taylor et al. (1973) found that *Thalassia* in Tampa Bay, Florida, tolerated harvesting twice a year. They speculate that this species might tolerate three harvests per year in more tropical areas.

Thorhaug (1974) has reported success in establishing *Thalassia* from seeds in south Florida.

Patriquin (1972) studied the nutrient sources and requirements of *Thalassia* in Barbados, and found that a reduced sediment layer is essential for good growth of turtle grass. Wood (1965) also noted this requirement and found that sea grasses stimulate the development of reducing conditions in the root layer.

Zieman (1974a) found a reduction of the finer components of the sediments in the disturbed *Thalassia* beds due to the lack of the stabilizing effect of the grass blades. This effect is even more pronounced in areas subject to continual boat traffic such as scoured ruts which are used by boaters as channels through the shallow beds and banks. Zieman also found a reduction in the pH and generally in the Eh of the disturbed sediments, yielding a more reducing environment. He suggested that *Thalassia* beds did not recover rapidly from physical disturbances such as boat traffic or dredging because: (1) the disturbed areas, whether sedimented in or scoured open, are not a suitable environment for rhizome growth; (2) when damaged, rhizomes normally require one year just to begin developing active new apices; and (3) rhizome apices, whether arising new from damaged *Thalassia* plants or simply growing from other portions of the grass bed, do not grow rapidly. The particular circumstances at each site determine which of these mechanisms is most important.

Recolonization of disturbed *Halodule beaudetti* (*Diplanthera wrightii*),

shoal grass, is not likely to be a serious problem. This plant does not have a well-developed deep rhizome system, grows well from seeds, and is capable of colonizing a damaged area in a short time. It does not seem as sensitive to sedimentation as *Thalassia*. The author has seen large mats of shoal grass attached to the substrate at only one edge, with the rest of the mat growing floating in the water column with roots and rhizomes unattached. *Halodule* can colonize an area in but a few weeks when conditions are correct, and exists as an annual plant in many estuarine areas. Once *Thalassia* becomes established, however, *Halodule* cannot compete, and is overgrown in all but the upper intertidal zones, or in areas in which turtle grass cannot become established. Den Hartog (1969) reported that in the tropical Indo-Pacific *H. uninervis* and *H. pinifolia* are the first sea grasses to appear following the disturbance of the sea bottom, and that they are gradually replaced by other species.

CONCLUSIONS

As knowledge of the structure and function of sea grass beds increases, it becomes evident that these areas are important components of the nearshore tropical marine ecosystem. A variety of studies have shown how grass beds act as nurseries, food producers, and stabilizers of the sea floor. As increasing development in the tropics begins to encroach upon these valuable resources, careful attention must be given to their fate, as many coastal people depend upon the resources of tropical grass beds for their food and sustenance, from the conch-fishermen of the Bahamas to the turtlers of the western Caribbean. In more developed areas, the grass beds are sources of sport-fishing and other forms of recreation.

Fortunately, recognition of the importance of grass beds to the ecology and ultimately to the economy is beginning to be recognized. The State of Florida, through its Coastal Coordinating Council, has declared grass beds to be preservation areas where no development is permitted. This is in part due to the recognition of the linkage between ecology and economy. An area that is dependent on its income from recreation, and primarily water-oriented recreation, must maintain high water quality standards and esthetic values.

Let us hope that other responsible decisions will result in the continued maintenance of the nearshore tropical marine environment.

CHAPTER 5

EFFECTS OF THERMAL POLLUTION ON TROPICAL-TYPE ESTUARIES, WITH EMPHASIS ON BISCAYNE BAY, FLORIDA

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GENERAL CONSIDERATIONS

Tropical estuarine regions are beginning to undergo the rapid development and industrialization which has damaged estuarine and neritic ecosystems of the temperate zone. Nearly all thermal pollution studies have been made in temperate waters and the body of quantitative information on the ecology of tropical marine and estuarine environments is still quite small.

The problem of thermal additions to the tropical oceans is similar qualitatively to that in temperate water. Any additions are potentially more harmful in the tropics, however, since tropical organisms live closer to their upper thermal limits. Usually, even in the tropics, the effluent will be warmer than the surface ocean water, and, if the intake is from an estuary, the salinity will normally be lower, so the heated water will tend to lie on the surface until mixed. Here it is subject to maximal evaporation, and mixing by wave and wind. Only in areas where there is a continual, onshore wind over long periods, is there likely to be any extensive heat damage to sedentary animal and plant communities, or near a high temperature—high salinity outfall from a combination power generation—desalination plant, where the effluent will sink to the bottom. Each ocean outfall will have its own properties, and should be the subject of a separate study; consideration will have to be given to wind, tide, and current patterns, distance from shore, relations of sandy beach and rocky headland, depth of water and the presence or absence of submarine canyons and upwelling.

Estuarine systems have been shown to be extremely important ecologically, but unfortunately engineers seem mesmerized by the possibility of discharging heated effluents into them, even where it is possible to put the waste into the ocean. One would think that thermal considerations alone would persuade those who desire to use seawater for cooling to direct their outfalls into the ocean so as to avoid any possibility of heating shallow areas, and raising the temperature of the intake water; however, this is often disregarded in practice. For example, in the original design and operation of the Turkey Point power station on South Biscayne Bay, Florida, the intake was situated to the north of the outfall in such a way that the warm water was

carried into the intake channel by prevailing northerly movements of the shallow water. This resulted in a virtual recirculation of the warmed water from the effluent in a small coastal lagoon less than 3 m deep. It would have been possible to site the power plant with easy access to an oceanic discharge point, with a strong current flowing past the outfall to insure quick mixing.

The term tropical-type estuaries has been used to signify those estuaries which may or may not be situated within the actual latitudes of the tropics but have the characteristics of tropical environments due to warm ocean currents moving close inshore. On the east coast of continents in both hemispheres, water circulation is such that tropical waters extend beyond the defined limits of the tropics (23°), so coasts of such regions as the northern half of New South Wales and southern Florida are still ecologically in the tropics. Thus, within the southernmost part of Florida, "there are many more species of West Indian marine life than in any one area of the Caribbean, with the exception of the islands of Hispaniola and Cuba and perhaps Jamaica" (Committee on Inshore and Estuarine Pollution, 1969), while Moreton Bay in southern Queensland has large numbers of species from the Coral Sea and the Solomon Islands over 2,000 km to the north. In shallow waters such as these, where there is high insolation and a low evaporation rate owing to high humidity, water temperatures will not be much, if any, lower than in truly tropical waters. Thus, in Australia, on the east coast, tropical species of sedentary plants and animals occur as far south as Long Reef, just north of Sydney, close to 34°S latitude, while Moreton Bay, at 27°S, is quite tropical in character. Florida waters have a still more tropical habitat due to the Florida Current derived from the Caribbean and to the prevailing onshore summer winds blowing over warm waters, though the latitude is north of 25°N.

Most tropical and subtropical estuaries are of the barrier lagoon type, though drowned valley estuarines do occur, such as those of Puerto Rico, Indonesia, and northern Australia. In these regions, coral reefs are often present. Because the terrain around the shores of drowned valleys tends to be somewhat rugged, industry prefers to build around the barrier lagoon type of estuary, where the hinterland tends to be flat. The world's largest rivers such as the Ganges, Congo, and Amazon are not the scene of much secondary industry which would occasion thermal pollution on any large scale, and possess a vast, though not unlimited, dilution capacity.

Industrial development in the true tropics, i.e. between latitudes 23°N and 23°S, has not resulted in studies of the effects of thermal pollution until recently. At present, therefore, we must be content with research carried out largely in quasitropical environments, and the most comprehensive studies thus far have been in South Biscayne Bay, Florida, which will be discussed below.

Let us consider first the ecological situation in several types of estuaries, and the likely effects of thermal pollution.

The barrier lagoon environment

Barrier lagoons are, in the main, shallow, elongated and narrow, with openings at intervals, and usually have short rivers coming in, though not necessarily opposite the openings to the ocean. The bottom may be silty or sandy, with, at times, ridges of more or less consolidated rock, generally parallel with the shoreline. At times some of these lagoons may be entirely out off from the sea and, when they break out again, the new opening may not be at the site of the previous entrance. These lagoons have usually a strong tendency to silt up, since suspended terrigenous sediments have restricted access to the sea. Lagoons of this type are exemplified by the Texas bays from the Laguna Madre at Port Isabel to Galveston Bay, and stretch from Florida Bay along much of the coast of the United States almost to the tip of Long Island, representing environments from tropical to cool-temperate. Similar strings of coastal lagoons, though not so continuous, stretch along the east coast of Australia, and are represented in Western Australia by Cockburn Sound and the shallow areas north of this.

In some lagoons, temperatures over 40°C may be encountered, as in Baffin Bay, near Corpus Christi, Texas, parts of Florida Bay, and Steward's Point on the Macleay River in New South Wales. Such temperatures are lethal to many algae, but are not sufficiently prolonged to kill the sea grass rhizomes, which persist in most cases, though exposed leaves often die off.

The shallow lagoons are rarely stratified, except during flash floods or heavy local rains, but salinities may vary considerably with the season. Parts of the Texas Bays may have salinities up to 60‰ (Oppenheimer and Wood, 1965), and Florida Bay also has very high salinities at times. After heavy rains, the salinity may drop to 0.1‰, so most of the organisms present are euryhaline. Thermal effluents, even in these shallow estuaries, produce stratification and, in areas with heavy summer rains, the cool, fresh water may flow over the top of the bay water, though the latter is heated by the effluent. This will cause a temperature inversion and insulation of the bottom water from the atmosphere and thus retard cooling.

Drowned valleys

In drowned valley estuaries, because of their greater depth, stratification is frequent, there is an annual turnover, and temperature and salinity gradients are not so steep. Many organisms can seek shelter from rapid changes by moving into the hypolimnion, so there is much greater chance of regeneration of species after local killing. One difficulty is that industrial cooling units usually take their water from the epilimnion and return it thereto with added heat so that stratification becomes more marked. Even so, there is much less risk of serious thermal pollution by industrial cooling in drowned

valleys than in coastal lagoons. However, the effects depend largely on the size of the industrial plant in relation to the area of the drowned valley or the portion of it available for the discharge. There is also the possibility of drawing cooler water from the hypolimnion, heating it, and returning it to the epilimnion at approximately the same temperature as the epilimnion. This must be considered carefully, however, using appropriate mathematical models.

River deltas

River deltas present special problems of a very complex nature, e.g. relations between depth, river volume and rate of flow, current and tide patterns at the mouth, flushing rates, the shape of the saltwater wedge and other factors. The degree of thermal pollution is most likely related to river flow rate.

BIOLOGICAL CONSIDERATIONS OF THERMAL POLLUTION IN THE TROPICS

Tropical environments often present quite different problems than the temperate ones. The first is that the organisms occurring in tropical waters are much closer to their thermal limits than those of temperate waters, especially in summer, may be either inhibitory or lethal, depending on their range, rate of change, and duration. Biebl (1962) has shown that the thermal death point of submerged algae in Puerto Rico is 32—35°C, only 4—6°C above the average summer maximum of 28°C, whereas for temperate algae from the north coast of France the upper limit is 27—30°C, which is 10.5—13.5°C above the average summer maximum of 16.5°C. The Committee on Inshore and Estuarine Pollution (1969) found that numerous "tropical marine organisms belonging to a widely divergent group of plants and animals all survive to temperatures of about 28°C, reduce their metabolism between 28° and 30°C, begin to die about 33—34°C. Mayer (1914, 1918b) was the first to recognize that tropical organisms normally live only a few degrees below their upper lethal temperatures, and Edmondson (1928) found that temperatures 5—6°C above summer temperatures killed 11 of 13 species of Hawaiian corals. Hohman and Tsuda (1973) found that above 28°C photosynthetic rates of *Caulerpa racemosa* decreased continuously, and that increasing the temperature from 28°C to 30°C doubled the respiration rate. Specimens of this algae incubated for 12.5 hours at 32°C had a P/R ratio much less than 1 and no plants at 34°C showed any net photosynthesis.

Biebl (1962) and Schwartz and Almodovar (1971) describe the effects of exposure and elevated temperature on a variety of coral reef algae. Thorhaug (1971) and Thorhaug and Fernandez (1973) report that changes in bioelectric potential across the cell membrane system of the giant celled macroalgae *Valonia* are a highly sensitive index of thermal stress. A variety of trop-

ical benthic green macro-algae tested in south Florida had abrupt thermal limits between 31° and 33°C (Bader et al., 1972). Temperature acclimation in these algae conferred adaptations of the order of only tenths of degrees.

Rupp (1973) found that temperatures of 34—36°C reduced fertilization and cleavage in a number of reef echinoderms. Early cleavage appeared to be more sensitive to elevated temperatures than fertilization. *Acanthaster planci* stop feeding and behave abnormally when the temperature reaches 33°C, and die within one week when held at 33—34°C (Yamaguchi, 1973).

Studies in Guam (Jones and Randall, 1973) have suggested the upper lethal limit for some corals is between 30° and 32°C (see Johannes, Chapter 2).

Graham (1971) found that the critical thermal maxima for three species of tropical Atlantic fish were higher than those of three closely related Pacific species.

In temperate waters, even lethal temperatures which kill off most of the existing local flora and fauna may not necessarily denude an area, as, under favorable conditions of currents and winds, the original association may be replaced by one derived from warmer regions, or by a resident population of warmer-water species living close to its temperature minimum, but restrained normally by competition with the local dominant populations (Naylor, 1965; North, 1966). Any artificial rise in temperature in tropical waters, however, will pose a greater threat to the biota, which, once eradicated, is largely not replaceable as long as the elevated temperature is sustained; there are very few replacement species able to colonize such environments, blue-green algae being an exception (see below).

This means that, if a thermal effluent is discharged into a tropical estuary, there is a strong possibility of permanent or long-term damage, the extent depending on the nature of the organisms, particularly the sessile plants and animals as opposed to the more motile forms, and on the relation between the area close to the discharge and the estuary as a whole.

Chadwick et al. (1950), Southward (1958), Singletary (1971), and Wood (1953) have shown that time is an important factor in defining the temperature tolerance of organisms. Most can stand considerable changes of temperature for a short time, but lesser changes over a longer period may be lethal. For this reason, diurnal fluctuations of temperature are usually of sufficiently short duration that the peak temperatures have little effect, except on the most vulnerable species.

Van Eepoel et al. (1971) reported that the winter—summer temperature range in parts of Lindberg Bay, St. Thomas, was reduced by about half at sampling stations in a plume of heated effluent. They noted that the environmental effects of limiting the seasonal water temperature range in the tropics is unknown.

With respect to the animals the sedentary species are likely to be more severely affected, as many, especially adult fishes and crustaceans, can

migrate. Jones and Randall (1973) found that fishes, crustaceans, and echinoderms characteristic of reef flat areas in Guam disappeared when heated water from a power plant was introduced. In addition there was considerable destruction of the sessile corals and marine algae.

Thorhaug et al. (1971) found upper temperature limits critical in the development of *Penaeus duorarum*. Maximum development rates were found between 25° and 31.5°C, survival was lessened at 33° to 34°C, and 18-h exposure to 37°C was lethal to all developmental stages.

Virnstein (1972) found that normal summer temperatures of 32—33°C in Tampa Bay restricted benthic fauna and caused a decrease in density and number of species, and that there was a large decrease in diversity and density of benthic infauna associated with the temperature of 34—37°C produced by a power plant. Glynn (1968) found high mortalities of echinoids, turtle grass, brachyuran crabs, chitons, and ophiuroids coincident with low-water conditions on shallow flats in Puerto Rico when temperatures rose to 35—40°C.

Some submerged algae such as *Gracilaria confervoides* (Wood, 1964), *Caulerpa* sp., *Cladophora* sp., and *Laurencia* sp. (Zieman, 1970), may be killed off in summer under natural conditions in shallow waters. Other species such as the coralline reds and calcareous green algal species, e.g., *Penicillus*, *Halimeda*, *Acetabularia* and *Melobesia*, are hardier, although Jones and Randall (1973) found heated effluents killed coralline algae.

Sea grasses are often less susceptible to thermal stress due to the rhizomes buried in the sediments (see Zieman, Chapter 6). The algae will be more easily killed by acute thermal stress and will disappear fairly rapidly from the scene at rates dependent on their thermal tolerances. Recolonization of algae from spores of plants which have been growing outside the thermal influence will occur as soon as the high temperature decreases. If the heating of the water in contact with the bottom is persistent enough and the temperature high enough to cause an increase in the temperature of the sediment, killing of the rhizomes can result (Zieman, 1970).

Sediments are poorer conductors of heat than seawater, so absorb heat more slowly. On the other hand, once the sediments become heated they will retain the heat much longer. Therefore, an intermittent heating of the water, if sufficiently great, can produce a lasting effect in the sediment, which will retain its heat after the water has cooled and act as a heat reservoir for the system. This fact is important for the sea grasses, but will also affect the rhizoids and spores of the seaweeds and will help to delay their regeneration after the water temperature has fallen again.

It is thus the duration of the heating as well as the temperature reached that will produce a given biological effect, and the heating, though intermittent, may have a cumulative result as sediment acts as an integrator of the fluctuating temperature. This is more likely to be critical for plant and animal populations in the tropics than in cooler waters and sediments. Legisla-

tion for regulating the effects of thermal effluents should include provisions for limiting the duration of heating as well as the temperature.

Reports of estuarine pollution more often describe the effects on plankton and larger, more obvious fish rather than on sedentary forms. In the case of phytoplankton, this emphasis is due to a basic misconception (see Johannes, Chapter 2). In studies which were made in two continents, Wood (1964) has found little evidence that phytoplankton is a major element in the food web of tropical estuaries. One rarely finds elements of phytoplankton in the gut of estuarine animals which prefer attached algae, either microscopic or macroscopic. Phytophagous fishes such as the mullets usually eat epiphytic algae, mainly diatoms and flagellates as well as some of the simpler macro-algae such as *Enteromorpha*.

These attached algae are, of course, much more abundant in the estuaries than the phytoplankton and, under ideal conditions, may present an annual production similar to that of the sea grasses (Wood, 1959, 1964), since the rate of reproduction, i.e. turnover, is far greater. Moreover, much of what is sometimes considered to be phytoplankton is frequently composed of detached epontic or benthic algae together with spores of macro-algae, many of which are motile. It is therefore of greater significance to study the benthic and epontic organisms than the phytoplankton when we are interested in the effects of thermal pollution.

Another difficulty in studying phytoplankton is that it fluctuates, both with regard to numbers and species, with time as well as with area (Wood, 1964). A bloom may occur in one part of an estuary while another bloom occurs several kilometers distant at the same time, and in stratified waters simultaneous blooms may occur one above the other.

The sedentary forms, of course, do not present a similar problem, and blooms of particular species tend to remain *in situ* for a week or more, and sometimes for months (Wood, unpublished). There are differences in the epontic community of estuaries from one part to another, and this has been reported by Wood (1964) from Lake Macquarie, New South Wales, and from the Texas bays. This must be taken into consideration in deciding the location of sampling stations and the provision of adequate controls.

In studying the structure of water at various temperatures Drost-Hansen (1969) found anomalies in the vicinity of 15, 30 and 45°C and noted abrupt changes in the activities of various organisms at these temperatures. Thorhaug (1969) showed a rapid decrease in the survival of *Valonia* and *Nitella* above 30°C, and Thorhaug et al. (1971) showed that the survival of several species of tropical algae and animals decreased in the range of 30—35°C.

With animals, the planktonic species, even if rapidly killed by heated water, may repopulate via larvae brought in from unheated areas, and from reproduction by survivors. Because of the haphazard chances of survival in a highly competitive environment, most marine animals produce a huge number of spawn, so that the death of a relatively large number may well result

in the survival of individuals that would normally die, and a very small diminution in the final population of the area. A heavy kill may, however, delay the reproductive peak of susceptible species, or allow the dominance of a competing species and thus upset the ecology of the system as a whole. The killing of certain ecologically important species can disrupt entire food chains as some species are selective in their feeding and may not readily accept as food other organisms which might appear to be quite similar to their favorite food.

Reeve and Cosper (1972) showed that *Acartia tonsa*, the dominant planktonic copepod in South Biscayne Bay, increased their tolerance to high temperature as the ambient temperature increased. The higher the ambient temperature to which the animal is adapted the less the mortality due to elevated temperatures. Reeve (1970) reported that when effluent temperatures exceeded 40°C the results were "immediate and catastrophic", with death of over 80% of the plankton. He also said that later work has "confirmed that the critical thermal maxima for many groups of plankton are in the mid-thirties".

Since the distribution of estuarine plankton is influenced strongly by circulation, a severe problem can develop if the circulation of a power plant is sufficiently large so as to dominate the normal estuarine circulation. In shallow bays and lagoons where this sometimes happens, the power plant can become a giant "predator" with the capacity to "consume" or destroy a large portion of the total plankton due to high temperature and recirculation.

In any estimation of the possibilities of damage to an estuary by thermal effluents, it is desirable, and should be considered essential, to make a study of the whole estuarine system before any industrial plant is constructed. However, except in remote areas, every estuary has, for many years, been subject to artificial stresses of one kind or another as long as there is any human population on its shores. There is no such thing as an unpolluted estuary in populated areas. It is still essential to fix a datum line for each estuary, and the sooner this is done the better.

Of course, such a program is wishful thinking, as few, if any, governments or other sponsors are willing to invest in a pollution research project which has no immediate effect on a recognized pollution problem. This is a tragedy, as the first defense of a polluter has often been that there is no adequate criterion of previous pollution, and therefore of how much he is contributing to the problem. He maintains that his pollution is minor when compared with the damage already done, and this may be very difficult to prove untrue.

A problem of still greater difficulty, and one that is almost completely unsolved, is the assessment of sub-lethal effects. Nearly all studies of necessity deal with acute, or short-term, severe effects. Studies dealing with chronic stresses, those that are sub-lethal in a short-time span but may prove lethal

if continued for a sufficient period, are scarce. One such effect, noted by Zieman (1970) for the sea grass *Thalassia*, will be discussed below, but studies of decreased swimming and feeding abilities, decreased reproductive capacity, and generally decreased competitive ability due to sub-lethal increases are few. These are exceptionally difficult problems to assess. Jones and Randall (1973), Mayer (1914), and Edmondson (1928) studied sub-lethal effects on corals and have found that sub-lethal temperature reduced the growth rate of corals in Guam (see Johannes, Chapter 2). Hohman and Tsuda (1973) found sub-lethal temperature increased respiration and reduced net photosynthesis in *Caulerpa racemosa* from Guam.

ASPECTS OF THERMAL POLLUTION IN A TROPICAL-TYPE ESTUARY — BISCAYNE BAY

The history of this study, mainly carried out by the University of Miami, is a classical example of frustration in science, especially when it is related to industrial expansion and the improvement of related facilities. A small but adequate proposal for an ecological study of South Biscayne Bay was submitted by the University of Miami as soon as the power plant at Turkey Point was projected, and before it was approved. This proposal was rejected after considerable delay, on the grounds that it was not extensive enough, and a much larger proposal was submitted to a federal agency, which then administered the desired funds. This was rejected because it was too large, and finally, a third proposal was accepted that included twelve senior research scientists instead of the three in the original proposal but with about a third of the funds originally asked for and was divided between several funding agencies.

This meant that a comprehensive, or even adequate study was impossible, and that one of the fossil fuel generators of 432 MW was already in operation when the study commenced. For this reason, no baseline information other than that available from aerial photography or obtainable from studying the history of recent sediments was available on the ecology of South Biscayne Bay prior to the starting of active operations by the Florida Power and Light Company.

As a result of the failure to establish a baseline, the scientists studying South Biscayne Bay have had to either select control areas well away from the part affected by the plume of hot water discharge or use gradient analysis.

Characteristics of South Biscayne Bay

Physical setting

Biscayne Bay is a shallow tropical estuarine lagoon situated on the extreme southwest portion of the Florida peninsula (Fig. 1). The bay as a

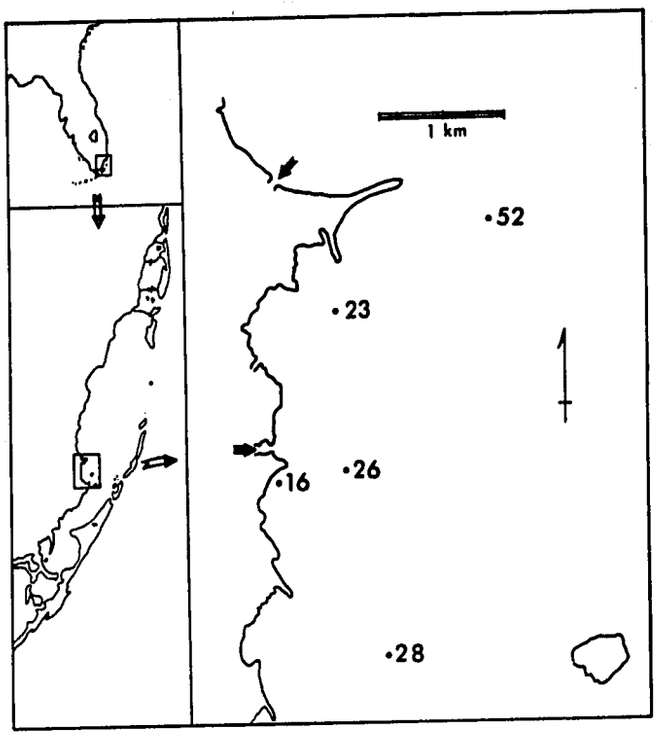

Fig. 1. Location map of south Florida showing Turkey Point and indicating several stations.

whole is about 56 km in length and averages about 8 km in width with a maximum width of about 16 km. Turkey Point is situated in Southern Biscayne Bay, which is divided from the northern and central portions by Featherbed Banks. Southern Biscayne Bay is about 16 by 20 km. It is bordered on the west by the mainland mangrove swamps and on the east by Elliot Key, which forms an unbroken barrier. Communication with the ocean is through Caesar's Creek at the southern end of Elliot Key, and through Boca Chica Cut at the north. There is some communication with the central portion of the bay around the west end of the Featherbed Banks, and across them at high tide, and with Card Sound to the south across Cutter Bank.

The local climate is tropical, due to warm currents moving north through the Straits of Florida from the Caribbean Sea. East of the Straits of Florida lie the Bahamas with shallow, warm waters, and east of them again is the northward-flowing Antilles Current derived from the North Equatorial Current. Thus, the trade winds blowing from the east are kept continually warm and near-saturated with moisture. The average summer air temperature is in the upper 20s and lower 30s, while the winter air temperatures are around 18–20°C. The summer humidity is consistently around 90–100%, and this

condition extends from May through October. In the summer, when the winds are from the east to southeast, the potential for extensive evaporative cooling from the surface of the bay is low. Precipitation is high, averaging about 150 cm per year, with 75—80% falling from June to November. The average at Homestead, just north of Turkey Point, is about 168 cm per year.

The tides are small, averaging 0.5 m. Circulation is generally tidal but may be overpowered by the wind. A large part of the subsequent problems surrounding Turkey Point was due to the failure to ascertain the true circulation in the vicinity of the power plant. It was originally believed that southern Biscayne Bay was well flushed because the tidal prism was 1/3 of the low-water volume of the bay, and the flow was predominantly NNE of the ebb and SSW of the flood (Richardson, 1966). Later studies, however, showed that this part of the bay is not well flushed.

Ambient temperatures in the inshore areas of southern Biscayne Bay average about 17°C in the winter and rise to 31°C in the summer. Minimum temperatures observed in the study area were 9—10°C during an extreme cold spell which caused considerable cold mortality. Maximum inshore temperatures were near 33°C but reached 36—37°C in shallow tide pools (Roessler and Zieman, 1969; Roessler, 1971). There was considerable modification of the normal thermal distribution by the effluent plume. Fig. 2 shows the average summer temperature distribution for the summer of 1967.

There is no large single source of freshwater input into the southern part of the bay. Freshwater comes from land runoff and numerous drainage canals. The drainage canals influence the near-shore salinity just north of Turkey Point, but there are none of these between Turkey Point and Card Sound. The salinity of inshore southern Biscayne Bay averages 27—33‰ in the summer and 31—34‰ in the winter; the salinity is often reduced following heavy rains to around 10‰ and salinities of 35—36‰ are reached during dry spells. The salinity distribution is one of longshore isohalines parallel to the shoreline and is not significantly altered by the Turkey Point effluent (Fig. 3).

Biological setting

The western shoreline of southern Biscayne Bay is a fringing mangrove swamp. The red mangrove *Rhizophora mangle* is dominant. Just seaward of the mangrove fringe, the sediments are 1—2 m thick, and are composed of a mixture of paralic and freshwater peats, calcitic mud, recent organic debris and detritus, shell hash and *Halimeda* plates. This shoreline forms a gradually sloping wedge which extends 350—400 m into the bay.

Among the algae, *Batophora oerstedi*, *Acetabularia crenulata*, and *Cladophora* sp. occur commonly throughout the zone. Occasional specimens of *Laurencia* sp., *Avrainvillea nigricans*, *Anadyomene stellata*, and *Digenia simplex* are found. During the winter, various ephemeral brown and red algae appear briefly and disappear.

Laurencia is frequently epiphytic, but is capable of independent existence,

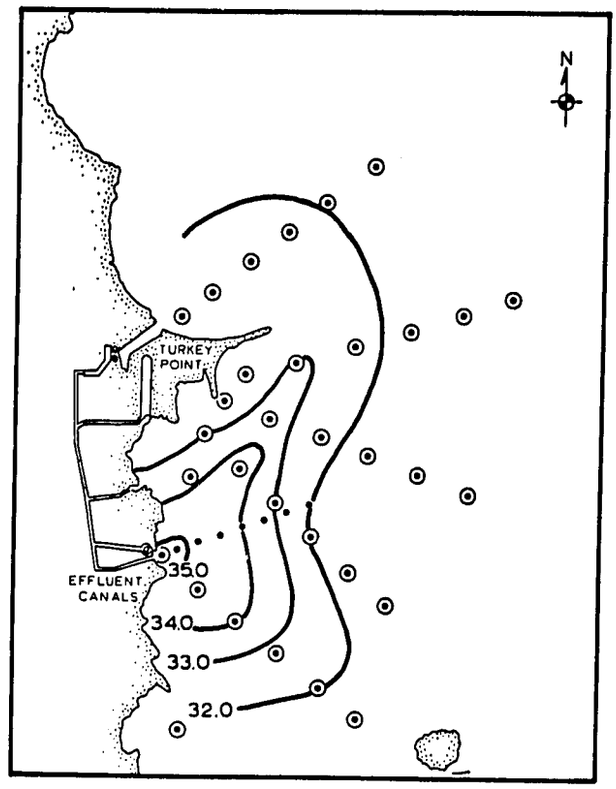

Fig. 2. Average summer temperature distribution off Turkey Point.

and tends to form parallel, generally longshore windrows and bundles when it is abundant. Its value in assessing pollution is very limited, as it is impossible to tell if a plant has existed in a particular locality or has recently been deposited there by the currents.

Beyond the inshore sediment wedge, two distinct community associations are found, which are regulated by the depth of the sediment. The first community consists of the *Thalassia* beds which are generally circular or oval in shape, occurring in areas of deep sediment (1—4 m) of remnant mangrove peat overlain with recent estuarine sediments (Zieman, 1972). These are often pure stands of *Thalassia* with few or no other higher plants in association, although some beds do have sporadic individuals of *Penicillus capitatus* associated with them. The *Thalassia* is also host to a number of epiphytes, including small macro-algae and large numbers of diatoms.

Other than the *Thalassia* beds, the area is nearly all Pleistocene limestone with little or no sediment. The thin veneer of sediment present is usually calcitic, with far less organic matter present than occurs in the inshore sediments. All of the algal species mentioned previously are present, though there is rarely a dominant species of fixed algae. *Udotea conglutinata*, *Rhipo-*

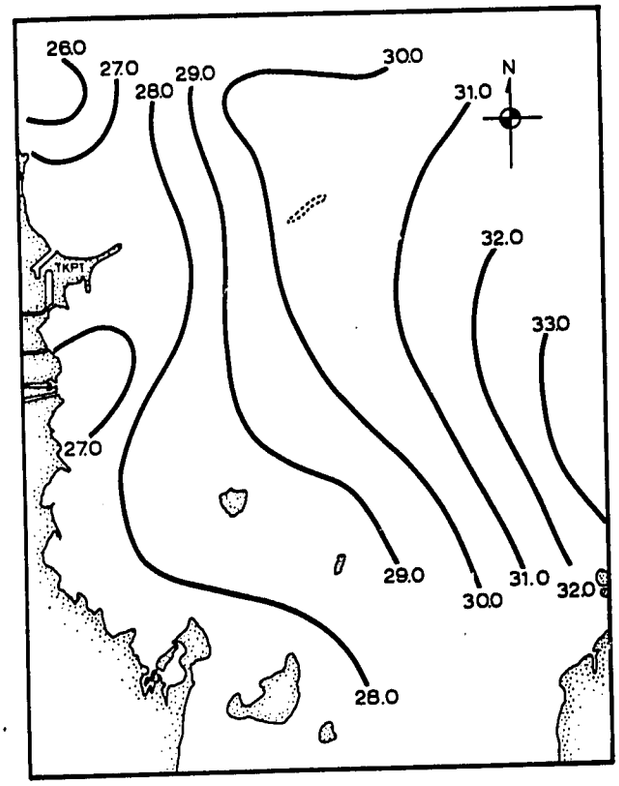

Fig. 3. Average salinity distribution off Turkey Point.

cephalus phoenix, *Sargassum pteropleuron*, and *Caulerpa paspaloides* appear abundantly. In addition there are more species of short-term or ephemeral reds and browns which appear primarily in the late winter months.

A detailed list of the plants and animals occurring in the area is given by Voss et al. (1969).

Thermal pollution studies at Turkey Point

Early operations

The Turkey Point power station is situated on the western shore of southern Biscayne Bay as shown in Fig. 1. The plant is designed to consist of two fossil fuel units, each generating 432 MW and requiring 5.9 m^3 /sec of cooling water heated about 5° C, and two nuclear units generating 760 MW each and requiring 13.8 m^3 /sec of water heated 6—8° C, in the original design. This design called for a discharge at full operations of an effluent of 39.6 m^3 /sec heated 6—8° C; however, as soon as the plant began operations with the small conventional units, it became obvious that there were some serious problems with the design of the cooling system.

As shown in Fig. 1, the intake for the power plant is just to the north of Turkey Point itself, in a barge canal. The initial discharge canals emptied most of the effluent about 1.5 km south of the intake at Turtle Point. This was the cooling system in use during the studies which will be discussed here. The ecologic damage occurred just off this discharge point. Initially this cooling-canal design was thought to be sufficient for the entire plant, including the nuclear units. However, insufficient attention was paid to the local current patterns until the oil-fired conventional generators were running, and it was discovered that the units were recirculating considerable amounts of their effluent.

Studies began in the Turkey Point vicinity about May of 1968. Early work was devoted largely to description of the area. It proved extremely important that studies commenced at this time. The first conventional unit began operation in the spring of 1967, with the second unit following about a year later. Although temperatures were beginning to rise with the onset of summer, and the increase in plant operations, the earliest surveys were made when the area was largely undisturbed. As opposed to some effects which will be related later, the benthic plant communities were initially the same just off the main discharge point as they were in areas to the immediate north and south. Damage at this time was limited to just a few hectares but increased throughout the summer and into the fall.

As the heat stress increased, the damage developed in a consistent pattern. Algae disappeared according to the thermal tolerances described below. The calcareous green algae and the sea grasses were the most heat-tolerant plants

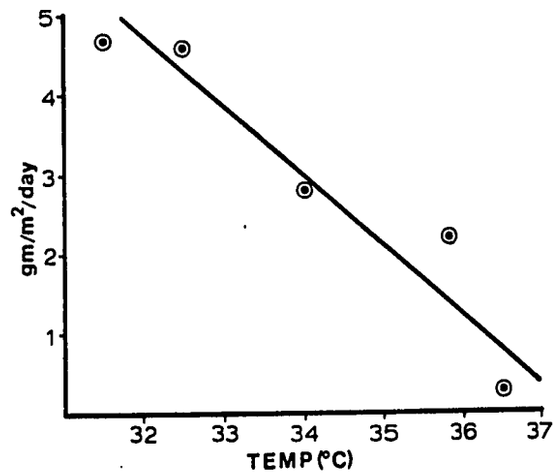

Fig. 4. Results of early studies at Turkey Point: decrease in productivity of *Thalassia* with increasing temperature.

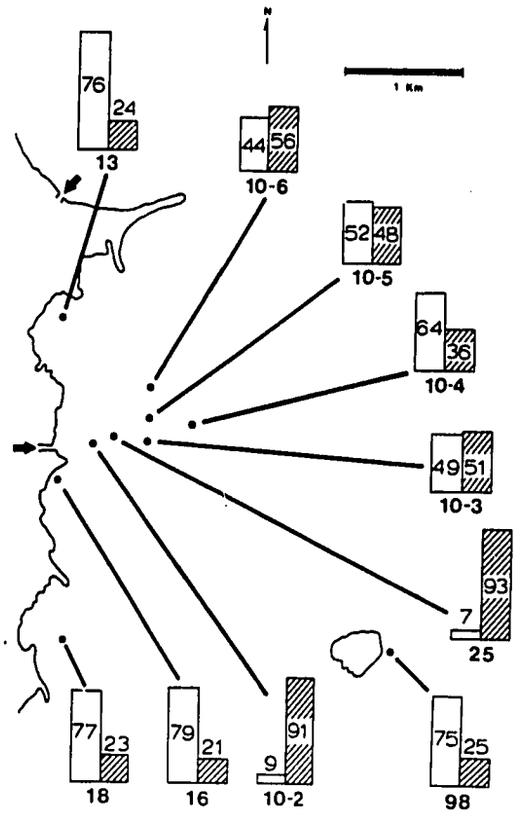

Fig. 5. Relationship between heat death of *Thalassia* leaves and the thermal effluent. Numbers on left (clear) are % of attached leaves which are alive, while numbers on right (slashed) are % of total leaves that are dead.

of the natural community. Ultimately, after much erosion due to the velocity of the effluent, the area was carpeted by a blue-green algal mat.

Preliminary studies showed the effect of temperature on the growth rate of *Thalassia* blades (Zieman, 1968; 1975). Growth rate decreased rapidly with temperatures above summer ambient (Fig. 4). There was also rapid increase in the death rate of *Thalassia* leaves with increases in temperature above ambient (Fig. 5), causing a decline in the abundance and density. By the end of September 1968, 14 ha of sea grasses had been destroyed.

Average summer temperatures for the Turkey Point region are shown in Fig. 2. By the summer of 1969 the area in which the average temperature was raised +5°C or more was 16—20 ha; 30 more ha were raised +4°C, 70 ha additional were raised +3°C and a further 120 ha were raised +2°C, in temperature (Fig. 6; Roessler and Zieman, 1969; Roessler, 1971).

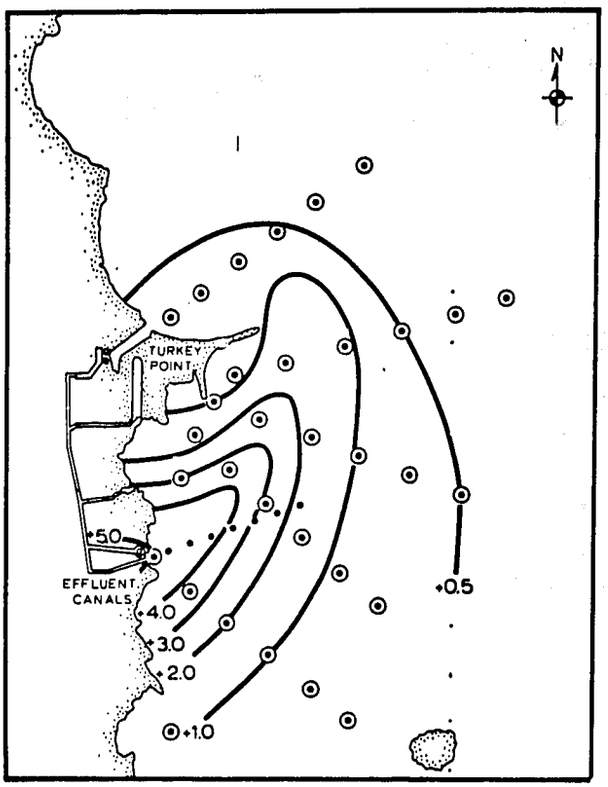

Fig. 6. Average temperature increase (ΔT) off Turkey Point.

The marine algae

The effects of pollution on primary producers especially is important, as they provide with food and shelter for the higher trophic levels. If they are severely damaged or destroyed, the entire trophic scheme collapses. In addition, the benthic plants are valuable indicator organisms, as they cannot avoid the stresses.

In the vicinity of Turkey Point, the distribution of algae (Roessler and Zieman, 1969; Zieman, 1970) was closely related to the distribution of the heated water (Fig. 6). Patterns of algal exclusion were closely correlated with the temperature elevation. Fig. 7A shows the distribution of algal species and Fig. 7B shows the pattern of the species diversity. In general, both algal species numbers and diversities increased until temperature reached about 30—31°C, and declined rapidly at higher temperatures. Blue-green algal distribution (Fig. 8) contrasted markedly with the above patterns. The areas where the natural communities were severely damaged by heat were replaced by blue-green algal mats consisting primarily of *Lyngbya majuscula* and *Schizothrix calcicola* (Hine, in DeSylva and Hine, 1972). Thus, the earlier findings of Cairns (1956) and Patrick (1969), that as temperatures increase

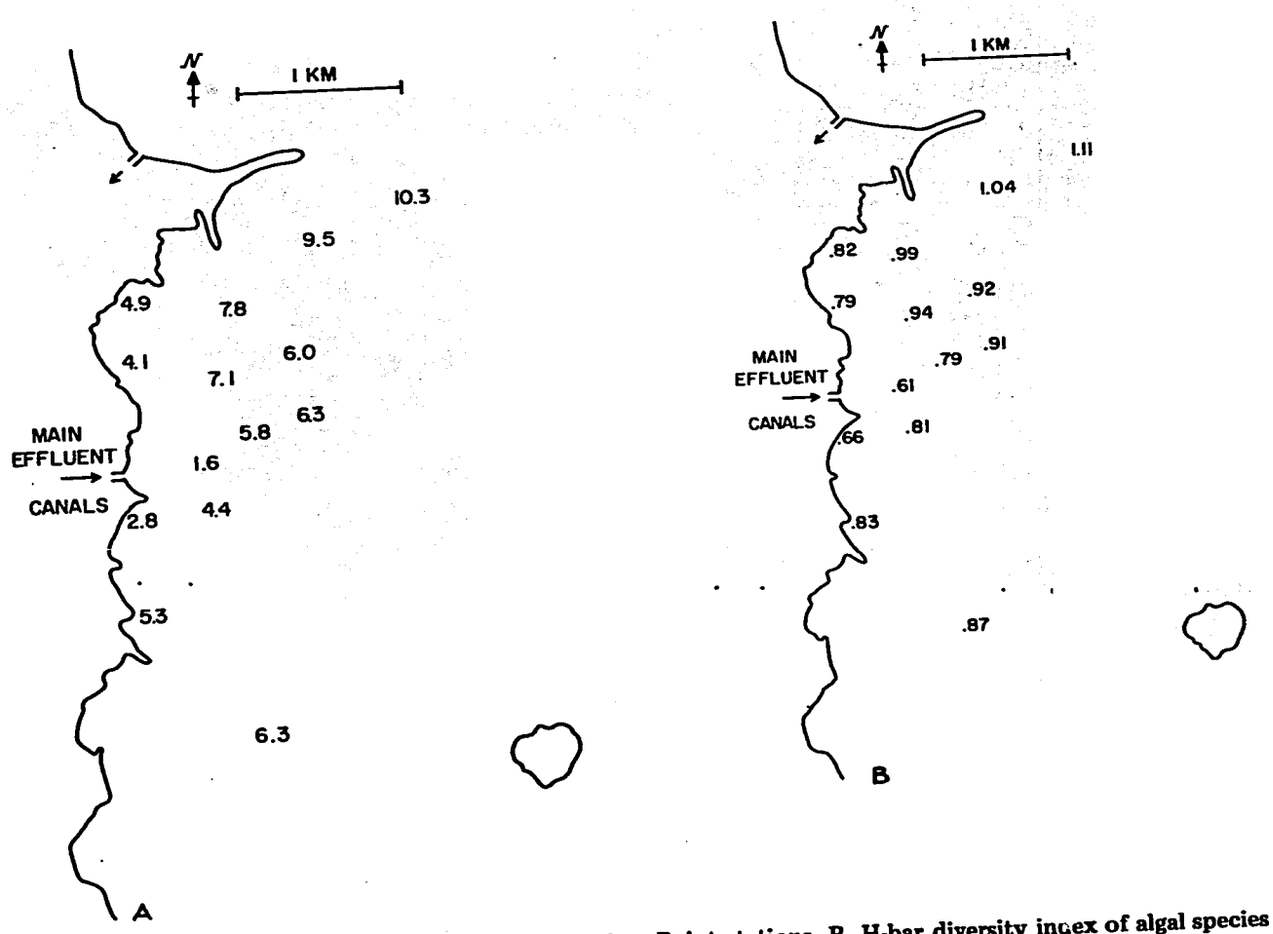

Fig. 7. A. Average numbers of algal species at Turkey Point stations. B. H-bar diversity index of algal species at Turkey Point stations.

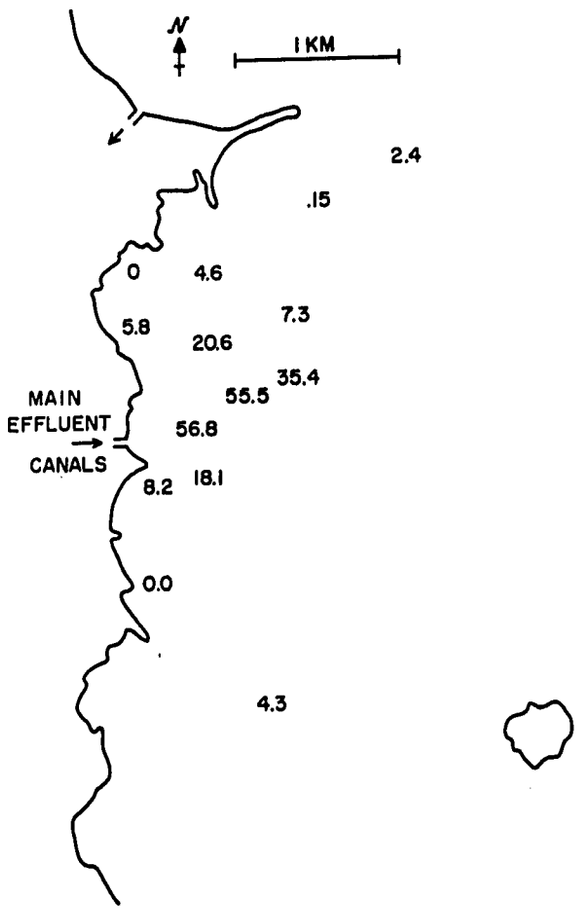

Fig. 8. Average yearly blue-green algal mat coverage off Turkey Point (% coverage).

normal algal communities are replaced by blue-greens, hold for marine as well as freshwater systems. Jones and Randall (1973) and Kolehmainen et al. (1973) also found blue-green algae replaced other species at elevated temperatures in Guam and Puerto Rico.

The shifts in distribution of four species of algae (*Penicillus capitatus*, *Halimeda incrassata*, *Digenia simplex*, and *Laurencia* spp.) are shown in Fig. 9. These clearly show the retreat of the algae due to heated water in the summer and incomplete recovery in the winter. Note the similarity of the pattern of retreat in the summer with the pattern of the raised temperatures (ΔT) from the effluent (Fig. 6). This also indicates the relative tolerance of heat by the algae, showing that *Penicillus* has a higher tolerance than the other species.

The relative tolerance of the algae, and the gradient of damage can be shown in another manner. Where Fig. 9 showed the spatial distribution of

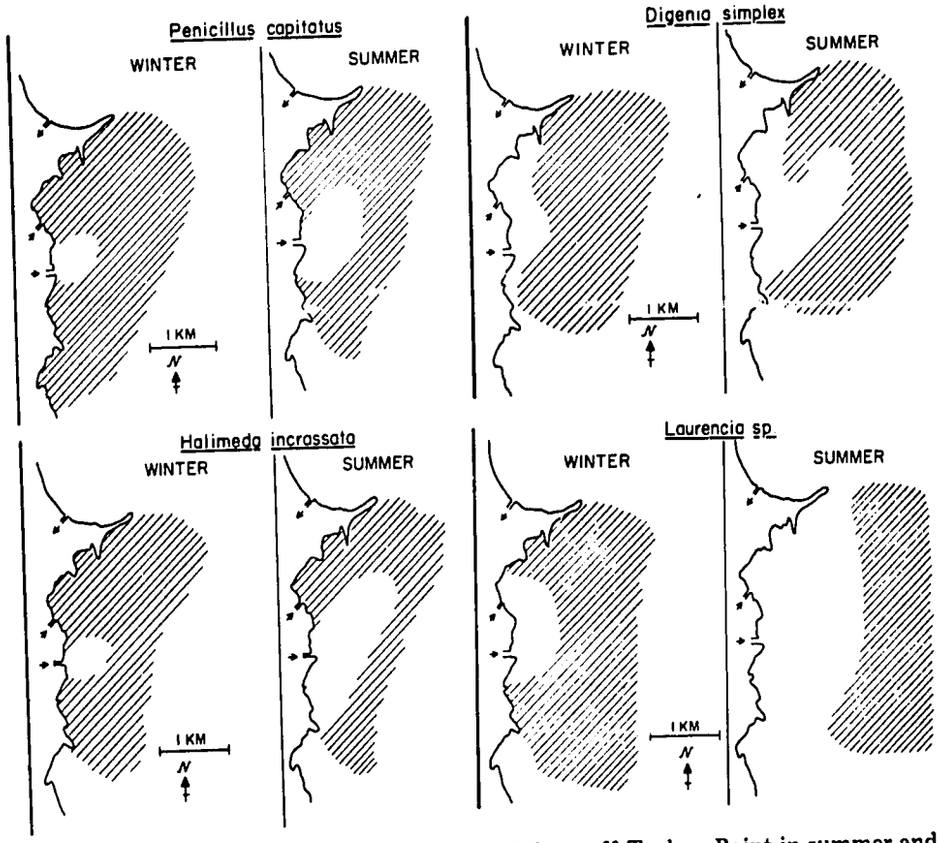

Fig. 9. Spatial distribution of four species of algae off Turkey Point in summer and winter.

certain algal species, Fig. 10 shows the temporal distribution of species at three stations at increasing distances from the mouth of the effluent canal. These represent the species present within a permanent 1-m^2 quadrat which was monitored every 2—3 weeks. At the station closest to the effluent (Station 26 — 600 m from effluent), all species of algae were killed during the summer of 1969, and only half recovered the following winter and half of these were killed the following summer. At Station 24, 900 m from the effluent along the axis of the plume, the distributional lines are more continuous, with several species persisting throughout the summer and others recovering the following winter. Station 42 is 2 km from the effluent canal and shows even less seasonality. Here the number of species which are present continually is considerably higher than at the closer stations.

From these studies it is possible to arrange the dominant algal species in order of thermal tolerance. *Acetabularia crenulata* is the most resistant algae and survived average temperatures 3.5—4° C above ambient. *Bataphora oerstedi* was nearly as resistant, surviving +3° C temperature increases, and *Peni-*

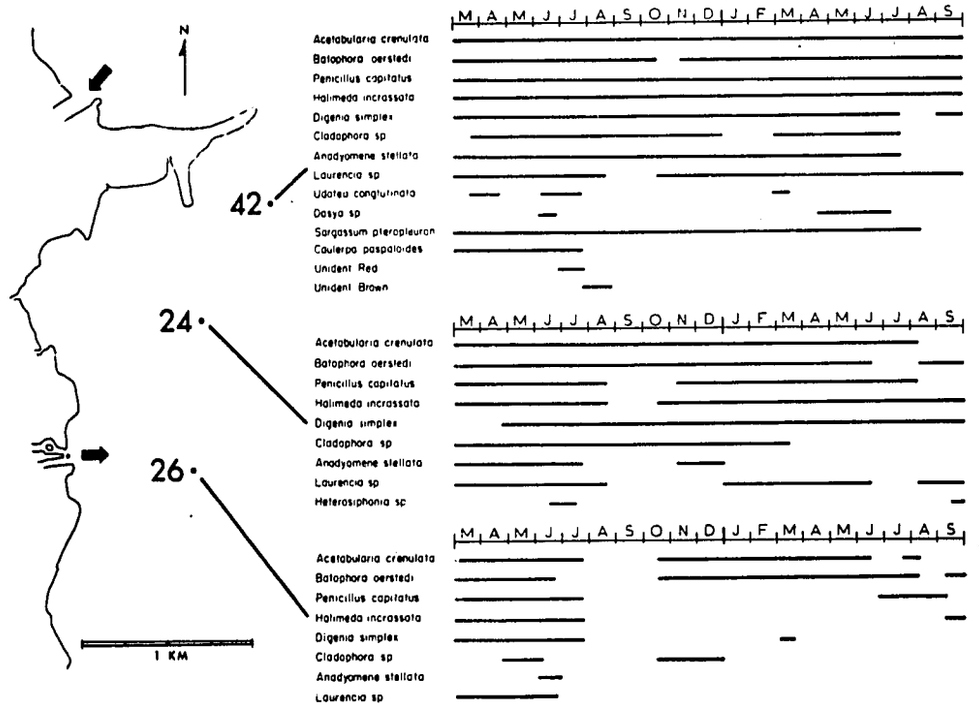

Fig. 10. Temporal distribution of algae at three stations off Turkey Point.

cillus capitatus disappeared with temperature increases of 2.5—3°C. *Halimeda incrassata*, *Digenia simplex*, and *Udotea conglutinata* disappeared at temperatures of +2°C, and *Anadyomene stellata* tolerated no more than 1°C temperature increases. *Laurencia* sp. and *Caulerpa* sp. were not resistant to temperature rises of even 1°C.

Sea grasses

Thalassia beds within the Turkey Point area tended to fluctuate between low values of 2,500 blades m^{-2} in the winter (January—March), and a high of 4,000 blades m^{-2} from May through October, after which they declined to their winter values. This corresponds approximately to average standing crops of leaves of 91 to 146 g dry weight per m^2 respectively. Leaf numbers increased until a temperature of about 30—31°C was reached, at which point they remained stable. Above these temperatures the density of the leaves declined. Net production by the leaves is also temperature-dependent (Fig. 4). *Thalassia* has a distinct temperature optimum in the immediate vicinity of 30°C.

The ratio of live/dead leaves on 10 September 1968 was 75%/25% at stations remote from the power plant, and decreased to approximately 50%/50% about 700—900 m from the mouth of the effluent canal, and to

9%/91% at a station 450 m from the canal mouth (Fig. 4). The initial decline of the sea grasses at this station was carefully documented. On 14 August 1968 there were approximately 2,300 healthy blades m^{-2}, by 4 September this was down to 400 blades m^{-2}, and by 24 September there were none. Studies of the rhizomes from this area showed that the plants had used all of their stored starch reserves. If stresses are alleviated before the starch reserves are exhausted, the plant can recover.

The sea grasses, which require high light levels, were further stressed in this area by the greatly increased turbidity of the effluent. This increased turbidity was due both to erosion at the canal mouth and to the washing out of material from the mangrove swamps in the areas adjacent to the effluent system on ebb tide. Because of the semipermanence of the robust rhizome system, which is 5—25 cm below the sediment surface, *Thalassia* is less subject to short-term fluctuations than other sea grasses or macro-algae, whose rhizomes and holdfasts are restricted to the upper few centimeters of the sediment. However, once the *Thalassia* is killed in an area, the recovery time can be years to decades.

Epiphytes

Early studies of epiphytes were made in the vicinity of Turkey Point utilizing wooden dowels placed in the main axis of the effluent at 100 m intervals. There was an increase in the thickness of the diatom mat outward from the canal mouth in summer, and a reverse in winter where the innermost sticks had the highest epiphytism. Wood found more then 240 species from 37 genera of diatoms alone.

Following these early attempts, studies by Wood (unpublished) and Sprogis (1971) showed a definite reduction in number of species colonizing glass slides with increasing temperature and closeness to the effluent (Fig. 11). They found the dominants were *Frustula saxonica* var. *rhomboides* and *Cocconeis placentula* with occasional high numbers of species of *Mastogloia* and *Synedra*. Sprogis (1971) found that the highest chlorophyll *a* values occurred within the effluent plume, but at the stations closest to the effluent canal, the chlorophyll dropped to extremely low values, possibly indicating that "the organisms are thriving at a temperature close to their critical level, and that any change of another parameter could drastically alter productivity".

Animals

Faunal responses to elevated temperature at Turkey Point were similar to floral patterns and showed similar patterns of temperature resistance, primarily in the sessile and less motile forms. There was a noticeable reduction in motile forms, such as fishes, due to avoidance of the high temperatures and the reduction of plant cover and food sources.

Nugent (1970) studied the fish population within the effluent canals and

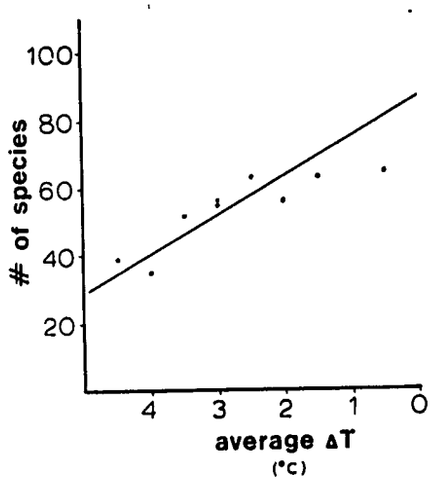

Fig. 11. Abundance of diatoms species vs. average station temperature off Turkey Point.

found that the grey snapper (*Lutjanus griseus*) and tarpon showed the least response to temperature increases. Fantail mullet (*Querimanaa trichodon*), white mullet (*Mugil curema*), and snook (*Centropomus undecimalis*) were not affected by the effluent during the winter, but decreased in abundance in the summer. Pinfish (*Lagodon rhomboides*) were abundant in the winter but decreased sharply in the summer when temperatures reached 40°C. In the intertidal zones, the settlement of the barnacle *Balanus eburneus* increased in the winter but was strongly inhibited in June and July.

Purkerson (1973; in U.S. Dep. of Int., 1970; Hagan and Purkerson, 1970) studied the effect of the heated effluent on the biota of the Turkey Point area, and recorded the effects of several particularly acute thermal stresses. He recorded the decline of the biota in 1968 and reported nearly complete destruction over an area of nearly 40 ha. Partial recovery was noted in the winter months.

Siderastrea sp. corals had expired as far as 540 m from the canals and some were dead at 725 m, although "small fish, shrimp, crabs, amphipods and brittle stars were noted as close as 185 m to the outfall. On 25 June temperatures reached 39.5°C 300 m off the canal and 36°C 1000 m off the canal and very turbid waters were found. Dead plants and animals were found 900 m east, 900 m north, and 450 m south of the canal. Dead animals included sponges (*Ircinia* and *Spheciosphongia*), mollusks (*Chione cancellata*, *Trachycardium egmontanum*, *Fasciolaria hunteria*, *Fasciolaria tulipa*, *Aquipecten givvus*, *Vermicularia spirata*, *Acmaea* sp., and others), blue crabs (*Callinectes sapidus*), stone crabs (*Menippe mercenaria*), other crustaceans, and "thousands of dead fish". This severe mortality was determined to cover at least 228 ha, and was also the cause of considerable sudden mortality of macro-algae (Zieman, 1970).

Roessler and Zieman (1969) found that temperatures of 4°C above ambient killed nearly all fauna normally present. In the area between the +4°C and +3°C isotherms there was a decrease in fish but some increase in mollusks and crustacea. This was found to be a transition zone with a temporary increase in detritus feeders due to the large amount of dead and decaying plant material.

Roessler (1971) later added that the optimum temperature for species diversity of animals and for maximum numbers of individuals was between 26° and 28°C. He found that the 50% exclusion temperature of invertebrates and fish caught by trawling was between 30° and 34°C, and the 75% exclusion temperature was between 35° and 37°C. Roessler found that the decline in abundance covered an area of about 120 ha, although he concluded that in about 50 ha the increased winter catches compensated for the summer decline.

CONCLUSIONS

The work described here has shown that thermal pollution is a real and immediate problem in tropical waters. Roessler and Zieman (1969) showed that heated water pumped into Biscayne Bay has reduced the diversity and abundance of algae and animals in a small area adjacent to the mouth of the effluent canals of Turkey Point. The region where virtually all plants and animals were killed or greatly reduced in numbers corresponded closely with the +4° isotherm and included an area of about 50 ha. A second zone in which algae were damaged and species numbers and diversity reduced corresponded generally to the +3°C isotherm and covered an area of about 68 additional ha. This was the extent of damage in the fall of 1969. Subsequent studies (Roessler et al., 1970; Zieman, 1970; Bader et al., 1972; Roessler, 1971) have shown it to be a just sub-equilibrium condition.

The figures for damage given by Roessler and Zieman (1969) and Zieman (1970) describe mean damage associated with averaged temperatures above ambient. The figures of Purkerson (1973) are maximum damage values associated with extreme short-term thermal maxima. Both are important. The average conditions poise the environment in the sense that they determine the potential components of which the community is composed. However, ultimately the realized community, that which is measured, is dictated by the short-term maxima and minima.

The damage described above was the result of the two conventional power generators at Turkey Point. It was feared that when the nuclear units became operational, the damage would increase dramatically. Added to this was the evident problem of recirculation of the heated effluent. To alleviate these situations, a 9-km canal was dug to Card Sound to the south; however, environmentalists and officials feared that this would simply shift the prob-

lem to a more distant location. As a result of this, construction was begun on a closed cooling system consisting of 270 km of 60 m wide cooling canals, covering some 2,800 ha of former mangrove and marsh area. Thus, it appears the bay has won a reprieve at the sacrifice of the mangroves and marshes.

Despite the fact that ambient temperatures are higher in tropical estuaries as opposed to temperate ones, the upper thermal lethal point of tropical estuarine organisms is not raised proportionally, so that there are often considerable mortalities due to high ambient temperatures caused by natural solar heating. This is coupled with high humidity, which greatly depresses evaporative cooling, and the fact that many tropical estuaries are shallow lagoonal environments which lose excess heat slowly.

Diverse studies in Florida (Roessler and Zieman, 1969; Zieman, 1970; Roessler, 1971), Guam (Jones and Randall, 1973), and Hawaii (Jokiel et al., quoted in Jones and Randall, 1973; Hohman and Tsuda, 1973) show temperatures of 2°C above ambient causing sub-lethal effects and some mortalities, and 4—6°C rises causing severe damage or total destruction of natural communities. These results indicate temperatures in excess of 30—32°C are likely to cause mortality in tropical environments.

They indicate that great care must be used when locating facilities which require extremely large amounts of cooling water, such as electric power plants, on tropical estuaries and lagoons.

In the future careful studies must be made when weighing alternatives of economic and engineering feasibility and potential environmental damage. In particular, more than perfunctory acknowledgement of alternate sites and plant configurations must be made, and they must be available for public scrutiny. In the past, prime consideration was given to the quiet acquisition of desirable land to avoid losses to speculators. Today other considerations must take precedence. In addition plant costs have risen so drastically — power plants often exceed $1 billion now — that land costs are no longer the concern they once were. To avoid repeats of the problems, delays, accusations, and recriminations of Turkey Point, and other power plants, such as the Pt. Aguirre power plant on the south coast of Puerto Rico, considerations must be made well in advance of *all* inputs: economic, engineering, ecological, and social.

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