

# **I. Anthropogenic and Natural Stresses on Coral Reefs in Hawaii: A Multi-Decade**

## **Synthesis of Impact and Recovery**

### *FINAL REPORT*

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**II. ABSTRACT:** In 2002, quantitative photo-transect surveys documenting coral community structure off six coastal sites in Hawaii were repeated to complete long-term data sets of 12 to 30 years duration. Study sites included areas fronting resort development, active and inactive sewage outfalls, and an area where there is no anthropogenic activity, but has been subjected to a variety of storm events. At the only site within a semi-enclosed embayment erosion from surrounding pineapple fields resulted in a decrease in living coral. Such periodic sedimentation in the Bay drives a cycle of damage and recovery that results in coral community structure different than other sheltered embayments in Hawaii. At the other five sites, located in open coastal waters, coral community structure was not adversely affected by shoreline development or discharge of treated sewage effluent. Long-term studies of pristine reefs under natural stress from episodic storms indicate that recovery along the successional continuum varies with time in the different reef zones.

The results of these studies provide a framework for effective and efficient coral reef management in Hawaii. Understanding patterns of natural and man-induced stress and recovery can provide a good model for management strategies, as anthropogenic impacts are superimposed over natural stresses. Our results provide good evidence that management efforts should be concentrated in embayments and areas with restricted circulation. Because such areas comprise less than 10% of the coastal areas, it is concluded that the overall condition of coral reefs in Hawaii is good, and should remain so. While concerns of catastrophic loss from anthropogenic impact to coral reefs are valid in some areas of the world, they do not accurately depict the overall health of coral reefs in Hawaii.

### **III. EXECUTIVE SUMMARY**

In 2002, quantitative photo-transect surveys documenting coral community structure off six coastal sites in Hawaii were repeated to complete long-term data sets of 12 to 30 years' duration. Study sites included areas fronting resort development, active and inactive sewage outfalls, and an area where there is no anthropogenic activity, but has been subjected to a variety of storm events. At Honolua Bay, West Maui, site of the Kapalua Resort, erosion from surrounding pineapple fields following a winter storm in early 2002 deposited sediment on the inner reef that remained in the Bay for at least six months. Between 1992 and our 2002 survey there were significant declines in coral cover on seven of the eight transects, and an overall reduction in coral cover of about 33% throughout the entire Bay as a result of sedimentation. Rainfall records indicate the storm was of relatively small magnitude, and there were no anomalies in wave climate to affect flushing of sediment from the Bay. Periodic sedimentation in the Bay drives a cycle of damage and recovery that results in coral community structure different than other sheltered embayments in Hawaii.

Similar time-series surveys off Mauna Lani Resort, on the West Coast of Hawaii (1983-2002), and Princeville Resort on the north shore of Kauai (1980-2002) revealed consistent pattern of increase in coral coverage at all stations. At these open coastal sites, anthropogenic effects are small in comparison to natural factors that control reef community structure. Lack of major wave events during the interval between surveys may account for the increase in coral cover. Activities from shoreline development appeared to have no effect on coral reef community structure.

Discharge of raw sewage for twenty-two years on the reef off of Sand Island resulted in complete devastation of the coral communities within an elliptical area

stretching about a km downcurrent from the outfall, and another zone of partial destruction that extended approximately 6 km from the point of discharge. Evaluation of reef community structure 25 years after abatement of sewage discharge indicates that coral coverage is presently similar throughout the area of former impact as at control sites. While coral recovery of the stressed area is essentially complete, the species composition of the entire south shore of Oahu has been altered by the wave impacts associated with hurricanes that occurred in 1982 and 1992.

Discharge of highly treated sewage effluent onto a reef environment adjacent to Sandy Beach Oahu, has not resulted in any detectable changes to reef coral community structure. Lack of particulate material in the effluent plume, and rapid mixing of the effluent with ambient water appears to prevent any changes to coral abundance, and the diffuser structures and armor rock are providing a settling site for corals and enhanced habitat for reef fish. Oscillations in coral abundance are similar at the control site and diffuser site, indicating that there is a temporal dynamic variation in coral abundance that is likely a result of damage from episodic storms and determinate life histories of the dominant coral species.

Time-series investigations of a pristine reef track on the West Coast of the Island of Hawaii (Kona Coast) from 1973 to 2002 are elucidating the temporal and spatial patterns of natural stress and recovery that typify open ocean coral reefs in Hawaii. Sequential surveys covering the successional sequence that is driven by storm wave damage and subsequent recovery during prolonged periods of low wave stress show that recovery is not uniform across the reef. Recovery is proceeding sequentially with distance from shore and depth of water, with the shallowest zones

essentially recovered within 20 years. Within the same time frame the deep zones have only begun to recover

The results of these studies provide a framework for effective and efficient coral reef management in Hawaii. Understanding patterns of natural stress and recovery can provide a good model for management strategies, as anthropogenic impacts are superimposed over natural stresses. With the exception of Honolulu Bay, all results from the long-term studies indicate that coral communities have increased, with no indication of any negative conditions. Our results provide good evidence that management efforts should be concentrated in embayments and areas with restricted circulation. Because such areas comprise less than 10% of the coastal areas, it is concluded that the overall condition of coral reefs in Hawaii is good, and should remain so. While concerns of catastrophic loss from anthropogenic impact to coral reefs are valid in some areas of the world, they do not accurately depict the health of coral reefs in Hawaii.

#### **IV. PURPOSE**

In recent years, coral reef ecosystems worldwide have experienced increasing stress associated with human population growth (e.g., Ginsburg 1993, Wilkinson 2000). This is particularly true in Southeast Asia and the Caribbean Basin where anthropogenic stress often exceeds impacts caused by natural disturbance. In Hawaii, and the Pacific Oceanic Islands in general, anthropogenic impacts undoubtedly exist, but the magnitude and extent of ecosystem alteration as in other oceans (Grigg and Birkeland 1997). Nevertheless, Pacific

Island coral reefs are frequently lumped together with reef areas experiencing severe degradation, and are often viewed as ecosystems in crisis (Hodgson and Liebeler 2002). In fact, in Hawaii, there is a strong perception that a variety of anthropogenic activities are seriously altering the structure and function of coral reefs. In part, this view has come about due to a lack of long-term studies on coral reefs and the difficulty in distinguishing impacts caused by natural vis-à-vis man-induced factors. This remains one of the most difficult objectives of coral reef science.

Coral reef ecosystems are normally long-lived, slow growing but ever changing ecosystems often requiring several to many decades to reach successional maturity (Grigg and Maragos 1974, Pearson 1981). Coral reef ecosystems are also notoriously patchy in their pattern of distribution and abundance in both space and time, and are sometimes referred to as spaci-temporal mosaics (Done 1992, Grassle 1972, Bak and Luckhurst 1980). Change in community structure may be subtle, due to long-term but chronic low grade stress (e.g., turbidity), or catastrophic, in the case of large scale episodic events (e.g., hurricane waves), which may have return periods of several to many decades (Dollar 1980, Dollar and Tribble 1993). In the context of time-series surveys, benthic assemblages are often the most useful biological assemblages for direct evaluation of environmental impacts to the marine environment. Because corals are generally long-lived, immobile, and intimately affected by input of potential pollutants, these organisms must either tolerate the surrounding conditions within the limits of adaptability or die.

These are several reasons why long-term data sets or time-series are necessary before the population and community dynamics of coral reefs can be more fully understood. Unfortunately, scientific studies are rarely funded for more than

several years, explaining why so few really long-term (decadal) data sets exist in Hawaii, or for that matter, anywhere.

Over the course of the last several decades, however, we have been fortunate to re-visit coral reef sites in the Hawaiian Islands that share commonalities in terms of their exposure to both natural and man-induced stress. Knowing the general history of both natural and anthropogenic change at these sites, we are now able to correlate both sources of stress to changes in coral reef community structure. In 2001, a grant was received from NOAA, through the HCRI program, which provided a further opportunity to re-visit sites to extend the length of investigation from one to three decades. While these data sets represent relatively long histories, they all suffer from the retrospective nature of the investigative planning. Had the long-term studies been originally planned, re-visits would have been more systematic and frequent. Therefore, the data have some statistical limitations, in terms of clearly establishing cause-and-effect. Nevertheless, the use of these empirical data sets is a unique opportunity to identify dominant patterns of coral communities that can be correlated with both anthropogenic factors and natural forcing.

In this report, the results for six investigations are presented. Three sites are located off areas in Hawaii where the shoreline land use is primarily resort operation. These sites are Honolua Bay on Maui, the reef in front of the Mauna Lani Resort on the Kona Coast of the Big Island of Hawaii, and the reef offshore Princeville Resort on Kauai. All three sites contain large resort complexes that include golf courses that extend to the coastline. The site on Maui is also surrounded by large-scale pineapple agriculture. Two of the sites are located in areas of past and present ocean sewage discharge. For twenty-two years (1955-1977) raw sewage was discharged onto the

reef off of the Sand Island Sewage Plant. Studies conducted in 1975 and 1979 characterized the impacts and initial stages of response to abatement of sewage discharge. A second post-abatement study in 2002 characterized the response of the reef community twenty-five years after cessation of sewage discharge. Since 1965, sewage effluent from East Oahu has been discharged from an outfall located in a coral reef habitat off of Sandy Beach. Currently treated at a tertiary level, the effluent has been directly exposed to the reef environment for nearly four decades. The results of continuing ongoing surveys are included as one of the sites in the present program.

The sixth study site is distinguished from the others in that it is not influenced by any anthropogenic factors. Rather, it provides a unique long-term data set to evaluate the rates of recovery of rich reef habitats from the natural stresses associated with storm waves. As such stresses are the most important determinant of coral reef structure on all open coastal reefs in Hawaii, investigating impact and recovery associated with natural phenomena provides a valuable insight with which to compare the effects of the activities of man. The site (Keawekaheka Point), located on the West Coast of the Island of Hawaii, has been investigated since 1973 and continuing to 2002, through the range of storm intensities that routinely impact Hawaii.

The investigative approach was to re-visit each site and conduct another survey of coral community structure in order to extend the length of the time-series. Changes in community structure were then correlated with site specific histories of both natural and anthropogenic sources of stress unique to each site. In this way, we can attempt to deconvolute anthropogenic from natural forcing and accordingly generate recommendations for coral reef management. Understanding the impacts of human activities on coral reefs is important not only as an environmental issue, but also for

economic considerations. The visitor industry is by far the leading factor in the economy of the State, generating at least \$10 billion annually for the last decade (DBEDT 2000). Understanding the impact of anthropogenic activities is imperative before effective management strategies can be developed.

## **V. APPROACH**

### *METHODS*

A photo-quadrat transecting method, modified after Kinzie and Snider (1978), was utilized to analyze benthic community structure at each study site. A 50-m long transect tape was first paid out along the bottom oriented perpendicular to the shoreline at a constant depth. Care was taken to place transects in "random" locations that were unbiased with regard to coral cover. A rectangular quadrat frame with dimensions of 1-m x 0.66-m was sequentially placed over ten random marks on the transect tape so that the tape bisects the long axis of the frame. The frame is fitted with four legs, which support a small platform on which a Nikonos camera with a super-wide angle lens (15 mm, 94° field of view) is mounted. At each mark, a color photograph was taken recording the segment of reef area enclosed by the quadrat frame. Mounting the camera on the frame ensures exact repeatability of quadrat area. This photographic methodology provides excellent resolution of the detail of the benthic community structure, to the degree that calices of individual corals are distinguishable. The photo-quadrat method also provides a permanent record in the form of photographs.

In addition to the photo-quadrats, diver-investigators with knowledge of the taxonomy of resident species (S. Dollar, R. Grigg) visually estimate, and record on



waterproof data sheets, the percent cover of corals, algae and barren substrata (i.e., sand, limestone, rubble) enclosed within the each quadrat frame. In the laboratory, area coverage of each component of bottom cover in the quadrat photographs is then determined using an overlay grid divided into 200 equally sized segments. The number of segments of each benthic species and substratum type within each grid is summed to calculate area coverage. Thus, for each transect, there is the equivalent of 2,000 data points that in sum contain data on 100% of the area of each quadrat.

Species identification is verified using the “ground truth” information collected in the field. The field data identifications are particularly useful for small and rare organisms. Hence, the method gives accurate estimates of abundance of both common and rare (inconspicuous) organisms. Because virtually 100% of the coverage of each quadrat becomes part of the data record, no information is lost. Few other methods provide for such accurate characterization of benthic community structure. Video transects, for example, which use tabulations of random point intercepts to determine quantitative aspects of reef structure utilize a very small area, even when large areas of reef are surveyed. Hence, they do not give an equally accurate determination of large-scale coral cover or accurate representations of small and rare species.

Results of the photo-quadrats and in-situ cover estimates were used to calculate indices of community structure, abundance and distribution (e.g., percent cover, number of species) and species cover diversity ( $H'c$ ) (Pielou 1966). Because each quadrat is a replicate each transect contains 10 samples. The nonparametric Wilcoxon matched-pairs signed-rank test was used to test for significance between

transects in both space and time (Siegal 1956). All field work and data analysis was performed by S.J. Dollar and R. W. Grigg.

## **VI. FINDINGS**

### *RESULTS AND DISCUSSION*

#### **A. RESORT SITES**

##### *1. Honolua Bay*

###### **a. Results**

Honolua Bay is located on the northwest tip of the Island of Maui (Figure 1). The Bay, along with adjacent Mokuleia Bay was designated as a Marine Life Conservation District (MLCD) in 1978. Honolua Stream flows into the most landward end of the Bay, and as a result there is a submerged paleostream channel through the center of the Bay. On both sides of the Bay, reef platforms extend from the shoreline. The reef platforms terminate in steeply sloping edges that extend to a sandy submerged stream channel.

Of the 140 hectares ( $1.4 \text{ km}^2$ ) comprising the developed watershed above Honolua Bay, approximately 110 ( $1.1 \text{ km}^2$ ) have been cultivated in pineapple since 1953. Prior to pineapple, the land consisted of pasture grazed by cattle. Most of the remaining 30 hectares presently consists of the Kapalua Resort (opened in 1976), which is comprised of three golf courses, three hotels, and substantial residential uses. Because of problems with storm runoff affecting the popular recreational area of Honolua Bay, the Maui Pineapple Company, with support from federal funding, completed twenty-two “Best Management Practices” in the Honolua watershed

between 1994 and 1996, including the construction of diversion ditches, terraces, filters, and siltation basins ((Maui Pineapple Co. 1999).

In 1990, Kapalua Land Co. contracted S. Dollar to implement a marine biological and water quality monitoring program in Honolua Bay in response to concerns that the Bay was experiencing negative impacts owing to shoreline development. In 1992 a second assessment of the coral community structure in the Bay was conducted. In July 2002, coral community in the Bay was again assessed, providing a twelve-year span between surveys (1990-2002). ). Honolua Bay was also one site in a study by Grigg (1994) in which reef community structure within MLCD's was compared to non-protected areas.

Surveys were conducted at four sites, two on each side of the reef platform separated by the submerged paleostream channel. At each site, two transects were completed; one on the top of the reef platform and one on the reef slope (Figure 1). In 1990, from 6 to 12 species occurred on transects, with mean coral cover ranging from  $38 \pm 6\%$  (s.e.) to  $89 \pm 5\%$  (Table 8). Cover was lower on the shallower reef flats and higher on the deeper reef slopes (Table 8, Figure 3). In 1992, mean cover ranged from  $69 \pm 4\%$  to  $89 \pm 3\%$ . When total coral cover of ten quadrats on each transect is compared using the non-parametric Wilcoxon matched-pairs signed-ranks test, there are significant increases on two of the eight transects in 1992 compared to 1990 (two-tailed test,  $P = 0.01$ )(Table 9). Coral cover did not decrease significantly on any transect between these two surveys. Coral cover diversity ( $H'c$ ) was higher on all transects in 1990 compared to 1992 (Table 8).

A winter storm with high rainfall in January 2002 resulted in substantial input of terrigenous sediment to the Bay through stream discharge. Erosion of soil from

cultivated pineapple fields with subsequent drainage to Honolua Bay, has been documented from at least 1964 (Maui Pineapple Co. 1999). Approximately six months after the 2002 storm, the third survey of coral community structure in the Bay was conducted. At the time of the survey (July 2002), deposits of muddy red sediment up to 10 cm thick covered the sand in the inner central channel, as well as some of the reef structure, including dead coral skeletons at the base of the reef slope at the location of Transect I-2 (Figure 4-A). No terrigenous sediment remained on the upper reef platforms, although dead coral was abundant (Figure 4-B, C).

In addition, dense growth of the “golden algae” *Chrysocystis fragilis* (family Chrysophyta) covered much of the outer areas of the reef in the vicinity of Site IV in July 2002. *Chrysocystis fragilis* is a delicate, gelatinous colonial alga that is found on reef flats throughout the Pacific, and reproduces primarily by asexual colony fragmentation (Lobban et al. 1995). In Hawaii, this alga is often observed attached to the base of fingers of *Porites compressa* at depths of 15-25 m during the summer months when wave action is minimal. The slightest water motion is adequate to dislodge the alga from its point of attachment and resuspend fragments in the water column. Normally, *C. fragilis* is removed from the reef during winter months by surge from long period swells, and does not reestablish until calm periods in the summer. During the 2002 survey, *C. fragilis* occurred in the densest aggregations ever observed by the authors on Hawaiian reefs, covering about 26% of the bottom on Transect IV-2 (Figure 4-D). It is possible that the dense aggregations of the alga were overgrowing living colonies of *Porites compressa* smothering the coral, although further detailed observations are necessary to substantiate if it caused coral mortality.

Mean coral cover in 2002 ranged from  $19 \pm 4\%$  to  $62 \pm 9\%$ , and decreased on all transects between 1992 and 2002 (Figure 3). Coral cover decreased significantly on seven of the eight transects between 1992-2002, and on four of the eight transects between 1990-2002 ( $P = 0.02$ ), (Table 9). The greatest decreases in total mean coral cover between 1990 and 2002 occurred at Transects I-2 (54%) and III-1 (60%). Both of these transects are located in the inner Bay where sediment deposition was the greatest. Coral cover diversity ( $H'$ ) showed the same pattern between surveys, with decreases on 4 of the 8 transects between 1992 and 2002, and 6 of the 8 transects between 1990 and 2002 (Table 8).

When the coral cover of all transects is pooled, the most abundant species was *Porites lobata* in the 1990 and 2002 surveys, and *Porites compressa* in 1992 (Table 10). *Montipora dilatata* ranked second in 1990 and 1992 and third in 2002. *Montipora* is known to be especially sediment resistant (Te 1998). *Porites compressa*, which is the normally dominant species in sheltered embayments, comprised only about 30% of total coral cover in Honolulu Bay (Table 10).

#### *b. Honolulu Bay Discussion*

Poor flushing of inner Honolulu Bay by restricted wave-induced currents resulted in prolonged trapping of sediment. The time-series data suggest that coral cover was reduced substantially between 1992 and 2002. Much of the mortality appeared to be a result of the sediment that remained on the reef since the last storm event in early 2002. If the documented reduction in coral occurred as a result of the 2002 storm, it is of interest to evaluate the magnitude of this event in the context of the time-series of

the coral surveys. Based on the remaining sediment in the Bay, and the level of coral mortality, it would be expected that the storm was of unusually high magnitude

In order to evaluate the relative magnitude of the 2002 storm, total monthly rainfall from a rain gauge located in the highest elevation pineapple field that drains into the Honolulu Stream watershed was tabulated from 1953 to 2002 (Field 45, data supplied by Maui Pineapple Co.). In addition, the highest 24-hour total rainfall for peak rainfall months was also noted (Figure 5). While the January 2002 storm produced a substantial flow of terrigenous sediment to Honolulu Bay, it was a relatively small event in the five decades that pineapple has been planted above Honolulu Bay. Total rainfall for January 2002 was 33.5 cm with a peak 24-hour accumulation of 12 cm. In contrast, there have been 55 months with total rainfall greater than 35 cm, and 5 months that have had total rainfall greater than 60 cm. Of these peak events, at least 14 had daily accumulations greater than occurred in January 2002 (Figure 5). The 1990 survey, which reported substantially higher coral cover than the 2002 survey occurred only about two years after the highest recorded 24-hour total rainfall accumulation of 56 cm (April 1988). Similarly, the 1992 survey occurred about one year after a storm that accumulated 19 cm of rain in a 24-hour period (April 1991)(Figure 5).

The record shows that the 2002 rainfall event associated with high coral mortality was not an unusually large storm. Thus, if storm intensity (in terms of rainfall and associated runoff) was the only factor causing coral mortality, it is likely that there would be few living corals in Honolulu Bay. Rather, it seems clear that rainfall *per se* is not the only factor causal of coral mortality. Retention of sediment thick enough to cause smothering is also a function of circulation resuspension, and flushing of sediment from the embayment.

Two other factors also important in flushing sediment from the Bay are long and short period waves. Long-period swell from the north during winter months produces high surf off Lipoa Point at the northern headland of Honolulu Bay. While most of the energy of breaking waves is dissipated prior to reaching the inner Bay, circulation is clearly increased during periods of large surf. Tabulated records of north swells classed by number of days per month of breaking surf between 12-17 feet and greater than 17 feet suggest a slight trend of decreasing frequency of large surf since about 1985 (Figure 6). The winter of 2002 had relatively few days with surf in the 12-17 foot range (maximum of 4 per month), but did have at least 4 days of surf greater than 17 feet (Figure 6). Hence, it does not appear that there was significantly less north swell wave action to flush sediment from Honolulu Bay in 2002 relative to other years.

The second factor that can affect circulation in Honolulu Bay is short-period waves generated by strong south-southwesterly (“Kona”) winds. These events are generated by local weather and occur for short periods primarily during the winter months. Funneling through the channel between the islands of Molokai and Lanai strengthens the intensity of Kona winds affecting Honolulu Bay. While short-period waves generated by Kona winds are generally smaller in size than long-period swells from the north, the angle of approach directly into the mouth of Honolulu Bay may result in more resuspension of sediment in shallow water. A histogram of wind velocity shows that the frequency of south-southwest wind is highly variable (Figure 7). It also suggests that the period following the January 2002 storm was a period of moderate Kona wind intensity. In fact, south-southwest winds were more prevalent during the remainder of the winter of 2002 than following two other winter storms in 1989 and 1991 which did not produce a large affect to coral community structure (Figure 7).

In summary, the time-series records indicate a reduction in coral cover within Honolulu Bay in 2002, but no single factor appears to be responsible. Moreover, it is not likely that the decline is unique in the recent history of the Bay, based on rainfall and wave records, as well as coral community structure. It is also surprising that the sediment impacts to the Bay occurred after 1996 when construction was completed of numerous drainage control structures to the watershed that drains to Honolulu Bay. It is also significant that coral community assemblages in the inner Bay contain a relatively high proportion of sediment resistant species. Overall, it appears that the declines in coral cover are part of a cycle of impact and recovery that has occurred in the Bay over at least the last 50 years. Subtle combinations of wind, rain, and surf may be critical factors regulating the residence time of sediment within inner Honolulu Bay, which appears to be the cause of coral mortality (intermediate disturbance) especially to sediment sensitive species.

## *2. Mauna Lani*

### *a. Results*

The Mauna Lani Resort is located in the South Kohala District on the West Coast of the Island of Hawaii (Figure 1). The Resort encompasses approximately five kilometers of coastline, and presently includes two hotels, two golf courses, and numerous private residences (Figure 2). Because the Mauna Lani Resort was one of the first major resorts on the South Kohala coast, the reef communities in the area have been subjected to the effects of the resort and associated golf courses for a period of about three decades.



The first marine community assessment was carried out by Dollar in 1983. This included five sets of coral transects encompassing the length of the Resort property. Each set included three transects, one in each major coral zone at depths of approximately 6, 10 and 20 m (Dollar 1982, Dollar and Tribble 1993). In preparation for the proposed construction of an inland marina (which was never built) the entire benthic survey was repeated in 1993. It was replicated a third time in August 2002 as part of the HCRI research program. During the 1993 and 2002 surveys, a sixth transect site was added off the community of Puako, to the north of the Mauna Lani property. This site was added to serve as a reverse control for possible impacts of sewage disposal. The entire coastal community of Puako utilizes cesspools and septic systems for domestic waste disposal since there is no municipal sewage system serving the area. Following treatment processing within the septic systems and cesspools, domestic sewage leaches to groundwater and eventually enters the coastal ocean. In contrast, at the Mauna Lani Resort all domestic sewage is processed at an on-site treatment plant and the effluent is used for golf course and landscape irrigation.

The western coastline of the Island of Hawaii is typically not exposed to long-period winter waves generated by north Pacific storms, owing to protection from the island of Maui. Occasional “Kona” storms from the southwest can and do impact the area (Dollar 1982) but none occurred between 1993 and 2002. As a result, the West Hawaii coastline has been relatively free of disturbance from physically destructive waves for the last two decades. In addition, the coastal region of West Hawaii is one of the driest areas of the entire state, and contains no permanent streams. When

intense rainfall events occur most of the rain percolates through the lava surface that comprises the coastal zone, with little surface runoff reaching the ocean.

The physical setting of the Mauna Lani reef tract consists primarily of a narrow, shallow basaltic bench that extends from the shoreline approximately 50-75 m offshore, where it terminates in a nearly vertical basaltic cliff face that drops to a deeper reef platform at a depth of approximately 8-10 m. The reef platform slopes gradually to the limit of coral growth (~25 m) and terminates in a sandy plain. Two of the transects sites (II and V) were located directly off man-made crescent beaches which have been nourished with calcium carbonate sand. At each site, three transects were established: one on the shallow bench at approximately 6 m, one at mid-depth at approximately 10 m on the reef platform near the juncture of the cliff; and a third one about 20 m deep on the outer slope of the reef platform. The zonation of coral community structure is typical of West Hawaii (Dollar 1983), with the shallow transect zone dominated by the pioneering species *Pocillopora meandrina* (Figure 4-E), the mid-depth transect zone dominated by various growth forms of *Porites lobata* (Figure 4-F), and the deep transect zone covered with a mixture of *P. lobata* and interconnected mats of *Porites compressa* (Figure 4-G).

Replicate surveys at three depths at six sites fronting the Mauna Lani Resort showed increases in mean coral cover at all 18 transects between both 1983-2002 and 1993-2002 (Table 4, Figure 8). Increases were significant ( $P = 0.02$ ) on 11 of the 18 transects between 1993-2002 (individual quadrat data were not available for the 1983 survey) (Table 2). There were no significant decreases in coral cover on any transect. Overall, increases in coral cover were greatest on the 6 and 10 m transects. Mean coral cover on the deep 20 m transects was between 92% and 98% of bottom

cover at all six sites in 2002. Of the three surveys, coral cover diversity ( $H'$ ) was highest in 2002 on ten of the eighteen transects, and highest in 1983 on the other eight transects (Table 3). Four transects, all of which were at depths of 20 m (I-3, III-3, IV-3, V-3), displayed a sequential increase in diversity over the three sampling periods. Two transects, both in the shallow 6-m depth zone (III-1, VI-1), had progressively decreasing diversity with time.

#### *b. Discussion*

The ranked abundance of pooled coral cover data showed a consistent pattern between all three surveys in order of *Porites lobata*, *Porites compressa* and *Pocillopora meandrina*. Together, these three species comprised 95% (1983), 97% (1993) 92% (2002) of coral cover. Compared to Honolua Bay, species of *Montipora* were far less abundant on the Mauna Lani transects, with a total pooled coral cover of about 1% in 1983, 2% in 1993 and 7% in 2003.

Coral community structure at Site IV, located off of Puako were consistent with the other sampling stations, with mean total cover of 49%, 86% and 98% at the 6, 10 and 20 m transects, respectively. Mean coral cover increased at all three depths between 1993 and 2002, with significant increases on the 10 and 20 m transects (Table 2). These results indicate that there is no apparent negative effect to the coral community from cesspools and septic systems along the shoreline.

The coral communities off the Mauna Lani Resort can be considered thriving, with little or no apparent impact from anthropogenic activities related to either resort or private residential development. Vast tracts of undisturbed *Porites compressa* indicate that destructive wave events have not occurred over the last two decades. The lack of

wave disturbance, along with no other impacts such as sediment input has allowed the community to attain a near climax stage of coral reef succession (Grigg and Maragos 1974) with nearly all available substratum colonized.

### 3. *Princeville*

#### a. *Results*

The Princeville Resort is located on the northern coastline of the Island of Kauai, which is the northernmost of the main Hawaiian Islands (Figure 1). The first phase of the resort included a hotel and golf course and was completed in 1971. As part of the Environmental Planning documentation for a second golf course, a study was commissioned in 1980 to compare the conditions of reefs off the existing golf course to control areas in order to evaluate impacts from the existing resort (Grigg, unpublished manuscript). The same study was repeated in 1995 for a projected new phase of resort development. A third investigation was conducted in June 2002 as part of the HCRI program.

The oceanographic setting of Princeville is strikingly different than either of the other two sites described in this paper, in that the reefs there are directly exposed to long-period swell generated from winter storms in the North Pacific. Hence, reef development is restricted to species assemblages that are tolerant of wave concussive forces and sediment scour that result from surf that occurs each winter.

The nearshore physiography off most of Princeville consists of a wide (up to several hundred m's') shallow (1-2 m) reef flat that was formed during the Holocene over the last 7,000 years (Easton and Olson 1976). The reef flat is covered with sand and rubble and is bisected by numerous sand filled channels that run offshore

perpendicular to the shoreline. There are also submerged streambeds from several existing streams that flow to the shoreline. Coral cover on the reef flat is sparse with living colonies comprising no more than about 2% of bottom cover. The seaward edge of the reef flat terminates in a narrow barrier reef that is subaerial at low tide, and absorbs much of the force of breaking waves. Seaward of the barrier reef, the reef front zone consists of spurs and grooves that extend to a depth of approximately 8-10 m. Between the spur and grooves, the bottom topography is relatively flat and gradually slopes into deeper water. Coral growth reaches a maximum at a depth of about 12 meters.

Six survey sites at a depth of about 12 m on the submarine terrace were evenly spaced along 8 km of coastline fronting Princeville from Hanalei Bay to Kalihiwai Bay (Site VI was not surveyed in 1995)(Figure 2). Only one transect was conducted at each site because coral cover was generally restricted to a single zone on the reef. Total mean coral cover increased on all transects on each succeeding survey (Table 5, Figure 9). Between 1980 and 1995 the increases in coral cover were significant on two of the five transects. Between 1980 and 2002 the increases were significant on all six transects (Table 2). Coral cover diversity showed no consistent pattern of change over the three surveys (Table 5).

#### *b. Discussion*

Ranking pooled coral cover showed *Porites lobata* comprised the highest percentage of coral cover in 1980 and 1995, while *Montipora patula* was the most abundant coral in 2002 (Table 3). Both of these species occurred primarily as flat encrustations on the limestone platform (Figure 4-H). *Porites compressa*, which was a major component of

the coral assemblages at Honolua Bay and Mauna Lani was relatively scarce at Princeville, comprising less than 10% of coral cover during all three surveys (Table 3).

As with Mauna Lani, these data indicate that anthropogenic activities on land are not affecting coral community structure, as there have been continuous increases in coral cover during the last two decades. While it could be argued that resort development is contributing to the increased coral coverage by mitigating surface runoff, it is probable that sediment effects on the reef have always been minor, since most of the sediment carried to the ocean by streams is deposited on the reef flat or carried out to sea during storm and high wave events. It is possible that the increases in coral cover are a response to a slight decrease in the frequency of large surf from winter swells since 1985 (Figure 6).

## B. SEWAGE OUTFALL SITES

### 1. *SAND ISLAND*

#### a. Background

Rapid increases in urban development and human populations in recent decades have intensified the problem of sewage disposal throughout the world. In the United States, the Federal Water Quality Control Act of 1972 (Act 92-500, the so-called “clean water bill”) required that all sewage discharged into navigable waters receive secondary treatment. Subsequent amendments to the bill allow for waivers of the secondary treatment condition if specific criteria are met regarding the treatment of effluent and effects of discharge to the receiving environment.

In Hawaii, the largest sewage treatment plant is located at Sand Island Oahu. From 1955 to mid 1977, about one-half the domestic sewage from the island was

discharged with virtually no treatment from the “old” outfall, which extended approximately 1,100 m (3,600 feet) from shore, and terminated at a depth of approximately 11 m (35 feet). As a consequence of the Federal Water Quality Control Act of 1972, the City and County of Honolulu completed construction of a new treatment plant and outfall in 1978. The new outfall discharges advanced primary treated sewage approximately 2,750 m (9,000 feet) from shore at a depth of 67 to 73 m (220-240 feet) (Figure 10). Since completion, the new deep-water outfall has been the sole source of sewage discharge in the Sand Island area, thereby removing the stress of sewage effluent discharge from the shallow marine communities.

In 1975, R. W. Grigg conducted a survey to evaluate quantitatively the impact of raw sewage on the coral reefs in the vicinity of the Sand Island Outfall (unpublished manuscript). In brief, results of the study revealed an area of significant impact, which extended approximately 1,000 m east, and about 4,000 m west of the point of discharge. Within this area, corals were totally lacking within 500 m of the outfall, and mounds of deposit-feeding worms (*Chaetopterus* sp.) dominated the epibenthos. The worms dominated over 1,000 acres (4 million m<sup>2</sup>) of bottom (both hard bottom and sand). The highest diversity of macrobenthos was observed at stations of intermediate distance from the outfall because species dominant in non-stressed communities were less abundant owing to moderate levels of sewage, while species directly associated with the outfall were also present. The information generated by the 1975 study serves as a “preabatement” database to determine conditions of the coral reef community during the period when sewage discharge was occurring.

In 1979, the Dept. of Public Works, City and County of Honolulu, funded the Water Resources Research Center at the University of Hawaii at Manoa to conduct a

second survey of the reef area off Sand Island to determine changes to benthic communities approximately two years after the diversion of sewage to the deep outfall (Dollar 1979). This study replicated the methods and sampling protocols used in the 1975 study. In brief, results of the first “post-abatement” study a clear pattern with respect to distance from the outfall. *Chaetopterus* worms, which were the dominant biota in the area affected by sewage discharge were completely absent within two years of sewage abatement. Following sewage abatement, a high-impact zone, which roughly corresponded with the area where *Chaetopterus* worms were previously present, was characterized by physical degradation of the reef framework. An intermediate impact zone was delineated by an area where the reef was clearly stressed, with many of the primary framework builders (e.g., corals) dead, although the physical structure of the reef remained at least partially intact. Settlement of new corals in the high impact area was also documented in 1979, with the highest abundance of new recruits settling on the old outfall structures and surrounding armor rock. The intermediate impact zone was characterized by a small percentage of living corals; rather the intact reef framework was almost totally covered with a veneer of red, encrusting coralline algae.

In 2002, as part of a project funded by the Hawaii Coral Reef Initiative, the area was surveyed a third time. The results of the second “post-abatement” survey conducted approximately twenty-five years after the cessation of sewage discharge reveal the long-term status of recovery of the reefs that were severely impacted by sewage stress.



## *b. Results*

Probably the most impressive result of the 2002 survey was the degree of coral colonization that has taken place on the old diffuser structure during the past 25 years. During the 1979 study several small colonies of corals were observed on the predominantly barren discharge structures and surrounding armor rock (Figure 11-A). In 2002, coral, predominantly *Pocillopora meandrina* (Figure 11-B, C), covered virtually all of the available space on the concrete diffusers. Similarly, the armor rock at the base of the outfall diffusers was nearly fully colonized by corals.

It is of interest that many of the larger colonies of *P. meandrina* were dead, and encrusted with calcareous algae (Figure 11-B, D). Similar observations of dead colonies of *P. meandrina* have been made at all of the other sites investigated for this study. As a pioneering species, *P. meandrina* is the first to settle new substratum, or settle in areas that may be too harsh for other species. The species, however, clearly exhibits a determinate growth form, with some as yet unidentified natural trigger that kills the colonies after a certain time, or upon reaching a certain size. The large percentage of dead colonies on the old Sand Island sewage structure suggests that the area is in what may be termed a “secondary recovery period.” The occurrence of numerous small living colonies suggests that the mortality is not related to physical/chemical characteristics of the habitat.

Quantitative benthic surveys were conducted at eight sites ranging from 4 km east and 6 km west of the old outfall (Figure 10). Sampling locations were all at the depth of the outfall (11 m), and replicated the stations from the 1975 and 1979 surveys as closely as was possible. Results of benthic transects for the three surveys are

shown in Table 6. Total coral cover, number of coral species and species-cover diversity are plotted as distance from the outfall in Figure 12.

The most conspicuous result that is evident in Figure 12 is that the zones of impact that were evident in 1975 and 1979 do not persist in 2002. The area of high impact (SI-2 –SI-5) characterized by a barren pitted limestone platform in 1979 (Figure 13-A) is now colonized by a variety of coral species (Figure 13-B). The region deemed to be the intermediate impact area (SI-6 – SI-7), was characterized by a veneer of calcareous red encrusting algae in 1979 (Figure 13-C). Twenty-five years later, the algal veneer remains over much of the reef surface, although corals have recolonized the area (Figure 13-D).

Number of coral species also is equal to, or higher in 2002 compared to previous studies at all of the transect stations within the zone of influence of the outfall (S-2 – S-7)(Figure 12). Species cover diversity in 2002 is nearly uniform at all eight transects (Figure 12). In contrast, during the preabatement survey (1975) diversity was highest at the boundaries of the zone of impact, owing to an assemblage of “normal” species and sewage tolerant species. In the first post-abatement survey (1979) diversity also peaked at the site of the diffusers, owing to initial colonization of the concrete diffuser structures.

Examination of percent cover of the four most common species (*Montipora capitata* and *Montipora patula* are combined) also provides some important information on the recolonization of the reef. *Porites compressa*, which was the second most abundant coral at the stations beyond the zone of influence during 1975 was absent from the reef at a depth of 11 m in 2002. On the other hand, *Pocillopora meandrina* , which was one of the dominant corals in 2002, was extremely rare at all

stations in 1975 and 1979. *Porites lobata*, which is generally the most ubiquitous coral on most Hawaiian reefs peaks in abundant at the stations beyond the influence of the outfall during all three studies (Figure 14).

### *c. Discussion*

The pattern of species abundance that has been characterized at the old Sand Island site may reflect physical stresses other than sewage discharge that have affected the reefs off the south shore of Oahu. Hurricanes Iwa (1982) and Iniki (1992) both produced anomalously large surf that caused serious damage to these reefs (Grigg 1994). Documentation of storm damage to Hawaiian reefs (Dollar 1992, Dollar and Tribble 1993) has shown that the most susceptible coral to concussive force is *Porites compressa*. In addition, the most resistant coral, as well as the species that recruits first to newly bared substratum is *Pocillopora meandrina*. *Porites lobata* and *Montipora* spp. occur in flat, encrusting growth forms that are also relatively wave tolerant. Hence, the pattern of coral abundance that presently (at least in 2002) occurs in the vicinity of the old Sand Island Sewage Outfall is likely more a response to relatively recent peak wave events than to the remnant effects of the sewage discharge.

The pattern of effects on benthic community structure close to the old outfall, and predominant current patterns provided convincing evidence that serious impacts were related to effluent discharge. Bioerosion, and calcareous algal colonization altered the framework of the substratum to varying degrees, and the degree of bioerosion was likely a controlling factor in the timing of the biotic response to the relaxation of sewage stress. However, with passage of several decades following the

termination of the stress, the conditions largely returned to a background setting that is similar to areas not influenced by the sewage discharge. It is also important that the outfall structure itself is presently supporting a community at least as rich as the surrounding natural reef substratum.

In light of the current regulatory climate, as well as common sense, it is highly unlikely that discharge of sewage effluent will occur again on the shallow reefs fronting Honolulu. The data collected over a two and half decade time-span indicate that the severe impacts to the reef from twenty-two years of discharge of raw sewage have largely been mitigated by the process of natural recovery. It is also apparent that the reef assemblages are also affected by the two hurricanes that have occurred since the cessation of sewage discharge, which have affected the entire south shore.

## *2. East Honolulu Wastewater Treatment Facility*

### *a. Background*

Since 1965, the privately operated East Honolulu Wastewater Treatment Facility (EHWWTF) has been discharging up to 15 million liters per day of blended secondary/tertiary treated domestic sewage effluent through a multi-port diffuser at a depth of approximately 12 m. The discharge is located less than one kilometer from a high-use recreational area at Sandy Beach (Figure 15). The habitat where the diffuser is located consists of a calcium carbonate platform populated by an abundant coral community. Between 1987 and 2002, S. Dollar has carried out 75 repetitive benthic monitoring surveys (every other month from 1987 to 1999; quarterly from 1999 to present) at transects in the vicinity of the outfall and control stations, as a requirement of the NPDES discharge permit. The repetitive monitoring program is designed to

determine if the discharge of sewage effluent results in changes to reef community structure that do not occur in similar communities which are not exposed to effluent. The frequency and duration of the monitoring are extensive enough that long-term changes can be recognized.

The EHWWTF provides an important contrast to the situation at Sand Island during the period when the old treatment plant and outfall were in use (1955-1977). During this period, untreated effluent was discharged on a shallow reef environment. In contrast, the effluent discharged from the EHWWTF is treated to a very high level, such that most of the particulate material is removed. While the receiving environments at Sand Island and Sandy Beach are somewhat different regarding physical and oceanographic conditions, it is still possible to evaluate the effects of discharge of untreated vs. highly treated effluent on coral reef habitats.

#### b. Results

Throughout the monitoring program, no particulate material has been observed on the ocean surface, nor was there any odor associated with sewage effluent noted from any location during the entire survey. During the survey there was a visible surface slick directly over the diffuser. Inspection of the effluent jets emanating from the diffuser ports revealed virtually no large particulates. Rather, the effluent jets appeared to consist entirely of uncolored freshwater plumes that were distinguishable primarily as a result of "shimmering" as the freshwater mixed with the saline ocean water. The effluent plumes rose slightly in the water column and dispersed within several meters of the diffuser. Because there was no current at the time of the survey, the entire water column in the vicinity of the diffuser was somewhat turbid compared to the

control area. The overall appearance of the effluent plumes had not changed to any noticeable extent compared to surveys conducted in the previous years (Figure 16).

Large schools of butterflyfish (*Chaetodon miliaris*), and mixed species schools of surgeonfish (Acanthurids) are consistently observed feeding in all of the effluent jets (Figure 16). Large schools of up to several thousand of the blue-lined snappers (*Lutjanus kasmira*) were observed in the immediate area of the diffuser pipe. Water clarity in the area of both the diffuser and the control site is generally similar during all surveys in 1995-2002, with no noticeable increase in turbidity from sewage discharge.

Coral community composition has been evaluated at three transect sites; one site is located immediately adjacent to and perpendicular to the end of the diffuser (transect DD); one site is located adjacent to and perpendicular to the outfall pipe just landward of the diffusers (SO); and a control site is located approximately 2 km to the southwest (CON)(Figure 15). Plots of percent cover of total coral and the three most abundant species as a function of time from September 1987 to November 2002 (the period when the current survey methodology has been utilized) are shown in Figure 17. During the most recent survey conducted in November 2002 survey, mean total coral cover was 34.9% at the deep diffuser, 36.6% at the shallow outfall, and 22.3% at the control site. Examination of Figure 17 shows that there is a distinct oscillation of coral abundance at all three survey sites over the fifteen years of monitoring. In order to evaluate if these oscillations indicate statistically significant changes in cover, linear regression statistics for total coral cover, as well as cover of the three most abundant species, were conducted (Table 6). Regression analyses were conducted for the entire data set (1987- 2002)(n=75), and four separate periods of surveys within the fifteen-year period of monitoring. Periods from which regression analyses were

conducted are from the initiation of monitoring in June 1987 to September 1993 (n=30); November 1993 to October 1998 (n=30); December 1998 to November 2000 (n=10), and the last two years of the program from November 2000 to November 2002 (n=6). One criteria for the selection of these periods was the occurrence of a severe storm that impacted the eastern shore of Oahu in November 1998. The effect of the storm on coral cover, depicted as a sharp decline in cover, is clearly evident in Figure 17.

Slopes of linear regression lines significantly different from zero ( $p < 0.05$ ) indicate that coral cover has decreased (negative slope) or increased (positive slope). Slopes not significantly different from zero indicate that mean coral cover has remained the same during the interval of time over which the test was conducted.

During the course of the entire monitoring program 1987-2002 (n=75 surveys), linear regression statistics indicate that total coral cover has not changed at the deep diffuser, but has decreased at the control station. Cover of the most abundant coral species, *Porites lobata*, showed a decrease at the deep diffuser, and also decreased at the control station in the 1987-2002 interval. The second and third most abundant species, *Pocillopora meandrina* and *Montipora patula* show no change at either the deep diffuser site or control site (Table 7).

Considering only the 1987 to 1993 surveys (n=30), linear regression analyses show significant decreases in total coral cover and individual species cover (except *Porites lobata*) at the deep diffuser site. At the control site total coral cover and cover of all species did not change significantly (Table 7). As storm effects would be expected to affect both the diffuser site and the control site, these results suggest that factors other than breakage from storm surf could be responsible for a decrease in

cover adjacent to the diffuser. An alternate explanation is that the control site was not subjected to the same storm effects as the region where the diffuser is located.

When regression analyses are conducted on the survey data collected between 1993 and 1998, the results are substantially different from the previous period. During the course of 30 surveys over the five-year interval, total coral cover and cover of the most abundant species (*P. lobata*) increased at all three sites. Of the other two species, cover increased at two of the three sites, and remained unchanged at the third site (Table 7). Hence, over this five-year period, any factors that might have previously caused decreases in coral cover at the diffuser sites were eliminated. Such elimination of negative environmental factors, as well as the lack of any severe storm events resulted in a period of recovery, with consistent significant increases in coral cover. As the increases in cover were similar at the control site and the sites in proximity to the diffuser, it is apparent that there was no negative impact from the sewage discharge to either total cover or individual species of corals.

Considering surveys conducted only in 1998-2000 (n=10) conducted after the storm of November 1998, linear regression statistics indicate a significant increase in total cover at the deep diffuse and shallow outfall stations, and no change at the control site. Changes in cover of individual species either increased or showed no change at the two sites adjacent to the diffuser, and showed consistent lack of change at the control site (Table 7).

Over the last two years, from November 2000 to 2002 (n=6) the pattern of change in coral abundance reversed again, with no increases in cover at any of the sites for either total cover or individual species (Table 7). Total cover at the deep diffuser remained unchanged, and decreased at both the shallow outfall and control



site. The most abundant species, *Porites lobata* decreased at both the deep diffuser and shallow outfall sites and remained unchanged at the control. Cover of *Pocillopora meandrina* showed no change at the diffuser sites, but was significantly reduced at the control site (Table 7).

The decreases in coral cover over the last two years, which are not related to the acute destructive force of an episodic storm event, appear similar to the gradual decrease noted in the 1987-1993 period. It is also apparent that the decreases are greater at the control site than at either of the sites adjacent to the diffuser. In particular, cover of *Pocillopora meandrina* has decreased far more at the control site than at either of the outfall sites. Over the last six surveys, cover of *P. meandrina* has progressively declined at the control site, from about 18% of bottom cover in March 2001 to 7% in November 2002. Observations of the reef at the control site revealed a substantial amount of dead, but intact, *P. meandrina* colonies. However, numerous dead colonies were not observed at the two survey sites adjacent to the diffuser. Transect data showed 20% of bottom cover at the control site consisted of dead *P. meandrina* colonies, for a live:dead ratio of about 1:3. In comparison, the live to dead ratio of *P. meandrina* at the deep diffuser site was about 13:1, and about 3:1 at the shallow outfall site.

The decline in cover of living *Pocillopora meandrina* at the control site does not appear to be the result of any man-induced changes in environmental conditions. Rather, the most plausible explanation for the decline of *P. meandrina* at the control site has to do with the natural life cycle of this species. *Pocillopora meandrina* is a “pioneering” species that is the first to colonize areas of newly bared substratum, and can exist in environments that may be too physically harsh for other species. In

addition, as described above, *P. meandrina* has a “determinant” life history, meaning that each colony appears to have a discrete life-span, and that the death of a colony will naturally occur at the end of this span if other environmental stressors do not cause mortality first. To illustrate, virtually no colonies of *P. meandrina* in Hawaii occur (dead or alive) that are larger than about 30 cm in diameter. Other genera that are common in Hawaii, such as *Porites* do not have such a determinate life span, and colonies can persist for decades, or even centuries, and can attain colony size of up to several meters if not disturbed by physical factors.

It is likely that the high percentage of dead *P. meandrina* colonies at the control site is a result of “year classes” of large colonies reaching the end of their life span nearly simultaneously. With the lack of a large storm to prune back coral cover over the last decade, growth of *P. meandrina* has progressed to the natural “end-point.” In contrast, cover of *P. meandrina* has been consistently lower at the deep diffuser and shallow outfall sites compared to the control site. Hence, the lower cover of smaller colonies has not exhibited the same natural decline as observed at the control site.

The overall results of the regression analyses suggest that there is an oscillation of coral cover that is a result of natural factors. It can be concluded that the oscillations are from factors other than discharge of sewage effluent because the oscillations are more pronounced at the control site than at either of the sites adjacent to the diffuser. In addition, oscillations are defined as alternating periods of decreases and increases in cover at all three sites. If the effluent was exerting a negative effect on the coral community structure, it would be expected that there would be a consistent decrease in cover, as the discharge of effluent has been constant over the monitoring period.

The long-term repetitive data set that has been compiled by the monitoring program provides data that confirms that the oscillation in coral abundance is a result of natural factors, including physical stress brought about by intermittent episodes of large surf, and determinant growth of a dominant species. As the patterns show changes that are not isolated at the diffuser site, it is clear that the fluctuations in coral abundance are not a response to discharge of sewage effluent.

The results of this survey indicate that the coral communities in the vicinity of the East Honolulu WWTF Ocean outfall do not show indications of impact from discharge of sewage effluent. Patterns of oscillations of coral abundance suggest that the coral community is not being negatively affected by the discharge of sewage effluent. Rather, the patterns in coral cover are the result natural factors, which affect both the outfall and control sites. A primary cause of variation in coral cover is damage from episodic storm waves. In addition, with a long between-storm interval, decline of cover at the control site is likely a result of the determinant growth mortality of a dominant species (*Pocillopora meandrina*).

Observations of apparently healthy corals growing on the diffuser pipe directly in the flow path of the effluent plumes also argue against negative impacts from the effluent on benthic community structure (Figure 16). In addition, the presence of the outfall structure and effluent jets appears to be responsible for aggregations of reef fish around the discharge ports that far exceed the control area. Effects to benthic organisms by the effluent are also minimized as the effluent plume consists of freshwater which rises in the water column with little contact with the bottom. In addition, there is no evidence of deposition of sewage material on the reef surface. As

a result, there does not appear to be any mechanism for causing alteration to coral growth.

There is also no visual indication of any disease or pathological abnormalities with any of the biota in the area of the discharge, and the outfall structures and effluent plumes continue to attract reef fish which use the diffuser structure as shelter, and may feed on particulates in the water column. Chlorination of the effluent result in a substantial reduction in indicator bacteria related to potential public health problems but does not have any apparent effect on marine biota in the vicinity of the discharge.

It is also important to note that there is no indication of increases in opportunistic species that appear in response to sewage stress, such as was observed during discharge of raw sewage from the old Sand Island outfall. In particular, species of macrothalloid algae and filter-feeding worms that have been associated with other ocean sewage discharge systems in Hawaii are totally absent from the East Honolulu Wastewater Treatment Facility Outfall discharge site. The absence of such organisms indicates that there continues to be no major changes in reef community structure from outfall-related factors.

## C. STORM IMPACT and RECOVERY

### 1. *Keawekaheka Point*

#### a. Background

The effects of severe disturbances have provided overwhelming evidence that coral reefs are not always stable, mature communities controlled only by biotic interactions in a space-limited environment (e.g. Endean 1976). Many coral reefs fit the description of "temporally varying mosaics" (Grassle 1973, Bak and Luckhurst 1980) in which the

community undergoes a continual cycle of disturbance (removal) and recovery (renewal). Superimposed on this cycle is either deposition or destruction of the calcium carbonate framework that defines reefs as geologic, as well as biotic, entities.

The effects of many severe physical and biotic disturbances that drive this cycle have been well documented for specific reef areas. Quantification of the destructive removal phase of the cycle depends largely on fortuitous circumstances: specifically on having established study sites prior to unpredictable disturbances. Follow-up surveys provide an instantaneous picture of the relationship between the intensity of the event and the change in the community.

The second phase of the removal-renewal cycle is much more difficult to evaluate. In some situations, a few years may be sufficient to encompass much of the recovery process. For example, some reefs in the Florida Keys appear to recover from hurricane damage within 5 years (Shinn 1976). However, because corals are relatively slow growing and long-lived, the successional process of most reefs takes place on a scale of decades (e.g. Grigg and Maragos 1974). Thus, evaluating recovery of many reefs from disturbances is a long-term process. Unfortunately, much of the existing literature documenting the recovery of reefs has been limited to the several years of regeneration, which does not always constitute a substantial part of the disturbance-recovery cycle. Hence, the predictive theories of reef recovery from severe disturbances have not been completely tested.

A predominant force shaping the structure and composition of reefs exposed to the open ocean is mechanical stress (breakage and abrasion) from storm waves. Wave energy has long been recognized as one of the most important controls of coral growth and subsequent reef development and coral island formation (Darwin 1842,

Storr 1964, Roberts 1974, Adey 1978). Yonge (1940) pointed out that wave energy and water turbulence in particular correlates to conspicuous reef zonation and segregation of organisms. Hurricane-driven seas frequently strike the leeward side of reefs where there is usually marked development of branching or fragile coral colonies, thus resulting in more severe effects than would be caused on windward reefs (Endean 1976). Qualitatively, impact and recovery from storms is different from events such as *Acanthaster* kills (e.g. Colgan 1987), and thermal damage (e.g. Brown and Suharsono 1990) in that the entire structural framework of the reef, and not just the living coral tissue, is often altered or destroyed in a very short span of time.

The effects of catastrophic wave stress are reported from many locations in the Atlantic (e.g. Stoddart 1963, 1969, 1974, Glynn et al. 1965, Perkins and Enos 1968, Ball et al. 1967, Shinn 1976, Hernandez-Avila et al. 1977, Highsmith et al. 1980, Rogers et al. 1982) and Pacific (e.g. Banner 1961, Blumenstock 1961, Blumenstock et al. 1961, Cooper 1966, Maragos et al. 1973, Randall and Eldredge 1977, Ogg and Koslow 1978, Dollar 1982, Pfeffer and Tribble 1985, Done et al. 1986, Harmelin-Vivien and Laboute 1986, Loubersac et al. 1988). The effects of Hurricane Allen, which struck the north coast of Jamaica in 1980, were thoroughly documented, because of preexisting reef study sites in the vicinity of the Discovery Bay Marine Laboratory and also because of the unusual nature of the event (first hurricane since 1944) (Knowlton et al. 1981, Porter et al. 1981, 1982, Woodley et al. 1981, Kjerfve et al. 1986, Liddell and Ohlhorst 1987).

In contrast, there are relatively few reports that quantitatively describe reef recovery from storm effects (e.g. Shinn 1976). Indeed, in a review of recovery and recolonization of coral reefs Pearson (1981) states, "In examining the literature it is

surprising to find that there are almost no quantitative studies of recovery following cyclone (=hurricane, typhoon) damage." Although reef community recovery depends on the magnitude and frequency of disturbance events, it is clear that the process also depends on a number of interrelated ecological elements, including environmental tolerances and life history traits of dominant species, and secondary disturbances such as disease, predation, and bioerosion (Hughes 1989, Brown and Suharsono 1990, Glynn 1990). Because of the potential complexity of such interactions, "simple" reef systems may provide a more intelligible natural laboratory to investigate disturbance-recovery cycles. The Hawaiian Islands provide such a setting. Owing to geographical isolation from Indo-Pacific centers of coral evolution (Grigg 1983), species composition is restricted to 44 hermatypes, with only 5 species being quantitatively significant components of reefs (Maragos 1972). As a result, coral communities in Hawaii are simple assemblages. In addition, most Hawaiian reefs are fringing and exposed to a wide range of sea and swell. Grigg and Maragos (1974) set out the premise that Hawaiian reefs are physically controlled communities in which species tend to be generalists with broad niches. As a consequence, community structure of coral reefs in Hawaii exposed to the open ocean is primarily a function of the length of recovery time between disturbances (Grigg 1983).

Dollar (1982) documented the effects of storm damage to coral reef communities off Keawekaheka Point on the West Coast of the island of Hawaii (known locally as the Kona Coast) in 1973, 1974 and 1980 (Figure 18). To determine the extent of change of the coral communities since 1980, the same area was surveyed again in 1992 and 1993, providing a data set that spanned a 20- year period. This time frame covered most of the spectrum of intensity associated with wave disturbance.

Dollar and Tribble (1993) described the relationships that existed during these two decades, between physical (wave disturbance), biotic (coral community structure), and geologic (deposition of calcium carbonate from coral growth that results in reef accretion) processes in an environment exposed to periodic, but catastrophic wave stress.

As part of the HCRI program, the same reef tract was again surveyed in 2002, providing a 30-year record of impact and recovery. This record probably provides the longest continuous investigation of a single area in the Hawaiian Islands, and thus provides a unique opportunity to examine impact and recovery from natural stresses on coral reefs in Hawaii.

As responses of reef systems to human-induced stress is superimposed on natural factors of impact and recovery, a good understanding the response of reef systems to natural stresses is an important aspect in evaluating the effects of human activities.

## c. Results

### *i. Wave Climate and Survey Sequence*

Detailed descriptions of the wave climate of west Hawaii are given by Dollar (1982). West Hawaii is completely sheltered from northeast tradewind-generated seas, but is exposed to four other types of waves: 1) south swells, 2) north swells, 3) locally generated storms known as "Kona storms", and 4) hurricanes. Figure 19 shows deep-water wave heights of each type of storm as a function of return period.

South swells of 1-3 m in height, generated by storms in the South Pacific in the northern hemisphere summer, are predictable seasonal events in West Hawaii that occur at least several times each summer. North Pacific storms routinely produce



waves during the winter months that reach Hawaii with heights up to 10 m. While large north swells regularly impact coastlines with northern exposures, the islands of Maui, Molokai and Lanai block most North Pacific swells from reaching west Hawaii. Only the occasional winter storms that form at unusually low latitudes and have strong westerly components produce large waves that affect the study area. Local storms in Hawaii that cause high wind and waves from the south through the west are termed "Kona storms". These storms are extremely unpredictable in frequency and intensity, and they can generate large destructive waves. Kona storms that develop to the south of the Island chain generate waves that can directly impact the study area. While not as frequent in occurrence as north swells, hurricanes also occur in the central Pacific near Hawaii. Since 1950, three hurricanes (Dot, Iwa, and Iniki) passed south of the Hawaiian chain, and also produced large surf on southern and western coastlines.

Weather records in Hawaii have been kept only since 1950, but prior to 1961 when the first satellite data became available, all records of storm events were anecdotal. As a result, there is only limited historical data on the occurrence of major storms, and the recorded information is primarily descriptions of property damage rather than reports on meteorological or oceanographic conditions. Compilations of significant storm events (Fletcher, unpub., Shaw 1981) document 8 incidents of sufficiently large waves from north swells to cause extensive property damage. The largest waves recorded in west Hawaii since 1950 were generated by an intense Kona storm which formed in January 1980 south-southwest of the Hawaiian Islands as the result of the joining of two strong low- pressure systems. Breakers with heights peaking at 6 m impacted the study area for about 5 days.

The initial two surveys (1973, 1974) took place 14-15 years after Hurricane Dot, and after four recorded large north swells from 1968 to 1974. In retrospect, the condition of the coral community during the initial phase of the program in 1973 provides a good "pre- storm" representation of coral community structure that developed following an extended period of nominal wave stress. The first storm event in 1974 resulting from a large north swell is characterized as an "intermediate storm event" (Dollar 1982). The third survey in 1980 was conducted one month after the Kona storm, and quantified the effects of an especially severe storm (Dollar 1982). The fourth survey (1992) was conducted 10 years after Hurricane Iwa and 4 years after the last significant north swell. The fifth survey (1993) was conducted within a year of both Hurricane Iniki and a north swell that was judged by long-time residents of the area to be the largest in a decade. The sixth survey was conducted in 2002 after a ten-year period when there were no significant storms. Thus, the investigation has extended over a 30-year period, and appears to have spanned the entire spectrum of storm wave conditions.

## *ii. Geologic Setting*

The geological and biological zonation of the study area from the shoreline to beyond the depths of coral occurrence is depicted is described in Dollar (1982) (Figure 20). The coastline is a lava cliff that drops from 3-7 m above sea level to 3-5 m below sea level. Basalt blocks and boulders resting on a gently sloping basalt basement dominate the base of the cliff. The boulders are round to subangular, 0.5 to 2 m in diameter, and extend up to 30 m from the shoreline cliff. A thin rind of coralline red algae, 1-3 mm thick, covers most of the basalt. The basalt basement extends 20-50 m

seaward of the boulders as a relatively flat "bench", and appears to terminate in a steep slope with an incline of 30-40°. During the three most recent surveys (1992, 1993, 2002) the slope was covered in sections with small elongate rubble fragments (2-10 cm in length, 1-2 cm in diameter) that are predominantly broken branches of *Porites compressa*. A hole dug approximately 0.5 m into the rubble at a water depth of 26 m did not penetrate the layer of rubble fragments to solid basement rock. Excavation of the rubble field triggered downslope movement of loose material. Discolored streaks in the rubble beds suggested recent slides of rubble, which exposed previously buried material. Below the water depth of rubble accumulation (35-40 m), the sloping bottom was covered with a uniform layer of white calcareous sand.

### *iii. Pre-Storm Coral Community Structure - 1973*

Coral community structure of west Hawaii is typified by four zones, each characterized by a single dominant species (Figures 21, 22, and 23). During the pre-storm survey (1973), clear-cut peaks in coral cover existed for each species in each zone (Figure 21). The nearshore boulder zone receives the brunt of wave forces, and is populated primarily with the branching species *Pocillopora meandrina* (Figure 22). In 1973, the solid flat pavement of the reef bench seaward of the boulder zone was populated by a variety of corals in an assortment of growth forms. The most abundant were massive colonies of *Porites lobata*, which contributed a large proportion of the living coral cover and provided a complex of calcareous surfaces for settlement by other species (Figure 22).

Interconnected thickets of *Porites compressa* dominated seaward of the shelf break, where water depth increases rapidly, the reef slope. With this growth form, the coral essentially provided its own substratum (Figure 22). The *P. compressa*-slope zone had the highest pre-storm mean transect coral cover (~75%) (Figure 21, Table 8), and the lowest mean diversity (0.45). Below the downslope limit of the *Porites compressa* thickets (~30 m), where normal wave surge is low, only scattered colonies of *Porites lobata* occurred on rubble fragments, resulting in a mean coral cover of only 3% (Figure 21, Table 8). Total coral cover of the three dominant species (*Pocillopora meandrina*, *Porites lobata*, and *P. compressa*) accounted for 94% of coral cover, and 49% of bottom cover (Figure 21, Table 8).

#### *iv. Storm Effects to Coral Community Structure - 1974-1980*

The intermediate and severe intensity storms in 1974 and 1980, respectively produced dramatically different effects on coral community structure and zonation (Dollar 1982). The 1974 intermediate storm caused coral mortality in all zones, but the primary effect was on the reef slope. Storm waves up to 4 m in height caused fragmentation and transport of parts of the fragile *Porites compressa* framework downslope. Most of the coral fragments were alive a month after the storm, with apparently healthy tissue covering the skeletons. The peak in living cover shifted 7 m farther offshore and 5-10 m deeper, resulting in expansion of the range of *Porites compressa*. As a percentage of bottom cover, coral cover on the reef decreased from 52% to 46% (Table 8).

The effects of the severe storm in 1980 were clearly catastrophic. Storm surf, consisting of steep plunging breakers up to 6 m in height appeared to suck up most of

the water on the reef bench before crashing over the entire reef, destroyed the zonation pattern (Dollar 1982). On the bench, *Pocillopora meandrina* colonies were either sheared off at the base of attachment or sustained extensive breakage of branches (Figure 22). Large *Porites lobata* colonies were broken and overturned, sustaining loss of most living tissue (Figure 22). Only flat encrustations of *Porites lobata* remained relatively unaffected.

Damage was most extensive on the reef slope. The entire expanse of *Porites compressa* thickets was reduced to a carpet of rubble with most fragments less than 5 cm in length. Scattered over the carpet were large broken plates of *Porites lobata* (Figure 22). Much of the damage to the *Porites compressa* thickets appeared to be a result of crushing as the large blocks of broken coral colonies and basalt boulders were tumbled downslope (Dollar 1982). Live coral cover decreased from about 46% to 10% (Figure 21, Table 8). The relative dominance of the three most abundant species changed substantially as a result of the storm. During the pre- storm and intermediate disturbance period, coral cover was rather evenly distributed between *Porites lobata* and *Porites compressa*. Following the severe storm, relative abundance of *Porites lobata* increased from about 35% to 87%, while *Porites compressa* dropped from about 40% to 5%. Percentage cover of *Pocillopora meandrina* decreased from 14% to 4% (Table 8).

Total coral cover over all transects was significantly reduced following both the intermediate and severe storms (Wilcoxon matched-pairs signed-rank test,  $p < 0.05$ ) (Table 9). When analyzed separately, *Porites lobata* and *Porites compressa* decreased significantly following only the severe storm, while cover of *Pocillopora meandrina* decreased significantly after both storms (Wilcoxon test,  $p < 0.05$ ) (Table 9).

Considering the entire reef, species cover diversity ( $H'_c$ ) did not change significantly following the either 1974 or 1980 storms (Wilcoxon matched pairs sign test,  $p < 0.05$ ) (Table 9). When analyzed by reef zone, mean diversity was significantly reduced in the *Porites lobata*-bench zone between 1973-1974 and 1974-1980, and increased in the *Porites compressa*-slope zone in 1973-1974. (t-test,  $p < 0.05$ ). The storm appeared to affect uncommon corals more than the most common species, resulting in a decrease in mean diversity in the *Porites lobata*-bench zone (Dollar 1982).

#### *v. Storm Effects to Coral Community Structure - 1992-1993*

Between 1980 and 1992, Hurricane Iwa (1982) and two north swells (1986, 1987) produced damaging surf along the coast of West Hawaii. Results of the 1992 survey indicated that the coral community was nowhere near the 1973 pre-storm level of development, either in terms of overall coral cover or of zonation pattern. In the shallow boulder zone, most of the living colonies of *Pocillopora meandrina* were small, recent recruits less than 5 cm in diameter. There were few undamaged living colonies of *Pocillopora meandrina*, which could be considered adults. There were, however, numerous stump-like remnants of colonies that appeared to have lost branches to the shearing action of large waves.

In the bench zone, there was little regeneration of the massive domed structures of *Porites lobata* that had provided the three-dimensional aspect to the reef bench prior to the 1980 storm. Most living corals consisted of flat encrustations or small knobby colonies of *Porites lobata* 5-10 cm in diameter on the basalt bench. A noticeable difference during the 1992 and 1993 surveys was the absence of loose

coral rubble that had filled many of the interstitial spaces on the bench immediately following the 1980 severe storm.

On the reef slope, there was no evidence of the interconnected lattice of *Porites compressa*. The slope was not substantially different in appearance from immediately following the 1980 severe storm, when the entire expanse of *Porites compressa* was reduced to a bed of small rubble fragments devoid of living tissue. Twelve years later, living colonies of *Porites compressa* on the reef slope remained scarce. It was difficult to determine if the few living colonies of *Porites compressa* were the result of planular settlement or regrowth from viable fragments.

Between 1980 and 1992, total bottom cover of living corals increased from 10.5% to 15%. Relative abundance of the three dominant species remained similar to 1980, with *Porites lobata* clearly the dominant species, comprising 70% of coral cover (Table 8). Over the entire reef, total coral cover and cover of the most abundant species (*Porites lobata*) did not change significantly ( $p < 0.05$ ) between 1980 and 1992 (Wilcoxon matched-pairs signed-rank tests)(Table 9). While the absolute increase in *Porites compressa* and *Pocillopora meandrina* cover were only about 1% (Table 8), these changes were statistically significant (Wilcoxon matched-pairs signed-rank test,  $p < 0.05$ ) (Table 9).

Comparing pooled coral cover between 1980 and 1992 transects revealed that none of the changes in zonal mean cover were significant (t-test,  $p < 0.05$ )(Table 10). However, species cover diversity across the reef increased significantly during the recovery period (Wilcoxon matched-pairs signed-rank test,  $p < 0.05$ ) (Table 9). Pooled diversity increased in all zones following the recovery period, with a significant increase in the *Porites compressa*-slope zone (t-test,  $p < 0.01$ ).

In 1992, Hurricane Iniki and large waves from a powerful north pacific storm struck the west Hawaii coastline. Between 1992 and 1993, total coral cover decreased from 15% to 12% (Table 8). Mean percent cover of each of the 3 dominant species also decreased from the 1992 survey (Table 6), with a significant decrease of *P. compressa* (Wilcoxon matched-pairs signed-rank test,  $p < 0.05$ ) (Table 9).

Comparing pooled coral cover between 1992 and 1993 transects revealed that there was a significant decrease in the *Porites lobata*-bench zone (t-test,  $p < 0.01$ ). Although not significant, there was a decrease in mean coral cover in the shallow *Pocillopora meandrina*-boulder zone of 2.4% (percentage decrease of 27.2%), and an increase in mean cover in the *Porites compressa*-slope zone (absolute increase 5.9%, percentage increase 45.1%). In the deepest *Porites lobata* rubble zone coral cover decreased by 0.1% (percentage decrease of 3.3) (Table 10).

#### *vi. Recovery Period 1993-2002*

The 2002 survey followed a period of about a decade without major storms. Observations of the reef track indicated that there was definite recovery of the reef, but that the recovery varied substantially between zones. Total coral cover and cover of the three dominant species all increased since the 1993 survey (Table 8, Figure 21). When total coral on each transect is compared using the Wilcoxon ranked sums test, there is a significant increase ( $P < 0.01$ ) over the entire reef (Table 9). The region of most impressive recovery was the nearshore *Pocillopora meandrina* boulder zone, where numerous “new” colonies covered the bottom (Figure 21). Mean coral cover in this zone (29.2%) increased significantly ( $P < 0.005$ ) from the previous survey in 1993 (Table 10). In addition the mean total coral cover in 2002 was virtually the same as in



1973 (31%), indicating that this area has essentially recovered to pre-storm values (Figures 21 and 23, Table 10).

Total coral cover in the *Porites lobata* bench zone (51%) also increased significantly ( $P=0.001$ ) over the previous survey (14%) (Table 10). While total cover in 2002 was similar to that in the pre-storm period, the assemblage of corals in this zone was substantially different than in the initial survey in 1973. During the pre-storm period, the predominant coral was large dome-shaped colonies of *Porites lobata* (Figure 22), which comprised about half of the total coral; the other dominant species was *Porites compressa*, which comprised about 30% of coral cover (Figure 21). In 2002, while *Porites lobata* still comprised about half of the total cover, the growth form is substantially different. Rather than large lobate colonies, most of the *P. lobata* consists of flat encrustations on the reef platform. Many large dome-shaped colonies were noted, however, overturned on the deep reef slope. These colonies had been disengaged from the reef bench by the severe concussive forces of the 1980 storm, and rolled down the reef slope. Rather than *P. compressa*, the other dominant coral presently inhabiting the reef bench is *Pocillopora meandrina*, covering nearly the same percentage of the bottom as in the nearshore boulder zone (Figure 21). *Porites compressa* was essentially absent from the reef bench in the 2002 survey.

The zone with the least recovery is the *Porites compressa* slope zone. Total coral cover on the slope increased slightly by about 10% (21% to 31%) over the last decade, but this change was not statistically significant (Table 10). Compared to total coral cover on the slope in 1973 (73%), coral cover is still about 40% less of the slope than before the 1980 Kona storm. While there are patches of living *Porites compressa*, most of the slope still consisted of a bed of small rubble fragments. In fact, much of

the area is essentially the same in appearance as immediately following the storm that occurred twenty-two years ago (Figure 22). The deepest zone, consisting of small colonies of corals, predominantly *Porites lobata* that has settled on rubble fragments has remained virtually unchanged over the course of the time-series surveys (Figures 21, 23).

### c. Discussion

From the few available case studies, it appears that reef community recovery processes, in terms of both time scale and resultant diversity patterns, are often site-specific, depending in part on the life-histories of dominant corals and the local physical regime (Done et al. 1991). The documented changes to the west Hawaii reef between 1973 and 2002 illustrate the extensive changes (impact and recovery) that can result from low frequency, severe disturbances. Reefs in other locales have suffered comparable effects. Six hurricanes, which struck French Polynesia during 1982-83, resulted in catastrophic damage remarkably similar to those noted in Hawaii. On outer reef slopes coral destruction varied from 50 to 100%. The most severe damage was inflicted on the well-developed, but more fragile colonies growing at greater depths. The outer slopes were transformed into a rubble zone covered with coarse sand and dead coral rubble. On steep slopes, it appeared that avalanches of colonies broken from the upper slopes, rather than direct wave energy, provided much of the destructive force to create the rubble fields (Harmelin-Vivien and Laboute 1986).

Hurricane Allen, which struck the north coast of Jamaica in 1980, devastated the reefs in Discovery Bay (e.g. Woodley et al. 1981) similar to that described for the severe storm in west Hawaii. Although the island of Jamaica experiences 60

hurricanes or tropical storms per 100 years, the north coast had not been subjected to catastrophic storm since 1944 (Kjerfve et al. 1986). In addition to the initial devastating effect of wave stress, secondary mortality, caused by undetermined factors (e.g. continued rolling, disease and fragmentation stress, predation) occurred up to 5 months following the storm. These effects were an order of magnitude more severe than the immediate effects for the predominant framework species, *Acropora cervicornis*, but not to less susceptible species (Knowlton et al. 1981).

The timing and chronology of secondary effects other than storm stress can also act as intermediate disturbance. Three years after Hurricane Allen struck Caribbean reefs, a second major change in the coral communities occurred when the sea urchin *Diadema antillarum* suffered mass mortality, apparently from disease (Hughes 1989). The marked reduction in herbivory that followed caused a dramatic increase in algal biomass. Coral cover, which was recovering steadily from hurricane damage, declined again to unprecedented low levels because of competition with algae. Had the sea urchin mortality occurred during a period when the reef was at a climax stage, it is likely that algal competition would have not been so effective (Hughes 1989). Similarly, secondary effects of bioerosion to Galapagos reefs appear to have intensified the effects from the 1982-3 ENSO event, and delayed recovery of reefs which suffered 95% mortality (Glynn 1990).

In 1993, Grigg developed a theoretical model of succession for coral reef communities based on the intermediate disturbance hypothesis (Connell 1978, Grigg 1983) (Figure 24). This name is somewhat misleading in that catastrophic, as well as intermediate, disturbances are key factors in the successional process. Severe disturbances reset the process to a near-zero state. Intermediate disturbances may

either increase or decrease diversity depending on the successional stage of the community at the time of the disturbance. The successional stages of the west Hawaii reef conform with the Grigg model. Community structure in 1973 can be considered the peak development state following an apparent extended period without major disturbance (Figure 24, #1), followed by the intermediate storm in 1974 which served to maintain peak diversity (Figure 24, #2). The severe Kona storm in 1980 set the successional state back to near zero (Figure 24, #3). The five intermediate events between 1980 and 1993 appear to have slowed or prevented recovery with respect to the fully developed community (Figure 24, #'s 4 and 5). The decade without storm impacts (1993-2002) is allowing the community to proceed back toward the undisturbed state, although it is evident that recovery is not uniform across the reef.

On the basis of estimations from dated lava flows, it has been projected that it will take about 50 years for Hawaiian reefs to regain peak diversity (Grigg and Maragos 1974), approximately equal to the expected frequency of maximum intensity disturbance events (Dollar 1982). Using the measured 5% increase of coral cover between 1980 and 1992 as a value to estimate exponential increase at the same rate with no other disturbance, then total cover could have reached the pre-storm level (50%) in about 40 years, essentially equal to the projected frequency of severe storms. If, however, coral cover could have increased linearly at the same rate of about 5% per decade, then recovery to the 1973 pre-storm level would require about 70 years. Given the lack of precision of these estimates, the projections of 40 and 70 years for recovery may not be substantially different.

The 1993 results, however, indicate that the recovery process was interrupted by two major storm events since 1992, with total coral cover decreasing from 15% to

12%. Statistically significant decreases in cover occurred in the zone with the highest coral cover (*Porites lobata*-bench) and in the species considered the most susceptible to wave damage (*Porites compressa*). These results indicate that the hurricane and north Pacific storms (intermediate storms) prior to "full recovery" have returned the coral community structure to a condition very similar to that following the severe storm in 1980. In the 13 years between 1980 and 1993, there appeared to be virtually no net recovery of the coral community.

Since 1993, however, the situation has changed considerably. Coral cover on the shallow boulder zone, inhabited by the pioneering species *Pocillopora meandrina* has essentially recovered to pre-storm status. Coral cover on the reef bench has increased significantly over the last decade, but the species assemblage is very different than the pre-storm composition, with a branching coral (*Porites compressa* replaced by *Pocillopora meandrina*). The reef slope, which had the highest coral cover during the undisturbed period, has shown the slowest recovery, with only the beginning of regeneration of the extensive mats of *P. compressa*. The relatively slow recovery of the reef slope, compared to the shallower zones is likely a response to several factors. First, the unconsolidated slope is not a favorable settling site for the most common reef species owing to both depth limitations (slower growth with lower ambient light). Secondly, instability caused by movement of the rubble bed during low intensity wave events may prevent coral settlement to a much greater degree than on the solid surfaces of the upper reef zones. The similar appearance of the rubble beds that comprise the slope zone over a twenty-two year period (1980-2002) suggests that the rubble is undergoing some dynamic processes. Because of the frequent movement of the rubble beds, it may be that complete regeneration of the *Porites*

*compressa* zone to predisturbance levels will depend more on asexual lateral growth of the existing patches of living *P. compressa*, than on settlement of new colonies on the rubble bed. If this is the case, it may be expected that recovery will take far longer on the slope than on the reef bench.

The observed pattern of damage and recovery in Hawaii over the last 20 years appears to be different than in other regions with different frequencies of severe wave stress, and different species composition. The damage-recovery cycle appears to proceed to completion relatively rapidly in some areas where hurricanes are more common (i.e. high frequency events). In this case, the effects of any one storm may be slight because the community has already adjusted to high wave stresses (Kjerfve et al. 1986, Perkins and Enos 1968, Randall and Eldredge 1977, Ogg and Koslow 1978). Ball et al. (1967) point out that on the average, in Florida, hurricanes have affected the entire reef once every 6 years during the Holocene (the past 10,000 years). Some reefs in the Florida Keys which have been severely affected by hurricanes have recovered completely within 6-year intervals (Shinn 1976).

Rapid recovery is largely a result of favorable adaptations of dominant species. Many of the Atlantic reefs subjected to frequent hurricanes are heavily populated by branching species of *Acropora*. These species appear to take advantage of wave activity to promote fragmentation, while morphology of the colonies appears to reduce fragment mortality. Branching *Acroporas* may also rapidly recover from damage through fragmentation, cementation, and high growth rates (e.g. Highsmith et al. 1980, Tunnicliffe 1981, Highsmith 1982). No coral species in Hawaii appears to have similarly adapted to rapid recovery from severe stress through re-cementation and regeneration of fragments. These communities with a rapid damage and recovery

cycle may be defined by a "high frequency" variation of the disturbance model where diversity (or cover) is graphically represented by a "sawtooth" pattern, with relatively short periods between peaks and valleys (Figure 24). The amplitude of the peak seems to depend on the ability of resident species to rapidly regenerate.

At a workshop on coral reef ecosystems and global change (D'Elia et al.1991), it was pointed out that coral reef scientists tend to select healthy reefs for study, thus biasing subsequent observations against possible "improvement", and in favor of "deterioration". The sequential studies off west Hawaii provide one of the first long-term field tests which can address both impacts (deterioration) and recovery (improvement) of reef community structure, as well as construction of the modern reef framework. While the long-term data base remains still remains limited even after 30 years, we are beginning to understand the interactions that control the "temporally varying mosaic" of disturbance and recovery of coral communities, as well as the constructional composition of reefs. While outcomes vary substantially on different reefs around the world, a common pattern appears to be delineated by the time between severe disturbance, and adaptations of component species.

## *CONCLUSIONS*

Within the last decade, the view that the coral reefs on a global scale are in serious jeopardy owing to anthropogenic activities has developed into one of the most compelling ecological conundrums facing marine scientists and ecosystem managers. While it appears unambiguous that humankind is affecting the marine environment, there are still widely differing viewpoints on how human interactions are influencing the

worldwide status of coral reefs (Grigg and Dollar 1990). One view asserts that coral cover on reefs is deteriorating steadily and rapidly from over-fishing, destructive fishing techniques, nutrient and other chemical inputs, freshwater runoff, extreme sedimentation from coastal development and poor land management, mining and dredging, oil and industrial pollution, and increased water temperature from global warming (Wilkinson 1992, Ginsburg 1993, Wilkinson and Buddemeier 1994). A second somewhat contrary view is that the coral reefs have displayed an impressive resilience by persisting through geologic time in the face of natural stressors such as storms and hurricanes, infestations of disease, climatic change, species shifts and sea-level change. This view acknowledges that serious impacts from the activities of man exist on a localized scale, (primarily near centers of human population), but it also recognizes that reefs are persistent and resilient over geologic time and global scales (Grigg and Dollar 1990, Buddemeier 1992, Smith and Buddemeier 1992, Grigg 2000).

In the “Status of the Coral Reefs of the World: 2000”, a compilation of regional reports, 11% of the worlds reefs are described as having been “destroyed” prior to 1998, with an additional 16% “destroyed” in 1998 (the editor provides the caveat that these 16% of reefs considered destroyed in 1998 by El Nino induced bleaching may eventually recover) (Wilkinson 2000). The report also predicts that 14% of reefs are at a critical stage for loss in the next 2-10 years, while 18% are threatened to be irreparably degraded in the next 10-30 years unless the stressors are removed, and large areas set aside as Marine Protected Areas. Taken as a worst case scenario, these estimates predict that in the next 30 years, about half of the worlds reefs could be lost. While this statistic is alarming and unequivocally signals a call for



implementation of effective management and enforcement, it also says that about half of the worlds reefs are not in imminent peril of being destroyed.

Most of the documented damage to reefs has, or is predicted to, occur in the Indian Ocean, Southeast and East Asia and the Caribbean/Atlantic regions. On the other hand, vast expanses of reef in the Pacific are in good health with a positive prognosis for the future, as long as management strategies addressing the effects of increasing populations are implemented (Maragos 1997). Hawaii also fits into this category. In a review of the status of coral reefs in Hawaii, Grigg (1997) concludes that..."with the exception of overfishing, about 90% of the coral reefs in the main Hawaiian Islands are healthy." In the remaining 10%, sedimentation and eutrophication are the most serious problems.

It has been well documented that growth and community structure of coral reefs in the main Hawaiian Islands are primarily under the control of wave forces (Grigg and Maragos 1974, Dollar 1983, Grigg 1988, Dollar and Tribble 1993, Grigg 1998). Community succession is continuously interrupted by intermediate level wave events, which prevent resource monopolization and maintain high diversity. Infrequent high intensity events can and do disrupt the entire community and set the successional process back to time zero (Grigg 1983). These studies of succession have shown that anthropogenic impacts on coral reefs in Hawaii are superimposed on these naturally controlling forces. The long-term investigations reported in this paper all provide additional evidence to support this conclusion. In general, anthropogenic impacts are only important in Hawaiian environments where wave forces are not the dominant controlling factor. These environments typically are embayments and lagoons that are protected from wave stress, resulting in relatively long residence time of the water

column. These environments constitute less than 10% of the coastline of the entire state (Grigg 1997).

The data from the six case studies reported in this paper reinforce the second view of status of coral reefs worldwide. The time-series surveys indicate that all of the open coastal sites, coral abundance has increased over the last several decades in the presence of extensive shoreline resort development, and sewage discharge. The coral communities off all of the sites, except Honolulu Bay, are thriving, with little or no apparent effect of land-based anthropogenic activities. The lack of destructive storm wave disturbance over the past decade, has resulted in a progression of the successional process at all of the open ocean sites. At Mauna Lani and Princeville, there has been a continuous increase in cover during the last two decades. It could be argued that the resort development is contributing to an increase in coral coverage by mitigating surface runoff, but it is more likely that sediment impacts on offshore reefs were have always been minor, as most of the sediment carried to the ocean by streams is deposited on the reef flat or rapidly dispersed by wave action. It is also possible that the increased coral cover off Princeville is a response to the slight decrease in frequency of large surf from winter swells since 1985. Recovery of the coral communities at the sites, which were severely impacted by human activities (Sand Island), and natural phenomena (Keaweakeheka) has also been documented to be proceeding along the successional pathways that occur under the physical parameters of the particular area.

On the other hand, anthropogenic impacts can be serious in localized areas where water circulation is restricted. In semi-enclosed Honolulu Bay, where a lack of wave-induced circulation limits resuspension and water exchange, sediment input can

exert a significant effect on coral community structure. Large-scale pineapple agriculture requires acreage with periodically exposed soil in a climate that produces occasional heavy rains. While sediment input into the Bay has occurred from long before human activity, it has undoubtedly changed with changing land use. The coral assemblages that are present today reflect such input, with abundant coverage of species that are “sediment-resistant.” Hodgson (1989) observed colonies of *Montipora verrucosa* (now called *M. capitata*) in Hawaii with substantial sediment accumulation that survived for several weeks, apparently without causing tissue damage. He suggested that resistance to sediment stress is a combination of sediment clearing efficiency and biological resistance to infection by microbes present in the sediment. *Montipora* spp. are the major component of the coral community in the inner portions of Honolulu Bay that are the most susceptible to sediment input and retention.

It is likely that the declines in coral cover in Honolulu Bay are part of a normal cycle of impact and recovery that has occurred there at least during the past 50 years.

The intermediate disturbance model described above may apply at Honolulu Bay where the combination of sediment input and water column flushing are the controlling factors. Sediment input following storms of sufficient intensity to deliver sediment loads that remain in the Bay, result in mortality to the reef community. Subsequent flushing of the sediment by wave-induced circulation provides new substratum for settlement and growth. High coral cover in the inner Bay during the 1990 and 1992 surveys, particularly with sediment-resistant species, verifies that sediment stress is not continuous or completely limiting to coral growth. If there were not such a cycle of recruitment and regrowth, no living coral would exist in the inner Bay. Future studies might differentiate between episodes of sediment runoff that result

in impact versus those that do not. Another investigation that would be of interest in understanding the effect of land use on coral community structure would be to core the reef in the inner Bay to determine if there were changes in species assemblages coinciding with the introduction of agriculture.

Considering resort development in Hawaii in general, a concern is that golf courses, which are a major component of resorts, cause degradation of the marine environment and coral reefs. Leaching of fertilizers to groundwater with subsequent discharge to the nearshore ocean may have the potential to affect systems. Such a situation has not been documented in Hawaii. Rather, Dollar and Atkinson (1992) found that nutrient subsidies to coastal waters from groundwater leaching of golf course fertilization doubled the background levels of nitrogen and increased phosphorus by about 20%. However, these subsidies resulted in no change to the nutrient dynamics within the coral reef communities. Along wave exposed coastlines, nutrients were rapidly mixed to background levels by surge and vigorous wave action. In wave-sheltered embayments, such as Honolua Bay and Keauhou Bay in West Hawaii, high nutrient groundwater formed a surface layer that flowed out the mouths of the bays with no contact with the benthos.

Similarly, the process of ocean sewage disposal is often cited as a potential source of reef degradation (e.g., Pastorak and Bilyard 1985). Indeed, in Hawaii the discharge of untreated effluent prior to 1977 had catastrophic effects to reef communities in Kaneohe Bay (Smith et al. 1981), and at Sand Island (reported in this paper). However, in both of these situations, the major effects were a result of the high particulate loading of the receiving environment, rather than solely a result of nutrient subsidies. With the federally mandated requirement of treatment of sewage, and the

relocation of outfalls to open ocean environments, particulate loading resulting in deposition of organic material on the reef surface has been eliminated. Thus, the major component of treated sewage effluent is dissolved nutrients (which are not reduced in the primary or secondary treatment process).

The effects of nutrient loading from anthropogenic activities in general is a topic that is presently under considerable debate within the scientific literature. Atkinson (in press) states that..."It is widely believed that any nutrient input to coral reefs is deleterious. The argument is actually based on an incorrect, historical view of how coral reefs recycle nutrients." He goes on to say that a dogma has developed that claims coral reefs recycle their nutrients through biologically-mediated mechanisms, and that any input of nutrients will alter, or perturb these processes. Based on rate constants for mass-transfer-limitation of nutrient uptake, it is physically impossible to recycle nutrients through the water column on such small scales (Atkinson in press). Corals can, and do grow well at elevated nutrient concentrations (Atkinson et al. 1995), and Smith and Buddemeier (1992) note that some reefs look healthy and are apparently doing well in a milieu of naturally high nutrient levels. Kinsey (1991) emphasized that it is incorrect to jump from the observation that coral reefs can do well under low nutrient conditions to the conclusion that coral reefs require such low nutrient environments.

The ENCORE experiment investigated responses of coral reef organisms and processes to controlled additions of dissolved nitrogen and phosphorus to a reef off Australia (Koop et al. 2001). Results of the two-year long experiment showed that reef organisms and processes were affected by elevated nutrients, although the impacts were generally sub-lethal and subtle, and the treated reefs at the end of the

experiment were visually similar to control reefs. At a community level, rapid nutrient changes are thought to cause phase shifts in structure and function. With an increase in nutrients, rapidly growing phytoplankton and benthic algae can gain a competitive edge, often overgrowing and eventually smothering them (Smith and Buddemeier 1992). However, it has been pointed out that despite widespread assumptions that algae are generally superior competitors over corals, especially under eutrophic conditions, there is little direct experimental evidence that corals and algae do compete, and little data on the processes and causality of their interactions (McCook 1999, 2001, McCook et al., 2001). McCook (1999) concluded that nutrient overloads can contribute to reef degradation, but that they are unlikely to cause phase shifts simply by enhancing algal growth (hence overgrowing corals), unless herbivory is unusually or artificially low. McCook goes on to suggest that nutrient increases alone are unlikely to tip the balance of competition in favor of macroalgae over corals, and that this view is a risky basis for management.

The results of the long-term investigation of the East Honolulu Wastewater Treatment Facility provides a good example that discharge of treated sewage effluent does not result in degradation of coral reef habitats. Other researchers (e.g., Grigg 1994, 1995, Russo 1982 and Russo et. al. 1981) have reported similar results for the outfalls presently discharging treated effluent off Kailua, Waianae, and Mamala Bay.

The point of this discussion is not to argue that effective management should ignore nutrient and sediment subsidies to reef environments. Rather, we suggest that effective reef management should be based on using objective scientific data. For example, Larcombe and Woolfe (1999) found that the rate of terrigenous sediment supply to the central Great Barrier Reef has probably increased in the last 200 years

as a result of changes in land use. Using geological data and information on sedimentary processes, they demonstrate that turbidity levels and sediment accumulation rates at most coral reefs will not be increased, because on the times scales that can reasonably be addressed by environmental management, these factors are not currently limited by sediment supply. These authors do not imply that sediment is not a potential problem to reefs (in fact the contrary). Rather, they suggest that future studies and management concerns focus on reef areas where sediment availability is a limiting factor for turbidity and sedimentation, such as immediately adjacent to point source inputs. The Great Barrier Reef study provides a good model for reef management issues in Hawaii. While sediment from anthropogenic sources may episodically enter the ocean along open coastal areas regardless of the extent of shoreline management, the impacts are likely to be minimal or nonexistent owing to resuspension and removal by wave action. Management concerns should concentrate on those specific areas where circulation is confined and sediment retention is likely (e.g., embayments).

In conclusion, the long-term data sets described in this paper provide empirical evidence that effective coral reef management should be based on case specific and objective scientific data. At present there is a strong consensus that coral reefs are threatened on a global scale by the activities of man. While such alarm is well justified on many reefs of the world, this view is not an accurate description of Hawaiian coral reefs. With the exception of Honolulu Bay, ALL of the sites investigated in the present study exhibit increased coral reef development over the last several decades. Our results indicate that natural factors, primarily associated with geographic location and oceanographic climate (wave exposure) far overshadow the effects of shoreline

development on open coastlines. Actual anthropogenic impacts on Hawaiian coral reefs (except for overfishing) are generally restricted to areas where ocean circulation is confined; hence these are the areas where management and mitigation should be concentrated.

These conclusions, however, do not mean that management of reefs in Hawaii is unimportant. Rather, focusing management effort on areas where it can be of real benefit (embayments) should strengthen the efficiency and effectiveness of coral reef conservation. As embayments comprise less than 10% of the coastal area of Hawaii, we conclude that these areas are the places where the major effort should be focused. Coral reefs off open wave exposed coastlines in Hawaii are generally quite healthy even though succession may be frequently disturbed by high wave events (hurricanes and long-period swell). Clearly it is not possible to prevent or manage this source of natural forcing.

## **VII. EVALUATION**

All project goals were fulfilled. This project provided a unique opportunity to synthesize long-term data sets that otherwise would not have been accomplished. The use of retrospective data going back several decades is also unique, as it is obviously not possible to recreate these situations. Research programs initiated in the present would require several decades into the future to provide the same magnitude of results. Project results will be disseminated in workshops currently planned for March 2003. In addition, results of a portion of the work have already been submitted for publication to a scientific journal; the remaining results will also be prepared as a separate manuscript for similar submittal.



**VIII. SIGNATURE OF PRINCIPAL INVESTIGATOR**

A handwritten signature in black ink, appearing to read "Shum Dallas". The signature is fluid and cursive, with a long horizontal stroke extending from the end.

## **ACKNOWLEDGMENTS**

We thank Wes Nohara of Maui Pineapple Co. for providing rainfall data, and Pat Caldwell for providing wind and surf data.

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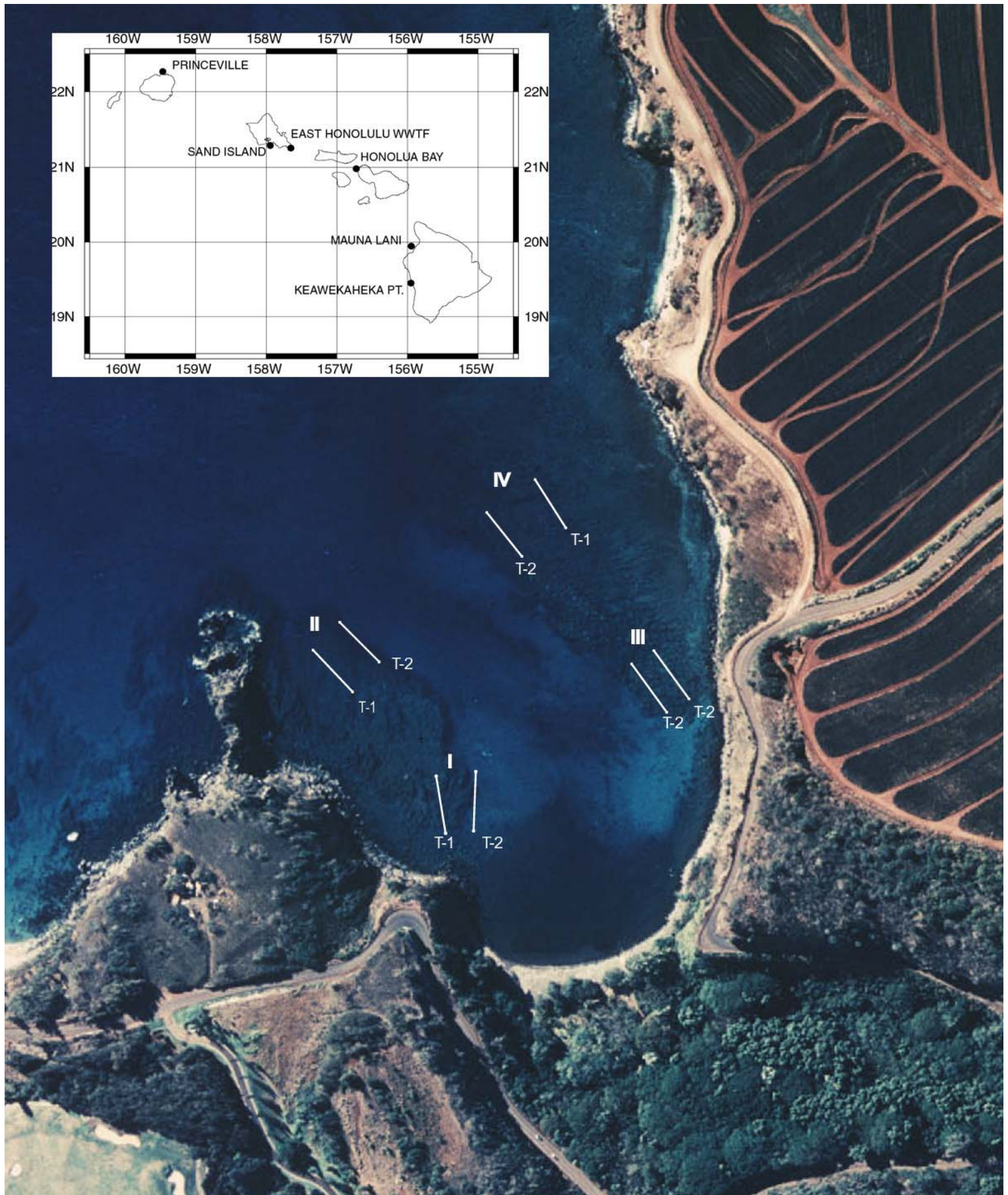


FIGURE 1. Aerial photo of Honolua Bay, Maui, showing locations of coral transects. Index map shows locations of six study sites in the main Hawaiian Islands.



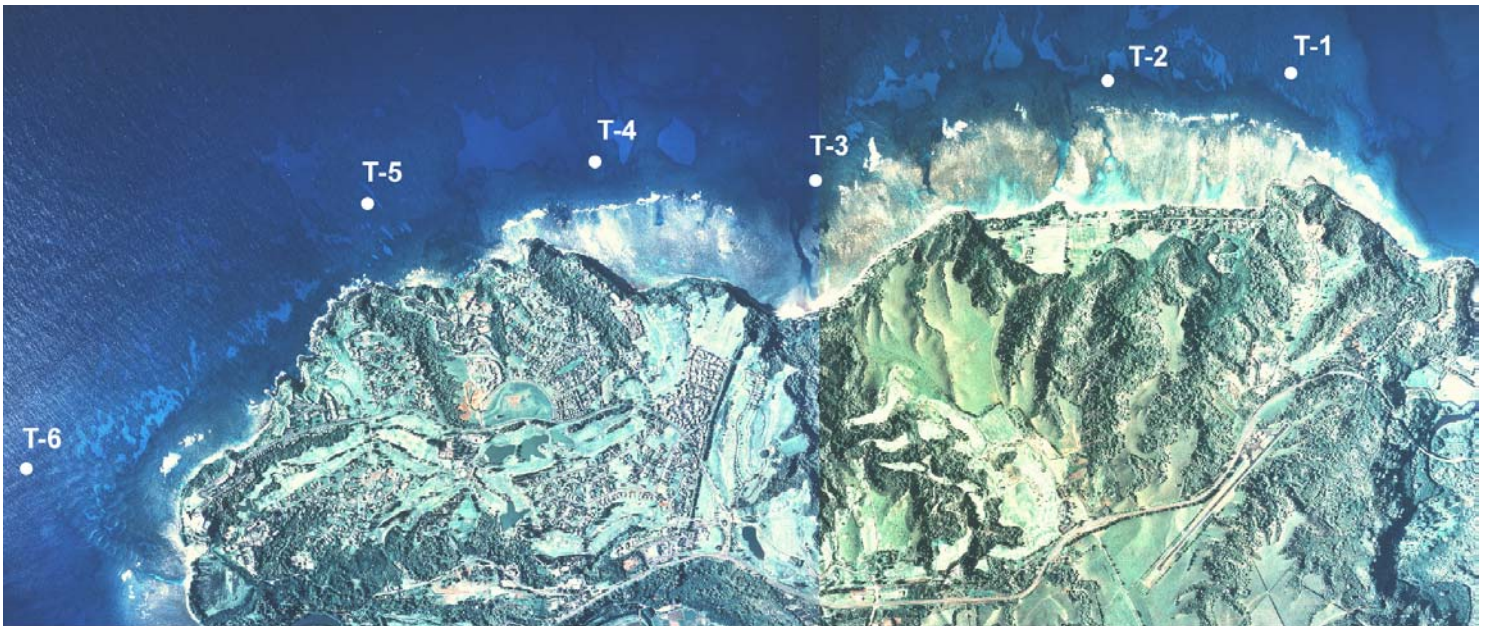


FIGURE 2. Aerial photographs of Mauna Lani Resort on the coastline of West Hawaii (top), and Princeville Resort on the north shore of Kauai (bottom). Also shown are locations of transect stations.

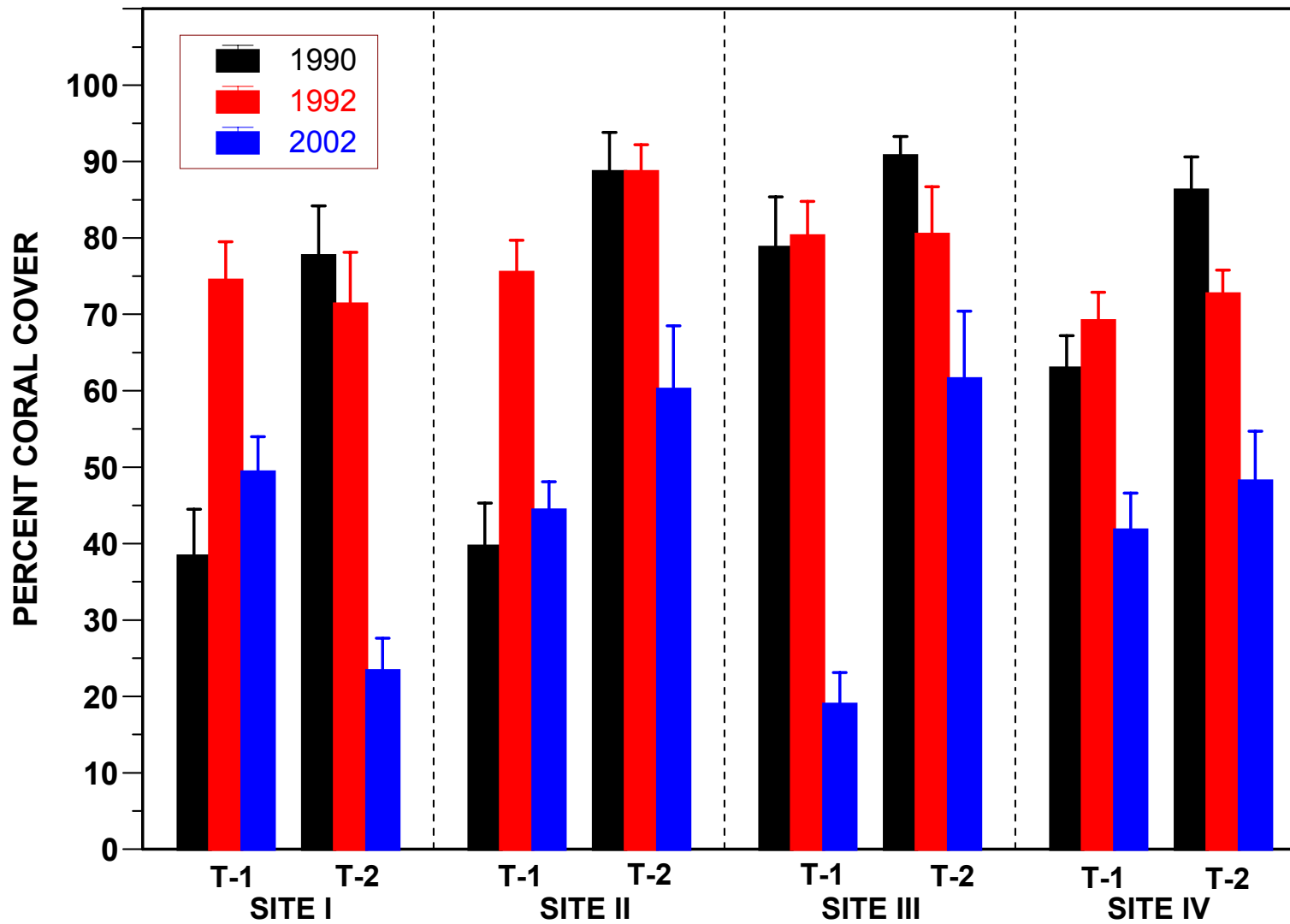


FIGURE 3. Histograms of coral cover (+ s.e.) from phtoquadtrat transects conducducted in Honolua Bay Maui in 1990, 1992 and 2002.

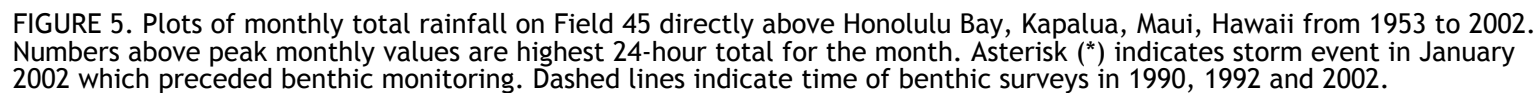


FIGURE 5. Plots of monthly total rainfall on Field 45 directly above Honolulu Bay, Kapalua, Maui, Hawaii from 1953 to 2002. Numbers above peak monthly values are highest 24-hour total for the month. Asterisk (\*) indicates storm event in January 2002 which preceded benthic monitoring. Dashed lines indicate time of benthic surveys in 1990, 1992 and 2002.



# North Shore Hawaii Surfer Scale Database 1968–2002

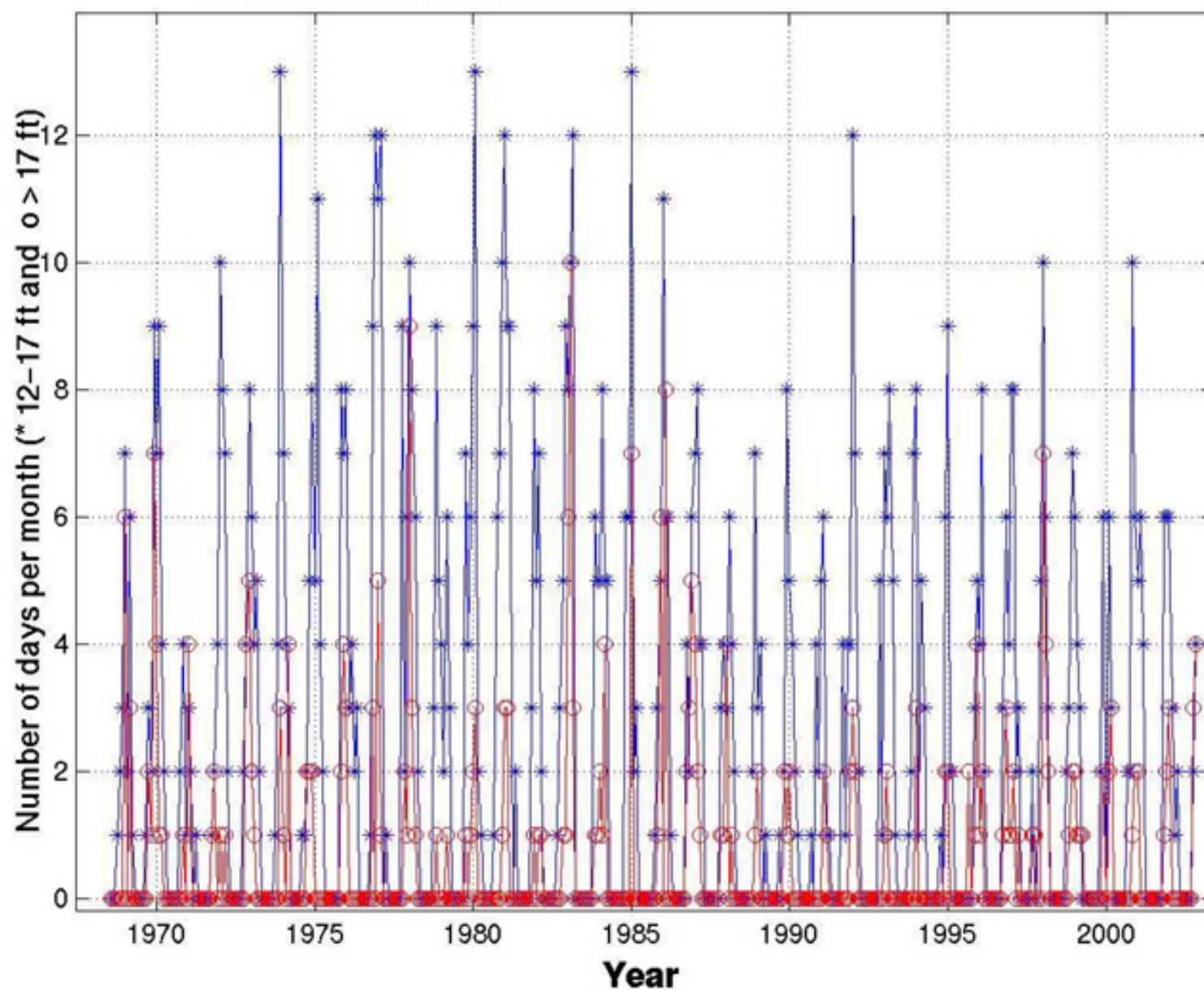


FIGURE 6. Number of days per month of surf between 12 and 17 feet, and greater than 17 feet reported on the north shore of Oahu from 1968 to 2002.

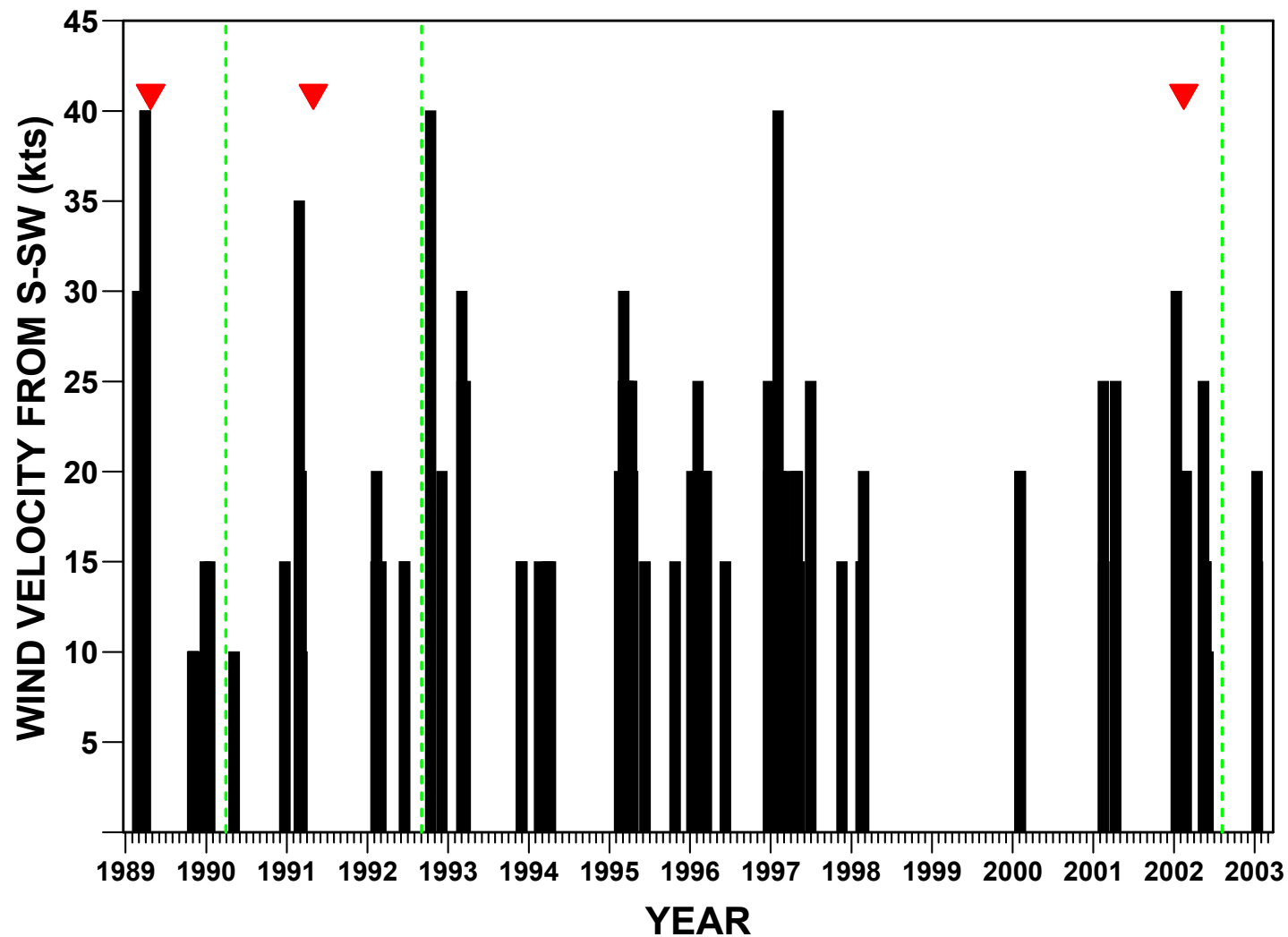


FIGURE 7. Histogram showing velocity of "Kona" winds out of the South and Southwest on Maui from 1989 to 2002. Small dashed lines indicate times when coral surveys were conducted in Honolulu Bay. Inverted triangles indicate large storm events with peak rainfall in April 1989, April 1991 and January 2002.

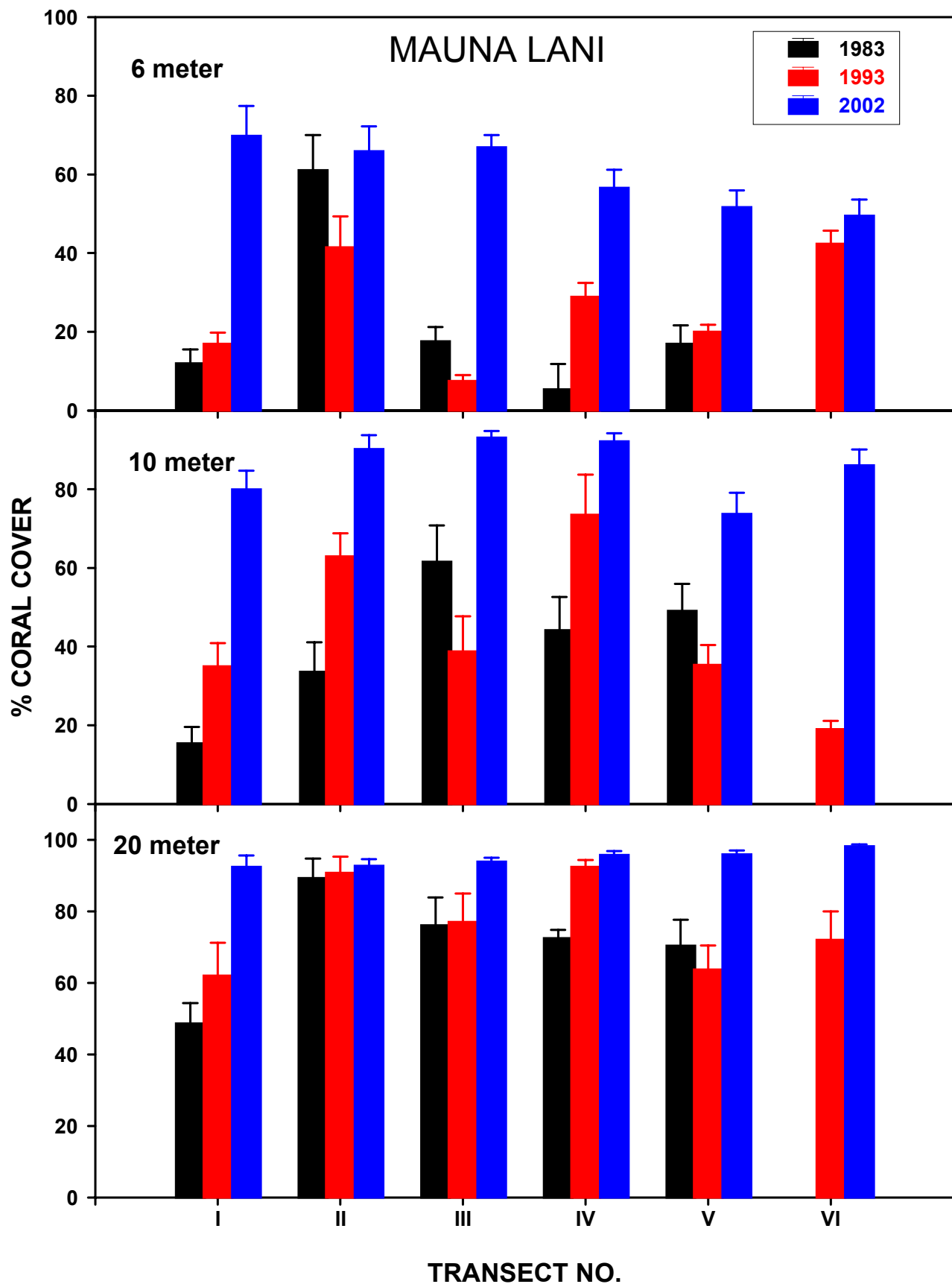


FIGURE 8. Histogram showing total mean coral cover (+ std. err.) from benthic transect survey off the Mauna Lani Resort in 1983, 1993, and 2002.



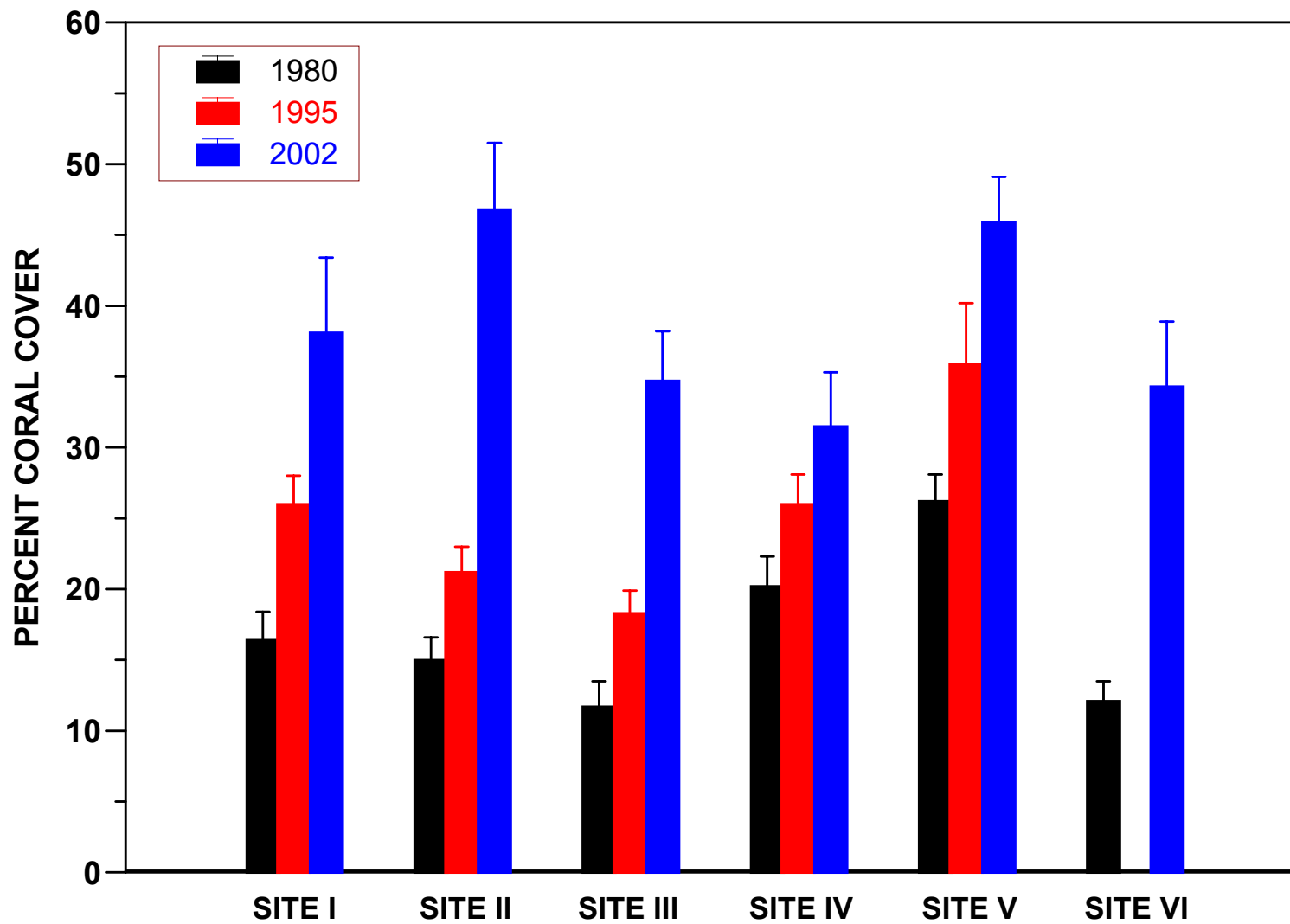


FIGURE 9. Histograms of coral cover (+ s.e) from photoquadrat transects conducted off of the Princeville Resort, Kauai, Hawaii in 1980, 1995 and 2002. See Figure 1 for Transect site locations.

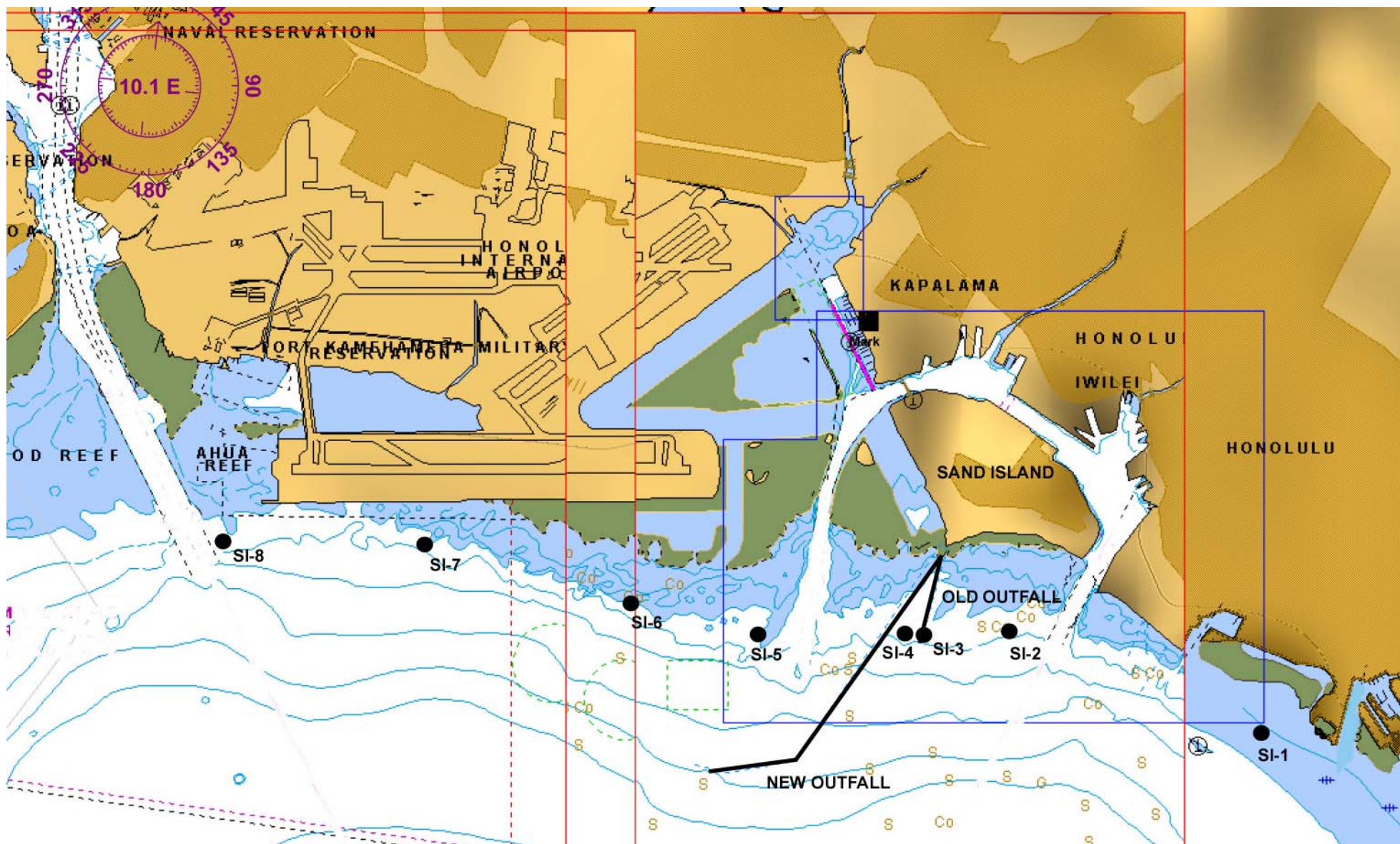


FIGURE 10. Chart of central Mamala Bay showing locations of old and new Sand Island Sewage Outfalls. Also shown are locations of eight transect sites where coral community structure was quantified during surveys conducted in 1975 while untreated sewage effluent was being discharged from the old outfall; 1979 approximately two years after sewage effluent was diverted to the new deep outfall; and in 2002, approximately 25 years after sewage was diverted to the new outfall.



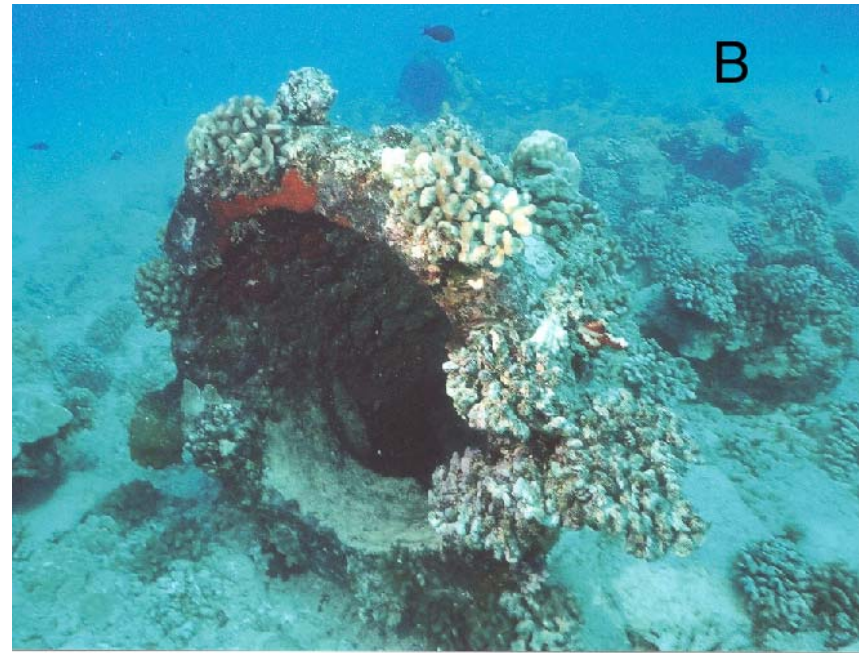
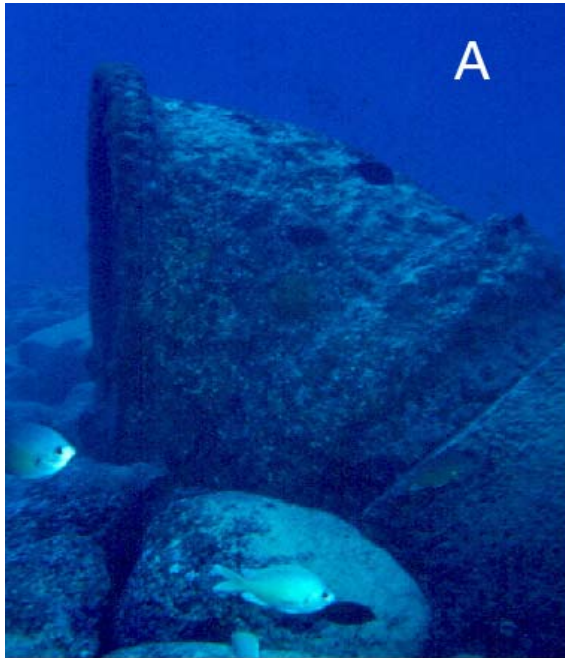


FIGURE 11. Old Sand Island Sewage Outfall Diffuser in 1979 (A) and 2002 (B and C). Transect photograph on armor rock of diffuser (Transect SI-3) shows numerous colonies of live and dead *Pocillopora meandrina* (D). Water depth in all photographs is approximately 10 m.

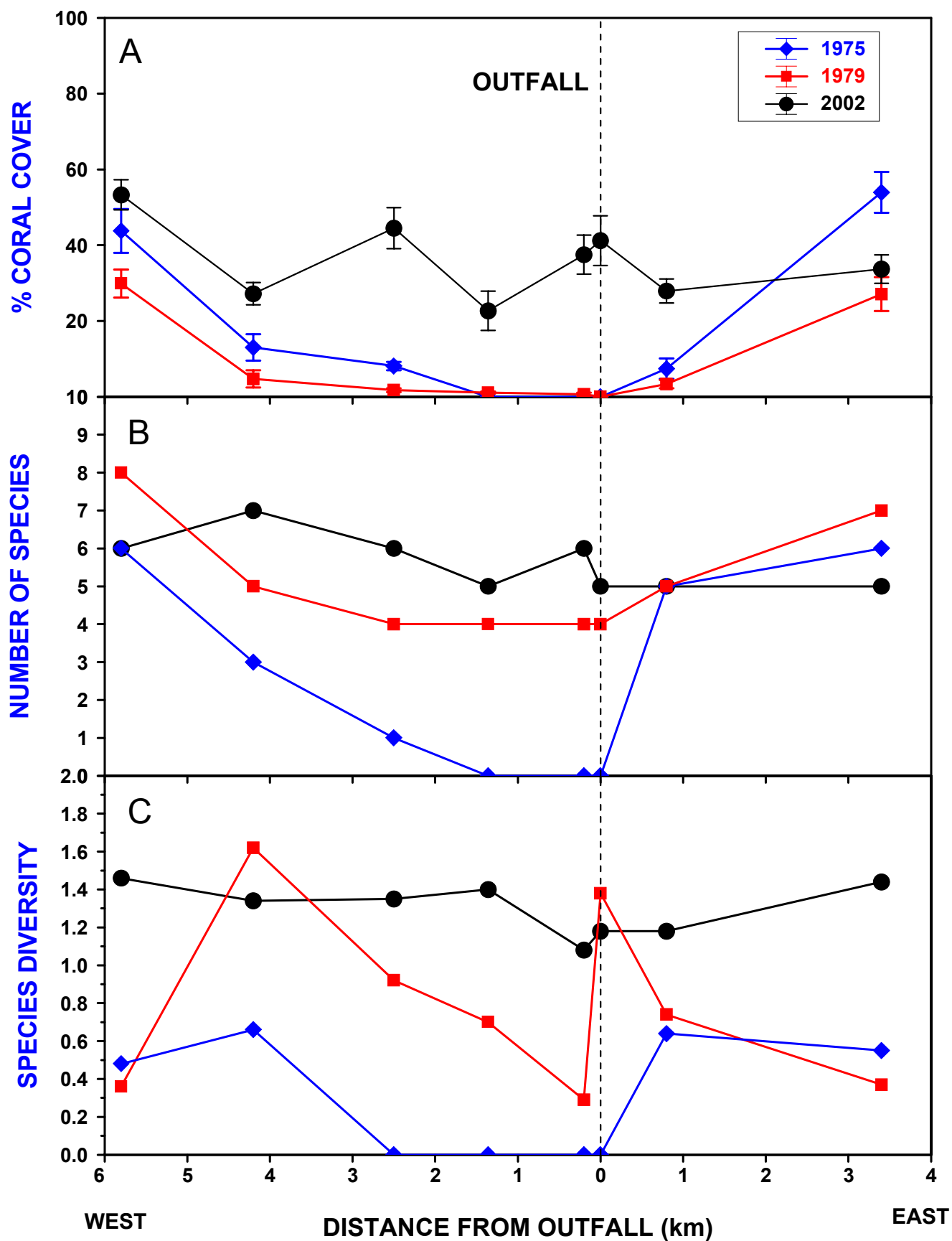


FIGURE 12. Total coral cover (+ s.e.) (A), number of coral species (B) and coral species diversity ( $H'_c$ ) (C) on transects in the vicinity of the old Sand Island Sewage Outfall, Mamala Bay, Oahu, Hawaii from surveys conducted in 1975 (while discharge from the outfall was occurring), 1979 (two years after cessation of sewage discharge) and in 2002 (25 years after cessation of discharge).



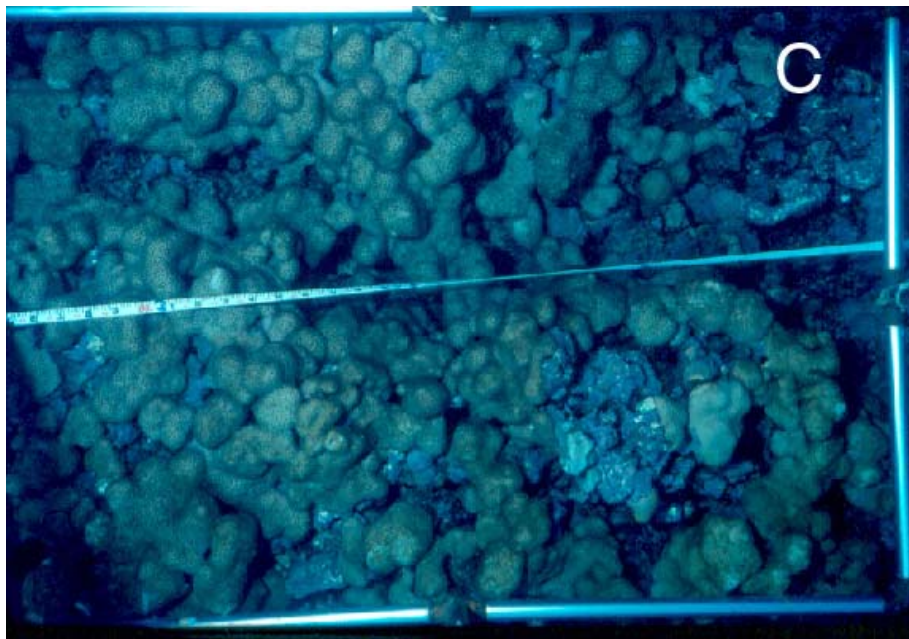
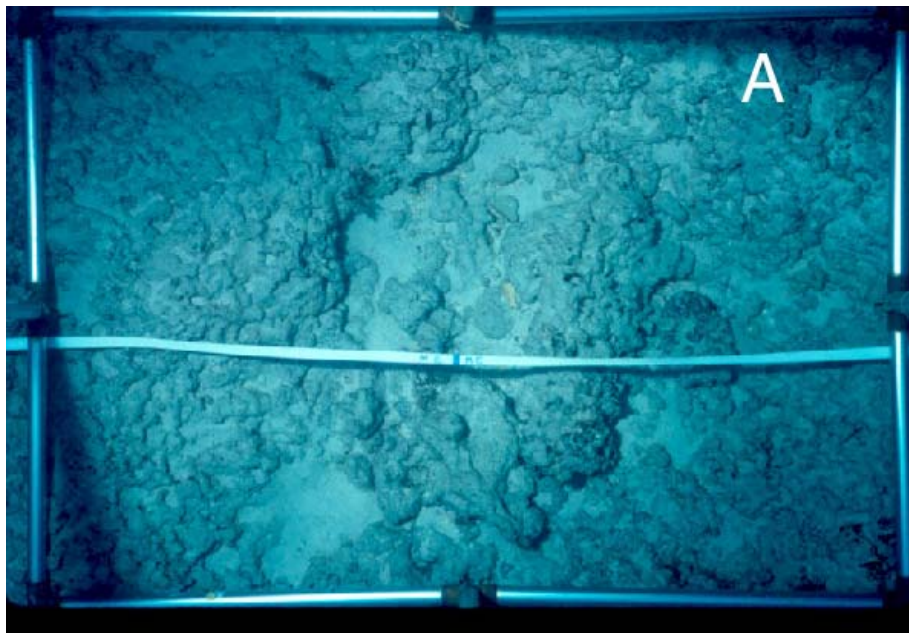


FIGURE 13. Photo-quadrats taken on transect SI-4 in 1978 (A) and 2002 (B). Transect SI-4 was in the high impact zone where bottom was essentially devoid of corals as a result of sewage discharge. In 2002 coral cover was similar to control area. Photoquadrats taken on transect SI-7 in 1978 (C) and 2002 (D) were located in intermediate impact zone, where calcareous algae dominated bottom cover after termination of discharge. Calcareous algae remains a significant bottom cover twenty-five years after cessation of discharge.

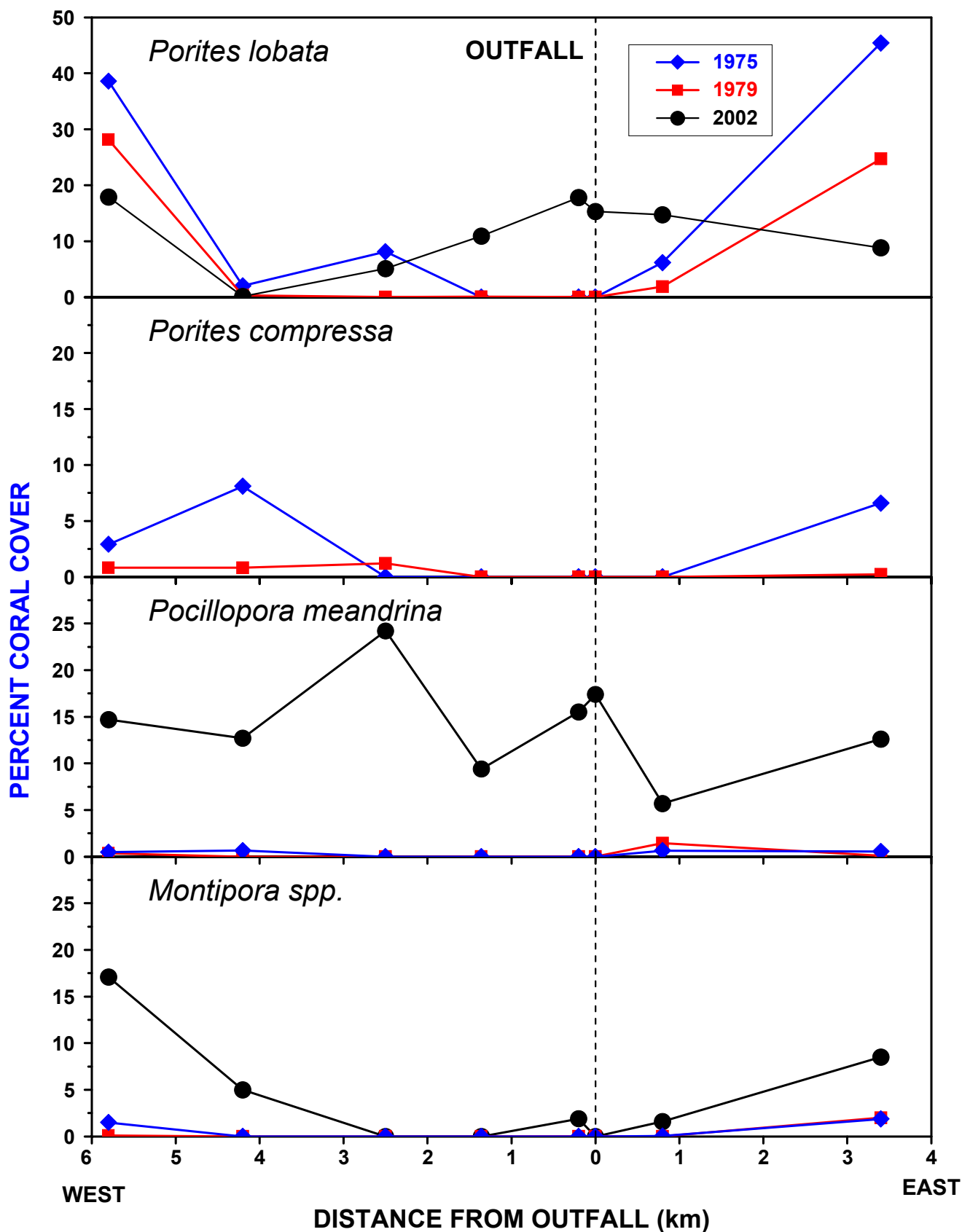


FIGURE 14. Coral cover of five most abundant species on transects in the vicinity of the old Sand Island Sewage Outfall, Mamala Bay, Oahu, Hawaii from surveys conducted in 1975 (while discharge from the outfall was occurring), 1979 (two years after cessation of sewage discharge) and in 2002 (25 years after cessation of discharge).



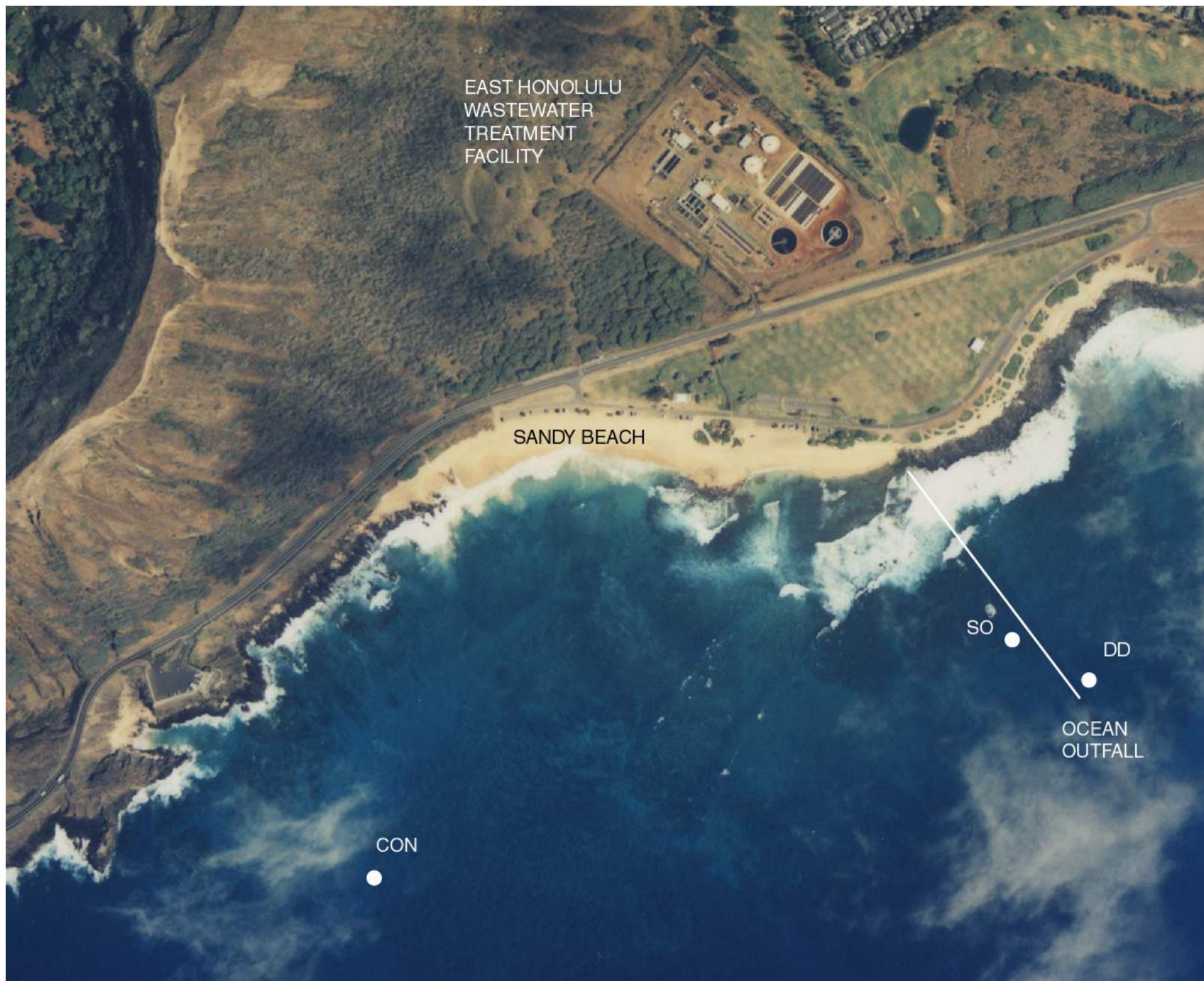


FIGURE 15. Aerial photography showing locations of East Honolulu Wastewater Treatment Facility and ocean outfall which terminates in a series of diffusers at a depth of about 12 m. Also shown are locations of three benthic transect monitoring stations: DD = deep diffuser; SO = shallow outfall; and CON = control.





FIGURE 16. Photographs of diffusers on the ocean outfall discharging sewage effluent from the East Honolulu Wastewater Treatment Facility. Water depth is approximately 11 m. Note corals growing on diffuser structures in the path of effluent discharge, and lack of macroalgae. Photo in upper right shows upward trajectory of effluent plume following discharge. All photos taken in July 2002.



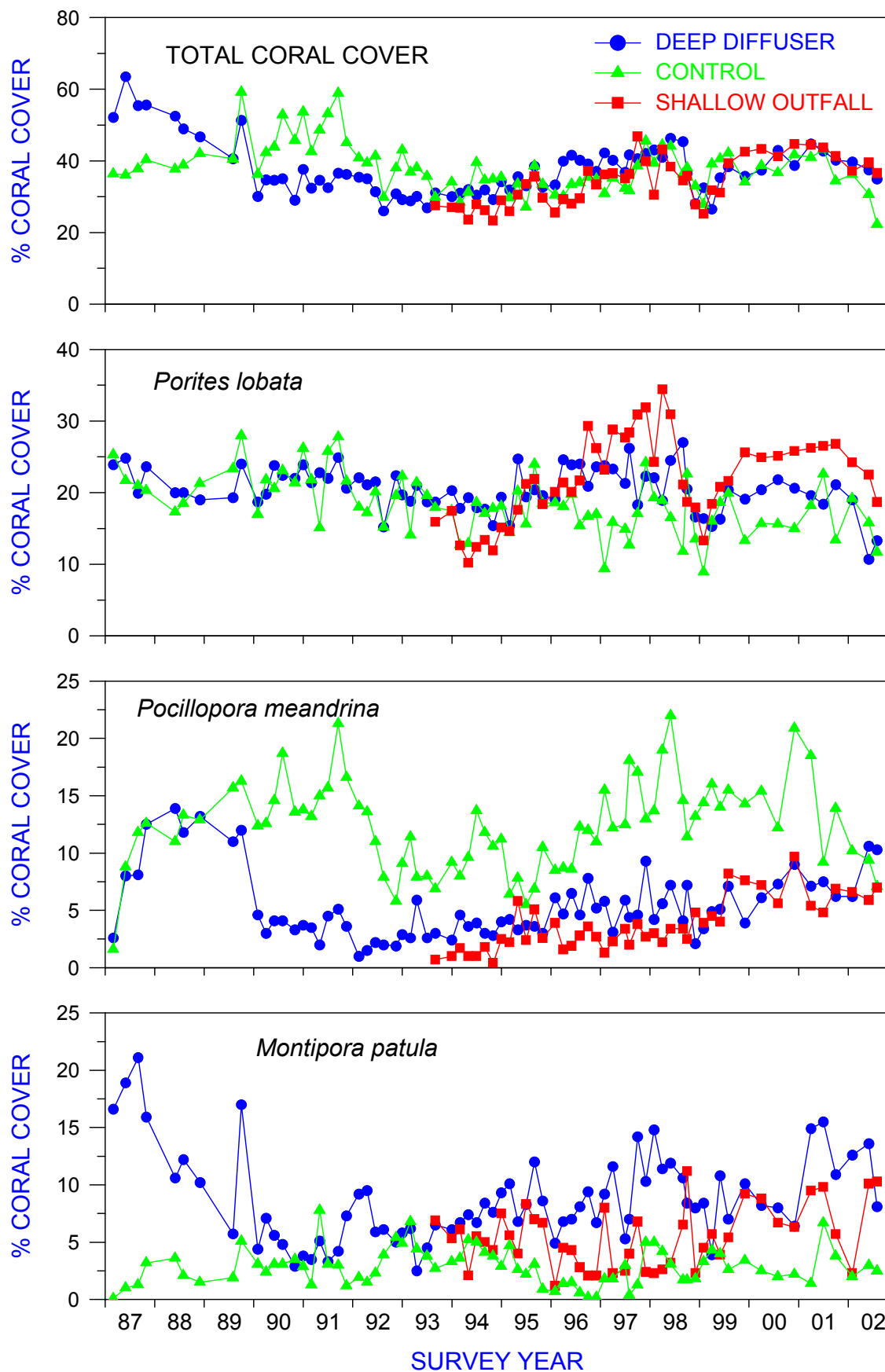


FIGURE 17. Percent coral cover at three benthic transect stations in the vicinity of the East Honolulu Wastewater Treatment Facility from surveys conducted from 1987 to 2002 (n = 74 for Deep Diffuser and Control Stations and 45 for Shallow Outfall Station. Note different scales on Y axes. See Figure 15 for locations of Stations.

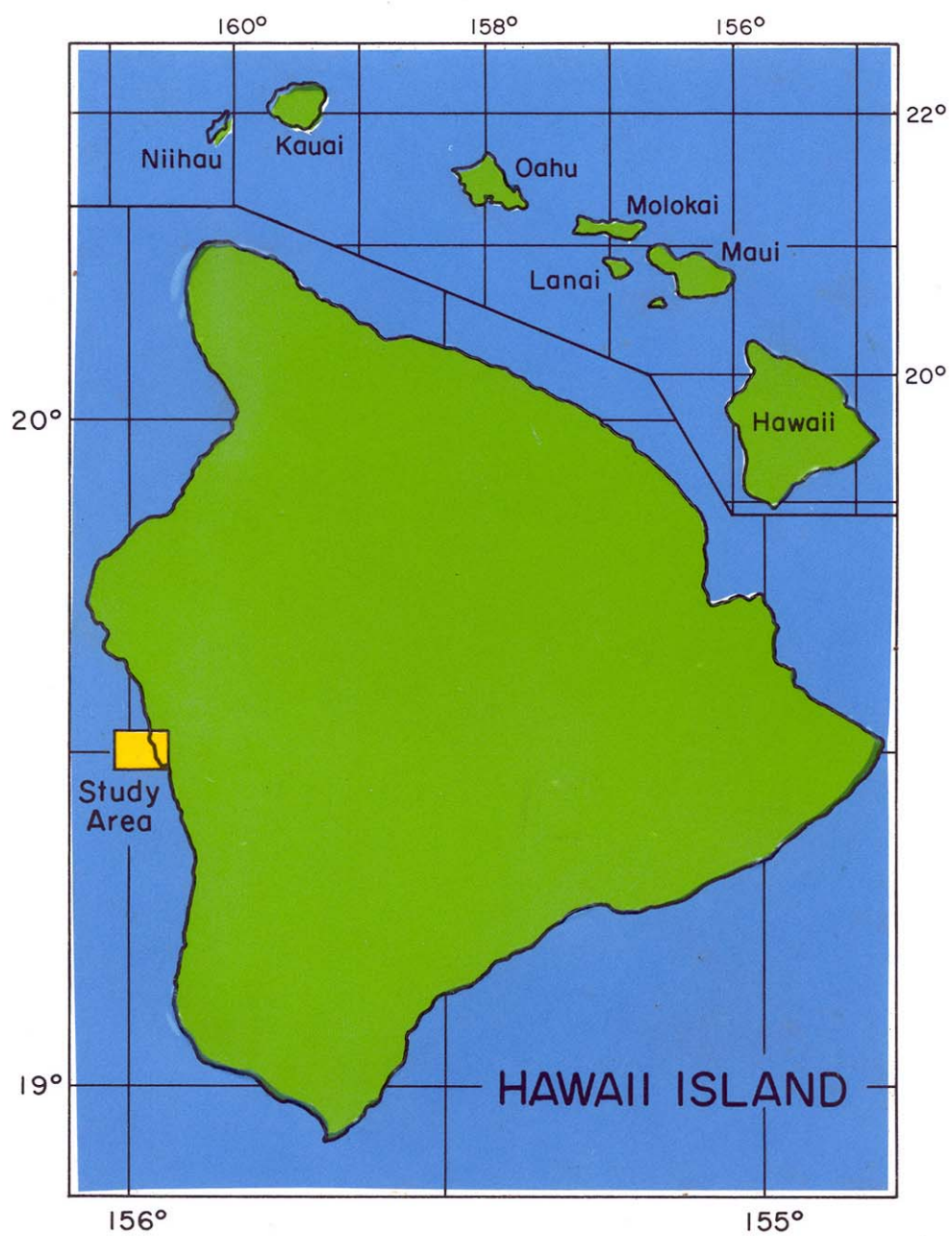


FIGURE 18. Map showing study area at Keawekaheka Point on West Coast of the Island of Hawaii.

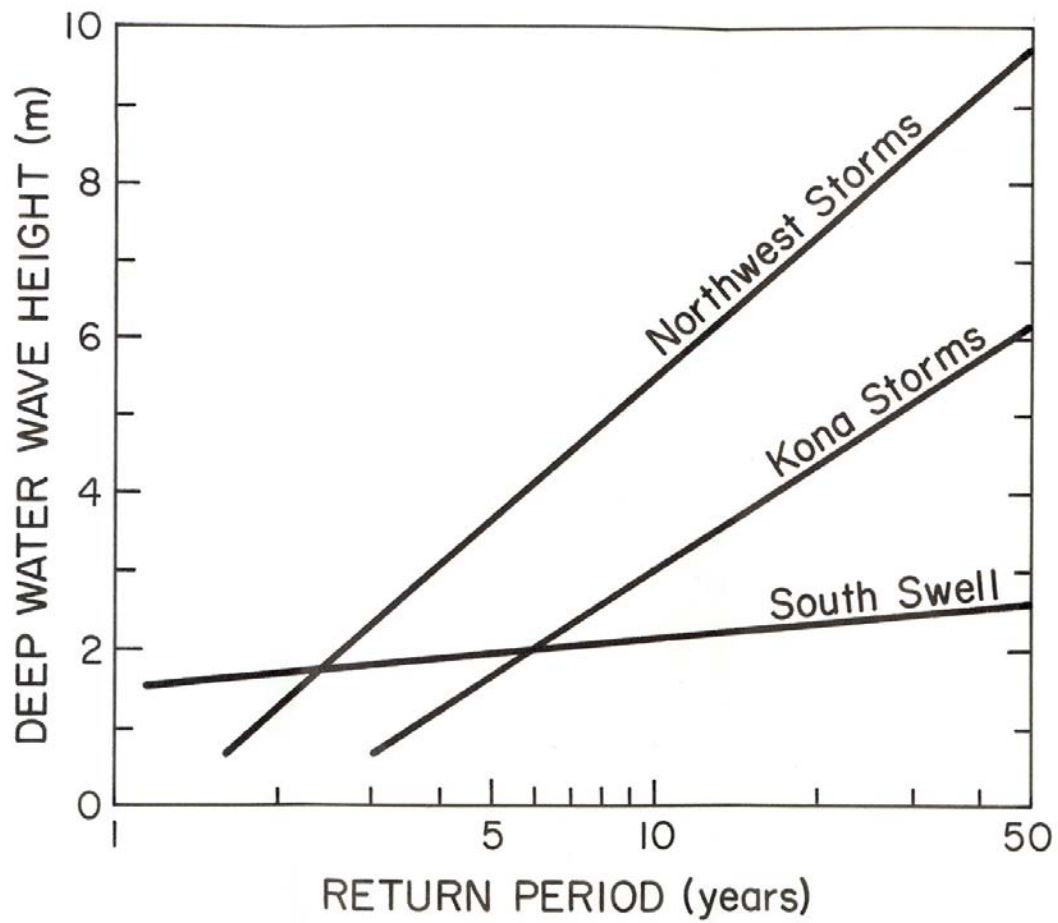


FIGURE 19. Plots of deepwater wave heights as functions of average return period of storm events (From Rocheleau and Sullivan, unpublished data, 1981).

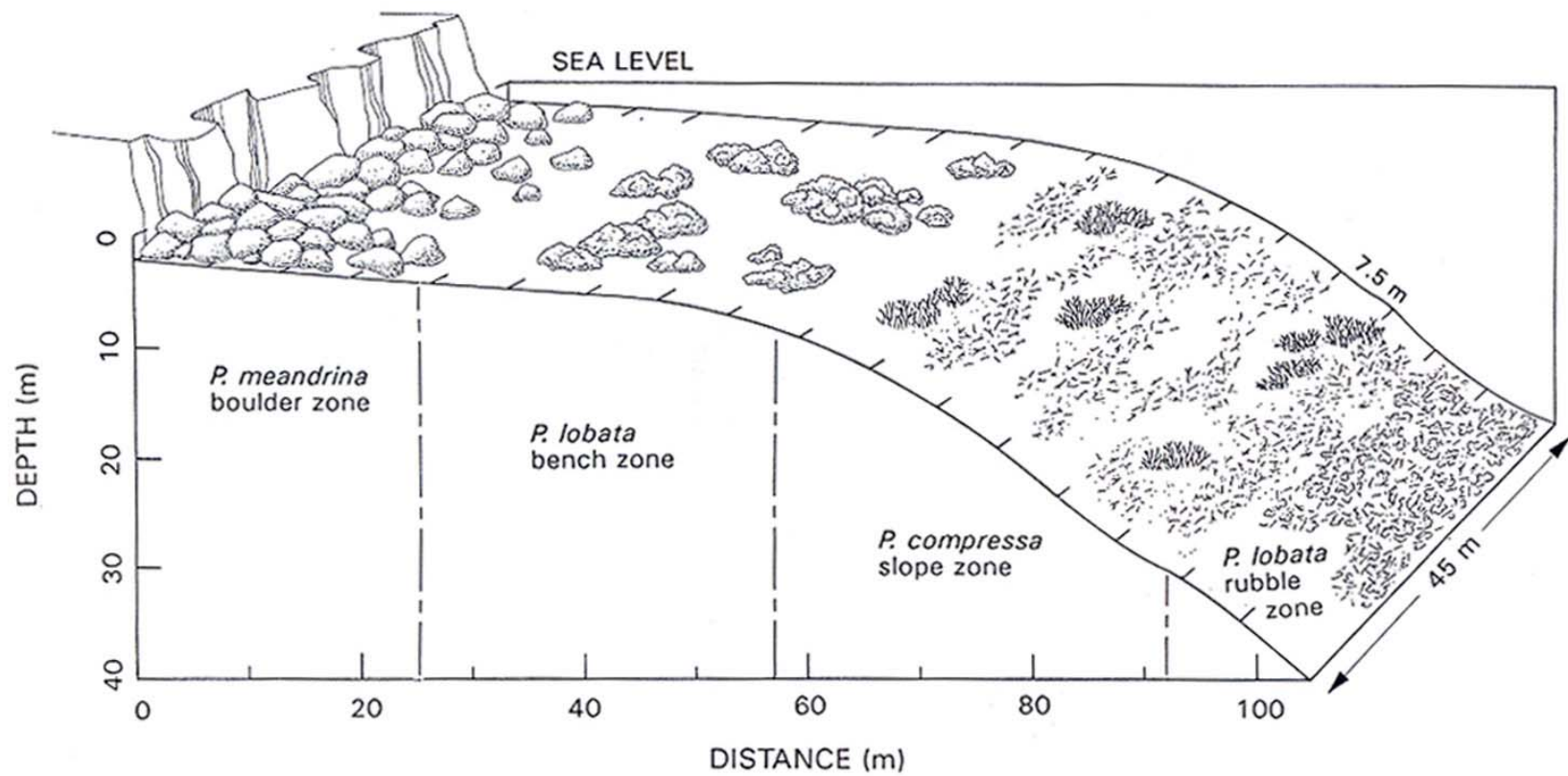


FIGURE 20. Schematic diagram of reef showing depth profile, approximate zone boundaries and transect orientation (tick marks at edges).

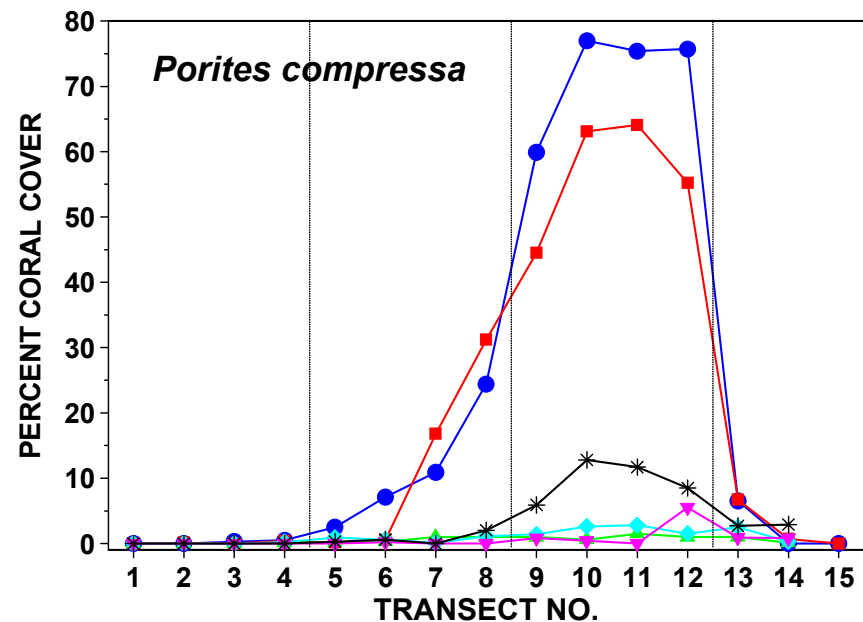
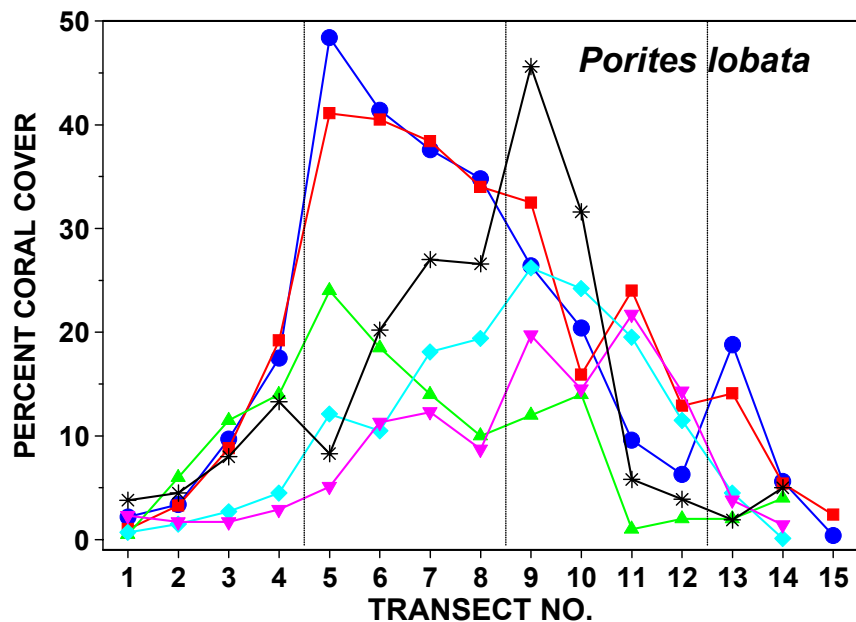
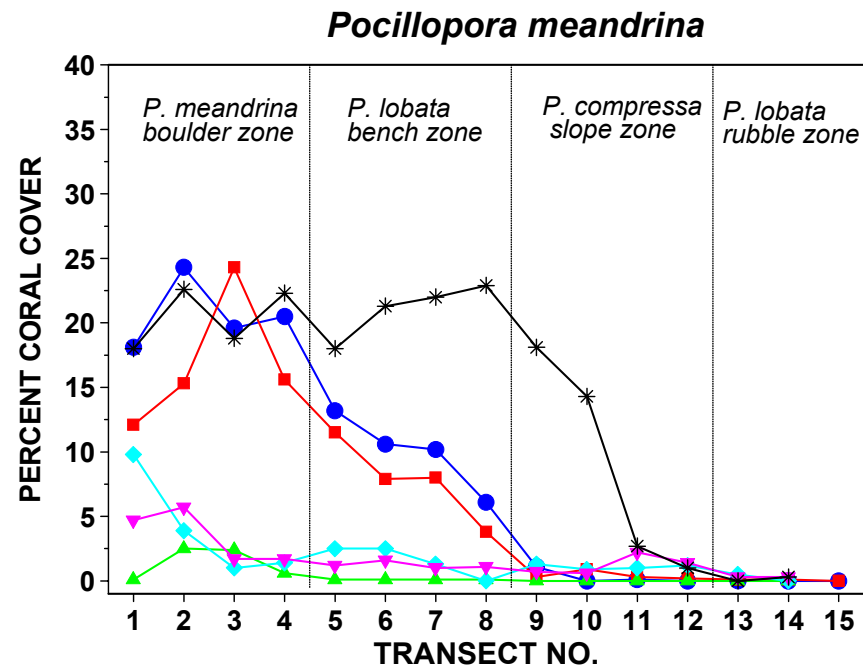
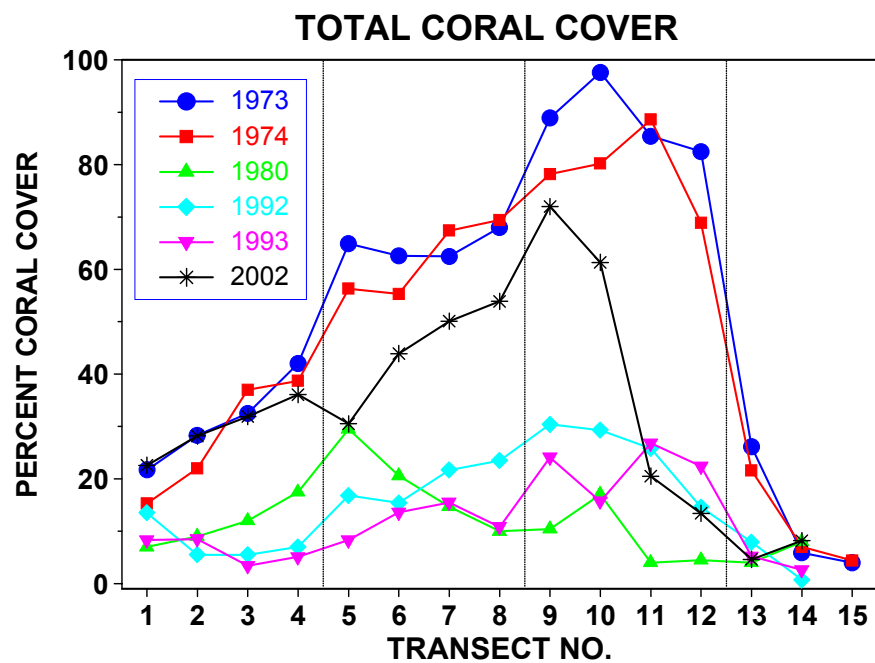


FIGURE 21. Plots of total coral cover and the three most abundant species on transects off of



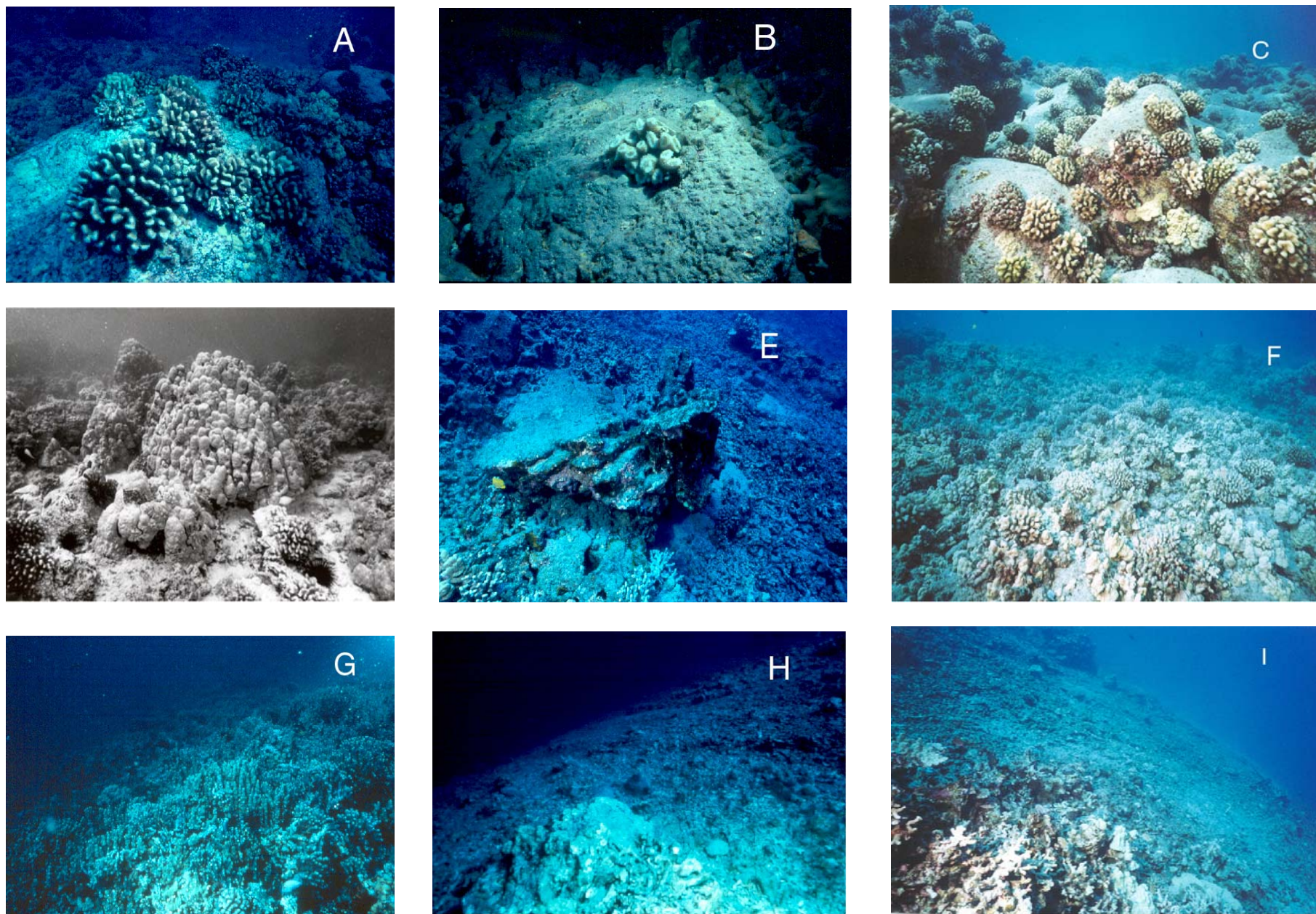


FIGURE 22. Underwater photographs of three major reef zones at Keawekaheka, Kona Hawaii during sequential surveys in 1973, 1980 and 2002. The survey in 1980 followed a major Kona storm event, while the 2002 survey followed about a decade that was free of major storms. Top row shows *Pocillopora meandrina* boulder zone in 1973 (A); 1980 (B) and 2002 (C). Middle row shows *Porites lobata* reef platform zone in '73 (D), large colonies that were broken from the reef and tumbled down the reef slope in '80 (E) and the current condition of the platform in 2002 (F). Note lack of large colonies in 2002 that covered the area in 1973. Bottom row shows *Porites compressa* slope zone in '73 (G) which was comprised of a interconnected mat of finger coral; the mat was turned to rubble in 1980 (H), which persists in 2002 (I).

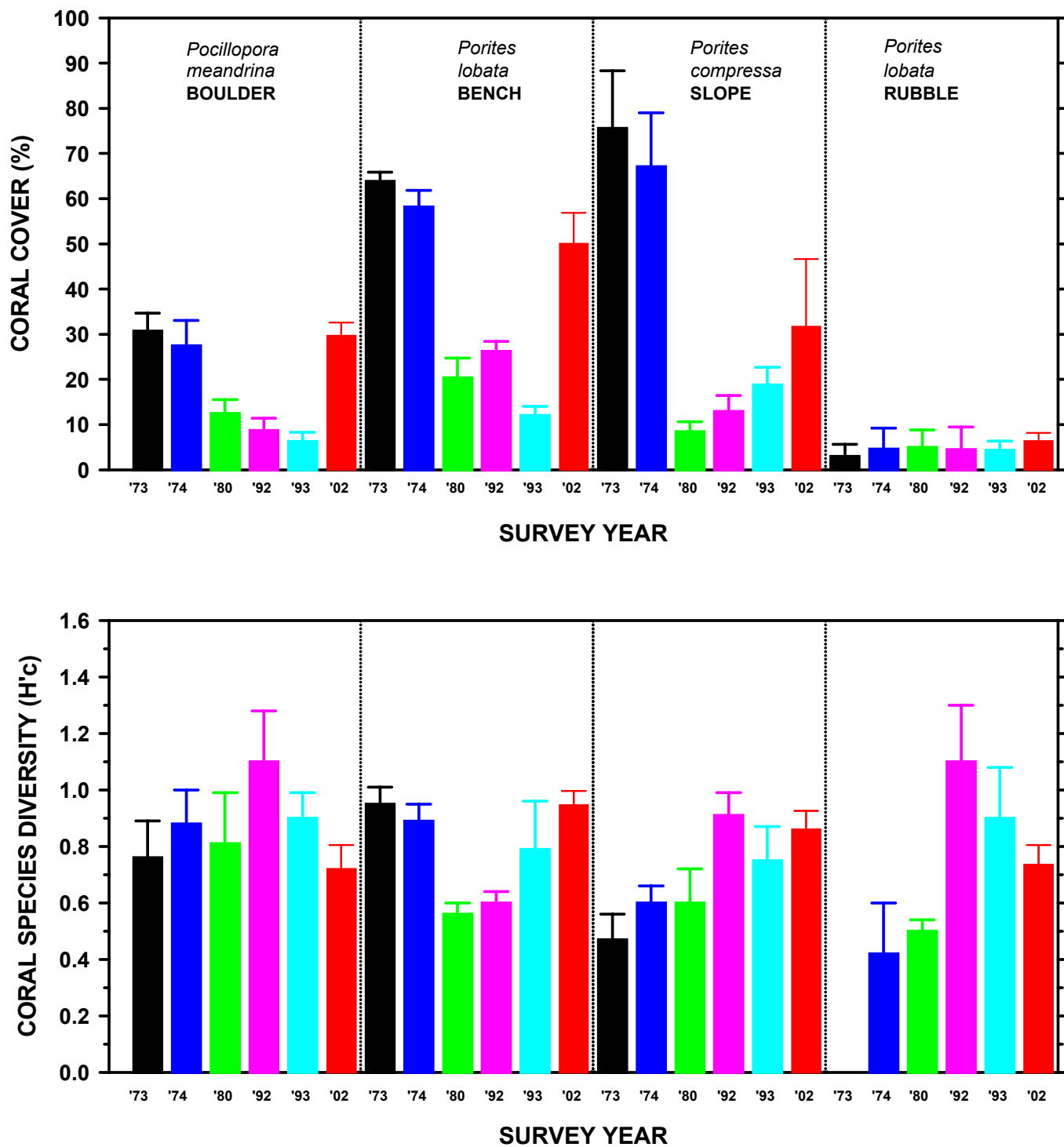


FIGURE 23. Histogram of total coral cover (top) and species cover diversity (bottom) of pooled transects each reef zone off Keawekaheka Point, Hawaii from 1973 to 2002. Error bars show standard error.

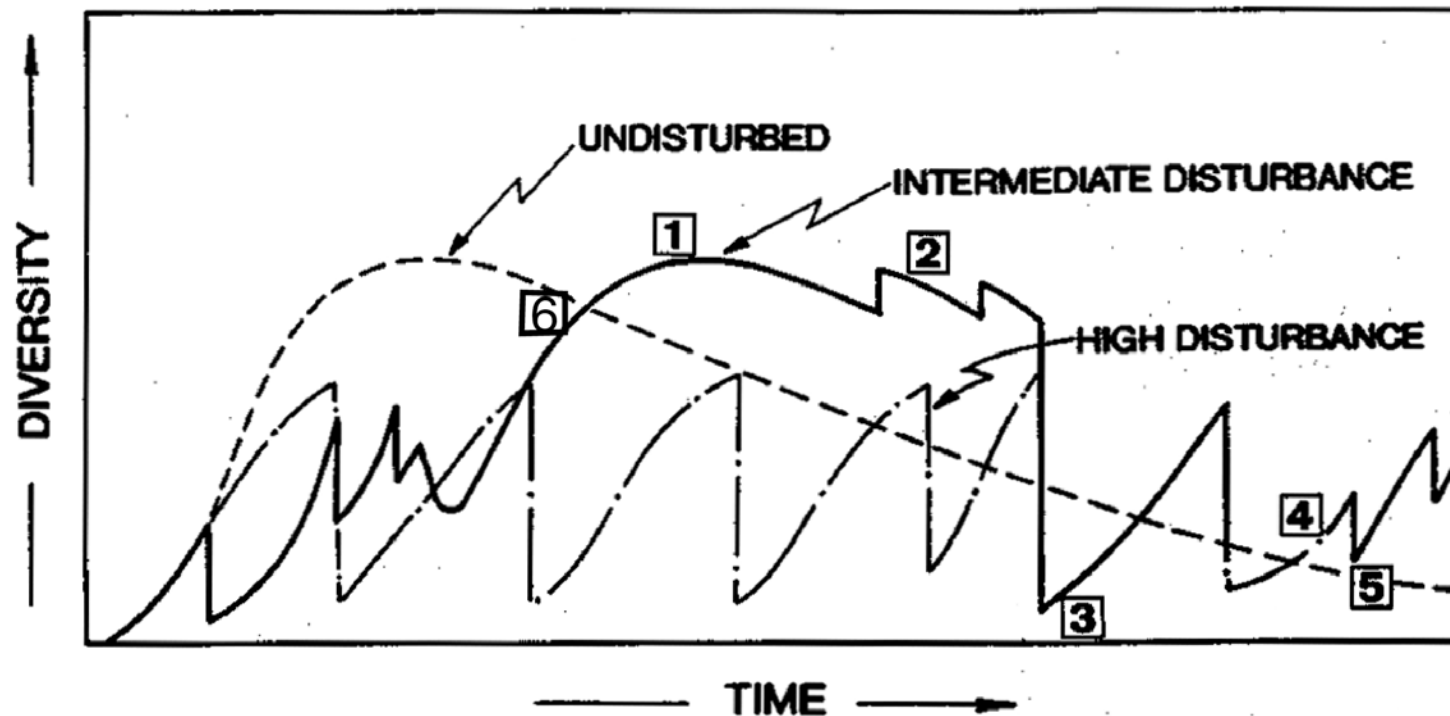


FIGURE 24. Theoretical model of succession of various coral reef communities showing relationship of diversity to intensity and frequency of disturbance. In the undisturbed case (dashed line), diversity increases during initial settlement and growth, and subsequently decreases as a result of competition for space when coral cover is maximal. Competitive exclusion by a single species results in decline in diversity. In the intermediate disturbance model (solid line), peak diversity is attained following a period of minimal disturbance; severe disturbances reset the process back to a near-zero state. Intermediate disturbance increase or decrease diversity depending on the successional stage. In the high disturbance model (dotted solid line), frequent severe disturbances and adaptive capabilities of species for rapid recovery result in a "sawtooth" pattern of diversity. Numbers 1-6 on intermediate disturbance model indicate approximate successional stages when surveys were conducted at the Keawekaheka, West Hawaii sites. Adapted from Grigg (1983).



TABLE 1. Percent cover, species number, and species cover diversity ( $H'$ ) for phototransects conducted in 1990, 1993, and 2002 in Honolua Bay, Maui, Hawaii.

TRANSECT	I-1			I-2			II-1			II-2		
SPECIES	1990	1992	2002	1990	1992	2002	1990	1992	2002	1990	1992	2002
<i>Porites lobata</i>	7.2	10.4	18.6	10.6	18.9	8.8	9.0	15.7	21.8	22.2	20.4	13.1
<i>Porites compressa</i>	4.2	3.5	1.9	12.2	8.8	4.6	2.3	8.9	6.3	46.3	44.7	29.0
<i>Porites brighami</i>	0.3						0.1					
<i>Pocillopora meandrina</i>			2.1		0.8	1.2	0.2	1.6	4.3			2.1
<i>Pocillopora eydouxi</i>	0.7			0.7			1.7					
<i>Montipora dilatata</i>	5.9	21.1	3.4	32.9	34.8	6.4	4.0	12.8	5.0	11.8	22.4	11.6
<i>Montipora patula</i>	4.9	18.6	2.2	12.9	6.9	2.4	7.5	18.9	5.1	4.6	1.1	4.0
<i>Montipora flabellata</i>	13.7	18.5	20.9	7.8			11.9	14.1	1.3			
<i>Pavona varians</i>	1.2	0.8	0.2	0.6	1.0		1.8	3.0	0.3	3.7	0.1	0.4
<i>Pavona duerdeni</i>		1.6	0.1				1.0	0.3				
<i>Leptastrea purpurea</i>							0.1		0.1			
<i>Palythoa tuberculosa</i>					0.1			0.2		0.1		
<i>Cyphastrea ocellina</i>	0.3				0.1		0.1					
TRANSECT TOTAL	38.4	74.5	49.4	77.7	71.4	23.4	39.7	75.5	44.2	88.7	88.7	60.2
Std. Err.	6.1	5.0	4.6	6.5	6.7	4.2	5.6	4.2	3.7	5.1	3.5	8.3
SPECIES NUMBER	9	7	8	7	8	5	12	9	8	6	5	6
SPECIES DIVERSITY	1.73	1.60	1.35	1.53	1.31	1.43	1.85	1.79	1.50	1.25	1.08	1.33

TRANSECT	III-1			III-2			IV-1			IV-2		
SPECIES	1990	1992	2002	1990	1992	2002	1990	1992	2002	1990	1992	2002
<i>Porites lobata</i>	27.6	50.6	8.0	14.0	0.1	19.3	35.1	27.7	34.7	28.8	2.5	0.7
<i>Porites compressa</i>	11.1	7.2	1.7	24.5	59.9	25.7	3.2	33.6	0.2	17.7	49.1	40.8
<i>Pocillopora meandrina</i>		3.0	1.5				0.3	0.4	2.4	1.3		
<i>Pocillopora eydouxi</i>							7.6			0.0		
<i>Montipora dilatata</i>	18.5	4.8	3.8	37.0	7.3	10.6	3.8	1.7	0.6	24.5	17.4	0.8
<i>Montipora patula</i>	15.7	11.5	2.3	14.4	6.0	4.8	5.4	5.7	3.1	12.1		4.5
<i>Montipora flabellata</i>		1.8	1.7		0.1		5.2		0.6	0.5		
<i>Pavona varians</i>	2.8	0.2		0.9	1.7	1.2	2.0			1.2	2.4	1.4
<i>Pavona duerdeni</i>	3.1	0.8					0.4				1.3	
<i>Leptastrea purpurea</i>		0.1							0.2	0.2		
<i>Psammocora stellata</i>					5.4							
<i>Cyphastrea ocellina</i>		0.1										
<i>Leptastrea bottae</i>								0.1				
TRANSECT TOTAL	78.8	80.1	19.0	90.8	80.5	61.6	63.0	69.2	41.8	86.3	72.7	48.2
Std. Err.	6.6	4.5	4.1	2.5	6.2	8.8	4.2	3.7	4.8	4.3	3.2	6.5
SPECIES NUMBER	6	10	6	5	7	5	9	6	7	8	5	5
SPECIES DIVERSITY	1.55	1.25	1.57	1.34	0.91	1.31	1.48	1.05	0.68	1.49	0.91	0.59

TABLE 2. Observed test criteria (T) for nonparametric Wilcoxon matched-pairs Signed-ranks test for related samples comparing total coral cover on 10 quadrats comprising transects at Honolua Bay, Mauna Lani and Princeville between sampling dates. "\*" indicates significant difference for two-tailed tests ( $P = 0.02$ ); "\*\*\*" indicates significance for two-tailed test ( $P = 0.01$ ). Underlined T criterion indicates significant decrease in cover; bold T criterion indicates significant increase in coral cover between surveys. "ND" indicates no data for Transect VI at Princeville in 1995. Individual quadrat data missing for Mauna Lani in 1980. For locations of Transects, see Figure 1.

HONOLUA BAY	TRANSECT								
	I-1	I-2	II-1	II-2	III-1	III-2	IV-1	IV-2	-
'90-'92	<b>0**</b>	23	<b>1**</b>	27	25	17	17	8	
'92-'02	<u>4*</u>	<u>0**</u>	<u>0**</u>	<u>4*</u>	<u>0**</u>	15	<u>2**</u>	<u>4*</u>	
'90-'02	15	<u>1**</u>	22	8	<u>0**</u>	<u>0**</u>	10	<u>1**</u>	
MAUNA LANI	I-6	I-10	I-20	II-6	II-10	II-20	III-6	III-10	III-20
'93-'02	<b>0**</b>	<b>3**</b>	<b>0**</b>	11	<b>2**</b>	28.0	<b>0**</b>	<b>0**</b>	10
	IV-6	IV-10	IV-20	V-6	V-10	V-20	VI-6	VI-10	VI-20
'93-'02	<b>0**</b>	12	26	<b>0**</b>	<b>0**</b>	13	16	<b>0**</b>	<b>4*</b>
PRINCEVILLE	I	II	III	IV	V	VI	-	-	-
'80-'95	<b>4*</b>	<b>5*</b>	<b>7</b>	8	7	ND			
'95-'02	15	<b>2**</b>	<b>2**</b>	17	15	ND			
'80-'02	<b>0**</b>	<b>0**</b>	<b>1**</b>	<b>2**</b>	<b>1**</b>	<b>0**</b>			

TABLE 3. Pooled coral cover data for transects in Honolua Bay, Mauna Lani, and Princeville showing relative percentage of coral cover (% cc) and percentage of total bottom cover (% bc) for three surveys. Order of species in table is based on highest rank of abundance in earliest survey.

#### HONOLUA BAY

SPECIES	1990		1992		2002	
	% cc	% bc	% cc	% bc	% cc	% bc
<i>Porites lobata</i>	27.42	19.31	23.87	18.29	35.98	15.63
<i>Montipora capitata</i>	24.57	17.30	25.13	19.25	11.92	5.18
<i>Porites compressa</i>	21.57	15.19	30.03	23.00	31.72	13.78
<i>Montipora patula</i>	13.76	9.69	11.21	8.59	8.18	3.55
<i>Montipora flabellata</i>	6.94	4.89	5.63	4.31	7.05	3.06
<i>Pavona varians</i>	2.52	1.78	1.50	1.15	0.89	0.39
<i>Pocillopora eydouxi</i>	1.90	1.34	0.00	0.00	0.00	0.00
<i>Pavona duerdeni</i>	0.80	0.56	0.60	0.46	0.09	0.04
<i>Pocillopora meandrina</i>	0.32	0.23	0.95	0.73	4.15	1.80
<i>Porites brighami</i>	0.07	0.05	0.00	0.00	0.00	0.00
<i>Cyphastrea ocellina</i>	0.07	0.05	0.07	0.05	0.00	0.00
<i>Leptastrea purpurea</i>	0.05	0.04	0.07	0.05	0.03	0.01
<i>Porites rus</i>	0.00	0.00	0.88	0.68	0.00	0.00
<i>Leptastrea bottae</i>	0.00	0.00	0.02	0.01	0.00	0.00
<i>Palythoa tuberculosa</i>	0.00	0.00	0.05	0.04	0.00	0.00
TOTAL	100.00	70.41	100.00	76.60	100.00	43.43

#### MAUNA LANI

SPECIES	1983		1993		2002	
	% cc	% bc	% cc	% bc	% cc	% bc
<i>Porites lobata</i>	68.92	37.20	62.61	30.55	53.92	43.27
<i>Porites compressa</i>	25.83	13.94	32.46	15.84	28.36	22.76
<i>Pocillopora meandrina</i>	3.67	1.98	1.97	0.96	9.55	7.66
<i>Montipora patula</i>	0.65	0.35	1.14	0.56	3.34	2.68
<i>Pavona varians</i>	0.52	0.28	0.43	0.21	1.31	1.05
<i>Montipora capitata</i>	0.32	0.17	1.16	0.57	3.45	2.77
<i>Leptastrea purpurea</i>	0.07	0.04	0.10	0.05	0.02	0.02
<i>Pocillopora eydouxi</i>	0.00	0.00	0.08	0.04	0.06	0.04
<i>Cyphastrea ocellina</i>	0.01	0.01	0.02	0.01	0.00	0.00
<i>Palythoa tuberculosa</i>	0.00	0.00	0.02	0.01	0.00	0.00
<i>Porites brighami</i>	0.00	0.00	0.00	0.00	0.01	0.01
<i>Pavona duerdeni</i>	0.00	0.00	0.01	0.01	0.00	0.00
TOTAL	100.00	53.97	100.00	48.80	100.00	80.25

#### PRINCEVILLE

SPECIES	1980		1995		2002	
	% cc	% bc	% cc	% bc	% cc	% bc
<i>Porites lobata</i>	40.6	6.9	27.1	6.9	20.4	7.9
<i>Montipora patula</i>	24.9	4.2	32.7	8.3	36.0	13.9
<i>Porites compressa</i>	9.3	1.6	2.4	0.6	4.9	1.9
<i>Montipora flabellata</i>	7.4	1.3	9.6	2.4	14.3	5.5
<i>Montipora capitata</i>	5.7	1.0	9.3	2.4	10.0	3.9
<i>Pocillopora meandrina</i>	4.5	0.8	10.6	2.7	5.5	2.1
<i>Pavona varians</i>	3.4	0.6	1.9	0.5	1.3	0.5
<i>Pavona duerdeni</i>	2.7	0.5	2.9	0.7	3.0	1.2
<i>Porites brighami</i>	0.3	0.1	0.0	0.0	0.0	0.0
<i>Palythoa tuberculosa</i>	1.1	0.2	1.6	0.4	3.8	1.5
<i>Fungia scutaria</i>	0.1	0.0	0.1	0.0	0.0	0.0
<i>Porites (S.) canvexa</i>	0.0	0.0	0.0	0.0	0.1	0.0
<i>Pocillopora eydouxi</i>	0.0	0.0	0.0	0.0	0.6	0.2
<i>Psammocora stellata</i>	0.0	0.0	1.0	0.3	0.0	0.0
<i>Pocillopora damicornis</i>	0.0	0.0	0.5	0.1	0.0	0.0
<i>Cyphastrea ocellina</i>	0.0	0.0	0.4	0.1	0.0	0.0
TOTAL	100.00	16.94	100.00	25.38	100.00	38.60

TABLE 4. Percent cover, number of species, and species cover diversity (H'<sub>c</sub>) for photo-quadrat transects conducted in 1975, 1979, and 2002 off the old Sand Island Sewage Outfall, Mamala Bay, Hawaii.

TRANSECT	SI-1			SI-2			SI-3			SI-4		
SPECIES	1975	1979	2002	1975	1979	2002	1975	1979	2002	1975	1979	2002
<i>Porites lobata</i>	45.40	24.70	8.80	6.17	1.87	14.70		0.01	15.30		0.02	17.80
<i>Porites compressa</i>	6.60	0.22										
<i>Pocillopora meandrina</i>	0.04	0.09	12.60	0.46	1.44	5.70		0.01	17.40		0.01	15.50
<i>Pocillopora eydouxi</i>			3.80			5.90			6.50			2.10
<i>Montipora patula</i>	0.47	0.01	6.80	0.02		0.30			1.90			1.60
<i>Montipora capitata</i>	1.40	2.00	1.70	0.20	0.01	1.30		0.01			0.01	0.30
<i>Montipora flabellata</i>												0.20
<i>Pavona varians</i>					0.01							
<i>Leptastrea purpurea</i>	0.02			0.55	0.01			0.01	0.10		0.61	
<i>Psammocora stellata</i>		0.01										
<i>Palthoa tuberculosa</i>		0.07										
TRANSECT TOTAL	53.93	27.10	33.70	7.40	3.34	27.90	0.00	0.04	41.20	0.00	0.65	37.50
std. Err.	5.4	4.5	3.8	2.7	1.1	3.2	0.0	0.0	6.6	0.0	0.4	5.2
Number of Species	6	7	5	5	5	5	0	4	5	0	4	6
Species Diversity	0.55	0.37	1.44	0.64	0.74	1.18	0.00	1.38	1.18	0.00	0.29	1.08

TRANSECT	SI-5			SI-6			SI-7			SI-8		
SPECIES	1975	1979	2002	1975	1979	2002	1975	1979	2002	1975	1979	2002
<i>Porites lobata</i>		0.05	10.90	8.12	0.03	5.10	2.02	0.33	0.10	38.60	28.14	17.90
<i>Porites compressa</i>					1.20		8.10	0.81		2.92	0.81	
<i>Porites brighami</i>			0.10									
<i>Pocillopora meandrina</i>		0.01	9.40			24.20			12.70	0.46	0.37	14.70
<i>Pocillopora eydouxi</i>			1.20									2.80
<i>Montipora patula</i>			1.10			4.50			4.80	0.82	0.07	13.80
<i>Montipora capitata</i>									0.20	0.67	0.02	3.30
<i>Pavona varians</i>					0.33	1.30		1.00	0.30		0.05	
<i>Pavona duerdeni</i>						7.50		0.96	6.30			0.80
<i>Leptastrea purpurea</i>		1.02									0.05	
<i>Palythoa tuberculosa</i>		0.01			0.20	1.90	2.90	1.63	2.80	0.30	0.37	
TRANSECT TOTAL	0.00	1.09	22.70	8.12	1.76	44.50	13.02	4.73	27.20	43.77	29.88	53.30
std. Err.	0.0	0.3	5.2	1.1	0.5	5.4	3.5	2.3	3.0	5.8	3.7	4.0
Number of Species	0	4	5	1	4	6	3	5	7	6	8	6
Species Diversity	0.00	0.70	1.40	0.00	0.92	1.35	0.66	1.62	1.34	0.48	0.36	1.46

Table 5. Linear regression statistics for coral cover on transects in the vicinity of the East Honolulu Wastewater Treatment Facility Ocean Outfall. Top table includes all surveys from the monitoring program (June 1987 to November 2002). Other tables include periods from 1987-1993, 1993-1998, 1998-2000, and 2000-2002. "INC" = significant increase in coral cover; "DEC" = significant decrease in coral cover; and "NC" = no significant change in coral cover ( $p < 0.05$ ).

1987-2002 (n = 75)	SPECIES	SLOPE	STUDENTS t CON. LIMITS		CHANGE
			LOWER	UPPER	
DEEP DIFFUSER	TOTAL CORAL	-0.028	-0.063	0.007	NC
	<i>P. lobata</i>	-0.018	-0.031	-0.004	DEC
	<i>P. meandrina</i>	-0.004	-0.014	0.014	NC
	<i>M. patula</i>	0.003	-0.015	0.021	NC
CONTROL	TOTAL CORAL	-0.050	-0.080	-0.019	DEC
	<i>P. lobata</i>	-0.040	-0.057	-0.024	DEC
	<i>P. meandrina</i>	0.011	-0.008	0.029	NC
	<i>M. patula</i>	0.000	-0.007	0.008	NC

6/87 - 9/93 (n=30)	SPECIES	SLOPE	STUDENTS t CON. LIMITS		CHANGE
			LOWER	UPPER	
DEEP DIFFUSER	TOTAL CORAL	-0.375	-0.495	-0.314	DEC
	<i>P. lobata</i>	-0.026	-0.064	0.010	NC
	<i>P. meandrina</i>	-0.116	-0.167	-0.065	DEC
	<i>M. patula</i>	-0.169	-0.228	-0.109	DEC
CONTROL	TOTAL CORAL	0.017	-0.106	0.139	NC
	<i>P. lobata</i>	-0.035	-0.095	0.076	NC
	<i>P. meandrina</i>	0.005	-0.063	0.074	NC
	<i>M. patula</i>	0.037	-0.010	0.060	NC

11/93 - 10/98 (n=30)	SPECIES	SLOPE	STUDENTS t CON. LIMITS		CHANGE
			LOWER	UPPER	
DEEP DIFFUSER	TOTAL CORAL	0.233	0.179	0.787	INC
	<i>P. lobata</i>	0.013	0.039	0.146	INC
	<i>P. meandrina</i>	0.010	-0.060	0.081	NC
	<i>M. patula</i>	0.069	0.024	0.114	INC
SHALLOW DIFFUSER	TOTAL CORAL	0.240	0.159	0.320	INC
	<i>P. lobata</i>	0.291	0.204	0.378	INC
	<i>P. meandrina</i>	0.026	0.002	0.050	INC
	<i>M. patula</i>	-0.016	-0.067	0.033	NC
CONTROL	TOTAL CORAL	0.370	0.054	0.272	INC
	<i>P. lobata</i>	0.055	0.022	0.083	INC
	<i>P. meandrina</i>	0.144	0.082	0.206	INC
	<i>M. patula</i>	-0.024	-0.056	0.007	NC

10/98 - 11/00 (n=10)	SPECIES	SLOPE	STUDENTS t CON. LIMITS		CHANGE
			LOWER	UPPER	
DEEP DIFFUSER	TOTAL CORAL	0.503	0.162	0.844	INC
	P. LOBATA	0.242	0.105	0.380	INC
	P. MEANDRINA	0.216	0.088	0.344	INC
	M. PATULA	-0.082	-0.244	0.208	NC
SHALLOW DIFFUSER	TOTAL CORAL	0.787	0.426	1.149	INC
	P. LOBATA	0.445	0.207	0.085	INC
	P. MEANDRINA	0.179	0.023	0.334	INC
	M. PATULA	0.171	-0.011	0.353	NC
CONTROL	TOTAL CORAL	0.257	-0.196	0.709	NC
	P. LOBATA	0.072	-0.262	0.407	NC
	P. MEANDRINA	0.135	-0.101	0.371	NC
	M. PATULA	-0.044	-0.131	0.042	NC

11/00 - 11/02 (n=6)	SPECIES	SLOPE	STUDENTS t CON. LIMITS		CHANGE
			LOWER	UPPER	
DEEP DIFFUSER	TOTAL CORAL	-0.293	-0.672	0.086	NC
	P. LOBATA	-0.422	-0.806	-0.038	DEC
	P. MEANDRINA	0.102	-0.161	0.366	NC
	M. PATULA	0.016	-0.521	0.553	NC
SHALLOW DIFFUSER	TOTAL CORAL	-0.420	-0.633	-0.207	DEC
	P. LOBATA	-0.321	-0.592	-0.049	DEC
	P. MEANDRINA	-0.061	-0.299	0.175	NC
	M. PATULA	-0.063	-0.406	0.532	NC
CONTROL	TOTAL CORAL	-0.883	-1.430	-0.033	DEC
	P. LOBATA	-0.174	-0.72	0.371	NC
	P. MEANDRINA	-0.614	-1.023	-0.205	DEC
	M. PATULA	-0.009	-0.078	0.267	NC

TABLE 6. Percent cover, number of species, and species cover diversity (H'<sub>c</sub>) for photo-quadrat transects conducted in 1975, 1979, and 2002 off the old Sand Island Sewage Outfall, Mamala Bay, Hawaii.

TRANSECT	SI-1			SI-2				SI-3			SI-4		
SPECIES	1975	1979	2002	1975	1979	2002	1975	1979	2002	1975	1979	2002	
<i>Porites lobata</i>	45.40	24.70	8.80	6.17	1.87	14.70		0.01	15.30		0.02	17.80	
<i>Porites compressa</i>	6.60	0.22											
<i>Pocillopora meandrina</i>	0.04	0.09	12.60	0.46	1.44	5.70		0.01	17.40		0.01	15.50	
<i>Pocillopora eydouxi</i>			3.80			5.90			6.50			2.10	
<i>Montipora patula</i>	0.47	0.01	6.80	0.02		0.30			1.90			1.60	
<i>Montipora capitata</i>	1.40	2.00	1.70	0.20	0.01	1.30		0.01			0.01	0.30	
<i>Montipora flabellata</i>												0.20	
<i>Pavona varians</i>					0.01								
<i>Leptastrea purpurea</i>	0.02			0.55	0.01			0.01	0.10		0.61		
<i>Psammocora stellata</i>		0.01											
<i>Palthoa tuberculosa</i>		0.07											
TRANSECT TOTAL	53.93	27.10	33.70	7.40	3.34	27.90	0.00	0.04	41.20	0.00	0.65	37.50	
std. Err.	5.4	4.5	3.8	2.7	1.1	3.2	0.0	0.0	6.6	0.0	0.4	5.2	
Number of Species	6	7	5	5	5	5	0	4	5	0	4	6	
Species Diversity	0.55	0.37	1.44	0.64	0.74	1.18	0.00	1.38	1.18	0.00	0.29	1.08	

TRANSECT	SI-5			SI-6			SI-7			SI-8		
SPECIES	1975	1979	2002	1975	1979	2002	1975	1979	2002	1975	1979	2002
<i>Porites lobata</i>		0.05	10.90	8.12	0.03	5.10	2.02	0.33	0.10	38.60	28.14	17.90
<i>Porites compressa</i>					1.20		8.10	0.81		2.92	0.81	
<i>Porites brighami</i>			0.10									
<i>Pocillopora meandrina</i>		0.01	9.40			24.20			12.70	0.46	0.37	14.70
<i>Pocillopora eydouxi</i>			1.20									2.80
<i>Montipora patula</i>			1.10			4.50			4.80	0.82	0.07	13.80
<i>Montipora capitata</i>									0.20	0.67	0.02	3.30
<i>Pavona varians</i>					0.33	1.30		1.00	0.30		0.05	
<i>Pavona duerdeni</i>						7.50		0.96	6.30			0.80
<i>Leptastrea purpurea</i>		1.02									0.05	
<i>Palythoa tuberculosa</i>		0.01			0.20	1.90	2.90	1.63	2.80	0.30	0.37	
TRANSECT TOTAL	0.00	1.09	22.70	8.12	1.76	44.50	13.02	4.73	27.20	43.77	29.88	53.30
std. Err.	0.0	0.3	5.2	1.1	0.5	5.4	3.5	2.3	3.0	5.8	3.7	4.0
Number of Species	0	4	5	1	4	6	3	5	7	6	8	6
Species Diversity	0.00	0.70	1.40	0.00	0.92	1.35	0.66	1.62	1.34	0.48	0.36	1.46

Table 7. Linear regression statistics for coral cover on transects in the vicinity of the East Honolulu Wastewater Treatment Facility Ocean Outfall. Top table includes all surveys from the monitoring program (June 1987 to November 2002). Other tables include periods from 1987-1993, 1993-1998, 1998-2000, and 2000-2002. "INC" = significant increase in coral cover; "DEC" = significant decrease in coral cover; and "NC" = no significant change in coral cover ( $p < 0.05$ ).

1987-2002 (n = 75)	SPECIES	SLOPE	STUDENTS t CON. LIMITS		CHANGE
			LOWER	UPPER	
DEEP DIFFUSER	TOTAL CORAL	-0.028	-0.063	0.007	NC
	<i>P. lobata</i>	-0.018	-0.031	-0.004	DEC
	<i>P. meandrina</i>	-0.004	-0.014	0.014	NC
	<i>M. patula</i>	0.003	-0.015	0.021	NC
CONTROL	TOTAL CORAL	-0.050	-0.080	-0.019	DEC
	<i>P. lobata</i>	-0.040	-0.057	-0.024	DEC
	<i>P. meandrina</i>	0.011	-0.008	0.029	NC
	<i>M. patula</i>	0.000	-0.007	0.008	NC

6/87 - 9/93 (n=30)	SPECIES	SLOPE	STUDENTS t CON. LIMITS		CHANGE
			LOWER	UPPER	
DEEP DIFFUSER	TOTAL CORAL	-0.375	-0.495	-0.314	DEC
	<i>P. lobata</i>	-0.026	-0.064	0.010	NC
	<i>P. meandrina</i>	-0.116	-0.167	-0.065	DEC
	<i>M. patula</i>	-0.169	-0.228	-0.109	DEC
CONTROL	TOTAL CORAL	0.017	-0.106	0.139	NC
	<i>P. lobata</i>	-0.035	-0.095	0.076	NC
	<i>P. meandrina</i>	0.005	-0.063	0.074	NC
	<i>M. patula</i>	0.037	-0.010	0.060	NC

11/93 - 10/98 (n=30)	SPECIES	SLOPE	STUDENTS t CON. LIMITS		CHANGE
			LOWER	UPPER	
DEEP DIFFUSER	TOTAL CORAL	0.233	0.179	0.787	INC
	<i>P. lobata</i>	0.013	0.039	0.146	INC
	<i>P. meandrina</i>	0.010	-0.060	0.081	NC
	<i>M. patula</i>	0.069	0.024	0.114	INC
SHALLOW DIFFUSER	TOTAL CORAL	0.240	0.159	0.320	INC
	<i>P. lobata</i>	0.291	0.204	0.378	INC
	<i>P. meandrina</i>	0.026	0.002	0.050	INC
	<i>M. patula</i>	-0.016	-0.067	0.033	NC
CONTROL	TOTAL CORAL	0.370	0.054	0.272	INC
	<i>P. lobata</i>	0.055	0.022	0.083	INC
	<i>P. meandrina</i>	0.144	0.082	0.206	INC
	<i>M. patula</i>	-0.024	-0.056	0.007	NC

10/98 - 11/00 (n=10)	SPECIES	SLOPE	STUDENTS t CON. LIMITS		CHANGE
			LOWER	UPPER	
DEEP DIFFUSER	TOTAL CORAL	0.503	0.162	0.844	INC
	P. LOBATA	0.242	0.105	0.380	INC
	P. MEANDRINA	0.216	0.088	0.344	INC
	M. PATULA	-0.082	-0.244	0.208	NC
SHALLOW DIFFUSER	TOTAL CORAL	0.787	0.426	1.149	INC
	P. LOBATA	0.445	0.207	0.085	INC
	P. MEANDRINA	0.179	0.023	0.334	INC
	M. PATULA	0.171	-0.011	0.353	NC
CONTROL	TOTAL CORAL	0.257	-0.196	0.709	NC
	P. LOBATA	0.072	-0.262	0.407	NC
	P. MEANDRINA	0.135	-0.101	0.371	NC
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11/00 - 11/02 (n=6)	SPECIES	SLOPE	STUDENTS t CON. LIMITS		CHANGE
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DEEP DIFFUSER	TOTAL CORAL	-0.293	-0.672	0.086	NC
	P. LOBATA	-0.422	-0.806	-0.038	DEC
	P. MEANDRINA	0.102	-0.161	0.366	NC
	M. PATULA	0.016	-0.521	0.553	NC
SHALLOW DIFFUSER	TOTAL CORAL	-0.420	-0.633	-0.207	DEC
	P. LOBATA	-0.321	-0.592	-0.049	DEC
	P. MEANDRINA	-0.061	-0.299	0.175	NC
	M. PATULA	-0.063	-0.406	0.532	NC
CONTROL	TOTAL CORAL	-0.883	-1.430	-0.033	DEC
	P. LOBATA	-0.174	-0.72	0.371	NC
	P. MEANDRINA	-0.614	-1.023	-0.205	DEC
	M. PATULA	-0.009	-0.078	0.267	NC

TABLE 8. Mean percentages of bottom cover and total coral cover of three dominant species, and percent bottom cover of all species.

Coral Species		SURVEY YEAR					
		1973	1974	1980	1992	1993	2002
<i>Porites lobata</i>	% bottom	17.8	17.6	9.1	10.5	8.4	13.9
	% coral cover	34.5	38.4	86.7	70.0	71	43.0
<i>Porites compressa</i>	% bottom	22.5	18.6	0.5	1.5	0.6	3.3
	% coral cover	43.6	40.5	4.8	10.0	5.3	10.3
<i>Pocillopora meandrina</i>	% bottom	8.4	6.5	0.4	1.8	1.6	13.5
	% coral cover	16.3	14.2	3.8	12.0	13.9	41.7
<b>Subtotal for three species</b>	% bottom	48.7	42.7	10.0	13.8	10.7	30.7
	% coral cover	94.4	93.1	95.3	92.0	90.6	95.0
<b>TOTAL</b>	% bottom	<b>51.6</b>	<b>45.8</b>	<b>10.5</b>	<b>15.0</b>	<b>11.8</b>	<b>32.24</b>



TABLE 9. Wilcoxon ranked sums (Ts) for mean percent toal coral cover, mean percent cover of the three most abundant species, and species diversity on 15 transects paired in successive surveys. For n=15, the critical value of Ts is 25 for alpha=0.05, and 15 for alpha=0.01 (two-tailed). Highlighted values indicate significance at P<0.05 level; highlighted and underlined values indicate significance at the P<0.1 level. Significant positive values indicate increases between successive surveys; significant negative values indicate decreases between successive surveys.

	SURVEY YEAR PAIR				
	73-74	74-80	80-92	92-93	93-02
Total Coral	<b>-20.0</b>	<b>-3.0</b>	44.5	-43.0	<b><u>12.0</u></b>
<i>Porites lobata</i>	-47.5	<b>-18.0</b>	50.0	-48.0	29.5
<i>Porites compressa</i>	-38.5	<b><u>-0</u></b>	<b><u>6.0</u></b>	<b>-20.0</b>	<b><u>15.0</u></b>
<i>Pocillopora meandrina</i>	<b>-23.5</b>	<b><u>-0</u></b>	<b>24.0</b>	39.5	<b><u>10.0</u></b>
Species diversity	35.0	36.0	<b>22.0</b>	-52.0	58.0

TABLE 10. Absolute and percentage (%) change in coral cover in each zone between each pair of successive surveys. Negative changes indicate decrease in coral cover, positive changes indicate increase in cover over the survey pair interval. Also shown are results of students t-test performed on values of absolute change in coral cover expressed as proportions, then acrsin square root transformed. Highlighted values indicate significance at  $P < 0.05$ ; highlighted and underlined values indicate significance at  $P < 0.01$ . n = number of transects per zone.

ZONE		SURVEY YEAR PAIR				
		73-74	74-80	80-92	92-93	93-02
<i>Pocillopora meandrina</i> boulder n=4 df=3	<i>Absolute change</i>	-2.9	-15.6	-2.7	-2.4	23.4
	<i>% change</i>	-9.6	-57.8	-23.5	-27.2	369
	t value	1.02	4.19	2.99	0.67	-5.95
	significance of t	0.382	<b>0.025</b>	0.058	0.551	<b>0.005</b>
<i>Porites lobata</i> bench n=4 df=3	<i>Absolute change</i>	-7.6	-35.0	7.5	-14.2	35.6
	<i>% change</i>	-10.9	-65.4	40.3	-54.1	246
	t value	2.77	5.09	-1.26	8.25	-6.73
	significance of t	0.070	<b>0.015</b>	0.296	<b>0.004</b>	<b>0.0012</b>
<i>Porites compressa</i> slope n=5 df=4	<i>Absolute change</i>	-6.4	-58.4	5.0	5.9	10.1
	<i>% change</i>	-8.8	-88.0	38.5	-45.1	46.9
	t value	1.49	5.75	-2.24	-1.28	-0.60
	significance of t	0.212	<b>0.004</b>	0.089	0.268	0.305
<i>Porites lobata</i> rubble n=2	<i>Absolute change</i>	0.4	2.7	-1.7	-0.1	2.5
	<i>% change</i>	14.5	76.0	-28.0	-3.3	64.1