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Reef Coral Populations and Growth on the Flower Garden Banks, Northwest Gulf of Mexico

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Abstract Since the 1970's, coral populations, cover, diversity, and growth rates have been monitored on high-latitude (28°N) reefs in the Gulf of Mexico. Random and repetitive photography, and direct measurements have been used to document change of reef corals on the Flower Garden Banks. The monitoring program addresses concerns over long-term impacts of hydrocarbon production, discrete and cumulative effects of mechanical impacts caused by other human activities, and long-term natural change caused by stochastic events. No deleterious changes in coral populations, cover, relative dominance, diversity, evenness, or coral growth rates have been documented since the late 1970's. These and other data on reef development, disease incidence, reproduction, and recruitment indicate the reefs are unique, viable, active, and fairly pristine.

Introduction

Environmental impacts caused by thermal, oil, chemical, or nutrient pollution can cause gradual deterioration of coral reefs by affecting coral health (e.g. Marszalek 1987). Similarly, chronic low level mechanical stresses imposed by coral collection, destructive fishing techniques, high levels of diver visitation or boat and ship activities are causes of gradual decline of coral populations (Salvat 1987).

Environmental threats posed by escalating hydrocarbon development and associated activities on the outer continental shelf (OCS) in the northwest Gulf of Mexico prompted the U.S. Department of Interior's Minerals Management Service to sponsor a long-term monitoring program at the Flower Garden Banks (Fig. 1). The East (EFG) and West (WFG) Flower Garden Banks are located on the edge of the continental shelf, slightly over 180 km south southeast of Galveston, Texas. The banks are topographic expressions of seafloor uplift caused by vertically migrating salt domes originating from Jurassic, Louann evaporite deposits 15 km below the seafloor (Rezak et al. 1985). The crests of these isolated banks, which are 19 km apart, support coral reefs that rise to within 15 m of the surface (Bright et al. 1984). Together, the bank zones containing the highest diversity coral reefs (above 36 m) cover over 1.3 km^2.

The Flower Garden Banks, at nearly 28°N, are very near the northern physiological limits for tropical reef-building corals in the Gulf of Mexico (Rezak et al. 1990). Only 18 of the 65 Western Atlantic reef-building coral species occur (Bright et al. 1984). Yet, reported abundance and growth rates are similar to those in more tropical locales at similar depths (Hudson and Robbin 1980).

In January 1992, the Flower Garden Banks were designated a National Marine Sanctuary by the National Oceanic and Atmospheric Administration. Recreational use of underwater areas has historically increased following their establishment as marine parks, preserves, sanctuaries, or authorities (Tilmant 1987). A long-term database, and standardized data collection and analysis techniques may enable identification of impacts caused by the expected increase in recreational use as well as those caused by escalating industrial activity in the region.

The objectives of the long-term monitoring study at the Flower Garden Banks were 1) to pro-

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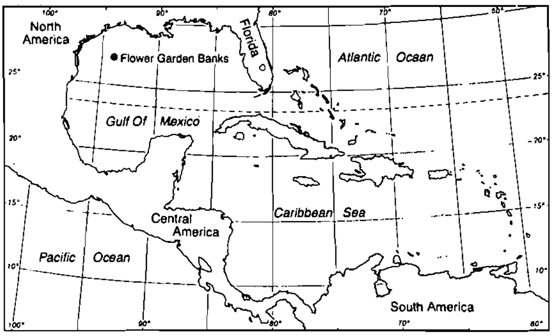

Fig. 1. The Caribbean Sea and the Gulf of Mexico, showing the location of the Flower Garden Banks.

vide relevant and timely environmental data to those responsible for developing policies concerning oil and gas exploration and production in the vicinity of these ecosystems; and 2) to document long-term changes in coral and associated communities at the Flower Gardens in light of natural variation and impact by petroleum exploration, development, and activities related to increasing recreational use. This paper reports on changes in coral populations, cover, and diversity, and changes in accretionary and encrusting growth rates.

Methods

Field Methods

The establishment of monitoring stations on the Flower Garden Banks in 1988 and early 1989 involved first delimiting one 100 by 100 m area on each bank containing reef communities representative of those inhabiting the high diversity zone of the banks (Bright et al. 1984). The total study site area represented approximately one percent of the total reef area on the two banks. This was followed by implanting and mapping 120 permanent stations to monitor encrusting (lateral) growth of *Montastrea annularis* and *Diploria strigosa* on each bank (60 stations for each species), and 30 stakes placed in *M. annularis* colonies on each bank to measure colony accretion rates (Continental Shelf Associates, Inc. (CSA) 1990; Gittings et al. 1992). Field work was conducted at both banks on six cruises at roughly six month intervals.

During each sampling effort, twenty, 10 by 0.5 m stratified random transects (n.b. sand flats were avoided) were photographed in each study area.

On each bank, the 30 spikes implanted in the tops of colonies of *M. annularis* were periodically measured to determine accretionary growth. Sclerochronology (Hudson et al. 1976) was used to obtain long-term records of accretionary growth rates. Cores were taken in May 1990 from four *M. annularis* colonies, two from each bank.

Stations for monitoring the encrusting growth of *M. annularis* and *D. strigosa* were established by implanting two stainless steel nails 23 cm apart near colony borders. A close-up framer attached to an underwater camera and placed directly over the nails allowed photography of a repeatable 13.3 by 19.7 cm area.

Analytical Methods

Percent cover data were acquired from random transect photos planimetrically (Gittings et al. 1992) for all coral species, leafy algae, sponges, and reefrock. The number of colonies of each species were also measured. Relative dominance of each coral species (percent cover relative to total coral cover), species diversity based on coral counts (H'_N; from the natural log form of the Shannon-Weaver Diversity Index), and evenness (E_N; species diversity divided by the maximum possible diversity; maximum diversity would be the natural log of the number of species present) were calculated. For species diversity and evenness based on coral cover (H'_C and E_C, respectively), p_i in the diversity formula $H' = -\Sigma\, p_i \ln p_i$ was relative dominance.

Spatial and temporal comparisons were made using the Kruskal-Wallis test. Where significant differences were found ($p<0.05$), Tukey's multiple range test was applied to identify similar sample groups (SAS 1985). Comparisons to historical data from the Flower Gardens were made using data of Tresslar (1974), Viada (1980), Kraemer (1982), and Bright et al. (1984).

Accretionary growth rates from 1910 to 1989 were determined planimetrically by measuring the average width of annual bands on X-radiographs of core slabs.

Encrusting growth and retreat were analyzed by first projecting sequential margins on close-up photographs onto the same surface using an image enlarging/reducing map projector. Areas of growth and retreat, as well as border lengths over which the changes occurred, were measured by planimetry (Gittings et al. 1988).

Results

Coral Community Parameters

Total coral cover did not differ significantly between study sites (46.0 ± 2.2% on the EFG and 46.5 ± 2.0% on the WFG; Table 1), or over time

Table 1. Comparisons between study sites of species diversity, species evenness, and percent coral cover. Subscript C denotes index calculations based on coral cover estimates. Subscript N denotes index calculations based on coral counts. Kruskal-Wallis Test values (F), degrees of freedom (df), and associated probability values (P-values) are presented. East = East Flower Garden Bank. W = West Flower Garden Bank. ns = no significant difference at α=0.05.

Index	df	F	P-value	Results
H'_C Diversity	1,236	5.84	0.0164	East>West
H'_N Diversity	1,236	14.01	0.0002	West>East
E_C Evenness	1,236	7.88	0.0054	East>West
E_N Evenness	1,236	21.18	0.0001	West>East
Percent Cover	1,236	0.22	0.6395	ns

($p>0.05$). Species diversity and evenness based on coral counts were significantly higher at the WFG study site. Species diversity and evenness based on coral cover showed the opposite pattern, being significantly higher at the EFG study site. Population levels (24.4 colonies m^{-2} at the EFG vs. 18.0 colonies m^{-2} at the WFG), and mean colony areas (size) of *Siderastrea siderea* and *Porites astreoides* differed significantly between sites. *S. siderea* was the largest of all species, and *P. astreoides* the most abundant.

Visual estimates of percent cover on the WFG were made in 1972 by divers in 26, 10m^2 circular quadrats (Tresslar, 1974). Cover averaged 38 ± 7% in an area near, but outside sites used in the present study. Data collected from 1978–1982 (Viada 1980; Kraemer 1982) reported cover of 50.4 ± 5.3% and 55.2 ± 6.6% for the EFG and WFG, respectively, but used a different field and analysis methods. These studies used overlapping photography along a transect tape underwater, and analysis using the line-intercept method rather than planimetry. Fewer samples were collected, and variability was somewhat higher. Nevertheless, cover data were not significantly different than those acquired in this study, nor were measures of diversity and evenness. In addition, cover and relative dominance for individual species reported in the 1978–1982 period were close to those found in this study (Fig. 2), though significant differences in relative dominance were found for two species on the EFG.

Coral Growth

Accretionary growth rates of *M. annularis*, measured using growth spikes between 1989 and 1991, averaged 6.8 mm/yr. Mean accretionary growth measured in cores from 1910 to 1989 was 6.6 mm/yr (range of 5.1 to 8.2 mm/yr; Fig. 3). A decrease in growth rates, and a concurrent increase in year-to-year variability was evident between 1957 and 1980.

Net growth rates on *M. annularis* and *D. strigosa* margins (average rate of change of advancing and retreating areas) were positive for nearly all semi-annual periods from 1989 through 1991 (Fig. 4). Net encrusting growth rates differed from data collected in the Florida Keys, which showed growth rates of essentially zero on apparently healthy adult *M. annularis* colonies (Gittings et al. 1988).

The average rates of advance and retreat (i.e. the magnitude of change) were not significantly different for either species (Fig. 5). Growth of *M. annularis* averaged 0.38 cm/6 mo. and retreat averaged −0.43 cm/6 mo. Respective rates for *D. strigosa* averaged 0.42 and −0.40. The rates of growth and retreat reported by Kraemer (1982) did not differ significantly for either species in this study.

Measurements of cumulative areas of advance and retreat showed that for every cm^2 of *D. strigosa* tissue that died between 1989 and 1991, approximately 1.5 cm^2 grew (retreat to advance ratio of 0.65). For every cm^2 of *M. annularis* lost, approximately 1.8 cm^2 grew (retreat to advance ratio of 0.55).

Tissue loss exceeded tissue gain only during the fall and winter between 1990 and 1991 (Fig. 6). This was a period of unusually low net growth rates (Fig. 4) It was followed by the highest levels of tissue gain in the study.

Analyses of the proportions of margin length advancing, retreating, and remaining stable between surveys showed that, on average, over half of the marginal tissue on *M. annularis* and *D. strigosa* advanced (Fig. 7; 63% of analyzed margin length for *M. annularis* and 56% for *D. strigosa*). By contrast, the proportions of retreating margins averaged only 22% for *M. annularis* and 26% for *D. strigosa.*

Discussion

Virtually no significant changes have been detected in coral reef populations, cover, or diversity at the

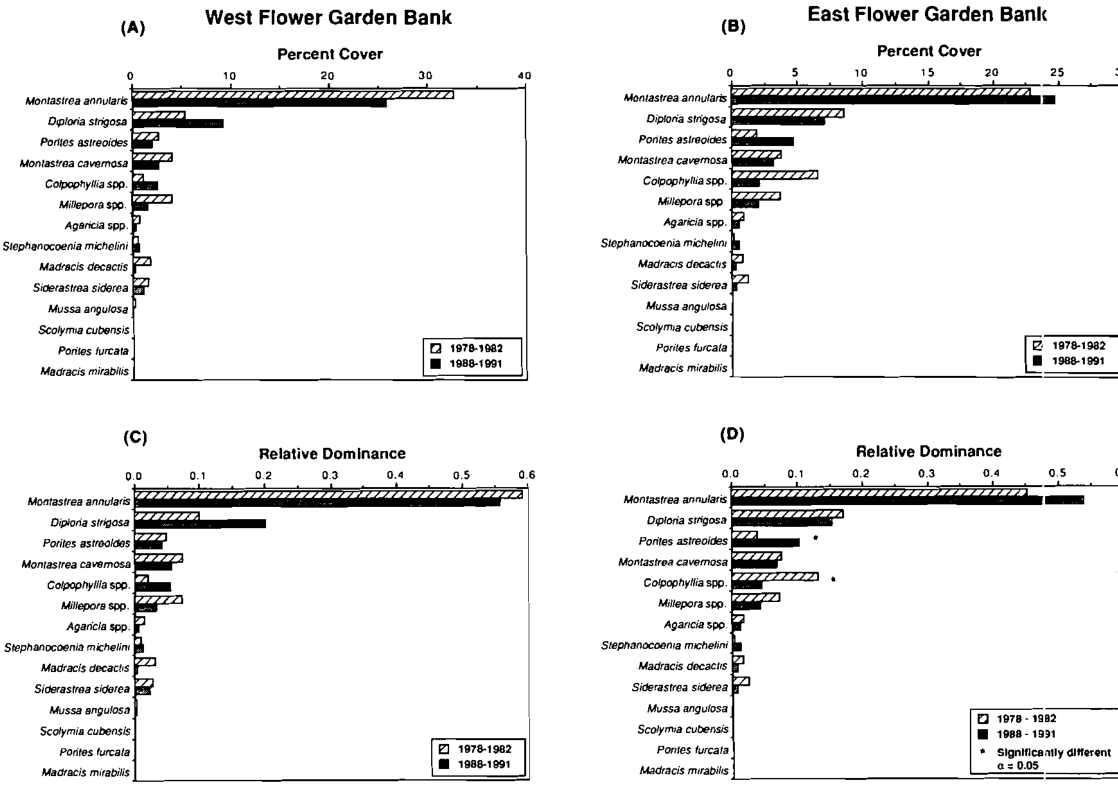

Fig. 2. Percent cover of corals on the West (A) and East (B) Flower Garden Banks and their relative dominance on the West (C) and East (D) Banks between 1978–1982 and 1988–1991

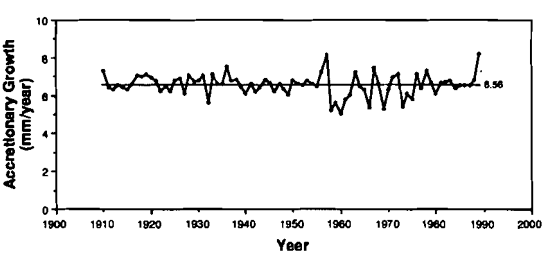

Fig. 3. Accretionary growth rates of *Montastrea annularis* from 1910 to 1989 at the Flower Garden Banks. Data were based on measurements from four cores. Middle line indicates mean (6.56 mm/yr)

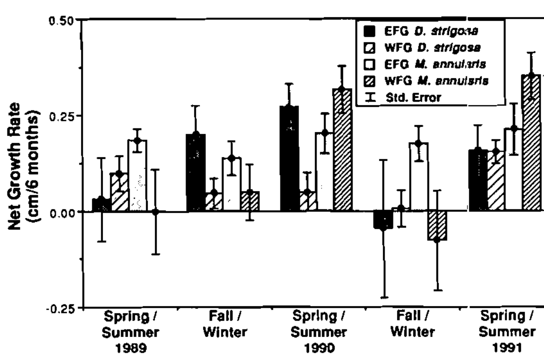

Fig. 4. Net lateral growth rates of *Diploria strigosa* and *Montastrea annularis* from 1989 to 1991 at the East and West Flower Garden Banks

Flower Garden Banks in the 20 years since quantitative surveys began. Differences in the present study, such as low net growth rates during the fall/winter 1990–1991 period, were transitory, and no observed differences were considered indicative of habitat deterioration.

Growth data suggest favorable conditions at the Flower Gardens. Tissue retreat under such conditions is likely to be dictated by natural factors, such as competition for space (Lang 1973), rather than man-induced stress (e.g. Marszalek 1987), and/or a situation where vectors such as disease are not widespread. Data from 80 repetitively photographed 8m² areas indicated that, while diseases at the Flower Gardens can lead to substantial tissue mortality, less than 67 (1.6%) of 4,160 colonies analyzed over a three-year period appeared to be affected (Gittings et al. 1992). Evidence of environmental deterioration caused by human activities

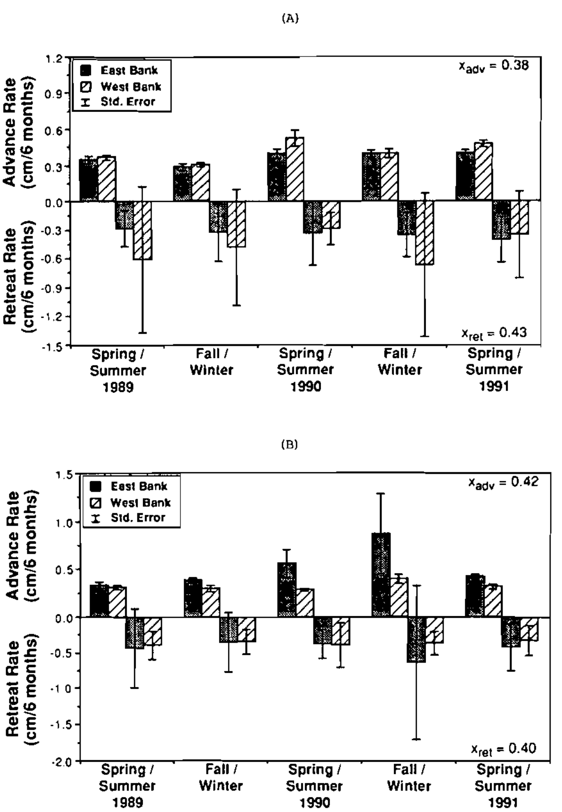

Fig. 5. Advance and retreat rates of *Montastrea annularis* (A) and *Diploria strigosa* (B) between 1989 and 1991. Note high standard error for retreats rate estimates compared to those of advance rates

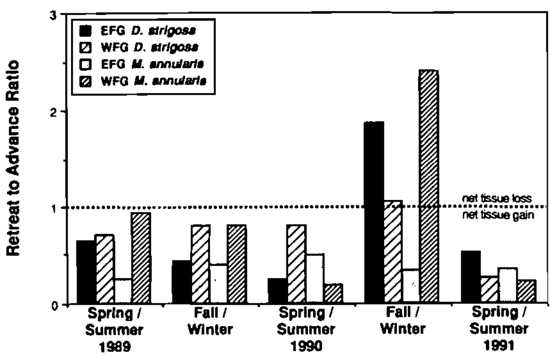

Fig. 6. Ratios of areas of retreat to areas of advance for *Diploria strigosa* and *Montastrea annularis* between 1989 and 1991 at the East and West Flower Garden Banks

would probably have included some of the following: accelerated retreat relative to advance rates; reduced advance and net growth rates; low proportions of advancing and stable margin length; increased retreating margin length; and higher disease incidence. None of these effects were observed at the Flower Garden Banks.

Where tissue mortality occurs, whether by disease or other causes, it generally retreats at a faster rate than it is capable of advancing. This is evident in the higher standard errors for retreat in Fig. 5. Similar growth and retreat rates at the Flower Gardens imply that areas of comparatively minor tissue retreat occur more frequently than areas of substantial retreat. Large areas of retreat are indicative of diseases, as well as recent mechanical impact (Gittings et al. 1988), neither of which are particularly common at the Flower Gardens.

Unlike this study, Gittings et al. (1988) found higher tissue retreat to advance ratios in the Florida Keys, averaging around 1.0 for most coral colonies. The Florida study involved mostly coral colonies affected by a ship grounding, yet control station retreat to advance ratios measured 1.08. They also found lower proportions of advancing margin length, and higher proportions of stable margin length at control stations in the Keys, suggesting less favorable conditions for coral growth.

For both *M. annularis* and *D. strigosa*, net tissue loss in the fall/winter 1990–1991 period (Fig. 6) may be attributable to transitory increases in the proportion of retreating margin length (Fig. 7). In neither species were growth rates significantly lower nor retreat rates significantly higher in fall/winter 1990–1991 than other periods (Fig. 5). Thus, while growth and retreat rates did not change during this period, less marginal tissue was growing.

Hudson and Robbin (1980) reported a 20% decrease in accretionary growth at the Flower Gardens between 1957 and 1978. Whatever the cause, recent data suggest the event was transitory (Fig. 3). Growth after 1980 was similar to the overall mean, and 1989 growth averaged the highest of any year since 1910.

The remote location of the Flower Garden banks has left them, for the most part, undisturbed by man. Discernible human impacts have been limited to occasional mechanical destruction caused by anchors, and debris (e.g. anchors, chains, and cables). Resource monitoring has demonstrated no substantial long-term changes in reef coral populations or growth rates that may be attributable to human activities. Potential industrial effects caused by offshore development in the northwest Gulf have been monitored (e.g. CSA 1985), but never detected, nor have long-term water quality changes (Deslarzes 1992). Monitoring studies indicate that the Flower Gardens have comparatively high coral cover, low disease incidence, and accretionary growth rates that are comparable to reefs in tropical settings.

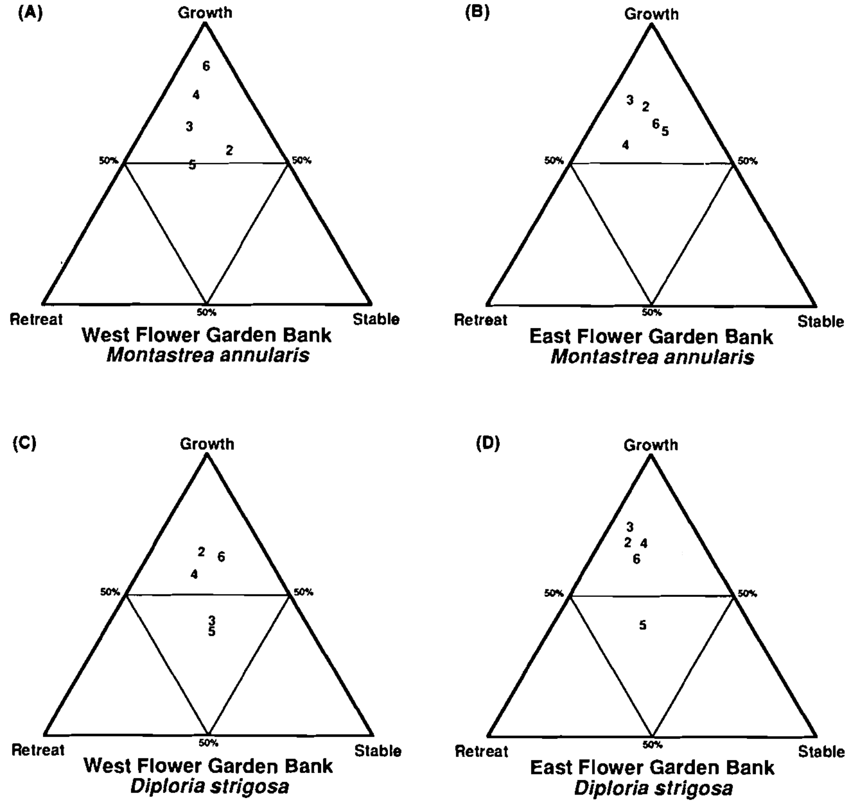

Fig. 7. Ternary diagrams showing the proportions of growing, retreating, and stable coral margin length for *Montastrea annularis* at the West (A) and East (B) Flower Garden Banks and *Diploria strigosa* at the West (C) and East (D) Banks. Numbers indicate time period as follows: 2 = spring/ summer 1989; 3 = following fall/winter; 4 = spring summer 1990; 5 = following fall/winter; 6 = spring/summer 1991

Collaborative research at the Flower Gardens has demonstrated high recruitment rates of brooding (Baggett and Bright 1985) and broadcasting corals (Bright et al. 1991), and mass coral spawning events (Gittings et al. 1992). These data suggest the reefs are fully functional, capable of self-seeding, and probably less impaired by their high latitude location than has been previously thought. It is anticipated that the recent designation of the Flower Garden Banks National Marine Sanctuary will foster continued resource protection and monitoring, and encourage increased research on important functional attributes of these unique and pristine coral reefs.

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